

ENCYCLOPEDIA OF
Soil Science

THIRD EDITION

Edited by
Rattan Lal



CRC Press
Taylor & Francis Group

ENCYCLOPEDIA OF
Soil Science
THIRD EDITION

Encyclopedias from the Taylor & Francis Group

Print	Online
-------	--------

Agriculture

Encyclopedia of Agricultural, Food, and Biological Engineering, 2nd Ed., 2 Vols. K10554 (978-1-4398-1111-5)	Pub'd. 10/21/10 K11382 (978-1-4398-2806-9)
Encyclopedia of Animal Science, 2nd Ed., 2 Vols. K10463 (978-1-4398-0932-7)	Pub'd. 2/1/11 K10528 (978-0-415-80286-4)
Encyclopedia of Biotechnology in Agriculture and Food DK271X (978-0-8493-5027-6)	Pub'd. 7/16/10 DKE5044 (978-0-8493-5044-3)
Encyclopedia of Pest Management DK6323 (978-0-8247-0632-6)	Pub'd. 5/9/02 DKE517X (978-0-8247-0517-6)
Encyclopedia of Plant and Crop Science DK1190 (978-0-8247-0944-0)	Pub'd. 2/27/04 DKE9438 (978-0-8247-0943-3)
Encyclopedia of Soil Science, 3rd Ed., 3 Vols. K26612 (978-1-4987-3890-3)	Publishing 2016 K26614 (978-1-4987-3893-4)
Encyclopedia of Water Science, 2nd Ed., 2 Vols. DK9627 (978-0-8493-9627-4)	Pub'd. 12/26/07 DKE9619 (978-0-8493-9619-9)

Business and Computer Science

Encyclopedia of Computer Sci. & Tech., 2nd Ed., 2 Vols. K21573 (978-1-4822-0819-1)	Publishing 2016 K21578 (978-1-4822-0822-1)
Encyclopedia of Information Assurance, 4 Vols. AU6620 (978-1-4200-6620-3)	Pub'd. 12/21/10 AUE6620 (978-1-4200-6622-7)
Encyclopedia of Information Systems and Technology, 2 Vols. K15911 (978-1-4665-6077-2)	Pub'd. 12/29/15 K21745 (978-1-4822-1432-1)
Encyclopedia of Library and Information Sciences, 4th Ed. K15223 (978-1-4665-5259-3)	Publishing 2017 K15224 (978-1-4665-5260-9)
Encyclopedia of Software Engineering, 2 Vols. AU5977 (978-1-4200-5977-9)	Pub'd. 11/24/10 AUE5977 (978-1-4200-5978-6)
Encyclopedia of Supply Chain Management, 2 Vols. K12842 (978-1-4398-6148-6)	Pub'd. 12/21/11 K12843 (978-1-4398-6152-3)
Encyclopedia of U.S. Intelligence, 2 Vols. AU8957 (978-1-4200-8957-8)	Pub'd. 12/19/14 AUE8957 (978-1-4200-8958-5)
Encyclopedia of Wireless and Mobile Communications, 2nd Ed., 3 Vols. K14731 (978-1-4665-0956-6)	Pub'd. 12/18/12 KE16352 (978-1-4665-0969-6)

Chemistry, Materials and Chemical Engineering

Encyclopedia of Chemical Processing, 5 Vols. DK2243 (978-0-8247-5563-8)	Pub'd. 11/1/05 DKE499X (978-0-8247-5499-0)
Encyclopedia of Chromatography, 3rd Ed. 84593 (978-1-4200-8459-7)	Pub'd. 10/12/09 84836 (978-1-4200-8483-2)
Encyclopedia of Iron, Steel, and Their Alloys, 5 Vols. K14814 (978-1-4665-1104-0)	Pub'd. 1/6/16 K14815 (978-1-4665-1105-7)
Encyclopedia of Plasma Technology, 2 Vols. K14378 (978-1-4665-0059-4)	Publishing 2016 K21744 (978-1-4822-1431-4)
Encyclopedia of Supramolecular Chemistry, 2 Vols. DK056X (978-0-8247-5056-5)	Pub'd. 5/5/04 DKE7259 (978-0-8247-4725-1)
Encyclopedia of Surface & Colloid Science, 3rd Ed., 10 Vols. K20465 (978-1-4665-9045-8)	Pub'd. 8/27/15 K20478 (978-1-4665-9061-8)

Print	Online
-------	--------

Engineering

Dekker Encyclopedia of Nanoscience and Nanotechnology, 3rd Ed., 7 Vols. K14119 (978-1-4398-9134-6)	Pub'd. 3/20/14 K14120 (978-1-4398-9135-3)
Encyclopedia of Energy Engineering and Technology, 2nd Ed., 4 Vols. K14633 (978-1-4665-0673-2)	Pub'd. 12/1/14 KE16142 (978-1-4665-0674-9)
Encyclopedia of Optical and Photonic Engineering, 2nd Ed., 5 Vols. K12323 (978-1-4398-5097-8)	Pub'd. 9/22/15 K12325 (978-1-4398-5099-2)

Environment

Encyclopedia of Environmental Management, 4 Vols. K11434 (978-1-4398-2927-1)	Pub'd. 12/13/12 K11440 (978-1-4398-2933-2)
Encyclopedia of Environmental Science and Engineering, 6th Ed., 2 Vols. K10243 (978-1-4398-0442-1)	Pub'd. 6/25/12 KE0278 (978-1-4398-0517-6)
Encyclopedia of Natural Resources, 2 Vols. K12418 (978-1-4398-5258-3)	Pub'd. 7/23/14 K12420 (978-1-4398-5260-6)

Medicine

Encyclopedia of Biomaterials and Biomedical Engineering, 2nd Ed. H7802 (978-1-4200-7802-2)	Pub'd. 5/28/08 HE7803 (978-1-4200-7803-9)
Encyclopedia of Biomedical Polymers and Polymeric Biomaterials, 11 Vols. K14324 (978-1-4398-9879-6)	Pub'd. 4/2/15 K14404 (978-1-4665-0179-9)
Encyclopedia of Biopharmaceutical Statistics, 3rd Ed. H100102 (978-1-4398-2245-6)	Pub'd. 5/20/10 HE10326 (978-1-4398-2246-3)
Encyclopedia of Clinical Pharmacy DK7524 (978-0-8247-0752-1)	Pub'd. 11/14/02 DKE6080 (978-0-8247-0608-1)
Encyclopedia of Dietary Supplements, 2nd Ed. H100094 (978-1-4398-1928-9)	Pub'd. 6/25/10 HE10315 (978-1-4398-1929-6)
Encyclopedia of Medical Genomics and Proteomics, 2 Vols. DK2208 (978-0-8247-5564-5)	Pub'd. 12/29/04 DK501X (978-0-8247-5501-0)
Encyclopedia of Pharmaceutical Science and Technology, 4th Ed., 6 Vols. H100233 (978-1-84184-819-8)	Pub'd. 7/1/13 HE10420 (978-1-84184-820-4)

Routledge Encyclopedias

Encyclopedia of Public Administration and Public Policy, 3rd Ed., 5 Vols. K16418 (978-1-4665-6909-6)	Pub'd. 11/6/15 K16434 (978-1-4665-6936-2)
Routledge Encyclopedia of Modernism Y137844 (978-1-135-00035-6)	Pub'd. 5/11/16
Routledge Encyclopedia of Philosophy Online RU22334 (978-0-415-24909-6)	Pub'd. 11/1/00
Routledge Performance Archive Y148405 (978-0-203-77466-3)	Pub'd. 11/12/12

Encyclopedia titles are available in print and online.
To order, visit <http://www.crcpress.com>
Telephone: 1-800-272-7737 Fax: 1-800-374-3401
E-Mail: orders@taylorandfrancis.com

ENCYCLOPEDIA OF
Soil Science
THIRD EDITION

Edited by
Rattan Lal



CRC Press

Taylor & Francis Group
Boca Raton London New York

CRC Press is an imprint of the
Taylor & Francis Group, an **informa** business

CRC Press
Taylor & Francis Group
6000 Broken Sound Parkway NW, Suite 300
Boca Raton, FL 33487-2742

© 2017 by Taylor & Francis Group, LLC
CRC Press is an imprint of Taylor & Francis Group, an Informa business

No claim to original U.S. Government works

Printed on acid-free paper
Version Date: 20160919

International Standard Book Number-13: 978-1-4987-8699-7 (Hardback)

This book contains information obtained from authentic and highly regarded sources. Reasonable efforts have been made to publish reliable data and information, but the author and publisher cannot assume responsibility for the validity of all materials or the consequences of their use. The authors and publishers have attempted to trace the copyright holders of all material reproduced in this publication and apologize to copyright holders if permission to publish in this form has not been obtained. If any copyright material has not been acknowledged please write and let us know so we may rectify in any future reprint.

Except as permitted under U.S. Copyright Law, no part of this book may be reprinted, reproduced, transmitted, or utilized in any form by any electronic, mechanical, or other means, now known or hereafter invented, including photocopying, microfilming, and recording, or in any information storage or retrieval system, without written permission from the publishers.

For permission to photocopy or use material electronically from this work, please access www.copyright.com (<http://www.copyright.com/>) or contact the Copyright Clearance Center, Inc. (CCC), 222 Rosewood Drive, Danvers, MA 01923, 978-750-8400. CCC is a not-for-profit organization that provides licenses and registration for a variety of users. For organizations that have been granted a photocopy license by the CCC, a separate system of payment has been arranged.

Trademark Notice: Product or corporate names may be trademarks or registered trademarks, and are used only for identification and explanation without intent to infringe.

Visit the Taylor & Francis Web site at
<http://www.taylorandfrancis.com>

and the CRC Press Web site at
<http://www.crcpress.com>

Brief Contents

Volume I

Accounting: Emergy Analysis	1	Arsenic: Southeast Asia	161
Acid Mine Drainage	6	Art and Soil	168
Acid Rain: Nitrogen Deposition	11	Atterberg Limits	171
Acid Sulfate Soils	17	Bacteria	175
Acid Sulfate Soils: Distribution and Extent	20	Biochar and Soil Characteristics	184
Acid Sulfate Soils: Formation	26	Biochar and Soil Quality	189
Acid Sulfate Soils: Management	31	Biochar: Soil Carbon and Fertility	193
Acid Sulfate Soils: Problems	36	Biodegradation and Bioremediation	198
Actinobacteria	40	Biodiversity and Sustainability	205
Aeration: Measurement	44	Bioenergy Crops: Carbon Balance Assessment	210
Aeration: Tillage Effects	47	Biofuels versus Agricultural Soils	213
Aggregates: Tensile Strength	51	Biological Crusts	216
Aggregation	55	Biological Nitrogen Fixation	220
Aggregation: Soil Organic Matter (SOM) and	58	Biological Nitrogen Fixation: Contributions to Agriculture	223
Aggregation: Structural Stability Measurement	61	Biological Nitrogen Fixation: Enhancement Techniques	227
Agricultural Lands: Flooding and Levee Breaches	65	Biological Nitrogen Fixation: Forms and Regulating Factors	232
Agricultural Management: Root Growth	72	Biota	235
Agricultural Soil Carbon: Trading	76	Biota and Disease-Suppressive Soils	239
Air Permeability	85	Biotechnology	243
Albedo	89	Boreal Forest	247
Alfisols	92	Boron and Molybdenum	252
Allophanes	97	Brick Making: Soil Degradation	255
Alpine Soils	101	Bulk Density	258
Aluminum	105	Bunch Planting: Water Conservation	261
Amazon Basin Soils	111	Calcification	264
Amendments and Ameliorants	114	Calcium	267
Ammonia: Volatilization from Agricultural Soils	117	Carbon Assessment: Spectroscopic Methods	270
Amoozometer	120	Carbon Sequestration: Fish Ponds	274
Anaerobic Processes	126	Carbon Sequestration: Geologic	279
Andisols	130	Carbon Sequestration: Iceland	284
Andisols: Iceland	134	Carbon Sequestration: Nutrient Management	288
Animals: Ecosystem Functioning	138	Carbon Sequestration: Sediments	294
Animals: Microbial Interactions and Nutrient Cycling	142	Carbon Sequestration: Semiarid Regions of India	299
Antimony	146	Carbon Sequestration: Urban Ecosystems	307
Ants	150	Carbon Sink Capacity: Soil Profile Characteristics	315
Archaeology	154	Carbon: Inelastic Neutron Scattering	319
Arid Soils	157	Carbon: Laser-Induced Breakdown Spectroscopy	323

Volume I (*cont'd.*)

Carbon: Physical Protection	327	Crop Evapotranspiration: Measurement System	488
Carbonates	331	Crop Management: Seasonally Frozen Soil	492
Carbonates: Secondary and Pedogenic	335	Crop Plants: Phosphorus Availability	494
Carbonates: Water and	339	Crop Residue Management	497
Care of Soil	342	Crop Root Response to Temperature: Temperate Zone	501
Cerrado	345	Crop Rotation and Farming Systems: Temperate Zone	504
Chemical Composition	348	Crop Rotation and Farming Systems: Tropical Smallholder Farm Systems	507
Chemisorption	353	Crop Rotation and Farming Systems: Tropics and Subtropics and Swelling Soils	512
Chernozems	356	Crop Yield: Compaction	516
Chlorine	362	Croplands	522
Classification Systems	367	Crops: Flooding Tolerance	526
Classification Systems: Arab Traditional, Moroccan Case	370	Cultivation: Raised-Bed	529
Classification Systems: Australian and New Zealand	375	Culture: Soil-less	533
Classification Systems: Brazilian	379	Cycling: Carbon and Nitrogen (C/N)	538
Classification Systems: Canadian	383	Debris Flow	544
Classification Systems: Chinese	389	Degradation	548
Classification Systems: French	391	Degradation: Biological	553
Classification Systems: Historical Developments	395	Degradation: Chemical	558
Classification Systems: Indian	398	Degradation: Critical Limits and Irreversible Degradation	561
Classification Systems: Indigenous	402	Degradation: Economic Incentives and Investments	565
Classification Systems: Major	406	Degradation: Farm-Level Economic Implications	568
Classification Systems: Netherlands	412	Degradation: Food Aid Needs in Low-Income Countries	571
Classification Systems: Russian	415	Degradation: Food Security and Poverty	574
Classification Systems: South African	418	Degradation: Greenhouse Effect	578
Clay Minerals	421	Degradation: Mapping by Normalized Difference Vegetation Index (NDVI)	581
Clay Minerals: Charge Properties	426	Degradation: Off-Farm Economic Implications	593
Clay Minerals: Weathering and Alteration of	430	Degradation: Physical	596
Climate	435	Degradation: Quality and	602
Climate Change: Gas Fluxes	438	Desertification: Biological Productivity Decline	608
Climate Change: Soil Carbon Sequestration	442	Desertification: Extent	616
Cobalt and Iodine	445	Desertification: Greenhouse Effect	620
Collembola	448	Desertification: Humid Environment in Iceland	623
Color	452	Desertification: Impact	628
Compaction	456	Desertification: Mapping Constraints and Challenges	633
Conditioning Index	459	Desertification: Prevention and Restoration	642
Conservation Tillage	462		
Conserved Lands: Stewardship	466		
Consistence	471		
Consolidation	474		
Controlled Traffic	477		
Copper	481		
Cover Crops	484		

Desertification: Productivity	645
Desertification: Restoration	648
Desertification: Reversal	654
Desertization: Irreversible Extension of Desert Land	657
Developing Countries: Economics of Soil Management	664
Diagnostic Horizons	668
Diffusivity: Measurement	672
Digital Soil Mapping	675
Drainage: Aeration and Trafficability	680
Drainage: Anaerobiosis	683
Drainage: Water Quality	689
Drought: Drought Resistance	693
Drought: Precipitation, Evapotranspiration, and Soil Moisture	698
Earthworms	701
Eco-Agriculture	706
Ecology: Cycling of Carbon and Nitrogen	711
Economics of Soil Management: U.S.	716
Ecosystems: Life Attributes and Environment	719
Enchytraeidae	723
Entisols	728
Environmentally Friendly Indigenous Technologies	732
Enzymes	737
Erosion	742
Erosion Management: Hill Soils	746
Erosion Tolerance	753
Erosion: Aggregate Breakdown Mechanisms	757
Erosion: Agronomic Practices	762
Erosion: Assessment	767
Erosion: Carbon Dioxide	777
Erosion: Controlling Irrigation-Induced	782
Erosion: Crop Yield	786
Erosion: Disturbed Lands	790
Erosion: Effects on Human Life	794
Erosion: Fly-Ash Spheres as Time Marker	798
Erosion: Global Change	804
Erosion: On-Site and Off-Site Impacts	811
Erosion: Overland Research and Measurement Techniques	814
Erosion: Perennial Crop Plantations	819
Erosion: Sedimentation Control Amendment Techniques	823
Erosion: Sedimentation Control Vegetative Techniques	827
Erosion: Snowmelt	831

Erosion: Soil Conservation	834
Erosion: Water Quality	838
Erosivity and Erodibility	845

Volume II

Ethics	848
Eutrophication	855
Evaporation	858
Evolution: Soil Seed Bank	868
Fauna	871
Fauna and Microflora: Macrofauna	877
Fertility Decline: Definitions and Assessment	880
Fertility: Environmentally Compatible Management	884
Fertility: Evaluation Systems	890
Fertility: Low Input and Traditional Maintenance Methods	894
Fertility: North China Plains	899
Fertilizers: Application Methods	903
Fertilizers: Leaching Losses	907
Fertilizers: Mineral	910
Fertilizers: Organic	916
Fertilizers: Types and Formulations	919
Fertilizers: Urban Waste and Sludges	923
Fire and Soils	926
Fluid Flow: Modeling	930
Food Crops: Zinc and Iron Biofortification	933
Food Security and Soil Water Management	939
Food Security: Agricultural Productivity and Resources	942
Food Security: Soil Degradation, Global Scale	945
Food Security: Soil Degradation, Local and Eco-Regional Scale	950
Forest Soils	955
Forest Soils: Calcium Depletion	959
Forest Soils: Major	963
Forest Soils: Nutrient Cycling	966
Forest Soils: Properties and Site Productivity	969
Freeze and Thaw: Diurnal	973
Frozen Soils: Water Relations	976
Fungi	979
Future of Soil Science	982
Gas and Vapor Phase Transport	986
Gelisols	989
Geologic Carbon Capture and Storage	992
Geology and Soil	997
Geomorphology	1000

Volume II (*cont'd.*)

Geophysical Methods	1004	Inorganic Carbon: Composition and Formation	1199
Geostatistics	1012	Inorganic Carbon: Global Carbon Cycle	1203
GIS Integration: Remote Sensing	1017	Inorganic Carbon: Global Stocks	1206
Global Resources	1020	Inorganic Carbon: Land Use Impacts	1210
Grass Strip Hydrology	1024	Inorganic Carbon: Modeling	1213
Grasslands Soils	1029	Insect Survival: Subzero Temperature	1217
Grazing Lands: Gaseous Emissions	1034	Integrated Nutrient Management	1221
Green Revolution: China North Eastern Plains	1042	Integrated Nutrient Management: Sustainability	1227
Greenhouse Effect	1048	International Soil Reference and Information Centre (ISRIC)	1232
Growing Media	1053	International Union of Soil Sciences	1237
Gulley Erosion	1059	Ion Exchange	1241
Gypsic Soils	1064	Iron Oxides	1245
Gypsic Soils: Gypsum Formation	1068	Irrigation	1250
Hard-Setting Soils	1072	Irrigation: Efficiency	1256
Health	1076	Irrigation: Erosion	1261
Heat and Water Movement	1079	Irrigation: Historical Perspective	1264
Heat Capacity	1083	Irrigation: Soil Salinity	1269
Heat Flux	1086	Jhum: Cultivation	1273
Heavy Metals	1089	Jhum: Shifting Agriculture	1281
History of Soil Science	1093	Land Capability Analysis	1285
History of Soil Science: 1927-2000	1098	Land Capability Classification	1288
History of Soil Science: Early to Mid-20 th Century	1103	Land Evaluation	1291
Histosols	1107	Land Forms	1294
Human Culture and Soils	1112	Land Husbandry	1297
Human Dimensions: Soil Sustainability	1115	Land Restoration	1304
Human Health and Soil	1118	Land Use: Historic	1308
Human Society and Soil	1123	Land Use: Land Cover	1313
Human Society and the Planet: Soil as a Heritage	1127	Land Use: Planning and Geographic Information System (GIS)	1318
Hydraulic Conductivity	1133	Landfill Sites: Revegetation	1322
Hydrologic Cycle	1137	Landscape	1327
Hydropedology	1140	Landscape: Classification	1331
Hydrophobic and Hydrophilic Compounds	1144	Landscape: Development and Erosion	1336
Hydrophobicity	1147	Landscape: Elements	1339
Inceptisols	1151	Landscape: Relationships	1344
India: Problems and Potentialities	1157	Landslide and Landcreep	1348
Indigenous Soil Management	1164	Law and Legal Issues	1352
Indo-Gangetic Plain: Agro-Ecological Regions	1168	Law and Legal Issues: Land as Property	1355
Industrial Waste	1178	Leaching and Illuviation	1358
Infiltration: Management	1183	Liming and Lime Materials	1362
Infiltration: Properties	1187	Loess	1365
Inorganic Carbon: Analysis	1191	Loess Plateau	1370
Inorganic Carbon: Climate and Time	1195	Low-Carbon Technology: Brazilian Semiarid Ecosystems	1377
		Lower Himalayas: Soil Management	1382

Macroporosity	1388	Nutrients: Deficiency and Toxicity Symptoms . .	1585
Magnesium	1392	Nutrients: Diffusion, Bioavailability, and	
Manganese: Brazilian Soils	1396	Plant Uptake	1588
Manure Management: Gaseous Emissions	1400	Nutrients: Interactions in Soil–Plant System . .	1594
Manure: Compost and Biosolids	1406	Nutrients: Water Interactions	1597
Mapping Soil Carbon	1409	Organic Agriculture	1600
Mapping Soil Carbon: Stocks in Turkey	1412	Organic Carbon: Assessment Methods	1604
Materials and Soils	1416	Organic Carbon: Sequestration Concepts	1609
Mediterranean Agriculture: Rainfed,		Organic Carbon: Subsoil Pools	1614
Nitrogen Use Efficiency	1421	Organic Farming: China	1618
Metal–Clay Interactions	1426	Organic Fertilization: Heavy Metal Uptake . . .	1621
Microbial Biomass: Measurement	1429	Organisms and Soil Food Webs	1624
Microbial Communities	1433	Organo-Mineral Relationships	1629
Micromorphology and Soil Quality	1439	Oxisols	1635
Micronutrients and Human Health	1443	Pampas, The	1638
Mine Soils: Measuring Geogenic Carbon	1449	Pantanal, The	1643
Mine Soils: Miscanthus Plantations	1458	Paramos Soils	1648
Mine Soils: Open Cut Mine Rehabilitation . . .	1462	Particle Density	1652
Mine Soils: Reclamation and Soil Carbon		Particle Packing	1654
Sequestration	1466	Particle Shape	1659
Minerals: Amorphous	1469	Particle Size	1661
Minerals: Primary	1473	Peatlands	1668
Minerals: Secondary	1476	Pedological Modeling	1674
Minerals: Solubility	1480	Pedometrics	1677
Minirhizotron	1484	Pedotransfer Functions	1682
Mites	1488	Permafrost: Soil Temperature and Special	
Modern Civilization and Soils	1491	Problems	1687
Mulch Farming	1495	Petrocalcic Horizons	1691
Mycorrhiza of Forest Ecosystems	1502	pH	1695
Mycorrhizae: Soil Quality	1505	Phosphorus	1700
Nanofertilizers	1511	Phosphorus: Manure and Diet Modification . .	1705
Natural Resources: Interaction with Soil	1516	Phosphorus: Plant Nutrition and	
Nematodes	1518	Microorganism Contribution	1708
Nematodes and Soil	1523	Phosphorus: Spatial Speciation in	
Nexus Approach: Resource Management for		Agricultural Soils	1712
Soil Productivity	1530	Phytoremediation	1717
Nitrate Leaching Index	1535	Plant Growth: Oxygen Diffusion Rate	1725
Nitrate Leaching Management	1538	Plant Nutrients	1728
Nitrogen and its Transformations	1541	Plant Nutrients: Intensity, Quantity, and	
Nitrogen Index	1545	Buffer Power	1731
Nitrogen-15: Utilization	1551	Plant Nutrients: Sufficiency and	
Nitrous Oxide Emissions: Agricultural Soils . .	1555	Requirements	1735
Nitrous Oxide Emissions: Paddy Soils	1559	Plastic Properties	1740
Nitrous Oxide Emissions: Sources, Sinks,		Plinthite and Petroplinthite	1743
and Strategies	1562	Podzols	1747
No Till	1566	Polar Regions, The	1753
Nutrient Management	1570	Pollution	1758
Nutrient Management: Fertility and	1576	Pollution: Industrial Waste	1762

Volume II (*cont'd.*)

Pollution: Non-Point Source	1765
Pollution: Organic and Inorganic	1768
Pollution: Persistent Organic	1771
Pollution: Point Source	1776

Volume III

Porosity: Pore Size Distribution	1782
Potassium	1786
Potassium: Dynamics and Mineralogy	1791
Potassium: Nutrition in Semi-Arid Tropics	1796
Power Plants: Carbon Dioxide Capture	1798
Precision Agriculture: Engineering Aspects	1804
Precision Agriculture: Fertilization	1808
Precision Agriculture: Nutrient Cycling	1811
Precision Agriculture: Remote Sensing	1815
Productivity: Natural and Anthropogenic Disturbances	1819
Protection: Policy of the European Union	1823
Protozoa	1828
Pyrite	1831
Pyrogenic Carbon	1834
Quality	1837
Quality Index: Soil	1841
Quality: Assessment by Interpolation Techniques	1844
Quality: Carbon and Nitrogen Gases	1848
Quality: Critical Limits and Standardization	1853
Quality: Erosion	1856
Quality: Indicators	1859
Quality: Productivity	1862
Quality: Soil and Water	1866
Radiometric Survey	1872
Radionuclides	1876
Raindrop Impact and Splash Erosion	1881
Rapid Soil Analysis: Diffuse Reflectance Spectroscopy	1886
Rare Earth Elements	1891
Redox Phenomena	1896
Rehabilitation: Indicators and Monitoring	1899
Religious Aspects of Soil–Human Relationships	1903
Rendzina	1907
Resilience	1909
Resilience: Land Use and Soil Management	1913
Resilience: Quality and	1918

Resilience: Restoration	1924
Respiration	1928
Restoration Ecology	1932
Restoration: Success and Completion Criteria	1935
Rice Paddies: Methane Emissions	1940
Rice Paddies: Methane Emissions, Mitigating	1943
Rice Paddies: Methanogenesis	1947
Rice Paddies: Soil Management	1950
Rocks	1953
Root Growth: Pressure at the Soil–Root Interface	1957
Root Monitoring System	1960
Salt-Affected Soils	1965
Salt-Affected Soils: Nitrous Oxide Emissions	1969
Satellite Mapping	1972
Scaling Effects	1976
Sediment: Influence on Aquatic Life	1981
Sedimentation Control and Erosion: Engineering Techniques	1986
Seepage	1990
Selenium	1995
Serpentinic Soils	1999
Sesquioxides	2003
Shrinkage	2008
Silica: Pedogenic Accumulation	2012
Silicon and Sodium	2015
Slaking, Dispersion, and Crust Formation	2021
Social and Behavioral Sciences' Contribution to Soil Science	2026
Sodic Soils	2029
Sodic Soils: Formation and Global Distribution	2032
Sodic Soils: Humid Regions	2037
Sodic Soils: Irrigation Farming	2041
Sodic Soils: Physical Properties and Behavior	2044
Sodic Soils: Processes, Characteristics, and Classification	2048
Sodic Soils: Reclamation	2051
Sodicity: Subsoil	2056
Soil Carbon Map: Africa	2059
Soil Carbon Sequestration: Ethiopia	2066
Soil Carbon: Compositional and Isotopic Analysis	2073
Soil Carbon: Farming and Economic Aspects	2078
Soil Conservation: Incentives	2081
Soil Conservation: Strategies	2084
Soil Conservation: U.S. Historical Perspective	2086
Soil Degradation: Global Assessment	2090

Soil Enzymes	2100	Surface Management	2259
Soil Organic Matter (SOM).	2108	Surface Water: Nitrogen Enrichment.	2263
Soil Organic Matter (SOM): Accumulation	2112	Surface Water: Pollution by Nitrogen Fertilizers	2268
Soil Organic Matter (SOM): Available Water Capacity.	2117	Sustainable Agriculture: Social Aspects	2273
Soil Organic Matter (SOM): Characterization	2123	Sustainable Agriculture: Soil Quality.	2276
Soil Organic Matter (SOM): Composition and Dynamics.	2126	Sustainable Development and Sustainability	2280
Soil Organic Matter (SOM): Global Carbon Cycle	2129	Tailings: Minerals Processing Residue Rehabilitation	2286
Soil Organic Matter (SOM): Global Distribution	2133	Taxonomies: U.S. Soil Taxonomy and Indian Scenario	2291
Soil Organic Matter (SOM): Historical Concepts.	2139	Taxonomy.	2295
Soil Organic Matter (SOM): Landscape	2142	Temperature Measurement	2302
Soil Organic Matter (SOM): Management.	2147	<i>Tepetates</i> : Mexico	2305
Soil Organic Matter (SOM): Mass Spectrometry Data Analysis	2154	Termites	2309
Soil Organic Matter (SOM): Modeling	2159	<i>Terra Preta</i> (Black Earths).	2312
Soil Organic Matter (SOM): Molecular Simulations.	2166	<i>Terra Preta de Indio</i>	2316
Soil Organic Matter (SOM): Nutrient Dynamics	2172	Testing	2320
Soil Organic Matter (SOM): Solvent-Based High Resolution Mass Spectrometry	2176	Texture	2323
Soil Organic Matter (SOM): Structure and Characterization	2180	Tillage Erosion: Description and Process	2330
Soil Organic Matter (SOM): Turnover	2186	Tillage Erosion: Measurement Techniques	2333
Soluble Salts: Translocation and Accumulation	2192	Tillage Erosion: Properties and Productivity Effects	2336
Solute Transport	2194	Tillage Erosion: Relationship to Water Erosion.	2339
Spectral Mixtures: Soil–Plant	2197	Tillage Erosion: Terrace Formation.	2342
Spodosols	2200	Tillage: Gas Exchange.	2348
Strontium	2203	Tilth	2351
Structure	2207	Time and Space: Soils in	2353
Structure: Belowground Management	2209	Torrential Erosion: Himalayas.	2358
Structure: Earthworms	2212	Trace Metal Contamination.	2364
Structure: Pedological Concepts and Water Flow.	2216	Tropics and Subtropics	2369
Structure: Plant Establishment	2221	Turbations	2373
Structure: Roots	2225	Turfgrass: Protection of Soil and Water.	2377
Sub-Irrigation.	2228	Ultisols	2380
Sugarcane Fields: Harvest Systems and Residue Management	2231	Uranium Contamination	2385
Sulfate and Sulfide Minerals	2238	Urban Lands: Abandoned.	2388
Sulfur	2241	Urban Lands: and Construction Sites	2395
Sulfur: Chemical Behavior in the Rhizosphere.	2247	Urban Lands: Management.	2400
Sulfur: Nutrition	2251	Urban Wastewater for Irrigation: Heavy Metals	2407
Surface Area: Specific.	2255	Urea: Subsoiling and Controlled Release	2411
		Value of Soil to Humans	2416
		Variability.	2419
		Variability: Spatial	2428
		Variable Charge Soils: Mineralogy and Chemistry.	2432
		Vertisols	2440

Volume III (*cont'd.*)

Vertisols: Surface Crack Management	2450
Vesicular Crusts	2456
Village Bamboos	2459
Waste Rock and Overburden Dumps: Rehabilitation	2464
Wastewater: Treatment and Utilization in Irrigation	2467
Water Conservation	2472
Water Content	2475
Water Content: Passive and Active Microwave Remote Sensing	2480
Water Erosion.	2485
Water Erosion: Empirical Models	2489
Water Erosion: Hybrid Models	2494
Water Erosion: Modeling	2499
Water Erosion: Process-Based Modeling	2504
Water Infiltration	2507
Water Management: Computer Modeling	2510
Water Management: Harvesting	2515
Water Management: Use Efficiency	2518
Water Measurement: Methods	2522
Water Range: Least-Limiting	2530
Water Retention	2535
Water: Plant-Available	2541
Water-Repellent Soils	2546
Watershed Approach	2550

Watershed Management: Remote Sensing and GIS Applications	2555
Weathering	2559
Weathering: Carbon Sequestration	2563
Wetlands.	2567
Wetlands: Biodiversity	2572
Wetlands: Carbon Sequestration	2576
Wetlands: Conservation Policy	2580
Wetlands: Economic Value	2584
Wetlands: Gaseous Emissions	2589
Wetlands: Methane Emission.	2596
Wetlands: Sedimentation and Ecological Engineering	2600
Wind Erosion: Climate Change.	2603
Wind Erosion: Control Measures	2609
Wind Erosion: Environmental Effects	2614
Wind Erosion: Field Measurement	2618
Wind Erosion: Global Hot Spots	2625
Wind Erosion: Micrometeorology	2628
Wind Erosion: Modeling	2632
Wind Erosion: Principles	2636
Wind Erosion: Soil Quality and Productivity	2640
Windblown Dust.	2644
Wind-Driven Rain: Erosion.	2647
World Reference Base for Soil Resources.	2650

Encyclopedia of Soil Science, Third Edition

Editor-in-Chief

Rattan Lal

*Director, Carbon Management and Sequestration Center
The Ohio State University, Columbus, Ohio, U.S.A.*

Editorial Advisory Board

Dr. Julio Alegre Orihuela

*National Agrarian University of La Molina,
Department of Soil Science – Soil Management
with Agroforestry Systems, Peru*

Dr. Rainer Horn

*Institute of Plant Nutrition and Soil Science,
Christian Albrechts University zu Kiel,
Olshausenstr, Germany*

Victor Chude

*Soil Science Society of Nigeria,
Nigeria*

Selim Kapur

*Departments of Soil Science,
Landscape Architecture, and Agricultural
Engineering, University of Cukurova,
Adana, Turkey*

Jorge Dionisio Etchevers Barra

*Postgraduate College,
Natural Resources Institute,
Montecillo, Mexico*

Dr. Takashi Kosaki

*Japanese Society of Soil Science and
Plant Nutrition, Tokyo Metropolitan University,
Tokyo, Japan*

Alfred E. Hartemink

*International Soil Reference and
Information Centre (ISRIC),
Wageningen, the Netherlands*

Professor Alex B. McBratney

*Department of Environmental Sciences,
University of Sydney, Sydney,
New South Wales, Australia*

Jeffrey E. Herrick

*Agricultural Research Service,
U.S. Department of Agriculture (USDA, ARS),
Las Cruces, New Mexico, U.S.A.*

Dr. Nikolla P. Qafoku

*Pacific Northwest National Laboratory,
Richland, Washington, U.S.A.*

Dr. Nancy J. Hess

*Environmental Molecular Sciences Laboratory,
Pacific Northwest National Laboratory,
Richland, Washington, U.S.A.*

Cherukumalli Srinivasa Rao

*Central Research Institute for Dryland Agriculture,
Indian Council of Agricultural Research (ICAR),
Hyderabad, India*

Moses Tenywa

*Department of Soil Science,
University of Makerere,
Kampala, Uganda*

Dr. Jae E. Yang

*Department of Biological Environment,
Kangwon National University,
Chuncheon, South Korea*

Dr. Pandi Zdruli

*Land and Water Resources Management
Department, International Center for
Advanced Mediterranean Agronomic Studies
(CIHEAM) of Bari, Valenzano, Italy*

Hailin Zhang

*College of Agronomy and Biotechnology,
China Agricultural University, Beijing, China*

Contributors

- Gayatri Acharya** / *World Bank, Washington, District of Columbia, U.S.A.*
- Veronica Acosta-Martinez** / *Cropping Systems Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Lubbock, Texas, U.S.A.*
- Viacheslav I. Adamchuk** / *Bioresource Engineering, McGill University, Ste-Anne-de-Bellevue, Quebec, Canada*
- Subhendu Adhikari** / *Central Institute of Freshwater Aquaculture, Bhubaneswar, India and Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.*
- Anna María Ágústsdóttir** / *Soil Conservation Service, Hella, Iceland*
- Laj R. Ahuja** / *U.S. Department of Agriculture (USDA), Fort Collins, Colorado, U.S.A.*
- Vasant A. Akala** / *Ohio State University, Columbus, Ohio, U.S.A.*
- Erhan Akça** / *Center of Remote Sensing, Adiyaman University, Adiyaman, Turkey*
- M. Elizabeth Alexander** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Barry J. Allred** / *Soil Drainage Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Columbus, Ohio, U.S.A.*
- Åsgeir Rossebø Almås** / *Department of Environmental Sciences, Norwegian University of Life Sciences, Aas, Norway*
- Miguel Altieri** / *Department of Environmental Science, Policy and Management, University of California—Berkeley, Berkeley, California, U.S.A.*
- Ashok Alva** / *Vegetable and Forage Crop/Production Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Prosser, Washington, U.S.A.*
- James Amonette** / *Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Aziz Amoozegar** / *Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.*
- Ronald G. Amundson** / *College of Natural Resources, University of California—Berkeley, Berkeley, California, U.S.A.*
- Amity Andersen** / *Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- D.W. Anderson** / *Department of Soil Science, University of Saskatchewan, Saskatoon, Saskatchewan, Canada*
- Konstantinos M. Andreadis** / *Jet Propulsion Laboratory, California Institute of Technology, Pasadena, California, U.S.A.*
- Wim Andriess** / *Green World Research, Wageningen University, Wageningen, the Netherlands*
- Samson D. Angima** / *Oregon State University, Corvallis, Oregon, U.S.A.*
- Reza Ardakanian** / *Institute for Integrated Management of Material Fluxes and of Resource, United Nations University, Dresden, Germany*
- Andres Arnalds** / *Soil Conservation Service, Hella, Iceland*
- Olafur Arnalds** / *Agricultural Research Institute, Keldnaholt, Iceland*
- Jeffrey G. Arnold** / *Grassland Soil/Water Research Laboratory, U.S. Department of Agriculture (USDA), Temple, Texas, U.S.A.*

- Richard W. Arnold** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Washington, District of Columbia, U.S.A.*
- Paul A. Arp** / *Faculty of Forestry and Environmental Management, University of New Brunswick, Fredericton, New Brunswick, Canada*
- K. Arulmozhiselvan** / *Department of Soil Science and Agricultural Chemistry, Tamil Nadu Agricultural University, Coimbatore, India*
- Levent Atatanır** / *Department of Soil Science and Plant Nutrition, Adnan Menderes University, South Campus, Aydın, Turkey*
- Peter August** / *Department of Natural Resources Science, University of Rhode Island, Kingston, Rhode Island, U.S.A.*
- Tamara Avellán** / *Institute for Integrated Management of Material Fluxes and of Resource, United Nations University, Dresden, Germany*
- Gönül Aydın** / *Department of Soil Science and Plant Nutrition, Adnan Menderes University, South Campus, Aydın, Turkey*
- L.A. Graham Aylmore** / *Soil Science and Plant Nutrition, University of Western Australia, Nedlands, Western Australia, Australia*
- Bryon W. Bache** / *Cambridge University, Cambridge, U.K.*
- Joerg Bachmann** / *Institute for Soil Study, University of Hannover, Hannover, Germany*
- M. Bagath** / *National Institute of Animal Nutrition and Physiology, Bangalore, India*
- Zhanquo Bai** / *World Soil Information, International Soil Reference and Information Centre (ISRIC), Wageningen, the Netherlands*
- James L. Baker** / *Department of Agricultural and Biosystems Engineering, Iowa State University, Ames, Iowa, U.S.A.*
- John M. Baker** / *U.S. Department of Agriculture—Agriculture Research Service (USDA-ARS), St. Paul, Minnesota, U.S.A.*
- Leslie Louise Baker** / *University of Idaho, Moscow, Idaho, U.S.A.*
- Manas R. Banerjee** / *Research and Development Division, Brett-Young Seeds Limited, Winnipeg, Manitoba, Canada*
- Kenneth A. Barbarick** / *Colorado State University, Fort Collins, Colorado, U.S.A.*
- Nikolaos I. Barbayiannis** / *Aristotle University of Thessaloniki, Thessaloniki, Greece*
- Richard D. Bardgett** / *Department of Biological Sciences, Institute of Environmental and Natural Sciences, University of Lancaster, Lancaster, U.K.*
- N.J. Barrow** / *Mt. Claremont, Western Australia, Australia*
- C.D. Barton** / *U.S. Department of Agriculture (USDA), Aiken, South Carolina, U.S.A.*
- Christel Baum** / *Institute of Land Use, University of Rostock, Rostock, Germany*
- Thomas Baumgartl** / *Institute for Plant Nutrition and Soil Science, Kiel University, Kiel, Germany*
- Susana Bautista** / *Department of Ecology, University of Alicante, Alicante, Spain*
- Mike H. Beare** / *Canterbury Agricultural and Science Centre, New Zealand Institute for Crop and Food Research, Christchurch, New Zealand*
- Dylan E. Beaudette** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Sonora, California, U.S.A.*
- Richard W. Bell** / *School of Environmental Science, Murdoch University, Perth, Western Australia, Australia*
- Jayne Belnap** / *Southwest Biological Science Center, U.S. Geological Survey (USGS), Moab, Utah, U.S.A.*
- Hassan Ben Jelloun** / *National School of Forest Engineers, Tabriquet, Morocco*

- Joshua W. Beniston** / *Jornada Experimental Range, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Las Cruces, New Mexico, U.S.A.*
- Jorge Benitez-Vega** / *Department of Agricultural and Forestry Science and Resources, University of Cordoba, Cordoba, Spain*
- Joseph G. Benjamin** / *Central Great Plains Research Station, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Akron, Colorado, U.S.A.*
- Raghavendra Bhatta** / *National Institute of Animal Nutrition and Physiology, Bangalore, India*
- Tapas Bhattacharyya** / *Dr Balasaheb Sawant Konkan Krishi Vidyapeeth, Dapoli, India*
- Jagtar S. Bhatti** / *Canadian Forest Service, Northern Forestry Centre, Edmonton, Alberta, Canada*
- Jerry M. Bigham** / *School of Environment and Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- David E. Bignell** / *School of Biological Sciences, Queen Mary and Westfield College, University of London, London, U.K.*
- David L. Bjorneberg** / *Northwest Irrigation and Soil Research, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Kimberly, Idaho, U.S.A.*
- Frederick Paxton Cardell Blamey** / *School of Land, Crop and Food Sciences, University of Queensland, St. Lucia, Queensland, Australia*
- Humberto Blanco-Canqui** / *School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- Elke Bloem** / *Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany*
- Wouter A. Blokhuis** / *International Soil Reference and Information Centre (ISRIC), Wageningen, the Netherlands*
- Kent Bloodsworth** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Winfried E.H. Blum** / *Institute of Soil Research, University of Natural Resources and Applied Life Sciences, Vienna, Austria*
- John Boardman** / *Department of Geographical and Environmental Science, University of Cape Town, Cape Town, South Africa and Environmental Change Institute, University of Oxford, Oxford, U.K.*
- James G. Bockheim** / *University of Wisconsin, Madison, Wisconsin, U.S.A.*
- Pascal Boeckx** / *Faculty of Agricultural and Applied Biological Sciences, University of Ghent, Ghent, Belgium*
- Ben Boer** / *School of Law, University of Sydney, Sydney, New South Wales, Australia*
- Janis L. Boettinger** / *Utah State University, Logan, Utah, U.S.A.*
- Patrick J. Bohlen** / *MacArthur Agro-Ecology Research Center, Lake Placid, Florida, U.S.A.*
- Julia Boike** / *Water and Environmental Research Center, University of Alaska, Fairbanks, Alaska, U.S.A.*
- Boris Boincean** / *Department of Sustainable Farming Systems, Selectia Research Institute of Field Crops, Balti, Republic of Moldova*
- Jaime Boixadera** / *Department of Agriculture, Livestock, and Fisheries, Government of Catalunya, Lleida, Spain*
- N.S. Bolan** / *Institute of Natural Resources, Soil and Earth Sciences, Massey University, Palmerston North, New Zealand*

- Michael J. Borda** / *Department of Plant and Soil Sciences, University of Delaware, Newark, Delaware, U.S.A.*
- Ricardo De Oliveira Bordonal** / *Department of Exact Sciences, College of Agricultural and Veterinarian Sciences, Sao Paulo State University, Sao Paulo, Brazil*
- Getachew Boru** / *Soil Drainage Research Unit and Department of Horticulture and Crop Science, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Columbus, Ohio, U.S.A.*
- Àngela D. Bosch Serra** / *Department of Environmental and Soil Sciences, University of Lleida, Lleida, Spain*
- Virginie Bouchard** / *School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- Biplab Brahma** / *Department of Ecology and Environmental Science, Assam University, Silchar, India*
- John T. Brake** / *Department of Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.*
- Babu S. Brar** / *Department of Soil Science, Punjab Agricultural University, Ludhiana, India*
- Eric C. Brevik** / *Department of Natural Sciences, Dickinson State University, Dickinson, North Dakota, U.S.A.*
- E.M. Bridges** / *International Soil Reference and Information Centre (ISRIC), Wageningen, the Netherlands*
- Sylvie M. Brouder** / *Department of Agronomy, Purdue University, West Lafayette, Indiana, U.S.A.*
- Mary Ann Bruns** / *Department of Ecosystem Science and Management, Pennsylvania State University, University Park, Pennsylvania, U.S.A.*
- Graeme D. Buchan** / *Centre for Soil and Environmental Quality, Lincoln University, Canterbury, New Zealand*
- Balu L. Bumb** / *International Fertilizer Development Center (IFDC), Muscle Shoals, Alabama, U.S.A.*
- Stanley W. Buol** / *Department of Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.*
- Steven N. Burch** / *University of Maryland, College Park, Maryland, U.S.A.*
- Wolfgang Burghardt** / *Soil Technology, University of Essen, Essen, Germany*
- Ellen Burnes** / *MWH Americas, Inc., Sacramento, California, U.S.A.*
- C. Lee Burras** / *Department of Agronomy, Iowa State University, Ames, Iowa, U.S.A.*
- Robert H. Burris** / *Department of Biochemistry, College of Agriculture and Life Sciences, University of Wisconsin, Madison, Wisconsin, U.S.A.*
- David Burrow** / *Agriculture Victoria, Tartura, Victoria, Australia*
- Rebecca Burt** / *National Soil Survey Laboratory, U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Lincoln, Nebraska, U.S.A.*
- Tim P. Burt** / *Department of Geography, University of Durham, Durham, U.K.*
- Alan Busacca** / *Department of Crop and Soil Sciences, Washington State University, Pullman, Washington, U.S.A.*
- Daniel E. Buschiazio** / *La Pampa University and National Institute of Agricultural Technology, Buenos Aires, Argentina*
- Frederick H. Buttel** / *Department of Rural Sociology, University of Wisconsin, Madison, Wisconsin, U.S.A.*

- Peter Buurman** / *Wageningen University, Wageningen, the Netherlands*
- Huanjie Cai** / *Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China*
- Cynthia A. Cambardella** / *National Soil Tilth Laboratory, Ames, Iowa, U.S.A.*
- Keith C. Cameron** / *Soil and Physical Science Group, Lincoln University, Canterbury, New Zealand*
- Iain Campbell** / *Land and Soil Consultancy Services, Nelson, New Zealand*
- Longxi Cao** / *Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China*
- Evaldo Luis Cardoso** / *Brazilian Corporation of Agricultural Research (EMBRAPA), Corumba, Brazil*
- Anne E. Carey** / *Ohio State University, Columbus, Ohio, U.S.A.*
- Jean Caron** / *Soil Science and Agrifood Engineering Department, Laval University, Ste-Foy, Quebec, Canada*
- Timothy R. Carr** / *Department of Geology and Geography, West Virginia University, Morgantown, West Virginia, U.S.A.*
- Florence Carré** / *Land Management and Natural Hazards Unit, Institute for Environment and Sustainability, Ispra, Italy*
- Martin R. Carter** / *Agriculture and Agri-Food Canada, Charlottetown, Prince Edward Island, Canada*
- Elemar Antonino Cassol** / *Department of Soils, Federal University of Rio Grande do Sul (UFRGS), Porto Alegre, Brazil*
- Delia C. Catacutan** / *Global Research Program, World Agroforestry Center, Malaybalay City, the Philippines*
- Paul B. Cavers** / *Plant Sciences, University of Western Ontario, London, Ontario, Canada*
- Artemi Cerdà** / *Department of Geography, University of Valencia, València, Spain*
- Carlos C. Cerri** / *University of Sao Paulo, Sao Paulo, Brazil*
- Abad Chabbi** / *National Institute of Agronomic Research (INRA) and Biodiversity (ACBB), Lusignan, France*
- Gilbert Yuk-sing Chan** / *Department of Applied Biology and Chemical Technology, Hong Kong Polytechnic University, Hong Kong, China*
- Kwong Yin Chan** / *Organic Wastes Recycling Unit, New South Wales Department of Primary Industries, Richmond, New South Wales, Australia*
- David Chandler** / *Plant, Soils and Biometeorology Department, Utah State University, Logan, Utah, U.S.A.*
- S. Chattaraj** / *National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), Nagpur, India*
- S. Chatterji** / *National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), Nagpur, India*
- Yona Chen** / *Department of Soil and Water Sciences, Faculty of Agricultural, Food and Environmental Quality Sciences, Hebrew University of Jerusalem, Rehovot, Israel*
- Claire Chenu** / *National Institute of Agronomic Research (INRA), Versailles, France*
- Michael Cherlet** / *European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy*
- Serge S. Chernyanskii** / *Moscow State University, Moscow, Russia*
- Sung Ju Cho** / *Department of Global Cooperation Research, Korea Rural Economic Institute, Naju-si, Korea*

- Jon Chorover** / *Department of Soil, Water and Environmental Sciences, University of Arizona, Tucson, Arizona, U.S.A.*
- Torben Røjle Christensen** / *Climate Impacts Group, Department of Ecology, Lund University, Lund, Sweden*
- Rosalie Chu** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Jock Churchman** / *Land and Water, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Adelaide, South Australia, Australia*
- Ahmet Çilek** / *Department of Landscape Architecture, University of Cukurova, Adana, Turkey*
- Jack Cline** / *Ohio State University, Columbus, Ohio, U.S.A.*
- Matthew Cochran** / *School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- Bruce Cockroft** / *Research Consultant, Kialla, Victoria, Australia*
- Neroli Pedro Cogo** / *Federal University of Rio Grande do Sul, Porto Alegre, Brazil*
- Janet Coit** / *Rhode Island Department of Environmental Management, Providence, Rhode Island, U.S.A.*
- Lisa Cole** / *Department of Biological Sciences, Institute of Environmental and Natural Sciences, University of Lancaster, Lancaster, U.K.*
- David C. Coleman** / *Institute of Ecology, University of Georgia, Athens, Georgia, U.S.A.*
- Nicholas Comerford** / *North Florida Research and Education Center, University of Florida, Quincy, Florida, U.S.A.*
- Robin D. Connolly** / *URS Pty. Ltd., East Perth, Western Australia, Australia*
- Mark K. Conyers** / *New South Wales Department of Primary Industries, Wagga Wagga Agricultural Institute, Wagga Wagga, New South Wales, Australia*
- Mariana Marotti Corradi** / *Department of Exact Sciences, College of Agricultural and Veterinarian Sciences, Sao Paulo State University, Sao Paulo, Brazil*
- Ray Correll** / *Commonwealth Scientific and Industrial Research Organisation (CSIRO), Adelaide, South Australia, Australia*
- Dennis L. Corwin** / *United States Soil Salinity Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Riverside, California, U.S.A.*
- Eric T. Craswell** / *Fenner School of Environment and Society, College of Medicine, Biology, and Environment, Australian National University, Canberra, Australian Capital Territory, Australia*
- Charles A. Cravotta III** / *Pennsylvania Water Science Center, U.S. Geological Survey (USGS), New Cumberland, Pennsylvania, U.S.A.*
- John W. Crawford** / *University of Abertay, Dundee, Scotland*
- Pierre Crosson** / *Senior Fellow and Resident Consultant, Resources for the Future, Frederick, Maryland, U.S.A.*
- Mehmet Ali Çullu** / *Department of Soil Science and Plant Nutrition, Harran University, Şanlıurfa, Turkey*
- Tony Jarbas Ferreira Cunha** / *Ministry of Agriculture, Livestock, and Food Supply, Brazilian Agricultural Research Corporation (Embrapa Semiárida), Brasília, Brazil*
- Nilton Curi** / *Federal University of Lavras, Lavras, Brazil*
- Martín Díaz-Zorita** / *Experimental Agricultural Station, National Institute of Technical Agriculture (EEA INTA), General Villegas, Argentina*
- A.C.S. da Costa** / *State University of Maringá, Maringá, Brazil*

- Alvaro Pires da Silva** / *Departamento de Solos e Nutrição de Plantas, University of Sao Paulo, Piracicaba, Brazil*
- Seth M. Dabney** / *National Sedimentation Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Oxford, Minnesota, U.S.A.*
- S.S. Dahiya** / *Chaudhary Charan Singh Haryana Agricultural University, Hisar, India*
- Ram C. Dalal** / *Department of Science, Information Technology, and Innovation, Queensland Government, Dutton Park, Queensland, Australia*
- Jacob H. Dane** / *Department of Agronomy and Soils, Auburn University, Auburn, Alabama, U.S.A.*
- Anup Das** / *Research Complex for North Eastern Hill Region, Indian Council of Agricultural Research (ICAR), Meghalaya, India*
- Ashesh Kumar Das** / *Department of Ecology and Environmental Science, Assam University, Silchar, India*
- Henk de Bakker** / *Netherlands Soil Survey Institute (StiBoKa), Wageningen, the Netherlands*
- Eduardo Barretto De Figueiredo** / *Department of Exact Sciences, College of Agricultural and Veterinarian Sciences, Sao Paulo State University, Sao Paulo, Brazil*
- Henrique de Oliveira** / *Brazilian Corporation of Agricultural Research (EMBRAPA), Corumba, Brazil*
- Eddy De Pauw** / *International Center for Agricultural Research in the Dry Areas (ICARDA), Aleppo, Syria*
- F.W.T. Penning de Vries** / *International Board of Soil Research and Management, Pretoria, South Africa*
- Wilfrida Decraemer** / *Department of Invertebrates, Royal Belgian Institute of Natural Sciences, Brussels, Belgium*
- Kathleen Delate** / *Departments of Agronomy and Horticulture, Iowa State University, Ames, Iowa, U.S.A.*
- Jorge A. Delgado** / *Soil Plant Nutrient Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.*
- Ambachew Demessie** / *College of Agriculture, Hawassa University, Awassa, Ethiopia*
- David Dent** / *Center for Sustainability Studies, Lund University, Lund, Sweden*
- M. Rifat Derici** / *Soil Science, University of Cukurova, Adana, Turkey*
- Susheelendra Desai** / *Central Research Institute for Dryland Agriculture, Indian Council of Agricultural Research (ICAR), Hyderabad, India*
- Markus Deurer** / *Institute of Soil Science, University of Hannover, Hannover, Germany*
- Anthony R. Dexter** / *Institute of Soil Science and Plant Cultivation (IUNG), Pulawy, Poland*
- G.S. Dheri** / *Department of Soil Science, Punjab Agricultural University, Ludhiana, India*
- Warren Dick** / *School of Natural Resources, Ohio State University, Wooster, Ohio, U.S.A.*
- Ian A. Dickie** / *School of Forest Resources, University of Minnesota, St. Paul, Minnesota, U.S.A.*
- M.F. Dignac** / *National Institute of Agronomic Research (INRA), National Center for Scientific Research (CNRS), Thiverval-Grignon, France*
- Peter Dillon** / *Commonwealth Scientific and Industrial Research Organisation (CSIRO), Adelaide, South Australia, Australia*
- Jinxiu Ding** / *Department of Public Economics, Xiamen University, Xiamen, China*
- Ruixia Ding** / *Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China*

- Craig Ditzler** / *National Leader for Soil Classification and Standards, Lincoln, Nebraska, U.S.A.*
- Endre Dobos** / *Department of Physical Geography and Environmental Sciences, University of Miskolc, Miskolc-Egyetemváros, Hungary*
- Vsevolod V. Dobrovolsky** / *Moscow State Pedagogical University, Moscow, Russia*
- Douglas J. Dollhopf** / *Department of Land Resources and Environmental Sciences, Montana State University, Bozeman, Montana, U.S.A.*
- Stefaan Dondeyne** / *Earth and Environmental Science, University of Leuven, Leuven, Belgium*
- Jim Doolittle** / *National Soil Survey Center, U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Newtown Square, Pennsylvania, U.S.A.*
- John W. Doran** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), University of Nebraska, Lincoln, Nebraska, U.S.A.*
- Malcolm Douglas** / *Consultant, York, U.K.*
- Richard R. Drees** / *Department of Soil and Crop Sciences, Texas A&M University, College Station, Texas, U.S.A.*
- Patrick Drohan** / *Department of Crop and Soil Sciences, Pennsylvania State University, University Park, Pennsylvania, U.S.A.*
- Sjoerd W. Duiker** / *Pennsylvania State University, University Park, Pennsylvania, U.S.A.*
- Jennifer A.J. Dungait** / *Rothamsted Research, North Wyke, Okehampton, U.K.*
- Michael Duniway** / *Jornada Experimental Range, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Las Cruces, New Mexico, U.S.A.*
- Michael H. Ebinger** / *Earth and Environmental Sciences Division, Los Alamos National Laboratory, Los Alamos, New Mexico, U.S.A.*
- Bahman Eghball** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Lincoln, Nebraska, U.S.A.*
- Reza Ehsani** / *Ohio State University, Columbus, Ohio, U.S.A.*
- Bettina Eichler-Löbermann** / *Institute of Land Use, University of Rostock, Rostock, Germany*
- Anna Ekberg** / *Department of Ecology, Plant Ecology, Climate Impacts Group, Lund University, Lund, Sweden*
- B.H. Ellert** / *Agriculture and Agri-Food Canada, Lethbridge, Alberta, Canada*
- William J. Elliot** / *U.S. Department of Agriculture—Forestry Service (USDA-FS), Moscow, Idaho, U.S.A.*
- Mike M. Ellsbury** / *Northern Grain Insect Research Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Brookings, South Dakota, U.S.A.*
- John Ronald Engel** / *Center for Humans and Nature, Beverly Shores, Indiana, U.S.A.*
- J.A. Entry** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Kimberly, Idaho, U.S.A.*
- Emrah Erdoğan** / *General Directorate of Agrarian Reform, Ministry of Food, Agriculture and Livestock, Ankara, Turkey*
- Gunay Erpul** / *Department of Soil Science, Ankara University, Ankara, Turkey*
- Sabit Erşahin** / *Department of Forestry Engineering, Çankırı Karatekin University, Çankırı, Turkey*
- Merve Ersoy** / *Department of Landscape Architecture, University of Cukurova, Adana, Turkey*
- Hari Eswaran** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS) (Retired), Washington, District of Columbia, U.S.A.*

- Jorge D. Etchevers** / *Postgraduate College, Natural Resources Institute, Montecillo, Mexico*
- Christien H. Ettema** / *Soil Biology Group, Wageningen Agricultural University, Wageningen, the Netherlands*
- Robert O. Evans** / *Department of Biological and Agricultural Engineering, North Carolina State University, Raleigh, North Carolina, U.S.A.*
- Anna Eynard** / *Ohio State University, Columbus, Ohio, U.S.A.*
- Delvin S. Fanning** / *Department of Environmental Science and Technology, University of Maryland, College Park, Maryland, U.S.A.*
- Norman R. Fausey** / *Soil Drainage Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Columbus, Ohio, U.S.A.*
- David Favis-Mortlock** / *Environmental Change Institute, University of Oxford, Oxford, U.K.*
- Thomas J. Feeley** / *National Energy Technology Laboratory, U.S. Department of Energy, Pittsburgh, Pennsylvania, U.S.A.*
- Christian Feller** / *Institute of Research for Development, Montpellier, France*
- Thomas E. Fenton** / *Soil Morphology and Genesis, Agronomy Department, Iowa State University, Ames, Iowa, U.S.A.*
- Martin V. Fey** / *Department of Soil Science, University of Stellenbosch, Matieland, South Africa*
- Joseph Fiksel** / *Center for Resilience, Ohio State University, Columbus, and Office of Research and Development, U.S. Environmental Protection Agency (EPA), Cincinnati, Ohio, U.S.A.*
- Amaury de Carvalho Filho** / *Soils and Crops Research Centre, Brazilian Corporation of Agricultural Research (EMBRAPA), Rio de Janeiro, Brazil*
- Antonio Pereira Filho** / *Federal University of San Francisco Valley, Juazeiro, Brazil*
- Jose Texeira Filho** / *Agricultural Engineering, University of Campinas, Campinas, Brazil*
- Thomas Fillmore** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Joshua B. Fisher** / *Jet Propulsion Laboratory, California Institute of Technology, Pasadena, California, U.S.A.*
- Dennis C. Flanagan** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), West Lafayette, Indiana, U.S.A.*
- Jurgen Fleckenstein** / *Institute of Plant Nutrients and Soil Science, Federal Agricultural Research Centre (FAL), Braunschweig, Germany*
- Gerald N. Flerchinger** / *Northwest Watershed Research Center, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Boise, Idaho, U.S.A.*
- Ronald F. Follett** / *Soil Plant Nutrient Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.*
- John E. Foss** / *University of Tennessee, Knoxville, Tennessee, U.S.A.*
- Neil W. Foster** / *Natural Resources Canada, Sault Ste. Marie, Ontario, Canada*
- Timothy Fout** / *National Energy Technology Laboratory, U.S. Department of Energy, Morgantown, West Virginia, U.S.A.*
- Thomas Fox** / *Forest Resources and Environmental Conservation, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, U.S.A.*
- Diego Antonio França de Freitas** / *Federal University of Lavras, Lavras, Brazil*
- Thomas G. Franti** / *Department of Biological Systems Engineering, University of Nebraska, Lincoln, Nebraska, U.S.A.*

- Alan J. Franzluebbers** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Watkinsville, Georgia, U.S.A.*
- Donald P. Franzmeier** / *Department of Agronomy, Purdue University, West Lafayette, Indiana, U.S.A.*
- Gary W. Frasier** / *U.S. Department of Agriculture (USDA), Fort Collins, Colorado, U.S.A.*
- Bruce E. Frazier** / *Department of Crop and Soil Sciences, Washington State University, Pullman, Washington, U.S.A.*
- D.M. Freebairn** / *Department of Natural Resources, Agricultural Production Systems Research Unit (APSRU), Toowoomba, Queensland, Australia*
- Robert S. Freeland** / *Biosystems Engineering and Soil Science Department, University of Tennessee, Knoxville, Tennessee, U.S.A.*
- John R. Freney** / *Plant Industry, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia*
- Serita Frey** / *School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- Wolfgang Friesl-Hanl** / *Health and Environment Department, Environmental Resources and Technology, Austrian Institute of Technology GmbH (AIT), Seibersdorf, Austria*
- Donald Gabriels** / *Department of Soil Management and Soil Care, Ghent University, Ghent, Belgium*
- Renata Gaj** / *Institute of Soil Science, Agricultural University, Poznan, Poland*
- John M. Galbraith** / *Crop and Soil Science, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, U.S.A.*
- Patricia M. Gallagher** / *Drexel University, Philadelphia, Pennsylvania, U.S.A.*
- Gary J. Gascho** / *Crop and Soil Sciences Department, University of Georgia, Tifton, Georgia, U.S.A.*
- Anja Gassner** / *Institute of Science and Technology, Malaysia Sabah University, Kota Kinabalu, Malaysia*
- Harold R. Geering** / *Department of Agricultural Chemistry, University of Sydney, Sydney, New South Wales, Australia*
- Alexander N. Gennadiyev** / *Faculty of Geography, Moscow State University, Moscow, Russia*
- Argyrios Gerakis** / *Laboratory of Agronomy, Faculty of Agriculture, Aristotle University of Thessaloniki, Thessaloniki, Greece*
- Maria Gerasimova** / *Faculty of Geography, Moscow Lomonosov University, Moscow, Russia*
- Martin H. Gerzabek** / *Institute of Soil Research, University of Natural Resources and Applied Life Sciences, Vienna, Austria*
- Hossein Ghadiri** / *School of Australian Environmental Studies, Griffith University, Nathan Campus, Nathan, Queensland, Australia*
- B.S. Ghuman** / *Department of Soils, Punjab Agricultural University, Ludhiana, India*
- Ramkrushna Gi** / *Research Complex for North Eastern Hill Region, Indian Council of Agricultural Research (ICAR), Meghalaya, India*
- Ken E. Giller** / *Department of Plant Sciences, Wageningen University, Wageningen, the Netherlands*
- Dale Gillette** / *U.S. Environmental Protection Agency (EPA), Research Triangle Park, North Carolina, U.S.A.*
- Daniel Gimenez** / *Department of Environmental Sciences, Rutgers University, New Brunswick, New Jersey, U.S.A.*

- Vanderlise Giongo** / *Ministry of Agriculture, Livestock, and Food Supply, Brazilian Agricultural Research Corporation (Embrapa Semiarid), Brasilia, Brazil*
- Gunnar Gissel-Nielsen** / *Head of Program Plant Ecosystems, Riso National Laboratory, Roskilde, Denmark*
- Zi-Tong Gong** / *Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China*
- Josef H. Görres** / *Department of Plant and Soil Science, University of Vermont, Burlington, Vermont, U.S.A.*
- Gerard Govers** / *Laboratory for Experimental Geomorphology, Catholic University of Leuven, Leuven, Belgium*
- Robert C. Graham** / *Soil and Water Sciences Program, University of California, Riverside, California, U.S.A.*
- Timothy C. Granata** / *Department of Civil and Environmental Engineering and Geodetic Science, Ohio State University, Columbus, Ohio, U.S.A.*
- Suzanne M. Gray** / *School of Environment and Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- Timothy R. Green** / *Great Plains Systems Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.*
- Richard S.B. Greene** / *Fenner School of Environment and Society, Australian National University, Canberra, Australian Capital Territory, Australia*
- Dennis J. Greenland** / *University of Reading, Reading, U.K.*
- David Gregg** / *Rhode Island Natural History Survey, Kingston, Rhode Island, U.S.A.*
- Ed G. Gregorich** / *Eastern Cereal and Oilseed Research Centre, Agriculture and Agri-Food Canada, Ottawa, Ontario, Canada*
- Bryan S. Griffiths** / *Soil Plant Dynamics Unit, Scottish Crop Research Institute, Dundee, Scotland*
- Katherine R. Grote** / *Geosciences and Geological and Petroleum Engineering Department, Missouri University of Science and Technology, Rolla, Missouri, U.S.A.*
- John Grove** / *University of Kentucky, Lexington, Kentucky, U.S.A.*
- Minakshi Grover** / *Central Research Institute for Dryland Agriculture, Indian Council of Agricultural Research (ICAR), Hyderabad, India*
- Nazim Gruda** / *Federal Office for Agriculture and Food and University of Bonn, Bonn, Germany*
- Noel J. Grundon** / *Land and Water, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Atherton, Queensland, Australia*
- Georg Guggenberger** / *Institute of Soil Science and Soil Geography, University of Bayreuth, Bayreuth, Germany*
- Satish C. Gupta** / *Department of Soil, Water, and Climate, University of Minnesota, St. Paul, Minnesota, U.S.A.*
- Umesh C. Gupta** / *Crops and Livestock Research Centre, Agriculture and Agri-Food Canada, Charlottetown, Prince Edward Island, Canada*
- Jose G. Guzman** / *Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.*
- Lawrence J. Hagen** / *Wind Erosion Research Unit, U.S. Department of Agriculture—Center for Grain and Animal Health Research (USDA-ARS-CGAHR), Manhattan, Kansas, U.S.A.*
- Claus Hager** / *Desert Research Foundation of Namibia, Windhoek, Namibia*
- George Hall** / *School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*

- C. Tom Hallmark** / *Department of Soil and Crop Sciences, Texas A&M University, College Station, Texas, U.S.A.*
- Ardell D. Halvorson** / *U.S. Department of Agriculture (USDA), Fort Collins, Colorado, U.S.A.*
- L.L. Hammond** / *Economic and Policy Development Program, Market Development Division, International Fertilizer Development Center (IFDC), Muscle Shoals, Alabama, U.S.A.*
- Qingfang Han** / *Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China*
- Silvia H. Haneklaus** / *Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany*
- Ian Hannam** / *Centre for Natural Resources, Department of Infrastructure, Planning and Natural Resources, Sydney, New South Wales, Australia*
- Jon Harbor** / *Department of Earth and Atmospheric Sciences, Purdue University, West Lafayette, Indiana, U.S.A.*
- Robin F. Harris** / *Department of Soil Science, University of Wisconsin, Madison, Wisconsin, U.S.A.*
- Ronny D. Harris** / *Earth and Environmental Sciences Division, Los Alamos National Laboratory, Los Alamos, New Mexico, U.S.A.*
- Alfred E. Hartemink** / *International Soil Reference and Information Centre (ISRIC), Wageningen, the Netherlands*
- Jerry L. Hatfield** / *National Laboratory for Agriculture and the Environment, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.*
- Stefan Hauser** / *Agronomy, International Institute of Tropical Agriculture, Ibadan, Nigeria*
- John L. Havlin** / *Department of Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.*
- Kris M. Havstad** / *Jornada Experimental Range, U.S. Department of Agriculture (USDA), Las Cruces, New Mexico, U.S.A.*
- Xiubin He** / *Institute of Mountain Hazards and Environment, Chinese Academy of Sciences, Chengdu, China*
- Alan D. Hecht** / *Office of Research and Development, U.S. Environmental Protection Agency (EPA), Washington, District of Columbia, U.S.A.*
- J. Douglas Helms** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Washington, District of Columbia, U.S.A.*
- Robert Hendershot** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Lancaster, Ohio, U.S.A.*
- Juan Herrero** / *Department of Soils and Irrigation, Laboratory for Agronomy and the Environment (DGA-CSIC), Zaragoza, Spain*
- David Francis Herridge** / *Environmental and Rural Sciences, University of New England, Armidale, New South Wales, Australia*
- Karl Herweg** / *Centre for Development and Environment (CDE), Institute of Geography, University of Bern, Bern, Switzerland*
- Nancy J. Hess** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Dean Hesterberg** / *Department of Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.*

- Irene L. Heuser** / *Department of International Affairs, State Chancellery of the State of Brandenburg, Berlin, Germany*
- A.E. Hewitt** / *Landcare Research, Christchurch, New Zealand*
- Claudia Hidalgo** / *Postgraduate College, Natural Resources Institute, Montecillo, Mexico*
- Roland Hiederer** / *European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy*
- Daniel Hillel** / *Center for Climate Systems Research, Columbia University, New York, New York, U.S.A.*
- Philippe Hinsinger** / *National Institute of Agronomic Research (INRA), Montpellier, France*
- Daniel Hirmas** / *Department of Geography and Atmospheric Science, University of Kansas, Lawrence, Kansas, U.S.A.*
- Kyle D. Hoagland** / *School of Natural Resources, University of Nebraska, Lincoln, Nebraska, U.S.A.*
- Richard Hobbs** / *Murdoch University, Murdoch, Western Australia, Australia*
- John A. Holt** / *James Cook University, Townsville, Queensland, Australia*
- Steve P. Hopkin** / *Division of Zoology, School of Animal and Microbial Sciences, Reading, U.K.*
- David W. Hopkins** / *Royal Agricultural University, Cirencester, U.K.*
- Jeffrey W. Hopkins** / *Resource Economics Division, U.S. Department of Agriculture—Economic Research Service (USDA-ERS), Washington, District of Columbia, U.S.A.*
- Jan W. Hopmans** / *Department of Land, Air and Water Resources, Hydrology, University of California—Davis, Davis, California, U.S.A.*
- Dave J. Horne** / *Massey University, Palmerston North, New Zealand*
- Lloyd R. Hossner** / *Soil and Crop Sciences Department, Texas A&M University, College Station, Texas, U.S.A.*
- Terry A. Howell** / *Conservation and Production Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Bushland, Texas, U.S.A.*
- Xiaotao Hu** / *Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China*
- Zhengyi Hu** / *Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China*
- Chihua Huang** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), West Lafayette, Indiana, U.S.A.*
- Pei Huang** / *Aurora Energy Research, Oxford, U.K.*
- J. Herbert Huddleston** / *Department of Soil Science, Oregon State University, Corvallis, Oregon, U.S.A.*
- Alfredo R. Huete** / *Department of Soil, Water and Environmental Sciences, University of Arizona, Tucson, Arizona, U.S.A.*
- Jeffrey C. Hughes** / *Soil Science, School of Environmental Sciences, University of KwaZulu-Natal, Scottsville, South Africa*
- Tom G. Huntington** / *U.S. Geological Survey (USGS), Augusta, Maine, U.S.A.*
- Craig Idso** / *Center for the Study of Carbon Dioxide and Global Change, Tempe, Arizona, U.S.A.*
- Keith E. Idso** / *Center for the Study of Carbon Dioxide and Global Change, Tempe, Arizona, U.S.A.*
- Sherwood E. Idso** / *U.S. Water Conservation Laboratory, Phoenix, Arizona, U.S.A.*
- Silvia Imhoff** / *Facultad de Ciencias Agrarias, National University of the Litoral, Esperanza, Argentina*

- Samuel J. Indorante** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Du Quoin, Illinois, U.S.A.*
- Terry A. Isbell** / *Davies Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Peoria, Illinois, U.S.A.*
- Muhammad Ishaq** / *Soil and Water Testing Laboratory, Soil Fertility Research Institute, Sahiwal, Pakistan*
- K.R. Islam** / *Ohio State University South Centers, Piketon, Ohio, U.S.A.*
- R. Cesar Izaurralde** / *Battelle Pacific Northwest National Laboratory, Washington, District of Columbia, U.S.A.*
- Alexandra Izosimova** / *Saint Petersburg Agricultural Physical Research Institute, Saint Petersburg, Russia*
- Pierre A. Jacinthe** / *Indiana University—Purdue University Indianapolis, Indianapolis, Indiana, U.S.A.*
- Tom J. Jackson** / *Hydrology Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Beltsville, Maryland, U.S.A.*
- Marc Jalbert** / *Auburn University, Auburn, Alabama, U.S.A.*
- Surinder Kumar Jalota** / *Department of Soils, Punjab Agricultural University, Ludhiana, India*
- Bruce R. James** / *University of Maryland, College Park, Maryland, U.S.A.*
- Henry H. Janzen** / *Lethbridge Research Centre, Agriculture and Agri-Food Canada, Lethbridge, Alberta, Canada*
- Philip M. Jardine** / *Oak Ridge National Laboratory, Oak Ridge, Tennessee, U.S.A.*
- David Jasper** / *Centre for Land Rehabilitation, University of Western Australia, Nedlands, Western Australia, Australia*
- Julie D. Jastrow** / *Environmental Research Division, Argonne National Laboratory, Argonne, Illinois, U.S.A.*
- James W. Jawitz** / *Soil and Water Science Department, University of Florida, Gainesville, Florida, U.S.A.*
- María Jesús Iglesias Briones** / *Department of Animal Ecology and Biology, University of Vigo, Vigo, Spain*
- Zhikuan Jia** / *Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China*
- Jodi Johnson-Maynard** / *University of Idaho, Moscow, Idaho, U.S.A.*
- Arwyn Jones** / *European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy*
- Srinivas K.** / *Central Research Institute for Dryland Agriculture, Indian Council of Agricultural Research (ICAR), Hyderabad, India*
- Uzi Kafkafi** / *Department of Field Crops, Faculty of Agriculture, Hebrew University of Israel, Rehovot, Israel*
- Yash P. Kalra** / *Northern Forestry Centre, Canadian Forest Service, Edmonton, Alberta, Canada*
- Nestor Kämpf** / *Federal University of Rio Grande do Sul, Porto Alegre, Brazil*
- Douglas Kane** / *Water and Environmental Research Center, University of Alaska, Fairbanks, Alaska, U.S.A.*
- Selim Kapur** / *Departments of Soil Science, Landscape Architecture, and Agricultural Engineering, University of Cukurova, Adana, Turkey*
- Anastasios D. Karathanasis** / *Department of Agronomy, University of Kentucky, Lexington, Kentucky, U.S.A.*

- Douglas L. Karlen** / *National Laboratory for Agriculture and the Environment, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.*
- Lev O. Karpachevsky** / *Moscow State University, Moscow, Russia*
- Mikhail Karpachevsky** / *Faculty of Soil Science, Moscow State University, Moscow, Russia*
- Md. Abul Kashem** / *Department of Soil Science, University of Chittagong, Chittagong, Bangladesh*
- Thomas C. Kaspar** / *National Soil Tilth Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.*
- Beverly D. Kay** / *Department of Land Resource Science, University of Guelph, Guelph, Ontario, Canada*
- Zulkuf Kaya** / *Department of Soil Science, University of Cukurova, Adana, Turkey*
- Jack Keller** / *Keller Bliesner Eng. LLC, Logan, Utah, U.S.A.*
- Katherine J. Kendrick** / *Western Earthquake Hazards, U.S. Geological Survey (USGS), Pasadena, California, U.S.A.*
- Vissarion Z. Keramidas** / *Laboratory of Soil Science, School of Agriculture, Aristotle University of Thessaloniki, Thessaloniki, Greece*
- Hossein Khademi** / *Isfahan University of Technology, Isfahan, Iran*
- Partap K. Khanna** / *Commonwealth Scientific and Industrial Research Organisation (CSIRO) (Retired), Canberra, Australian Capital Territory, Australia*
- John M. Kimble** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Lincoln, Nebraska, U.S.A.*
- Kevin W. King** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Temple, Texas, U.S.A.*
- Peter I.A. Kinnell** / *School of Resource, Environmental and Heritage Sciences, University of Canberra, Holt, Australian Capital Territory, Australia*
- Gunnar Kirchhof** / *New South Wales Department of Agriculture, Tamworth, New South Wales, Australia*
- Eileen J. Kladvik** / *Agronomy Department, Purdue University, West Lafayette, Indiana, U.S.A.*
- Andreas Klik** / *Department of Water, Atmosphere and Environment, University of Natural Resources and Applied Life Sciences, Vienna, Austria*
- Patrik Klintenberg** / *Desert Research Foundation of Namibia, Windhoek, Namibia*
- Hans Klompen** / *Museum of Biological Diversity, Ohio State University, Columbus, Ohio, U.S.A.*
- Susanne Klose** / *University of California—Davis, Davis, California, U.S.A.*
- Xiangbin Kong** / *College of Resources and Environmental Science, China Agricultural University and Key Laboratory of Farmland Quality, Monitoring and Control, National Ministry of Land Resources, Beijing, China*
- Rai Kookana** / *Commonwealth Scientific and Industrial Research Organisation (CSIRO), Adelaide, South Australia, Australia*
- Jan Kopecký** / *Crop Research Institute, Prague, Czech Republic*
- Tayfun Korucu** / *Biosystems Engineering Department, Kahramanmaraş Sutcu Imam University, Kahramanmaraş, Turkey*
- Deborah A. Kozlowski** / *Edenton, North Carolina, U.S.A.*
- Sylvia L. Kratz** / *Institute of Plant Nutrients and Soil Science, Federal Agricultural Research Centre (FAL), Braunschweig, Germany*

- Helmut H. Krause** / *University of New Brunswick, Fredericton, New Brunswick, Canada*
- Lars Krogh** / *Geographical Institute, University of Copenhagen, Copenhagen, Denmark*
- William F. Kuenstler** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Fort Worth, Texas, U.S.A.*
- Sumanta Kundu** / *Central Research Institute for Dryland Agriculture, Indian Council of Agricultural Research (ICAR), Hyderabad, India*
- William P. Kustas** / *Hydrology and Remote Sensing Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Beltsville, Maryland, U.S.A.*
- Jennifer Kyle** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Newton La Scala Jr.** / *Department of Exact Sciences, College of Agricultural and Veterinarian Sciences, Sao Paulo State University, Sao Paulo, Brazil*
- John M. Laflen** / *(Retired), U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Buffalo Center, Iowa, U.S.A.*
- David A. Laird** / *Department of Agronomy, Iowa State University, Ames, Iowa, U.S.A.*
- K. Lal** / *Central Soil Salinity Research Institute, Karnal, India*
- Rattan Lal** / *Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.*
- Maria del Carmen Lamas** / *Institute of Soil Science, University of Buenos Aires, Buenos Aires, Argentina*
- Judith Lancaster** / *Desert Research Institute, Reno, Nevada, U.S.A.*
- A. Landi** / *Shahid Chamran University, Ahvaz, Iran*
- Robert J. Lascano** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Lubbock, Texas, U.S.A.*
- Alexander Laskin** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Jayanta Layek** / *Research Complex for North Eastern Hill Region, Indian Council of Agricultural Research (ICAR), Meghalaya, India*
- Yves Le Bissonnais** / *Orleans Research Center, Science of the Sun Unit, National Institute of Agronomic Research (INRA), Orleans, France*
- Brad D. Lee** / *Department of Plant and Soil Sciences, University of Kentucky, Lexington, Kentucky, U.S.A.*
- Jean-Claude Lefeuvre** / *Laboratory of the Evolution of Natural and Modified Systems, University of Rennes, Rennes, France*
- Jean-Paul Legros** / *Environment and Agronomy Department, National Institute of Agronomic Research (INRA), Montpellier, France*
- Johannes Lehmann** / *Department of Crop and Soil Science, Cornell University, Ithaca, New York, U.S.A.*
- Henry Noel LeHou  rou** / *Functional and Evolutionary Ecology Center (CEFE), National Center for Scientific Research (CNRS), Montpellier, France*
- Luiz Fernando Carvalho Leite** / *Brazilian Agricultural Research Corporation (Embrapa Mid-North), Teresina, Brazil*
- Reynald Lemke** / *Agriculture and Agri-Food Canada, Swift Current, Saskatchewan, Canada*
- Rocky Lemus** / *Mississippi State University, Starkville, Mississippi, U.S.A.*
- Narendra Kumar Lenka** / *Indian Institute of Soil Science, Bhopal, India*
- Igo F. Lepsch** / *Soil Science Department, Luiz de Queiroz College of Agriculture (USP-ESALQ), Piracicaba, Brazil*

- Renato Levien** / *Federal University of Rio Grande do Sul, Porto Alegre, Brazil*
- John Leys** / *Department of Land and Water Conservation, Center for Natural Resources, Gunnedah, New South Wales, Australia*
- Zengjia Li** / *Chinese National Engineering Research Center for Slow/Controlled Release Fertilizers, Shandong Agricultural University, Taian, China*
- Zhijun Li** / *Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China*
- Yin Liang** / *Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China*
- Holger Lilienthal** / *Institute of Plant Nutrition and Soil Science, Federal Agricultural Research Centre (FAL), Berlin, Germany*
- Henry Lin** / *Department of Crop and Soil Sciences, Pennsylvania State University, University Park, Pennsylvania, U.S.A.*
- David L. Lindbo** / *V. James Research and Extension Center, North Carolina State University, Plymouth, North Carolina, U.S.A.*
- Dennis R. Linden** / *Soil Water and Climate Department, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), St. Paul, Minnesota, U.S.A.*
- Michael J. Lindstrom** / *USDA-ARS-MWA, North Central Soil Conservation Research Laboratory, Morris, Minnesota, U.S.A.*
- John T. Litynski** / *National Energy Technology Laboratory, U.S. Department of Energy, Morgantown, West Virginia, U.S.A.*
- Ruiqiang Liu** / *Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.*
- David A. Lobb** / *Department of Soil Science, University of Manitoba, Winnipeg, Manitoba, Canada*
- F. Javier López-Bellido** / *University of Castilla–La Mancha, Ciudad Real, Spain*
- Luis López-Bellido** / *Department of Agricultural and Forestry Science and Resources, University of Cordoba, Cordoba, Spain*
- Rafael J. López-Bellido** / *Department of Agroforestry, University of Huelva, Huelva, Spain*
- Klaus Lorenz** / *Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.*
- Nicola Lorenz** / *Ohio State University, Columbus, Ohio, U.S.A.*
- Jau-Yau Lu** / *Department of Civil Engineering, National Chung Hsing University, Taichung, Taiwan*
- John Ludwig** / *Ecosystem Sciences, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Winnellie, Northern Territory, Australia*
- A. Lufafa** / *Makerere University, Kampala, Uganda*
- Sven B. Lundstedt** / *School of Public Policy and Management, Ohio State University, Columbus, Ohio, U.S.A.*
- William Berry Lyons** / *Ohio State University, Columbus, Ohio, U.S.A.*
- Jian Feng Ma** / *Research Institute for Bioresources, Okayama University, Kurashiki, Japan*
- Liwang Ma** / *Great Plains Systems Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.*
- J.A. MacLeod** / *Agriculture and Agri-Food Canada, Charlottetown, Prince Edward Island, Canada*
- Rory O. Maguire** / *Virginia Polytechnic Institute and State University, Blacksburg, Virginia, U.S.A.*
- Michael Maher** / *Kinsealy Research Centre, Teagasc, Dublin, Ireland*
- J.G.M. Majaliwa** / *Department of Soil Science, University of Makerere, Kampala, Uganda*

- Badapmain Makdoh** / *Research Complex for North Eastern Hill Region, Indian Council of Agricultural Research (ICAR), Meghalaya, India*
- S.K. Malhotra** / *Ministry of Agriculture, Government of India, New Delhi, India*
- R.S. Malik** / *Department of Soil Science, CCS Haryana Agricultural University, Hisar, India*
- Douglas D. Malo** / *Biostress Center of Excellence, Brookings, South Dakota, U.S.A.*
- Brendan Malone** / *Department of Environmental Sciences, University of Sydney, Sydney, New South Wales, Australia*
- Asit Mandal** / *Indian Institute of Soil Science, Bhopal, India*
- Sandip Mandal** / *Research Complex for North Eastern Hill Region, Indian Council of Agricultural Research (ICAR), Meghalaya, India*
- M.C. Manna** / *Indian Institute of Soil Science, Bhopal, India*
- Paul Mapfumo** / *Department of Soil Science and Agricultural Engineering, University of Zimbabwe, Harare, Zimbabwe*
- Helaine W. Markewich** / *U.S. Geological Survey (USGS), Atlanta, Georgia, U.S.A.*
- Abraham Marmur** / *Department of Chemical Engineering, Technion-Israel Institute of Technology, Haifa, Israel*
- Jose Marques Jr.** / *Department of Soil and Fertilizers, Sao Paulo State University, Sao Paulo, Brazil*
- Jay F. Martin** / *Department of Food, Agricultural, and Biological Engineering, Ohio State University, Columbus, Ohio, U.S.A.*
- Paul Martin** / *Australian Centre for Agriculture and Law, University of New England, Armidale, New South Wales, Australia*
- José A. Martínez-Casasnovas** / *University of Lleida, Lleida, Spain*
- Christopher John Matocha** / *Agronomy Department, University of Kentucky, Lexington, Kentucky, U.S.A.*
- Douglas G. Maynard** / *Natural Resources Canada, Pacific Forestry Centre, Victoria, British Columbia, Canada*
- Galina Mazhitova** / *Komi Science Center, Russian Academy of Sciences, Syktyvkar, Russia*
- Alex B. McBratney** / *Department of Environmental Sciences, University of Sydney, Sydney, New South Wales, Australia*
- Ray McBride** / *Department of Land Resource Science, University of Guelph, Guelph, Ontario, Canada*
- Bruce A. McCarl** / *Agricultural Economics, Texas A&M University, College Station, Texas, U.S.A.*
- Gregory W. McCarty** / *Environmental Quality Laboratory, U.S. Department of Agriculture (USDA), Beltsville, Maryland, U.S.A.*
- Donald K. McCool** / *U.S. Department of Agriculture (USDA), Pullman, Washington, U.S.A.*
- Paul A. McDaniel** / *University of Idaho, Moscow, Idaho, U.S.A.*
- Leslie D. McFadden** / *Department of Earth and Planetary Sciences, University of New Mexico, Albuquerque, New Mexico, U.S.A.*
- Des McGarry** / *Natural Resource Sciences, Queensland Department of Natural Resources and Mines, Indooroopilly, Queensland, Australia*
- Sean McGinn** / *Agriculture and Agri-Food Canada, Lethbridge, Alberta, Canada*
- Douglas Robert McGuffog** / *McGuffog & Co Pty Ltd, Noosa Heads, Queensland, Australia*

- Howard G. McIlvried** / *National Energy Technology Laboratory, Science Applications International Corporation, Pittsburgh, Pennsylvania, U.S.A.*
- Kevin McInnes** / *Department of Soil and Crop Sciences, Texas A&M University, College Station, Texas, U.S.A.*
- Gregory McIsaac** / *Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.*
- David C. McKenzie** / *Precision Land Management, Orange, New South Wales, Australia*
- Ronald G. McLaren** / *Soil, Plant, and Ecological Sciences Division, Lincoln University, Canterbury, New Zealand*
- Mike J. McLaughlin** / *Land and Water, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Glen Osmond, South Australia, Australia*
- Shawn McVey** / *National Soil Survey Center, U.S. Department of Agriculture (USDA), Lincoln, Nebraska, U.S.A.*
- Mallavarapu Megharaj** / *Commonwealth Scientific and Industrial Research Organisation (CSIRO), Adelaide, South Australia, Australia*
- John J. Meisinger** / *U.S. Department of Agriculture (USDA), Beltsville, Maryland, U.S.A.*
- Michael D. Melville** / *University of New South Wales, Sydney, New South Wales, Australia*
- Neal William Menzies** / *School of Land, Crop and Food Sciences, University of Queensland, St. Lucia, Queensland, Australia*
- Ahmet R. Mermut** / *Department of Soil Science, University of Saskatchewan, Saskatoon, Saskatchewan, Canada*
- Clifton W. Meyer** / *Earth and Environmental Sciences Division, Los Alamos National Laboratory, Los Alamos, New Mexico, U.S.A.*
- Yuxin Miao** / *College of Resources and Environmental Sciences, China Agricultural University, Beijing, China*
- Somayyeh Rezzaghi Miavaghi** / *Department of Soil Science and Plant Nutrition, University of Cukurova, Adana, Turkey*
- Rienk Miedema** / *Department of Soil Science and Geology, Wageningen University, Wageningen, the Netherlands*
- Fred P. Miller** / *School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- A.J. Mills** / *Department of Soil Science, University of Stellenbosch, Matieland, South Africa*
- Katsu Minami** / *Environmental Resources Division, National Institute of Agro-Environmental Sciences, Tsukuba, Japan*
- Budiman Minasny** / *Department of Environmental Sciences, University of Sydney, Sydney, New South Wales, Australia*
- Douglas W. Ming** / *Johnson Space Center, National Aeronautics and Space Administration (NASA), Houston, Texas, U.S.A.*
- P.S. Minhas** / *Punjab Agricultural University, Ludhiana, India*
- Rabi K. Misra** / *Faculty of Science, University of the Sunshine Coast, Sunshine Coast, Queensland, Australia*
- Mara Regina Moitinho** / *Department of Exact Sciences, College of Agricultural and Veterinarian Sciences, Sao Paulo State University, Sao Paulo, Brazil*
- Delbert Mokma** / *Department of Crop and Soil Sciences, Michigan State University, East Lansing, Michigan, U.S.A.*
- H. Curtis Monger** / *Department of Agronomy and Horticulture, New Mexico State University, Las Cruces, New Mexico, U.S.A.*

- Luca Montanarella** / *European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy*
- Roberta Bottino Montolar Sparovek** / *Luiz de Queiroz College of Agriculture (ESALQ) Piracicaba, University of Sao Paulo, Piracicaba, Brazil*
- Lynn E. Moody** / *Earth and Soil Sciences Department, California Polytechnic State University, San Luis Obispo, California, U.S.A.*
- James Moran** / *Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Lois Wright Morton** / *Department of Sociology, College of Agriculture and Life Sciences, Iowa State University, Ames, Iowa, U.S.A.*
- David A. Mouat** / *Division of Earth and Ecosystem Sciences, Desert Research Institute, Reno, Nevada, U.S.A.*
- Atanu Mukherjee** / *Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.*
- Jan Mulder** / *Department of Plant and Environmental Sciences, Norwegian University of Life Sciences (UMB), Aas, Norway*
- Verónica Muñoz-Romero** / *Department of Agriculture and Forestry Science and Resources, University of Cordoba, Cordoba, Spain*
- Brian Murphy** / *New South Wales Department of Land and Water Conservation, Cowra, New South Wales, Australia*
- James A. Murphy** / *Department of Plant Science, Rutgers University, New Brunswick, New Jersey, U.S.A.*
- Stephanie L. Murphy** / *Soil Testing Laboratory, Rutgers University, New Brunswick, New Jersey, U.S.A.*
- S.K. Nag** / *Fishery Resource and Environmental Management Division, Central Inland Fisheries Research Institute, Barrackpore, Indian Council of Agricultural Research (ICAR), Kolkata, India*
- Takanori Nagano** / *Graduate School of Agricultural Science, Kyoto University, Kobe, Japan*
- Ravendra Naidu** / *Commonwealth Scientific and Industrial Research Organisation (CSIRO), Adelaide, South Australia, Australia*
- Ravi Naidu** / *Global Centre for Environmental Remediation (GCER), University of South Australia, Callaghan, New South Wales, Australia*
- Toru Nakajima** / *Department of Agriculture, Meiji University, Kawasaki, Japan*
- M.R. Nanni** / *State University of Maringa, Maringa, Brazil*
- Ram Phal Narwal** / *Ministry of Agriculture, Government of India, New Delhi, India*
- Ed Nater** / *Department of Soil, Water and Climate, University of Minnesota, St. Paul, Minnesota, U.S.A.*
- Arun Jyoti Nath** / *Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A. and Ecology and Environmental Science, Assam University, Silchar, India*
- Mark A. Nearing** / *Watershed Research Center, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Tucson, Arizona, U.S.A.*
- Junfeng Nie** / *Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China*
- John L. Nieber** / *Department of Biosystems and Agricultural Engineering, University of Minnesota, St. Paul, Minnesota, U.S.A.*
- David C. Nielsen** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Akron, Colorado, U.S.A.*

- Niels Erik Nielsen** / *Plant Nutrition and Soil Fertility Lab, Department of Agricultural Sciences, Royal Veterinary and Agricultural University, Frederiksberg, Denmark*
- Tangyuan Ning** / *Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A. and Chinese National Engineering Research Center for Slow/Controlled Release Fertilizers, Shandong Agricultural University, Taian, China*
- Todd Nissen** / *University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.*
- Egide Nizeyimana** / *Department of Agronomy and Environmental Resources Research Institute, Pennsylvania State University, University Park, Pennsylvania, U.S.A.*
- Ephraim Nkonya** / *Environment and Production Technology Division, International Food Policy Research Institute (IFPRI), Washington, District of Columbia, U.S.A.*
- Andrew Noble** / *Integrated Water and Land Management and Ecosystems Program, International Center for Agricultural Research in the Dry Areas (ICARDA), Amman, Jordan*
- Lee C. Nordt** / *Department of Geology, Baylor University, Waco, Texas, U.S.A.*
- M. Lee Norfleet** / *Natural Resources Conservation Service, U.S. Department of Agriculture (USDA-NRCS), Temple, Texas, U.S.A.*
- Lindsey Norgrove** / *Department of Environmental Sciences, University of Basel, Basel, Switzerland*
- Charlotte E. Norris** / *Department of Renewable Resources, University of Alberta, Edmonton, Alberta, Canada*
- Stephen Nortcliff** / *Department of Soil Science, University of Reading, Reading, U.K.*
- L. Darrell Norton** / *National Soil Erosion Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), West Lafayette, Indiana, U.S.A.*
- Jeffrey M. Novak** / *Coastal Plain Soil, Water and Plant Conservation Research Center, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Florence, South Carolina, U.S.A.*
- Tarcisius Nyobe** / *Institute of Agricultural Research for Development, Nkolbisson, Cameroon*
- Vincent Obade** / *School of Environment and Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- R.D. O'Brien** / *Department of Soil Science, University of Stellenbosch, Matieland, South Africa*
- Gudny Okkenhaug** / *Norwegian Geotechnical Institute (NGI), Oslo, Norway*
- Arne E. Olsen** / *University of Florida, Gainesville, Florida, U.S.A.*
- Carolyn G. Olson** / *National Leader Soil Survey Investigations, Lincoln, Nebraska, U.S.A.*
- Kenneth R. Olson** / *Department of Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.*
- Ken A. Olsson** / *Tatura, Victoria, Australia*
- Jacob Opadeyi** / *Department of Surveying and Land Information, University of West Indies, Saint Augustine, Trinidad and Tobago*
- Ibrahim Ortaş** / *Department of Soil Science and Plant Nutrition, University of Cukurova, Adana, Turkey*
- Khan Towhid Osman** / *Department of Soil Science, University of Chittagong, Chittagong, Bangladesh*
- D.L. Oswalt** / *(Retired), Plymouth, Michigan, U.S.A.*
- Adam O'Toole** / *Climate and Environment Division, Norwegian Institute of Bioeconomy Research (NIBIO), Aas, Norway*
- Yakov A. Pachepsky** / *Hydrology Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Beltsville, Maryland, U.S.A.*

- Eswaran Padmanabhan** / *Geoscience, University Teknologi Petronas, Bandar Seri Iskandar, Malaysia*
- John Page** / *Bond University, Gold Coast, Queensland, Australia*
- Stefano Pagiola** / *Environment Department, World Bank, Washington, District of Columbia, U.S.A.*
- Dilip Kumar Pal** / *Division of Soil Resource Studies, National Bureau of Soil Survey and Land Use Planning, Nagpur, India*
- Alan Rodrigo Panosso** / *Department of Mathematics, College of Engineering of Ilha Solteira, Sao Paulo State University, Sao Paulo, Brazil*
- Kerstin Panten** / *Institute of Plant Nutrition and Soil Science, Federal Agricultural Research Centre (FAL), Berlin, Germany*
- P. Paramasivam** / *Tamil Nadu Agricultural University, Coimbatore, India*
- Soojin J. Park** / *Department of Geography, Seoul National University, Seoul, Korea*
- David R. Parker** / *Department of Soil and Environmental Sciences, University of California—Riverside, Riverside, California, U.S.A.*
- Ljiljana Paša-Tolić** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- N.G. Patil** / *National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), Nagpur, India*
- A.K. Patra** / *Indian Institute of Soil Science, Bhopal, India*
- Ashok K. Patra** / *Indian Council of Agricultural Research, Indian Institute of Soil Science, Bhopal, India*
- Keith Paustian** / *Natural Resources Ecology Lab, Colorado State University, Fort Collins, Colorado, U.S.A.*
- Judith F. Pedler** / *Department of Environmental Sciences, University of California—Riverside, Riverside, California, U.S.A.*
- Mark B. Peoples** / *Plant Industry, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia*
- Edmund Perfect** / *Department of Agronomy, University of Kentucky, Lexington, Kentucky, U.S.A.*
- Gary W. Petersen** / *Department of Agronomy, College of Agricultural Sciences, Pennsylvania State University, University Park, Pennsylvania, U.S.A.*
- Ian R. Phillips** / *School of Environmental Engineering, Griffith University, Nathan, Queensland, Australia*
- Joseph L. Pikul** / *Northern Grain Insect Research Lab (NGIRL), U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Brookings, South Dakota, U.S.A.*
- David Pimentel** / *Department of Entomology, Cornell University, Ithaca, New York, U.S.A.*
- Alain F. Plante** / *Science of the Sun Unit, National Institute of Agronomic Research (INRA), Versailles, France*
- Sean I. Plasynski** / *National Energy Technology Laboratory, U.S. Department of Energy, Pittsburgh, Pennsylvania, U.S.A.*
- Peter W. Plumstead** / *North Carolina State University, Raleigh, North Carolina, U.S.A.*
- Rosa M. Poch Claret** / *University of Lleida, Lleida, Spain*
- Pascal Podwojewski** / *Institute for Research and Development (IRD), Hanoi, Vietnam*
- Wilfred M. Post** / *Oak Ridge National Laboratory, Oak Ridge, Tennessee, U.S.A.*
- Jérôme Poulénard** / *Sciences of the Sun Laboratory, Interdisciplinary Scientific Center of Montagne, University of Savoie, Savoy, France*

- Munoo Prasad** / *Bord na Móna Horticulture, Co., Kildare, Ireland*
- Rajendra Prasad** / *Division of Agronomy, Indian Council of Agricultural Research (ICAR), New Delhi, India*
- C. Prat** / *Research Institute for Development (IRD), Montpellier, France*
- Franco Previtali** / *Department of Environmental Science, University of Milan—Bicocca, Milan, Italy*
- Odo Primavesi** / *Brazilian Corporation of Agricultural Research (EMBRAPA), Sao Paulo, Brazil*
- Giorgio Prosdocimi Gianquinto** / *Department Agricultural Sciences, University of Bologna, Bologna, Italy*
- Pascale Puget** / *School of Engineering and Agriculture (ESITPA), Rouen, France*
- Nikolla P. Qafoku** / *Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- P. Quantin** / *Research Institute for Development (IRD), Dijon, France*
- Sylvie A. Quideau** / *Department of Renewable Resources, University of Alberta, Edmonton, Alberta, Canada*
- Timothy A. Quine** / *School of Geography and Archaeology, University of Exeter, Exeter, U.K.*
- Robert Quirk** / *Private Consultant and Sugar Cane Farmer, Duranbah, New South Wales, Australia*
- Martin C. Rabenhorst** / *Department of Natural Resource Sciences, University of Maryland, College Park, Maryland, U.S.A.*
- Nageswararao C. Rachaputi** / *Queensland Department of Primary Industries, Kingaroy, Queensland, Australia*
- Mark Radosevich** / *University of Delaware, Newark, Delaware, U.S.A.*
- Mohammad Mahmudur Rahman** / *Global Centre for Environmental Remediation (GCER), University of South Australia, Callaghan, New South Wales, Australia*
- John Raison** / *Mullion Group, Canberra, Australian Capital Territory, Australia*
- P.S. Ramakrishnan** / *School of Environmental Sciences, Jawaharlal Nehru University, New Delhi, India*
- A. Subba Rao** / *Indian Institute of Soil Science, Bhopal, India*
- B. Krishna Rao** / *Research Center, Indian Institute of Soil and Water Conservation, Vasad, India*
- Cherukumalli Srinivasa Rao** / *Central Research Institute for Dryland Agriculture, Indian Council of Agricultural Research (ICAR), Hyderabad, India*
- Abdul Rashid** / *Pakistan Atomic Energy Commission, Islamabad, Pakistan*
- Daniel Rasse** / *Climate and Environment Division, Norwegian Institute of Bioeconomy Research (NIBIO), Aas, Norway*
- Barry G. Rawlins** / *British Geological Survey, Nottingham, U.K.*
- Walter J. Rawls** / *Hydrology Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Beltsville, Maryland, U.S.A.*
- S.K. Ray** / *National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), Nagpur, India*
- Ramón Redondo** / *Autonomous University of Madrid, Madrid, Spain*
- Randall C. Reeder** / *Food, Agricultural and Biological Engineering Department, Ohio State University, Columbus, Ohio, U.S.A.*
- D. Wayne Reeves** / *J. Phil Campbell Sr. Natural Resource Conservation Center, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Watkinsville, Georgia, U.S.A.*

- James B. Reeves III** / *Animal Manure and By-Products Laboratory, U.S. Department of Agriculture (USDA), Beltsville, Maryland, U.S.A.*
- Paul Reich** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Washington, District of Columbia, U.S.A.*
- Jose Miguel Reichert** / *Federal University of Santa Maria, Santa Maria, Brazil*
- Donald C. Reicosky** / *North Central Soil Conservation Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Morris, Minnesota, U.S.A.*
- Pichu Rengasamy** / *Department of Soil and Water Science, University of Adelaide, Waite, South Australia, Australia*
- Zdenko Rengel** / *Head, Soil Science and Plant Nutrition, University of Western Australia, Nedlands, Western Australia, Australia*
- E. Danielle Rhine** / *Department of Plant and Soil Sciences, University of Delaware, Newark, Delaware, U.S.A.*
- James D. Rhoades** / *Agricultural Salinity Consulting, Riverside, California, U.S.A.*
- Fred E. Rhoton** / *National Sedimentation Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Oxford, Mississippi, U.S.A.*
- Charles W. Rice** / *Department of Agronomy, Kansas State University, Manhattan, Kansas, U.S.A.*
- Jimmie L. Richardson** / *Department of Soil Science, North Dakota State University, Fargo, North Dakota, U.S.A.*
- Dominique Righi** / *Hydrology, Clay, Soils and Alterations, National Center for Scientific Research (CNRS), Poitiers, France*
- Joe T. Ritchie** / *Department of Crop and Soil Sciences, Michigan State University, East Lansing, Michigan, U.S.A.*
- Wayne P. Robarge** / *Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.*
- Errol Robinson** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Philippe Rochette** / *Soils and Crops Research Centre, Agriculture and Agri-Food Canada, Saint-Foy, Quebec, Canada*
- Volker Roemheld** / *Institute for Plant Nutrition, University of Hohenheim, Stuttgart, Germany*
- Helmut Rogasik** / *Leibnitz-Center for Agricultural Landscape and Land Use Research (ZALF) e.V., Berlin, Germany*
- Jutta Rogasik** / *Institute of Plant Nutrients and Soil Science, Federal Agricultural Research Centre (FAL), Berlin, Germany*
- Dennis E. Rolston** / *Land, Air, and Water Resources, University of California—Davis, Davis, California, U.S.A.*
- D.E. Romig** / *Energy, Minerals and Natural Resources Department, State of New Mexico, Santa Fe, New Mexico, U.S.A.*
- Calvin W. Rose** / *Faculty of Environmental Sciences, Griffith University, Nathan Campus, Nathan, Queensland, Australia*
- Mark W. Rosegrant** / *Environment and Production Technology Division, International Food Policy Research Institute (IFPRI), Washington, District of Columbia, U.S.A.*
- Stacey Rosen** / *U.S. Department of Agriculture—Economic Research Service (USDA-ERS), Washington, District of Columbia, U.S.A.*

- Clayton Rubec** / *Center for Environmental Stewardship and Conservation, Ottawa, Ontario, Canada*
- Alain Ruellan** / *National Superior School of Agronomy of Rennes, Montpellier, France*
- Cornelia Rumpel** / *National Institute of Agronomic Research (INRA), National Center for Scientific Research (CNRS), Thiverval-Grignon, France*
- Silke Ruppel** / *Institute of Vegetable and Ornamental Crops, Grossbeeren, Germany*
- John Ryan** / *International Center for Agricultural Research in the Dry Areas (ICARDA), Aleppo, Syria*
- Mohamed Sabir** / *National School of Forest Engineers, Tabriquet, Morocco*
- Paul G. Saffigna** / *Tropical Forestry, University of Queensland, Gatton, Queensland, Australia*
- Uriel N. Safriel** / *Department of Ecology, Evolution and Behavior, Hebrew University of Jerusalem and Center of Environmental Conventions, Jacob Blaustein Institutes for Desert Research, Jerusalem, Israel*
- Marketa Ságová-Marecková** / *Crop Research Institute, Prague, Czech Republic*
- Kanwar L. Sahrawat** / *International Crops Research Institute for the Semi-Arid Tropics (ICRISAT), Patancheru, India*
- Lipismita Samal** / *College of Agriculture, Odisha University of Agriculture and Technology, Bhubaneswar, India*
- David Sanders** / *World Association of Soil and Water Conservation, Bristol, U.K.*
- Tayane de Lima Santos** / *Federal University of Ceara, Fortaleza, Brazil*
- Debankur Sanyal** / *Department of Soil Science, Punjab Agricultural University, Ludhiana, India*
- Dibyendu Sarkar** / *Research Complex for North Eastern Hill Region, Indian Council of Agricultural Research (ICAR), Manipur, India*
- G.S. Saroa** / *Ohio State University, Columbus, Ohio, U.S.A.*
- Ronald L. Sass** / *Department of Ecology and Evolutionary Biology, Rice University, Houston, Texas, U.S.A.*
- Thomas J. Sauer** / *National Soil Tilth Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.*
- Mary C. Savin** / *Department of Crop, Soil, and Environmental Sciences, University of Arkansas, Fayetteville, Arkansas, U.S.A.*
- Dimitrios Savvas** / *Laboratory of Vegetable Production, Agricultural University of Athens, Athens, Greece*
- Kenneth D. Sayre** / *International Maize and Wheat Improvement Center (CIMMYT), Mexico City, Mexico*
- Peter Schad** / *Soil Science, Technical University of Munich, Freising-Weihenstephan, Germany*
- Carlos Ernesto Reynauld G. Schaefer** / *Soil Department, Federal University of Vicosa, Vicosa, Brazil*
- Brian M. Schafer** / *Agriculture and Horticulture, University of Queensland, Blue Mountain Heights, Queensland, Australia*
- Kenneth Scheffe** / *National Soil Survey Center, U.S. Department of Agriculture (USDA), Lincoln, Nebraska, U.S.A.*
- Sara J. Scherr** / *Forest Trends, Washington, District of Columbia, U.S.A.*
- David L. Schertz** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Washington, District of Columbia, U.S.A.*

- William H. Schlesinger** / *Department of Geology and Botany, Duke University, Durham, North Carolina, U.S.A.*
- Ewald Schnug** / *Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany*
- Philip Schoeneberger** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Lincoln, Nebraska, U.S.A.*
- Susanne Schroetter** / *Institute of Plant Nutrients and Soil Science, Federal Agricultural Research Centre (FAL), Braunschweig, Germany*
- Tom E. Schumacher** / *Department of Plant Science, South Dakota State University, Brookings, South Dakota, U.S.A.*
- A. Paul Schwab** / *Department of Agronomy, Purdue University, West Lafayette, Indiana, U.S.A.*
- Kai Schwärzel** / *Institute for Integrated Management of Material Fluxes and of Resource, United Nations University, Dresden, Germany*
- H.D. Scott** / *Center for Agribusiness and Environmental Policy, Mount Olive College, Mount Olive, North Carolina, U.S.A.*
- Mary Seely** / *Desert Research Foundation of Namibia, Windhoek, Namibia*
- Veerasamy Sejian** / *Animal Physiology Division, National Institute of Animal Nutrition and Physiology, Bangalore, India*
- Adam Louis Selhorst** / *Environmental Studies, Ashford University, San Diego, California, U.S.A.*
- Hamid Shahandeh** / *Texas A&M University, College Station, Texas, U.S.A.*
- A.V. Shanwal** / *Department of Soil Science, Chaudhary Charan Singh Haryana Agricultural University, Hisar, India*
- Shahla Shapouri** / *Markets and Trade Economics Division, U.S. Department of Agriculture—Economic Research Service (USDA-ERS), Washington, District of Columbia, U.S.A.*
- Brenton S. Sharratt** / *Land Management and Water Conservation Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Pullman, Washington, U.S.A.*
- Joey N. Shaw** / *Agronomy and Soils, Auburn University, Auburn, Alabama, U.S.A.*
- Francis Shaxson** / *Consultant, Dorset, U.K.*
- J. Shen** / *Department of Plant Nutrition, China Agricultural University, Beijing, China*
- Yufeng Shen** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Keith D. Shepherd** / *World Agroforestry Center, International Centre for Research in Agroforestry (ICRAF), Nairobi, Kenya*
- Hao-Jung Shieh** / *Taiwan Slope Land Disaster Prevention Association, Taichung, Taiwan*
- Martin Shipitalo** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.*
- Layla Shiva** / *Department of Agricultural Economics, Texas A&M University, College Station, Texas, U.S.A.*
- Yashbir Singh Shivay** / *Division of Agronomy, Indian Council of Agricultural Research (ICAR), New Delhi, India*
- Paul Short** / *Canadian Sphagnum Peat Moss Association, St. Albert, Alberta, Canada*
- Manoj K. Shukla** / *School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- William D. Shuster** / *National Risk Management Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency (EPA), Cincinnati, Ohio, U.S.A.*

- G.S. Sidhu** / *Ministry of Agriculture, Government of India, New Delhi, India*
- Tomas Simon** / *Department of Plant Nutrition, Crop Research Institute, Prague, Czech Republic*
- James Thomas Sims** / *Institute of Soil and Environmental Quality and Delaware Water Resources Center, University of Delaware, Newark, Delaware, U.S.A.*
- Charles F. Sing** / *Department of Human Genetics, University of Michigan, Ann Arbor, Michigan, U.S.A.*
- David B. Sing** / *Department of Human Genetics, University of Michigan, Ann Arbor, Michigan, U.S.A.*
- Michael J. Singer** / *Department of Land, Air, and Water Resources, University of California—Davis, Davis, California, U.S.A.*
- Ajay Singh** / *Department of Biology, University of Waterloo, Waterloo, Ontario, Canada*
- Bal Ram Singh** / *Department of Environmental Sciences, Norwegian University of Life Sciences, Aas, Norway*
- Muneshwar Singh** / *Indian Institute of Soil Science, Bhopal, India*
- S.K. Singh** / *National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), Nagpur, India*
- S.P. Singh** / *National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), New Delhi, India*
- Tarunjit Singh Butalia** / *Civil and Environmental Engineering, Ohio State University, Columbus, Ohio, U.S.A.*
- Nishant K. Sinha** / *Indian Council of Agricultural Research, Indian Institute of Soil Science, Bhopal, India*
- Diego S. Siqueira** / *Department of Soil and Fertilizers, Sao Paulo State University, Sao Paulo, Brazil*
- Johan Six** / *Department of Agronomy and Range Science, University of California—Davis, Davis, California, U.S.A.*
- Andrew K. Skidmore** / *Division of Agriculture, Conservation and Environment, International Institute for Aerospace Survey and Earth Sciences, Enschede, the Netherlands*
- Brian Slater** / *School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- Neil E. Smeck** / *School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.*
- Keith R.J. Smettem** / *Soil Science and Plant Nutrition, University of Western Australia, Nedlands, Western Australia, Australia*
- G.D. Smith** / *Agricultural Production Research Systems Unit, Queensland Department of Natural Resources and Mines, Indooroopilly, Queensland, Australia*
- Peter Smith** / *Institute of Biological and Environmental Sciences, School of Biological Sciences, University of Aberdeen, Aberdeen, Scotland*
- Alvin J.M. Smucker** / *Department of Crop and Soil Sciences, Michigan State University, East Lansing, Michigan, U.S.A.*
- Hwat Bing So** / *School of Agriculture and Food Sciences, University of Queensland, St. Lucia, Queensland, Australia*
- Robert E. Sojka** / *Northwest Irrigation and Soils Research Lab, U.S. Department of Agriculture (USDA), Kimberly, Idaho, U.S.A.*
- J. Somasundaram** / *Indian Council of Agricultural Research, Indian Institute of Soil Science, Bhopal, India*
- Deborah A. Soukup** / *Geomatrix Consultants, Inc., Bakersfield, California, U.S.A.*

- Graham P. Sparling** / *Landcare Research, Hamilton, New Zealand*
- Gerd Sparovek** / *Luiz de Queiroz College of Agriculture (ESALQ) Piracicaba, University of Sao Paulo, Sao Paulo, Brazil*
- Kurt Spokas** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), St. Paul, Minnesota, U.S.A.*
- Victor R. Squires** / *Adelaide University, Adelaide, South Australia, Australia*
- Rameshwar D. Srivastava** / *National Energy Technology Laboratory, Science Applications International Corporation, Pittsburgh, Pennsylvania, U.S.A.*
- Witold Stepniewski** / *Technical University of Lublin, Lublin, Poland*
- Larry D. Stetler** / *Department of Geology and Geological Engineering, South Dakota School of Mines and Technology, Rapid City, South Dakota, U.S.A.*
- B.A. Stewart** / *Department of Agricultural Sciences, West Texas A&M University, Canyon, Texas, U.S.A.*
- Michael Stocking** / *School of Developmental Studies, University of East Anglia, Norwich, U.K.*
- Mark H. Stolt** / *Department of Natural Resources Science, University of Rhode Island, Kingston, Rhode Island, U.S.A.*
- Daniel G. Strawn** / *University of Idaho, Moscow, Idaho, U.S.A.*
- Donald L. Suarez** / *U.S. Salinity Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Riverside, California, U.S.A.*
- R.U. Suganthi** / *National Institute of Animal Nutrition and Physiology, Bangalore, India*
- Matthew O. Sullivan** / *Department of Food, Agricultural and Biological Engineering, Ohio State University, Columbus, Ohio, U.S.A.*
- Jaya N. Surya** / *Regional Centre Delhi, National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), New Delhi, India*
- David Susko** / *Environmental Science, Department of Natural Sciences, University of Michigan at Dearborn, Dearborn, Michigan, U.S.A.*
- Alon Tal** / *Mitrani Department of Desert Ecology, Jacob Blaustein Institute for Desert Research, Ben Gurion University, Beer-Sheva, Israel*
- John P. Tandarich** / *Hey and Associates, Inc., Chicago, Illinois, U.S.A.*
- Jorge Tarchitzky** / *Department of Soil and Water Sciences, Faculty of Agricultural, Food and Environmental Quality Sciences, Hebrew University of Jerusalem, Rehovot, Israel*
- Charles Tarnocai** / *Agriculture and Agri-Food Canada, Ottawa, Ontario, Canada*
- Moses M. Tenywa** / *Department of Soil Science, University of Makerere, Kampala, Uganda*
- Malak M. Tfaily** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Janice E. Thies** / *Department of Crop and Soil Sciences, Cornell University, Ithaca, New York, U.S.A.*
- D.P. Thoma** / *University of Minnesota, St. Paul, Minnesota, U.S.A.*
- Grant W. Thomas** / *University of Kentucky, Lexington, Kentucky, U.S.A.*
- Robin N. Thwaites** / *School of Natural Resource Sciences, Queensland University of Technology, Brisbane, Queensland, Australia*
- Gary L. Tibke** / *U.S. Department of Agriculture—Wind Erosion Research Unit (USDA-WERU), Kansas State University, Manhattan, Kansas, U.S.A.*
- D.R. Tilley** / *Department of Environmental Science and Technology, University of Maryland, College Park, Maryland, U.S.A.*
- Ralph W. Tiner** / *U.S. Fish and Wildlife Service (FWS), Hadley, Massachusetts, U.S.A.*

- Judy Tisdall** / *Department of Agricultural Sciences, LaTrobe University, Bundoora, Victoria, Australia*
- Nikola Tolic** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- David Tongway** / *Ecosystem Sciences, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia*
- Cássio A. Tormena** / *Departamento de Agronomia, State University of Maringa, Maringa, Brazil*
- Dan Towery** / *Natural Resource Specialist, Conservation Technology Information Center, West Lafayette, Indiana, U.S.A.*
- Clinton C. Truman** / *Southeast Watershed Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Tifton, Georgia, U.S.A.*
- Jeff N. Tullberg** / *Farm Mechanisation, CTF Solutions, Gatton, Queensland, Australia*
- Ronald F. Turco** / *Department of Agronomy, Purdue University, West Lafayette, Indiana, U.S.A.*
- Yuksel Tuzel** / *Department of Horticulture, Ege University, Bornova, Turkey*
- Goro Uehara** / *Department of Agronomy and Soil Science, University of Hawaii, Honolulu, Hawaii, U.S.A.*
- April L. Ulery** / *Agronomy and Horticulture, New Mexico State University, Las Cruces, New Mexico, U.S.A.*
- Paul W. Unger** / *Conservation and Production Research Laboratory, U.S. Department of Agriculture (USDA), Bushland, Texas, U.S.A.*
- Norman Uphoff** / *Cornell University, Ithaca, New York, U.S.A.*
- David A.N. Ussiri** / *Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.*
- Rowena A. Valmonte-Santos** / *Environment and Production Technology Division, International Food Policy Research Institute (IFPRI), Washington, District of Columbia, U.S.A.*
- Oswald Van Cleemput** / *Faculty of Agricultural and Applied Biological Sciences, University of Ghent, Ghent, Belgium*
- Harold M. van Es** / *Department of Crop and Soil Sciences, Cornell University, Ithaca, New York, U.S.A.*
- M.E.F. van Mensvoort** / *Laboratory of Soil Science and Geology, Wageningen University and Research Center, Wageningen, the Netherlands*
- R. Scott Van Pelt** / *Wind Erosion and Water Conservation Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Big Spring, Texas, U.S.A.*
- Eric Van Ranst** / *Geology and Soil Science, Ghent University, Ghent, Belgium*
- Ken Van Rees** / *Department of Soil Science, University of Saskatchewan, Saskatoon, Saskatchewan, Canada*
- Jan van Schilfgaarde** / *U.S. Department of Agriculture (USDA), Fort Collins, Colorado, U.S.A.*
- George F. Vance** / *Department of Renewable Resources, University of Wyoming, Laramie, Wyoming, U.S.A.*
- Tara T. VanToai** / *Soil Drainage Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Columbus, Ohio, U.S.A.*
- Mariappan Velayutham** / *National Bureau of Soil Survey and Land Use Planning, Nagpur, India*

- Dalmo A.N. Vieira** / *National Sedimentation Laboratory, Arkansas State University, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Jonesboro, Arkansas, U.S.A.*
- Raphael A. Viscarra Rossel** / *Bruce E. Butler Laboratory, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia*
- Dale H. Vitt** / *Department of Plant Biology and Center for Ecology, Southern Illinois University, Carbondale, Illinois, U.S.A.*
- Joel T. Walker** / *Department of Food, Agricultural and Biological Engineering, Ohio State University, Columbus, Ohio, U.S.A.*
- Markus G. Walsh** / *World Agroforestry Centre (ICRAF), Kisumu, Kenya*
- Michelle Wander** / *Department of Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.*
- Dong Wang** / *Department of Soil, Water, and Climate, University of Minnesota, St. Paul, Minnesota, U.S.A.*
- Fanghao Wang** / *College of Resources and Environmental Sciences, China Agricultural University, Beijing, China*
- Xingxiang Wang** / *Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China*
- R.H. Wanjari** / *Indian Institute of Soil Science, Bhopal, India*
- Owen P. Ward** / *Department of Biology, University of Waterloo, Waterloo, Ontario, Canada*
- Sam C. Ward** / *Bicton, Western Australia, Australia*
- Eric D. Warner** / *Environmental Resources Research Institute, Pennsylvania State University, University Park, Pennsylvania, U.S.A.*
- Andrew Warren** / *Department of Geography, University College London, London, U.K.*
- Glenn A. Weesies** / *National Soil Erosion Research Laboratory, U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), West Lafayette, Indiana, U.S.A.*
- Johannes Bernhard Wehr** / *School of Land, Crop and Food Sciences, University of Queensland, St. Lucia, Queensland, Australia*
- Walter W. Wenzel** / *Institute of Soil Research, University of Natural Resources and Applied Life Sciences, Vienna, Austria*
- Larry T. West** / *National Soil Survey Center, U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS) (Retired), Fayetteville, Arkansas, U.S.A.*
- Ian White** / *Australian National University, Canberra, Australian Capital Territory, Australia*
- Walter G. Whitford** / *Jornada Experimental Range, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Las Cruces, New Mexico, U.S.A.*
- Dennis Wichelns** / *International Water Management Unit, Colombo, Sri Lanka*
- Penny E. Widdison** / *Department of Geography, University of Durham, Durham, U.K.*
- Keith D. Wiebe** / *Resource Economics Division, U.S. Department of Agriculture (USDA), Washington, District of Columbia, U.S.A.*
- Lucian Wielopolski** / *Environmental Sciences Department, Brookhaven National Laboratory, Upton, New York, U.S.A.*
- Giles F.S. Wiggs** / *Department of Geography, University of Sheffield, Sheffield, U.K.*
- Larry P. Wilding** / *Department of Soil and Crop Sciences, Texas A&M University, College Station, Texas, U.S.A.*
- Jimmy R. Williams** / *Texas A&M University, Temple, Texas, U.S.A.*

- Michael A. Wilson** / *U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Lincoln, Nebraska, U.S.A.*
- Verena Winiwarter** / *Centre for Environmental History, Institute of Social Ecology, Interdisciplinary Research and Continuing Education (IFF), Vienna, Austria*
- William E. Wolfe** / *Department of Civil and Environmental Engineering and Geodetic Sciences, Ohio State University, Columbus, Ohio, U.S.A.*
- Ming H. Wong** / *Institute for Natural Resources and Land Management, Hong Kong Baptist University, Hong Kong, China*
- J.H.M. Wösten** / *Alterra Research Institute, Wageningen University and Research Center, Wageningen, the Netherlands*
- Graeme C. Wright** / *Department of Primary Industries, Queensland Department of Primary Industries, Kingaroy, Queensland, Australia*
- S.R. Wright** / *Ohio State University South Centers, Piketon, Ohio, U.S.A.*
- Chia Chun Joseph Wu** / *National Pingtung University of Science and Technology, Neipu, Taiwan*
- Pute Wu** / *Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China*
- Douglas Wysocki** / *Research Soil Scientist, National Soil Survey Center, Lincoln, Nebraska, U.S.A.*
- Guohua Xu** / *College of Resources and Environmental Sciences, Nanjing Agricultural University, Nanjing, China*
- Ren-kou Xu** / *State Key Laboratory of Soil and Sustainable Agriculture, Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China*
- Ximeng Xu** / *Institute of Soil and Water Conservation, Northwest A&F University, Yangling, China*
- Jianfu Xue** / *College of Agronomy and Biotechnology, China Agricultural University, Beijing, China*
- Dan H. Yaalon** / *Institute of Earth Sciences, Hebrew University of Jerusalem, Jerusalem, Israel*
- R.P. Yadav** / *National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), New Delhi, India*
- Ram Prashad Yadav** / *Regional Center, National Bureau of Soil Survey and Land Use Planning, New Delhi, India*
- Kazuyuki Yagi** / *National Institute for Agro-Environmental Sciences, Tsukuba, Japan*
- Laila Yesmin** / *Research and Development Division, Brett-Young Seeds Limited, Winnipeg, Manitoba, Canada*
- Dana D. York** / *Watershed and Landscape Programs Division, Natural Resources Conservation Service, U.S. Department of Agriculture (USDA-NRCS), Washington, District of Columbia, U.S.A.*
- Alper Yorulmaz** / *Department of Soil Science and Plant Nutrition, Adnan Menderes University, South Campus, Aydın, Turkey*
- Chin-Hsien Yu** / *Institute of Development Studies, Southwestern University of Finance and Economics, Chengdu, China*
- Don F. Yule** / *Department of Natural Resources, Rural Water Use Efficiency Initiative, Indooroopilly, Queensland, Australia*
- C. William Zanner** / *University of Nebraska, Lincoln, Nebraska, U.S.A.*
- Pandi Zdruli** / *Land and Water Resources Management Department, International Center for Advanced Mediterranean Agronomic Studies (CIHEAM) of Bari, Valenzano, Italy*

- Fusuo S. Zhang** / *College of Resources and Environmental Sciences, China Agricultural University, Beijing, China*
- Fusuo Zhang** / *College of Resources and Environmental Sciences, China Agricultural University, Beijing, China*
- Hailin Zhang** / *College of Agronomy and Biotechnology, China Agricultural University, Beijing, China*
- Jianhuan Zhang** / *Ohio State University, Columbus, Ohio, U.S.A.*
- Lulu Zhang** / *Institute for Integrated Management of Material Fluxes and of Resource, United Nations University, Dresden, Germany*
- Min Zhang** / *Chinese National Engineering Research Center for Slow/Controlled Release Fertilizers, Shandong Agricultural University, Taian, China*
- Taolin Zhang** / *Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China*
- X.-C. (John) Zhang** / *Grazinglands Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), El Reno, Oklahoma, U.S.A.*
- Rui Zhao** / *Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.*
- Xin Zhao** / *College of Agronomy and Biotechnology, China Agricultural University, Beijing, China*
- Fenli Zheng** / *Institute of Soil and Water Conservation, Northwest A&F University, Yangling, China*
- Y.-G. Zhu** / *Research Centre for Eco-Environmental Sciences, Chinese Academy of Sciences, Beijing, China*
- Erik Zillmann** / *Institute of Plant Nutrition and Soil Science, Federal Agricultural Research Centre (FAL), Braunschweig, Germany*
- Yuri Lopes Zinn** / *Federal University of Lavras, Lavras, Brazil*
- Ted M. Zobeck** / *U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS) (Retired), Lubbock, Texas, U.S.A.*
- Michael A. Zoebisch** / *German Agency for International Cooperation, Tashkent, Uzbekistan*
- Yufeng Zou** / *Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China*
- Claudio Zucca** / *Integrated Water and Land Management (IWLMP) Research Program, International Center for Agricultural Research in the Dry Areas (ICARDA), Amman, Jordan*

Contents

<i>Editorial Advisory Board</i>	xiii
<i>Contributors</i>	xv
<i>Topical Table of Contents</i>	lxiii
<i>Preface</i>	lxxxiii
<i>About the Editors</i>	lxxxv

Volume I

Accounting: Emergy Analysis / Jay F. Martin and D.R. Tilley	1
Acid Mine Drainage / Jerry M. Bigham and Charles A. Cravotta III	6
Acid Rain: Nitrogen Deposition / George F. Vance	11
Acid Sulfate Soils / Delvin S. Fanning	17
Acid Sulfate Soils: Distribution and Extent / Wim Andriesse and M.E.F. van Mensvoort	20
Acid Sulfate Soils: Formation / Martin C. Rabenhorst, Delvin S. Fanning, and Steven N. Burch	26
Acid Sulfate Soils: Management / Michael D. Melville, Ian White, and Robert Quirk	31
Acid Sulfate Soils: Problems / Martin C. Rabenhorst and Delvin S. Fanning	36
Actinobacteria / Marketa Ságová-Marecková and Jan Kopecký	40
Aeration: Measurement / Robert E. Sojka and H.D. Scott	44
Aeration: Tillage Effects / Dave J. Horne and Robert E. Sojka	47
Aggregates: Tensile Strength / Humberto Blanco-Canqui and Rattan Lal	51
Aggregation / Sjoerd W. Duiker	55
Aggregation: Soil Organic Matter (SOM) and / Cynthia A. Cambardella	58
Aggregation: Structural Stability Measurement / Edmund Perfect, Martín Díaz-Zorita, and John Grove	61
Agricultural Lands: Flooding and Levee Breaches / Kenneth R. Olson and Lois Wright Morton	65
Agricultural Management: Root Growth / Susanne Schroetter, Jutta Rogasik, and Ewald Schnug	72
Agricultural Soil Carbon: Trading / Bruce A. McCarl, Sung Ju Cho, Jinxiu Ding, Pei Huang, Layla Shiva, and Chin-Hsien Yu	76
Air Permeability / Manoj K. Shukla and Rattan Lal	85
Albedo / Endre Dobos	89
Alfisols / Rienk Miedema	92
Allophanes / Leslie Louise Baker and Jodi Johnson-Maynard	97
Alpine Soils / Jérôme Poulenard and Pascal Podwojewski	101
Aluminum / Johannes Bernhard Wehr, Frederick Paxton Cardell Blamey, and Neal William Menzies	105
Amazon Basin Soils / Stanley W. Buol	111
Amendments and Ameliorants / David C. McKenzie	114
Ammonia: Volatilization from Agricultural Soils / Paul G. Saffigna and John R. Freney	117
Amoozemeter / Aziz Amoozegar	120
Anaerobic Processes / Paul A. McDaniel and Daniel G. Strawn	126
Andisols / Jon Chorover	130
Andisols: Iceland / Olafur Arnalds	134

Volume I (*cont'd.*)

Animals: Ecosystem Functioning / <i>Alan J. Franzluebbers</i>	138
Animals: Microbial Interactions and Nutrient Cycling / <i>Lisa Cole and Richard D. Bardgett</i>	142
Antimony / <i>Gudny Okkenhaug and Jan Mulder</i>	146
Ants / <i>Walter G. Whitford</i>	150
Archaeology / <i>John E. Foss</i>	154
Arid Soils / <i>H. Curtis Monger</i>	157
Arsenic: Southeast Asia / <i>Mohammad Mahmudur Rahman and Ravi Naidu</i>	161
Art and Soil / <i>Christian Feller</i>	168
Atterberg Limits / <i>Thomas Baumgartl</i>	171
Bacteria / <i>Mary Ann Bruns</i>	175
Biochar and Soil Characteristics / <i>Atanu Mukherjee and Rattan Lal</i>	184
Biochar and Soil Quality / <i>David A. Laird and Jeffrey M. Novak</i>	189
Biochar: Soil Carbon and Fertility / <i>Adam O'Toole and Daniel Rasse</i>	193
Biodegradation and Bioremediation / <i>Owen P. Ward and Ajay Singh</i>	198
Biodiversity and Sustainability / <i>Odo Primavesi</i>	205
Bioenergy Crops: Carbon Balance Assessment / <i>Rocky Lemus and Rattan Lal</i>	210
Biofuels versus Agricultural Soils / <i>David Pimentel</i>	213
Biological Crusts / <i>Jayne Belnap</i>	216
Biological Nitrogen Fixation / <i>Robert H. Burris</i>	220
Biological Nitrogen Fixation: Contributions to Agriculture / <i>Mark B. Peoples</i>	223
Biological Nitrogen Fixation: Enhancement Techniques / <i>David Francis Herridge</i>	227
Biological Nitrogen Fixation: Forms and Regulating Factors / <i>Ken E. Giller and Paul Mapfumo</i>	232
Biota / <i>Lev O. Karpachevsky and Mikhail Karpachevsky</i>	235
Biota and Disease-Suppressive Soils / <i>Nicola Lorenz</i>	239
Biotechnology / <i>Manas R. Banerjee and Laila Yesmin</i>	243
Boreal Forest / <i>Galina Mazhitova</i>	247
Boron and Molybdenum / <i>Umesh C. Gupta and J.A. MacLeod</i>	252
Brick Making: Soil Degradation / <i>G.S. Dheri, Babu S. Brar, and Debankur Sanyal</i>	255
Bulk Density / <i>Kwong Yin Chan</i>	258
Bunch Planting: Water Conservation / <i>B.A. Stewart</i>	261
Calcification / <i>Janis L. Boettinger</i>	264
Calcium / <i>Zdenko Rengel</i>	267
Carbon Assessment: Spectroscopic Methods / <i>James B. Reeves III and Gregory W. McCarty</i>	270
Carbon Sequestration: Fish Ponds / <i>Subhendu Adhikari and Rattan Lal</i>	274
Carbon Sequestration: Geologic / <i>Eswaran Padmanabhan</i>	279
Carbon Sequestration: Iceland / <i>Anna Maria Ágústsdóttir</i>	284
Carbon Sequestration: Nutrient Management / <i>M.C. Manna, Muneshwar Singh, R.H. Wanjari, Asit Mandal, and A.K. Patra</i>	288
Carbon Sequestration: Sediments / <i>Eswaran Padmanabhan</i>	294
Carbon Sequestration: Semiarid Regions of India / <i>Cherukumalli Srinivasa Rao, Sumanta Kundu, and Srinivas K.</i>	299
Carbon Sequestration: Urban Ecosystems / <i>Adam Louis Selhorst</i>	307
Carbon Sink Capacity: Soil Profile Characteristics / <i>Yuri Lopes Zinn</i>	315
Carbon: Inelastic Neutron Scattering / <i>Lucian Wielopolski</i>	319
Carbon: Laser-Induced Breakdown Spectroscopy / <i>Michael H. Ebinger, Ronny D. Harris, and Clifton W. Meyer</i>	323

Carbon: Physical Protection / Jennifer A.J. Dungait and David W. Hopkins	327
Carbonates / Douglas W. Ming	331
Carbonates: Secondary and Pedogenic / Ahmet R. Mermut and A. Landi	335
Carbonates: Water and / Michael Duniway	339
Care of Soil / Delia C. Catacutan	342
Cerrado / Igo F. Lepsch	345
Chemical Composition / Goro Uehara	348
Chemisorption / N.J. Barrow	353
Chernozems / Boris Boincean and Rattan Lal	356
Chlorine / Uzi Kafkafi and Guohua Xu	362
Classification Systems / Stephen Nortcliff	367
Classification Systems: Arab Traditional, Moroccan Case / Mohamed Sabir and Hassan Ben Jelloun	370
Classification Systems: Australian and New Zealand / Terry A. Isbell and A.E. Hewitt	375
Classification Systems: Brazilian / A.C.S. da Costa and M.R. Nanni	379
Classification Systems: Canadian / Charles Tarnocai	383
Classification Systems: Chinese / Zi-Tong Gong	389
Classification Systems: French / Jean-Paul Legros	391
Classification Systems: Historical Developments / Dan H. Yaalon	395
Classification Systems: Indian / Mariappan Velayutham, Tapas Bhattacharyya, and Dilip Kumar Pal	398
Classification Systems: Indigenous / Erhan Akça and Selim Kapur	402
Classification Systems: Major / Lars Krogh	406
Classification Systems: Netherlands / Alfred E. Hartemink and Henk de Bakker	412
Classification Systems: Russian / Maria Gerasimova	415
Classification Systems: South African / Jeffrey C. Hughes	418
Clay Minerals / C.D. Barton and Anastasios D. Karathanasis	421
Clay Minerals: Charge Properties / Christopher John Matocha	426
Clay Minerals: Weathering and Alteration of / Anastasios D. Karathanasis	430
Climate / Alexander N. Gennadiyev and Serge S. Chernyanskii	435
Climate Change: Gas Fluxes / Pascal Boeckx and Oswald Van Cleemput	438
Climate Change: Soil Carbon Sequestration / Sherwood E. Idso and Keith E. Idso	442
Cobalt and Iodine / Ronald G. McLaren	445
Collembola / Steve P. Hopkin	448
Color / Robin N. Thwaites	452
Compaction / William E. Wolfe	456
Conditioning Index / Ted M. Zobeck, M. Lee Norfleet, and Douglas L. Karlen	459
Conservation Tillage / Jack Cline and Robert Hendershot	462
Conserved Lands: Stewardship / Peter August, Janet Coit, and David Gregg	466
Consistence / Ray McBride	471
Consolidation / Tarunjit Singh Butalia	474
Controlled Traffic / Randall C. Reeder	477
Copper / Judith F. Pedler and David R. Parker	481
Cover Crops / Jorge A. Delgado, D. Wayne Reeves, and Ronald F. Follett	484
Crop Evapotranspiration: Measurement System / Huanjie Cai, Pute Wu, Zhijun Li, Xiaotao Hu, and Yufeng Zou	488
Crop Management: Seasonally Frozen Soil / Brenton S. Sharratt	492
Crop Plants: Phosphorus Availability / Bettina Eichler-Löbermann and Ewald Schnug	494

Volume I (*cont'd.*)

Crop Residue Management / <i>David L. Schertz and Dan Towery</i>	497
Crop Root Response to Temperature: Temperate Zone / <i>Thomas C. Kaspar</i>	501
Crop Rotation and Farming Systems: Temperate Zone / <i>Martin R. Carter</i>	504
Crop Rotation and Farming Systems: Tropical Smallholder Farm Systems / <i>Stefan Hauser, Lindsey Norgrove, and Tarcisius Nyobe</i>	507
Crop Rotation and Farming Systems: Tropics and Subtropics and Swelling Soils / <i>D.M. Freebairn, G.D. Smith, Robin D. Connolly, and Jeff N. Tullberg</i>	512
Crop Yield: Compaction / <i>Muhammad Ishaq and Rattan Lal</i>	516
Croplands / <i>B.A. Stewart</i>	522
Crops: Flooding Tolerance / <i>Tara T. VanToai, Getachew Boru, and Jianhuan Zhang</i>	526
Cultivation: Raised-Bed / <i>Kenneth D. Sayre</i>	529
Culture: Soil-less / <i>Nazim Gruda, Giorgio Prosdocimi Gianquinto, Yuksel Tuzel, and</i> <i>Dimitrios Savvas</i>	533
Cycling: Carbon and Nitrogen (C/N) / <i>Sylvie M. Brouder and Ronald F. Turco</i>	538
Debris Flow / <i>Jau-Yau Lu</i>	544
Degradation / <i>Rosa M. Poch Claret and José A. Martínez-Casasnovas</i>	548
Degradation: Biological / <i>Ibrahim Ortaş</i>	553
Degradation: Chemical / <i>M. Rifat Derici</i>	558
Degradation: Critical Limits and Irreversible Degradation / <i>Anthony R. Dexter and</i> <i>Michael A. Zoebisch</i>	561
Degradation: Economic Incentives and Investments / <i>Dennis Wichelns</i>	565
Degradation: Farm-Level Economic Implications / <i>Dennis Wichelns and Ellen Burnes</i>	568
Degradation: Food Aid Needs in Low-Income Countries / <i>Shahla Shapouri and Stacey Rosen</i>	571
Degradation: Food Security and Poverty / <i>Pierre Crosson</i>	574
Degradation: Greenhouse Effect / <i>Eddy De Pauw and Michael A. Zoebisch</i>	578
Degradation: Mapping by Normalized Difference Vegetation Index (NDVI) / <i>Zhanquo Bai</i> <i>and David Dent</i>	581
Degradation: Off-Farm Economic Implications / <i>Dennis Wichelns and Ellen Burnes</i>	593
Degradation: Physical / <i>Michael A. Zoebisch and Anthony R. Dexter</i>	596
Degradation: Quality and / <i>Rattan Lal</i>	602
Desertification: Biological Productivity Decline / <i>Uriel N. Safriel</i>	608
Desertification: Extent / <i>Victor R. Squires</i>	616
Desertification: Greenhouse Effect / <i>Craig Idso and Sherwood E. Idso</i>	620
Desertification: Humid Environment in Iceland / <i>Andres Arnalds</i>	623
Desertification: Impact / <i>David A. Mouat and Judith Lancaster</i>	628
Desertification: Mapping Constraints and Challenges / <i>Pandi Zdruli, Michael Cherlet, and</i> <i>Claudio Zucca</i>	633
Desertification: Prevention and Restoration / <i>Claudio Zucca, Susana Bautista, and</i> <i>Franco Previtali</i>	642
Desertification: Productivity / <i>Kris M. Havstad</i>	645
Desertification: Restoration / <i>Alon Tal</i>	648
Desertification: Reversal / <i>David Tongway and John Ludwig</i>	654
Desertization: Irreversible Extension of Desert Land / <i>Henry Noel LeHouérou</i>	657
Developing Countries: Economics of Soil Management / <i>Stefano Pagiola</i>	664
Diagnostic Horizons / <i>E.M. Bridges</i>	668
Diffusivity: Measurement / <i>Toru Nakajima</i>	672

Digital Soil Mapping / Budiman Minasny, Alex B. McBratney, and Florence Carré	675
Drainage: Aeration and Trafficability / Norman R. Fausey	680
Drainage: Anaerobiosis / Robert O. Evans	683
Drainage: Water Quality / James L. Baker	689
Drought: Drought Resistance / Graeme C. Wright and Nageswararao C. Rachaputi	693
Drought: Precipitation, Evapotranspiration, and Soil Moisture / Joshua B. Fisher and Konstantinos M. Andreadis	698
Earthworms / Patrick J. Bohlen	701
Eco-Agriculture / Norman Uphoff and Janice E. Thies	706
Ecology: Cycling of Carbon and Nitrogen / Alan J. Franzluebbers	711
Economics of Soil Management: U.S. / Jeffrey W. Hopkins	716
Ecosystems: Life Attributes and Environment / Dennis R. Linden	719
Enchytraeidae / María Jesús Iglesias Briones	723
Entisols / Larry T. West	728
Environmentally Friendly Indigenous Technologies / Selim Kapur and Erhan Akça	732
Enzymes / Veronica Acosta-Martinez and Susanne Klose	737
Erosion / Dennis C. Flanagan	742
Erosion Management: Hill Soils / Narendra Kumar Lenka and B. Krishna Rao	746
Erosion Tolerance / David L. Schertz and Mark A. Nearing	753
Erosion: Aggregate Breakdown Mechanisms / Yves Le Bissonnais	757
Erosion: Agronomic Practices / Glenn A. Weesies, David L. Schertz, and William F. Kuenstler	762
Erosion: Assessment / John Boardman	767
Erosion: Carbon Dioxide / Pierre A. Jacinthe and Rattan Lal	777
Erosion: Controlling Irrigation-Induced / Robert E. Sojka and David L. Bjorneberg	782
Erosion: Crop Yield / Michael Stocking	786
Erosion: Disturbed Lands / William J. Elliot	790
Erosion: Effects on Human Life / Neroli Pedro Cogo and Renato Levien	794
Erosion: Fly-Ash Spheres as Time Marker / Kenneth R. Olson and Alexander N. Gennadiyev	798
Erosion: Global Change / Xingxiang Wang, Taolin Zhang, Longxi Cao, and Yin Liang	804
Erosion: On-Site and Off-Site Impacts / Jerry L. Hatfield	811
Erosion: Overland Research and Measurement Techniques / Chia Chun Joseph Wu	814
Erosion: Perennial Crop Plantations / Alfred E. Hartemink	819
Erosion: Sedimentation Control Amendment Techniques / X.-C. (John) Zhang	823
Erosion: Sedimentation Control Vegetative Techniques / Samson D. Angima	827
Erosion: Snowmelt / Donald K. McCool	831
Erosion: Soil Conservation / Eric T. Craswell	834
Erosion: Water Quality / Andreas Klik	838
Erosivity and Erodibility / Peter I.A. Kinnell	845

Volume II

Ethics / John Ronald Engel	848
Eutrophication / Thomas G. Franti and Kyle D. Hoagland	855
Evaporation / William P. Kustas	858
Evolution: Soil Seed Bank / Paul B. Cavers and David Susko	868
Fauna / Mary C. Savin	871
Fauna and Microflora: Macrofauna / Judy Tisdall	877

Volume II (*cont'd.*)

Fertility Decline: Definitions and Assessment / Alfred E. Hartemink	880
Fertility: Environmentally Compatible Management / Bal Ram Singh	884
Fertility: Evaluation Systems / Ram C. Dalal and A. Subba Rao	890
Fertility: Low Input and Traditional Maintenance Methods / Miguel Altieri	894
Fertility: North China Plains / Xiangbin Kong and Rattan Lal	899
Fertilizers: Application Methods / John Ryan	903
Fertilizers: Leaching Losses / Paul G. Saffigna and Ian R. Phillips	907
Fertilizers: Mineral / L.L. Hammond and Balu L. Bumb	910
Fertilizers: Organic / Gary J. Gascho	916
Fertilizers: Types and Formulations / Douglas Robert McGuffog	919
Fertilizers: Urban Waste and Sludges / Rory O. Maguire and James Thomas Sims	923
Fire and Soils / Partap K. Khanna and John Raison	926
Fluid Flow: Modeling / John L. Nieber	930
Food Crops: Zinc and Iron Biofortification / Yashbir Singh Shivay and Rajendra Prasad	933
Food Security and Soil Water Management / Mark W. Rosegrant, Ephraim Nkonya, and Rowena A. Valmonte-Santos	939
Food Security: Agricultural Productivity and Resources / Keith D. Wiebe	942
Food Security: Soil Degradation, Global Scale / Michael A. Zebisch and Eddy De Pauw	945
Food Security: Soil Degradation, Local and Eco-Regional Scale / Sara J. Scherr	950
Forest Soils / Nicholas Comerford and Thomas Fox	955
Forest Soils: Calcium Depletion / Tom G. Huntington	959
Forest Soils: Major / Ken Van Rees	963
Forest Soils: Nutrient Cycling / Neil W. Foster and Jagtar S. Bhatti	966
Forest Soils: Properties and Site Productivity / Paul A. Arp and Helmut H. Krause	969
Freeze and Thaw: Diurnal / Joseph L. Pikul	973
Frozen Soils: Water Relations / John M. Baker	976
Fungi / Serita Frey	979
Future of Soil Science / Eric C. Brevik, Samuel J. Indorante, Dylan E. Beaudette, and Richard W. Arnold	982
Gas and Vapor Phase Transport / Dennis E. Rolston	986
Gelisols / James G. Bockheim	989
Geologic Carbon Capture and Storage / Sean I. Plasynski, John T. Litynski, Timothy R. Carr, Howard G. McIlvried, and Rameshwar D. Srivastava	992
Geology and Soil / Kenneth R. Olson	997
Geomorphology / Robin N. Thwaites	1000
Geophysical Methods / Barry J. Allred, Viacheslav I. Adamchuk, Raphael A. Viscarra Rossel, Jim Doolittle, Robert S. Freeland, Katherine R. Grote, and Dennis L. Corwin	1004
Geostatistics / Anja Gassner and Ewald Schnug	1012
GIS Integration: Remote Sensing / Egide Nizeyimana	1017
Global Resources / Paul Reich and Hari Eswaran	1020
Grass Strip Hydrology / Hossein Ghadiri and Calvin W. Rose	1024
Grasslands Soils / Douglas D. Malo	1029
Grazing Lands: Gaseous Emissions / Lipismita Samal, Veerasamy Sejian, M. Bagath, R.U. Suganthi, Raghavendra Bhatta, and Rattan Lal	1034
Green Revolution: China North Eastern Plains / Xiangbin Kong and Rattan Lal	1042
Greenhouse Effect / Rattan Lal	1048

Growing Media / Nazim Gruda, Jean Caron, Munoo Prasad, and Michael Maher	1053
Gulley Erosion / Fenli Zheng, Chihua Huang, and Ximeng Xu	1059
Gypsic Soils / Juan Herrero and Jaime Boixadera	1064
Gypsic Soils: Gypsum Formation / Ahmet R. Mermut and Hossein Khademi	1068
Hard-Setting Soils / Richard S.B. Greene	1072
Health / Robin F. Harris and D.E. Romig	1076
Heat and Water Movement / Gerald N. Flerchinger	1079
Heat Capacity / Surinder Kumar Jalota and B.S. Ghuman	1083
Heat Flux / Thomas J. Sauer	1086
Heavy Metals / Mike J. McLaughlin	1089
History of Soil Science / Eric C. Brevik and Artemi Cerdà	1093
History of Soil Science: 1927–2000 / Fred P. Miller	1098
History of Soil Science: Early to Mid-20th Century / John P. Tandarich	1103
Histosols / David L. Lindbo and Deborah A. Kozlowski	1107
Human Culture and Soils / Daniel Hillel	1112
Human Dimensions: Soil Sustainability / Mary Seely, Patrik Klintonberg, and Claus Hager	1115
Human Health and Soil / Charles F. Sing and David B. Sing	1118
Human Society and Soil / Rattan Lal	1123
Human Society and the Planet: Soil as a Heritage / Fred P. Miller	1127
Hydraulic Conductivity / Jacob H. Dane, Marc Jalbert, and Jan W. Hopmans	1133
Hydrologic Cycle / Keith R.J. Smettem	1137
Hydropedology / Henry Lin	1140
Hydrophobic and Hydrophilic Compounds / Tomas Simon	1144
Hydrophobicity / Joerg Bachmann, Abraham Marmur, and Markus Deurer	1147
Inceptisols / John M. Galbraith, Kenneth Scheffe, and Shawn McVey	1151
India: Problems and Potentialities / S.K. Singh, S. Chattaraj, N.G. Patil, S.K. Ray, and S. Chatterji	1157
Indigenous Soil Management / Karl Herweg	1164
Indo-Gangetic Plain: Agro-Ecological Regions / Jaya N. Surya, R.P. Yadav, G.S. Sidhu, and S.K. Singh	1168
Industrial Waste / Warren Dick	1178
Infiltration: Management / Jutta Rogasik, Kerstin Panten, Ewald Schnug, and Helmut Rogasik	1183
Infiltration: Properties / Walter J. Rawls	1187
Inorganic Carbon: Analysis / Richard R. Drees and C. Tom Hallmark	1191
Inorganic Carbon: Climate and Time / Lee C. Nordt	1195
Inorganic Carbon: Composition and Formation / H. Curtis Monger and Larry P. Wilding	1199
Inorganic Carbon: Global Carbon Cycle / William H. Schlesinger	1203
Inorganic Carbon: Global Stocks / Larry P. Wilding, Lee C. Nordt, and John M. Kimble	1206
Inorganic Carbon: Land Use Impacts / Donald L. Suarez	1210
Inorganic Carbon: Modeling / Leslie D. McFadden and Ronald G. Amundson	1213
Insect Survival: Subzero Temperature / Mike M. Ellsbury	1217
Integrated Nutrient Management / Bal Ram Singh and Ram C. Dalal	1221
Integrated Nutrient Management: Sustainability / G.S. Saroa and Rattan Lal	1227
International Soil Reference and Information Centre (ISRIC) / David Dent	1232
International Union of Soil Sciences / Stephen Nortcliff	1237
Ion Exchange / Bryon W. Bache	1241
Iron Oxides / Nikolla P. Qafoku and James Amonette	1245
Irrigation / Jack Keller	1250

Volume II (*cont'd.*)

Irrigation: Efficiency / Terry A. Howell	1256
Irrigation: Erosion / Robert E. Sojka and David L. Bjorneberg	1261
Irrigation: Historical Perspective / Robert E. Sojka, David L. Bjorneberg, and J.A. Entry	1264
Irrigation: Soil Salinity / James D. Rhoades	1269
Jhum: Cultivation / Arun Jyoti Nath, Biplab Brahma, Rattan Lal, and Ashesh Kumar Das	1273
Jhum: Shifting Agriculture / P.S. Ramakrishnan	1281
Land Capability Analysis / Michael J. Singer	1285
Land Capability Classification / Thomas E. Fenton	1288
Land Evaluation / J. Herbert Huddleston	1291
Land Forms / Carolyn G. Olson	1294
Land Husbandry / Francis Shaxson and Malcolm Douglas	1297
Land Restoration / Richard W. Bell	1304
Land Use: Historic / J. Douglas Helms, Hari Eswaran, and Paul Reich	1308
Land Use: Land Cover / Andrew K. Skidmore	1313
Land Use: Planning and Geographic Information System (GIS) / Egide Nizeyimana and Jacob Opadeyi	1318
Landfill Sites: Revegetation / Gilbert Yuk-sing Chan and Ming H. Wong	1322
Landscape / George Hall	1327
Landscape: Classification / Philip Schoeneberger	1331
Landscape: Development and Erosion / Jon Harbor	1336
Landscape: Elements / Douglas Wysocki and C. William Zanner	1339
Landscape: Relationships / Timothy A. Quine	1344
Landslide and Landcreep / Jau-Yau Lu and Hao-Jung Shieh	1348
Law and Legal Issues / Ian Hannam and Ben Boer	1352
Law and Legal Issues: Land as Property / Paul Martin and John Page	1355
Leaching and Illuviation / Lynn E. Moody	1358
Liming and Lime Materials / Mark K. Conyers	1362
Loess / Fred E. Rhoton and Helaine W. Markewich	1365
Loess Plateau / Fenli Zheng, Ximeng Xu, and Xiubin He	1370
Low-Carbon Technology: Brazilian Semiarid Ecosystems / Antonio Pereira Filho, Vanderlise Giongo, Tony Jarbas Ferreira Cunha, Jose Texeira Filho, Tayane de Lima Santos, and Luiz Fernando Carvalho Leite	1377
Lower Himalayas: Soil Management / Anup Das, Ramkrushna Gi, Badapmain Makdoh, Dibyendu Sarkar, Jayanta Layek, Sandip Mandal, and Rattan Lal	1382
Macroporosity / Daniel Gimenez and Daniel Hirmas	1388
Magnesium / N.S. Bolan, K. Arulmozhiselvan, and P. Paramasivam	1392
Manganese: Brazilian Soils / Diego Antonio França de Freitas, Nilton Curi, and Nestor Kämpf	1396
Manure Management: Gaseous Emissions / Veerasamy Sejian, Lipismita Samal, M. Bagath, R.U. Suganthi, Raghavendra Bhatta, and Rattan Lal	1400
Manure: Compost and Biosolids / Bahman Eghball and Kenneth A. Barbarick	1406
Mapping Soil Carbon / Budiman Minasny, Brendan Malone, and Alex B. McBratney	1409
Mapping Soil Carbon: Stocks in Turkey / Gönül Aydın, Mehmet Ali Çullu, Sabit Erşahin, Erhan Akça, Emrah Erdoğan, Levent Atatanır, Alper Yorulmaz, Ahmet Çilek, Merve Ersoy, Somayyeh Rezzaghi Miavaghi, Selim Kapur, and Rattan Lal	1412
Materials and Soils / Selim Kapur, Erhan Akça, and Takanori Nagano	1416

Mediterranean Agriculture: Rainfed, Nitrogen Use Efficiency / <i>Àngela D. Bosch Serra</i>	1421
Metal—Clay Interactions / <i>Dean Hesterberg</i>	1426
Microbial Biomass: Measurement / <i>K.R. Islam and S.R. Wright</i>	1429
Microbial Communities / <i>K.R. Islam and S.R. Wright</i>	1433
Micromorphology and Soil Quality / <i>Ahmet R. Mermut</i>	1439
Micronutrients and Human Health / <i>Ram Phal Narwal, R.S. Malik, S.K. Malhotra, and Bal Ram Singh</i>	1443
Mine Soils: Measuring Geogenic Carbon / <i>David A.N. Ussiri and Rattan Lal</i>	1449
Mine Soils: Miscanthus Plantations / <i>Jose G. Guzman and Rattan Lal</i>	1458
Mine Soils: Open Cut Mine Rehabilitation / <i>Douglas J. Dollhopf</i>	1462
Mine Soils: Reclamation and Soil Carbon Sequestration / <i>Vasant A. Akala and Rattan Lal</i>	1466
Minerals: Amorphous / <i>April L. Ulery</i>	1469
Minerals: Primary / <i>Nikolaos I. Barbayiannis and Vissarion Z. Keramidas</i>	1473
Minerals: Secondary / <i>Deborah A. Soukup</i>	1476
Minerals: Solubility / <i>Wayne P. Robarge</i>	1480
Minirhizotron / <i>Verónica Muñoz-Romero, Luis López-Bellido, F. Javier López-Bellido, and Rafael J. López-Bellido</i>	1484
Mites / <i>Hans Klompen</i>	1488
Modern Civilization and Soils / <i>Kenneth R. Olson</i>	1491
Mulch Farming / <i>Rattan Lal</i>	1495
Mycorrhiza of Forest Ecosystems / <i>Ian A. Dickie</i>	1502
Mycorrhizae: Soil Quality / <i>Ibrahim Ortas</i>	1505
Nanofertilizers / <i>Ruiqiang Liu and Rattan Lal</i>	1511
Natural Resources: Interaction with Soil / <i>Dennis J. Greenland</i>	1516
Nematodes / <i>Christien H. Ettema</i>	1518
Nematodes and Soil / <i>Wilfrida Decraemer</i>	1523
Nexus Approach: Resource Management for Soil Productivity / <i>Kai Schwärzel, Reza Ardakanian, Tamara Avellán, and Lulu Zhang</i>	1530
Nitrate Leaching Index / <i>Harold M. van Es and Jorge A. Delgado</i>	1535
Nitrate Leaching Management / <i>John J. Meisinger, Jorge A. Delgado, and Ashok Alva</i>	1538
Nitrogen and its Transformations / <i>Oswald Van Cleemput and Pascal Boeckx</i>	1541
Nitrogen Index / <i>Jorge A. Delgado</i>	1545
Nitrogen-15: Utilization / <i>Jorge Benitez-Vega, Luis López-Bellido, Ramón Redondo, and Rafael J. López-Bellido</i>	1551
Nitrous Oxide Emissions: Agricultural Soils / <i>John R. Freney</i>	1555
Nitrous Oxide Emissions: Paddy Soils / <i>Jianfu Xue, Xin Zhao, and Hailin Zhang</i>	1559
Nitrous Oxide Emissions: Sources, Sinks, and Strategies / <i>Katsu Minami</i>	1562
No Till / <i>Paul W. Unger</i>	1566
Nutrient Management / <i>Jorge A. Delgado</i>	1570
Nutrient Management: Fertility and / <i>John L. Havlin</i>	1576
Nutrients: Deficiency and Toxicity Symptoms / <i>Noel J. Grundon</i>	1585
Nutrients: Diffusion, Bioavailability, and Plant Uptake / <i>Niels Erik Nielsen</i>	1588
Nutrients: Interactions in Soil—Plant System / <i>Fusuo S. Zhang, J. Shen, and Y.-G. Zhu</i>	1594
Nutrients: Water Interactions / <i>Ardell D. Halvorson</i>	1597
Organic Agriculture / <i>Kathleen Delate</i>	1600
Organic Carbon: Assessment Methods / <i>K.R. Islam</i>	1604
Organic Carbon: Sequestration Concepts / <i>Kenneth R. Olson</i>	1609
Organic Carbon: Subsoil Pools / <i>Klaus Lorenz and Rattan Lal</i>	1614

Volume II (*cont'd.*)

Organic Farming: China / Yuxin Miao, Fusuo Zhang, and Fanghao Wang	1618
Organic Fertilization: Heavy Metal Uptake / Ewald Schnug, Alexandra Izosimova, and Renata Gaj	1621
Organisms and Soil Food Webs / David C. Coleman	1624
Organo-Mineral Relationships / Claire Chenu, Alain F. Plante, and Pascale Puget	1629
Oxisols / Stanley W. Buol	1635
Pampas, The / Martín Díaz-Zorita and Daniel E. Buschiazso	1638
Pantanal, The / Henrique de Oliveira, Amaury de Carvalho Filho, Carlos Ernesto Reynauld G. Schaefer, and Evaldo Luis Cardoso	1643
Paramos Soils / Pascal Podwojewski and Jérôme Poulenard	1648
Particle Density / Keith R.J. Smettem	1652
Particle Packing / Satish C. Gupta and D.P. Thoma	1654
Particle Shape / Brian M. Schafer	1659
Particle Size / Hwat Bing So and Harold R. Geering	1661
Peatlands / Dale H. Vitt and Paul Short	1668
Pedological Modeling / Ronald G. Amundson	1674
Pedometrics / Alex B. McBratney and Budiman Minasny	1677
Pedotransfer Functions / J.H.M. Wösten	1682
Permafrost: Soil Temperature and Special Problems / Douglas Kane and Julia Boike	1687
Petrocalcic Horizons / Alain Ruellan	1691
pH / Grant W. Thomas	1695
Phosphorus / John Ryan and Abdul Rashid	1700
Phosphorus: Manure and Diet Modification / Rory O. Maguire, John T. Brake, and Peter W. Plumstead	1705
Phosphorus: Plant Nutrition and Microorganism Contribution / Bettina Eichler-Löbermann, Christel Baum, Silke Ruppel, and Ewald Schnug	1708
Phosphorus: Spatial Speciation in Agricultural Soils / Anja Gassner and Ewald Schnug	1712
Phytoremediation / Khan Towhid Osman and Md. Abul Kashem	1717
Plant Growth: Oxygen Diffusion Rate / Witold Stepniewski	1725
Plant Nutrients / Volker Roemheld	1728
Plant Nutrients: Intensity, Quantity, and Buffer Power / Ram C. Dalal	1731
Plant Nutrients: Sufficiency and Requirements / Kanwar L. Sahrawat	1735
Plastic Properties / Gunnar Kirchhof	1740
Plinthite and Petroplinthite / Eswaran Padmanabhan and Hari Eswaran	1743
Podzols / Peter Buurman	1747
Polar Regions, The / Charles Tarnocai and Iain Campbell	1753
Pollution / Zulkuf Kaya	1758
Pollution: Industrial Waste / Winfried E.H. Blum, Walter W. Wenzel, Wolfgang Friesl-Hanl, and Martin H. Gerzabek	1762
Pollution: Non-Point Source / Ravendra Naidu, Mallavarapu Megharaj, Peter Dillon, Rai Kookana, Ray Correll, and Walter W. Wenzel	1765
Pollution: Organic and Inorganic / A. Paul Schwab	1768
Pollution: Persistent Organic / Mark Radosevich and E. Danielle Rhine	1771
Pollution: Point Source / Ravendra Naidu, Mallavarapu Megharaj, Peter Dillon, Rai Kookana, Ray Correll, and Walter W. Wenzel	1776

Volume III

Porosity: Pore Size Distribution / Keith C. Cameron and Graeme D. Buchan	1782
Potassium / Philippe Hinsinger	1786
Potassium: Dynamics and Mineralogy / A.V. Shanwal and S.S. Dahiya	1791
Potassium: Nutrition in Semi-Arid Tropics / D.L. Oswalt	1796
Power Plants: Carbon Dioxide Capture / Sean I. Plasynski, Thomas J. Feeley, Timothy Fout, Howard G. McIlvried, and Rameshwar D. Srivastava	1798
Precision Agriculture: Engineering Aspects / Joel T. Walker, Reza Ehsani, and Matthew O. Sullivan ...	1804
Precision Agriculture: Fertilization / Silvia H. Haneklaus and Ewald Schnug	1808
Precision Agriculture: Nutrient Cycling / Robert J. Lascano	1811
Precision Agriculture: Remote Sensing / Kerstin Panten, Holger Lilienthal, Erik Zillmann, Silvia H. Haneklaus, and Ewald Schnug	1815
Productivity: Natural and Anthropogenic Disturbances / Douglas G. Maynard and Charlotte E. Norris	1819
Protection: Policy of the European Union / Irene L. Heuser and Luca Montanarella	1823
Protozoa / Bryan S. Griffiths	1828
Pyrite / Michael J. Borda	1831
Pyrogenic Carbon / Barry G. Rawlins	1834
Quality / Ed G. Gregorich	1837
Quality Index: Soil / Toru Nakajima	1841
Quality: Assessment by Interpolation Techniques / Vincent Obade	1844
Quality: Carbon and Nitrogen Gases / Philippe Rochette, Reynald Lemke, and Sean McGinn ...	1848
Quality: Critical Limits and Standardization / Martin R. Carter	1853
Quality: Erosion / Craig Ditzler	1856
Quality: Indicators / Graham P. Sparling	1859
Quality: Productivity / Mike H. Beare	1862
Quality: Soil and Water / Todd Nissen and Michelle Wander	1866
Radiometric Survey / Barry G. Rawlins and Raphael A. Viscarra Rossel	1872
Radionuclides / Philip M. Jardine	1876
Raindrop Impact and Splash Erosion / Hossein Ghadiri	1881
Rapid Soil Analysis: Diffuse Reflectance Spectroscopy / Keith D. Shepherd and Markus G. Walsh	1886
Rare Earth Elements / Zhengyi Hu, Gerd Sparovek, Silvia H. Haneklaus, and Ewald Schnug	1891
Redox Phenomena / Bruce R. James	1896
Rehabilitation: Indicators and Monitoring / David Jasper	1899
Religious Aspects of Soil—Human Relationships / Verena Winiwarter and Winfried E.H. Blum ...	1903
Rendzina / Thomas E. Fenton	1907
Resilience / Moses M. Tenywa, J.G.M. Majaliwa, and A. Lufafa	1909
Resilience: Land Use and Soil Management / Soojin J. Park	1913
Resilience: Quality and / Rattan Lal	1918
Resilience: Restoration / F.W.T. Penning de Vries and Eric T. Craswell	1924
Respiration / Pierre A. Jacinthe and Rattan Lal	1928
Restoration Ecology / Richard Hobbs	1932
Restoration: Success and Completion Criteria / Sam C. Ward	1935
Rice Paddies: Methane Emissions / Ronald L. Sass	1940
Rice Paddies: Methane Emissions, Mitigating / Kazuyuki Yagi	1943

Volume III (*cont'd.*)

Rice Paddies: Methanogenesis / Ronald L. Sass	1947
Rice Paddies: Soil Management / Xin Zhao, Jianfu Xue, and Hailin Zhang	1950
Rocks / Vsevolod V. Dobrovolsky	1953
Root Growth: Pressure at the Soil—Root Interface / Rabi K. Misra	1957
Root Monitoring System / Zhikuan Jia, Pute Wu, Qingfang Han, Junfeng Nie, Ruixia Ding, and Yufeng Zou	1960
Salt-Affected Soils / Anna Eynard, Rattan Lal, and Keith D. Wiebe	1965
Salt-Affected Soils: Nitrous Oxide Emissions / G.S. Dheri, Babu S. Brar, and Debankur Sanyal	1969
Satellite Mapping / Bruce E. Frazier	1972
Scaling Effects / Yakov A. Pachepsky, Walter J. Rawls, and John W. Crawford	1976
Sediment: Influence on Aquatic Life / Suzanne M. Gray	1981
Sedimentation Control and Erosion: Engineering Techniques / Jose Miguel Reichert and Elemar Antonino Cassol	1986
Seepage / Patricia M. Gallagher	1990
Selenium / Gunnar Gissel-Nielsen	1995
Serpentinitic Soils / Rebecca Burt and Michael A. Wilson	1999
Sesquioxides / Joey N. Shaw and Larry T. West	2003
Shrinkage / Des McGarry and Don F. Yule	2008
Silica: Pedogenic Accumulation / Katherine J. Kendrick	2012
Silicon and Sodium / Jian Feng Ma	2015
Slaking, Dispersion, and Crust Formation / Hwat Bing So	2021
Social and Behavioral Sciences' Contribution to Soil Science / Sven B. Lundstedt	2026
Sodic Soils / Pichu Rengasamy	2029
Sodic Soils: Formation and Global Distribution / Brian Murphy	2032
Sodic Soils: Humid Regions / Samuel J. Indorante	2037
Sodic Soils: Irrigation Farming / David Burrow	2041
Sodic Soils: Physical Properties and Behavior / Hwat Bing So	2044
Sodic Soils: Processes, Characteristics, and Classification / Pichu Rengasamy	2048
Sodic Soils: Reclamation / Jock Churchman	2051
Sodicity: Subsoil / Kwong Yin Chan	2056
Soil Carbon Map: Africa / Arwyn Jones, Roland Hiederer, and Luca Montanarella	2059
Soil Carbon Sequestration: Ethiopia / Ambachew Demessie, Bal Ram Singh, and Rattan Lal	2066
Soil Carbon: Compositional and Isotopic Analysis / James Moran, M. Lizabeth Alexander, and Alexander Laskin	2073
Soil Carbon: Farming and Economic Aspects / Bruce A. McCarl	2078
Soil Conservation: Incentives / David Sanders	2081
Soil Conservation: Strategies / Winfried E.H. Blum	2084
Soil Conservation: U.S. Historical Perspective / Dana D. York	2086
Soil Degradation: Global Assessment / Selim Kapur and Erhan Akça	2090
Soil Enzymes / Cherukumalli Srinivasa Rao, Minakshi Grover, Sumanta Kundu, and Susheelendra Desai	2100
Soil Organic Matter (SOM) / Rattan Lal	2108
Soil Organic Matter (SOM): Accumulation / Sylvie A. Quideau	2112
Soil Organic Matter (SOM): Available Water Capacity / Tom G. Huntington	2117

Soil Organic Matter (SOM): Characterization / Christian Feller	2123
Soil Organic Matter (SOM): Composition and Dynamics / Cornelia Rumpel, M.F. Dignac, and Abad Chabbi	2126
Soil Organic Matter (SOM): Global Carbon Cycle / Keith Paustian	2129
Soil Organic Matter (SOM): Global Distribution / Wilfred M. Post	2133
Soil Organic Matter (SOM): Historical Concepts / Christian Feller	2139
Soil Organic Matter (SOM): Landscape / Henry H. Janzen, B.H. Ellert, and D.W. Anderson	2142
Soil Organic Matter (SOM): Management / R. Cesar Izaurralde and Carlos C. Cerri	2147
Soil Organic Matter (SOM): Mass Spectrometry Data Analysis / Nikola Tolic, Malak M. Tfaily, Yufeng Shen, Rosalie Chu, Thomas Fillmore, Rui Zhao, Kent Bloodsworth, Errol Robinson, Ljiljana Paša-Tolić, and Nancy J. Hess	2154
Soil Organic Matter (SOM): Modeling / Peter Smith	2159
Soil Organic Matter (SOM): Molecular Simulations / Amity Andersen	2166
Soil Organic Matter (SOM): Nutrient Dynamics / Charles W. Rice	2172
Soil Organic Matter (SOM): Solvent-Based High Resolution Mass Spectrometry / Malak M. Tfaily, Yufeng Shen, Nikola Tolic, Jennifer Kyle, Rosalie Chu, Thomas Fillmore, Rui Zhao, Kent Bloodsworth, Errol Robinson, Ljiljana Paša-Tolić, and Nancy J. Hess	2176
Soil Organic Matter (SOM): Structure and Characterization / Georg Guggenberger	2180
Soil Organic Matter (SOM): Turnover / Johan Six and Julie D. Jastrow	2186
Soluble Salts: Translocation and Accumulation / Jimmie L. Richardson	2192
Solute Transport / Arne E. Olsen and James W. Jawitz	2194
Spectral Mixtures: Soil—Plant / Alfredo R. Huete	2197
Spodosols / Delbert Mokma	2200
Strontium / Silvia H. Haneklaus and Ewald Schnug	2203
Structure / Eileen J. Klodivko	2207
Structure: Belowground Management / Ken A. Olsson and Bruce Cockcroft	2209
Structure: Earthworms / Martin Shipitalo and Tayfun Korucu	2212
Structure: Pedological Concepts and Water Flow / Clinton C. Truman and Donald P. Franzmeier	2216
Structure: Plant Establishment / Douglas L. Karlen	2221
Structure: Roots / Alvin J.M. Smucker	2225
Sub-Irrigation / Norman R. Fausey	2228
Sugarcane Fields: Harvest Systems and Residue Management / Newton La Scala Jr., Ricardo De Oliveira Bordonal, Eduardo Barretto De Figueiredo, Alan Rodrigo Panosso, Mara Regina Moitinho, and Mariana Marotti Corradi	2231
Sulfate and Sulfide Minerals / Delvin S. Fanning and Steven N. Burch	2238
Sulfur / Ewald Schnug, Silvia H. Haneklaus, and Elke Bloem	2241
Sulfur: Chemical Behavior in the Rhizosphere / Zhengyi Hu, Silvia H. Haneklaus, and Ewald Schnug	2247
Sulfur: Nutrition / Silvia H. Haneklaus, Elke Bloem, and Ewald Schnug	2251
Surface Area: Specific / L.A. Graham Aylmore	2255
Surface Management / Kwong Yin Chan	2259
Surface Water: Nitrogen Enrichment / Tim P. Burt and Penny E. Widdison	2263
Surface Water: Pollution by Nitrogen Fertilizers / Gregory McIsaac	2268
Sustainable Agriculture: Social Aspects / Frederick H. Buttel	2273
Sustainable Agriculture: Soil Quality / John W. Doran and Ed G. Gregorich	2276
Sustainable Development and Sustainability / Alan D. Hecht and Joseph Fiksel	2280
Tailings: Minerals Processing Residue Rehabilitation / Lloyd R. Hossner and Hamid Shahandeh	2286

Volume III (*cont'd.*)

Taxonomies: U.S. Soil Taxonomy and Indian Scenario / <i>Tapas Bhattacharyya and Dilip Kumar Pal</i>	2291
Taxonomy / <i>Mark H. Stolt</i>	2295
Temperature Measurement / <i>Kevin McInnes</i>	2302
Tepetates: Mexico / <i>Jorge D. Etchevers, Claudia Hidalgo, C. Prat, and P. Quantin</i>	2305
Termites / <i>David E. Bignell and John A. Holt</i>	2309
Terra Preta (Black Earths) / <i>Kurt Spokas, Jose Marques Jr., Newton La Scala Jr., Ed Nater, and Diego S. Siqueira</i>	2312
Terra Preta de Indio / <i>Johannes Lehmann</i>	2316
Testing / <i>Yash P. Kalra and Jagtar S. Bhatti</i>	2320
Texture / <i>Harold R. Geering and Hwat Bing So</i>	2323
Tillage Erosion: Description and Process / <i>Michael J. Lindstrom</i>	2330
Tillage Erosion: Measurement Techniques / <i>David A. Lobb</i>	2333
Tillage Erosion: Properties and Productivity Effects / <i>Tom E. Schumacher</i>	2336
Tillage Erosion: Relationship to Water Erosion / <i>Gerard Govers</i>	2339
Tillage Erosion: Terrace Formation / <i>Seth M. Dabney and Dalmo A.N. Vieira</i>	2342
Tillage: Gas Exchange / <i>Donald C. Reicosky</i>	2348
Tilth / <i>Jerry L. Hatfield</i>	2351
Time and Space: Soils in / <i>Neil E. Smeck and C. Lee Burras</i>	2353
Torrential Erosion: Himalayas / <i>Ram Prashad Yadav</i>	2358
Trace Metal Contamination / <i>Åsgeir Rossebø Almås and Bal Ram Singh</i>	2364
Tropics and Subtropics / <i>Hari Eswaran and Eswaran Padmanabhan</i>	2369
Turbations / <i>Brad D. Lee and Robert C. Graham</i>	2373
Turfgrass: Protection of Soil and Water / <i>James A. Murphy and Stephanie L. Murphy</i>	2377
Ultisols / <i>Joey N. Shaw</i>	2380
Uranium Contamination / <i>Ewald Schnug, Roberta Bottino Montolar Sparovek, Maria del Carmen Lamas, Sylvia L. Kratz, Jurgen Fleckenstein, and Susanne Schroetter</i>	2385
Urban Lands: Abandoned / <i>Joshua W. Beniston and William D. Shuster</i>	2388
Urban Lands: and Construction Sites / <i>Wolfgang Burghardt</i>	2395
Urban Lands: Management / <i>Klaus Lorenz</i>	2400
Urban Wastewater for Irrigation: Heavy Metals / <i>P.S. Minhas and K. Lal</i>	2407
Urea: Subsoiling and Controlled Release / <i>Tangyuan Ning, Rattan Lal, Zengjia Li, and Min Zhang</i>	2411
Value of Soil to Humans / <i>Rattan Lal</i>	2416
Variability / <i>Josef H. Görres</i>	2419
Variability: Spatial / <i>Brian Slater</i>	2428
Variable Charge Soils: Mineralogy and Chemistry / <i>Eric Van Ranst, Nikolla P. Qafoku, Andrew Noble, and Ren-kou Xu</i>	2432
Vertisols / <i>Wouter A. Blokhuis</i>	2440
Vertisols: Surface Crack Management / <i>J. Somasundaram, Nishant K. Sinha, Ashok K. Patra, and Rattan Lal</i>	2450
Vesicular Crusts / <i>Patrick Drohan</i>	2456
Village Bamboos / <i>Arun Jyoti Nath, Ashesh Kumar Das, and Rattan Lal</i>	2459
Waste Rock and Overburden Dumps: Rehabilitation / <i>Martin V. Fey, R.D. O'Brien, and A.J. Mills</i>	2464
Wastewater: Treatment and Utilization in Irrigation / <i>Yona Chen and Jorge Tarchitzky</i>	2467

Water Conservation / <i>Paul W. Unger</i>	2472
Water Content / <i>Tom J. Jackson</i>	2475
Water Content: Passive and Active Microwave Remote Sensing / <i>Eric D. Warner and Gary W. Petersen</i>	2480
Water Erosion / <i>David Favis-Mortlock</i>	2485
Water Erosion: Empirical Models / <i>John M. Laflen</i>	2489
Water Erosion: Hybrid Models / <i>Jeffrey G. Arnold, Kevin W. King, and Jimmy R. Williams</i>	2494
Water Erosion: Modeling / <i>Calvin W. Rose</i>	2499
Water Erosion: Process-Based Modeling / <i>Mark A. Nearing</i>	2504
Water Infiltration / <i>Manoj K. Shukla and Rattan Lal</i>	2507
Water Management: Computer Modeling / <i>Laj R. Ahuja and Liwang Ma</i>	2510
Water Management: Harvesting / <i>Gary W. Frasier</i>	2515
Water Management: Use Efficiency / <i>David C. Nielsen</i>	2518
Water Measurement: Methods / <i>Des McGarry</i>	2522
Water Range: Least-Limiting / <i>Alvaro Pires da Silva, Beverly D. Kay, Cássio A. Tormena, Silvia Imhoff, Douglas L. Karlen, and Joseph G. Benjamin</i>	2530
Water Retention / <i>Satish C. Gupta and Dong Wang</i>	2535
Water: Plant-Available / <i>Argyrios Gerakis and Joe T. Ritchie</i>	2541
Water-Repellent Soils / <i>Anna Eynard, Rattan Lal, and Keith D. Wiebe</i>	2546
Watershed Approach / <i>Timothy R. Green and Jan van Schilfgaarde</i>	2550
Watershed Management: Remote Sensing and GIS Applications / <i>A.V. Shanwal and S.P. Singh</i>	2555
Weathering / <i>Dominique Righi</i>	2559
Weathering: Carbon Sequestration / <i>William Berry Lyons, Jerry M. Bigham, Anne E. Carey, and Rattan Lal</i>	2563
Wetlands / <i>Ralph W. Tiner</i>	2567
Wetlands: Biodiversity / <i>Jean-Claude Lefeuvre and Virginie Bouchard</i>	2572
Wetlands: Carbon Sequestration / <i>Virginie Bouchard and Matthew Cochran</i>	2576
Wetlands: Conservation Policy / <i>Clayton Rubec</i>	2580
Wetlands: Economic Value / <i>Gayatri Acharya</i>	2584
Wetlands: Gaseous Emissions / <i>S.K. Nag</i>	2589
Wetlands: Methane Emission / <i>Anna Ekberg and Torben Røjle Christensen</i>	2596
Wetlands: Sedimentation and Ecological Engineering / <i>Timothy C. Granata and Jay F. Martin</i> ...	2600
Wind Erosion: Climate Change / <i>Alan Busacca and David Chandler</i>	2603
Wind Erosion: Control Measures / <i>Gary L. Tibke</i>	2609
Wind Erosion: Environmental Effects / <i>R. Scott Van Pelt</i>	2614
Wind Erosion: Field Measurement / <i>R. Scott Van Pelt and Ted M. Zobeck</i>	2618
Wind Erosion: Global Hot Spots / <i>Andrew Warren</i>	2625
Wind Erosion: Micrometeorology / <i>Giles F.S. Wiggs</i>	2628
Wind Erosion: Modeling / <i>Lawrence J. Hagen</i>	2632
Wind Erosion: Principles / <i>Larry D. Stetler</i>	2636
Wind Erosion: Soil Quality and Productivity / <i>John Leys</i>	2640
Windblown Dust / <i>Dale Gillette</i>	2644
Wind-Driven Rain: Erosion / <i>Gunay Erpul, L. Darrell Norton, and Donald Gabriels</i>	2647
World Reference Base for Soil Resources / <i>Peter Schad and Stefaan Dondeyne</i>	2650



Taylor & Francis

Taylor & Francis Group

<http://taylorandfrancis.com>

Topical Table of Contents

Agriculture

Agricultural Management: Root Growth / Susanne Schroetter, Jutta Rogasik, and Ewald Schnug	72
Agricultural Soil Carbon: Trading / Bruce A. McCarl, Sung Ju Cho, Jinxiu Ding, Pei Huang, Layla Shiva, and Chin-Hsien Yu	76
Biofuels versus Agricultural Soils / David Pimentel	213
Bunch Planting: Water Conservation / B.A. Stewart	261
Culture: Soil-less / Nazim Gruda, Giorgio Prosdocimi Gianquinto, Yuksel Tuzel, and Dimitrios Savvas	533
Growing Media / Nazim Gruda, Jean Caron, Munoo Prasad, and Michael Maher	1053
No Till / Paul W. Unger	1566
Organic Fertilization: Heavy Metal Uptake / Ewald Schnug, Alexandra Izosimova, and Renata Gaj	1621
Plant Growth: Oxygen Diffusion Rate / Witold Stepniewski	1725
Root Growth: Pressure at the Soil—Root Interface / Rabi K. Misra	1957
Root Monitoring System / Zhikuan Jia, Pute Wu, Qingfang Han, Junfeng Nie, Ruixia Ding, and Yufeng Zou	1960
Sugarcane Fields: Harvest Systems and Residue Management / Newton La Scala, Jr., Ricardo De Oliveira Bordonal, Eduardo Barretto De Figueiredo, Alan Rodrigo Panosso, Mara Regina Moitinho, and Mariana Marotti Corradi	2231
Sustainable Agriculture: Social Aspects / Frederick H. Buttel	2273
Sustainable Agriculture: Soil Quality / John W. Doran and Ed G. Gregorich	2276
Tillage: Gas Exchange / Donald C. Reicosky	2348
Tilth / Jerry L. Hatfield	2351

Biological Effects

Biological Crusts / Jayne Belnap	216
Biotechnology / Manas R. Banerjee and Laila Yesmin	243
Organisms and Soil Food Webs / David C. Coleman	1624
Organo-Mineral Relationships / Claire Chenu, Alain F. Plante, and Pascale Puget	1629
Spectral Mixtures: Soil—Plant / Alfredo R. Huete	2197

Animals

Animals: Ecosystem Functioning / Alan J. Franzhuebbers	138
Animals: Microbial Interactions and Nutrient Cycling / Lisa Cole and Richard D. Bardgett	142
Fauna / Mary C. Savin	871
Fauna and Microflora: Macrofauna / Judy Tisdall	877

Biochar

Biochar and Soil Characteristics / Atanu Mukherjee and Rattan Lal	184
Biochar and Soil Quality / David A. Laird and Jeffrey M. Novak	189
Biochar: Soil Carbon and Fertility / Adam O'Toole and Daniel Rasse	193
Fire and Soils / Partap K. Khanna and John Raison	926

Biological Effects (*cont'd.*)

Fungi

Fungi / <i>Serita Frey</i>	979
Mycorrhiza of Forest Ecosystems / <i>Ian A. Dickie</i>	1502
Mycorrhizae: Soil Quality / <i>Ibrahim Ortaş</i>	1505

Insects

Ants / <i>Walter G. Whitford</i>	150
Collembola / <i>Steve P. Hopkin</i>	448
Insect Survival: Subzero Temperature / <i>Mike M. Ellsbury</i>	1217
Mites / <i>Hans Klompen</i>	1488
Termites / <i>David E. Bignell and John A. Holt</i>	2309

Microbes

Actinobacteria / <i>Marketa Šágová-Marecková and Jan Kopecký</i>	40
Bacteria / <i>Mary Ann Bruns</i>	175
Biota / <i>Lev O. Karpachevsky and Mikhail Karpachevsky</i>	235
Biota and Disease-Suppressive Soils / <i>Nicola Lorenz</i>	239
Microbial Communities / <i>K.R. Islam and S.R. Wright</i>	1433
Protozoa / <i>Bryan S. Griffiths</i>	1828

Plants

Village Bamboos / <i>Arun Jyoti Nath, Ashesh Kumar Das, and Rattan Lal</i>	2459
---	------

Soil Organic Matter

Soil Organic Matter (SOM) / <i>Rattan Lal</i>	2108
Soil Organic Matter (SOM): Accumulation / <i>Sylvie A. Quideau</i>	2112
Soil Organic Matter (SOM): Available Water Capacity / <i>Tom G. Huntington</i>	2117
Soil Organic Matter (SOM): Characterization / <i>Christian Feller</i>	2123
Soil Organic Matter (SOM): Composition and Dynamics / <i>Cornelia Rumpel, M.F. Dignac, and Abad Chabbi</i>	2126
Soil Organic Matter (SOM): Global Carbon Cycle / <i>Keith Paustian</i>	2129
Soil Organic Matter (SOM): Global Distribution / <i>Wilfred M. Post</i>	2133
Soil Organic Matter (SOM): Historical Concepts / <i>Christian Feller</i>	2139
Soil Organic Matter (SOM): Landscape / <i>Henry H. Janzen, B.H. Ellert, and D.W. Anderson</i>	2142
Soil Organic Matter (SOM): Management / <i>R. Cesar Izaurralde and Carlos C. Cerri</i>	2147
Soil Organic Matter (SOM): Mass Spectrometry Data Analysis / <i>Nikola Tolic, Malak M. Tfaily, Yufeng Shen, Rosalie Chu, Thomas Fillmore, Rui Zhao, Kent Bloodsworth, Errol Robinson, Ljiljana Paša-Tolić, and Nancy J. Hess</i>	2154
Soil Organic Matter (SOM): Modeling / <i>Peter Smith</i>	2159
Soil Organic Matter (SOM): Molecular Simulations / <i>Amity Andersen</i>	2166
Soil Organic Matter (SOM): Solvent-Based High Resolution Mass Spectrometry / <i>Malak M. Tfaily, Yufeng Shen, Nikola Tolic, Jennifer Kyle, Rosalie Chu, Thomas Fillmore, Rui Zhao, Kent Bloodsworth, Errol Robinson, Ljiljana Paša-Tolić, and Nancy J. Hess</i>	2176
Soil Organic Matter (SOM): Structure and Characterization / <i>Georg Guggenberger</i>	2180
Soil Organic Matter (SOM): Turnover / <i>Johan Six and Julie D. Jastrow</i>	2186

Worms

Earthworms / <i>Patrick J. Bohlen</i>	701
Enchytraeidae / <i>María Jesús Iglesias Briones</i>	723
Nematodes / <i>Christien H. Ettema</i>	1518
Nematodes and Soil / <i>Wilfrida Decraemer</i>	1523
Structure: Earthworms / <i>Martin Shipitalo and Tayfun Korucu</i>	2212

Carbon

Bioenergy Crops: Carbon Balance Assessment / <i>Rocky Lemus and Rattan Lal</i>	210
Carbon Assessment: Spectroscopic Methods / <i>James B. Reeves III and Gregory W. McCarty</i>	270
Carbon Sink Capacity: Soil Profile Characteristics / <i>Yuri Lopes Zinn</i>	315
Carbon: Inelastic Neutron Scattering / <i>Lucian Wielopolski</i>	319
Carbon: Laser-Induced Breakdown Spectroscopy / <i>Michael H. Ebinger, Ronny D. Harris, and Clifton W. Meyer</i>	323
Carbon: Physical Protection / <i>Jennifer A.J. Dungait and David W. Hopkins</i>	327
Carbonates / <i>Douglas W. Ming</i>	331
Carbonates: Water and / <i>Michael Duniway</i>	339
Cycling: Carbon and Nitrogen (C/N) / <i>Sylvie M. Brouder and Ronald F. Turco</i>	538
Ecology: Cycling of Carbon and Nitrogen / <i>Alan J. Franzluebbers</i>	711
Geologic Carbon Capture and Storage / <i>Sean I. Plasynski, John T. Litynski, Timothy R. Carr, Howard G. McIlvried, and Rameshwar D. Srivastava</i>	992
Low-Carbon Technology: Brazilian Semi-arid Ecosystems / <i>Antonio Pereira Filho, Vanderlise Giongo, Tony Jarbas Ferreira Cunha, Jose Texeira Filho, Tayane de Lima Santos, and Luiz Fernando Carvalho Leite</i>	1377
Mine Soils: Measuring Geogenic Carbon / <i>David A.N. Ussiri and Rattan Lal</i>	1449
Quality: Carbon and Nitrogen Gases / <i>Philippe Rochette, Reynald Lemke, and Sean McGinn</i>	1848
Soil Organic Matter (SOM): Global Carbon Cycle / <i>Keith Paustian</i>	2129

Carbon Sequestration

Carbon Sequestration: Fish Ponds / <i>Subhendu Adhikari and Rattan Lal</i>	274
Carbon Sequestration: Geologic / <i>Eswaran Padmanabhan</i>	279
Carbon Sequestration: Iceland / <i>Anna María Ágústssdóttir</i>	284
Carbon Sequestration: Nutrient Management / <i>M.C. Manna, Muneshwar Singh, R.H. Wanjari, Asit Mandal, and A. K. Patra</i>	288
Carbon Sequestration: Sediments / <i>Eswaran Padmanabhan</i>	294
Carbon Sequestration: Semi-arid Regions of India / <i>Cherukumalli Srinivasa Rao, Sumanta Kundu, and Srinivas K.</i>	299
Carbon Sequestration: Urban Ecosystems / <i>Adam Louis Selhorst</i>	307
Carbonates: Secondary and Pedogenic / <i>Ahmet R. Mermut and A. Landi</i>	335
Climate Change: Soil Carbon Sequestration / <i>Sherwood E. Idso and Keith E. Idso</i>	442
Mine Soils: Reclamation and Soil Carbon Sequestration / <i>Vasant A. Akala and Rattan Lal</i>	1466
Pyrogenic Carbon / <i>Barry G. Rawlins</i>	1834
Soil Carbon Sequestration: Ethiopia / <i>Ambachew Demessie, Bal Ram Singh, and Rattan Lal</i>	2066
Weathering: Carbon Sequestration / <i>William Berry Lyons, Jerry M. Bigham, Anne E. Carey, and Rattan Lal</i>	2563
Wetlands: Carbon Sequestration / <i>Virginie Bouchard and Matthew Cochran</i>	2576

Soil Inorganic Carbon

Inorganic Carbon: Analysis / <i>Richard R. Drees and C. Tom Hallmark</i>	1191
Inorganic Carbon: Climate and Time / <i>Lee C. Nordt</i>	1195
Inorganic Carbon: Composition and Formation / <i>H. Curtis Monger and Larry P. Wilding</i>	1199

Carbon Sequestration (*cont'd.*)

Inorganic Carbon: Global Carbon Cycle / William H. Schlesinger	1203
Inorganic Carbon: Global Stocks / Larry P. Wilding, Lee C. Nordt, and John M. Kimble	1206
Inorganic Carbon: Land Use Impacts / Donald L. Suarez	1210
Inorganic Carbon: Modeling / Leslie D. McFadden and Ronald G. Amundson	1213

Soil Organic Carbon

Organic Carbon: Assessment Methods / K.R. Islam	1604
Organic Carbon: Sequestration Concepts / Kenneth R. Olson	1609
Organic Carbon: Subsoil Pools / Klaus Lorenz and Rattan Lal	1614

Chemical Attributes

Allophanes / Leslie Louise Baker and Jodi Johnson-Maynard	97
Chemical Composition / Goro Uehara	348
Chemisorption / N.J. Barrow	353
Enzymes / Veronica Acosta-Martinez and Susanne Klose	737
Fertilizers: Mineral / L.L. Hammond and Balu L. Bumb	910
Ion Exchange / Bryon W. Bache	1241
Liming and Lime Materials / Mark K. Conyers	1362
Minerals: Amorphous / April L. Ulery	1469
Minerals: Primary / Nikolaos I. Barbayiannis and Vissarion Z. Keramidas	1473
Minerals: Secondary / Deborah A. Soukup	1476
Minerals: Solubility / Wayne P. Robarge	1480
pH / Grant W. Thomas	1695
Pyrite / Michael J. Borda	1831
Redox Phenomena / Bruce R. James	1896
Sesquioxides / Joey N. Shaw and Larry T. West	2003
Soil Enzymes / Cherukumalli Srinivasa Rao, Minakshi Grover, Sumanta Kundu, and Susheelendra Desai	2100
Soluble Salts: Translocation and Accumulation / Jimmie L. Richardson	2192
Solute Transport / Arne E. Olsen and James W. Jawitz	2194
Urea: Subsoiling and Controlled Release / Tangyuan Ning, Rattan Lal, Zengjia Li, and Min Zhang	2411
Variable Charge Soils: Mineralogy and Chemistry / Eric Van Ranst, Nikolla P. Qafoku, Andrew Noble, and Ren-kou Xu	2432

Clays

Clay Minerals / C.D. Barton and Anastasios D. Karathanasis	421
Clay Minerals: Charge Properties / Christopher John Matocha	426
Clay Minerals: Weathering and Alteration of / Anastasios D. Karathanasis	430

Metals

Aluminum / Johannes Bernhard Wehr, Frederick Paxton Cardell Blamey, and Neal William Menzies	105
Boron and Molybdenum / Umesh C. Gupta and J.A. MacLeod	252
Calcification / Janis L. Boettinger	264
Calcium / Zdenko Rengel	267
Cobalt and Iodine / Ronald G. McLaren	445
Heavy Metals / Mike J. McLaughlin	1089
Iron Oxides / Nikolla P. Qafoku and James Amonette	1245
Magnesium / N.S. Bolan, K. Arulmozhiselvan, and P. Paramasivam	1392

Manganese: Brazilian Soils / <i>Diego Antonio França de Freitas, Nilton Curi, and Nestor Kämpf</i>	1396
Metal–Clay Interactions / <i>Dean Hesterberg</i>	1426
Organic Fertilization: Heavy Metal Uptake / <i>Ewald Schnug, Alexandra Izosimova, and Renata Gaj</i>	1621
Selenium / <i>Gunnar Gissel-Nielsen</i>	1995
Strontium / <i>Silvia H. Haneklaus and Ewald Schnug</i>	2203
Trace Metal Contamination / <i>Åsgeir Rossebø Almås and Bal Ram Singh</i>	2364
Urban Wastewater for Irrigation: Heavy Metals / <i>P.S. Minhas and K. Lal</i>	2407

Other Elements

Antimony / <i>Gudny Okkenhaug and Jan Mulder</i>	146
Arsenic: Southeast Asia / <i>Mohammad Mahmudur Rahman and Ravi Naidu</i>	161
Chlorine / <i>Uzi Kafkafi and Guohua Xu</i>	362
Gypsic Soils / <i>Juan Herrero and Jaime Boixadera</i>	1064
Gypsic Soils: Gypsum Formation / <i>Ahmet R. Mermut and Hossein Khademi</i>	1068
Nitrate Leaching Index / <i>Harold M. van Es and Jorge A. Delgado</i>	1535
Nitrate Leaching Management / <i>John J. Meisinger, Jorge A. Delgado, and Ashok Alva</i>	1538
Rare Earth Elements / <i>Zhengyi Hu, Gerd Sparovek, Silvia H. Haneklaus, and Ewald Schnug</i>	1891

Sodic Soils

Salt-Affected Soils / <i>Anna Eynard, Rattan Lal, and Keith D. Wiebe</i>	1965
Sodic Soils / <i>Pichu Rengasamy</i>	2029
Sodic Soils: Formation and Global Distribution / <i>Brian Murphy</i>	2032
Sodic Soils: Humid Regions / <i>Samuel J. Indorante</i>	2037
Sodic Soils: Physical Properties and Behavior / <i>Hwat Bing So</i>	2044
Sodic Soils: Processes, Characteristics, and Classification / <i>Pichu Rengasamy</i>	2048
Sodic Soils: Reclamation / <i>Jock Churchman</i>	2051
Sodicity: Subsoil / <i>Kwong Yin Chan</i>	2056

Sulfur and Sulfates

Sulfate and Sulfide Minerals / <i>Delvin S. Fanning and Steven N. Burch</i>	2238
Sulfur / <i>Ewald Schnug, Silvia H. Haneklaus, and Elke Bloem</i>	2241
Sulfur: Chemical Behavior in the Rhizosphere / <i>Zhengyi Hu, Silvia H. Haneklaus, and Ewald Schnug</i>	2247
Sulfur: Nutrition / <i>Silvia H. Haneklaus, Elke Bloem, and Ewald Schnug</i>	2251

Classifications

Taxonomies: U.S. Soil Taxonomy and Indian Scenario / <i>Tapas Bhattacharyya and Dilip Kumar Pal</i>	2291
Taxonomy / <i>Mark H. Stolt</i>	2295

Classification Systems

Classification Systems / <i>Stephen Nortcliff</i>	367
Classification Systems: Arab Traditional, Moroccan Case / <i>Mohamed Sabir and Hassan Ben Jelloun</i>	370
Classification Systems: Australian and New Zealand / <i>Terry A. Isbell and A.E. Hewitt</i>	375
Classification Systems: Brazilian / <i>A.C.S. da Costa and M.R. Nanni</i>	379
Classification Systems: Canadian / <i>Charles Tarnocai</i>	383
Classification Systems: Chinese / <i>Zi-Tong Gong</i>	389
Classification Systems: French / <i>Jean-Paul Legros</i>	391
Classification Systems: Historical Developments / <i>Dan H. Yaalon</i>	395

Classifications (*cont'd.*)

Classification Systems: Indian / Mariappan Velayutham, Tapas Bhattacharyya, and Dilip Kumar Pal	398
Classification Systems: Indigenous / Erhan Akça and Selim Kapur	402
Classification Systems: Major / Lars Krogh	406
Classification Systems: Netherlands / Alfred E. Hartemink and Henk de Bakker	412
Classification Systems: Russian / Maria Gerasimova	415
Classification Systems: South African / Jeffrey C. Hughes	418

Types of Soils

Alfisols / Rienk Miedema	92
Andisols / Jon Chorover	130
Andisols: Iceland / Olafur Arnalds	134
Arid Soils / H. Curtis Monger	157
Chernozems / Boris Boincean and Rattan Lal	356
Entisols / Larry T. West	728
Gelisols / James G. Bockheim	989
Grasslands Soils / Douglas D. Malo	1029
Hard-Setting Soils / Richard S.B. Greene	1072
Histosols / David L. Lindbo and Deborah A. Kozlowski	1107
Inceptisols / John M. Galbraith, Kenneth Scheffe, and Shawn McVey	1151
Loess / Fred E. Rhoton and Helaine W. Markewich	1365
Loess Plateau / Fenli Zheng, Ximeng Xu, and Xiubin He	1370
Oxisols / Stanley W. Buol	1635
Peatlands / Dale H. Vitt and Paul Short	1668
Permafrost: Soil Temperature and Special Problems / Douglas Kane and Julia Boike	1687
Petrocalcic Horizons / Alain Ruellan	1691
Plinthite and Petroplinthite / Eswaran Padmanabhan and Hari Eswaran	1743
Podzols / Peter Buurman	1747
Rendzina / Thomas E. Fenton	1907
Serpentinitic Soils / Rebecca Burt and Michael A. Wilson	1999
Spodosols / Delbert Mokma	2200
Terra Preta (Black Earths) / Kurt Spokas, Jose Marques Jr., Newton La Scala Jr., Ed Nater, and Diego S. Siqueira	2312
Terra Preta de Indio / Johannes Lehmann	2316
Ultisols / Joey N. Shaw	2380
Vertisols / Wouter A. Blokhuis	2440
Vertisols: Surface Crack Management / J. Somasundaram, Nishant K. Sinha, Ashok K. Patra, and Rattan Lal	2450

Desertification

Desertification: Biological Productivity Decline / Uriel N. Safriel	608
Desertification: Extent / Victor R. Squires	616
Desertification: Greenhouse Effect / Craig Idso and Sherwood E. Idso	620
Desertification: Humid Environment in Iceland / Andres Arnalds	623
Desertification: Impact / David A. Mouat and Judith Lancaster	628
Desertification: Mapping Constraints and Challenges / Pandi Zdruli, Michael Cherlet, and Claudio Zucca	633
Desertification: Prevention and Restoration / Claudio Zucca, Susana Bautista, and Franco Previtali	642
Desertification: Productivity / Kris M. Havstad	645

Desertification: Restoration / <i>Alon Tal</i>	648
Desertification: Reversal / <i>David Tongway and John Ludwig</i>	654
Drought: Drought Resistance / <i>Graeme C. Wright and Nageswararao C. Rachaputi</i>	693
Drought: Precipitation, Evapotranspiration, and Soil Moisture / <i>Joshua B. Fisher and Konstantinos M. Andreadis</i>	698

Ecological Concerns

Amendments and Ameliorants / <i>David C. McKenzie</i>	114
Biodiversity and Sustainability / <i>Odo Primavesi</i>	205
Climate / <i>Alexander N. Gennadiyev and Serge S. Chernyanskii</i>	435
Ecology: Cycling of Carbon and Nitrogen / <i>Alan J. Franzluebbers</i>	711
Ecosystems: Life Attributes and Environment / <i>Dennis R. Linden</i>	719
Environmentally Friendly Indigenous Technologies / <i>Selim Kapur and Erhan Akça</i>	732
Fire and Soils / <i>Partap K. Khanna and John Raison</i>	926
Landslide and Landcreep / <i>Jau-Yau Lu and Hao-Jung Shieh</i>	1348
Leaching and Illuviation / <i>Lynn E. Moody</i>	1358
Resilience / <i>Moses M. Tenywa, J.G.M. Majaliwa, and A. Lufafa</i>	1909
Resilience: Quality and / <i>Rattan Lal</i>	1918
Resilience: Restoration / <i>F.W.T. Penning de Vries and Eric T. Craswell</i>	1924
Sustainable Development and Sustainability / <i>Alan D. Hecht and Joseph Fiksel</i>	2280

Climate Change

Climate Change: Gas Fluxes / <i>Pascal Boeckx and Oswald Van Cleemput</i>	438
Climate Change: Soil Carbon Sequestration / <i>Sherwood E. Idso and Keith E. Idso</i>	442
Erosion: Global Change / <i>Xingxiang Wang, Taolin Zhang, Longxi Cao, and Yin Liang</i>	804
Greenhouse Effect / <i>Rattan Lal</i>	1048

Conservation

Conservation Tillage / <i>Jack Cline and Robert Hendershot</i>	462
Conserved Lands: Stewardship / <i>Peter August, Janet Coit, and David Gregg</i>	466
Protection: Policy of the European Union / <i>Irene L. Heuser and Luca Montanarella</i>	1823
Soil Conservation: Incentives / <i>David Sanders</i>	2081
Soil Conservation: Strategies / <i>Winfried E.H. Blum</i>	2084
Soil Conservation: U.S. Historical Perspective / <i>Dana D. York</i>	2086

Greenhouse Gases

Ammonia: Volatilization from Agricultural Soils / <i>Paul G. Saffigna and John R. Freney</i>	117
Erosion: Carbon Dioxide / <i>Pierre A. Jacinthe and Rattan Lal</i>	777
Nitrous Oxide Emissions: Agricultural Soils / <i>John R. Freney</i>	1555
Nitrous Oxide Emissions: Paddy Soils / <i>Jianfu Xue, Xin Zhao, and Hailin Zhang</i>	1559
Nitrous Oxide Emissions: Sources, Sinks, and Strategies / <i>Katsu Minami</i>	1562
Power Plants: Carbon Dioxide Capture / <i>Sean I. Plasynski, Thomas J. Feeley, Timothy Fout, Howard G. McIlvried, and Rameshwar D. Srivastava</i>	1798
Rice Paddies: Methane Emissions / <i>Ronald L. Sass</i>	1940
Rice Paddies: Methane Emissions, Mitigating / <i>Kazuyuki Yagi</i>	1943
Salt-Affected Soils: Nitrous Oxide Emissions / <i>G.S. Dheri, B.S. Brar, and Debankur Sanyal</i>	1969
Wetlands: Methane Emission / <i>Anna Ekberg and Torben Røjle Christensen</i>	2596

Ecological Concerns (*cont'd.*)

Pollution

Pollution / <i>Zulkuf Kaya</i>	1758
Pollution: Industrial Waste / <i>Winfried E.H. Blum, Walter W. Wenzel, Wolfgang Friesl-Hanl, and Martin H. Gerzabek</i>	1762
Pollution: Non-Point Source / <i>Ravendra Naidu, Mallavarapu Megharaj, Peter Dillon, Rai Kookana, Ray Correll, and Walter W. Wenzel</i>	1765
Pollution: Organic and Inorganic / <i>A. Paul Schwab</i>	1768
Pollution: Persistent Organic / <i>Mark Radosevich and E. Danielle Rhine</i>	1771
Pollution: Point Source / <i>Ravendra Naidu, Mallavarapu Megharaj, Peter Dillon, Rai Kookana, Ray Correll, and Walter W. Wenzel</i>	1776
Radionuclides / <i>Philip M. Jardine</i>	1876
Surface Water: Pollution by Nitrogen Fertilizers / <i>Gregory McIsaac</i>	2268
Uranium Contamination / <i>Ewald Schnug, Roberta Bottino Montolar Sparovek, Maria del Carmen Lamas, Sylvia L. Kratz, Jurgen Fleckenstein, and Susanne Schroetter</i>	2385
Windblown Dust / <i>Dale Gillette</i>	2644

Restoration and Rehabilitation

Biodegradation and Bioremediation / <i>Owen P. Ward and Ajay Singh</i>	198
Land Restoration / <i>Richard W. Bell</i>	1304
Landfill Sites: Revegetation / <i>Gilbert Yuk-sing Chan and Ming H. Wong</i>	1322
Phytoremediation / <i>Khan Towhid Osman and Md. Abul Kashem</i>	1717
Rehabilitation: Indicators and Monitoring / <i>David Jasper</i>	1899
Restoration Ecology / <i>Richard Hobbs</i>	1932
Restoration: Success and Completion Criteria / <i>Sam C. Ward</i>	1935
Tailings: Minerals Processing Residue Rehabilitation / <i>Lloyd R. Hossner and Hamid Shahandeh</i>	2286
Waste Rock and Overburden Dumps: Rehabilitation / <i>Martin V. Fey, R.D. O'Brien, and A.J. Mills</i>	2464

Erosion

Erosion / <i>Dennis C. Flanagan</i>	742
Erosion Management: Hill Soils / <i>Narendra Kumar Lenka and B. Krishna Rao</i>	746
Erosion Tolerance / <i>David L. Schertz and Mark A. Nearing</i>	753
Erosion: Aggregate Breakdown Mechanisms / <i>Yves Le Bissonnais</i>	757
Erosion: Agronomic Practices / <i>Glenn A. Weesies, David L. Schertz, and William F. Kuenstler</i> ...	762
Erosion: Assessment / <i>John Boardman</i>	767
Erosion: Carbon Dioxide / <i>Pierre A. Jacinthe and Rattan Lal</i>	777
Erosion: Crop Yield / <i>Michael Stocking</i>	786
Erosion: Disturbed Lands / <i>William J. Elliot</i>	790
Erosion: Effects on Human Life / <i>Neroli Pedro Cogo and Renato Levien</i>	794
Erosion: Global Change / <i>Xingxiang Wang, Taolin Zhang, Longxi Cao, and Yin Liang</i>	804
Erosion: On-Site and Off-Site Impacts / <i>Jerry L. Hatfield</i>	811
Erosion: Overland Research and Measurement Techniques / <i>Chia Chun Joseph Wu</i>	814
Erosion: Perennial Crop Plantations / <i>Alfred E. Hartemink</i>	819
Erosion: Soil Conservation / <i>Eric T. Craswell</i>	834
Erosion: Water Quality / <i>Andreas Klik</i>	838
Erosivity and Erodibility / <i>Peter I.A. Kinnell</i>	845
Gully Erosion / <i>Fenli Zheng, Chihua Huang, and Ximeng Xu</i>	1059
Landscape: Development and Erosion / <i>Jon Harbor</i>	1336
Quality: Erosion / <i>Craig Ditzler</i>	1856

Sedimentation

Erosion: Sedimentation Control Amendment Techniques / <i>X.-C. (John) Zhang</i>	823
Erosion: Sedimentation Control Vegetative Techniques / <i>Samson D. Angima</i>	827
Sedimentation Control and Erosion: Engineering Techniques / <i>Jose Miguel Reichert and Elemar Antonino Cassol</i>	1986

Tillage

Tillage Erosion: Description and Process / <i>Michael J. Lindstrom</i>	2330
Tillage Erosion: Measurement Techniques / <i>David A. Lobb</i>	2333
Tillage Erosion: Properties and Productivity Effects / <i>Tom E. Schumacher</i>	2336
Tillage Erosion: Relationship to Water Erosion / <i>Gerard Govers</i>	2339
Tillage Erosion: Terrace Formation / <i>Seth M. Dabney and Dalmo A.N. Vieira</i>	2342

Water

Erosion: Controlling Irrigation-Induced / <i>Robert E. Sojka and David L. Bjorneberg</i>	782
Erosion: Snowmelt / <i>Donald K. McCool</i>	831
Raindrop Impact and Splash Erosion / <i>Hossein Ghadiri</i>	1881
Torrential Erosion: Himalayas / <i>Ram Prashad Yadav</i>	2358
Water Erosion / <i>David Favis-Mortlock</i>	2485
Water Erosion: Empirical Models / <i>John M. Laflen</i>	2489
Water Erosion: Hybrid Models / <i>Jeffrey G. Arnold, Kevin W. King, and Jimmy R. Williams</i>	2494
Water Erosion: Modeling / <i>Calvin W. Rose</i>	2499
Water Erosion: Process-Based Modeling / <i>Mark A. Nearing</i>	2504

Wind

Erosion: Fly-Ash Spheres as Time Marker / <i>Kenneth R. Olson and Alexander N. Gennadiyev</i>	798
Wind Erosion: Climate Change / <i>Alan Busacca and David Chandler</i>	2603
Wind Erosion: Control Measures / <i>Gary L. Tibke</i>	2609
Wind Erosion: Environmental Effects / <i>R. Scott Van Pelt</i>	2614
Wind Erosion: Field Measurement / <i>R. Scott Van Pelt and Ted M. Zobeck</i>	2618
Wind Erosion: Global Hot Spots / <i>Andrew Warren</i>	2625
Wind Erosion: Micrometeorology / <i>Giles F.S. Wiggs</i>	2628
Wind Erosion: Modeling / <i>Lawrence J. Hagen</i>	2632
Wind Erosion: Principles / <i>Larry D. Stetler</i>	2636
Wind Erosion: Soil Quality and Productivity / <i>John Leys</i>	2640
Wind-Driven Rain: Erosion / <i>Gunay Erpul, L. Darrell Norton, and Donald Gabriels</i>	2647

Fertility

Biochar: Soil Carbon and Fertility / <i>Adam O'Toole and Daniel Rasse</i>	193
Crop Plants: Phosphorus Availability / <i>Bettina Eichler-Löbermann and Ewald Schnug</i>	494
Cycling: Carbon and Nitrogen (C/N) / <i>Sylvie M. Brouder and Ronald F. Turco</i>	538
Fertility Decline: Definitions and Assessment / <i>Alfred E. Hartemink</i>	880
Fertility: Environmentally Compatible Management / <i>Bal Ram Singh</i>	884
Fertility: Evaluation Systems / <i>Ram C. Dalal and A. Subba Rao</i>	890
Fertility: Low Input and Traditional Maintenance Methods / <i>Miguel Altieri</i>	894
Fertility: North China Plains / <i>Xiangbin Kong and Rattan Lal</i>	899
Nutrient Management: Fertility and / <i>John L. Havlin</i>	1576
Phosphorus / <i>John Ryan and Abdul Rashid</i>	1700
Phosphorus: Manure and Diet Modification / <i>Rory O. Maguire, John T. Brake, and Peter W. Plumstead</i>	1705

Fertility (*cont'd.*)

Phosphorus: Manure and Diet Modification / Rory O. Maguire, John T. Brake, and Peter W. Plumstead	1705
Phosphorus: Plant Nutrition and Microorganism Contribution / Bettina Eichler-Löbermann, Christel Baum, Silke Ruppel, and Ewald Schnug	1708
Phosphorus: Spatial Speciation in Agricultural Soils / Anja Gassner and Ewald Schnug	1712
Potassium / Philippe Hinsinger	1786
Potassium: Dynamics and Mineralogy / A.V. Shanwal and S.S. Dahiya	1791
Potassium: Nutrition in Semi-Arid Tropics / D.L. Oswalt	1796

Biological Nitrogen Fixation

Biological Nitrogen Fixation / Robert H. Burris	220
Biological Nitrogen Fixation: Contributions to Agriculture / Mark B. Peoples	223
Biological Nitrogen Fixation: Enhancement Techniques / David Francis Herridge	227
Biological Nitrogen Fixation: Forms and Regulating Factors / Ken E. Giller and Paul Mapfumo	232
Nitrogen and its Transformations / Oswald Van Cleemput and Pascal Boeckx	1541
Nitrogen Index / Jorge A. Delgado	1545
Nitrogen-15: Utilization / Jorge Benitez-Vega, Luis López-Bellido, Ramón Redondo, and Rafael J. López-Bellido	1551

Fertilizers

Fertilizers: Application Methods / John Ryan	903
Fertilizers: Leaching Losses / Paul G. Saffigna and Ian R. Phillips	907
Fertilizers: Organic / Gary J. Gascho	916
Fertilizers: Types and Formulations / Douglas Robert McGuffog	919
Fertilizers: Urban Waste and Sludges / Rory O. Maguire and James Thomas Sims	923
Liming and Lime Materials / Mark K. Conyers	1362
Manure Management: Gaseous Emissions / Veerasamy Sejian, Lipismita Samal, M. Bagath, R.U. Suganthi, Raghavendra Bhatta, and Rattan Lal	1400
Manure: Compost and Biosolids / Bahman Eghball and Kenneth A. Barbarick	1406
Nanofertilizers / Ruiqiang Liu and Rattan Lal	1511

Micronutrients

Animals: Microbial Interactions and Nutrient Cycling / Lisa Cole and Richard D. Bardgett	142
Carbon Sequestration: Nutrient Management / M.C. Manna, Muneshwar Singh, R.H. Wanjari, Asit Mandal, and A.K. Patra	288
Integrated Nutrient Management / Bal Ram Singh and Ram C. Dalal	1221
Integrated Nutrient Management: Sustainability / G.S. Saroa and Rattan Lal	1227
Micronutrients and Human Health / Ram Phal Narwal, R.S. Malik, S.K. Malhotra, and Bal Ram Singh	1443
Nutrient Management / Jorge A. Delgado	1570
Nutrient Management: Fertility and / John L. Havlin	1576
Nutrients: Deficiency and Toxicity Symptoms / Noel J. Grundon	1585
Nutrients: Diffusion, Bioavailability, and Plant Uptake / Niels Erik Nielsen	1588
Nutrients: Interactions in Soil—Plant System / Fusuo S. Zhang, J. Shen, and Y.-G. Zhu	1594
Nutrients: Water Interactions / Ardell D. Halvorson	1597
Plant Nutrients / Volker Roemheld	1728
Plant Nutrients: Intensity, Quantity, and Buffer Power / Ram C. Dalal	1731
Plant Nutrients: Sufficiency and Requirements / Kanwar L. Sahrawat	1735
Precision Agriculture: Nutrient Cycling / Robert J. Lascano	1811

Soil Organic Matter (SOM): Nutrient Dynamics / Charles W. Rice	2172
Variable Charge Soils: Mineralogy and Chemistry / Eric Van Ranst, Nikolla P. Qafoku, Andrew Noble, and Ren-kou Xu	2432

Minerals

Boron and Molybdenum / Umesh C. Gupta and J.A. MacLeod	252
Copper / Judith F. Pedler and David R. Parker	481
Food Crops: Zinc and Iron Biofortification / Yashbir Singh Shivay and Rajendra Prasad	933
Iron Oxides / Nikolla P. Qafoku and James Amonette	1245
Selenium / Gunnar Gissel-Nielsen	1995
Silica: Pedogenic Accumulation / Katherine J. Kendrick	2012
Silicon and Sodium / Jian Feng Ma	2015

General Topics

Accounting: Emergy Analysis / J.F. Martin and D.R. Tilley	1
Care of Soil / Delia C. Catacutan	342
Economics of Soil Management: U.S. / Jeffrey W. Hopkins	716
Ethics / John Ronald Engel	848
Geology and Soil / Kenneth R. Olson	997
Land Forms / Carolyn G. Olson	1294
Landscape / George Hall	1327
Landscape: Classification / Philip Schoeneberger	1331
Landscape: Elements / Douglas Wysocki and C. William Zanner	1339
Landscape: Relationships / Timothy A. Quine	1344
Materials and Soils / Selim Kapur, Erhan Akça, and Takanori Nagano	1416
Rocks / Vsevolod V. Dobrovolsky	1953
Time and Space: Soils in / Neil E. Smeck and C. Lee Burras	2353

History

Future of Soil Science / Eric C. Brevik, Samuel J. Indorante, Dylan E. Beaudette, and Richard W. Arnold	982
History of Soil Science / Eric C. Brevik and Artemi Cerdà	1093
History of Soil Science: 1927–2000 / Fred P. Miller	1098
History of Soil Science: Early to Mid–20th Century / John P. Tandarich	1103
Land Use: Historic / J. Douglas Helms, Hari Eswaran, and Paul Reich	1308

Human Relationship to Soil

Archaeology / John E. Foss	154
Art and Soil / Christian Feller	168
Degradation: Food Security and Poverty / Pierre Crosson	574
Erosion: Effects on Human Life / Neroli Pedro Cogo and Renato Levien	794
Food Security and Soil Water Management / Mark W. Rosegrant, Ephraim Nkonya, and Rowena A. Valmonte-Santos	939
Health / Robin F. Harris and D.E. Romig	1076
Human Culture and Soils / Daniel Hillel	1112
Human Dimensions: Soil Sustainability / Mary Seely, Patrik Klintonberg, and Claus Hager	1115
Human Health and Soil / Charles F. Sing and David B. Sing	1118
Human Society and Soil / Rattan Lal	1123
Human Society and the Planet: Soil as a Heritage / Fred P. Miller	1127
Law and Legal Issues / Ian Hannam and Ben Boer	1352
Law and Legal Issues: Land as Property / Paul Martin and John Page	1355

General Topics (*cont'd.*)

Micronutrients and Human Health / <i>Ram Phal Narwal, R.S. Malik, S.K. Malhotra, and Bal Ram Singh</i>	1443
Modern Civilization and Soils / <i>Kenneth R. Olson</i>	1491
Religious Aspects of Soil—Human Relationships / <i>Verena Winiwarter and Winfried E.H. Blum</i>	1903
Social and Behavioral Sciences' Contribution to Soil Science / <i>Sven B. Lundstedt</i>	2026
Value of Soil to Humans / <i>Rattan Lal</i>	2416

Resources

Evolution: Soil Seed Bank / <i>Paul B. Cavers and David Susko</i>	868
Global Resources / <i>Paul Reich and Hari Eswaran</i>	1020
International Soil Reference and Information Centre (ISRIC) / <i>David Dent</i>	1232
International Union of Soil Sciences / <i>Stephen Nortcliff</i>	1237
Natural Resources: Interaction with Soil / <i>Dennis J. Greenland</i>	1516
Nexus Approach: Resource Management for Soil Productivity / <i>Kai Schwärzel, Reza Ardakanian, Tamara Avellán, and Lulu Zhang</i>	1530
World Reference Base for Soil Resources / <i>Peter Schad and Stefaan Dondeyne</i>	2650

Land Use

Land Use: Land Cover / <i>Andrew K. Skidmore</i>	1313
Land Use: Planning and Geographic Information System (GIS) / <i>Egide Nizeyimana and Jacob Opadeyi</i>	1318
Resilience: Land Use and Soil Management / <i>Soojin J. Park</i>	1913

Croplands and Grazing

Bioenergy Crops: Carbon Balance Assessment / <i>Rocky Lemus and Rattan Lal</i>	210
Cover Crops / <i>Jorge A. Delgado, D. Wayne Reeves, and Ronald F. Follett</i>	484
Crop Evapotranspiration: Measurement System / <i>Huanjie Cai, Pute Wu, Zhijun Li, Xiaotao Hu, and Yufeng Zou</i>	488
Crop Management: Seasonally Frozen Soil / <i>Brenton S. Sharratt</i>	492
Crop Plants: Phosphorus Availability / <i>Bettina Eichler-Löbermann and Ewald Schnug</i>	494
Crop Residue Management / <i>David L. Schertz and Dan Towery</i>	497
Crop Root Response to Temperature: Temperate Zone / <i>Thomas C. Kaspar</i>	501
Crop Rotation and Farming Systems: Temperate Zone / <i>Martin R. Carter</i>	504
Crop Rotation and Farming Systems: Tropical Smallholder Farm Systems / <i>Stefan Hauser, Lindsey Norgrove, and Tarcisius Nyobe</i>	507
Crop Rotation and Farming Systems: Tropics and Subtropics and Swelling Soils / <i>D.M. Freebairn, G.D. Smith, Robin D. Connolly, and Jeff N. Tullberg</i>	512
Croplands / <i>B.A. Stewart</i>	522
Crops: Flooding Tolerance / <i>Tara T. VanToai, Getachew Boru, and Jianhuan Zhang</i>	526
Cultivation: Raised-Bed / <i>Kenneth D. Sayre</i>	529
Eco-Agriculture / <i>Norman Uphoff and Janice E. Thies</i>	706
Grazing Lands: Gaseous Emissions / <i>Lipismita Samal, Veerasamy Sejian, M. Bagath, R.U. Suganthi, Raghavendra Bhatta, and Rattan Lal</i>	1034
Land Husbandry / <i>Francis Shaxson and Malcolm Douglas</i>	1297
Mulch Farming / <i>Rattan Lal</i>	1495
Nanofertilizers / <i>Ruiqiang Liu and Rattan Lal</i>	1511
Organic Agriculture / <i>Kathleen Delate</i>	1600

Food Crops

Food Crops: Zinc and Iron Biofortification / <i>Yashbir Singh Shivay and Rajendra Prasad</i>	933
Food Security: Agricultural Productivity and Resources / <i>Keith D. Wiebe</i>	942
Rice Paddies: Methane Emissions / <i>Ronald L. Sass</i>	1940
Rice Paddies: Methane Emissions, Mitigating / <i>Kazuyuki Yagi</i>	1943
Rice Paddies: Methanogenesis / <i>Ronald L. Sass</i>	1947
Rice Paddies: Soil Management / <i>Xin Zhao, Jianfu Xue, and Hailin Zhang</i>	1950

Precision Agriculture

Precision Agriculture: Engineering Aspects / <i>Joel T. Walker, Reza Ehsani, and Matthew O. Sullivan</i>	1804
Precision Agriculture: Fertilization / <i>Silvia H. Haneklaus and Ewald Schnug</i>	1808
Precision Agriculture: Nutrient Cycling / <i>Robert J. Lascano</i>	1811
Precision Agriculture: Remote Sensing / <i>Kerstin Panten, Holger Lilienthal, Erik Zillmann, Silvia H. Haneklaus, and Ewald Schnug</i>	1815

Forest Land

Boreal Forest / <i>Galina Mazhitova</i>	247
Forest Soils / <i>Nicholas Comerford and Thomas Fox</i>	955
Forest Soils: Calcium Depletion / <i>Tom G. Huntington</i>	959
Forest Soils: Major / <i>Ken Van Rees</i>	963
Forest Soils: Nutrient Cycling / <i>Neil W. Foster and Jagtar S. Bhatti</i>	966
Forest Soils: Properties and Site Productivity / <i>Paul A. Arp and Helmut H. Krause</i>	969
Mycorrhiza of Forest Ecosystems / <i>Ian A. Dickie</i>	1502

Mine Land

Acid Mine Drainage / <i>Jerry M. Bigham and Charles A. Cravotta III</i>	6
Mine Soils: Miscanthus Plantations / <i>Jose G. Guzman and Rattan Lal</i>	1458
Mine Soils: Open Cut Mine Rehabilitation / <i>Douglas J. Dollhopf</i>	1462
Mine Soils: Measuring Geogenic Carbon / <i>David A.N. Ussiri and Rattan Lal</i>	1449
Mine Soils: Reclamation and Soil Carbon Sequestration / <i>Vasant A. Akala and Rattan Lal</i>	1466

Urban Land

Carbon Sequestration: Urban Ecosystems / <i>Adam Louis Selhorst</i>	307
Fertilizers: Urban Waste and Sludges / <i>Rory O. Maguire and James Thomas Sims</i>	923
Industrial Waste / <i>Warren Dick</i>	1178
Urban Lands: Abandoned / <i>Joshua W. Beniston and William D. Shuster</i>	2388
Urban Lands: and Construction Sites / <i>Wolfgang Burghardt</i>	2395
Urban Lands: Management / <i>Klaus Lorenz</i>	2400
Urban Wastewater for Irrigation: Heavy Metals / <i>P.S. Minhas and K. Lal</i>	2407

Wetlands

Wetlands / <i>Ralph W. Tiner</i>	2567
Wetlands: Biodiversity / <i>Jean-Claude Lefevre and Virginie Bouchard</i>	2572
Wetlands: Conservation Policy / <i>Clayton Rubec</i>	2580
Wetlands: Economic Value / <i>Gayatri Acharya</i>	2584
Wetlands: Gaseous Emissions / <i>S.K. Nag</i>	2589
Wetlands: Sedimentation and Ecological Engineering / <i>Timothy C. Granata and Jay F. Martin</i> . . .	2600
Wetlands: Methane Emission / <i>Anna Ekberg and Torben Røjle Christensen</i>	2596
Wetlands: Carbon Sequestration / <i>Virginie Bouchard and Matthew Cochran</i>	2576

Managing Constraints

Acidity

Acid Mine Drainage / Jerry M. Bigham and Charles A. Cravotta III	6
Acid Rain: Nitrogen Deposition / George F. Vance	11
Acid Sulfate Soils / Delvin S. Fanning	17
Acid Sulfate Soils: Distribution and Extent / Wim Andriesse and M.E.F. van Mensvoort	20
Acid Sulfate Soils: Formation / Martin C. Rabenhorst, Delvin S. Fanning, and Steven N. Burch	26
Acid Sulfate Soils: Management / Michael D. Melville, Ian White, and Robert Quirk	31
Acid Sulfate Soils: Problems / Martin C. Rabenhorst and Delvin S. Fanning	36

Air

Aeration: Measurement / Robert E. Sojka and H.D. Scott	44
Aeration: Tillage Effects / Dave J. Horne and Robert E. Sojka	47
Air Permeability / Manoj K. Shukla and Rattan Lal	85
Anaerobic Processes / Paul A. McDaniel and Daniel G. Strawn	126
Respiration / Pierre A. Jacinthe and Rattan Lal	1928

Carbon

Agricultural Soil Carbon: Trading / Bruce A. McCarl, Sung Ju Cho, Jinxiu Ding, Pei Huang, Layla Shiva, and Chin-Hsien Yu	76
Biochar: Soil Carbon and Fertility / Adam O'Toole and Daniel Rasse	193
Mapping Soil Carbon / Budiman Minasny, Brendan Malone, and Alex B. McBratney	1409
Mapping Soil Carbon: Stocks in Turkey / Gönül Aydın, Mehmet Ali Çullu, Sabit Erşahin, Erhan Akça, Emrah Erdoğan, Levent Atatanır, Alper Yorulmaz, Ahmet Çilek, Merve Ersoy, Somayyeh Rezzaghi Miavaghi, Selim Kapur, and Rattan Lal	1412
Soil Carbon Map: Africa / Arwyn Jones, Roland Hiederer, and Luca Montanarella	2059
Soil Carbon Sequestration: Ethiopia / Ambachew Demessie, Bal Ram Singh, and Rattan Lal	2066
Soil Carbon: Compositional and Isotopic Analysis / James Moran, M. Lizabeth Alexander, and Alexander Laskin	2073
Soil Carbon: Farming and Economic Aspects / Bruce A. McCarl	2078

Compaction

Compaction / William E. Wolfe	456
Crop Yield: Compaction / Muhammad Ishaq and Rattan Lal	516

Drainage

Acid Mine Drainage / Jerry M. Bigham and Charles A. Cravotta III	6
Drainage: Aeration and Trafficability / Norman R. Fausey	680
Drainage: Anaerobiosis / Robert O. Evans	683
Drainage: Water Quality / James L. Baker	689

Water

Turfgrass: Protection of Soil and Water / James A. Murphy and Stephanie L. Murphy	2377
Water Conservation / Paul W. Unger	2472
Water Content / Tom J. Jackson	2475
Debris Flow / Jau-Yau Lu	544
Water Content: Passive and Active Microwave Remote Sensing / Eric D. Warner and Gary W. Petersen	2480
Water Infiltration / Manoj K. Shukla and Rattan Lal	2507

Water Management: Computer Modeling / Laj R. Ahuja and Liwang Ma	2510
Water Management: Use Efficiency / David C. Nielsen	2518
Water Measurement: Methods / Des McGarry	2522
Water Range: Least-Limiting / Alvaro Pires da Silva, Beverly D. Kay, Cássio A. Tormena, Silvia Imhoff, Douglas L. Karlen, and Joseph G. Benjamin	2530
Water Retention / Satish C. Gupta and Dong Wang	2535
Water: Plant-Available / Argyrios Gerakis and Joe T. Ritchie	2541
Water-Repellent Soils / Anna Eynard, Rattan Lal, and Keith D. Wiebe	2546
Watershed Approach / Timothy R. Green and Jan van Schilfgaarde	2550
Watershed Management: Remote Sensing and GIS Applications / A.V. Shanwal and S.P. Singh	2555

Measurements, Testing, and Modeling

Color / Robin N. Thwaites	452
Conditioning Index / Ted M. Zobeck, M. Lee Norfleet, and Douglas L. Karlen	459
Consistence / Ray McBride	471
Diagnostic Horizons / E.M. Bridges	668
Geomorphology / Robin N. Thwaites	1000
Geophysical Methods / Barry J. Allred, Viacheslav I. Adamchuk, Raphael A. Viscarra Rossel, Jim Doolittle, Robert S. Freeland, Katherine R. Grote, and Dennis L. Corwin	1004
Geostatistics / Anja Gassner and Ewald Schnug	1012
Hydraulic Conductivity / Jacob H. Dane, Marc Jalbert, and Jan W. Hopmans	1133
Land Capability Analysis / Michael J. Singer	1285
Land Capability Classification / Thomas E. Fenton	1288
Land Evaluation / J. Herbert Huddleston	1291
Pedotransfer Functions / J.H.M. Wösten	1682
Plastic Properties / Gunnar Kirchhof	1740
Radiometric Survey / Barry G. Rawlins and Raphael A. Viscarra Rossel	1872
Rapid Soil Analysis: Diffuse Reflectance Spectroscopy / Keith D. Shepherd and Markus G. Walsh	1886
Soil Organic Matter (SOM): Mass Spectrometry Data Analysis / Nikola Tolic, Malak M. Tfaily, Yufeng Shen, Rosalie Chu, Thomas Fillmore, Rui Zhao, Kent Bloodsworth, Errol Robinson, Ljiljana Paša-Tolić, and Nancy J. Hess	2154
Testing / Yash P. Kalra and Jagtar S. Bhatti	2320
Texture / Harold R. Geering and Hwat Bing So	2323
Variability / Josef H. Görres	2419
Variability: Spatial / Brian Slater	2428

Mapping

Digital Soil Mapping / Budiman Minasny, Alex B. McBratney, and Florence Carré	675
GIS Integration: Remote Sensing / Egide Nizeyimana	1017
Satellite Mapping / Bruce E. Frazier	1972

Measurements

Amoozemeter / Aziz Amoozegar	120
Atterberg Limits / Thomas Baumgartl	171
Diffusivity: Measurement / Toru Nakajima	672
Microbial Biomass: Measurement / K.R. Islam and S.R. Wright	1429
Minirhizotron / Verónica Muñoz-Romero, Luis López-Bellido, F. Javier López-Bellido, and Rafael J. López-Bellido	1484
Pedometrics / Alex B. McBratney and Budiman Minasny	1677

Measurements, Testing, and Modeling (*cont'd.*)

Modeling

Fluid Flow: Modeling / John L. Nieber	930
Inorganic Carbon: Modeling / Leslie D. McFadden and Ronald G. Amundson	1213
Pedological Modeling / Ronald G. Amundson	1674
Soil Organic Matter (SOM): Modeling / Peter Smith	2159
Water Erosion: Empirical Models / John M. Laflen	2489
Water Erosion: Process-Based Modeling / Mark A. Nearing	2504
Water Management: Computer Modeling / Laj R. Ahuja and Liwang Ma	2510
Wind Erosion: Modeling / Lawrence J. Hagen	2632

Particle Measurements and Structure

Bulk Density / Kwong Yin Chan	258
Macroporosity / Daniel Gimenez and Daniel Hirmas	1388
Micromorphology and Soil Quality / Ahmet R. Mermut	1439
Particle Density / Keith R.J. Smettem	1652
Particle Packing / Satish C. Gupta and D.P. Thoma	1654
Particle Shape / Brian M. Schafer	1659
Particle Size / Hwat Bing So and Harold R. Geering	1661
Porosity: Pore Size Distribution / Keith C. Cameron and Graeme D. Buchan	1782
Structure / Eileen J. Kladviko	2207
Structure: Belowground Management / Ken A. Olsson and Bruce Cockroft	2209
Structure: Earthworms / Martin Shipitalo and Tayfun Korucu	2212
Structure: Pedological Concepts and Water Flow / Clinton C. Truman and Donald P. Franzmeier	2216
Structure: Plant Establishment / Douglas L. Karlen	2221
Structure: Roots / Alvin J.M. Smucker	2225
Surface Area: Specific / L.A. Graham Aylmore	2255

Quality

Quality / Ed G. Gregorich	1837
Quality Index: Soil / Toru Nakajima	1841
Quality: Assessment by Interpolation Techniques / Vincent Obade	1844
Quality: Critical Limits and Standardization / Martin R. Carter	1853
Quality: Indicators / Graham P. Sparling	1859
Quality: Productivity / Mike H. Beare	1862
Quality: Soil and Water / Todd Nissen and Michelle Wander	1866

Temperature

Heat and Water Movement / Gerald N. Flerchinger	1079
Heat Capacity / Surinder Kumar Jalota and B.S. Ghuman	1083
Heat Flux / Thomas J. Sauer	1086
Temperature Measurement / Kevin McInnes	2302

Physical Attributes

Albedo / Endre Dobos	89
Controlled Traffic / Randall C. Reeder	477
Eutrophication / Thomas G. Franti and Kyle D. Hoagland	855
Gas and Vapor Phase Transport / Dennis E. Rolston	986
Productivity: Natural and Anthropogenic Disturbances / Douglas G. Maynard and Charlotte E. Norris	1819

Scaling Effects / <i>Yakov A. Pachepsky, Walter J. Rawls, and John W. Crawford</i>	1976
Sediment: Influence on Aquatic Life / <i>Suzanne M. Gray</i>	1981
Seepage / <i>Patricia M. Gallagher</i>	1990
Shrinkage / <i>Des McGarry and Don F. Yule</i>	2008
Slaking, Dispersion, and Crust Formation / <i>Hwat Bing So</i>	2021
Surface Management / <i>Kwong Yin Chan</i>	2259
Turbations / <i>Brad D. Lee and Robert C. Graham</i>	2373
Vesicular Crusts / <i>Patrick Drohan</i>	2456
Weathering / <i>Dominique Righi</i>	2559

Processes

Aggregation

Aggregates: Tensile Strength / <i>Humberto Blanco-Canqui and Rattan Lal</i>	51
Aggregation / <i>Sjoerd W. Duiker</i>	55
Aggregation: Soil Organic Matter (SOM) and / <i>Cynthia A. Cambardella</i>	58
Aggregation: Structural Stability Measurement / <i>Edmund Perfect, Martín Díaz-Zorita, and John Grove</i>	61
Erosion: Aggregate Breakdown Mechanisms / <i>Yves Le Bissonnais</i>	757

Consolidation

Consolidation / <i>Tarunjit Singh Butalia</i>	474
--	-----

Degradation

Brick Making: Soil Degradation / <i>G.S. Dheri, Babu S. Brar, and Debankur Sanyal</i>	255
Degradation / <i>Rosa M. Poch Claret and José A. Martínez-Casasnovas</i>	548
Degradation: Biological / <i>Ibrahim Ortas</i>	553
Degradation: Chemical / <i>M. Rifat Derici</i>	558
Degradation: Critical Limits and Irreversible Degradation / <i>Anthony R. Dexter and Michael A. Zebisch</i>	561
Degradation: Economic Incentives and Investments / <i>Dennis Wichelns</i>	565
Degradation: Farm-Level Economic Implications / <i>Dennis Wichelns and Ellen Burnes</i>	568
Degradation: Food Aid Needs in Low-Income Countries / <i>Shahla Shapouri and Stacey Rosen</i>	571
Degradation: Food Security and Poverty / <i>Pierre Crosson</i>	574
Degradation: Greenhouse Effect / <i>Eddy De Pauw and Michael A. Zebisch</i>	578
Degradation: Mapping by Normalized Difference Vegetation Index (NDVI) / <i>Zhanquo Bai and David Dent</i>	581
Degradation: Off-Farm Economic Implications / <i>Dennis Wichelns and Ellen Burnes</i>	593
Degradation: Physical / <i>Michael A. Zebisch and Anthony R. Dexter</i>	596
Degradation: Quality and / <i>Rattan Lal</i>	602
Food Security: Soil Degradation, Global Scale / <i>Michael A. Zebisch and Eddy De Pauw</i>	945
Food Security: Soil Degradation, Local and Eco-Regional Scale / <i>Sara J. Scherr</i>	950
Soil Degradation: Global Assessment / <i>Selim Kapur and Erhan Akça</i>	2090

Regions

Alpine Soils / <i>Jérôme Poulénard and Pascal Podwojewski</i>	101
Developing Countries: Economics of Soil Management / <i>Stefano Pagiola</i>	664
Indigenous Soil Management / <i>Karl Herweg</i>	1164
Polar Regions, The / <i>Charles Tarnocai and Iain Campbell</i>	1753
Tropics and Subtropics / <i>Hari Eswaran and Eswaran Padmanabhan</i>	2369

Regions (*cont'd.*)

Africa

Soil Carbon Map: Africa / Arwyn Jones, Roland Hiederer, and Luca Montanarella	2059
Soil Carbon Sequestration: Ethiopia / Ambachew Demessie, Bal Ram Singh, and Rattan Lal . . .	2066

Americas

Amazon Basin Soils / Stanley W. Buol	111
Cerrado / Igo F. Lepsch	345
Low-Carbon Technology: Brazilian Semiarid Ecosystems / Antonio Pereira Filho, Vanderlise Giongo, Tony Jarbas Ferreira Cunha, Jose Texeira Filho, Tayane de Lima Santos, and Luiz Fernando Carvalho Leite	1377
Manganese: Brazilian Soils / Diego Antonio França de Freitas, Nilton Curi, and Nestor Kämpf	1396
Pampas, The / Martín Díaz-Zorita and Daniel E. Buschiazso	1638
Pantanal, The / Henrique de Oliveira, Amaury de Carvalho Filho, Carlos Ernesto Reynauld G. Schaefer, and Evaldo Luis Cardoso	1643
Paramos Soils / Pascal Podwojewski and Jérôme Poulenard	1648
Tepetates: Mexico / Jorge D. Etchevers, Claudia Hidalgo, C. Prat, and P. Quantin	2305

Asia

Arsenic: Southeast Asia / Mohammad Mahmudur Rahman and Ravi Naidu	161
Fertility: North China Plains / Xiangbin Kong and Rattan Lal	899
Green Revolution: China North Eastern Plains / Xiangbin Kong and Rattan Lal	1042
India: Problems and Potentialities / S.K. Singh, S. Chatteraj, N.G. Patil, S.K. Ray, and S. Chatterji	1157
Indo-Gangetic Plain: Agro-Ecological Regions / Jaya N. Surya, R.P. Yadav, G.S. Sidhu, and S.K. Singh	1168
Jhum: Cultivation / Arun Jyoti Nath, Biplab Brahma, Rattan Lal, and Ashesh Kumar Das . . .	1273
Jhum: Shifting Agriculture / P.S. Ramakrishnan	1281
Lower Himalayas: Soil Management / Anup Das, Ramkrushna Gi, Badapmain Makdoh, Dibyendu Sarkar, Jayanta Layek, Sandip Mandal, and Rattan Lal	1382
Organic Farming: China / Yuxin Miao, Fusuo Zhang, and Fanghao Wang	1618

Europe

Andisols: Iceland / Olafur Arnalds	134
Mapping Soil Carbon: Stocks in Turkey / Gönül Aydın, Mehmet Ali Çullu, Sabit Erşahin, Erhan Akça, Emrah Erdoğan, Levent Atatanır, Alper Yorulmaz, Ahmet Çilek, Merve Ersoy, Somayyeh Rezzaghi Miavaghi, Selim Kapur, and Rattan Lal	1412
Mediterranean Agriculture: Rainfed, Nitrogen Use Efficiency / Àngela D. Bosch Serra	1421

Soil Water

Acid Mine Drainage / Jerry M. Bigham and Charles A. Cravotta III	6
Drainage: Aeration and Trafficability / Norman R. Fausey	680
Drainage: Anaerobiosis / Robert O. Evans	683
Drainage: Water Quality / James L. Baker	689
Evaporation / William P. Kustas	858
Freeze and Thaw: Diurnal / Joseph L. Pikul	973
Frozen Soils: Water Relations / John M. Baker	976
Grass Strip Hydrology / Hossein Ghadiri and Calvin W. Rose	1024
Hydrologic Cycle / Keith R.J. Smettem	1137
Hydropedology / Henry Lin	1140

Hydrophobic and Hydrophilic Compounds / Tomas Simon	1144
Hydrophobicity / Joerg Bachmann, Abraham Marmur, and Markus Deurer	1147
Infiltration: Management / Jutta Rogasik, Kerstin Panten, Ewald Schnug, and Helmut Rogasik	1183
Infiltration: Properties / Walter J. Rawls	1187
Surface Water: Nitrogen Enrichment / Tim P. Burt and Penny E. Widdison	2263
Water Management: Harvesting / Gary W. Frasier	2515

Irrigation

Agricultural Lands: Flooding and Levee Breaches / Kenneth R. Olson and Lois Wright Morton	65
Crops: Flooding Tolerance / Tara T. VanToai, Getachew Boru, and Jianhuan Zhang	526
Irrigation / Jack Keller	1250
Irrigation: Efficiency / Terry A. Howell	1256
Irrigation: Erosion / Robert E. Sojka and David L. Bjorneberg	1261
Irrigation: Historical Perspective / Robert E. Sojka, David L. Bjorneberg, and J.A. Entry	1264
Irrigation: Soil Salinity / James D. Rhoades	1269
Sodic Soils: Irrigation Farming / David Burrow	2041
Sub-Irrigation / Norman R. Fausey	2228
Urban Wastewater for Irrigation: Heavy Metals / P.S. Minhas and K. Lal	2407
Wastewater: Treatment and Utilization in Irrigation / Yona Chen and Jorge Tarchitzky	2467



Taylor & Francis

Taylor & Francis Group

<http://taylorandfrancis.com>

Preface

Six important challenges facing humanity at the start of the twenty first century are: (1) a world population of 7.4 billion in 2016 projected to be 9.7 billion by 2050 and 11 billion by 2100; (2) a food-insecure population of 795 million in 2015 mostly in sub-Saharan Africa and South Asia; (3) severe and extreme forms of soil degradation affecting more than 300 million hectares (Mha) of agricultural lands, continuing unabated in developing countries where the resource-poor farmers cannot afford to invest in soil restoration; (4) an anthropogenic increase in atmospheric concentration of greenhouse gases such as CO₂, with current levels of 400 ppm in 2015 and increasing at the rate of 0.48% per year; CH₄ currently at 1.83 ppm and increasing at the rate of 0.49% per year; N₂O at 327 ppb and increasing at the rate of 0.34% per year; and atmospheric C pool at 840 Pg in 2016 and increasing at the rate of 4.4 Pg/yr; (5) a projected 40% decrease in the global per capita arable land area between 2000 and 2050; and (6) the threat of a decreasing, renewable fresh-water supply, affecting 2.3 billion people in 2000 and 3.5 billion by 2025. World soils, and their management, plays an important role in addressing these and other issues, especially those related to food security and climate change. Yes, soils matter.

Natural ecosystems have been drastically altered by anthropogenic perturbations due to increase in human population from about 0.2 billion in 1 A.D. to one billion in 1850, 6 billion in 1998, 7 billion in 2011, and 7.4 billion in 2016. Consequently, world forest resources have decreased from 55.3×10^6 Km² to 43.9×10^6 Km², world savannas from 33.4×10^6 Km² to 26.7×10^6 Km², and shrub land from 17.9×10^6 Km² to 15.9×10^6 Km² from prehistoric times to 1990. The present rate of tropical deforestation is about 7.8 million hectare per year with remaining forest area of <4 billion ha by 2015. Use of chemicals in agricultural ecosystems has increased drastically and, by 2012 chemical use included 125×10^6 Mg per year of N, 46×10^6 Mg per year for P and 25×10^6 Mg per year of potash. The annual need for N fertilizer is projected to be 135×10^6 Mg by the year 2020 and 236×10^6 Mg by 2050. Global water consumption has increased from 579 Km³ per year in 1900 to 3973 Km³ by 2000 of which 66% is for agriculture. Soil degradation affects 1.9 billion ha and is increasing due to anthropogenic activities and the projected global warming. Land degradation affects 24% of the global land area as of 2008. Therefore, need for understanding of soil processes is critical to restoring degraded/polluted soils, identifying sustainable systems of soil for food security and climate change.

The solution to these challenges lies in developing management strategies based on understanding the nature, properties, and dynamics of the life-support processes of the most basic of all natural resources ... the soil. This thin upper surface of the Earth's crust supports life in numerous ways—moderation of climate, filtration of chemicals, and the purification of water, detoxification of pollutants, restoration and resilience of ecosystems, and cycling of basic elements such as carbon, nitrogen, phosphorus, and sulfur. These life support functions of soil are precisely summed up in an old Chinese proverb that states “Man ... despite his artistic pretensions and many accomplishments ... owes his existence to a six-inch layer of topsoil and the fact that it rains.”

Traditionally, soil was regarded as a medium for plant growth, a repository of gene pool, an engineering foundation, an industrial raw material, and an archive of planetary history. However, to meet the challenges of the post-modern era of the twenty first century world soils must now also (1) maximize long-term productivity; (2) moderate moisture storage and distribution; (3) moderate climate and proxy for interpretation of past global changes and cultural heritage; (4) improve water resources; and (5) minimize environmental perturbation by functioning as a repository for waste disposal, an ameliorator of water quality through bioremediation, a sink for atmospheric carbon and other greenhouse gases, a component of biogeochemical cycles and other ecological processes, and a medium for immersing and loading infrastructure. To enhance soil's capacity to perform these functions, it is important to understand factors and processes affecting soil quality and health under growing and conflicting modes of land uses (among them agricultural, forestry, urban, industrial, military, aesthetic and cultural).

It is necessary that state-of-the-knowledge information about these complex attributes of soil and its multi-faceted functions of interest to humans is readily available to the scientific community, policy makers, and the public at large in an easily accessible compendium such as this Encyclopedia.

The *Third Edition* of the *Encyclopedia of Soil Science* has been compiled by over 500 experts in their fields. The *Third Edition* has been updated and includes more than 600 entries, compared with 458 in the *Second Edition* and 346 in the *First Edition*. These entries include discussions of important subjects prepared by soil scientists from around the world. The topics cover all branches of soil science, including pedology, mineralogy, physics, soil mechanics, hydrology, chemistry, biology, ecology and management, and restoration of problem or degraded soils, and address numerous challenges including soil structure, tillage methods, and mulch farming, irrigation, drainage and water table management, fertilizer and nutrient management, erosion control, climate change, soil carbon sequestration, and management of soil organic matter. Entries are alphabetically arranged within a theme, and a subject is included for easy access.

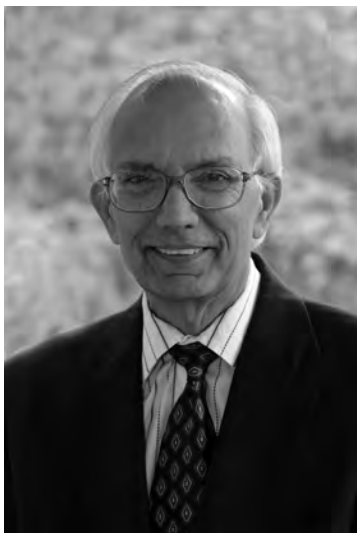
Preparation of this state-of-the-knowledge compendium on soil science has been made possible by hard work, vision, and commitment of hundreds of soil scientists from around the world who have contributed as members of the advisory board, as topic editors, or as authors. I thank all of them for their outstanding efforts to prepare the Third Edition in record time. I am thankful to all topic editors for identifying expert authors, for coordinating the writing with minimal duplication, and for carefully reviewing the manuscripts to ensure high quality. I also thank all the authors for their outstanding efforts in documenting and presenting, in a timely fashion, the scientific information supporting the current understanding of their field of specialization. Their efforts in the preparation of this volume provide an up-to-date information on the nature, properties, functions, and dynamics of soil.

Thanks are also due to the staff of Taylor and Francis for their efforts in making this information available to the world community. Ms. Sapna Maloor helped in the organization of the First Edition, Ms. Susan Lee in preparation of the Second Edition, Mr. Michael Powers and Mr. Daniel Vazquez and Ms. Claire Miller for the Third Edition. Their professionalism, commitment to excellence, and dedication to the mission is much appreciated. I also thank Mrs. Laura Conover of the Carbon Management and Sequestration Center at the Ohio State University for her efforts in handling the flow of manuscripts in the review process of the Third Edition. These efforts have produced a Third Edition of the *Encyclopedia of Soil Science* that contains information useful to meeting the challenges of the twenty first century and beyond.

Rattan Lal, Editor-in-Chief
Director, Carbon Management and Sequestration Center
The Ohio State University
Columbus, Ohio, U.S.A.

About the Editors

Dr. Rattan Lal



Rattan Lal, Ph.D., is a Distinguished University Professor of Soil Science and Director of the Carbon Management and Sequestration Center, The Ohio State University, and an Adjunct Professor of University of Iceland. With completion of education from Punjab Agricultural University, Ludhiana (B.Sc, 1963), Indian Agricultural Research Institute, New Delhi (M.Sc, 1965), and the Ohio State University, Columbus (Ph.D, 1968), he served as Sr. Research Fellow with the University of Sydney, Australia (1968–69), Soil Physicist at IITA, Ibadan, Nigeria (1969–87), and Professor of Soil Science at OSU (1987–to date). His current research focus is on climate-resilient agriculture, soil carbon sequestration, sustainable intensification, enhancing use efficiency of agroecosystems, and sustainable management of soil resources of the tropics. He is a fellow of the American Society of Agronomy (ASA, 1985), Soil Science Society of America (SSSA, 1986), Third World Academy of Sciences (1992), American Association for the Advancement of Sciences (1996), Soil and Water Conservation Society (SWCS, 1997), Indian Natl. Academy of Agricultural Sciences (1998), and Rothamsted, U.K. (2013). He

received the Hugh Hammond Bennett Award of the SWCS, 2005; Borlaug Award (2005), and Liebig Award (2006) of the International Union of Soil Sciences, M.S. Swaminathan Award (India) of 2009, and COMLAND Award (Germany, 2009). He received honorary degree of Doctor of Science from Punjab Agricultural University (2001), the Norwegian University of Life Sciences, Aas (2005), Alecu Russo Balti State University, Moldova (2010), and Technical University of Dresden Germany (2015) and cited in Thomas Reuters 2014 and 2015 list of World's Most Influential Scientific Minds. He was president of the World Association of the Soil and Water Conservation (1987–1990), the International Soil Tillage Research Organization (1988–91), the Soil Science Society of America (2005–2007), and is President of the International Union of Soil Sciences, Vienna, Austria (2017–2018). He was a member of the Federal Advisory Committee on National Assessment of Climate Change-NCADAC (2010–2014), member of the SERDP Scientific Advisory Board of the Department of Defense (2011–), Senior Science Advisor to the Global Soil Forum of IASS, Potsdam, Germany (2010–2012), member Steering Committee of the Global Soil Week, IASS, Potsdam, Germany (2012–), member Advisory Board of FACCE-JPI of the European Council (2013–2016), Chair Advisory Committee to UNU-FLORES, Dresden, Germany (2014–), and member of the Scientific Committee to implement “4 Pour Mille” program proposed at COP21 in Paris. Prof. Lal was a lead author of IPCC (1998–2000), and was awarded Nobel Peace Prize Certificate by IPCC in 2007, and Global Dry Land Champion of the United Nations Convention to Combat Desertification, Bonn, Germany in 2013. He has mentored 105 graduate students and 54 postdoctoral researchers, and hosted 152 visiting scholars from around the world. He has authored/co-authored more than 2023 research publications including 798 refereed journal articles and 436 book chapters, has written 19 and edited/co-edited 63 books.



Taylor & Francis

Taylor & Francis Group

<http://taylorandfrancis.com>

Encyclopedia of Soil Science

Third Edition

Volume I

Accounting through Erosivity
Pages 1–847

Accounting –
Alpine

Aluminum –
Bioenergy

Biofuels –
Carbon

Carbonates –
Classification

Clay –
Crops

Cultivation –
Desertification

Desertization –
Enzymes

Erosion –
Erosivity



Taylor & Francis

Taylor & Francis Group

<http://taylorandfrancis.com>

Accounting: Emergy Analysis

Jay F. Martin

Department of Food, Agricultural, and Biological Engineering, Ohio State University, Columbus, Ohio, U.S.A.

D.R. Tilley

Department of Environmental Science and Technology, University of Maryland, College Park, Maryland, U.S.A.

Abstract

This entry explains the use of emergy analysis to evaluate systems and compare resource use, environmental loading, and sustainability. This method allows the free contributions of the environment to be compared directly with the purchased inputs from the economy. Results summarized from agriculture and forestry systems indicate that extensive amounts of indirect energy were dissipated to yield agricultural and forest products and that timber typically requires less indirect energy than crops. Timber production had higher average yields than agricultural crops, demonstrating that forests yield a greater amount of solar emergy per unit of economic input than crop systems. The average environmental loading of agricultural crops was greater than that of forests, revealing that crop production relies much more heavily on non-renewable and purchased inputs than forestry. Finally, the sustainability of forestry systems was much greater than crops, indicating that forests are more sustainable than cropland.

INTRODUCTION

The work of nature sustains the global life-support system for humans, contributes directly to human welfare, and is ultimately responsible for a large portion of the human economy. Quantifying the sustainability of human activities with an accounting system that values ecosystem work and the human economy on a comparable basis could improve environmental decision making. For example, modern agriculture in developed countries must supply more food, fiber, and fuels to an expanding world population under tightening environmental and energy constraints. Agriculture needs to reverse the trend of the 20th century when use of non-renewable resources increased faster than yields. Because systems like agriculture depend heavily on the unpaid work of nature, environmental accounting methods must be able to compare all the driving factors on a common basis. While the values of economic contributions are routinely quantified in economic analyses, such approaches often underestimate or fail to measure environmental contributions to production. If environmental inputs are not properly accounted for relative to economic inputs, optimum use of resources may not be achieved, and decisions will be based on incomplete information. In the 1970s, Odum published a pioneering, comprehensive look at how energy accounting could be applied to evaluate the

multiple inputs used to produce the world's food. Odum's novel approach went beyond an analysis of the energy inputs traditionally included—direct solar irradiance and fuel consumption—the contributions from the environment (e.g., dry air, water, and soil organic matter) and the economy (e.g., labor and services) as embodied solar energy (emergy in later nomenclature). Following his 1967 world food article, Odum focused on perfecting emergy accounting by formalizing its theoretical basis and conducting evaluations of systems around the world.^[1,2]

Emergy analysis is a form of energy analysis that quantifies values of natural and economic resources on a common basis to derive the contribution of nature to the human economy.^[1] Solar emergy is used to determine the contribution of environmental and human work to a system by transforming all inputs to the ultimate amount of solar energy required to produce each. Because of its ability to measure environmental and economic resources with a common metric, emergy analysis can quantify environmental sustainability. The solar emergy of products and services is calculated by multiplying energy by energy transformation ratios (transformities), mass by mass transformation ratios (specific emergy), and dollar by dollar transformation ratios (solar emergy $\$^{-1}$). In this entry, we introduce the concept of emergy accounting and review how it has been applied to assess environmental sustainability of agricultural and forested ecosystems.

DEFINITIONS OF TERMS AND INDICES

The following list defines key terms and indices used in energy analysis (Fig. 1). Emergy indices are calculated from aggregated data and used to quantify economic and environmental contributions to production, net process yield, environmental loading, and sustainability.^[1]

- **Emergy:** the available energy (exergy) of one kind used in transformations directly and indirectly to make a product or service. Emergy is measured in solar emjoules (sej). Sunlight, fuel, electricity, human service, and all resource flows are quantified as sej.
- **Transformity:** the amount of emergy required to produce a unit of available energy (exergy). For example, the solar transformity of wood is 20,000 sej J⁻¹ because 20,000 solar emjoules of environmental inputs were required to generate a joule of wood. The solar transformity of sunlight absorbed by the earth is defined as 1 sej J⁻¹. Transformities have been calculated for a wide variety of resources, commodities, and renewable energies.^[1,3-6]
- **Specific emergy:** emergy per mass, usually expressed as sej g⁻¹.
- **Emergy per unit money:** emergy supporting the generation of one unit of economic product (expressed as currency). Average emergy per money is calculated by dividing the total emergy use of an economy by its gross economic product (sej \$⁻¹).
- **Empower:** flow of emergy per time, usually expressed as solar empower (sej yr⁻¹).
- **Emergy yield ratio (EYR = Y/F):** emergy yield produced ($Y = R + N + F$) per unit of emergy contributed from the economy (F) (sej sej⁻¹).
- **Environmental loading ratio (ELR = (N + F)/R):** emergy contributed from non-renewable and economic sources per unit of emergy contributed from renewable

resources (sej sej⁻¹). It is an indicator of the pressure of agricultural systems on the environment and may be considered as a measure of ecosystem stress.

- **Emergy sustainability index (ESI = EYR/ELR):** the ratio of yield to environmental load, which measures system production relative to environmental pressure.
- **Emergy investment ratio (EIR = F/(N + R)):** emergy purchased and contributed from the economy (F) per unit of emergy contributed free from the environment whether renewable or non-renewable (R + N).

ENVIRONMENTAL DECISION MAKING WITH EMERGY

Emergy analysis begins with the construction of a systems diagram^[4,6] to define the boundary, identify resource inputs, and conceptualize relationships among components, inputs, and outputs. The emergy analysis table^[1] is constructed directly from the systems diagram using inflows and outflows crossing the system boundary as row headings. At first the annual amount of flow of each input and output is quantified in physical units (i.e., joules, grams, and dollars), then the annual solar emergy of each flow is calculated by multiplying the respective solar transformity, specific solar emergy, or solar emergy-to-money ratio. The flows are aggregated into categories of renewable resources (i.e., replenished within a year or less), non-renewable resources (i.e., replenished after multiple years), non-indigenous purchased resources (i.e., paid for and brought from outside system), and exports (or yield).

AGRICULTURAL PRODUCTION

As part of an emergy analysis of Italy, Ulgiati, Odum, and Bastianoni^[7] calculated emergy indices for several Italian crops and assessed the role of agriculture in Italy. The ELR for crop and livestock production in Italy was 2.5 and 3.3, respectively. Comparing these values with the Italian economy's mean, ELR of 9.5 indicated that Italian agriculture relied more heavily on natural environmental energy inputs than the economy as a whole. Table 1 compares the transformity and emergy indices for selected Italian crops. Almonds had the greatest transformity, indicating that more emergy was needed to produce one joule of almonds than any other crop. The relatively low ELR for almonds, wheat, rice, and forage indicated that these crops rely heavily on renewable emergy and lightly on purchased and non-renewable emergy. In contrast, sunflowers, oranges, and lemons required greater inputs of purchased and non-renewable emergy and had greater ELRs. Greater EYRs for crops such as almonds, wheat, rice, and forage indicated a greater yield for these crops

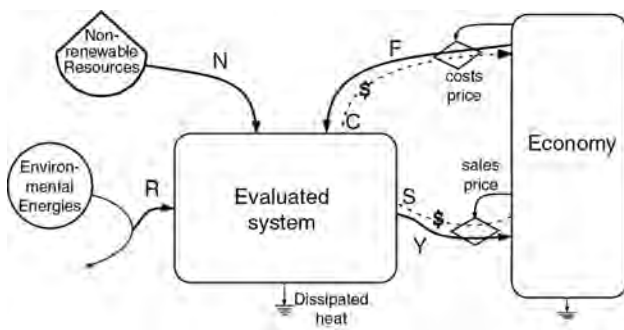


Fig. 1 Energy system diagram of the aggregated inputs (R, renewable; N, non-renewable; and F, purchased) to an evaluated system that produces a yield (Y) to the economy and generates sales revenue (S) that pays for purchased costs (C).

Table 1 Emergy indices for selected crops in Italian agriculture.

Crop	Solar transformity (1 E4 sej J ⁻¹)	ELR	EYR	ESI
Rice	7.78	2.86	1.38	0.48
Forage	8.00	1.45	1.76	1.21
Sugar beet	8.49	7.33	1.15	0.16
Corn	8.52	5.63	1.19	0.21
Wheat	15.90	3.38	1.32	0.39
Fruits	28.74	9.37	1.11	0.12
Vineyard	34.11	5.33	1.20	0.23
Oranges and lemons	38.17	11.82	1.09	0.09
Olive	53.03	4.40	1.24	0.28
Sunflower	79.12	27.78	1.04	0.04
Almonds	84.28	3.10	1.35	0.44

Source: From Ulgiati, Odum, et al.^[7]

relative to required economic investment. The ESI showed that almonds, wheat, rice, and forage crops were more sustainable than sunflower, orange, and lemon production.

Another study estimated that corn solar transformities declined from 8.41 to 5.11 E4 sej J⁻¹ in the United States from 1945 to 1994. This decline was due to more efficient use of inputs and less use of renewable resources. The ESI declined from 1.12 in 1945 to 0.34 in 1994, indicating that efficiency gains were overshadowed by increased non-renewable inputs. To increase sustainability of U.S. corn production, the authors suggested increasing production per emergy input, more reliance on local renewable resources, and less use of purchased inputs such as fertilizers and pesticides. Emergy was also used to evaluate cropping options in Southwestern Australia^[8] and to compare the sustainability of Swedish agriculture in 1927 and 1999.^[9]

FOREST PRODUCTION

Table 2 summarizes six emergy evaluations of timber yielding forests, including tropical, temperate, natural, and plantation systems operated since the 1970s under various harvest practices from clear-cutting to select-harvest. Timber production was the highest in the rain forest of Papua New Guinea (PNG) and the lowest in the ecologically managed Southern Appalachian forests of North Carolina. Environmental (i.e., renewable) inputs ranged from 605×10^{12} sej ha⁻¹ yr⁻¹ for the Jari, Brazil, forests to $26,074 \times 10^{12}$ sej ha⁻¹ yr⁻¹ for the primary growth PNG rain forest. One goal of timber yielding systems is to have a high return on investment, which is measured by the EYR. The sago palm system had the highest EYR, followed by the

Siris plantation of Puerto Rico, the tropical forest of Brazil, and the pine plantation of New Zealand. With the exception of sago palm, the EYR for timber production (1.5–2.8) was less than known EYRs of oil, coal, natural gas, and geothermal systems.^[1]

Another useful index for gauging the economic success of an operation that relies on natural resource inputs is the EIR. A process with a lower EIR than the larger scale economy that encompasses the process indicates a profitable investment, whereas an EIR higher than the mean economy EIR suggests that a greater proportion of resources must be diverted from the general economy to make the operation successful. For the timber yielding forests shown in Table 2, the sago palm system had the lowest EIR (0.15), while the PNG rain forest, which was a primary forest, had the highest EIR (1.63). Four of the evaluated forests had EIRs <1.0, showing that natural processes contributed more than the economy did.

The rain forest of PNG had the highest ELR due to the harvest of primary forests, while the majority of forests had an ELR below 1.0, which indicates that the relative load on the environment was small. The index of sustainability (ESI) indicated that the sago palm forest was the most sustainable. The Puerto Rico, North Carolina, Brazil, and New Zealand systems had ESIs of 5.0, 4.59, 3.68, and 3.07, respectively, which were all much greater than the least sustainable forest (PNG).

Ecological sources of renewable fuel, such as forests, most often have a lower EYR than non-renewable fuels, such as coal. This is partly because non-renewable resources are concentrated in space with a higher energy density and more accumulated solar emergy. Conversely, renewable fuel sources accumulate emergy slowly and are more dilute on the landscape. Odum found that the EYR of ecological sources of fuel, such as corn, sugarcane, pine plantations, and old-growth forests, was directly related to the replacement time of the biomass (p. 144).^[1] Corn with a replacement time of 1 year had an EYR of 1.1, while a 90-year-old Swedish spruce forest had an EYR of 4.1. The gathered evidence indicated that the EYR was linearly related to replacement time; each 50-year increase in replacement time increased the EYR by 1.75 units. Odum^[1] concluded that as the frequency of harvest diminished, the work of nature increased in proportion to the work of humans, resulting in a higher EYR.

CONCLUSION

Emergy analysis estimates the amount of solar energy required to operate a system for production, enumerating all flows of energy, material, and information in units of sej. The aggregation of inputs according to renewability and source location is used to create various indices. Emergy evaluations of agriculture and forest production systems

Table 2 Emergy evaluations of forests managed for timber production.

Input item	Tropical forest (Jari, Brazil) ^a	Pine plantation (New Zealand) ^c	Sago palm (Papua New Guinea) ^b	Siris plantation (Puerto Rico) ^c	Rain forest (Papua New Guinea) ^b	Southern Appalachians (North Carolina, U.S.A.) ^d
Renewable	$1 \times 10^{12} \text{ sej ha}^{-1} \text{ yr}^{-1}$					
Sunlight	42					50
Precipitation		1,092	2,400		1,827	1,603
Transpiration	493			770		440
Land cycle	112			1,620		425
Timber					26,074	217
Village labor			16,900			
Phosphorus, weathering				18		
Renewables (R)	605	1,092	19,300	2,408	26,074	217
Purchased or imported from economy	$1 \times 10^{12} \text{ sej ha}^{-1} \text{ yr}^{-1}$					
Fuel + lubricants	28			97	2,802	151
Goods + services	382	318	2,800	430	3,580	153
Phosphorus, fertilizer		503				
Labor				57	32,500	13
Machinery				11	3,698	
Seedling costs				195		
Capital costs				565		
Purchased (F)	410	821	2,800	1,355	42,580	317
Total solar emergy (Y)	1,015	1,913	22,100	3,763	68,654	534
Harvest ($1 \times 10^6 \text{ J ha}^{-1} \text{ yr}^{-1}$)	23,000	167,000	168,000	167,000	2,960,000	4,100
Costs: silviculture + harvest ($\$ \text{ ha}^{-1} \text{ yr}^{-1}$)	29	57		1,314	4,319	110
Solar transformity (sej J^{-1})	44,130	16,380	131,600	22,500	23,200	53,000
EYR (sej sej^{-1})	2.5	2.3	8.0	2.8	1.5	1.7
ELR (sej sej^{-1})	0.68	0.75	0.15	0.56	23.3	0.37
EIR (sej sej^{-1})	0.68	0.75	0.15	0.38	1.63	1.46
ESI	3.68	3.07	53.3	5.00	0.064	4.59

Source: Data from ^aChristianson,^[10] ^bDoherty & Brown,^[11] ^cScatena, Doherty, et al.,^[12] ^dTilley & Swank,^[6] and ^eOdum & Odum.^[13]

demonstrated that the multiple driving sources, with their various units of measure, could be aggregated to the amount of solar energy responsible for their existence. This allowed the free contributions of the environment to be compared directly with the purchased inputs from the economy. Past environmental work required to provide a modern flow of exergy embodied in agricultural products was reported as the solar transformity. Agricultural products had solar transformities ranging from 51,100sejJ⁻¹ for corn to 843,000sejJ⁻¹ for almonds, while timber was between 22,400 and 131,600sejJ⁻¹, indicating that extensive amounts of indirect energy were dissipated to yield agricultural and forest products and that timber typically requires less indirect energy than crops. Timber production had a higher average EYR (2.16 ± 0.55) than crop (1.26 ± 0.20) production, demonstrating that forests yield a greater amount of solar emergy per unit of

economic input than crop systems. The average ELR of crop fields (7.0) was greater than that of forests (0.50), revealing that crop production relies much more heavily on non-renewable and purchased inputs than forestry. The median ESI of forests (4.1) was much higher than crops (0.23), indicating that forests are over 10 times more sustainable than cropland.

REFERENCES

1. Odum, H.T. *Environmental Accounting: Emergy and Environmental Decision Making*; Wiley: New York, 1996.
2. Brown, M.T.; Ulgiati, S. Energy quality, emergy, and transformity: H.T. Odum's contributions to quantifying and understanding systems. *Ecol. Model.* **2004**, *178*, 201–213.

3. Ulgiati, S. A comprehensive energy and economic assessment of biofuels: When “green” is not enough. *Crit. Rev. Plant Sci.* **2001**, *20*, 71–106.
4. Martin, J.F. Emergy valuation of diversions of river water to marshes in the Mississippi River Delta. *Ecol. Eng.* **2002**, *18*, 165–186.
5. Kangas, P.C. *Handbook of Emergy Evaluation: A Compendium of Data for Emergy Computation Issued in a Series of Folios. Folio 5. Emergy of Landforms*; Center for Environmental Policy, University of Florida: Gainesville. Available at <http://www.ees.ufl.edu/cep/>.
6. Tilley, D.R.; Swank, W.T. Emergy-based environmental systems assessments of a multi-purpose temperate mixed-forest watershed of the Southern Appalachian Mountains, USA. *J. Environ. Manage.* **2003**, *69*, 213–227.
7. Ulgiati, S.; Odum, H.T.; Bastianoni, S. Emergy use, environmental loading and sustainability. An emergy analysis of Italy. *Ecol. Model.* **1994**, *73*, 215–268.
8. Lefroy, E.; Rydberg, T. Emergy evaluation of three cropping systems in Southwestern Australia. *Ecol. Model.* **2003**, *161*, 195–211.
9. Rydberg, T.; Jansen, J. Comparison of horse and tractor traction using emergy analysis. *Ecol. Eng.* **2002**, *19*, 13–28.
10. Christianson, R.A. Emergy perspectives on plantation and pulp mill at Jari. In *Emergy Systems Overview of the Amazon Basin*; Odum, H.T., Brown, M.T., Christianson, R.A., Eds.; Center for Wetlands, University of Florida: Gainesville, 1986.
11. Doherty, S.J.; Brown, M.T. *Emergy Synthesis Perspectives, Sustainable Development, and Public Policy Options for Papua New Guinea*, CFW #93-06; Center for Water and Wetland Resources, University of Florida: Gainesville, 1993.
12. Scatena, F.N.; Doherty, S.J.; Odum, H.T.; Karecha, P. *An Emergy Evaluation of Puerto Rico and the Luquillo Experimental Forest*, General Technical Report IITF-GTR-9; United States Department of Agriculture, Forest Service, International Institute of Tropical Forestry: San Juan, 2002.
13. Odum, H.T.; Odum, E.C. *Emergy System of New Zealand and the Use of Embodied Emergy for Evaluating Benefits of International Trade*; Joint Centre for Environmental Sciences, University of Canterbury: Christchurch, 1980.

Acid Mine Drainage

Jerry M. Bigham

School of Environment and Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Charles A. Cravotta III

Pennsylvania Water Science Center, U.S. Geological Survey (USGS), New Cumberland, Pennsylvania, U.S.A.

Abstract

Acid mine drainage (AMD) consists of metal-laden solutions produced by the oxidative dissolution of iron sulfide minerals exposed to air, moisture, and acidophilic microbes during the mining of coal and metal deposits. The pH of AMD is usually in the range of 2–6, but mine-impacted waters at circumneutral pH (5–8) are also common. Mine drainage usually contains elevated concentrations of sulfate, iron, aluminum, and other potentially toxic metals leached from rock that hydrolyze and coprecipitate to form rust-colored encrustations or sediments. When AMD is discharged into surface waters or groundwaters, degradation of water quality, injury to aquatic life, and corrosion or encrustation of engineered structures can occur for substantial distances. Prevention and remediation strategies should consider the biogeochemical complexity of the system, the longevity of AMD pollution, the predictive power of geochemical modeling, and the full range of available field technologies for problem mitigation.

INTRODUCTION

Acid mine drainage (AMD) refers to metal-rich sulfuric acid (H_2SO_4) solutions released from mine tunnels, shafts, open pits, and waste rock piles. Similar solutions can be produced by the excavation of roadways through certain rocks, dredging of harbor sediments, or drainage of coastal wetlands.

Spoils and soils exposed to AMD tend to be sparsely vegetated and susceptible to erosion. When AMD enters waterways, increases in pH can result in the precipitation of metal hydroxide sediments, which smother aquatic habitats and clog the gills of fish and other aquatic life (Fig. 1). The corrosion of concrete and metal structures in contact with AMD, such as bridge pilings and pipes, is also greatly accelerated. Once initiated, AMD may persist for decades, making it a challenging problem to solve.

AMD results from the rapid oxidation of iron (Fe)-bearing sulfide minerals such as pyrite (cubic FeS_2), marcasite (orthorhombic FeS_2), pyrrhotite (Fe_{1-x}S , $x = 0$ to 0.2), chalcopyrite (CuFeS_2), and arsenopyrite (FeAsS). These minerals are commonly found in coal and ore deposits and are stable until exposed to oxygen and moisture. Sulfide oxidation produces H_2SO_4 , which can dissolve associated metals. This release of acid and metals can occur locally as a form of natural mineral weathering (Natural Acid Rock Drainage) but is exacerbated by mining because of the sudden, large-scale exposure of sulfides in unweathered rock to atmospheric conditions.

MINE DRAINAGE CHEMISTRY

AMD, in which mineral acidity exceeds alkalinity, typically contains elevated concentrations of sulfate (SO_4), Fe, manganese (Mn), aluminum (Al), and other solutes and may or may not have a low $\text{pH}^{[1]}$ (Table 1). AMD typically has pH from 2 to 6; however, sites, such as Iron Mountain, California, have produced extreme pH values as low as -3.6 .^[2,3] Neutral to alkaline mine drainage (NAMD), with pH from 5 to 8 and a surplus of alkalinity, is common in areas where the surrounding geologic units contain carbonate rocks to buffer acidity^[1,3,4] (Table 1). Because water discharged from mines generally is not at equilibrium with atmospheric conditions, the pH tends to be unstable—the hydrolysis of dissolved Fe, Al, and Mn can decrease the pH, while the outgassing of dissolved carbon dioxide (CO_2) can increase the pH. The “hot peroxide acidity” indicates the ultimate potential for the pH to be acidic or circumneutral and the amount of base required to neutralize the fully oxidized solution.^[4] Given sufficient time for equilibration with the atmosphere and oxidation and hydrolysis of dissolved Fe and Mn, AMD ultimately will have $\text{pH} \leq 4.5$, whereas NAMD ultimately will have $\text{pH} > 4.5$ (Table 1).

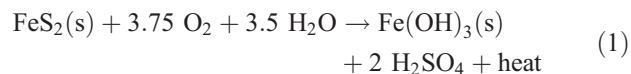
AMD forms by complex biogeochemical processes involving oxidation–reduction, hydrolysis, precipitation, and dissolution reactions as well as microbial catalysis.^[2] The entire sequence is commonly represented by Reaction 1, which describes the overall oxidation of pyrite by oxygen in the presence of water to form solid iron hydroxide



Fig. 1 (A) Discharge from the Old Forge Borehole (41.3591–75.75137), which drains a flooded underground coal mine in northeastern Pennsylvania, imparts a characteristic orange–brown turbidity to the Lackawanna River due to voluminous precipitates of iron hydroxide minerals. (B) Confluence of the Susquehanna River (at left) and Lackawanna River (at right) near Duryea, Pennsylvania (41.3412–75.7930), nearly 3 miles downstream from the Old Forge Borehole.

Source: Google Earth image dated 9/11/2012.

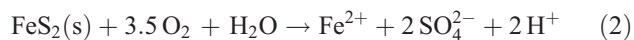
$[\text{Fe}(\text{OH})_3]$ and H_2SO_4 with the spontaneous release of energy as heat.



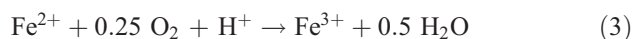
Chemolithotrophic microbes, described in more detail below, utilize the energy from the oxidation of Fe and sulfur (S).

Pyrite and related iron sulfide minerals contain both Fe and S in reduced oxidation states. When exposed to oxygen and water, the S moiety is oxidized first, releasing H_2SO_4 and Fe^{2+} to solution (Reaction 2). The rate of oxidation is dependent on environmental factors such as

temperature, pH, Eh, and relative humidity as well as mineral surface area and microbial catalysis.



Reaction 2 can be strictly abiotic or mediated by S-oxidizing bacteria.^[5] The Fe^{2+} is subsequently oxidized by oxygen, as described by Reaction 3.

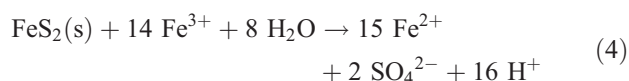


If acidity generated by Reaction 2 exceeds the buffering capacity of the system, the pH eventually decreases. Below

Table 1 Summary of mine drainage chemistry for 140 abandoned coal mines in Pennsylvania, 1999.^[1]

	AMD (hot acidity ≥ 10 mg/L; N = 88)	NAMD (hot acidity < 10 mg/L; N = 52)
	Median (range)	Median (range)
pH, fresh (units)	4.1 (2.7 to 6.2)	6.2 (5.2 to 7.3)
pH, aged (units)	3.2 (2.2 to 4.6)	7.8 (4.7 to 8.6)
Alkalinity (mg/L as CaCO ₃)	0 (0 to 190)	125 (8 to 510)
Hot acidity (mg/L as CaCO ₃)	124 (10 to 1590)	-52 (-417 to 7.5)
Cl (mg/L)	4.9 (0.1 to 460)	20.5 (0.7 to 270)
SO ₄ (mg/L)	560 (34 to 2000)	475 (96 to 1400)
Ca (mg/L)	85 (3.3 to 410)	100 (11 to 320)
Mg (mg/L)	41 (3.6 to 210)	37 (8.5 to 140)
Na (mg/L)	6.9 (0.69 to 460)	63 (0.89 to 500)
K (mg/L)	2.6 (0.5 to 12)	3.2 (0.78 to 6.3)
Silica (mg/L)	20 (6.4 to 67)	15 (5.8 to 22)
Fe (mg/L)	41 (0.046 to 512)	23 (0.049 to 101)
Al (mg/L)	7.7 (0.014 to 108)	0.022 (0.007 to 1.5)
Mn (mg/L)	3.4 (0.39 to 74)	1.5 (0.02 to 9.2)
As (mg/L)	0.0010 (< 0.00003 to 0.064)	0.0052 (< 0.00003 to 0.015)
Cu (mg/L)	0.0039 (0.00045 to 0.19)	0.0008 (0.0004 to 0.017)
Ni (mg/L)	0.145 (0.02 to 3.2)	0.0225 (0.0026 to 0.12)
Pb (mg/L)	0.00067 (< 0.00001 to 0.011)	< 0.00001 (< 0.00001 to 0.00068)
Zn (mg/L)	0.27 (0.011 to 10)	0.0088 (0.0006 to 0.25)

pH 3, Fe³⁺ solubility increases and a second mechanism of pyrite oxidation becomes important (Reaction 4).

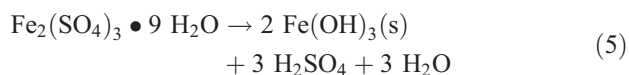


In this case, pyrite is oxidized by Fe³⁺ resulting in the generation of even greater acidity than when oxygen is the primary oxidant. Pyrite decomposition is thus controlled by the rate at which Fe²⁺ is converted to Fe³⁺ at low pH.^[6] At pH < 5 , Fe²⁺ oxidation by Reaction 3 is very slow unless catalyzed by populations of acidophilic, Fe-oxidizing bacteria^[6,7] that oxidize Fe²⁺ as a means of generating energy. In doing so, they supply soluble Fe³⁺ at a rate equal to or slightly greater than the rate of pyrite oxidation by Fe³⁺. Pyrite oxidation then regenerates Fe²⁺ (Reaction 4) creating a cyclic situation that leads to vigorous acidification of mine drainage water.

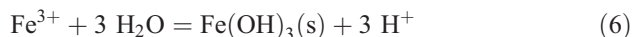
MINE DRAINAGE MINERALOGY

Sulfide mineral oxidation typically takes place in humid air above the groundwater table or at the land surface, where reactants are present and variably soluble products including secondary sulfate minerals may accumulate. Commonly observed secondary sulfates associated with weathered pyrite include melanterite (FeSO₄ • 7H₂O), rozenite (FeSO₄ • 4H₂O), szomolnokite (FeSO₄ • H₂O),

copiapite (Fe^{II}Fe^{III}₄(SO₄)₆(OH)₂ • 20H₂O), and coquimbite (Fe₂(SO₄)₃ • 9H₂O).^[3,8,9] Such secondary SO₄ minerals can also form by the evaporation of AMD during dry periods. Dissolution of these relatively soluble “acid sulfate” minerals by infiltrating water or runoff can result in the rapid release of “stored acidity.”^[3,9] For example, H₂SO₄ is a product of the dissolution of coquimbite (Reaction 5).



The Fe³⁺ released by dissolution of such SO₄²⁻ minerals may interact with pyrite (Reaction 4) or hydrolyze and precipitate as a variety of insoluble Fe minerals, generally represented as [Fe(OH)₃] (Reaction 6).



These rust-colored (yellow-to-red-to-brown) solids can impart turbidity to the water or coat streambeds and aquatic plants and, thus, are often the most obvious indicators of mine drainage contamination. The actual mineralogy of the precipitates is determined by solution parameters such as pH, SO₄, and metal concentration and can vary both spatially and temporally. Some of the most common mine drainage minerals are goethite (α-FeOOH), ferrihydrite (Fe₅HO₈ • 4H₂O), schwertmannite [Fe₈O₈(OH)₆SO₄], and jarosite [(H₃O, K, Na)Fe₃(OH)₆(SO₄)₂].^[10] Goethite is a crystalline oxyhydroxide that is relatively stable over

a wide pH range and may represent a final transformation product of other mine drainage minerals.^[10,11] Ferrihydrite is a poorly crystalline ferric oxide that forms in higher pH (> 6.5) environments. Schwertmannite is commonly found in drainage waters with pH ranging from 2.8 to 4.5 and with moderate to high SO₄ contents. Schwertmannite may be the dominant phase controlling Fe and trace metal concentrations in most AMD.^[12] Jarosite group minerals form in more extreme environments, often with pH < 3, very high SO₄ concentrations, and in the presence of appropriate cations such as sodium (Na) and potassium (K).^[10]

MINE DRAINAGE MICROBIOLOGY

The most studied bacterial species in mine drainage systems belong to the genus *Acidithiobacillus* (formerly *Thiobacillus*).^[13] Species such as *Acidithiobacillus thiooxidans* and *Acidithiobacillus ferrooxidans* are important in S and Fe oxidation in acid drainage; however, many other microorganisms, such as *Leptospirillum* species, may also be involved.^[3] Bacteria have been found in close association with pyrite grains and may play a direct role in mineral oxidation,^[5] but they most likely function indirectly through oxidation of dissolved Fe²⁺ as described previously. In low pH systems (< 3), *A. ferrooxidans* can increase the rate of Fe oxidation as much as five orders of magnitude relative to strictly abiotic rates.^[6,7]

Fe-oxidizing bacteria are chemolithotrophic, meaning they oxidize inorganic compounds, such as Fe²⁺, to generate energy and use CO₂ as a source of carbon. Fe oxidation, however, is a very low energy yielding process compared to heterotrophic processes. It has been estimated that the oxidation of 90.1 moles of Fe²⁺ is required to assimilate 1 mole of C into biomass.^[14] Thus, large amounts of Fe²⁺ must be oxidized to achieve even modest growth.

In addition to mediating Fe oxidation, bacteria may play an additional role in mineral formation. Bacterial cells in mine drainage systems are often partially encrusted with mineral precipitates.^[15] Bacterial cell walls provide reactive sites for the sorption of metal cations, which can accumulate and subsequently develop into precipitates using the bacterial surface (living or dead) as a template.^[16]

ENVIRONMENTAL IMPACTS OF MINE DRAINAGE

AMD from mine openings, such as tunnels or shafts, or from mine spoil can impact aquatic environments substantial distances downstream. Chemical precipitates can smother streambed habitats, obstruct water flow, dramatically increase turbidity, and ruin stream aesthetics. Dissolved metals and acidity can also be toxic to plants and animals.

Besides Fe, dissolved Al is common in AMD. The primary source of Al is the acid dissolution of aluminosilicates found in soil, spoils, tailings deposits, and gangue material.^[10] At moderate concentrations (<1 mg/L), Al can be toxic

to plants, and colloidal Al precipitates can clog or irritate the gills of fish and other aquatic organisms causing suffocation. Al occurs as a dissolved species at low pH but rapidly hydrolyzes at about pH 5 to form “amorphous” feldsparite [Al₄(SO₄)(OH)₁₀ • 4H₂O] or gibbsite [Al(OH)₃].^[3,10] Al precipitates are white in color but are readily masked by associated Fe compounds.

Elevated concentrations of trace elements such as arsenic (As), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn) may be released during the oxidation of sulfide minerals and be present in AMD and NAMD solutions (Table 1). These elements can play a role in mineralization processes by forming coprecipitates with iron sulfate and hydroxide minerals^[9,17] but occur primarily as sorbed species with the hydroxides.^[8,12] Mine drainage precipitates can retain both anions and cations, depending on pH. While coprecipitation and sorption function to immobilize trace elements by removing dissolved ions from solution, this effect may not be permanent. Dissolution of precipitates and shifts in pH can result in the release of sorbed species, providing a latent source of pollution.^[18]

DEALING WITH MINE DRAINAGE

Successful control of mine drainage usually involves varying degrees of prevention and treatment.

Prevention

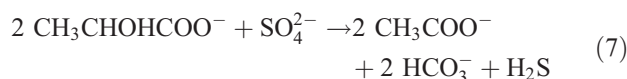
Prevention techniques include sealing mine shafts to prevent discharge, burying or submerging spoil piles to reduce contact by oxygen, reining and recycling of waste materials to eliminate the sulfide minerals, and the addition of bactericides to limit catalysis by Fe-oxidizing bacteria. These techniques often have limited success. Sealing of mines is extremely difficult due to fractures and the permeability of surrounding rocks. Covering spoil with soil material can decrease the degree of sulfide oxidation by limiting exposure to oxygen, but establishment of a vegetative cover is necessary to prevent erosion from reexposing the spoil. Submerging mine waste in water may be effective where oxidation has not initiated; however, previously formed oxidation products can be dissolved with acidity, metals, and SO₄ released to solution (Reaction 5). Reining and recycling may not be economically viable. Inhibition of Fe-oxidizing bacteria with bactericides can decrease sulfide mineral oxidation and reduce metal mobility; however, reapplication is necessary and adequate distribution to all affected areas is difficult. In addition, target bacteria may develop resistance and beneficial bacteria may be harmed.^[5]

Treatment

The traditional approach to treatment of AMD involves neutralization of acidity by the addition of alkaline chemical agents such as caustic soda (NaOH) or hydrated

lime [Ca(OH)₂]. This method is effective in rapidly increasing pH and precipitating dissolved metals; however, it requires continuous oversight and produces large amounts of metal-rich sludge that require disposal. Alternative remediation strategies focus on low-maintenance treatment methods. For example, diverting AMD through limestone drains coupled with settling ponds or aerobic wetlands have shown some promise as passive remediation technologies.^[19] In these systems, drainage is channeled through either oxic or anoxic limestone substrates to neutralize active acidity. Dissolved metals are then allowed to hydrolyze and precipitate in ponds or wetland cells. A major difficulty is the coating of limestone surfaces by precipitates of Fe and Al that may eventually obstruct flow. When drainage is naturally net alkaline, engineered aeration systems can be effective in promoting CO₂ outgassing, which increases pH and the rate of abiotic Fe(II) oxidation, with the rapid removal of dissolved Fe by mineral precipitation.^[20]

Compost wetlands are designed to stimulate the development of anaerobic microbial populations, particularly SO₄-reducing bacteria. The bacteria use the compost as an organic substrate and remove SO₄ from solution either by converting it to hydrogen sulfide (H₂S), which is lost to the atmosphere, or by forming insoluble iron sulfides (Reactions 7 and 8).



Bicarbonate is formed as a by-product of SO₄ reduction and functions to buffer acidity. These systems have also shown limited success in the field because the SO₄ removal rates are usually low (<10%), and pH often remains unchanged or decreases within the system.^[11] Nevertheless, bioreactors involving SO₄ reduction can be effective for removing potentially toxic trace elements such as Zn and Cu that form sulfide precipitates.

REFERENCES

- Cravotta, C.A., III. Dissolved metals and associated constituents in abandoned coal-mine discharges, Pennsylvania, USA – 2. Geochemical controls on constituent concentrations. *Appl. Geochem.* **2008**, *23* (2), 203–226.
- Nordstrom, D.K.; Blowes, D.W.; Ptacek, C.J. Hydrogeochemistry and microbiology of mine drainage: An update. *Appl. Geochem.* **2015**, *57*, 3–16.
- Nordstrom, D.K. Mine waters: Acidic to circumneutral. *Elements* **2011**, *7* (6), 393–398.
- Kirby, C.S.; Cravotta, C.A., III. Net alkalinity and net acidity I: Theoretical considerations. *Appl. Geochem.* **2005**, *20* (10), 1920–1940.
- Lu, X.; Wang, H. Microbial oxidation of sulfide tailings and the environmental consequences. *Elements* **2011**, *8* (2), 119–124.
- Singer, P.C.; Stumm, W. Acidic mine drainage: The rate determining step. *Science* **1970**, *167*, 1121–1123.
- Kirby, C.S.; Thomas, H.M.; Southam, G.; Donald, R. Relative contributions of abiotic and biological factors in Fe(II) oxidation in mine drainage. *Appl. Geochem.* **1999**, *14* (4), 511–530.
- Jambor, J.L.; Nordstrom, D.K.; Alpers, C.N. Metal-sulfate salts from sulfide mineral oxidation. In *Sulfate Minerals; Crystallography, Geochemistry and Environmental Significance; Reviews in Mineralogy and Geochemistry*; Alpers, C.N., Jambor, J.L., Nordstrom, D.K., Eds.; The Mineralogical Society of America: Washington, DC, 2000; Vol. 40, 303–350.
- Hammarstrom, J.M.; Seal, R.R., II; Meierb, A.L.; Kornfeld, J.M. Secondary sulfate minerals associated with acid drainage in the eastern US: Recycling of metals and acidity in surficial environments. *Chem. Geol.* **2005**, *215* (1–4), 407–431.
- Bigham, J.M.; Nordstrom, D.K. Iron and aluminum hydroxysulfates from acid mine waters. In *Sulfate Minerals; Crystallography, Geochemistry and Environmental Significance*; Alpers, C.N., Jambor, J.L., Nordstrom, D.K., Eds.; Reviews in Mineralogy and Geochemistry; The Mineralogical Society of America: Washington, DC, 2000; Vol. 40, 351–403.
- Gagliano, W.B.; Brill, M.R.; Bigham, J.M.; Jones, F.S.; Traina, S.J. Chemistry and mineralogy of ochreous sediments in a constructed mine drainage wetland. *Geochim. Cosmochim. Acta.* **2004**, *68*, 2119–2128.
- Webster, J.G.; Swedlund, P.J.; Webster, K.S. Trace metal adsorption onto an acid mine drainage iron(III) oxy hydroxy sulfate. *Environ. Sci. Technol.* **1998**, *32*, 1361–1368.
- Kelly, D.P.; Wood, A.P. Reclassification of some species of *thiobacillus* to the newly designated genera *Acidithiobacillus* gen. nov., *Halothiobacillus* gen. nov. and *Thermithiobacillus* gen. nov. *Int. J. Syst. Evol. Micr.* **2000**, *50* (2), 511–516.
- Ehrlich, H.L. *Geomicrobiology*, 3rd Ed.; Marcel Dekker, Inc.: New York, 1996; 293–295.
- Clarke, W.; Konhouser, K.O.; Thomas, J.; Bottrell, S.H. Ferric hydroxide and ferric hydroxysulfate precipitation by bacteria in an acid mine drainage lagoon. *FEMS Microbiol.* **1997**, *20* (3–4), 351–361.
- Konhauser, K.O. Diversity of bacterial iron mineralization. *Earth-Sci. Rev.* **1998**, *43*, 91–121.
- Carlson, L.; Bigham, J.M.; Schwertmann, U.; Kyek, A.; Wagner, F. Scavenging of As from acid mine drainage by schwertmannite and ferrihydrite: A comparison with synthetic analogues. *Environ. Sci. Technol.* **2002**, *36* (8), 1712–1719.
- Lee, G.; Faure, G.; Bigham, J.M.; Williams, D.J. Metal release from bottom sediments of Ocoee Lake No. 3, a primary catchment area for the Ducktown, Mining District. *J. Environ. Qual.* **2008**, *37* (2), 344–352.
- Cravotta, C.A., III; Trahan, M.K. Limestone drains to increase pH and remove dissolved metals from acidic mine drainage. *Appl. Geochem.* **1999**, *14*, 581–606.
- Cravotta, C.A., III. Monitoring, field experiments, and geochemical modeling of Fe(II) oxidation kinetics in a stream dominated by net-alkaline coal-mine drainage, Pennsylvania, U.S.A. *Appl. Geochem.* **2015**, *62*, 96–107.

Acid Rain: Nitrogen Deposition

George F. Vance

Department of Renewable Resources, University of Wyoming, Laramie, Wyoming, U.S.A.

Abstract

Air pollution has occurred naturally since the formation of the earth's atmosphere; however, the industrial era has resulted in human activities greatly contributing to global atmospheric pollution. One of the more highly publicized and controversial aspects of atmospheric pollution is that of acidic deposition. Acidic materials can be transported long distances, some as much as hundreds of kilometers. Acidic deposition can impact buildings, sculptures, and monuments that are constructed using weatherable materials like limestone, marble, bronze, and galvanized steel. While acid soil conditions are known to influence the growth of plants, agricultural impacts related to acidic deposition are of less concern due to the buffering capacity of these types of ecosystems. When acidic substances are deposited in natural ecosystems, a number of adverse environmental effects are believed to occur, including damage to vegetation, particularly forests, and changes in soil and surface water chemistry.

INTRODUCTION

Air pollution has occurred naturally since the formation of the earth's atmosphere; however, the industrial era has resulted in human activities greatly contributing to global atmospheric pollution.^[1,2] One of the more highly publicized and controversial aspects of atmospheric pollution is that of acidic deposition. Acidic deposition includes rainfall, acidic fogs, mists, snowmelt, gases, and dry particulate matter.^[3] The primary origin of acidic deposition is the emission of sulfur dioxide (SO₂) and nitrogen oxides (NO_x) from fossil fuel combustion; electric power-generating plants contribute approximately two-thirds of the SO₂ emissions and one-third of the NO_x emissions.^[4]

Acidic materials can be transported long distances, some as much as hundreds of kilometers. For example, 30%–40% of the sulfur (S) deposition in the northeastern United States originates in industrial Midwestern U.S. states.^[5] After years of debate, United States and Canada have agreed to develop strategies that reduce acidic compounds originating from their countries.^[5,6] In Europe, the small size of many countries means that emissions in one industrialized area can readily affect forests, lakes, and cities in another country. For example, approximately 17% of the acidic deposition falling on Norway originated in Britain, and 20% in Sweden came from eastern Europe.^[5]

The U.S. Environmental Protection Agency National Acid Precipitation Assessment Program (NAPAP) conducted intensive research during the 1980s and 1990s that resulted in the “Acidic Deposition: State of the Science and Technology” that was mandated by the Acid Precipitation Act of 1980.^[6] NAPAP Reports to Congress have been developed in accordance with the 1990 amendment to the 1970 Clean Air Act and present the expected benefits of the

Acid Deposition Control Program.^[6,7] Mandates include an annual 10 million ton or approximately 40% reduction in point-source SO₂ emissions below 1980 levels, with national emissions limit caps of 8.95 million tons from electric utility and 5.6 million tons from point-source industrial emissions. A reduction in NO_x of about 2 million tons from 1980 levels has also been set as a goal; however, while NO_x has been on the decline since 1980, projections estimate a rise in NO_x emissions after the year 2000. In 1980, the U.S. levels of SO₂ and NO_x emissions were 25.7 and 23.0 million tons, respectively.

Acidic deposition can impact buildings, sculptures, and monuments that are constructed using weatherable materials like limestone, marble, bronze, and galvanized steel.^[7,8] While acid soil conditions are known to influence the growth of plants, agricultural impacts related to acidic deposition are of less concern due to the buffering capacity of these types of ecosystems.^[5] When acidic substances are deposited in natural ecosystems, a number of adverse environmental effects are believed to occur, including damage to vegetation, particularly forests, and changes in soil and surface water chemistry.^[9,10]

SOURCES AND DISTRIBUTION

Typical sources of acidic deposition include coal- and oil-burning electric power plants, automobiles, and large industrial operations (e.g., smelters). Once S and nitrogen (N) gases enter the earth's atmosphere, they react very rapidly with moisture in the air to form sulfuric acid (H₂SO₄) and nitric acid (HNO₃).^[2,3] The pH of natural rainfall in equilibrium with atmospheric carbon dioxide (CO₂) is about 5.6; however, the pH of rainfall is less than 4.5 in

many industrialized areas. The nature of acidic deposition is controlled largely by the geographic distribution of the sources of SO_2 and NO_x (Fig. 1). In the Midwestern and northeastern United States, H_2SO_4 is the main source of acidity in precipitation because of the coal-burning electric utilities.^[2] In the western United States, HNO_3 is of more concern because utilities and industries burn coal with low S contents and populated areas are high sources of NO_x .^[2]

Emissions of SO_2 and NO_x increased in the 20th century due to the accelerated industrialization in the developed countries and antiquated processing practices in some undeveloped countries. However, there is some uncertainty as to the actual means by which acidic deposition affects our environment^[11,12] (<http://nadp.sws.uiuc.edu/isopleths/maps1999/>). Chemical and biological evidence, however, indicates that atmospheric deposition of H_2SO_4 caused some New England lakes to decrease in alkalinity.^[13,14] Many scientists are reluctant to overgeneralize cause and effect relationships in an extremely complex environmental problem. However, the National Acid Deposition Assessment Program has concluded that there were definite

consequences due to acidic deposition that warrant remediation.^[6,7] Since 1995, when the 1990 Clean Air Act Amendment's Title IV reduction in acidic deposition was implemented, SO_2 and NO_x emissions have, respectively, decreased and remained constant during the late 1990s.^[4]

Both H_2SO_4 and HNO_3 are important components of acidic deposition, with volatile organic compounds and inorganic carbon also components of acidic deposition-related emissions. Pure water has a pH of 7.0, natural rainfall about 5.6, and severely acidic deposition less than 4.0. Uncontaminated rainwater has a pH of 5.6 due to CO_2 chemistry and the formation of carbonic acid (H_2CO_3). The pH of most soils ranges from 3.0 to 8.0.^[2] When acids are added to soils or waters, the decrease in pH that occurs depends greatly on the system's buffering capacity, the ability of a system to maintain its present pH by neutralizing added acidity. Clays, organic matter, oxides of aluminum (Al) and iron (Fe), and calcium (Ca) and magnesium (Mg) carbonates (limestones) are the components responsible for pH buffering in most soils. Acidic deposition, therefore, will have a greater impact on sandy, low organic

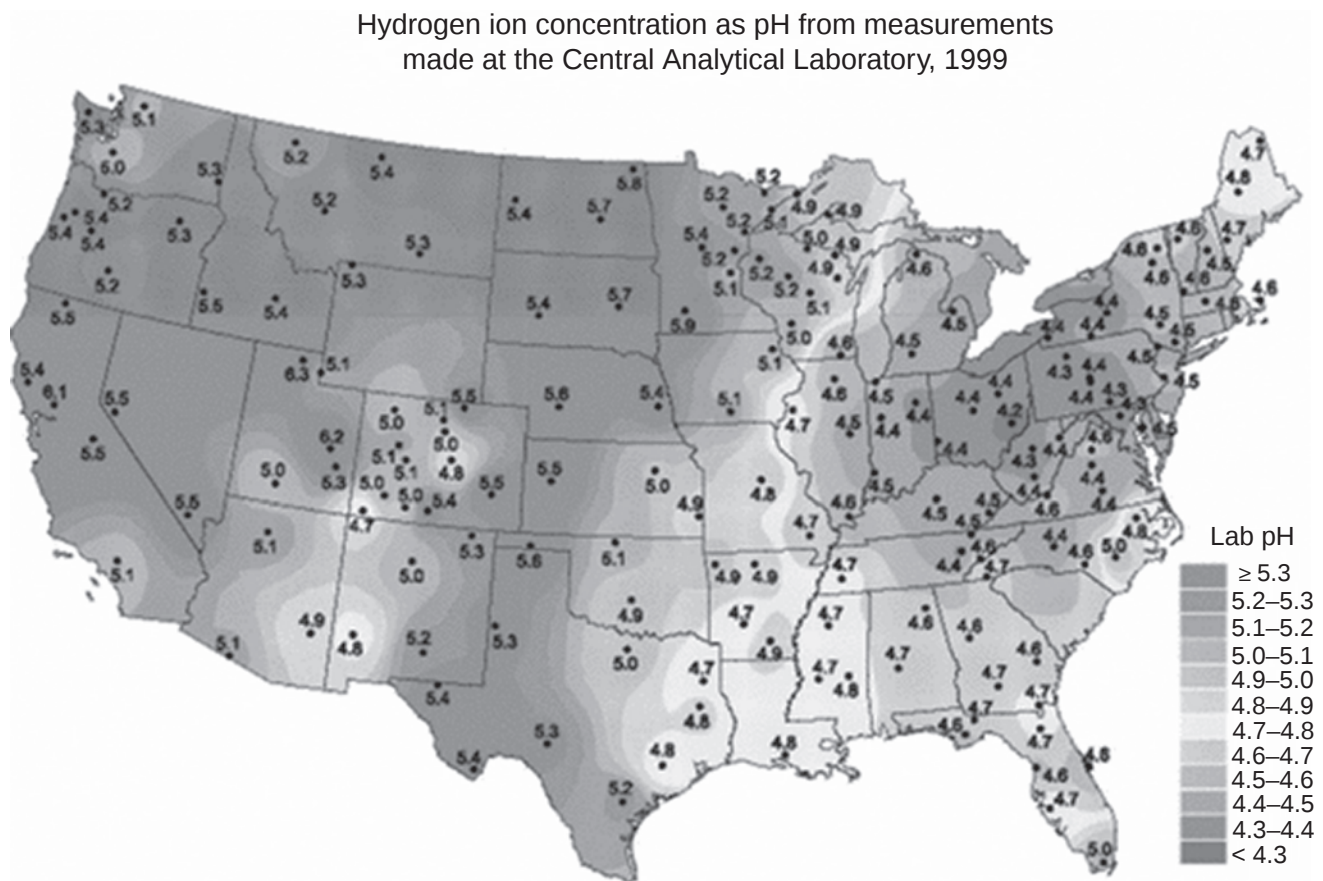
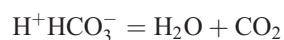


Fig. 1 Acidic deposition across the United States during 1999.

Source: From the National Atmospheric Deposition Program (NRSP-3)/National Trends Network.^[11]

matter soils than on those higher in clay, organic matter, and carbonates. In freshwaters, the primary buffering mechanism is the reaction of dissolved bicarbonate ions (HCO_3^-) with hydrogen ion (H^+) according to the following equation:



HUMAN HEALTH EFFECTS

Few direct human health problems have been attributed to acidic deposition. Long-term exposure to acidic deposition precursor pollutants such as ozone and NO_x , which are respiratory irritants, can cause pulmonary edema.^[5,6] SO_2 is also a known respiratory irritant, but is generally absorbed high in the respiratory tract.

Indirect human health effects due to acidic deposition are more important. Concerns center around contaminated drinking water supplies and consumption of fish that contain potential toxic metal levels. With increasing acidity (e.g., lower pH levels), metals such as mercury, aluminum, cadmium, lead, zinc, and copper become more bioavailable.^[2] The greatest human health impact is due to the consumption of fish that bioaccumulate mercury; freshwater pike and trout have been shown to contain the highest average concentrations of mercury.^[5,15] Therefore, the most susceptible individuals are those who live in an industrial area, have respiratory problems, drink water from a cistern, and consume a significant amount of freshwater fish.

A long-term urban concern is the possible impact of acidic deposition on surface-derived drinking water. Many municipalities make extensive use of lead and copper piping, which raises the question concerning human health effects related to the slow dissolution of some metals (lead, copper, zinc) from older plumbing materials when exposed to more acidic waters. Although metal toxicities due to acidic deposition impacts on drinking waters are rare, reductions in S and N fine particles expected by 2010 based on Clean Air Act Amendments resulted in annual public health benefits valued at \$50 billion with reduced mortality, hospital admissions, and emergency room visits.^[16]

STRUCTURAL IMPACTS

Different types of materials and cultural resources can be impacted by air pollutants. Although the actual corrosion rates for most metals have decreased since the 1930s, data from three U.S. sites indicate that acidic deposition may account for 31–78% of the dissolution of galvanized steel and copper.^[7,8] (<http://www.nnmc.noaa.gov/CENR/NAPAP/>). In urban or industrial settings, increases in atmospheric acidity can dissolve carbonates (e.g., limestone and marble) in buildings and other structures. Deterioration of stone

products by acidic deposition is caused by: 1) erosion and dissolution of materials and surface details; 2) alterations (blackening of stone surfaces); and 3) spalling (cracking and spalling of stone surfaces due to accumulations of alternation crusts.^[8] Painted surfaces can be discolored or etched, and there may also be degradation of organic binders in paints.^[8]

ECOSYSTEM IMPACTS

It is important to examine the nature of acidity in soil, vegetation, and aquatic environments. Damage from acidification is often not directly due to the presence of excessive H^+ , but is caused by changes in other elements. Examples include increased solubilization of metal ions such as Al^{3+} and some trace elements (e.g., Mn^{2+} and Pb^{2+}) that can be toxic to plants and animals, more rapid losses of basic cations (e.g., Ca^{2+} and Mg^{2+}), and the creation of unfavorable soil and aquatic environments for different fauna and flora.

Soils

Soil acidification is a natural process that occurs when precipitation exceeds evapotranspiration.^[2] “Natural” rainfall is acidic (pH of ~5.6) and continuously adds a weak acid (H_2CO_3) to soils. This acidification results in a gradual leaching of basic cations (Ca^{2+} and Mg^{2+}) from the uppermost soil horizons, leaving Al^{3+} as the dominant cation that can react with water to produce H^+ . Most of the acidity in soils with pH between 4.0 and 7.5 is due to the hydrolysis of Al^{3+} .^[17,18] (<http://www.epa.gov/airmarkets/acidrain/effects/index.html>). Other acidifying processes include plant and microbial respiration that produces CO_2 , mineralization and nitrification of organic N, and the oxidation of iron disulfide (FeS_2) in soils disturbed by mining or drainage.^[2] In extremely acidic soils (pH < 4.0), strong acids such as H_2SO_4 are a major component.

The degree of accelerated acidification depends on both the buffering capacity of the soil and the use of the soil. Many of the areas subjected to the greatest amount of acidic deposition are also areas where considerable natural acidification occurs.^[19] Forested soils in the northeastern United States are developed on highly acidic, sandy parent materials that have undergone tremendous changes in land use in the past 200 years. However, clear-cutting and burning by the first European settlers have been almost completely reversed, and many areas are now totally reforested.^[5] Soil organic matter that accumulated over time represents a natural source of acidity and buffering. Similarly, greater leaching or depletion of basic cations by plant uptake in increasingly reforested areas balances the significant inputs of these same cations in precipitation.^[20,21] Acidic deposition affects forest soils more than agricultural or urban soils because the latter are routinely limed to neutralize acidity.

Although it is possible to lime forest soils, which is done frequently in some European countries, the logistics and cost often preclude this except in areas severely impacted by acidic deposition.^[5]

Excessively acidic soils are undesirable for several reasons. Direct phytotoxicity from soluble Al^{3+} or Mn^{2+} can occur and seriously injure plant roots, reduce plant growth, and increase plant susceptibility to pathogens.^[21]

The relationship between Al^{3+} toxicity and soil pH is complicated by the fact that in certain situations organic matter can form complexes with Al^{3+} that reduce its harmful effects on plants.^[18] Acid soils are usually less fertile because of a lack of important basic cations such as potassium ion (K^+), Ca^{2+} , and Mg^{2+} . Leguminous plants may fix less N_2 under very acidic conditions due to reduced rhizobial activity and greater soil adsorption of molybdenum by clays and Al and Fe oxides.^[2] Mineralization of N, phosphorus, and S can also be reduced because of the lower metabolic activity of bacteria. Many plants and microorganisms have adapted to very acidic conditions (e.g., pH < 5.0). Examples include ornamentals such as azaleas and rhododendrons and food crops such as cassava, tea, blueberries, and potatoes.^[5,22] In fact, considerable efforts in plant breeding and biotechnology are directed toward developing Al- and manganese-tolerant plants that can survive in highly acidic soils.

Agricultural Ecosystems

Acidic deposition contains N and S that are important plant nutrients. Therefore, foliar applications of acidic deposition at critical growth stages can be beneficial to plant development and reproduction. Generally, controlled experiments require the simulated acid rain to have pH of 3.5 or less in order to produce injury to certain plants.^[22] The amount of acidity needed to damage some plants is 100 times greater than natural rainfall. Crops that respond negatively in simulated acid rain studies include garden beets, broccoli, carrots, mustard greens, radishes, and pinto beans, with different effects for some cultivars. Positive responses to acid rain have been identified with alfalfa, tomato, green pepper, strawberry, corn, lettuce, and some pasture grass crops.

Agricultural lands are maintained at pH levels that are optimal for crop production. In most cases, the ideal pH is around pH 6.0–7.0; however, pH levels of organic soils are usually maintained at closer to pH 5.0. Because agricultural soils are generally well buffered, the amount of acidity derived from atmospheric inputs is not sufficient to significantly alter the overall soil pH.^[2] N and S soil inputs from acidic deposition are beneficial, and with the reduction in S atmospheric levels mandated by 1990 amendments to the Clean Air Act, the S fertilizer market has grown. The amount of N added to agricultural ecosystems as acidic deposition is rather insignificant in relation to the 100–300 kg N/ha/yr required for most agricultural crops.

Forest Ecosystems

Perhaps the most publicized issue related to acidic deposition has been widespread forest decline; e.g., in Europe estimates suggest that as much as 35% of all forests have been affected.^[23] Similarly, in the United States many important forest ranges such as the Adirondacks of New York, the Green Mountains of Vermont, and the Great Smoky Mountains in North Carolina have experienced sustained decreases in tree growth for several decades.^[6] Conclusive evidence that forest decline or dieback is caused solely by acidic deposition is lacking and complicated by interactions with other environmental or biotic factors. However, NAPAP research^[6] has confirmed that acidic deposition has contributed to a decline in high-elevation red spruce in the northeastern United States. In addition, N saturation of forest ecosystems from atmospheric N deposition is believed to result in increased plant growth, which in turn increases water and nutrient use followed by deficiencies that can cause chlorosis and premature needle-drop as well as increased leaching of base cations from the soil.^[24]

Acidic deposition on leaves may enter directly through plant stomates.^[1,22] If the deposition is sufficiently acidic (pH ~3.0), damage can also occur to the waxy cuticle, increasing the potential for direct injury of exposed leaf mesophyll cells. Foliar lesions are one of the most common symptoms. Gaseous compounds such as SO_2 and sulfur trioxide present in acidic mists or fogs can also enter leaves through the stomates, form H_2SO_4 upon reaction with H_2O in the cytoplasm, and disrupt many metabolic processes. Leaf and needle necrosis occurs when plants are exposed to high levels of SO_2 gas, possibly due to collapsed epidermal cells, eroded cuticles, loss of chloroplast integrity and decreased chlorophyll content, loosening of fibers in cell walls and reduced cell membrane integrity, and changes in osmotic potential that cause a decrease in cell turgor.

Root diseases may also increase in excessively acidic soils. In addition to the damages caused by the exposure to H_2SO_4 and HNO_3 , roots can be directly injured or their growth rates impaired by increased concentrations of soluble Al^{3+} and Mn^{2+} in the rhizosphere^[2,25] (<http://nadp.sws.uiuc.edu>). Changes in the amount and composition of these exudates can then alter the activity and population diversity of soilborne pathogens. The general tendency associated with increased root exudation is an enhancement in microbial populations due to an additional supply of carbon (energy). Chronic acidification can also alter nutrient availability and uptake patterns.^[8,22]

Long-term studies in New England suggest that acidic deposition has caused significant plant and soil leaching of base cations,^[1,21] resulting in decreased growth of red spruce trees in the White Mountains.^[6] With reduction in about 80% of the airborne base cations, mainly Ca^{2+} but also Mg^{2+} , from 1950 levels, researchers suggest that forest growth has slowed because soils are not capable of

weathering at a rate that can replenish essential nutrients. In Germany, acidic deposition was implicated in the loss of soil Mg^{2+} as an accompanying cation associated with the downward leaching of SO_4^{2-} , which ultimately resulted in forest decline.^[2] Several European countries have used helicopters to fertilize and lime forests.

Aquatic Ecosystems

Ecological damage to aquatic systems has occurred from acidic deposition. As with forests, a number of interrelated factors associated with acidic deposition are responsible for undesirable changes. Acidification of aquatic ecosystems is not new. Studies of lake sediments suggest that increased acidification began in the mid-1800s, although the process has clearly accelerated since the 1940s.^[15] Studies indicate that there is significant S mineralization in forest soils impacted by acidic deposition and that the SO_4^{2-} levels in adjacent streams remain high, even though there has been a decrease in the amount of atmospheric S deposition.^[24]

Geology, soil properties, and land use are the main determinants of the effect of acidic deposition on aquatic chemistry and biota. Lakes and streams located in areas with calcareous geology resist acidification more than those in granitic and gneiss materials.^[16] Soils developed from calcareous parent materials are generally deeper and more buffered than thin, acidic soils common to granitic areas.^[2] Land management decisions also affect freshwater acidity. Forested watersheds tend to contribute more acidity than those dominated by meadows, pastures, and agronomic ecosystems.^[8,14,20] Trees and other vegetation in forests are known to “scavenge” acidic compounds in fogs, mists, and atmospheric particulates. These acidic compounds are later deposited in forest soils when rainfall leaches forest vegetation surfaces. Rainfall below forest canopies (e.g., throughfall) is usually more acidic than ambient precipitation. Silvicultural operations that disturb soils in forests can increase acidity by stimulating the oxidization of organic N and S and reduced S compounds such as FeS_2 .^[2]

A number of ecological problems arise when aquatic ecosystems are acidified below pH 5.0 and particularly below pH 4.0. Decreases in biodiversity and primary productivity of phytoplankton, zooplankton, and benthic invertebrates commonly occur.^[15,16] Decreased rates of biological decomposition of organic matter have occasionally been reported, which can then lead to a reduced supply of nutrients.^[20] Microbial communities may also change, with fungi predominating over bacteria. Proposed mechanisms to explain these ecological changes center around physiological stresses caused by the exposure of biota to higher concentrations of Al^{3+} , Mn^{2+} , and H^+ and lower amounts of available Ca^{2+} .^[15] One specific mechanism suggested involves the disruption of ion uptake and the ability of aquatic plants to regulate sodium ion (Na^+), K^+ , and Ca^{2+} export and import from cells.

Acidic deposition is associated with declining aquatic vertebrate populations in acidified lakes and, under conditions of extreme acidity, of fish kills. In general, if the water pH remains above 5.0, few problems are observed; from pH 4.0 to 5.0, many fish are affected, and below pH 3.5, few fish can survive.^[23] The major cause of fish kill is due to the direct toxic effect of Al^{3+} , which interferes with the role Ca^{2+} plays in maintaining gill permeability and respiration. Ca has been shown to mitigate the effects of Al^{3+} , but in many acidic lakes, the Ca^{2+} levels are inadequate to overcome Al^{3+} toxicity. Low pH values also disrupt the Na^+ status of blood plasma in fish. Under very acidic conditions, H^+ influx into gill membrane cells both stimulates excessive efflux of Na^+ and reduces influx of Na^+ into the cells. Excessive loss of Na^+ can cause mortality. Other indirect effects include reduced rates of reproduction, high rates of mortality early in life or in reproductive phases of adults, and migration of adults away from acidic areas.^[16] Amphibians are affected in much the same manner as fish, although they are somewhat less sensitive to Al^{3+} toxicity. Birds and small mammals often have lower populations and lower reproductive rates in areas adjacent to acidified aquatic ecosystems. This may be due to a shortage of food due to smaller fish and insect populations or due to physiological stresses caused by consuming organisms with high Al^{3+} concentrations.

REDUCING ACIDIC DEPOSITION EFFECTS

Damage caused by acidic deposition will be difficult and extremely expensive to correct, which will depend on our ability to reduce S and N emissions; e.g., society may have to burn less fossil fuel, use cleaner energy sources, and/or design more efficient “scrubbers” to reduce S and N gas entering our atmosphere. Despite the firm conviction of most nations to reduce acidic deposition, it appears that the staggering costs of such actions will delay implementation of this approach for many years. The 1990 amendments to the Clean Air Act are expected to reduce acid-producing air pollutants from electric power plants. The 1990 amendments established emission allowances based on a utilities’ historical fuel use and SO_2 emissions, with each allowance representing 1 ton of SO_2 that can be bought, sold, or banked for future use.^[4,6,7] (<http://www.nnmc.noaa.gov/CENR/NAPAP/>). Short-term remedial actions for acidic deposition are available and have been successful in some ecosystems. Liming of lakes and some forests (also fertilization with trace elements and Mg^{2+}) has been practiced in European countries for over 50 years.^[16,23] Hundreds of Swedish and Norwegian lakes have been successfully limed in the past years. Lakes with short mean residence times for water retention may need annual or biannual liming; others may need to be limed every 5–10 years. Because vegetation in some forested ecosystems has adapted to acidic soils, liming (or overliming) may result in an unpredictable and undesirable redistribution of plant species.

REFERENCES

1. Smith, W.H. Acid rain. In *The Wiley Encyclopedia of Environmental Pollution and Cleanup*; Meyers, R.A., Dittrick, D.K., Eds.; Wiley: New York, 1999; 9–15.
2. Pierzynski, G.M.; Sims, J.T.; Vance, G.F. *Soils and Environmental Quality*; CRC Press: Boca Raton, 2000; 459 pp.
3. Wolff, G.T. Air pollution. In *The Wiley Encyclopedia of Environmental Pollution and Cleanup*; Meyers, R.A., Dittrick, D.K., Eds.; Wiley: New York, 1999; 48–65.
4. U.S. Environmental protection agency. *Progress Report on the EPA Acid Rain Program*; EPA-430-R-99-011; U.S. Government Printing Office: Washington, 1999; 20 pp.
5. Forster, B.A. *The Acid Rain Debate: Science and Special Interests in Policy Formation*; Iowa State University Press: Ames, 1993.
6. *National Acid Precipitation Assessment Program Task Force Report*. National Acid Precipitation Assessment Program 1992 Report to Congress; U.S. Government Printing Office: Pittsburgh, 1992; 130 pp.
7. National Science and Technology Council. *National Acid Precipitation Assessment Program Biennial Report to Congress: An Integrated Assessment*; 2005. Available at <http://www.esrl.noaa.gov/csd/AQRS/reports/napapreport05.pdf>.
8. Charles, D.F., Ed. The acidic deposition phenomenon and its effects: Critical assessment review papers. In *Effects Sciences*; EPA-600/8-83-016B; U.S. Environmental Protection Agency: Washington, 1984; Vol. 2.
9. McKinney, M.L.; Schoch, R.M. *Environmental Science: Systems and Solutions*; Jones and Bartlett Publishers: Sudbury, 1998.
10. United Nations, World Bank and World Resources Institute. *World Resources: People and Ecosystems—The Fraying Web of Life*; Elsevier: New York, 2000.
11. *National Atmospheric Deposition Program (NRSP-3)/National Trends Network*; Isopleth Maps. NADP Program Office, Illinois State Water Survey, 2204 Griffith Dr., Champaign, IL, 61820, 2000.
12. Council on Environmental Quality. *Environmental Quality, 18th and 19th Annual Reports*; U.S. Government Printing Office: Washington, 1989.
13. Charles, D.F., Ed. *Acid Rain Research: Do We Have Enough Answers?* In Proceedings of a Speciality Conference, Studies in Environmental Science #64; Elsevier: New York, 1995.
14. Kamari, J. *Impact Models to Assess Regional Acidification*; Kluwer Academic Publishers: London, 1990.
15. Charles, D.F., Ed. *Acidic Deposition and Aquatic Ecosystems*; Springer-Verlag: New York, 1991.
16. Mason, B.J. *Acid Rain: Its Causes and Effects on Inland Waters*; Oxford University Press: New York, 1992.
17. U.S. Environmental Protection Agency. Effects of acid rain: Human health. In *EPA Environmental Issues Website*. Available at <http://www2.epa.gov/learn-issues> (accessed June 26, 2001).
18. Marion, G.M.; Hendricks, D.M.; Dutt, G.R.; Fuller, W.H. Aluminum and silica solubility in soils. *Soil Sci.* **1976**, *121*, 76–82.
19. Kennedy, I.R. *Acid Soil and Acid Rain*; Wiley: New York, 1992.
20. Reuss, J.O.; Johnson, D.W. *Acid Deposition and the Acidification of Soils and Waters*; Springer-Verlag: New York, 1986.
21. Likens, G.E.; Driscoll, C.T.; Buso, D.C. Long-term effects of acid rain: Response and recovery of a forest ecosystem. *Science* **1996**, *272*, 244–246.
22. Linthurst, R.A. *Direct and Indirect Effects of Acidic Deposition on Vegetation*; Butterworth Publishers: Stoneham, 1984.
23. Bush, M.B. *Ecology of a Changing Planet*; Prentice-Hall: Upper Saddle River, 1997.
24. Alawell, C.; Mitchell, M.J.; Likens, G.E.; Krouse, H.R. Sources of stream sulfate at the Hubbard Brook experimental forest: Long-term analyses using stable isotopes. *Biogeochemistry* **1999**, *44*, 281–299.
25. National Atmospheric Deposition Program. *Nitrogen in the Nation's Rain*; NADP Brochure 2000–01a.

Acid Sulfate Soils

Delvin S. Fanning

*Department of Environmental Science and Technology, University of Maryland,
College Park, Maryland, U.S.A.*

Abstract

Acid sulfate soils are a special group of mineral soils. They either have ultralow pH in near surface horizons or they have the potential to become ultra acid upon the exposure of sulfide minerals that they contain to oxidizing conditions, or they have been ultra acid in the past as evidenced by the presence in their profiles of the yellow iron hydroxysulfate mineral, jarosite, or other special characteristics. Sulfuric acid formed from the oxidation of iron sulfides, in most cases from the mineral pyrite, is the acid of concern with these soils and with waters that emanate from them.

INTRODUCTION

Pons^[1] reported that acid sulfate soils have been known for ages, which began to receive scientific attention in the 18th century when Linnaeus recognized them in the Netherlands with terms such as *argilla vitriolacea*, meaning “clay with sulfuric acid.” Terms such as “kattekleigronden” in Dutch, “cat clay soils” in English, and “Maibolt” in German, which are used to imply hayfields affected by an evil spirit or “Gifterde” for poison earth, were applied to these soils to connote the mysterious and evil circumstances surrounding the difficulty in producing crops on them. The term “acid sulfate soils” is of more modern usage. It has gained popularity with the four international acid sulfate soils symposia^[2–5] that have been held under the sponsorship of the International Land Reclamation Institute (ILRI), Wageningen, the Netherlands, and other organizations, and usage of the term is apparent in many other publications^[6–8] on these soils.

DEFINITION AND MAIN KINDS

In the broad sense, acid sulfate soils include all soils in which sulfuric acid may be produced, is being produced, or has been produced in amounts that have a lasting effect on main soil characteristics.^[1,9] This definition includes potential, active, and postactive acid sulfate soils, which are broad genetic kinds as described in the following three paragraphs.

Potential acid sulfate soils are anaerobic soils, commonly occurring in, or at one time formed in, coastal (tidal) sedimentary environments affected by sulfidization. Potential acid sulfate soils contain sulfide minerals at such levels in near surface horizons/layers that they are expected to generate, upon exposure to oxidizing conditions, sufficient sulfuric acid to drive the pH of these horizons/layers to

ultralow levels. Under these conditions, most plants would be unable to grow on the soils, and an active acid sulfate soil would then be recognized.

Active acid sulfate soils form where sulfide minerals (most typically iron sulfides and the mineral pyrite) have oxidized in near surface horizons and formed enough sulfuric acid, with insufficient neutralization, in order to make the pH drop to ultralow levels. Most commonly the pH, as measured in water, is 3.5 or less, such that a “sulfuric horizon,” as defined later, is recognized. Some soil scientists would only require the pH to be 4.0 or below to recognize the active stage.

In postactive acid sulfate soils, weathering and pedogenesis have proceeded beyond the active stage to where sulfide minerals are no longer present in surface soil horizons, and the pH in these horizons has risen to levels above that which would cause them to be recognized as sulfuric horizons.

A simpler definition of acid sulfate^[8] is simply that they are soils that contain iron sulfides. This definition calls attention to the minerals that cause acid sulfate soils to become acid; however, it neglects to consider that the degree of acidity of the soils depends on the balance between the acid-forming substances (mainly iron sulfides) and the substances (minerals) that neutralize acidity, most commonly calcium carbonate minerals. This simpler definition also does not recognize the genetic distinction among the potential, active, and postactive stages of development of acid sulfate soils.

CLASSIFICATION IN SOIL TAXONOMY

Potential, active, and early postactive acid sulfate soils receive special recognition in *Soil Taxonomy*.^[10–12] The taxonomic definition of “sulfidic materials” also permits the recognition of these acid-forming materials, regardless

of their depth in the soil–geologic column. This is useful for their recognition in construction activities such as mining, highway construction, and dredging operations, which can bring active acid sulfate soils into existence by exposing (to oxidation) sulfidic materials that previously occurred deep in the column or under deep water.

Diagnostic Characteristics

Sulfidic Materials

Soil Taxonomy^[10] gives the following definition:

Sulfidic materials contain oxidizable sulfur compounds. They are mineral or organic soil materials that have a pH value of more than 3.5 and that, if incubated as a layer 1 cm thick under moist aerobic conditions (field capacity) at room temperature, show a drop in pH of 0.5 or more units to a pH value of 4.0 or less (1 : 1 by weight in water or in a minimum of water to permit measurement) within 8 weeks.

Sulfidic materials were defined to permit the recognition of soil materials that lead to the formation of active acid sulfate soils if exposed to aerobic conditions at land surfaces. Sulfidic materials, when they are mineral soil materials, typically have Munsell color chromas ≤ 1 and values ≤ 4 .^[12]

Sulfuric Horizon

Following is the definition of sulfuric horizon as given in *Soil Taxonomy*.^[10,11]

The sulfuric (*L. sulfur*) horizon is 15 cm or more thick and is composed of either mineral or organic soil material that has a pH value of 3.5 or less (1 : 1 in water or in a minimum of water to permit measurement) and shows evidence that the low pH value is caused by sulfuric acid. The evidence is *one or more* of the following: (a) Jarosite concentrations; (b) directly underlying sulfidic materials; or (c) 0.05 percent or more water-soluble sulfate.

The definition of the sulfuric horizon permits the recognition of active acid sulfate soils and soil materials affected by active sulfuricization.

Classes

Potential and active acid sulfate soils are classified in *Soil Taxonomy* in the orders of Entisols, Inceptisols, and Histosols. Within these orders, acid sulfate soils are recognized at the great group and subgroup levels, which are intermediate categories—third and fourth in level from the top (order) level of the six categories of *Soil Taxonomy*. Mineral potential acid sulfate soils that have sulfidic materials within 50 cm of their surfaces without an overlying sulfuric horizon

are classified in the great group of Sulfaquepts. Organic potential acid sulfate soils that have sulfidic materials within 100 cm of their surfaces without a sulfuric horizon within 50 cm of their surface are classified as either Sulphemists or Sulfisapristis.

Active acid sulfate soils are recognized in *Soil Taxonomy* as soils that have a sulfuric horizon with its upper boundary within 50 cm of the soil surface. Those that are mineral soils are classified as either Sulfaquepts or Sulfudepts, whereas those that are organic soils are classified as either Sulfohemists or Sulfosapristis. Most of these soils would be wet soils, belonging to an Aqu-suborder or are organic soils that formed under wet conditions, those with the “ist” stem; however, the Sulfudepts are better drained and would be the active acid sulfate soils of upland exposures, such as on mined land spoils that were not reclaimed properly.

Postactive acid sulfate soils are recognized in *Soil Taxonomy* only if they are early postactive. For example, Sulfuric Endoaquepts are Endoaquepts that have either sulfidic materials or a sulfuric horizon or “a horizon 15 cm or more thick that has all of the characteristics of a sulfuric horizon, except that it has a pH between 3.5 and 4.0” within 150 cm of the mineral soil surface.^[10,11] Many other soils beyond those recognized by *Soil Taxonomy* are considered to be postactive, such as certain Alfisols and Ultisols in Texas^[13] and certain Ultisols in Maryland.^[14] It is important to recognize such soils because they do usually contain sulfidic materials at a depth that could give rise to new active acid sulfate soils if unearthed by land-moving operations and to understand better the morphology, genesis, and other aspects of these soils.^[15]

CLASSIFICATION IN OTHER SYSTEMS

Dent^[6] has presented the ILRI system for the classification of acid sulfate soils, which considers acidity and potential acidity, salinity, soil composition and texture, degree of physical ripening, and profile form (the depth zone at which various properties occur). The Food and Agriculture Organization (FAO) of the United Nations groups potential and active acid sulfate soils together in thionic classes for purposes of showing these soils on the FAO/United Nations Educational, Scientific and Cultural Organization soil map of the world.^[16] Thionic Fluvisols, thionic Gleysols, and thionic Histosols are recognized.^[7]

CONCLUSION

Acid sulfate soils are a special group of mineral soils, although some organic soils are included. They either have ultralow pH (3.5, measured in water) in near surface horizons (active acid sulfate soils) or they have the potential to become ultra acid upon the exposure of sulfide minerals that they contain to oxidizing conditions (potential acid

sulfate soils), or they have been ultra acid in the past as evidenced by the presence in their profiles of the yellow iron hydroxysulfate mineral jarosite or other special characteristics (postactive acid sulfate soils). Sulfuric acid formed from the oxidation of iron sulfides, in most cases from the mineral pyrite, FeS_2 , is the acid of concern with these soils and with waters that emanate from them. Special characteristics, sulfidic materials and the sulfuric horizon, have been defined in *Soil Taxonomy* to enable the recognition and classification of these soils. The soils receive special classification in other classification systems as well.

REFERENCES

1. Pons, L.J. Outline of the genesis, characteristics, classification and improvement of acid sulphate soils. In *Acid Sulphate Soils*, Proceedings of the International Symposium, Wageningen, the Netherlands, Aug 13–29, 1972; Dost, H., Ed.; International Land Reclamation Institute Pub. 18; Wageningen, 1973; Vol. 1, 3–27.
2. Dost, H. Acid sulphate soils. In *Proceedings of the International Symposium*. Wageningen, the Netherlands, Aug 13–29, 1972, International Land Reclamation Institute Pub. 18, International Institute for Land Reclamation and Improvement: Wageningen, 1973.
3. Dost, H.; van Breemen, N. Eds. *Proceedings of the Second International Symposium on Acid Sulphate Soils: Second International Symposium on Acid Sulphate Soils*, Bangkok, Thailand, Jan 8–24, 1981, International Land Reclamation Institute Pub. 31, International Institute for Land Reclamation and Improvement: Wageningen, 1982.
4. Dost, H. *Selected Papers of the Dakar Symposium on Acid Sulphate Soils*, Dakar, Senegal, Jan 1986, International Land Reclamation Institute Pub. 44, International Institute for Land Reclamation and Improvement: Wageningen, 1988.
5. Dent, D.L.; van Mensvoort, M.E.F., Eds. *Selected Papers of the Ho Chi Minh City Symposium on Acid Sulphate Soils*, Ho Chi Minh City, Vietnam, Mar, 1982, International Land Reclamation Institute Pub. 53, International Institute for Land Reclamation and Improvement: Wageningen, 1993.
6. Dent, D. *Acid Sulphate Soils: A Baseline for Research and Development*; International Land Reclamation Institute Pub. 39, International Institute for Land Reclamation and Improvement: Wageningen, 1986; 82–83.
7. Kittrick, J.A.; Fanning, D.S.; Hossner, L.R. *Acid Sulfate Weathering*, Special Pub. No. 10; Soil Science Society of America: Madison, 1982; 127–146.
8. Sammut, J. *An Introduction to Acid Sulfate Soils*; New South Wales Department of Agriculture: Wollongbar, 1997; 23 pp.
9. Fanning, D.S.; Burch, S.N. Coastal acid sulfate soils. In *Reclamation of Drastically Disturbed Lands*, Agronomy 41; Barnhisel, R.I., Darmody, R.G., Daniels, W.L., Eds.; American Society of Agronomy: Madison, 2000; 921–937.
10. Soil Survey Staff. *Soil Taxonomy*, U.S. Department Agriculture Handbook, 2nd Ed.; U.S. Government Printing Office: Washington, 1999.
11. Soil Survey Staff. *Keys to Soil Taxonomy*, 9th Ed.; U.S. Department of Agriculture, Natural Resources Conservation Service: Washington, 2003. Available at http://soils.usda.gov/technical/classification/tax_keys/keysweb.pdf
12. Fanning, D.S.; Rabenhorst, M.C.; Bigham, J.M. Colors of acid sulfate soils. In *Soil Color*; Bigham, J.M., Ciolkosz, E.J., Eds.; Soil Science Society of America Spec. Pub No 31; Soil Science Society of America: Madison, 1993; 91–108.
13. Carson, C.D.; Fanning, D.S.; Dixon, J.B. Alfisols and ultisols with acid sulfate weathering features in Texas. In *Acid Sulfate Weathering*; Kittrick, J.A., Fanning, D.S., Hossner, L.R., Eds.; Soil Science Society of America: Madison, 1982; 127–146.
14. Wagner, D.P.; Fanning, D.S.; Foss, J.E.; Patterson, M.S.; Snow, P.A. Morphological and mineralogical features related to sulfide oxidation under natural and disturbed land surfaces in Maryland. In *Acid Sulfate Weathering*; Kittrick, J.A., Fanning, D.S., Hossner, L.R., Eds.; Soil Sci. Soc. Amer. Spec. Pub. No. 10; Soil Science Society of America: Madison, 1982; 109–125.
15. Fanning, D.S.; Burch, S.N. Acid sulphate soils and some associated environmental problems. In *Soils and Environment*; Auerswald, K., Stanjek, H., Bigham, J.M., Eds.; Advances in Geocology 30; Catena Verlag: Reiskirchen, 1997; 145–158.
16. FAO/UNESCO. *Soil Map of the World, Revised Legend*; Food and Agriculture Organization of the United Nations: Rome, 1990.

Acid Sulfate Soils: Distribution and Extent

Wim Andriessse

Green World Research, Wageningen University, Wageningen, the Netherlands

M.E.F. van Mensvoort

Laboratory of Soil Science and Geology, Wageningen University and Research Center, Wageningen, the Netherlands

Abstract

This entry discusses the global and regional distribution of acid sulfate soils and their extent. Acid sulfate soils, in spite of their notoriety, do not cover large acreages of land worldwide. They are important, however, in terms of the specific management they require in order to avoid catastrophic acidification upon drainage and in terms of their specific occurrence that is genetically linked to coastal areas. Mostly, these coastal lands comprise densely populated areas of high economic importance.

INTRODUCTION

International scientific attention to acid sulfate soils has been channeled through four international symposia^[1–4] under the aegis of the International Institute for Land Reclamation and Improvement (ILRI) in Wageningen, The Netherlands. This entry discusses the global and regional distribution of acid sulfate soils and their extent. Acid sulfate soils, in spite of their notoriety, do not cover large acreages of land worldwide. They are important, however, in terms of the specific management they require in order to avoid catastrophic acidification upon drainage and in terms of their specific occurrence that is genetically linked to coastal areas.^[5,6] Mostly, and this is particularly the case in South and Southeast Asia, West, and Southern Africa, and along the South American and Australian coastlines, these coastal lands comprise densely populated areas of high economic importance. Here, alternative land for food production and cash crops is scarce. See the related entries *Acid Sulfate Soils* (p. 17–19), *Acid Sulfate Soils: Formation* (p. 26–30), *Acid Sulfate Soils: Management* (p. 31–35), and *Acid Sulfate Soils: Problems* (p. 36–39) for more information.^[6,7]

GLOBAL DISTRIBUTION AND EXTENT

Estimates of the total area of potential and actual acid sulfate soils in the world are in the order of 12–13 millionha (Table 1). Approximately, 10 millionha are known to occur in tropical regions.^[5,8–10] Over much larger areas (potential) acid sulfate soil materials occur under thick covers of peat (mainly in Indonesia and Malaysia) or under non-pyritic alluvial sediments. In addition, Pleistocene, Tertiary,

or still-older pyritic sediments, often originating from tidal environments, occur in inland positions. Examples have been found in Canada and the United States (many authors), the Soviet Union, Uganda, Great Britain, Denmark, Zimbabwe, Germany, and The Netherlands.^[11–17]

In the FAO–UNESCO Legend for the Soil Map of the World^[18] acid sulfate soils, both potential and actual are called Thionic Fluvisols. Beek et al.,^[5] ISRIC,^[9] and Dent^[6] describe the world distribution of Thionic Fluvisols based on the work of Kawalec^[8] and FAO.^[18] Large extensions occur in South and Southeast Asia, in West Africa, and along the northeastern coast of South America (the Guyana's, the Orinoco, and Amazon deltas). Smaller areas are found in coastal regions of Eastern and Southern Africa, particularly in Madagascar, along the Australian coastline and in the Caribbean. In Southeast Asia, the bulk of the acid sulfate soils (approximately 5 millionha) is found in Indonesia, Thailand, and Vietnam. Millions of hectares of peat land in Indonesia (Kalimantan and Sumatra) are underlain by potentially acid sediments. These peat lands do not qualify as Thionic Fluvisols properly, and their extent has not been included in the tables. It should be noted that, after a major revision, the FAO Legend recognizes Thionic Gleysols and Thionic Histosols as well as Thionic Fluvisols.^[19]

Table 1 has been compiled on the basis of data from the FAO–UNESCO Soil Map of the World.^[18] In Table 2, these data have been updated, based on an extensive literature survey^[10] by Langenhoff in 1986. In the last column of Table 2, the authors have again revised the acreages of potential and actual acid sulfate soils based on new information and on expert knowledge, including an accuracy assessment of the available information. Their new estimate amounts to a world total of over 17 millionha. This higher figure is mainly due to the “discovery” of acid sulfate soils in Australia since the 1980s.

Table 1 Regional distribution of acid sulfate soils.

Region	Area (million ha)	Area (million ha) per length of growing period (day)			
		<90	90–180	180–300	>300
Africa	3.7	0.4	0.7	1.5	1.1
Near and middle East	—	—	—	—	—
Asia	6.7	—	0.2	5.1	1.4
Australia	—	—	—	—	—
Latin America	2.1	—	0.1	0.8	1.2
North America	0.1	—	0.1	—	—
Europe	—	—	—	—	—
World total	12.6	0.4	1.1	7.4	3.7

Note: Soil extents based on the FAO–UNESCO Soil Map of the World: Lengths of growing periods according to FAO agro-ecological zones project.

Source: From FAO.^[18,20]

REGIONAL DISTRIBUTION

At landscape level, saline and brackish-water tidal swamps and marshes form the ecological environment for the formation of potential acid sulfate soils. The prevailing dense vegetation fuels the process of pyrite formation, the tidal cycle brings in sediment, renews the supply of dissolved sulfate, and removes soluble by-products.^[6,21] Slight differences within this general setting may cause important spatial variations in potential acidity, even over small distances.

According to Pons et al.,^[22,23] regional soil patterns are related to changes in relative sea level and the rate of sedimentation, as well as to changes in climate, hydrology, and chemical composition of floodwaters. In general, where sedimentation keeps pace with the rising sea level, broad and stable tidal swamp zones build up in which thick layers of sulfidic materials accumulate. If sea levels rise slowly and heavy sedimentation continues, coastal swamp formation extends rapidly in seaward direction and sediments are

Table 2 World distribution and extent of potential and actual acid sulfate soils.

Region/country	Area (1,000 ha)			
	Source: FAO, 1974 ^[18]		Revised estimates	
			Other sources	1986 ^a
Africa				
Benin	—	10 ^[24]	10	10
Cameroon	35	<200 (“mangrove swamps” ^[25])	100	100
Cte d'Ivoire	10	10 ^[26]	10	10
Ethiopia	150	—	150	150
Gabon	350	—	350	350
Gambia	100	500 ^[27–31]	500	500
Guinea	305	375 ^[32,33]	375	375
Guinea Bissau	530	500 ^[34]	530	500
Kenya	—	50 ^[35]	50	50
Liberia	—	30 (“mangrove swamps” ^[36])	15	15
Madagascar	530	500 ^[37]	530	500
Mauritania	130	—	130	130
Mozambique	170	—	170	170
Nigeria	765	900 (“mangrove swamps” ^[38,39])	765	900
Senegal	375	600 ^[28–30,39]	600	600
Sierra Leone	195	120 ^[40]	120	120
Tanzania	10	—	10	10
Total: Africa	3,655	4,415	4,490	
Asia				
Bangladesh	350	700, ^[41] 200–300 ^[42]	350	350
Burma	250	180 ^[43]	180	180
Cambodia	205	200 ^[44]	200	200

(Continued)

Table 2 World distribution and extent of potential and actual acid sulfate soils. (Continued)

Region/country	Area (1,000 ha)			
	Source: FAO, 1974 ^[18]	Other sources	Revised estimates	
			1986 ^a	2002 ^b
China	—	65; ^[44] 116 ^[45]	65	115
India	560	110 (Kerala only ^[46]); >80; ^[47] 400 ^[48]	400	400
Indonesia	1,075	2,000 ^[49,50]	1,500	2,000
Japan	—	10 ^[51,52]	—	10
Malaysia	480	>140; ^[53–55] 200 ^[56]	200	200
Philippines	100	<500; ^[57] 200 ^[58]	200	200
South Korea	—	5 ^[44]	5	5
Sri Lanka	—	—	—	15
Thailand	1,030	1,000; ^[59,60] 840; ^[61] 670 ^[62]	670	840
Vietnam	1,560	1,500; ^[63] 1,800 (Mekong Delta ^[64]); 1,000; ^[65] >200 (Red River Delta ^[66])	2,000	2,000
Total: Asia	5,610	5,770	6,515	
Australia	—	3,000 ^[67]	—	3,000
New Zealand	—	—	—	—
Latin America and the Caribbean				
Brazil	1,100	—	1,100	1,100
French Guyana	75	150 ^[68,69]	500 (Mara deposits covered with peat of varying depth ^[70])	500
Guyana	150	—		
Surinam	210	—		
Uruguay	35	—	35	35
Venezuela	500	—	500	500
Caribbean Isles (total)	—	—	—	15
Central America	650	—	650	650
Total: Latin America and the Caribbean	2,720	2,785	2,800	
North America (United States and Canada)	100	—	100	100
Europe				
Netherlands	5	5 ^[71]	5	5
Germany	—	—	—	5
Denmark	—	—	—	5
Finland	—	160 ^[72,73]	—	160
France	—	—	—	5
United Kingdom	—	5	5	
Sweden	—	50 ^[74]	—	50
Total: Europe	5	10	235	
Total: World	12,090	13,080	17,100	

^aLangenhoff.^[10]^bUpdated expert judgment by present authors.

generally thinner and have lower pyrite contents. This is mainly caused by the unfavorable conditions for development of dense mangrove and reed vegetation. Favorable conditions for the formation of pyritic sediments as

outlined above occur in coastal deltas, sheltered estuaries, and coastlines protected by offshore islands and bars (e.g., the lagoons along the West African coastline between Côte d'Ivoire and Nigeria).

REFERENCES

1. Dost, H., Ed. Acid sulphate soils. In *Proceedings of the First International Symposium*, Wageningen 1972; ILRI Publication 18 (2 Vols); International Institute for Land Reclamation and Improvement: Wageningen, 1973; 295–406.
2. Dost, H.; Van Breemen, N. In *Proceedings of the Bangkok Symposium on Acid Sulphate Soils*, Bangkok, 1981. ILRI Publication 31; International Institute for Land Reclamation and Improvement: Wageningen, 1982; 450 pp.
3. Dost, H. *Selected Papers of the Dakar Symposium on Acid Sulphate Soils*, Dakar, Senegal, Jan 1986; ILRI Publication 44; International Institute for Land Reclamation and Improvement: Wageningen, 1988; 251 pp.
4. Dent, D.L.; Van Mensvoort, M.E.F. *Selected Papers of the Ho Chi Minh City Symposium on Acid Sulphate Soils, Ho Chi Minh City, Vietnam, Mar 1982*; ILRI Publication 53; International Institute for Land Reclamation and Improvement: Wageningen, 1993; 425 pp.
5. Beek, K.J.; Blokhuis, W.A.; Driessen, P.M.; Van Breemen, N.; Brinkman, R.; Pons, L.J. Problem soils: Their reclamation and management. In *Land Reclamation and Water Management, Developments, Problems and Challenges*; ILRI, Ed; ILRI Publication 27; International Institute for Land Reclamation and Improvement: Wageningen, 1980; 191.
6. Dent, D.L. *Acid Sulphate Soils: A Baseline for Research and Development*; ILRI Publication 39; International Institute for Land Reclamation and Improvement: Wageningen, 1986; 204 pp.
7. Brinkman, R. *Directions for Further Research on Acid Sulphate Soils*; In *Proceedings of Bangkok Symposium on Acid Sulphate Soils*; Dost, H., Van Breemen, N., Eds.; ILRI Publications 31: Wageningen, 1982; 12–20.
8. Kawalec, A. World distribution of acid sulphate soils: References and map; In *Proceedings of International Symposium*; Dost, H., Ed.; ILRI Publications 31: Wageningen, 1973; 293–295.
9. ISRC. *Thionic Fluvisols. Soil Monolith Paper 1*; International Soil Reference and Information Centre: Wageningen, 1981.
10. Langenhoff, R. *Distribution, Mapping, Classification and Use of Acid Sulphate Soils in the Tropics. A Literature Study*; Internal Communication Report Series 74; Netherlands Soil Survey Institute (STIBOKA): Wageningen, 1986; 133 pp.
11. Verigina, K.V. Processes of the movement and accumulation of iron during soil formation. *Trudy Pochvennogo Instituta imeni V.V. Dokuchaeva. Akademiya nauk SSSR* **1950**, *34*, 190–201.
12. Chenery, E.M. Acid sulphate soils in central Africa. In *Trans. Fifth International Congress of Soil Science*, Leopoldville, Belgian Congo, 1954; Vol. 4, 195–198.
13. Doubleday, G.P. *Proceedings of English Soils Discussion Group No. 6*, 1969; 21–25.
14. Petersen, L.; Rasmussen, K.; Tovborg Jansen, A. *Soil Problems and Tree Growth after Lignite Strip-Mining*; Yearbook 1967; Royal Veterinarian and Agricultural College: Copenhagen, 1967.
15. Thomson, J.G. Occurrence of acid sulphate soil in Rhodesia. *Rhod. J. Agr. Res.* **1972**, *10*, 153–157.
16. Buurman, P.; Van Breemen, N.; Jongmans, A.G. *A Fossil Acid Sulphate Soil in Ice-Pushed Tertiary Deposits Near Uelsen, Germany*; *Proc. Int. Symp. Acid Sulph. Soils*, Wageningen. Dost, H., Ed.; 1973; 53–75.
17. Poelman, J.N.B. *Soil Material Rich in Pyrite in Non-Coastal Areas*; Dost, H., Ed.; *Proceedings of the International Symposium*, Wageningen, Vol. 18, 1973; 197–207.
18. FAO. *FAO–UNESCO Soil Map of the World, 1:5,000,000*; 10 Vols.; UNESCO: Paris, 1974.
19. FAO/UNESCO/ISRIC. *FAO–UNESCO Soil Map of the World (1:5,000,000). Revised Legend*; FAO World Soil Resources Report 60; Food and Agriculture Organisation: Rome, 1990.
20. FAO. *Agro-Ecological Zones Project*; Food and Agriculture Organisation: Rome, 1976.
21. Dent, D.L. Acid sulphate soils: Morphology and prediction. *J. Soil Sci.* **1980**, *31* (1), 87–99.
22. Pons, L.J.; Van Breemen, N. *Factors Influencing the Formation of Potential Acidity in Tidal Swamps*; In *Proceedings of the Second International Symposium on Acid Sulfate Soils*, ILRI Publication No. 31; Dost, H., Van Breemen, N., Eds.; International Institute for Land Reclamation and Improvement: Wageningen, 1982; 37–51.
23. Pons, L.J.; Van Breemen, N.; Driessen, P. Coastal sedimentary environments influencing the development of potential acidity in tidal swamps. In *Acid Sulfate Weathering*; Soil Science Society of America: Madison, 1982; 1–18.
24. Volkoff, B.; Willaime, P. *Carte Pédologique de Reconnaissance de la République Populaire du Bénin à 1:200,000. Feuille de Porte-Novo*; Notice Explicative 66; Institut. Français de Recherche Pour le Développement en Coopération (ORSTOM): Paris, 1976.
25. Martin, D.; Ségalen, P. *Carte Pédologique du Cameroun Oriental au 1:1,000,000*; Notice Explicative 51, 53, 62; Français de Recherche Pour le Développement en Coopération (ORSTOM): Paris, 1966.
26. Roose, E.; Cheroux, M. *Les Sols du Bassin Sédimentaire de Côte d'Ivoire. Série Pédologie*; Mémoires ORSTOM 63; Cahiers ORSTOM, Vol. 4.2; Institut. Français de Recherche Pour le Développement en Coopération (ORSTOM): Paris, 1966.
27. Giglioli, M.E.C.; Thornton, I. Mangrove swamps in the Gambia. *J. Appl. Ecol.* **1965**, *2*, 81–103.
28. Michel, P. *Les Bassins des Fleuves Sénégal et Gambie. Étude Géomorphologique*; Mémoires ORSTOM 63; Institut. Français de Recherche Pour le Développement en Coopération (ORSTOM): Paris, 1973; 752.
29. Marius, C. *Acid Sulphate Soils of the Mangrove Areas of Senegal and the Gambia*, In *Proceedings of the Bangkok symposium on acid sulphate soils: Second International Symposium on Acid Sulphate Soils*, Bangkok, Thailand, Jan 18–24; Dost, H., Van Breemen, N., Eds.; 1982; 103–136.
30. Marius, C. *Mangroves de Sénégal et de la Gambie. Ecologie, Pédologie, Géochimie et Mise en Valeur et Aménagement*; Collection Travaux et Documents ORS-TOM no 193; Institut. Français de Recherche Pour le Développement en Coopération (ORSTOM): Paris, 1985.

31. Thomas, P.; Varley, J.A. *Soil Survey of Tidal Sulphidic Soils in the Tropics. A Case Study*; Dost, H., Van Breemen, N., Eds.; Proceedings of Bangkok Symposium on Acid Sulphate Soils ISSC, Thailand, 1982; 52–72.
32. FAO/HEC. *Etude d'un Programme d'aménagement Hydro-agricole des Terres Rizicultivables de la Basse-Guinée*. Rapport Final, Pédologie; Food and Agriculture Organisation: Rome; Ministère de Agriculture de Guinée: Conakry, 1969; Vol. 2.
33. Sow, M. *Programme Mangroves*; Min. de la Pêche et de l'Élevage. République de Guinée. Conakry, Guinée, et Instit. Français de Recherche Pour le Développement en Coopération (ORSTOM): Paris, 1998; 17 pp.
34. da Silva Texeira, A.J. *Os Solos da Guine Portuguesa. (Soil Maps at Scale 1: 500.000; in Portuguese)*; Junta de Investigações do Ultramar: Luanda, 1962.
35. Sombroek, W.G.; Braun, H.M.H.; van der Pouw, B.J.A. *Exploratory Soil Map and Agro-Climatic Zone Map of Kenya (Scale 1:1,000,000)*; Exploratory Soil Survey Report No. E1; Kenya Soil Survey: Nairobi, 1982.
36. Reed, W.E. *Reconnaissance Soil Survey of Liberia*; USDA Agricultural Information Bulletin 66; United States Department of Agriculture: Washington, 1951.
37. SECA/CML. *Mangroves of Africa and Madagascar. Conservation and Reclamation*. Société d'éco-aménagement/ Centre for Environmental Studies; University of Leiden: Leiden, 1987.
38. Buckle, C. *Landforms in Africa. An Introduction to Geomorphology*; Longman Group Ltd.: Harlow, 1982; 249 pp.
39. Sylla, M.; Andriesse, W. An agro-ecological characterization of mangrove ecosystems in West Africa, with special emphasis on rice cultivation. In *Soil Salinity and Acidity: Spatial Variability and Effects on Rice Production in West Africa's Mangrove Zone*; Sylla, M., Ed.; Ph.D. Thesis; Wageningen Agricultural University: Wageningen, 1994; 19–50.
40. Birchall, C.J.; Bleeker, P.; Cusani-Visconti, C. *Land in Sierra Leone: A Reconnaissance Survey and Evaluation for Agriculture*; Technical Report 1; FAO/UNDP: Free-town, 1979.
41. FAO/UNDP. *Bangladesh Agricultural Development Possibilities*. Soil Survey Project AGL: SF/PAK 6, Technical Report No. 2; Food and Agriculture Organisation; Rome; Ministry of Agriculture: Bangladesh, 1971.
42. Bloomfield, C.; Coulter, J.K. Genesis and management of acid sulphate soils. *Adv. Agron.* **1973**, *25*, 265–326.
43. Goung, Y.; Win, K.; Tin, W. Rice soils of birma. In *Soils and Rice*; IRRI, Ed; International Rice Research Institute: Los Baños, 1977; 57–71.
44. Van Breemen, N.; Pons, L.J. Acid sulphate soils and rice. In *Soils and Rice*; IRRI, Ed; International Rice Research Institute: Los Baños, 1977; 739–761.
45. Wu, W.J. Environmental impacts of acid sulphate soils and preventive countermeasures. *Trop. Subtrop. Soil Sci.* **1998**, *7* (4), 314–318.
46. Yadav, J.S.P. Saline, alkaline and acid sulphate soils in India and their management. *Fert. News* **1976**, *76*, 15–23.
47. Murthy, R.S. *Acid Sulphate Soils of India*; FAO World Soil Resources Report No. 41; Food and Agriculture Organisation: Rome, 1971; 24–29.
48. Goswami, N. Distribution and characteristics of problem soils in India and their management for crop production. *Trop. Agr. Res. Ser.* **1982**, *15*, 25–40.
49. Driessen, P.M.; Soepraptohardjo, M. *Soils for Agricultural Expansion in Indonesia*; Bulletin 1; Soil Research Institute: Bogor, 1977; 63 pp.
50. NEDECO/Euroconsult/BIEC. *Nation-Wide Study of Coastal and Near Coastal Swamp Land in Sumatra, Kalimantan and Irian Jaya, Indonesia. Tidal Swampland Development Project (P3S)*; Directorate-General of Water Resources Development, Min. of Public Works: Jakarta, 1984.
51. Murakami, H. Characteristics and improvement of acid sulphate soils. *Jpn. J. Sci. Soil Manure* **1967–1968**, *38*, 112–120.
52. Kobayashi, T. Studies on the low productive soils in reclaimed lands. The soils in the Hane-Ko Polder, Shimane prefecture. *Jpn. J. Sci. Soil Manure* **1952**, *5*, 293–296.
53. Paramanantan, S. *Reconnaissance Soil Survey of Malacca*; Malayan Soil Survey Report No. 3; Dept. of Agriculture: Kuala Lumpur, 1967.
54. Andriesse, J.P. *The Soils of West Sarawak*; Dept. of Agriculture: Sarawak, 1972.
55. Kanapathy, K. *Acid Sulphate Soils*; Soils and Analytical Services Bulletin 6; Kuala Lumpur, 1976.
56. Zahari, A.B.; Wahab, N.; Chang, S.; Rahim, M. Distribution, characteristics and utilisation of problem soils in Malaysia. *Trop. Agr. Res. Ser.* **1982**, *15*, 41–61.
57. Brinkman, R.; Singh, V.P. *Rapid Reclamation of Brackish-Water Fishponds in Acid Sulphate Soils*, In Proceedings of the Bangkok symposium on acid sulphate soils: Second International Symposium on Acid Sulphate Soils, Bangkok, Thailand, Jan 18–24; Dost, H., Van Breemen, N., Eds., 1982; 318–330.
58. Briones, A.A. The nature, distribution and management of some problem soils in the Philippines. *Trop. Agr. Res. Ser.* **1982**, *15*.
59. Pons, L.J.; van der Kevie, W. *Acid Sulphate Soils in Thailand; Morphology, Genesis and Agricultural Potential*; Soil Survey Report No. 81; Soil Survey Division, Dept. of Agriculture: Bangkok, 1969; 1–65.
60. van der Kevie, W. Morphology, genesis, occurrence and agricultural potential of acid sulphate soils in central Thailand. *Thai J. Agr. Sci.* **1972**, *5*, 165–182.
61. Dent, F.J. *Reconnaissance Soil Survey of Peninsular Thailand*. Soil Survey Report No. 94; Dept. of Land Development: Bangkok, 1972; 12 pp.
62. Panichapong, S. Distribution, characteristics and utilisation of problem soils in Thailand. *Trop. Agric. Res. Ser.* **1982**, *15*, 83–96.
63. Moormann, F.R. *The Soils of the Republic of Vietnam*; Ministry of Agriculture: Saigon, 1961; 66 pp.
64. Tram, H.V.; Pham, N.L. Problem soils in the Mekong Delta, Vietnam: A general description and implication for rice cultivation. In International Rice Research Conference; IRRI: Los Baños, 1975.
65. Van Mensvoort, M.E.F.; Xuan, V.-T. *VH-10 Project Evaluation Report; Management of Acid Sulphate Soils in Vietnam*; Can Tho University: Can Tho, 1984.

66. Ton, T.C.; Nguyen, C.P.; Ngyuen, V.N.; Tran, A.P.; Pham, Q.K. *Soils of the Mekong Delta, Vietnam. Explanatory Note to the Soil Map 1:250,000 (in Vietnamese)*; National Inst. of Agric. Planning and Projection, Ministry of Agriculture: Hanoi, 1991.
67. White, I.; Melville, M.D.; Wilson, B.P.; Sammut, J. Reducing acid discharges from coastal wetlands in eastern Australia. *Wetl. Ecol. Manag.* **1997**, *5*, 55–72.
68. Lévêque, A. *Mémoire Explicatif de la Carte des sols de Terres Basses de Guyane Française*; Institut Français de Recherche Pour le Développement en Coopération (ORS-TOM): Paris, 1962.
69. Turenne, J.F. *Carte Pédologique de Guyana. Feuille Mana Saint Laurent à 1:50,000*; Institut Français de Recherche Pour le Développement en Coopération (ORS-TOM): Paris, 1973.
70. Brinkman, R.; Pons, L.J. *A Pedo-Geomorphological Classification and Map of the Holocene Sediments in the Coastal Plain of the Three Guyana's*; The Netherlands Soil Survey Institute (STIBOKA): Wageningen, 1968.
71. Steur, G.G.L.; de Vries, F.; van Wallenburg, C. *Bodemkaart van Nederland 1:250.000. (Soil Map of the Netherlands 1:250.000; in Dutch)*; Netherlands Soil Survey Institute (Stiboka): Wageningen, 1985.
72. Kivinen, E. Sulphate soils and their management in Finland. In Fourth International Congress of Soil Science, Amsterdam, 1950; Vol. II, 259–261.
73. Palko, J.; Yli-Halla, M. Assessment and management of acidity release upon drainage of acid sulphate soils in Finland. In *Selected Papers of the Ho Chi Minh City Symposium on Acid Sulphate Soils*. International Institute for Land Reclamation and Improvement Publication, Vol. 53, 1993; 411–418.
74. Öborn, I. *Morphology, Chemistry, Mineralogy and Fertility of Acid Sulphate Soils in Sweden*; Department of Soil Science Reports and Dissertations No. 18; Swedish University of Agricultural Science: Uppsala, 1994; 235 pp.

Acid Sulfate Soils: Formation

Martin C. Rabenhorst

Department of Natural Resource Sciences, University of Maryland, College Park, Maryland, U.S.A.

Delvin S. Fanning

Department of Environmental Science and Technology, University of Maryland, College Park, Maryland, U.S.A.

Steven N. Burch

University of Maryland, College Park, Maryland, U.S.A.

Abstract

Soils containing sulfide minerals that have not yet been oxidized through acid sulfate weathering are referred to as potential acid sulfate soils (SS). The processes involved in the formation and accumulation of sulfide minerals in soils leading to the formation of potential acid SS will be discussed first. Subsequently, processes related to the oxidation of sulfides in the formation of active acid SS will be examined.

INTRODUCTION

The extent of acid sulfate soils (SS) worldwide has been estimated to be approximately 12–15 Mha. (For details, see the entry *Acid Sulfate Soils: Distribution and Extent*, p. 20–25.)

POTENTIAL ACID (SS) AND SULFIDE MINERAL FORMATION—SULFIDIZATION

Biogeochemistry of Sulfide Mineral Formation

Several factors are required for sulfate reduction. These include a source of sulfate, a source of oxidizable carbon, reducing conditions, and the presence of sulfate-reducing bacteria.^[1] Any of these components could theoretically limit sulfate reduction. In saturated soil or sedimentary environments where the required factors are present, heterotrophic microbes utilize sulfate as an electron acceptor that becomes reduced to sulfide according to Eq. 1.



Sulfate

Provided that the other required factors are met, the quantity of sulfate may limit the rate of sulfate reduction. Goldhaber and Kaplan^[2] reported sulfate reduction to be independent of concentration when sulfate levels are above 10 mM (320 mg/l). Work by Haering,^[3] in Chesapeake Bay, indicated that sulfate levels may begin to limit sulfur (S)

accumulation in marsh soils when levels drop below 1 mM (32 mg/l). Some degree of sulfate reduction will continue as long as sulfate is present at minimal levels ($> 5\text{--}20 \mu\text{M}$, 0.16–0.6 mg/l).^[4] Because sulfate-reducing bacteria are better able to compete for electron-donating substrates than are methane-generating bacteria, methanogenesis is of minimal significance so long as sulfate levels are above 0.03–0.4 mM.^[5] Therefore, sulfate reduction dominates in brackish systems. In freshwater, sulfate reduction may become overshadowed by methanogenesis as sulfate is depleted.

Oxidizable organic carbon

The oxidation of organic matter provides the energy microorganisms need to facilitate sulfate reduction. Plant materials rich in labile components are more easily decomposed than humified soil organic matter or peat. In sediments low in organic matter, sulfate reduction may be limited by the paucity of oxidizable carbon. This can be demonstrated in thin sections from mineral horizons in tidal marsh soils where iron sulfide (FeS) minerals have accumulated in pores occupied by decaying plant roots (Fig. 1). The intimate association of pyrite (FeS₂) with the decomposing organic minerals, and its near absence from the surrounding soil matrix, suggests that organic matter is limiting the formation of sulfide.^[6]

Reducing/saturated conditions

Because diffusion of gases through saturated soils and sediments is very slow, oxygen becomes depleted

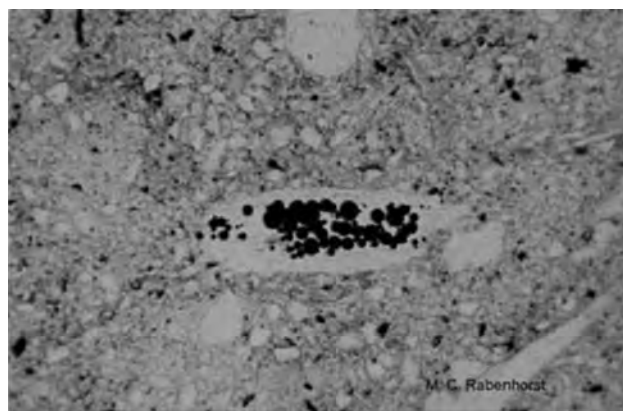


Fig. 1 Micrograph of a thin section from the mineral (Cg) horizon of a tidal marsh soil illustrating accumulation of FeS_2 framboids in the channel occupied by decaying plant roots. Plane polarized light; frame length 1.2 mm.

under saturated conditions and microbes that utilize other electron acceptors become active. Nitrate, Mn(IV) , and Fe(III) are so utilized as the environment becomes progressively reduced (Fig. 2). If the conditions permit the entry of oxygen, then redox potentials may never become sufficiently low to foster sulfate reduction. More typically, diffusion of oxygen into a saturated soil or sediment is sufficiently slow, and if other necessary factors are present, sulfate reduction will occur. Fig. 2 illustrates that pH, as well as E_h , must be specified in assessing S phase equilibria; e.g., as the pH increases from 5 to 7, the minimum E_h at which sulfate reduction

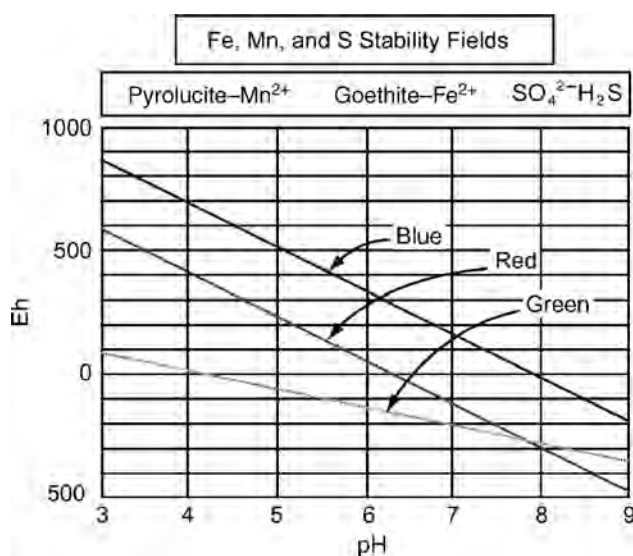


Fig. 2 pe-pH diagram illustrating the location of stability fields for redox-sensitive components. The sulfate-sulfide lines (green) is based on a (SO_4^{2-}) concentration of 10 mM and a pH_2S of 0.001 atm. (Blue line separates the pyrolucite- Mn^{2+} stability fields; red line separates the goethite- Fe^{2+} stability fields.)

is expected decreases from approximately -50 to -200 (based on a SO_4^{2-} concentration of 10 mM and a pH_2S of 0.0001 atm).

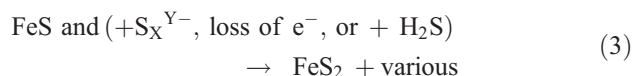
Sulfate-reducing bacteria

Some 15 genera of bacteria have been recognized as sulfate reducers including *Desulfovibrio*, *Desulfotomaculum*, and *Desulfobacter*.^[7,8] These organisms thrive under strongly reducing conditions, but many are able to persist in aerobic conditions for significant periods of time. Thus, if the other factors necessary for sulfate reduction are present, sulfate-reducing bacteria will also become active.

As with most heterotrophic bacteria, rates of sulfate reduction are temperature dependent. Optimum temperature for most sulfate reducers is $30\text{--}40^\circ\text{C}$,^[7] and the rate of sulfate reduction generally increases with temperature across this range. Some groups of sulfate reducers are thermophilic and can function at temperature up to 85°C . Thus, in tropical coastal wetlands, sulfate reduction occurs all year round. In higher latitudes, where soil and sediment temperatures may approach biological zero, rates may become very slow during winter.

Reactive Fe

Once formed, sulfide is available to form a variety of minerals provided that there is adequate reactive iron (Fe) present. Most of the Fe enters coastal environments as Fe oxides sorbed to the surface of clay and silt particles. When Fe oxides in the sediments and marsh soils become reduced to Fe(II) , they can form FeS minerals. While monosulfide species may form first (Eq. 2), and minerals such as greigite (Fe_3S_4) may persist in recent sediments, disulfide forms such as FeS_2 are energetically more stable and will form at the expense of the monosulfides.



Mechanisms for FeS_2 formation may follow several possible pathways including 1) reaction of monosulfide with polysulfide; 2) partial oxidation of monosulfide; and 3) reaction of monosulfides with H_2S ^[9] (Eq. 3). Sulfide itself has the ability to reduce Fe(III) to Fe(II) on the surface of Fe oxides.^[10] FeS_2 can occur either as small ($<2\text{ }\mu\text{m}$) individual crystals or as spherical clusters of crystals called framboids. In low organic mineral sediments, reactive Fe is usually present in excess, resulting in a low degree of pyritization.^[11] However, in organic-rich soils, Fe may limit the accumulation of sulfide minerals, and the degree of pyritization is generally high. This has been demonstrated experimentally in salt marsh Histosols.^[12]

Environments of Sulfide Formation and Accumulation

It is clear that in environments that provide a source of oxidizable carbon and sulfate and that are sufficiently saturated to enhance reducing conditions, sulfate-reducing bacteria will generate sulfide. If reactive Fe is present, then solid-phase ferrous minerals will accumulate. This process of *sulfidization*^[13] is shown schematically in Fig. 3. The obvious settings for these processes are coastal marine and brackish environments, where sulfate is abundant. Under permanently submersed conditions, detrital carbon is added by flora and fauna to the sediment. In shallow water settings (<3 m) where various pedogenic processes are at work, these accumulated sediments have been recognized as subaqueous soils^[14] and are classified to reflect the sulfide components.

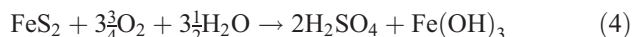
The soils of coastal marshes (in temperate environments) and mangroves (in tropical settings) also are ideal for sulfide formation and accumulation. The high primary productivity of these ecosystems (up to $3 \text{ kg m}^{-2} \text{ yr}^{-1}$ in marshes and up to $5 \text{ kg m}^{-2} \text{ yr}^{-1}$ in mangroves)^[15] makes these an exceptionally good environment for sulfate reduction. Such soils may contain up to 20–30 g/kg of S in FeS_2 , and estimates of S in FeS_2 accumulation rates in estuarine marshes are as high as $7 \text{ g m}^{-2} \text{ yr}^{-1}$.^[16]

Sulfate reduction can occur in other settings, so long as a source of sulfate is available. While generally small, atmospheric deposition of sulfate may be enough to induce sulfate reduction in the sediments of some interior freshwater lakes. Sulfate reduction has also been documented in prairie potholes where sulfate has apparently been contributed by the weathering of S-bearing shales.^[17]

FORMATION OF ACTIVE ACID SS—SULFURICIZATION

Chemistry of Sulfide Oxidation

Sulfides begin to oxidize once they are exposed to more oxidizing conditions. This occurs most often as a result of such human activities as drainage or dredging of sulfide-bearing soils or sediments, or the mining of sulfide-bearing coal, but may also occur due to tectonic uplift or oceanic regression. Under humid or moist aerobic conditions, sedimentary sulfide minerals can oxidize chemically,^[18] but this is a slow process, probably due to particular rate-limiting reactions. Various microorganisms are adapted to oxidize sulfides either directly through S transformations or by facilitating (catalyzing) such rate-limiting reactions as the oxidation of Fe(II) to Fe(III) .^[19] While there are many possible intermediate reactions in the oxidation of FeS_2 , the overall reaction is summarized in Eq. 4. One mole of FeS_2 eventually yields two moles of sulfuric acid and a mole of Fe hydroxide.



The oxidation of FeS_2 proceeds along two fronts (Fig. 4). First, the S is oxidized (through intermediates) to sulfate yielding sulfuric acid and the remaining Fe(II) . The generated Fe(II) sulfate salts are very soluble and potentially mobile. Second, Fe(II) is oxidized to Fe(III) , which when hydrolyzed produces additional acid. At high pH, the oxidation of sulfide is accomplished with oxygen, but under low pH conditions sulfide is oxidized by Fe(III) . Microorganisms such as *Thiobacillus ferrooxidans* facilitate this

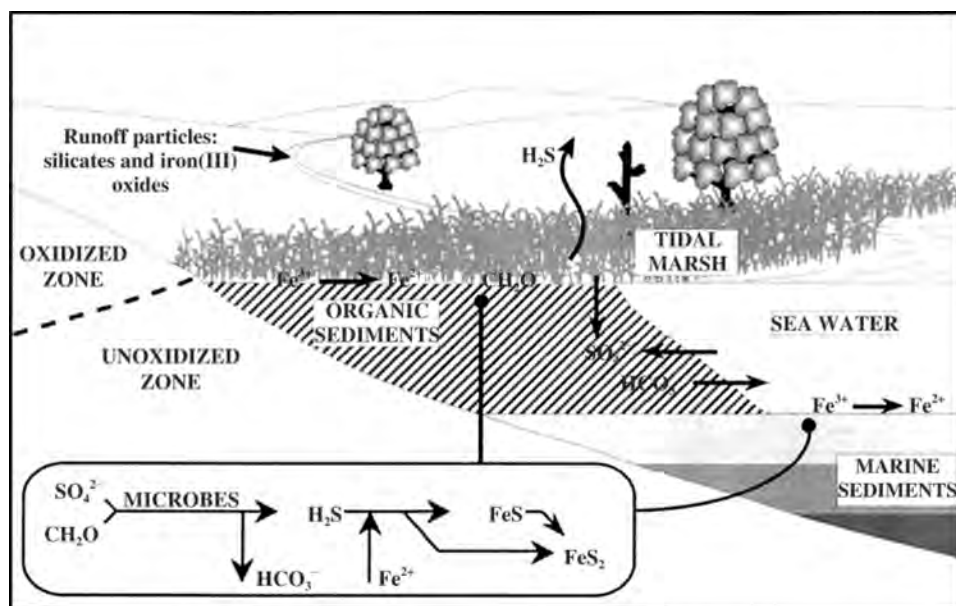


Fig. 3 Schematic diagram illustrating the generalized process of sulfidization that leads to the formation of FeS minerals and potential acid SS.

Source: From Ritsema, van Mensvoort, et al.^[23]

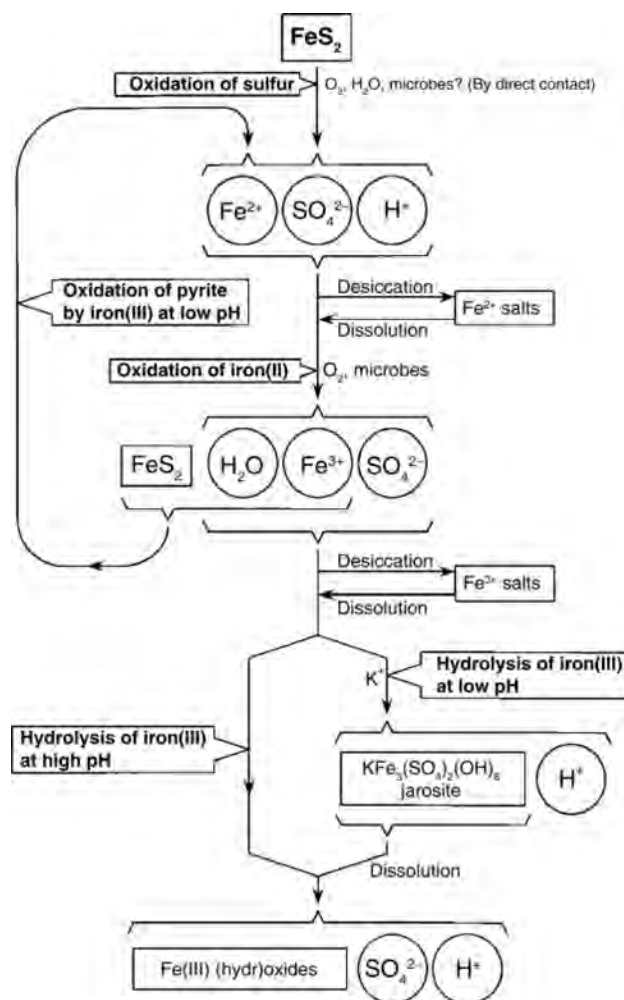


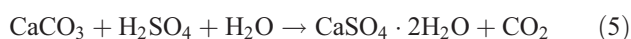
Fig. 4 Schematic diagram illustrating the generalized process of sulfurification that involves the oxidation of FeS minerals and the production of acidity and the formation of new sulfate and other minerals.

Source: From Fanning.^[20]

reaction by oxidizing Fe(II) to Fe(III). For more details, refer to the entry *Acid Mine Drainage*, p. 6–10.

Other Aspects of Sulfurification and Properties of Acid SS

Sulfurification is the overall process by which sulfide-bearing minerals are oxidized, minerals are weathered by the sulfuric acid produced, and new mineral phases are formed from the dissolution products.^[13] (See also the entry *Acid Mine Drainage*, p. 6–10.) When CaCO_3 minerals are present, the sulfuric acid reacts with them to form the mineral gypsum, according to Eq. 5.



As long as sufficient CaCO_3 is present, the pH is prevented from becoming very low and the soil does not become acid.

When insufficient acid-neutralizing minerals are present, the oxidation of FeS_2 in soils will lower the pH. The pH of active SS commonly drops to below four and in extreme can go below two. As Fe is oxidized and hydrolyzed, various Fe minerals form in the soil including ferrihydrite, schwertmannite, and goethite. If the soil pH falls below four while maintaining an oxidizing environment ($E_h > 400$ mV), then the mineral jarosite ($\text{KFe}_3(\text{SO}_4)_2(\text{OH})_6$) can form.^[21] Because ($\text{KFe}_3(\text{SO}_4)_2(\text{OH})_6$) forms under conditions of high E_h and very low pH, which can only develop from the generation of sulfuric acid, it is considered a diagnostic mineral for acid SS (See also the entry *Sulfate and Sulfide Minerals*, p. 2238.).

($\text{KFe}_3(\text{SO}_4)_2(\text{OH})_6$) has been reported in soils, which are not extremely acid and which may even contain carbonates.^[22] These are interpreted to be “postactive” acid SS, meaning that earlier in their pedogenic history, they had undergone acid sulfate weathering. Subsequently, the soil pH has risen due to weathering of silicate minerals or addition of eolian carbonates. Because the redox potential has remained strongly oxidized, the ($\text{KFe}_3(\text{SO}_4)_2(\text{OH})_6$) has persisted as a metastable species. A review of acid SS including a discussion of the modeling of associated processes has been published.^[23]

REFERENCES

1. Rabenhorst, M.C.; James, B.R.; Magness, M.C.; Shaw, J.N. Iron removal from acid mine drainage in wetlands by optimizing sulfate reduction. In *The Challenge of Integrating Diverse Perspectives in Reclamation*; Proc. Am. Soc. Surf. Mining Reclam; Spokane: Washington, 1993; 678–684.
2. Goldhaber, M.B.; Kaplan, I.R. Controls and consequences of sulfate reduction rates in recent marine sediments. *Soil Sci.* **1975**, *119*, 42–54.
3. Haering, K.C. *Sulfur Distribution and Partitionment in Chesapeake Bay Tidal Marsh Soils*; M.S. Thesis. University of Maryland, College Park, 1986; 172.
4. Ingvorsen, K.; Zehnder, A.J.B.; Jorgensen, B.B. Kinetic of sulfate and acetate uptake by *Desulfobacter postgatei*. *Appl. Environ. Microb.* **1984**, *47*, 403–408.
5. Smith, D.W. Ecological actions of sulfate-reducing bacteria. In *The Sulfate-Reducing Bacteria: Contemporary Perspectives*; Odom, J.M., Singleton, R., Jr. Eds.; Springer: New York, 1993; 161–188.
6. Rabenhorst, M.C.; Haering, K.C. Soil micromorphology of a Chesapeake Bay Tidal Marsh: Implications for sulfur accumulation. *Soil Sci.* **1989**, *147*, 339–347.
7. Fauque, G.D. Ecology of sulfate-reducing bacteria. In *Sulfate-Reducing Bacteria*; Barton, L.L., Ed.; Plenum Press: New York, 1995; 217–241.
8. Postgate, J.R. *The Sulphate-reducing Bacteria*, 2nd Ed.; Cambridge University Press: London, 1984; 107–122.
9. Rickard, D.; Schoonen, M.A.A.; Luther, G.W., III. Chemistry of iron sulfides in sedimentary environments. In *Geochemical Transformations of Sedimentary Sulfur*; Vairavamurthy, M.A., Schoonen, M.A.A., Eds.; American Chemical Society: Washington, 1995; 168–193.

10. Ghiorse, W.C. Microbial reduction of manganese and iron. In *Biology and Anaerobic Microorganisms*; Zehnder, A.J. B., Ed.; John Wiley and Sons: New York, 1988; 305–331.
11. Griffin, T.M.; Rabenhorst, M.C. Processes and rates of pedogenesis in some Maryland Tidal Marsh soils. *Soil Sci. Soc. Am. J.* **1989**, *53*, 862–870.
12. Rabenhorst, M.C. Micromorphology of induced iron sulfide formation in a Chesapeake Bay (USA) Tidal Marsh. In *Micromorphology: A Basic and Applied Science*; Douglas, L.A., Ed.; Elsevier: Amsterdam, 1990; 303–310.
13. Fanning, D.S.; Fanning, M.C.B. *Soil Morphology, Genesis, and Classification*; John Wiley and Sons: New York, 1989; 395 pp.
14. Demas, G.P.; Rabenhorst, M.C. Subaqueous soils: Pedogenesis in a submersed environment. *Soil Sci. Am. J.* **1999**, *63*, 1250–1257.
15. Mitsch, W.J.; Gosselink, J.G. *Wetlands*, 2nd Ed.; Van Nostrand Reinhold: New York, 1993; 722 pp.
16. Hussein, A.H.; Rabenhorst, M.C. Modeling of sulfur sequestration in coastal marsh soils. *Soil. Sci. Soc. Am. J.* **1999**, *63*, 1954–1963.
17. Arndt, J.L.; Richardson, J.L. Geochemistry of hydric soil salinity in a recharge-throughflow-discharge Prairie-Pothole wetland system. *Soil. Sci. Soc. Am. J.* **1989**, *53*, 848–855.
18. Borek, S.L. Effect of humidity of pyrite oxidation. In *Environmental Geochemistry of Sulfide Oxidation*; Alpers, C.N., Blowes, D.W., Eds.; American Chemical Society: Washington, 1994; 31–44.
19. Nordstrom, D.K. Aqueous pyrite oxidation and the consequent formation of secondary iron minerals. In *Acid Sulfate Weathering*; Kittrick, J.A., Fanning, D.S., Hossner, L.R., Eds.; Soil Sci. Soc. Am. Spec. Pub. No. 10; Science Society of America: Madison, 1982; Vol. 37, 56 pp.
20. Fanning, D.S.; Rabenhorst, M.C.; Burch, S.N.; Islam, K.R.; Tangren, S.A. Sulfides and sulfates. In *Soil Mineralogy with Environmental Applications*; Dixon, J.B., Schulze, D.G., Eds.; Soil Science Society of America Book Series #7: Madison, 2002; 229–260.
21. van Breeman, N. Genesis, Morphology, and classification of acid sulfate soils in coastal plains. In *Acid Sulfate Weathering*; Kittrick, J.A., Fanning, D.S., Hossner, L.R., Eds.; Soil Sci. Soc. Am. Spec. Pub. No. 10; Science Society of America: Madison, 1982; 95–108.
22. Carson, C.D.; Fanning, D.S.; Dixon, J.B. Alfisols and ultisols with acid sulfate weathering features in Texas. In *Acid Sulfate Weathering*; Kittrick, J.A., Fanning, D.S., Hossner, L.R., Eds.; Soil Sci. Soc. Am. Spec. Pub. No. 10; Soil Sci. Soc. Am.: Madison, 1982; 127–146.
23. Ritsema, C.J.; van Mensvoort, M.E.F.; Dent, D.L.; Tan, Y.; van den Bosch, H.; van Wijk, A.L.M. Acid sulfate soils. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 1999; G121–G154.

Acid Sulfate Soils: Management

Michael D. Melville

University of New South Wales, Sydney, New South Wales, Australia

Ian White

Australian National University, Canberra, Australian Capital Territory, Australia

Robert Quirk

Private Consultant and Sugar Cane Farmer, Duranbah, New South Wales, Australia

Abstract

The issue of acid sulfate soils (ASSs) is of global extent, and their management must encompass all of those landscapes containing sulfurous material, both soft sediments and hard rocks, located on coasts and inland of land masses because there has been a marked shift in understanding the links that various disciplines share about the geochemical sulfur cycle and aspects of its management. Thus, soil scientists are joining mining geologists and engineers, and others in sharing experience and potential techniques for better managing these sulfurous landscapes. Nevertheless, the emergence of the “pyrite-heave” problem with construction, particularly in Canada and Ireland, shows that there must be greater linkages; education and communication between such disciplines; and politicians, regulators, and all stakeholders must be involved for better ASS management. There are three fundamental questions underlying the use of ASS sites and landscapes. Firstly, how best to use the existing environment productively while minimizing sulfide mineral oxidation. Secondly, how to sustainably neutralize any existing or newly created acidity within a particular land use. Thirdly, how to eliminate or minimize downstream impacts from ASS drainage waters. Such an holistic approach might best be achieved by an acceptance by all stakeholders of a sense of environmental stewardship for environmental management.

INTRODUCTION

Acid sulfate soils (ASSs) is the name given to all those soils or unconsolidated sediments that contain reduced sulfur minerals and compounds that have been or can be oxidized and thereby produce acidity.^[1] Thus, we are interested in the existence and behavior of the iron disulfide mineral, pyrite.^[2] In the previous two editions of this encyclopedia,^[3] we discussed some of the problems and relationships between ASSs and acid rock drainage (ARD). In much of the initial research about ASS, the emphasis was on coastal locations in Holocene-age materials (<10,000 yr). However, it is clear that much older, inland materials must also be considered.^[4] Subsequently, we expanded on this widened theme and described management approaches that were appropriate to minimize the problems associated with these sulfurous materials. It is appropriate to also include some of the problems associated with what might be best termed “construction and development” involving sulfurous materials because we have become aware of this very large and important issue and options that have been or are being adopted for their best management. It seems that as the global population increases, there is an increasing need for production of food and development of infrastructure to

accommodate the larger population, and this development must be completed at a larger scale, more rapidly, and relatively more cheaply. The management options available for particular sites depend upon the specific problem, its scale, and the particular economics of the situation. Nevertheless, there are three fundamental questions underlying the development of a particular ASS site or landscape. Firstly, how best to productively use the existing environment while minimizing any additional sulfide mineral oxidation. Secondly, how to effectively neutralize any existing or new acidity created during the development. Thirdly, how to eliminate or minimize downstream environmental impacts from ASS drainage waters, such as acidification, deoxygenation, or release of metals and nutrients. Best management must therefore address these questions.

EMERGING PROBLEMS FOR MANAGEMENT

The agronomic problems of ASS have been recognized for centuries in Europe and management techniques developed, particularly following the large-scale polder developments in The Netherlands. The experiences from there were transferred to developments in more tropical

environments, and the initial international symposia, after 1973, emphasizing these interests were organized by International Institute for Land Reclamation and Improvement (ILRI) from Wageningen. Australia had very limited scientific interest in ASS until the 1987 flood on the Tweed River where the entire 23 km of the estuary became clarified and sterilized of fish and benthic organisms. The cause and source of this problem were identified (as from the floodplain ASS) by the astute observations of the local government entomologist, Clive Easton,^[5] who had for many years been manipulating floodplain agricultural drains so as to better control problem mosquito populations. We consider Easton's paper seminal to the subsequent development of Australian scientific, environmental, governmental, and public interest in ASS. This interest has developed in Australia so that ASSs have become an essential management requirement regulated by most levels of government across the nation, but this has been generally a cooperative learning process.^[6] This development has been greatly advanced by much research publication, personnel training and education, and National ASS Conferences in Australia. Beginning with the 4th International ASS Symposium organized by ILRI in Ho Chi Minh City, Vietnam, in 1992, the environmental and engineering aspects of ASS and ARD have become the leading interests of subsequent international ASS/ARD symposia and the ASS Working Group discussions at the International Soil Science Symposia.

Events of acidification and deoxygenation of ASS drainage and receiving waters are seen around the world that lead to major fish kills but also to the increased concentration of dissolved iron at micromolar concentrations, and the associated deleterious phytoplankton blooms have been linked to ASS discharges.^[7] Discharge of metals in agricultural ASS drainage systems of Finland (the largest area of ASS in Europe) greatly exceed that from industrial effluents, and much research and development has been undertaken to reduce this impact.^[8,9] Emissions from ASS of toxic and greenhouse gases have also been reported.^[10,11] Hydrological changes to ASS landscapes, such as the increase in evapotranspiration and lowering of the water table, can allow oxidation of sulfides in the sediment and the associated acidification of contained pore water. Such changes can be by either natural causes or by imposed drainage systems. Our major research site at McLeods Ck is an ASS Tweed River estuarine floodplain, almost entirely under sugarcane cultivation. Robert Quirk has developed and implemented a wide range of ASS management options so as to minimize acidity export problems but also to increase sugarcane productivity.^[12,13] We have found it difficult to distinguish between natural and artificial drainage impacts on sulfidic oxidation, but the latter provides the conduit by which acidity from whatever cause is transferred to receiving waters and therefore requires careful management.^[14] In some locations, such as in the Mekong Delta of Vietnam, increased abstraction of water flow in the

upstream nations along the Mekong River has led to increased ASS problems and tidal incursions into distributary channels and canals draining the Delta. Thus, floodgates on all distributary channels and canals have been installed to stop this incursion with an intended increase in rice production across the Delta. Unfortunately, many of the local farmers depend upon seasonal brackish water incursions to enable the better cash crop from shrimp production. This dichotomy of purpose was explained to us as “those imposing the floodgates view seawater as their enemy, these farmers see seawater as their friend.”

In southeastern Australia, the severe drought from 2005 to about 2009 markedly reduced delivery of discharge to, and therefore desiccation in, the outflow lakes and other wetlands along the lower Murray River. This caused oxidation of the sulfide minerals contained in the previously saturated sediments, and severe acidification and environmental problems ensued.^[15] Subsequent reflooding with freshwater did not immediately reverse these problems.^[16] While natural climatic variation was the main cause here, the imposition of flow restrictions and abstraction to enable irrigation in upstream reaches of the Murray Darling Basin (MDB) exacerbated the effect as too did the effect of using wetlands along the river for disposal of sulfate-containing irrigation tailwater drainage. Understanding and better managing this problem is not straightforward.^[17]

In our section of the first edition (2002) of this encyclopedia, we reported the large-scale and expensive use of geotextile vertical wicks to enable the consolidation of sulfidic clay gels in floodplain sections of the main highway linking Sydney and Brisbane on the east coast of Australia. It is pleasing to report that this technique, including acidity management provisions, is successful. However, there have been some limited problems with vegetation-stabilizing batter embankment where deep cuttings through bedrock hills encountered rock materials with very low sulfide content, which was below the natural geological weathering front of the landscape. Fortunately, the acid management systems for the adjacent floodplain sections of the highway were able to neutralize any acid export from the hill sections.

We have become aware of a major problem in building construction, where crushed sulfidic rock was used as fill. The sulfidic material oxidized and when it reacted with contained or added calcium carbonate, gypsum, and other related swelling minerals were formed.^[18] Major cracking of building walls and heave of concrete foundation slabs occurred, forcing occupants to vacate their homes and await repairs. In Ireland, there are more than 20,000 newly constructed houses affected. The Irish situation was only realized in about 2007 following the “Celtic Tiger” burst of economic activity that drove an unprecedented level of construction, mainly residential. Although there appear to have been more than 20,000 claims for damages from the problem, the final number may be much greater because it can take up to 20 years for the damage to become

evident.^[18] The average estimated repair is about 45,000 euros per house and it appears that only the Irish national government has the capacity to cover these costs. This pyrite oxidation swelling problem has also occurred in other parts of the world and particularly in Canada where, again, more than 10,000 houses were affected.^[19,20] It is disappointing that these problems occurred because there is a considerable history of such problems and published literature (e.g., see the works of Maher and Gray^[21] and Britpave^[22]). The pyrite content of quarry rock involved in the Canadian problems was about 1–2%, in the Irish situation, it was about 3–4% (see Fig. 10 in the study by Maher and Gray^[21]). These concentrations are comparable with those we have found in Holocene-age estuarine sediments that produce ASS. However, in the footnote of Table 1 of Britpave's study^[22] that lists the geological strata with a potential for high sulfate/sulfide, Quaternary deposits are said to be “unlikely to contain high sulfate/sulfide levels because conditions during deposition were not anaerobic.” Obviously, Britpave^[22] was not considering the very young materials (aerial and subaqueous) with which the ASS community are involved. Clearly, there is a great need for wider communication between the various disciplines manipulating geological material.

The European Commission of the European Union has developed and published criteria for the definition and management of “end-of-waste” that probably should have been involved with the management of material in this pyrite-heave problem in Ireland. In Australia, the National Coastal Acid Sulfate Soil Coordinating Group (“NatCASS”; that also considers inland ASS) is investigating the development of criteria for definition of sulfurous material and education, qualification, and certification of individuals to provide for the materials' best management. The United Kingdom's Contaminated Land: Application in Real Environments (see www.claire.co.uk) is being investigated initially as a model.

MANAGEMENT APPROACHES

The best management of ASS involves a range of activities that addresses both the issues of minimizing the creation of new acidity from sulfide mineral oxidation and managing the existing acidity in the landscape. Across the range of land uses on ASS landscapes in North Coast of New South Wales (NSW), the existing acidity in the sulfuric layer of the ASS profiles averages about 50 t of sulfuric acid/hectare.^[14] The potential acidity in the deeper sulfidic material of many meters is many times greater than this. Nevertheless, the annual discharge of acidity is <0.5 t/ha.^[23] The degree to which the existing acidity is natural or caused by imposed drainage or application of nitrogenous fertilizer is uncertain.^[14] However, the drainage systems, which markedly reduce the duration of floodplain inundation, provide the conduit for acidity export. Thus, better drainage

management is essential and the NSW sugar industry has adopted this as a mandatory code of practice for all sugarcane growers. Compliance to this code is externally audited annually and only the few instances of inadvertent omissions have been quickly rectified.

Best management of ASS requires knowledge of their occurrence and distribution; the depth from the surface of the sulfidic/sulfuric layer boundary; the existing acidity store; the hydrological behavior of the ASS profiles, landscape, and drainage system; the existing and probable climatic regimes; and, for estuarine ASS floodplains, the magnitude, tidal characteristics, and water quality of the receiving waters.^[24] Frequently, not all of these factors are considered.

EDUCATION AND ASSESSMENT

Although Australian soil scientists and others were slow to recognize the widespread occurrence of ASS, and their importance to better environmental management, this changed rapidly and widely following the 1987 Tweed River fish kill and Easton's seminal paper.^[5]

Mapping of ASS has proven an important tool for better management in Australia. The existence and characteristics of coastal ASS have been mapped by each State (see, e.g., in NSW at: <http://mapdata.environment.nsw.gov.au>, accessed June 2015), and across Australia, and this information is provided in the Atlas of Australian Acid Sulfate Soils that is accessible online through the Australian Soil Resource Information System (<http://www.aris.csiro.au>, accessed June 2015).

In Australia, the assessment and quantification of ASS risk are provided as a guideline in the ever-evolving Queensland Acid Sulfate Soil Technical Manual (version 3.8, accessed June 2015). However, NatCASS is developing a guideline document that will have acceptance across all Australian jurisdictions (personal communication, NatCASS Chair, Steve Appleyard). Such documents are intended to provide all of the information necessary to enable best management of ASS and that will be freely available online to everyone. Obviously, the rapid growth of online documentation of guidelines and reports should greatly facilitate better ASS management because there are increasing issues and problems being encountered by groups and individuals from a great many disparate disciplines and jurisdictions (e.g., see Tichy^[25] and Johnson and Hallberg^[26]).

Education and training of individuals involved in assessing and managing ASS have become important in Australia and several organizations, such as Southern Cross University, provide these as courses in various locations and times. Information on such courses is advertised in the free online biannual publication “ASSAY” (Issues 1–66, available at: <http://www.dpi.nsw.gov.au/aboutus/resources/periodicals/newsletters/assay/>). Free subscription

from email: simon.walsh@dpi.nsw.gov.au includes “subscribe ASSAY” in subject line. This periodical is intended to provide international coverage of articles and research but that depends upon subscribers communicating with Editor Walsh. It was pleasing to see in ASSAY No. 65 that a large number of participants at the 4th National ASS Conference in Perth, WA, in May 2014, described themselves as “consultants.” Also, many of the delegates had previously attended such conferences and/or had undertaken ASS training courses.

AVOIDANCE

A primary preventative consideration with ASS should be an avoidance strategy. Such strategies include the decision not to drain wetlands containing sulfidic material and to divert or relocate a proposed land use or development to another site. If the use of the site is unavoidable, then a management plan incorporating treatment so as to prevent sulfide mineral oxidation and export of any existing acidity is necessary.

Given that most soil scientists and others working with ASS consider that material containing more than 0.25% sulfide require an ASS management plan, it seems surprising and disappointing that the quarry operations in Canada and Ireland were permitted to provide crushed rock fill for construction with a pyrite content greatly exceeding 1%.^[21] Better education and communication about ASS are therefore needed. In assessing the need for avoidance or otherwise, land use planners will consider the relative land values and alternative costs as well as engineering criteria and political necessity. This is a complex assessment process.

OXIDATION PREVENTION

There are a number of possible options here.^[25,26] Separation of sulfidic material and its disposal by capping with material that is impermeable to oxygen and rainfall infiltration is used in the mining industry but sulfidic clays for capping material must be avoided. Decisions to not drain sulfidic wetlands are intended not only to avoid the creation of new acidity from sulfide oxidation but also to address the issue of retaining any existing acidity. Management techniques, such as reflooding to raise the landscape water table, are a means of protecting sulfide minerals from oxidation. However, this technique must also consider the hydrological changes that ensue, such as increasing the export of existing acidity during rainfall-induced discharges. While maintenance of high water tables has beneficial effects, in agricultural land uses with certain crops, this may adversely impact productivity and economic sustainability. As has been seen in the drought problem in the MDB, it is not necessarily easy to always accommodate natural climatic variability. In a rapidly

growing human population, the social impacts must be considered.

ACIDITY CONTAINMENT AND NEUTRALIZATION

Neutralization of ASS acidity by the alkalinity of seawater is the ultimate natural process in the geochemical sulfur cycle. Certainly during the Quaternary sea-level fluctuations, many/most coastal estuary embayments containing sulfidic/sulfuric sediments were excavated to offshore storage during low sea levels and the contained acidity was neutralized. During rising and high sea levels, these embayments were refilled from offshore sediment stores and terrestrial erosion and this material accumulated sulfide minerals from the dissolved sulfate of seawater. The use of this seawater neutralization process with human-induced acidification needs a cautious approach because the downstream ecology of the receiving waters often depends upon the natural-contained alkalinity. The NSW sugar industry, in their required canefield drain management, must apply additional neutralization (usually by agricultural limestone) of any disturbed materials from drains. Since much of the acidity exported in drains from ASS is sourced from very close to the drains,^[14] the elimination of some drains that is possible with laser grading, and egress-pump management to suitable water level above the depth of the ASS sulfidic layer, has been shown to greatly reduce acidity export but also to improve productivity.^[12] In attempting to greatly reduce acidity export from ASS canelands, we have trialed the application of techniques used in the mining industry.^[13,27] The application of other possible options^[25,26] may follow.

CONCLUSION

The understanding of the need and techniques for best management of ASS has markedly changed in the last 20 years, from one predominantly concerned with improvement of their agronomic usefulness to one concerning their environmental impacts. The management of what was previously considered an issue for soil scientists and agronomists must involve environmentalists and particularly the stakeholders. As well, the boundary between the previous “ASS” and “acid rock/mine drainage” has tended to disappear as recognition is made that the chemical and hydrological processes for these two disciplines are associated with the common sulfide mineral oxidation and acidity and dissolved metal drainage processes. The emergence of the “pyrite-heave” problem with using sulfidic fill in the construction industry suggests that there is need for an even wider involvement of researchers, practitioners, regulators, and stakeholders in understanding sulfurous materials. The political and regulatory environment is quite rapidly changing as the problems of ASS are recognized, and it is to be hoped that there is adoption

by all stakeholders and land managers of a culture of environmental stewardship so as to reduce the problems arising from ASS.

REFERENCES

1. Fanning, D.S. Acid sulfate soils. In *Encyclopedia of Soil Science*, 2nd Ed.; Lal, R., Ed.; Taylor and Francis: New York, 2006; 11–13.
2. Rickard, D. *Pyrite*; Oxford University Press: Oxford and New York, 2015; 1–340.
3. Fitzpatrick, R.; Shand, P., Eds. *Inland Acid Sulfate Soil Systems Across Australia*, CRC LEME Open File Report 249; CRC LEME: Perth, Australia, 2008; 1–303. Available at CRC LEME, Open File Reports Index (accessed June 2015).
4. Melville, M.D.; White, I. Acid sulfate soils: Management. In *Encyclopedia of Environmental Management*; Taylor and Francis: New York, 2013; 55–60.
5. Easton, C. The trouble with the Tweed. *Fishing World*, March, **1989**, 58–59.
6. White, I.; Melville, M.D.; Macdonald, B.; Quirk, R.; Hawken, R.; Tunks, M.; Buckley, D.; Beattie, R.N.; Williams, J.; Heath, L. From conflicts to wise practice agreement and national strategy: Cooperative learning and coastal stewardship in estuarine floodplain management, Tweed River, Eastern Australia. *J. Clean. Prod.* **2007**, *15*, 1545–1558.
7. Albert, S.; O'Neil, J.M.; Udy, J.W.; Ahern, K.S.; O'Sullivan, C.M.; Dennison, W.C. Blooms of cyanobacterium *Lyngbya majuscula* in coastal Queensland, Australia: Disparate sites, common factors. *Mar. Pollut. Bull.* **2005**, *51*, 428–437.
8. Faltmarsch, R.M.; Astrom, M.E.; Vuori, K.M. Environmental risk of metals mobilized from acid sulphate soils in Finland: A literature review. *Boreal Environ. Res.* **2008**, *13*, 444–456.
9. Ministry of Agriculture and Forestry, and Ministry of the Environment. *Guidelines for Mitigating the Adverse Effects of Acid Sulphate Soils in Finland Until 2020*, March 18, 2011; 1–27. Available at http://mmm.fi/documents/1410837/1720912/2_2011_mmm_hapsu-eng-WEB.pdf/9fa91ff2-6c93-41cb-996a-12c05af2a586 (accessed June 2015).
10. Macdonald, B.C.T.; Denmead, O.T.; White, I.; Melville, M.D. Natural sulfur dioxide emissions from acid sulfate soils. *Atmos. Environ.* **2004**, *38*, 1473–1480.
11. Hicks, W.; Fitzpatrick, R. Greenhouse gas emissions and toxic gas emissions from soil organic matter and carbonates associated with acid sulfate soils. In *Inland Acid Sulfate Soil Systems across Australia*, CRC LEME Open File Report 249; Fitzpatrick, R., Shand, P., Eds.; CRC LEME: Perth, 2008; 137–148.
12. Quirk, R.G.; Melville, M.D.; White, I.; Macdonald, B.C.T.; Keene, A.F. Better management of acid sulfate soils improves drainage water quality and sugarcane yields. In *International Society of Sugar Cane Technologists, XXVI Congress*, Durban, SA, Aug. 1, 2007; Hogarth, D.M., Ed.; AGP 29, 485–489.
13. Quirk, R.G.; Kinsela, A.S.; White, I.; Macdonald, B.C.T.; Melville, M.D. Assessment of a constructed wetland to reduce acidity discharge in caneland drains. In *Australian Society of Sugar Cane Technologists, 32 Congress*, Bundaberg, QLD, May 11–14, 2010; 278–285.
14. Kinsela, A.S.; Melville, M.D. Mechanisms of acid sulfate soil oxidation and leaching under sugarcane cropping. *Aust. J. Soil Res.* **2004**, *42*, 569–578.
15. Mosely, L.M.; Fitzpatrick, R.W.; Palmer, D.; Leyden, E.; Shand, P. Changes in acidity and metal geochemistry in soils, groundwater, drain and river water in the Lower Murray River after a severe drought. *Sci. Total. Environ.* **2014**, *485–486*, 281–291.
16. Creeper, N.L.; Shand, P.; Hicks, W.; Fitzpatrick, R.W. Pore-water geochemistry of inland acid sulfate soils with sulfuric horizons following post-drought re-flooding with freshwater. *J. Environ. Qual.* **2015**, *44*, 989–1000.
17. Baldwin, D.S.; Capon, S.J. *Sulfidic Sediments in Inland Waterways*, Waterline Report Series N. 55; National Water Commission: Canberra, Sept. 2011. (ISBN 978-921853-32-6; accessed June 2015).
18. McCabe, B.A.; McKeon, E.P.; Virbukiene, R.J.; Mannion, P.J.; O'Connell, A.M. Pyritiferous mudstone-siltstone: Expansion rate measurement and prediction. *Quat. J. Eng. Geol. Hydrol.* **2015**, *48*, 41–54.
19. Ballivy, G.; Rivard, P.; Pepin, C.; Tonguay, M.G.; Dion, A. Damages to residential buildings related to pyritic rockfills: Field results of an investigation on the south shore of Montreal, Quebec, Canada. *Canadian J. Civil Eng.* **2002**, *29*, 246–255.
20. Hawkins, A.B., Ed. *Implications of Pyrite Oxidation for Engineering Works*; Springer International Publishing: Switzerland, 2014; 1–314.
21. Maher, M.I.J.; Gray, C. Aggregates prone to causing pyrite-induced heave: How they can be avoided. In *Proceedings of 17th Extractive Industry Geology Conference*; Hunger, E., Brown, T.J., Lucas, G., Eds.; EIG Conferences Ltd, Edge Hill University: Ormskirk, 2014; 58–66.
22. Britpave. *Guidelines for Stabilization of Sulfate-bearing Soils*; British Cementitious Paving Association: Surrey, 2005; 1–10. Available at http://www.britpave.org.uk/uploads/documents/originals/sulfate_soils.pdf (accessed June 2015).
23. Wilson, B.P.; White, I.; Melville, M.D. Floodplain hydrology, acid discharge and change in water quality associated with a drained acid sulfate soil. *Mar. Freshwater Res.* **1999**, *50*, 149–157.
24. White, I.; Melville, M.D.; Wilson, B.P.; Sammut, J. Reducing acidic discharges from coastal wetlands in Eastern Australia. *Wetlands Ecol. Manag.* **1997**, *5*, 55–72.
25. Tichy, R.L. Treatment of solid material containing inorganic sulfur compounds. In *Environmental Technologies to Treat Sulfur Pollution: Principles and Engineering*, Chapter 14; Lens, P.N.L., Pol, L.H., Eds.; IWA Publishing: London, 2010; 329–354.
26. Johnson, D.B.; Hallberg, K.B. Acid mine drainage options: A review. *Sci. Total. Environ.* **2005**, *338*, 3–14.
27. Green, R.; Waite, T.D.; Melville, M.D.; Macdonald, B.C.T. Effectiveness of an open limestone channel in treating acid sulfate soil drainage. *Water Air Soil Poll.* **2008**, *191*, 293–304.

Acid Sulfate Soils: Problems

Martin C. Rabenhorst

Department of Natural Resource Sciences, University of Maryland, College Park, Maryland, U.S.A.

Delvin S. Fanning

Department of Environmental Science and Technology, University of Maryland, College Park, Maryland, U.S.A.

Abstract

Acid sulfate soils (acid SS) are, or have the potential to become, extremely acidic due to the oxidation of naturally occurring, reduced sulfur compounds. Because its formation requires sulfate to be present, most acid SS occur in coastal areas. Sulfides within geological deposits remain stable indefinitely in anaerobic environments. Active acid SS form when environmental shifts or alterations expose sulfides to an oxidizing environment. This can be a natural process, such as coastal regression or tectonic coastal uplift, or it may be the result of human activity, such as drainage of coastal areas or earthmoving activities during construction or mining. Whatever the cause, once placed in an aerobic environment, particular bacteria oxidize sulfide minerals and produce sulfuric acid.

INTRODUCTION

What Are Acid Sulfate Soils?

Acid sulfate soils (acid SS) are, or have the potential to become, extremely acidic due to the oxidation of naturally occurring, reduced sulfur compounds. Oxidation forms sulfuric acid causing extreme soil acidity when there is inadequate buffering or neutralizing capability within the soil. In acid SS, pH values < 4 are common, and in extreme instances, pH values below 2 may develop. Soils are sometimes referred to as potential acid SS prior to oxidation of the sulfide minerals, or active acid SS once the extreme acidity has developed (see *Acid Sulfate Soils*, p. 17–19).

In most acid SS, the reduced sulfur compound is the mineral pyrite (FeS_2), but less commonly, elemental sulfur, monosulfide, or polysulfide minerals can cause the same problem. Therefore, acid SS may form wherever sulfide minerals accumulate. The environmental requirements for the formation of sulfide have been specified by numerous workers (see *Acid Sulfate Soils: Formation*, p. 26–30). Because its formation requires sulfate to be present, most acid SS occur in coastal areas although there are cases where acid SS have formed from older rocks or sediments, which at one time were near the coast but are now located in interior settings.

Sulfides within geological deposits remain stable indefinitely in anaerobic environments. Active acid SS form when environmental shifts or alterations expose sulfides to an oxidizing environment. This can be a natural process such as coastal regression or tectonic coastal uplift, or it may be the result of human activity such as drainage of

coastal areas or earthmoving activities during construction or mining. Whatever the cause, once placed in an aerobic environment, particular bacteria oxidize sulfide minerals and produce sulfuric acid. When these soil materials lack minerals to adequately buffer or neutralize the acidity generated, extremely low pH conditions may develop leading to the formation of active acid SS (see *Acid Sulfate Soils: Formation*, p. 26–30).

ENVIRONMENTAL PROBLEMS ASSOCIATED WITH ACID SS

Land Disturbance

Mining

Large-scale earthmoving activity is one important cause of acid SS. Mining activity was one of the first recognized.^[1] Many coal deposits formed in coastal environments contain pyrite either within the coal itself or in the surrounding rocks. Prior to recognition of the acid SS problem, pyrite-bearing soil was commonly left unattended resulting in the formation of active acid SS. Soil surveys in the Appalachian region of the United States include map units for active acid SS such as Sulfudepts. These soils have very low pH in upper horizons, large components of rock fragments, and a high density due to compaction by heavy equipment. For these reasons, extensive areas of abandoned mined lands remained nearly unvegetated for decades. In addition to the problem of acid soils, water (see *Acid Mine Drainage*, p. 6–10) from such acid SS degrades aquatic environments.

Massive additions of lime are usually needed to neutralize soil acidity so that hardy plants may become established. In the last few decades, progress has been made in the United States in recognizing acid SS problems during mining, and both federal (Surface Mining and Control Act of 1977, SMCRA) and state regulations require non-pyritic “topsoil” to be stored separately from the sulfide-bearing materials. During land reclamation, the topsoil is returned to the soil surface providing a better environment for plant establishment and growth.

New mineland reclamation strategies have largely eliminated the problem of barren acid SS on mined land, however, AMD persists. In addition to sulfuric acid, drainage waters carry high levels of iron, aluminum, and manganese, which detrimentally impact the microbial and invertebrate communities of streams.^[2,3] Such waters often become uninhabitable by crustaceans, fish, or other organisms further up the food chain.

Urban development

Geological weathering generally removes sulfides from the upper zone. However, pyrite may be found at depths ranging between 2 and 20 m in some portions of the mid-Atlantic region of the United States. Excavation and leveling associated with building homes, commercial sites, or the construction of highways may sometimes expose sulfide-bearing deposits. Dark gray colors associated with sulfide-bearing materials^[4] and their near-neutral pH when unoxidized sometimes mislead people into viewing this potential acid SS material as good “topsoil.” Unaware of the hazard, sulfide-bearing materials have sometimes been spread unintentionally over the land surface, leading to the formation of active acid SS.

Acid SS formed during urban development pose the same problems as mined lands. Extreme soil acidity presents difficulties in establishing turfgrass and ornamentals. In addition, corrosion of utility structures can be serious when concrete pipes, piers, drainage ways, foundations, streets, and sidewalks are readily attacked by sulfuric acid. The combination of acidity and salinity is particularly corrosive to steel. These acid SS may accumulate ferrous sulfate salts during dry periods and release them into surface waters during rainfall events. There are anecdotal (if not documented) accounts of fish kills in creeks and streams in urbanizing watersheds, where acidity from acid SS has been the likely cause.

Dredging

Opening and maintaining channels for marine traffic represent a nearly continuous operation in many port cities. In the eastern United States, major cities including Richmond, Washington, Baltimore, Philadelphia, and New York were built near the upper limit of navigable waters. When the water bodies are saline or brackish (containing sulfate), materials dredged from the channels often contain sulfide minerals and

represent potential acid SS. Where industry has contributed metals into the estuarine environment, acid SS problems in dredged materials (DM) may be further complicated by the presence of heavy metals in the acid soil environment.

Overboard disposal of DM can lead to release of sorbed and sulfide-bound metals.^[5] Application of DM to land, or creation of new land, may be viewed as good alternatives to overboard disposal of DM, but serious problems remain. Acid SS formed in materials dredged from Baltimore Harbor, e.g., had pH values of 2.5 and contained alarmingly high levels of zinc, nickel, and copper. In the 1980s and 1990s, approximately 600 ha of land were created in the northern part of Chesapeake Bay through the deposition of DM, in an area known as Hart-Miller Island, with similar acid SS problems.

Agriculture

Effects on plant growth

Reclamation of coastal wetlands containing sulfides has been accomplished successfully in the Netherlands and northern Germany, especially where carbonates in the sediments neutralize the sulfuric acid generated through sulfide oxidation. However, there are also cases where the attempted reclamation of coastal wetlands has been disastrous, with soils developing both acidity and salinity problems (such as in Guinea-Bissau in West Africa).^[6,7] Portions of Southeast Asia also have extensive agriculture in areas plagued with acid SS problems, such as in the delta regions of VietNam, Thailand, Malaysia, Indonesia, China, and India,^[8] and similar problems also occur in Australia. The agricultural problems of acid SS are often associated with metal toxicity and nutrient deficiencies.

Acid drainage

Even in areas where agricultural use of coastal acid SS may not pose immediate problems for crops, there may still be environmental difficulties. For example, in areas of sugarcane production in Australia on acid SS, the cane survives by sending roots down into the unoxidized (and non-acidic) portion of the soil. Nevertheless, acidity and metals from these acid SS may be transported through drainage waters into the rivers and estuaries, where dramatic fish kills have been reported.^[9] Strategies to minimize acid drainage from agricultural areas are under development.^[10,11]

REMEDIATION STRATEGIES FOR ACID SS PROBLEMS

Non-disturbance

The old adage “an ounce of prevention is worth a pound of cure” is especially applicable to acid SS, particularly where

the problems are related to land disturbance. It is far better and far easier to avoid the exposure and oxidation of sulfide-bearing soil materials than it is to try to deal with the acidic aftermath. Using geomorphology and modeling techniques, efforts have been made to predict the depths at which sulfides are likely to be encountered within upland landscapes.^[12] This can help prevent the inadvertent exposure of sulfide-bearing materials during construction activity. In cases such as coal and lignite mining, where disturbance may be unavoidable, it has been effective (and is mandated) to separate the sulfide-bearing material from that which is not sulfidic. During landscape reclamation, the non-sulfidic materials are replaced at the soil surface, preventing the development of acid SS in the upper rooting zone for plants.

Heavy Liming

Extremely high rates of liming have, in some cases, been proven effective in neutralizing the sulfuric acid in acid SS. In order to accomplish this, however, liming rates of 25–75 tons/acre may be needed and, in some cases, rates as high as 150 tons/acre have been used. It is important that sufficient lime be added to neutralize both the active acidity and also the potential acidity that remains in yet unoxidized sulfide minerals. A limitation of liming is that it usually can neutralize only acidity within the uppermost portion of the soil (15–30 cm), leaving the subsoil extremely acidic.

Natural Processes of Ripening or Flooding

Rather than trying to quickly neutralize acidity that is being generated through acid sulfate weathering, some have advocated permitting, or even enhancing, the natural oxidation of pyrite in acid SS. This process has been termed *ripening*. Allowing the sulfides to oxidize naturally for a period of time (estimates range from 10 to 50 years) makes the soil more easily neutralized by liming. On the other hand, some have advocated altering soil drainage conditions by flooding or saturation to cause the sulfate to revert to sulfide. While such strategies may be able to prevent or slow the oxidation of sulfides, generally there is insufficient energy (unoxidized organic matter) in the systems to induce significant sulfate reduction in soils that have already become acidic.

Strategies for Acid Drainage

Neutralization

One way to remediate acid drainage has been through direct neutralization of the surface water, either in ponds where precipitation products (usually iron oxides) may accumulate or within the streams themselves. The first efforts involved lining the streambed with limestone gravel. The

high pH of the oxygenated, iron-rich water resulted in the precipitation of iron and aluminum oxide coatings directly on the gravel. This *armoring* of the gravel quickly reduced their effectiveness in neutralizing acidity. Automatic lime dosers have been developed, which periodically add pulverized limestone to a stream based on monitored water pH values. The fine-grained nature of the lime eliminates the problem of armoring, raises the pH of the water, and lowers the levels of soluble metals. The North Branch of the Potomac River, which was once terribly polluted by AMD, has been successfully restored as a recreational trout fishery through the use of lime dosers. Efforts have attempted to use some non-traditional or waste materials to neutralize acidified waters.^[13] Anoxic limestone drains, a variation on neutralization using limestone, were designed to intercept discharging groundwater before it emerged at the land surface. By isolating limestone gravel from oxygenated water below ground using impermeable plastic, it is thought that armoring of the gravel can be prevented.^[14]

Wetlands

Over the last years, constructed wetlands have emerged as a treatment for AMD. Much of the published work emphasized treatment through such processes as neutralization and oxidation, which one might expect to be better accomplished in more aerobic environments than wetlands.^[15] Some have focused more on wetland processes such as sulfate reduction,^[16] and it has been shown that properly designed and engineered wetland systems can effectively remediate the negative effects of AMD through sulfidization and generation of bicarbonate.^[17]

REFERENCES

1. Barnhisel, R.I.; Powell, J.L.; Akin, G.W.; Ebelhar, M.W. Characteristics and reclamation of acid sulfate mine spoils. In *Acid Sulfate Weathering*; Kittrick, J.A., Fanning, D.S., Hossner, L.R., Eds.; Spec. Pub. Ser. No. 10; Soil Science Society of America: Madison, 1982; 225–234.
2. Niyogi, D.K.; Lewis, A.W.M.; McKnight, D.M. Litter breakdown in mountain streams affected by mine drainage: Biotic mediation of abiotic controls. *Ecol. Appl.* **2000**, *11*, 506–516.
3. Cherry, D.S.; Currie, R.J.; Coucek, D.J.; Latimer, H.A.; Trent, G.C. An Integrative assessment of a watershed impacted by abandoned mined land discharges. *Environ. Poll.* **2001**, *111*, 377–388.
4. Fanning, D.S.; Rabenhorst, M.C.; Bigham, J.M. Colors of acid sulfate soils. In *Soil Color*; Bigham, J.M., Ciolkosz, E.J., Eds.; Spec. Pub. Ser. No. 31; Soil Science Society of America: Madison, 1992; 91–108.
5. Simpson, S.L.; Apte, S.C.; Batley, G.E. Effect of short-term resuspension events on the oxidation of cadmium, lead, and zinc sulfide phases in anoxic estuarine sediments. *Environ. Sci. Tech.* **2000**, *34*, 4533–4537.

6. Fanning, D.S.; Burch, S.N. Acid sulphate soils and some associated environmental problems. *Adv. Geo Ecol.* **1997**, *30*, 145–158.
7. Fanning, D.S.; Burch, S.N. Coastal acid sulfate soils. In *Reclamation of Drastically Disturbed Lands*; Barnhisel, R.I., Darmody, R.G., Daniels, W.L., Eds.; Agron. Monogr. 41; ASA, CSSA, SSSA: Madison, 2000; 921–937.
8. Mathew, E.K.; Panda, R.K.; Nair, M. Influence of subsurface drainage on crop production and soil quality in a low-lying acid sulphate soil. *Agr. Water Manage.* **2001**, *47*, 191–209.
9. Cook, F.J.; Hick, W.; Gardner, E.A.; Carlin, G.D.; Froggatt, D.W. Export of acidity in drainage water from acid sulphate soils. *Mar. Poll. Bull.* **2000**, *41*, 319–326.
10. Yang, X.H.; Zhou, Q.M.; Melville, M. An integrated drainage network analysis system for agricultural drainage management—part 2: The application. *Agr. Water Manage.* **2000**, *45*, 87–100.
11. Blunden, B.G.; Indraratna, B. Evaluation of surface and groundwater management strategies for drained sulfidic soil using numerical simulation models. *Austr. J. Soil Res.* **2000**, *38*, 569–590.
12. Valladares, T.M. *Estimating Depth to Sulfide-Bearing Sediments in the Maryland Coastal Plain: A Pedogeomorphic Modeling Approach*. MS Thesis, University of Maryland, College Park, 1998; 232 pp.
13. Chtaini, A.; Bellaloui, A.; Ballivy, G.; Narasiah, S. Field Investigation of controlling acid mine drainage using alkaline paper mill waste. *Water Air Soil Poll.* **2001**, *125*, 357–374.
14. Skousen, J.G.; Sextone, A.; Ziemkiewicz, P.F. Acid mine drainage control and treatment. In *Reclamation of Drastically Disturbed Lands*; Barnhisel, R.I., Darmody, R.G., Daniels, W.L., Eds.; Agron. Monogr. 41; ASA, CSSA, SSSA: Madison, 2000; 131–168.
15. Henrot, J.; Wieder, R.K.; Heston, K.P.; Nardi, M.P. Wetland treatment of coal mine drainage: Controlled studies of iron retention in model wetland systems. In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural*; Hammer, D.A., Ed.; Lewis Publishers: Chelsea, 1989; 793–800.
16. McIntire, P.E.; Edenborn, H.M. The use of bacterial sulfate reduction in the treatment of drainage from coal mines. In *Proceedings of the 1990 Mining and Reclamation Conference and Exhibition*; Skousen, J., Sencindiver, J., Samuel, D., Eds.; West Virginia University: Morgantown, 1990; Vol. II, 409–415.
17. Rabenhorst, M.C.; James, B.R.; Magness, M.C.; Shaw, J.N. Iron removal from acid mine drainage in wetlands by optimizing sulfate reduction. In *The Challenge of Integrating Diverse Perspectives in Reclamation*; Proceedings of the 10th National Meeting of the American Society of Surface Mining and Reclamation; ASSMR: Spokane, 1993; 678–684.

Actinobacteria

Marketa Šágová-Marecková

Jan Kopecký

Crop Research Institute, Prague, Czech Republic

Abstract

Actinobacteria constitute an important part of soil microbial communities and are widely distributed in contrasting environments. Their communities seem mostly structured by pH, but other environmental factors such as soil moisture and organic matter content and quality are also important. A major activity of actinobacteria in soil is decomposing recalcitrant organic matter. Actinobacteria are also prominent producers of secondary metabolites including commercially used antibiotics.

DISPERSAL OF ACTINOBACTERIA

Actinobacteria occupy multiple niches in all environments. They are very important in surface soils and represent up to 63% of the bacterial communities.^[1] Actinobacteria amount to 10^7 colony-forming units (CFU) per gram of soil in grasslands^[2] and to 10^4 CFU per gram of soil in cropped fields.^[3] Actinobacteria occur in soils of all latitudes, freshwater and marine habitats, deep mines, plants, and animals. They exist in less studied soils, e.g., under tropical rainforest, in extremely acidic or alkaline soils, or in sand dunes.

The wide distribution of actinobacteria, however, does not prove the even distribution of particular populations.^[4] It rather seems that the actinobacterial communities are distinct not only over the large areas^[4] but also within one sampling location.^[5]

FACTORS INFLUENCING DISTRIBUTION

Soil bacterial communities are structured both at the microscale and at the macroscale by factors such as soil moisture, soil nutrients, particle size, soil organic matter (SOM) content, and other organisms.^[6] Actinobacteria prefer medium water content although some genera are typically observed in both dry and waterlogged soils.^[7] Some actinobacteria can survive in extreme dry conditions because they predominate in desiccated areas or deserts. Soil moisture content is also related to the spatial variation of nutrients [i.e., nitrogen (N), phosphorus, carbon (C), calcium, and magnesium). Thus, different preferences of actinobacterial species to soil moisture may also be related to their nutrition. In general, cool and nutrient-poor soils narrow the number of actinobacterial genera, with minimum diversity occurring in high mountains (Table 1). Some specific actinobacteria are abundant at high pH,

while others live in acidic soils.^[8] Occurrence of actinobacteria is positively correlated with smaller soil and SOM particles^[9] and negatively correlated with plant root biomass in forest^[10] but enriched in the rhizosphere in grasslands.^[11] Specific actinobacteria are also associated with particular plant species. Actinobacteria are usually saprophytic, so their quantity and diversity often follow that of SOM. Contrasting was their response to fertilizers where an increased proportion of actinobacteria was observed both in an unfertilized site and at a site receiving a high rate of mineral fertilizer.^[12] Actinobacteria are mostly considered K selected, which means that they are adapted to limiting conditions and possess a great ability to compete in stable environments. However, fast-growing actinobacteria with high adaptation ability are also known to live in fluctuating environments.

SEASONALITY OF ACTINOBACTERIA DISTRIBUTION

Seasonal shifts in actinobacteria communities are attributed to changes in SOM composition and water content^[13] and are more pronounced in subsurface than in surface soil, due to higher fluctuations in nutrient availability. In temperate forest, where the biomass of microorganisms is less in summer than in spring and autumn, differences among seasons are even more pronounced for actinobacteria.^[13] The highest proportion of actinobacteria in the microbial community is observed in winter,^[14] and despite a stable count of bacteria throughout the year, actinobacteria are much less frequent in summer.^[15]

ACTINOBACTERIAL GENE POOL

Actinobacteria have a versatile gene pool resulting from frequent rearrangements and gene uptake via horizontal

Table 1 Actinobacterial diversity in relationship to bacterial diversity at selected sites.

Sites	Bacteria			Actinobacteria		
	OTUs	Simpson's 1/D	Simpson's E	OTUs	Simpson's 1/D	Simpson's E
Peat bog (pH 3.2)	37	17.67	0.48	31	8.98	0.29
Mountain meadow (pH 4.9)	44	16.09	0.37	12	7.24	0.60
Meadow (pH 7.9)	39	17.23	0.44	29	9.34	0.32
Steppe (pH 7.9)	32	12.04	0.38	38	10.50	0.28
Pine forest (pH 3.7)	49	20.00	0.41	25	9.87	0.39
Pine forest (pH 7.5)	48	27.40	0.57	23	15.24	0.66
Pine forest (pH 8.0)	33	21.32	0.65	30	17.52	0.58
Deciduous forest (pH 4)	22	9.43	0.43	36	17.14	0.48
Deciduous forest (pH 6.1)	18	10.77	0.60	23	12.41	0.54
Deciduous forest (pH 7.6)	40	25.68	0.64	24	13.98	0.58

Notes: Operational taxonomic units (OTUs) represent a number of terminal restriction fragment lengths detected at the particular site for total bacteria and actinobacteria. Simpson's 1/D and Simpson's E are indices of diversity and evenness, respectively, calculated from terminal restriction fragment lengths profiles.

gene transfer events. The actinobacterial genome sizes reflect the complexity of environments they inhabit and need for adaptation.^[16] The extremes are obligate pathogens such as *Tropheryma* sp. (1 Mbp) or *Mycobacterium tuberculosis* (2 Mbp) at the low end and soil actinobacteria such as *Streptomyces* sp. (up to 10 Mbp) at the high end. Closely related species may differ substantially in their gene pools. For example, two members of the genus *Streptomyces* may share only 65% of their genes, while 30% remain strain specific. Even genome size varies among related species depending on their life strategy; e.g., in the genus *Frankia*, which forms N₂-fixing root nodules, the genome sizes range from 5.43 to 9.04 Mbp, with the larger genome size associated with an increased host range.^[17]

VERSATILITY OF ACTIVITIES

Actinobacteria can colonize the plant rhizosphere, where they can parasitize plant roots or benefit from plant exudates, which represent easily available food source. Soil is also a reservoir for many actinobacteria that are important human or animal pathogens and which are represented by strains of the genera *Mycobacterium*, *Nocardia*, *Actinomadura*, or *Rhodococcus*.^[18] Actinobacteria support the growth of plants, e.g., by N fixation or antagonism to fungal pathogens.^[19] They are highly adaptive because they can assist in the development of the climax state of plant communities by supporting the growth of plants and associated bacteria. Actinobacteria represent a reservoir of strains for: 1) biodegradation of pollutants; 2) bio-control inoculants in commercial greenhouse; and 3) symbiotic association with actinorhizal plants that aid

in remediation of degraded soils. In addition, actinobacteria are pharmaceutically important because they are used for production of a majority of antibiotics as well as antibacterial, antiviral, antiparasitic, antitumor, and immunomodulating compounds. Secondary metabolites of actinobacteria are also used in agriculture as herbicides or insecticides.

DOMINATING FUNCTIONS IN SOIL ENVIRONMENT

A primary function of actinobacteria in soil is the decomposition of recalcitrant SOM. Actinobacteria use plant, animal, and fungal detritus as C and energy sources, and they are efficient degraders of organic polymers such as lignin, cellulose, and chitin. Solubilization of lignin is an important activity in C cycling.^[20] Fungi and actinobacteria are key players in decomposition of litter, and their proportion in the microbial community reflects the chemical composition of the litter and the state of the SOM decomposition.^[13] The specific relationship of actinobacteria to SOM quality is reflected by their vertical distribution in the soil profile. Surface soil has a higher number of cultivable actinobacteria than subsoil in grassland,^[2] and the proportion of actinobacteria increases with an increase in depth.^[21] Their numbers follow the site-specific distribution of nutrients in all situations.

SECONDARY METABOLITE PRODUCTION

Actinobacteria are major producers of secondary metabolites, a structurally diverse group of substances, whose functions are debatable.^[22] In the natural environments,

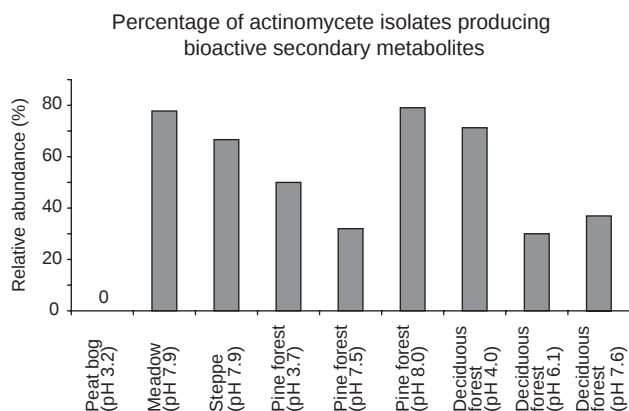


Fig. 1 Percentage of antibiotic-producing actinobacteria strains isolated from differing sites. The antibiotic activity was tested against strains of *Escherichia coli* and *Kocuria rhizophila*. The indicated percentage is a sum of strains exhibiting either of the two tested activities.

secondary metabolites have the potential to modify the structure and functioning of microbial communities by antibiosis or as communication agents. Secondary metabolites of actinobacteria also constitute a large number of clinically used antibiotics. However, the great diversity of genes coding for secondary metabolite biosynthetic pathways is not fully recognized until the first actinobacteria genome sequencing projects. The actinobacteria strains, previously known as producers of a limited number of secondary metabolites, have genetic potential to synthesize a vast number of secondary metabolites. The genetic pool of secondary metabolism is, thus, much more abundant and diverse than the previously estimated and represents great potential for discovery of new or modified bioactive compounds of medical, industrial, and agricultural importance. Further, each soil environment is specific with regard to the secondary metabolites formed and also differs in the percentage of actinobacteria that produce secondary metabolites, specifically antibiotics (Fig. 1).

ACKNOWLEDGMENT

This work was supported by the Ministry of Education, Youth and Sports of the Czech Republic, Project No. ME 10137.

REFERENCES

- Hackl, E.; Zechmeister-Boltenstern, S.; Bodrossy, L.; Sessitsch, A. Comparison of diversities and compositions of bacterial populations inhabiting natural forest soils. *Appl. Environ. Microb.* **2004**, *70*, 5057–5065.
- Cho, S.T.; Tsai, S.H.; Ravindran, A.; Selvam, A.; Yang, S.S. Seasonal variation of microbial populations and biomass in Tatachia grassland soils of Taiwan. *Environ. Geochem. Hlth* **2008**, *30*, 255–272.
- Shirokikh, I.G.; Zenova, G.M.; Zvyagintsev, D.G. Actinomyces in the rhizosphere of barley grown on acid soddy podzolic soil. *Microbiology* **2002**, *71*, 445–459.
- Wawrik, B.; Kutliev, D.; Abdivasieva, U.A.; Kukor, J.J.; Zylstra, G.J.; Kerkhof, L. Biogeography of actinomycete communities and type II polyketide synthase genes in soils collected in New Jersey and central Asia. *Appl. Environ. Microb.* **2007**, *73* (9), 2982–2989.
- Bredholdt, H.; Galatenko, O.A.; Engelhardt, K.; Fjaervik, E.; Terekhova, L.P.; Zotchev, S.B. Rare actinomycete bacteria from the shallow water sediments of the Trondheim fjord, Norway: Isolation, diversity and biological activity. *Environ. Microbiol.* **2007**, *9*, 2756–2764.
- Nunan, N.; Wu, K.; Young, I.M.; Crawford, J.W.; Ritz, K. Spatial distribution of bacterial communities and their relationships with the micro-architecture of soil. *FEMS Microbiol. Ecol.* **2003**, *44*, 203–215.
- Williams, S.T.; Shameemullah, M.; Watson, E.T.; Mayfield, C.I. Studies on the ecology of actinomycetes in soil—VI. The influence of moisture tension on growth and survival. *Soil Biol. Biochem.* **1972**, *4*, 215–225.
- Zakalyukina, Y.V.; Zenova, G.M.; Zvyagintsev, D.G. Acidophilic soil actinomycetes. *Microbiology* **2002**, *71*, 342–345.
- Sessitsch, A.; Weilharter, A.; Gerzabek, M.H.; Kirchmann, H.; Kandeler, E. Microbial population structures in soil particle size fractions of a long-term fertilizer field experiment. *Appl. Environ. Microb.* **2001**, *67*, 4215–4224.
- Brant, J.B.; Myrold, D.D.; Sulzman, E.W. Root controls on soil microbial community structure in forest soils. *Oecologia* **2006**, *148*, 650–659.
- Giri, B.; Giang, P.H.; Kumari, R.; Prasad, R.; Varma, A. Microbial diversity in soils. In *Microorganisms in Soils: Roles in Genesis and Functions*; Buscot, F., Varma, A., Eds.; Springer: Heidelberg, 2005; 19–58.
- Zhang, Q.C.; Wang, G.H.; Yao, H.Y. Phospholipid fatty acid patterns of microbial communities in paddy soil under different fertilizer treatments. *J. Environ. Sci. (China)* **2007**, *19*, 55–59.
- Myers, R.T.; Zak, D.R.; White, D.C.; Peacock, A. Landscape-level patterns of microbial community composition and substrate use in upland forest ecosystems. *Soil Sci. Soc. Am. J.* **2001**, *65*, 359–367.
- Lipson, D.A.; Schmidt, S.K. Seasonal changes in an Alpine soil bacterial community in the Colorado rocky mountains. *Appl. Environ. Microb.* **2004**, *70*, 2867–2879.
- Yang, S.-S.; Fan, H.-Y.; Yang, C.-K.; Lin, I.-C. Microbial population of spruce soil in Tatachia mountain of Taiwan. *Chemosphere* **2003**, *52*, 1489–1498.
- Ventura, M.; Canchaya, C.; Tauch, A.; Chandra, G.; Fitzgerald, G.F.; Chater, K.F.; van Sinderen, D. Genomics of *Actinobacteria*: Tracing the evolutionary history of an ancient phylum. *Microbiol. Mol. Biol. R.* **2007**, *71* (3), 495–548.
- Normand, P.; Lapierre, P.; Tisa, L.S.; Gogarten, J.P.; Alloisio, N.; Bagnarol, E.; Bassi, C.A.; Berry, A.M.; Bickhart, D.M.; Choise, N.; Couloux, A.; Cournoyer, B.; Cruveiller, S.; Daubin, V.; Demange, N.; Francino, M.P.;

- Goltsman, E.; Huang, Y.; Kopp, O.R.; Labarre, L.; Lapidus, A.; Lavire, C.; Marechal, J.; Martinez, M.; Mastrorunzio, J.E.; Mullin, B.C.; Niemann, J.; Pujic, P.; Rawnsley, T.; Rouy, Z.; Schenowitz, C.; Sellstedt, A.; Tavares, F.; Tomkins, J.P.; Vallenet, D.; Valverde, C.; Wall, L.G.; Wang, Y.; Medigue, C.; Benson, D.R. Genome characteristics of facultatively symbiotic *Frankia* sp. strains reflect host range and host plant biogeography. *Genome Res.* **2007**, *17*, 7–15.
18. De Groote, M.A.; Pace, N.R.; Fulton, K.; Falkinham, J.O. Relationships between *Mycobacterium* isolates from patients with pulmonary mycobacterial infection and potting soils. *Appl. Environ. Microb.* **2006**, *72* (12), 7602–7606.
19. Coombs, J.T.; Michelsen, P.P.; Franco, C.M.M. Evaluation of endophytic actinobacteria as antagonists of *Gaeumannomyces graminis* var. *tritici* in wheat. *Biol. Control* **2004**, *29*, 359–366.
20. Ball, A.S.; Betts, W.B.; McCarthy, A.J. Degradation of lignin-related compounds by actinomycetes. *Appl. Environ. Microb.* **1989**, *55*, 1642–1644.
21. Fierer, N.; Schimel, J.P.; Holden, P.A. Variations in microbial community composition through two soil depth profiles. *Soil Biol. Biochem.* **2003**, *35*, 167–176.
22. Davies, J.; Spiegelman, G.B.; Yim, G. The world of subinhibitory antibiotic concentrations. *Curr. Opin. Microbiol.* **2006**, *9*, 445–453.

Aeration: Measurement

Robert E. Sojka

Northwest Irrigation and Soils Research Lab, U.S. Department of Agriculture (USDA), Kimberly, Idaho, U.S.A.

H.D. Scott

Center for Agribusiness and Environmental Policy, Mount Olive College, Mount Olive, North Carolina, U.S.A.

Abstract

Soil aeration is important to soil processes and plant growth. It can be characterized at various levels of complexity in terms of capacity, intensity, or transport rate. Quantification of oxygen (O₂) transport rates and measurement of concentrations of other important gases besides O₂ provide the best overall characterization of soil aeration for modern investigations, but simpler measurements can be valuable as rapid diagnostics for land managers. Modern instrumentation has made sophisticated characterization of soil aeration attainable at reasonable cost.

INTRODUCTION

Soil oxygen (O₂) enables aerobic respiration of plant roots and soil micro- and meso-flora and micro- and meso-fauna. Its availability can be limited by soil wetness, compaction, discontinuous pores, or high respiration in moist soil due to elevated soil temperature or the incorporation of fresh organic substrate. With O₂ depletion, soil redox potential shifts from oxidative to reducing conditions, hampering plant growth because of less efficient metabolic pathways and release into soil of toxic by-products of reduction chemistry or anaerobic respiration. Several texts are excellent sources for fundamental soil aeration concepts.^[1–3]

Measurements of soil aeration fall into the following three categories: “capacity,” volume of gas-filled void space; “intensity,” partial pressure or concentration of O₂ (or other gases) in the voids; and “transport rate,” the rapidity at which O₂ can be supplied to a point in the soil. Difficulty in measurement increases in the following order: capacity < intensity < rate, as do the value and insight of the measurements.

CAPACITY MEASUREMENT

Capacity describes the ability of soil to contain air. Soil capacity parameters include total porosity, void ratio, relative saturation, and air-filled porosity. These parameters are calculated from simple measurements of soil volume, particle density, bulk density, and water content. Capacity has been used to understand plant growth and yield for over a century. A “rule of thumb” associates impaired plant

growth with <10% soil air volume. Soil attributes affecting capacity include texture, structure, water content, clay mineralogy, and sodium adsorption ratio. Sandy soils have less total porosity than clays, but in sands, the pores are large and well connected. Percent void space tends to increase in finer-textured (more clay), less-compact soils. Because clays have many small unconnected pores and retain water more readily than coarser textured soils (sands), plants tend to suffer O₂ limitation more commonly in clays despite their greater porosity. High soil sodium content or smectitic clay mineralogy (swelling clays) can exacerbate this tendency. Well-aggregated clays often avoid poor aeration because structure enhances macroporosity. Interaggregate pores are larger, better connected, and better drained than smaller intra-aggregate pores which usually contain more water.

INTENSITY MEASUREMENT

Intensity (partial pressure or concentration) measurements have been facilitated by instrumentation, allowing rapid measurement of O₂ and other gas concentrations. The ambient atmosphere is 78% nitrogen (N) and 21% O₂. The remaining 1% contains other gases. Ambient carbon dioxide (CO₂) is about 0.037% and increasing at the rate of 1.5 ppmv/yr. In soil air, the O₂ concentration is <21%. The drop in O₂ below 21% generally corresponds to the CO₂ concentration increase due to the respiration of roots and soil organisms. In soil air, CO₂ is commonly 0.3–1% but can be much higher. In warm wet soil with freshly incorporated organic matter, or where carbonates are abundant, CO₂ can rise above 10%, and where drainage is

restricted, it can reach 20%. When waterlogging occurs and reducing conditions prevail, a few percent by volume of gaseous products of reducing chemistry or non-oxidative metabolism, e.g., methane (CH_4) or nitrous oxide (N_2O), can be present.

Soil air O_2 concentration can be measured using various analytical techniques, mostly depending on whether air is withdrawn from the soil or analyzed *in situ*.^[2,4,5] Withdrawn soil air samples have an interpretation problem stemming from the overrepresentation of macropore composition and/or mixing during convective extraction from the heterogeneous pore sites within the soil matrix. Buried diffusion cavities (artificial porous voids) are sometimes used for sampling points, but questions remain as to representativeness of cavity-equilibrated air. Once withdrawn, soil air samples can be analyzed by wet chemistry, paramagnetic or polarographic methods, or using gas chromatography.^[6,7] Paramagnetic and polarographic instruments can be used *in situ*, but they have limitations. Paramagnetic analyzers require gas flows of tens of cubic centimeters over the sensors. Polarographic soil O_2 sensors are usually covered by membrane. Double membrane probes are used *in situ*, to overcome calibration shifts caused by condensation on sensors. Sensors can be small enough to measure O_2 between small aggregates or within large aggregates.^[8] Samples withdrawn from soil allow for analysis of gases besides O_2 . Knowing the soil O_2 concentration does not indicate if the O_2 consumption rate can be satisfied by the rate of O_2 convection and diffusion through soil.

Soil O_2 concentration <10% by volume indicates poor aeration. However, O_2 concentration per se is an imperfect predictor of plant response. The composition of other soil gases, such as CO_2 , ethylene, and CH_4 , can affect response to a given O_2 concentration. Furthermore, O_2 concentration gives little information about the amount (mass) of O_2 (volume $\times$ concentration) in soil, or the rate at which it can move through tortuous soil pores or across water films and root membranes to reach metabolic sites where it is reduced.^[9] Specific composition of soil atmospheres vary with organic matter and mineral content, soil redox potential, and pH, making it hard for generalizations about soil air trace gas composition as soils become wetter and O_2 concentration decreases.^[1,2]

TRANSPORT RATE MEASUREMENT

Measurements of the rate of gaseous transport in soil are of the following two types: diffusion and convection. To characterize diffusion, Lemon and Erickson^[10] proposed a method to place a small platinum (Pt) wire microelectrode in soil to electrically reduce O_2 . The Pt electrode simulates a respiring root to which O_2 diffuses through air- and water-filled pores and then through a water film surrounding the root. The microelectrode measures the effect of restrictions along this pathway on the O_2 supply to the electrode

surface. The current measured through an electrode tip of known geometry, supplied at steady potential, is related to the steady-state reduction of O_2 supplied by diffusion to root surfaces of similar geometry. The technique became known as the soil ODR measurement (from “oxygen diffusion rate”). Lemon and Erickson’s seminal concept was perfected and field-adapted by others,^[11–13] providing a standardized approach to the technology and a unified interpretive framework. Eventually, commercialization provided the mass production of uniform reliable Pt electrodes and compact portable semiautomated multi-probe instrumentation.

ODR is probably the most common characterization of *in situ* soil O_2 status conducted, and it is well correlated with plant physiological, nutritional, and growth responses. ODR is the only soil O_2 status measurement suited to prolonged remote and nearly continuous measurement of dynamic soil O_2 status.^[14] Factors affecting ODR include water content, electrode contact with soil, presence of reducible compounds, salinity, temperature, and O_2 concentration.^[5,6] Numerous studies^[1,2] have identified an ODR value of $20 \mu\text{g}/\text{m}^2/\text{s}$ as a threshold for a variety of plant physiological, nutritional, and growth responses to limited soil O_2 .

Subsequent ODR advances have been achieved. Electrode miniaturization allows O_2 flux measurements within roots and root microstructures and within intact aggregates.^[15,16] Also, due to potential poisoning of the Pt microelectrodes during long-term exposure to soil by oxides, alternatives to Pt have been developed. Wax-impregnated graphite electrodes (WIGEs) have greater current efficiencies and a wider current plateau than Pt electrodes.^[17] In moist soil, the current plateau region was a function of soil water content, and the response was linear in atmospheres containing as much as 60% O_2 . The WIGEs are less susceptible to oxide poisoning, can be fabricated easily, and are less expensive.

Measurements of soil aeration based on the convection of gas use simple flow permeameters to accurately measure mass flow directly through soil,^[18] or to measure the total air pressure or the difference in air pressure between the atmosphere and soil.^[19] The convection of soil air arises from spatial differences in total air pressures due to abrupt changes in air pressure of the atmosphere, the effects of temperature differences on gas properties, the infiltration and redistribution of water in the soil profile, and the microbial production of gases such as CO_2 , nitric oxide, N_2O , and CH_4 .^[3] Flüher and Laser^[8] developed a hydrophobic membrane probe (HMP) to measure total pressure and partial pressure in soil atmospheres. The HMP consisted of a membrane-covered chamber having a small volume and two teflon capillaries connected to a differential pressure transducer or to an O_2 electrode. A water-repellent non-rigid teflon membrane excluded wet soil from the continuous gas phase of the HMP. The total air pressure in the HMP is compared with the soil surface air pressure to

determine pressure difference between the two locations. Renault et al.^[20] developed an absolute pressure probe to measure *in situ* air pressure fluctuations at the soil surface and at varying depths within the profile. Their probe had negligible signal drift and an accuracy of 10 Pa. The sensitive component of the probe is a differential pressure sensor that functions from -14,000 to 14,000 Pa and senses a pressure differential of 2500 Pa. The signal results from resistance changes in an arsenic-doped silicon membrane that functions as a Wheatstone bridge. The sensor requires 1.5 mA d.c. and its output is -40 to 40 mV. Circuitry converts the input voltage (24 V) into the stabilized current output (4–20 mA), providing a linear relationship between the differential air pressure and the output signal at a given temperature. The characteristics of the air pressure probe allow for *in situ* calibrations.

CONCLUSION

Soil aeration is important to soil processes and plant growth. It can be characterized at various levels of complexity in terms of capacity, intensity, or transport rate. The quantification of O₂ transport rates and measurement of concentrations of other important gases besides O₂ provide the best overall characterization of soil aeration for modern investigations, but simpler measurements can be valuable as rapid diagnostics for land managers. Modern instrumentation has made sophisticated characterization of soil aeration attainable at reasonable cost.

REFERENCES

1. Kozlowski, T.T., Ed. *Flooding and Plant Growth*; Academic Press Inc.: Orlando, 1984; 356 pp.
2. Gliniski, J.; Stepniewski, W. *Soil Aeration and Its Role for Plants*; CRC Press Inc.: Boca Raton, 1985; 229 pp.
3. Scott, H.D. *Soil Physics: Agricultural and Environmental Applications*; Iowa State Press: Ames, 2000.
4. Fatt, I. *Polarographic Oxygen Sensors*; CRC Press, Inc.: Cleveland, 1976.
5. Phene, C.J. Oxygen electrode measurement. In *Methods of Soil Analysis. Part 1. Physical and Mineralogical Methods. Monograph 9*, 2nd Ed.; Klute, A., Ed.; American Society of Agronomy: Madison, 1986; Vol. 9, 1137–1159.
6. Tackett, J.L. Theory and application of gas chromatography in soil aeration research. *Soil Sci. Soc. Am. Proc.* **1968**, *32* (3), 346–350.
7. Patrick, W.H. Oxygen content of soil air by a field method. *Soil Sci. Soc. Am. J.* **1977**, *41* (3), 651–652.
8. Flühler, H.; Laser, H.P. A hydrophobic membrane probe for total pressure and partial pressure measurements in the soil atmosphere. *Soil Sci.* **1975**, *120* (2), 85–91.
9. Hutchins, L.M. Studies on the oxygen supplying power of the soil together with quantitative observations on the oxygen-supplying power requisite for seed germination. *Plant Physiol.* **1926**, *1* (2), 95–150.
10. Lemon, E.R.; Erickson, A.E. The measurement of oxygen diffusion in the soil with a platinum microelectrode. *Soil Sci. Soc. Am. Proc.* **1952**, *16* (2), 160–163.
11. Letey, J.; Stolzy, L.H. Measurement of oxygen diffusion rates with the platinum microelectrode. I. Theory and equipment. *Hilgardia* **1962**, *35*, 545–576.
12. Stolzy, L.H.; Letey, J. Characterizing soil oxygen conditions with a platinum microelectrode. *Adv. Agron.* **1964**, *16*, 249–279.
13. McIntyre, D.S. The platinum microelectrode method for soil aeration measurement. *Adv. Agron.* **1970**, *22*, 235–283.
14. Phene, C.J.; Campbell, R.B.; Doty, C.W. Characterization of soil aeration *in situ* with automated oxygen diffusion measurements. *Soil Sci.* **1976**, *122* (5), 271–281.
15. Hook, D.D.; McKevlin, M.A. Use of oxygen microelectrodes to measure aeration in the roots of intact tree seedlings. In *The Ecology and Management of Wetlands. Volume 1: Ecology of Wetlands*; Hook, D.D., Ed.; Croom Helm: London, 1988; 467–476.
16. Sextone, A.J.; Revsbech, N.P.; Parkin, T.B.; Tiedje, J.M. Direct measurement of oxygen profiles and denitrification rates in soil aggregates. *Soil Sci. Soc. Am. J.* **1985**, *49* (3), 645–651.
17. Shaikh, A.U.; Hawk, R.M.; Sims, R.A.; Scott, H.D. Graphite electrode for the measurement of redox potential and oxygen diffusion rate in soil. *Nucl. Chem. Waste Man.* **1985**, *5*, 237–243.
18. Evans, D.D. Gas movement. In *Methods of Soil Analysis, Part 1. Physical and Mineralogical Properties, Including Statistics of Measurement and Sampling, Agronomy Monograph 9*; Black, C.A., Evans, D.D., White, J.L., Ensminger, L.E., Clark, F.E., Eds.; American Society of Agronomy: Madison, 1965; 319–330.
19. Flühler, H.; Peck, A.J.; Stolzy, L.H. Air pressure measurement. In *Methods of Soil Analysis. Monograph 9. Part 1. Physical and Mineralogical Methods*, 2nd Ed.; Klute, A., Ed.; American Society of Agronomy: Madison, 1986; 1161–1172.
20. Renault, P.; Mohrath, D.; Gaudu, J.-C.; Fumanal, J.-C. Air pressure fluctuations in a prairie soil. *Soil Sci. Soc. Am. J.* **1998**, *62* (3), 553–563.

Aeration: Tillage Effects

Dave J. Horne

Massey University, Palmerston North, New Zealand

Robert E. Sojka

Northwest Irrigation and Soils Research Lab, U.S. Department of Agriculture (USDA), Kimberly, Idaho, U.S.A.

Abstract

Adequate soil aeration is important to crop production and environmental protection. Tillage may profoundly influence soil aeration. Although the effect of tillage on soil aeration is dependent on a range of factors and is, therefore, hard to predict, often surface soil aeration is more favorable under well-managed conventional cultivation than no-tillage. Often, these improvements are confined to shallow soil depths and relatively short-time intervals. Where appropriate no-till equipment is used, direct-drilling may have greater potential to improve soil aeration, both in terms of uniformity throughout the soil profile and permanence in time.

INTRODUCTION

Few land management practices have the potential to impact upon soil aeration as directly or rapidly as tillage. Indeed, often, the reason for performing tillage is to modify or improve soil physical properties including aeration. The problems associated with inadequate aeration have been comprehensively reviewed elsewhere.^[1,2] Important effects of limited soil aeration in crop production are as follows: altered nutrient dynamics, a shift from oxidative to reductive chemical/biological reactions, impaired plant growth, and changes in gas equilibria affecting both soil and ambient atmospheres. For example, consider the soil nitrogen cycle that aeration effects via its influence on denitrification and gaseous nitrogen losses, decreased nitrogen mineralization rate, and a reduction in nodulation and symbiotic fixation by leguminous plants.^[3] If the oxygen supply is sufficiently limited, and anerobiosis sets in, then the products of reduction reactions may accumulate to toxic levels. In addition, a depleted oxygen supply may constrain root form and function, such as water and nutrient uptake, and therefore plant shoot performance even when many other soil physical factors are favorable.^[4] Unfortunately, relatively short periods of oxygen shortage can seriously compromise crop performance if they coincide with critical stages of crop growth.^[1] Finally, there are the effects of gas sources and sinks in the soil and transformations of soil gaseous components, and the exchange between soil and aboveground air, on the atmosphere, e.g., diminished soil aeration may enhance the emission of greenhouse gases.^[5]

While the tillage-related literature is voluminous, little of it directly addresses soil aeration. Of necessity, this

short entry critiques only research that has measured aeration status directly—particularly indices of concentration and rate—and will make little or no attempt to draw inferences about the effect of tillage on soil aeration from studies reporting other related soil characteristics. Although bulk density, moisture content, and pore size distribution are related to soil aeration, and so may be indicative of aeration status, their direct relevance to a nuanced understanding of soil aeration is problematical. For instance, measurements of pore space convey little about pore continuity, tortuosity, or stability,^[6] whereas these effects are largely integrated de facto in measurements of oxygen diffusion rate (ODR).

EFFECTS OF SURFACE TILLAGE

Conventional tillage of virgin soil or soils growing permanent or long-term pasture for livestock grazing often improves soil aeration.^[7] In this situation, it is relatively straightforward to prepare a seedbed of good tilth and alleviate any compaction associated with animal or vehicular traffic. However, the timing of such plowing is important, as cultivation of wet soil leads to a dramatic reduction in soil aeration.^[8] If conventional tillage is poorly managed or executed in difficult circumstances (e.g., fine textured soils in humid climates) then relative to the permanent pasture datum, soil aeration may decline within a three- or four-year period.^[9] In the world's major cropping areas, tillage of pastureland is an infrequent occurrence; so a more interesting, if difficult, question is: what are the comparative effects of different tillage practices on soil aeration?

Accurately quantifying the effects of surface tillage on aeration has usually proved difficult. There are issues of both methodology and interpretation to contend with. For example, it has been argued that values for oxygen concentration in the seedbed say little about the quality of the soil air, or its rate of renewal where the roots or organisms are located, and that these factors are likely to be of greater importance than the average level for the whole soil.^[10] Also, some plants can compensate for low soil aeration by the supply of oxygen from one part of the root systems to another via internal diffusion. Another example relates to scale; what effect does a tillage practice have on intra-aggregate aeration in contrast to changes in aeration in the pore space between structural units where roots easily penetrate?

Interestingly, use of the ODR meter, arguably the best technique for field measurement of aeration status, is often unable to routinely and consistently discriminate between tillage treatments. This measure seems better able to reflect wet soil conditions associated with high rainfall and impeded drainage.^[11,12] Yet a significant relationship between ODR and soybean yield has been observed, suggesting that it is only through critical stages of growth that low ODR will negatively impact on crop yield.^[12] Perhaps there are some implications here for the *in situ* monitoring of the effects of tillage on aeration.

The difficulties mentioned above, and the sometimes conflicting research results that have been reported, mean that caution is required when attempting to draw broad conclusions about the effect of surface tillage on soil aeration. The impacts vary dramatically depending on: spatial heterogeneity, the depth in the soil profile under consideration (i.e., above or beneath the depth corresponding to plowing), soil type, the type or configuration of implements, climate, the cropping history of the field, the crop, the crop rotation, and the competency of the operator, especially regarding the timeliness of field work. The quantity and quality of crop residue may influence soil aeration; this aspect has received relatively little attention. Furthermore, interactions between aeration and other soil and crop properties make predictions about the exact effect of tillage on aeration risky. For example, aeration status may interact with earthworm populations and residue levels to effect seedling emergence.^[13]

The above discussion notwithstanding, most published literature has shown that surface soil is better aerated under conventional than other forms of tillage or that plowing enhances surface soil aeration to a greater extent than no tillage.^[14–16] The soil disturbance or loosening associated with tillage implements that invert or mix soil increases air movement in surface soil. In structurally degraded and/or fine textured soil, this increase in aeration may be confined to the inter-aggregate pore space. In an Indian study, tillage gave greater ODR than untilled (control) areas, and the greatest ODR was under moldboard

plow and it was lowest with zero tillage.^[7] Likewise, in a compacted soil in New Zealand, rotary tillage of the soil surface improved ODR rates at 5-, 10-, and 15-cm depths compared with no-tillage.^[11] Enhanced rates of soil carbon oxidation are further evidence of improved aeration under cultivation.

The advantage that conventional tillage affords surface soil aeration may be somewhat transitory in nature both within seasons and across years. Enhanced soil aeration in a conventionally tilled seedbed may be short-lived as a result of soil reconsolidation,^[7] and may disappear by the time the crop becomes well established. The advantage of conventional tillage is likely to be more prominent early in the cropping cycle of a field: after many years of continuous cropping, no-tillage may be more beneficial to aeration status.^[17]

Although conventional tillage may generate more short-term macropores in the surface soil than no-tillage, they may not be as continuous down the profile and may be more tortuous.^[18–20] Therefore, no-tillage may improve aeration more uniformly throughout the soil profile particularly in the horizon just below the depth that corresponds to “plow level.” Where sufficiently detailed measurements have been made, it would appear that not only may air permeability be more continuous or uniform down the profile under no-till, but it may also be more constant across “in rows” and between row positions.^[19]

Implement type will also have an important influence on the effect of tillage on soil aeration. In a summary of a number of New Zealand studies, the effects of some different drill openers on soil aeration are discussed.^[13] They demonstrate the benefits of some drill configurations (e.g., inverted T) relative to others (e.g., triple disk) and illustrate the importance of soil aeration to seedling establishment. In part, differences in tillage equipment, particularly direct-drilling machinery, may account for much of the disparity in the literature on the effects of tillage on aeration. For example, permeability measurements made under no-till treatments established using a triple disc are nearly always markedly inferior to those made under plowing.^[21]

EFFECTS OF SUBSURFACE TILLAGE

Subsurface tillage is practiced to improve aeration deeper in the profile where soils are naturally compact and/or have poor internal drainage. Where deep tillage is employed to improve drainage, it is important that there is an outlet for excess water or deep tillage may exacerbate aeration problems. In a study comparing deep tillage implements, the paraplow (angled shanks operating at 0.5-m depth) achieved more consistent improvement in air permeability at sowing, particularly in the important depth of 15–25 cm, than did subsoilers with straight shanks tilling at either shallow (0.25 m) or deeper (0.5 m).^[11] In addition, all

subsurface tillage treatments increased ODR values to a depth of 40 cm compared with the control. Due to reconsolidation in this humid climate, these advantages had disappeared by harvest.

The hypothesis that, as an alternative to conventional tillage, subsoiling in combination with no-tillage seeding would improve soil aeration, other physical properties, and crop yield in a soil that has become severely compacted following many years of cultivation has been tested.^[11] Some aspects of this study and a subsequent investigation^[22] suggest that, for these New Zealand soils and conditions, despite the significant loosening achieved at depth by subsoiling implements, it was direct seeding that was the important component in the proposed system, and subsoiling contributed or added relatively little to the enhancement of crop yield. In part, this is due to the short-lived nature of the structural improvements generated by subsoiling.^[23]

TILLAGE, AERATION, AND GREENHOUSE GASES

In soil aeration research, there has been a shift in focus away from the study of the movement and use of oxygen and its effects on crop nutrition and growth to the role of soil aeration in environmental protection, e.g., greenhouse gas production and the susceptibility of nutrients to leaching. Indeed, concern about, and investigations of, the relationships between tillage and fluxes, composition and transformations of air into and out of soil are prominent on the environmental agenda. Of particular interest is the effect of tillage practices on the release of the major greenhouse gases: carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O).^[5,24] In addition, there is the suggestion that conservation tillage practices may enhance carbon sequestration.^[25]

Relative to no-tillage, conventional cultivation may result in greater CO₂ fluxes from the soil.^[26] No-tillage may increase the frequency of N₂O emissions and the net fixation of carbon by decreasing CO₂ emissions.^[5,27] Periods of low or zero CO₂ fluxes and very high N₂O fluxes under no-tillage may be associated with periodic reduced gas diffusivity and air-filled porosity, both of which are often caused by heavy rainfall. In general, peak N₂O emissions are mainly associated with heavy rainfall following fertilization particularly with no-tilled soils.^[5] Plowing may decrease the oxidation rate of atmospheric CH₄ in aerobic soils, but this is unlikely to be sufficient to offset the lower N₂O emissions from the soil.^[5]

CONCLUSIONS

Adequate soil aeration is important to crop production and environmental protection. Tillage may profoundly influence soil aeration. Although the effect of tillage on soil

aeration is dependent on a range of factors and is, therefore, hard to predict, often surface soil aeration is more favorable under well-managed conventional cultivation than no-tillage. Often, these improvements are confined to shallow soil depths and relatively short-time intervals. Where appropriate no-till equipment is used, direct-drilling may have greater potential to improve soil aeration, both in terms of uniformity throughout the soil profile and permanence in time. Tillage practices that result in anaerobic soil conditions are likely to promote N₂O production, while those that disturb larger volumes of soil and promote aeration may result in greater emissions of CO₂.

REFERENCES

1. Glinski, J.; Stepniewski, W. *Soil Aeration and Its Role for Plants*; CRC Press Inc.: Boca Raton, 1985; 229 pp.
2. Scott, H.D. *Soil Physics: Agricultural and Environmental Applications*; Iowa State Press: Ames, 2000; 421 pp.
3. Lipiec, J.; Stepniewski, W. Effects of soil compaction and tillage systems on uptake and losses of nutrients. *Soil Till. Res.* **1995**, *35*, 37–52.
4. Letey, J.; Stolzy, L.H.; Valoras, N.; Szuszkiewicz, T.E. Influence of soil oxygen on growth and mineral concentration of barley. *Agron. J.* **1962**, *54*, 538–540.
5. Ball, B.C.; Scott, A.; Parker, J.P. Field N₂O, CO₂ and CH₄ fluxes in relation to tillage, compaction and soil quality in Scotland. *Soil Till. Res.* **1999**, *53*, 29–39.
6. Francis, G.S.; Cameron, K.C.; Kemp, R.A. A comparison of soil porosity and solute leaching after six years of direct drilling or conventional cultivation. *Aust. J. Soil Res.* **1988**, *26*, 637–649.
7. Khan, A.R. Influence of tillage on soil aeration. *J. Agron. Crop Sci.* **1996**, *177* (4), 253–259.
8. Hodgson, A.S.; MacLeod, D.A. Oxygen flux, air-filled porosity and bulk density as indices of vertisol structure. *Soil Sci. Soc. Am. J.* **1989**, *53*, 540–543.
9. Shepherd, T.G.; Dando, J.L. Physical indicators of soil quality for environmental monitoring. In *Proceedings of Soil and Land Indicators*; Hawkes Bay Regional Council New Zealand Technical Report EMT 96/3; 33–42.
10. Dexter, A.R. Physical properties of tilled soils. *Soil Till. Res.* **1997**, *43*, 41–63.
11. Sojka, R.E.; Horne, D.J.; Ross, C.W.; Baker, C.J. Subsoiling and surface tillage effects on soil physical properties and forage oat stand and yield. *Soil Till. Res.* **1997**, *40*, 125–144.
12. Flowers, M.D.; Lal, R. Axle load and tillage effects on soil physical properties and soybean grain yield on a mollic ochraqualf in northwest Ohio. *Soil Till. Res.* **1998**, *48* (1–2), 21–35.
13. Baker, C.J.; Chaudhary, A.D.; Springett, J.A. Barley seedling establishment and infiltration from direct drilling in a wet soil. *Proc. Agron. Soc. New Zealand* **1987**, *17*, 59–66.
14. Ball, B.C.; O'Sullivan, M.F.; Hunter, R. Gas diffusion, fluid flow and derived pore continuity indices in relation to vehicle traffic and tillage. *J. Soil Sci.* **1988**, *3*, 327–339.

15. Ball, B.C.; Campbell, D.J.; Douglas, J.T.; Henshall, J.K.; O'Sullivan, M.F. Soil structural quality, compaction and land management. *Eur. J. Soil Sci.* **1997**, *48*, 593–601.
16. Rasmussen, K.J. Impact of ploughless soil tillage on yield and soil quality: A Scandinavian review. *Soil Till. Res.* **1999**, *53*, 3–14.
17. Voorhees, W.B.; Lindstrom, M.J. Long-term effects of tillage method on soil tilth independent of wheel traffic compaction. *Soil Sci. Am. J.* **1984**, *48*, 152–156.
18. Boone, F.R.; Van der Werf, H.M.G.; Kroesbergen, B.; Ten Hag, B.A.; Boers, A. The effect of compaction of the arable layer in sandy soils on the growth of maize for silage I. Critical matric water potential in relation to soil aeration and mechanical impedance. *Neth. J. Agr. Sci.* **1986**, *34*, 155–171.
19. Herad, J.R.; Klavivko, E.J.; Mannering, J.V. Soil macroporosity, hydraulic conductivity and air permeability of silty soils under long-term conservation tillage in Indiana. *Soil Till. Res.* **1987**, *11*, 1–18.
20. Carter, M.R. Characterizing the soil physical conditions in reduced tillage systems for winter wheat on a fine sandy loam using small cores. *Can. J. Soil Sci.* **1992**, *72*, 395–402.
21. Schjonning, P.; Rasmussen, K.J. Soil strength and soil pore characteristics for direct-drilled and ploughed soils. *Soil Till. Res.* **2000**, *57*, 69–82.
22. Hamilton-Manns, M. The effects of no-tillage and subsoil loosening on soil physical properties and crop performance. In M.Sc. Thesis, Massey University, New Zealand, 1998.
23. Busscher, W.J.; Sojka, R.E. Enhancement of subsoiling effect on soil strength by conservation tillage. *T. ASAE* **1987**, *30* (4), 888–892.
24. Soane, B.D.; van Ouwerkerk, C. Implications of soil compaction in crop production for the quality of the environment. *Soil Till. Res.* **1995**, *35*, 5–22.
25. Lal, R. Residue management, conservation tillage and soil restoration for mitigating greenhouse effect by CO₂ enrichment. *Soil Till. Res.* **1997**, *43*, 81–107.
26. Reicosky, D.C.; Reeves, D.W.; Prior, S.A.; Runion, G.B.; Rogers, H.H.; Raper, R.L. Effects of residue management and controlled traffic on carbon dioxide and water loss. *Soil Till. Res.* **1999**, *52* (3–4), 153–165.
27. Aulakh, M.S.; Rennie, D.A.; Paul, E.A. Gaseous nitrogen losses from soils under zero-till as compared to conventional-till management systems. *J. Environ. Qual.* **1984**, *13* (1), 130–136.

Aggregates: Tensile Strength

Humberto Blanco-Canqui

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

A soil aggregate is a conglomeration of organic and inorganic particles that cohere to each other more than to the neighboring particles. Tensile strength of soil aggregates refers to the stress or force per unit area required to break an aggregate. It is the inherent ability of aggregates to resist the maximum disruptive force without being mechanically fractured along the weakest failure planes. Tensile strength reflects the amount of aggregation in a soil and is a dynamic mechanical property of aggregates changing readily with the wetting and drying cycle of the soil. Intra-aggregate porosity (within aggregate), intra-aggregate bonds, abundance of rupture planes, internal microcracks, aggregate size, clay content and mineralogy, the presence of cementing agents, and overall soil management determine the tensile strength. Aggregate strength depends on the macro- and microstructure of the soils and is a key indicator of soil structural attributes. When an aggregate ruptures in a systematic manner from macroaggregates into microaggregates, it indicates that aggregate hierarchy exists. When an aggregate ruptures directly into silt and clay particles, it indicates that the aggregate hierarchy does not exist. Tensile strength of individual aggregates along with the interaggregate porosity (between aggregates) defines the mechanical nature and behavior of the whole soil.

IMPORTANCE OF AGGREGATE STRENGTH

Tensile strength of soil aggregates is of fundamental importance to soil tillability and plant growth. Aggregate strength affects seedbed preparation, root growth, seedling emergence, activity of soil organisms, and organic matter decomposition.^[1] Soil aggregation and fragmentation depend on the strength of individual soil aggregates.^[2] Aggregate strength also influences the macroscale physical behavior of soils including erosion, infiltration, and permeability. However, the importance of aggregate strength to soil and crop management is often overlooked when dealing with soil-related problems. Furthermore, most soil properties are often determined on the bulk soil rather than on individual aggregates, which are the building units of soil structure.

AGGREGATE STRENGTH AND TILLAGE

Tensile strength of aggregates is a very sensitive indicator of changes in soil structure.^[1] Therefore, any mechanical disturbance of the soil is portrayed in the tensile strength of individual aggregates. Tillage implements exert large loads to fracture and crush aggregates, while vehicular traffic compacts the soil and increases the aggregate strength.

Tensile strength is an important factor affecting the choice of an appropriate tillage method designed to minimize soil disruption. Information about the strength of individual aggregates is useful in understanding the disruptive load exerted by tillage to break soil aggregates. The strength of aggregates is the lowest immediately after tillage and increases gradually following tillage as a result of rainfall-induced consolidation and reduced organic matter content.^[3] Aggregates created by tillage at moisture contents above the plastic limit have higher tensile strength than those created below the plastic limit.^[4] Decline in aggregate strength by tillage can compromise soil quality and long-term sustainability of agriculture. Changes in aggregate strength among different tillage, crop, and soil management practices are not well documented.

IMPACT OF AGGREGATE STRENGTH ON CROP AND ENVIRONMENT

The strength of individual aggregates in interaction with the whole soil affects crop production and environmental quality.^[5] Soil structure is a spatial integrator of all the heterogeneous mechanical properties of aggregates and is made up of individual ever-changing aggregates. Thus, aggregate strength is a useful indicator of the suitability of the soil

structural condition for favorable crop production. Soils with high aggregate strength can reduce seed germination and root growth, thereby reducing crop yield.^[1] High strength may also reduce the uptake of nutrients and water by plants because of reduced porosity and permeability.

Aggregate strength is also indirectly connected to important ecological and environmental processes. Because soil erodibility depends on aggregate strength, the magnitudes of soil erosion and runoff increase in compacted soils with high aggregate strength, whereas soils with low aggregate strength have favorable infiltration and drainage and low runoff. Intensive cultivation in wet soils disperses clay and, following tillage, forms strong aggregates, reducing soil permeability and thus increasing runoff and soil loss.^[5] Data on aggregate strength are useful to identify soils susceptible to compaction and erosion. Management problems related to crop production including soil compaction and plow-pan formation also depend on the integrated effect of aggregate strength. Aggregate strength is thus of paramount importance to the interactions between soil structure and plant growth for favorable seedbed conditions, crop production, and overall environmental quality.

RELATIONSHIP OF AGGREGATE STRENGTH TO SOIL PROPERTIES

There exists a strong relationship between aggregate strength and other soil properties, including moisture (Fig. 1) and texture.^[3] Aggregate strength decreases with increasing moisture content because the bonding strength of organic and mineral particles in aggregates is weakened by the increase in moisture content. Tensile strength of dry aggregates increases with increasing clay content.^[4] Frequently, soils with similar texture and moisture content may differ in their aggregate tensile strength because of differences in organic matter content, clay mineralogy, and soil solution.^[6] Aggregate strength can be sensitive^[7]

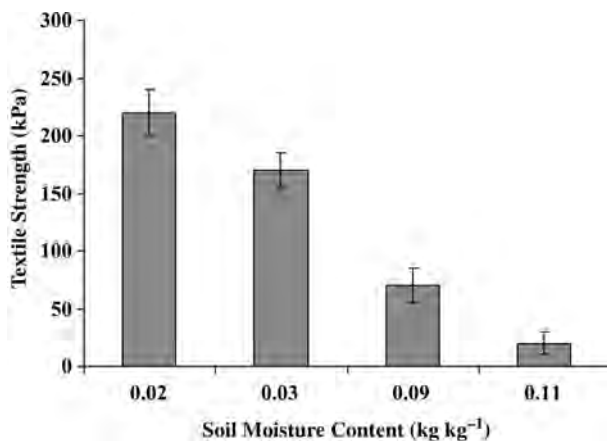


Fig. 1 Effect of soil moisture content on aggregate strength.

Source: From Horn & Dexter.^[3]

or insensitive^[1,5] to soil organic carbon (SOC) depending on soil and crop management (Table 1). Increase in SOC concentration may reduce the aggregate strength in the wet state but not in the dry state in which clay content may be the controlling mechanism.

Aggregate strength is also a function of aggregate size (Fig. 2). Strength often decreases with increasing size of aggregates because large aggregates have more planes of failure and weaker matrix points compared with small aggregates. The relationship between aggregate strength and aggregate size can be estimated using Eq. 1, known as the index of friability:

$$\log Y = -k \log V + A \quad (1)$$

where Y is the aggregate strength (kPa), A is the predicted log strength (kPa), V is the volume (mm³) of aggregates, and k is the index of friability.^[2] Wet aggregate strength of soils with high SOC is often higher than that in tilled soils because of greater organic bonding that reduces the slumping of aggregates in water. A simple equation relating aggregate strength to SOC is shown in Eq. 2.

$$\text{Aggregate strength} = a - b \cdot \text{SOC} \quad (2)$$

where a and b are the intercept and slope of the regression model, respectively. The negative slope indicates that aggregate strength decreases with SOC concentration.

Macropores reduce the strength of aggregates. Macropores, planes of failure, and weak spots are abundant in macroaggregates.^[4] The SOC-enriched macroaggregates have lower dry aggregate strength than SOC-depleted macroaggregates. Applied stress concentrates on air-filled macropores and microcracks, propagating these cracks until failure occurs. Aggregates with continuous pores are thus likely to fail more rapidly than those with tortuous pores. Aggregates tend to have higher tensile strength than bulk soil due to their increased bonding of contact points. Interaggregate contacts commonly exhibit weaker planes of failure than intra-aggregates.^[4] Aggregate strength is related to capillary forces, bonding energy between organic and inorganic particles, microbial processes, and soil solution dynamics. Aggregate strength is the result of interactive effects of soil properties and cannot be solely explained by a single soil property in the study by Horn and Dexter.^[3]

MEASUREMENT OF AGGREGATE STRENGTH

Tensile strength of aggregates can be determined using direct or indirect tension (crushing test) tests.^[8,9] The direct tension test quantifies the fracture of soil aggregates perpendicular to the force applied. A challenge in the use of the direct test is the difficulty in preparing samples. In addition, the direct test is more suitable for large aggregates or clods, which are either remolded or grooved into cups.^[2,8] Therefore, indirect tension test is commonly used to measure

Table 1 Relationship of aggregate strength to SOC for different soil types and management

Soil	Management	Strength (kPa)	SOC (g kg ⁻¹)	SOC pool (Mg ha ⁻¹)	References
Fine silty loam	Fallow	305	11	20	[5]
	Conventional tillage	270	15	26	
	Ley rotation	220	21	33	
	Reseeded grass	225	28	39	
	Permanent grass	215	32	44	
Silt loam	Soil + peat	260	0	0	[10]
		240	10	16	
		215	30	47	
		205	50	78	
		160	80	127	
Clay	Soil + peat	575	0	0	
		660	10	14	
		655	30	43	
		650	50	71	
		650	80	114	

Source: Adapted from Watts & Dexter^[5] and Zhang.^[10]

strength on individual aggregates. Simple apparatus^[10] or sophisticated equipment^[2,8] can be used for the crushing test. The indirect test consists of placing an aggregate between two parallel plates and then applying a known load of force at a constant rate of displacement across the aggregate until it fractures. The tensile strength, Y (kPa), is computed by using Eq. 3:^[7]

$$Y = k \frac{F}{d^2} \quad (3)$$

where F is the breaking force (N) applied to break the aggregate, d is the mean aggregate diameter (m), and k is a coefficient (~ 0.576). Aggregate diameter can be

estimated by 1) passing soil through sieves of known diameter; 2) measuring the longest and smallest diameter with a caliper; 3) calculating the geometric mean of the longest and smallest diameter; 4) adjusting the mean sieving diameter with aggregate mass; and 5) adjusting the mean sieving diameter with aggregate density.^[9] Eq. 3 assumes that the aggregates are homogeneous, are isotropic, and have a uniform deformation and elastic behavior.^[7]

Because tillage is usually performed under high soil moisture, aggregate tensile strength must be measured at moisture contents similar to those in the field at the time of tillage.^[2] Strength is often determined on oven-dried aggregates, which may not reflect the tensile failure of soil during the seedbed preparation. Indeed, drying soil aggregates at 105°C irreversibly alters soil properties. A better approach is to determine strength at known matric potentials by drying aggregates gradually from saturation in Büchner funnels and pressure plate apparatus to simulate field soil moisture dynamics.

CONCLUSION

Tensile strength of aggregates refers to their capacity to resist disruption and is highly sensitive to changes in soil structure. It is a dynamic indicator of soil structural behavior and is important to tillage operations and plant growth. The magnitude of failure of aggregates during tillage can be estimated by determining the tensile strength of individual aggregates. Tensile strength is a valuable indicator for the choice of an appropriate tillage method designed to minimize soil disruption. Aggregate strength affects seedling emergence, root growth, biological activity, SOC

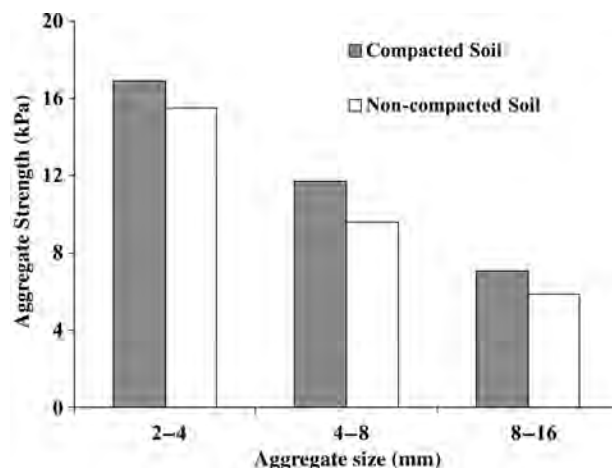


Fig. 2 Relationship of the aggregate strength to the size of aggregates.

Source: From Munkholm & Kay.^[2]

concentration, erosion, infiltration, runoff, and permeability of soils. It affects crop growth and environmental quality. Aggregate strength depends on soil moisture content, SOC concentration, size of aggregates, macroporosity, and clay mineralogy. The most commonly used method to determine the tensile strength of aggregates is the crushing test where failure of an individual aggregate is recorded under a load of force applied. Aggregate strength may not only differ from that of the bulk soil but can also determine the mechanical behavior of the whole soil.

REFERENCES

1. Causarano, H. Factors affecting the tensile strength of soil aggregates. *Soil Till. Res.* **1993**, *28*, 15–25.
2. Munkholm, L.J.; Kay, B.D. Effect of water regime on aggregate-tensile strength, rupture energy, and friability. *Soil Sci. Soc. Am. J.* **2002**, *66*, 702–709.
3. Horn, R.; Dexter, A.R. Dynamics of soil aggregation in an irrigated desert loess. *Soil Till. Res.* **1989**, *13*, 253–266.
4. Nearing, M.A.; Parker, S.C.; Bradford, J.M.; Elliot, W.J. Tensile strength of thirty-three saturated repacked soils. *Soil Sci. Soc. Am. J.* **1991**, *55*, 1546–1551.
5. Watts, C.W.; Dexter, A.R. The influence of organic matter in reducing the destabilization of soil by simulated tillage. *Soil Till. Res.* **1997**, *42*, 253–275.
6. Horn, R.; Baumgartl, T.; Kayser, R.; Baasch, S. Effect of aggregate strength on strength and stress distribution in structured soils. In *Soil Structure: Its Development and Function*; Hartge, K.H., Stewart, B.A., Eds.; CRC Press, Inc. & Lewis Publishers: Boca Raton, 1995; 31–51.
7. Rogowski, A.S.; Moldenhauer, W.C.; Kirham, D. Rupture parameters of soil aggregates. *Soil Sci. Soc. Am. Proc.* **1968**, *32*, 720–724.
8. Dexter, A.R.; Watts, C.W. Tensile strength and friability. In *Soil and Environmental Analysis: Physical Methods*; Smith, K.A., Mullins, C.E., Eds.; Marcel Dekker: New York, 2001; 405–433.
9. Dexter, A.R.; Kroesbergen, B. Methodology for determination of tensile strength of soil aggregates. *J. Agr. Eng. Res.* **1985**, *31*, 139–147.
10. Zhang, H. Organic matter incorporation affects mechanical properties of soil aggregates. *Soil Till. Res.* **1994**, *31*, 263–275.

Aggregation

Sjoerd W. Duiker

Pennsylvania State University, University Park, Pennsylvania, U.S.A.

Abstract

Soil aggregation is the organization and binding of soil particles into distinct units that can be separated by the application of energy to soil. When increasing levels of energy are applied, aggregates of most soils break down into smaller aggregates in a stepwise manner. Conceptual models have been formulated to explain this apparent hierarchy of soil aggregation. Different binding agents play roles at specific hierarchical levels. In this entry, the predominant model of aggregate hierarchy and known aggregating agents are discussed.

AGGREGATE HIERARCHY

A common model of aggregation distinguishes the following four levels:^[1–3]

1. Domains, quasicrystals (also called clay tactoids), and assemblages (<2 μm);
2. Clusters (2–20 μm);
3. Microaggregates (20–250 μm); and
4. Macroaggregates (250–2000 μm).

This classification is based on the research in temperate and Mediterranean regions and also seems to apply to moderately weathered tropical soils and volcanic ash soils.^[4] In highly weathered tropical soils with very stable microgranular structure, only one hierarchical level can be distinguished.^[2,5]

MECHANISMS OF AGGREGATE FORMATION

Roots and Vesicular–Arbuscular Mycorrhizal (VAM)

Both plant roots and VAM hyphae associated with roots are important binding agents at the scale of microaggregates and macroaggregates.^[6] Roots physically hold aggregates together, exert a drying effect, and may excrete organic substances that act as glues. The direct effect of roots on aggregation is high with perennial grass species due to the enmeshment of their extensive fine root systems with soil. Annual crops have smaller root systems and therefore a less positive direct effect on aggregation.^[1]

Fungi and Bacteria

Fungi have direct effect on aggregate stability, more than bacteria or actinomycetes. Fungal hyphae act as nets

holding microaggregates together, and fungi and bacteria produce polysaccharides that favor aggregation.^[7] Dead fungi and bacteria act as centers of clusters.^[1] Clay particles adhere to the walls of living fungal hyphae, apparently due to the gluing action of fungal exudates. In sandy soils, roots and fungal hyphae may be the only components responsible for aggregation because sand particles are large and inert and do not interact with charged organic ions.^[8]

Transient and Persistent Organic Matter

Transient organic matter is mostly composed of polysaccharides. Polysaccharides are produced by microorganisms or excreted by roots and bind clay-sized particles together into clusters and microaggregates.^[1] The effect of transient polysaccharides on aggregation is short-lived (generally less than 2 months). Persistent organic matter consists of highly humified organic molecules or polysaccharides protected against the decomposition by aluminosilicate clays or metal oxides.^[9] Persistent organic matter is an important aggregating agent at <2 μm level.

Earthworms

The casts of earthworms become highly stable aggregates that are resistant to slaking and dispersion. The casts contain more silt, clay, organic matter, nitrogen, and exchangeable nutrients and have a higher porosity and stability than the surrounding soil. Besides creating soil aggregates by their casts, earthworms create many pores in the soil as is explained elsewhere in the encyclopedia.

Termites

Termites bring large quantities of clay-sized particles from the subsurface to the surface of tropical soils to build their

mounds. The clay particles are glued together by a sticky fluid excreted by the termites.^[10] The microaggregates thus formed are 0.03–1 mm in diameter and are stable to dispersion by raindrop impact.^[5] Termite activity may be a contributor to the “pseudosand” or “pseudosilt” texture of clayey tropical soils.^[2,5]

Iron and Aluminum Oxides

Iron and aluminum (hydr)oxides are responsible for the very stable structure of many highly weathered tropical soils. However, there is limited evidence to support this conclusion. There is some evidence in which poorly crystalline iron oxides favor aggregation more than crystalline iron oxides.^[11–13] In some cases, aluminum oxides may be more important than iron oxides for aggregation.^[14] The role of these oxides needs more research before far-reaching conclusions can be drawn.

Inorganic Cations

Positively charged cations are attracted to negatively charged aluminosilicate clay surfaces and form bridges between the clay particles or between clay particles and organic matter.^[3] The role of cations in aggregation is primarily at the quasicrystal and domain scale. Differences between cations in their effects on clay flocculation can be attributed to their charge, concentration, size, and hydration.^[15] The ability to flocculate clays decreases in the following order: $\text{Al}^{3+} > \text{Ca}^{2+} > \text{Mg}^{2+} > \text{K}^+ > \text{Na}^+$.^[9]

Aluminosilicate Clays

Clay mineralogy determines to a large extent the basic soil structure of a soil. The most important clay minerals in soils are kaolinite, montmorillonite, and illite. Packs of montmorillonite clays have been termed “quasicrystals” (also called “clay tactoids”), those of illite “domains,” and those of kaolinite as “assemblages.”^[16] Clay particles in quasicrystals are more perfectly aligned than in domains, whereas they are randomly ordered in assemblages. Quasicrystals have a width/thickness ratio >100 , domains <10 , and assemblages <2 . Montmorillonite plates in quasicrystals can come apart when water molecules enter the interlamellar space, explaining the shrink–swell behavior of these clays.

Wetting and Drying and Freezing and Thawing

Wetting and drying cycles are needed to form aggregates. The effect of wetting and drying may be partly due to its effect on microbial activity and the orientation of clay platelets.^[17] Freezing and thawing cycles can break up cloddy soil if water contents are high enough

to separate soil particles. A limited number of freeze–thaw cycles (up to 4) can increase aggregate stability because of localized drying and precipitation of binding agents.^[18]

CONCLUSION

Most soils are built up of a hierarchy of aggregates. Different components determine aggregation at different levels. At the macro- and microaggregate levels, roots and fungal hyphae appear to be the primary factors responsible for aggregation. Clusters are composed of fungal hyphae, and bacterial remains encrusted with clay and silt particles. At the lowest level of aggregation ($<2 \mu\text{m}$), persistent organic matter and inorganic components are responsible for bonds between clay particles. Biological, chemical, and physical processes play a role in aggregation, including the crops grown, earthworms, termites, wetting and drying, and freeze–thaw cycles.

REFERENCES

1. Tisdall, J.M.; Oades, J.M. Organic matter and water-stable aggregates in soils. *J. Soil Sci.* **1982**, *33*, 141–163.
2. Oades, J.M.; Waters, A.G. Aggregate hierarchy in soils. *Aust. J. Soil Res.* **1991**, *29*, 815–828.
3. Edwards, A.P.; Bremner, J.M. Microaggregates in soils. *J. Soil Sci.* **1967**, *18*, 64–73.
4. Maeda, T.; Takenaka, H.; Warkentin, B.P. Physical properties of allophane soils. *Adv. Agron.* **1977**, *29*, 229–264.
5. Trapnell, C.G.; Webster, R. Microaggregates in red earths and related soils in east and central Africa, their classification and occurrence. *J. Soil Sci.* **1986**, *37*, 109–123.
6. Thomas, R.S.; Franson, R.L. Bethelenfalvay. Separation of vesicular—arbuscular mycorrhizal fungus and root effects on soil aggregation. *Soil Sci. Soc. Am. J.* **1993**, *57*, 77–81.
7. Chaney, K.; Swift, R.S. studies on aggregate stability. II. The effect of humic substances on the stability of re-formed soil aggregates. *J. Soil Sci.* **1986**, *37*, 337–343.
8. Bond, R.D.; Harris, J.R. The influence of the microflora on physical properties of soils. I. Effects associated with filamentous algae and fungi. *Aust. J. Soil Res.* **1964**, *2*, 111–122.
9. Emerson, W.W.; Greenland, D.J. Soil aggregates—Formation and stability. In *Soil Colloids and Their Association in Aggregates*; De Boodt, M.F., Hayes, M., Herbillon, A., Eds.; Plenum Press: New York, 1990; 485–511.
10. Jungerius, P.D.; Van Den Ancker, J.A.M.; Múcher, H.J. The contribution of termites to the microgranular structure of soils on the uasin gishu plateau, Kenya. *Catena* **1999**, *34*, 349–363.
11. Desphande, T.L.; Greenland, D.J.; Quirk, J.P. Changes in soil properties associated with the removal of iron and aluminum oxides. *J. Soil Sci.* **1968**, *19*, 108–122.

12. Schahabi, S.; Schwertmann, U. Der einfluß von synthetischen eisenoxiden auf die aggregation zweier Löß bodenhorizonte. *Z. Pflanzenernaehr. Bodenk.* **1970**, *125*, 193–204.
13. Duiker, S.W. *Soil Erodibility in Southern Spain and the Southern USA with Special Emphasis on the Role of Iron Oxides*. Ph.D. dissertation, The Ohio State University, Columbus.
14. El-Swaify, S.A.; Emerson, W.W. Changes in the physical properties of soil clays due to precipitated aluminum and iron hydroxides. I. Swelling and aggregate stability after drying. *Soil Sci. Soc. Am. Proc.* **1975**, *39*, 1056–1063.
15. Sumner, M.E. The electrical double layer and clay dispersion. In *Soil Crusting: Chemical and Physical Processes*; Sumner, M.E., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 1992; 1–31.
16. Quirk, J.P.; Aylmore, L.A.G. Domains and quasi crystalline regions in clay systems. *Soil Sci. Soc. Am. Proc.* **1971**, *35*, 652–654.
17. Utomo, W.H.; Dexter, A.R. Changes in soil aggregate water stability induced by wetting and drying cycles in non-saturated soil. *J. Soil Sci.* **1982**, *33*, 623–637.
18. Lehrs, G.A. Freeze–thaw cycles increase near-surface aggregate stability. *Soil Sci.* **1998**, *163*, 63–70.

Aggregation: Soil Organic Matter (SOM) and

Cynthia A. Cambardella

National Soil Tilth Laboratory, Ames, Iowa, U.S.A.

Abstract

Soil organic matter (SOM) consists of plant, animal, and microbial residues and organic decomposition products that are associated with the inorganic soil matrix. SOM turnover is coupled to the aggradation/degradation of soil aggregates through the activity of soil microorganisms. In order to understand how soil structure can mediate SOM turnover and formation, it is important to understand how soil primary particles and organic matter interact with each other.

INTRODUCTION

A soil aggregate is formed when closely packed sand, silt, clay, and organic particles adhere more strongly to each other than to the surrounding particles.^[1] The arrangement of these aggregates and the pore space between them is referred to as soil structure. Soil structural stability is the ability of aggregates to remain intact when exposed to different stresses.^[2]

AGGREGATE STRUCTURE

In the late 1950s, it was proposed that clay particles were organized into units called clay domains,^[3] which are defined as groups of clay crystals that behave in water as a single unit. Clay domains are joined to quartz particles through organic matter (OM)-mediated bonds, and these organomineral complexes can be further structured into larger discrete units held together by OM–polyvalent cation bridges. Based on the experiments using ultrasonic vibration to disperse soils, Edwards and Bremner^[4] suggested a model that classified aggregates into the following two groups based on their relative size and stability: macroaggregates (>250 μm) and microaggregates (<250 μm). They observed that the bonds that hold together macroaggregates were easily disrupted, but that microaggregates were quite stable and their structure was disrupted only by applying mechanical energy (sonication) or by chemical treatment. They further postulated that the OM contained within microaggregates is inaccessible to microbial attack and, therefore, stabilized against decomposition.

In 1982,^[5] two Australian scientists, Drs. Judy Tisdall and Malcolm Oades, presented a hierarchical conceptual model of aggregate formation and soil organic matter (SOM) turnover that further refined the ideas published in the late 1960s. The model was based on the observation that aggregates broke down in a stepwise fashion;

macroaggregates >250 μm broke down into microaggregates of 20–250 μm before particles <20 μm were released. These results have been confirmed by a number of researchers^[6–8] and provide the basis for the concept of aggregate hierarchy. The idealized model of Tisdall and Oades^[5] consists of three hierarchical levels of aggregates:

$$2-20\ \mu\text{m} \rightarrow 20-250\ \mu\text{m} \rightarrow >250\ \mu\text{m}$$

Microaggregates of 2–20 μm diameter may be formed by the flocculation of fine silt or clay particles. The flocculation of negatively charged clay particles is increased in the presence of polyvalent cations such as Al^{+3} and Ca^{+2} . OM may enhance aggregation at this level when it forms complexes with the clay and polyvalent cations. It is believed that the OM associated with aggregates <20 μm is primarily of microbial origin, and there is little evidence that any plant debris exists in aggregates of this size.^[9]

The next stage in the hierarchy is 20–250 μm microaggregates. The core of such aggregates may occasionally contain a fragment of plant debris; however, Oades and Waters^[9] reported that the cores of many of these aggregates, especially those between 20 and 90 μm , were void. They suggested that the empty core was due to the complete degradation of the plant debris they originally contained. The stability of this aggregate was maintained by the decomposition products of the organic core.

Macroaggregates (>250 μm in diameter) consist of microaggregates, primary particles, and particulate organic matter (POM) held together by a network of fine roots and vesicular arbuscular mycorrhizae (VAM) hyphae. The surface of the roots and hyphae is covered with extracellular polysaccharides to which microaggregates may become bound. Macroaggregate stability is related to the amount of POM carbon (C) in the soil,^[10] and aggregate formation and stabilization processes in soil are directly related to the decomposition of roots and the dynamics of root-derived POM C in soil.^[11]

The model of Tisdall and Oades^[5] is often interpreted to mean that microaggregates are gathered together by the ramifying effect of roots and stabilized by the enmeshing effect of roots and hyphae. It also suggests that microaggregates are very stable, and the OM associated with them is biologically recalcitrant. A number of researchers have presented an alternative view in which microaggregates are formed within existing macroaggregates.^[12–15] According to this theory, macroaggregates form first through the ramifying effect of plant roots. Fragments of OM, originating as small pieces of plant root, fungal hyphae, or other organic debris may be included in the macroaggregate structure. During the decomposition process, the organic fragment becomes encrusted with clay particles and microbially produced mucilage. The decomposition of the organic core may be retarded. The result is a newly formed, stable microaggregate.

AGGREGATE STABILIZATION AND ORGANIC BINDING AGENTS

Tisdall and Oades^[5] emphasized the importance of organic binding agents to soil structural stability and divided these agents into the following three broad classes based on their temporal persistence in the soil: persistent (humic material associated with polyvalent metal cations), transient (predominantly polysaccharides), and temporary (roots, root hairs, and fungal hyphae).

Humic substances are large, relatively immobile, aromatic molecules produced during the microbial decomposition of plant and animal residues. Persistent organic bonds occur when a water molecule forms a bridge between an anionic functional group (e.g., carboxyl or hydroxide) on the humic molecule and a polyvalent cation (Al^{+3} , Fe^{+3} , and Ca^{+2}) attached to the clay surface. Because of their chemical nature and chelation with polyvalent cations, humic substances are resistant to decomposition but not completely inert.

The origin of soil polysaccharides includes plant and animal tissue; microbial cells; and exudates secreted by roots, hyphae, and bacteria. Plant-originating polysaccharides may be easily decomposed by microorganisms, which in turn secrete polysaccharides themselves. Most polysaccharides are negatively charged, and polyvalent cations serve as a bridge between polysaccharides and clay particles. Unlike humic substances, polysaccharides can be rapidly metabolized by microorganisms; hence, they are considered to be transitory binding agents. If the binding effect of polysaccharides is to persist, they must be continually replenished, or they must be protected from decomposition.

Plant roots, root hairs, mycorrhizae, and fungal hyphae may also serve as temporary binding agents within and between aggregates. Roots form an extensive network throughout the soil, and as they grow, they may enmesh microaggregates and pull them together. In the rhizosphere,

the hyphae of VAM may contribute further to the aggregating effect as they grow into small pores and bind soil particles together.^[16] The hyphae of filamentous fungi associated with decomposing fragments of OM are likely to have a similar effect in the bulk soil.

A summary of the different binding agents and their role in soil aggregation is given in Table 1. It is generally agreed that the humic substances and polysaccharides are the primary organic binding agents for the formation of stable microaggregates. It is important to note that these binding agents are relatively immobile in the soil. The mobile component in the formation of microaggregates appears to be fine clay particles, which may be carried by water through soil pores toward plant roots by matrix or gravity suction. As the fine clays move, it appears that they align themselves parallel to and encrust roots or OM fragments. They may also surround fungal hyphae or encapsulate bacteria. Thus, it is envisioned that microaggregate formation occurs primarily in zones of high microbial activity because this is where the humic substances are being produced. Microaggregates formed in this way would have a core of plant-derived or microbially derived OM.

AGGREGATE STRUCTURE AND OM DECOMPOSITION

In general, it is thought that the humified OM that stabilizes microaggregates is fairly recalcitrant, while the temporary and transient binding agents are more easily mineralized.

Table 1 Summary of the major organic binding agents in the stabilization of aggregates.

Aggregating agents	Aggregation process	Major scale of aggregation
Humic substances	Form strong bonds with soil mineral components	Basis of microaggregate formation
Polysaccharides	Act as gelatinous gluing agents Form organomineral associations	Involved in the stabilization of both micro- and macroaggregates
Plant roots	Enmesh soil aggregates Exude polysaccharides	Agents of macroaggregate formation and short-term binding
Fungal hyphae	Enmesh soil aggregates Exude polysaccharides	Agents of macroaggregate formation and short-term binding
Earthworms	Mix OM and clay colloids together Mix decaying detritus with the bulk soil	Agents of macroaggregate formation

Source: From Haynes & Beare.^[17]

The actual rate at which organic materials are mineralized, however, depends not only on their chemical qualities but also on their availability to microorganisms. The OM bound within soil aggregates may be physically or chemically protected from decomposition.

The physical protection of OM occurs when physical barriers exist between otherwise decomposable substrates and decomposers. The primary evidence that aggregate structure physically protects OM from microbial decomposition is provided by studies in which aggregates are crushed or ground. C and nitrogen mineralization rates increase greatly when the aggregate structure is disrupted. This is attributed to the exposure of OM, which was previously inaccessible to microbial attack.

Chemical stabilization occurs by the chemical interaction of substrate with the mineral part of the soil. As polysaccharides dry, they become denatured and are more chemically resistant to decomposition. In addition, many aggregates are composed of organic cores encrusted by clay particles. During the decomposition process, oxygen is consumed and the cores may eventually become anaerobic. Under these conditions, the solubility of polyvalent cations (Cu^{+2} , Fe^{+3} , Zn^{+2} , and Ca^{+2}) may increase with the result that polysaccharides form complexes with the polyvalent cations more easily. These structures may protect polysaccharides from decomposition.

Differences in the relative stability of macro- and microaggregates are a function of their relative size, as well as the location and lability of the organic binding agents within the aggregates. Because the OM binding microaggregates together is generally assumed to be recalcitrant and in smaller pores that cannot be accessed by microorganisms, it is expected that OM associated with microaggregates will decompose more slowly than that associated with macroaggregates. A number of research studies have supported the assertion that SOM decomposition rates are higher in macroaggregates than in microaggregates for a variety of soils and land use management protocols.^[6,14,18,19]

REFERENCES

1. Martin, J.P.; Martin, W.P.; Page, J.B.; Raney, W.A.; DeMent, J.D. Soil aggregation. *Adv. Agron.* **1955**, *7*, 1–37.
2. Kay, B.D.; Angers, D.A.; Baldock, J.A.; Groenevelt, P.H. Quantifying the influence of cropping history on soil structure. *Can. J. Soil Sci.* **1988**, *68*, 359–368.
3. Emerson, W.W. The structure of soil crumbs. *J. Soil Sci.* **1959**, *10*, 235–244.
4. Edwards, A.P.; Bremner, J.M. Microaggregates in soils. *J. Soil Sci.* **1967**, *18*, 64–73.
5. Tisdall, J.M.; Oades, J.M. Organic matter and water-stable aggregates in soil. *J. Soil Sci.* **1982**, *33*, 141–163.
6. Elliott, E.T. Aggregate structure and carbon, nitrogen and phosphorus in native and cultivated soils. *Soil Sci. Soc. Am. J.* **1986**, *50*, 627–633.
7. Cambardella, C.A.; Elliott, E.T. Carbon and nitrogen distribution in aggregates from cultivated and native grassland soils. *Soil Sci. Soc. Am. J.* **1993**, *57*, 1071–1076.
8. Angers, D.A.; Giroux, M. Recently deposited organic matter in soil water-stable aggregates. *Soil Sci. Soc. Am. J.* **1996**, *60*, 1547–1551.
9. Oades, J.M.; Waters, A.G. Aggregate hierarchy in soils. *Aust. J. Soil Res.* **1991**, *29*, 815–828.
10. Cambardella, C.A.; Elliott, E.T. Particulate soil organic matter changes across a grassland cultivation sequence. *Soil Sci. Soc. Am. J.* **1992**, *56*, 777–783.
11. Gale, W.J.; Cambardella, C.A.; Bailey, T.B. Root-derived carbon and the formation and stabilization of aggregates. *Soil Sci. Soc. Am. J.* **2000**, *64*, 201–207.
12. Oades, J.M. Soil organic matter and structural stability: Mechanisms and implications for management. *Plant Soil* **1984**, *76*, 319–337.
13. Elliott, E.T.; Coleman, D.C. Let the soil work for us. *Ecol. Bull.* **1988**, *39*, 23–32.
14. Beare, M.H.; Cabrera, M.L.; Hendrix, P.F.; Coleman, D.C. Aggregate-protected and unprotected organic matter pools in conventional- and no-tillage soils. *Soil Sci. Soc. Am. J.* **1994**, *58*, 787–795.
15. Gale, W.J.; Cambardella, C.A. Carbon dynamics of surface residue- and root-derived organic matter under simulated no-till. *Soil Sci. Soc. Am. J.* **2000**, *64*, 190–195.
16. Miller, R.M.; Jastrow, J.D. Hierarchy of root and mycorrhizal fungal interactions with soil aggregation. *Soil Biol. Biochem.* **1990**, *22*, 579–584.
17. Haynes, R.J.; Beare, M.H. Aggregation and organic matter storage in meso-thermal, humid soils. In *Structure and Organic Matter Storage in Agricultural Soils*; Carter, M.R., Stewart, B.A., Eds.; Lewis Publishers, CRC Press, Inc.: Boca Raton, 1996; 213–262.
18. Gupta, V.V.S.R.; Germida, J.J. Distribution of microbial biomass and its activity in different soil aggregate size classes as affected by cultivation. *Soil Biol. Biochem.* **1988**, *20*, 777–786.
19. Jastrow, J.D.; Boutton, T.W.; Miller, R.M. Carbon dynamics of aggregate-associated organic matter estimated by carbon-13 natural abundance. *Soil Sci. Soc. Am. J.* **1996**, *60*, 801–807.

Aggregation: Structural Stability Measurement

Edmund Perfect

Department of Agronomy, University of Kentucky, Lexington, Kentucky, U.S.A.

Martín Díaz-Zorita

Experimental Agricultural Station, National Institute of Technical Agriculture (EEA INTA), General Villegas, Argentina

John Grove

University of Kentucky, Lexington, Kentucky, U.S.A.

Abstract

This entry focuses on fragmentation procedures and associated indices for assessing soil aggregation on the basis of the size distribution and stability of fragments after mechanical disruption. When choosing between fragment size distribution and structural stability, procedures will depend on the use to be made of the indices. In general, if the procedure is to characterize the field soil condition without consideration of disruptive processes, then measurement of soil fragment size distribution after low-energy input is recommended. Structural stability measurements are most useful for characterizing soil susceptibility to wind and water erosion. In determining the stability of the soil to mechanical disruption, it is important to measure the distribution of soil fragments relative to an initial undisturbed state.

INTRODUCTION

Soil aggregation can be viewed as the arrangement of primary soil particles into hierarchical structural units, which is identified on the basis of varying failure zone strengths reflecting the characteristics of both the void and the solid phases.^[1]

Direct characterization of soil aggregation can be done by describing morphological features in the field,^[2] using image analysis techniques,^[3] or by measuring the size distribution and connectivity of pores.^[4] Other procedures are based on the partial breakdown of structural units by dispersion or fragmentation and evaluation of the resulting fragment size distribution. In soils, where dispersion–flocculation processes dominate the arrangement of secondary soil units, the use of fragmentation procedures to assess soil aggregation is less precise than the use of dispersion methods.^[5] This entry focuses on fragmentation procedures and associated indices for assessing soil aggregation on the basis of the size distribution and stability of fragments after mechanical disruption.

Aggregates, Fragments, and Particles

Aggregate size and stability are interrelated concepts in the description of soil structure using fragmentation procedures. Assuming separation of aggregates occurs at planes

of weakness surrounding coherent structural units, application of mechanical stress results in soil breakdown into fragments with stabilities greater than the applied stress (Fig. 1). With application of low-energy stress, the resulting fragments will be similar in size to those naturally occurring aggregates. However, with increasing fragmentation energy, the rate of reduction in fragment size will be related to structural stability (bonding energy) between aggregation units (clay packages, microaggregates, etc.). With application of high-energy stress, the resulting fragment size distribution will be independent of further increments in stress and more closely related to the sizes of primary soil particles (e.g., sand, silt, and clay) than the aggregates. Soil fragmentation rarely implies complete disruption of aggregates and can be performed under dry, moist, or saturated conditions.

SAMPLING AND SAMPLE PREPARATION

The processes of breaking down involved in fragmentation methodologies are continuous, commencing at sampling and continuing until the final size distribution of fragments is determined. Thus, the purpose of any aggregation measurement must be well understood before sampling to determine the sampling and handling procedures to be employed. Recommended tools for taking undisturbed samples are shovels or soil core samplers.^[6,7] Samplers

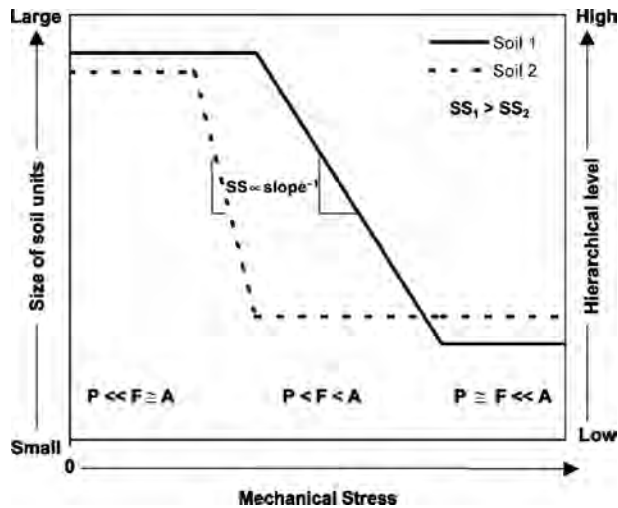


Fig. 1 Idealized changes in soil fragment size distribution as a function of applied mechanical stress. P, median particle size; F, median fragment size; A, median aggregate size; SS, soil structural stability. The subscript indicates the soil type 1 or 2.

with low wall area to volume ratios minimize compression. Air-drying of samples without initial fragmentation promotes the formation of clods, which may not be present in the natural state, particularly when the sampled layer does not dry excessively under field conditions. Low-energy fragmentation of moist samples before air-drying is common. “Gentle” hand manipulation is widely used, but it provides a nonmeasurable energy input that is strongly operator dependent. We favor the application of a standardized low-energy fragmentation procedure such as drop shatter.^[8] Fragmentation and/or compaction can continue during transportation and storage, so adequately sized containers are required. Cohesion of soil increases with drying/storage time.^[9] To be comparable, samples should be treated in the same manner and measurements to be done on samples with the same water potential whenever possible.^[10]

FRAGMENTATION PROCEDURES

Dry Sieving

Dry fragment separation is done with a nested set of rotary sieves or with vibratory or oscillatory movements imposed on a nest of flat sieves. Rotary sieving is accepted as the standard for dry aggregate size distribution measurements for wind erosion assessment.^[11,12] When flat sieves are used, it is important to determine the sieving duration to avoid fragment abrasion. Several studies recommend 30 seconds of vibratory, horizontal, or horizontal-vertical sieving for adequate fragment separation with minimal abrasion.^[13]

Wet Sieving

Wet sieving refers to the separation of soil fragments from an undisturbed sample or a single initial size fraction in the presence of free water or other liquids. Methods that involve sieving a single initial fragment size fraction are preferred because they are less time consuming. However, the selection of appropriately sensitive initial fragment sizes varies among soils. A better estimate of water-stable fragments is obtained by sieving an undisturbed sample over a nest of sieves. The soil water content prior to immersion is a major factor in determining wet aggregate stability. Based on the wetting procedure, different mechanisms of aggregate breakdown can be identified: slaking owing to the fast wetting of dry aggregates, micro-cracking after slow wetting, and mechanical breakdown of prewetted aggregates.^[14] The use of “field moist” samples introduces another variable that may lead to erroneous conclusions.^[15]

Energy Input

Both dry and wet sieving subject soil to forces that are rarely found in the field and which cannot be easily quantified. Several attempts to quantify soil fragmentation energy, on the basis of ultrasonic dispersion or drop shatter, have been described.^[8,16]

SIZE DISTRIBUTION INDICES

The mean weight diameter is a commonly used index for expressing aggregation with a single value.^[17,18] The literature also contains a variety of functions to parameterize the measured mass size distribution of soil fragments (Table 1). Indices based on a single fragment size class have been related to soil processes (e.g., wind erosion^[11]) but are not useful in describing fragment size distributions. When performing the calculations, special care must be taken to avoid inclusion of primary particles (e.g., sand) with sizes similar to that of the measured fragments. Fragment sizes rarely follow a normal distribution, so parameters from the log-normal or Rosin-Rammler distribution models are preferred. A new approach, using fractal theory, has been proposed.^[19]

SOIL STRUCTURAL STABILITY

Soil structural/aggregate stability is a measure of the ability of the soil structural units to resist change (retain structure) in response to application of a mechanical stress (i.e., dry or moist fragmentation, disruption in water, etc.). Although there are a great variety of methods to assess aggregate stability, the ratio or difference in the fragment sizes before and after application of mechanical energy is generally used (Table 2). Defining the low- and high-energy

Table 1 Indices for soil fragment size distribution quantification.

Distribution	Function	Parameters		
		Reference size	Spread of distribution	References
Normal	$P(x < X) = \frac{1}{\sigma\sqrt{2\pi}} \exp \left[-\frac{(x-MD)^2}{2SD^2} \right]$	MD	SD	[20,21]
Log-normal	$P(x < X) = \frac{1}{\log GSD \sqrt{2\pi}} \exp \left[-\frac{(\log(x) - GMD)^2}{2(\log GSD)^2} \right]$	GMD	log GSD	[20]
Rosin–Rammler	$P(x < X) = \exp \left[-\left(\frac{x}{\alpha}\right)^\beta \right]$	α	β	[22]
Gaudin–Schuhmann	$P(x < X) = \left(\frac{x}{x_0}\right)^m$	x_0	m	[23]

Notes: MD, mean diameter; GMD, geometric mean diameter; α , fragment diameter corresponding to the 36.78 percentile of the cumulative probability function; x_0 , diameter of the largest fragment; SD, standard deviation; log GSD, log of the geometric standard deviation; β , Rosin–Rammler exponent; m, Gaudin–Schuhmann exponent. $P(x < X)$ is the fraction of soil structural units smaller than sieve size X.

input levels requires special attention because fragment size depends on both energy input and structural stability (Fig. 1). Aggregate stability indices differ from indices used to describe the fragment size distribution because the former reflects the extent of breakdown relative to an initial condition, while the latter is independent of the soil's initial state.

Structural stability indices are empirical, and comparisons among treatments, soil properties, and/or processes have meaning only when similar procedures are used. The

choice of stability index should be carefully considered in relation to the type(s) of comparison desired.

CONCLUSION

When choosing between fragment size distribution and structural stability, procedures will depend on the use to be made of the indices. In general, if the procedure is to characterize the “field” soil condition without consideration of disruptive processes, then measurement of soil fragment size distribution after low-energy input is recommended. Structural stability measurements are most useful for characterizing soil susceptibility to wind and water erosion. In determining the stability of the soil to mechanical disruption, it is important to measure the distribution of soil fragments relative to an initial undisturbed (low-energy input) state.

REFERENCES

- Kay, B.D.; Angers, D.A. Soil structure. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 1999; 229–276.
- Manichon, H. Observation morphologique de l'état structural et mise en évidence D'effets De Compactage des horizons travaillés. In *Soil Compaction and Regeneration*; Monnier, G., Goss, M.J., Eds.; Balkema: Rotterdam, 1987; 39–52.
- Dexter, A.R. Advances in characterization of soil structure. *Soil Till. Res.* **1988**, *11*, 199–238.
- Wilson, G.V.; Luxmoore, R.J. Infiltration, macroporosity, and mesoporosity distributions on two forested watersheds. *Soil Sci. Soc. Am. J.* **1988**, *52*, 329–335.
- Emerson, W.W. Inter-particle bonding. In *Soils: An Australian Viewpoint*; The Division of Soils CSIRO, Ed.; CSIRO/Academic Press: Melbourne, 1983; 477–498.
- McIntyre, D.S. Soil sampling techniques for physical measurements. In *Methods for Analysis of Irrigated Soils*; Loveday, J., Ed.; Commonwealth Agricultural Bureaux: Victoria, 1974; 12–20.

Table 2 Summary of soil structural/aggregate stability indices for fragmentation procedures.

Index	Equation	Specific conditions	References
ΔMWD	A–B	Dry and wet mean weight diameters	[24]
Henin index, I_s	A/B	Water, ethanol, and benzene stable aggregates	[25]
Stability index	B/A	Water stable aggregates ≥ 3.0 mm	[26]
WAS and DCF	B/A	Water stable aggregates and dispersible clay in 1–2 mm fraction	[27]
Stability index	B/A	Water stable aggregates ≥ 0.25 mm	[7]
Soil erodibility	B/A	Dry stable aggregates ≥ 0.84 mm	[11]
Dry structural stability	B/A	Mass fraction distribution after 1 and 5 minutes of sieving	[28]
Stability constant (k)	$(\ln B - \ln A)/E_s$	Ultrasonic dispersion	[16]

Note: ΔMWD, change in mean weight diameter; WAS, wet aggregate stability; DCF, dispersible clay fraction (dispersible clay/total clay); A, size of fragments under low energy stress; B, size of fragments under high-energy stress; E_s , sonic energy level.

7. Kemper, W.D.; Rosenau, R.C. Aggregate stability and size distribution. In *Methods of Soil Analysis. Part 1. Physical and Mineralogical Methods*; Klute, A., Ed.; American Society of Agronomy and Soil Science Society of America: Madison, 1986; 425–442.
8. Hadas, A.; Wolf, D. Refinement and re-evaluation of the drop-shatter soil fragmentation method. *Soil Till. Res.* **1984**, *4*, 237–249.
9. Kemper, W.D.; Rosenau, R.C. Soil cohesion as affected by time and water content. *Soil Sci. Soc. Am. J.* **1984**, *48*, 1001–1006.
10. Gollany, H.T.; Schumacher, T.E.; Evenson, P.D.; Lindstrom, M.J.; Lemme, G.D. Aggregate stability of an eroded and desurfaced typic argiustoll. *Soil Sci. Soc. Am. J.* **1991**, *55*, 811–816.
11. Chepil, W.S. Improved rotary sieve for measuring state and stability of dry soil structure. *Soil Sci. Soc. Am. Proc.* **1952**, *16*, 113–117.
12. Zobeck, T.M. Soil properties affecting wind erosion. *J. Soil Water Conserv.* **1991**, *46*, 112–118.
13. Braunack, M.V.; McPhee, J.E. The effect of initial soil water content and tillage implement on seedbed formation. *Soil Till. Res.* **1991**, *20*, 5–17.
14. Le Bissonnais, Y. Soil characteristics and aggregate stability. In *Soil Erosion, Conservation, and Rehabilitation*; Agassi, M., Ed.; Marcel Dekker: New York, 1996; 41–60.
15. Yoder, R.E. A direct method of aggregate analysis of soils and a study of the physical nature of erosion losses. *J. Am. Soc. Agron.* **1936**, *28*, 337–351.
16. Fuller, L.G.; Goh, T.B. Stability–energy relationships and their application to aggregation studies. *Can. J. Soil Sci.* **1992**, *72*, 453–466.
17. Puri, A.N.; Puri, B.R. Physical characteristics of soils. II. Expressing mechanical analysis and state of aggregation of soils by single values. *Soil Sci.* **1939**, *47*, 77–81.
18. Youker, R.E.; McGuinness, J.L. A short method of obtaining mean weight–diameter values of aggregate analyses of soils. *Soil Sci.* **1957**, *83*, 291–294.
19. Perfect, E. Fractal models for the fragmentation of rocks and soils: a review. *Eng. Geol.* **1997**, *48*, 185–198.
20. Gardner, W.R. Representation of soil aggregate size distribution by a logarithmic–normal distribution. *Soil Sci. Soc. Am. Proc.* **1956**, *20*, 151–153.
21. Allen, T. *Particle Size Measurement. Volume 1. Powder Sampling and Particle Size Measurement*; Chapman & Hall: Cornwall, 1997; 525 pp.
22. Irani, R.R.; Callis, C.F. *Particle Size: Measurement, Interpretation, and Application*; Wiley: New York, 1963; 165 pp.
23. Gaudin, A.M.; Meloy, T.P. Model and comminution distribution equation for single fracture. *T. AIME* **1962**, *223*, 40–43.
24. De Leenheer, L.; De Boodt, M. Determination of aggregate stability by the change in mean weight diameter. In *Proceedings of the International Symposium on Soil Structure*; Rijkslandouwhogeschool: Ghent, 1959; 290–300.
25. Henin, S.; Monnier, G.; Combeau, A. Methode pour l'étude de la stabilité structurale des sols. *Ann. Agron.* **1958**, *9*, 71–90.
26. Bryan, R.B. The development, use and efficiency of indices of soil erodibility. *Geoderma* **1968**, *2*, 5–26.
27. Pojasok, T.; Kay, B.D. Assessment of a combination of wet sieving and turbidimetry to characterize the structural stability of moist aggregates. *Can. J. Soil Sci.* **1990**, *70*, 33–42.
28. Buschiazzo, D.E.; Aimar, S.B.; Babinec, F.J.; Ferramola, L. Un método para la determinación de estabilidad de agregados en seco en suelos de la región semiárida pampeana central (Argentina). *Ciencia del Suelo* **1994**, *12*, 32–34.

Agricultural Lands: Flooding and Levee Breaches

Kenneth R. Olson

Department of Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.

Lois Wright Morton

Department of Sociology, College of Agriculture and Life Sciences, Iowa State University, Ames, Iowa, U.S.A.

Abstract

Whenever levees on rivers are breached, there are soil and crop damages in the flooded bottomland areas that impact agricultural management capacities and crop productivity. Earthen levees and floodwalls can be undermined by sand boils, fail after weeks of high floodwater pressure and soil saturation, or even be topped. When a levee fails, there is often a crater lake created adjacent to the levee breach with gullies and land scouring extending into the previously protected lands. As the water spreads out and slows down, sand and sediments are deposited on the bottomlands and in road and drainage ditches. Floodwaters may drown crops and coat the entire flooded land surface with sediments containing a variety of pollutants, nutrients, and contaminants. Restoration of craters, gullies, land-scoured areas, and contaminated sediment depositional sites is necessary if agricultural lands are to be returned to some level of productivity.

INTRODUCTION

Alluvial soils are developed from fine-textured clay and silt sediments deposited in floodplains when rivers overflow their natural banks and flood into adjacent bottomlands. These water-saturated lands experience annual flooding for many months each year as the river levels vary with local and upstream precipitation and snowmelt. Fast moving floodwaters can also transport and deposit sand and gravel onto alluvial bottomlands. When these lands are drained and leveed to protect from river flooding, they are some of the most fertile and productive agricultural soils in the world. Whenever levees on rivers are breached, there are soil and crop damages in the flooded bottomland areas that impact agricultural management capacities and crop productivity. Earthen levees and floodwalls can be undermined by sand boils, fail after weeks of high floodwater pressure and soil saturation, or even be topped. When a levee fails, there is often a crater lake created adjacent to the levee breach with gullies and land scouring extending into the previously protected lands. As the water spreads out and slows down, sand and sediments are deposited on the bottomlands and in road and drainage ditches. Floodwaters may drown crops and coat the entire flooded land surface with sediments containing a variety of pollutants, nutrients, and

contaminants. Restoration of craters, gullies, land-scoured areas, and contaminated sediment depositional sites is necessary if agricultural lands are to be returned to some level of productivity.

River Bottomlands, Agriculture, and Levees

In the United States, the Mississippi River and tributaries drain 41% of the continental land mass including millions of hectares of agricultural lands protected by thousands of kilometers of earthen levees and floodwalls. By 1926, an extensive system of private and public levees along the Mississippi River was engineered to secure agricultural lands and river cities against flooding^[1] from the confluence of the Ohio and Mississippi rivers at Cairo, Illinois, all the way south to the Gulf of Mexico. The U.S. Army Corps of Engineers (USACE) and local levee districts continue to actively manage river levels to maintain navigation and protect against flooding. Prior to the construction of these levees, most river bottomlands experienced crop loss every time the river flooded, but the flooding caused little if any soil damage. Although these massive fortress-like structures seem impermeable, levees do fail for a variety of reasons and allow floodwater to flow through breaches with an intensity that can do substantive damage to agricultural crops. Not only does the crop drown if flooded during

growing season, but considerable soil damage can also occur as a result of the levee breach and lead to the creation of crater lakes, gullies that extend into agricultural lands, land scouring, and sand and sediment deposition on bottomlands as well as drainage and road ditches. Further, as these fast moving waters slow down, the drowned crop and land surface are coated with silt, clay, organic matter, and other chemicals that the water carried. This flooding of crops and soils and the maintenance, repair, and strengthening of degraded levees cost millions of private and public dollars after every levee breach.

Changes in climate,^[2] such as shifts in the long-term seasonality and frequency of extreme weather events, can result in record flooding and droughts and increase the vulnerability and risks associated with managing levee-protected agricultural lands. To understand the impacts of natural and induced levee breaching during high water and flood events, the levee system, the processes and mechanisms by which land scouring and sediment deposition occur, and the remedial activities that are used to repair and guard against levee breaching are discussed in the following sections.

Earthen Levees and Floodwalls

The levee is a massive earthworks designed to contain and channelize the river at flood stage and protect agriculture and other land uses against flooding. It has a flat crown with 3:1 sloped sides. The texture of the materials used in earthen levee construction can vary from silty clay to sandy. Grasses with thick dense roots are planted on the levee to hold the soil in place and reduce the erosive effect of water.

Breaks in the levee, called breaches, can occur when a portion of the levee is eroded or breaks from a subsurface weakness. The higher the levee, the greater the force of the river on the protected land when a breach occurs; thus, a levee as high as a three- or four-story building can explode with the same power and suddenness of a dam bursting. Silty clay levees with a sand core can be affected by vegetation and animal burrowing, which in turn influence the susceptibility of the levee to erode and naturally breach. However, the greatest danger to levee failure is constant water pressure against the levee.^[1,3] The weight of the river can push water underneath the levee and create sand boils that undermine the strength of the levee and its capacity to hold back water. Another type of levee is a floodwall constructed of concrete. These are often built in urban areas at the most erosive points in the river, usually the bend where strong currents and constant pressure of flowing water can erode an earthen levee. Floodwalls may also replace earthen levees when a large quantity of desired soil material is not available or is too costly to transport.

Levee Saturation and Topping

The water-holding capacities of the soil in earthen levees affect its strength to withstand the pressure of the river. During record flooding, the levee can be saturated for a prolonged period and fail, or the floodwater can be higher than the levee and top (run over) the levee crest (Fig. 1). When floodwater starts over the top of the levee, it can cut an erosion channel into and through the levee. Once the floodwater flows over or through a break in the levee, the

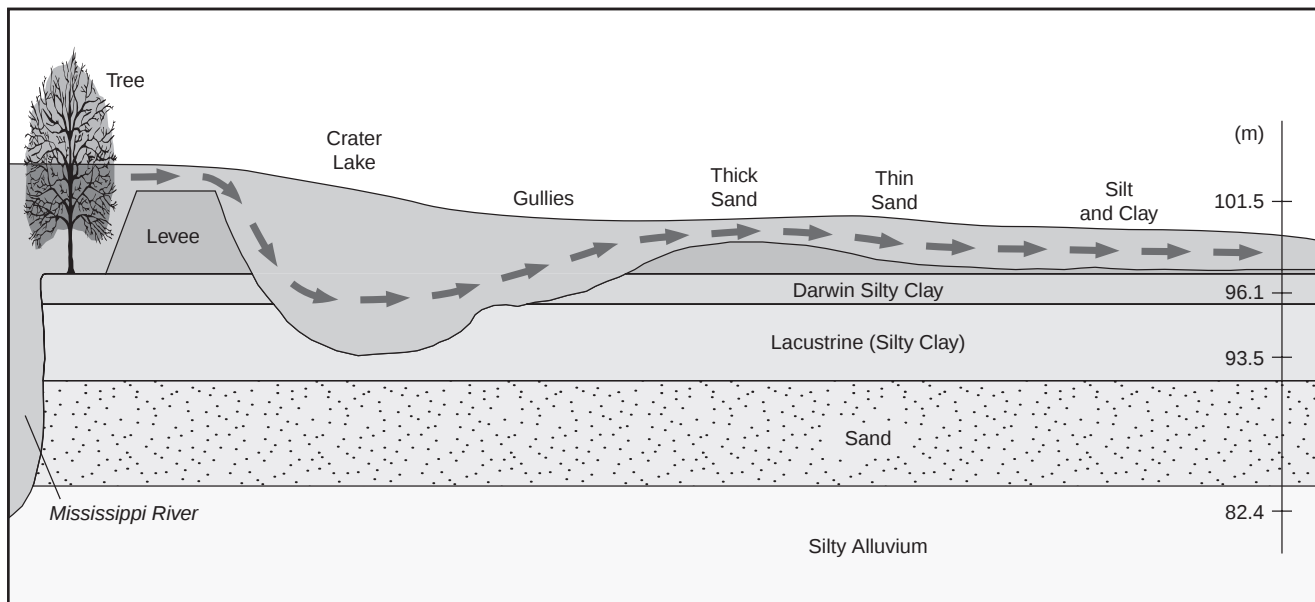


Fig. 1 Earthen levee topped by floodwaters resulting in a crater lake.

Source: Reprinted with permission from Olson & Morton.^[5]

river water drops with great force and cuts a crater lake on the inside of the levee. As more water flows through the eroding crack, the floodwaters' speed increases and widens the breach by removing sections of the adjacent levee.^[4] The breach can end up at 100 m or more in width and result in a deep crater lake.

Sand Boils

When floodwaters put significant pressure on floodwalls and levee systems, seepage and sand boils can occur, especially if there are sandy soils underneath the floodwall (Fig. 2) or earthen levee.^[3] Sand boils, including a mega sand boil, occur when the river is at or above flood stage. The bottomland inside the levee acts like an empty sunken basin with the higher floodwater outside the basin creating a hydraulic gradient (Fig. 2). As the water seeks to equalize the pressure on both sides of the levee, a stream of water (called piping) can force its way through even a tiny opening in the riverside of the levee

or floodwall.^[4] Once the water is piped through the soil into the basin, the pressure is released and water shoots up through the porous earth or sand creating a churning or boiling action from which the sand boil is derived.^[6]

Uncontrolled seepage, a major cause of levee failure, creates instability when high water pressure and soil saturation cause the earth materials to lose their strength. Most small sand boils are treated with a 1.5-m-high sandbag ring and filled with water.^[4] The sandbag dike is normally a temporary measure to increase the depth and weight of water over the sand boil in order to decrease the hydraulic gradient across the seepage path and reduce the potential for erosion of earth materials along the piping path.^[6] However, a sand boil can start small and quickly turn into a high-energy sand boil (Fig. 2). In a few hours, it can enlarge dramatically from a few centimeters to 0.6 m in diameter. In the case of a mega sand boil, the crew often has to construct a 15-m ring berm to a height of 2 m or more. When the counterweight of water alone is insufficient to control a mega

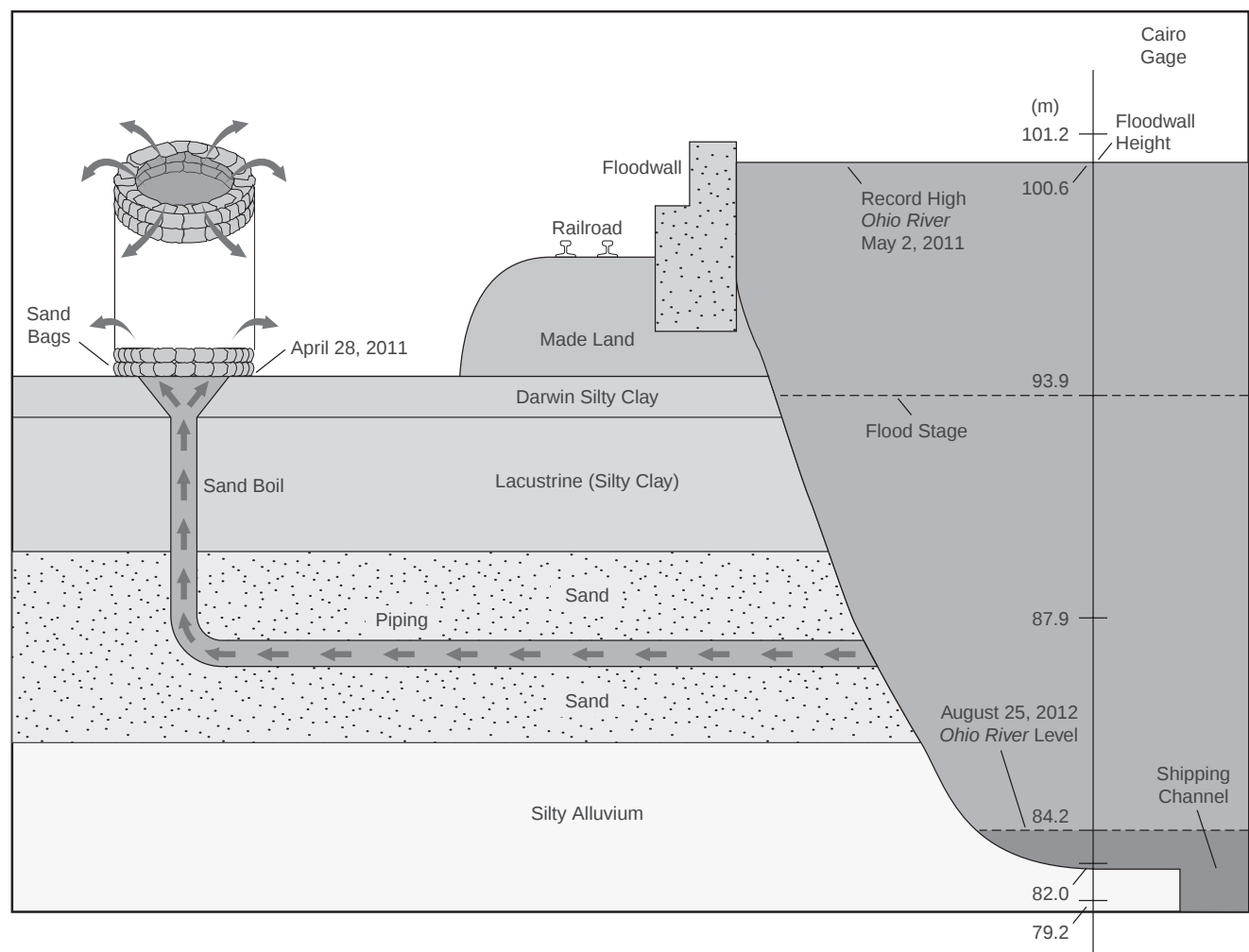


Fig. 2 Anatomy of a sand boil.

Source: Reprinted with permission from Morton & Olson.^[7]



Fig. 3 Crater lake extending through 100-m levee breach.

sand boil, it is covered with a few meters of fly ash cinders or other available materials. The treatment of a mega sand boil can require big earth moving equipment such as bulldozers, backhoes, loaders, excavators, and dump trucks and a large crew to contain the mega sand boil and keep watch until the flood stage recedes.

Crater Lakes, Adjacent Gullies, and Thick Sand Deposits

As the floodwater tops or breaks through an earthen levee, it often drops many meters to the bottomland inside the levee and causes deep erosion of the soil and underlying geological parent material (Fig. 1). Most crater lakes (Fig. 3) are several meters deep.^[3] As the floodwater flows rapidly into the previously protected bottomland, gullies originating adjacent to the crater lake are cut into the soil and can extend 10–100 m beyond. Geologic materials, soil, sand bars, and sediment carried down river and from the degraded earthen levee are deposited in the bottomland after the floodwaters slow down (Fig. 1).^[3] The thick sand can be deposited on agricultural crops (Fig. 4) and other



Fig. 4 Sand delta covering corn field with tree residue from between river and levee.



Fig. 5 Gully fields on O'Bryan Ridge with levee in background.

vegetation in the land surface as well as in drainage and road ditches.

Land Scouring and Gully Fields

After a levee breach occurs and the fast current of the water has created crater lakes, extended gullies into the bottomland (Fig. 5), and scoured the land surface, the speed of the advancing floodwater begins to slow and deposit sand particles on the bottomland in large sand deltas. As the floodwaters continue to slow, the silts drop and then the clay and organic particles settle out and coat plant residues and the land surface.^[8,9] Significant land scouring can result in many hundreds of hectares of land losing centimeters of topsoil after each levee breach.

Levee-protected bottomlands normally have very little slope and are almost flat but can contain higher natural levees formed from old oxbows cut off from the river (Fig. 6). An oxbow is the wide curve portion of a meandering river channel. The floodwaters will pond in front of these meander scars or natural levees. When water flows over a natural levee ridge and drops down to another alluvial bottom, it concentrates and creates an eroded channel or waterway. This erosion process is called hydraulic jumping. As high-energy floodwater flows into the newly created waterway, it can cut into the ridge and carve additional new channels and gullies.

These deeply eroded soils or gully fields (Fig. 8) are extremely difficult and costly to reclaim.^[8] Often bulldozers are used to push in the tops of the vertical gully walls to fill in the bottom and then grade the side slopes. The pushing of topsoil into the gullies puts the soil material on slopes that are highly erodible, and the exposed subsoil and parent material of the scrapped areas lower the productivity of the original soils. Topsoil must be hauled in to raise the soil organic carbon content of the soil if the land is to be returned to the previous level of agricultural productivity. Terracing is another option for reshaping the side slopes above the gully

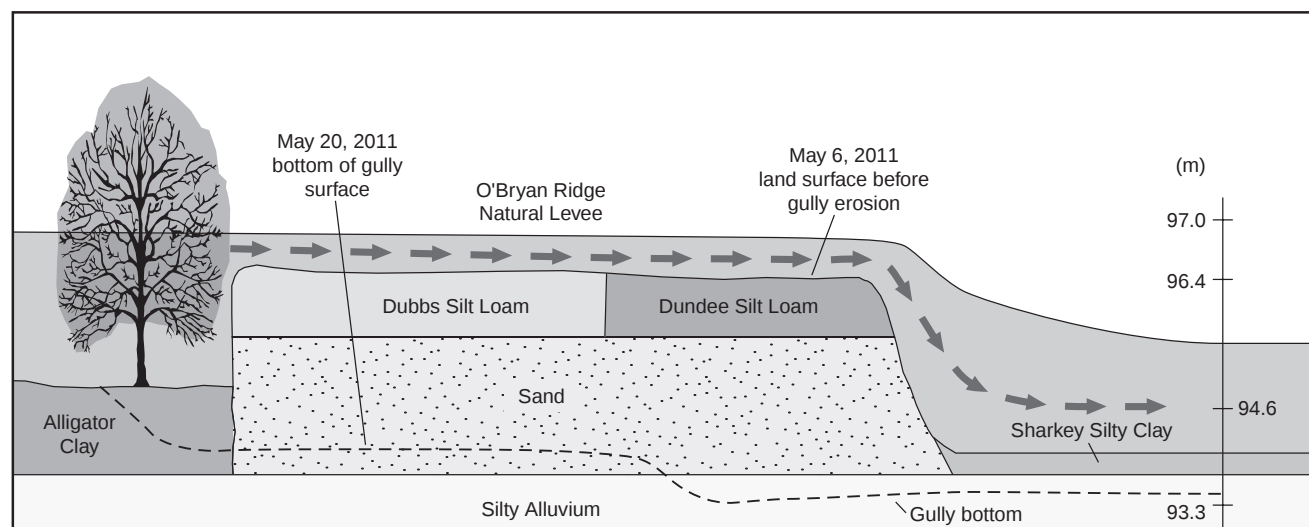


Fig. 6 Land scouring and gully formation when floodwaters flow over a natural levee ridge.

bottom. This approach will still result in the loss of long-term soil productivity and crop yields because the newly created soils will be less productive than the original soils as a result of mixing topsoil with parent material low in soil organic carbon. Reclamation efforts to restore these land-scoured ridges and gully fields can cost hundreds of thousands of dollars and are likely to still result in lowered soil productivity and crop yields when compared to original crop yields. When gully fields are created by levee breaching, the land use may change. Gullies that are not reclaimed will collect water and become wetlands. This loss of agricultural land to ponds and wetlands results in a net loss of agricultural productivity in the region.

Effect of Growing Crops and Residue on Erosion and Deposition

Crops grown in the Mississippi and Ohio River bottomlands are primarily wheat, corn, soybean, and forages. Depending on the time of year, the land cover may include these crops in various stages of growth or only plant residues remaining from the previous year's crop such as soybean stubble, corn stalks, and wheat stubble. When spring floods occur, winter wheat and forages are growing and are likely to drown. However, these fully developed plants can hold the soil in place and prevent the land scouring. Sediment carried by floodwaters is caught by the wheat and forage vegetation and deposited on the crop. If the levee breaching and flooding occur in the spring before corn and soybeans are planted, only previous crop plant residues are protecting the bottomland soils. These plant residues are often picked up and carried along by floodwaters and lose their protective capacity to prevent land scouring. When the flooding

and levee breaching occur later in the growing season, the soybean and corn plants help slow the speed of the floodwaters and anchor the soil. After the floodwaters recede and the land drains, the soil can either be dried and planted or be left idle but tilled to eventually mix in the sediments to help dry out the fields depending on the timing of the flooding. Thin layers of silt and clay deposits can be treated by sunlight, drying, and tillage to incorporate into the plow layer. Tillage equipment, such as chisel plows and moldboard plows, can be used on the areas with thin sand deposits (<15 cm) in an attempt to mix the sand into the underlying bottomland with silty and clayey topsoil.

Crop damages depend on the type of crops commonly grown in the area and the timing of the levee breach and subsequent flooding. If flooding occurs in the growing season of a crop and the plants are under floodwater for 24–36 hours, the submerged crop will drown and can result in a total crop failure for that year. If flooded early in the growing season, the crop can be replanted but usually results in lower yields. Crop insurance can provide replacement for a portion of the income for farmers who have purchased it.^[10,11]

Sediment Deposition in Road and Drainage Ditches

After a levee breach and flood event, road and drainage ditches in the area are filled with sediments and sand, sometimes as much as 1–2 m deep. Excavators are usually brought in by either the county or drainage districts or Natural Resources Conservation Service to remove debris and sediment that block drainageways and ditches to speed up the drainage process and to accelerate the drying out of low lying areas.^[8,9] The sediment in private drainageways

of most qualified landowners can sometimes be partially financed by the U.S. Department of Agriculture Emergency Services Agency's Conservation program. The local drainage district often provides additional matching funds for these projects.

Levee Repair, Sand Delta Removal, and Crater Lake Filling

If the funds are available, the USACE or the farmer levee and drainage districts usually begin reconstructing levees immediately after the floodwaters have drained. Sometimes, the levee is repaired in stages if the funds are insufficient for the entire reconstruction of the levee. Restoration and repair of the levee to the original height or higher is usually a concern of the landowners flooded by the levee breach.^[9] The sand deposits are often collected and used to fill in the crater lake and then topsoil is trucked in and spread over the crater lake to restore the previous land use. The thick deltaic sand deposits can be 3–20 ha in size and between 0.1 and 1 m deep. Both the crater lakes and the thick sand deposits can result in a permanent loss of agricultural productivity^[2,11] if they are not filled in or removed. The transported topsoil may come from other levees on smaller tributaries and drainage ditches or other adjacent soil deposits. The reconstructed soil profile will still be less productive than the original soils.

Damage to Farm Building and Homes

When a levee breach occurs, there is always the risk that lives and property may be lost or seriously harmed.^[1] When a levee breach is eminent, the U.S. National Guard usually sweep the area to make sure the people living and working in levee-protected bottomlands are evacuated. There is also a high risk of damage to the homes, barns, and other structures on these flooded bottomlands.^[12] Buildings can be impacted by the force of the flowing floodwater and become fully or partially submerged in the floodwater. Water pressure can result in the loss of the lower half of entire walls, damage wooden floors, or destroy structures completely. In addition, farm structures (sheds, barns, and grain bins) can be damaged, and depending on insurance coverage, only a few of these structures may be repaired.

Protected and Unprotected Agricultural Lands

River bottomland areas that are not protected by levees usually receive a thin layer of silt, clay, and organic matter during flood events. The crop is lost if flooding occurs in the growing season, but soils do not usually suffer permanent damage.^[13] This is not the case where levees fail. Levees can fail as a result of a sand boil or by levee topping. Blowout holes or craters can be created between 1 and 10 ha in size and can hold water. Fast-flowing water can remove hundreds of meters of

levee embankments and can erode thousands of cubic meters of soils and underlying outwash parent material to depths of many meters below the base of the earthen levee when the levee breaks.^[4] The force of the rushing water can uproot trees growing between the river and the levee and deposit them, as residue, on the previously protected floodplain. Deep gullies can extend a few hundred meters into the bottomland, and hundreds of mature trees can be transported hundreds to thousands of meters. Deltaic sand deposits up to 1 m thick cover many hectares on the floodplain at each breach site with additional hectares covered with a thin layer of sand. The remaining hundreds of hectares of previously protected floodplain soils receive a thin coating of silt and clay and can remain under floodwaters long enough to drown the year's crop if it was planted and not already removed by the wall of advancing floodwater. After a few weeks, the floodwater usually drains from the bottomland and back into the river or slowly evaporates and infiltrates bottomland soils sufficiently for the local landowners to begin the task of moving the trees from near the blowout holes and floodplain and begin filling in the craters and gullies.^[4]

CONCLUSION

Climate change will amplify the risks associated with snowmelt, rainfall, runoff patterns, and river flooding. As the odds for certain types of weather extremes increase in a warming climate, farmers, rural residents, and supporting institutions as well as public and private levee districts will need short- and long-term strategies to sustain their system of levees, address breaching events and reclamation of agricultural lands, and put in place adaptive management plans that anticipate events. Levees are complex engineered systems coupled with natural and social systems. Levee redesigns must account for not only the risks to the engineered system but how to make the social, economic, and environmental systems more resilient. One need is to better understand how the soils are affected by flooding and levee breaching and their capacities to support to agricultural productivity.

The following three recommendations are offered to provide valuable data in assessing soil degradation and to guide restoration in making levee-protected agricultural bottomlands more resilient: 1) improve characterization and measurement of eroded soils and distribution of sediment contaminants after levee breaching; 2) assess contamination effects on soil productivity and long-term agricultural production in order to understand the impacts of flooding on agricultural soils; and 3) evaluate reconstruction investments needed to repair levees based on return of the land to productivity and increased landscape resilience by reducing vulnerability to future flooding and levee breaching stress.

REFERENCES

1. Barry, J.M. *Rising Tide: The Great Mississippi Flood of 1927 and How It Changed America*; Simon & Schuster: New York, 1997.
2. Palmer, M.A.; Lettenmaier, D.P.; Poff, N.L.; Postel, S.L.; Richter, B.; Warner, R. Climate change and river ecosystems: Protection and adaptation options. *Environ. Manage.* **2009**, *44* (6), 1053–1068.
3. Olson, K.R.; Morton, L.W. The effects of 2011 Ohio and Mississippi River Valley flooding on Cairo, Illinois area. *J. Soil Water Conserv.* **2012**, *67* (2), 42A–46A.
4. Olson, K.R. Impacts of 2008 flooding on agricultural lands in Illinois, Missouri, and Indiana. *J. Soil Water Conserv.* **2009**, *64* (6), 167A–171A.
5. Olson, K.R.; Morton, L.W. Impacts of Len Small levee breach on private and public Illinois lands. *J. Soil Water Conserv.* **2013**, *68* (4), 89A–95A.
6. Veesaert, C.J. Inspection of embankment dams. Session X in Embankment Dams. Bureau of Reclamation, 1990. Available at http://www.michigan.gov/documents/deq/deq-p2ca-embankmentdaminspection_281088_7.pdf (accessed December 8, 2012).
7. Morton, L.W.; Olson, K.R. Sinkholes and sand boils during 2011 record flooding in Cairo, Illinois. *J. Soil Water Conserv.* **2015**, *70* (3), 49A–54A.
8. Olson, K.R.; Morton, L.W. The impacts of 2011 man-induced levee breaches on agricultural lands of the Mississippi River Valley. *J. Soil Water Conserv.* **2012**, *67* (1), 5A–10A.
9. Olson, K.R.; Morton, L.W. Restoration of 2011 flood-damaged Birds Point-New Madrid Floodway. *J. Soil Water Conserv.* **2013**, *68* (1), 13A–18A.
10. Morton, L.W.; Olson, K.R. Point-new Madrid floodway: Redesign, reconstruction and restoration. *J. Soil Water Conserv.* **2013**, *68* (2), 35A–40A.
11. Lowery, B.; Cox, C.; Lemke, D.; Nowak, P.; Olson, K.R.; Strock, J. The 2008 midwest flooding impact on soil erosion and water quality: Implications for soil erosion control practices. *J. Soil Water Conserv.* **2009**, *64*, 166A.
12. Camillo, C.A. *Divine Providence: The 2011 Flood in Mississippi River and Tributaries Project*; Mississippi River Commission: Vicksburg, 2012.
13. Olson, K.R.; Morton, L.W. Soil and crop damages as a result of levee breaches on Ohio and Mississippi rivers. *J. Earth Sci. Eng.* **2013**, *3*, 139–158.

Agricultural Management: Root Growth

Susanne Schroetter

Institute of Plant Nutrients and Soil Science, Federal Agricultural Research Centre (FAL), Braunschweig, Germany

Jutta Rogasik

Institute of Plant Nutrients and Soil Science, Federal Agricultural Research Centre (FAL), Berlin, Germany

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

Roots represent the belowground component of vascular plants and are one of the most important plant organs responding to changes in growth conditions. The growth of the root system is genetically determined. However, the root formation of crops can strongly be affected and controlled by agricultural management. Roots grow by a simple branching process leading to lateral roots emerging from the main root. Generally, the root biomass may comprise almost half of the plant biomass.

ROOT GROWTH

Roots are cylindrical organs without nodes and internodes. The growing point is located at the root tip, followed by a small section containing numerous fine roots or root hairs. Because of the small size and multitude of these root hairs, the explored soil volume and uptake of water and ions by the plant roots increase drastically.^[1] The roots build a root system in the soil, with a total length far greater than that of the aboveground shoot system. Predominantly, the roots are located within upper soil layers, which have a sufficient water and nutrient supply. However, they are able to penetrate the soil very deeply to reach the groundwater.

Dicotyledonous plants have an allorhizous root system, consisting of a gravitropically growing main root that continuously generates new lateral roots. These secondary roots of first order grow plagiotropically in horizontal direction. Monocotyledonous plants have a homorhizous root system. Seminal roots initially developed during germination from the embryonic nodes die back, and morphologically equivalent adventitious roots developed from lower stem nodes replace them. The life span of roots varies between a few days and the whole life period of the plant.^[2,3]

ROOT FUNCTIONS

The main functions of roots are anchorage of the plant in the soil and absorption of water and nutrients. Furthermore,

roots transform and distribute absorbed ions, store photosynthetic products, and synthesize growth regulators.^[4] Their effectiveness depends on the extent of the root system and varies with age and physiological condition of the plant. Plant roots are capable of entirely changing the soil chemical milieu in the rhizosphere by their metabolic activity.^[5] Generally, the nutrient availability for roots depends on the actual amount of dissolved ions and the capability of the soil to replenish ions from the organic and mineral nutrient pool. Roots are also producing plant growth regulators, thereby influencing shoot growth, photosynthesis, senescence, water balance, and phototropic and geotropic reactions.^[6]

FACTORS INFLUENCING ROOT GROWTH

Agricultural crops were selected for rapid growth according to increasing nutrient and water use efficiency. Crops respond to optimal resource availability with an increase in growth and yield. Arable crops are characterized by a relatively low root-to-shoot ratio. In the course of plant breeding, cultivated crops lose their natural high degree of phenotypic and physiological variability. Therefore, agricultural crops may be severely influenced by external stress factors such as soil compaction, water and nutrient deficiencies, and variations in temperature or diseases.

Roots of annual plants can reach up to 2 m depth. However, a sufficient supply of nutrients and water in the upper soil horizons may result in lower rooting

depths. Before exploring deeper soil horizons, roots absorb water from the upper soil layers.^[7] Mechanical soil compaction below the surface soil (e.g., plough pan) impairs the soil porosity and its water and oxygen content and thus the root penetrability. Consequently, shallower roots develop in soils with physical constraints. Furthermore, the number and the total length of roots decrease, and the regeneration of fine lateral roots and their activity are suppressed; e.g., the root diameter of cereals increased by 25% with an increase in soil depth of compacted soils.^[8] Roots mechanically stressed consume more assimilates and have low nutrient absorption, higher mortality, and a higher root turnover.^[9]

The soil water content is particularly important to root growth. Short-term water deficiency leads to the stagnation of the root growth with direct consequences to shoot growth and yield.^[10] Prolonged low water supply promotes root expansion to deeper soil layers. Extensive root branching patterns, however, enable plants to cope with long-term water stress. Under drought conditions, a favorable potassium (K) supply reduces the effects of water stress; e.g., beans grown under suboptimal soil moisture conditions increased the root dry weight up to 80% and the root length up to 35% with the application of K.^[11]

The stimulating effect of nitrogen (N) on root branching enables plants to use resources with a higher efficiency. This response is especially important to the acquisition of more immobile nutrients such as phosphorus.^[12,13]

In a field experiment, balanced fertilization predominantly affected the rooting intensity of winter wheat

(*Triticum aestivum* L.) within the upper soil horizons. At flowering, the optimal nutrient supply by combined mineral and straw/farmyard fertilization stimulated the development of finer roots with more branches, without significant differences in the total dry weight (Fig. 1). The input of carbon (C)- and N-containing organic material by root and crop residues was approximately higher by 1 t/ha in plots with organic fertilizer compared to mineral fertilization.

The root system development of cereals follows a similar pattern. During the tillering and shooting phase when the plants show the most intensive shoot development, the active root mass increases until flowering. In the topsoil, however, the amount of living roots permanently decreases during the following growing stages. The rooting depths of cereal plants can extend from 60 to 200 cm, influenced by the soil compaction and the water conditions.^[14,15]

Generally, up to 90% of the total root biomass is located within 30-cm depth, whereas the most intensive rooting occurs in the upper 10 cm of the soil. Below the topsoil (the 30-cm horizon), the root density proportionally decreases with the depth. In sandy soils, where nutrient losses by leaching can be high, growing a catch crop contributes to the enhancement of the soil fertility, e.g., by the improvement in the soil structure, maintenance of organic matter content, and the reduction of nitrate leaching. Catch crops differ in rooting depth and root dry matter (DM) production (Fig. 2). The amount of N stored temporarily within roots and shoots may reach up to 160 kg/ha for oil radish (*Raphanus sativus* L.), 120 kg/ha for phacelia

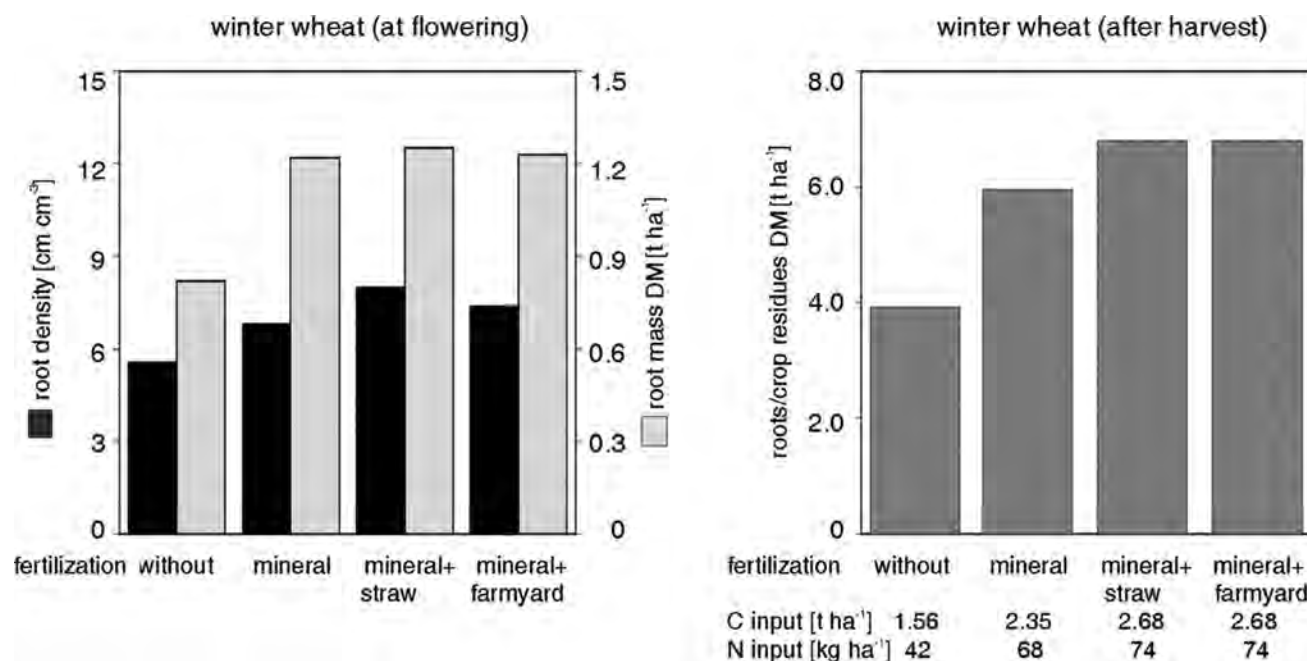


Fig. 1 Root density and root DM of winter wheat (*Triticum aestivum* L.) within the topsoil (0–30 cm) of a loamy sand and C and N inputs by crop residues after harvest.

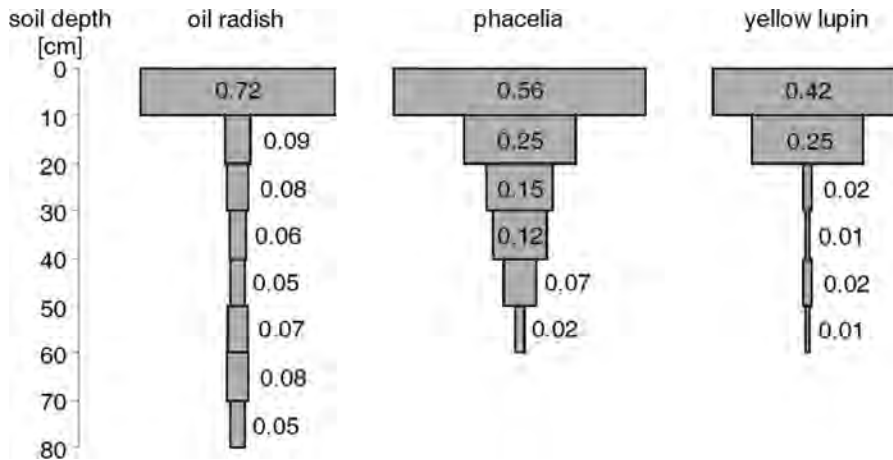


Fig. 2 Root DM (t/ha) of catch crops in a sandy soil at the end of the growth period.

Source: From Rogasik, Smukalski, et al.^[16]

(*Phacelia juss*), and 100 kg/ha for lupines (*Lupinus luteus*), respectively.^[16]

The annual average input of organic matter from roots and crop residues to agricultural soils of the temperate zone ranges between 0.56 and 6.06 t/ha/yr, respectively, depending on the crop species.^[10] Postharvest residues including plant roots are fundamental to the maintenance of soil organic matter stock.

Soil tillage strongly affects the soil physical properties and thus the development of the root system. In long-term field experiments in Germany, conservation tillage encouraged more intensive rooting within the upper 30 cm soil depth compared to conventional tillage including ploughing (Fig. 3). Ploughing of compacted soils can increase root biomass. However, differences have been observed between the root length density of

cereals (*Secale cereale* L.) and beans (*Vicia faba* L.) by different tillage methods. Cereals have a widely branching root system, consisting of adventitious and many fibrous roots, whereas the root system of beans is comprised of tap root systems with laterals.

CONCLUSION

Roots are continuously produced and turned over during the entire vegetation period. For their growth, appropriate nutrient supply, soil organic matter content, and the physical soil conditions within the rooting zone play an important role. The balanced application of suitable amounts of mineral and organic fertilizers can stimulate the root development. Roots are an important input of easily decomposable organic matter into arable soils. Roots also contain very stable organic compounds like lignin. During the decomposition processes, the nutrients organically bound within the root biomass are released over a longer period and are available again for plant nutrition. Crop residues and roots remaining on the field after harvest act as a temporary C and N pool.

REFERENCES

1. Hopkins, W.G. *Introduction to Plant Physiology*; John Wiley & Sons: New York, 1995.
2. Lüttge, U.; Kluge, M.; Bauer, G. *Botanik. Ein grundlegendes Lehrbuch*; VCH: Weinheim, 1989.
3. Waisel, Y.; Eshel, A. Multifunctional behavior of various constituents of one root system. In *Plant Roots—The Hidden Half. Books in Soils, Plants, and the Environment*, 1st Ed.; Waisel, Y., Eshel, A., Kafkafi, U., Eds.; Marcel Dekker, Inc.: New York, Hong Kong, 1991; 39–51.
4. Fitter, A. Characteristics and functions of root systems. In *Plant Roots—The Hidden Half. Books in Soils, Plants, and the Environment*, 3rd Ed.; Waisel, Y., Eshel, A., Kafkafi, U., Eds.; Marcel Dekker, Inc.: New York, 2002; 15–32.

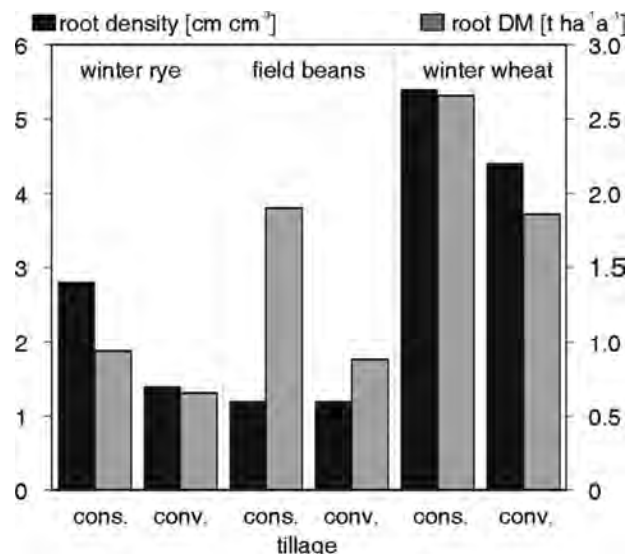


Fig. 3 Effects of soil tillage on the root development of different cultivated crops in the topsoil (0–30 cm) at flowering (cons.: conservation tillage without ploughing; conv.: conventional tillage including ploughing).

5. de Willigen, P.; van Noordwijk, M. *Roots, Plant Production and Nutrient Efficiency*; Doctoral thesis, Agriculture University: Wageningen, 1987.
6. Itai, C.; Birnbaum, H. Synthesis of plant growth regulators by roots. In *Plant Roots—The Hidden Half. Books in Soils, Plants, and the Environment*, 1st Ed.; Waisel, Y., Eshel, A., Kafkafi, U., Eds.; Marcel Dekker, Inc.: New York, Basel, Hong Kong, 1991; 163–177.
7. Klepper, B. Root growth and water uptake. In *Irrigation of Agricultural Crops; Agron. Mongr.*; Stewart, B.A., Nielsen, D.R., Eds.; ASA-CSSA-SSSA: Madison, 1990; Vol. 30, 281–322.
8. Himmelbauer, M.; Cepuder, P.; Loiskandl, W.; Kastanek, F. Field study on root soil nitrogen relations. *J. Balkan Ecol.* **2002**, 5 (1), 31–40.
9. Helal, H.M.; Müller, A.; Sauerbeck, D. Auswirkungen von bodenverdichtungen auf die wurzelfunktionen und den nährstoffhaushalt. *Landbauforsch Volk* **1994**, SH 147, 167–177.
10. Klimanek, E.-M. *Ernte- und Wurzelrückstände landwirtschaftlich genutzter Fruchtarten*; Forschungszentrum f. Bodenfruchtbarkeit Müncheberg, AdL: Berlin, 1987.
11. Sangakkara, U.R.; Hartwig, U.A.; Nösberger, J. Response of root branching and shoot water potentials of French beans (*Phaseolus vulgaris* L.) to soil moisture and fertilizer potassium. *J. Agron. Crop Sci.* **1996**, 177, 165–173.
12. Barraclough, P.B. Root growth, macronutrient uptake dynamics and soil fertility requirements of a high yielding winter oilseed rape crop. *Plant Soil* **1989**, 119, 59–70.
13. Davies, W.J.; Dobson, C.; Rados-Blanus, T. Plant factors and opportunities for the improvement of root functioning. In *Proceedings of the International Fertiliser Society, Dahlia Greidinger Symposium*, Lisbon, Portugal, Mar 5, 2001; 387–402, 468.
14. Böhm, W. Untersuchungen zur Wurzelentwicklung bei Winterweizen. *Z Acker Pflanzenbau* **1978**, 147, 264–269.
15. Roth, D.; Werner, D.; Krumbiegel, D. Modellparameter und Modellberechnungen zur Ermittlung des Niederschlags- und Zusatzwasserbedarfes landwirtschaftlicher Fruchtarten in Abhängigkeit von Standort und Witterung. *Arch Acker Pfl Boden* **1987**, 31 (8), 531–541.
16. Rogasik, J.; Smukalski, M.; Obenauf, S. Cover crops on sandland in Germany: Husbandry and fate of nitrogen. *Asp. Appl. Biol.* **1992**, 30, 309–316.

Agricultural Soil Carbon: Trading

Bruce A. McCarl

Agricultural Economics, Texas A&M University, College Station, Texas, U.S.A.

Sung Ju Cho

Department of Global Cooperation Research, Korea Rural Economic Institute, Naju-si, Korea

Jinxu Ding

Department of Public Economics, Xiamen University, Xiamen, China

Pei Huang

Aurora Energy Research, Oxford, U.K.

Layla Shiva

Department of Agricultural Economics, Texas A&M University, College Station, Texas, U.S.A.

Chin-Hsien Yu

Institute of Development Studies, Southwestern University of Finance and Economics, Chengdu, China

Abstract

In the 1990s, prospects arose for income from carbon sequestration with the end of the decade seeing formulation of the Kyoto Protocol. Discussions of the potential for soil carbon trading were widespread. However, in the ensuing 15+ years, the actual incidence of trading soil carbon for money has been quite low. Here we explore the background and motivations for soil carbon trading, mechanisms for trading, realized experience in markets, reasons for low levels of trading, and possible actions that might enhance chances.

INTRODUCTION

In the 1990s, prospects arose for income from carbon sequestration with the end of the decade seeing formulation of the Kyoto Protocol. Discussions of the potential for soil carbon trading were widespread. However, in the ensuing 15+ years, the actual incidence of trading soil carbon for money has been quite low. Here we explore the background and motivations for soil carbon trading, mechanisms for trading, realized experience in markets, reasons for low levels of trading, and possible actions that might enhance chances.

BACKGROUND: WHY TRADE SOIL CARBON

The reasons for considering trading soil carbon come in large part from an understanding of the carbon cycle and the link between climate change and atmospheric concentrations of greenhouse gases (GHGs). The motivation for

and logic behind GHG control has been well discussed in the series of Intergovernmental Panel on Climate Change (IPCC) documents^[1–3] along with many other publications including the U.S. National Academy of Sciences.^[4] These arguments will not be reproduced here as there is already a voluminous literature.

In terms of soil carbon sequestration, many arguments have been made. Thus, rather than repeat them, we refer the reader to the many pieces by Lal and associates, the conference proceedings by Rosenberg, Izaurralde, and Malone,^[5] and the IPCC working group III chapters on agricultural mitigation^[6] along with likely other chapters in this book.

Additionally beyond GHG motivations, there are other reasons why one might wish to engage in soil carbon trading. Just to list a few of these:

- Many of the mechanisms for increasing soil carbon are well-known practices that have also been advocated for soil conservation, erosion control, runoff reduction, and water quality improvement (Lal et al.^[7] and Follett and

Kimble^[8] discuss these items as likely do other chapters in this book).

- Increasing soil carbon content is often associated with increasing soil organic matter with accompanying benefits for crop yields, nutrient holding, and water retention plus a number of other benefits that are well known to those who would read this book (for more again, see Lal et al.,^[7] Follett and Kimble,^[8] or other chapters in this book).
- Market trading of soil carbon for money would enhance farm income and provide an alternative form of income support to farmers. Furthermore, the agricultural role in the market for offsets is likely to be small and thus the demand for soil carbon offsets is likely to be much more elastic than typical agricultural commodity demands. This means that most of the benefits of those markets will go to producers rather than mainly accruing to consumers as is commonly found in analyses of new agricultural supply enhancements.
- Many have discussed cobenefits in terms of soil and water quality, biodiversity, and other items (see Lal et al.,^[7] Follett and Kimble,^[8] Elbakidze and McCarl,^[9] or other chapters in this book).
- Agricultural soil carbon offsets have been argued to be a potentially cheap source of GHG offsets^[10,11] and thus may allow society to achieve atmospheric GHG reductions at a lower price, greatly lower in the cost of climate change action.^[12,13]

THE CONCEPT OF TRADING

Trading soil carbon is a means of achieving reductions in environmental damage by harnessing the power of the marketplace as suggested by Baumol and Oates.^[14] In particular, suppose policymakers develop a program to limit society's net emissions of GHGs and require all emitters to cut back by some given percentage. Under such circumstances, some emitters may find it very expensive to cut back emissions, whereas other parties may find it substantially cheaper or may have cheap ways of employing sequestration activities. If a marketplace is established, which allows high-cost emitters to meet their emission reduction obligations by, among other actions, buying emission reductions or sequestration increases that are carried out by other parties, then the overall societal cost of reducing net emissions would be reduced. Such an approach was advocated in the Kyoto Protocol.^[15]

In the soil carbon setting, the basic trading concept is to allow soil sequestration to be sold into the marketplace to provide emitting entities with an alternative way of meeting their emissions reduction obligations by paying farmers a smaller amount to increase soil sequestration. This would introduce another commodity that agriculture could sell.

WHY HAS TRADING BEEN LOW

Now let us turn attention to the questions of why there has not been much trading. This really boils down to three reasons:

- Trading volumes and prices have been low.
- Agriculture has not really been let in the door.
- Soil carbon brings with it some unique challenges.

Each of these is discussed individually.

Trading Volumes and Prices Have Been Low

It is almost uniformly a fact that in general GHG markets once up and running, the price has been smaller than expected. Toward the end of its run, the Chicago Climate Exchange (CCX) prices were in the pennies, the European price in the 2015 market fell to three euros, and the Environmental Defense Fund indicates that prices from the first year of California trading are lower than expected.^[16] There is also the possibility of trading underneath the Clean Development Mechanism and in trading systems in New Zealand, Australia, and Alberta. However, in those cases soil carbon trades have either been non-existent or exhibit certain problems as are discussed below.

The largest exchange and one often pointed to is the European emission trading system (ETS); however, this certainly has had some troubles. Burtraw^[17] argues that the ETS is “hobbled” mainly because it exhibits low prices. Burtraw^[17] attributes the low prices to overestimates of the costs of compliance, the economic downturn, the influence of carbon taxes and other complementary policies in European states, an influx of international credits, the banking of excess emission reductions from prior years, and early auctioning of next year's allowances. The incidence of low prices certainly limits the demand for potential offsets like those from soil.

However, when considering why U.S. credits have not been trading there are other causes. Certainly in the United States, we have seen a decision to not ratify the Kyoto Protocol, a lack of passage of any cap-and-trade-related bills in Congress, a failure of the voluntary CCX, the incidence of low prices under the Northeast Regional Greenhouse Gas Initiative scheme forcing an early 2014 redesign and a slow evolution of the California market.

Generally then trading volumes have been low because of low prices and/or a lack of some sort of regulation, which provides an incentive for emissions reductions. As a consequence, there has not been much expressed market demand for bringing agricultural soil into the picture. However, there is some hope of increased U.S. trading as stimulated by the EPA proposal regarding GHG emissions from power plants.

Agriculture Has Not Really Been Let in the Door

While the trading volumes have been low in general, they have been yet lower in agriculture. This basically is because agriculture has been virtually universally treated as an offsets sector, not a capped sector, and one that is either precluded from some of the larger schemes or included but only in very selective ways. Furthermore, where included, the trading rules have not necessarily guaranteed GHG net emission reduction success. Reasons grouped into physical factors, market demand factors, and attitudes are discussed in this section and the rule question in the next section.

In terms of agriculture being treated as an offset sector, many agricultural interests have generally argued that agriculture should not be subject to a cap on emissions. This offsets role has been adopted in virtually every trading environment and means that agriculture is not to be forced to reduce its emissions but rather to operate as a source of offsets. This would involve, for example, reducing manure-based emissions or increasing sequestration and in turn being able to create saleable offset credits that can enter into the marketplace and purchased by, for example, large industrial or electrical power generating concerns for use in meeting their obligations. However, the ability to do this has been limited by eligibility criteria as discussed in the next paragraph.

In terms of agriculture being precluded from some of the larger schemes, the largest volume market is the ETS and there are no provisions for any agricultural participation in that scheme. IPCC^[3] also indicates that agricultural participation in the New Zealand scheme has been delayed indefinitely and that most of the other schemes only allow agricultural participation on a restrictive basis (as reviewed in IPCC^[3] in Chapters 11 and 15). This largely involves afforestation/reforestation (generally ignoring soil carbon) and manure methane along with some agricultural residues for biomass energy. The main reasons for this are the likely ease of monitoring of the standing tree, crop inputs to bioenergy, and manure emission recovery coupled with the point nature of emissions from the allowed practices. California has a draft rice protocol that also has some soil carbon-related features but is mainly focused on methane. Alberta has features regarding control of nitrous oxide (N₂O) emissions and fuel use associated with the management of fertilizer, manure, and crop residues. Australia also has features under the carbon farming initiative, although the first possibility for payment for land moving from crops to pasture is just emerging.

Soil Carbon Brings With it Some Unique Challenges

While trading and agricultural participation therein has been limited, there are also some characteristics of soil sequestration, which complicate and have been seen to limit trading. These characteristics involve physical factors,

demand factors, and attitudes toward soil sequestration. Each is discussed below.

Physical factors

The main physical features, we discuss here are spatial distribution of carbon, timing of sequestration accumulation, heterogeneity of sequestration, and uncertainty in accumulation.

Soil carbon is spatially ubiquitous with it dispersed throughout the landscape. Furthermore, unlike sequestered carbon in standing trees or carbon injected then sequestered in a deep underground aquifer, it is not directly observable by eye or simple flow instruments. Rather one can sample and measure it at sample points and then infer based on the samples stored in a particular area. However, it is not practically possible to ever measure the total amount in storage. This means that the amount of carbon sequestration has similar characteristics to a non-point pollutant in that it is difficult to know exactly how much there is and where it is.

Second, there are issues regarding the rate of accumulation. The amount of carbon that a given unit of soil will ultimately hold depends on the relationship between the rate of carbon addition and the rate of decomposition. This means that when a practice is changed resulting in an increase in carbon addition, the amount of soil carbon sequestered will rise until the rate of carbon decomposition matches the rate of addition. West and Six^[18] discuss this at length. What it means from a practical standpoint is that given a new adoption of a sequestration-enhancing practice, a change in sequestered carbon will accumulate for a while but then reach some sort of new equilibrium. West and Post^[19] review experimental data on tillage system changes and indicate that carbon accumulation generally occurs for somewhat less than 20 years with longer time periods for other sorts of changes. Furthermore, for all practices, they examine the rate of sequestration addition eventually falls to near zero. This means that one cannot rely on continual and constant levels of long-term annual soil carbon sequestration additions unlike is the case with manure management methane.

In addition, most actions that increase soil sequestered carbon have the characteristic that if the action is reversed (i.e., no till is discontinued) the soil carbon is released relatively rapidly. This then means that the sequestration level under most practices will continue for a while but then will approach a zero annual increment. It also means that the practice must be maintained perhaps at a cost. To our knowledge, the only exception to this involves biochar where the form of the carbon involved is highly stable in the soils and is not subject to practice reversal plus does not saturate nearly as fast. See, for example, the book by Lehmann and Joseph^[20] for discussion. Kim, McCarl, and Murray^[21] show in the face of this that carbon prices for sequestration are likely to be discounted by the market.

Third, sequestration is heterogeneous across different places depending upon physical and climate characteristics with the effects of practices highly dependent on local conditions. This means that it is difficult to know exactly how much one would get from a particular practice change at a place without some kind of local study.

Fourth, agricultural production in general is uncertain between years and so will be the rate of carbon addition to the soil. Furthermore, the spatial dispersion and heterogeneity bring about uncertainty. Smith et al.^[6] classify uncertainty as having two components: mechanism uncertainty and measurement uncertainty. The mechanism uncertainty reflects the complex biological and ecological influences on the rate of sequestration while the measurement uncertainty involves the difficulty of estimating exactly how much is accumulated. Such variability can be reduced with the use of spatially diversified sites.^[22] Kim and McCarl^[23] indicate that a contract containing widespread portfolio of soils over space and time exhibits substantially less variation in carbon accumulation.

Fifth, there is producer risk issue where, for example, once a practice is undertaken, there may be some good reasons to abandon it. For example, use of no till may need to be discontinued at least temporarily, if a large weed infestation occurs and/or critical herbicides used in weed control become ineffective. This frequently leaves agricultural producers desirous of leasing rather than selling carbon (see the discussion by Bennett^[24]).

Sixth, GHG offsets are often multi-GHGs (carbon, methane, and N₂O) in nature involving a complex chain of potential actions. For example, Schlesinger^[25] criticizes justifying actions based on soil carbon sequestration alone when they also involve added emissions in terms of other gases. He illustrates this point in the case of nitrogen fertilization and argues that the N₂O emissions coming from increased fertilization more than offset the sequestration additions. Moreover, the net GHG impact of stimulating soil sequestration may also alter fuel use and fertilizer needs altering emissions of fossil fuel-based emissions and those involved with fertilizer and pesticide manufacture. This means complete accounting across all gasses likely using life cycle analysis-based techniques^[26] are needed rather than concentration on on-site soil carbon only.

Seventh, cobenefits are involved where actions to manage soil carbon sequestration can create ancillary benefits plus increase willingness to pay for projects.^[27,28] For example, actions to increase soil sequestration mitigation can reduce erosion and runoff, improve biodiversity, and improve water quality. Elbakidze and McCarl^[9] argue that these should be considered carefully as there are cobenefits from alternative forms of GHG mitigation and just focusing on those involved with soil-related actions ignores benefits from other strategies. In a case study, they find the air quality cobenefits from reduced coal usage are as large as the benefits of expanding soil sequestration. Furthermore,

they indicate that it is costly to fully assess cobenefits across a wide variety of strategies.

Market demand factors

There are also a number of considerations on behalf of those purchasing carbon offsets—the demanders. The Kyoto Protocol, which is one of the principal documents that describes desirable features of a carbon trading market, identifies the characteristics of key GHG offsets. In particular, it raises additionality, permanence, leakage, and uncertainty. There are also concerns regarding transaction costs.

The concern about additionality involves whether or not the GHG content of the atmosphere is in fact reduced when one purchases a soil carbon offset. Basically the Kyoto Accord recommends only paying for practices that generate GHG offsets that are on net additional to the offset that would have happened in the absence of the payment. This raises the broad issue of a baseline where one needs to have a reasonable estimate of what the carbon sequestration in the soil would have been in the absence of the payment^[29–31] and then only pay for the incremental amount. Moreover, baseline establishment can be difficult. The adequacy of baseline determination has been known to cause the failure of suggested mitigation approaches as reviewed in Millard-Ball and Ortolano.^[32] There have been proposed rules for additionality, which in cases preclude practices that are already in use in the region. For example, the commonly discussed financial or investment additionality concept requires that the activity generating the purchased offsets to not be financially viable without the carbon payments.^[33–35]

The additionality situation becomes complex when there are existing uses of the practice in the region both in terms of those using the practice wishing to be paid for their prior good acts (see the discussion below) and in terms of incentive design. Additionality may well lead to a discounted price for soil carbon (see the exploration in Smith et al.^[36]). There is also the question of diminished incentives, if for example one reduces the payment by say some proportion, which is an estimate of the non-additional amount (see Feng^[37] for investigation). Some of the schemes such as that under the CCX and the one in Alberta do not require prior land use history and thus could have additionality problems. Anecdotal in talks, we have heard that their participants were dominated by people who were using conservation tillage before the beginning of the payments.

The concern about permanence involves whether once soil carbon is paid for that it, in fact, remains sequestered in the soil forever or whether there is some sort of limited term to the contract in the form of a lease. This arises since, for example, the actions reducing tillage intensity may well cause an increase in sequestered carbon, but the increases in tillage intensity would lead to a release of the sequestered carbon. Farmers may also wish to guarantee sequestration

only (providing a lease) for a given amount of time as indicated by Bennett.^[24] This, however, means that at the end of that time the purchaser would have to go and buy additional carbon from some other source. Hence the mitigation effects in the agricultural sector are not necessarily permanent, and some form of a grading standard or price may be needed to reflect actions to secure carbon for a longer duration. Kim, McCarl, and Murray^[21] investigated this issue and found that the permanence discount for soil carbon can be high.

The concern about leakage refers to whether emission reduction efforts from adopting sequestration enhancing or emissions reducing practices might be offset if they reduce conventional crop production and in turn stimulate production outside of the project region through the marketplace (as discussed in Murray, McCarl, and Lee^[38]). For example, replacing corn with perennial energy crops may well reduce net emissions but can cause emissions increasing land use change elsewhere.^[39,40] Certain studies have shown that leakage can be significant but various in sectors. For example, Wear and Murray^[41] show substantial leakage regarding displaced timber harvest but Murray, McCarl, and Lee^[38] indicate that it is likely small for tillage-induced soil carbon.

The uncertainty concern involves the supply uncertainty considerations discussed above coupled with the fact that potential purchasers face penalties if they are out of compliance. Sometimes these penalties can be substantial. For example, when the U.S. sulfur dioxide trading scheme was operating, it contained a penalty for excess emissions above the permits held that exceeded 10 times the trading price. This creates a desire to buy offsets that can be safely relied upon to exceed the regulatory commitments. This likely means that credits bought would not represent the amount that would occur on average but rather reflect a safety margin in the form of an uncertainty discount. Kim and McCarl^[23] provide further justification and a procedure for computing such a discount and show that it can be reduced with broad portfolios over time and space. The carbon trading schemes such as those in Alberta, the CCX, and Australia mention such safety margins and/or were designed with conservative permitted amounts reflecting consideration of the uncertainty factor.

Finally, we come to transaction costs. This is the amount of money needed to carry out the trades and is a gap between what the ultimate purchaser pays and the offset producer receives. Transaction costs have been identified as one of the greatest hurdles for tradable permit systems.^[42] Their magnitude will have important consequences for the ability to trade an item. Cases have arisen where transaction costs have caused markets to fail.^[43] In the soil case, the transaction costs largely would deal with assembling the group of producers to sell the contract, project design, measurement and monitoring of sequestration volume search and negotiation, approval, project management, enforcement, and insurance.^[44,45]

Measurement and monitoring costs arise from compliance and practice continuation verification. The measurement costs will mainly involve the costs of some mixture of aerial sensing, on-site measurement, and modeling.^[6] Several studies have examined the magnitude of these costs, yielding some disparity in conclusions regarding their size.^[22,46,47] Such costs generally vary with the price of carbon credits, the quantity of carbon sequestered, the extent of the sequestration area, and methodological advances along with other factors.^[6,22] Millard-Ball and Ortolano^[32] argue that the costs would be increased by the expenditures on preparing the measurement and project design document plus emission monitoring.

Assembly costs may be the larger item. Alston and Hurd^[48] estimate that the transaction costs of administering the farm program ranged from 25 to 50 cents for each dollar distributed. McCann and Easter^[49] find transaction costs of the magnitude of 38% of total costs or over 50% of direct payments. Need for a contract for an annual offset of 100,000 tons of carbon dioxide from farms would involve over 200 U.S. farmers in a group (assuming 1 ton of carbon dioxide per acre per year and average farm size of 400 acres) and thousands of small farmers in developing countries imposing a likely costly set of transaction costs that might preclude small farms.^[44,47]

Higher transaction costs could well prevent the entrance of smallholders.^[6] Butt and McCarl^[50] suggest that multi-year contracts with the same group of carbon credit suppliers might help to reduce the transaction cost and in turn to keep the overall compliance costs low; moreover, bundled projects and payments could also reduce these costs.^[47]

Attitudes

There are also a number of attitudes and opinions that influence the possibility of soil carbon trading. The ones we feature here involve opposition to soil carbon sequestration and feelings about market structure, which are not consistent with some of the Kyoto-based concerns.

One set of attitudes involves opposition to using biological carbon sequestration. Doubts based on permanence and uncertainty issues have been expressed by some scientists^[51] and a number of particular European environmental groups.^[52] Another aspect involves the feeling that use of sequestration legitimizes fossil fuel use and that emissions must be the focus of control.^[53]

In terms of feelings about market structure, there are two fundamental issues. The first is reflected in the position taken a few years back by a number of farm groups that all sequestration should be paid for regardless of the land-use history including preprogram, “early adopters.” However, given the “saturation or approach to zero” findings discussed above, it is clear that the land use history determines the volume of sequestration and whether or not sequestration gains will be realized. Paying early adopters

under a strict Kyoto Protocol interpretation would not meet the additionality test.

Second, there is the opinion that farmers are unwilling to make long-term commitments as reflected in Bennett^[24] and that therefore short-term, lease-like, contracts for sales need to be used. That opinion flies in the face of the buyers' desire for permanence. There is also an allied desire for farmers to avoid long-term obligations against their farming practices and property. For example, a farmer may undertake soil carbon sequestration-enhancing practices under a 100 year contract and then in turn sell the land to someone else who must maintain those practices perhaps in the absence of carbon payments if saturation has occurred. Again this favors leasing and is in conflict with permanence desires. Marland, McCarl, and Schneider^[29] discuss this and other property rights issues.

SOME ACTIONS

We have now reached nearly the end of the entry with the findings that the general level of carbon trading has been somewhat lower than expected, the level of agricultural trading has been a lot lower than expected, and that fully Kyoto compliant soil carbon trading has been virtually nonexistent with the only trades we are aware of (CCX and Alberta) potentially having large additionality concerns. This raises the obvious question that if we wish to trade soil carbon for money what sort of actions might we undertake. There are likely a wide number of non-mutually exclusive alternatives, several of which are discussed here.

Approach one: Own the flaws. Given the vocal resistance to soil carbon (based on permanence and uncertainty), it may be best if scientists and offset producers fully acknowledge that soil sequestration is not a permanent item that is available in a certain quantity. In turn to address the permanence concerns, soil carbon producers might offer contracts for non-permanent leased items for lower than market prices. This would be accompanied by the argument that there is value in temporarily providing soil carbon credits but with recognition that this value will not be on par with permanent offsets (such as burning a methane emission—see discussion in Smith et al.^[36]). This can also be accompanied by the argument that carbon is a bridge to the future as mentioned in Marland, McCarl, and Schneider,^[29] and delays the amount of time at which the offset purchasers will have to make technological changes allowing time for innovation. Also to address uncertainty, soil carbon producers could develop contracts that contain carbon from widely spatially dispersed sequestration sites for a cumulative amount over multiple years to help manage the uncertainty (note Kim and McCarl^[23] show such contracts have the potential to greatly reduce the uncertainty). Producers would also be well served if evidence could be assembled that documents the narrowing of uncertainty under spatial and temporally diverse contracts in field

observations. Finally in the contracts, it would well be worthwhile to advocate a squaring up after some period of time to compare the actual level of increase in sequestered carbon versus that projected allowing producers to potentially gain back some of the payments for the safety margin. Also one may include offsets from other groups like cattle feeders to create a more diversified and lower risk portfolio.

Approach two: Address additionality. Prior land use history is an important factor in determining the volume of sequestration plus the total cost of the program. In particular, if all practice users are to be paid then the program may well cost more per unit carbon than under alternative offsets diminishing the attractiveness of soil carbon. Soil producer groups either need to reach an agreement with the purchasers where the purchasers are willing to pay an excess amount for the net amount sequestered carbon or somehow include a *prior* land use history requirement in the contract. Experience seems to indicate that the trading market acceptance of excess payments is somewhat unlikely and this certainly lowers the desirability of soil carbon to the purchasers. Yet this seems to be the position of most farm groups and needs to be recognized by both sides in a contracting discussion. Personally we doubt that an approach of ignoring the prior land use additionality concern can be viable but perhaps negotiations can make it work. Naturally an alternative is to only allow payments for those newly switching practices and developing procedures to document and guarantee that.

Approach three: Design-phased carbon amounts for practices in contracts. While economic studies such as Mooney et al.^[22] show that payment per ton is more economically efficient than payment for practice adoption, they ignore transaction costs. In implementation, programs such as the now defunct CCX put in a constant amount that they would pay for practices across all land in wide geographic areas. This amount is not the amount that will be sequestered each year, as it is likely to be larger at the beginning of the practice and smaller after the practice is fully mature. Perhaps a phased amount should be developed giving a tonnage for the first 5 years, a new practice is in use and a different amount for the second 5 years, then yet another amount for the third 5 years, and so on. Perhaps there should also be a maintenance payment after the practice saturates. Developing this more complex carbon schedule would more realistically represent what was happening and perhaps reduce resistance to trading soil carbon. Including the maintenance payment would also address some of the permanence issue and give farmers a stake in maintaining the carbon allowing longer term contracts.

Approach four: Figure ways to manage the transaction costs. While we believe the assertions like those in Mooney et al.,^[22] which indicate that the measurement and monitoring costs may be small, we also believe that the assembly, funds distribution, compliance monitoring, and

other costs could be quite large. In the crop insurance case that has a lot of similarity, local aggregating agents keep 25% of the money and pass 75% on to the crop insurer. This implies that the transaction costs could be large and perhaps groups need to evolve strategies to manage this like using farm cooperatives, farm groups, or public agencies. On the other hand, we would note that the Iowa Farm Bureau did this under the CCX and was heard to state that the costs were too large relative to the 10% fee they were charging. This was in part due to low carbon prices but still shows the need to engage a cheaper assembly process. A pure private market solution is likely to charge upward of 25% in transaction costs and cause a pretty large wedge. Furthermore, for smallholders in developing countries these costs could be very large precluding participation.

Approach five: Look for ways to sell carbon outside of a trading market. Practices like tillage conversion or grassland establishment that are carbon sequestering have been the recipients of payments under environmental programs such as the conservation reserve program or various soil erosion programs. Perhaps it would be best to advocate the inclusion of carbon payments in such practice subsidy-based programs and not try to trade carbon.

Approach six: Advocate grading standards in the sale of carbon much like we have them for other commodities. Agricultural commodities have involved grading standards on broken kernels and foreign matter or moisture for a long time. Certainly alternative offset forms that could enter the carbon market have different characteristics that could be differentiated by grades. For example, if methane is captured out of a manure lagoon and immediately combusted generating electrical power, then the methane can be metered and this is one-time permanent reduction in methane emissions with the methane destroyed, so it cannot be released in the future. As such, it has more value to one purchasing a carbon offset and that represents an uncertain amount of carbon sequestered in the soil that could be rereleased to the atmosphere if the practices were reversed. Thus, for example, soil carbon might fit into one grading class and could trade but at a discounted price relative to other options.

Approach seven: Advocate forms of climate change control. Soil carbon certainly will not trade in markets unless there is a market-making binding cap or substantial value to participating in a voluntary market. Perhaps those wishing to trade soil carbon should advocate the need for climate change mitigation and the value of trading as a means of lowering costs and stimulating innovative solutions. It is interesting that while many in the farming sector doubt the human influence on the climate, they also may be advocates of trading carbon for money. A widespread consistent mitigation advocacy approach would enhance the chances of trading.

Approach eight: Adopt a position that the seller will assume shortfall liability. One could facilitate trading by creating contracts with stronger guarantees of carbon

retention and carbon amounts. This could involve entering into more binding agreements regarding non-reversible practices and by using uncertainty discounts and thus assuring that the carbon being sold is in fact being sequestered. This could also be guaranteed with, for example, an offer by the sequestration producer to assume any penalties incurred by the buyer under shortfalls in compliance. In turn, individual farmers could choose whether or not they are willing to bear the risks when they decide whether or not to participate in the program.

Approach nine: Band together with other elements of agriculture to advocate an expanded agricultural role. Here one would advocate for a place for agriculture in trading. This could be coupled with documented arguments that agriculture provides a low-cost alternative relative to say the cost of emissions control in fossil fuel-using activities.

CONCLUDING COMMENTS

Trading soil carbon has not been an extremely widespread venture to date and likely will not be in the future unless different approaches and advocacy positions are taken. We have discussed the main reasons for the small role. This involves a lack of widespread markets especially in countries like the United States, the common lack of inclusion of agriculture in trading schemes, the inconsistency of soil carbon characteristics with purchasers' desires as manifest in the Kyoto Protocol, and producer/purchaser/environmental group attitudes. For trading soil carbon to become greater, we believe first that an active GHG offset trading market needs to be accessible to the producer and second that some different strategic approaches to the advocacy of and contractual basis for soil carbon trades need to arise.

REFERENCES

1. IPCC. *Climate Change 2001: Mitigation. Contribution of Working Group III to the Third Assessment Report of the Intergovernmental Panel on Climate Change*; Metz, B., Davidson, O., Swart, R., Pan, J., Eds.; Cambridge University Press: Cambridge and New York, 2001. Available at http://www.grida.no/publications/other/ipcc_tar/ (accessed August 1, 2014).
2. IPCC. *Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*; Metz, B., Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A., Eds.; Cambridge University Press: Cambridge and New York, 2007. Available at http://www.ipcc.ch/publications_and_data/publications_ipcc_fourth_assessment_report_wg3_report_mitigation_of_climate_change.htm (accessed August 1, 2014).
3. IPCC. *Climate Change 2014: Mitigation. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*; Cambridge

- University Press: Cambridge and New York, 2014. Available at <http://www.ipcc.ch/report/ar5/wg3/> (accessed August 1, 2014).
4. Fri, R.; Brown, M.; Arent, D.; Carlson, A.; Carter, M.; Clarke, L.; Chesnaye, F.d.I.; Eads, G.; Giuliano, G.; Hoffman, A.; Keohane, R.O.; Lutzenhiser, L.; McCarl, B.A.; McFarland, M.C.; Nichols, M.D.; Rubin, E.S.; Tietenberg, T.; Trainham, J.; Geller, L.; Crane, A.; Menzies, T.; Freeland, S. *America's Climate Choices: Limiting the Magnitude of Future Climate Change*. National Academy Report; The National Academies Press: Washington, DC, 2010.
 5. Rosenberg, N.J.; Izaurrealde, R.C.; Malone, E.L. Carbon Sequestration in Soils: Science, Monitoring, and Beyond. In *Proceedings of the St. Michaels*. Battelle Press: Columbus, OH, 1999.
 6. Smith, P.; Martino, D.; Cai, Z.; Gwary, D.; Janzen, H.; Kumar, P.; McCarl, B.; Ogle, S.; O'Mara, F.; Rice, C.; Scholes, B.; Sirotenko, O. Agriculture. In *Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*; Metz, B., Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A., Eds.; Cambridge University Press: Cambridge and New York, 2007; 497–540.
 7. Lal, R.; Kimble, J.M.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; CRC Press: Boca Raton, FL, 1998.
 8. Follett, R.F.; Kimble, J.M. *The Potential of U.S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect*; CRC Press: Boca Raton, FL, 2000.
 9. Elbakidze, L.; McCarl, B.A. Sequestration offsets versus direct emission reductions: Consideration of environmental co-effects. *Ecol. Econ.* **2007**, *60* (3), 564–571.
 10. McCarl, B.A.; Schneider, U.A. U.S. Agriculture's role in a greenhouse gas emission mitigation world: An economic perspective. *Rev. Agr. Econ.* **2000**, *22* (1), 134–159.
 11. McCarl, B.A.; Schneider, U.A. Greenhouse gas mitigation in U.S. agriculture and forestry. *Science* **2001**, *294* (5551), 2481–2482.
 12. Fawcett, A.A. EPA analysis of the american clean energy and security act of 2009 H.R.. In *2454 in the 111th Congress Office of Atmospheric Programs*, June, U.S. Environmental Protection Agency, 2009. Available at http://www.epa.gov/climatechange/Downloads/EPAactivities/HR2454_Analysis.pdf (accessed August 1, 2014).
 13. Fawcett, A.A.; Calvin, K.V.; de la Chesnaye, F.C.; Reilly, J.M.; Weyant, J.P. Overview of EMF 22 U.S. transition scenarios. *Energ. Econ.* **2009**, *41* (9), S198–S211.
 14. Baumol, W.J.; Oates, W.E. *The Theory of Environmental Policy: Externalities, Public Outlays, and the Quality of Life*; Prentice Hall Incorporated: Englewood Cliffs, NJ, 1975.
 15. United Nations Framework Convention on Climate Change. *Kyoto Protocol: Climate Change Secretariat (UNFCCC)* 1998. Available at <http://unfccc.int/resource/docs/convkp/kpeng.pdf> (accessed August 1, 2014).
 16. Hsia-Kiung, K.; Reyna, E.; O'Connor, T. Carbon market california – a comprehensive analysis of the golden state's cap-and-trade program, year one: 2012–2013: Environmental defense fund, 2014. Available at http://www.edf.org/sites/default/files/content/ca-cap-and-trade_1yr_22_web.pdf (accessed August 1, 2014).
 17. Burtraw, D. Saving Europe's Key Weapon against Climate Change. *Resources* **2014**, *2014* (187), 10–11.
 18. West, T.; Six, J. Considering the influence of sequestration duration and carbon saturation on estimates of soil carbon capacity. *Climatic Change* **2007**, *80* (1–2), 25–41.
 19. West, T.O.; Post, W.M. Soil organic carbon sequestration rates by tillage and crop rotation. *Soil Sci. Soc. Am. J.* **2002**, *66* (6), 1930–1946.
 20. Lehmann, J.; Joseph, S. *Biochar for Environmental Management: Science and Technology*; Earthscan: London, 2009.
 21. Kim, M.-K.; McCarl, B.A.; Murray, B.C. Permanence discounting for land-based carbon sequestration. *Ecol. Econ.* **2008**, *64* (4), 763–769.
 22. Mooney, S.; Antle, J.; Capalbo, S.; Paustian, K. Influence of project scale and carbon variability on the costs of measuring soil carbon credits. *Environ. Manage.* **2004**, *33* (1), S252–S263.
 23. Kim, M.-K.; McCarl, B.A. Uncertainty discounting for land-based carbon sequestration. *J. Agr. Appl. Econ.* **2009**, *41* (01), 1–11.
 24. Bennett, J.F. A Commentary on Marland, G., B. A. McCarl, and U. A. Schneider, Soil carbon: Policy and economics. In *Carbon Sequestration in Soils: Science, Monitoring, and Beyond: Proceedings of the St. Michaels Workshop*; Rosenberg, N.J., Izaurrealde, R.C., Malone, E.L., Eds.; Battelle Press: Columbus, OH, 1999; 168–170.
 25. Schlesinger, W.H. Carbon sequestration in soils. *Science* **1999**, *284* (5423), 2095.
 26. Wang, M.; Hong, M.W.; Huo, H. Life-cycle energy and greenhouse gas emission impacts of different corn ethanol plant types. *Environ. Res. Lett.* **2007**, *2* (2), 024001.
 27. Glenk, K.; Colombo, S. Designing policies to mitigate the agricultural contribution to climate change: An assessment of soil based carbon sequestration and its ancillary effects. *Climatic Change* **2011**, *105* (1–2), 43–66.
 28. Rodríguez-Entrena, M.; Espinosa-Goded, M.; Barreiro-Hurlé, J. The role of ancillary benefits on the value of agricultural soils carbon sequestration programmes: Evidence from a latent class approach to Andalusian olive groves. *Ecol. Econ.* **2014**, *99*, 63–73.
 29. Marland, G.; McCarl, B.; Schneider, U. Soil carbon: Policy and economics. *Climatic Change* **2001**, *51* (1), 101–117.
 30. Chomitz, K.M. Baseline, leakage and measurement issues: How do forestry and energy projects compare? *Climate Policy* **2002**, *2* (1), 35–49.
 31. Murray, B.; Sohngen, B.; Ross, M. Economic consequences of consideration of permanence, leakage and additionality for soil carbon sequestration projects. *Climatic Change* **2007**, *80* (1–2), 127–143.
 32. Millard-Ball, A.; Ortolano, L. Constructing carbon offsets: The obstacles to quantifying emission reductions. *Energ. Policy* **2010**, *38* (1), 533–546.
 33. Baumert, K.A. Understanding additionality. In *Promoting development while limiting greenhouse gas emissions: trends & baselines*; Goldemberg, J., Reid, W.V., Eds.; United Nations Development Programme: New York, 1999; 135–145.
 34. Greiner, S.; Michaelowa, A. Defining investment additionality for CDM projects—practical approaches. *Energ. Policy* **2003**, *31* (10), 1007–11015.

35. Gillenwater, M. *What is Additionality? Part 1: A Long Standing Problem*, Version 03 Ed.; Greenhouse Gas Management Institute: Silver Spring, 2012.
36. Smith, G.A.; McCarl, B.A.; Li, C.S.; Reynolds, J.H.; Hammerschlag, R.; Sass, R.L.; Parton, W.J.; Ogle, S.M.; Paustian, K.; Holtkamp, J.A.; Barbour, W. *Harnessing Farms and Forests in the Low-carbon Economy: How to Create, Measure, and Verify Greenhouse Gas Offsets*; Willey, Z., Chameides, W., Eds.; Duke University Press: Durham, 2007; 46–51.
37. Feng, S. *Three Essays on Agricultural and Forestry Offsets in Climate Change Mitigation*, PhD dissertation, Department of Agricultural Economics, Texas A&M University, 2012.
38. Murray, B.C.; McCarl, B.A.; Lee, H.-C. Estimating leakage from forest carbon sequestration programs. *Land Econ.* **2004**, *80* (1), 109–124.
39. Searchinger, T.; Heimlich, R.; Houghton, R.A.; Dong, F.; Elobeid, A.; Fabiosa, J.; Tokgoz, S.; Hayes, D.; Yu, T.-H. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. *Science* **2008**, *319* (5867), 1238–1240.
40. Fargione, J.; Hill, J.; Tilman, D.; Polasky, S.; Hawthorne, P. Land clearing and the biofuel carbon debt. *Science* **2008**, *319* (5867), 1235–1238.
41. Wear, D.N.; Murray, B.C. Federal timber restrictions, interregional spillovers, and the impact on US softwood markets. *J. Environ. Econ. Manage.* **2004**, *47* (2), 307–330.
42. Hahn, R.W.; Hester, G.L. Marketable permits – lessons for theory and practice. *Ecol. Law Quart.* **1989**, *16* (2), 361–406.
43. Atkinson, S.; Tietenberg, T. Market failure in incentive-based regulation – the case of emissions trading. *J. Environ. Econ. Manage.* **1991**, *21* (1), 17–31.
44. Post, W.M.; Amonette, J.E.; Birdsey, R.; Garten, C.T.; Izaurralde, R.C.; Jardine, P.M.; Jastrow, J.; Lal, R.; Marland, G.; McCarl, B.A.; Thomson, A.M.; West, T.O.; Wulschleger, S.D.; Metting, F.B. Terrestrial biological carbon sequestration: science for enhancement and implementation. In *Carbon Sequestration and Its Role in the Global Carbon Cycle*; Mepheron, B.J., Sundquist, E.T., Eds.; American Geophysical Union: Washington, DC, 2013; 73–88.
45. Cacho, O.J.; Lipper, L.; Moss, J. Transaction costs of carbon offset projects: A comparative study. *Ecol. Econ.* **2013**, *88*, 232–243.
46. Smith, P. Monitoring and verification of soil carbon changes under Article 3.4 of the Kyoto protocol. *Soil Use Manage.* **2004**, *20* (2), 264–270.
47. Cacho, O.J.; Marshall, G.R.; Milne, M. Transaction and abatement costs of carbon-sink projects in developing countries. *Environ. Dev. Econ.* **2005**, *10* (5), 597–614.
48. Alston, J.M.; Hurd, B.H. Some neglected social costs of government spending in farm programs. *Am. J. Agr. Econ.* **1990**, *72* (1), 149–156.
49. McCann, L.M.J.; Easter, K.W. Estimates of public sector transaction costs in NRCS programs. *J. Agr. Appl. Econ.* **2000**, *32* (3), 555–563.
50. Butt, T.A.; McCarl, B.A. Farm and forest carbon sequestration: Can producers employ it to make some money? *Choices* **2004**, *19* (3), 27–32.
51. van Kooten, G.C. Biological carbon sequestration and carbon trading re-visited. *Climatic Change* **2009**, *95* (3–4), 449–463.
52. Nicholls, M. Credits for sinks – Understanding the case against. Ecosystem Marketplace, 2014. Available at http://www.ecosystemmarketplace.com/pages/dynamic/article.page.php?page_id=474§ion=home&eod=1 (accessed October 2014).
53. Mackey, B.; Prentice, I.C.; Steffen, W.; House, J.I.; Lindenmayer, D.; Keith, H.; Berry, S. Untangling the confusion around land carbon science and climate change mitigation policy. *Nature Climate Change* **2013**, *3* (6), 552–557.

Air Permeability

Manoj K. Shukla

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

The process of exchange of air between soil and the atmosphere, called aeration, is important to plant growth because it maintains oxygen (O_2) in the root zone at the level needed for root and microbial respiration. The O_2 in soil air also governs the chemical reactions that provide the necessary conditions for oxidation of reduced elements (Fe^{2+} and Mn^{2+}), which may otherwise be toxic to plant growth. The air circulation in and out of soil matrix also regulates the temperature of the soil. In addition to plant growth, soil air composition alters the production and emission of trace gases [e.g., methane (CH_4) and nitrous oxide (N_2O)].

INTRODUCTION

Soil air differs from atmospheric air in several respects. The composition of soil air varies greatly from place to place in the soil matrix, as plants consume some gases and microbial processes release others. The carbon dioxide (CO_2) content in soil air is much higher and the oxygen (O_2) content is much lower than the atmospheric air. The soil air is also relatively moister than the atmospheric air. The amount and composition of soil air is determined by the water content of soil unless soil is very dry. The O_2 content in a well-aerated soil is higher than that of a poorly aerated soil. The latter has higher concentrations of CO_2 , methane (CH_4), and nitrous oxide (N_2O) than atmospheric air. A soil is considered healthy if the air-filled pore spaces are about 50% of the total porosity and composition of the soil air is similar to that of atmospheric air.

MEASUREMENT OF SOIL AERATION

Measurement of soil aeration involves assessing: 1) composition of soil air and 2) rate of diffusion of O_2 from atmosphere into the soil. Measurement of the relative concentrations of O_2 , CO_2 , and other gases in the soil air provides important information on the aeration and soil structure. Depletion of O_2 level content in soil air is a good indicator of the restricted gas exchange in the soil matrix. This method, although static, is better than the measurement of air volume alone. However, it requires extraction of a sample that is large enough to provide a measurement, but at the same time small enough to be representative. Another

drawback of this method is soil disturbance and contamination or mixing of air from the atmosphere. The measurement of depletion of O_2 or increase in CO_2 can be made both *in situ* or in a laboratory by gas chromatography technique which provides reliable measurements. The repeated measurements of O_2 concentrations in soil air without extracting a sample can also be obtained by electrode methods. The measurement of soil aeration is carried out at known or standardized soil water suction, at about 50 cm of water.

AIR PERMEABILITY

Air permeability is defined as the ease with which air enters or passes through a bulk mass of soil and has the dimensions of area (square liter). Air permeability of porous media is governed by both the convective transport and the diffusional transport of air. The convective transport occurs under a pressure gradient, whereas diffusional transport occurs due to the change in concentration gradients or the partial pressures of the various components of soil air. Under normal circumstances, both convection and diffusion processes occur simultaneously, provided the concentration and pressure gradients exist concurrently. The mass flow of gas is important when differences in pressure are due to the change in barometric pressure, temperature, or soil water content. Measurement of air permeability of soil is important because it provides valuable information on: 1) soil structure; 2) pore size network; 3) changes in soil structure due to land use and land management practices; and 4) saturated hydraulic conductivity of soil.

Table 1 The measured average gas permeability at highest soil air content for a range of soil water suction interval with respect to clay content of soils.

Clay (%)	k (μm^2)	Soil water suction (cm H_2O)
2.5	14.3	$16 > h > 100$
3.7–4.3	36.0	$10 > h > 500$
5.9	22.8	$50 > h > 500$
16	1225.0	$20 > h > 3000$
21–24	525.0	$10 > h > 1090$

Source: Adapted from Sabatier, Hess, et al.^[1]

Table 1 presents typical air permeability values measured on soil cores.^[2]

BASIC PRINCIPLE

According to Darcy's law for laminar flow (no acceleration along line of flow), velocity of a given fluid is proportional to the pressure difference and inversely proportional to the length of the flow path. Therefore, Darcy's law is applicable for the airflow through soils and air permeability (k_a) can be defined by the following relationship:

$$k_a = q\eta_a \frac{dx}{dp} \quad (1)$$

where q is the volume flux per unit area ($\text{L}^3\text{L}^{-2}\text{T}^{-1}$), η_a the dynamic viscosity of air ($\text{ML}^{-1}\text{T}^{-1}$), p the pressure of air ($\text{ML}^{-1}\text{T}^{-2}$), and x the distance in direction of flow (L).^[3] The dimensions of air permeability coefficient, k_a as L^2 , are similar to the intrinsic permeability of soil, and therefore, k_a is also referred to as intrinsic permeability of air.

MEASUREMENT OF AIR PERMEABILITY

The methods of measuring air permeability can be broadly divided into steady-state methods, unsteady state (or transient) methods, and acoustic methods.

Steady-State Methods

The steady-state methods are commonly categorized into i) constant pressure gradient method and ii) constant flux method.^[4] In constant pressure gradient method, air is blown across a soil core (or repacked soil column) at a constant pressure above the atmospheric pressure (Fig. 1), and the rate of flow of air at the inlet end of the soil core is measured. The constant pressure gradient method is suitable for soil cores at high water content. In constant flux method, a constant air flux is applied at one end of a soil core, and pressure difference across the

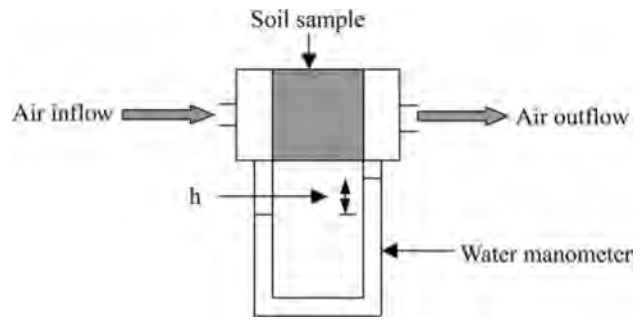


Fig. 1 Schematic of the apparatus to measure air permeability (k_a).

soil core is measured. The soil core is placed inside a chamber and is connected to a water manometer or a pressure gauge to measure pressure difference across the sample. The constant flux method is simple and straightforward and is useful for soils of low air permeability.

Conservation of moisture status of the core sample is an important requirement for air permeability measurement. The water content of the soil core is a function of the pressure difference between both air and water, and the pressure gradient between these two competing fluids should be equal in magnitude and direction. One of the methods is stationary liquid method,^[5] in which air flows upward and through the sample in response to a pressure gradient equal to that in a static liquid. The other method uses both fluids flowing with equal pressure gradient in any direction. The steady-state laboratory methods described till now are reasonably accurate and reproducible for disturbed or rock samples. The merits and demerits of air permeability measurement methods are given in Table 2.

Transient Methods

Transient methods are practical, rapid, economic, and easy to use. They can be employed for air permeability measurements both in the laboratory on soil cores and directly in the fields. They also require less volume of air to pass through the soil core or soil volume for a given permeability determination. The duration of the tests is shorter, and soil desaturation or drying is less as compared to the steady-state method. In transient *in situ* method (Figs. 2 and 3), one end of a soil core is connected to a close pressurized air tank and the other end is kept open.^[3] The rate of drop of air pressure in the tank is measured as the air flows out through the other end and is used to calculate the air permeability of soil.^[6] The field air permeameter has been modified to include a gas cylinder and a water manometer to replace the float.^[7] The apparatus is suitable for air permeability measurements *in situ*, on-site, and in laboratory on soil cores. Traditionally, air permeability calculations are made for isothermal conditions. However,

Table 2 Merits and demerits of air permeability measurement methods.

Steady-state laboratory method		Transient method		Acoustic method
Pressure gradient	Flux method	Core method	Field method	Field method
Constant pressure gradient	Constant flux	Drop in pressure in air tank	Drop in pressure in air tank	Reflection and transmission of audio frequency
Easy and simple	Constant flux and gradients are difficult to attain	Rapid and easy	Practical, rapid, economical, and easy	Rapid but requires skilled labor
Suitable for highly permeable soils	Suitable for less permeable soil	Suitable for both	Suitable for both	Suitable for homogeneous soils
Does not alter water content	Water content is altered	Does not alter water content	Does not alter water content	Does not alter water content
Disadvantage of airflow between soil and core	Disadvantage of airflow between soil and core	Disadvantage of airflow between soil and core	—	—
—	Soil shrinkage	—	—	—
Well developed	Well developed	Well developed	Well developed	Underdevelopment

air permeability can also be calculated for the nonisothermal conditions, arising due to temperature changes inside the gas cylinder.^[6]

Acoustic Methods

Sound reflected from a soil surface interferes with the incident sound and causes an interference pattern in the total sound field. If the acoustic properties of porous media are known, interference patterns can be modeled from theory of sound propagation. Analytic approximations for calculating the sound levels from outdoor sound sources on or near the ground in terms of acoustic properties of soil are available.^[8] For a homogeneous porous medium, these acoustic properties depend on the air-filled porosity of soil surface.^[9] The acoustic techniques involve the measurement of both the reflection and the transmission of audio frequency broadband sound by the soil. The sound reflection measurements give qualitative indications of relative air permeability. Sound transmission measurements are made by inserting a probe microphone at a given depth and keeping another vertically separated above ground. Theoretical predictions for homogeneous soils are fitted to the

measured reflection, and quantitative information on surface air-filled porosity and air permeability is obtained. The acoustic techniques have been validated on a series of trial plots for a variety of soils and have been found within 10% of those obtained conventionally and have shown a potential for the measurement and monitoring of management induced or seasonal changes in soil surface properties.^[2]

CONCLUSION

Air permeability is sensitive to changes in soil structure resulting from different land use and soil management practices. Air permeability of soils is recognized as an important parameter for soil aeration and contaminant remediation techniques and is fundamental to our understanding of environmental problems in vadose zone (or root zone

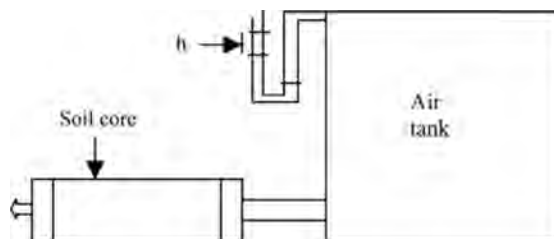


Fig. 2 Schematic of an air permeability apparatus on soil cores for transient method.

Source: From Kirkham.^[3]

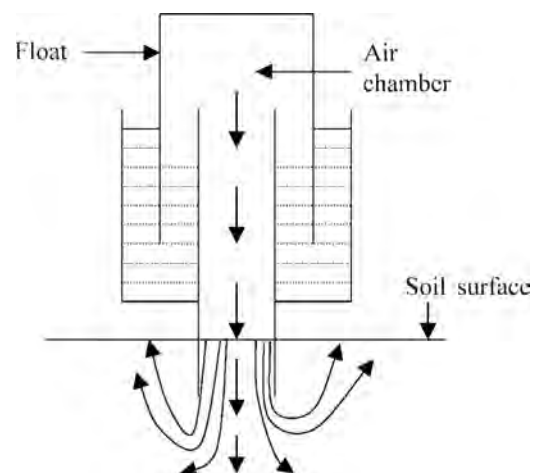


Fig. 3 Schematic of an *in situ* air permeability apparatus for transient method.

Source: From Grover.^[10]

depth). Knowledge of air-filled pores, pore size distribution, tortuosity, air permeability, and their variation along the cross section or depth is important to describe aeration, structure, and compaction of the soil. Air permeability can be measured both in the laboratory and on the field by steady-state as well as transient methods. Transient methods are better than steady-state methods, as they are more practical, rapid, economic, and easy to use.

REFERENCES

1. Sabatier, J.M.; Hess, H.; Arnott, W.P.; Attenborough, K.K.; Romkens, M.J.M.; Grissinger, E.H. *In situ* measurements of soil physical properties by acoustic techniques. *Soil Sci. Soc. Am. J.* **1990**, *54*, 658–672.
2. Moldrup, P.; Poulsen, T.G.; Schjonning, P.; Olsen, T.; Yamaguchi, Y. Gas permeability in undisturbed soils: Measurement and predictive models. *Soil Sci.* **1998**, *163* (3), 180–189.
3. Kirkham, D. Field methods for determination of air permeability of soil in its undisturbed state. *Soil Sci. Soc. Am. Proc.* **1946**, *11*, 93–99.
4. Grover, B.L. Simplified air permeameter for soil in place. *Soil Sci. Soc. Am. Proc.* **1955**, *19*, 414–418.
5. Brooks, R.H.; Corey, A.T. *Hydraulic Properties of Porous Media*. Hydrology Paper No. 3; Colorado State University: Fort Collins, 1964.
6. Smith, J.E.; Robin, M.J.L.; Elrick, R.R. A source of systematic error in transient flow permeameter measurements. *Soil Sci. Soc. Am. J.* **1997**, *61*, 1563–1568.
7. Iversen, B.V.; Schjonning, P.; Poulsen, T.G.; Moldrup, P. *In situ*, on-site and laboratory measurements of soil air permeability: Boundary conditions and measurement scale. *Soil Sci.* **2000**, *166* (2), 97–106.
8. Attenborough, K. Acoustic impedance models for outdoor ground surfaces. *J. Sound Vib.* **1985**, *99*, 521–544.
9. Attenborough, K. On the acoustic slow wave in air filled granular media. *J. Acoust. Soc. Am.* **1987**, *81*, 93–102.
10. Grover, B.L. Simplified air permeameter for soil in place. *Soil Sci. Soc. Am. Proc.* **1956**, *19*, 414–418.

Albedo

Endre Dobos

*Department of Physical Geography and Environmental Sciences,
University of Miskolc, Miskolc-Egyetemváros, Hungary*

Abstract

The fraction of the incident radiation that is reflected from the surface is called the albedo. Albedo plays a major role in the energy balance of the earth's surface, as it defines the rate of the absorbed portion of the incident solar radiation. Soil albedo is a complex feature, which is determined by many soil dependent and independent (environmental) characteristics. The process that results in the reflected radiation is called reflectance, whereby the energy of radiation is reradiated by the chemical constituents (e.g., atoms or molecules) of the surface layer approximately half the thickness of wavelength. The portion of solar radiation not reflected by the earth's surface is absorbed by the soil or the vegetation, which interacts with the incident radiation. The absorbed energy can increase the soil temperature or the rate of evapotranspiration from the surface of the soil–vegetation system. Some of the energy that is absorbed and transformed into heat is reradiated at a longer wavelength than the incoming radiation. That is why the peak terrestrial radiation occurs in the infrared spectrum, while the peak incident radiation occurs in the blue–green portion of the visible spectrum.

INTRODUCTION

The albedo value ranges from 0 to 1. The value of 0 refers to a blackbody, a theoretical media that absorbs 100% of the incident radiation. Albedo ranging from 0.1 to 0.2 refers to dark-colored, rough soil surfaces, while the values around 0.4–0.5 represent smooth, light-colored soil surfaces. The albedo of snow cover, especially the fresh, deep snow, can reach as high as 0.9. The value of 1 refers to an ideal reflector surface (an absolute white surface) in which all the energy falling on the surface is reflected. The mean albedo of the earth system is 0.36 ± 0.06 (Table 1).^[1]

FACTORS AFFECTING ALBEDO

Albedo varies diurnally and seasonally due to the changing sun angle.^[2,3] In general, the lower the sun angle, the higher the albedo. Besides the sun angle, many of the surface characteristics have large impact on the albedo. The most significant factors affecting the soil albedo are the type and condition of the vegetation covering the soil surface, soil moisture content, organic matter content, particle size, iron oxides, mineral composition, soluble salts, and parent material.^[4]

The type and the condition of the vegetation have a strong impact on the surface albedo. Forest vegetation with multilevel canopy has a low albedo because the incident radiation can penetrate deeply into the forest canopy where it bounces back and forth between the

branches and leaves and get trapped by the canopy.^[5] The albedo for grassland and cropland ranges between 0.1 and 0.25.^[6–9]

Changes in soil moisture content change the absorbance and reflectance characteristics of the soil. Increase in soil moisture content increases the portion of the incident solar radiation absorbed by the soil system. This relationship is well known and used for soil color differentiation when the Munsell color chart is used. The colors of dry and moist soil samples are always different. The higher the soil moisture content, the darker the color and the lower the albedo. However, this relationship is valid only for soil moisture contents up to the field capacity. Beyond field capacity, the increase in soil moisture content does not darken the color anymore but starts building up a water sheet on the aggregate surface, creating a shiny and better reflecting surface, which increases the reflectance and thus the albedo. This phenomenon is the major reason for differences in the albedo among soils of different textural classes. Clayey soils can maintain high moisture content in the presence of water supply, while the sandy textured soils drain and dry out much more rapidly. Due to the differences in the resulting soil moisture content between the texture classes, there are differences in the reflectance and absorbance characteristics and so in the albedo (Fig. 1).

Surface roughness defines the type of reflection. Shiny, smooth surfaces, like water body, plant leaves, or wet soil surfaces may be near-perfect, specular reflectors, which may reflect well and show relatively high albedo for lower sun angles. Rough surfaces represent

Table 1 The approximated ranges of albedo of natural surfaces.

Natural surface types	Approximated albedo
Blackbody	0
Forest	0.05–0.2
Grassland and cropland	0.1–0.25
Dark-colored soil surfaces	0.1–0.2
Dry sandy soil	0.25–0.45
Dry clay soil	0.15–0.35
Sand	0.2–0.4
Mean albedo of the earth	0.36
Granite	0.3–0.35
Glacial ice	0.3–0.4
Light-colored soil surfaces	0.4–0.5
Dry salt cover	0.5
Fresh, deep snow	0.9
Water	0.1–1
Absolute white surface	1

lower albedo values, especially when sun angle is low and the shading effect lowers the reflection. There are measurable differences in the surface roughness among soil textural classes. Fine-textured, dry soils with small particle size produce high albedo due to relatively

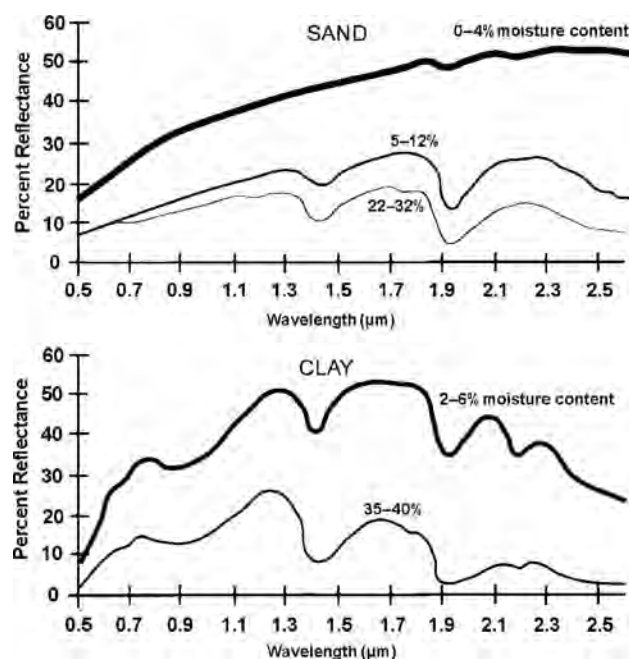


Fig. 1 The higher the moisture content the lower the reflectance throughout the visible and near-infrared region, especially along the water absorption bands at 1.4 and 1.7 μm . Notice the differences in the reflectance characteristics between the clayey and sandy soils.

Source: From Hoffer.^[10]

smooth surface. However, clayey soils are often wet, and soil moisture absorbs the incident radiation and decreases albedo. Conversely, dry, coarse-textured soils with relatively large particles (sand grains) reflect the larger portions of incident radiation than clayey soils.

Surface color is determined by the interaction of the surface material with the visible spectra of the incident solar radiation. Soil color is a differentiating factor in all the soil classification systems. It reflects many of the most important soil physical and chemical characteristics. One of the most significant coloring agents of the soils is the soil organic matter content. Soil organic matter content increases the absorbance of the soil. Thus, the higher the organic matter content, the lower the albedo. Iron oxides increase the reflectance in the red portion of the spectrum while causing a decrease in the blue-green and infrared portion. Salt crust on the surface increases the albedo dramatically. That is why mapping of salt-affected area with remotely sensed images is a very powerful tool for soil surveyors.

MEASUREMENT OF ALBEDO

The theoretical concept of measuring albedo is simple. A radiation sensor (pyranometer) is pointed upward to measure the incident radiation and then quickly flipped downward to measure the reflected radiation. For deriving the albedo, the quantity of the reflected radiation has to be divided by the one for the incident radiation. In fact, the actual measurement of surface albedo under natural condition is rather complex. The problem is threefold. Firstly, the incident radiation does not only come from the radiation source directly but also from diffused light from other directions. Secondly, the reflector surfaces do not reflect equally in all directions, and thirdly, the sensors gather light only from a small range of angles. Thus, our measurements of reflectance are only samples of the bidirectional reflectance distribution function. Albedo is often defined as an overall average reflection coefficient of an object. More precisely, the terms of spectral and total albedo are differentiated. The spectral albedo refers to the reflectance in a given wavelength, while the albedo is calculated as an integral of the spectral reflectivity times the radiation, over all wavelengths in the visible spectrum. A good estimation of the surface albedo can be done using clear-sky satellite measurements.^[11]

CONCLUSION

Albedo measures the overall reflectance of the surface. It provides lots of useful information about the soil system and helps to better understand the soil energy balance. But different wavelengths of sunlight are normally not equally reflected, which gives rise to a variable color of

surfaces and differences in reflectance of certain wavelengths due to differences in physical or chemical characteristics of the soil surface. Differences in soil albedo can be measured with radiometers.

REFERENCES

1. Weast, R.C.; Astle, M.J., Eds. *Handbook of Chemistry and Physics*, 63rd Ed.; CRC Press: Boca Raton, 1982.
2. Matthews, E. Vegetation, land-use and seasonal albedo data sets. In *Global Change Data Base Africa Documentation*. Appendix D; NOAA/NGDC, 1984.
3. Kotoda, K. Estimation of river basin evapotranspiration. Environmental Research Center Papers; University of Tsukuba, 1986; Vol. 8, p. 166.
4. Baumgardner, M.F.; Sylva, L.F.; Biehl, L.L.; Stoner, E.R. Reflectance properties of soils. *Adv. Agron.* **1985**, 38, 1–44.
5. Geiger, R. *The Climate near the Ground*, 4th Ed.; Harvard University Press: Cambridge, 1965; 501 pp.
6. Jones, H.G. *Plants and Microclimate: A Quantitative Approach to Environmental Plant Physiology*, 2nd Ed.; The Cambridge University Press: Cambridge, 1992.
7. Jensen, M.E.; Burman, R.D.; Allen, R.G. Evapotranspiration and irrigation water requirements. In *ASCE Man. Rep. Pract. 70*; American Society of Civil Engineers: New York, 1990.
8. Oke, T.R. *Boundary Layer Climates*; Methuen: New York, 1978.
9. Van Wijk, W.R.; Scholte Ubing, D.W. Radiation. In *Physics of Plant Environment*; Van Wijk, W.R., Ed.; North-Holland Publishing Co.: Amsterdam, 1963; 62–101.
10. Hoffer, R.M. Biological and physical considerations in applying computer-aided analysis techniques in remote sensing data. In *Remote Sensing: The Quantitative Approach*; Swain, P.H., Davis, S.M., Eds.; McGraw Hill: San Francisco, 1978; 227–289.
11. Li, Z.; Garand, L. Estimation of surface albedo from space: A parameterization for global application. *J. Geophys. Res.* **1994**, 99, 8335–8350.

Alfisols

Rienk Miedema

Department of Soil Science and Geology, Wageningen University, Wageningen, the Netherlands

Abstract

Alfisols occur almost everywhere, provided that conditions suitable for clay illuviation are met, either presently or in the past. The subdivisions of Alfisols are based on climate. Medium-textured, decalcified, weakly acidic parent materials like loess and glacial deposits provide suitable conditions for clay illuviation, while the leaching of basic cations has not progressed too far and a high cation exchange capacity is maintained.

INTRODUCTION

The definition^[1] of the soil order of Alfisols is as follows: Soils do not have a plaggen epipedon and have either 1) an argillic or kandic horizon (base saturation 35% or more) or a natric horizon, or 2) a fragipan with clay films of thickness 1 mm or more in some parts. The argillic, kandic, and natric horizons are formed as illuvial subsurface horizons, characterized by a significantly higher percentage of clay than the overlying eluvial horizon. The fine fraction of clay is strongly involved in the clay illuviation process (the ratio of fine clay to total clay is the highest in the argillic horizon).

The argillic and the kandic horizons differ because the kandic horizon has over more than half its thickness an apparent cation exchange capacity (CEC) of 16 cmol(+) or less per kg clay.

The natric horizon is an argillic horizon with: 1) columnar or prismatic structures in its upper part with or without uncoated silt and sandgrains in the top of the horizon and 2) an exchangeable sodium percentage of 15 or more, a sodium (Na) adsorption ratio of 13 or more in the upper part of the horizon, or more exchangeable magnesium plus Na plus exchangeable acidity in the upper part of the horizon. Alfisols, distinguished on the basis of a fragipan (firm rupture resistance and slaking of water-submerged air-dry fragments), also show clear evidence of clay illuviation,^[2] but the relation between fragipans and argillic horizons is controversial.^[3–5] Alfisols are very reviewed.^[6] The objective of this short communication is to reflect, in a general way, on the (palaeo)environmental conditions under which clay illuviation, the main soil-forming process in Alfisols, takes place. The areal extent and geographical distribution of Alfisols are summarized.

THE CLAY ILLUVIATION PROCESS

Particulate dispersed (fine) clay is liberated in the surface soil, is transported down the profile, and accumulates

(precipitates) in the subsurface horizon. Which conditions promote the liberation of dispersed (fine) clay from (micro)-aggregates in the surface soil?

These conditions can be considered from a (bio)physical and biochemical point of view. The clay mineralogy is also of importance: Smectitic clays disperse to finer clay particles than kaolinitic clays (as in the kandic horizon).

From the (bio)physical point of view, the moisture content of the soil material is important. Suddenly wetted dry aggregated soil material may experience “air explosion.” The wetting front proceeds to the center of the dry aggregate, the trapped air is compressed, the aggregate explodes, and fine and coarse material is released. The porosity and chemical characteristics of the original aggregates determine whether air explosion happens. Very porous aggregates do not experience air explosion. Soil material with many Ca^{2+} or Al^{3+} bonds forms stable aggregates that do not produce fine clay upon air explosion.

The ratio of water to solid is important.^[7] When the water content is extremely high, this may promote the liberation of fine clay. From the (bio)chemical point of view, aggregate stability/clay flocculation is important. Soil material with a high percentage of exchangeable Ca^{2+} or Al^{3+} clay remains flocculated even with extremely high ratios of water to solid. Humic substances play a role in the liberation of fine clay, related to the clay mineralogy.^[8,9]

It is possible to indicate pH ranges in the soil, where liberation of fine clay is possible.^[10–12] In soils with a pH below 5, the amount of Al^{3+} in the soil solution and the amount of exchangeable Al^{3+} are so high that the clay remains flocculated.

In soils with a pH of 7–8, the amount of Ca^{2+} in the soil solution and on the exchange complex is so high that the clay remains flocculated. In soils with a high pH (8.5 or more), Na^+ is strongly dominant and the liberation of dispersed fine clay is promoted due to a very extended electric double layer (natric horizon).

Hence, the prerequisites for soils to become Alfisols are as follows: 1) decalcification of originally calcareous parent

materials; 2) not very acidic parent materials; and 3) strongly alkaline parent materials.

Downward transport of the fine clay occurs as laminar flow with water along the pore walls.^[13,14] Turbulent flow moves coarser particles downward.

Accumulation (precipitation) of the liberated (fine) clay with adsorbed iron (Fe) may occur through a number of mechanisms.

Chemically, a calcareous subsurface layer (Fig. 1B) forces the fine clay to flocculate and precipitate on the pore wall.

Stratification may trigger the accumulation of the (fine) clay at boundaries of finer and coarser textured soil materials. In sandy soils, this leads to the characteristic “banded B horizons” with bridges of fine clay between the sandgrains.^[16,17]

Water absorption in a dry subsoil from pores may also prompt precipitation of fine clay on the pore wall.

Recognition of Illuvial Clay

Macromorphologically, clay coatings in pores or on pedfaces, distinguished by their more shiny appearance, are noted in the profile description. To judge the composition of the coatings with the naked eye or with a hand lens is difficult.

The accumulation of fine clay in the argillic horizon leads to the parallel arrangement of clay particles (continuous orientation) and distinctive optical properties such as strong birefringence and extinction phenomena (Fig. 2A and B) in thin sections (30 μm thick) of undisturbed soils.^[19,20]

In macroscopically identified argillic horizons, and sometimes in micromorphologically, very little clay coatings are observed.^[21,22] Recognizing and counting

fragments of clay coatings and distinguishing between illuvial clay coatings and stress coatings are not always easy. Strong swell–shrink activity of smectitic clay in certain soils leads to argillic horizons “without clayskins.”^[23]

Quantification of Clay Illuviation

In micromorphological quantification of clay illuviation,^[15,24,25] different trained operators obtain different results on the same samples (coefficients of variance of 50%).^[24,25]

Volume percentages of micromorphologically illuviated clay in argillic horizons range from about 1 to 20%. Well-developed argillic horizons in loess typically contain 4–7% illuviated fine clay micromorphologically (Fig. 1A and B).

Pedological Translocations/Transformations of Clay Coatings

The accumulated fine clay is present as films coatings or pore walls. Subsequent biological activity may translocate fragments of the coatings into the groundmass of the soil.

The accumulation of clay in the subsurface horizon may lead to problems with the hydraulic conductivity in that horizon. Redox processes (pseudogley) in the upper part and along cracks and pores in the argillic horizon lead to degradation features^[3,26,27] in the argillic horizon: clay coatings (partly) covered with Fe/manganese precipitates with less Fe content or with grainy clay coatings (clay destruction) by ferrololysis.^[28] In Mediterranean^[29] and semiarid areas, argillic horizons are found with clay illuviation and subsequent recarbonation from eolian dust (secondary calcite coatings covering fine clay coatings).

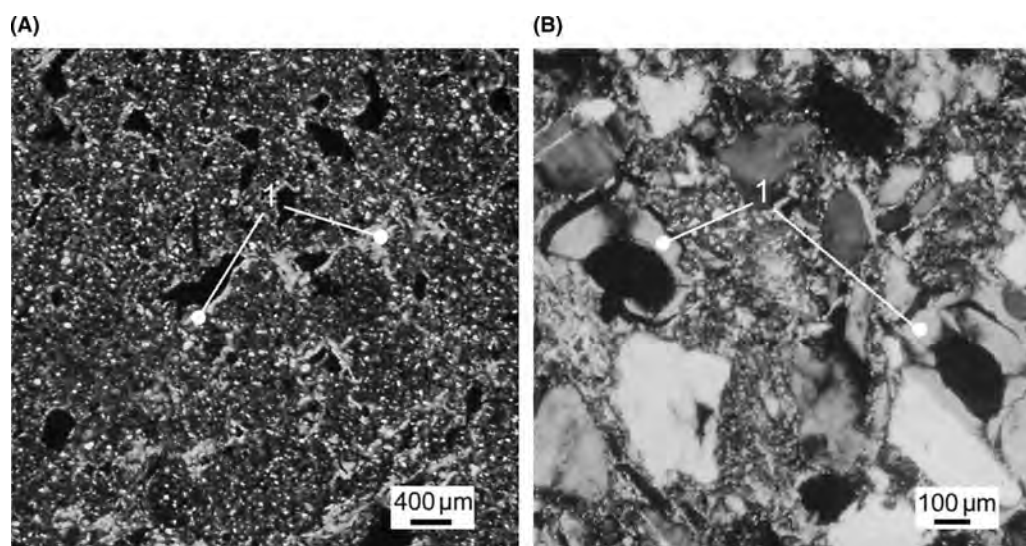


Fig. 1 Micromorphological quantification of clay coatings and granulometric clay content in (A) an udalf in loess from the Netherlands and (B) in an ustalf in loess from Argentina.

Source: Adapted from Miedema & Slager.^[15]

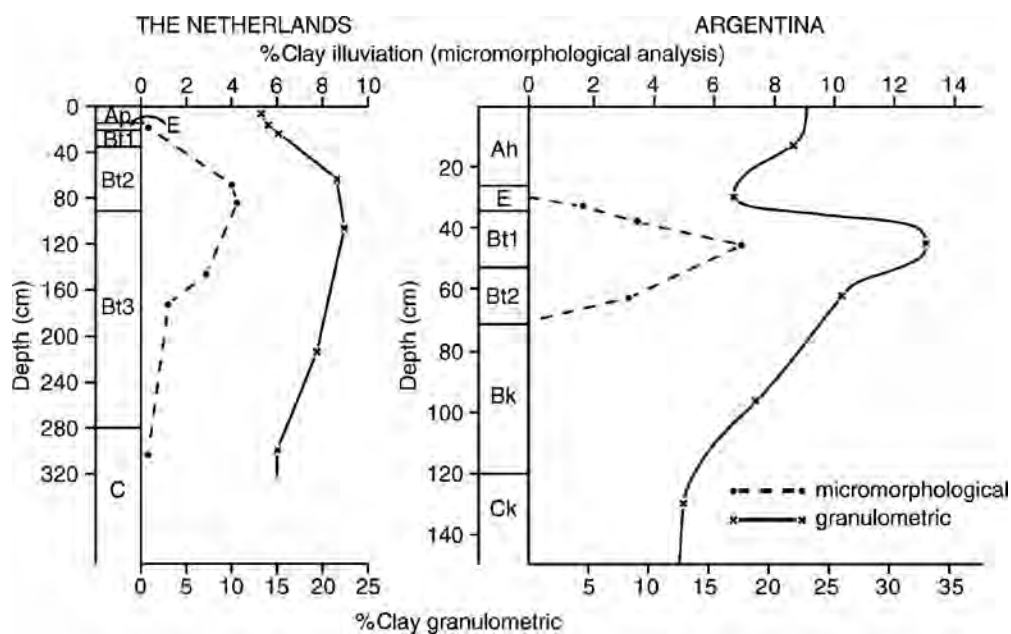


Fig. 2 Micromorphological images (crossed polarizers) of illuvial clay coatings (1) in (A) a glossudalf from Russia and (B) an ochraqalf from the Netherlands.

Source: Adapted from Miedema, Koulechova, et al.^[18]

TIME OF FORMATION OF THE ARGILLIC HORIZON

The optimum conditions for the occurrence of clay illuviation (argillic horizon) are much disputed. One school of thought considers clay illuviation to occur in

interglacial periods, such as the Holocene period.^[4,19,30–32] Widespread semideciduous and mixed broadleaf–coniferous natural forest vegetation (through humic acids in their topsoil) is thought to create conditions optimally suitable for the liberation of fine clay in the topsoil.

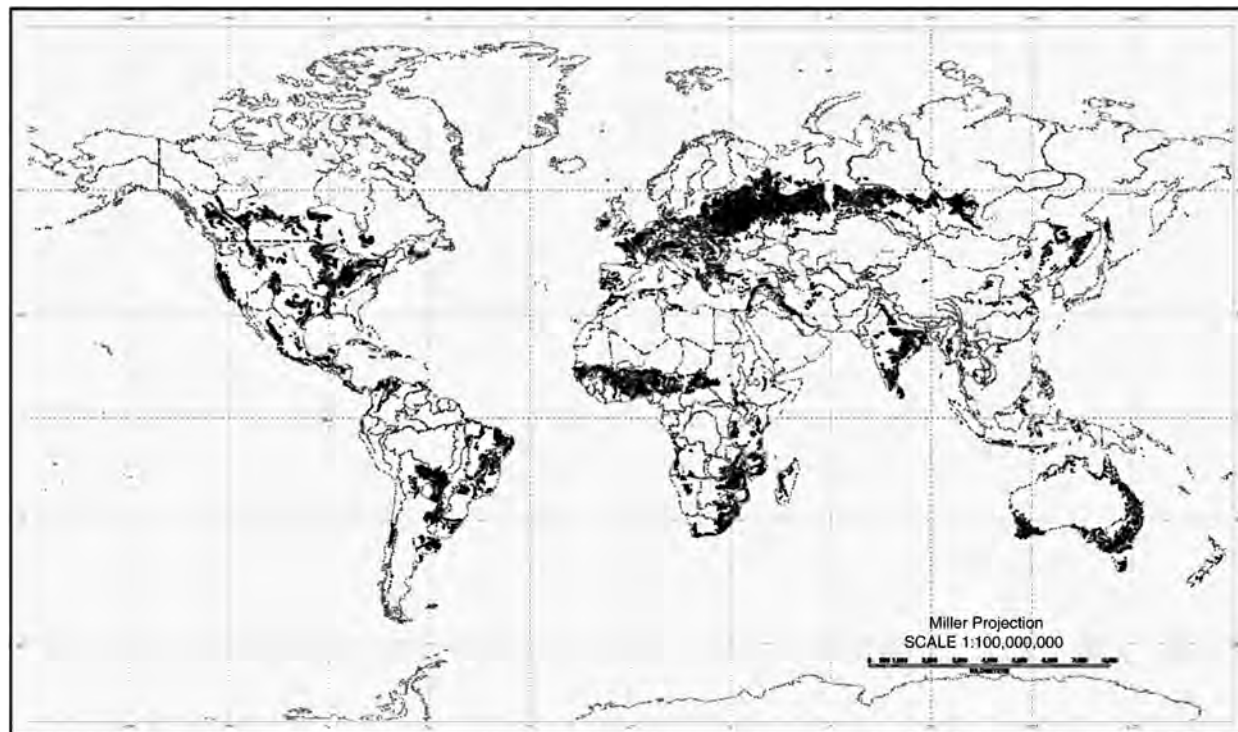


Fig. 3 Geographical distribution of Alfisols.

Source: U.S. Department of Agriculture, Nature Resources Conservation Service, Soil Survey Division, World Soil Resources, © U.S. Department of Agriculture.

Another school of thought^[33–36] considers the optimum period for clay illuviation to occur at the end of a glacial period or at the beginning of an interglacial or interstadial period. Following rapid Late Glacial decalcification of originally calcareous parent materials, the conditions for liberation of fine clay occur in the melting topsoil. This topsoil (active layer) is freeze-dried in frozen conditions, and the melting of ice lenses produces sudden wetting, air explosion, and liberation of fine clay.

Relict argillic horizons occur in Late Pleistocene loess soils below tidal marshes (submergence due to postglacial sea level rise) in Maryland.^[37]

The Late Glacial conditions are also conducive for the formation of a fragipan or fragic characteristics^[34,38] in loess and glacial till in North America,^[2] Western Europe,^[34] and Russia.^[18] Distinction is made between argillic horizons and paleo-argillic horizons.^[39] Paleo-argillic horizons are not associated with soil-forming conditions.^[40–42]

GEOGRAPHICAL DISTRIBUTION AND AREAL EXTENT

Alfisols (total areal extent of which is 9.6% of the ice-free land surface of the earth) occur almost everywhere, provided that conditions suitable for clay illuviation are met, either presently or in the past (Fig. 3). The subdivisions of Alfisols are based on climate (temperature and precipitation). Suborders comprise the cryalfs of Arctic areas, the udalfs of temperate humid areas and the ustalfs and xeralfs of semiarid and/or dry areas. A large proportion of the Alfisols occurs in the temperate and Mediterranean zone. Medium-textured, decalcified, weakly acidic parent materials like loess and glacial deposits provide suitable conditions for clay illuviation, while the leaching of basic cations has not progressed too far, and a high CEC is maintained.

Landscape position (degradation of Alfisols) is accommodated in the suborder of the aqualfs.^[43,44]

REFERENCES

1. Soil Survey Staff. In *Keys to Soil Taxonomy*, 8th Ed.; USDA-NRCS US Government Printing Office: Washington, 1998.
2. Smeck, N.E.; Ciolkosz, E.J., Eds. *Fragipans: Their Occurrence, Classification and Genesis*; Soil Sci. Soc. Am. Spec. Publ. 24; Soil Sci. Soc. Am. Spec. Publ.: Madison, 1989.
3. James, H.R.; Ransom, M.D.; Miles, R.J. Fragipan genesis in polygenetic soils on the springfield plateau of Missouri. *Soil Sci. Soc. Am. J.* **1995**, *59*, 151–160.
4. Lindbo, D.L.; Rhoton, F.E.; Bigham, J.M.; Hudnall, W.H.; Jones, F.S.; Smeck, N.E.; Tyler, D.D. Loess toposequences in the lower Mississippi river valley. I. Fragipan morphology and identification. *Soil Sci. Soc. Am. J.* **1995**, *59*, 487–500.

5. Lindbo, D.L.; Rhoton, F.E.; Hudnall, W.H.; Smeck, N.E.; Bigham, J.M. Loess stratigraphy and fragipan occurrence in the lower Mississippi river valley. *Soil Sci. Soc. Am. J.* **1997**, *61*, 195–210.
6. Hallmark, C.T.; Franzmeier, D.P. Alfisols. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 1999; E338–E359.
7. Gombeer, R.; d'Hoore, J. Induced migration of clay and other moderately mobile constituents. III. Critical soil/water dispersion ratio, colloid stability and electrophoretic mobility. *Pédologie* **1971**, *21*, 311–342.
8. Kaplan, D.I.; Bertsch, P.M.; Adriano, D.C.; Miller, W.P. Soil-borne mobile colloids as influenced by water flow and organic carbon. *Environ. Sci. Technol.* **1993**, *27*, 1193–1200.
9. Kretzschmar, R.; Robarge, W.P.; Weed, S.B. Flocculation of kaolinitic soil clays: Effect of humic substances and iron oxides. *Soil Sci. Soc. Am. J.* **1993**, *57*, 1277–1283.
10. Jenny, H.; Smith, G.D. Colloid chemical aspects of clay pan formation in soil profiles. *Soil Sci.* **1935**, *39*, 377–389.
11. Beery, M.; Wilding, L.P. The relationship between soil pH and base saturation percentage for surface and subsoil horizons of selected mollisols, alfisols and ultisols in Ohio. *Ohio J. Sci.* **1971**, *71*, 43–55.
12. van Breemen, N.; Buurman, P. *Soil Formation*; Kluwer Academic Publishers: Dordrecht, 1998.
13. Hudson, B.D. Cohesive water films as a factor in clay translocation. *Soil Surv. Horizons* **1977**, *18*, 9–15.
14. Dalrymple, J.B.; Theocharopoulos, S.P. Intrapedal cutans—an experimental production of depositional (illuviation) channel argillans. *Geoderma* **1984**, *33*, 237–243.
15. Miedema, R.; Slager, S. Micromorphological quantification of clay illuviation. *J. Soil Sci.* **1972**, *23*, 309–314.
16. Dijkerman, J.C.; Cline, M.G.; Olson, G.W. Properties and genesis of textural subsoil lamellae. *Soil Sci.* **1967**, *104*, 7–16.
17. Torrent, J.; Nettleton, W.D.; Borst, G. Clay illuviation and lamella formation in a Psammentic Haploxeralf in southern California. *Soil Sci. Soc. Am. J.* **1980**, *44*, 363–369.
18. Miedema, R.; Koulechova, I.N.; Gerasimova, M.I. Soil formation in greyzems in Moscow district: micromorphology, chemistry, clay mineralogy and particle size distribution. *Catena* **1999**, *34*, 315–347.
19. McKeague, J.A. Clay skins and argillic horizons. In *Soil Micromorphology*; Bullock, P., Murphy, C.P., Eds.; Academic Publishers: Berkhamsted, 1983; Vol. I, 367–388.
20. Bullock, P.; Thompson, M.L. Micromorphology of alfisols. *Soil Sci. Soc. Am. Spec. Publ.* **1985**, *15*, 17–48.
21. McKeague, J.A.; Guertin, R.K.; Pagé, F.; Valentine, K.W.G. Micromorphological evidence of illuvial clay in horizons designated Bt in the field. *Can. J. Soil Sci.* **1978**, *58*, 179–186.
22. Bronger, A. Argillic horizons in modern loess soils in an ustic moisture regime: Comparative studies in forest-steppe and steppe areas from Eastern Europe and the United States. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer: New York, 1991; Vol. 15, 41–90.
23. Nettleton, W.D.; Flach, K.W.; Brasher, B.B. Argillic horizons without clay skins. *Soil Sci. Soc. Am. Proc.* **1969**, *33*, 121–125.
24. McKeague, J.A.; Guertin, R.K.; Valentine, K.W.; Belisle, J.; Bourbeau, G.A.; Michalyna, W.; Hopkins, L.; Howell, L.

- Pagé, F.; Bresson, L.M. Variability of estimates of illuvial clay in soils by micromorphology. *Soil Sci.* **1980**, *129*, 386–388.
25. Murphy, C.P. Point counting pores and illuvial clay in thin sections. *Geoderma* **1983**, *31*, 133–150.
 26. Bullock, P.; Milford, M.H.; Cline, M.G. Degradation of argillic horizons in udalf soils of New York state. *Soil Sci. Soc. Am. Proc.* **1974**, *38*, 621–628.
 27. Ransom, M.D.; Smeck, N.E.; Bigham, J.M. Stratigraphy and genesis of polygenetic soils on the illinoian till plain of southwestern Ohio. *Soil Sci. Soc. Am. J.* **1987**, *51*, 135–141.
 28. Brinkman, R.; Jongmans, A.G.; Miedema, R.; Maaskant, P. Clay decomposition in seasonally wet acid soils: Micromorphological, chemical and mineralogical evidence from individual argillans. *Geoderma* **1973**, *10*, 259–270.
 29. Fedoroff, N. Clay illuviation in red Mediterranean soils. *Catena* **1997**, *28*, 171–189.
 30. Catt, J.A. Recent work on quaternary paleosols in Britain. *Quat. Int.* **1996**, *34–36*, 183–190.
 31. Mûcher, H.J. *Aspects of Loess and Loess-derived Slope Deposits: An Experimental and Micromorphological Approach*. PhD Thesis; University of Amsterdam: Amsterdam.
 32. Hoffman, R.; Blume, H.P. Holocene clay migration as a soil forming process in loamy soils of the moraine landscapes of North Germany. *Catena* **1977**, *4*, 359–368.
 33. Hoeksema, K.J.; Edelman, C.H. *The Role of Biological Homogenization in the Formation and Transformation of Gray-Brown Podzolic Soils*. Transactions of 7th International Congress Soil Science; Madison, 1960; Vol. VI, 402–405.
 34. Van Vliet-Lanoë, B.; Langohr, R. Correlation between fragipans and permafrost with special reference to silty deposits in Belgium and Northern France. *Catena* **1981**, *8*, 137–154.
 35. Van Vliet-Lanoë, B. The genesis and age of the argillic horizon in weichselian loess of Northwestern Europe. *Quat. Int.* **1990**, *5*, 49–56.
 36. Miedema, R. Processus de formation des sols tardiglaciaires et holocènes sur les terrasses alluviales du rhin aux pays Bas. *Sci. du Sol.* **1992**, *30*, 149–168.
 37. Stolt, M.H.; Rabenhorst, M.C. Micromorphology of argillic horizons in an upland/tidal marsh Catena. *Soil Sci. Soc. Am. J.* **1991**, *55*, 443–450.
 38. FitzPatrick, E.A. An indurated horizon formed in permafrost. *J. Soil Sci.* **1956**, *7*, 248–254.
 39. Bullock, P.; Murphy, C.P. Evolution of a paleo-argillic brown earth (paleudalf) from Oxfordshire, England. *Geoderma* **1979**, *22*, 225–252.
 40. Kemp, R.A. Micromorphology of loess–paleosol sequences: A record of paleoenvironmental change. *Catena* **1999**, *35*, 179–196.
 41. Jongmans, A.G.; Feijtel, T.C.J.; Miedema, R.; van Breemen, N.; Veldkamp, A. Soil formation in a quaternary terrace sequence of the allier, limagne, france. macro- and micromorphology, particle size distribution, chemistry. *Geoderma* **1991**, *49*, 215–239.
 42. Dorronsoro, C.; Alonso, P. Chronosequence in almar river fluvial terrace soil. *Soil Sci. Soc. Am. J.* **1994**, *58*, 910–925.
 43. Smith, H.; Wilding, L.P. Genesis of argillic horizons on ochraqualls derived from fine-textured till deposits in Ohio. *Geoderma* **1972**, *24*, 1–16.
 44. Coventry, R.J.; Williams, J. Quantitative relationships between morphology and current soil hydrology in some alfisols in semi-arid tropical Australia. *Geoderma* **1984**, *33*, 191–218.

Allophanes

Leslie Louise Baker

Jodi Johnson-Maynard

University of Idaho, Moscow, Idaho, U.S.A.

Abstract

Allophane is a nanospherical aluminosilicate, the aluminum (Al): silicon (Si) ratio of which can range continuously from values of 2 to 0.9. It forms in several different environments but is most characteristic of parent materials influenced by volcanic tephra, where it is often associated with the related nanotubular aluminosilicate imogolite. The structure of both these materials is based upon a rolled octahedral Al (gibbsite-like) sheet, with individual orthosilicate tetrahedra bonded to the interior; at progressively higher Si contents, the silica is polymerized and the gibbsite sheet may be fragmental or incomplete. The hollow nanoball structure of allophane influences its physical properties, giving it a low bulk density, high water-holding capacity, and high sorption capacity for organic matter, phosphorus, and other nutrients. The unique and distinctive properties of allophanic soils require specific management practices and result in the creation of the Andisol soil order.

INTRODUCTION

Allophanes are poorly crystalline nanoscale aluminosilicates that are especially prevalent in soils formed from volcanic ash. They are frequently found in association with imogolite, most commonly as products of the weathering of volcanic tephra. Allophane minerals impart unique chemical and physical properties to soils in which they are a predominant mineralogical component, and those properties have implications for management of soils. The influence of allophanes on soil behavior is largely responsible for the development of the Andisol soil order, highlighting the importance of allophane minerals in defining the chemical and physical properties of soil and therefore the soil's response to management. The following discussion will focus on the structure and resulting properties of end members, as defined by aluminum (Al):silicon (Si) ratio, of the allophane group. Despite having well-defined crystalline structure in one direction, imogolite will be included in this discussion because it is commonly found in association with allophane.

STRUCTURE

Allophane and imogolite exhibit rolled morphologies with nanometer-scale diameters, and therefore lack the long-range ordered orthogonal lattices observed in macroscopically crystalline materials, although imogolite exhibits long-range order in one direction. Although allophane and imogolite are difficult to characterize due to their small size and lack of macroscopic crystal ordering, research has illuminated much about their structure and formation.

Imogolite

Imogolite exhibits nanotubular morphology. Nanotubes are typically 2 nm in exterior diameter and the strands may be tens of nanometers in length (Fig. 1). These tubes are thought to consist of a rolled gibbsite-like sheet of octahedrally coordinated Al (Fig. 2) with isolated silicon tetroxide tetrahedra bonded to the tube interior to yield an overall Al:Si ratio of 2:1.^[2] Each orthosilicate unit is bonded to three Al atoms and no silica polymerization is observed. As a result of its tubular morphology, imogolite has no long-range structure in the cross-tube direction but does possess long-range structure in the along-tube direction. Computer modeling has confirmed this structure and reproduced the X-ray diffractogram of natural imogolite samples.^[3,4] In synthesis experiments, imogolite has been observed to evolve from nanospherical allophane-like structures upon aging for periods of >2 weeks.^[5]

Allophanes

Owing to their lack of long-range order, allophanes have been difficult to characterize by standard methods such as X-ray diffraction,^[6] and most of the information known about their chemical structures has been determined by methods such as infrared (IR) and nuclear magnetic resonance (NMR) spectroscopy or by computer modeling. The detailed methodologies that are used to study short-range ordered minerals are beyond the scope of this discussion. For information, the reader is directed to the excellent reviews of Dahlgren,^[7] Wada,^[8] and Parfitt.^[6]

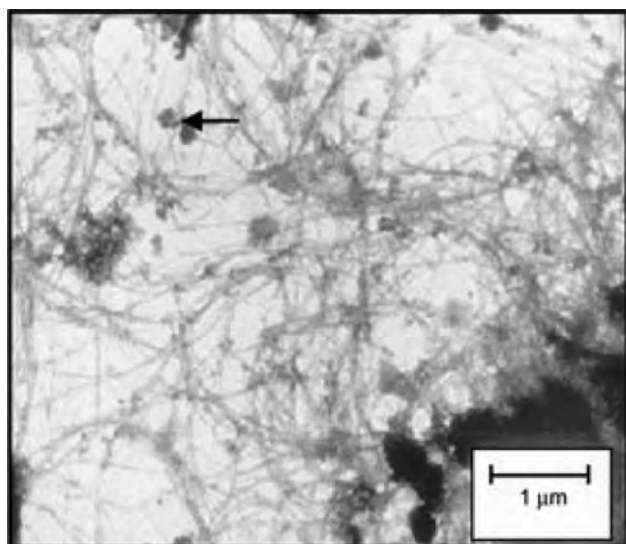


Fig. 1 Electron micrograph showing the fine tubular morphology of imogolite. The arrow indicates spherical allophane units. **Source:** Photo courtesy of R. Dahlgren.

Natural allophanes display a continuous variation in Al:Si ratio and a corresponding variation is observed in their structures. High-Al or proto-imogolite allophane exhibits an Al:Si ratio of approximately 2:1. It is thought to have a hollow, nanospherical structure that is similar in cross section to the cross-tube structure of imogolite.

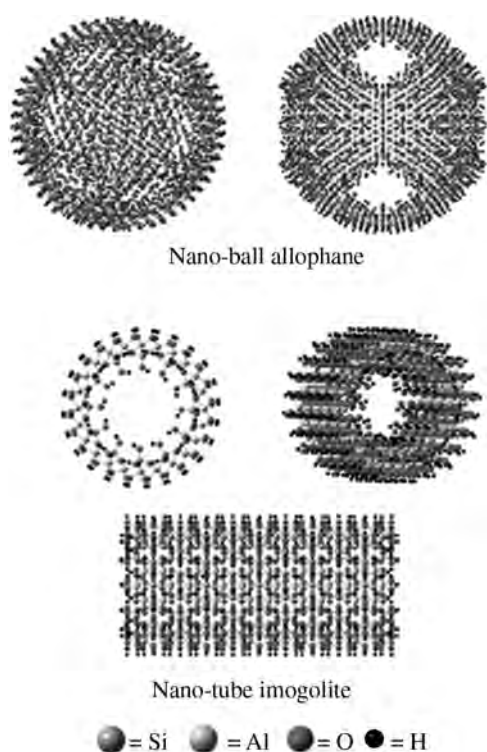


Fig. 2 Modeled allophane porous nanoball (top) and imogolite nanotube (bottom) structures, after Abidin, Matsue, and Henmi.^[1]

In the proposed structure, an exterior gibbsite-like sheet of octahedrally coordinated Al is rolled into a nanoball of diameter 3.5–6 nm (Fig. 2). Orthosilicate units are bonded to the nanoball interior as in imogolite. Computer modeling suggests that four to six pores are present in the nanoball structure.^[1,9] These pores may permit exchange of ions or molecules into or out of the sphere and so are likely to affect the unusual sorption properties of allophanes.

Allophanes with higher Si contents have Al:Si ratios as low as 0.9.^[6] NMR and Fourier transform infrared spectroscopy analyses suggest that high-Si allophanes formed in soil environments have a similar fundamental structure to high-Al allophanes, and that the additional silica is accommodated in a polymerized form in the nanoball interior. The lowest Al:Si ratios are found in “stream-deposit” allophanes that precipitate from silica-saturated stream or spring water. In these highly Si-enriched allophanes, the framework is proposed to be a nanospherical shell of polymerized Si, with a partial or fragmental octahedral Al shell and with some Al substitution in tetrahedral sites.^[10]

OCCURRENCE

Allophane and imogolite are most commonly found in Andisols and tephra-derived Spodosols, although occurrences have been described in other settings.^[6] In general, they precipitate from soil solutions containing Si and Al dissolved from parent material. Laboratory syntheses have produced allophane and imogolite precipitates from millimolar to decimolar solutions of Si and Al. The solution composition affects the composition of the allophane formed with higher silicic acid concentrations leading to lower Al:Si ratios. Alkali and alkaline earth cations in solution have also been found to affect the relative stability of allophane and imogolite by interfering with the dissociation of orthosilicic acid.^[1]

These experimental results illuminate observations of natural systems, summarized by Parfitt.^[6] High-Al and proto-imogolite-type allophanes are characteristics of sil-andic soils, although some soil allophanes display Al:Si ratio close to 1. Si isotopic analyses suggest that allophane precipitates from solution after opaline silica and that its formation is controlled by Al availability, as influenced by Al complexation with organic matter.^[11] Allophane may also form in the absence of tephra, from weathering of aluminosilicate minerals. The presence of organic acids may interfere with allophane formation at lower pH by complexing Al^{3+} ions. This may lead to the formation of alu-andic horizons, which largely lack allophane.

In addition to terrestrial soils, allophane and possibly imogolite have been identified as possible phases on Mars, where they are considered to be likely indicators of tephra weathering under earth-like conditions.^[12]

PROPERTIES

Probably the most well-noted property of allophane minerals is their high water-holding capacity. Imogolite, for example, was shown to have 1.5 times higher water retention than pure sodium (Na)-montmorillonite at -650 J/kg, and water retention of allophane also exceeded that of Na-montmorillonite at above -400 J/kg.^[13] Owing to their porous structures, allophanes have water contents (structural and hydration) ranging from 35.5% to 37.5% by weight.^[14] Allophanic soils, therefore, tend to have relatively high plant-available water contents that are desirable for agricultural and agroforestry production systems. Andisols of the inland Pacific Northwest that support high productivity forests have greater amounts of plant-available water compared to nonandic soils with similar textures (Fig. 3).

Low bulk density is a definitive feature of the Andisol soil order. As Andisols typically have particle densities ranging between 2.5 and 2.7 g/cm³, the relatively low bulk density has been attributed to the development of porous structure.^[15] Bulk density values of allophanic Andisols with relatively low-organic matter content (<3%) decrease with an increase in the allophane content, thereby suggesting that allophane is the main noncrystalline material that results in porous structure.^[16]

It has been noted that Andisols exhibit irreversible changes in water retention, clay dispersibility, and liquid limit upon drying, with the degree of change strongly correlated to oxalate-extractable Al, iron, and Si.^[16] Air-dried allophane, for example, has water contents that range from 23% to 29% of fresh allophane.^[13] The alteration of physical properties after drying has been attributed to

irreversible aggregation.^[13] Allophane, even after treatment to enhance dispersion, tends to form chain-like flocs or domains that alter its physical properties.^[17]

Because of its charge characteristics and high surface area, allophane acts as an important sorbent in soils. Strong sorption of phosphorus (P) on allophane has particularly important implications for nutrient cycling in soils. It has been argued that P sorbs to reactive Al–OH groups in allophane.^[6] However, several studies suggest rather that P sorbs to allophane by displacing weakly bonded orthosilicate groups, damaging the allophane structure.^[18,19] Total P sorption depends on allophane Al:Si ratio, and imogolite sorbs less P than allophanes.

Allophanic soils accumulate relatively large amounts of organic matter when compared with other soils, and interaction between allophanic materials and soil organic matter may be critical in long-term stabilization of carbon in many soils.^[20] Some evidence, however, suggests carbon stabilization by allophane may occur relatively slowly and may be selective for more decomposed carbon compounds.^[6] Allophane has been shown to stabilize organic matter through both direct and indirect mechanisms. Organic matter can be sorbed onto the surface of allophane minerals through ligand exchange reactions. Indirectly, allophanes release Al into soil solution as a result of weathering. Subsequent formation of Al–humus complexes causes the retardation of organic matter decomposition.^[6] The accumulation of organic matter contributes to the water retention capacity of Andisols.

CONCLUSION

Allophane and imogolite are unique and difficult to study, but a combination of analytical, experimental, and modeling work is beginning to elucidate their structures. They lend distinctive properties to soils, most notably high capacities for retention of both soil water and nutrient anions such as phosphate and for long-term stabilization of organic matter. They are primarily found in soils formed on volcanic tephra, and the special characteristics of these minerals and their strong influence on soil properties were recognized with the formation of the Andisol order, which is one of two orders defined primarily by parent material. It is important to carefully consider the properties of these minerals when managing allophanic soils.

REFERENCES

1. Abidin, Z.; Matsue, N.; Henmi, T. Differential formation of allophane and imogolite: Experimental and molecular orbital study. *J. Comput-Aided Mater.* **2007**, *14* (1), 5–18.
2. Cradwick, P.D.G.; Farmer, V.C.; Russell, J.D.; Masson, C.R.; Wada, K.; Yoshinaga, N. Imogolite, a hydrated aluminum silicate of tubular structure. *Nat. Phys. Sci.* **1972**, *240*, 187–189.

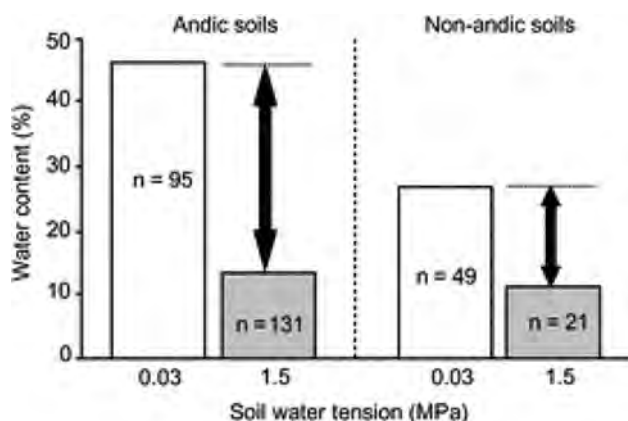


Fig. 3 Mean water content of inland Pacific Northwest Andisols and nonandic soils with similar textures at field capacity (0.03 MPa) and permanent wilting point (1.5 MPa). Vertical arrows indicate the range of plant-available water (water content 0.03–1.5 MPa).

Source: Figure courtesy of P. McDaniel. Data from National Cooperative Soil Survey Soil Characterization Database, available online at <http://ncsslabdatamart.sc.egov.usda.gov/>

3. Creton, B.; Bougeard, D.; Smirnov, K.S.; Guilment, J.; Poncelet, O. Molecular dynamics study of hydrated imogolite. 1. Vibrational dynamics of the nanotube. *J. Phys. Chem. C* **2008**, *112* (27), 10013–10020.
4. Guimarães, L.; Enyashin, A.N.; Frenzel, J.; Heine, T.; Duarte, H.A.; Seifert, G. Imogolite nanotubes: Stability, electronic, and mechanical properties. *ACS Nano* **2007**, *1* (4), 362–368.
5. Levard, C.; Rose, J.; Thill, A.; Masion, A.; Doelsch, E.; Maillet, P.; Spalla, O.; Olivi, L.; Cognigni, A.; Ziarelli, F.; Bottero, J.-Y. Formation and growth mechanisms of imogolite-like aluminogermanate nanotubes. *Chem. Mater.* **2010**, *22* (8), 2466–2473.
6. Parfitt, R.L. Allophane and imogolite: Role in soil biogeochemical processes. *Clay Miner.* **2009**, *44* (1), 135–155.
7. Dahlgren, R.A. Quantification of allophane and imogolite. In *Quantitative Methods in Soil Mineralogy*; Amonette, J.E., Zelazny, L., Eds.; Soil Science Society of America, Inc: Madison, WI, 1994; 430–451.
8. Wada, K. Allophane and imogolite. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; Soil Science Society of America: Madison, 1989; 1051–1087.
9. Creton, B.; Bougeard, D.; Smirnov, K.S.; Guilment, J.; Poncelet, O. Structural model and computer modeling study of allophane. *J. Phys. Chem. C* **2008**, *112* (2), 358–364.
10. Childs, C.W.; Parfitt, R.L.; Newman, R.H. Structural studies of Silica Springs allophane. *Clay Miner.* **1990**, *25* (3), 329–341.
11. Opfergelt, S.; Georg, R.B.; Burton, K.W.; Guicharnaud, R.; Siebert, C.; Gislason, S.R.; Halliday, A.N. Silicon isotopes in allophane as a proxy for mineral formation in volcanic soils. *Appl. Geochem.* **2011**, *26*, S115–S118.
12. Rampe, E.B.; Kraft, M.D.; Sharp, T.G.; Golden, D.C.; Ming, D.W.; Christensen, P.R. Allophane detection on mars with thermal emission spectrometer data and implications for regional-scale chemical weathering processes. *Geology*. **2012**, *40* (11), 995–998.
13. Karube, J.; Abe, Y. Water retention by colloidal allophane and imogolite with different charges. *Clay Miner.* **1998**, *46* (3), 322–329.
14. MacKenzie, K.; Bowden, M.; Meinhold, R. The structure and thermal transformations of allophanes studied by ²⁹Si and ²⁷Al high resolution solid-state NMR. *Clays Clay Miner.* **1991**, *39* (4), 337–346.
15. Biolders, C.; De Backer, L.; Delvaux, B. Particle density of volcanic soils as measured with a gas pycnometer. *Soil Sci. Soc. Am. J.* **1990**, *54* (3), 822–826.
16. Nanzyo, M.; Shoji, S.; Dahlgren, R. Physical characteristics of volcanic ash soils. In *Volcanic Ash Soils: Genesis, Properties and Utilization*; Shoji, S., Nanzyo, M., Dahlgren, R., Eds.; Elsevier: Amsterdam, 1993; 189–207.
17. Karube, J.; Nakaishi, K.; Sugimoto, H.; Fujihara, M. Size and shape of allophane particles in dispersed aqueous systems. *Clays Clay Miner.* **1996**, *44* (4), 485–491.
18. Johan, E.; Matsue, N.; Henmi, T. Phosphate adsorption on nano-ball allophane and its molecular orbital analysis. *Clay Sci.* **1997**, *10* (3), 259–270.
19. Okada, K.; Nishimuta, K.; Kameshima, Y.; Nakajima, A.; MacKenzie, K.J.D. Reaction of phosphate compounds with a high-silica allophane. *Clays Clay Miner.* **2005**, *53* (4), 372–379.
20. Basile-Doelsch, I.; Balesdent, J.; Rose, J. Are interactions between organic compounds and nanoscale weathering minerals the key drivers of carbon storage in soils? *Environ. Sci. Technol.* **2015**, *49* (7), 3997–3998.

Alpine Soils

Jérôme Poulénard

*Sciences of the Sun Laboratory, Interdisciplinary Scientific Center of Montagne,
University of Savoy, Savoy, France*

Pascal Podwojewski

Institute for Research and Development (IRD), Hanoi, Vietnam

Abstract

Alpine soils are found in mountainous regions above the natural subalpine tree line. This high-altitude belt is characterized by a lack of trees and dominance of a continuous grass carpet. The global land area covered by alpine soils is fragmented into many mountain regions (Rocky Mountains, Alps, Himalayas, Atlas, Andes, East Africa Mountains, New Guinea Highlands, New Zealand Alps, etc.) and approaches approximately 4×10^6 km². As a function of time, topographic conditions, and parent materials, a large range of soil types exists. However, alpine belt environment specificity leads to common features for all alpine soils. Because of the last glaciation, alpine soils are young (<10,000 years BP) and strongly influenced by a periglacial environment. On steep slopes, soils are often thin, regularly truncated, and in a constant process of rejuvenation. However, deep soils are found in some alpine grasslands on highly weatherable parent materials. Organic matter accumulation, acidification process, and a great role of aeolian dust deposition are other general characteristics of alpine soils genesis. The alpine soils are fragile and subjected to severe environmental threats, such as overgrazing, acid deposition, and climate change.

THE ALPINE BELT ENVIRONMENT

The altitude of the tree line (i.e., the zone of transition between the continuous forest and the alpine grassland) varies strongly with latitude. Close to the equator, this tree line is found at approximately 3600 m a.s.l., whereas in the regions above 60°, this limit is virtually at sea level (Table 1). This limit also varies in space with slope exposition and in time according to both climatic changes and human activities. The alpine ecosystem is often called “alpine grassland,” “alpine tundra,” or “alpine meadow.” In tropical zones, this belt is often known by local names: “paramos” and “punas” in the Andes and “afroalpine belt” in East Africa. Other than graminoid tussock, alpine vegetation is composed mainly of dwarf shrubs and herbaceous dicotyledonous species, which typically occur in a complex mosaic of communities as a function of topographic position, soil properties, and distribution of snow.^[1]

The alpine soil temperature regime^[2] is generally cryic with an average decrease of 0.6°C for every 100 m of elevation increase. The alpine belt experiences considerable fluctuations in temperature between day and night, with frequent night frost in tropical as well as in temperate zones. In addition to the diurnal cycles, deeper soil freezing and thawing may occur following seasonal cycles in temperate zones.

Rainfall generally increases with altitude. However, in high mountains, condensation occurs well before air masses attain the summit, leading generally to a lower annual rainfall in the alpine belt than in the subalpine forest.

Substantial cloud moisture input is another important characteristic of alpine climate. The snow regime has a great impact on preventing soil and plant exposure to low-temperature extremes. The general pattern of alpine climate may be disrupted by a number of local factors (e.g., the “foehn effect” with opposition between the very wet windward and the very dry leeward).^[1,3]

Erosional (“cirques,” “U-shaped valley”) and depositional (“moraines”) glacial landforms are prominent in the alpine belt. Steep upper slopes, usually consisting of rough rock outcrops, and extensive lower slopes of rock debris are commonly observed (Fig. 1).

FACTORS OF ALPINE SOILS FORMATION

Erosion and Rejuvenation

Steep slopes are subject to strong erosion, especially if stabilizing vegetation is absent. In these slopes where soil losses are higher than the rate of soil formation, alpine soils are regularly truncated and in a constant process of rejuvenation with shallow depth, high skeleton soil content, and low fine earth fraction (Fig. 2).^[3]

Periglacial Phenomena

Among the physical effects on alpine soil formation, the processes caused by freezing play a central role. Permafrost

Table 1 Latitudinal distribution of the areas of alpine life zone and main mountain ranges with alpine environment.

Latitude range (°)	Approximate altitudinal boundaries of alpine life zone (m. a.s.l.)	Area within alpine life zone (km ²)	Main mountain ranges
<60°N	0–500	824,000	Brooks Range, Alaska Range (United States), Kjollen Range (Norway, Sweden), Gory Putorana, Chersky Range, Verkhoyansk Range, and Kolyma Mountains (Russia)
60°N–50°N	1,000–2,500	428,000	Rocky Mountains (Canada), Saigan Mountains, and Yablonovy Range (Russia)
50°–40°N	2,000–3,500	724,000	Cascade Range, Rocky Mountains (United States), Alps (France, Italy, Switzerland, and Austria), Apennines (Italy), Pyrenees (Spain and France), Carpathians (Slovakia, Ukraine and Romania), Caucasus (Armenia, Georgia, Azerbaijan, Turkey, etc.), Altaï (China, Mongolia, Russia, and Kazakhstan), and Tien Shan (China)
40°N–30°N	3,000–4,500	1,088,000	Sierra Nevada, Rocky Mountains (United States), Atlas Mountains (Morocco and Algeria), Zagros Mountains (Turkey, Iraq, and Iran), Hindu Kush, Pamirs, Karakoram, Kunlun Mountains, Plateau of Tibet, Himalayas, Ningling Shan, Dxue Shan, and Bayan Schan (Afghanistan India, China, Pakistan, Kyrgyzstan, and Uzbekistan)
30°N–20°N	3,250–4,750	208,000	Sierras Madre Occidental and Oriental (Mexico), Himalayas, and Plateau of Tibet (India, Nepal, Bhutan, and China)
20°N–0°	3,500–5,000	18,000	Andes (Colombia and Ecuador), Sierra Madre del Sur (Mexico, Guatemala, Salvador, and Honduras) Cordilleras de Talamanca (Costa Rica), Ethiopian Highlands (Ethiopia and Eritrea), and East African Highlands (Kenya)
0°–20°S	3,250–4,500	280,000	Andes (Ecuador, Peru, and Bolivia), East African Highlands (Kenya and Tanzania), and Pegunungan Maoke (Indonesia and New Guinea)
20°S–30°S	2,750–4,000	150,000	Andes (Bolivia, Chile, and Argentina) and Drakensburg (South Africa)
30°S–40°S	2,000–3,000	60,000	Andes (Chile and Argentina)
>40°S	1,000–2,000	220,000	Andes (Chile and Argentina) and New Zealand Alps (New Zealand)

and their related Cryosols^[4] or Gelisols^[2] are found only in the nival belt above the upper limit of higher plant distribution.^[1] In the alpine belt, soils are strongly affected by freeze/thaw cycles with fragmentation of rocks, creeping, solifluction, and cryoturbation phenomena, which disrupt

and dislocate horizons, displace and incorporate materials from other horizons, and mechanically sort soil particles^[1,3] (Fig. 3). In flat terrain, deep frost tables can induce water logging.



Fig. 1 Narrow alpine belt strongly submitted to the active erosion processes of the rock outcrops (Banff National Park, Canadian Rocky Mountains).

Source: Photo from L. Trosset.



Fig. 2 First development of alpine soil formation on “roches moutonnées” after glacier withdrawal (Rousses, French Alps).

Source: Photo from J. Poulenard.

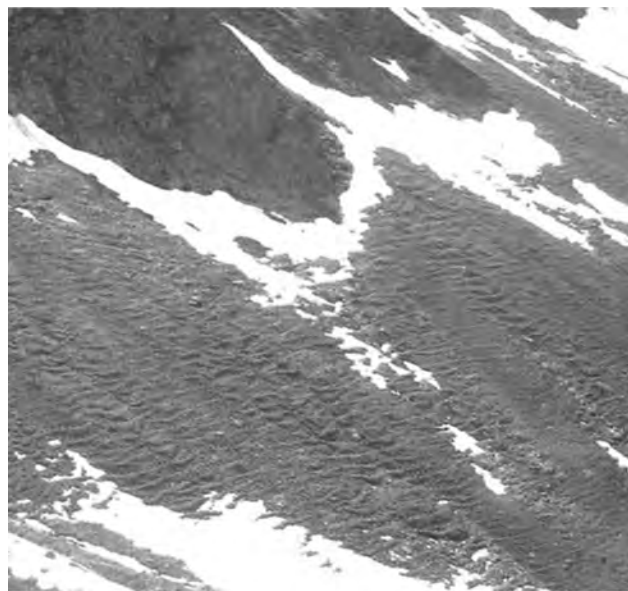


Fig. 3 Solifluction and grazing step terrassettes in alpine slope (Vanoise, French Alps).

Source: Photo from J. Poulenard.

AEOLIAN DUST DEPOSITION

Aeolian dust is an important soil-forming factor in alpine soil pedogenesis.^[3] In acidic materials (quartzitic residuum, granite, and gneiss), the steady supply of eolian carbonates through wind-blown materials greatly contributes to raise the pH of surface horizons close to neutrality.^[5,6] The original particle size of the soil may be changed, if the allochthonous particles mix with autochthonous ones.^[3] In volcanic regions (Northern Andes, East African Mountains, New Zealand, etc.), alpine soils are developed from volcanic pyroclastic materials, and volcanic ash can strongly influence alpine soil genesis even in the case of non-volcanic parent materials.^[7]

Weathering and Soil Formation Rate

After the glacier retreat, the greatest change in soil chemistry of alpine soils occurred within the first 3000–4000 years of soil development.^[3] Despite the cold climate, chemical weathering was intense^[3] and soil development proceeded more rapidly in the alpine zone than in the subalpine and mountain forests.^[6] However, it is the degree of parent material fragmentation that largely controls the rate of alpine soil formation.^[3]

In parent rocks containing carbonates, solubility phenomena are especially important. High precipitation and low temperature of the percolating water increase carbonate solubility. Rapid dissolution of CaCO_3 is a common feature of alpine soil weathering.^[3,8]

Soil Organic Matter and Biological Activity

Alpine soils are often characterized by relatively high organic matter content ($>50 \text{ g kg}^{-1}$) in the surface soil with a good incorporation of organic matter through the profile linked with abundant below-ground production of meadow grasses.^[1,9] Both substrate properties with high organic content resistant to mineralization and climate conditions prolong the mean residence time of organic matter.^[1] Faunal and microbial activity of carbon (C) mineralization is clearly reduced under alpine field conditions.^[1,10] Soil biodiversity and activity have been found to vary strongly with characteristic seasonal microclimatic conditions (snow regime, soil moisture, etc.) and spatial distribution of vegetation.^[1]

MAJOR SOILS IN ALPINE BELT AREAS

Using the two international systems of classification,^[2,4] one can find a large variety of alpine soil types over very short distances. However, in the former U.S. soil classification system, the soils of grassy meadows above the timberline were all classified in the “alpine meadow soils” overgroup of the intrazonal order.^[11]

In the steep slopes and in the zone with constant rejuvenation of the profiles, Leptosols (rendzic Leptosols over calcareous materials and umbric Leptosols over acid rocks) and Regosols^[4] are the main soil types found in the alpine belt (Fig. 1). The stability of the slope and the rate of water erosion are the main factors controlling the occurrence of these poorly differentiated soils in the alpine zone.^[3] On stable slopes and over parent materials, highly weatherable and providing weathered materials rich in clay (e.g., some micashist), Cambisols,^[4] or Inceptisols,^[2] and frequently deep soils are found.^[3,9] On stable slopes and over granular crystalline acid rocks, we can sometimes observe the occurrence of podzols in the alpine belt.^[3] However, the podzolization process is often limited by aeolian dust of calcareous materials.^[6,7] Over limestone and parent materials containing both large amounts of phyllosilicates and calcite (calcschale, calcareous micashist, etc.), the rapid and intense decalcification process leads to the formation of a large range of alpine soil from more or less decalcified Cambisols^[3] to locally true podzols.^[8]

The occurrence of poorly drained soils (Gleysols) in alpine environment is common.^[5] Their distribution is a function of topography and they show a morphology that is primarily determined by the effect of frost. They may have peat surfaces. Histosols are also frequent in flat topographic situations (Fig. 4).^[1]

In wet climate conditions, when the aluminum (Al) availability of the parent rock is high, especially on volcanic ash but also in non-volcanic areas, non-allophanic Andisols with high amounts of Al–humus complexes and great accumulation of organic matter have a great extension in the alpine belt.^[7,12]



Fig. 4 Cushion plants formation developed over Histosol (see the entry on *Paramos Soils*, p. 1648–1651) (Chimborazo Volcano, Ecuadorian Andes).

Source: Photo from P. Podwojewski.

ALPINE SOIL LAND USE AND MANAGEMENT

Alpine soils play a major role in the functioning and conservation of the unique alpine ecosystem^[1] and in the hydrologic function of mountain belts.^[13] Thus change of alpine soil properties linked with land use change has raised concerns.^[1,13] Traditional, man-made alpine pasture near the tree line has been common in all mountainous regions for at least 7000 years.^[1,13] In parts of Europe, abandonment of these pastures affects soil dynamics.^[1,8] However, in most of the world, alpine soils are submitted to increasing grazing pressure, leading to rapid degradation and erosion.^[13] Other more localized form of alpine land use is the construction of ski runs, which frequently leads to the complete destruction of natural soils. Acid deposition, with the low buffering capacities of thin soil layers in alpine areas, is an other important threat for alpine soils.^[1,13]

ALPINE SOIL AND GLOBAL CLIMATE CHANGE

High mountain ecosystems are generally considered to be particularly sensitive to climate warming. Therefore they appear to be useful “ecological indicators,” and extensive work has been done to study climate changes in alpine ecosystems.^[1,13,14] Overall warming and associated change in precipitation patterns and snow cover will drastically influence alpine vegetation with change in diversity and abundance of certain species. The possible impact of projected climate change on alpine soils is then firstly linked with the alpine vegetation change.^[13,14] As in other regions but possibly with higher intensity, the climate change will influence both C balance (C mineralization vs. C sequestration) and mineral balance (weathering vs. erosion) in alpine soils with apparent contradictory

effects. The increase in temperature could substantially increase the depth of soil’s active layer, leading to: 1) higher rates of soil organic C decomposition but with a longer growth season and 2) increase of C inflow (increase of the C pool in alpine soils). In the same way, the consequences of climate change on the mean soil depth are unclear. With higher temperature, an increase of the weathering rate is anticipated. However, evidences are accumulating that as heavy rainfall events (associated with global warming) become more frequent, erosion of alpine soils is likely to be enhanced.^[1,13,14]

REFERENCES

1. Körner, C. *Alpine Plant Life: Functional Plant Ecology of High Mountain Ecosystems*; Springer-Verlag: Berlin, 1999.
2. Soil Survey Staff. *Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys*; U.S. Department of Agriculture Handbook No. 436. USDA-NRCS: Washington, 1999.
3. Legros, J.P. Soils of alpine mountains. In *Weathering, Soils and Paleosols*; Martini, I.P., Chesworth, W., Eds.; Elsevier: Amsterdam, 1992; 155–181.
4. FAO. *World Reference Base for Soil Resources*; 84 World Soil Resources Reports; FAO, ISRIC, ISSS: Rome, 1998.
5. Litaor, M.I. The influence of Eolian dust on the genesis of Alpine soils in the front range, Colorado. *Soil Sci. Soc. Am. J.* **1987**, *51*, 142–147.
6. Bockheim, J.G.; Munroe, J.S.; Douglass, D.; Koerner, D. Soil development along an elevational gradient in the South-eastern Uinta mountains, Utah, USA. *Catena* **2000**, *39*, 169–185.
7. Allen, C.E.; Burns, S.F. Characterization of Alpine soils, Eagle cap, Wallowa mountains, Oregon. *Phys. Geogr.* **2000**, *20* (3), 212–222.
8. Buurman, P.; Van Der Plas, L.; Slager, S. A toposequence of alpine soils on calcareous Micaschists, Northern Adula region, Switzerland. *J. Soil Sci.* **1976**, *27*, 395–410.
9. Bartoli, F.; Burtin, G. Etude de quatre séquences sol-végétation à l’étage alpin. *Doc. Cartogr. Ecol.* **1979**, *XXI*, 79–93.
10. Mancinelli, R.L. Population dynamics of Alpine tundra soil bacteria, niwot ridge, Colorado front range, U.S.A. *Arct. Alp. Res.* **1984**, *16*, 185–192.
11. Thorp, J.; Smith, G.D. Higher categories of soil classification, order, suborder, and great soil groups. *Soil Sci.* **1949**, *67*, 117–126.
12. Poulenard, J.; Podwojewski, P.; Herbillon, A.J. Characteristics of non-allophanic andisols with hydric properties from the Ecuadorian Páramos. *Geoderma* **2003**, *117*, 267–281.
13. Messerli, B.; Ives, J.D. *Mountains of the World—A Global Priority*; The Parthenon Publishing Group: London, 1997.
14. Steininger, K.W.; Weck-Hannemann, H. *Global Environmental Change in Alpine Regions. Recognition, Impact, Adaptation and Mitigation*; Edwar Elgar Publishing: Cheltham, 2002.

Aluminum

Johannes Bernhard Wehr

Frederick Paxton Cardell Blamey

Neal William Menzies

School of Land, Crop and Food Sciences, University of Queensland, St. Lucia, Queensland, Australia

Abstract

Aluminum (Al), a light metal, is ubiquitous in soils. At low pH, toxic Al species can form that inhibit plant root growth and thereby limit ecosystem productivity. The occurrence of toxic Al species in soils and the effect of these forms on plant physiology and root growth are reviewed. Agricultural management practices aimed at limiting Al toxicity and plant-based Al resistance mechanisms are also discussed.

INTRODUCTION

Aluminum (Al) is the most common metallic element in the earth's crust (8%) and the third most common element after oxygen and silicon on the earth. The dissolution of the minerals, namely gibbsite and boehmite,^[1] releases Al and ultimately controls the concentration of Al in solution in many soils,^[2–5] although factors, such as dissolved organic matter, also influence the total Al concentration present. Al plays an important role in the environment as many soils (Spodosols, Alfisols, Inceptisols, and Histosols in cold and temperate climates and Oxisols and Ultisols in tropical climates) are acidic either naturally (due to high rainfall leaching of base cations) or due to anthropogenic (use of ammonium fertilizers, legumes in cropping systems, acid rain) influences.^[6] It is estimated that approximately 30% of the world ice-free land area is occupied by acid soils of which approximately 180 million hectares are used for agricultural production.^[6] In solution, Al is highly toxic to plants, *Rhizobium* bacteria in soils, aquatic organisms in fresh waters,^[7] and to a lesser extent soil fauna. Al has been implicated among others in Alzheimer's disease and dialysis dementia in humans.^[8,9]

OCCURRENCE AND USES

Al is a stable element with 100% abundance as the ²⁷Al isotope. The pure metal, which is non-magnetic, does not occur in nature and Al is only found as a constituent in minerals such as mica, feldspar, and some gemstones. Al-containing minerals weather initially to 2:1 layer aluminosilicate clay minerals (e.g., vermiculite, montmorillonite), which upon further weathering form 1:1 layer aluminosilicates such as kaolinite. Further weathering of clay minerals

leads to leaching of silica and base cations leaving behind hydrous aluminum oxide (e.g., gibbsite) in the form of bauxite, which is an important Al ore. Bauxite is digested with hot alkali (Bayer process), yielding insoluble impurities (red mud) and aluminate, which is precipitated as Al(OH)₃ and electrolyzed in molten cryolite to yield metallic Al. The metal has a low specific gravity (2.7 g cm⁻³) and is resistant to corrosion due to formation of an oxide film on the surface. Consequently, the metal is extensively used in the building and construction industries; automotive, shipping, and aerospace industries; manufacture of power lines; and for food packaging. Al salts and compounds are used for water purification, as catalysts in the chemical industry and as ingredients in cosmetics.

CHEMISTRY OF AL

Al is an amphoteric element, dissolving both at low and high pH with minimal solubility around pH 6–7.^[10] Soluble Al³⁺, which is not redox reactive, is classified as a “hard” acid due to its very small ionic radius of 0.53 Å and reacts strongly with “hard” ligands such as oxygen and fluoride.^[10,11] It also forms stable complexes with di- and multidentate ligands (chelate effect).^[10] The dissolution of Al minerals at low pH releases octahedral Al³⁺, which is hexacoordinated with water molecules.^[11] Octahedral Al³⁺ gives a broad peak on ²⁷Al nuclear magnetic resonance (NMR) spectra.^[12,13] As the pH of an acidic Al solution is increased to 4, hydrolysis of Al occurs, giving rise to a series of Al hydroxy species (AlOH²⁺, AlOH₂⁺, and Al(OH)₃),^[11] which may undergo aggregation (polymerization) via OH bridges if the Al concentration is sufficiently high (>10 μM).^[14] As the pH is raised above 6, Al changes its coordination number to 4 and yields tetrahedral

aluminate (AlOH_4^-),^[11] with a sharp and well-defined peak at ~63 ppm upfield of Al^{3+} on ^{27}Al NMR spectra.^[12,13]

POLYMERIZATION OF Al

Depending on the pH and Al concentration in solution, hydrolyzed Al species may aggregate and form polymeric (polycationic) Al species such $\text{Al}_2(\text{OH})_2^{4+}$, $\text{Al}_3(\text{OH})_4^{5+}$, $\text{Al}_8(\text{OH})_{20}(\text{H}_2\text{O})_x^{4+}$, the “gibbsite fragment” model forms, $\text{Al}_6(\text{OH})_{12}(\text{H}_2\text{O})_{12}^{6+}$ through $\text{Al}_{54}(\text{OH})_{144}(\text{H}_2\text{O})_{36}^{18+}$, $\text{Al}_{13}\text{O}_4(\text{OH})_{24}(\text{H}_2\text{O})_{12}^{7+}$ (Al_{13}), and $\text{Al}_2\text{O}_8\text{Al}_{28}(\text{OH})_{56}(\text{H}_2\text{O})_{26}^{18+}$ species.^[12,15–17] While many more Al polymers have been proposed, the experimental evidence in support of these species is limited. The lack of suitable analytical approaches for the determination of polymeric Al species may mean that other species form but cannot be detected with sufficient accuracy.^[12] The polymeric Al species are metastable with a half-life for Al_{13} of several hundred hours,^[18] which imply that the species undergo depolymerization or crystallization with time. It is generally accepted that sulfate, phosphate, fluoride, silicate, and organic acids prevent or reverse Al_{13} formation,^[12,19–21] and crystalline gibbsite may induce crystallization and depolymerization of Al_{13} .^[22] As sulfate and silicate ions are prevalent even in acid soils, the natural occurrence of Al_{13} in soil solution is considered unlikely.^[12,23] However, there is evidence that Al_{13} may form in the cell wall of plant roots^[24–26] even if the conditions in bulk solution are not favorable for Al_{13} formation. Clearly, this area requires further research.

TOXICITY OF Al

Toxicity of Al to plants is mainly manifested by an inhibition of root growth and while it is considered that Al^{3+} and Al_{13} are the most toxic species,^[27,28] studies correlating Al species and their activity with degree of root growth inhibition have given poor results. This is due to the several Al-hydroxy species that coexist within a narrow pH band and cannot be investigated in isolation. Furthermore, the activities of individual species must be calculated from equilibrium data that may be uncertain.^[29] The critical Al concentration at which root elongation is inhibited is as low as 5–20 μM in solutions of low ionic strength representative of acid soils. Exact critical values depend on the plant species and the conditions in which the plants are grown. Generally, root hairs are more sensitive to Al toxicity than roots.^[30]

The forms of Al potentially present in solutions extracted from acid soils are listed in Table 1. These forms differ markedly in their effects on plant growth, from having no effect (microcrystals and amorphous precipitates) to being toxic at a concentration <10 μM (Al_{13} and Al^{3+}). It is, therefore, essential that methods be used to discriminate among the various forms that may be extracted.

Table 1 Forms of Al potentially present in solution extracted from soils.

Form	Examples	Phytotoxicity
Microcrystalline	Kaolinite, Gibbsite	Low
Aluminate	AlOH_4^-	
Amorphous precipitates	Al_2SiO_5 , $\text{Al}(\text{OH})_3$, AlPO_4	
Organic complexes		
Low relative molecular mass	Al-citrate, Al-oxalate	FX
High relative molecular mass	Al-fulvate, Al-humate	
Inorganic complexes	AlF^{2+} , AlF_2^+ , AlSO_4^+	
Inorganic monomers	Al^{3+} , AlOH^{2+} , $\text{Al}(\text{OH})_2^+$	
Small polycations	$\text{Al}_2(\text{OH})_2^{4+}$, $\text{Al}_3(\text{OH})_4^{5+}$	
Large polycations	$\text{AlO}_4\text{Al}_{12}(\text{OH})_{24}(\text{H}_2\text{O})_{24}^{7+}$	High

Source: Adapted from Asher, Blamey, et al.^[31]

PHYSIOLOGICAL BASIS OF Al TOXICITY IN PLANTS

Root growth is more inhibited than shoot growth by Al. Consequently, yield may be drastically reduced by Al in acid soils without developing clear and diagnostic foliar symptoms. Root growth inhibition results in short stubby roots and absence of root hairs. The main site affected by Al is the elongation zone of roots (1–4 mm behind the root tip).^[32,33]

The inhibition of root growth by Al is a very rapid process, occurring within 30 minutes of exposure to Al.^[33] This points to involvement of the apoplast (i.e., the cell wall and the exterior side of the plasma membrane) in fast Al responses.^[34] Owing to the proximity of the cell wall and plasma membrane in addition to physical interaction between the two (e.g., by cell wall associated kinases^[35–37]), it is beyond contemporary experimental approaches to distinguish between cell wall and plasma membrane effects *in vivo*, while *in vitro* approaches are plagued by artifacts.

Binding of Al to negative charges of the cell wall (contained on hemicelluloses, pectin, and proteins) interferes with processes necessary for cell expansion and thus root elongation.^[38,39] For instance, binding of Al to pectin renders pectin resistant to degradation by enzymes,^[40] in turn reducing mobility of enzymes necessary to cleave load bearing hemicellulose molecules,^[41,42] resulting in stiffening of the cell wall.^[38,43] The desorption of bound Al with organic acids, which chelate Al,^[44] leads to a recovery in cell and root elongation.^[39] While extensions, a group of proteins responsible for cell wall loosening, are inhibited by 1 mM Al *in vitro*,^[41,45] it is uncertain if inhibition also occurs at Al concentrations known to inhibit root growth

(namely 5–20 μM). Furthermore, binding of Al to the cell wall reduces the negative charge (CEC) of the cell wall and affects accumulation of ions in the cell wall. This can lead to a decreased state of hydration of the cell wall^[46] and decreased transmembrane potential.^[47] Interactions of Al with sodium (Ca), potassium, and anion channels in the plasma membrane have been described^[48–50] with implications for the transmembrane potential.^[51] Rapid transient changes in the cytosolic (intracellular) Ca concentration have been observed, which can interfere with the cytoskeleton^[48,52] and induce callose formation.^[52] Al also affects the fluidity of the plasma membrane^[53–55] and may trigger the generation of reactive oxygen species,^[56–58] which in turn can crosslink cell wall polymers.^[59] Cytosolic Al effects have been postulated for the Ca homeostasis, oxidative stress responses, cell division, and energetic processes.^[51] However, these processes are affected sometimes after the initial inhibition of root elongation, suggesting that the primary site of Al toxicity is in the cell wall. For reviews dealing with Al toxicity, the reader is referred by Rengel and Zhang,^[52] Barceló and Poschenrieder,^[60] Horst,^[61] Kochian et al.,^[62] and Zheng and Yang.^[63]

RESISTANCE AND TOLERANCE MECHANISMS IN PLANTS

The ability of a plant to withstand high Al concentrations in soil can be mediated by either resistance mechanisms (i.e., exclusion of Al from the apoplast and cytoplasm) or by tolerance mechanisms (i.e., accumulation of Al in the apoplast and symplast). Considering the economic importance of Al toxicity, a great deal of research has been conducted into Al resistance and tolerance mechanisms, and there are detailed reviews of this topic available.^[60,64–69] Several plant species naturally occurring on acid soils are tolerant toward Al and even accumulate Al in leaves and plant tops [e.g., tea (*Camellia sinensis*) and some eucalypts and rain-forest plants]^[67,68,70–74] in the form of organic or inorganic Al complexes or bound to the cell wall. Al resistance can be imparted by organic acid exudation from plant roots and it has been demonstrated that genetic engineering for increased organic acid synthesis and exudation conferred Al resistance to previously sensitive species.^[75,76] Complexation of Al by peptides^[77] and exuded organic acids, in particular malic, citric, and oxalic acid, has been found for a number of Al resistant species; however, the importance of this strategy is debated.^[78–81]

A further resistance mechanism has been suggested in the modification of the rhizosphere pH by plant roots.^[82,83] An alkalization of the area immediately surrounding the zone of root elongation may hydrolyze or precipitate Al and thereby prevent formation of rhizotoxic Al species.^[84,85] Finally, there is evidence suggesting that the affinity of plant root cell walls for Al may be different between Al-sensitive and resistant plants.^[25,46,86,87]

AMELIORATION OF Al TOXICITY

Generally, applying dolomite or calcite limestone is the most effective way of dealing with Al toxicity since it raises the soil pH (through precipitation of $\text{Al}(\text{OH})_3$) and supplies Ca and magnesium ions, which compete with Al for binding by roots and soil particles. The application of single superphosphate has a similar effect through the precipitation of Al phosphate and formation of less toxic AlSO_4 . Agronomically, use of Al tolerant crop plants in conjunction with liming is the most effective way of dealing with acid soils. Organic matter addition may complex soluble Al and alleviate Al toxicity,^[88] although large quantities may be required depending on their ash alkalinity.^[89]

REFERENCES

1. Lindsay, W.L. *Chemical Equilibria in Soils*; John Wiley: New York, 1979.
2. Simonsson, M. Interactions of aluminium and fulvic acid in moderately acid solutions: Stoichiometry of the $\text{H}^+/\text{Al}^{3+}$ exchange. *Eur. J. Soil Sci.* **2000**, *51*, 655–666.
3. Skyllberg, U. pH and solubility of aluminium in acidic forest soils: A consequence of reactions between organic acidity and aluminium alkalinity. *Eur. J. Soil Sci.* **1999**, *50*, 95–106.
4. Adams, M.L.; Hawke, D.J.; Nilsson, N.H.S.; Powell, K.J. The relationship between soil solution pH and Al^{3+} concentrations in a range of South Island (New Zealand) soils. *Aust. J. Soil Res.* **2000**, *38*, 141–153.
5. Lofts, S.; Woof, C.; Tipping, E.; Clarke, N.; Mulder, J. Modelling pH buffering and aluminium solubility in European forest soils. *Eur. J. Soil Sci.* **2001**, *52*, 189–204.
6. von Uexküll, H.R.; Mutert, E. Global extent, development and economic impact of acid soils. In *Plant Soil Interactions at Low pH*; Date, R.A., Grundon, N.J., Rayment, G.E., Probert, M.E., Eds.; Kluwer Academic: Dordrecht, 1995; 5–19.
7. Gensemer, R.W.; Playle, R.C. The bioavailability and toxicity of aluminum in aquatic environments. *Crit. Rev. Environ. Sci. Technol.* **1999**, *29*, 315–450.
8. Yokel, R.A. The toxicology of aluminum in the brain: A review. *Neurotoxicology* **2000**, *21*, 813–828.
9. Nayak, P. Aluminum: Impacts and disease. *Environ. Res.* **2002**, *89*, 101–115.
10. Martell, A.E.; Hancock, R.D.; Smith, R.M.; Motekaitis, R.J. Coordination of $\text{Al}(\text{III})$ in the environment and in biological systems. *Coord. Chem. Rev.* **1996**, *149*, 311–328.
11. Martin, R.B. Ternary complexes of Al^{3+} and F^- with a third ligand. *Coord. Chem. Rev.* **1996**, *141*, 23–32.
12. Bertsch, P.M.; Parker, D.R. Aqueous polynuclear aluminum species. In *The Environmental Chemistry of Aluminum*, 2nd Ed.; Sposito, G., Ed.; CRC Lewis Publishers: Boca Raton, 1996; 117–168.
13. Bi, S.P.; Yang, X.D.; Zhang, F.P.; Wang, X.L.; Zou, G.W. Analytical methodologies for aluminium speciation in environmental and biological samples—A review. *Fresenius J. Anal. Chem.* **2001**, *370*, 984–996.

14. Furrer, G.; Trusch, B.; Muller, C. The formation of polynuclear aluminum under simulated natural conditions. *Geochim. Cosmochim. Acta* **1992**, *56*, 3831–3838.
15. Brown, P.L.; Sylva, R.N.; Batley, G.E.; Ellis, J. The hydrolysis of metal ions. Part 8. Aluminium (III). *J. Chem. Soc. Dalton Trans.* **1985**, *1985*, 1967–1970.
16. Orvig, C. The aqueous coordination chemistry of aluminum. In *Coordination Chemistry of Aluminum*; Robinson, G.H., Ed.; VCH: New York, 1993; 85–122.
17. Wang, M.; Muhammed, M. Novel synthesis of Al₁₃-cluster based alumina materials. *Nanostruct. Mater.* **1999**, *11*, 1219–1229.
18. Furrer, G.; Gfeller, M.; Wehrli, B. On the chemistry of the Keggin Al₁₃ polymer: Kinetics of proton-promoted decomposition. *Geochim. Cosmochim. Acta* **1999**, *63*, 3069–3076.
19. Casey, W.H. Large aqueous aluminum hydroxide molecules. *Chem. Rev.* **2005**, *106*, 1–16.
20. Yamaguchi, N.U.; Hiradate, S.; Mizoguchi, M.; Miyazaki, T. Formation and disappearance of Al tridecamer in the presence of low molecular weight organic ligands. *Soil Sci. Plant Nutr.* **2003**, *49*, 551–556.
21. Masion, A.; Thomas, F.; Tchoubar, D.; Bottero, J.Y.; Tekely, P. Chemistry and structure of Al(OH)/organic precipitates. A small-angle X-ray scattering study. 3. Depolymerization of the Al₁₃ polycation by organic ligands. *Langmuir* **1994**, *10*, 4353–4356.
22. Sanjuan, B.; Michard, G. Aluminum hydroxide solubility in aqueous solutions containing fluoride ions at 50°C. *Geochim. Cosmochim. Acta* **1987**, *51*, 1823–1831.
23. Hiradate, S.; Taniguchi, S.; Sakurai, K. Aluminum speciation in aluminum-silica solutions and potassium chloride extracts of acidic soils. *Soil Sci. Soc. Am. J.* **1998**, *62*, 630–636.
24. Xia, H.; Rayson, G.D. Investigation of aluminum binding to a *Datura innoxia* material using ²⁷Al NMR. *Environ. Sci. Technol.* **1998**, *32*, 2688–2692.
25. Masion, A.; Bertsch, P.M. Aluminium speciation in the presence of wheat root cell walls: A wet chemical study. *Plant Cell Environ.* **1997**, *20*, 504–512.
26. Kopittke, P.M.; Menzies, N.W.; Blamey, F.P.C. Rhizotoxicity of aluminate and polycationic aluminium at high pH. *Plant Soil* **2005**, *266*, 177–186.
27. Kinraide, T.B. Reconsidering the rhizotoxicity of hydroxyl, sulphate, and fluoride complexes of aluminium. *J. Exp. Bot.* **1997**, *48*, 1115–1124.
28. Kinraide, T.B. Identity of the rhizotoxic aluminium species. In *Plant-Soil Interactions at Low pH*; Wright, R.J., Baligar, V.C., Murrmann, R.P., Eds.; Kluwer Academic Publishers: Dordrecht, 1991; 717–728.
29. Boudot, J.P.; Maitat, O.; Merlet, D.; Rouiller, J. Occurrence of non-monomeric species of aluminium in undersaturated soil and surface waters: Consequences for the determination of mineral saturation indices. *J. Hydrol.* **1996**, *177*, 47–63.
30. Brady, D.J.; Edwards, D.G.; Asher, C.J.; Blamey, F.P.C. Calcium amelioration of aluminium toxicity effects on root hair development in soybean *Glycine max*. *New Phytol.* **1993**, *123*, 531–538.
31. Asher, C.J.; Blamey, F.P.C.; Kerven, G.L. Recent research on soil acidity: Identifying toxic aluminium in soils. In *La Investigacion Edafologia en Mexico 1991–1992. Memorias del 15 Congreso Nacional, Acapulco*; Tovar, S.J.L., Quintero, L., Eds.; Sociedad Mexicana de la Ciencia del Suelo: Mexico, 1992; 61–68.
32. Sivaguru, M.; Horst, W.J. The distal part of the transition zone is the most aluminum-sensitive apical root zone of maize. *Plant Physiol.* **1998**, *116*, 155–163.
33. Blamey, F.P.C.; Nishizawa, N.K.; Yoshimura, E. Timing, magnitude, and location of initial soluble aluminum injuries to mungbean roots. *Soil Sci. Plant Nutr.* **2004**, *50*, 67–76.
34. Rengel, Z. Tansley review no 89 – Uptake of aluminium by plant cells. *New Phytol.* **1996**, *134*, 389–406.
35. Sivaguru, M.; Ezaki, B.; He, Z.-H.; Tong, H.; Osawa, H.; Baluska, F.; Volkmann, D.; Matsumoto, H. Aluminum-induced gene expression and protein localization of a cell wall-associated receptor kinase in Arabidopsis. *Plant Physiol.* **2003**, *132*, 2256–2266.
36. Horst, W.J.; Schmohl, N.; Kollmeier, M.; Baluska, F.; Sivaguru, M. Does aluminium affect root growth of maize through interaction with the cell wall—plasma membrane—cytoskeleton continuum? *Plant Soil* **1999**, *215*, 163–174.
37. Anderson, C.M.; Wagner, T.A.; Perret, M.; He, Z.-H.; He, D.; Kohorn, B. WAKs: Cell wall-associated kinases linking the cytoplasm to the extracellular matrix. *Plant Mol. Biol.* **2001**, *47*, 197–206.
38. Tabuchi, A.; Matsumoto, H. Changes in cell-wall properties of wheat (*Triticum aestivum*) roots during aluminum-induced growth inhibition. *Physiol. Plant.* **2001**, *112*, 353–358.
39. Ma, J.F.; Yamamoto, R.; Nevins, D.J.; Matsumoto, H.; Brown, P.H. Al binding in the epidermis cell wall inhibits cell elongation of okra hypocotyl. *Plant Cell Physiol.* **1999**, *40*, 549–556.
40. Wehr, J.B.; Menzies, N.W.; Blamey, F.P.C. Inhibition of cell wall autolysis and pectin degradation by cations. *Plant Physiol. Biochem.* **2004**, *42*, 485–492.
41. Kenjebaeva, S.; Yamamoto, Y.; Matsumoto, H. The impact of aluminium on the distribution of cell wall glycoproteins of pea root tip and their Al-binding capacity. *Soil Sci. Plant Nutr.* **2000**, *47*, 629–636.
42. Takeda, T.; Fry, S.C. Control of xyloglucan endotransglycoylase activity by salts and anionic polymers. *Planta* **2004**, *219*, 722–732.
43. Ma, J.F.; Shen, R.; Nagao, S.; Tanimoto, E. Aluminum targets elongating cells by reducing cell wall extensibility in wheat roots. *Plant Cell Physiol.* **2004**, *45*, 583–589.
44. Wehr, J.B.; Menzies, N.W.; Blamey, F.P.C. Model studies on the role of citrate, malate and pectin esterification on the enzymatic degradation of Al- and Ca-pectate gels: Possible implications for Al-tolerance. *Plant Physiol. Biochem.* **2003**, *41*, 1007–1010.
45. McQueen-Mason, S.; Durachko, D.M.; Cosgrove, D.J. Two endogenous proteins that induce cell wall extension in plants. *Plant Cell* **1992**, *4*, 1425–1433.
46. Jozefaciuk, G.; Szatanik, K.A. Aluminium-induced changes in the surface and micropore properties of wheat roots: A study using the water vapor adsorption-desorption technique. *Plant Soil* **2001**, *233*, 95–108.
47. Shomer, I.; Novacky, A.J.; Pike, S.M.; Yermiyahu, U.; Kinraide, T.B. Electrical potentials of plant cell walls in response to the ionic environment. *Plant Physiol.* **2003**, *133*, 411–422.

48. Sivaguru, M.; Pike, S.; Gassmann, W.; Baskin, T.I. Aluminum rapidly depolymerizes cortical microtubules and depolarizes the plasma membrane: Evidence that these responses are mediated by a glutamate receptor. *Plant Cell Physiol.* **2003**, *44*, 667–675.
49. Etherton, B.; Heppner, T.J.; Cumming, J.R.; Nelson, M.T. Opposing effects of aluminum on inward-rectifier potassium currents in bean root-tip protoplasts. *J. Membr. Biol.* **2004**, *198*, 15–22.
50. Kawano, T.; Kadono, T.; Fumoto, K.; Lapeyrie, F.; Kuse, M.; Isobe, M.; Furuichi, T.; Muto, S. Aluminium as a specific inhibitor of plant TPC1 Ca^{2+} channels. *Biochem. Biophys. Res. Commun.* **2004**, *324*, 40–45.
51. Ahn, S.J.; Rengel, Z.; Matsumoto, H. Aluminum-induced plasma membrane surface potential and H^{+} -ATPase activity in near-isogenic wheat lines differing in tolerance to aluminum. *New Phytol.* **2004**, *162*, 71–79.
52. Rengel, Z.; Zhang, W.-H. Role of dynamics of intercellular calcium in aluminium-toxicity syndrome. *New Phytol.* **2003**, *159*, 295–314.
53. Ishikawa, S.; Wagatsuma, T.; Takano, T.; Tawaraya, K.; Oomata, K. The plasma membrane intactness of root-tip cells is a primary factor for Al-tolerance in cultivars of five species. *Soil Sci. Plant Nutr.* **2001**, *47*, 489–501.
54. Ofei, M.P.; Wagatsuma, T.; Ishikawa, S.; Tawaraya, K. The plasma membrane strength of the root cells and root phenolic compounds are correlated with Al tolerance in several common woody plants. *Soil Sci. Plant Nutr.* **2001**, *47*, 359–375.
55. Verstraeten, S.V.; Villaverde, M.S.; Oteiza, P.I. Al^{3+} -mediated changes on membrane fluidity affects the activity of PI-PLC but not of PLC. *Chem. Phys. Lipids* **2003**, *122*, 159–163.
56. Devi, S.R.; Yamamoto, Y.; Matsumoto, H. An intracellular mechanism of aluminum tolerance associated with high antioxidant status in cultured tobacco cells. *J. Inorg. Biochem.* **2003**, *97*, 59–68.
57. Yamamoto, Y.; Kobayashi, Y.; Devi, S.R.; Rikiishi, S.; Matsumoto, H. Oxidative stress triggered by aluminum in plant roots. *Plant Soil* **2003**, *255*, 239–243.
58. Darko, E.; Ambrus, H.; Stefanovits-Banyai, E.; Fodor, J.; Bakos, F.; Barnabas, B. Aluminium toxicity, Al tolerance and oxidative stress in an Al-sensitive wheat genotype and in Al-tolerant lines developed by *in vitro* microspore selection. *Plant Sci.* **2004**, *166*, 583–591.
59. Otte, O.; Barz, W. Characterization and oxidative *in vitro* cross-linking of an extensin-like protein and a proline-rich protein purified from chickpea cell walls. *Phytochemistry* **2000**, *53*, 1–5.
60. Barceló, J.; Poschenrieder, C. Fast root growth responses, root exudates, and internal detoxification as clues to the mechanisms of aluminium toxicity and resistance: A review. *Environ. Exp. Bot.* **2002**, *48*, 75–92.
61. Horst, W.J. The role of the apoplast in aluminium toxicity and resistance of higher plants: A review. *Z. Pflanzenernähr. Bodenk.* **1995**, *158*, 419–428.
62. Kochian, L.V.; Pineros, M.A.; Hoekenga, O.A. The physiology, genetics and molecular biology of plant aluminum resistance and toxicity. *Plant Soil* **2005**, *274*, 175–195.
63. Zheng, S.J.; Yang, J.L. Target sites of aluminum phytotoxicity. *Biol. Plant* **2005**, *49*, 321–331.
64. Samac, D.A.; Tesfaye, M. Plant improvement for tolerance to aluminum in acid soils—A review. *Plant Cell Tiss. Org. Cult.* **2003**, *75*, 189–207.
65. Rengel, Z. Genetic control of root exudation. *Plant Soil* **2002**, *245*, 59–70.
66. Kochian, L.V.; Pence, N.S.; Letham, D.L.D.; Piñeros, M.A.; Magalhaes, J.V.; Hoekenga, O.A.; Garvin, D.F. Mechanisms of metal resistance in plants: Aluminum and heavy metals. *Plant Soil* **2002**, *247*, 109–119.
67. Jansen, S.; Broadley, M.R.; Robrecht, E.; Smets, E. Aluminum hyperaccumulation in angiosperms: A review of its phylogenetic significance. *Bot. Rev.* **2002**, *68*, 235–269.
68. Jansen, S.; Watanabe, T.; Caris, P.; Geuten, K.; Lens, F.; Pyck, N.; Smets, E. The distribution and phylogeny of aluminium accumulating plants in the ericales. *Plant Biol.* **2004**, *6*, 498–505.
69. Ma, J.F.; Furukawa, J. Recent progress in the research of external Al detoxification in higher plants: A mini review. *J. Inorg. Biochem.* **2003**, *97*, 46–51.
70. Watanabe, T.; Osaki, M.; Yoshihara, T.; Tadana, T. Distribution and chemical speciation of aluminium in the Al accumulator plant *Melastoma malabathricum*. *Plant Soil* **1998**, *201*, 165–173.
71. Carr, H.P.; Lombi, E.; Kupper, H.; McGrath, S.P.; Wong, M.H. Accumulation and distribution of aluminium and other elements in tea (*Camellia sinensis*) leaves. *Agronomie* **2003**, *23*, 705–710.
72. Webb, L.J. Aluminium accumulation in the Australian-New Guinea flora. *Aust. J. Bot.* **1954**, *2*, 176–196.
73. Cuenca, G.; Herrera, R.; Mérida, T. Distribution of aluminium in accumulator plants by X-ray microanalysis in *Richeria grandis* leaves from a cloud forest in Venezuela. *Plant Cell Environ.* **1991**, *14*, 437–441.
74. Shen, R.F.; Iwashita, T.; Ma, J.F. Form of Al changes with Al concentration in leaves of buckwheat. *J. Exp. Bot.* **2004**, *55*, 131–136.
75. Delhaize, E.; Ryan, P.R.; Hebb, D.M.; Yamamoto, Y.; Sasaki, T.; Matsumoto, H. Engineering high-level aluminum tolerance in barley with the ALMT1 gene. *Proc. Nat. Acad. Sci. USA* **2004**, *101*, 15,249–15,254.
76. Anoop, V.M.; Basu, U.; McCammon, M.T.; McAlister-Henn, L.; Taylor, G.J. Modulation of citrate metabolism alters aluminum tolerance in yeast and transgenic canola overexpressing a mitochondrial citrate synthase. *Plant Physiol.* **2003**, *132*, 2205–2217.
77. Basu, U.; McDonald-Stephens, J.L.; Archambault, D.J.; Good, A.G.; Briggs, K.G.; Taing-Aung; Taylor, G.J. Genetic and physiological analysis of doubled-haploid, aluminium-resistant lines of wheat provide evidence for the involvement of a 23 kD, root exudate polypeptide in mediating resistance. *Plant Soil* **1997**, *196*, 283–288.
78. Ishikawa, S.; Wagatsuma, T.; Sasaki, R.; Ofei-Manu, P. Comparison of the amount of citric and malic acids in Al media of seven plant species and two cultivars each of five plant species. *Soil Sci. Plant Nutr.* **2000**, *46*, 751–758.
79. Piñeros, M.A.; Shaff, J.E.; Manslank, H.S.; Carvalho-Alves, V.M.; Kochian, L.V. Aluminum resistance in maize cannot be solely explained by root organic acid exudation. A comparative physiological study. *Plant Physiol.* **2005**, *137*, 231–241.

80. Nian, H.; Yang, Z.M.; Huang, H.; Yan, X.L.; Matsumoto, H. Citrate secretion induced by aluminum stress may not be a key mechanism responsible for differential aluminum tolerance of some soybean genotypes. *J. Plant Nutr.* **2004**, *27*, 2047–2066.
81. Parker, D.R.; Pedler, J.F. Probing the “malate hypothesis” of differential aluminum tolerance in wheat by using other rhizotoxic ions as proxies for Al. *Planta* **1998**, *205*, 389–396.
82. Pellet, D.M.; Papernik, L.A.; Jones, D.L.; Darrah, P.R.; Grunes, D.L.; Kochian, L.V. Involvement of multiple aluminium exclusion mechanisms in aluminium tolerance in wheat. *Plant Soil* **1997**, *192*, 63–68.
83. Hinsinger, P.; Plassard, C.; Tang, C.X.; Jaillard, B. Origins of root-mediated pH changes in the rhizosphere and their responses to environmental constraints: A review. *Plant Soil* **2003**, *248*, 43–59.
84. Degenhardt, J.; Larsen, P.B.; Howell, S.H.; Kochian, L.V. Aluminum resistance in the *Arabidopsis* mutant *alr-104* is caused by an aluminum-induced increase in rhizosphere pH. *Plant Physiol.* **1998**, *117*, 19–27.
85. Wenzl, P.; Patiño, G.M.; Chaves, A.L.; Mayer, J.E.; Rao, I.M. The high level of aluminum resistance in signalgrass is not associated with known mechanisms of external aluminum detoxification in root apices. *Plant Physiol.* **2001**, *125*, 1473–1484.
86. Heim, A.; Luster, J.; Brunner, I.; Frey, B.; Frossard, E. Effects of aluminium treatment on Norway spruce roots: Aluminium binding forms, element distribution, and release of organic substances. *Plant Soil* **1999**, *216*, 103–116.
87. Schildknecht, P.H.P.A.; Vidal, B.C. A role for the cell wall in Al³⁺ resistance and toxicity: Crystallinity and availability of negative charges. *Int. Arch. Biosci.* **2002**, *1*, 1087–1095.
88. Haynes, R.J.; Mokolobate, M.S. Amelioration of Al toxicity and P deficiency in acid soils by addition of organic residues: A critical review of the phenomenon and the mechanisms involved. *Nutr. Cyc. Agroecosys.* **2001**, *59*, 47–63.
89. Noble, A.D.; Zenneck, I.; Randall, P.J. Leaf litter ash alkalinity and neutralisation of soil acidity. *Plant Soil* **1996**, *179*, 293–302.

Amazon Basin Soils

Stanley W. Buol

*Department of Soil Science, North Carolina State University, Raleigh,
North Carolina, U.S.A.*

Abstract

The Amazon basin is approximately 7×10^6 km² in area. The watershed encompasses parts of Brazil, Venezuela, Colombia, Ecuador, Peru, and Bolivia. About 80% of the area is vegetated with tropical rain forest. The paucity of roads and sparse human habitation continue to inhibit extensive investigations, although limited studies have clearly established that significant differences in soil composition are present within the basin. The extent and geographical distribution of soil properties is only partially known as vast areas of the basin hide their soil properties under dense vegetative canopies that have fostered popular generalizations about the infertile composition of all the soils because of the severity of leaching in the humid tropical environment. It is known that significant areas of relatively fertile soils, related to the composition of the initial parent materials, are present throughout the basin.

SOIL TEMPERATURE AND MOISTURE REGIMES

Lying within tropical latitudes, soils in the basin share one common characteristic of nearly uniform temperature throughout the year, i.e., isothermic (15–22°C mean annual soil temperature) soil temperature regimes (STRs) at higher elevations and isohyperthermic (mean annual soil temperatures greater than 22°C) STRs at lower elevations. Perudic soil moisture regimes (SMRs) are present where rainfall exceeds potential evaporation every month of the year on the lower slopes of the Andean mountains in the western portion of the basin. Udic SMRs are present in much of the western and eastern portions of the basin where the soil moisture is available for plant growth almost every month each year. Ustic SMRs are present in the central and southern portions of the basin where a lack of rainfall limits plant growth for 3–6 months each year.^[1] Soils that are saturated in a portion of each year, aquic soil moisture conditions, are prevalent along flood plains and in other broad-level areas throughout the basin.

UPLAND SOILS (TERRA FIRME)

Soil texture, mineralogy, and chemical composition contrast in response to the composition of sedimentary material that filled the basin. Much of the basin consists of fluvial and lacustrine sedimentary materials of contrasting texture (Fig. 1). As these sediments are dissected by meandering river valleys, broad, flat terrace-like upland plateaus are formed and vegetated by upland species that hide significant soil textural contrasts (Fig. 2).

A major contrast in composition of sediments is present in the basin. The Guiana and Brazilian shields contributed

infertile sediment almost devoid of weatherable minerals from Precambrian rock to the eastern portion of the basin. During the Tertiary period, the uplift of the Andean mountains reversed the flow of water within the basin, and erosion from the Andes contributes less weathered materials, often lime-rich and weathered volcanic materials to the western portions of the basin and modern flood plains.^[2] The Andean sediments are present west of the Rio Negro and Madeira rivers in Brazil^[3] and extend into Peru,^[5,6] Columbia, Ecuador, and Bolivia. The sediments derived from the Andes contain more weatherable minerals and are more nutrient-rich than those to the east that are derived from the Guyana and Brazilian shields. Ultisols^[4] [*Podzólicos Vermelho-Amarelo Distrófico*; throughout this entry, soil names from *Soil Taxonomy*^[4] are followed by the *Brazilian soil classification*^[3]] predominate in the upper basin, while Oxisols (*Latossolos*) predominate in the lower portion of the basin. Although Ultisols and most Oxisols are nutrient-poor, significant inclusions of more nutrient-rich material are present within the basin. Although long undetected in the poorly explored and sparsely inhabited basin, large areas of Alfisols (*Podzólicos Vermelho-Amarelo Eutrófico*) are formed in base-rich sediments in western Brazil primarily in the states of Rondonia and Acre and surrounding countries. The presence of these base-rich soils belies the extensive leaching often cited as the cause of low fertility and acidity of soils in the basin. Mollisols and high base status Inceptisols, some with free calcium carbonate present within a few centimeters of the surface, have been observed in upland areas with more than 5 m of annual precipitation. The Brazilian soil classification system recognizes Eutrófico as having high base saturation percentage and Distrófico as having low base saturation percentage, at the phase level of both *Latossolos* and



Fig. 1 Aerial view of meander scars along Rio Marañon, a major tributary of the Amazon River in Peru. The tree-vegetated levees are sandy while the inundated areas are of clayey texture.

Podzólicos.^[3] Cursory translations of Brazilian literature that equate Latossolos as Oxisols and Podzólicos as Ultisols often concealed the significance of base saturation percentage. Canopies of dense tropical forest vegetation blanket almost all upland areas in the basin, and base saturation percentage is not visually evident from aerial observations. Significant areas of the more chemically fertile soils are discovered only when more detailed soil surveys, supported by laboratory data, are conducted. Subsistent farmers become aware of the more base-rich areas only after a few years of crop production. The more base-rich soils tend to remain in crop production and cattle grazing, while adjacent low base saturated soils are abandoned to forest regrowth.

East of the confluence of the Rio Negro and Rio Madeira near the city of Manaus, the predominant upland soils near the main channel of the Amazon River and also further south in the state of Mato Grosso are xanthic Oxisols (*Latossolos Amarelo, Distrófico*). These soils have a distinctive reddish-yellow to yellow color and have been found to contain less



Fig. 2 Aerial view of a river and oxbow lake formed as a tributary of the Amazon River entrenches within the Amazon basin in Peru.

iron oxide and fix less phosphate fertilizer than more reddish Oxisols.^[7,8] Extensive areas of Ultisols (*Podzólicos Vermelho-Amarelo Distrófico*) are also present in association with the Oxisols on uplands both north and south of the Amazon River. The Oxisols appear to be formed in transported and preweathered sediments, while the Ultisols are formed from residual rock and sediments that have slightly higher weatherable mineral content. Where the mineral composition of the rock is mafic (diorite), Alfisols (*Podzólicos Vermelho-Amarelo Eutrófico*) are present. A rather extensive area of the high base status soils is readily seen in association with shallow soils (*Solos Litólicos Distróficos e Eutróficos*) in the south central area of Para State.^[3] Often, these Alfisols (*Podzólicos Vermelho-Amarelo Eutrófico*) are a darker red and have been observed to occupy small areas associated with Ultisols (*Podzólicos Vermelho-Amarelo Distrófico*), illustrating that soils formed in parent rock of basic composition retain a high base saturation percentage under humid tropical leaching.

Some of the upland soils in the basin are formed in very sandy sediments. Soils formed in the sandy sediments are very infertile Quartzipsamments (*Solo Arenoquartzosos Profundos*) where the water table is deep, while Spodosols (*Podzols*) form on sandy sediments where the water table is near the soil surface. It has been estimated that perhaps 10% of the Amazon basin has these kinds of soils.^[9] One major area of sandy soils is in the headwaters of the Rio Negro in the north central part of the basin where it merges with the Orinoco watershed. Smaller areas of sandy soils are scattered throughout the basin, often detectable by the organic stained black, “coffee colored” waters of the rivers that drain these sandy watersheds (Fig. 3).

FLOOD PLAINS

The flood plains along the numerous rivers in the Amazon basin are composed of recent sediments. Many of the



Fig. 3 “Black” water from the Rio Negro as it enters the “white” water of the Amazon River near Manaus, Brazil.

sediments are quite fertile, and montmorillonite clay content is high especially in those sediments derived from base-rich materials. Other areas are infertile where the sediments are derived from siliceous materials. Most of these soils in the flood plains are aquents and aquepts (*Solos Gley*, *Distróficos*, and *Eutróficos*). The base saturation percentage of these soils ranges from high to low depending on the composition of the material in the source area of the individual river. Most flood plains are often quite broad and subject to flooding and stream-bank erosion from seasonal rise and fall of water level in the major rivers. Rivers provide routes for travel, and most human settlements are located on flood-free uplands near the rivers. The presence of anthropogenic soils (*Terra Preta do Índio*) formed from the accumulation of charcoal and other human wastes reveal that agrarian settlements may have existed in the basin for several centuries.^[10]

Road maintenance is extremely difficult in the interior of the Amazon basin where these broad flood plains are seasonally flooded. A seldom-mentioned constraint to road building is the lack of coarse aggregate in the sediments in the interior of the basin. Also, extreme difficulty is experienced when attempting to construct and maintain roads on saturated clayey sediments of high shrink and swell montmorillonite clays.

CONCLUSION

Much of the Amazon basin is poorly explored because of sparse population and lack of infrastructure. The nearly unbroken cover of tropical forest vegetation and the pedogenic dogma of extensive leaching in humid tropical environments have entrapped scientists in the assumption that a uniformity of infertile soil capable only of supporting shifting cultivation is present. This paradigm is changing as transportation infrastructure to support the market needs of continuous agriculture and cattle ranching have facilitated detailed soil studies that

reveal significant contrasts in soil properties not readily apparent from remote aerial observations hampered by dense jungle vegetation.

REFERENCES

1. Van Wambeke, A.; Newhall, F. Calculated soil moisture and temperature regimes of South America: A compilation of soil climatic regimes calculated by using a mathematical model developed by F. Newhall. In *SMSS Tech. Monograph #2*; Cornell University: Ithaca; Soil Mgt. Support Services, USDA-SCS: Washington, 1981.
2. James, D.E. The evolution of the Andes. *Sci. Am.* **1973**, *229*, 61–69.
3. EMBRAPA. *Mapa de solos do Brasil. (Escala 1: 5,000,000)*; Serviço Nacional de Levantamento e Conservação de Solos: Geocarta, 1981.
4. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; Agric. Handbook No. 436; U.S. Dept. of Agric. Natural Resources Conservation Service: Washington, 1999.
5. Osher, L.J.; Buol, S.W. Relationship of soil properties to parent material and landscape position in Eastern Madre de Dios, Peru. *Geoderma* **1998**, *83*, 143–166.
6. Tyler, E.J.; Buol, S.W.; Sanchez, P.A. Genetic association of soils in the Upper Amazon Basin of Peru. *Soil Sci. Soc. Am. J.* **1978**, *42*, 771–776.
7. Smyth, T.J.; Cravo, M.S. Critical phosphorus levels for corn and cowpea in a Brazilian amazon oxisol. *Agron. J.* **1990**, *82*, 309–312.
8. Buol, S.W.; Eswaran, H. Oxisols. *Adv. Agron.* **2000**, *68*, 151–195.
9. Cochrane, T.T.; Sanchez, L.G.; de Aseuedo, L.G.; Porras, J.A.; Carver, C.L. *Land in Tropical America*; Centro Internacional de Agricultura Tropical, CIAT: Cali, 1985; Vol. 3.
10. Heckenberger, M.J.; Kuikuro, A.; Kuikuro, U.T.; Russell, J.C.; Schmidt, M.; Fausto, C.; Franchetto, B. Amazonia 1493: Pristine forest or cultural parkland? *Science* **2003**, *301*, 1710–1714.

Amendments and Ameliorants

David C. McKenzie

Precision Land Management, Orange, New South Wales, Australia

Abstract

Soil amendments/ameliorants can be used to overcome many physical problems in soil. For improving soil structural stability, the best-known ameliorants are gypsum (calcium sulfate) and, to a lesser extent, finely ground limestone (calcium carbonate). Other useful inputs include natural and synthetic organic materials.

SOIL STRUCTURE LIMITATIONS

Key Processes

Good soil structure is vital for the growth of most plants (paddy rice is an exception), particularly where water application is either uncontrolled (rainfed) or not easily controlled (flood irrigated). Soil with excellent structural form accepts water readily, drains excess water quickly, and has a broad range of water contents over which waterlogging (poor aeration) and excessive hardness do not limit seedling emergence or root growth.^[1]

Structural form refers to the arrangement of the solid components of soil and the associated pore space.^[2] Structural stability is the ability of soil to retain its structural form when subjected to disruptive forces such as immersion in water.

There are two aspects of soil stability in water: slaking and dispersion.^[3] Slaking refers to the collapse of air-dry soil aggregates into subunits (diameter of about 0.2 mm) when placed in rainwater. It indicates that the bonds created by materials such as organic matter are not strong enough to withstand the forces associated with rapid wetting. If the subunits created by slaking break down further, leading to a separation of clay, sand, and silt, dispersion is said to have occurred. An associated limitation is excessive subsoil swelling. Formation of a dispersed surface seal under wet conditions leads to poor aeration and inadequate water penetration, and the runoff water may then cause erosion. Hard crusts develop when dispersed soil is dried, particularly where the shrink/swell potential is poor.^[2]

As the amount of sodium adsorbed on the surfaces of clay particles increases, so does dispersion. This problem is aggravated by a lack of electrolyte in soil solution. Clay type also is important. Illite disperses more readily than smectite clay, and kaolinite clay become more dispersive as pH increases. The presence of exchangeable magnesium on clay surfaces aggravates

soil dispersion. Several direct measures of soil dispersibility are available.^[4,5]

Management Options

Poor structure can be overcome indirectly by special management inputs. Options include:

1. Raised beds and/or extra nitrogen fertilizer to deal with waterlogging limitations; and
2. The use of drip irrigation systems to apply water in a way that maintains a very narrow range of soil water contents, where neither waterlogging nor soil strength will limit growth.

However, it is usually less expensive and less risky to improve structural stability by adding ameliorants to the soil. The benefits of gypsum application to sodic (dispersive) soil have been recognized since the early 1900s in the United States.^[6] The practice of applying lime (chalk) to improve the structure of clay soil has been carried out for thousands of years in England.^[7] In developing countries, ash and organic household waste often are used to fertilize nutrient-depleted soil^[8] but may also improve soil structure.

Factors to consider before selecting an amelioration strategy include:

1. Soil condition should be measured so that amelioration is based on objective data about the severity of soil degradation.
2. The types and amounts of soil ameliorants that are applied will be strongly influenced by their cost in relation to the requirements and value of the plants being grown.

Secondary benefits, such as overcoming sulfur deficiency with gypsum and reducing acidity limitations with lime, should be considered when choosing an ameliorant. If the

soil has been compacted by heavy machinery, deep ripping may also be required.

AMELIORANTS FOR IMPROVING SOIL STRUCTURE

Gypsum

Gypsum is the main compound used for the reclamation of soil that is dispersive and/or prone to excessive swelling. The benefits of using gypsum to improve soil structural stability are due to both electrolyte concentration and cation exchange effects (replacement of sodium ions attached to the clay surfaces with calcium ions).^[9] Gypsum is available as a mined product or can be obtained as a by-product of industrial processes. Mined gypsum, which has a coarser particle size than by-product gypsum, tends to be less soluble. This is beneficial in situations where a slow-release source of electrolyte and calcium is required. Gypsum can be either spread as a solid or dissolved in irrigation water. A typical application rate on sodic clay soil is 2.5 t/ha for dryland wheat and 7.5 t/ha for higher value irrigated crops.

Soil that is most likely to show an economically viable response to gypsum application has a high clay content (greater than 30%) and is sodic with an exchangeable sodium percentage (ESP) greater than 5.^[11] If the salt concentration in soil is very low, the critical ESP for dispersion may be as low as 2 (as found in the surface of some hard-setting soil types).

Overcoming a soil structure problem with gypsum may introduce new problems. For example, the correction of drainage limitations in a sodic clay soil may lead to a leaching of nitrates and other contaminants beyond the root zone. Therefore, soil water management needs to be adjusted after soil structural stability has been improved. Possible impurities in gypsum products also need to be considered. Mined gypsum often contains traces of clay, while some industrial gypsum may be contaminated by toxins such as cadmium and fluoride compounds. It is important to obtain data about the quality of ameliorants prior to application.

Lime

When the soil pH (measured in water) is less than approximately 7, the application of ground limestone is likely to release substantial amounts of electrolyte and calcium. Lime often is introduced to the soil naturally via irrigation water. Even in soil with an average pH > 7, there may be pockets of soil adjacent to roots with a much lower pH that will encourage the dissolution of lime. Liming products (ground limestone, calcium oxide, and calcium hydroxide) are a more concentrated form of calcium than gypsum. The addition of acidifying

materials^[12] can release calcium by dissolving lime that occurs naturally in soil.

Another option is to apply a blend of gypsum and ground limestone.^[11] The gypsum provides an immediate source of calcium and electrolyte, while the less-soluble lime provides a longer term supply.

Organic Matter

For soil that is prone to slaking, the application of organic matter is likely to be beneficial.^[3] Organic ameliorants are particularly useful for non-swelling topsoil that is prone to hardsetting. Natural organic products include crop residues, animal and poultry manure, and sewage sludge (biosolids). Synthetic soil conditioners such as polyacrylamide are also effective, but cost and poor persistence of benefits limits their use. Organic matter application can reduce dispersion if accompanied by calcium ions.^[13] Organic mulches protect the soil surface from the disruptive effects of raindrop impact. They also encourage soil fauna such as earthworms, which improve soil structure with their burrows and exudates.

CHALLENGES

Modelling the Effects of Ameliorants on Soil Condition and Plant Growth

Numerous field trials have been conducted throughout the world to determine the most effective rates and forms of gypsum (and related materials) for soil that is structurally unstable. However, the results generally are location-specific^[4] and are strongly influenced by climatic conditions. A model is needed to predict the most profitable rates of gypsum and other ameliorants for various land use systems on soil that is prone to slaking and/or dispersion.

Integration of Soil Amelioration Strategies with “Precision Agriculture” Technology

The introduction of crop yield and quality mapping^[14] will help land managers to identify soil structure problems within subsections of management units. This should allow ameliorants to be applied more efficiently via variable-rate application equipment, rather than as blanket applications using conventional machinery. Research is needed to optimize soil sampling strategies for yield map interpretation and to determine the extent to which within-field variability of soil properties can be minimized economically. More attention should also be given to the development of rapid and inexpensive field measurement techniques that allow the ameliorant requirements of soil to be mapped more accurately.

REFERENCES

1. Letey, J. Relationship between soil physical properties and crop production. *Adv. Soil S.* **1985**, *1*, 277–294.
2. Kay, B.D. Rates of change of soil structure under different cropping systems. *Adv. Soil Sci.* **1990**, *12*, 1–52.
3. Emerson, W.W. Structural decline of soils, assessment and prevention. *Aust. J. Soil Res.* **1991**, *29*, 905–921.
4. Rengasamy, P.; Churchman, G.J. Cation exchange capacity, exchangeable cations and sodicity. In *Soil Analysis: An Interpretation Manual*; Peverill, K.I., Sparrow, L.A., Reuter, D.J., Eds.; CSIRO: Collingwood, 1999; 147–157.
5. Field, D.J.; McKenzie, D.C.; Koppi, A.J. Development of an improved vertisol stability test for soilpak. *Aust. J. Soil Res.* **1997**, *35*, 843–852.
6. Richards, L.A., Ed. *Diagnosis and Improvement of Saline and Alkali Soils*; United States Department of Agriculture: Washington, 1954.
7. Russell, E.W. *Soil Conditions and Plant Growth*, 9th Ed.; Longman: London, 1961; 452 pp.
8. Okigbo, B.N. *Development of Sustainable Agricultural Production Systems in Africa*; International Institute of Tropical Agriculture: Ibadan, 1991.
9. Loveday, J. Relative significance of electrolyte and cation exchange effects when gypsum is applied to a sodic clay soil. *Aust. J. Soil Res.* **1976**, *14*, 361–371.
10. Levy, G.J.; Sumner, M.E. Mined and by-product gypsum as soil amendments and conditioners. In *Handbook of Soil Conditioners: Substances that Enhance the Physical Properties of Soil*, 1st Ed.; Wallace, A., Terry, R.E., Eds.; Marcel Dekker: New York, 1998; 187–215.
11. Abbott, T.S.; McKenzie, D.C. *Improving Soil Structure with Gypsum and Lime*; NSW Agriculture: Orange, 1996.
12. Somani, L.L.; Totawat, K.L. Mined and industrial waste products capable of generating gypsum in soil. In *Handbook of Soil Conditioners*; Wallace, A., Terry, R.E., Eds.; Marcel Dekker: New York, 1998; 257–291.
13. Muneer, M.; Oades, J.M. The role of Ca-organic interactions in soil aggregate stability II. field studies with ^{14}C -labelled straw, CaCO_3 and $\text{CaCO}_4 \cdot 2\text{H}_2\text{O}$. *Aust. J. Soil Res.* **1989**, *27*, 401–409.
14. Robert, P.C.; Rust, R.H.; Larson, W.E., Eds. *Proceedings of the Fourth International Conference on Precision Agriculture*; American Society of Agronomy: Madison, 1999.

Ammonia: Volatilization from Agricultural Soils

Paul G. Saffigna

Tropical Forestry, University of Queensland, Gatton, Queensland, Australia

John R. Freney

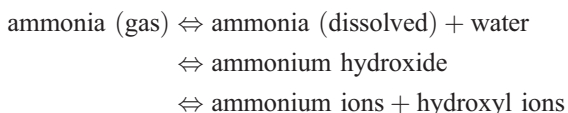
Plant Industry, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia

Abstract

Ammonia can be in the form of native organic matter, which decomposes to release ammonia, or fertilizer such as anhydrous ammonia, ammonium salts, or urea. Urea, either from animal urine or fertilizer, is rapidly hydrolyzed to ammonium carbonate in soil. This urease catalyzed reaction results in localized areas of high pH. Apart from the application of anhydrous ammonia, these sources tend to add ammonium ions rather than ammonia to the soil. Therefore, the conversion of ammonium ions to ammonia controls the loss of ammonia.

INTRODUCTION

The exchange of ammonia between soils, plants, waters, and the atmosphere is an important part of the terrestrial nitrogen cycle. The process by which ammonia is lost from the earth's surface to the atmosphere is termed volatilization. The primary source of ammonia for loss is the natural microbial decomposition of amino acids and proteins in dead plants, animals, and microorganisms in soils and waters, but substantial amounts come from the excreta of animals and the use of nitrogen fertilizers.^[1,2] Ammonia (NH₃) has a strong affinity for water and it readily dissolves in it to form ammonium (NH₄⁺) hydroxide:



The ammonia and ammonium ions are in equilibrium and the reaction may be displaced to the left or right depending on the conditions. For example, if the pH of the system is increased by addition of alkali (hydroxyl ions), the reaction is displaced to the left and ammonia gas is formed and lost to the atmosphere. As ammonia is a gas at normal temperatures and pressures, and as the concentration in the atmosphere is usually low, it can be readily lost to the atmosphere. However, volatilization is a complex process affected by a combination of biological, chemical, and physical factors, and the loss process may be hindered.^[1,2]

THE MECHANISM OF AMMONIA VOLATILIZATION

Before volatilization can occur there must be a source of ammonia. This can be in the form of native organic matter,

which decomposes to release ammonia, or fertilizer such as anhydrous ammonia, ammonium salts, or urea. Urea, either from animal urine or fertilizer, is rapidly hydrolyzed to ammonium carbonate in soil. This urease catalyzed reaction results in localized areas of high pH. Apart from the application of anhydrous ammonia, these sources tend to add ammonium ions rather than ammonia to the soil. Therefore, the conversion of ammonium ions to ammonia controls the loss of ammonia.^[1,2]

The relative concentrations of ammonium and ammonia in solution are strongly affected by pH (acidity or alkalinity) and temperature. For example, the percentage of ammonia present at pH 6, 7, 8, and 9 is approximately 0.1, 1, 10, and 50.^[3] Thus the higher the pH, the greater is the potential for ammonia loss from soil. The loss of ammonia from an application of ammonium sulfate increased from nil at pH 7 to 87% at pH 10.5.

The main driving force for ammonia volatilization is the difference in concentration between ammonia gas in the soil and ammonia in the atmosphere. Increasing windspeed increases the rate of volatilization by promoting more rapid transport of ammonia away from the soil surface. A four-fold increase in wind speed resulted in a ten-fold increase in ammonia loss. Ammonia volatilization and water loss from soils are directly related; no ammonia is emitted until evaporation commences.

Any factor which affects the ammonium ion concentration in soil will also affect the ammonia gas concentration and the loss process. Consequently, plant uptake, immobilization by microorganisms, nitrification, and leaching will reduce the amount of nitrogen available for volatilization, whereas increasing the rate of application of ammonium or ammonium producing fertilizers or organic residue will increase the potential for volatilization.^[1,2]

Other factors which control ammonia volatilization are cation exchange capacity, buffer capacity, presence of calcium carbonate, water content of soil, soil texture, plant residues, fertilizer form, radiation, and atmospheric ammonia concentration.^[3] A number of models incorporating these factors have been developed to describe the volatilization of ammonia.^[4]

MEASUREMENT

A number of workers have used canopies over field crops and pastures, combined with acid traps, to measure ammonia loss, but the canopies affect the temperature, moisture, and wind speed in the immediate environment of the plant, and thus the result obtained may not reflect ammonia loss from the natural environment. Simplified micrometeorological techniques have been developed to measure ammonia volatilization in the unconfined field situation, thus allowing the assessment of the importance of ammonia loss, and the factors controlling loss in different agricultural systems with minimum labor, equipment, and skills.^[1,4]

AMMONIA EMISSIONS

Animals and Their Wastes

Waste from farm animals is the principal source of atmospheric ammonia. Ammonia concentrations in the air range from as low as $1 \mu\text{g Nm}^{-3}$ over oceans to 5 in rural areas, 15 in urban areas, 50 in areas of intense animal husbandry, and 1000 over a field shortly after spreading animal waste as a slurry. The importance of animals as a source of ammonia is well illustrated by the situation in Europe, where nearly three quarters of the total emissions are from animals and their wastes (stables 34%, surface spreading 32%, and grazing 8%), and only one quarter comes from a combination of fertilizer application (12%), industry (0.5%), crops (5%), and miscellaneous (8%, e.g., treatment of waste water and sludge, pets, humans, and refrigeration).^[1]

The large loss of ammonia from livestock systems is due to the low conversion of dietary nitrogen into animal protein. More than 75% of the nitrogen intake is excreted in forms that give rise to ammonia emissions. Nitrogen is excreted mostly in urine with some present in droppings as microbial cell constituents and undigested food. Urine contains 70–90% urea which is rapidly hydrolyzed to ammonia by naturally occurring urease. The amount excreted depends on the feed composition—the better the quality of the food the less nitrogen excreted. Factors influencing ammonia emissions include manure properties, weather, soil attributes, and application measures (incorporation, dilution, and soil preparation). Where animals are grazing pastures, about 10% of the excreted nitrogen is lost as ammonia—mostly from urine.^[1]

Cropping Systems

There has been a widespread move to urea as the major form of nitrogen fertilizer used in cropping systems because of its relatively low manufacturing cost and low transportation cost per unit of nitrogen. However, large losses of ammonia have been detected from rainfed and irrigated crops and flooded rice in many countries following applications of urea (Table 1). Ammonia loss from cropping systems is affected by many of the factors discussed above, fertilizer composition, rate, time, and method of application, and factors unique to the crop.

In flooded rice, up to 56% of the applied nitrogen is lost from the system by ammonia volatilization as a result of the growth of photosynthetic algae, which markedly increases the pH of the floodwater during daytime. The new practice of retaining tops and leaves of cut sugarcane plants on the soil surface following green cane harvesting has created problems for farmers when they apply urea. So as not to disturb the residue cover, which has many advantages, including weed control, many farmers apply urea by broadcasting onto the surface of the residue. The sugarcane residue has high urease activity and low ammonia retention capacity, so when the residue layer is moistened by dewfall, rainfall, or condensation of evaporated soil water, some of the urea dissolves, is hydrolyzed, and when the water evaporates, between 30% and 40% of the applied nitrogen is lost as ammonia. Bananas have a very high requirement for nitrogen, frequent applications are made and more than $500 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ is applied. Direct measurements of ammonia volatilization from a banana crop in tropical Australia showed that, when urea was applied onto wet soil, 20% of the applied N was lost even though 90 mm of rain fell during the study. The extensive canopy of banana plants restricted rainwater from falling on the fertilized area and washing the urea into the soil.

Plants can absorb ammonia from the air or release it to the atmosphere. It has been established that plants have an

Table 1 Ammonia volatilized (% of N applied) from different cropping systems fertilized with urea.

Plant	Treatment	Loss
Bananas	Surface applied	20
Corn	Surface applied	22
Rice	Broadcast into floodwater	10–56
	Broadcast into floodwater and incorporated	10–43
	Incorporated before flooding	5–16
	Broadcast 12 days after transplanting	21
	Broadcast at panicle initiation	3
Sugarcane	Broadcast on trash	30–40
Wheat	Surface applied	36
	Buried	7

Source: Adapted from Peoples, Freney, et al.^[3]

ammonia compensation point, which is a finite ammonia concentration in the intercellular air spaces of plant leaves. Plants absorb or lose ammonia depending on whether the ambient ammonia concentration is above or below the compensation point. Ammonia losses of about $1 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ have been determined from fields of barley, maize, and wheat.^[1]

Biomass Burning

During combustion, considerable plant nitrogen is converted into gaseous forms including ammonia. Most biomass burning (~90%) occurs in the tropics as a result of forest clearing, savanna and sugarcane fires, and burning of agricultural wastes and firewood. According to Intergovernmental Panel on Climate Change,^[5] 8700 Mt of biomass is burned every year, and agriculture accounts for half of this. About 4% of the biomass nitrogen is released as ammonia during combustion, with the result that biomass burning contributes between 4.5 and 5.9 Mt N per yr^{-1} globally to the atmosphere.^[6,7]

GLOBAL SIGNIFICANCE

The global emission of ammonia was estimated to be 54 Mt N yr^{-1} in 1990.^[7] The contributions from the major sources were given as: 1) excreta from animals, 21.7 Mt; 2) synthetic fertilizers, 9.0 Mt; 3) oceans, 8.2 Mt; 4) biomass burning, 5.9 Mt; 5) crops, 3.6 Mt; 6) human population and pets, 2.6 Mt; 7) soils under natural vegetation, 2.4 Mt; 8) industrial processes, 0.2 Mt; and 9) fossil fuels, 0.1 Mt. About half of the global emission originates in Asia, and approximately 70% is associated with food production.

When ammonia is emitted into the atmosphere, some is absorbed by vegetation, some is dissolved in atmospheric water, converted to aerosols and transported long distances (>1000 km), and some is deposited nearby.^[4,8] Model estimates indicate that about 50% of the emitted ammonia is deposited within 50 km of the source.^[8] High ammonia concentrations close to point sources such as cattle feedlots can damage vegetation, and deposition of ammonia and ammonium can result in acidification of soils and lakes, increased carbon storage in pristine areas, and increased emission of the greenhouse gas nitrous oxide;^[9] measures need to be instituted to reduce losses.

MITIGATION

Techniques proposed for reducing loss of ammonia from animal slurries applied to soils include incorporation or injection of the slurry into the soil, application with trail

hoses, acidification before application, applying during rainfall, at night, or in winter, and matching nitrogen supply to the demand of the crop. Decreasing the water content of the slurry and delaying application until a substantial canopy has developed (to reduce wind speeds) would also appear to have a large impact on ammonia loss.^[10] A logical option for limiting ammonia volatilization from cropping systems is to drill the fertilizer into the soil. Other recommendations include spreading the fertilizer just prior to rain, application in irrigation water, optimizing split application schemes, changing the fertilizer type to suit the conditions, and better matching of nitrogen supply with crop demand. Controlled release fertilizers provide an opportunity to match supply and demand while protecting the remainder of the fertilizer from release. Adoption of some or all of these practices will allow the farmer to reduce inputs and reduce the impact on the environment.^[5]

REFERENCES

1. ECETOC. *Ammonia Emissions to Air in Western Europe*; Technical Report No. 62; ECETOC (European Centre for Ecotoxicology and Toxicology of Chemicals); Brussels, 1994; 196 pp.
2. Nelson, D.W. Gaseous losses of nitrogen other than through denitrification. In *Nitrogen in Agricultural Soils*; Stevenson, F.J., Ed.; American Society of Agronomy: Madison, 1982; 327–363.
3. Peoples, M.B.; Freney, J.R.; Mosier, A.R. Minimizing gaseous losses of nitrogen. In *Nitrogen Fertilization in the Environment*; Bacon, P.E., Ed.; Marcel Dekker: New York, 1995; 565–602.
4. Denmead, O.T. Progress and challenges in measuring and modelling gaseous nitrogen emissions from grasslands: an overview. In *Gaseous Nitrogen Emissions from Grasslands*; Jarvis, S.C., Pain, B.F., Eds.; CABI: Wallingford, UK, 1997; 423–438.
5. Watson, R.T.; Zinyowera, M.C.; Moss, R.H. IPCC (Intergovernmental Panel on Climate Change). In *Climate Change 1995. Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses*; Cambridge University Press: Cambridge, 1996; 1–878.
6. Schlesinger, W.H.; Hartley, A.E. A global budget for atmospheric NH_3 . *Biogeochemistry* **1992**, *15*, 191–211.
7. Bouwman, A.F.; Lee, D.S.; Asman, W.A.H.; Dentener, F.J.; Van der Hoek, K.W.; Olivier, J.G.J. A global high-resolution emission inventory for ammonia. *Global Biogeochem. Cycles* **1997**, *11*, 561–587.
8. Ferm, M. Atmospheric ammonia and ammonium transport in europe and critical loads—A review. *Nutr. Cycl. Agroecosyst.* **1998**, *51*, 5–17.
9. Smil, V. Nitrogen in crop production: an account of global flows. *Global Biogeochem. Cycles* **1999**, *13*, 647–662.
10. Jarvis, S.C.; Pain, B.F. *Gaseous Nitrogen Emissions from Grasslands*; CABI: Wallingford, 1997.

Amoozometer

Aziz Amoozegar

Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.

Abstract

Hydraulic conductivity, defined as a measure of the ability of soil to transmit water, is an important dynamic soil property that is needed to solve agricultural, hydrological, and environmental problems. Since unsaturated hydraulic conductivity (K_{unsat}) measurements are difficult and time-consuming, saturated hydraulic conductivity (K_{sat}) is used to estimate K_{unsat} or assess water flow in the vadose zone using mathematical models. The constant-head well permeameter method is the most versatile procedure for determining K_{sat} in the vadose zone. In this procedure, the steady-state rate of water flow into the soil at the bottom of a cylindrical hole of known diameter ($2r$) under a constant depth of water (H) is measured, and K_{sat} is calculated using an appropriate model. The Amoozometer is a portable and easy-to-use permeameter for measuring K_{sat} of the vadose zone from the soil surface to 2 m depth. Measurements can be performed using a 4–10 cm diameter auger hole without excavating a pit. Using an additional set of constant-head tubes, measurement depth can be increased to 4 m. The Amoozometer has a 5-L water capacity and can be used on any landscape position. The Glover model, developed based only on the saturated water flow from the hole, is recommended for calculating K_{sat} because it is simple, does not require estimation of any soil property, and results in consistent values for all conditions. In addition, the Glover model compares well with other models that consider both saturated and unsaturated flow for H/r values ≥ 5 .

INTRODUCTION

Water flow through the saturated and unsaturated zones, governed by Darcy's law, is one of the major components of the hydrologic cycle. Water transmission is one of the most important dynamic processes in the soil, and it impacts many of the soil's properties and processes. Hydraulic conductivity is defined as a measure of the ability of soil to transmit water. It is also the proportionality factor in Darcy's law relating flux density or flux to the hydraulic gradient. Unsaturated hydraulic conductivity (K_{unsat}) depends on soil water content (θ) or pressure head (h); i.e., $K_{\text{unsat}} = K(\theta)$ or $K(h)$; however, obtaining a complete unsaturated hydraulic conductivity curve will be impractical for most applications. As an alternative, saturated hydraulic conductivity, generally denoted as K_{sat} (or K_s), is used in mathematical models to estimate K_{unsat} for modeling purposes.^[1] Although soil hydraulic conductivity is a highly variable soil property, K_{sat} can be taken to be a constant for a given time and space within the soil, and its value for the vadose (unsaturated) zone is often needed to solve many agricultural, hydrological, and environmental problems. Examples include designing a drainage system for controlling seasonal high water table and determining groundwater mounding under wastewater application areas or retention ponds. For most practical applications, K_{sat} of the vadose zone from the soil surface to depths exceeding a few meters can be measured conveniently by the constant-head well permeameter method using an Amoozometer.

CONSTANT-HEAD WELL PERMEAMETER METHOD

There are a number of procedures for *in situ* determination of soil K_{sat} above a water table (in the vadose zone) and below a water table (within an aquifer).^[2,3] The constant-head well permeameter method, also known as the shallow well pump-in technique or the borehole permeameter method, is perhaps the most versatile procedure used throughout the world for measuring the K_{sat} of the vadose zone. This procedure is completed in two steps; field measurement of the steady-state water flow rate into the soil through a cylindrical hole under a constant head and calculation of the K_{sat} using an appropriate model. For field measurement, a cylindrical hole of known radius (r) is dug to the desired depth. After establishing a constant depth of water in the hole, referred to as head of water (H), the steady-state rate of water flow from the hole into the soil (Q) is measured for calculating K_{sat} . A constant head of water at the bottom of the hole can be maintained using a small float,^[4] or a Mariotte bottle system.^[2,3,5] The Amoozometer (Fig. 1), also known as the compact constant-head permeameter (CCHP),^[6] is an instrument that can be used to measure K_{sat} by this procedure.

DESCRIPTION OF THE AMOOZEMETER

The Amoozometer is a portable, versatile, and easy to operate device that performs two functions: it maintains a



Fig. 1 Photograph of the Amoozometer.

Source: Photo courtesy of Ksat, Inc.

constant depth of water at the bottom of a cylindrical hole, and allows measurements of the rate of water entering the hole (equal to the rate of water infiltrating the soil at the bottom of the hole). With an Amoozometer, K_{sat} can be measured from the soil surface to 2 m depth without excavating a pit. Using an additional set of constant-head tubes (an accessory), the depth of measurement can be easily extended to 4 m.

An Amoozometer (hereafter referred to as CCHP) is composed of a set of four constant-head tubes, a 4-L water reservoir, a flow-measuring reservoir, a water-dissipating unit, and a base that houses a three-way valve that connects the two reservoirs and the water-dissipating unit together (Fig. 2).

The four constant-head tubes are based on the Mariotte principle and are constructed of clear polycarbonate tubing. They develop up to -200 cm of constant water pressure (vacuum) for maintaining the water level in an auger hole at a fixed distance below the device placed next to the hole on the land surface. Each constant-head tube is equipped with a “bubble tube” and must be filled with clean water (Fig. 2). The bubble tube in the first constant-head tube is adjustable and provides from 0 to -50 cm of water pressure. The bubble tube in each of the other three

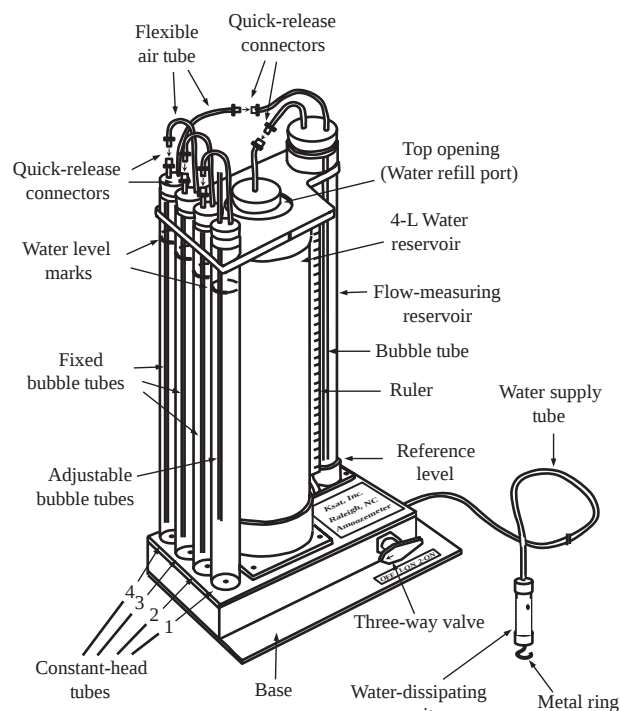


Fig. 2 Schematic diagram of the Amoozometer.

Source: Photo courtesy of Ksat, Inc.

constant-head tubes is fixed and provides approximately -50 cm of water pressure. During operation, the height of water above the end of the bubble tube in each of the fixed constant-head tubes is approximately 50 cm. The negative pressure (up to -200 cm of water), corresponding to the desired distance between the level of water in the auger hole and the reference level on the base of the CCHP placed at the soil surface, is obtained by serially connecting an adequate number of constant-head tubes and adjusting the bubble tube in the first constant-head tube.

The 4-L reservoir at the center of the CCHP is constructed of durable polyvinyl chloride (PVC) pipe. It maintains the center of gravity above the middle of the base of the unit. The flow-measuring reservoir is constructed of clear polycarbonate tube and is equipped with a bubble tube. The tip of this bubble tube is set at a fixed position in the reservoir and is used as the reference level for maintaining a constant depth of water at the bottom of the hole. These two reservoirs are connected at the bottom through a three-way valve housed inside the base of the device. A large opening on top of the 4-L reservoir allows rapid filling of the CCHP with water. For operation, the bubble tube in the flow-measuring reservoir is connected to the series of constant-head tubes, and the tops of the two reservoirs are connected together to allow the air pressures above the water levels in the reservoirs to be the same.

The outlet of the three-way valve is connected to a water-dissipating unit through a 2.5-m long flexible plastic tubing (Fig. 2). The water-dissipating unit allows water to

be applied to the bottom of the hole uniformly without scouring the auger hole wall. A metal ring at the end of the water-dissipating unit minimizes the blockage of the bottom of the (cylindrical) hole during measurements.

The constant-head tube set for increasing the depth of measurement to 4 m below the land surface is similar to the setup of the four constant-head tubes on the CCHP. The only difference is that all the four bubble tubes are fixed, and each constant-head tube provides pressure equal to approximately -50 cm of water. For measurements, an adequate number of constant-head tubes from the set are serially connected to the four constant-head tubes on the permeameter on one side and to the flow-measuring reservoir on the other. In addition, for measurements below 2-m depth, the flexible tubing that connects the water-dissipating unit to the three-way valve is replaced with a 4.5-m long flexible plastic tube.

FIELD DATA COLLECTION

In addition to the CCHP, a set of augers, measuring tape, stopwatch, data sheet (Fig. 3), small shovel, and more than 5 L of water are needed. At the location of interest, an auger hole of known diameter ($2r$) is dug to the desired depth. A 6-cm diameter hole is highly recommended, but auger holes ranging from 4 to 10 cm in diameter can be used depending on the application and type of soil. (A large diameter hole should only be used in a soil horizon that is thick and has a relatively low K_{sat} .) A larger hand auger or a mechanical auger can be used to dig the hole to approximately 50 cm above the desired depth. The bottom 50 cm of the hole must be dug using a hand auger capable of boring the hole with the desired diameter. After digging the hole to the desired depth, a planer auger of the same diameter as the auger is used to cut the bottom of the hole and form a cylindrical-shaped cavity. A round nylon brush or another device^[3] may be used to remove smearing from the auger hole sidewall. Under certain conditions (e.g., the soil being too wet), however, brushing the hole may not reduce smearing. Various sizes of augers, planer augers, and brushes are commercially available.

The CCHP is placed on a flat area next to the auger hole, and the vertical distance from the bottom of the hole to the reference level on the CCHP is measured (distance D in Fig. 4). After selecting the desired depth of water in the hole (H), an adequate number of constant-head tubes are serially connected to the flow-measuring reservoir, and the bubble tube in the first constant-head tube is adjusted such that the cumulative distance between the tip of the bubble tube and the water level in the constant-head tubes in the series equals the distance from the reference level on the CCHP to the desired water level in the hole (distance d in Fig. 4). In Fig. 4, the proper connections of the constant-head tubes and the flow-measuring reservoir for measurements between a 100 and 150 cm depth are shown. Leaving the

top of the 4-L reservoir open, the three-way valve is fully opened to fill the flexible tube of the water-dissipating unit with water from both reservoirs. Then, after closing the three-way valve, and connecting the tops of the two reservoirs together, the water-dissipating unit is inserted to the bottom of the hole and the three-way valve is fully opened to allow water from both reservoirs to fill the hole. Shortly thereafter, a constant depth of water will be established in the hole. Finally, the adjustable bubble tube is used to adjust the depth of water in the hole to the desired level. At this point, all measured data are recorded in the data sheet (Fig. 3).

Water is allowed to move from the hole into the soil until a steady-state flow rate is achieved. If the flow rate of water into the hole is small (e.g., 50 cm³/hr), the three-way valve is turned to the flow-measuring reservoir only for more accurate measurements of water flow from the CCHP. The depth of the water in the hole must be checked (through checking distance d) frequently to make sure that a constant depth of water is maintained in the hole. For most practical applications, the steady-state condition is reached when three consecutive measurements of the flow rate under a constant depth of water are the same. After determining the steady-state infiltration rate, the K_{sat} value is calculated using the Glover equation or another appropriate model.

MODELS FOR CALCULATING K_{SAT}

There are a number of models for calculating K_{sat} using field data.^[3,4,7-10] One of the simplest models for calculating K_{sat} is the Glover equation:^[7]

$$K_{sat} = \left\{ \sinh^{-1} \left(\frac{H}{r} \right) - \left[\left(\frac{r}{H} \right)^2 + 1 \right]^{1/2} + \frac{r}{H} \right\} \frac{Q}{(2\pi H^2)} \quad (1)$$

where r , H , and Q are as defined before, and $\sinh^{-1}$ is the inverse hyperbolic sine function. This equation is for cases where the distance between the bottom of the hole and an impermeable layer below the hole (distance s in Fig. 4) is $\geq 2H$. To use the Glover equation, Amoozegar^[11] has suggested that H/r must be ≥ 5 . When s is $< 2H$, the equation

$$K_{sat} = \frac{3Q \ln \left(\frac{H}{r} \right)}{\pi H (3H + 2s)} \quad (2)$$

can be used for calculating K_{sat} . In Eqs. 1 and 2, r , H , and s have units of length (e.g., cm), Q has units of volume over time (e.g., cm³/hr), and K_{sat} has units of length over time (e.g., cm/hr).

The Glover model, Eq. 1, is based only on the saturated flow around the hole. Some of the other models, on the other hand, consider both saturated and unsaturated flow of water away from the hole. Stephens and Neuman^[10] presented an empirical equation based on saturated-unsaturated flow

SAMPLE DATA SHEET

Measurement No. #6 Conducted by Christopher Niewoehner Page 1 of 1
 Location Lake Wheeler Center, Wake County, NC Date 5/18/2004
 Weather Condition Sunny Temperature 80 °F
 Horizon Bt Source of Water City of Raleigh Tap Water

Hole depth 51 cm Radius of the hole (r) 3 cm
 Distance between reference level and soil surface 8 cm Measured (Actual) water level in hole
 Initial 15.5 cm
 Distance from the hole bottom to the reference level (D) 59 cm Final 15.5 cm
 Desired water depth in hole (H) 15 cm Clock time
 Start saturation 9:15 a.m.
 Constant-head tube setting (d) 44 cm Steady-state reading 10:30 a.m.

Reservoirs Used for Measurement of the Steady-State Flow Rate

Flow Measuring Reservoir Only _____ Conversion Factor (C.F.) = 20 cm²
 Both Flow Measuring and Main Reservoirs X Conversion Factor (C.F.) = 105 cm²

(To obtain flow volume multiply change in water level by the appropriate C.F. from above)

Clock Time h:min	Reservoir Reading cm	Δt min	Change in Water Level cm	Flow Volume cm ³	Q cm ³ /min	Q cm ³ /h	K _{sat} cm/h
<u>9:15</u>	<u>41.9</u>	_____	_____	_____	_____	_____	_____
<u>9:30</u>	<u>40.2</u>	<u>15</u>	<u>1.7</u>	<u>179</u>	<u>11.9</u>	<u>714</u>	<u>0.72</u>
<u>9:45</u>	<u>38.7</u>	<u>15</u>	<u>1.5</u>	<u>158</u>	<u>10.5</u>	<u>630</u>	<u>0.63</u>
<u>10:00</u>	<u>37.2</u>	<u>15</u>	<u>1.5</u>	<u>158</u>	<u>10.5</u>	<u>630</u>	<u>0.63</u>
<u>10:15</u>	<u>35.9</u>	<u>15</u>	<u>1.3</u>	<u>137</u>	<u>9.1</u>	<u>546</u>	<u>0.55</u>
<u>10:30</u>	<u>34.6</u>	<u>15</u>	<u>1.3</u>	<u>137</u>	<u>9.1</u>	<u>546</u>	<u>0.55</u>
<u>10:45</u>	<u>33.4</u>	<u>15</u>	<u>1.2</u>	<u>126</u>	<u>8.4</u>	<u>504</u>	<u>0.51</u>
<u>11:00</u>	<u>32.2</u>	<u>15</u>	<u>1.2</u>	<u>126</u>	<u>8.4</u>	<u>504</u>	<u>0.51</u>
<u>11:15</u>	<u>31.0</u>	<u>15</u>	<u>1.2</u>	<u>126</u>	<u>8.4</u>	<u>504</u>	<u>0.51</u>
_____	_____	_____	_____	_____	_____	_____	_____

Average of last three measurements: K_{sat} = 0.51 cm/h 12.2 cm/day (other units)

COMMENTS: _____

Fig. 3 A sample data sheet showing recording of data using an Amoozemeter.

Source: Photo courtesy of Ksat, Inc.

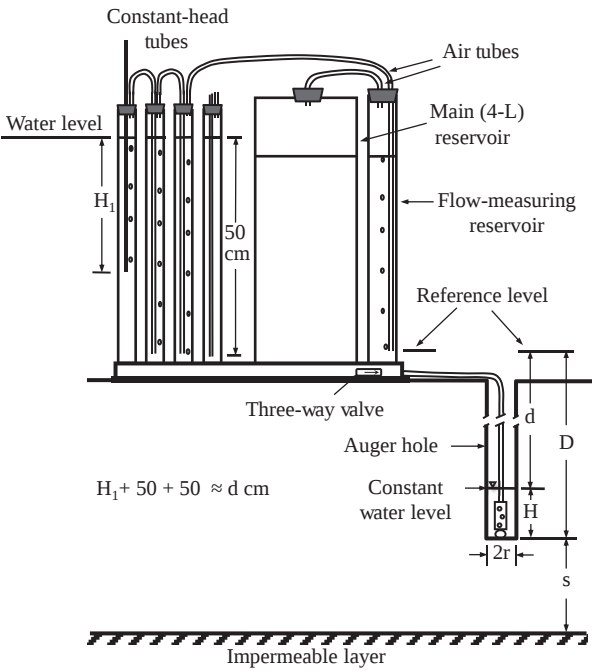


Fig. 4 Schematic diagram of the CCHP showing the proper connection of the constant-head tubes and the two reservoirs for measurements between 100 and 150 cm depth.
Source: Photo courtesy of Ksat, Inc.

through isotropic soils for estimating K_{sat} in a vadose zone with deep water table. Philip^[8] and Reynolds, Elrick, and Clothier^[9] used similar assumptions and developed models for calculating K_{sat} using the empirically developed equation relating the unsaturated hydraulic conductivity $K(h)$ to K_{sat} by^[1]

$$K(h) = K_{sat} \exp(\alpha h) \quad (3)$$

where h is the soil water pressure head and α (units of length^{-1} , e.g., cm^{-1}) is a measure of capillary properties of the soil, referred to as the sorptive number. For

determining K_{sat} , the parameter α must be determined independently or be estimated based on the soil texture and structure.^[12,13] One of the approaches for determining K_{sat} by the Reynolds, Elrick, and Clothier^[9] model is the simultaneous equation approach.^[3] Amoozegar^[11] and Salverda and Dane^[14] have shown that this approach is unreliable because it may result in negative K_{sat} values. Estimating α based on soil properties also may result in errors if the operator misestimates the soil properties in the field. Based on the author's evaluation, the Glover model, used when H/r is ≥ 5 , results in K_{sat} values that are very close to the calculated values by Philip^[8] and Reynolds, Elrick, and Clothier^[9] models for most practical applications. Considering the assumptions made for the later models (e.g., isotropic conditions and formation of a saturated bulb around the hole) and similarity between the calculated K_{sat} values, the Glover model is recommended for calculating K_{sat} because it is simple and results in consistent hydraulic conductivity values. Figure 3 shows field data and the calculated K_{sat} for one measurement at a site near Raleigh, North Carolina. For this measurement, radius of the hole $r = 3$ cm, the constant depth of water in the hole $H = 15.5$ cm, and the steady-state rate of water flow into the hole $Q = 504$ cm^3/hr . Plugging these values in Eq. 1, the calculated $K_{sat} = 0.51$ cm/hr or 12.2 cm/d.

COMPARISON OF METHODS FOR MEASURING K_{SAT}

Hydraulic conductivity is a highly variable soil property with coefficient of variation ($100 \times \text{SD}/\text{mean}$) that generally exceeds 100%. Also, because of the dynamic nature of water flow in soils, and differences in the soil volume involved in various *in situ* hydraulic conductivity measurement procedures, it is impractical to choose a field method as the benchmark procedure for measuring K_{sat} . Schoeneberger, Amoozegar, and Buol^[15] reported the *in situ* K_{sat} of

Table 1 Arithmetic mean, SD, and number of observations (n) for saturated hydraulic conductivity (K_{sat}) of three horizons of a Cecil soil at three landscape positions measured *in situ* under 15 cm head and determined in the laboratory using intact core samples collected in vertical orientation.

Horizon		Ridge top		Shoulder		Ridge nose	
		<i>In situ</i> (10^{-6} m/s)	Core (10^{-6} m/s)	<i>In situ</i> (10^{-6} m/s)	Core (10^{-6} m/s)	<i>In situ</i> (10^{-6} m/s)	Core (10^{-6} m/s)
Bt	Mean	0.28	1.62	0.86	2.19	0.24	1.56
	SD	0.32	1.03	0.82	1.57	0.20	0.85
	n	3	8	3	7	3	7
B/C	Mean	0.23	0.07	0.57	0.23	0.27	0.19
	SD	0.31	0.05	0.20	0.09	0.19	0.16
	n	3	18	3	8	3	5
C	Mean	0.87	0.81	2.04	1.02	1.27	1.57
	SD	0.80	0.29	0.09	0.59	0.04	0.95
	n	3	24	3	27	3	23

three different horizons of a Cecil soil (a clayey, kaolinitic, and thermic Typic Kanhapludult), measured at three different geomorphic (landscape) positions under 15 and 25 cm heads using the Amoozometer and the Glover model (Eq. 1), along with the laboratory measured K_{sat} of intact core samples collected from the same horizons in different orientations. The *in situ* results for 15 cm head and laboratory-determined vertical K_{sat} reported by them are presented in Table 1. Their results showed good agreement between laboratory and *in situ* determined K_{sat} values for saprolite (C horizon) with no soil structure and few tubular pores or planar voids. The average laboratory values for the Bt horizon containing tubular and planar voids, on the other hand, were consistently higher than the corresponding *in situ* values. For the B/C horizon, although the average values for *in situ* K_{sat} were numerically higher than the corresponding values for cores, there were no significant differences between the *in situ* and laboratory measured K_{sat} for all three landscape positions.

CONCLUSION

The constant-head well permeameter method is perhaps the best procedure for measuring saturated hydraulic conductivity (K_{sat}) of the vadose (unsaturated) zone. The Amoozometer is a portable and versatile device that allows measurement of K_{sat} by this procedure from the soil surface to depths exceeding a few meters using only a small diameter hole. The device is constructed of durable materials, does not require field assembly, and is easy to use. The soil K_{sat} at the depth of interest can be calculated using the relatively simple Glover model. For most practical applications, K_{sat} can be measured in 1–3 hours using less than 5 L of water.

ACKNOWLEDGMENTS

The original research and development of Amoozometer were supported by the North Carolina Agricultural Research Service. The use of trade name in this entry does not imply endorsement by the North Carolina Agricultural Research Service of the product named or criticism of similar ones not mentioned.

REFERENCES

- Gardner, W.R. Some steady-state solutions of the unsaturated moisture flow equation with application to evaporation from a water table. *Soil Sci.* **1958**, *85* (4), 228–232.

- Amoozegar, A.; Wilson, G.V. Methods for measuring hydraulic conductivity and drainable porosity. In *Agricultural Drainage*, Agronomy No. 38; Skaggs, R.W., van Schilfgaarde, J., Eds.; American Society of Agronomy, Inc.: Madison, 1999; 1149–1205.
- Reynolds, W.D.; Elrick, D.E. Constant head well permeameter (vadose zone). In *Methods of Soil Analysis, Part 4. Physical Methods*, Book Series No. 5; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America: Madison, 2002; 844–858.
- Stephens, D.B.; Lambert, K.; Watson, D. Regression models for hydraulic conductivity and field test of the borehole permeameter. *Water Resour. Res.* **1987**, *23* (12), 2207–2214.
- Amoozegar, A.; Warrick, A.W. Hydraulic conductivity of saturated soils: Field methods. In *Methods of Soil Analysis, Part 1. Physical and Mineralogical Methods*, Agronomy No. 9, 2nd Ed., Klute, A., Ed.; ASA and SSSA: Madison, 1986; 735–770.
- Amoozegar, A. A compact constant-head permeameter for measuring saturated hydraulic conductivity of the vadose zone. *Soil Sci. Soc. Am. J.* **1989**, *53* (5), 1356–1361.
- Zangar, C.N. *Theory and Problems of Water Percolation*. Engineering Monograph No. 8, Bur. Reclamation; U.S. Department of Interior: Denver, 1953.
- Philip, J.R. Approximate analysis of the borehole permeameter in unsaturated soil. *Water Resour. Res.* **1985**, *21* (7), 1025–1033.
- Reynolds, W.D.; Elrick, D.E.; Clothier, B.E. The constant head well permeameter: Effect of unsaturated flow. *Soil Sci.* **1985**, *139* (2), 172–180.
- Stephens, D.B.; Neuman, S.P. Vadose zone permeability tests: Steady state results. *J. Hydrol. Div. ASCE* **1982**, *108* (HY5), 640–659.
- Amoozegar, A. Comparison of the Glover solution with the simultaneous-equations approach for measuring hydraulic conductivity. *Soil Sci. Soc. Am. J.* **1989**, *53* (5), 1362–1367.
- Elrick, D.E.; Reynolds, W.D.; Tan, K.A. Hydraulic conductivity measurements in the unsaturated zone using improved well analyses. *Ground Water Monit. R.* **1989**, *9* (3), 184–193.
- Radcliffe, D.E.; West, L.T. Relating saturated hydraulic conductivity to percolation and borehole permeameter tests. *Soil Surv. Horizon.* **2000**, *41* (4), 99–103.
- Salverda, A.P.; Dane, J.H. An examination of the Guelph permeameter for measuring the soil's hydraulic properties. *Geoderma* **1993**, *57*, 405–421.
- Schoeneberger, P.J.; Amoozegar, A.; Buol, S.W. Physical property variation of a soil and saprolite continuum at three geomorphic positions. *Soil Sci. Soc. Am. J.* **1995**, *59* (5), 1389–1397.

Anaerobic Processes

Paul A. McDaniel

Daniel G. Strawn

University of Idaho, Moscow, Idaho, U.S.A.

Abstract

Anaerobic processes in soils occur when available molecular oxygen concentrations are limited. These processes include a myriad of biogeochemical processes that typically occur in soils that are saturated with water for extended periods. Anaerobic processes result in soil characteristics such as increased organic matter content, changes in biological activity, and formation of redoximorphic features.

INTRODUCTION

Availability of oxygen (O_2) in soils, or lack thereof, has a profound effect on soil biogeochemical processes. When molecular O_2 is absent in soils or present in very small quantities, anaerobic biological processes occur that create distinct soil chemical properties and morphological features as compared to aerobic soils.

ANAEROBIC CONDITIONS AND O_2 AVAILABILITY

A common means of assessing O_2 status of soils is measuring redox potential (E_h). E_h can be thought of as a measure of the electron activity of a system.^[1] E_h can be predicted using theoretical relationships or measured directly in the soil using a redox electrode. The exact relationship between O_2 and E_h within soils is dependent upon a number of specific soil properties; the general relationship illustrated in Fig. 1 shows that as O_2 levels in soils decline, E_h decreases (provided biological respiration is occurring). Levels of O_2 availability in soils are generally described using three categories (Fig. 2):

- *Oxic* or *aerobic* soils— O_2 is readily available; E_h range of ~300–800 mV at pH 7.
- *Suboxic* soils—limited O_2 availability; E_h values of ~100–300 mV at pH 7.
- *Anoxic* or *anaerobic* soils— O_2 is absent or otherwise unavailable to organisms (may occur in localized zones of the soil); E_h values less than ~100 mV at pH 7.

It is important to note that in soils, anaerobic conditions may occur on a microscale. The O_2 content of soil aggregates may change dramatically across a distance of a few millimeters.^[5] Due to slow gas exchange, the centers of wet aggregates as small as 8 mm in diameter can remain anaerobic even when dried and exposed to ambient air O_2 concentrations.

While the O_2 content of earth's atmosphere is approximately 21%, the O_2 content in soil air is typically lower. The main reservoir of O_2 in soils is air that occupies pore spaces between mineral particles, but O_2 is also dissolved in soil water. The gaseous and dissolved O_2 present in soils serves a critical biological function—it is consumed during aerobic respiration by soil microorganisms and by the roots of most plants. In aerobic respiration, organisms obtain energy through the oxidation of organic substrates, a process whereby electrons are passed through a series of carriers and ultimately accepted by O_2 .^[6]

DEVELOPMENT OF ANAEROBIC CONDITIONS

Several factors contribute to the decrease in O_2 availability and the development of anaerobic conditions in soils. Aerobic respiration consumes O_2 and generates carbon dioxide (CO_2). Therefore, without rapid gas exchange between the soil and the atmosphere, the soil atmosphere will become depleted in O_2 and enriched in CO_2 . The rate of gas exchange is dependent upon soil texture, structure, and the quantity, size, and connectivity of soil pores.^[7] In general, slower gas exchange occurs in fine-textured soils and those with poor soil structure. As soils become very wet and approach saturation, air is physically excluded from soil pores, thereby reducing the amount of air present and inhibiting gas exchange. The amount of O_2 that can be dissolved in soil water decreases as water temperature increases. Soil temperature also affects microbial activity in soils, and thus temperature is a key factor controlling soil redox processes.

ANAEROBIC CONDITIONS AND BIOLOGICAL ACTIVITY

Most plants are adversely affected by anaerobic conditions because root respiration decreases, thereby limiting

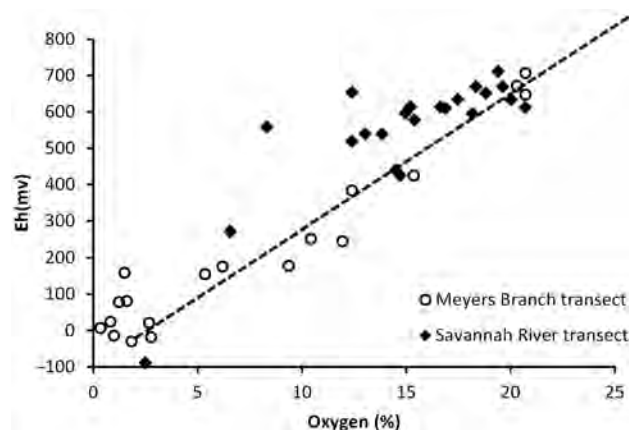


Fig. 1 Observed relationship between soil E_h (corrected to pH 5) and measured soil O_2 content from two wetland transects. Each point is an average of 22 monthly measurements at 4 cm depths from 10 sites.

Source: Reproduced from Megonigal, Patrick, et al.^[2] ©1993 with permission.

growth. As a result of decreased root activity, the uptake of water and nutrients also decreases.^[7] Some plants, however, have developed special adaptations to anaerobic soils and can transport O_2 from the aboveground into the roots.

When O_2 becomes limiting in soils, microbial populations shift to organisms capable of anaerobic and fermentative respiration, and soils become anaerobic.^[6] During anaerobic respiration, organisms use alternative electron acceptors. In soils, these are typically nitrogen,

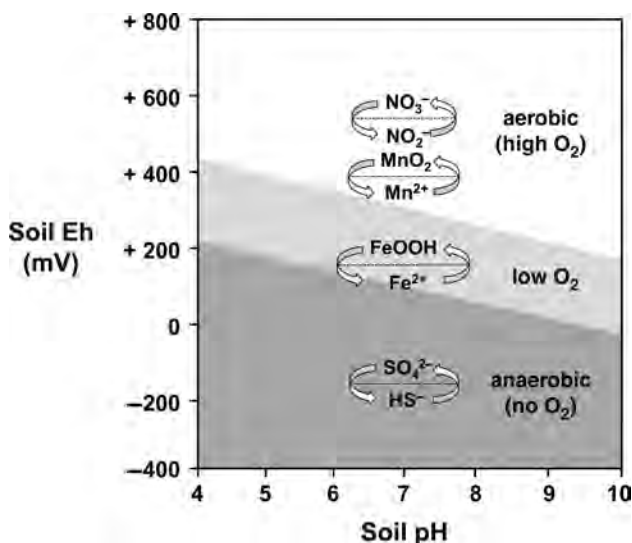


Fig. 2 Generalized relationship between soil O_2 availability, E_h , and pH. Shading patterns indicate the general aeration status of the soil. E_h values at which selected soil constituents are reduced at pH 7 are shown.

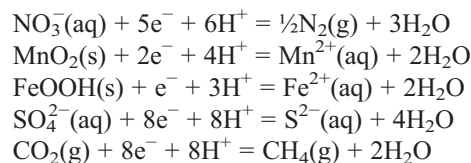
Source: Adapted from Evangelou.^[3] ©1998 John Wiley & Sons and Bass Becking, Kaplan, et al.^[4] ©1960.

manganese (Mn), iron (Fe), and sulfur (S). The E_h at which these reduction reactions occur in soils is dependent on a number of variables, including pH and the form of the element that is present. The theoretical E_h values at which some of these reduction reactions occur at pH 7.0 are shown in Fig. 2. As E_h decreases, the theoretical sequence of terminal electron acceptor use is: NO_3^- , Mn^{4+} , Fe^{3+} , and SO_4^{2-} .

A second consequence of the change in microbial activity from aerobic to anaerobic conditions is that the biodegradation of soil organic material decreases. Under aerobic conditions, the decomposition of organic materials can proceed rapidly.^[8] However, as the O_2 supply becomes limited, organic matter decomposition occurs via anaerobic respiration and fermentation. These are less efficient metabolic processes, resulting in slower decomposition of organic substrates. In soils with limited O_2 availability, this leads to a greater net accumulation of organic matter.

ANAEROBIC REACTIONS IN SOILS

Utilization of alternative or secondary electron acceptors during anaerobic respiration causes important changes in the speciation and behavior of many soil chemicals. Reduction of NO_3^- , Mn^{4+} , Fe^{3+} , SO_4^{2-} , and CO_2 are common reactions that occur in anaerobic soils. Possible half reactions for reduction of these species are



The electrons in these reactions come from an oxidizing reaction, typically heterotrophic oxidation of reduced carbon (i.e., $CH_2O + H_2O = CO_2 + 4e^- + 4H^+$). The half reaction involving nitrogen is a summary reaction that involves several steps and the formation of a variety of compounds, including NO_2^- , N_2 , and nitrous oxide (N_2O).^[9] Loss of gaseous nitrogen as N_2 or N_2O , a process known as denitrification, reduces nitrogen availability for plant uptake and is a concern for agriculture management of flooded soils such as rice paddies and other soils that have high water tables.

The reduction of insoluble mineral forms of Mn and Fe to soluble forms is characteristic of low- O_2 conditions.^[10] The half reactions shown for reduction of Mn oxide and goethite ($FeOOH$) are reductive dissolution reactions. Reduced Fe^{2+} and Mn^{2+} are much more soluble than the oxidized species, and therefore mobile in anaerobic soils, forming zones of Fe and Mn depletion and enrichment. As a result, reduction reactions involving Fe and Mn are of particular significance because they result in the formation of observable morphological

features that can be used as indicators of anaerobic conditions.

Reduction of SO_4^{2-} occurs at low E_h conditions in soils and will typically give rise to sulfide [hydrogen sulfide (H_2S), HS^- , or metal sulfide minerals].^[10] In flooded soils, H_2S gas that is released has a rotten egg smell.

At extremely low E_h conditions, the fermentation of organic compounds produces CO_2 and methane (CH_4).^[3,6] Methanogenesis, the formation of CH_4 , can occur by multiple pathways, one consuming hydrogen and CO_2 (shown in the half reaction above).

Other elements not listed above are also susceptible to reduction in soils under low E_h conditions, thereby changing their speciation and solubility. This is particularly important for many trace elements in soils such as selenium [$\text{Se(VI)} \rightarrow \text{Se(IV)} \rightarrow \text{Se(0)} \rightarrow \text{Se(-II)}$], arsenic [$\text{As(V)} \rightarrow \text{As(III)}$], chromium [$\text{Cr(VI)} \rightarrow \text{Cr(III)}$], and mercury [$\text{Hg(II)} \rightarrow \text{Hg(I)} \rightarrow \text{Hg(0)}$]. In anaerobic soils contaminated with these elements, reduction dramatically changes their behavior.

ANAEROBIC CONDITIONS AND SOIL MORPHOLOGY

Anaerobic conditions associated with water tables and saturated conditions are often seasonal occurrences in soils. Field scientists therefore commonly rely on soil morphological features as indicators of seasonal anaerobic conditions and by extension, high water tables. Two features commonly used for this purpose are the color patterns created by Fe and Mn compounds and the accumulation of soil organic matter.^[9]

The microbial reduction of Fe and Mn that occurs under conditions of limited O_2 is a critical step in the formation of redoximorphic features. Redoximorphic features result from alternating periods of reduction and oxidation of Fe and Mn compounds in soils.^[11] As Fe and Mn are reduced, they form soluble and generally colorless compounds. In the more mobile reduced state, Fe and Mn are subjected to redistribution and, in some cases, are removed from soils. As soil water moves to more oxidizing zones or the soil dries out, Fe and Mn are reoxidized and precipitate out of soil solution, giving rise to distinct zones of depletion and accumulation. Under oxidizing conditions, various oxide forms of Fe impart characteristic red, yellow, and brown colors to soils. The zones of Fe and Mn enrichment and depletion are referred to as redox concentrations and redox depletions.^[11] Redox depletions appear as low-chroma, grayish zones from which Fe and Mn have been at least partially removed. Redox concentrations appear as brightly colored soft masses or hard nodules composed of Fe and Mn oxide forms (Fig. 3A and B).^[11]

Localized zones of clay depletion are also indicative of anaerobic conditions and associated processes.^[12] These

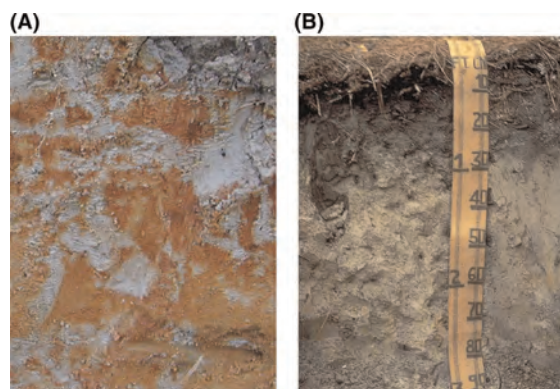


Fig. 3 (A) Close-up view of redox concentrations and redox depletions in a seasonally anaerobic soil. The mottled color pattern consists of gray redox depletions and reddish-orange redox concentrations. Field of view is approximately 10 cm across. (B) Morphological expression of anaerobic conditions in an Aquult from the southeastern United States. The thick, dark surface layer reflects the accumulation of organic matter. Note gray subsoil colors. Scale shows depth in feet and centimeters.

depletions typically have lighter and/or grayer colors than the soil matrix, reflecting the loss of pigmenting agents such as Fe and Mn oxides and clays.^[12] Layer silicate clays are degraded by the alternating oxidation and reduction of Fe compounds, a process known as ferrolysis.^[13] As a result, these zones of clay depletion contain less clay than does the soil matrix.

Under strongly anaerobic conditions, the soil matrix is reduced and appears dull gray and may exhibit a bluish or greenish tint—these are referred to as gray colors. Upon exposure to air, these reduced matrices may develop brighter colors as Fe oxidizes.^[9,12]

Cycling and turnover of organic materials are directly related to O_2 availability in soils. Biodegradation of organic residues is inhibited under anaerobic conditions, and organic matter may accumulate rapidly under such conditions. This results in the formation of a very dark surface horizon in mineral soils (Fig. 3B). Under more prolonged anaerobic conditions, organic materials may accumulate to the extent that soils are classified as organic soils (Histosols). Histosols contain a minimum of 20% organic matter on a weight basis to a depth of at least 40 cm.^[11] These soils typically occupy the wettest landscape positions, such as depressions and low-lying areas with high water tables.

CONCLUSION

Anaerobic processes in soils occur when O_2 concentrations are limiting. These biogeochemical processes are typically associated with soils that are saturated with water for extended periods of time. Anaerobic processes include the biologically mediated reduction of Fe and

Mn and subsequent formation of distinctive morphological features that are used by soil scientists to infer hydrological conditions. Changes in microbial activity also occur with anaerobic conditions and lead to the accumulation of organic matter.

REFERENCES

1. James, B.R.; Brose, D.A. Oxidation-reduction phenomena. In *Handbook of Soil Sciences: Properties and Processes*, 2nd Ed.; Huang, P.M., Li, Y., Sumner, M.E., Eds.; CRC Press: Boca Raton, 2012; 14-1–14-24.
2. Megonigal, J.P.; Patrick, W.H.J.; Faulkner, S.P. Wetland identification in seasonally flooded forest soils—Soil morphology and redox dynamics. *Soil Sci. Soc. Am. J.* **1993**, *57*, 140–149.
3. Evangelou, V.P. *Environmental Soil and Water Chemistry*; John Wiley & Sons: New York, 1998.
4. Bass Becking, L.G.M.; Kaplan, I.R.; Moore, D. Limits of the natural environment in terms of pH and oxidation – reduction potentials. *J. Geol.* **1960**, *68*, 243–284.
5. Sexstone, A.J.; Revsbech, N.P.; Parkin, T.B.; Tiedje, J.M. Direct measurement of oxygen profiles and denitrification rates in soil aggregates. *Soil Sci. Soc. Am. J.* **1984**, *49*, 645–651.
6. Coyne, M. *Soil Microbiology: An Exploratory Approach*; Delmar Publishers: Albany, 1999.
7. Brady, N.C.; Weil, R.B. *Elements of the Nature and Properties of Soils*, 3rd Ed.; Prentice Hall: Upper Saddle River, 2010.
8. Duchaufour, P. *Handbook of Pedology*; A.A. Balkema: Rotterdam, 1998.
9. Hurt, G.W.; Whited, P.M.; Pringle, R.F., Eds. *Field Indicators of Hydric Soils in the United States, Version 4.0*; USDA-NRCS: Fort Worth, 1998.
10. Sposito, G. *The Chemistry of Soils*, 2nd Ed.; Oxford University Press: New York, 2008.
11. Soil Survey Staff. *Keys to Soil Taxonomy*, 12th Ed.; USDA-NRCS: Washington, 2014.
12. Schoeneberger, P.J.; Wysocki, D.A.; Benham, E.C. Soil Survey Staff. In *Field Book for Describing and Sampling Soils, Version 3.0*; USDA-NRCS: Lincoln, 2012.
13. Schaetzl, R.J.; Anderson, S. *Soils. Genesis and Geomorphology*; Cambridge University Press: New York, 2005.

Andisols

Jon Chorover

*Department of Soil, Water and Environmental Sciences, University of Arizona,
Tucson, Arizona, U.S.A.*

Abstract

Andisols are deep soils of low bulk density that are most often derived from volcanic parent materials. They are classified on the basis of unique mineralogical, chemical, and physical properties. This soil order, which is most prevalent along the tectonically active Pacific Ring of Fire, covers approximately 120 million hectares or 1% of the earth's surface. Andisols are capable of supporting agricultural production and human populations that are large relative to the spatial extent of the soils. The prefix, "An," refers to the dark color of the surface horizon(s) that results from high concentrations of humified organic matter stabilized by chemical interaction with poorly crystalline secondary minerals. The morphology of the soils typically includes multiple sequences of A and Bw horizons that are buried successively because of intermittent deposition events.

CLASSIFICATION OF ANDISOLS

The classification of Andisols^[1] proposed by Smith^[2] and was refined through the efforts of the International Committee on the Classification of Andisols. Parent material must be weathered to fall within a range of characteristics pertaining to bulk density, particle size, phosphorus adsorption capacity, and within a range of mass concentrations of organic carbon (C), volcanic glass, and oxalate extractable iron (Fe) and aluminum (Al).^[1] Specifically, the soil material must contain no more than 250 g/kg organic C and either oxalate extractable $\text{Al} + 0.5 \text{ Fe} = 20 \text{ g/kg}$, bulk density (at 33 kPa) $= 0.9 \text{ Mg/m}^3$, and phosphate retention $= 85\%$, or phosphate retention $= 25\%$, particles between 0.02 and 2 mm in size $= 300 \text{ g/kg}$, oxalate extractable $\text{Al} + 0.5\text{Fe} = 4 \text{ g/kg}$, and volcanic glass in the 0.02–2 mm fraction $= 362 - 15.6 \times (\text{oxalate extractable } \text{Al} + 0.5\text{Fe})$, where all units are grams per kilogram. Soils classified as Andisols are, therefore, dominantly those developing in volcanic ash, pumice, cinders, and other volcanic ejecta. Suborders are based on climate regimes, whereas great groups and subgroups are based on soil physical–chemical properties such as clay and organic matter content.

PEDOGENIC FACTORS

Most Andisols develop from chemical weathering of volcanic ejecta in combination with organic matter humification.^[3] Common parent materials range, in physical form, from fine ash to viscous lava flows, and in mineralogical composition from andesitic and rhyolitic to basaltic rock types. Basalt weathers more rapidly than andesitic rock

and, because of the availability of reactive surface, finely divided ash weathers more rapidly than lava. Sufficient weathering of parent material is required to produce the poorly crystalline Al, Fe, and silicon (Si) compounds in the quantities needed to classify a soil as an Andisol. Thus, these soils commonly occur in humid regions.^[4] Andisols are formed from parent materials other than volcanic ejecta, provided that sufficiently high levels of reactive Al, Fe, and organic matter are present, but volcanic parent materials are most common. The high specific surface area and reactivity of poorly crystalline minerals result in greater organic matter retention than occurs in most other soil orders except Histosols. The sequence of soil horizons is typically A–Bw–C (Fig. 1). The surface A horizon often contains very high concentrations of organic matter (typically greater than 100 g/kg) that is stabilized by complexation with Al and Fe. The B horizon is normally dominated by poorly crystalline aluminosilicates and hydrous oxides (allophane, imogolite, and ferrihydrite). Periodic eruptions in volcanically active areas can produce a series of buried A–Bw sequences, which is one cause of deep humus penetration in Andisols.^[5] This factor also contributes to the significant depth of Andisol profiles.

PHYSICAL–CHEMICAL PROPERTIES OF ANDISOLS

Mineralogy

The unique properties of Andisols are strongly linked to the chemical properties of constituent solid phases.^[4,6] Primary silicate minerals in andesitic and rhyolitic flows

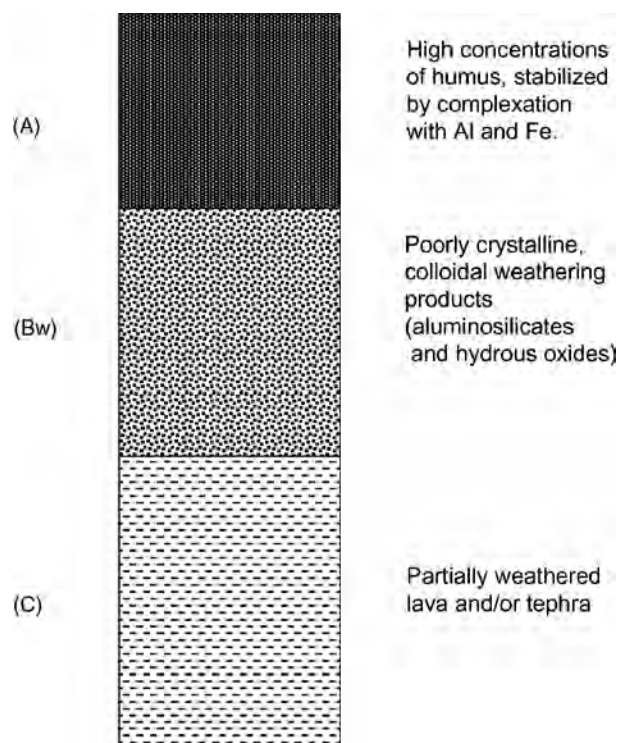


Fig. 1 Schematic representation of a typical Andisol profile.

include quartz, hornblende, and biotite, whereas olivines, pyroxenes, and plagioclase are more common in basalts. Rapid weathering of volcanic ejecta (containing aluminosilicates, volcanic glass, and smaller amounts of ferromagnesian minerals) results in the accumulation of soluble silica, Al, and Fe to high concentrations in the soil solution of young Andisols. In the near surface (e.g., A horizon), Fe and Al are often immobilized into humic complexes, whereas Si may polymerize to form opaline silica. At greater depth (e.g., Bw horizon), or if humus concentrations are low, hydrolysis and polymerization of Al, Fe, and Si result in precipitation of allophane ($x\text{SiO}_2 \cdot \text{Al}_2\text{O}_3 \cdot y\text{H}_2\text{O}$, where $x = 0.8\text{--}2$ and $y = 2.5$), ferrihydrite ($5\text{Fe}_2\text{O}_3 \cdot 9\text{H}_2\text{O}$), opaline silica ($\text{SiO}_2 \cdot n\text{H}_2\text{O}$), amorphous aluminum hydroxide [$\text{Al}(\text{OH})_3$], and/or imogolite ($\text{SiO}_2 \cdot \text{Al}_2\text{O}_3 \cdot 2.5\text{H}_2\text{O}$), depending upon solution composition. Allophane and ferrihydrite, which do not exhibit the long-range crystal structure necessary to yield well-defined X-ray diffraction patterns, are commonly termed short-range order (SRO) solids. These minerals and imogolite are "metastable" thermodynamically. That is, over the long term, Ostwald ripening of soil solids transforms SRO phases into more crystalline forms such as halloysite, goethite, and gibbsite. As the accumulation of more crystalline phases causes soil characteristics to diverge from those that are diagnostic for the Andisol order, Andisols represent an intermediate stage of pedogenesis, the lifetime of which depends on interacting state factors that control mineral transformation rates.

The important properties of Andisols are generally ascribed to the preponderance of allophane and X-ray amorphous compounds of Al, Si, and humus,^[2] but high concentrations of Al and Fe humus complexes can confer to non-allophanic soils, the key diagnostic properties that are common in allophanic Andisols.^[7] It is, therefore, recognized that important SRO minerals of Andisols include not only allophane but also imogolite, ferrihydrite, poorly crystalline $\text{Al}(\text{OH})_3$, and Al/Fe-humus complexes. The SRO minerals in Andisols exhibit very high specific surface areas (ca. $10^5\text{--}10^6 \text{ m}^2/\text{kg}$) and reactive site densities ($5\text{--}20 \mu\text{mol}/\text{m}^2$). Unit particles of allophane are roughly spherical and 3.5–5.5 nm in diameter, whereas imogolite appears as smooth and curved threads of 10–30 nm diameter, and ferrihydrite is composed of spherical particles of 3–7 nm in diameter.^[8,9]

Chemistry

Chemical properties of Andisols are dominated by the main pedogenic weathering products of volcanic debris in terrestrial ecosystems: X-ray amorphous SRO compounds of Al, Si, and Fe, and their complexes with organic matter.^[10] These constituents are the target species measured during acid ammonium oxalate treatment of the soil, and their prevalence is diagnostic for an Andisol. Humus concentrations correlate with SRO minerals and extend to depth accordingly. Allophane, imogolite, and ferrihydrite contain a prevalence of surface hydroxyl groups that: 1) retain and dissociate protons (H^+) in response to changes in soil solution pH and 2) can form strong complexes with humic substances and oxyanions such as phosphate.^[4,9] In particular, the strong binding of phosphate can lead to plant P deficiencies, when these soils are used for crop production. These surface reactions confer significant variable charge on the soils. With increasing pH, cation adsorption increases and anion adsorption decreases. The pH value where moles of adsorbed cation and anion charge are equivalent is termed the point of zero net charge (PZNC). The total surface charge at any point is a function of solution chemistry, soil mineralogy, and organic matter chemistry. In general, increasing amounts of Fe and Al oxides will increase the PZNC, whereas increasing amounts of SRO silica and organic matter will decrease it. Andisols contain relatively low quantities of 2:1 layer-type clay minerals and, therefore, permanent (structural) charge represents a small fraction of the total surface charge.^[11]

As SRO solids are more soluble than well-crystallized phases, they can support relatively high equilibrium concentrations of dissolved Al. The prevalence of Al in Andisols can maintain acidic soil pH (4.5–6.0) via Al hydrolysis reactions and diminish base saturation via displacement of non-hydrolyzing cations Ca^{2+} , Mg^{2+} , K^+ , and Na^+ from cation exchange sites. In addition, complexation with humic substances of monomeric, polymeric, and colloidal

forms of Al and Fe serves to coagulate and stabilize organic matter in Andisols, leading to long turnover times of organic C.^[12]

Physical Properties

Allophane, imogolite, ferrihydrite, and organic matter are responsible for the unique properties of soils derived from volcanic parent materials. The charge characteristics, size, and shape of these constituents are important determinants of soil physical properties. Andisols containing large quantities of SRO constituents have specific surface areas as high as $6 \times 10^5 \text{ m}^2/\text{kg}$. The density of allophane particles—ca. 2.7 Mg/m^3 —is slightly higher than more crystalline aluminosilicates such as kaolinite and montmorillonite. Therefore, the much lower bulk densities encountered in Andisols ($<0.85 \text{ Mg/m}^3$), where SRO minerals are dominant, are attributed to the high porosity of microaggregates formed, when unit particles interact with each other and with organic matter. Aggregation of SRO constituents, and their interaction with humic substances, gives rise to large void volumes (25–45%) in the micropore and mesopore size ranges. In conjunction with the significant quantity of hydration water associated with primary particles, this gives Andisols the capacity to retain as much as 1.8 kg water per kilogram soil solids and to support lush vegetation even at low rainfall. Hydraulic conductivity values are high because of low bulk density values and granular structure. Hydraulic properties can be altered by changing chemical conditions to induce dispersion of variable-charge soil colloids. For example, saturated hydraulic conductivity values can be reduced dramatically by displacing soil pH several units from the point of zero charge. Because of the hydrated nature of SRO phases and microaggregates, soil drying to high tensions results in irreversible changes to many of these soil physical properties. Physical properties of Andisols have been reviewed by Maeda and coworkers^[13,14] and Wada.^[10]

LAND USE AND MANAGEMENT

Andisols can provide exceptional media for plant growth in comparison to other mineral soils.^[4] They tend to have a thick solum with unrestricted rooting zones, high organic matter contents, and abundant available water. The supply of lithogenic nutrients is provided through rapid weathering of residual volcanic ash and lava. In regions of high rainfall, however, loss of cationic nutrients may be facilitated by intense leaching. Under such conditions, and as a result of reactive SRO mineral surfaces, the sustainable management of Andisols for agriculture and forestry may also be limited by phosphorus deficiency and low cation exchange capacity, among

other constraints. Nitrogen (N) availability may also limit plant production to the extent that N mineralization is limited by chemical stabilization of humus into metal–humus or mineral–humus complexes. The low bulk density, friable consistence, and granular structure, particularly in surface horizons, facilitate tillage operations such as seed-bed preparation and plowing. High porosity and plant available water favor seedling emergence, root development, and ramification. These soils resist compaction, even when cultivated continuously, and tend to regain physical properties after wetting and drying cycles once compacted by machinery. High hydraulic conductivities and water retention capacities promote infiltration and diminish erosive overland flow. Binding of SRO minerals and humic substances gives rise to strong soil aggregates that resist mechanical disruption by raindrops that could otherwise lead to particle detachment and erosion.

CONCLUSION

Weathering of volcanic ejecta under humid conditions results in supersaturation of soil solutions with respect to metastable, poorly crystalline aluminosilicates, $\text{Al}(\text{OH})_3$, and Fe hydroxides. Accumulation of these high specific surface area solids gives rise to the unique chemical and physical properties characteristic of the Andisol order. Organic C accumulations result from complexation with dissolved Al and Fe and from adsorption to the high surface area secondary minerals. Given the prevalence of weakly acidic functional groups associated with hydrous oxides and organic matter, Andisols exhibit variable surface charge properties that are highly dependent on solution pH and ionic composition. Andisols are also characterized by low bulk density and high water retention because of the extensive porosity of particle micro- and macroaggregates.

REFERENCES

1. Soil Survey Staff. *Keys to Soil Taxonomy*, 8th Ed.; USDA–NRCS: Washington, 1998.
2. Smith, G.D. *A Preliminary Proposal for the Reclassification of Andepts and Some Andic Subgroups (the Andisol Proposal 1978)*; New Zealand Soil Bureau Record; DSIR: Lower Hutt, 1978; 96 pp.
3. Van Breeman, N.; Buurman, P. *Soil Formation*; Kluwer: Dordrecht, 1998.
4. Shoji, S.; Nanzyo, M.; Dahlgren, R.A. *Volcanic Ash Soils. Genesis, Properties and Utilization*; Elsevier: Amsterdam, 1993.
5. Ping, C.L.; Shoji, S.; Ito, T. Properties and classification of three volcanic ash-derived pedons from Aleutian Islands and Alaska Peninsula, Alaska. *Soil Sci. Soc. Am. J.* **1988**, *52*, 455–462.

6. Wada, K. Mineralogical characteristics of Andisols. In *Soils with Variable Charge*; Theng, B.K.G., Ed.; New Zealand Society of Soil Science: Lower Hutt, 1980; 87–108.
7. Shoji, S.; Ono, T. Physical and chemical properties and clay mineralogy of Andisols from Kitakami. *Jpn. Soil Sci.* **1978**, *126*, 297–312.
8. Schwertmann, U.; Taylor, R.M. Iron oxides. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; Soil Science Society of America: Madison, 1989; 379–438.
9. Harsh, J.B.; Chorover, J.; Nizeyimana, E. Allophane and imogolite in soils and their effects on environmental processes. In *Soil Mineralogy Environmental Applications*; Dixon, J.B., Schulze, D., Eds.; Soil Science Society of America: Madison, 2002; 291–322.
10. Wada, K. The distinctive properties of Andisols. *Adv. Soil Sci.* **1985**, *2*, 173–229.
11. Chorover, J.; Amistadi, M.K.; Chadwick, O.A. Surface charge evolution of mineral-organic complexes during pedogenesis in Hawaiian basalt. *Geochim. Cosmochim. Acta* **2004**, *68*, 4859–4876.
12. Torn, M.S.; Trumbore, S.E.; Chadwick, O.A.; Vitousek, P.M.; Hendricks, D.M. Mineral control of soil organic carbon storage and turnover. *Nature* **1997**, *389*, 170–173.
13. Maeda, T.; Takenaka, H.; Warkentin, B.P. Physical properties of Allophane soils. *Adv. Agron.* **1977**, *29*, 229–269.
14. Warkentin, B.P.; Maeda, T. Physical and mechanical characteristics of Andisols. In *Soil with Variable Charge*; Theng, B.K.G., Ed.; New Zealand Society of Soil Science: Lower Hutt, 1980; 281–302.

Andisols: Iceland

Olafur Arnalds

Agricultural Research Institute, Keldnaholt, Iceland

Abstract

Icelandic soils and their environment have some attributes seldom found in volcanic areas, such as steady eolian flux of basaltic tephra materials, intense cryoturbation, and extensive deserts in a humid environment, dominated by vitric surface materials. Soils of Icelandic wetlands are predominantly Andisols, but Histosols are of limited extent despite the subarctic climate. The objective of this entry is to give a brief overview of Icelandic soils and to describe the major factors shaping their development.

INTRODUCTION

The surface of Iceland is covered by volcanic soils, representing the largest area of such soils in Europe and perhaps >5% of Andisols in the world.^[1] Worldwide distribution of Andisols is generally related to volcanic parent materials, but they have also formed in older volcanic rocks reworked by glacial activity such as in central France.^[2] Andic soil properties, which define Andisols, result from the presence of short-range-order clay minerals such as allophane, ferrihydrite, and imogolite and the presence of metal–humus complexes. Vitric materials (tephra) are also used for the definition of Andosols.

PHYSIOGRAPHY

Iceland is a volcanic island of about 100,000 km² on the Mid-Atlantic Ridge. Temperature regime is mostly cryic as a result of subarctic oceanic and moist climate, with mild winters and cool summers. Many freeze-thaw cycles occur each winter, causing intense cryoturbation. Volcanic eruptions occur on average every 4–5 years and produce both solid lavas and volcanic tephra. Since glaciation during the Quaternary period, the surface of Iceland has been extensively mantled by eolian and tephra materials in which Andisols have developed. Lavas and glaciofluvial sediments also cover large areas. Most glacial sediments are mainly composed of volcanic glass.

Erosion has removed large proportion of the soils that formed in the eolian and tephra sediments, which has created large desert areas.^[3] For a more detailed description of the Icelandic physiography, please refer to Arnalds^[1] and the bibliography at <http://www.rala.is/desert>.

CLASSIFICATION AND SOIL CHARACTERISTICS

A new classification scheme for Icelandic soils^[1] is related to the Food and Agriculture Organization World Reference

Base (WRB) for soils,^[4] and it has been used for making a 1:500,000 soil map of Iceland (see www.rala.is/desert). The major soil types according to the system are Vitrisols, Histosols, Histic Andosols, Gleyic Andosols (GA), Brown Andosols (BA), and Leptosols (Table 1). The Andosol soil types have a single name in Icelandic, although two are used in English for clarity. Soil types under the Icelandic scheme are branded in italics to prevent confusion with WRB or Soil Taxonomy. The method of separating the soils acknowledges the dominant influence of three factors in shaping soil properties in Iceland: 1) the rate of eolian and tephra deposition; 2) the drainage; and 3) the presence of deserts with coarse-grained soils with low organic content.

The first two factors, eolian deposition and drainage, influence the amount and type of colloid materials (allophane, ferrihydrite, and metal–humus complexes) and the total organic content of the Andosols and Histosols. This relationship is shown in Fig. 1. Rapid eolian deposition tends to favor the formation of BA and reduces the organic content of the soils. Decreasing drainage (or permanent wetness) and slower eolian input lead to higher organic content with Histosols in wet positions that receive little eolian sedimentation.

When considering soil properties such as water retention and bulk density, the dominance of organic content becomes evident [Fig. 2; symbols grouped according to carbon (C) content]. Water retention at 15-bar suction is very high and reaches 300% in some organic horizon, but 15-bar water content is commonly 50–100% in the andic soil horizons (1.2–20% C). Vitric horizons with low organic content (<1.2% C) have relatively much lower 15-bar water content. The influence of organic matter (that includes metal–humus complexes) masks the influence of clays on the 15-bar water-holding capacity. Bulk density shows similar trend with the lowest bulk density of about 0.2 g cm⁻³ in organic horizons (>20% C). However, bulk density remains low in inorganic horizons with considerable clay content (allophane and ferrihydrite). Other soil properties are also typical of Andisols, such as high

Table 1 Extent of Icelandic soil types and complexes of soil types.

Soil type	Symbol	Extent (km ²)	Percentage of soils (%)
Histosols	H	1,090	1.2
Histic Andosols	HA	4,920	5.5
GA	WA	2,390	2.6
BA	BA	13,360	14.8
Leptosols	L	7,310	8.1
Cambic Vitrisols	MV	17,640	19.5
Arenic Vitrisols	SV	4,550	5.0
Cryosols and GA	C-WA	140	0.2
BA and GA	BA-WA	28,080	31.1
Cambic Vitrisols and Arenic Vitrisols	MV-SV	6,000	6.6
Arenic Vitrisols and Leptosols	SV-L	4,790	5.3
<i>Total</i>		<i>90,270</i>	<i>100</i>

Source: From Arnalds.^[1]

phosphate retention and cation exchange capacity, and most Icelandic soils have pH measured in NaF greater than 11.^[1]

Wetland Soils

The Andisols of Iceland include extensive wetland soils (>22,000 km²) that are characterized by both clays (allophane and ferrihydrite) and metal–humus complexes. Because of the dominance of andic materials when present

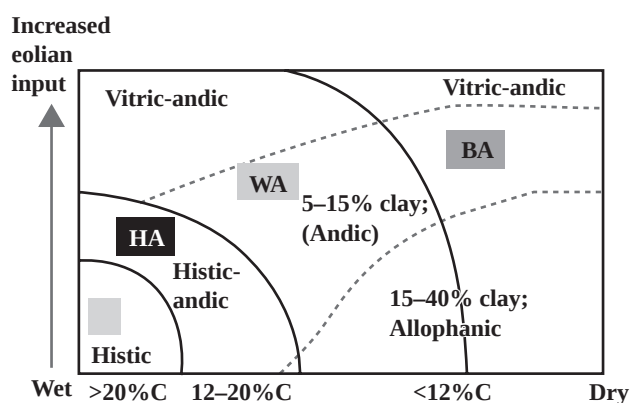


Fig. 1 Schematic figure showing the influence of drainage and eolian flux on properties and classification of Icelandic soils. BA are found in freely drained positions. They are vitric near active eolian sources, but have relatively high clay content (allophane, ferrihydrite, and imogolite) far from such sources. GA are found in poorly drained positions with high flux of eolian materials, but there is a gradient from poorly (with vitric GA) and andic GA to allophanic GA in somewhat poorly drained sites with little flux of eolian materials. As soils become less influenced by eolian input, soils become more Histic, with Histosols as the end member at wet sites far from eolian sources (lower left corner).

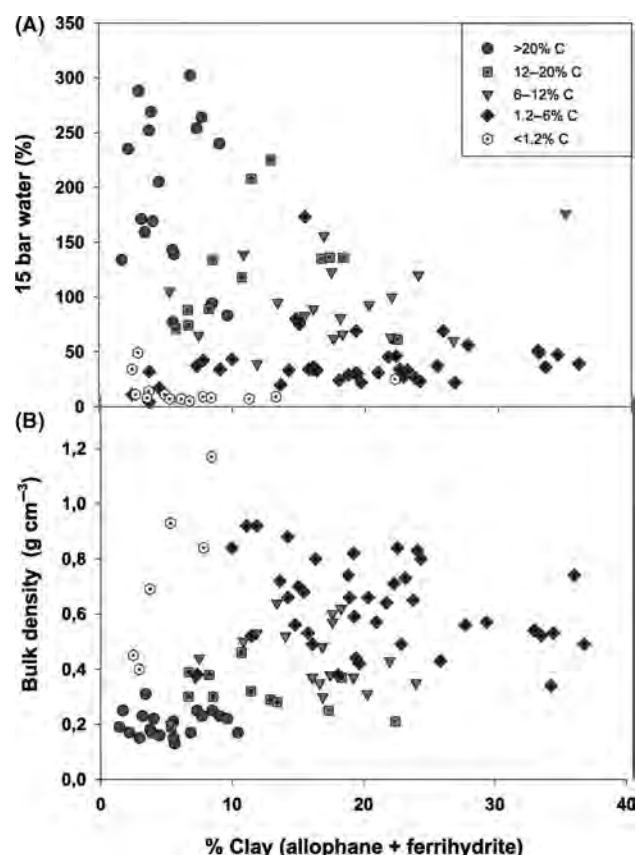


Fig. 2 (A) Water retention of Icelandic Andisols as a function of clay content. Horizons are grouped by organic C%. Organic soils have highest water retention, but some of the andic horizons with both high organic C content (12–20% C) and allophane (>2% Si₀) also have >100% 15-bar water content. Vitrisols with <1% Si₀ have the lowest water retention. (B) Bulk density of Icelandic Andisols as a function of clay content, horizons grouped by organic C%. Organic horizons (>20% C) have bulk density of about 0.2 g cm⁻³. Soils with low C% (vitric horizons) have low clay content and often relatively high bulk density (>0.8 g cm⁻³), but the soils have, in general, low bulk density characteristic of Andisols (predominately <0.8 g cm⁻³).

in soils, Andosols/Andisols can have up to 20% C if other criteria for andic properties are met according to the WRB,^[4] but 25% C according to Soil Taxonomy.^[5] The 20% limit is used in Iceland. Histosols, with >20% C, are found only far from active eolian sources and mostly on Tertiary bedrock, which has slow water permeability. Their extent (1.2%) is surprisingly low considering the subarctic location of Iceland and the great extent of wetlands. Histic Andosols (4920 km²) are wetland soils with high organic content (12–20% C), but in addition to organic matter, the colloid fraction is dominated by metal–humus complexes, allophanes, and ferrihydrite. Histic Andosols receive sufficient eolian input to lower the organic content below the 20% C diagnostic limit.

The GAs (estimated as 16,000 km²) are soils of wetlands with relatively low organic content in surface horizons

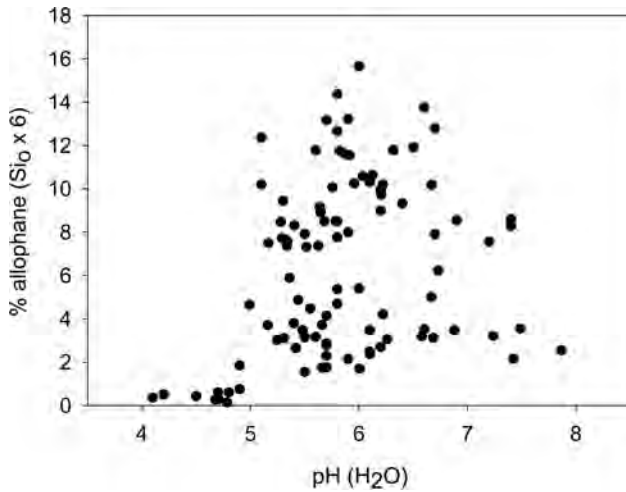


Fig. 3 The relationship between pH (measured in water) and allophane content in soils from 34 sites from all parts of Iceland. Surface horizons only, sampled from 0–5, 5–10, and 10–15 cm depth intervals. Soils with pH < 4.9 do not contain appreciable amounts of allophane.

(<12% C). Intense eolian input reduces the organic content and results in the formation of Andisols with properties dominated by allophane, ferrihydrite, and metal–humus complexes. GA are also found on young parent materials (fluvial and eolian) in poorly drained positions. The eolian input, with easily weathered basaltic tephra, recharges the system with bases, resulting in relatively moderate pH of 6–7, which favors the formation of allophane. Soil reaction is lower in the Histic Andosols and Histosols, where the formation of metal–humus complexes dominates. Our data indicate that the formation of allophane is clearly hindered when pH is lower than 4.9 (Fig. 3), as has been shown for Andisols elsewhere.^[6]

Freely Drained Soils

The BA are the typical freely drained Andisols of Iceland, containing appreciable amounts of allophane and ferrihydrite (total of 10–40%).^[1,7] These soils often contain dark basaltic and light-colored rhyolitic tephra that have been dated.

The soils of Icelandic deserts are predominately Andisols according to Soil Taxonomy.^[8] However, the desert soils are classified either as Andosols or as Leptosols according to the WRB, which requires a minimum soil depth for Andosols, resulting in Leptosol classification of shallow soils. These soils have different characteristics and functions from the Andosols and Histosols (Icelandic classification scheme). The great extent (> 30,000 km²) and uniqueness warrant their separation at the highest level of the Icelandic classification scheme as Vitrisols. The Vitrisols are coarse grained, with low contents of allophane and organic matter. Nitrogen levels are low,

making the reestablishment of vegetation difficult, but low water-holding capacity and intense frost action also contribute to their infertility. Future work is likely to result in further division of the Vitrisols into 2–3 categories comparable to those used for Andosols according to the Icelandic scheme, as is performed in Table 1.

Leptosols (9700 km²) are mainly very thin soils (1–10 mm) of (Holocene) lava fields that lack the eolian mantle in which Andosols form. They also include extensive scree slopes in mountainous areas.

VITRISOL CLASSIFICATION PERSPECTIVES

Central to classification of Andisols/Andosols in Soil Taxonomy^[5] and the WRB^[4] is the determination of their colloidal constituents by acid oxalate extraction of aluminum (Al), iron (Fe), and silica (Si; Al_o, Fe_o, and Si_o). Andisols have >2% Al_o + (1/2)Fe_o. The presence of vitric materials (various types of tephra) is also recognized by lowering this limit to as low as 0.4%, depending on the abundance of vitric materials. Most of the Icelandic Vitrisols meet the > 0.4% Al_o + (1/2)Fe_o criterion for Andisols, but this limit seems arbitrary in Iceland, and it can be doubted that it does provide meaningful separation of soils with abundance of vitric materials. On a world basis, vitric soils are poorly accounted for and recognized as weakly developed soils or parent material (tephra), with [Al_o + (1/2)Fe_o] < 0.4%. The uniqueness of these parent materials is recognized in some other classification systems such as for New Zealand with “Pumice Soils.”^[9]

ANDIC SOILS AND CRYOTURBATION

Cryoturbation occurs with great intensity in Iceland.^[1,10] The evidence is seen by cryoturbation in the soils and as various surface features such as “thufur” (hummocks) and a variety of active solifluction features. Cryoturbation in Iceland is enhanced by several factors. Water is pumped from shallow water table in wetlands. Freeze-thaw cycles are numerous, and the soil temperature stays near 0°C for extensive periods, which enhances stationary freezing front. The physical properties of Andosols are important as well. Both saturated and unsaturated hydraulic conductivity of the soils are rapid, which enhances water transfer to the freezing front. The great water-holding capacity (often >40% at 15-bar suction in freely drained soils) also contributes water that freezes in the soil. An additional factor is the thixotropic nature of many soils, as this allows easy deformation of the soil by the ice formation.

SOIL EROSION

The climate of Iceland is cold and subjected to periodic cold spells that exert stress on Icelandic ecosystems, as

do ashfall events during volcanic eruptions. The vegetation of Iceland evolved in the absence of grazing animals. These factors render Icelandic ecosystems particularly sensitive to disturbance and reduced resilience caused by grazing and other land use. Soil erosion in Iceland has remained intense for the past 1200 years since the settlement by Vikings. A unique feature of soil erosion in Iceland is that the full depth of the soil mantle that has formed in eolian and tephra sediments is truncated, leaving barren surfaces (deserts) behind. A survey of erosion in Iceland has been published^[11] showing the great extent and severity of soil erosion in Iceland.

The Icelandic soils are extremely vulnerable to erosion, by both wind and water for many reasons. The formation of stable silt size clusters enhances saltation movement of soil particles, making the soils susceptible to wind erosion. The lack of cohesion while wet and the thixotropic nature of the soils intensify water erosion and landslides. These andic properties that have reduced resistance against erosion are important contributing factors to the severe soil erosion in Iceland.

CONCLUSION

A classification scheme specific to Iceland has been developed, and it has been used to produce a new 1:500,000 soil map.^[11] Icelandic soils are primarily Andosols and Vitrisols according to the Icelandic scheme, which are classified as Andisols according to Soil Taxonomy. The Icelandic soil environment is characterized by cold humid climate and a steady flux of vitric materials to the surface which, together with drainage, are major factors dominating soil formation in Iceland. The extensive Icelandic desert environments are unique considering the humid climate of Iceland. The soils of the deserts are characterized by basaltic vitric materials and are termed Vitrisols.

The andic soil characteristics of the Andisols, such as high water-holding capacity, rapid hydraulic conductivity, thixotropy, and stable silt-sized aggregates, have important implications for intensive cryoturbation and soil erosion in Iceland.

REFERENCES

1. Arnalds, O. Volcanic soils of Iceland. In *Volcanic Soil Resources. Occurrence, Development, and Properties*; Arnalds, O., Stahr, K., Eds.; Elsevier: Amsterdam, 2004.
2. Quantin, P. Volcanic soils of France. In *Volcanic Soil Resources. Occurrence, Development, and Properties*; Arnalds, O., Stahr, K., Eds.; Elsevier: Amsterdam, Catena, **2004**, 56 (1), 1–2.
3. Arnalds, O. Sandy deserts of Iceland. *J. Arid Environ.* **2001**, 47 (3), 359–371.
4. FAO. *World Reference Base for Soil Resources*; World Soil Resources Reports No. 84; FAO: Rome, 1998; Vol. 88.
5. Soil Survey Staff. *Keys to Soil Taxonomy*, 8th Ed.; USDA-NRCS: Washington, 1998.
6. Parfitt, R.L.; Kimble, J.M. Conditions for formation of allophane in soils. *Soil Sci. Soc. Am. J.* **1989**, 53 (3), 971–977.
7. Arnalds, O.; Hallmark, C.T.; Wilding, L.P. Andisols from four different regions of Iceland. *Soil Sci. Soc. Am. J.* **1995**, 58 (1), 161–169.
8. Arnalds, O.; Kimble, J. Andisols of deserts in Iceland. *Soil Sci. Soc. Am. J.* **2001**, 65 (6), 1778–1786.
9. Hewitt, A.E. *New Zealand Soil Classification*, 2nd Ed.; Landcare Research Science Series; Manaaki Whenua Press: Lincoln, 1998; Vol. 1.
10. Van Vliet-Lanoe, B.; Bourgeois, O.; Dauteuil, O. Thufur formation in northern Iceland and its relation to Holocene climate change. *Permafrost. Periglacial. Proc.* **1998**, 9 (4), 347–365.
11. Arnalds, O.; Thorarinsdottir, E.F.; Metusalemsson, S.; Jonsson, A.; Gretarsson, E.; Arnason, A. *Soil Erosion in Iceland*; Soil Conservation Service and Agricultural Research Institute: Reykjavik, 2001.

Animals: Ecosystem Functioning

Alan J. Franzluebbers

U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
Watkinsville, Georgia, U.S.A.

Abstract

Soil animals are represented by a diverse array of creatures living in or on soil for at least a part of their life cycle. Many animals have influences on soil properties but should not be considered soil dwellers since only a minor portion of their life cycle is spent in the soil. Soil animals can be classified according to body size, where they inhabit the soil, and feeding activity.

INTRODUCTION

Soil animals (i.e., fauna) are represented by a diverse array of creatures living in or on soil for at least a part of their life cycle. Many animals have influences on soil properties but should not be considered soil dwellers since only a minor portion of their life cycle is spent in the soil (Fig. 1).

Based on body size, soil animals can be divided into the following three categories:

1. Microfauna (< 200 μm length and <100 μm width) including protozoa, rotifers, and nematodes
2. Mesofauna (0.2–10 mm length and 0.1–2 mm width) including tardigrades, collembola, and mites
3. Macrofauna (>10 mm length and >2 mm width) including millipedes, spiders, ants, beetles, and earthworms

Soil animals can also be classified according to where they inhabit the soil. The aquatic fauna (e.g., protozoa, rotifers, tardigrades, and some nematodes) live primarily in the water-filled pore spaces and surface water films covering soil particles. Earthworms are divided into species that occupy the surface litter of soil (epigeic), that are found in the upper soil layers (endogeic), or that burrow deep into the soil profile (anecic).

A further classification of five groups of soil animals is based on feeding activity, which can be useful in distinguishing how different groups affect soil ecosystem functions:

1. Carnivores feed on other animals. This group can be subdivided into the following two categories: 1) predators (e.g., centipedes, spiders, ground beetles, scorpions, ants, and some nematodes), who normally engulf and digest their smaller prey and 2) parasites (e.g., some flies, wasps, and nematodes), who feed on or within their typically larger host organism.
2. Phytophages feed on living plant materials, including those that feed on aboveground vegetation (e.g., snails

and butterfly larvae), roots (e.g., some nematodes, fly larvae, beetle larvae, rootworms, and cicadas), and woody materials (e.g., some termites and beetle larvae).

3. Saprophages feed on dead and decaying organic material and include many of the earthworms, enchytraeids, millipedes, dung beetles, and collembola (or spring-tails). They are often referred to as scavengers, debris feeders, or detritivores.
4. Microphytic feeders consume bacteria, fungi, algae, and lichens. Typical microphytic feeders include mites, collembola, ants, termites, nematodes, and protozoa.
5. Miscellaneous feeders are not restrictive in their diet and consume a range of the previously mentioned sources of food. This group includes certain species of nematodes, mites, collembola, and fly larvae.

The arrangement of these feeding groups can be visualized as a soil food web with multiple trophic levels, beginning with the autotrophic flora (Fig. 2). Trophic levels describe the order in the food chain. The first trophic level is composed of photosynthetic organisms, including plants, algae, and cyanobacteria, which fix carbon dioxide (CO_2) from the atmosphere into organic compounds. Organisms that consume the photosynthesizers are in the second trophic level, which includes bacteria, actinomycetes, fungi, root-feeding nematodes and insects, and plant pathogens and parasites. The third trophic level feeds on the second trophic level, including many of the dominant soil animals, including bacterial- and fungal-feeding arthropods, nematodes, and protozoa. The soil food web can be continued to include various vertebrates, including amphibians, reptiles, and mammals.

SPATIAL DISTRIBUTION OF SOIL ANIMALS

Soil animals are not uniformly distributed in soil. Unlike the soil microflora, which could be considered ubiquitous,

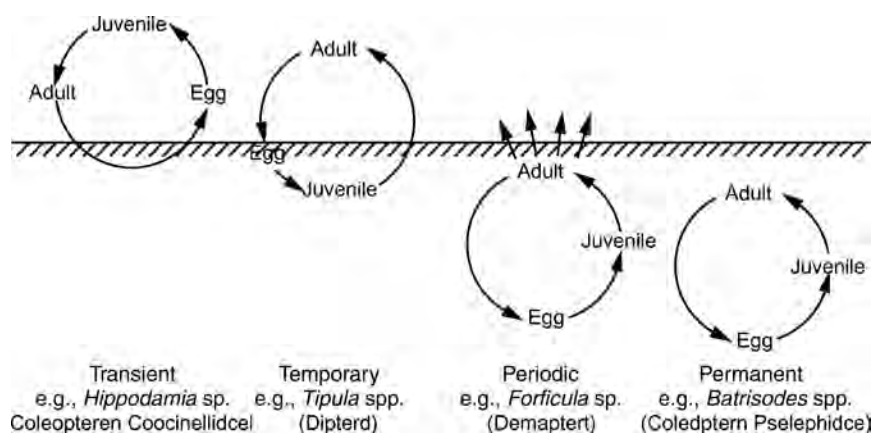


Fig. 1 Categories of soil animals defined according to the degree of presence in soil, as illustrated by some insect groups.

Source: From Wallwork.^[1]

the proliferation of soil animal communities is more sensitive to environmental disturbances and ecological interactions. Gross climatic differences afford opportunities for unique assemblages of organisms. Even within a specific climatic region, large differences occur in the community of organisms present based on the type of vegetation, soil, availability of water, land use, and the presence of xenobiotics. Within the confines of a seemingly uniform pedon, “hot spots” of soil organism activity can be isolated based on the localized availability of resources and environmental conditions (Fig. 3).

INFLUENCE OF SOIL ANIMALS ON SOIL FUNCTIONS

Decomposition and Nutrient Cycling

Soil animals work directly and indirectly with the soil microflora (i.e., bacteria, actinomycetes, fungi, and algae) to decompose organic matter and mineralize nutrients.^[3] The primary consumers of organic materials are the soil microflora. Soil animals, like many of the microflora, are heterotrophs and therefore consume organic materials to

gain energy for growth and activity. Soil animals make important contributions to decomposition by:

1. Shredding organic materials, thereby exposing a greater surface area for enhancing the activities of other organisms, especially microorganisms;
2. Consuming resistant plant materials that would decompose slowly otherwise, such as wood, roots, and dung, and transforming these materials into more decomposable constituents;
3. Dispersing soil microorganisms (i.e., inoculation) within the soil profile by transporting them on their bodies and through their intestinal tracts;
4. Creating burrows in soil to increase aeration, which stimulates microbial activity;
5. Transporting organic materials from the soil surface to deeper in the soil profile, thereby improving environmental conditions for decomposition and increasing biological interactions deeper in the soil profile;
6. Consuming bacteria and fungi, thereby releasing nutrients and stimulating the regeneration of microbial populations; and
7. Providing unique food sources themselves for consumption by other soil fauna and microflora.

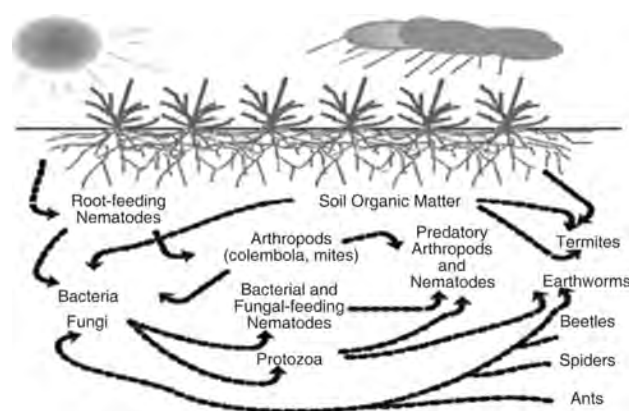


Fig. 2 Generalized diagram of a soil food web.

Water Cycling

Soil animals are active participants in the formation of soil structure, which is an important characteristic that influences water infiltration, soil water retention, and percolation.^[4] The biochemical activity of soil organisms transforms organic materials into soil-stabilizing cementing agents, which bind the primary soil particles (i.e., sand, silt, and clay) into aggregates. In addition, the burrowing activity of soil animals creates larger pores alongside water-stable aggregates to increase total porosity of soil, which aids water flow without decreasing overall water retention capacity and improves the plant rooting environment.

Both aggregates and porosity are important components of soil structure. Poor soil structure due to disruption of

system. They are able to decompose cellulose in wood because they harbor various microorganisms (protozoa, bacteria, or fungi) to aid in decomposition. Better drainage and aeration of termite mounds may be beneficial to nearby plant growth in soils with a high water table. Stable macro-channels created by termites can improve water infiltration into soils that otherwise would form impermeable surface crusts.

Protozoa

Protozoa are single-celled animals that generally consume bacteria and soluble organic matter. They are more numerous in marine and freshwater environments but do occur widely in water films of many soils.^[5] Their principal soil function is predation on soil bacteria, which releases nutrients for potential plant uptake; increases decomposition and soil aggregation by stimulating their bacterial prey; and prevents some bacterial pathogens from developing on plant roots.

SOIL BIODIVERSITY

There has been a great deal discovered about soils and the organisms that live in them, yet it is estimated that <10% of the species of soil organisms have been identified globally. A rich diversity of genetic information resides in soil. An awaiting challenge is to discover the ecological consequences of soil biodiversity. A critical understanding of how organisms interact has begun, yet there will be much more to learn about how species diversity interacts with functional diversity. There may be soil functions provided by organisms that we are unaware of.

How biodiversity relates to ecosystem functioning is an intensive area of modern research in both aboveground and belowground ecology.^[6,7] The experimental evidence

suggests that the loss of species richness due to perturbations may not always lead to the loss of ecosystem functioning, especially in initially species diverse functional groups.^[8] Ecosystem functions can be performed by a number of different species within a trophic level, suggesting that functional redundancy is a mechanism to insure stability. However, the loss of functional groups in trophic levels closest to the base of the detrital food web would be the most detrimental to ecosystem stability. Future work on soil biodiversity should unravel the relative importance of species richness on the resistance and resilience of ecological processes under various short- and long-term stresses and environmental conditions.

REFERENCES

1. Wallwork, J.A. *Ecology of Soil Animals*; McGraw-Hill: London, 1970; 283 pp.
2. Beare, M.H.; Coleman, D.C.; Crossley, D.A., Jr; Hendrix, P.F.; Odum, E.P. A hierarchical approach to evaluating the significance of soil biodiversity to biogeochemical cycling. *Plant Soil* **1995**, *170* (1), 5–22.
3. Coleman, D.C.; Crossley, D.A., Jr. *Fundamentals of Soil Ecology*; Academic Press: San Diego, 1996; 205 pp.
4. Brady, N.C.; Weil, R.R. *The Nature and Properties of Soils*, 12th Ed.; Prentice Hall: Upper Saddle River, 1999; 4–8.
5. Dabyshire, J.F., Ed. *Soil Protozoa*; CAB International: Wallingford, 1994; 209.
6. Tilman, D.; Knops, J.; Wedin, D.; Reich, P.; Ritchie, M.; Siemann, E. The influence of functional diversity and composition on ecosystem processes. *Science* **1997**, *277*, 1300–1302.
7. Wardle, D.A. How soil food webs make plants grow. *Trends Ecol. Evol.* **1999**, *14*, 418–420.
8. Laakso, J.; Setälä, H. Sensitivity of primary production to changes in the architecture of belowground food webs. *Oikos* **1999**, *87*, 57–64.

Animals: Microbial Interactions and Nutrient Cycling

Lisa Cole

Richard D. Bardgett

Department of Biological Sciences, Institute of Environmental and Natural Sciences,
University of Lancaster, Lancaster, U.K.

Abstract

Although most mineralization of nutrients is governed directly by the activities of bacteria and fungi, their ability to do this is strongly affected by soil animals. The effects of biotic interactions on the structure and activity of microbial communities, and the consequences of this regards nutrient mineralization, are discussed for the common groups of macro- and mesobiota found in soils, namely the earthworms, microarthropods, and enchytraeid worms.

INTRODUCTION

The decomposer food web has a primary role in determining the mineralization of nutrients in soil and hence plant nutrient acquisition and plant productivity.^[1] Although most mineralization of nutrients is governed directly by the activities of bacteria and fungi, their ability to do this is strongly affected by soil animals. Soil animals affect microbial communities either directly through selectively feeding on fungi and bacteria or indirectly by comminution of organic matter, dissemination of microbial propagules, and the alteration of nutrient availability.^[2] Combined, these interactions between microorganisms and animals drive the ecosystem-level processes of energy flow and nutrient cycling and, therefore, contribute to primary production.^[3–5] The effects of these biotic interactions on the structure and activity of microbial communities, and the consequences of this regards nutrient mineralization, will be discussed for the common groups of macro- and mesobiota found in soils, namely the earthworms, microarthropods, and enchytraeid worms.

SELECTIVE FEEDING BY SOIL FAUNA

Although soil animals are considered to be both generalist and opportunist in their feeding behavior, two main groups of feeder that have been distinguished are bacterivorous and fungivorous. The structure of the microbial community is to some extent controlled by the selective feeding activities of these animal groups; selection of prey is based on their palatability, which varies with species, age, and physiological status; e.g., the collembolan *Folsomia candida* has been shown to feed on metabolically active hyphae in preference to dead or inactive hyphae^[6] and also selects regions of the fungal thallus with high

nitrogen (N) content.^[7] Preferential grazing is also thought to arise as a result of the avoidance of toxins that are produced by some fungal colonies,^[8] and some species of Collembola have been shown to locate and select their fungal food source by volatile compounds released from fungal mycelium.^[9]

Intermediate levels of grazing have been shown to enhance the activity and growth of selected fungi.^[10] Stimulation of fungal growth, which is often referred to as compensatory growth, is due to new fungal growth after senescent hyphae are grazed and regrowth after periodic grazing of actively growing mycelia. For example, grazing of the fungus *Mortierella isabellina* by the collembolan *Onychiurus armatus* induced switching from a “normal” hyphal mode, with appressed growth and sporulating hyphae, to fan-shaped sectors of fast-growing and non-sporulating mycelium with extensive areas of aerial hyphae.^[11] In addition, activities of specific amylase (starch-degrading enzymes) were several times higher in grazed cultures than in those cultures that were ungrazed.^[11] Heavy grazing of fungal communities can reduce their activity. For example, heavy grazing of mycorrhizae has been shown to counteract the mutualistic relationship between mycorrhizae and their host plant.^[12]

Selective grazing by soil fauna can alter the composition of the soil microbial community by altering the competitive relationships between species of microorganisms. Selective grazing by the collembolan *Orthogonius latus* on the fungus *Marasmius androsaceus* in coniferous leaf litter resulted in a reduction in the activity of this palatable fungi and an increase in an unpalatable fungus *Mycena galopus* present in the litter.^[13,14] Because *M. galopus* decomposes litter at a slower rate than *M. androsaceus*, the presence of *O. latus* reduced the decomposition of leaf litter.^[13,14]

The ingestion and exposure of microbes to intestinal fluids can influence microbial activity. Microbial

communities have been shown to be more abundant and active following passage through the gut of earthworms.^[15] The passage of microorganisms through earthworm (e.g., *Lumbricus terrestris*) guts has been shown to activate dormant bacteria due to the removal of endospore cell coats and subsequent germination of bacterial spores.^[16] Conversely, it has been suggested that the earthworm gut is a hostile environment for soil fungi;^[17] the germination of fungal spores (*Fusarium lateritium*, *Agrocybe temulenta*, *Trichoderma* sp., and *Mucor hiemalis*) was reduced following ingestion by *L. terrestris* and *Aporrectodea longa*. It was demonstrated that the germination of fungal spores was enhanced by physical abrasion by soil particles in the gut, but that subsequent exposure to intestinal fluid significantly reduced germination.^[17]

EFFECTS ON NUTRIENT CYCLING AND PLANT GROWTH

The most studied interactions between invertebrates and the microbial community in soils involve those that result in the release of nutrients from the microbial biomass as a result of the feeding activities (grazing) of animals. In general, soil fauna have lower carbon (C)/N ratios than the microbes upon which they graze, and therefore they excrete nutrients that are not required for production in forms that are biologically available (i.e., inorganic forms—"mineralized"). This release of nutrients into the soil system is effectively remobilization of the nutrients that were bound up in the microbial biomass and has been termed the "microbial loop."^[18] In addition, this grazing of microorganisms often leads to changes in the structure, size, and activity of the microbial community in the soil; as discussed above, these changes in microbial communities indirectly affect the processes of nutrient mineralization.

Many studies demonstrate the positive effects of soil fauna on C turnover and N mineralization in soil and litter materials. A study that used litter bags buried in the field^[19] demonstrated that the exclusion of enchytraeids and dip-teran larva from the litter bags resulted in lower rates of decomposition of litter. Likewise, laboratory microcosm studies have shown that the presence of different groups of animals in soil enhances the processes of nutrient mineralization; e.g., the grazing activities of enchytraeids on microorganisms that colonize litter have been shown to result in increased rates of leaching of ammonium and dissolved organic C^[20,21] associated with increased microbial activity.^[21] The earthworm *Aporrectodea caliginosa* has been shown to increase C and N mineralization in soils amended with birch litter,^[22] despite a reduction in the microbial biomass. The grazing of fungi on litter by the Collembola *F. candida*^[23] and *Tomocerus minor*^[24] also increased the mineralization of N. These positive effects of soil fauna have been shown to translate into increased uptake of nutrients and growth of plants. It has been shown

that Collembola and nematodes increased the concentrations of ammonium in the soil solution of an upland organic soil, leading to increased plant nutrient uptake by an upland grass species, *Nardus stricta*.^[4] It has also been demonstrated that although grazing by the earthworm *A. caliginosa* reduced the root biomass of the grass *Hordelymus europaeus*, this was associated with an increased N content in its shoot and roots, due to increased availability of mineral N in the presence of the earthworm.^[5] The leaf, stem, and shoot biomass of birch seedlings (*Betula pendula*) was increased when grown in the presence of a diverse soil fauna.^[3] Further studies have demonstrated that reductions in microbial biomass through grazing activities of soil fauna can benefit plant growth, most likely as a result of increased activity of the microorganisms. For example, grazing of ectomycorrhizal fungi associated with *B. pendula* by a diverse soil fauna resulted in enhanced growth of and nutrient uptake by *B. pendula*, despite reductions in the biomass of ectomycorrhiza.^[25]

The effects of interactions between fauna and microorganisms on nutrient release are often regulated by resource quality. In studies of N-limited soil, Collembola did not increase the amount of nitrate and ammonium released into soil solution, because the extra N that was released as a result of animal grazing was rapidly re-utilized by the growing fungus that was nutrient limited.^[10] Similarly, a microcosm study showed that collembolan grazing on a leaf litter-inhabiting fungus had no effect on nutrient release due to efficient re-utilization of nutrients by the fungus.^[26] These studies, however, were conducted in the absence of plants, which are effective competitors with microbes for nutrients in N-limited soil.^[27] Studies of ecosystem-scale N cycling point to the importance of these animal-microbial interactions in controlling seasonal patterns of plant-microbial competition for nutrients in nutrient-limited ecosystems.^[27]

Some works suggest that the effects of these biotic interactions on nutrient release are complex and involve a diversity of species in more than one trophic group;^[28,29] e.g., the effects of feeding of fungal-feeding Collembola on nutrient cycling have been shown to become apparent only when they are interacting with another trophic group of soil fauna, namely microbial-feeding nematodes.^[4] Likewise, it has been demonstrated that combinations of soil animals, as opposed to a single group of soil fauna, had a synergistic effect on the microbial community in microcosms of coniferous forest humus, resulting in enhanced leaching of mineral nutrients.^[30] There is also evidence for top-down regulation of microbial biomass and nutrient mineralization rates in studies with microbes, microbivorous nematodes, and predatory organisms.^[31] However, some microcosm studies^[32] show that lower trophic levels in soil food webs are considerably more important than higher ones in regulating ecosystem productivity, as measured by plant growth. What is clear from these studies is that further testing of the effects of soil fauna on plant-microbe nutrient

competition needs to examine the influence of more complex trophic relations.

Although not a direct interaction, the ability of soil fauna to comminute or fragment organic materials indirectly influences the activity of microorganisms that are involved in decomposition and nutrient cycling by increasing the surface area available for microbial colonization.^[19,33]

DISPERSAL OF MATERIALS

The movement of animals through the soil provides a passive means of transport for microbial propagules to new sites in the soil, both through the attachment and transport of microbes on the body surface and through the ingestion and excretion of microbes at new sites in the soil. Mites, Collembola, and earthworms have been shown to carry propagules of several species of saprotrophic fungi and arbuscular mycorrhizal fungi.^[34,35] Soil fauna also transport plant pathogens; the earthworm *L. terrestris* is known to enhance the dispersal of *Syntrichium endobioticum*, the causal agent of wart disease of potato.^[36] This has led to concerns that soil fauna may provide a means for the dispersal of genetically modified organisms. The beneficial role of soil fauna for the delivery of biocontrol agents to the rhizosphere is also a research theme in the control of root pathogens.

REFERENCES

1. Wardle, D.A. How soil food webs make plants grow. *Trends Ecol. Evol.* **1999**, *14*, 418–420.
2. Griffiths, B.S.; Bardgett, R. Interactions between microbe-feeding invertebrates and soil microorganisms. In *Modern Soil Microbiology*; van Elsas, J.D., Wellington, E.M.H., Trevors, J.T., Eds.; Marcel Dekker: New York, 1997; 165–182.
3. Setälä, H.; Huhta, V. Soil fauna increase *Betula pendula* growth: Laboratory experiments with coniferous forest floor. *Ecology* **1991**, *72*, 665–671.
4. Bardgett, R.D.; Chan, K.F. Experimental evidence that soil fauna enhance nutrient mineralization and plant nutrient uptake in montane grassland ecosystems. *Soil Biol. Biochem.* **1999**, *31*, 1007–1014.
5. Alpehi, J.; Bonkowski, M.; Scheu, S. Protozoa, nematoda and lumbricidae in the rhizosphere of *Hordelymus europaeus* (Poaceae): Faunal interactions, response of microorganisms and effects on plant growth. *Oecologia* **1996**, *106*, 111–126.
6. Moore, J.C.; St. John, T.V.; Coleman, D.C. Ingestion of vesicular-arbuscular mycorrhizal hyphae and spores by soil microarthropods. *Ecology* **1985**, *66*, 1979–1981.
7. Leonard, M.A. Observations on the influence of culture conditions on the fungal feeding preference of *Folsomia candida* (Collembola; Isotomidae). *Pedobiologia* **1984**, *26*, 361–367.
8. Parkinson, D.; Visser, S.; Whittaker, J.B. Effects of collembolan grazing on fungal colonization of leaf litter. *Soil Biol. Biochem.* **1979**, *11*, 529–535.
9. Bengtsson, G.; Erlandsson, A.; Rundgren, S. Fungal odour attracts soil collembola. *Soil Biol. Biochem.* **1988**, *20*, 25–30.
10. Bardgett, R.D.; Whittaker, J.B.; Frankland, J.C. The effect of collembolan grazing on fungal activity in differently managed upland pastures: A microcosm study. *Biol. Fert. Soils* **1993**, *16*, 255–262.
11. Hedlund, K.; Boddy, L.; Preston, C.M. Mycelial responses of the soil fungus *Mortierella isabellina*, to grazing by *Onychiurus armatus* (Collembola). *Soil Biol. Biochem.* **1991**, *23*, 361–366.
12. Warnock, A.J.; Fitter, A.M.; Usher, M.B. The influence of a springtail *Folsomia candida* (insecta, collembola) on the mycorrhizal association of leek *Allium porrum* and the vesicular-arbuscular mycorrhizal endophyte *Glomus fasciculatum*. *New Phytol.* **1982**, *90*, 285–292.
13. Newell, K. Interaction between two decomposer basidiomycetes and a collembolan under sitka spruce: Distribution, abundance and selective grazing. *Soil Biol. Biochem.* **1984**, *16*, 227–233.
14. Newell, K. Interaction between two decomposer basidiomycetes and a collembolan under sitka spruce: Grazing and its potential effects on fungal distribution and litter decomposition. *Soil Biol. Biochem.* **1984**, *16*, 235–239.
15. Brown, G.G. How do earthworms affect microfloral and faunal community diversity? *Plant Soil* **1995**, *170*, 209–231.
16. Fischer, K.; Hahn, D.; Hönerlage, W.; Zeyer, J. Effect of passage through the gut of the earthworm *Lumbricus terrestris* L. on *Bacillus megaterium* studied by whole cell hybridization. *Soil Biol. Biochem.* **1997**, *29*, 1149–1152.
17. Moody, S.A.; Pearce, T.G.; Dighton, J. Fate of some fungal spores associated with wheat straw decomposition on passage through the guts of *Lumbricus terrestris* and *Aporrectodea longa*. *Soil Biol. Biochem.* **1996**, *28*, 533–537.
18. Clarholm, M. Interactions of bacteria, protozoa and plants leading to mineralization of soil nitrogen. *Soil Biol. Biochem.* **1985**, *17*, 181–187.
19. Standen, V. The influence of soil fauna on decomposition by micro-organisms in blanket bog litter. *J. Anim. Ecol.* **1978**, *47*, 25–38.
20. Förster, B.; Römbke, J.; Knacker, T.; Morgan, E. Microcosm study of the interactions between micro-organisms and enchytraeid worms in grassland soils and litter. *Eur. J. Soil Biol.* **1995**, *31*, 21–27.
21. Cole, L.; Bardgett, R.D.; Ineson, P. Enchytraeid worms (Oligochaeta) enhance mineralization of carbon in organic upland soils. *Eur. J. Soil Sci.* **2000**, *51*, 185–192.
22. Saetre, P. Decomposition, microbial community structure and earthworm effects along a birch-spruce soil gradient. *Ecology* **1998**, *79*, 834–846.
23. Ineson, P.; Leonard, M.A.; Anderson, J.M. Effect of collembolan grazing upon nitrogen and cation leaching from decomposing leaf litter. *Soil Biol. Biochem.* **1982**, *14*, 601–605.

24. Teuben, A. Nutrient availability and interactions between soil arthropods and microorganisms during decomposition of coniferous litter: A mesocosm study. *Biol. Fert. Soils* **1991**, *10*, 256–266.
25. Setälä, H. Growth of birch and pine seedlings in relation to grazing by soil fauna on ectomycorrhizal fungi. *Ecology* **1995**, *76*, 1844–1851.
26. Visser, S.; Whittaker, J.B.; Parkinson, D. Effects of collembolan grazing on nutrient release and respiration of a leaf litter inhabiting fungus. *Soil Biol. Biochem.* **1981**, *13*, 215–218.
27. Jaeger, C.H., III; Monson, R.K.; Fisk, M.C.; Schmidt, S.K. Seasonal partitioning of nitrogen by plants and soil microorganisms in an alpine ecosystem. *Ecology* **1999**, *80*, 1883–1891.
28. Bardgett, R.D.; Cook, R. Functional aspects of soil animal diversity in agricultural grasslands. *Appl. Soil Ecol.* **1998**, *10*, 263–276.
29. Mikola, J.; Setälä, H. No evidence of trophic cascades in an experimental microbial-based soil food web. *Ecology* **1998**, *79*, 153–164.
30. Setälä, H.; Tyynismaa, M.; Martikainen, E.; Huhta, V. Mineralization of C, N and P in relation to decomposer community structure in coniferous forest soil. *Pedobiologia* **1991**, *35*, 285–296.
31. Brussaard, L.; Noordhuis, R.; Geurs, M.; Bouwman, L.A. Nitrogen mineralization in soil in microcosms with and without bacterivorous nematodes and nematophagous mites. *Acta Zool. Fenn.* **1995**, *196*, 15–21.
32. Laakso, J.; Setälä, H. Sensitivity of primary production to changes in the architecture of belowground food webs. *Oikos* **1999**, *87*, 57–64.
33. Martin, A.; Marinissen, J.C.Y. Biological and physico-chemical processes in excrements of soil animals. *Geoderma* **1993**, *56*, 331–347.
34. Behan-Pelletier, V.M.; Hill, S.B. Feeding habits and spore dispersal of oribatid mites in the north american arctic. *Révue d'Écologie et de Biologie du Sol* **1978**, *15*, 497–516.
35. Reddell, P.; Spain, A.V. Earthworms as vectors of viable propagules of mycorrhizal fungi. *Soil Biol. Biochem.* **1991**, *23*, 767–774.
36. Hampson, M.C.; Coombes, J.W. Pathogenesis of *Syntrichium Endobioticum* VII: earthworms as vectors of wart disease on potato. *Plant Soil* **1989**, *116*, 147–150.

Antimony

Gudny Okkenhaug

Norwegian Geotechnical Institute (NGI), Oslo, Norway

Jan Mulder

Department of Plant and Environmental Sciences, Norwegian University of Life Sciences (UMB), Aas, Norway

Abstract

Antimony (Sb), a non-essential, toxic metalloid, is classified as a pollutant of priority interest by the U.S. Environmental Protection Agency. In soils, Sb is commonly oxidized to pentavalent antimonate, $\text{Sb}(\text{OH})_6^-$. Iron and manganese oxides are important sorbents, but $\text{Sb}(\text{OH})_6^-$ is relatively mobile in neutral and alkaline soils. In reducing conditions, Sb may form $\text{Sb}(\text{OH})_3$ or Sb sulfides. $\text{Sb}(\text{OH})_3$ is sorbed to oxides over a wide pH range. Due to the increased industrial use of Sb, contamination of the soil is of growing concern. In the most contaminated soils, the negative effects of Sb on plant growth and nitrification may be expected.

INTRODUCTION

The natural sulfide of antimony (Sb) was known and used in biblical times as medicine and cosmetic. Pliny (A.D. 50) referred to the metal as stibium, from which the chemical symbol Sb is derived. The name antimony is attributed to the Greek words *anti* and *monos*, which means “enemy of solitude” or “metal not found alone.”^[1] Sb is toxic and classified as a pollutant of priority interest by the U.S. Environmental Protection Agency.^[2] Due to the increased industrial use of Sb, contamination of the soil is of growing concern. Here, we discuss the most important aspects of Sb contamination of soils, including its sources, geochemistry, common concentration ranges, and toxicity.

CHEMISTRY OF Sb

Sb, a metalloid in group 15 of the periodic table, exists in a variety of oxidation states (−III, 0, III, V), with oxidation states III and V being the most common in the natural environment. In the lithosphere, Sb occurs as Sb sulfides, metal antimonides, and antimonoxides. The principal Sb-containing ore minerals are stibnite (Sb_2S_3) and valentinite [antimony trioxide (Sb_2O_3)].^[1] The physical and chemical properties of Sb are given in Table 1.^[1,3]

The predominant inorganic Sb species in soils are Sb(III) [as $\text{Sb}(\text{OH})_3^0$] and Sb(V) [as $\text{Sb}(\text{OH})_6^-$]. These species behave differently; thus, the redox state is important for the fate of Sb in the environment (Fig. 1). Sb is present as soluble $\text{Sb}(\text{OH})_6^-$ in oxic systems and as soluble $\text{Sb}(\text{OH})_3^0$ in more reduced systems. In reducing conditions, $\text{Sb}_2\text{S}_3(\text{s})$ may form at low pH. Inorganic species of Sb predominate over organic species in the most environment systems.^[5]

ABUNDANCE AND SOURCES OF Sb IN SOIL

In 2009, Sb was the ninth most exploited metal worldwide with approximately 187,000 tons being mined.^[6] World reserves of Sb are estimated to be in excess of two million tons and are located principally in Bolivia, China, Russia, South Africa, and Mexico. Sb is primarily used as a hardener in lead (Pb) for storage batteries but also in solders and other alloys (e.g., used in bullets). Sb_2O_3 , the most important industrially used Sb compound, is primarily used in flame retardants but also in paint pigments, glass, and ceramics. Organoantimony compounds are commonly used to treat tropical protozoan diseases (e.g., leishmaniasis).^[7]

Sb compounds are increasingly released into the environment during incineration of waste, combustion of fossil fuels, e.g., coal and petroleum, and smelting of metal ore. Other sources of Sb contamination of soils include ammunition at shooting ranges, road traffic (dust from brake linings and tires), and older battery-producing plants.^[8] Background concentrations of Sb in unpolluted soils (Table 2) are low (a few milligrams per kilogram). Higher concentrations are generally related to anthropogenic sources. Filella et al. have carried out an extensive literature review of the occurrence of Sb in the environment.^[8,9,10]

MOBILITY OF Sb IN SOIL

The fate and mobility of Sb in soil depend on speciation, soil characteristics, and environmental conditions. In soil, Sb is commonly oxidized to Sb(V), which is the dominant form observed in contaminated soils near smelters and at shooting ranges.^[11,12] Oxidation is enhanced by iron (Fe) and manganese (Mn) oxides.^[5,13,14] For example, in soil

Table 1 Physical and chemical properties of Sb.

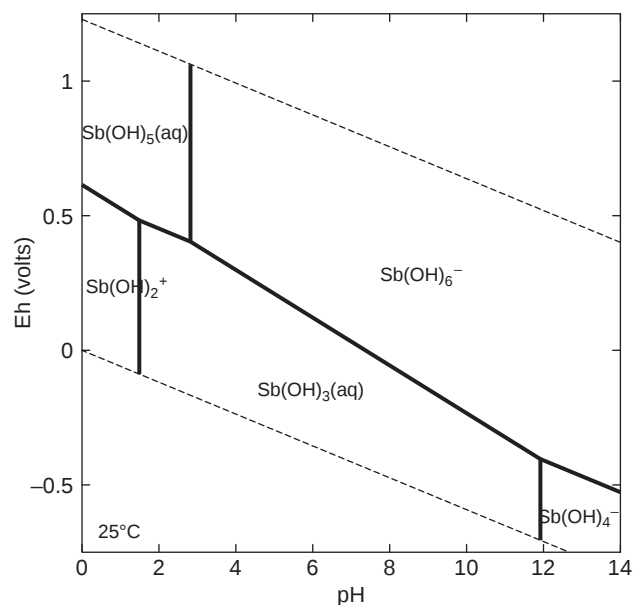
Property	Value
General	
Atomic number	51
Periodic table group	15
Group name	Pnictogen
Electronic configuration	
Ground state electron configuration	[Kr].4d ¹⁰ .5s ² .5p ³
Shell structure	2.8.18.18.5
Atomic mass (relative to ¹² C = 12.0000)	121.75
Naturally occurring isotopes	¹²¹ Sb (57.21%) ¹²³ Sb (42.79%)
Ionization energies (kJ/mol)	
M → M ⁺	833.7
M ⁺ → M ²⁺	1794
M ²⁺ → M ³⁺	2443
M ³⁺ → M ⁴⁺	4260
M ⁴⁺ → M ⁵⁺	5400
Electron affinity (kJ/mol)	101
Electron negativity	
Pauling scale	2.05
Allred–Rochow scale	1.82
Allen scale	1.984
Melting point (K)	903.89
Density (kg/m ³)	6691 (293 K)
Ionic radii (Å)	
Sb ⁵⁺	0.62
Sb ³⁺	0.76
Sb ^{3−}	2.45

Source: Adapted from Norman^[1] and Filella, Belzile, et al.^[3]

water 5 m downward the mine pit at the closed-down Ichinokawa Sb mine in Japan, Mitsunobu et al.^[12] found Sb to be in the oxidized form. In contrast, even at 1 cm depth, arsenic (As) mainly occurred in the reduced state (Fig. 2^[12]).

Due to its anionic character, Sb(V) is relatively mobile in the environment, having a low affinity for clay minerals.^[8] Natural Fe and Mn oxides are the most important sorbents in soil.^[5] The strongest sorption of Sb(V) in soil occurs at pH < 6 (Fig. 3^[13]). At pH > 6, the sorption of Sb shows a strong decreasing trend and is small compared with that of As and phosphorus (P). In contrast, the neutral Sb(III) is strongly sorbed to Fe oxides over a wide pH range of 3–12. The role of Sb interaction with organic phases is largely unexplored.^[5]

The sorption of Sb, commonly expressed by the distribution coefficient (K_d), depends on soil properties. Values reported in the literature vary greatly. For example, in agricultural soils in Japan, K_d values ranged from 1 to

**Fig. 1** Eh–pH diagram for the soluble Sb species at 25°C and a Sb concentration of 10^{−8} mol/L.

Source: From *The Geochemist's Workbench*.^[4]

2065 L/kg, with a mean of 62 L/kg. K_d values increased with decreasing pH and decreased with increasing phosphate concentration.^[15] For soils (down to 60 cm) at shooting ranges, K_d values varying between 158 L/kg and 2512 L/kg were found.^[16] At high contamination levels, precipitation of Ca-antimonate may control the concentrations of Sb(V) in soil water.^[10]

Table 2 Some published Sb concentrations in soils.

Soils	Total Sb (mg/kg dry weight)
Unpolluted soil, B horizon, the United States (1000 samples)	<1–8.8
Market-garden soils, marsh (surface soil), the Netherlands	1.85–2.11
Soil, Scotland, the United Kingdom (10 samples)	0.29–1.3
Humus layer of Norwegian soils (10 regions)	0.17–2.20
Sandstone and sands, Zimbabwe (arsenical soils)	50–5000
Soil near stack at gold-refining site, Yellowknife, Northwest Territories, Canada	280
Soils close to a Pb smelting complex (0.4–19 km from smelter, top layer), Kellogg, Idaho, the United States	5–260
Soils close to six different Copper smelter sites, the United States	3–204
Soils in proximity to Sb smelter (100 m from smelter), River Tyne, the United Kingdom	170–360

Source: From Filella, Belzile, et al.^[3]

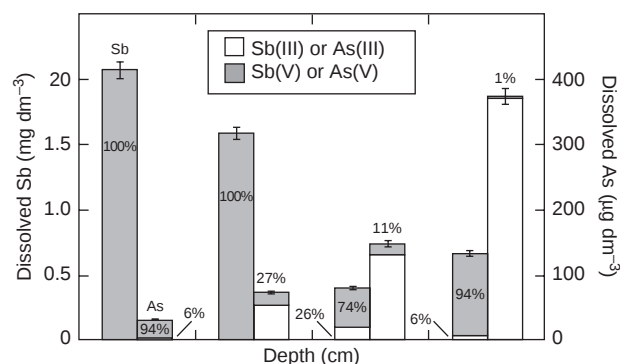


Fig. 2 Depth profile of Sb(III) and Sb(V) concentrations in soil water downward the mine pit at Ichinokawa mine, Japan.

Source: Reprinted in part with permission from Mitsunobu, Harada, et al.^[12]

Biologically mediated methylation of Sb, e.g., to monomethylstibine (CH_3SbH_2), has been detected in soils and plants. In urban soil samples from the German Ruhr basin, the highest concentrations of methylated Sb were observed in agricultural and garden soils, due to cultivation and a high biological activity (up to 1.6% of the total Sb content).^[17] Methylated Sb concentrations in the environment are low and are not considered to be of great concern.^[8]

TOXICITY AND BIOLOGICAL EFFECTS

Sb, a non-essential element, is toxic at high concentrations. The toxicity is a function of water solubility and oxidation state, and in general, Sb(III) is more toxic than Sb(V).^[8] The World Health Organization drinking water limit for Sb is set at 20 $\mu\text{g/L}$.^[18]

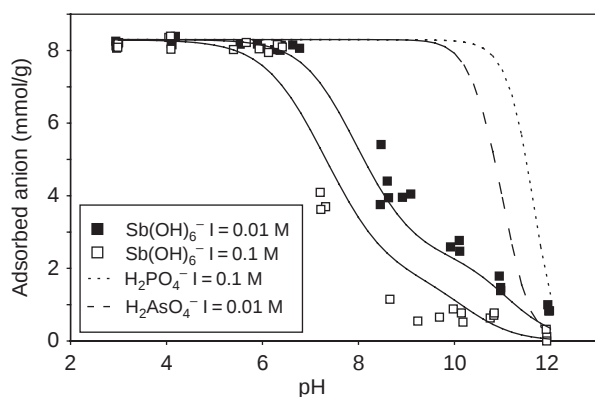


Fig. 3 Sorption of Sb(V) on goethite at $I = 0.01 \text{ M}$ (solid symbols) and $I = 0.1 \text{ M}$ (open symbols). Experimental conditions: $[\text{Sb(V)}]_0 = 4.15 \mu\text{M}$, 0.5 g/L goethite, 25°C. Dashed lines show predicted adsorbed P and As concentrations for the same initial concentrations of anions and goethite.

Source: Reprinted in part with permission from Leuz, Monch, et al.^[13]

The knowledge about the toxicity of Sb in soils is limited. In laboratory toxicity tests, half maximal effective concentration values for Sb range from 70 to >1000 mg/kg.^[19,20] Reported values depend on end point, soil, and the Sb form tested. Thresholds for the toxicity of Sb (V) with respect to plant growth and nitrification were over 100-fold larger than background concentrations in the European soil (0.1–1.9 mg/kg, Table 2).^[14]

It is not clear whether bioaccumulation of Sb occurs.^[9] In Sb-mining areas, high Sb concentrations have been found in vegetation (up to 1400 mg/kg).^[21] Furthermore, studies on invertebrates and mammals from grassland in the vicinity of a Sb smelter showed elevated Sb concentrations in tissues. On the other hand, other studies in such Sb-contaminated areas have shown low uptake and bioaccumulation in plants and microorganisms, due to low solubility of Sb in water. Similarly, plants growing in contaminated soils (concentrations up to 500 mg/kg) have about the same level of Sb uptake as plants growing in uncontaminated soils.^[16]

CONCLUSION

Sb occurs at increasing concentrations in soils, due to its growing industrial use. In oxic conditions, Sb occurs mainly as Sb(V) $[\text{Sb(OH)}_6^-]$. Fe and Mn oxides are the most important sorbents for Sb(V), particularly in acidic soils. In neutral and alkaline soils, Sb(V) is relatively mobile. Sb(III), important in reducing conditions, is sorbed strongly to oxides. Due to its increased use, elevated concentrations in the environment, and toxicity, further research on Sb is warranted.

REFERENCES

1. Norman, N.C., Ed. *Chemistry of Arsenic, Antimony and Bismuth*; Blackie Academic & Professional: London, 1998.
2. U.S. EPA. *Water-Related Fate of the 129 Priority Pollutants*; USEPA: Washington, 1979. Available at www.epa.gov/waterscience/methods/pollutants.htm.
3. Filella, M.; Belzile, N.; Chen, Y.-W. Antimony in the environment: A review focused on natural waters I. Occurrence. *Earth Sci. Rev.* **2002**, 57 (1–2), 125–176.
4. *The Geochemist's Workbench® (GWB) Standard*; Release 8.0. Datasets: thermo.com.v8.r6+.dat, thermo_minteq.dat.
5. Wilson, S.C.; Lockwood, P.V.; Ashley, P.M.; Tighe, M. The chemistry and behaviour of antimony in the soil environment with comparisons to arsenic: A critical review. *Environ. Pollut.* **2010**, 158, 1169–1181.
6. U.S. Geological Survey. *Mineral Commodity Summaries 2010*; U.S. Geological Survey: Washington, 2010.
7. Buttermann, W.C.; Carlin, J.F. *Mineral Commodity Profiles, Antimony*. Open-File Report 03–019; U.S. Department of the interior, U.S. Geological Survey: 2004. Available at <http://pubs.usgs.gov/of/2003/of03019/index.html#minerals>.

- usgs.gov/minerals/pubs/commodity/antimony/ (accessed September 2008).
8. Filella, M.; Belzile, N.; Chen, Y.-W. Antimony in the environment: A review focused on natural waters II. Relevant solution chemistry. *Earth Sci. Rev.* **2002**, *59* (1–4), 265–285.
 9. Filella, M.; Belzile, N.; Lett, M.-C. Antimony in the environment: A review focused on natural waters III. Microbiota relevant interactions. *Earth Sci. Rev.* **2007**, *80* (3–4), 195–217.
 10. Filella, M.; Williams, P.A.; Belzile, N. Antimony in the environment: knowns and unknowns. *Environ. Chem.* **2009**, *6*, 95–105.
 11. Johnson, C.A.; Moench, H.; Wersin, P.; Kugler, P.; Wenger, C. Solubility of antimony and other elements in samples taken from shooting ranges. *J. Environ. Qual.* **2005**, *34* (1), 248–254.
 12. Mitsunobu, S.; Harada, T.; Takahashi, Y. Comparison of antimony behavior with that of arsenic under various soil redox condition. *Environ. Sci. Technol.* **2006**, *40* (23), 7270–7276.
 13. Leuz, A.K.; Monch, H.; Johnson, C.A. Sorption of Sb(III) and Sb(V) to goethite: Influence on Sb(III) oxidation and mobilization. *Environ. Sci. Technol.* **2006**, *40* (23), 7277–7282.
 14. Oorts, K.; Smolders, E.; Degryse, F.; Buekers, J.; Gascó, G.; Cornelis, G.; Mertens, J. Solubility and toxicity of antimony trioxide (Sb₂O₃) in soil. *Environ. Sci. Technol.* **2008**, *42* (12), 4378–4383.
 15. Nakamaru, Y.; Tagami, K.; Uchida, S. Antimony mobility in Japanese agricultural soils and the factors affecting antimony sorption behavior. *Environ. Pollut.* **2006**, *141* (2), 321–326.
 16. Olivé, I.X. Mobility of Lead and Antimony in Shooting Range Soils; Doctoral Thesis ETH No. 16689; Swiss Federal Institute of Technology: Zurich, 2006.
 17. Duester, L.; Diaz-Bone, R.A.; Kusters, J.; Hirner, A.V. Methylated arsenic, antimony and tin species in soils. *J. Environ. Monitor.* **2005**, *7* (12), 1186–1193.
 18. WHO. *Antimony in Drinking-water. Background Document for Development of WHO Guidelines for Drinking-water Quality*; World Health Organisation, 2003. Available at http://www.who.int/water_sanitation_health/dwq/chemicals/0304_74/en/index.html (accessed September 2008).
 19. Kuperman, R.G.; Checkai, R.T.; Simini, M.; Phillips, C.I.; Speicher, J.A.; Barclift, D.J. Toxicity benchmarks for antimony, barium, and beryllium determined using reproduction endpoints for *Folsomia candida*, *Eisenia fetida*, and *Enchytraeus crypticus*. *Environ. Toxicol. Chem.* **2006**, *25*, 754–762.
 20. Hammel, W.; Steubing, L.; Debus, R. Assessment of the ecotoxic potential of soil contaminants by using a soil-algae test. *Ecotox. Environ. Safe.* **1998**, *40*, 173–176.
 21. Tschan, M.; Robinson, B.H.; Schulin, R. Antimony in the soil-plant system – A review. *Environ. Chem.* **2009**, *6*, 106–115.

Ants

Walter G. Whitford

Jornada Experimental Range, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Las Cruces, New Mexico, U.S.A.

Abstract

Ants contribute to heterogeneity in soil properties by the construction of subterranean nests and by accumulating organic materials in and around nests. The construction and maintenance of nests affect soil turnover, macroporosity, and mixing of soil profiles. Foraging and food processing behaviors affect soil nutrient concentrations and soil microflora and microfauna. The magnitude of the effects of ants on soils is dependent upon soil type and topographic position.

INTRODUCTION

Ants are among the most ubiquitous insects on the planet. They inhabit in all biomes except for the extreme polar regions. In the biomes where ants are abundant, they affect many soil processes that contribute to the creation of patch mosaics that characterize the soils and vegetation of many landscapes. The abundance and diversity of soil-nesting ants vary from as high as 7000 colonies per hectare in tropical savanna to as few as 3–4 colonies per hectare on some periodically flooded, fine-textured soil, desert landscape units.^[1] Soil-nesting ants affect critical ecosystem processes such as nutrient cycling and water redistribution. Ant nest mounds vary from a few centimeters in height and diameter to >1 m in height and >2 m diameter.^[2] Ant nests consist of underground and branched networks of galleries and chambers. Surficial chambers are connected to lower chambers by vertical galleries with branching lateral galleries. Galleries and chambers vary in size and number, depending upon the species of ant. For example, *Lasius neoniger*, an abundant ant species in temperate North America, constructs tubular galleries of 1.5–5.0 mm diameter and chambers of 10–20 mm diameter and 30–50 mm length. The volume of *L. neoniger* nests ranges from 20 to 250 cm³ and the nests are confined to the upper 70 cm of soil.^[3] Other species construct nests to depths ranging from 50 cm to greater than several meters, depending upon species-specific behavior, soil type, and landscape position. Soil profile mixing, texture, physical and chemical property modification of mound soils, soil macroporosity, and geomorphological attributes of ant nest mounds vary with species-specific colony longevity, body size, and numbers of workers of a colony, soil type, and landscape position. The perturbation effects of ants therefore depend upon the species composition of the ant community, geomorphic history, soil properties, and topographic position of a landscape unit.

Because most studies concerning the effects of ants on soils have focused on one or two species, a comprehensive analysis of the combined effects of all ant species on the soils of an ecosystem cannot be made.

MICROTOPOGRAPHY

In areas that are periodically flooded or where the water table is close to the surface, some species of soil-nesting ants build mounds that create favorable microhabitats for themselves and also a habitat for some species of plants that are confined to the aerated soils of the ant mounds. Soil-nesting ants create hummock microtopography in some wet meadow fens and tropical wet savannas.^[4] In the Chaco region of South America (parts of Paraguay, Bolivia, Argentina, and Brazil), nest mounds of *Campotonotus punctulatus* occur at densities of between 200 and 1000 mounds/ha. These conical mounds average a height of 0.62 m (with a maximum of 1.85 m) with a mean basal diameter of 1.2 m. The mound soils are lighter textured than surrounding soils, reflecting the amount of materials transported from surrounding subsurface soil during mound construction.^[5] *Formica podzolica* mounds in a Montana fen are thought to contribute to the hummock-hollow microtopography of peat lands. Abandoned *F. podzolica* mounds provide drier and warmer microsites that are enriched with some soil nutrients.^[4] The mounds of *Lasius flavus* contribute to the microtopography of some European grasslands and salt marshes.^[6] Mima-type earth mounds up to a height of 1.5 m with a diameter of 20 m in Buenos Aires Province, Argentina, are produced by horizontal translocation of soil to the colony sites of black fire ants, *Solenopsis richteri*. Continued occupation of the mounds by successive generations of ants gradually increases the size of the mounds to mima-type size.^[7] Ants (*Formica* spp. and *Myrmica* spp.) are

important agents in the process of development and maintenance of hummock microtopography of subarctic peatlands. Hummock retrogression is accelerated by the tunneling activity of ants.^[8]

HETEROGENEITY OF PHYSICAL AND CHEMICAL PROPERTIES OF THE SOIL

Many species of ants alter the texture and chemistry of the soil in the nest mounds. The nutrients most frequently reported to be at higher concentrations in ant mound soils include nitrogen, phosphorus, potassium, calcium, magnesium, manganese, and iron.^[9] The effect of soil-nesting ants on soil nutrient patchiness and on vegetation varies as a function of landscape position, soil type, and the biology of the ant species. Nutrient enrichment of mound soils has been reported for several species of seed-harvesting ants and omnivorous species of ants that collect seeds, prey on insects, or collect insect carrion. Species of soil-nesting ants that enrich the nutrient content of mound soils are characterized by relatively long-lived colonies (>5 years) and the behavior of depositing chaff and unwanted insect parts on and around the nest mound or disk. Nutrient enrichment of mound soils by a species may not occur on all soils on a watershed or landscape. For example, *Pogonomyrmex rugosus* nest disks in desert shrubland and mixed shrub grassland were nutrient enriched, but the nest disks of this species in a piedmont grassland were not nutrient enriched.^[10] *Formica* spp. mounds in forest were nutrient enriched, but *Formica* spp. mounds in meadows and grasslands were not.^[11]

The variability in soil nutrient enrichment of ant mounds has been documented in several species of leaf-cutting ants. In remnant Cerrado (woodland savanna), Brazil, leaf-cutting ants (*Atta* spp.) had no detectable effect on nutrient enrichment.^[12] In northern Patagonia, soils associated with the leaf-cutting ant, *Acromyrmex lobicornis*, had higher concentrations of nitrogen, phosphorus, and organic matter than reference soils.^[13] The location of nutrient-rich organic refuse produced by leaf-cutting ant colonies varies among species. *Atta cephalotes* deposit organic refuse in subterranean chambers, whereas *Atta colombica* place organic refuse on the soil surface near the nest. The location of organic refuse is a major factor affecting nutrient concentrations and the composition, abundance, and activity of soil microflora and microfauna.^[14] In the Orinoco Llanos savanna, Venezuela, *Atta laevigata* nests had higher concentrations of nitrogen, magnesium, calcium, and organic carbon, but other soil nutrients and properties were not affected by ant mounds.^[15]

In an Australia Vertisol, ant nest soils had greater concentration of coarse and particulate organic matter, lower fine particulate soil organic matter (SOM)/coarse

particulate SOM ratios, larger sand content, and lower clay content than surrounding soils.^[16] Nutrient enrichment of nest mound soils of funnel ants (*Aphaenogaster barbigula*) was attributed to entrapment of organic materials around the nest entrances. Reexcavation of nest chambers after rainfall buries trapped litter, resulting in higher concentrations of nitrogen, organic matter, and some cations compared to nest-free soils.^[17] In humid tropical savanna, ant mounds of *Camponotus* spp. had higher clay and coarse sand content than surrounding soils.^[9] Even exotic or alien species of ants change the chemical and physical properties of nest mound soils. Mounds of imported fire ants (*Solenopsis invicta*) had higher concentrations of clay, phosphorus, and potassium and lower concentration of SOM than reference soils. The effect of *S. invicta* on calcium concentrations relative to reference soils was dependent upon the characteristics of the unmodified soil.^[18]

Ants change the nutrient concentration of mound soils, but the physical and chemical properties of mound soils can also affect mineralization processes. Nitrogen mineralization rates were reduced in nest mound soils in moss sedge, sedge, and alder peat habitats.^[19]

SOIL TURNOVER

The longevity and turnover rates of nests and nest mounds of ants in a community frequently follow a distribution gradient from high turnover (<3 months) to long-lived (>10 years). The importance of ants in the transport of subsurface horizon materials to the surface varies with the density and diversity of the ant community on a landscape unit. In Chihuahuan Desert grasslands, soil-nesting ants are an order of magnitude more abundant on sand and sandy loam soils than on fine-textured soils. Ants were estimated to move between 21.3 and 85.8 kg/ha/yr on sandy and sandy loam soils and between 0.1 and 3.4 kg/ha/yr on clay and clay-loam soils.^[20] The estimated annual soil turnover by ants in an *Atriplex vesicaria* shrubland in the semiarid region of Australia was 350–420 kg/ha/yr.^[9] Soil that is excavated by ants in the construction of galleries and chambers and deposited on mounds around nest entrances is generally eroded by water and wind within a year unless the mound is protected from raindrop splash erosion by gravel, stones, or wood fragments. Nest mound soils may be replenished by the belowground expansion of galleries and chambers. Ant nest mounds in sparsely vegetated arid regions are prone to wind erosion. On an Australian Aeolian soil, funnel ants' (*A. barbigula*) nests were active for approximately 9 months and the ants changed location approximately twice per year. Soil transport was estimated to be 33.6 kg/ha, and it was estimated that 92% of the soil volume would be turned over by these ants in 100 years.^[21] In Western Australia, ant communities on gray soils of semiarid woodlands were

estimated to turnover 46.5 kg ha/yr and on yellow soils, the soil-nesting ant community was estimated to turnover 22.3 kg/ha/yr.^[22] In a humid savanna environment, one abundant ant species, *Paltothyreus tarsatus*, was estimated to transport approximately 30 g/m²/yr of sand particles and soil aggregates. This ant species increased the concentrations of clay, carbon, iron oxides, and coarse sand in the A horizon.^[9] The amount of soil transported to the surface by *Pogonomyrmex occidentalis* in pinon-juniper woodland and ponderosa pine forest was estimated to be 650 kg/ha.^[23] Soil turnover by the ant community in New England forest soil was estimated to be over 50 kg/ha/yr. It was concluded that the translocation of B-horizon materials to the soil surface by soil-nesting ants was an important process in podzol formation in New England forest soils.^[24]

Some long-lived species of soil-nesting ants relocate their nests one or more times a year. The construction of new nests results in the transport of a volume of soil equal to the volume of galleries and chambers to the soil surface. Most of that soil originates in lower soil horizons and contributes to soil profile homogenization. The relocation of nests by some species of ants results in lower estimates of soil turnover than occurs in some environments.

SOIL WATER RELATIONS

The structure of nests of soil-nesting ants provides extensive macroporosity to the soil in which the nests are constructed. The macropores constructed by ants affect rates of infiltration and rates of percolation. In some environments, extremely high densities of nest entrances can have a dramatic effect on infiltration. In semiarid Western Australia, ant biopores were found to transmit water down the soil profile only when the soil was saturated and water was ponding on the surface.^[25] On Aeolian sand soils in Australian semiarid woodland, densities of nest entrances of funnel ants (*A. barbigula*) were estimated at 88,000 per hectare. Steady-state water infiltration on soils with nest entrances averaged 23.3 mm/min, in comparison to an infiltration rate of 5.9 mm/min on nest-entrance-free soil.^[26] In semiarid woodland of Eastern Australia on red earth soil, ponded steady-state infiltration averaged 1026 mm/hr on soil with nest entrances of *A. barbigula*, but only 120 mm/hr on soils without nest entrances.^[27] Bulk flow along nest galleries provides an important route of recharge of deep soil moisture in arid and semiarid environments.

Ant gallery macropores are not always avenues for bulk flow. In a study of a mesic Typic Quartzipsamment, there was no preferential flow down ant galleries. The lack of an effect on hydraulic conductivity was attributed to the sandy soil.^[28] In another study of a sandy soil, the estimated saturated soil matrix hydraulic conductivity of nest

burrows was approximately eight times smaller than that of the bulk sandy soil. This reduction in hydraulic conductivity was attributed to the ants' in-filling of gallery walls with fine materials.^[28]

EFFECTS ON OTHER SOIL BIOTA

Soil around relatively long-lived ant colonies may be enriched with microflora, microfauna, and mesofauna. The soils of nest disks of western harvester ants, *P. occidentalis*, are enriched with vesicular-arbuscular mycorrhizal fungi.^[29] In areas of North America dominated by the red imported fire ant, *S. invicta*, the species composition and abundance of soil yeast within mounds are altered by changes in soil properties produced by fire ants.^[30] Mound soils of *Formica aquilonia* are dominated by bacteria-feeding microfauna and have a higher microbial biomass than the surrounding soils.^[31] Species-specific differences in the effect of ants on soil microflora of mounds are related to the feeding strategies of the species and nest architecture. Three ant species, *Myrmica scabrinodis*, *Lasius niger*, and *L. flavus*, differ greatly in foraging strategies and methods of mound construction. Microbial functional diversity and evenness were higher in mound soils of *M. scabrinodis* and *L. niger* than in reference soils but were not different from reference soils in the mounds of *L. flavus*. Different functional groups of microorganisms were activated in the mounds of the different species. Carbon mineralization was higher in mound soils of all three species.^[32]

CONCLUSION

Ants contribute to heterogeneity in soil properties by the construction of subterranean nests and by accumulating organic materials in and around nests. The construction and maintenance of nests affect soil turnover, macroporosity, and mixing of soil profiles. Foraging and food processing behaviors affect soil nutrient concentrations and soil microflora and microfauna. The magnitude of the effects of ants on soils is dependent upon soil type and topographic position.

REFERENCES

1. Holldobler, B.; Wilson, E.O. *The Ants*; The Belknap Press of Harvard University Press: Cambridge, 1990.
2. Green, W.P.; Pettry, D.E.; Switzer, R.E. Formicariious pedons, the initial effect of mound-building ants on soils. *Soil Surv. Horizons* **1995**, *39* (2), 33–44.
3. Wang, D.; McSweeney, K.; Lowery, B.; Norman, J.M. Nest structure of the ant *Lasius neoniger* emery and its implications to soil modification. *Geoderma* **1995**, *66* (3–4), 259–272.

4. Lesica, P.; Kanno, P.B. Ants create hummocks and alter structure and vegetation of a Montana fen. *Am. Midl. Nat.* **1998**, *139* (1), 58–68.
5. Pire, E.F.; Torres, P.F.; Romagnoli, O.D.; Lewis, J.P. The significance of ant-hills in depressed areas of the Great Chaco. *Rev. Biol. Trop.* **1991**, *39* (1), 71–76.
6. King, T.J. Ant-hills and grassland history. *J. Biogeogr.* **1981**, *8*, 329–334.
7. Cox, G.W.; Mills, J.N.; Ellis, B.A. Fire ants (Hymenoptera: Formicidae) as major agents of landscape development. *Environ. Entomol.* **1992**, *21* (2), 281–286.
8. Luken, J.O.; Billings, W.D. Hummock-dwelling ants and the cycling of microtopography in an Alaskan peatland. *Can. Field Nat.* **1986**, *100* (1), 69–73.
9. Lobry de Bruyn, L.A.; Conacher, A.J. The role of ants and termites in soil modification a review. *Aust. J. Soil Res.* **1990**, *28* (1), 55–95.
10. Whitford, W.G.; DiMarco, R. Variability in soils and vegetation associated with harvester ant (*Pogonomyrmex rugosus*) nests on a Chihuahuan desert watershed. *Biol. Fert. Soi.* **1995**, *20*, 169–173.
11. Culver, D.C.; Beattie, A.J. Effects of ant mounds on soil chemistry and vegetation patterns in a Colorado montane meadow. *Ecology* **1983**, *64* (3), 485–492.
12. Schoereder, J.H.; Howse, P.E. Do trees benefit from nutrient rich patches created by leaf-cutting ants? *Stud. Neotrop. Fauna Environ.* **1998**, *33* (2–3), 111–115.
13. FarjiBrener, A.G.; Ghermandi, L. Influence of nests of leaf-cutting ants on plant species diversity in road verges of Northern Patagonia. *J. Veg. Sci.* **2000**, *11* (3), 453–460.
14. FarjiBrener, A.G.; Medina, C.A. The importance of where to dump the refuse: Seed banks and fine roots in nests of the leaf-cutting ants *Atta cephalotes* and *A. colombica*. *Biotropica* **2000**, *32* (1), 120–126.
15. Brener, A.G.F.; Silva, J.F. Leaf-cutting ants and forest groves in a tropical parkland savanna of Venezuela: Facilitated succession. *J. Trop. Ecol.* **1995**, *11* (4), 651–669.
16. Hulugalle, N.R. Effects of ant hills on soil physical properties of a vertisol. *Pedobiology* **1995**, *39* (1), 34–41.
17. Eldridge, D.J.; Myers, C.A. Enhancement of soil nutrients around nest entrances of the funnel ant *Aphaenogaster barbigula* (Myrmicinae) in semi-arid Eastern Australia. *Aust. J. Soil Res.* **1998**, *36* (6), 1009–1017.
18. Green, W.P.; Pettry, D.E.; Switzer, R.E. Impact of imported fire ants on the texture and fertility of Mississippi soils. *Comm. Soil Sci. Plant Anal.* **1998**, *29* (3–4), 447–457.
19. Petal, J. The influence of ants on carbon and nitrogen mineralization in drained fen soils. *Appl. Soil Ecol.* **1998**, *9* (1–3), 271–275.
20. Whitford, W.G.; Forbes, G.S.; Kerley, G.I. Diversity, spatial variability, and functional roles of invertebrates in desert grassland ecosystems. In *The Desert Grassland*; McClaran, M.P., Van Devender, T.R., Eds.; University of Arizona Press: Tucson, 1995; 152–195.
21. Eldridge, D.J.; Pickard, J. Effects of ants on sandy soils in semiarid Eastern Australia. 2. Relocation of nest entrances and consequences for bioturbation. *Aust. J. Soil Res.* **1994**, *32* (2), 323–333.
22. Lobry de Bruyn, L.A.; Conacher, A.J. The bioturbation activity of ants in agricultural and naturally vegetated habitats in semiarid environments. *Aust. J. Soil Res.* **1994**, *32* (3), 555–570.
23. Carlson, S.T.; Whitford, W.G. Ant mound influence on vegetation and soils in a semiarid mountain ecosystem. *Am. Midl. Nat.* **1991**, *126* (1), 125–139.
24. Lyford, W.H. *Importance of Ants to Brown Podzolic Soil Genesis in New England*; Harvard Forest Paper No. 7; 1963; 1–18.
25. Lobry de Bruyn, L.A.; Conacher, A.J. The effect of ant biopores on water infiltration in soils in undisturbed brushland and farmland in a semi-arid environment. *Pedobiology* **1994**, *38* (3), 193–207.
26. Eldridge, D.J. Effect of ants on sandy soils in semiarid Eastern Australia: Local distribution of nest entrances and their effect on infiltration of water. *Aust. J. Soil Res.* **1993**, *31* (4), 509–518.
27. Eldridge, D.J. Nests of ants and termites influence infiltration in a semiarid woodland. *Pedobiology* **1994**, *38* (6), 481–492.
28. Wang, D.; Lowery, B.; Norman, J.M.; McSweeney, K. Ant burrow effects on water flow and soil hydraulic properties of Sparta sand. *Soil Till. Res.* **1996**, *37* (2–3), 83–93.
29. Friese, C.F.; Allen, M.F. The interaction of harvester ants and vesicular arbuscular mycorrhizal fungi in a patchy semi-arid environment: The effects of mound structure on fungal dispersion and establishment. *Funct. Ecol.* **1993**, *7* (1), 13–20.
30. Ba, A.S.; Phillips, S.A.; Anderson, J.T. Yeasts in mound soil of the red imported fire ant. *Mycol. Res.* **2000**, *104* (8), 966–973.
31. Laasko, J.; Setälä, H. Composition and trophic structure of detrital food web in ant nest mounds of *Formica aquilonia* and in the surrounding forest soil. *Oikos* **1998**, *81* (2), 266–278.
32. Dauber, J.; Wolters, V. Microbial activity and functional diversity in mounds of three ant species. *Soil Biol. Biochem.* **2000**, *32* (1), 93–99.

Archaeology

John E. Foss

University of Tennessee, Knoxville, Tennessee, U.S.A.

Abstract

Pedologists have grown increasingly interested in work on archaeological sites. Although we aid archaeologists in understanding the soils and pedologic features they carefully excavate, we have learned a great deal about weathering rates, horizon formation, and landscape development by teamwork with archaeologists and geologists.

INTRODUCTION

Geology has had a long period of interaction with archaeology, but the use of soil investigation in archaeology has a rather short history. In 1942, Nikiforoff^[1] used the term *archeopedology* for those soil scientists working with fossil soils or Paleosols. Early studies of soils at archaeological sites were concerned mainly with soil chemical properties.^[2–4] An early book by Cromwall^[5] also played an important role in demonstrating the usefulness of soil–archaeological interactions. The past has seen a substantial increase in the multidisciplinary effort between these two sciences and have involved more subdisciplines of soil science.

ROLE OF SOIL SCIENCE IN ARCHAEOLOGY

As archaeologists become more interested in a complete understanding of the chronology and environmental history of sites, a multidisciplinary effort is absolutely necessary. Team members commonly include scientists from soil science, geology, botany, zoology, palynology, and other specializations. Soil science, especially the pedology area (i.e., the study of soil formation and classification), has been particularly active in evaluation of archaeological sites. Pedology, geology, and other earth sciences often work in the specialized field of geoarchaeology, which means using earth science principles to study archaeological sites. Fig. 1 shows a landscape of Tikal, Guatemala (Mayan site); this site is one of the many important archaeological sites that has required the expertise of pedologists to help interpret chronology and land use.^[6–9]

The study of soils and landscapes is an integral part of many archaeological investigations. Some federal and state regulations that require geologic and soil input on archaeological sites have also been responsible for including earth scientists in these studies. Publications in the 20th century indicated the interest of pedologists, geologists, and archaeologists in evaluating soils and landscapes as part of overall archaeological investigations.

Publications such as *Soils in Archaeology* edited by V.T. Holliday,^[8] *Soil Science and Archaeology* by Scudder, Foss, and Collins,^[9] Soil Science Society of America Special Pub. No. 44 on *Pedological Perspectives in Archaeological Research*,^[10] and articles in the *Proceedings of Conferences on Pedo-Archaeology*^[11,12] have raised pedologists' and archaeologists' awareness of the potential contributions of soil studies to site evaluation. The periodical *Geoarchaeology: An International Journal* has also been valuable in promoting earth science activity in archaeological investigations.

Some of the major pedologic contributions to field archaeology have included the following.^[13]

1. Determining site delimitation
2. General pedologic stratigraphy
3. Soil–landscape relationships
4. Identification of geologic parent material
5. Correlating soil morphology and archaeological levels
6. Identifying lithologic (parent material) and pedologic (soil weathering) discontinuities
7. Approximating soil age
8. Identifying Paleosols (fossil soils)
9. Contributing to the overall interpretation of site

In the 20th century, many of the above pedologic contributions to archaeology were made during the final phase of archaeological field work. Pedologists have been more involved in phase 1 activity of archaeological investigations. The early identification of major stratigraphic zones, preliminary analysis of landscape and soil age, and model of site development have resulted in more efficient archaeological excavations and interpretations.

FIELD STUDIES

Archaeological sites occur in many different geologic provinces and landscape positions. Determination of the site context is thus the most important initial stage in pedoarchaeology. Geologic maps can provide general knowledge of a region, but detailed soil surveys provide the most useful introduction to a study area. These maps produced by the



Fig. 1 Landscape view of Mayan city of Tikal, Guatemala.

National Resource Conservation Service, in cooperation with the Land Grant Institutions, are usually on a county-wide basis using an air photo base with a scale of 1:15,840–1:24,000. At this scale, there is not sufficient detail to relate the morphology of individual soil mapping units to specific horizons encountered at an archaeological site. The landscape and physiographic position of each soil mapping unit, however, are useful in preliminary analysis of archaeological sites.

The most important and informative archaeological sites occur in landscapes where sediment is added to a pre-existing surface, thereby protecting the artifacts and soil horizons. Those buried sites may occur in the following areas or situations:

1. Alluvial deposition
2. Volcanic activity
3. Eolian deposition
4. Colluvial slopes
5. Mass movement or slumping
6. Seismic areas
7. Artificial deposition or destruction

These situations provide the opportunity for soil burial (subsequently termed Paleosols) and archaeological levels. The buried surfaces (A horizons) of these Paleosols are particularly good sources of artifacts and living surfaces, when the events above took place in the Holocene. Holliday^[8] provides an excellent background in the use of paleopedology in archaeology.

Soil Morphology

Soil morphology (e.g., a detailed description of soil profiles) provides the key to understanding and interpreting soils and landscapes at archaeological sites. The unique soil morphology of a given region and site results from the weathering processes regulated by the interaction of soil-forming factors.^[14] These factors are climate, biotic, geology, topography, and length of time that the weathering processes have been operating. The morphologic properties of soils usually described in excavations and their interpretation for archaeological sites are given in Table 1. A great deal of experience

and technique is needed to provide an accurate and informative soil description. Evaluation of the age of soils, e.g., requires an integration of all the morphologic features that are detailed in Table 1. Certain soil horizons provide general age estimates based on the length of time needed for weathering processes to develop specific features (e.g., argillic horizons). As noted in Table 1, a minimal argillic horizon can form in 4000 years. Other age estimates of soil horizons have been published previously.^[9,15]

One of the most useful applications of soil morphology in archaeological site interpretation is that of distinguishing “natural” from “artificial” or “man-influenced” horizons. Some natural horizons—such as a spodic (Bh) with a dark-colored, organic-rich matrix—may appear as a buried surface or midden. Some albic E horizons could be interpreted as ash layers. Other characteristics that are related to

Table 1 Morphologic properties of soils and their interpretive value at archaeological sites.

Soil property	Useful interpretative features
Texture	Lithologic and pedologic discontinuities; classification of geologic materials; determination of argillic horizons; determine relative energy of alluvial sedimentation
Structure	Relative abundance of macropores and potential artifact movement; degree of development is an indicator of soil age; development of clay or organic coatings on argillic horizons (e.g., continuous clay coatings on pedologic faces indicate 10,000 years of development while discontinuous coatings may indicate 4,000–5,000 years of weathering in Southeastern United States)
Color	Indicator of organic matter and free iron content; classification of sediments; delineation of horizons; drainage characteristics (redoximorphic features)
Boundary	Abrupt boundary indicator of Ap (plow zone) or modern deposition; boundary becomes more diffuse with age
Consistence	Indicator of structural development, cementation, or consolidation (e.g., modern alluvium usually very friable or loose)
Clay coatings	Coatings on peds or in pores indicate the state of development and age
Carbonate	Secondary CaCO ₃ leaching, coatings, pore filling, and cementation can provide soil age estimates and climatic implications
Horizon identification	Indicates many weathering processes occurring in profile, e.g., A=organic matter accumulation; E=leached zone; Bt=argillic horizon with minimal 4000 year age; distinguish natural vs. artificial horizons; horizon thickness (solum) is a measure of length of weathering time; diagnostic horizons useful indicator for archaeological interpretation (argillic, cambic, fragipan, spodic, etc.)

Source: Adapted from Foss, Lewis, et al.^[13]

soil genesis, such as redoximorphic features (i.e., mottling or gleying), result from water table fluctuations and often cause confusion in the interpretation of color in archaeological levels. Horizons with calcium carbonate (CaCO_3) filling (Bk or Ck) have sometimes been identified as plaster filled.

LABORATORY

Laboratory soil characterization for archaeological interpretations is used to verify and supplement field morphology. Laboratory analysis without complete soil morphology is generally of minimal value for archaeological interpretation. Complete sampling of all soil horizons, columns, or archaeological levels is also important to realize the full benefit of the additional cost and labor of soil analysis.

Those laboratory analyses that are frequently applied in pedoarchaeology are organic carbon,^[16] particle size distribution,^[17] and elemental composition.^[18] Other soil analysis may include pH, electrical conductivity, mineralogy, free iron, scanning electron microscopy (SEM), energy dispersive X ray (EDAX), CaCO_3 , and micromorphology. The micromorphologic studies by Goldberg^[19] and Macphail and Goldberg^[20] have been especially useful in interpreting site stratigraphy and pedologic and geologic events.

FUTURE

Pedologists have grown increasingly interested in the work on archaeological sites, and it is likely that this trend will continue well in the future. Although we aid archaeologists in understanding the soils and pedologic features they carefully excavate, we have learned a great deal about weathering rates, horizon formation, and landscape development by teamwork with archaeologists and geologists. The use of additional techniques or applications by geoarchaeologists, such as X-ray diffraction, SEM, EDAX, electrical resistivity, ground-penetrating radar, magnetic susceptibility, and micromorphology, will improve soil interpretation work in the future. Despite advances in analytical tools, the key to archaeological site interpretation is the accurate, complete soil morphologic descriptions.

REFERENCES

1. Nikiforoff, C.C. Introduction to paleopedology. *Am. J. Sci.* **1943**, *41*, 194–200.
2. Dietz, E.F. Phosphorus accumulation in soil of an Indian habitation site. *Am. Antiquity* **1957**, *22*, 405–409.
3. Cook, S.F.; Heizer, R.F. Studies on the chemical analysis of archaeological sites. *Univ. Calif. Pub. Anthropol.* **1965**, *2*, 1–102.
4. Sokoloff, V.P.; Carter, G.F. Time and trace metals in archaeological sites. *Science* **1952**, *116*, 1–5.
5. Cromwall, I.W. *Soils for the Archeologist*; Phoenix House Ltd.: London, 1958.
6. Olson, G.W. *Soils and the Environment: A Guide to Soil Surveys and Their Application*; Chapman and Hall: New York, 1981.
7. Foss, J.E. Paleosols of pompeii and oplontis. In *Studia Pompeiana and Classics*; Curtis, R.L., Ed.; Orpheus Pub. Inc.: London, 1988; 127–144.
8. Holliday, V.T., Ed. *Soils in Archaeology, Landscape Evolution and Human Occupation*; Smithsonian Institution Press: Washington, 1992.
9. Scudder, S.J.; Foss, J.E.; Collins, M.E. Soil science and archaeology. *Adv. Agron.* **1996**, *57*, 1–76.
10. Collins, M.E.; Carter, B.J.; Gladfelter, B.G.; Southard, R.J. *Pedological Perspectives in Archaeological Research*; Soil Sci. Soc. Am. Special Pub.; Soil Science Society of America: Madison, 1995; Vol. 44; 157 pp.
11. Foss, J.E.; Timpson, M.E.; Morris, M.W., Eds. *Proceedings of the First International Conference on Pedo-Archaeology*. University of Tennessee Agricultural Experiment Station: Knoxville, 1993; Vol. 93–03; 210 pp.
12. Goodyear, A.C.; Foss, J.E.; Sassaman, K.E. Proceedings of the Second International Conference on Pedo-Archaeology. South Carolina Institute of Archaeology and Anthropology, Univ. of South Carolina, Anthro. Studies: Columbia. 1994; Vol. 10, 157.
13. Foss, J.E.; Lewis, R.J.; Timpson, M.E.; Morris, M.W.; Ammons, J.T. Pedologic approaches to archaeological sites of contrasting environments and ages. In *Proceedings of the First International Conference on Pedo-Archaeology*; Foss, J.E., Timpson, M.E., Morris, M.W., Eds.; Agr. Exp. Sta. Spec. Pub.; University of Tennessee: Knoxville, 1992; Vol. 93–03, 19–22.
14. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
15. Foss, J.E.; Lewis, R.J.; Timpson, M.E. Soils in alluvial sequences: Some archaeological implications. In *Pedological Perspectives in Archaeological Research*; Collins, M.E., Carter, B.J., Gladfelter, B.G., Southard, R.J., Eds.; Soil Sci. Soc. Am. Special Pub.: Madison, 1995; Vol. 44, 1–14.
16. Stein, J.K. Organic matter in archaeological contexts. In *Soils in Archaeology*; Holliday, V.T., Ed.; Smithsonian Institution Press: Washington, 1992; 193–216.
17. Timpson, M.E.; Foss, J.E. The use of particle-size analysis as a tool in pedological investigations of archaeological sites. In *Proceedings of the First International Conference on Pedo-Archaeology*; Collins, M.E., Carter, B.J., Gladfelter, B.G., Southard, R.J., Eds.; Univ. of Tenn. Agr. Exp. Sta. Spec. Pub.: Knoxville, 1992; Vol. 93–03, 69–80.
18. Schuldenrein, J. Geochemistry, phosphate fractionation, and the detection of activity areas at prehistoric North American sites. In *Pedologic Perspectives in Archaeological Research*; Collins, M.E., Ed.; Soil Sci. Soc. Am. Special Pub.: Madison, 1995; Vol. 44, 107–132.
19. Goldberg, P. Micromorphology, soils, and archaeological sites. In *Soils in Archaeology*; Holliday, V.T., Ed.; Smithsonian Institution Press: Washington, 1992; 145–167.
20. Macphail, R.; Goldberg, P. Recent advances in micromorphological interpretation of soils and sediments from Archaeological sites. In *Archaeological Sediments and Soils*; Barham, A.J., Macphail, R.I., Eds.; Institute of Archaeology, University College: London, 1995; 1–24.

Arid Soils

H. Curtis Monger

*Department of Agronomy and Horticulture, New Mexico State University,
Las Cruces, New Mexico, U.S.A.*

Abstract

Humans have lived on arid soils for millennia. Arid soils are important to humans for livestock grazing, irrigated agriculture, and urban development. In many arid regions of the world, human land use has resulted in desertification and diminishing groundwater supplies, both of which are increasingly important social and scientific issues as the human population increases.

INTRODUCTION

Scarcity of rain is the dominant characteristic of arid soils. While age, parent material, carbonate, and salt content may vary from arid soil to arid soil, dryness is common to all. Of the total ice-free land area on earth (130,797,000 km²), about 22% or 28,703,000 km² is occupied by soils with aridic moisture regimes.^[1]

Although arid (Latin: “aridus,” dry) signifies lack of moisture, technical definitions of arid vary. In some cases, the arid–semiarid boundary is placed at 25 cm (10 in.) of annual rainfall.^[2] In other systems, such as the Köppen–Geiger–Pohl and Meigs systems, the arid (desert)–semiarid (steppe) boundary is based on a combination of rainfall and temperature.^[3,4] Still other systems, such as those by Strahler and Soil Survey Staff, use soil moisture to define arid zones because the availability of moisture to plants is more important than annual precipitation itself.^[4,5] In all cases, however, rainfall is insufficient to maintain perennial streams. Soils in these regions are unique because relatively little water percolates deep enough to reach groundwater. As a result, carbonates, gypsum, and more soluble salts accumulate in the profiles of many arid soils.

ARID SOILS OF RIVER FLOODPLAINS

Floodplain soils along rivers that flow through arid climates were sites of several ancient and eminent civilizations. Sumerian (ca. 3600 B.C.) and later Babylonian (ca. 2000 B.C.) civilizations grew into centers of trade and government as a result of irrigated agriculture on the Tigris and Euphrates river floodplains.^[6] Likewise, soils and irrigated agriculture along the Nile of ancient Egypt, the Indus of ancient India, and the Hoang-Ho (Yellow River) of ancient China made it possible for civilizations to create notable schools, calendars, armies, mathematics, medicine, literature, philosophy, science, and art. Hohokam, Aztec,

and Inca societies emerged in arid and semiarid environments in the western hemisphere as well.^[7]

These civilizations existed because floodplain soils are well suited for irrigated agriculture if groundwater tables are sufficiently deep and salts do not accumulate. In the case of the Nile prior to dam construction, the river would rise and spill over its banks, flood the adjacent plain, deposit sediment, and leach salts.^[7] In the case of the Tigris and Euphrates, however, drainage canals were needed to carry away leached salts and with their demise soils became saline.

ARID SOILS ON UPLANDS

Most soils in arid regions are not on floodplains but occur in vast upland areas composed of the following three major landforms: mountains, piedmont slopes, and basin floors.^[8] These major landforms, in turn, are composed of smaller, component landforms. Typically, soil boundaries correspond to component landforms. In mountains, e.g., soil boundaries match the boundaries of colluvial wedges, valley fills, and pediments.^[9] On piedmont slopes, soil boundaries parallel the boundaries of alluvial fans, ballenas, and fan skirts. On basin floors, which characteristically have little topographic relief, soil boundaries generally follow landforms produced by wind, such as deflation blowouts, dunes, and eolian plains, or landforms produced by pluvial lakes, such as lake plains, playas, and beach plains.

Of the five soil-forming factors (climate, time, biota, topography, and parent material), climate is the defining factor of arid soils although time is an important factor as well. The impact of time on arid soils is revealed by the carbonate and clay accumulations in soils of progressively older geomorphic surfaces (Fig. 1). Carbonate in non-gravelly soils, e.g., progresses from carbonate filaments in middle Holocene soils to carbonate nodules in late Pleistocene soils to carbonate-indurated horizons in middle Pleistocene soils.^[10] Clay likewise accumulates with time

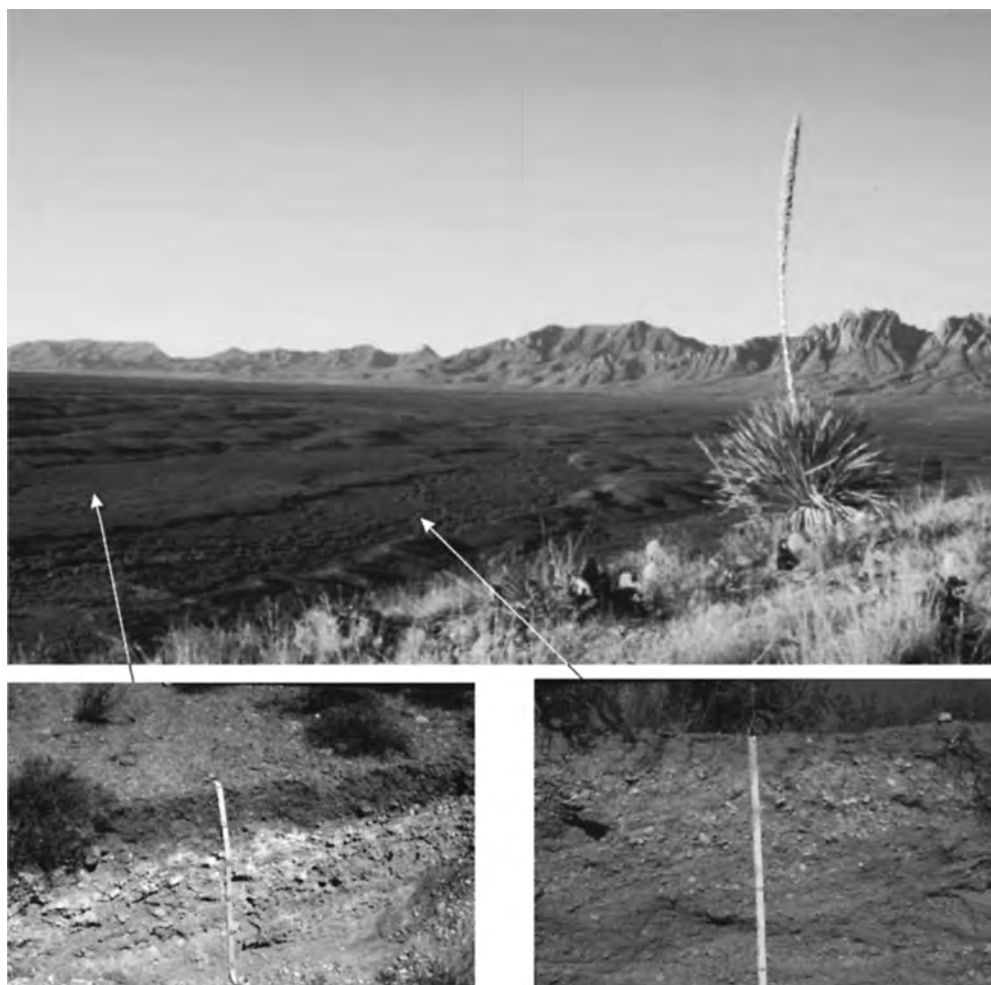


Fig. 1 Desert piedmont slope rising to a mountain chain in Southern New Mexico. Progressively older geomorphic surfaces with progressively greater soil development are the typical features of piedmont slopes. The younger soil on the right (about 3000 years old) has a small amount of carbonate (white zone in profile). The older soil to the left (25,000–150,000 years old) has substantially more carbonate. In addition, the older soil has an argillic horizon overlying the carbonate horizon.

to form argillic horizons. However, the correlation of clay accumulation with time is less robust than that of carbonate accumulation with time because many ancient soils that have calcretes do not have argillic horizons.^[11] This indicates that argillic horizons are more vulnerable to obliteration by erosion and bioturbation than calcic or petrocalcic horizons.

Arid soils are unique not only because carbonate, gypsum, and soluble salt accumulate, but also because many have vesicular A horizons covered by desert pavement^[12] or microbiotic crust.^[13] In addition, inadequate water and nitrogen suppress biomass production on arid soils to about one-tenth the biomass of temperate forest soils.^[14] Nevertheless, soil animals such as rodents, ants, and termites are common. Ants, e.g., can transfer 80 g m⁻² of desert soil to the land surface per year, which is as much as ants transfer in more mesic environments.^[15]

TYPES OF ARID SOILS

The main criterion for the classification of arid soils is soil dryness, or the aridic (torric) moisture regime,

which is defined as soils too dry for agricultural crops unless irrigated.^[16] Further taxonomic subdivisions are based on diagnostic horizons. In contrast to the notion that arid soils are poorly developed, as written in some soil science books, many arid soils are strongly developed with a variety of diagnostic subsurface horizons.^[17] These horizons include the argillic, natric, salic, gypsic, petrogypsic, calcic, and petrocalcic horizons and the duripan.^[5] Diagnostic surface horizons include the ochric epipedon with minor occurrences of the mollic and anthropic epipedons.

Arid soils that have diagnostic subsurface horizons are generally classified as Aridisols. These include many of the older soils on piedmont slopes, basin floors, and mountain uplands. Various types of Aridisols occur on the landscape because of lateral changes in the particle size, truncation of diagnostic horizons, degradation of diagnostic horizons, moisture heterogeneity across the landscape, and age differences, which can range from Historical to Pliocene within small geographical areas.^[9,18]

Arid soils that lack diagnostic subsurface horizons are generally classified as Entisols, which fall into the azonal

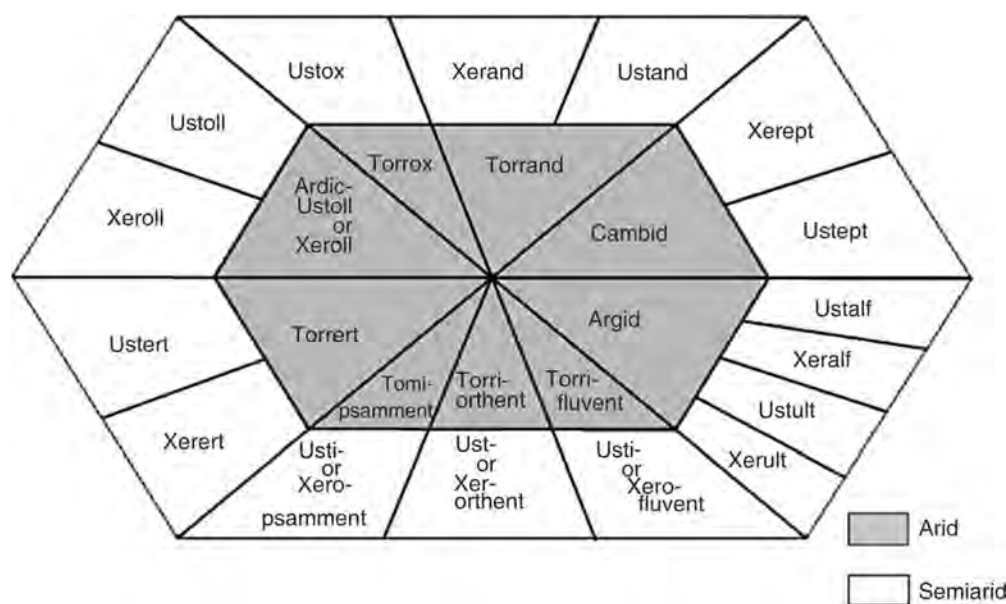


Fig. 2 The illustration of Soil Taxonomy Suborders and Great Groups that have aridic moisture regimes (shaded) and their moister counterparts that have ustic and xeric moisture regimes. **Source:** From Soil Survey Staff.^[5]

concept of Sibirtsev.^[19] These include many of the younger soils on floodplains, dunes, and erosional surfaces. In Soil Taxonomy, floodplain soils are mainly classified as fluvents or, more specifically, torrifluvents.^[5] Arid soils associated with dunes are commonly torripsamments, and those associated with erosional surfaces are commonly torriorthents.

Aridisols (14,942,000 km²) and Entisols (12,682,000 km²) are the dominant soil types in arid regions although other soil types include Vertisols (889,000 km²) and Oxisols (31,000 km²) and very minor amounts of Mollisols, Andisols, Histosols, and Spodosols.^[1] Arid soils grade into semiarid soils across the following three climatic transects: laterally into wetter regions, upslope into wetter climates at higher elevations, or downslope into run-in areas with wetter microclimates. Taxonomically, changes in soil types from dry region aridic to wetter region ustic or xeric moisture regimes are expressed at the Suborder and Great Group level (Fig. 2). Linked to this climatic transition is a progressive change in vegetation—desert shrublands give way to grasslands that in turn give way to woodlands.

Also across this transition, soils have progressively deeper carbonate horizons. In the Chihuahuan Desert, for instance, carbonate zones are 50 cm deep at 230 mm of annual rainfall and 100 cm deep at 320 mm of annual rainfall.^[20] Likewise, gypsum zones progressively deepen from about 50 cm depth at 150 mm of annual rainfall to about 100 cm depth at 250 mm annual rainfall.^[21] Accompanying an increase in rainfall is an increase in soil organic matter. Although the amount of organic matter depends on the clay content, organic matter ordinarily increases from less than 0.5% in A horizons of arid shrubland soils to 2–5% in semiarid and subhumid grassland soils.^[22]

ECOLOGICAL SIGNIFICANCE

Biodiversity in arid regions is linked to habitat diversity. Habitat diversity, in turn, is created by various microclimates caused by topographic factors and soil properties.^[23]

Thus, soils help mold and are molded by ecosystems. In many arid regions, such as the Southwestern United States, soils have been impacted by ecosystem changes of the Holocene and late Pleistocene when wetter climates alternated with drier climates.^[24–27] According to this model, landscape stability was greater during wetter climates because denser vegetative cover reduced erosion. With reduced erosion, soil formation occurred. In contrast, instability was greater during drier climates because sparse vegetative cover gave rise to more bare ground and increased erosion. With increased erosion, soil formation was inhibited. This oscillation between stability and instability is recorded as stacked sequences of buried Paleosols in depositional environments and as stepped sequences of fan terraces in areas that grade to fluctuating river base levels.

Globally, arid soils affect atmospheric dust, rain chemistry, ocean fertilization, albedo, denitrification, and the carbon cycle as both sinks and sources of carbon dioxide.^[28,29] Carbonate-carbon, for instance, is the second largest terrestrial carbon pool, totaling approximately 50–60 Pg C in the dryland zones of the United States^[30] and approximately 750–950 Pg C in the dryland zones of the world.^[31]

Humans have lived on arid soils for millennia. In fact, the oldest known hominid tools are in arid East Africa and date back 2.5 million years.^[32] Arid soils are important to humans for livestock grazing, irrigated agriculture, and urban development. In many arid regions of the world, human land use has resulted in desertification and diminishing groundwater supplies, both of which are increasingly important social and scientific issues as human population increases.

ACKNOWLEDGMENT

The author is very grateful to Haiyang Xing and Marco Inzunza for making the figures and Rebecca Krammer for reviewing the manuscript.

REFERENCES

- Wilding, L.P. Introduction: General characteristics of soil orders and global distributions. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; E175–E182.
- Bull, W.B. *Geomorphic Responses to Climatic Change*; Oxford University Press: New York, 1991; 326 pp.
- Dick-Peddie, W.A. Semiarid and arid lands: A worldwide scope. In *Semiarid Lands and Deserts*; Skujins, J., Ed.; Marcel Dekker: New York, 1991; 3–32.
- Strahler, A.N.; Strahler, A.H. *Modern Physical Geography*, 3rd Ed.; Wiley: New York, 1987; 544 pp.
- Soil Survey Staff. *Soil Taxonomy—A Basic System of Soil Classification for Making and Interpreting Soil Surveys, 2nd Version*; USDA Agriculture Handbook Number 436; U.S. Government Printing Office: Washington, 1999.
- Durant, W. *Our Oriental Heritage*; Simon and Schuster: New York, 1935; 1049 pp.
- Dregne, H.E. *Soils of Arid Regions*; Elsevier: Amsterdam, 1976; 237 pp.
- Peterson, F.F. *Landforms of the Basin and Range Province Defined for Soil Survey*; Nevada Agricultural Experiment Station, Tech. Bull. 28; University of Nevada: Reno, 1981; 52 pp.
- Gile, L.H.; Hawley, J.W.; Grossman, R.B. *Soils and Geomorphology in the Basin and Range Area of Southern New Mexico—Guidebook to the Desert Project*; Memoir 39; New Mexico Bureau of Mines and Mineral Resources: New Mexico, 1981; 222 pp.
- Gile, L.H.; Peterson, F.F.; Grossman, R.B. Morphology and genetic sequences of carbonate accumulation in desert soils. *Soil Sci.* **1966**, *101*, 347–360.
- Gile, L.H. Eolian and associated pedogenic features of the Jornada Basin floor, southern New Mexico. *Soil Sci. Soc. Am. J.* **1999**, *63*, 151–163.
- McFadden, L.D.; Wells, S.G.; Jercinovich, M.J. Influences of eolian and pedogenic processes on the origin and evolution of desert pavements. *Geology* **1987**, *15*, 504–508.
- Kidron, G.J.; Yaalon, D.H.; Vonshak, A. Two causes for runoff initiation on microbiotic crusts: Hydrophobia and pore clogging. *Soil Sci.* **1999**, *164*, 18–27.
- Ludwig, J.A. Primary productivity in arid lands: Myths and realities. *J. Arid Environ.* **1987**, *13*, 1–7.
- Whitford, W.G.; Schaefer, D.; Wisdom, W. Soil movement by desert ants. *Southwest. Nat.* **1986**, *31*, 273–274.
- Smith, G.D. *The Guy Smith Interviews: Rationale for Concepts in Soil Taxonomy*; Soil Management Support Service Monograph No. 11; U.S. Government Printing Office: Washington, 1986.
- Ahrens, R.J.; Eswaran, H. The international committee on aridisols: Deliberations and rationale. *Soil Surv. Horizons.* **2000**, *41*, 110–117.
- Gile, L.H. Causes of soil boundaries in an arid region; I. Age and parent materials. *Soil Sci. Soc. Am. Proc.* **1975**, *39*, 316–323.
- Sibirtsev, N.M. *Selected Works, Soil Science*; Issued in Translation by the Israel; Program for Scientific Translation: Jerusalem, 1966; Vol. 1, 354 pp.
- Gile, L.H. Holocene soils and soil-geomorphic relations in a semi-arid region of southern New Mexico. *Quat. Res.* **1977**, *7*, 112–132.
- Cooke, R.; Warren, A.; Goudie, A. *Desert Geomorphology*; UCL Press: London, 1993; 526 pp.
- Birkeland, P.W. *Soils and Geomorphology*, 3rd Ed.; Oxford University Press: New York, 1999; 430 pp.
- McAuliffe, J.R. Landscape evolution, soil formation, and ecological patterns and processes in Sonoran Desert bajadas. *Ecol. Monogr.* **1994**, *64*, 111–148.
- Ruhe, R.V. Age of the Rio Grande Valley in southern New Mexico. *J. Geol.* **1962**, *70*, 151–167.
- Gile, L.H.; Hawley, J.W. Periodic sedimentation and soil formation on an alluvial-piedmont in southern New Mexico. *Soil Sci. Soc. Am. Proc.* **1966**, *30*, 261–268.
- Monger, H.C.; Cole, D.R.; Gish, J.W.; Giordano, T.H. Stable carbon and oxygen isotopes in quaternary soil carbonates as indicators of ecogeomorphic changes in the northern Chihuahuan Desert, USA. *Geoderma* **1998**, *82*, 137–172.
- Buck, B.J.; Monger, H.C. Stable isotopes and soil-geomorphology as indicators of Holocene climate change, northern Chihuahuan Desert. *J. Arid Environ.* **1999**, *43*, 357–373.
- Schlesinger, W.H.; Reynolds, J.F.; Cunningham, G.L.; Huenneke, L.F.; Jarrell, W.M.; Virginia, R.A.; Whitford, W.G. Biological feedbacks in global desertification. *Science* **1990**, *247*, 1043–1048.
- Schlesinger, W.H. *Biogeochemistry: An Analysis of Global Change*, 2nd Ed.; Academic Press: New York, 1997.
- Monger, H.C.; Martinez-Rios, J.J. Inorganic carbon sequestration in grazing lands. In *The Potential of U.S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect*; Follette, R.F., Kimble, J.M., Lal, R., Eds.; CRC Press: Boca Raton, 2001; 87–118.
- Eswaran, H.; Reich, P.F.; Kimble, J.M.; Beiroth, F.H.; Padmanabhan, E.; Moncharoen, P. Global carbon sinks. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Ed.; CRC Press: Boca Raton, 2000; 15–26.
- Ambrose, S.H. Paleolithic technology and human evolution. *Science* **2001**, *291*, 1748–1753.

Arsenic: Southeast Asia

Mohammad Mahmudur Rahman

Ravi Naidu

*Global Centre for Environmental Remediation (GCER), University of South Australia,
Callaghan, New South Wales, Australia*

Abstract

Groundwater contaminated with arsenic (As) and its associated health dangers for people have been reported worldwide, particularly in the countries of Southeast Asia. This study presents a literature-based overview of the As contamination in this region where groundwater is contaminated with naturally occurring As. As a consequence of using contaminated water for irrigation, As is also deposited in agricultural soils and potentially contaminates food crops grown in this region. To minimize the risk of As contamination from affected groundwater and soil, simple, practical, and cost-effective solutions need to be implemented urgently.

INTRODUCTION

Arsenic (As) is a metalloid widely distributed in the earth's crust. It is recognized as a toxic element and has been classified as a human carcinogen affecting skin, lungs, and other internal organs.^[1] Elemental As is a member of Group VA on the periodic table, with nitrogen, phosphorus, antimony, and bismuth. Its atomic number is 33, and it has an atomic mass of 74.91. It can exist in four valence states: -3 , 0 , $+3$, and $+5$. Under reducing conditions, arsenite [As(III)] is the dominant form; arsenate is generally the stable form found in oxygenated environments.^[1] Arsenic and its compounds occur in crystalline, powder, and amorphous or vitreous forms. It usually occurs in trace quantities in all rock, soil, water, and air.^[1]

The bioavailability and toxicity of As depend on its chemical form. In general, the inorganic forms of As are much more toxic than the organo-As forms. Organo-As species are common in marine organisms. The major As compound in marine animals is arsenobetaine. Two minor As compounds (tetramethylarsonium and arsenocholine) are commonly found in marine animals. However, some organic As species such as monomethylarsonous acid (MMA[III]) and dimethylarsinous acid (DMA[III]), which have been identified in urine, are also very toxic.^[1]

The occurrence of elevated concentrations of As has been detected in numerous aquifers around the world where As concentrations in groundwater exceeded the World Health Organization (WHO)-recommended value of As ($10 \mu\text{g/L}$) in drinking water. Significant As exposure mostly occurs through drinking As-contaminated water and food crops irrigated with As-contaminated water. Inorganic As exposure at chronic levels adversely impacts on human health, causing skin disorders, cardiovascular disease,

neurological complications, reproductive disorders, respiratory effects, diabetes mellitus, as well as various types of cancers including skin, lung, bladder, and kidney.^[2,3] The Southeast Asian countries that are affected by As in groundwater include Bangladesh, several states of India (i.e., West Bengal, Bihar, Uttar Pradesh, Jharkhand, Assam, Chhattisgarh, and Manipur), Nepal, Myanmar, Pakistan, Vietnam, Lao People's Democratic Republic (Lao PDR), Cambodia, lowlands of Sumatra in Indonesia, and several provinces of China.^[4–6] Several European countries, e.g., Spain, Portugal, and Hungary, are also As-contaminated.^[7] In South and North America, the most notable problems with As in groundwater have been detected in several areas of the United States of America, Canada, Mexico, Nicaragua, Ecuador, Chile, Argentina, Peru, Brazil, and Uruguay.^[8] In the Ganges Delta region of Bangladesh and West Bengal, As in groundwater has emerged as the largest environmental health disaster; it has been estimated that more than 100 million people are at risk from As poisoning. A study conducted by the World Bank found that about 0.7 million people have so far been affected by As-related diseases in South and East Asia.^[9] In this entry, we present a literature-based overview of major As contamination in groundwater and soils from various regions located in Southeast Asian countries.

EXTENT AND SEVERITY OF As IN GROUNDWATER OF BENGAL DELTA

Bangladesh

In a nationwide survey conducted by the British Geological Survey along with the Department of Public Health

Engineering (DPHE), Bangladesh, 3534 tube well samples were analyzed from all over Bangladesh except Chittagong Hill Tract. It was reported that 46% and 27% of the samples exceeded 10 and 50 $\mu\text{g/L}$ of As, respectively.^[10] It was also estimated that 57 million and 35 million people could be drinking As-contaminated water with concentration above 10 and 50 $\mu\text{g/L}$, respectively.^[10] Reports emerged of As concentrations ranging from 0.07 to 640 $\mu\text{g/L}$, based on 112 groundwater samples from throughout Bangladesh.^[11] Another study analyzed 6500 tube wells samples from the Araihaazar area of Narayanganj in 2000–2001 and concluded that half of the samples contained As above 10 $\mu\text{g/L}$ and one-quarter contained As above 50 $\mu\text{g/L}$.^[12]

In a large study, it was reported that 27.2% and 42.1% of the tube wells had As above 50 and 10 $\mu\text{g/L}$, respectively, based on an analysis of 52,202 water samples from hand tube wells from all 64 districts of Bangladesh since 1996.^[13] This involved employing flow injection–hydride generation atomic absorption spectrometry method for As measurement.^[13] An analysis of scenarios of As status including As-related health effects in Bangladesh was reported based on the authors' 21-year study.^[14]

A study showed that chronic As exposure through drinking water was associated with an increase in the mortality rate based on the Cox proportional hazards model to estimate hazard ratios mortality.^[15] In that study, authors interviewed in person and clinically assessed 11,746 participants (aged 18–75 years) from the Araihaazar area. The study found that 407 deaths were ascertained between October 2000 and February 2009. Multivariate adjusted hazard ratios for all-cause mortality in a comparison of As at concentrations of 10.1–50.0, 50.1–150.0, and 150.1–864.0 $\mu\text{g/L}$ with at least 10.0 $\mu\text{g/L}$ in well water were 1.34 [95% confidence interval (CI), 0.99–1.82], 1.09 (95% CI, 0.81–1.47), and 1.68 (95% CI, 1.26–2.23), respectively.^[15]

West Bengal

One study reported that 48.1% had As above 10 $\mu\text{g/L}$ and 23.8% had As above 50 $\mu\text{g/L}$, based on 140,150 water samples analysis from tube wells in all 19 districts, and the estimated population drinking As-contaminated water with concentration above 10 and 50 $\mu\text{g/L}$ were 9.5 and 4.2 million, respectively.^[16] Another large-scale survey was carried out by the Public Health Engineering Department (PHED), Government of West Bengal, with the support of United Nations Children's Emergency Fund (UNICEF). This involved eight districts (i.e., North 24 Parganas, South 24 Parganas, Murshidabad, Nadia, Bardhaman, Hawrah, Hooghly, and Malda) and water was analyzed using the silver diethyldithiocarbamate method.^[17] Altogether, water samples from 132,262 tube wells were analyzed, and 25.5% of them contained As greater than 50 $\mu\text{g/L}$ and 57.9% greater than 10 $\mu\text{g/L}$.^[17]

ARSENIC CONTAMINATION IN GROUNDWATER IN OTHER REGIONS OF SOUTHEAST ASIA

Other Areas in India

In a study, it was reported that around 20,000 tube well water samples were analyzed from 12 districts of Bihar state; 32.7% samples contained As above 10 $\mu\text{g/L}$, while 17.8% had As above 50 $\mu\text{g/L}$.^[18] It was also estimated that around 3.1 million and 1.7 million people are potentially at risk from consuming As-contaminated water from Bihar.^[18] In Uttar Pradesh, around 5500 tube well water samples were examined from five districts; 44.4% samples contained As above 10 $\mu\text{g/L}$, and 26.2% had As above 50 $\mu\text{g/L}$.^[18] Approximately 0.6 million people are exposed to As-contaminated water with concentration above 50 $\mu\text{g/L}$.^[18] It has been reported for Jharkhand that around 3500 tube well water samples investigated from five districts and 36.1% of them contained As above 10 $\mu\text{g/L}$; furthermore, 15.4% had As above 50 $\mu\text{g/L}$.^[18] In Assam and Manipur, 1590 and 628 tube well water samples were analyzed. About 42.3% and 63.3% of samples had As above 10 $\mu\text{g/L}$, while 19.1% and 23.2% had As above 50 $\mu\text{g/L}$ from Assam and Manipur, respectively.^[18]

The quality of groundwater with respect to As from 33 different sources in the predominantly rural area of Golaghat subdivision in Golaghat district of Assam (India) was determined during the premonsoon (February) and postmonsoon (October) seasons of 2008.^[19] The As concentrations varied from not detectable to 220 $\mu\text{g/L}$ during premonsoon and from 1 to 187 $\mu\text{g/L}$ during postmonsoon. The high values of As were observed in Forkating, Fatual (220 $\mu\text{g/L}$), and Bahbari (187 $\mu\text{g/L}$) in premonsoon and postmonsoon seasons, respectively.^[19]

China (Including Taiwan)

The distribution and chemical characteristics of high-As groundwater, hydrogeological settings of high-As groundwater aquifers, and the geochemical processes controlling As distribution in typical areas of China were reviewed (the study by Guo et al.^[20] and references therein). It was reported that out of 34 provinces, high As was reported in the groundwater of 20 provinces. Generally, there are two types of areas where high-As groundwater naturally occurs. One is the arid–semiarid inland basins, and the other is the river deltas.^[20] Concentrations of As above 50 $\mu\text{g/L}$ have been detected in groundwater in aquifers of the Hohhot basin (depth: 4–400 m; As range: <1–1860 $\mu\text{g/L}$) and the Hetao basin (depth: 10–30 m; As range: <1–969 $\mu\text{g/L}$) of Inner Mongolia.^[20] The ranges of As in aquifers of the Datong (depth: 20–50 m), the Taiyuan (depth: 50–200 m), and the Yuncheng (depth: 20–300 m) basins of Shanxi Province were reported as <1–1300, 0.1–116, and 0.1–500 $\mu\text{g/L}$, respectively. The concentrations of As in aquifers of Songnen basin in Jilin

Province and Yinchuan basin in Ningxia Province ranged from <1 to 360 µg/L and <0.1 to 200 µg/L, respectively. It was also reported that the aquifer of Dzungaria basin of Xinxiang Province had As concentration between 70 µg/L and 850 µg/L.^[20]

Elevated concentrations of As have been found in several river deltas, e.g., Yangtze River delta of Shanghai and Nanjing, Yellow River delta of Shandong Province, Pearl River delta of Guangdong Province, Chianan alluvial plain of Taiwan, and Lanyang alluvial plain of Taiwan (the study by Guo et al.^[20] and references therein). The concentrations of As in river deltas varied from 0.22 to 3590 µg/L. High levels of As contamination up to 2160 µg/L have also been detected in the mountain areas such as Qinghai, Gansu, Sichuan, and Yunnan.^[20]

Indonesia

A study analyzed 97 samples of five distinct groups of groundwater composition that reflect a variety of geological sources and/or chemical conditions (pH and redox) from Sumatra.^[6] Arsenic concentrations showed a heterogeneous distribution over the study area, with 12 out of 97 samples from the tube wells exceeding the WHO guideline value for drinking water (10 µg/L). Elevated As levels were mostly present in the groundwater of tube wells located in Holocene deposits. The lowest concentrations were measured in groundwater (oxidizing conditions), while the highest values occurred in areas with Holocene swamp and alluvial sedimentation. The average (range) As concentrations in five areas detected were 5.8 (<0.1–34), 0.5 (<0.1–3.8), 7.2 (<0.1–35), 9.8 (0.2–65), and 2.6 (<0.1–22) µg/L, respectively.^[6]

Lao PDR

Arsenic concentrations in tube well water collected from several provinces (e.g., Attapeu, Bolikhamxay, Champasak, Savannakhet, Saravane, and Vientiane) of Lao PDR were determined, and the results showed that As-contaminated areas were mostly floodplains in the central and southern parts of the country. Total As concentrations ranged from <0.5 to 278 µg/L, with over half exceeding the WHO guideline concentration of 10 µg/L and As(III) were present in 46% of water samples.^[21]

Myanmar

A study confirmed the presence of elevated levels of As in groundwater of Myanmar.^[22] Altogether 55 wells water were analyzed by visual field kit measurements, and the results reveal that a total of 23 wells met the WHO guideline concentration for As (10 µg/L), another 9 wells had As concentrations up to 50 µg/L, and 23 wells recorded >50 µg/L of As. It was also reported that elevated concentrations of As (50–630 µg/L) were determined in wells up

to 60 m deep. Another study assessed samples from 18 drinking water wells (12 in Myingyan city and 6 in nearby Tha Pyay Thar village) of Myanmar for As and reported that the ranges of As concentrations in Myingyan city and nearby Tha Pyay Thar village were 1–22 and 10–134 µg/L, respectively.^[23]

Nepal

In 2003, it was reported that 23% of samples exceeded As >10 µg/L, and 5% were above the “Nepal Interim Arsenic Guideline” concentration of 50 µg/L, based on an analysis of 15,000 tube wells.^[24] It was estimated that around 0.5 million people in Terai were at risk of As poisoning (>50 µg/L).^[24] An analysis of 42 deep and 13 shallow groundwater samples from Kathmandu valley showed that As levels in shallow groundwater are usually <10 µg/L, but 52% of the deep groundwater had As >10 µg/L.^[25] Screening of tube wells in the Terai region of Nepal started in 2001, and several organizations tested 737,009 groundwater samples from 25 districts.^[26] Of these, 82.6% contained As ≤10 µg/L, 7.6% had As >10–50 µg/L, and 2.6% had As >50 µg/L.^[26] It was also reported that in 2011 about 2.29 million and 0.37 million people are expected to drink water with As concentrations of 10–50 and >50 µg/L, respectively.^[26] In 2011, it was reported that out of 24,674 tested wells in the Nawalparasi district, approximately 7896 wells (32%) had As concentrations greater than 10 µg/L, and 3676 wells (14.9%) had As concentrations greater than 50 µg/L.^[27]

Pakistan

The concentration of As in groundwater (n = 153) ranged from 13 to 106 µg/L in Jamshoro district of Pakistan.^[28] The study concluded that As may originate from coal combustion at brick factories and power generation plants, and it was mobilized promotionally by the alkaline nature of the understudy groundwater samples.^[28]

Thailand

Groundwater samples were collected from 20 different sampling points between August 2009 and February 2010 in two areas (i.e., Amphoe Khemmarat and Ubon Ratchathani) of Thailand, and the concentration of As in the groundwater of Amphoe Khemmarat ranged from 0.7 to 20.2 µg/L with an average value of 4.79 µg/L.^[29]

Vietnam and Cambodia

Groundwater samples were analyzed during 2007–2008 in the Mekong River delta in Vietnam and revealed that 26% of samples had above 10 µg/L of As.^[30] Concentrations of As ranging from <0.1 to 1351 µg/L and elevated levels of As were found in groundwater at sampling sites close to the

Mekong River and in wells less than 60–70 m deep. Sediment samples from An Giang and Dong Thap had 18 and 38 mg/kg, respectively, of As concentrations. Arsenic sediment occurred mainly in the poorly crystalline iron (Fe) oxide phases.^[30] Concentrations of As in the groundwater collected at Gia Lam district and Thanh Tri district, suburban areas of Hanoi, Vietnam, ranged from <0.10 µg/L to 330 µg/L, with about 40% of these exceeding the WHO drinking water guideline concentration of 10 µg/L.^[31] One study presented an overview of groundwater As pollution in the Mekong River delta; As concentrations ranged from 1 to 1610 µg/L in Cambodia (average: 217 µg/L) and 1 to 845 µg/L in southern Vietnam (average: 39 µg/L), respectively.^[32] Of the studied wells, 48% had As concentrations >10 µg/L in shallow aquifers in the south of Phnom Penh situated at the Cambodian floodplain.^[33] Another study investigated the concentrations of As in groundwater from three villages in the Kandal Province of Cambodia and reported that As concentrations ranged from 6.64 µg/L to 1543 µg/L, with average and median concentrations of 552 and 353 µg/L, respectively.^[34] About 86% out of 15 samples contained As concentrations exceeding the WHO drinking water guideline concentration of 10 µg/L.^[34]

Arsenic concentrations in groundwater ranged from 0.1 µg/L to 1340 µg/L, with 37% of the studied wells exceeding the WHO guideline As concentration of 10 µg/L, based on its transnational groundwater survey of the 62,000 km² Mekong River delta floodplain (southern Vietnam and bordering Cambodia).^[35] Arsenic contamination in groundwater in four villages (i.e., Vinh Tru, Bo De, Hoa Hau, and Nhan Dao) of Ha Nam Province in northern Vietnam revealed that concentrations of As in three of these villages (i.e., Vinh Tru, Bo De, and Hoa Hau) significantly exceeded the Vietnamese drinking water standard for As (10 µg/L) with average concentrations of 348, 211, and 325 µg/L, respectively.^[36] It was also determined the distribution of As in the groundwater from two regions (i.e., Tien Giang Province and Dong Thap Province) of the Mekong River delta and reported that the concentration of total As in the groundwater, which is used for the drinking water supply, ranged from 0.9 to 321 µg/L, and 27% of the shallow well water samples exceeded the WHO provisional guideline concentration of 10 µg/L.^[37]

ARSENIC IN SOILS

Arsenic concentrations in soils were reported to be as high as 57 mg/kg in four different districts (i.e., Chapai Nawabganj, Kushtia, Pabna, and Jamalpur) in Bangladesh.^[38] The range of As in paddy soil was 4.7–52 mg/kg with an average of 17.2 mg/kg (mean As concentrations in highland and medium highland soils were 14 and 21 mg/kg, respectively), based on 263 paddy soils (both highland and medium highland) irrigated with As-contaminated water (average: 130 µg/L; range: <2–458 µg/L) from Tala of

Satkira, Bangladesh.^[39] The study concluded that soil As in highland paddy soils was related to As and Fe levels of irrigation water and duration of contaminated wells in operation.^[39] A study reported that out of 31 soil samples evaluated from various locations of As-contaminated districts in Bangladesh, results showed that all soils contained As below 10 mg/kg.^[40] In 2003, it was reported that As levels in surface paddy soils (0–15 cm) of Bangladesh varied between 3.1 and 42.5 mg/kg.^[41] The highest levels of As were reported in districts to the west of Bangladesh, particularly Meherpur, Chuadanga, Kushtia, and Rajbari, all having one or more soil samples with As >30 mg/kg. Concentrations of As in soil in the range of 20–30 mg/kg are distributed throughout the central belt of Bangladesh.^[41] The concentrations of As in soils from the command area of the tube wells situated in southwestern Bangladesh ranged from 4.5 mg/kg to 68 mg/kg; more than 85% of these soils contained <20 mg/kg of As.^[42] At the Marua and Samta villages in the Jessore district of Bangladesh, the As concentrations ranged from 1.2 mg/kg to 14.8 mg/kg in surface soil.^[43]

A study showed that the average As concentrations were 33.15 and 6.10 mg/kg in the soils collected from Faridpur and Dhamrai, respectively.^[44] The average As concentration in Faridpur was more than 3 times higher than the world standard (10 mg/kg) for soil.^[44] Average As concentration was 13.3 mg/kg at Samta village, one of the As-rich areas of Bangladesh.^[45] A study determined As contamination of 30 selected soils at three depths (i.e., 0–15, 15–30, and 30–45 cm) in 10 thanas of four As-affected districts (i.e., Gopalganj, Faridpur, Rajbari, and Madaripur) of Bangladesh.^[46] The ranges of As in the soils of all four districts were 3.96–25.09, 0–38.67, 1.32–36.99, and 1.32–38.19 mg/kg, respectively.^[46]

The average concentration of As in rice field (irrigated with As-rich groundwater) was 101 mg/kg in West Bengal, India.^[47] The average As concentrations in soils was 5.31 mg/kg (range: 2.68–6.79 mg/kg) and 10.7 mg/kg (range: 3.34–31.6 mg/kg) collected from fallow lands (n = 86) and agricultural lands (n = 30) of Domkal block in the Murshidabad district of West Bengal.^[48] The mean As concentrations in surface soils, soils from the roots of plants, soils below ground level (0–30 cm), and all soils from the four agricultural lands were 14.2 mg/kg (range: 9.5–19.4 mg/kg, n = 99), 13.7 mg/kg (range: 7.56–20.7 mg/kg, n = 99), 14.8 mg/kg (range: 8.69–21 mg/kg, n = 102), and 14.2 mg/kg (range: 7.56–21 mg/kg, n = 300), respectively.^[49]

Soil samples (n = 73) from the Singe Tsangpo (upstream of the Indus River), Yarlung Tsangpo (upstream of the Brahmaputra River), and other drainage basins in Tibet were analyzed in 2008.^[50] The average As concentration in soils was 44 ± 27 mg/kg (n = 28; range: 12–84 mg/kg) for the Singe Tsangpo, and it was 30 ± 34 mg/kg (n = 21; range: 6–173 mg/kg) for the Yarlung Tsangpo.^[50] The soil As levels from an alluvial plain of the Mangyeong River,

Korea, ranged from 3.9 to 9.9 mg/kg.^[51] The total soil concentrations of As in soil collected from paddy fields in the suburban areas of some major cities in Fujian Province, southeast China, ranged from 1.29 to 25.28 mg/kg with an average of 6.04 mg/kg.^[52] The total As concentration in 11 tested soil samples ranged from 11.8 to 112 mg/kg from the northern part of the Chianan Plain in southwestern Taiwan.^[53] The As concentrations in all soil samples except one notably exceeded the background level of As (18 mg/kg) in the rural soils of Taiwan. Another study found total As content ranging from 7.92 to 12.7 mg/kg in surface soils collected from the coastal part of Chianan Plain in southwestern Taiwan.^[54]

Total As in the soils collected from 3 As-contaminated areas (18 samples per area) at different distances (0, 400, and 800 m) in Iran ranged from 105.4 to 1500 mg/kg.^[55] The As concentrations of 0–20 cm and the 20–40 cm soil layers were comparable, and there was no consistent decline in soil As concentration when distance increased.^[55]

A study measured Arsenic levels in agricultural soils (surface: 0–5 cm and subsurface: 20–25 cm) located in the Red River delta in northern Vietnam.^[56] Soil samples were collected from 18 paddy and 6 upland fields on both sides of the river. The total As contents of approximately 80% of the surface paddy and upland soils exceeded the maximum allowable limit for Vietnamese agricultural soils (12 mg/kg). Arsenic contents higher than 35 mg/kg were found in soils from the Hung Yen and Ha Nam Provinces, where high As levels in the groundwater have also been reported.^[56] Another study investigated the concentration of total As in soil (0–15 cm depth) from 15 selected locations in Phumi Khleang, Kandal Province, Cambodia. The As concentrations in soils varied from 5.34 to 27.81 mg/kg.^[57] A study evaluated the As content of soils from five provinces (i.e., Kandal, Prey Veng, Battambang, Banteay Meanchey, and Kampong Thom) in Cambodia's rice-growing regions and reported that the total soil As concentrations ranged from 0.8 µg/g to 18 µg/g. Significant differences were documented for these provinces ($P < 0.0001$), with Banteay Meanchey and Battambang having lower soil As than Kandal and Prey Veng.^[58] Sediments of 12–40 m deep cores from the Red River delta of Cambodia contained As levels of 2–33 µg/g (average 7 µg/g, dry weight) and demonstrated a remarkable correlation with sediment-bound Fe.^[33]

CONCLUSION

Naturally occurring As has been detected in groundwater in many parts of the world, and it is evident that the countries of Southeast Asia are highly contaminated area. The soils irrigated with As-contaminated water also contained considerable amounts of As. Consequently, crops and in particular paddy rice that is cultivated on contaminated lands accumulate high levels of As. Effective remedial

strategies for water, soil, and food crops grown in contaminated soils are urgently required to prevent or minimize the exposure of As to humans.

REFERENCES

1. IPCS. *IPCS Environmental Health Criteria. 224: Arsenic and Arsenic Compounds*; WHO: Geneva, Switzerland, 2001; 1–521.
2. Rahman, M.M.; Ng, J.C.; Naidu, R. Chronic exposure of arsenic via drinking water and its adverse health impacts on humans. *Environ. Geochem. Hlth.* **2009**, *31* (1), 189–200.
3. Kapaj, S.; Peterson, H.; Liber, K.; Bhattacharya, P. Human health effects from chronic arsenic poisoning—A review. *J. Environ. Sci. Heal. A.* **2006**, *41* (10), 2399–2428.
4. Mukherjee, A.; Sengupta, M.K.; Hossain, M.A.; Ahamed, S.; Das, B.; Nayak, B.; Lodh, D.; Rahman, M.M.; Chakraborti, D. Arsenic contamination in groundwater: A global perspective with emphasis on the Asian scenario. *J. Health Popul. Nutr.* **2006**, *24* (2), 142–163.
5. Mandal, B.K.; Suzuki, K.T. Arsenic round the world: A review. *Talanta* **2002**, *58* (1), 201–235.
6. Winkel, L.; Berg, M.; Stengel, C.; Rosenberg, T. Hydrogeological survey assessing arsenic and other groundwater contaminants in the lowlands of Sumatra, Indonesia. *Appl. Geochem.* **2008**, *23* (11), 3019–3028.
7. Smedley, P.L.; Kinniburgh, D.G. A review of the source, behaviour and distribution of arsenic in natural waters. *Appl. Geochem.* **2002**, *17* (5), 517–568.
8. Bundschuh, J.; Litter, M.I.; Parvez, F.; Román-Ross, G.; Nicolli, H.B.; Jean, J.-S.; Liu, C.-W.; López, D.; Armienta, M.A.; Guilherme, L.R.G.; Cuevas, A.G.; Cornejo, L.; Cumbal, L.; Toujaguez, R. One century of arsenic exposure in Latin America: A review of history and occurrence from 14 countries. *Sci. Total Environ.* **2012**, *429*, 2–35.
9. Kemper, K.E.; Minnatullah, K.M. *Towards a More Effective Operational Response: Arsenic Contamination of Groundwater in South and East Asian Countries*; World Bank and Water and Sanitation Program: South and East Asia, 2005.
10. Kinniburgh, D.; Smedley, P. Arsenic contamination of groundwater in Bangladesh. British Geological Survey: Nottingham, 2001.
11. Frisbie, S.H.; Ortega, R.; Maynard, D.M.; Sarkar, B. The concentrations of arsenic and other toxic elements in Bangladesh's drinking water. *Environ. Health Persp.* **2002**, *110* (11), 1147.
12. Van Geen, A.; Cheng, Z.; Jia, Q.; Seddique, A.A.; Rahman, M.W.; Rahman, M.M.; Ahmed, K.M. Monitoring 51 community wells in Araihaazar, Bangladesh, for up to 5 years: Implications for arsenic mitigation. *J. Environ. Sci. Heal. A.* **2007**, *42* (12), 1729–1740.
13. Chakraborti, D.; Rahman, M.M.; Das, B.; Murrill, M.; Dey, S.; Mukherjee, S.C.; Dhar, R.K.; Biswas, B.K.; Chowdhury, U.K.; Roy, S. Status of groundwater arsenic contamination in Bangladesh: A 14-year study report. *Water Res.* **2010**, *44* (19), 5789–5802.
14. Chakraborti, D.; Rahman, M.M.; Mukherjee, A.; Alauddin, M.; Hassan, M.; Dutta, R.N.; Pati, S.; Mukherjee, S.C.; Roy, S.; Quamruzzman, Q. Groundwater arsenic contamination

- in Bangladesh-21 years of research. *J. Trace Elem. Med. Bio.* **2015**, *31*, 237–248.
15. Argos, M.; Kalra, T.; Rathouz, P.J.; Chen, Y.; Pierce, B.; Parvez, F.; Islam, T.; Ahmed, A.; Rakibuz-Zaman, M.; Hasan, R.; Sarwar, G.; Slavkovich, V.; van Geen, A.; Graziano, J.; Ahsan, H. Arsenic exposure from drinking water, and all-cause and chronic-disease mortalities in Bangladesh (HEALS): A prospective cohort study. *Lancet* **2010**, *376* (9737), 252–258.
 16. Chakraborti, D.; Das, B.; Rahman, M.M.; Chowdhury, U.K.; Biswas, B.; Goswami, A.; Nayak, B.; Pal, A.; Sengupta, M.K.; Ahamed, S. Status of groundwater arsenic contamination in the state of West Bengal, India: A 20-year study report. *Mol. Nutr. Food Res.* **2009**, *53* (5), 542–551.
 17. Nickson, R.; Sengupta, C.; Mitra, P.; Dave, S.; Banerjee, A.; Bhattacharya, A.; Basu, S.; Kakoti, N.; Moorthy, N.; Wasuja, M. Current knowledge on the distribution of arsenic in groundwater in five states of India. *J. Environ. Sci. Heal. A.* **2007**, *42* (12), 1707–1718.
 18. Chakraborti, D.; Rahman, M.M.; Das, B.; Nayak, B.; Pal, A.; Sengupta, M.K.; Hossain, M.A.; Ahamed, S.; Sahu, M.; Saha, K.C. Groundwater arsenic contamination in Ganga–Meghna–Brahmaputra plain, its health effects and an approach for mitigation. *Environ. Earth Sci.* **2013**, *70* (5), 1993–2008.
 19. Borah, M.; Misra, A. Arsenic level in ground and surface water in predominantly rural areas of Golaghat sub-division in Golaghat district, Assam, India. *J. Environ. Sci.* **2011**, *53* (1), 75–80.
 20. Guo, H.; Wen, D.; Liu, Z.; Jia, Y.; Guo, Q. A review of high arsenic groundwater in Mainland and Taiwan, China: Distribution, characteristics and geochemical processes. *Appl. Geochem.* **2014**, *41*, 196–217.
 21. Chanpiwat, P.; Sthiannopkao, S.; Cho, K.H.; Kim, K.-W.; San, V.; Suvanthong, B.; Vongthavady, C. Contamination by arsenic and other trace elements of tube-well water along the Mekong River in Lao PDR. *Environ. Pollut.* **2010**, *159* (2), 567–576.
 22. van Geen, A.; Win, K.H.; Zaw, T.; Naing, W.; Mey, J.L.; Mailloux, B. Confirmation of elevated arsenic levels in groundwater of Myanmar. *Sci. Total Environ.* **2014**, *478*, 21–24.
 23. Bacquart, T.; Frisbie, S.; Mitchell, E.; Grigg, L.; Cole, C.; Small, C.; Sarkar, B. Multiple inorganic toxic substances contaminating the groundwater of Myingyan Township, Myanmar: Arsenic, manganese, fluoride, iron, and uranium. *Sci. Total Environ.* **2015**, *517*, 232–245.
 24. Shrestha, R.R.; Shrestha, M.P.; Upadhyay, N.P.; Pradhan, R.; Khadka, R.; Maskey, A.; Maharjan, M.; Tuladhar, S.; Dahal, B.M.; Shrestha, K. Groundwater arsenic contamination, its health impact and mitigation program in Nepal. *J. Environ. Sci. Heal. A.* **2003**, *38* (1), 185–200.
 25. Chapagain, S.K.; Shrestha, S.; Nakamura, T.; Pandey, V.P.; Kazama, F. Arsenic occurrence in groundwater of Kathmandu Valley, Nepal. *Desalin. Water Treat* **2009**, *4* (1–3), 248–254.
 26. Thakur, J.K.; Thakur, R.K.; Ramanathan, A.; Kumar, M.; Singh, S.K. Arsenic contamination of groundwater in Nepal—An overview. *Water* **2010**, *3* (1), 1–20.
 27. Yadav, I.C.; Dhuldhaj, U.P.; Mohan, D.; Singh, S. Current status of groundwater arsenic and its impacts on health and mitigation measures in the Terai basin of Nepal: An overview. *Environ. Rev.* **2011**, *19* (NA), 55–67.
 28. Baig, J.A.; Kazi, T.G.; Arain, M.B.; Afridi, H.I.; Kandhro, G.A.; Sarfraz, R.A.; Jamal, M.K.; Shah, A.Q. Evaluation of arsenic and other physico-chemical parameters of surface and ground water of Jamshoro, Pakistan. *J. Hazard. Mater.* **2009**, *166* (2–3), 662–669.
 29. Pattanapitpaisal, P.; Suraruk, P. Groundwater quality and arsenic contamination in Amphoe Khemmarat, Ubon Ratchathani, Thailand. *J. Environ. Sci. Eng. A.* **2012**, *1* (2), 133–141.
 30. Hoang, T.H.; Bang, S.; Kim, K.-W.; Nguyen, M.H.; Dang, D.M. Arsenic in groundwater and sediment in the Mekong River delta, Vietnam. *Environ. Pollut.* **2010**, *158* (8), 2648–2658.
 31. Agusa, T.; Kunito, T.; Fujihara, J.; Kubota, R.; Minh, T.B.; Trang, P.T.K.; Iwata, H.; Subramanian, A.; Viet, P.H.; Tanabe, S. Contamination by arsenic and other trace elements in tube-well water and its risk assessment to humans in Hanoi, Vietnam. *Environ. Pollut.* **2006**, *139* (1), 95–106.
 32. Berg, M.; Stengel, C.; Trang, P.T.K.; Viet, P.H.; Sampson, M.L.; Leng, M.; Samreth, S.; Fredericks, D. Magnitude of arsenic pollution in the Mekong and Red River Deltas—Cambodia and Vietnam. *Sci. Total Environ.* **2007**, *372* (2), 413–425.
 33. Buschmann, J.; Berg, M.; Stengel, C.; Sampson, M.L. Arsenic and manganese contamination of drinking water resources in Cambodia: Coincidence of risk areas with low relief topography. *Environ. Sci. Technol.* **2007**, *41* (7), 2146–2152.
 34. Luu, T.T.G.; Sthiannopkao, S.; Kim, K.-W. Arsenic and other trace elements contamination in groundwater and a risk assessment study for the residents in the Kandal Province of Cambodia. *Environ. Int.* **2009**, *35* (3), 455–460.
 35. Buschmann, J.; Berg, M.; Stengel, C.; Winkel, L.; Sampson, M.L.; Trang, P.T.K.; Viet, P.H. Contamination of drinking water resources in the Mekong delta floodplains: Arsenic and other trace metals pose serious health risks to population. *Environ. Int.* **2008**, *34* (6), 756–764.
 36. Nguyen, V.A.; Bang, S.; Viet, P.H.; Kim, K.-W. Contamination of groundwater and risk assessment for arsenic exposure in Ha Nam province, Vietnam. *Environ. Int.* **2009**, *35* (3), 466–472.
 37. Shinkai, Y.; Truc, D.V.; Sumi, D.; Canh, D.; Kumagai, Y. Arsenic and other metal contamination of groundwater in the Mekong River Delta, Vietnam. *J. Health. Sci.* **2007**, *53* (3), 344–346.
 38. Alam, M.; Sattar, M. Assessment of arsenic contamination in soils and waters in some areas of Bangladesh. *Water Sci. Technol.* **2000**, *42* (7–8), 185–192.
 39. Ahmed, Z.U.; Panaullah, G.M.; DeGloria, S.D.; Duxbury, J.M. Factors affecting paddy soil arsenic concentration in Bangladesh: Prediction and uncertainty of geostatistical risk mapping. *Sci. Total Environ.* **2011**, *412–413*, 324–335.
 40. Islam, M.; Salminen, R.; Lahermo, P. Arsenic and other toxic elemental contamination of groundwater, surface water and soil in Bangladesh and its possible effects on human health. *Environ. Geochem. Hlth.* **2000**, *22* (1), 33–53.
 41. Meharg, A.; Rahman, M. Arsenic contamination of Bangladesh paddy field soils: Implications for rice

- contribution to arsenic consumption. *Environ. Sci. Technol.* **2003**, *37* (2), 229–234.
42. Rahman, M.S.; Abdul Mazid Miah, M.; Khaled, H.M.; Islam, A.; Panaullah, G.M. Arsenic concentrations in groundwater, soils, and irrigated rice in Southwestern Bangladesh. *Commun. Soil Sci. Plan.* **2010**, *41* (16), 1889–1895.
 43. Kurosawa, K.; Egashira, K.; Tani, M.; Jahiruddin, M.; Moslehuddin, A.Z.M.; Rahman, Z.M. Groundwater–soil–crop relationship with respect to arsenic contamination in farming villages of Bangladesh – A preliminary study. *Environ. Pollut.* **2008**, *156* (2), 563–565.
 44. Ahsan, D.; DelValls, T.; Blasco, J. Distribution of arsenic and trace metals in the floodplain agricultural soil of Bangladesh. *B. Environ. Contam. Tox.* **2009**, *82* (1), 11–15.
 45. Alam, M.G.M.; Snow, E.T.; Tanaka, A. Arsenic and heavy metal contamination of vegetables grown in Samta village, Bangladesh. *Sci. Total Environ.* **2003**, *308* (1), 83–96.
 46. Hossain, M.M.; Sattar, M.A.; Hashem, M.A.; Islam, M.R. Arsenic contamination in some selected soils of Bangladesh. *J. Biol. Sci.* **2001**, *1* (10), 989–992.
 47. Norra, S.; Berner, Z.A.; Agarwala, P.; Wagner, F.; Chandrasekharam, D.; Stüben, D. Impact of irrigation with As rich groundwater on soil and crops: A geochemical case study in West Bengal Delta Plain, India. *Appl. Geochem.* **2005**, *20* (10), 1890–1906.
 48. Roychowdhury, T.; Uchino, T.; Tokunaga, H.; Ando, M. Arsenic and other heavy metals in soils from an arsenic-affected area of West Bengal, India. *Chemosphere* **2002**, *49* (6), 605–618.
 49. Roychowdhury, T.; Tokunaga, H.; Uchino, T.; Ando, M. Effect of arsenic-contaminated irrigation water on agricultural land soil and plants in West Bengal, India. *Chemosphere* **2005**, *58* (6), 799–810.
 50. Li, S.; Wang, M.; Yang, Q.; Wang, H.; Zhu, J.; Zheng, B.; Zheng, Y. Enrichment of arsenic in surface water, stream sediments and soils in Tibet. *J. Geochem. Explor.* **2013**, *135*, 104–116.
 51. Sahoo, P.; Zhu, W.; Kim, S.-H.; Jung, M.; Kim, K. Relations of arsenic concentrations among groundwater, soil and paddy from an alluvial plain of Korea. *Geosci. J.* **2013**, *17* (3), 363–370.
 52. Huang, R.-Q.; Gao, S.-F.; Wang, W.-L.; Staunton, S.; Wang, G. Soil arsenic availability and the transfer of soil arsenic to crops in suburban areas in Fujian Province, Southeast China. *Sci. Total Environ.* **2006**, *368* (2–3), 531–541.
 53. Hsu, W.-M.; Hsi, H.-C.; Huang, Y.-T.; Liao, C.-S.; Hseu, Z.-Y. Partitioning of arsenic in soil-crop systems irrigated using groundwater: A case study of rice paddy soils in southwestern Taiwan. *Chemosphere* **2012**, *86* (6), 606.
 54. Kar, S.; Das, S.; Jean, J.-S.; Chakraborty, S.; Liu, C.-C. Arsenic in the water–soil–plant system and the potential health risks in the coastal part of Chianan Plain, Southwestern Taiwan. *J. Asian Earth Sci.* **2013**, *77*, 295–302.
 55. Zandsalimi, S.; Karimi, N.; Kohandel, A. Arsenic in soil, vegetation and water of a contaminated region. *Int. J. Environ. Sci. Te.* **2011**, *8* (2), 331–338.
 56. Phuong, N.M.; Kang, Y.; Sakurai, K.; Iwasaki, K.; Kien, C.N.; Van Noi, N.; Son, L.T. Arsenic contents and physicochemical properties of agricultural soils from the Red River Delta, Vietnam. *Soil Sci. Plant Nutr.* **2008**, *54* (6), 846–855.
 57. Hamzah, A.; Wong, K.; Hasan, F.; Mustafa, S.; Khoo, K.; Sarmani, S. Determination of total arsenic in soil and arsenic-resistant bacteria from selected ground water in Kandal Province, Cambodia. *J. Radioanal. Nucl. Ch.* **2013**, *297* (2), 291–296.
 58. Seyfferth, A.; Mccurdy, S.; Schaefer, M.; Fendorf, S. Arsenic concentrations in paddy soil and rice and health implications for major rice-growing regions of Cambodia. *Environ. Sci. Technol.* **2014**, *48* (9), 4699–4699.

Art and Soil

Christian Feller

Institute of Research for Development, Montpellier, France

Abstract

This entry is a chronological short history of Western art (mainly paintings) from prehistory to the contemporary period through the word “Soil.” The conclusion is that the vision of Soil (in a scientific meaning), as an independent work of art, is contemporary.

SOIL IN ART

It is widely accepted that humans have always considered the natural environment a subject of great interest to art. Early pictorial examples include cave paintings done by Cro-Magnon man during the Upper Paleolithic, about 30,000–40,000 years ago. However, the vision of Soil (in a scientific meaning), as an independent work of art is contemporary and extremely rare in the world of painting. For many years, artists have depicted actual or imaginary landscapes from which the trained eye of a pedologist, agronomist, or geographer can recognize a schematic view of what is commonly called soil but not Soil (as a Soil profile). This communication will show: 1) the representations of Soil or soil in Western art from the Paleolithic to the modern era; and 2) some contemporary artworks where the Soil is considered as the main subject and has its goal to present Soil in art from Genesis (the Bible) to Pedogenesis (the scientific approach of the Soil formation from the Greek word *Pedon* meaning soil).^[1,2]

When Soil Is Depicted By Chance in the Landscape: The Soil as a Surface

In the biblical Genesis story of world creation, the whole of humanity is “soil” as Adam—meaning “soil” in Hebrew—was created from red dust and returned back to it. To a large extent, the representation of soils, as even a single line of soil surface, is neglected in Upper Paleolithic cave paintings.

The few extremely schematic representations inherited from the Assyrian civilization (from the 11th century to 7th century B.C.) depict natural scenes in which the soil surface is represented by schematic rocks and hillocks, drawn as shaped curves.^[3] In Grecian art, very few traces of soil representation have been found, except for frescoes of Aegean art coming from Santorini dating from the 5th century B.C.^[4] Wall paintings were widespread in the Roman civilization as decorative art designed in a truly realistic style that would never be seen again until the Early

Renaissance. At Pompeii, nature was represented through flowers and birds as well as other animals. However, relatively few representations of the landscape stood the test of time.^[4] They were probably simply lost.

From the Byzantine period of the early Middle Ages (the 6th century), many mosaics depicted rocky landscapes. However, between the 5th and 12th centuries, the representations of soil surface or landscapes are very often strongly schematized with undulating lines or hillocks, as in religious miniatures.

The Florentine painter Giotto (1266–1337) made a decisive break with the static Byzantine style, introducing realism. His paintings of rocky landscapes were among the first that included some perspective. Other Italian painters from the contemporary Giotto’s Sienese School developed a similar naturalistic style: Duccio di Buoninsegna (1260–1318), Simone Martini (1284–1344), and the Lorenzetti Brothers (1280/1285–1348).

During the Renaissance, the soil, as a surface, is generally represented in a highly realistic way, even for symbolic and/or imaginary landscapes. Such realism sometimes allows one to discern the Soil profile with different colors given to the surface soil and to deep horizons, as, e.g., in the works by Hans Memling (1430–1494) of “The Last Judgment” (ca. 1470)^[5] and Hieronymus Bosch with his “St. John the Baptist” (ca. 1500).

When Soil Is Depicted By Choice in the Landscape: The Soil as a Profile

The following three reasons motivated the representation of a soil profile: to explain the resurrection of the dead, to display the roots, and to show ploughing.

In “The Last Judgment” by Rogier Van der Weyden (1432), the resurrection of dead required the artist to show the Soil profile. The complete painting exhibits numerous such soil profiles.

In the paintings of the Renaissance, the representation of a ditch or a soil cut in a painting served very often as an excuse to picture roots. In the “St. John the Baptist” by

Hieronymus Bosch (1450–1516), the figure of St. John leans toward a sharp vertical exposure of soil, which includes a strange large root.^[6] A large root also appears in “The Tempest” painted by Giorgione (1477/1478–1510)^[7] and in “The Fall of Icarus” by Pieter Bruegel the Elder (1525–1569), just at right and behind the ploughman. These works were just some examples of paintings in which large forked roots were made evident. The representation of roots was not due to chance, but chosen for its symbolic value, and referred to the mandragora as suggested by Marijnissen and Ruyffelaere.^[8] The perception of mandragora as the subject of superstitions was presented in the *Encyclopédie des Symboles*^[9] through the following comments:

Mandragora is a plant with a high symbolic value, inspiring both fear and fascination. Its forked root which crudely resembles the human form has been credited since ancient times with a divine origin. It is considered as a universal medicine. The mandrake grows only at night, releasing some toxins (hyoscyamine, atropine, scopolamine) with a narcotic effect. For this reason, the root was used by medieval witches to concoct potions, and it played a remarkable role in the occult practices. According to the legend, the root grew only beneath gallows trees, as it was believed to be produced from the semen involuntarily ejaculated by a hanged man. It has to be gathered with high caution, and it was said that the mandrake gave forth an extremely piercing and fatal cry. It was uprooted, therefore, by a dog that died immediately after. During antiquity, the mandrake was considered as one of the attributes of the sorceress Circe. The root was used by the Jews to overcome infertility. In general, mandrake was associated with black and supernatural forces that man would approach with many precautions.

From the 14th to the 15th centuries, especially in the “Très Riches Heures” and the “Calendriers” (calendars), we see the representations of agricultural tasks and toils. Here, the soil is depicted with a clear concern of realism and technical specificity, including the tilling of the soil. In addition to this example, Pieter Bruegel the Elder (1525/30–1569) might be newly cited for “The Fall of Icarus.” Icarus is the tiny figure at the bottom on the right-hand corner, with only his legs visible, while in forefront of the canvas, attention is centered on the good Flemish ploughman tilling furrows. That was the triumph of daily working life over Utopia (“falling from the sky”).

THE SOIL BY HIERONYMUS BOSCH AND HIS DISCIPLES

The work of Hieronymus Bosch (ca. 1450–1516) abounds in “earth” and “bare soil” representations as seen in the “The Temptation of St. Anthony.” The soil in Bosch’s work was represented not only as a surface, but often either as a Soil profile in a slope cut (see above) or in adobes that are

associated with thatched roofs. Here, the artist emphasized the decomposition and decay of the sides of huts, in the same way that plant debris is decomposing on the top of the soil. The whole of Bosch’s work is influenced by “decomposition,” and soil depiction contributes to this process. The work of Bosch would deserve an independent study of his “soil” or “Soil” vision.

As Hieronymus Bosch is said to have been an inspiration for the surrealist movement of the 20th century, some surrealists might be considered as his disciples throughout their vision of soil or earth’s uses, e.g.:

- Salvador Dalí with “Soft Construction with Boiled Beans: Premonition of Civil War” (1936. Oil on canvas), Museum of Art, Philadelphia,^[10] “The Spectre of Sex-Appeal” (1934. Oil on canvas), Gala-Salvador Dalí Foundation, Figueres, Spain,^[11] or the “Metamorphosis of Narcissus”^[12];
- Yves Tanguy with the “Extinction of Useless Lights” (“Extinction des lumières inutiles”; 1927. Oil on canvas);
- Jean Dubuffet with “The Magician” (1954. Slag and roots, including slag base), both on view at the Museum of Modern Art, New York.^[13]

THE SOIL: HISTORY

Paintings during the 16th, 17th, and 18th centuries reached realistic excellence in representing the soil surface that could never be equaled. During this period and until the 20th century, artists did not consider the Soil as a chief subject, not even the impressionists who rendered it only as a landscape component.

During the 20th century, the soil surface was well represented in the land art movement. Soil is depicted as an object per se especially by some naturalists, agronomists, pedologists, and others who have developed a substantial artistic talent besides their scientist’s profession. Many works of contemporary artists—paintings, sculptures, performances, or art installations—centered on the Soil can be seen at Wessolek’s Web site,^[1] particularly, women artists, such as Ulrike Arnold, Pablia Velarde, Janis Lang, Betty Beier, Birgit Kratzhaller, and Kathryn Miller, who have led the way in using soil and geologic material as a means of artistic expression.

CONCLUSION

Our look at soil or Soil in art confirmed that in Western culture, most artists did not view soil as the complex and subtly beautiful medium that holds the interest of agronomists or pedologists. It was usually only considered as a surface. The belowground layers were generally not represented, while rocks and other natural objects fascinated artists. However, as underlined by the conclusion in

Jenny^[14]: “Whoever said that soils lack beauty is behind the times. Soil in art has arrived. It is an enrichment of art that is here to stay.”

Finally, it is predicted that we will increasingly see more artists and soil scientists interacting at the interface of their expertise and consciousness to produce images and objects that will capture the attention of audiences. If a successful exhibition on “the Soil” could be hosted by the Grand Palais in Paris or the Museum of Modern Art in New York, the soil science texts would surely become best sellers the day after the opening reception!

Never stop dreaming...

ACKNOWLEDGMENT

The authors thank Edward Landa for assisting with the English language review and editing this entry.

REFERENCES

1. Available at <http://www.kunstundboden.de/>.
2. Alfred, E. *Hartemink Home Page*. Available at <http://www.alfredhartemink.nl/various.htm>.
3. Parrot, A. *Assur: Univers Des Formes, Coll. Dirigée par A. Malraux et G. Salles*; NRF Gallimard: Paris, 1961; 40 pp.
4. Carli, E. *Le Paysage Dans L'art. Traduit De L'italien Par M. Orcel*; Nathan: Paris, 1980; 12, 21, 24.
5. Memling, H. *The Last Judgment*. Available at [http://en.wikipedia.org/wiki/The_Last_Judgment_\(Memling\)](http://en.wikipedia.org/wiki/The_Last_Judgment_(Memling)).
6. Bosch, H. *St. John the Baptist in the Wilderness*, http://en.wikipedia.org/wiki/St._John_the_Baptist_in_the_Wilderness.
7. Giorgione. *The Tempest (painting)*. Available at [http://en.wikipedia.org/wiki/The_Tempest_\(painting\)](http://en.wikipedia.org/wiki/The_Tempest_(painting)).
8. Marijnissen, R.H.; Ruyffelaere, Jérôme Bosch P. *Tout L'oeuvre Peint Et Dessiné. Fonds Mercator*; Albin Michel: Paris, 1987; 513 pp.
9. Encyclopédie des Symboles. *Article «Mandragore», 392. traduit de l'allemand par F. Périgaut, G. Marie et A. Tondat. Edition française établie sous la direction de M. Cazenave*; Librairie Générale Française, Pochoth'que, Livre de Poche, 1996; 818 pp.
10. Dali, S. *Soft Construction with Boiled Beans (Premonition of Civil War)*. Available at <http://www.philamuseum.org/collections/permanent/51315.html>.
11. Dali, S. *The Spectre of Sex Appeal*. Available at <http://www.salvador-dali.org/museus/teatre-museu-dali/the-collection/143/the-spectre-of-sex-appeal>.
12. Dali, S. *Metamorphosis of Narcissus*. Available at http://upload.wikimedia.org/wikipedia/en/2/21/Metamorphosis_of_Narcissus.jpg.
13. Museum of Modern Art (MoMA). New York. Available at <http://www.moma.org/collection>.
14. Jenny, H. The image of soil in landscape art, old and new. In *Study Week on Organic Matter and Soil Fertility*, Pontificiae Academiae Scientiarum Scripta Varia 32; North Holland Publ. and Wiley Interscience: Amsterdam, New York, 1968; 947–979.

Atterberg Limits

Thomas Baumgartl

Institute for Plant Nutrition and Soil Science, Kiel University, Kiel, Germany

Abstract

Soil is exposed to different states of stability depending on the amount of water that it contains. This characteristic is described as consistency and specifies the state of a remolded and cohesive soil in the range from the liquid (when wet) to plastic and finally solid (when dry) state. Different soils contain a specific amount of water at these different states of stability. In 1911, the Swedish soil physicist Atterberg developed a classification system and method with which these states of consistency could be determined. The method is based on the determination of the water content [calculated as: (mass of water)/(dry mass of soil)] at distinct transitions between different states of consistency of soil. These transitions are defined as liquid limit, plastic limit, and shrinkage limit and are generally referred to as Atterberg limits. The values for these limits are dependent on various soil parameters, e.g., particle size, specific surface area of the particles which is able to take up water, and hence its particle size distribution. These limits are used to derive indexes, e.g., index of plasticity and index of consistency, and are often used for the mechanical characterization of soils.

DEFINITION OF LIMITS AND THEIR DETERMINATION

Liquid Limit (Upper Plastic Limit; w_L)

The liquid limit describes the transition from a viscous liquid to a plastic state. Soils with a water content at the liquid limit barely flow under an applied force. The associated capillary forces of the water menisci in the unsaturated pore system are equivalent to $pF \sim 0.5$ (~ 0.3 kPa matric potential).^[1]

The liquid limit is determined by a method and device developed by Casagrande. The principle is to find the water content (kg/kg) at which a soil sample starts to liquefy under a small applied stress. In practice, a groove is cut into soil samples with different water contents.

These soil samples are then exposed to a small standardized force by repeatedly dropping the Casagrande cup over a distance of 10 mm until the groove is close to ca. 10 mm. A semilogarithmic plot of the number of blows as a function of water content will result in a straight line. The liquid limit is defined as the water content at 25 blows (Fig. 1).

Plastic Limit (Lower Plastic Limit; w_P)

The plastic limit determines the transition from a plastic (cohesive) to a semirigid or brittle state. Under an applied force cracking will occur. In sandy soils, the plastic limit often cannot be determined. The matric potential at the plastic limit is the main cohesive stress and ranges between 63 and 200 kPa (pF 2.8–3.3).^[1]

The plastic limit is determined by forming a moist ball from 2 to 3 g of soil, which is then rolled on—a piece of frosted glass to a rod of thickness ca. 3 mm. The remolding and rolling are repeated until the 3 mm rod starts to break up into pieces of 10–20 mm. The gravimetric water content (kg/kg) at this point gives the plastic limit.

Shrinkage Limit (w_S)

Cohesive and remolded soils reduce their soil volume with the loss of water due to capillary forces. If in a drying process the reduction of the total soil volume equals the volume of water loss, then the soil shows normal shrinkage. Below a certain water content, the further shrinkage of the soil volume is restricted due to a high number of particle contact points and high effective stresses. This restricted shrinkage pattern is called residual shrinkage (Fig. 2). The transition from normal to residual shrinkage defines the shrinkage limit.

The shrinkage limit is often calculated by the following formula:

$$w_S = 0.65 \times w_P \quad (1)$$

Index of Plasticity (I_P)

The I_P is the amount of water between plastic and liquid limit and is calculated by the following formula:

$$I_P = w_L - w_P \quad (2)$$

It describes the sensitivity in the mechanical behavior of a soil toward changes in water content. However, it does not explain mechanical stability, as hydraulic parameters

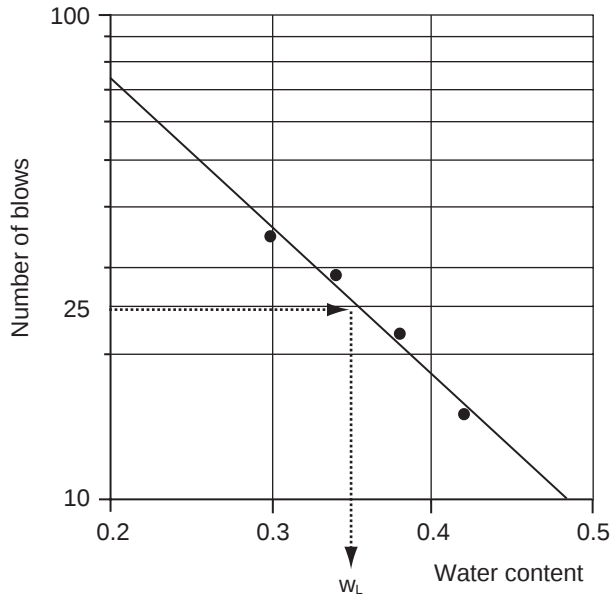


Fig. 1 The determination of liquid limit.

are not included which are necessary as water flow becomes important when stresses are applied on a soil sample. Values for the index may range from 0 (no plastic behavior) for sandy material to 1 (100%) for clay.

Index of Consistency (I_C)

The I_C is determined as the ratio of the difference between the liquid limit and actual water content and the I_p as follows:

$$I_C = \frac{w_L - w}{w_L - w_P} = \frac{w_L - w}{I_p} \quad (3)$$

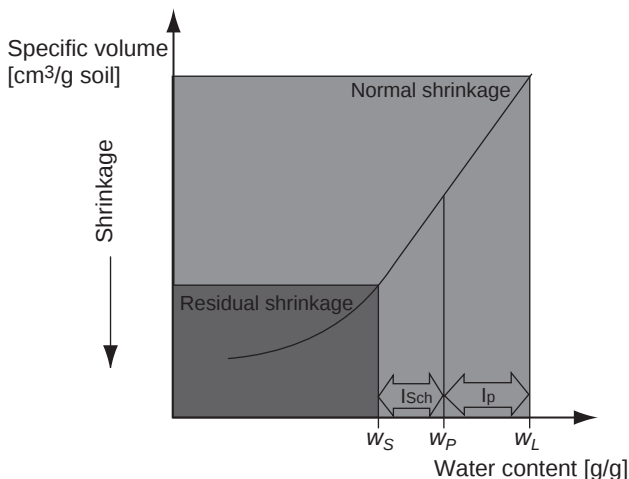


Fig. 2 Shrinkage behavior with change in water content in relation to Atterberg limits w_S , w_P , w_L , and Atterberg indexes.

Source: From Kretschmer.^[3]

The water content was such provides no information about the consistency of soils. The same water content in a sandy soil may reflect a liquid state, whereas in a clay soil the behavior may be brittle. The I_C normalizes the water contents and characterizes whether the soil is close to the plastic limit ($\max I_C = 1$) or the liquid limit ($\min I_C = 0$).

Index of Shrinkage (I_{Sch})

The difference between the plastic limit and the shrinkage limit results in the I_{Sch} by the following formula:

$$I_{Sch} = w_P - w_S \quad (4)$$

Soils with water contents within this range are most suitable for cultivation (see Fig. 2). Table 1 summarizes some general values for the Atterberg limits and indexes.^[2]

FACTORS INFLUENCING THE ATTERBERG LIMITS

Many mechanical processes are linked to hydrological properties of soils. Therefore, values of the limits and indexes are influenced by factors, which are generally important for the water retention curve, e.g., the capacity of swelling and shrinkage, clay content, type of clay minerals, and organic matter.

Generally, the values of the limits and indexes increase with their clay content. As the liquid limit increases in comparison to the plastic limit, the I_p also increases. The swelling and shrinkage intensity are dependent not only on the amount but also on the type of clay mineral. Skempton introduced the following factor described as the activity of clay:^[3,4]

$$A = \frac{I_p}{\% \text{ clay content}} \quad (5)$$

The values of A can be classified as follows: 1) $A > 1.25$: active soil with high capacity of swelling and shrinking [calcium montmorillonite ($A \sim 1.5$), sodium (Na) montmorillonite ($A \sim 7.5$), smectite, and salt influenced clays]; 2) $0.75 < A < 1.25$: normal soils (illite); and 3) $A < 0.75$: inactive clay with only little swelling–shrinking activity (kaolinite).

Table 1 Consistency limits (g water/g soil) for different soil textures.

Texture	Sand	Silt	Clay
Consistency limits			
Liquid limit	0.15–0.20	0.30–0.40	0.40–1.50
Plastic limit	0	0.20–0.25	0.25–0.5
I_p	0	0.10–0.15	0.10–1.00
Shrinkage limit	0.12–0.18	0.14–0.25	0.08–0.25

Source: From Kezdi.^[2]

Organic matter increases both the plastic and liquid limits but does not have a big effect on the I_P . Organic substances in a soil matrix seem therefore to increase the surface hydration. Once this pool for water uptake is saturated, the soil shows the same mechanical behavior toward changes in water content as when organic matter is absent, only at a higher level of water content. Thus, the I_C is higher.^[1]

With the exception of organic matter and clays, the amount and type of exchangeable cations have a significant effect on the value of the limits.^[5–7] Na-saturated soils reduce the liquid limit but increase the shrinkage limit. Soils therefore have the tendency to show crust formation at an earlier stage and will slake at lower water contents.^[1,2]

APPLICATION

Soil characteristics are inherent in the values of the Atterberg limits. Therefore, Atterberg limits are correlated with soil properties. For specific soils, investigations have been carried out which correlate the total particle surface with the plastic limit.^[8,9] Soil strength can be defined by its compressibility, and compressibility is correlated to the Atterberg limits.^[10] As soil strength is influenced to a great extent by the energy status of the capillary water, Atterberg limits reflect the soil water potential and show the dependency on texture and the water retention curve.^[11]

At a broader scale, the Atterberg limits can be used for the evaluation of the trafficability and cultivation of soils. Table 2 lists the limits and derived mechanical properties and qualities of soil substrates.

From this classification (Table 2), it is evident that an optimal range of water contents for agricultural use can be determined. This range is present when the soil is stiff and has a compression strength of >100 kPa and an I_C between 0.75 and 1. Drier soils increase the energy input needed for cultivation, which can be a serious problem for clay-rich soils as plowing can become difficult. In the case of lower than optimal plasticity indexes, the soil structure can be destroyed easily when the soil is kneaded by trafficking resulting in ecological problems. As a result, the hydraulic conductivity and gas flow as well as nutrient uptake of plants can be reduced. Hence, cultivation at I_C smaller than

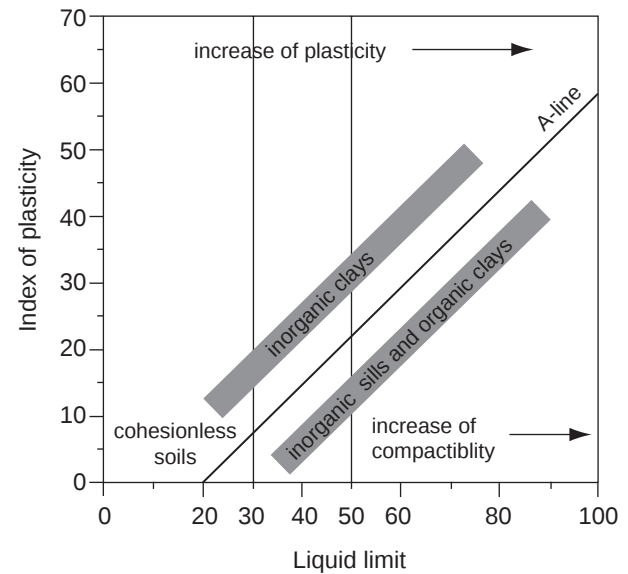


Fig. 3 The classification of soils according to Atterberg limits and Casagrande A-line.

0.75 can have a serious effect on plant growth and soil biological activity.

Although the values of the limits and indexes are not independent values, they can be related to each other (e.g., I_P and w_L). Classifications with respect to particle size distribution, geological origin of material, and suitability under soil mechanical point of view can be derived thereafter. With the ratio of liquid limit and I_P , a linear relationship was found by Casagrande and described as A-line,^[12] following the equation:

$$I_P = 0.73(w_L - 0.2) \quad (6)$$

It distinguishes soil with content of organic matter of $<5\%$ above this line from those in the range of 5–30% below the line. Certain groups of soils and soil substrates can be classified as shown in Fig. 3.

Soil mechanical parameters including the angle of internal friction and cohesion (the principle parameters for defining soil mechanical stability) can be related to the Atterberg limits as well.^[13,14] Generally, the angle of internal friction increases with decreasing soil substrate

Table 2 Mechanical properties of soils and Atterberg limits.

I_C	0	0.25	0.5	0.75	1	1.3	
Symbol	w _L				w _P	w _S	
Index			I _P			I _{Sch}	
Description	Slurry	Very soft	Soft	Deformable	Stiff	Medium hard	Hard
Unconfined compression strength (kPa)		<25	25–50	50–100	100–200	200–400	>400
Cultivation	–	–		–	+		–

Note: +, positive; –, negative.

Source: From Chancellor^[4] and Kretschmer.^[3]

Table 3 Mechanical parameters (angle of internal friction and cohesion) dependent on texture and I_p

Texture	I_p	Angle of internal friction ϕ' (°)	Cohesion (kPa)
Clay of high plasticity: $w_L > 0.5$	0.50–0.75	17.5	0
	0.75–1.00	17.5	10
	1.00–1.30	17.5	25
Clay and silt of medium plasticity: $0.35 < w_L < 0.5$	0.50–0.75	22.5	0
	0.75–1.00	22.5	5
	1.00–1.30	22.5	10
Clay and silt of low plasticity: $w_L < 0.35$	0.50–0.75	27.5	0
	0.75–1.00	27.5	2
	1.00–1.30	27.5	5

Source: Adapted from Kretschmer.^[3]

plasticity, i.e., with decreasing liquid limits. Cohesion increases with increasing I_p ; however, the increase is less with decreasing plasticities (see Table 3).

LIMITATIONS

Atterberg limits are empirical values developed for engineering purposes. They describe in approximation the change of the properties of soil material. The limits reflect not a sudden change in the mechanical state of soils but rather transitions. The determination of the limits does not describe processes but only categories for the classification of the soil. As denoted, capillary forces, tensile stresses, and volume change due to external and internal stresses are the parameters that determine the material behavior of the soil substrates. The determination of the Atterberg limits requires the destruction of structural units within the soil and remolding. Usually structure formation increases the strength of soils mainly by reorientation of particles and formation of bonding between particles. The interpretation of the Atterberg limits under agricultural–technical aspects should take these considerations into account.

REFERENCES

1. Baver, L.D.; Gardner, W.H.; Gardner, W.R. *Soil Physics*; Wiley: New York, 1972; 498 pp.
2. Kezdi, A. Bodenphysik. In *Handbuch der Bodenmechanik*; VEB Verlag fuer Bauwesen: Berlin, 1969; 259 pp.
3. Kretschmer, H. Koernung und konsistenz. In *Handbuch der Bodenkunde*; Blume, H.P., Felix-Henningsen, P., Fischer, W.R., Frede, H.-G., Horn, R., Stahr, K., Eds.; Landsberg/Lech: Ecomed, 1997; 1–45.
4. Chancellor, W.J. Soil physical properties. In *Advances in Soil Dynamics*. ASAE Monograph; DeVore-Hansen, P., Ed.; 12th Ed.; American Society of Agricultural Engineers: St Joseph, 1994; Vol. 1.
5. Dexter, A.R.; Chan, K.Y. Soil mechanical properties as influenced by exchangeable cations. *J. Soil Sci.* **1991**, *42* (2), 219–226.
6. Gill, W.R.; Reaves, C.A. Relationship of Atterberg limits and cation exchange capacity to some physical properties of soil. *Soil Sci. Soc. Am. J.* **1957**, *21*, 491–494.
7. Smith, C.W.; Hadas, A.; Dan, J.; Koyumdjisky, H. Shrinkage and Atterberg limits in relation to other properties of principal soil types in Israel. *Geoderma* **1999**, *35* (1), 47–65.
8. Farrar, D.M.; Coleman, J.D. The correlation of surface area with other properties of nineteen British clay soils. *J. Soil Sci.* **1967**, *18*, 118–124.
9. Hammel, J.E.; Sumner, M.E.; Burema, J. Atterberg limits as indices of external surface areas of soils. *Soil Sci. Soc. Am. J.* **1983**, *47*, 1054–1056.
10. McBride, R.A.; Bober, M.L. A re-examination of alternative test procedures for soil consistency limit determination: I. A compression-based procedure. *Soil Sci. Soc. Am. J.* **1989**, *53*, 178–183.
11. McBride, R.A. A re-examination of alternative test procedures for soil consistency limit determination: II. A simulated desorption procedure. *Soil Sci. Soc. Am. J.* **1989**, *53*, 184–191.
12. Terzaghi, K.; Peck, R.B. *Soil Mechanics in Engineering Practice*; Wiley: New York, 1967; 729 pp.
13. Kanji, M.A. The relationship between drained friction angles and Atterberg limits of natural soils. *Geotechnique* **1974**, *24* (4), 671–674.
14. Voight, B. Correlation between Atterberg plasticity limits and residual shear strength of natural soils. *Geotechnique* **1973**, *23* (2), 265–267.

Bacteria

Mary Ann Bruns

Department of Ecosystem Science and Management, Pennsylvania State University,
University Park, Pennsylvania, U.S.A.

Abstract

Bacteria are the most numerous microorganisms in soil, comprising billions per gram of topsoil but decreasing quickly with soil depth (to hundreds of thousands per gram). Bacteria make up an important living fraction of soil organic matter. Densities of bacteria are greater in soil associated with plant roots than in soil unaffected by plants. Bacteria use sugars and organic acids released by plant roots as carbon and energy sources. Some bacteria produce growth factors and other stimulatory compounds that improve plant growth and reduce stress. Working in concert with soil fungi, aerobic bacteria decompose organic matter, consume oxygen, build cell material, and respire carbon dioxide. During decomposition, chemical bonds in organic macromolecules are broken, and inorganic nutrients like ammonium and hydrogen phosphates become available for plant uptake. Active bacteria produce extracellular polysaccharides that help bind soil mineral grains into microaggregates. These in turn combine to form macroaggregates in well-structured soils having many macropores. Thousands of bacterial species coexist in soil, and the vast majority are beneficial rather than disease-causing. Specific functional groups of soil bacteria are responsible for key transformations of nitrogen (N), including atmospheric dinitrogen fixation, N mineralization, nitrification, and denitrification. Soil pH is an important factor influencing bacterial community composition.

INTRODUCTION

Soils contain a myriad of microscopic organisms belonging to all three domains of life (*Bacteria*, *Archaea*, and *Eukarya*), but bacteria are the most numerous. A single gram of garden soil may contain billions of individual bacteria coexisting with millions of archaea and thousands of Eukarya. *Bacteria* and *Archaea* are prokaryotic (i.e., single cells with nuclei that are not membrane-bound), while *Eukarya* are eukaryotic (i.e., single-celled or multicellular organisms having membrane-bound nuclei and other organelles). Among *Eukarya*, soil-dwelling taxa include protozoans and fungi, which themselves serve as hosts for bacteria inside or outside their cells.

As the most numerous microorganisms in soils, bacteria are important drivers of terrestrial carbon (C) and nitrogen (N) cycling. Soil bacteria act in concert with fungi in decomposing plant, animal, and microbial residues to obtain C and energy for growth (chemoheterotrophy), and their organic degradation products speed rock weathering and clay formation during soil development. The bacteria responsible for most organic matter decomposition in soils are aerobic chemoheterotrophs [i.e., they respire oxygen (O₂) to break chemical bonds in organic compounds to obtain energy and C]. Such nutrients as ammonium (NH₄⁺) and phosphate (H₂PO₄⁻), contained within plant litter as it enters the soil, cannot be taken up by plant roots until decomposer bacteria and fungi break the

organic–inorganic chemical bonds. Recycling of nutrients to support primary production and plant growth is therefore a crucial function of soil bacteria. Although aerobic heterotrophs make up a large fraction of soil bacterial communities, other bacteria with diverse physiologies also abound in soils.

This entry gives an overview of how our understanding of bacterial life has advanced over the years, from the use of microscopy and the study of pure cultures to Carl Woese's pivotal discovery of three domains of life based on ribosomal ribonucleic acid (rRNA). Since that discovery, the application of deoxyribonucleic acid (DNA)- and RNA-based methods has revealed the existence of many bacterial taxa (genetically related groups) in soils that have never been cultured in the laboratory. This entry describes characteristics and functions of the major phyla of bacteria found in soils and how their spatial distribution and activities are affected by edaphic (soil-related) factors. This entry also describes how DNA sequencing analyses of soil microbial genomes (extant genes) and transcriptomes (expressed genes) are helping to elucidate relationships between soil bacterial community compositions and biogeochemical functions. These functions include diverse transformations of C, N, and pollutants affecting bioavailability and turnover, greenhouse gas fluxes, and ecological interactions influencing plant health and agricultural productivity.

HISTORICAL ADVANCES

Classical Microbiology

Bacteria viewed under the microscope range in size from 0.2 to 1 μm , with most appearing as round (coccus) or rod-shaped (bacillus) cells in singles, doublets, clusters, or chains.^[1,2] Similarity in physical appearance of bacteria, however, masks astounding variety in genetic compositions, biochemical capabilities, and evolutionary histories. In surface soils having favorable conditions of moisture, aeration, organic matter, and other nutrients, bacterial densities can reach up to 10^{10} cells per gram.^[3] Soil bacteria are principal subjects of study in the field of soil microbiology, defined as “the branch of soil science concerned with soil-inhabiting microorganisms, their functions, and activities.”^[4]

Two historical developments made learning about bacteria in soils possible: the microscope developed by Antonie van Leeuwenhoek (1632–1723) and laboratory culture methods developed by Louis Pasteur (1822–1895) and Robert Koch (1843–1910). Culturing methods required sterile growth media and aseptic handling procedures to exclude other contaminating microorganisms and enable the study of genetically pure strains of bacteria.^[1] Such advances opened the way for extensive research on pure and mixed cultures by pioneers Sergei Winogradsky (1856–1953), the Father of Soil Microbiology, and Martinus Beijerinck (1851–1931), known as the Father of Microbial Ecology.^[5]

Sergei Winogradsky elucidated many different biochemical processes performed by bacteria in soils. He demonstrated chemolithotrophic sulfur oxidation, autotrophic nitrification, and N_2 fixation, respectively, at the University of Strasburg, Swiss Polytechnic Institute, and Imperial State Institute in St. Petersburg.^[6] Students learn about bacterial physiologies by creating biochemical gradients and observing bacterial growth in Winogradsky columns.^[1] Martinus Beijerinck, from the Delft School of Microbiology in the Netherlands, isolated cultures of bacteria that could fix atmospheric N_2 either symbiotically in association with plant roots (e.g., *Rhizobium*) or as free-living organisms in the soil (e.g., *Azotobacter*). Robert Koch’s advances in culturing pathogenic (disease-causing) bacteria enabled the study of many other bacteria from the environment.^[1]

Culture-Based Microbiology

Robert Koch’s techniques for isolating pathogens first involved placing infectious material on the cut surfaces of potatoes or solidified gelatin, which were nearly sterile (free of other living organisms) when handled aseptically. These substrates often became liquefied as the bacteria multiplied, causing growth to become mixed and making purification difficult. Subsequently, bacteria were grown in petri dishes containing gels made with agar, a less degradable carbohydrate made from seaweed, enabling outgrowth and multiplication of individual cells into well-separated

colonies (i.e., visible masses of bacterial clones arising from an individual cell). The repeated process of transferring growth from a well-isolated “colony-forming unit” for distribution on fresh sterile medium (serial transfer) is required to obtain cultures that appear genetically pure on microscopic examination.

Pure culture isolation was important in the formulation of Koch’s postulates for confirming that specific microorganisms (e.g., *Mycobacterium tuberculosis*) caused infectious disease.^[1] Successful pathogen control achieved through the study of pure cultures led to an emphasis during the 20th century on the isolation and study of bacteria from environmental samples. During that time, bacterial taxonomy (classification) was based on phenotypic (observable) characteristics, such as cell size and shape, growth characteristics, and basic metabolisms (e.g., aerobic, anaerobic, autotrophic, chemoheterotrophic, chemolithotrophic, and phototrophic).^[1] One particularly useful phenotypic trait is cell wall structure, as demonstrated by either purple or pink coloration of cells viewed under the microscope after Gram-staining. Cell walls of gram-positive bacteria (purple color) have a single, thick layer of peptidoglycan (amino-sugar polymer), while cell walls of gram-negative bacteria (pink color) have a thin layer of peptidoglycan surrounded by a periplasmic space and an outer cell membrane of lipopolysaccharide.^[1,5] Both types of bacteria make up important fractions of soil microbial communities.

Classification of bacterial isolates based on phenotype was difficult and often contradictory. Recognition also arose early in the 20th century that the study of cultured bacterial isolates would yield only an incomplete picture of soil microbial diversity.^[7] When soil smears were examined under the microscope, as many as 100 times more cells could be observed per gram than the number of colony-forming units calculated after dilution and plate counting.^[8] Discrepancies between plate counts and microscopic observations were at first attributed to dead and injured cells. Subsequent use of fluorescent stains^[9] to distinguish live cells from dead cells under the microscope indicated high proportions of viable, but nonculturable bacteria in many environments, including soils.^[8,10] Some unculturable bacteria may actually require unknown, alternative growth conditions (e.g., trace elements, ionic strength, and oxidation–reduction status) or substrates produced by different microorganisms (i.e., syntrophs). Some bacteria may undergo cell alteration to form dormant structures (e.g., spores and cysts),^[11] while others may be inactive for long periods interspersed with pulses of activity.^[11] Therefore, the number of detectable bacterial taxa at any given time in soil may far exceed the number of active taxa.^[12]

Process-Based Microbiology

Recognition that culture-based methods did not recover all microorganisms from soils led to a greater focus during the

1960s–1990s on measuring end products of collective processes carried out by whole soil microbial communities (i.e., a “black box” approach to soil microbiology).^[13] Aerobic respiration, e.g., is a key C cycling process whereby microorganisms generate energy from the oxidation of reduced C compounds (electron donors) to carbon dioxide (CO₂) and using the electrons to reduce the electron acceptor (O₂) to H₂O. Aerobic respiration by soil communities is thus measured as CO₂ emission rates and is directly related to C mineralization. Net C mineralization rates (micrograms of C–CO₂ per gram of soil⁻¹ d⁻¹) may be measured in the field or under optimized, controlled conditions in soil mesocosms in the laboratory (to estimate “potential” rates).^[11,14] Soil respiration rates following addition of labile C (e.g., glucose) to soils (substrate-induced respiration assays) also are used to determine the size of the soil microbial biomass. Group-selective antibiotics (i.e., cycloheximide to inhibit fungi or streptomycin to inhibit bacteria)^[11] permit estimation of the relative contributions by bacteria and fungi, respectively, to respiration. Gas fluxes from sterilized soils also are measured to rule out effects due to abiotic factors.

Bacterial respiration can also occur in the absence of O₂, such as when soils become anoxic upon water saturation.^[5] Under these conditions, facultative anaerobic bacteria (capable of growing in the presence or absence of O₂) can transfer electrons from C substrates to such acceptors as nitrate (NO₃⁻), ferric iron (Fe³⁺), or sulfate (SO₄⁻). Anaerobic respiration can be tracked by measuring concentrations of oxidized or reduced forms of these electron acceptors coupled to CO₂ emission. Other process measurements related to C cycling include fermentation (use of partially oxidized organic compounds as sources of energy) and methane production and consumption.^[1,3] Measurements of N cycling processes include ¹⁵N isotope tracing to measure N₂ fixation to intracellular amino-nitrogen, N mineralization (net production of NH₄⁺ and NO₃⁻ during decomposition of organic matter), nitrification [oxidation of ammonia (NH₃) to NO₂⁻ and NO₃⁻],^[13] and denitrification (reduction of NO₃⁻ or NO₂⁻ to N₂ or the trace gas nitrous oxide).^[15] Chromatographic and mass spectrometric instruments, which provide fast and accurate measurements of specific compounds, also are used to track end products of degradation processes.^[3]

In lieu of culturing or microscopic enumeration, total soil microbial biomass content can be estimated by extracting and measuring certain chemical compounds from whole soils. Extraction of microbial phospholipids is one such approach, while measuring changes in extractable C before and after fumigating soils with chloroform is another.^[3] As a rule, soil microbial biomass is directly proportional to soil organic matter (SOM) content, making up about 1% of total soil organic C. Since stable SOM is estimated to contain 58% organic C,^[16] a soil containing 3.5% organic matter, e.g., would have about 20 g organic C kg⁻¹ and 0.2 g microbial biomass C kg⁻¹ soil. Increasing SOM content by amending soils with cover crop residues,

animal manures, or composts, e.g., is the most direct way to increase microbial biomass in agricultural soils.^[5]

Molecular (DNA-Based and RNA-Based) Microbiology

A universal classification system for bacteria arose during the 1970s–1980s from the work of Carl Woese, who collected as many pure cultures as possible to compare their genetic variations.^[17] Woese based his phylogenetic (evolutionary) tree on sequence variations in the rRNA strands found in ribosomes, the protein-synthesizing components of living cells.^[1] These rRNAs (and the corresponding rRNA genes that encode them on chromosomes) are useful as “molecular clocks” because they are universal (i.e., found in all organisms) and highly conserved (slow to change over time because they are essential to cell survival). Among the different rRNAs found in prokaryotic ribosomes, the 16S rRNA in the small ribosomal subunit has a convenient length for sequence alignments and comparisons (e.g., about 1540 nucleotides in the reference bacterium, *Escherichia coli*). In eukaryotes, the corresponding rRNA is encoded by the 18S rRNA gene (e.g., 1800–1900 base pairs).^[1] Woese’s phylogenetic tree showed how organisms formerly thought to be bacteria (e.g., methane-producing methanogens) are actually archaea, belonging to a domain that is as genetically distinct from bacteria as it is from animals and other eukaryotes.^[17]

DNA-Based Classification of Cultured Isolates

Today, classifying pure cultures of bacteria requires determining the nucleotide sequences of their 16S rRNA genes. This is done readily with polymerase chain reaction (PCR) amplification of genomic (chromosomal) DNA with rRNA-gene-specific oligonucleotide primers.^[1] Resulting sequences are then compared to others (i.e., accessions) in Internet-accessible databases, which are searched for highest nucleotide identities using computer algorithms like Basic Local Alignment Search Tool.^[18] Public databases include GenBank, maintained by the National Center for Biotechnology Information of the U.S. National Institutes of Health,^[19] the Ribosomal Database Project,^[20] and the European Nucleotide Archive (in Data Library for the European Molecular Biology Laboratory, <http://www.embl.de/>).

When a nearly full-length 16S rRNA gene sequence of a new isolate is found to have >98% nucleotide identity to that of an accession from a described culture, the two are considered to belong to the same species.^[21] Two isolates with 100% identical 16S rRNA sequences may represent distinctive genomic variants or strains within a species due to genomic variations outside of the 16S rRNA gene. (Species are defined as having 70% genome sequence identity.^[22]) In many searches, new DNA sequences exhibit low nucleotide identities to database accessions. Recognized taxonomic thresholds for 16S rRNA nucleotide identities are 94.5% for genus, 86.5% for family, 82% for order, 75.5% for class, and

75% for phylum.^[21] Within the domain *Bacteria*, a total of 89 phyla are recognized. This total includes 29 established phyla having cultured representatives and 60 candidate phyla, known to exist based on recovered DNA sequences but not represented by cultured isolates.^[22] It should be noted that the definition of a bacterial phylum is subjective and that classification systems are subject to change as new accessions are continually added to public databases (several hundred thousand sequences annually).^[21]

DNA-Based Characterization of Communities Without Culturing

In addition to their use in classifying cultured isolates, DNA-based methods enable characterization of microbial community composition in environmental samples without culturing.^[12] Such cultivation-independent methods involve analysis of mixtures of microbial DNA (i.e., metagenomes) extracted directly from a sample after lysing the indigenous organisms by chemical or physical breakage.^[23] Torsvik, Daae, and Goksoyr, at the University of Bergen, Norway, were the first to analyze whole DNA extracts to assess bacterial richness (numbers of different species) in soils. They measured reannealing rates of DNA mixtures to estimate that 1 g of forest soil harbored 4000–10,000 different bacterial genomes.^[24] Following this pioneering insight, many studies employed PCR amplification to prepare 16S rRNA clone libraries from extracts of soil microbial community DNA.^[12] Subsequent development of next-generation (automated, high-throughput) DNA sequencing technologies allowed generation of massive numbers of rRNA gene amplicons (e.g., 2000–20,000 short sequences, or reads, per soil sample).

Increasingly, rRNA gene sequences have been used to characterize soil microbial diversity on the basis of “operational taxonomic units” (OTUs), rather than species. Typically, 16S rRNA gene sequences that are 97–98% similar to each other are grouped within an OTU.^[25] Whereas soil bacterial richness refers to the number of different OTUs detected in a specified volume of soil, diversity indices take into account richness as well as relative abundances of sequences in different OTUs. Widespread and reproducible recovery of identical 16S rRNA gene sequences from diverse soils revealed the existence of uncultured bacteria representing candidate phyla in the bacterial phylogenetic tree.^[21,22] The application of high-throughput sequencing studies has enabled better understanding of the biogeography of soil bacteria^[26–28] and how soil bacterial diversity varies with scale.^[29]

Analyzing DNA or RNA provides different types of information about community composition. At any given time of soil sampling, most bacteria are *metabolically inactive* because of nutrient limitation. DNA-derived sequences thus represent bacterial taxa that are *present* in soils, while RNA-derived sequences provide better coverage of *actively transcribing* taxa, which tend to have higher ribosomal

contents per cell.^[1] Soil RNA extracts contain both rRNAs and messenger RNAs (representing functional gene transcripts prior to their translation into proteins). Extracted RNAs can be reverse-transcribed to generate copy DNAs from a metatranscriptome that represents genes being *expressed* at the time of soil sampling.^[1] Even without culturing, therefore, information about physiologies and metabolisms of extant members of candidate phyla can be obtained. Targeted high-throughput sequencing yields large numbers of amplicons from specific genes, while “whole genome shotgun” (WGS) sequencing yields short reads from whole genome mixtures. Short DNA reads (50–250 bp) can be computer-assembled into longer, contiguous fragments of genomes and sorted with bioinformatics tools.^[28] Classifications based on 16S rRNA genes may be modified using identities of housekeeping genes (which code for proteins with essential functions) or whole genomes.

Bacterial Diversity in Soils

Soil bacterial communities are composed of diverse species belonging to a wide range of phyla. Table 1 lists phyla reported in two extensive reviews of soil bacterial diversity and the percentages of total reads determined for each phylum in these two studies.^[7,28] Five established phyla were ubiquitous and abundant in soils (>5% of rRNA gene reads): *Proteobacteria*, *Acidobacteria*, *Actinobacteria*, *Bacteroidetes*, and *Verrucomicrobia*. Five established phyla were ubiquitous but less abundant (1–5% of reads): *Firmicutes*, *Planctomycetes*, *Chloroflexi*, *Gemmatimonadetes*, and *Cyanobacteria*. Other established phyla that were consistently detected but present at even lower percentages (<0.5%) were *Nitrospirae*, *Thermi*, *Chlorobi*, *Fibrobacteres*, as well as the candidate phyla OP11 and TM7. Bacterial taxa present at <0.5% are described as belonging to the “rare biosphere” and may exhibit distinctive growth responses in soils.^[11]

In general, only about 1% of soil microbial community DNA is typically derived from the *Archaea*, which have cell membranes and wall structures distinct from *Bacteria*.^[1,22] Fundamental knowledge about the physiological characteristics of pure cultures in established phyla is described in the multivolume resource, *Bergey's Manual of Systematic Bacteriology*.^[30–34] Table 1 cites the *Bergey's* volume containing relevant information about each phylum. Even without culturing, however, the increased use of WGS sequencing of environmental DNA is yielding important insights about unculturable organisms in such candidate phyla as TM7 and OP11 (Table 1). These taxa have small cell sizes (<0.2 micron), small genomes, and unusual ribosome compositions.^[35] Genome databases such as Genomes on Line Database^[36] and Genomic Encyclopedia of Bacteria and Archaea are yielding useful insights into how members of all phyla contribute to soil processes and life on Earth.^[22]

Of the 10 most abundant, established bacterial phyla found in soils, only 4 are readily culturable and represented by large numbers of isolates: *Proteobacteria*,

Table 1 Major bacterial phyla detected by their 16S rRNA genes in DNA extracts from surface soils

Phylum	Average (range) ^[7]	Average (range) ^[28]	Subgroups found in soils and examples of known characteristics and functions (list is not exhaustive)
Proteobacteria ^[30] well-represented by cultured isolates, but many remain uncultured	39 (10–77)	20 (6–31)	Gram-negative cell walls; four classes are abundant in soils (no <i>Epsilonproteobacteria</i>)
	19	9 (1–15)	<i>Alphaproteobacteria</i> [fix N ₂ ; endophytes; prey on bacteria; oxidize nitrite; oxidize methane (CH ₄)]
	11	4 (1–11)	<i>Betaproteobacteria</i> [fix N ₂ ; oxidize ammonia (NH ₃), denitrify; prey on bacteria; oxidize CH ₄]
	8	3 (1–10)	<i>Gammaproteobacteria</i> (denitrify; ammonify nitrite; plant pathogens; solubilize phosphate)
	3	3 (0.1–11)	<i>Deltaproteobacteria</i> (many anaerobes; reduce sulfate; prey on other bacteria)
Acidobacteria ^[31] Subdivision 1 members are readily cultured	20 (5–46)	20 (0.1–46)	Gram-negative cell walls; comprise at least eight subgroups, few cultured representatives; Subgroups 1, 4, and 6 most abundant in soils, with Subgroup 3 (<i>Solibacteres</i>) abundant in some soils; all aerobic except for Subgroup 8
Actinobacteria ^[32] (formerly called “actinomycetes”)	13 (0–34)	14 (2–41)	Gram-positive cell walls; filamentous or pleomorphic morphologies; three subgroups common in soil: <i>Actinobacteridae</i> (most abundant, many cultured, aerobic heterotrophs, produce antibiotics); <i>Acidimicrobidae</i> (few cultured, include acidophiles that oxidize ferrous iron); <i>Rubrobacteridae</i> (few cultured aerobic heterotrophs, possibly other physiologies)
Verrucomicrobia ^[31]	7 (0–21)	13 (0.2–40)	Five major classes; Class 2, <i>Spartobacteria</i> most common, aerobic heterotrophs, some are nematode symbionts; many uncultured members with unknown physiologies
Bacteroidetes ^[31] (CFB group)	5 (0–18)	11 (2–40)	Gram-negative cell walls; of four subgroups, three common in soil: <i>Sphingobacteria</i> most common (aerobes, anaerobes, or facultative; degrade chitin); <i>Flavobacteriia</i> (aerobic heterotrophs); <i>Cytophagia</i> (aerobic heterotrophs, degrade cellulose); <i>Bacteroidetes</i> (uncommon, but may be symbionts or commensals in soil fauna)
Chloroflexi ^[33]	3 (0–16)	2 (0.6–3)	Phylum may be as diverse as Proteobacteria; possibly eight major subgroups; only one isolate from soil (aerobic heterotroph)
Planctomycetes ^[31]	2 (0–8)	2 (0.1–4)	Lack cell walls; budding morphology; may possess intracellular, membrane-bound compartments; two major subgroups: Phycisphaerae (rhizosphere) and Planctomycetia (includes soil isolates; anammox bacteria that oxidize NH ₃ anaerobically)
Firmicutes ^[34]	2 (0–8)	0.5 (04)	Gram-positive cell walls; form heat-resistant spores; two major subgroups: <i>Bacilli</i> (readily cultured aerobically; lactic fermenters) and <i>Clostridia</i> (anaerobes)
Gemmatimonadetes ^[31]	2 (0–4)	2 (0.1–4)	Gram-negative cell wall; aerobic heterotrophs; one major group; few cultured; little known physiologies
Cyanobacteria ^[33]	< 1	3 (0–25)	Photosynthetic; fix N ₂ ; unicellular, colonial or filamentous morphologies; Gram-negative cell walls; may form biofilms on soil surfaces; symbiotic partners with fungi in lichens

Following “low-abundance” phyla are cosmopolitan but detected at frequencies of < 0.5% of DNA/RNA reads; also called “rare phyla”^[11,27]

Nitrospirae ^[33]	Includes <i>Nitrospira</i> spp. that oxidize nitrite and <i>Leptospirillum</i> spp. that oxidize ferrous iron for energy
Deinococcus-Thermi ^[33]	Includes radiation-resistant, thermophilic members; physiologies and functions in soil unknown
Chlorobi ^[33]	“Green sulfur bacteria” anaerobic photoautotrophs that oxidize sulfide or sulfur; physiologies and functions in soil unknown
Fibrobacteres ^[31]	Anaerobic, some chitinolytic, includes termite-gut inhabitants
OP11, TM7	Lineages in candidate phyla radiation characterized by small cell size (pass through 0.2-micron filters; small genomes indicative of fermentative physiologies ^[35])
Candidate phyla ^[22]	BRC1, NKB19, OP10, OS-K, SC3, SC4, TM6, WS2, WS3 (no cultured isolates)

Note: CFB, Cytophaga-Flavobacterium-Bacteroides group.

Actinobacteria, *Bacteroidetes*, and *Firmicutes* (Table 1).^[7] In soils, readily cultured bacteria are described as having “r-selected” growth strategies. Such organisms compete well for available C, grow quickly under high-nutrient (copiotrophic) conditions, and decline rapidly when nutrients are depleted.^[36] Members of the less readily cultured phyla, such as *Acidobacteria* and *Verrucomicrobia*, may therefore have “k-selected” lifestyles characterized by slower growth and better adaptation to low-nutrient (oligotrophic) conditions. Despite few cultured isolates, the phylum *Acidobacteria* is composed of as many as eight major classes,^[33,37] possibly encompassing as much phylogenetic breadth as *Proteobacteria*. The latter phylum comprises six major classes (alpha-, beta-, gamma-, delta-, epsilon-, and zeta proteobacteria) with more than 550 recognized genera,^[22,30] ranging from chemolithotrophs using reduced inorganic compounds for energy (e.g., NH_3 oxidizers and hydrogen oxidizers) to obligate symbionts that can only reproduce inside specific host organisms.^[1,30] Despite high physiological diversity observed among cultured *Proteobacteria*, the majority of their DNA sequences in soils belong to taxa that contain no cultured representatives.^[38]

BACTERIAL LIFE IN SOILS

Soil Microenvironment and Ecological Interactions

Bacteria are non-uniformly distributed in soils, being more abundant in surface horizons near plant roots and in zones rich in organic matter.^[3] Bacterial biomass in soil thus tends to reflect organic matter distribution, with bacterial numbers quickly declining with soil depth.^[3,5] Most bacteria live in soils as microcolonies (2–10 cells) tightly attached to surfaces,^[39] rather than as planktonic (free-floating) organisms in soil solution. By producing mucilaginous polysaccharides (i.e., mucigels), bacteria can become enmeshed with clays and organic colloids to initiate soil aggregation.^[3,5] Bacteria adhering to soil surfaces inside pores (Fig. 1) are protected from desiccation and grazing by protozoan and other faunal predators.

Soil bacterial habitats are shaped by soil structure, which is the 3-D arrangement of sand, silt, clay, and organic matter into aggregates and pores.^[16] Optimal bacterial growth conditions are provided by soils with favorable organic matter, water-holding capacity, and aeration, all of which are supplied by well-structured soils with a high proportion of macroaggregates (>250 microns) and macropores (>60 microns).^[16] Macroaggregate formation is hierarchical and biologically facilitated. Inner microaggregates, formed through the binding of clays by bacterial exopolysaccharides, are held together through the action of fungal hyphae and root hairs (Fig. 1). Extensive formation of soil macroaggregates increases macroporosity, thereby enhancing air and water infiltration and positive feedback effects on

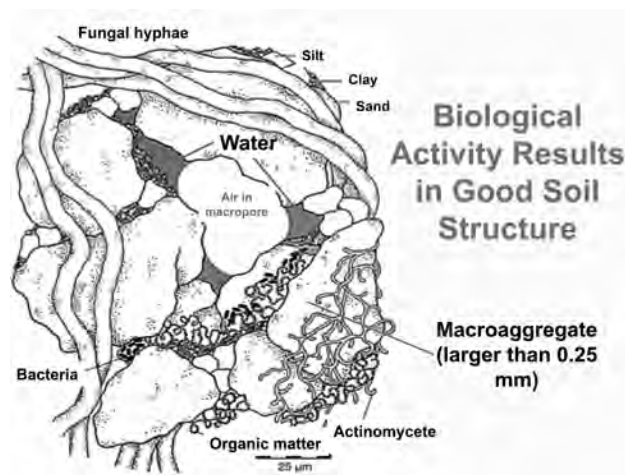


Fig. 1 Scale drawing of a typical soil aggregate. Sand (>50 μm), silt (2–50 μm), and clay (<2 μm) particles become cemented together with organic matter and microbial substances as a result of microbial activity. Note the pore in the center of the aggregate and the meniscus of water surrounding the airspace inside the pore. Bacteria are shown as dark rods and filaments (actinomycetes). Fungal filaments (hyphae) surround the aggregate. This illustrates how an aggregate can offer a diverse set of microsites for bacterial habitation over very short distances.

Source: From Sylvia, Fuhrmann, et al.^[5] ©2005 Prentice Hall.

root and microbial activity. Well-aggregated soils may provide more varied niches to support diverse bacterial metabolisms, because large aggregates can exhibit steep gradients in O_2 and redox potential.^[3,5]

Most soil bacteria require external sources of reduced C and N for growth and reproduction.^[1] Cyanobacteria are an exception in that they can use light energy to reduce CO_2 during photosynthesis and tap the resulting photosynthate for energy to reduce (fix) N_2 gas from the atmosphere.^[33,40] Cyanobacteria thus play critical roles as pioneer species on rock surfaces, particularly in symbioses with fungi (as lichens) or in associations with moss mats.^[41,42] The CO_2 respired by pioneer communities dissolves in rainwater to form carbonic acid, which helps solubilize and weather rocks. Essential metal nutrients released during rock weathering can form ligands with other organic acids produced by bacteria and their partners. Organic–metal ligands keep nutrients in solution and bioavailable, and they also serve to initiate formation of colloidal secondary minerals important for soil development.^[42]

Bacteria in soils participate in many diverse ecological interactions: as predators of other bacteria;^[43] as food sources for protozoans; as pathogens, parasites, or antagonists of other organisms (e.g., antibiotic producers); and as plant commensals or symbionts.^[3,44]

Establishment of interactions between bacteria and organisms in higher trophic groups clearly requires that these latter groups be maintained within soil habitats. Compared to quiescent soils under perennial vegetation, soils that undergo regular physical disturbance (e.g., by agricultural tillage)

provide less favorable habitats for larger organisms that are more desiccation-sensitive, thus resulting in less trophic complexity. Perennial vegetation also supplies soils with root exudates for longer periods of time than annual crops, so that greater substrate availability supports greater microbial biomass and activity. Cooperative interactions between bacteria also occur as a result of quorum-sensing chemicals^[44] or the transfer of genes via plasmids.^[45]

Bacterial Functions in Soil-Plant Systems

Even though bacteria have great functional importance in driving biogeochemical cycles, bacteria occupy no more than 0.4% of the total pore space of soils.^[39] Bacterial growth in most soils is limited by soluble C,^[3] although N availability may also influence activity.^[46] Although the bacterium's small size and high surface-to-volume ratio^[1] make it highly efficient in taking up nutrients, its limited mobility renders it dependent on the immediate environment for nutrient supply. It has been estimated that total C inputs from plant litter and root exudates in an agricultural soil may support only four to five bacterial growth cycles per year.^[3] In soils, average bacterial cell size (0.3–0.4 micron) can be less than half that observed for cultured bacteria growing in nutrient-rich medium in the laboratory.^[2,10]

Certain bacterial populations (e.g., in the genera *Pseudomonas* spp. and *Burkholderia* spp. of the *Gamma-proteobacteria* and *Betaproteobacteria*, respectively) are typically 10–100 times more numerous in zones immediately surrounding plant roots (i.e., rhizospheres rich in root-derived nutrients) than in bulk soils (not influenced by roots).^[3,5] Higher microbial densities in rhizosphere soils (soil adhering to excavated roots after physical shaking) relative to bulk soils are described as the rhizosphere effect. Plant roots exude or secrete chemicals that stimulate bacterial movement toward roots by chemotaxis. Root-associated bacteria carry out several functions that can be beneficial to plant nutrition and health, such as H_2PO_4^- mineral solubilization, N_2 fixation, iron chelation, phytohormone production, ethylene suppression, and facilitation of root colonization by mycorrhizal fungi.^[44,47,48]

Soil bacteria may colonize root surfaces or enter the roots through epidermal junctures to become endophytes. Some soilborne bacteria can be pathogenic (disease-causing) to plants (e.g., *Agrobacterium tumefaciens*, which causes crown gall, and *Erwinia carotovora*, which causes soft rots). However, many bacterial pathogens are introduced into soil with infected seeds or other planting materials, and bacteria as a group cause less overall economic damage to crops than pathogenic fungi, protists, or viruses.^[49,50] Some soils exhibiting high biological activity have been characterized as disease-suppressive, due to the action of microbial competitors and antagonists of plant pathogens.

Interest has increased in the possibility of using plant-growth-promoting rhizobacteria as seed or soil inoculants.^[44] High densities of bacteria (e.g., 10^6 viable cells per

milliliter), however, are generally required for efficacy.^[50] Although significant measurable results after inoculation of seeds or roots have been documented under controlled greenhouse conditions, comparable outcomes may not be reproduced in the field due to the great spatial heterogeneity of soils and their management histories.^[47] The ability of viable cells in an inoculated culture to survive and grow in a plant–soil system is dependent on diverse outcomes involving the physicochemical status of the soil and competition or cooperation with resident soil organisms. Inoculants must therefore contend with the entire assemblage of life in a soil, which is sometimes referred to as the “edaphon,” from the Greek word for “ground” or “soil,”^[4] encompassing all other bacteria, archaea, viruses, fungi, protists, nematodes, microarthropods, earthworms, and other soil fauna.

In subsoils below the root zone, bacteria are present but at lower densities (10^3 – 10^5 g^{-1}), and they may survive extended periods of soil nutrient deficit by invoking the following mechanisms: induction of low metabolic rates to reduce energy requirements,^[51] accumulation of intracellular polymers for C storage, production of extracellular polysaccharides to envelope and protect cells from desiccation, release of extracellular enzymes to solubilize nutrients, and formation of spores, cysts, and other resting-state structures.^[1] Some bacteria survive for decades in dried soils.^[5] Our knowledge that thousands of different bacterial species can coexist within 1 cm^3 of soil reflects a high degree of spatial and temporal heterogeneity in soil *micro-niches*, where appropriate conditions for growth may occur only sporadically during the growing season.^[11]

Despite the high complexity and heterogeneity of soil microbial communities, species diversity measures adopted from macroecology can be useful in their characterization; e.g., R. H. Whittaker's^[52] concept of gamma diversity (all species in a landscape encompassing multiple habitats) could be applied to all microbial taxa in a soil unit (e.g., one soil type in an agricultural field, pasture, or forest). Mean alpha diversity measures (species within habitats) could be determined separately for bacterial taxa in rhizospheres, bulk soils in surface horizons, and subsoils, with differences in these taxa used to characterize beta diversity (species differences across different soil habitats) of the soil unit.^[28]

Scientific motivation for understanding soil microbial community composition is based on the need to learn how soil microbiota vary across the landscape, particularly in response to management, and how biotic responses in turn affect ecosystem function and global change.^[53] The most important environmental factor influencing soil bacterial composition appears to be soil pH.^[27,28] Other factors such as vegetation, latitude, soil texture, and organic matter content do not appear to be correlated with total bacterial diversity. In general, neutral soils (pH 6–7) contain much greater bacterial diversity than soils with more extreme pH (<4 and >9). The availability of labile C, however, can influence changes in abundance of taxa associated with

r-selected and k-selected lifestyles. Addition of sucrose to forest soils, e.g., resulted in reproducible increases in *Betaproteobacteria* and *Bacteroidetes* (considered more copiotrophic) and decreases in *Acidobacteria* (more oligotrophic).^[54] Knowledge of how community composition is related to soil function will require integration of DNA sequence analyses with soil process measurements, knowledge of metabolic capabilities, and metadata (information about soils and sampling conditions).^[36]

CONCLUSION

Soils contain billions of bacteria: microscopic, single-celled prokaryotes that carry out diverse processes in nutrient cycling, including organic matter decomposition, secondary production, weathering and mineral nutrient release for plant uptake, and N₂ fixation. Although some bacteria cause diseases of animals and plants, most bacteria are beneficial and absolutely necessary for maintaining the biogeochemical balance of our environment. As technological advances enable more in-depth understanding of bacterial physiologies based on whole genomes, bacterial roles in soils will be more fully elucidated. Integrated approaches will be needed to understand the relationships between soil bacterial community composition and soil function.

REFERENCES

- Madigan, M.T.; Martinko, J.M.; Bender, K.S.; Buckley, D.H.; Stahl, D.A.; Brock, T. *Brock Biology of Microorganisms*, 14th Ed.; Benjamin Cummings: San Francisco, 2015.
- Portillo, M.C.; Leff, J.W.; Lauber, C.L.; Fierer, N. Cell size distributions of soil bacteria and archaea taxa. *Appl. Environ. Microb.* **2013**, *79* (24), 7610–7617. DOI:10.1128/AEM.02710-13.
- Paul, E.A. *Soil Microbiology, Ecology, and Biochemistry*, 4th Ed.; Elsevier: New York, 2015.
- Soil Science Society of America. *Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 2008.
- Sylvia, D.M.; Fuhrmann, J.J.; Hartel, P.G.; Zuberer, D.A. *Principles and Applications of Soil Microbiology*, 2nd Ed.; Prentice Hall: Upper Saddle River, 2005.
- Dworkin, M. Sergei Winogradsky: A founder of modern microbiology and the first microbial ecologist. *FEMS Microbiol. Rev.* **2012**, *36*, 364–379.
- Janssen, P.H. Identifying the dominant soil bacterial taxa in libraries of 16S rRNA and 16S rRNA genes. *Appl. Environ. Microb.* **2006**, *72* (3), 1719–1728. DOI:10.1128/AEM.72.3.1719-1728.2006.
- Olsen, R.A.; Bakken, L.R. Viability of soil bacteria: Optimization of plate-counting technique and comparison between total counts and plate counts within different size groups. *Microb. Ecol.* **1987**, *13* (1), 59–74. DOI:10.1007/BF02014963.
- Kepner, R.L.; Pratt, J.R. Use of fluorochromes for direct enumeration of total bacteria in environmental samples: Past and present. *Microbiol. Rev.* **1994**, *58*, 603–615.
- Bakken, L.R.; Olsen, R.A. The relationship between cell size and viability of soil bacteria. *Microb. Ecol.* **1987**, *13* (2), 103–114. DOI:10.1007/BF02011247.
- Aanderud, Z.T.; Jones, S.E.; Fierer, N.; Lennon, J.T. Resuscitation of the rare biosphere contributes to pulses of ecosystem activity. *Front. Microbiol.* **2015**, *6* (24), 1–11.
- Hugenholtz, P.; Goebel, B.M.; Pace, N.R. Impact of culture-independent studies on the emerging phylogenetic view of bacterial diversity. *J. Bacteriol.* **1998**, *180*, 4765–4774.
- Tiedje, J.M.; Asuming-Brempong, S.; Nüsslein, K.; Marsh, T.L.; Flynn, S.J. Opening the black box of soil microbial diversity. *Appl. Soil Ecol.* **1999**, *13* (2), 109–122. DOI: 10.1016/S0929-1393(99)00026-8.
- Robertson, G.P.; Wedin, D.; Groffman, P.M.; Blair, J.M.; Holland, E.A.; Nadelhoffer, K.J.; Harris, D. Soil carbon and nitrogen availability-nitrogen mineralization, nitrification, and soil respiration potentials. In *Standard Soil Methods for Long-Term Ecological Research*; Robertson, G.P., Coleman, D.C., Bledsoe, C.S., Sollins, P., Eds.; Oxford University Press: New York, 1999; 258–271.
- Tiedje, J.M. Denitrifiers. In *Methods of Soil Analysis. II. Microbiological and Biochemical Properties*; Weaver, R.W., Angle, J.S., Bottomley, P.S., Eds.; Soil Science Society of America: Madison, 1994; 245–267.
- Brady, N.C.; Weil, R. *The Nature and Properties of Soils*, 14th Ed.; Prentice Hall: Upper Saddle River, 2007.
- Woese, C.R.; Kandler, O.; Wheelis, M.L. Towards a natural system of organisms: Proposal for the domains Archaea, Bacteria, and Eucarya. *P. Natl. Acad. Sci. USA.* **1990**, *87* (12), 4576–4579.
- Altschul, S.F.; Gish, W.; Miller, W.; Myers, E.W.; Lipman, D.J. Basic local alignment search tool. *J. Mol. Biol.* **1990**, *215* (3), 403–410.
- Benson, D.A.; Karsch-Mizrachi, I.; Lipman, D.J.; Ostell, J.; Sayers, E.W. GenBank. *Nucleic Acids Res.* **2010**, *38* (Database), D46–D51.
- Cole, J.R.; Wang, Q.; Fish, J.A.; Chai, B.; McGarrell, D.M.; Sun, Y.; Brown, C.T.; Porras-Alfaro, A.; Kuske, C.R.; Tiedje, J.M. Ribosomal database project: Data and tools for high throughput rRNA analysis. *Nucleic Acids Res.* **2014**, *42* (D1), D633–D642.
- Yarza, P.; Yilmaz, P.; Pruesse, E.; Glöckner, F.O.; Ludwig, W.; Schleifer, K.H.; Whitman, W.B.; Euzéby, J.; Amann, R.; Rosselló-Móra, R. Uniting the classification of cultured and uncultured bacteria and archaea using 16S rRNA gene sequences. *Nat. Rev. Microbiol.* **2014**, *12* (9), 635–645. DOI:10.1038/nrmicro3330.
- Wu, D.; Hugenholtz, P.; Mavromatis, K.; Pukall, R.; Dalin, E.; Ivanova, N.N.; Kunin, V.; Goodwin, L.; Wu, M.; Tindall, B.J.; Hooper, S.D.; Pati, A.; Lykidis, A.; Spring, S.; Anderson, I.J.; D'haeseleer, P.; Zemla, A.; Singer, M.; Lapidus, A.; Nolan, M.; Copeland, A.; Han, C.; Chen, F.; Cheng, J.F.; Lucas, S.; Kerfeld, C.; Lang, E.; Gronow, S.; Chain, P.; Bruce, D.; Rubin, E.M.; Kyripides, N.C.; Klenk, H.P.; Eisen, J.A. A phylogeny-driven genomic encyclopedia of Bacteria and Archaea. *Nature* **2009**, *462*, 1056–1060.
- Bruns, M.A.; Buckley, D.H. Isolation and purification of microbial community nucleic acids from environmental samples. In *Manual of Environmental Microbiology*, 2nd Ed.; Hurst, C.J., Crawford, R.L., Knudsen, G.R.,

- McInerney, M.J., Stetzenbach, L.D., Eds.; American Society for Microbiology Press: Washington, 2002; 564–572.
24. Torsvik, V.; Daae, F.L.; Goksoyr, J. High diversity in DNA of soil bacteria. *Appl. Environ. Microb.* **1990**, *56*, 782–787.
 25. Schloss, P.D.; Westcott, S.L. Assessing and improving methods used in operational taxonomic unit-based approaches for 16S rRNA gene sequence analysis. *Appl. Environ. Microb.* **2011**, *77* (10), 3219–3226. DOI:10.1128/AEM.02810-10.
 26. Fierer, N.; Jackson, R.B. The diversity and biogeography of soil bacterial communities. *P. Natl. Acad. Sci. USA*. **2006**, *103* (3), 626–631. DOI:10.1073/pnas.0507535103.
 27. Lauber, C.L.; Hamady, M.; Knight, R.; Fierer, N. Pyrosequencing-based assessment of soil pH as a predictor of soil bacterial community structure at the continental scale. *Appl. Environ. Microb.* **2009**, *75* (15), 5111–5120. DOI: 10.1128/AEM.00335-09.
 28. Fierer, N.; Leff, J.W.; Adams, B.J.; Nielsen, U.N.; Bates, S.T.; Lauber, C.L.; Owens, S.; Gilbert, J.A.; Wall, D.H.; Caporaso, J.G. Cross-biome metagenomics analyses of soil microbial communities and their functional attributes. *P. Natl. Acad. Sci. USA*. **2012**, *109*, 21390–21395.
 29. Martiny, J.B.H.; Eisen, J.A.; Penn, K.; Allison, S.D.; Horner-Devine, M.C. Drivers of bacterial β -diversity depend on spatial scale. *P. Natl. Acad. Sci. USA*. **2011**, *108* (19), 7850–7854. DOI:10.1073/pnas.1016308108.
 30. Brenner, D.J.; Krieg, N.R.; Staley, J.T.; Garrity, G.M., Eds. *Bergey's Manual of Systematic Bacteriology*, 2nd Ed.; Springer-Verlag: New York, 2005; Vol. 2, parts A, B and C.
 31. Krieg, N.R.; Staley, J.T.; Brown, D.R.; Hedlund, B.P.; Paster, B.J.; Ward, N.L.; Ludwig, W.; Whitman, W.B., Eds. *Bergey's Manual of Systematic Bacteriology*, 2nd Ed.; Springer-Verlag: New York, 2010; Vol. 4.
 32. Whitman, W.B.; Goodfellow, M.; Kämpfer, P.; Busse, H.-J.; Trujillo, M.E.; Ludwig, W.; Suzuki, K.-I., Eds. *Bergey's Manual of Systematic Bacteriology*, 2nd Ed.; Springer-Verlag: New York, 2012; Vol. 5, parts A and B.
 33. Garrity, G.M.; Boone, D.R.; Castenholz, R.W., Eds. *Bergey's Manual of Systematic Bacteriology*, 2nd Ed.; Springer-Verlag: New York, 2001; Vol. 1.
 34. Vos, P.; Garrity, G.; Jones, D.; Krieg, N.R.; Ludwig, W.; Rainey, F.A.; Schleifer, K.-H.; Whitman, W. *Bergey's Manual of Systematic Bacteriology*, 2nd Ed.; Springer-Verlag: New York, 2009; Vol. 3.
 35. Brown, C.T.; Hug, L.A.; Thomas, B.C.; Sharon, I.; Castelle, C.J.; Singh, A.; Wilkins, M.J.; Wrighton, K.C.; Williams, K.H.; Banfield, J.F. Unusual biology across a group comprising more than 15% of domain Bacteria. *Nature* **2015**, *523* (7559), 208–211. DOI:10.1038/nature14486.
 36. Liolios, K.; Chen, I.-M.A.; Mavromatis, K.; Mavromatis, K.; Tavernarakis, N.; Hugenholtz, P.; Markowitz, V.M.; Kyrpides, N.C. The genomes on line database (GOLD) in 2009: Status of genomic and metagenomic projects and their associated metadata. *Nucleic Acids Res.* **2010**, *38* (Database), D346–D354. DOI:10.1093/nar/gkp848.
 37. Eichorst, S.A.; Breznak, J.A.; Schmidt, T.M. Isolation and characterization of soil bacteria that define Terriglobus gen. nov. in the phylum Acidobacteria. *Appl. Environ. Microb.* **2007**, *73* (8), 2708–2717. DOI:10.1128/AEM.02140-06.
 38. Spain, A.M.; Krumholz, L.R.; Elshahed, M.S. Abundance, composition, diversity and novelty of soil Proteobacteria. *ISME J.* **2009**, *3* (8), 992–1000. DOI:10.1038/ismej.2009.43.
 39. Foster, R.C. Microenvironments of soil microorganisms. *Biol. Fert. Soils*. **1988**, *6* (3), 189–203. DOI:10.1007/BF00260816.
 40. Pankratova, E.M. Functioning of cyanobacteria in soil ecosystems. *Eurasian Soil Sci.* **2006**, *39* (S1), S118–S127. DOI:10.1134/S1064229306130199.
 41. Banfield, J.F.; Barker, W.W.; Welch, S.A.; Taunton, A. Biological impact on mineral dissolution: Application of the lichen model to understanding mineral weathering in the rhizosphere. *P. Natl. Acad. Sci. USA*. **1999**, *96* (7), 3404–3411. DOI:10.1073/pnas.96.7.3404.
 42. Jackson, T.A. Weathering, secondary mineral genesis, and soil formation caused by lichens and mosses growing on granitic gneiss in a boreal forest environment. *Geoderma* **2015**, *251*–252, 78–91. DOI:10.1016/j.geoderma.2015.03.012.
 43. Casida, L.E., Jr. Minireview: Nonobligate bacterial predation of bacteria in soil. *Microb. Ecol.* **1988**, *15* (1), 1–8. DOI:10.1007/BF02012948.
 44. Goh, C.H.; Veliz Vallejos, D.F.; Nicotra, A.B.; Mathesius, U. The impact of beneficial plant-associated microbes on plant phenotypic plasticity. *J. Chem. Ecol.* **2013**, *39* (7), 826–839. DOI:10.1007/s10886-013-0326-8.
 45. Daane, L.L.; Molina, J.A.E.; Berry, E.C.; Sadowsky, M.J. Influence of earthworm activity on gene transfer from *Pseudomonas fluorescens* to indigenous soil bacteria. *Appl. Environ. Microb.* **1996**, *62*, 515–521.
 46. Colman, B.P.; Schimel, J.P. Drivers of microbial respiration and net N mineralization at the continental scale. *Soil Biol. Biochem.* **2013**, *60*, 65–76. DOI:10.1016/j.soilbio.2013.01.003.
 47. Cook, R.J.; Thomashow, L.S.; Weller, D.M.; Fujimoto, D.; Mazzola, M.; Banger, G.; Kim, D.S. Molecular mechanisms of defense by rhizobacteria against root disease. *P. Natl. Acad. Sci. USA*. **1995**, *92* (10), 4197–4201. DOI:10.1073/pnas.92.10.4197.
 48. Frey-Klett, P.; Garbaye, J.; Tarkka, M. The mycorrhiza helper bacteria revisited. *New Phytol.* **2007**, *176* (1), 22–36. DOI:10.1111/j.1469-8137.2007.02191.x.
 49. Kennedy, B.W.; Alcorn, S.M. Estimates of U.S. crop losses to prokaryotic plant pathogens. *Plant Dis.* **1980**, *64* (7), 674–676. DOI:10.1094/PD-64-674.
 50. Vidaver, A.K.; Lambrecht, P.A. Bacteria as plant pathogens. In *The Plant Health Instructor*; The American Phytopathological Society: St. Paul, 2004. DOI: 10.1094/PHI-I-2004-0809-01.
 51. Morita, R.Y. Bioavailability of energy and the starvation state. In *Starvation in Bacteria*; Kjelleberg, S., Ed.; Plenum Press: New York, 1993; 20–33.
 52. Whittaker, R.H. Evolution and measurement of species diversity. *Taxon* **1972**, *21* (2–3), 213–251. DOI:10.2307/1218190.
 53. Schmidt, T.M.; Waldron, C. Microbial diversity in soils of agricultural landscapes and its relation to ecosystem function. In *The Ecology of Agricultural Landscapes: Long-Term Research on the Path to Sustainability*; Hamilton, S.K., Doll, J.E., Robertson, G.P., Eds.; Oxford University Press: New York, 2015; 135–157.
 54. Fierer, N.; Bradford, M.A.; Jackson, R.B. Toward an ecological classification of soil bacteria. *Ecology*. **2007**, *88* (6), 1354–1364. DOI:10.1890/05-1839.

Biochar and Soil Characteristics

Atanu Mukherjee

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Biochar is a carbonaceous substance which can be obtained by the process of pyrolysis. Since mid-2000, extensive research on biochar has started and is being carried on because of its intended use as a soil amendment. Although there are uncertainties and concerns about biochar usage under field scale, studies claim improvement in soil quality. This entry outlines impacts of various biochars on selected soil chemical and physical properties, and trends in the results are identified, wherever possible. Selected data from 2010 to 2016 indicate that pH and cation exchange capacity (CEC) of the biochar-amended soils can be increased by up to 73% and 2-folds, with an average of 28% and 30%, respectively, compared to the control. A trend could be established that soil pH can be increased by biochar as amendment but such a trend could not be established with CEC due to conflicting results and lack of field-scale experiments. Data also confirm that higher biochar application rate could be more beneficial toward some soil qualities such as surface area (SA), however, no clear trend has been established. The SA and water-holding capacity (WHC) of the amended soils could be enhanced by up to 5.5- and 1.5-fold, with an average of 2-fold and 31%, respectively, compared to control soils. The WHC of biochar-amended soils is improved under laboratory incubation but not consistently under field studies. A clear trend of decrease in bulk density by up to 24% of the amended soils is observed. Although these data are to be used as representative, however, research priority of biochar amendment studies under field scale and over time is needed to establish trends in soil quality under biochar.

LITERATURE REVIEW

Since the mid-2000s, a number of studies have been done on biochar material itself as well as its effects on soil properties, greenhouse gas (GHG) emissions, and primary productivity, when amended to soil. Biochar is the carbonaceous product of biomass obtained by removing water and other volatile components when plant or animal biomass is subjected to heat treatment. This may occur in the presence of oxygen (hereafter referred to as combustion) or without oxygen (referred to as pyrolysis). However, all biochars are not the same and biochar characteristics mainly depend on the pyrolysis conditions they were made of and biomass types they were derived from.^[1] Thus, when biochars are amended with soils, the resultant soil/biochar mixtures behave differently in properties. This is the primary reason why intensive physical and chemical characterizations of biochar should be made before amending any particular soil type to investigate its effects on the properties of the amended soil. Researchers have made significant progress in understanding biochar's physical and chemical properties and factors such as combustion temperature, heating duration and rate, and biomass species combusted have all been found to influence the properties of biochars produced.

Biochars derived from animal origin (animal bone, poultry or cow manure/litter, and animal waste) are

characteristically different than biochars pyrolyzed from plant biomass.^[2] The key differences are higher phosphorus (P) and other metal availability and pH in the animal-derived biochars than those of plant-derived ones. However, for the plant-derived biochars, generally, lower temperature (<650°C) biochars are characteristically acidic in pH, high in volatile matter and surface functional groups, and low in surface area (SA) and ash content, while higher temperature (>650°C) biochars are usually high in pH, SA, and ash content with low in volatile matter and surface functional groups.^[1] However, some properties of biochar also depend on plant biomass types. The key properties governed by plant biomass types are cation or anion exchange capacities (CEC or AEC), amount of toxic metals, SA, and GHG emissions.^[1,3] A potential biochar should always be amended with the right soil. For an example, an acidic soil should be amended with a biochar with high pH and vice versa. However, due to the lack of consensus, biochar/soil studies have not been designed the way that the outcome becomes most beneficial and lack of field-scale data and the high cost of biochar production have been identified as the constraints of large-scale biochar application.^[4] Noting this in mind, an overview of changes in selected soil chemical and physical properties under biochar amendment is discussed briefly in the following sections based on 2010–2016 data. Although all the soil physicochemical properties are

not given equal attention, an attempt is made to find trends under selected properties.

Biochar Impacts on Soil Chemical Properties

A number of biochar/soil studies demonstrated that biochar can significantly influence amended soils' chemical properties (Table 1). Soil fertility can be improved by biochar addition, reduction of soil acidity, and enhancement of plant-available nutrients. For example, biochars derived from a number of plant and animal origin increased amended soils' pH by up to 73% with an average increase of 28% (Table 1), depending on the application rate.^[5–10] However, although there is conceived knowledge that biochar-amended soil would have higher pH than control soil, it is not always necessarily true. Some biochars (sometimes having pH < 7.0) tend to decrease the pH of the amended soils. For an example, poultry manure biochar pyrolyzed at 300°C decreased pH of amended calcareous soil in a pot study and beech wood biochar produced at 500°C decreased pH of amended loamy silt soil by 1.3% compared to control.^[10,11] However, from the published data, a trend could be drawn that pH of most soils could be increased by biochar addition which is supported by a meta-analysis (n = 146, p < 0.001) indicating that biochar works best in acidic compared to alkaline soil to increase the amended soil's pH.^[12]

The major required soil nutrients for plants are nitrogen (N), P, and potassium (K). Various greenhouse and laboratory experiments exhibited enhancement of soil nutrients level by biochar addition. Biochar also either increased or retained soil nutrients as documented in field and pot experiments.^[11,13] For example, poultry manure biochar pyrolyzed at 300°C decreased the concentration of plant-available iron of the amended calcareous soil but increased plant-available P, zinc, copper, and manganese concentrations and biochar increased the level of exchangeable cations (K, calcium, and magnesium) in the amended soil under a greenhouse pot experiment.^[11] In another field study, compost biochar significantly increased water-soluble N and total and dissolved organic carbon in the amended plots compared to the control calcareous soil in two consecutive years.^[14] Similarly, significant increases of K⁺ and CEC upon the addition of 2.5–10% (w/w) biochar for soils from three regions were observed under corn cob biochar pyrolyzed at 350°C.^[7] Although biochar itself contains nutrients and serves as an adsorbent of the soil nutrients, it also can release nutrients slowly over time and the amount and forms of the nutrients released from biochar depend on biomass type and production temperature of the biochar. For example, biochars derived from a grass type and pyrolyzed at lower temperatures released higher nutrients compared to the biochars derived from woody biomass and pyrolyzed at higher temperatures.^[15]

The extent of nutrient retention or sorption and slow release or leaching over time depends on sorption and the

exchange capacity of biochar/soil mixtures, when amended. The CEC of pure biochar made from diverse biomass sources ranged from 10.0 cmol_c kg⁻¹ to 69.2 cmol_c kg⁻¹ showing an increase in CEC of amended soil by up to 2-folds (Table 1) compared to the control.^[5,7,9,10] However, very few studies documented AEC of biochar-amended soil, which is considered a research priority. On the other hand, very little enhancement or even decadence of CEC by up to 41% (Table 1, negative sign indicates decrease) was also reported in some cases. For example, biochar derived from beech (*Fagus* spp.) and oak (*Quercus* spp.) and pyrolyzed at 500°C, 250°C, and 650°C actually decreased CEC of the amended soils by 7.6%, 25.7%, and 41%, respectively, when amended to loamy silt and forest Spodosols.^[9,10] A biochar derived from poultry litter and pyrolyzed at 450°C decreased the CEC of Frederick silt loam soil under forage and pepper in the first year; however, CEC was increased by up to 9.5% during the second year of the experiment, identifying the importance of biochar field aging,^[16] as aged biochars demonstrate characteristics that significantly improve the soil properties in a longer run as biochar becomes aged over time.^[8,9] Nevertheless, more field-aging studies need to be done focusing on both CEC and AEC of biochar-amended soils, as very few data are available to make any credible interpretations. While a little enhancement of CEC was observed in a number of cases, many times the increment of CEC was not significantly higher compared to the control and thus a clear trend of change in CEC cannot be established.

Biochar Impacts on Soil Physical Properties

While a lot of attention has been given to effects of biochar on soil chemical properties, data are scarce on biochar impacts on soil physical properties, especially under field scale.^[3] Biochars usually are characteristically very light materials with high porosity and SA, which are likely to alter soil physical properties such as bulk density (BD), water-holding capacity (WHC) or available water capacity, SA, and penetration resistance (PR). Only BD and WHC have been investigated in most studies, while a few studies documented SA and PR of biochar-amended soils (Table 1). Although biochars possess high SA, there is a lack of data reported on SA of biochar-amended soils. Increase in SA of the amended soils by up to 5.5-fold (Table 1) has been documented,^[17–19] which indicates advantage of sorption and water/nutrient storage capacity of the biochar-amended soils due to greater extent of SA available for nutrient sorption or storage, when amended. A rate-dependent SA increment trend was noted by at least two different studies. For example, oak and hickory (*Carya* spp.) mixed woods when pyrolyzed at 500°C and mixed with Clarion fine loamy soil increased SA of the amended soil by 3.8% when biochar application rate was increased from 0.5% to 1% (w/w) and jarrah (*Eucalyptus marginata*) woods pyrolyzed at 600°C increased SA of the amended sandy soil dramatically by

Table 1 Changes in soil quality parameters under biochar (numbers in parenthesis indicate average values, and negative sign indicates decrease).

Properties	Biochar types	Soil types	Application rate (w/w)	Range (average)			References	
				Properties				
				Biochar	Soil	Amended soil		
pH	Coffee at 400°C, corn cob at 350°C, beech at 500°C, white lead at 700°C, white lead at 700°C, and oak at 650°C	Acid mine drainage, Mongu, Kaoma, Mkushi, loamy silt, Typic Paleudults, Typic Paleudults, and Crosby silt loam	0.5–10 (4.8)	7.0–9.9 (8.7)	4.0–7.5 (5.2)	4.7–7.5 (6.5)	–1.3–72.7 (27.9)	[1–5]
CEC (cmol _c kg ^{–1})	Beech at 500°C, white lead at 700°C, oak at 250°C, oak at 650°C, grass at 250°C, grass at 650°C, poultry litter at 450°C, and corn cob at 350°C	Loamy silt, Typic Paleudults, forest Spodosol, sandy Entisol, Frederick silt loam, Mongu, Kaoma, and Mkushi	0.2–10 (2.6)	10.0–69.2 (37.2)	1.4–22.5 (13.5)	1.8–25.0 (11.7)	–41.0–100 (30.2)	[2–4,6,7]
CO ₂ -SA (m ² g ^{–1})	Oak and hickory at 500°C, oak at 650°C, and Jarrah wood at 600°C	Clarion fine loamy, Crosby silt loam, and sandy	0.5–2.7 (1.1)	4.4–604 (206.7)	1.3–130 (63.1)	2.7–153 (69.9)	2.3–546.2 (99.9)	[5,8,9]
BD (kg m ^{–3})	Hardwoods at 400°C, corn cob at 350°C, white lead at 700°C, and oak at 650°C	Calcareous, fluvo-aquic loamy, sandy loam, Mongu, Kaoma, Mkushi, Typic Paleudults, and Crosby silt loam	0.5–10 (4.1)	0.2–0.7 (0.5)	1.0–1.7 (1.3)	0.8–1.6 (1.2)	–23.5–0 (–12.2)	[2,4,5,10–12]
WHC (%)	Corn stover at 350°C, oak at 550°C, hardwoods at 400°C, corn cob at 350°C, and oak at 650°C	Tokomaru silt loam, Egmont silt loam, weld silt loam, Normania loam, sandy loam, Mongu, Kaoma, Mkushi, and Crosby silt loam	0.5–10 (4.6)	—	4.5–61 (41.5)	7.1–73 (35.7)	–14.6–148.9 (30.5)	[2,5,12–15]

2.1-fold when the biochar application rate was increased from 0.45% to 2.7% (w/w).^[17,18] These data confirm that higher biochar application rate may be more beneficial toward improvement of some soil characteristics such as SA. However, any trend with SA under biochar amendment is to be established without having enough field data and long-term studies.

The BD of the biochar-amended soils is likely to be reduced due to high porosity of the biochars, and a decrease in BD of a range of amended soils was observed by up to 24% (Table 1, negative sign indicates decrease) compared to control soils.^[5,7,19–21] For example, wheat straw derived biochars pyrolyzed at 350–550°C, when amended with variety of soils decreased BD by up to 14% depending on the rate and duration of the experiment.^[20,21] However, significant decrease in BD of the amended soil was not observed in some cases. For example, corn cob biochar pyrolyzed at 350°C was not able to reduce the BD of the amended Mongu soil with 2.5% application rate; however, progressive reduction of BD was observed when application rate increased up to 10% (w/w).^[7] It was concluded earlier that a biochar application rate of 2% (w/w) could be sufficient to reduce the BD of the amended soils,^[3] but the data^[7] indicate more work needs to be carried out on a variety of soil/biochar combinations, especially under field scale. However, despite the complexity of application rates, decrease in BD of amended soils is likely as an observed trend from the data.

On the other hand, many studies indicated that WHC of the biochar-amended soils can be significantly improved by up to 1.5-fold compared to control except for a few cases where WHC of the amended soils decreased to some extent (Table 1). For example, several soil incubation studies indicated that biochars derived from corn stover, oak, and mixed hardwoods enhanced WHC of the resultant soils by up to 33%, 26%, and 20%, respectively, depending on biochar pyrolysis temperatures or application rates.^[7,8,13,19,22,23] A similar response was also observed under field-scale experiments. The amended soils had improved WHC when corn cob biochar pyrolyzed at 350°C applied to three different soil types under conservation tillage system and three application rates (2.5–10%, w/w) with progressive higher WHC observed with a higher application rate,^[7] and a similarly progressive increase in WHC with higher biochar application rate was observed in other studies by Case et al.^[22] and Herath, Camps-Arbestain, and Hedley.^[23] In contrast, a biochar amendment study documented decrease in WHC of the amended soil. Oak biochar pyrolyzed at 650°C decreased WHC of the amended soils by 15% in the first year,^[19] but increased the same by 22% in the second year of the field trial,^[8] although the increment was not significant ($p > 0.05$). These results demonstrate that: 1) the right biochar should be used for the right soil to see the desired benefit; and 2) biochar field aging is important and should be considered in research studies. Additionally, biochar

amendment studies, while aiming for improving soil quality, should have their main goal as increasing crop yield, and significant uncertainties exist at this point.^[4] A meta-analysis (data collected from 23 studies until March 1, 2010) indicates a wide range of outcomes (–28% decrease to +39% yield increase) with a mean yield increase of 10%, and the greatest positive results were observed from acidic to neutral soils with coarse to medium textures indicating liming and improved WHC benefits of soil under biochar.^[24]

SUMMARY

This entry identifies that clear trends under biochar amendment could possibly be established for only pH and BD of the amended soils. Lack of field data on CEC, soil nutrients, aggregation, SA, and WHC is a constraint to validate any trend on these soil attributes. Representative data also indicate that biochar application rate and field aging have significant impacts on improvement of most soil quality parameters. However, the biochar production process is expensive and unless a specific threshold application rate is established, large-scale recommendation is not possible.

REFERENCES

1. Mukherjee, A.; Zimmerman, A.R.; Harris, W.G. Surface chemistry variations among a series of laboratory-produced biochars. *Geoderma* **2011**, *163* (3–4), 247–255.
2. Vassilev, N.; Martos, E.; Mendes, G.; Martos, V.; Vassileva, M. Biochar of animal origin: A sustainable solution to the global problem of high-grade rock phosphate scarcity? *J. Sci. Food Agr.* **2013**, *93* (8), 1799–1804.
3. Mukherjee, A.; Lal, R. Biochar impacts on soil physical properties and greenhouse gas emissions. *Agronomy* **2013**, *3* (2), 313–339.
4. Mukherjee, A.; Lal, R. The biochar dilemma. *Soil Res.* **2014**, *52* (3), 217–230.
5. Jien, S.H.; Wang, C.S. Effects of biochar on soil properties and erosion potential in a highly weathered soil. *Catena* **2013**, *110*, 225–233.
6. Kim, M.-S.; Min, H.-Gi.; Namin, K.; Jeongsik, P.; Lee, S.-H.; Bak, G.-I.; Kim, J.-G. The effectiveness of spent coffee grounds and its biochar on the amelioration of heavy metals-contaminated water and soil using chemical and biological assessments. *J. Environ. Manage.* **2014**, *146* (0), 124–130.
7. Martinsen, V.; Mulder, J.; Shitumbanuma, V.; Sparrevik, M.; Børresen, T.; Cornelissen, G. Farmer-led maize biochar trials: Effect on crop yield and soil nutrients under conservation farming. *J. Plant Nutr. Soil Sc.* **2014**, *177* (5), 681–695.
8. Mukherjee, A.; Lal, R.; Zimmerman, A.R. Impacts of 1.5-year field aging on biochar, humic acid, and water treatment residual amended Soil. *Soil Sci.* **2014**, *179* (7), 333–339.

9. Mukherjee, A.; Zimmerman, A.R.; Hamdan, R.; Cooper, W.T. Physicochemical changes in pyrogenic organic matter (biochar) after 15 months of field aging. *Solid Earth* **2014**, *5* (2), 693–704.
10. Prommer, J.; Wanek, W.; Hofhansl, F.; Trojan, D.; Offre, P.; Urich, T.; Schleper, C.; Sassmann, S.; Kitzler, B.; Soja, G.; Hood-Nowotny, R.C. Biochar decelerates soil organic nitrogen cycling but stimulates soil nitrification in a temperate arable field trial. *PloS One* **2014**, *9* (1), e86388.
11. Inal, A.; Gunes, A.; Sahin, O.; Taskin, M.B.; Kaya, E.C. Impacts of biochar and processed poultry manure, applied to a calcareous soil, on the growth of bean and maize. *Soil Use Manage.* **2015**, *31* (1), 106–113.
12. Biederman, L.A.; Harpole, W.S. Biochar and its effects on plant productivity and nutrient cycling: A meta-analysis. *Global Change Biol. Bioenerg.* **2013**, *5* (2), 202–214.
13. Zheng, J.Y.; Stewart, C.E.; Cotrufo, M.F. Biochar and nitrogen fertilizer alters soil nitrogen dynamics and greenhouse gas fluxes from two temperate soils. *J. Environ. Qual.* **2012**, *41* (5), 1361–1370.
14. SÁnchez-García, M.; Sánchez-Monedero, M.A.; Roig, A.; López-Cano, I.; Moreno, B.; Benitez, E.; Cayuela, M.L. Compost vs biochar amendment: A two-year field study evaluating soil C build-up and N dynamics in an organically managed olive crop. *Plant Soil* **2016**, 1–14. DOI:10.1007/s11104-016-2794-4.
15. Mukherjee, A.; Zimmerman, A.R. Organic carbon and nutrient release from a range of laboratory-produced biochars and biochar soil mixtures. *Geoderma* **2013**, *193–194*, 122–130.
16. Revell, K.T.; Maguire, R.O.; Agblevor, F.A. Field trials with poultry litter biochar and its effect on forages, green peppers, and soil properties. *Soil Sci.* **2012**, *177* (10), 573–579.
17. Dempster, D.N.; Gleeson, D.B.; Solaiman, Z.M.; Jones, D.L.; Murphy, D.V. Decreased soil microbial biomass and nitrogen mineralisation with Eucalyptus biochar addition to a coarse textured soil. *Plant Soil* **2012**, *354* (1–2), 311–324.
18. Laird, D.A.; Fleming, P.; Davis, D.D.; Horton, R.; Wang, B.; Karlen, D.L. Impact of biochar amendments on the quality of a typical Midwestern agricultural soil. *Geoderma* **2010**, *158* (3–4), 443–449.
19. Mukherjee, A.; Lal, R.; Zimmerman, A.R. Effects of biochar and other amendments on the physical properties and greenhouse gas emissions of an artificially degraded soil. *Sci. Total Environ.* **2014**, *487*, 26–36.
20. Zhang, A.F.; Bian, R.; Pan, G.; Cui, L.; Hussain, Q.; Li, L.; Zheng, J.; Zheng, J.; Zhang, X.; Han, X.; Yu, X. Effects of biochar amendment on soil quality, crop yield and greenhouse gas emission in a Chinese rice paddy: A field study of 2 consecutive rice growing cycles. *Field Crop. Res.* **2012**, *127*, 153–160.
21. Zhang, A.F.; Liu, Y.; Pan, G.; Hussain, Q.; Li, L.; Zheng, J.; Zhang, X. Effect of biochar amendment on maize yield and greenhouse gas emissions from a soil organic carbon poor calcareous loamy soil from Central China Plain. *Plant Soil* **2012**, *351* (1–2), 263–275.
22. Case, S.D.C.; McNamara, N.P.; Reay, D.S.; Whitaker, J. The effect of biochar addition on N₂O and CO₂ emissions from a sandy loam soil—The role of soil aeration. *Soil Biol. Biochem.* **2012**, *51*, 125–134.
23. Herath, H.; Camps-Arbestain, M.; Hedley, M. Effect of biochar on soil physical properties in two contrasting soils: An Alfisol and an Andisol. *Geoderma* **2013**, *209*, 188–197.
24. Jeffery, S.; van der Velde, M.; Bastos, A.C. A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis. *Agr. Ecosyst. Environ.* **2011**, *144* (1), 175–187.

Biochar and Soil Quality

David A. Laird

Department of Agronomy, Iowa State University, Ames, Iowa, U.S.A.

Jeffrey M. Novak

*Coastal Plain Soil, Water and Plant Conservation Research Center,
U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
Florence, South Carolina, U.S.A.*

Abstract

Biochar is a form of char that is used for environmental applications, principally as a soil amendment or for water treatment. It is produced by the incomplete combustion of biomass and is primarily a mixture of single- and condensed-ring aromatic carbon with lesser amounts of aliphatic carbon, hydrogen, oxygen, nitrogen, and other elements. The addition of biochar to soils returns most of the nutrients that are removed from the soil when biomass is harvested. Biochar has the capacity to retain bioavailable water and is a very strong adsorbent of plant nutrients and dissolved organic compounds. It is an effective soil conditioner that reduces soil bulk density and increases the capacity of soils to retain and supply nutrients and water to growing plants. Because of these positive effects on soil quality, most studies have reported enhanced plant growth for soils amended with biochar; however, caution is urged as biochar quality is critical. Some biochars have the capacity to tie up nutrients whether by immobilization, by nutrient adsorption, or by raising the soil pH to the point that the nutrients are no longer bioavailable.

Biochar, also known as charcoal, black carbon (C), soot, and char, is a broad class of materials produced from the incomplete combustion or pyrolysis of organic materials such as wood, wood by-products, plant residue, manure, petroleum, and petroleum by-products.^[1] Of the terms mentioned, char is the most general and simply implies a particulate pyrogenic (formed by fire) material dominated by a mixture of single- and condensed-ring aromatic C with lesser amounts of aliphatic C, hydrogen (H₂), oxygen (O), nitrogen (N), and other elements. Black C implies a particulate material found in soils and sediments, which resembles char but cannot unequivocally be shown to be pyrogenic. Soot is reserved for char particles that form by condensation from exhaust vapors of open combustion or an internal combustion engine. Charcoal is char that is used as a fuel for cooking or to produce heat. And, biochar is char that is used for environmental applications, principally as a soil amendment or for water treatment.

Biochar has been used as a soil amendment for several thousand years. Extensive regions of Terra Preta de Indio soils in the Amazon basin are Anthrosols made by pre-Columbian Amerindian farmers.^[2,3] In addition to high concentrations of biochar, the Terra Preta soils also contain large amounts of biogenic (formed by living organisms) humic materials and high concentrations of plant nutrients, notably phosphorus (P), which is often deficient in the Oxisols from which the Terra Preta soils were made. Perhaps the most important feature of the Terra Preta soils

is their enhanced capacity to retain and recycle both nutrients and water, and hence their greater fertility and capacity to sustainably support agricultural production relative to the native Oxisols.^[3,4] The pre-Columbian Amerindian farmers of the Amazon are believed to have practiced a form of slash-and-char agriculture whereby they produced and incorporated large quantities of biochar along with fish bones and various organic manures into the native Oxisols in a deliberate attempt to enhance the soil quality. More than 500 years after the cessation of the practices that led to their genesis, the Terra Preta soils are some of the most productive soils of the Amazon and require lower inputs of nutrients for sustainable agricultural and horticultural production compared with the Oxisols from which they were made.

The properties of biochar vary substantially depending on the source of biomass, the rate at which it is heated, the maximum temperature of heating, and the extent to which volatiles produced during pyrolysis are separated from the biochar prior to cooling. The level of aromaticity directly influences the stability of biochar in soil environments. Other properties of biochar that influence soil quality are particle size, porosity, surface area, the density and types of surface functional groups, the concentrations of biologically available and biologically active compounds in the biochar, and the concentrations and forms of inorganic bases that are admixed with the biochar.

During pyrolysis, most plant nutrients in a biomass feedstock are concentrated in the biochar fraction. Inorganic bases, including calcium ions (Ca^{2+}), magnesium ions (Mg^{2+}), and potassium ion (K^+), and most trace elements form oxides and/or hydroxides that are occluded within biochar pores or admixed as discrete particles with the biochar. Approximately, one-half of the N present in the biomass feedstock is partitioned into the biochar during pyrolysis. The other half of the N is volatilized during pyrolysis and then condensed with the bio-oil fraction. Much of the N in biochar is present as heterocyclic N, which adds base functionality to biochar, and is a slowly released source of N fertility in soils. Less is known about the forms and function of sulfur in biochar. Thus, the use of biochar as a soil amendment partially closes the nutrient cycle by returning most of the nutrients that are removed from the soil when biomass is harvested.

The base oxides and hydroxides in biochar are both a source of plant nutrients and a liming agent when the biochar is applied to the soil. Thus, both the fertility contribution and the calcium carbonate equivalent of biochar vary with the concentration of base oxides and hydroxides in the biochar, which primarily depends on the concentration of the bases in the initial biomass feedstock. Base concentrations are typically low in biochars produced from softwoods such as pine, intermediate in biochars produced from hardwoods, and high in biochars produced from grasses and crop residues.^[5]

Freshly made biochar is hydrophobic. On exposure to water and O in soil environments, the surfaces of fresh biochar particles are oxidized leading to the formation of O-containing surface functional groups especially carboxylate and hydroxyl groups.^[6,7] Carboxylate groups are weak acids with pK_a values typically between 4 and 7. Hence in alkaline, neutral, and mildly acidic soils, the carboxylate groups disassociate and produce a negative charge on the surfaces of biochar particles. The high surface area and high density of negative surface charge add cation exchange capacity (CEC) to soils, and together with other polar surface functional groups that form during surface oxidation, the carboxylate groups transform the biochar from hydrophobic to hydrophilic. The rate at which this transformation occurs has not been quantified and will likely depend on other properties of the biochar such as the level of condensed aromatic C and concentrations of O, N,

and other elements that are covalently bound to the aromatic C matrix. Measuring the CEC of fresh biochar is complicated because the CEC is dependent on both pH and the extent of surface oxidation, both of which vary with time after the biochar has been placed in an aqueous solution. Furthermore, the adsorption of both organic and inorganic compounds by biochar in soil environments alters the surface properties of the biochar.^[8] The complexation of polyvalent metal cations by carboxylate and/or other O-containing surface functional groups can transform a surface charge site from negative to positive, effectively creating anion exchange sites. Amphiphilic organic molecules may be adsorbed on biochar particles and oriented with the lipophilic portions of the organic molecules interacting with non-polar regions of the biochar and the polar portions of the organic molecules oriented toward the aqueous phase. Thus, adsorption of organic molecules may contribute additional surface charge to biochar particles and/or alter the hydrophilic–hydrophobic character of surfaces of biochar particles. Because of the processes discussed above, the surface properties of biochar change both with the length of time biochar has been in the soil and with the chemical properties of the soil. Although directly quantifying meaningful properties such as CEC of fresh biochar is difficult, the impact of biochar additions on soil properties can be readily measured after the biochar has had an opportunity to equilibrate with the soil environment. Several studies have measured significantly higher CECs for soils amended with biochar than control soils.^[8,9]

Biochar in various forms is widely used as an adsorbent for removing both inorganic and organic contaminants from wastewater and for purification of drinking water. When mixed with soil, biochar functions in the same way. This is illustrated simply by looking at the color of leachate collected from soil columns following a swine manure amendment (Fig. 1). After the manure amendment, water leaching from the control columns containing soil without biochar was turbid indicating high levels of dissolved organic material, while the water leaching from soil columns amended with as little as 5 g kg^{-1} biochar was nearly clear. The biochar also reduced leaching of nitrate, K^+ , Ca^{2+} , Mg^{2+} , P, and several other nutrients.^[10] Of particular note, leaching of P was reduced by 40–70% relative to the controls, as the rate of biochar amendment increased from 5 to 20 g kg^{-1} . The reductions in nutrient



Fig. 1 Leachate collected from soil columns 2 days after a manure amendment. Column treatments are as follows: C0, no manure and no biochar; C0M, 5 g kg^{-1} manure and no biochar; C1M, 5 g kg^{-1} manure and 5 g kg^{-1} biochar; C2M, 5 g kg^{-1} manure and 10 g kg^{-1} biochar; and C3M, 5 g kg^{-1} manure and 20 g kg^{-1} biochar.

Source: From Laird, Fleming, et al.^[9,10]

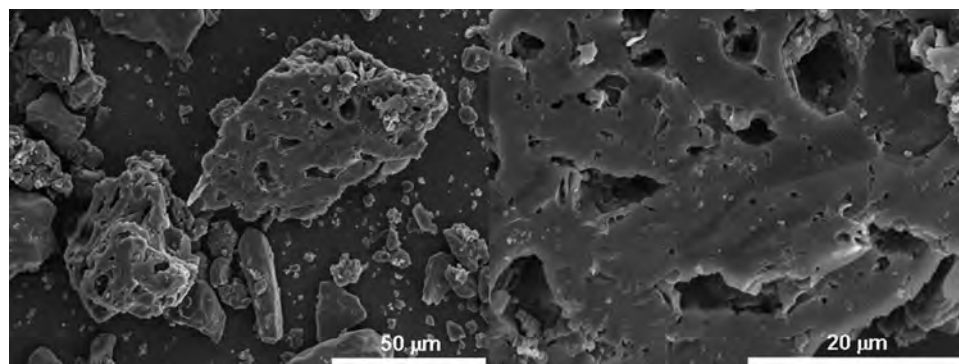


Fig. 2 Scanning electron micrographs showing two biochar particles that were made from pyrolysis of pecan shells (left) and a close-up of the surface of one of the biochar particles (right). The images show a network of internal pores.

leaching were associated with increases in bioavailable nutrients in the soils measured 500 days after the biochar amendments.^[9] Comparisons of nutrient leaching under Terra Preta soils with adjacent control soils have shown similar nutrient leaching reductions.^[3,4]

Porosity is a key property of biochar (Fig. 2). Some of the pores in the original biologic tissue are preserved in the residual biochar. Furthermore, the desiccation of biologic tissue and the pyrolytic emission of structural water, carbon dioxide (CO₂), carbon monoxide, and H₂ from biologic tissue during pyrolysis create even more internal porosity within biochar particles. Internal porosity lowers the effective particle density and greatly increases the effective surface area of biochar. In soil environments, the low particle density and the high internal porosity of biochar help to reduce the bulk density of soils and increase the capacity of soils to retain bioavailable water, nutrients, and organic molecules, and the pores can provide a habitat for soil microorganisms that facilitate nutrient cycling.

The ability of biochar amendments to reduce the bulk density of soils is an important mechanism by which biochar enhances the physical condition of soils. The bulk density of tightly packed biochar powder (<1 mm particle size) is about 0.5 g cm⁻³, but the bulk density of biochar varies substantially depending on the ash content, internal porosity, and particle size. The mixing of biochar with a mineral soil reduces the bulk density of the soil by a factor that is significantly greater than that would be predicted from the volumetric mixing of the low-density biochar with the higher density mineral soil. The cause of this synergistic effect is not entirely clear.

One of the more remarkable features of biochar is the apparent enhancement of microbial activity, especially mycorrhizal fungi,^[11] which increase soil organic matter turnover and nutrient cycling. The cause of this increase in soil microbial activity is not clear; however, several factors such as lower bulk densities, improved aeration, higher pH values due to the liming effect of biochar, sorption of bioavailable dissolved organic matter by biochar particles, and the porosity in biochar providing a favorable habitat for soil microorganisms all may contribute to the observed enhanced soil microbial activity. Enhanced soil microbial activity is indicated by increased CO₂ emissions resulting

from mineralization of biologic tissue and soil organic matter. Wardle, Nilsson, and Zackrisson^[12] suggested that biochar-induced increased mineralization of soil organic matter might have an adverse effect on the levels of soil organic matter. However, any analysis of the net effect of biochar on the levels of soil organic matter must include both the rate at which existing soil organic matter is mineralized and the rate at which new soil organic matter is being formed. Increased nutrient cycling associated with enhanced microbial activity increases nutrient availability to plants, which typically results in increased plant growth and increased input of new residue to soils. Furthermore, by adsorbing dissolved organic compounds (Fig. 1) during the decomposition of organic residues, biochar may increase the efficiency by which new residues are transformed into stabilized soil organic matter. Scientific studies that consider the effects of biochar on both inputs and losses of soil organic matter are lacking. However, the presence of high levels of both biochar and biogenic soil organic matter in the Terra Preta soils suggests that the net effect of biochar is to sustain high levels of biogenic soil organic matter.

Most studies report small to substantial yield increases following soil biochar applications,^[13,14] however, there are also a few reports of crop yield decreases. The cause of these yield decreases is not always clear. However, biochar quality appears to be very important. Biochars that contain substantial amounts of bioavailable C have the potential to immobilize N and induce N deficiencies. Biochar binds micronutrients such as copper very tightly, and it may be possible to induce micronutrient deficiencies in some soils by adding too much biochar. Most biochar is a liming agent, and when applied to calcareous soils or when too much biochar is applied to a neutral or acidic soil, biochar could increase the soil pH to the point that some micronutrients are rendered insoluble and thereby induce micronutrient deficiencies. Furthermore, biochars that are exposed to pyrolysis vapors during or after the production of the biochar may contain significant levels of phytotoxic phenolic compounds, and the release of such phenolic compounds from biochar could inhibit plant growth. On the contrary, most biochars strongly adsorb phenolic compounds, thereby reducing their bioavailability and leaching.^[15] For example, Garnett et al.^[16]

observed that biochar adsorbed phenolic compounds from leaf litter extracts and enhanced seedling growth relative to controls exposed to leaf litter extracts without the biochar. Therefore, anyone considering the use of biochar as a soil amendment should carefully consider the quality of the biochar before applying it to soil; some biochars will inhibit plant growth while others will enhance plant growth.

SUMMARY

Biochar has the capacity to retain bioavailable water and is a very strong adsorbent of plant nutrients and dissolved organic compounds. Biochar is an effective soil conditioner that reduces soil bulk density and increases the capacity of soils to retain and supply nutrients and water to grow plants. Because of these positive effects on soil quality, most studies have reported enhanced plant growth for soils amended with biochar; however, caution is urged as biochar quality is critical. Some biochars have the capacity to tie up nutrients whether by immobilization, by nutrient adsorption, or by raising the soil pH to the point that the nutrients are no longer bioavailable.

REFERENCES

1. Antal, M.J., Jr.; Grønli, M. The art, science, and technology of charcoal production. *Ind. Eng. Chem. Res.* **2003**, *42*, 1619–1640.
2. Glaser, B.; Balashov, E.; Haumaier, L.; Guggenberger, G.; Zech, W. Black carbon in density fractions of anthropogenic soils of the Brazilian Amazon region. *Org. Geochem.* **2000**, *31*, 669–678.
3. Glaser, B.; Haumaier, L.; Guggenberger, G.; Zech, W. The ‘Terra Preta’ phenomenon: A model for sustainable agriculture in the humid tropics. *Naturwissenschaften* **2001**, *88*, 37–41.
4. Lehmann, J.; da Silva, J.P., Jr.; Steiner, C.; Nehls, T.; Zech, W.; Glaser, B. Nutrient availability and leaching in an archaeological Anthrosol and a Ferralsol of the Central Amazon basin: Fertilizer, manure and charcoal amendments. *Plant Soil* **2003**, *249*, 343–357.
5. Baerthaler, G.; Zischka, M.; Haraldsson, C.; Oberberger, I. Determination of major and minor ash-forming elements in solid biofuels. *Biomass Bioenerg.* **2006**, *30*, 983–997.
6. Cheng, C.H.; Lehmann, J.; Thies, J.E.; Burton, S.D.; Engelhard, M.H. Oxidation of black carbon by biotic and abiotic processes. *Org. Geochem.* **2006**, *37*, 1477–1488.
7. Cheng, C.-H.; Lehmann, J.; Engelhard, M.H. Natural oxidation of black carbon in soils: Changes in molecular form and surface charge along a climosequence. *Geochim. Cosmochim. Ac.* **2008**, *72*, 1598–1610.
8. Liang, B.; Lehmann, J.; Solomon, D.; Kinyangi, J.; Grossman, J.; O’Neill, B.; Skjemstad, J.O.; Thies, J.; Luizão, F.J.; Petersen, J.; Neves, E.G. Black carbon increases cation exchange capacity in soils. *Soil Sci. Soc. Am. J.* **2006**, *70*, 1719–1730.
9. Laird, D.A.; Fleming, P.D.; Davis, D.D.; Horton, R.; Wang, B.; Karlen, D.L. Impact of biochar amendments on the quality of a typical Midwestern agricultural soil. *Geoderma* **2010**, *158*, 443–449.
10. Laird, D.A.; Fleming, P.D.; Karlen, D.; Wang, B.; Horton, R. Biochar impact on nutrient leaching from a Midwestern agricultural soil. *Geoderma* **2010**, *158*, 436–442.
11. Warnock, D.D.; Lehmann, J.; Kuyper, T.W.; Rillig, M.C. Mycorrhizal responses to biochar in soil—concepts and mechanisms. *Plant Soil* **2007**, *300*, 9–20.
12. Wardle, D.A.; Nilsson, M.C.; Zackrisson, O. Fire-derived charcoal causes loss of forest humus. *Science* **2008**, *320*, 629.
13. Chan, K.Y.; Van Zwieten, L.; Meszaros, I.; Downie, A.; Joseph, S. Agronomic values of greenwaste biochar as a soil amendment. *Soil Res.* **2007**, *45*, 629–634.
14. Glaser, B.; Lehmann, J.; Zech, W. Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal—a review. *Biol. Fert. Soils* **2002**, *35*, 219–230.
15. Keech, O.; Carcaillet, C.; Nilsson, M.C. Adsorption of allelopathic compounds by wood-derived charcoal: The role of wood porosity. *Plant Soil* **2005**, *272*, 291–300.
16. Garnett, E.; Jonsson, L.M.; Dighton, J.; Murnen, K. Control of pitch pine seed germination and initial growth exerted by leaf litters and polyphenolic compounds. *Biol. Fert. Soils* **2004**, *40*, 421–426.

Biochar: Soil Carbon and Fertility

Adam O'Toole

Daniel Rasse

Climate and Environment Division, Norwegian Institute of Bioeconomy Research (NIBIO),
Aas, Norway

Abstract

Biochar is a multipurpose soil amendment for addressing two key global problems, namely soil degradation and climate change. The unique properties of biochar reside in the nature of its carbon (C) structure, which is more persistent in soils than that of uncharred organic matter. When made and applied correctly, biochar can remain in soil for hundreds of years. This characteristic gives biochar a comparative advantage over alternative methods of soil C sequestration in terms of both storage duration and its compatibility with conventional tillage operations. Parallel to C sequestration effects, biochar application appears to promote soil quality and crop productivity. Studies show that biochar influences the fertility of soils via a variety of mechanisms including soil pH increases in acidic soils, direct plant nutrient supply, and increases in soil water retention. Indirect effects on plant growth have also been found due to the unique surface properties of biochar, which promote a buildup of plant-available nutrients in soils. Biochar technology is largely at the research and pilot stage. Operational implementation will require robust and cost-efficient methods for product quality and C sequestration assessment, better understanding of effects on soil biota, and mechanisms for paying farmers for the ecosystem services of biochar.

INTRODUCTION

Biochar is the term given to define charcoal when it is used for sequestering soil carbon (C) and improving soil fertility. In a soil science context, the study of biochar overlaps with the previous research on pyrogenic C in soil derived from vegetation fire. The study of biochar has gained significant attention post-2000 with the pressing need to find mitigation options for climate change. Biochar is produced via the thermal process of pyrolysis, which heats biomass at temperatures $>350^{\circ}\text{C}$ in an oxygen (O_2)-depleted atmosphere. During pyrolysis, molecular structures within the feedstock undergo chemical transformation where light compounds are released in the form of gas and oils, leaving behind more complex compounds, such as lignin, which are transformed into polyaromatic C structures. These polyaromatic structures are difficult for microbes to consume as an energy source. Therefore, converting biomass to biochar generates a form of organic matter that can remain in soil for a long time. The persistence of biochar C in soil depends upon a variety of factors including climatic and soil conditions, mode of application, and how the biochar was made (production temperature, pyrolysis technology, and feedstock type).

In addition to its mitigation function for climate change, biochar has also shown to improve the fertility of soils. Research into the agronomic benefits of biochar has been partly inspired by the discovery of the fertile

black soils in the Amazon known as *Terra preta*. *Terra preta* are Anthrosols that are typified by a profile rich in soil organic C, pottery remnants, and biochar, the latter making up $>90\%$ of the soil organic C in investigated *Terra preta* soils.^[1] *Terra preta* is remarkable for its enduring fertility and ability to retain high levels of soil organic matter despite moist and warm tropical conditions, which normally lead to rapid soil C loss. Soil scientists have made a concerted effort within the 20th century to determine to what extent the “*Terra preta* effect” of enduring soil fertility and high C content is due to the addition of biochar.

Modern biochar trials have only been started within the 20th century, with many of them not older than a few growing seasons, and therefore, we can only report on short-term agronomic effects. Whether biochar increases soil fertility depends on whether it enhances the chemical, physical, and/or biological conditions that are necessary for optimal plant growth. Here, we summarize the main findings of biochar research and knowledge as regard to its use to mitigate climate change and to improve soil quality and crop yields around the world.

SOIL C SEQUESTRATION, C CYCLING EFFECTS, AND BIOCHAR C STABILITY

It has been known for a long time that vegetation fires can generate long-lived charcoal forms in soils;^[2] however,

the yield of charcoal formation in a vegetation fire is very low, estimated around 5%.^[3] The reason for the low charcoal yield generated by a vegetation fire is that most of the biomass is fully oxidized while the inside of larger stems often does not reach sufficient temperatures due to insufficient heating times. In contrast, modern pyrolysis technologies can control temperature, O₂ exposure, and heating time to produce higher and more stable yields of biochar C.

In theory, C sequestration with biochar technology presents multiple advantages as compared to traditional methods for organic matter management such as composting or plowing of crop residues.^[4] The main advantage is that the resulting biochar is much more persistent in soils than the original feedstock. Although a wide range of estimated residence time for biochar in soils has been reported depending on pyrolysis conditions, many studies show that the constitutive C of well-engineered biochar remains in soil well over the 100-year threshold, which is used as a convention for the permanent removal of carbon dioxide (CO₂). By contrast, most of the C in fresh biomass, such as straw, is lost to the atmosphere in the form of CO₂ within a few years after incorporation into soil. A small fraction of fresh biomass C becomes part of a more protected pool of soil organic C.^[5] However, even soil organic C itself has a mean residence time in soils of about only 50 years,^[6] i.e., well below that of biochar. Fig. 1 illustrates the large potential for increased C storage in soils through pyrolysis of plant residues and incorporation of the resulting biochar into soil. The soil organic C content of a cereal field with plowed-in residues reaches a simulated equilibrium of 30 tons C ha⁻¹ in its Ap horizon, under the assumption that 12% of shoot biomass is transformed into soil organic matter with a turnover rate of 2% per year. Exporting the shoot biomass off-farm rather than

plowing them in would result in a progressive decrease of nearly 50% of the organic C content of the soil. By contrast, if we assume that 50% of the shoot biomass is converted to biochar (with a conservative mineralization rate of 1% per year), we would nearly quadruple the amount of C sequestered into the Ap horizon, reaching about 120 tons C ha⁻¹ after 250 years (Fig. 1). This biochar effect on C sequestration would be even larger if we logically assume that the increased soil organic C content has a moderate positive effect on plant yield and biomass production. This simple simulation exercise illustrates that it is not necessary to assume millennial-scale stability for biochar products in soil in order to increase several fold the amount of organic C being sequestered in agricultural soils.

The C sequestration effect obtained with biochar addition is insensitive to changes in management practices because well-engineered biochars decompose slowly in soil. This is not the case for other C sequestration methods. For example, all C sequestration gains with a plantation forest can be lost very quickly in a fire or storm event. Soil C stocks can be increased with continuous amendments of compost and manure, no-till practices, and conversion of arable lands to pasture. However, once these C sequestration measures are discontinued, the soil organic matter stocks rapidly decrease due to the inherent degradability of non-pyrogenic organic matter in soil. By contrast, the enhanced stability of biochar C in soil gives a farmer the possibility to continue ploughing if they wish while still retaining soil C.

More streamlined accounting for soil C in soil C credit/offset programs is another potential advantage of biochar technology.^[4] In theory, all that is needed are records of applied biochar quantities and an estimate of its mineralization rate in soil. This gives biochar an advantage over other soil-based methods, which require expensive and regular monitoring campaigns in order to verify the changes in soil C content. However, little research has been conducted in documenting the physical transport and fate of biochar in soil. Due to the low density of biochar and its initial hydrophobicity when first made, it is likely that some of the biochar will be eroded or leached from the soil. Downstream environmental effects of this loss have not been well documented, and more research is needed here.

In spite of its many potential advantages, i.e., permanent removal, flexibility of land management, and ease of implementation in C accounting schemes, biochar remains mostly at the research and pilot stage. Encouraging biochar use in modern farming will require research to identify where biochar will lead to added value in terms of agronomy, energy, and/or economy. Carbon financing is likely needed from public and private funds to give farmers an incentive to convert their biomass to biochar and store more C in soil. But before this happens, a scientific consensus is needed on the best

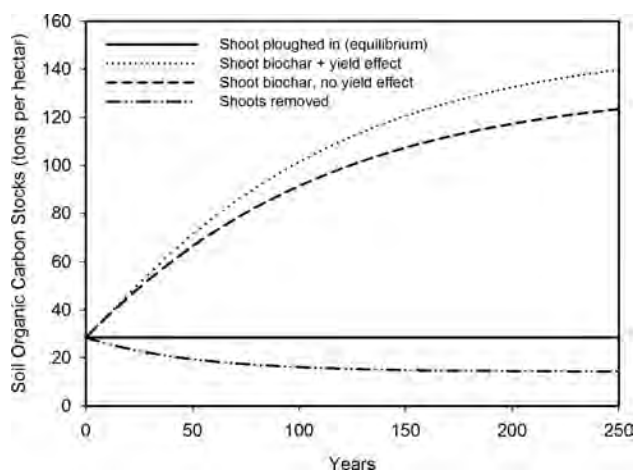


Fig. 1 1. Simulated evolution of soil C stocks comparing straw restitution (12% transformed into soil organic C, which mineralizes at 2% per year) and straw pyrolysis into biochar before restitution (50% biochar yield and 1% mineralization per year).

method/methods to estimate the stability of biochar in soil. These methods need to be robust and cost-efficient. One method proposed by an international panel uses the molar hydrogen/organic C ratio of biochar as a proxy for biochar C stability.^[7]

BIOCHAR'S EFFECT ON REDUCING NON-CO₂ SOIL GREENHOUSE GAS EMISSIONS

In order to evaluate the total greenhouse gas mitigation potential of biochar, it is imperative to determine its effect on the other two major greenhouse gasses—methane (CH₄) and nitrous oxide (N₂O). N₂O has a global warming potential 298 times that of CO₂ on a 100-year time horizon, and, therefore, even limited changes in N₂O emissions induced by biochar application could significantly eat away at the C sequestration benefits. Fortunately, the vast majority of studies show that biochar reduces N₂O emissions, especially in poorly drained soils where denitrification takes place.^[8] In these soils, N₂O is a by-product of microbial processes and occurs mostly when nitrate is not fully denitrified to nitrogen gas.

The story regarding biochar's effect on CH₄ is uncertain, with studies showing both increases and decreases in CH₄ emissions due to biochar addition. Under some experiments, biochar has increased CH₄ uptake to soil by increasing aeration of the soil and making soil pH conditions more favorable for CH₄-consuming methanotrophs.^[9] In another study, biochar has increased CH₄ at higher moisture rates due to its ability to extend the periods of water saturation and anoxia, which encourages CH₄-producing methanogens.

THE ROLE OF BIOCHAR IN IMPROVING SOIL FERTILITY IN AGRICULTURAL SYSTEMS

Plant growth is dependent on a combination of chemical, physical, and biological conditions in the soil. The extent to which biochar modifies these three main soil parameters

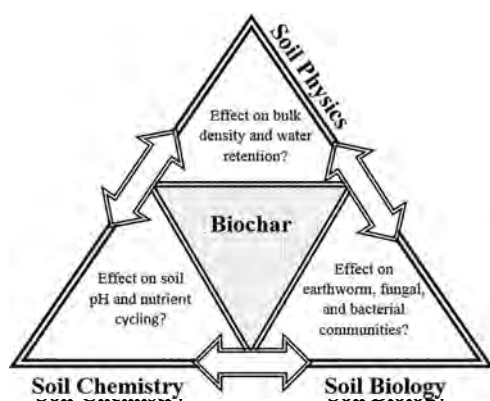


Fig. 2 Scientific efforts are investigating the effects of biochar on soil physics, chemistry, and biology.

depends on the inherent biochar properties, as influenced by the type of feedstock and the pyrolysis conditions, application dose rates, and the degree to which it interacts with the inorganic, organic, and living matter in the soil (Fig. 2).

Here, we will outline some of the main findings from biochar related to its effects on plant yield and nutrition, soil physics, and soil biology.

PLANT YIELD AND NUTRITION

A meta-analysis of biochar research showed that on average biochar additions to soil increase yield; soil pH; soil C, nitrogen (N), and phosphorus (P); and concentrations of plant tissue potassium (K).^[10] Another meta-study in 2011 reported an average increase in crop yield of 10%, which was more pronounced in acidic and coarse-textured soils.^[11] In highly degraded soils, which receive little or no fertilizer, biochar incorporation directly contributes to plant nutrient supply notably through its ash content. The nutrient content of biochar depends on the feedstock it is made from. For example, biochar made from poultry litter is high in P, while straw- and grass-based biochars are often high in K. Nitrogen-rich feedstocks such as poultry litter and biosolids can be used for making biochar, but a significant proportion of the N content is volatilized during pyrolysis, and the remaining N is not easily mineralizable.^[12] This should be taken into consideration when evaluating the total environmental footprint of a biochar pyrolysis system. More important than the nutrient content of the biochar is its ability to capture nutrients when mixed with other nutrient-rich sources. Studies have shown beneficial synergies by incorporating biochars into compost, biogas, and manure management systems. Here, it is possible to take advantage of the large pore space (Fig. 3) of the biochar to store nutrients and water and reduce the loss of N to groundwater and the atmosphere.

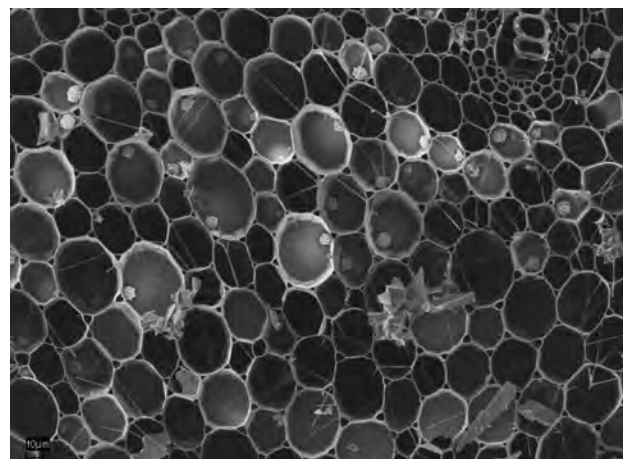


Fig. 3 Cross section of the stem of wheat straw biochar. **Source:** NIBIO.

SOIL PHYSICAL EFFECTS

Biochar can also modify soil physical properties, such as soil water retention and bulk density. Biochar has a low bulk density ranging from 0.2 to 1.0 g cm³, depending on the feedstock it was made from.^[13] Thus, additions of biochar to mineral soil (which usually has a bulk density of 1.3 g/cm³) can reduce the overall density of the soil. Biochar is a highly porous material somewhat like a sponge and can adsorb many times its own weight in water. This property can be of benefit for increasing soil water retention in sandy soils, especially during drought conditions.^[14,15] In heavy clay soils, biochar additions have shown to increase the stability of macroaggregates and aid in reducing the mechanical strength and penetration resistance of hardsetting soils, which is beneficial for root growth.^[16] In horticulture, biochar has been successfully used in high concentrations (50% to 75%) as a replacement for peat, which is considered a non-renewable resource.^[17] Biochar is similar to peat in that it can hold a lot of water, but has an advantage over peat in that it does not shrink when it dries out.

SOIL BIOTA

While biochar, due to its higher aromatic C content, is unlikely to be a significant source of energy for microbes, there have been a number of reports that biochar can indirectly affect microbial communities. This could be for a number of reasons, including the following:

1. Biochar is usually alkaline and raises soil pH, which can have significant effects on microbial processes;
2. The porous structure of the biochar provides a refuge for microbes, which may alter predation levels and thus trophic turnover; and
3. The presence of volatile C compounds on the surface of biochar, which may have either inhibiting or stimulating effect on microbial communities.

Generalizing biochar effects on soil biota is difficult due to the heterogeneity of biochars and their chemical and physical properties. Short-term effects might not last in the long term. For example, biochar application can induced short-term increases in gross N mineralization, nitrification, and immobilization rates; however, this effect appears to fade with time.^[18] Faunal activity is also affected by biochar, both through direct pH and chemical effects and through indirect changes in bacteria populations, which are a food source for the soil fauna. A meta-analysis on the effect of biochar on earthworm populations showed a short-term negative effect but no long-term impact.^[19] Further studies investigating soils with historically large biochar contents may shed more light on the long-term effects of biochar on microbial

populations and their ecosystem functioning. For example, one study compared *Terra preta* Anthrosols rich in biochar with adjacent soils sharing the same mineralogy and texture and found large differences in bacterial community composition.^[20]

CONCLUSION

Biochar is the subject of intense research within the soil science community as a multipurpose material for both storing more C and improving the fertility of soils. Biochar has been tested in most of the world's major agricultural countries and climatic zones, and much has been learned about how it affects soil chemistry, physics, and biology. The applicability of biochar as a robust method for storing more C in soils seems well proven, but large-scale implementation will require more research and development on intrinsic biochar properties and resulting effects on agroecosystems.

ACKNOWLEDGMENT

The authors would like to acknowledge the financial support of the Norwegian Ministries of Agriculture and Environment that makes funding available for them to disseminate scientific knowledge to a wider audience on agricultural and climate change-related topics.

REFERENCES

1. Mao, J.D.; Johnson, R.L.; Lehmann, J.; Olk, D.C.; Neves, E.G.; Thompson, M.L.; Schmidt-Rohr, K. Abundant and stable char residues in soils: Implications for soil fertility and carbon sequestration. *Environ. Sci. Technol.* **2012**, *46*, 9571–9576.
2. Preston, C.M.; Schmidt, M.W.I. Black (pyrogenic) C: A synthesis of current knowledge and uncertainties with special consideration of boreal regions. *Biogeosciences* **2006**, *3* (4), 397–420.
3. Alexis, M.A.; Rasse, D.P.; Rumpel, C.; Bardoux, G.; Péchot, N.; Schmalzer, P.; Drake, B.; Mariotti, A. Fire impact on C and N losses and charcoal production in a scrub oak ecosystem. *Biogeochemistry* **2007**, *82*, 201–216.
4. Lehmann, J. A handful of carbon. *Nature* **2007**, *447*, 143–144.
5. Rasse, D.P.; Rumpel, C.; Dignac, M.-F. Is soil C mostly root C? Mechanisms for a specific stabilization. *Plant Soil* **2005**, *269*, 341–356.
6. Schmidt, M.W.I.; Torn, M.S.; Abiven, S.; Dittmar, T.; Guggenberger, G.; Janssens, I.A.; Kleber, M.; Kögel-Knabner, I.; Lehmann, J.; Manning, D.A.C.; Nannipieri, P.; Rasse, D.P.; Weiner, S.; Trumbore, S.E. Persistence of soil organic matter as an ecosystem property. *Nature* **2011**, *478*, 49–56.
7. Budai, A.; Zimmerman, A.R.; Cowie, A.L.; Webber, J.B.W.; Singh, B.P.; Glaser, B.; Masiello, C.A.; Andersson, D.; Shields, F.; Lehmann, J.; Camps Arbestain, M.; Williams,

- M; Sohi, S.; Joseph, S. *Biochar C Stability Test Method: An Assessment of Methods to Determine Biochar C Stability. International Biochar Initiative, 2013*. Available at http://www.biochar-international.org/sites/default/files/IBI_Report_Biochar_Stability_Test_Method_Final.pdf (accessed January 2014).
8. Cayuela, M.L.; van Zwieten, L.; Singh, B.P.; Jeffrey, S.; Roig, A.; Sánchez-Monedero, M.A. Biochar's role in mitigating soil nitrous oxide emissions: A review and meta-analysis. *Agr. Ecosyst. Environ.* **2014**, *191*, 5–16.
 9. Feng, Y.; Xu, Y.; Yu, Y.; Xie, Z.; Lin, X. Mechanisms of biochar decreasing methane emission from Chinese paddy soils. *Soil Biol. Biochem.* **2012**, *46*, 80–88.
 10. Biederman, L.; Harpole, S.W. Biochar and its effects on plant productivity and nutrient cycling: A meta-analysis. *Glob. Change Biol. Bioenerg.* **2013**, *5*, 202–214.
 11. Jeffery, S.; Verheijen, F.G.A.; van der Velde, M.; Bastos, A.C. A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis. *Agr. Ecosyst. Environ.* **2011**, *144* (1), 175–187.
 12. Hossain, M.K.; Strezov, V.; Chan, K.Y.; Ziolkowski, A.; Nelson, P.F. Influence of pyrolysis temperature on production and nutrient properties of wastewater sludge biochar. *J. Environ. Manage.* **2011**, *92* (1), 223–228.
 13. Downie, A.; Crosky, A.; Munroe, P. Physical properties of biochar. In *Biochar for Environmental Management: Science and Technology*; Lehmann, J., Joseph, S., Eds.; Earthscan: London, 2009; 14–32.
 14. O'Toole, A.; Knoth de Zarruk, K.; Steffens, M.; Rasse, D.P. Characterization, stability, and plant effects of kiln-produced wheat straw biochar. *J. Environ. Qual.* **2013**, *42* (2), 429–436.
 15. Mulcahy, D.N.; Mulcahy, D.L.; Dietz, D. Biochar soil amendment increases tomato seedling resistance to drought in sandy soils. *J. Arid Environ.* **2013**, *88*, 222–225.
 16. Chan, K.Y.; Van Zwieten, L.; Meszaros, I.; Downie, A.; Joseph, S. Agronomic values of greenwaste biochar as a soil amendment. *Aust. J. Soil Res.* **2007**, *45* (8), 629–634.
 17. Steiner, C.; Harttung, T. Biochar as growing media additive and peat substitute. *Solid Earth Discuss.* **2014**, *6* (1), 1023–1035.
 18. Nelissen, V.; Rütting, T.; Huygens, D.; Ruysschaert, G.; Boeckx, P. Temporal evolution of biochar's impact on soil nitrogen processes – A ¹⁵N tracing study. *Glob. Change Biol. Bioenerg.* **2014**, *4*, 635–645.
 19. Weyers, S.L.; Spokas, K.A. Impact of biochar on earthworm populations: A review. *Appl. Environ. Soil Sci.* **2011**, *2011*, 1–12.
 20. Grossman, J.M.; O'Neill, B.E.; Tsai, S.M.; Liang, B.; Neves, E.; Lehmann, J.; Thies, J.E. Amazonian anthrosols support similar microbial communities that differ distinctly from those extant in adjacent, unmodified soils of the same mineralogy. *Microb. Ecol.* **2010**, *60* (1), 192–205.

Biodegradation and Bioremediation

Owen P. Ward

Ajay Singh

Department of Biology, University of Waterloo, Waterloo, Ontario, Canada

Abstract

Biodegradation processes mediate recycling of organic materials in the environment. In bioremediation processes, microorganisms and plants participate in the biodegradation and removal of hazardous contaminants to restore the polluted environment. Although microorganisms thrive in aqueous environments, the majority of organic chemicals and environmental contaminants are hydrophobic and sorb to soil particles. To access these contaminants, microbes may interact directly with the contaminant or soil particulates or both or secrete biosurfactants to mobilize the contaminant into the aqueous phase. A variety of *in situ* and *ex situ* bioremediation strategies are employed, aimed at promoting the biodegradation of chemical pollutants present in the contaminated medium.

INTRODUCTION

In their most general sense, biodegradation processes are central to all biological systems. In these processes, cells transform and recycle biological and synthetic chemicals and materials into smaller molecules to generate energy and/or metabolic intermediates needed for cell functioning including maintenance, biosynthesis, and growth. Indeed, these processes also contribute to the unique adaptation properties of organisms, e.g., enabling microbes to adjust to changing environments by making the requisite biochemical and physiological changes mediated by biodegradative and biosynthetic metabolic processes. Bioremediation represents an important applied segment among these biodegradation processes in which technologies are generally engineered and optimized to exploit the same biodegradation capabilities especially of microbes, for the removal of organic contaminants and, sometimes, metal contaminants from the environment. The principal objective of this entry is to provide an overview of bioremediation processes and systems as well as a perspective on developments and future prospects. General principles of biodegradation processes are first considered to provide a context for addressing bioremediation technologies.

BIODEGRADATION OVERVIEW

On a macro scale, biodegradation processes often mediate recycling of organic materials, including animal, plant, and microbial materials, present in the environment. Enzymes secreted by microbes and other species attack and depolymerize the principal biological polymeric structures, namely complexes containing carbohydrates, proteins, nucleic acids, lipids, and their derivatives into oligomeric

or monomeric components, which may be taken up and metabolized by cells. These processes are promoted in the digestive systems of humans, animals, and other species mediated by endogenous enzymes from these organisms or supported by the microbial populations present or both. These processes also occur everywhere in the environment where microbes can thrive, most notably in soil and in aqueous media. Processes in soil are enhanced around plant roots where nutrients released from the roots support the growth of plant through growth-promoting bacteria. Such processes are also actively promoted in the practices of agriculture and horticulture and in myriad solid and liquid waste treatment systems. Some examples of general environmental processes and practices for the biodegradation of natural organic materials include processes for silage making, composting, and wastewater treatment (Table 1). The reader is referred to references^[1–3] for further discussion of these topics.

BIOREMEDIATION OVERVIEW

Industrial-scale processes for the recovery of petroleum and the manufacturing, distribution, and use of chemicals invariably lead to the transfer of chemical contaminants into the environment, most notably into soils, groundwater, shorelines, and water bodies. These contaminants range from alkanes and mono- and polycyclic aromatic hydrocarbon (PAH) compounds, naturally present in crude oil or generated through combustion, to high-volume synthetic chemicals including chlorine-, nitrogen-, and phosphate-containing organic chemicals and many others. In addition, soils become contaminated with inorganic chemicals. Just as microbes have the ability to concentrate and accumulate metals in wastewater treatment processes, so too specific

Table 1 Example processes for the biodegradation of natural organic materials.

- *Silage-making processes*, where fibrous plant materials are rendered more digestible for animal feeding by the conversion of sugars to organic acids and some enzymatic cellulose/hemicellulose degradation, mediated by microbial anaerobic digestion.
- *Composting processes*, used to aerobically transform plant and other biological and organic materials, most frequently derived from agricultural, horticultural, food, and municipal wastes. A variety of microbial species, most notably fungi and bacteria including mesophiles and thermophiles, participate in these exothermic biodegradative processes to generate beneficial organic fertilizers devoid of pathogens, which are killed as thermophilic conditions are achieved.
- *Biological wastewater treatment processes*, where the objective is to remove contaminants from domestic, municipal, or industrial waste streams. Microorganisms are primarily responsible for aerobic and anaerobic degradation and assimilation of the organic components together with some removal/concentration of heavy metals. The resulting accumulated biosolids may be further processed and transformed into beneficial organic fertilizers for use in agriculture.

microbes can accumulate or transform some of these inorganic species or both.

The extent of the threat to the environment posed by petroleum and organic chemicals may be appreciated by reference to the production volumes of these products. Global crude oil production and world refining capacity are approaching 80 million bbl per day. Estimates for global production volumes of synthetic polymers and plastics approximate to 140 million tons per annum.^[4] In the United States alone, the annual production volumes of each of ethylene, propylene, vinyl chloride (VC), and benzene range from 5 to 20 million tons, with volumes of many other organic chemicals ranging from 1 to 5 million tons.^[5]

Some of the most widely distributed organic environmental contaminants are aliphatic and aromatic hydrocarbons, including alkanes, mono- and polycyclic aromatics, present in crude oils or in refined oil fuels or petrochemicals; halogenated aliphatic, especially chlorinated species including chlorinated methanes, ethanes, and ethenes such as trichloroethylene, dichloroethene, perchloroethene, and VC; mono- and polyhalogenated phenols and benzoates, including pentachlorophenol; polychlorinated biphenyls (PCBs); chloroaromatics including 2,4-dichlorophenoxyacetate, dichlorodiphenyltrichloroethane, and related compounds; nitroaromatics including nitrobenzene and 2,4,6-trinitrotoluene (TNT); organophosphates; nitrate esters (including glycerol trinitrate); nitrogen heterocycles (including the explosive compound hexahydro-1,3,5-trinitrotriazine or RDX); and polyesters, polylactates, polyurethanes, and nylon.^[6,7]

The recognition that microbes in soil could degrade, transform, or accumulate many of the chemicals polluting the soil provided momentum for the undertaking of research to exploit these activities to clean up contaminated soil and other media, using processes termed as bioremediation. Indeed, the observed effectiveness of bioremediation as a process for the degradation of oils contaminating the shorelines at Prince William Sound, Alaska, due to the *Exxon Valdez* oil spill led to the development of a variety of strategies for the application of bioremediation processes for soil remediation.^[8,9] Bioremediation is generally considered to be the low-cost option in comparison to physical and chemical alternatives.^[10]

Bioremediation technologies may be adapted to a variety of different contaminants and environmental conditions, and in particular, conditions can be developed for the selection and enrichment of microbial populations having the requisite and unique degradation abilities to remediate the specific contaminants present at a particular site.^[11] Bioremediation technologies are also advantageous where contaminants have become diluted or widely dispersed over time.

A vast array of physical environments exist where biodegradation and bioremediation processes may thrive. Broadly speaking, these environments will all contain water or at least be moist because the presence of water is a prerequisite for microbial growth and biodegradation activity. The nature of the microbial type or population that develops will be influenced by a wide range of factors including the physical properties of the medium; salt concentration; solid content, surface properties, and particle size; nature and concentration(s) of contaminant(s); nature and concentration of nutrients present to support microbial growth; relative humidity; presence of oxygen or other electron acceptors; pH; and temperature.

CONTAMINANT BIODEGRADABILITY

Although many organic chemicals are biodegradable by microbes present in the environment, other natural or synthetic chemical structures exhibit resistance or recalcitrance to biodegradation and tend to persist in the environment for prolonged periods of time. The microbial transformation of organic pollutants is mediated primarily by enzyme-catalyzed reactions but often with the support of other cell components, associated with cell surfaces, surfactants, chelators, vesicles, and organelles. The biotransforming enzymes may be located within the cell protoplasm or be membrane associated (or a combination of these) or indeed in a few cases may be extracellular. A good example of a combined intracellular/membrane-bound system is the *alk* gene system for alkane degradation, whereas extracellular fungal peroxidases and laccase are non-specific mechanisms for the attack of organic pollutants, mediated by the generation of free radicals.^[11] The microbial enzyme

transformation of contaminants may lead to complete contaminant mineralization or to the production of breakdown products that may be assimilated into the cells' central pathways for energy or biomass production or both. Alternatively, contaminants may be partially degraded or transformed into metabolites, which the cell cannot further utilize, whereby such metabolites may accumulate in the cell or be excreted. These metabolites may be further transformed by other cell types, among the multitude of microbes present in the ecosystem, which may have different metabolic capacities. Alternatively, the metabolites may be generally recalcitrant to biodegradation and may bioaccumulate. Contaminants in well-aerated environments will predominantly be degraded by aerobic metabolic processes, whereas contaminants in a non-aerated subsurface environment may be anaerobically degraded.

A review of aerobic pathways for the biodegradation of organic contaminants including petroleum hydrocarbons, chlorinated hydrocarbons, nitroaromatics and nitrogen heterocycles, organophosphate derivatives, and plastics reveals the following reaction types: dioxygenases, monooxygenases, hydroxylases, ligninases, and reductases among others. Anaerobic biodegradations typically involve initial activation mechanisms such as carboxylation, methylation, hydroxylation, and dehydrogenation followed by additional reactions, anaerobic reductions, and hydrolytic reactions.^[12]

Microbes have the potential to remediate metals in contaminated media by exploiting a combination of mechanisms including precipitation, sorption, biosurfactant complexation, biofilm entrapment, and metabolic uptake. Some microbes can also transform metals including radionuclides, e.g., by reducing mercury into a volatile form, which can be removed. In metal bioremediation applications, there is particular interest in exploiting microbes, which have been isolated from mine tailings and heavy metal-contaminated sites, as these strains exhibit superior metal-scavenging capacities, greater resistance to metal toxicity (sometimes mediated by metal efflux mechanisms), and ionizing radiation.^[13] Examples of interesting strains for these applications include certain *Pseudomonas* species, including recombinant strains, *Ralstonia eutropha*, *Thiobacillus ferrooxidans*, and *Deinococcus radiodurans*.

CONTAMINANT TOXICITY

The principal concern regarding chemical contamination of the environment relates to contaminant toxicity. Many of the chemical contaminants, but especially the more recalcitrant chemicals, are toxic to higher life forms or are known as carcinogens or both. As such, their presence represents a threat to human and animal health. Many of these chemicals are classified as hazardous by national regulatory agencies, including the U.S. Environmental

Protection Agency (USEPA). For example, 16 PAHs are considered to be priority pollutants by the USEPA and 7 of these are classified as probable human carcinogens.^[14]

As a result of chemical contamination of the environment, toxicity concerns do not just relate to the toxicities of the original chemical compounds transferred. There are many examples where the "parent" contaminant may get chemically or biologically transformed into reaction products or catabolites, which are as toxic as or even much more toxic than the parent compound. For example, surfactants present in detergents, such as nonylphenol ethoxylates, are often biotransformed into nonylphenol end products with endocrine-disrupting activity as a result of microbial transformation.^[15] The presence of strong chemical oxidizing or reducing agents in the contaminated medium can give rise to non-specific conversions of the contaminants into more reactive and more toxic products.^[16]

In heavily contaminated media, the toxicity of the medium may hinder the implementation of bioremediation strategies as a result of the contaminant, at a higher concentration, being toxic to the microbial population that may have the metabolic capability to biotransform the contaminant. In a manner akin to antibiotic resistance, some microbial cells have developed resistance to specific toxic organic compounds such as toluene, by using efflux pumps to extrude toxic contaminants from the cell cytoplasm or periplasmic space.^[17] It is also thought that, where toxic contaminants are only partially transformed by one species, efflux pumps may have the potential to excrete the metabolites such that they may be made available to other members of a consortium with the capacity for the degradation of excreted metabolites. In metal bioremediation applications, there is particular interest in exploiting microbes, which have been isolated from mine tailings and heavy metal-contaminated sites, as these strains exhibit greater resistance to metal toxicity (sometimes mediated by metal efflux mechanisms) and ionizing radiation.^[13]

The extent of toxicity of a known toxic contaminant cannot be simply related to its concentration in the contaminated medium. Actual toxicity may be diminished by the sorption of contaminant to soil particles or organic matter. In addition, biological species present in or added to the contaminated medium may adsorb the contaminant or secrete biological molecules including chelators or surfactants, which effectively sequester the contaminant, thereby reducing its toxicity. Arguably, the presence of other components in the contaminated medium or indeed produced by the biological organisms present could enhance toxicity. For example, a hydrophobic contaminant, with a tendency to sorb to soil, might become mobilized and solubilized by the presence of lipids or surfactants and possibly become more toxic to the organisms present.

Scientists have developed toxicity-testing methods, whereby contaminant-sensitive organisms, including

microbes, algae, plants, fish, macroinvertebrates, and cultured cell lines, are used as biomonitors of pollutant accumulation in soil and aquatic systems and also for following the reduction in the toxicity of the contaminated media as a result of the implemented remediation processes. These biomonitoring tests rely on observing whole-organism biological impacts, related, e.g., to growth, behavior, reproduction, or mortality. Alternatively, toxicity tests may involve the measurement of intracellular biomarkers, e.g., heat shock proteins; enzymes including antioxidant enzymes, DNA repair enzymes, acetylcholinesterase, cytochrome P450 monooxygenase, and glutathione S-transferase; and DNA mutations or chromosomal defects.^[18]

CONTAMINANT BIOAVAILABILITY

Microbes generally thrive and grow in aqueous phases and are most efficient at accessing nutrients that are water soluble. However, petroleum components, as well as most other chemical contaminants, are water insoluble or hydrophobic and hence less bioavailable to microbes. In addition, soil is the most dominant medium for these contaminants, and hydrophobic contaminants tend to bind strongly to soil particles, thereby further reducing their bioavailability to microbes capable of biodegrading the contaminants and rendering them more persistent in the environment. Soil bioremediation processes require adequate amounts of water present in the soil to support microbial activity, and the contaminant-biodegrading organisms have the means to access the hydrophobic or soil particle-bound contaminants or both. Two general natural microbial mechanisms are known to facilitate this contaminant access. First, cells may sorb and uptake the contaminant by direct interaction between the microbes and the hydrophobic substance/particulate matter. Alternatively, the microbes may secrete biosurfactants into the surrounding environment, which may pseudosolubilize and mobilize the contaminant through micelle formation, which facilitates contaminant contact with the microbes and promotes cellular uptake of the hydrophobic substrate. As amphipathic molecules, biosurfactants contain both hydrophilic and hydrophobic ends. In the presence of hydrophobic contaminants in an aqueous environment, micelle formation results when the biosurfactant molecules associate with and solubilize the hydrophobic contaminant in the center of the micelle, while the hydrophilic ends of the biosurfactant molecules remain on the outside of the micelle in contact with the aqueous phase.^[19,20] These contrasting processes were well illustrated by culturing oil-degrading strains of *Pseudomonas* sp. and *Rhodococcus* sp. separately and together in aqueous media supplemented with crude oil. The *Pseudomonas* was located predominantly in the aqueous phase and did not associate with oil droplets, whereas the *Rhodococcus* cells were concentrated at the

oil–water interface.^[21] The greatest degradation occurred where both mechanisms were in play, when a coculture of the strains was used.

In bioremediation processes, the natural mechanisms of contaminant mobilization and pseudosolubilization, promoted by biosurfactants, may be replaced or supplemented by the use of added chemical surfactants. The chemical, namely nonylphenol ethoxylate surfactant, substantially increased crude oil biodegradation by a consortium of oil-degrading organisms.^[22] However, biosurfactants or chemical surfactants may also inhibit the degradation of contaminants, whereby isolating the contaminant in the center of the micelles effectively may prevent uptake by the degrading microbes. For example, Billingsley et al.^[23] observed that at chemical surfactant concentrations above the critical micelle value, the presence of an anionic surfactant promoted, whereas non-ionic surfactants inhibited, biotransformation as compared to no surfactant controls. These and related topics are discussed in more detail by Ward.^[24]

GENERAL MICROBIAL APPROACHES TO THE REMEDIATION OF CONTAMINANTS

Two general strategies for the implementation of bioremediation processes were identified around the time of the *Exxon Valdez* oil spill and cleanup, biostimulation, and bioaugmentation. The objective of biostimulation processes is to promote the growth of the segment of microbes present in the contaminated environment, which can degrade the contaminants. If the contaminated environmental medium does not contain microbes, which can degrade the contaminant, bioaugmentation with a selected inoculum already acclimated to grow on the contaminant may be needed.

Biostimulation

Most organic contaminants in the environment are carbon rich, and the growth of organisms that can use this carbon source is usually limited by the availability of other nutrients, supplements, and electron acceptors. Nutrients that are most widely used in aerobic bioremediation processes are nitrogen and phosphate, and getting oxygen to the medium is often the most challenging engineering component. In surface soil bioremediation processes, oxygen is supplied from the atmosphere, and atmospheric oxygen is also exploited in reactor-based processes, engineered soil piles, and *in situ* processes by sparging air to the contaminated soil or slurry.^[25] Other strategies involve the supply of pure oxygen gas, ozone, or manganese peroxide. In anaerobic bioremediation, the delivery of adequate amounts of suitable electron acceptors to the medium is a priority strategy. Examples of electron acceptors are sulfate, nitrate, and ferric iron.

There are also other more specialized biostimulation approaches. Hydrophobic contaminants are less accessible to microbes, surfactants, biosurfactants, or indeed, easily biodegraded solvents and vegetable oils, agents may be added to solubilize these contaminants, releasing them from particulate matter and rendering them more amenable to biodegradation.^[19,20,26] Where contaminants are typically degraded by cometabolic processes requiring the presence of primary substrates, the remediation of contaminant may be stimulated by the addition of a primary substrate. A further example of biostimulation, in case of the aerobic bioremediation of clays having low porosities that resist aeration/oxygen delivery systems, is to add low-density bulking agents such as plant materials, sawdust, and wood chips to increase the porosity of the medium and allow easy delivery of air.

Bioaugmentation

Examples where a bioaugmentation strategy has been beneficial include the bioremediation of very recalcitrant contaminants such as TNT, phthalate esters, and carbon tetrachloride.^[27,28] This strategy is also necessary where the contaminants are not degraded by natural organisms but where recombinant organisms have been engineered, which are capable of degrading the contaminant. Bioaugmentation is also applicable where highly accelerated biodegradation processes are required, e.g., for accelerated reactor-based processes for the remediation of hydrocarbon sludges.^[29] In many cases, bioaugmentation has been shown not to enhance biodegradation rates beyond what may be achieved through biostimulation, as was most notably illustrated in case of the studies during the *Exxon Valdez* oil spill cleanup.

BIOREMEDIATION TECHNOLOGIES

There are a wide range of possible approaches to the bioremediation of contaminated environments. They include soil bioremediation processes involving natural attenuation, *in situ* subsurface treatments and use of biopiles and composting approaches, and landfarming and bioreactor treatments. Crude oil spills at sea may be addressed through shorelines or deep-sea bioremediation strategies.

Monitored Natural Attenuation or Intrinsic Remediation

This process is the simplest and least-invasive bioremediation approach.^[30] The process presumes that the environment contains sufficient nutrients to support the indigenous microbial population in the biodegradation of contaminants. Oxygen availability is often a rate-limiting factor although anaerobic biodegradation may also be important.

Intrinsic remediation processes may proceed for years, if not decades, as was shown with the remediation of PCBs in the Hudson riverbed.

In situ Subsurface Bioremediation

This approach involves designing a subsurface configuration to allow growth and contaminant biodegradation by the microbes present through the efficient supply of nutrients, oxygen, or other electron acceptors through injector wells and air-sparging systems into the unsaturated or saturated zone, generally below the contaminated plume.^[31] Processes may include a soil vapor extraction or other vapor containment system to prevent the release of volatile contaminants from the site to the atmosphere.

Engineered Biopiles

Engineered soil biopiles are typically fitted with a series of aeration pipes to distribute air to the contaminated soil via positive or negative pressure.^[31] With soils contaminated with volatile hydrocarbons, the air pumps generally draw a vacuum such that air enters at the surface of the pile drawing any undegraded volatiles into the pipework system where they may be directed to a biofilter or other volatile organic carbon (VOC) treatment or collection system. The addition of bulking agents to the soil may be required to facilitate airflow through the soil, and care must be taken to control soil humidity such that growth and biodegradation processes are facilitated.

Soil Composting

In the application of composting strategies to soil bioremediation, biomaterials including wood chips or other plant materials are mixed with the contaminated soil in approximately equal amounts. As in general composting, aerobic biodegradation processes result in heat generation. Contaminants are degraded by the bacteria and fungi associated with the compost, and it is known that contaminants may also react irreversibly with plant and humic substance components.^[31]

Landfarming

This bioremediation approach involves surface atmospheric aeration processes enhanced by frequent tilling and maintenance of moisture levels at 40–60% saturation via sprinklers or other means. The process is applicable to *in situ* soils contaminated near the surface or to spreading and treating excavated soils at a designated landfarming location.^[31] Weathering, sorption, evaporation or volatilization, leaching, and photo-oxidation processes may cause the removal of certain hydrocarbon compounds during bioremediation, resulting in the overestimation of the extent of biodegradation.^[7]

Soil Bioreactors

Soil slurry bioreactors have advantages for bioremediation processes similar to the general fermentation and other bioprocesses, where those conditions in the mixture can approach homogeneity, and processes can be optimally controlled to promote microbial growth and contaminant biodegradation.^[26] Although capital and energy costs are high in these processes, high rates and extents of degradation may be achieved with high levels of process dependability and reliability.^[29]

Bioremediation of Shorelines

Marine shoreline bioremediation processes are challenging due to the potential for changing tides to wash away nutrients and microbes from the contaminated shore. The capacities of microbes to remediate contaminated soil, coastlines, and water bodies were first demonstrated on a large scale in the cleanup of the crude oil, which spilled from the *Exxon Valdez* tanker that ran aground off the coast of Alaska, near Prince William Sound. The remediation of hydrocarbons was promoted by the addition of specific oleophilic nitrogen fertilizers, which bound to the oil contaminants to support microbial growth and biodegradation.^[8,9] In contrast, the addition of microbial inoculants was ineffective.

Deep-Sea Bioremediation

The capacity of microbes to degrade crude oil at sea was illustrated during one of the world's largest environmental disasters, when the explosion at British Petroleum's Deepwater Horizon oil rig resulted in an estimated 4.9 million bbl of crude oil spilling into the Gulf of Mexico. On August 4, 2010, just 3 weeks after the well was capped, the U.S. government estimated that 50% of the oil was already gone from the Gulf, and the rest was being rapidly degraded. Dr. Kenneth Lee puts all of this in a scientific perspective as follows: "The consensus now is that microbial degradation was a major process responsible for cleaning up the Deepwater Horizon spill" (<http://www.dfo-mpo.gc.ca/science/publications/article/2012/01-25-12-eng.html>).

PHYTOREMEDIATION

In phytoremediation processes, the removal of inorganic or organic contaminants from media such as soil is promoted by plants or associations between plant roots and microbes in soil and aqueous media.^[32,33] In these processes, specific contaminants may be extracted, degraded, or immobilized. The mechanisms involved include the following: phytodegradation, where root exudates from the plant enhance microbial growth and contaminant biodegradation in the root surroundings; rhizofiltration,

where soluble contaminants are sorbed onto or into the roots; and phytoextraction involving uptake of metals by roots and their translocation in the plants. The plants may be harvested for disposal, and phytovolatilization, where contaminants taken up by the plant, are possibly, but not always, modified and transpired into the atmosphere.

AIR BIOFILTRATION

Contaminants in an air stream—e.g., VOCs—may be remediated through air biofiltration, whereby the contaminated air passes through a porous medium, which supports the growth of microbes as attached biofilms.^[34] The contaminants and oxygen are transferred from the air stream to the biofilm where biodegradation occurs, thereby decontaminating the air stream. Biofilter media include wood branches, bark or chips, peat, compost, synthetic media, coated ceramic or other particulates, and various combinations of these constituents. For effective performance, the biofilter environment must support efficient bacterial growth and facilitate effective transfer and sorption of contaminants from the air stream into the biofilm and access to the biodegrading microbes. The air stream or filtration medium must be appropriately humidified to insure that adequate water activity is maintained to support microbial growth and biodegradation while not oversaturating the medium such that air flow rate is retarded. The flow rate of the mobile phase (air in this case) is controlled to achieve a retention time in the biofilter sufficient to remove and biodegrade the contaminants in the air stream to a desired specification.

CONCLUSION

The purpose of surface or groundwater remediation is not only to enhance the biodegradation, transformation, or detoxification of pollutants but also to protect the quality of soil to maintain environmental quality and sustain biological productivity within the ecosystem boundaries. Although biological processes are typically implemented at a relatively low cost, the implementation of bioremediation technologies requires knowledge of interdisciplinary sciences involving microbiology, chemistry, engineering, ecology, and hydrogeology. Bioremediation methods have been successfully used to treat polluted soils, oily sludges, and groundwater contaminated by industrial chemicals and solvents, petroleum hydrocarbons, pesticides, and heavy metals.

REFERENCES

1. Bitton, G. *Wastewater Microbiology*, 4th Ed.; Wiley-Blackwell: Hoboken, NJ, 2011.
2. Kutzner, H.J. Microbiology of composting. In *Biotechnology*, 2nd Ed.; Rehm, H.-J., Reed, G., Eds.; Wiley-VCH: Weinheim, 2001; Vol. 3C, 35–94.

3. Storm, I.M.; Kristensen, N.B.; Raun, B.M.; Smedsgaard, J.; Thrane, U. Dynamics in the microbiology of maize silage during whole-season storage. *J. Appl. Microbiol.* **2010**, *109*, 1017–1026.
4. Shimao, M. Biodegradation of plastics. *Curr. Opin. Biotechnol.* **2001**, *12*, 242–247.
5. Singh, A.; Ward, O.P. Soil bioremediation and phytoremediation – An overview. In *Applied Bioremediation and Phytoremediation, Soil Biology Series*; Singh, A., Ward, O.P., Eds.; Springer-Verlag: Berlin, 2004; Vol. 1, 1–12.
6. Megharaj, M.; Ramakrishnan, B.; Venkateswarlu, K.; Sethunathan, N.; Naidu, R. Bioremediation approaches for organic pollutants: A critical perspective. *Environ. Intern.* **2011**, *37*, 1362–1375.
7. Singh, A.; Ward, O.P., Eds. *Biodegradation and Bioremediation, Soil Biology Series*; Springer-Verlag: Berlin, 2004; Vol. 2.
8. Prince, R.C. Bioremediation of marine oil spills. *Trends Biotechnol.* **1997**, *15*, 158–160.
9. Head, I.M.; Martin Jones, D.; Röling, W.F.M. Marine microorganisms make a meal of oil. *Nat. Rev. Microbiol.* **2006**, *4*, 173–182.
10. U.S. EPA. *Cleaning Up the Nations Waste Sites: Markets and Technology Trends*. EPA 542-R-04-015, Technology Innovation and Field Services Division, Office of Solid Waste and Emergency Response, 2004.
11. Van Hamme, J.; Singh, A.; Ward, O.P. Recent advances in petroleum microbiology. *Microbiol. Mol. Biol. Rev.* **2003**, *67*, 503–549.
12. Van Hamme, J.D. Bioavailability and biodegradation of organic pollutants – A microbial perspective. In *Biodegradation and Bioremediation, Soil Biology Series*; Singh, A., Ward, O.P., Eds.; Springer-Verlag: Berlin, 2004; Vol. 2, 37–56.
13. Marques, A.P.G.C.; Rangel, A.O.S.S.; Castro, P.M.L. Remediation of heavy metal contaminated soils: An overview of site remediation techniques. *Crit. Rev. Environ. Sci. Technol.* **2011**, *41*, 879–914.
14. U.S. Environmental Protection Agency. *Provisional Guidance for Quantitative Risk Assessment of Polycyclic Aromatic Hydrocarbons*, EPA/600/R-93/089; USEPA: Washington, 1993.
15. Vazquez-Duhalt, R.; Marquez-Rocha, F.; Ponce, E.; Licea, A.F.; Viana, M.T. Nonylphenol, an integrated vision of a pollutant (scientific review). *Appl. Ecol. Environ. Res.* **2005**, *4*, 1–25.
16. Crawford, R.L.; Hess, T.F.; Paszczynski, A. Combined biological and abiological degradation of xenobiotic compounds. In *Biodegradation and Bioremediation, Soil Biology Series*; Singh, A., Ward, O.P., Eds.; Springer-Verlag: Berlin, 2004; Vol. 2, 251–278.
17. Ramos, J.L.; Duque, E.; Godoy, P.; Segura, A. Efflux pumps involved in toluene tolerance in *Pseudomonas putida* DOT-T1E. *J. Bacteriol.* **1998**, *180*, 3323–3329.
18. Singh, A.; Kuhad, R.C.; Shareefdeen, Z.; Ward, O.P. Methods for monitoring and assessment of bioremediation processes. In *Biodegradation and Bioremediation, Soil Biology Series*; Singh, A., Ward, O.P., Eds.; Springer-Verlag: Berlin, 2004; Vol. 2, 279–304.
19. Van Hamme, J.D.; Singh, A.; Ward, O.P. Surfactants in microbiology and biotechnology: Part 1. Physiological aspects. *Biotechnol. Adv.* **2006**, *24*, 604–620.
20. Singh, A.; Van Hamme, J.D.; Ward, O.P. Surfactants in microbiology and biotechnology: Part 2. Application aspects. *Biotechnol. Adv.* **2007**, *25*, 99–121.
21. VanHamme, J.D.; Ward, O.P. Physical and metabolic interactions of *Pseudomonas* sp. Strain-B45 and *Rhodococcus* sp. F9-D79 during growth on crude oil and the effect of chemical surfactants thereon. *Appl. Environ. Microbiol.* **2001**, *67*, 4874–4879.
22. VanHamme, J.; Ward, O.P. Influence of chemical surfactants on the biodegradation of crude oil by a mixed culture. *Can. J. Microbiol.* **1999**, *45*, 130–137.
23. Billingsley, K.A.; Backus, S.M.; Wilson, S.; Singh, A.; Ward, O.P. Remediation of PCBs in soil by surfactant washing and biodegradation in the wash by *Pseudomonas* sp. LB400. *Biotechnol. Lett.* **2002**, *24*, 1827–1832.
24. Ward, O.P. Biosurfactants in biodegradation and bioremediation. *Adv. Exper. Med. Biol.* **2010**, *672*, 65–74.
25. Ward, O.P.; Singh, A.; Van Hamme, J.D.; Voordouw, G. Petroleum microbiology. In *Encyclopedia of Microbiology*, 3rd Ed.; Schaechter, M., Ed.; Elsevier: Amsterdam, 2009; 443–456.
26. Yapa, C.L.; Gana, S.; Ng, H.K. Application of vegetable oils in the treatment of polycyclic aromatic hydrocarbons-contaminated soils. *J. Hazard. Mater.* **2010**, *177*, 28–41.
27. Tyagi, M.; Fonseca, M.M.R.; de Carvalho, C.C.C.R. Bioaugmentation and biostimulation strategies to improve the effectiveness of bioremediation processes. *Biodegradation* **2011**, *22*, 231–241.
28. Ye, J.; Singh, A.; Ward, O.P. Biodegradation of nitroaromatics and other nitrogen-containing xenobiotics. *World J. Microbiol. Biot.* **2004**, *20*, 117–135.
29. Ward, O.P.; Singh, A.; Van Hamme, J. Accelerated biodegradation of petroleum hydrocarbon waste. *J. Ind. Microbiol. Biot.* **2003**, *30*, 260–270.
30. Delisle, S.; Greer, C.W. Remediation of organic pollutants through natural attenuation. In *Applied Bioremediation and Phytoremediation, Soil Biology Series*; Singh, A., Ward, O.P., Eds.; Springer-Verlag: Berlin, 2004; Vol. 1, 159–186.
31. Ward, O.P.; Singh, A. Evaluation of current soil bioremediation technologies. In *Applied Bioremediation and Phytoremediation, Soil Biology Series*; Singh, A., Ward, O.P., Eds.; Springer-Verlag: Berlin, 2004; Vol. 1, 187–214.
32. Macek, T.; Kotrba, P.; Svatos, A.; Novakova, M.; Demnerova, K.; Mackova, M. Novel roles for genetically modified plants in environmental protection. *Trends Biotechnol.* **2008**, *26*, 146–152.
33. Gerhardt, K.E.; Xiao-Dong Huang, X.-D.; Glick, B.R.; Bruce, M.; Greenberg, B.M. Phytoremediation and rhizoremediation of organic soil contaminants: Potential and challenges. *Plant Sci.* **2009**, *176*, 20–30.
34. Shareefdeen, Z.; Singh, A., Eds. *Biotechnology for Odour and Air Pollution Control*; Springer-Verlag: Berlin, 2005.

Biodiversity and Sustainability

Odo Primavesi

Brazilian Corporation of Agricultural Research (EMBRAPA), Sao Paulo, Brazil

Abstract

The discussion on sustainability and biodiversity is focused at the level of the economic, agricultural, and policy sectors. However, there are different interpretations about this discussion, depending on the point of view of the party presenting the argument. In this entry, a global view is given to better understand the specificity based on the ecological principles that run nature. These principles remain as the source of inspiration and reference for the development of useful concepts to promote life and its quality.

BIODIVERSITY

What Is Biodiversity?

The U.N. Convention on Biological Diversity defines biological diversity as “the variability among living organisms from all sources, including terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species, and of ecosystems.”^[1] Table 1 shows the global measured and estimated biodiversity in the world ecosystem.

Diversity is considered at different levels, between individuals, subspecies, species (most useful level), biological communities, and ecosystems. Species richness increases from colder to warmer latitudes. This is also true for the deep-sea species diversity.^[2]

The biological diversity is organized in a food web, with plants as base and humans as top of the pyramid and where the individuals act as producers or consumers or as recyclers or decomposers.^[3]

The soil is one of the most diverse habitats on earth and contains one of the most diverse assemblages of living organisms, mainly in the humid tropics; e.g., a single gram of soil may contain millions of individuals and several thousand species of bacteria.^[4] Table 2 shows a 100 × 30-cm-deep soil life in a pasture under temperate climate, remembering that in tropical conditions, there is a greater presence of termites, ants, larvae of beetles, and flies (Table 3). But in a broader view, it is advisable to consider soil as the undissociable soil–plant interaction. This will improve the degree of soil biodiversity, including the rooting system architectures or the so-called weeds, which are visible indicators of the degree of soil health.^[5]

In both natural and agricultural ecosystems, the different groups of soil biota interacting with plants (Table 4)

and their debris (Table 5) are responsible for, or strongly influence, the soil properties and optimize processes, including soil genesis; soil structure; carbon, nutrient, and water cycles; agrochemical movement or breakdown; and plant protection and growth^[3,6] They act in the processes of synthesis or production, transformation and decomposition, and consumption of organic material, affecting abiotic and biotic components, transportation, and soil engineering.

Therefore, soil biodiversity is a ground stone for sustainable agriculture, and it could be used as a good indicator of agroecosystem or soil health. Soil biodiversity does not necessarily refer to the number of individuals or species, but to the ratio of functional groups (Table 6) and the result or the tool of their activities, such as the presence and intensity of enzymatic activity.^[5]

What Is Its Importance?

In general, the arguments for biodiversity conservation and their importance are optimized environmental services (water cleaning up, recycling, biodegradation, soil permeabilizing, fertilization, etc.), food supplies, natural products, materials, medicines, fertilizers, pesticides, biological control agents, warning signs, genes, model systems for science, interesting wildlife, future options.^[2] Considering the process nature uses to develop a site or to recover its resting soil, it could be seen that biodiversity is the key tool used because it allows the complementary activity of individuals with different structures, needs, wastes, and functions to flourish in one of the diverse habitats created by the diverse interaction of the abiotic and emergent biotic factors occurring in this site. Nature uses biodiversity to produce the maximum of life and biomass per square meter and the available energy unity per year. But biodiversity is also the result of this settlement process of nature. At the same time, nature, by

Table 1 Global diversity of species.

Groups	Cataloged	Estimated	Note
Total	1.4 million	5–50 million	
Mammals	4,000	90–95%	
Reptiles and amphibians	10,848	90–95%	
Birds	9,040	94–100%	2–3 times in the tropics
Fish	19,056	83–100%	
Plants	322,311	67–100%	
	Trees	(50,000)	estimated for the tropics
Invertebrates	1,020,561	3–27%	
	Beetles		40% of arthropods
	Hemipterans	78,656	37.5% 10% of all insects
Microorganisms	5,760	3–27%	

Source: From Canada Agriculture and Agri-Food/Environment Bureau.^[1]

developing a short food chain into a complex food net, allows for a greater food availability and diversity for the individuals on the top of the food web or food pyramid, such as the humans, and also ensures their sustainability. When the biodiversity of the food web is disrupted by the establishment of a monoculture and when the environment is submitted to a degradation process, with malnourished plants and greater meso- and microclimate variability, the population outbreak of the more resistant members of the food web may occur as the so-called parasites and pathogens.^[5]

The reason for the species richness of the tropics is not known. Some ideas proposed take longer time to develop a new species and require greater supply of solar energy allowing more biomass production or more organisms per unit area.^[2]

Furthermore, the shallower soils in temperate climates show a greater chemical fertility, water-holding capacity, and clay activity, and the cold switch-off of biological activity will control it. In tropical conditions, the deep soils with low fertility, low water-holding capacity, low clay

activity, and the higher temperatures throughout the year will result in a greater number of interactions of water/drought × water table depth × nutrient × temperature × strong rains × wind × fire × photoperiod × oxygen (because of faster respiration rates); therefore, habitat variability occurs, with specificities settled in by the different plant species, the first component of the food web. The diversity of litter, defense substances, and root exudates produced by these different plant species and correlated fauna need to be recycled by a greater number of invertebrates and microorganisms in soil, because of their specificity in producing from 1 (bacteria) to 4 (fungi, insects) degradation enzymes. The high recycling activity in soil need to be considered because of the great importance of organic material (50–90%) as nutrient source for higher plants, besides the microbial activity of rock solubilizers, nitrogen (N)-fixing bacteria, or root surface expanding fungi (Mycorrhizae): biological fertility. In the tropics, with the great variability of habitats, the biodiversity of species and genes is the keystone for high biomass yield per high variable unit area, making the food net of an ecosystem very complex.

Table 2 Soil fauna number and weight under a pasture in temperate climate.

Groups	Minimum	Maximum	Optimum	Weight (g) of the optimum
Protozoa	—	—	1,551,000,000	10
Nematoda	1,800,000	120,000,000	21,000,000	40
Acarina	20,000	400,000	100,000	10
Collembola	10,000	440,000	50,000	20
Formicidae	200	500	—	—
Oligochaeta	600	2,000	800	400
Enchytraeids	10,000	200,000	200,000	26
Mollusca	20	1,000	50	30

Source: From Dunger,^[7] Kevan,^[8] and Primavesi.^[5]

Table 3 Soil macroinvertebrates (number/m²) in tropical environment, at the first 10-cm layer, under the litter.

Groups	Semideciduous forest		Brachiaria pasture	
	Dry season	Rainy season	Dry season	Rainy season
Isoptera (termites)	0	746	4344	362
Formicidae (ants)	1091	99	1059	48
Annelida (rainworms)	48	51	174	253
Coleoptera (beetles)	35	32	123	11
Coleoptera larvae	29	32	22	16
Diplopoda	96	35	21	45
Chilopoda	74	0	16	3
Arachnida (spiders)	29	35	19	10
Hemiptera	0	0	54	14
Mollusca (snails)	70	80	3	0
Thysanura	23	0	17	0
Crustacea	3	0	8	8
Lepidoptera larvae	10	0	2	0
Blattodea	3	0	3	0
Dermaptera	0	0	0	2

Source: From Brigante.^[9]

Biodiversity reaches the maximum level when the environment offers enough water and energy and low-to-medium level of nutrients, such as N and phosphorus, avoiding the very hard inter- and intraspecific competition from some more responsive or demanding species, occurring in high-fertility and very high-fertility soils, or the growth-restrictive conditions in very low-fertility soils, such as soils with high aluminum content.

Table 4 Effect of wheat rhizosphere on microbial population in soil, under temperate climate.

Microorganism	Number	
	Soil far from root	Soil next to root surface
Total	57,700,000	1,121,000,000
Fungi	120,000	1,160,000
Protozoa	990	2,410
Algae	26,900	4,500
Nitrifier	100,000	100,000
Denitrifier	140,000	12,650,000
Ammonifier	1,800,000	100,000,000
Sporogenic bacteria	575,000	927,000
Aerobic cellulolytic bacteria	2,700	720,000
Anaerobic cellulolytic bacteria	1,800,000	9,100

Source: From Katznelson^[10] and Primavesi.^[5]

SUSTAINABILITY

What Is Sustainability?

In 1987, the World Commission on Environment and Development established a definition of sustainability, known as the Brundtland Report. It stated that sustainable development meets the needs of the present without compromising the ability of future generations to meet their own needs. Although this definition has become widely publicized, the term sustainability is not limited to one precise definition.^[11]

Several authors have discussed the real meaning of sustainability and sustainable development.^[9,12] Munoz^[13] has presented a theoretical model of true sustainability or sustainable development ideas with global to local implications based on the need to balance social, economic, and environmental goals; stakeholders's interests; issues to induce or determine fairer or more appropriate development solutions, options, and actions. However, the social component can be seen as part of the environmental component, and the interaction of both will generate the economic component. So the environmental component is the keystone, and only its improvement and quality will allow us to reach a stable social welfare and sustainable economic profit.

What Is Nature Teaching Us?

Nature teaches us that the development of a natural primary environment (rocky) or degraded (soil in rest for some years to restore its productive potential) area, to an on-soil-based natural climax environment (mainly forest), with a high production potential, occurs with the development or restoration of a permeable soil, protected by a diversified vegetation with an active rooting system, and the return to soil of diversified organic material, the energy source for the diversified and active soil life. The soil vegetation and associated biodiversity interaction could improve the available resident water of a site and a longer water cycle. The more the resident water, the more the vegetation and the more permeable the soil. With more available resident water-permeable soil-diversified vegetation and soil life, there is an increase of relative air humidity and a decrease of the maximum temperature and the thermal amplitude, characteristic for desert environments. This friendlier mesoclimate helps more sensible plant and animal species to establish, and it improves biodiversity, with their additive and emergent characteristics.

Ecology considers desert ecosystems sometimes as sustainable as tropical forest ecosystems. So what level of environmental sustainability is desirable? The biological carrying capacity (BCC) is considered as the representative of this level. It represents the concept of primary productivity of an ecosystem, or the rate in which energy

Table 5 Physiologic groups of microbes in soil, incubated with and without composted rice straw, fertilized with ammonium sulfate.

Microorganism	Number per gram of incubated soil			
	Straight after mixture		Four-month incubation	
	Without straw	With straw	Without straw	With straw
Total bacteria	800,000	500,000	250,000	800,000
Total fungi	750,000	1,400,000	750,000	900,000
Aerobic N fixing	0	0	25	13
Anaerobic N fixing	7,500	45,000	2,500	95,000
Nitrifiers	450	450	110	16
Nitritifiers	45	95	600	2,500
Ammonifiers	4,500,000,000	9,500,000,000	5,000,000,000	4,500,000,000
Aerobic cellulolytic	300	1,500	2,500	2,500
Anaerobic cellulolytic	115	950	250	2,500
Reducing sulfobacteria	250	1,500	250	2,500
Oxidizing sulfobacteria	2,500	2,500	250	950
Organic S mineralizers	0	0	9	0

Source: From Inamatsu^[14] and Primavesi.^[5]

is stored by photosynthetic or chemosynthetic activity of the producer organisms as organic substances, food for the food web.^[3] The BCC considers also, e.g., the feeding capacity of grazing animals (0.5 or 6 animal units of 450 kg live weight/ha yr) or grain equivalent available for humans (4 or 16 persons/ha yr, with a minimum need of 1000 cal/day), calculated as available digestive energy or calories per surface unit and year. The BCC depends on the recovering capacity of a site to produce biomass, after yield, extraction, degradation, or pollution activities. Considering the exuberant flora and biomass production by the Amazonian forest, the following question arises: Which is the BCC of the mostly sandy soils in the Amazonian basin without that vegetation and the aggregated or dependent mesoclimate? Something similar to the Sahara? This example shows that the BCC may be managed in a certain range and sustainability improving the tripod WSB. Considering that the main objective of all human activity, from global to local scale, is to promote life and its quality, mainly the human one, the key tool in the economic system, as a producer and a consumer, the environmental

sustainability will be reached when the BCC is adequate to supply the minimal health life requirements of a given human population. The increase in the BCC level will allow a rise in the human population density. What is occurring is the destruction of the main natural resources—the decrease in the BCC with an increase in the human population density. If nothing is carried out to revert and/or prevent such destruction, then a global disaster with acute local consequences is possible.

CONCLUSION

Looking for the macroscale procedures nature uses to develop the BCC of a sustainable site and its sustainability implications, it could be seen that the increase of resident water, the production or restoration of a surface-protected permeable soil, and the development of a diversified flora and fauna are reflected on the whole biodiversity of the soil macro-, meso-, and microlife and the soil–plant complex. These procedures, based on ecological principles, need to be globally forwarded/passed to the whole human population through formal and informal environmental education programs, to increase the awareness of their vital tie to the environment and to the need to improve or maintain its health to guarantee life and its quality for the existing and future generations.

REFERENCES

1. Canada Agriculture and Agri-Food/Environment Bureau. *Biodiversity*. Available at http://www.agr.gc.ca/policy/environment/biodiv_e.phtml (accessed October 2002).

Table 6 Soil fauna ratio changes with soil degradation in the tropics.

Biotype	Acarina		Collembola		Ratio
	Number	%	Number	%	
Riparian forest	57,242	79.5	10,123	14.1	5.65
Dryland forest	49,749	78.1	11,509	18.1	4.32
Brachiaria pasture	56,144	63.7	26,973	30.6	2.08
Sunflower field	23,144	55.5	15,485	37.1	1.49

Source: From Maldague^[15] and Primavesi.^[5]

2. Bryant, P.J. *Biodiversity and Conservation: A Hypertext Book*. Available at <http://darwin.bio.uci.edu/~sustain/bio65/Titlpage.htm> (accessed February 2003).
3. Odum, E.P. *Fundamentals of Ecology*, 2nd Ed.; W. B. Saunders Co: Philadelphia, 1959.
4. FAO. *Soil Biodiversity Portal*; Montanez, 2000. Available at <http://www.fao.org/ag/AGL/agll/soilbiod/soilbiod.htm> (accessed October 2002).
5. Primavesi, A.M. *Manejo Ecológico Do Solo: A Agricultura Em Regiões Tropicais*; Nobel: São Paulo, 1980.
6. Dumanski, J.; Gameda, S.; Pieri, C. *Indicators of Land Quality and Sustainable Land Management: An Annotated Bibliography*; The World Bank: Washington, 1998.
7. Dunger, W. *Tiere im boden*. A. Ziemsen: Wittenberg, 1964.
8. Kevan, L. Cation interactions of trapped electrons in irradiated alkaline ice. *J. Amer. Chem. Soc.* **1965**, *87* (7), 1481–1483.
9. Brigante, J. *A Substituição Do Sistema Natural Por Sistema de Pastagens e Seus Efeitos Sobre as Comunidades Microbiológicas e de Macrofauna Invertebrada, Em Um Latossolo Tropical*; Ph.D. Thesis, Universidade Federal de São Carlos—PPG Ecologia e Recursos Naturais, São Carlos, 2000.
10. Katznelson, H.; Lochhead, A.G.; Timonin, M.I. Soil micro-organisms and the rhizosphere. *Botanical Rev.* **1948**, *14* (9), 543–586.
11. Alliance for Global Sustainability. *Sustainability*. Available at <http://www.global-sustainability.org/Education/Definitions/> (accessed October 2002).
12. FAO. *Land Quality Indicators and Their Use in Sustainable Agriculture and Rural Development*; Land and Water Bulletin 5; FAO, UNDP, UNEP, World Bank: Rome, 1996.
13. Munoz, L. Understanding sustainability versus sustained development by means of a WIN development model. In *Sustainability Review*; Flint, W., Ed.; Five E's Unlimited: Pungoteague, 1999. Available at <http://www.interchange.ubc.ca/munoz> (accessed October 2002).
14. Inamatsu, K.; Tardieux, A.; Moussin, M.H.; Pochon, J. Influence of rice straw compost on the humic substances and microbes of a soil planted to mulberries. *Rev. Ecol. Biol. Sol.* **1974**.
15. Maldague, M.E. *Relations entre le couvert végétal et la microfaune: leur importance dans la conservation biologique des sols tropicaux*. Publications de l'Institut National pour l'Étude Agronomique du Congo. Vromant: Brussels, 1961.

Bioenergy Crops: Carbon Balance Assessment

Rocky Lemus

Mississippi State University, Starkville, Mississippi, U.S.A.

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University,
Columbus, Ohio, U.S.A.

Abstract

Bioenergy crops are perennial crops with the potential to sequester carbon (C) into the biomass and soil while providing an alternative source of energy. The decrease in crop productivity due to soil degradation and the increased risks of global warming warrant assessing alternative sources of energy produced on degraded soils. New strategies to mitigate atmospheric carbon dioxide by increasing soil organic matter and biomass productivity are some of the alternatives through bioenergy crops.

OVERVIEW

Since the 1990s, several species have been assessed as potential energy sources and for carbon (C) sequestration. Switchgrass (*Panicum virgatum* L.), a herbaceous species, and two short-rotation woody crops (SRWCs), namely hybrid poplar (*Populus* spp.) and willow (*Salix* spp.), are promising for C sequestration and biofuel production. Using these species as energy crops can partly offset carbon dioxide (CO₂) emitted by fossil fuel combustion.

There are approximately 60 million ha of degraded soil (severely eroded and mined land) in the United States that can be sown to bioenergy crops to decrease soil erosion and increase C sequestration.^[1] The Midwest and the Southeast regions of the United States are the potential areas where bioenergy crops are most likely to compete with other traditional crops for land resources.^[2] The use of bioenergy crops in these areas offers an opportunity to replenish the soil organic C (SOC) pool depleted by tillage and soil degradation.

Biomass fuels used in a sustainable manner can result in no net increase in atmospheric CO₂. Indeed, sustainable use of biomass can result in a net decrease in the rate of enrichment of atmospheric CO₂. This is based on the assumption that all the CO₂ emitted by the use of biomass fuels is absorbed from the atmosphere by photosynthesis. Increased substitution of fossil fuels with biomass-based fuels reduces the risk of global warming by enrichment of atmospheric CO₂. This entry discusses the importance of energy crops in offsetting fossil fuel combustion through C sequestration in biomass and soil.

INFLUENCE OF BIOENERGY CROPS AT THE TEMPORAL SCALE

Long-term tillage and continuous cropping can reduce SOC pool^[3] and increase atmospheric concentration of CO₂ (Fig. 1).^[4] The conversion of natural to agricultural ecosystems results in the net release of C into the atmosphere.^[5] Conservation practices, such as the introduction of perennial crops on degraded agricultural soils and growing bioenergy crops, constitute a direct link between sink (SOC) and the source of CO₂ (fossil fuel). Bioenergy crops are the sink/source transition because the C incorporated into their biomass and root system has a high potential of being incorporated into the soil organic matter (SOM) pool. Most of the C is incorporated into the biomass and soil by the deep and extensive root systems. The SOC pool attains a new equilibrium after grassland restoration, but the time required for SOC to reach equilibrium is variable, especially under diverse climatic conditions.^[6] Most models used to estimate C sequestration in bioenergy crops suggest that a period longer than 10 years may be necessary for pronounced soil-quality improvements.

BENEFITS OF BIOENERGY CROPS

The SOC pool and its dynamics are the major indicators of soil quality and crop productivity.^[7] Bioenergy crops can improve soil quality through increase in SOM, nutrient dynamics, erosion control, and improvement in soil structure and porosity. Most perennial crops and SRWCs have extensive root systems that reduce soil erosion and non-point-source pollution.

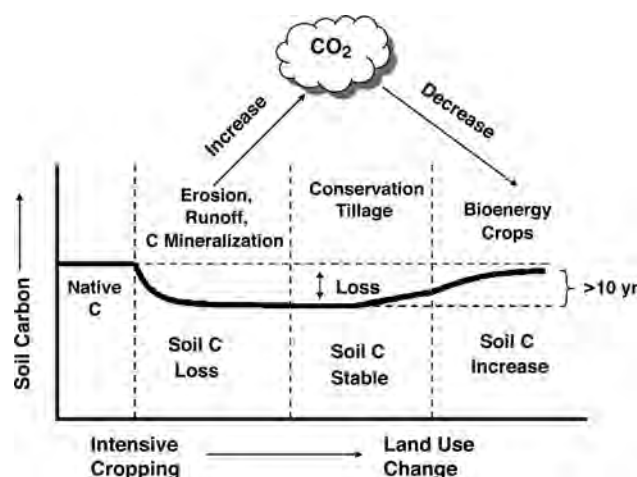


Fig. 1 Influence of bioenergy crops on C sequestration and CO₂ mitigation with changes in agricultural practices.
Source: Adapted from Janzen, Campell, et al.^[8]

A shift from traditional agricultural crops into perennial bioenergy crops stabilizes agricultural soils, reduces erosion, and improves water quality.^[9] Most of these benefits are due to the elimination of tillage leading to a significant decrease in both erosion and chemical runoff especially nitrate (NO₃-N). Soil erosion and nitrate runoff are reduced in 2-year-old stands of switchgrass when compared to no-till corn (Table 1). Conservation effectiveness of switchgrass is attributed to high root biomass, which stabilizes the soil,^[10] depletes excess nitrogen (N) and phosphorus,^[11] and increases microbial activity.^[12]

POTENTIAL OF C SEQUESTRATION

Bioenergy crops have the potential to store large quantities of C. Exploring their biomass potential through N fertilization and soil management are some of the proposed strategies to offset CO₂ emissions by fossil fuel combustion. However, a bioenergy crop is not a closed system, and only some portion of the C sequestered might be conserved through the production–utilization cycle. Most of the C sequestered in the biomass is utilized for energy production, allowing some C release back into the atmosphere.

This system does not cause negative impacts, as most of the C returned to the atmosphere is recaptured by the plants during the subsequent season.

Assessing the net C sequestration of bioenergy compared to agricultural crops requires analyses of C sequestration across the soil profile. C sequestration is influenced by biological processes, such as root biomass and crop species, and by soil physical and chemical aspects, such as texture, bulk density, and pH. The amount of C in these types of management options is affected by soil perturbation which could increase SOM oxidation, exacerbating losses, especially at the soil surface. The estimations of C sequestration on land under the Conservation Reserve Program have ranged from 0.6 to 1.0 Mg C ha⁻¹ yr⁻¹ in SOM,^[13] compared to 2.0 Mg C ha⁻¹ yr⁻¹ in SRWC.^[14] Root biomass and its relationship to SOC are always positive but may be influenced by soil depth and plant species.^[15] Switchgrass improves soil quality via reduced nutrient loss (especially NO₃-N) and increases C sequestration through its deep rooting system, high biomass production, and perenniality.^[16] Switchgrass root biomass can be as high as 16.8 Mg ha⁻¹ up to 3.3 m deep with some root mass variability among soil type.^[12]

The sustainability of soil and crop systems is typically affected by changes in the SOC pool. The largest gain in SOC pool occurs in the upper 30-cm layer. There are differences in the amount of C being sequestered by herbaceous crops and SRWCs, but they have the potential to increase C both in the soil and aboveground biomass. The data in Fig. 2 show that after 5 years SOC pool at 60-cm depth was 97.5 Mg ha⁻¹ under willow, 80.5 Mg ha⁻¹ under switchgrass, and 78.5 Mg ha⁻¹ under corn. The mean rate of SOC sequestration, vis-à-vis corn, over 5 years was 3.8 Mg C ha⁻¹ yr⁻¹ under willow and 0.4 Mg C ha⁻¹ yr⁻¹ under switchgrass (Fig. 2). The SOC fluctuations are more drastic in the topsoil layers probably because of the greater effect of precipitation, soil temperature, larger root biomass, and greater microbial activity.

CONCLUSION

Bioenergy crops can influence the global C cycle. An important challenge in using bioenergy crops is assessing how

Table 1 Environmental benefits of switchgrass (SWG) compared to no-till corn (NTC) overtime.

Year	Soil erosion			Water runoff			Nitrate concentration		
	SWG (Mg ha ⁻¹)	NTC (Mg ha ⁻¹)	Ratio SWG/NTC ^a	SWG (Mg ha ⁻¹)	NTC (Mg ha ⁻¹)	Ratio SWG/NTC	SWG (ppm)	NTC (ppm)	Ratio SWG/NTC
1	2.80	0.70	4.00	10.70	2.60	4.11	3.41	0.57	5.98
2	0.14	0.19	0.74	0.70	1.40	0.50	0.72	2.18	0.33
3	0.06	0.08	0.75	0.30	0.90	0.33	0.77	0.90	0.86

^aThe SWG/NTC ratio of <1 indicates conservation effectiveness of a SWG system during the 2nd and 3rd year of its establishment.

Source: Adapted from McLaughlin, De La Torre Ugarte, et al.^[17]

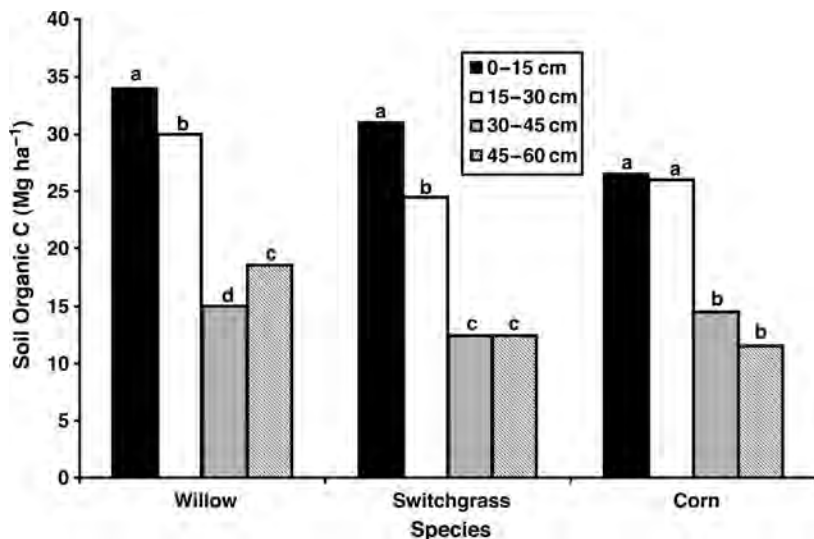


Fig. 2 Comparison of C sequestration between bioenergy crops and corn at different soil depths 5 years after establishment.

Source: From Mehdi, Zan, et al.^[18]

much C is being sequestered and how much is being put back into the atmosphere by the co-firing process to produce electricity or other sources of energy. Available data indicate that perennial crops are moving in the right direction in improving soil quality while providing an alternative energy source. A combination of soil C sequestration and bioenergy crops with high biomass capability and deep perennial root systems is a useful strategy of reducing the rate of enrichment of atmospheric CO₂. The use of bioenergy crops increases soil C and improves soil quality by eliminating C losses associated with annual cultivation. However, these improvements depend on the rate of soil C additions, the long-term capacity of the soil for C storage, and the stability or permanence of the sequestered C overtime.

REFERENCES

- Kort, J.; Collins, M.; Ditsch, D. A review of soil erosion potential associated with biomass crops. *Biomass Bioenerg.* **1998**, *14* (4), 351–359.
- Walsh, M.E.; Ince, P.J.; De la Torre Ugarte, D.G.; Alig, R.; Mills, J.; Spelter, H.; Skog, K.; Slinsky, S.P.; Ray, D.E.; Graham, R.L. Potential of short-rotation wood crops as a fiber and energy source in the U.S. In *Proceedings of the Fourth Biomass Conference of the Americas*; Elsevier Science Ltd.: Oxford, 1999; 195–206.
- Kern, J.S.; Johnson, M.G. Conservation tillage impacts on national soil and atmospheric carbon levels. *Soil Sci. Soc. Am. J.* **1993**, *57*, 200–210.
- Schlamadinger, B.; Marland, G. The role of forest and bioenergy strategies in the global carbon cycle. *Biomass Bioenerg.* **1996**, *10*, 275–300.
- Schulze, E.D.; Wirth, C.; Heimann, M. Managing forests after Kyoto. *Science* **2000**, *289*, 2058–2059.
- Potter, K.N.; Tober, H.A.; Johnson, H.B.; Tischler, C.R. Carbon storage after long-term grass establishment on degraded soils. *Soil Sci.* **1999**, *164*, 718–725.
- Mann, L.; Tolbert, V. Soil sustainability in renewable biomass plantings. *Ambio* **2000**, *29* (8), 492–498.
- Janzen, H.H.; Campell, C.A.; Izaurrealde, R.C.; Ellert, B.H.; Juma, N.; McGill, W.B.; Zenter, R.P. Management effects on soil C storage on the Canadian prairies. *Soil Till. Res.* **1998**, *47*, 181–195.
- McLaughlin, S.B.; Walsh, M.E. Evaluating environmental consequences of producing herbaceous crops for bioenergy. *Biomass Bioenerg.* **1998**, *14* (4), 317–324.
- Gilley, J.E.; Eghball, B.; Kramer, L.A.; Moorman, T.B. Narrow grass hedge effects on runoff and soil loss. *J. Soil Water Conserv.* **2000**, *55*, 190–196.
- Eghball, B.; Gilley, J.E.; Kramer, L.A.; Moorman, T.B. Narrow grass hedge effects on phosphorus and nitrogen in runoff following manure and fertilizer application. *J. Soil Water Conserv.* **2000**, *55*, 172–176.
- Ma, Z.; Wood, C.W.; Bransby, D.I. Soil management on soil C sequestration by switchgrass. *Biomass Bioenerg.* **2000**, *18*, 469–477.
- Follet, R.F. Soil management concepts and carbon sequestration in zircon cropland soils. *Soil Till. Res.* **2001**, *61*, 77–92.
- Tuskan, G.A. Short-rotation woody crop supply systems in the United States: What do we know and what do we need to know? *Biomass Bioenerg.* **1998**, *14* (4), 307–315.
- Slobodian, N.; Van Rees, K.; Pennock, D. Cultivation-induced effects on belowground biomass and organic carbon. *Soil Sci. Soc. Am. J.* **2002**, *66*, 924–930.
- Sladden, S.E.; Bransby, D.I.; Aiken, G.E. Biomass yield, composition, and production costs for eight switchgrass varieties in Alabama. *Biomass Bioenerg.* **1991**, *1* (2), 119–122.
- McLaughlin, S.B.; De La Torre Ugarte, D.G.; Garten, C.T.; Lynd, L.R.; Sanderson, M.A.; Tolbert, V.R.; Wolf, D.D. High-value renewable energy from prairie grasses. *Environ. Sci. Technol.* **2002**, *36*, 2122–2129.
- Mehdi, B.; Zan, C.; Girouard, P.; Samson, R. *Soil Carbon Sequestration Under Two Dedicated Perennial Bioenergy Crops*; Research Reports; Resource Efficient Agricultural Production (REAP): Canada, 1998. Available at <http://www.reap-canada.com/Reports/reportsindex.htm> (accessed April 2004).

Biofuels versus Agricultural Soils

David Pimentel

Department of Entomology, Cornell University, Ithaca, New York, U.S.A.

Abstract

The rapidly growing world population and rising consumption of biofuels are increasing demand for both foods and biofuels. This exaggerates both food and fuel shortages and increases soil erosion. Using food crops such as corn grain to produce ethanol raises major agricultural, nutritional, and ethical concerns. Nearly 60% of humans in the world are malnourished, so the need for grains and other basic foods is critical. Growing crops for fuel squanders land, water, and energy resources vital for the production of food for human consumption.

INTRODUCTION

The world population is adding a quarter million people each day to the 6.7 billion people who are already present. Soon, it will increase to 13 billion humans on earth. Along with diverse human activities, industrial and transportation development are removing land and other basic resources that provide us with our food, water, and energy.

Conserving land for crops and livestock must become a priority to ensure a sustainable food supply now and for the future, as the world population continues to increase. The protection of farmlands is complicated by the government interest in converting corn crops and ultimately grassland into ethanol.

This entry focuses on conserving land essential for crops, pastures, and forests. Following is an evaluation of how the proposed conversion of biomass crops for ethanol production could impact the U.S. agricultural system and our valuable environmental resources for the future.

CONSERVING SOIL

More than 99% of our food (calories) comes from the land while less than 1% comes from the oceans and other aquatic ecosystems. Therefore, maintaining fertile cropland must be a primary concern of all who wish to ensure a sustainable food supply for humans and wildlife now and for the future.

Wind and rain displace and remove layers of topsoil and diminish the nutrients and the water holding capacity of the soil; this, in turn, reduces the productivity of the soil. A ton of fertile topsoil averages 1–6 kg of nitrogen, 1–3 kg of phosphorus, and 2–30 kg of potassium, whereas the soil on eroded land has average nitrogen levels of only 0.1–0.5 kg/t.^[1] Eroded soil carries away vital plant

nutrients such as nitrogen, phosphorus, potassium, and calcium. Typically, the eroding soil contains about three times more nutrients than are left in the remaining soil.

When soil nutrient resources are depleted by erosion, plant growth is stunted and overall productivity declines.^[2,3]

Nutrient deficient soils produce 15–30% lower crop yields than uneroded soils.

Land areas covered by plant biomass, living or dead, are protected and experience relatively little soil erosion because raindrops and wind energy are dissipated by the biomass layer. For example, in Utah and Montana, as the amount of ground cover decreased from 100% to <1%, erosion rates increased ~200 times.

On average, the soil erosion rate on U.S. cropland is about 10 t/ha/yr.^[4] However, these rates of erosion greatly exceed the average rate of natural soil formation from the parent material. Under agricultural conditions, soil formation ranges from 0.5 to 1 t/ha/yr.^[5,6] Thus, U.S. cropland is losing soil more than 10 times faster than its replacement rate.^[4]

In the United States, soil erosion is severe on some of the most productive agricultural ecosystems. For instance, one half of the fertile topsoil of Iowa has been lost by erosion during the last 150 years of farming. These high rates of erosion continue there at a rate of about 30 t/ha/yr both because of the rolling hill topography and type of agriculture practiced.^[7] Similarly, 40% of the rich soil of the Palouse region in the northwestern United States has been eroded during the past years of cultivation. In both of these regions, intensive agriculture is employed and monocultural plantings are common. Also, many such areas are left unplanted during the late fall and winter months, further exposing the soil to erosion. Yearly in the United States, several thousand hectares of once valuable cropland are abandoned, because rain and wind erosion have made them unproductive.

To offset some of the nutrient losses caused by erosion, large quantities of fossil-based fertilizers often are applied.

If the soil base is relatively deep, about 300 mm, and if only from 10 t to 20 t of soil is lost per hectare per year, the lost nutrients can be replaced with the application of commercial fertilizers and/or livestock manure.^[2,3] For example, of the ~900 L of fossil energy required to produce 1 ha of corn, about 300 L of this energy is for fertilizers.^[8] However, this replacement strategy is expensive for the farmer and nation and usually not affordable by poor farmers. The estimate is that the lost soil nutrients cost U.S. agriculture \$10–\$20 billion annually. Furthermore, not only are the fertilizer inputs fossil-energy dependent, these chemicals can harm human health and pollute the environment.^[9]

Soil erosion in the United States is reported to cause \$45 billion in environmental and public health impacts each year.^[2] Removing about half of the biomass, including all crop residues, would intensify the soil loss and more than double the economic losses from soil erosion to >\$100 billion per year.

Although erosion rates on U.S. cropland have decreased during the past decades, erosion rates on rangelands remain relatively high or about 6 t/ha/yr.^[9] Such high erosion rates are typical on more than half of the world's rangelands including those in the United States. Soil erosion on U.S. pastures and rangelands is having a negative impact on livestock production, which relies on pasture grasses.

BIOFUEL PRODUCTION

Basic to producing all biomass (food crops, including corn, sugarcane, soybeans, and cereal grains) are large land areas, water supplies, and fossil energy. For ethanol and biodiesel production, these resources are used both in the production of the biomass and subsequently in the processing of the biomass to produce ethanol or biodiesel.

To guarantee increased production of corn ethanol, the U.S. government spends about \$6 billion in subsidies. The corn ethanol subsidies per gallon are 60 times greater than the subsidies per gallon of gasoline!

For ethanol production, vast quantities of corn must be grown. Twenty-two pounds of corn are required to produce one gallon of ethanol. A total of 1700 gal of water is required to produce 1 gal of ethanol. Water use for ethanol processing plants is draining aquifers that support some local communities. Especially, large quantities of water are required for corn production. In the United States, corn production uses 5,600,000 L of water per hectare in one growing season. If irrigation is necessary, about twice that amount is required. Each gallon of ethanol produced also creates about 12 gal of sewage effluent that must be discarded into the local sewage system. Another challenge is that the ethanol must most often be transported by truck and/or rail.

Pimentel and Patzek report (2007) that 46% more fossil energy is required to produce 1 L of ethanol from corn than

the energy in the ethanol;^[8] in contrast, some in the U.S. Department of Agriculture (USDA) report anywhere from a 21% to a 64% net return in corn ethanol.

Because corn is a food for cattle, hogs, and chickens, livestock producers are faced with increased costs and the consumer is faced with increased prices of milk, meat, and eggs. In addition, over many decades U.S. corn has been a staple food for many poor people in the world.

In 2006, the United States is producing 5 billion gallons of ethanol, and this production is using 20% of all U.S. corn and represents only 1% of U.S. petroleum use. If 100% of U.S. corn were to be used, it would provide only 7% of U.S. petroleum use (Table 1).

The USDA reports a 50% greater return using corn stover to produce ethanol than using only the corn grain. This is extremely surprising, when it is known that corn stover has only half of the starches and sugars that corn grain has. In addition, to release the starches and sugars held in the lignin in the corn stover, either an acid or special enzyme is needed to dissolve the lignin.^[8]

Perlack et al. claim that 1.5 billion tons of biomass can be harvested each year from U.S. land and converted into biofuels providing the nation with 30% of its liquid fuel needs.^[10] This suggests that one half of the 3 billion tons of all biomass (such as crops, forests, and grasses) produced annually in the United States can be utilized. No consideration was given to the further needs of agriculture or the needs of wildlife and other organisms. Under growing pressure to replace dwindling fossil-fuel supplies, focus is on grassland and forest biomass to produce ethanol. Tilman, Hill, and Lehman^[11] and Perlack et al.^[10] recommend the harvesting of crop residues such as corn stover for the biofuels program. This would be especially disastrous for agricultural ecosystems. Note that there are 100 million cattle, 7 million sheep, and 4 million horses grazing on grasses—what will these livestock be fed if all or most of the grass is used for cellulosic ethanol?

Without the protection of crop residues such as corn stover, soil loss in crop production would increase from 10-fold to 100-fold. Already the U.S. crop system is losing soil 10 times faster than sustainability.^[9] Soil formation rates are extremely slow, ranging from 0.5 to 1.0 t/ha/yr.^[1,9] Increased erosion will facilitate soil carbon oxidation and contribute to the greenhouse problem.^[12]

Table 1 Energy inputs and outputs of some U.S. biofuels per liter of 99.5% ethanol or biodiesel.

Crop	Input (kcal/L)	Output (kcal/L)	Gain or loss (%)
Corn	7,474	5,130 (ethanol)	−46
Switchgrass	8,634	5,130 (ethanol)	−68
Sugarcane	4,578	5,130 (ethanol)	+04
Soybean	12,564	9,000 (biodiesel)	−40

Source: From Pimentel & Patzek.^[8]

Such an intensification of the U.S. soil erosion problem would significantly reduce not only corn, but also other crop production.^[1] Corn production already causes more soil erosion than any crop produced in the United States.^[9] Removing most of the corn stover, or any other crop residues, would devastate corn and all other crops, especially row crops. Note that soil loss from row crops, such as corn and soybeans used as biofuels, is about 50 times higher than the soil erosion rates with sod-type crops, such as pastures.

CONCLUSION

The continued expansion of biomass for ethanol production will devastate vast land areas and expose more soil to erosion. Based on existing biomass production and processing methods, the ethanol produced will not make the U.S. oil independent. Meanwhile, billions of government funds are being spent to encourage ethanol production.

The conversion of millions of hectares of land for biomass and ethanol production is doing long-lasting damage to vital soil and water resources of the United States.

ACKNOWLEDGMENT

The author acknowledges his sincere gratitude to the Cornell Association of Professors' Emeriti for the partial support of our research through the Albert Podell Grant Program.

REFERENCES

1. Troeh, F.R.; Hobbs, J.A.; Donahue, R.L. *Soil and Water Conservation*; Prentice Hall: Englewood Cliffs, 2004.
2. Pimentel, D.; Harvey, C.; Resosudarmo, P.; Sinclair, K.; Kurtz, D.; McNair, M.; Crist, S.; Spritz, L.; Fittton, L.; Saffouri, R.; Blair, R. Environmental and economic costs of soil erosion and conservation benefits. *Science* **1995**, *267*, 1117–1123.
3. Pimentel, D. Soil erosion: A food and environmental threat. *Environ. Dev. Sust.* **2006**, *8*, 119–137.
4. USDA. *Changes in Average Annual Soil Erosion by Water on Cropland and CRP Land, 1992–1997*; Natural Resources Conservation Service, USDA, Washington, DC, 2000.
5. Troeh, F.R.; Thompson, L.M. *Soils and Soil Fertility*, 5th Ed.; Oxford University Press: New York, 1993.
6. Lal, R. Water management in various crop production systems related to soil tillage. *Soil Till. Res.* **1994**, *30*, 169–185.
7. USDA. *The Second RCA Appraisal. Soil, Water, and Related Resources on Nonfederal Land in the United States: Analysis of Conditions and Trends*; U.S. Department of Agriculture: Washington, DC, 1989.
8. Pimentel, D.; Patzek, T. Ethanol production using corn, switchgrass, and wood: Biodiesel production using soybean. In *Biofuels, Solar and Wind as Renewable Energy Systems: Benefits and Risks*; Pimentel, D., Ed.; Springer: Dordrecht, 2008; 373–394.
9. NAS. *Frontiers in Agricultural Research: Food, Health, Environment, and Communities*; National Academy of Sciences Press: Washington, DC, 2003.
10. Perlack, R.; Wright, L.; Turlow, A.; Graham, R.; Stokes, B.; Erback, D. *Biomass as Feedstock and Bioproducts Industry: The Technical Feasibility of Billion-Ton Annual Supply*; Energy Efficiency and Renewable Energy, Office of the Department of Energy Program and USDA: Oak Ridge, 2005.
11. Tilman, D.; Hill, J.; Lehman, C. Carbon-negative biofuels from low-input high-diversity grassland biomass. *Science* **2006**, *314*, 1598–1600.
12. Lal, R. Global potential of soil C sequestration to mitigate the greenhouse effect. *Crit. Rev. Plant Sci.* **2003**, *22*, 151–184.

Biological Crusts

Jayne Belnap

Southwest Biological Science Center, U.S. Geological Survey (USGS), Moab, Utah, U.S.A.

Abstract

Desert soil surfaces are generally covered with biological soil crusts, a group of organisms dominated by cyanobacteria, lichens, and mosses. Despite their unassuming appearance, these tiny organisms are surprisingly critical to many processes in past and present desert ecosystems and are vital in creating and maintaining fertility of desert soils. They fix both carbon and nitrogen, much of which is leaked to the soils below. They stabilize soils, capture nutrient-rich dust, and can stimulate plant growth. These organisms must tolerate extreme temperatures, drought, and solar radiation, despite having relatively few wet hours for metabolic activity. Under most circumstances, they are extremely vulnerable to climate change and disturbances such as off-road vehicles and livestock grazing. Unfortunately, recovery times are generally measured in decades or centuries.

INTRODUCTION

Biological soil crusts (BSCs; Fig. 1) are a community of cyanobacteria, microfungi, lichens, or mosses that occur on the soil surface in most dryland and subhumid regions as well as in other ecosystems where light can reach the soil surface (e.g., treefall gaps and pine barrens; see the review by Belnap and Lange^[1]). Well-developed BSCs can dominate the living cover in dryland regions and can contain more diversity than the surrounding vascular plant community. The external morphology of BSCs is determined by climate and species composition, with four main types being recognized (Fig. 2); cyanobacterially dominated *smooth crusts* are found in hyperarid regions [no frost heaving, very high potential evapotranspiration (PET)]. *Rugose crusts*, with a microtopography <3 cm, are found in arid deserts (no frost heaving, high PET, and very low moss–lichen cover). *Pinnacled crusts* occur in semiarid cool deserts (moderate PET, lichen–moss cover <40%, and freezing soils) and have a microtopography of up to 15 cm. *Rolling crusts* are found in semiarid cool/cold deserts (low PET, freezing soils, and >40% lichen–moss cover), but the cohesion of the high moss–lichen cover restricts the pinnacles to <5 cm. The external morphology of BSCs is very important, as it influences how and whether materials such as dust, water, organic matter, and seeds move across or are retained by the soil surface.

The desert soil surface is one of the most extreme environments on earth. Temperatures can range from –20°C to +70°C. Solar radiation is high, and rainfall is infrequent and sparse. The key to survival for BSC organisms is their ability to desiccate and terminate or slowdown the metabolic processes. In addition, many BSC species photosynthesize at

high and low temperatures, have high water-holding capacities, require only low amounts of moisture to begin metabolic activities, and can cope with high radiation loads. Many BSC taxa are similar around the world, despite their occurrence in a wide range of climates and vegetation types. Unrelated taxa often have comparable structures, implying that soil surface conditions have produced convergent evolutionary trends.

Biological crusts perform many critical roles in the ecosystems in which they occur. Well-developed, late successional BSCs (supporting lichens and mosses) are better able to perform the functions discussed below than BSCs kept at an early successional stage by climate or disturbance.

WEATHERING

BSCs increase water retention, thus enhancing freeze–thaw action and the dissolution of minerals in rocks. The metabolic activities of BSCs increase soil pH approximately by 2 pH units (e.g., ~8 to ~10.5), accelerating weathering rates of siliceous rock by up to 100 times.^[2]

SOIL PHYSICAL STRUCTURE AND STABILIZATION

Polysaccharides extruded by BSCs and the anchoring structures of mosses and lichens bind soil particles together and form soil aggregates (Fig. 3). Soil aggregates enhance soil nutrient transformation rates, soil aeration, and gas/water movement. In addition, BSCs greatly increase the resistance of soils to wind and water erosion.^[3,4]

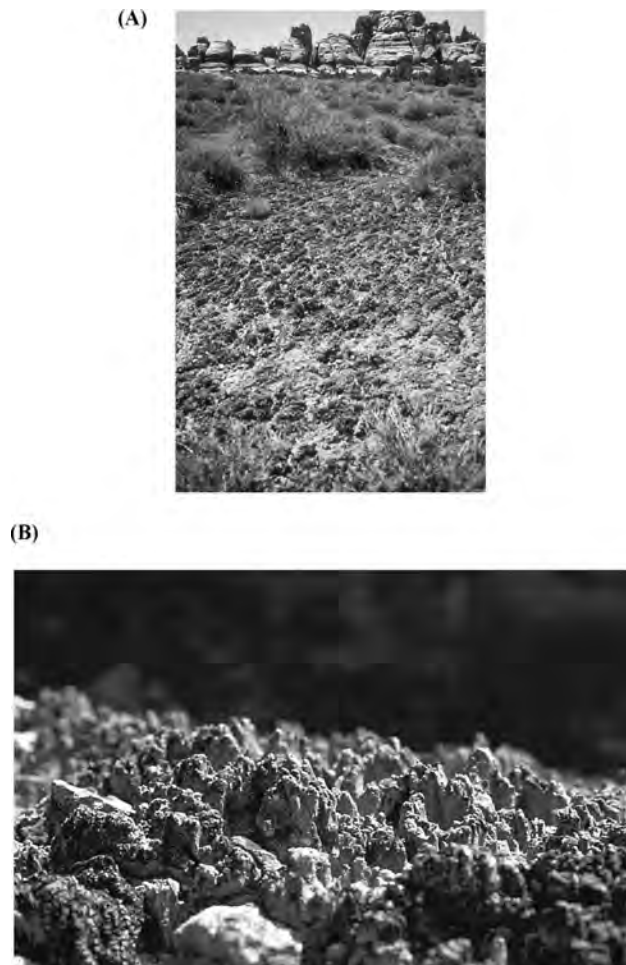


Fig. 1 (A) Soil crusts cover the large interspaces between vascular plants. (B) Close-up view shows crust mounds that greatly roughen soil surfaces, enhancing retention of water, organic matter, dust, and seeds.

SOIL–WATER RELATIONS

In smooth and rugose crusts, BSC organisms reduce surface porosity while not substantially enhancing surface roughness; thus, they reduce water infiltration. In contrast, pinnated and rolling BSCs greatly roughen the soil surface and thus increase water infiltration. On heavy clay soils, BSCs have little effect on infiltration. Because of these differences, data on the effect of BSCs on soil water relations are conflicting, and more work is needed.^[5]

ALBEDO

Biologically crusted surfaces decrease surface energy flux by up to $\sim 40 \text{ J m}^{-2} \text{ s}^{-1}$, increasing dry surface temperatures by 10–14°C.^[1] Surface temperatures regulate many ecosystem functions, including rates of nitrogen and carbon fixation, microbial activity, plant nutrient uptake and growth, evaporation, and seed germination. The timing of these events is often critical for desert plants, and relatively small alterations can affect community structure. In addition, many animals depend on temperature to regulate their foraging times and burrowing depths.

SOIL FERTILITY AND VASCULAR PLANTS

The increased surface roughness and sticky polysaccharide sheaths exuded by BSCs increase the capture of dust, enriching soil nitrogen, phosphorus, potassium, and (depending on the original soil texture) water-holding capacity.^[6] In addition, the negatively charged polysaccharides reduce leaching by binding nutrients, especially cations. Biological crusts secrete chelating compounds,

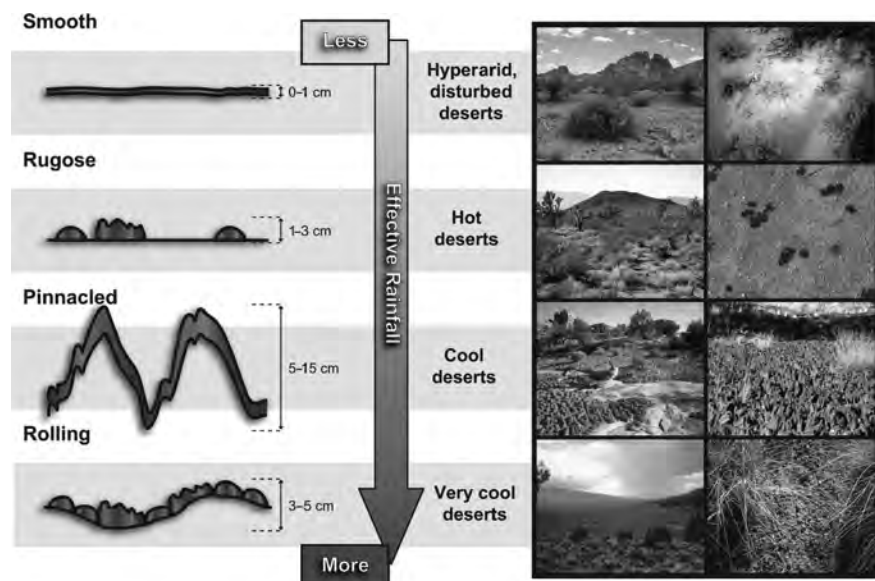


Fig. 2 The four groupings for the external morphology of BSCs.

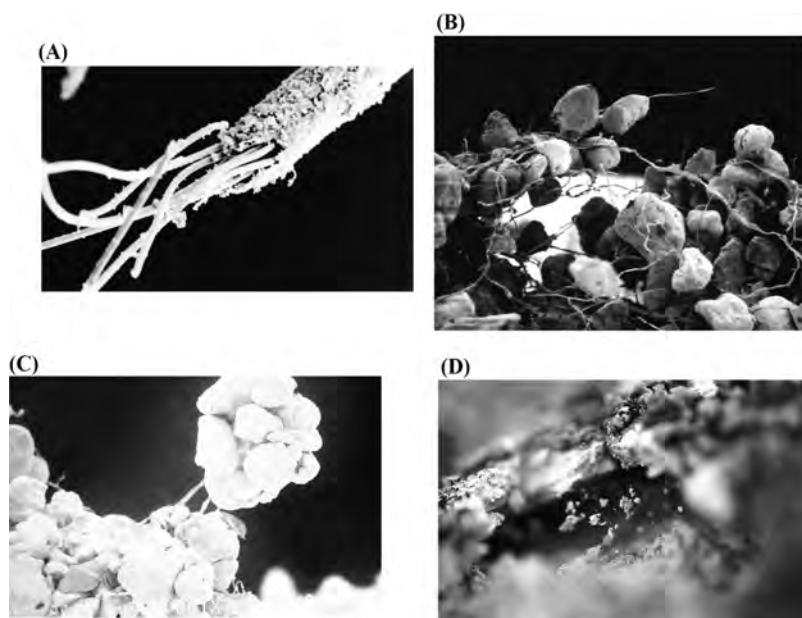


Fig. 3 Scanning electron micrographic images of BSCs: (A) *Microcoleus vaginatus* in desert soils, $\times 700$. (B) *M. vaginatus* sheaths, winding through sand grains, $\times 90$. (C) Cyanobacterial filaments increase soil aggregate structure. (D) Close-up view of a soil surface, sliced vertically. Note the spider web-like cyanobacterial fibers holding the soil aggregates in place.

which are important for keeping metals bioavailable in high pH soils. BSCs also contribute nitrogen ($1\text{--}10\text{ kg yr}^{-1}\text{ ha}^{-1}$) and carbon ($1\text{--}37\text{ g m}^{-2}\text{ yr}^{-1}$) to soils, much of which is released from BSCs upon wetting and utilized by nearby plants, fungi, actinomycetes, and bacteria. Soil fauna are more diverse and abundant in BSC-covered soils.^[1]

Smooth and rugose BSCs generally increase movement of seeds, water, and organic matter from plant interspaces to nearby obstacles (e.g., rocks and plants), whereas pinnated and rolling crusts can increase the retention of materials in the interspaces between plants. Most studies show BSCs enhance or do not affect native seed germination, whereas they retard germination of at least some exotic annual grasses in many ecosystems. As BSCs cannot compete for light with vascular plants, BSC cover is reduced with greater plant cover.^[1]

THE RESISTANCE AND RESILIENCE OF BSCS

Many existing and future conditions threaten BSCs, including soil surface disturbance, increased temperatures, exotic annual grass invasions, and increased fire.^[1] Although BSCs are highly tolerant of stresses with which they evolved (e.g., radiation and drought), they have little resistance to the compressional forces generated by livestock or human activities. These forces essentially pulverize the BSCs, with lichens and mosses being the most susceptible to loss. Annual grasses occupy the plant interspaces once covered by BSCs, reducing BSC cover. Rodents and increased fire that accompany annual grasses also lead to a reduction in BSC cover. As millions of hectares of rangelands are dominated by exotic annual grasses, we stand to lose large amounts of well-developed lichen–moss BSCs throughout deserts of the world.

All BSCs are metabolically active only when wet, and their rates of physiological functioning are highly responsive to temperature. Existing climate models predict higher temperatures and altered precipitation patterns, which will affect the structure and function of BSCs. Increased summer rain will very likely increase carbon deficits in BSCs, as they will dry too quickly to replace respired carbon. Less winter rain will reduce carbon gain. Less carbon gain translates into less nitrogen fixation, as this process requires photosynthetic products, and, together, these decreases will reduce soil fertility. With reduced access to adequate carbon and nitrogen, crusts will be less able to avoid or repair radiation damage, increasing mortality and most likely altering the existing species distribution patterns.

Recovery rates depend on the availability of inoculant (natural or cultured, in the case of cyanobacteria) and on the characteristics of the site, including soils, climate, and disturbance. In coarse soils with low stability, fertility, and water-holding capacity, BSCs recover more slowly than when on fine-textured soils. On stable soils (e.g., low slope, low wind deposition of sand, and embedded rocks), BSCs recover faster than on less stable sites (e.g., steep slopes, high sand deposition, and rolling rocks). Because BSC organisms are metabolically active only when wet, microhabitats (e.g., shrub canopies) and regions with lower PET recover more quickly than drier places. When disturbance is severe or frequent enough to remove crust material, recovery is slower than if organisms are crushed but left in place.

Cyanobacteria recolonize most rapidly. The larger, filamentous cyanobacteria (e.g., *Microcoleus vaginatus*) appear first. Once they stabilize soils, the less mobile and smaller cyanobacteria and green algae appear. In regions where PET is low enough, early successional lichen and moss species appear next (e.g., the cyanolichen *Collema*).

As PET decreases further, mid- and late successional desert lichens, most with green algal phycobionts, colonize next. Recolonization of these later successional species can be very slow, especially if disturbed areas are large. In regions with low PET, recovery can be fairly fast (20 years), while recovery in high PET regions can be extremely slow (1000+ years). Intact crust material salvaged from one area and reapplied to another has long been used to speed recovery in small areas. Growth of cyanobacterial inoculant and application in the field have been attempted for some time. While there are several successful methods for growing the cyanobacteria, past field applications have not been overly successful.^[7] However, techniques developed in China show great promise.^[8]

CONCLUSION

Soil fertility and stability are heavily influenced by BSCs, especially in dryland environments. Unfortunately, many human activities are incompatible with their well-being. The cyanobacteria fibers that confer such tensile strength to soils cannot withstand the compressional stresses generated by animals or vehicles. Exotic annual grasses and increased fire often follow surface disturbance, further threatening BSCs. Compromised crusts contribute less to soil fertility and are less able to offer protection from wind or water erosion. Unlike vascular plant cover, biological crust cover is not reduced in drought, and unlike physical soil crusts, they are present under all soil moisture conditions. Consequently, biological crusts, over time and under adverse conditions, offer many services to ecosystems that are often low in resources. For more information, see www.soilcrust.org.

REFERENCES

1. Belnap, J.; Lange, O.L., Eds. *Biological Soil Crusts: Structure, Function, and Management*, 2nd Ed.; Ecological Studies Series 150; Springer-Verlag: Berlin, 2003.
2. Schwartzman, D.W.; Volk, T. Biotic enhancement of weathering and the habitability of Earth. *Nature* **1989**, *340* (6233), 457–460.
3. Belnap, J.; Gardner, J.S. Soil microstructure in soils of the Colorado Plateau: The role of the cyanobacterium *Microcoleus vaginatus*. *Great Basin Nat.* **1993**, *53* (1), 40–47.
4. Marticorena, B.; Bergametti, G.; Gillette, D.; Belnap, J. Factors controlling threshold friction velocity in semiarid and arid areas of the United States. *J. Geophys. Res.* **1997**, *102* (D19), 23277–23287.
5. Warren, S.D. Biological soil crusts and hydrology cycles in North American deserts. In *Biological Soil Crusts: Structure, Function, and Management*, 2nd Ed.; Belnap, J., Lange, O., Eds.; Ecological Studies Series 150; Springer-Verlag: Berlin, 2003; 327–337.
6. Reynolds, R.; Belnap, J.; Reheis, M.; Lamothe, P.; Luiszer, F. Aeolian dust in Colorado Plateau soils: Nutrient inputs and recent change in source. *Proc. Natl. Acad. Sci.* **2001**, *98* (13), 7123–7127.
7. Buttars, S.M.; St. Clair, L.L.; Johansen, J.R.; Sray, J.C.; Payne, M.C.; Webb, B.L.; Terry, R.E.; Pendleton, B.K.; Warren, S.D. Pelletized cyanobacterial soil amendments: Laboratory testing for survival, escapability, and nitrogen fixation. *Arid. Soil Res. Rehab.* **1998**, *12* (2), 165–178.
8. Li, X.-R.; Wang, X.-P.; Li, T.; Zhang, J.-G. Microbiotic soil crust and its effect on vegetation and habitat on artificially stabilized desert dunes in Tengger Desert, North China. *Biol. Fert. Soils* **2002**, *35* (3), 147–154.

Biological Nitrogen Fixation

Robert H. Burris

Department of Biochemistry, College of Agriculture and Life Sciences, University of Wisconsin, Madison, Wisconsin, U.S.A.

Abstract

The symbiotic process of nitrogen (N_2) fixation by leguminous plants plus associated root nodule bacteria (*Rhizobium* species) is of the greatest practical importance to agriculture, but there are also a number of free-living bacteria capable of N_2 fixation. There are well over 10,000 legume plant species that fix N_2 and about 200 nonlegumes.

INTRODUCTION

For centuries farmers practiced mixed cropping of leguminous and nonleguminous plants without knowing the basis of the empirically observed benefit from the practice. The first clear recognition that leguminous plants could utilize nitrogen (N_2) from the air is attributed to Boussingault's report in 1838.^[1] Skepticism remained until Hellriegel and Wilfarth in Germany in 1886 reported definitive experiments demonstrating that pea plants (*Pisum sativum*) in association with bacteria borne in their root nodules were capable of utilizing N_2 .^[1]

SYMBIOTIC AND FREE-LIVING N_2 FIXERS

The symbiotic process of N_2 fixation by leguminous plants plus associated root nodule bacteria (*Rhizobium* species) is of greatest practical importance to agriculture, but there are also a number of free-living bacteria capable of N_2 fixation. The first of these recognized was the anaerobic organism *Clostridium pasteurianum* as reported by Winogradsky in 1893.^[2] In 1901, Beijerinck recorded that the aerobic bacterium *Azotobacter chroococcum* also fixed N_2 .^[2] These early observations were followed by practical applications, and the practice of inoculation of leguminous seeds with cultures of the rhizobia became widespread. The bacteria were grown commercially and farmers applied these bacteria to leguminous seeds at the time of planting. There is a specificity between the plants and bacteria, and this gave rise to the recognition of cross-inoculation groups, i.e., groups of plants all of which are nodulated by the same species of rhizobia. For example, *Rhizobium leguminosarum* infects garden peas, sweet peas (*Lathyrus odoratus*), common vetch (*Vicia faba*), hairy vetch (*Vicia villosa*), and lentil (*Lens culinaris*), whereas *Rhizobium japonicum* is rather specific for soybeans (*Glycine max*). Not all strains of root nodule bacteria that infect a plant are equally

effective in N_2 fixation, and this is referred to as strain variation. Because of strain variation, commercial inoculants often are a mixture of effective strains. Inoculants are grown as large batches of effective organisms in liquid culture, and then, these cultures are mixed on a peat base. The finely ground peat retains moisture, and the microorganisms remain viable for long periods. The peat culture is mixed with seeds before planting, and the viable organisms are in proximity when the seeds germinate and produce roots. The bacteria induce curling of root hairs and invade the plant through the root hairs. After invasion, they induce the production of root nodules and proliferate in the nodules. The bacteria undergo morphological and metabolic changes in the nodule and are referred to as bacteroids. Bacteroids in the nodule are capable of fixing N_2 from the air.

Claims of fixation of N_2 by cultures of the rhizobia outside the host plant in general could not be verified. In 1975, three laboratory groups reported success in achieving such fixation, by furnishing succinic acid as the substrate for growth and by culturing the organisms under a very low pressure of oxygen.^[3]

The rhizobia function at a low oxygen concentration in the nodule. The nitrogenase enzyme system is labile to oxygen, but there must be sufficient oxygen to support the generation of adenosine triphosphate (ATP). Hemoglobin in nodules achieves this necessary balance in the oxygen level, keeping it adequate to support the action of cytochrome oxidase that generates ATP.

There are well over 10,000 legume plant species that fix N_2 and about 200 nonlegumes. Only one of these nonlegumes has been reported to be nodulated by a *Rhizobium* species such as nodulate the legumes: this is *Parasponia andersonii*, a tree of the elm group. The other nonleguminous plants, such as the alder (e.g., *Alnus glutinosa*), fix N_2 in association with actinomycetes. It proved extremely difficult to culture these actinomycetes, but since methodology to isolate and culture these organisms was

developed in 1977,^[3] information on them has expanded substantially. Some of the actinorhizal associations have found extensive use, particularly in tropical and low-rainfall areas.

BIOCHEMISTRY OF N₂ FIXATION

Early studies on biological N₂ fixation emphasized the practical applications of the process and the development of effective inoculants. Since 1925, there has been interest in the biochemistry of the N₂ fixation process. Wilson^[1] utilized nodulated red clover plants (*Trifolium pratense*) to demonstrate that half-maximum fixation occurred at a pN₂ (partial pressure of N₂) of about 0.05 atmosphere (approximately 50 millibar). High levels of O₂ inhibited the fixation non-competitively. While growing clover plants under controlled atmospheres, Wilson^[1] made the unexpected observation that fixation was inhibited by hydrogen (H₂) gas. H₂ is a specific and competitive inhibitor of N₂ fixation. The H₂ inhibition of N₂ fixation results from an ordered, sequential process in which the H₂ must bind to the nitrogenase enzyme before the N₂.

There was speculation about the first or key product of N₂ fixation. Winogradsky^[2] suggested that ammonia (NH₃) was the first product. Virtanen,^[2] on the other hand, supported hydroxylamine as the key intermediate based on his claim that leguminous plants excreted aspartic acid and that the hydroxylamine formed by fixation reacted with oxaloacetic acid to form an oxime that then was converted to aspartic acid. Other investigators failed to confirm Virtanen's work. The use of the stable isotope ¹⁵N as a tracer resolved the argument. ¹⁵N-enriched N₂ was supplied to a culture of *Azotobacter vinelandii* fixing N₂ vigorously, and among the amino acids recovered from the hydrolysate of the cells, the highest enrichment in ¹⁵N among the isolated amino acids was in glutamic acid^[2] rather than in aspartic acid, which would have been supportive of Virtanen's hydroxylamine hypothesis. If NH₃ were the key product of fixation, it could logically be incorporated into glutamic acid by the action of known enzymes. Similar experiments were performed with species of bacteria from the genera *Clostridium*, *Chromatium*, *Chlorobium*, and *Rhodospirillum*, as well as the blue-green alga *Nostoc muscorum* and excised soybean nodules. In all cases, the highest concentration of ¹⁵N among the isolated amino acids was in glutamic acid.

CELL-FREE N₂ FIXATION

As the metabolism of nitrogenous compounds in intact microorganisms can be rather complex, it was important to verify the path of fixation in cell-free preparations where reactions can be defined more specifically. ¹⁵N as a tracer furnished a sensitive tool for seeking reliable cell-free fixation. Carnahan and coworkers in 1960^[3] developed a

method for recovering active preparations consistently from *C. pasteurianum*. Such cell-free preparations supplied ¹⁵N-enriched N₂-yielded NH₃ with over 50 atom% ¹⁵N excess, a very convincing verification that NH₃ is the "key" intermediate in biological N₂ fixation.

PURIFICATION OF NITROGENASE

Mortenson^[3] demonstrated that the cell-free enzyme system consisted of two proteins. When the two proteins were purified, it turned out that one component was a molybdenum-iron (MoFe) protein, and the other was an iron protein. In purification on a chromatographic column, the MoFe protein was eluted before the Fe protein, so the MoFe protein was often referred to as protein 1 or component 1 and the Fe protein as protein 2 or component 2. A number of other terminologies were used but abandoned. Descriptive terms are dinitrogenase for the MoFe protein, dinitrogenase reductase for the Fe protein, and nitrogenase for the complex of the two components. The primary function (it also has other functions) of the Fe protein is to transfer electrons to the MoFe protein and hence the designation dinitrogenase reductase.

REQUIREMENT FOR ATP

In 1960, it was reported that ATP was inhibitory to N₂ fixation in cell-free preparations of nitrogenase, but in 1962 McNary^[3] demonstrated that ATP is an absolute requirement for fixation by such preparations. The N-N bond is stable, so N₂ fixation is energy demanding whether it is accomplished chemically at high temperature and pressure or whether it is accomplished by a biological system. A minimum of 16 ATPs are required for the reduction of one N₂ to 2NH₃. Sixteen ATPs are required under ideal conditions, and the requirement is 20 to 30 ATPs under the most conditions in nature.

SUBSTRATES OTHER THAN N₂

Nitrogenase is a versatile enzyme and is capable of reducing substrates other than N₂.^[3] It was recognized early that H₂ is a by-product of the nitrogenase reaction. The next substrate to be recognized was nitrous oxide that is reduced to N₂. Cyanide, methyl isocyanide, and azide can also be reduced by the enzyme system. Nitrogenase can reduce acetylene to ethylene. Other substrates demonstrated include diazine, diazirine, cyanamide, and cyclopropene. Acetylene reduction to ethylene has become a widely used measure of nitrogenase activity. The substrate is easy to prepare, the ethylene formed as a product can be measured readily and accurately by gas chromatography, and the simple method has been applied to studies in the glasshouse and field.

CONTROL OF NITROGENASE

As N_2 fixation is energy demanding, it is advantageous for the fixing organisms to turn off the system if they have access to a source of fixed N_2 . Ludden^[3] found that *Rhodospirillum rubrum* turned its nitrogenase off when supplied fixed N_2 such as NH_3 or when it was placed in the dark and thus was deprived of light energy. When NH_3 was exhausted or when light was restored, it turned nitrogenase back on. Inactivation is achieved by ADP ribosylation of arginine 100 of one of the two equivalent subunits of dinitrogenase reductase, and reactivation is achieved by the removal of this ADP-ribosyl group.

TERTIARY STRUCTURE OF NITROGENASE

The tertiary structure of the nitrogenase components was established in 1992.^[2] This aids one in visualizing how the dinitrogenase and dinitrogenase reductase fit together and how the active Fe and MoFe sites are aligned.

ASSOCIATIVE N_2 FIXATION

The leguminous plants and their associated rhizobia are of great practical importance in agriculture, but the non-leguminous actinorhizal systems often fix N_2 vigorously and occupy important niches in ecosystems. Certain non-legumes can benefit from association with bacteria that

occupy their root surface or multiply within the plant without forming nodules.^[3,4] In Brazil, it has been demonstrated that sugarcane can derive N_2 from fixation by *Acetobacter diazotrophicus* carried inside the cane. *A. diazotrophicus* is a remarkable bacterium, as it can grow in a 25% sucrose solution and fix N_2 at a pH as low as 3.0.

GENETICS OF THE NITROGENASE SYSTEMS

The genetics of biological N_2 -fixing systems has occupied many investigators, as genetic manipulations may offer ways to improve the process or adapt it to new crops. Progress in our knowledge of the genetics of N_2 -fixing organisms has led to a substantial reorganization of classification of these organisms.

REFERENCES

1. Wilson, P.W. *The Biochemistry of Symbiotic Nitrogen Fixation*; The University of Wisconsin Press: Madison, 1940; 302 pp.
2. Triplett, E.W., Ed. *Prokaryotic Nitrogen Fixation*; Horizon Scientific Press: Wymondham, 2000; 800 pp.
3. Stacey, G.; Burris, R.H.; Evans, H.J., Eds. *Biological Nitrogen Fixation*; Chapman and Hall: New York, 1992; 943.
4. Postgate, J.R. *The Fundamentals of Nitrogen Fixation*; Cambridge University Press: Cambridge, 1982; 252 pp.

Biological Nitrogen Fixation: Contributions to Agriculture

Mark B. Peoples

Plant Industry, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia

Abstract

Symbiotic associations between legumes and rhizobia are responsible for the greatest contributions of biological nitrogen fixation (BNF) in agricultural systems. Data collected from pulses, legume oilseeds, and pastures growing in farmers' fields generally indicate the levels of BNF much lower than the potential values observed under experimental conditions. Strategies are available to improve BNF beyond what is being achieved. For example, enormous benefits in terms of crop production and dinitrogen fixed can be derived from the application of good agronomic principles. But the ability to overcome constraints at the farm level may be limited because either the relevant technologies are not in the hands of the farmers, or they cannot readily adopt them because of lack of knowledge and information, economic constraints, or operational imperatives. There are a number of environmental advantages in relying on BNF over fertilizer nitrogen to produce such large quantities of high-quality protein.

INTRODUCTION

Although dinitrogen (N_2) gas represents almost 80% of the earth's atmosphere, it is not a source of nitrogen (N) that is readily available to plants. However, a number of prokaryotic microorganisms have evolved that utilize the enzyme nitrogenase to reduce atmospheric N_2 to ammonia, which can subsequently be used to support their growth. Biological N fixation (BNF) by some diazotrophs can occur in a free-living state, or via associative relationships with plants, while others can fix N_2 only in symbiosis with specific plant hosts.^[1] Although calculations of global contributions of BNF are subject to enormous approximations, annual inputs of fixed N into arable agroecosystems have been conservatively estimated to be 35–55 million metric tonnes of N, with a further 40–45 million tonnes of N per year occurring in permanent pastures.^[2] This compares to 75–80 million tonnes of N applied to crops and grasslands as fertilizer each year.^[2]

SOURCES OF FIXED N IN AGRICULTURAL SYSTEMS

The N_2 fixation process can directly contribute to agricultural production where the fixed N is harvested in grain or other food for human or animal consumption. However, BNF can also represent an important renewable source of N that can help maintain or enhance the fertility of many agricultural soils. Examples of experimental estimates of amounts of N fixed by various N_2 -fixing organisms are presented in Table 1. Values for associative and symbiotic

systems have always been determined from measures of plant shoot biomass. Belowground contributions of fixed N have generally been ignored. However, research suggests that N associated with roots might represent between 25% and 60% of the total N accumulated by crops and pastures.^[3–5] Therefore, total inputs of fixed N could be 50–100% greater than has been traditionally determined from shoot-based measurements, such as those depicted in Table 1.

Contributions by Different N_2 -Fixing Organisms

Free-living N_2 fixers probably contribute only small amounts of N to agricultural systems (Table 1). The data tend to be inconclusive concerning the role of diazotrophs associated with non legumes in temperate agriculture, but studies have demonstrated significant inputs of fixed N by tropical grasses and crops such as sugarcane (*Saccharum officinarum*).^[1,6] Symbiotic associations between legumes and specific soil bacteria (*Rhizobium*, *Bradyrhizobium*, *Allorhizobium*, *Azorhizobium*, *Mesorhizobium*, or *Sinorhizobium* spp.) in specialized root structures (nodules) are generally responsible for the largest amounts of fixed N in farming systems (Table 1).

Inputs of Fixed N by Legumes

BNF by legume systems that play a key role in world crop production is irrefutable. The ability of legumes to progressively improve the N status of soils has been utilized for thousands of years in crop rotations and traditional farming systems.^[1,2,7] The 163 million hectares of legume oilseeds (soybean—*Glycine max*; and

Table 1 Experimental estimates of the amounts of N₂ fixed in different agricultural systems.

N ₂ -fixing organism	Range measured (kg N/ha per crop or per year)	Range commonly observed (kg N/ha per crop or per year)
Free-living		
Crops	0–80	0–15
Associative		
Tropical grasses	10–45	10–20
Crops	0–240	25–65
Symbiotic		
Azolla	10–150	10–50
Green manure legumes	5–325	50–150
Pasture/forage legumes	1–680	50–250
Crop legumes	0–450	30–150
Trees/shrubs	5–470	100–200

groundnut—*Arachis hypogea*) and pulses sown globally each year, legume components of the 200 million hectares under temporary pastures or fodder crops, and the 10–12 million hectares of perennial legume cover crops in rubber (*Hevea brasiliensis*) and oil palm (*Elaeis guineensis*) plantations contribute fixed N to farming systems. Most modern methods used to quantify inputs of fixed N by legumes separate the plant N into fractions originating from soil N or N₂ fixation.^[1,5,8] Once the legume N can be partitioned into that proportion derived from atmospheric N₂ (%Nd_{fa}, sometimes also described as %P_{fix}) and that coming from the soil, the amounts of N₂ fixed can be calculated from measures of shoot dry matter (DM) and N content. The formation of the symbiosis between legume and rhizobia is dependent on many factors and cannot be assumed to occur as a matter of course. This is reflected in the range of values presented in Table 1. Such large variations in reported estimates of N₂ fixation make it difficult to generalize about how much N may be fixed by different legume species. Collectively, the data suggest maximum rates of N₂ fixation of 3–4 kg shoot N/ha/day^[5] and potential inputs of fixed N by many legumes of several hundred kilograms of shoot N/ha each year (Table 1). However, much of the information in Table 1 was derived from research trials in which specific treatments were imposed to generate differences in %Nd_{fa} values and legume growth as an experimental means of studying factors which regulate BNF. Therefore, these data may be of little relevance to what might actually be occurring in farmers' crops and pastures. Fortunately, measurement procedures have been developed which allow on-farm measures of legume N₂ fixation to be conducted with confidence.^[5,7,8]

Levels of N Fixation Achieved in Farmers' Fields

Examples of the types of information which can be generated about BNF in farmers' fields are presented in the tables for farming systems in different regions of the world. These on-farm data and observations can be used to develop a picture of N₂ fixation within an individual country or region and provide insights into contributions of BNF to agriculture on a global scale. Collectively, the results in Table 2 indicate that the potential for BNF inputs can differ between legumes and countries, but they also suggest many commonalities. Although wide ranges in %Nd_{fa} values have been observed, it seems that, on average, most winter pulses (e.g., chickpea—*Cicer arietinum*; lentil—*Lens culinaris*; field pea—*Pisum sativum*; faba bean—*Vicia faba*; and lupin—*Lupinus albus*) relatively satisfy higher proportions of their growth requirements from N₂ fixation (>65%) than do the summer legumes (e.g., mung bean—*Vigna radiata*; mash bean—*Vigna mungo*; soybean; and groundnut) where %Nd_{fa} values were commonly less than 60% (Table 2). Poor or variable nodulation observed in some summer legumes and the resulting increased reliance upon soil N may reflect greater N mineralization during summer, water stress, and/or low vegetative biomass accumulated by short duration legume crops.^[8,9]

Table 2 Summary of the proportion of plant N derived from N₂ fixation (%Nd_{fa}) and the amounts of N₂ fixed by farmers' legume crops and pastures in different geographical regions.

Country and legume	Number of fields	Mean Nd _{fa} (%)	Total N fixed (kg N/ha) ^a
Winter pulse crops			
Pakistan	126	78	79
Nepal	27	79	78
Syria	46	67	na ^b
Australia	90	65	170
Summer legume crops			
Pakistan	63	47	42
Nepal	50	55	77
Thailand	13	75	78
Vietnam	45	48	125
South Africa	14	58	na ^b
Australia	33	53	267
Annual pastures			
Australia	300	75	na ^b
Perennial pastures			
Australia	110	64	na ^b

Note: Nd_{fa}, nitrogen derived from atmospheric N₂.

^aIncludes an estimate of fixed N from the roots and nodules which assumes that belowground N represents 33% of total plant N.

^bData not available for all fields.

Table 3 Key factors influencing inputs of fixed N by legumes in farmers' fields.

Country	System	BNF regulated by		
		DM	Soil nitrate	Primary factors
Pakistan	Winter crop	+++		Rainfall nutrition, weed control
	Summer crop	+++	+++	Fertilizer N, no inoculation, insects, disease
Nepal	Winter crop	+++		Rainfall, nutrition
	Summer crop	+++	+	Total soil N, mineralized N, available P, legume species
Syria	Winter crop	+++	++	Soil nutrients, insects, disease
Thailand	Summer crop	+++		Available P
Vietnam	Summer crop	+++		Plant density, soil pH, available P, legume species
South Africa	Summer crop	++	++	Effective inoculation, nutrition, rotation, water availability
Australia	Winter crop	+++	+++	Rainfall, fallowing, legume species
	Summer crop	++	+++	Crop rotation, tillage, rainfall
	Pasture	+++	+	Soil pH, available P, legume density, grazing management

Note: BNF, biological N₂ fixation; DM, dry matter; P, phosphorus.

In grazed pastures, the competition for mineral N between legumes and companion grasses or vigorous broadleaf weeds growing within the pasture sward results in low levels of plant-available soil N throughout the growing season.^[4,7] As a consequence, %Nd_fa by the legume components of pastures tends to be high (Table 2). The slightly lower %Nd_fa values detected in perennial legume species, such as alfalfa (*Medicago sativa*) and white clover (*Trifolium repens*), presumably result from a greater ability to scavenge soil mineral N from a larger rooting zone compared with annual pasture species.^[4]

Although the levels of %Nd_fa are important, the amounts of N₂ fixed are usually regulated by legume growth rather than %Nd_fa in most farming systems,^[7] and many legumes appear to fix approximately 20 kg of shoot N for every metric ton of shoot DM accumulated.^[4,7,8]

Impact of Management

Factors that either enhance or depress N₂ fixation (Table 3) can generally be summarized in terms of environmental or management constraints to crop growth (e.g., basic agronomy, nutrition, water supply, diseases, and pests). A number of strategies can be employed that specifically enhance BNF through increased legume biomass. These include the use of legume genotypes adapted to the prevailing edaphic and environmental conditions, procedures to improve legume plant density, irrigation (if available), the amelioration of soil nutrient toxicities or deficiencies, and the control of weeds and pests.^[1,4,10] However, as the formation of an active symbiosis is dependent on the compatibility of both the diazotrophic microorganism and the legume host, local practices that limit the presence of effective rhizobia (no inoculation, poor inoculant quality) will also be crucial in determining the legume's capacity to fix N (Table 3), as will any management decisions that directly affect soil N

fertility (excessive tillage, extended fallows, fertilizer N, and rotations), since mineral N is a potent inhibitor of the N₂ fixation process.^[1,7,9,10]

CONCLUSION

Symbiotic associations between legumes and rhizobia are responsible for the greatest contributions of BNF in agricultural systems. Research trials suggest potential annual inputs of fixed N by most legumes equivalent to several hundreds of kilograms of N per hectare. However, data collected from pulses, legume oilseeds, and pastures growing in farmers' fields generally indicate the levels of BNF much lower than the potential values observed under experimental conditions. So while legumes should routinely be fixing >100 kg/ha each year, in reality, they usually do not. Strategies are available to improve BNF beyond what is being achieved. For example, provided that a legume crop is abundantly nodulated and effectively fixing N₂, enormous benefits in terms of crop production and N₂ fixed can be derived from the application of good agronomic principles (see *Biological Nitrogen Fixation*, p. 227–231). But the ability to overcome constraints at the farm level may be limited because either the relevant technologies are not in the hands of the farmers, or they cannot readily adopt them because of lack of knowledge and information, economic constraints, or operational imperatives. While the global inputs of fixed N by legumes may be considerably less than their genetic potential and is lower than the amounts of N applied as fertilizer each year, some 15 million tonnes of N are harvested annually from legume crops, and many million tonnes more will be consumed by animals in legume-based forage, and there are a number of environmental advantages in relying on BNF over fertilizer N to produce such large quantities of high-quality protein.^[11,12]

REFERENCES

1. Giller, K.E. *Nitrogen Fixation in Tropical Cropping Systems*, 2nd Ed.; CABI Publishing: Wallingford, 2001.
2. Peoples, M.B.; Herridge, D.F.; Ladha, J.K. Biological nitrogen fixation: An efficient source of nitrogen for sustainable agricultural production? *Plant Soil* **1995**, *174*, 3–28.
3. Khan, D.F.; Peoples, M.B.; Chalk, P.M.; Herridge, D.F. Quantifying below-ground nitrogen of legumes. 2. A comparison of ^{15}N and non isotopic methods. *Plant Soil* **2002**, *239*, 273–289.
4. Peoples, M.B.; Baldock, J.A. Nitrogen dynamics of pastures: Nitrogen fixation inputs, the impact of legumes on soil nitrogen fertility, and the contributions of fixed nitrogen to Australian farming systems. *Aust. J. Exp. Agr.* **2001**, *41*, 327–346.
5. Unkovich, M.J.; Pate, J.S. An appraisal of recent field measurements of symbiotic N_2 fixation by annual legumes. *Field Crop. Res.* **2001**, *65*, 211–228.
6. Boddey, R.M.; de Oliveira, O.C.; Urquiaga, S.; Reis, V.M.; de Olivares, F.L.; Baldani, V.L.D.; Döbereiner, J. Biological nitrogen fixation associated with sugar cane and rice: Contributions and prospects for improvement. *Plant Soil* **1995**, *174*, 195–209.
7. Peoples, M.B.; Bowman, A.M.; Gault, R.R.; Herridge, D.F.; McCallum, M.H.; McCormick, K.M.; Norton, R.M.; Rochester, I.J.; Scammell, G.J.; Schwenke, G.D. Factors regulating the contributions of fixed nitrogen by pasture and crop legumes to different farming systems of eastern Australia. *Plant Soil* **2001**, *228*, 29–41.
8. Peoples, M.B.; Herridge, D.F. Quantification of biological nitrogen fixation in agricultural systems. In *Nitrogen Fixation: From Molecules to Crop Productivity*; Pedrosa, F.O., Hungria, M., Yates, M.G., Newton, W.E., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 519–524.
9. Herridge, D.F.; Robertson, M.J.; Cocks, B.; Peoples, M.B.; Holland, J.F.; Heuke, L. Low nodulation and nitrogen fixation of mungbean reduce biomass and grain yields. *Aust. J. Exp. Agr.* **2005**, *45*, 269–277.
10. Peoples, M.B.; Ladha, J.K.; Herridge, D.F. Enhancing legume N_2 fixation through plant and soil management. *Plant Soil* **1995**, *174*, 83–101.
11. Crews, T.E.; Peoples, M.B. Legume versus fertilizer sources of nitrogen: Ecological tradeoffs and human needs. *Agr. Ecosys. Environ.* **2004**, *102*, 279–297.
12. Jensen, E.S.; Hauggaard-Nielsen, H. How can increased use of biological N_2 fixation in agriculture benefit the environment? *Plant Soil* **2003**, *252*, 177–186.

Biological Nitrogen Fixation: Enhancement Techniques

David Francis Herridge

Environmental and Rural Sciences, University of New England, Armidale,
New South Wales, Australia

Abstract

Legume biological nitrogen fixation (BNF), resulting from the symbiotic relationship between legumes and rhizobia, a soil bacteria, is a key process in agriculture with the crop legumes alone supplying about 24 million tons of nitrogen (N) annually for global grain production. The specific amount of N fixed by a legume crop in a farmer's field essentially depends on the productivity of the crop and the level of soil mineral N in the field. Productivity can be enhanced by optimizing crop use of growing season and space, and nutrient and water resources, with careful attention to species and cultivar choice, planting time, row spacing and plant density, pest, disease and weed control, and crop nutrition. The suppressive effects of soil mineral N on BNF can be reduced through practices that lead to reductions in soil mineral N, such as no-tillage. Inoculation of legumes at sowing is another practice that can enhance BNF. It is virtually impossible to obtain accurate figures, but it is likely that about 50 million hectares of crop legumes, equivalent to about 25% of the area sown globally, are inoculated each year.

INTRODUCTION

Legume biological nitrogen fixation (BNF) is a key process in agriculture with the crop legumes alone supplying about 24 million tons of nitrogen (N) annually for the world's grain production. Soybean is the most widely grown of the crop legumes with 111 million hectares harvested in 2013 and almost half the area in just two countries—Brazil and Argentina. Soybean makes a substantial contribution to global N fixation with an estimated 17.4 million tonnes fixed annually. Notwithstanding these impressive statistics, legume BNF, resulting from the symbiotic association between legumes and rhizobia, a soil bacteria, remains essentially unmanaged by farmers. In many parts of the world, the legumes are considered secondary crops and are not provided the inputs and management given to major crops, usually cereals. In other situations, economic factors mitigate against the use of appropriate inputs. This lack of management, coupled with environmental influences beyond the control of farmers, results in wide variations in BNF, commonly between nil and approximately 400 kg N/ha.

FACTORS AFFECTING BNF

The factors accounting for variations in BNF are quite simple. Provided that there are adequate numbers of highly effective rhizobia in the soil in which the legume is growing, BNF is essentially determined by the growth of the legume (i.e., the legume's N demand) and the proportion of legume N derived from BNF (%Nd_{fa}), as follows:

$$\text{Legume BNF (kg/ha)} = \frac{\text{Total N yield (kg/ha)}}{100} \times \% \text{Nd}_{\text{fa}}$$

Thus, legume BNF can be enhanced by increasing total N yield and/or increasing the %Nd_{fa}. With the majority of legumes, there is sufficient genetically based capacity for BNF to supply the entire amount of N that is required for plant growth.^[1,2] For example, the data of Evans et al.^[3] highlighted the very strong relationship between shoot dry matter and N₂ fixed for the temperate crop legumes, lupin (*Lupinus angustifolius*) and pea (*Pisum sativum*), and indicated sufficient BNF capacity of lupin to support shoot dry matters of up to 14 t/ha.

There are cases, however, where insufficient BNF capacity limits legume yield, the most notable being common or dry bean (*Phaseolus vulgaris*). Redden and Herridge,^[4] after reviewing the genetic capacity for BNF of >1000 genotypes of *P. vulgaris*, concluded that BNF could supply only a fraction (<30%) of the N demand with the remainder supplied by soil or fertilizers. Clearly, BNF of this species is weak, and management for yield will only have limited benefits for BNF.

LEGUME YIELD AND BNF

Just how much influence does legume yield have on BNF? The simple answer is a great deal. Researchers in Australia's northern grain belt^[5] reported that 96% of the variation in BNF of alfalfa (*Medicago sativa*) was related to the variation in legume aboveground biomass with 28 kg shoot N fixed for every ton of biomass produced. A study of BNF

of pulse (pea) and pasture species [alfalfa and annual medic (*Medicago truncatula*)] showed that 99% of the variation in BNF was associated with differences in biomass production (23 kg shoot N fixed/t biomass).

Similar close relationships between biomass and BNF were reported by Evans et al.^[3] in a network of 21 experiments at 10 locations of the cool-season pulses, lupin and pea, in southeastern Australia. They reported variations in BNF of 36–400 kg N/ha and 22–250 kg N/ha for the two species. The original data were presented as fixed N in shoots only; whole-plant BNF was calculated by multiplying shoot N by 1.4 to account for N in below-ground parts.^[6] Values for %Ndfa varied from 29% to 97% for lupin and 20% to 95% for pea. Almost 70% of the variation in BNF was associated with variation in legume biomass yield; an additional 20 kg shoot N/ha was fixed for every extra ton of biomass produced. Relationships between legume biomass production and BNF were summarized by Unkovich et al.^[6] as 17 kg shoot N fixed/t biomass for crop and pasture legumes combined, 21 kg shoot N fixed/t biomass for crop legumes alone, and 16 kg shoot N fixed/t biomass for pasture legumes alone.

Soil Water

In the rainfed environments where the majority of agricultural legumes are grown, the overriding influence on plant growth (biomass yield) is the availability of water, which is either stored in the soil during the precrop fallow or falling as rain during growth. With the lupin and pea study referred to previously, 83% of variation in lupin biomass could be explained by in-crop rainfall.^[3] In an ancillary study, additional inputs of 75 and 125 mm water in the form of supplementary irrigation resulted in the enhancement of BNF at rates of 0.4–0.8 kg N/mm water. With the extra 125 mm water, BNF increased from 60 to 210 kg N/ha for lupin and from 47 to 150 kg N/ha for pea. The water use efficiency values for lupin and pea BNF were similar to published values of 0.3–0.5 kg N/mm for chickpea (*Cicer arietinum*).

These data highlight the close relationship between water availability, biomass production, and BNF. Irrigation is not often an option for farmers in dryland agriculture, and they have no control over seasonal weather. What they have some control over is the efficiency with which water is infiltrated into and stored in the soil, coupled with the efficiency with which the water is used by the legume.

Rhizobial Inoculation

Legume inoculation with rhizobia is a long established and successful practice, especially in countries in which legumes are widely grown, such as the United States, Australia, Canada, Brazil, Argentina, and India. It is virtually impossible to obtain accurate figures, but it is likely that about 50 million hectares of crop legumes, equivalent to about 25% of the area sown globally, are inoculated each year.

Since 1970, many experiments in different parts of the world have examined the need to inoculate. Results from 214 experiments of 10 legume species conducted by the University of Hawaii's NifTAL project indicated inoculation-enhanced yields in 53% of cases through enhanced BNF.^[7] Soybean (*Glycine max*) responded to inoculation in 65% of experiments, which is not surprising because of its specific requirements for rhizobia. In the same study, analysis of 305 soil samples from 17 countries indicated that only 14% of soils contained rhizobia that were effective on the highly bred American soybean.

These figures highlight the uncertainty regarding the benefits of and need for inoculation. In countries where inoculants are readily available, inoculation is generally recommended as a practice for all legume sowings. Unfortunately, few farmers in less-developed countries have access to high-quality inoculants or use inoculants as a normal part of their legume cultivating practices. In these situations, potential yield is lost unless soil N uptake is supplemented with fertilizer N (with additional cost to the farmer), or the soil already contains adequate numbers of effective rhizobia.

Legume Agronomy

Strategies to enhance legume biomass and legume BNF include optimizing crop use of growing season and space, and nutrient and water resources with careful attention to species and cultivar choice, planting time, row spacing and plant density, pest, disease and weed control, and crop nutrition. In the following sections, experiments are discussed in which legume BNF was enhanced through management for biomass yield. The examples are specific for particular sites and seasons and are not meant to be prescriptive; rather, they serve to illustrate how management practices can be changed to enhance BNF.

Species/Cultivar Choice

Matching the species and cultivar of legume to the edaphic (soil) and environmental conditions can lead to enhanced BNF. In the same lupin and pea study referred to previously, BNF of lupin was approximately 80% greater than that of pea on the acidic soils because of substantially increased biomass yield and marginally improved %Ndfa (Table 1). The reverse was the case on the alkaline soils in the study; on those soils, pea outyielded lupin by approximately 60% and fixed approximately 80% more N.

Planting Time

Optimizing planting time to take full advantage of rainfall and temperatures in the growing season and to minimize deleterious effects of pest and disease cycles can provide options for enhancing BNF. In the Australian pea experiment (Table 1), BNF was increased from 64 to 180 kg N/ha

Table 1 Examples of enhancement of legume BNF through practices to increase legume yield.

Agronomic practice ^a	Crop N ^b (kg/ha)	% Nd _f	Total N fixed (kg/ha)	References
Choice of species				
Acid soil (pH 4.3)				
Lupin	253	91	230	[3]
Pea	166	78	130	
Alkaline soil (pH 7.8)				
Lupin	104	57	53	
Pea	166	57	95	
Time of planting				
Pea experiment (Australia)				
Early (0 days)	238	76	180	[8]
Mid (+26 days)	166	64	106	
Late (+54 days)	120	53	64	
Chickpea experiment (Syria)				
Winter sown	177	81	143	[9]
Spring sown	40	18	7	
Plant density (alfalfa plants/m ²)				
40	548	55	300	[10]
20	442	53	234	
10	346	51	176	
5	246	52	128	
Pest and weed control				
Control of <i>Sitona</i> weevil in lentil				
+Carbofuran	200	75	150	[9]
–Carbofuran	174	71	124	
Control of grassy weeds in clover				
+Herbicide	400	89	356	[10]
–Herbicide	316	92	290	
Nutrient additions (subterranean clover)				
Nil	80	94	75	[11]
+P	136	90	122	
+Lime	88	87	77	
+P + lime	182	90	164	

^aIn most cases, values shown are the means of several sites and/or seasons.

^bOriginal values for shoot N recalculated as crop N using multiplication factors of 2.0 for chickpea, alfalfa, and subterranean clover, and 1.4 for pea, lupin, and lentil.^[6]

by planting 54 days earlier. Most of the increase was because of the larger biomass crops.^[8]

In a more radical departure from traditional agronomic practice, Beck et al.^[9] in Syria increased BNF of chickpea, from 7 to 143 kg N/ha, by planting in winter rather than in spring. This was made possible by the introduction of cold-tolerant, ascochyta blight [*Ascochyta rabei* (Pass.) Labrousse] resistant varieties. The longer growing season

allowed the crop to utilize rainfall more efficiently and produce greater biomass, which resulted in the enhanced BNF.

Plant Density and Row Spacing

Within the limits of site and environment, narrow row spacing and/or high plant density can result in enhanced BNF. In on-farm surveys of 51 chickpea and faba bean (*Vicia faba*) crops in Australia's northern grain belt, Schwenke et al.^[12] reported that crop biomass was greatest in narrow rows and that %Ndfa was positively correlated with plant density. In pasture systems in southeastern Australia,^[10] increasing alfalfa numbers from 5 to 40 plants/m² resulted in producing more than double the amount of crop biomass (from 246 to 548 kg N/ha) and BNF (from 128 to 300 kg N/ha; Table 1).

Pest Control

Pests and diseases will reduce legume biomass. For example, a major constraint to lentil (*Lens culinaris*) production is the *Sitona* weevil, the larvae of which feed on the N₂-fixing root nodules. Destruction of nodules can be as high as 80%. Application of carbofuran (2,3-dihydro-2,2-dimethyl-7-benzofuranol methylcarbamate) during sowing controls the pest. In the Syrian study reported by Beck et al.,^[9] use of the insecticide resulted in a 20% increase in both biomass and BNF (Table 1).

Weeds compete with the legumes for light, water, and nutrients, and their control benefits the production of legume biomass. Herbicide spraying of grassy weeds in subterranean clover (*Trifolium subterraneum*) pastures in southeastern Australia resulted in a 27% increase in biomass and a 23% enhancement of BNF (Table 1). Similar results were reported^[13] for a mixed annual medic–alfalfa pasture in which winter cleaning of grassy weeds increased BNF by 110 kg N/ha.

Other Soil Constraints and Plant Nutrients

Other soil constraints include acidity, salinity, nutrient toxicities, and nutrient deficiencies. Such constraints must be addressed if potential legume biomass production is to be realized. Research has also established that the N₂-fixing legumes may have additional edaphic and nutritional requirements, compared with the non-N₂-fixing plants.^[14] Examples are the higher requirement by N₂-fixing legumes for calcium, boron, iron, and molybdenum.

Soil acidity and phosphorus (P) deficiency are common constraints to legume BNF. In a 3-year study at three sites in southeastern Australia, N yields and BNF of subterranean clover-dominated pastures were increased from 65% to 70% with P fertilizer and from 120% to 130% with a combination of lime and P (Table 1). Lime increased pH and reduced extractable aluminum and manganese, both of which are toxic to legumes and rhizobia at elevated

concentrations. In an ancillary study, also involving subterranean clover at three sites, crop biomass N and BNF were increased by 80 and 70 kg N/ha, respectively, with fertilizer P.

%NDFA AND BNF

At any level of yield, increasing %Nd_{fa} results from practices to reduce soil nitrate suppression of BNF. This can be achieved in a number of ways, most dramatically by land use, and, to a lesser extent, by tillage practice and stubble and mulch management.

Land Use and Crop Sequence

Rainfed chickpea grown immediately after sorghum (*Sorghum bicolor*) fixed 183 kg N/ha, compared with 54 kg N/ha by chickpea in previously fallowed soil (Table 2). The Nd_{fa} values were 84% and 19%, respectively. The differences in BNF were because of the large differences in soil nitrate levels at sowing—17 and 226 kg N/ha—affecting the %Nd_{fa} values.

Soybean BNF was similarly enhanced when grown on land that had previously grown unnodulated soybean, rather than on land used for nodulated soybean cultivation or on fallowed land (Table 2). Effects on BNF were again mediated through soil nitrate levels affecting plant nodulation and %Nd_{fa}. Unnodulated soybean reduced soil nitrate levels more than nodulated soybean because the residues of the former were N deficient and would have immobilized some of the mineral N, as it was released from organic matter. Many other studies relating BNF to crop sequence and land use report similar findings, including on-farm surveys of 33 soybean crops^[18] and 51 chickpea and faba bean crops^[12] in Australia’s eastern grain belt.

Tillage

Cultivation accelerates the mineralization of organic N to plant-available mineral N in soils, with the result that cultivated soils often have an additional 20–25 kg nitrate–N/ha in the root zone, compared with untilled (or no-tilled) soils. For cereals under no-tillage, additional fertilizer N may be required to supplement the reduced soil nitrate N. For legumes, however, the lower nitrate N should result in enhanced BNF. In a series of experiments comparing effects of tillage practice on soybean production, the no-tilled soybean produced more nodules and more biomass, had higher %Nd_{fa}, and fixed more N (320 vs. 246 kg/ha) than the cultivated crops (Table 2).

Although soil nitrate levels were not measured in this experiment, it is highly likely that reduced soil nitrate resulted in the higher %Nd_{fa} value for the untilled soybean. It was interesting that the untilled crops produced 8% more biomass N than the cultivated crops. Untilled soils are often wetter than cultivated soils, which may be crucial in arid to semiarid, rainfed environments. An additional 40 mm plant-available water could mean an extra 25 kg biomass N/ha. Thus, although the extra 74 kg N fixed/ha should be mainly attributed to the reduced nitrate suppression (and enhanced %Nd_{fa}), the wetter untilled soils may have contributed to increased growth.

CONCLUSION

The previous examples show clearly that management options for enhancing BNF are available to farmers. The options roughly fall into two categories—those that increase biomass yield and those that reduce soil nitrate. As stated earlier, the examples are site and species specific and are not meant to be prescriptive. The means by which

Table 2 Examples of enhancement of legume BNF through practices to increase %Nd_{fa}.

Agronomic practice ^a	Soil nitrate (kg/ha; 1.2 m depth)	Nodulation (mg/plant)	Crop N (kg/ha) ^b	Ndfa (%)	Total N fixed (kg/ha)	References
Cropping sequence: Chickpea following						
Sorghum	17	n.d.	218	84	183	[15]
Fallow	226	n.d.	284	19	54	
Cropping sequence: Soybean following						
Unnod soybean	70	306	310	59	183	[16]
Nod soybean	140	298	317	50	159	
Fallow	260	139	290	22	64	
Tillage practice: Soybean grown with						
No-tillage	n.d.	139	363	88	320	[17]
Cultivated	n.d.	86	337	73	246	

n.d., not determined.
^aIn most cases, values shown are the means of several sites and/or seasons.
^bOriginal values for shoot N recalculated as crop N using multiplication factors of 2.0 for chickpea and 1.5 for soybean.

BNF is enhanced for a particular legume crop or pasture will depend on the type of the crop or soil management practices that can be used to increase legume biomass yield and/or reduce soil nitrate levels.

REFERENCES

- Herridge, D.F.; Peoples, M.B.; Boddey, R.M. Global inputs of biological nitrogen fixation in agricultural systems. *Plant Soil* **2008**, *311*, 1–18.
- Peoples, M.B.; Ladha, J.K.; Herridge, D.F. Enhancing legume N₂ fixation through plant and soil management. *Plant Soil* **1995**, *174*, 83–101.
- Evans, J.; O'Connor, G.E.; Turner, G.L.; Coventry, D.R.; Fettel, N.; Mahoney, J.; Armstrong, E.L.; Walscott, D.N. N₂ fixation and its value to soil N increases in lupin, field pea and other legumes in southeastern Australia. *Aust. J. Agric. Res.* **1989**, *40*, 791–805.
- Redden, R.J.; Herridge, D.F. Evaluation of genotypes of navy and culinary bean (*Phaseolus vulgaris* L.) selected for superior growth and nitrogen fixation. *Aust. J. Exp. Agric.* **1999**, *39*, 975–980.
- Hossain, S.A.; Waring, S.A.; Strong, W.M.; Dalal, R.C.; Weston, E.J. Estimates of nitrogen fixations by legumes in alternate cropping systems at Warra, Queensland, using enriched-¹⁵N dilution and natural ¹⁵N abundance techniques. *Aust. J. Agric. Res.* **1995**, *46*, 493–505.
- Unkovich, M.J.; Baldock, J.; Peoples, M.B. Prospects and problems of simple linear models for estimating symbiotic N₂ fixation by crop and pasture legumes. *Plant Soil* **2010**, *329*, 75–89.
- Singleton, P.W.; Bohlool, B.B.; Nakao, P.L. Legume response to rhizobial inoculation in the tropics: Myths and mealities. In *Myths and Science of Soils of the Tropics*; Lal, R., Sanchez, P.A., Eds.; Soil Science Society of America and American Society of Agronomy Special Publication: Madison, 1992; Vol. 29, 135–155.
- O'Connor, G.E.; Evans, J.; Fettel, N.A.; Bamforth, I.; Stuchberry, J.; Heenan, D.P.; Chalk, P.M. Sowing date and varietal effects on the N₂ fixation of field pea and implications for improvement of soil nitrogen. *Aust. J. Agric. Res.* **1993**, *44*, 151–163.
- Beck, D.P.; Wery, J.; Saxena, M.C.; Ayadi, A. Dinitrogen fixation and nitrogen balance in cool-season food legumes. *Agron. J.* **1991**, *83*, 334–341.
- Peoples, M.B.; Gault, R.R.; Scammell, G.J.; Dear, B.S.; Virgona, J.; Sandral, G.A.; Paul, J.; Wolfe, E.C.; Angus, J.F. Effect of pasture management on the contributions of fixed N to the N economy of Ley-farming systems. *Aust. J. Agric. Res.* **1998**, *49*, 459–474.
- Peoples, M.B.; Lilley, D.M.; Burnett, V.F.; Ridley, A.M.; Garden, D.L. Effects of surface application of lime and superphosphate to acid soils on growth and N₂ fixation by pasture clover in mixed pasture swards. *Soil Biol. Biochem.* **1995**, *27*, 663–671.
- Schwenke, G.D.; Peoples, M.B.; Turner, G.L.; Herridge, D.F. Does nitrogen fixation of commercial, dryland chickpea and faba bean crops in north-west New South Wales maintain or enhance soil nitrogen. *Aust. J. Exp. Agr.* **1988**, *38*, 61–70.
- McCallum, M.H.; Peoples, M.B.; Connor, D.J. Contributions of nitrogen by field pea (*Pisum sativum* L.) in a continuous cropping sequence compared with lucerne (*Medicago sativa* L.)-based pasture Ley in the Victorian Wimmera. *Aust. J. Agric. Res.* **2000**, *51*, 13–22.
- O'Hara, G.W.; Boonkerd, N.; Dilworth, M.J. Mineral constraints to nitrogen fixation. *Plant Soil* **1988**, *108*, 93–110.
- Doughton, J.A.; Vallis, I.; Saffigna, P.G. Nitrogen fixation in chickpea: 1. Influence of prior cropping or fallow, nitrogen fertilizer and tillage. *Aust. J. Agric. Res.* **1993**, *44*, 1403–1413.
- Herridge, D.F.; Bergersen, F.J.; Peoples, M.B. Measurement of nitrogen fixation by soybean in the field using the ureide and natural ¹⁵N abundance methods. *Plant Physiol.* **1990**, *93*, 708–716.
- Hughes, R.M.; Herridge, D.F. Effect of tillage on yield, nodulation and N₂ fixation of soybean in far north-coastal New South Wales. *Aust. J. Exp. Agr.* **1989**, *29*, 671–677.
- Peoples, M.B.; Gault, R.R.; Lean, B.; Sykes, J.D.; Brockwell, J. Nitrogen fixation by soybean in commercial irrigated crops of central and southern New South Wales. *Soil Biol. Biochem.* **1995**, *27*, 553–561.

Biological Nitrogen Fixation: Forms and Regulating Factors

Ken E. Giller

Department of Plant Sciences, Wageningen University, Wageningen,
the Netherlands

Paul Mapfumo

Department of Soil Science and Agricultural Engineering, University of Zimbabwe,
Harare, Zimbabwe

Abstract

Nitrogen (N_2) fixation is the basis of the global N cycle. Therefore, it is not surprising that the ability to fix atmospheric N_2 evolved in the “primeval soup” and is deeply rooted in the evolutionary tree of life. Despite this, nitrogenase remains an enzyme exclusive to prokaryotes; no eukaryote has been described that can fix N_2 except through a symbiotic relationship.

VARIOUS TYPES OF BIOLOGICAL NITROGEN FIXATION

Members of both the Archaea and Eubacteria can fix nitrogen (N_2), and, even within the Eubacteria, nitrogenase is widely distributed among the various bacterial divisions.^[1] The most widely distributed form of nitrogenase is known as the molybdenum nitrogenase, which, as its name suggests, contains molybdenum. Other nitrogenases in which molybdenum is substituted by iron or vanadium have been found in free-living bacteria such as *Azotobacter*. The structure and the synthesis of nitrogenases are reviewed in detail by Smith.^[2] A novel mechanism for N_2 fixation was described by Ribbe, Gadkari, and Meyer^[3] in *Streptomyces thermoautotrophicus*. This nitrogenase is very unusual: it apparently uses superoxide as its reductant and contains a molybdopterin cofactor as the active site.

The diversity of types of N_2 fixation arises from the multiple modes by which bacteria and eukaryotes have coevolved to meet the rather exacting demands of active N_2 fixation. Associations range from loose relationships of free-living heterotrophic bacteria with plant roots to highly evolved symbioses that involve complex morphological differentiation of both partners (Table 1). Symbioses between N_2 -fixing bacteria and eukaryotes range from the *Cyanobacteria* (often referred to as blue-green algae) with fungi in lichens, Cycads, and *Gunnera*; actinomycetes (generally placed in the genus *Frankia*) with a range of angiosperms; and the rhizobia with legumes (and the single non-legume genus *Parasponia*). N_2 -fixing bacteria tend to occupy habitats abundant in carbon (C) but, despite many claims, these cannot be considered to be true symbioses. Examples are N_2 -fixing bacteria in plant rhizospheres (often referred to in the literature as

“associative symbioses”) and endophytic N_2 -fixing bacteria present in the vascular tissues of graminaceous plants such as sugarcane.^[5]

INTRODUCTION OF N_2 -FIXING ORGANISMS

Obviously, if N_2 -fixing organisms are not present within the ecosystem, they cannot contribute to the N cycle. In the case of agriculture, the identification of new niches for introduction of legumes into cropping systems is one of the most important means of increasing inputs from N_2 fixation.^[6] Another example of the introduction of N_2 -fixing organisms that is practiced on a commercial scale is inoculation with rhizobia; this industry has existed for almost 100 years. Legumes that warrant rhizobial inoculation are fairly specific in their symbiosis and often only require inoculation when they are transported to regions outside their centers of diversity or to where they are not traditionally grown.^[7]

ENVIRONMENTAL FACTORS THAT REGULATE N_2 FIXATION

As with all biological processes, environmental conditions such as temperature and moisture availability regulate the rates of N_2 fixation. There is a range of adaptation among N_2 -fixing bacteria and symbioses. As indicated previously, many methanogenic archaeobacteria can fix N_2 and some of these are found in hot springs. By contrast, lichens are the dominant, primary producers in cold tundra. The optimum conditions for N_2 fixation therefore differ strongly, obviating generalizations in description of ideal environments. Extreme temperatures, whether hot or cold, and extremes

Table 1 Examples of the range of N₂-fixing organisms.

Status of organism	N ₂ -fixing organism	Symbiont
Heterotrophs		
Free living		
Anaerobic	<i>Clostridium</i> , <i>Methanosarcina</i>	
Microaerophilic	<i>Frankia</i> , <i>Azospirillum</i> , <i>Bradyrhizobium</i>	
Aerobic	<i>Azotobacter</i> , <i>Derxia</i>	
Root associated		
Microaerophilic	<i>Azospirillum</i> , <i>Paenibacillus</i>	
Endophytic	<i>Herbaspirillum</i> , <i>Acetobacter</i>	Sugar cane (<i>Saccharum</i>), tropical grasses
Symbiotic	<i>Frankia</i>	<i>Alnus</i> , <i>Myrica</i> , <i>Casuarina</i> , etc.
	<i>Bradyrhizobium</i> , <i>Mesorhizobium</i> , <i>Rhizobium</i> , <i>Sinorhizobium</i>	Many legumes, <i>Parasponia</i>
	<i>Azorhizobium</i>	<i>Sesbania rostrata</i>
	<i>Methylobacterium</i>	<i>Crotalaria</i> spp.
Autotrophs		
Free living		
Microaerophilic	<i>Rhodospirillum</i> , <i>Bradyrhizobium</i>	
Aerobic	<i>Cyanobacteria</i>	
Symbiotic	<i>Anabaena azollae</i>	<i>Azolla</i> spp.
	<i>Cyanobacteria</i> , <i>Bradyrhizobium</i>	Fungi (lichens), Cycads, <i>Gunnera</i>
		<i>Aeschynomene</i> spp.

Source: Adapted from Ledgard & Giller.^[4] ©1995 Marcel Dekker.

of drought or water logging are important constraints to N₂ fixation in particular climates. The focus in this entry is placed on environmental variables that commonly act as limiting factors for N₂ fixation at different scales.

Regulation at the Physiological Scale

The triple dinitrogen bond is highly stable and large amounts of energy are required for N₂ fixation in the form of adenosine triphosphate and electrons. Aerobic respiration is necessary to provide sufficient energy for high rates of N₂ fixation, yet the nitrogenase enzyme is denatured irreversibly on exposure to molecular oxygen. This is what has been termed as the “oxygen paradox” of N₂ fixation and a host of mechanisms has evolved to circumvent this

problem (Table 1).^[8] In legume/rhizobium symbioses, it is believed that the response of N₂ fixation to the majority of environmental stresses is regulated by altering the supply of oxygen to the N₂-fixing “bacteroids.” This is mediated by the presence of an “oxygen-diffusion barrier” that has a variable permeability to oxygen.^[9]

A common feature is the repression of nitrogenase activity by the presence of available N in combined mineral forms (NH₄⁺ and NO₃⁻). In cultures, active N₂ fixation by free-living bacteria is only strongly induced when the mineral N becomes limiting for bacterial growth. In N₂-fixing symbioses, availability of large amounts of mineral N can act to repress N₂ fixation in several ways. Nodule formation and development are inhibited by combined N (in what is sometimes referred to as “autoregulation”), or the activity of N₂-fixing nodules can be suppressed (in what is termed N feedback regulation).^[10] This can be viewed as an evolutionary demand of the symbiosis, as it demonstrates that the extent of development of colonization of plant tissues is under the firm control of the host, effectively preventing parasitism.

Nutrient Limitations

A range of nutrients are required for N₂ fixation.^[6,10] Among the potential nutritional constraints to N₂ fixation, availability of phosphorus (P) is often overriding in both agricultural and natural systems. This is true even in aquatic systems where P limits the growth and N₂ fixation of Cyanobacteria and *Azolla*.^[11] Deficiency of P forms part of the “soil acidity complex” that limits N₂ fixation in acid soils, particularly in old, weathered soils of the tropics^[12] or in volcanic soils.^[13] These are precisely the conditions that favor deficiency of molybdenum, which is an essential component of the nitrogenase enzyme. Molybdenum deficiency is relatively uncommon except in older, weathered tropical soils but can be locally important. Many other major- and micronutrients have special roles in N₂ fixation; for instance, calcium is important for infection and nodule formation in legumes and cobalt is required by N₂-fixing bacteria for electron transport.^[6,10]

Toxicities

As with deficiencies, toxic effects can be manifested on both N₂-fixing bacteria and their symbionts. Toxicity of aluminum is of particular note: It appears to interfere specifically with the process of infection and nodule formation in legumes and is a major problem for survival of N₂-fixing bacteria in acid soils.^[6] Heavy metals appear to be particularly toxic to N₂-fixing bacteria in soil. This is true for free-living heterotrophic N₂-fixing bacteria, *Cyanobacteria*, and rhizobia that are all more sensitive to heavy metal toxicity than are host legumes.^[14]

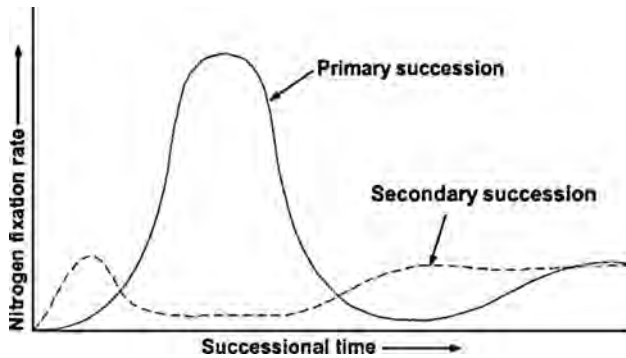


Fig. 1 Changes in the relative importance of N_2 fixation at different stages in vegetation succession.

Source: From Gorham, Vitousek, et al.^[16] ©1979.

Regulation of N_2 Fixation at the Ecosystem Scale

The feedback of abundant combined N on regulating rates of N_2 fixation by both free-living bacteria and N_2 -fixing symbioses also acts to regulate rates of N_2 fixation at the scale of the (agro)ecosystem.^[15] In natural ecosystems, N_2 fixation is more important in early successional phases (unless other nutrients are strongly limiting; Fig. 1) but becomes relatively unimportant as the amount of N in the soil gradually accumulates.^[16] A resurgence of the importance of N_2 fixation at later stages of primary or secondary succession is supported when the system becomes more abundant in C and limited in N. Agricultural systems are also influenced in the same way: Rates of N_2 fixation are largest when N is limiting but other nutrients plentiful. In systems dominated by legumes, such as fodder banks, rates of N_2 fixation decline as the system accumulates N in the soil. Given that most agricultural production systems have high rates of N removal in harvested products, and hence slow rates of N accumulation, a large demand for high rates of N_2 fixation is the normal situation.

REFERENCES

1. Young, J.P.W. Phylogenetic classification of nitrogen fixing organisms. In *Biological Nitrogen Fixation*; Stacey, G., Burris, R.H., Evans, H.J., Eds.; Chapman and Hall: New York, 1992; 43–85.

2. Smith, B.E. Structure, function, and biosynthesis of the metallosulfur clusters in nitrogenases. *Adv. Inorg. Chem.* **1999**, *47*, 159–218.
3. Ribbe, M.; Gadhari, D.; Meyer, O.J. N_2 fixation by *Streptomyces thermoautotrophicus* involves a molybdenum dinitrogenase and a manganese superoxide oxidoreductase that couple N_2 reduction to the oxidation of superoxide produced from O_2 by a molybdenum-CO dehydrogenase. *J. Biol. Chem.* **1997**, *272* (26), 627–633.
4. Ledgard, S.J.; Giller, K.E. Atmospheric N_2 fixation as an alternative nitrogen source. In *Nitrogen Fertilization and the Environment*; Bacon, P., Ed.; Marcel Dekker: New York, 1995; 443–486.
5. Boddey, R.M.; Döbereiner, J. Nitrogen fixation associated with grasses and cereals: Recent progress and perspectives for the future. *Fert. Res.* **1995**, *42*, 241–250.
6. Giller, K.E. *Nitrogen Fixation in Tropical Cropping Systems*, 2nd Ed.; CAB International: Wallingford, 2001; 352 pp.
7. Eaglesham, A.R.J. Global importance of *Rhizobium* as an inoculant. In *Microbial Inoculation of Crop Plants*; Campbell, R., Macdonald, R.M., Eds.; IRL Press: Oxford, 1989; 29–48.
8. Postgate, J. *Nitrogen Fixation*; Cambridge University Press: Cambridge, 1998.
9. Minchin, F.R. Regulation of oxygen diffusion in legume nodules. *Soil Biol. Biochem.* **1997**, *29*, 881–888.
10. O'Hara, G.W. Nutritional constraints on root nodule bacteria affecting symbiotic nitrogen fixation. *Aust. J. Exp. Agr.* **2001**, *41*, 417–433.
11. Roger, P. *Biological Management of the Floodwater Ecosystem in Wetland Ricefields*; IRR/ORSTOM: Los Baños, The Philippines/Paris, 1995; 250 pp.
12. Sanchez, P.A. *Properties and Management of Soils in the Tropics*; John Wiley & Sons: New York, 1976; 618 pp.
13. Vitousek, P.M. Nutrient limitation to nitrogen fixation in young volcanic sites. *Ecosystems* **1999**, *2*, 505–510.
14. Giller, K.E.; Witter, E.; McGrath, S.P. Toxicity of heavy metals to microorganisms and microbial processes in agricultural soils—A review. *Soil Biol. Biochem.* **1998**, *30*, 1389–1414.
15. Hartwig, U.A. The regulation of symbiotic N_2 fixation: A conceptual model of N feedback from the ecosystem to the gene expression level. *Perspect. Plant Ecol.* **1998**, *1*, 92–120.
16. Gorham, E.; Vitousek, P.H.; Reiners, W.A. The regulation of chemical budgets over the course of terrestrial ecosystem succession. *Ann. Rev. Ecol. Syst.* **1979**, *10*, 53–84.

Biota

Lev O. Karpachevsky

Moscow State University, Moscow, Russia

Mikhail Karpachevsky

Faculty of Soil Science, Moscow State University, Moscow, Russia

Abstract

Biota—the totality of living organisms—is an important factor of soil formation, which creates organic matter, loosens rock, and modifies its composition by transforming into soil. In the absence of organisms, rocks are transformed into loose sedimentary materials similar to lunar regolith. Autotrophs, plants (their biomass on the earth is 2.4×10^{12} ton), as well as sulfur bacteria, iron bacteria, etc. build up organic matter. Heterotrophs, animals, and microorganisms (0.02×10^{12} ton) consume organic matter and decompose it to carbon dioxide and water. According to the type of feeding, they are divided into phytophages (feed on living plants), zoophages (predators), necrophages (eat animal corpses), and saprophages (consume decaying plants). The biota plays an essential role in creating the anisotropy of soils.

ROLE OF PLANTS

The biota is responsible for the redistribution of nutrients within the pedosphere, soils, and their accumulation in the upper layer of soil. This is especially true for such nutrients as carbon (C), nitrogen (N), phosphorus (P), and sulfur (S). Practically, all biological cycles are primarily driven by plants. Plants uptake chemical elements from soil, water, and atmosphere and buildup phytomass—the source of soil humus. The phytomass deposits on or in soil. The annual fall of plant litter, mortmass, on the soil surface ranges from 1 ton/ha to 30 ton/ha. Reserves of roots in soil may reach 20 ton/ha in tropical forest and 8–12 ton/ha in grasslands, temperate deciduous forests, and southern taiga. Up to 30% roots die annually. Roots of herbaceous plants and partly fine tree roots promote humus formation in the soddy horizon, undershrub and root horizon, and, partly, humic A horizon. These horizons are the soil factory of humus and it can form only in them.

The roots make up only 0.1–1% of soil by weight, while they directly affect soil in the vicinity up to 3 mm from the root surface, within the rhizosphere—the zone close to fine sucking roots. However, this generally has very little effect on the soil properties. Typically, the rhizosphere is densely populated with microorganisms, protozoans, and other invertebrates associated with these organisms.

If soil is continuously frozen, phytomass decomposition at the stage of the formation of litter (mortmass) slows down. The reserves of phytomass reach their maximum in forests—tropical, subtropical, temperate, deciduous, and southern taiga. The reserves of mortmass—fresh and partly decomposed litter—generally decrease from tundra and

northern taiga to steppes, accompanied by increasing humus content in soil (soil “mortmass”).

The biogeochemical importance of plants is also due to the fact that they uptake nutrients. The comparison of coefficients of enrichment (biological uptake) of plants relative to lithosphere and soil shows that plants add in the trophic chain additional amounts of sodium (Na), magnesium (Mg), potassium (K), calcium (Ca), copper (Cu), boron (B), P, S, bromine (Br), caesium (Cs), and gold (Au). Plants, when compared with soils, contain noticeably less amounts of fluorine (F), arsenic (As), and cadmium. At the same time, soil, when compared to lithosphere and sedimentary rocks, concentrates beryllium, S, chromine, zinc (Zn), As, molybdenum (Mo), silver (Ag), antimony, tin, iodine (I), Cs, Au, and Br. Soils on calcareous material contain much less Ca and barium. Plants, when compared to soil, accumulate more Na, Mg, K, Ca, manganese, cobalt, Cu, Zn, rubidium, Mo, Ag, and I. This accumulation is significant for B, P, S, Br, Cs, and especially Au. The enrichment of a geological layer with these elements and impoverishing in F, Na, Mg, chlorine, K, Ca, Cu, Rubidium (Rb), mercury, lead (Pb), and uranium (U) attests that this rock experienced a phase of soil formation.

Besides changing the chemical composition of rocks and accumulating humus, plants together with animals create the anisotropy of soils—a regular change in the system of soil horizons downward and spatial variation in properties of upper horizons (the soil pattern). Each particular plant and their group exert a specific effect on soil. Long-living immobile organisms, like trees, modify the soil during the whole life. However, their effect is not the same at various distances from the edifying plants, like large trees. Typically, soil pH in boreal ecosystems is lower near tree

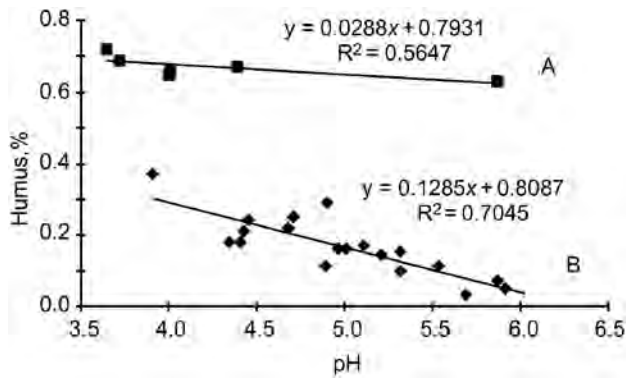


Fig. 1 The correlation between humus content and soil acidity in eluvial horizons of sandy aluminum–iron podzols in pyrogenic pine forests of Surgut Woodland, Western Siberia. Samples from pink-colored eluvial horizons—the material from deeper horizons exposed by tree windthrow after wildfire and being actively bleached. Samples from sugar-white eluvial horizons—long bleached material of upper horizons.

trunks, while litter reserves and humus content are higher than under the canopy. The death of an individual tree or its group also has an important effect on soil formation. Thus, windthrow of trees mixes the soil, creating specific pit and mound topography. The traces of uprooting in soil can be recognized for decades. Fig. 1 shows the differences between two types of eluvial horizons of podzols found in pine forest, Western Siberia. Pink-colored eluvial horizons are the richer material from deeper horizons exposed by tree uprooting after wildfire and being actively bleached. Sugar-white eluvial horizons are long bleached material of upper horizons.

Local disturbances, like the death of a tree, may maintain the stability of ecosystem, while dynamically changing its internal structure through gap mosaic. Extended disturbances, like wildfires, create an opportunity for new species to colonize the landscape. Such plant communities in the course of their development (succession) alter an environment, making it more suitable to newer species. Long-term changes in soil properties following wildfire can be traced for hundreds of years as shown by Fig. 2. Thus, plants change soil properties (Table 1), structure soil, and increase its porosity relative to a parent material. Different plants and plant communities transform soil material in different way.

ROLE OF ANIMALS

The distribution of animal weight is similar to plants. Animals living on the ground comprise less than 0.5% of the weight of all animals in ecosystem. In turn, animals dwelling in litter and organic horizons (mortmass) make up about 90% of all animals by weight in tundra and northern taiga ecosystems. In steppe ecosystems, the weight of animals dwelling in mortmass decreases to 2–5% of the total

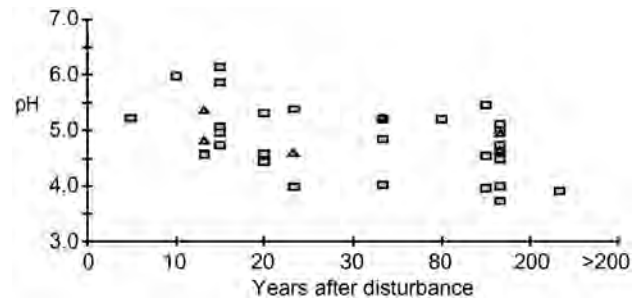


Fig. 2 Change in pH of drained and poorly drained aluminum–iron podzols in Surgut Woodland, Western Siberia, in the course of successions after wildfire.

zoomass as litter reserves decrease, while up to 90–98% of the zoomass occurs in the soil. Most animals in grasslands and deserts live in soil, while in taiga they occur in forest litter. The distribution of microbial mass coincides with that of zoomass—in taiga its maximum occurs in litter, while further south it shifts to soil.

Major components of soil fauna are protozoans (amoeba, infusoria, etc.), worms, beetle larvae, arachnoids, slugs, etc. Saprophages play a prominent role in soil formation, especially burrowing animals—moles, ground squirrels, marmots, and mice. Soil-dwelling animals are subdivided into groups: geobionts (constantly live in soil), geophiles (spend a major part of their life cycle in soil), and geoxenes (occasionally dwell in soil or use it as a shelter).

Earthworms (*Oligochaeta* and a group of the families Megadrili) are represented by the families Lumbricidae, Criodrilidae, Ocnerodrilidae, Moniligastridae, Megascolecidae, etc. Most widespread are species of the family Lumbricidae. Earthworms are saprophages, part of them live in litter, others in soil. Litter-dwelling species, e.g., *Dendrobaena attemsi*, *Dechloromonas hortensis*, *Dendrodrilus rubidus*, *Allobophora parva*, *Eisenia submontana*, also live in decaying trees, dispersing organic material and partially mixing it with mineral soil. Earthworms dwelling in soil (species *Nicodrilus caliginosus*, *Allobophora chlorotica*, *Acaabaria biserialis*, *Allolobophora sturanyi*, *Kritidrilus calarensis*, etc.) pass soil through their gastrointestinal tract feeding on soil organic matter. Various ecological groups of the family Lumbricidae dwell on the ground; in soil and litter; in humus horizon; some large species make deep burrows and tunnels in soil. Earthworms produce coprolites—excrements rich in organic compounds, microorganisms, and N compounds; they often have high physical resistance. The upper layer of chernozems of old gullies and gray forest soils (Dnepropetrovsk Oblast, Ukraine) are 60–80% made up by earthworm coprolites. Groups of earthworms living in soil and litter and on the ground can penetrate far north; burrowing earthworms dwell even in dry forests of Mediterranean type. According to the type of feeding, earthworms are further subdivided into two ecological groups: feeding on leaf litter and detritus on the ground and proper soil forms feeding on

Table 1 Properties of soils (0–30 cm) in forest, virgin grassland, and arable land (according to N.S. Oreshkina).

Land type	Soil solids density (g/cm ³)	Porosity (vol %)	Specific surface (m ² /g)	Infiltration (mm/min)	Humus (% of mass)	
Soddy-podzolized soil, Malinki, Moscow Oblast						
Meadow	1.37	2.64	48	n.d.	2	3.4
Forest	1.19	2.62	54	46	130	1.1
Field	1.47	2.62	44	54	17	n.d.
Typical chernozems, Kursk Oblast						
Virgin grassland	1.08	2.62	56	114	68	4.5
Forest	1.05	2.59	58	94	148	5.4
Field	1.17	2.65	54	91	83	2.9
Dark chestnut soil, Dzhanibek, Western Kazakhstan						
Virgin grassland	1.06	2.58	60	122	2	3.7
Forest	1.08	2.62	58	n.d.	190	3.9
Field	1.30	2.66	51	137	12	3.1

Note: n.d., not determined.

raw humus. The number of earthworms in soils varies from one to hundreds earthworms per square meter. Earthworms feeding on litter on average consume up to 5 mg/g living mass per day. Earthworms may accelerate litter decomposition by 2 to 3 times. In the presence of earthworms (*N. caliginosus*, 20 worms and *Lumbricus terrestris*, 3 worms), the raw humus layer (H, or Oh) and humus layer, A, up to 5 cm thick develop. When earthworms are absent, only the F (Of) layer forms.

Other representatives of soil fauna are beetle larvae, ticks, annelids, collembolids, etc. basically dwelling in litter and the A horizon. Worms of the class Enchitridae, which ingest organic matter, are up to 10–25 mm long. In acid peat soils, there can be from 85,000 to 200,000 worms per square meter of soil, with a total weight of 0.3–30 g. Nematodes are small (up to 1 mm) segmented worms, whose number may reach several million individuals per square meter, feeding on decaying remnants of animals and plants, soil microflora, alga, and fungal mycelium. Sloths, with a size of 0.2–0.3 mm, dwell in mosses, feeding on small animals and plant cells. Myriapods, e.g., small symphylids, eat decaying plants and few are predators. Diplopoda, including millepedes, *Rossius kessleri*, eating dead plants can be up to 10–17 cm long in the tropics. Their exoskeletons accumulate Ca, strontium, U, Pb, etc. Few species are predators. Almost 95% of insects are dependent on soil during their life cycle. Some of them are phytophages—Elateridae, mole cricket, cutworm, phylloxera, May beetle larva, etc.—others are saprophages (larvae of Diptera—flies and mosquitoes: Bibionidae, Licoryidae, and Tipulidae). They decompose organic mass, transforming it into raw humus. Besides, beetle larvae (e.g., May beetle larvae) and beetles themselves live in soil.

Ants, which mix soil, are widespread in natural ecosystems (up to 50 kg/ha). They are partly predators and necrophages.

Wood louse, mainly saprophagous, is also common. Termites (5–22 g/m²) live in southern soils under conditions of deep groundwater. With the help of microorganisms dwelling in their alimentary tracts, they digest cellulose and burrow channels down to the depth of 8 m from the soil surface.

Animals 0.1–3 mm long are referred to as microfauna. The most abundant are ticks (up to 50,000 species and especially those with exoskeleton, beetle mites) feeding on fungus hyphae and decaying plant debris. Their number in forest litter reaches 200,000–300,000 per square meter with the weight of 20 kg/ha. They annually discharge (mg/m²): 1 in tundra, up to 6 in taiga, up to 8 in temperate deciduous forests, 2.2 in steppes, 1 in semi-deserts, 0.1 in deserts, and 13 in humid subtropics. Collembolids are lower wingless insects eating small plants (alga) dispersing plant debris. They give several generations per year with the total number of 1–50 million/m² and biomass of 0.2–6.4 g/m².

Protozoans are single-cell organisms inhabiting all soils, totaling up to 20 billion/m².

Rhizopoda, flagellates, and infusoria are 2–20 µm long and widely occur in soil. Their primary food is bacteria. Microorganisms are also considered part of the biota. They dwell on plant leaves, in soil, litter, and alimentary tract of all animals, participating in transformation of organic matter of ecosystems. Microorganisms also promote N fixation in soil (*Azotobacter*, *Clostridium pasterianum*, and *Rhizobium*), methane emission from soil, etc. In soils, they are adsorbed on mineral particles. Microorganisms participate in all processes of organic matter transformation at all trophic levels of the food chain. The number of microorganisms in soils is estimated in tens of thousands to billions of cells per gram of soil.

More than 1500 alga species occur in soil—more than 600 species of green alga, Chlorophyta, more than 400 species of cyanobacteria, Cyanophyta, nearly 300 species

of diatomic alga, Bacillariophyta, and nearly 200 species of yellow green alga, Xantophyta. The total biomass of soil alga is estimated to be 0.5–1 ton/ha. Alga fix N from the atmosphere and accumulate C, thus making the primary trophic level.

Fungi and actinomycetes play a considerable role in organic matter decomposition. The length of the mycelium of actinomycetes can reach 100–300 m/g of soil, increasing up to 6000 m/g in peatlands. The number of cells is estimated to be 2–300 million/g. Fungi decompose wood and litter and accelerate C cycling in ecosystems.

Animals contain a noticeably less variety of elements in their bodies and the total amount of elements [except C, O, N, and hydrogen (H)] are much less compared to plants. However, there is one part of animal activity, which is by scale comparable or even exceeds the biogeochemical activity of plants. This is the role of animals in reallocation and mixing of soil (burrows, disturbed grounds, passages in soil, etc.) and organic matter decomposition.

According to Abaturonov,^[1] mole discards may occupy up to 50% of plot area, totaling to 50 ton/ha. Typically, animals discard the material from deeper horizons on the soil surface—E, EB, and B in forests and AB, B, and BC in steppes and forest steppe. In taiga soils, discarded material is enriched with Ca and clay. In steppe soils, it is enriched with calcium carbonates, salts, and gypsum. The addition of elements to the upper layer of eluvial soils by animals often exceeds their input with plant litter. The discarded material becomes involved in soil formation. In the 0–7 cm layer, the humus content increases from 0.8% C during the first year after discard to 4% in 17 years. Animals like wild boar can actively reallocate soil matter—when abundant they may disturb 5–10% of the ground. Furthermore, they mix the upper and lower layers of soil down to the depth of 20 cm. Thus, in pine forests polluted during the Chernobyl accident, the radioactivity of soil decreased from 43 to 25 mR/h from 1987 to 1990, while in places disturbed by wild boar it decreased to 20 mR/h during the first year.

Dams made by beavers provoke soil waterlogging on midslopes and peat formation. Herbivores, including elephants trampling grasslands and savannas, make soil denser and decrease the content of humus in it. Excrements of mice in tundra and taiga lower the soil acidity permitting to survive pathogenic microorganisms infesting leptospirosis and pseudotuberculosis (*Hebdomadis*, *Grippytyphosa*, etc.).

Invertebrates and fungi are major destroyers of dead plant material. Fungi destroy wood and break down plant litter to trash. Observations show that trunks of beech, oak, and spruce with the diameter at breast height of 30–40 cm become fully decomposed on the soil surface in 100 years. Decomposing trunks, besides fungi and alga, are inhabited with mosses, higher plants, including woody species. In Northern European Russia, partly decomposed dead trunks are a place where seeds of spruce and fir may germinate, because large deadwood is not so quickly inhabited by

mosses and undershrubs (e.g., bilberry). In Kamchatka, saplings of larch also prefer deadwood, because trunks of decomposing trees are characterized by high moisture content and nutrient reserve, thus promoting young trees to grow on these substrata. Decomposing trunks are often inhabited with ants. Saprophages can be divided into those which feed only on organic substratum and whose excrements consist of 80–90% organic matter and organisms, which together with organic matter consume mineral substratum (many species of earthworms). The excrements of earthworms (coprolites) contain much more N and organic matter than the A horizon. Thus, the coprolites of *Eisenia nordeskioldi* contain 29% organic matter and 0.95% N, while those of *Dendrobena octaedra* contain 53% organic matter and 1.21% N. However, they contain more mineral matter than the excrements of wood louse (51–80% organic matter and 1.9% N), Bibionidae larvae (58–83% and 1.8%), and millipedes (76% and 1.5%). Fresh litter has about 90% organic matter and 1.5–1.9% N.

Although saprophages eating litter increase the ash content from 2–8% to 7–15%, they do not form soil humus. The humus matrix—humus tightly fixed on the surface of soil particles—is primarily formed by earthworms, which by three times accelerate decomposition of litter of various woody species. Soil worms mix organic matter with soil, thus promoting formation of the humus matrix in soil. Other animals and litter worms create raw humus, dispersed organic matter with higher ash content than fresh litter.

REFERENCE

1. Abaturonov, B.D. *The Functional Role of Animals (in Russian)*; Nauka: Moscow, 1980.

BIBLIOGRAPHY

1. Gilarov, M.S. Distribution peculiarities of soil-dwelling invertebrates in different zonal genetic soil types. In *Proceedings of 8th International Congress of Soil Science*, Bucharest, 1964; 3, 551–560.
2. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
3. Karpachevsky, L.O. *Ecological Soil Science (in Russian)*; Mosk. Gos. Univ.: Moscow, 1993.
4. Sukachev, V.N. *The Principles of Forest Biogeocenology (in Russian)*; Nauka: Moscow, 1964.
5. van der Drift, J.; Doeksen, J., Eds. *Soil Organisms*; North-Holland: Amsterdam, 1963.
6. Vsevolodova-Perel, T.S. *Range and Regularities in the Distribution of Earthworms of the USSR Fauna (in Russian)*; Nauka: Moscow, 1979.
7. Wilde, S.A. *Woodlands of Wisconsin*; University of Wisconsin-Madison-Extension: Madison, 1976.
8. Zvyagintsev, D.G. Structure and functions of the soil complex. In *Structural and Functional Role of Soil in the Biosphere (in Russian)*; GEOS: Moscow, 1999.

Biota and Disease-Suppressive Soils

Nicola Lorenz

Ohio State University, Columbus, Ohio, U.S.A.

Abstract

It has been estimated that globally 13% of crop yield is lost due to common plant diseases such as wilt, blight, and damping off triggered by soil-borne pathogens like *Fusarium* spp., *Phytophthora* spp., and *Pythium* spp. or *Rhizoctonia solani*. Plants affected by disease have been commonly treated with synthetic biocides, a method which is highly criticized in many countries. Thus, research has been focusing on soils showing resistance toward plant diseases to identify factors preventing plant disease outbreak. Soil disease suppression occurs when beneficial soil microorganisms such as *Pseudomonas* spp., *Gliocladium* spp., *Trichoderma* spp., *Bacillus* spp., and *Streptomyces* spp. are abundant and active in soils. Besides these known and well-described microorganisms, still unknown soil disease suppressive microorganisms are expected to be discovered in the near future using modern molecular techniques. Important biotic mechanisms for disease suppression triggered by soil organisms are: i) antibiosis, ii) competition for trace elements, nutrients and microsites, iii) predation or parasitism, and iv) hypovirulence. Advantageous for biological disease suppression are also soil organic carbon (SOC)-rich soils. Thus, studies have been focusing on amending soils with organic matter (OM) to enhance disease suppression. This entry highlights well-established and recent findings on biota and soil conditions supporting disease suppression.

INTRODUCTION

In conventional agriculture, the use of mineral fertilizers has generally replaced organic fertilizers, cover crops, and manure. Tillage depth and intensity have increased, which in return have led to decreasing soil organic carbon (SOC) stocks in agricultural soils.^[1] Concomitantly, the occurrence of soil-borne plant diseases increased triggering a rise in synthetic pesticide applications worldwide.^[2] In non-disease suppressive soil, soil-borne plant pathogens like *Fusarium* spp. and *Phytophthora* spp. are causing wilt and blight in vegetable crops. *Pythium* spp. are known to trigger damping-off, which is affecting seeds or seedlings. Another prominent fungal plant pathogen is *Rhizoctonia solani*, causing serious loss in potato (*Solanum tuberosum* L.) production worldwide. Higher organisms like soil-inhabiting nematodes, for example, *Meloidogyne* spp. and *Pratylenchus* spp. triggering root knot and root lesion as well as protozoa have the potential to substantially damage plants. Most plant pathogens are difficult to control, since they are increasingly resistant toward synthetic pesticides.

In response to this dilemma, studies have focused on alternatives to chemical pesticides, analyzing naturally disease suppressive soils by studying soil microbial biomass, diversity, and activity.^[3] Others have focused on SOC content or soil organic matter (OM) amendment and its role in supporting plant health thus restoring natural soil disease suppression.^[4] Numerous reviews and scientific studies

have been published on disease suppression in soils. This entry highlights some examples of disease suppressive biota in soils (*Pseudomonas* spp., *Gliocladium* spp., and *Trichoderma* spp.), followed by a discussion on the role of OM in disease suppression.

DISEASE SUPPRESSION IN SOILS

Disease suppressive soils are defined as soils in which certain diseases are suppressed due to the presence of microorganisms antagonistic to a pathogen.^[5] As outlined below, key soil microorganisms are involved in disease suppression; in addition, soil OM amendments have the potential to enhance soil disease suppression by promoting antagonistic microorganisms.

Soil Biota Suppressing Disease

Naturally disease suppressive soils are important subjects for research because the indigenous microflora is able to effectively protect plants from diseases. Soil disease suppression can be characterized as “general suppression,” where the entire microbial community is directly or indirectly involved in disease suppression. In other cases, certain members/taxa of the microbial community act as the main disease suppressors, this phenomenon is referred to as “specific disease suppression.” For example, specific disease suppressive microbial species or groups

such as *Pseudomonas fluorescens*, *Streptomyces* spp., *Agrobacterium radiobacter*, and fungi like *Gliocladium* spp. and *Trichoderma* spp. act to suppress *Pythium* spp. or *R. solani*.^[6] Those organisms use antagonistic mechanisms against plant pathogens. Mechanisms of plant protection differ from organism to organism, and relationships and mechanisms steered by environmental conditions are highly complex. A basic principle holds true when the antagonist controls the pest: It is a matter of tipping the balance toward disease suppression, which depends on the whole microbial community (general suppression), the type of pathogen and antagonist (specific suppression), the place and time of the pathogen's and antagonist's occurrence, their metabolic activity,^[7] as well as the soil condition (SOC content, pH, and nutrient contents^[8]).

Basic mechanisms of biological disease suppression are: 1) antibiosis (inhibition or destruction of the pathogen by a metabolic product of the antagonist, e.g., lytic enzymes); 2) competition for trace elements, nutrients, and microsites; 3) parasitism or predation; and 4) hypovirulence.

Antibiosis occurs when antagonists are metabolically active producing metabolites or enzymes harmful for the pathogen(s). As a result of competition for trace elements, disease suppression occurs, this happens when antagonists are more successful in using nutrients for their growth and thus effectively outcompeting pathogens. Predation typically refers to species that feed at higher trophic levels; however, it has also been used for the actions of microbes, for example, protists and mesofauna, such as fungal feeding nematodes and microarthropods, which consume pathogen biomass. Hypovirulence refers to the reduction of virulence of a pathogenic strain.^[7]

Pseudomonas spp.

The suppression of root diseases by *P. fluorescens* strain CHA0 was related to the production of the secondary metabolite 2,4-diacetylphloroglucinol (DAPG).^[9] Several studies have shed light on the regulation of DAPG, an antifungal compound being released by *Pseudomonas*. Both, pathway-specific and general regulators have been reported controlling the biosynthesis of DAPG. It was observed that DAPG production by *P. fluorescens* strain CHA0 can be affected by plant phenolics indicating that plant-derived compounds have the potential to regulate the activity of microorganisms involved in disease suppression. It was also observed that diverse *Pseudomonas* spp. populations correlated with increased antagonistic activity measured by the production DAPG and other antibiotics, which significantly decreased fungal pathogens. In addition, a review on the rhizosphere ecology of tobacco (*Nicotiana glutinosa* L.) revealed that suppressiveness to *Thielaviopsis basicola*-mediated black rot is largely attributed to DAPG-producing *Pseudomonas* spp.^[10] In a *R. solani*-suppressive

soil, *Pseudomonas* spp. and *Trichoderma* spp. have been identified as some of the most active microbial taxa engaged in disease suppression^[6]).

Gliocladium spp. and *Trichoderma* spp.

The soil-borne fungi *Gliocladium* spp. and *Trichoderma* spp. have been studied comprehensively for their suppressiveness of plant pathogens. Mainly three mechanisms of biotic disease suppression are used by these fungi to prevent disease outbreak, namely antibiosis, competition, and mycoparasitism. An example for mycoparasitism is *Gliocladium roseum*, which is known as a parasite for fungi and nematodes. In addition, *G. roseum* produces a wide range of volatile organic compounds that are toxic to organisms including other fungi, bacteria, and insects and is of interest as a biological pest control agent.^[11]

Several strains of *Trichoderma* have been developed as biocontrol agents against fungal diseases of plants,^[6] since they have been shown to effectively suppress soil-borne fungal plant diseases. The various mechanisms include antibiosis, parasitism, inducing host-plant resistance, and competition. *Trichoderma* spp. produce enzymes (chitinases, glucanases, cellulases, and proteases) that are highly effective in suppressing *Pythium* spp. or *R. solani* in soils.^[8] Most biocontrol agents are derived from the species *Trichoderma harzianum*, *Trichoderma viride*, and *Trichoderma hamatum*. The biocontrol agent generally grows in its natural habitat on the root surface and thus specifically affects root disease but can also be effective against foliar diseases.

Research has highlighted the importance of previously unknown fungi for disease suppression, which revealed that disease suppressive soils were hosting mycoparasites such as *Xylaria* spp. with known antibiotic activity against plant pathogenic fungi. Biota plays a significant role in disease suppression, but more research is needed to fully understand the mechanisms and specific microorganisms controlling diseases. Because soils are complex ecosystems, the factors controlling disease are therefore highly diverse and not only restricted to biotic interactions but also steered by soil properties.^[12]

Soil OM Amendments and Disease Suppression

Compost application to soils has been shown to be a successful OM amendment to prevent plant disease outbreak. Analyzing 18 different composts, Termorshuizen et al.^[13] revealed that compost has generally a positive effect or no effect on disease suppression and very rarely stimulated disease. Another study has shown that complex organic amendments (a mixture of compost, slurry, and dung) have resulted in greater *Fusarium* wilt suppression compared to adding them separately. In a review, Metha

et al.^[14] highlighted that compost application to soils is a promising tool to prevent plant disease outbreak. However, since microbial communities in compost are the main drivers of disease suppression, it is a matter of promoting antagonistic microorganisms in compost to affectively control pests. Studies enriching compost with beneficial microorganisms such as *Trichoderma* spp. have been conducted and shown increased disease suppressiveness. But there are expectations for beneficial but unknown microorganisms (especially non-cultivable microorganisms) to be discovered in compost by modern molecular approaches. A better understanding of compost-inhabiting microorganisms will increase the use of compost in enhancing soil disease suppression.

Different complementary mechanisms have been proposed to explain the suppressive capacity of OM amendments like compost. For example, these include enhanced activities of antagonistic microbes, increased competition against pathogens for resources hampering fungal growth, or induction of resistance in the host plants.^[15]

However, despite the potential benefits of organic soil amendments, there are several concerns about its use, since several studies have indicated that the effects of compost on disease suppression are variable. In several cases, compost and other organic amendments either enhance disease severity or show no effect.^[13] Negative effects of OM amendment were often associated with increased inoculum and growth of pathogenic fungi and oomycetes. In addition, it was discussed that the release of phytotoxic compounds could damage plant roots and predisposes them to pathogen attack. The inconsistent disease control results obtained with compost amendments, with both disease suppressive and conducive (disease increasing) effects, show that more research is needed to fully understand disease suppressive mechanisms of composts.^[13]

Besides compost, biochar has been discussed for its potential to decrease disease outbreak. While being mixed with compost and being amended to soils, it was shown that mixed biochar amendments affected soil mycorrhizal associations and thus prevented *Fusarium* rot in asparagus (*Asparagus officinalis* L.) suggesting that biochar has the potential to stimulate antagonistic microbial communities preventing disease outbreak.^[16] The reduction of application of synthetic pesticides by biochar application on soils appears plausible but more research is needed to fully understand the effect of biochar on disease suppression.^[17]

CONCLUSION

Disease suppressive soils have been shown to suppress common diseases such as wilt, blight, and damping off triggered by *Fusarium* spp., *Phytophthora* spp., and *Pythium* spp. It has been shown that disease suppressive

soils host key soil microorganisms involved in plant disease suppression such as *Trichoderma* spp., *Gliocladium* spp., *Pseudomonas* spp., *Bacillus* spp., and *Streptomyces* spp. Besides these known and well-described microorganisms, more, unknown soil dwelling microorganisms are expected to be involved in disease suppression. Disease suppressive microorganisms use several mechanisms, namely antibiosis, competition, parasitism/predation, and hypovirulence, which hamper disease outbreak. Overall, successful disease suppression in soils is highly dependent on the type of pathogen and antagonist (specific suppression), the place and time of the pathogen's and antagonist's occurrence, their metabolic activity, and the soil condition (SOC content, pH, and nutrient contents). In general, SOC-rich soils offer the most favorable conditions for disease suppression.

Compost has been shown to promote disease suppressive soil; however, the effects of compost are not always beneficial. It has been revealed that compost can promote plant pathogens and can trigger the release of phytotoxic compounds damaging the plant root, thus predisposing plants to disease. More research is needed to fully understand the highly complex disease suppressive mechanisms in soils.

REFERENCES

- Freibauer, A.; Rounsevell, M.D.A.; Smith, P.; Verhagen, J. Carbon Sequestration in the agricultural soils of Europe. *Geoderma* **2004**, *122*, 1–23.
- Pimentel, D. Green revolution agriculture and chemical hazards. *Sci. Total. Environ.* **1996**, *188* (Suppl.1), S86–S98.
- Mendes, R.; Garbeva, P.; Raaijmakers, J.M. The rhizosphere microbiome: Significance of plant beneficial, plant pathogenic, and human pathogenic microorganisms. *FEMS Microbiol. Rev.* **2013**, *37*, 634–663.
- Bonanomi, G.; Antignani, V.; Pane, C.; Scala, F. Suppression of soilborne fungal diseases with organic amendments. *J. Plant Pathol.* **2007**, *89* (3), 311–324.
- Shurtleff, M.C.; Averre, C.W., III. *Glossary of Plant-Pathological Terms*; American Phytopathological Society Press: St. Paul, 1997.
- Hicks, E.; Bienkowski, D.; Braithwaite, M.; McLean, K.; Falloon, R.; Stewart, A. *Trichoderma* strains suppress *Rhizoctonia* diseases and promote growth of potato. *Phytopathol. Mediterr.* **2014**, *53* (3), 235–247.
- Chernin, L.; Chet, I. Microbial enzymes in the biocontrol of plant pathogens and pests. In *Enzymes in the Environment. Activity, Ecology and Applications*; Burns, R.G., Dick, R.P., Eds.; Marcel Dekker Inc.: New York, 2002; 195–258.
- Liu, B.; Gumpertz, M.L.; Hu, S.; Ristaino, J.B. Long-term effects of organic and synthetic soil fertility amendments on soil microbial communities and the development of southern blight. *Soil Biol. Biochem.* **2007**, *39*, 2302–2316.

9. Keel, C.; Schnider, U.; Maurhofer, M.; Voisard, C.; Laville, J.; Burger, U.; Wirthner, P.; Haas, D.; Défago, G. Suppression of root diseases by *Pseudomonas fluorescens* CHA0: Importance of the bacterial secondary metabolite 2,4-diacetylphloroglucinol. *Mol. Plant Microbe. In.* **1992**, *5* (1), 4–13.
10. Almario, J.; Moënne-Loccoz, Y.; Muller, D. Monitoring of the relation between 2,4-diacetylphloroglucinol-producing *Pseudomonas* and *Thielaviopsis basicola* populations by real-time PCR in tobacco black root-rot suppressive and conducive soils. *Soil Biol. Biochem.* **2013**, *57*, 144–155.
11. Harman, G.E.; Kubicek, C.K., Eds. *Trichoderma and Gliocladium*, Vol. 2; Taylor and Francis: London, 1998.
12. Penton, C.R.; Gupta, V.V.S.R.; Tiedje, J.M.; Neate, S.M.; Ophel-Keller, K.; Gillings, M.; Harvey, P.; Pham, A.; Roget, D.K. Fungal community structure in disease suppressive soils assessed by 28 S LSU Gene sequencing. *Plos One* **2014**, *9* (4), 1–12.
13. Termorshuizen, A.J.; van Rijn, E.; van der Gaag, D.J.; Alabouvette, C.; Chen, Y.; Lagerlöf, J.; Malandrakins, A.A.; Paplomatas, E.J.; Rämert, B.; Ryckeboer, J.; Steinberg, C.; Zmora-Nahum, S. Suppressiveness of 18 composts against 7 pathosystems: Variability in pathogen response. *Soil Biol. Biochem* **2006**, *38*, 2461–2477.
14. Metha, C.M.; Palni, U.; Franke-Whittle, I.H.; Sharma, A.K. Compost: Its role, mechanism and impact on reducing soil-borne plant diseases. *Waste Manage* **2014**, *34*, 607–622.
15. Senechkin, I.V.; van Overbeek, L.S.; van Bruggen, A.H.C. Greater Fusarium wilt suppression after complex than after simple organic amendments as affected by soil pH, total carbon and ammonia-oxidizing bacteria. *Appl. Soil Ecol.* **2014**, *73*, 148–155.
16. Elmer, W.H.; Pignatello, J.J. Effect of biochar amendments on mycorrhizal associations and Fusarium crown and root rot of asparagus in replant soils. *Plant Dis.* **2011**, *95*, 960–966.
17. Wiedner, K.; Glaser, B. Biochar-fungi interactions in soils. In *Biochar and Soil Biota*; Ladygina, N., Rineau, F., Eds.; CRC Press: Boca Raton, 2013; 69–99.

Biotechnology

Manas R. Banerjee

Laila Yesmin

Research and Development Division, Brett-Young Seeds Limited, Winnipeg, Manitoba, Canada

Abstract

Although there are several genetically modified products commercially available, biotechnology has no magic solution to all our problems. It has tremendous promise, but to attain its full potential it must be environmentally compatible, economically viable, and socially responsible. This is an ever-emerging technology that needs proficient management of its development. However, biotechnology may be the single most sought-after tool at our disposal capable of making significant contribution toward the food security and sustainable development.

INTRODUCTION

Biotechnology is an applied field that can be viewed from many perspectives. To some, it is the age-old technique of selecting better cultivars and wine production, and to others, it is the recombinant DNA technique that creates the basis of modern biotechnology. The term “biotechnology” may suggest a single subject, but in reality it is a multidisciplinary approach to utilize science for the benefit of our society. The word “biotechnology” originates from Greek words “bios” and “technologos,” which means living and technical study, respectively. The literal meaning of biotechnology is, thus, the technical study of living things. The simplest way to look at biotechnology is the use of living organisms to produce goods and services for industrial purposes. *Chambers Science and Technology Dictionary* defines biotechnology as “the use of organisms or their components in industrial or commercial process, which can be aided by the techniques of genetic manipulation in developing, e.g., novel plants for agriculture or industry.”^[1] Furthermore, the *Macmillan Dictionary of Biotechnology* defines the term as “the application of organisms, biological systems, or biological processes to manufacturing and service industries.”^[2] This definition actually includes any process in which organisms, tissues, cells, organelles, or isolated enzymes are used to convert biological or other raw materials to products of greater value, as well as the design and use of reactors, fermenters, downstream processing, and analytical and control equipment associated with biological manufacturing processes.^[2] Biotechnology has been used for centuries in fermentation, cheese, and bread making, but it involves innovative devices like molecular techniques, *in vitro* techniques, genetic engineering,

cloning, protein manipulation, monoclonal antibodies, and stem cells. Biotechnology is no longer the applied branch of biological sciences rather it is a strategic technology having diverse application on the socioeconomic development and sustainability of the industrialized countries (Fig. 1).

APPLICATIONS AND PROSPECTS

Biotechnology has made remarkable progress over the last decades and is well integrated with technical improvement of processing and instrumentations. For example, penicillin manufacturing has changed over the years with the development of efficient strain and improved operational techniques. The same is true for many pharmaceutical products, drugs, fine chemicals, biopolymers, energy, and processed food products. In developed world, major biotech industries are in pharmaceutical, food, agriculture, environment, and health sectors, whereas in developing countries, food and agriculture biotechnology are progressing rapidly.

With the introduction of genetic engineering in the agricultural sector, biotechnology has made fabulous advances in crop yield and quality. Agronomic traits have been successfully developed or being developed through genetic manipulation of crop plant. For instance, “roundup-ready” soybean and canola were developed to protect from herbicide and “Bt” cotton and corn to provide protection against insect damage. Genetically modified (GM) crops are commercially grown in more than 40 countries, with over 110 million ha under cultivation.^[3]

This technology reached almost one billion acres of farmland in 2005, and millions of farmers are using it. Crops like soybean, oilseed rape, corn, and cotton have

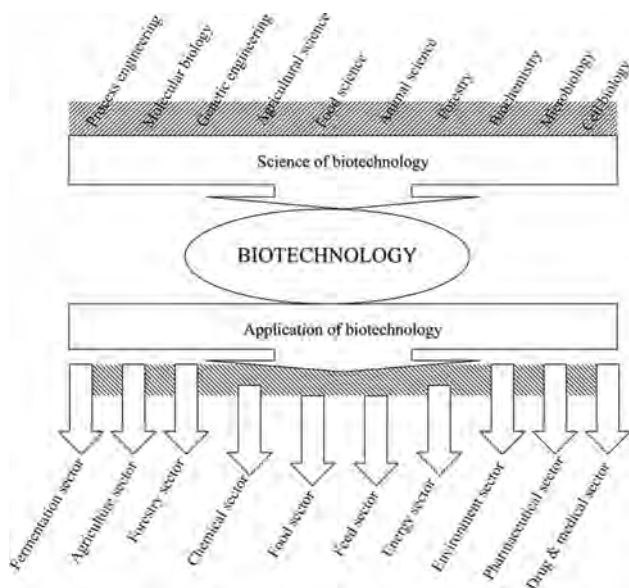


Fig. 1 The contribution of science toward the application of biotechnology.

the largest acreages and are primarily grown in countries like the United States, Canada, Argentina, and China. Biotechnological applications in agriculture have a potential market estimated at \$67 billion per year.^[4] Many developing countries are embracing agriculture biotechnology with the hope to increase their food production to meet the demand of growing population, as food security is their main concern. Food and Agriculture Organization of United Nations recognizes the potential of increasing production in agriculture, fisheries, and forestry through GM technology. This might not completely eradicate world food shortage, but it could definitely contribute to the food security particularly in the developing world. With increasing yield, GM technologies have also been shown to reduce pesticide use and increase farm profitability. The technology has been extended to local crops like banana, cassava, plantain, rice, and sorghum for abiotic stress tolerance and quality. “Golden rice” capable of producing provitamin A (beta-carotene) and iron-rich rice is also the example of this novel technology. As such, more challenging works in the areas of stress resistance, crop yield, and quality enhancement are on the horizon. Some microbiologically derived biopesticides and biofertilizers are already in the marketplace making strides in boosting agricultural production. Besides, biotechnology indirectly aids in carbon sequestration by reducing tillage that minimizes soil erosion, helps water retention, and reduces carbon escape from soil. Introducing mycorrhizal fungi to increase plant growth that enhance moisture and nutrient uptake resulting in plant capacity to sequester more carbon is another example. Biotechnology is also contributing to the improvement of biodiversity and restoring ecological health via the reduction of carcinogenic chemical use. Advances on effective identification

of genes and genomic and proteomic research play a tremendous role in improving biodiversity as well. The preservation of terrestrial ecosystem is an important step to maintain biodiversity as ecological damage by deforestation already wiped out and/or threaten the existence of many valuable species. If GM technology could produce more food per unit of existing farmland, then more land will be protected for wildlife and forest species. Biotech research based on gene called “cellulose binding domain” that accelerates tree growth (superfast GM tree) would give hope to reforestation and damage control. The GM technology also opens the door for plant-based edible vaccines for preventable human diseases like tetanus, diphtheria, measles, and cholera. This cost-effective vaccine would have tremendous impact for global population health especially in developing and underdeveloped countries. For example, potato-derived hepatitis B vaccine would likely be the first edible vaccine that could save millions of lives.^[5]

As the public’s awareness of “green technology” is increasing, so does their reluctance of using harmful products. With biotech innovation, non-degradable toxic products are being replaced with biodegradable and less toxic products, and the applications are extended into plastics, food processing, textiles, polymers, and energy and mining industries. Environmental sector uses biotechnology in pollution control, waste management, and renewable resource areas. Although the use of transgenic in bioremediation is limited, the potential could be substantial in the foreseeable future. Biotechnology can also be applied in the conventional fossil fuel and renewable energy sector. For instance, a non-toxic biopolymer of xanthan gum and cellulose avoids the problems of conventional drilling mud via better viscosity and binding in the drilling process. The production of renewable fuel-like ethanol can be achieved more efficiently by using modified cellulase enzyme that maximizes the conversion of cellulose into fermentable sugar and in turn helps in reducing greenhouse gas emissions.

Biotechnology is extensively used in the food and food processing industry. A range of essential food products and food additives requires microbes or microbial source. *Lactobacillus*, *Streptococcus*, and *Leuconostoc* are commonly used in dairy industry for flavor, consistency, and reducing spoilage time. Various techniques have been used to modify the functional properties of foods such as spreadability of butter and enhancement of fruit and vegetable qualities. Fermentation technology is the other key area of this applied science. Although pharmaceutical and food industries rely heavily on the advancement of this technology, almost every sector has its use. Many important drugs, antibiotics, steroids, vaccines, hormones, enzymes, diagnostic products, solvents, food and food processing products, animal feeds, animal vaccines, biopesticides, and inoculants are the results of this technology. Improvements on bioreactors, computer-

controlled fermentation processes, and downstream processing have added a new dimension to the biotech industry. With the evidence of new diseases and many recurring diseases, the drug and pharmaceutical sector is rapidly expanding in areas of antibiotics, vaccines, hormones, enzymes, and diagnostic tests using monoclonal antibodies. Biotechnology also makes it possible to mass-produce these substances that might otherwise be cumbersome and expensive to produce. New methods allow producing purer drugs like human insulin from GM bacteria that does not cause any allergic reactions on patients. Evidently, biotechnology is becoming more diverse in application and beginning to make substantial contributions to the health and well-being of our society.

DILEMMAS AND CONCERNS

Like any other science, biotechnology is also not free from some obvious concerns. Environmental release of transgenes can impact food safety, human health, and environment because of its vast applications in the food and agricultural industry. There is also public perception that the use of GM products or continued biotech research is ethically or morally wrong. Whether they are scientifically based or not, it is important that all these issues need to be carefully considered.

Herbicide and Insecticide Resistance

The possibility of transferring transgenes from GM crops to wild relatives to create “superweeds” and insects that have developed tolerance to insecticides to create “superbugs” is not an unreal threat. Herbicide-resistant crops are voluntarily coming up in different fields creating huge problems. However, agronomic practices like crop rotation, hybrid rotation, and integrated pest management could reduce these risks.

Transfer of Allergens

To develop soybeans with higher methionine content, a Brazil nut gene was inserted, which is responsible for methionine-rich protein, into soybean. The transgenic soybean was supposed to be utilized as animal feed. But the product was found to be allergenic, and the soybean was never put on the market.^[6] StarLink corn was released for animal feed only. The corn contains Cry9C protein, a potential allergen. The genes were found in human food and seed corn causing many corn-based foods to be recalled although no health problems associated with the corn is reported.^[4] However, there is a real concern of accidental mix-up with the human foods.

Antibiotic-Resistant Genes

Antibiotic-resistant genes originally isolated from bacteria are used as selectable markers for transforming plants. By the use of such genes, cells that have been modified can grow in the presence of specific antibiotic, for instance, kanamycin-resistant corn. If these genes are transferred into human or other bacteria, they may increase antibiotic resistance in human or bacterial population. As such worldwide, an estimation of \$9 billion is wasted each year on ineffective use of antibiotics.^[7]

Gene Escape and Genetic Pollution

Preventing the movement of transgenes to the other plants is a concern for biotechnologists to avert the genetic pollution. There are some gene containment techniques like male sterility, terminator technology, apomixis, cleistogamy, chloroplast transformation, and transgenic mitigation, which can reduce gene transfer,^[8] but these are used for theoretical or laboratory purposes only.^[9]

Social, Moral, and Ethical Issues

Multinationals have heavily invested in biotechnology, and they would naturally want a return on their investment. This is a legitimate concern to think that these companies will either enforce technology fees or monopolize the marketplace. However, studies have revealed that farmers also enjoy a financial benefit from GM crops.^[6] This is mainly because of less pesticide use and enhanced production. In Canada, GM canola farmers experienced a 40% reduction in herbicide costs and a 10% increase in yield.^[10] Besides, other issues like increased reliance on developed countries by developing countries, control of food production by a special few, foreign exploitation of natural resources, and infringe on the natural organism's inherent traits are of great concern.^[3] Does it morally right to do gene transfer from bacteria to plant, animal to another animal, and human to animal? Are these ethically correct? These dilemmas are warranted rationale answer.

CONCLUSION

Although there are several GM products commercially available, biotechnology has no magic solution to all our problems. It has tremendous promise, but to attain its full potential it must be environmentally compatible, economically viable, and socially responsible. This is an ever-emerging technology that needs proficient management of its development. However, biotechnology may be the single most sought-after tool at our disposal capable of making significant contribution toward the food security and sustainable development.

REFERENCES

1. Walker, M.B., Ed. *Chambers Science and Technology Dictionary*; W&R Chambers Ltd: Edinburgh, 1991; 1–1008.
2. Coombs, J. *Macmillan Dictionary of Biotechnology*; Macmillan Press: London, 1986; 1–330.
3. Venter, B. Modified crops: The pros and cons. In *Pretoria News*; Pretoria, 2004. Available at <http://www.pretorianews.co.za/index.php?SectionID=665&ArticleID=2150709> (accessed September 2004).
4. Borem, A.; Santos, F.R.; Bowen, D.E. *Understanding Biotechnology*; Prentice Hall, PTR: New Jersey, 2003; 1–216.
5. Marsa, L. Garden variety vaccines may be edible alternative. In *Los Angeles Times*; Los Angeles, 2005. Available at <http://www.agbioworld.org> (accessed April 2005).
6. CropBiotechnet. *Biotechnology Myths & Facts*; Global Knowledge Center on Crop Biotechnology: Manila, 2002; 1–37. Available at <http://www.isaaa.org/kc> (accessed August 2004).
7. Grace, E.S. Biotechnology of the body. In *Biotechnology Unzipped: Promises & Realities*; Trifolium Books Inc.: Toronto, 1997; 55–96.
8. Daniell, H. Molecular strategies for gene containment in GM crops. *Nat. Biotechnol.* **2002**, *20*, 581–586.
9. Slatter, A.; Scott, N.; Fowler, M. Future prospects of GM crops. In *Plant Biotechnology, The Genetic Manipulation of Plants*; Oxford University Press: Oxford, New York, 2003; 305–333.
10. Canola Council of Canada. *An Agronomic and Economic Assessment of Transgenic Canola*. Canola Council of Canada, 2001; 1–95. Available at http://www.canola-council.org/production/gmo_main.htm (accessed September 2004).

Boreal Forest

Galina Mazhitova

Komi Science Center, Russian Academy of Sciences, Syktyvkar, Russia

Abstract

This entry is based on English and Russian publications and deals with soils of the circumpolar boreal biome, including the Subarctic forest–tundra and sparse forest. Environmental conditions are described. Major features of soil formation under boreal conditions, geographical distribution of soils in major boreal regions, land use, and global ecological significance of boreal soils are discussed.

INTRODUCTION

The boreal forest or taiga is a northern forest dominated by conifers (Fig. 1). The forest dominates the boreal biome, where it alternates with bogs and other ecosystems. The biome is circumpolar in the northern hemisphere (Fig. 2), and it is subdivided into low boreal, mid-boreal, and high boreal. The transition between the high boreal and the Arctic composed of forest-tundra/sparse taiga is called the Subarctic. Besides plains and lowlands, the boreal forest occupies a number of mountain systems: the Fennoscandian Range, the Ural Mountains, the mountains of southern and northeastern Siberia, and the northern extent of the North American Cordillera. The boreal forest covers 58.6 Mha in Scandinavia, 709.0 Mha in the Former Soviet Union, 42.4 Mha in Alaska, and 221.8 Mha in Canada.^[1–6] In addition, the Subarctic comprises 340.0 Mha in Russia and 171.2 Mha in Canada.^[4,6]

ENVIRONMENTAL CONDITIONS

Large boreal regions, with the exception of East Siberia, experienced multiple continental ice sheets in the Pleistocene, which predetermined the prevalence of geologically young parent materials and distribution of periglacial phenomena. Climate was highly dynamic during the Holocene, causing shifts of the northern tree line and southern border of the permafrost and the polygenetic nature of many subarctic soils.^[7,8] The northern tree line roughly corresponds to the July isotherm of 10°C. The climate of the boreal forest is characterized by long, cold, and snowy winters; however, it varies widely across the biome (Table 1).^[9–13] East Canada is humid with relatively mild winters, whereas East Siberia is semiarid and reveals the highest thermal continentality on earth. Either continuous or discontinuous permafrost

occurs everywhere in the Subarctic and in the closed taiga of East Siberia and Alaska. The rest of the biome is characterized by seasonal ground freezing. The most widely distributed boreal forests in the world are larch forests; in Russia, they occupy 269 Mha.^[14] Other important tree species are spruce, pine, fir, and birch.

SOIL CLASSIFICATION

In terms of Soil Taxonomy, the boreal biome comprises the largest area of Spodosols and Histosols, is the only biome where the Arctic and Alpine contain Gelisols, and is an important area of Alfisol formation. Inceptisols are also widespread. In terms of the world reference base (WRB) for soil resources, the boreal biome contains most of the Albeluvisols, Podzols, and Histosols and the second largest area of Cryosols after the Arctic. Luvisols, Cambisols, Gleysols, Regosols, Leptosols, and Fluvisols also occur.

MAJOR FEATURES OF SOIL FORMATION

Most soils in the boreal biome either are Gelisols, i.e., permafrost-affected soils, or have cryic thermal regime. Aquic or udic soil moisture regimes are common. Ice-cemented permafrost, where present, adds to soil saturation. In East Siberia, however, salts accumulate in some soils as a result of the semiarid climate. The cold climate hinders chemical weathering. The combinations of moisture regime, low pH, and non-neutralized organic compounds foster leaching, clay illuviation, and podzolization in many soils. The latter, though not confined to the boreal biome, is best pronounced and widely expressed throughout it. Permafrost and, in many regions, seasonal frost cause various cryogenic features in soils. Nutrient turnover is slow. Only a small fraction of the ecosystem nutrients are contained in the vegetation, and



Fig. 1 Mid-boreal spruce forest in European Russia.

Source: Photo courtesy of E. Lopatin.

the most are contained in soils.^[15,16] The most biologically active nutrient fraction is cycled in litterfall to the forest floor. In the Subarctic, because the trees are shallow rooted, the forest floor is the main source of nutrients for tree growth. However, boreal ecosystems are well adapted to the conditions of their growth. Conifers

need a small amount of nutrients and are able to grow on highly leached acid soils. About half of the world's wetland area is in the boreal biome, and most wetlands are bogs, that is, their soils are Histosols and Histels. In the area of continuous permafrost, peat plateaus and polygonal peatlands are typical. In the discontinuous permafrost area, palsa peatlands develop, and in the zone of seasonal frost, string mires and domed bogs are characteristic.^[17,18]

SOILS OF THE MAIN BOREAL REGIONS

Scandinavia

Spodosols, mostly typic Haplocryods, and Histosols dominate the soil cover. Many Spodosols in Lapland are cryoturbated, possibly in the absence of permafrost. Frost heave occurs everywhere in the region of the seasonal frost in Finland. The only Gelisols recognized are sphagnum or terric fibristsels in the palsas of Lapland.^[17]

European Russia

Cryalfs and, in the northern taiga, Aqualfs (Albeluvisols in terms of WRB) on loamy materials^[19] and Spodosols derived from coarse-textured parent materials^[20] dominate the area with small inclusions of Histosols (Sphagnum domed bogs). Similar to Scandinavia, no mineral Gelisols form within the biome; however, Histels develop in the forest-tundra under peat plateaus or palsas. West-facing slope and foothills of the Ural Mountains support the least human-affected boreal forests in Europe, which grow on Cryalfs and Cryepts.



Fig. 2 Global boreal forest biome.

Table 1 Information from boreal weather stations.

Station	Region	Mean annual temperature (°C)	Mean July temperature (°C)	Mean January temperature (°C)	Annual precipitation (mm)
Palojarvi ^a	Lapland	−2.1	12.1	−12.4	413
Syktyvkar ^b	European Russia	0.4	16.6	−15.1	514
Oymyakon ^c	East Siberia	−16.7	13.6	−47.5	233
Fairbanks ^d	Alaska	−2.7	16.0	−22.2	284
Moosonee ^e	Ontario, Canada	−1.3	15.1	−20.5	700
St. John's ^f	Newfoundland, Canada	4.7	15.4	−4.3	1482

^aPeriod of 1991–1997.^b1889–1989.^c1943–1980.^d1961–1983.^e1951–1980.^f1961–1990.

Source: Meteorological Yearbook of Finland,^[9] *Scientific-Applied Climate Reference Book of the USSR. 1. Air and Soil Temperature*,^[10] Ping,^[11] Atmospheric Environment Service. Canadian Climate Normals 1951–1980, Temperature and Precipitation: Ontario,^[12] and Atmospheric Environment Service. Canadian Climate Normals 1961–1990. Temperature and Precipitation: Atlantic Provinces.^[13]

West Siberia

West Siberia contains one of the world's largest lowland areas (128 Mha). Continuous permafrost occurs in the forest–tundra and is discontinuous in the northern taiga. Soils of the region have never been classified according to the Soil Taxonomy, except for several pedons in the Subarctic.^[21] The analysis of Russian literature^[21–23] indicates that not only histels but also mineral Gelisols develop in the Subarctic and in the east of the high boreal. To the south of these areas, Cryalfs and Cryods dominate well-drained sites, whereas Cryaqualfs and Cryaquods with deep albic horizons and often ortstein develop in poorly drained sites affected by groundwater. String-and-fen bogs with fibrists and hemists are typical for permafrost-free areas.

Central and East Siberia

The largest part of East Siberia has continuous low-temperature permafrost and is dominated by Gelisols.^[24–26] The Russian soil classification^[27] recognizes the soil type “Cryozem,” which is widely distributed here (oxyaquic Cryosols in WRB; typic haploturbels in the Soil Taxonomy). Oxyaquic features develop in fine-textured materials due to the presence of oxygen in the water of melting ground frost aided by the drainage provided by frost cracking.^[28] In Central Yakutia, unique haglels showing evidence of solidification develop.

Alaska

Eighty percent of boreal Alaska has sparse or discontinuous permafrost. Turbels, with some haglels and histels,

dominate permafrost-affected areas. Haplocryods and Humicryods are common for the Cordillera in south-eastern Alaska, whereas in southern Alaska, Andisols are the dominant soils.^[29]

Canada

About 30% of the land area in the Subarctic is covered with wetlands.^[18] In the continental high subarctic region, most wetlands are affected by permafrost, and their soils include histels. In the areas adjacent to Hudson Bay and in the east of the Canadian Subarctic, unfrozen fen and palsa complexes are common with fibrists, hemists, and histels. In addition, mineral Gelisols (both haglels and turbels) develop here as opposed to more southern boreal regions.^[29,30] South from the Subarctic, Cryepts and to a smaller extent Cryalfs dominate the western part of the territory with more Spodosols and Cryalfs dominating the eastern part.^[31]

LAND USE

The main use of boreal lands is for forestry. In Scandinavia, large areas of Histosols are artificially drained to cultivate forests. Main agricultural use is for seasonal pastures, with very small areas of ploughed lands. In major economic divisions of European Russia, the area of ploughed lands varied in 1981 from 8% to 39%,^[32] whereas it was incomparably smaller in Siberia. Most boreal soils are acid and nutrient poor and require liming and organic and inorganic fertilizers to grow crops.

GLOBAL ECOLOGICAL SIGNIFICANCE OF BOREAL SOILS

Global change ecological relationships within the taiga could assume global importance, because the biome greatly affects the world's carbon balance. Boreal forests contain 200 Pg organic carbon in soils, which is three times more than the estimated biomass pool.^[33] This accounts for 15% of global terrestrial detrital carbon pool.^[34,35] In the past, boreal regions, along with the Arctic ones, have been the net sink of carbon. This is primarily due to slow decomposition of soil organic matter. However, under global change, the biome's carbon pool is one of a few most vulnerable carbon pools in the Earth system. The boreal forest may become a major source of carbon due to anthropogenic activities and the increased rate of decomposition due to soil warming.

CONCLUSION

The boreal forest or taiga dominates the boreal biome, where it alternates with bogs and other ecosystems. Boreal biome is circumpolar in the northern hemisphere with large areas in both Eurasia and North America. Most boreal soils are permafrost affected or experience seasonal freezing. The boreal biome has the largest area of Spodosols and Histosols, is the only biome where the Arctic and Alpine contain Gelisols, and is an important area of Alfisol formation. The main land use of the boreal biome is forestry. Agricultural land use is limited to seasonal pastures and small areas of crop lands. Global change ecological relationships within the boreal biome could assume global importance, because the biome greatly affects the world's carbon balance. Three-fourth of the biome's organic carbon is located in soils.

REFERENCES

1. Statistics Finland/Yleisjulkaisut; 1998.
2. Statistics Norway/Natural Resources and Environment; 1999.
3. Statistics Sweden/Section: Agriculture, Forestry, and Fishery; 2000.
4. Kolchugina, T.P.; Vinson, T.S.; Gaston, G.G.; Rozhkov, V.A.; Schlentner, S.F. Carbon pools, fluxes, and sequestration potential in soils of the former soviet union. In *Soil Management and Greenhouse Effect*; Lal, R., Kimble, J., Levine, E., Stewart, B., Eds.; Lewis Publishers: Boca Raton, London, Tokyo, 1995; 25–40.
5. van Cleve, K.; Chapin, F.S. III; Dyrness, C.T.; Viereck, L.A. Element cycling in taiga forest: State-factor control. *Bio-Science* **1991**, *41* (2), 78–88.
6. Tarnocai, C. Carbon pools in soils of the Arctic, Subarctic, and Boreal regions of Canada. In *Global Climate Change and Cold Regions Ecosystems*; Lal, R., Kimble, J., Stewart, B., Eds.; Lewis Publishers: Boca Raton, London, New York, Washington, 2000; 91–103.
7. Aleksandrovsky, A.L. *Soil Evolution in the European Plain in Holocene*; Nauka: Moscow, 1983; 152 pp. (In Russian).
8. Tarnocai, C. Paleosols of the interglacial climates in Canada. *Geogr. Phys. Quatern.* **1990**, *44* (3), 363–374.
9. *Meteorological Yearbook of Finland*; Finnish Meteorological Institute: Helsinki, 1991–1997. (Annual Issues).
10. *Scientific-Applied Climate Reference Book of the USSR. 1. Air and Soil Temperature*; Hydrometeoizdat: Leningrad, 1989. 1(2) and 24(3) (In Russian).
11. Ping, C.L. Soil temperature profiles of two alaskan soils. *Soil Sci. Soc. Am. J.* **1987**, *51* (4), 1010–1018.
12. *Atmospheric Environment Service. Canadian Climate Normals 1951–1980. Temperature and Precipitation: Ontario*; Environment Canada, Atmospheric Environmental Service: Downsview, 1982; 254.
13. *Atmospheric Environment Service. Canadian Climate Normals 1961–1990. Temperature and Precipitation: Atlantic Provinces*; Environment Canada, Atmospheric Environmental Service: Downsview, 1993; 105.
14. Buktynov, A.D.; Groshev, B.I.; Krylov, G.V. *Forests*; Mysl: Moscow, 1981; 318 pp. (In Russian).
15. Rodin, L.E.; Bazilevich, N.I. *Dynamics of the Organic Matter and Biological Turnover of Ash Elements and Nitrogen in the Main Types of the World Vegetation*; Nauka: Moscow, 1965; Vol. 254. (In Russian).
16. Powers, R.F.; Van Cleve, K. Long-term ecological research in temperate and boreal forest ecosystems. *Agron. J.* **1991**, *83* (1), 11–24.
17. Seppala, M. Palsas and related forms. In *Advances in Periglacial Geomorphology*; Clark, M.J., Ed.; John Wiley: Chichester, 1988; 247–278.
18. Zoltai, S.C.; Tarnocai, C.; Mills, G.F.; Veldhuis, H. Wetlands of subarctic Canada. In *National Wetlands Working Group Canada Committee on Ecological Land Classification*; Ecological Land Classification Series, 1988; Vol. 24, 56–96.
19. Tonkonogov, V.D. *Texture-Differentiated Soils of European Russia*; V.V. Dokuchaev Soil Science Institute: Moscow, 1996; 156 pp. (In Russian).
20. Aparin, B.F.; Zaboieva, I.V.; Lipkina, G.S.; Nogina, N.A.; Rudneva, E.N.; Rusakova, T.V.; Sloboda, A.V.; Urusevskaya, I.S. *Podzolic Soils of the Centre and East of the European Territory of the USSR*; Nauka: Leningrad, 1981; 200 pp. (In Russian).
21. Mazhitova, G.; Lapteva, E., Eds. *Trans-Ural Polar Tour. Guidebook*; 2nd Ed.; Publishing Service, Institute of Biology Komi SC UrD RAS: Syktyvkar, 2004.
22. Karavaeva, N.A. *Paludification and the Evolution of Soils*; Nauka: Moscow, 1982; 296 pp. (In Russian).
23. Avetov, N.A.; Trofimov, C.Y. Soil cover of the taiga and floodplain landscapes in the Pur river basin, West Siberia. *Pochvovedenie* **1997**, *1*, 31–35. (In Russian).
24. Elovskaya, L.G.; Petrova, E.I.; Teterina, L.V. *Soils of Northern Yakutia*; Nauka: Novosibirsk, 1979; 303 pp. (In Russian).

25. Mazhitova, G.G. Soil formation on the Yedomas of the Kolyma lowland. *Sov. Soil Sci.* **1990**, 22 (3), 12–23.
26. Yershov, Yu.I. Features proper to forest soil formation in the north of middle Siberia. *Pochvovedenie* **1995**, 7, 805–810. (In Russian).
27. Shishov, L.L.; Tonkonogov, V.D.; Lebedeva, L.I.; Gerasimova, M.I. *Russian Soil Classification System*; Dokuchaev Soil Science Institute: Moscow, 1997. (Translated into English).
28. Sokolov, I.A. Hydromorphic non-gley soil formation. *Pochvovedenie* **1980**, 1, 21–33. (In Russian).
29. Moore, J.P.; Swanson, D.K.; Fox, C.A.; Ping, C.L. *International Correlation Meeting on Permafrost-Affected Soils*; Guidebook—Alaska Portion, USDA Soil Conservation Service: Lincoln, 1993.
30. Zoltai, S.C.; Tarnocai, C. Some nonsorted patterned ground types in northern Canada. *Arctic Alpine Res.* **1981**, 13 (2), 139–151.
31. Lacelle, B.; Tarnocai, C.; Waltman, S.; Kimble, J.; Swanson, D.; Naumov, Ye.M.; Jacobson, B.; Broll, G. *Northern Circumpolar Soil Map*; Agriculture and Agrifood Canada, USDA, Dokuchaev Soil Institute, Institute of Geography—University of Copenhagen, Institute of Landscape Ecology—University of Muenster, 1998.
32. Fridland, V.M., Ed. *Soil Map of the RSFSR, Scale 1 : 2,500,000*; GUGK: Moscow, 1988.
33. Apps, M.J.; Kurz, W.A.; Luxmoore, R.J.; Nilsson, L.O.; Sedjo, R.J.; Schmidt, R.; Simpson, L.G.; Vinson, T. The changing role of circumpolar boreal forest and tundra in global C cycle. *Water Air Soil Poll.* **1993**, 70 (39), 39–54.
34. Schlesinger, W.H. Carbon balance in terrestrial detritus. *Annu. Rev. Ecol. Syst.* **1977**, 8, 51–81.
35. Post, W.M.; Emanuel, W.R.; Zinke, P.J.; Stangenberger, A.G. Soil carbon pools and world life zones. *Nature* **1982**, 298, 156–159.

Boron and Molybdenum

Umesh C. Gupta

Crops and Livestock Research Centre, Agriculture and Agri-Food Canada, Charlottetown, Prince Edward Island, Canada

J.A. MacLeod

Agriculture and Agri-Food Canada, Charlottetown, Prince Edward Island, Canada

Abstract

Boron (B) and molybdenum (Mo) are among the chief micronutrients essential for plant growth that exist in soil solutions as anions. B deficiencies occur in many countries of the world with coarse-textured soils in humid regions where leaching and heavy cropping have diminished the soil B reserves. B toxicity occurs in soils which are inherently high in B or as a result of high B fertilization. Mo deficiencies are common in many countries where soils are acidic and coarse in texture. Mo toxicities in crops tend to be rare in distribution and occur only when unusually high quantities of Mo are present.

INTRODUCTION

Boron (B) and molybdenum (Mo) are among the chief micronutrients essential for plant growth that exist in soil solutions as anions. B deficiencies occur in many countries of the world with coarse-textured soils in humid regions where leaching and heavy cropping have diminished the soil B reserves. B toxicity occurs in soils which are inherently high in B or as a result of high B fertilization. Such may be the case, e.g., if a B sensitive crop were to follow a crop that received a high rate of B, or through the use of irrigation water high in B. Crops sensitive to B, e.g., peanuts (*Arachis hypogaea* L.), could result in B toxicity^[1] at B levels as low as 2 kg ha⁻¹.^[2]

Mo deficiencies are common in many countries where soils are acidic and coarse in texture. Mo toxicities in crops tend to be rare in distribution and occur only when unusually high quantities of Mo are present. Soils high in Mo have been reported on some poorly drained soils, e.g., in Australia, Scotland, and the United States.^[3]

SOIL FACTORS AFFECTING PLANT UPTAKE

Boron

Soil reaction and pH

Generally, B becomes less available to plants with increasing soil pH. Visual symptoms of B deficiency were found to be accentuated by calcium (Ca) deficiency and were less evident when Ca was added in excess.^[4]

Macronutrients

Nitrogen (N) is of utmost importance in affecting B uptake by plants. Liberal N applications have been found to decrease plant B uptake. Increasing rates of phosphorus (P) and potassium (K) have also been found to decrease B uptake.

Soil properties

Higher quantities of available B are generally found in fine-textured than in coarse-textured soils. The lower amounts of B in sandy soils are associated with higher leaching of B. Organic matter is one of the chief sources of B in acid soils, as relatively little B adsorption occurs at low pH. Irrigation waters with high B can result in B toxicity in crops. B applied in bands is more effective than when applied broadcast in increasing B uptake by crops. Method of plowing (e.g., with moldboard plow) and ridge till resulted in higher B in plants than with no till and beds.^[5] Highly weathered acid soils are also responsible for causing B deficiency.^[6]

Other factors include environment, genotypes, and plant species, e.g., legumes can accumulate more B than grasses.^[7] Plant age and plant parts can affect the B composition of plants.^[8]

Molybdenum

Parent material and soil pH

Soils formed on sandstones that have experienced heavy leaching losses are most likely to be low in total

Mo. Alfisols derived from mixed shale may be higher in Mo than Spodosols formed from mixed schist. Lateritic soils derived from granite and basalt rocks are very high in total Mo. Soil solution MoO_4^{2-} is the most available form to plants. Its availability increases a hundredfold for each unit increase in pH. Alkaline soils have a relatively large proportion of Mo in the soil-solution phase. Some studies^[9] have indicated that the severity of Mo deficiency in lentil (*Lens culinaris* Medic) can be reduced by liming on Mo-deficient acid alluvial soils.

Soil properties

Coarse-textured loams and silty loams low in organic matter, as well as severely eroded and/or heavily weathered soils, are particularly low in Mo. The amount of Mo adsorbed may be closely related to the organic matter content. Soil wetness is one of the chief factors affecting the availability of Mo. Peats and mucks are products of a wet environment and have been associated with high Mo in certain regions of the world. Well-drained soils, e.g., podzols, are low in Mo. The availability of Mo to plants is also reduced by sulfur fertilization.

PHYSIOLOGICAL ROLE, DEFICIENCY, AND TOXICITY SYMPTOMS

Boron

B plays a key role in seed production. B deficiency can severely impede seed production particularly in legumes.^[10] B is involved in the transport of sugars across cell membranes and in the synthesis of cell wall material. Some studies suggest that the selection of transgenic cultivars with an increased sugar alcohol content can result in increased B uptake by translocating it as a complex sugar alcohol in the phloem.^[11] It promotes the elongation of epicotyls and hypocotyls and increased height of seedlings grown under aluminum stress.^[12] B deficiency can inhibit the growth of seedlings.^[13] B plays an important role in maintaining the integrity of plasma membranes of leaf cells and in alleviating the damage of membrane caused by low temperatures.^[13]

B deficiency in crops is more widespread than the deficiency of any other micronutrient. Because of its slow mobility in plants, B deficiency symptoms generally first appear on the young leaves at the top of the plant. Crops most sensitive to a B deficiency include rutabaga (*Brassica napobrassica* L.), other brassicas, forage legumes, and root crops.

The B toxicity symptoms are also similar among most plants. Generally, they consist of marginal and tip chlorosis, which is quickly followed by a necrosis.^[8]

Molybdenum

Mo is an essential constituent of several important molybdoenzymes. Nitrate reductase, which is involved in nitrate reduction in plants, and nitrogenase, which is involved in N fixation by legumes, are most important. Studies on beans in Brazil showed that foliar applications of Mo greatly enhanced nitrogenase and nitrate reductase activities resulting in increased N content in the shoots.^[14] Their results demonstrated that in certain soils, N fertilization may be replaced by small amounts of Mo as foliar application. Other roles of Mo include increased organic N, protein content, and drought resistance of plants. Lack of Mo can inhibit tasseling in corn and decrease sucrose and ascorbic acid concentrations in cauliflower (*Brassica oleracea* L. Botrytis Group) and legumes.

Deficiency symptoms for most micronutrients appear on the young leaves at the top of the plant because most micronutrients are not readily translocated. Mo is an exception in that it is readily translocated, and its deficiency symptoms generally appear as yellowing of the whole plant. Symptoms associated with Mo deficiency are closely related to N metabolism. In brassicas, Mo deficiency can result in cupped leaves, i.e., rolling or upward curling of leaves. Crops that commonly suffer due to Mo deficiency are forage legumes, brassicas, and soybeans [*Glycine max* (L.) Merr.]. Mo toxicity in crops is uncommon and is found only when unusually high concentrations of Mo are present. The most striking symptom of Mo toxicity as reported in alfalfa (*Medicago sativa* L.) is yellow or orange-yellow chlorosis, with some brownish tints that start in the youngest leaves. Further symptoms of Mo toxicity damage include moribund buds, thick stems, and development of axillary buds and sometimes succulent older leaves. Details of Mo deficiency and toxicity symptoms can be found elsewhere.^[3]

METHODS OF CORRECTING DEFICIENCY

Boron

Sodium borates of varying degrees of hydration and H_3BO_3 are principal sources of B fertilizers. Application rates of B generally range from 0.25 to 3.0 kg ha⁻¹ depending on crop requirements and the method of application. Higher rates are required for broadcast applications than for banded soil application or foliar sprays. On clay loam soils, rates of up to 8.8 kg ha⁻¹ are not toxic, while on sandy loam soils, this rate was toxic to cauliflower.^[15] However, on crops such as peanuts, sensitive to excess B, 2 kg B ha⁻¹ was found to be toxic.^[1] Grasses are highly tolerant of B, e.g., 45 kg B ha⁻¹ caused only 12% yield reduction.^[16] Most B sources are applied to soils with NPK fertilizers, and B may be incorporated or bulk blended with granular fertilizers or mixed with fluid fertilizers.^[17]

Table 1 Sufficiency levels of B and Mo in several crops.

Crop	Plant part	Sufficiency level (mg kg ⁻¹)	
		B	Mo
Alfalfa	Whole tops	20–40	0.12–1.29
Cauliflower	Leaves	11–97	0.19–0.25
Corn	Leaves	10	0.20
Red clover	Whole tops	21–45	0.3–1.59
Rutabagas	Leaves	40	—
Soybeans	Leaves	—	0.5–1.0
Wheat	Boot stage tissue	8	0.09–0.18

Source: From Gupta.^[3,8]

Molybdenum

Methods of controlling Mo deficiency in crops include band or broadcast application (generally of Mo contained in P or NPK fertilizers) to soil, foliar sprays, and seed treatment. Amounts applied to soil range from 50 g to 1 kg ha⁻¹ depending on the crop and soil pH. However, seed treatment is the most common method because the recommended rates are low, ranging from 50 to 400 g ha⁻¹. Mo sources can be applied to seeds as liquid or slurry, and some type of sticking or conditioning agents may be included. Because legume seeds, such as soybeans, peas (*Pisum sativum* L.), alfalfa, and clovers (*Trifolium* spp.), are treated with a bacterial inoculant, the Mo source for seed treatment must be compatible with the inoculant. Foliar rates of application range from 60 to 218 g Mo ha⁻¹.^[18,19] It should also be pointed out that on soils containing adequate Mo, liming soils to pH 6–6.5 can overcome a Mo deficiency in crops. Sufficiency levels of B and Mo for a few crops are shown in Table 1.

REFERENCES

1. Rashid, A.; Rafique, E.; Ali, N. Micronutrient deficiencies in rainfed calcareous soils of Pakistan II. Boron nutrition of the peanut plant. *Commun. Soil Sci. Plant Anal.* **1997**, *28*, 149–159.
2. Gupta, U.C.; Jame, Y.W.; Campbell, C.A.; Leyshon, A.J.; Nicholaichuk, N. Boron toxicity and deficiency: A review. *Can. J. Soil Sci.* **1985**, *65*, 381–409.
3. Gupta, U.C., Ed. *Molybdenum in Agriculture*; Cambridge University Press: Cambridge, 1997; 1–276.
4. Chatterjee, C.; Sinha, P.; Nautiyal, N.; Agarwala, S.C.; Sharma, C.P. Metabolic changes associated with boron-

- calcium interaction in maize. *Soil Sci. Plant Nutr.* **1987**, *33*, 607–617.
5. Lal, R.; Fausey, N.R.; Brown, L.C. Drainage and tillage effects on leaf tissue nutrient contents of corn and soybeans on crosby-kokomo soils in Ohio. In *Drainage in the 21st Century: Food Production and the Environment*; International Drainage Symposium, Orlando, FL, 1998; 465–471.
6. Fageria, N.K.; Baligar, V.C. Response of common bean, upland rice, corn, wheat, and soybean to soil fertility of an oxisol. *J. Plant Nutr.* **1997**, *20*, 1279–1289.
7. Adarve, M.J.; Hernandez, A.J.; Gil, A.; Pastor, J. Boron, zinc, iron, and manganese content in four grassland species. *J. Environ. Qual.* **1998**, *27*, 1286–1293.
8. Gupta, U.C. *Boron and Its Role in Crop Production*; CRC Press: Boca Raton, 1993; 237 pp.
9. Mandal, B.; Pal, S.; Mandal, L.N. Effect of molybdenum, phosphorus, and lime application to acid soils on dry matter yield and molybdenum nutrition on lentil. *J. Plant Nutr.* **1998**, *21*, 139–147.
10. Mozafar, A. Role of boron in seed production. In *Boron and Its Role in Crop Production*; Gupta, U.C., Ed.; CRC Press: Boca Raton, 1993; 185–206.
11. Bellaloui, N.; Brown, P.H.; Dandekar, A.M. Manipulation of *in vivo* sorbitol production alters boron uptake and transport in tobacco. *Plant Physiol.* **1999**, *119*, 735–741.
12. Yang, Y.H.; Zhang, H.Y. Boron amelioration of aluminum toxicity in mungbean seedlings. *J. Plant Nutr.* **1998**, *21*, 1045–1054.
13. Wang, Z.Y.; Tang, Y.L.; Zhang, F.S.; Wang, H. Effect of boron and low temperature on membrane integrity of cucumber leaves. *J. Plant Nutr.* **1999**, *22*, 543–550.
14. Vieira, R.F.; Vieira, C.; Cardoso, E.J.B.N.; Mosquim, P.R. Foliar application of molybdenum in common bean. II. Nitrogenase and nitrate reductase activities in a soil of low fertility. *J. Plant Nutr.* **1998**, *21*, 2141–2151.
15. Batal, K.M.; Granberry, D.M.; Mullinix, B.G., Jr. Nitrogen, magnesium, and boron applications affect cauliflower yield, curd mass, and hollow stem disorder. *Hort. Sci.* **1997**, *32*, 75–78.
16. Wilkinson, S.R. Response of kentucky-31 tall fescue to broiler litter and composts made from broiler litter. *Commun. Soil Sci. Plant Anal.* **1997**, *28*, 281–299.
17. Mortvedt, J.J.; Woodruff, J.R. Technology and application of boron fertilizers for crops. In *Boron and Its Role in Crop Production*; Gupta, U.C., Ed.; CRC Press: Boca Raton, 1993; 157–176.
18. Mortvedt, J.J. Sources and methods of molybdenum fertilization of crops. In *Molybdenum in Agriculture*; Gupta, U.C., Ed.; CRC Press: Boca Raton, 1997; 171–181.
19. Adams, J.F. Yield responses to molybdenum by field and horticultural crops. In *Molybdenum in Agriculture*; Gupta, U.C., Ed.; CRC Press: Boca Raton, 1997; 182–201.

Brick Making: Soil Degradation

G.S. Dheri

Babu S. Brar

Debankur Sanyal

Department of Soil Science, Punjab Agricultural University, Ludhiana, India

Abstract

Bricks, a primary tool of urbanization, are made of soils. In the age of urbanization, a large amount of bricks are needed, which in turn destroy agricultural soils. Due to desurfacing, the most fertile portion of a soil profile is getting degraded. Most importantly, soil organic matter along with primary nutrients, such as nitrogen, phosphorus, and potassium, is lost from the soil, destroying soil microbial activities and depleting soil fertility as well as productivity. Hence, this important but ignored aspect of land degradation needs comprehensive studies to understand the extent of problem and remedial measures. In this entry, economic cost of nutrients lost due to desurfacing is estimated. Some management strategies for rehabilitation of desurfaced soils have also come up from few studies. Further studies will reveal more unknown facts about soil degradation by brick making and its management.

INTRODUCTION

Bricks, the cells of urban society, are made of a very common material, soil. Land, a gift of nature, we use it to grow foods and also modern, developed society. In this 21st century, we need more and more bricks to build more and more houses, in order to shelter the expanding population. But we have neglected a crucial part here. We are using a large amount of fertile agricultural soil to make bricks, degrading our lands, and thus converting fertile land to barren. Few studies have also been done on this aspect, several projects have also been started, but this alarming situation is quite neglected. A time has come to take a look at the scenario.

Ministry of Food and Agriculture, Government of India (1962) defined the term “land degradation” as “the lands available for cultivation but not taken up for cultivation or abandoned after a few years for some reason or other.” Land degradation can be considered in terms of the loss of actual or potential productivity or utility as a result of natural or anthropogenic factors.^[1] The widespread existence of brick kilns in the rural–urban fringes of the third world cities represents one of such important centers of anthropic activities with a potential to alter the ecology of the surrounding vegetation and soil.^[2] From 1958 to 2009, this has led to a decrease of agricultural area in the core zone of the cities by 60%.^[3] Accordingly, demand for construction materials, such as bricks, has increased as their production provides an employment opportunity for the urban poor.^[4] In India, bricks are usually made of clay and are generally produced in traditional, unorganized small-scale industries. Brick industries are essential for economic development, as they provide employment to nearly 12 million people.^[5]

Millions of hectares of lands and tons of carbon, nitrogen (N), phosphorus (P), and microbes are being losing every year through brick making, which not only destroying the agriculturally most important topsoils but also hindering microbially mediated essential soil processes and inducing the global warming and greenhouse effects.^[6]

DETRIMENTAL EFFECTS ON SOIL PROPERTIES AND NUTRIENTS

After the removal of surface soil, a very low-quality soil, characterized by high strength, very low permeability, and poor fertility, remains. Common crops, even weeds grow poorly on this soil. Degradation affects the productivity^[7] and this may be assessed in a number of ways. First, nutrients are lost in sediment and runoff. Generally, nutrients associated with organic matter, i.e., N, P, and the cation exchange of soil colloids, such as potassium (K) and calcium, are most at risk.^[8] Secondly, as soil degrades, its infiltration capacity worsens and consequently, there is drastic change in depth to water level. Few selected herb and shrub species can grow on burnt soil due to brick industry.^[9] Removal of plants’ cover and litter exposes soil to increased kinetic energy of rain drops, causing soil movement as well as increased erosion.

The greatest loss is undoubtedly removal of nutrients, as most of the available nutrients and organic carbon remains at the surface. In a study, it has been shown that brick kiln dust is alkaline with pH ranging from 8.2 to 10.5^[10] which in turn makes the neighboring soil pH higher. Desurfacing decreased organic carbon from 0.47% to 0.31%. Bulk

density values for 0–15 cm and 15–30 cm depths in desurfaced soil were greater than adjoining normal soil whereas available P and K were decreased.^[11] In a different study, it has been reported that the average loss of valuable soil components due to burning of the agricultural topsoils amounted to 63% for soil organic matter, 56–86% for total N, and 23–88% for total P, K, and S, respectively.^[12] The content of soil microbial biomass carbon was several fold higher in the unburnt (agricultural land) soil profiles than those of the burnt (soil around brick kilns) profiles, regardless of regions, climate and plant types or vegetation.^[6] According to a report by Grewal and Kuhad,^[11] due to desurfacing, the pH increased from 8.1 to 8.4, electrical conductivity decreased from 0.79 to 0.15 dSm⁻¹, organic carbon from 0.48% to 0.34%, available P from 3.56 to 1.78 kg ha⁻¹, and available K from 470 to 320 kg ha⁻¹.

Desurfacing of soil reduced available water holding capacity^[13] and deteriorated aggregate stability as compared with normal soil which may be attributed to lower soil organic matter, reduction of total porosity, and poor soil structure.^[14] Crop grown on desurfaced soils had small plant height, plant population, and yield.^[11,15–17] Additional application of fertilizers to compensate the fertility differences between normal and degraded soils was not able to sustained yield loss.^[17] Field adjacent to desurfaced soils remained at high risk of erosion during rainy season and irrigation.^[18]

GLOBAL SCENARIO OF BRICK MAKING

A study by Narayan and Gupta^[2] revealed that a huge chunk of agriculturally fertile lands are consumed by each brick kiln (3–4 ha), which get converted into wastelands as a result of industrial operation in the life of a brick kiln (8–12 yr). There are approximately 1,00,000 large-scale kilns with fixed chimneys, ca. 1,900 operating in India and 6,000 in Bangladesh. Latin America also has a considerable number of brick production facilities, including 6,898 in Brazil, 300 in Chile, 2,453 in Colombia, 17,000 in Mexico, and 2,222 in Peru.^[19] About 1,500 billion bricks are produced every year, 1 billion of those are produced in China and altogether Asia generates almost 90% of global production.^[13]

ECONOMIC IMPACTS

Economic impacts of soil degradation through brick making can be estimated using two approaches, that is, replacement cost approach (RCM) and productivity changes approach (PCA). The estimated economic costs of desurfacing the soil for brick making using PCA were higher compared to RCM, due to the ignorance of PCA to the loss of certain unquantifiable, qualitative properties of topsoil which lead to yield loss.^[20] Francisco and Angeles^[21] developed relationship between corn yield and soil loss ($Y_t = 1,360 - 12.5 \times \text{soil loss}$) and estimated that

Table 1 Annual loss of nutrients in brick making.

Nutrients ^a	Loss per brick (g)	Total amount lost (Tera g)	Total monetary loss (approximately)	
			Rs. ^a (billion)	\$ ^a (billion)
N	15–45	22.5–67.5	264.15–792.45	4.40–13.21
P	0.3–6	0.45–9	52–1040	0.86–7.18
K	0.6–6	0.9–9	30.13–301.32	0.50–5.04

^aThe the source of N is urea (46% N) at Rs. 541 per quintal, source of P₂O₅ is single super phosphate (16% P₂O₅) at Rs. 800 per quintal and the source of K₂O is muriate of potash (60% K₂O) at Rs. 1670 per quintal and it is assumed that 1\$ = Rs. 60.

corn yield decline equivalent to 12.5 kg ha⁻¹ with a ton of soil loss.

It was found that for making one brick, approximately 3 kg of clay is used and production of 47 bricks leads to degradation of 4 ft² land.^[22] So, if it is assumed that an agricultural soil contains about 0.5% organic carbon on an average, then, about 22.5 tera g (1 tera = 10¹²) of organic carbon is being lost from soil due to brick making annually as per reports mentioned. In general, for a normal fertile soil, the values of N, P, and K are in the range of 0.5–1.5%, 0.01–0.2%, and 0.02–0.2%,^[23] respectively, and the calculated loss of nutrients in amount and monetary terms are presented in Table 1.

MANAGEMENT STRATEGIES

Soil desurfacing to make bricks is increasing exponentially posing a threat to soil protection and soil productivity. Rapid restoration of soil productivity and management of desurfaced soils are of great concern but generally neglected. Desurfacing makes the soil unsuitable for crop production as subsoil horizons that are generally rich in calcium carbonate, high in bulk density, and poor in organic carbon, available nutrients, biological activity, and available water holding capacity are exposed to the surface.^[11] The first step in designing proper management techniques is to point out the dominant factors that limit crop production. The renovation of soil structure through organic amendments and agronomic practices or their integrated use is considered to be useful for the amelioration of desurfaced soils. Because available nutrient status is poor in desurfaced soils, use of higher doses (about 25% to 50% more) of fertilizers and water than that of normal soils for better crop growth can be beneficial. Alternative uses of desurfaced soils for social forestry, soil waste disposal, farm ponds, etc., need to be explored.^[11] Desurfaced soils can be effectively revived by using organic manures (poultry manure, animal waste, press mud, farmyard manure, and vermicompost); incorporation of crop residue and biochar; adopting green manuring and leguminous crops in crop rotation, deep tillage, gypsum application; using bio-fertilizers,

social forestry, fishery ponds, balance fertilization and extra 25–50% supplementary doses of fertilizers.^[24] Remote sensing data coupled with analytical and field-based data are useful in proper planning for management of degraded soils. Interpretation of satellite imagery recognizes brick kilns spatially that open way for management using other data toward sustainable development in the area.^[9] Such interdisciplinary approach for management of degraded soil could be useful for any area.

CONCLUSION

One thing is clear from the discussion, scientists have to find a way to defend this enormous destruction of fertile agricultural lands for making bricks. To support the ever-expanding population pressure, we are in need of more and more lands to produce foods and we cannot afford this huge loss of soil as well as nutrients. This aspect of land degradation is neglected throughout the world. Only limited studies have been conducted in this aspect. A lot of research is needed to find out better cropping systems, cultural practices, fertilizer managements, and agronomic practices in order to stepwise rehabilitate the degraded agricultural soils.

REFERENCES

1. Eswaran, H. Recommendation in the proceedings of the 2nd international conference on land degradation, Khon Kaen, Thailand, Jan 25–29, 1999; 9 pp.
2. Narayan, R.; Gupta, S. Brick kiln industry in long-term impacts biomass and diversity structure of plant communities. *Current Sci. India* **2010**, *99* (1), 72–79.
3. Schumacher, J.; Luedeling, E.; Gebauer, J.; Saied, A.; El-Siddig, K.; Buerkert, A. Spatial expansion and water requirements of urban agriculture in Khartoum, Sudan. *J. Arid. Environ.* **2009**, *73*, 399–406.
4. Aubry, C.; Ba, A.; Dabat, M.H.; Ramamonjisoa, J. Urban agriculture and sustainable urban landscape. An applied research on two case studies (Madagascar and Senegal). In *99th European IFSA Symposium*, Vienna, Austria, July 4–7, 2010.
5. Budhwar, R.; Bihari, V.; Mathur, N.; Srivastava, A.; Kumar, S. DNA protein crosslinks as a biomarker of exposure to solar radiation: A preliminary study brick-kiln workers. *Biomarkers* **2003**, *8* (2), 162–166.
6. Khan, H.R.; Rahman, K.; Rouf, A.J.M.A.; Sattar, G.S.; Akhtar, M.S.; Oki, Y.; Adachi, T. Effects of brick burning on microbial biomass and C/N ratio in selected soil profiles in the eastern region of Bangladesh. *J. Fac. Environ. Sci. Tech.* **2006**, *12* (1), 83–89.
7. Lutz, E.; Pagiola, S.; Reiche, C. The cost and benefits of soil conservation: The farmer's viewpoint. *World Bank Res. Obser.* **1994**, *9* (2), 273–295.
8. Stocking, M. Soil erosion and land degradation. In *Environmental Science for Environmental Management*; Timothy, O'Riordan, T. Ed.; Longman Scientific & Technical Publications: Singapore, 1995.
9. Yadav, S.K. Remote sensing based management of degraded soil due to brick industry for sustainable development—a case study. *J. Hum. Ecol.* **2003**, *14* (6), 451–455.
10. Upadhyay, E. *Impact of Particulate Air Pollutants on Brassica Juncea L. and Linum Usitatissimum L.*, Ph.D. Thesis; Aligarh Muslim University: Aligarh, India, 2004; 130 pp.
11. Grewal, M.S.; Kuhad, M.S. Soil desurfacing—impact on productivity and its management. In *Proceedings of the 12th ISCO Conference*, Tsinghua University Press, Beijing, May 26–31, 2002.
12. Khan, H.R.; Rahman, K.; Rouf, A.J.M.A.; Sattar, G.S.; Oki, Y.; Adachi, T. Assessment of degradation of agricultural soils arising from brick burning in selected soil profiles. *Int. J. Environ. Sci. Te.* **2007**, *4* (4), 471–480.
13. Sadler, J.M. Effects of topsoil loss and intensive cropping on soil properties related to the crop production potential of a Podzolic Grey Luvisol. *Can. J. Soil Sci.* **1984**, *64*, 533–543.
14. Gollany, H.; Schumacher, T.E. Aggregate stability of an eroded and desurfaced typic argiustoll. *Soil Sci. Soc. Am. J.* **1991**, *55*, 811–816.
15. Carter, D.L.; Berg, R.D.; Sanders, B.J. The effect of furrow irrigation erosion on crop productivity. *Soil. Sci. Soc. Am. J.* **1985**, *49*, 207–211.
16. Pettry, D.E.; Wood, C.W.; Soileau, J.M. Effect of topsoil thickness and horizonation of a virgin coastal plain soil on soybean yields. In *Natl Symp Erosion and Soil Productivity*, New Orleans, LA, Dec 10–11, 1984; St. Joseph, MI: American Society of Agricultural Engineers; 75–82.
17. Gollany, H.; Schumacher, T.E. Topsoil depth and desurfacing effects on properties and productivity of a typic argiustoll. *Soil Sci. Soc. Am. J.* **1992**, *56*, 220–225.
18. Das, R. Causes and consequences of land degradation in and around the brick kilns of khejuri CD blocks over Coastal Medinipur in West Bengal. *Int. J. Innov. Res. Dev.* **2015**, *4* (2), 184–194.
19. Climate and Clean Air Coalition, Mitigating black carbon and other pollutants from brick production. Available at www.icimod.org/resource/16874.
20. Bansal, S.; Kathuria, V.; Balasubramanian, R. Environmental cost of using top-soil for brick-making a case study from Tamil Nadu, India. *Rev. Market Ins.* **2013**, *5* (2), 171–201.
21. Francisco, H.A.; de Los Angeles, M.S. *Soil Resource Depreciation and Deforestation: Philippine Case Study in Resource Accounting; Planning and Statistics Branch, Policy and Planning Division, Forestry Department; Food and Agriculture Organization: Rome*, 1998.
22. Vyas, H.; Vyas, A.; Khan, R. A study of land degradation due to brick industries in Ujjain (India). *J. Environ. Res. Dev.* **2009**, *3* (4), 1174–1177.
23. Allen, S.E. *Chemical Analysis of Ecological Materials*; Blackwell Scientific Publications: Oxford, 1989.
24. Priyanka, Singh; Devi, R.; Hooda, R.S. Dynamics of soil desurfacing due to brick kilns and suggestive management techniques. *Int. J. Multidiscip. Res. Dev.* **2015**, *2* (4), 292–298.

Bulk Density

Kwong Yin Chan

Organic Wastes Recycling Unit, New South Wales Department of Primary Industries, Richmond, New South Wales, Australia

Abstract

Bulk density is a key soil physical property commonly used as an indicator of soil compactness, soil structure, and aeration status. It is also required for converting soil water content and nutrient values from the gravimetric to the volumetric basis. Depending on soil types, conditions of sampling, and degree of accuracy required, there are direct and indirect methods for the determination of bulk density.

INTRODUCTION

Bulk density (ρ_b) is defined as the ratio of the mass of a given soil sample (M) to its bulk volume (V) as follows:

$$\rho_b = \frac{M}{V} \quad (1)$$

The mass is obtained by drying the sample to a constant weight at 105°C, and the bulk volume is that of soil particles plus pore space at the time of sampling.^[1] The commonly used SI unit for bulk density is Mg/m³, which is numerically the same as g/cm³. Bulk density is a key soil physical property and is required for: 1) determining the degree of compactness and hence as a measure of soil structure; 2) using as an indicator of soil aeration status (together with soil water content); and 3) converting soil water content and nutrient values from the gravimetric to the volumetric basis.

Bulk density of soil depends on the composition as well as the structural conditions. It therefore tends to vary with the soil texture, which determines the packing of the soil particles. Sandy soils tend to have higher bulk density (ranged 1.4–1.9 Mg/m³) than clays (0.9–1.4 Mg/m³; Table 1). Volcanic soils and peaty soils tend to have bulk density <1.0 Mg/m³ because of the high organic matter levels and special clay minerals for the former and higher organic matter levels for the latter. Furthermore, as field soils are not just mixtures of primary particles in different degrees of packing, bulk density is also modified by soil aggregation (soil structure) and soil organic matter levels. Organic matter tends to reduce the degree of compaction. For 58 samples of surface soils in Scotland, a highly significant effect of organic matter is represented statistically by the following relationship:

$$\rho_{\max} = 1.86 - 0.055 \times \text{OM} \quad (2)$$

where $\rho_{\max}$ is the maximum bulk density over a range of 1.3–1.8 Mg/m³ and OM is the content of organic matter over a range of 2–10% that was obtained.^[3]

SOME RELATED TERMS

1. Wet bulk density (ρ_w) is the wet mass of soil per unit volume and is related to ρ_b as follows:

$$\rho_w = \rho_b(1 + \theta_g) \quad (3)$$

where θ_g is gravimetric water content, kg/kg soil.

- i. Porosity (ε) is the total pore space per unit volume of soil, m³/m³, and is related to ρ_b as follows:

$$\varepsilon = 1 - \frac{\rho_b}{\rho_s} \quad (4)$$

where ρ_s is soil particle density, Mg/m³.

- ii. Air-filled porosity (ε_a) is the air-filled pore space per unit volume of soil, m³/m³, and is related to ρ_b as follows:

$$\varepsilon_a = \varepsilon - \theta_v = 1 - \frac{\rho_b}{\rho_s} - \rho_b \theta_g \quad (5)$$

where θ_v is volumetric water content, m³/m³.

METHODS FOR DETERMINATION

Methods commonly used for bulk density determination can be grouped into the following two categories.

Direct Methods

Direct methods involve measurement of the sample mass and volume, following Eq. 1. The most widely used procedure is the “core method” in which a cylindrical sampler is hammered into the soil.^[4,5] As the volume of the cylinder is known, trimming of the soil core flush with the ends of the cylinder and drying of the soil at 105°C for determining its weight allows the bulk density to be calculated. Accuracy of the core method is affected by compression and

Table 1 Range of bulk density and porosity values in different soils and particle mixtures.

Soil/particles mix	Bulk density (Mg/m ³)	Porosity (%)
Surface soil of wet clay	1.12	0.58
Surface soil of loam	1.28	0.52
Sandy subsoil	1.61	0.39
Sandy loam compacted by heavy traffic	1.90	0.28
Spheres of uniform size in open packing	1.39	0.48
Spheres of uniform size in close packing	1.96	0.26
Sandstone	2.12	0.20
Peaty soil	0.3	—
Volcanic soil	<0.9	—

Note: For swelling soils, bulk density is a function of soil water content.

^aParticle density (ρ_s) is taken as 2.65 Mg/m³.

Source: Adapted from Marshall & Holmes.^[2]

shattering, which are dependent on sample size and soil water content.^[4,5] Samplers with too small diameter are unsuitable because of the risk of compression. A thin-walled sampler with the diameter of 75–100 mm has been proposed as a compromise.^[4] The core method works best in soft cohesive soils sampled at water contents in the region of field capacity. The method is not suitable for non-cohesive soils, i.e., sands, gravels, and gravelly soils. In the latter cases, excavation methods that involve digging a burrow in the soil and determining the volume of the hole (using rubber balloon or sand displacement methods) and the oven-dried weight of the excavated soil are preferred.^[5,6] For the sand displacement method, the volume of the hole is determined by filling it with sand from a precalibrated sand funnel.^[1]

For soil samples of irregular shape, e.g., clods and aggregates, bulk density can be measured using the “clod method” which involves coating with paraffin wax or resin (e.g., Saran resin) and liquid displacement for the determination of volume.^[4,5] Saran resin coating has been used for the determination of bulk density changes of swelling soils over a range of soil water contents on a single clod.^[7] As the resin has the unique property of being permeable to only water vapor molecules but not liquid water molecules, it allows the determination of shrinkage curve on one single clod, thus greatly reducing sampling variability.

With proper precautions taken, the field determination of bulk density using the core method has a relatively low coefficient of variation (<10%) for uniform soils.^[8] Therefore, about four samples should be sufficient to estimate the mean bulk density to within 10% of the true value, 95% of the time.

Radiation Methods

Radiation methods are based on the empirical relationships between radiation and soil wet bulk density. The most common methods involve the attenuation and scattering of gamma radiation by soil, both of which increase with bulk density.^[5] Because of the interaction of gamma radiation with water, simultaneous measurement of soil water content is required for the calculation of bulk density using Eq. 3. Radiation methods are potentially at least equal in accuracy to any of the direct methods^[5,9] and are simpler and quicker to use especially where measurements at depth are required. It has the added advantage of being non-destructive and therefore allows repeated measurements at the same location; however, the equipment is more expensive and involves safety issues associated with the use of radioactive radiation.

APPLICATION OF BULK DENSITY MEASUREMENTS

As Indicator of Soil Structure and Soil Quality

Bulk density has been included in a minimum data set for monitoring soil quality, as an indicator of both soil structure and soil strength.^[10] Changes in bulk density reflect changes in soil structure because of the relationship between bulk density and total porosity (Eq. 4). However, total porosity gives no indication of the pore size distribution and pore continuity, which are important attributes of soil structure and associated functions.^[11] Nevertheless, with the exception of swelling soils, soil compaction is often measured by the increases in bulk density. As bulk density of swelling soils changes with soil water content, comparison is only valid at the same soil water content.

Bulk density affects plant growth because of its effect on soil strength and soil porosity. With increasing bulk density, strength tends to increase and porosity tends to decrease; both tend to be limiting to root growth at some critical values. Critical bulk density for root growth was found to vary with different textures; e.g., for sunflower, the critical value was 1.75 Mg/m³ for sandy soil and 1.46–1.63 Mg/m³ for clays.^[12,13]

To account for the texture difference so that crop yield can be related to soil compaction (bulk density) over a whole range of soils, the concept of degree of compactness has been proposed.^[14] The latter refers to the ratio of the bulk density of the soil in the field to that of the same soil in a compacted reference state of 200 kPa. As an average of 100 field experiments on a range of soil types (%clay = 2–60%), the optimal degree of compactness for grain yield was 0.87 above which adverse effects start to occur.

As a Measure of Aeration Status

Together with gravimetric water content and using Eq. 5, air-filled porosity (ϵ_a) can be calculated. ϵ_a values ranging between 0.05 and 0.15 (m^3/m^3) are commonly used as the critical limit below which aeration is limiting to root growth.^[15]

Conversion of Gravimetric Data to Volumetric Data

Laboratory analyses for soil organic matter, nitrogen contents, water contents, and so on are commonly expressed on a gravimetric or weight basis; however, for comparison in the field in terms of management practices, the amount per volume of soil or per area is preferred. For the conversion, soil bulk density, which frequently varies with management, depth of sampling, and time of the year, is needed. For example, to calculate the mass of organic carbon (OC) in t/ha to 0.10 m depth from gravimetric OC (g/kg), the following equation can be used:

$$\text{OC(t/ha)} = \text{OC(g/kg)} \times \rho_b(\text{Mg/m}^3) \quad (6)$$

The inclusion of bulk density in a set of basic soil indicators for the proper interpretation of the change in magnitude in chemical and biochemical soil components has been proposed.^[10]

Estimate of Mass of Large Volume of Soil

To estimate total mass of soil in 1 ha of land to 0.10 m depth, knowledge of soil bulk density is needed

$$\text{Mass(t/ha)} = \rho_b(\text{Mg/m}^3) \times 1000 \quad (7)$$

CONCLUSION

Bulk density is a key soil physical property commonly used as indicator of soil compactness, soil structure, and aeration status. It is also required for converting soil water content and nutrient values from the gravimetric to the volumetric basis. Depending on soil types, conditions of sampling, and degree of accuracy required, there are direct and indirect methods for the determination of bulk density.

REFERENCES

1. Blake, G.R.; Hartge, K.H. Particle density. In *Methods of Soil Analysis Part I*; Klute, A., Ed.; Soil Science Society of America: Madison, 1986; 363–375.
2. Marshall, T.J.; Holmes, J.W. Composition of soil. In *Soil Physics*; Cambridge University Press: Cambridge, 1979; 10.
3. Soane, B.D. Studies on some physical properties in relation to cultivation and traffic. In *Soil Physical Conditions and Crop Production*; Tech. Bull. No. 29; Min. Agr. Fish and Food: London, 1975; 115–126.
4. McIntyre, D.S.; Loveday, J. Bulk density. In *Methods for Analysis of Irrigated Soils*; Loveday, J., Ed.; Commonwealth Agricultural Bureaux: Farnham Royal, 1974; 38–42.
5. Campbell, D.J.; Henshall, J.K. Bulk density. In *Soil Analysis—Physical Methods*; Smith, K.A., Mullins, C.E., Eds.; Marcel Dekker: New York, 1991; 329–366.
6. Freitag, D.R. Methods of measuring soil compaction. In *Compaction of Agricultural Soils*; Barnes, K.K., Carleton, W.W., Taylor, H.R., Throckmorton, R.I., Vanden Ber, G.E., Eds.; American Society of Agricultural Engineers: St Joseph, 1971; 47–103.
7. Chan, K.Y. Shrinkage characteristics of soil clods from a grey clay under intensive cultivation. *Aust. J. Soil Res.* **1982**, *20*, 65–74.
8. Warrick, A.W.; Neilson, D.R. Spatial variability of soil physical properties in the field. In *Application of Soil Physics*; Hillel, D., Ed.; Academic Press: New York, 1980; 319–344.
9. Soane, B.D.; Campbell, D.J.; Herkes, S.M. Hand held gamma-ray transmission equipment for the measurement of bulk density of field soils. *J. Agr. Eng. Res.* **1971**, *16*, 146–156.
10. Doran, J.W.; Parkin, T.B. Defining and assessing soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., Eds.; SSSA: Madison, 1994; 3–21.
11. Kay, B.D. Rates of change of soil structure under different cropping systems. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer-Verlag: Heidelberg, 1990; Vol. 12, 1–52.
12. Veihmeyer, F.J.; Hendrickson, A.H. Soil density and root penetration. *Soil Sci.* **1948**, *65*, 487–493.
13. Jones, C.A. Effect of soil texture on critical bulk density for root growth. *Soil Sci. Soc. Am. J.* **1983**, *47*, 1208–1211.
14. Hakansson, I. A method for characterizing the state of compactness of the plough layer. *Soil Till. Res.* **1990**, *16*, 105–120.
15. Stepniewski, W.; Glinski, J.; Ball, B.C. Effects of soil compaction on soil aeration properties. In *Soil Compaction in Crop Production*; Soane, B.D., van Ouwerkerk, C., Eds.; Elsevier Science B.V.: The Netherlands, 1994; 167–189.

Bunch Planting: Water Conservation

B.A. Stewart

Department of Agricultural Sciences, West Texas A&M University, Canyon, Texas, U.S.A.

Abstract

Stored soil water and growing season precipitation generally support early season growth of grain sorghum (*Sorghum bicolor* L. Moench) in dryland areas, but severe water stress is the norm during the later growth stages. Consequently, grain yield is generally low. Further studies have shown that growing grain sorghum in clumps rather than being equally spaced reduces tillering and early season vegetative growth, thereby conserving more soil water for use during the later growth stages. This results in an increase in the harvest index and a higher yield of grain. The architecture of clumped plants is also notably different from uniformly spaced plants. The leaves of both the tillers and main stalk grow outward, exposing essentially all of the leaf area to sunlight and wind. In contrast, clumped plants grow upward with the leaves partially shading one another and reducing the effect of wind, thereby reducing water use.

INTRODUCTION

Significant gains have been made in storing more precipitation as soil water during fallow periods and increasing the use efficiency of growing season precipitation by leaving crop residues on the soil surface.^[1] The fact remains, however, that dryland crop production in many semiarid regions are severely constrained nearly every year by scarcity of growing season precipitation. Water becomes particularly lacking during the later growth stages for grain crops such as wheat and grain sorghum. The later stages are the reproductive and grain filling, and during these stages, yield and quality are drastically reduced without water. Craufurd et al.^[2] reported that water stress during the stages of booting and flowering resulted in grain yield reduction of up to 85% for grain sorghum.

Crops are often grown under hostile climatic conditions where water limits yield every year. In semiarid regions such as the U.S. Southern High Plains, average precipitation for every month of the year is less than one-half of the potential evapotranspiration. Therefore, growing dryland crops such as grain sorghum (*Sorghum bicolor* L. Moench) requires that a significant quantity of plant-available water is stored in the soil profile at the time of seeding to supplement the growing season precipitation. Grain sorghum is generally seeded in these areas during the time of year when the probability of precipitation is high so that plant establishment can be achieved. Thus, with the precipitation during stand establishment and the stored plant, available water in the soil profile, plant-growing conditions during the early growth stages are generally favorable. However, water stress during the reproduction and grain-filling stages occurs practically every year and results in low grain yields. Many farmers use low plant populations as a strategy for using less of the stored soil water during the early growth

stages so that more will be available during the later stages. This strategy is often not effective because widely spaced grain sorghum plants commonly produce one to several tillers that utilize water and nutrients early in the season but produce little or no grain, particularly in years when growing season precipitation is low. Therefore, most or all of the expected benefits of using low plant population are often negated.

Stewart^[3] and Bandaru et al.^[4] conducted studies to determine if growing plants in bunches, or clumps, would affect early season growth and subsequent grain yield when compared with uniformly spaced plants. They hypothesized that growing grain sorghum plants in clumps would result in fewer tillers and less vegetative growth so that more soil water would be available during the grain-filling period.

EXPERIMENTAL FINDINGS

A summary of the methods and results reported by Bandaru et al.^[4] relating to the hypothesis stated above is presented here. The treatments used in the 2004 study are shown in Fig. 1. The primary focus of the treatments was to determine how growing plants in clumps would affect tillering and water use during the season, and there was also a treatment where the tillers were removed by hand to provide information about whether plants growing in clumps with few or no tillers when compared with equally spaced plants with or without tillers. Similar studies were conducted for three different slope positions—upper, middle, and bench position. Runoff from the upper position ran across the middle position, and runoff from the middle position was captured on the bench position. Therefore, there were different water regimes for the three positions,

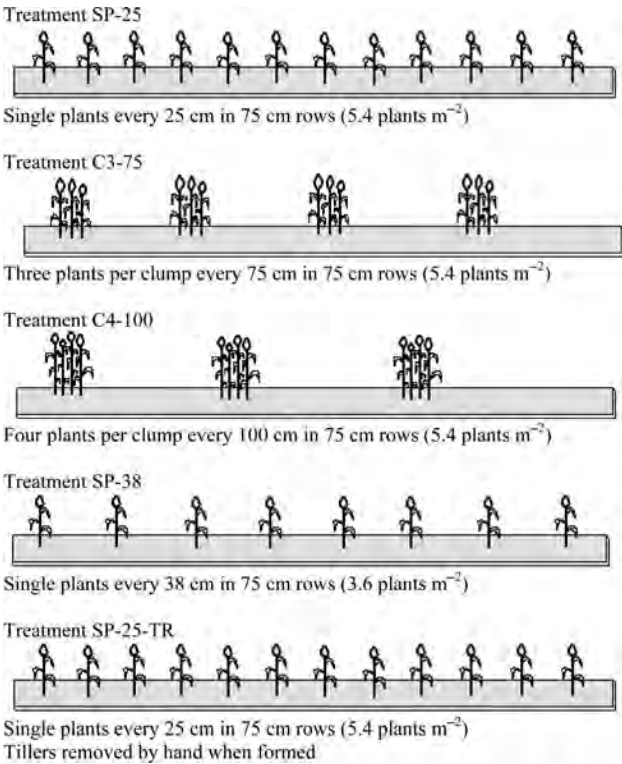


Fig. 1 Schemes showing plant geometries for the treatments used in the 2004 experiments at Bushland, Texas, and Tribune, Kansas.

but they were not controlled. The upper position was the driest, the bench position was the wettest, and the middle position somewhat in between them.

The results reported in Table 1 show clearly that plants growing in clumps produced significantly fewer tillers than equally spaced plants. Tillering is a complex phenomenon, and it is not clear why clumping reduced the number of tillers so dramatically. Studies have shown that the production of tillers is reduced, as the ratio of red to far red light is decreased.^[5–7] In Bandaru et al.’s^[4] study, clumped plants did not receive as much light at the base as individually spaced plants. Also, when there were four plants per clump, C4-100, there were fewer tillers produced than when there were three plants per clump, C3-75 (Table 1). This suggests that the amount of light at the base decreases as the number of plants in a clump increases.

Aboveground biomass and leaf area produced during the initial 42 days of growth were closely related to tiller production (Table 1). Treatment SP-25 produced ~75% more dry matter and leaf area than the C4-100 treatment. The SP-25-TR treatment that had the tillers removed as they were formed to produce essentially the same amounts of dry matter and leaf area as the C4-100 treatment, supporting the hypothesis that the increased dry matter and leaf area for the SP-25 treatment were the results of more tillers.

Table 1 Mean values for measurements of grain sorghum as affected by five planting geometries in 75-cm rows in 2005 Bushland, Texas, study on upper, middle, and bench positions of a stubble-mulched area.

Planting geometry ^a	LAI 42		Panicles (m ⁻²)	Grain (kg ha ⁻¹)	Harvest index
	Tillers (plant ⁻¹ 28 DAP)	DAP (kg ha ⁻¹)			
Upper					
SP-25	2.3A ^b	1.31A	8.1A	2385C	0.28C
C3-75	0.7B	1.01B	7.7A	2976B	0.36B
C4-100	0.3B	0.87C	6.2B	3563A	0.41A
SP-38	3.1A	1.30A	8.0A	2702BC	0.30C
SP-25-TR	Removed	0.85C	5.4C	2964B	0.39A
Middle					
SP-25	2.0A	1.41A	10.6A	3180C	0.34C
C3-75	1.1B	1.12C	9.5B	4013A	0.41AB
C4-100	0.5C	0.90D	8.1B	3952A	0.44A
SP-38	2.6A	1.37B	10.0A	3610AB	0.38B
SP-25-TR	Removed	0.88D	5.4C	3563BC	0.40B
Bench					
SP-25	2.3A	1.44A	12.0A	4743A	0.41B
C3-75	1.2B	1.18C	9.9BC	4902A	0.46A
C4-100	0.8C	0.93D	8.8C	4810A	0.46A
SP-38	2.9B	1.42B	10.8B	4911A	0.41B
SP-25-TR	Removed	0.91E	5.4C	4247B	0.42B

Notes: DAP, days after planting. Separate but identical experiments were conducted on three positions that had different amounts of stored soil water at the time of seeding and different amounts of runoff or run-on during the cropping season.

^aPlanting geometries were SP-25 (plants every 25 cm), C3-75 (clumps of three plants every 75 cm), C4-100 (clumps of four plants every 100 cm), SP-38 (plants every 38 cm), and SP-25-TR (plants every 25 cm with tillers removed by hand) in 75 cm rows.

^bMeans in columns for a position on the benched terrace followed by the same letter are not significantly different according to a protected LSD mean separation ($P < 0.5$ level); each position represents a separate experiment and cannot be compared statistically.

Source: Adapted from Bandaru, Stewart, et al.^[4]

Although there were large differences in the number of tillers for the various treatments (Table 1), the differences in the number of panicles produced were much smaller. This was because a large percentage of the tillers, particularly for the experiment located on the upper position of the benched terrace where water stress was more severe, did not produce panicles.

Grain yields were relatively high for experiments located on the bench position (Table 1) because this position received runoff from experiments located on other slope positions. There were no differences in yields between the clump treatments, C3-75 and C4-100, and the spaced plant treatments, SP-25 and SP-38, for experiments on the bench. However, the yield of the SP-25-TR

treatment that had tillers removed was reduced. This reduction was most likely caused by an insufficient number of panicles for the yield level achieved with the favorable water conditions.

The yield increase from clumped plants was primarily due to higher harvest indexes. This indicates that under water-stressed growing conditions, as is almost always the case in the semiarid regions like the U.S. Southern Great Plains, growing grain sorghum in clumps rather spaced uniformly reduces tillering and early season vegetative growth and conserves soil water until later in the season and may enhance grain yield. The benefit of clumps decreased as grain yields increased, and there was even a slight decrease when yields exceeded 6000 kg ha⁻¹. There was also a noted difference in the architecture of the plants. Uniformly spaced plants produced more tillers, and the leaves of both the tillers and main stalk grow outward, exposing essentially all of the leaf area to sunlight and wind. In contrast, clumped plants grow upward with the leaves partially shading one another and reducing the effect of wind, thereby reducing water use. Growing grain sorghum in clumps in semiarid areas as shown in Fig. 2 appears to be a useful strategy with little downside risk.



Fig. 2 Dryland grain sorghum growing in clumps at Bushland, Texas, in 2007.

CONCLUSION

Grain sorghum planted in clumps rather than spaced uniformly in rows in the semiarid dryland U.S. Southern Great Plains resulted in higher grain yields in years of lower than average precipitation. Tiller formation was reduced, and there was less early season vegetative growth conserving a greater portion of the stored soil water for use during the reproductive and grain-filling growth stages. This strategy may be applicable to similar dryland environments of the world.

REFERENCES

1. Schiere, H.B.; Baumhardt, R.L.; Van Keulen, H.; Whitbread, A.M.; Bruinsma, A.S.; Goodchild, A.V.; Gregorini, P.; Slingerland, M.A.; Hartwell, B. Mixed crop-livestock systems in semiarid regions. In *Dryland Agriculture*, American Society of Agronomy Monograph Series No. 23, 2nd Ed.; Peterson, G.A., Unger, P.W., Payne, W.A., Eds.; American Society of Agronomy: Madison, 2006; 227–291.
2. Craufurd, P.Q.; Flower, D.J.; Peacock, J.M. Effect of heat and drought stress on sorghum (*Sorghum bicolor*). I. Panicle development and leaf appearance. *Exp. Agric.* **1993**, *29*, 61–76.
3. Stewart, B.A. Growing dryland crops in clumps: What are the benefits? In *Proceedings of 28th Southern Conservation Systems Conference*, Amarillo, Texas, Jun 26–28, 2006; Schwartz, R.C., Baumhardt, R.L., Bell, J.M., Eds.; USDA-ARS Conservation and Production Research Laboratory: Bushland, 2006; 47–56.
4. Bandaru, V.; Stewart, B.A.; Baumhardt, R.L.; Ambati, S.; Robinson, C.A.; Schlegel, A. Growing dryland grain sorghum in clumps to reduce vegetative growth and increase yield. *Agron. J.* **2006**, *98*, 1109–1120.
5. Casal, J.J.; Deregibus, V.A.; Sanchez, R.A. Variations in tiller dynamics and morphology in *Lolium multiflorum* Lam. Vegetative and reproductive plants as affected by differences in red/far-red irradiation. *Ann. Bot. (London)* **1985**, *56*, 553–559.
6. Davis, M.H.; Simmons, S.R. Tillering response of barley to shifts in light quality caused by neighboring plants. *Crop Sci.* **1994**, *34*, 1604–1610.
7. Deregibus, V.A.; Sanchez, R.A.; Casal, J.J. Effects of light quality on tiller production in *Lolium* spp. *Plant Physiol.* **1983**, *72*, 900–902.

Calcification

Janis L. Boettinger

Utah State University, Logan, Utah, U.S.A.

Abstract

Calcification is the general process by which calcium carbonate (CaCO_3) accumulates in soils. Most commonly, CaCO_3 accumulates in subsurface horizons of soils in subhumid, semiarid, or arid regions. The accumulation of CaCO_3 in subsoil horizons is often coupled with its removal from overlying horizons, linked by the downward translocation of the soluble ions that precipitate to form CaCO_3 . However, the properties and processes that govern water flow in soils also affect calcification. Calcification can operate at a rate and duration such that subsoil horizons become plugged with CaCO_3 , effectively limiting penetration by plant roots and reducing the conductivity of gases, solutions, and suspended solids. The objective of this entry is to describe the environmental factors, components, mechanisms, and manifestations of calcification in soils.

COMPONENTS OF CALCIFICATION

For calcification to occur in soil, there must first be sources of calcium carbonate (CaCO_3) or the ions that precipitate to form CaCO_3 . Calcium (Ca^{2+}) is the cation required, and bicarbonate (HCO_3^-) or, less commonly, carbonate (CO_3^{2-}) are the required anions.

The most direct source of CaCO_3 for calcification is calcareous soil parent materials, which already contain CaCO_3 . Examples include limestone,^[1] calcareous shales and sandstones, and unconsolidated sediments derived from calcareous rocks (i.e., alluvium and lacustrine sediments). However, calcification can occur in soils that lack obvious calcareous parent materials. A likely external source of CaCO_3 in arid and semiarid regions is the eolian deposition of fine-grained calcareous sediments onto the soil surface ("calcareous dust").^[2,3] These calcareous eolian sediments originate from areas rich in calcareous sediments that can be easily entrained by the wind, such as playas or basins that contain dry lake beds.^[4]

Calcium-bearing, primary silicate minerals may also be a source for the Ca in the CaCO_3 that accumulates in soil.^[5] Silicates that can weather to provide a source of Ca for calcification include plagioclase feldspars (e.g., anorthite, $\text{CaAl}_2\text{Si}_2\text{O}_8$; oligoclase, $\text{Na}_{0.8}\text{Ca}_{0.2}\text{Al}_{1.2}\text{Si}_{2.8}\text{O}_8$; etc.) and Ca-bearing amphiboles (commonly hornblende) and pyroxenes.

Carbon dioxide (CO_2) gas is the ultimate source of bicarbonate and carbonate in soils. CO_2 , which attains a higher partial pressure in soils than in the atmosphere because of respiration by plant roots and heterotrophic microorganisms, dissolves in water to form carbonic acid (H_2CO_3):



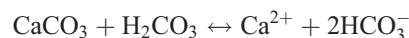
H_2CO_3 is a weak acid that dissociates to form bicarbonate (HCO_3^-) that dissociates very slightly to form carbonate (CO_3^{2-}):



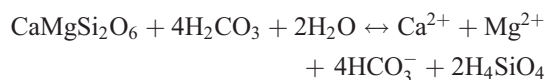
The dominance of these species in soils depends on the pH. Bicarbonate is the dominant species in solutions of most soils, whereas H_2CO_3 dominates only in strongly acid soils and carbonate dominates only in strongly alkaline soils.

MECHANISMS OF CALCIFICATION

The first step in calcification is the chemical weathering of minerals in upper soil horizons. In soils that contain calcareous parent materials or receive calcareous eolian deposits, CaCO_3 must first dissolve. Some authors refer to this process as "decalcification."^[6] Dissolution of CaCO_3 is accelerated by the presence of weak acids in soil solution because (CO_3^{2-}) associates readily with H^+ to form (HCO_3^-). H_2CO_3 is the most common solvent of CaCO_3 in natural systems.^[7]



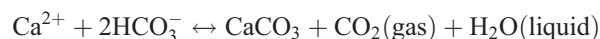
H_2CO_3 also accelerates the dissolution of primary silicate minerals in soils and other natural systems. For example, diopside, a Ca-bearing pyroxene ($\text{CaMgSi}_2\text{O}_6$), can undergo H_2CO_3 weathering in soil solution to produce Ca^{2+} and (HCO_3^-) as well as Mg^{2+} and silicic acid (H_4SiO_4):



Any process that increases CO_2 in the soil solution can increase the rate of H_2CO_3 weathering.

The second step in calcification involves the movement of Ca^{2+} and (HCO_3^-) ions in solution to the soil horizon(s) where the conditions allow the chemical precipitation of CaCO_3 . Because atmospheric precipitation is limited in arid and semiarid climates, products of mineral dissolution are not effectively leached from soils. Instead, Ca^{2+} and (HCO_3^-) generally move downward with percolating solutions to deeper soil horizons. However, soil properties and processes that affect the magnitude and direction of saturated or unsaturated water flow in soils will influence where CaCO_3 ultimately precipitates and accumulates. For example, the downward movement of soil solution may be inhibited by an underlying slowly permeable layer (e.g., clay-rich horizon or bedrock) or by a textural discontinuity (e.g., silt loam or loam overlying loamy sand or gravelly sand). Alternatively, the soil solution may move upward in response to a water potential gradient set up by abundant water in the subsoil, such as a seasonally high water table, and evapotranspiration of water from near-surface soil in arid, semiarid, or seasonally dry climates.

The third step in calcification involves the chemical precipitation of CaCO_3 ; this step is defined exclusively by some authors as the process of “calcification”.[6]



The removal of H_2O directly increases the concentration of Ca^{2+} and (HCO_3^-) in the soil solution and causes the precipitation of CaCO_3 . Decreasing the partial pressure of CO_2 will also enhance the precipitation of CaCO_3 . In the case of downward moving solution in arid and semiarid climates, deeper horizons of soils are generally drier and have lower partial pressures of CO_2 because of reduced biological activity. These conditions alone may favor the precipitation of CaCO_3 . However, the removal of H_2O by plant roots via transpiration directly causes the precipitation of CaCO_3 . Evapotranspiration of H_2O causes precipitation of CaCO_3 in soils in which the solution moves upward into drier horizons that have increasing concentrations of plant roots.

Soil microorganisms can play a direct role in the precipitation of pedogenic CaCO_3 .^[8] In soil horizons with Ca-rich solutions, some bacteria and fungi excrete excess Ca^{2+} . If the soil is sufficiently alkaline and moist, the CO_2 respired by heterotrophic microorganisms and plants dissolves and dissociates to form an abundance of (HCO_3^-) . Excreted Ca^{2+} combines with (HCO_3^-) in solution to precipitate CaCO_3 on the exterior surface of the microorganism.

The continued precipitation of CaCO_3 at about the same soil depth interval over hundreds or, perhaps, thousands of years results in the accumulation of CaCO_3 . CaCO_3 precipitates mainly in larger soil voids, which dry more quickly and have lower partial pressure of CO_2 than smaller micropores.^[9] CaCO_3 also acts as a template for further precipitation of CaCO_3 . The mineral form of pedogenic CaCO_3 is

almost exclusively calcite, including that which is microbially precipitated.^[8]

MANIFESTATIONS OF CALCIFICATION

The morphology of CaCO_3 concentrations is varied.^[2,10] Carbonates can be disseminated or finely dispersed throughout the soil matrix such that specific carbonate features are not visible. Carbonates can also be segregated into visible morphological features such as filaments, soft masses, concentrically layered concretions, and irregularly shaped nodules. Carbonates can occur as coatings and pendants (coatings restricted to the undersides) on sand grains and rock fragments.

The manifestations of accumulating CaCO_3 with increasing time are different in gravelly and non-gravelly soils.^[2] In gravelly soils, CaCO_3 first appears as thin, discontinuous coatings on rock fragments where water flows preferentially. With accumulation over time, coatings become continuous, followed by CaCO_3 filling in the interstices between rock fragments (Fig. 1). Eventually, CaCO_3 can plug the interstices and effectively cement the soil fabric, which reduces water, gas, and solute flow through the horizon. In non-gravelly soils, which possess smaller pores and a larger total pore volume, CaCO_3 is first disseminated, then appears as filaments and coatings on sand grains. Nodules of CaCO_3 form and increase in abundance and size with time. The soil fabric between the nodules eventually fills in with CaCO_3 , effectively plugging and cementing the horizon. At this stage, in both gravelly and nongravelly soils, the downward moving solution is restricted and CaCO_3 accumulates at the upper boundary of the plugged horizon, forming a very hard, laminated layer.

As mentioned before, the properties and processes that affect water flow in soils influence the patterns of CaCO_3 accumulation in soil. Similarly aged soils (ca. 13,000 years) formed in calcareous lacustrine sediments of Pleistocene pluvial Lake Bonneville in a seasonally dry climate exhibit different vertical distributions of CaCO_3 (Fig. 2).^[11] The different patterns of CaCO_3 accumulation are strongly influenced by different depths to a seasonally high water table. The dominant direction of water flow is downward in soils that are not influenced by a water table, and CaCO_3 removed from upper horizons accumulates near the average maximum depth of wetting (e.g., Parleys, Fig. 2). Much of the water movement is upward in soils that have a seasonal high water table near the soil surface, and relatively insoluble CaCO_3 accumulates at about the depth of the water table (e.g., Salt Lake, Fig. 2). Soils that experience downward percolation of water and have a seasonal water table present in the subsoil exhibit both removal of CaCO_3 from upper horizons and the precipitation and accumulation of CaCO_3 at about the depth that the downward moving solution and the water table meet (e.g., Nibley) (Fig. 2).



Fig. 1 Profile of a soil with CaCO_3 accumulation in the subsoil. The soil formed in the seasonally dry, semiarid climate in late Pleistocene-aged deltaic sediments derived from pluvial Lake Bonneville and alluvium. Carbonates occur as coatings and pendants on rock fragments and are disseminated between rock fragments. The CaCO_3 is concentrated below the textural discontinuity between alluvial gravelly loam and underlying deltaic very gravelly sandy loam and sand.

REFERENCES

1. Rabenhorst, M.C.; Wilding, L.P. Pedogenesis on the edwards plateau texas. I. nature and continuity of parent material. *Soil Sci. Soc. Am. J.* **1986**, *50* (3), 678–687.
2. Gile, L.H.; Petersen, F.F.; Grossman, R.B. Morphological and genetic sequences of carbonate accumulation in desert soils. *Soil Sci.* **1966**, *101* (5), 347–360.
3. Reheis, M.C.; Goodmacher, J.C.; Harden, J.W.; McFadden, L.D.; Rockwell, T.K.; Shroba, R.R.; Sowers, J.M.; Taylor, E.M. Quaternary soils and dust deposition in southern Nevada and California. *Geol. Soc. Am. Bull.* **1995**, *107* (9), 1003–1022.
4. McFadden, L.D.; Wells, S.G.; Dohrenwend, J.C. Influences of quaternary climatic change on processes of soil

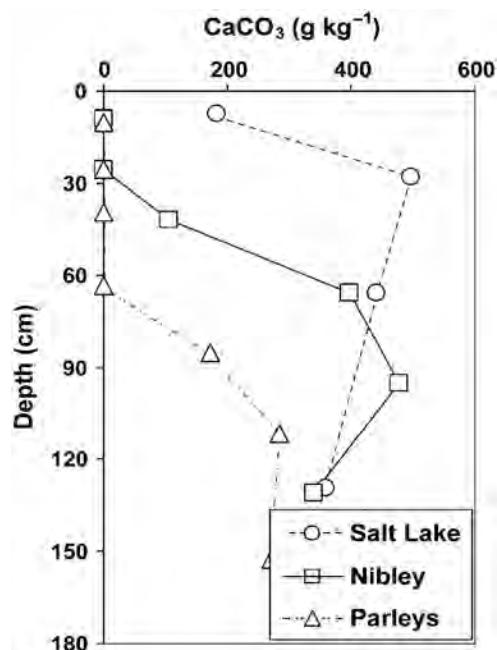


Fig. 2 Vertical distribution of CaCO_3 in three soils that have different depths to a seasonally high water table. The seasonal high water table is near the surface in the soil of the Salt Lake series and is between 75 and 100 cm in the soil of the Nibley series. The soil of the Parleys series is unaffected by a water table. **Source:** From Erickson & Mortensen.^[11]

development on desert loess deposits of the cima volcanic field, California. *Catena* **1986**, *13* (4), 361–389.

5. Boettinger, J.L.; Southard, R.J. Silicate and carbonate sources for aridisols on a granitic pediment, western mojave desert. *Soil Sci. Soc. Am. J.* **1991**, *55* (4), 1057–1067.
6. Buol, S.W.; Hole, F.D.; McCracken, R.J.; Southard, R.J. *Soil Genesis and Classification*, 4th Ed.; Iowa State University Press: Ames, 1997; 527 pp.
7. Krauskopf, K.B. *Introduction to Geochemistry*, 2nd Ed.; McGraw-Hill: New York, 1979; 617 pp.
8. Monger, H.C.; Daugherty, L.A.; Linderman, W.C.; Liddell, C.M. Microbial precipitation of pedogenic calcite. *Geology* **1991**, *19*, 997–1000.
9. Chadwick, O.A.; Hendricks, D.M.; Nettleton, W.D. Silica in duric soils: A depositional model. *Soil Sci. Soc. Am. J.* **1987**, *51* (4), 975–982.
10. Schoeneberger, P.J.; Wysocki, D.A.; Benham, E.C.; Broderick, W.D. *Field Book for Describing and Sampling Soils, Version 1.1*; National Soil Survey Center, Natural Resources Conservation Service, US Department of Agriculture: Lincoln, 1998; 185 pp.
11. Erickson, A.J.; Mortensen, V.L. *Soil Survey of Cache Valley Area, Utah*; US Government Printing Office: Washington, 1974; 192 pp.

Calcium

Zdenko Rengel

*Head, Soil Science and Plant Nutrition, University of Western Australia, Nedlands,
Western Australia, Australia*

Abstract

Calcium (Ca) has numerous roles in soils, plants, microflora, and fauna. In particular, Ca influences soil chemical and physical properties, maintains the structure and function of the cell plasma membrane, and is directly involved in cell signaling (as a secondary messenger).

CALCIUM IN SOIL

The average Calcium (Ca) content in the earth's crust is about 36 g/kg.^[1] In contrast, the average Ca content in soils is around 10 g/kg, which is about five times lower than the Ca content in igneous and sedimentary rocks, reflecting the ease of Ca leaching in the weathering process and in well-drained soils. Calcium (Ca) in igneous and metamorphic rocks occurs mainly in the plagioclase series of feldspars, with the high end of the plagioclase series weathering rapidly. Non-calcareous soils contain Ca in the form of high sodium plagioclase, augite, hornblende, and epidote.^[2] These minerals weather slowly and therefore are not good sources of Ca for living organisms. Calcareous soils have been formed from limestone or marl parent materials and frequently contain 50% or more calcium carbonate, with little Ca leaching over time owing to arid climates.

Leaching of Ca from soils in wet climates can be substantial, amounting up to around 300 kg Ca/ha/yr. Some fertilizers (e.g., ammonium) can increase Ca^{2+} leaching by displacing Ca^{2+} from the soil cation exchange complex. Losses of Ca^{2+} from soils by leaching are generally greater than the amounts taken out in farming products.

Exchangeable Ca

The crystallographic (dehydrated) radius of Ca^{2+} cation is 0.099 nm, which is smaller than the radii of soil monovalent cations and many divalent cations (e.g., Sr^{2+} and Ba^{2+}) but not smaller than the magnesium ion (Mg^{2+}) radius (0.066 nm).^[3] However, the hydrated radius of Ca^{2+} (0.43 nm) is greater than that of monovalent cations and is also greater than the radii of Mg^{2+} and Ba^{2+} .^[4]

Exchangeable cations are attracted to negative charges on the soil colloidal particles, forming an ion swarm in the diffuse double layer surrounding these particles. The specific adsorption increases with an increase in ion charge and with decreasing hydrated ion radius (because water

molecules act as an insulator). Exchangeable cations are present in the ratio $\text{Ca}^{2+} > \text{Mg}^{2+} > \text{K}^+ = \text{Na}^+$ in productive agricultural soils. Ca occupies the major part of the cation exchange complex in the neutral soils (e.g., up to 84% in Mollisols in Russia at pH 7.0) but also represents the most frequently occurring cation on the cation exchange complex in the acidic soils (e.g., 48% of the exchange complex in Lanna soil in Sweden).^[3] Only highly acidic, sodic, or serpentine soils may not have exchangeable Ca^{2+} as the dominant cation on the exchange complex; instead, they may be dominated by H^+ (acidic), Na^+ (sodic), and Mg^{2+} (serpentine soils).

Cation exchange capacity (CEC) refers to the sum of negative sites on soil solid phase that can bind cations and therefore protect them from being leached but keep them in a state that will allow replenishment of cations depleted from the soil solution. With increased soil acidity, there is a decrease in CEC.^[5] As a consequence, greater amounts of cations (especially basic cations like Ca^{2+} and Mg^{2+}) are present in soil solution and are therefore prone to leaching into deeper layers of the soil profile^[5] and eventually out of reach of roots. Liming of acidic soils results in an increase in effective CEC, with extra cation exchange sites mostly neutralized by Ca^{2+} .^[6]

Soil Solution Ca

An average Ca^{2+} concentration in the soil solution of temperate region soils is between 0.8 and 19.4 mM (30–730 mg/L),^[3,7] with the median values around 1.5 mM.^[8] Optimal growth of monocots, however, is already achieved at $<10 \mu\text{M}$ in flowing solution culture.^[9] Concentrations between 10 and 1000 μM of Ca are generally needed for optimal growth of dicotyledonous species.

The concentration of soil solution Ca^{2+} is regulated by the exchangeable Ca^{2+} , but the relationship varies for different soils. The ratio between exchangeable Ca^{2+} on the soil solid particles and Ca^{2+} in soil solution represents Ca buffer power; it ranges from 19 to 107 for various soils,^[10]

indicating a high capacity to replenish Ca^{2+} in soil solution.

Given the relatively high concentration of Ca^{2+} in the soil solution, the predominant transport pathway of Ca^{2+} toward roots is by mass flow rather than by diffusion. However, diffusion of Ca^{2+} toward roots occurs when uptake of Ca^{2+} is faster than the resupply by mass flow. In contrast, when mass flow exceeds uptake by roots, Ca^{2+} accumulates in the rhizosphere, creating a diffusion gradient away from roots. Most plants would take up less Ca^{2+} than the amount supplied via mass flow (e.g., ryegrass *Lolium rigidum*, subclover *Trifolium subterraneum*, and capeweed *Arctotheca calendula*), whereas some plant species can deplete Ca^{2+} in the rhizosphere by taking it up in excess of the supply by mass flow (e.g., lupin *Lupinus digitatus*).^[11] Therefore, the relative contribution of diffusion to Ca^{2+} transport toward roots depends on the plant's Ca requirement and on Ca^{2+} concentration in soil solution. If accumulation of Ca^{2+} takes place in the rhizosphere, precipitation of calcium carbonates or calcium sulfates may occur,^[10] especially in poorly leached, arid soils. Hence, soluble Ca^{2+} concentration in many arid soils is regulated not just by exchangeable Ca^{2+} but also by solubility of calcium sulfate and/or calcium carbonate.

Ca/Mg Ratio in Soil

Some authors have promoted the balance of soil exchangeable cations, particularly Ca^{2+} and Mg^{2+} , as an important criterion of soil fertility and productivity. The recommended range of the exchangeable Ca:Mg ratio in soil is 3–7.^[12] However, fertilization strategies based on cation saturation ratios (aimed at achieving a balance of soil cations instead of applying sufficient amounts of each nutrient) are not widely accepted.^[13] Although the concept has a good theoretical basis, it is undisputable that optimal plant growth may occur at a wide range of exchangeable Ca:Mg ratios (e.g., ratios ranging from 0.5 to 16 have produced constant yields for a variety of plant species and soil types).^[14,15]

Ca AS SOIL AMELIORANT

Amelioration of Acidic Soils

The widely accepted ameliorative measure for acid soils is liming, whereby Al held on the exchange complex is neutralized.^[16] Incorporating lime into the topsoil increases soil pH.^[3,4,16] In contrast, incorporation of lime into deeper horizons to ameliorate acidic subsoil is technically difficult and is frequently unprofitable.^[17] Alternatively, the surface-applied lime may in some cases allow sufficient leaching of Ca^{2+} down the profile to ameliorate subsoil acidity; this approach works well only in light-textured, sandy soils and requires lime rates sufficiently high not just

to neutralize topsoil acidity but also to provide surplus alkalinity to be leached into deeper horizons.^[16]

Lime reaction in soil is dependent on the nature and the fineness of the liming material. Dolomite [$\text{CaMg}(\text{CO}_3)_2$, with >15% Mg] has lower solubility but a higher surface area (smaller particle size) compared with calcite (CaCO_3); hence, the two types of material may have similar effectiveness in ameliorating acidic soil.

Amelioration of Sodic Soils

Sodic soils have high exchangeable Na^+ content and low hydraulic conductivity because Na promotes deflocculation and swelling of clay particles and thus dispersion of soils, especially those containing expanding montmorillonite clays.^[3,4] Amelioration of sodic soils hinges on replacing exchangeable Na^+ by Ca^{2+} and the restoration of a granular physical structure. For non-calcareous sodic soils, gypsum is the Ca-containing material of choice.

CA UPTAKE BY PLANTS

Ca deficiency in soils is rare and occurs only on serpentine soils derived from Mg-rich materials or on highly leached acidic Al-saturated soils. Even under those conditions, Ca deficiency in plants may be owing to Mg- or Al-repressing Ca uptake rather than owing to absolute Ca deficiency.

Symptoms of Ca deficiency are leaf distortion and petiole collapse (e.g., in *Glycine max*), rolled young leaves (e.g., in small grains), collapsed midrib, terminal dieback and defoliation (e.g., in *Prunus persica* seedlings), or reduced rate of wood formation and decreased amount of functional sapwood and live crown.^[18]

An average Ca concentration in the plant material is around 5 g/kg dry weight, which is about 125,000 times greater than for molybdenum as the least abundant nutrient and about 1/8 of nitrogen as the most abundant nutrient in plant tissue. Greater tissue concentrations of Ca are required in dicot compared to monocot plants.^[19] Most of Ca is bound in the cell wall (about 60%) or sequestered in different organelles.^[20] The activity of free cytosolic Ca^{2+} is therefore very low (10^{-7} to 10^{-6} M),^[21] preventing reaction of Ca^{2+} with inorganic phosphates and thus Ca^{2+} cytotoxicity.

Plant roots have CEC similarly to soils.^[22] Therefore, one cation exchanger (root) is in contact with the other (soil), resulting in competition for cations based on physicochemical principles. Plant capacity to take up cations is at least partly influenced by the root CEC^[23] because ionic environment at the plasma membrane surface (where uptake of all ions takes place) depends on the strength of the electrical field developed in interaction of two charged surfaces (root and soil particles).

Ca uptake is via an apoplastic pathway and is thus hindered by the Casparian strip or any suberization in the root

cell walls in general. Therefore, the most effective transport of Ca^{2+} is at the root tip.^[24,25] Transport of Ca^{2+} across the root cell plasma membrane into the cytosol is down the concentration gradient (high in the apoplast and low in the cytosol) via Ca^{2+} -selective channels.^[25,26]

Ca is transported mainly in the xylem sap, whereas its mobility in the phloem sap is poor. As a consequence, concentration of Ca is greater in mature and old leaves than in the young ones.^[27] Similarly, transport of Ca into fruits and grain is poor. The typical example of inadequate supply of Ca to developing fruits is blossom-end rot in capsicum and tomatoes^[28] and bitter pits in apples and pears.^[29]

Ca content in soils determines distribution of plant species in the natural ecosystems. Calcicole species are adapted to soils rich in Ca (these species are also sensitive to Al). In contrast, calcifuge species have a low Ca demand and are therefore adapted to acid, low-Ca soils (these species are also tolerant to Al).^[8,18]

CONCLUSION

As the predominant soil exchangeable cation, Ca^{2+} determines chemical and physical properties of soil. Ca can easily be leached into deep soil layers, causing topsoils to acidify. Ca lost by leaching and by export in farming products should be replenished. Liming is used to restore Ca content as well as to neutralize acidity.

REFERENCES

1. Glendinning, J.S. *Australian Soil Fertility Manual*; CSIRO Publishing: Collingwood, 2000; 154 pp.
2. Bear, F.E. *Chemistry of the Soil*, 2nd Ed.; Van Nostrand Reinhold Company: New York, 1969; 515 pp.
3. Bohn, H.L.; McNeal, B.L.; O'Connor, G.A. *Soil Chemistry*, 2nd Ed.; Wiley-Interscience: New York, 1985; 341 pp.
4. Tan, K.H. *Principles of Soil Chemistry*, 3rd Ed.; Marcel Dekker Inc.: New York, 1998.
5. Blake, L.; Goulding, K.W.T.; Mott, C.J.B.; Johnston, A.E. Changes in soil chemistry accompanying acidification over more than 100 years under woodland and grass at Rothamsted Experimental Station, U.K. *Eur. J. Soil Sci.* **1999**, *50*, 401–412.
6. Hochman, Z.; Edmeades, D.C.; White, E. Changes in effective cation exchange capacity and exchangeable aluminium with soil pH in lime-amended field soils. *Aust. J. Soil Res.* **1992**, *30*, 177–187.
7. Adams, F. Soil solution. In *The Plant Root and Its Environment*; Carson, E.W., Ed.; University Press of Virginia: Charlottesville, 1974; 441–481.
8. Runge, M.; Rode, M.W. Effects of soil acidity on plant associations. In *Soil Acidity*; Ulrich, B., Sumner, M.E., Eds.; Springer: Berlin/Heidelberg, 1991; 183–202.
9. Loneragan, J.F.; Snowball, K. Calcium requirement of plants. *Aust. J. Agr. Res.* **1969**, *20*, 465–478.
10. Barber, S.A. *Soil Nutrient Bioavailability. A Mechanistic Approach*, 2nd Ed.; John Wiley & Sons: New York, 1995.
11. Barber, S.A.; Ozanne, P.G. Autoradiographic evidence for the differential effect of four plant species in altering the Ca content of the rhizosphere soil. *Soil Sci. Soc. Am. Proc.* **1970**, *34*, 635–637.
12. Kinsey, N.; Walters, C. *Neal Kinsey's Hands-on Agronomy*; Acres: Metairie, 1995.
13. Dahnke, W.C.; Olson, R.A. Soil test correlation, calibration and recommendation. In *Soil Testing and Plant Analysis*; Westerman, R.L., Ed.; Soil Science Society of America: Madison, 1991; 65 pp.
14. Simson, C.R.; Corey, R.B.; Sumner, M.E. Effect of varying Ca: Mg ratios on yield and composition of corn (*Zea mays*) and alfalfa (*Medicago sativa*). *Commun. Soil Sci. Plant Anal.* **1979**, *10*, 153–162.
15. McLean, E.O.; Hartwig, R.C.; Eckert, D.J.; Triplett, G.B. Basic cation saturation ratios as a basis for fertilising and liming agronomic crops. II. Field studies. *Agron. J.* **1983**, *75*, 635–639.
16. Rengel, Z. *Handbook of Soil Acidity*; Marcel Dekker Inc: New York, 2002.
17. Tang, C.; Rengel, Z.; Diatloff, E.; Gazey, C. Responses of wheat and barley to liming on a sandy soil with subsoil acidity. *Field Crops Res.* **2003**, *80*, 235–244.
18. Rengel, Z. Role of calcium in aluminium toxicity. *New Phytol.* **1992**, *121*, 499–513.
19. Marschner, H. *Mineral Nutrition of Higher Plants*, 2nd Ed.; Academic Press: London, 1995.
20. Clarkson, D.T. Calcium transport between tissues and its distribution in the plant. *Plant Cell Environ.* **1984**, *7*, 449–456.
21. Bush, D.S. Calcium regulation in plant cells and its role in signalling. *Annu. Rev. Plant Physiol. Plant Mol. Biol.* **1995**, *46*, 95–122.
22. Rengel, Z.; Robinson, D.L. Determination of cation exchange capacity of ryegrass roots by summing exchangeable cations. *Plant Soil* **1989**, *116*, 217–222.
23. Rengel, Z. Competitive Al^{3+} inhibition of net Mg^{2+} uptake by intact *Lolium multiflorum* roots. II. Plant age effects. *Plant Physiol.* **1990**, *93*, 1261–1267.
24. Ferguson, I.B.; Clarkson, D.T. Ion transport and endodermal suberization in the roots of *Zea mays*. *New Phytol.* **1975**, *75*, 69–79.
25. White, P.J.; Broadley, M.R. Calcium in plants. *Ann. Bot.* **2003**, *92*, 487–511.
26. Piñeros, M.; Tester, M. Calcium channels in higher plant cells: Selectivity, regulation and pharmacology. *J. Exp. Bot.* **1997**, *48*, 551–577.
27. Drossopoulos, J.B.; Bouranis, D.L.; Bairaktari, B.D. Patterns of mineral nutrient fluctuations in soybean leaves in relation to their position. *J. Plant Nutr.* **1994**, *17*, 1017–1035.
28. Ho, L.C.; Belda, R.; Brown, M.; Andrews, J.; Adams, P. Uptake and transport of calcium and the possible causes of blossom-end rot in tomato. *J. Exp. Bot.* **1993**, *44*, 509–518.
29. Raese, J.T.; Drake, S.R. Effect of calcium spray materials, rate, time of spray application, and rootstocks on fruit quality of 'RED' and 'Golden Delicious' apples. *J. Plant Nutr.* **2000**, *23*, 1435–1447.

Carbon Assessment: Spectroscopic Methods

James B. Reeves III

Animal Manure and By-Products Laboratory, U.S. Department of Agriculture (USDA), Beltsville, Maryland, U.S.A.

Gregory W. McCarty

Environmental Quality Laboratory, U.S. Department of Agriculture (USDA), Beltsville, Maryland, U.S.A.

Abstract

Diffuse reflectance spectroscopy based on near-infrared radiation [near-infrared reflectance spectroscopy (NIRS)] has become an important method for analyzing agricultural products, where large numbers of measurements are needed to detect variations in product composition that impact market value (i.e., protein in grains and food/feed composition) or to monitor and measure spatial and temporal variation in environmental parameters (i.e., soil carbon and other soil properties). Diffuse reflectance spectroscopy, using Fourier transform mid-infrared [diffuse reflectance infrared Fourier transform spectroscopy (DRIFTS)], has also been shown to be capable of rapid quantitative analysis of agricultural products and environmental samples. Through a process termed “chemometrics,” information contained in the entire spectrum is related to the property of interest by the use of multivariate statistical techniques such as stepwise regression, principle component analysis, or partial squares least regression. Mathematical models produced by these techniques form the calibration models that are used to predict properties of unknown samples. With the development of robust calibrations, NIRS and DRIFTS have the ability to analyze samples rapidly and simultaneously for multiple properties with virtually no consumables and minimal sample preparation.

QUANTITATIVE ANALYSIS BY DIFFUSE REFLECTANCE

Basics on Diffuse Reflectance

From a historical perspective, the analytical use of spectroscopy has often involved measurement of light transmission through a sample held in, e.g., a cuvette, such as in colorimetric determinations, or through a sample squeezed between salt plates in the case of mid-infrared (mid-IR) spectroscopy. Over the last decades, a quantitative analytical approach has been developed based on a process called diffuse reflectance (Fig. 1), and this approach has come to dominate the analysis of many agriculture products.^[1] In this method, samples can be scanned in an open sample cup (sealed cups are used for some instrument configurations depending on the spectral range) with the surface prepared by either scrapping across the surface of the cup with a straight edge or smoothing in a similar manner with a spatula.

Examination of Fig. 1 reveals that light radiation hitting a particulate sample in a cell can interact in several ways. Light directly reflects off either the cell window (A) or sample (B) undergoing specular (mirror-like) reflection which provides no compositional information. This is also

true for light that enters but never leaves the sample (D). However, light, which penetrates sample particles and is then reflected back to be collected by some detector arrangement, (C) is said to be diffusely reflected and provides compositional information based on the absorption pattern.

Most of this work has been performed using radiation in the near-infrared (NIR) spectral range, 9000–4000 cm^{−1}, because of the belief that spectral distortions caused by specular reflection in the mid-IR^[2] made quantitative analysis, using Diffuse reflectance infrared Fourier transform spectroscopy (DRIFTS), impossible despite the greater wealth of spectral information present. However, it has been demonstrated that despite the spectral distortions present in DRIFTS spectra, quantitative analysis can be performed^[3–8] with an accuracy often greater than that achieved using near-infrared reflectance spectroscopy (NIRS).^[7–9]

Chemometrics

Unlike quantitative analysis using colorimetry, where only one or two absorbing components are changing at any given time and thus only one or two wavelengths need to be monitored, quantitative analysis using spectra such as

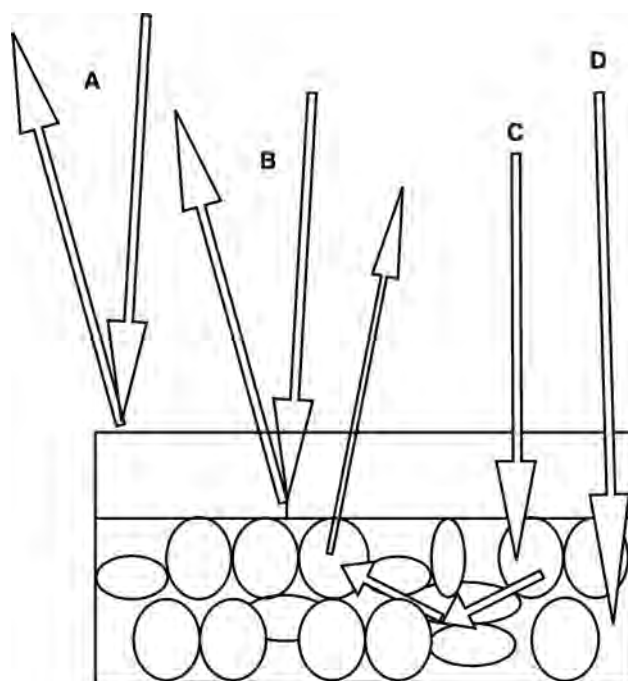


Fig. 1 Diffuse reflectance for a ground sample. (A) Reflection of cell window, (B) specular reflection of sample surface, (C) diffuse reflection giving rise to spectral information, (D) total absorption.

those shown in Fig. 2 requires many spectra with a range of values for the analyte in question, because many components are absorbed at the same spectral locations and the concentrations of many components are simultaneously changing from one sample to the next. The process of developing and testing calibrations using multivariate statistics for quantitative analysis has come to be known as chemometrics.^[1,10] The statistical methods used are said to be multivariate in nature, in that many wavelengths (stepwise regression) or even all (i.e., principle component analysis, partial least squares analysis, and neural nets) are used to relate the analyte information to the spectral data. Because of the many spectral values often available (100–1000s of values per spectrum), it is also very easy to produce spurious correlations if only a few 10s of samples are used. This is called overfitting, and the resulting equation is useless for predicting new samples. The use of too few samples for calibration development is also probably the biggest error made by novice users of chemometric techniques.

Simultaneous Determination of Multiple Analytes

Spectral information is a characteristic of the sample and does not change with the analyte of interest, only the

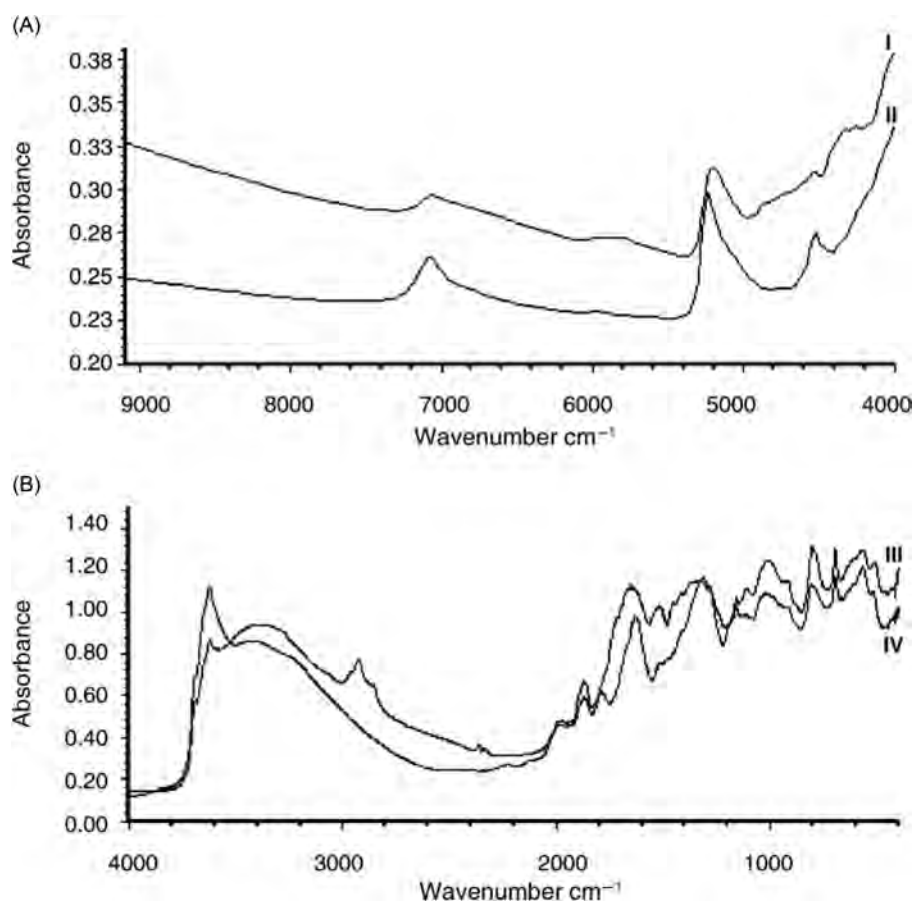


Fig. 2 Near-infrared spectra (A) of low C (I) and high C (II) and mid-infrared spectra (B) of high C (III) and low C (IV) soil samples (approximately 0.1 and 3% organic C contents).

specific spectral data needed for calibration changes with the analyte of interest, i.e., different absorptions relate to carbon (C) as opposed to nitrogen (N), etc. Thus, once a spectrum is obtained, the same spectrum can be used to determine different analytes, assuming calibration equations are available, i.e., one or a hundred analytes can be determined in a matter of moments from the same spectrum. Thus, NIRS and DRIFTS can greatly reduce the time for sample analysis when many samples and analytes per sample are going to be determined.

Potential for Soil Analysis

The chemometric approach using NIRS and DRIFTS works best when large numbers of samples are involved both from the standpoint of the need to avoid overfitting and from the point of being worth the time to develop the calibrations. Because of the possibility of using C sequestration in soil as a means to remove carbon dioxide from the atmosphere, there is increased interest in detecting changes in the amounts of C present in soil. Because of differences in the lability of various forms of soil C, there is also interest in determining differences in the composition of the C present. The “gold standard” for the determination of total, organic, and inorganic C in soils is combustion that requires two runs to determine both organic and inorganic C (in the form of carbonates) and cannot distinguish forms of C such as charcoal different from other forms of organic C. As a result, it cannot determine labile as opposed to non-labile organic C, unless extractions are performed and the C content of the extracts determined. NIRS and DRIFTS offer the possibility of determining the forms of organic C present while simultaneously determining the organic C vs. inorganic C and do so at a greatly reduced cost and speed many times the conventional methods. Similarly, the advent of data-intensive technologies such as site-specific agriculture has generated the need for large databases, and the analysis of large numbers of samples so that spatial structure in agricultural fields can be characterized.

Soil Calibration Example

In Table 1, results are presented from a set of 237 soils collected over a wide range of the Western United States.^[8] These samples were available in acidified (carbonates removed) and non-acidified forms. Calibrations were developed using three-fourths of the samples and then used to determine the remaining one-fourth. As demonstrated, results using DRIFTS were much better than the corresponding NIR results, with results overall very similar to those found for the calibration set. These results support the conclusion that a

Table 1 PLS results using three-fourths of samples to develop calibration and remaining one-fourth as an independent test set.

Assay	Calibration results		Test results		Bias ^a	RD ^b
	R ²	RMSD	R ²	RMSD		
Mid-infrared spectra ^c						
Organic C ^d	0.985	1.62	0.967	2.42	0.552	18.0
Organic C ^e	0.949	3.03	0.942	3.15	0.016	23.4
Inorganic C ^e	0.991	1.11	0.983	1.20	0.328	22.6
Total C ^e	0.967	2.89	0.946	3.42	0.381	18.2
Near-infrared spectra ^c						
Organic C ^d	0.829	5.54	0.800	5.77	−1.11	42.8
Organic C ^e	0.844	5.28	0.822	5.47	0.084	40.6
Inorganic C ^e	0.968	2.03	0.872	3.14	−0.141	59.2
Total C ^e	0.873	5.71	0.861	5.41	−0.321	28.8

Note: RMSD = root mean squared deviation (%).

^aActual – predicted mean value.

^bRMSD/Mean as percent.

^cSpectral ranges for mid- and near-infrared, 4000–10,000 and 10,000–400 wavenumbers, respectively.

^dAcidified samples to remove carbonates.

^eNon-acidified samples.

DRIFTS calibration could be developed using these samples, which could be used to determine the C values for other samples from the same geographical locations, thus greatly reducing the need for combustion analysis.

CONCLUSION

Results from a variety of data sets have demonstrated that NIRS and DRIFTS show great potential for the determination of soil composition, especially soil C. While mid-IR calibrations for soil C appear to be more accurate and robust, the use of DRIFTS for quantitative analysis is not nearly as developed as that of NIRS and may be more difficult to make portable. With reasonable input of resources, spectroscopy has the potential to generate the extensive databases on soil properties needed for producing spatial structure maps for soil properties that can then be used for site-specific management of agricultural lands. Results have also shown that accurate calibrations can be developed using NIR or mid-IR spectra for the determination of a number of compositional parameters including total C, total N, pH, and many measures of biological activity as reflected by enzyme activities and measures of biologically active N. While efforts at determining the robustness of mid-IR calibrations indicated that mid-IR soil calibrations generally performed in a manner similar to NIR calibrations, differences found indicate that the basis for mid-IR calibrations may at times be different.

REFERENCES

1. Roberts, C.; Workman, J.J.; Reeves, J.B., III *Near-Infrared Spectroscopy in Agriculture*; ASA-CSSA-SSSA: Madison, 2004.
2. Reeves, J.B., III; Francis, B.F.; Hamilton, S.K. Specular reflection and diffuse reflectance spectroscopy of soils. *Appl. Spectrosc.* **2005**, *59*, 39–46.
3. Reeves, J.B., III Improvement in Fourier near- and mid-infrared diffuse reflectance spectroscopic calibrations through the use of a sample transport device. *Appl. Spectrosc.* **1996**, *50*, 965–969.
4. Reeves, J.B., III Mid-infrared diffuse reflectance spectroscopy: Is sample dilution with KBr necessary, and if so, when? *Am. Lab.* **2003**, *35* (8), 24–28.
5. Janik, L.J.; Merry, R.H.; Skjemstad, J.O. Can mid-infrared diffuse reflectance analysis replace soil extractions? *Aust. J. Exp. Agr.* **1998**, *38*, 681–696.
6. Reeves, J.B., III; McCarty, G.W.; Reeves, V.B. Mid-infrared diffuse reflectance spectroscopy for the quantitative analysis of agricultural soils. *J. Agr. Food Chem.* **2001**, *49*, 766–772.
7. Reeves, J.B., III; McCarty, G.W.; Mimmo, T.V.; Reeves, V.B.; Follett, R.F.; Kimble, J.M. Spectroscopic calibrations for the determination of carbon in soils. In *17th WCSS*, Thailand, Aug 14–21, 2002; 398-1–398-9.
8. McCarty, G.W.; Reeves, J.B., III; Reeves, V.B.; Follett, R.F.; Kimble, J.M. Mid-infrared and near-infrared diffuse reflectance spectroscopy for soil carbon measurement. *Soil Sci. Soc. Am. J.* **2002**, *66*, 640–646.
9. Malley, D.F.; Martin, P.D.; Ben-Dor, E. Application in analysis of soils. In *Near-Infrared Spectroscopy in Agriculture*; Roberts, C., Workman, J.J., Reeves, J.B., III, Eds.; ASA-CSSA-SSSA: Madison, 2004; 729–784.
10. Naes, T.; Isaksson, T.; Fearn, T.; Davies, T. *A User Friendly Guide to Multivariate Calibration and Classification*; NIR Publications: Chichester, 2002.

Carbon Sequestration: Fish Ponds

Subhendu Adhikari

*Central Institute of Freshwater Aquaculture, Bhubaneswar, India
and Carbon Management and Sequestration Center, Ohio State
University, Columbus, Ohio, U.S.A.*

Rattan Lal

*Carbon Management and Sequestration Center, Ohio State University,
Columbus, Ohio, U.S.A.*

Abstract

The carbon (C) accretion rate in different aquaculture ponds ranges from 0.28 to 4.37 Mg/ha/yr with an average of 1.56 Mg/ha/yr. The culture of tilapia and carp has a high C accretion rate while freshwater prawn may have a low rate. Aquaculture ponds could sequester 17.2 Tg/yr of C from 11.1 million hectares (Mha) of aquaculture ponds globally.

INTRODUCTION

With the increase of atmospheric carbon dioxide (CO₂) concentration, considerable attention has been devoted to terrestrial carbon (C) storage in soils, forests, and grasslands. The depositional environments (i.e., lakes and impoundments) represent short- to long-term scale storage of atmospheric CO₂. Globally, this storage of organic C (OC) in the sediments of natural lakes ranges from 30 to 70 Tg C/yr (Tg = teragram = 10¹² g = 1 million metric ton).^[1–3] The burial in impoundments is much larger ranging from 150 to 220 Tg C/yr.^[4] These rates of C burials are comparable to the OC storage in the sediments of the global oceans estimated at 120–240 Tg C/yr.^[5–7] The annual magnitude of OC stored in lakes and reservoirs is also comparable to the delivery of OC by rivers to the ocean, about 400 Tg C/yr.^[8–10] However, these rates of OC storage in lakes and agricultural impoundments are modest compared to the total storage of OC terrestrially that ranges from 1000 to 4000 Tg C/yr.^[11] Wetlands also accumulate significant amounts of C in their soils, compared to adjacent upland sites.^[12] Strategies to reduce C and other greenhouse gas emissions from agriculture and enhance C sinks on farms/wetlands have also been identified,^[13–16] but options for different aquaculture farming systems have not been widely assessed.

There are 11.1 million hectares (Mha) of aquaculture ponds globally.^[17] Manures, fertilizers, feed, and other agricultural wastes are applied to ponds for higher production, and these inputs stimulate OC production by phytoplankton photosynthesis in ponds.^[18] Aquaculture ponds do not have large external sediment loads like

reservoirs or watershed ponds in agricultural or other rural areas. Further, sediments in aquaculture ponds are eroded by rain, waves, and water currents generated by mechanical aerators, activities of different culture species, and harvesting operations. Thus, coarse soil particles suspended by internal erosion settle near the edge of ponds while smaller particles tend to settle in deeper areas of the ponds.^[19] Uneaten feed, organic fertilizers, organic matter (OM) from dead plankton, and excreta from different species settle at pond bottoms and gradually mix with soil particles. When ponds are drained for harvest, organic detritus is discharged from the ponds, and after draining, pond bottoms are exposed to sunlight for drying that enhance soil aeration and accelerate decomposition of labile OM.^[20] Despite these practices in aquaculture ponds, a layer of sediments with a higher OC concentration than that in the original soil develops at the pond bottom.^[21] Aquaculture ponds store as much as 0.21% of the annual global C emissions of about 10 Pg/yr and represent a small sink for C that is an important ancillary benefit in considerations of the global C budget.^[22] However, detailed data on C storage by aquaculture ponds are scanty globally. Thus, this entry will be dealt with the potentialities of the soil C sequestration in fish ponds.

MATERIALS AND METHODS

Inputs in Aquaculture Activities

Aquaculture activities involve a variety of inputs for fish production including manures, fertilizers, feed, and a

combination of all these things. In addition to this, lime is applied, if necessary. With high fertilization rate, the nutrients assimilated in fish biomass were estimated to be less than 20–30% for nitrogen (N), 10–15% for phosphorus (P), 10–20% for OC.^[23,24] Most of those lost nutrients are distributed in water, fish biomass, and sediments of the pond systems. It is generally believed that a large proportion of nutrients received in ponds end up in pond muds and discharged effluents.^[25] The organic fertilizer (cow dung) contained 40–42% C (on dry weight basis), while feed contained 46–48% C (on dry weight basis). The carps, freshwater prawn (scampi), shrimp, and tilapia contained 11.0–13.5% C (on live weight basis).^[26]

Sediment Core Analysis

The average depths of fish ponds vary between 1.0 and 2.5 m. Sediment cores could be collected manually from the ponds.^[27] Sediment accumulation rate could be estimated by dividing the sediment depth by the age of the pond. Wet sediment samples could be weighed, a measured quantity dried for 24 hours at 105°C, cooled in desiccators, and weighed again for dry bulk density estimation. A part of sediment samples were air-dried, pulverized to pass a 0.25-mm screen, and could be analyzed for OC using dichromate oxidation technique by rapid titration method. The OC accretion rate could be estimated by multiplying the annual rate of accumulation of dry sediment by the sediment dry bulk density and the

OC concentration in the sediment. The soil OC accretion could be computed using the following equation:^[13]

$$\text{Mg C/ha} = (\%C \times \text{corrected bulk density} \times d \times 104 \text{ m}^2/\text{ha})/100$$

where Mg C/ha represents mega gram of C per hectare (1 Mg = 10⁶ g = 1 metric ton), soil bulk density is represented in mega gram per cubic meter, and d is the depth measured in meters.

RESULTS AND DISCUSSION

Rate of Sediment Accumulation

The rate of sediment accumulation ranged from 0.9 to 1.20 cm/yr in shrimp ponds, 1.10 to 1.54 cm/yr in freshwater prawn ponds, 1.10 to 2.33 cm/yr in tilapia ponds, 188 cm/yr in polyculture ponds, 1.00 to 3.70 cm/yr in carp ponds, 1.20 cm/yr in channel catfish ponds, 2.50 cm/yr in Clarias catfish ponds, 0.45 cm/yr in yellow perch ponds, and 0.80 cm/yr in bait minnow ponds, respectively (Table 1). Overall, the sediment rate accumulation ranged from 0.45 to 3.70 cm/yr in these ponds. When ponds are drained, considerable amount of sediments are lost from the bottom of the ponds through outflow or runoff. Ponds are usually drained once in every 3 years till the upper layer cracks to reduce the concentration of toxic gases in the pond bottom. The dry bulk density (g/cm³) of sediment in these ponds ranged from 0.49 to 0.63 in shrimp, 0.18 to 0.46 in prawn,

Table 1 C accretion from the data of sediment core analysis for different aquaculture systems.

Species	Country	Sediment accumulation (cm/yr)	Sediment dry bulk density (g/cm ³)	OC (%)	C sequestration (Mg/ha/yr)	References
Shrimp	Honduras	0.90	0.50	2.10	0.95	[22]
	India	1.62	0.49	1.22	0.97	[26]
	Thailand	1.20	0.63	3.66	2.76	[28]
Freshwater prawn	India	1.54	0.46	1.20	0.86	[26]
	Thailand	1.10	0.18	1.38	0.28	[29]
Tilapia	Brazil	1.25	0.51	4.17	2.65	[22]
	South Africa	2.33	0.56	3.35	4.37	[22]
	Thailand	1.10	0.35	3.00	1.15	[30]
India major carps with prawn (polyculture)	India	1.88	0.58	1.37	1.43	[26]
Carps	India	1.00	1.10	1.10	1.21	Unpublished
	Thailand	3.70	0.28	3.02	3.12	[29]
Channel catfish	Alabama, U.S.A.	1.20	0.38	2.92	1.33	[22]
Clarias catfish	Thailand	2.50	0.18	1.46	0.66	[29]
Yellow perch	Piketon, U.S.A.	0.45	0.84	2.80	1.06	[31]
Bait minnow	Arkansas, U.S.A.	0.80	0.47	1.58	0.59	[32]

0.34 to 0.56 in tilapia, 0.58 in polyculture, 0.28 to 1.10 in carps, 0.38 in channel catfish, 0.18 in Clarias catfish, 0.84 in yellow perch, and 0.47 in bait minnow ponds (Table 1). Overall, the bulk density of soils of these ponds ranged from 0.18 to 1.10 g/cm³.

OC Concentration in Sediment

The OC concentration (% g/g) in sediment ranged from 1.22 to 3.66 in shrimp, 1.20 to 1.38 in freshwater prawn, 3.00 to 4.17 in tilapia, 1.37 in polyculture, 1.10 to 3.02 in carps, 2.92 in channel catfish, 1.46 in Clarias catfish, 2.80 in yellow perch, and 1.58 in bait minnow ponds (Table 1). Overall, the OC in sediment varied from 1.10% to 4.17% in different ponds of the world. There were no clear trends in OC concentration related to culture species or intensity of culture. Significant quantities of OM can accumulate in bottom sediments in these aquaculture ponds. Feeds applied to these ponds to increase fish/prawn/shrimp settle at the bottom where it is mostly eaten by aquatic species. Inorganic nutrients released into the water from microbial decomposition of uneaten feed and feces of these species stimulate heavy plankton blooms. Phytoplankton cells have a short life span and continually die and settle to the bottom. In some cases, water supplies contain solids of appreciable OM content that may deposit in ponds. High rates of water exchange may result in large sediment inputs. Mechanical erosion may often erode bottom particles where water currents are higher and sedimentation occurs where currents are slower. Such a current flow pattern alters the shape of the bottom of the pond, reduces pond volume, and provides organic substrate for microorganisms.^[33]

Rate of C Sequestration

The annual C accretion rate estimated from sediment accumulation rate, dry bulk density, and percentage OC in sediment ranged from 0.95 to 2.76 Mg/ha/yr in shrimp ponds, 0.28 to 0.86 Mg/ha/yr in prawn ponds, 1.15 to 4.37 Mg/ha/yr in tilapia ponds, 1.43 Mg/ha/yr in

polyculture ponds, 1.21 to 3.12 Mg/ha/yr in carp ponds, 1.33 Mg/ha/yr in channel catfish ponds, 0.66 Mg/ha/yr in Clarias catfish ponds, 1.06 Mg/ha/yr in yellow perch ponds, and 0.59 Mg/ha/yr in bait minnow ponds (Table 1). Overall, the C sequestration rate in these ponds ranged from 0.28 to 4.37 Mg/ha/yr with an average of 1.56 Mg/ha/yr. The culture of tilapia and carps recorded the highest C storage while freshwater prawn recorded the lowest. The freshwater prawn prefers clear bottom (sandy) for their growth. In general, tilapia and carp stir pond sediment in search of food organisms, and this action possibly incorporates deposited OM into sediment to minimize its loss during pond drainage for harvest.^[22]

The applied nutrients from different sources in the culture systems and also the nutrients entering from the terrestrial into aquatic ecosystems through runoff have an important role in C sequestration by the sediment in fish ponds. Fertilizers and feeds are generally the largest inputs of N and P to fish ponds. Feeds also supply C to fish ponds. Organic fertilizers (fermented rice straw, cow dung, etc.) are often added to ponds to boost fish yields by increasing primary productivity through released inorganic nutrients or by providing OC through heterotrophic pathways. Depending on the species, fish feed directly on attached or planktonic algae, detritus/fungal flocs, or smaller animals such as zooplankton and snails that feed on algae and detritus.^[34] Ponds also receive nutrients from the regulated inflow water.

The estimated average annual C sequestration rate for aquaculture ponds monitored in this study is lower than that of agricultural impoundments and large river reservoirs but higher than that of natural lakes and inland seas (Table 2). Aquaculture ponds can sequester 17.2 Tg/yr of C globally. Aquaculture ponds store less C than that by large reservoirs (160–280 Tg/yr^[1,37]) and agricultural impoundments (163 Tg/yr^[36]). The aquaculture ponds store C at a lower rate than agricultural impoundments and large reservoirs because of lower input of external sediment and associated OM to aquaculture ponds than in other impoundments.^[22] Moreover, aquaculture pond management minimizes OM accumulation.

Table 2 Areas of global inland water bodies and annual rates and amounts of OC storage in these systems.

Inland water body	Global area (Mha)	C sequestration rate (Mg/ha/yr)	Global C storage (Tg/yr)	References
Aquaculture ponds				
Freshwater	8.75	1.56	13.6	[35]
Brackish water	2.33	1.56	3.6	[35]
Large lakes	118	0.05	6.0	[1]
Small lakes	32	0.72	23.0	[1]
Inland seas	100	0.05	5.0	[1]
Large river reservoirs	40	4.00	160	[1]
Agricultural impoundments	7.7	21.20	163	[36]

For example, ponds are dried at least once in 3 years to reduce the gaseous emissions from the bottom and also to flush out sediments using pressurized water. This intervention is needed because a thick layer of the sediment reduces the productivity of the pond. Although aquaculture ponds sequester comparatively small amount of C, globally pond farming systems under different management practices can sequester a large amount over long term.

REFERENCES

- Dean, W.E.; Gorham, E. Magnitude and significance of carbon burial in lakes, reservoirs, and peat lands. *Geology* **1998**, *26*, 535–538.
- Einsele, G.; Yan, J.; Hinderer, M. Atmospheric carbon burial in modern lake basins and its significance for the global carbon budget. *Global Planet. Change* **2001**, *30*, 167–195.
- Mullholland, P.J.; Elwood, J.W. The role of lake and reservoir sediments as sinks in the perturbed global carbon cycle. *Tellus* **1982**, *34*, 490–499.
- Stallard, R.F. Terrestrial sedimentation and the C cycle: Coupling weathering and erosion to carbon storage. *Global Biogeochem. Cy.* **1998**, *12*, 231–237.
- Van Oost, K.; Quine, T.A.; Govers, G.; De Gryze, S.; Six, J.; Harden, J.W.; Ritchie, J.C.; McCarty, G.W.; Heckrath, G.; Kosmas, C.; Giraldez, J.V.; da Silva, J.R.; Merckx, R. The impact of agricultural soil erosion on the global carbon cycle. *Science* **2007**, *318*, 626–629.
- Duarte, C.M.; Middelburg, J.J.; Caraco, N. Major role of marine vegetation on the oceanic carbon cycle. *Biogeosci. Discuss.* **2004**, *1*, 659–679.
- Sundquist, E.T. The global carbon dioxide budget. *Science* **2003**, *259*, 934–935.
- Meybeck, M. Riverine transport of atmospheric carbon: Sources, global typology and budget. *Water Air Soil Pollut.* **1993**, *70*, 443–463.
- Lal, R. Soil erosion and the global carbon budget. *Environ. Int.* **2003**, *29*, 437–450.
- Probst, J.L. The role of continental erosion and river transports in the global carbon cycle. *Geochim. Cosmochim. Acta* **2002**, *69*, A725.
- Randerson, J.R.; Chapin, F.S.; Harden, J.W.; Neff, J.C.; Harmon, M.E. Net ecosystem production: A comprehensive measure of net carbon accumulation by ecosystems. *Ecol. Appl.* **2002**, *12*, 937–947.
- Bernal, B.; Mitsch, W.J. A comparison of soil carbon pools and profiles in wetlands in Costa Rica and Ohio. *Ecol. Eng.* **2008**, *34*, 311–323.
- Lal, R.; Kimble, J.M.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Ann Arbor Press: Chelsea, 1998.
- Robertson, G.P.; Paul, E.A.; Harwood, R.R. Greenhouse gases intensive agriculture: Contributions of individual gases to radiative warming of the atmosphere. *Science* **2000**, *289*, 1922–1925.
- Mitsch, W.J.; Tejada, J.; Nahlik, A.; Kolmann, B.; Bernal, B.; Hernandez, C.E. Tropical wetlands for climate change research, water quality management and conservation education on a university campus in Costa Rica. *Ecol. Eng.* **2008**, *34*, 276–288.
- Li, Y.; Wang, L.; Zhang, W.; Zhang, S.; Wang, H.; Fu, X.; Le, Y. Variability of soil carbon sequestration capability and microbial activity of different types of salt marsh soils at Chongming Dingtan. *Ecol. Eng.* **2010**, *36*, 1754–1760.
- Verdegem, M.C.J.; Bosma, R.H. Water withdrawal for brackish and inland aquaculture, and options to produce more fish in ponds with present water use. *Water Policy* **2009**, *11*, 52–68.
- Boyd, C.E.; Tucker, C.S. *Pond Aquaculture Water Quality Management*; Kluwer: Boston, 1998.
- Boyd, C.E. *Bottom Soils, Sediment, and Pond Aquaculture*; Chapman and Hall: New York, 1995.
- Ayub, M.; Boyd, C.E.; Teichert-Coddington, D. Effects of urea application, aeration, and drying on total carbon concentrations in pond bottom soils. *Prog. Fish Cult.* **1993**, *55*, 210–213.
- Munsiri, P.; Boyd, C.E.; Hajek, B.F. Physical and chemical characteristics of bottom soil profiles in ponds at Auburn, Alabama, and a proposed method for describing pond soil horizons. *J. World Aquacult. Soc.* **1995**, *26*, 346–377.
- Boyd, C.E.; Wood, C.W.; Chaney, P.L.; Queiroz, J.F. Role of aquaculture pond sediments in sequestration of annual global carbon emissions. *Environ. Pollut.* **2010**, *158*, 2537–2540.
- Edwards, P. Environmental issues in integrated agriculture-aquaculture and waste water-fed fish culture systems. In *Environment and Aquaculture in Developing Countries*; Pullin, R.S.V., Rosenthal, H., Maclean, J.L., Eds.; ICLARM Conf. Proc. 31; ICLARM: Manila, 1993; 139–170.
- Adhikari, S.; Sahu, B.C.; Dey, L. Nutrients budget and effluent characteristics in polyculture of scampi (*Macrobrachium rosenbergii*) and Indian major carps ponds using organic inputs. *Water Sci. Technol.* **2012**, *66*, 1540–1548.
- Lin, C.K.; Yi, Y.; Diana, J.S. The Effects of Pond Management Strategies on Nutrient Budgets: Thailand. In *Pond Dynamics/Aquaculture CRSP*; Burke, D., Goetze, B., Clair, D., Egna, H., Eds.; Fourteenth Annual Technical Report; Oregon State University: Corvallis, 1997; 19–24.
- Adhikari, S.; Lal, R.; Sahu, B.C. Carbon sequestration in the bottom sediments of aquaculture ponds of Orissa, India. *Ecol. Eng.* **2012**, *47*, 198–202.
- Steeby, J.A.; Hargreaves, J.A.; Tucker, C.S.; Kingsbury, S. Accumulation, organic carbon and dry matter concentration of sediment in commercial channel catfish ponds. *Aquacult. Eng.* **2004**, *30*, 115–126.
- Boyd, C.E.; Wood, C.W.; Thunjai, T.; Sonnenholzner, S. Pond soil characteristics and dynamics of soil organic matter and nutrients. In *Pond Dynamics/Aquaculture CRSP*, Seventeenth Annual Technical Report, 1996–1997; McElwee, Burke, K., Niles, D., Cummings, M., Egna, H., Eds.; Office of International Research and Development, Oregon State University: Corvallis, 1999; 1–8.
- Wudtisn, I.; Boyd, C.E. Physical and chemical characteristics of sediments in catfish, freshwater prawn, and

- carp ponds in Thailand. *Aquacult. Res.* **2006**, *37*, 1202–1214.
30. Thunjai, T.; Boyd, C.E.; Boonyaratpalin, M. Bottom soil quality in tilapia ponds of different age in Thailand. *Aquacult. Res.* **2004**, *35*, 698–705.
 31. Adhikari, S.; Lal, R.; Wang, H.-P. Carbon sequestration in the soils of aquaculture ponds, crop land, and forest land in southern Ohio, USA. *Environ. Monit. Assess.* **2014**, *186*, 1569–1574.
 32. Tepe, Y.; Boyd, C.E. Sediment quality in Arkansas bait minnow ponds. *J. World Aquacult. Soc.* **2002**, *33*, 221–232.
 33. Boyd, C.E. Shrimp pond bottom soil and sediment management. In *Proceedings of the Special Session on Shrimp Farming*, Orlando, Florida, May 22–25, 1992; Wyban, J., Ed.; Aquaculture Society: Baton Rouge, 1992; 166–181.
 34. Colman, J.A.; Edwards, P. Feeding pathways and environmental constraints in waste-fed aquaculture: Balance and optimization. In *Detritus and Microbial Ecology in Aquaculture*, Proceedings of the Conference on Detrital Systems for Aquaculture, Aug 26–31, 1985, Bellagio, Como, Italy; Moriarty, D.J.W., Pullin, R.S.V., Eds.; International Center for Living Aquatic Resources Management: Manila, 1987; 240–281.
 35. Smith, S.V.; Renwick, W.H.; Bartley, J.D.; Buddemeier, R.W. Distribution and significance of small, artificial water bodies across the United States landscape. *Sci. Total Environ.* **2002**, *299*, 21–36.
 36. Downing, J.A.; Cole, J.J.; Middelburg, J.J.; Striegl, R.G.; Durante, C.M.; Kortelainen, P.; Prairie, Y.T.; Laube, K.A. Sediment organic carbon burial in agriculturally eutrophic impoundments over the last century. *Global Biogeochem. Cy.* **2008**, *22*, 1–10.
 37. Cole, J.J.; Prairie, Y.T.; Caraco, N.F.; McDowell, W.H.; Tranvik, L.J.; Striegl, R.G.; Duarte, C.M.; Kortelainen, P.; Downing, J.A.; Middelburg, J.J.; Melack, J. Plumbing the global carbon cycle: Integrating inland waters into the terrestrial carbon budget. *Ecosystems* **2007**, *10*, 171–184.

Carbon Sequestration: Geologic

Eswaran Padmanabhan

Geoscience, University Teknologi Petronas, Bandar Seri Iskandar, Malaysia

Abstract

Carbon (C) has been sequestered naturally in soils and geological formations. The efficiency of the sequestration in soils depends on several factors. There are several challenges to maintaining the C pool in soils. Overall, C sequestration in soils can be hampered by mismanagement. Geological sequestration of carbon dioxide can be done in a few ways as well. However, high costs and the need for monitoring for possible leakage appear to be some of the major concerns regarding this type of sequestration. In comparison with soil sequestration that happens mostly in a natural manner, geological sequestration is anthropogenic in nature, involves high costs, and has a risk factor associated with it since the integrity of the containment will always be of our main concern. Nevertheless, as anthropogenically produced C is very high and is expected to be much higher, technology is driven to excel in the field of sequestration for long terms. The success of this technology can only be accessed with time.

INTRODUCTION

Carbon (C) has been sequestered naturally in the various components such as geological formations, soil, oceans, and wetlands. The three major pools of C are ocean, terrestrial, and atmosphere. The oceans contribute approximately 39,000 Pg (10^{15} g) of C, the terrestrial system about 2500 Pg, and the atmosphere about 750 Pg. The various estimates for C pool in soils range from 1220 Pg to 1576 Pg. Human activities are believed to have changed the atmospheric composition, including increasing the atmospheric carbon dioxide (CO_2) concentration, leading to global warming. Crowley,^[1] Karl and Trenberth,^[2] Manabe and Stouffer,^[3] Johns et al.,^[4] and Cox et al.^[5] pointed out that the terrestrial biosphere has lost the ability to act as a C sink as temperature rises. It is beyond the scope of this entry to discuss the reasons as to why the terrestrial biosphere has lost the ability to act as a C sink. However, the reader is referred to entries on soil degradation in this encyclopedia for further information on this subject matter. Many authors have indicated that reducing the impact of CO_2 emissions on the atmosphere and, therefore, global climate change would be a challenging task.^[6–11]

Soil C sequestration specifically deals with the transfer of CO_2 from the atmosphere into the soil via crop residues and other organic solids in a form that is not easily released back into the atmosphere. The rates of accumulation of C in soils are also known to be different.

Oil and gas companies have been injecting CO_2 in geological formations for many years. Traditionally, the CO_2 that is produced from anthropogenic factors is released to the atmosphere. In order for this type of sequestration to

reduce global emissions of CO_2 to preindustrial levels, the volume involved in the process must be huge.

Geological storage is achieved by injecting CO_2 into porous rock formations.^[12–14] Reservoirs that can be used for geological storage of CO_2 include sedimentary basins, depleted oil reservoirs, and non-economic coal beds. The stability of the impermeable caprock is important in such storage projects since a breach in the cap will release the buoyancy-driven, upward-migrating CO_2 into surrounding marine waters and eventually back into the atmosphere. It has been stated that caprocks are the primary seal during the life cycle of the CO_2 geological storage in a reservoir.^[15,16]

In essence, geological sequestration implies injecting CO_2 at depths of about 1–2 km, whereas sequestration of CO_2 in soils occurs in the solum. The rate of sequestration in soils tends to vary based on several factors such as soil type, temperature, organic C pool, pedological conditions, and humidity. In contrast, the ability to sequester CO_2 in geological strata depends on factors such as permeability, porosity, pore throat size, resident pressure, temperature at depth of sequestration, and mineral stability as well as other conditions that influence solid–liquid interactions. It is evident from the preceding discussion that CO_2 can be sequestered differently in different media. This entry discusses these basic differences and highlights the need to conserve and sustain the natural resources in order to better manage the increasing CO_2 contents on a global basis.

DEFINITIONS

It is well known that the produced CO_2 needs to be prevented from entering the earth's atmosphere for a

period of time sufficient to allow climate stabilization. Unfortunately, the optimum period of time that is required for this stabilization is not well understood. Generally, it is believed that the optimum period would be hundreds, if not thousands of years. As such, there is a need to store, in a safe manner, the produced CO₂ for extended periods of time.

The terms storage and sequestration have been occasionally used interchangeably and therefore merit a proper distinction. Mere physical storage would not seem to meet the stringent requirements of being able to hold back the stored CO₂ from entering the earth's atmosphere for thousands of years. Therefore, the storage mechanism is required to incorporate a chemical fixation of the CO₂ in a manner that the release would not occur easily and that too for a long time. This then provides the basic difference between CO₂ storage and CO₂ sequestration, the latter being the preferred long-term isolation of produced CO₂.

Essentially, sequestration means the transfer of atmospheric CO₂ into a form that has a high mean residence time that is long enough so that the CO₂ is not reemitted into the atmosphere. There are three types of C sequestration:^[17] 1) those that involve photosynthesis of plants that convert atmospheric CO₂ into biomass, soil organic matter, and other components; 2) those that involve engineering techniques; and 3) those that involve chemical transformations.

Another term used in this entry is EOR. EOR stands for enhanced oil recovery. Basically, this refers to the recovery of oil that is not easily recoverable. In such cases, additional effort is needed to expel the oil from the reservoir. Several methods are used in EOR endeavors. They include gas flooding and water-alternating gas or the use of surfactants. Chemical flooding is another EOR technique used (this includes, e.g., rock-fluid chemical reactions, application of new chemicals). Lastly, thermal methods and unconventional methods such as application of solvents and chemicals to steam flooding, modelling, and simulation are used in some cases as enhanced recovery techniques.

C SEQUESTRATION IN THE PEDOSPHERE

The natural processes of sequestering are generally cost-effective and tend to enhance the quality of the ecosystem.^[18] Sequestration in terrestrial systems is high in protected areas as well as in well-managed agricultural systems. Factors such as soil types, vegetation, and climatic conditions also play an important role in the rate of sequestration. Sustainable management of wetlands ensures optimum natural sequestration in aquatic ecosystems.

The bulk of the CO₂ emitted into the atmosphere is sequestered naturally in terrestrial ecosystems. Lal^[19]

estimated that about 25% of the amount of C released into the atmosphere by human activities is sequestered in soil organic matter on an annual basis. Plants play an active role in sequestration in terrestrial and wetlands, and therefore, it is not surprising to note that organic C is added to the soil regularly by plants and animals as well as from agricultural practices.

Soils with degraded organic C pool represent a unique challenge to scientists as the ability of the soil to function in a normal capacity to sequester C is impaired. In the first instance, issues such as erosion, acidification, nutrient mining, degradation of soil structure, and even contamination from excessive use of fertilizers or pesticides, herbicides, and other chemicals require continuous and intensive attention in order to keep the soil quality within the resilience capacity. The tropics host about 750 million hectares of degraded lands.^[20] The capacity of soils to sequester C depends on our ability to restore the soil to the original status.

There are numerous challenges that need to be overcome in order to enhance the C storage capacity of soils. The soil organic carbon (SOC) pool has to be improved especially in agricultural soils since these lands make up a huge proportion of our land space. Humification is a process of converting biomass C into stable humic substances, thereby improving the storage capacity of C in soils. Unfortunately, this process is very complicated and requires extensive research in order to implement or even improve this in degraded lands.

Another associated challenge in trying to enhance the C storage capacity is in intensifying the organo-mineral complexes especially in degraded soils. Organo-mineral complexes are by their very own nature very complicated in terms of their structure, bonds, and chemical tendencies. In addition to this, the sequestering capacity of soils can be altered simply by changing the type of land use. This basically means that a good cost-benefit analysis needs to be done in every farming system in order to assess the values in improving the C balance in such systems.

It has been mentioned that a combination of nanotechnology and biotechnology is very useful in the restoration of degraded soils and associated ecosystems, thereby enhancing the soil organic matter pool. The techniques range from the use of nanoporous materials, use of genetically modified organisms, and bioremediation of degraded lands. The very act of restoring degraded lands gives many benefits such as increased available water-holding capacity, elevated nutrient retention capacity, increased water infiltration capacity, and also increased C sequestration. More research needs to be done in this area, as the type of degraded lands present, especially in the tropics, is large and each represents a unique problem. It has been estimated that the gross rate of SOC sequestration ranges from 500 to 800 kg/ha/year in cold and humid regions and 100 to 300 kg/ha/year in dry and

warm regions in the Midwestern region of the United States.^[21] Such differences have to be taken into account when remediation techniques are implemented. The reader is referred to other parts of the encyclopedia for further information on soil remediation.

Despite knowing that soils and the rest of the terrestrial system can sequester huge amounts of C, it is very important to recognize that the C pool in soils can be easily destroyed by mismanagement of the land. The reader is referred to other sections of this encyclopedia for further information on this subject.

GEOLOGIC C SEQUESTRATION

Oldenburg^[22] has shown that the physical state of CO₂ is dependent on temperature and pressure. At room pressure and temperature, CO₂ is in a gaseous state. At greater depths, the gas can become a liquid and even reach a supercritical state at very high pressures. As such, CO₂ is usually injected in a liquid form, but the material transforms into other states as explained, when it moves into the sedimentary formation at various confining pressures and temperatures.

Large sedimentary basins are best suited for CO₂ sequestration mainly because of the large pore volume and connectivity.^[23] This is essential to maximize the amount of gas that can be injected into the formation. A suitable formation is more than 800 m deep, has thick and extensive seal, sufficient porosity, and permeability to allow large volume of gas injection at high flow rates without requiring high pressures.^[14] The need to reduce the pressure of injection is critical so as not to breach the seal for the formation.

There are about five ways to sequester CO₂ in geological media (Fig. 1). First, the CO₂ can be utilized in EOR operations to displace hydrocarbon and be sequestered at the same time. The CO₂ can be disposed of in disused oil and gas reservoirs or used to displace methane in coal

beds. The CO₂ can also be injected in deep saline aquifers and/or be stored in salt caverns.

There are various estimates for the capacity to sequester CO₂ by these different methods. According to Intergovernmental Panel on Climate Change,^[24] depleted oil and gas reservoirs have the capacity to sequester an estimated 675–900 Gt C. Saline aquifers have been estimated to be able to sequester about 1000–10,000 Gt C, whereas deep and unminable coal beds have the lowest potential to sequester (3–300 Gt C). Mineral carbonation involves the transformation of industrial CO₂ into calcium carbonate, magnesium carbonate, or other stable carbonate minerals. Basalt aquifers occur in many oceans and could sequester 2.3–11.5 Tt-C along seismic ridges and 5.9–29.6 Tt-C along aseismic ridges.^[25] Although these estimates vary from place to place, it is to be noted that the conditions within and external to the formation are not the same globally. Therefore, the actual sequestration capacity becomes a function of locality and time.

CO₂ is a reactive gas and as such can interact with both the rock and formation fluids in injected reservoirs. A number of studies have attempted to predict the behavior of CO₂ under a range of physical and chemical conditions.^[26,27] It is interesting to note that CO₂ solubility depends on temperature, pressure, and composition of formation fluids.

The rate of injection, surface area of CO₂–water interaction, and the degree of mixing control the proportion of injected volume that dissolves. Dissolution of this CO₂ results in mineral dissolution and precipitation.^[28] The precipitation of C-bearing minerals provides a long-term sequestration of CO₂. Studies show that about 1% of the CO₂ precipitated as carbonate minerals, the dissolved phase was 15–20 times more than that, and the rest stayed as an immiscible phase.

Apart from these factors, pressure and hydrodynamic and geothermal regimes in a basin have a significant impact on its potential for CO₂ storage. It is well known that fluids and gases do not remain stationary in the subsurface but migrate at rates related to their composition, state, and the resident pressures in the basin.

The geological fill of a basin and the internal heterogeneities at all scales, including faults and fractures, depositional patterns, and porosity–permeability trends, affect the capacity and effectiveness of CO₂ storage. The buoyancy-driven migration of immiscible CO₂ is controlled by barriers and increases the CO₂–water–rock interaction. This internal heterogeneity can also have a negative impact on storage effectiveness. It has outlined that four trapping mechanisms that can contribute to the long-term retention, i.e., structural, capillary, solubility, and mineral trapping.^[24] The reader is encouraged to review this document to gain further insight on these mechanisms.

In the actual sequestration process, engineering techniques^[7,29,30] used include injecting compressed and

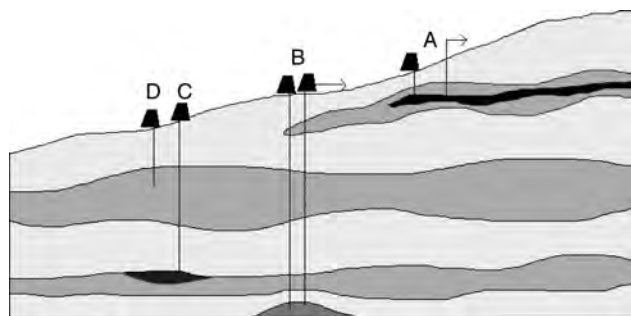


Fig. 1 The various types of geological carbon sequestration. A. CO₂ in enhanced coal bed methane recovery. B. CO₂ in EOR projects. C. Injection into deep saline formations D. Sequestration in depleted oil and gas reservoirs.

liquefied CO₂ beneath the ocean, into sedimentary formations, abandoned hydrocarbon reservoirs, saline aquifers, and brown fields as an EOR technique and to displace coal bed methane in unminable coal seams.^[31,32] The main concerns with geologic sequestration revolve around high costs^[18] and CO₂ leakage. Despite technological advancements, the concerns regarding breach of the seals due to abnormal pressures, reactivation of fissures and fractures, and seepage through old and new fissures require new protocols to be continually developed and tested for the measurement, monitoring, and verification. The issue of stability of the caprock seals is beyond the scope of this entry.

CONCLUSION

This entry discusses the sequestration potential of geological formation as opposed to soils. It is evident from the foregoing discussion that the success of the various methods can only be assessed with time. This is simply because there is some uncertainty on the long-term stability of the solid–gas interface phases. Large sedimentary basins can only be available for sequestration when the field is declared as a brown field or when EOR methods have been decided to be implemented in the said field. In addition to this, a good understanding of the reservoir properties is essential in ensuring a success in the sequestration procedure.

In a similar argument, maintaining the stability of the organomineral complexes in soils under various types of land management is a challenge. The basic idea of sequestration lies in the ability of the system to retain the gas for a long period of time. As such, an additional challenge would be to increase the retention rates to achieve this target.

It is therefore concluded that the naturally sequestered CO₂ in soils needs to be safeguarded in as much as the ability of the soils to continue to sequester the same gas. Geological sequestration of CO₂ raises several questions with regard to the level of technology that needs to be used in order to ensure long-term stability and sequestration of the gas.

REFERENCES

1. Crowley, T.J. Causes of climate change over the past 1000 years. *Science* **2000**, 289 (5477), 270–277.
2. Karl, T.R.; Trenberth, K.E. Modern global climate change. *Science* **2003**, 302 (5651), 1719–1723.
3. Manabe, S.; Stouffer, R.J. Multiple-century response of a coupled ocean-atmosphere model to an increase of atmospheric carbon dioxide. *J. Climate*. **1994**, 7 (1), 5–23.
4. Johns, T.C.; Gregory, J.M.; Ingram, W.J.; Johnson, C.E.; Jones, A.; Lowe, J.A.; Mitchell, J.F.B.; Roberts, D.L.; Sexton, D.M.H.; Stevenson, D.S.; Tett, S.F.B.; Woodage, M.J. Anthropogenic climate change for 1860 to 2100

- simulated with the HadCM3 model under updated emissions scenarios. *Clim. Dynam.* **2003**, 20 (6), 583–612.
5. Cox, P.M.; Bets, R.A.; Jones, C.D.; Spall, S.A.; Totterdell, I.J. Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. *Nature* **2000**, 408, 184–187.
6. Gunter, W.D.; Bachu, S.; Law, D.H.S.; Marwaha, V.; Drysdale, D.L.; MacDonald, D.E.; McCann, T.J. Technical and economic feasibility of CO₂ disposal in aquifers within the Alberta sedimentary basin, Canada. *Energ. Convers. Manage.* **1996**, 37, 1135–1142.
7. Lackner, K.S. A guide to CO₂ sequestration. *Science* **2003**, 300 (5626), 1677–1678.
8. Pacala, S.; Socolow, R. Stabilization wedges: Solving the climate problem for the next 50 years with current technologies. *Science* **2004**, 305 (5686), 968–972.
9. Oelkers, E.H.; Schott, J., Eds. Geochemical aspects of CO₂ sequestration. *Chem. Geol.* **2005**, 217 (3–4), 183–405.
10. Broecker, W.S. Global warming: Take action or wait? *Jökull* **2005**, 55, 1–16.
11. Schrag, D.P. Confronting the climate – energy challenge. *Elements* **2007**, 3, 171–178.
12. Holloway, S. Storage of fossil fuel-derived carbon dioxide beneath the surface of the earth. *Annu. Rev. Energ. Env.* **2001**, 26, 145–166.
13. Friedmann, S.J. Geological carbon dioxide sequestration. *Elements* **2007**, 3, 179–184.
14. Benson, S.M.; Cole, D.R. CO₂ sequestration in deep sedimentary formations. *Elements* **2008**, 4, 325–331.
15. Emberley, S.; Hutcheon, I.; Shevalier, M.; Durocher, K.; Gunter, W.D.; Perkins, E.H. Geochemical monitoring of fluid–rock interaction and CO₂ storage at the Weyburn CO₂-injection enhanced oil recovery site, Saskatchewan, Canada. *Energy* **2004**, 29 (9–10), 1393–1401.
16. Xu, T.F.; Apps, J.A.; Pruess, K. Numerical simulation of CO₂ disposal by mineral trapping in deep aquifers. *Appl. Geochem.* **2004**, 19 (6), 917–936.
17. Lal, R. Sequestration of atmospheric CO₂ into global carbon pool. *Energy Environ. Sci.* **2008**, 1, 86–100.
18. McKinsey & Company. *Pathways to a Low Carbon Economy. Version 2 of the Global Greenhouse Gas Abatement Cost Curve*; McKinsey & Company: London, 2009.
19. Lal, R. Global potential of soil carbon sequestration to mitigate the greenhouse effect. *Crit. Rev. Plant Sci.* **2003**, 22, 151–184.
20. Grainger, A. Modeling the anthropogenic degradation of drylands and the potential to mitigate global climate change. In *Global Climate Change by Combating Land Degradation*; Squires, V.R., Glen, E.P., Ayoub, T.A., Eds.; UNEP: Nairobi, 1995; 193–199.
21. Lal, R. Soil carbon dynamics in cropland and rangeland. *Environ. Pollut.* **2001**, 116 (2002), 353–362.
22. Oldenburg, C. Migration mechanisms and potential impacts of CO₂ leakage and seepage. In *Carbon Capture and Sequestration: Integrating Technology, Monitoring, Regulation*; Wilson, E.J., Gerard, D., Eds.; Blackwell Publishing: Iowa, 2007; 127–146.
23. Bachu, S. Screening and ranking of sedimentary basins for sequestration of CO₂ in geological media in response to climate change. *Environ. Geol.* **2003**, 44, 277–289.

24. IPCC. Underground geological storage. In *IPCC Special Report on Carbon Dioxide Capture and Storage. Prepared by Working Group III of the Intergovernmental Panel on Climate Change*; Mertz, B., Davidson, O., de Coninck, H.C., Loos, M., Meyer, L.A., Eds.; Cambridge University Press: Cambridge and New York, 2005; 195–276.
25. Goldberg, D.; Slagle, A.L. A global assessment for deep-sea basalt sites for carbon sequestration. *Energ. Procedia*. **2009**, *1*, 3675–3682.
26. Czernichowski-Lauriol, I.; San Juan, B.; Rochelle, C.; Bateman, K.; Pearce, J.; Blackwell, P. Analysis of the geochemical aspects of the under-ground disposal of CO₂. In *Deep Injection Disposal of Hazardous and Industrial Waste*; Apps, J.A., Tsang, C.E., Eds.; Academic Press Inc.: New York, 1996; 565–583.
27. Gunter, W.D.; Wong, S.; Cheel, D.B.; Sjostrum, G. Large CO₂ sinks: Their role in the mitigation of greenhouse gases from an international, national (Canadian) and provincial (Alberta) perspective. *Appl. Energ.* **1998**, *61*, 209–227.
28. Bachu, S.; Gunter, W.D.; Perkins, E.H. Aquifer disposal of CO₂: Hydrodynamic and mineral trapping. *Energ. Convers. Manage.* **1994**, *35*, 269–279.
29. Koonin, S. The challenge of CO₂ stabilization. *Elements* **2008**, *4* (5), 293–296.
30. Broecker, W.S. CO₂ capture and storage: Possibilities and perspectives. *Elements* **2008**, *4* (5), 295–297.
31. Chu, S. Carbon capture and sequestration. *Science* **2009**, *325* (5948), 1595.
32. Haszeldine, R.S. Carbon capture and storage: How green can back be? *Science* **2009**, *325* (5948), 1647–1651.

Carbon Sequestration: Iceland

Anna María Ágústsdóttir

Soil Conservation Service, Hella, Iceland

Abstract

Mitigation of human-induced changes to the climate of the earth is among the most pressing environmental challenges. Reducing emissions of greenhouse gases (GHGs) and returning some of the carbon (C) back to earth are both essential tools if countries are to become C neutral with regard to the greenhouse effects. Terrestrial C sequestration involves capturing the atmospheric C through photosynthesis and subsequently storing them in soils, wetlands, and biota. Being C neutral refers to reducing national GHG emissions as far as possible and offsetting the remaining emissions by reduction elsewhere.

INTRODUCTION

The concept of carbon (C) neutrality is fairly new and refers to reducing national greenhouse gas (GHG) emissions as far as possible and offsetting the remaining emissions by reduction elsewhere (Fig. 1). It is a voluntary action taken by countries, companies, or individuals without legal requirements, setting emission reduction targets well beyond targets set by international agreements such as the United Nations Framework Convention on Climate Change (UNFCCC) and its Kyoto Protocol. The Vatican is the only C neutral state. Iceland, New Zealand, Norway, and Costa Rica were the first four countries to join the Climate Neutral Network, launched in 2008 by the United Nations Environment Programme (<http://www.unep.org/climateutral>). Costa Rica aims to reach C neutrality by 2021, Norway by 2030, and New Zealand in 2025–2040.

Governmental initiative is important to provide leadership and to foster the implementation of innovative solutions to reduce atmospheric C. It is important that offsetting is not viewed as encouraging ethical carelessness but attributing climate responsibility and credibility to the term “C neutrality.”

C sequestration is a method of net emission reduction that is immediately available using nature's technology. Early action increases the likelihood of avoiding the most severe consequences of global climate change according to the findings of the Intergovernmental Panel on Climate Change Fourth Assessment Report: Climate Change Mitigation.^[1] Sustainable land use improves land health, making it more capable of tolerating changes of future climate. When using soil C sinks, it is important to consider multiple objectives, avoiding large-scale monotonous plantations limited to climatic benefits, being aware of the issue of land rights and

having a clear focus on social justice. Voluntary emission goals can have more “value-added” attributes than those within international agreements or the compliance market such as developmental aims and environmental/sustainability benefits.

Financing revegetation is relatively cheap compared with the other more experimental projects. The Stern report was very particular about the importance of forestry and land use to the climate debate and to climate change mitigation, as a highly cost-effective way of reducing GHG emissions and with a potential to offer significant reductions quickly.^[2]

The time frame of achieving C neutrality may be difficult to envision. Over what time frame does the amount of C emitted have to be fully offset for the C balance to be zero? This depends very much on the balance between emission reductions, the rate of technological development and its employment, and the potential rate and amount of C sequestration.

THE ICELANDIC PROFILE

Iceland offers a good example of a country which could become C neutral within a few decades by combining reductions of emissions and restoring land quality for multiple benefits including C sequestration on some of the vast areas in Iceland that have become degraded over the 20th century.

The profile of GHG emissions for Iceland is unusual in many respects (Fig. 2).^[3] Firstly, electricity production and space heating are based on renewable energy sources (80% of total energy budget), resulting in very low emissions from these sectors. Secondly, more than 80% of emissions from the energy sector stem from mobile sources (transport, mobile machinery,

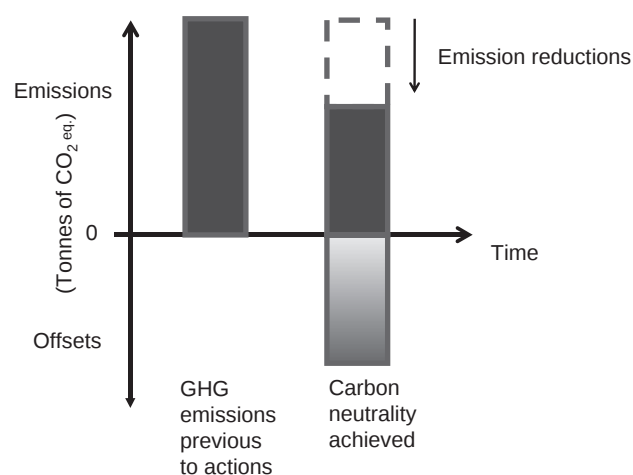


Fig. 1 C neutrality, which refers to that GHG emissions to the atmosphere are reduced as much as possible, and the remaining emissions are offset.

and fishing vessels). Thirdly, emissions from the land use, land use change and forestry sector are relatively high. Also, research has indicated that there are significant emissions of carbon dioxide (CO₂) from drained wetlands.^[3]

The Potential for Sequestration

The potential for sequestration lies in the land areas that could be revegetated for which the predominant soil type in Iceland is Andosol, which has a capacity to immobilize atmospheric CO₂. Iceland has suffered from severe soil erosion from the time of its settlement (874 A.D.; Fig. 3).

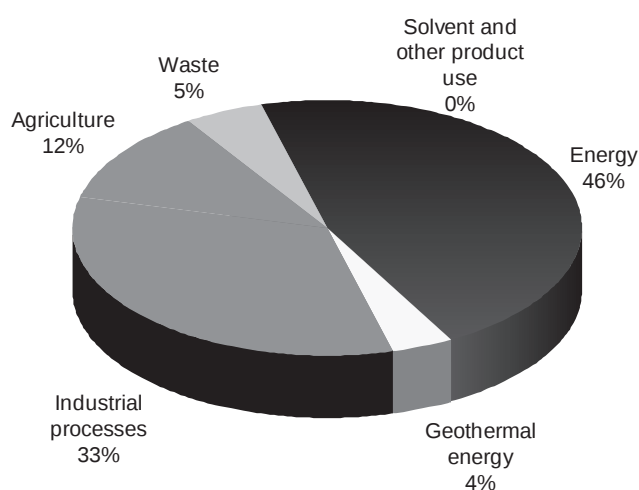


Fig. 2 Emissions of GHGs in Iceland by UNFCCC sector in 2007.

Source: From Arnalds, Thórarinsdóttir, et al.^[4]

Areas with considerable to extremely severe erosion cover about 40% of the country.^[4] In Iceland, C sequestration due to revegetation amounted to 534 Gg CO₂ or 12% of total emissions in 2007.^[3] These C offsets can potentially be increased significantly.

Revegetation in Iceland refers to restoring vegetation on barren land that has no agricultural value, leading to self-sustainable ecosystems with greatly improved ecosystem services. This land cannot recuperate on its own and needs managerial input to overcome ecological thresholds. Reclamation of soil and vegetation on eroded land is therefore considered to be a win-win strategy, linking restoration of soil fertility, biodiversity, ecosystems and their services, increased food security, and habitat conservation.

The legal basis and requirements for emission reduction in Iceland are weak. No laws or regulations cover commerce of C credits, and thus, no national C market is in place. This prevents land users and large-scale emitters in cooperating toward lowering emissions of GHG through C sequestration and therefore needs to be remedied.

Financial assistance or incentives are important for stimulating land improvement. This could include the following:

1. Strengthening policies and measures (the legal basis)
2. Direct policy changes, reclassifying subsidies, land-use zones, or indirect policy changes, e.g., changes in agricultural programs
3. Economic incentives, e.g., payments for environmental services, or disincentives, e.g., taxes
4. Direct infrastructure changes, e.g., damming of canals on peat land

A description of some of the means in use (but could be enhanced) or possibilities that could be employed to increase C sequestration in Iceland is detailed in the following sections.

Sustainable Land Use, Cooperation with Farmers, and Funding

“Regulation on sustainable utilization of land” is important to establish a code of practices for both public and private lands. In 2000, the Icelandic government and sheep farmers signed a voluntary contract on quality control in agriculture, including sustainable land use for which farmers can receive up to 22.5% more subsidies. If the land use criteria were applied stringently, they could lead to land improvement with associated increase in C sequestration.

The “Farmers Heal the Land” project provides incentives for better land health, assisting farmers to halt erosion, revegetate degraded land, and make it available



Fig. 3 Erosion escarpments at Arnardrangur in W-Skaftafellssýsla County show vividly the magnitude of soil erosion. The erosion has resulted in the complete removal of the soil profile down to bedrock or the lava surfaces, leaving isolated islands or areas with some remaining soil and degraded vegetation. The complete lack of vegetation on the eroded surfaces shows that erosion in this site is severe.

Source: Photo from Björn Barkarson, Soil Conservation Service of Iceland.

for sustainable use. The Soil Conservation of Iceland cooperates with 25% of Icelandic farmers in this program, providing consultation and financial support for seed and fertilizer, whereas farmers perform the actual work.

The “Land Improvement Fund,” “Landbótasjóður,” aims to move responsibility, initiative, and execution of soil conservation projects to local authorities, land owners, and non-governmental organizations. The Soil Conservation Service of Iceland provides consultation, funding, and supervision of projects.

Afforestation

Afforestation of treeless landscapes results in net C sequestration in biomass. In 1990, the woodland cover was 1% in Iceland and has great potential to be increased. In 2005, sequestration by woodlands amounted to 78 Gg CO₂ or 1.7% of total national emissions.^[3] In the forestry legislation, the goal is set to increase woodland cover to at least 5% of the lowland surface area below 400-m altitude by 2050.

Restoration of Wetlands

Extensive drainage of wetlands occurred in Iceland for cultivation purposes between 1945 and 1985 due to governmental subsidies. Some of the drained wetlands are used for cultivation or grazing, while others have been abandoned. Conditions for wetland restoration in Iceland are considered favorable, as most of the drained wetlands have not been intensively cultivated. Research has shown that reclaimed wetlands can sequester C, while drained wetlands are big emitters of CO₂. These drained wetlands emit about 1800 Gg CO₂ yr⁻¹, which

was calculated using UNFCCC default emission factors. A typical undisturbed wetland in Iceland sequesters about 0.051–0.099 Gg CO₂ km⁻² yr⁻¹ (H. Óskarsson, 2007, personal communication). If, for example, half of this land was to be restored, instead of emitting 900 Gg CO₂ yr⁻¹, the land could potentially sequester about 120–220 Gg CO₂ yr⁻¹. Restoration activities should be subsidized to speed up the recovery, thereby halting emission from drained wetlands, preventing nutrient loss from the land, and thereby restoring the land for future use.

C Sequestration with Geological Processes

Additional possibility is geological C sequestration. Mineral sequestration studies are underway in Iceland attempting to dispose the CO₂ emissions associated with geothermal energy production. Reinjection of CO₂-saturated fluids into wells or basaltic aquifers can acidify the groundwater, which may be neutralized by water–rock reactions and precipitation of carbonate minerals, thereby sequestering CO₂ as a solid phase in basalt.

CONCLUSION—CAN C NEUTRALITY BE ACHIEVED?

The global process of climate change may influence ecological, economic, and social activities and development at the regional and local level in all countries, making it one of the more profound challenges to sustainable development and poverty eradication.

The goal of C neutrality can only be achieved in a positive political environment. The climate issues have gained

higher visibility at the international and national scales. Increased public interest and scientific evidence for climate changes may prioritize funding for mitigation actions concerning climate issues.

Iceland may have the option of becoming a C neutral country, with emission reductions, with its renewable energy resources, with the vast potential for C sequestration by the restoration of degraded land, and with its more sustainable land use. Options need to be explored in a synergic manner, reaching other environmental and social goals at the same time. C offsetting is not the one and only panacea, but a way forward that is immediately available until new technology leads the way toward low-C societies. Iceland has been used here as an example; elsewhere, other options are likely to be more applicable toward achieving C neutrality.

REFERENCES

1. IPCC. *IPCC Fourth Assessment Report (AR4) Working Group III Report: Mitigation of Climate Change*, 2007. Available at <http://www.ipcc.ch/> (accessed October 2007).
2. Stern, N. *The Stern Review: The Economics of Climate Change*; Cambridge University Press: Cambridge, 2006; 712 pp.
3. Hallsdóttir, B.S.; Harðardóttir, K.; Guðmundsson, J.; Snorrason, A. *National Inventory Report Iceland 2009*; Submitted under the United Nations Framework Convention on Climate Change; UST:07. Environment and Food Agency, 2009; 190 pp.
4. Arnalds, O.; Thórarinsdóttir, E.F.; Metúsalemsson, S.; Jónsson, A.; Gretarsson, E.; Arnason, A. *Soil Erosion in Iceland* (English translation, originally published in Icelandic 1997); Soil Conservation Service, Agriculture Research Institute: Reykjavík, 2001; 121 pp.

Carbon Sequestration: Nutrient Management

M.C. Manna

Muneshwar Singh

R.H. Wanjari

Asit Mandal

A.K. Patra

Indian Institute of Soil Science, Bhopal, India

Abstract

The carbon (C) sequestration is a cost-effective strategy to mitigate climate change during the first few decades of the 21st century. There are five global C pools, and the third largest pool exists in soil and is estimated at 2.5 trillion tons (1-m depth). The conversion of natural ecosystems to agricultural ecosystems disturbs the soil ecological balance, soil processes, organic C, and biotic C pools. Extractive farming practices, low external inputs, and soil degrading land use all deplete terrestrial C pools. Approximately, 1.6 Pg C yr⁻¹ depletion can be attributed to deforestation and land use change in agriculture. In tropical agriculture, the application of manures at 10–15 Mg ha⁻¹ yr⁻¹ along with nitrogen, phosphorus, and potassium (NPK) increased soil organic C sequestration at the rate of 50.7–900 kg ha⁻¹ yr⁻¹ over 28–33 years of management. Globally, agricultural soils are estimated to potentially sequester 0.4–0.8 Pg C yr⁻¹ by the adoption of recommended management practices on croplands, 0.01–0.03 Pg C yr⁻¹ on irrigated soils, and 0.01–0.3 Pg C yr⁻¹ on grasslands. In rangelands, integrated nutrient management in horticulture crops, C sequestration rate was estimated at 58–1190 kg ha⁻¹ yr⁻¹. Improved nutrient management in existing agroforestry system sequestered C about 0.012 Tg C yr⁻¹. Globally, there is a C crisis in soil especially in tropical and subtropical ecosystems because of increased carbon dioxide emissions from soil. Therefore, there is huge challenge to maintain the soil C status. Nutrient management is a viable option for soil C sequestration and the mitigation of C emissions.

INTRODUCTION

The increase in concentration of greenhouse gases in the atmosphere is a major threat to agricultural food security and climate change in the 21st Century. Anthropogenic activities have led to notable changes over the 20th century in the earth's climate including the increase in global temperature by $0.6 \pm 0.2^{\circ}\text{C}$ at an average rate of increase of $0.17^{\circ}\text{C}/\text{decade}$. Since 1950, there has been a rise in sea level by 0.1–0.2 m, increase in precipitation by 0.5–1.0%/decade, coupled with an increase in frequency of extreme events and heavy precipitation by 2–4% over the 20th century.^[1] The conversion of natural ecosystems to agricultural and forestry ecosystems can disturb the natural ecological balance, carbon (C) pools, soil processes, and biotic C pools. Tropical and subtropical ecosystems have a great impact on global C cycle, environmental concerns, and socioeconomic issues. The numerous experiments from around the world have demonstrated that the conversion of tropical ecosystems to agricultural, pastoral, and silvicultural land uses substantially enhanced soil C pools. The magnitude of this variation, however, depends on the method of removing the

natural vegetation, agronomic practices, soil and nutrient management, and cropping systems.

C sequestration is an important strategy for the mitigation of climate change effects by means of storing C in soils and biomass. Soil cultivation with agricultural production may lead to significant loss of C from soil.^[2] Stagnating and declining crop yields are attributed to decline in soil quality caused by removal or burning of crop residues, imbalanced fertilization, and inappropriate use of irrigation water. Most agricultural soils have lost 50–70% of their original soil organic carbon (SOC) pool, and the depletion is exacerbated by further soil degradation and deforestation/desertification.^[3] The restoration of degraded soils, the conversion of agriculturally marginal lands to appropriate land use, and the adoption of recommended management practices (RMPs) on agricultural soils can reverse the land degradative trends and lead to SOC sequestration. Technological options for C sequestration on agricultural soils include adoption of conservation tillage, use of manures, and compost as per integrated nutrient management, precision farming strategies, conversion of monoculture to complex diverse cropping systems, meadow-based rotations winter cover crops, and

establishing perennial vegetation on contours and steep slopes. Proper soil and nutrient management may play key role for improving the C sequestration in the soil. The impact of ecosystem processes on global C cycle is comprehensive, but systematic information is needed on soil nutrient management for C sequestration. The objective of this entry is to discuss global C potential and its importance in agriculture and to discuss the best nutrient management practices for enhancing soil C sequestration in agriculture.

GLOBAL C POTENTIAL

There are five global C pools of which the largest oceanic pool is estimated at 38,400 Pg and is increased at the rate of 2.3 Pg C yr⁻¹ followed by the fossil/fuel (4130 Pg). The third largest pool is in the pedologic soil and is estimated at 2500 Pg to 1-m depth. The biotic pool is 602 Pg, and atmospheric pools are estimated at 800 Pg. The soil C pool has two distinct components: SOC pool estimated at 1550 Pg and soil inorganic C pool at 950 Pg.^[4] The SOC pool includes highly active humus and relatively inert charcoal C. Most methods for determining SOC do not account for resistant forms, such as inert charcoal, and thus, they find it difficult to quantify this source of organic C in global C budgets.

IMPORTANCE OF TERRESTRIAL C POTENTIAL

C concentration in the atmosphere is increasing at the rate of about 4 billion tons (2 parts per million) per year with transfer primarily from the fossil fuel, biotic, and soil pools. Fossil fuel combustion emits about 8 million t C yr⁻¹ annually, and deforestation and land use conversion emit another 1.6–2 million t C yr⁻¹, for a total of 9.6–10.80 billion t C emission yr⁻¹. Thus, roughly 8% C being photosynthesized by the biosphere is retained in the soil and biotic pools.

The long-term use of extractive farming practices and conversion of natural ecosystem into crop lands decline in soil quality and exacerbate food insecurity. Therefore, judicious use of fertilizer and integrated nutrient management as well as application of biosolids and other agricultural waste are of paramount importance to improve soil health, nutrient use efficiency, and crop productivity; to increase soil biodiversity; and to enhance the environment quality. Hence, the rate of soil C sequestration depends on the use of precious inputs such as crop residues, fertilizer, manure, compost, and good quality of irrigation water for which farmers must be compensated. Rate of SOC sequestration (kg ha⁻¹ yr⁻¹) may be 500–1000 for croplands, 50–500 for grazing lands, and 2–10 as pedogenic carbonates in drylands.^[5]

C SEQUESTRATION POTENTIAL UNDER DIFFERENT NUTRIENT MANAGEMENT IN AGRICULTURE

C emissions from agricultural activities contribute to enrichment of atmospheric carbon dioxide (CO₂), but C sequestration in agricultural soils, through the use of proper management practices, can mitigate this trend. Different sources of organic, either alone or with, chemical fertilizer application enhance C sequestration potential in arable crops, agroforestry, forage and fodder, and horticulture crops.

Nutrient Management in Arable Cropping System for C Sequestration

The restoration of SOC pool in arable lands represents a potential sink for atmospheric CO₂. The cropping system and soil type influence crop biomass under different fertilizers and manures application. Data from different long-term field experiments on rice–wheat, maize–wheat, soybean–wheat, sorghum–wheat, rice–wheat–jute, and maize–wheat–cowpea were analyzed to assess the impact of fertilization practices on soil C sequestration rate in Inceptisols, Alfisols, and Vertisols (Table 1).^[2,6–13] Long-term application of NPK or farm yard manure (FYM) significantly increased the C sequestration rate in rice–wheat system (55% higher SOC in FYM plots and 70% higher in NPK plots) than in maize–wheat cropping system.^[6] The incorporation of green manure with FYM sequestered relatively low organic C as compared to green manure with FYM and crop residue. Manna et al.^[14] observed that C sequestration rate was greater in Vertisol followed by Inceptisol than Alfisols under integrated nutrient management.

The humification process can be severely constrained by the lack of N, P, sulfur, and other building blocks of soil humus. The efficiency of C sequestration is reduced when C and N are not adequately balanced.^[14,15] Therefore, the SOC sequestration rate is enhanced by an increase in the application of biomass C and N.^[16] Liebig et al.^[17] observed that high N rate treatments increased C sequestration rate by 1.0–1.4 Mg ha⁻¹ yr⁻¹. The application of FYM at 10–15 Mg ha⁻¹ yr⁻¹ along with NPK increased SOC sequestration at the rate of 50.7–900 kg ha⁻¹ yr⁻¹ over 28–33 years.^[2,6–8,11,14]

Some of the strategies for enhancement of C sequestration potential are no-till farming with crop residue mulch and cover cropping (conservation agriculture), integrated nutrient management including use of compost and manure, and liberal use of biosolids. Globally, agricultural soils are estimated to potentially sequester 0.4–0.8 Pg C yr⁻¹ by adopting conservation agricultural practices, which represent 33.3–100% of the total potential of C sequestration in world soils.^[18]

Biofertilizers are an essential component of organic farming. They contain living microorganisms that

Table 1 Long-term effects of different nutrient managements on C sequestration rate under different cropping systems.

Cropping system	Nutrient management	C sequestration rate (kg ha ⁻¹)	Soil type and depth	References
Rice–wheat (33 years)	NPK	260	<i>Typic Ustochrept</i>	[6]
	FYM	310	(0–60)	
Maize–wheat (33 years)	NPK	80	<i>Typic Ustochrept</i>	[6]
	FYM	140	(0–60)	
Soybean–wheat (32 years)	NPK	332.5	<i>Typic Haplustept</i>	[7]
	NPK + FYM	936	(0–45)	
Soybean–wheat (30 years)	NPK	327	<i>Typic Haplustept</i>	[8]
	NPK + FYM	900	(0–45)	
Rice–wheat–Jute (30 years)	NPK	18.2	<i>Typic Entrochrept</i>	[9]
	NPK + FYM	50.7	(0–15)	
Sorghum–wheat (15 years)	NPK	189	<i>Typic Haplustert</i>	[9]
	NPK + FYM	295	(0–15)	
Maize–wheat–cowpea (31 years)	NPK	84	<i>Typic Haplustept</i>	[10]
	NPK + FYM	149	(0–15)	
Maize–wheat–cowpea (34 years)	NPK	380	<i>Typic Haplustept</i>	[11]
	NPK + FYM	513	(0–15)	
Soybean–wheat–maize (38 years)	NPK	146	<i>Typic Haplustert</i>	[2]
	NPK + FYM	333	(0–15)	
Soybean–wheat–maize (28 years)	NPK	89.2	<i>Typic Haplustert</i>	[12]
	NPK + FYM	228.5	(0–15)	
Rice–fallow (10 years)	FYM	110	<i>Aeric Endoaquept</i>	[13]
	FYM + GM	90	(0–15)	
	FYM + GM + crop residue	120		

Source: From Manna, Bhattacharyya, et al.^[2] ©2013 Elsevier; Kukal & Rehana-Rasool Benbi^[6] ©2009 Elsevier; Bhattacharyya, Pandey, et al.^[7] ©2010 Elsevier; Bhattacharyya, Prakash, et al.^[8] ©2010 Elsevier; Manna & Singh^[9] ©2001 Elsevier; Masto, Chhonkar, et al.^[10] ©2008 Springer; Mandal, Patra, et al.^[11] ©2007 Elsevier; Hati, Swarup, et al.^[12] ©2007 Elsevier; and Bhattacharyya, Roy, et al.^[13] ©2012 Elsevier.

colonize the rhizosphere and promote plant growth by increasing the supply of nutrients through N fixation or enhancing the availability of primary nutrients to the host plant by solubilizing P and other nutrients. The microorganisms in biofertilizers restore the soil's natural nutrient cycle and help in building soil organic matter. The studies indicated that biofertilizers sequestered about 1.4 t C ha⁻¹ yr⁻¹.

Several experiments in Europe have shown that the rate of C sequestration is greater with the application of manure than chemical fertilizer alone. Long-term use of manure when compared with chemical fertilizer was found 10% to have greater C sequestration over 100 years in Denmark,^[19] 100% over 144 years in Rothamsted,^[20] and 44% over 21 years in Sweden.^[21] Uhlen and Tveitnes^[22] reported that manure application would increase SOC sequestration at the rate of 70–227 kg ha⁻¹ over 37–74 years.

For sustainability of intensive cropping systems, it is desirable to grow a sequence of crop on a soil type. For example, rice or wheat crop sequence in Indo-Gangetic

Plain has to be replaced by introducing legume crops in the system in different ways like replacing rice by pigeon pea (*Cajanus cajan* L.) in summer or wheat by lentil in winter or introducing a summer green manure crop dhaincha (*Sesbania aculeata*) after the harvest of wheat and before the planting of rice. Growing of pea (*Pisum sativum* L.) or pigeon pea as compared with maize and gram or black gram as compared with soybean (*Glycine max* L. Merrill) substantially increased SOC.

Nutrient Management in Grassland and Fodder Crops for C Sequestration

The disturbance of grassland by means of removing biomass, changing the vegetation, or altering soil function is an integral part of traditional grassland management systems. Practices like overgrazing and burning can also deplete C sequestration in grassland systems.^[23] Harvesting a large proportion of plant biomass increases yields of useful material but at the cost of C inputs to the soil.^[24] It is widely accepted that continuous excessive grazing is

Table 2 C sequestration in different grassland species.

Tree/crop/cropping system C sequestration (Mg ha ⁻¹ yr ⁻¹)	C sequestration rate (Mg ha ⁻¹ yr ⁻¹)
<i>Vetiveria zizanioides</i>	15.24
Lemongrass	5.38
Palmarosa	6.14
Vetch (V)–maize (M)–oat(O)–soybean (S)–wheat (W)–soybean(S)	7.26
O–M–W–S	8.56
V–M–W–S	7.58
Ryegrass–M–R–S	8.44

Source: From Kaul, Mohren, et al.^[28] ©2010 Springer and Santos, Dieckow, et al.^[29] ©2011 Elsevier.

detrimental to plant communities and soil C sequestration.^[25] When management practices deplete the reversed soil C stocks, grassland ecosystem C stocks can be rebuilt and sequester atmospheric CO₂.^[26]

Agriculture management techniques such as livestock forage production also have the potential to augment soil C stocks through fertilization, irrigation, intensive grazing management, and sowing of favorable forage grasses and legumes. The global potential of SOC sequestration is estimated at 0.6–1.2 Pg C yr⁻¹, comprising 0.4–0.8 Pg C yr⁻¹ through the adoption of RMPs on cropland soils, 0.01–0.03 Pg C yr⁻¹ on irrigated soils, and 0.01–0.3 Pg C yr⁻¹ through improvements of rangelands and grasslands.^[27] The synthesis by Smith et al.^[23] suggest that adding manure or biosolids to soil could sequester C at the rates of between 0.42 and 0.76 t C ha⁻¹ yr⁻¹ depending on the region. A comparison of C sequestration of different grassland and fodder crops is presented in Table 2.^[28,29] The potential for C sequestration by *Vetiveria zizanioides* was the highest (15.24 Mg ha⁻¹ yr⁻¹) followed by oat–maize fodder–wheat–soybean in rotation (8.56 Mg ha⁻¹ yr⁻¹). The ryegrass–fodder followed by rice–soybean in rotation substantially improved C sequestration (8.44 Mg ha⁻¹ yr⁻¹).

Nutrient Management in Horticultural Crops for C Sequestration

Management practices for horticulture are more complex because of multiple species possessing varied phenological, physiological, and agronomic requirements. The conversion of long-term arable cropland to agro-horticulture resulted in a significant increase in C sequestration (Table 3).^[9] Under a system of different intercropped fruit trees, the cultivation of fruit trees—i.e., coconut (*Cocos nucifera* L.) intercropped with guava (*Psidium guajava* L.)—enhanced the soil biological activities approximately twofold after 38 years over 10 years of the same intercropped system, and C sequestration rate varied

Table 3 C sequestration rate under different horticulture-based systems.

Horticulture crop	C sequestration rate (kg ha ⁻¹ yr ⁻¹)
Site A (10 years)^a	
1. Coconut	58
2. Coconut + sapota	318
3. Vegetable	373
4. Coconut + guava	337
Site B (38 years)^a	
5. Coconut	160
6. Coconut + guava	910
7. Coconut + banana	1,060
8. Coconut + custard apple	780
9. Coconut + sapota	1,190
10. Coconut + litchi	1,130

^aNutrient such as organic manure (FYM) was applied for 3 years initially at the rate of 25 kg per pit mix with 0.225 kg, 0.450 kg, and 0.225 kg of N, P₂O₅, and K₂O per pit, respectively. In vegetables, crops manure was applied at the rate of 10 Mg ha⁻¹, and chemical fertilizer was applied at 80, 50, and 50 kg ha⁻¹ of N, P₂O₅, and K₂O per pit, respectively.

Source: From Manna & Singh^[9] ©2001 Elsevier.

from 58 to 373.0 and 160 to 1190 kg ha⁻¹ yr⁻¹ after 38 and 10 years, respectively.^[9]

Nutrient Management in Agroforestry for C Sequestration

C sequestration in agroforestry systems can be broadly divided into sequestration in aboveground and below-ground plant parts. The aboveground C sequestration rates in some major agroforestry systems around the world are highly variable, ranging from 0.29 to 15.21 Mg ha⁻¹ yr⁻¹^[30] and differ greatly depending on a number of factors, such as the agroclimatic region, type of system, site quality, previous land use, and management practices adopted. Agroforestry enhances C uptake by lengthening the growing season, expanding the niches from which water and soil nutrients are drawn and, in the case of N-fixing species, enhancing soil fertility. Improved management in existing agroforestry systems could sequester 0.012 Tg C yr⁻¹, while conversion of 630 million hectares of unproductive or degraded croplands and grasslands to agroforestry could sequester as much as 0.59 Tg C annually by 2040.^[31]

In a typical agroforestry system, about 69% of soil C in the profile was confined to the upper 40 cm soil layer wherein C stock ranged from 0.87 to 13.7 Mg C ha⁻¹ yr⁻¹ (Table 4).^[32] In a *Leucaena* agri-silvi system, C sequestration was the highest (13.7 Mg C ha⁻¹ yr⁻¹) followed by *Prosopis* silvi-pasture system (2.36 Tg C ha⁻¹ yr⁻¹). A mix of agroforestry with crop fields is a promising option to enhance C sequestration in soils. In general, agroforestry

Table 4 C sequestration rates in different agroforestry systems.

System	C sequestration (Mg ha ⁻¹ yr ⁻¹)
<i>Leucaena</i> agri-silvi system	0.87
<i>Anogeissus</i> agri-silvi system	1.36
<i>Casuarina</i> agri-silvi system	1.45
<i>Leucaena</i> agri-silvi system	13.7
<i>Eucalyptus</i> agri-silvi system	7.5
<i>Prosopis</i> silvi-pasture system	2.36
<i>Acacia</i> silvi-pasture system	1.29
<i>Dalbergia sissoo</i> silvi-pasture system	1.68

Source: From Srinivasarao, Venkateswarlu, et al.^[32] ©2013 Academic Press.

systems on the arid, semiarid, and degraded sites have a lower C sequestration potential than those on fertile humid sites, and the temperate agroforestry systems have relatively lower sequestration potential compared with the tropical systems under conservation agriculture. In agroforestry systems, the C stored in soil ranged from 30 to 300 Mg C ha⁻¹ up to 1-m depth.^[30]

FUTURE RESEARCH

Soil-stabilized C is the key component for enhancing soil quality. It could mitigate soil degradation caused by several biotic and abiotic factors. C sequestration in tropical ecosystems remains a challenging task for soil scientists. In this context, proper action, such as increasing SOC storage through adoption of RMPs, should be taken to save our soil from the soil degradation. These practices include no-till farming, compost on soil, legume-based rotations, use of mulch farming and conservation tillage, integrated nutrient management and manuring, implementation of agroforestry systems, restoration of eroded and salinized soils, and conversion of agriculturally marginal lands into restorative land uses. A natural way of C conservation can be achieved through maximum organic residues, cover crops, and no-tillage and with minimum emissions from fossil fuel combustion. Further, there is an additional potential of C sequestration in biomass especially by forest and other biota. Biosequestration of C, both by soil and by biota, is a truly win-win situation. While improving crop biomass productivity, these options also improve water quality and mitigate climate change by reducing the emission of CO₂ from soil.

REFERENCES

- IPCC. *Climate Change 2007: Mitigation of Climate Change*; Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate; Cambridge University Press: Cambridge, 2007.
- Manna, M.C.; Bhattacharyya, P.; Adhya, T.K.; Singh, M.; Wanjari, R.H.; Ramana, S.; Tripathi, A.K.; Singh, K.N.; Reddy, K.S.; Rao, A.S. Carbon fractions and productivity under changed climate scenario in soybean-wheat system. *Field Crop. Res.* **2013**, *145*, 10–20.
- Jenny, H.; Raychaudhuri, S.P. *Effect of Climate and Cultivation on Nitrogen and Organic Matter Reserves in Indian Soils*; ICAR: New Delhi, 1960; 126 pp.
- Batjes, N.H. Total carbon and nitrogen in the soils of the world. *Eur. J. Soil Sci.* **1996**, *47*, 151–163.
- Lal, R. Soil and India's food security. *J. Indian Soc. Soil Sci.* **2008**, *56* (2), 129–138.
- Kukal, S.S.; Rehana-Rasool Benbi, D.K. Soil organic carbon sequestration in relation to organic and inorganic fertilization in rice-wheat and maize-wheat systems. *Soil Till. Res.* **2009**, *102* (1), 87–92.
- Bhattacharyya, R.; Pandey, S.C.; Chandra, S.; Kundu, S.; Saha, S.; Mina, B.L.; Srivastva, A.K.; Gupta, H.S. Fertilization effects on yield sustainability and soil properties under irrigated wheat-soybean rotation of an Indian Himalayas upper valley. *Nutr. Cycl. Agroecosys.* **2010**, *86*, 255–268.
- Bhattacharyya, R.; Prakash, V.; Kundu, S.; Srivastva, A.K.; Gupta, H.S. Long term effects of fertilization on carbon and nitrogen sequestration and aggregate associated carbon and nitrogen in the Indian sub-Himalayas. *Nutr. Cycl. Agroecosys.* **2010**, *86*, 1–16.
- Manna, M.C.; Singh, M.V. Long-term effects of intercropping and bio-litter recycling on soil biological activity and fertility status of sub-tropical soils. *Bioresour. Technol.* **2001**, *76* (2), 143–150.
- Masto, R.E.; Chhonkar, P.K.; Singh, D.; Patra, A.K. Alternative soil quality indices for evaluating the effect of intensive cropping, fertilisation and manuring for 31 years in the semi-arid soils of India. *Environ. Monit. Assess.* **2008**, *136*, 419–435.
- Mandal, A.; Patra, A.K.; Singh, D.; Swarup, A.; Masto, R.E. Effect of long-term application of manure and fertilizer on biological and biochemical activities in soil during crop development stages. *Bioresour. Technol.* **2007**, *98*, 3585–3592.
- Hati, K.M.; Swarup, A.; Dwivedi, A.K.; Misra, A.K.; Bandyopadhyay, K.K. Changes in soil physical properties and organic carbon status at the topsoil horizon of a vertisol of central India after 28 years of continuous cropping, fertilization and manuring. *Agr. Ecosyst. Environ.* **2007**, *119*, 127–134.
- Bhattacharyya, P.; Roy, K.S.; Neogi, S.; Adhya, T.K.; Rao, K.S.; Manna, M.C. Effects of rice straw and nitrogen fertilization on greenhouse gas emissions and carbon storage in tropical flooded soil planted with rice. *Soil Till. Res.* **2012**, *124*, 119–130.
- Manna, M.C.; Swarup, A.; Wanjari, R.H.; Ravankar, H.N.; Mishra, B.; Saha, M.N.; Singh, Y.V.; Sahid, D.K.; Sarap, P.A. Long-term effect of fertilizer and manure application on soil organic carbon storage, soil quality and yield sustainability under sub-humid and semi-arid tropical India. *Field Crop. Res.* **2005**, *93*, 264–280.
- Paustian, K.; Collins, H.P.; Paul, E.A. Management controls on soil carbon. In *Soil Organic Matter in Temperate*

- Agroecosystems: Long-Term Experiments in North America*; Paul, E.A., Paustian, K., Elliot, E.T., Cole, E.V., Eds.; CRC Press: Boca Raton, 1997; 15–49.
16. Ma, L.; Yang, L.Z.; Xia, L.Z.; Shen, M.X.; Yin, S.X.; Li, Y.D. Long-term effects of inorganic and organic amendments on organic carbon in a paddy soil of the Taihu Lake Region, China. *Pedosphere* **2011**, *21*, 186–196.
 17. Liebig, M.A.; Varvel, G.E.; Doran, J.W.; Wienhold, B.J. Crop sequence and nitrogen fertilization effects on soil properties in the western corn belt. *Soil Sci. Soc. Am. J.* **2002**, *66*, 596–601.
 18. Lal, R. Soil carbon sequestration impacts on global climate change and food security. *Science* **2004**, *304*, 1623–1627.
 19. Christensen, S. Decomposability of recalcitrant soil carbon assessed by denitrification. *FEMS Microbiol. Lett.* **1991**, *86*, 267–274.
 20. Jenkinson, D.S.; Andrew, S.P.S.; Lynch, J.M.; Goss, M.J.; Tinker, P.B. The turnover of organic carbon and nitrogen in soil [and discussion]. *Philos. T. R. Soc. B.* **1990**, *329*, 361–368.
 21. Witter, E.; Mårtensson, A.M.; Garcia, F.V. Size of the soil microbial mass in a long-term field experiment as affected by different N-fertilizers and organic manures. *Soil Biol. Biochem.* **1993**, *25*, 659–669.
 22. Uhlen, G.; Tveitnes, S. Effects of long-term crop rotation, fertilizers, farm manure and straw on soil productivity. *Norw. J. Agr. Sci.* **1995**, *9*, 143–161.
 23. Smith, P.; Martino, D.; Cai, Z.; Gwary, D.; Janzen, H.; Kumar, P.; McCarl, B.; Ogle, S.; O'Mara, F.; Rice, C.; Scholes, B.; Sirotenko, O.; Howden, M.; McAllister, T.; Pan, G.; Romanenkov, V.; Schneider, U.; Towprayoon, S.; Wattenbach, M.; Smith, J. Greenhouse gas mitigation in agriculture. *Philos. T. R. Soc. B.* **2008**, *363*, 789–813.
 24. Wilts, A.R.; Reicosky, D.C.; Allmaras, R.R.; Clapp, C.E. Long-term corn residue effects: Harvest alternatives, soil carbon turnover, and root derived carbon. *Soil Sci. Soc. Am. J.* **2004**, *68*, 1342–1351.
 25. Conant, R.T.; Paustian, K. Potential soil carbon sequestration in overgrazed grassland ecosystems. *Global Biogeochem. Cy.* **2002**, *16*, 1143.
 26. Follett, R.F.; Kimble, J.M.; Lal, R., Eds. *The Potential of U.S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect*; CRC Press: Boca Raton, 2001; 401–430.
 27. Lal, R.; Follett, R.F.; Stewart, B.A.; Kimble, J.M. Soil carbon sequestration to mitigate climate change and advance food security. *Soil Sci.* **2007**, *172* (12), 943–956.
 28. Kaul, M.; Mohren, G.M.J.; Dadhwal, V.K. Carbon storage and sequestration potential of selected tree species in India. *Mitig. Adapt. Strateg. Glob. Change* **2010**, *15*, 489–510.
 29. Santos, N.Z.; Dieckow, J.; Bayer, C.; Molin, R.; Favaretto, N.; Pauletti, V.; Piva, J.T. Forages, cover crops and related shoot and root additions in no-till rotations to C sequestration in a subtropical Ferralsol. *Soil Till. Res.* **2011**, *111*, 208–218.
 30. Nair, P.K.R.; Nair, V.D.; Mohan, K.B.; Showalter, J.M. Carbon sequestration in agroforestry systems. *Adv. Agron.* **2010**, *108*, 237–307.
 31. IPCC. Emissions scenarios. In *Special Report on Emissions Scenarios*; Nakicenovic, N., Swart, R., Eds.; Cambridge University Press: Cambridge, 2000; 20 pp.
 32. Srinivasarao, C.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Kundu, S. Sustainable management of soils of dryland ecosystems of India for enhancing agronomic productivity and sequestering carbon. In *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press: Burlington, 2013; 253–329.

Carbon Sequestration: Sediments

Eswaran Padmanabhan

Geoscience, University Teknologi Petronas, Bandar Seri Iskandar, Malaysia

Abstract

The rise in global temperatures over the last few decades has been linked to the unprecedented increase in greenhouse gases. Storage or elimination of produced carbon dioxide (CO₂) has become a challenge since the 1970s. Various methods and options exist that will cater to excess C being removed from the ecosystem. However, the relative stability of these options is a challenge to the sequestration team. Sediments in onshore areas can be found in several types of environments. Of these, wetlands are recognized as being able to hold at least a quarter of the global organic C content. Spatial and temporal variability in C contents as well as sequestration potential are important issues to be considered when dealing with sediments as a sequestration media. Offshore methods have a different set of challenges. The density and dissolution kinetics of CO₂ are critical in determining a proper sequestration with minimal need for postinjection monitoring for leakages. Despite advances in technology, public perception of sequestration projects is not devoid of skepticism and distrust. The main concerns revolve around release of large amounts of CO₂ into the natural waters or atmosphere culminating in irreparable damage to the ecosystem. As the production of CO₂ is projected to increase with time, it is inevitable that management policies and strategies need to be reformulated in order to successfully curb the impacts of excess CO₂ in the atmosphere.

INTRODUCTION

Global temperatures are rising. The temperature increase has been linked to more intense precipitation events, decrease in ocean thermohaline circulation, and also rising sea levels due to glacial melting. Rising sea levels can cause flooding of low-lying coastal areas that can result in loss of life and property. Global temperature increase could also affect global ecosystems resulting in unpredictable plant distribution and colonization, extinction of several species, and cause outbreaks of diseases.

The study of ice cores from the Antarctica has established a strong relationship between increase in atmospheric temperature and rise in carbon dioxide (CO₂) content in the air. Studies have shown that as temperature rises, solubility of CO₂ in oceans decreases thereby inducing CO₂ to diffuse out of the oceans. This increases the concentration of CO₂ in the atmosphere. Therefore, it is important to know the best means of reducing CO₂ concentration in the atmosphere. One such way is known as CO₂ sequestration.

Carbon (C) is sequestered naturally in soils and sediments. The distinction between soils and sediments lies in the ability of the former to sustain life. The term “sediment,” on the other hand, is traditionally reserved for loose and unconsolidated material that is found at the earth’s surface. In terms of sequestration, there is a lot of similarities in the mechanisms of accumulation, storage, sequestration, and eventually depletion through destruction.

Storing CO₂ in deep-sea sediments was first proposed by Koide et al.^[1] who proposed storing a mixture of CO₂, clay, ash solutions, and CO₂(l) in three depth regimes: 1) shallow subseabed (shallower than 300 m); 2) deep subseabed (300–3,700 m); and 3) super deep subseabed (more than 3,700 m).

This entry focuses on C sequestration in sediments and excludes any discussion of sequestration at or above the ocean floor.

C SEQUESTRATION IN TERRESTRIAL SEDIMENTS

Cultivation of soils causes depletion of organic C pool. This depletion is explained as being due to oxidation or mineralization due to breakdown of aggregates leading to exposure of C, and change in temperature and moisture regimes,^[2,3] leaching and translocation as dissolved organic C or particulate organic C,^[2] and accelerated erosion by water runoff or wind.^[4,5]

In coastal environments, it is believed that a significant fraction of CO₂ is sequestered for centuries by certain coastal habitats, e.g., seagrasses, mangroves, and salt marshes. These systems should be managed and conserved for C sequestration. However, these coastal systems are being lost due to pollution and mismanagement. Thirty-five percent of the mangrove systems in the world have been lost while about 29% of the seagrass system globally have been lost. Annual loss of coastal systems is estimated

at about 2%. This loss basically means that a very efficient and natural C mitigation mechanism is being lost gradually.

Degradation of the coastal ecosystems initiates a rapid release of stored C. It has been estimated that draining a coastal wetland such as a marsh or a mangrove releases about $0.25 \text{ Mt CO}_2 \text{ km}^{-2} \text{ m}^{-1}$ of sediment lost.

The estimated C stocks in salt marshes are about $1.3\text{--}4.9 \text{ t C ha}^{-1}$ for material above the ground and about $0.9\text{--}13.9 \text{ t C}^{-1}$ for below the ground biomass. Measured C contents are somewhat higher in their values compared to the estimated values. Unfortunately, the exact areal extent of these marshes worldwide is unknown. In addition to that, the exact sequestration and mechanism of C dynamics in such systems are also poorly understood.

Deserts on the other hand have been estimated to have a mean C content of 64 t C ha^{-1} .^[6] It is recognized that the C stocks in deserts can be high since the aridity perturbs microbial decomposition (typical stocks of $14\text{--}270 \text{ t C ha}^{-1}$). Naturally, deserts in tropical environments have lower stocks compared to temperate and boreal regions. However, the C balance in deserts remains a gap in knowledge.^[7]

Wetlands occupy about 5% of the land globally and are typified by water-logging conditions. Productivity among wetlands varies depending on the type of the wetland, climatic condition, and vegetation communities. Climate (temperature and moisture enhanced microbial activity) and quality (composition) of organic matter determine the accumulation potential of C in wetlands. These characteristics make wetlands one of the most effective ecosystems for storing soil C.^[8] Wetlands may hold up to 20–25% of world's organic C.^[9] Since C fluxes and pools are spatially and temporally variable in different types of wetlands, the actual quantity of C stored in each system also varies.

Peat lands (Fig. 1) sequester significant quantities of the world's C. The total amount of C in standing vegetation and peat soil has been estimated between 20% and 35% of the



Fig. 1 Peatlands being cleared for land development.

total terrestrial C.^[10] It has been estimated that northern peat lands alone contain more than 500,000 million tons of C. Many peat lands produce only 20% of the methane produced by shallow water wetlands.^[10] In addition, processes vary at different levels with a peat deposit. The lower levels of peat produce methane while the upper levels at least partially oxidize methane released from the lower levels. Although drainage of peat lands has been shown to reduce methane production, other studies have indicated that this may be more than compensated by the methane production in the associated drainage ditches. CO_2 release will increase dramatically to levels as high as $15 \text{ t C ha}^{-1} \text{ yr}^{-1}$ in the temperate zone and $50 \text{ t C ha}^{-1} \text{ yr}^{-1}$ in the tropics through decomposition of peat after drainage.^[11]

River microterraces can show large variations in the amount and type of C sequestered. In Malaysia, a river terrace sequence developed from the weathered sediments of granite and sedimentary sequences shows differences in height as well as internal drainage.^[12] The sediments in this terrace were grouped (Fig. 2) under three classes: 1) low-level microterrace (drainage classes ranging from poor to very poor drainage); 2) intermediate-level microterrace (drainage classes ranging from well-drained to moderately well-drained drainage); and 3) high-level microterrace (drainage classes ranging from somewhat imperfectly drained to somewhat poorly drained drainage).

The sediments of the low-level terrace have very low C contents. The upper layers are dominated by clay whereas fine sand dominates layers below the lithologic discontinuity. Porosity and clay percentage decreases with depth. In accordance with this, the moisture retained has a general decreasing trend with depth as well. High amounts of moisture retained at the surface horizons are attributed to the high organic C content. The surface horizon has a high organic matter content which decreases with depth. In a slightly elevated position, the sediments still experience poor drainage conditions. In most parts, the sediment has a clayey texture. The surface horizon has a high content of organic matter typically about 4%. Lower horizons have very low organic matter. The high concentration of organic

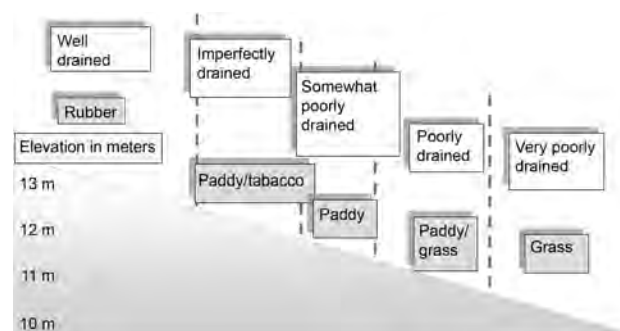


Fig. 2 Variation of internal drainage in sediments of river terraces. The figure also illustrates the type of land use practiced in Malaysia.

matter in the surface is attributed to poor drainage conditions.

In the intermediate-level microterrace, the sediments have a somewhat poorly drained internal drainage. The surface horizon is exposed to seasonal flooding and water logging. The material is generally clayey and is subject to fluctuating water tables. In situations where drainage may improve locally, the sediment would have a much lower organic matter content compared with the lower microterrace level sediments. This indicates that drainage has a control on the amount of organic matter and C in terraces. Organic matter decreases with depth. The water retention at various pressures and available water are quite low in the top horizon and change erratically with depth. This has been attributed to the decreasing clay contents.

Sediments at the high-level microterraces have better internal drainage. The internal drainage is classified as somewhat imperfectly drained. This material has a clayey surface layer and a lighter textured subsurface layer. Organic matter content shows a maximum at the surface with concomitant decrease with depth. The sediments here have generally a much lower organic matter content compared to all other topographic positions discussed so far. This is attributed to the better drainage conditions thereby inducing better turnover of the organic matter.

An alternative technology is being tried out. The application of biochar (charcoal or biomass-derived black C) could lead to a significant, long-term sink for atmospheric CO₂ in terrestrial ecosystems. Biochar apparently improves fertility and increases crop production. Application of biochar could lead to sequestration of about 50% of the initial C compared to the 3% or less amounts retained after burning. The C derived in this manner is claimed to be more stable than those derived from burning or direct application of biomass on land.

C SEQUESTRATION IN MARINE SEDIMENTS

CO₂(l) is denser than water at high pressures and low temperatures due to its high degree of compressibility. It has been stated^[13] that the injected CO₂(l) at depths of 3,000 m sinks and forms a lake of CO₂(l) on the seafloor. Ocean currents can mix the injected CO₂(l) and release the CO₂ into the atmosphere.^[14] The combination of high pressures and low temperatures that is needed to compress CO₂(l) to a density greater than the pore fluid creates a condition that is similar to the condition necessary for CO₂ hydrates to form. The region where this condition exists is also known as the hydrate formation zone (HFZ). The HFZ starts from this point and extends downward until the temperature rises above the boundary of the hydrate stability field.^[15]

In order to prevent this from happening, CO₂ is to be injected below the seafloor. The injected CO₂ is denser than the pore fluids. This difference in density creates a buoyancy effect that results in gravitational stability.

Subsequent to injection, the CO_{2(g)} will slowly dissolve into CO_{2(aq)} that is denser than the surrounding pore fluid.^[16] CO₂ requires a lot of pore fluids to fully dissolve. Interaction of this CO₂ with calcareous sediments in the host rock determines the CO_{2(aq)}, H₂CO₃, HCO₃⁻, and CO₃²⁻ balance. The total dissolution of carbonate minerals is predicted to be small (i.e., <1%) before the pore fluids become saturated. After injection, the CO₂(l) injected below the HFZ is expected to flow upward until it reaches the bottom the HFZ.^[15] These authors believe that CO₂ hydrates will form once the CO₂(l) reaches the bottom of the HFZ. The formation of the hydrates tends to clog up pore channels thereby creating a cap of limited permeability. Additional CO₂ flowing up from the injection point is expected to become physically trapped beneath the hydrate cap.^[15] This trapping mechanism forces the additional CO₂ to spread laterally thereby resulting in a larger storage area.

The stability of the hydrates is maintained as long as it is in contact with pore fluids saturated with CO_{2(aq)}. It has been clarified further^[15] that if the CO₂(l) to CO_{2(aq)} dissolution kinetics were rapid, then the pore fluid in contact with pure CO₂(l) plume will be saturated in CO_{2(aq)} until the entire plume of CO₂(l) dissolves. As such, they stated that the CO₂ hydrate cap will not dissolve until the CO₂(l) plume has fully dissolved.

ENVIRONMENTAL IMPACTS

C sequestration has been perceived with mixed feelings by the society. The main concerns have been with long-term stability of seals in sequestration projects, impact on fauna and flora, as well as stability of coral platforms and geological storage media. In addition to this, there has been some concern about sudden release of CO₂ plumes into the atmosphere from both offshore and onshore sequestration sites. These concerns have been the reason for a leap in technological advancements designed to minimize these effects.

In the initial stages, it was estimated that a large quantity of CO₂ may be dissolved in deep ocean waters. These estimates were made at a time when the knowledge on ocean biogeochemistry was still incomparable to the wealth of information we had. We know that the pH of the surface ocean has been reduced by approximately 0.1 units since preindustrial times. Further addition of C into the ocean is estimated to decrease the average ocean pH by about 0.3 units. Studies have shown that the pH of the ocean waters increases away from the point of injection. The nonswimming marine organisms (e.g., zooplankton, bacteria, and benthos) residing at depths of about 1,000 m or greater will be affected by both the level of pH change and the duration of exposure to the lowered pH conditions. As such, it is pertinent that any sequestration of CO₂ in marine environment be done preferably in deep sediments or sedimentary formations.

Therefore, it is evident that environmental impact assessment is very important in any sequestration project. Ultimately, this may also be the most important factor that determines the viability of any sequestration projects in offshore areas. The viability of such projects also depends on social and political considerations and that this strategy will require all stakeholders (private, public, and non-governmental organizations) be included in research and development of best sequestration strategies.

MANAGEMENT STRATEGIES TO MAINTAIN/ENHANCE C SEQUESTRATION IN SEDIMENTS

Essentially, it is important that we prevent the release of stored C in sediments as a consequence of mismanagement or mishandling. There obviously is a need to maintain a high sequestration capacity in sediments or enhance the sequestration capacity when the resident capacity is very low. Knowledge obtained from managing soils to increase C sequestration could also be used in managing sediments for the same purpose. The fact that sediments tend to have low contents of C is the first factor that encourages C sequestration. The estimates indicate that sediments could, if other conditions were right, sequester close to 1 Gt C yr⁻¹. Cooler and wetter conditions tend to favor C sequestration. It is also important to remember that the quantity of C stored is finite and the process itself is reversible.

Other techniques to increase C sequestration in terrestrial sediments include mulching, maintaining water balance (moisture), proper nutrient management in cropping systems, implementation of agroforestry practices in suitable areas, abstinence from indiscriminate drainage designs, species introduction, erosion control, and organic as well as nutrient amendments.

Conservation and management of sediments in coastal marine ecosystems could be an efficient and cheap natural method for removing CO₂ from the atmosphere. Regrettably, there are insufficient policies globally at this moment to support the management of sediments of the marine ecosystems to sequester this gas. Some research questions have come forth when considering the sequestration rates of C in these systems. It is unclear as to what is the impact of degradation on the specific sequestration rates. A popular research question asked is how are the sequestration rates and C stocks in sediments affected by climatic changes.

The requirements can only be realized if certain policies are implemented globally. Degradation and loss of natural habitats require urgent attention in many countries, especially in the tropics. Ecosystem resilience in relation to C pools requires a careful assessment through conservation and restoration. This will reduce the instances of desertification in many locations.

CONCLUSION

Onshore sediments tend to behave differently with regard to sequestration. In all cases, moisture and temperature are very important in determining the amount and length of sequestration. Other factors are also recognized. The use of charred materials appears to be an effective means of C sequestration.

The discussion on sequestration in sediments below 3,000 m at high pressure and low temperature suggests a possible permanent solution to CO₂ sequestration problems in offshore areas. The success of this approach requires CO₂ to be present below a layer of more buoyant pore fluid. The existing theories predict that the sequestered CO₂ needs to dissolve into the pore fluids. It is predicted that the dissolving of CO₂ into the pore fluids will take thousands of years. Concurrently, the CO_{2(aq)} will sink that of the pore fluids. If this theory works, then CO_{2(l)} could be stored in deep-sea sediments at high pressures and low temperatures without any investment in monitoring or verification technology. This would then provide untold financial benefits to the industry.

The environmental concerns associated with C sequestration have been received with mixed feelings. Most concerns are with the C being released in huge amounts somewhere in the near future. As outlined in this entry, any abnormal amounts of C released can result in fatality of fauna and flora. Therefore, it is imperative that the technology employed is able to contain the excess C for the intended lengthy period, i.e., to last thousands of years. To achieve this, certain specific management policies are outlined and relevant strategies have to be initiated globally.

REFERENCES

1. Koide, H.; Shindo, Y.; Tazaki, Y.; Iijima, M.; Ito, K.; Kimura, N.; Omata, K. Deep seabed disposal of CO₂—The most protective storage. *Energ. Convers. Manage.* **1997**, *38*, 253–258.
2. Lal, R. Global soil erosion by water and carbon dynamics. In *Soils and Global Change*; Lal, R., Kimble, J.M., Levine, E., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1995; 131–142.
3. Lal, R. Soil carbon dynamics in cropland and rangeland. *Environ. Pollut.* **2002**, *116*, 353–362.
4. DeJong, E.; Kochanoski, R.G. The importance of erosion in the carbon balance of prairie soils. *Can. J. Soil Sci.* **1988**, *68*, 111–119.
5. Rhoton, F.E.; Tyler, D.D. Erosion-induced changes in properties of a fragipan soil. *Soil Sci. Soc. Am. J.* **1990**, *54*, 223–228.
6. Trumper, K.; Bertzky, M.; Dickson, B.; van der Heijden, G.H.; Jenkins, M.; Manning, P. *The Natural Fix? The Role of Ecosystems in Climate Mitigation*; Cambridge: United Nations Environment Programme, UNEP-WCMA, 2009; 67 pp.

7. Schlesinger, W.H.; Belnap, J.; Marion, G. On carbon sequestration in desert ecosystems. *Glob. Change Biol.* **2009**, *15* (6), 1488–1490.
8. Schlesinger, W.H. *Biogeochemistry: An Analysis of Global Change*, 2nd Ed.; Academic Press: San Deigo, 1997.
9. Gorham, E. Northern peatlands: Role in the carbon cycle and probable responses to climate warming. *Ecol. Appl.* **1991**, *52*, 182–195.
10. IGBP Terrestrial Carbon Working Group. The terrestrial carbon cycle: Implications for the kyoto protocol. *Science* **1998**, *280*, 1393–1394.
11. Immirzi, C.P.; Maltby, E. *The Global Status of Peatlands and their Role in the Carbon Cycle*. Wetlands Ecosystems Research Group, Report 11; Department of Geography, University of Exeter: London, 1992; 118 pp.
12. Padmanabhan, E.; Eswaran, H. Variations in N₂ fluxes in some hydrosequences in Malaysia. In *Proceedings of International Seminar on Increased Agricultural Nitrogen Circulation in Asia: Technological Challenge to Mitigate Agricultural Nitrogen Emissions*, Taipei, Taiwan, Sept 27–28, 2011; Chen, Z.S., Shindo, J., Eds.; GIS Convention Center of National Taiwan University: Taipei, 2011; 137–148.
13. Fer, I.; Haugan, P. Dissolution from a liquid CO₂ lake disposed in the deep ocean. *Limnol. Oceanogr.* **2003**, *48*, 872–883.
14. Jain, A.K.; Cao, L. Assessing the effectiveness of direct injection for ocean carbon sequestration under the influence of climate change. *Geophys. Res. Lett.* **2005**, *32*, L09609.
15. House, K.Z.; Schrag, D.P.; Harvey, C.F.; Lackner, K.S. Permanent carbon dioxide storage in deep-sea sediments. *PNAS* **2006**, *103* (38), 12291–12295.
16. Song, Y.; Chen, B.; Nishio, M.; Akai, M. The study on density change of carbon dioxide seawater solution at high pressure and low temperature. *Energy* **2005**, *30*, 2298–2307.

Carbon Sequestration: Semiarid Regions of India

Cherukumalli Srinivasa Rao

Sumanta Kundu

Srinivas K.

Central Research Institute for Dryland Agriculture, Indian Council of Agricultural Research (ICAR), Hyderabad, India

Abstract

Severe soil organic carbon (SOC) depletion is a major constraint in semiarid regions of India because it directly influences soil quality, crop productivity, and sustainability. The magnitude of SOC stocks in the semiarid bioclimate is estimated at 2.9 Pg. Sorghum, finger millet, pearl millet, maize, rice, groundnut, soybean, cotton, food legumes, etc. are the predominant crop production systems. Data from long-term experiments on major rainfed production systems in India show that higher amount of crop residue carbon (C) input ($\text{Mg ha}^{-1} \text{ yr}^{-1}$) is returned to soil in soybean–saflflower (3.37) system practiced in Vertisols of central India. The long-term addition of chemical fertilizer and organic amendments improved the SOC stock. For every 1 Mg ha^{-1} increase in SOC stock in the root zone, grain yield (kg ha^{-1}) increased by 13, 101, 90, 170, 145, 18, and 160 for groundnut, finger millet, sorghum, pearl millet, soybean, lentil, and rice, respectively. Long-term cropping without using any organic amendment and/or mineral fertilizers resulted in severe depletion of the SOC stock, which was highest in groundnut–finger millet system ($0.92 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$) in Alfisols. The critical levels of C input requirements for maintaining SOC at the antecedent levels ranged from 1.1 to 3.5 $\text{Mg C ha}^{-1} \text{ yr}^{-1}$, depending on soil type and production system. Promotion of sustainable use of soil and water resources includes prohibiting residue burning, reducing deforestation, promoting integrated farming systems, and facilitating payments for ecosystem services can improve soil quality through increase in SOC sequestration and agronomic productivity of rainfed agroecosystems.

INTRODUCTION

The global area of semiarid tropics or dryland tropics, estimated at around 650 million hectares, covers 55 countries and is home to more than 2 billion people.^[1] According to the latest estimates, global rainfed cropland is 1132 million hectares, which is 2.78 times of the world's net irrigated area (407 Mha).^[2] About three fourths of the area under arid, semiarid, and dry-humid bioclimatic zones is rainfed and accounts for 57% of the net cultivated area in India. These areas contribute to 40% of national food grain production and support two-third of the total livestock population of the country. Rainfed areas contribute the production of 87% of coarse grain cereals and pulses, 77% of oilseeds, 60% of cotton, 90% of soybean, and 50% of fine cereals including rice (*Oryza sativa*), wheat (*Triticum aestivum*), maize (*Zea mays*), and sorghum (*Sorghum bicolor*). Although double cropping in rainfed regions is rare, there is a vast potential of rainfed area in terms of food grain production. Region specific investments and management strategies are required for rainfed agroecologies, as they are complex, diverse, fragile, risk prone, and under-invested. The net sown area in the country was 141.6 Mha during 2010–2011 and it did not increase much during 1970–2010, but the human and livestock populations have been steadily

increasing. The slowdown in productivity growth has been attributed to resource degradation and falling factor productivity in major production systems.

Soil organic carbon (SOC) concentration is a key indicator of soil quality and productivity. Restoring the quality of degraded soils necessitates increasing SOC concentration in the root zone. The term “soil C sequestration” implies removal of atmospheric carbon dioxide (CO_2) by plants through photosynthesis and transfer of biomass C into soil as humus. The strategy is to increase SOC density, improve depth distribution of SOC, and stabilize SOC by encapsulating it within stable microaggregates so that C is protected from microbial processes and has a long mean residence time (MRT). In this context, adopting recommended management practices (RMPs) in agroecosystems that create a positive soil C budget is an important strategy for SOC sequestration. Thus, land use change can be an important instrument for SOC sequestration. However, the SOC sink capacity depends on the antecedent level of SOC, climate, profile characteristics, and management. Incorporation of SOC into the subsoil by establishing plants with a deeper root systems or illuviation through bioturbation and pedogenesis can increase MRT of SOC. The SOC concentration in the surface layer usually increases with increasing inputs of biosolids,^[3] although the specific empirical

relation depends on soil moisture and temperature regimes, nutrient availability [nitrogen (N), phosphorous (P), potassium, and sulfur], texture, and climate. In addition to the quantity of input, quality of biomass can also be important in determining the SOC pool. Most of the research done thus far on SOC sequestration in soils of agroecosystems is confined to cold and temperate regions. Some of the studies were conducted in tropical and subtropical regions of India.^[4–15] The objective of this entry is to synthesize the available information on carbon stocks in different soil types in semiarid India, carbon sequestration scenarios in major rainfed production systems, minimal carbon input requirement for zero change in SOC stock, and important management practices and strategies that influence C sequestration and policy interventions required in Indian context.

SOIL CARBON STOCKS OF DRYLAND AGROECOSYSTEMS IN INDIA

The SOC storage is strongly related to climate (temperature and rainfall). Semiarid bioclimate consists of major parts of central and southern Peninsula, which extends up to western and northwestern part of the country and comprises of 17 agroecological subregions and 7 agroclimatic zones. By and large, mean annual temperature in semiarid bioclimate ranges between 25°C and 27°C and mean annual precipitation (MAP) ranges between 500 and 1000 mm. This bioclimate is characterized by a type of vegetation, which ranges from bushy thorns and grasses to deciduous forests. The total SOC stock in the semiarid bioclimate is 2.9 Pg, which is nearly 30% of the total SOC stock of the country.^[13] SOC stock varied among major rainfed soil types of India. Vertisols and associated soils contain higher carbon stocks, followed by Inceptisols, Alfisols, and Aridisols.^[16] The SOC stocks range from 26.69 to 59.71 Mg ha⁻¹ with a mean of 43.74 Mg ha⁻¹ in Inceptisols, from 23.28 to 49.83 Mg ha⁻¹ with a mean of 30.82 Mg ha⁻¹ in Alfisols, from 28.60 to 95.90 Mg ha⁻¹ with a mean of 46.38 Mg ha⁻¹ in Vertisols, and from 20.10 to 27.36 Mg ha⁻¹ with a mean of 23.73 Mg ha⁻¹ in Aridisols. Vertisols with smectite as dominant mineral have larger SOC stocks than illitic Inceptisols and kaolinitic Alfisols.

CARBON SEQUESTRATION POTENTIAL OF DIFFERENT RAINFED CROPPING SYSTEMS UNDER VARIED NUTRIENT MANAGEMENT

Some of the long-term manurial experiments established in late 1970s to early 1990s under the auspices of the All India Coordinated Research Project on Dryland Agriculture are located in different ecoregions in semiarid climates such as Anantapur (Andhra Pradesh), Bangalore (Karnataka), Solapur (Maharashtra), Indore (Madhya Pradesh), Sardar Krushinagar (Gujarat), and Varanasi (Uttar Pradesh;

Table 1). Common treatments across seven experiments are control (no fertilizer or organics), 100% recommended dose of fertilizers (RDF), 50% RDF + 50% organics, and 100% organics.^[11,13]

Nutrient Management Effects on SOC Stock and SOC Sequestration Rate

Nutrient management plays a vital role on SOC sequestration. Better nutrient management, particularly the application of organics not only increases the carbon input directly but also increases the plant biomass, particularly root biomass. Results from long-term experiments show the highest SOC stock (Mg ha⁻¹) in 50% RDF + 4 Mg ha⁻¹ groundnut shell (47.2), 10 Mg ha⁻¹ FYM + 100% NPK (85.7), FYM + 100% NPK (73.0), 25 kg N (sorghum residue) + 25 kg N (*Leucaena*) (68.5), 50% RDN (fertilizer) + 50% RDN (FYM) (25.5), 6 Mg FYM + N₂₀P₁₃ ha⁻¹ (69.9), and 100% organic (FYM) (27.5) in groundnut (*Arachis hypogaea*), finger millet (*Eleusine coracana*), groundnut–finger millet rotation, sorghum, pearl millet (*Pennisetum glaucum*), soybean (*Glycine max*), and rice-based systems, respectively. Improved nutrient management practices were identified in major rainfed production systems on the basis of the mean rate of SOC sequestration. The average SOC sequestration rate (Mg C ha⁻¹ yr⁻¹) with different management treatments was as follows: 0.57 for 50% RDF + 4 Mg ha⁻¹ groundnut shell, 0.57–0.72 for 10 Mg ha⁻¹ FYM + 100% NPK, 0.65 for 25 kg N ha⁻¹ (sorghum residue) + 25 kg N (*Leucaena* clippings), 0.24 for 50% RDN (fertilizer) + 50% RDN (FYM), 0.79 for 6 Mg ha⁻¹ FYM + 20 kg N + 13 kg P, and 0.32 for 100% organic (FYM) (Table 1).

Productivity Enhancement due to Increase in SOC Stock in the Root Zone

Increase in agronomic productivity through increased SOC stock in the root zone was evaluated in major rainfed production systems. Results showed that 1 Mg C ha⁻¹ yr⁻¹ increase in SOC stock in the root zone led to a significant increase in yield in several rainfed crops. These increases were 13, 101, 90, 170, 145, 18, and 160 kg ha⁻¹ in groundnut, finger millet, sorghum, pearl millet, soybean, lentil (*Lens esculenta*), and rice, respectively.^[11] These increases of yield of different rainfed crops in India are comparable to other parts of the globe under rainfed/irrigated conditions. In most of the cases, these are higher than other findings. For example, an increase in SOC stock by 1 Mg ha⁻¹ increased grain yield by 27 kg ha⁻¹ in wheat in North Dakota, U.S.A.,^[17] 40 kg ha⁻¹ in wheat in the semiarid pampas of Argentina,^[18] 17 kg ha⁻¹ in maize in Thailand,^[19] and 10 kg ha⁻¹ in maize and 1 kg ha⁻¹ in cowpea (*Vigna unguiculata*) in western Nigeria.^[20] Even these increases are much higher compared to irrigated crops in India. For example, Kanchikerimath and Singh^[21] reported

Table 1 SOC sequestration rates with different integrated nutrient management practices in important rainfed production systems.

Production system#/location* duration ^a /no of years+ of permanent manurial experiments	Treatment details	Best management practices	SYI	Mean annual C input (Mg C ha ⁻¹ yr ⁻¹)	Mean C depletion rate in control (Mg C ha ⁻¹ yr ⁻¹)	SOC seques- tration (Mg C ha ⁻¹)	SOC seques- tration rate (Mg C ha ⁻¹ yr ⁻¹)	Critical C input requirement (Mg C ha ⁻¹ yr ⁻¹)
Groundnut/Anantapur/ 1985–2004/20 years	T ₁ = control (no fertilizer), T ₂ = 100% RDF (20:40:40 N, P ₂ O ₅ , K ₂ O), T ₃ = 50% RDF + 4 Mg GNS ha ⁻¹ , T ₄ = 50% RDF + 4 Mg FYM ha ⁻¹ , T ₅ = 100% organic (5 Mg FYM ha ⁻¹)	T ₃ = 50% RDF + 4 Mg GNS ha ⁻¹	0.48	3.45	0.18	11.43	0.57	1.12
Finger millet/Bangalore/ 1978–2004/27 years	T ₁ = control, T ₂ = 10 Mg FYM ha ⁻¹ , T ₃ = 10 Mg FYM ha ⁻¹ + 50% NPK, T ₄ = 10 Mg FYM ha ⁻¹ + 100% NPK, T ₅ = recommended NPK (50:50:25 kg NPK ha ⁻¹ —fingermillet)	T ₄ = 10 Mg FYM ha ⁻¹ + 100% NPK	0.59	3.10	0.92	15.4	0.82	1.13
Finger millet–Groundnut/ Bangalore//1991–2004/13 years	T ₁ = control, T ₂ = 10 Mg FYM ha ⁻¹ , T ₃ = 10 Mg FYM ha ⁻¹ + 50% NPK, T ₄ = 10 Mg FYM ha ⁻¹ + 100% NPK, T ₅ = recommended NPK (25:50:25 kg NPK ha ⁻¹ —groundnut; 50:50:25 kg NPK ha ⁻¹ —fingermillet)	T ₄ = 10 Mg FYM ha ⁻¹ + 100% NPK	Groundnut: 0.21; fingermillet: 0.76	3.03	0.25	9.3	1.64	1.62
Rabi sorghum/Solapur/ 1985–2006/22 years	T ₁ = control, T ₂ = 25 kg N ha ⁻¹ (urea), T ₃ = 50 kg N ha ⁻¹ (urea), T ₄ = 25 kg N ha ⁻¹ through sorghum residue (CR), T ₅ = 25 kg N ha ⁻¹ through FYM, T ₆ = 25 kg N ha ⁻¹ (CR) + 25 kg N ha ⁻¹ (urea), T ₇ = 25 kg N ha ⁻¹ (FYM) + 25 kg N ha ⁻¹ (urea), T ₈ = 25 kg N ha ⁻¹ (CR) + 25 kg N ha ⁻¹ (Leucaena clippings), T ₉ = 25 kg N ha ⁻¹ (Leucaena), T ₁₀ = 25 kg N ha ⁻¹ (Leucaena) + 25 kg N ha ⁻¹ (urea)	T ₈ = 25 kg N ha ⁻¹ (CR) + 25 kg N ha ⁻¹ (Leucaena clippings)	0.48	3.40	0.23	14.4	0.89	1.10

(Continued)

Table 1 SOC sequestration rates with different integrated nutrient management practices in important rainfed production systems. (Continued)

Production system#/location years+ of permanent manurial experiments	Treatment details	Best management practices	SYI	Mean annual C input (Mg C ha ⁻¹ yr ⁻¹)	Mean C depletion rate (Mg C ha ⁻¹ yr ⁻¹)	SOC sequestration (Mg C ha ⁻¹ yr ⁻¹)	SOC sequestration rate (Mg C ha ⁻¹ yr ⁻¹)	Critical C input requirement (Mg C ha ⁻¹ yr ⁻¹)
Pearlmillet/S.K. Nagar/ 1988–2006/18 years	T ₁ = control, T ₂ = 100% RDNF,	T ₅ = 50%	Pmillet: 0.30	1.86	0.67	-4.4	-0.24	3.30
	T ₃ = 50% RDNF; T ₄ = 50%	recommended N	Cbean: 0.69					
	recommended N (FYM), T ₅ = 50%	(fertilizer) + 50%						
	recommended N (fertilizer) + 50%	recommended N	Castor: 0.46					
Soybean/Indore/1992– 2007/15 years	recommended N (FYM), T ₆ = farmers method (5 Mg of FYM ha ⁻¹ once in 3 years)	(FYM)						
	T ₁ = control, T ₂ = 20 kg N + 13 kg P, T ₃ = 30 kg N + 20 kg, T ₄ = 40 kg N + 26 kg, T ₅ = 60 kg N + 35 kg P,	T ₆ = 6 Mg FYM ha ⁻¹ + N ₂₀ P ₁₃	Soybean: 0.48	6.99	0.47	11.9	1.26	3.47
	T ₆ = 6 Mg FYM ha ⁻¹ + N ₂₀ P ₁₃ , T ₇ = 5 Mg soybean residue ha ⁻¹ + N ₂₀ P ₁₃ , T ₈ = 6 Mg FYM ha ⁻¹ , T ₉ = 5 Mg soybean residues ha ⁻¹ .		Safflower: 0.45					
	T ₁ = control, T ₂ = 100% RDF (inorganic), T ₃ = 50% RDF (FYM), T ₅ = 50% organic (FYM), T ₆ = 50% RDF + 50% (foliar), T ₇ = 50% organic (FYM) + 50% RDF, T ₈ = farmers practice	T ₄ = 100% organic (FYM)	Rice: 0.26 Lentil: 0.26	5.55	0.15	6.6	0.32	2.47

Note: SYI, sustainable yield index; RDNF: recommended dose of N through mineral fertilizer; GNS: groundnut shells; RDF: recommended dose of fertilizer; ‘-ive’ sign indicates depletion.

6 kg ha⁻¹ in wheat and 3 kg ha⁻¹ in maize in alluvial soils of northern India. Lal^[17] estimated that adoption of RMPs could increase SOC stock by 1 Mg ha⁻¹ yr⁻¹, which can increase food grain production by 32 million Mg yr⁻¹ in developing countries.

Minimal Carbon Input Requirements for Arresting C Depletion

“Critical carbon input” is the quantity of carbon input required to maintain SOC level of the soil to its antecedent level. This information is important to formulate long-term carbon management strategies. Continuous cropping without using any organic amendment and/or mineral fertilizers causes depletion of the SOC stock. This depletion ranges from 0.15 Mg C ha⁻¹ yr⁻¹ in rice-based system to 0.92 Mg C ha⁻¹ yr⁻¹ in groundnut–finger millet rotation. The highest rate of depletion (0.92 Mg C ha⁻¹ yr⁻¹) was observed in groundnut–finger millet rotation in semiarid Alfisols followed by 0.67 Mg C ha⁻¹ yr⁻¹ in pearl millet-based system in Entisols and 0.47 Mg C ha⁻¹ yr⁻¹ in soybean-based system in Vertisols. The SOC depletion was lowest in rice-based system. There was a negative relationship between mean annual C inputs and mean SOC depletion rates across the locations, soil types, and production systems. However, the MAP plays a significant role in SOC depletion and an inverse relationship exists between MAP and SOC depletion rate. Globally, rates of C sequestration by different types of management ranged from 0.11 to 3.04 Mg C ha⁻¹ yr⁻¹, with a mean of 0.54 Mg C ha⁻¹ yr⁻¹, and are highly influenced by soil type and climate.^[22,23] Total C input including leaf, stubble, root, nodules, and rhizodeposition, as well as external inputs through FYM/groundnut shell/sorghum residue/*Leucaena* clippings, and cumulative C inputs into soil under different treatments during the long-term experiments are given in Table 1. The highest C inputs (6.99 Mg ha⁻¹ yr⁻¹) were added in soybean and lowest 1.86 Mg ha⁻¹ yr⁻¹ in pearl millet system. The positive linear relationship between the changes in SOC stock and the total cumulative C inputs to the soil (external organics plus crop residue) over the years was highly significant and indicates that even after 13–27 years of C input ranging from 0.2–1.9 Mg C ha⁻¹ yr⁻¹ in pearl millet-based system to 1.9–7.0 Mg C ha⁻¹ yr⁻¹ in soybean–safflower system, the C sink capacity was not saturated. Therefore, these severely depleted soils have a high C sink capacity. However, sink capacity and/or sequestration rate cannot continue indefinitely.^[24] Each soil with a different C loading may reach a new steady state of SOC stock over time. The global potential of SOC sequestration and restoration of degraded/desertified soils is estimated at 0.6–1.2 Pg C yr⁻¹ for about 50 years with a cumulative sink capacity of 30–60 Pg (petagram = 10¹⁵ g = billion metric ton),^[23] comprising 0.4–0.8 Pg C yr⁻¹ through adoption of RMPs on cropland (1350 Mha), 0.01–0.03 Pg C yr⁻¹ on irrigated soils (275 Mha), and 0.01–0.3 Pg C yr⁻¹ through improvements

of rangelands and grasslands (3700 Mha). Maintaining a constant level of SOC stock (zero change) requires C input of 1.10 Mg C ha⁻¹ yr⁻¹ in Vertisols under winter sorghum system to 3.47 Mg C ha⁻¹ yr⁻¹ under a soybean-based system. The rate of C input required for groundnut, finger millet, and winter sorghum system is much lower than those reported by Kong et al.^[25] (3.1 Mg ha⁻¹ yr⁻¹) in Davis, California, U.S.A. and 4.59 Mg ha⁻¹ yr⁻¹ by Majumder et al.^[26] for rice–wheat–jute system, 3.56 Mg ha⁻¹ yr⁻¹ by Majumder et al.^[27] for irrigated rice–wheat systems of the Indo-Gangetic plains, and 2.92 Mg ha⁻¹ yr⁻¹ by Mandal et al.^[28] for rice-based system in some soils of subtropical India. The lower input of C needed to maintain a constant level in long-term experiments done in India may be due to lower initial SOC levels (1.4–3.9 g kg⁻¹ soil).^[16] In the abovementioned studies, the initial SOC concentrations were approximately 3 to 6 times higher (>6–15 g kg⁻¹ soil) than those in this entry. In case of soybean, pearl millet, and upland rice, this rate is comparatively higher.^[9–14] In case of soybean–safflower sequence cultivated in Vertisols, initial SOC concentration of the soil was relatively high (6.2 g kg⁻¹). New and well-designed experiments are needed on different rates of biomass C input so that the threshold level can be established for different soils. Similar experiments are needed for deriving the rates of increase in crop yields per unit increase in SOC stock in the root zone.

CARBON MANAGEMENT STRATEGIES

The adoption of appropriate/diversified crop rotations by including legumes in cereal-based systems, balanced use of chemical fertilizers, and integration with organic manures and conservation agricultural practices can maintain the required level of SOC in the soil.^[13] Quantity of soil organic matter is a critical component of soil productivity. Residue management, particularly the retention of full/partial residue along with improved fertilization, can play a significant role in increasing the carbon input into the soil.^[29] However, availability of crop residue is a major problem in India due to competing alternative uses, but in some crops such as groundnut (shells), pigeonpea, cotton, castor, and soybean, residues are available for soil application, as they do not have major alternate uses. However, the application of some of these residues is difficult, as they have very hard stems (pigeonpea, cotton, etc.). Some residues like pigeonpea leaves are difficult to retain in the field, as they easily blown away by wind.

The strategies for a positive C balance are termed as RMPs in comparison with traditional practices in agriculture. Some of these RMPs are alteration of cropping systems, improved varieties with high biomass production, recycling organic waste, judicious use of chemical fertilizers, use of bioamendments, and improved soil and water management. These practices contribute not only toward soil conservation but also enhance the amount of SOC and

reduce CO₂ emissions^[30] as well as maintain a steady state of SOC for longer term.^[31] Conservation agriculture (CA) based on three major principles, namely minimum soil disturbance, residue retention, and crop rotation, is gaining importance day by day as a sustainable crop production technology in the context of increased climatic vulnerability. Experts have recognized its role in C storage and sustainable ecosystem services, but at the same time they have different opinions about C sequestration under CA system.^[15] Although, the adoption of CA worldwide shows a positive C balance in soils according to many case studies,^[22,31] there are several studies that show no change or decrease in SOC stock after practicing CA for several years. It has been estimated that conversion of all cropland to CA globally could sequester 25 Gt C over the next 50 years. This might mitigate C emission to the extent of 1833 Mt CO₂eq yr⁻¹.^[32] The applications of biochar or charcoal also have higher GHG mitigation potential. Many studies suggest that biochar is an effective soil amendment for improving soil conditions and increasing C sequestration.^[33]

Several management options are being promoted by Central Research Institute for Dryland Agriculture in farmer's participatory action mode in Telangana and Andhra Pradesh state. Some RMPs being promoted include crop residue retentions, FYM, biofertilizers, inclusion of legumes in the cropping sequence or as intercrops, green manure crops, green leaf manuring, tank silt addition, and vermicomposting along with chemical fertilizers.^[34] Among them, green leaf manure (GLM) with *Gliricidia sepium* is the most promising technology. *Gliricidia* can thrive in dry moist, acidic soils, or even on degraded soils under rainfed conditions. Growing *Gliricidia* plants on farm bunds serves dual purpose of producing GLM rich in N and in conserving soil.^[34] Farm bunds could be productively used for growing N-fixing shrubs and trees to generate N-rich loppings. For example, growing *G. sepium* at a close spacing of 75 cm on farm bunds could provide 28–30 kg N ha⁻¹ yr⁻¹ in addition to being a valuable source of organic matter.^[35]

Using GLM of *Gliricidia* can enhance soil productivity and increase crop yields of several rainfed crops. Impact of mulch-cum-manuring with *Gliricidia* on the productivity of castor showed a threefold increase compared with that of the control and one-and-a-half-fold increase in comparison with application of FYM and fertilizer. Yield response of different rainfed crops to *Gliricidia* GLM has been positive in many regions including that of finger millet in red soils of Karnataka, groundnut in red soils of Andhra Pradesh, pearl millet in light-textured soils of Gujarat, and sorghum in medium to deep black soils of Maharashtra. At Bhubaneswar, in acid red and lateritic soils, maize yield improved from 1.7 to 2.1 Mg ha⁻¹ by *Gliricidia* GLM equivalent to 20 kg N ha⁻¹.^[36] Based on a long-term experiment, Sharma et al.^[37] reported that the conjunctive use of urea and organics such as loppings of *Leucaena* and *Gliricidia* (1:1 ratio

on N equivalent) increased sorghum grain yield to 1.69–1.72 Mg ha⁻¹, respectively, and thus revealed that a minimum of 50% N requirement of sorghum can be easily met from farm-based organic sources. This research information can be used to supplement fertilizer N up to 50% by using green loppings of *Gliricidia maculata* and *Leucaena leucocephala* and will be useful while planning for the cultivation of organic produce. Furthermore, the SOC concentration was significantly increased by the application of crop residues such as sorghum stover and *Gliricidia* at the rate of 2 Mg ha⁻¹ under minimum and conventional tillage systems for rainfed sorghum–castor rotation on Alfisols.^[37]

CONCLUSION

SOC is a strong determinant of soil quality, which ultimately plays a pivotal role in maintaining productivity on a sustainable basis. It is more important in arid and semiarid conditions as the soils under these conditions are deficient in many nutrients. Carbon storage in the soil profile not only improves fertility but also abates global warming. SOC stocks in soil profiles across India showed wide variations and followed the order Vertisols > Inceptisols > Alfisols > Aridisols. Data from experiments involving long-term cropping, fertilization, and manuring show that mineral fertilizer application alone can neither sustain the productivity nor maintain the SOC stock. The depletion of organic C occurs without organics irrespective of production system and soil type. Application of organics along with mineral fertilizer can sustain productivity and sequester significant amount of soil C. The rate of depletion of SOC is in the range from 0.15 Mg C ha⁻¹ yr⁻¹ in rice-based system to 0.92 Mg C ha⁻¹ yr⁻¹ in groundnut–finger millet system. To curtail this depletion, C input of 1.10–3.47 Mg C ha⁻¹ yr⁻¹ is required as a maintenance dose, depending on soil type, ecoregion, and land use. There are many changes in agricultural management practices, which have positive impacts on SOC. Large scope exists in the adoption of C-positive management practices at farm level. In particular, reduction in tillage and retention of organic residues play key role in improving soil structure and hydrological properties as well as maintaining or increasing yields. Practices such as legume cover crops, vermicompost, tank silt application, crop residue recycling through mulching and compost, green leaf manuring, integrated nutrient management and site-specific nutrient management along with soil water conservation measures, and integrated farming systems are some of the high priority technologies for maintaining soil health and enhancing C sequestration in rainfed ecosystems in India.

RESEARCH AND POLICY NEEDS

There is a need to launch long-term field experiments to quantify the influence of best management practices

on the soil carbon sequestration in various ecosystems and crop production systems.

There is a need for refinement of the existing data on critical C input requirement as well as to derive this type of data set for other crop production systems in diverse soil types.

Quantification is required for C stabilization (%) from different types of residues and organic manures.

Monitoring and verification of the rate of SOC sequestration in a transparent, cost-effective, and credible manner are key requirement for developing a user-friendly carbon trading system.

Quantification of the country's soil carbon sequestration potential in terms of various land use and management scenarios using soil carbon models.

Information on the recommended land use and management practices for soil C sequestration and corresponding rates under "on-farm" conditions.

REFERENCES

1. International Crops Research Institute for the Semi-Arid Tropics (ICRISAT). *ICRISAT Report*; ICRISAT: Patancheru, 2013.
2. Biradar, C.M.; Thenkabail, P.S.; Noojipady, P.; Li, Y.; Venkateswarlu, D.; Turrall, H.; Velpuri, M.; Gumma, M.K.; Gangalakunta, O.R.P.; Cai, X.L.; Xiao, X.; Schull, M.A.; Alankara, R.D.; Gunasinghe, S.; Mohideen, S. A global map of rainfed cropland areas (GMRCA) at the end of last millennium using remote sensing. *Int. J. Appl. Earth Obs.* **2009**, *11* (2), 114–129.
3. Graham, M.H.; Haynes, R.F.; Meyer, J.H. Soil organic matter content and quality: Effects of fertilizer applications, burning and trash retention on a long-term sugarcane experiment in South Africa. *Soil Biol. Biochem.* **2002**, *34*, 93–102.
4. Velayutham, M.; Pal, D.K.; Bhattacharyya, T. Organic carbon stock in soils of India. In *Advances in Soil Science: Global Climatic Change and Tropical Ecosystems*; Lal, R., Kibbe, J.M., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2000; 71–95.
5. Srinivasarao, Ch.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Kundu, S.; Vittal, K.P.R.; Balaguruvaiah, G.; Vijaya Shankar Babu, M.; Ravindra Chary, G.; Prasadbabu, M.B.B.; Yellamanda Reddy, T. Soil carbon sequestration and agronomic productivity of an Alfisol for a groundnut based system in a semi-arid environment in South India. *Eur. J. Agron.* **2012**, *43*, 40–48.
6. Srinivasarao, Ch.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Kundu, S.; Vittal, K.P.R.; Gajanan, G.N.; Ramachandrappa, B.K. Yield sustainability and carbon sequestration potential of groundnut–finger millet rotation in Alfisols under semi-arid tropical India. *Int. J. Agric. Sus.* **2012**, *10* (3), 1–15.
7. Srinivasarao, Ch.; Venkateswarlu, B.; Singh, A.K.; Vittal, K.P.R.; Kundu, S.; Gajanan, G.N.; Ramachandrappa, B.; Chary, G.R. Critical carbon inputs to maintain soil organic carbon stocks under long-term finger-millet (*Eleusine coracana* [L.] Gaertn.) cropping on Alfisols in semiarid tropical India. *J. Plant Nutr. Soil Sci.* **2012**, *175* (5), 681–688.
8. Srinivasarao, Ch.; Deshpande, A.N.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Kundu, S.; Vittal, K.P.R.; Mishra, P.K.; Prasad, J.V.N.S.; Mandal, U.K.; Sharma, K.L. Grain yield, carbon sequestration potential of post monsoon sorghum cultivation in vertisols in the semi-arid tropics of central India. *Geoderma* **2012**, *175–176*, 90–97.
9. Srinivasarao, Ch.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Kundu, S.; Vittal, K.P.R.; Sharma, S.K.; Sharma, R.A.; Jain, M.P.; Ravindra Chary, G. Sustaining agronomic productivity and quality of a Vertisolic Soil (Vertisol) under soybean-safflower cropping system in semi-arid central India. *Can. J. Soil Sci.* **2012**, *92* (5), 771–785.
10. Srinivasarao, Ch.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Vittal, K.P.R.; Kundu, S.; Singh, S.R.; Singh, S.P. Long-term effects of soil fertility management on carbon sequestration in a rice–lentil cropping system of the Indo-Gangetic plains. *Soil Sci. Soc. Am. J.* **2012**, *76* (1), 168–178.
11. Srinivasarao, Ch.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Kundu, S. Sustainable management of soils of dry-land ecosystems of India for enhancing agronomic productivity, sequestering carbon. In *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press: Burlington, 2013; 253–329.
12. Srinivasarao, Ch.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Kundu, S.; Jakkula, V.S. Carbon sink capacity and agronomic productivity of soils of the semiarid regions of India. In *Principles of Sustainable Soil Management in Agroecosystems*; Lal, R., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2013; 423–476.
13. Srinivasarao, Ch.; Lal, R.; Kundu, S.; Prasad Babu, M.B.B.; Venkateswarlu, B.; Singh, A.K. Soil carbon sequestration in rainfed production systems in the semiarid tropics of India. *Sci. Total Environ.* **2014**, *487*, 587–603.
14. Srinivasarao, Ch.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Kundu, S.; Vittal, K.P.R.; Patel, J.J.; Patel, M.M. Long-term manuring and fertilizer effects on depletion of soil organic carbon stocks under pearl millet-cluster bean-castor rotation in western India. *Land Degrad. Dev.* **2014**, *25*, 173–183.
15. Srinivasarao, Ch.; Lal, R.; Kundu, S.; Thakur, P.B. Conservation agriculture and soil carbon sequestration. In *Conservation Agriculture*. Part. V; Farooq, M., Siddique, K.H.M., Eds.; Springer International: Switzerland, 2015; 479–524.
16. Srinivasarao, Ch.; Vittal, K.P.R.; Venkateswarlu, B.; Wani, S.P.; Sahrawat, K.L.; Marimuthu, S.; Kundu, S. Carbon stocks in different soils under diverse rainfed production systems in tropical India. *Commun. Soil Sci. Plan.* **2009**, *40* (15), 2338–2356.
17. Lal, R. Enhancing crop yields in the developing countries through restoration of the soil organic carbon pool in agricultural soil. *Land Degrad. Dev.* **2006**, *17*, 197–209.
18. Diaz-zorita, M.; Gustavo, A.D.; Grove, J.H. A review of no-till system, soil management for sustainable crop production in the sub-humid, semi-arid Pampas of Argentina. *Soil Till. Res.* **2002**, *65*, 1–18.
19. Petchawee, S.; Chaitep, W. *Organic Matter Management in Upland Systems in Thailand*; Australian Centre for International Agricultural Research: Canberra, 1995; 21–26.

20. Lal, R. Soil erosion problems on Alfisols in Western Nigeria. VI. Effects of erosion on experimental plots. *Geoderma* **1981**, 25, 215–230.
21. Kanchikerimath, M.; Singh, D. Soil organic matter and biological properties after 26 years of maize–wheat–cowpea cropping as affected by manure, fertilization in a Cambisol in semi-arid region of India. *Agr. Ecosyst. Environ.* **2001**, 86, 155–162.
22. West, T.O.; Post, W.M. Soil organic carbon sequestration by tillage and crop rotation: A global data analysis. *Soil Sci. Soc. Am. J.* **2002**, 66, 1930–1946.
23. Lal, R. Soil carbon sequestration to mitigate climate change. *Geoderma* **2004**, 123, 1–22.
24. Six, J.; Feller, C.; Denef, K.; Ogle, S.M.; de Moraes Sa, J.C.; Albrecht, A. Soil organic matter, biota, and aggregation in temperate and tropical soils—Effects of no-tillage. *Agronomy* **2002**, 22, 755–775.
25. Kong, A.Y.Y.; Six, J.; Bryant, D.C.; Denison, R.F.; van Kessel, C. The relationship between carbon input, aggregation, and soil organic carbon stabilization in sustainable cropping systems. *Soil Sci. Soc. Am. J.* **2005**, 69, 1078–1085.
26. Majumder, B.; Mandal, B.; Bandyopadhyay, P.K.; Chaudhury, J. Soil organic carbon pools and productivity relationships for a 34 year old rice–wheat–jute agroecosystem under different fertilizer treatments. *Plant Soil* **2007**, 297, 53–67.
27. Majumder, B.; Mandal, B.; Bandyopadhyay, P.K.; Gangopadhyay, A.; Mani, P.K.; Kundu, A.L.; Majumder, D. Organic amendments influence soil organic carbon pools and crop productivity in a 19 years old rice–wheat agroecosystems. *Soil Sci. Soc. Am. J.* **2008**, 72, 775–785.
28. Mandal, B.; Majumder, B.; Bandyopadhyay, P.K.; Hazra, G.C.; Gangopadhyay, A.; Samantaray, R.N.; Mishra, A.K.; Chaudhury, J.; Saha, M.N.; Kundu, S. The potential of cropping systems and soil amendments for carbon sequestration in soils under long-term experiments in subtropical India. *Glob. Change Biol.* **2007**, 13, 357–369.
29. Campbell, C.A.; Bowren, K.E.; Schnitzer, M.; Zentner, R.P.; Townley-Smith, L. Effect of crop rotations and fertilization on soil organic matter and some biochemical properties of a thick Black Chernozem. *Can. J. Soil Sci.* **1991**, 71, 377–387.
30. Follett, R.F.; Varvel, G.E.; Kimble, J.; Vogel, K.P. No-till corn after brome grass: Effect on soil carbon and soil aggregates. *Agron. J.* **2009**, 101, 261–268.
31. Govaerts, B.; Verhulst, N.; Navarrete, C.; Sayre, A.; Dixon, K.D.; Dendooven, J.L. Conservation agriculture and soil carbon sequestration: Between myth and farmer reality. *Crit. Rev. Plant Sci.* **2009**, 28, 97–122.
32. Baker, J.M.; Ochsner, T.E.; Venterea, R.T.; Griffis, T.J. Tillage and soil carbon sequestration—what do we really know? *Agr. Ecosyst. Environ.* **2007**, 118, 1–5.
33. Sohi, S.P.; Krull, E.; Lopez-Capel, E.; Bol, R. A review of biochar and its use and function in soil. In *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press Inc.: San Diego, 2010; 47–82.
34. Srinivasarao, Ch.; Venkateswarlu, B.; Dixit, S.; Kundu, S.; Gayatri Devi, K. *Livelihood Impacts of Soil Health Improvement in Backward and Tribal Districts of Andhra Pradesh*; Central Research Institute for Dryland Agriculture: Hyderabad, 2011; 119 pp.
35. Wani, S.P.; Pathak, P.; Jangawad, L.S.; Eswaran, H.; Singh, P. Improved management of Vertisols in semiarid tropics for increased productivity and soil carbon sequestration. *Soil Use Manage.* **2003**, 19, 217–222.
36. All India Coordinated Research Project On Dryland Agriculture (AICRPDA). *Annual Reports of AICRP on Dryland Agriculture (various issues)*; CRIDA: Hyderabad, 1991–2009.
37. Sharma, K.L.; Srinivas, K.; Das, S.K.; Vittal, K.P.R.; Kusuma Grace, J. Conjunctive use of inorganic and organic sources of nitrogen for higher yield of sorghum in dryland Alfisol. *Indian J. Dryland Agric. Res. Dev.* **2002**, 17 (2), 79–88.

Carbon Sequestration: Urban Ecosystems

Adam Louis Selhorst

Environmental Studies, Ashford University, San Diego, California, U.S.A.

Abstract

As global climate change (GCC) continues to be an international concern, researchers are looking at the potential of urban soils to sequester atmospheric carbon (C). With the rapid expansion of global urbanization, the subsequent development of urban green space may prove to be a vital C sink. Urban turfgrass systems have shown the ability to sequester C at rates of 0.69–3.58 Mg C ha⁻¹ yr⁻¹, providing an overall potential sink capacity of 20.8–96.3 Mg C ha⁻¹. However, sequestration potential is dependent on climatic properties with optimal soil organic carbon (SOC) pools located in cooler regions with precipitation levels between 60 and 70 cm yr⁻¹. Furthermore, soil physical properties also influence SOC sequestration as sink capacity is positively correlated with increases in bulk density (ρ_b) up to 1.5 Mg m⁻³ and increased soil nitrogen concentrations. However, the hidden carbon costs (HCCs) due to urban soil management practices result in emissions of 189 kg carbon equivalent (Ce) ha⁻¹ yr⁻¹ for mowing practices and 63.6 kg Ce ha⁻¹ yr⁻¹ for fertilization application in home lawns. In more intensively managed soils such as golf courses, HCCs accounted for as much as 302 kg Ce ha⁻¹ yr⁻¹. Thus, HCCs converted home lawn soils from sinks to sources in 66–199 years, depending on management intensity. In order to maximize the potential of urban soils to sequester C, low C-intensive management practices should be utilized to maintain ideal soil physical properties in regions with optimal climatic conditions.

INTRODUCTION

As rapid expansion of the human population continues, both urbanization and industrialization have increased globally. These increases have been coupled with significant alterations in worldwide land use and a growing combustion of fossil fuels. Such changes have resulted in the anthropogenic enrichment in atmospheric carbon dioxide (CO₂), leading to a host of environmental concerns, most notably global climate change (GCC). In an effort to slow the increase in atmospheric carbon (C) enrichment, researchers are beginning to note the potential of urban soils to sequester C and mitigate the effects of GCC.

With the migration of global populations moving toward city centers, urban soils are becoming an increasingly important part of the landscape. The enhancement of green space is seen as highly desirable, often coming in the form of parks, athletic fields, golf courses, and home lawns. While providing recreational and aesthetic benefits, much research has shown the potential of urban soils to sequester large amounts of C. However, the extent of sequestration may be affected by climatic factors, soil properties, and turf management regimes. This entry examines the past and ongoing research on the ability of urban soils to sequester C. Additionally, it reviews the impacts of temperature, precipitation, and various soil properties on sequestration potential and views all results in light of the various hidden carbon costs (HCCs) that are often generated due to turfgrass maintenance practices.

C SEQUESTRATION POTENTIAL IN URBAN SOILS

As the global population continues to trend upward, urban areas are occupying an increasingly large portion of the world's landscape. Urban land is increasing twice as fast as population, and it is estimated that by 2030 land dedicated for urban use will be triple 2000 levels, increasing by as much as 1.2×10^6 km².^[1] Such prolific land use change has led to global concerns regarding the negative environmental impacts associated with rapid urbanization including its contribution to GCC.

Urbanization and its subsequent land use change have shown the ability to alter the C dynamics within a system. Globally, soils hold an estimated C of 2500 Pg; however, significant perturbations can lead to the depletion of soil organic matter pools, resulting in the direct release of soil organic carbon (SOC) into the atmosphere in the form of CO₂.^[2] Although there is concern over the loss of SOC to the atmosphere, formerly disturbed soils provide an immense opportunity to resequester a large portion of previously emitted SOC and thus might prove to be an effective tool in GCC mitigation. Each year, 120 Pg of C is released to the atmosphere by soil and plant respiration. If 10% of this C could be retained within soils, it could offset the growing C emissions released through fossil fuel burning.^[2] Thus, soils have the potential to sequester C and reduce the rate of atmospheric CO₂ enrichment.

Table 1 Soil organic carbon sequestration rate following land use conversion in various sites throughout the world.

Source	Site location	Soil organic carbon sequestration rate (Mg C ha ⁻¹ yr ⁻¹)
Tyson et al. ^[4]	United Kingdom	0.82 ^a
Haynes, Swift, and Steohen ^[5]	New Zealand	1.11 ^a
Mensah, Schoenau, and Malhi ^[6]	California, U.S.A.	0.86 ^a
Gebhart et al. ^[7]	U.S.A.	0.47 ^a
Qian and Follett ^[8]	Colorado, U.S.A.	0.90 ^b
Bandaranayake et al. ^[9]	Colorado, U.S.A.	0.15 ^b
Huh et al. ^[10]	New Zealand	0.69 ^b
Qian, Follett, and Kimble ^[11]	Nebraska, U.S.A.	0.32–0.78 ^c
Selhorst and Lal ^[12]	Ohio, U.S.A.	2.47–3.58 ^d
Selhorst and Lal ^[13]	U.S.A.	0.90–5.4 ^e

Notes: ^acropland converted to grassland.

^bgrassland converted to golf course.

^cprairie converted to golf course.

^dcropland converted to golf course.

^egrassland converted to home lawn.

Urban soils may be of particular interest due to the growing area of green space being utilized by city planners. As they are areas of high productivity, green areas have been shown to function as C sinks.^[3] Research has illustrated how unproductive soils can provide long-term C sequestration potential upon conversion to grasslands with C turnover rates as high as 2200 years.^[4] Additionally, studies throughout the world have confirmed that reversion of C-depleted cropland to grasslands can result in soil C sequestration at rates between 0.86 and 1.11 Mg C ha⁻¹ yr⁻¹^[4–6] (Table 1).

While reversion of depleted soils to prior land use does show significant sequestration potential, urban soils contain higher concentrations of SOC than their suburban counterparts.^[14] In general, this is due to the intensive management and subsequent high productivity of turfgrasses maintained in urban settings. Golf courses provide an example of one of the most intensively managed urban soils. The proliferation of highly productive grasses and the regular fertilization and irrigation they receive allow newly designed courses to sequester C for between 20 and 90 years at rates between 0.15 and 3.58 Mg C ha⁻¹ yr⁻¹ depending on original land use.^[8–12] This means that a golf course constructed on previously depleted cropland has the potential to sequester as much as 3517 Mg of C.^[13]

While golf course area comprises a small component of urban soil land use, turf proliferation in the form of parks and commercial and residential lawns may provide a more representative sample of the ability of urban soils to sequester SOC. Research indicates that home lawns can sequester C at rates between 0.9 C ha⁻¹ yr⁻¹ and 5.4 Mg C ha⁻¹ yr⁻¹.^[13]

Additionally, total sequestration potential has been measured at as much as 144 Mg C ha⁻¹, depending on management intensity and soil location.^[15] Overall, home lawns throughout the United States hold the potential to harness as much as 496.3 Tg of C.^[13]

In addition to the direct sequestration of C due to turfgrass establishment, urban soils may provide secondary benefits allowing for the additional sequestration of C. First, soil erosion is responsible for a loss of SOC. Establishment of turf on areas highly susceptible to erosion may prevent such loss and lead to the retention of soil C in urban areas. Furthermore, the incorporation of trees and other foliage into urban green space plans may provide another venue for the sequestration of C as plants have illustrated the potential to act as C sinks.^[16] More specifically, tree biomass has been shown to accumulate C at a rate of 1.7 Mg C ha⁻¹ yr⁻¹ in urban parks accounting for as much as 45.9 Mg C ha⁻¹.^[17] It is also important to note that mulched areas have shown the ability to increase both SOC content and microbial biomass activity leading to SOC concentrations on par and in many cases higher than in neighboring turf areas.^[18] Thus, when possible, mulching of urban soils may prove to be a valid sequestration strategy. Thus, the incorporation of a mixture of plants, mulched areas, and grasses into the urban landscape may provide an optimal C sequestration strategy for urban soils.

Finally, it is important to note that all data presented here reference turf soils that are managed with minimal disruption to the soil structure. This is an important distinction as regularly resurfaced turfgrass systems such as athletic fields have shown no C sequestration benefits due to the regular removal of topsoil, aboveground and belowground biomass, and mixing of soil structure.^[19] As such, to maximize the C sequestration potential in urban soils, management practices must be cautious to avoid significant disturbances to the soil structure in order to evade the unintentional release of SOC to the atmosphere.

CLIMATIC EFFECTS ON C SEQUESTRATION IN URBAN SOILS

While urban soils have shown the ability to enhance the global C pool, the extent of their sequestration potential may be affected by climatic conditions. While past research has shown both positive and negative interactions between mean annual temperature (MAT) and soil C concentration rates, research in home lawn systems has shown a direct correlation between lower temperatures and higher SOC pools (Fig. 1).^[20] This is most likely due to decreased plant growth at higher temperatures coupled with increased SOC decomposition rates in response to enhanced microbial activity.^[21] Along with decreased net photosynthetic rates at temperatures greater than 30°C,^[22] elevated MAT is associated with decreased root development and overall turf quality in grass systems.^[23]

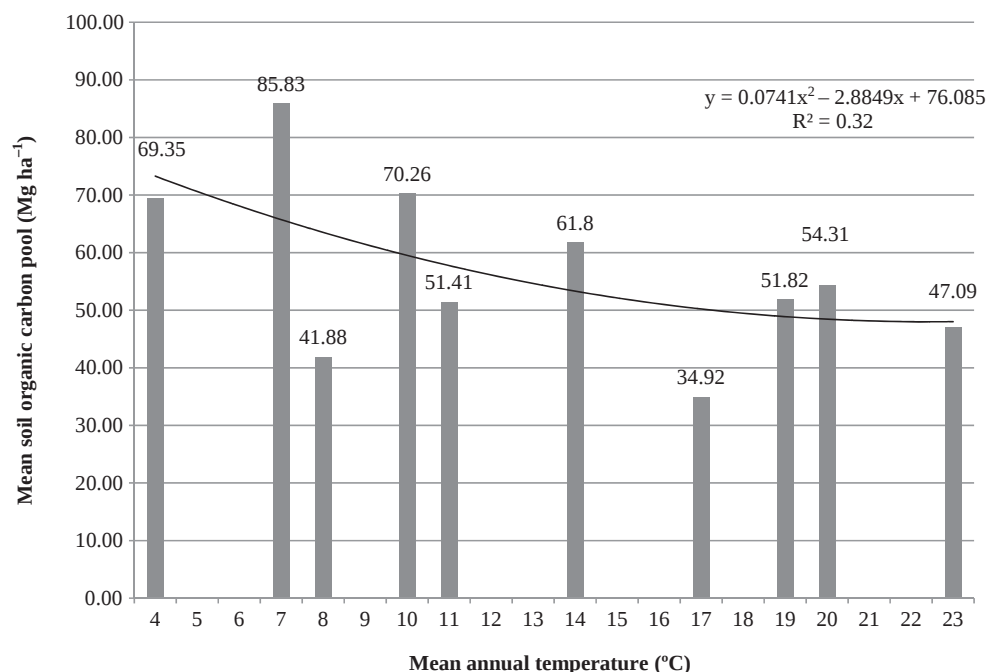


Fig. 1 Mean soil organic carbon pool for U.S. home lawns located in various mean annual temperatures following initial soil organic carbon sequestration to a depth of 15 cm (analysis of variance for interaction of soil organic carbon pool and mean annual temperature: $p < 0.001$, degrees of freedom (DF) = 341, $N = 342$).

Moisture content can also alter soil-based C as mean annual precipitation (MAP) has been shown to affect SOC concentrations. In dry climates, the lack of vegetation can limit soil C storage; thus, increased MAP in dry areas has shown the ability to enhance SOC pools.^[24] However, research illustrates that continued soil saturation can lead to inhibition of SOC storage, and lower overall C concentrations are found in saturated climates.^[25] Analysis of urban soils under home lawn management has shown highest levels of SOC concentration in climates where precipitation levels are between 60 cm yr⁻¹ and

70 cm yr⁻¹ (Fig. 2).^[20] These findings may provide justification for the addition of irrigation water in dry climates and the avoidance of supplemental watering in wetter climates. However, grass species type may also play a vital role in determining the optimal moisture level for turf quality and SOC sequestration. Home lawns seeded with drought-tolerant species such as St. Augustine and Bermuda grass may ultimately require lower levels of irrigation, while less tolerant species such as Zoysiagrass may require greater application of supplemental water.^[26]

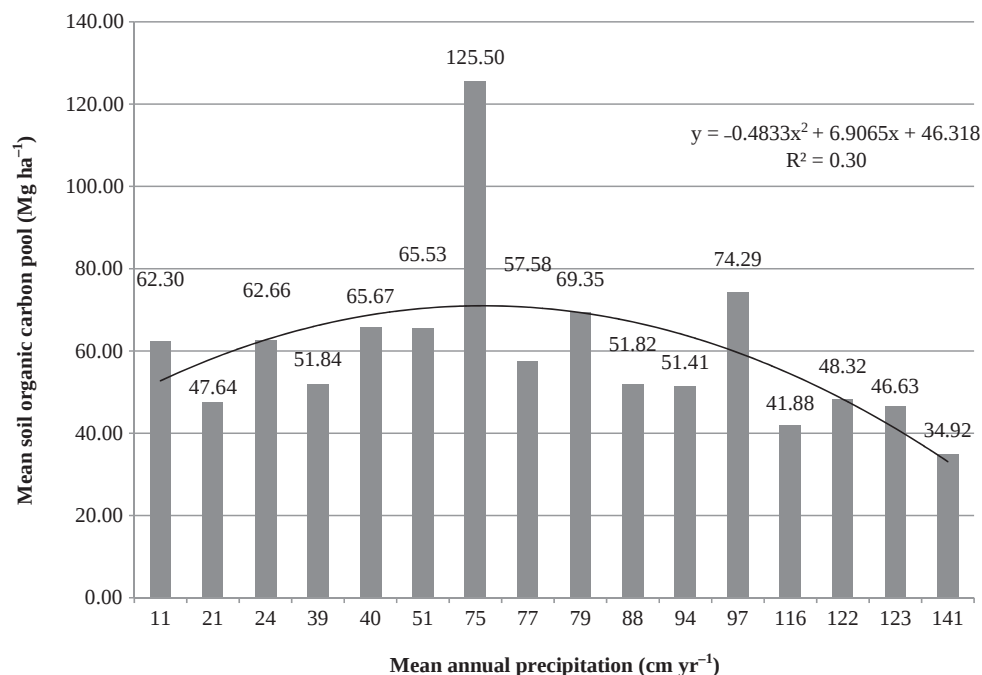


Fig. 2 Mean soil organic carbon pool for U.S. home lawns located in various mean annual precipitations following initial soil organic carbon sequestration to a depth of 15 cm (analysis of variance for interaction of soil organic carbon pool and mean annual temperature: $p < 0.001$, DF = 350, $N = 351$).

Based on the research presented, maximization of the C sequestration potential of urban soils under turfgrass management may be obtained by considering the climatic conditions during urban planning. In extremely arid and hot climates, the use of mulched soils with drought-tolerant foliage may provide a better alternative to turfgrasses that will require ample irrigation and minimal sequestration benefits. However, for soils located in cooler climates with moderate MAP levels, expansion of turfed areas may prove to be an effective method of enhancing the overall SOC pool and maximizing the potential benefits of urban soils.

INFLUENCE OF SOIL PROPERTIES ON C SEQUESTRATION IN URBAN SOILS

In conjunction with climatic factors, soil properties can also greatly influence the ability of urban soils to sequester C. Conversion of land from natural to urban uses can either increase or decrease C storage due to changes in the

soil properties and structure. One important property associated with C sequestration in urban ecosystems is bulk density (ρ_b). In general, urban soils show high levels of soil compaction and thus have elevated bulk densities.^[27] This compaction has been shown to decrease SOC content through limiting of fresh organic matter inputs into lower levels of the soil profile.^[27] However, this does provide an opportunity for enhancing the SOC sequestration potential of urban soils as maintenance practices that result in a decrease in compaction can enhance the soils' ability to sequester C. Furthermore, research has shown optimal levels of C sequestration in urban home lawns when ρ_b is maintained between 1.4 Mg m^{-3} and 1.5 Mg m^{-3} (Fig. 3).^[20] Thus, maintenance of turfgrass systems directed at optimal compaction could enhance the ability of urban soil to sequester C.

In addition to ρ_b , clay content and pH may also alter the sequestration potential in urban soils. Fine soils with elevated levels of silt + clay have been shown in certain conditions to result in increased SOC pools.^[28] However, with

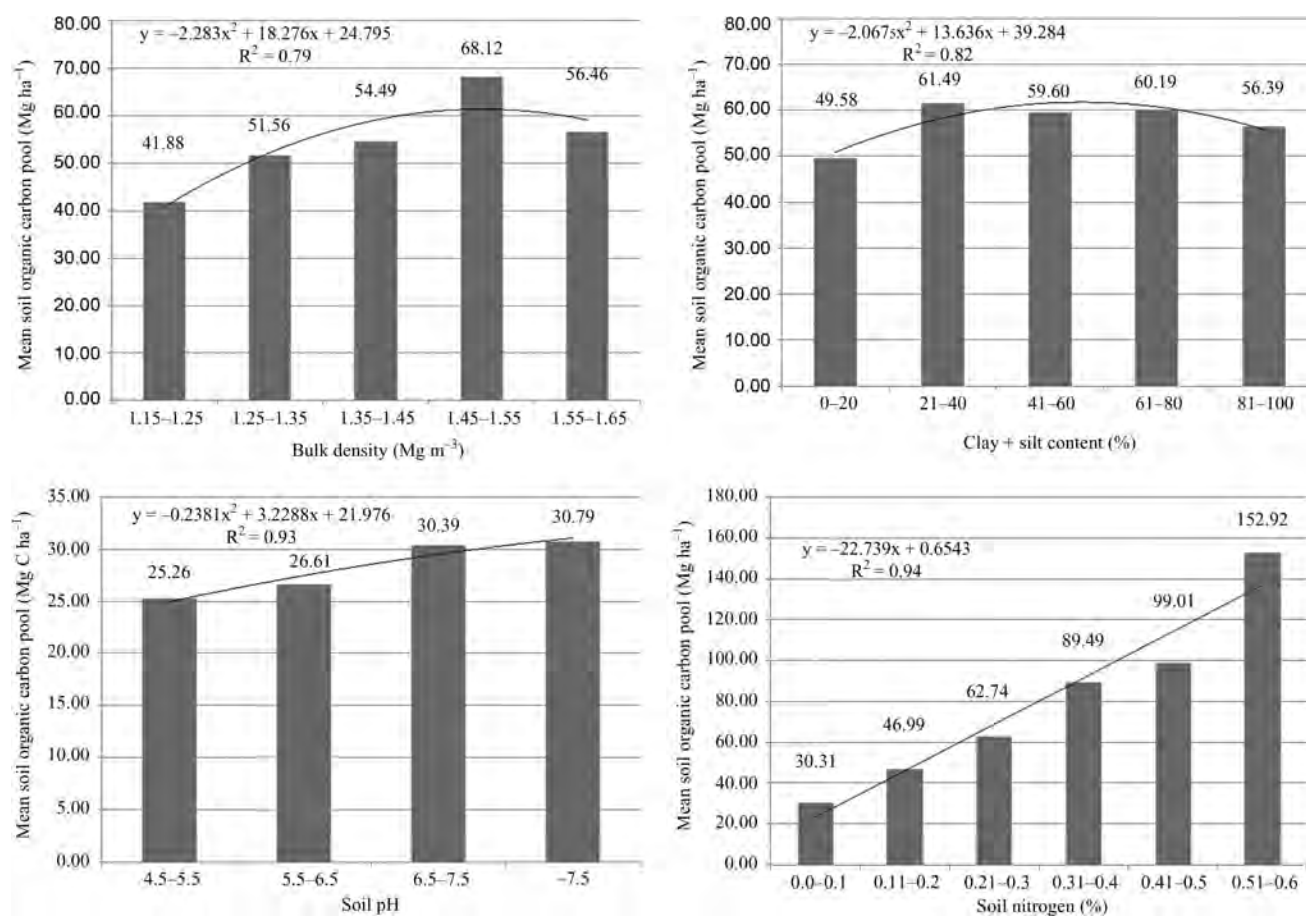


Fig. 3 The effect of soil bulk density, clay + silt content, pH, and nitrogen on the mean soil organic carbon pool for U.S. home lawns following initial soil organic carbon sequestration to a depth of 15 cm [analysis of variance (ANOVA) for interaction of soil organic carbon pool and soil bulk density: $p < 0.001$, $DF = 341$, $N = 342$; ANOVA for interaction of soil organic carbon pool and soil clay + silt content: $p = 0.237$, $DF = 341$, $N = 342$; ANOVA for interaction of soil organic carbon pool and soil pH: $p = 0.410$, $DF = 341$, $N = 342$; ANOVA for interaction of soil organic carbon pool and soil nitrogen: $p < 0.001$, $DF = 340$, $N = 341$].

proper moisture, coarse particles can account for as much as 40% of the SOC.^[29] Additionally, pH has been shown to alter microbial communities which are responsible for the decomposition of biomass, which can have substantial effects on SOC pools.^[28] However, research conducted nationwide on home lawns showed no correlation between either soil texture or pH and SOC sequestration ability (Fig. 3).^[20] Thus, texture and pH levels may have little overall effect on the ability of urban soils under turfgrass regimes to sequester atmospheric C.

Alternatively, research into soil nitrogen (N) concentrations did show a significant effect on SOC content with each 0.1% increase in N content leading to a 0.99% increase in SOC (Fig. 3).^[20] Increased N levels have been shown to decrease litter decomposition, leading to enhanced overall C pools.^[30] This is especially important to note in highly managed urban turfgrass systems as the addition of N fertilizer can greatly enhance SOC sequestration potential. In terrestrial ecosystems, anthropogenic N can increase SOC pools at rates as much as 0.1–0.3 Pg C yr⁻¹.^[31] Based on these findings, proper management strategies may include the regular addition of inorganic N to maintain optimal turfgrass decomposition rates to maximize soil C sequestration potential.

HIDDEN COSTS OF URBAN SOIL MANAGEMENT

The bulk of this entry has reviewed the high potential of turf-covered urban soils to sequester C; however, results must be viewed in light of the HCCs due to turf and soil maintenance. Although increased turf management practices can directly lead to enhanced soil C sequestration, the emissions produced through these management practices must be considered when determining the overall potential of urban turfgrass soils to sequester C. Additionally, management due to mowing and the application of fertilizer, pesticides, insecticides, and irrigation can be both time-consuming and financially costly for a landowner. For these reasons, the continued use of fossil fuels and chemicals to maintain healthy looking turfgrasses may negate much of the enhanced sequestration potential of urban soils.

The addition of artificial chemicals has shown the ability to enhance soil and turf health in urban ecosystems; however, the emissions created through their use must be weighed against the benefits provided. For each 1 kg of fertilizer added to turf, 0.1–0.8 kg of carbon equivalent (Ce) is released into the atmosphere. Emissions for insecticides and fungicides are even greater, releasing 1.2–12.6 kg Ce for each 1 kg of active ingredient.^[32] For intensively managed systems such as golf courses, this can lead to significant atmospheric emissions ranging from 0.64 kg Ce ha⁻¹ for insecticides to 32.03 kg Ce ha⁻¹ for N-based fertilizers (Table 2).^[12] Thus, highly managed green spaces

Table 2 Total carbon emissions per hectare for golf course maintenance practices.

Emission source	Total emissions (kg Ce ha ⁻¹)
Unleaded fuel	77.36
Diesel fuel	140.20
Irrigation	13.38
Insecticide	0.64
Herbicide	4.40
Fungicide	29.44
Nitrogen fertilizer	32.03
Phosphorous fertilizer	2.05
Potassium fertilizer	2.95
Total Emissions	302.44

such as golf courses may not provide the most efficient strategy for the sequestration of atmospheric C.

Due to the decreased HCCs and higher proportion of land area, less intensively managed urban turf soils may provide a more efficient means of sequestering atmospheric C. For home lawns, emissions due to fertilization at the industry-recommended rate account for an average annual emission of 63.6 kg Ce ha⁻¹ yr⁻¹ (Fig. 4).^[13] Although much lower than in golf systems, this represents a significant annual output of C due to management practices. Additionally, secondary nitrous oxide (N₂O) emissions from such areas due to fertilization are significantly elevated, releasing as much as 0.1–0.3 g m⁻² yr⁻¹.^[19] This is especially important to note, as N₂O is a greenhouse gas with a global warming potential 310 times greater than that of CO₂.

In addition to supplemental chemical use, regular mowing of turfgrasses with fossil fuel generated equipment can further limit the net sequestration potential of some urban soils. For each 1 kg of fuel burned, 0.85 kg Ce is emitted into the atmosphere.^[33] For golf courses that utilize regular

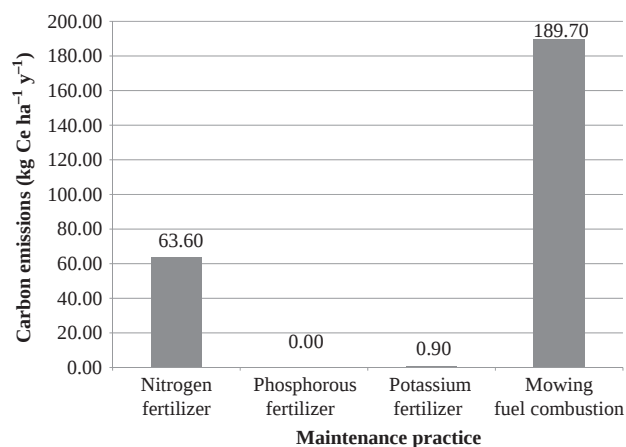


Fig. 4 Carbon emissions due to home lawn turf maintenance practices.

daily mowing and other fossil fuel burning equipment as part of their management practices, this can account for as much as 217.56 kg Ce ha⁻¹ yr⁻¹ (Table 2).^[12] Again, such emissions are greatly reduced for soils maintained for home lawn use with emissions due to mowing accounting for an estimated 189.7 kg Ce ha⁻¹ yr⁻¹ (Fig. 4).^[13]

Furthermore, irrigation emissions are common in urban ecosystems, especially in dry climates. In these areas, excess emissions are produced due to the pumping of water and other environmental concerns during periods of water shortage. For intensively managed systems such as golf courses, supplemental water usage can account for as much as 13.38 kg Ce ha⁻¹ (Table 2).^[12] Furthermore, the installation of irrigation equipment can release as much as 121 kg Ce ha⁻¹ due to fossil fuel burning^[34] and might provide additional concerns as installation often disrupts the soil profile and can lead to the release of previously sequestered SOC.

Additional secondary environmental concerns may further lessen the positive impact of increased C sequestration in urban soils. It has been observed that the majority of urban turfgrass systems have become contaminated through the use of chemical additions. Up to 53% of the N fertilizer added to turfgrasses is lost as surface runoff,^[35] and 15% of all pesticides found in groundwater leached from home lawns.^[36] Not only does the loss of these chemicals from the soil lead to negative environmental externalities, but also to increased cost for the homeowner and the community.

Accounting for the emissions generated due to fossil fuel use, application of maintenance chemicals, and supplemental water use through irrigation, the net C sequestration potential of urban soils is greatly reduced. For intensively managed turfgrass soils such as golf courses, the significant emissions produced shift courses from sinks to sources just 30 years post-establishment, thus limiting the overall potential of the these soils to mitigate GCC.^[23] However, due to the increased acreage and reduced management emissions, less intensively maintained systems provide a greater potential to mitigate the increased rate of atmospheric C enrichment.

Based on the available data, combined U.S. home lawn acreage has a C sink capacity of 496.3 Tg of C (Table 3).^[13] However, in order to determine the GCC mitigation potential of these soils, C emitted to the atmosphere through management practices is subtracted from the total sink capacity to determine the net C sequestration potential. While maintenance emissions are significantly lower for home lawns than for golf systems, lawn soils emit between 2.5 and 7.55 Tg Ce yr⁻¹ (low vs. high lawn management strategies; Table 3). Taking into account the C sink capacity and standard maintenance emissions, home lawns throughout the United States illustrate a potential to act as sinks for 66–199 years before shifting to sources. If fossil-fuel-based mowing emissions are removed, C sink time could be significantly lengthened to 130–775 years (Table 3).

Table 3 Potential organic carbon sink capacity, hidden carbon costs (HCCs) due to maintenance emissions, and time for sequestration potential to be negated by emissions for U.S. home lawn soils under various maintenance regimes.

Management intensity	Potential soil organic carbon sink capacity (Tg C)	HCCs of maintenance emissions (Tg Ce yr ⁻¹)	Time until sequestration = emissions (years)
Standard	496.3	2.50–7.55	66–199
Mowing only	496.3	1.87–3.74	133–265
Fertilization only	496.3	0.64–3.81	130–775

While this is impractical in many cases, large quantities of home lawns throughout the United States are small enough to utilize non-fossil-fuel-based rotary mowers. Additionally, developments of solar-generated mowing equipment could allow for a large-scale reduction in mowing emissions. It is important to note, however, that these values refer to the sequestration potential of newly established home lawns. Over time, turfgrass soils become C saturated and no longer illustrate the potential to sequester C within the soil. As many home lawns have been established for some time, their potential to sequester C may be partially or entirely eliminated, thus lowering the total potential C sink capacity reported.

In order to maximize the potential of urban soils to sequester C, a low management regime is suggested. This would consist of fertilizer and other chemical application at the minimum rate necessary to sustain a healthy lawn. Excess application results in emissions, runoff, and leaching of chemicals into groundwater. Additionally, proper irrigation strategies that utilize supplemental water application only when necessary limit pumping emissions, N₂O production, and excessive water use. Furthermore, avoidance of mowing can reduce fossil fuel use and in many cases increase root stability and turf health. This could result in decreased emissions coupled with an increase in the C sequestration potential of the soil. Finally, the establishment of turfgrasses in areas that require high water and/or chemical demands should be avoided. Extremely arid soils may benefit more from the incorporation of xeriscapes or through the establishment of drought-tolerant plant species with mulching incorporated than from the establishment of home lawns or urban parks. Quality land management practices and urban design can serve to maximize the potential benefits of urban soils while reducing the impending drawbacks.

CONCLUSION

Examination of global research findings illustrates a clear ability of urban soils to sequester atmospheric C. Additionally, the establishment of turfgrasses on such soils may

successfully enhance the C sequestration potential of urban ecosystems. This potential may be enriched or hindered based on the climatic and soil properties of the site. Turf-grasses established in cooler climates with moderate rainfall may provide optimal conditions for soil C sequestration. Establishment of grasses on soils with a p_b of 1.45–1.55 Mg m⁻³ and sufficient levels of soil N may further bolster the soil's potential to sequester C. As such, the ability of urban soils to sequester C may be maximized by maintaining soil properties in the desired ranges and focusing urban turfgrass development in regions with optimal climates.

Based upon the research reviewed in this entry, the establishment of green space through the use of urban parks and the planting of home lawns could provide a sink for excess atmospheric C. However, sequestration potential must be weighed against the HCCs generated through turfgrass maintenance practices. While newly established urban turfgrasses are likely to sequester C for a period of time, the continued mowing and C-intensive fertilization of these turfs may negate all of the sequestration benefits. As such, urban green space should be maximized in areas with optimal climatic conditions and in areas where soil properties are ideal for C sequestration. Additionally, a research aimed at determining the optimal level of maintenance necessary to maximize sequestration potential while minimizing the HCCs could prove vital. With such information, turf managers and homeowners alike could utilize research-generated best practices in order to maximize the ability of urban soils to mitigate GCC.

ACKNOWLEDGMENTS

This entry is based in part on work done for the completion of the PhD in Environmental Science from The Ohio State University. As such, I would like to thank The Ohio State University and the Environmental Science Graduate Program for their financial and academic support. Additionally, I would like to express my gratitude to Scotts Lawn Service for providing additional funding and advice throughout the program. Finally, this entry is based in part upon work supported by the National Science Foundation under grant DGE-0638669. Any opinions, findings, and conclusions or recommendations expressed in this entry are those of the author and do not necessarily reflect the views of the National Science Foundation.

REFERENCES

1. Seto, K.C.; Güneralp, B.; Hutyra, L.R. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *P. Natl. Acad. Sci.* **2012**, *109* (40), 16083–16088.
2. Lal, R. Soil carbon sequestration to mitigate climate change. *Geoderma* **2004**, *123*, 1–22.
3. McHale, M.R.; McPherson, E.G.; Burke, I.C. The potential of urban tree plantings to be cost effective in carbon credit markets. *Urban For. Urban Gree.* **2007**, *6*, 49–60.
4. Tyson, K.C.; Roberts, D.H.; Clement, C.R.; Garwood, E.A. Comparison of crop yields and soil conditions during 30 years under annual tillage or grazed pasture. *J. Agr. Sci.* **1990**, *115*, 29–40.
5. Haynes, R.J.; Swift, R.S.; Steohen, R.C. Influence of mixed cropping rotations (pasture-arable) on organic matter content, water stable aggregation and clod porosity in a group of soils. *Soil Till. Res.* **1991**, *19*, 77–81.
6. Mensah, F.; Schoenau, J.J.; Malhi, S.S. Soil carbon changes in cultivated and excavated land converted to grasses in east-central Saskatchewan. *Biogeochemistry* **2003**, *63*, 85–92.
7. Gebhart, D.L.; Johnson, H.B.; Mayeux, H.S.; Polley, H.W. The CRP increases soil organic carbon. *J. Soil Water Conserv.* **1994**, *49*, 488–492.
8. Qian, Y.L.; Follett, R.F. Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. *Agron. J.* **2002**, *94*, 930–935.
9. Bandaranayake, W.; Qian, Y.L.; Parton, W.J.; Ojima, D.S.; Follett, R.F. Estimation of soil organic carbon changes in turfgrass systems using the CENTURY model. *Agron. J.* **2003**, *95*, 558–563.
10. Huh, K.Y.; Deurer, M.; Sivakumaran, S.; McAuliffe, K.; Bolan, N.S. Carbon sequestration in urban landscapes: The example of a turfgrass system in New Zealand. *Aust. J. Soil Res.* **2008**, *46*, 610–616.
11. Qian, Y.L.; Follett, R.F.; Kimble, J.M. Soil organic carbon input from urban turfgrasses. *Soil Sci. Soc. Am. J.* **2010**, *74*, 366–371.
12. Selhorst, A.L.; Lal, R. Carbon budgeting in golf course soils of central Ohio. *Urban Ecol.* **2011**, *14*, 771–781.
13. Selhorst, A.L.; Lal, R. Carbon budgeting in golf course soils of Central Ohio. *Environ. Manage.* **2013**, *51* (1), 198–208.
14. Shanghai Bureau of Statistics. *Shanghai Statistical Yearbook*; China Statistics Press: Beijing, 2007.
15. Pouyat, R.V.; Yesilonis, I.D.; Nowak, D.J. Carbon storage by urban soils in the United States. *J. Environ. Qual.* **2006**, *35* (4), 1566–1575.
16. Terumasa, T.; Yoshihiro, A.; Kayo, K.; Kobayashi, T. Carbon content of soil in urban parks in Tokyo, Japan. *Landsc. Ecol. Eng.* **2008**, *4* (2), 139–142.
17. Schwendenmann, L.; Mitchell, N.D. Carbon accumulation by native trees and soils in an urban park Auckland. *New Zeal. J. Ecol.* **2014**, *38* (2), 213–220.
18. Tiquia, S.M.; Lloyd, J.; Herms, D.A.; Hoitink, H.A.J.; Michel, F.C. Effects of mulching and fertilization on soil nutrients, microbial activity and rhizosphere bacterial community structure determined by analysis of TRFLPs of PCR-amplified 16 S rRNA genes. *Appl. Soil Ecol.* **2002**, *21*, 31–48.
19. Townsend-Small, A.; Czimczik, C. Carbon sequestration and greenhouse gas emissions in urban turf. *Geophys. Res. Lett.* **2010**, *37*, 1–5.
20. Selhorst, A.; Lal, R. Effects of climate and soil properties on U.S. home lawn soil organic carbon concentration and pool. *Environ. Manage.* **2012**, *50* (6), 1177–1192.
21. Sanderman, J.; Amundson, R.; Baldocchi, D.D. Application of eddy covariance measurements to the temperature

- dependence of soil organic matter mean residence time. *Global Biogeochem. Cy.* **2003**, *17*, 1061–1075.
22. Crafts-Brandner, S.J.; Salvucci, M.E. Rubisco activase constrains the photosynthetic potential of leaves at high temperature and CO₂. *P. Natl. Acad. Sci. USA* **2000**, *97*, 13430–13435.
 23. Xu, Q. Morphological and physiological characteristics associated with heat tolerance in creeping bentgrass. *Crop Sci.* **2001**, *41*, 127–133.
 24. Shen, C.; Wang, W.C.; Peng, Y.; Zheng, J. Variability of summer precipitation over eastern China during the last millennium. *Clim. Past* **2008**, *4*, 611–643.
 25. Derner, J.D.; Schuman, G.E. Carbon sequestration and rangelands: A synthesis of land management and precipitation effects. *J. Soil Water Conserv.* **2007**, *62* (2), 77–85.
 26. Carrow, R.N. Drought resistance aspects of turfgrasses in the southeast: Root-shoot responses. *Crop Sci.* **1996**, *36*, 687–694.
 27. Brevik, E.C.; Fenton, T.E.; Moran, L.P. Effect of soil compaction on organic carbon amounts and distribution, south-central Iowa. *Environ. Pollut.* **2002**, *116*, 137–141.
 28. Hassink, J. The capacity of soils to preserve organic C and N by their association with clay and silt particles. *Plant Soil.* **1997**, *191*, 77–87.
 29. Yang, X.M.; Kay, B.D. Impacts of tillage practices on total, loose- and occluded-particulate, and heavy organic carbon fractions in soils within a field in southern Ontario. *Can. J. Soil Sci.* **2001**, *81*, 149–156.
 30. Magnani, F.; Mencuccini, M.; Borghetti, M.; Berbigier, P.; Berninger, F.; Delzon, S.; Grelle, A.; Hari, P.; Jarvis, P.G.; Kolari, P.; Kowalski, A.S.; Lankreijer, H.; Law, B.E.; Lindroth, A.; Loustau, D.; Manca, G.; Moncrieff, J.B.; Rayment, M.; Tedeschi, V.; Valentini, R.; Grace, J. The human footprint in the carbon cycle of temperate and boreal forests. *Nature* **2007**, *447*, 848–850.
 31. Vitousek, M.; Aber, J.D.; Howarth, R.W.; Likens, G.E.; Matson, P.A.; Schindler, D.W.; Schlesinger, W.H.; Tilman, D.G. Human alteration of the global nitrogen cycle: Sources and consequences. *Ecol. Appl.* **1997**, *7*, 737–750.
 32. Lal, R. Carbon emissions from farm operations. *Environ. Int.* **2004**, *30*, 981–990.
 33. Fluck, R.C. Energy in farm production. In *Energy in World Agriculture*, 6th Ed.; Fluck, R.C., Ed.; Elsevier: New York, 1992; 218–267.
 34. Batty, J.C.; Keller, J. Energy requirements for irrigation. In *Handbook of Energy Utilization in Agriculture*; Pimentel, D., Ed.; CRC Press: Boca Raton, 1980; 35–44.
 35. Petrovic, A.M. The fate of nitrogenous fertilizers applied to turfgrass. *J. Environ. Qual.* **1990**, *19*, 1–14.
 36. Gold, A.J.; Groffman, P.M. Leaching of agrichemicals from suburban areas. *ACS Sym. Ser.* **1993**, *522*, 182–190.

Carbon Sink Capacity: Soil Profile Characteristics

Yuri Lopes Zinn

Federal University of Lavras, Lavras, Brazil

Abstract

Soils are increasingly considered as viable, effective carbon (C) sinks that play a major role in the global C cycle. Most research work on this issue focused on the effect of external factors such as climate, soil management, and vegetation on soil C sink capacity, whereas the influence of the soil profile characteristics has been generally overlooked. This entry reviews how soil organic carbon (SOC) and soil inorganic carbon (SIC) sink capacities are affected by a soil's inherent properties. The role of soil structure, particle size distribution, clay mineralogy, and depth on SOC retention was reviewed, whereas soil pH and base cation availability were discussed in regard to their control on SIC sink capacity.

INTRODUCTION

Increasing concern about global warming due to anthropogenic carbon dioxide (CO₂) emissions to the atmosphere has raised interest on soils as potential carbon (C) sinks. In fact, soils of the world store C in amounts higher than those in biomass or the atmosphere, although lower than those in geological deposits.^[1] In soils, C is stored either as soil organic carbon (SOC) or as soil inorganic carbon (SIC). The turnover times for both forms are generally much shorter than those for hydrocarbons or carbonates of sedimentary rocks, but on the other hand, they are longer than those for C stored in plant biomass. In other words, soils are not the largest C pool on earth, but they represent a C sink capacity that can be achieved relatively faster and at lower costs than geological pools and in more stable forms than those in wood or forest biomass. A great deal of research has been conducted on how environmental or external factors such as climate, vegetation, and management affect the retention of C, especially as SOC.^[2,3] This entry presents a brief review on a different perspective, i.e., on which and how soil internal factors, or profile characteristics, affect soil C sink capacity.

SOC SINK CAPACITY

The cycling of dead organic matter is one of the main ecological services provided by soils, releasing mineral nutrients to plants during the decomposition of those residues. This is a complex process that encompasses numerous decomposition reactions and rates^[3] for each organic functional group and decomposer organism, with turnover times ranging from hours to centuries or even millennia. Furthermore, a fast, full mineralization to CO₂ is common only for the simplest organic compounds, and partial

decay of large macromolecules such as cellulose and lignins constantly produces intermediate compounds, in a process called humification.^[3] These by-products are the highly altered humic substances, comprise ca. 2–20% of the original C input,^[1] and are stored in soils altogether with fresh solid residues and smaller, more soluble intermediates. This mix of compounds varies widely in particle or molecular size, in surface or charge properties, and in their affinity for water, resulting in the inherently intricate nature of SOC. In consequence, it follows that different SOC components are stored in different soil compartments, by means of various mechanisms, which depend on the soil profile characteristics described in the following sections.

Soil Structure

SOC concentration is determined in fine earth samples, excluding materials >2 mm. The coarsest (sand-sized) components of SOC chiefly consist of fresh or slightly altered plant debris, although partially burnt and old, recalcitrant materials also often occur. These materials are commonly named as particulate organic matter (POM) and can exist either occluded within soil aggregates or interspersed among the aggregates (“free” POM).^[4] Soil aggregates are the secondary particles built up by primary (sand, silt, and clay) particles and SOC, in different shapes and sizes, which compose soil structure.^[5] The occlusion of POM within aggregates is a process that slows down its decomposition by reducing aeration and access to decomposers, leading to slower turnover^[4] and a general perception that soils with a higher degree of aggregation store more SOC. In fact, one of the main reasons why soils are tilled is to promote SOC decomposition and release nutrients to crops,^[6] an effect ascribed to the disruption of

aggregates, enhancing oxygenation and exposing labile POM to decomposers. This is the main reason why continuous annual plowing often leads to SOC depletion, in comparison with no-till farming.^[5] By the same token, a positive feedback between SOC sink capacity and soil structure exists,^[7] which can be seen as an improvement in soil aggregate stability as SOC levels increase.

It is difficult to ascribe precisely how much POM–SOC is actually free or occluded, because: 1) sieving samples <2 mm releases POM originally occluded within large macroaggregates; 2) aggregate size distribution changes seasonally and with cultivation practices; 3) measurement protocols, commonly based on density separation or particle size fractionations, vary widely in the literature, making interstudies comparison difficult. However, there is evidence that most POM in forest or native soils occurs actually outside aggregates (Fig. 1), but with continuous cultivation, this situation is reversed as most free POM rapidly decays.^[4] It can be generally expected that, for the same climate and drainage conditions, soils with more water-stable aggregates will hold more bulk SOC than those with less stable aggregation or smaller aggregates,^[5] although this effect may be restricted to surface layers where aggregation factors (e.g., wetting–drying cycles and soil fauna) are more active.^[7]

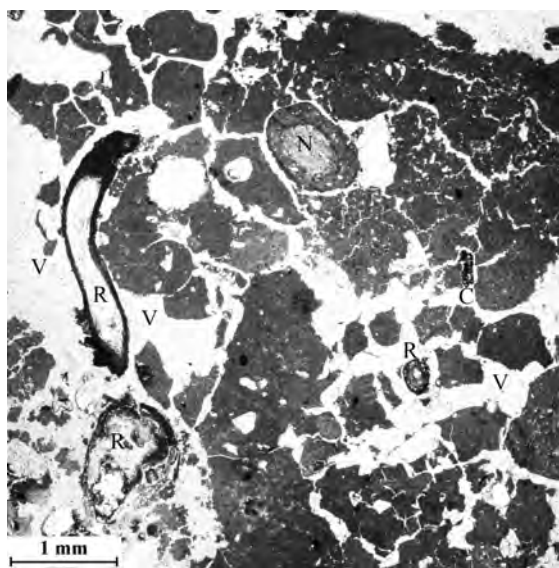


Fig. 1 Surface horizon of a gibbsitic Oxisol under native savanna vegetation in Brazil. Most POM occurs as roots (R) in the voids (V) among the granular structure, i.e., outside aggregates, although a charcoal (C) fragment can be seen occluded within an aggregate. The root-like feature in the upper center is actually a gibbsite nodule (N). Although this soil has a total organic carbon content of 4%, it has a light reddish color due to the significant presence of Fe and Al oxides masking the dark color otherwise expected for soils with such high C concentrations.

Source: Photo in planar polarized light by Marla A. Araujo, Federal University of Lavras, Brazil.

Soil Particle Size Distribution

In well-drained soils of many parts of the world, SOC concentrations are highly correlated with the concentrations of soil particles of size smaller than clay (2 μm) or silt (20 or 50 μm). This effect has been ascribed to various mechanisms, including inactivation of SOC-oxidizing enzymes, greater water retention leading to higher plant biomass and C inputs, stronger aggregation, and more cation bridges between SOC and clays.^[3,8] However, since the clay content of a soil strongly controls its specific surface area, it is more likely that the major mechanism involved in this textural control is the greater availability of mineral surfaces able to sorb soluble and colloidal SOC, in soils of finer textures.^[8] In fact, when different textured soils have their particle size fractions separated and analyzed, the higher SOC sink capacity in the finer soils can be ascribed to larger C pools in the clay size, whereas the sand and silt size C pools vary little with soil texture.^[8] This trend demonstrates that the main effect of higher clay contents is the stabilization^[3] of soluble and colloidal SOC by sorptive effects and has the critical consequence that the clay fraction generally holds more of the total SOC than the silt and sand fractions. Soil structure has an indirect effect on this textural control, as a higher degree of clay flocculation and cementation is likely to improve SOC stabilization.

Clay Mineralogy

A direct consequence of the sorption-based textural control described above is that the nature of clay minerals and their surfaces strongly affect SOC sink capacity. A general rule states that the smaller the size of the crystals, the higher the specific surface area and the higher the potential for sorption. Thus, amorphous silicon, aluminum (Al), and iron (Fe) phases common in Andisols and Spodosols favor SOC sequestration^[3] in comparison with well-crystallized clay minerals with lower surface areas. However, there is a large variability in terms of surface properties, specific area, and charge density, not only among the different clay mineral types but also for different provenances of the same mineral. Studies on sorption of dissolved organic C by clays are many, but just like those on soil structure, a comprehensive understanding is hindered by non-standardization of analytical procedures and disparate experimental conditions. Nevertheless, it is evident that highly active smectite clays are more efficient than kaolinite in sorbing SOC when sorption results are expressed on a mass basis, but when calculated in a specific surface area basis, kaolinite is more effective than smectites, perhaps due to its Al–OH surfaces.^[9] There is also evidence of an even greater SOC affinity to crystalline Fe oxides such as goethite; since in some cases, SOC and Fe dithionite concentrations show even higher correlation than with clay + silt.^[8] This effect is not restricted to aerobic soils, having been reported that Fe oxides respond for ca. 20% of total organic C retention

in oceanic sediments.^[10] Goethite is known to greatly increase SOC sorption to kaolinite and to decrease its desorption, but this effect is null for smectites and illites,^[9] which may suggest that kaolinitic, Fe-rich tropical soils have a high C sink capacity. In addition, an indirect effect of clay mineralogy (and texture) on SOC sink capacity can be exerted through aggregation, which is more stable in soils rich in Fe and Al oxides.^[5,7] In such soils, even high concentrations of SOC have little or no darkening effect on the soil color (Fig. 1), which in Fe-poor soils of temperate regions generally result in black colors.

Soil Depth and Bulk Density

Soil sampling in subsurface depths has always been limited in SOC studies,^[8] which is not only due to procedural difficulties but also due to a general expectancy that changes in SOC are restricted to the soil surface. Soil depth is a profile characteristic that influences SOC sink capacity in at least two ways, the most obvious being that deeper soils provide more mineral surfaces to sorb SOC than shallow soils. SOC can reach deeper layers by root colonization and activity, but especially by downward flow of dissolved SOC less strongly bound to soil minerals.^[11] This effect is not restricted to A and B horizons, since C horizons have very low SOC concentrations (<0.1%) but can be so deep and dense that total SOC stocks can reach 300 Mg ha⁻¹ in >30 m deep profiles, even under relatively dry Mediterranean climate.^[12] The downward migration of SOC also defines the spodic and sombric horizons, and although these features are not causally ascribed to depth as a factor per se, their location often below 1 m is another example of how depth can influence SOC stocks in regional estimates and actual C sink capacity.

Soil depth apparently also affects SOC sink capacity by determining how deep and thoroughly SOC can migrate downward and thus be removed from topsoil giving an erroneous idea of lower C sink capacity. This effect is also determined by soil bulk density and structure patterns. In the humid tropics, very deep, highly porous soils such as Oxisols with granular structure often have lower SOC concentrations in topsoil in comparison with shallower, more dense Ultisols and Inceptisols. However, in the latter soils, SOC concentrations tend to decrease sharply with depth, whereas in Oxisols such decrease is less evident and concentrations of 1% at a depth of 2 m are not uncommon.^[13] More research is needed on this effect of depth, including not only deep samplings but also a mathematical analysis and modeling of involved factors and their impact on SOC stratification.

SIC SINK CAPACITY

In this text, SIC refers only to carbonate minerals generated in soil environments, i.e., excluding those present within

soils but inherited from calcareous parent materials upon weathering. Due to its synthesis (and resynthesis), SIC is often of a much finer grain size than inherited carbonates, and quantitative assessments of how much carbonate-C is actually pedogenic are difficult and involve microscopic or isotopic techniques.^[14,15] It has been estimated that SIC is sequestered at rates of 0.005–0.15 Mg ha⁻¹ yr⁻¹, i.e., much slower than 0.1–1.0 Mg ha⁻¹ yr⁻¹ attributed to SOC.^[1] In addition, ca. 950 Pg of SIC is stored in soils of the world, to a depth of 1 m,^[1] a figure that comprises only two-thirds of the amounts estimated to be stored as SOC. Nevertheless, SIC is relevant as a C sink since its occurrence is favored in the semiarid and arid lands that occupy ca. 37% of the earth's land surface.^[14] Semiarid or drier climatic regimes are generally associated with the two soil profile characteristics that allow for the genesis and stability of SIC, i.e., the existence of near-neutral to alkaline pH and the availability of basic cations.

Neutral and Alkaline pH

The principal reactions involving carbonates in soils are well known,^[16] but since they are multiple and cannot be treated in detail here, they are summarized as follows:^[14,15]



This reversible reaction basically shows that calcite, the main calcium carbonate in soils and rocks, dissolves due to the acidity (H⁺) generated by dissociation of carbonic acid, a weak acid, formed by the subreaction between CO₂ and H₂O (carbonation). In addition, the equation shows that the dissolved ions can precipitate again in the soil by desiccation and that increasing humidity and CO₂ pressure promote carbonate dissolution. Moisture also has the indirect, longer term effect of promoting plant growth and thus the formation of various organic acids and humic substances that protonate and dissolve carbonates. The equilibrium of the reaction varies swiftly with Ca²⁺, water, and CO₂ concentrations as well as with interference of factors such as dissolved organic C and other ions, so it is quite difficult to discern when carbonates are forming or dissolving in net terms.^[16] However, in the long term, SIC is formed and is relatively stable at slightly alkaline soil conditions (pH > 7.2) and is not expected in soils with pH values <6.5. As a consequence, SIC occurs in Aridisols, Mollisols, Vertisols, and some Alfisols but not in acidic soils such as Oxisols, Ultisols, and Histosols.

Base Cations Availability

As noted in the equation above, base cations (especially Ca²⁺, but also Mg²⁺, Na⁺, K⁺, and Fe²⁺)^[15] are required either dissolved or sorbed by clay surfaces in order to precipitate SIC with CO₂-rich soil solutions. To promote net SIC sequestration, these cations must come from sources

other than dissolution of preexisting carbonates. Deposition of gypsum dust and Ca^{2+} -laden rainwater can be important,^[17] but the weathering of silicate minerals (e.g., feldspars, micas, amphiboles, pyroxenes, and olivines), which form the bulk of earth's crustal rocks, release most of these cations. In semiarid or drier regions where bedrock is comprised of basic to intermediate rocks, these minerals can be abundant in soils as sand and silt grains. After the infrequent rain events, such grains undergo chemical weathering, mostly by hydrolysis with the H^+ produced by carbonation, and base cations are released in solution. Upon evaporation, these cations precipitate with the HCO_3^- and CO_3^{2-} ions, mostly forming fine-grained calcite, magnesium–calcite, and others on and near partially weathered silicate grains, easily visible in soil thin sections.^[16] Alkaline dissolution of silicates in contact with these carbonates in soils is visible as serrated grain surfaces,^[16] which further accelerates cation release and carbonate precipitation. The multiplication and growth of SIC crystals promote cementation of soil aggregates and layers (caliches, calcretes, “k” horizons, etc.),^[17] implying that, unlike SOC, SIC sequestration may not increase soil quality since it can restrict root development. Due to the inherent low contents of clay minerals and Fe/Al oxides in semiarid or drier soils, there is little apparent effect of soil texture, mineralogy, and structure on SIC sequestration, as is the case for SOC sequestration.

CONCLUSION

The main soil profile characteristics affecting the C sink capacity either as organic or as inorganic compounds are reviewed and interpreted envisaging a novel understanding on how soil C dynamics can be managed. However, such initiatives are complicated by lack of standard analytical procedures, by insufficient sampling depth, and by mutual, indirect effects of all profile characteristics on the C retention mechanism of each other. The research on soils as sinks or sources of C must provide improved, thorough soil characterization in order to better provide insights into how profile characteristics interact with external controls of C retention, such as climate and site management. It is also recommended that the C sink capacity of charcoal within the soil matrix (Fig. 1) be studied with regard to whether and how it is affected by those profile characteristics.

ACKNOWLEDGMENT

The author would like to thank Ms. Marla A. Araujo, Federal University of Lavras, Brazil, for the micrograph given in Fig. 1.

REFERENCES

1. Lal, R. Soil carbon sequestration impacts on global climate change and food security. *Science* **2004**, 304, 1623–1627.
2. Jenny, H. *Factors of Soil Formation. A System of Quantitative Pedology*; McGraw-Hill: New York, 1941.
3. Quideau, S.A. Organic matter accumulation. In *Encyclopedia of Soil Science*; Lal, R., Ed.; Marcel Dekker: New York, 2002; 370–373.
4. Besnard, E.; Chenu, C.; Balesdent, J.; Puget, P.; Arrouays, D. Fate of particulate organic matter in soil aggregates during cultivation. *Eur. J. Soil Sci.* **1996**, 47, 495–503.
5. Bronick, C.J.; Lal, R. Soil structure and management: A review. *Geoderma* **2005**, 124, 3–22.
6. Lal, R. Evolution of the plow over 10,000 years and the rationale for no-till farming. *Soil Till. Res.* **2007**, 93, 1–12.
7. Zinn, Y.L.; Lal, R.; Bigham, J.M.; Resck, D.V.S. Edaphic controls on soil organic carbon retention in the Brazilian Cerrado: Soil structure. *Soil Sci. Soc. Am. J.* **2007**, 71, 1215–1224.
8. Zinn, Y.L.; Lal, R.; Bigham, J.M.; Resck, D.V.S. Edaphic controls on soil organic carbon retention in the Brazilian Cerrado: Texture and mineralogy. *Soil Sci. Soc. Am. J.* **2007**, 71, 1204–1214.
9. Saidy, A.R.; Smernik, R.J.; Baldock, J.A.; Kaiser, K.; Sanderman, J. The sorption of organic carbon onto different clay minerals in the presence and absence of hydrous iron oxide. *Geoderma* **2013**, 209–210, 15–21.
10. Lalonde, K.; Mucci, A.; Ouellet, A.; Gélinas, Y. Preservation of organic matter in sediments promoted by iron. *Nature* **2012**, 483, 198–200.
11. Kaiser, K.; Kalbitz, K. Cycling downwards – dissolved organic matter in soils. *Soil Biol. Biochem.* **2012**, 52, 29–32.
12. Harper, R.J.; Tibbett, M. The hidden organic carbon in deep mineral soils. *Plant Soil* **2013**, 368, 641–648.
13. Zinn, Y.L.; Guerra, A.R.; Oliveira, G.C.; Marques, J.J.; Silva, A.C.; Curi, N. Perfis de carbono orgânico do solo nas regiões Sul e Serra do Espinhaço Meridional, Minas Gerais: Modelagem em profundidade. *Rev. Bras. Ci. Solo.* **2012**, 36, 1395–1406.
14. Lal, R.; Kimble, J.M. Pedogenic carbonates and the global carbon cycle. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2000; 1–14.
15. Durand, N.; Curtis Monger, H.; Canti, M.G. Calcium carbonate features. In *Interpretation of Micromorphological Features of Soils and Regoliths*; Stoops, G., Marcelino, V., Mees, F., Eds.; Elsevier: Amsterdam, 2010; 149–194.
16. Doner, H.E.; Grossl, P.R. Carbonates and evaporites. In *Soil Mineralogy with Environmental Applications*; Dixon, J.B., Schulze, D.G., Eds.; SSSA: Madison, 2002; 199–228.
17. Buol, S.W.; Southard, R.J.; Graham, R.C.; McDaniel, P.A. *Soil Genesis and Classification*; Wiley-Blackwell: Chichester, 2010.

Carbon: Inelastic Neutron Scattering

Lucian Wielopolski

Environmental Sciences Department, Brookhaven National Laboratory, Upton,
New York, U.S.A.

Abstract

The growing importance of accurately determining soil carbon (C) is attributable to several factors concerning the quality and fertility of soil and to society's emerging need to broker C sequestration via soil C credits under the U.S. Department of Energy's Voluntary Greenhouse Gas Reporting Program, established by section 1605(b) of the Energy Policy Act of 1992. Thus, the two principal justifications for quantifying belowground soil C are: 1) the recognition of the effects of C on the quality and fertility of the soil for precision agriculture; and 2) photosynthetic processes resulting in belowground C sequestration that is critically important to temporarily mitigate the consequences of anthropogenic CO₂ emissions into the atmosphere. However, belowground studies pose special challenges because underground structures, organisms, and processes are invisible to humans. Soil is a dynamic mix of living and dead plant matter, fungi, macro- and microfauna, plant exudates, and animal waste products, embedded in a matrix of solid, liquid, and gases. Thus, tinkering with this "black box" of soil *mélange* to study its components alters its function.

INTRODUCTION

The systematic study of belowground ecosystems requires new instrumentation that will allow *in situ*, non-destructive analysis of soil. The standard method entails extracting a soil core and subsequently analyzing it in the laboratory *ex situ*.^[1] Despite this traditional approach being labor-intensive, destructive, slow, and, consequently, very limited in its utility and scope, it is the accepted standard for soil analysis. Two newly emerging technologies, laser-induced breakdown spectroscopy (LIBS)^[2] and near- and mid-infrared spectroscopy,^[3] being field deployable offer significant advantages over the conventional method, although both are destructive and not *in situ* methods. A third novel method for soil elemental analysis is based on the spectroscopy of gamma rays induced by inelastic neutron scattering (INS) from the elements present in the soil.^[4–6] As the system based on the INS method is hovering about 30 cm above the ground (it is mounted on wheels), it is the only truly non-invasive *in situ* method that can be operated in a stationary mode or can be used for contiguous scanning of large fields. All of the newly emerging techniques are in their infancy and require further development and characterization; however, early reports suggest very attractive and unique features for each of the methods that cannot be embraced by a single device. A brief description of the INS system and its characteristics is given here.

BASIS OF THE INS SYSTEM

The INS method is one of the three alternatives of neutron activation analysis (NAA), a widely established tool for

elemental analysis. Of the three, the most widespread method is that of delayed NAA wherein a thermal neutron, a neutron in thermal equilibrium with the surroundings, approximately 0.025 eV, is absorbed by the nucleus of an element, thereby creating a radioisotope that decays to its ground state by a delayed emission of particles and/or gamma rays. Alternatively, by capturing a thermal neutron, an excited nucleus may decay instantaneously to its ground state by the emission of gamma rays: this process is referred to as prompt gamma ray analysis. In the third modality, a fast neutron is inelastically scattered by a nucleus leaving it in an excited state, which decays promptly to the ground state by emitting gamma rays. For this INS process to occur, the incident fast neutrons must be with energies that are above the threshold energies of approximately 5 MeV, depending on the element. Naturally, the scattered neutrons differ in their energies from the incident ones. Furthermore, the process of fast neutrons that lose energy via elastic scatterings may take up to few hundred microseconds depending on the scattering medium. Therefore, the first two processes are delayed after the INS process. Fig. 1 depicts these three processes schematically where one of them is marked as the INS that is used for carbon (C) detection. It is important to point out that during delayed neutron activation the irradiation with neutrons and the analysis of the sample occur at two different places, while in the two prompt processes, both irradiation and analysis take place, simultaneously resulting in high count rates and elevated background owing to data acquisition in the vicinity of the radiation source.

A typical INS system consists of a source of fast, 14-MeV, neutrons provided by a pulsed neutron generator (PNG).^[7] Gating the data acquisition system during the

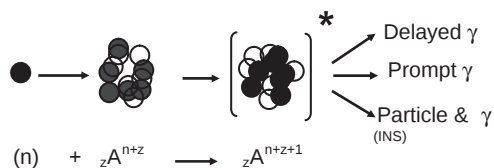


Fig. 1 Absorption processes with thermal and fast neutrons. Asterisk (*) denotes a nuclear excited state.

neutron pulse reduces the background resulting from gamma rays because of thermal neutron capture with the net effect of improving the signal-to-noise ratio. The additional components include a gamma ray detection system, nuclear spectroscopy with data acquisition system, and a mobile cart with a power generator. The operating conditions are such that 14-MeV neutrons produced by the PNG are directed toward the soil's surface, where they penetrate and undergo inelastic scatterings with C and other elements; they also undergo elastic scatterings losing energy until they thermalize and get absorbed. During the INS processes, characteristic gamma radiation at 4.44 MeV is emitted from the C nucleus and is detected by an array of NaI detectors. The detected signal is processed and acquired in a form of a gamma ray spectrum in which all the gamma rays are recorded and their peak intensities analyzed. The peak intensities are proportional to the concentrations of the elements present in the soil. The counting time is typically set for 30 minutes, although the duration will depend on the required precision and minimum detection limits. Our instrument was initially calibrated, and measurements were taken by placing the entire system on the sand. However, in Fig. 2, the latest prototype of a mobile system set at an optimized height above the ground for detecting C is shown.

RESULTS

The initial system was calibrated with homogeneous mixtures of soil and 0, 2.5, 5, and 10 wt.% C. A sandpit 132 cm wide, 152 cm long, and 46 cm deep was filled, in turn,

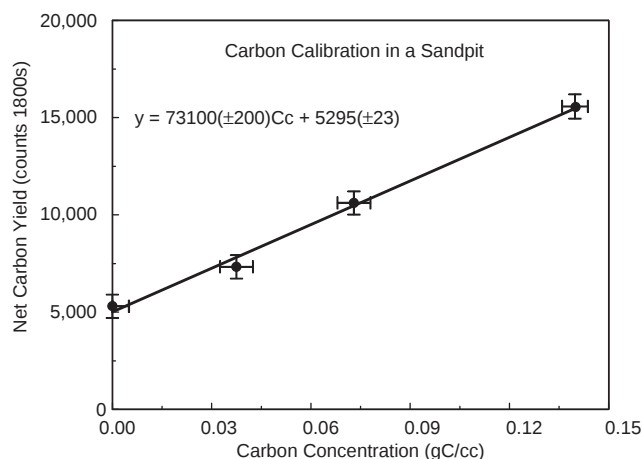


Fig. 3 Calibration line of C yield vs. C concentration.

with each of the mixtures, and gamma ray spectra were acquired for 30 minutes. However, after measurement of the sand bulk density, the abscissa in the calibration line in Fig. 3 was expressed in units of C concentration gC/cc instead of % gC/g. The rationale for this conversion was that, in principle, the instrument detects C from a constant volume as delineated by the neutron flux and the solid angle subtended by the detection system.

Two gamma ray spectra were acquired for 30 minutes: one was in coincidence with the neutron pulse and corresponded to the INS events, Fig. 4A, and the second was acquired in between the neutron pulses, thus corresponding to prompt gammas following thermal neutron capture, Fig. 4B.

Field measurements in a pine stand, in an oak forest, and in a sandy patch were conducted with the initial system lying on the top of the soil. Using the calibration line shown in Fig. 2, these measurements were compared with the results of C analysis of five cores sampled from each site. The core samples were 5 cm in diameter and divided into five sections 5, 5, 10, 10, and 10 cm long. As the C distribution in soil is seldom uniform with depth, and the process of proper averaging C concentration in soil is complicated as a first order of approximation, the INS results were

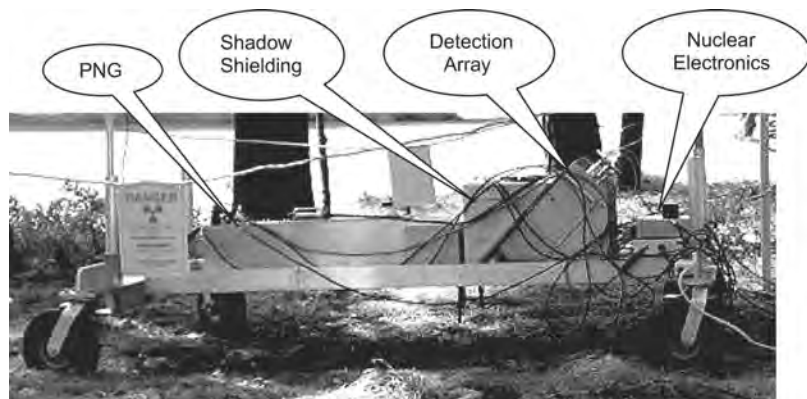


Fig. 2 A mobile system for C detection deployed for field measurements.

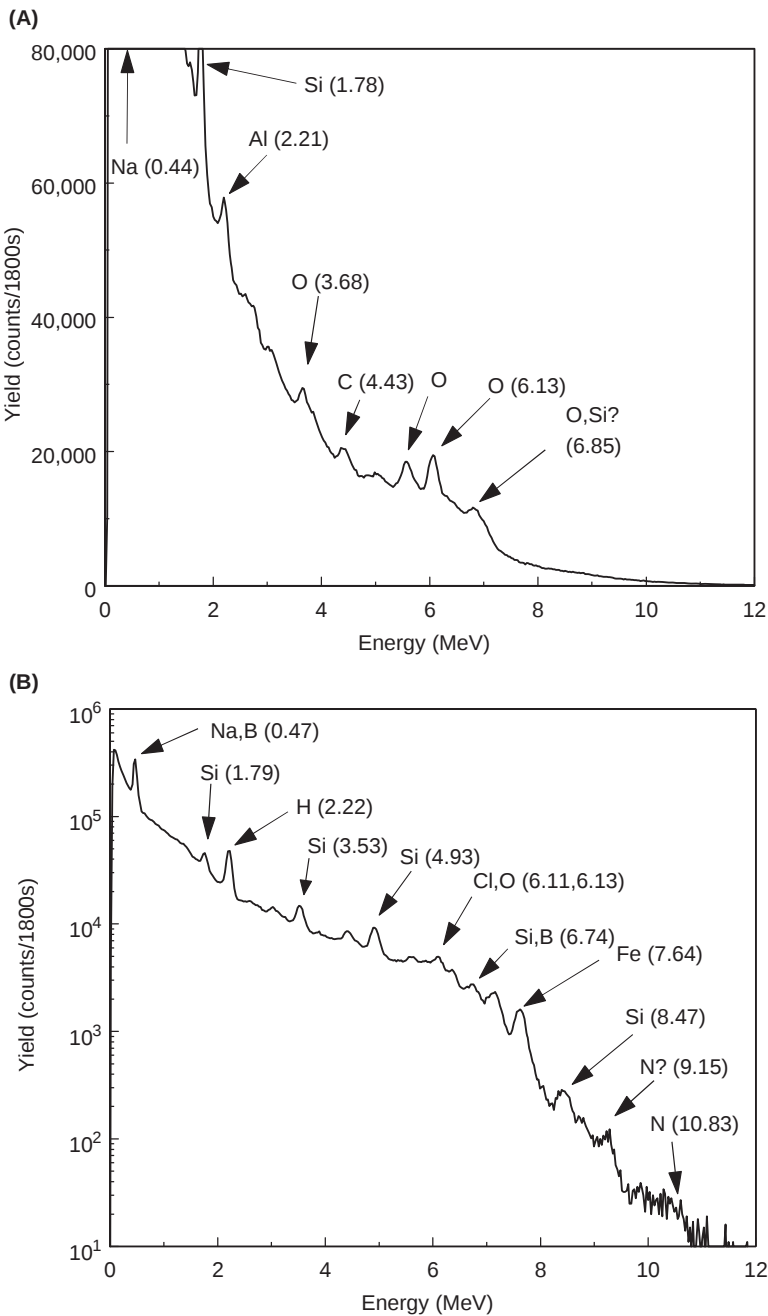


Fig. 4 Typical “inelastic” (A) and “prompt” (B) gamma spectra measured in a pine stand.

Table 1 Comparison of C analysis by INS from the test volume vs. chemical analysis from the top 5 cm.

Site	INS (gC/cc)	Chemical analysis (gC/cc)
Pine stand (w, l)	0.099 ∓ 0.005	—
Pine stand (w/on, l)	0.079 ∓ 0.005	0.073 ∓ 0.021
Oak forest (w/o, l)	0.072 ∓ 0.004	0.085 ∓ 0.017
Sandy patch	0.026 ∓ 0.003	0.025 ∓ 0.002
Sandy soil	0.091 ∓ 0.007	0.104 ∓ 0.019
Sandpit (Cal.)	0.00	0.0004 ∓ —

compared only with the chemical analysis of the top 5-cm layer of the soil samples, i.e., the mean value of the top 5-cm layer from the five cores in each site. The results are summarized in [Table 1](#).

CONCLUSION

These limited results demonstrate the viability of the INS method for measuring C in soil *in situ*. Its unique attributes make it a one-of-a-kind system that is completely non-destructive. Furthermore, as the processes involved in

acquiring the measurements are very fast (picoseconds), the system can be operated in a stationary mode or used for scanning large areas using a system as shown in Fig. 2. The operating depth of the INS system is about 20–30 cm, which may depend on soil conditions. Clearly, with these unknowns, the system needs to be recalibrated with a much more realistic C depth distribution in soil. An additional unique attribute of the INS system is that the sample size in a stationary mode is large, over 30 L. Thus, small-scale lateral variations in the C distribution are averaged, so that fewer samples are required to represent a given area. Consequently, a more realistic calibration procedure using large volume excavations should be used in various soil types instead of core samples. Because the system provides multi-elemental analysis, in principle, it should be possible to partition the measured total C to organic and inorganic components by measuring calcium and magnesium and using stoichiometry to determine the inorganic C component. Other elements of interest that can be measured are nitrogen, potassium, and possibly phosphorus. Water content appears to have small effect on the measurements, because the hydrogen elastic scattering probability at these energies is small.

In summary, a novel multi-elemental analysis system for *in situ* non-destructive analysis of large volumes or large areas of soil was described. Further developments of the INS system will satisfy all the requirements posed for detecting C levels in soil.

REFERENCES

1. Taylor, H.M.; Upchurch, D.R.; Brown, J.M.; Rogers, H.H. Some methods of root investigations. In *Plants Roots and Their Environment*; McMichael, B.L., Persson, H., Eds.; Elsevier Science Publisher B.V.: Amsterdam, 1991; 553–564.
2. Cremers, D.A.; Ebinger, M.H.; Breshears, D.D.; Unkefer, P.J.; Kammerdiener, S.A.; Ferris, M.J.; Catlett, K.M.; Brown, J.R. Measuring total soil carbon with laser-induced breakdown spectroscopy (LIBS). *J. Environ. Qual.* **2001**, *30*, 2202–2206.
3. McCarty, G.W.; Reeves, J.B., III; Reeves, V.B.; Follett, R.F.; Kimble, J.M. Mid-infrared and near-infrared diffuse reflectance spectroscopy for soil carbon measurement. *Soil Sci. Soc. Am. J.* **2002**, *66*, 640–646.
4. Wielopolski, L.; Orion, I.; Hendrey, G.; Roger, H. Soil carbon measurements using inelastic neutron scattering. *IEEE T. Nucl. Sci.* **2000**, *47*, 914–917.
5. Wielopolski, L.; Mitra, S.; Hendrey, G.; Rogers, H.; Torbert, A.; Prior, S. Non-destructive *in situ* soil carbon analysis: Principle and results. In *Proceedings of the Second Annual Conference on Carbon Sequestration*, May 3–5, 2003, Alexandria, VA, Article No. 225.
6. Wielopolski, L.; Mitra, S.; Hendrey, G.; Orion, I.; Prior, S.; Rogers, H.; Runion, B.; Torbert, A. *Non-Destructive Soil Carbon Analyzer (ND-SCA)*; BNL Report No. 72200-2004; Brookhaven National Lab: Upton, NY, 2004.
7. Csikai, J. *CRC Handbook of Fast Neutron Generators V1*; CRC Press: Boca Raton, 1987.

Carbon: Laser-Induced Breakdown Spectroscopy

Michael H. Ebinger

Ronny D. Harris

Clifton W. Meyer

*Earth and Environmental Sciences Division, Los Alamos National Laboratory,
Los Alamos, New Mexico, U.S.A.*

Abstract

Advanced methods of soil carbon (C) analysis are required to provide measurements of soil C at levels of accuracy and precision relevant to C trading and climate change research. Laser-induced breakdown spectroscopy uses a laser to interrogate a small soil sample from within intact soil cores. This method of analysis takes only a few seconds to accomplish, is cost-effective compared with conventional analysis methods, and can provide data on soil C distribution at 1 mm resolution in soil cores. This method correlates well with conventional analysis methods of the same soils and shows considerable promise as a cost-effective means to measure, monitor, and verify C sequestration in soils.

INTRODUCTION

The methods of carbon (C) analysis provide a set of analytical tools required to estimate changes in soil organic carbon (SOC) stocks with some precision.^[1] However, advanced analytical methods offer improved detection limits, increased ease of operation, and the potential to analyze samples in the field.^[2–6] These advanced methods could improve SOC measurements significantly and reduce uncertainties in C inventories. Improved accuracy and precision of C measurements are needed to support national and international policies on C emissions and C trading, but these improved methods must be more cost-effective than the existing methods. When fully developed and tested, advanced methods must optimize the amount of information about SOC pools per dollar spent and deliver C assessment data at less than 10% of the total cost of sequestration practices.^[7]

In this entry, we report preliminary calibrations of a new spectroscopic method for measuring total soil C that is based on atomic emission spectroscopy using laser-induced breakdown spectroscopy (LIBS).^[3,6,8–10] In this method, a laser is focused on a solid sample and forms a microplasma that emits light characteristic of the elemental composition of the sample. The emitted light is collected, spectrally resolved, and detected to monitor concentrations of elements via their unique spectral signatures. When calibrated, this method provides quantitative measurements of soil C in a sample. This method is readily adaptable to field-portable instrumentation, which would provide investigators a means to measure soil C in near real time and in remote locations^[11] or in a laboratory setting with high-throughput analysis. We evaluated LIBS

for its potential to measure total soil C at 1-mm intervals along a soil profile. C distribution data at this resolution can provide whole-profile C inventory estimates as well as new estimates of measurement uncertainty.

HIGH-RESOLUTION LIBS METHODOLOGY

Strong carbon C(I) emission lines at 247.8 nm are selected for the evaluation of LIBS.^[3,6,12,13] An Nd:YAG laser (Spectra-Physics Lasers, Mountain View, California, U.S.A.) at a wavelength of 1064 nm (50-mJ pulses of 10 ns) is focused with a lens of 50-mm focal length on each soil sample (Fig. 1). The emitted light is collected through a fused silica fiber optic cable pointed at the plasma from a distance of about 50-mm. A spectrograph of 0.5 m focal length resolves the light that is then detected using a gated intensified photodiode array detector. Intact soil cores are placed in the instrument for analysis. A stepping motor and movable stage are coupled to transport the cores through the LIBS instrument and allow collection of spectra each 1 mm along the length of the core. Peak areas are integrated to estimate signal intensities for each spectrum. The background signal was obtained from a shot into a blank sample and was subtracted from the spectrum. The C signal was ratioed to the signal of Si at 251 nm to minimize shot-to-shot instrumental variation. Typical measurement volumes for the LIBS analysis are 1–5 mm³.

C concentrations from each 1 mm along the core are obtained by collecting 20 shots from different points within the 1-mm band. The stepping motor moves the core slightly after each shot to prevent multiple analyses of a single

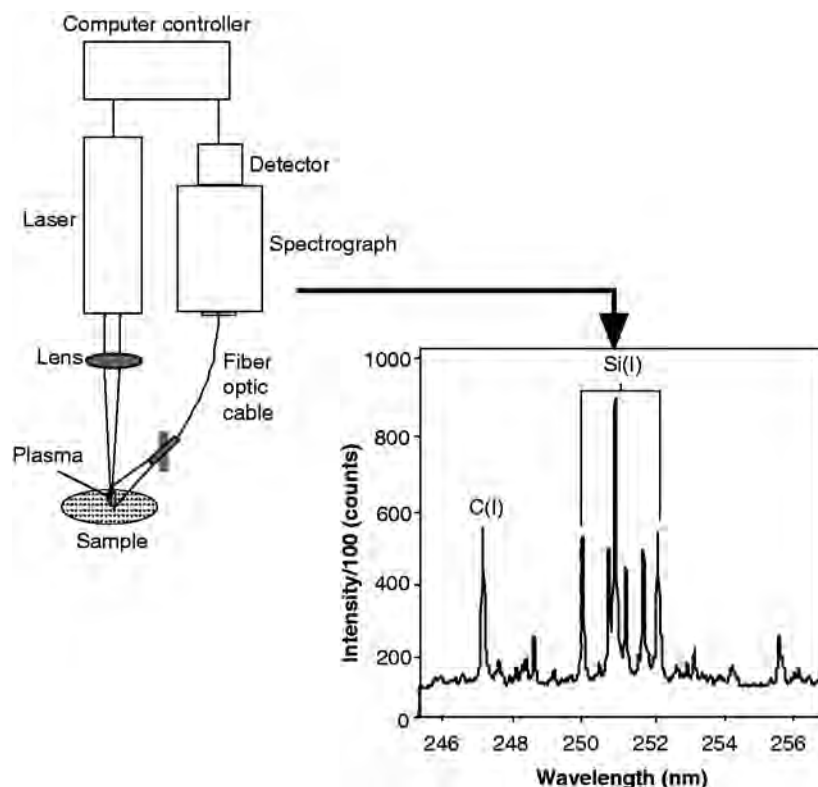


Fig. 1 Schematic representation of LIBS instrument, showing microplasma collection, detection, and spectral resolution of a sample.

location. C peak areas from the 20 shots are averaged into the point estimate of C concentration at each 1-mm location. Finally, an average C concentration for each 2.5-cm interval is computed using the estimates from all 1 mm measurements and by averaging these over the 2.5-cm interval. The averaging over the 2.5-cm interval allows comparison of LIBS data with dry combustion data from the same 2.5-cm intervals taken from core samples after LIBS analysis.

Intact soil cores are extracted using a hand-operated coring apparatus. Any method to remove an intact core should be adequate, including a Giddings probe or core from shallow drill rig operations. The core is collected in a plastic tube. After description of the horizons, a slit of approximately 10 mm is cut in the plastic tube to allow laser access. The core is then cycled through the LIBS instrument, and data are collected each 1 mm along the core. Twenty shots are averaged per step.

C peak areas are estimated from the raw data and plotted with core depth. The LIBS data are then summed over a 2.5 cm core depth, and the geometric mean of the LIBS measurements is computed for each interval. The geometric mean is used to minimize the effects of large variance in the LIBS-measured C concentration due to heterogeneity in the C distribution in each core (Fig. 2). After LIBS analysis, the core is sectioned into 2.5-cm intervals and prepared for dry combustion analysis. Comparison between the LIBS data and the dry combustion data is reasonably consistent, as shown in the regression of LIBS data against the dry combustion data (Figs. 3 and 4).

RESULTS

An example of raw LIBS data with depth is shown in Fig. 2. These data indicate the variation of the C signal with depth, a feature that is not collected with conventional methods. This variation can be used to estimate the C concentration in the core as well as the uncertainty of the concentration estimate. In the raw form, the data are not comparable with conventional analysis data. Therefore, we integrated the

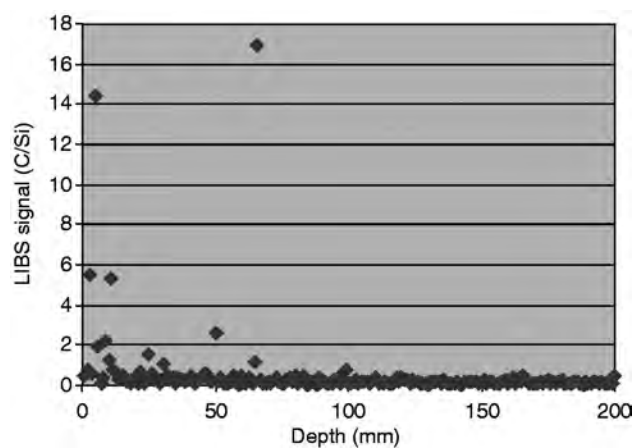


Fig. 2 Raw data from LIBS data at 1 mm step size in an intact soil core. Since Si is ubiquitous in soils and present at much greater total concentrations, small variations in Si concentration did not appreciably affect the C/Si ratio. Thus, differences in C/Si values are interpreted as changes in C concentration.

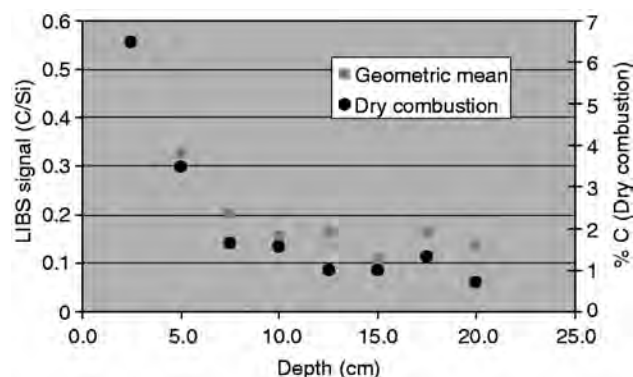


Fig. 3 Integrated LIBS data and dry combustion data from the same core. LIBS values were estimates computed using the measurement made at each 1 mm along the core.

LIBS data into 2.5-cm intervals to compute average C concentrations within a given interval. Dry combustion analyses obtained from the same 2.5-cm intervals gave the data by which we could compare the two methods.

The data in Fig. 3 show a reasonable agreement between the geometric mean of the LIBS data from each 2.5-cm interval and the dry combustion data (Fig. 3). The correlation found regressing the two data sets was good for the core used in this example ($R^2 = 0.982$; Fig. 4) and ranged from 0.7 to 0.98 for 12 other cores (data not shown).

SUMMARY

These initial data show promising application of the high-resolution method for the analysis of intact soil cores with LIBS. The LIBS data are generally comparable to the data from conventional methods, but acquisition of the LIBS data is considerably more rapid and at higher resolution. Extraction and collection of the spectra from each of the cores take about 30 minutes with the LIBS instrument, whereas an additional time period of 7–10 days is required to complete the analysis via dry combustion. The cost

savings, in terms of man-hours, of the LIBS analysis compared favorably to dry combustion. Sample extraction and preparation require only minutes using LIBS, and analytical results are usable within an hour of sampling, whereas days to weeks of laboratory preparation and handling are required for dry combustion analysis.

High-resolution analysis of intact cores can also provide routine estimates of measurement uncertainty, whereas uncertainty estimates from dry combustion are difficult because of the need for multiple subsamples. The variability of C concentrations from one sample step to the next is clear (Fig. 2). Part of this variation is due to the heterogeneity of the formation of soil C in the soil, and part is due to C from included material (e.g., roots) or dilution of C with material that does not contain C (e.g., quartz grains). The amount of data collected from a core using LIBS is basically a C measurement each 1 mm; when composited into 2.5-cm intervals, the LIBS data yield an uncertainty estimate based on a high density of measurements. Dry combustion, on the other hand, requires dozens of measurements from the same interval of the core to approximate the estimate from the LIBS data.

The LIBS analyses are usually conducted on samples directly extracted from the soil with no additional preparation. The advantages of this method are in its speed and its apparent accuracy and precision. One disadvantage of the method is the cost of building a LIBS instrument or adapting a commercial instrument to this application. Efforts are underway to commercialize the LIBS instrument for high-resolution C analysis, and such a product was expected by 2007. A second disadvantage is the inclusion of all components of the soil core in the analysis. Thus, roots, rock fragments, and other material in the soil are part of the C analysis for the entire core. This feature explains some of the variability in the C signal as mentioned earlier.

The high-resolution method requires large-scale, systematic evaluation before it is accepted and used as a method for C analysis of soils. The promise shown in these results, however, suggests that such a study needs to be undertaken at an earliest opportunity.

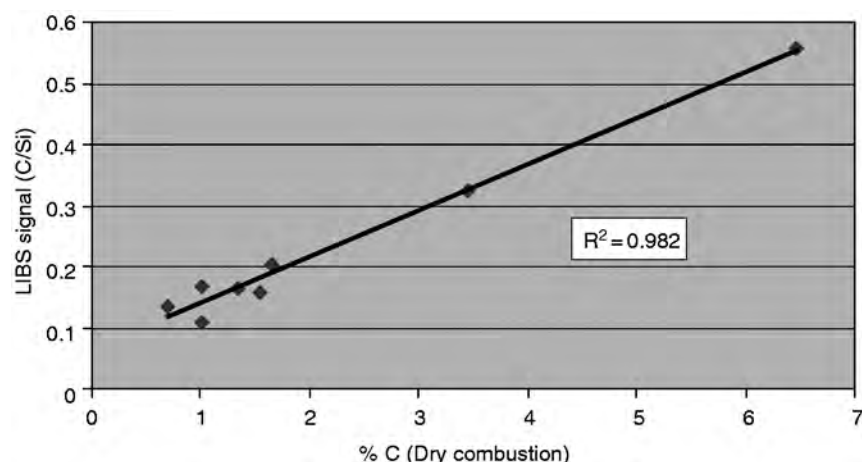


Fig. 4 Linear regression analysis of LIBS data with dry combustion data.

ACKNOWLEDGMENTS

The authors completed the reported work with funding from the U.S. Department of Energy National Energy Technology Laboratory and Office of Science. This is Los Alamos Report LA-UR-05-0480.

REFERENCES

1. Rossell, R.A.; Gasparoni, J.C.; Galantini, J.A. Soil organic matter evaluation. In *Assessment Methods for Soil Carbon*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2001; 311–322.
2. Wielopolski, L.; Orion, I.; Hendrey, G.; Roger, H. Soil carbon measurements using inelastic neutron scattering. *IEEE T. Nucl. Sci.* **2000**, *47*, 914–917.
3. Ebinger, M.H.; Lee Norfleet, M.; Breshears, D.D.; Cremers, D.A.; Ferris, M.J.; Unkefer, P.J.; Lamb, M.S.; Goddard, K.L.; Meyer, C.W. Extending the applicability of laser-induced breakdown spectroscopy for total soil carbon measurement. *Soil Sci. Soc. Am. J.* **2003**, *67*, 1616–1619.
4. McCarty, G.W.; Reeves, J.B., III. Development of rapid instrumental methods for measuring soil organic carbon. In *Assessment Methods for Soil Carbon*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2001; 371–380.
5. McCarty, G.W.; Reeves, J.B., III; Reeves, V.B.; Follett, R.F.; Kimble, J.M. Mid-infrared and near-infrared diffuse reflectance spectroscopy for soil carbon measurement. *Soil Sci. Soc. Am. J.* **2002**, *66*, 640–646.
6. Cremers, D.A.; Ebinger, M.H.; Breshears, D.D.; Unkefer, P.J.; Kammerdiener, S.A.; Ferris, M.J.; Catlett, K.M.; Brown, J.R. Measuring total soil carbon with laser-induced breakdown spectroscopy (LIBS). *J. Environ. Qual.* **2001**, *30*, 2202–2206.
7. U.S. Department of Energy (DOE). *Carbon Sequestration Technology Roadmap and Program Plan—2004*. National Energy Technology Laboratory, DOE Office of Fossil Energy. Available at www.netl.doe.gov (accessed September 1, 2004).
8. Rusak, D.A.; Castle, B.C.; Smith, B.W.; Winefordner, J.D. Fundamentals and applications of laser-induced breakdown spectroscopy. *Crit. Rev. Anal. Chem.* **1997**, *27*, 257–290.
9. Moenke-Blankenburg, L. *Laser Microanalysis*; Wiley: New York, 1989.
10. Radziemski, L.J.; Cremers, D.A. Spectrochemical analysis using laser plasma excitation. In *Applications of Laser-Induced Plasmas*; Radziemski, L.J., Cremers, D.A., Eds.; Marcel Dekker: New York, 1989; 295–325.
11. Cremers, D.A.; Ferris, M.J.; Davies, M. Transportable laser-induced breakdown spectroscopy (LIBS) instrument for field-based soil analysis. *Proc. Soc. Photo-Opt. Instrum. Eng.* **1996**, *2835*, 190–200.
12. Alkemade, C.T.J.; Snelleman, W.; Boutilier, G.D.; Pollard, B.D.; Winefordner, J.D.; Chester, T.L.; Omenetto, N. A review and tutorial discussion of noise and signal-to-noise ratios in analytical spectrometry—I. Fundamental principles of signal-to-noise ratios. *Spectrochim. Acta B* **1978**, *33*, 383–399.
13. Boutilier, G.D.; Pollard, B.D.; Winefordner, J.D.; Chester, T.L.; Omenetto, N. A review and tutorial discussion of noise and signal-to-noise ratios in analytical spectrometry—II. Fundamental principles of signal-to-noise ratios. *Spectrochim. Acta B* **1978**, *33*, 401–416.

Carbon: Physical Protection

Jennifer A.J. Dungait

Rothamsted Research, North Wyke, Okehampton, U.K.

David W. Hopkins

Royal Agricultural University, Cirencester, U.K.

Abstract

The protection of soil organic carbon (SOC) from mineralization by physical factors arises because of the multiple interactions of SOC with the other components of the soil matrix and the physical environment. It has been investigated by considering the properties of SOC in different physical fractions (including macroaggregates, microaggregates, and organic-mineral complexes) of soils, by attempts to visualize the distribution of SOC in soils and relating this to the factors controlling the presence and activity of decomposer organisms, and as components in simulation models. The main mechanisms whereby SOC is physically protected are: 1) encapsulation within aggregates where it is isolated from decomposer organisms; 2) stabilization by sorption of SOC at reactive colloid (e.g., clay or amorphous iron or aluminum) surfaces; and 3) where the environmental conditions, such as oxygen availability, restrict decomposition, or because of the extremes of physical conditions in the soils as a whole, such as extremes of temperature or very high water/very low oxygen content. Physical protection is not, however, a permanent state because the protection mechanisms tend to retard rather than completely halt decomposition, and because aggregate formation and disruption is a dynamic process, sorption can be followed by desorption, and environmental conditions change. Nonetheless, the emerging view is that accessibility of SOC to decomposer organisms is the dominant factor governing recalcitrance of SOC rather than inherent resistance to decay.

INTRODUCTION

Soil organic carbon (SOC) is the dead and decaying remains of plant, animal, and microbial organisms and their wastes. Heterotrophic soil organisms can utilize SOC directly as a source of organic carbon for biosynthesis and oxidize it to release energy, a process variously referred to as carbon mineralization, heterotrophic respiration, or just soil respiration. Carbon mineralization gives rise to a variety of intermediate products with different mean residence times in the soil, and, under aerobic conditions, ultimately carbon dioxide (CO₂) which diffuses into the atmosphere. In the context of this entry, physical protection refers to any set of conditions which limits the rate at which heterotrophic organisms can utilize SOC and the potential loss of carbon as CO₂ from the soil carbon pool. The tendency for soil to protect physically SOC from mineralization is affected by the interactions between the SOC and the soil mineral fraction and the spatial arrangements of the soil mineral (inorganic) components around the SOC (Fig. 1). Soils with large clay contents, and therefore a substantial surface area on which molecules can be absorbed, tend to have a larger SOC content than those with a lower clay content, and soils in which stable aggregates form (which may be related to clay content also) will tend to hold a greater SOC content relative to soils with less stable

structure. In both cases, the physical protection, i.e., the protection from biological attack leading to carbon mineralization, is related to the spatial arrangement and interactions of the SOC and the soil mineral components.^[1] Land use by humans affects the rates of carbon mineralization in soils, usually causing acceleration of SOC decomposition, due to disruption of the mechanisms of physical protection. Unprotected SOC that is dissolved or suspended in water may also be lost from the soil by leaching or in surface runoff from top soils and this too may result from physical protection being removed as a result of the action of water in either dispersing soil aggregates or components of the SOC dissolving in it.

PHYSICALLY DEFINED FRACTIONS OF SOC AND RELATIONSHIP TO PHYSICAL PROTECTION

The practice of considering SOC on the basis of physically defined fractions is well established. In these approaches, soils are usually dispersed to stable microaggregates and separated into fractions on the basis of size or density.^[2,3] In reality, it is rarely the “clean” organic matter fractions that are separated into the different fractions, and the fractions usually contain varying amounts of inorganic soil material. Nonetheless, physical fractionation provides a useful way

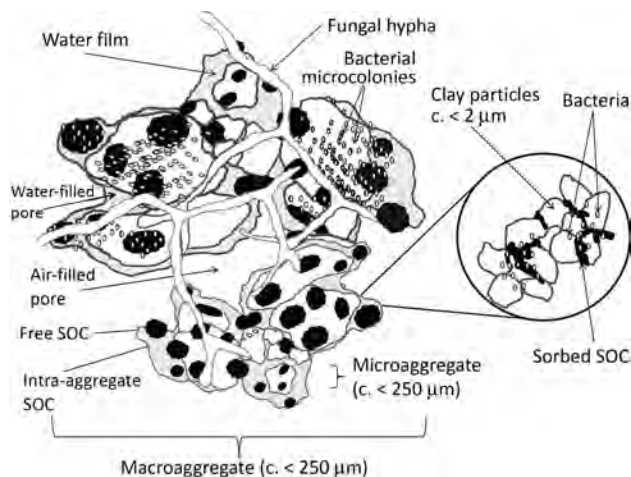


Fig. 1 Spatial arrangement of soil organic and inorganic components and soil microorganisms and water in soil macroaggregates and microaggregates and, at the smallest scale, SOC sorbed on clay (colloid) surfaces.

to understand SOC decomposition processes and to detect the effects of different soil management on SOC. Separation using procedures that discriminate on the basis of size, solubility, or density, and sometimes several in combination, follows the rationale that the fragment size of SOC decreases with time and the density of the fraction in which SOC is recovered increases with time because of increasing association with relatively inorganic soil components. The predominantly insoluble, particulate components of fresh plant litter are represented by particulate organic matter (POM) in some schemes and in the sand-sized [i.e., 50 μm (or 63 μm) to 2000 μm equivalent spherical diameter] fraction in other schemes. It is essential to realize that the SOC recovered in the sand-sized fraction does not necessarily have an association with actual sand. POM is often characterized by having relatively low density (close to 1 g cm^{-3}) and it is therefore frequently recovered as a low density fraction, or the so-called light fraction (LF). Although defined by different operational procedures, POM, sand-sized SOC, and LF SOC share several similar characteristics—they are usually relatively large-sized materials, relatively labile, and have little, if any, intimately associated inorganic material. SOC is subject to comminution (often by the feeding processes of soil organisms) and fragmentation as a result of physical disintegration, biological degradation, and assimilation into microbial biomass and extracellular exudates, so that the overall fragment size declines with time. Thus, older SOC is recovered in progressively smaller (silt- and then clay-sized) fractions. Unless very harsh dispersion procedures are used, these fractions are likely to be aggregations of sand and/or silt particles with SOC rather than isolated silt or clay particles. As a result of residence in the soil and exposure over time to reactive inorganic surfaces, SOC develops progressively more intimate associations with inorganic

materials, particularly clay particles and amorphous colloids. As the inorganic components tend to have greater density than SOC, older SOC tends to be recovered in fractions with greater density and smaller size. Simultaneous with the passage of carbon in plant litter from large-sized and low density fractions to smaller-sized and higher density fractions, aggregations of soil particles are formed and breaking down continuously. This occurs because of the influence, for example, of feeding by soil fauna (e.g., earthworms, collembola, or mites), freezing and thawing, wetting, and drying and management, such as tillage, and is superimposed on the size and density journey of SOC. Thus, fresh plant residues or recent exudates which may be valuable microbial substrates can become encapsulated in soil aggregates where they may be isolated from microorganisms or located in locations where the environmental conditions retard decomposition.^[4] This can lead to old but potentially labile SOC being recovered in relatively dense fractions; Dungait et al.^[5] reviewed the recalcitrance of plant and microbial derived SOC and concluded that accessibility of SOC to decomposer organisms was the dominant factor governing recalcitrance of SOC rather than the inherent properties of the SOC at the molecular level.

MECHANISMS OF PHYSICAL PROTECTION

Physical protection of SOC occurs because either the biological decomposition agents (soil organisms or their extracellular enzymes) are spatially isolated from the SOC or because local physical conditions prevent them from using the SOC in their vicinity.

Spatial Isolation

Spatial separation of SOC from heterotrophic organisms operates at different scales, which correspond to the size and motility of the organisms and the ability of the organisms to disrupt the mechanisms of protection and access the SOC.

At the molecular scale, the interactions between SOC and soil mineral colloids are complex. The behavior of both SOC and soil inorganic colloids (clay minerals and amorphous iron and aluminum colloids) is dominated by the physical and chemical properties distributed over their relatively vast surface areas. Most mineral and organic colloids in soil carry a net negative surface charge and thus, at close proximity, are repelled from each other. Opposing this repulsion, van der Waal's forces of attraction operate over slightly longer ranges. Therefore, the proximity of SOC to a mineral colloid surface is determined by the balance between the two opposing forces. Bacteria also behave as organic colloids and this influences their ability to access SOC at this most intimate scale. At the nanoscale, the interlamellar spaces of certain clays have hygroscopic properties which draw in and trap dissolved SOC. The SOC

is physically protected from decomposition if the dimensions of these spaces are too small for soil microorganisms or extracellular enzymes to access the SOC. At the micro- and submillimeter scale, adsorption of SOC onto mineral surfaces of clay and silt particles because of a range of properties including adhesion, cation bridging, and hydrogen bonding leads to persistent complexes between SOC and mineral surfaces.

Clay and silt particles, upon which SOC is sorbed, may further combine to form microaggregates, encapsulating or occluding SOC and thereby increasing its physical protection. Microaggregates are comprised of inorganic and organic components bound together under the influence, at least in part, of biological phenomena including, on the one hand, the adhesive properties of the SOC, microbial extracellular polymeric substances and rhizoxudates, and mucilage from soil invertebrates, and on the other, the enmeshing action of fungi and fine roots. The SOC encapsulated in microaggregates is rendered relatively stable because of the spatial isolation from external soil microorganisms and internal physical conditions that limit microbial activity within aggregates. However, soil microaggregates are transient because the biological phenomena that contribute to stability are dependent on the persistence of the metabolic activity of organisms whose contributions subside as their substrate (i.e., SOC) is exhausted. Larger-scale aggregation, in a continuum of size ranging from microaggregates to peds, is progressively controlled by physical processes that impose an upper size limit, i.e., water infiltration, erosion, compaction, and tillage. The size of soil aggregates is usually inversely proportional to the amount of energy needed to disrupt them; thus, microaggregates afford the most protection to encapsulated SOC.

The activity of earthworms is of particular importance due to the huge mass of casts produced, e.g., 20 to 30 kg m⁻² yr⁻¹ in temperate pastures. Cast production initially enhances carbon mineralization and nutrient release because of the stimulation of the microorganisms ingested with the organic residues by the introduction of moisture and mucus and physical fragmentation by grinding with ingested mineral particles during passage through the gut. These effects are, however, transient with progressive stabilization of organic carbon in aging casts as labile organic components are mineralized or become physically unavailable through interactions with coingested mineral particles, thereby providing a route for the stabilization of SOC.

Physical Constraints on SOC Decomposition

Even under circumstances where both SOC and heterotrophic soil organisms are simultaneously colocated, physical conditions including temperature, water availability, and oxygen availability may restrict carbon mineralization, because biological processes can only operate within a range of environmental tolerances. Where conditions in the soil exceed these thresholds, heterotrophic activity may be

repressed, but if inputs from photosynthesis can proceed, SOC may accumulate (see below).

Biological processes, including carbon mineralization in soils, respond to changes in temperature. Assuming that there is no water limitation, the optimum temperature range for metabolic activity is around typically 25–30°C. The magnitude of change to the rate of SOC decomposition as temperature increases by 10 K is described by the Q10 parameter and usually has a value of around 2, except at the limits of biological activity. The upper limit for biological activity in soils is around 50°C, because most proteins that control metabolic function (including the enzymes that catalyze SOC decomposition) denature above this limit and Q10 falls rapidly as the upper temperature limit is approached. At low temperatures, biological activity increases rapidly as the temperature increases away from 0°C. Although some biological processes can be detected even at subzero temperatures, their contribution to net carbon mineralization is negligible. Moreover, under natural circumstances, it is difficult to separate the effect of extreme temperatures from water availability in soils which is limiting at both high and low temperatures. Where photosynthesis proceeds but carbon mineralization is retarded by temperature, SOC may accumulate. Thus, the large stocks of SOC in the Cryosols of boreal and tundra regions at high latitudes are particularly vulnerable to warming, and in the boreal regions which contain large SOC stocks protected from carbon mineralization by low temperatures, the effect of warming will be particularly marked.

Under conditions of non-extreme temperature, the availability of water and oxygen is closely linked. In general, the amount of water in the soil is inversely related to the amount of oxygen in the soil. Most soil organisms are obligate aerobes, requiring oxygen to act as the terminal electron acceptor during respiration. When soils are waterlogged, the rate of diffusion of oxygen through water to biological organisms is too slow to allow aerobic respiration to proceed. However, facultative and obligate anaerobes become active under these conditions allowing reduced rates of carbon mineralization to proceed by fermentation, which ultimately leads to the production of gaseous methane, and other incompletely oxidized terminal products in solution, such as ethanol.

At the scale of the individual aggregate, water and oxygen are not uniformly distributed. Water tends to occupy the pores with the smallest neck diameters under non-saturated conditions in soils, drawn and held strongly in place by capillarity, restricting the movement of oxygen (Fig. 2). Consequently, locations at the interior of aggregates may be persistently anoxic, regardless of the moisture content of the wider soil environment. Anoxic conditions are likely to arise rapidly in newly formed aggregates where fresh organic carbon is newly encapsulated. These include the casts of earthworms which are important macroinvertebrate “soil engineers” in many soils.

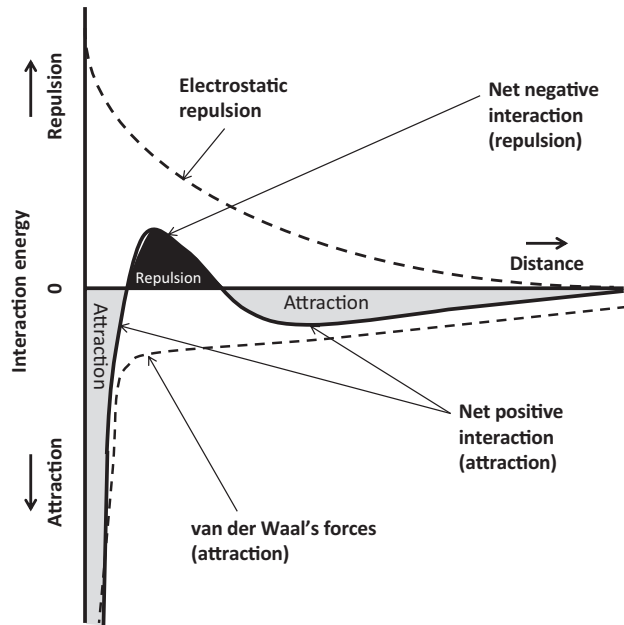


Fig. 2 The DLVO^[6,7] model showing the decay with distance of electrostatic repulsion and van der Waal's forces of attraction and the net interaction energy. At very close proximity (typically at the submicrometer scale), the model predicts net attraction between surfaces that display colloidal properties (e.g., clay particles, SOC, microorganisms, and extracellular enzymes), while at slightly greater distances, there is net repulsion, before at greater distances the attractive van der Waal's forces completely overwhelm electrostatic repulsion. The distances over which electrostatic repulsion operates is influenced by electrolyte concentrations because positively charged counterions neutralize some of the negative surface charges. The two regions where net attractive forces apply can form a diffuse double layer at colloid surfaces. Here, microorganisms and SOC may be simultaneously present and may interact leading potentially to carbon mineralization, but toxic compounds may also be held in the same vicinity which may suppress carbon mineralization.

CONCLUSION

Physical protection of SOC is not a single process, but a set of processes operating at different scales that collectively lead to the rate of decomposition of SOC being restricted compared to that which would occur in their absence. The key components of physical protection are encapsulation of the SOC at sites where it is isolated from microorganisms or where the environmental conditions retard decomposition, or stabilization on colloid surfaces retarding microbial attack, or because of the overall restrictions on decomposition activity imposed by the wider environmental conditions such as temperature or oxygen availability. Many soil management practices, such as tillage activities and irrigation, disrupt the mechanisms of physical protection of SOC, which may be observed as the loss of aggregate structure in

poorly managed cultivated soils. The breakage of soil aggregates exposes the encapsulated SOC to the agents of carbon mineralization, and the removal of organic carbon and nutrients starves the organisms in the soil that promote aggregate formation. In the absence of structure, the soil may slake when wetted because the aggregates are less stable in water, producing a dense layer impermeable to water and air. Soil compaction by heavy machinery or livestock can crush soil aggregates and reduce the number of large pores for water and air infiltration, creating an anaerobic compaction layer in the soil in which the nutrient turnover that is driven by carbon mineralization is reduced. Thus, in addition to being of fundamental interest to advance the understanding of SOC dynamics, appreciation of the sensitivity of SOC to changes in physical protection mechanisms is important for soil management.

ACKNOWLEDGMENTS

This work is part of the Institute Strategic Research Programmes on “Designing Sustainable Systems” and “Cropping Carbon” supported by the UK Biotechnology and Biological Sciences at Rothamsted Research, which is acknowledged, as is the support from The Royal Agricultural University.

REFERENCES

1. Baldock, J.A.; Skjemstad, J.O. Role of the soil matrix and minerals in protecting natural organic materials against biological attack. *Org. Geochem.* **2000**, *31*, 697–710.
2. Gregorich, E.G.; Kachanoski, R.G.; Voroney, R.P. Carbon mineralization in soil size fractions after various amounts of aggregate disruption. *J. Soil Sci.* **1989**, *40*, 649–659.
3. Six, J.; Elliott, E.T.; Paustian, K. Soil macroaggregate turnover and microaggregate formation: A mechanism for C sequestration under no-tillage agriculture. *Soil Biol. Biochem.* **2000**, *32*, 2099–2103.
4. Golchin, A.; Oades, J.M.; Skjemstad, J.O.; Clarke, P. Study of free and occluded particulate organic matter in soils by solid-state ¹³C CP/MAS NMR spectroscopy and scanning electron microscopy. *Aust. J. Soil Res.* **1994**, *32*, 1043–1068.
5. Dungait, J.A.J.; Hopkins, D.W.; Gregory, A.S.; Whitmore, A.P. Soil organic matter turnover is governed by accessibility not recalcitrance. *Glob. Change Biol.* **2012**, *18*, 1781–1796.
6. Derjaguin, B.; Landau, L. Theory of the stability of strongly charged lyophobic sols and of the adhesion of strongly charged particles in solutions of electrolytes. *Acta Physicochim. URS* **1941**, *14*, 633.
7. Verwey, E.J.W.; Overbeek, J.T.G. *Theory of the Stability of Lyophobic Colloids*; Elsevier: Amsterdam, 1948.

Carbonates

Douglas W. Ming

*Johnson Space Center, National Aeronautics and Space Administration (NASA),
Houston, Texas, U.S.A.*

Abstract

Carbonate dissolution and precipitation are among the most important chemical reactions in soils. Carbonates precipitate as coatings on soil particles and in soil pores that may result in the cementation of particles. Other important reactions involving carbonate minerals are adsorption and/or precipitation of some anions and cations on their surfaces which have an impact on the mobility of essential plant growth elements (e.g., ferrous ion, ferric ion, zinc ion, hydrogen phosphate, and dihydrogen phosphate) and elements of environmental concern [e.g., cadmium ion and plumbous ion (Pb^{2+})]. Carbonates (e.g., calcite and dolomite) are also added to raise the pH of acid soils (i.e., carbonates as liming agents) or to immobilize heavy metals (e.g., Pb^{2+}) in contaminated soils through the precipitation of relatively insoluble carbonates (e.g., lead carbonate). These important processes have led to many studies of carbonate reactions and occurrences in soil environments.

INTRODUCTION

There are approximately 150 carbonate minerals that occur in nature; however, most of these carbonates are relatively rare. The most common rock-forming carbonates are calcite (CaCO_3) and dolomite ($\text{CaMg}(\text{CO}_3)_2$), which account for over 90% of natural carbonates.^[1] CaCO_3 , $\text{CaMg}(\text{CO}_3)_2$, siderite (FeCO_3), and ankerite ($\text{Ca}(\text{Mg,Fe})(\text{CO}_3)_2$) are the most common carbonates that occur in sedimentary environments.^[2] Carbonates are very common in soils, and they either form during soil formation (i.e., paedogenic) or are inherited from parent materials (i.e., detrital or lithogenic). The most common carbonates that occur in soils are CaCO_3 , magnesium (Mg)-calcite ($\text{Ca}_x\text{Mg}_{1-x}(\text{CO}_3)$), and $\text{CaMg}(\text{CO}_3)_2$. Aragonite (CaCO_3) occasionally occurs in soils, and FeCO_3 has been reported to occur in some soils.^[3]

STRUCTURE AND COMPOSITION

The carbonate ion (CO_3^{2-}) is the basic structural unit for all carbonate minerals. The large planar CO_3 group occurs as a rigid structure with the carbon (C) atom occupying the center, which is surrounded by three oxygen atoms arranged in an equilateral triangle. The linkage of the carbonate group to other cations has minimal effect on the ideal C–O bond angle of 120° in the CO_3 group.^[4] The non-hydrous carbonates can be grouped according to their structure into the calcite group, dolomite group, or the aragonite group (see Table 1). Hydrated carbonates have entirely different crystal structures, and most crystallize in the monoclinic crystal system (Table 1).

Calcite group minerals crystallize in the hexagonal crystal system. The structure of CaCO_3 , which is the most common calcite-group mineral, is a distorted sodium chloride type structure where calcium (Ca) alternates with CO_3 . This distorted structure places Ca atoms in a face-centered rhombohedral unit cell (Fig. 1), which is the morphologic or cleavage cell. The smaller true unit cell is an elongated rhombohedral unit cell, which is easily compared to the hexagonal unit cell (Fig. 1). Magnesite, FeCO_3 , rhodochrosite (MnCO_3), and smithsonite (ZnCO_3) are isostructural with CaCO_3 (Table 1).

Aragonite-group minerals crystallize in the orthorhombic crystal system. In this group, large divalent cations (e.g., ionic radii > 0.1 nm) do not permit a stable sixfold coordination and, therefore, result in a ninefold coordination configuration. Aragonite, which is a higher temperature and pressure CaCO_3 polymorph (CaCO_3 is the stable polymorph at room temperature), has CO_3 groups in two structural planes rather than the single structural plane of CaCO_3 . Witherite (BaCO_3), strontianite (SrCO_3), and cerussite (PbCO_3) are isostructural with aragonite (Table 1).

The third major group of carbonate minerals, the dolomite group, crystallizes in the hexagonal system. Their structure is similar to calcite-group minerals, except that different cations (e.g., Ca–Mg, Ca–Fe, and Ca–Mn) alternate along the c-axis. Because of this arrangement of alternating cations, the symmetry of dolomite-group minerals is reduced compared to the calcite-group minerals. $\text{CaMg}(\text{CO}_3)_2$ has Ca and Mg alternating along the c-axis. $\text{Ca}(\text{Mg,Fe})(\text{CO}_3)_2$ and kutnahorite ($\text{CaMn}(\text{CO}_3)_2$) are isostructural with $\text{CaMg}(\text{CO}_3)_2$ (Table 1).

Various cations may substitute into the carbonate structure. For example, Mg^{2+} may substitute for Ca^{2+} in the

Table 1 Representative unit cell formulae and selected properties of common carbonates.

Carbonate group	Ideal unit cell formula	Crystal system	Specific gravity
<i>Calcite group</i>			
Calcite	CaCO ₃	Hexagonal (rhombohedral)	2.71
Magnesite	MgCO ₃	Hexagonal	3.00
Siderite	FeCO ₃	Hexagonal	3.97
Rhodochrosite	MnCO ₃	Hexagonal	3.70
Smithsonite	ZnCO ₃	Hexagonal	4.43
Otavite	CdCO ₃	Hexagonal	4.96
Gaspéite	NiCO ₃	Hexagonal	4.39
<i>Aragonite group</i>			
Aragonite	CaCO ₃	Orthorhombic	2.95
Witherite	BaCO ₃	Orthorhombic	4.3
Strontianite	SrCO ₃	Orthorhombic	3.7
Cerussite	PbCO ₃	Orthorhombic	6.55
<i>Dolomite group</i>			
Dolomite	CaMg(CO ₃) ₂	Hexagonal	2.85
Ankerite	CaFe(CO ₃) ₂	Hexagonal	3.01
Kutnahorite	CaMn(CO ₃) ₂	Hexagonal	3.12
Minrecordite	CaZn(CO ₃) ₂	Hexagonal	3.45
<i>Other carbonates</i>			
Azurite	Cu ₃ (CO ₃) ₂ (OH) ₂	Monoclinic	3.77–3.89
Malachite	Cu ₂ CO ₃ (OH) ₂	Monoclinic	3.7–4.1
Huntite	Mg ₃ Ca(CO ₃) ₄	Hexagonal	2.70
Landsfordite	MgCO ₃ · 5H ₂ O	Monoclinic	1.69
Artinite	Mg ₂ (OH) ₂ CO ₃ · 3H ₂ O	Monoclinic	2.04
Nesquehonite	MgCO ₃ · 3H ₂ O	Monoclinic	1.84
Hydromagnesite	Mg ₅ (CO ₃) ₄ (OH) ₂ · 4H ₂ O	Monoclinic	2.25
Nyerereite	Na ₂ Ca(CO ₃) ₂	Orthorhombic	2.54
Gaylussite	Na ₂ Ca(CO ₃) ₂ · 5H ₂ O	Monoclinic	1.99
Nahcolite	NaHCO ₃	Monoclinic	2.19
Thermonatrite	Na ₂ CO ₃ · H ₂ O	Orthorhombic	2.26
Soda (Natron)	Na ₂ CO ₃ · 10H ₂ O	Monoclinic	1.46
Trona	NaCO ₃ · NaHCO ₃ · 2H ₂ O	Monoclinic	2.13

Source: Adapted from Chang, Howie, et al.^[5] and the JCPDS—International Centre for Diffraction Data.

CaCO₃ structure. These Mg-substituted calcites are called magnesian calcites or Mg-calcites. Mg-calcites containing up to 20 mol% MgCO₃ are found in skeletal materials of marine organisms, and even higher concentrations of Mg (e.g., 45 mol% MgCO₃) in the CaCO₃ structure have been synthesized at high pressure and temperature. Most soil Mg-calcites, however, contain

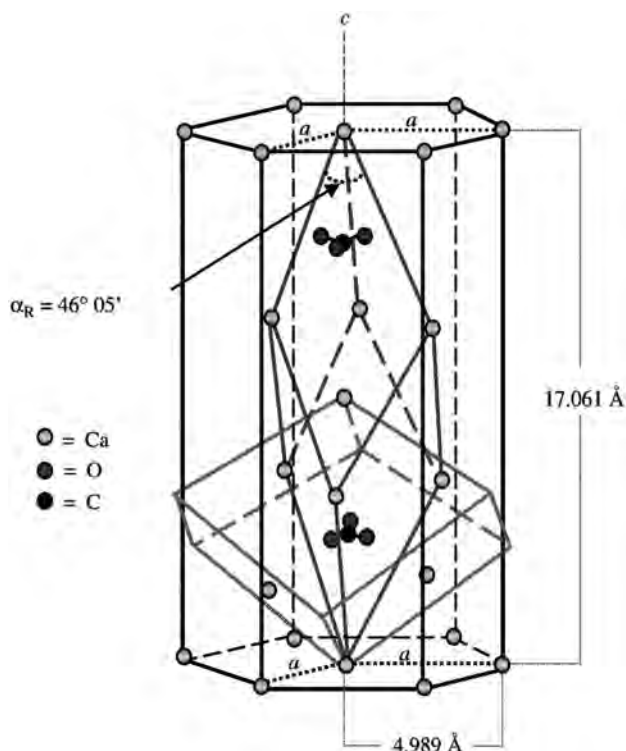


Fig. 1 Structure of CaCO₃. Unit cell parameters for a and c are shown for the hexagonal unit cell. The elongated rhombohedral, true unit cell contains two CaCO₃; the two CO₃ groups are shown ($\alpha_R = \alpha_{\text{rhombohedral-cell}}$). The morphological or cleavage cell contains four CaCO₃ (face-centered rhombohedral cell). Note that ionic radii for Ca, C, and O are not drawn to scale.

Source: Adapted from Hurlbut Jr. & Klein Jr.^[6]

less than 10 mol% MgCO₃.^[3] Due to the smaller ionic radius of Mg²⁺ as compared to Ca²⁺, the a and c unit cell axes decrease in length, as the substitution of Mg²⁺ increases in the structure of CaCO₃.^[7] Detailed discussion on the substitution of cations into carbonate structures can be found in the studies by Chang et al.^[5] and Reeder.^[8]

OCCURRENCE IN SOILS

Carbonates are common in many soils of the world, particularly in soils of subhumid to arid regions. CaCO₃ is the most common carbonate in soils, but CaMg(CO₃)₂, aragonite, and FeCO₃ are also found in certain soils. Carbonates may be either inherited into soil or form directly via precipitation processes. The most common sources and mechanisms for carbonate occurrences in soils are as follows: 1) inheritance from parent material; 2) precipitation following dissolution of soluble Ca-bearing minerals; 3) precipitation by reaction of CO₂-charged water with Ca²⁺ released by decomposition of plant residues; 4) deposition of windblown carbonates at the soil surface and subsequent translocation within the soil by solution and precipitation; 5) precipitation by interaction of Ca²⁺ in

rainwater entering the soil and combining with HCO_3^- from CO_2 -charged water; 6) precipitation from CO_2 -charged natural surface or irrigation water containing Ca^{2+} ; and 7) precipitation in soils from groundwater that has moved through carbonate-containing soils or sediments.^[3]

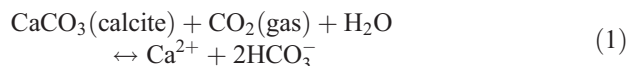
CaCO_3 and Mg-calcite are the two carbonates that commonly precipitate in soils. Although all of the carbonates commonly found in soils may be inherited, aragonite and $\text{CaMg}(\text{CO}_3)_2$ are almost always inherited. Paedogenic $\text{CaMg}(\text{CO}_3)_2$ has been reported to occur in some saline, alkaline soils, but generally, its formation in soil is very rare.^[9] Mg-calcite with high MgCO_3 mole% (e.g., > 10 mol%) is typically inherited in soils from marine sediments (i.e., from biogenic Mg-calcite). FeCO_3 is relatively unstable in soil environments and is not known to form directly in soils. Mg carbonates are very rare in soils, but hydrated Mg carbonates (e.g., hydromagnesite and nesquehonite) may form at the surface during the evaporation of solutions high in Mg^{2+} and HCO_3^- .^[10] Mechanisms by which carbonates precipitate in soils are provided in the following section.

The distribution of carbonates in a soil profile depends on the climate, soil chemical properties, and degree of leaching through the profile. In many arid soils, carbonates are common secondary products in the top horizons (e.g., A and B horizons). As a general rule, carbonates tend to be leached or removed from the upper horizons with increasing leaching or rainfall. Carbonates leached from upper horizons may precipitate as secondary carbonates in the lower horizons where they may form carbonate-cemented layers (e.g., petrocalcic horizons).

Sodium carbonates may occur at the soil's surface in arid regions where soil salinity is high. Highly soluble sodium carbonates precipitate at the surface during the periods of evaporation, and such conditions produce soil pH values of higher than 8.5.

FORMATION

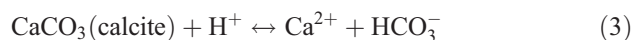
Carbonate dissolution or precipitation is dependent on several factors, such as the partial pressure of dissolved CO_2 in solution and temperature. A typical reaction for the dissolution of CaCO_3 may be expressed as a function of CO_2 as follows



The equilibrium constant (K^0) for this reaction at 25°C is as follows

$$K^0 = \frac{(\text{Ca}^{2+})(\text{HCO}_3^-)^2}{P_{\text{CO}_2}} = 10^{-5.90} \quad (2)$$

where P_{CO_2} is the partial pressure of CO_2 and brackets denote activities.^[11] The reaction may also be expressed as a function of H^+ as follows



with the respective equilibrium constant at 25°C being

$$K^0 = \frac{(\text{Ca}^{2+})(\text{HCO}_3^-)}{(\text{H}^+)} = 10^{+1.92} \quad (4)$$

Eqs. 2 and 4 indicate that the dissolution of CaCO_3 (and other carbonates) is favored with increasing CO_2 pressure and decreasing pH. Conversely, the degassing or removal of CO_2 from solution or an increase in pH will favor CaCO_3 participation. The equilibrium pH of a solution in contact with CaCO_3 at atmospheric CO_2 pressure ($10^{-3.5}$ bar P_{CO_2}) is 8.3. Detailed discussion on carbonate solution chemistry can be found in the studies by Lindsay^[11] and Langmuir.^[12]

PROPERTIES

Calcium carbonate has an influence on several soil properties, e.g., soil pH, adsorption-desorption, and soil cementation. Most soils that contain calcium carbonate will have a pH in the range of 7.1–8.5, in which the carbonate acts as a pH buffer. The surfaces of CaCO_3 are reactive, and various ions may adsorb or interact at the crystal's surface. For example, Mg^{2+} , zinc ion (Zn^{2+}), Cu^{2+} , ferrous ion (Fe^{2+}), and Al^{3+} may replace Ca^{2+} on exposed surface lattice sites. The reactive surfaces of carbonates may adsorb essential plant ions such as hydrogen phosphate, dihydrogen phosphate, Fe^{2+} , ferric ion, Zn^{2+} , and Mn^{2+} and adversely affect their availability for plant uptake. Iron deficiency chlorosis in plants has been attributed to the interaction of Fe and HCO_3^- in calcareous soils.^[13,14] Other factors, such as quantity, mineralogy, and crystallinity of iron oxide phases in the soil, may also contribute to plant Fe chlorosis in calcareous soils.^[15]

The precipitation of carbonates on soil particles and in soil pores may form layers (i.e., calcic horizons) that impede the movement of water.^[16] Some layers may become indurated or cemented (i.e., petrocalcic horizons) and force water to move laterally in the soil.

Soil carbonates play an important role in the global C cycle although their role in the greenhouse effect is not well understood.^[17] Soil inorganic C (i.e., lithogenic and paedogenic carbonates) accounts for about one-third of the total C in soils; the remaining two-thirds of soil C is comprised of organic C. Soil organic C is the primary C pool in humid regions, whereas carbonates are the predominant C pool in arid and semiarid regions. The role of paedogenic carbonates in atmospheric C sequestration is not clear; however, the flux of paedogenic carbonates with the atmosphere has been estimated as $0.023 \text{ Pg C yr}^{-1}$, with a turnover time in the range of 30,000–90,000 years.^[18]

Carbonates such as CaCO_3 have been suggested for use in the remediation of plumbous (Pb)-contaminated soils where plumbous ion reacts with HCO_3^- to form insoluble Pb carbonates. Other soil contaminants (e.g., barium ion

and cadmium ion) may also be immobilized through carbonate precipitation with the addition of more soluble carbonate amendments.^[19]

In summary, carbonates are of great importance in soils because they have considerable influence on the physical and chemical properties of soils, they have an important role in the global C cycle, and, occasionally, carbonates are the predominant mineral phases that occur in soils of arid and semiarid regions.

REFERENCES

1. Reeder, R.J. Crystal chemistry of the rhombohedral carbonates. In *Carbonates: Mineralogy and Chemistry*; Reeder, R.J., Ed.; Mineralogical Society of America: Washington, 1983; Vol. 11, 1–47.
2. Lippmann, F. *Sedimentary Carbonate Minerals*; Springer: Secaucus, 1973; 228 pp.
3. Doner, H.E.; Lynn, W.C. Carbonate, halide, sulfate, and sulfide minerals. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; Soil Science Society of America: Madison, 1989; 279–330.
4. Effenberger, H.; Mereiter, K.; Zemann, J. Crystal structure refinements of magnesite, calcite, rhodochrosite, siderite, smithsonite, and dolomite with discussion of some aspects of the stereochemistry of calcite-type carbonates. *Z. Krist.* **1981**, *156*, 233–243.
5. Chang, L.L.Y.; Howie, R.A.; Zussman, J. *Rock-forming Minerals, Non-Silicates: Sulphates, Carbonates, Phosphates, Halides*, 2nd Ed.; Longman: Essex, 1996; Vol. 5B, 383 pp.
6. Hurlbut, C.S., Jr.; Klein, C. Jr. *Manual of Mineralogy (after James D. Dana)*, 19th Ed.; Wiley: New York, 1977; 532 pp.
7. Paquette, J.; Reeder, R.J. Single-crystal X-ray structure refinements of two biogenic magnesian calcite crystals. *Am. Mineral.* **1990**, *75*, 1151–1158.
8. Reeder, R.J., Ed. Carbonates: Mineralogy and chemistry. In *Reviews in Mineralogy*; Mineralogical Society of America: Washington, 1983; Vol. 11, 394 pp.
9. Kohut, C.; Muehlenbachs, K.; Dudas, M.J. Authigenic dolomite in a saline soil in Alberta, Canada. *Soil Sci. Soc. Am. J.* **1994**, *59*, 1499–1504.
10. Ming, D.W.; Franklin, W.T. Synthesis and characterization of lansfordite and nesquehonite. *Soil Sci. Soc. Am. J.* **1985**, *49*, 1303–1308.
11. Lindsay, W.L. *Chemical Equilibria in Soils*; Wiley: New York, 1979; 449 pp.
12. Langmuir, D. *Aqueous Environmental Geochemistry*; Prentice-Hall: Upper Saddle River, 1997; 600 pp.
13. Lindsay, W.L.; Thorne, D.W. Bicarbonate ion and oxygen level as related to chlorosis. *Soil Sci.* **1954**, *77*, 271–279.
14. Inskeep, W.P.; Bloom, P.R. Soil chemical factors associated with soybean chlorosis in calciaquolls of Western Minnesota. *Agron. J.* **1987**, *79*, 779–786.
15. Loeppert, R.H.; Hallmark, C.T. Indigenous soil properties influencing the availability of iron in calcareous soils. *Soil Sci. Soc. Am. J.* **1985**, *49*, 597–603.
16. Stuart, D.M.; Dixon, R.M. Water movement and caliche formation in layered arid and semi-arid soils. *Soil Sci. Soc. Am. Proc.* **1973**, *37*, 323–324.
17. Lal, R.; Kimble, J.M.; Eswaran, H.; Stewart, B.A. *Global Climate Change and Pedogenic Carbonates*; Lewis Publishers: Boca Raton, 2000; 305 pp.
18. Lal, R.; Kimble, J.M. Inorganic carbon and the global C cycle: Research and development priorities. In *Global Climate and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2000; 291–302.
19. Doner, H.E.; Grossl, P.R. Carbonates and evaporates. In *Soil Mineralogy with Environmental Applications*; Dixon, J.B., Schulze, D.G., Eds.; Soil Science Society America: Madison, 2002; 199–228.

Carbonates: Secondary and Pedogenic

Ahmet R. Mermut

Department of Soil Science, University of Saskatchewan, Saskatoon, Saskatchewan, Canada

A. Landi

Shahid Chamran University, Ahvaz, Iran

Abstract

Secondary or pedogenic carbonates (PCs) are those that are formed through the processes responsible for the soil development. The PC is an important part of the global carbon cycle. Global estimates are far from complete. The stable isotope technique is used to calculate the amount and rate of secondary carbonates.

INTRODUCTION

Secondary or pedogenic carbonates (PCs) are those that are formed through the processes responsible for the soil development. They are found in relatively dry soils and developed under natural good drainage and vegetation comprising grass and shrubs mixture.^[1] Total soil inorganic carbonate (SIC) has been estimated globally to be about 800 Pg in Aridisols and Entisols of arid regions, mainly in the caliche layers.^[2] Other estimates of total SIC in world soils include 720 Pg C,^[3] 695–748 Pg,^[4] and 750–947 Pg.^[5] The differences among estimates are a result, at least in part, of the difficulty of differentiating between primary carbonates (lithogenic carbonates) and carbonates of secondary origin or PCs.^[6] Reliable estimates on PC are to be calculated.

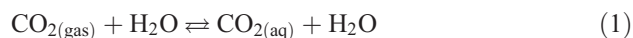
In soils, secondary carbonate species are calcite and magnesian calcites ($\text{Ca}_x\text{Mg}_{1-x}$).^[7] They occur as filaments, nodules, seams, or diffuse impregnations in non-gravelly materials; in gravelly materials, carbonates occur as pebble coatings, bridges, diffuse interstice fillings, and pendants secondary carbonates on the lower surface of pebbles, and cobbles form carbonate pendants with a complex, layered structure.^[8] Secondary carbonates, as a more stable pool than organic carbon (C), can sequester substantial amounts of C and are important constituents of many soils.

SECONDARY CARBONATE FORMATION

The partial pressure of carbon dioxide (CO_2) gas and pH of the soil solution play major roles in the precipitation of carbonates. The partial pressure of CO_2 in the atmosphere is about 3×10^{-5} MPa ($\sim 3 \times 10^{-3}$ atm). In the soil, because of

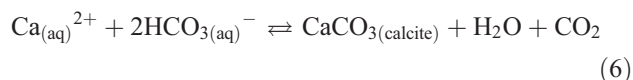
microbial activity and oxidation of organic matter, the partial pressure of CO_2 is much higher than in the atmosphere and it varies with time.

The reactivities between CO_2 and water are shown in Eqs. 1–5 as follows:



Secondary carbonate accumulations are commonly found in the pH range of 7.3–8.5. To form calcite in soils, a sufficient amount of calcium must be present in the soil solution.

Calcite forms through the following reaction:^[9]



As shown in Fig. 1,^[10] the ratio of $\text{H}_2\text{CO}_3/\text{HCO}_3^-$ is unity at about pH 6.4. This ratio increases tenfold for each unit decrease in pH. Below pH 6, almost all the dissolved carbonate species are in the form of H_2CO_3 . Equimolar of CO_3^{2-} and HCO_3^- occurs at pH 10.3, and between pH 6.4 and 10.3, the dominant carbonate ion species is HCO_3^- .^[11]

Calcite accumulation may develop relatively rapidly in gravelly soils. The process is rather slow in non-gravelly, sandy, or loamy soils. Continuous accumulation results in the formation of a calcic and/or petrocalcic horizons.

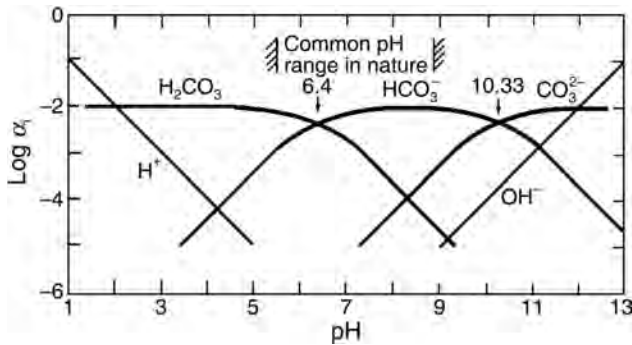


Fig. 1 Activities of different carbonate species as affected by pH.
Source: From Drever.^[10]

MORPHOLOGY OF SECONDARY CARBONATES

The morphology of secondary carbonate crystals is controlled mainly by the rate of crystallization and by Mg^{2+} and Na^+ contents of the precipitating waters.^[12] Most of the visible individual PC crystals in soils^[13] have diameters of 0.3–1.0 μm . In the soil matrix, soil particles inhibit the crystal growth, but in pores, the crystals tend to be larger in size. Fine crystal size is also attributed to the relatively high solubility of soil carbonates or the flush of high concentration of Ca^{2+} introduced to the soil system.

The surface morphologies of carbonate pendants and lithogenic carbonate pebbles have different features (Fig. 2). PCs occur in the form of microscopic calcite crystals, which are strongly clustered, with individual crystal sizes ranging from about 0.1 mm to crystal clusters of 3 mm diameter. Depending on soil matrix composition, microsparite, sparite, and micritic fabrics can form.^[14]

CALCULATION OF PC IN SOIL USING STABLE ISOTOPES

The C isotope geochemistry technique is used to distinguish between pedogenic and inherited carbonate.^[15]

Carbon isotope ratios are given as relative deviation in per mil from the isotope ratio in the Pee Dee Belemnite (PDB; specimen of *Belemnitell americana* found in the Pee Dee formation of South Carolina) standard.^[16] Because PDB has been exhausted, most of the researchers report $\delta^{13}C$ values relative to Vienna PDB as follows:

$$\delta^0/_{oo} = [(R_s - R_{std})/R_{std}] \times 1000 \quad (7)$$

where R_s is the isotope ratio for an element in a compound and R_{std} is the corresponding isotope ratio in a standard.

The difference between $\delta^{13}C$ values of parent material carbonate and PC can be used as an indicator of dissolution–precipitation process taking place in soil horizons. The amount of newly formed carbonates can be calculated from the C isotopic composition:^[16]

$$\% \text{Pedogenic carbonate} = \frac{\delta^{13}C(\text{soil}) - \delta^{13}C(\text{pm})}{\delta^{13}C(\text{new}) - \delta^{13}C(\text{pm})} \times 100 \quad (8)$$

where $\delta^{13}C(\text{soil})$, $\delta^{13}C(\text{pm})$, and $\delta^{13}C(\text{new})$ represent the stable C isotopic composition of the carbonate in the bulk soil, parent material, and the pure PC, respectively.

Stable isotope ratios of $^{13}C/^{12}C$ and $^{18}O/^{16}O$ are used to study carbonate dissolution, transportation, and precipitation processes and rates.^[17] They provide additional records of past environments that include shifts in vegetation, climate, and atmospheric circulation.^[18,19] As a result, studies of carbonate isotope chemistry have become an important component of paleoecology and global change research. Substantial variations in the ratio of these two isotopes are important to trace and quantify sources, sinks, and flux rates within the C cycle.^[20]

RATE OF PC FORMATION

A simulation model, CALDEP, was developed by Marion and Schlesinger^[21] to predict the amount of PC deposition

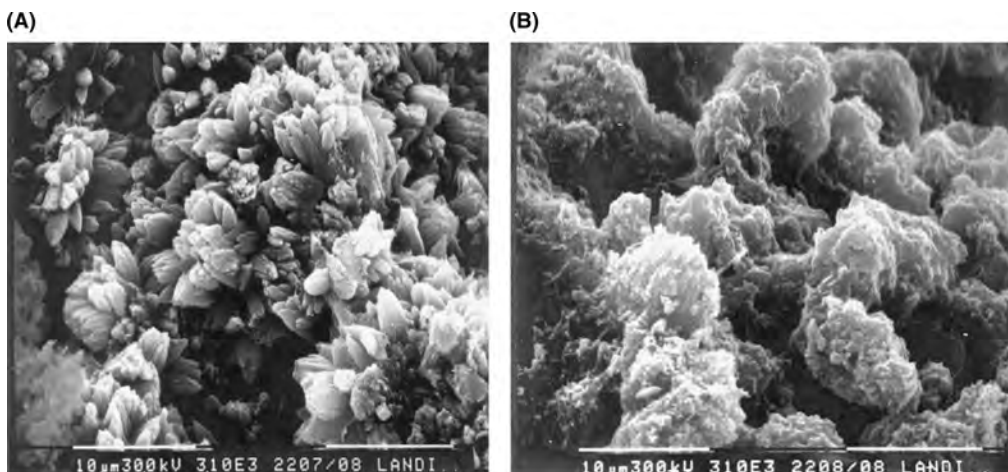


Fig. 2 Scanning electron micrographs of PC from the inner (A) and outer (B) layers of a carbonate pendant.

Table 1 Rates of PC deposition in the soils of the southwestern deserts of the United States and Saskatchewan.

Location	Rate of deposition (g CaCO ₃ m ⁻² yr ⁻¹)	Location	Rate of deposition (g CaCO ₃ m ⁻² yr ⁻¹)
Nevada, Mormon Mesa	1.73–10.85	Brown soils, SK, Canada	8.3
California, Whipple Mountain alluvium	1.0	Dark brown soils, SK, Canada	11.4
Arizona, Avra Valley	3.87–5.67	Black soils, SK, Canada	11.2
New Mexico, Rio Grande Valley	1.0–12.0		
New Mexico, Rio Grande Valley	2.2–5.1		

Source: From Schlesinger^[23] and Landi.^[24]

in soils (caliche) of the deserts of the Southwestern United States. The model assumed that most PC formed under climates with cool and wet winters during the Pleistocene. The predicted CaCO₃ deposition rate was 1–5 g m⁻² yr⁻¹ (0.12–0.6 g C m⁻² yr⁻¹) with an input of CaCO₃ as atmospheric dust of 0.51 g m⁻² yr⁻¹. The model also predicted that an increase in rainfall resulted in an increase in CaCO₃ precipitation.

The rate of PC accumulation calculated by Machette^[22] for non-calcareous parent materials ranges from 1.4 to 5.1 g m⁻² yr⁻¹ (0.17–0.61 g C m⁻² yr⁻¹) for three areas in New Mexico and Utah. Using ¹⁴C, ²³⁰Th, and ²³⁴U dates of calcic horizons of Aridisols in the Mojave Desert, Schlesinger^[23] has calculated the rate of deposition of PC to be between 1.0 and 3.5 g CaCO₃ m⁻² yr⁻¹ (0.12–0.42 g C m⁻² yr⁻¹). In semiarid Saskatchewan, the rate ranges between 8.3 and 11.4 g CaCO₃ m⁻² yr⁻¹ (0.99–1.34 g C m⁻² yr⁻¹; Table 1). The rate increases with increasing precipitation.

CONCLUSION

Secondary or PCs are those that are formed through the processes responsible for the soil development. The PC is an important part of the global carbon cycle. Global estimates are far from complete. The stable isotope technique is used to calculate the amount and rate of secondary carbonates. The rate of secondary carbonate formation in arid regions of California, Arizona, and Mexico states in the United States ranges from 1 to 5 g C m⁻² yr⁻¹, and it increases up to 11.2 g C m⁻² yr⁻¹ with increasing precipitation in semiarid Saskatchewan, Canada. Studies from other parts of the world are needed to make a reliable estimate of global PC.

REFERENCES

- Gile, L.H.; Peterson, F.F.; Grossman, R.B. The K horizon: A master soil horizon of carbonate accumulation. *Soil Sci.* **1965**, *99*, 74–82.
- Schlesinger, W.H. Carbon storage in the caliche of arid soils. A case study from Arizona. *Soil Sci.* **1982**, *133*, 247–255.
- Sombroek, W.G.; Nachtergale, F.O.; Heble, A. Amounts, dynamics, and sequestration of C in tropical and subtropical soil. *Ambio* **1993**, *22*, 417–426.
- Batjes, N.H. Total carbon and nitrogen in the soils of the world. *Eur. J. Soil Sci.* **1996**, *47*, 151–163.
- Eswaran, H.; Reich, P.F.; Kimble, J.M.; Beinroth, F.H.; Padmanabhan, E.; Moncharoen, P. Global carbon stocks. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2000; 15–25.
- Mermut, A.R.; Amundson, R.; Cerling, T.E. The use of stable isotopes in studying carbonates dynamics in soils. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2000; 65–85.
- St. Arnaud, R.J.; Herbillon, A.J. Occurrence and genesis of secondary magnesium-bearing calcites in soils. *Geoderma* **1973**, *9*, 279–298.
- Rabenhorst, M.C.; Wilding, L.P.; West, L.T. Identification of pedogenic carbonates using stable carbon isotope and microfabric analyses. *Soil Sci. Soc. Am.* **1984**, *48*, 125–132.
- Doner, E.H.; Lynn, W.C. Carbonate, halide, sulfate and sulfide minerals. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society America: Madison, 1989; 279–330.
- Drever, J.I. *The Geochemistry of Natural Waters*; Prentice Hall Inc.: Englewood Cliffs, 1982; 388 pp.
- Lindsay, W.L. *Chemical Equilibria in Soils*; Wiley: New York, 1979; 449 pp.
- Volk, R.L. The natural history of crystalline calcium carbonate: Effect of magnesium content and salinity. *J. Sediment. Petrol.* **1974**, *44*, 40–53.
- Mermut, A.R.; St. Arnaud, R.J. A study of microcrystalline pedogenic carbonates using submicroscopic techniques. *Can. J. Soil Sci.* **1981**, *61*, 261–272.
- Wieder, M.; Yaalon, D.H. Effect of matrix composition on carbonate nodule crystallization. *Geoderma* **1974**, *11*, 95–121.
- Magaritz, M.; Amiel, A.J. Calcium carbonate in a calcareous soil from the Jordan valley, Israel: Its origin as revealed by the stable carbon isotope method. *Soil Sci. Soc. Am. J.* **1980**, *44*, 1059–1062.
- Salomons, W.; Mook, W.G. Isotope geochemistry of carbonate dissolution and reprecipitation in soils. *Soil Sci.* **1976**, *122*, 15–24.

17. Amundson, R.G.; Lund, L.J. The stable isotope chemistry of a native and irrigated Typic Natrargid in the San Joaquin valley of California. *Soil Sci. Soc. Am. J.* **1987**, *51*, 761–767.
18. Cerling, T.E. The stable isotopic composition of modern soil carbonate and its relationship to climate. *Earth Planet. Sci. Lett.* **1984**, *71*, 229–240.
19. Amundson, R.G.; Chadwick, O.A.; Kendall, C.; Wang, Y.; DeNiro, M. Isotopic evidence for shifts in atmospheric circulation patterns during quaternary in mid-North America. *Geology* **1996**, *24*, 23–26.
20. Boutton, T.W. Stable carbon isotope ratios of natural materials: II. Atmospheric, terrestrial, marine, and freshwater environments. In *Carbon Isotope Techniques*; Coleman, D.C., Fry, B., Eds.; Academic Press: New York, 1991; 173–185.
21. Marion, G.M.; Schlesinger, W.H. Quantitative modeling of soil forming process in deserts: The CALDEP and CALGYP models. In *Quantitative Modeling of Soil Forming Processes*; Bryant, R.B., Arnold, R.W., Eds.; Soil Science Society of America (SSSA) Special Publication: Madison, 1994; Vol. 39, 129–145.
22. Machette, M.N. Calcic soils of the southwestern United States. In *Soils and Quaternary Geomorphology of Southwestern United States*; Weide, D.L., Ed.; Geological Society of America, 1985; Vol. 203, 1–21.
23. Schlesinger, W.H. The formation of caliche in soils of Mojave desert, California. *Geochim. Cosmochim. Acta* **1985**, *49*, 57–66.
24. Landi, A. *Carbon Balance in the Major Soil Zones of Saskatchewan*. Ph.D. Thesis, University Saskatchewan, Canada, 2002.

Carbonates: Water and

Michael Duniway

Jornada Experimental Range, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Las Cruces, New Mexico, U.S.A.

Abstract

The soil carbonates occurring as solidified masses or dispersed particles can alter the soil water dynamics compared to that expected from noncarbonate soil properties. Carbonate minerals in the soil can be derived from high-carbonate parent material, additions in the form of carbonate mineral containing rain or dust, high-carbonate groundwater, and formed *in situ* as secondary minerals. Soil carbonates found in the fine earth fraction are predominately derived from precipitates and are usually of the clay and silt size classes. Carbonate coarse fragments can be derived from carbonate parent material detritus or from pedogenic sources such as nodules and petrocalcic rubble. Parts of the profile can be continuously indurated with carbonates due to either the high-carbonate bedrock or the formation of a petrocalcic horizon. The accumulation of dispersed soil carbonates appears to increase the available water-holding capacity (AWHC) in coarse-textured soils and some fine-textured soils. Due to the relatively high AWHC of carbonate rock-like material ($0.03\text{--}0.22\text{ m}^3\text{ m}^{-3}$), soil profile AWHC will likely be underestimated if fragments of indurated carbonate material or clasts of carbonate parent material are ignored. The high water-holding capacity of petrocalcic horizons, high-carbonate bedrock, and the presence of water within these cemented soil horizons indicate their potential importance as a plant's water source, especially during dry periods.

INTRODUCTION

Carbonate minerals occur commonly in soils and have long been recognized for their importance in soil nutrient availability and soil structure. The soil carbonates occurring as solidified masses or dispersed particles, however, can alter the soil water dynamics compared to that expected from the noncarbonate soil properties. Carbonate minerals in the soil can be derived from high-carbonate parent material, additions in the form of carbonate mineral containing rain or dust, high-carbonate groundwater, and formed *in situ* as secondary minerals. Although many carbonate minerals occur, calcite (calcium carbonate) accounts for the vast majority of carbonates in soils (see the entry *Inorganic Carbon: Composition and Formation*, p. 1199) and will herein be referred to as carbonates. Soil carbonates in the fine earth fraction are predominately derived from precipitates and are usually of the clay and silt size classes.^[1] Carbonate coarse fragments can be derived from carbonate parent material detritus or from pedogenic sources such as nodules and petrocalcic rubble. Parts of the profile can be continuously indurated with carbonates due to either high-carbonate bedrock or the formation of a petrocalcic horizon. All sizes and morphologies of soil carbonates have important implications in soil water availability and dynamics.^[2–6]

FORMATION

Whether the source is carbonate dust, rain, parent material, or weathering of primary minerals, carbonate minerals tend

to move down the soil profile in solution. Because carbonates can be leached out of the soil profile, soils with the greatest amounts of carbonate occur in areas where evapotranspiration losses exceed precipitation, including arid, semiarid, and subhumid climates (see the entry *Inorganic Carbon: Composition and Formation*, p. 1199).

In poorly leached soils, carbonate minerals precipitate where the soil solution dries and reactants are concentrated. Fine carbonate crystals ($\sim 2\text{--}10\text{ }\mu\text{m}$) initially precipitate along roots, fungal hyphae, and soil particle surfaces and progressively fill soil pores.^[7] With sufficient time, carbonates can completely plug soil pores, producing an indurated horizon and a distinct laminar carbonate cap (petrocalcic horizon). The formation of laminar morphology is attributed to the restriction of downward soil water movement and the precipitation of carbonates in the accumulated soil water. Petrocalcic horizons are characterized by high densities and bulk carbonate contents.^[3]

DISPERSED CARBONATES

The accumulation of soil carbonates generally increases the available water-holding capacity (AWHC) of coarse-textured soils but has little effect or reduces the AWHC of fine-textured soils. Based on micromorphology, plugging of pores with carbonates can change a coarse-textured soil from a matrix of large pores to one dominated by fine pores.^[7] Soil water release curves (SWRCs) measured for a high-carbonate calcic horizon resemble those of soils with higher clay contents than those of soils with the

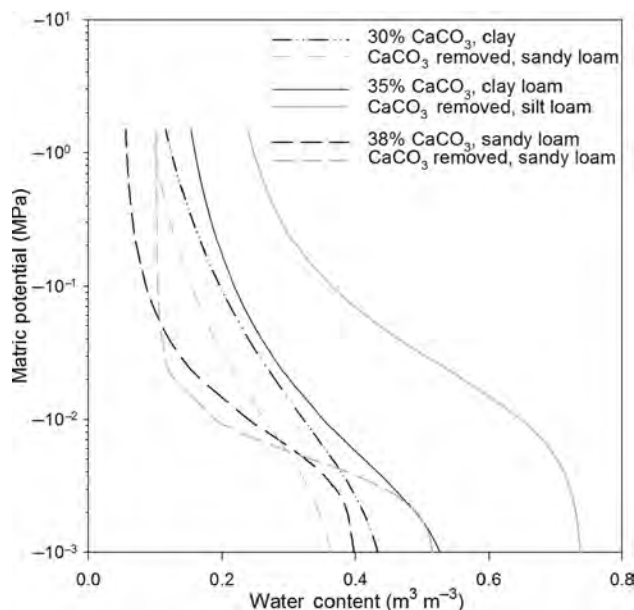


Fig. 1 SWRCs of nonindurated high-carbonate soils (black) and of soils after carbonates were removed (gray). Gray and black lines in the same pattern are SWRCs of the same samples with and without carbonates, respectively. Carbonate content and soil textural class with and without carbonates are noted in the legend. **Source:** Adapted from Baumhardt & Lascano^[2] and Stakman & Bishay.^[8]

noncarbonate horizon texture (sandy clay loam; Fig. 1).^[2] Conversely, Stakman and Bishay^[8] found soil carbonates to reduce the amount of water held across all potentials, except in very coarse-textured soils (Fig. 1). The carbonate content was found to be negatively correlated with soil water retained at water potentials near saturation (-1 and -10 kPa) in a predictive model developed for high-carbonate glacial till subsoils.^[9]

Extensive water extraction by several row crop species from a strong calcic horizon has been documented.^[10] For some species, plant water availability and specific root water-uptake rates were higher in the calcic horizon than in the overlying soil layers. Similarly, fairly high hydraulic conductivity values have been reported for calcic horizons, and values that are larger than would be expected based on the texture alone.^[2] Despite the importance of soil carbonates for soil water-holding capacity and dynamics, methods for measuring soil particle size distribution, the soil property often used to predict soil water-holding capacity and dynamics, often recommend pretreatment to remove carbonates.

INDURATED CARBONATE NODULES AND CLASTS

The water-holding capacity of carbonate nodules, fragments, and parent material clasts has been found to be

significant^[3,4] and often higher than that of other coarse fragments, either pedogenic or those inherited from the parent material.^[11–14] AWHC for calcareous rock fragments is generally lower (0.03 – 0.14 $\text{m}^3 \text{m}^{-3}$)^[4] than that reported for fragments from a petrocalcic horizon (0.04 – 0.22 $\text{m}^3 \text{m}^{-3}$).^[3] Limestone clast SWRCs are similar to those of laminar morphology petrocalcic material of similar bulk densities (Fig. 2). Petrocalcic fragments of plugged morphology, which are cemented by carbonates within the original soil matrix, have much higher AWHC than either laminar morphologies or limestone materials. Because plugged morphologies of petrocalcic horizons contain some of the noncarbonate soil matrix, bulk density as well as carbonate content appears to be important for water retention (Fig. 2).^[3] Soil volume occupied by clasts, either carbonate or otherwise, however, is almost always excluded when quantifying soil profile available water and soil water-holding capacity.

CARBONATE BEDROCK AND PETROCALCIC HORIZONS

Limestone bedrock and petrocalcic horizons have been shown to contain significant amounts of water that is potentially available to plants.^[5,6] In many soils, these layers occur within the rooting depths of deep-rooted shrubs and in some areas within the rooting zone of grasses and row crops. Although carbonate-cemented soil horizons appear to be water restricting, there is evidence that these horizons

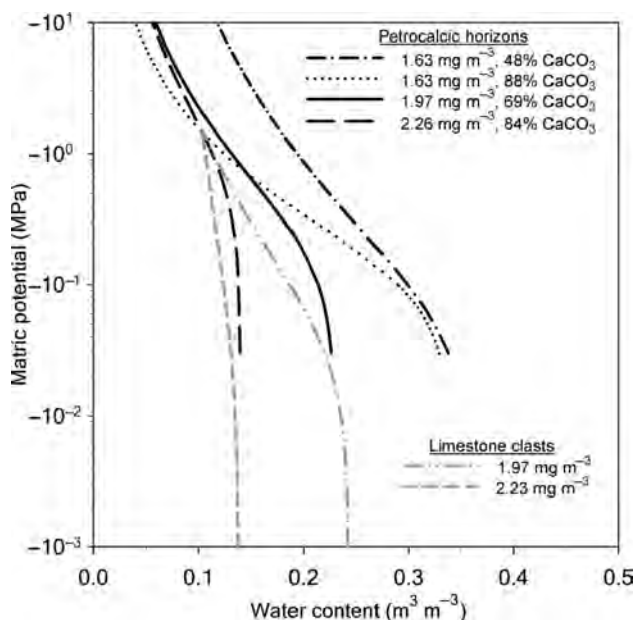


Fig. 2 SWRCs for petrocalcic horizon material and limestone clasts from calcareous glacial till soils. Bulk density and carbonate content are noted in the legend.

Source: Adapted from Duniway, Herrick, et al.^[3] and Cousin & Nicoullaud.^[4]

do absorb soil moisture. Petrocalcic horizons can rapidly absorb and retain large volumes of soil water with a measured field capacity of up to $0.36 \text{ m}^3 \text{ m}^{-3}$.^[6] When compared with similar depths in deep sandy soils, shallow petrocalcic horizons (~0.5 m deep) in sandy soils have been shown to absorb and retain larger quantities of winter and summer rains and retain water at plant-available tensions more consistently.^[6] Following a wet summer, the petrocalcic horizon monitored retained very high plant-available water contents (16–18% volumetric) through early spring of the following year, more than double the amount retained by similar depths in the deep sandy soil.^[6] Native perennial grasses growing on similar petrocalcic soil in a semiarid region were found to have lower mortality rates than grasses growing on coarse-textured soils lacking shallow petrocalcic horizons.^[15] Similarly, relatively high soil water contents have been reported for limestone bedrock in a dry tropical forest.^[5] In the same system, roots of native tree species were observed within the softer limestone along with isotopic evidence of utilization of the contained water.

CONCLUSION

High-carbonate soil horizons are common in arid, semiarid, and subhumid regions around the world. Understanding soil AWHC and dynamics is of particular importance in these dry climates. The accumulation of soil carbonates appears to increase AWHC and improve soil water relations in coarse-textured soils^[3,6,8] and some fine-textured soils.^[2] Soil profile AWHC will likely be underestimated if fragments of indurated carbonate material or clasts of carbonate parent material are ignored.^[3,4] In coarse-textured soils, the water retention provided by the fine porosity in these fragments could be of particular importance during extended dry periods. Similarly, the water-holding capacity of petrocalcic horizons and high-carbonate bedrock indicates their potential importance as a plant's water source, especially during drought. Retention properties of these high-carbonate layers do limit the downward loss of soil water. Instead of posing an insurmountable obstacle to soil water movement as generally believed, however, carbonate-cemented soils and parent materials appear to potentially function as a reservoir of water in some systems. Water in some of these denser materials could be made available to plants through either bulk flow to roots in contact with the outer surface or mycorrhizal relationships. More work is needed to further understand the importance of water contained in carbonate rock and rock-like material and the importance of this water for existing and potential plant communities.

REFERENCES

1. Bui, E.N.; Loeppert, R.H.; Wilding, L.P. Carbonate phases in calcareous soils of the western United States. *Soil Sci. Soc. Am. J.* **1990**, *54* (1), 39–45.
2. Baumhardt, R.L.; Lascano, R.J. Physical and hydraulic properties of a calcic horizon. *Soil Sci.* **1993**, *155* (6), 368–375.
3. Duniway, M.C.; Herrick, J.E.; Monger, H.C. The high water-holding capacity of petrocalcic horizons. *Soil Sci. Soc. Am. J.* **2007**, *71* (3), 812–819.
4. Cousin, I.; Nicoulaud, B.; Coutadeur, C. Influence of rock fragments on the water retention and water percolation in a calcareous soil. *Catena* **2003**, *53* (2), 97–114.
5. Querejeta, J.I.; Estrada-Medina, H.; Allen, M.F.; Jiménez-Osornio, J.J.; Ruenes, R. Utilization of bedrock water by *Brosimum alicastrum* trees growing on shallow soil atop limestone in a dry tropical climate. *Plant Soil* **2006**, *287* (1–2), 187–197.
6. Duniway, M.C.; Herrick, J.E.; Monger, H.C. Spatial and temporal variability of plant-available water in calcium carbonate-cemented soils and consequences for arid ecosystem resilience. *Oecologia* **2010**, *163* (1), 215–226.
7. Monger, H.C.; Daugherty, L.A.; Gile, L.H. A microscopic examination of pedogenic calcite in an aridisol of southern New Mexico. In *Occurrence, Characteristics, and Genesis of Carbonate, Gypsum, and Silica Accumulations in Soils*; Nettleton, W.D., Ed.; SSSA Special Publication; Soil Science Society of America (SSSA): Madison, 1991; 37–60.
8. Stakman, W.P.; Bishay, B.G. Moisture retention and plasticity of highly calcareous soils. *Neth. J. Agric. Sci.* **1976**, *24*, 43–57.
9. Jensen, N.H.; Balstrom, T.; Breuning-Madsen, H. The relations between soil water retention characteristics, particle size distributions, bulk densities and calcium carbonate contents for Danish soils. *Nord Hydrol.* **2005**, *36* (3), 235–244.
10. Geogren, P.G.; Davis-Carter, J.; Taylor, H. Root-growth and water extraction patterns from a calcic horizon. *Soil Sci. Soc. Am. J.* **1991**, *55* (1), 210–215.
11. Flint, A.L.; Childs, S. Physical properties of rock fragments and their effect on available water in skeletal soils. In *Erosion and Productivity of Soils Containing Rock Fragments*; Box, J.E., Ed.; SSSA Special Publication; Soil Science Society of America (SSSA): Madison, 1984; 91–103.
12. Jones, D.P.; Graham, R.C. Water-holding characteristics of weathered granitic rock in chaparral and forest ecosystems. *Soil Sci. Soc. Am. J.* **1993**, *57* (1), 256–261.
13. Tokunaga, T.K.; Olson, K.R.; Wan, J. Moisture characteristics of hanford gravels: Bulk, grain-surface, and intragranular components. *Vadose Zone J.* **2003**, *2* (3), 322–329.
14. Brouwer, J.; Anderson, H. Water holding capacity of ironstone gravel in a typic plinthoxeralf in southeast Australia. *Soil Sci. Soc. Am. J.* **2000**, *64* (5), 1603–1608.
15. Herbel, C.H.; Ares, F.N.; Wright, R.A. Drought effects on a semidesert grassland range. *Ecology* **1972**, *53* (6), 1084–1093.

Care of Soil

Delia C. Catacutan

Global Research Program, World Agroforestry Center, Malaybalay City, the Philippines

Abstract

The term “soil care” reflects the human and social dimension of soil management, based on the growing realization that the problems of soil degradation cannot only be single-handedly addressed by technical solutions but also through engagement of multiple stakeholders, whose day-to-day decisions affect the health of the soil. From its roots in Australia, landcare is widely adopted as a multistakeholder approach to integrated natural resource management, where multitudes of farmers and land users work together to care for the land and improve their livelihoods. Landcare has shown that a “land ethic” is fundamental to successful conservation efforts.

INTRODUCTION

“Soil” is one, if not the most important, input to production. With constantly increasing demand for production of food and fiber and environmental services, “care of soil” has never been urgent. Soil degradation is a physical process; hence, conventional soil management focused on technological solutions. However, the underlying causes of soil degradation are deeply rooted in complex sociocultural, economic, and political contexts.^[1] Therefore, institutional and policy solutions are equally pressing as the technical solutions to soil degradation.

The term “soil care” resonates the human and social dimension of soil management, based on the growing realization among scientists, practitioners, scholars, advocates, policy makers, local communities, and most importantly farmers, that the earth’s soil is not just a biophysical unit, requiring discrete management techniques, but largely a social unit where multiple decisions of multiple stakeholders have a direct impact on its health. As part of an integrated natural resource management (NRM) strategy, “care of soil” requires a nuanced process of incorporating complex aspects of soil management into a sustainable system that meets the explicit goals of production, profitability, and risk reduction by multiple stakeholders. This means that soil care is no longer the exclusive domain of soil scientists; in general, NRM calls for “integration” across disciplines and objectives, scales, stakeholders, structures, and components.^[2]

This entry discusses soil care from a social perspective. Landcare is presented as a multistakeholder approach to NRM. It concludes that the bottom line of soil care is a “land ethic” developed among stakeholders within and across sectors and levels of society.

MULTISTAKEHOLDER ENGAGEMENT IN SOIL CARE

Soil degradation is a physical process, but soil management is largely a social process. As a key natural resource, soil health is influenced by the day-to-day management decisions of large numbers of different types of stakeholders at various scales.^[1,3] These stakeholders often (e.g., small and large holder farmers, policy makers, managers, administrators, businesses, scientists, communities, other economic and social sectors, etc.) have contrasting objectives, based on their specific circumstances and positions in society.^[3] For farmers, trade and price policies are crucial because their decisions are commonly responsive to relative prices and to price variability.^[4] Because of variability of production inputs (e.g., soil fertility) and volatility of outputs, the farmers constantly struggle to meet their production goals while managing soil fertility, and governments are stifling its ability to enforce regulatory measures on land use. As soil fertility drops, productivity declines, and incomes fall, poor farmers are being blamed for environmental degradation.^[1,5] This is unforgiving, given the lack of incentives to farmer adoption of conservation technologies due to the cycle of poverty and skewed patterns of economic development.^[1,6,7]

The whole gamut of soil management is complex, requiring active involvement of various stakeholders. However, multiple stakeholder engagement is not straightforward. Many of the difficulties experienced in engaging stakeholders are the result of poor policy analysis and program design.^[1,8] The underlying issue is that many economies employ a reductionist approach with institutions specializing in either production or conservation, or regulation and control.^[1] This structural constraint reduces the chances for innovation and integration. Furthermore,

stakeholder decisions are often formulated based on political considerations; hence, engaging multiple stakeholders in soil care is a formidable task.^[3]

The Landcare Approach

Landcare is a multistakeholder approach that employs innovative solutions to NRM challenges, linking farmers with the broader community and helping them influence resource policy.^[1,9] It centers on formation of landcare groups, supported to varying degrees through partnerships with government, non-government, and private sector stakeholders.^[6,9,10] Landcare groups are involved in farm care, catchment care, vegetation and coastal management, property planning, capacity building and research, and resource generation for livelihood financing.^[1]

Beginning in mid-1980s, the growth of Australian Landcare has been explosive, with over 4000 landcare groups formed.^[1,6] Bipartisan political support and the broad popular appeal of landcare contributed to success, but the presence of catastrophic land degradation across vast swathes of Australia triggered the landcare movement.^[6] Australian soils are declining with annual soil loss averaging between 5 and 10 tons ha⁻¹, with sugarcane as the most erosive crop (150 tons ha⁻¹ yr⁻¹).^[11] The country is basically short of good soils, with less than 300,000 km² (less than 4%) classified as good or very good, in terms of quality for broadscale cropping.^[12] However, significant progress has been achieved through landcare, and it has been highlighted as part of the fabric for achieving sustainable ecosystems.^[6]

Some 17 countries or multilateral organizations in the Pacific, Africa, America, Europe, the United Kingdom, and Southeast Asia are either independently implementing landcare programs or receiving limited support to initiate them.^[3] The genesis of landcare in these countries was different, evolving through different pathways, but the problems that landcare groups are trying to address are similar,

adhering to the same principle, which is enriching human and social capital to mobilize local action, to reverse land degradation and improve rural livelihoods.^[1] Regardless of differences in circumstances, the essential requirements to facilitate this process are also common, i.e., a good balance between community efforts and public-private partnerships.

Evidence suggests that landcare has significantly contributed to improving NRM and livelihood outcomes in areas where it is active (Fig. 1). Landcare associations around the world have developed more resilient capacities to transform farming systems to mitigate and adapt to climate change through practices that increase carbon sequestration.^[1] In the Philippines, landcare provides a coping mechanism for communities to achieve resiliency.^[13] In Uganda, Australia, New Zealand, Germany, Iceland, and South Africa, landcare associations are actively implementing sustainable farming and catchment-wide management projects.^[1]

A distinctive strength of landcare is the involvement of research and development (R&D) institutions. In New Zealand, landcare is implemented by the NZ Landcare Trust, a research-based non-governmental organization (NGO), while the World Agroforestry Centre (International Centre for Research in Agroforestry—ICRAF) pioneered landcare research in the Philippines and in Eastern Africa, and in the United States, Virginia Tech University facilitates landcare R&D linkages. Landcare has thus demonstrated a mechanism for linking scientific knowledge with actions from the local to the global level.^[1]

Building a Land Ethic

There are many aspects, with which to identify the success of landcare. Among them is improved human and social capital for farmers to apply conservation technologies and stakeholder engagement in decision making.^[1] The high level of community participation indicates profound

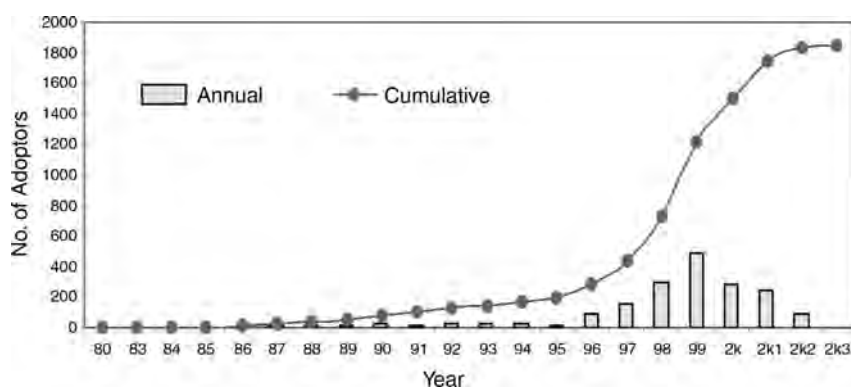


Fig. 1 Annual and cumulative adoption of contour hedgerows using natural vegetative strips (NVSS) and agroforestry practices in Claveria, Misamis Oriental, in the southern Philippines. Rapid adoption of NVS and agroforestry was observed beginning in 1996, when farmers had started to actively disseminate the technology throughout the municipality. Rapid increase in adoption was unprecedented in Claveria despite previous efforts of other agencies that promoted conservation farming in the Philippine uplands. Farmers were technically assisted by local agricultural technicians and ICRAF researchers. This marked the beginning of landcare in the Philippines.

Source: From ICRAF Philippines' database.

cultural changes in people's understanding on the land and their relationship with their environments.^[6] Stakeholder engagement across geographic scales (farms–catchments); sectors and structures (farmer–government agencies–NGO–business–R&D); and disciplines, interests, and objectives (farm conservation–water management–research–capacity building–livelihood) fostered the emergence of a “land ethic” based on the spirit of stewardship and volunteerism, complimented with various types of support and incentives.^[14]

CONCLUSION

Care of soil has never been resolute than in the past decades. The greatest challenge in human history has been developing management systems that meet present and future needs of food and fiber and ecosystem services. Integrating soil science solutions across other disciplines has become more important, not only because the soil is a complex biophysical unit but also because it is a dynamic social unit when all factors of utilization are considered. Multiple sources of knowledge are needed to draw multiple solutions, through engagement of different types of stakeholders at various scales. The landcare experience has shown that reversing land degradation and broad-level NRM outcomes can only be achieved through building a land ethic among the global community.

ACKNOWLEDGMENTS

The author is grateful to many landcare colleagues for sharing their experiences, on which this entry is based. Special acknowledgment is due to the Iceland Soil Conservation Service for providing the opportunity to share the landcare experience at the 2007 Global Forum on Soils, Society, and Global Change, with which this topic was deemed relevant to soil science and to the Soil Science Encyclopedia.

REFERENCES

1. Catacutan, D.; Tanui, J. Engaging stakeholders in integrated natural resource management: Approaches and guidelines from landcare. In *Proceedings of the Global Forum on Soils, Society and Global Change*, Selfoss, Iceland, Aug 30–Sept 4, 2008; Soil Conservation Service of Iceland: Gunnarsholt, 2008.
2. Lal, P. The adaptive decision-making process as a tool for INRM: Focus, attitudes, and approach. *Conserv. Ecol.* **2001**, 5 (2), 11.
3. Van Noordwijk, M.; Tomich, T.P.; Verbist, B. Negotiation support models for integrated natural resource management in tropical forest margins. In *Integrated Natural Resource Management-Linking Productivity, the Environment and Development*; Campbell, B.M., Sayer, J.A., Eds.; CAB International: Cambridge, 2003.
4. Coxhead, I.; Demeke, B. How do relative price changes alter land use decisions? Panel Data Evidence from the Manupali Watershed, Philippines. In *Land Use Change in Tropical Watersheds: Evidence, Causes and Remedies*; Coxhead, I., Shively, G., Eds.; CABI Publishing: Cambridge, 2005; 78–88.
5. Boada, E.L. Incentive policies and forest use in the Philippines. In *Public Policies and the Misuse of Forest Resources*; Repetto, R.C., Gillis, M., Eds.; University of Cambridge: Cambridge, 1998; 165–204.
6. Catacutan, D.C. *Scaling Up Landcare in the Philippines: Issues, Methods and Strategies*; World Agroforestry Centre; Southeast Asia Regional Research Programme; 2007.
7. Nelson, R.A. *Bioeconomic Analysis of Hedgerow Intercropping in the Philippine Uplands*. PhD Thesis, University of Queensland, Brisbane, Queensland, 1996.
8. Gleeson, T.; Crawford, P.; Douglas, J. *Guide to Australian Landcare Management*; Australian Landcare Management Systems Ltd: Brisbane, Queensland, 2004. Available at http://www.regional.org.au/au/apen/2005/1/3159_gleeson.htm?print=1.
9. Cramb, R.A.; Culasero, Z. Landcare and livelihoods: The promotion and adoption of conservation farming systems in the Philippine uplands. *Int. J. Agric. Sustain.* **2003**, 1 (2), 141–154.
10. Bumacas, D.; Catacutan, D.; Chibememe, G.; Odigha, O.; Rhoades, C. Enabling local communities to developing and scaling-up agriculture: A grassroots perspective. In *Farming with Nature: The Science and Practice of Eco-Agriculture*; Mc Neely, J., Scherr, S., Eds.; Island Press: Washington, 2007; 286–307.
11. <http://oregonstate.edu/~muirp/erosion.htm>.
12. <http://home.alltel.net/bsundquist1/se2d.html#Ee>.
13. Villanueva, J.; Espaldon, M.V.; Lasco, R.; Perez, R.; Catacutan, D. Abstract of Papers, No. 38, International Conference of Asian Scholars (ICAS5), KLCC, Malaysia, Aug 6–10, 2007.
14. Robertson, D.; Gabbard, C.; Hull, B.; Moles, J.; Stoke, J.; Burge, S.; Waldon, J.; Lowe, D. Landcare in America. In *Landcare: Local Action-Global Progress*; Catacutan, D., Neely, C., Johnson, M., Poussard, H., Youl, R., Eds.; Landcare International: Melbourne, 2008; 110–125.

Cerrado

Igo F. Lepsch

Soil Science Department, Luiz de Queiroz College of Agriculture (USP-ESALQ),
Piracicaba, Brazil

Abstract

The Cerrado is a South American tropical savanna ecosystem that originally covered about two million square kilometers. The major areas are on in Central Brazil where annual rainfall is between 1100 and 1600 mm concentrated in a period of 6 or 7 months. About 45% of the Cerrado's soils are the well-drained Oxisols (also named as Ferralsols in the FAO soil map of the world and latossolos in the Brazilian soil classification system) occurring either on extensive, almost flat, tableland tops, or on gently undulating low hill surfaces. Other Cerrado's soils are the Ultisols (16%), Entisols (20%), and Cambisols (3%). The Oxisols were developed on stable geomorphic surfaces under thick layers (more than 2 m) of pre-weathered parent materials. Their high contents of aluminum and iron oxides made them very permeable and with good physical properties for mechanized agriculture even with high amounts of clay. These tropical, naturally nutrient-poor, deep acid soils with low cation exchange capacity were long considered unsuitable for crop production. However, research in soil science in Brazil has provided knowledge for the utilization of the Cerrado's vast areas of Oxisols as extremely productive agricultural lands. This achievement, in the last years, is considered the world's single largest increase in farmland since the settlement of Midwestern United States that began in 1850.

INTRODUCTION

The Cerrado biome is a kind of edaphic tropical savannah which, before being occupied by intensive agriculture, covered approximately two million square kilometers (Fig. 1). It is typical of the high plateaus that preserve old geomorphic surfaces in central Brazil where vegetation has evolved to survive for millions of years adapting to long dry periods and having nutrient-poor and very acidic soil conditions. Its flora is considered to be the richest in biodiversity among the world's savannahs.^[1] Almost half of the land originally covered by the Cerrado vegetation has been transformed into cultivated pastures and extensive crops.

CLIMATE, VEGETATION, AND GEOLOGY

The Cerrado climate is limited to two dominant seasons throughout the year—wet and dry, with the dry season during the southern winter (approximately April–September). Annual temperatures average between 22°C and 27°C, and average precipitation is between 800 and 2000 mm for over 90% of the area.^[1]

This large-scale vegetal province possesses distinctive flora with hundreds of endemic species. There is much evidence that the Cerrado biome has existed in prototypical form since the Tertiary period.^[1,2] During this time, the plant species of the Cerrado have evolved under alternating

conditions of subhumid and semiarid climates and on highly weathered, acidic, and extremely nutrient-poor soils.^[2] The vegetation physiognomies include forest savannah (*cerradão*), wooded savannah (*cerrado strictu sensu*), park savannah (*campo cerrado*), gramineous-woody savannah (*campo sujo*), and grasslands (*campo limpo*). Some gallery forests around wetlands are also included. The *cerrado strictu sensu* is a mosaic of sparse vegetation of trees and shrubs with twisted trunks and crooked or irregular branching, often possessing rigid leaves and thick wrinkled or cracked barks; the herbaceous stratum is usually short hard grasses.^[3]

The parent rocks of the Cerrado soils are mostly pre-Cambrian rocks of the Guiana and Brazilian shields (gneisses, schists, and granites) and Paleozoic and Mesozoic basalts, sandstones, and shales. However, the parent material of most of the present soils was weathered on former non-glaciated land surfaces and redistributed during many episodes of erosion, especially during the shifting from humid to semiarid climates (that coincided with interglacial and glacial periods of lower latitudes) of early Cenozoic time.^[4] As a result, few micas, feldspars, apatites, and other weatherable minerals containing potassium (K), calcium (Ca), magnesium (Mg), or phosphorus (P) remained, having long since been dissolved while in previous locations and/or during transport. Most present-day soils formed in such material that had been exposed to many cycles of weathering, erosion, and deposition; almost no nutrient-bearing minerals were present and sand-sized



Fig. 1 Geographic location of the Cerrado region in Brazil.

quartz and clay-sized iron and aluminum oxides, as well as, kaolinite are the dominant primary minerals remaining in the parent material within which the present soils have formed.^[5,6]

PRINCIPAL SOILS OF THE CERRADO

The principal soils of the Cerrado are represented by the well-drained “non-plinthic” Oxisols,^[7] also named Ferralsols in the Food and Agriculture Organization (FAO) soil map of the world^[8] and Latossolos in the Brazilian Soil Classification System,^[9] and they comprise about 45% of the Cerrado’s area and are well-drained soils without iron concretions, occurring either on the extensive, almost flat, tableland tops (locally called “chapadas”) or on gently undulating low hill surfaces. These Oxisols are deep and well drained presenting no physical barriers to root development. Other Cerrado’s soils are classified^[7] as Ultisols (16%), Entisols (20%), and Cambisols (3%). The Entisols are either very sandy (Quartzpsamments) or shallow soils. About 9% of the area includes soils rich in soft (plinthite) or hard (petroplinthite) iron concretions (the former called “laterite”^[5,6]).

General soil landscape studies involving the Cerrado soils were reported by Feuer,^[10] Queiroz Neto,^[2] and Lepsch and Buol.^[6]

SOIL-RELATED CONSTRAINTS TO BIOMASS PRODUCTION

Soils that support cerrado vegetation are considered as some of the most chemically infertile in the world. Because of the nutrient-poor soils, the natural vegetation of the gramineous-woody Cerrado strata is so poor in nutrients that it was considered marginal even for

extensive domestic cattle pasture. There are no accounts indicating that herds of native large mammals are present in similar appearing but in nutrient-rich and water-stressed savannahs of Africa and prairies of Midwestern U.S. prior to European settlement. In the Cerrado, domestic cattle often died from broken bones after grazing only Ca- and P-deficient grasses, and ranchers came to refer to Cerrado as a place to lose cattle rather than graze cattle.^[5]

The highly weathered latossolos^[9] of the Cerrado are devoid of the plant macronutrients such as Ca, Mg, K, and P as well as some micronutrients. The association of kaolinitic clays with iron and aluminum oxides has the ability to adsorb P in forms unavailable to plants. However, this same mineral assemblage forms a strong fine and very fine granular structure often called “pseudosand.” The result is that very small voids within the granules retain water at high tension, and the large voids between the granules allow water to rapidly percolate under the force of gravity. This accounts for the characteristic of these latossolos—even with high amounts of clay, they are very permeable and have good physical properties such as good tilth capability.^[11]

MANAGEMENT OPTIONS FOR SUSTAINABLE LAND USE

Although the Latossolos originally under Cerrado vegetation can be considered as some of the most chemically infertile in the world, their cropped areas are among one of the most productive lands in the world. In the year 2014, about 14 million hectares were set into annual crops and 60 million hectares into improved pastures. Before 1960, very little farming or intensive grazing took place, and with a very low population density, the land purchase costs were cheap. Among the most important soil factors contributing to low yields are Ca, P deficiency, aluminum toxicity, and low soil water storage capacity.^[12]

Field experiments testing recommendations for liming and fertilizers were established shortly after the new capital, Brasilia, was built in the center of the Cerrado region, in the 1960s.^[5,12] Progressively, experienced farmers from south of Brazil moved into the area, bringing commercial farming practices and grain production based on these research findings.

The uniform temperatures and the dry season allowed for ideal conditions for sowing and harvesting the grain over extended periods of time, resulting in maximum efficiency of machinery operations and reducing time and cost to dry the harvested grain. In these lands, two cycles of crops per year are possible, even without irrigation if no-till systems—now a common practice—are used. Much of the production of grain and meat taking place there today is the result of the scientific understanding that some of the chemical properties of the soil had to be changed before

grain production and cattle grazing could take place.^[12] The two primary changes needed were to provide plant-available P and the replacement of extractable aluminum with Ca on the cation exchange capacity (CEC) sites in the soil.^[5,12]

The amount of surface area on the iron oxides present in high amounts in topsoil was responsible for adsorbing large amounts of P added as fertilizer. Field studies determined that initial applications of lime and relatively great amounts of P sustained maximum grain yields through several years^[5] and thus are economically feasible. The aluminum toxicity and Ca and Mg deficiency can be overcome with modest lime applications. Also, Ca when supplied as gypsum has been found to move downward more rapidly than calcium carbonate forms. With continued applications of lime and gypsum, the naturally Ca deficient and aluminum toxic subsoil will acquire exchangeable Ca below the depth of cultivation and provide for deeper rooting and thus buffer crops better from drought damage.^[5,12]

Modern soil science finds an example of the high yields obtained in the deep, highly weathered, and low CEC soils of the Cerrado that proves that vast tropical areas of previously uncultivated soil, long considered unsuitable for human food production, can be transformed into highly productive agricultural land. Naturally, acid soils with high contents of aluminum and iron oxides and low CEC values and organic matter contents, long considered insurmountable obstacles to crop production in tropical latitudes, could be extremely productive. This phenomenal achievement over a span of less than 50 years is the world's single largest increase in farmland since the settlement of mid-west U.S. that began in 1850 and has been hailed as a far reaching achievement and milestone in agricultural science.^[13]

REFERENCES

1. Ratter, J.A.; Ribeiro, J.F.; Bridgewater, S. The Brazilian cerrado vegetation and threats to its biodiversity. *Ann. Bot.* **1997**, *80*, 223–230.
2. Queiroz Neto, J.P. Solos da região do cerrado e interpretações. *Rev. Bras. Ciênc. Solo.* **1982**, *6*, 1–12.
3. Eiten, G. The cerrado vegetation of Brazil. *Bot. Rev.* **1972**, *38*, 201–341.
4. Orme, A.R. Tectonism, climate, and landscape change. In *The Physical Geography of South America*; Veblen, T.T., Young, K.R., Orme, A.R., Eds.; Oxford University Press: New York, 2007; 23–44.
5. Buol, S.W. Soils and agriculture in central-west and north Brazil. *Sci. Agr.* **2009**, *66* (5), 697–707.
6. Lepsch, I.F.; Buol, S.W. Oxisol-landscape relationships in Brazil. In *International Soil Classification Workshop 8: Proceedings*; Beinroth, F.H., Camargo, M.N., Eswaran, H., Eds.; SNLCS, EMBRAPA, SMSS-SCS-USDA: Rio de Janeiro, 1988; 174–189.
7. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Survey*; Soil Survey Staff: Washington, DC, 1975; 754 pp.
8. FAO/UNESCO. *FAO Soil Map of the World: 1: 500,000, Legend*; UNESCO: Paris, 1974; 1 pp.
9. Embrapa. *Centro Nacional de Pesquisa de Solos; Sistema Brasileiro de Classificação de Solos*; Brasília, 1999; 157 pp.
10. Feuer, R. *An Exploratory Investigation of the Soils and Agricultural Potential of the Soils of Federal District in Central Plateau of Brazil*. PhD Thesis, Cornell University, Ithaca, 1956; 432 pp.
11. Lepsch, I.F. Status of soil surveys and demand for soil series descriptions in Brazil. *Soil Hor.* **2013**, *54*, 1–5.
12. Lopes, A.S. Soils under Cerrado: A success story in soil management. *Better Crops Int.* **1996**, *10*, 9–15.
13. Seedquest. *World Food Prize Winners Opened Brazil's "Closed Lands."* Available at <http://www.seedquest.com/News/releases/2006/october/17249.htm> (accessed May 2015).

Chemical Composition

Goro Uehara

Department of Agronomy and Soil Science, University of Hawaii, Honolulu, Hawaii, U.S.A.

Abstract

Soil chemical composition is the product of the interactions of climate, topography, and biota operating over geologic time on a parent rock that is open to gains and losses of matter. Physical disintegration and chemical weathering transform existing unstable minerals into secondary or pedogenic minerals that may differ in kind and composition with depth in the profile. Soil chemical composition is important only in so far as it can be related to the behavior and performance of soils.

INTRODUCTION

Every student of soil science learns to recognize and appreciate soil differences and to understand how these differences affect soil behavior and performance. Soils can differ in particle size distribution or texture, in the way the individual particles are arranged and cemented to form compound particles or peds, in the amount and type of organic matter and in the mineralogy of the inorganic particles. Variations in any of the above will determine whether a soil will behave as a sponge to absorb and hold water, as a sieve and filter to purify water as it flows through the soil, or as a plastic or elastic rheologic body to withstand the pressure of a tractor wheel. How a soil behaves will, in turn, determine whether it will perform well as a soybean field, building foundation, playground, or waste disposal site.

The soil differences just described generally go hand in hand with differences in chemical composition. A soil that is 95% quartz sand or silicon dioxide will clearly behave and perform differently in many ways from another that is 95% coral sand or calcium (Ca) carbonate. These two soils would in turn behave and perform very differently from an organic soil, and all three would again differ greatly from a tropical soil rich in iron oxides. How do soils with such diverse chemical makeup originate? The purpose of this entry is to summarize the biogeochemical processes and factors that lead to differences in the chemical composition of soils.

SOIL-FORMING PROCESSES

Changes in the chemical composition of the rock and the soil that forms from it are the result of four soil-forming processes: transformations, translocations, additions, and losses.

Transformations

The process of weathering takes place because the minerals that make up the rock are thermodynamically unstable under the conditions prevailing at the site. Under these conditions, minerals exposed to the elements undergo hydration, hydrolysis, dissolution, and oxidation–reduction. Ions that go into solution, if not washed out to sea, seek new equilibria to form secondary or pedogenic minerals.^[1,2] In dry regions, one of the first secondary minerals to form is calcite. As rainfall and leaching intensity increase, sodium (Na), potassium (K), Ca, and magnesium (Mg) leave the system and secondary, layered aluminosilicates become more prominent. And with time and intense leaching, desilication transforms even the secondary silicates to metal oxides and hydrated oxides. It is no accident that bauxite deposits are more common in the tropics than elsewhere in the world. An example of concentration of aluminum through weathering is illustrated by the data in [Table 1](#).

Translocations

Termites, earthworms, and burrowing animals move huge quantities of soil material to the surface. Surface soil materials can also fall between cracks in soils that shrink and fracture when dried. Repeated wetting and drying result in churning and inversion of the soil creating surface patterns called gilgai. These inverting soils are aptly named Vertisols. Particles suspended in water can also be transported to greater depth and deposited on walls of water-conducting pores. The wall coatings are called argillans or clay skins and can significantly increase the clay content of the subsoil horizons called argillic horizons.

Additions

The addition of water to soils is taken for granted, but water enables most other soil processes to occur. Water is

Table 1 Chemical compositions of (A) melilite-mepheline basalt and (B) soil formed from it.

Chemical composition	%	Chemical composition	%	Chemical composition	%	Chemical composition	%
(A) <i>Melilitite-mepheline basalt</i> ^a							
SiO ₂	35.85	MgO	13.98	CO ₂	0.09	BaO	0.16
Al ₂ O ₃	9.76	CaO	13.78	TiO ₂	3.68	H ₂ O+	3.59
Fe ₂ O ₃	5.09	Na ₂ O	2.22	P ₂ O ₅	1.05	<i>Total</i>	99.76
FeO	9.59	K ₂ O	0.83	MnO	0.10		
Chemical composition	Depth (cm)						
	0–38	38–48	48–75	75–100	100–155		
(B) <i>Soil</i> ^b							
SiO ₂	8.5	2.9	1.0	0.4	0.4		
Al ₂ O ₃	31.3	20.6	20.8	28.5	30.1		
Fe ₂ O ₃	45.5	48.2	46.4	42.3	41.3		
MgO	0.0	0.0	0.0	0.0	0.0		
CaO	0.0	0.0	0.0	0.0	0.0		
Na ₂ O	0.6	0.7	0.7	0.7	0.0		
K ₂ O	0.23	0.23	0.1	0.0	0.0		
TiO ₂	8.7	8.0	8.3	7.3	6.8		
P ₂ O ₅	0.70	0.63	0.60	0.61	0.64		
MnO	0.07	0.07	0.04	0.09	0.08		
H ₂ O+	5.0	19.7	20.5	20.7	20.2		
<i>Total</i>	100.1	100.4	97.8	100.0	99.5		

Source: ^aFrom MacDonald, Davis, et al.^[3] and ^bSoil Conservation Service, U.S. Department of Agriculture.^[4]

necessary for life and the production of soil organic matter. Weathering that may begin as a geochemical reaction soon turns into a biogeochemical process with the first appearance of lichens on rock surfaces. Additions of dissolved gases and electrolytes play critical roles in soil formation. Rainfall also scrubs particulate matter from the atmosphere. Tropospheric dust from Asia containing quartz and mica is deposited over the northern Pacific, but its concentration on land is highly dependent on rainfall.^[3] Close to their sources, aerosolic dusts can form thick loess deposits.

Losses

One soil's loss can be another soil's gain. Wind erosion that generates aerosolic dust exposes underlying materials to the elements to renew the weathering processes. Landslides and water erosion do the same and create new parent material downstream.

Less obvious soil losses occur through solution in drainage waters as evidenced by the salty sea. Desilication or leaching of silica from soils into streams and oceans is evident in the data presented in Table 1. Less obvious are gaseous losses through microbial activity. Carbon dioxide (CO₂), methane, and nitrous oxide fluxes into and out of

soil are measured with renewed interest because of their role as greenhouse gases.

SOIL-FORMING FACTORS

On a geologic timescale, soils are transient bodies on the landscape. What we see today may have little resemblance to what was there in the beginning and what will remain in the end. The birth of a soil might begin with the solidification of molten lava to form solid rock or the exposure of seafloor from geologic uplift. The end could come with a landslide or during accelerated soil erosion on a cultivated hillside. But between beginning and end, major chemical transformation usually occurs so that the end product would be unrecognizable from the parent rock. In addition to time and the parent rock, three additional factors operate over time to produce differences in the soil chemical composition.^[5] These three factors are climate, topography, and biota. All five of these factors have been discussed earlier, but will now be covered with emphasis on soil chemical composition.

Parent Rock

One would expect the chemical composition of the parent rock to have a significant effect on the chemical

composition of the soil that forms from it. It does—but the departure from initial rock composition is generally greater than one would expect (see Table 1). This departure from parent rock composition occurs because soils are open systems that gain new substances and gradually lose their original minerals primarily through erosion, dissolution, and leaching. Rock-forming minerals, however, differ in their susceptibility to weathering.^[6] Acid igneous rocks containing K feldspars, muscovite, and quartz tend to be more resistant to weathering than basic igneous rocks containing plagioclase feldspars, biotite, and amphiboles. Igneous rock erupted as lava or tefra having identical chemical compositions can and will most likely weather into very different chemical end products. Tefra consisting mainly of glassy particles tends to weather more rapidly than lava rock and to develop into soils that fall in the unique Andisol soil order.

Time

Mohr,^[7] a Dutch scientist investigating soils in Indonesia during colonial times marveled at the soils he found there, and to organize his findings, categorized soils as being juvenile, mature, or senile. Mohr’s juvenile soils still retained features of their parent rock and had not yet developed distinct features that set them apart from other soils. The mature soils were those favored by farmers. In mature soils, the parent rock had been weathered to clay, but nutrient elements such as Ca, Mg, K, and phosphorus were still plentiful. But what intrigued Mohr most were the senile soils. Prolonged weathering under warm and humid conditions had stripped the soils of life-sustaining nutrients and rendered them unsuitable for farming. What remained on the landscape were the insoluble residues of iron and aluminum oxides along with resistant rock-forming minerals such as quartz. Senile soils are more likely to occur near the equator because soil changes faster under warm tropical climates.

Climate

The influences of climate on soil chemical composition can best be observed along steep climate gradients located on a single parent rock of known geologic age. Such conditions occur on the Hawaiian Islands, and researchers are taking advantage of them to study the role of rock age,^[8] Island age,^[9–11] and climate.^[12] The islands offer temperatures associated with balmy beaches to snow-capped mountains and precipitation that range from rainforests to deserts, all compressed in a small area. A combination of steep rainfall gradient, warm tropical climate, and a basaltic parent rock containing easily weatherable minerals contributes to large soil chemical differences over short distances and in a relatively short period. The data in Tables 2 and 3

Table 2 Effect of climate on chemical composition of two soils developed from volcanic ash: (A) in a dry zone (mean annual rainfall: 800 mm; mean annual temperature 17°C) and (B) wet zone (mean annual rainfall: 3400 mm; mean annual temperature: 18°C).

Chemical composition	Depth (cm)				
	0–13	13–25	25–50	50–63	125–163
(A) Dry zone ^a					
SiO ₂	32.3	32.9	36.1	38.6	55.7
Al ₂ O ₃	21.1	23.9	24.6	24.0	21.1
Fe ₂ O ₃	14.5	15.0	15.4	15.1	4.8
MgO	3.30	2.86	2.03	2.07	0.77
CaO	3.44	3.55	3.45	3.44	1.68
Na ₂ O	1.76	2.24	2.26	2.25	4.19
K ₂ O	0.80	0.87	0.88	0.82	2.66
H ₂ O ⁺	18.3	14.7	11.7	10.0	7.6
Chemical composition	Depth (cm)				
	0–18	18–35	35–50	50–63	148–163
(B) Wet zone ^b					
SiO ₂	13.1	12.2	11.9	10.5	13.1
Al ₂ O ₃	17.5	20.6	24.6	24.0	22.8
Fe ₂ O ₃	23.1	24.3	19.5	21.4	29.7
MgO	0.50	0.24	0.29	0.27	trace
CaO	0.17	0.08	0.0	0.0	0.0
Na ₂ O	1.11	0.05	0.03	trace	0.10
K ₂ O	0.71	0.70	0.46	0.38	0.72
H ₂ O ⁺	39.0	36.1	37.1	36.1	30.2

Source: ^{a,b}From Soil Conservation Service, U.S. Department of Agriculture.^[4]

illustrate how differences in climate can produce soils of very different compositions and properties from identical parent rock.

Topography

Soil composition is affected by topography primarily through drainage and hydrology. Soil in depressions tend to be wetter, cooler, and richer in organic matter, clay, and bases. These differences occur naturally but are amplified by cultivation. Soils on high points on the landscape tend to be net losers of fine particles and dissolved matter, whereas bottom lands tend to be net accumulators of the same. Soils in between form a toposequence of composition that is repeated in a predictable pattern over the landscape. Topography and drainage control wetness, oxygen supply, oxidation–reduction potential, leaching, and pore water composition. All of the above affect biological diversity and activity and the biogeochemistry of the system. Soil chemical compositional differences arising from topographic differences are the easiest to recognize in the landscape.

Table 3 Influence of climate on soil chemical properties as illustrated by a difference in (A) and (B) corresponding to soil (A) and (B) in Table 2.

Soil property	Depth (cm)				
	0–13	13–25	25–50	50–63	125–163
(A) <i>Dry zone</i>					
Organic carbon (%)	7.53	3.69	1.88	1.02	0.11
Nitrogen (%)	0.32	0.16	0.11	0.09	–
C/N ratio	24	23	17	11	–
Extractable Ca (cmol/kg)	27.9	28.0	32.6	34.6	33.3
Extractable Mg (cmol/kg)	7.8	6.4	7.1	9.8	14.4
Extractable Na (cmol/kg)	0.40	0.40	0.80	1.10	6.10
Extractable K (cmol/kg)	4.80	5.60	4.80	3.00	1.30
CEC (cmol/kg)	55.2	51.6	50.2	51.7	47.7
pH (H ₂ O)	6.6	7.1	7.3	7.5	8.2
pH (INKCl)	5.7	6.2	6.2	6.3	6.9
Soil property	Depth (cm)				
	0–18	18–35	35–50	50–63	148–163
(B) <i>Wet zone</i>					
Organic carbon (%)	11.70	6.55	9.36	8.49	3.33
Nitrogen (%)	0.90	0.45	0.44	0.43	0.31
C/N ratio	13	15	21	20	11
Extractable Ca (cmol/kg)	9.4	0.8	0.5	trace	trace
Extractable Mg (cmol/kg)	2.6	0.3	0.3	0.1	0.6
Extractable Na (cmol/kg)	0.3	0.2	0.2	0.2	0.2
Extractable K (cmol/kg)	0.5	0.2	0.1	0.2	0.2
CEC (cmol/kg)	53.1	33.7	39.3	29.9	14.4
pH (H ₂ O)	5.4	5.2	5.4	5.4	5.2
pH (INKCl)	4.5	4.5	4.7	5.0	5.1

Biota

Plants clearly have an effect on mineral weathering.^[13] How different would a geochemically formed soil be from one formed biogeochemically? The presence or absence of organic matter would make a difference, but removing biota and organic matter from soils as we know them would not approximate one developed without living organisms. Not only do soil organisms such as termites and burrowing animals move huge quantities of soil material horizontally and vertically, but also plants produce residue that serves as an energy source for other organisms and recycles inorganic nutrients. The organic residues also act as cementing agents to hold clay particles into stable compound particles or peds. It is unlikely that a soil can withstand the erosive impact of raindrops without the cementing action of humus or the high infiltration rate soils acquired through the formation of large water-conducting pores

between large water-stable peds. Organic matter also chelates metal ions such as iron and aluminum and facilitate their downward movement, but it is their contribution to the physical and hydraulic properties of soils that biota influence long-term stability of soil composition.

CONCLUSIONS

Soil chemical composition is the product of the interactions of climate, topography, and biota operating over geologic time on a parent rock that is open to gains and losses of matter. Physical disintegration and chemical weathering transform existing unstable minerals into secondary or pedogenic minerals that may differ in kind and composition with depth in the profile. Soil chemical composition is important only in so far as it can be related to the behavior and performance of soils.

REFERENCES

1. Garrels, R.M.; Christ, C.L. *Solutions, Minerals, and Equilibria*; Harper and Row: New York, 1965.
2. Jackson, M.L.; Level, T.W.M.; Syers, J.K.; Rex, R.W.; Clayton, R.M.; Sherman, G.D.; Uehara, G. Geomorphological relationships of tropospherically derived quartz in the soils of the Hawaiian Islands. *Soil Sci. Soc. Am. Proc.* **1971**, *35*, 515–525.
3. MacDonald, G.A.; Davis, D.A.; Cox, D.C. *Geology and Ground-Water Resources of the Island of Kauai, Hawaii*; Hawaii Division of Hydrography: Honolulu, 1960.
4. Soil Conservation Service, U.S. Department of Agriculture. *Soil Survey Laboratory Data and Descriptions for Some Soils of Hawaii*. Soil Survey Investigations Report No. 29, HAES and HSPA: Honolulu, 1976.
5. Jenny, H. *Factors of Soil Formation: A System of Quantitative Pedology*; Dover: Mineola, 1941.
6. Barshad, I. Chemistry of soil development. In *Chemistry of the Soil*; Bear, F.E., Ed.; Reinhold: New York, 1964; 1–70.
7. Mohr, E.C.I. Tropical soil forming processes and the development of tropical soils. In *National Geological Survey of China*; National Geological Society of China: Peiping, 1930.
8. Chororor, J.; DiCharo, M.J.; Chadwick, O.A. Structural charge and cesium retention in a chronosequence of tephritic soils. *Soil Sci. Soc. Am. Proc.* **1999**, *63*, 169–177.
9. Vitousek, P.M.; Chadwick, O.A.; Crews, T.E.; Fownes, J.H.; Hendricks, D.M.; Hebert, D. Soil and ecosystem development across the Hawaiian Islands. *U.S.A. Today* **1997**, *7*, 1–8.
10. Kennedy, M.J.; Chadwick, O.A.; Vitousek, P.M.; Derry, D.A.; Hendricks, D.M. Changing sources of base cations during ecosystem development, Hawaiian Islands. *Geology* **1998**, *26*, 1015–1018.
11. Chadwick, O.A.; Derry, L.A.; Vitousek, P.M.; Huebert, B.J.; Hedin, L.O. Changing sources of nutrients during four million years of ecosystem development. *Nature* **1999**, *397*, 491–497.
12. Chadwick, O.A.; Olson, C.G.; Hendricks, D.M.; Kelly, E.F.; Gavenda, R.T. *Quantifying Climatic Effects on Mineral Weathering and Neof ormation in Hawaii*, Proceedings of the 15th International Soil Science Congress 8a; Acapulco: Mexico, 1994; 94–105.
13. Kelly, E.F.; Chadwick, O.A.; Hilinski, T. The effect of plants on mineral weathering. *Biogeochemistry* **1998**, *42*, 53–72.

Chemisorption

N.J. Barrow

Mt. Claremont, Western Australia, Australia

Abstract

Many substances react with soil. Unless precipitation of ions from solution is involved, they initially react with the outside of soil particles in a process referred to as *adsorption*. When reaction continues and involves diffusion of the reactant into the interior of the soil particles, we use the more general word *sorption* to cover the whole process: adsorption plus diffusion. For some reactants, the attraction to the soil particles is largely electrical, e.g., the reaction of K^+ with a negatively charged surface. For other reactants, a strong inherent or chemical attraction is involved. This attraction may be so strong that ions overcome electrical repulsion and react with like-charged surfaces, e.g., the reaction of phosphate (P) ions with negatively charged surfaces. We use the word *chemisorption* to describe such reactions. Many nutrients such as P, borate, copper, and zinc are chemisorbed by soil, and this restricts their supply to plants. Similarly, many pollutants, including selenite, arsenate, mercury, and lead, are chemisorbed, and this slows their entry into the food chain.

SPECIFICITY AND SORPTION

It follows that chemisorption involves specificity. That is, an ion may be sorbed despite the presence of a large excess of some other like-charged ion in solution. An example is sorption of phosphate (P) from a solution containing chloride or nitrate. However, specificity does not always involve chemisorption. Preference may be a matter of valency or of differences in the tendency of ions to retain a sheath of water molecules. The terms also differ in that *specific absorption* describes an observation, whereas chemisorption requires some interpretation.

The strength of the chemical bond involved in chemisorption varies according to the ions involved. It therefore follows that specific adsorption is a graded property, with particular ions falling along a gradient of specificity.

CHEMICAL BONDING

For chemisorption to occur, there must be an attraction between the ion in solution and atoms on the surface of the soil particles. This attraction may show itself in other ways; e.g., both iron (Fe) and aluminum (Al) form low-solubility compounds with P. Similarly, metals, such as zinc (Zn), form hydroxides with low solubility. This has suggested that precipitates are important in the soil chemistry of such ions. However, discrete compounds are only formed when concentrated solutions leach out of fertilizer particles and have a limited life in soil. The ions eventually react with atoms, or groups of atoms, on the surface of soil particles: P and several other anions with Fe and Al atoms; and Zn and other metals with hydroxyls or with organic complexes.

SOIL COMPONENTS INVOLVED

Several kinds of soil particles may have appropriate groups of atoms on their surfaces. These include the edges of clay minerals, organic matter, and metal oxides. Fe oxides are the most widespread, with goethite being the most important and most frequently studied.

Even though chemisorption involves chemical bonds, these bonds must form between charged particles (ions) and charged surfaces. Two characteristics of the surfaces are important in this respect. One is that the charge on the surfaces varies with pH. In the case of metal oxides, this is because the metal atoms on the surface attempt to complete their electron shells by reacting with water molecules. These molecules lose or gain protons depending on the pH and the concentration of all ions in the solution in which reaction occurs. They are variable-charge surfaces. The charge is also variable in the spatial sense. Different particles, or different patches on the same particles, may carry different charge densities and therefore have different potentials. Direct evidence for this is that it is possible to measure both positive and negative charges in the one soil.

CHARACTERISTICS OF THE REACTING IONS

Two characteristics of the reacting ions are also important. First, many of the relevant acids have pK values in or near the range of soil pH values. Changes in pH therefore result in large changes in the ions present, with consequent effects on the reaction. Second, when the ions react with the surface, their charge must be conserved. Some of the charge is conveyed to the surface for reaction with P; the surface becomes more negative. The remainder of the charge is

returned to the solution for reaction with P; hydroxide ions may be desorbed (or protons adsorbed). The balance between these two options changes in a complex way with changes in pH and in the amount of sorption. Because the surface charge is consequently more negative, the affinity of the surface for more P ions decreases. Analogous effects occur for other ions.

This brief summary equips us to comprehend a great deal of the information on chemisorption; e.g., the Langmuir equation, although widely used to describe the relationship between sorption and concentration, is appropriate only in special circumstances. Soils do not provide one (or even two) uniform surfaces, and the feedback effect of the change in charge negates a basic assumption of this equation. A more realistic approach is to accept the range of surface characteristics and to apply a distribution to describe them. This distribution can be divided into slices, and a Langmuir-like equation that takes feedback into account can be applied to each slice. The net behavior is then given by the sum of the slices.

EXPLAINING THE EFFECTS OF CONCENTRATION

Consider how this model describes the most common observation in sorption studies: i.e., the relationship between sorption and concentration. In many studies, sorption curves are produced over a limited range of concentrations, about 100-fold. Over such a range, the Freundlich equation usually describes the data well. This means that a plot of log sorption against log concentration approximates a straight line. However, over a greater range, such as that illustrated in Fig. 1, curvature is obvious and is reproduced by the model outlined above. This curvature is inevitable because, at low concentrations, sorption is confined to the

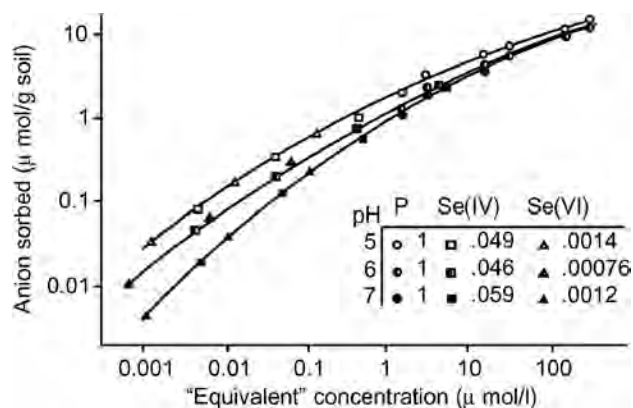


Fig. 1 Sorption of P, Se(IV), and Se(VI) at pH values 5, 6, and 7, respectively, plotted on common axes. The concentrations were multiplied by the values in the table to give equivalent concentrations. Thus, at pH 5, the value for phosphate is 1; for Se(IV), 0.049; and for Se(VI), 0.0014.

Source: From Barrow.^[1]

most favored end of the spectrum of sites and is therefore on a nearly uniform set of sites. Further, because the amount of sorption is small, the feedback effect is insignificant. This is one of the unusual situations in which the Langmuir equation is appropriate. However, at very low concentrations, the form of the equation is such that sorption is virtually proportional to concentration. That means that on a log-log scale, the slope approaches unity. It is, therefore, much steeper than at higher concentrations.

EXPLAINING THE EFFECTS OF pH

Consider the interactions between pH and the concentration of the background solution on the sorption of some common inorganic anions (Fig. 2).

These interactions occur because there is an initial adsorption reaction of ions on a variable-charge surface. An increase in salt concentration increases the number of counter ions close to the surface and therefore changes the way that electric potential changes, as we move away from the surface. At any particular distance from the surface, an increase in salt concentration decreases the absolute value of the electric potential. For example, when a reaction is with a negatively charged surface, the potential becomes less negative and anion sorption increases. A convenient, though less precise, way of thinking about this is that an increase in the cation concentration near a negatively

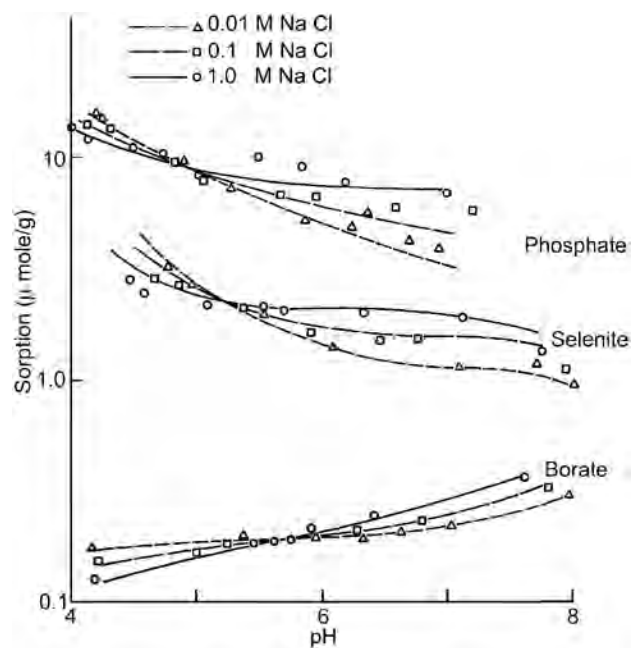


Fig. 2 The effect of pH and background electrolyte concentration on the sorption of phosphate, Se(IV), and borate by a soil. In each case, sorption was measured at a solution concentration of 100 μ M. The lines were obtained by fitting the data into the model described in the text.

Source: From Barrow.^[1]

charged surface makes it easier for anions to react with the surface. The corollary is that, when increasing salt concentration increases the sorption of anions, the reacting surface must be negatively charged. In such cases, there must be a clear chemical component to sorption. By a similar argument, when sorption of anions is decreased by an increase in salt concentration, the reacting surface is positively charged. The results in Fig. 2, therefore, show that the reaction of borate, selenite [Se(IV)], and P at medium to high pH occurred with a negative surface and reaction at low pH occurred with a positive surface. Thus, all three species reacted with a variable-charge surface.

BEHAVIOR OF SOME IONS

Before considering the ions in detail, note that selenate [Se(VI)], sulfate, P, Se(IV), and arsenate all form bidentate, inner-sphere complexes with the surface. That is, two of the oxygen atoms provide direct chemical links to the surface atoms. When a reactant forms a bidentate link to the surface, it is appropriate to refer to the divalent ion in solution.^[1] Similarly, in the case of a monodentate link, as with borate, it is appropriate to refer to the monovalent ion.

Boric acid is fairly weak, with pH of about 9. Therefore, in the normal range of soil pH, the proportion of monovalent borate ion increases 10-fold for each unit increase in pH. The effects of pH on the charge and potential and the effects on acid dissociation therefore oppose each other: the increasingly negative electric potential favors decreased desorption; the increasing dissociation favors increased adsorption. Because the ion is monovalent, the effects of surface charge are not quite large enough to exceed the effects of the increasing value of the dissociation term. Thus, sorption increases with increasing pH.

Selenious acid is a diprotic acid with pK_1 at 2.7 and pK_2 at 8.5 in very dilute solution. The main species present in the range of soil pH values are $HSeO_3^-$ and SeO_3^{2-} , with the divalent ion increasing with increasing pH. However, because the relevant ion is divalent, the effects of the increasingly negative charge exceed the effects of increasing dissociation, and sorption therefore decreases with increasing pH.

Phosphoric acid is a triprotic acid, but the third dissociation (pH_3) is well beyond the range of soil pH. The species present in soil solutions are therefore $H_2PO_4^-$ and HPO_4^{2-} . The divalent ion increases 10-fold for each unit increase in pH up to just below pH_2 , which is about 7. Its behavior is similar to that of Se(IV), and the explanation is analogous.

EXPLAINING THE EFFECTS OF TIME

We have thus far ignored the effects of the reaction period. However, many reactants continue to react with soil, and also with goethite, for a very long period. This continuing reaction is of great importance in agriculture where it has a

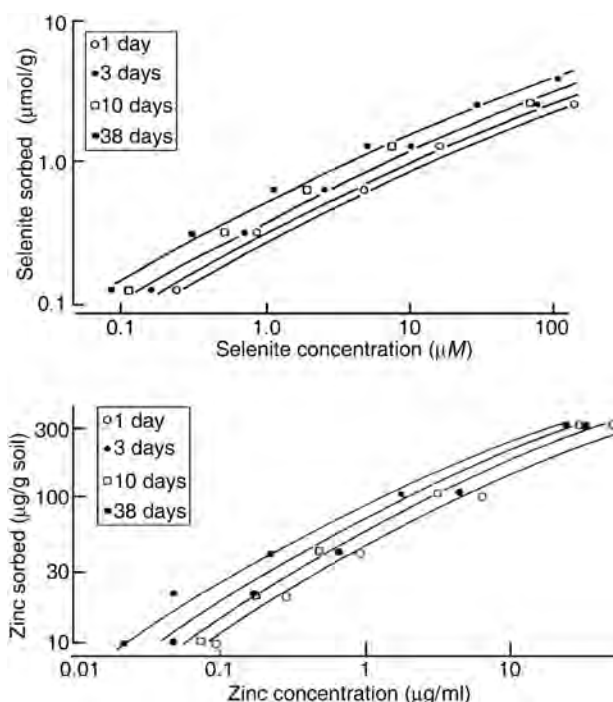


Fig. 3 The effect of period of reaction at 25°C for Se(IV) and Zn on the relationship between solution concentration and amount of sorption. The lines were obtained by fitting the model described in the text to the data.

Source: From Barrow.^[1]

large effect on the need to reapply fertilizer to land. Consequently, sorption is better described by a surface, with time as the extra dimension, rather than by a line (Fig. 3).

Ions differ in the extent to which they continue to react, so that one can produce a sequence as follows: $P > Se(IV) > \text{fluoride} > Se(VI)$. For the anions, the greater the chemical attraction for the surface, the more marked the continuing reaction. The continuing reaction is caused by the diffusion of adsorbed ions into the absorbing particle. This is a slow process but can be accelerated by raising the temperature. Consequently, increased temperature increases sorption and/or decreases solution concentration.

DESORPTION

The more marked the continuing reaction, the more deeply buried the ion becomes. Although desorption curves do not seem to follow the same track as sorption curves, they can be fully predicted if the continuing reaction is taken into consideration.

REFERENCE

1. Barrow, N.J. The four laws of soil chemistry: The leeper lecture 1998. *Aust. J. Soil Res.* **1999**, 37, 787–830.

Chernozems

Boris Boincean

Department of Sustainable Farming Systems, Selectia Research Institute of Field Crops, Balti, Republic of Moldova

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Chernozems are the most fertile soils of the world. Besides having higher stocks of soil organic matter (SOM), they have a granular soil structure of the humus layer with high base saturation and a calcareous subsoil. The discrepancy between the conditions of soil formation under natural ecosystems and agroecosystems has caused several types of degradation in Chernozems: dehumidification, destructuration, compaction, accelerated soil erosion, etc. Consequently, agronomic productivity of crops has decreased. Nevertheless, Chernozems are highly resilient because of inherent physical and mineralogical properties, thickness of topsoil, and the mechanisms of self-restoration similar to that of a living organism. Permanent recycling of SOM is a determinant of soil functionality and its capacity to provide ecosystem and social services. Restructuring of modern industrial farming systems by using natural ecosystems as models of agricultural intensification can contribute to soil fertility restoration, increasing the competitiveness of agricultural producers, resilience to global warming, and stability of rural communities.

INTRODUCTION

Chernozems occupy 240 million hectare (Mha) in the steppes of Eurasia and North and South America. They are the most fertile soils of the world.

Chernozems are formed under steppe vegetation. Under natural ecosystems, the amounts of energy and nutrients removed from the soil are equivalent to the amount returned to the soil, creating a dynamic equilibrium and closed cycling of energy and nutrients. Plowing of Chernozems, along with replacement of perennial by annual vegetation, has drastically changed the turnover of soil organic matter (SOM). Because the input of energy with crop residues and manure is significantly lower than the amount of energy harvested with the aboveground biomass and that in uncompensated mineralization losses of SOM, the system has a strong negative energy budget and is not sustainable. Dehumidification of Chernozems has many other negative consequences on soil quality—destructuration, compaction, soil erosion, droughts, etc. Degraded Chernozems can't provide good ecosystem and social services, and their productivity is in a continuous decline.

Soil productivity is determined by its functionality. The latter depends on soil biodiversity of the whole trophic chain. The ratio between fresh organic matter (raw organic matter) and the effective and stable humus changes the physical, chemical, and biological functions of SOM. Regular additions and adequate amount of the fresh organic

matter of good quality to compensate loss by annual mineralization of SOM are essential strategies for creation of healthy and climate-resilient soils.

Agriculture requires, more than ever before, the ecological and biological restructuring of farming systems. Technological modernization of agriculture, without taking into consideration of soil as a living organism, cannot lead to a sustainable agriculture. Under these conditions, soil fertility is depleted more rapidly than it can be restored. Soils are also a non-renewable natural resource on the human time scale. Thus, agricultural intensification should be based on new agroecological principles for sustainable soil and farm management.

CHERNOZEMS

Chernozems are rich in SOM. The depth of humus layer ranges from 30 to 160 cm, and the stocks of SOM from 200 up to 650–700 Mg/ha. Thus, Chernozems are very fertile soils.

Chernozems can be grouped according to the following criteria:

- a. Depth of humus layer:
 - Shallow: <40 cm
 - Medium: 40–80 cm
 - Deep: 80–120 cm

b. Humus concentration:

- Low: <6%
- Medium: 6–9%
- High: >9%

Several theories have been proposed regarding the genesis of Chernozems: marine, marshy, and terrestrial. It was at the end of the 19th century when the famous Russian scientist V. V. Docucaeav published the book *Russian Chernozem*.^[1] He documented that the formation of Chernozem is a result of decomposition of steppe grasses, but not of forest vegetation, by a close interaction between climate, duration of soil formation, vegetation, and relief of the place and parental material. Chernozems are developed under the influence of meadow grasses and associated steppes.

Agriculture, in order to be productive and sustainable, must take into consideration all these factors and account for soil biological diversity, functionality, and productivity.

The hydrothermal conditions in the Chernozem regions are favorable to the decomposition of crop residues with the formation of humic substances. The most favorable conditions for Chernozem formation are observed in the forest-steppe region (podzolized, leached, and typical Chernozems), with more abundant vegetation, better drainage, and higher rates of humidification. Under the steppe conditions, both drainage and abundance of vegetation are not so favorable as in the previous zone. Further, the root system doesn't penetrate deep into the solum, and rates of humidification are low (calcareous, common, southern Chernozems).

Besides high stocks of SOM, Chernozems also have a granular structure of the humus layer with high base saturation and a calcareous subsoil. One of the predominant characteristics of Chernozems is the formation of organo-mineral constituents rich in nitrogen (N), phosphorus (P), sulfur, calcium, and other elements, including humate of calcium.

Chernozems occupy 240 million hectare (Mha) in the steppes of Eurasia and North and South America. Eurasia includes some western and south eastern countries of Europe up to Mongolia and China. The North American region includes prairies of Midwestern United States and southern provinces of Canada. South American continent includes regions in the southern parts of Argentina and pre-mountainous southern parts of Chile.

The Chernozem family includes different subtypes of Chernozems. In more humid regions, Chernozem grade into dark gray forest soils or Phaeozem in the World Reference Base. These soils lack the calcareous subsoil. Phaeozem occupy ~190 Mha in the forest steppes in Europe, Northern China, the prairies of North America, and pampas of South America.

A Chernozem profile is characterized by a dark humus A horizon. Virgin Chernozems (unplowed) under steppe vegetation have the A0 horizon derived from steppe grass residues. Some Chernozems also have combined humus

A + B1 horizon, where A is the most humidified horizon (the black earth topsoil) and B1 is the lower part of the humus layer (transition horizon). The other B2 horizon can exist as a separate horizon, with residual humic substances. Beneath the humus layer, there lies a calcareous horizon, which leads into the transition to parental material (C horizon).

The modern concepts regarding genesis of Chernozem are based on specific characteristics of biological turnover of energy and nutrients under steppe vegetation. Plowing of Chernozem can strongly alter the conditions of soil formation, which are discussed below in the entry.

COMPOSITION AND PROPERTIES OF CHERNOZEMS

Among inherent characteristics of Chernozems derived from the parent material include the particle size distribution or texture. It provides the physical framework and largely determines the soil's capacity to supply water and nutrients and to develop soil structure and its resistance to erosion. The textural triangle is commonly used to set the limits for each textural class (used by the USDA–NRCS, Soil Survey Division). In this method, the textural assessment is based on the relative proportions of clay, silt, and sand in a soil sample.

Majority of Chernozems are clay loams and heavy clay loams comprising a mixture of the finest clay particles, middle-sized silt particles, and coarse sand particles. The texture of Chernozem, the water-stable microstructure, the water-holding capacity, permeability, and resistance to erosion are important attributes of Chernozems. The most stable aggregates are formatted under grasslands in the presence of sufficient clay, fresh organic matter, and calcium carbonates.

Mineralogy of Chernozems depends on the parent material. Chernozems contain feldspar, mica, and illite, which are rich in potassium. That's why the effectiveness of potassic fertilizers on Chernozems is low. The clay mineralogy and colloidal humus of the Chernozems provide a high adsorptive capacity and buffering capacity or the ability to resist degradation processes.

The reaction of Chernozems is slightly acidic or neutral (pH 6.0–7.5). Chernozems are well drained with low salinity, but secondary salinization can result from irrigation with saline water. The dominant anions in the soil solution are bicarbonates in the upper layers and sulfates and chlorides in the sub-soil.

Chernozem also contains high SOM and total N concentrations. Accumulation of nitrates in soil profile can lead to their leaching, especially in a short period of time in the spring, when the evaporation is lower than the rainfall.

Soil is a living organism. The amount of organisms per 1 ha of virgin Chernozems consists of large biodiversity:^[2–7]

- 3 Mg of bacterium,
- 3 Mg of microscopic fungi,
- 1.5 Mg of actinomycetes,
- 100 kg of algae,
- 100 kg of protozoa,
- 500 kg of earthworms (their population can reach more than 300 Mg/ha),
- 50 kg of nematodes,
- 40 kg of arthropods,
- 40 kg of molluscs,
- 20 kg of reptiles and rodents, etc.
- Each cubic centimeter of soil contains 7–10 millions of microorganisms.

The diversity of life in the soil determines the functionality and the fertility of Chernozems. The biological aspect of soil fertility was neglected in agronomical science in favor of chemical fertilizers.

Charles Darwin in his monograph^[8] emphasized the importance of earthworms in ingesting and processing the soil, which is passing many times through the intestines of worms. Their contribution to increased soil porosity, aeration, water infiltration and drainage, water-stable aggregates, etc. is crucial to natural soil formation, especially in arable lands. Soil structure, bulk density, and porosity are relatively stable under natural vegetation but degrade under cultivation.

The texture, mineralogical, and chemical composition of soil are generally stable. The most changeable component of soil is SOM in spite of relatively low amount (3–6% by weight of cultivated soils). First and foremost, sustainable management of soil involves judicious management of SOM in cultivated soils. Synthesis and decomposition of SOM are critical to high soil fertility, including that of

Chernozems. In natural ecosystems, the amount of energy and nutrients removed from the soil are equivalent to the amount returned back to the soil that establishes a dynamic equilibrium and closed cycling of energy and nutrients. Turnover of the more easily decomposed fractions of SOM takes only a few months to a few years, but turnover of recalcitrant fractions occurs over hundreds or thousands of years. By replacing perennial vegetation by annual vegetation, there has been a drastic reversal of the natural accumulation of C and N in soils.

The analytical data obtained from a long-term field experiment with crop rotations and permanent crops at the Selectia Research Institute of Field Crops^[9] indicate a large deficit of energy in crop rotations with higher percent of row crops and permanent crops, especially with the so-called black fallow (Table 1).

A large part of biomass energy accumulated by the crops is harvested in plant/animal product from the field and lost from the system. Because the input of energy with crop residues and manure is significantly lower than that harvested in the aboveground biomass and the energy in uncompensated mineralization losses of SOM, the energy balance is strongly negative. Consequently, agriculture can't be sustainable in a system with perpetual energy deficit, just as is the case with every living organism. In several scenarios, losses of SOM by accelerated erosion exceed those by mineralization. In such situations, it is important to take into consideration the erosional losses of SOM.

In Chernozems, accelerated soil erosion and droughts are two sides of the same coin. It is a paradox that drought is one of the limiting factors in Chernozems, yet the abundance of water during seasons of torrential rains causes a significant degradation of soil. Because of the discrepancy between soil formation in natural ecosystems and soil

Table 1 The annual balance of energy in the long-term field experiments at Selectia Research Institute of Field Crops, average for 30 years (10³ Mdj/ha).^[9]

Variants			Output			Input			Balance	
			Taken up by aboveground biomass	Non-compensated deficit of SOM	Total	With crop residues	With manure	Total	Mdj/ha	Annual energy deficit (%)
Crop rotations, % of row crops	40 (with perennial leguminous crops)		99.3	10.4	109.7	64.8	1.9	66.7	-43.0	39.2
	50		95.2	12.4	107.6	50.4	0.3	50.7	-56.9	52.9
	60		116.9	11.5	128.4	60.2	3.7	63.9	-64.5	50.2
	70		112.6	11.5	124.1	63.6	2.4	66.0	-58.1	46.8
Continuous crops	Corn for grain	Unfertilized	100.4	20.0	120.4	35.8	—	35.8	-84.6	70.3
		Fertilized	141.0	16.8	157.8	51.1	4.4	55.5	-102.3	64.8
	Winter wheat	Unfertilized	57.4	20.0	77.4	21.0	—	21.0	-56.4	72.9
		Fertilized	80.3	15.2	95.5	29.4	4.4	33.8	-61.7	64.6
Black (bare) fallow	Unfertilized			32.7	32.7	0	0	0	-32.7	100
	Fertilized			26.9	26.9	0	4.4	4.4	-22.5	83.6

management in agroecosystems, Chernozems are degraded by numerous processes. In general, natural ecosystems should serve as a model for sustainable management of agroecosystems. Predominant use of the industrial model of intensification of agroecosystems by using inputs from non-renewable sources of energy and their by-products has exacerbated the degradation of Chernozems and accentuated the danger of disappearance of Chernozems over a long-term period. Thus, Chernozems may be termed endangered soils.^[10–13]

Regular and long-term addition of sufficient quantities of fresh organic matter is crucial to enhance biodiversity and soil functionality. It is important to understand that soils in natural conditions are always covered with vegetation but in agroecosystems they remain uncovered and are tilled for a longer or shorter period of time, depending on the crops.

Several researchers^[14–16,17] have documented that natural ecosystems have many other advantages relatively to agroecosystems: a more complex trophic interaction, a higher species and genetic diversity, a closer turnover of energy and nutrients, a higher stability and longer sustainability over time, etc.

Intensive farming practices widely adopted since 1970s included excessive soil tillage by moldboard plow; monoculture or narrow specialized crop rotations; indiscriminate use of synthetic fertilizers, especially of N; chemical pest, weed, and disease control; excessive irrigation by flooding; and manipulation of plant genomes with inappropriate root and shoot characteristics. All these practices have contributed to physical, chemical, and biological degradation of cultivated Chernozems. Among the widespread processes of soil degradation are destructure, dehumidification, compaction, accelerated soil erosion, etc.^[11,18]

Destructure (degradation) means deterioration of granular structure of Chernozems because of excessive and long-term soil tillage. For example, in virgin typical Chernozems of the Balti steppe, Republic of Moldova, the dominant fraction of water stable aggregates is 1–5 mm. Soil macroaggregates (>10 mm) and microaggregates (<0.25 mm) are negligible. In contrast, aggregate size distribution is just the opposite in cultivated Chernozems. For example, the data of dry sieving analyses for aggregate size distribution of experiments from Moldova indicate that 40–70% of soil aggregates are >10 mm in the range of clods. Furthermore, the layers beneath the plow layer (25–35 cm) are highly compacted, with prismatic structural aggregates. Principal causes of structural degradation of cultivated Chernozems are the dehumidification of the cultivated layer because of intensive soil tillage with heavy machinery.

Soil destructure and compaction reduce water infiltration capacity and exacerbate losses by water erosion. Soil tillage, and especially with the moldboard plow, also increases the rate of decomposition of crop residues and

humus. Increase in soil erosion and the rate of mineralization of SOM depletes the humus and degrades soil structure and the overall quality.

Severe deterioration of physical properties of soils indicates the inefficiency of existing farming systems. These unsustainable systems are undermining the capacity of Chernozems to maintain the productive capacity for a longer period of time. Cultivation of alfalfa (*Medicago sativa*) alone cannot significantly change the degradation trends. When alfalfa is grown in mixture with grasses, it can make a difference and improve soil quality and functionality.^[20]

Dehumidification or Depletion of SOM Contents in Cultivated chernozems is a Serious Issue

In natural ecosystems, formation of new SOM and decomposition are in dynamic equilibrium. The humus content is depleted when the rate of decomposition exceeds that of its formation. It is the SOM and its quality which enhance life support processes in the soil. The value of SOM lies in its dynamic nature—a continuous cycle of synthesis and decomposition.^[9,20–22]

It is well known that the most intensive mineralization of SOM occurs during the initial period of conversion of virgin soil to agroecosystems. In the book dedicated to 100 years from the publication of “Russian Chernozem” written by V.V. Docuchaev, it was established that losses of humus for different geographical regions in Russia range from 51–71 to 150–180 Mg/ha.^[1] The annual rate of uncompensated losses of SOM is 0.5–0.9 Mg/ha. Organic matter is the source of the energy without which the plant food elements cannot be changed into a usable form. Thus, stocks of SOM have decreased for the entire soil profile over 100 years by as much as 40–50% and even more relative to the antecedent stocks. The data from some long-term field experiments indicate the trend of humus concentration of 11% at 50–60 cm depth and 8% at 140–150 cm depth.

The long-term Sanborn experiments in Missouri, U.S.A., compared the properties of soils under undisturbed virgin prairie to that of an adjoining field cultivated to corn, wheat, and oats for 60 years without the addition of fertilizers. The cultivated field was not affected by accelerated soil erosion. The losses of SOM on cultivated fields were as much as 38%. Consequently, the granular structure of soil was degraded, it was highly compacted, its capacity to provide crops with nutrients was curtailed, and the agronomic yield was severely reduced.

Similarly, long-term field experiments conducted with different crop rotations, monoculture, irrigation, and different systems of irrigation and fertilization on typical Chernozems from Balti steppe, in the Northern part of the Republic of Moldova, have shown the adverse impacts of each of these factors alone on the SOM stocks in the upper layers and in the whole soil profile.^[10,13] The positive

impacts of the abovementioned factors of agricultural intensification on crop yields are not equivalent to their influence on soil fertility. Organo-minerals and organic systems of fertilization in crop rotation with 50% of row crop reduce the stocks of SOM to a lesser extent relative to the use of chemical fertilizers. The period of heavy loss of SOM is followed by stabilization of the losses by mineralization when the losses are lower than those in the initial period. Reduction in the concentration of SOM and total N contributes to the stabilization and even decline in crop productivity. Similar trends have been observed in many long-term field experiments all over the world and in productivity of majority of crops.^[10,23–25]

The equilibrium in the SOM concentration is determined by the climate of the region and by the farm (field) management practices. In the northern latitudes, with more precipitations and lower temperatures, ecological factors of SOM restoration are favorable than those in the southern regions.

No single factor of intensification used alone can compensate the annual losses of SOM from arable land. Integration of crop rotation with perennial legumes and grasses, use of farmyard manure, and conversion of plowing to conservation agriculture can maintain crop productivity and improve soil fertility. A judicious combination of the moldboard plowing and conservation tillage can be efficient in crop rotation with perennial legumes and grasses along with sufficient application of farmyard manure or other supplementary sources of organic amendments.^[13,17]

General awareness about these facts must be enhanced globally regarding the rapid rate of SOM depletion and to emphasize the need for SOM restoration. Predominant farming system in each country must be designed to at least maintain if not increase the stocks of SOM. Implementation of this strategy at the national level will determine the sustainability of land use together with different ecosystem services provisioned by agriculture for each society.

Water and wind erosion are the principal determinants of soil degradation and SOM depletion. Soil degradation can be reversed by mitigating soil erosion. When soil is effectively covered by vegetation, crop, or crop residues, it has lower risks of soil erosion and degradation.

Improvement in structure decreases soil losses, because of high water infiltration even on steep slope gradients. Reducing the intensity and frequency of moldboard plowing is crucial to the reduction of soil erosion. Simultaneously, the proportion of area under sod/cover crops should be increased especially in regions with torrential rains and hilly terrain. The higher is the degree of soil erosion, the lower is the level of productivity. Agronomic productivity of principal crops can be reduced by 20–30% on less eroded soils, 40–60% on moderately eroded, and 60–80% on strongly eroded soils.^[9,21,12,26] In severely eroded soils, even the B2 horizon is washed away. The degree of erosion on cultivated soils in Moldova is evaluated [Table 2](#).

The ecological functions of soils (thermodynamical, hydrological, geological, biogeochemical, agropedological, etc.) are strongly determined by the SOM concentration. The energy content of humus in Chernozems of the steppe regions is 8–9 times higher than that of crop and vegetation.^[28] The primary energy potential of Chernozems is located in the root zone.

Soils are the most important link in turnover of water and in carbon sequestration. The biodiversity of soils flora and fauna is significantly richer than that of the terrestrial biomes. However, the knowledge about the soil biodiversity and its functions is rather sketchy even with the modern science. Properly managed soils also have a high capacity to recycle nutrients. The ratio of diphosphorus pentoxide in 0–20 cm to that in 50–100 cm soil layers is >1.0, which implies that crops have the ability to “pump” P from deeper soil into the surface layers. Similar trends are observed for some trace elements.

Chernozems are highly resilient—because of the inherent physical and mineralogical properties, thickness of topsoil, and the mechanisms of self-restoration similar to that of a living organism. Soil maintains its life-support processes by continuous recycling of SOM. Thus, farming systems and management practices must be specifically designed to enhance recycling of SOM for achieving sustainability. Ecological principles must be strictly observed for achieving a more sustainable development of agriculture. Therefore, restructuring of modern farming systems of Chernozems must be realized by following the holistic

Table 2 The effects of severity of soil erosion on agronomic productivity.^[18]

The degree of erosion	The degree of losses for genetic horizons	The most probable content of humus relatively to undegraded soil, in % for soil layers		The distribution of carbonates in soil profile
		0–20 cm	0–100 cm	
Low	Less than ½ of A	75–80	65–70	At the depth of 30–50 cm in the northern part
Average	More than ½ of A or the whole A horizon	65	45–55	From the surface, very seldom at the simple deeper in the northern part
Strong	Partly or the whole horizon B1	50	35–45	From the surface

(systemic) vision rather than a reductionist (simplistic) approach. The holistic vision may include the followings:

- using landscape approach to land organization with differentiation of land use according the ecological characteristics of soils and crops;
- adopting crop rotations with the mixture of perennial leguminous crops and grasses, with a higher diversity rather than a narrow range of crops;
- reducing or avoiding the inversion system of soil tillage while retaining more residues on soil surface, more mixed crops, and cover crops;
- integrating crops and animal husbandries; and
- promoting cooperation on horizontal and vertical levels among farmers, farmers and processing industry, between all links within the whole food chain to ensure equitability between all of them.

Achieving sustainability of the entire food chain, from production to consumption and return, is essential in maintaining soil quality, minimizing risks of degradation of soil and other natural resources, and contributing to rebuilding rural communities and renewing society.

REFERENCES

1. Docuchaev, V.V. *Russian Chernozem*, 2nd Ed.; State Publisher of Agricultural Literature: Moscow, 1952; 634 pp. (Russian).
2. Ursu, A. *Soils of Moldova*; Știința: Chișinău, 2011; 323 pp. (Romanian).
3. Boincean, B.; Lal, R. Conservation agriculture on Chernozems in the Republic of Moldova, 2014. In *Soil Management of Small holder Agriculture: Advances in Soil Sciences*; Lal, R., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2014; Vol. 6, 203–222.
4. Whalen, J.K.; Sampedro, L. *Soil Ecology and Management*; CAB International: Wallingford, 2010; 296 pp.
5. Magdoff, F.; van Es, H. *Building Soils for Better Crops*, 2nd Ed.; Sustainable Agriculture Network: Burlington, 2000; 229 pp.
6. Doran, J.W.; Coleman, D.C.; Bezdicek, D.F.; Stewart, B.A. *Defining Soil Quality for a Sustainable Environment*. SSSA Special Publication, Number 35; Soil Science Society of America and American Society of Agronomy: Madison, 1994; 244 pp.
7. Doran, J.W.; Jones, A.J. *Methods for Assessing Soil Quality*. SSSA Special Publication, Number 49; Soil Science Society of America and American Society of Agronomy: Madison, 1996; 410 pp.
8. Charles, D. *The formation of vegetal layer by the earthworms and observation on their activity*. Translation in Russian by Menzbir M.A. 1882, (Moscow).
9. Boincean, B. *Ecological Farming in the Republic of Moldova (Crop Rotation and Soil Organic Matter)*; Știința: Chișinău, 1999; 269 pp. (Russian).
10. Dent, D., Ed. *Soils as World Heritage*; Springer: Dordrecht, 2014; 502 pp.
11. Krupenicov, I.A.; Boincean, B.P.; Dent, D. *The Black Earth. Ecological Principles for Sustainable Agriculture on Cernozem Soils*; Springer: Dordrecht, 2011; 143 pp.
12. Kononova, M.M. *Soil organic Matter*. Academy of Sciences of the USSR: Moscow, 1999; 313 pp. (Russian).
13. Brady, N.C.; Well, R.R. *The Nature and Properties of Soils*; Pearson Education: New Jersey, 2008; 975 pp.
14. Goldstein, W.; Boincean, B. *Sustainable Agriculture in The Forest Steppe and Steppe Zones of Moldova, Ukraine And Russia*; Ekoniva: Moscow, 2000; 267 pp. (Russian).
15. Odum, E.P. *Basic Ecology*; CBS College Publishing: Philadelphia, 1983; Vol. 1–2.
16. Gliessman, S.R. *Agroecology. Ecological Processes in Sustainable Agriculture*; Lewis Publishers: Boca Raton, 2000; 337 pp.
17. Soule, J.D.; Piper, J.K. *Farming in Nature's Image. An Ecological Approach to Agriculture*; Foreword by Wes Jackson, Island Press: Washington, DC, 1992; 286 pp.
18. Nour, D.D., Ed. *Soil Erosion. The Essence of the Process. Consequences, Minimalisation and Stabilization*; Pontos: Chisinau, 427 pp. (Russian).
19. Altieri, M.A.; Nicholls, C.I. *Agroecology and the Search for a Truly Sustainable Agriculture*; University of California: Berkeley, 2005; 290 pp.
20. United States Department of Agriculture. *Soils and Men: Yearbook of Agriculture*; United States Government Printing Office: Washington, D.C., 1938; 1232 pp.
21. Kovda, V.; Samoilova, E.M. *The Russian Chernozem, 100 Years after Docuchaev*; Nauka: Moscow, 1983; 303 pp. (Russian).
22. Brown, L.R.; Lenssen, N.; Kane, H. *Vital Signs, 1995. The Trends that are Shaping Our Future*; Worldwatch Institute: New York, 1995; 176 pp.
23. Viliams, V.R. *The Complete Works in 12 Volumes; State Publisher of Agricultural Literature*; Moscow, 1949. (Russian).
24. Mulvaney, R.L.; Khan, S.A.; Ellsworth, T.R. Synthetic nitrogen fertilizers deplete soil nitrogen: A global dilemma for sustainable cereal production. *J. Environ. Qual.* **2009**, *38*, 2295–2314.
25. Brown, L.R. *State of the World. A World Watch Institute; Report on Progress Toward a Sustainable Society*; W.W. Norton and Company: New York and London, 1996; 249 pp.
26. Lîcov, A.M.; Esikov, A.I.; Novikov, M.N. *Soil Organic Matter in Arable Non-chernozem Soils*; Moscow, 2004; 630 pp. (Russian).
27. Magdoff, F.; Weil, R.R. *Soil Organic Matter in Sustainable Agriculture*; CRC Press: Boca Raton, 2004; 398 pp.
28. Volobuev, V.P. *Introduction to the Energetic of Soil Formation*; Nauka: Moscow, 1974. (Russian).

Chlorine

Uzi Kafkafi

Department of Field Crops, Faculty of Agriculture, Hebrew University of Israel, Rehovot, Israel

Guohua Xu

College of Resources and Environmental Sciences, Nanjing Agricultural University, Nanjing, China

Abstract

Chloride ion is abundant in nature. Plants need it as a micronutrient. Its deficiency in plants is observed in rainy inland regions. Chloride salinity is widespread all over the world in irrigated areas. Plant species and cultivars vary in their sensitivity to excess Cl^- in the soil solution. The transport of Cl from the soil to the plant and its transport inside plant cells involve chloride channels and energy investment. It accumulates in leaves and to a lesser extent in fruits and seeds. There is a competition in the uptake and accumulation between nitrate and chloride, the two anions that are taken in large quantities by plants. Continuous presence of nitrate in the soil solution near the roots reduces chloride uptake by a factor of 2–6, enabling sensitive plants to overcome Cl salinity. The main reasons for excessive Cl in soils are excess evaporation from irrigated lands, sea inundations, and use of saline water for irrigation.

INTRODUCTION

Chlorine is found in nature in the form of negatively charged (Cl^-), highly water-soluble anion. Most of the world's Cl^- is found either in oceans or in salt deposits left by evaporation of old inland seas, which are found in deep quarries. Field and glasshouse studies in the mid-1800s to the early 1900s have shown that the influence of Cl^- on plant growth varies with the plant variety.^[1] Lipman^[2] demonstrated the beneficial effect of Cl^- on buckwheat (*Triticum aestivum* L.) growth. Arnon and Whatley^[3] suggested that Cl^- is an essential cofactor in the O_2 evolution during photosynthesis. The first complete definition of Cl^- , as a plant essential micronutrient, was described by Broyer et al.^[4]

The function of Cl^- in crop yield has been largely neglected,^[5] as it becomes a limiting factor for plant growth only in areas of high precipitation far from the sea. The negative effects of high Cl^- concentrations in soil and irrigation water on crop production are observed in coastal, arid, and semiarid areas, where freshwater sources are often scarce and the available groundwater is saline. The dependence of modern agriculture on irrigation and fertilization causes more concern on excess of Cl^- rather than on its deficiency.^[6]

CHLORIDE IN SOIL

The concentrations of Cl^- in natural sources are listed in Table 1. The Cl^- content in the soil is not an intrinsic

property of the soil but rather a result of soil management, rainfall and evaporation, and irrigation and fertilization.

The Cl^- concentration in rainwater close to seashore ranges from about 20 to 50 g/m^3 and 2 to 6 g/m^3 in inner continental areas.^[5,7] The annual amount of Cl^- deposition on land ranges from 17 to 175 kg/ha. Midcontinental areas such as the Great Plains of North America receive <1.0 kg Cl^- /ha annually through precipitation.^[5]

The chloride anion is not sorbed on soil particles at neutral and basic pH values, and therefore is easily leached. However, in acid soils containing variable-charge clays, a slight specific sorption of Cl^- is involved.^[9]

Plants can take up substantial amounts of Cl^- when soil Cl^- levels are high. At peak accumulation, the Cl^- content of spring wheat (*T. aestivum* L.) was 18 and 61 kg/ha on sites testing low and high in Cl^- , respectively.^[5]

PLANT YIELD AND QUALITY RESPONSES TO CHLORIDE

The small amount of Cl^- in precipitation, combined with the limited use of potassium chloride (KCl) fertilizer in some potassium (K)-rich inland soils, results in low levels of Cl^- in soil testing.^[5] Substantial positive responses to Cl^- -containing fertilizers have been reported for different crops in many parts of the world:^[5,6,10] barley (*Hordeum vulgare* L.), coconut (*Cocos nucifera* L.), kiwifruit (*Actinidia deliciosa* L.), oil palm (*Elaeis guineensis* Jacq.), potato (*Solanum tuberosum* L.), wheat (*T. aestivum* L.), tobacco

Table 1 Chloride concentrations in some natural sources.

Source	Chloride (g kg ⁻¹)
Earth crust	1.50
Lithosphere	0.48
Basalt rocks	0.50
Syenite	0.98
Igneous rocks	0.23
Shale	0.16
Sandstone	0.02
Limestone	0.37
Dolomite	0.50
Soils	0.10
Plants	1.0–10.0
Ocean	19.0 ^a
Dead Sea water	218.0 ^a
Low to medium saline water	0.10–0.30 ^a
High to very high saline water	0.30–1.20 ^a
Table salt (NaCl)	607
Potassium chloride (KCl)	475
Potassium chloride fertilizers	450–570

^aUnit: kg m⁻³.Source: From Yaalon.^[7] ©1963 and Flowers.^[8] ©1988.

(*Nicotiana tabacum* L.), and sugar beet (*Beta vulgaris* L.). Typical symptoms of Cl⁻ deficiency include wilting of leaves, curling of leaflets, bronzing and chlorosis, and severe inhibition of root growth. The so-called “Cl⁻ deficient leaf spot syndrome,” the leaf spots and tissue necrosis in wheat, is caused by inadequate Cl⁻ nutrition (about 1 g/kg DM) rather than pathogen infection.^[10] The concentration range of Cl⁻ deficiency in plants varies between 0.13 and 5.7 g/kg (Table 2). Wheat yield benefits from Cl⁻ were attributed to either partial suppression of root or foliar diseases or to the enhancement of host tolerance to the “disease.”

Sensitivity to high Cl⁻ concentrations varies widely between plant species and cultivars. Generally, many woody plant species and legumes are susceptible to Cl⁻ toxicity, whereas most non-woody crops tolerate excessive Cl⁻.^[30] The critical toxicity concentration of Cl⁻ (CTCC) in saturated soil extracts varies from 10 to 15 mM for sensitive crops like common bean (*Phaseolus vulgaris* L.), grape (*Vitis vinifera* L. ssp. *Vinifera*), lettuce (*Lactuca sativa* L.), potato, and strawberry (*Fragaria vesca*) and to 70–80 mM for crops, such as barley, cotton (*Gossypium hirsutum* L.), sorghum (*Sorghum bicolor* L.), and sugar beet.^[6,30] The CTCC in plant is about 4–7 g/kg and 15–50 g/kg for Cl-sensitive and Cl-tolerant plant species, respectively (Table 2). It seems likely that factors associated with vigorous growth or Cl⁻ compartmentation within the cell could offset the inhibitory effects of Cl⁻ accumulation. The tolerance of a crop to Cl⁻ varies with the growth stage as well as climatic conditions.

The specific influences of Cl⁻ on the quality of agricultural products are not clear. Chloride generally accumulates in the vegetative parts, mainly in the leaves, while its content in grains, fruits, and seeds is very low and is hardly affected by the Cl⁻ concentration in the soil.^[5,6] Therefore, the quality of leafy vegetables is relatively sensitive to Cl⁻ such as Chinese cabbage (*Brassica rapa* L.), lettuce, and tobacco, while fruit quality is generally tolerant to high Cl⁻ levels in the soil.

PLANT CHLORIDE UPTAKE AND DISTRIBUTION

Uptake of Cl⁻ by the plant roots is an active process that requires energy. The Cl⁻ transport through the cell membrane involves the nH⁺:Cl⁻ symporter^[31] or occurs via antiport with hydroxyl ions energized by adenosine triphosphate (ATP).^[30] Specific protein channels energized by ATP for Cl⁻ transport are found both in the plasma-membrane and in the tonoplast. The fluxes of Cl⁻ in intact plants are very different from those measured in plants with excised roots. One of the schemes used to describe the control of Cl⁻ uptake and transport in plants is a homeostatic mechanism that senses vacuolar NO₃⁻ plus Cl⁻ or total anion concentration.^[32]

Chloride compartmentation appears to be highly regulated. In the chloroplast, Cl⁻ concentration remains relatively constant regardless of whether the plant growth medium is Cl⁻ deficient or excessive.^[30] Harmful effects of high leaf Cl⁻ concentrations could be avoided, if salt excluding varieties are used as rootstocks.

BIOCHEMICAL AND PHYSIOLOGICAL FUNCTIONS OF CHLORIDE IN PLANTS

Chloride has a stimulatory effect on asparagine synthetase, which uses glutamate as a substrate.^[33] The proton-pumping ATPase at the tonoplast is stimulated directly by Cl.^[34] The chlorinated indole-3-acetic acid (IAA) found in substantial amounts in some legume species seeds enhances hypocotyl elongation at 10 times the rate of IAA itself, probably because of the resistance of the former compound to degradation by peroxides.^[35] Chloride is required for the water-splitting system of photosystem II.^[3] Binding of Cl⁻ to membranes is needed for activation of the O₂-evolving enzyme. The preference for Cl⁻ over Br⁻, NO₃⁻, or I⁻ with regard to its ability to promote O₂ evolution is due to its specific ionic volume.

The ability of Cl⁻ to move rapidly across cell membranes against an electrochemical gradient and its relatively low biochemical activity are two important properties that make Cl⁻ particularly suited to serve as a key osmotic solute in plants. The accumulation of Cl⁻ by plants contributes greatly to an increase in cell hydration and turgor pressure, both of which are essential for cell elongation and stomata opening.^[16] The osmoregulatory function of Cl⁻ in

Table 2 Chloride concentrations in plants.

Plant family	Crop	Latin name	Plant part	Concentration ranges of tissue Cl (g kg ⁻¹ DM)			References
				Deficient	Normal	Toxicity ^a	
Actinidiaceae	Kiwifruit	<i>A. deliciosa</i> L.	Leaves	2.1	6.0–13.0	>15.0	[11,12]
Arecaceae	Coconut palm	<i>C. nucifera</i> L.	Leaves	2.5–4.5	>6.0–7.0		[13]
Asteraceae	Lettuce	<i>L. sativa</i> L.	Leaves	>0.14	2.8–19.8	>23.0	[14]
Chenopodiaceae	Spinach	<i>S. oleracea</i> L.	Shoot	>0.13			[15]
Chenopodiaceae	Sugarbeet	<i>B. vulgaris</i> L.	Leaves	0.71–1.78			[16]
Chenopodiaceae	Sugarbeet	<i>B. vulgaris</i> L.	Petioles	<5.7	>7.1–7.2	>50.8	[6,16]
Fabaceae	Alfalfa	<i>Medicago sativa</i> L.	Shoot	0.65	0.9–2.7	6.1	[17,18]
Fabaceae	Peanut	<i>Arachis hypogaea</i> L.	Shoot		<3.9	>4.6	[19]
Fabaceae	Red clover	<i>Trifolium pratense</i> L.	Shoot	0.15–0.21			[20]
Fabaceae	Soybean	<i>Glycine max</i> L. Merr.	Leaves		0.3–1.5	16.7–24.0033	[21,22]
Fabaceae	Subterranean clover	<i>Trifolium subterraneum</i> L.	Shoot	>1.0			[17]
Gramineae	Barley	<i>H. vulgare</i> L.	Heading shoot	1.2–4.0	>4.0		[5,9]
Gramineae	Corn	<i>Zea mays</i> L.	Ear leaves		1.1–10.0	>32.7	[23]
Gramineae	Corn	<i>Z. mays</i> L.	Shoots	0.05–0.11			[14]
Gramineae	Rice	<i>Oryza sativa</i> L.	Shoot	<3.0		>7.0–8.0	[24]
Gramineae	Rice	<i>O. sativa</i> L.	Mature straw		5.1–10.0	>13.6	[6]
Gramineae	Spring wheat	<i>T. aestivum</i> L.	Heading shoot	1.5	3.7–4.7	>7.0	[5,19]
Gramineae	Wheat	<i>T. aestivum</i> L.	Heading shoot	1.2–4.0	>4.0		[5,9]
Lauraceae	Avocado	<i>Persea americana</i> Mill.	Leaves		~1.5–4.0	~7.0	[25]
Malvaceae	Cotton	<i>G. hirsutum</i> L.	Leaves		10.0–25.0	>25.0–33.1	[6]
Rosaceae	Apple	<i>Malus domestica</i> L.	Leaves	0.1		>2.1	[18]
Rosaceae	Peach	<i>Prunus persica</i> L.	Leaves		0.9–3.9	10.0–16.0	[18,26]
Rosaceae	Pear	<i>Pyrus communis</i> L.	Leaves		<0.50	>10.0	[26]
Rosaceae	Strawberry	<i>F. vesca</i> L.	Shoot		1.0–5.0	>5.3	[19,26]
Rutaceae	Citrus	<i>Citrus sp.</i> L.	Leaves		~2.0	~4.0–7.0	[25]
Solanaceae	Potato	<i>S. tuberosum</i> L.	Mature shoot	<1.0	2.0–3.3	12.2	[27]
Solanaceae	Potato	<i>S. tuberosum</i> L.	Petioles	0.71–1.42	18.0	44.8	[27,28]
Solanaceae	Sweet pepper	<i>Capsicum annuum</i> L.	Shoot	10–40			^b
Solanaceae	Tobacco	<i>N. tabacum</i> L.	Leaves		1.2–10.0	>10.0	[6,18]
Solanaceae	Tomato	<i>Lycopersicon esculentum</i> Mill	Shoot	0.25		~30.0	[4,29]
Vitaceae	Grapevine	<i>V. vinifera</i> L. ssp. <i>vinifera</i>	Petioles		0.7–8.0	10.0–11.0	[6,18]

^aThe plant yields decline or the plant shows visible scorching symptoms in leaves.

^bXu and Kafkafi, unpublished data.

plants seems to operate at different levels.^[8] The Cl⁻ concentration range usually found in plants (50–150 mM of tissue water) exceeds its critical deficiency level by 1–2 orders of magnitudes. At low concentration of Cl⁻ (1 mM or less in the whole plant tissue), the osmoregulatory functions of Cl⁻ are presumably confined to specialized compartments in tissues or cells. These are the elongation zones of roots and shoots, the pulvini, stigmata, and guard cells, where the concentrations of Cl⁻ might be much higher than the average Cl⁻ concentration in the bulk tissue.^[6]

Activation of an H⁺ pump in the plasma membrane initiates K⁺ and Cl⁻ influx, accompanied by malate synthesis, resulting in osmotic water flow into the guard cells, and consequent stomata opening. The relative contribution of Cl⁻ and malate may vary among species and depend on the availability of external Cl⁻ and plant growth environment.^[36] In plant species such as onion (*Allium cepa* L.), which lack the functional chloroplasts for malate synthesis in the guard cells, Cl⁻ is essential for stomata functioning.^[36,37] Members of the *Palmaceae*, such as coconut and oil palm,

which might possess starch-containing chloroplasts in their guard cells, also require Cl^- for stomata function.^[13]

INTERACTION OF CHLORIDE UPTAKE WITH THE UPTAKE OF OTHER NUTRIENTS

Ammonium is taken up by plant as a cation, and therefore relatively more anions have to be taken up to maintain the electrical neutrality of the uptake process. Plants fertilized with NH_4^+ usually contain higher Cl^- levels in the tissue than plants fertilized with NO_3^- or with both N sources, irrespective of the Cl^- concentration in the nutrient solution.^[6] Thus, when Cl^- is present in the root medium, NH_4^+ uptake may increase the salt sensitivity of the plants.

The antagonism between NO_3^- and Cl^- uptake has been well demonstrated in many crops.^[6] When both Cl^- and NO_3^- anions are taken up by the root against their electrochemical gradient, Cl^- maintains its negative charge, while NO_3^- is metabolized and loses its negative charge. The accumulation of Cl^- reduces its further uptake since the Cl^- electrochemical potential gradient builds up during its accumulation in the cell.^[25] Nitrate can prevent Cl^- toxicity of avocado at a concentration of up to 16 mM in the root medium.^[25] On the other hand, Cl^- application may also be used as a strategy to decrease the NO_3^- content of leafy vegetables, as spinach (*Spinacia oleracea* L.), lettuce, and cabbage (*Brassica oleracea* L.), which are classified as NO_3^- accumulators.^[6]

Increasing concentrations of Cl^- in the root medium has generally no consistent effect on K concentrations in the plants. Potassium concentrations in the leaves of kiwifruit are significantly higher for vines receiving KCl than for vines receiving K_2SO_4 as the kiwifruit uses Cl^- rather than organic anions for charge balance, and thus maintains a high K uptake.^[11]

CHLORIDE MANAGEMENT IN IRRIGATION AND FERTILIZATION

Large amounts of Cl^- enter the field through irrigation. The amount of Cl^- added to a field with 500 mm of irrigation water containing only 200 g Cl/m^3 is only 1,000 kg/ha. This is four times more than the amount of Cl^- applied by fertilization with KCl at 500 kg/ha. The irrigation system influences the distribution of Cl^- salts in the soil. Irrigation with saline water is managed by an excess of irrigation to meet the leaching requirement for avoiding salt accumulation in the root zone.^[38] The amount of water required to wash salts out of the root zone can be estimated from the electrical conductivity (EC) of the irrigation water and the mean EC of the saturated soil extract at which no crop yield reduction occurs under sprinkler or surface irrigation.^[38]

Fertilization with potassium nitrate (KNO_3) under saline conditions might reduce the toxic effect of salinity of some woody plants even if it is associated with increased soil

osmotic potential.^[25] Under high salinity and Cl^- conditions, KNO_3 or potassium sulfate as K fertilizers are preferred due to their lower salt index and absence of Cl^- . Addition of adequate phosphorus can also be helpful in alleviating salt stress.^[6]

CONCLUSION

Chloride ion is essential for plant growth as a micro-nutrient. It is taken up by plants in large quantities when its concentration in the soil is elevated due to irrigation, fertilization, and evaporation of water from the soil surface. Cl^- deficiency was monitored in inland regions with ample rainfall. There is a wide range of Cl^- concentrations needed by plants (from Cl-sensitive plants like avocado to tolerant plants like coconuts). The main load of Cl^- is observed when irrigation with saline water is employed. The main Cl^- -rich fertilizer is KCl that is normally applied without deleterious effects.

REFERENCES

1. Tottingham, W.E. A preliminary study of the influence of chlorides on the growth of certain agricultural plants. J. Am. Soc. Agron. **1919**, *11*, 1–32.
2. Lipman, C.B. Importance of silicon, aluminum, and chlorine for higher plants. Soil Sci. **1938**, *45*, 189–198.
3. Arnon, D.L.; Whatley, F.R. Is chloride a coenzyme of photosynthesis? Science **1949**, *110*, 554–556.
4. Broyer, T.C.; Carlton, A.B.; Johnson, C.M.; Stout, P.R. Chloride—A micronutrient element for higher plants. Plant Physiol. **1954**, *29*, 526–532.
5. Fixen, P.E. Crop responses to chloride. Adv. Agron. **1993**, *50*, 107–150.
6. Xu, G.H.; Magen, H.; Tarchitzky, J.; Kafkafi, U. Advances in chloride nutrition of plants. Adv. Agron. **2000**, *68*, 97–150.
7. Yaalon, D.H. The origin and accumulation of salts in groundwater and in soils in Israel. Bull. Res. Coun. Isr. **1963**, *11G*, 105–131.
8. Flowers, T.J. Chloride as a nutrient and as an osmoticum. Adv. Plant Nutr. **1988**, *3*, 55–78.
9. Wang, J.H.; Yu, T.R. Release of hydroxyl ions during specific adsorption of chloride by variable-charge soils. Z. Pflanz. Bodenkunde. **1998**, *161*, 109–113.
10. Engel, R.E.; Bruckner, P.L.; Mathre, D.E.; Brumfield, S.K. Z. A chloride deficient leaf spot syndrome of wheat. Soil Sci. Soc. Am. J. **1997**, *61*, 176–184.
11. Buwalda, J.G.; Smith, G.S. Influence of anions on the potassium status and productivity of kiwifruit (*Actinidia deliciosa*) vines. Plant Soil **1991**, *133*, 209–218.
12. Prasad, M.; Burge, G.K.; Spiers, T.M.; Fietje, G. Chloride induced leaf breakdown in kiwifruit. J. Plant Nutr. **1993**, *16*, 999–1012.
13. von Uexkull, H.R. Chloride in the nutrition of coconut and oil palm. Trans. Int. Congr. Soil Sci. **1990**, *IV* (14), 134–139.

14. Johnson, C.M.; Stout, P.R.; Broyer, T.C.; Carlton, A.B. Comparative chloride requirements of different plant species. *Plant Soil* **1957**, *8*, 337–353.
15. Robinson, S.P.; Downton, W.J.S. K, Na and Cl content of isolated intact chloroplasts in relation to ionic compartmentation in leaves. *Arch. Biochem. Biophys.* **1984**, *228*, 197–206.
16. Ulrich, A.; Ohki, K. Chloride, bromine, and sodium as nutrients for sugar beet plants. *Plant Physiol.* **1956**, *31*, 171–181.
17. Ozanne, P.G.; Woolley, J.T.; Broyer, T.C. Chloride and bromine in the nutrition of higher plants. *Aust. J. Biol. Sci.* **1957**, *10*, 66–79.
18. Eaton, F.M. Chlorine. In *Diagnostic Criteria for Plants and Soils*; Chapman, H.D., Ed.; University of California: Riverside, 1966; 98–135.
19. Wang, D.Q.; Guo, B.C.; Dong, X.Y. Toxicity effects of chloride on crops. *Chin. J. Soil. Sci.* **1989**, *30*, 258–261.
20. Whitehead, D.C. Chlorine deficiency in red clover grown in solution culture. *J. Plant Nutr.* **1985**, *8*, 193–198.
21. Parker, M.B.; Gaines, T.P.; Gascho, G.J. The chloride toxicity problem in soybean in Georgia. In *Special Bulletin on Chloride and Crop Production*, 2nd Ed.; Jackson, T.L., Ed.; Potash & Phosphate Institute: Atlanta, 1986; 100–108.
22. Yang, J.; Blanchar, R.W. Differentiating chloride susceptibility in soybean cultivars. *Agron. J.* **1993**, *85*, 880–885.
23. Parker, M.B.; Gaines, T.P.; Gascho, G.J. Chloride effects on corn. *Commun. Soil Sci. Plan.* **1985**, *16*, 1319–1333.
24. Yin, M.J.; Sun, J.J.; Liu, C.S. Contents and distribution of chloride and effects of irrigation water of different chloride levels on crops. *Soil Fert.* **1989**, *1*, 3–7. (Chinese).
25. Bar, Y.; Apelbaum, A.; Kafkafi, U.; Goren, R. Relationship between chloride and nitrate and its effect on growth and mineral composition of avocado and citrus plants. *J. Plant Nutr.* **1997**, *20*, 715–731.
26. Robinson, J.B. Fruits, vines and nuts. In *Plant Analysis—An Interpretation Manual*; Reuter, D.J., Robinson, J.B., Eds.; Inkata Press: Sydney, 1986; 120–147.
27. Corbett, E.G.; Gausman, H.W. The interaction of chloride and sulfate in the nutrition of potato plants. *Agron. J.* **1960**, *52*, 94–96.
28. James, D.W.; Weaver, W.H.; Reeder, R.L. Chloride uptake by potatoes and the effects of potassium chloride, nitrogen and phosphorus fertilization. *Soil Sci.* **1970**, *109*, 48–53.
29. Kafkafi, U.; Valoras, N.; Letay, J. Chloride interaction with NO_3^- and phosphate nutrition in tomato. *J. Plant Nutr.* **1982**, *5*, 1369–1385.
30. Maas, E.V. Physiological responses to chloride. In *Special Bulletin on Chloride and Crop Production*, No. 2; Jackson, T.L., Ed.; Potash & Phosphate Institute: Georgia, 1996; 4–20.
31. Felle, H.H. The H^+/Cl^- symporter in root-hair cells of *Sinapis alba*. An electrophysiological study using ion-selective microelectrodes. *Plant Physiol.* **1994**, *106*, 1131–1136.
32. Glass, A.D.M.; Siddiqi, M.Y. Nitrate inhibition of chloride influx in barley: Implications for a proposed chloride homeostat. *J. Exp. Bot.* **1985**, *36*, 557–566.
33. Rognes, S.E. Anion regulation of lupin asparagine synthetase: Chloride activation of the glutamine-utilizing reaction. *Phytochemistry*. **1980**, *19*, 2287–2293.
34. Churchill, K.A.; Sze, H. Anion-sensitive, H^+ -pumping ATPase of Oat roots. *Plant Physiol.* **1984**, *76*, 490–497.
35. Hofinger, M.; Bottger, M. Identification by GC-MS of 4-chloroindolylacetic acid and its methyl ester in immature *Vicia faba* broad bean seeds. *Phytochemistry* **1979**, *18*, 653–654.
36. Talbott, L.D.; Zeiger, E. Central roles for potassium and sucrose in guard-cell osmoregulation. *Plant Physiol.* **1996**, *111*, 1051–1057.
37. Schnabl, H.; Raschke, K. Potassium chloride as stomatal osmoticum in *Allium cepa* L. (onion), a species devoid of starch in guard cells. *Plant Physiol.* **1980**, *65*, 88–93.
38. Keller, J.; Bliesner, R.D. Trickle irrigation planning factor. In *Sprinkler and Trickle Irrigation*; Keller, J., Bliesner, L.R.D., Eds.; Van Nostrand Reinhold: New York, 1990; 453–477.

Classification Systems

Stephen Nortcliff

Department of Soil Science, University of Reading, Reading, U.K.

Abstract

This entry briefly describes some of the general principles of soil classification including selection of the soil properties to be involved in the classification, a brief explanation of why it is necessary to classify soils, and the general principles of single or general purpose classifications. A small number of examples of general purpose classifications at international and national levels are provided along with a short discussion on the importance of soil classification as a means of communication between soil scientists and users of soil science information.

BACKGROUND

Soil varies across the earth's surface. This variation occurs both laterally and vertically and is a result of the complex interactions of the processes of soil formation in the environmental context, particularly climate, parent material, and topography. These processes have operated over varying time periods. As a consequence, some soils on the earth's surface may have been developing for thousands and possibly millions of years and are well developed and in equilibrium with their environment; others will show only the initial stages of soil development and may be only a few years or tens of years old. Often the characteristics of this latter group of soils are very strongly influenced by the nature of the relatively unaltered parent material.

An important feature of the soil is that it is not static, rather it is a dynamic natural body interacting in a complex manner with its environment. As a consequence, the soil changes through time and in space, as a response to environmental changes. As a result of this, the soil will show variation at different times within the development of a landscape. But that the soil will vary from place to place at any given time is of considerably greater importance. Taking this argument to its extreme conclusion, we might consider that no two places on the earth's surface have identical soils! Rather than accepting this extreme scenario, it is generally agreed that it is possible to group soils together into classes, within which many properties of the soil may be expected to be similar. Soil classification seeks to group these soils into soil classes or taxonomic units, and the related practice of soil survey seeks to map the soils into spatially homogeneous groups so that they may be portrayed on maps. Before a map can be produced; however, the soil surveyor needs, at the minimum, a basic soil classification system. Frequently, the soil surveyor producing soil maps will endeavor to produce soil-mapping units that correspond to the taxonomic units of the soil classification.

Soil classification is not new. In the 12th century, Yahya Ibn Mohammed, writing in his compendium entitled *The Book of Agriculture*^[1] noted that “the first step in agriculture is recognizing the soil and knowing how to differentiate between the good and the poor one.” It is interesting that there is very little documented consideration of soil classification from this time until the 19th century. In the second half of the 19th century, Dokuchaiev, working in Russia, highlighted the need to consider the nature and pattern of soils in the landscape and suggested the possibility of being able to group soils into distinct classes based on their properties and environmental context.^[2] From this time, throughout the 20th century and through the 21st century, there have been numerous attempts to classify soils, some in the context of a local landscape of a few kilometers, others on a global scale.

WHICH PROPERTIES SHOULD BE CHOSEN FOR CLASSIFICATION?

It is unclear whether the approach to soil classification in the 12th century involved only the top few centimeters or greater thicknesses of soil material, but many of the early documented classification focused on the upper soil layers, with little attention being given to underlying layers. As the knowledge concerning soils and their development have increased, so has the awareness of the need to consider the underlying layers. We identify four broad groups of these layers, from the surface downwards the O, A, B, and C, which we call soil horizons. The nature and significance of soil horizons are discussed in the entry *Diagnostic Horizons* (p. 668).

It has already been stated that part of the purpose of soil classification is to provide a degree of order to the considerable diversity of soils and soil properties found across the earth's surface. The classes are normally defined with

respect to one or a group of properties, with a particular soil having membership of the class if the values for the selected properties fall within defined ranges. The problem firstly arises in selecting the properties and secondly the ranges of properties to be used in determining class membership. In some cases, the properties selected for classification have not been properties of the soil but properties of the environment. In using environmental factors in the classification of the soils, it is assumed that there are very strong environment–soil property relationships. This was a major feature of the classification of soil by Dokuchaiev and co-workers in 19th century in Russia.

While the organization of our knowledge by means of a soil classification is a major progress toward an understanding of the nature of soils and soil distributions, there remains the often-problematical decision of choosing the right properties of soil to determine and identify the soil classes. As our knowledge of soils has increased, so has the range of properties upon which we might base a classification of soils. It is essential that there is a clear rationale for the selection of the small number of soil properties chosen often to differentiate the soil classes and to determine the class limits.

WHY CLASSIFY SOILS

The principal reason for classifying soils is to organize the vast amount of information about soils, to produce classes that have either similar properties and/or similar responses to external inputs, both natural and man-made. The production of a classification should save the investigator time and simplify the description of the soil.^[3] If many of the soils that we encounter have three or four properties in common (e.g., uniform texture down the profile, strong aggregated clay, and red color), it may be sensible to use one short name to describe them all. Having produced a framework for the classification of soils, this may provide the basis for the prediction of individual properties and behavior from knowledge of the defined class characteristics. In this respect, soil classification becomes an essential means of communication among soil scientists. It is also important in cases of identifying the most appropriate use of the soil, estimating the production, extrapolating knowledge gained at one location to other often relatively little known locations, and providing a basis for future research needs.

SINGLE OR GENERAL-PURPOSE CLASSIFICATION

One of the matters that have concerned those involved with classification of soils is whether the classification should be with respect to a single objective, a narrow range of purposes, or whether the classifier should seek to produce a general-purpose classification. The debate between those promoting either single or general-

purpose classifications has been widespread, yet the aims of these classifications are very different. Sneath^[4] argues that classification based on as many features as possible is the most “natural” classification and that such a classification will be best suited for general purposes. Certainly, a classification will have the greatest range of predictive powers, although for any specific purpose a special classification based on the features of importance to that purpose is likely to be more precise. Confusion frequently arises because the classifications have been undertaken according to the principles of general-purpose classification but then used for very narrow ranges of specific purpose situations. While the classifications may be very good at the general level, they will generally be less effective for specific needs. If the applications to specific purposes are inappropriate, they may result in erroneous predictions. The general-purpose classifications are the most widely used type of classification, both at the international and national levels. In Table 1, some of the general-purpose soil classifications are listed.

SOIL CLASSIFICATION AS A MEANS OF COMMUNICATION

Soil classification facilitates communication about soils, both among the soil scientists and between soil scientists and non-soil scientists. Because of this need for communication, it is essential that those persons using a classification are aware of the criteria used to define the classes, and the context within which these criteria were chosen. The outcomes of the classification must be understood in the context of selecting the differentiating criteria. If a soil classification is to be used as a means of communication, the principles upon which it is based and the criteria used to select the properties for class discrimination must be understood by those using it. While this is important for the communication among the soil scientists, it is of particular importance when soil scientists communicate with non-soil scientists. It is also important for the user to be aware whether there is any

Table 1 International and national general-purpose soil classifications.

Soil classification	References
International classification	<i>World Reference Base for Soil Resources</i> ^[5]
International/national classification	U.S.A.— <i>Soil Taxonomy</i> ^[6]
National classification	Australia— <i>The Australian Soil Classification</i> ^[7] France— <i>Référentiel Pédologique</i> ^[8] Russia— <i>Soil Classification System</i> ^[9] South Africa— <i>A Taxonomic System for South Africa</i> ^[10]

perceived covariation between the properties chosen for discrimination and other soil properties, as frequently classifications based on one property or one set of properties are used by other people, although in the original specification of the classification no such covariation had been assumed. Such misuses have the potential to result in substantial misunderstanding and major errors.

In summary, therefore, the key benefits that result from soil classification may be summarized as follows:

1. Soil classifications provide a framework for soil survey and for the observations made on soils.
2. Soil classifications provide an understanding of the relationships between (selected) soil properties and the significance of these properties with respect to the usage.
3. Soil classifications provide a means of organizing research information and the communication of research results.^[11]
4. The allocation of soils to classes should provide a framework within which the laws relating to the nature and development of the soil can be discovered.^[12]
5. Soil classifications provide a means of technology transfer, where the knowledge and behavior of soils in response to different management strategies may be transferred to soils that are considered to be similar with respect to the classification.^[13,14]

CONCLUSION

It is obvious that the vast amount of information that can be gathered for each soil requires some form of organization and management. To accomplish this, some form of allocation of soils into classes is essential. To simplify the wealth of information in this manner will greatly assist the communication between groups of soil scientists and between soil scientists and others. The key to a successful classification is to identify the purpose of classification and to select criteria selected as differentiating properties that are appropriate for this purpose. Frequently, the soil classifications are misused because no account is taken of why it was devised in the first place. Using an appropriate classification can provide greater insight into soil relationships, whereas the use of inappropriate classification may result in confusion.

REFERENCES

1. Johnson, W.M. Soil classification and design of soil surveys. In *Soil-Resource Data for Agricultural Development*; Swindale, L.D., Ed.; Hawaii Agricultural Experimental Station, University of Hawaii: Honolulu, 1978; 3–11.
2. Dokuchaiev, V.V. *The Russian Chernozem*. PhD Thesis; St. Petersburg, 1889. (In Russian).
3. Leeper, G.W. The classification of soils. *J. Soil Sci.* **1956**, *7*, 59–64.
4. Sneath, P.H.A. The application of computers to taxonomy. *J. Gen. Microbiol.* **1957**, *17*, 201–206.
5. Deckers, J.A.; Nachtergaele, F.O.; Spaargaren, O.C., Eds. Introduction. In *World Reference Base for Soil Resources*, 1st Ed.; ISSS/ISRIC/FAO, Acco: Leuven, 1998; 165 pp.
6. Soil Survey Staff. *Soil Taxonomy—A Basic System of Soil Classification for Making and Interpreting Maps*; Agricultural Handbook Number 436; USDA Natural Resources Conservation Service: Washington, 1999; 869 pp.
7. Isbell, R.F. *The Australian Soil Classification*; CSIRO Publishing: Melbourne, 1998; 143 pp.
8. Baize, D. *A Sound Reference Base for Soils. The Référentiel Pédologique*; INRA Editions: Paris, 1998; 322 pp. (In English).
9. Tonkonogov, V.D.; Lebedeva, I.I.; Shishov, L.L.; Gerasimora, M. *Russian Soil Classification System*; (English Version, Translated by Gerasimora, M., and Edited by Arnold, D.); Dokuchaev Soil Science Institute: Moscow, 2001; 220 pp.
10. Soil Classification Working Group. *Soil Classification. A Taxonomic System for South Africa*, 2nd Ed.; Memoirs on the Agricultural Natural Resources of South Africa No. 15; Department of Agricultural Development: Pretoria, 1991; 257 pp.
11. Runge, E.C.A.; McCracken, R.J. The role of soil classification in research planning. In *Soil Taxonomy—Achievements and Challenges*; Grossman, R.B., Rust, R.H., Eswaran, H., Eds.; SSSA Special Publication 14; SSSA: Madison, 1984; 15–28.
12. Smith, G.D. Lectures on soil classification. *Pedologie* **1965**, Special Ed. 4, 1–135.
13. Beinroth, F.H.; Uehara, G.; Silva, J.A.; Arnold, R.W.; Cady, F.B. Agrotechnology in the tropics based on soil taxonomy. *Adv. Agron.* **1980**, *33*, 303–339.
14. Buol, S.W.; Denton, H.P. The role of soil classification in technology transfer. In *Soil Taxonomy—Achievements and Challenges*; Grossman, R.B., Rust, R.H., Eswaran, H., Eds.; SSSA Special Publication 14; SSSA: Madison, 1984; 29–44.

Classification Systems: Arab Traditional, Moroccan Case

Mohamed Sabir

Hassan Ben Jelloun

National School of Forest Engineers, Tabriquet, Morocco

Abstract

Soil science, a relatively new discipline, compared to other fields, such as botany, biology, zoology, and mineralogy, is lacking an internationally accepted taxonomic system. Traditionally, a local scientific system of soil classification was lacking in most countries of the Arab world. However, soils were mostly designated by vernacular names that stem from local agricultural practices and soil use.

BACKGROUND

Different systems of soil classification have been developed throughout the world.^[1] In the Maghreb Arab countries (Morocco, Algeria, and Tunisia), the first efforts related to soil classification were initiated in 1934 by Del Villar, who was the president of the Mediterranean subcommission of the 1st International Society of Soil Science.^[2] His principal and most important pedological studies were conducted in Morocco.

The vernacular names are used among farmers and even among agricultural scientists, and detailed soil classification system are inspirations from the ones developed in Western Europe, Canada, and the United States, especially the systems of soil classification adopted by the Commission of Pedology and Cartography of Soil of France (CPCS, 1967),^[3] FAO/UNESCO, Soils of the World (ELSEVIER/ISRIC, 1989),^[4] Soil Taxonomy (1975),^[5] and the World Reference Base (1994).^[6]

The object of this entry is to present a brief review about the soil classification scheme used in some areas of Morocco.

ANCIENT ARAB SOIL CLASSIFICATION

Knowledge about land and agricultural practices stems from before Roman times. Men in ancient times have already used many of our farming practices (manuring, liming, and crop rotations with legumes).^[7]

Most of the actual knowledge farmers used during the long period from the fall of Roman civilization to the French Revolution and for some time afterward was the local knowledge of farmers.

At the time where the people of Europe were disorganized and lived in a dark age of diseases, famine, and war for more than a thousand years, the Arabian culture flourished in the Near East, Northern Africa, and southern

Spain, where farming was reasonably good, especially under irrigation.

Some of soil classification systems and farming practices were well explained in the handbook of agriculture prepared by Ibn-al-Awam,^[7] with a Moorish scholarship, in the 12th century. Ibn-al-Awam's^[8] book of agriculture classified different land types according to their agricultural qualities as follows:

Black earth: A warm earth that gives high agricultural yields.

Red earth: A moderately warm earth with moderate agricultural yields.

Yellow earth and white earth: A cold earth that gives low agricultural yields.

Dry earth: It has two species or divisions.

Sandy earth: It has a low fertility level.

Muddy earth: It has a marly-clay texture, it is sticky and plastic, and it becomes very hard and very compact when dry.

MOROCCAN VERNACULAR SOIL NAMES

Soil classification was introduced for the first time to the Mediterranean region in 1925 through the efforts of Spanish pedologist Del Villar ([Table 1](#)).^[9] Previously, most of the soil types were designated by their color, texture, structure, degree of fertility, and water-holding capacity, despite the fact that their meaning may differ from a region to another. In the northern humid regions, soils are distinguished by their fertility. Contrariwise, in the southern arid regions, soils are distinguished by their water-holding capacity following irrigation.

Among the Maghreb West Arab countries, Morocco, Algeria, and Tunisia show climatic, geologic, and geomorphologic similarities.^[10] Therefore, some of the vernacular

Table 1 Del Villar soil classification system.

Soil types	Soil cycles	Soil sectors
1-Homocyclic type	1.1. Sialferric cycle: soil dynamics are dominated by colloidal silicon and aluminum and by ferrous sesquioxides. 1.2. Calcareous cycle: pedogenic processes related to calcareous parent rock or carbonate-rich material are dominant. Leaching of carbonate may give K horizon at depth. Distinguished soils are: soil with AC profile; soil with AKC profile (Rendzina soil, Terra-rossa soil). 1.3. Sodic cycle: pedogenic processes are dominated by sodium (Na).	S1. Oxyhumic sector: unsaturated soil with acid humus. Leaching or epigenic metabolism is strong (e.g., gley soil). S2. Siallic sector: humus is not acidic, the ratio $\text{SiO}_2/\text{Al}_2\text{O}_3 > 2$, $\text{pH} < 7$. It shows two subdivisions according to parent material and energy of epigenic metabolism: parent material: siliceous or calcareous; energy of metabolism: podzolic profile with distinct A2 horizon, podzolic profile with A2 not very distinct, Illuvial profile. S3. Siallo-ferric sector: $\text{SiO}_2/\text{Al}_2\text{O}_3 > 2$; Fe_2O_3 is high. S4. Allitic sector: $\text{SiO}_2/\text{Al}_2\text{O}_3$ is very low. Leaching is important. Podzolization is possible. Ferruginous accumulation is possible (laterite soil).
1-Heterocyclic type: pedogenic metabolism responsible for soil type is not based on known chemical process. Chemical characteristics are mixed or variable; it is the regime which is responsible of soil type individualization.		S1. Saline sector: Na is in chloride form to which other soluble salts and gypsum are added. Saline soils may be epigenic or hypogenic. Epigenic metabolism of Na (leaching) is dominant (e.g., coast marshes). Hypogenic metabolism: salts are lifted from below to the surface. Examples are: thermal sources; salty lagoons; black saline soil: gley Solontchak soil; local deposited salt sebkhas of Arab countries; saline soil with surface crust; soil with salt at depth (infraline soil); soil with low salt concentration at surface (subsoltchak soil). S2. Alkaline sector: hypogenic metabolism process is low, but endogenic process resulting from precipitation, epigenic process, and percolation is dominant (Solonetz soil).
Distinguished soils are: non-sodic-hydro-epigenic soil (alluvial soil); soil with subamphigenic or subbalanced regime (prairie soil); soil with amphigenic or		

(Continued)

Table 1 Del Villar soil classification system. (Continued)

Soil types	Soil cycles	Soil sectors
balanced regime (chernosium soil); soil with hydro-hypogenic regime (clayey–gley soil); soil with hydro-hypogenic calcification; oligogenic soil (thin A horizon); Ambrogenic or holohypogenic soil = crypto-hypogenic and pheno-hypogenic soil (soils of desert climate. Below-ground water table and lithic material are the dominant genetic factors.); mixed transitional soil types.		

soil names (Table 2) used in Morocco may, to some extent, be encountered in Algeria and Tunisi.
In Morocco (Fig. 1), several local soil names are used.

- Region I: North Western Rif Mountains.
Ferich: A thin soil formed on pelitic rock material with high percentage of flysch plates.
Rmel and ferich: Complex soil, mixture of sandy (rmel), and ferich materials.

Table 2 Vernacular soil names and their equivalent in soil taxonomy.

Vernacular soil names	Soil taxonomy equivalent names
Ferich (R1 ^a), Hajar (R1), Sgguine (R5 ^a), and Salsale (R5), Azegzou (R7 ^a)	Entisols
Rmel (R3 ^a and R5)	Psaments (Entisols)
Rmel (R1), Abiad (R1), Amlil (R1), and Sahl (R1)	Udepts (Inceptisols)
Rmel jayef (R1)	Mixed Entisol–Udepts
Biad (R2 ^a) and El Hrach (R5)	Xerocrepts (Inceptisols)
Hrach (R2 and R3)	Stony Xerocrepts
Biadi (R3)	Xerocrepts over marl
Teine (R1)	Pelluderts (Vertisols)
Tirs (R3), Tirst (R4 ^a), and Tirste (R5)	Chromoxererts (Vertisols)
Hrach (R6 ^a), Itikki or Hamri (R6), Tamakalte (R6), and Terrist (R7)	Orthids (Aridisols)
Amarigh (R7)	Salorthids (Aridisols)
Azougakh (R7) and Amerdoun mouharmel (R7)	Argids (Aridisols)
Ard Kbira (R1) and Toires (R1)	Vertic Udolls (Mollisols)
Rtab (R4) and Hamri (R5)	Xerolls (Mollisols)
Safra (R4)	Aquolls (Mollisols)
Rmel (R2 and R4), Hamri (R2, R3, and R5), and Ahamri (R4)	Xeralfs (Alfisols)

^aR1, R2, R3, R4, R5, R6, and R7 are Moroccan regions where local soil names are used.

- Abiad: White soil color (abiad means white), developed on marl and marly–flysch material.
Hajar: Non-soil with sandstone outcrops (hajar means rock outcrop and stones).
Rmel or rmel jayef: Sandy soil (rmel means sand) with red-yellowish color. This soil has low fertility, sometimes has small rock grains, and is not hard when dry.
Teine: Vertic soil with high expanding clay content and high water retention (teine means clay).
Hamri: Red soil with high content of ferric oxides.
Amlil: Lithic calcareous soil with generally white color.
Sahl: Less developed soil on alluvial sediments.
Ard kbira: Black soil with high fertility level and high agricultural productivity and is suitable for all agricultural uses with low rock content and low water infiltration rate (ard kbira means big soil).
Toiress: Brown soil with low fertility but still good crop production. Rock content of this soil is moderate. Fairly hard when dry with particular texture.
- Region II: Temara (South of Rabat on the Coast Side).
Rmel: Sandy soil with brown color. Highly permeable, not hard when dry, has low percent of stones, and may reach 2 m depth.
Hamri: Red soil with high content of ferric oxides in mixture with clay; it is weakly permeable, hard, has low percentage of stones, and may reach 1 m depth.
El hassa: Gray soil with high percentage of stones, fairly permeable, and stones may reach 0.5 m depth.
Biad: White-colored soil with low agricultural productivity and good water permeability.
Hrach: Yellow-colored soil with low agricultural productivity, good water permeability, and some stones (hrach means rough).

- Region III: Romani (South East of Rabat)
Tirs: Black to dark brown soil with high fertility, needs large quantities of water for irrigation, has good agricultural potential, contains high percentage of expanding clay and low stones, has good permeability, and is resistant to erosion.

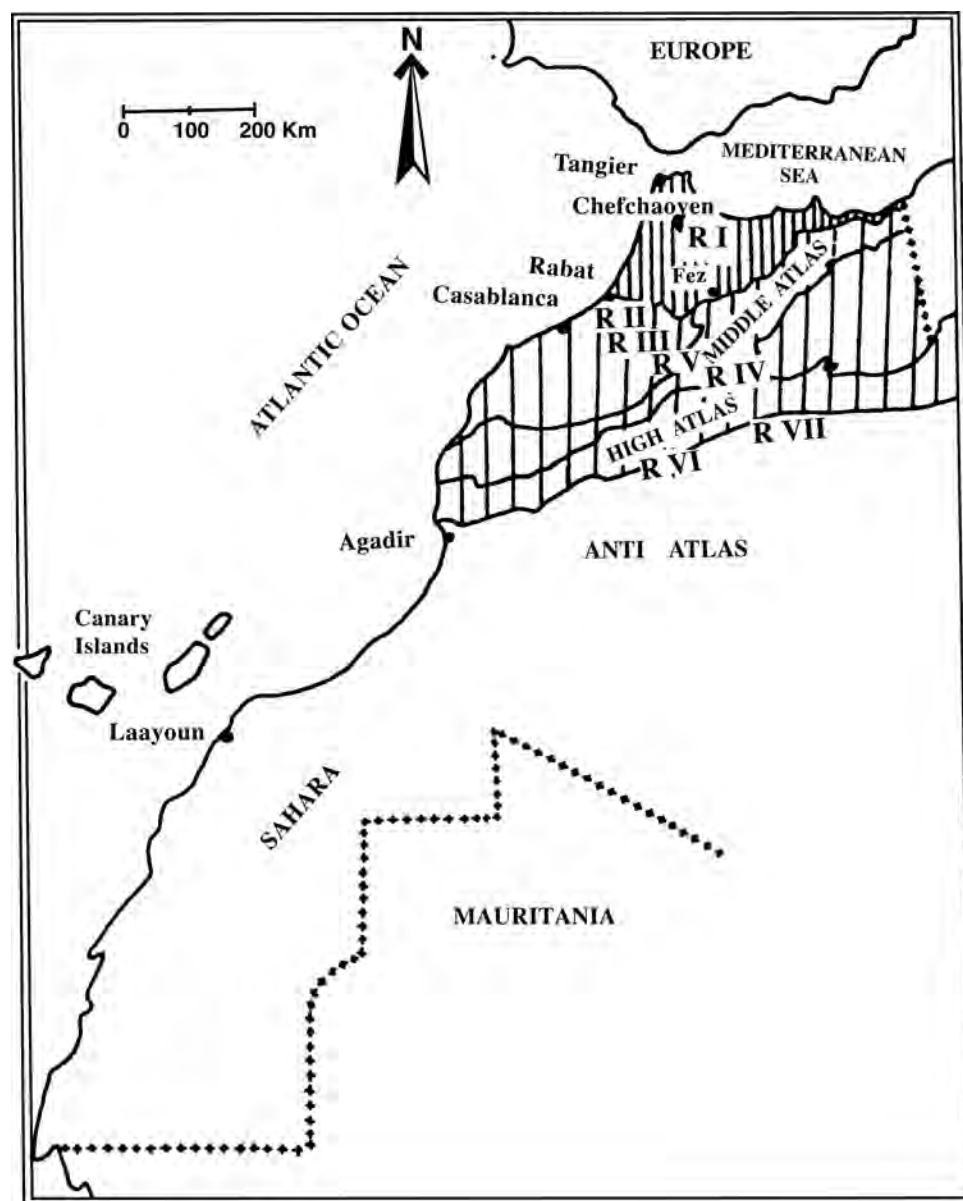


Fig. 1 Map of Morocco, geographical situation of different regions where traditional soil names were collected (RI: North Western Rif; RII: Temara; RIII: Romani; RIV: Midelt; RV: Moulay Bou Azza; RVI: Bou Malne Daddes; RVII: Erfound).

Brach: Grayish soil that requires less water with medium agricultural potential, is slightly permeable, and has large particles and stones.

Hamri: Red soil that has particles of loam and clay diameter in mixture with stones, needs much water for irrigation with very good agricultural potential.

Biadi: White soil with dust and stone on the surface.

Rmel: Brownish soil with sandy texture, high permeability, and low water-holding capacity, no stones, and deep.

Region IV: Midelt (High Moulouya Region).

Ahamri: Red soil that has clayey texture, compact with no stones, a high water-holding capacity, gets rapidly dry and warm, and has good crop production capacity.

Tirst: This soil is black, has clayey texture, and is less or more compact with no stones. Water-holding capacity of this soil is fairly good, and it is good soil for agriculture. It shows more resistance to dryness than hamri. On slopes, when dry, may show wind erosion hazard. Parent rocks are sediments of 10 cm depth. Agricultural yields are of 1000–1200 kg/ha/yr (less than world average).

Rtab: This is dusty soil with medium texture and no stones. It is 1 m thick and holds water for 20–30 days after showers or irrigation.

Brach: Brach is dark brown compact soil with large particles and stones. It is 0.5–1 m thick and holds water for 8–10 days after showers or irrigation.

Rmel: It has silver-gray color, light-sandy texture, and low water-holding capacity.

Safra: Yellow calcareous soil (safra means yellow color) with clayey texture, hard, and shows some water stagnation, and shallow (about 15 cm thick).

Region V: Moulay Bouazza

Tirste: Black soil with slightly stony–fine texture, low permeability, high water-holding capacity, and shows little cracks when dry and good biomass production vegetation development.

Hamri: It is a red compact soil, slightly stony texture, low to moderate water permeability, and cracks when dry. This soil needs much water for irrigation and shows medium potential for agriculture.

El Hrach: This soil has reddish-gray color with stones in the surface layer, is fairly permeable, and requires low quantities of water for irrigation.

Rmel: Yellow soil of sandy texture, with no stones, and highly permeable.

Sgguine: White-colored soil with hard stony–fine texture, permeability is low, and agricultural potential is poor.

Salsale: Gray soil with low stones and has friable consistence, low permeability, and poor quality.

Region VI: Bou Malne de Daddes (arid and Saharan region).

Rmel: It has yellow soil color, fine texture with no stones, not hard when dry, fertility is medium, and water-holding capacity is very low.

Hrach: Faint red soil, very hard stony–loamy texture, fertility is moderate, water-holding capacity is low, workability is difficult, sometimes has water stagnation, and agricultural potential is medium.

Itikki or hamri: Red soil with very hard compact clayey texture, has a high water-holding capacity, its dryness is very slow, and shows a good agricultural potential.

Tamakalte: Black gray soil with a fine clayey texture, resistant to dryness, has a high water retention capacity, and has a good agricultural potential.

Region VII: Erfoud (arid and Saharan region).

Terrist: White–yellow soil, without stones, no salt, and agricultural potential is low.

Amarigh: Different colors of soil (white, yellow, red, and black) and has salinity problems.

Azegzou: Schistous non-soil and agriculture is not practicable.

Azougakh: Red soil, may have or may have not stones, and good for agriculture.

Amerdoum mouharmel: Very heavy clayey soil, dryness is very slow, and has spontaneous vegetation.

CONCLUSION

In the Maghreb Arab world, soil science, in general, and soil classification systems, in particular, have followed the same evolution as in all the circum-Mediterranean countries. In this region, agricultural practices go back to the prehistoric period, and soil names used before the development of a scientific soil classification scheme were mostly related to local agricultural practices and were based essentially on color, texture, structure, fertility, richness of stones, and water-holding capacity.

In Morocco, as likely in the other countries of the Arab world, because of the multiplicity of ethnic tribes and natural conditions (climates, geological material, and vegetation), most of these names are maybe the same to qualify different soils.

REFERENCES

1. Finkl, C.W. Soil classification. Benchmark Pap. Soil Sci. **1982**, *1*, 391.
2. Del Villar, E.H. *Types de sol de l'Afrique du Nord*; Soil Types of North Africa; Fascicule II: Tunis-Rabat, 1947; 137–288.
3. Commission of pedology and cartography of soils (CPCS). *Reprinted from Classification des Sols*; INRA, Laboratoire de Géologie-Pédologie: Paris, 1967; *1*, 5–13, 15.
4. FAO-UNESCO. *Soils of the World*; Elsevier Science Publishers: Amsterdam, 1987.
5. U.S. Department of Agriculture. Soil Survey Staff. Soil taxonomy, a basic system of soil classification for making and interpreting soil surveys. In *Agriculture Handbook No. 436*; Soil Conservation Service, U.S. Department of Agriculture: Washington, 1975; 754 pp.
6. *Atelier sur les bases de données SOTER dans les pays de l'UMA*; Workshop on SOTER Data Base in UMA Countries; Organisation des Nations Unies pour l'Alimentation et l'Agriculture, Bureau Sous-Régional pour l'Afrique du Nord, SNEA: Tunis, 2001; 132 pp.
7. *Soil, the Yearbook of Agriculture*; U.S. Department of Agriculture: Washington, 1957; 784 pp.
8. Ibn-al-Awam. Le livre de l'agriculture (Kita[^]b al Fila[^]ha). In *Traduction from Arab by J.J. Clément-Muller Introduction of Mohammed El Faïz, "Thesaurus"*; Actes Sud/Sindbad, 2000; 1027 pp.
9. Del Villar, E.H. *Méthodes de Classification et Analyse des Sols: Base Scientifique Pour Leur Cartographie Harmonique Universelle* (Methods of Classification and Soil Analysis. A Scientific Basis for Their Cartography), 1953; 193 pp.
10. Refleh, Ph.; Chami, A.M. *Geography of the Arab World*; Annahda Library: Egypt, 1962; 375 pp.

Classification Systems: Australian and New Zealand

Terry A. Isbell

Davies Laboratory, U.S. Department of Agriculture—Agricultural Research Service
(USDA-ARS), Peoria, Illinois, U.S.A.

A.E. Hewitt

Landcare Research, Christchurch, New Zealand

Abstract

This entry provides a brief review of the development of two national soil classification systems: Australian and New Zealand. Of the 15 New Zealand soil orders that are defined, at least two are virtually absent in Australian soil landscapes. Conversely, some widespread Australian soils are only partly represented in the New Zealand system. There are some soils common to both the countries that probably could be satisfactorily classified at the order level by the other system.

INTRODUCTION

Although Australia and New Zealand are relatively close neighbors both are independent island states and this probably accounts for some degree of isolation between the two countries. In terms of size, climate, geology, vegetation, and land forms, there are striking differences, and it is not surprising, that the New Zealand soil pattern is mostly very different from that of much of Australia. It also follows for these factors, and other national differences, that separate soil classification schemes have always been in use.

It is widely accepted that classification is an essential part of any scientific discipline and is a necessary part of the language of science. No classification scheme can remain static; as new knowledge is gained, soil classifications need to be updated and improved. The history of national soil classifications in Australia^[1,2] is a good example of how benefits are obtained from modifications in the light of new knowledge. This, in effect, largely explains why Australia has rightly had a succession of national classification schemes. It is also of interest to note that two of the commonly used schemes—the *Factual Key* of Northcote^[3] and the so-called *A Handbook of Australian Soils*^[4]—were a direct result of the decision to hold the Ninth International Congress of Soil Science in Adelaide, Australia, in 1968. This meeting served as a catalyst to increase rapidly the knowledge about Australian soils by means of a targeted approach to the mapping of the continent at a published scale of 1:2 million using the *Factual Key*.^[3] This was achieved on time in spite of the size and soil diversity of the Australian continent. The *Factual Key* is a bifurcating, hierarchical scheme with five categorical levels. All classes are mutually exclusive, and

the keying morphological attributes are determined in the field, including the pH and the presence of carbonate as determined by a simple field test. Most class names are descriptive, e.g., “Hard Pedal Mottled Red Duplex Soils.”

The associated *A Handbook of Australian Soils*^[4] was a separate exercise to the soil mapping project and is a compendium of morphological and laboratory data of 94 soil profiles that were visited on the 1968 Congress field excursions, plus some extra representative soils not visited in the field tours. The “Handbook” is not a formal classification but is an assemblage of “great soil groups” largely derived from Stephens.^[5] This was more or less current at the time and was essentially based on the earlier U.S. Department of Agriculture schemes that were in vogue prior to the formal advent of the early *Soil Taxonomy Approximations* of the 1960s. Both the *Factual Key* and the *Handbook* are widely used in Australia, the former mainly because of the Australia-wide map coverage at 1:2 million, while the *Handbook* is very useful because of the detailed accounts of the morphology, micromorphology, and laboratory data for a large number of widespread Australian soils. A disadvantage of the *Handbook* is the lack of a key to the soil groups. The most comprehensive of the earlier New Zealand national soil classification systems is the New Zealand Genetic Soil Classification^[6] reviewed by Hewitt^[7] in 1992. In his classification, Taylor^[6] developed what would be called soil landscape models, in which soil groups were related to environmental factors employing the zonal concepts of the Russians. The classification by Taylor was first published in 1948 as a legend to the first National Soil Map, but with time, a number of weaknesses became apparent. Examples

include the failure of the genetic scheme when applied at the scale of the farm paddock and the difficulty of soil correlation between regions. It became obvious that a new model was required that made provision for important classes of New Zealand soils.

NATIONAL SOIL CLASSIFICATION SYSTEMS IN THE MODERN ERA

In the 1970s and 1980s, there was an increase in soil surveys in some Australian states, particularly Queensland and the Northern Territory, where the then-current Northcote and Stace, et al. systems were often found to be inadequate to cater for many “new” Australian soils, particularly in the wet tropics and in the subtropics. A soil classification committee with an Australia-wide charter was set up in the early 1980s to remedy the lack of an adequate, up-to-date national soil classification. Extensive field travel was carried out in most parts of Australia to gather field and laboratory data to build up a database, which would eventually contain some 14,000 published and unpublished profiles, many with laboratory data; mainly chemical. The “First Approximation” of a new national scheme was first compiled in 1989 and was widely circulated in Australia for comment. Two further approximations were produced and circulated before the published version appeared in 1996 titled *The Australian Soil Classification* (ASC).^[2]

It should be noted that due assessment was made of the two so-called international systems, namely, *Soil Taxonomy*^[8] and the subsequent revisions of most of its orders, and the other major “international” system that of the FAO–UNESCO (1990) *Soil Map of the World* and its updated (1998) version titled *World Reference Base for Soil Resources*.^[9] While these two so-called international schemes are of considerable value and interest for comparative purposes, it is more likely that most Australian pedologists would agree on the need for an appropriate national system that is regularly updated. The possibility of using methods of numerical classification was also examined but was thought to be inappropriate because of the great variability and inconsistencies in the soil datasets, e.g., lack of representation of important groups of soils.

The Australian Soil Classification

This scheme^[2,10] is widely used in most Australian soil surveys and pedological studies, primarily in an identification role. The scheme may be described as a multicategorical, hierarchical (order, suborder, great group, subgroup, and family), general purpose scheme with classes defined on the basis of diagnostic horizons or materials and their arrangement in vertical sequence as seen in an exposed profile. The classes are mutually exclusive,

and the allocation of new or unknown individuals to the classes is by means of keys. The scheme is open-ended and new classes can be defined as knowledge increases, although these may not necessarily be added in sequence to the present list. Fourteen orders are defined (Fig. 1). An innovation is the development of an interactive key to *The Australian Soil Classification* available on CD-ROM.^[11]

The structure of the ASC has strong similarities with that of *Soil Taxonomy*, and although the emphasis is on field morphology, laboratory data have been used as appropriate. All technical terms are defined in a glossary. In a companion publication,^[10] further descriptions of diagnostic horizons and materials as well as the general occurrence of each order are given, except for Anthropeposols where data were unavailable. The major classes of each order and the use of major attributes in the subdivision of the orders are discussed. The main chemical properties used are pH and exchangeable cations, with pH commonly being determined in the field. A table is provided giving approximate correlations between ASC orders, three other Australian classifications, and *Soil Taxonomy* orders.

Perhaps the major benefit of the ASC is that it provides a means for efficient communication and a systematic framework for understanding and learning about the properties and inter-relationships of soils. Experience with the *Factual Key*^[3] suggested retention of the use of color and generalized texture profiles at high levels in the hierarchy of the new system, although formal justification of such decisions is often difficult to demonstrate. Another feature of the ASC that partly overcomes some of the problems associated with hierarchies is the use of family criteria at any level of the system.

New Zealand Soil Classification

With the advent of *Soil Taxonomy*, soil scientists of New Zealand became heavily committed to their international development, involving a 5-year testing program in New Zealand. As in Australia, difficulties became apparent, particularly with regard to its complexity. The results of the investigations showed that *Soil Taxonomy* made inadequate provision for important classes of New Zealand soils, particularly in the case of Inceptisols. Much of the new order of Andisols was based on New Zealand, where Guy Smith spent 12 months studying these and other soils. The inadequacies of *Soil Taxonomy* and the older New Zealand genetic classification to serve as an up-to-date national classification led to the development and publication of new material, largely through the efforts of Hewitt.^[7,12–15] The latest version^[13] has a helpful introductory section in which concepts, objectives, and principles are outlined. The scheme is a hierarchical one (order, group, subgroup, and series), with class distinctions based on diagnostic horizons and materials. Keys are provided to enable easy class

ALL SOILS

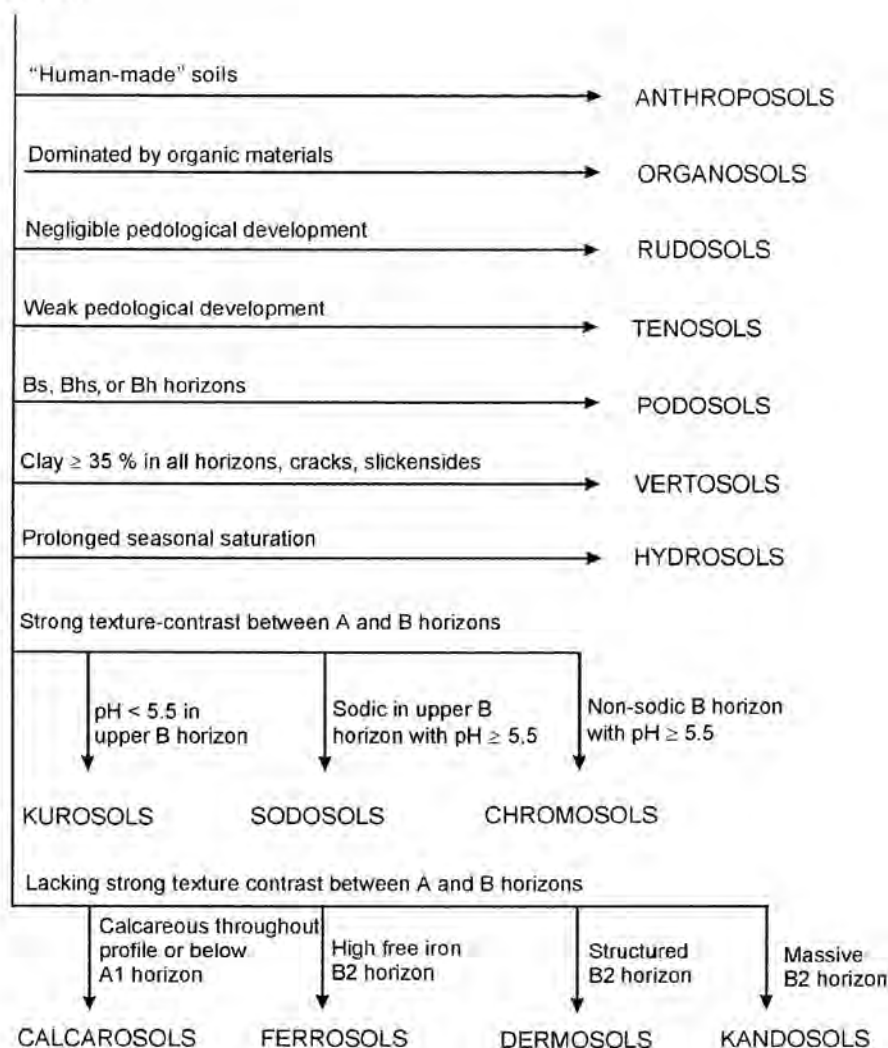


Fig. 1 Schematic summary of the 14 orders of the Australian soil classification.

Source: From Isbell, McDonald, et al.^[10]

Note: This figure is not to be used as a key.

definitions and identification. Total number of orders is 15: allophanic soils, anthropic soils, brown soils, gley soils, granular soils, melanic soils, organic soils, oxidic soils, pallic soils, Podzols, pumice soils, raw soils, recent soils, semiarid soils, and ultic soils. Accessory properties of the orders as well as concept, occurrence, and correlation with the earlier New Zealand Genetic Soil Classification and appropriate classes of *Soil Taxonomy* are also listed. The scheme is open-ended, and new classes can be incorporated into the hierarchy.

CONCLUSION

This brief review of the development of two national soil classification systems indicates the need for some taxa which are required for one country but not the other. Of the 15 New Zealand soil orders that have been defined, at least two (Allophanic Soils and Pumice Soils) are virtually absent in Australian soil landscapes. Conversely, some widespread Australian soils are only partly represented in

the New Zealand system. Particular examples are the widespread Australian semiarid soils characterized by profiles that feature a clear or abrupt change to a textural B horizon, which is frequently sodic. Finally, there are some soils common to both the countries that probably could be satisfactorily classified at the order level by the other system. A few such examples include arthropic soils and Arthroposols, gley soils and Hydrosols, organic soils and Organosols, oxidic soils and Ferrosols, and Podzols and Podosols.

REFERENCES

1. Isbell, R.F. A brief history of national soil classification in Australia since the 1920's. *Aust. J. Soil Res.* **1992**, *30*, 825–842.
2. Isbell, R.F. *The Australian Soil Classification*; CSIRO Publishing: Collingwood, 1996.
3. Northcote, K.H. *A Factual Key for the Recognition of Australian Soils*; 4th Ed.; Rellim Technical Publications: South Australia, 1979.

4. Stace, H.C.T.; Hubble, G.D.; Brewer, R.; Northcote, K.H.; Sleeman, J.R.; Mulcahy, M.J.; Hallsworth, E.G. *A Handbook of Australian Soils*; Rellim Technical Publications: South Australia, 1968.
5. Stephens, C.G. *A Manual of Australian Soils*; 3rd Ed.; CSIRO: Melbourne, 1962.
6. Taylor, N.H. *Soil Map of New Zealand, 1:2,027,520 Scale*; DSIR: Wellington, 1948.
7. Hewitt, A.E. Soil classification in New Zealand: Legacy and lessons. *Aust. J. Soil Res.* **1992**, *30*, 843–854.
8. Soil Survey Staff. *Soil Taxonomy, a Basic System of Soil Classification for Making and Interpreting Soil Surveys*; *USDA Agriculture Handbook No. 436*; U.S. Government Printing Office: Washington, 1975.
9. FAO. *World Reference Base for Soil Resources*; FAO World Soil Resources Report 84; Food and Agriculture Organization of The United Nations: Rome, 1998.
10. Isbell, R.F.; McDonald, W.S.; Ashton, L.J. *Concepts and Rationale of the Australian Soil Classification, ACLEP*; CSIRO Land and Water: Canberra, 1997.
11. Jacquier, D.W.; McKenzie, N.J.; Brown, K.L.; Isbell, R.F.; Paine, T.A. *The Australian Soil Classification: An Interactive Key*; CSIRO Publishing: Collingwood, 2001.
12. Hewitt, A.E. *New Zealand Soil Classification (Version 2.0)*; DSIR Division of Land and Soil sciences Technical Record DN 2; Department of Scientific and Industrial Research: Wellington, 1989.
13. Hewitt, A.E. *New Zealand Soil Classification; Landcare Research Science Series No. 1*; Manaaki Whenua Press: Lincoln, 1998.
14. Hewitt, A.E. *Methods and Rationale of the New Zealand Soil Classification; Landcare Research Science Series No. 2*; Manaaki Whenua Press: Lincoln, 1993.
15. Clayden, B.; Webb, T.H. *Landcare Research Science Series No. 1*; Manaaki Whenua Press: Lincoln, 1994.

Classification Systems: Brazilian

A.C.S. da Costa

M.R. Nanni

State University of Maringá, Maringá, Brazil

Abstract

Both technical and natural soil classification systems have been developed in Brazil for governmental and agricultural purposes, roads and building construction, erosion hazards, etc. Natural systems of classification are usually based on major intrinsic characteristics of the population being classified. Soils have their own genesis, morphology, physical, chemical, and mineralogical properties that can be used to distinguish and group them in classes with similar characteristics.

INTRODUCTION

Brazil comprises about 8.5 million km². Because of the large geographic area, diversity in climate (tropical to temperate), parent rock, and vegetation, it contains one of the largest collections of soils in the world. With the exception of the Andisols, all soil orders in the U.S. system of soil classification have been identified. The proper use of this diverse natural resource can only be accomplished if intrinsic properties are identified and used to distinguish groups of soils with similar behavior, leading to some sort of classification.

HISTORY

Soil classification systems used in Brazil, before 1947, were prepared by geologists and geomorphologists, rather than by pedologists, working in small groups without much exchange of information. The group that worked at the Agricultural Institute of Campinas probably did the earliest study on soils in Brazil. Based on this and other research, Setzer^[1] published the first soil survey of the state of São Paulo in 1949. This survey used a primitive system of soil classification that had almost no hierarchy. The class definitions were vague and difficult to use elsewhere. In this survey, soils were identified using local or regional names and grouped according to their parent rock and texture (Table 1).

The National Commission of Soils (CNS) and the Brazilian Soil Science Society (SBCS) were founded by a group of pedologists in 1947. The CNS was assigned responsibility for the inventory of soils all over the country, while the SBCS became responsible for the organization of a scientific journal, the Brazilian Journal of Soil Science, that would publish technical papers related to soil distribution, properties, and management.

After the works of Baldwin et al.^[2] and Thorp and Smith^[3] were published, the primitive soil classification system used in Brazil was abandoned and another was developed by the CNS for conducting soil surveys in the state of Rio de Janeiro^[4] and to reclassify soils in the state of São Paulo.^[5] This system employed a hierarchical arrangement, and the Great Group level was used as a reference to establish the most important taxonomic units. This was also the first time that diagnostic criteria were used to classify soils, especially diagnostic B horizons such as B latossólico and B textural, that are comparable, in part, to the Oxic and Argillic horizons of the U.S. soil classification system.^[6]

Soil classes such as Brunizém Avermelhado, Gleissolo, Solo Aluvial, Solonchak, Solonetz, Planossolo, and Podzólico Vermelho Amarelo, among others, have their origin in these entries and are identified in more than 90% of the soil maps of Brazil. Some of these classes are no longer used, but some others, such as the Latossolos, have been maintained.^[7]

The organization of soil surveys all over the country and the development of soil science groups in different research institutes and universities resulted in the definition of soil attributes necessary for the separation of soils at the Great Group level. Diagnostic characteristics included base saturation, clay activity, cation exchange capacity, sodium saturation, calcium carbonate content, and several other distinctive morphological characteristics. The criteria became more and more complex, and, with time, several new classes of soils were incorporated into the system (Table 2).

The National Soil Survey and Conservation Service (SNLCS), a federal agency that followed the National Commission of Soils, organized a Soil Classification Commission in 1978. Under the coordination of the pedologist Marcelo Nunes Camargo, a national project on the development of a revised Brazilian system of soil classification was initiated. This project resulted in the publication of

Table 1 Popular names and parent rock materials of soils in the primitive Brazilian system of soil classification.

Popular name	Parent rock material
Salmorão	Gneiss and schist
Massapé	Phyllite, micaceous schists
Terra Arenosa Branca	Sandstone
Terra Arenosa Catanduva	Sandstone
Terra Siálicosa	Sandstone and siliceous rocks
Terra Roxa de Campo	Sandstone and diabase
Terra Roxa Misturada	Diabase and sandstone
Terra Roxa Legítima	Diabase

Source: From Setzer.^[1]

three approximations in 1980, 1981, and 1988.^[7] In 1987, the SBCS also published a Special Bulletin^[8] where the Brazilian system of soil classification was summarized with the information gathered up to that time. This document was used as a standard reference for soil classification over the following decade.

From 1988 to 1995, the system suffered from neglect and a new strategy was proposed.^[9] In 1993, the SNLCS was reorganized as the National Centre for Soil Research (CNPq), one of the research centers of Empresa Brasileira de Pesquisa Agropecuária (EMBRAPA), the Brazilian federal agricultural research corporation, located in Rio de Janeiro. The new strategy involved the organization of a committee structure led by an Executive Committee, followed by a National Assistant Committee and Regional Committees. This organization permitted an active and effective participation of pedologists and other scientists interested in the development of the Brazilian System of Soil Classification. The fourth approximation, according to EMBRAPA,^[7] was based mostly on the third approximation (1988) of the system.

BRAZILIAN SYSTEM OF SOIL CLASSIFICATION

Development of a fourth approximation began in 1995. Information regarding soil profiles and classification were obtained by the Regional Committees installed throughout the country. For 2 years, the Regional Committees had several meetings under the supervision of the National Advisory Committee and the Executive Committee, which, in 1997, launched a fourth approximation during the XXVI Brazilian Soil Science Congress. At that time, the fourth approximation constituted a provisory model and was submitted to pedologists with restricted circulation for the purpose of receiving suggestions and corrections.

In 1999, after final revisions, a new Brazilian System of Soil Classification (SiBCS) was released. The SiBCS is a hierarchical, multicategorical, and open system that allows the inclusion of new soil classes.

Table 2 The introduction of new soil classes in the Brazilian system of soil classification.

Soil class	Year of definition	Explanation
Rubrozém	1956	Acid soils with humic surface horizon followed by a reddish B textural
Hidromórfico Cinzento	1958	Hydromorphic soils with a Gley and B textural horizons
Latossolo Amarelo	1959	Soils with Fe ₂ O ₃ content < 7%, brownish colors, no magnetic attraction, Ki ^a 1.5–2.2
Latossolo Bruno	1967	Soils with Fe ₂ O ₃ content > 11%, topsoil with dark brownish colors followed by reddish colors, virtually no magnetic attraction, Ki 0.2–2.2
Areias Quartzosas	1969	Soils with an AC profile derived from sandstone
Solos Litólicos	1970	Soils with AR or ACR profile
Podzólico Acinzentado	1973	Soils with a grayish B-textural horizon
Latossolo Variação Una	1977	Soils with Fe ₂ O ₃ content > 11%, dark brownish colors with no magnetic attraction, Ki 0.2–2.0
Podzólico Amarelo	1979	Soils with B textural, Fe ₂ O ₃ content < 7%, Ki < 2.2
Terra Bruna Estruturada	1979	Soils with brownish colors, Fe ₂ O ₃ > 10%, no magnetic attraction, Ki 1.70–2.10
Plintossolo	1980	Soils with Plintic horizon
Latossolo Ferrífero	1982	Soils with Fe ₂ O ₃ content > 36%, strong magnetic attraction, Ki 0.06–0.9
Podzólico Vermelho-escuro	1982	Soils with a B textural, Fe ₂ O ₃ content ≤ 15%, TiO ₂ ≤ 1.70

^aKi is the number of moles of SiO₂/number of moles of Al₂O₃, determined in the < 2 mm soil fraction after total acid dissolution.

Source: From National Center for Soil Inquiries.^[7]

This system maintains 13 classes at the first categorical (order) level, as defined in the third approximation system, and also creates a new class—NITOSSOLOS.^[7] The soil orders are presented in alphabetical order without relation to their extent in the Brazilian territory in Table 3. The other levels are designated as suborder (second level), great group (third level), and subgroup (fourth level).

Some orders include soils that previously constituted separate classes in previous classification systems (EMBRAPA).^[7] As an example, the NEOSSOLOS contain soils formerly designated as Regossolos, Solos Litólicos, Solos Aluviais, and Areais Quartzosas (Table 4).

The new system also incorporates two more categorical levels: level 5 (families) and level 6 (series), for which no

Table 3 The 14 soil orders in the Brazilian system of soil classification and the number of units at the suborder, great group, and subgroup levels.

Order	Suborder	Great group	Subgroup	Soil survey staff	FAO
Alissolos	2	5	12	Ultisols	Alisols
Argissolos	4	10	85	Ultisols/Alfisols	Acrisols
Cambissolos	3	17	69	Inceptisols	Cambisols
Chernossolos	4	10	31	Mollisols/Alfisols	Chernozems/Phaeozems
Espodossolos	2	6	27	Spodosols	Podzols
Gleissolos	4	15	53	Variable	Gleysols
Latossolos	4	22	90	Oxisols	Ferralsols
Luvissolos	2	5	23	Alfisols	Luvisols
Neossolos	4	18	59	Inceptisols/Entisols	Regosols
Nitossolos	2	7	22	Alfisols/Ultisols	Nitisols
Organossolos	4	9	42	Histosols	Histosols
Planossolos	3	10	44	Inceptisols/Alfisols/Ultisols	Planosols
Plintossolos	3	8	28	Variable	Plinthosols
Vertissolos	3	11	40	Vertisols	Vertisols
Total	44	153	625		

Note: Closest approximation of the Brazilian soil orders to the American and FAO systems of soil classification.

Source: From Oliveira,^[9] Soil Survey Staff,^[10] and FAO, ISRIC.^[11]

Table 4 Comparison of the soil classification system used by EMBRAPA before and after 1998.

Brazilian system of soil classification	Previous classification of the soils
Alissolos	Rubrozem, Podzólico Bruno-Acinzentado Distrófico or Álico, Podzólico Vermelho-Amarelo Distrófico or Álico Ta, and some Podzólicos Vermelho-Amarelos Distróficos or Álicos Tb
Argissolos	Podzolic Vermelho-Amarelo Tb, part of the Terra Roxa Estruturada, Terra Roxa Estruturada Similar, Terra Bruna Estruturada and Terra Bruna Estruturada Similar, and the Podzólico Vermelho-Escuro Tb with B textural and the Podzólico Amarelo
Cambissolos	Cambissolos Eutróficos, Distróficos, and Álicos. Cambissolos Eutróficos, Distróficos, and Álicos
Chernossolos	Brunizem, Rendzina, Brunizem Avermelhado, and Brunizem Hidromórfico
Espodossolos	Podzol including Podzol Hidromórfico
Gleissolos	Glei Pouco Húmico, Glei Húmico, part of the Hidromórfico Cinzento, Glei Tiomórfico, and Solonchak
Latossolos	Latossolos except the Latossolos Plinticos

(Continued)

Table 4 Comparison of the soil classification system used by EMBRAPA before and after 1998. (Continued)

Brazilian system of soil classification	Previous classification of the soils
Luvissolos	Bruno Não Cálculo, Podzólico Vermelhoamarelo Eutrófico Ta, Podzólico Bruno-Acinzentado Eutrófico, and the Podzólicos Vermelho-Escuros Eutróficos Ta
Neossolos	Litossolos, Solos Litólicos, Regossolos, Solos Aluviais, and Areias Quartzosas
Nitossolos	Terra Roxa Estruturada, Terra Roxa Estruturada Similar, Terra Bruna Estruturada, Terra Bruna Estruturada Similar, some Podzolicos Vermelho-Escuros Tb, and some Podzólicos Vermelho-Amarelos Tb
Organossolos	Solos Organicos, Solos Semiorgânicos, Solos Tiomórficos Turfosos, and part of the Solos Litólicos Turfosos
Planossolos	Planossolos, Solonetz-Solodizado, and Hidromórficos Cinzentos
Plintissolos	Lateritas Hidromórficas, part of the Podzólicos Plínticos, part of the Glei Húmico, and Glei Pouco Húmico Plínticos and some of the Latossolos Plínticos
Vertissolos	Vertissolos

Source: From National Center for Soil Inquiries^[7] and EMBRAPA and Oliveira.^[9]

members have been defined. According to Oliveira,^[9] there is a project of CNPS to establish the classes at both levels.

NOMENCLATURE OF THE BRAZILIAN SYSTEM OF SOIL CLASSIFICATION

The names of the soil orders are formed by the association of a formative element with the termination “solo.” Table 5 presents the names of the classes, their respective formative elements, and meanings.

The edition of the SiBCS uses the definitions, horizon notations, and basic knowledge of morphological characteristics used in earlier approximations. However, in the new system, there are differences in many of the diagnostic

criteria and parameters. As an example, diagnostic attributes such as Ácrico, Álico, Epiáquico, Ebânico, Crômico, and Hipocrômico have been created.

CONCLUSION

The SiBCS, as in previous approximations, consists of an open system with possibilities for change. It is expected that this system will experience alterations in editions as the system is used and criticisms are evaluated.

REFERENCES

1. Setzer, J. *Os Solos do Estado de São Paulo*; Instituto Brasileiro de Geografia e Estatística, Conselho Nacional de Geografia: Rio de Janeiro, 1949; 1–387.
2. Baldwin, M.; Kellog, C.E.; Thorp, J. *Soil Classification. Soils and Men*; United States Department of Agriculture: Washington, 1938; 979–1001.
3. Thorp, J.; Smith, G.D. Higher categories of soil classification: Order, suborder and great groups. *Soil Sci. Baltimore* **1949**, *67*, 177–226.
4. Ministério da Agricultura. Centro Nacional de Ensino e Pesquisa agrônômicas. In *Levantamento de Reconhecimento dos Solos do Estado do Rio de Janeiro e Distrito Federal*; Boletim; SNPA: Rio de Janeiro, 1958; Vol. 11, 1–305.
5. Ministério da Agricultura. Centro Nacional de Ensino e Pesquisa Agrônômicas. In *Levantamento de Reconhecimento dos Solos do Estado de São Paulo*; Boletim; SNPA: Rio de Janeiro, 1960; Vol. 12, 1–634.
6. Soil Survey Staff. *Soil Taxonomy, A Basic System of Soil Classification for Making and Interpreting Soil Surveys*; United States Department Agriculture, Handbook No. 436; U.S. Government Printing Office: Washington, 1975; 1–754.
7. National Center for Soil Inquiries. *Sistema Brasileiro de Classificação de Solos*; EMBRAPA: Rio de Janeiro, 1999; 1–412.
8. Camargo, M.N.; Klamt, E.; Kauffman, J.H. Classificação de solos usada em levantamentos pedológicos no Brasil. *Bol. Inf. Soc. Bras. Cienc. Solo* **1987**, *12* (1), 11–33.
9. Oliveira, J.B. O novo sistema brasileiro de classificação de solos. *Agronomico* **2001**, *53* (1), 8–10.
10. Soil Survey Staff. *Keys to Soil Taxonomy*; Agency for International Development; United States Department Agriculture; Soil Conservation Service; Soil Management Support Services; Technical Monograph; Pocahontas Press: Blacksburg, 1994; Vol. 19, 1–541.
11. FAO, ISRIC. *ISSS World Reference Base for Soil Resources; World Soil Resources*. World Soil Resources Report; FAO: Rome, 1998; Vol. 84, 1–88.

Table 5 Classes, formative elements, and meanings of terms used at the order level of the SiBCS.

Soil class	Formative elements	Significance of the terms
Neossolo	NEO	New. Little development
Vertissolo	VERTI	Vertic. Vertico horizon, high shrink swell
Cambissolos	CAMBI	Cambiar. Incipient B horizon
Chernossolo	CHERNO	Chernozemic. Black, rich in bases
Luvissolo	LUVI	Saturated. Accumulation of clays with high chemical activity (Ta ^a)
Alissolo	ALI	Aluminic. High Al content
Argissolo	ARGI	Podzolic. B-textural horizon, clays with low activity (Tb ^b)
Nitossolo	NITO	Nitic. B-nítico horizon ^c
Latossolo	LATO	Latosol. B-latossólico horizon ^d
Espodossolo	ESPODO	Spodos. Spodic B horizon
Planossolo	PLANO	Planic. B-plânico horizon ^e
Plintossolo	PLINTO	Plinthite. Plinthic horizon
Gleissolo	GLEI	Gley. Glei horizon
Organossolo	ORGANO	Organic. H or O horizons

^aCation exchange capacity > 27 cmol/kg of clay.

^bCation exchange capacity ≤ 27 cmol/kg of clay.

^cMineral subsurface horizon, clay, or heavy clay texture, small increase in clay content with depth.

^dMineral subsurface horizon with high degree of weathering with < 4% primary minerals or 6% muscovite, thickness > 50 cm, and cation exchange capacity < 17 cmol/kg of clay.

^eMineral subsurface horizon with abrupt textural change between A or E to B horizon, very dense, high percentage of dispersible clay, low water permeability, dark or grayish soil colors.

Source: From National Center for Soil Inquiries.^[7]

Classification Systems: Canadian

Charles Tarnocai

Agriculture and Agri-Food Canada, Ottawa, Ontario, Canada

Abstract

The Canadian classification system recognizes 10 soil orders based on soil properties that reflect the dominant soil-forming processes. These soil orders are divided into great groups based on the differentiating criteria of the order to which they belong. The great groups are subdivided into subgroups based on the arrangement of the soil horizons. Finally, the subgroups are subdivided into families and the families into series. The type and degree of development of the soil horizons within the pedon, the basic soil unit, are used to determine the classification of the soil.

INTRODUCTION

Classifying soils in Canada began with the first soil survey in Ontario in 1914. This classification was based largely on geological material and texture. The first taxonomic system of soil classification was developed in 1955 and is the basis of the soil classification used. This classification has been considerably revised as additional soil information has become available, resulting in both more precise definitions to the taxa at all levels and increasing emphasis on soil properties as taxonomic criteria.

The system recognizes 10 soil orders based on soil properties that reflect the dominant soil-forming processes. These soil orders are divided into great groups based on the differentiating criteria of the order to which they belong. The great groups are subdivided into subgroups based on the arrangement of the soil horizons. Finally, the subgroups are subdivided into families and the families into series. The type and degree of development of the soil horizons within the pedon, the basic soil unit, are used to determine the classification of the soil.

BASIS OF THE SOIL CLASSIFICATION

Pedon, The Basic Soil Unit

The pedon is the basic soil unit used to classify Canadian soils.^[1] It is defined as the smallest 3-D body of soil that has lateral dimensions large enough to include representative variations in the shape and relation of soil horizons and in composition of the soil.^[2] The lateral dimension of the pedon is 1 m, if the soil horizons vary within this distance. If the soil horizons are cyclical or intermittent within a lateral distance of 2–7 m, the lateral dimension of the pedon is 1–3.5 m (half of the cycle). The vertical dimension of the pedon is the depth of the control section.

Although the pedon concept applies to the classification of all soils, it is especially relevant to soils exhibiting cyclic variation, such as Cryosols, Vertisols, and forest soils having hummocky microtopography due to blow-down of trees. Examples of pedons for these soils are given in Figs. 1–3, respectively.

Soil Horizons

Soil horizons are basic to the soil classification because the taxa are defined primarily by the kinds, degree of development, and sequence of soil horizons and layers in a

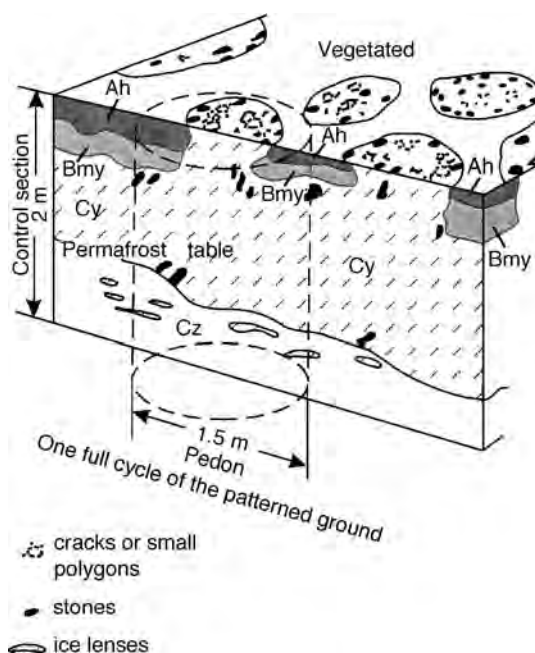


Fig. 1 Pedon of Orthic Turbic Cryosol in area of non-sorted circles.

Source: From Soil Classification Working Group.^[1]

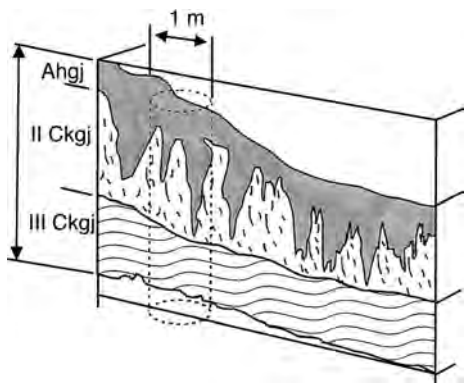


Fig. 2 Pedon of Gleyed Vertic Black Chernozem with tonguing Ah horizon.
Source: From Soil Classification Working Group.^[1]

pedon. The major mineral soil horizons are designated A, B, and C. The major organic horizons, which are derived primarily from well to imperfectly drained upland forest vegetation at various stages of decomposition, are designated L, F, and H, and those derived mainly from poorly to very poorly drained wetland vegetation are designated O. The non-soil layers are designated R (rock) and W (water). Brief descriptions of these major horizons and layers are given in Table 1. Subdivisions of the major soil horizons are labeled by adding lowercase suffixes (e.g., Ah, Bm, and Bt). Brief descriptions of some lowercase suffixes are presented in Table 2.

CRITERIA FOR DEFINING TAXA

The criteria used to differentiate the taxa at the various levels are based on soil properties and the soil-forming processes. The criteria for the various hierarchical levels from highest to lowest are as follows:

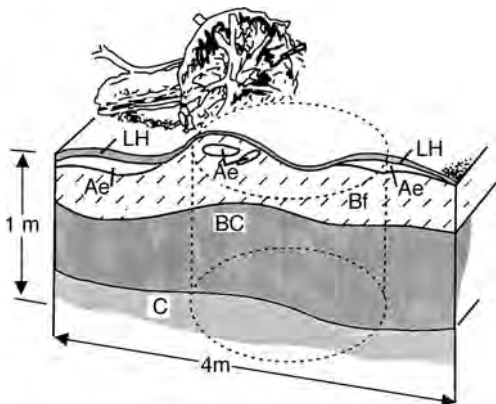


Fig. 3 Pedon of Orthic Humo-Ferric Podzol, turbic phase, in hummocky terrain resulting from blowdown of trees.
Source: From Soil Classification Working Group.^[1]

Table 1 Descriptions of mineral and organic soil horizons and mineral layers.

Horizon or layer	Description
Mineral horizons and layers	
A	A horizon formed at or near the surface in the zone of accumulation of organic matter (Ah) or leaching of materials by solution (Ae).
B	A horizon characterized by enrichment in organic matter (Bf), sesquioxides (Bf), or clay (Bt); or by development of soil structure and/or color (Bm); or by significant amounts of sodium (Bn); or by mottling and gleying (Bg).
C	A horizon (parent material) comparatively unaffected by pedogenic processes, except for gleying (Cg) and accumulation of calcium (Cca).
R	A consolidated bedrock layer.
W	A water (hydric) layer that may occur in organic, gleysolic, and cryosolic soils. It occurs in the form of ice (Wz) in cryosolic soils.
Organic horizons	
L	A well to imperfectly drained, undecomposed horizon, composed primarily of leaves, twigs, and woody materials in which the original structure is easily recognizable.
F	A well to imperfectly drained, moderately decomposed horizon in which the original structures are difficult to recognize.
H	A well to imperfectly drained, well-decomposed horizon in which the original structure is unrecognizable.
O	A poor to very poorly drained horizon derived primarily from wetland vegetation that is subdivided on the basis of the degree of decomposition into undecomposed (Of), moderately decomposed (Om), well-decomposed (Oh), and limnic (Oco) materials.

Source: From Soil Classification Working Group.^[1]

- Order: Taxa are based on properties of the pedon that reflect the soil environment and the dominant soil-forming processes.
- Great group: Taxa are differentiated on the basis of soil properties that reflect differences in the strengths of dominant processes, or a major contribution of a process in addition to the dominant one.
- Subgroup: Taxa are differentiated on the basis of the kind and arrangement of soil horizons that indicate conformity to the central concept of the great group (e.g., Orthic).
- Family: Taxa are differentiated on the basis of characteristics of the parent material (e.g., particle size, mineralogy, calcareousness, acidity, and soil climatic factors).
- Series: Taxa are differentiated on the basis of properties of the pedon. Pedons belonging to a series have

Table 2 Descriptions of the most common lowercase suffixes for mineral and organic soil horizons.

Lowercase suffix	Description
Associated with mineral horizons	
ca	Enriched by secondary carbonate.
e	Eluviated of clay, Fe, Al, or organic matter.
f	Enriched with amorphous material, mainly Al and Fe.
g	With gray color (gleyed), mottling, and reduced conditions.
h	Enriched with organic matter.
m	Slightly altered by hydrolysis and/or oxidation to produce a change in color and/or structure.
n	With high Na content and columnar or prismatic structure.
p	Affected by human activity (e.g., cultivation or logging).
t	Enriched with silicate clay.
v	Affected by vertic turbation because of shrinking and swelling clay.
y	Affected by cryoturbation.
z	Affected by permafrost (perennially frozen).
Associated with organic horizons	
f	Consisting of undecomposed organic material (rubbed fiber content >40%).
m	Consisting of moderately decomposed organic material (rubbed fiber content 10–40%).
h	Consisting of well-decomposed organic material (rubbed fiber content <10%).
co	Consisting of coprogenous earth deposited by aquatic organisms.

Source: From Soil Classification Working Group.^[1]

similar kinds and arrangements of soil horizons whose color, texture, structure, consistence, thickness, reaction, and composition fall within a narrow range.

OUTLINE OF THE CANADIAN SYSTEM OF SOIL CLASSIFICATION

Brief descriptions of the 10 soil orders, representing the highest level of the Canadian system of soil classification,^[1] are presented in [Table 3](#), and detailed descriptions are given by Soil Classification Working Group^[1] and examples are shown in [Figs. 4–11](#).

In mineral soils, the central concept of the great group is represented by the Orthic subgroup. The other subgroups within the great group are named according to the kind and arrangement of soil horizons (e.g., Eluviated—presence of an Ae horizon), intergradation

Table 3 Brief descriptions of the soil orders of the Canadian system of soil classification.

Order	Description
Brunisolic	Soils with a Bm horizon, indicating weak to moderate soil development. They occur in a wide range of climatic and vegetation conditions including forest, shrub, grassland, and alpine.
Chernozemic	Soils with an Ah horizon >10 cm thick, color value darker than 5.5 dry and 3.5 moist, chroma <3.5 moist, and >80% base saturation where Ca is the dominant cation. They are restricted to soils with mean annual soil temperatures of 0°C or higher and soil moisture regimes dryer than humid. They are associated with grassland vegetation.
Cryosolic	Soils formed on either mineral or organic materials that have permafrost within 1 m of the surface, or within 2 m if the pedon is strongly cryoturbated. They occur in Boreal and Arctic regions.
Gleysolic	Soils with gleyed horizons and horizons with mottles, indicating the influence of periodic or sustained reducing conditions. They are associated with meadow and wetland vegetation.
Luvisolic	Soils with light-colored eluvial (Ae) horizons and B horizons in which clay has accumulated (Bt). They are associated with forest vegetation.
Organic	Soils that contain >17% organic carbon by weight and have developed on organic materials associated with peatlands or on deep upland humus (folic) of forest origin.
Podzolic	Soils with a B horizon in which the dominant accumulation product is composed mainly of humified organic material combined with Al and Fe. They develop mainly on coarse- to medium-textured acid parent materials, under forest or heath vegetation in cool to very cold, humid to perhumid climates.
Regosolic	Weakly developed soils that, thus, have only thin B horizons, if present. They occur under a wide range of vegetation and climates.
Solonetzic	Soils that have developed on saline materials and have Bn horizons that are very hard when dry and swell to a sticky mass when wet. They are associated with saline-tolerant vegetation.
Vertisolic	Soils that have vertic Bv horizons and occur in heavy-textured materials (=60% clay) that have shrink–swell characteristics. They are associated with grasslands.

Source: From Soil Classification Working Group.^[1]

toward soils of another order (e.g., Gleyed and Brunisolic), or additional special features (e.g., Turbic—presence of a Bmy horizon and Vertic—presence of Bv or Bvg horizons).



Fig. 4 Eutric Brunisol with a thin eluvial horizon (Ae). Boreal forest area, Mackenzie Valley, Northwest Territories.

Source: Photo by Charles Tarnocai.



Fig. 6 Turbic Cryosol associated with an earth hummock. Note the ice layers in the lower part of the profile. North Slope, Yukon Territory.

Source: Photo by Charles Tarnocai.

In organic soils, the subgroups are defined by the depth to the mineral contact (typic and terric), the degree of decomposition of the organic materials (e.g., fibric, mesic, and humic, except for Folisols, which are hemic, humic, and Histic), or special features (e.g., hydric and lignic). Detailed descriptions of the subgroups are given by Soil Classification Working Group.^[1]



Fig. 5 Black Chernozem with a thick, black, organic-rich surface horizon (Ah). Grasslands, Saskatchewan.

Source: Photo by Charles Tarnocai.



Fig. 7 Gray Luvisol with a brownish Bt horizon. Forested area, Manitoba.

Source: Photo by Robert G. Eilers, Agriculture and Agri-Food Canada, Manitoba.



Fig. 8 Mesisol associated with a deep peat deposit. Boreal forest area, Manitoba.

Source: Photo by Charles Tarnocai.



Fig. 9 Humo-Ferric Podzol with a reddish Bf horizon enriched by iron and organic matter. Princeton area, British Columbia.

Source: Photo by Elizabeth Kenney, Agriculture and Agri-Food Canada, British Columbia.



Fig. 10 Solonetzic soil with columnar structure and a high salt-content soil horizon (Bn). Southeastern Saskatchewan.

Source: Photo by Robert G. Eilers, Agriculture and Agri-Food Canada, Manitoba.

CORRELATION WITH OTHER SOIL CLASSIFICATION SYSTEMS

The approximate equivalents of the Canadian taxa to those of the American system^[3] and the WRB, an international system,^[4] are given in by Soil Classification Working Group (p. 155).^[1]

The closest equivalent Canadian and American taxa are the organic soils (Canada) and Histosols (U.S.A.) and

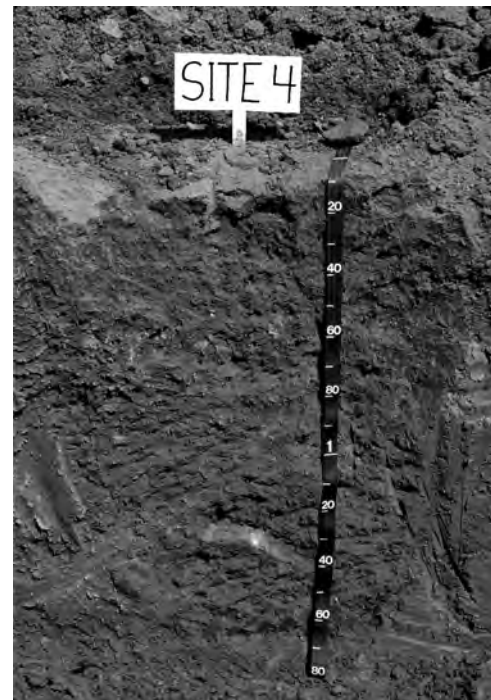


Fig. 11 Vertisol developed on clay soils in southern Saskatchewan. Note the slickenside at a depth of ~40 cm.

Source: Photo by Charles Tarnocai.

the cryosolic soils (Canada) and Gelisols (U.S.A.). The Canadian great groups and American suborders of these two orders are nearly equivalent. For the remaining orders, however, differences in taxa are greater. For example, most of the chernozemic soils (Canada) are Mollisols (U.S.A.), but some are Aridisols, Gleysols, or Solonetz (U.S.A.).

There is a great deal of similarity between the Canadian and WRB taxa at the highest level. The closest equivalents are for the Solonetzic, Luvisolic, Podzolic, Regosolic, Gleysolic, and Vertisolic orders (Canada). The corresponding equivalents in the WRB are the Solonetz, Luvisol, Podzol, Regosol, Gleysol, and Vertisol major soil units. At the lower levels, however, differences between the two systems are greater.

CONCLUSION

The Canadian system of soil classification,^[1] which was published in 1998, is based on a system that was established in 1955. This hierarchical system consists of five categories, or levels—order, great group, subgroup, family, and series—with the classification of soils at all levels being based on soil properties. As more soil information becomes available, *The Canadian System of Soil Classification*^[1] is revised to give more precise definitions to the taxa at all levels, with new orders being added as necessary.

With the development of continental and multicontinental databases, it is increasingly necessary to harmonize the soil classifications used by various countries into a more uniform, usable form. In addition, as new soil information becomes available both nationally and internationally, the various classification systems, including the Canadian system, need to be continually updated to ensure they meet the needs of the time and reflect the state of the scientific knowledge.

REFERENCES

1. Soil Classification Working Group. *The Canadian System of Soil Classification*, 3rd Ed.; Agriculture and Agri-Food Canada Publication No. 1646 (Revised); NRC Research Press: Ottawa, 1998; 1–187. Available at http://sis.agr.gc.ca/cansis/references/1998sc_a.html (accessed September 2003).
2. Soil Survey Staff. *Soil Taxonomy*. U.S. Dept. of Agriculture Handbook No. 436; U.S. Government Printing Office: Washington, 1975; 1–754.
3. Soil Survey Staff. *Keys to Soil Taxonomy*, 8th Ed.; U.S. Department of Agriculture, Natural Resources Conservation Service: Washington, 1998; 1–326.
4. ISSS Working Group RB. *World Reference Base for Soil Resources*; International Society of Soil Science (ISSS), International Soil Reference and Information Centre (ISRIC) and Food and Agriculture Organization of the United Nations (FAO) World Soil Resources Report 84; FAO: Rome, 1998; 1–91.

Classification Systems: Chinese

Zi-Tong Gong

Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China

Abstract

Chinese Soil Taxonomy (CST) is the taxonomic soil classification of China after a long history of related research from ancient to modern times and peculiarized with several main points concerning *Anthrosols*, *Aridosols*, *Ferrosols*, *Alpine Region soils*, their nomenclature, and mapping. CST can be correlated to worldwide popular systems like *Soil Taxonomy* and World Reference Base.

HISTORY

Soil Classification in Ancient China

China is one of the most ancient civilizations in the world with a long history of agricultural activities. In about 2500 B.P., the earliest soil classification in China appeared in two books, *Yugong* and *Guanzi Diyuan pian*.^[1]

Soil Classification in Modern China

It can be subdivided into the following three main stages: 1) In the 1930s, the concept of soil great groups was adopted and more than 2000 soil series were established by combining C.F. Marbut's system with Chinese practice; 2) in 1954, a Chinese soil classification was formed by following the Russian system that was further developed throughout the first National General Soil Survey (1958–1961), modified in 1978 and then accepted widely as the official system used in the Second Soil Survey (1979–1994); 3) with the increasing international exchanges, Soil Taxonomy (ST) and Food and Agriculture Organization of the United Nations–United Nations Educational, Scientific and Cultural Organization Soil Map Legend were introduced into China beginning in the 1980s. A research group of 35 universities and institutions led by ISSAS step by step, worked out a soil taxonomic classification of China—Chinese Soil Taxonomy (CST).^[2]

SPECIFIC FEATURES

Diagnostic Horizons

There are 11 diagnostic surface horizons grouped as organic, humic, anthropic, and crustic epipedons. They consist of histic, mattic, mollic, umbric, ochric, siltigic, cumulic, fimic, anthrostatic, aridic epipedons, and salic crust. There are 20 diagnostic subsurface horizons, namely

albic, glossic, cambic, ferralic, lower activity caly (LAC)-ferric, plinthic, spodic, agric, hydragric, argic, salic, sulfuric, alkaline, hypersalic, gypsic, calcic, hypercalcic horizons, and special cemented pans: clay-pan, calcipan, phosphipan, and salipan.

Diagnostic Characteristics

There are diagnostic characteristics setup in CST. They are organic soil materials, lithologic characters, lithic contact, paralithic contact, anthro-silting materials, vertic features, anthroturbic layer, soil moisture regimes, gleyic features, redoxic features, soil temperature regimes, permafrost layer, frost-thawic features, n value, isohumic property, humic property, andic property, ferric property, allitic, property, phosphic property, sodic property, calcareous property, base saturation, and sulfuric materials.

Anthrosols

Based on anthropogenic horizons such as an anthrostatic epipedon and a hydragric horizon, a fimic epipedon and phos-agric horizon, and a siltigic epipedon or cumulic epipedon, Anthrosols are grouped into static Anthrosols, fimi-orthic Anthrosols, cumuli-orthic Anthrosols, and siltigi-orthic Anthrosols.^[3–5]

Aridosols

Aridosols are classified by the aridic epipedon and one or more of the following diagnostic horizons: salic, hypersalic, salipan, gypsic, hypergypsic, calcic, hypercalcic, calcipan, argic, and cambic horizon whose upper boundaries are within 100 cm of the mineral soil surface.^[6]

Ferrosols

In tropical and subtropical China, Ferrosols are based on LAC-ferric subhorizon that has lower activity, clays, and

Table 1 Correlation between CST, ST, and WRB systems.

CST ^[1]	ST ^[9]	WRB ^[10]
Histosols	Histosols	Histosols
Anthrosols	—	Anthrosols ^a
Spodosols	Spodosols	Podosols
Andosols	Andosols	Andosols, ^a Cryosols ^b
Ferrosols	Oxisols	Ferralsols, ^a Plinthosols, ^b Acrisols ^b
Vertosols	Vertosols	Vertisols
Aridosols	Aridosols	Calcisols, Gypsisols
Halosols	Aridosols, ^b Alfisols, ^b Inceptisols ^b	Solonchaks, Solonetz
Gleyosols	Inceptisols, Gelisols, Entisols	Gleysols, ^a Cryosols ^b
Isohumosols	Mollisols	Chernozems, Kastanozems, Phaeozems
Ferrosols	Ultisols, ^a Alfisols, ^b Inceptisols ^b	Acrisols, ^a Lixisols, ^b Plinthosols, ^b Nitisol ^b
Argosols	Alfisols, ^a Ultisols, ^b Mollisols ^b	Luvisols, ^a Alisols ^b
Cambosols	Inceptisols, ^a Mollisols, ^b Gelisols ^b	Cambisols ^a
Primosol	Entisols, ^a Gelisols ^b	Fluvisols, Leptosols, Arenosols, Regosols, Cryosols

^aMostly.

^bPartly corresponding.

rich in free iron oxides [cation exchangeable capacity (CEC₇) 16–24 cmol(+)/kg⁻¹; (free iron/total iron) = 0.4 or more] with a total area of 2 million square kilometer.

Alpine Region Soils

Tibet region is called the third polar, which is cold and dry. The soils there are classified as cryic Aridisols that have a cryic soil temperature regime, and celic Cambosols that have a cryic or colder soil temperature regime and have frost-thawic features.

Nomenclature and Mapping

Similar to ST, the principle of a segmental continuous name has been adapted in CST. The first segment includes order,

suborder, group, and subgroup. The family name provides additional information on particle size class, mineral composition, and soil temperature regime. The Series is a second segment named independently. A 1:12 M scale National Soil Map of China based upon CST is already available.

CORRELATION

CST has become gradually known to the world since being proposed by the Chinese Soil Science Society in 1996, translated into Japanese in 1997,^[7] adopted by World Reference Base (WRB) especially for the Anthrosols in 1998, and introduction in the *Handbook of Soil Science*^[8] in 1999. The correlation between CST, ST, and WRB systems is shown in Table 1.

REFERENCES

1. Gong, Z.T. *Chinese Soil Taxonomy (in Chinese)*; Science Press: Beijing, 1999; 1–903.
2. Cooperative Research Group on Chinese Soil Taxonomy. *Chinese Soil Taxonomy*; Science Press: Beijing, 2001; 1–203.
3. Gong, Z.T. Pedogenesis of paddy soil and its significance in soil classification. *Soil Sci.* **1983**, 35 (1), 5–10.
4. Gong, Z.T.; Zhang, G.; Luo, G. Diversity of anthrosols in China. *Pedosphere* **1999**, 9 (3), 193–204.
5. Gong, Z.T.; Zhang, X.; Luo, G.; Shi, H.; Spaagaren, O. Extractable phosphorus in soil with fmic epipedon. *Geoderma* **1997**, 75, 289–296.
6. Eswaran, H.; Gong, Z.T. Properties, classification and distribution of soil with gypsum. In *Occurrence, Characteristics and Genesis of Carbonate, Gypsum and Silica Accumulation in Soil*; Nettleton, W.D., Ed.; Special Publication No. 26; SSSA: Madison, 1991; 89–119.
7. Cooperative Research Group on Chinese Soil Taxonomy (Revised proposal) (in Japanese). JIRCAS Working Re No. 6; JIRCAS: Ibaraki, 1997; 1–128.
8. Sumner, M.E., Ed. *Handbook of Soil Science*; CRC Press: Boca Raton, 1999; 161–166.
9. USDA. *Soil Taxonomy*, 2nd Ed.; Agriculture Handbook No. 436; Government Printing Office: Washington, 1999; 1–869.
10. FAO; ISRIC; ISSS. *World Reference Base for Soil Resources*; World Soil Resources Reports 84; FAO: Rome, 1998; 1–68.

Classification Systems: French

Jean-Paul Legros

Environment and Agronomy Department, National Institute of Agronomic Research (INRA), Montpellier, France

Abstract

This entry summarizes very briefly the evolution of the different systems used to classify the soils in France through the 20th century. It describes the French *Référentiel Pédologique* in comparison with the World Reference Base. It indicates similarities and differences and presents the comparison between both the systems.

INTRODUCTION

France is a very small country. But it was previously the head of an empire covering a part of Africa and Asia. During the International Exhibition of 1900, in Paris, Dokuchaev explained his concepts concerning pedology, attracting the interest of the French scientists on this subject. The two facts explain the interest of French scientists in soil classification.

BRIEF HISTORY

From the beginning of pedology, the French scientists used genetic classifications of soils in relation to the work of the Russian school of Dokuchaev (1846–1903) and Glinka (1867–1927). For example, Valérien Agafonoff (1863–1955), a Russian soil scientist, came to Paris fleeing his country during the revolution of 1917. He built one of the first soil maps of France with the corresponding legend (created in 1928 but published mainly in 1936 after improvements).^[1–3] Using the previous and successive works of Lagatu (1862–1942),^[4,5] Demolon (1881–1954), Kubiěna (1897–1970), Erhart (1898–1982), and several German authors, Georges Aubert (born in 1913) and Philippe Duchaufour (1912–2000) presented a first French classification in Paris, in 1956, during the Sixth International Congress of Soil Science. A second version of this system was presented in 1962 and is known as “Aubert–Duchaufour classification.” In 1960, Duchaufour published his famous *Précis de Pédologie*,^[6] which popularized his classification system to a much wider audience. Then, in 1967, the whole French scientific community collaborated and completed the system which became the “Classification of the CPCS” (*Commission de Pédologie et de Cartographie des Sols*). This document was used

for 20 years and popularized a vocabulary used in the ordinary pedological language of French scientists, e.g., *sol brun*, *ranker*, *Rendzine*, etc.

During this period, the U.S. Department of Agriculture system and the Food and Agriculture Organization Soil Legend were developed (1960 and 1981). Simultaneously, the development of computer science and statistical science demonstrated that the soils could be classified in a more objective way by searching for the mathematical similarities between taxa and objects rather than trying to enter a soil in a more or less adapted taxon of a rigid classification system. In this context, since 1986, the French community met, worked, and presented a first version of *Référentiel Pédologique* (RP) in 1992. A second version was produced by Baize and Girard,^[7] with an English translation done by Baize,^[8] and followed by Italian and Russian translations (2000). Because several French scientists were involved in the development of both the new French trials and the World Reference Base (WRB),^[9] the two systems have many similarities in both philosophy and organization, although not necessarily in the details of vocabulary. Thus, it is logical to review the RP in comparison with the WRB.

COMPARING THE RP AND THE WRB

The following are the principal points of comparison:

- The 40 diagnostic horizons of the WRB are replaced by the 73 “Horizons de Référence” of the RP.
- The 30 reference soil groups of the WRB are replaced by the 30 “Grands Groupes de Références” (major groups of references or “GER”). But these 30 categories do not match exactly because, in the French system, the tropical soils are not considered.

This last point is regretted^[10] (10), considering the large vast experience of French scientists in Africa. These 30 *Groupes* are divided into 102 “Sols de Référence.”

- The WRB “diagnostic properties,” “diagnostics materials,” “formative elements,” and “prefixes” correspond to the “qualificatifs” (qualifiers) of the RP. In the RP, all these elements are grouped in a single list of 235 terms.

The more original point of the RP is that a diagnostic horizon alone is in general not sufficient to recognize a Référence. Several diagnostic horizons are associated to identify a *solum* (profile development as A, E, B, and C including a part of the underlying rock).^[11] For this reason, the Horizons de Référence, in the RP, are defined considering strongly their relative development (A, B, C horizons, etc.). This indicates clearly that a part of the previous genetic approach remains inside the RP. Its organization is presented in Fig. 1.

Even if it is not fully completed, the RP, richer in taxa and qualifiers for a smaller part of the world, may be considered to be more precise than the WRB. It is organized in three logical levels that are not truly hierarchical. It allows the user to identify a soil as an intermediate between two taxa, e.g., a “Regosol–Cryosol.”

Table 1 presents the approximate links between WRB and RP at the level of GER, Références, and groups.

The most specific Références of the RP compared with WRB are presented in Table 2 with short explanations.

All these demonstrates that it is very easy to go from one system to the other at the level of groups and Références even if “Calcisol” is a false cognate. (Calcisol means calcareous in WRB but saturated and not calcareous in the RP.)

The situation is more complicated at the level of soil types. Many of the qualifiers seem similar, but this may not truly be the case. Table 3 shows some of the difficulties which may be encountered.

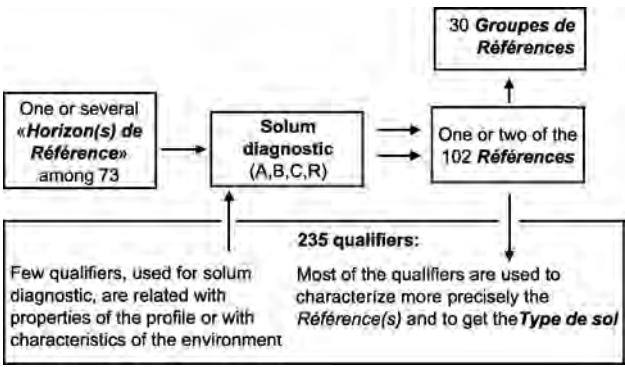


Fig. 1 Organization of the RP.

Table 1 Relationships between the WRB groups (left) and the GER or the Références of the RP (right).

WRB	RP
Histosol	Histosol
Cryosol	Cryosol
Anthrosols	Anthroposols
Leptosols	Lithosols (superficial)
	Rankosols (on acid rocks)
	Organosols (rich in organic matter)
	Rendosols (on calcareous rocks)
Vertisols	Leptismectisols (A/C or A/R profile)
	Vertisols (with B horizon)
Fluvisols	Fluviosols
	Thalassosols (estuarine/marine soils)
Solonchaks	Salisols
Gley soils	Réductisols (dominant reduction)
	Rédoxisol (dominant oxidation)
Andosols	Andosols
	Vitrosols (with glass)
Podzols	Podzosols
Plinthosols	Not already studied
Ferralsols	Not already studied
Solonetz	Sodisols
Planosols	Planosols
Chernozems	Chernosols
Kastanozem	
Phaeozem	Phaeosols
(Greyzems)	Grisols
Gypsisols	Gypsosols
Durisols	Not already studied
Calcisols	Rendosol (Rendzine)
	Rendisols (same morphology, saturated)
	Calcosols (calcareous with B)
	Dolomitosols (with MgCO ₃)
	Calcisols (noncalcareous, saturated)
	Magnesisols (with Mg ²⁺ on clay)
	Calcarisols (calcaric within 25cm)
Albeluvisols	Luvisols (for a part)
Alisols	Not already studied
Nitisols	Fersialsols (for a part)
Acrisols	Not already studied
Luvisol	Luvisol
Lixisols	Not already studied
Umbrisols	Alocrisols humiques, Rankosols
Cambisols	Brunisols
	Pelosols (rich in clay but not 2/1)
Arenosol	Arenosol
Regosol	Régosol

Table 2 Specific Références in the RP.

Alocrisols	Acidic but without <i>argic</i> horizon (i.e., different from the WRB “Acrisols”)
Colluviosols	From colluvium, i.e., on slopes, in parallel with Fluvisols
Peyrosols	From Peyre = stone in some local French languages, i.e., with important coarse fraction
Veracrisols	Sort of Acrisols rich in earth worms

Table 4 characterizes the French soils using the RP system. The data were kindly provided by the *Service d'Etude des Sols et de la Carte Pédologique de France* (Orléans).

DISCUSSION

To pass from a purely genetic classification to an international reference system, such as the WRB, was rather a long journey for the French pedological community.^[12] The construction of the RP, initiated by Denis Baize and Michel-Claude Girard, was a good way to test the possibility of this large change in the French method of thinking.

The French community of soil science fell in agreement (vote in an Administrative Council of Association Française pour l'Étude du Sol in June 2005) on the following ideas: 1) the RP system will be updated adding the tropical soils that were so precisely studied in the period of the French African Empire and 2) this new version of the RP will be built taking in consideration the WRB to get a full compatibility between the two systems. Working in such a way, French scientists will conserve their national system allowing them to focus on such taxa for regional reasons but preserving the international dialog through the WRB.

Table 4 Inventory of the French soils using the RP (from INRA-SESCPF).

RP	% of France	Corresponding WRB group
Calcosols, Brunisols	48.5	Cambisols
Luvisols	14.5	Luvisols
Rendosols	8.3	Calcisols (p.p)
Fluvisols	7.8	Fluvisols
Podzolized Luvisols	6.4	Albeluvisols
Podzols	5.6	Podzols
Lithosols	2.3	Leptosols (p.p)
Rankosols	1.8	Leptosols (p.p)
Andosols, Vitrosols	1.0	Andosols
Arénosols	0.8	Arenosols
Rédoxisols, Réductisols	0.4	Gleysols
Régosols	0.4	Regosols
Salisols	0.4	Solontchaks
Histosols	0.3	Histosols
Phaeosols	0.1	Phaeozems
Vertisols	0.1	Vertisols
Planosols	0.1	Planosols
Autres sols et surfaces	1.2	Others soils and surfaces

CONCLUSION

For the French scientific community, the development of the RP is a good opportunity to work together on the concepts of the soil classifications and to study the genesis and the functioning of the soils identified in France. Moreover, in the RP volume, the texts that present the Andosols (the soils with hydromorphic features and different kinds of humus) are valuable contributions to pedology. Nevertheless, the interest in the RP, at an international level, seems limited because of its great similarity with the WRB.

Table 3 Differences between qualifiers in WRB and RP.

Case	Problem	Examples	
		WRB terminology	RP terminology
1	Term specific of one of the two systems (scarce case)	Carbic	Clinohumic (isohumic)
2	Same meaning but different terms (scarce case)	Alumic (Al sat>50%)	Aluminic (Al ³⁺ dominant)
3	Same name but slightly different meanings (general case)	Magnesian Ca ²⁺ /Mg ²⁺ <1	Magnesian 0.2<Ca ²⁺ /Mg ²⁺ <2
4	Same name but very different meaning (scarce case)	Ruptic	Ruptic
		Vertical discontinuity	Lateral discontinuity
5	Different orthography but same meaning (scarce case)	Bathi (prefix)	Bathy
		Between 100 and 200 cm	Found at depth

REFERENCES

1. Agafonoff. *Les Sols De France Au Point De Vue Pédologique*; Dunod: Paris, 1936; 154 pp.
2. Boulaine, J. *Histoire Des Pédologues Et De La Science Des Sols*; INRA Editions: Paris, 1989; 285 pp.
3. Legros, J.P. *Cartographies Des Sols, De L'analyse Spatiale À La Gestion Des Territoires*; Presses Polytechniques et Universitaires Romandes: Lausanne, 1996; 321 pp.
4. Boulaine, J.; Legros, J.P. Lagatu Henri, la pédologie Lui Doit Beaucoup. *Revue Agr.* **1988**, 528, 18–20.
5. Boulaine, J.; Legros, J.P. *D'Olivier De Serres À René Dumont, Portraits D'Agronomes*; Coll. Tec/Doc, Lavoisier: Paris, 1998; 320 pp.
6. Duchaufour, Ph. *Précis De Pédologie*; Masson et Cie: Paris, 1960; 438 pp.
7. Baize, D.; Girard, M.C. *Coord. Référentiel Pédologique*; INRA-Editions: Paris, 1995; 332 pp.
8. Baize, D. *A Sound Reference Base for Soils, The Référentiel Pédologique (text in English)*; INRA Editions: Paris, 1998; 322 pp.
9. WRB. *World Word Reference Base for Soil Resources*, FAO Report No. 84; ISSS, ISRIC, FAO, 1998; 88 pp.
10. Isbell, R.F. Book Review: A sound Reference Base for Soils—"référentiel pédologique" 1998. *Catena* **2001**, 43, 157–159.
11. Baize, D. Place of horizon in the New French "référentiel pédologique." *Catena* **1993**, 20, 383–394.
12. Baize, D. Typologies et types en pédologie. *Sci. du Sol.* **1992**, 30 (2), 95–115.

Classification Systems: Historical Developments

Dan H. Yaalon

Institute of Earth Sciences, Hebrew University of Jerusalem, Jerusalem, Israel

Abstract

Although ancient farmers and agriculturists recognized the diversity and differential quality of their soils by giving them names, soil classification—except for taxation purposes—was developed only at the end of 19th century, following the Dokuchaev and followers' recognition of soil profiles as specific natural bodies. National or local soil classification systems were commonly used when mapping soil resources. Only the hierarchical six-levels of U.S. Department of Agriculture Soil Taxonomy, which promotes quantitative limits with the concept of diagnostic horizons, and the International Society of Soil Science (renamed as International Union of Soil Science), which sponsored two-tier and a less comprehensive world reference base for soil resources, can claim expanding worldwide use.

INTRODUCTION

Soil classification is an important tool in the studies of soils, enabling the establishment of order and possible relationships between pedons and polypedons at various locations. It is a major tool for the transfer of knowledge and scientific communication in the use of soils for many different purposes. Soils are one of the more complicated systems in nature, possessing both basic properties resistant to alteration and properties subject to relatively rapid, seasonal, periodic, or human-induced changes. The logic of soil classification recognizes several possibilities or ways of grouping the great variation in soil properties, from purely technical soil classification systems involving only a few selected properties, needed to be recognized for specifically designed use, e.g., salinity content when planning for irrigation, to comprehensive soil profile and its surroundings classifications, enabling the interpretation of past soil scape development and possible changes in the future.

Like modern pedology recognizing soil as an independent body derived through processes controlled by five defined soil-forming factors, soil classification is a young enterprise without deep roots in antiquity. Although attempts were made to arrange and classify soils according to these perceived genetic processes, the consensus is that basic soil classification must avail itself of recognized properties and features from the complete soil profile and chosen for their pedogenetic significance. The two books compiled by Finkl^[1] and in 2003 by Eswaran et al.^[2] contain a wealth of articles excellently illustrating the manifold approaches suggested and taken during the development of soil classification since the end of the 19th century. Other background materials will also be considered in this entry.

BEGINNING OF CLASSIFICATION

Soil classification, like all classifications, is a human construct made to organize better manifold information for a definite purpose, e.g., to prepare them for use, to simplify communication between soil users and scientists, and to establish order and relationships between soil classes. Because soils are essentially deterministic bodies, it does not entail discovery or theory but rather observations based on an established conceptual framework, which may vary as science develops. There is no natural classification. In soil science, its major use to establish and monitor soil resources at various scales, besides serving as a baseline for the study and evaluation of the nature, i.e., properties and processes, of its soils.

Ever since the modest beginnings of agriculture, some 10,000 years ago the ancient farmers recognized the differences in the productivity or value of the soils depending on their texture, color, thickness, stoniness, and other properties. The first official classification was probably established in China during the Han dynasty, listing some 30 soil types for taxation purposes. The ancient Chinese also mapped the soils and produced a written soil symbol that recognizes the difference between topsoil and subsoil. But this had little influence on the development of soil classification. In the ancient Middle East, Southern Europe, or Central and South America, though high-level agriculture flourished during the various historic civilizations, differences in soils were duly recognized and named. But the ancient and classical scholars did not deal with soils as a worthy object of study and interest.

During the Enlightenment (17th and 18th centuries), learned societies began to flourish in Western Europe, including various efforts to improve agriculture, which required the recognition and mapping of the land and soil

resources and the beginning of an orderly soil classification.^[3] The Swedish naturalist Carl Linnaeus (1707–1778), the father of binomial plant classification (*Species Plantarum*, 1753), extended this similarly to animals and minerals (*Systema Naturae*, 1758), which also included some soils. However, Carl Linnaeus's terms *argilla communis*, *arena nobilis*, and *humus ruralis*, like those of his contemporary mineralogist–chemist Johann Wallerius (1709–1785) in Uppsala—*humus rubra* or *umbra*—did not catch on. It must be remembered that for both these savants, *humus* meant topsoil, just like *vegetable mold*, later for Charles Darwin (1809–1882) in his pioneering earthworm studies (1835, 1881), was also meant as arable topsoil. An essentially agrogeological concept recognizing a few source materials for the soils or evaluating its productivity for taxation purposes prevailed in recognizing various soil types.

When new ideas about the nature and origin of soils as independent bodies in nature, derived and evolved from the interaction of external and internal factors, emerged in the second half of the 19th century by Dokuchaev (1846–1903) and Hilgard (1833–1916), this was followed rapidly by incorporating the new conceptual framework into an orderly soil classification system. Dokuchaev presented his system in 1879^[4] to be slightly modified by his student and collaborator Sibirtzev (1860–1899) and also disseminated by him^[5] through his widely used textbook of soil science.

The two ideas introduced and widely used ever since were that the unit of soil classification is the soil profile (without defining its spatial extension) and the zonal concept defining the major soil types in the zonal (normal) order from the tundra in the north to the desert steppe in the south, with intrazonal and azonal soil groups as special orders. The scheme provides a top-down hierarchy. It defined a relatively small number of soil types as the major genetic unit, essentially equivalent to plant *genus*, to be subdivided into subtypes as needed or gradually recognized. Numerous folk names for soils, like Chernozem and Podzol, were introduced, which continue to be used in various transformations. In North America, where the zonal regime from east to west is more prevalent, the pioneering pedological work of Hilgard resulted in the recognition of the mid-continental border between the eastern leached humid soils and the western acidic soils with CaCO₃, later incorporated in the influential Marbut (1863–1935) scheme of the U.S. Department of Agriculture (USDA) classification as *Pedalfers* and *Pedocals*.^[6] The new, essentially zonal genetic schemes replaced the Russian soil type term by the *Great Soil Group* category, reflecting the control of soil-forming processes over time.^[7] It became the major unit for knowledge transfer in the American and in many other countries' classification systems,^[1,2] described by modal or centrist properties. At the same time, the American soil classification continued to use the concept of *soil series*, introduced in 1903 (instead of soil type) for the principal unit of detailed mapping, for soil bodies of similar

profile morphology, and for similarity in observed and described property features.^[8] It is thus most nearly akin to the concept of *species* in the plant kingdom. They have geographical names according to places where the polypedon was first described; their number in the United States reaches tens of thousands. Related soil series are grouped into *soil families* and *subgroups*, thus providing a down-to-up hierarchy, the opposite of the Russian classification.

The idea of the climatogenetic origin of the zonal soils dominated the concepts of classification up to the middle of the 20th century, even where the zonal grouping was not as clearly evident as in Russia. Strongly weathered and well-developed soils in the tropics and subtropics also caused problems.^[2] In soil survey mapping, only natural soil properties not affected by human activity or degradation were considered, causing another problem in disregarding altered (metapedogenetic and anthropogenic) soils in all classifications. Soil classification schemes and their logic were frequently discussed,^[1,2] especially at International Soil Science Congresses, but no international consensus emerged and most countries developed their own local classification. There is no doubt that this impeded the development of pedology to some extent, especially in the ease of communication with scientists in related fields and with users of soil maps.

MODERN SOIL CLASSIFICATIONS

In the 1950s, the leaders of the USDA (Charles Kellogg, 1902–1977) decided to develop a new comprehensive soil classification and charged Smith (1902–1981) with the task. He did this over a period of 25 years, developing a completely new system with the help and cooperation of a number of American and international colleagues through a series of seven approximations and international committees, publishing it in 1975 as *Soil Taxonomy—A Basic System of Soil Classification for Making and Interpreting Soil Surveys*. It is an open-ended system subject to continuous testing and revision. A *Second Edition* has been issued in 1999. It is a six-level hierarchical system with 12 soil orders at the first level. Using a completely new nomenclature, originally subject to considerable criticism because of this, these new terms have gradually entered common use and especially the modern textbooks. Its major innovations are the consistent use of quantitative limits (rather than “modal values”) of observed properties and especially the use of diagnostic horizons (see the entry *Diagnostic Horizons*, p. 668) as the major differentiating feature at the higher levels of the hierarchy. Though originally a strictly properties classification, alleged genetic processes interpretations have gradually become incorporated.^[2]

During the last decades, the International Union of Soil Science (IUSS, previously International Society of Soil Science) sponsored an international working group and meetings to develop the world reference base for soil

resources (WRB) that would enable correlation with many national soil classification systems. The WRB system was developed essentially from the revised FAO/UNESCO World Soil Map Legend and is the second system with a claim of a more or less worldwide acceptance. (See the entry *World Reference Base for Soil Resources*, p. 2650, for more information.) It is a two-tier comprehensive system with 30 reference soil groups at the first level, also using diagnostic horizons as a differentiating feature. The second-level qualifiers include many anthropogenic properties that are one of the major differentiating features from the USDA Soil Taxonomy. It has been officially adopted by the IUSS in 1998 for general correlation with other soil classifications.

OTHER CLASSIFICATIONS

Besides the above-mentioned basic soil classifications, it should be remembered that the pedogenic reference profiles are not the only one in use. Although enabling expansive interpretations and behavior predictions, especially when related and neighboring soils are also identified, comprehensive profile examinations are time consuming and costly. For many specific use or limited information needs, a technical classification of a limited number of properties may be equally useful. Thus, several limited property surveys and selected physical determinations are used, especially in engineering classifications for construction purposes. Similarly, low- to medium-level sampling for selected analyzed elements is used in geochemical surveys. When plotted on maps, these have often provided good baselines against which more specific geochemical variations of these elements, reflecting natural or human-induced variations, can be appraised. In all these cases, the range and desirable or non-desirable limits of the examined properties must be known *a priori* or determined separately.

Other developments were various attempts to use numerical classification methods and dynamic computer simulation models to shorten the time-consuming profile analysis, including kriging geostatistics, fuzzy, and fractal analysis, for the interpretation and quantification of spatial problems in recognizing actual soil bodies in alleged soil continua and in predicting interpolative properties needed for better classification of soil landscapes. These computer-assisted methods are more applicable to soil mapping and are not likely to replace the basic need for comprehensive soil classification schemes enabling multifaceted interpretations and predictions.

CONCLUSION

Reviewing the developmental history of pedological soil classification, it becomes evident that at least two well-elaborated systems make claim for a more or less worldwide acceptance. General consensus was not reached.^[1] National or regional soil classification schemes are most likely to continue to be used,^[2] especially as the need for more and better evaluation and interpretation of soil resources becomes necessary. The problem of delimiting soil bodies and their actual 2-D and 3-D limits as the base for the classification used will probably stay with us for a long time.

REFERENCES

1. Finkl, C.W., Jr., Ed. *Soil Classification: Benchmark Papers in Soil Science*; Hutchinson Ross: Stroudsburg, 1982; XIV + 393 pp.
2. Eswaran, H.; Rice, T.; Ahrens, R.; Stewart, B.A., Eds. *Soil Classification—A Global Desk Reference*; CRC Press: Boca Raton, 2003; 263 pp.
3. Fussel, G.E. Soil classification in the 17th and 18th centuries. *Pochvovedenie* **1933**, 5, 409–419.
4. Dokuchaev, V.V. Short historical description and critical analysis of the more important soil classifications (in Russian). *Trav. Soc. Nat. St. Petersburg* **1879**, 10, 64–67.
5. Sibirtzev, N.M. Classification of soils as applied to Russia (in Russian). *Pochvovedenie* **1902**, (1), supplement table.
6. Marbut, C.F. *Soils: Their Genesis and Classification*; USDA Graduate School Lectures; Soil Science Society of America: Madison, 1928; 134 pp.
7. Baldwin, M.; Kellogg, C.E.; Thorp, J. Soil classification. In *Soils and Men, USDA Yearbook of Agriculture*; USDA: Washington, DC, 1938; 979–1001.
8. Simonson, R.W. Evolution of soil series and type concepts in the United States. *Adv. Geocol.* **1997**, 29, 79–108.

BIBLIOGRAPHY

1. Paton, T.R.; Humphreys, G.S. A critical evaluation of the zonalistic foundations of soil science in the United States. Part I: The beginning of soil classification. *Geoderma* **2007**, 139, 257–267.
2. Paton, T.R.; Humphreys, G.S. A critical evaluation of the zonalistic foundations of soil science in the United States. Part II: The pragmatism of Chales Kellogg. *Geoderma* **2007**, 139, 268–276.

Classification Systems: Indian

Mariappan Velayutham

National Bureau of Soil Survey and Land Use Planning,
Nagpur, India

Tapas Bhattacharyya

Dr Balasaheb Sawant Konkan Krishi Vidyapeeth, Dapoli, India

Dilip Kumar Pal

Division of Soil Resource Studies, National Bureau of Soil Survey
and Land Use Planning, Nagpur, India

Abstract

In ancient India, the agriculturists were quite conscious of the nature of soils and its relation to the production of specific crops of good economic return. Therefore, native soil classification was made on the basis of fertility, climate, and the land revenue system. This vast knowledge of the agriculturists became tradition, and the same had been passed on from generation to generation and was transformed into pastoral songs, maxims, and proverbs that acted as guidance to the farmers. Scientific interest in the characteristics of Indian soils began in 1846, and in 1898, four major groups of soils (Indo-Gangetic alluvial soils, black or *regur* soils, red soils, and laterite and lateritic soils) were recognized. Starting from 1932, various soil maps of India were prepared, and in 1954, a revised soil map of India was prepared on the scale, 1 in. = 70 mi. and it showed 20 broad soil classes. This map was revised, which provided the extent and distribution of the different soil classes and their equivalents available in the U.S. soil classification system. Based on the data of the soil resource mapping project of various states, a simplified three-level classification of soils of India can be systematized for both classification of soils and analogous transfer of agrotechnology generated at various stations to similar soils as classified and delineated in the soil map.

INTRODUCTION

Soil classification deals with the grouping of soils according to their suitability to produce plants and crops of economic importance and their use for habitation, recreation, and industry. Agriculture was the mainstay of the people in ancient India and the agriculturists then were quite conscious of the nature of soils and its relation to the production of specific crops of good economic return. According to recorded information, in ancient India, in the period 2,500 B.C. to 600 A.D., a vast knowledge acquired by the then agriculturists by experience has become tradition, and the same has been passed on from generation to generation. However, a major part of this has been forgotten and thus has become a story of the past. Experiences transformed into instructions that had been molded intelligently and ably in the form of pastoral songs, maxims, and proverbs that guided farmers for farming. In this way, the farmers were trained enough to make a choice of a particular soil for a specific crop.

SOIL FERTILITY-BASED CLASSIFICATION OF SOILS

On the basis of fertility, soils were made into two classes, *urvara* (fertile) and *anurvara* or *usara* (sterile). *Urvara mrittika* (fertile soil) was again subdivided into different kinds in view of crops such as *yava* (barley), *tila* (sesamum), *vrihi* (rice), and *mandiena* (mung). On the other hand, *anurvara mrttika* (sterile soil) was subdivided into *usara* (sand ground) and *maru* (desert). In addition, the soils that were irrigated by river and rainfed were known as *nadimatrika* and *devamatrika*, respectively.

SUITABILITY OF CROPS AND SOIL CLASSIFICATION

In the *Arthasashtra* (ca. 300 B.C.), soils were said to be classified on the basis of suitability of crops. Lands that used to be beaten by foam near the river bank were considered suitable for growing pumpkin and gourd. Lands

that were frequently flooded with water were considered suitable for long pepper, grapes, and sugarcane. Lands near wells were cultivated to vegetables and root crops. Lowlands (lakes) were used for green crops, and marginal furrows between any rows of crops were considered suitable for fragrant plants and medicinal herbs.

CLIMATE-BASED SOIL CLASSIFICATION

Caraka divided the lands into three classes, namely, *Jangala*, *Anupa*, and *Sadharana*, according to the nature of the soil and climate.^[1]

Jangala region means dry places/plants (xerophytes).

Anupa region indicates marshy or swampy and watery plants (littoral or inland). These areas are thickly overgrown with forests, bowers, and trees, and flowers encircled by verdant trees and tender creepers.

Sadharana region literally means the ordinary plants (mesophytes). These regions are endowed with creepers, plants, and trees of both classes, i.e., the *vanaspati* and *vanaspatvas*.

As suggested by Misra Chakrapani^[1] in *Visva-Vallabha*, land is described as arid, wet (i.e., marshy), and moderate (i.e., neither too dry nor too wet) and is distinguished by six tastes through color. It is described that gray-colored, pale white, black, white, red, and yellow soils are sweet, sour, salty, bitter, pungent, and astringent in taste, respectively. It also described that land that is littered with anthills, pits, and stones is saline and gravelly, has water at great depth, and is considered to be poisonous so far as planting of trees is concerned. In a region where trees and plants are blighted with frost and in a place littered with stumps, laying a garden is not advisable.

Jala-Bhumi: Krishni-Sukti, a comprehensive book on “agriculture science” attributed to Kasyapa (Misra Chakrapani^[1]), classified land into: 1) wet lands for paddy fields named as *suli bhumi*, *jala-bhumi*, and *sasya bhumi*; and 2) dry lands called *adhaka bhumi*, *tara bhumi*, and *usa bhumi*.

Land with a mild color like the sapphire or the plumage of a parrot, or is in the color of conch or the moon or is bright like molten gold, is considered as excellent.

REVENUE SYSTEM OF LAND CLASSIFICATION

In ancient India, as in modern times, land revenue was based on income from land. In other words, the revenue was rated according to the productivity and kind of soil.

Manu, the *Arthasashtra*, and the *Sukraniti* presuppose a gradation of land, survey, and measurement, calculation of outturn, as well as expenses per unit of land. It is indicated in the *Arthasashtra* and the *Smrtis* that laid stringent rules about leaving not only a good producer's surplus but also a

classification of soil on the basis of fertility. The king's share did not necessarily mean a fixed share. It was determined on the basis of soil fertility and by the needs of the state or of the cultivator. Measurement and survey and the differentiation of soils according to their productivity thus indicate that land revenue assessment was not permanent, and it used to be revised at times, although a constant revision was not felt necessary. Megasthenes, in his travel notes, recorded that Maurya officers were most likely concerned with the measurement and supervision of alluvial deposits for revenue purposes.^[2]

During the 16th century in India, the land assessment classification was based on the suitability of soils for crops.^[1] It also considered other factors such as texture and color of soils, availability of water, slope of the land, and yield of crops. Considering all these, in addition to marketing facilities for the produce, fair estimates of land values were arrived. The land that was dependent solely on rainfall was called *barani*, watered by wells was *chahi*, irrigated by canal was *nahri*, and moistened by river percolation was *sailabi*. In addition, local names of soils that conform closely to the soil classes were also developed. An example of this from Chhattisgarh indicates four classes of soils like *matasi*, *dorsa*, *kanhar*, and *bhata*. These soils are in a catenary sequence, wherein *matasi* are yellow soils on the upland or level land with loam to clay loam and loamy clay in texture and yield good paddy. *Dorsa* soils are on the slopes, darker, and with the same texture as of *matasi* and yield good paddy. *Kanhar* soils are in lowland areas, dark and slightly heavier than *matasi* and *dorsa*, and yield good paddy and wheat. *Bhata* soils are barren wastelands on uplands with gravelly sandy soils and are reddish yellow.

Red soils of Jhansi district of Uttar Pradesh are known as *parwa* and *rakar*; the former is a brownish gray soil with loam to sandy clay loam texture, whereas the latter is not useful for cultivation.

The red soils of Telangana district of Andhra Pradesh are known as *chalkas*, which are sandy loams located on highlands and are cultivated for kharif crops. In Andhra Pradesh, the pale brown to brown soils are known as *dubba* soils. These soils have low fertility and suffer severe erosion, and thus are submarginal lands and more suited for pasture and forage crops. Similarly, in Orissa state, the soils are classified as *at*, *mal*, *berna*, and *bahal* according to their topographic situation. The revenue system of soil classification and these local names provide an index of soil fertility.

Initiatives During the 19th to 21st Century

Scientific interest in the characteristics of Indian soils began when the Geological Survey of India started studying the soils and the underlying strata in 1846.^[1] In 1898, Leather recognized four major groups of soils in the country: 1) Indo-Gangetic alluvial soils; 2) black or *regur* soils; 3) red soils; and 4) laterite and lateritic soils.

Table 1 The extent and distribution of the different soil classes of India as represented in the soil maps on 1:250,000 scale along with their equivalent to U.S. Department of Agriculture, U.S. nomenclature system.

Major soils (traditional name)	Extent		Distribution in states	Soil orders U.S. soil taxonomy
	*000 ha	Percentage		
Alluvial	100,006	30.4	J&K, HP, Punjab, Haryana, Delhi, UP, Gujarat, Goa, MP, MS, AP, Karnataka, TN, Kerala, Puducherry, Bihar, Odisha, WB, ArP, Assam, Nagaland, Manipur, Mizoram, Tnpura, Meghalaya, A&N	Inceptisols, Entisols, Alfisols, Andisols
Coastal alluvial	10,049	3.1	AP, Karnataka, TN, Kerala, WB, Gujarat, Odisha, Puducherry, Lakshadweep, A&N	Andisols. Inceptisols. Entisols
Red	87,98	26.8	AP, Karnataka, Kerala, TN, Puducherry, Rajasthan, MP, MS, Gujarat, Goa, ArP, Assam, Manipur, Meghalaya, Nagaland, Mizoram, Tripura, Delhi, UP, HP, A&N	Alfisols, Ultisols, Entisols, Inceptisols, Mollisols, Andisols
Laterites	18,094	5.5	AP, Karnataka, Kerala, TN, Puducherry, MS, Odisha, WB	Alfisols, Ultisols, Inceptisols
Brown forest	540	0.2	Karnataka, Maharashtra	Mollisols. Inceptisols
Hill	2,262	0.7	Manipur, Odisha, WB, Tripura, Nagaland	Inceptisols, Entisols
Terai	326	0.1	UP, Sikkim	Mollisols, Entisols
Mountain meadow	60	—	J&K	Mollisols
Sub-montane	104	—	J&K	Alfisols
Black	54,682	16.6	MP, MS, Rajasthan, Puducherry, TN, UP, Bihar, Odisha, AP, Gujarat	Vertisols, Mollisols, Inceptisols, Entisols, Aridisols
Desert	26,283	8	Rajasthan, Gujarat, Haryana, Punjab	Aridisols, Inceptisols, Entisols
Others [†]	28,305	8.6	—	—
Total	328,700	100	—	—

[†]Includes glaciers (0.4%), sand dunes (0.01%), mangrove swamps (0.04%), salt waste (0.01%), water bodies (0.1%), rock land (0.25%) and rock outcrops (7.8%). MP, Madhya Pradesh; MS, Maharashtra; UP, Uttar Pradesh; J&K, Jammu and Kashmir; TN, Tamil Nadu; AP, Andhra Pradesh; ArP, Arunachal Pradesh; WB, West Bengal; HP, Himachal Pradesh; A&N, Andaman and Nicobar Islands.

The first soil map of India was prepared by Schokalsky in 1932.^[1] Wadia, Krishnan, and Mukherjee^[3] mentioned that soil groups were nearly coterminous with the boundaries of the geological outcrops. Vishwanath and Ukil^[4] prepared a soil map of India portraying the different climatic types on the basis of N.S. quotients by adopting color and texture as units of classification, and these were correlated with four major climatic zones (arid, semiarid, humid, and perhumid). In 1954, a revised map was prepared at Indian Agricultural Research Institute, New Delhi, on the scale 1 in. = 70 mi, and it showed 20 broad soil classes.

Integrating the effects of climate, vegetation, and topography, 16 major and 108 minor soil regions were identified and brought under 27 units by Raychaudhuri et al.^[5]

The soil map was revised by Raychaudhuri^[6] and later on by Raychaudhuri and Govindarajan.^[7] This map provided the extent and distribution of the 23 soil classes in the map and their equivalents available in the U.S. Department of Agriculture system.

Velayutham^[8] gave the idea of simplified three-level classification with the 1st level of 23 major soil groups and within each major soil group,^[5,6] the corresponding and nearest soil families (1,247 families) and series can be grouped in the next two levels, respectively. This can be

modeled for various states of India, from the data of the soil resource mapping project^[9] (1985–1996) of various states, based on key functional parameters (just as functional genomics) of soils in relation to sustainable land use and management. Such a national soil classification, which is systematic and systemic in character, can also be consistent enough to relate our (Indian) soils with other schemes of soil classification for international correlation and reference and comprehension.^[10]

Bhattacharyya et al.^[11] have given extent and distribution of 11 major soils as represented in the soil maps on 1:250,000 scale along with soil orders of the U.S. Soil Taxonomy (Table 1).

CONCLUSION

To develop a soil classification scheme, consistent research on pedology to understand soil-forming processes is necessary to link soil parameters with level/category of soil classes and scale of mapping. The U.S. Soil Taxonomy was officially adopted as a scheme of Indian soil classification, and the soil scientists engaged in soil survey and mapping used it for all research purpose.

Moreover, the soil taxonomy, being an open system, permitted revision of soil groups as and when new soils are identified in the field. However, for practical purposes, the 23 major soil groups classified and the local names of soils borne out of indigenous technical knowledge as discussed above prevail and appear useful for common understanding and extension advisory purpose.

REFERENCES

1. Raychaudhuri, S.P. Evolution of classification of soils of India. *Indian Agric.* **1975**, *19* (1), 163–173.
2. Bose, A.N. *Social and Rural Economy of Northern India (c. 600 BC–A.D. 200)*; Firma, K.L., Ed.; Mukhopadhyay: Calcutta, 1961; 154 pp.
3. Wadia, D.N.; Krishnan, M.S.; Mukherjee, P.N. Introductory note on the geological formation of the soils of India. *Rec. Geol. Surv. India* **1935**, *68*, 369–391.
4. Vishwanath, B.; Ukil, A.C. *Soil Map of Indian Agricultural Research Institute*; IARI: New Delhi, 1943.
5. Raychaudhuri, S.P.; Aggarwal, R.R.; Datta Biswas, N.R.; Gupta, S.P.; Thomas, P.R. *Soils of India*; Indian Council of Agricultural Research: New Delhi, 1963.
6. Raychaudhuri, S.P. Land resources of India. In *Indian Soils—Their Classification, Occurrence and Properties*; Committee on Natural Resources, Planning Commission, Govt. of India: New Delhi, 1964; Vol. I.
7. Raychaudhuri, S.P.; Govindarajan, S.V. Soils of India. In *Indian Council of Agricultural Research. Technical Bulletin (Agr.) No. 25*; ICAR: New Delhi, 1971; 1–45.
8. Velayutham, M. Available soil information and the need for a systematic classification of soils of India. *J. Indian Soc. Soil Sci.* **2000**, *48* (4), 683–689.
9. Bhattacharyya, T.; Nagar, A.P.; Nimkhedkar, S.S.; Sarkar, D.; Sehgal, J.; Velayutham, M.; Gajbhiye, K.S. *Soil Taxonomic Database of India and the States (1:250,000 scale)*. NBSS & LUP pubn no:143; National Bureau of Soil Survey and Land Use Planning: Nagpur, India, 2009; 266 pp.
10. Dudal, R. Correlation of soil classification units used in different continents. *Trans. 8th Int. Congr. Soil Sci., Bucharest, Romania* **1964**, 253–260.
11. Bhattacharyya, T.; Pal, D.K.; Mandal, C.; Chandran, P.; Ray, S.K.; Sarkar, D.; Velmourougane, K.; Srivastava, A.; Sidhu, G.S.; Singh, R.S.; Sahoo, A.K.; Dutta, D.; Nair, K.M.; Srivastava, R.; Tiwary, P.; Nagar, A.P.; Nimkhedkar, S.S. Soils of India: Historical perspective, classification and recent advances. *Curr. Sci.* **2013**, *104* (10), 1308–1323.

Classification Systems: Indigenous

Erhan Akça

Center of Remote Sensing, Adiyaman University, Adiyaman, Turkey

Selim Kapur

Departments of Soil Science, Landscape Architecture, and Agricultural Engineering, University of Cukurova, Adana, Turkey

Abstract

Numerous developing countries, with agricultural and natural resource management traditions dating back to several thousand years, have generated, via hundreds of ethnic groups living in dozens of different ecological zones, an immense collection of indigenous natural resource management and agricultural knowledge as well as ethnopedology, i.e., soil type properties developed by indigenous technical knowledge (ITK). These soil types have been the basis for the “sustainable land management” (SLM) concept. This knowledge on soil classification, unique to a given culture or society, is unfortunately largely ignored in the so-called developing world in spite of the increasing number of studies conducted by international and locally functioning establishments, such as the World Bank, International Crops Research Institute for the Semi-Arid Tropics, International Board for Soil Research and Management, the Institute of Indigenous Technology of the Iowa State University, United Nations Educational, Scientific and Cultural Organization, and United Nations Environment Programme. ITK obtained throughout the relevant world programs is concerned with a wide range of subjects from improved farm implements, to soil classification and SLM.

INDIGENOUS SOIL CLASSIFICATION AND LAND USE

Indigenous soil classification systems should be regarded as complementary information to science-based soil maps, which bear the advantage of being widely recognized and communicated between farmers, extension services, and scientists.^[1,2] The indigenous soil classification of the Mossi of Burkina Faso is a good example, for having been largely obtained by interviewers, for the relevant and widely accepted soil attributes for use in the construction of contemporary land use management activities. Moreover, Dialla's^[3] study on soil classification recognizes four major classes of soils based on texture, whereas topographical position is the major grouping parameter, with texture as a lower-order diagnostic property, in some West African indigenous soil classifications.^[4–6] Similarly, many farmers in African drought-prone areas, such as Senegal, Mauritania, Burkina Faso, and Mali, have developed distinctions between soils bound to their major limitations such as flooding frequencies,^[2] whereas Maglinao's^[7] indigenous soil classification represents an approach on a central concept (Table 1).

Maglinao's concept is concerned with depth, color, structure, and base saturation of surface and subsurface horizons like the Mossi farmers^[3] four major classes of soils (Table 2) used for interpreting crop potential. In fact, this approach itself is the concept of soil series yielding

information on productivity, in particular to the traditional use of soils as well as their quality, which is reported to be closely related to the development of risk minimization strategies along with slope, water, and pastoral resource management.^[8]

THE PHYSICAL DIMENSION

The physical dimension, which is the primary approach in indigenous classification (as in all visual research systems), includes color and touch (feeling) in assessing soil texture—as the Yoruba people in Nigeria have rubbed soil between fingers to tell if they are sandy (Yanrin), clayey (Bole), loamy (Alaadun), or loamy clay (Bole and Alaadun).^[9,10] Moreover, texture was used by ancient Peruvians as basis to designate eight major soils, in the same way that bulk density was classified by weight and ease of penetration of the cutlass.^[9] Color and equally refined physical dimension of indigenous soil classification were used by the Baruya people of Wonenera, New Guinea, as a source of pigment consisting of nine different colors for ceremonies and six others for agricultural soils.^[11] As dark soils are considered more fertile than their lighter-colored counterparts the farmers of Niger have designated soils with different color classes in relation to land degradation as: 1) fertile black soils (labu biri) with high organic matter; 2) white soils (labu kware) with

Table 1 Indigenous soil classification and their equivalents in soil taxonomy and World Reference Base (WRB).

Local classification	Central concept	Soil order and soil name
Blatong pantad	Thick, dark-colored, well-structured surface horizon; high base saturation	Mollisols, Cambisols (rich in organic matter)
Kalibugan	Brown clayey soil; deep; subsurface clay horizon; high base saturation	Alfisols, Luvisol (brown forest soil)
Puro Pantad	Brown or reddish color; clayey; deep; low base saturation; have subsurface clay horizon	Ultisols, Acrisol, Nitisol (red mountain soils)
Pra ambao	Strongly weathered; reddish and yellow colored; low exchange capacity; lack of weatherable minerals; absence of clay illuviation; very low natural fertility	Oxisols, Ferralsols (strongly weathered soils)

Source: From Maglinao,^[7] USDA,^[12] and Lambert, Daroussin, et al.^[13]

plant nutrient depletion by cultivation and erosion; and 3) further-degraded red soils (labu kirey).^[14,15] Taste has been another major criterion to assess acidity and salinity among Malaysian and Thai farmers with the sweet, the neutral, and the sour relating to the Western concept of soil pH.^[16,17] The Lari people in Peru have been able to separate taxons for eatable soils. The high Tundra soils are

Table 2 The Mossi farmers' (Africa) indigenous soil classes.

Major classes	Zşkugri (stony soil)	Bşisri (sandy soil)	Bolle (clay soil)	Bşoogo (loamy soil)
Derived soils	Zşka (lateritic soil)	Bşs-miuugu (red sandy soil)	Dagre (hard clay soil)	Zş-bugri (very soft soil)
	Rasempuiiga (gravelly soil)	Bşs-sabille (black sandy soil)	Zş-sabille (black soil)	Zş-naare (wet loamy clay soil)
	Tşnga (mountainous soil)	Zş-peelee (white soil)	Zş-kotşka (a clay soil in a low land where water stagnates)	Kşongo (black soil with a dense growth of bushes as vegetal cover.
	Zş-miuugu (red soil)			

Source: From Dialla.^[3]

rich in minerals and salt and are eaten by the Laris as condiments or used as fatteners for animals.^[18]

THE PERCEPTUAL APPROACH

The perceptual approach of soil classification may include features other than the physical dimension, i.e., the utilitarian color–touch–texture domain stated by Posey^[19] and Weinstock.^[16] These features reflect the environmental priorities on soils based on their suitability, particularly for managing pastures or indigenous particular crops as in the case of Anatolian–Turkish land use context (Z.A. Savatlı, private communication, Adana Farmers Association, Adana, Turkey; Table 3).^[20]

Table 3 The Anatolian indigenous soil classification and their equivalents in WRB.

Local soil classification	LU context	Soil name WRB (2001)
Keli (fertile soils)	Sandy loam to loam, good drainage and irrigable lands of the Mediterranean region	Cambisols, Fluvisols
Dambası (fertile soils)	Loamy to sandy loam, very good drainage and irrigable lands of the Mediterranean region nearby villages	Cambisols, Fluvisols
Malaz	Deep clayey soils developed on marls in need of drainage and conservation management/tillage	Fluvisols, Cambisols
Dölek	Deep clayey soils of sedimentary origin in need of drainage and conservation management/tillage	Fluvisols, Cambisols
Kepir	Slightly sloping to flat land, i.e., calcrete terraces with shallow to moderately deep red to reddish brown soils suitable for agroforestry with indigenous tree crops (olives, carobs, figs, vineyards and stone pine)	Calcisols, Cambisols, Luvisols
Cökek	Water logged playas to be managed as wetlands	Vertisols, Fluvisols
Bozkır	Slightly degraded land with primary saline areas of Inner Anatolian grazelands	Cambisols, Calcisols
Dazkır	Degraded Bozkır soils of the mild aridic Inner Anatolian indigenous overgrazed pastures in need of management	Cambisols, Calcisols
Corak	Soils of Inner Anatolia with primary and secondary salinity suitable for halophyte and/or biodiversity	Fluvisols, Cambisols, Calcisols

Source: From Lambert, Daroussin, et al.^[13]

Similarly, the Melanesians of the Trobriand islands have classified soils for the suitability of yam or taro (their main diet) crops: dumya soil is good for dry season taro or, but not for yam; butuma soil is excellent for yam, but unsuitable for taro; malala soil is unsuitable for taro, but good for hardy yam; sawewo soil is good for large yam; galaluwa soil is good for all cultivation; and the very fertile kwala soil is good for all crops. (B. Malinowski, cited in Ettama.^[16]) Mixed cropping is also an extensive practice in indigenous societies, especially in the form of shifting cultivation, giving rise to soil classifications based on vegetation covers rather than on soil properties.^[21] Such biological indicators, i.e., vegetation and soil macrofauna, which actually reflect soil suitability, are also common to

Table 4 Indigenous use of biological indicators of soil quality: vegetation and macrofauna.

Vegetation	
Malaysia	<i>kedukuk</i> bush (<i>Melostoma</i>) indicates high Al level <i>Pohon bakan</i> (<i>Hanguana</i>) tree indicates acid soil with stagnant water <i>Imperata</i> grass, <i>keriang</i> berry bushes and cashew indicate low soil fertility
Shipibo, Peru	Use indicator plants for soil hydrology
Caatinga, Brazil	Thinly wooded vegetation indicates imperfect drainage
S. Mexico	Sparse vegetation is general indication for <i>tierra delgada</i> , thin soil
Maya, Mexico	Dark-colored vegetation high soil fertility
Kekchi, Guatemala	Use indicator plants for site suitability for <i>Milpa</i> agriculture
Mebengokre, Brazil	Use indicator plants for general site suitability
Gaborone, Botswana	Use indicator plants for soil fertility
Yoruba, Nigeria	<i>Odundun</i> (<i>Kalanchoe</i> sp.) indicates high soil fertility, while <i>Eran</i> (<i>Digitaria horizontalis</i>), <i>Okan</i> (<i>Combretum platypterum</i>), and <i>Pepe</i> (<i>Mallotus oppositifolius</i>) indicate poor fertility
Niger	Dark, dry roots of millet seedlings indicate 'sick' soil which is not fertile
Soil Fauna	
Yoruba, Nigeria	Earthworm casts indicate fertile soil
Sukuma, Tanzania	Termite soil is fertile, soil classification based on their presence/absence
Niger, Sierra Leone	Soil close to ant and termite hills is fertile and planted with special crops
Ecuador	Earthworm casts and grub casts (?) indicate good soil
Thailand	Soil from termite hills is used as soil fertility improver

Source: From Ettama.^[15]

non-shifting cultivation in many parts of the world (Table 4).^[15] These indigenous societies employ specific types of crops to determine the level of soil fertility, drainage characteristics, and acidity.

CONCLUSION

Indigenous soil classification systems may generally seem to be more shallow when compared to contemporary classifications.^[9,11,21] Indigenous soil classifications primarily bear a functionality for land use, whereas common contemporary classification systems divide soils mainly based on knowledge on pedogenesis.^[21,22] Second, indigenous soil taxa are derived from surface horizon properties, while the diagnostic features of differentiation of contemporary soil taxa are the character and sequence of soil horizons, i.e., the perception of taxonomic units is 2-D in indigenous classification and 3-D in the contemporary. Further studies undertaken by Bellon and Taylor^[23] revealed that soil quality-wise ranking by indigenous perception and scientific method gave similar results, showing that the distinctions made by local people were all scientifically valid and statistically testable.

REFERENCES

1. Tabor, J.A. Ethnopedology: Using indigenous knowledge to classify soils. *Arid Lands Newsl.* **1990**, *30*, 28–29.
2. Tabor, J.A. *Soil Surveys and Indigenous Soil Classification*; IK Monitor Articles, 1993. Available at www.nuffic.nl/ciran/ikdm/1-1/tabor.html.
3. Dialla, B.E. The Mossi indigenous soil classification in Burkina Faso. *Indig. Knowl. Dev. Monit.* **1993**, *1* (3), 17–18.
4. Richards, P. *Indigenous Agricultural Revolution: Ecology and Food Production in West-Africa*; Westview Press: Boulder, 1985.
5. Carney, J. Indigenous soil and water management in Senegambian rice farming systems. *Agr. Hum. Values* **1991**, *8* (1/2), 37–48.
6. Schutjes, A.H.M.; Van Driel, W.F. La classification locale des terres par les Mossi: paysan et pedologues—ils le meme langage? In *Premier Colloque International de l'AOCASS: gestion durable de sols et de l'environnement en Afrique Tropicale*; Burkina Faso: Ouagadougou, 1994.
7. Maglinao, A.R. Indigenous technical knowledge (ITK) on soil management in the Philippines: A preliminary review. In *Assembly Meeting of the Managing Soil Erosion Consortium (MSEC)*, Nan, Thailand, Jan 29–Feb 2, 1997; 1997. (Mimeograph).
8. Mathias-Mundy, E.; Muchena, O.; McKiernan, G.; Mundy, P. *Indigenous Technical Knowledge of Private Tree Management*; Final Draft Report; FAO: Rome and CIKARD: Ames, 1990.
9. Osunade, M.A. Optimisation of traditional systems of soil resources inventory to achieve increased

- sustainable production. Third World Plan. Rev. **1989**, *11* (1), 97–108.
10. Osunade, M.A. The significance of colour in indigenous soil studies. *Int. J. Environ. Stud.* **1992**, *40*, 185–193.
 11. Ollier, C.D.; Drover, D.P.; Godelier, M. Soil knowledge among the Baruya of Wonenara, New Guinea. *Oceania* **1971**, *42* (1), 33–41.
 12. USDA. *Soil Taxonomy. A System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; Agriculture Handbook; Soil Survey Staff: Washington, 1999; Vol. 436.
 13. Lambert, J.J.; Daroussin, J.; Eimberck, M.; Jamagne, M.; King, D. *Instructions Guide for the Elaboration of the Soil Geographical Database of Euro-Mediterranean Countries at 1:1 Million Scale*; (WRB) EU, European Soil Bureau of the Institute of Environment and Sustainability; EU-JRC: Ispra, 2000; 47 pp. Version 4.0.
 14. Taylor-Powell, E.; Manu, A.; Geiger, S.C.; Ouattara, M.; Juo, A.S.R. Integrated management of agricultural watersheds: Land tenure and indigenous knowledge of soil and crop management. *Trop. Soils Bull.* **1991**, *91-04*, 30.
 15. Ettama, C.H. *Indigenous Soil Classification*; 1994. Available at www.itc.nl/~rossiter/Docs/Misc/IntroToEthnopedology.pdf (accessed February 2003).
 16. Weinstock, J. Getting the right feel for soil: Traditional methods of crop management. *Ecologist* **1984**, *14* (4), 146–149.
 17. Marten, G.G.; Vityakon, P. Soil management in traditional agriculture. In *Traditional Agriculture in Southeast Asia. A Human Ecology Perspective*; Marten, G.G., Ed.; Westview Press: Boulder, 1986; 199–225.
 18. Furbee, L. A folk expert system: Soil classification in the Colca Valley, Peru. *Anthropol. Quart.* **1989**, *62* (2), 83–102.
 19. Posey, D.A. Hierarchy and utility in a folk botanical system: Patterns in the classification of arthropods by the Kayapo Indians of Brazil. *J. Ethnobiol.* **1984**, *4*, 123–134.
 20. Caglar, K.Ö. *Soil Science (in Turkish)*; Fac. of Agr. Pub., University of Ankara: Ankara, 1949; Vol. 10.
 21. Williams, B.J.; Ortiz-Solorio, C.A. Middle American folk soil taxonomy. *Ann. Assoc. Am. Geogr.* **1981**, *71* (3), 335–358.
 22. FitzPatrick, E.A. *Horizon Identification*; Research Report; European Soil Bureau, 2002; Vol. 7.
 23. Bellon, M.R.; Taylor, J.E. “Folk” soil taxonomy and the partial adoption of new seed varieties. *Econ. Dev. Cult. Change* **1993**, *41* (4), 762–786.

Classification Systems: Major

Lars Krogh

Geographical Institute, University of Copenhagen, Copenhagen, Denmark

Abstract

Numerous systems have been developed and exist for the classification of soils and soil materials, serving a variety of purposes. As international references, however, the agriculturally biased, morpho-genetic systems, Soil Taxonomy and World Reference Base, for soil resources have gained widespread acceptance. With increasing demands on small-scale quantitative soil information and a reality where soil information must be able to address a broad range of environmental issues other than agricultural production, the soil research community is faced with a major challenge in meeting this demand and thinking along new lines to develop new soil information systems.

INTRODUCTION

Soil classification is a tool for stratifying, generalizing, remembering, predicting, mapping, and communicating information on soil resources for a specific purpose and, in addition, it serves as a basis for further enquiry into soil processes.

Soil cover varies continuously vertically and horizontally at many scales in response to the interactive effects of climate, vegetation, relief, parent material, time, and the human activities. Soils have many users and the use of soils also varies according to climate, technology, land cover, agricultural systems, trade possibilities, and traditions, and therefore the concepts of soils are very diverse. In addition, the very nature of soils—slowly evolving three-dimensional bodies with no distinct borders, and thus no true “individual” object for classification—in combination with the late “discovery” of soil science probably explains why numerous and diverse systems of soil classification have evolved over time: from the ancient Chinese and Egyptian tax assessment systems, to the scientific, national, and partly global systems of the Western world and to the advanced, geostatistical, fuzzy logic systems.

The American classification system *Soil Taxonomy*^[1] and the joint FAO, ISRIC, and ISSS system *World Reference Base (WRB) for Soil Resources*,^[2] which is a further development of the FAO–UNESCO *Soil Map of the World*^[3] and its update from 1988,^[4] are the two most widely accepted and used soil classification systems. In addition, the Australian,^[5,6] Brazilian,^[7] Canadian,^[8,9] Chinese,^[10] French,^[11] and Russian^[12–14] systems might also be considered major systems in that they are used on a large proportion of the land surface of the earth. However, these systems have all been developed for national purposes only.

SOIL TAXONOMY

Building upon available soil concepts and classification systems in the United States, the development of *Soil Taxonomy* was started in 1951 and through a series of modifying steps the *Seventh Approximation* was published in 1960^[15] followed by the publication of the definitive system for the first time in 1975, under the name *Soil Taxonomy*.^[16,17] The system has been amended nine times since then (published as keys) and is continuously undergoing modifications to encompass results of new research and changing soil concepts, and the second comprehensive edition was published in 1999.^[1,18] The original purpose of the system was to serve as a tool for making and interpreting soil surveys in the United States, predominantly in relation to agricultural production, but in years it has evolved into a means of communication in soil science. *Soil Taxonomy* differs from most other systems by its rather special names for describing soils, which were created to avoid older traditional names that have ambiguous meanings. At the time of publication, it was among the very few systems that separated soils on the basis of precise quantitative definitions of observable and measurable properties.

Soil Taxonomy is a hierarchical, six-level, morphogenetic classification system, in which soils can be arranged—with decreasing rank and with increasingly narrower variation in properties—in 12 orders, 64 suborders, about 300 great groups, about 2400 subgroups, about 8000 families, and (in the United States) about 20,000 series.

Soil Taxonomy has defined a number of diagnostic horizons (Table 1) and diagnostic soil characteristics, which are used to identify and assign soils into its categories by the means of a key. Diagnostic horizons are recognized as reflecting common and widespread results of major

Table 1 Brief description of the 8 diagnostic epipedons and the 20 diagnostic subsurface horizons in *Soil Taxonomy*

	Description
Epipedons	
Anthropic (Gr. <i>anthropos</i> , human being)	Manmade, thick, dark, formed by long continued residence or application of organic residues. Often with high phosphorus content
Folistic (L. <i>folia</i> , leaf)	More than 20 cm thick, high in organic matter of cool humid regions under forest vegetation
Histic (Gr. <i>histos</i> , tissue)	Between 20 and 40 cm thick, high in organic matter that is saturated with water for long periods
Melanic (Gr. <i>melas</i> , black)	Thick, black, high in organic matter of soils formed from volcanic ash. The organic matter is associated with short-range order minerals or Al-humus complexes
Mollic (L. <i>mollis</i> , soft)	Thick, dark, high in organic matter, well structured with over 50% base saturation, predominantly from divalent cations. Typical for steppe soils
Ochric (Gr. <i>ochros</i> , pale)	Thin, light, or low in organic matter, thereby failing to be mollic or umbric
Plaggen (Ger. <i>plaggen</i> , sod)	Manmade, more than 50 cm thick, high in organic matter, and raised above the original soil surface. Formed by spreading of a mixture of manure and sod, the latter used for bedding livestock in medieval times
Umbric (L. <i>umbra</i> , shade = dark)	Thick, dark, high in organic matter, well structured with under 50% base saturation
Subsurface horizons	
Agric (L. <i>ager</i> , field)	Illuvial horizon below the plow layer containing silt, clay, and organic matter. Formed under (old-world) cultivation
Albic (L. <i>albus</i> , white)	Eluvial, bleached and light-colored horizon from which clay and Fe-oxides have been removed
Argillic (L. <i>argilla</i> , white clay)	Horizon with significantly higher clay content than horizon above. Formed through clay illuviation and/or <i>in situ</i> weathering
Calcic (L. <i>calcis</i> , lime)	Illuvial horizon with secondary CaCO ₃ as powder or concretions. Mostly found in arid environments
Cambic (L. <i>cambiare</i> , to exchange)	Horizon in which weak structure formation, weathering of primary minerals, and precipitation of Fe-oxides have taken place
Duripan (L. <i>durus</i> , hard)	Horizon cemented by illuvial silica thereby impeding root and water movement
Fragipan (L. <i>fragilis</i> , brittle)	Compact, high bulk density, non-cemented horizon impeding root and water movement
Glossic (Gr. <i>glossa</i> , tongue)	Transitional horizon with vertical bleached tongues formed by movement and destruction of clay and Fe-oxides in existing argillic horizon
Gypsic (L. <i>gypsum</i> , gypsum)	Illuvial horizon with secondary gypsum. Mostly found in arid environments
Kandic (from kandite)	Horizon with significantly higher clay content than horizon above and dominated by LAC clays. Formed through clay illuviation and/or <i>in situ</i> weathering
Natric (L. <i>natrium</i> , sodium)	Horizon with significantly higher clay content than horizon above and 15% or more exchangeable Na. Formed through clay illuviation and/or <i>in situ</i> weathering
Ortstein (Ger.)	Cemented horizon consisting of complexes of Al and organic matter with or without Fe (spodic materials)
Oxic (F. <i>oxide</i>)	Fine textured, thoroughly weathered horizon with clay consisting of LAC and sesquioxides and having a low CEC. Mostly found in old tropical environments
Petrocalcic (Gr. <i>Petra</i> , rock and calcic)	Indurated illuvial horizon with secondary CaCO ₃ . Mostly found in arid environments. Barrier to roots and water
Petrogypsic (Gr. <i>petra</i> , rock and gypsic)	Indurated illuvial horizon with secondary gypsum. Mostly found in arid environments. Barrier to roots and water
Placic (Gr. <i>plax</i> , flat stone)	Thin (<25 mm), black to reddish horizon cemented by Fe (and Mn) and organic matter
Salic (L. <i>sal</i> , salt)	Horizon with accumulation of salts more soluble than gypsum and in quantities harmful to roots. Mostly found in arid environments
Sombric (F. <i>sombre</i> , dark)	Horizon with illuvial concentrations of organic matter that is not in combination with Al. Mostly found in cool, mountainous environments
Spodic (Gr. <i>spodos</i> , wood ash)	Illuvial horizon with amorphous material consisting of organic matter and Al with or without Fe
Sulfuric (L. <i>sulphur</i> , sulfur)	More than 15 cm thick mineral or organic horizon with low pH (pH = 3.5) formed by artificial drainage of sulfide rich materials

pedogenetic processes and are defined by observable and measurable morphological and chemical properties. Diagnostic horizons present at the surface are termed epipedons and those present below the surface are termed subsurface horizons. Diagnostic soil characteristics, some unique to mineral soils and some unique to both mineral and organic soil, are also defined by specific morphological and chemical properties. They do not, however, necessarily make up a coherent horizon, and they also include soil water and temperature characteristics as defined by several soil moisture and soil temperature regimes. For a soil to be assigned into a category at any level of the system, a soil must meet the criteria specified in the definitions of the category.

Soil orders are separated on the basis of stable and long-term development processes and properties reflected by diagnostic horizons (Table 2); suborders are separated on the basis of mostly dynamic properties controlling the soil-forming processes, such as soil moisture regime and the presence or absence of water or vegetation; great groups are separated on the basis of generally static properties with minor or additional control on soil-forming processes; subgroups are separated on the basis of a central concept and divergence from this; families are separated on the basis of properties that relate to plant growth or the potential for further change; series, which are the mapping unit carrying local names, are separated on the basis of properties that reflect relatively narrow ranges of soil-forming factors. To illustrate the special nomenclature of *Soil Taxonomy*, the name Typic Argiaquolls

(subgroup level) signifies that the soil is a typical base rich soil with a deep, organic matter-rich A-horizon (Mollisol) that is relatively wet (Aquic conditions) and with clay accumulation in the subsoil (argillic subsurface horizon).

THE WRB FOR SOIL RESOURCES

The *WRB for Soil Resources* was developed jointly by the ISSS, FAO, and ISRIC through a series of steps and published in 1998.^[2,19–21] It is not an entirely new system, but rather a revision of the FAO–UNESCO *Soil Map of the World* from 1974^[3] and the so-called FAO Revised Legend from 1988,^[4,22] to some extent, it is also inspired by *Soil Taxonomy* and the French *Référentiel Pédologique*.^[12] It was launched to provide scientific depth and background to the FAO system, as it became clear that important new insight and knowledge about the world's soils had been generated. Additional objectives are to serve as a link between national classification systems, to serve as a basic language in soil science, to strengthen the application of soil science, and to facilitate the use of soil data for other professionals in soil and land management. Together with the superseded FAO system, WRB is the only system specifically designed to accommodate global soil resources.

The WRB system is a two level morphogenetic system. At the highest level, it has 30 reference soil groups that represent the major soil regions of the world (Table 3).

Table 2 Brief description of the 12 orders in *Soil Taxonomy* and their surface area in percent.

Order	Formative name element and meaning	Brief description of order	Percentage of ice-free land surface ^a
Alfisols	Alf (M. ^b <i>Pedalfer</i>)	Weakly weathered soils with argillic or kandic horizon and high base saturation in subsoil	9.6
Andisols	And (Jap. <i>an</i> , dark)	Soils formed from volcanic ash and with short-range order minerals	0.7
Aridisols	Id (L. <i>aridus</i> , dry)	Weakly developed soils of dry regions	12.0
Entisols	Ent (M. <i>recent</i>)	Soils with very weak pedogenic development	16.2
Gelisols	El (L. <i>gelare</i> , to freeze)	Soils with perennial permafrost within 100 cm of the surface	8.6
Histosols	Ist (Gr. <i>histos</i> , tissue)	Periodically wet soils with more than 30% organic matter to a depth of 30 cm	1.2
Inceptisols	Ept (L. <i>inceptum</i> , beginning)	Soils with incipient development in terms of weathering, color, and structure	9.8
Mollisols	Oll (L. <i>mollis</i> , soft)	Soils with mollic horizon and high base saturation	6.9
Oxisols	Ox (F. <i>oxide</i> , oxide)	Highly weathered red soils of the tropics	7.5
Spodosols	Od (Gr. <i>spodos</i> , wood ash)	Acidic soils with spodic horizon	2.5
Ultisols	Ult (L. <i>ultimus</i> , last)	Strongly weathered soils with argillic or kandic horizon and low base saturation in subsoil	8.5
Vertisols	Ert (L. <i>vertere</i> , to turn)	Soils with more than 30% expandable clays in all horizons and development of cracks during dry periods	2.4

^aIce-free land surface = 129,788,231 km².

^bM = meaningless syllable.

Table 3 Brief description of the 30 reference soil groups in *WRB for Soil Resources* and their surface area in percent.

Reference soil group	Formative element	Brief description of reference soil groups	Percentage of ice-free land surface ^a
Histosols	Gr. <i>histos</i> , tissue	Periodically wet soils with more than 30% organic matter to a depth of 30 cm	2.2
Cryosols	Gr. <i>kraios</i> , cold ice	Soils with perennial permafrost	12.3
Anthrosols	Gr. <i>anthropos</i> , man	Soils transformed by human activity through deep working, application of plaggan manure, or sediment-rich irrigation water	?
Leptosols	Gr. <i>leptos</i> , thin	Weakly developed shallow soils with hard rock within 25 cm of surface	11.5
Vertisols	L. <i>vertere</i> , to turn	Soils with more than 30% expandable clays in all horizons and development of cracks during dry periods	2.3
Fluvisols	L. <i>fluvius</i> , river	Soils developed on stratified floodplain deposits	2.4
Solonchaks	Russ. <i>sol</i> , salt; <i>chak</i> , salty area	Soils with accumulation of soluble salts	2.1
Gleysols	Russ. <i>gley</i> , wet soil	Poorly drained waterlogged soils with mottles and anaerobic conditions	5.0
Andosols	Jap. <i>an</i> , dark; <i>do</i> , soil	Soils formed from volcanic ash and with short-range order minerals	0.8
Podzols	Russ. <i>pod</i> , under; <i>zola</i> , ash	Acidic soils with accumulation of amorphous material consisting of organic matter and Al with or without Fe in the subsoil (spodic horizon)	3.4
Plinthosols	Gr. <i>plinthos</i> , brick	Soils with mottled appearance because of segregation of iron oxides, which harden upon exposure to the air	0.4
Ferralsols	L. <i>ferrum</i> , iron; <i>alumen</i> , aluminum	Highly weathered red soils of the tropics with high content of Fe/Al oxides and LAC clays	5.2
Solonetz	Russ. <i>sol</i> , salt; <i>etz</i> , strongly	Soils with high content of exchangeable sodium	0.9
Planosols	L. <i>planos</i> , flat	Soils on flat terrain with abrupt textural change from sandy surface to clayey subsoil, which causes seasonal water stagnation	0.9
Chernozems	Russ. <i>chern</i> , black; <i>zemlja</i> , earth	Dark-colored loess soils of the grass steppe regions with high content of organic matter and accumulation of calcium carbonate in subsoil	1.6
Kastanozems	L. <i>kastanea</i> , chestnut; Russ. <i>zemlja</i> , earth	Brown-colored soils of the grass steppe regions with high content of organic matter and accumulation of calcium carbonate or gypsum in subsoil	3.2
Phaeozems	Gr. <i>phaios</i> , dusky; Russ. <i>zemlja</i> , earth	Dark-colored soils of the steppe regions with high content of organic matter and leaching of calcium carbonate	1.3
Gypsisols	L. <i>gypsum</i> , gypsum	Soils with accumulation of secondary gypsum	0.6
Durisols	L. <i>durus</i> , hard	Well-drained, arid region soils with compact and silica-cemented layer	?
Calcisols	L. <i>calx</i> , lime	Soils with accumulation of secondary calcium carbonate as soft powder or as cemented layers	5.6
Albeluviols	L. <i>albus</i> , white; <i>eluere</i> , to wash out	Soils with clay accumulation in subsoil penetrated by tongues of coarse-textured and lighter material of eluvial horizon	2.2
Alisols	L. <i>alumen</i> , alum	Acidic soils with clay accumulation in subsoil, high CEC, and low base saturation	0.7
Nitisols	L. <i>nitidus</i> , shiny	Low latitude, deep, well-drained soils with clay accumulation in subsoil and shiny ped-surfaces	1.4
Acrisols	L. <i>acer</i> , strong acid	Acidic, weathered soils with clay accumulation in subsoil, low CEC, and low base saturation	7.0
Luvisols	L. <i>luere</i> , to wash	Soils with clay accumulation in subsoil, high CEC, and high base saturation	4.5
Lixisols	L. <i>lixia</i> , washing	Strongly weathered soils with clay accumulation in subsoil, low CEC, and high base saturation	3.0
Umbrisols	(L. <i>umbra</i> , shade = dark	Well-drained soils with dark-colored, organic-rich, acidic surface horizon (umbric horizon)	0.7
Cambisols	L. <i>cambiare</i> , to change	Soils with incipient weathering and slight development of color and structure	10.5
Arenosols	Gr. <i>arena</i> , sand	Soils with sandy texture throughout	6.3
Regosols	Gr. <i>rhegos</i> , blanket	Weakly developed fine-textured soils	1.8

^aIce-free land surface = 129,788,231 km².

Table 4 Nearest WRB equivalents to the 12 orders in *Soil Taxonomy*.

Soil Taxonomy orders	WRB reference soil groups
Alfisol	Luvissols, Lixissols, Planossols, and Solonetz
Andisol	Andosols
Aridisol	Leptosols, Gypsisols, Durisols, Calcisols, Solonchaks
Entisol	Arenossols, Fluvisols, Regossols, and Gleysols
Gelisols	Cryosols
Histosols	Histosols
Inceptisol	Cambisols; Fluvisols, Gleysols, and Solonchaks
Mollisol	Chernozems, Kastanozems, Phaeozems
Oxisol	Ferralsols, Plintossols
Spodosol	Podzols
Ultisol	Alisols; Acrisols; and Albeluvisols and Nitisols
Vertisol	Vertisols

In Table 4, the nearest WRB equivalents to the 12 soil orders in *Soil Taxonomy* are shown.

At the second level, the WRB is a classification system where combinations of 121 qualifiers may be added to the reference groups, thus allowing very precise characterization of individual soil profiles. Traditional soil names have been retained to the greatest extent possible, e.g., Podzols and Solonetz, but in case of ambiguity of traditional names, new names derived from various linguistic roots were selected, e.g., Albeluvisols instead of Podzoluvisols.

For describing the reference soil groups, WRB has defined 40 diagnostic horizons, 12 diagnostic properties [assemblages of soil characteristics, e.g., texture, color, soil reaction, and cation exchange capacity (CEC)], and 7 diagnostic soil materials (original parent materials that are only slightly altered, e.g., anthropogenic material, fluvic material, or sulfidic material). Diagnostic horizons and properties are the expression of pedogenic processes and have been defined as representing generally agreed pathways of soil formation and importance for soil management and use. Many of the differentiating criteria have been adopted from *Soil Taxonomy* either directly and simplified, e.g., the mollic, umbric, and ochric horizons, or modified and simplified, e.g., the argic B horizon (from the argillic subsurface horizon) and the ferrallic B horizon (from the oxic subsurface horizon). In contrast to *Soil Taxonomy*, the use of soil climate is not part of the system, except for the effects it may have on soil properties. Assignment of soils into the WRB categories is by means of a key. To illustrate how second level classification systems of WRB work, a reddish-colored Vertisol with a surface-near calcic horizon would be termed a Chromic Vertisol.

CONCLUSION

Numerous systems have been developed and exist for classification of soils and soil materials, serving variety of purposes. As international references, however, the agriculturally biased, morpho-genetic systems, Soil Taxonomy and WRB, for soil resources have gained widespread acceptance. With increasing demands on small-scale quantitative soil information and a reality where soil information must be able to address a broad range of environmental issues other than agricultural production, the soil research community is faced with a major challenge in meeting this demand and thinking along new lines to develop new soil information systems.

REFERENCES

1. Soil Survey Staff. Soil taxonomy. In *A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; United States Department of Agriculture, Natural Resources Conservation Service, Agriculture Handbook No. 436; U.S. Government Printing Office: Washington, DC, 1999.
2. FAO-ISRIC-ISSS. *World Reference Base for Soil Resources*; World Soil Resources Report 84; FAO: Rome, 1998.
3. FAO-UNESCO. *FAO-UNESCO Soil Map of the World. 1: 5,000,000*; Legend; UNESCO: Paris, 1974; Vol. I.
4. FAO. *FAO-UNESCO Soil Map of the World, Revised Legend, with Corrections and Updates*. World Soil Resources Report 60; FAO: Rome, 1988. Reprinted with Updates as Technical Paper 20; ISRIC: Wageningen, 1997.
5. Isbell, R.F. *The Australian Soil Classification*, Rev. Ed.; Australian Soil and Survey Hand Book; CSIRO Publishing: Collingwood, 2002.
6. Isbell, R.F.; McDonald, W.S.; Ashton, L.J. *Concepts and Rationale of the Australian Soil Classification*; ACLEP, CSIRO Land and Water: Canberra, 1997.
7. DoPrado, H. *Manual de Classificação de Solos do Brasil*, 3rd Ed.; FUNEP: Brasília, 1996.
8. Canada Soil Survey Committee, Subcommittee on Soil Classification. *The Canadian System of Soil Classification*; Canadian Department of Agriculture, Publication No. 1646; Supply and Services Canada: Ottawa, 1978.
9. Soil Classification Working Group. *The Canadian System of Soil Classification*, 3rd Ed.; Research Branch, Agriculture Canada, Publication 1646; NRC Research Press: Ottawa, 1998.
10. Gong, Z. *Chinese Soil Taxonomic Classification*; Institute of Soil Science, Academia Sinica: Nanjing, 1994.
11. Baize, D.; Girard, M.C. *Référentiel Pédologique*; Institut National de la Recherche Agronomique: Paris, 1995.
12. VASKhNIL (Dokuchaev Institute of Soil Science). *Classification and Diagnostics of Soils of the USSR*; Translated by S. Viswanathan; Amerind Publishing Co: New Delhi, 1986.
13. Shishov, L.L.; Sokolov, I.A. Genetic classification of soils in the USSR. In *Soil Classification, Reports of the*

- International Conference on Soil Classification*; Rozanov, B.G., Ed.; USSR State Committee for Environmental Protection: Moscow, 1990; 77–93.
14. Stolbovoi, V. *Soils of Russia: Correlated with the Revised Legend of the FAO Soil Map of the World and World Reference Base for Soil Resource*; IIASA: Laxenburg, 2000.
 15. Soil Survey Staff. *Soil Classification—A Comprehensive System—7th Approximation*; Soil Conservation Service, United States Department of Agriculture, Agriculture Handbook; U.S. Government Printing Office: Washington, DC, 1960.
 16. Soil Survey Staff. *Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys*; Soil Conservation Service, United States Department of Agriculture, Agriculture Handbook No. 436; U.S. Government Printing Office: Washington, DC, 1975.
 17. Smith, G.D. *The Guy Smith Interviews: Rationale for Concepts in Soil Taxonomy*; Soil Management Support Services, Department of Agronomy, Cornell University: Ithaca, 1986.
 18. <http://www.soils.usda.gov/technical/classification/technical>.
 19. Bridges, E.M.; Batjes, N.H.; Nachtergaele, F.O. *ISSS Working Group RB, World Reference Base for Soil Resources: Atlas*; ISRIC–FAO–ISSS–Acco: Leuven, 1998.
 20. Deckers, J.A.; Nachtergaele, F.O.; Spargaren, O.C. *ISSS Working Group RB, World Reference Base for Soil Resources: Introduction*, 1st Ed.; ISRIC–FAO–ISSS–Acco: Leuven, 1998.
 21. <http://www.fao.org/landandwater/agll/wrb/default.stm>.
 22. Driessen, P.M.; Dudal, R. *The Major Soils of the World*; Agricultural University: Wageningen: Katholieke Universiteit Leuven: Belgium, 1991.

Classification Systems: Netherlands

Alfred E. Hartemink

International Soil Reference and Information Centre (ISRIC), Wageningen, the Netherlands

Henk de Bakker

Netherlands Soil Survey Institute (StiBoKa), Wageningen, the Netherlands

Abstract

Although people have always classified soils, only since the mid-19th century has soil classification emerged as an important topic within soil science. It forced soil scientists to think systematically about soils and its genesis, and it was developed to facilitate communication between soil scientists. It has also been the cause for much debate and confusion, but two internationally accepted systems have emerged: Soil Taxonomy and the World Reference Base for Soil Resources (WRB). In addition, many national soil classification systems exist whose methods and approaches are based on one of the international soil classification systems or which have influenced these systems. In this entry, we describe the Dutch soil classification system, which was essentially developed after the fourth International Congress of Soil Science in Amsterdam in 1950.

INTRODUCTION

The Netherlands is a low-lying country with the lowest point at nearly 7 m below mean sea level just north of Rotterdam. The highest point is 321 m above mean sea level and located in the southern part of the country. About half of the country is below sea level and would be inundated without dikes and dunes. It is also a wet country, and more than 90% of the soils have groundwater within 140 cm of the soil surface during winter. As a result, most Dutch soils are hydromorphic and require artificial drainage when taken in use. There is no consolidated rock, and the parent material is alluvial (marine or fluvial), or aeolian, glacial, or organic. About one-third of the country consists of embanked forelands from either the North Sea or the rivers Scheldt, Rhine, and Meuse; these polders have Holocene loamy and mostly clayey soils. About 40% of the Dutch soils have Pleistocene sands as parent material and 2% loess. Peat areas comprise about 25% of the Netherlands.

BRIEF HISTORY OF SOIL MAPPING AND SOIL CLASSIFICATION

W.C.H. Staring (1808–1877) made the first geological map of the Netherlands at a scale 1:200,000 in the 1850s. At the request of a teacher's society, the map was simplified and used in education on basic schools for over a century. Until the 1940s, there was little activity in soil surveying, although much work was done in the Dutch East Indies by E.C.J. Mohr (1873–1970), and there was intensive and

detailed soil fertility research by D.J. Hissink (1874–1956) in Groningen.

The focus of Staring and his followers was on the topsoil, and relatively little was known about the whole solum, soil genesis, and geography. This changed in the 1930s when W.A.J. Oosting (1898–1942) started soil mapping around Wageningen. Inspired by the work of Oosting, C.H. Edelman (1903–1964) started soil surveying with his students in the riverine clay area during World War II. The Dutch Soil Survey Institute (StiBoKa) was founded by Edelman in 1945. Soil maps were very much in demand in areas with severe war damage such as, e.g., polders that were inundated with seawater, minefields, and for restoring airfields to agricultural use. His book *Soils of the Netherlands* was published on the occasion of the 1950 Congress of the International Society of Soil Science (ISSS) in Amsterdam.^[1] The accompanying map (scale 1:400,000) reflects the main Staring classes (i.e., sea clay, river clay, sand, loess, and peat), but for the subdivision physiographic criteria are used. The Edelman approach was strongly based on geology and the landscape, which resulted in the Dutch school of physiographic soil mapping. The fieldwork for the map at a scale 1:200,000 took place between 1952 and 1954, and it was published in 1960. In 1952, the Soil Survey Institute started to map the whole country at 1:50,000 (110 sheets), which was published between 1964 and 1995.^[2]

The physiographic approach for soil mapping was less suitable for soil classification, and the legends of the map provided more insight into geogenesis than into pedogenesis. In the 1950s and 1960s, soil classification was widely discussed because of an increasing number of soil scientists

working in soil mapping, increased international interaction following the Congress of ISSS in 1950, and the development of soil classification in the United States. Under the guidance of the main author of *Soil Taxonomy*, G.D. Smith, Dutch soil scientists discussed the series of approximations and a committee started to frame a system of soil classification using inherent soil properties as differentiating criteria rather than physiographic and geological criteria. It formed the base for the Dutch system of soil classification.^[3]

THE DUTCH SYSTEM OF SOIL CLASSIFICATION

At the highest level, soils are differentiated by soil-forming processes, and as such, the system has a profound pedogenic base. Units in the classification system are defined by morphometric properties and characteristics of the soil profile. There are five orders, and a brief description including the equivalents in *Soil Taxonomy* is listed in Table 1. A Podzol or textural B horizon and the formation of an A horizon are diagnostic criteria at the order level. In addition, the absence of soil formation or the presence of peat as parent material is used at the order level. No soil chemical criteria are used at the order level because of the heavy use of manure and inorganic fertilizer applications in the Netherlands.

The five orders are subdivided into 13 suborders, 23 groups, and 58 subgroups (Table 2). Criteria to distinguish the lower levels of classification include texture, organic

Table 1 The five orders and their diagnostic properties and equivalents in Soil Taxonomy and the WRB.

Orders	Main diagnostic property	Approximate equivalent in Soil Taxonomy	Approximate equivalent in the WRB
Peat soils	>40 cm peat within 80 cm depth	Histosols	Histosols
Podzol soils	Soils with Podzol B	Spodosols (aquods and orthods)	Podzols
Brick soils	Soils with a “brick” layer	Alfisols (hapludalfs)	Luvisols and Planosols
Earth soils	Soils with a mineral earthy layer	Mollisols, Inceptisols (aquic soils with a mollic or anthropic epipedon)	Anthrosols
Vague soils	Without foregoing diagnostic horizons	Entisols and Inceptisols	Fluvisols, Gleysols, Anthrosols, Regosols, Cambisols, and Arenosols

Table 2 The soil classification of the Netherlands at the higher levels.

Order	Suborder	Group	Subgroup
Peat soils	Earthy peat soils	Clayey earthy peat soils	“Aar” peat soils
			“Koop” peat soils
		Clay-poor earthy peat soils	“Bo” peat soils
			“Made” peat soils
			“Vliet” peat soils
	Raw peat soils	Initial raw peat soils	“Weide” peat soils
			“Waard” peat soils
		Ordinary raw peat soils	“Meer” peat soils
			“Vlier” peat soils
Podzol soils	Moder Podzol soils	Moder Podzol soils	“Holt” Podzol soils with a sand cover
			“Loo” Podzol soils
			“Hoek” Podzol soils
			“Horst” Podzol soils
			“Holt” Podzol soils
			“Moer” Podzol soils with a clay cover
			“Moer” Podzol soils with a sand cover
			“Dam” Podzol soils
			“Moer” Podzol soils
			“Veld” Podzol soils with a clay cover
	Hydropodzol soils	Peaty Podzol soils	“Veld” Podzol soils with a sand cover
			“Laar” Podzol soils
			“Veld” Podzol soils
			“Haar” Podzol soils with a sand cover
			“Kamp” Podzol soils
			“Heuvel” Podzol soils
			“Haar” Podzol soils
		Ordinary Hydropodzol soils	“Beemd” brick soils
			“Kuיל” brick soils
			“Berg” brick soils
			“Del” brick soils
			“Rooi” brick soils
Brick soils	Hydrobrick soils	Hydrobrick soils	“Daal” brick soils
			“Rade” brick soils
			Brown “enk” earth soils
			Black “enk” earth soils
			“Tuin” earth soils
	Xerobrick soils	Xerobrick soils	
Earth soils	Thick earth soils	“Enk” earth soils	

(Continued)

Table 2 The soil classification of the Netherlands at the higher levels. (Continued)

Order	Suborder	Group	Subgroup
Vague soils	Hydroearth soils	Peaty earth soils	“Plas” earth soils
			“Broek” earth soils
		Sandy hydroearth soils	Brown “beek” earth soils
			“Goor” earth soils
			Black “beek” earth soils
		Clayey hydroearth soils	“Lied” earth soils
			“Tocht” earth soils
			“Woud” earth soils
	Xeroearth soils	Sandy xeroearth soils	“Leek” earth soils
			“Akker” earth soils
		Clayey xeroearth soils	“Kant” earth soils
	Initial vague soils	Clayey hydrovague soils	“Hof” earth soils
			“Gors” vague soils
			“Slik” vague soils
		Sandy hydrovague soils	“Vlak” vague soils
			“Drecht” vague soils
			“Nes” vague soils
Vague soils	Hydrovague soils	Clayey hydrovague soils	“Polder” vague soils
			“Krijt” vague soils
			“Sandy” xerovague soils
	Xerovague soils	Clayey xerovague soils	“Duin” vague soils
			“Vorst” vague soils
			“Ooi” vague soils

Source: From de Bakker & Schelling.^[3]

matter content, hydromorphic characteristics, peaty topsoil, plaggen epipedon, and ripening class. Processes such as gley formation, ripening, and the influence of cultivation are considered at the suborder level, and hydromorphic properties are profoundly present at the suborder level as in many other systems of soil classification. At the group level, differences in parent material are considered as are the presence of peat layers and the stage of ripening. At the lowest level (subgroup), topsoil properties are important. Although the subgroup level is the lowest level in the Dutch Soil Classification System, lower classification levels are used in the 1:50,000 soil map of the Netherlands. It is

interesting to note that the 1:50,000 soil map reflects Starling's main classes of the mid-1800s.^[4]

SOIL NOMENCLATURE

The highest categories (order, suborder, and group) have names adopted from the existing terminology, but in some cases, artificial terms are chosen. At the subgroup level, names have been chosen that are a combination of Dutch toponyms and the name of the order; e.g., “Aar” peat soils are found around the villages of Langeraar and Ter Aar in the province of South Holland, whereas most of the “koop” peat soils are names after villages ending in koop (e.g., Boskoop and Teckop). Some of the names stem from medieval reclamation (e.g., “rooi” from uprooting shrubs or trees). Other names are from rivers and lakes or low-lying areas (e.g., “tocht” and “daal”) or from the dominant type of land use (“weide” = pasture and “akker” = arable land).^[3] An overview on toponyms and soil nomenclature is given by Siderius and de Bakker.^[5]

CONCLUSIONS

The Dutch Soil Classification System was developed in the 1960s following decades of physiographic soil mapping. The system has a strong pedogenic base at the highest level, and parent material and hydromorphic properties are important. There are 5 orders, 13 suborders, 23 groups, and 58 subgroups. This system was used as the background of the legend for the 1:50,000 soil map published in 110 sheets between 1964 and 1995. Although the system stems from the 1980s and no attempts have been made to update or revise the system, it continues to provide useful information on the genesis and geography of soils in the Netherlands.

REFERENCES

1. Edelman, C.H. *Soils of the Netherlands*; North-Holland Publishing Company: Amsterdam, 1950.
2. Steur, G.G.L. The soil map of the Netherlands, scale 1: 50, 000. Some aspects of the grouping of the legend and the nomenclature of the mapping units. *Boor Spade* **1966**, *XV*, 43–58.
3. de Bakker, H.; Schelling, J. *Systeem van Bodemclassificatie voor Nederland, de Hogere Niveaus*; PUDOC: Wageningen, 1989.
4. de Bakker, H. Purposes of soil classification. *Geoderma* **1970**, *4*, 195–208.
5. Siderius, W.; de Bakker, H. Toponymy and soil nomenclature in the Netherlands. *Geoderma* **2003**, *111*, 521–536.

Classification Systems: Russian

Maria Gerasimova

Faculty of Geography, Moscow Lomonosov University, Moscow, Russia

Abstract

The first soil classification systems appeared in Russia on the eve of the 20th century, and the imprint of their ideology and conceptual priorities has remained very strong in the later systems. Both early Russian schemes developed by Dokuchaev and Sibirtsev are traditionally regarded as factor genetic, i.e., they comprise the notions on the origin of soils, soil-forming processes, and soil position in the system of geographical zones. However, some importance was attributed to soil properties, moisture and temperature regimes, and the stage of soil development. These major features of pedogenetic manifestations and environment have proved to be key points in the further development of the classification schemes in Russia. Subsequent schemes differed in the priorities given either to the environment (zones) or to soil genesis or to soil properties.

BACKGROUND INFORMATION

The diversity of approaches to soil classification was always inherent to Russian pedology. Unlike the early and existing American systems, all Russian schemes started from “above”: the lowest taxa were derived from the upper ones, the names of which were retained in the low-category soil name; in other words, there was no similar concept to that of the American Series in the Russian approach. The majority of systems were hierarchical, with the soil type being the central unit. The definition of the soil type serves to illustrate the fundamental principles underlying the classification (Table 1). During the development of the various classification systems, different sets of criteria for the upper taxonomic levels were proposed: soil genesis in terms of soil-forming processes, or soil age; soil chemical and/or morphological properties; soil horizons; and soil-forming agents. The categories higher in the system than soil type differed in definition depending upon the conceptual approach of the author(s), while the below-type categories have remained basically unchanged since they were first specified in the late 1950s. Not all classification systems have distinguished categories at levels higher than soil type.

The “discovery” of soil science by Dokuchaev^[1] resulted in a simple and descriptive zonal soil classification system, which was the *first* in Russia. It was presented as a table with short comments in 1886. All soils—14 groups were subdivided into normal—*zonal* soils, *transitional* (boggy-meadow soils, Rendzinas, and Solonchets), and *abnormal* ones (e.g., alluvial and eolian). Groups of zonal soils were ascribed to zones and characterized in terms of processes, sediments, climate, vegetation, fauna, and topography. The system of Sibirtsev^[2] was slightly more advanced; the soils known in those times were grouped into *zonal*, *intrazonal*, *incomplete*, and *surface*

geologic formations, and lower categories were outlined as related to soil texture and parent rocks.

The ecological–genetic system of Gerasimov et al.,^[3] under supervision of Prassolov, is widely recognized to be based on the strictly defined notion of “genetic soil type,” with subdivision related to the potential drainage and/or position in a catena. Soil sequences were identified as automorphic, hydromorphic, and semihydromorphic. The definition involves processes of substance migration–accumulation–transformation, geographic environments, soil evolution, and pedogenetic processes, which are specifically emphasized. A first categorization of subtypes and species was proposed; species—stages of pedogenetic process development—were later transformed into quantitative groupings. Units of the lowest category were identified in relation to land use and soil degradation (old-arable limed, old-irrigated, secondary saline, etc.).

In 1956, Ivanova^[4] published a classification for world soils, which was widely criticized by foreign and some Russian soil scientists because of its over reliance upon climatic subdivisions. Indeed, the highest taxa were differentiated with respect to climate and vegetation, the names of soil types were supplemented with landscape characteristics (e.g., red soils of dry savanna). The degree of soil hydromorphism (automorphic, transitional, and hydromorphic series) was also taken into account at the higher (above-type) level. Given this complexity, it is perhaps not surprising that the number of genetic soil types reached 80. The system was termed evolutionary genetic–geographical and highly appreciated in the USSR for its geographical essence.

Two more classification systems illustrate the diversity of concepts that have prevailed in the former USSR. Both were concerned with the above-type categories, and they have common approaches to the choice of soil properties serving as differentiating criteria.

Table 1 Scheme of taxonomic units in some classification systems in USSR/Russia.

	Author(s) (year)					
	Dokuchaev ^[1]	Gerasimov et al. ^[3]	Official systems (1967, 1977)	Kovda et al. ^[5]	Glazovskaya ^[6]	New system ^[10]
Criteria for the upper level	Zonality	No upper level		Soil age, budget of substances	Eh, pH, weathering, humus type	Pedogenesis vs. lithogenesis
(Genetic) soil type						
Criteria for lower levels	Subtype	Catenary and plants effects	Additional process	Not discussed		Diagnostic features
	Genus	—	Parent material			CEC, type of salinity
	Species	Stages	Quantitative characteristics			
	Variety	Texture				
	Phase	Human impacts	Degree of erosion	Not discussed		Parent material

^aOnly two levels were presented in this pioneer system, both related to zonality.

Perception of soils as results of broad geochemical mass–energy exchange processes interacting in time parallel to the evolution of landscapes served as background for the world system elaborated by Kovda et al.^[5] The upper category is represented by broad soil geochemical formations differentiated by major trends of weathering and clay minerals, types of humus, acid–base properties, supplemented with characteristics of climate and vegetation. For example: “Formation of acid, strongly freezing or cryogenic soils. Weak weathering. Clay of illite–hydromica type. Main processes: humus and peat accumulation, weak podzolization, gley, cryogenesis. Taiga vegetation. Cold boreal climate” (1975). The second entry—stadial groups of soils—corresponded to hypothetical evolution stages from submerged sites to excessively drained ones. The following stadal groups were proposed: hydroaccumulative, hydromorphic, mesohydromorphic, paleohydromorphic, proterohydromorphic, automorphic (including mountain soils), and paleoautomorphic. The number of stadal groups and of soil geochemical formations varied in different versions of this system; moreover, climatic zones or belts were sometimes introduced, and climatic facies appeared in the final version. The latter was published as the legend of the World Soil Map, scale 1:10 M, 1975. The system was severely criticized for a high degree of clumsiness, insufficient paleogeographical substantiations, and internal inconsistency.

A similar geochemical approach was implemented by Glazovskaya^[6] in the classification of world soils in late 1960s. Most of her genetic–taxonomic ideas were applied in the legend of the World Soil Map (produced for higher level educational use at universities, colleges, and advanced secondary schools) at a scale of 1:15 M, in 1981. This system is strictly hierarchical: there are three above-type levels: associations, generations (classes), and families. Soil types are arranged in families in accordance with distinct rules. Lower categories were preserved as traditional. Criteria for differentiation at the highest category are pH

and soil features indicative of redox conditions; members of the second category are differentiated by a range of process-related properties, including broad processes of organic matter accumulation, formation of secondary minerals, translocation of substances, and effects of groundwater. The difference in the composition of pedogenetic accumulations (humus, illuvial horizons, concretions, etc.) allows the specification of families within classes that were the main soil units on the world map.

Official systems—an initial approximation produced in 1967 and the final version of 1977(a) derived from the system of Ivanova of 1956. Both systems lacked any above-type categories, whereas the lower ones were clearly and adequately defined. The 1977 version is best known in Russia and abroad. It was translated into English (1986) and has been efficiently used in soil survey and small-scale mapping since then. The ecological–genetic principle of the system was applied practically for all hierarchical categories and soil diagnostics. The upper category corresponds to genetic soil types, of which there are more than 100. The subtypes are very large, both conceptually and in number. Many subtypes were required to cover many and diverse soil features owing to their dual nature: there are subzonal and facies subtypes, both variants being more geographic than taxonomic units. The concept of subzonal subtypes as “central images” of types and intergrades corresponding to subzones did not contradict the traditional interpretation of subtypes as modifications of types by superimposed processes, while the facies subtypes are really virtual phenomena. They were recognized only by climatic parameters (total air and topsoil temperatures > 10°C, depth and duration of soil freezing), and their names were like this: “very cold deeply (freezing to a depth in excess of 1 m) and long freezing ...”

The third—genus—category contained much information on parent rocks, groundwater, and soil history, and its differentiating criteria were far from being universal for different types. Species (or kinds in some translations) and

subspecies, unlike genera, were more consistent in specifying the quantitative parameters of pedogenetic processes (not horizon's properties!). For example, the thickness of humus-accumulative horizon and humus content in the 0–10 cm layer served to qualify the intensity of the “chernozemic process.” Texture characteristics and erosion phases were also accommodated in this system.

The traditional logic and consistency of this system made it very popular in the Soviet Union. It really indicates the essence of soil formation in different zones (in accordance with the central images of types) and allows soil development in a changing environment to be forecast; it uses traditional nomenclature of Russian soils and its influence can be traced in many national classifications. However, this classification was considered by others to be outdated and is considered by some to be inappropriate—given the new knowledge—on the soils of Russia. Moreover, the system embraced only soils of agriculturally suitable areas. Little attention was paid to human-modified soils. Using the basic factor approach, this classification system could not provide distinct morphological diagnostics of soil types and subtypes in these environments. Finally, it was considered to be based on a static understanding of the nature of soils and soil-forming environments and could not accommodate soils that did not fit the subtype criteria; therefore, Siberian and northern soils could not be inserted into the system without considerable changes in its principles and structure. These drawbacks of the official system and the scarcity of appropriate information on the western classification systems provided the impetus for a new search in the field of soil classification in the late 1970s and 1980s.

The categorization of pedogenetic processes initiated by Neustruev in 1922 was further developed by his pupil Gerasimov,^[7] who, sticking to his “factors–processes–properties” triad, actively supported the idea of profile formulae (a similar profile formula approach using horizon sequences was suggested by Fitzpatrick).^[8] Such formulae presented sequences of horizons diagnostic for the majority of soil types. This was a change toward a substantive-genetic ideology in classifying soils.

Fridland^[9] initiated the development of a new composite soil classification system based on three components: 1) regime; 2) petrographic–mineralogical; and 3) profile-genetic component. The latter component presumed the assessment of soil properties via diagnostic horizons, in the definition of above-type categories. It evolved into the new soil classification of 1997.^[10]

Classification of Soils of Russia^[11] is an open, substantive-genetic system, embracing soils of all regions. Soil properties related to genesis are used as criteria for the highest taxonomic categories (Trunks and Orders), and soil types, traditionally being central units of the system, are specified by sequences of diagnostic soil horizons.

These horizons are defined by the integrity of substantive soil properties, whose choice is controlled by pedogenetic processes. Environmental agents, including climatic parameters, are virtually excluded from the diagnostics of most soil taxa. Special attention is paid to human-modified soils. Together with corresponding natural soils, they are perceived as a conceptual or spatial continuum: from natural soils to modified natural-anthropogenic soils and, finally, to non-soil surface formations. The taxonomic position of human-modified soils does not take into account the goals and character of impacts on soil and the level of soil fertility; it is fully dictated by the morphology of soil profile. Traditional soil names were preserved and supplemented by new constructions for human-modified soils and non-soils.

The new Russian system has a number of features in common with the International and American systems in terms of methodology, in particular, in the attitude to soil horizons. However, in spite of many efforts, the correlation of soils between the systems remains broadly inadequate, and this may be a challenge in the future.

REFERENCES

1. Dokuchaev, V.V. Classification of soils of professor Dokuchaev, V.V. (Northern Hemisphere). *Pochvovedeniye* **1900**, 2, Appendix.
2. Sibirtsev, N.M. *Soil Classification as Applied to Russia (Table)*; St. Petersburg, 1896.
3. Gerasimov, I.P.; Zavalishin, A.A.; Ivanova, E.N. A new scheme of a general soil classification for the USSR. *Pochvovedeniye* **1939**, 7, 10–43.
4. Ivanova, E.N. Essai de classification générale des sols. In *Rapports VI Congrès International de la Science du Sol*, Paris, France, 1956; Vol. E (Commission) V, 387–394.
5. Kovda, V.A.; Lobova, E.V.; Rozanov, B.G. Problem of world soils classification. II. Historical-genetic systematics of soils. *Pochvovedeniye* **1967**, 7, 3–16.
6. Glazovskaya, M.A. *Soils of the World*; Moscow University Press: Moscow, 1972; Vol. 1, 231 pp.
7. Gerasimov, I.P. Elementary pedogenetic processes as a basis for genetic diagnostics of soils. *Pochvovedeniye* **1973**, 5, 102–113.
8. Fitzpatrick, E.A. *Pedology: A Systematic Approach to Soil Science*; Oliver and Boyd: Edinburgh, 1971.
9. Fridland, V.M. *Major Principles and Elements of a Basic Soil Classification System and the Program of Its Development*; Dokuchaev Soil Institute: Moscow, 1982; 149 pp.
10. *Classification and Diagnostics of the Soils of the USSR*; USDA and National Science Foundation: Washington, 1986; Kolos Publication: Moscow, 1997; 222 pp.
11. Tonkonogov, V.D.; Lebedeva, I.I.; Shishov, L.L. *Classification of Soils of Russia*; Dokuchaev Soil Institute: Moscow, 1997; English Version: Arnold, R.W., Ed. *Russian Soil Classification System*; Moscow, 2001; 220.

Classification Systems: South African

Jeffrey C. Hughes

Soil Science, School of Environmental Sciences, University of KwaZulu-Natal,
Scottsville, South Africa

Abstract

This entry presents the South African classification system. The available soil classification has its roots in a number of important field studies. Concepts and characteristics brought in from other systems include the luvic character of B horizons, the placic pan, and the use of the plasticity index to distinguish between vertic and non-vertic topsoils. However, the system has retained its local character.

INTRODUCTION

The new soil classification has its roots in a number of important field studies, the most influential of which covered 11,000 square miles of the Tugela (Thukela) river basin in KwaZulu-Natal.^[1] Two coauthors of that survey, C.N. MacVicar (later, Director, Natal Region, Department of Agriculture) and J.M. de Villiers (later, Chair of Soil Science at the University of Natal), became two of the most influential figures in soil classification in South Africa. The classification presented in 1969, in the publication *Soils of the Tugela Basin*,^[1] laid the foundation for those who followed.

THE TUGELA BASIN CLASSIFICATION

The soils of the survey area were subdivided into 29 soil forms at the highest level in the classification and into 110 soil series at the most detailed level. Thus, just two levels in the hierarchy were recognized—a feature that has remained in all subsequent changes. The concept of the soil form was “developed to include all soils that have the same kind and number of diagnostic horizons, arranged in the same vertical order.”^[1] The soil series were distinguished from the forms by using soil properties that were either not used to define the forms or, where properties were used, a narrower range of variation was employed at the series level. The major criteria used for defining the soil series were texture, base status, and calcareousness. In individual soil forms, characteristics such as pH and surface crusting were used. The classification also recognized that not all the possible soil series were present within the basin and that there would be others found elsewhere in the country. Gaps were therefore left so that the series could be slotted in as they were identified.

Eighteen diagnostic horizons were defined with their general connotation, i.e., the meaning behind the definition and where they were most likely to be found. All the soil

forms and series were named after the geographic location where the “type” of soil was identified. One of the major features of this classification was its simplicity since almost all the properties needed to classify a soil were identifiable in the field. The main exception to this was base status (dystrophic, mesotrophic, or eutrophic), although this could often be inferred from other characteristics such as the vegetation, climate, and soil structure. Base status, together with other field-estimated or measured properties such as pH and texture, needed confirmation by laboratory measurement to finalize the classification of some soils. The degree of success that this system achieved can be measured by so many of its features having been retained in future editions of the classification.

THE BINOMIAL SYSTEM

Soil Classification—A Binomial System for South Africa

This was the first detailed system that allowed for the classification of the soils of South Africa,^[2] as opposed to individual classifications developed for specific areas of the country. Its publication resulted only after considerable consultations with the major role players in South Africa and extensive in-field testing. The authors also recognized in the preface that “This is the first edition; it is certain that the system as presented here is not the final word.”^[2] The classification in many ways followed the same format as the Tugela Basin survey. The name “binomial” came from the fact that soils were classified into two classes: soil form and soil series. In total, 41 soil forms and 504 soil series were recognized and described. As in the Tugela Basin survey, the soil forms were defined on the basis of a unique vertical sequence of diagnostic horizons. A total of 5 topsoil and 15 subsoil horizons were defined, and, as before, each had a connotation in terms of presumed genesis and provenance. The soil series (and soil forms) had unique

geographic names, and the series were also given a number within their soil form, which in turn had a unique two-letter code. Thus, e.g., the Griffin form, Farmhill series has a notation Gf 13; the Cartref form, Cranbrook series has a notation of Cf 22. This greatly facilitated soil mapping, soil survey, and communication. The main soil series criteria were again texture, base status, and calcareousness, with horizon color (especially the distinction between defined red and non-red colors) and pH important in some soil forms. The format followed that begun in the Tugela survey in that each soil form was represented by a full-color plate and the series criteria were given on the facing page. This classification also included some useful features, notably that each soil form was given possible correlations to the Food and Agriculture Organization (FAO)^[3] and U.S. Department of Agriculture^[4] classifications. Although, of necessity, these were only broadly equivalent, they did introduce the local users of the classification of these two more widely known classifications and, internationally, allowed readers of the binomial system to relate to terminology with which they would be familiar. The system also included a full glossary. Perhaps most importantly, the system retained its user-friendly nature in that it remained possible to classify almost all soils to series level *in the field* (with the same provisos as earlier), without resorting to expensive laboratory analyses. A criticism leveled at the binomial system was that, because it was based on the evidence of previous soil surveys, it contained no differentiae that those surveys had not recognized.^[5] Thus, it lacked the previous flexibility. However, as stated earlier, the authors recognized that it was just “the first edition” and that as knowledge expanded the classification would have to be modified. Perhaps a more important limitation to the binomial system was that it was based strongly on the knowledge built up within the Province of KwaZulu-Natal and drew heavily on the Tugela Basin survey information. This resulted in the (very different) soils of the Cape Province, especially, being largely restricted to two Podzol forms and to the calcareous series of a few of the other soil forms. These restrictions made it inevitable that a second edition would be needed.

Soil Classification—A Taxonomic System for South Africa

The new soil classification^[6] is the second (revised) edition of the binomial system and retains many of the features of that classification although there are some fundamental differences. As stated in the preface, the main pressure for a second edition came from soil scientists in the Cape Province (see above). The major change to the classification itself was the decision to “include only those classes which accommodate similar, naturally occurring, more or less uniform soil bodies (polypedons), and to exclude arbitrarily chosen classes (mainly texture) which cause uniform soil bodies to be split artificially by class boundaries.”^[6]

This reflects a change in philosophy from the Tugela survey where, in explaining the use of the term “soil individual,” the authors stated “The U.S.D.A. Soil Survey Staff has defined discrete individuals of the soil population in terms of three-dimensional bodies of soil (polypedons). We fail to see the necessity, or logic of this.”^[1] The decision to exclude texture as a differentiating criterion, given that it was the major feature that distinguished the soil series in the binomial system, meant that soil series were omitted from the classification, and the lowest level became the soil family. It was also felt that the information needed to define the series satisfactorily was not generally available. By excluding the most detailed level of the classification, the soil series, the authors also tacitly recognized that the criticism leveled at the binomial system by Butler^[5] had some basis. Despite texture being excluded from the new classification, there is a recommendation that the texture of the *topsoil* should be appended to the formal communication of any particular soil. This emphasis on the topsoil texture only makes this classification unusual (unique?) since others generally use the texture of the subsoil as a differentiating feature. The classification thus consists of two levels, namely soil form and soil family. As in the first edition, all form and family names are from geographic locations, and none of the earlier series names is used in the second edition. Similarly, the forms are designated by unique two-letter codes and the families by four-digit numbers, e.g., the Estcourt form, Zastron family is designated Es 1100; the Clovelly form, Brereton family is designated by the code Cv 1200. The number of forms recognized has increased to 73, with a total of 404 soil families. There are 5 diagnostic topsoil horizons and 25 diagnostic subsoil horizons or materials (Fig. 1).

Concepts and characteristics brought in from other systems include the luvisc character of B horizons, the placic pan, and the use of the plasticity index to distinguish between vertic and non-vertic topsoils. However, the system has retained its local character in that the terminology of many diagnostic horizons is different from that used elsewhere. Examples include Podzol B (not spodic), hardpan carbonate horizon (not calcrete), and a particular usage of the term “saprolite.” The classification book also includes the methods of analysis as appendices used for the distinguishing criteria, an outline of the diagnostic horizons from the FAO^[7] and the U.S. Department of Agriculture^[8] soil classifications, and a full glossary. However, lacking from the second edition is any attempt to correlate the South African system with either of these more generally used international systems. The increase in the number of forms is mainly accounted for by the increase in the podzolic soils from 2 in the first edition to 8, the recognition of 19 arid-zone soils [either with calcareous B horizons or underlain by dorbank (a hard subsurface horizon cemented by silica)]—these two categories largely cater for the soils of the Cape Province—and 3 more soils underlain by wet materials. The families are distinguished by a total of

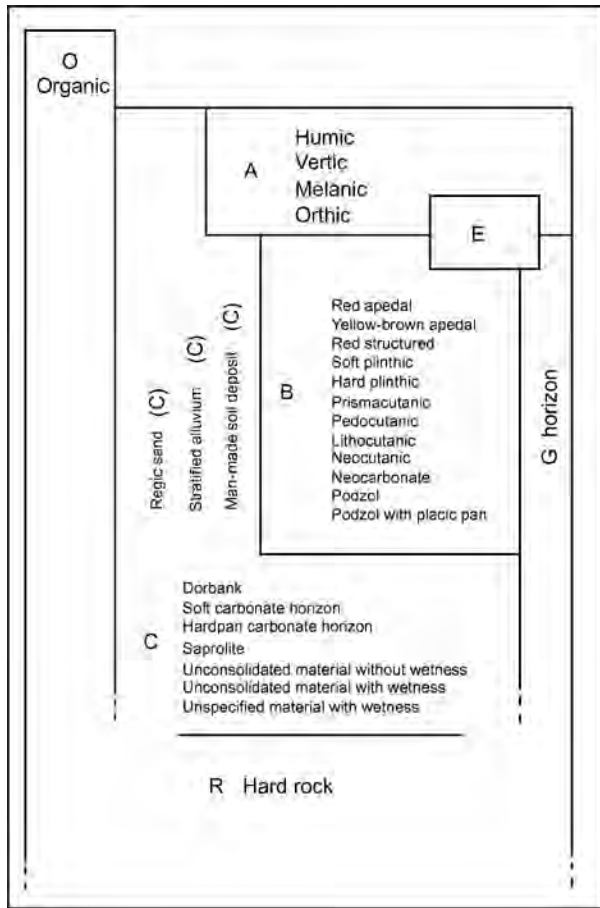


Fig. 1 Diagnostic horizons and materials.

Source: From Courtesy of the Director, Institute for Soil, Climate and Water, Pretoria and Soil Classification Working Group.^[6]

19 characteristics, of which color, base status, calcareousness, luvic character in the B horizon, and signs of wetness in the subsoil are the most important. The classification has retained many of the positive features of the earlier ones remaining one of the most user-friendly classifications, with it being a very large field classification. However, the addition of measurable characteristics such as the plasticity index to distinguish vertic soils and the addition of a cation exchange capacity criterion to distinguish borderline structured from apedal subsoils mean that laboratory data are more necessary now than in the past.

CONCLUSION

Discussions are underway among the South African soil science community with a view to decide on the next edition of the soil classification and what changes are required. The main item of contention is whether we are in a position to define soil series satisfactorily and, if so, what differentiae should be used. The decision not to reuse any of the earlier soil series names in the new classification means that they could be brought back into any future edition, provided that their characteristics can be agreed upon. What is certain is that the new classification will not be the final word as South African soil science develops, and more knowledge is gained about the country's soil mantle.

REFERENCES

1. Van der Eyk, J.J.; MacVicar, C.N.; de Villiers, J.M. *Soils of the Tugela Basin. A Study in Subtropical Africa; Natal Town and Regional Planning Reports Volume 15, Town and Regional Planning Commission; Natal: Pietermaritzburg, 1969; 1–263.*
2. MacVicar, C.N.; de Villiers, J.M.; Loxton, R.F.; Verster, E.; Lambrechts, J.J.N.; Merryweather, F.R.; Le Roux, J.; Van Rooyen, T.H.; Von, M.; Harmse, H.J. *Soil Classification. A Binomial System for South Africa; Department of Agricultural Technical Services: Pretoria, 1977; 1–150.*
3. Dudal, R. *Definitions of Soil Units for the Soil Map of the World; World Soil Resources Reports No. 33; World Soil Resources Office, Land and Water Development Division, FAO: Rome, 1968; 1–72.*
4. Soil Survey Staff. *Soil Classification. A Comprehensive System (7th Approximation); SCS, USDA; U.S. Government Printing Office: Washington, 1960; 1–265.*
5. Butler, B.E. *Soil Classification for Soil Survey; Monographs on Soil Survey; Clarendon Press: Oxford, 1980; 120–122.*
6. Soil Classification Working Group. *Soil Classification. A Taxonomic System for South Africa; Memoirs on the Agricultural Natural Resources of South Africa No. 15; Department of Agricultural Development: Pretoria, 1991; 1–257.*
7. FAO-UNESCO. *Soil Map of the World, 1:5,000,000; Vol. 1, Legend; UNESCO: Paris, 1974.*
8. Soil Survey Staff. *Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys; SCS, U.S. Department of Agriculture Handbook No. 436; U.S. Government Printing Office: Washington, 1975; 1–754.*

Clay Minerals

C.D. Barton

U.S. Department of Agriculture (USDA), Aiken, South Carolina, U.S.A.

Anastasios D. Karathanasis

Department of Agronomy, University of Kentucky, Lexington, Kentucky, U.S.A.

Abstract

Clay minerals refer to a group of hydrous aluminosilicates that predominate the clay-sized ($<2\ \mu\text{m}$) fraction of soils. These minerals are similar in chemical and structural composition to the primary minerals that originate from the earth's crust; however, transformations in the geometric arrangement of atoms and ions within their structures occur due to weathering. Primary minerals form at elevated temperatures and pressures and are usually derived from igneous or metamorphic rocks. Inside the earth, these minerals are relatively stable, but transformations may occur once exposed to the ambient conditions of the earth's surface. Although some of the most resistant primary minerals (quartz, micas, and some feldspars) may persist in soils, other less resistant minerals (pyroxenes, amphiboles, and a host of accessory minerals) are prone to breakdown and weathering, thus forming secondary minerals. The resultant secondary minerals are the culmination of either alteration of the primary mineral structure (incongruent reaction) or neoformation through precipitation or recrystallization of dissolved constituents into a more stable structure (congruent reaction). These secondary minerals are often referred to as phyllosilicates because, as the name implies (Greek: *phyllon*, leaf), they exhibit a platy or flaky habit, while one of their fundamental structural units is an extended sheet of silicate tetrahedra.

STRUCTURE OF CLAY MINERALS

The properties that determine the composition of a mineral are derived from its chemical foundation, geometric arrangement of atoms and ions, and the electrical forces that bind them together.^[1] Given that there are eight elements that constitute over 99% of the earth's crust (Table 1), the inclusion of these in the elemental makeup of soil minerals is understandable. Notwithstanding, the prevalence of silicon and oxygen in the phyllosilicate structure is logical. The SiO_4 tetrahedron is the foundation of all silicate (SiO_4) structures. It consists of four O^{2-} ions at the apices of a regular tetrahedron coordinated to one Si^{4+} at the center (Fig. 1). An interlocking array of these tetrahedral connected at three corners in the same plane by shared oxygen anions forms a hexagonal network called the tetrahedral sheet.^[2] When external ions bond to the tetrahedral sheet, they are coordinated to one hydroxyl and two oxygen anion groups. An aluminum (Al), magnesium, or iron ion typically serves as the coordinating cation and is surrounded by six oxygen atoms or hydroxyl groups resulting in an eight-sided building block termed an octohedron (Fig. 1). The horizontal linkage of multiple octahedra comprises the octahedral sheet. The minerals brucite ($\text{Mg}(\text{OH})_2$) and gibbsite ($\text{Al}(\text{OH})_3$) are similar to the octahedral sheets found in many clay minerals; however, phyllosilicates may contain coordinating anions

other than hydroxyls. Cations in the octahedral layer may exist in a divalent or trivalent state. When the cations are divalent (Mg , Fe^{2+}), the layer exhibits a geometry similar to brucite, such that electrical neutrality is maintained. In this arrangement, the ratio of divalent cations to oxygens is 1:2, and all three possible cation sites in the octahedron are occupied. This configuration and the respective sheet formed from an array of such as octahedral are referred to as trioctahedral. When the cations are trivalent (Al , Fe^{3+}), the charge balance is maintained by leaving one of every three octahedral cation sites empty. Under this configuration, the ratio of trivalent cations to oxygens is 1:3, and the layer exhibits a gibbsite-like dioctahedral arrangement. A combination of tetrahedral and di- or trioctahedral sheets bound by shared oxygen atoms forms aluminosilicate layers that comprise the basic structural units of phyllosilicates (Fig. 2). Sheet arrangement within the aluminosilicate layers varies between clay mineral types, resulting in variable physical and chemical properties that differentiate the clay mineral classes.

ISOMORPHOUS SUBSTITUTION

The structural arrangement of the elements described above forms the template for the SiO_4 clay minerals. However, the composition varies frequently due to substitution of ions

Table 1 Common elements in earth’s crust and ionic radius.

Element	Crustal average (g kg ⁻¹)	Ionic radius (nm)	Volume (%)
O ²⁻	466.0	0.140	89.84
Si ⁴⁺	277.2	0.039	2.37
Al ³⁺	81.3	0.051	1.24
Fe ²⁺	50.0	0.074	0.79
Mg ²⁺	20.9	0.066	0.60
Ca ²⁺	36.3	0.099	1.39
Na ²⁺	28.3	0.097	1.84
K ⁺	25.9	0.133	1.84

Source: Adapted from Klein & Hurlbut.^[1]

within the mineral structure. Weathering allows for the substitution of Si⁴⁺, Al³⁺, and Mg²⁺ with cations with comparable ionic radii in their respective tetrahedral and octahedral sheets (Table 1). Consequently, Si⁴⁺ may be replaced by Al³⁺ in the center of the tetrahedron without changing the basic structure of the crystal. Moreover, cations such as Fe^{3+/2+} and Zn²⁺ (ionic radius = 0.074 nm) may replace Al³⁺ and Mg²⁺ in the octahedra. The process of replacing one structural cation for another of similar size is referred to as isomorphous substitution. This replacement represents the primary source of both negative and positive charges in clay minerals. For example, the substitution of one Al³⁺ for a Si⁴⁺ in the tetrahedron results in a gain of one negative charge.

Alternatively, replacement of a lower valence cation by one with a higher valence (Fe²⁺ by Fe³⁺) results in a gain of one positive charge. Some clay minerals exhibit substitutions that result in both positive and negative charges. A balance of electron loss and gain within the structure determines the net charge of the mineral. In most soils, however,

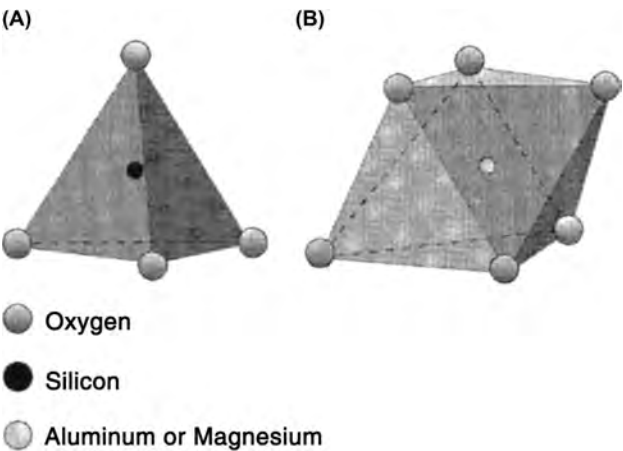


Fig. 1 The basic structural components of clay minerals: (A) a single four-sided tetrahedron and (B) a single eight-sided octahedron.

Source: From Grim.^[3]

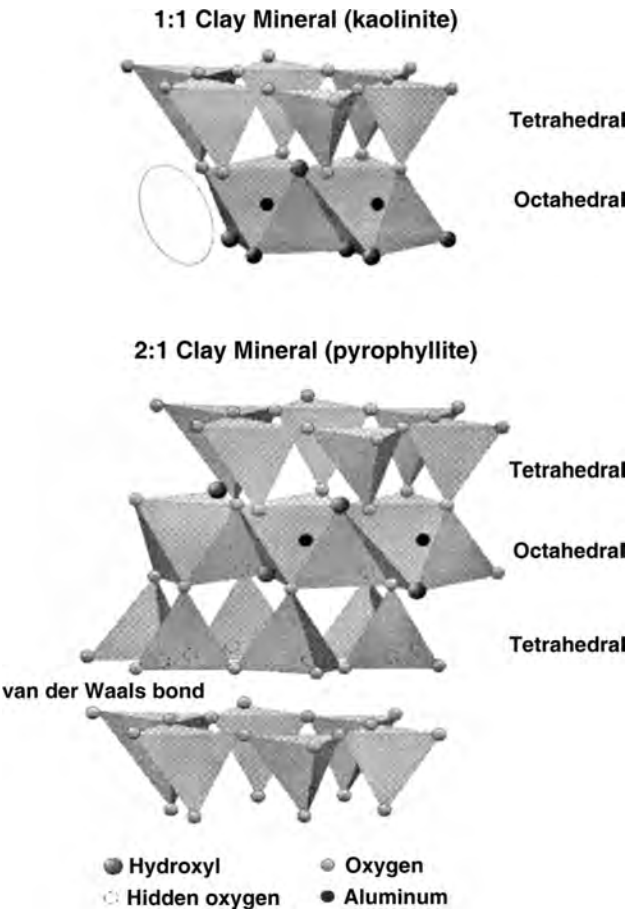


Fig. 2 Diagrammatic sketch of a 1:1 clay mineral consisting of one tetrahedral sheet bonded to an octahedral sheet (kaolinite) and a 2:1 clay mineral consisting of an octahedral sheet bound between two tetrahedral sheets (pyrophyllite). Source: From Grim.^[3]

substitutions that result in net negative charge exceed those producing a positive charge.

CLAY MINERAL CLASSIFICATION

Clay minerals are generally classified into three layer types based upon the number and arrangement of tetrahedral and octahedral sheets in their basic structure. These are further separated into five groups that differ with respect to their net charge (Table 2).

1:1 Clay Minerals

The 1:1 layer minerals contain one tetrahedral and one octahedral sheet in their basic structural unit (Fig. 2.) This two-sheet mineral type is represented by the kaolin group, with the general formula Al₂Si₂O₅(OH)₄. Kaolinite, the most common mineral in this group, is dioctahedral, exhibiting Al³⁺ octahedral and Si⁴⁺ tetrahedral coordination. The sheets are held together by van der Waals bonds

Table 2 Properties of clay mineral groups.

Group	Layer type	Net negative charge (cmol kg ⁻¹)	Surface area (m ² g ⁻¹)	Basal spacing (nm)
Kaolinite	1:1	2–5	10–30	0.7
Fine-grained mica	2:1	15–40	70–100	1.0
Smectite	2:1	80–120	600–800	1.0–2.0
Vermiculite	2:1	100–180	550–700	1.0–1.5
Chlorite	2:1:1	15–40	70–100	1.4

Source: Adapted from Brady.^[4] ©1990 Macmillan Publishing Co.

between the basal oxygens of the tetrahedral sheet and the hydroxyls of the octahedral sheet. Layers are held together tightly by hydrogen bonding, which restricts expansion and limits the reactive area to external surfaces. Isomorphic substitution for Si⁴⁺ and Al³⁺ in this mineral is negligible. As such, soils dominated by 1:1 minerals exhibit a low capacity for adsorbing cations and have low fertility. The serpentine group, with the general formula Mg₃Si₂O₅(OH)₄, represents the trioctahedral version of the 1:1 layer minerals.

2:1 Clay Minerals

The joining of two tetrahedral sheets (one from each side) to one octahedral sheet produces a three-sheet mineral type, which is called 2:1 and is represented by the mica, smectite, and vermiculite groups. Talc [Mg₃Si₄O₁₀(OH)₂] and pyrophyllite [Al₂Si₄O₁₀(OH)₂] are typical representatives of electrically neutral 2:1 type minerals in which adjacent layers are joined to each other by van der Waals bonds (Fig. 2). Although these two minerals are found infrequently in soils,^[2] their structure serves as a model for discussing transitions leading to the formation of other more common 2:1 clay minerals.

The true micas have a similar structure to that of talc and pyrophyllite, except that substitution of Al³⁺ for Si⁴⁺ in every fourth tetrahedral site results in an excess of one negative charge per formula unit. The negative charge is satisfied by monovalent cations, primarily K⁺, that reside on interlayer sites between the 2:1 layers. The interlayer cation forms a strong bond between adjoining tetrahedral sheets, which limits expansion of the mineral. The mica group is subdivided into tri- and dioctahedral minerals according to cation substitutions in the octahedral sheet and within the interlayer. The trioctahedral group of micas contains interlayer K⁺ cations and is represented by phlogopite [KMg₃(AlSi₃O₁₀)(OH)₂], with Mg²⁺ occupying the octahedral sites, and biotite, which contains both Fe²⁺ and Mg²⁺ in the octahedron. Muscovite [KAl₂(AlSi₃O₁₀)(OH)₂] is a dioctahedral mica containing Al³⁺ in the octahedral sheet and K⁺ in the interlayer, while paragonite exhibits a similar

dioctahedral coordination with interlayer K⁺ and Na⁺ cations. In most soils, micas are generally inherited from the parent material and occur in a relatively unweathered state in the sand and silt fractions. Mica in the clay fraction usually exhibits poorer crystallinity, lower K⁺ content, higher water content, and possible substitutions of Fe²⁺ and Fe³⁺ in the octahedral sheets and Ca²⁺ in the interlayer. Manganese, vanadium, lithium, chromium, titanium, and several other cations are also known to occur in varying amounts in these fine-grained or clay-sized micas.^[1] Illite and glauconite (dioctahedral iron illites) are commonly associated with the clay-sized micas; however, the structures of these minerals are poorly defined and likely to be representative of a mixture of weathered micas. Expandable 2:1 clay minerals exhibit a similar layer structure to that described for mica but vary widely in layer charge and interlayer spacing due to the presence of weakly bound cations, water, or polar organic molecules in the interlayer region.

Smectites generally refer to a group of expandable dioctahedral 2:1 minerals with a charge of 0.2–0.6 per formula unit. Montmorillonite, the most common member of this group, derives its charge from the octahedral substitution of Mg²⁺ for Al³⁺. Beidellite and nontronite, which are less abundant in soils, derive much of their charge from tetrahedral substitutions. Nontronite is distinguished from beidellite by the presence of iron in the octahedral sheet. The 2:1 layers in smectites are held together by van der Waals bonds and weak cation-to-oxygen linkages. The presence of exchangeable cations located between water molecules in the interlayer allows for expansion of the crystal lattice as the mineral hydrates. When the mineral is saturated with water, the basal spacing between layers can approach 2 nm, while under dry conditions, the basal spacing may be reduced to less than 1 nm (Fig. 3). This expansion and contraction trait found in smectites, often referred to as shrink–swell potential, is problematic to engineers and farmers alike due to the propensity for crack formation and general instability of the soil surface.

The weathering of mica, via replacement of interlayer K⁺ with hydrated exchangeable cations, results in the formation of vermiculite. In soils, vermiculite exists as an Al³⁺-dominated dioctahedral and, to a lesser extent, Mg²⁺-dominated trioctahedral mineral. A charge of 0.6–0.9 per formula unit is derived in these minerals by tetrahedral substitution of Al³⁺ for Si⁴⁺. Consequently, vermiculites exhibit a high cation exchange capacity, particularly for weakly hydrated cations such as K⁺, NH₄⁺, and Cs⁺.^[5] Because of the tetrahedral charge origin, water molecules and exchangeable cations—primarily Mg²⁺ and Ca²⁺—are strongly adsorbed within the interlayer space of vermiculites. Unlike in smectites, the strong bonding of the interlayer cations holds the 2:1 layers together in vermiculites, thus limiting expansion of the basal spacing to 1.5 nm (Fig. 3).

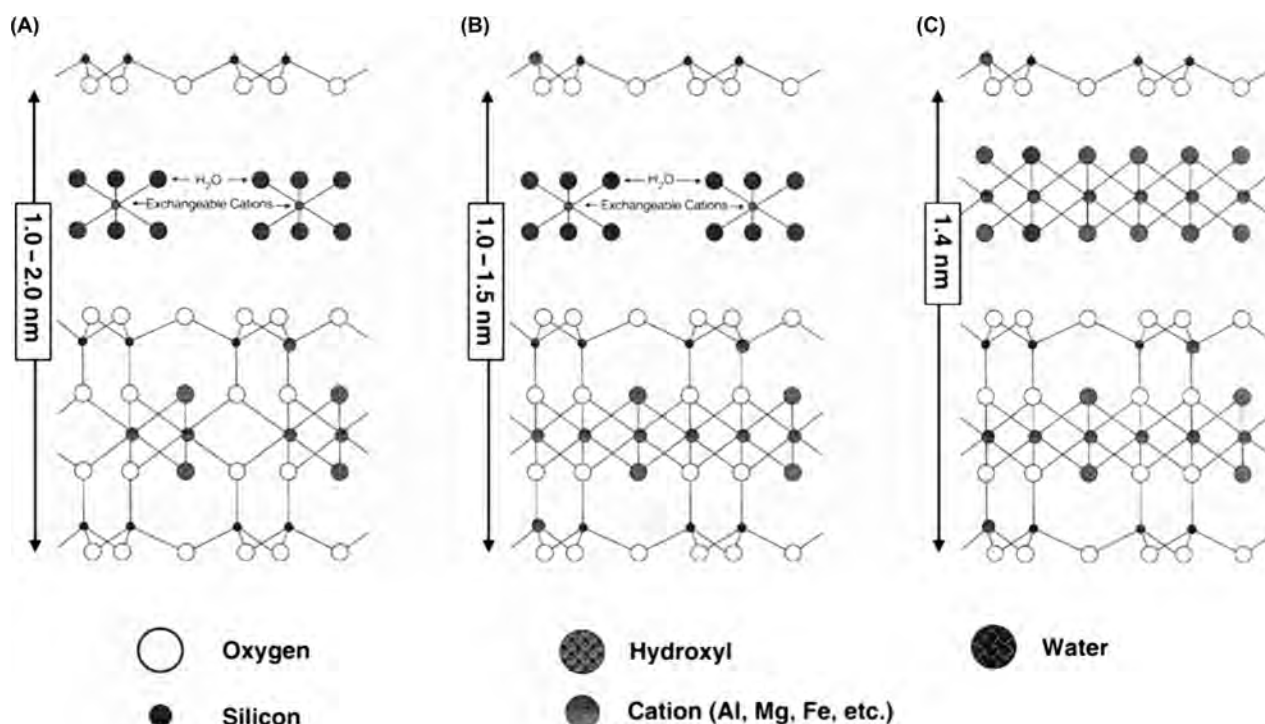


Fig. 3 Schematic representation of (A) expandable dioctahedral montmorillonite; (B) expandable trioctahedral vermiculite; and (C) non-expandable trioctahedral chlorite.

2:1:1 Clay Minerals

Chlorites are a group of minerals that exhibit a basic 2:1 layer structure similar to that described for talc or pyrophyllite, but with an interlayer brucite- or gibbsite-like sheet, which forms a 2:1:1 structural arrangement. Isomorphic substitutions within the interlayer hydroxide sheet create a net positive charge that balances the negative charge arising from the 2:1 layers. A typical formula for the interlayer sheet is $(\text{MgFeAl})(\text{OH})_6^+$; however, a variety of cation species may exist in these brucite or gibbsite-like islands that contribute to a large number of mineral species within this group. There is no water adsorption within the interlayer space; thus, chlorites are considered non-expansive minerals (Fig. 3).

Hydroxy-Al Interlayered Vermiculites and Smectites

Chlorites exist in soils as primary minerals that weather to form vermiculite and smectite. In contrast, hydroxy-Al interlayered vermiculites or smectites are considered to be exclusively secondary minerals forming as intermediate mineral weathering products or from deposition of hydroxy-Al polymeric components within the interlayer space of expanding minerals.^[6] These hydroxy-Al polycations—with the general formula $\text{Al}_6(\text{OH})_{15}^{3+}$ —balance a portion of the charge but they are not exchangeable.

Because the level of hydroxy-Al occupancy within the interlayer space is variable, the CEC of the expandable 2:1 clays is reduced as a function of the quantity and valence of the hydroxy-Al polymer residing within the interlayer.

Mixed Minerals

In the soil environment, clay minerals rarely occur as homogeneous mixtures of single groups, types, or mineral phases. Instead, they comprise complex assemblages of primary minerals and weathering intermediates of multiple structural and compositional combinations. The potential also exists for a discrete mineral grain to be composed of more than one clay type or contain components that are intermediate between two known minerals. These minerals are referred to as mixed-layer or interstratified minerals. Examples of such mixed-layer mineral sequences include mica-vermiculite, mica-smectite, chlorite-vermiculite, and kaolinite-smectite, among others. The sequence of differing layers making up the mixed mineral occurs at both regular and random intervals. Consequently, efforts to fully characterize clay mineralogy within certain soils may be hindered considerably.

Other Clay Minerals

In addition to the phyllosilicates described, certain oxides, hydroxides, and hydroxy-oxides (sesquioxides) of Si, Al,

Fe, as well as some poorly crystalline aluminosilicates may also be found in small quantities in the soil clay fraction.^[7,8] Typical representatives of Si-oxide minerals that are often found in the clay fraction are quartz—due to its high resistance to weathering—and opal—a poorly crystalline polymorph of quartz, which precipitates from Si-supersaturated solutions or is of volcanic or biogenic origin. Gibbsite ($\text{Al}(\text{OH})_3$) is the main Al-hydroxide representative in clay fractions of highly weathered soils, while goethite (FeOOH) is the most common clay fraction Fe mineral. The poorly crystalline aluminosilicate clay minerals allophane and imogolite are typically found in clay fractions of soils derived from volcanic materials. Extensive discussions about these minerals and their properties are provided in other soil mineral sections of this encyclopedia.

Clay Mineral Identification and Quantification

The basic methods used for clay mineral identification and quantification involve X-ray diffraction (XRD) and thermal analyses.^[9] Electron microscopy is also employed for complementary mineral characterizations. In XRD analysis, the basal spacing (or d-spacing) between equivalent crystal planes is measured from the specific angle at which they diffract X rays of known wavelength according to Bragg's Law: ($d/n = \lambda / 2 \sin \theta$), where d = basal spacing, n = order of diffraction, usually set to 1, λ = X-ray wavelength, and θ = diffraction angle. Therefore, diffraction peaks at specific angles corresponding to fixed d-spacing characteristic of specific minerals are used for mineral identification, while diffraction peak intensities are utilized for quantitative estimations. The identification of minerals with overlapping d-spacing regions (smectite, vermiculite, and chlorite) is based on specific diffraction peak shifts, following solvation with specific ions and/or heating treatments.

Thermal methods are based on characteristic temperature regions in which specific minerals lose their structural OH water (thermogravimetric analysis) and the enthalpy associated with the dehydroxylation reaction (differential scanning calorimetry). Mineral quantities are estimated from comparisons of the OH- or enthalpy fractions that have been measured and standard quantities of representative pure minerals.

REFERENCES

1. Klein, C.; Hurlbut, C. Systematic mineralogy part IV: Silicates. In *Manual of Mineralogy*, 20th Ed.; John Wiley & Sons: New York, 1985; 366–467.
2. Schulze, D.G. An introduction to soil mineralogy. In *Minerals in Soil Environments*; Dickson, J.B., Weed, S.B., Eds.; Soil Science Society of America: Madison, 1989; 1–34.
3. Grim, R. *Clay Mineralogy*, 2nd Ed.; McGraw-Hill Book Co.: New York, 1968; 596 pp.
4. Brady, N. Soil colloids: Their nature and practical significance. In *The Nature and Properties of Soils*, 10th Ed.; Macmillan Publishing Co.: New York, 1990; 177–212.
5. Douglas, L. Vermiculites. In *Minerals in Soil Environments*; Dixon, J., Weed, S., Eds.; Soil Science Society of America: Madison, 1989; 635–676.
6. Barnhisel, R.I.; Bertsch, P.M. Chlorites and hydroxy-interlayered vermiculite and smectite. In *Minerals in Soil Environments*, 2nd Ed.; Dixon, J.B., Weed, S.B., Eds.; Soil Science Society of America: Madison, 1989; 729–788.
7. Amonette, J.E.; Zelazny, L.W., Eds. *Handbook of Soil Science*; CRC Press: Boca Raton, 2000.
8. Amonette, J.E.; Zelazny, L.W., Eds. *Origin and Mineralogy of Clays: Clays and the Environment*; Springer-Verlag: New York, 1995.
9. Amonette, J.E.; Zelazny, L.W., Eds. *Quantitative Methods in Soil Mineralogy*; Soil Science Society of America Miscellaneous Publication: Madison, 1994.

Clay Minerals: Charge Properties

Christopher John Matocha

Agronomy Department, University of Kentucky, Lexington, Kentucky, U.S.A.

Abstract

Surfaces of soil particles are classified as having either permanent charge or pH-dependent charge. A permanent charge is derived from isomorphous substitutions in the mineral structure and is indifferent to the conditions surrounding the mineral. Isomorphous substitution involves the replacement of constituent metal ions of the lattice by cations of lower charge, generating a net negative charge on the mineral, which is compensated by the positive charge of adsorbed (exchangeable) cations. The substitution can occur during or after crystal formation. Surface functional groups that may be exposed on the periphery of minerals protonate and deprotonate depending on the solution pH, generating pH-dependent surface charge. Most soils carry a net negative charge because of negative charges on phyllosilicate minerals, but some weathered soils can develop a net positive charge at sufficiently low pH due to the presence of hydrous iron and aluminum oxides. The net negative charge on soil minerals produces a positive electrostatic attraction for cations in soil solution that are retained on the surface. The total quantity of cations retained in this way is called the cation exchange capacity because they are easily exchanged with other cations. The focus of this entry is the surface charge properties of 1:1 and 2:1 phyllosilicate minerals because of their abundance and significant influence on agricultural and environmental processes.

ORIGINS OF CHARGE

Phyllosilicate minerals consist of aluminosilicate tetrahedral and octahedral sheets stacked one above the other and differentiated by the number and combination of sheets and the cation substitution in each sheet.^[1,2] In 1:1 phyllosilicate minerals, such as kaolinite, the full-cell chemical formula is $\text{Si}_4\text{Al}_4\text{O}_{10}(\text{OH})_8$ where one silica tetrahedral sheet is combined with one aluminum octahedral sheet and individual layers stack by hydrogen bonding (O–H–O) in the c-dimension (perpendicular to the plane of the sheets) with a combined thickness of 0.7 nm. Generally, no isomorphous substitution occurs in either sheet; thus, kaolinite has no significant permanent negative structural charge on basal sites and little cation exchange capacity (CEC).^[1] Where crystal growth stops and no more unit cells are added, broken bonds exist on the edges of clay minerals and a surface is obtained (Fig. 1). These “edge” sites exhibit pH-dependent charge that results from protonation/deprotonation of aluminol ($>\text{AlOH}$) and silanol ($>\text{SiOH}$), functional groups that are exposed on the periphery or edges of the mineral.^[4] The acidity or dissociation constants ($\text{p}K_{\text{a}}$ s) for the aluminol edge sites are $\text{p}K_{\text{a}1} = 6.3$ and $\text{p}K_{\text{a}2} = 8.7$, meaning that at $\text{pH} < 6$, these edge sites are positively charged.^[5] This explains the early findings of Schofield and Samson,^[6] who documented Cl^- sorption at low pH. Kaolinite has a significant concentration of edge sites because it tends to stack in the c-dimension.^[4]

The 2:1 phyllosilicate minerals consist of two tetrahedral sheets that sandwich one octahedral sheet (Fig. 1). The ideal full-cell chemical formula for 2:1 clay would be $\text{Si}_8^{\text{IV}}\text{Al}_4^{\text{VI}}\text{O}_{20}(\text{OH})_4$, where the roman numerals refer to the tetrahedral (IV) and octahedral (VI) sheets, respectively.^[7] This formula is represented by the mineral pyrophyllite, which is neutral because no cation substitution occurs in either sheet. More common 2:1 phyllosilicate minerals include the smectite and vermiculite groups. The number of cation positions occupied within the octahedral sheet determines whether a clay mineral is di- or trioctahedral. If Al^{3+} is present in the octahedral sheet, then the mineral is dioctahedral, and if Mg^{2+} is present, then the mineral is trioctahedral. Within the smectite group of minerals, dioctahedral montmorillonites have the majority of structural negative charges. These are derived primarily from octahedral site substitutions of Fe^{2+} or Mg^{2+} for Al^{3+} with a structural formula of $\text{Si}_8^{\text{IV}}\text{M}_{0.66}\text{H}_2\text{OAl}_{3.34}^{\text{VI}}(\text{Fe}^{2+}, \text{Mg}^{2+})_{0.66}\text{O}_{20}(\text{OH})_4$, where M refers to the exchangeable cation balancing the permanent charge produced by the substitution of Fe^{2+} or Mg^{2+} for Al^{3+} .^[3] The octahedral layer charge for montmorillonite, or the number of moles of excess electron charge per formula unit,^[1] would be -0.66 , balanced by $\text{M}_{0.66}$ exchangeable cations adsorbed near the basal plane of the tetrahedral sheet. In dioctahedral beidellites, tetrahedral substitutions of Al^{3+} for Si^{4+} exceed substitutions in the octahedral sheet (Fig. 1). Nontronite is similar to beidellite except it has redox-active Fe^{3+} in octahedral coordination (Fig. 1). The vermiculite group differs from

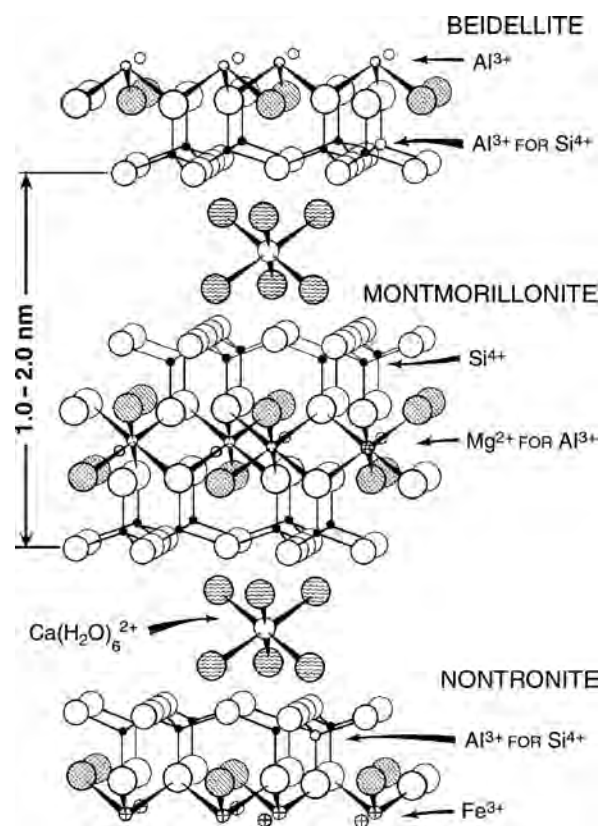


Fig. 1 Representative 2:1 phyllosilicate minerals indicating charge sites and different locations of substitution.

Source: From Borchardt.^[3]

the smectite group because the primary source of negative charge arises from a large degree of Al^{3+} for Si^{4+} substitution in the tetrahedral layer, giving rise to a layer charge greater than that in smectites, ranging from 1.2 to 1.8.

The basal plane of a tetrahedral sheet composed of oxygen atoms is called the siloxane surface, and its reactivity depends on the location and type of isomorphous substitution.^[8] In a tetrahedrally substituted clay such as beidellite, the excess negative charge is localized and functions as a strong electron donor because of the lone pair, non-bonding electrons on the siloxane oxygens.^[8,9] Siloxane oxygens not adjacent to the site of substitution, as is the case for octahedrally substituted montmorillonite, are weaker electron donors because the negative charge is distributed over at least 10 surface oxygens.^[9]

One characteristic that the smectite and vermiculite groups have in common with kaolinite is the presence of edge sites, although in lower concentration because stacking occurs predominantly in the horizontal dimensions (a, b) rather than the vertical c-dimension. A distinguishing feature of the smectite and vermiculite groups from kaolinite is the presence of an interlayer, occupied by exchangeable cations such as Ca^{2+} and water (Fig. 1). Thus, the 2:1 clay mineral particles have external planar

sites, internal sites, and edge sites, while kaolinite contains mainly external edge sites. The active interlayer and the higher level of isomorphous substitution give rise to greater CEC values for smectite and vermiculite minerals ($\text{CEC} \sim 0.80\text{--}1.50 \text{ mEq g}^{-1}$) when compared to kaolinite ($\text{CEC} \sim 0.02\text{--}0.05 \text{ mEq g}^{-1}$).

SURFACE CHARGE DENSITY OF CLAY MINERALS

It is important to note at this point that the clay size fraction is, by definition, any mineral with a particle size $<2 \mu\text{m}$. Thus, the clay size fraction has a large amount of surface area per unit mass when compared to the silt and sand fractions. This is clearly shown when one derives the specific surface area, or the area per unit mass. For uniform, spherical particles, this ratio equals

$$S = 3/\rho R$$

where S is the specific surface area ($\text{cm}^2 \text{g}^{-1}$) having a radius R (cm) and a particle density ρ (g cm^{-3}). For a given particle size, an equal mass of smectite or vermiculite would have a greater specific surface area when compared to kaolinite because of the presence of the interlayer. A typical value of S for montmorillonite would be $700 \text{ m}^2 \text{g}^{-1}$, while for kaolinite S is considerably less ($\sim 10 \text{ m}^2 \text{g}^{-1}$). The surface density of charge, obtained by normalizing the CEC to S , is similar irrespective of clay mineral type because S tends to scale with CEC.^[10] Higher S values usually indicate greater potential for reactivity.

AGRICULTURAL AND ENVIRONMENTAL IMPLICATIONS OF SURFACE CHARGE

High-Activity and Low-Activity Clays

Small amounts of smectite minerals can greatly influence overall soil charge properties due to their greater specific surface area. In the context of charge, clay activity classes have been defined as the ratio of CEC/clay content.^[11] High-activity clay minerals, such as smectite or vermiculite, have ratios >0.7 , while low-activity clays like kaolinite have ratios <0.3 . These indices are useful in estimating mineralogy when no specific mineralogy data are available.^[11]

Adsorption of Plant Nutrients and Contaminants

As mentioned earlier, the cations balancing the net negative charge generated from isomorphous substitutions in order to maintain electroneutrality are exchangeable. They maintain their inner hydration shell, thus forming the so-called outer-sphere complexes.^[8] An example of an outer-sphere

complex would be exchangeable Ca^{2+} in the interlayer space of 2:1 minerals (Fig. 1). Exchangeable cations are agronomically important because they are available to plants. In contrast, cations that lose H_2O from their inner hydration shell to form a direct bond with a mineral surface functional group are inner-sphere surface complexes.^[8] The type of surface complex formed influences the fate of plant nutrients and contaminants in the soil environment.

Tetrahedrally substituted clay minerals can form stronger adsorption complexes with cations than octahedrally substituted clay minerals.^[1] Cations with the largest ionic radius and lowest hydration energy are preferentially adsorbed on permanent charge sites.^[10] Vermiculites and beidellites, with more tetrahedral charge sites, can fix K^+ more readily by forming inner-sphere bonds with siloxane oxygens. The fixation of K^+ makes it unavailable to plants because formation of these bonds causes the collapse of adjacent silicate layers.^[6] The proximity of tetrahedral charge to the interlayer space of 2:1 phyllosilicates explains the greater K^+ -fixing ability of tetrahedrally charged clays.^[12]

Varying pH and background electrolyte concentration is an approach used to assess the contribution of pH-dependent charge edge sites and permanent charge sites in controlling metal contaminant solubility on montmorillonite. Spectroscopic investigations confirmed that Pb^{2+} and Cu^{2+} adsorbed as inner-sphere surface complexes on hydroxyl edge sites in montmorillonite, particularly when pH and ionic strength increased, and as outer-sphere surface complexes on permanent charge sites at low pH and less ionic strength.^[13,14] The inner-sphere Cu^{2+} complexes coordinated to the hydroxyl edge sites were resistant to desorption, whereas outer-sphere Cu^{2+} surface complexes on permanent charge sites were exchangeable and potentially more mobile. Therefore, the practical benefit achieved in considering the difference between outer-sphere and inner-sphere surface complexation would be in predicting the fate of plant nutrients and contaminants in the soil environment.

Flocculation Processes

Clay particle–particle interactions regulate dispersion and flocculation processes and are important to soil quality. Flocculation, by definition, is the formation of particle aggregates from the association of individual particles in water that settle by gravity and promote aggregation and optimal soil structure.^[10] Positively charged clay mineral edges contribute to flocculation, because of the electrostatic attraction between positive edges and negative planar surface faces of adjacent minerals. Studies of the Schofield and Samson^[5] showed that kaolinite can be deflocculated by the addition of montmorillonite particles. Other factors influencing the degree of flocculation include charge of exchangeable cation, ionic strength of the soil solution, and

cation size. High-charge exchangeable cations and high ionic strength promote flocculation of clay mineral suspensions dominated by permanent charge minerals.^[10] High Ca^{2+} status and near-neutral pH are associated with stable aggregation in less weathered soils carrying a net negative charge. There is a direct relationship between cation size and ability to form inner-sphere surface complexes with clay mineral sites. For instance, high concentrations of exchangeable Na^+ promote dispersion and poor structure because it is weakly adsorbed, likely bound as an outer-sphere surface complex with octahedral charge sites on smectites.^[11] The greater hydration of Na^+ , a function of its small ion size, explains the weaker retention of Na^+ when compared to larger cations like Cs^+ .^[10]

Clay Swelling

Water penetrates the interlayer of 2:1 phyllosilicates and forces them apart, causing the clay to swell, which is an important natural soil phenomenon. High-activity clay minerals, such as montmorillonite, influence the engineering behavior of soils because of their ability to take up and adsorb large amounts of water, when compared to low-activity clay minerals like kaolinite.^[11] Water adsorption is an exothermic process, thus “heat of wetting” values are typically greater for montmorillonite than for kaolinite. In addition, the type of exchangeable cation and location and distribution of charge strongly influence hydration characteristics of the clay mineral. The amount of water adsorbed and internal surface hydrated on montmorillonite is positively correlated with hydration energy of the exchangeable cation.^[15] Sodium (Na) montmorillonite exhibited greater adsorption–desorption hysteresis of water when compared to calcium (Ca) montmorillonite and magnesium (Mg) montmorillonite, which was ascribed to differences in hydration energy for the cations.^[16] Well-hydrated interlayer Ca and Mg serve to prop open the interlayer when compared to Na montmorillonite. The ability of montmorillonite to swell in water more than vermiculite, despite the greater layer charge in the latter mineral, is explained by the location of the charge. The localized charge in tetrahedrally substituted vermiculite apparently causes the exchangeable cations to pull the layers together and limit the expansion.^[10]

REFERENCES

1. Sposito, G.; Skipper, N.T.; Sutton, R.; Park, S.H.; Soper, A.K.; Greathouse, J.A. Surface geochemistry of the clay minerals. *Proc. Natl. Acad. Sci.* **1999**, *96*, 3358–3364.
2. Swartzen-Allen, S.L.; Matijevic, E. Surface and colloid chemistry of clays. *Chem. Rev.* **1974**, *74*, 385–400.
3. Borchardt, G. Smectites. In *Minerals in Soil Environments*; Dixon, J.B., Weeds, S.B., Eds.; Soil Science Society of America: Madison, 1989; 675–728.

4. White, G.N.; Zelazny, L.W. Analysis and implications of the edge structure of dioctahedral phyllosilicates. *Clay. Clay Miner.* **1988**, *36*, 141–146.
5. Wieland, E.; Stumm, W. Dissolution kinetics of kaolinite in acidic aqueous solutions at 25 degrees C. *Geochim. Cosmochim. Acta* **1992**, *56*, 3339–3355.
6. Schofield, R.D.; Samson, H.R. The deflocculation of kaolinite suspensions and the accompanying change over from positive to negative chloride adsorption. *Clay Miner. Bull.* **1953**, *2*, 45–51.
7. Sparks, D.L. *Environmental Soil Chemistry*; Academic Press: San Diego, 1995; 267 pp.
8. Sposito, G. *The Surface Chemistry of Soils*; Oxford University Press: New York, 1984.
9. Farmer, V.C.; Russell, J.D. Interlayer complexes in layer silicates. The structure of water in lamellar ionic solutions. Water on particle surfaces. *T. Faraday Soc.* **1971**, *67*, 2737–2749.
10. McBride, M.B. *Environmental Chemistry of Soils*, 1st Ed.; Oxford University Press: New York, 1994; 406 pp.
11. Olson, C.G.; Thompson, M.L.; Wilson, M.A. Phyllosilicates. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; F77–F123.
12. Bouabid, R.; Badraoui, M.; Bloom, P.R. Potassium fixation and charge characteristics of soil clays. *Soil Sci. Soc. Am. J.* **1991**, *55*, 1493–1498.
13. Strawn, D.G.; Sparks, D.L. The use of XAFS to distinguish between inner- and outer-sphere lead adsorption complexes on montmorillonite. *J. Colloid Interf. Sci.* **1999**, *216*, 257–269.
14. Morton, J.D.; Semrau, J.D.; Hayes, K.F. An X-Ray absorption spectrometry study of the structure and reversibility of copper adsorbed to montmorillonite clay. *Geochim. Cosmochim. Acta* **2001**, *65*, 2709–2722.
15. Sposito, G.; Prost, R. Structure of water adsorbed on smectites. *Chem. Rev.* **1982**, *82*, 553–573.
16. Xu, W.; Johnson, C.T.; Parker, P.; Agnew, S.F. Infrared study of water sorption on Na-, Li-, Ca-, and Mg-exchanged (SWy-1 and SAZ-1) montmorillonite. *Clay. Clay Miner.* **2000**, *48*, 120–131.

Clay Minerals: Weathering and Alteration of

Anastasios D. Karathanasis

Department of Agronomy, University of Kentucky, Lexington, Kentucky, U.S.A.

Abstract

Mineral weathering encompasses the physical and chemical breakdown of minerals that are unstable under the conditions prevailing at the earth's surface. It is driven by physical, chemical, and biological gradients, which cause gradual structural and compositional mineral alterations that allow them to attain more stable states. Most weathering reactions lead into complete dissolution of the original mineral structure to soluble components, which resynthesize to form a new mineral with different composition. A significant number of mineral alterations, however, result in modified residual mineral structures without going through a complete dissolution phase. Water (H_2O) is the main mineral weathering agent. Under its dominant influence, physical forces (wetting and drying, freezing and thawing, wind and H_2O erosion, thermal expansion, and salt crystallization) combine with chemical forces (hydration–dehydration, acidification, oxidation–reduction, and chelation) to produce new mineral phases more conforming to the existing environment.

MINERAL RESISTANCE TO WEATHERING

Weathering affects both primary and secondary minerals. Primary minerals having formed under high temperature and pressure conditions are less stable under ambient conditions and tend to transform into secondary minerals. Their resistance to weathering is inversely related to the temperature of their crystallization from cooling magmas. The higher the temperature of formation, the lower their stability at the surface of the earth. Based on this principle, Goldich^[1] proposed a stability series of common primary minerals (Table 1), which was experimentally confirmed with long-term mineral dissolution comparisons.^[2] Mineral stability is also a function of structural bond strength with which atoms and ions are held together in the crystal lattice of each mineral. Most primary minerals are silicates or aluminosilicates, with strong silicon–oxygen (Si–O) bonds, moderately strong aluminum (Al)–O bonds, and weaker metal–O bonds in their crystal lattice. Their susceptibility to weathering depends on the number of weak bonds and the strength of the Si–O bond, which is a function of the Si–O ratio in the structure. The lower the ratio (increased sharing of O between adjacent silica tetrahedrons) the stronger the overall mineral structure. Quartz, e.g., having the maximum sharing of tetrahedral O, is one of the most resistant minerals. Olivine is one of the most weatherable minerals because there is no sharing of O in its structure. The stability of other minerals is controlled by the number of shared O and the type of cation bonds satisfying isomorphic substitutions within the structure. In addition, the presence of iron (Fe) and manganese (Mn) in the lattice usually induces higher weatherability because oxidation–reduction transitions create charge imbalances within the structure. Keller^[3] and Birkeland^[4] used the following mineral

stability sequence to rank the silicate mineral groups according to bond strength considerations: nesosilicates ($\text{Si–O} = 1/4$) < single-chain inosilicates ($\text{Si–O} = 1/3$) < double-chain inosilicates ($\text{Si–O} = 4/11$) < phyllosilicates ($\text{Si–O} = 2/5$) < tectosilicates ($\text{Si–O} = 1/2$). Even though some discrepancies exist between the crystallization temperature and the Si–O bond strength sequences, these two schemes are used extensively for making mineral weathering stability interpretations.

Weathering continues after the transformation of primary to secondary minerals because some of them may be stable only within a certain range of solution ionic gradients. As the concentration of soluble Si, Al, and other cations decreases through leaching processes, initially stable secondary minerals may weather further to more stable states. Therefore, secondary minerals may be the products of alteration of primary or secondary minerals subjected to multiple weathering cycles. Jackson and Sherman^[5] proposed a mineral weathering sequence based on the relative degree of soil development with continuous leaching and the types of minerals commonly found in the soil clay fraction (Table 2). This weathering scheme or stages of it may be applicable to many soils, but it does not specify the physicochemical gradients that could trigger a certain set of transformations, or whether a given soil under specific environmental conditions will go through every one of the predicted stages. Nevertheless, it has been an effective tool in describing soil development stages and processes mainly because the clay fraction minerals play a greater role and are more actively involved than their sand and silt size counterparts in the changes occurring in the soil over time. Furthermore, the clay minerals due to their smaller size, high surface area, and chemical activity are in closest agreement with the surrounding soil solution composition, which

Table 1 Mineral stability table.^a

Mafic minerals	Felspathic minerals
Olivine	Calcic plagioclase
Augite	Calcic-alkalic plagioclase
Hornblende	Alkali-calcic plagioclase
Biotite	Alkalic plagioclase
K feldspar	
Muscovite	
Quartz	

^aStability increasing from top to bottom.Source: From Goldich.^[1]

is the driving force for mineralogical changes. The mineral groups of the clay size fraction listed in Table 2 indicate progressively increasing stages of soil development. A certain mineral suite may not include all the minerals of a specific weathering stage or index or may not necessarily dominate the clay fraction in order to be a fairly reliable indicator of the overall weathering or maturity status of a soil. A given soil clay fraction often contains a mixture of minerals covering several adjacent weathering stages. A quantitative weathering mean representing the weighted average of mineral composition by soil horizon depth is used to estimate the overall weathering stage of soil development.^[6] In most soils, this mean generally shifts from lower to higher numerical weathering stages as the particle size decreases, as we move from the parent material to the soil surface, and as the soil advances in maturity. With continuous weathering and sufficient time, the effect of distinct climatic and geographic settings should be more evident in the distribution of soil clay minerals. Indeed,

Table 2 Weathering sequence for clay size minerals.^a

1	Gypsum (also halite, Na nitrate, ammonium chloride, etc.)
2	Calcite (also dolomite, aragonite, apatite, etc.)
3	Olivine-hornblende (also pyroxenes, diopside, etc.)
4	Biotite (also glauconite, magnesium chlorite, antigorite, nontronite, etc.)
5	Albite (also anorthite, stilbite, microcline, orthoclase, etc.)
6	Quartz (also cristobalite, etc.)
7	Muscovite (also illite, etc.)
8	Vermiculite (also interstratified 2:1 layer silicates)
9	Montmorillonite (also beidellite, saponite, etc.)
10	Kaolinite (also halloysite, hydroxyinterlayered vermiculite and smectite, etc.)
11	Gibbsite (also boehmite, allophane, etc.)
12	Hematite (also goethite, limonite, etc.)
13	Anatase (also zircon, rutile, ilmenite, leucoxene, corundum, etc.)

^aStability increasing from 1 to 13.Source: From Jackson & Sherman.^[5]

these relationships have been the foundation for developing various soil classification schemes, grouping soils with similar behavior, including Soil Taxonomy.^[7]

MINERAL WEATHERING PATHWAYS

Regardless of the mineralogical and structural composition of the minerals undergoing weathering, the degree of alteration and mineral weathering products depend on the intensity and extent of leaching by water (H₂O). Differences in the chemical composition between the leaching H₂O and the mineral structures provide the chemical energy to initiate and sustain mineral changes. The rate of leaching controls the composition of the aqueous solution by dictating the rate of removal of the dissolved solutes, and hence the course of the mineral alteration. In regions of limited rainfall, where evapotranspiration exceeds precipitation, the downward flow of H₂O through a soil-containing soluble salts and carbonates is sufficient only in removing the most soluble weathering products, such as sodium (Na) salts. Less soluble minerals tend to form and accumulate in lower soil depths because of limited leaching. Soluble potassium (K) and magnesium (Mg) may form secondary aluminosilicates, while soluble calcium (Ca) may reprecipitate as calcite or gypsum. Secondary calcite precipitation is induced by lower carbon dioxide (CO₂) pressure and higher pH below the soil surface. The accumulation of secondary carbonates in the soil results in considerable inhibition of silicate mineral weathering because of their control of pH and soluble Ca concentrations.

The main weathering pathway for silicate minerals is desilication. Part of the Si released from the mineral structure reacts with Al and to a lesser extent with Fe or Mg to form other clay minerals, while the remainder is subject to leaching. Consequently, most soils experience a loss of Si and basic cations during weathering, which results in a relative increase and residual accumulation of Al- and Fe-enriched mineral structures (ferrallitization; Fig. 1). Desilication is more pronounced in humid tropical environments, but it occurs with lesser intensity in temperate regions as well. Upon desilication and release of Mg and Fe, olivines weather mainly to serpentine and subsequently smectite and goethite.^[8] Similarly, pyroxenes and amphiboles weather through loss of Mg, Fe, and Ca to chlorite and smectite. Under stronger leaching conditions, all three of these mineral groups may produce more advanced weathering mineral products, such as kaolinite, halloysite, and various oxyhydroxides.^[7] Trioctahedral micas (biotite and phlogopite) weather more readily than dioctahedral micas (muscovite and illite) through K release and charge reduction to form interstratified mica-vermiculite, vermiculite, and subsequently smectite.^[8,9] Upon stronger weathering under tropical and subtropical conditions, trioctahedral micas may ultimately form kaolinite and halloysite as well as goethite and hematite. The weathering stages of

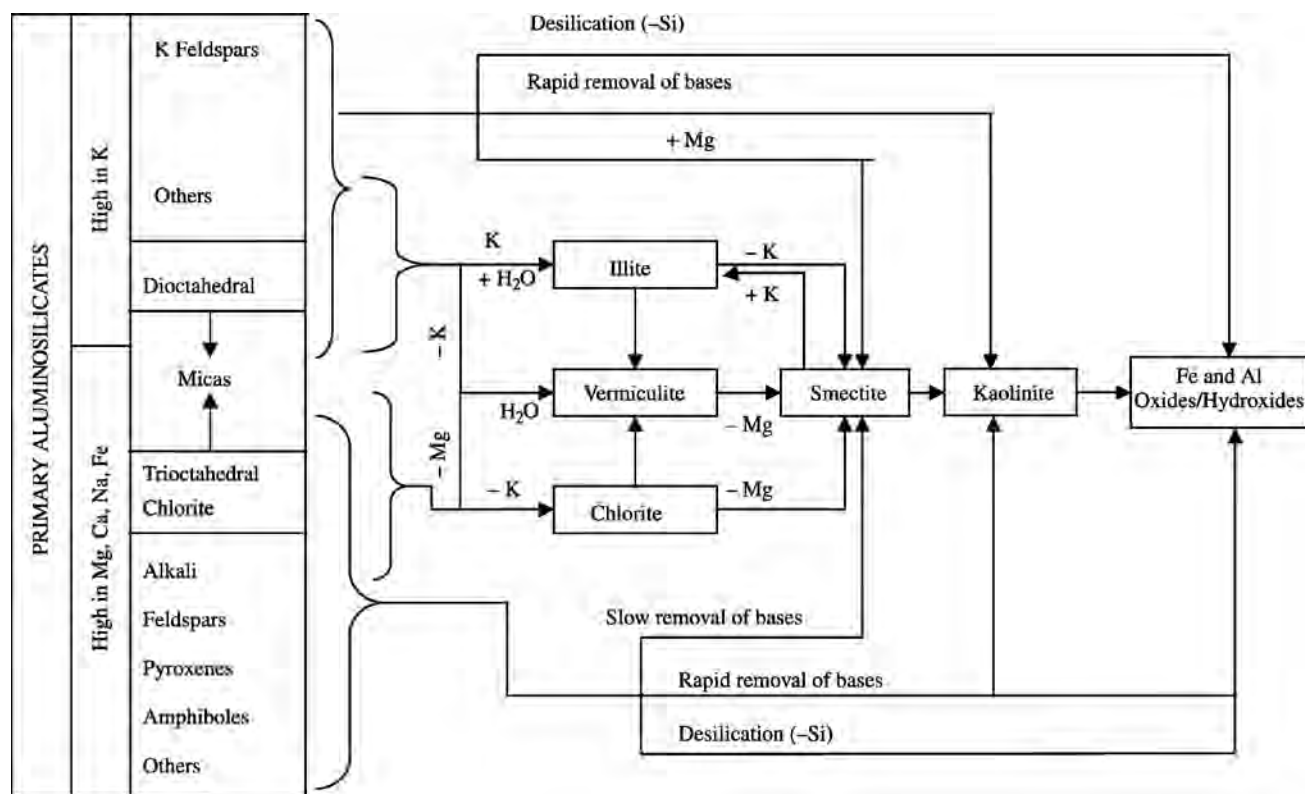


Fig. 1 A generalized mineral weathering/formation pathway scheme.

trioctahedral chlorites are analogous to those of trioctahedral micas, producing interstratified chlorite-vermiculite, vermiculite, smectite, halloysite, kaolinite, and Fe-oxyhydroxides.^[7-9] Dioctahedral micas (muscovite and illite) undergo similar transformations upon weathering to those discussed for trioctahedral micas. However, their intermediate and advanced weathering products are clearly dioctahedral. Furthermore, under the conditions of active leaching, moderately acidic conditions, and low organic carbon content, their vermiculite and smectite weathering products may transform into Al-hydroxyinterlayered versions as a result of Al-hydroxide polymerization within their interlayer spaces.^[7] These minerals, sometimes known as pedogenic chlorites, have a lower charge but display considerably greater stability in the weathering environment.^[10]

Feldspar weathering could lead into a variety of mineral products, including sericite (a micaceous mineral), interstratified mica-smectite, smectite, and a variety of 2:1 aluminosilicates.^[7,11,12] Most often, however, feldspars weather to kaolinite, halloysite, gibbsite, and even quartz.^[9,12] Slow rates of leaching through the feldspar grains usually form 2:1 phyllosilicates, while high rates of leaching induce gibbsite formation.

While vermiculite is considered mostly as an intermediate product of the transformation of primary or secondary mica minerals, smectites can be inherited from parent materials or form either by transformation of other

2:1 phyllosilicates (micas and vermiculite) or by precipitation from saturated solutions.^[7,9] Transformations from other 2:1 minerals usually occur under high leaching acidic environments, whereas neoformation is induced by poor drainage conditions and the accumulation of necessary cations and silica. Under prolonged acidic conditions, vermiculites may transform to smectites through interlayering, while smectites may alter to kaolinite or halloysite following various stages of interstratification.^[9,13] Smectite to kaolinite transformations may involve several stages of interlayering of smectite with various kaolinite precursor phases.^[7,13] Both vermiculite and smectite may also form Al-hydroxyinterlayer phases, which enhance their stability in intense weathering environments. Kaolinite may also form via precipitation from solution under warm or temperate humid conditions, while halloysite neoformation has been often associated with weathering of volcanic materials.^[7] Under extreme desilication conditions and low residence time of Si in the soil solution, gibbsite instead of kaolinite may form. The reverse transformations of gibbsite to kaolinite and kaolinite to smectite in environments experiencing resilication have also been reported. The nature of Fe oxides and oxyhydroxides formed through mineral weathering processes depends mostly on the environmental conditions and less on the structure of the dissolved mineral. Accumulation occurs in well-oxidized environments in the Fe(III) form, while remobilization occurs under anaerobic conditions in the Fe(II) form.^[14] Goethite

formation is usually favored under cool temperate conditions with moderate organic carbon content, while hematite prefers low organic carbon, warmer, and drier environments. Finally, the most stable titanium (Ti) and zirconium (Zr) minerals are the ultimate products of weathering, which accumulate residually in the soil as a result of the dissolution of silicate minerals containing Ti and Zr mainly as impurities in their structure.^[7] Because of their high stability, these minerals are used as indexes of soil development stages and soil age.

THERMODYNAMIC FORMULATIONS OF MINERAL WEATHERING

Based on the principle that the stability of a mineral in the weathering environment depends on the balance or imbalance of its structural ions with the activities of the common ions in the soil solution, Kittrick^[15] developed a thermodynamic approach in predicting mineral stability. The approach involves the formulation of mineral stability lines in terms of major ionic constituents from dissolution–precipitation reactions and plotting them in a stability diagram. For relatively simple aluminosilicate minerals, such diagrams are plotted as functions of $\text{pH} + 1/3 \log \text{Al}^{3+}$ and $\log \text{H}_4\text{SiO}_4$ supported by different minerals from which their stability is deduced under specific ranges of solution ionic conditions (Fig. 2). For minerals with additional structural elements, a fixed nominal range in their activity is assumed, so that their stability lines are expressed in terms of major structural ions only (Al and Si) and pH, and be comparable to those of other minerals. The stability of a given mineral under a specific soil environment is determined from the solution composition coordinates in terms of $\text{pH} + 1/3 \log \text{Al}^{3+}$ and $\log \text{H}_4\text{SiO}_4$ plotted in the diagram. Solution points plotted along a mineral stability line are considered in equilibrium with the mineral, while points plotted above (also to the right) or below (also to the left) the line represent environments in which the specific mineral is stable or unstable, respectively. Since desilication

generally produces more stable minerals, as a convention, the most stable mineral in a stability diagram is the one supporting the lowest level of $\text{pH} + 1/3 \log \text{Al}^{3+}$ for a given level of $\log \text{H}_4\text{SiO}_4$. In the stability diagram of Fig. 2, at $\log \text{H}_4\text{SiO}_4 = -3.5$, the mineral supporting the lowest $\text{pH} + 1/3 \log \text{Al}^{3+}$ and, therefore, the most stable is kaolinite. Similar thermodynamic stability concepts have been used for non-silicate minerals in a variety of geochemical and environmental applications.^[16]

MECHANISMS AND RATES OF MINERAL WEATHERING

Important mechanisms of mineral weathering include dissolution–precipitation, hydrolysis, oxidation–reduction, hydration–dehydration, chelation, and ion exchange.^[4] Dissolution reactions could be congruent or incongruent. During congruent dissolution, the mineral goes into solution completely with no concurrent precipitation of other minerals. In contrast, with incongruent dissolution, all or some of the ions released by weathering precipitate to form new minerals. Relatively soluble minerals (salts, gypsum, and carbonates) and most silicate minerals dissolve congruently, while most aluminosilicate minerals weather through incongruent dissolution reactions. Hydrolysis involves the interaction and replacement of structural cations with H^+ ions dissociated from H_2O . This exchange has a disrupting effect on the crystal surface because of the high charge to ion size ratio of H^+ and weakens the rigidity of the mineral structure. Hydrolysis is the main mechanism in the weathering of feldspars and other aluminosilicates. Minerals containing Fe, Mn, or sulfur (S) are especially susceptible to oxidation–reduction reactions. Oxidation involves the loss of electrons, and therefore, changes in charge. The disruption of the crystal electrostatic neutrality along with the size change of the same element at different oxidation states causes considerable instability in the crystal lattice and makes the mineral more susceptible to weathering. Hydration and dehydration involve the addition or removal of H_2O molecules in the crystal structure of a mineral. They are not as important weathering mechanisms because they affect very few minerals. Examples are the formation of gypsum and anhydrite or halloysite and kaolinite following hydration–dehydration. Complexation is the process by which chelating agents produced through soil biological processes interact with structural cations to form stable ligands and remove them into solution. The process, which affects many clay minerals, destabilizes the mineral structure and facilitates further decomposition. Ion exchange between the solution and mineral surfaces does not produce drastic alterations, since it leaves the mineral structure unchanged, but hydration differences between exchanged ions may cause considerable changes in the physicochemical properties of the interlayer space.

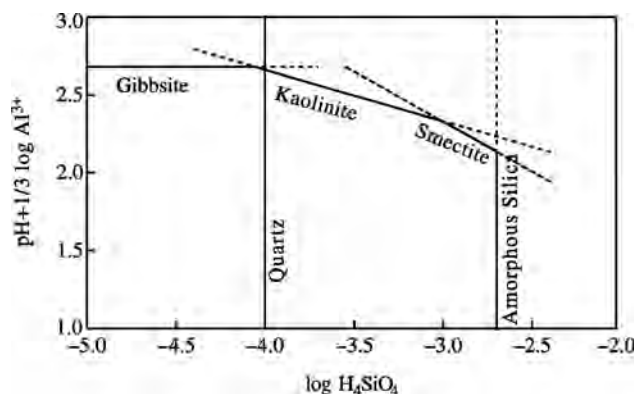


Fig. 2 A typical thermodynamic mineral stability diagram for common soil clay minerals.

Temperature, moisture flux, and solution gradients are the major environmental variables affecting mineral weathering rates. Additional factors include mineral particle surface area and biological activity. Generally, weathering rates increase with temperature, intense leaching and removal of soluble products, surface area, and biological activity. The rate of weathering of silicate minerals is very slow, with average lifetimes ranging from 2.3×10^3 for olivine to 2.4×10^7 for quartz considering 1-mm diameter crystals under pH = 5.0 and 25°C.^[17] This suggests that the bulk mineralogy of a soil is not expected to change considerably during the life span of any given individual. Relative weathering rates are generally consistent with the mineral stability sequences reported earlier, but natural rates are usually several orders of magnitude lower than laboratory estimated rates.^[17] One of the main reasons is that natural systems experience variable moisture fluxes and periodic replenishment compared to the steady flows used during laboratory experiments. Also, the presence and distribution of structural defects and crystal imperfections enhance significantly mineral dissolution rates. Therefore, quite variable dissolution rates may be obtained for the same mineral depending on its degree of crystallinity.

PRACTICAL IMPLICATIONS OF MINERAL WEATHERING

Mineral weathering sequences have been important models in explaining soil development stages over time and understanding respective pedochemical processes. Thermodynamic approaches of mineral weathering have allowed the assessments of existing mineral stability trends and the predictions of future changes in soil ecosystems, as they are impacted by shifts in use and management.^[16] Mineral weathering is also the process contributing several macronutrients, such as Ca, Mg, K, P, and S and most micronutrients to the soil, as they are released from dissolving minerals. Ultimately, these elements are used and recycled by plant communities or leached out of the rooting zone, thus providing an important link between the soil fertility status and degree of mineral weathering. Weathering of silicate minerals also plays an important role in buffering watershed acidification from atmospheric inputs. This is the only process by which the long-term neutralization of produced acidity is achieved through continuous release and replenishment of basic cations from mineral weathering. Global climatic changes are also affected by mineral weathering processes.^[18] On geological timescales, atmospheric CO₂ levels have been primarily controlled by the balance of volcanic inputs from the earth's interior and silicate weathering outputs at the earth's surface. This balance has been disturbed by burning large quantities of fossil

fuel. It will be interesting to see whether the greenhouse effect will be able to trigger high enough silicate weathering rates able to moderate or buffer larger global climatic changes.

REFERENCES

1. Goldich, S.S. A study in rock weathering. *J. Geol.* **1938**, *46*, 17–58.
2. Franke, W.A.; Teschner-Steinhardt, R. An experimental approach to the sequence of the stability of rock-forming minerals towards chemical weathering. *Catena* **1994**, *21*, 279–290.
3. Keller, W.D. Environmental aspects of clay minerals. *J. Sediment. Res.* **1970**, *40*, 788–813.
4. Birkeland, P.W. *Pedology, Weathering and Geomorphological Research*; Oxford University Press: New York, 1974.
5. Jackson, M.L.; Sherman, G.D. Chemical weathering of minerals in soils. In *Advances in Agronomy*; Norman, A.G., Ed.; Academic Press: New York, 1953; 221–317.
6. Marshall, C.E. *The Physical Chemistry and Mineralogy of Soils: Soils in Place*; John Wiley and Sons: New York, 1977.
7. Dixon, J.B.; Weed, S.B. *Minerals in Soil Environments*; Soil Science Society of America: Madison, 1989.
8. Loughnan, F.C. *Chemical Weathering of the Silicate Minerals*; Elsevier: New York, 1969.
9. Churchman, G.J. The alteration and formation of soil minerals by weathering. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; F3–F76.
10. Karathanasis, A.D. Compositional and solubility relationships between AL-hydroxyinterlayered soil smectites and vermiculites. *Soil Sci. Soc. Am. J.* **1988**, *52*, 1500–1508.
11. Millot, G. *Geology of Clays*; Springer-Verlag: New York, 1970.
12. Eswaran, H. The alteration of plagioclases and augites under different environmental conditions. *J. Soil Sci.* **1979**, *30*, 547–555.
13. Karathanasis, A.D.; Hajek, B.F. Transformation of smectite to kaolinite in naturally acid systems: structural and thermodynamic considerations. *Soil Sci. Soc. Am. J.* **1983**, *47*, 158–163.
14. Schwertmann, U.; Fitzpatrick, R.W. Iron minerals in surface environments. *Catena Supp.* **1992**, *21*, 7–30.
15. Kittrick, J.A. *Soil Mineral Weathering*; Van Nostrand Reinhold: New York, 1986.
16. Karathanasis, A.D. Mineral equilibria in environmental soil systems. In *Environmental Soil Mineralogy*; Dixon, J.B., Schulze, D., Eds.; Soil Science Society of America: Madison, 2002; 109–152.
17. White, A.F.; Brantley, S.L. Chemical weathering rates of silicate minerals. In *Reviews in Mineralogy*; Mineralogical Society of America: Washington, 1995; Vol. 31.
18. Lasaga, A.C.; Soler, J.M.; Canor, J.; Burch, T.E.; Nagy, K.L. Chemical weathering rate laws and global geochemical cycles. *Geochim. Cosmochim. Acta.* **1994**, *58*, 2361–2386.

Climate

Alexander N. Gennadiyev

Faculty of Geography, Moscow State University, Moscow, Russia

Serge S. Chernyanskii

Moscow State University, Moscow, Russia

Abstract

Climate is a versatile factor that affects virtually all pedogenetic phenomena. Climatic conditions specify the amounts of solar radiation and precipitation getting to the soil surface. Solar radiation is the main source of energy for most of the processes in the pedosphere. The degree of transformation of soil mineral mass, the rate and character of decomposition of organic remains in the soil, and the nature and functioning of soil biota depend on the supply of solar radiation to the soil, provided that the soil gets a sufficient amount of moisture. Annual precipitation and its distribution by separate seasons together with the evapotranspiration value dictate the degree of soil moistening, the depth of water percolation, the reserves of water in plants, and the water supply of soil microbiota. The dependence of soils on climate is manifested by the climatic zonality of the soil cover, i.e., by the distribution of particular soils over the earth's surface in agreement with thermal belts and regions of different moisture levels. Climatic zonality is one of the main regularities in the development of soil cover patterns.

HISTORICAL BACKGROUND

Since the end of the 19th century, special in-depth studies of the role of climate as a factor that controls the supply of energy and moisture to soil have been conducted by soil scientists. In 1883, Dokuchaev advanced the concept of soil forming factors in his classical monograph *Russian Chernozem*.^[1] Along with climate, he considered parent rocks, living organisms, relief, and time to be equally important obligatory factors for soil formation. At the same time, it was climate that received Dokuchaev's primary attention. The student and follower of Dokuchaev, N.M. Sibirtsev considered climate as the most important soil forming factor. The system of soil classification developed in 1898 by Sibirtsev^[2] stressed the priority of climatic factors. At approximately the same time the works of E.W. Hilgard on the role of climate in soil formation appeared in the United States. Hilgard paid special attention to the study of soil temperature and moisture conditions and their effect on soil processes. The results of these studies were summarized in his works. *A Report on the Relations of Soil to Climate*^[3] and *Soils: Their Formation, Properties, Composition, and Relation to Climate and Plant Growth in the Humid and Arid Regions*.^[4] Later, numerous empirical relationships between various soil properties (the contents of humus, carbonates, nitrogen, clay, saturation capacity, acidity, silica-to-alumina ratio, etc.) and the main climatic parameters (annual precipitation and mean annual temperature) were analyzed by H. Jenny. Jenny suggested the equation of soil forming factors in which the contributions of the

inner (soil) and outer (atmosphere) climate to soil formation were considered, respectively, as the components of the internal and external potentials of pedogenesis.^[5,6] V.R. Volobuev suggested the concept of hydrothermic sequences of soils. He revealed global relationships between the distribution of soils and ecosystems and several climatic parameters (annual precipitation, mean annual temperature, radiation balance, and evaporation capacity).^[7] V.A. Kovda stressed that, in contrast to all the other factors of soil formation having terrestrial origin, the climate is governed by cosmic factors.^[8] He emphasized the need to study soil evolution as related to climate changes in the earth's history. S.W. Buol, F.D. Hole, and R.J. McCracken considered climate (together with living organisms) in the group of *flux factors* responsible for the transfer across a surface of interface of energy, fluids, particles, and other entities.^[9] M.A. Glazovskaya and A.N. Gennadiyev analyzed the contribution of climate to the energy state of soils, their material basis, and the dynamics of soil forming processes.^[10]

It is firmly established that the great diversity of soils on our planet is connected to the diversity of climatic conditions.

DISCUSSION

The annual sum of active (above 10°C) air temperatures (accumulated active temperature), mean annual temperatures, and the indices of radiation balance increases by tens and even hundreds of times from polar regions to the

equator. This results in considerable differences between polar and tropical regions with respect to the thermal state of soils and the climatic potential of pedogenesis. According to Van't Hoff's Law, a 10°C rise in temperature accelerates the rate of chemical reactions by 2 to 3 times. It means that the intensity of chemical reactions, including the weathering of minerals, decomposition of organic matter, and the synthesis of new organomineral compounds, in the soils of tropical regions is tens of times higher than in cold polar regions. Therefore, the thickness of soil profiles and weathering crusts in the tropics is much greater than that in temperate and cold regions of the earth.

The input of solar radiation to the earth's surface proceeds with varying intensity, displaying diurnal, seasonal, annual, and long-term fluctuations (cycles). Therefore, periods of soil heating alternate with periods of soil cooling; in many regions, the alternation between soil thawing and freezing is observed. Various combinations of these processes together with particular temperature conditions in the soil specify the *heat (or temperature) regimes of soils*.

In soils that remain in a frozen state for a long period, the possibilities for the development of soil forming processes are strongly restricted in time: at low temperatures, chemical reactions proceed very slowly or cease, the activity of living organisms is decelerated, and the migration of soil solutions stops. At the same time, soil material may be subjected to cryogenic churning (*cryoturbation*). In contrast, long periods with relatively high temperature favor the development of diverse pedogenetic processes.

The soil temperature regime is conditioned by the direct supply of solar radiation, as well as by the peculiarities of atmospheric circulation, including the advection of warm or cool air masses.

The pedogenetic and biological effects of the heat transmitted from the Sun to the earth's surface can only manifest themselves in full measure if the moisture supply of the soil is sufficient. Atmospheric precipitation is the main factor controlling the solution, leaching, and migration of mobile compounds in the soils. The hydrolysis of primary minerals and the synthesis of secondary clay minerals in soils occur due to the supply of water. Atmospheric water that reaches the soil is retained in soil horizons and consumed by plants in the course of photosynthesis. The biomass produced by plants serves as a source of energy and nutrients for animals and microorganisms. Thus, precipitation is directly and indirectly related to the processes of humus formation in the soils. Eventually, the degree of soil water supply dictates the development of the main genetic horizons of soils, i.e., humus-accumulative, eluvial (horizons with predominant leaching), metamorphic, and illuvial (whose accumulation of substances leached from the above-lying horizons takes place) horizons. Different types of *soil water regime* can be distinguished on the basis of data on the depth and intensity of water migration and the predominance of ascending or descending water flows in the soil. The many-sided role of atmospheric precipitation in the

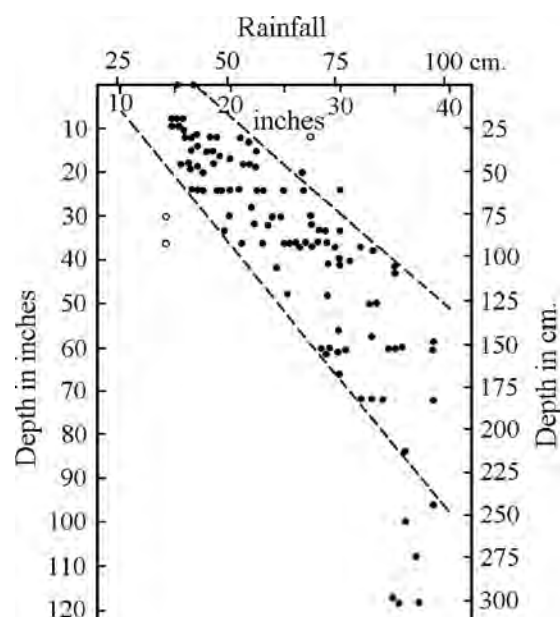


Fig. 1 Relation between depth of carbonate accumulation and rainfall in loessial soils. Every point represents the depth of the beginning of the concretion zone in a soil cut investigated.

Source: From Jenny.^[5]

pedogenetic process is shown in Figs. 1 and 2. The effect of precipitation on soil properties is so significant that it can be indirectly reflected in soil classification. Thus, C. Marbut suggested that all soils could be subdivided at the highest level into two groups, namely, Pedalfers and Pedocals.^[11] Pedalfers included the soils of tundra, taiga, and tropical rainforests, i.e., territories with sufficient or excessive moistening. Intensive leaching of soluble compounds in Pedalfers leads to residual accumulation of less soluble sesquioxides (aluminum and iron). Pedocals

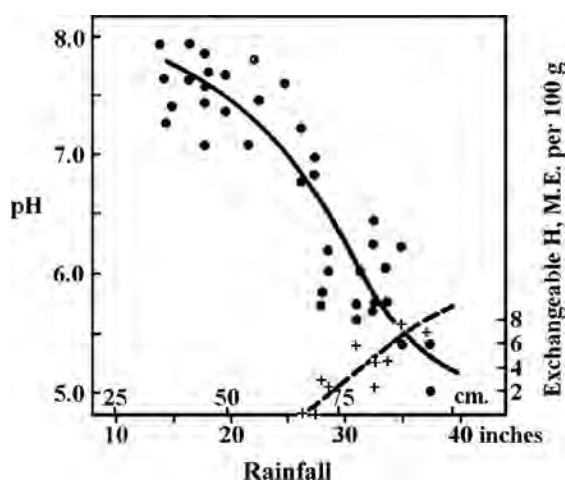


Fig. 2 Relation between soil acidity and rainfall for surface soils derived from similar parent material (loess). Circles stand for pH and crosses for exchangeable hydrogen ions.

Source: From Jenny.^[5]



Fig. 3 Marbut's classification of soils into Pedocals and Pedalfers.

Source: From Jenny.^[5]

included the soils of steppes, semideserts, and deserts with insufficient moistening and accumulation of calcium carbonates in the profile (Fig. 3). The leading role of the moisture factor in soil development is especially stressed by I.A. Sokolov.^[12] According to Sokolov, two fundamentally different worlds of soils can be distinguished: the world (pedocosmos) of arid pedogenesis and the world of humid pedogenesis. Arid pedocosmos consists of accumulative, saturated, neutral or alkaline, calcareous, and/or saline xeromorphic soils. Humid pedocosmos is the world of eluvial, leached, acid, and unsaturated soils.

Variations in annual precipitation across the globe are very considerable. A general trend toward an increase in precipitation from polar regions to the equator (i.e., in the same direction as mean annual air and soil temperatures) is observed. However, the input of solar radiation, and, hence, the heat supply of soils, has a rather distinct latitudinal (zonal) pattern, whereas the zonal distribution of precipitation is strongly altered by the particular geomorphic conditions (the distance from the sea, the location of mountain chains), general circulation of the atmosphere, ocean currents, etc. The overall effect of precipitation and temperature on soil formation has an extremely complex character.

Initially, the most general regularities of soil distribution dependent on spatial changes in moisture and temperature conditions were formulated by Dokuchaev in the form of the Law of Horizontal (for plain territories) and Vertical (for mountains) Soil Zonality. Later, with acquisition of new factual data, this initial perception of soil zonality was refined and shaped into a more complex form. In 1933, I. P. Gerasimov formulated the concept of bioclimatic soil facies. He stressed that local provincial (facial) peculiarities of the climate conditioned by local thermodynamic processes in the atmosphere complicate the latitudinal pattern of soil zonality.

In works devoted to geographic division (regionalization) of the soil cover of the planet and particular regions, much attention has been paid to climatic criteria of such division. Thus, M.A. Glazovskaya^[13] separated the

pedosphere into *pedobioclimatic belts* encompassing horizontal soil zones and vertical soil sequences (mountainous soil provinces) with similar radiation and thermal conditions (e.g., polar, subboreal, and tropical belts). In turn, these belts were subdivided into *pedobioclimatic regions* composed of horizontal soil zones and mountainous soil provinces that not only had similar radiation and thermal conditions but also similar moistening and degree of climatic continentality (e.g., the regions of boreal forests, subboreal steppes, subtropical semideserts, etc.).

The economic activity of humans is an important factor affecting microclimatic conditions. Some human impacts are directly aimed at the transformation of soil climate, i.e., soil water and temperature regimes. Thus, drainage and irrigation, shading the soil surface with a dense canopy of crops, destruction of the root mat on the soil surface by plowing, and changes in the direction and velocity of local winds caused by human constructions or artificial planting of trees (wind-breaking strips or shelter belts) lead to considerable changes in the provision of soils with heat and moisture. All these measures are aimed at maintaining the optimum parameters of soil water and heat supply during the growing season.

REFERENCES

1. Dokuchaev, V.V. Russian chernozem. In *Selected Works*; USSR Academy of Sciences Publishers: Moscow, 1948.
2. Sibirtsev, N.M. Pochvovedeniye (soil science). In *Selected Works*; Sel'khozgiz: Moscow, 1951; Vol. 1.
3. Hilgard, E.W. A report on the relations of soil to climate. U. S. Dep. Agr. Weather Bull. **1892**, 3, 1–59.
4. Hilgard, E.W. *Soils: Their Formation, Properties, Composition, and Relation to Climate and Plant Growth in the Humid and Arid Regions*; MacMillan: New York, 1906.
5. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
6. Jenny, H. Derivation of state equations of soil and ecosystems. Soil Sci. Soc. Am. Proc. **1961**, 25, 385–388.
7. Volobuev, V.R. *Pochvy i Klimat (Soils and Climate)*; SSR: Baku, 1953; 486 pp.
8. Kovda, V.A. *Osnovy Ucheniya o Pochvakh (Basics of Soil Science)*; Nauka: Moscow, 1973; 448 pp.
9. Buol, S.W.; Hole, F.D.; McCracken, R.J. *Soil Genesis and Classification*; Iowa State University Press: Ames, 1989; 446 pp.
10. Glazovskaya, M.A.; Gennadiyev, A.N. *Geografiya Pochv s Osnovami Pochvovedeniya (The Geography of Soils with the Basics of Soil Science)*; Izd. Mosk. Gos. Univ.: Moscow, 1995; 246 pp.
11. Marbut, C.F. *Soils of the United States*; Atlas of American Agriculture, Part III, Advance Sheets, No. 8; US Department of Agriculture: Washington, 1935.
12. Sokolov, I.A. *Pochvoobrazovanie i Ekzogenez (Soil Formation and Exogenesis)*; Dokuchaev Soil Science Institute: Moscow, 1997; 244 pp.
13. Glazovskaya, M.A. *Obshcheye Pochvovedeniye i Geografiya Pochv (General Soil Science and Soil Geography)*; Vysshaya Shkola: Moscow, 1981; 400 pp.

Climate Change: Gas Fluxes

Pascal Boeckx

Oswald Van Cleemput

Faculty of Agricultural and Applied Biological Sciences, University of Ghent, Ghent, Belgium

Abstract

Increasing concentrations of atmospheric gases, such as carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O), contribute to radiative forcing. Activities related to land use are importantly responsible for the build up of CO₂, CH₄, and N₂O in the atmosphere. A survey of techniques will be described for measuring fluxes of greenhouse gases from soils. Subsequently, sampling techniques, sample handling, and means of detection will be presented.

SAMPLING TECHNIQUES, SAMPLE HANDLING, AND ANALYSIS

Chamber Techniques

Flux chambers are simple inverted containers, which form an enclosure for gases emitted from the soil surface.^[1] Both closed (static) and open (dynamic) chambers can be used. Advantages and disadvantages of chamber techniques are listed in [Table 1](#).

Closed and Open Chambers

Emissions of carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O) are more variable, both spatially and temporally. It has not been determined whether the size of the flux chambers has an influence on this variability. Nevertheless, at least six chambers should be used per campaign. Flux chambers (cylindrical, or square or rectangular and box-like) can either be installed as a complete assembly, eventually for a short period and then removed until the next sampling occasion ([Fig. 1A](#)), or they can exist out of a basal part, which is installed for the entire duration of the experiment with a gas-tight chamber attached to it for short periods ([Fig. 1B](#)). This last variant is often used in flooded systems (e.g., paddy soils). The normal procedure for installation is to make a slit in the soil with a metal-cutting edge, the same size and shape as the collar of the chamber, and to insert the chamber collar for about 3 cm into the slit.

When a closed chamber is fixed in place, gas samples can be taken from the headspace at different time intervals. The change in concentration in the chamber over time is used to calculate the gas flux. The calculation of the flux goes through a linear regression analysis of the gas concentration increase with time (corrected for

eventual temperature changes) and a calculation of the chamber volume and area. Typical expressions of fluxes are g GHG ha⁻¹ d⁻¹. A minimum of three measurements should be made to check the linearity of concentration increase in the chamber. A non-linear increase could indicate an inadequate sealing of the chamber or an important increase of pressure in the chamber due to increase in temperature.^[2] However, venting of the closed chambers can create large errors. The chamber cover should be removed once the final sample has been taken to minimize disturbance of the environmental conditions of the area covered by the chamber.

Open chambers can be used as well ([Fig. 2](#)). In open chambers outside air flows into the chamber via an inlet and is forced to flow over the enclosed soil surface before leaving the chamber via an outlet. The concentration of the respective gas is measured at the in- and outlet sides. The gas flux from the soil surface can be calculated from the concentration difference between the in- and outlet, gas flow rate, and volume and area covered by the chamber.

Manual or Automated Sampling and Analysis

Using closed chambers, the headspace can be sampled by syringe. The gas samples are transferred to the laboratory into sealed containers (evacuated vials fitted with rubber septa) for analysis. To obtain a representative sample of the headspace, a diffusive mixing over 30 minutes is adequate, unless a mixing fan is used. Gas samples should be taken from the headspace immediately after sealing and at equal time intervals, thereafter, during which, gas concentrations increase linearly.

With automatic sampling, manual gas sampling from the chamber is replaced by a gas flow system providing a periodic sample transfer to a detector. The basic

Table 1 Advantages and disadvantages of the closed chamber, open chamber, and micrometeorological methods to measure gas fluxes from soils.

Method	Advantage	Disadvantage
Closed chamber	Simple and low cost	Labor intensive
	Multiple gases can be sampled	Small area is covered
	Small fluxes can be measured	Only a short-term emission event is monitored (1–2 hours)
	Manual and automated gas sampling can be used	Disturbance of the emitting surface upon installation
		Altered conditions of temperature and soil atmosphere exchange
		Different functioning of plants in the chamber
Open chamber	Relatively simple	Small area is covered
	Environmental condition close to uncovered field	Disturbance of the emitting surface upon installation
	Continuous long-term monitoring possible	Pressure deficits can cause artificially high fluxes
		Automated sampling is required
Micrometeorological	Useful for diurnal and seasonal variations	Expensive and sophisticated instrumentation needed
	Large areas can be monitored (aggregate flux)	Dependence on a uniform, large surface and constant atmospheric conditions
	Minimal disturbance of the emitting surface	

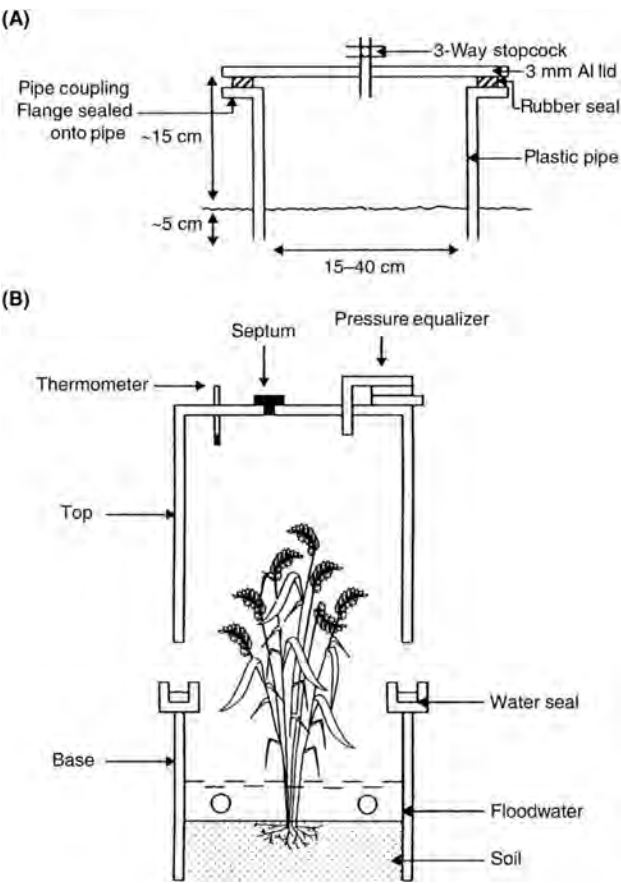


Fig. 1 (A) Typical closed chamber used for GHG flux measurements from aerobic soils. (B) Schematic drawing of closed chamber used for GHG flux measurements in flooded soil.
Source: (A) Adapted from IAEA^[2] and (B) Lindau, Bollich, et al.^[3]

elements of an automatic system consist of closed chambers (equipped with lids that open and close automatically) or open chambers, gas flow systems (tubing and pump); a sampling unit; an analytical unit (detector); a time controller; and a data acquisition system. Automatic sampling is more expensive and it needs to be done in the vicinity of the analysis device. However, this technique is helpful when extensive data sets need to be collected over longer periods of time. It increases the reliability of the emission data obtained, because the number of manipulations is reduced. In most cases, open chambers are sampled automatically.



Fig. 2 Field set-up of an open chamber with removable lid to measure NO fluxes from forest soils: in the front, the outlet with the air sampling tube; in the back, the air inlet with the air sampling tube and the NO analyzer with data acquisition system.

ANALYTICAL ASPECTS

Gas Chromatography

Gas samples are injected into the gas chromatograph (GC) either manually or through the use of a sample loop (automated).^[2] Depending on the type of gas to be analyzed, specific GC settings and columns are used. CO₂ is detected using a thermal conductivity detector (TCD). CH₄ is detected using a flame ionization detector. N₂O is detected using a ⁶³Ni electron capture detector (ECD). For analysis of N₂O, care must be taken to remove both CO₂ and water vapor. In experiments with acetylene, to block the last step in the denitrification process, one may need to bypass the ECD with the acetylene in the sample after it exits from the chromatographic column. Acetylene alters both the sensitivity and stability of some ECDs.

Photo-Acoustic-Infrared Detector (PAID)

The measuring principle is based on photo-acoustic detection of the absorption of infrared light.^[4] This means that any gas that absorbs infrared light of a specific wavelength can be measured. The most common PAID is the *Brüel and Kjær multigas monitor*. The PAID is equipped with a pump, which draws air from the flux chamber into an analysis cell inside the gas monitor. This cell can be sealed hermetically. Light from an infrared source is pulsed by a mechanical chopper and then passes through one of the optical filters of the filter carousel. The filter produces an infrared wavelength, which is selectively absorbed by the gas being monitored. The absorption of infrared light by the gas in the closed analysis cell causes the temperature to increase. Because the infrared light is pulsating, the gas temperature increases and decreases. In the closed cell, this results in an increase and decrease of the pressure, which can be measured via two microphones. This acoustic signal is proportional to the concentration of the gas

monitored. The *Brüel and Kjær multigas monitor* allows the analysis of five different gases and water vapor from one sample in ~120 seconds. The device has the ability to compensate for temperature fluctuations and water vapor interference. Using an automated sampling system, careful calibration of the multigas monitor and the sampling unit is required.^[4]

Micrometeorological Methods

The basic concept of micrometeorological methods for measuring trace gas fluxes to or from the soil surface is that gas transport is accomplished by the eddying motion of the atmosphere, which displaces parcels of air from one level to another.^[5,6] Transport of a gas through the free atmosphere is provided by turbulent diffusion and convection in which the displacement of individual eddies is the basic transport process. Micrometeorological methods are based on the assumption that the flux to or from the soil surface is identical to the vertical flux measured at the reference level some distance above the surface. Therefore, a flat and homogeneous terrain is needed. The flux measured at the reference level provides the average flux over the upwind area (fetch), provided that sampling point at the reference level is in the height range in which the vertical flux is constant with height (fetch ~100 H, where H = height of the sampling point). In the simplest of the micrometeorological methods, the flux may be measured by sensing the concentrations and velocities of components of the turbulence.

Two general micrometeorological techniques are used to measure trace gas fluxes: the eddy correlation and flux-gradient technique (Fig. 3). Application of both approaches is limited to situations in which the air analyzed has passed over a homogeneous exchange surface for a long distance so that profiles of gas concentration in the air are in equilibrium with the local rates of exchange. These methods also require that horizontal concentration gradients are

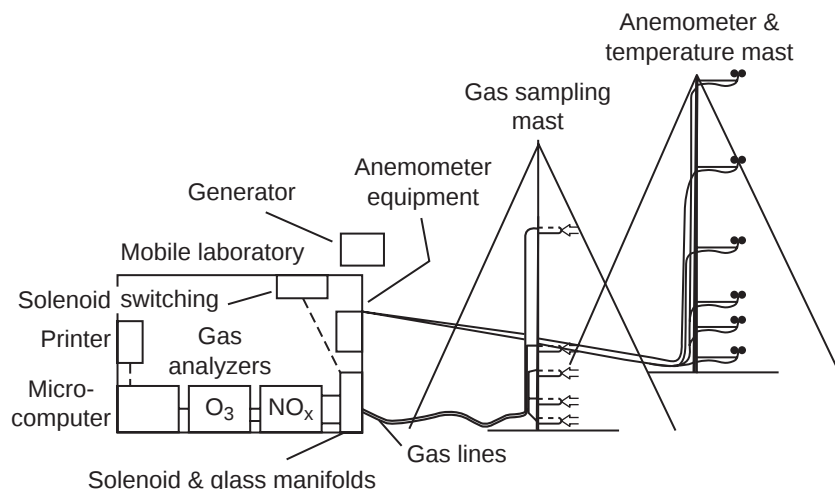


Fig. 3 Typical instrument for flux gradient measurements of GHG fluxes in the field.
Source: Adapted from Denmead.^[5]

negligible. Eddy correlation methods require a fast response detector. The tuneable diode laser technique is based on infrared absorption spectrometry, whereby the absorption depends upon path length, line strength, and absorber concentration. Liquid nitrogen temperature diodes are commercially available to cover the infrared spectrum from about 2 to 10 μm , the region where most trace gases have absorption spectra. Detailed information about these methods can be found by Denmead^[5] and Fowler and Duyzer.^[6] Advantages and disadvantages of micrometeorological methods are listed in Table 1.

Nonisotopic Tracer Methods

Tracer methods involve the release of an inert tracer gas, most commonly sulfur hexafluoride, from an emitting surface.^[2] The tracer gas is released at a known rate in a pattern similar to the release pattern of the greenhouse gas (GHG), perpendicular to the direction of the prevailing wind. This method can be applied when a definite plume of the GHG can be readily detected in the ambient environment. Under these conditions, the plume of the dispersed emission is located based on analyses of upwind and downwind air samples. The flux rate is computed, using the ratio of the plume concentration of the tracer and the GHG and the known release rate of the tracer. The advantage of this technique is that aggregate gas emissions can be collected from heterogeneous areas, such as landfills, circumventing the problem of spatial heterogeneity. However, the high costs, dependence on meteorological conditions, and the potential for interfering sources limit its application.

Ultra-Large Chambers with Long-Path Infrared Spectrometers

Infrared absorption spectrometers are available that can give an average value for the gas concentration over distances of tens or hundreds of meters.^[2] They are useful for

measurements of average emissions from a whole experimental plot, by covering the plot temporarily with a large canopy to act as a chamber and retain the gas emitted from the soil. The following two systems are available: 1) the Fourier transform infrared spectrometer with a mirror system, which allows multiple reflections and thus a total path of up to 1 km that is capable of measuring GHG concentration changes down to a fraction of 1 ppb and 2) a simpler, less-sensitive IR spectrometer with the capacity to detect a concentration change of about 25 ppbv of N_2O and 10 ppbv of CH_4 .

REFERENCES

1. Hutchinson, G.L.; Livingstone, G.P. Use of chamber techniques to measure trace gas fluxes. In *Agricultural Ecosystems on Trace Gases and Global Change*; ASA Special Publication 55; ASA: Madison, 1993; 63–78.
2. IAEA. *Manual on Measurement of Methane and Nitrous Oxide Emissions from Agriculture*; IAEA-TECDOC-674; IAEA: Vienna, 1992; 91 pp.
3. Lindau, C.W.; Bollich, P.K.; De Laune, R.D.; Patrick, W.H., Jr.; Law, V.J. Effect of urea fertilizer and environmental factors on CH_4 emissions from a Louisiana USA rice field. *Plant Soil* **1991**, *136*, 195–203.
4. De Visscher, A.; Goossens, A.; Van Cleemput, A. Calibration of a multipoint sampling system connected with a photoacoustic detector. *Int. J. Environ. An. Ch.* **2000**, *76*, 115–133.
5. Denmead, O.T. Micrometeorological methods for measuring gaseous losses of nitrogen in the field. In *Gaseous Loss of Nitrogen from Plant-Soil Systems*; Freney, J.R., Simpson, J.R., Eds.; Nyhoff Junk Publication: The Hague, 1983; 133–157.
6. Fowler, D.; Duyzer, J. Micrometeorological techniques for the measurement of trace gas exchange. In *Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere*; Andreae, M.O., Schimel, D.S., Eds.; John Wiley and Sons: Chichester, 1989; 189–207.

Climate Change: Soil Carbon Sequestration

Sherwood E. Idso

U.S. Water Conservation Laboratory, Phoenix, Arizona, U.S.A.

Keith E. Idso

Center for the Study of Carbon Dioxide and Global Change, Tempe, Arizona, U.S.A.

Abstract

One of the most promising ways of reducing the rate of rise of the air's carbon dioxide (CO₂) content is to encourage land management policies that promote plant growth, which removes CO₂ from the atmosphere and sequesters its carbon, first in vegetative tissues and ultimately in soils. Some of these policies deal with managed forests and agroecosystems, while others apply to natural ecosystems, such as unmanaged forests and grasslands. In all instances, however, questions abound. Can carbon inputs to soils really be enhanced or carbon losses reduced? Can carbon storage in recalcitrant fractions of soil organic matter be increased, making it possible to successfully maintain new stores of sequestered carbon for long periods? And what if global warming runs wild? Will the ensuing rise in temperature stimulate plant and microbial respiration rates, returning even more CO₂ to the air than is removed by photosynthesis and leading to a negative net ecosystem exchange of carbon? These important questions rank high on the priority lists of many research organizations concerned about the planet's future climate and the sustainability of the biosphere.

INTRODUCTION

Concomitant with mankind's growing numbers and the progression of the Industrial Revolution, there has been a significant increase in the burning of fossil fuels (coal, gas, and oil) over the past 200 years, the carbon dioxide (CO₂) emissions from which have led to ever-increasing concentrations of atmospheric CO₂. This "large-scale geophysical experiment," to borrow the words of two of the phenomenon's early investigators,^[1] is still ongoing and expected to continue throughout the 21st century. Furthermore, this enriching of the air with CO₂ is looked upon with great concern, because CO₂ is an important greenhouse gas, the augmentation of which is believed by many to have the potential to produce significant global warming. Therefore, and because of perceived serious consequences, such as the melting of polar ice; rising sea levels, coastal flooding; and more frequent and intense droughts, floods, and storms,^[2] a concerted effort is underway to slow the rate at which CO₂ accumulates in the atmosphere, with the goal of stabilizing its concentration at a level that would prevent dangerous anthropogenic interference with the planet's climate system.

REMOVING CARBON FROM THE AIR

The Role of Man

There are only two ways to significantly increase the natural flux of carbon from the atmosphere to the biosphere within the time frame required for effective

ameliorative action if the ongoing rise in the air's CO₂ content is indeed a bona fide global warming threat: 1) increasing the rate of vegetative CO₂ assimilation (photosynthesis) per unit leaf area and 2) increasing the total plant population of the globe, i.e., leaf area per unit land area. Additionally, these things must be done without increasing the rate at which carbon is lost from the soil.

Man can do certain things to promote both of these phenomena while meeting the latter requirement as well. He can, e.g., increase the rate of CO₂ assimilation per unit leaf area in agroecosystems by supplying additional nutrients and water to his crops. As has been noted, however, that there are significant carbon costs associated with the production and application of fertilizers, as well as the transport of irrigation water, and factoring the CO₂ emissions of these activities into the equation often results in little net CO₂ removal from the atmosphere via these intensified agricultural interventions.^[3]

Man can also draw more CO₂ out of the air by increasing the acreage of land devoted to growing crops, but this approach simultaneously releases great stores of soil carbon built up over prior centuries. When the plow exposes buried organic matter and it is oxidized, e.g., prodigious amounts of CO₂ are produced and released to the atmosphere. But if a transition to less intensive tillage is made on fields that have a long history of conventional management and have thus been largely depleted of carbon, there is a good opportunity for nature to rebuild previously lost stores of soil organic matter.^[4]

This approach to carbon sequestration is doubly beneficial for its result in a net removal of CO₂ from the

atmosphere at the same time that it enhances a whole host of beneficial soil properties.^[4,5] Also, abandoned farmlands will gradually replenish their carbon stores, both above- and belowground, as native vegetation gradually reestablishes itself upon them. And, of course, the process can be hastened and made even more effective if trees are planted on such lands. Even without trees, it has been estimated that agricultural “best management practices” that employ conservation tillage techniques have the potential to boost the U.S. farm and rangeland soil carbon sequestration rate of 20 million metric tons of carbon per year to fully 200 million metric tons per year,^[6] which is approximately 13% of the country’s yearly carbon emissions.^[7]

Commercial forests also offer excellent opportunities for CO₂ removal from the air for considerable periods of time, especially when harvested wood is used to produce products that have long lifetimes. In addition, since some species of trees, such as many of those found in tropical rainforests,^[8] can live in excess of a thousand years, CO₂ can be removed from the atmosphere and sequestered within their tissues—if man protects the trees from logging—either until long after the age of fossil fuels has run its course or until significant changes in energy systems have reduced our dependence on fossil fuels and the CO₂ content of the air has returned to a level no longer considered problematic. Furthermore, carbon transferred to the soil beneath the trees via root exudation and turnover has the potential to remain sequestered even longer.

The Role of Nature

The fact that the biosphere has maintained itself over the eons in the face of a vast array of environmental perturbations (albeit with significant modifications) suggests that earth’s plant life has great resiliency and may even be able to exert a restraining influence on climate change.

A particularly important negative feedback of this type is the biosphere’s ability to intensify its rate of carbon sequestration in the face of rising atmospheric CO₂ concentrations, as this phenomenon slows the rate of rise of the air’s CO₂ content and thereby reduces the degree of intensification of the atmosphere’s greenhouse effect. This particular climate-moderating influence of atmospheric CO₂ enrichment was first described in quantitative terms by Idso.^[9,10] It begins when the aerial fertilization effect produced by the rising CO₂ content of the atmosphere elicits an increase in plant CO₂ assimilation rate per unit leaf area and when the concomitant plant water use-efficiency-enhancing effect of the elevated CO₂ leads to an increase in the total plant population of the globe, due to the ability of more water use-efficient plants to live and successfully reproduce in areas where it was formerly too dry for them to survive. In fact, these two effects are so powerful, they may actually be able to stabilize the CO₂ content of the atmosphere sometime during the 21st century, but only if anthropogenic CO₂ emission rates do not rise by an inordinate amount in the

interim.^[9,10] At the very least, together with the things man can do, they have the potential to “buy time” until other less-CO₂-emitting technologies become available.^[11]

KEEPING CARBON IN THE SOIL

As more carbon is added to soils via CO₂-enhanced root growth, turnover, and exudation, as well as from CO₂-induced increases in leaf litter and other decaying plant parts, the trick of significantly augmenting soil carbon sequestration is to keep at least the same percentage of this carbon in the soil as has historically been the case and to do so in the face of potential global warming.

A number of studies have addressed various aspects of this subject, with most of them finding that atmospheric CO₂ enrichment has little to no significant effect on plant litter decomposition rates. Furthermore, in nearly all of the cases where elevated CO₂ was observed to impact this phenomenon, the extra CO₂ was found to actually slow down the rate of plant decomposition.^[12] Much of the same results have been obtained when analogous studies have used temperature as the independent variable. Warming either has had no effect on CO₂ evolution from the soil or has led to an actual decrease in CO₂ loss to the atmosphere.^[13] Hence, the balance of evidence obtained from these studies suggests that the same—or a greater—percentage of plant material produced in a world of elevated atmospheric CO₂ concentration (and possibly higher mean air temperature) would indeed be retained in the soils of the terrestrial biosphere.

Even more compelling are the results of experiments where scientists have made direct measurements of changes in soil carbon storage under conditions of elevated atmospheric CO₂. Nearly every such study has observed increases in soil organic matter. In a free-air CO₂ enrichment (FACE) experiment where portions of a cotton field were exposed to a 50% increase in atmospheric CO₂, e.g., Leavitt et al.^[14] found that 10% of the organic carbon present in the soil below the CO₂-enriched plants at the conclusion of the three-year experiment came from the extra CO₂ supplied to the FACE plants. In addition, some of the stored carbon had made its way into a very recalcitrant portion of the soil organic matter that had an average soil residence time of 2200 years.

Here, too, most experiments indicate that concomitant increases in temperature do not negate the increased carbon storage produced by atmospheric CO₂ enrichment. In a two-year study of perennial ryegrass grown at ambient and twice-ambient atmospheric CO₂ concentrations, as well as ambient and ambient + 3°C temperature levels, e.g., Casella and Soussana^[15] determined that the elevated CO₂ increased soil carbon storage by 32% and 96% at low and high levels of soil nitrogen supply, respectively, “with no significant increased temperature effect.” Hence, as in the case of studies of plant decomposition rates, the balance of evidence obtained from these studies also suggests that the

same—or a greater—percentage of plant material produced in a world of elevated atmospheric CO₂ concentration (and possibly higher mean air temperature) would indeed be retained in the soils of the terrestrial biosphere.

CONCLUSION

As the air's CO₂ content continues to rise, there will almost certainly be a significant upward trend in the yearly production of terrestrial vegetative biomass, due to the growth-enhancing aerial fertilization effect of atmospheric CO₂ enrichment and the concomitant CO₂-induced increase in plant water use efficiency that enables plants to grow where it is too dry for them. Experimental evidence further suggests that at least the same percentage—but in all likelihood more—of this yearly increasing mass of plant tissue will be sequestered in earth's soils. Consequently, it is almost impossible to conclude that the carbon sequestering prowess of the planet will not be greatly enhanced in the years ahead, even without any overt actions on the part of man. Hence, if the nations of the earth were to implement even a modicum of carbon-conserving measures—such as 1) using minimum tillage techniques wherever possible in agricultural settings; 2) allowing abandoned agricultural land to revert to its natural vegetative state; 3) allowing stands of trees that can grow to very old age to actually do so; and 4) employing wise forestry practices to produce wood for making products that have long lifetimes—it is possible that the antiwarming feedback produced by the subsequent removal of CO₂ from the atmosphere would

be sufficient to keep the risk of potential greenhouse gas-induced global warming at an acceptable level. Estimates of the carbon sequestering power of some of these “best management practices” are given in Table 1.

REFERENCES

1. Revelle, R.; Suess, H.E. Carbon dioxide exchange between atmosphere and ocean and the question of an increase of atmospheric CO₂ during the past decades. *Tellus* **1957**, *9*, 18–27.
2. Intergovernmental panel on climate change. *Climate Change 2001: The Scientific Basis, Summary for Policy Makers and Technical Summary of the Working Group I Report*; Cambridge University Press: Cambridge, 2001; 1–98.
3. Schlesinger, W.H. Carbon sequestration in soils: some cautions amidst optimism. *Agr. Ecosyst. Environ.* **2000**, *82*, 121–127.
4. Lal, R.; Kimble, J.M.; Follett, R.F.; Cole, C.V. *The Potential for U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Sleeping Bear Press: Chelsea, 1998; 1–128.
5. Idso, S.B. *Carbon Dioxide and Global Change: Earth in Transition*; IBR Press: Tempe, 1989; 292 pp.
6. Jawson, M.D.; Shafer, S.R. Carbon credits on the Chicago board of trade? *Agr. Res.* **2001**, *49* (2), 2.
7. Comis, D.; Becker, H.; Stelljes, K.B. Depositing carbon in the bank. *Agr. Res.* **2001**, *49* (2), 4–7.
8. Chambers, J.Q.; Higuchi, N.; Schimel, J.P. Ancient trees in Amazonia. *Nature* **1998**, *391*, 135–136.
9. Idso, S.B. The aerial fertilization effect of CO₂ and its implications for global carbon cycling and maximum greenhouse warming. *B. Am. Meteorol. Soc.* **1991**, *72*, 962–965.
10. Idso, S.B. Reply to comments of L.D. Danny Harvey, Bert Bolin, and P. Lehmann. *B. Am. Meteorol. Soc.* **1991**, *72*, 1910–1914.
11. Izaurralde, R.C.; Rosenberg, N.J.; Lal, R. Mitigation of climatic change by soil carbon sequestration: issues of science, monitoring, and degraded lands. *Adv. Agron.* **2001**, *70*, 1–75.
12. Nitschelm, J.; Luscher, A.; Hartwig, U.; van Kessel, C. Using stable isotopes to determine soil carbon input differences under ambient and elevated atmospheric CO₂ conditions. *Glob. Change Biol.* **1997**, *3*, 411–416.
13. van Ginkel, J.H.; Whitmore, A.P.; Gorissen, A. Lolium perenne grasslands may function as a sink for atmospheric carbon dioxide. *J. Environ. Qual.* **1999**, *28*, 1580–1584.
14. Leavitt, S.W.; Paul, E.A.; Kimball, B.A.; Hendrey, G.R.; Mauney, G.R.; Rauschkolb, R.; Rogers, H.; Lewin, K.F.; Nagy, J.; Pinter, P.J. Jr.; Johnson, H.B., Jr. Carbon isotope dynamics of free-air CO₂-enriched cotton and soils. *Agr. For. Meteorol.* **1994**, *70*, 87–101.
15. Casella, E.; Soussana, J.-F.; Loiseau and P. Long-term effects of CO₂ enrichment and temperature increase on the carbon balance of a temperate grass sward. *J. Exp. Bot.* **1997**, *48*, 1309–1321.
16. Follett, R.F.; Kimble, J.M.; Lal, R. *The Potential of U.S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect*; Lewis Publishers, Boca Raton, 2001; 1–442.

Table 1 Potential rates of carbon sequestration (kilograms carbon per hectare per year) due to land management practices that could be employed for this purpose.

Improved rangeland management	50 to 150
Improved pastureland management	
Commercial fertilizer applications	100 to 200
Manure applications	200 to 500
Use of improved plant species	100 to 300
Improved grazing management	300 to 1300
Nitrogen fertilization of mountain meadows	100 to 200
Restoration of eroded soils	50 to 200
Restoration of mined lands	1000 to 3000
Conversion of cropland to pasture	400 to 1200
Conversion of cropland to natural vegetation	600 to 900
Conversion from conventional to conservation tillage	
No till	500
Mulch till	500
Ridge till	500

Source: Adapted from data reported by Follett, Kimble, et al.^[16] and Lal, Kimble, et al.^[4]

Cobalt and Iodine

Ronald G. McLaren

Soil, Plant, and Ecological Sciences Division, Lincoln University, Canterbury, New Zealand

Abstract

Cobalt (Co) and iodine (I) are two trace elements that are generally considered to be non-essential for the growth of higher plants. However, both elements are essential nutrients for animals, particularly in the case of ruminants (sheep and cattle), and deficiencies of Co and I in grazing animals are not uncommon. Co is also essential for nitrogen fixation by microorganisms such as rhizobium and blue-green algae and can actually be toxic to plants if present at high concentrations in the soil. Most investigations into Co and I in the soil have concentrated on factors affecting the plant availability of these elements, with the aim of diagnosing potential deficiencies.

SOIL COBALT (Co)

The Co concentration in soils depends primarily on the parent materials (rocks) from which they were formed and on the degree of weathering undergone during soil development.^[1,2] Co tends to be most abundant as a substituent ion in ferromagnesian minerals and therefore has relatively high concentrations in mafic and ultramafic rocks (rocks containing high or extremely high proportions of ferromagnesian minerals). Conversely, Co concentrations are relatively low in felsic rocks (rocks containing large amounts of silica-rich minerals) such as granite and in coarse-textured quartz-rich sedimentary rocks (sandstones). Higher concentrations of Co may be associated with finer textured sediments (shales) in which Co has become surface-adsorbed by, or incorporated into, secondary layer silicates by isomorphous substitution.^[3] Typical concentrations of Co reported in different rock types are shown in [Table 1](#). As a result of the large range in Co concentrations of soil parent materials, and variation in the degree of weathering, total soil Co concentrations also vary widely. However, the mean values reported for agricultural soils from many countries appear to have a somewhat restricted range of between approximately 2 and 20 mg/kg ([Table 1](#)).

Forms of Co in Soils

Co in soils, whether released from parent materials during soil development or derived from anthropogenic contaminant sources, occurs in several different forms or associations. Co may be present as: 1) the simple Co^{2+} ion, or as complexes with various organic or inorganic ligands in the soil solution; 2) exchangeable Co^{2+} ions; 3) specifically adsorbed Co, bound to the surfaces of inorganic soil colloids [clays and oxides/hydrous oxides of aluminum (Al),

iron (Fe), and manganese (Mn)]; 4) Co complexed by soil organic colloids; 5) Co occluded by soil oxide materials; and 6) Co present within the crystal structures of primary and secondary silicate minerals.^[6] In some soils, there appears to be a particularly strong association between Co and Mn oxides, especially in soils where Mn oxides occur as distinct nodules or coatings.^[7,8]

Plant Availability of Soil Co

The immediate source of Co for plant uptake is the soil solution. However, Co concentrations in the soil solution are extremely low, generally much less than 0.1 $\mu\text{g/ml}$.^[1] Soil solution Co appears to be in equilibrium with Co adsorbed at the surfaces of soil colloids, and the distribution between solution and surface phases is strongly influenced by soil pH. As pH increases, soluble Co decreases.^[6] Similarly, soils with high capacities to adsorb Co, particularly those soils with high Mn oxide contents, also have low solution Co concentrations.^[9] Thus, in addition to soils with low total Co concentrations, the plant availability of Co may be low in soils with high pH or high Co adsorption capacities. Co availability is also influenced by soil moisture status, increasing under waterlogged conditions.^[10]

Determination of soil Co dissolved by various extractants is the most common way of assessing the plant availability of soil Co. Ideally, the forms of Co extracted should include any soluble and exchangeable Co, together with any solid-phase forms of Co that are able to move readily into the soil solution in response to changes in solution Co concentrations.^[6] The two extractants used most commonly for this purpose are 2.5% acetic acid and solutions of ethylenediaminetetraacetic acid. However, the ability of these extractants to accurately assess Co availability appears to be somewhat limited.^[11,12]

Table 1 Co and I concentrations in rocks and soils.

	Co concentration (mg/kg)	I concentration (mg/kg)
Rock type		
Ultramafic (e.g., serpentinite)	100–300	0.01–0.5
Mafic (e.g., basalt and gabbro)	30–100	0.08–50
Intermediate (e.g., diorites)	1–30	0.3–0.5
Felsic (e.g., granites and gneiss)	<1–10	0.2–0.5
Sandstones	0.3–10	0.5–1.5
Shales/argillites	11–40	2–6
Limestones	0.1–3.0	0.5–3.0
Soils		
Complete range	0.1–300	<0.1–25.4
Range of mean values	2–21.5	1.1–13.1

Source: From Kabata-Pendias & Pendias,^[1] Aubert & Pinta,^[4] and Swaine.^[5]

Co DEFICIENCY

Co deficiency in grazing sheep and cattle was first diagnosed in the 1930s, initially in New Zealand, Australia, and Scotland.^[13–15] The condition causes animals to lose their appetite, become weak and emaciated, suffer severe anemia, and eventually die. Subsequently, it was shown that Co is a constituent of both vitamin B₁₂ and a closely related coenzyme and that Co deficiency is in effect a deficiency of vitamin B₁₂.^[16] It has been concluded from field studies that pasture containing Co below 0.08 mg/kg for sheep or below 0.04 mg/kg for cattle is unlikely to meet nutritional requirements in terms of maintaining adequate serum and liver vitamin B₁₂ concentrations and healthy growth rates.^[17,18]

Co deficiency is commonly treated by applying Co-containing fertilizers to pastures, usually at very low rates, e.g., 350 g/ha/yr of CoSO₄·7H₂O. However, on some soils such treatments appear to be ineffective.^[11] Alternative treatments for Co deficiency involve injecting the animal with vitamin B₁₂, the use of Co drenches, or the insertion of slowly dissolving Co “bullets” in the animal’s rumen.

SOIL I

The range of I concentrations found in rocks and soils is shown in Table 1. I occurs as a minor constituent in various minerals, where it can replace anions such as OH[−], SiO₄^{4−}, and CO₃^{2−}, and has relatively low concentrations in most types of rock.^[19] Highest I concentrations are generally found in fine-grained sedimentary rocks (Table 1). However, soil I concentrations are generally higher than that in the rocks from which they have been derived, a fact

attributed predominantly to atmospheric accessions of I.^[20] I is known to be present in the atmosphere in vapor form and associated with particles of dust. In coastal areas, accession of I may also be related to sea spray.^[1]

Forms of I in Soils

Information of the forms of I in soils is limited, most published analyses of soils have determined total I concentrations only. Of the three most common forms of I, namely elemental I (I₂), iodide (I[−]), and iodate (IO₃[−]), it seems likely that most I in soils is present as I[−] or possibly as I₂. The presence of IO₃[−] in soils has also been postulated, but would require high oxidation conditions in neutral or alkaline soils.^[20] Indeed, there is some evidence that when IO₃[−] is added to soils, it is rapidly reduced to I₂ or I[−] by soil organic matter.^[21] There is also evidence that, under some conditions, I₂ can be volatilized from soils.^[22]

Most I in soils appears to be associated with soil organic matter and Fe and Al oxides, materials by which both I[−] and I₂ are known to be strongly adsorbed.^[21,23] Indeed, the distribution of I in soil profiles appears to follow the distribution of these soil components.^[24] The atmospheric accessions of I and the affinity between I and organic matter often result in maximum concentrations of I in the surface horizons of soils.^[20,24]

Plant Availability of Soil I

Interest in soil analysis as a means of assessing plant availability of I has been minimal,^[20] and relationships between the I status of soils and the concentrations of I in plants appear to be poor.^[1] Even soil extractants designed to determine the most soluble forms of I in soils do not provide a good indication of I availability to plants.^[25] Plants are capable of absorbing I directly from the atmosphere,^[1] and plant species or varietal differences appear to have a greater influence on plant I concentration than soil I status.^[20,26] Dicotyledonous pasture species (e.g., clovers) generally have higher I contents than do grasses.^[26] Concentrations of I in plants may be reduced by liming,^[25] by the application of nitrogen fertilizer,^[26] and by the application of farmyard manure.^[25] Seasonal effects on pasture I concentrations have also been observed, with decreases in the summer and slight increases in the autumn.^[26]

I DEFICIENCY

Low concentrations of I in food and water have been associated with the occurrence of endemic goitre (enlargement of the thyroid gland) in humans and farm livestock.^[27] Early work suggested a close relationship between goitre incidence and low soil levels of I; however, it is recognized that other factors are also involved. In particular, the

presence of a group of substances known as goitrogens, which occur in various plant species, has been shown to reduce thyroid hormone synthesis and metabolism.^[20] In the absence of goitrogens, it is considered that diets containing 0.5 mg I/kg DM will more adequately meet the I requirements of all classes of animals, while levels as low as 0.15 mg/kg might be sufficient to meet the requirements for growing animals.^[28] In the presence of goitrogens, I requirements may be as high as 2 mg I/kg DM.^[28] Attempts to increase pasture I concentrations with I-containing fertilizers have been generally ineffective, with very low recoveries of the applied I.^[25,26] Direct supplementation of livestock is normally the preferred way to increase I intakes.

CONCLUSION

Co and I deficiencies in livestock result from low soil concentrations and/or low availability of these trace elements for uptake by pasture plants. Plant availabilities of Co and I are determined by several factors including soil forms, soil sorption properties, soil pH, soil moisture status, season, plant species, and fertilizer applications. Deficiencies of Co and I can be prevented by the application of fertilizers (Co) to the soil or by direct treatment of livestock (Co and I).

REFERENCES

- Kabata-Pendias, A.; Pendias, H. *Trace Elements in Soils and Plants*; CRC Press: Boca Raton, 1984; 238–246.
- Mitchell, R.L. Cobalt in soil and its uptake by plants. In *Atti del IX Simposio Internazionale di Agrochimica su La Fitonutrizione Oligominerale*; Punta Ala, Argentina, Oct 2, 1972; 521–532.
- Hodgson, J.F. Chemistry of micronutrient elements in soils. *Adv. Agron.* **1963**, *15*, 119–159.
- Aubert, H.; Pinta, M. *Trace Elements in Soils, Development in Soil Science*; Elsevier Scientific Publishing Co.: Amsterdam, 1977; 395 pp.
- Swaine, D.J. *The Trace Element Content of Soils*; CAB: Harpenden, 1955; 157 pp.
- McLaren, R.G.; Lawson, D.M.; Swift, R.S. The forms of cobalt in some Scottish soils as determined by extraction and isotopic exchange. *J. Soil Sci.* **1986**, *37*, 223–234.
- Taylor, R.M.; McKenzie, R.M. The association of trace elements with manganese minerals in Australian soils. *Aust. J. Soil Res.* **1966**, *4*, 29–39.
- Jarvis, S.C. The association of cobalt with easily reducible manganese in some acidic permanent grassland soils. *J. Soil Sci.* **1984**, *35*, 431–438.
- Tiller, K.G.; Honeysett, J.L.; Hallsworth, E.G. The isotopically exchangeable form of native and applied cobalt in soils. *Aust. J. Soil Res.* **1969**, *7*, 43–56.
- Adams, S.N.; Honeysett, J.L. Some effects of soil waterlogging on Co and Cu status of pasture plants grown in pots. *Aust. J. Agric. Res.* **1964**, *15*, 357–367.
- McLaren, R.G.; Lawson, D.M.; Swift, R.S.; Purves, D. The effects of cobalt additions on soil and herbage cobalt concentrations in some S.E. Scotland pastures. *J. Agric. Sci. Camb.* **1985**, *105*, 347–363.
- McLaren, R.G.; Lawson, D.M.; Swift, R.S. The availability to pasture plants of native and applied soil cobalt in relation to extractable soil cobalt and other soil properties. *J. Sci. Food Agr.* **1987**, *39*, 101–112.
- Grange, L.I.; Taylor, N.H. Bush sickness. Part IIA. The distribution and field characteristics of bush-sickness soils. *Bull. N. Z. Dept. Sci. Ind. Res.* **1932**, *32*, 21.
- Underwood, E.J.; Filmer, J.F. The determination of the biologically potent element (cobalt) in limonite. *Aust. Vet. J.* **1935**, *11*, 84–92.
- Corner, H.H.; Smith, A.M. The influence of cobalt on pine disease in sheep. *Biochem. J.* **1938**, *32*, 1800–1805.
- Smith, R.M.; Gawthorne, J.M. The biochemical basis of deficiencies of zinc, manganese, copper and cobalt in animals. In *Trace Elements in the Soil-Plant-Animal System*; Nicholas, D.J.D., Egan, A.R., Eds.; Academic Press: New York, 1975; 243–258.
- Andrews, E.D. Observations on the thrift of young sheep on a marginally cobalt deficient area. *N. Z. J. Agric. Res.* **1965**, *8*, 788–817.
- Gardner, M.R. *Cobalt in Ruminant Nutrition: A Review*; Department of Agriculture: Western Australia, 1977; 602–620.
- Goldschmidt, V.M. *Geochemistry*; Oxford University Press: Oxford, 1954; 730.
- Fleming, G.A. Essential micronutrients. II. Iodine and selenium. In *Applied Soil Trace Elements*; Davies, B.E., Ed.; John Wiley and Sons, Ltd.: Chichester, 1980; 199–234.
- Whitehead, D.C. The influence of organic matter, chalk, and sesquioxides on the solubility of iodide, elemental iodine, and iodate incubated with soil. *J. Soil Sci.* **1974**, *25*, 461–470.
- Whitehead, D.C. The volatilisation, from soils and mixtures of soil components, of iodine added as potassium iodide. *J. Soil Sci.* **1981**, *32*, 97–102.
- Whitehead, D.C. The sorption of iodide by soils as influenced by equilibrium conditions and soil properties. *J. Sci. Food. Agric.* **1973**, *24*, 547–556.
- Whitehead, D.C. Iodine in soil profiles in relation to iron and aluminium oxides and organic matter. *J. Soil Sci.* **1978**, *29*, 88–94.
- Whitehead, D.C. Uptake by perennial ryegrass of iodide, elemental iodine and iodate added to soil as influenced by various amendments. *J. Sci. Food. Agr.* **1975**, *24*, 361–367.
- Hartmans, J. Factors affecting the herbage iodine content. *Neth. J. Agr. Sci.* **1974**, *22*, 195–206.
- Underwood, E.J. *Trace Elements in Human and Animal Nutrition*, 3rd Ed.; Academic Press: New York, 1971; 543 pp.
- Agricultural Research Council. *The Nutrient Requirements of Ruminant Livestock*; Commonwealth Agricultural Bureau: Slough, 1980; 351 pp.

Collembola

Steve P. Hopkin

Division of Zoology, School of Animal and Microbial Sciences, Reading, U.K.

Abstract

Collembola are widespread and abundant in most soils. Their biomass is low but they contribute to nutrient cycling by consuming fungal hyphae. Deposition of fecal pellets throughout the soil profile contributes to soil fertility. Collembola are beneficial, only rarely being pests of aboveground vegetation.

INTRODUCTION

Collembola (commonly known as “springtails”) are considered to be a monophyletic class of the phylum Arthropoda,^[1] although their exact taxonomic position is the subject of some debate. Many authors treat Collembola as insects. However, many of the old ideas on arthropod evolution are being reassessed in the light of modern evolutionary theory and molecular phylogeny so their placement may change.

There are three main Orders of Collembola. Members of the *Arthropleona* (about 6000 species) have a more or less elongated body shape and range from highly active surface-dwelling species to those that live out all their lives in the depths of the soil. An example of this Order is *Folsomia candida* (Fig. 1), which belongs to the Family Isotomidae. It is familiar to ecotoxicologists as one of a suite of “standard test soil organisms,” which are used to assess the toxicity of new chemicals before they are released into the environment. The *Symphyleona* (about 1000 species) have a much more rounded body shape and are mostly attractively patterned, surface-living species. A typical example is *Dicyrtomina ornata* (Fig. 2) (subfamily Dicyrtominae). The *Neelipleona* are very small soil-inhabiting springtails (typically 0.5 mm in length) with only about 25 species known in the world. They have a rounded body shape and bear a superficial resemblance to Symphyleona.

MORPHOLOGY

The oldest fossil Collembola (indeed the oldest fossil insects) are specimens preserved in Rhynie Chert of more than 400 million years in age. Collembola are present in amber in which remarkable detail is preserved (Fig. 3). The most obvious feature of Collembola is the jumping organ or furca, which is clearly visible in the specimen shown in Fig. 3 projecting behind the animal in its “sprung” state. The furca evolved through the basal fusion

of a pair of appendages on the fourth abdominal segment and is capable of propelling some springtails many times their own body length in a fraction of a second. The spring developed as an escape mechanism to avoid predators. Species of Collembola confined to the soil have a reduced furca to ease their movement between soil particles and tightly packed leaf litter. Some have lost the jumping organ altogether. The maximum number of ocelli in each eye is eight, but these are often reduced and many soil-dwelling and cave species are blind.

All Collembola have a *ventral tube* that consists of eversible sacs derived from a pair of appendages on the first abdominal segment. This organ is extremely important in fluid balance but can also function as a sticky appendage to enable springtails to adhere to slippery surfaces. In some species, the vesicles of the ventral tube may extend more than twice the length of the body and be used for self-righting after a jump. Lubbock introduced the scientific name for springtails in 1873.^[2] He rightly considered the ventral tube to be the most characteristic feature of the group and gave them the name Collembola based on the Greek *Colle* (=glue) and *embo-lon* (=piston).

Collembola are small animals. Most are only a few millimeters long although the Central European species *Tetradontophora bielensis* can reach 9 mm in length, and some members of the Subfamily Uchidanurinae grow to 10 mm and bear brightly colored “spines.” The tiniest and least-pigmented species tend to be those that live permanently in the spaces between particles of soil or sand. The blood of some Collembola possesses chemicals that act as powerful feeding deterrents to predators.^[3]

ECONOMIC IMPORTANCE

The majority of springtails feed on fungal hyphae or decaying plant material. In the soil, they may influence the growth of mycorrhizae^[4] and control fungal diseases



Fig. 1 Adults and juveniles of *F. candida*. The largest specimen is 2 mm in length.

of some plants.^[5] In general, these effects are beneficial; however, there are a few species, including *Sminthurus viridis* the “Lucerne flea,” which feed directly on plant material. In Australia, a country to which *S. viridis* has been introduced, the species can cause significant economic damage.^[6] Some springtails are carnivorous and eat nematodes, rotifers, and even other Collembola.

LONGEVITY

Springtails are generally short-lived. Few survive as adults for more than a year or two. The documented longevity record is held by *Pseudosinella decipiens*, which survived for up to 67 months in the laboratory, although some cave species, or those in very cold climates, may live longer. Most Collembola continue to moult after reaching reproductive maturity and may alternate reproductive and “sexually dormant” instars through several cycles. Several species are parthenogenetic (males absent) including *F. candida* (Fig. 1).



Fig. 2 *D. ornata* (1.5 mm in length).

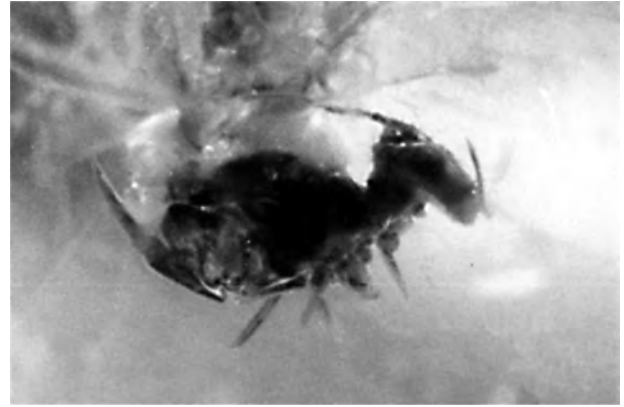


Fig. 3 Symphypleonid springtail (1 mm in length) preserved in Baltic amber of 40 million years in age.

Source: Specimen supplied by Andrew Ross, Natural History Museum, London.

HOW MANY SPECIES?

Approximately 7000 species of Collembola have been described although it is difficult to give an exact figure as there are many species yet to be discovered, especially in countries such as Australia and New Zealand where there is a high level of endemism. The estimate of the total number of species of all insects on the earth quotes a figure of between 5 and 10 million, less than 20% of which have been described.^[7] In view of their cryptic behavior and relative lack of scientific study, it is likely that at least 50,000 species of springtail exist on our planet.

DISTRIBUTION

Collembola have a very wide global distribution. They are abundant on every continent, including Antarctica, where *Biscoia sudpolaris* and *Antarctophorus sudpolaris* have been found crawling among lichen at a latitude of 84°47'S, the most southerly location for any invertebrate.^[8] *Aackia karakoramensis* occurs on newly fallen snow in the Himalayas at an altitude of 7742 meters.^[9] *Folsomides arnoldi* is abundant in Australian deserts.^[10]

Collembola are common on the seashore. *Anurida maritima* is a marine species and is one of the most familiar invertebrates of the littoral zone in Europe.^[11] Several species live almost permanently on the surface of freshwater, including the common and widespread *Podura aquatica*, a frequent sight on puddles after rain, sometimes in huge numbers. Species of *Hypogastrura* are abundant and are important in clearing growths from the percolation filters of sewage beds.

Many species live all their lives in the soil, where they penetrate more than 150 cm below the surface. Others live on trees and are abundant in rain forest canopies. In one

study, about one million Collembola of 16 species were collected from 100 m² of dry forest in Mexico by insecticide fogging.^[12] Some Collembola are specialized for living in sand.^[13]

DIVERSITY

Habitats with extreme climates such as deserts and polar regions support few species of Collembola, but sites with many niches have a diverse springtail fauna. Collembola seem to follow the general rule that diversity is inversely related to latitude; that is, there are more species in tropical than in temperate zones. In tropical rain forests, more than 130 species/m² have been found in soil, leaf litter, and aboveground vegetation.^[14] In more temperate forests, diversity is lower, but it is not unusual to find more than 40 species in deciduous woodland.^[15] Collembola exhibit dominance patterns typical of most groups of terrestrial arthropods. The majority of individuals are usually represented by a small number of common species. In most populations, a large fraction of the species (usually >50%) is rare with dominance values of <1%.

“SWARMING”

Collembola are most obvious when they swarm. Most reports are of species in the family Hypogastruridae. They occur following synchronized reproduction in conditions of ideal humidity and temperature and abundant food supply. There are numerous references in the literature to swarming, particularly on snow and glaciers (e.g., Hagvar^[16]). Swarms certainly can be huge, often comprising several millions of individuals. The reasons for this behavior are not completely understood, although in most cases the Collembola are probably searching for a more favorable habitat after becoming overcrowded.

ABUNDANCE

Collembola are extremely abundant in soil and leaf litter. In most terrestrial ecosystems, they occur in high numbers, typically between 10⁴ and 10⁵/m². Densities of springtails of more than 10⁵/m² have been found in pine forests in India and Japan, moorland in England, and dry meadows in Norway. Collembola are particularly abundant in agricultural soils that are farmed “organically.”^[17] In the rain forests of Seram, Indonesia, Collembola comprise about 20% of the total number of arthropods on tree trunks and 50% and 60% of the total from soil and leaf litter, respectively.^[18] However, because of their small size the contribution of Collembola to total soil animal biomass and respiration is low, typically between 1% and 5% in temperate ecosystems, but up to about 10% in some arctic sites

and as much as 33% of total soil fauna respiration in ecosystems in the early stages of succession. Typical values for the dry weight of springtails in temperate ecosystems are 0.15 g/m² in deciduous woodland and 0.3 g/m² in limestone grassland.

EFFECTS ON SOIL STRUCTURE

Despite their relatively low biomass, springtails are extremely important in influencing the structure of some soils. For example, “alpine pitch Rendzinas” on limestone are composed mainly of a deep black humus layer of 15–20 cm in depth that is formed almost entirely of Collembola feces.^[19] Most soils contain millions of collembolan fecal pellets/m², and these must be beneficial in slowly releasing essential nutrients to plant roots as the pellets are broken down by microbes.

ROLE IN DECOMPOSITION

The main effect of Collembola on decomposition and “soil respiration” is through feeding on fungal hyphae. At certain densities of Collembola, grazing of mycorrhizae on roots can stimulate growth of the symbiont and improve plant growth.^[4] In other situations, Collembola may reduce disease by consuming pest fungi.^[5]

Selective grazing by springtails may be an important factor limiting the distribution of certain species of basidiomycete fungi in the field. However, many of these effects are density-dependent, and too little information is available for quantifying accurately the specific contribution of Collembola to “indirect” or “catalytic” decomposition. Nevertheless, the influence of springtails on decomposition and nutrient availability must be significant in many ecosystems.

CONCLUSION

Collembola are widespread and abundant in most soils. Their biomass is low but they contribute to nutrient cycling by consuming fungal hyphae. Deposition of fecal pellets throughout the soil profile contributes to soil fertility. Collembola are beneficial, only rarely being pests of aboveground vegetation.

Further Reading

Further information can be found in the review by Hopkin, which contains a list of the most important references on Collembola published before 1996. A list of Collembola publications from 1995 onward is available at <http://www.stevehopkin.co.uk/collembolapapers/>. A very detailed Collembola site is maintained by Frans Janssens at <http://www.collembola.org>. This includes a

world checklist of species together with identification keys to genus level for most families. Since the publication of Hopkin a number of important reviews have been published that should be consulted for developments in taxonomy, ecotoxicology and general biology. These are as follow:

- a major revision of the Iberian Collembola by Jordana et al.
- a second edition of the key to North American Collembola by Christiansen and Bellinger
- the first volume of a key to the Nordic Collembola by Fjellberg
- three volumes in the series on Palaearctic Collembola
- a review of developments in Collembola taxonomy
- a discussion of the use of 'standard' springtail *F. candida* in laboratory experiments.

REFERENCES

1. Cameron, S.L.; Miller, K.B.; D'Haese, C.A.; Whiting, M.F.; Barker, S.C. Mitochondrial genome data alone are not enough to unambiguously resolve the relationships of Entognatha, Insecta and Crustacea *sensu lato* (Arthropoda). *Cladistics* **2004**, *20*, 534–557.
2. Lubbock, J. *Monograph of the Collembola and Thysanura*; Ray Society: London, 1873.
3. Messer, C.; Walther, J.; Dettner, K.; Schulz, S. Chemical deterrents in podurid Collembola. *Pedobiologia* **2000**, *44*, 210–220.
4. Gange, A. Arbuscular mycorrhizal fungi, collembola and plant growth. *Trends Ecol. Evol.* **2000**, *15*, 369–372.
5. Sabatini, M.A.; Innocenti, G. Effects of collembola on plant-pathogenic fungi interactions in simple experimental systems. *Biol. Fert. Soils* **2001**, *33*, 62–66.
6. Bishop, A.L.; Harris, A.M.; McKenzie, H.J. Distribution and ecology of the lucerne flea, *Sminthurus viridis* (L.) (Collembola: Sminthuridae), in irrigated lucerne in the hunter dairying region of New South Wales. *Aust. J. Entomol.* **2001**, *40*, 49–55.
7. Odegaard, F. How many species of arthropods? Erwin's estimate revised. *Biol. J. Linn. Soc.* **2000**, *71*, 583–597.
8. Block, W. Terrestrial microbiology, invertebrates and ecosystems. In *Antarctic Ecology*; Laws, R.M., Ed.; Academic Press: London, 1984; Vol. 1, 163–236.
9. Yosii, R. Snow collembola of the siachen glacier in Karakoram. Results of the Kyoto University Scientific Expedition to the Karakoram and Hindukush **1966**, *8*, 407–410.
10. Suhardjono, Y.R.; Greenslade, P. *Folsomides arnoldi* n.s.p. (Isotomidae): a new collembolan abundant in arid Australia, with a redescription of *Folsomides denisi* (Womersley) In Proceedings of the Linnean Society of New South Wales, 1994; Vol. 114, 21–27.
11. McMeechan, F.K.; Manica, A.; Foster, W.A. Rhythms of activity and foraging in the intertidal insect *Anurida maritima*: coping with the tide. *J. Mar. Biol. Assoc.* **2000**, *80*, 189–190.
12. Palacios-Vargas, J.G.; Gonzalez, V. Two new species of *Deuterosminthurus* (Bourletiellidae), epiphytic collembola from the neotropical region with a key for the American species. *Fla. Entomol.* **1995**, *78*, 286–294.
13. D'Haese, C. Is psammophily an evolutionary dead end? A phylogenetic test in the Genus *Willemia* (Collembola: Hypogastruridae). *Cladistics* **2000**, *16*, 255–273.
14. Deharveng, L.; Bedos, A.; Leksawasdi, P. Diversity in tropical forest soils: The Collembola of Doi Inthanon (Thailand). In *Third International Seminar on Apterygota*; Dallai, R., Ed.; University of Siena: Siena, 1989; 317–328.
15. Lauga-Reyrel, F.; De Conchat, M. Diversity within the collembola community in fragmented coppice forests in South-Western France. *Eur. J. Soil Biol.* **1999**, *35*, 177–187.
16. Hagvar, S. Navigation and behaviour of four Collembola species migrating on the snow surface. *Pedobiologia* **2000**, *44*, 221–223.
17. Axelsen, J.A.; Kristensen, K.T. Collembola and mites in plots fertilised with different types of green manure. *Pedobiologia* **2000**, *44*, 556–566.
18. Stork, N.E.; Blackburn, T.M. Abundance, body size and biomass of arthropods in tropical forest. *Oikos* **1993**, *67*, 483–489.
19. Kubiena, W.L. *The Soils of Europe*; Thomas Murby: London, 1953.

BIBLIOGRAPHY

1. Hopkin, S.P. *Biology of the Springtails (Insecta : Collembola)*; Oxford University Press: Oxford, 1997.
2. Jordana, R.; Arbea, J.I.; Simón, C.; Lucíañez, M.J. Collembola, poduromorpha. In *Fauna Ibérica*; Ramos, M.A., Ed.; Museo Nacional de Ciencias Naturales, CSIC: Madrid, 1997; Vol. 8.
3. Christiansen, K.; Bellinger, P. *The Collembola of North America, North of the Rio Grande*, 2nd Ed.; Grinnell College: Grinnell, 1998.
4. Fjellberg, A. *The Collembola of Fennoscandia and Denmark Part I. Poduromorpha*; Fauna Entomologica Scandinavica: Brill, 1998; Vol. 35.
5. Bretfeld, G. Synopses on palaearctic collembola: Symphypleona. *Abhandlungen und Berichte des Naturkundemuseums Görlitz* **1999**, *71*, 1–318.
6. Potapov, M. Synopses on palaearctic collembola: Isotomidae. *Abhandlungen und Berichte des Naturkundemuseums Görlitz* **2001**, *73* (2), 1–603.
7. Thibaud, J.M.; Schulz, H.J.; da Gama Assalino, M.M. Synopses on palaearctic collembola: Hypogastruridae. *Abhandlungen und Berichte des Naturkundemuseums Görlitz* **2004**, *75* (2), 1–287.
8. Deharveng, L. Recent advances in collembola systematics. *Pedobiologia* **2004**, *48*, 415–433.
9. Fountain, M.T.; Hopkin, S.P. *Folsomia candida* (Collembola): A 'standard' soil arthropod. *Annu. Rev. Entomol.* **2005**, *50*, 201–222.

Color

Robin N. Thwaites

*School of Natural Resource Sciences, Queensland University of Technology,
Brisbane, Queensland, Australia*

Abstract

Soil color, defined through a standard system, plays an important role in all major soil classification systems and for all means of soil interpretation. Although soil color per se has no known direct effect on soil behavior (except for heat absorption at the surface), it provides a valuable indication of soil conditions, composition, and genesis. It can indicate the conditions of aeration (therefore drainage), organic matter content, fertility, and degree of weathering. Color is also a major criterion to differentiate horizons within a soil as a means to its classification and interpretation.

INTRODUCTION

Soil color is one of the most evident characteristics observed on viewing a soil exposure. It is often used in lay terms as a descriptor for any soil type, e.g., “red soils,” “black soil,” and “mottled soils,” and it is always recorded as part of a soil description for inventory purposes. The pioneering Russian school of pedology noted variations in soil color before the turn of the 19th century, but it was not until the end of the 1920s before a satisfactory attempt had been made in the United States to standardize colors of dry soil samples.^[1] Soil color, defined through a standard system, plays an important role in all major soil classification systems and for all means of soil interpretation. Although soil color per se has no known direct effect on soil behavior (except for heat absorption at the surface), it provides a valuable indication of soil conditions, composition and genesis. It can indicate the conditions of aeration (therefore drainage), organic matter content, fertility, and degree of weathering. Various minerals at different stages of weathering, particularly the iron (Fe) oxides, lend their color to the soil matrix and to individual soil grains. Color is also a major criterion to differentiate horizons within a soil as a means to its classification and interpretation. It can occur as whole-colored matrix of the soil, as mottles (blotches and specks), in marbled or streaked patterns, as precipitates around mineral grains (lining root holes), or as nodules and other segregations.

MEASURING SOIL COLOR

Soil colors vary from very pale blues and greens and grays, even white, through browns, yellows, oranges, and reds, to dark forms of these as well as completely black. There is a general perception that soil color cannot be measured with any precision or accuracy, but color has been recorded

consistently and assiduously in both the field and the laboratory for many years. The routine determination of soil color in the field is the less precise and less accurate form of measurement, although consistent results can be achieved with experience. It is determined through visual comparison of a small, fresh soil sample with color chips in standard color charts. The most familiar and widely used are the Munsell Soil Color Charts.^[2] The Munsell color scheme is an objective three-coordinate system that defines color through its *hue* (shade), *value* (lightness), and *chroma* (intensity/saturation; Fig. 1). For soil colors, the Munsell chart ranges in value from 0 to 10, in chroma from 0 to 8, and in hue (in units of principal colors—which are primarily red and yellow for soils) as 10R (red), 2.5YR (yellow–red), 5YR, 7.5YR, 10YR, 2.5Y (yellow), and 5Y. Red soil colors are usually 5YR or redder (with chromas >1), and yellow soils are hues 7.5YR or yellower. Dark soil colors have values <3 and chromas <2 for any hue. Some soil colors have been recorded outside this standard range (e.g., 5R in Central Australia), but these occurrences are rare. Some “gley” hues (5Y, 5GY, 5G, 5BG, and 5B) are also included in the standard charts to account for pale yellow, green, and blue colors from hydromorphic, or waterlogged, soils with reduced ferrous Fe conditions. Color matching is usually undertaken on moist, or moistened, soil samples, although dry soil colors are needed for some classification purposes. Moistening a soil can darken the value by up to three units and up to two units in chroma.^[3]

Soil colors measured this way rarely match with the standard color system perfectly, and training is recommended to perform consistent and accurate color matching.^[4] The factors that most affect visual perception of soil color are as follows: 1) the light source characteristics illuminating the sample; 2) the surface characteristics of the sample; and 3) the variable spectral response characteristics of the human eye.^[5] Even among professional soil scientists agreement on the same color “chip” in the

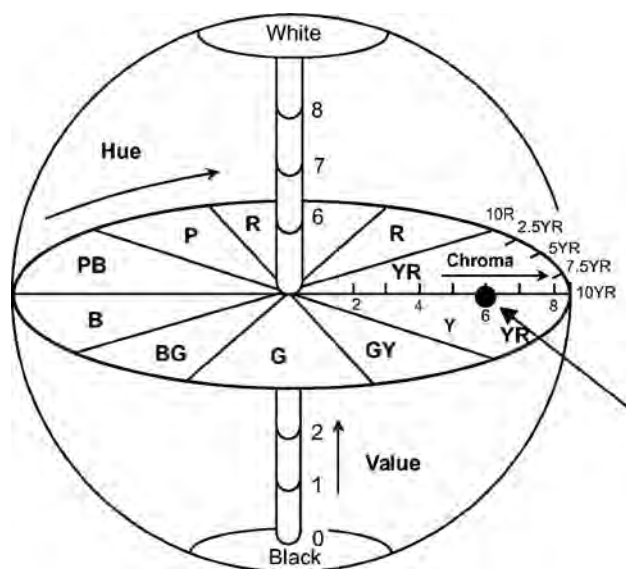


Fig. 1 The Munsell 3-D color concept. The hue and chroma ranges for a value of five are shown. Thus, the pointed arrow represents a color of 10YR 5/6.

Munsell chart for all three-color components is around 50% for a variety of soil color samples, although this rises to around 70% for individual color components.^[4]

In the laboratory, soil color can be ascertained best by spectral reflectance from prepared, disturbed soil samples via a diffuse reflectance spectrophotometer. The reflectance data provided by this process can be converted to a three-figure “tristimulus” value that defines the visually perceived color. This, in turn, can then be converted to a standard color system such as the Munsell notation.^[6] Photoelectric colorimeters can also be used, but results are deemed to be less accurate.^[6]

WHAT CAUSES SOIL COLORS?

Variations in soil color may be due to soil forming processes or inherited from the parent material. It was

recognized in the earliest known study devoted to soil color in 1911^[7] that the color of soil depends on the content of organic matter and the type of Fe oxides. Several Fe oxide mineral types can be present in soil, and they each possess distinctive colors (Table 1).^[8] The type of Fe mineral present is influenced by the parent material and the immediate soil environment. The type and amount of organic matter will influence the darkness of the topsoil. It must also be realized that some soil colors may be relics from prior climates and landscape processes that do not affect the soil.

There is a broad relationship between the darker color of surface soils and organic matter content. However, the relative darkness does not reflect absolute levels of organic matter content. Higher levels of humification will produce darker colors as it will increase wetness: The near black color of some peats and mucks is due to intense humification. The dark color of many black soils (e.g., dark clays with high calcium carbonate contents) may not be attributable to high levels of organic matter but rather due to the clay mineralogy and stage of humification.^[9] The nature of the organic compounds will usually be clay–Fe–(manganese)–organic matter complexes. Intense black coloration, often as mottles, is due to the presence of manganese oxides.

The bright brown, red, and yellow colors of many subsoils are caused by oxidized (often hydrated) Fe products, Fe(III) oxides (or “Fe oxides”), in the fine clay-sized fraction. Contrary to common belief, it is not the *amount* of Fe oxides in the soil that determines the nature of the color but the *type* and *nature* of Fe oxides. The Fe mineral responsible for most non-organic soil coloration is goethite (α -FeO–OH), which imparts colors from reddish-brown through brown to yellow with increasing hydration. Goethite, the most common soil Fe oxide, gives Munsell hues of 10YR–7.5YR if it is finely dispersed; variations in this color come from variations in crystal size and cementation.^[8] The cementation of goethite into nodules and ferricretes darkens the appearance. The strong red colors of tropical and subtropical soils under aerobic conditions are due to the dominance of hematite (α -Fe₂O₃). This mineral

Table 1 The soil colors and soil environments associated with the common Fe oxides.

Fe oxide: formula	Munsell color range	Soil environment
Goethite: α -FeOOH	7.5YR–2.5YR (brown–reddish yellow–reddish brown–dark red)	Weathering of all soils in all regions with Fe release
Lepidocrocite: γ -FeOOH	5YR–7.5YR, value ≥ 6 (light brown–reddish yellow–light reddish brown–pink)	Non-calcareous, aerobic/anaerobic conditions in temperate regions
Hematite: α -Fe ₂ O ₃	7.5YR–5YR (brown–reddish yellow–yellowish red–reddish brown)	Aerobic conditions of the tropics and subtropics. High Fe release rate, high soil temperature, rapid biomass turnover, low leaching activity
Maghemite: γ -Fe ₂ O ₃	2.5YR–5YR (red–reddish brown–reddish yellow–yellowish red)	Tropical and subtropical regions. May be a product of fires
Ferrihydrite: Fe ₅ HO ₈ · 4H ₂ O	5YR–7.5YR, value ≤ 6 (reddish brown–yellowish red–brown–dark brown)	Rapid oxidation in humic environments in gleys and Podzols (Spodosols)

Source: Adapted from Schwertmann.^[8]

gives Munsell hues of 5YR–5R, but, again, variations are due to crystal size: larger hematites, in the micrometer range, tend to purple colors and may even reach Munsell RP hue.^[8] Other Fe oxides that may impart distinctive colors to the soil are as follows: 1) lepidocrocite (γ -FeO–OH), which gives dull yellow to orange mottle colors in anaerobic/aerobic, gleyed conditions; 2) ferrihydrite ($\text{Fe}_5\text{HO}_8 \cdot 4\text{H}_2\text{O}$) giving rusty red colors to specks, mottles, and root holes in otherwise wet or waterlogged (anaerobic) conditions, including the reddish tinge to the Podzol (spodic) B horizon (Bs and Bhs); and 3) maghemite (γ -Fe₂O₃), often the result of fire affecting the soil transforming goethite, produces colors between those of goethite and hematite.^[8]

Other Fe compounds impart distinctive colors to subsoils. A special case is that of jarosite [$\text{KFe}_3(\text{OH})_6(\text{SO}_4)_2$] which occurs commonly as yellow (5Y) mottles in acid sulfate soils. Reduced Fe compounds (through anaerobic, often waterlogged, conditions) lack the red and yellow pigmentation. The soil matrix under these conditions attains a range of shades between gray, green, and blue, as the Fe oxides are completely removed after mobilization by microbial reduction. These “gley” colors form in ferriferrous clay minerals through partial reduction of structural Fe.^[8] It is possible that some of the greenish tinge is also due to the presence of the so-called “green rusts,” which are a group of Fe(II, III) hydroxy salts. If this is the case, then the minerals lose their color rapidly upon exposure to air.^[8]

In some cases, subsoil colors may be inherited from distinctively colored parent materials, e.g., red–brown soils from hematite-rich red sandstone, green tinges from glauconite-rich substrates, red soils from pre-weathered volcanic ash deposits, and sooty black soils from manganese-rich, ore-bearing substrates.

SOIL CONDITIONS INDICATED BY COLOR

Dark brown colors near the surface are associated with high levels of organic matter, thus higher than average nutrient fertility.

Bright yellows and reddish colors in the subsoil signal the presence of Fe(III) oxides, therefore indicating good drainage and aeration conditions. Red soils are usually better aerated/drained than yellow soils. The strong surface charge of these Fe oxides also contributes to the good aggregation properties of the soil.^[9]

Very pale gray to white colors, particularly in earthy fabric topsoils, indicate significant leaching and low organic matter. Fertility status is, therefore, low. A distinctively paler sub-topsoil horizon, characterized by leaching, is known as an E (eluviated), or A₂, horizon and is distinctive in Podzols (Spodosols).

Orange or dull yellow mottles (lepidocrocite) in gray, bluish, or greenish (reduced) ferriferrous clay matrix (gley) signify prolonged anaerobic conditions, permanent or seasonal waterlogging.

Table 2 Simple indications given by some soil color characteristics and examples of possible soil types, for illustrative purposes only.

Color characteristic	Indication	Soil type example
Brown, whole-colored profile	Well drained, moderately weathered, high organic matter, high fertility	ST: Haplustoll; FAO: Chernozem
Red, whole-colored profile	Well drained, highly leached, intensely weathered (high proportion of sesquioxides), low fertility	ST: Durargid, Humox; FAO: Ferralsol
Yellow, whole-colored profile	Moderately drained, moderately leached, moderately weathered (goethite dominant), moderate fertility	ST: Haplustalf, Paleustult; FAO: Xanthic Ferralsol, Ferric Luvisol
Black, whole-colored profile	Moderately to imperfectly drained, little weathered, little leached, high organic carbon, high fertility	ST: Ustert, Haploxeroll; FAO: Pellic Vertisol, Rendzina
Black topsoil, whole-colored	Imperfectly to poorly drained, high organic matter, unweathered, unleached, high fertility	ST: Umbrept; FAO: Humic Cambisol, Humic Podzol
Gray topsoil, whole-colored	Well drained, leached, low organic carbon, low fertility	ST: Palexeralf; FAO: Dystric Cambisol
Pale-white subsoil	Intensely leached, no organic carbon and low cations, very low fertility	ST: Haplorthod; FAO: Podzol, Albic Luvisol
Yellow subsoil, whole-colored	Moderately drained, moderately weathered, high Fe content (probably goethite), moderate fertility	ST: Paleustalfs; FAO: Luvisol, Acrisol
Green–gray mottled subsoil, Green–blue mottled subsoil	Poorly drained, often saturated by groundwater, anaerobic/reducing conditions (Fe(II) oxides), poor fertility	ST: Aquult; FAO: Gleyic Acrisol, Gleysol
Red–yellow mottled subsoil	Imperfectly drained, fluctuating water status (may be relict), regular oxidizing of wet soils (Fe(III) oxides), mod-poor fertility	ST: Paleustult, Palexeralf; FAO: Plinthic Luvisol, Chromic Luvisol
Orange–rusty mottled topsoil	Regular surface waterlogging and draining of surface, oxidizing mineral matter, high organic matter, mod-high fertility	ST: Pellustert; FAO: Chromic Vertisol

Note: ST, soil taxonomy; FAO, FAO/UNESCO soil map of the world.

Black matrix material or mottles in wet conditions with a hydrogen sulfide (rotten eggs) odor (the anaerobic decay of organic matter) indicate severe permanent waterlogging to the surface (either perched or from groundwater).^[9]

Red, orange, and yellow mottles in pale (sometimes colored) matrix, often in a marbled pattern, may well be relict features of previous weathering environments associated with fluctuating water tables. Such features are common to the so-called “lateritized” soil profiles on older geomorphic surfaces where existing weathering environments are not responsible for the observed, dominant color patterns.

Other indications are given in [Table 2](#).

REFERENCES

1. Simonson, R.W. Soil color standards and terms for field use—History of their development. In *Soil Color*; Bigham, J.M., Ciolkosz, E.J., Eds.; SSSA: Madison, 1993; 1–20.
2. Munsell Color Company. *Munsell Soil Color Charts*; Special Publication 31; Munsell Color Company: Baltimore, 1975.
3. Hunt, C.B. *Geology of Soils. Their Evolution, Classification, and Uses*; W.H. Freeman: San Francisco, 1972.
4. Post, D.F.; Bryant, R.B.; Batchily, A.K.; Huete, A.R.; Levine, S.J.; Mays, M.D.; Escadafal, R. Correlations between field and laboratory measurements of soil color. In *Soil Color*; Bigham, J.M., Ciolkosz, E.J., Eds.; Special Publication 31; SSSA: Madison, 1993; 35–49.
5. Melville, M.D.; Atkinson, G. Soil color: Its measurement and its designation in models of uniform color space. *J. Soil Sci.* **1985**, *36*, 495–512.
6. Torrent, J.; Barrón, V. Laboratory measurement of soil color: Theory and practice. In *Soil Color*; Bigham, J.M., Ciolkosz, E.J., Eds.; Special Publication 31; SSSA: Madison, 1993; 21–33.
7. Robinson, W.O.; McCaughey, W.J. *The Color of Soils*; U.S. Government. Printing Office: Washington, 1911.
8. Schwertmann, U. Relations between iron oxides, soil color, and soil formation. In *Soil Color*; Bigham, J.M., Ciolkosz, E.J., Eds.; SSSA: Madison, 1993; 51–69.
9. Fitzpatrick, R.W.; McKenzie, N.; Maschmedt, D.J. Soil morphological indicators and their importance to soil fertility. In *Soil Analysis. An Interpretation Manual*; Peverill, K.I., Sparrow, L.A., Reuter, D.J., Eds.; CSIRO: Collingwood; 55–69.

Compaction

William E. Wolfe

Department of Civil and Environmental Engineering and Geodetic Sciences, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Compaction is perhaps the oldest technique for enhancing the engineering properties of a soil mass. Experience in compacting a large number of soil types has shown that, in many soils, adequate engineering properties can be expected, if a minimum density that can be related to the laboratory Proctor density is achieved. This knowledge allows the geotechnical engineer to reduce the testing that must be done to establish the performance of compacted soil profiles.

INTRODUCTION

A determination of the behavior of a soil mass subjected to structural loads is performed by a geotechnical engineer who evaluates soil properties and makes design recommendations. Satisfactory performance is usually achieved when ground movements are small enough that the structure placed on the prepared soil behaves the way the owners want it to. This is accomplished by specifying minimum performance requirements. Often, on-site soils are not ideally suited for the proposed project, and the specified performance cannot be achieved unless the soil's properties are modified.

Modification can be accomplished in a number of ways. The two most commonly used classes of modification are chemical stabilization and mechanical stabilization. Chemical methods involve the addition of a stabilizing agent, usually lime or cement, to the soil. The chemically modified soil has a higher strength and lower compressibility than the untreated soil. Chemical additives have also been effective in reducing the swelling potential in clays. A mechanically modified soil has had its properties improved either by the inclusion of foreign material (usually reinforcing elements) or by densification. Densification is most commonly achieved through preloading or compaction. The two methods differ in the means employed to achieve the necessary increase in density. Preloading is used primarily to force water out of a saturated soil. Because preloading causes a reduction in volume by forcing water to flow away from areas of load, preloading a site requires the imposition of large loads over extended periods of time. The principles governing the rate and quantity of volume decrease accomplished by preloading are discussed in the section on consolidation.

In the compaction process, the volume of void space is reduced by applying high loads over small areas to force the

air out of an unsaturated soil mass. Because the area covered by the compaction load is much smaller than the one used to preload a site, the volume of soil stabilized during each application of the compaction load is much less. So, to be successful, the treatment must be moved from place to place and applied to thin layers. The most common method involves driving heavy vehicles, a specified number of times (passes) over the soil surface, as the soil is being placed. Typically, each 8–12 in. (20–30 cm) thick layer of the soil is compacted and the amount of compaction measured. Since the procedure specifically targets the air fraction of the soil mass, the water content is not affected by compaction.^[1]

COMPACTION PRINCIPLES

Although compaction may not change the water content of a soil, the maximum density obtainable does depend on the amount of water present in the soil mass. Although the engineering properties of all soils are affected by the soil density, the response of a coarse-grained soil to compaction is different and needs to be considered separately from the response of a fine-grained soil.

Coarse-grained soil—because the individual particles in a coarse-grained soil are chemically inert, there is little interaction between the solid and liquid fractions of the soil mass. The effect of water in the deposit is largely through the formation of capillary tensions, which result in the particles being tightly bound in a matrix that resists rearrangement. As the degree of saturation increases, the capillary forces are destroyed, and compaction energy becomes more effective in densifying the soil. An example of a compaction curve obtained for a coarse-grained soil is presented as [Fig. 1](#). Vibratory loading has been found to be the most effective method for densifying coarse-grained soils.^[2,3]

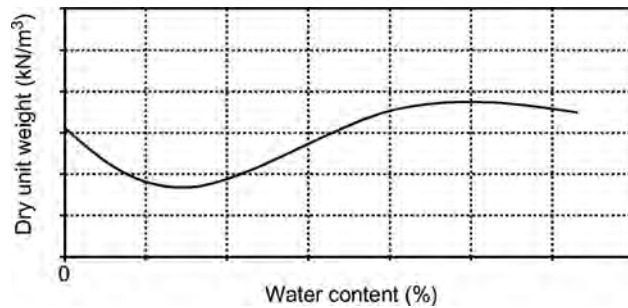


Fig. 1 Typical compaction curve for a coarse-grained soil.

Fine-grained soil—compaction tests performed on fine-grained soil samples have clearly shown that for every soil and compactive effort, an optimum water content yielding the greatest density of the solid fraction (dry density) can be identified. At water contents both above and below the optimum, the same compactive effort results in lower soil densities. Typical curves showing the dependence of density on water content and compactive effort are presented in Fig. 2.

Although the density decreases above and below the optimum water content, the behavior of the soil is much different on the two sides of the optimum condition. On the dry side of optimum, the electrical charges on the sheet-like solid particles encourage an edge to face orientation of the individual soil grains with the resultant structure looking something like a honeycomb. This particle arrangement results in a fairly open structure, but one in which the high negative pressures in the water resulting from capillary tension cause the individual particles to be tightly bound together. As water content increases, the capillary tensions decrease. This decrease in capillarity reduces the force holding grains together and increases the effectiveness of the compactive energy. The particles are forced into more orderly patterns. As the water content continues to increase, the air volume fraction becomes small enough that air voids are discontinuous and so air cannot be easily expelled. The maximum density is quickly reached, and further increases in water content merely increase the water adsorbed onto the solid grains reducing contact strength and density.^[3]

TESTING

It is the job of the geotechnical engineer to determine if the soil, whether on-site or imported, is strong or stiff enough to support the design loads. The most direct measure of the ability of a soil to perform as designed would be to measure the strength, stiffness, and/or hydraulic conductivity at many locations in the field. In most cases, field testing of these important engineering properties is not practical, and an indirect measure of how the soil will behave when loaded is appropriate.

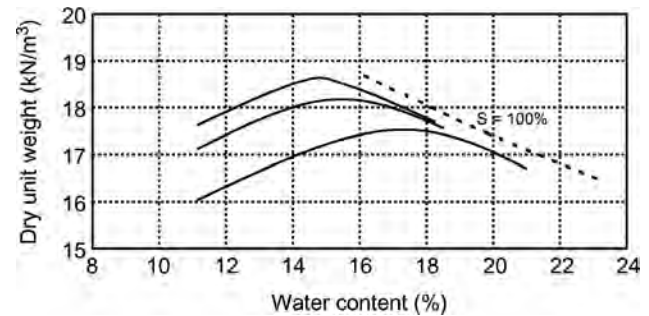


Fig. 2 Typical compaction curves for a fine-grained soil.

Density and moisture content are the most common substitutes for the actual soil properties upon which design was based.

Laboratory Tests

The actual combination of maximum dry density and optimum moisture content is different for each soil. Therefore, a laboratory obtained moisture content vs. density relationship such as one of those shown in Fig. 2 should be developed for each different soil or mixture of soils to be used on the site. A moisture-density curve is generated by plotting the results of five separate compaction tests performed on different samples of the same soil at five different moisture contents. Several test methods have been standardized by the American Society for Testing and Materials (ASTM).^[4] The most commonly used are referred to as the Standard Proctor Test (ASTM D-698) and the Modified Proctor Test (ASTM D-1557), after R.R. Proctor, who developed the procedures for conducting repeatable compaction tests in the 1920s and 1930s.

Ideally, the engineer then performs the appropriate engineering tests on laboratory samples compacted to the optimum moisture content and density. Once the data showing a correlation between soil density and its strength and stiffness have been generated for each soil type, the density and water content become surrogates for the true design parameters. Because these two parameters are easy to determine and can be related to the significant engineering properties, the measurement of density and moisture content has become the preferred method for verifying adequate preparation of the soil.

Field Tests

With the laboratory data obtained as described above, field verification of the appropriate density and moisture content is the extent of the field testing necessary to be confident that the required strength and stiffness have been achieved. The two methods most commonly used to check density are the sand cone method (ASTM D-1556) and the nuclear gauge (ASTM D-2922).



Fig. 3 Troxler Model 3450 moisture/density gauge in the direct transmission mode.

Sand Cone

In this method, a test hole is excavated in the soil whose density is to be determined and all the excavated material weighed. The hole is filled with a clean, uniform sand of a known density, and the amount of sand used to fill the hole is determined. A fraction of the sample of the soil from the hole is dried in an oven at 105°C to determine the water content. The field density and water content can then be compared with the optimum conditions obtained in the laboratory.

Nuclear Gauge

The principle behind the use of the nuclear gauge is that measuring the attenuation of gamma radiation can accurately determine the total density and water content of a near surface soil. In the direct transmission method, a radiation source is placed in a predrilled hole, usually about 300 mm below the surface. A gamma radiation detector is placed on the ground surface. In the backscatter method, both the radiation source and the detector are

placed on the ground surface.^[5] Fig. 3 depicts a nuclear gauge (Troxler Model 3450) measuring the density of clay material in the direct transmission mode. The nuclear gauge has become a very popular way to make density and water content determinations because it is much faster than other methods, less prone to operator error and, in the backscatter mode, is non-destructive, since no excavation needs to be made. However, since the method is an indirect measure of density, the instrument must be calibrated at the start of each day's work, and the calibration must be kept in the permanent record for the gauge. In addition, the precision of even properly calibrated gauges may be adversely affected by a number of elements in the soil including organic matter. The U.S. Army Corps of Engineers^[6] specifies that readings from a nuclear gauge must be frequently compared with direct measurements for density (sand cone) and water content (conventional oven).

CONCLUSION

Compaction is perhaps the oldest technique for enhancing the engineering properties of a soil mass. Experience in compacting a large number of soil types has shown that, in many soils, adequate engineering properties can be expected, if a minimum density that can be related to the laboratory Proctor density is achieved. This knowledge allows the geotechnical engineer to reduce the testing that must be done to establish the performance of compacted soil profiles.

REFERENCES

1. Hilf, J.W. Compacted fill. In *Foundation Engineering Handbook*; Fang, H.Y., Ed.; 2nd Ed.; Van Nostrand Reinhold Company: New York, 1991; 249–316.
2. Schroeder, W.L.; Dickenson, S.E. *Soils in Construction*, 4th Ed.; Prentice Hall: Englewood Cliffs, 1996.
3. Lambe, T.W.; Whitman, R.V. *Soil Mechanics*; John Wiley and Sons: New York, 1969.
4. American Society for Testing and Materials ASTM Standards, Book of Standards Volume 04.08, Soil and Rock (I): D420–D5779, Mar 2000.
5. Department of the Army, US Army Corps of Engineers, Engineer Manual 1101-2-1906, Appendix VI, Compaction Tests, Aug 20, 1986.
6. Department of the Army, US Army Corps of Engineers, Engineer Manual 1110-2-1911, Construction Control for Earth and Rock-Fill Dams, Sept 30, 1995.

Conditioning Index

Ted M. Zobeck

*U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS)
(Retired), Lubbock, Texas, U.S.A.*

M. Lee Norfleet

*Natural Resources Conservation Service, U.S. Department of Agriculture
(USDA-NRCS), Temple, Texas, U.S.A.*

Douglas L. Karlen

*National Laboratory for Agriculture and the Environment, U.S. Department
of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.*

Abstract

Soils are a critical natural resource for the future development and sustainability of humankind. Natural resource assessment tools are often used to evaluate the effects of management on soil properties and processes to ensure the sustainable use of our limited soil resources. The soil conditioning index (SCI) is a soil assessment tool used to predict the consequences of management actions on the state of soil organic matter. Although SCI is correlated with particulate (labile) organic matter and residue returned to the soil, it is not a precise index. Calculated SCI values may vary by about ± 0.2 to ± 0.3 . This entry describes the use, determination, and evaluation of the SCI on agricultural lands in the United States.

INTRODUCTION

Soils are a critical natural resource for the future development and sustainability of humankind. Not only soils are used to produce our food and fiber, but also they serve as critical environmental filters for waste treatment and recycling and act as important sources and sinks for nutrients, gases, and other materials essential for proper functioning of life processes. Natural resource assessment tools are often used to evaluate the effects of management on soil properties and processes to ensure the sustainable use of our limited soil resources.

SOIL ORGANIC MATTER (SOM)

SOM is an important constituent in the soil surface and provides several important functions. Strictly speaking, organic matter is any material derived from living organisms. Organic matter in soil also originates from biological material and is composed chiefly of molecules containing carbon (C) from plants and a variety of other soil organisms such as arthropods, microbes, and others. As these living organisms die and decompose, they are transformed by biological, physical, and chemical processes into decomposed organic matter called humus. SOM is important in the formation of stable soil structure that resists wind- and water-induced erosion and promotes increased water infiltration into the soil. SOM improves soil fertility and acts as

a reservoir for soil nutrients by increasing the soil's ability to absorb and hold nutrients and water. SOM is a primary indicator of soil quality and an important factor in C sequestration and global climate change. Knowledge of how land management changes SOM is critical to understand how management affects important soil properties and functions.

SOIL CONDITIONING INDEX (SCI)

The SCI is a soil assessment tool used to predict the consequences of management actions on the state of SOM. The Natural Resources Conservation Service (NRCS), part of the U.S. Department of Agriculture (USDA) has adopted the SCI to evaluate cropland management systems in the United States. The SCI is a dimensionless index that predicts qualitative changes (positive or negative) in SOM in the top 10 cm (4 in.) of soil based on the combined effects of three dimensionless determinants of SOM using the following equation:

$$SCI = (OM \times 0.4) + (FO \times 0.4) + (ER \times 0.2) \quad (1)$$

where OM is the organic material (biomass) from animal or plant sources produced and returned to the soil, FO represents field operations including tillage and other cultural practices that stimulate organic matter breakdown and decomposition, and ER corresponds to the influences of wind and water erosion.^[1] Both OM and FO each account

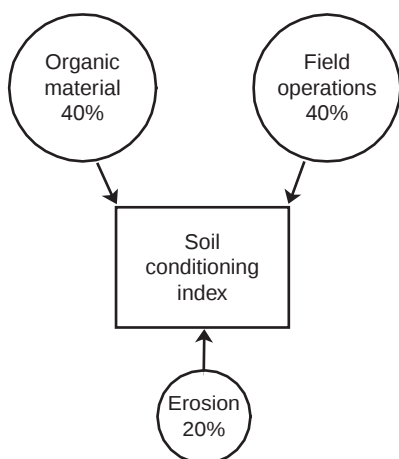


Fig. 1 The SCI considers OM input, field operations and other soil disturbance, and wind and water erosion.

for 40% of the final SCI value (total of 80% combined), and wind and water erosion represent 20% (Fig. 1). The SCI assumes that field operations reduce SOM by stimulating decomposition and that maintaining organic residues with no or very limited tillage will maintain and increase soil organic levels.

THE USE OF SCI

The SCI is an important soil management index and is required by several USDA–NRCS soil or crop management guidelines, including the Conservation Crop Rotation guideline, and as an additional criterion in the Residue and Tillage Management—No Till/Strip Till/Direct Seed guidelines, and is specified for use in the Conservation Security Program of the Farm Security and Rural Investment Act of 2002. The USDA–NRCS uses SCI to develop sustainable conservation plans at the field, watershed, and larger scales.

DEVELOPMENT OF THE SCI

The SCI was originally developed based on research conducted from 1948 to 1959 in a humid region with high clay soils at Renner, Dallas, Texas, U.S.A. The SCI was first published in 1964 as a soil rating system for soils of the Western United States as USDA Conservation Agronomy Technical Note No. 27. This note was revised in 1967 and 1974, and a shorter version of this rating system was later prepared for use by the USDA Soil Conservation Service, Southern National Technical Center.^[2] Further modification of the concept was provided using data from Iowa and Montana.^[1,2] The USDA Soil Quality Institute made final adjustments to determine the amount of residue needed to maintain SOM at constant levels in 2000. The SCI was incorporated into the Revised Universal Soil Loss Equation

(RUSLE2), a computer model to estimate water erosion on cropland, in 2003.^[1]

The SCI value is usually determined by applying Eq. 1 to specific field locations using RUSLE2, which uses information on the local climate, soils, crop, and tillage to estimate annual water erosion and the SCI value. Since wind and water erosion are both included in the ER subfactor of Eq. 1, wind erosion may also need to be determined if it occurs. However, wind erosion is not determined by RUSLE2 and must be provided by another method. Wind erosion is estimated using an MS Excel spreadsheet program,^[3] based on the USDA Wind Erosion Equation (WEQ). Where it is a factor, the estimated wind erosion total is manually entered into RUSLE2 before making the final determination of SCI.

SCI TESTING

The SCI was tested by USDA–NRCS staff scientists using data from 36 long-term crop production research sites to determine how well it could detect the trend in organic matter content. This study showed that SCI could identify the general trend in organic matter accumulation or depletion in 97% of the tests. Fig. 2 shows how the SCI varied with the annual amount of change in organic matter for each test. When the SCI was less than 0, the annual change in SOM was negative. The annual change in SOM was positive when the SCI was greater than 0. Another evaluation of SCI using nine long-term cropping studies also found that positive trends in SOM followed positive trends in SCI, and negative SCI trends were associated with negative SOM trends.^[4] Correlations between SOM and SCI were improved when data were separated by states.

A test of SCI on 52 fields in the Southern High Plains of western Texas successfully associated the conservation grasslands, native rangelands, and no-tillage, limited (minimum) tillage, and high-residue croplands with positive SCI values and the conventionally tilled fields with negative SCI values.^[5] The conservation grasslands had the

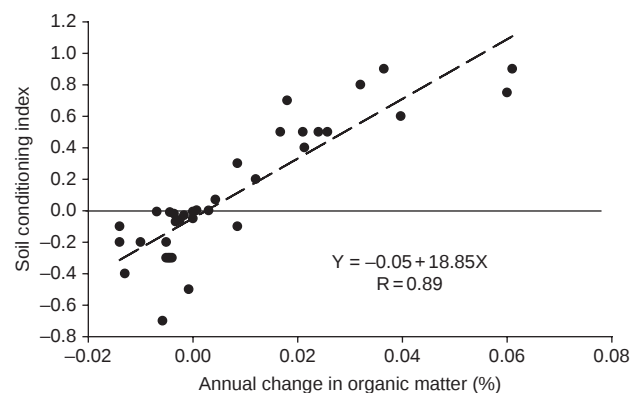


Fig. 2 Comparison of SCI with percent annual change in organic matter for 36 long-term tillage study sites.

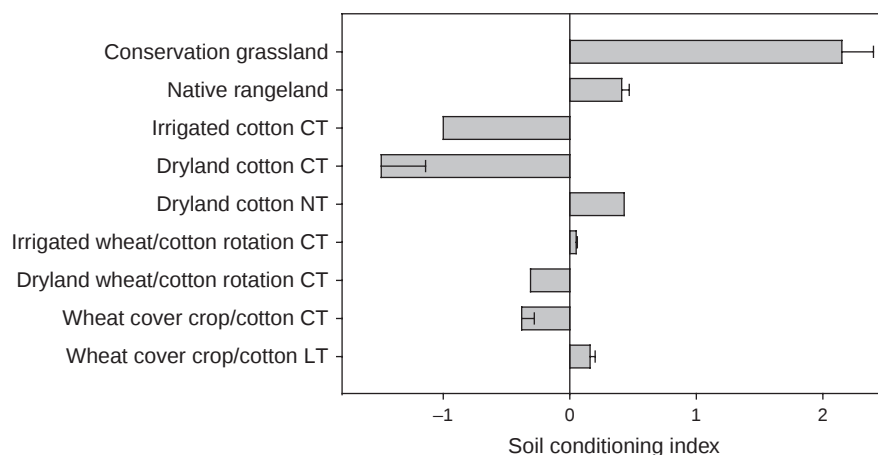


Fig. 3 SCI by agroecosystem. CT = conventional tillage, NT = no tillage, LT = limited tillage. Error bars are standard errors.

highest SCI value, and the conventionally tilled dryland cotton had the most negative SCI value. The SCI values by agroecosystem tested are shown in Fig. 3. Rather, subtle changes in soil management were detected by using RUSLE2 and a spreadsheet version of the WEQ to determine the SCI.^[5] Although the stated purpose of the SCI is to predict the consequences of management actions on the state of SOM, the SCI values were not strongly correlated with total SOM. The SCI values were more strongly associated with a specific and more labile form of soil organic C called particulate organic matter C. The SCI was even more strongly correlated with a measure of residue production, which serves to add organic matter to the soil and protect the soil from the forces of erosion. This study suggests that although SCI is a good estimate of organic matter, it is not a precise index. It may be advisable to have a buffer of ± 0.2 or ± 0.3 when assigning SCI values. This buffer may be particularly necessary in Western states where the OM subfactor in SCI may often be less than 0, even in situations with adequate cover.

A more comprehensive soil quality assessment tool evaluating biological, chemical, and physical indicators is offered by the Soil Management Assessment Framework (SMAF). This tool provides a consistent approach for evaluating and, if desired, combining various soil biological, chemical, and physical indicator measurements for an overall assessment of dynamic soil quality.^[6] The SMAF was used to synthesize information from cropping system comparisons throughout the U.S. Great Plains.^[7]

CONCLUSION

The SCI is a soil assessment tool used to predict the consequences of management actions on SOM trends. The SCI is determined using wind and water erosion models. The information needed to run the models includes field location, type of crops grown, yields, tillage, and other management information. Much of the information needed to run the models is available in public databases. Although

SCI is correlated with particulate (labile) organic matter and residue returned to the soil, it is not a precise index. Calculated SCI values may vary about ± 0.2 to ± 0.3 .

REFERENCES

1. NRCS (Natural Resources Conservation Service). *Interpreting the Soil Conditioning Index: A Tool for Measuring Soil Organic Matter Trends*. U.S. Department of Agriculture, Natural Resources Conservation Service, Soil Quality-Agronomy Tech. Note No. 16, 2003. Available at http://soils.usda.gov/sqi/management/files/sq_atn_16.pdf.
2. RCS (Natural Resources Conservation Service). *Soil conditioning index for cropland management systems*. NRCS National Agronomy Manual, Directive No. 190-V-NAM, 2002. Available at http://policy.nrcs.usda.gov/media/pdf/M_190_NAM.pdf.
3. Sporcic, M.; Keep, T.; Nelson, L. *WEQ Management Period Method Wind Erosion Model Worksheet*, 1998. Available at <http://www.nm.nrcs.usda.gov/technical/tech-notes/agro/ag55.xls>.
4. Hubbs, M.D.; Norfleet, M.L.; Lightle, D.T. Making conservation tillage conventional: Building a future on 25 years of research. In *Interpreting the Soil Conditioning Index*, Proceedings of 25th annual southern conservation tillage conference for sustainable agriculture, June, 24–26, 2002; van Santen, E., Ed.; Spec. Rept No. 1; Alabama Agricultural Experiment Station and Auburn University: Auburn, 2002; 192–196.
5. Zobeck, T.M.; Crownover, J.; Dollar, M.; Van Pelt, R.S.; Acosta-Martinez, A.; Bronson, K.F.; Upchurch, D.R. Investigation of soil conditioning index values for southern high plains agroecosystems. *J. Soil Water Conserv.* **2007**, *62* (6), 433–442.
6. Andrews, S.S.; Karlen, D.L.; Mitchell, J.P. A comparison of soil quality indexing methods for vegetable production systems in northern California. *Agr. Ecosyst. Environ.* **2002**, *90*, 25–45.
7. Wienhold, B.J.; Pikul, J.L., Jr.; Liebig, M.A.; Mikha, M.M.; Varvel, G.E.; Doran, J.W.; Andrews, S.S. Cropping system effects on soil quality in the great plains: Synthesis from a regional project. *Renew. Agr. Food Syst* **2006**, *21*, 49–59.

Conservation Tillage

Jack Cline

Ohio State University, Columbus, Ohio, U.S.A.

Robert Hendershot

*U.S. Department of Agriculture—Natural Resources Conservation Service
(USDA-NRCS), Lancaster, Ohio, U.S.A.*

Abstract

Soil erosion is a natural process that has been a major problem for all civilizations since recorded time began. Conservation tillage or crop residue management systems range from no-till or direct-seeding methods to reduced tillage systems that disturb the soil but leave the required amount of crop residue on the soil surface after planting. Conservation tillage can greatly reduce water- and wind-driven soil erosion. In spite of its long history, the science of conservation tillage is contemporary.

INTRODUCTION

History of Conservation Tillage

Conservation tillage has been around since humans first planted a seed by making a hole in the ground or spreading seeds on the surface of the soil. Conservation tillage can greatly reduce water- and wind-driven soil erosion. In spite of its long history, the science of conservation tillage is contemporary. The relationship between plant residues and the forces of soil erosion and soil quality parameters has been studied in depth only during this generation. As used in this entry, conservation tillage is an agronomic production system that leaves at least 30% of the soil surface covered with plant residue after the planting process. Laflen^[1] showed that 30% crop residue cover reduces soil erosion by 60%. This conservation practice can be utilized on both cropland and grazing lands. Protecting the soil from erosion is important for the maintenance of long-term productivity of the soil resource base. Conservation tillage also improves the quality of the soil by potentially increasing the soil organic matter and generally improving the soil quality.

Concept of Conservation Tillage

Conservation tillage or crop residue management systems range from no-till or direct-seeding methods to reduced tillage systems that disturb the soil but leave the required amount of crop residue on the soil surface after planting. Equipment used for no-till planting till only a narrow strip of soil or push aside the residue, so the planting can be completed with good soil-to-seed contact. Forage and

cereal crops can be broadcasted on the soil surface where they rely on the forces of nature such as frost or rain to incorporate the seeds into the soil. Reduced tillage equipment utilizes chisel plows, field cultivators, or disks that incorporate a portion of previous crop residue. The number of trips, speed, depth, and type of point used on the cultivation tool all affect the amount of crop residue incorporated. Sweeps and straight chisel points leave more residue on the surface than twisted chisel points.

SOIL EROSION

Soil erosion is a natural process that has been a major problem for all civilizations since recorded time began. Lowden^[2] had pointed out the tremendous impact soil erosion has had on great civilizations around the world. The loss of soil productivity and the effect the resulting sediment from the eroding fields of both cropland and grazing lands have, in many cases, hastened or even caused the demise of some of these past societies. Modern governments have begun to recognize that many practices lead to the accelerated soil erosion that affects the productivity of the land unit, and the displaced sediment causes many other problems. Many recognize the importance of sustaining the soil resources not only for food and fiber production, but also for the economic savings to communities. Sediment removal is a costly endeavor for many communities and governments in order to maintain drainage ditches, irrigation canals, shipping and boating channels, water storage reservoirs, and lakes for flood control. Sediment also lowers water quality and increases water treatment costs for drinking and other uses. The deposition

of eroded soil interferes with the utilization of the land resources and often requires removal or incorporation into the existing soil.

THE ECONOMICS OF CONSERVATION TILLAGE

Conservation tillage has been a major part of the conservation program in the United States since the 1970s. It is the primary conservation practice used to control soil erosion on cropland. Conservation tillage is also used to control soil erosion on grazing lands when establishing a forage crop or introducing a new forage species to an existing pasture or rangeland. Conservation tillage systems generally require more management and have less opportunity for adjustment in decisions, but they are considered to be the most cost-effective tillage systems due to the often significant reduction in fuel and labor costs as well as investment in machinery.

CONSERVATION TILLAGE EFFECTS ON SOIL QUALITY

Conservation tillage improves soil quality by improving soil structure and soil tilth by increasing the soil organic matter. Tilling the soil oxidizes organic matter and exposes it to erosion. The less tillage performed, the more organic matter will be in the soil profile. Humus is the stable portion of the soil organic matter. Humus aids in binding soil aggregates together and increases the cation exchange capacity. These strong soil aggregates create a porous soil that allows both water and air to move freely. This porous condition is ideal for plant root growth. The increase in soil organic matter increases the water-holding capacity of the soil, increases the water infiltration rate, and decreases runoff.^[3] Soil organisms flourish under conservation tillage systems.^[4] The more plant residue left in the field, the more soil organisms are there in the soil. Bacteria, fungi, algae, protozoa, nematodes, earthworms, and insects are all working in the soil profile breaking down organic matter and providing a biodiverse community. The plant residues provided by conservation tillage feed the soil biology.

SOIL CONSERVATION BY GRASSLANDS

In addition to the many techniques described above for conserving soil and water resources, the use of forages, either as a permanent soil cover or in some rotational system with crops for direct human use, may be the most effective of all techniques for soil and water conservation. Historically and prehistorically, natural grasslands and shrublands have supported large populations of plant eaters (herbivores), which have grazed and/or browsed these resources. These plant eaters have then been the source of food for a variety of carnivores (meat eaters) and omnivores

(mixed diet of plant and meat). Grazing lands include the tundra of the arctic regions, the great temperate region grasslands, and the savannas of the tropical regions. These growing plants convert solar energy by the process of photosynthesis into chemical energy (primarily carbohydrates), and then, some are transformed by complex biochemical processes into lipids, amino acids and proteins, and vitamins, which are the nutrients needed by all multicellular life.

GRASSLAND UTILIZATION

The most abundant carbohydrates in nature are the structural compounds of cellulose and hemicellulose and in mature plants increasing levels of the derived lignin. Higher animals do not digest these compounds, but the anaerobic intestinal microorganisms can degrade cellulose and hemicellulose to usable end products, while lignin, though somewhat degraded, cannot be utilized at all. Herbivores have an enormous population of these anaerobic microorganisms either in the forestomachs (before the true gastric stomach, therefore called pregastric digesters) or in the caecum and colon (therefore called postgastric digesters). These microorganisms produce short-chain fatty acids, gases, and microbial cells. The fatty acids provide 60–80% of the host animal's energy, the gases are waste products, and the microbial cells are digested to amino acids, carbohydrate, fats, minerals, vitamins, or excreted in feces, depending on the pre- or postgastric anatomy. Thus, these structural carbohydrates not digestible by humans and higher animals are converted into usable products like meat, milk, cheese, hides, and wool for human utilization and benefit.

GRASSLAND HISTORIC BENEFITS

In the natural environment, wild herbivores were migratory, and many species traveled as much as 1200–1500 miles on their migratory routes. This meant that grazing pressure was heavy for a relatively short period as the animals passed through, but that extensive periods without grazing allowed plants to recover and maintain high levels of root energy reserves. These grasslands therefore accumulated very significant organic matter in the soils, and most of the high organic matter soils of the world developed under grasslands (which is an important mechanism in sequestering carbon). The notable exception to this statement is a forest growing in swamps or bogs where the leaves and small branches fall into water in an anaerobic or reduced oxygen environment, as opposed to the normal dry aerobic environment where oxidative processes re-release the sequestered carbon with significantly reduced yearly gain. Arctic areas have low oxidation rates for much of the year because of low temperatures and therefore reduced the rates of organic matter degradation.

OVERGRAZING PROBLEMS

Domestication and fencing of land have destroyed migratory patterns in much of these grasslands and rangelands. This is especially true in those areas of sufficient rainfall for relatively high productivity. The tendency therefore has been to graze continually, inevitably resulting in the depletion of root energy reserves, reduced soil organic matter during much of the year, and, therefore, reduced rates of organic matter production. The density of plant cover is reduced, and the time needed for plant growth to recover after grazing is increased. These processes increase the probability and quantity of soil erosion and reduce the efficiency of water infiltration into the soil. Continuous overgrazing is one of the major causes of desertification, and either the reduction of animal concentration or reduction in the length of the grazing period is necessary to prevent overgrazing.

RESEARCH

Long-term research has been conducted at the U.S. Department of Agriculture North Appalachian Experimental Watershed near Coshocton, Ohio, where precipitation, surface water runoff, and sediment loss in that runoff have been measured. In Table 1, which is composed of data from two articles by Owens, Edwards, and Van Keuren,^[5,6] precipitation, surface runoff, and sediment load were measured. Data from an unimproved pasture ungrazed (2 years), grazed only during the grazing season (3 years), and grazed during the growing season and winter hay feeding (6 years) were taken and summarized here. A similar sloped wooded watershed nearby was also sampled for the same measures. Note that by comparing the 3-year data, it is apparent that less surface runoff occurred from unfertilized pasture grazed during the growing season only (203 mm/yr) than from a similar sloped, wooded watershed nearby (261 mm/yr). Also note that for the 6-year data presented by Owens, Edwards, and Van Keuren,^[5,6] there was less annual surface runoff (137 mm/yr) from pasture (grazed and imported winter fed hay) than from the wooded watershed (173 mm/yr). The lower values for runoff recorded in the pasture watershed as compared to the wooded watershed occurred even though precipitation levels in the pasture watershed were higher than that in the wooded

watershed. Also note that the data from the storm of September 13, 1979, recorded a significantly higher sediment load that was carried in surface water runoff from the wooded watershed (1032 kg/ha) as compared to the load from the grazed pasture watershed (739 kg/ha), even though surface runoff from the pasture was 6 mm greater.

MANAGEMENT INTENSIVE GRAZING (MIG)

Subsequent research studied moderately fertilized and heavy fertilized pasture with rotational grazing as well as continual grazing. Many interesting conclusions reached from these studies and others in the eastern and Midwestern areas of the United States have led to the development of what are commonly called MIG systems. MIG systems are described below. The principles followed in MIG are designed to easily meet the animals' nutrient needs and thus their productivity levels while at the same time to keep pasture plants in the immature or boot-prebud stage of growth; when root reserves have been restored, rapid regrowth can be expected. To accomplish this goal, sufficient cells or grazing paddocks are needed, so that animals remain in individual paddocks no more than three days to prevent animals from grazing new regrowth, and so that at least 3–4 in. of leaf remain, although regrowth is rapid. The plant species used are a function of moisture supply, latitude, fertility, etc., but the MIG system increases plant diversity and maximizes ground cover over time.

MIG BENEFITS—CONSERVATION AND ECONOMICS

The advantages of MIG include greatly reduced soil erosion with the improved air and water quality this entails. MIG greatly improves plant vigor, production, and diversity and improved fish, game bird, and wildlife habitat. The increased organic matter accumulating in soils allows a higher percentage of rainfall to infiltrate where it can be used to stimulate plant growth rather than runoff and to increase sedimentation problems in streams. The animal wastes (manure and urine) are more evenly distributed by MIG than by other grazing systems, allowing those nutrients to be recycled through the plants to the animals without getting into runoff water. In research performed at the

Table 1 Sediment transport and supporting hydrologic data.

	Unimproved pasture watershed				Wooded watershed		
	2 years	3 years	6 years	September 13, 1979 ^a	3 years	6 years	September 13, 1979 ^a
Precipitation (mm/yr)	991	1204	1108	116	1120	1034	121
Surface runoff (mm/yr)	123	203	137	55	261	173	49
Sediment, average annual (kg/ha)	228	934	2088	739	347	301	1032

^aData from the storm of September 13, 1979—the worst storm in more than a century.

North Appalachian Experimental Watershed, water quality in pastured watershed was found to be as good or better than both surface and groundwater from an adjacent woodland watershed. In total, grazing lands properly used are one of the best tools to conserve water and soil resources, to sequester significant quantities of atmospheric carbon dioxide,^[7] and to provide an excellent economic return to the producer by converting for ages into higher valued animal products for human use.

REFERENCES

1. Laflen, J.M. Tillage and residue effect on erosion from cropland. In *Proceedings of the Natural Resources Modeling Symposium*, Pingree Park, Oct 16–20, 1983; DeCoursey, D.G., Ed.; USDA–ARS, 1985; 438–441.
2. Lowdermilk, W.C. *Conquest of the Land Through 7000 Years*; Agriculture Information Bulletin Number 99; Reprint of the 1953 Report; Natural Resources Conservation Service, USDA: Washington, 1999.
3. Swan, J.B.; Lowery, B.; Cruse, R.; Kasper, T.; Lindstrom, M.J.; Moncrief, J.F.; Staricka, J. Interactions of crop residue with soil and climate. In *Crop Residue Management to Reduce Erosion and Improve Soil Quality*; Moldenhauer, W.C., Mielke, L.N., Eds.; USDA–Agricultural Research Service Conservation Research Report Number 42; USDA, North Central Region: Washington, 1995; 73–77.
4. Soil and Water Conservation Society. *Farming for a Better Environment a White Paper*; Soil and Water Conservation Society: Madison, 1995.
5. Owens, L.B.; Edwards, W.M.; Van Keuren, R.W. Surface run-off water quality comparisons between unimproved pasture and woodland. *J. Environ. Qual.* **1983**, *12*, 518–522.
6. Owens, L.B.; Edwards, W.M.; Van Keuren, R.W. Sediment and nutrient losses from an unimproved, all year grazed watershed. *J. Environ. Qual.* **1989**, *18*, 232–238.
7. Owen, L.B.; Hothem, D.L. Carbon stored in soils under eastern grasslands. In *Proceedings of the Second Eastern Native Grass Symposium*; USDA Agricultural Research Service and Natural Resources Conservation Service: Beltsville, MD, 1999; 231–234.

Conserved Lands: Stewardship

Peter August

Department of Natural Resources Science, University of Rhode Island, Kingston, Rhode Island, U.S.A.

Janet Coit

Rhode Island Department of Environmental Management, Providence, Rhode Island, U.S.A.

David Gregg

Rhode Island Natural History Survey, Kingston, Rhode Island, U.S.A.

Abstract

Land that has been conserved for natural resource protection requires careful and ongoing management. There are many factors that can degrade protected lands. These include changes in native plant or animal communities caused by pests, pathogens, or invasive species; poaching; illegal resource harvesting; disturbance to plants and animals from intensive human use; malicious destruction of resources; and ecological changes in nearby landscapes. Every protected property, regardless of size, must have an explicit conservation goal, a management plan, stewardship consistent with the management plan, monitoring to determine whether management goals are being achieved, and adaptive adjustment of management plans if goals are not being met or new threats to the integrity of the protected land emerge.

INTRODUCTION

Acquiring ownership of, or development rights to, land is one of the most effective ways of conserving natural and cultural resources at local, regional, and national scales. Once acquired, the protected land requires constant stewardship. Land protection is done for many reasons: to protect the fauna and flora of a site, to protect cultural resources, to ensure that ecosystem services are maintained, to allow sustainable use of natural resources of the site, and to afford public access for recreational purposes.^[1,2] Land conservation occurs at multiple scales, in all ecosystems (terrestrial, aquatic, and marine), and is accomplished by many kinds of institutions and public agencies. National governments protect watersheds, forests, and rangelands for resource protection and, in many cases, for sustainable resource harvesting. National governments also conserve large national park systems for the benefit of biota and present and future generations of citizens. Conservation occurs at smaller jurisdictions as well, e.g., states, counties, and towns. Large nonprofit organizations, such as The Nature Conservancy and the Audubon Society in the United States, are very effective in protecting land as are small nonprofit organizations such as local land trusts and conservancies. Regardless of the size of the conservation organization or the property preserved, all protected lands require ongoing management and stewardship.

The consequences of not stewarding protected lands can jeopardize the very resources that are meant to be preserved. Developing and implementing a management plan for protected land are good ways of cataloging and prioritizing stewardship responsibilities, and we review steps in management planning here. Threats to protected lands can rapidly change; therefore, monitoring protected lands is an important part of the process. Finally, the technical knowledge within a conservation organization, the availability of staff and equipment, and funding for protected land management are frequently inadequate for the magnitude of the task; thus, innovative collaborations and efficient implementation of management plans are necessary.

Protected land stewardship is a challenging endeavor and has many components, which will be reviewed here. Large government agencies or other organizations that own and manage large areas of protected lands typically have dedicated programs for stewardship activities. For example, the National Park Service (NPS) oversees a large and complex network of protected areas in the United States. Every park in the NPS system has a general management plan, which articulates the management needs of a park and how the park will meet those needs.^[3] Furthermore, the NPS has a sophisticated program—the NPS Inventory and Monitoring Program—to systematically monitor environmental conditions in parks to know whether management and stewardship activities are having the desired results and to be vigilant to new or unforeseen threats to the ecological

integrity of a park.^[4] Other conservation agencies that oversee large areas of land have similar programs such as the U.S. Fish and Wildlife Services,^[5] U.S. Bureau of Land Management,^[6] and U.S. Forest Service.^[7] Small conservation organizations, such as local conservancies and land trusts, typically do not have dedicated staff or program resources for protected land stewardship, yet they own significant areas of land. In the United States, e.g., there are 1700 different land trusts that control over 150,000 km² of land.^[5] The focus of this entry is to describe the conservation land stewardship process that would be followed by a small conservation organization.

LAND PROTECTION AND STEWARDSHIP

There are a variety of reasons to protect land and many stewardship strategies that can be used to meet conservation goals (Table 1). Conservation lands are typically protected by securing fee simple ownership, control over future development rights, or establishment of permitted and restricted uses of the land through zoning controls.^[2]

Table 1 Examples of common conservation goals and management activities that achieve them.

Goal	Management and stewardship actions
Preserve biodiversity	Protect large tracts of land. Connect separate refuges with corridors. Increase the size of refuges. Enforce policies against poaching wildlife or harvesting plants. Ensure adjacent land uses are not a source of invasive species, pests, or pathogens. Monitor for invasive species, pests, and pathogens. Monitor population levels of key or indicator species of plants and animals.
Preserve cultural resources	Protect sites or regions of interest; limit public access to sensitive sites.
Preserve ecosystem services	Ensure that conservation land boundaries encompass whole watersheds, maintain diversity of habitats, and allow public access and nondestructive forms of recreation so that patrons can benefit from aesthetic values.
Preserve aesthetic or recreational values	Provide trails and interpretive services for public access; encourage hiking, hunting, and fishing if appropriate for the site; and manage views and soundscapes to preserve natural conditions.
Ensure sustainable resource use	If a site permits, allow controlled harvest of sustainable resources, such as wood products, game and fish, plants, fruits, berries, and fungi.
Agricultural preservation	Obtain development rights and easements to working lands and waters to ensure they will remain in agricultural land use.

However a property is controlled, the development and implementation of an ongoing management plan for the protected property are essential. There are many threats to the ecological integrity of protected lands (Table 2 and Fig. 1). Pests, pathogens, and invasive species can impact native fauna and flora of a site. Illegal poaching or harvesting natural resources can diminish the biota. Soil compaction, erosion, and vegetation disturbance in fragile ecosystems caused by motor vehicle riding (e.g., all-terrain vehicles and motorcycles) can result in serious environmental damage. Malicious acts, such as garbage dumping, littering, theft of cultural resources, and vandalism, can diminish the aesthetic value of a site. A carefully designed management plan will be attentive to these threats. Protected land management should happen in a systematic manner (Fig. 2). The basic steps are discussed in detail in the coming sections.

Site Assessment and Baseline Inventory

An essential first step in conservation land management is a site (the parcel) and landscape (what is around it) survey to inventory existing conditions and catalog important habitats. The purpose of site assessment is to evaluate the presence and condition of important natural resources and to identify threats to the focal resources and the ecosystem as a whole. Target resources can be species, habitats, rivers, landforms, views, farms, cultural resources, groundwater, or ecosystem services and ecological processes.^[8,9] Site assessment can be a complex activity and require personnel familiar with local ecological conditions. Assessment involves field survey and consolidation of relevant geospatial information such as data from geographic information systems (wetlands, rare species occurrences, land use, soils, landform, etc.), digital imagery, and the results of previous reconnaissance of the area if available. The initial site assessment establishes a baseline condition for easement monitoring and tracking changes in the condition of

Table 2 Examples of common threats to resources on conserved lands.

Resource	Threat
Biodiversity	Habitat destruction, poaching; illegal harvest of plants or animals; spread of invasive species, pests, and pathogens; loss of fitness due to small population sizes; and illegal motor vehicle access in sensitive habitats.
Cultural resources	Vandalism, theft, and alteration of landscape context (e.g., views, soundscapes).
Ecosystem services	Groundwater withdrawal outside refuge, habitat destruction on borders of refuge, and high-impact land uses outside refuge in watershed.
Aesthetic or recreational values	Trespassing, illegal motor vehicle access, dumping, and degradation of natural views or soundscapes.

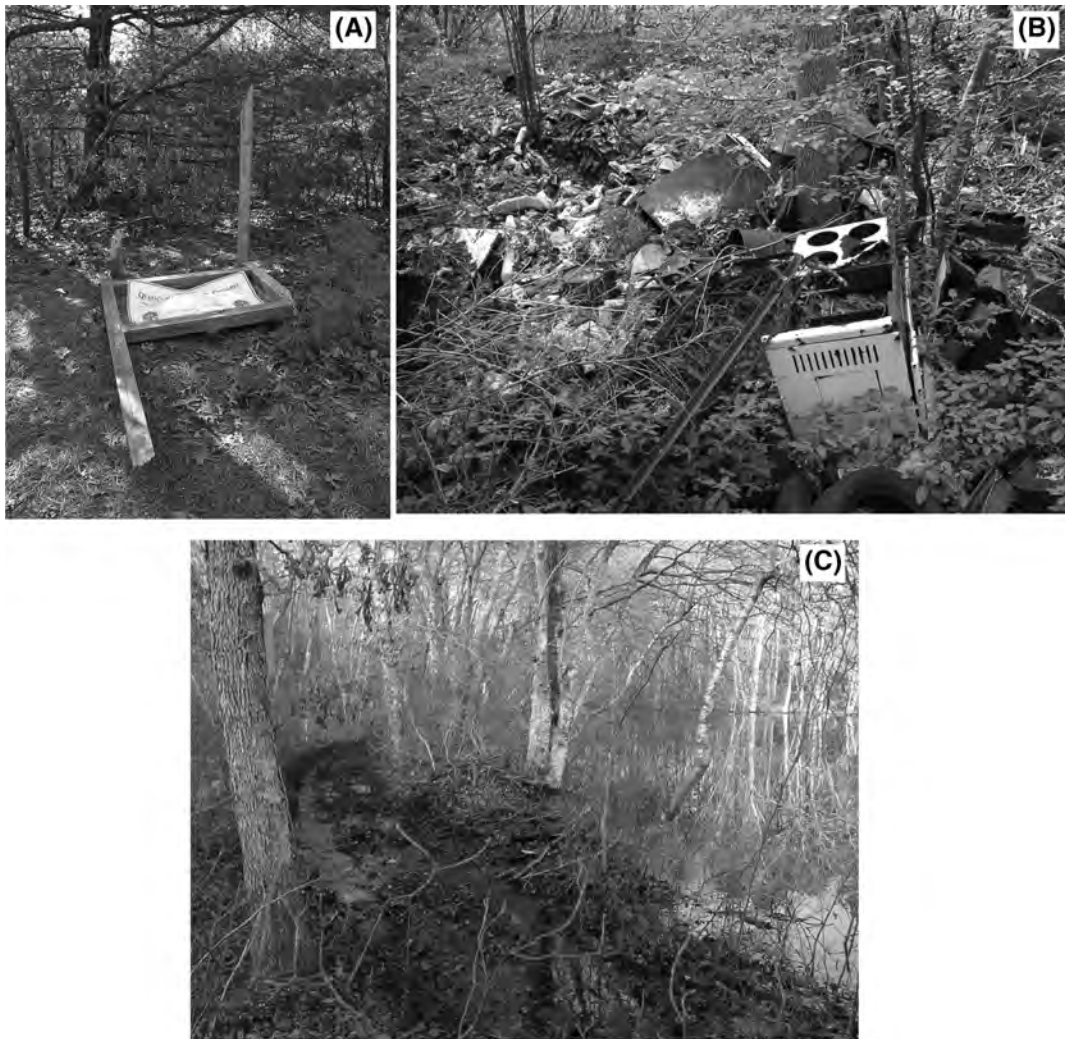


Fig. 1 Examples of threats to protected lands: (A) vandalism (destruction of signage); (B) illegal dumping of refuse; and (C) all-terrain vehicle track damage to wetland habitat.

Source: Photo courtesy of The Rhode Island Chapter of The Nature Conservancy.

the property. Landscape analysis provides a regional context for conservation and provides insight into gains and losses of dispersal corridors, up watershed threats that would jeopardize site-level habitats or species, and land use changes on nearby properties that might enhance or diminish the conservation value of a particular parcel.^[10] There is a growing number of protocols that have been advanced to perform a baseline inventory.^[11,12]

Development of Management Goals and Plan

Development of management goals is an essential step, and the goals of the management plan will reflect the values of the conservation institution and the purpose of acquiring the conservation land.^[13] Common management goals include stewarding the land for water (ground and/or surface) protection, conservation of biodiversity, forestry, farming, aesthetic values, fish and wildlife management, and recreation

(hiking, paddling, fishing, hunting, etc.). Management goals of one property can affect the viability of nearby conservation properties owned by others; hence, there is a need for coordinated management when protected lands are in a mosaic of small parcels managed by different institutions. It is important that a management plan establishes a basis or rationale for prioritizing resources and stewardship actions so that scarce resources applied over time can still achieve large, long-term goals. Once goals for a property are established, a management plan should be developed and clear measures of success must be identified. These measures are the basis for ongoing monitoring and are the indicators of success or failure of the management plan.

Implementation of Management Plan

Management plans can vary in complexity; simple management plans for conservation lands may require little activity

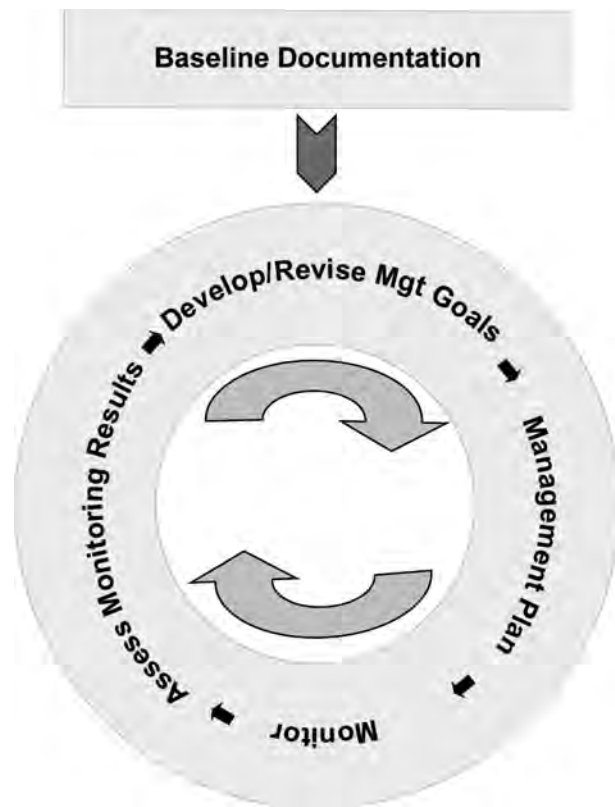


Fig. 2 Management cycle of protected lands: Steps involved in managing protected lands.

beyond periodic monitoring. Other stewardship activities, however, could require considerable work, expertise, and investment in personnel, supplies, equipment, and time in the field. Examples of expensive and complex management tasks include habitat restoration, creating and maintaining trail systems, forest management (selective cutting to maintain the age and target species composition of the forest), and removal or control of invasive species.^[14,15] The implementation of a management plan can be a challenge for conservation organizations. Large land owners do not always have the staff or resources to manage extensive properties. Similarly, small, local conservation organizations, such as land trusts, rarely have the technical wherewithal or financial ability to take on complex management activities. Partnerships and collaborative projects are one way to perform complex stewardship tasks.

Monitoring

Vigilant monitoring is required of conservation properties and the land surrounding them. Easements must be monitored on a regular basis to ensure that fee owners are managing properties in a manner that is consistent with the easement that is held by the conservation organization. Violations in easements will have legal and policy ramifications, which need to be addressed.

All properties must be visited on a regular basis to protect against adverse possession claims and to identify inappropriate human activities occurring on or near them. Vandalism, dumping, erosion caused by motor vehicles, illegal hunting, and wood cutting are, unfortunately, common on conservation lands (Fig. 1). Monitoring for disturbances such as these is relatively straightforward for small properties but requires constant attention. Monitoring over large, expansive conservation lands can be logistically difficult, especially if access is difficult.^[16] Ecological monitoring must be done on and around conservation lands to ensure that invasive plants and animals have not become established, pests or pathogens are not present, stewardship activities are yielding desired outcomes, target species are still present, and ecosystem health remains high. Regional changes in the creation of impervious surfaces, which increase storm water runoff, water withdrawal in the watershed, land use conversion, and habitat fragmentation can degrade the condition and viability inside conservation lands.

Synthesis, Reflection, and Adaptive Stewardship

Monitoring provides the data from which decisions about the efficacy of the management plan can be made. A careful analysis of monitoring data determines whether the management plan is working and the ecological condition of the property is changing. If the desired results are not occurring, the management plan should be modified to meet the conservation goals of the property. This is a critical step and follows the logic of the adaptive management paradigm.^[17] New, unanticipated stewardship challenges can emerge rapidly. Management plans must be dynamic documents and capable of changing as knowledge or needs require.

CONCLUSION

Acquiring land is an effective way to protect natural resources, and careful stewardship of conserved lands ensures that the condition of the natural resources is preserved. Stewardship is an ongoing process and a long-term commitment. Many factors can diminish the value of protected land, and these must be monitored and, when present, mitigated. The process of protected land management has a number of steps and begins with a careful resource inventory of the protected property and establishment of a suite of management goals that will direct stewardship activities. Ongoing monitoring to ensure that management goals are being met is an essential component of the process.

Large land owners, such as U.S. NPS or U.S. Forest Service, have very complex management programs to ensure that the property in their care retains the environmental and cultural resources the lands were obtained to protect. Small land owners, such as land trusts and local

conservancies, frequently do not have dedicated staff resources, knowledge, or budgets to steward their land, but innovative partnerships are one way the resources of many institutions can be leveraged to achieve effective land stewardship even by small land owners. One model is the Rhode Island Conservation Stewardship Collaborative (RICSC),^[18] an alliance of federal, state, municipal, and nonprofit conservation organizations who partner to (from its mission statement) “. . . advance long-term protection and stewardship of terrestrial, aquatic, coastal, estuarine, and marine areas in Rhode Island that have been conserved by fee, easement, or other means.” The RICSC tackles systemic, state-wide, impediments to good conservation land stewardship and provides training materials, protocols, and technical capacity to assist conservation land owners in their stewardship challenges.

ACKNOWLEDGMENTS

The following individuals have provided us thoughtful guidance and support for our work in conservation land management and stewardship: Julie Sharpe, Peggy Sharpe, Rupert Friday, Cathy Sparks, Scott Ruhren, Sharon Marino, Larry Taft, Jennifer Pereira, Charles Vandemoer, and Kathleen Wainwright. The Rhode Island Conservation Stewardship Collaborative has challenged us to develop procedures, protocols, and guidance for conservation agencies, especially local land trusts, in supporting their protected land management. The University of Rhode Island Cooperative Extension program, URI Department of Natural Resources Science, and USDA Renewable Resources Extension Act have steadily supported our work in conservation land management and stewardship.

REFERENCES

- Daily, G.C.; Ehrlich, P.R.; Goulder, L.H.; Alexander, S.; Lubchenco, J.; Matson, P.A.; Mooney, H.A.; Postel, S.; Schneider, S.H.; Tilman, D.; Woodwell, G.M. Ecosystem services: Benefits supplied to human societies by natural ecosystems. *Issues Ecol.* **1997**, *2* (1), 1–18.
- Duerksen, C.; Snyder, C. *Nature Friendly Communities: Habitat Protection and Land Use Planning*; Island Press: Washington, 2005; 421 pp.
- National Park Service. *General Management Plan Dynamic Sourcebook*. Available at <http://planning.nps.gov/GMPSourcebook/GMPHome.htm> (accessed February 2012).
- Oakley, K.L.; Thomas, L.P.; Fancy, S.G. Guidelines for long-term monitoring protocols. *Wildlife Soc. B.* **2003**, *31* (4), 1000–1003.
- United States Fish and Wildlife Service. *Comprehensive Conservation Planning Process*. Available at <http://www.fws.gov/policy/602fw3.html> (accessed February 2012).
- United States Bureau of Land Management. *Land Use Planning*. Available at <http://www.blm.gov/planning/index.html> (accessed February 2012).
- United States Forest Service. *Strategic Planning and Resource Assessment*. Available at <http://www.fs.fed.us/plan> (accessed February 2012).
- Turner, R.K.; Daily, G.C. The ecosystem services framework and natural capital conservation. *Environ. Resour. Econ.* **2008**, *39* (1), 25–35.
- Lokocz, E.; Ryan, R.L.; Sadler, A.J. Motivations for land protection and stewardship: Exploring place attachment and rural landscape character in Massachusetts. *Landscape Urban Plan.* **2011**, *99* (2), 65–76.
- Theobald, D.M. Targeting conservation action through assessment of protection and exurban threats. *Conserv. Biol.* **2003**, *17* (6), 1624–1637.
- Land Trust Standards and Practice. *Land Trust Alliance*, 2004. Available at <http://www.lta.org> (accessed October 2011).
- Ruhren, S. *Baseline documentation and inventory protocol. Rhode Island Conservation Stewardship Collaborative*, 2011. Available at http://www.ricsc.org/Projects/Docs_2009/CSC_BDIP.pdf (accessed October 2011).
- Carwardine, J.; Wilson, K.A.; Watts, M.; Etter, A.; Klein, C.J.; Possingham, H.P. Avoiding costly conservation mistakes: The importance of defining actions and costs in spatial priority setting. *PLoS ONE* **2008**, *3* (7), e2586.
- Anderson, P. Ecological restoration and creation: A review. *Biol. J. Linn. Soc.* **1995**, *56* (S1), 187–211.
- Mack, R.N.; Simberloff, D.; Lonsdale, W.M.; Evans, H.; Clout, M.; Bazzaz, F.A. Biotic invasions: Causes, epidemiology, global consequences and control. *Ecol. Appl.* **2000**, *10* (3), 689–710.
- Upgren, A.; Bernard, C.; Clay, R.P.; de Silva, N.; Foster, M.N.; James, R.; Kasecker, T.; Knox, D.; Rial, A.; Roxburgh, L.; Storey, R.J.; Williams, K.J. Key biodiversity areas in wilderness. *Int. J. Wilder.* **2009**, *15* (2), 14–17.
- Walters, C. *Adaptive Management of Renewable Resources*; MacMillan Publishers: New York, 1986; 374 pp.
- RICSC. *Rhode Island Conservation Stewardship Collaborative*. Available at <http://www.ricsc.org> (accessed October 2011).

Consistence

Ray McBride

Department of Land Resource Science, University of Guelph, Guelph, Ontario, Canada

Abstract

Soil consistence and consistency are closely related terms used by pedologists and soil engineers, respectively, to describe the degree of resistance to deformation or rupture exhibited by a structured soil when subjected to externally applied mechanical stresses. This aspect of a soil's rheological behavior, however, has only ever been characterized with semiquantitative criteria. Soil profiles are routinely characterized for their consistence by horizon as part of a soil morphological description, which can aid in making pertinent soil survey interpretations for common land uses, where knowledge of soil strength is an important consideration. Soil consistency limits are gravimetric soil water contents that correspond to arbitrary boundaries defining the liquid, plastic, and semisolid states as measured by simple laboratory tests. The term *consistency*, however, was originally used by soil engineers with reference to a simple field test of soil resistance to penetration by a thumb or thumbnail, when the specimen was at field water content.

INTRODUCTION

Soil consistence and consistency are closely related terms used by pedologists and soil engineers, respectively, to describe the degree of resistance to deformation or rupture exhibited by a structured soil when subjected to externally applied mechanical stresses. This aspect of a soil's rheological behavior, however, has only ever been characterized with semiquantitative criteria. One of these criteria is that the test soil should be under one or more prescribed (and stated) conditions of initial soil wetness (e.g., dry, moist, and wet), as these properties are highly dependent on soil water content (Fig. 1). These soil properties are a manifestation of the combined forces of adhesion/cohesion that contribute to: 1) the attraction between soil particles and pore water and 2) the attraction of soil particles to one another. As a result, particle-size distribution, organic matter content, clay mineralogy, soil solution chemistry (adsorbed cations), soil fabric, sample pretreatment, etc. also have a large influence on these properties. Classes or ratings of soil consistence are commonly determined by pedologists with simple manual field tests that involve the application of compressive forces to a structurally intact soil specimen (e.g., clod) until *rupture* occurs. Conversely, soil consistency classes or ratings are routinely used by soil engineers in the classification of soils used for engineering purposes, but in this case the field tests usually involve defining the soil resistance to *penetration* by a blunt object (e.g., thumb/thumbnail, blunt end of a pencil, etc.).

SOIL CONSISTENCE

Soil profiles are routinely characterized for their consistence by horizon as part of a soil morphological description, which can aid in making pertinent soil survey interpretations for

common land uses (e.g., agriculture), where knowledge of soil strength is an important consideration (e.g., tillage and traffic). Simple field tests for soil consistence at prescribed water contents have been developed by the U.S. Department of Agriculture–Soil Conservation Service^[1] and have been adopted by pedologists in many other countries including Australia^[2] and Canada.^[3]

The principal field test involves compressing an unconfined, block-like specimen between the extended thumb and forefinger, between both hands, or between the foot and a non-resilient flat surface (Table 1).^[1] Rupture resistance to compressive stress is indirectly related to the tensile strength (i.e., the load per unit area at which an unconfined cylindrical specimen will fail in a simple

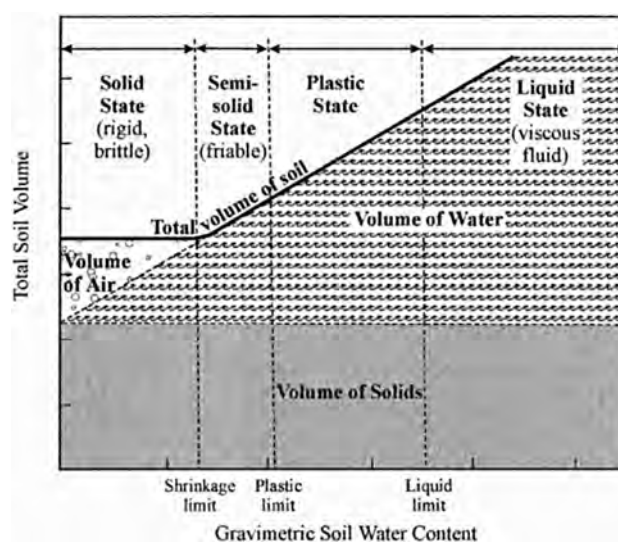


Fig. 1 Generalized schematic of consistency states and shrinkage behavior of clay-rich soils.

Table 1 Terms for describing consistence (rupture resistance) of block-like specimens.

Soil consistence classes at prescribed soil wetness conditions			Field test description	
Moderately dry and very dry	Slightly dry and wetter	Air-dried, then submerged	Operation	Approximate force/energy applied ^a
Loose	Loose	Not applicable	Specimen not obtainable	—
Soft	Very friable	Non-cemented	Fails under very slight force applied slowly between thumb and forefinger	<8 N
Slightly hard	Friable	Extremely weakly cemented	Fails under slight force applied slowly between thumb and forefinger	8–20 N
Moderately hard	Firm	Very weakly cemented	Fails under moderate force applied slowly between thumb and forefinger	20–40 N
Hard	Very firm	Weakly cemented	Fails under strong force applied slowly between thumb and forefinger (80 N is about the maximum force that can be applied)	40–80 N
Very hard	Extremely firm	Moderately cemented	Cannot be failed between thumb and forefinger but can be between both hands or by placing on a non-resilient surface and applying gentle force underfoot	80–160 N
Extremely hard	Slightly rigid	Strongly cemented	Cannot be failed in hands but can be underfoot by full body weight (approximately 800 N) applied slowly	160–800 N
Rigid	Rigid	Very strongly cemented	Cannot be failed underfoot by full body weight but can be by a <3 J blow	800 N–3 J
Very rigid	Very rigid	Indurated	Cannot be failed by a blow of <3 J	≥3 J

^aBoth units of force (Newtons, N) and energy (joules, J) are used.

Source: From Soil Survey Division Staff.^[1]

tension test).^[4] In Table 1, approximate values of applied force/energy for the various classes are provided. If induration with cementing agents is suspected (e.g., organic substances, calcite, silica, and sesquioxides), the specimen is air-dried and then submerged in water for at least 1 hour before manual compression testing takes place (Table 1). Apart from brittle fracture (rupture), however, soil specimens can fail in different manners (e.g., plastic deformation, flow, and smearing) and meaningful categories can also be determined manually in the field (Fig. 1).^[1]

Other related tests that can be performed on structured, confined soil include resistance to penetration with a commercially available, inexpensive “pocket penetrometer” (i.e., the pressure required to push a 6.4-mm diameter cylindrical rod a distance of 6.4 mm into the soil within 1 second).^[1] Related field tests on puddled, unconfined soils include plasticity (permanent deformation at constant volume without rupturing as force is applied), toughness (manual force needed to perform the plastic limit roll test), and stickiness (capacity of a soil to adhere to other objects). The plasticity and toughness tests used by pedologists were originally derived from soil engineering practice.^[5]

SOIL CONSISTENCY

Soil consistency (or Atterberg) “limits” are gravimetric soil water contents that correspond to arbitrary boundaries defining the liquid, plastic, and semisolid states as measured by

simple laboratory tests^[6] (Fig. 1). The term “consistency,” however, was originally used by soil engineers with reference to a simple field test of soil resistance to penetration by a thumb or thumbnail, when the specimen was at field water content.^[5] There has been no attempt by the American Society for Testing and Materials to standardize their test

Table 2 Terms for describing consistency (penetration resistance) of cohesive material at field water content.

Soil consistency term	Field test description	Approximately undrained shear strength (kN/m ²)
Very soft	Easily penetrated, several centimeters by fist	<20
Soft	Easily penetrated several centimeters by thumb	20–40
Soft to firm	—	40–50
Firm	Moderate effort needed to penetrate by thumb	50–75
Firm to stiff	—	75–100
Stiff	Great effort needed to penetrate by thumb	100–150
Very stiff	Readily indented by thumbnail	150–200
Hard	Very difficult to indent with thumbnail	>200

Source: Adapted from ASTM.^[5]

method using a pocket penetrometer or similar device,^[5] but approximate values of undrained shear strength have been assigned to consistency classes of cohesive soils (Table 2). Soil engineering tests for cementation^[5] are performed manually by applying pressure on structured, unconfined specimens, as in Soil survey division staff,^[1] but do not involve submersion in water prior to testing.

Only soils with significant clay contents (i.e., “plastic” or “cohesive” soils) have measurable Atterberg limits, and they are among the most meaningful and widely interpreted of soil engineering test indices. Applications include being used to estimate the shear strength and bearing capacity, compressibility, permeability, swelling potential, and specific surface of soils.

REFERENCES

1. Soil Survey Division Staff. *Soil Survey Manual*. USDA-SCS Agricultural Handbook 18; U.S. Government Printing Office: Washington, 1993.
2. McDonald, R.C.; Isbell, R.F.; Speight, J.G.; Walker, J.; Hopkins, M.S. *Australian Soil and Land Survey Field Handbook*, 2nd Ed.; CSIRO Land and Water Division, Inkata Press: Melbourne, 1998.
3. Expert Committee on Soil Survey. *The Canada Soil Information System (CanSIS): Manual for Describing Soils in the Field*; Day, J.H., Ed.; Land Resource Research Institute (LRRI) Contribution No. 82–52; Expert Committee on Soil Survey, Supply and Services Canada, Research Branch, Agriculture Canada: Ottawa, 1982.
4. SSSA. *Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 1997.
5. ASTM. Standard practice for description and identification of soils (visual–manual procedure) (D2488–93). In *2000 Annual Book of ASTM Standards*; American Society for Testing and Materials: West Conshohocken, 2000; Vol. 04.08, 249–259.
6. ASTM. Standard test methods for liquid limit, plastic limit, and plasticity index of soils (D 4318–98). In *2000 Annual Book of ASTM Standards*; American Society for Testing and Materials: West Conshohocken, 2000; Vol. 04.08, 546–558.

Consolidation

Tarunjit Singh Butalia

Civil and Environmental Engineering, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Consolidation is the reduction in volume of soil caused by the movement of water out of a soil mass. The term is commonly referred to as primary compression. The rate of consolidation is an important property of a soil mass because water has to move in the direction of decreasing potential for consolidation to occur. It is the movement of water rather than the compression of air-filled voids, which distinguishes consolidation from compaction. The magnitude of consolidation for coarse-grained soils (sand and gravel) is much less compared to fine-grained soils (clay and silt). However, because fine-grained soils exhibit low coefficient of permeability, the rate of consolidation in fine-grained soils is much slower than coarse-grained soils.

THEORIES AND ANALYSES

Consolidation analysis generally focuses on saturated soils. When an external load is applied on soil or if the vertical stresses increase for saturated soils, the pore water pressure increases immediately by an amount equal to the increase in applied stress. Gradually, some of the pore water moves out of the soil mass in the direction of decreasing potential, thus transferring the stresses from the water particles to the soil skeleton. This results in reduced volume of the soil. During the consolidation process, the volume of the soil particles remains unchanged, while the volume of the water-filled voids is reduced.

One of the first soil consolidation laboratory experiments was presented by Frontard.^[1] He placed clay samples in metal containers and loaded the samples with a piston. Meantime, Forchheimer^[2] was the first to propose a mathematical model for consolidation analysis. While the mathematical formulation was simple, it was not very accurate because of incomplete understanding of the concept of effective stress. Terzaghi,^[3] one of Forchheimer's former students, was the first person to recognize clearly the concept of effective stress and its application to the soil consolidation problem.

Terzaghi's theory of consolidation, relating soil compression and effective stress as well as the rate of consolidation, is considered to be the advent of modern soil mechanics. Terzaghi's simple theory assumed a saturated clayey soil, the void ratio–logarithm of effective stress relationship to be linear, and that the soil properties do not change during the consolidation process. The analysis was carried out for 1-D (axial) settlement with no radial deformations. For a homogeneous soil layer of thickness H with initial void ratio e_0 , the

amount of 1-D consolidation settlement due to Δe decrease in void ratio is

$$\Delta H = \frac{\Delta e H}{1 + e_0} \quad (1)$$

In most cases, the decrease in void ratio is generally expressed in terms of compression index or coefficient of compressibility and change in effective stress. The rate of consolidation is affected by several factors including soil, stratum, and loading properties. Soil properties of interest include soil permeability, void ratio, and compressibility. Stratum factors include the thickness of the layer under consideration as well as the drainage conditions surrounding the layer. Loading factors include the ratio of the new loading to original loading. The soil properties are commonly modeled through the coefficient of consolidation (c_v) as follows:

$$c_v = \frac{k(1 + e)}{a_v \gamma_w} \quad (2)$$

where k is the coefficient of permeability, e is the in-place void ratio, a_v is the coefficient of compressibility, and γ_w is the density of the pore fluid (typically water). The governing equation of 1-D diffusion as applied to transient water flow is as follows:

$$\frac{\partial u}{\partial t} = c_v \frac{\partial^2 u}{\partial z^2} \quad (3)$$

Eq. 3 is solved to obtain the rate at which consolidation occurs.

Oedometer or 1-D consolidation laboratory test (ASTM D2435)^[4] is used to evaluate the consolidation characteristics of a soil sample. The test involves applying varying



Fig. 1 Consolidation test apparatus.

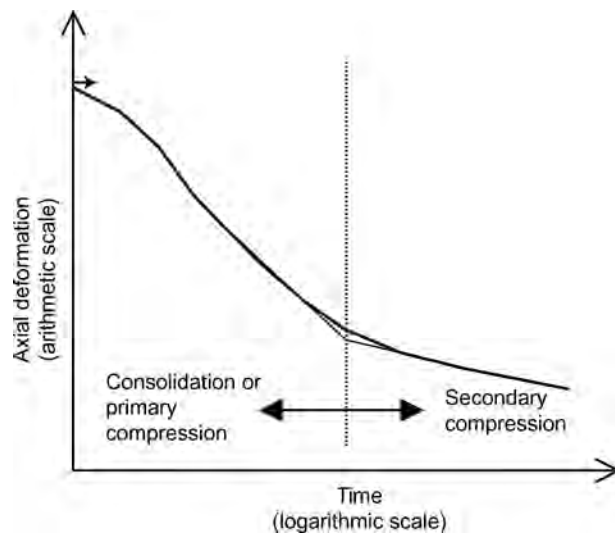


Fig. 2 Idealized axial compression vs. time plot.

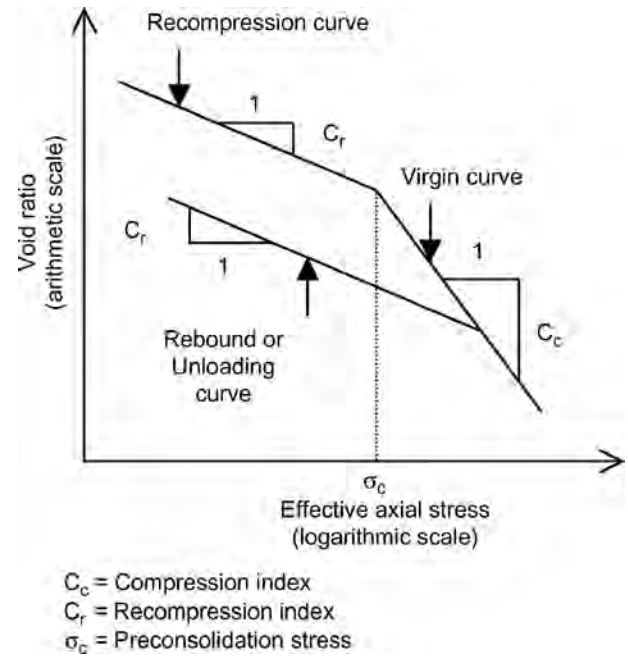


Fig. 3 Idealized void ratio vs. axial effective stress plot.

levels of stress on a soil sample that is confined in the radial direction and allowed to drain and compress axially (Fig. 1). Data collection for the test mainly consists of axial compression vs. time readings for each stress loading and unloading process (Fig. 2). The ultimate void ratio for each loading and unloading step can then be plotted against the logarithm of the vertical (axial) effective stress (Fig. 3). The slope of the virgin curve is referred to as the compression index (C_c), while the slope of the

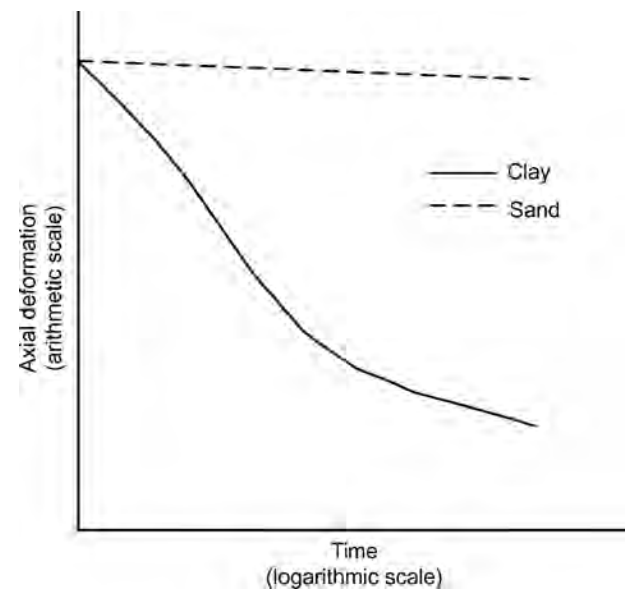


Fig. 4 Typical axial compression vs. time plot for clay and sand at specified effective axial stress.

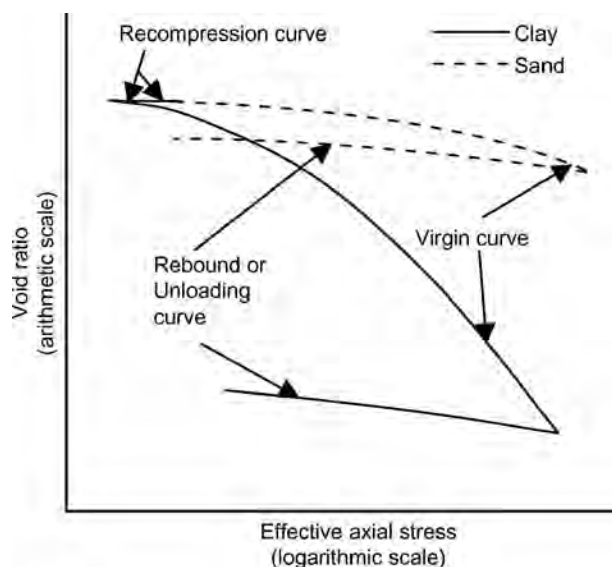


Fig. 5 Typical void ratio vs. axial effective stress plot for clay and sand.

recompression or unloading curve is referred to as the recompression index (C_r). Typical consolidation test curves for clay and sand samples are shown in Figs. 4 and 5. Compression and recompression index values less than 0.2 represent soils with low compressibility. Values between 0.2 and 0.4 indicate moderately compressible soils, while values larger than 0.4 represent highly compressible soils. Accurate laboratory evaluation of coefficient of consolidation values can be challenging because of its dependence on permeability coefficient (k), coefficient of compressibility (a_v), and potential sample disturbance. The average coefficient of consolidation varies from 10^{-8} to 10^{-6} m²/s. For controlled-strain loading testing of soil samples, test method ASTM D4186^[5] should be used.

Terzaghi's theory and the 1-D laboratory experiment based on it are used extensively by engineers for consolidation analysis. However, the stringent soil behavior and modeling considerations in Terzaghi's theory limit the use of the theory for predicting pore water pressure responses and settlements (amount and rate) in complex situations. Alternatively, non-linear consolidation models proposed by Schiffman,^[6] Gibson,^[7] Tse,^[8] Scott,^[9] and Leroueil^[10] can be used. Many of these models allow for a non-linear variation of void ratio with logarithmic effective stress and the change in soil permeability as consolidation occurs. Additional technical

information on soil consolidation can be obtained from the studies by Wu,^[11] Mitchell,^[12] Terzaghi,^[13] Coduto,^[14] McCarthy,^[15] and DeBoer.^[16]

REFERENCES

1. Frontard, J. Notice sur l'accident de la digue de charmes. *Ann. Ponts et Chaussees* **1914**, 23 (9), 173–280. (in French).
2. Forchheimer, P. *Hydraulik*; Teubner: Leipzig, 1914; 494–495. (in German).
3. Terzaghi, K. *Erdaumechanik auf Bodenphysikalischer Grundlage*; Deuticke: Vienna, 1925.
4. ASTM D2435. *Standard Test Method for One-dimensional Consolidation Properties of Soils. Book of Standards*; American Society for Testing and Materials, 2001; Vol. 04.08.
5. ASTM D4186. *Standard Test Method for One-dimensional Consolidation Properties of Soils Using Controlled-strain Loading. Book of Standards*; American Society for Testing and Materials, 2001; Vol. 04.08.
6. Schiffman, R.L. The use of visco-elastic stress-strain laws in soil testing. *ASTM Spec. Tech. Publ.* **1959**, 254, 131–155.
7. Gibson, R.E.; Schiffman, R.L.; Cargill, F.W. The theory of one-dimensional consolidation of saturated clays. II. Finite non-linear consolidation of thick homogenous layers. *Can. Geotech. J.* **1981**, 18, 280–293.
8. Tse, E.C. *Influence of Structure Change on Pore Pressure and Deformation Behavior of Soft Clays Under Surface Loadings*. Ph.D. dissertation, Department of Civil Engineering, University of California, Berkeley, 1985.
9. Scott, R.F. consolidation of sensitive clay as a phase change process. *J. Geotech. Eng.-ASCE* **1989**, 115 (10), 1439–1458.
10. Leroueil, S.; Magnan, J-P.; Tavenas, P. *Embankments on Soft Clays*; Ellis Horwood Series in Civil Engineering, D.M. Wood (Trans); Ellis Horwood: New York, 1990.
11. Wu Tien, H. *Soil Mechanics*; Allyn and Bacon, Inc.: Worthington, 1982.
12. Mitchell James, F. *Fundamentals of Soil Behavior*; Wiley: New York, 1993.
13. Terzaghi, K.; Peck, R.B.; Mesri, G. *Soil Mechanics in Engineering Practice*; Wiley: New York, 1996.
14. Coduto Donald, P. *Geotechnical Engineering: Principles and Practices*; Prentice Hall: Upper Saddle River, 1999.
15. McCarthy David, F. *Essentials of Soil Mechanics and Foundations: Basic Geotechnics*; Prentice Hall: Upper Saddle River, 1998.
16. DeBoer, R.; Schiffman, R.L.; Gibson, R.E. The origins of the Terzaghi-Fillunger dispute. *Geotechnique* **1996**, 46 (2), 175–186.

Controlled Traffic

Randall C. Reeder

*Food, Agricultural and Biological Engineering Department, Ohio State University,
Columbus, Ohio, U.S.A.*

Abstract

Controlled traffic eliminates worries about soil compaction and improves overall farm efficiency and economics. Auto-steering is recommended, especially for large equipment, because it makes a controlled traffic system easier to set up and follow. Controlled traffic systems work with no-till and other tillage systems and with most major crops.

INTRODUCTION

Controlled traffic is a method to manage soil compaction. Compaction is “managed” by assuring that all heavy traffic is confined to specific lanes through the crop, year after year. The lanes become compacted, and the soil between lanes is never driven on.

Controlled traffic is an ideal match for most conservation tillage systems. A global positioning system-based guidance system makes it possible to till the entire surface and follow the permanent lanes. With controlled traffic, field efficiency and crop yields increase and the cost of inputs (pesticides, seed, labor, and fuel) decrease. Many farmers convert all of their machinery to controlled traffic at one time, while others choose to reduce or eliminate the extra cost by moving to controlled traffic gradually as equipment is replaced. Maximum advantages of controlled traffic occur when the operating width of all equipment is matched and tire spacings are set to minimize the trafficked area.

The first step to controlled traffic is to make all equipment cover the same width or multiples of that width. For big grain farms, the combine harvester will likely be the determining factor; e.g., if the grain head is 36 ft (11 m), the drill or air seeder might be 35 ft (10.7 m), and the sprayer either 70 ft (21.3 m) or 105 ft (32 m) wide. For a farm with row crops [e.g., corn in 30-in. (750 mm) rows], the combine could be 12-row, the planter either 12- or 24-row, and the sprayer either 60 ft (18.3 m) or 90 ft (27.4 m) wide. A farm operation with a 16-row corn head would have similar multiples for the planter and sprayer. If the same farm raises small grains or soybeans, the drill, air seeder, and grain head must match these widths. For smaller operations, the basic width might be 15 ft (4.5 m) or 20 ft (6 m).

The second step is to look at the tires (or tracks) of all vehicles in the field. The goal is to minimize the number of traffic lanes and the width of those lanes. Any equipment (sprayer and cultivator) driven through a growing crop must use the traffic lanes, and the tires should fit between the rows.

A combine or four-wheel drive tractor with wide tires is satisfactory, but the goal would be to go to narrower tires (maybe split duals) using the same traffic lanes ([Table 1](#)).

Controlled traffic improves traction, flotation, and timeliness of planting, spraying, and harvesting while minimizing potential yield losses from compaction. Compaction is managed, not eliminated. Controlled traffic takes the advantage of compaction where it is beneficial. Compacted soil under wheel tracks provides better flotation and improved traction, especially on wet soil. Controlled traffic eliminates compaction where it is a detriment (under and close to rows) and leads to better water infiltration, root development, and fertilizer uptake.

Controlled traffic eliminates overlaps and skips during pesticide and fertilizer application and while seeding crops. Overlaps often waste 10–15% of chemicals, drilled seed, fuel, and all other machine operation expenses. Precisely placed traffic paths can eliminate the waste and more than offset the expense of establishing and maintaining the controlled traffic system.

NO-TILL

One concern farmers have about no-till is the potential for compaction and ruts caused by driving on wet soil. Controlled traffic removes most of that concern. Controlled traffic provides for potential yield increases by restricting compaction to trafficked lanes. The stalks or skip rows remaining from the previous year's crop provide visual guidance at planting ([Fig. 1](#)). Auto-steering systems are extremely accurate and can eliminate any need for visual clues.

RIDGE TILL

Controlled traffic is an integral part of a ridge till system and is responsible for part of the yield advantages.

Table 1 Example traffic patterns for row crops.

No. of rows	Tractor (in.)	Combine (in.)	No. of paths	% Trafficked
30-in. (750-mm) row spacing				
6	60	120	4	44
6	120	120	2	22
8	120	120	2	17
8	60 and 120	120 and 180	6	50
12	60	120 (6-row)	4	22
16	60 and 120	120 and 180 (8-row)	8	33
24	60 and 120	120 and 180 (12-row)	12	33
36-in. (900-mm) row spacing				
6	72	144	4	37
8	72	144	4	28
12	72	144 (6-row)	4	18

Example wheel spacings for tractor and combine, single and dual tire options. Percent of the field trafficked assumes that each path is 20 in. (500 mm) wide.

All tires must be narrow enough and spaced properly to fit between the rows. Driving on the row with a combine or other heavy load significantly reduces yield in those rows the following year. If the ground is frozen solid, driving on the row does not harm the soil.

MULCH TILL

Auto-steering systems make it possible to use controlled traffic with mulch till systems. To maintain the full benefits of controlled traffic, tillage implements could be adapted by removing shanks that follow the tire tracks. The tillage implement would have to be the same width as the planter or drill.



Fig. 1 Permanent traffic lanes are easy to follow at planting.

BENEFITS

Timeliness of planting and harvesting is a major advantage of controlled traffic. Compacted paths provide a firm base for tractor and combine tires. Both flotation and traction are improved. Although not recommended, farmers with controlled traffic sometimes drive through standing water to plant without sinking or destroying soil structure. With controlled traffic, the absorption of water by the soil is greater except in the traffic lanes. Because the soil can store more water in the root zone, yields may improve in dry seasons, and runoff and erosion may be reduced.

Benefits related to chemical application may outweigh any other. Precisely spaced traffic lanes eliminate overlaps and uneven guess rows. Traffic lanes make it easy to spray at night, which usually improves the efficiency of the plant protection product. Eliminating waste from overlaps, and reduced yields from any gaps in application, may quickly pay for the extra investment in controlled traffic.

Controlled traffic helps retain any long-term benefits of subsoiling to alleviate compaction. With controlled traffic, repeated subsoiling is not needed because root zones under row crops are not compacted. Otherwise, as few as two passes of a tractor in the spring for shallow tillage and planting after fall subsoiling may recompact soil in the tire tracks to the same porosity as in non-subsoiled, non-trafficked areas. Wheel tracks become compacted even when relatively light machinery is used. With a typical plow-based cropping system, random trafficking would cover 75–90% of a field by the time planting is completed the second year, nearly eliminating the benefits of subsoiling.

Cotton research with no-till has shown yield reductions caused by shallow compaction, especially in a layer 2–4 in. (50–100 mm) below the surface.^[1] This surface compaction resulting from random wheel traffic could be minimized by controlled traffic.

With an automatic guidance system, the operator can devote attention to a machine’s performance instead of steering. The operator could literally look backward 90% of the time. Ease of driving minimizes fatigue allowing for longer work days.

IMPLEMENTING CONTROLLED TRAFFIC

Controlled traffic can work with, but is not limited to the following systems:

- continuous row crops;
- row crops in rotation with drilled beans or grain (requires skip rows);
- continuous small grains (using skip rows); and
- row crops/small grains, with cover crops.

Because the goal of controlled traffic is to drive on the least amount of soil, it is important that tire width be minimized and that wheels run in the fewest possible row middles (Table 1).^[2]

Corn and other Row Crops

Getting all wheels spaced properly is the major challenge with available equipment. For corn and other crops planted in rows, machinery could include a grain cart, manure spreader, fertilizer spreader, and sprayer. For continuous cotton, or continuous small grains, there are usually fewer machines to get aligned.

Cotton

Typical cotton rows are spaced at 38–40 in. (1 m). The wide rows allow adequate space for large tires. Where cotton is grown on beds or ridges, the maximum tire width would be about 10 in. less than the row spacing. A tire spacing of 80 in. (2 m) is a fairly easy adjustment for all equipment. Bigger tractors may have split duals spaced at 80 and 160 in. (2 and 4 m). Some cotton is grown in 30-in. (750-mm) rows making the wheel alignments similar to most corn production systems.

Site-specific subsoiling often improves cotton yields.^[3] Tilling just deep enough to eliminate the hard pan is compatible with a controlled traffic system and can provide the same yield advantage as subsoiling at a uniform depth.

Cotton is often planted in the same row position year after year, so controlled traffic is ideal. With controlled traffic, planters and cultivators would likely be twice as wide as the cotton harvester. Harvesters are usually 4-row, with some 6-row units.

Small Grains

For small-grain production, a major benefit from controlled traffic is in sprayer operation efficiency. Controlled traffic lanes allow the sprayer operator to drive accurately, even at night.

Examples

An example of an optimum system is the tractor and combine on single tires with the same spacing, making only two tracks with 8-row equipment (Figs. 2 and 3). The section width of the tires should be several inches less than the row spacing (e.g., 22 in. (550 mm) for 30-in. (750 mm) rows). Less than 20% of the soil would be trafficked, leaving more than 80% of the soil in good condition for root growth. Even a combine with split duals and a tractor with narrow wheel spacing (six tracks with 8-row equipment) would result in only about half of the soil being driven on. The



Fig. 2 An 8-row combine with single tires is a key part of one controlled traffic system.

rows between adjacent tracks would have a strip of uncompacted soil about 8 in. (20 mm) wide.

With narrow row spacings of 20 or 15 in. (50 or 38 mm), installing tires that can fit between the rows and still carry the load is a challenge; e.g., a tractor with split duals for 30 in. (750 mm) rows would likely need split triples for the narrower row spacing. This results in four rows with tire traffic on both sides. An option with 15-in. rows is to leave 30 in. spaces for tractor tires. Another is to keep the wide tires, realizing that for any operations after the crop emerges some damage will occur to the plants driven over. This last option may be better than the wide skip rows because of poor weed control in the unshaded space. Even a very low yield from the plants driven over is better than no yield (and weed problems) from a skip row.

All field equipment needs to cover multiples of the same width. Lighter implements, such as a sprayer or rotary hoe may cover double or triple the width of the planter or drill. The combine or cotton harvester might be one-half, or in some cases one-third, the planter width.

Break long fields in the middle for crop hauling unless a grain cart is used for on-the-go unloading. Any trip through



Fig. 3 The tractor and grain cart (and a sprayer) all have single tires spaced 10 ft. on centers, matching the combine.

the field should continue to the end, or to a cross road, instead of turning around a fraction of the way through.

PLANNING AHEAD

Adopting controlled traffic is not a simple change, but rather a transition that can take several years to complete. Consider controlled traffic in all major machinery buying decisions. Selecting wheels and tires to avoid running on rows adds little to the cost and can provide immediate benefits. Better yet, design a controlled traffic plan appropriate for your operation. Perhaps tires spaced 120 in. (3 m) would fit a farm with large machinery, and 80- or 90-in. (2 or 2.3 m) spacing would be ideal for a smaller farm. The combine header may define the operating width, with the planter being the same or, perhaps, double the width of the combine. (With a precise controlled traffic pattern, the combine can conveniently harvest half the planter width, centered on guess rows.)

The following are tips to consider when buying new equipment:

Choose a combine with tires that match the row spacing. If standard single tires are too wide, use split duals or tall single tires. Costs vary widely by manufacturer. Some combines require several mechanical modifications to accommodate tire changes, others do not. For ridge till, the maximum recommended tire width is 8 in. (20 mm) less than the row width. For no-till or mulch till, tires could be wider, perhaps 4 in. (10 mm) less than row width, but rows with traffic on both sides may lose yield.

Select a small-grain platform with the same width as the corn head on combines.

Buy a drill with the same width as the planter.

Choose pesticide sprayers and fertilizer applicators that cover the planter width, or double or triple that width.

Match grain cart tire spacings with the combine. Plan to run the cart in the same tracks as the previous pass of the combine. This may require an extension on the combine unloading auger.

An auto-steering system makes controlled traffic much easier. The additional cost may be minimal because auto-steering eliminates the need for row markers and foam markers, e.g., which are quite expensive for wide equipment.

CONCLUSION

Controlled traffic eliminates worries about soil compaction and improves overall farm efficiency and economics. Auto-steering is recommended, especially for large equipment, because it makes a controlled traffic system easier to set up and follow. Controlled traffic systems work with no-till and other tillage systems and with most major crops (corn, soybeans, cotton, wheat, and other small grains).

REFERENCES

1. Burmester, C.H.; Patterson, M.G.; Reeves, D.W. *Challenges of No-Till Cotton Production on Silty Clay Soils in Alabama*; Conservation Tillage Systems for Cotton Special Report 169; University of Arkansas: Fayetteville, 1995; 5 pp.
2. Grisso, R.D.; Jasa, P.J.; Jones, A.J.; Peterson, T.A. *Equipment Wheel Spacing for Ridge-Till and No-Till Row Crops*; University of Nebraska Cooperative Extension EC 96-780; University of Nebraska: Lincoln, 1996; 1 pp.
3. Raper, R.L.; Wayne Reeves, D.; Shaw, J.N.; Van Santen, E.; Mask, P.L. Site-specific subsoiling: Benefits for coastal plain soils. In *Proceedings of the 26th Southern Conservation Tillage Conference*, Auburn, AL, 2004; 96 pp.

Copper

Judith F. Pedler

*Department of Environmental Sciences, University of California—Riverside,
Riverside, California, U.S.A.*

David R. Parker

*Department of Soil and Environmental Sciences, University of California—Riverside,
Riverside, California, U.S.A.*

Abstract

Copper (Cu) is a metallic trace element with two naturally occurring isotopes (^{63}Cu and ^{65}Cu) and is essential for the growth of all life. Cu is naturally present in all soils and is found in excessive and potentially toxic concentrations in mine spoils, in waste products from industrial and agricultural activities, and in some agricultural soils due to historic use of Cu sprays (e.g., Bordeaux mix) for disease control. Cu deficiency occurs in many soils around the world, and the addition of Cu fertilizer is required for productive crop growth.

COPPER (Cu) IN SOILS

The average total concentration of Cu in the earth's crust is estimated to be 70 mg kg^{-1} although levels of $20\text{--}30 \text{ mg kg}^{-1}$ are prevalent in average soils.^[1] Common primary minerals include Cu sulfides, with Cu largely in the positive iodine oxidation state, which dissolve by weathering processes. Secondary minerals of Cu(II) include oxides, carbonates (malachite), silicates, sulfates, and chlorides, most of which are relatively soluble. Cu(II) may substitute for iron (Fe), magnesium (Mg), and manganese (Mn) in an assortment of minerals, especially silicates and carbonates.^[1]

The copper ions (Cu^{2+}) can form strong inner-sphere complexes and are thus immobilized by carboxylic, carbonyl, or phenolic functional groups, even at low pH. Exchangeable and weak acid-extractable Cu represents a small percentage of total Cu in most soils. The bulk of the Cu is complexed by organic matter, occluded in oxides, and substituted in primary and secondary minerals. Organic matter and Mn oxides are the most likely materials to retain Cu in a non-exchangeable form in soils. Alkali extraction techniques that remove organic matter from soils usually release large fractions of the total soil Cu.^[2] The addition of organic matter to soils and biological exudation of organic acids may increase dissolved organic carbon, thus solubilizing Cu from mineral forms, increasing the total dissolved Cu in soil solution,^[3] but predictive models of humic acid binding of Cu in soil solution are generally inadequate.^[4] Overall, Cu is one of the least mobile of the trace elements, maintained in a form sufficiently available to plants but relatively resistant to movement by leaching.^[1]

Free Cu^{2+} in soil solution decreases with increasing pH, reaching a minimum above pH 10. In the absence of organic ligands, Cu speciation is dominated by free

Cu^{2+} and increasingly by carbonate and hydroxy complexes as pH rises above 6.5.^[1] The dissolved organic carbon found in the most surface soils has a strong affinity for Cu, but estimates of the percentage of soluble Cu that is organically complexed can vary widely.

Cu AND PLANTS

Plant uptake of Cu appears to be directly related to the concentration of the free ion, Cu^{2+} , but may also be influenced by the total concentration in soil solution, including organic complexes.^[5] As with most trace metals, it is not known whether Cu is passively absorbed or actively taken up across the root-cell membrane.^[6] Rates of absorption are generally low, on the order of 1 nmol h^{-1} (g root dry weight)⁻¹.^[7] The activity of free Cu^{2+} required in nutrient solution for optimal plant growth is just 10^{-14} to 10^{-16} M .^[5] Cu absorption is generally halted by metabolic inhibitors and uncoupling agents which disrupt the normal transmembrane potential.^[7]

Uptake of Cu is strongly affected by pH: Increasing the concentration of hydrogen ions decreases the absorption of Cu ions by plant roots.^[8] Uptake is also affected by the presence of Ca, and to a lesser extent by Mg, both of which compete with Cu for binding sites at the root plasma-membrane.^[9] The effects of other trace metals on Cu uptake have frequently been seen as inhibitory (zinc), or stimulatory (Mn), but not necessarily under well-defined conditions.^[2]

Cu AS AN ESSENTIAL ELEMENT

Cu is an essential element for plant growth and is a component of many enzymes, including plastocyanin, and thus

is an indispensable prosthetic group in Photosystem 2. Cu-containing proteins are also important in respiration (cytochrome *c* oxidase is the terminal oxidase of the mitochondrial electron chain), in the detoxification of superoxide radicals (superoxide dismutase), and in lignification (polyphenol oxidase). Ascorbate oxidase, which contains eight Cu^{2+} ions, has been proposed as an indicator of plant Cu status, although its relevant biological function has yet to be determined.^[10]

The critical concentration of Cu in shoot tissue for optimal growth does not vary greatly between plant species, ranging from 1 to 6 $\mu\text{g g}^{-1}$ dry weight of young leaf tissue.^[11] Most crops are recorded as having a requirement of 3–5 $\mu\text{g g}^{-1}$.^[11] The average concentration of Cu in plant parts varies with age and with the level of Cu and nitrogen (N) supply. The translocation of Cu to plant shoots increases with an increasing supply of N. In the xylem and phloem saps, Cu is probably complexed by amino acids.^[2] Cu is usually described as having “variable” phloem mobility in plants,^[2] as the re-translocation of Cu from older tissues is regulated by both Cu supply and N status. Lack of a sufficiently long-lived radioisotope makes study of Cu transport and translocation problematic.

PLANT GROWTH ON Cu-DEFICIENT SOILS

Cu deficiency most often occurs on organic soils where excessive leaching has occurred, or on calcareous sands. In general, crops grown on mineral soils with Cu contents less than 4–6 $\mu\text{g g}^{-1}$ or on organic soils with less than 20–30 $\mu\text{g g}^{-1}$ are the most likely to suffer Cu deficiency^[12] although this varies with specific soil type and the crop grown.

Symptoms of Cu deficiency include chlorosis, necrosis, leaf distortion, and terminal dieback and are most evident in new leaf growth. Wilting can also occur, indicating structural weaknesses due to reduced lignification of the xylem elements. These symptoms are not entirely specific to Cu deficiency and can be observed in plants under a variety of stresses. The most profound symptoms of Cu deficiency are those seen in the reproductive cycles of many sensitive species: delay of flowering, and/or reductions in seed and fruit yield as a consequence of sterile pollen or reduced floret numbers. Because these latter symptoms are not observed until maturity or harvest, Cu deficiency is often termed a “hidden hunger.” Rice, citrus, and cassava are sometimes referred to as indicator species that are more sensitive to Cu deficiency but are not reported to require more than 5–6 $\mu\text{g g}^{-1}$ Cu to avoid Cu deficiency. Cereal rye and canola are crops more tolerant of Cu deficiency, requiring only 1–2 $\mu\text{g g}^{-1}$ Cu.^[11]

Cu deficiency in legumes depresses nodulation and N_2 fixation, leading to N deficiencies. Unlike molybdenum and cobalt, there seems to be no specific Cu requirement

for N_2 fixation in nodules beyond that required for plant growth and the production of carbohydrates.^[13]

Cu deficiency decreases polyphenol oxidase activity and thus lignification.^[14] Susceptibility of Cu-deficient plants to pathogenic attack may be increased due to reduced lignification of xylem elements or due to impaired lignification in response to pathogenic invasion (wounding response). The application of Cu to soil, at rates too low to directly affect the pathogen, has controlled powdery mildew in wheat.^[15] It has also been suggested that, where Cu in soils is more than sufficient, the accumulation of additional Cu in roots provides a fungistatic defense against pathogens.^[16] Conversely, where Cu is deficient, roots are more vulnerable to pathogenic invasion.

There are genetic differences in the absorption of Cu by plant roots. Rye is able to take up significantly more Cu from soil than wheat and is thus viewed as being more Cu-efficient. Triticale, the wheat-rye hybrid, inherits the efficiency factor.^[7] Cu efficiency could be a useful trait in breeding crops for regions where soils are commonly Cu-deficient.^[17] However, the mechanism of the efficiency factor is not clear.

PLANT GROWTH ON HIGH-Cu SOILS

As some plant species have adapted to soils of low Cu status, others have evolved tolerance to Cu-toxic conditions. The 16th century author, Agricola, wrote of indicator plants that grow on naturally Cu-rich soils. The natural revegetation of mine spoils has been shown to reflect rapid genetic evolution of Cu tolerance by grasses and other plants.^[14] There seem to be several possible mechanisms of Cu tolerance, although exclusion from shoots is a common feature. The exceptions are a few Cu-accumulator species which may contain in excess of 1000 $\mu\text{g g}^{-1}$ Cu in shoot tissue.^[12] In other Cu-tolerant species, the root compartmentation or immobilization of Cu may be achieved by immobilization in cell walls, by complexation with intracellular proteins, or by removal of Cu to the vacuole.^[14]

With non-tolerant taxa, plant growth is likely to be depressed when Cu concentrations in the whole shoots exceed $\sim 20 \mu\text{g Cu g}^{-1}$. Symptoms of Cu toxicity include poorly developed and discolored root systems, reduced shoot vigor, and leaf chlorosis. Toxicity thresholds (e.g., for a 10% yield reduction) seem to vary widely, probably because of the low translocation of Cu from roots to shoots. Only when roots are overwhelmed by Cu, rhizotoxicity does sufficient Cu to reach shoots to affect growth and function. Other syndromes, such as Fe deficiency, can readily occur as secondary consequences of excess Cu.^[14]

The exclusion of Cu from the shoot protects photosynthetic activity, which is highly sensitive to excess Cu. Photosynthetic electron transport is blocked by high levels of free Cu, at the oxidizing side of Photosystem 2, and at the reducing side of Photosystem 1. Excess Cu supply results

in reduced lipid content and noticeable changes in the fatty acid composition of tomato roots and primary leaves, indicating enhanced activity of enzymes that catalyze lipid peroxidation.^[18]

Concentrated Cu sprays have historically been used to control foliar pathogens, especially in vineyard and orchard crops. These fungicidal sprays often included limestone to reduce their phytotoxicity and to make them more rain fast. The accumulation of Cu in the soils under these crops has not regularly caused Cu toxicity, which indicates the remarkable ability of most plant roots to accumulate Cu, while regulating its flow to the shoots. Both Cu deficiency and toxicity may result in non-specific symptoms of plant stress. Assessment of the Cu status of a soil, or of crop plants, is most accurate when soil type, soil history, and soil and plant analyses for Cu are considered.

REFERENCES

1. McBride, M.B. Forms and distribution of copper in solid and solution phases of soil. In *Copper in Soils and Plants*; Loneragan, J.F., Robson, A.D., Graham, R.D., Eds.; Academic Press: Sydney, 1981; 25–45.
2. Loneragan, J.F. Distribution and movement of copper in plants. In *Copper in Soils and Plants*; Loneragan, J.F., Robson, A.D., Graham, R.D., Eds.; Academic Press: Sydney, 1981; 165–188.
3. Sanders, J.R.; McGrath, S.P. Experimental measurements and computer predictions of copper complex formation by soluble soil organic matter. *Environ. Pollut.* **1988**, *49*, 63–79.
4. Robertson, A.P.; Leckie, J.O. Acid/base copper binding, and $\text{Cu}^{2+}/\text{H}^+$ exchange properties of a soil humic acid, an experimental and modelling study. *Environ. Sci. Technol.* **1999**, *33*, 786–795.
5. Bell, P.B.; Angle, J.S.; Chaney, R.L. Free metal activity and total metal concentrations as indices of micronutrient availability to barley [*Hordeum vulgare* (L.) cv. 'Klages']. *Plant Soil* **1991**, *130*, 51–62.
6. Strange, J.; Macnair, M.R. Evidence for a role for the cell membrane in copper tolerance of *Mimulus guttatus* Fischer ex DC. *New Phytol.* **1991**, *119*, 383–388.
7. Graham, R.D. Absorption of copper by plant roots. In *Copper in Soils and Plants*; Loneragan, J.F., Robson, A.D., Graham, R.D., Eds.; Academic Press: Sydney, 1981; 141–163.
8. Lexmond, Th.M.; van der Vorm, P.D.J. The effect of pH on copper toxicity to hydroponically grown maize. *Neth. J. Agr. Sci.* **1981**, *29*, 209–230.
9. Parker, D.R.; Pedler, J.F.; Thomason, D.N.; Li, H.Y. Alleviation of copper rhizotoxicity by calcium and magnesium at defined free metal-ion activities. *Soil Sci. Soc. Am. J.* **1998**, *62*, 965–972.
10. Maksymiec, W. Effect of copper on cellular processes in higher plants. *Photosynthetica* **1997**, *34*, 321–342.
11. Reuter, D.J.; Robinson, J.B., Eds. *Plant Analysis: An Interpretation Manual*; CSIRO Press: Collingwood, Australia, 1997; 83–566.
12. Jarvis, S.C. Copper concentrations in plants and their relationship to soil properties. In *Copper in Soils and Plants*; Loneragan, J.F., Robson, A.D., Graham, R.D., Eds.; Academic Press: Sydney, 1981; 265–285.
13. Römheld, V.; Marschner, H. Function of micronutrients in plants. In *Micronutrients in Agriculture*, 2nd Ed.; Mortvedt, J.J., Cox, F.R., Shuman, L.M., Welch, R.M., Eds.; SSSA: Madison, 1991; 297–328.
14. Woolhouse, H.W.; Walker, S. The physiological basis of copper toxicity and copper tolerance in higher plants. In *Copper in Soils and Plants*; Loneragan, J.F., Robson, A.D., Graham, R.D., Eds.; Academic Press: Sydney, 1981; 235.
15. Graham, R.D. Susceptibility to powdery mildew (*Erysiphe graminis*) of wheat plants deficient in copper. *Plant Soil* **1980**, *56*, 181–185.
16. Graham, R.D.; Webb, M.J. Micronutrients and disease resistance and tolerance in plants. In *Micronutrients in Agriculture*; Mortvedt, J.J., Cox, F.R., Shuman, L.M., Welch, R.M., Eds.; Soil Science Society of America: Madison, 1991; 329–370.
17. Owuochi, J.O.; Briggs, K.G.; Taylor, G.J. The efficiency of copper use by 5A/5RL wheat-rye translocation lines and wheat (*Triticum aestivum* L.) cultivars. *Plant Soil* **1996**, *180*, 113–120.
18. Ouairiti, O.; Boussama, N.; Zarrouk, M.; Cherif, A.; Ghorbal, M.H. Cadmium and copper-induced changes in tomato membrane lipids. *Phytochemistry* **1997**, *45*, 1343–1350.

Cover Crops

Jorge A. Delgado

Soil Plant Nutrient Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.

D. Wayne Reeves

J. Phil Campbell Sr. Natural Resource Conservation Center, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Watkinsville, Georgia, U.S.A.

Ronald F. Follett

Soil Plant Nutrient Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.

Abstract

Humanity is facing a large number of challenges in the 21st century, including population growth and a changing climate. The benefits of cover crops (CCs) are very extensive, and CCs can provide a large number of ecosystem services. CCs can be used to cover the surface soil and reduce the off-site transport of soil particles, nutrients, organic matter, and agrochemicals. This makes CCs key tools for maintaining and/or improving soil quality and soil health and great tools to increase carbon sequestration. They are nutrient management tools that can help scavenge nitrate (NO₃), cycle nitrogen (N) to the following crop, mine NO₃ from groundwater, and increase N use efficiency of cropping systems. CCs will be key soil and water conservation tools that could be used across different worldwide agroecosystems to protect soil and water quality and to help in climate change mitigation and adaptation during the 21st century.

INTRODUCTION

Cover crops (CCs) are key soil and water conservation tools that can be used across a large number of worldwide agroecosystems to protect soil and water quality.^[1] The benefits of CCs are very extensive. CCs can be used to provide a large number of ecosystem services. They are tools that can be used for different objectives to provide positive agricultural and environmental impacts. A typical use of CCs is their use to maintain the soil surface cover to reduce soil erosion potential from wind and/or water. CCs can be used to reduce the off-site transport of soil particles, nutrients, organic matter, and agrochemicals.^[2,3] This makes CCs key tools for maintaining and/or improving soil quality and soil health.

CCs are also excellent tools for nutrient management.^[4] They can help scavenge residual soil nitrate (NO₃), reduce NO₃ leaching, and even mine NO₃ from groundwater, helping to reclaim groundwater. CCs can be used to increase cropping system nitrogen (N) use efficiency to protect water quality.^[1,4] CCs can also contribute to increased populations of beneficial insects, weed management and suppression, and pest management. By serving as biocontrol tools, CCs can contribute

to decreased use of agrochemicals and protect water and environmental quality.^[1,5–7]

CCs can be used as a precision conservation practice to manage spatial and temporal variability to increase conservation effectiveness.^[8] CCs can be one of the practices used with a 7 Rs approach to increase nutrient use efficiency and reduce transport of nutrients off-site.^[9] They can be used as scavenger crops and be one of the conservation/nutrient management tools to increase nutrient use efficiencies spatially. CCs can show spatial variation in the amount of scavenged N as well as in the amount of N cycled to the subsequent crop. There is potential to use management zones and precision information to consider these spatial differences when using CCs to account for different N budgets to increase crop N use efficiency while reducing NO₃ leaching losses to the environment and to protect water quality.^[10]

Summer CCs with limited irrigation can potentially be used to increase yields, crop quality, and nutrient and water use efficiencies while protecting the environment.^[11,12] Similarly, winter CCs can also contribute to protection of the environment by reducing the transport of nutrients off-site, and for some cropping systems, they can contribute to increased yields and nutrient use efficiencies.^[4,13] However, summer or winter CCs or CCs in general, independent

of the time of planting, could also potentially contribute to reduced yields of the subsequent crop.^[4,12,13] This could be due to an allelopathic effect, nutrient and water availability, length of the CC season, or a potentially shorter growing season of the subsequent crop, among other possibilities.^[4,12,14] The growing season of the grain (e.g., corn and soybean) may not provide enough time to plant the CC. The user of the CCs needs to evaluate what CC to use and how to manage the CC to avoid potential negative impacts.

There are several factors that can contribute to a negative impact; one of them is incorporation of a CC that has a high carbon-to-nitrogen ratio into the rotation, which could potentially immobilize available N for the subsequent crop, having a negative impact on yields. A CC could use available water stored in the soil profile, reducing available water for the subsequent crop and negatively impacting yields, especially for cropping systems grown in drier climates. Time needed for seeding and establishment of the CCs and for killing and incorporating the CC and preparing the land for the following crop could potentially shorten the growing season, also impacting yields. Fortunately, there are different management practices that can be used to help CCs avoid impacting, or even help them increase, the yields of the following crop while providing a large number of ecosystem services.^[13]

Leguminous CCs such as vetch can be used to add N to a system, since they can fix atmospheric N. Non-leguminous grain CCs such as winter cover rye can be used to scavenge residual soil NO₃ from the deep layers of the soil profile. There are different management strategies to use CCs under different management systems such as no till, minimum tillage, and conventional tillage systems.^[13] CCs can be grazed, harvested for hay, used as mulch or incorporated as a green manure.^[13] CCs can be killed earlier when they have a lower carbon-to-nitrogen ratio or can be killed at a later date to let the CC increase the amount of biomass before it is harvested, or incorporated as a green manure, or left over the surface to make a mulch.^[1,5,13]

One of the hardest management problems to overcome may be the available water used by a CC under a drier system.^[14] In general, the benefits of CCs are so numerous that nutrient managers, conservation practitioners, and others use them as tools to increase nutrient use efficiencies; reduce off-site transport of soil particles, nutrients, and agrochemicals; recover soil NO₃-N from the soil profile and cycle to the next crop; improve soil, air, and water quality; and reduce pests and weeds. The potential of CCs to improve yields is one reason CCs are a key soil and water conservation practice.^[1,5,13] Fortunately, for the majority of cropping systems, management can potentially be used to minimize the potential negative impacts of CCs on yields by using the right CC for the right crop situation for a given combination of soil, weather, and cropping system.^[13] CC management is essential to provide all the benefits of CCs while maintaining and/or increasing yields of the subsequent crop.^[13]

CONSERVATION OF WATER QUALITY

CCs can be one of the tools to apply the “right product, right rate, right method, right practice, right place, right scale, and right time: the 7 Rs of nutrient management and conservation.”^[9] In a 7 Rs approach, using a CC as a scavenger crop in a rotation we can apply the right product (fertilizer), at the right fertilizer rate, with the right method of fertilizer application for the crop. CCs could be the right conservation practice to scavenge NO₃ and reduce erosion potential. Considering spatial variability, a leguminous CC could be planted in some areas of the field with lower leaching potential, whereas a non-leguminous CC could be planted in the other areas of the field with a higher leaching potential, thus applying a conservation practice at the right place, and at the right scale of conservation practice by applying the CCs across the whole field. It will be necessary to apply both the fertilizer and the conservation practice at the right time. CCs could be planted right after harvesting the crop to maximize the potential of the CC to be used as a catch crop to recover the residual soil NO₃ while reducing the potential for erosion. Summer or winter CCs could be used to apply the right conservation practice at the right time.

CCs can have a significant impact on conservation of water quality across the United States.^[1,5] Delgado et al.^[11]

Reported on the potential regional impacts of CCs on conservation of groundwater quality. CCs can potentially be used to minimize NO₃ leaching to groundwater and even mine NO₃ from groundwater, serving as a reclamation tool.

Kladivko et al.^[15] reported on the positive impacts of CCs on water quality of the Mississippi River watershed. The Mississippi River basin contributes to loss of NO₃ to the Gulf of Mexico and to the hypoxia problem in the Gulf of Mexico. Kladivko et al.^[15] discussed the potential adoption of CCs to reduce NO₃ leaching from the Midwestern United States.^[15,16] Kladivko et al.^[15] reported that CCs can have significant positive impacts on water quality across Ohio, Indiana, Illinois, Iowa, and Minnesota. Their analysis of 10 representative counties across these five states showed that from 34% to 81% of the agricultural land planted to corn and soybean rotations could use fall-planted winter cover rye to reduce NO₃ leaching to the Mississippi River by 20%. The winter cover rye could be established with overseeding at main crop maturity in continuous corn and corn-soybean systems on tile-drained lands managed with no-till, spring-till, or fall-till that could be transitioned to spring tillage.^[15] Winter cover rye was one of the CCs reported to recover NO₃ from deep in the soil profile, reducing NO₃ leaching and mining NO₃ from groundwater.^[10] Modeling tools such as NLEAP-GIS 4.2 (Nitrogen Loss and Environmental Assessment Package with GIS Capabilities) and RZWQM (Root Zone Water Quality Model) can be used to assess the benefits of CCs across regions.

CONSERVATION OF SOIL QUALITY

CCs can contribute to conservation of soil quality and soil health.^[6] A key way that CCs contribute to soil health and quality is by reducing erosion. Raindrops impacting a bare soil surface can contribute to the breaking down of aggregates and loosening of soil particles, facilitating their movement in surface water runoff. As water flows over the soil surface, it can also gain energy and increase the kinetic force to dislodge more particles and transport more soil particles off-site. By covering the soil surface with CCs, the plant biomass intercepts and reduces the force of raindrops. The CC biomass that covers the surface slows down the water runoff, reducing its force and velocity as it moves across the surface, contributing to protection of the soil. CC protection to slow down water movement is especially important in those areas of the field with steeper slopes where water may flow downhill and there is more susceptibility to erosion. The belowground root system is also important and can help stabilize the soil and hold it together. Another pathway for surface transport is wind erosion, which can also contribute to the movement of fine particles out of the soil. CCs that cover the soil surface can also protect against wind erosion forces; reducing the off-site transport of soil particles, nutrients, and soil organic matter; and contributing to conservation of soil quality and soil health.

CCs can contribute to increased carbon sequestration and improved soil quality and soil health.^[6,13,17] Increases in carbon sequestration from CCs contribute to improvements in soil quality and health with greater nutrient cycling, more soil aggregates, improved water holding capacity, and higher cation exchange capacity.^[6] CCs also have positive impacts on soil biology, increasing the diversity of microorganisms.^[6]

CONSERVATION OF AIR QUALITY

CCs protect air quality by reducing wind erosion and the aerial transport of soil particles and agrochemicals that may be tied to this process. Emissions of greenhouse gases are also reported to negatively impact air quality. One of the important greenhouse gases is nitrous oxide (N_2O), the emissions of which contribute to the greenhouse effect and climate change. CCs could potentially contribute to reduction of N_2O emissions.^[18] A meta-analysis of the use of CCs found that non-leguminous CCs reduce N_2O emissions. Although leguminous crops have the potential to increase N_2O emissions, when N_2O measurements were monitored over the whole year, on average the impacts of CCs on N_2O emissions ranged from negligible to a small increase in N_2O emissions.^[18] It is also important to point out that leguminous CCs could fix atmospheric N, which can be cycled to the next crop, reducing the amount of N fertilizer applied to the next crop. CCs also reduced the

potential for NO_3 leaching, reducing indirect emissions of N_2O . The data suggest that there is potential to use CCs to reduce N_2O emissions.^[18]

Delgado, Del Grosso, and Ogle^[19] conducted an analysis of the N losses from inorganic fertilizer versus the N losses from CCs' N content using published ^{15}N data and simulation modeling. Delgado, Del Grosso, and Ogle^[19] found that the average percentage of N loss (N that is not found in the following crop or in the soil profile) is 33%. The data from the ^{15}N -labeled CC residue showed that only 10% was lost from the system. The average amount of N fertilizer lost from the system is 66% greater than the N lost from the CC. So the losses via N_2O emissions from the CC were assumed to be lower than those from the fertilizer when comparing losses from the ^{15}N -labeled N fertilizer versus losses from the added ^{15}N -labeled CC residue. This was in agreement with model simulations of the ^{15}N -labeled N fertilizer and of the ^{15}N -labeled CC that showed lower N_2O emissions with the CC system. There is an opportunity to use CCs as a climate change mitigation and adaptation practice.^[19]

CONCLUSION

CCs are great tools to improve soil quality and health^[6] and great tools to increase carbon sequestration.^[17] They are nutrient management tools that can help scavenge NO_3 , cycle N to the following crop, mine NO_3 from groundwater, and increase N use efficiency of cropping systems. N cycling from CCs is more efficient than from inorganic N fertilizer, and CCs act like a slow-release fertilizer, with 66% lower losses than the losses from inorganic N fertilizer.¹⁹ CCs can be grazed or harvested for hay, used as a mulch, or incorporated as green manures. CCs can contribute to suppression of weeds and pests.^[7] CCs are excellent tools for soil and water conservation and climate change adaptation and mitigation. To maximize these positive effects of CCs and to avoid any negative impacts on yields, CC users need to determine the right CC for the right crop rotation and how to manage the CC. In other words, "There is a need to use the right cover crop or cover crop mixture, select the right time to plant and harvest or kill the cover crop, and apply the right cover crop management practices at the right location, to increase the benefits of cover crops (the 4Rs for cover crops)."^[20]

REFERENCES

1. Dabney, S.M.; Delgado, J.A.; Reeves, D.W. Using winter cover crops to improve soil and water quality. *Commun. Soil Sci. Plan.* **2001**, *32*, 1221–1250.
2. Reeves, D.W. Cover crops and rotations. In *Crops Residue Management*; Hatfield, J.L., Stewart, B.A., Eds.; Advances in Soil Science; Lewis Publishers: Boca Raton, FL, 1994; 125–172.

3. Langdale, G.W.; Blevins, R.L.; Karlen, D.L.; McCool, D.K.; Nearing, M.A.; Skidmore, E.L.; Thomas, A.W.; Tyler, D.D.; Williams, J.R. Cover crop effects on soil erosion by wind and water. In *Cover Crops for Clean Water*; Hargrove, W.L., Ed.; Soil and Water Conservation Society: Ankeny, 1991; 15–21.
4. Thorup-Kristensen, K.; Magid, J.; Jensen, L.S. Catch crops and green manures as biological tools in nitrogen management in temperate zones. *Adv. Agron.* **2003**, *79*, 227–302.
5. Dabney, S.M.; Delgado, J.A.; Meisinger, J.J.; Schomberg, H.H.; Liebig, M.A.; Kaspar, T.; Mitchell, J.; Reeves, W. Using cover crops and cropping systems for nitrogen management. In *Advances in Nitrogen Management for Water Quality*; Delgado, J.A., Follett, R.F., Eds.; Soil and Water Conservation Society: Ankeny, 2010; 230–281.
6. Sarrantonio, M. Building soil fertility and tilth with cover crops. In *Managing Cover Crops Profitably*, 3rd Ed.; Clark, A., Ed.; Sustainable Agriculture Research and Education, United Book Press, Inc.: College Park, MD, 2007; 16–24.
7. Phatak, S.C.; Diaz-Perez, J.C. Managing pest with cover crops. In *Managing Cover Crops Profitably*, 3rd Ed.; Sustainable Agriculture Research and Education, United Book Press, Inc, 2007; 25–33.
8. Berry, J.; Delgado, J.A.; Khosla, R.; Pierce, F. Precision conservation. *J. Soil Water Conserv.* **2003**, *58* (6), 332–339.
9. Delgado, J.A. 4 Rs are not enough: We need 7 Rs for nutrient management and conservation to increase nutrient use efficiency and reduce off-site transport of nutrients. In *Soil-Specific Farming Precision Agriculture*; Lal, R., Stewart, B.A., Eds.; CRC Press: Boca Raton, FL, 2015; 89–126.
10. Delgado, J.A. Sequential NLEAP simulations to examine effect of early and late planted winter cover crops on nitrogen dynamics. *J. Soil Water Conserv.* **1998**, *53*, 241–244.
11. Delgado, J.A.; Dillon, M.A.; Sparks, R.T.; Essah, S.Y.C. A decade of advances in cover crops: Cover crops with limited irrigation can increase yields, crop quality, and nutrient and water use efficiencies while protecting the environment. *J. Soil Water Conserv.* **2007**, *62*, 110A–117A.
12. Essah, S.Y.C.; Delgado, J.A.; Dillon, M.; Sparks, R. Cover crops can improve potato tuber yield and quality. *HortTechnology* **2012**, *22*, 185–190.
13. Clark, A. *Managing Cover Crops Profitably*, 3rd Ed.; Sustainable Agriculture Research and Education; United Book Press, Inc.: Beltsville, 2007.
14. Unger, P.W.; Vigil, M.F. Cover crop effects on soil water relationships. *J. Soil Water Conserv.* **1998**, *53*, 200–207.
15. Kladvko, E.J.; Kaspar, T.C.; Jaynes, D.B.; Malone, R.W.; Singer, J.; Morin, X.K.; Searchinger, T. Cover crops in the upper midwestern United States: Potential adoption and reduction of nitrate leaching in the Mississippi River Basin. *J. Soil Water Conserv.* **2014**, *69*, 279–291.
16. Malone, R.W.; Jaynes, D.B.; Kaspar, T.C.; Thorp, K.R.; Kladvko, E.; Ma, L.; James, D.E.; Singer, J.; Morin, X.K.; Searchinger, T. Cover crops in the upper midwestern United States: Simulated effect on nitrate leaching with artificial drainage. *J. Soil Water Conserv.* **2014**, *69*, 292–305.
17. Sainju, U.M.; Singh, B.P.; Whitehead, W.F. Long-term effects of tillage, cover crops, and nitrogen fertilization on organic carbon and nitrogen concentrations in sandy loam soils in Georgia, USA. *Soil Till. Res.* **2002**, *63*, 167–179.
18. Basche, A.D.; Miguez, F.E.; Kaspar, T.C.; Castellano, M.J. Do cover crops increase or decrease nitrous oxide emissions? A meta-analysis. *J. Soil Water Conserv.* **2014**, *69*, 471–482.
19. Delgado, J.A.; Del Grosso, S.J.; Ogle, S.M. ¹⁵N Isotopic crop residue cycling studies suggest that IPCC methodologies to assess N₂O-N emissions should be reevaluated. *Nutr. Cycl. Agroecosyst.* **2010**, *86*, 383–390.
20. Delgado, J.A.; Gantzer, C.J. The 4Rs for cover crops and other advances in cover crop management for environmental quality. *J. Soil Water Conserv.* **2015**, *70*, 142A–145A.

Crop Evapotranspiration: Measurement System

Huanjie Cai

Pute Wu

Zhijun Li

Xiaotao Hu

Yufeng Zou

*Institute of Water Saving Agriculture in Arid Areas of China,
Northwest A&F University, Yangling, China*

Abstract

Crop evapotranspiration is an important parameter in research of water conservancy, meteorology, soil science, natural geography, and other related disciplines and fields. The system for measuring crop evapotranspiration mainly consists of three parts: test pit, lysimeter, and standardized evapotranspiration measurement test field. It is characterized by the following features: 1) two large weighing lysimeters are installed, not only to realize normalized automatic measurement and observation, but also to conduct data correction by measuring a large area with high precision; 2) the test pit and lysimeter cylinders are installed with a high-precision system for the measurement of soil water content, soil water potential, and soil temperature via the water balance method can be employed for the self-correction of measured data; 3) the large-size test pit groups and high-precision lysimeters are used to conduct a comparative study on different evapotranspiration measurement methods and to perform a mutual verification between the field measurement of crop evapotranspiration and by other methods; and 4) relatively large numbers of test pit groups and rainproof systems are used to conduct the evapotranspiration test on different moisture, nutrient gradients, and different crops and to combine the evapotranspiration measurements with the meteorological data to determine the locally representative and important parameters such as crop water requirement, crop coefficient, and irrigation system.

INTRODUCTION

Crop evapotranspiration refers to the sum of the soil/watering surface evaporation between plants and the transpiration of plants.^[1] Generally, four methods are employed to measure the evapotranspiration of dry land crops. This involves a combination of the lysimeter method, the pit checking method, and the field measurement method, which follow the water balance principle to measure evapotranspiration of crops, and the instrument measurement method. The lysimeter method measures the evapotranspiration of crops cultivated in soil tanks through weighing, and the pit checking method and the field measurement method both use specially built test pits or fields to monitor the water balance equation of the soil. The instrument measurement method introduces systems such as the eddy covariance and Bowen ratio system to measure the evapotranspiration by applying the principles of energy balance and water vapor diffusion.

Of the four methods, the lysimeter method has high precision and can rapidly and accurately reflect the variation process of crop evapotranspiration. However, the design of lysimeter measurement and control system and the corresponding means of measurement exert a

significant influence on the results. The pit checking method has a relatively high precision and is used to measure evapotranspiration at weekly or monthly time scales, in which case the selection of the number and form of test pits is of great importance. The field measurement method is carried out in the field by the water balance method, therefore the test fields must be representative of the growing conditions, as the accuracy of the results measured under restricted conditions in a small area cannot be guaranteed.^[2] The field instrument measurement method requires uniform and open test fields and can be subject to the influence of surrounding environment.

Establishing a precise crop evapotranspiration measurement system is of great importance for studying the water consumption law and the spatial transformation of precipitation (irrigation water) to soil water, underground water, plant water, and subsequently to atmospheric water.

1. Crop water requirement measurement system using a lysimeter

The lysimeter is a large-size vessel that is loaded with soil, placed in the fields or under the ground, and exposed on the surface or covered with plants to simulate the

growth environment of the field to measure the evapotranspiration of growing crops or the evaporation of uncovered soil. In essence, the device measures the transformation of water bodies under restricted 3-D boundary conditions. The system consists of three parts, the measurement apparatus, data transmission, and terminal display. The measurement apparatus is constituted of soil, steel cylinder, weighing system platform, weighing system, weighing sensor, and displacement sensor. The data transmission part is constituted of the front-end amplifier and data conversion module. Finally, the terminal display part is constituted of lysimeter host, backup power source, and computer bundled software.

a. Lysimeter measurement principle

The water balance equation of the soil column of the lysimeter is calculated as follows:

$$\Delta S = P + I + Q - \Delta R - ET \quad (1)$$

ΔS represents the variation of water stored by the soil, P represents the precipitation, I represents the irrigation, Q represents the underground water flow, ΔR represents the net surface runoff, and ET represents the evapotranspiration. All of the above variables adopt millimeter as their unit.

For a lysimeter, generally ΔR can be neglected, so Eq. (1) can be simplified into:

$$ET = P + I - Q - \Delta S \quad (2)$$

P and I can be directly measured with a rain gauge and water meter. ΔS represents the increase in moisture after precipitation (or irrigation) or the loss of moisture caused by evapotranspiration. Given that this can be a difficult variable to measure, a high-precision weighing system has been developed for the measurement of ΔS .

Q represents the water supplied into and drained out of the soil column through the water supply/drainage system of the lysimeter. In cases where the underground water level remains constant, the water supplied into the soil column is the replenishment of soil water by underground water (E_g) and is represented by $E_g = Q$. The water drained out of the soil column is the replenishment of underground water (R_g) and is represented by $R_g = Q$. When the underground water levels vary in the field, a certain amount of water must be supplied or drained from the soil column to ensure that the water level of the instrument and the fields are consistent. In these cases, the replenishment of soil water can be calculated indirectly.

If the underground water level has risen by ΔH :

$$E_g = Q - a \cdot \Delta H \quad (3)$$

If the underground water level has declined by ΔH :

$$E_g = b \cdot \Delta H - Q \quad (4)$$

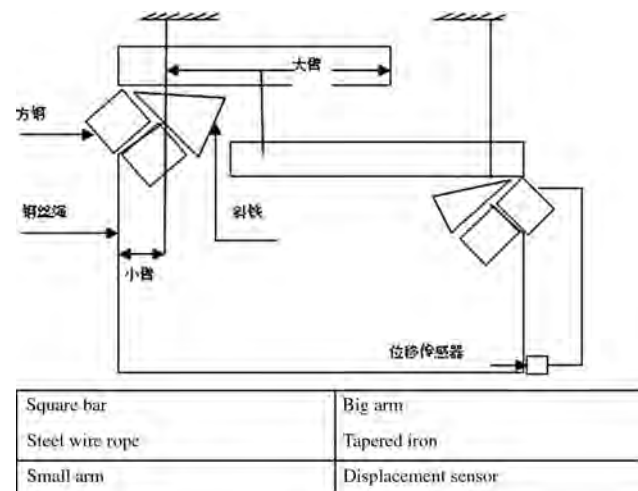


Fig. 1 Schematic diagram of the weighing mechanism principle.

Where a and b are the coefficients measured in the water absorption and dehydration tests. If $a = b = \Delta Q$, then ΔQ represents the variation of water content of underground water levels and can be measured with the moisture meter.

b. Weighing system principle

Fig. 1 depicts the weighing system principle. It is clamped by two square bars in the middle of the steel wire rope and works in concert with two tapered iron blocks and the balance arm to create the working state. When the balance arm is at the horizontal level, the steel wire rope produces a vertical dislocation (datum zero). When the weight of the soil mass varies, the balance arm swings up and down to produce a variation of swing angle, which is then transformed into linear displacement via the rigid link mechanism. Following that, a high-resolution displacement sensor is used to measure the variation of displacement, thus measuring the weight variation of soil box.

c. Data collection system principle

The signal amplifier can collect the signals of the displacement sensor and percolation sensor, amplify them into digital signals, and transmit these signals to the data collector via RS485. The data collector adopts single-chip microcomputer as CPU control board and uses Flash ROM as data-recording medium. This system can continuously store 30,000 groups of data, each of which consisting of displacement, percolation, temperature compensation, as well as the corresponding time point of the measurement. The host computer can obtain data from the data collector via RS-232 on a real-time or archival basis and compile them into data, curves, or sheets in standard formats, which are then to be analyzed and filed by the experimenter.

There is a linear relationship between the weight variation of the lysimeter's soil box and the displacement variation of the displacement sensor. The automatic data

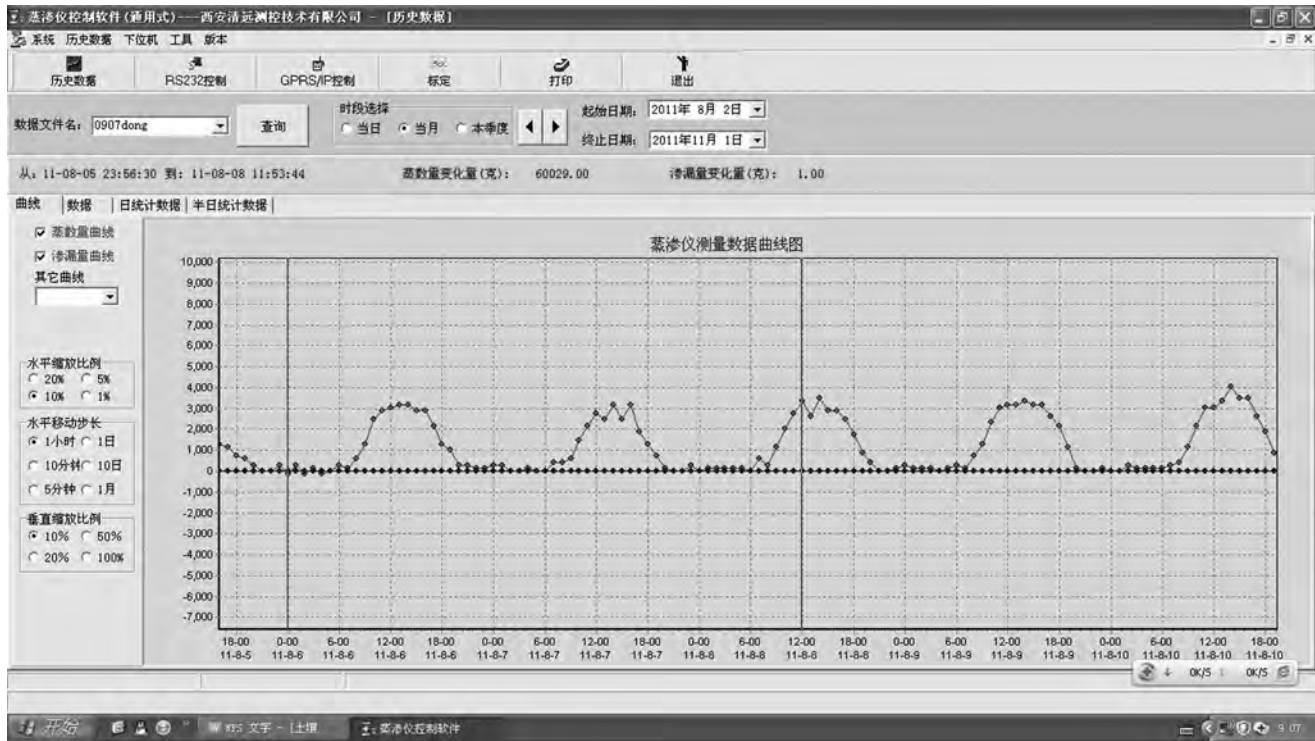


Fig. 2 Evapotranspiration variation curve.

collection system consists of signal amplifier and data collector. After obtaining the displacement variation through a group of displacement sensors, the coefficient obtained through calibration is then used to calculate the weight variation of the corresponding soil box, which corresponds to the variation of the water storage.

A percolation sensor is installed at the bottom outlet of the soil box to measure the variation of percolation. The evapotranspiration is calculated from water storage and percolation. The evapotranspiration is obtained after subtracting the variation of percolation from the variation of water storage. As can be seen from Eq. (2), the actual transpiration of the system includes precipitation and irrigation; see the measured figure lines in Fig. 2.

2. Crop evapotranspiration and irrigation water circulation measurement system of test pits

The system is built with a total of 44 test pits, 4 of which are bottomless test pits. The test pit system consists of standardized test pits, data collection corridor, and automatic mobile rain shed, occupying a total construction area of 640 m² and is installed with underground soil solution collection device and time domain reflectometry (Trime-IPH) TDR soil moisture measurement device (see the upper and lower parts of test pits in Fig. 3).

Each test pit has an area of 2 × 3.33 m² and a depth of 3 m, is enclosed on all sides with reinforced concrete, and provided with waterproof treatment. The wall of the test pit



Fig. 3 Aboveground and underground parts of test pits.

exposed above the ground has a height of 8 cm. The soil is backfill soil, and the bottom of the test pit is paved with an inverted filter layer 20 cm in thickness and consisting of sand and gravels. The base is provided with a lateral drain-pipe, connected with the drainage collection system. Soil backfilling must be done in layers that are in strict accordance with the unit weight of the original soil layers. The soil in the bottomless test pits must be undisturbed soil, as this is a key link in ensuring the representativeness of the test results of these test pits. For the purpose of precipitation control, mobile rainproof facilities are installed above the test pits. Each test pit is installed inside with a TDR water content measurement tube 2.5 m in depth, used for regular or irregular soil moisture measurement. The outer wall of each test pit is also installed with eight soil sampler bottles of different depths and with 13 automatic soil moisture measurement systems. The moisture variation inside test pits is obtained through calculating the difference between the soil water contents measured twice in succession.

3. Field measurement of standardized crop evapotranspiration

Based on the test pits and the lysimeters, a standardized crop evapotranspiration measurement field of 15,000 m² in area is built. Besides being used for the basic observation of field evapotranspiration, the field is also installed with an eddy covariance system (also used as automatic meteorological station), Bowen ratio system, and soil moisture measurement equipment, which together constitute a complete and high-level field crop evapotranspiration measurement system.

The open-type carbon dioxide (CO₂) eddy covariance system (LI-7500A; Fig. 4) adopts the micrometeorological turbulent eddy motion covariance method and can automatically measure and store near-earth air layer measurement of parameters such as 3-D wind velocity fluctuation, temperature fluctuation, H₂O fluctuation, CO₂ fluctuation, CO₂ flux, water vapor flux, sensible heat flux, air momentum flux, energy exchange flux, frictional wind velocities, and other micrometeorological characteristic variables between the ground surface and the atmosphere during their interactions, with the purpose of determining the fluxes of CO₂ and H₂O. The distance between the eddy covariance system and the test area boundary ranges between 375 and 1,700 m, meeting the requirement that the ratio of the fetch length to the installation height should be greater than 100:1. Through measuring the fluctuations of vertical wind velocity and water vapor density,



Fig. 4 The eddy covariance system.

the eddy covariance method can directly observe the crop evapotranspiration from a meteorological perspective and conduct long-term, continuous, nondestructive, and fixed-point monitoring over ground surface ET. Compared with the lysimeter method and other methods, this method features a relatively short measurement step size, and its equipment can obtain a lot of high-temporal resolution ET and environmental variation information within a short period of time. This system can also be contrasted with the Bowen ratio system in the test field for further measurement and studies.

The crop evapotranspiration monitoring can generate systematic and multiscale measurements of crop evapotranspiration that fully takes into account the influence of climate, soil moisture status, topography, crop, and other factors on the measurement precision. Through modifying various observation and measurement methods and fully applying computer technology and sensor technology, there has been an extremely significant increase of measurement precision, integrating several measurement methods in a convenient, rapid, and labor-saving manner to generate measurements for studies of crop evapotranspiration on different scales.

REFERENCES

1. Zhang, H.C.; He, D.C.; Liu, Z.X. Advances in crop water consumption. *Mod. Agri. Tech.* **2006**, *3*, 52–54.
2. Aiwang, D.; Junfu, X.; Yifu, S. *Research Method on Irrigation Experiment*; China Agriculture Press: Beijing, 2015; 10 pp.

Crop Management: Seasonally Frozen Soil

Brenton S. Sharratt

Land Management and Water Conservation Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Pullman, Washington, U.S.A.

Abstract

The thermal regime of soils during winter and spring can have a profound influence on civil engineering designs, watershed hydrology, and agricultural productivity. For example, alfalfa (*Medicago sativa* L.) and winter wheat (*Triticum aestivum* L.) can die as a consequence of heaving as soils freeze and thaw. Heaving may lift plants above the soil surface where there is a greater risk for exposure of crown and root tissues to lethal temperatures of about -20°C . In addition, the presence of frozen layers within the soil profile can impede water infiltration during snowmelt and rain events as well as retard drainage as the soil thaws in spring. Drainage, as well as soil temperatures, can also affect the timing of field operations such as planting. Agricultural productivity, therefore, depends on moderating soil temperatures during winter and rapidly thawing the soil profile in spring. Soil temperature and thaw can be regulated to some extent through crop residue and soil management.

INTRODUCTION

The thermal regime of seasonally frozen soils is dependent on the interrelated processes of heat and water transfer at the soil surface and within the soil profile. The processes of heat and water transfer at the soil surface are governed by atmospheric conditions and physical properties of the soil at the soil–atmosphere interface. Atmospheric conditions such as air temperature, humidity, solar radiation, wind, and precipitation influence the energy available for processes such as soil heating and evaporation. In cold regions, snow cover will modify the atmospheric conditions at the soil surface. Texture, density, water content, and ice content are important physical properties of soils affecting the transmission of heat through a frozen soil. Ice formation typically occurs in soils at temperatures below 0°C due to capillary and adsorption forces that reduce the free energy status of water in soils. The process of freezing and thawing affects the quantity of heat transmitted through soils since heat (334 J g^{-1}) is liberated as pore ice freezes and must be absorbed to melt pore ice. The process of ice formation and melting requires nearly 100 times the heat (4.2 J g^{-1}) expended in warming or cooling water by 1°C . Thus, heat transfer processes in soils are influenced by phase transitions that occur as soils freeze and thaw. Atmospheric conditions and physical properties of the soil at the soil–atmosphere interface can also be modified by the quantity and orientation of crop residue on the soil surface as well as by the depth and type of tillage.

CROP RESIDUE

Crop residues can mitigate soil erosion, protect plants from winter temperature extremes, and provide a favorable

environment for seed germination in the spring. Residues act as a barrier to heat and water transfer between the atmosphere and soil, thereby retarding heat loss from the soil during winter and hindering the warming of soil in spring. This thermal retardation is shown in Fig. 1 for a clear day in late autumn, winter, and spring. On warm autumn and spring days, soil without residue cover was warmer during the daytime and cooler at night than soil with residue cover. On cold winter days, however, soil with residue cover remained warmer throughout the day than soil without residue cover.

Residue management alters the amount and orientation of crop residue on the soil surface. Various techniques employed in managing crop residues include cutting stubble at various heights, burning residue, removing residue from the seed row, and altering the color of the residue. The winter thermal regime of soil can be dramatically affected by the height of stubble, especially in regions where strong winds redistribute snow. Taller stubble traps more snow and thereby provides additional insulation to the soil that can reduce frost penetration, hasten thawing of the soil profile, and elevate soil temperatures (Table 1). Burning crop residue in autumn or spring can hasten soil warming in the spring by removing residue from the soil surface and temporarily elevating soil temperatures by 100°C or more.^[1] These high temperatures, however, are only sustained for a few minutes during the burn and near the soil surface. In addition, crop residue can be removed along the seed row to bolster soil temperatures during spring. Daily temperatures can rise by 2°C as the width of the band increases from 0 to 20 cm but are unaffected by bands greater than 20 cm.^[2] Residue color can alter soil temperatures, but only on clear days without snow cover. Daily temperatures can be as much as $1\text{--}2^{\circ}\text{C}$ higher for soils covered with black than with natural straw.^[3]

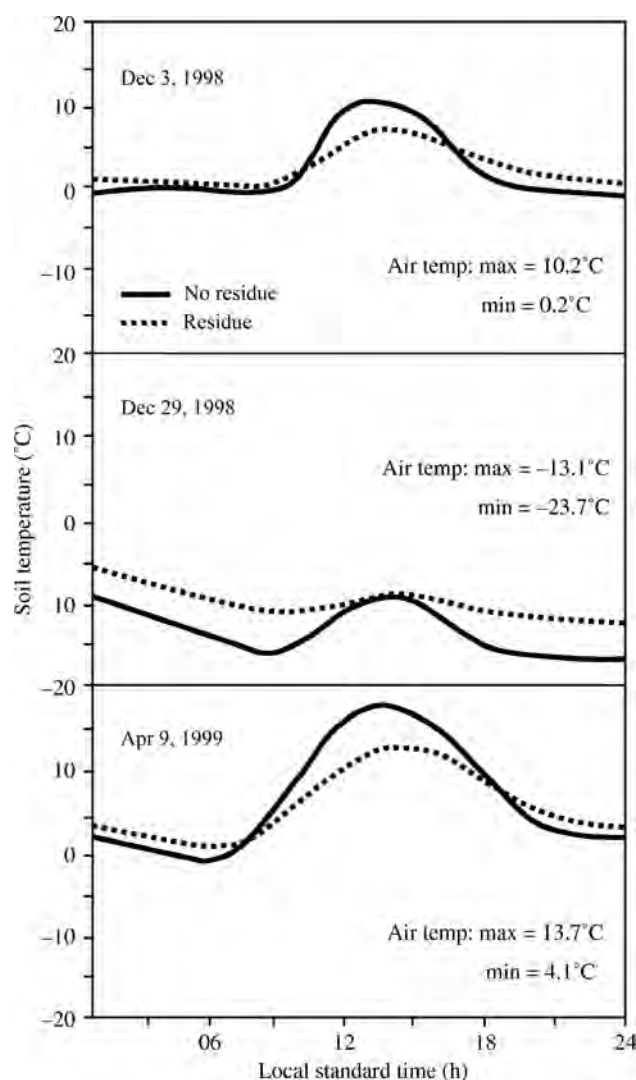


Fig. 1 Temperature at 1-cm depth of a soil with and without soybean (*Glycine max* L.) residue cover and without snow cover on a clear day in late autumn, winter, and spring near Morris, Minnesota.

SOIL MANAGEMENT

Tillage is used to prepare a seedbed, alter the physical properties of soil to curtail erosion, and optimize the thermal regime of the seed zone. In cold regions, methods are

Table 1 Depth of frost penetration, day of year of complete soil thaw, and minimum soil (1-cm depth) temperature as influenced by corn (*Zea mays* L.) stubble height over three winters near Morris, Minnesota.

	Stubble height			
	No stubble/residue	15 cm	30 cm	60 cm
Frost depth (cm)	91	78	46	22
Day of thaw	122	120	98	87
Soil temperature (°C)	-12.5	-10.0	-6.5	-6.0

Table 2 Average soil temperature at 1-cm depth on various aspects of a ridged soil surface for a clear, spring day at Fairbanks, Alaska^a and Morris, Minnesota.

Location	Date	Aspect (°C)				
		North	South	West	East	Level
Fairbanks	May 6, 1990	4.9	9.0	7.5	7.3	8.0
Morris	March 22, 1999	-2.0	2.5	0.6	0.4	0.7

Source: From Sharratt.^[4]

sought that roughen, darken, and reduce the amount of residue on the soil surface to hasten soil warming in the spring. No tillage is advocated to conserve the soil resource, but this method often retards warming of the soil. Other methods such as strip tillage, ridge tillage, chisel plow, and moldboard plow are alternatives to managing the soil thermal regime. Strip tillage is accomplished in autumn or spring by cultivating in bands, thus resulting in a residue-free band. Little is known concerning spring temperatures achieved using strip tillage, but temperatures may be similar to those of residue-free bands. Ridge tillage can dramatically affect soil temperature in cold regions.^[4] Daily temperatures on a southerly slope can be elevated as much as 2°C over those on a level surface and 5°C over those on a northerly slope (Table 2). Spring soil temperatures are generally higher for moldboard and chisel plow owing to the rougher, darker, and smaller amount of residue on the soil surface compared with other tillage methods. Daytime temperatures can be 15°C higher, while nighttime temperatures can be 5°C lower, for moldboard plow than no tillage.^[5]

Crop and soil management practices that retard heat loss from soils in winter also slow the warming of soil in spring. Discovery of new residue management or tillage techniques that enhance soil warming during winter and spring is essential to the viability of agriculture in cold regions. These techniques will be identified only by improving our understanding of those physical properties of residue and soil that influence heat and water transfer between the soil and atmosphere.

REFERENCES

1. Rasmussen, P.E.; Rickman, R.W.; Douglas, C.L. Jr. Air and soil temperatures during spring burning of standing wheat stubble. *Agron. J.* **1986**, *78*, 261–263.
2. Shinnars, K.J.; Nelson, W.S.; Wang, R. Effects of residue-free band width on soil temperature and water content. *Trans. ASAE* **1994**, *37*, 39–49.
3. Sharratt, B.S.; Flerchinger, G.N. Straw color for altering soil temperature and heat flux in the subarctic. *Agron. J.* **1995**, *87*, 814–819.
4. Sharratt, B.S. Soil temperature, water content, and barley development of level vs. ridged subarctic seedbeds. *Soil Sci. Soc. Am. J.* **1996**, *60*, 258–263.
5. Gupta, S.C.; Larson, W.E.; Linden, D.R. Tillage and surface residue effects on soil upper boundary temperatures. *Soil Sci. Soc. Am. J.* **1983**, *47*, 1212–1218.

Crop Plants: Phosphorus Availability

Bettina Eichler-Löbermann

Institute of Land Use, University of Rostock, Rostock, Germany

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

The economical use of phosphorus (P) is a key issue for sustainable agriculture because its natural resources are among the first to be depleted in a predictable time span. P supply to crop plants is generally assured by fertilizer application; however, less attention has been paid to the active energy supply and adaptation mechanisms, enabling the plants to mobilize P in soils. This entry focuses on the P efficiency of crop plants and possibilities to improve the utilization of P in agriculture. In this context, special attention is given to catch crops as a means of improving the P supply of the following main crops.

INTRODUCTION

Plants respond to their soil environment and are able to influence the nutrient availability in the rhizosphere.^[1] Among others, there are mechanisms that enable plants to acquire P from the soil. Some of these mechanisms are specific for P, such as the excretion of phosphatases by the roots.^[2] Others affect a wide range of nutrients, such as the development of root hairs and the enhancement of the root to shoot ratio.^[3]

IMPROVEMENT OF P EFFICIENCY

Although soils may contain 400–1000 kg/ha P in total, the amount of plant available P in soil is usually low. Up to 90% of the total soil P is found in the non-labile pool, as a result of immobilization into organic and inorganic components.^[4]

Improvement of P efficiency is accomplished by either a raise of the P uptake or a better utilization of the existing P in the plant. Crop plants have different strategies to influence the P uptake. Some plants adapt to low-P soil by changing their root morphology, while others excrete P-solubilizing compounds.^[5] There are major differences between plant species (Table 1), but even genotypes within the same species may differ in their P efficiency.^[6,7]

Under P-deficient conditions, plants usually increase their root system in relation to the shoot. Because of this increased root to shoot ratio, the nutrient need and the nutrient influx of the single root decrease. Especially monocotyle plants such as maize (*Zea mays*) and grass are known to have a large root length and root surface. Thus, they can utilize a greater soil volume, which is of vital importance for P utilization, as it has a low mobility in soil.^[3,8]

Plant root exudates mainly consist of a complex mixture of amino acids, organic acids, organic and inorganic ions, sugars, vitamins, nucleosides, enzymes, and root border cells, which have major direct or indirect effects on the acquisition of mineral nutrients required for plant growth.^[9–11] Besides the acidifying effect of the excreted organic acids such as citric and malic acids, there are also chelating effects. Acidification of the soil enhances the solubility of P in basic soils, while the chelating compounds bind cations and thus release P from primarily not available Ca, Fe, and Al phosphates.^[12]

The capacity to excrete organic acids is pronounced for some Proteaceae, which do not form mycorrhizal associations but have proteoid roots. Under P deficiency, lupine (*Lupinus albus*) excretes much more malate and citrate from its proteoid roots than under sufficient P supply.^[13] Oilseed rape (*Brassica napus*) has a higher P uptake from rock phosphate than sunflower (*Helianthus annuus*) because of higher excretion of malate and citrate.^[14] For corn, regarding the root excreted sugars, Schilling et al.^[5] found that under a lack of P the portions of pentoses excreted increase while the amounts of hexoses and sucrose decrease.

Root exudates also contain chemical molecules that control the development of plant–fungal symbiosis, as they provide signals that alert the mycorrhizal fungi to the presence of a host plant.^[10]

Mainly soluble inorganic soil P can be taken up by plants. Therefore, the organic P compounds have to be mineralized into inorganic P forms to become available. This process is promoted by phosphatases. Acid and alkaline phosphatases are known, which occur in dependence of the pH value of soils.^[15] Acid phosphatase in soil originates from many sources, including plant root exudates, fungi,

Table 1 Features of different crop plants (cultivated as catch crops in three different trials) and their influence on P contents in soil and P uptake of the following main crop

Crop	Root length ^a (m/pot)	Root/shoot ratio ^a (m/g DM)	Activity of acid phosphatase ^a (p-nitrophenyl, µg/g/h)	Cation/anion ratio of nutrient uptake ^b	P content in soil solution ^c (PO ₄ -P, mg/l)	P content in soil ^c (PDL, mg/100 g soil)	P uptake main crops ^d
Oilseed rape	107.5 A	4.9 A	227.8 C	5.3 AB			
Maize	578.5 C	13.8 C	276.7 D	8.4 C			
Phacelia	88.1 A	9.3 B	122.6 B	13.7 D	0.35 D	4.91 B	24.7 B
Buckwheat	274.3 B	10.2 B	262.1 D	9.4 C	0.30 C	4.41 A	23.3 A
Comm. Ryegrass	205.7 B	14.4 B	73.0 A	7.7 BC	0.22 A	4.40 A	23.0 A
Lupine	49.5 A	9.3 B	217.1 C	9.5 C			
Pea	65.9 A	5.1 A	231.8 C	6.9 B			
Serradella	68.7 A	9.7 B	302.6 E	8.1 BC	0.32 C	4.80 B	23.9 AB
Amaranth	42.1 A	7.6 AB	301.5 E				
Oil narved				6.3 B			
Oil radish				4.0 A	0.25 B	4.79 B	23.5 A
Yellow mustard				4.2 A			

Note: ANOVA, analysis of variance. Different capital letters (A–E) represent the significant differences between the variants (ANOVA, $\alpha = 0.05$).

^aPot trial A [6 kg of loamy sand/pot (low in plant available P)] after 7 weeks of vegetation time under greenhouse conditions.

^bPot trial B [6 kg of sandy loam per pot (extreme low in plant available P)] in a two-year pot trial under greenhouse conditions. Average of the 2 years.

^cField experiment (loamy sand), average of six sampling dates, from spring 1999 to autumn 2001.

^dField experiment (loamy sand), average of P uptake of main crops (summer oilseed rape, summer barley, and summer wheat) from 1999 to 2001.

mycorrhizal fungi, and bacteria. Alkaline phosphatase is produced by soil microorganisms and soil fauna, whereas higher plants are devoid of alkaline phosphatase.^[16] In a pot experiment, the highest activity of acid phosphatase was found in soil under serradella (*Ornithopus sativus*), amaranth (*Amaranthus hypochondriacus*), and maize; by far the lowest activity of acid phosphatase was found in soil under ryegrass (*Lolium westerwoldicum*) and phacelia (*Phacelia tanacetifolia*) (Table 1).^[17]

The soil pH is important for the P solubility, which is higher in neutral soils than in acid or alkaline soils.^[11] Plants affect the soil pH not only by the excretion of organic acids but also by the amount of inorganic ions absorbed by roots.^[18] Plant uptake of anions in excess of cations often causes the secretion of HCO₃[−], which leads to increased pH values. On the other hand, the uptake of cations in excess to anions can cause roots to exude H₃O⁺ and lower the pH value. This could be shown for phacelia (Table 1).^[19]

Catch cropping has a positive influence on soil fertility and can reduce nutrient losses from soil.^[20,21] Furthermore, plants used as catch crops or inter crops can improve the P supply of the following main crops directly by increasing the plant-available P amount in soil or indirectly when used as green fertilizers.^[22] The organic bound P in catch crops becomes available to the main crops after decomposition. In different experiments, catch crops, such as phacelia and serradella, were found to mobilize P in soil. They substantially increased the P content in soil and soil solution in P poor soils, and the P uptake of the main crops was enhanced (Table 1). On P-poor sandy soils, buckwheat (*Fagopyrum*

esculentum) was found to have a high P uptake.^[19] In a pot experiment, a green fertilization with phacelia increased the P uptake of the following maize more than a fertilization with triple-super P or manure.^[19] However, catch cropping does not replace the P fertilization, but it should be seen as an opportunity to improve the utilization of P.

The active mobilization of P sources by plants is an important factor to counteract the loss of P solubility by fixation and adsorption and thus a source of energy for high P utilization. The mobilization of P in soil can contribute to save commercial fertilizers. Green fertilization with catch crops that have a high P uptake efficiency may help to improve P supply of the main crops.

REFERENCES

- Schilling, G. *Pflanzenernährung und Düngung*; Eugen Ulmer: Stuttgart, 2000.
- Gaume, A.; Mächler, F.; De León, C.; Narro, L.; Frossard, E. Low-P tolerance by maize (*Zea mays* L.) genotypes: significance of root growth, and organic acids and acid phosphatase root exudation. *Plant Soil* **2001**, *228*, 253–264.
- Otani, T.; Ae, N. Sensitivity of phosphorus uptake to changes in root length and soil volume. *Agron. J.* **1996**, *88*, 371–375.
- Stevenson, F.J.; Cole, M.A. *Cycles of Soil—Carbon, Nitrogen, Phosphorus, Sulfur, Micronutrients*; John Wiley & Sons: Hoboken, NJ, 1986.
- Schilling, G.; Gransee, A.; Deubel, A.; Lezovic, G.; Ruppel, S. Phosphorus availability, root exudates, and

- microbial activity in the rhizosphere. *Z. Pflanzenernähr. Bodenk.* **1998**, *161*, 465–478.
6. Schnug, E.; Strampe, U. Sortentypische Unterschiede der Nährelementkonzentration bei Winterweizen. *J. Agron. Crop Sci.* **1988**, *160*, 163–172.
 7. Brennan, R.F.; Bolland, M.D.A. *Lupinus luteus* cv. *Wodjil* takes up more phosphorus and cadmium than *Lupinus angustifolius* cv. *Kalya*. *Plant Soil* **2003**, *248*, 167–185.
 8. Rosolem, C.A.; Assis, J.S.; Santiago, A.D. Root growth and mineral nutrition of corn hybrids as effected by phosphorus and lime. *Commun. Soil Sci. Plan.* **2004**, *25*, 2491–2499.
 9. Ae, N.; Arihara, J.; Okada, K.; Yoshihara, T.; Johansen, C. Phosphorus uptake by pigeonpea and its role in cropping systems of the Indian subcontinent. *Science* **1990**, *248*, 477–480.
 10. Dakora, F.D.; Phillips, D.A. Root exudates as mediators of mineral acquisition in low-nutrient environments. *Plant Soil* **2002**, *245*, 35–47.
 11. Staunton, S.; Leprince, F. Effect of pH and some organic anions on the solubility of soil phosphate: implications for P bioavailability. *Eur. J. Soil Sci.* **1996**, *47*, 231–239.
 12. Lambers, H.; Stuart Chapin, F., III; Pons, T.L. *Plant Physiological Ecology*; Springer: New York, 1998.
 13. Gerke, J.; Römer, W.; Jungk, A. The excretion of citric and malic acid by proteid roots of *Lupinus albus* L.: effects on soil solution concentration of phosphate, iron and aluminium in the proteid rhizosphere in samples of an oxisol and a luvisol. *Z. Pflanzenernähr. Bodenk.* **1994**, *157*, 289–294.
 14. Hoffland, E. Mobilization of Rock Phosphate by Rape. Dissertation, University of Wageningen, 1991.
 15. Eivazi, F.; Tabatabai, M.A. Phosphatases in soils. *Soil Biol. Biochem.* **1977**, *9*, 167–172.
 16. Tarafdar, J.C.; Claassen, N. Organic phosphorous compounds as a phosphorus source through the activity of phosphatase produced by plant roots and microorganisms. *Biol. Fertil. Soil* **1988**, *5*, 308–312.
 17. Eichler, B.; Caus, M.; Schnug, E.; Köppen, D. Soil acid and alkaline phosphatase in relation to crop species and fungal treatment. *Landbauforschung Völkenrode* **2004**, *54* (1), 1–6.
 18. Haynes, R.J. Active ion uptake and maintenance of cation-anion balance: A critical examination of their role in regulating rhizosphere pH. *Plant Soil* **1990**, *126*, 247–264.
 19. Eichler, B. Möglichkeiten zur Einflussnahme auf Phosphorkreisläufe für die Gestaltung nachhaltiger Bodennutzungssysteme. Habilitationsschrift, University of Rostock, 2004.
 20. Martin, P.; Papy, F.; Souchère, V.; Capillon, A. Surface runoff control and modelling cropping practices. *Cahiers d'études et de Recherches Francophones/Agricultures* **1998**, *7* (2), Article 111.
 21. Merbach, W.; Wurbs, A.; Jacob, H.J.; Latus, C. Temporary biological conservation by winter oilseed turnip (*Brassica rapa* L.) and its influence on following crops and N-percolation. *Isotops Environ. Health Stud.* **1997**, *33*, 39–43.
 22. Cavigelli, M.A.; Thien, S.J. Phosphorus bioavailability following incorporation of green manure crops. *Soil Sci. Soc. Am. J.* **2003**, *67*, 1186–1194.

Crop Residue Management

David L. Schertz

*U.S. Department of Agriculture—Natural Resources Conservation Service
(USDA-NRCS), Washington, District of Columbia, U.S.A.*

Dan Towery

*Natural Resource Specialist, Conservation Technology Information Center,
West Lafayette, Indiana, U.S.A.*

Abstract

Crop residue management is a general term used to describe tillage management practices that leave a significant portion of the soil surface covered with the previous crop's residue to reduce wind and water erosion. Residue management practices include no-till/strip-till, mulch-till, and ridge-till. Although all residue management practices can favorably impact soil, water, and air quality, they will vary in the degree of this benefit, depending on the amount of soil disturbance, surface residue cover, soil, and climate. This entry focuses on no-till/strip-till because it provides the greatest beneficial environmental impacts compared to the other residue management practices.

INTRODUCTION

Residue management practices are defined as follows:

1. No-till/strip-till: Managing the amount, orientation, and distribution of crop and other plant residues on the surface year-round while growing crops in narrow slots or tilled or residue-free strips in soil previously untilled by full-width tillage implements.
2. Mulch-till: Managing the amount, orientation, and timing of crop or other crop residues on the soil surface year-round while growing crops where the entire field surface is tilled before planting.
3. Ridge-till: Managing the amount, orientation, and distribution of crop and other plant residues on the soil surface year-round while growing crops on preformed ridges alternated with furrows protected by crop residue.

CROP RESIDUE MANAGEMENT SURVEY DATA

Conservation tillage was used only on 4 million ha or 3% of all cropland acres in the United States in 1970.^[1] The decade of 1990s saw a pronounced increase in the adoption of conservation tillage. It increased from 29.6 million ha or 26% of all annually planted cropland acres in 1990 to 43.8 million ha or 36.3% of all annually planted cropland acres in 2000 (Fig. 1). This increase of 14.2 million ha in conservation tillage acres has come almost entirely from the no-till category. Soybeans and corn accounted for 75% of all no-till acres in 2000. During the 1990s, no-till experienced a 210% increase, growing from

6.8 million ha to 21.1 million ha. Ridge-till and mulch-till have remained basically unchanged during this same time period. No-till growth was more pronounced from 1990 to 1994 when it averaged a 2.2 million ha increase every year.^[2]

SOIL, CLIMATE, AND CROP RELATIONSHIPS

The successful adoption of conservation tillage and no-till, in particular, is dependent on the soil type, crop rotation, and climate of a given region. No-till is widely adopted in an area only if the local management strategies minimize risk and maintain or increase yields. Some of the key soil properties, which may influence no-till adoption, include soil drainage, soil texture, slope, and organic matter levels. Key climatic factors include total rainfall, rainfall distribution, temperature, and length of growing season. Crops vary in their adaptability to conditions typically found in a no-till environment. Commonly, a no-till seedbed will be cooler and will have a higher moisture level than the soil previously tilled, which may be an advantage or a disadvantage to achieving the desired stand. In addition, the crop rotation is more important in a no-till environment. Rotating crops minimizes potential allelopathic effects and reduces insect, weed, and disease pressure.

ENVIRONMENTAL BENEFITS

Key environmental benefits of residue management practices, although site specific, generally include erosion

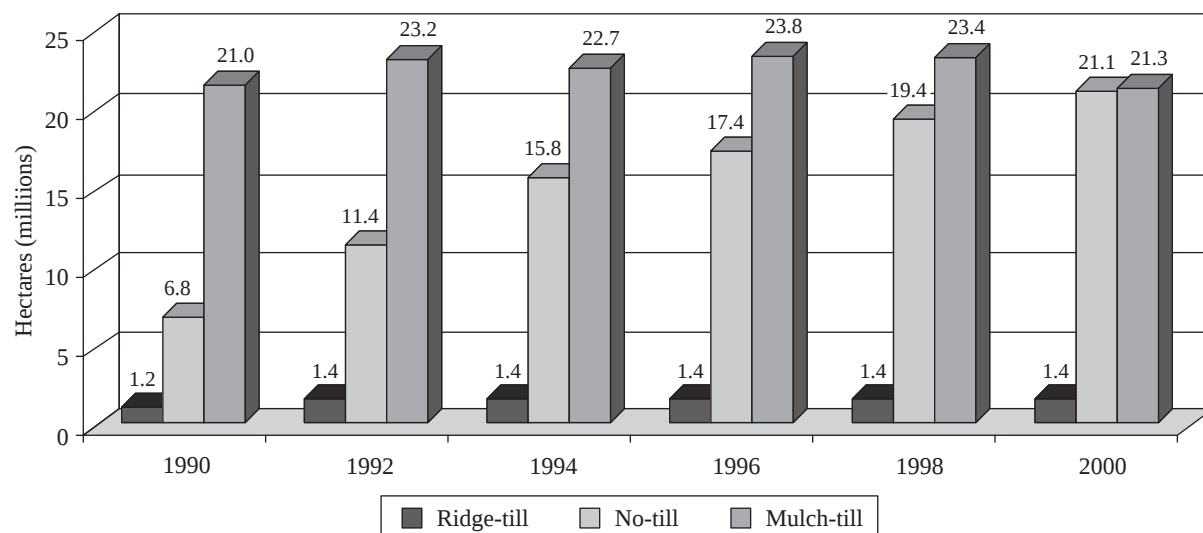


Fig. 1 Conservation tillage adoption in the United States, 1990–2000.

Source: From CTCIC.^[2]

reduction, improved soil, water, and air quality, and reduction in sediment. In addition, other agronomic benefits include improved moisture conservation, reduced soil compaction, improved soil structure, and reduced surface crusting. This entry will briefly discuss the key environmental benefits of crop residue management practices, including soil erosion reduction and soil, water, and air quality improvements.

Soil Erosion Reduction

Crop residue left on the soil surface significantly reduces each of the water erosion processes, including soil detachment, transport, and deposition. When rain falls on a bare soil surface, soil particles are dislodged from soil aggregates by the explosive action of falling raindrops. Surface residue cover intercepts the falling raindrop and dissipates its erosive energy. Surface residue can also form small dams that slow surface runoff. When the surface runoff is slowed, it has a greater opportunity to infiltrate the soil surface. The benefit of surface residue in reducing inter-rill and rill erosion is shown in Fig. 2.^[3] No-till leaves the most surface residue cover of the residue management practices and is the most beneficial of these practices in reducing soil erosion. With no-till, the amount of surface residue cover can approach 80–90% after high residue-producing crops and 30–50% after low residue-producing crops.

Crop residue left on the soil surface also significantly reduces each of the wind erosion processes, including surface creep, saltation, and suspension. The threshold wind velocity required to begin the wind erosion process is higher across surfaces protected by crop residue. Also, crop residue is more effective in reducing wind erosion if it is left standing compared to lying flat. However, in most no-till situations, the orientation of residue is not critical

because large quantities of surface residue are present. With low-residue producing crops, planting crops perpendicular to the prevailing wind and leaving crop residue standing become more important.

Soil, Water, and Air Quality

Sediment is the number one pollutant in the United States.^[4] It not only creates physical problems but also is detrimental to plants, animals, and aquatic species. No-till significantly reduces soil erosion, increases infiltration, improves aggregate stability, and increases organic matter. As a result, the amount of sediment reaching surface water is reduced.

Soil organic matter is probably the most important soil quality indicator. Reducing soil erosion has a direct positive impact on soil quality because the amount of soil organic matter lost through erosion is greatly reduced. No-till can have a significant impact on increasing soil organic matter, while full-width tillage minimizes the opportunity to increase soil organic matter. The largest increases in soil organic matter with intensive cropping result from continuous no-till. If no-till is used 1 year followed by full-width tillage, the increase in organic matter will be limited.^[5] Soil organic matter increases primarily through avoidance of tillage, leaving the root structure undisturbed.^[6] Even with continuous no-till, however, the increase in soil organic matter is a slow process.

Crop residue provides an energy source for microorganisms. They use crop residue for their life processes and return humus to the soil, resulting in an increase in soil carbon. The population of microorganisms in the soil is directly related to the amount of energy or food available. When residue is incorporated in the soil, microorganisms consume it rapidly, leaving little or no energy source available for top-feeding organisms. As a result, the energy

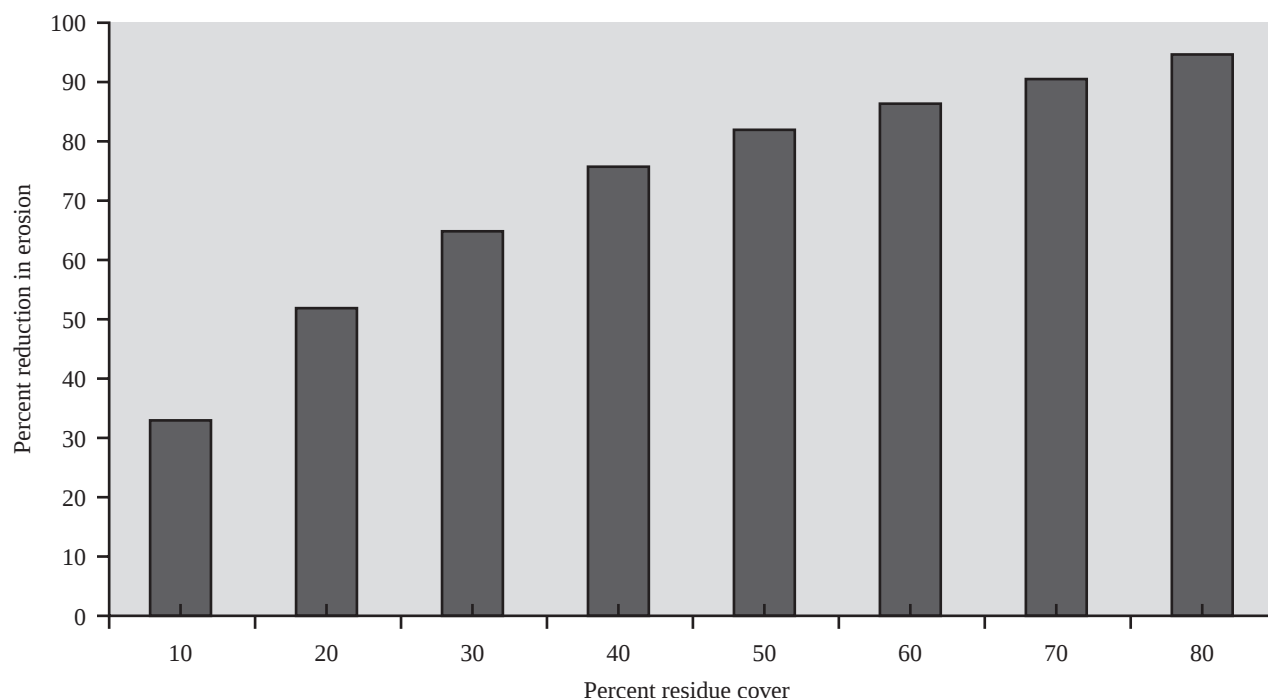


Fig. 2 Approximate effect of residue cover on any day in reducing sheet and rill erosion compared to conventional tillage without surface residue cover.

Source: From Renard, Foster, et al.^[3]

source is quickly depleted, and the number of beneficial microorganisms decreases. When crop residue is left on the soil surface, microorganisms consume it more slowly, remain active for longer periods, and significantly contribute to improving soil humus.^[7]

Nutrients, such as phosphorus, that attach to soil particles are slow to move in the soil profile but can move with surface runoff and sediment. Soluble nutrients that have not infiltrated the soil can move freely in surface runoff. Because residue management practices reduce erosion, improve infiltration, and reduce runoff, they play a key role in reducing the transport of nutrients across the surface and potential contamination of surface water.

Pesticides can present a potential water quality concern similar to nutrients by either moving off target in surface water or attaching to soil particles. However, since residue management practices reduce erosion and surface runoff, these practices help reduce the potential for off-site contamination.

With no-till, herbicides replace tillage for weed control. However, the total amount of herbicides used in comparison to full-width tillage systems is not significantly different.^[8] Herbicide chemistry has changed dramatically providing new crop protection chemicals that are applied in milliliters rather than liters per acre and are less toxic to non-target species. As a result, these new products are considered more environmentally friendly. In addition, biotechnology has made weed control easier, cheaper, and more environmentally friendly for certain crops.^[8]

In addition, biotechnology has helped to facilitate the adoption of no-till with certain crops.

Extensive macropores in no-till, especially if open to the surface, have raised some concern about providing a direct conduit for potential groundwater contaminants. Some macropores, especially earthworm channels, contain large amounts of organic matter along their walls that can absorb some of these chemicals and help retain them in the upper portions of the profile. Earthworm channels also have increased microorganism activity that can significantly increase the breakdown and degradation of chemicals and reduce their potential for groundwater contamination.

Particulate matter of 10 μm (PM-10) in size has been identified as a potential health hazard.^[9] These fine particle sizes can be generated from wind erosion, tillage, and harvesting operations. No-till/strip-till, ridge-till, and mulch-till practices used with high residue-producing crops should provide sufficient residue cover to significantly reduce air quality hazards from PM-10 by reducing wind erosion and tillage trips.

CONCLUSION

Crop residue management practices provide numerous environmental benefits, are economical for growers to implement, and are the most widely used conservation practices in the world. Although increases in no-till adoption in the United States have been the primary reason for

the increase in crop residue management practices in the 1990s, the adoption rate has slowed in the later part of this decade. More emphasis must be placed on the use of continuous no-till if growers and society are to achieve the benefits described in this entry.

REFERENCES

1. Schertz, D.L. Conservation tillage: An analysis of acreage projections in the United States. *J. Soil Water Conserv.* **1988**, 43, 256–258.
2. CTIC. *2000 National Crop Residue Management Survey*; Conservation Technology Information Center: West Lafayette, 2000; 1–69.
3. Renard, K.G.; Foster, G.R.; Weesies, G.A.; McCool, D.K.; Yoder, D.C. *Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE)*. Agricultural Handbook 703; U.S. Department of Agriculture, Agricultural Research Service: Washington, 1997; 1–384.
4. Anderson, M.; Magleby, R. *1996–97 Edition of Agricultural Resources and Environmental Indicators*; Agricultural Handbook 712; U.S. Department of Agriculture, Economic Research Service: Washington, 1997; 83–96.
5. Langdale, G.W.; West, L.T.; Bruce, R.R.; Miller, W.P.; Thomas, A.W. Restoration of eroded soil with conservation tillage. *Soil Technol.* **1992**, 5, 81–90.
6. Balesdent, J.; Balabane, M. Major contribution of roots to soil carbon storage inferred from maize cultivated soils. *Soil Biol. Biochem.* **1996**, 28 (9), 1261–1263.
7. Natural Resources Conservation Service. *Core 4 Conservation Practices Training Guide, CD-ROM Version 1.0*; U. S. Department of Agriculture, Natural Resources Conservation Service: Washington, 1999; 1–395.
8. Gianessi, L.P.; Carpenter, J.E. *Agricultural Biotechnology: Benefits of Transgenic Soybeans*; National Center for Food and Agricultural Policy: Washington, 2000; 1–103.
9. Karstadt, M. *The Plain English Guide to the Clean Air Act*; EPA 400-K-93-001; United States Environmental Protection Agency, Office of Air and Radiation: Washington, 1993; 7–22.

Crop Root Response to Temperature: Temperate Zone

Thomas C. Kaspar

National Soil Tilth Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.

Abstract

Root system expansion is a function of three temperature-dependent processes: growth, development, and orientation. Temperature affects root growth through its influence on root weight, root length, and root diameter. Root development is influenced by the effect of temperature on root initiation and root turnover. Finally, temperature controls root orientation through its impact on the direction of root growth.

INTRODUCTION

In temperate regions, the average temperature of the soil profile rises from winter minimums to summer maximums as the growing season progresses. At the time of planting for most summer annual crops, a temperature gradient has developed in the soil profile with temperature decreasing with depth. As the soil surface absorbs thermal energy throughout the growing season, deeper soil layers become increasingly warmer.^[1] Because temperature influences a wide variety of plant functions, it is not surprising that soil temperature affects root growth in many ways,^[1–5] and that the expansion of crop root systems in temperate regions is limited by cool soil temperatures.^[1,6]

ROOT GROWTH

Root dry weight is the most commonly measured growth parameter. For most plant species, the temperature response of root dry weight follows a typical temperature response curve (Fig. 1).^[1] The optimum, minimum, and maximum growth temperatures of the response curves differ among plant species and among genotypes within species. Generally, crop species that are winter annuals or cool-season species, such as oat (*Avena sativa* L.), rye (*Secale cereale* L.), wheat (*Triticum aestivum* L.), and rape (*Brassica napus* L.), have lower optimum, minimum, and maximum growth temperatures than crops such as cotton (*Gossypium hirsutum* L.) or maize (*Zea mays* L.), which are summer annuals.

Crop root length is an important factor in determining water and nutrient uptake. Root elongation responds to temperature in much the same way as root dry weight (Fig. 2).^[2] Similarly, plant species and genotypes within species differ in their response to soil temperature.

Root length, however, often responds more strongly to soil temperature than does root dry weight. For example, Loffroy et al.^[7] found that cotton taproot length at a root temperature of 17°C was less than 40% of its length at 27°C, whereas root dry weight was 68% of that at 27°C. Presumably, this occurred because the mean taproot diameter was greater at 17°C than at 27°C.

Root diameter along with root length determines root surface area. Additionally, the amount of carbohydrate required for root system expansion is determined by root diameter. Unfortunately, the effect of temperature on root diameter is not clear. Part of the confusion arises because of the differences between mean root diameter of an entire root system and mean root diameter of the roots in a branching order of roots within a root system. Root branching orders are determined by the point of origin of the root. For example, a first-order root originates from the seed, whereas second-order roots originate from first-order roots. Generally, higher order roots have smaller diameters than the lower order roots from which they originate. Thus, an increase in root initiation or branching results in more roots with smaller diameters. If temperature increases root initiation, then mean root diameter of the root system is likely to decrease. Another complicating factor is that root diameters of individual roots tend to decrease as they elongate. Therefore, the generally shorter roots at cooler temperatures would most likely have greater mean diameters than the longer roots grown at warmer temperatures. In summary, the mean root diameter of an entire root system will decrease with increasing temperature, if root elongation and root branching increase with increasing temperature. In contrast, the root diameter of individual roots that are from the same branching order and are of equal length may have similar root diameters at both the higher and the lower temperatures.^[1]

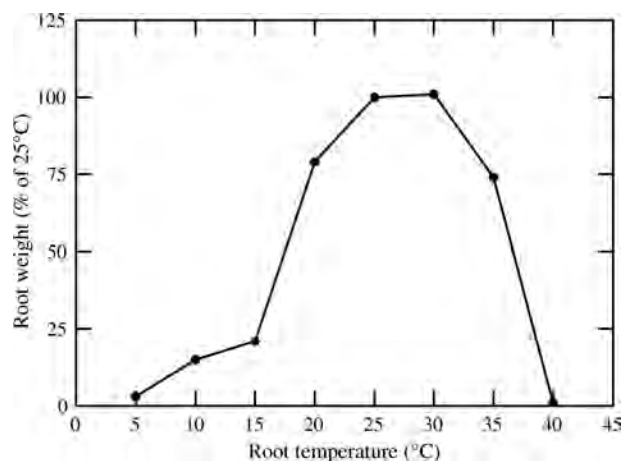


Fig. 1 Root temperature response of maize (*Z. mays* L.) root dry weight at 24 days after germination.

Source: From Brouwer.^[2]

ROOT DEVELOPMENT

Root initiation is a developmental process that responds to soil temperature. The number of lateral roots of cotton and sunflower (*Helianthus annuus* L.) seedlings changed in response to soil temperature in a manner similar to that of root length or root dry weight.^[5] The number of lateral roots increased from zero at low temperatures to maxima at 27°C for sunflower and 35°C for cotton. Above the optimum temperature, the numbers of laterals decreased rapidly. Root initiation can also be described in terms of accumulated degree-days or thermal time for temperatures below the optimum (Fig. 3),^[1] which is normally the case in temperate regions. Accumulated degree-days are calculated by summing up over a period of days, and the difference for each day between the daily mean temperature

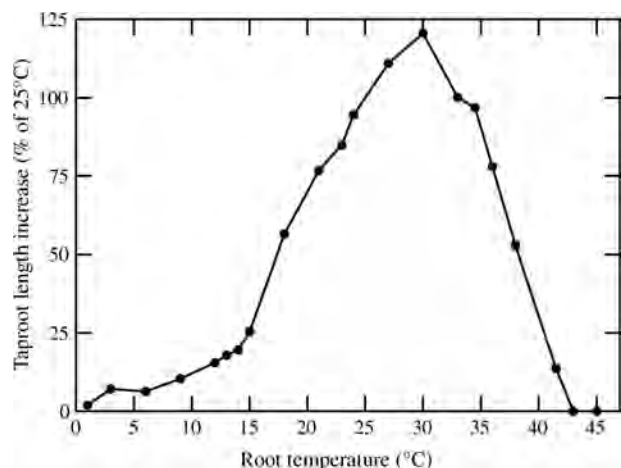


Fig. 2 Increase in taproot length of pecan (*Carya illinoensis*) seedlings after 4 days of growth at the treatment soil temperature.

Source: Data from Woodroof & Woodroof.^[8]

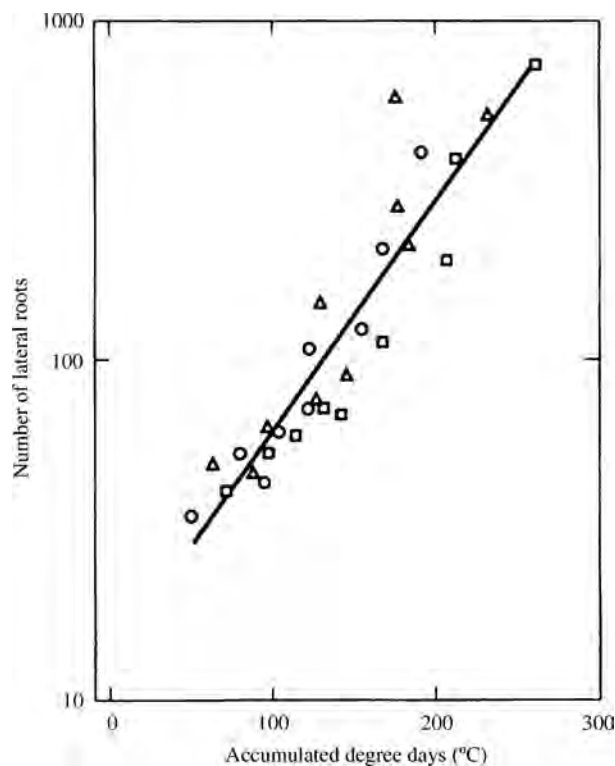


Fig. 3 The relationship between the number of lateral roots of pearl millet (*P. typhoides* S. & H.) and accumulated degree-days (base 12°C) at the shoot meristem during the period 4–22 days after planting.

Source: From Gregory.^[9]

and the minimum temperature required for root initiation. Studies of pearl millet (*Pennisetum typhoides* S. & H.), for which the minimum temperature of initiation is 12°C, have shown that the numbers of lateral roots and adventitious roots were strongly correlated to accumulated degree-days.^[9] In other words, both the length of time and the temperature during that time controlled the number of lateral and adventitious roots. Interestingly, the numbers of seminal roots of oat, wheat, and maize are not determined by soil temperature because the seminal root initials are already present in the seed embryos at the time of planting.^[1]

There have been few studies that have examined the effect of soil temperature on root maturation and turnover. In general, roots at low temperatures remain white and succulent longer than roots grown at high temperatures, which become brown and shriveled more quickly. In one study, root longevity of several grass species was observed to be greater at a site with cooler soil temperatures than at a similar site with warmer soil temperatures.^[10] Root production and standing numbers of roots, however, were greater at the warmer site. Another indication of root maturation is the suberization of endodermal cells. For maize roots of similar length, suberization occurred closer to the apex at higher

soil temperatures, indicating that low temperatures delayed root maturation.^[1]

ROOT ORIENTATION

Roots of many plant species grow out from the plant's main axis at a specific angle from the vertical rather than straight down. The angle at which these roots grow is temperature sensitive.^[1] In general, cooler temperatures tend to cause more horizontal growth, and warmer temperatures result in more downward growth. Additionally, maize and soybean (*Glycine max* [L.] Merr.) genotypes differ in the temperature response and sensitivity of the growth direction of their roots.^[1]

ROOT RESPONSE TO SOIL TEMPERATURE GRADIENTS

In temperate regions, soil temperatures below the depth of 30 cm are usually less than the optimum temperature for root growth of summer annual crops like cotton, maize, and soybean at the start of the growing season. As the growing season continues, the temperatures of the deeper soil layers continue to increase until canopy closure and decreasing air temperatures halt the process. Because warming of soil layers progresses downward from the surface, some deep soil layers may never reach the optimum temperature for root growth, and at greater depths, the soil may only warm to a few degrees above the minimum root growth temperature. As a result, the downward extension of crop roots may be limited by the cool temperatures of the subsoil layers.^[1,6] A controlled environment experiment in which temperature gradients were established in soil columns has shown that the rate of downward growth of cotton and soybean roots is limited by temperature gradients and the rate at which soil warming progresses from the surface.^[6] These results corroborate field observations of soybean rooting depth increases that seemed to follow the 17°C isotherms downward from the surface.^[1]

CONCLUSION

Root system expansion is a function of three temperature-dependent processes: growth, development, and orientation. Temperature affects root growth through its influence on root weight, root length, and root diameter. Root development is influenced by the effect of temperature on root initiation and root turnover. Finally, temperature controls root orientation through its impact on the direction of root growth.

REFERENCES

1. Kaspar, T.C.; Bland, W.L. Soil temperature and root growth. *Soil Sci.* **1992**, *154*, 290–299.
2. Brouwer, R. *Influence of Temperature of the Root Medium on the Growth of Seedlings of Various Crop Plants*; Jaarb. Inst. Biol.: Scheik, 1962; 11–18.
3. Cooper, A.J. *Root Temperature and Plant Growth: A Review*; Research Review No. 4; Commonwealth Bureau of Horticulture and Plantation Crops; Agricultural Bureau: Farnham Royal, 1973; 1–73.
4. Bowen, G.D. Soil temperature, root growth, and plant function. In *Plant Roots: The Hidden Half*; Waisel, Y., Eshel, A., Kafkafi, U., Eds.; 1st Ed.; Marcel Dekker, Inc.: New York, 1991; 309–330.
5. McMichael, B.L.; Burke, J.J. Temperature effects on root growth. In *Plant Roots: The Hidden Half*; Waisel, Y., Eshel, A., Kafkafi, U., Eds.; 2nd Ed.; Marcel Dekker, Inc.: New York, 1996; 383–396.
6. Bland, W.L. Cotton and soybean root system growth in three soil temperature regimes. *Agron. J.* **1993**, *85*, 906–911.
7. Loffroy, O.; Hubac, C.; da Silva, J.B.V. Effect of temperature on drought resistance and growth of cotton plants. *Physiol. Plant* **1983**, *59*, 297–301.
8. Woodroof, J.G.; Woodroof, N.C. Pecan root growth and development. *J. Agr. Res.* **1934**, *49*, 511–530.
9. Gregory, P.J. Response to temperature in a stand of Pearl Millet (*Pennisetum typhoides* S. & H.). III. Root development. *J. Exp. Bot.* **1983**, *34*, 744–756.
10. Eissenstat, D.M.; Yanai, R.D. The ecology of root lifespan. In *Advances in Ecological Research*; Begon, M., Fitter, A.H., Eds.; Academic Press, Ltd.: New York, 1997; Vol. 27, 1–60.

Crop Rotation and Farming Systems: Temperate Zone

Martin R. Carter

Agriculture and Agri-Food Canada, Charlottetown, Prince Edward Island, Canada

Abstract

Crop rotation and farming systems play major roles in soil structure management, especially in temperate zones where moderate climates provide a potential for accumulation of soil organic matter and related improvements in soil structure. *Soil structure* can be defined as the spatial arrangement of both soil particles and voids, and improvements in its quality are positively related to crop and soil management practices, such as organic matter inputs (e.g., continuous cropping, organic amendments), soil nutrient management (e.g., fertilizers, calcium amendments), and conservation tillage practices. Increasing organic matter inputs through crop rotation, which enhances the supply of structure-forming agents (e.g., humic substances, plant roots, fungal hyphae, and soil fauna), is the major contributor to soil structure management in temperate soils.

SOIL STRUCTURE MANAGEMENT IN TEMPERATE AGRICULTURE

Soil structure can be described in the broadest sense as the spatial arrangement of both soil particles (aggregates) and voids (pores), which has significant effects on air–water relationships in soil.^[1–3] The shape, size, and stability of aggregates in cultivated soils can be greatly modified by crop rotation and farming practices.

Soil Structure in Temperate Zones

A useful approach to understanding the management of soil structure in temperate agriculture, which accommodates both aggregates and pores, characterizes structure on the following basis:^[4]

- Structural form—arrangement and size of the pore space
- Structural stability—ability to retain the distribution and size of aggregates after exposure to various stresses (e.g., external forces of impact, shear, abrasion, and slaking)
- Structural resiliency—ability of a soil to recover its pore-space arrangement after the removal of a specific stress (e.g., compaction)

On a macroscopic scale, soil structural form can be characterized by soil bulk density. Pore-size distribution, however, provides a more detailed measure of structural form. Pore space can be categorized on the basis of the dominant water process (i.e., water transmission or storage pores).

Although both the pore and aggregate viewpoints are useful, the latter usually dominates through ease of

measurement. It assesses soil structural stability on the basis of the stability of a certain aggregate size. The level of dispersed clay can also provide a useful index of soil structural stability, especially for processes involving reduced permeability, sodicity, and soil crusting.^[2]

Soil structural resiliency is characterized by assessing a soil's ability to self-generate structure through natural processes. Biological processes, such as the activity of soil fauna; physical processes, such as freezing–thawing and wetting–drying, are the main agents involved in structure recreation at the soil surface.^[4] These “self-mulching” soils are mainly clay to clay loams that undergo sufficient swelling to allow structure formation.

Soil Structure-Forming Agents in Crop Rotation and Farming Systems

The formation of soil aggregates is dependent upon both abiotic and biotic factors, the former being related mainly to soil clay content and the capacity for natural structure-forming processes (e.g., alternating shrink–swell, freeze–thaw, and wet–dry). In the temperate zone, organic matter (i.e., biotic factor) plays the major role in soil structure formation and management. The structure-forming process occurs with some degree of order.^[2,4] This ordering or arrangement of particles and aggregates can be viewed conceptually as a hierarchy consisting of three main orders:^[2] clay microstructures (<2 μm diameter), microaggregates (2–250 μm diameter), and macroaggregates (>250 μm diameter).

In clay microstructures, clay–organic matter complexes are stabilized by humic acids and inorganic ions [e.g., calcium (Ca)]. Microaggregates are stabilized directly by microbial materials, such as polysaccharides,

hyphal fragments, and bacterial cells or colonies.^[2,5] In comparison, the formation of macroaggregates and their temporary stabilization is the result of a combination of mechanisms related to plant roots and the activity of soil fungi and fauna^[5] (Table 1). Soil fauna, strongly associated with structure formation, is a major determinant of soil processes influencing nutrient cycling, aggregate formation, and permeability of soil.^[6] Macrofauna (e.g., earthworms) influence the formation of large soil pores and play an important role in preferential flow.

Importance of Soil Structural Quality in Temperate Farming Systems

Management of soil structure is needed to ensure the maintenance of soil structural quality, which is important for many soil functions, including serving as a medium for root growth and development and regulation of water and air.^[1] Inherent soil properties, such as clay content, influence the magnitude of the soil water-holding capacity. This is also augmented by the organic matter content. Dynamic soil structural properties, such as soil density and porosity, control the soil storage capacity for water and air. Pore size distribution, considered a good indicator of the soil structural condition, has proven useful in predicting water infiltration rates, water availability to plants, soil water storage capacity, and soil aeration status. Macroporosity, air permeability, and/or oxygen diffusion rate provide a measure of soil macropore continuity and organization. Soil strength and aggregate stability reflect a soil's ability to resist compaction and other stresses that can lead to loss of structure.

Table 1 Benefit of crop rotation on soil structure in temperate soils: agents, processes, and scale of structure formation.

Structure-forming agent	Structure-forming process	Scale of structure
Humic substances; hydroxides of iron and aluminum; polyvalent metal cations (e.g., Ca); clays	Allow bonding between soil mineral and organic components	Microaggregation
Polysaccharides	Gelatinous glue; organomineral bonding	Micro- and macroaggregation
Plant roots; fungal hyphae	Enmesh soil aggregates; exude polysaccharides	Macroaggregation
Soil fauna (e.g., earthworms)	Mix organic matter with soil colloids; form large pore or gallery networks	Macroaggregation

Source: From Carter & Stewart.^[2]

BENEFITS OF CROP ROTATION ON SOIL STRUCTURE

In the temperate zone, soil structure and organic matter accumulation are interrelated: organic matter (or fractions thereof) is basic to the structure-forming process, while organic matter sequestered within microaggregates is protected against decomposition. However, crop rotation, land use and farming system, and soil and crop management practices influence the degree of soil structure.

Perennial Forages and Pastures

Water-stable macroaggregation increases rapidly when perennial forages and grasses are utilized in crop rotations. Maximum stability is often achieved after 3 to 5 years.^[7,8] Grass roots and fungal hyphae are the main aggregating agents involved in forming and stabilizing macroaggregates.^[9] Early improvement in water-stable aggregates is related to soluble carbohydrates and microbial biomass.^[2] Choice of perennial forage crop species and cultivar can also influence the extent of aggregate formation and stabilization.

Short-Term Rotations and Cover Crops

Crop rotation, an ordered sequence of crops on the same land, and the use of cover crops mainly in row crop systems [i.e., maize (*Zea mays* Linnaeus) and potato (*Solanum tuberosum* L.)] provide important benefits to soil structure in temperate agriculture. The degree of structure modification, especially improvement of water-stable aggregation, is dependent on crop species and, in particular, the amount of residue that each crop of the rotation returns to the soil. Carbon inputs vary widely among crop species. Generally, an annual input of 2–3 Mg C ha⁻¹ is required in temperate soils to prevent a decline in soil organic matter concentration and soil structure.^[2]

Organic Additions, Fertilization, and Ca

Many organic materials are applied to agricultural soils, such as manure, compost, and sewage sludge. The most widespread is farmyard manure. Organic additions and inputs can lead to the formation of water-stable macroaggregates.^[2,10] However, organic materials vary widely in composition and, consequently, differ in their rate of decomposition and resultant benefit on soil structure and organic matter content.

Mineral fertilizer, nitrogen in particular, can have both positive and negative effects on soil structure and organic matter. The potential improvement in soil aggregation associated with increased production of roots due to fertilization may not be fully realized if fertilization also causes increased rates of mineralization of the binding agents.^[4]

Common Ca amendments (e.g., lime, gypsum) can lead to soil structure formation and stabilization. The Ca functions as an agent that promotes bridging of microbial cells and clay minerals to form organomineral microstructures.^[2]

Soil Cultivation and Conservation Tillage

Long-term studies have shown that cultivation of native soils has resulted in a loss of soil organic carbon and a subsequent decline in soil structure.^[11] Some of this loss would be associated with a change in vegetation or cropping (e.g., from native grassland to annual grains) and the resulting decrease in organic matter inputs to the soil. Reducing the amount and degree of tillage intensity and retaining a soil cover of crop residues (i.e., conservation tillage; >30% soil residue cover) can lead to an accumulation of organic matter in the surface layer (5–10 cm). This increase in organic matter yields major benefits for improved soil structural stability and soil physical quality.^[2,5] However, in comparison, the mixing effect of tillage increases the association of mineral and organic particles, resulting in the beneficial formation of organomineral microstructures.^[12] Although in many cropping practices (especially conservation tillage), the mixing activity of soil macrofauna may complement, to some degree, the role of tillage.^[6]

REFERENCES

1. Gregorich, E.G.; Carter, M.R. *Soil Quality for Crop Production and Ecosystem Health*; Elsevier: Amsterdam, 1997.
2. Carter, M.R.; Stewart, B.A. *Structure and Organic Matter Storage in Agricultural Soils*; Lewis/CRC Press: Boca Raton, 1996.
3. Burke, W.; Gabriels, D.; Bouma, J. *Soil Structure Assessment*; A.A. Balkema: Rotterdam, 1986.
4. Kay, B.D. Rates of change of soil structure under different cropping systems. *Adv. Soil Sci.* **1990**, *12*, 1–52.
5. Carter, M.R.; Gregorich, E.G.; Angers, D.A.; Beare, M.H.; Sparling, G.P.; Wardle, D.A.; Voroney, R.P. Interpretation of microbial biomass measurements for soil quality assessment in humid temperate regions. *Can. J. Soil Sci.* **1999**, *79*, 507–520.
6. Lavelle, P.; Bignell, D.; Lepage, M.; Wolters, V.; Roger, P.; Ineson, P.; Heal, O.W.; Dhillon, S. Soil function in a changing world: The role of invertebrate ecosystem engineers. *Eur. J. Soil Biol.* **1997**, *33*, 159–193.
7. Low, A.J. Improvements in the structural state of soil under leys. *J. Soil Sci.* **1955**, *6*, 177–199.
8. Angers, D.A. Changes in soil aggregation and organic carbon under corn and alfalfa. *Soil Sci. Soc. Am. J.* **1992**, *56*, 1244–1249.
9. Tisdall, J.M.; Oades, J.M. Organic matter and water stable aggregates in soils. *J. Soil Sci.* **1982**, *33*, 141–163.
10. Aoyama, M.; Angers, D.A.; N'Dayegamiye, A. Particulate and mineral-associated organic matter in water-stable aggregates as affected by mineral fertilizer and manure applications. *Canadian J. Soil Sci.* **1999**, *79*, 295–302.
11. Mann, L.K. Changes in soil carbon storage after cultivation. *Soil Sci.* **1986**, *142*, 279–288.
12. Angers, D.A.; Bolinder, M.A.; Carter, M.R.; Gregorich, E.G.; Drury, C.F.; Liang, B.C.; Voroney, R.P.; Simard, R.R.; Donald, R.G.; Beyaert, R.P.; Martel, J. Impact of tillage practices on organic carbon and nitrogen storage in cool, humid soils of eastern Canada. *Soil Till. Res.* **1997**, *41*, 191–201.

Crop Rotation and Farming Systems: Tropical Smallholder Farm Systems

Stefan Hauser

Agronomy, International Institute of Tropical Agriculture, Ibadan, Nigeria

Lindsey Norgrove

Department of Environmental Sciences, University of Basel, Basel, Switzerland

Tarcisius Nyobe

Institute of Agricultural Research for Development, Nkolbisson, Cameroon

Abstract

Typically tropical smallholder farm systems are characterized by the use of fire as a clearance tool, the incorporation of a fallow phase, the lack of mechanization and external purchased inputs, and a high reliance of manual labor. While land husbandry practices are therefore relatively benign, farmers can be forced to cultivate marginal land, such as steep slopes, which might increase the risk of soil structural degradation. We outline the role of fallow and land husbandry techniques in determining soil structure. We then outline methods used by smallholder farmers that can minimize soil structural degradation. We then give examples of local methods used by smallholder farmers to rehabilitate structurally degraded or marginal soils to permit cultivation.

INTRODUCTION

Soil structure is the manner in which soil mineral particles and organic matter are spatially arranged into groups or aggregates and the arrangement of voids between the solid materials. It depends upon soil texture, clay type, the quality and quantity of soil organic matter inputs, and the activity of soil biota (e.g., microflora, plant roots, and soil fauna). Soil structure is characterized by porosity, pore size distribution, and pore continuity. These can be assessed by measurements of penetrometer resistance, soil bulk density, water infiltration rate, hydraulic conductivity, and soil aggregate stability.

SMALLHOLDER AGRICULTURE AND THE IMPORTANCE OF SOIL STRUCTURE

Most smallholders in the tropics practice shifting cultivation or other forms of “slash and burn” agriculture, so these are the focus of this entry. Slash and burn agriculture is a generic term for agricultural systems in which the fallow vegetation is manually slashed, left to dry, and cleared from the field by burning before crop cultivation.^[1] After a cropping phase, the land is abandoned to a fallow phase. Later, the cycle is repeated.

Relative to undisturbed forest or fallow land, generally, soil structure is degraded through clearance to agriculture. In South America, compared with undisturbed forest, continuous sugar cane production for 20 years on both Oxisols

and Alfisols resulted in reductions in wet aggregate stability at low water contents, threefold reductions in pore tortuosity, and thus higher unsaturated hydraulic conductivity. In the Oxisol, total porosity was reduced under cultivation, but in the Alfisol, land use change had no significant effect.^[2] Other studies in the region on Ultisols^[3] and Oxisols^[4] have found higher soil bulk density^[3,4] and low soil moisture content^[3] after slash and burn, compared to undisturbed forest. In West Africa, clearance and burning resulted in decreases in soil macroporosity and infiltration rates after both short and long fallows.^[5] In a drier environment in Southern Africa, clearance of miombo woodland to cropland generally decreased soil aggregate stability and resulted in reduced soil infiltration rates on clay but not on sandy soils.^[6]

However, such structural changes may not be the limiting factors in crop production. Many soils in the humid tropics are constrained by low nutrient reserves rather than by soil physical properties. For example, there were few differences in soil physical properties between “productive” and “non-productive” land as classified by smallholder farmers in central Kenya, although there were substantial differences in soil chemical properties.^[7] In most tropical soils, most nutrients are concentrated in the shallow topsoil, subtended by the nutrient-poor and acid subsoil. Thus, the retention of the topsoil is of paramount importance for sustained crop production. Land use practices that cause structural collapse reduce the topsoil’s ability to absorb and conduct water. On sloping land, this leads to surface runoff and soil erosion, which can completely

remove the topsoil layer, exposing infertile subsoil unsuitable for agriculture and unlikely to be reclaimed by vegetation.

The Importance of Fallow

Slash and burn smallholder systems can be subdivided according to the fallow length employed and the climatic zone in which they are practiced. The basis of the processes leading to soil aggregation during fallow phases is biomass production as an energy supply. Litterfall and root turnover apply and incorporate biomass into the soil. Therefore, generally soil organic matter levels increase with time under fallow.^[8] The fallow vegetation covers the land, preventing particle detachment by wind and rain. Shade and the increased soil organic matter levels encourage macrofaunal activity.^[9] Mites, collembola, earthworms, termites, and ants burrow in the soil generating macropores connected to the soil surface. These pores provide for rapid water infiltration during intensive rainstorms.^[10]

The Importance of Fallow Length

In some areas of the humid tropics, farmers clear primary or secondary forest and return to the same land after a forest fallow phase of 10–30 years. These systems are perceived to be sustainable as fallow recovery is complete.^[1] Indeed, in West and Central Africa, farmers most frequently rank fallow age first of the criteria they use to determine which parcel to recultivate.^[11] However, many smallholder farmers face a shortage of land, so fields are cleared after shorter, suboptimal fallow periods,^[12] with insufficient regeneration.

The Impact of Climatic Zone

With decreasing rainfall, the climax vegetation shifts from forest to savanna, shrubland, and grassland. A minimum amount of time is required to reestablish the climax vegetation after clearing and cropping. In areas with low rainfall, biomass production is limited. Soil protection relies on other, mainly physical measures preventing or reducing soil exposure to forces degrading soil structure. Windbreaks, stone bunds, ridges, and mulching are measures used by farmers to reduce wind erosion, some of them indirectly through water harvesting and retention.

HOW SMALLHOLDER FARMERS CAN INFLUENCE SOIL STRUCTURE

Smallholders can modify soil structure only through the promotion of soil aggregating factors and processes increasing soil porosity. These factors are largely biological processes, e.g., plant root growth, aboveground and belowground plant biomass decomposition, the breakdown of

organic matter by soil microbes, forming humic substances, and soil mesofauna and macrofauna, such as collembola, mites, termites,^[13] and earthworms,^[14] which ingest and mix mineral soil particles and organic matter. The selection of the land, clearing method employed, whether the land is burned or mulched and tilled or not, the use of inputs, and the management of fallow are the smallholder's tools to balance soil structural degradation vs. structural regeneration.

Site Selection: Avoiding Slopes and Soil with Low Structural Resilience

Slash and burn farmers have topographic preferences. In Southern Cameroon, on structurally unstable Ultisols, farmers prefer the plateaux and avoid cropping even on gentle slopes.^[1] However, farmers' choices are increasingly limited by the type of land available, so unsuitable land is taken into crop production, increasing the risk of degradation.

Clearing Method

Smallholder farmers generally clear bush or forest manually, using machetes and, in some cases, chainsaws. Compaction, which would occur with mechanized clearing by bulldozer, is avoided. For example, in the Amazon basin, the effects of mechanical and manual clearance were compared.^[15] Prior to clearance, infiltration rates were 420 mm h⁻¹. Manual clearance reduced infiltration rates to 304 mm h⁻¹, whereas mechanical clearing reduced infiltration rates to 32 mm h⁻¹. All clearing methods increased topsoil bulk density to 15 cm depth; however, only mechanical clearing caused significant increases to 25 cm depth. There was no effect upon soil texture. Similar results were obtained in analogous experiments on an Ultisol in Southern Nigeria.^[16]

In the humid forest zone, farmers traditionally retain certain trees in their fields. In Southern Cameroon, persistent pioneers such as *Ceiba pentandra*, *Triplochiton scleroxylon*, and *Terminalia superba* are most often retained to create a level of shade deemed appropriate for some of the more shade-tolerant food crops cultivated, such as plantain. Farmers also consider that such trees maintain soil fertility through leaf litterfall and subsequent decomposition.^[17] Retention of living trees, as compared to clear-felling them, was shown to maintain lower soil bulk density and higher infiltration rates on a Southern Cameroonian Ultisol.^[18]

Burning vs. Mulching

In burning experiments conducted in Southern Cameroon, burning had no effect on bulk density and penetrometer resistance.^[1] It has been reported in the literature that the burn loosens the soil and facilitates tillage and planting. Direct measurements in Southern Cameroon could not confirm these reports, and the reports might refer to the lack of litter and shallow root biomass, rather than any changes in the soil. Where hot burns are attained, such as under a log,

compaction and soil structural collapse can occur due to a relative excess of monovalent cations, leading to a breakdown of cation bridges between clay minerals, the combustion of soil organic carbon and thus substances binding soil particles, and the elimination of soil macrofauna and microfauna mixing soil particles and organic materials to form stable aggregates.

After land clearing and burning, the soil surface is exposed to the impact of raindrops. The retention of biomass would reduce the impact of raindrops and through decomposition would contribute to soil structural maintenance. Mulch layers prevent detachment of soil particles and are conducive for soil biota such as earthworms, which produce stable soil aggregates by casting.

In semiarid zones, where mulch is applied to crusted soil, termite activity can be triggered, and this can have positive effects upon soil structure. In Burkina Faso, in plots mulched with a wood–straw mixture, saturated soil hydraulic conductivity was $3.6 \times 10^{-5} \text{ m s}^{-1}$ when termites were present and was $0.34 \times 10^{-5} \text{ m s}^{-1}$ when termites were excluded.^[19]

However, mulching is not a generally applicable solution, because not all crops respond well to a mulch layer and require a “clean” soil surface, and there is also incompatibility between the use of tillage and *in situ* mulch.^[20] However, for other crops, such as plantain, mulching can be universally recommended.^[21] In systems without fallow phases, mulch has to be produced during cropping phases by a cover crop or intercropped tree or shrub hedgerows (see the section “Improved Planted Fallows”).

Tillage vs. No Tillage

Farmers have various reasons for tillage, depending on site conditions and the intended crop. Tillage can loosen crusted and compacted soil, increase the depth of the loose soil layer, mix ash and residue into the soil to avoid nutrient losses by runoff, and concentrate topsoil through mounding. Breaking-up of soil aggregates increases the decomposition rate of soil organic matter, previously physically protected, and thus contributes to increased nutrient availability. Accordingly, the purposes of tillage range from water harvesting to nutrient supply and accumulation in small areas. It can generally be assumed that tillage does not contribute to soil aggregation. Its loosening effect is temporary, and rain may disintegrate soil aggregates further, leading to more severe crusting and compaction. Some crops, however, do require tillage, and this exposes the soil and can lead to structural degradation.

Under humid conditions, tillage is mainly serving the requirements of certain crops. Groundnut (*Arachis hypogaea*), for instance, requires its pegs (fertilized ovules) to penetrate the soil. To form grains, the crop has to take up Ca^{2+} through the pod walls. Therefore, the soil is tilled with hand hoes, mixing Ca^{2+} -rich ash with the soil at seeding. Other crops may require a deeper layer of loose soil than is

available. This requirement is achieved by mounding for crops such as yam (*Dioscorea spp.*), which form a tuber that grows with a blunt end vertically into the soil. Any obstacle will stop tuber growth and lead to deformed, unmarketable tubers. Other crops planted in smaller mounds are sweet potatoes (*Ipomoea batatas*) and tannia (*Xanthosoma sagittifolium*), which may not require a deeper layer of loose soil but may positively respond to an increased amount of nutrients. Furthermore, tubers are more easily harvested from mounds than flat soil, especially during dry seasons, when soils are hard.

A study summarizing more than 20 years of research in semiarid Southern Africa on tillage in maize systems assessed the impact of “traditional” animal draught tillage versus manual “conservation” tillage in maize concluded that its impact was dependent on the general yield level of the farm and the rainfall in the area.^[22] At low yields (less than 2.5 Mg ha^{-1}) and low rainfall (less than 500 mm yr^{-1}), conservation tillage conferred a yield advantage. Above these levels, conventional tillage was advantageous, although results were variable. In the forest–savanna transition zone of central Cameroon, tractor tillage in tomato systems resulted in significant reductions in aggregate stability, lower aggregate mean size, and higher meso-aggregate proportions, compared with manual tillage and no tillage. The effects of the last two techniques could not be distinguished.^[23]

Manuring, Compost, and Fertilizer Application

Although primarily used to apply nutrients, the high organic matter content in manures and composts has positive effects on soil aggregation. This is mainly caused by the processes of decomposition and ingestion and egestion by soil mesofauna and macrofauna. In drier areas, manure is usually incorporated by tillage (see the section “Tillage”) to facilitate biological activity which would be hampered at the surface due to drought. The major constraints are the low quantities of manure and compost available and the high labor requirement for transport and incorporation. Organic matter inputs will increase if crop biomass production increases. For most crops, only a proportion of the plant is harvested and various amounts of biomass will remain in the field (unless required for livestock feeding). Judicious fertilizer use can increase biomass production, crop yields, and thus the amount of organic matter returned to the soil. Furthermore, a fertilized crop may grow more vigorously and cover the soil earlier and, more intensively, reduce the negative impact of rain.

Improved Planted Fallows

If shortened natural regrowth fallow is not capable of restoring soil properties to a level at which adverse effects on long-term suitability of the soil for crop production can be avoided, planted fallows may be an alternative. There

are two main types of planted fallow: herbaceous legume fallow and tree-based or shrub-based fallow. In southwestern Nigeria, under *Pueraria phaseoloides* live mulch and *Leucaena leucocephala* alley cropping, aggregate stability, measured as mean weight diameter (MWD), was higher than in natural regrowth system when both were continuously cropped.^[24] The differences between the fallow types diminished with increasing fallow length. MWD variation over time was less under *P. phaseoloides* live mulch. On a Southern Cameroonian Ultisol, increased infiltration rates, reduced soil temperature, and reduced penetrometer resistance occurred in a tilled alley cropping system using *Senna spectabilis* compared with a tilled, no-tree control.^[25] However, planted fallows do not always have more positive effects on soil physical properties than on the natural regrowth.^[24]

EXAMPLES OF INDIGENOUS SYSTEMS OF SOIL STRUCTURAL MANAGEMENT

Traditional smallholder management techniques can be divided into those that, whether inadvertently or directly, aim to maintain soil structure and avoid degradation and those that are used in already-degraded environments or on marginal soils to try to aid soil rehabilitation. From a systems perspective, changes in smallholder systems are probably non-linear and prone to hysteresis effects,^[26] and thus, small changes in management might provoke a cascade of effects resulting in a downward degradative spiral. Once a system is degraded, huge efforts might be required to return to achieve the previous state. Many of the techniques in the former category, such as maintaining trees in fields, mulching, and using planted fallows, have been dealt with in the previous sections. Some of those in the latter category will be described below.

In the Sahel, smallholder cereal farmers use a traditional technique of pits referred to as *Zai* in Burkina Faso, *Tassa* in Niger and *Towalen* in Mali.^[27] This is used on soils, known as *zipellé* in Burkina Faso, which have lost their structure and have become sealed and crusted. Farmers dig small holes of 15–30 cm diameter and about 10–20 cm deep. During the dry season, fine sand and litter are trapped in these holes. Farmers apply animal manure and plant residues, attracting termites. The termites build stable channels in the soil under the manure to access it and distribute some of the organic material. Once the rains commence, the holes collect water, rapidly conducted in the termite channels, thus avoiding runoff. The fine sand can retain more water than coarse sand, while the organic layer reduces evaporation, thus reducing the risk of seeds failing to germinate. Sorghum, millet, and cowpea are commonly planted in the *Zai*.^[28] Crop yields can be more than tenfold higher than in no *Zai* controls, and the holes can be reused for 3 years.^[29] Exclusion experiments have demonstrated the critical role of termites in the system as their activity

increased hydraulic conductivity and soil porosity, and these differences were amplified when additional mulch was added to further stimulate termite activity.^[19]

In Tanzania, similar pits called “Ngoro” have been traditionally used for hundreds of years by the Matengo people in southern Tanzania to permit cultivation of Oxisols and Ultisols on steep slopes.^[30] The pits are larger than those used in the Sahel and are traditionally 1.5 m × 1.5 m square by down to 0.5 m deep, although sizes are variable. Plant residues such as cut grasses are buried under the pits, and maize is cultivated on the ridges of the pit where the soil from the pit is piled. Although runoff and erosion occur, the losses are small because most of the soil is deposited in the pits. Ngoro production systems have been documented to effectively conserve water, soil, and nutrients on sloping land.^[30]

ACKNOWLEDGMENT

Lindsey Norgrove is supported by the Swiss National Science Foundation through a Marie Heim-Vögtlin Fellowship.

REFERENCES

1. Hauser, S.; Norgrove, L. Slash and burn agriculture, effects of. In *Encyclopedia of Biodiversity*, 2nd Ed.; Levin, S., Ed.; Academic Press: Waltham, 2013; Vol. 6, 551–562.
2. Caron, J.; Espindola, C.R.; Angers, D.A. Soil structural stability during rapid wetting: Influence of land use on some aggregate properties. *Soil Sci. Soc. Am. J.* **1996**, *60* (3), 901–908.
3. Comte, I.; Davidson, R.; Lucotte, M.; de Carvalho, C.J.R.; de Assis Oliveira, F.; da Silva, B.P.; Rousseau, G.X. Physicochemical properties of soils in the Brazilian Amazon following fire-free land preparation and slash-and-burn practices. *Agr. Ecosyst. Environ.* **2012**, *156*, 108–115.
4. Perrin, A.S.; Fujisaki, K.; Petitjean, C.; Sarrazin, M.; Godet, M.; Garric, B.; Horth, J.-C.; Balbino, L.C.; Silveira Filho, A.; Machado, P.L.O.A.; Brossard, M. Conversion of forest to agriculture in Amazonia with the chop-and-mulch method: Does it improve the soil carbon stock? *Agr. Ecosyst. Environ.* **2014**, *184*, 101–114.
5. Are, K.S.; Oluwatosin, G.A.; Adeyolanu, O.D.; Oke, A.O. Slash and burn effect on soil quality of an Alfisol: Soil physical properties. *Soil Till. Res.* **2009**, *103* (1), 4–10.
6. Mapanda, F.; Munotengwa, S.; Wuta, M.; Nyamugafata, P.; Nyamangara, J. Short-term responses of selected soil properties to clearing and cropping of miombo woodlands in central Zimbabwe. *Soil Till. Res.* **2013**, *129*, 75–82.
7. Murage, E.W.; Karanja, N.K.; Smithson, P.C.; Woomer, P.L. Diagnostic indicators of soil quality in productive and non-productive smallholders' fields of Kenya's central highlands. *Agr. Ecosyst. Environ.* **1999**, *79* (1), 1–8.
8. Kass, D.C.L.; Somarriba, E. Traditional fallows in Latin America. *Agroforest. Syst.* **1999**, *47*, 13–36.

9. Hauser, S. Distribution and activity of earthworms and contribution to nutrient recycling in alley cropping. *Biol. Fert. Soils* **1993**, *15*, 16–20.
10. Hauser, S.; Asawalam, D. Effects of fallow system and cropping frequency upon quality and composition of earthworm casts. *Z. Pflanz. Bodenkunde* **1998**, *161*, 23–30.
11. Norgrove, L.; Hauser, S. Biophysical criteria used by farmers for fallow selection in West and Central Africa. *Ecol. Indic.* **2016**, *61* (1), 141–147.
12. Van Vliet, N.; Mertz, O.; Heinemann, A.; Langanke, T.; Pascual, U.; Schmook, B.; Adams, C.; Schmidt-Vogt, D.; Messerli, P.; Leisz, S.; Castella, J.-C.; Jørgensen, L.; Birch-Thomsen, T.; Hett, C.; Bech-Bruun, T.; Ickowitz, A.; Chi, V.K.; Yasuyuki, K.; Fox, J.; Padoch, C.; Dressler, W.; Ziegler, A.D. Trends, drivers and impacts of changes in swidden cultivation in tropical forest-agriculture frontiers: A global assessment. *Global Environ. Chang.* **2012**, *22* (2), 418–429.
13. Mando, A. The impact of termites and mulch on the water balance of crusted Sahelian soil. *Soil Technol.* **1997**, *11* (2), 121–138.
14. Hauser, S.; Norgrove, L.; Asawalam, D.; Schulz, S. Effect of land use change, cropping systems and soil type on earthworm cast production in West and Central Africa. *Eur. J. Soil Biol.* **2012**, *49*, 47–54.
15. Alegre, J.C.; Cassel, D.K.; Bandy, D.E. Effects of land clearing and subsequent management on soil physical properties. *Soil Sci. Soc. Am. J.* **1986**, *50* (6), 1379–1384.
16. Ghuman, B.S.; Lal, R.; Shearer, W. Land clearing and use in the humid Nigerian tropics: I. soil physical properties. *Soil Sci. Soc. Am. J.* **1991**, *55* (1), 178–183.
17. Carrière, S.M. Orphan trees of the forest: Why do Ntumu farmers of Southern Cameroon protect trees in their swidden fields. *J. Ethnobiology* **2002**, *22* (1), 133–162.
18. Hulugalle, N.R.; Ndi, J.N. Contributory factors to soil spatial variability in an ultisol. II. Retention of living trees *in situ* following land clearing. *Commun. Soil Sci. Plan.* **1993**, *24*, 1409–1419.
19. Mando, A.; Van Rheenen, T. Termites and agricultural production in the Sahel: From enemy to friend? *Neth. J. Agr. Sci.* **1998**, *46* (1), 77–85.
20. Erenstein, O. Smallholder conservation farming in the tropics and sub-tropics: A guide to the development and dissemination of mulching with crop residues and cover crops. *Agr. Ecosyst. Environ.* **2003**, *100* (1), 17–37.
21. Norgrove, L.; Hauser, S. Improving plantain (*Musa* spp. AAB) yields on smallholder farms in West and Central Africa. *Food Secur.* **2014**, *6* (4), 501–514.
22. Nyakudya, I.W.; Stroosnijder, L. Conservation tillage of rainfed maize in semi-arid Zimbabwe: A review. *Soil. Till. Res.* **2015**, *145*, 184–197.
23. Tueche, J.R.; Norgrove, L.; Hauser, S.; Cadisch, G. Tillage and varietal impacts on tomato (*Solanum lycopersicum* L.) production on an ultisol in central Cameroon. *Soil. Till. Res.* **2013**, *128*, 1–8.
24. Salako, F.K.; Babalola, O.; Hauser, S.; Kang, B.T. Soil macroaggregate stability under different fallow management systems and cropping intensities in Southwestern Nigeria. *Geoderma* **1999**, *91*, 103–123.
25. Hulugalle, N.R.; Ndi, N.J. Effects of no-tillage and alley cropping on soil properties and crop yields in a typical kandiudult of southern Cameroon. *Agroforest. Syst.* **1993**, *22*, 207–220.
26. Tittonell, P. Livelihood strategies, resilience and transformability in African agroecosystems. *Agr. Syst.* **2014**, *126*, 3–14.
27. Lahmar, R.; Bationo, B.A.; Lamso, N.D.; Guéro, Y.; Tittonell, P. Tailoring conservation agriculture technologies to West Africa semi-arid zones: Building on traditional local practices for soil restoration. *Field Crop. Res.* **2012**, *132*, 158–167.
28. Maatman, A.; Sawadogo, H.; Schweigman, C.; Ouedraogol, A. Application of zaï and rock bunds in the northwest region of Burkina Faso: Study of its impact on household level by using a stochastic linear programming model. *Neth. J. Agr. Sci.* **1998**, *46* (1), 123–136.
29. Roose, E.; Kabore, V.; Guenat, C. Zaï practice: A West African traditional rehabilitation system for semiarid degraded lands, a case study in Burkina Faso. *Arid Soil Res. Rehab.* **1999**, *13* (4), 343–355.
30. Malley, Z.J.U.; Kayombo, B.; Willcocks, T.J.; Mtakwa, P.W. Ngoro: An indigenous, sustainable and profitable soil, water and nutrient conservation system in Tanzania for sloping land. *Soil. Till. Res.* **2004**, *77* (1), 47–58.

Crop Rotation and Farming Systems: Tropics and Subtropics and Swelling Soils

D.M. Freebairn

Department of Natural Resources, Agricultural Production Systems Research Unit (APSRU), Toowoomba, Queensland, Australia

G.D. Smith

Agricultural Production Research Systems Unit, Queensland Department of Natural Resources and Mines, Indooroopilly, Queensland, Australia

Robin D. Connolly

URS Pty. Ltd., East Perth, Western Australia, Australia

Jeff N. Tullberg

Farm Mechanisation, CTF Solutions, Gatton, Queensland, Australia

Abstract

Cropping swelling soils in the tropics and subtropics is a challenge of managing climate variability and developing systems that can respond to this variability. Fixed rotations may not suit this combination of soil and climate. As water supplier is a commonly limiting factor in sustainable crop production, optimizing water storage and use is a central theme of good systems. Optimizing water entry is achieved by retaining a surface cover on the soil, especially when high energy rainfall is expected. Systems that are responsive to water availability make the best use of rainfall and minimize off-site impacts of erosion. Inclusion of a pasture phase in rotations can be important in maintaining soil organic carbon and structure.

INTRODUCTION

Large areas of swelling clay soils (Vertisols) occur in arid and semiarid tropical and subtropical regions of Africa, India, and Australia and are increasingly being relied on to produce food for the growing population. Farmers of Vertisols are often dealing with poor resources and face many risks and constraints associated with soil and weather interactions. These regions usually have wet and dry seasons, but rainfall may be highly variable between years and within seasons. Intense rains may follow long dry periods. In some areas, a high proportion of annual rainfall may be in light falls that do not wet below the surface and are quickly lost by evaporation. In other areas, there may be long wet periods in which waterlogging and weed growth present major problems for farmers.

Vertisols have long been recognized as difficult to farm, because they are soft and sticky when wet and hard, and intractable when dry. The swelling characteristics arise from the high proportion of montmorillonite clays. Forces associated with the interaction between water and the highly active clay particles are extremely powerful and usually predetermine soil structural relations in the soils. Management inputs will have the best effect if they work in

harmony with these forces. For example, the aggregation characteristics strongly influence the effects of tillage and the nature of pore spaces available for roots and water entry.

Cropping Vertisols alter water and carbon balances. For example, changes in soil cover and hydrology as a result of cultivation can result in doubling of annual runoff and increasing soil erosion by a factor of 100.^[1] Also, tillage reduces organic matter in the soil surface,^[2] while the use of large tractors compacts subsurface soil.^[3] Fallowing is a key management tactic in rain-fed farming. A distinguishing feature of Vertisols is their very high infiltration capacity when dry and cracked, in contrast to low infiltration rates when wet.^[3] Low infiltration rates are generally a result of a crusted or saturated surface, a wet profile,^[4] or a poor internal drainage due to compaction or loss of structural voids. High erosion rates associated with bare soil fallows are clearly unsustainable, far exceeding likely soil formation rates.^[1] Loss of productivity associated with shallow soils and nutrient depletion results in lower biomass production and declines in organic stores in soil. While all soils are vulnerable to compaction, the ability of clay soils to store water for long periods makes them particularly susceptible, with subsoils being above the plastic limit for much of the time.^[5] The hidden nature of

subsoil compaction and variable crop responses make the diagnosis and quantification of impacts of compaction difficult.

OPTIONS FOR MANAGING SOIL STRUCTURE

Retaining Soil Cover

Soil cover (from growing crops and their residues) and reduced tillage maintain infiltration by reducing aggregate disruption by raindrops.^[6,7] The amount of crop residue available to provide surface cover is determined by crop type, yield, and tillage method.^[8] Reduced tillage and no-till practices result in less stubble breakdown and minimize the deleterious features of tillage while maximizing water storage.

Maintenance of soil cover can reduce erosion rates by an order of magnitude,^[1] thus protecting the soil resource base. Tillage systems with high nutrition levels and crop residue retention can also maintain higher organic carbon levels.^[9]

Rotations to Manage Soil Moisture Deficit

Soil moisture content is the most important factor in determining infiltration into cracking clay soils. Although seasonal conditions have a strong and uncontrollable influence on soil moisture, the sequence and number of crops grown have a major effect on the timing of soil water deficits during the year and hence on runoff.^[10] Rotations that grow more crops, or crops grown during the wetter season, have lower runoff and deep drainage. One element of a system to improve water use efficiency includes planting of crops before the onset of the wet season to make better use of incident rainfall.^[11] Increasing cropping intensity can also result in greater total productivity although the yields per crop may be reduced.^[12] The concept of “opportunity” or “response” cropping involves matching cropping intensity with rainfall expectancy and moisture conditions. Crop rotations are based more on existing conditions rather than on a fixed pattern, thus exploiting above average rainfall conditions when they occur. Rotations that maintain a higher soil water deficit are also less prone to compaction damage due to being below the plastic limit for a greater proportion of the time.

Exploiting Soil Cracking

The cracking nature of Vertisols offers potential to capture intense falls of rain that might otherwise be lost as runoff. Rapid water movement to depth via cracks can occur during high intensity rainfall and provides a means for improving the efficiency of water storage. Tillage should be avoided as it may close cracks that

would otherwise channel water below the highly evaporative surface soil.

Tillage and Roughness

Tillage, although superimposed on structural arrangements produced by wetting and drying, influences aggregate and pore size distribution, roughness, residue cover, and strength of the surface soil.^[13] Tillage can increase infiltration by breaking surface crusts and by increasing surface porosity and surface storage capacity.^[14] In some situations, smooth soil surfaces associated with no-till systems can result in higher runoff compared to tilled soil.^[6] Depression storage can be created using specialized tillage equipment. Its function is to store excess rainfall, allowing greater time for infiltration. Such storage is variously referred to as furrow dikes, tied ridges, or pits. A particular example of surface configuration used to improve productivity and reduce risk is the broad bed and furrow system developed to allow double-cropping on Vertisols at the International Crops Research Institute for the Semi-Arid Tropics.^[14] This system reduces the risk of waterlogging and, by controlling runoff, reduces soil erosion.^[15]

Deep tillage (20–40 cm depth) has been practiced to reduce runoff, an improved water storage and root growth, but results have been variable. Deep ripping (35 cm) prior to wheat and sorghum on a poorly structured clay in southern Queensland has only short-term benefits, not sufficient to justify the cost.^[16] In contrast, deep tillage reduced runoff and erosion on a steep (14%) cracking soil in Southern Italy.^[17] The variability in results between studies suggests caution in applying deep tillage.

Zonal Wheel Traffic to Reduce Compaction

The possibilities of conserving energy and time associated with confining wheel traffic to permanent laneways are being explored to determine whether there were improvements in soil physical conditions through removal of tractor wheel loads and compaction.^[18] The system involves creating permanent tracks for most machinery operations to follow. Global positioning system technology is being employed to guide machinery operations. Advantages of this “controlled traffic” system are as follows: less overlap of implements leading to greater efficiency, lower draft requirements,^[3] more timely planting and spray operations, and less compaction of the majority of the crop-growing soil mass. A trafficked micro-catchment yielded considerably more runoff (30–60 mm) compared to an untrafficked area.^[19] The energy implications of controlled traffic also make the system attractive by improving the efficiency in the application of seed and chemicals. Soil benefits are not always expressed as improvements in yield^[19] although consistent yield advantages have been observed on a clay loam.^[20]

Pasture Phases to Rebuild Organic Carbon and Structure

Soil organic matter declines with cultivation, with associated loss of nutrient supply capacity and soil structure.^[2] Incorporating a pasture ley in a cropping system has the capacity to increase soil organic matter and improve infiltration characteristics.^[21,22] The use of pasture leys in Mediterranean cropping systems is well established but is not common in clay soils in the subtropics.

With many interactions between soil, weather, and crops in cropping systems, the design of improved cropping systems is difficult to explore when based wholly on experimentation. Results from a sequence of seasons may not be representative, and options are restricted by logistics. Cropping system models can be valuable tools in the exploration of new options for improving both the economic and natural resource sustainability of alternative rotations and management systems. As an example, the APSIM-SWIM model^[23,24] was used to explore interactions between changes in soil properties resulting from rotations of crops and pastures, providing predictions of variables such as soil organic carbon and wheat crop yield.^[25] Such models allow extrapolation of experimental results beyond the experience and locations where detailed experiments have been conducted. In general, cropping systems that have high fertilizer inputs and less tillage result in improvements in soil organic matter, infiltration characteristics, and yield. Analysis of time series of costs and returns then allows for a robust analysis of economic performance.

CONCLUSION

Soil structure in swelling soils is closely related to soil-weather-plant interactions. These interactions need to be taken into consideration when designing improved soil-crop strategies. The manipulation of soil cover and moisture content can be used to improve infiltration and moisture conservation, leading to better production and reduced soil erosion. Tillage systems that reduce soil compaction improve both production efficiency and soil function. With the incorporation of pasture leys in cropping systems, an improvement in the organic carbon status can be achieved. Relatively slow soil responses to pasture phases make such management options economically difficult for many land managers. System models provide a means to explore some of the biophysical interactions in cropping-tillage systems. When used in conjunction with field experimentation, new management systems can be efficiently developed.

REFERENCES

- Freebairn, D.M.; Wockner, G.H. A study of soil erosion on vertisols of the eastern darling downs, Queensland. I. The effect of surface conditions on soil movement within contour bay catchments. *Aust. J. Soil Res.* **1986**, *24*, 135–158.
- Dalal, R.C.; Mayer, R.J. Long-term trends in fertility of soils under continuous cultivation and cereal cropping in southern Queensland. II. Total organic carbon and its rate of loss from the soil profile. *Aust. J. Soil Res.* **1986**, *24*, 281–292.
- Freebairn, D.M.; Loch, R.J.; Cogle, A.L. Tillage methods and soil and water conservation in Australia. *Soil Till. Res.* **1993**, *27*, 303–325.
- Freebairn, D.M.; Loch, R.J.; Glanville, S.F.; Boughton, W.C. Use of simulated rain and rainfall-runoff data to determine final infiltration rates for a heavy clay. In *Properties and Utilisation of Cracking Clay Soils*; McGarity, J., Hoult, E.H., So, H.B., Eds.; Reviews in Rural Science No. 5; Univ. of New England: New South Wales, 1984; 348–351.
- McGarry, D.; Sharp, G.; Bray, S.G. *The Current Status of Soil Structural Degradation in Queensland Cropping Soils*; Project Report DNRQ990092; Department of Natural Resources: Queensland, 1999.
- Hewitt, J.S.; Dexter, A.R. Effects of tillage and stubble management on the structure of a swelling soil. *J. Soil Sci.* **1980**, *31*, 203–215.
- Loch, R.J. Effects of fallow management and cropping history on aggregate breakdown under rainfall for a range of Queensland soils. *Aust. J. Soil Res.* **1994**, *32*, 1125–1139.
- Sallaway, M.M.; Lawson, D.; Yule, D.F. Ground cover during fallow of wheat, Sorghum and sunflower stubble under three tillage practices in central Queensland. *Soil Till. Res.* **1988**, *12*, 347–364.
- Marley, J.M.; Littler, J.W. Winter cereal production on the darling downs—An 11 year study of fallowing practices. *Aust. J. Exp. Agr.* **1989**, *29*, 807–827.
- Adams, J.E.; Henderson, R.C.; Smith, R.C. Interpretations of runoff and erosion from field-scale plots on Texas Blackland soil. *Soil Sci.* **1959**, *87*, 232–238.
- Krantz, B.A. Technological packages and approaches to management for rainfed agriculture in semi-arid tropics. In *Proceedings of the Agricultural Sector Symposia*, Jan 7–11, 1980; 1–29.
- Berndt, R.D.; White, B.J. A simulation-based evaluation of three cropping systems on cracking clay soils in a summer-rainfall environment. *Agr. Meteorol.* **1976**, *16*, 211–229.
- Smith, G.D.; Yule, D.F.; Coughlan, K.J. Soil physical factors in crop production on vertisols in Queensland, Australia. In *Proceedings of the International Workshop on Soils*, Sept 12–16, 1983, Townsville, Queensland. Australian Centre for International Agricultural Research: Canberra, 1983; 87–104.
- El-Swaify, S.A.; Pathak, P.; Rego, T.J.; Singh, S. Soil management for optimized productivity under rainfed conditions in the semi-arid tropics. *Adv. Soil Sci.* **1985**, *1*, 1–64.
- Kampen, J.; Hari Krishna, J.; Pathak, P. Rainy season cropping on deep vertisols in the semi-arid tropics—Effects on hydrology and soil erosion. In *Tropical Agricultural Hydrology*; Lal, R., Russell, E.W., Eds.; Wiley: New York, 1981; 257–271.
- Radford, B.J.; Gibson, G.; Nielsen, R.G.H.; Butler, D.G.; Smith, G.D.; Orange, D.N. Fallowing practices soil water storage, soil nitrate accumulation and wheat performance in South West Queensland. *Soil Till. Res.* **1992**, *22*, 73–93.

17. Postiglione, L.F.; Basso, M.; Amato, M.; Carone, F. Effects of soil tillage methods on soil losses, soil characteristics and crop productions in hilly areas of southern Italy. In *Advanced Soil Mechanics and Related Soil Properties*; Proceedings of the NATO Conference; Larson, W.E., Blake, G.R., Allmaras, R.R., Voorhes, W.B., Gupta, S.C., Eds.; Series E: Applied Science Series; Kluwer Academic: London, 1988; Vol. 172.
18. Yule, D.F.; Tullberg, J.N., Eds. Proceeding of a National Controlled Traffic Conference, Sept 13–14, 1995; Department of Primary Industries: Rockhampton, 1985; 209 pp.
19. Tullberg, J.N.; Ziebarth, P.J.; Li, Y. Tillage and traffic effects on runoff. *Aust. J. Soil Res.* **2001**, *39*, 249–257.
20. Tullberg, J.N. Controlled traffic for sustainable cropping. In Proceedings of the 10th Australian Agronomy Conference, Hobart, 2001; 12 pp.
21. Connolly, R.D.; Freebairn, D.M.; Bridge, B.J. Change in infiltration characteristics associated with cultivation history of soils in southeastern Queensland. *Aust. J. Soil Res.* **1997**, *35*, 1341–1358.
22. Connolly, R.D.; Freebairn, D.M.; Bell, M.J. Change in soil infiltration associated with leys in south-eastern Queensland. *Aust. J. Soil Res.* **1998**, *36*, 1057–1072.
23. McCown, R.L.; Hammer, G.L.; Hargreaves, J.N.G.; Holzworth, D.P.; Freebairn, D.M. APSIM: A novel software system for model development model testing and simulation in agricultural systems research. *Agr. Syst.* **1996**, *50*, 255–271.
24. Verberg, K., Ed. *Methodology in Soil Water and Solute Balance Modelling: An Evaluation of the APSIM-SoilWat and SWIMv2 Models*; Divisional Report No. 131; CSIRO Division of Soils: Canberra, 1996.
25. Connolly, R.D.; Freebairn, D.M.; Bell, M.J.; Thomas, G. Effects of rundown in soil hydraulic condition on crop productivity in south eastern Queensland — A simulation study. *Aust. J. Soil Res.* **2001**, *39*, 1111–1129.

BIBLIOGRAPHY

1. Freebairn, D.M. Erosion control – some observations on the role of soil conservation structures and conservation. *Nat. Res. Man.* **2004**, *7*, 8–13.
2. Freebairn, D.M.; Connolly, R.D.; Dimes, J.; Wylie, P. Crop sequencing. In *Sustainable Crop Production in the Subtropics: An Australian Perspective*; Clarke, A.L., Wylie, P.B., Eds.; Department of Primary Industries Information: Brisbane, 1998; 289–308.
3. Freebairn, D.M.; King, C.A. Reflections on collectively working toward sustainability: Indicators for indicators! *Aust. J. Exp. Agr.* **2003**, *43*, 223–238.
4. Keating, B.A.; Carberry, P.S.; Hammer, G.L.; Probert, M.E.; Robertson, M.J.; Holzworth, D.; Huth, N.I.; Hargreaves, J.N.G.; Meinke, H.; Hochman, Z.; McLean, G.; Verburg, K.; Snow, V.; Dimes, J.P.; Silburn, M.; Wang, E.; Brown, S.; Bristow, K.L.; Asseng, S.; Chapman, S.; McCown, R.L.; Freebairn, D.M.; Smith, C.J. An overview of APSIM, a model designed for farming systems simulation. *Eur. J. Agron.* **2003**, *18*, 267–288.

Crop Yield: Compaction

Muhammad Ishaq

Soil and Water Testing Laboratory, Soil Fertility Research Institute, Sahiwal, Pakistan

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Soil compaction is a global problem and is a severe factor in crop production. This entry describes the consequences of soil compaction on crop yields, and discusses methods and strategies of alleviating and minimizing risks of soil compaction. Soil compaction is the process of rearrangement of soil particles, leading to decrease in total pore space and pore size, and increase in bulk density. It is expressed in terms of dry bulk density, porosity, hydraulic conductivity, soil strength, infiltration rate, and crop yield. It may be tillage-induced or traffic-induced. Compaction in the plow layer is largely related to the contact pressure of the tires, and that in the subsoil to the axle load. Compaction affects soil properties and processes, and influences long-term soil productivity as well as resource economy and environmental quality. Effects of compaction on plant growth and yield depend on soil, crop, and climatic conditions. The crop response to the traffic varied between sites and years, and there was a persistent negative crop response to high axle-load traffic. Compaction increases soil strength that adversely affects root growth and reduces crop yields. Use of large and heavy farm equipment increases risks of reduction in crop yields by 5–75% due to soil compaction. Strategies of managing compaction include soil and tillage management. This can be accomplished through the use of no till, controlled traffic system, crop rotations, mulching, incorporation of deep-rooted leguminous crops in rotation, and periodic chiseling. Alleviation of soil compaction can increase yields by 20–70%.

INTRODUCTION

Soil compaction or densification is the process of rearrangement of soil particles, leading to decrease in total pore space and pore size, and increase in bulk density. A large portion of soil air is forced out of the root zone, and the channels of greatest continuity and least resistance to air and water movement and root penetration are eliminated. Compaction below the depth of normal tillage operation is generally called subsoil compaction. Soil compaction may be tillage induced or traffic induced. The tillage-induced compaction, caused by primary tillage under less than optimum soil moisture conditions and by excessive secondary tillage, leads to densification of the plow layer. The traffic-induced compaction, caused by wheel traffic associated with farm operations, occurs in the subsoil. Compaction in the plow layer is largely related to the contact pressure of the tires, and that in the subsoil to the axle load. Subsoil compaction is not easily alleviated by self-correcting processes (i.e., freezing–thawing and wetting–drying cycles). The depth of subsoil compaction depends upon vehicle weight, axle load, and wheel arrangement or ground contact pressure. Compaction can exist for hundreds of years even when there is annual soil freezing and thawing. Soil compaction is expressed in terms of dry bulk density, relative bulk density, porosity, air permeability, hydraulic

conductivity, soil strength, cone resistance, plant height, infiltration rate, water retention, and crop yield.

Soil compaction is a global problem and is a severe factor in crop production. Most research on machinery-induced compaction has been carried out in humid and temperate regions. However, it is a problem everywhere and needs to be studied worldwide, as its evaluation requires local experimental data. In developed countries, this concern about soil compaction, both in the plow layer and in the subsoil, is closely related to the adverse effects on soil structure by the use of large and heavy farm equipment, increased specialization in crop production, increased traffic and tillage necessary for application and incorporation of fertilizers and other chemicals, and early seedbed preparation and planting of many crops when soils are often wet and susceptible to compaction. However, soil compaction problems in the tropics arise from the use of heavy machinery in the land clearing and in the subsequent crop production. Adverse impacts of densification on crop yields are further accentuated by intense tropical rains, the presence of naturally occurring compacted layers, and the widespread occurrence of soil horizons with large percentage of gravel and plinthite. Soil compaction and their consequences on crop production have not been much researched in tropics as in the temperate region. The magnitude of soil compaction depends on soil type, moisture content, type of

running gear, and tyre-inflation pressure. The objective of this entry is to describe consequences of soil compaction on crop yields, and discuss methods and strategies of alleviating and minimizing risks of soil compaction.

CONSEQUENCES OF SOIL COMPACTION

Soil compaction affects chemical, physical, and biological properties and processes, and influences long-term soil productivity as well as resource economy and environmental quality.^[1] Economic impact of soil compaction may be difficult to quantify because of the confounding effects of other interacting factors, e.g., antecedent soil conditions, soil management, cropping systems, weather conditions, and seasonality.^[2] Economic effects of soil compaction may be direct due to reduction in crop yield or indirect due to less effective use of resources or inputs.^[3,4] Environmental impacts of soil compaction are related to its effects on accelerated soil erosion, transport of sediments and sediment-borne agrochemicals, and possible emission of radioactively active gases into the atmosphere. Compaction-induced anaerobiosis may accentuate emission of methane, nitrous oxide, and nitrogen oxide gases from soil.^[5] Losses of nitrogen due to nitrification are usually more on compacted than uncompacted soils.

CROP RESPONSE TO SOIL COMPACTION

Soil compactability and its effects on crop growth depend on texture, cation exchange capacity, pH, soil organic matter content, weather conditions, and crop characteristics. A series of experiments conducted by a Working Group of International Soil Tillage Research Organization (ISTRO) showed that crop response to the traffic varied considerably between sites and years, and there was a persistent negative crop response to high axle-load traffic.^[6–11] Several experiments conducted in North America and Europe also showed yield reduction up to 50%.^[12] Under extreme conditions of subsoil compaction, yield reductions of maize were up to 70%^[13] (Table 1) and adverse effects persisted for several years.^[4,10] However, in some soils, the compaction effects on yield may be none or even positive.^[21,22] The effect of different axle loads on yields of silage maize in sandy soils at four locations in the Netherlands showed that heavy axle loads reduced yields by up to 38%.^[15] In Morocco, Oussible, Crookston, and Larson^[14] found that compacting a clay loam soil to a density of 1.52 Mg m⁻³ below 0.10 m depth reduced grain and straw yields of wheat by 12–23% and 9–20%, respectively. For a loam soil in Norway, Riley^[8] observed that grain yield of barley decreased from 5.7 Mg ha⁻¹ for uncompacted control to 5.2 Mg ha⁻¹ with four passes of 26 Mg axle load, and adverse effects persisted for 2–3 years.

Soil compaction can adversely impact the establishment, growth, and yields of crops in the tropics^[23] (Table 2). An intensive mechanized cultivation of wheat in large-scale

Table 1 Effect of soil compaction on crop yields in soils of temperate climate.

Location	Soil type	Crop	Decrease in yield (%)	References
Canada	Clayey	Maize	70	[13]
Finland	Mollicgley	Oat, wheat, and barley	1–4	[9]
Morocco	Clay loam	Wheat	23	[14]
Netherlands	Sandy	Maize silage	38	[15]
Spain	Loam	Seed cotton	28	[16]
Sweden	Loam	Wheat	11	[17]
United States	Clayey	Maize	24	[4]
United States	Clayey	Sorghum	39	[4]
United States	Clayey	Oat	31	[4]
United States	Silt loam	Barley	14	[7]
United States	Silt loam	Pea	28	[7]
United States	Silt loam	Maize	14	[18]
United States	Clay loam	Maize	30	[19]
United Kingdom	Sandy loam	Potato	36	[20]

irrigation projects can lead to the development of a dense tillage pan at about 0.15–0.20 m depths. The pan restricts root penetration resulting in reduced grain yields.^[31–33] Experiments conducted in Punjab, Pakistan^[27,28,34,35] showed that compacting a sandy clay loam soil to a density of 1.93 Mg m⁻³ below 0.15 m depth reduced grain yield of wheat by 12–38% and of sorghum fodder by 14–22%.

Table 2 Effect of soil compaction on crop yields in soils of tropic and subtropics.

Location	Soil type	Crop	Decrease in yield (%)	References
Argentina	Silty loam	Ryegrass/white clover	74	[11]
Australia	Alfisol	Cereal	25	[24]
Australia	Alfisol	Cereal	30	[25]
Brazil	Latosol	Sorghum	50	[26]
Nigeria	Alfisol	Maize	62	[23]
Nigeria	Alfisol	Cassava	38	[23]
Pakistan	Sandy clay loam	Wheat	38	[27]
Pakistan	Sandy clay loam	Sorghum fodder	22	[27]
Pakistan	Clay loam	Maize fodder	46	[28]
Pakistan	Silt loam	Wheat grain	5–48	[29]
Tanzania	Sandy loam	Bean	50	[30]

Ishaq^[36] reported that subsoil compaction effects persisted for 3–4 years and decreased yields of wheat by 12–18% and cotton by 7%.

MANAGEMENT OF SOIL COMPACTION

Minimizing Soil Compaction Hazards

One of the most direct methods for avoiding compaction is to adopt the strategy of controlled traffic. A controlled traffic system restricts wheel traffic to specific lanes. As a result, most of the soil area remains uncompacted compared to the uncontrolled traffic pattern. Other possible ways to manage soil strength are the use of no till, alley cropping, crop rotation or addition of manure or other organic materials, and promotion of earthworm activity.^[37]

Amelioration of Compacted Soils

Mechanical and/or biological measures can be used for alleviating soil compaction.

Mechanical loosening of compacted soil

Mechanical loosening of soil is necessary to alleviate soil compaction. The negative effects of compaction on soil

productivity require decades for amelioration.^[1–3] A study conducted in Australia by Hamza and Anderson^[38] showed that deep ripping to 30 cm depth in combination with tine spacing of 30 cm resulted in increased stored soil water and infiltration rates, decreased soil bulk density and soil strength, and increased grain yields of wheat and barley on both sandy and clayey soils. On the other hand, in Denmark, Olesen and Munkholm^[39] reported that subsoil loosening in crop rotation for organic farming eliminated plough pan with mixed effects on the yields of wheat. In Orange Free State, South Africa, Bennie and Botha^[40] reported that deep ripping of a continuously plowed soil increased maize and wheat yields. The beneficial effects of mechanical loosening are often short lived, and soils may resettle to a high density due to subsequent excessive traffic and other management.^[41] Periodic chiseling to a depth of about 50 cm in the row zone followed by sowing of a deep-rooted leguminous cover can avoid resetting of the loosened soil and improve infiltration and root development in the subsoil.^[37,42] Subsoiling in the row zone can be effective to alleviate the effect of dense layer (Table 3).^[49,56–59]

Biological loosening of compacted soil

Soil compaction can be alleviated by growing deep-rooted annuals such as mucuna or pigeon^[60] pea, or deep-rooted

Table 3 Amelioration of compacted soils through mechanical loosening.

Location	Soil type	Treatment	Crop	Increase in yield (%)	References
Africa	Aeolian sandy	Deep ripping	Maize	30	[40]
Africa	Aeolian sandy	Deep ripping	Wheat	19	[40]
Australia	Alfisol	Deep plowing	Soybean	9	[43]
Morocco	Clay loam	Subsoiling	Wheat	48	[44]
Newzealand	Silty clay	Subsoiling	Oat fodder	18	[45]
Nigeria	Alfisol	Chiseling	Maize	9	[46]
Nigeria	Alfisol	Paraplough	Maize	30	[46]
Pakistan	Silt loam	Deep plowing	Wheat	21	[47]
Sudan	Sandy clay	Chiseling	Sorghum	72	[48]
United States	Sandy loam	Subsoiling	Maize	27	[49]
United States	Loamy sand	Subsoiling	Maize	17	[49]
United States	Sandy clay loam	Subsoiling	Maize	5	[49]
United States	Loamy	Chiseling	Maize	9	[50]
United States	Loamy sand	Subsoiling	Maize	36	[51]
United States	Loamy sand	Subsoiling	Stover	40	[51]
United States	Clay loam	Trenching	Wheat	21	[52]
United States	Clay loam	Trenching	Maize	64–180	[53]
United States	Loamy sand	Subsoiling	Maize	25	[54]
United States	Silt loam	Subsoiling	Seed cotton	17	[55]
United States	Sandy loam	Subsoiling	Seed cotton	10	[55]
United States	Clay loam	Subsoiling	Seed cotton	19	[55]

Table 4 Amelioration of compacted soils through biological loosening.

Location	Crop	Beneficial effects	References
Africa	Cover crops	Increased aeration, infiltration rate, hydraulic conductivity, and decreased bulk density	[61–64]
Africa	Cover crops	Increased organic carbon, total N, pH, and CEC	[62,64]
Brazil	Cover crops	Increased infiltration rate	[65,66]
		Friable soil structure	
Canada	Cover crops (<i>Secalecereale</i>) (<i>Loliummultiflorum</i>)	Improved aggregation	[67]
		Better soil structure	
Iran	Use of arbuscular mycorrhiza	Reduced the stressful effects of soil compaction on maize growth and enhanced the nutrient uptake	[68,69]
Nigeria	Deep-rooted perennials (<i>Leucaenaleucocephala</i>)	Improved soil physical properties	[60]
Nigeria	Deep-rooted annuals (<i>Mucunautilis</i>)	Improved soil physical properties	[70]
Nigeria	Leguminous crops (<i>Acacia leptocarpa</i>) (<i>Acacia auriculiformis</i>) (<i>Senna siamea</i>)	Ameliorated soil compaction and soil strength	[71]
		Decreased soil bulk density	

perennials such as *Leucaenaleucocephala* (Table 4).^[61,62] Improvement in soil physical properties by legume covers is attributed to increase in the transmission or micropores (>5 µm), created by the taproot systems of legumes.^[60,63] Roots of alfalfa, sweet clover, and guar can penetrate hard pans, but improvement in yield is not clear.^[72] The roots of bahia grass can penetrate soil layers, whereas roots of cotton cannot, and the yield of cotton grown following bahia grass is often improved. Lal, Wilson, and Okigbo^[63] recommended growing *Stylosanthesguiansis* on Alfsoils in Nigeria. Growing cover crops can increase biological activity of earthworms and thus improve soil physical conditions or tilth, through the addition of organic matter and the formation of root channels.^[65,66,73,74] The beneficial effects of legume covers are greater in soils that have been subjected to low levels of compaction.^[70]

CONCLUSION

Use of large and heavy farm equipment increases risks of reduction in crop yields by 5–75% due to soil compaction. Effects of compaction on plant growth and yield depend on soil, crop, and climatic conditions. Compaction increases soil strength that adversely affects root growth and reduces crop yields. Compaction in the plow layer is related to contact pressure of the tire, whereas in subsoil, it is related to total axle load. Depth of compaction increases as both the soil moisture content and axle load increase. Strategies of managing compaction include soil and tillage management. This can be accomplished through the use of no till, controlled traffic system, crop rotations, mulching, incorporation of deep-rooted leguminous crops in rotation, and periodic chiseling. Alleviation of soil compaction can increase yields by 20–70%.

REFERENCES

1. Batey, T. Soil compaction and soil management—A review. *Soil Use Manage.* **2009**, 25, 335–345.
2. Farrakh, M.F.; Bourrié, G.; Trolard, F. Soil compaction impact and modeling: A review. *Agron. Sustain. Dev.* **2013**, 33, 291–309.
3. Hoefler, G.; Hartge, K.H. Subsoil compaction: Cause, impact, detection, and prevention. In *Soil Engineering, Soil Biology 20*; Dedousis, A.P., Bartzanas, T., Eds.; Springer-Verlag: Berlin Heidelberg, 2010; 121–145.
4. Lal, R. Axle load and tillage effects on crop yields on a mollicochraqualf in northwest Ohio. *Soil Till. Res.* **1996**, 37, 143–160.
5. Lal, R.; Fausey, N.R.; Eckert, D.J. Land use and soil management effects on greenhouse gas emissions from two soils in Ohio. In *Soil Management and Greenhouse Effect*; Lal, R., Kimble, J., Levine, E., Steward, B.A., Eds.; Lewis: Boca Raton, 1995; 41–60.
6. Håkansson, I.; Reeder, R.C. Subsoil compaction by vehicles with high axle load—Extent, persistence and crop response. *Soil Till. Res.* **1994**, 29, 277–304.
7. Hammel, J.E. Effect of high axle load traffic on subsoil physical properties and crop yields in the Pacific Northwest, USA. *Soil Till. Res.* **1994**, 29, 195–203.
8. Riley, H. The effect of traffic at high axle load on crop yields on a loam soil in Norway. *Soil Till. Res.* **1994**, 29, 211–214.
9. Alakukku, L.; Elonen, P. Finish experiment on subsoil compaction by vehicles with high axle load. *Soil Till. Res.* **1994**, 29, 151–155.
10. Etana, A.; Håkansson, I. Swedish experiments on the persistence of subsoil compaction caused by vehicles with high axle load. *Soil Till. Res.* **1994**, 29, 167–172.
11. Jorajuria, D.; Draghi, L.; Aragon, A. The effect of vehicle weight on the distribution of compaction with depth and the yield of *Lolium/trifolium* grassland. *Soil Till. Res.* **1997**, 41, 1–12.

12. Raghavan, G.S.V.; Alvo, P.; McKyes, E. Soil compaction in agriculture: A review toward managing the problem. *Adv. Soil Sci.* **1990**, *11*, 1–36.
13. Gameda, S.G.; Raghavan, S.V.; McKyes, E.; Watson, A.K.; Mehuys, G. Long-term effects of a single incidence of high axle load compaction on a clay soil in Quebec. *Soil Till. Res.* **1994**, *29*, 173–177.
14. Oussible, M.; Crookston, P.K.; Larson, W.E. Subsurface compaction reduces the root and shoot growth and grain yield of wheat. *Agron. J.* **1992**, *84*, 34–38.
15. Alblas, J.; Wanink, F.; van der Akker, J.; van der Werf, H.M. G. Impact of traffic induced compaction of sandy soils on the yields of silage maize in the Netherlands. *Soil Till. Res.* **1994**, *29*, 157–165.
16. Coelho, M.B.; Mateos, L.; Villalobos, F.J. Influence of a compacted loam subsoil layer on growth and yield of irrigated cotton in southern Spain. *Soil Till. Res.* **2000**, *57*, 129–142.
17. Arvidsson, J.; Hakansson, I. Do effects of soil compaction persist after ploughing? Results from 21 long-term field experiments in Sweden. *Soil Till. Res.* **1996**, *39*, 175–197.
18. Lal, R.; Ahmadi, M. Axle load and tillage effects on crop yield for two soils in central Ohio. *Soil Till. Res.* **2000**, *54*, 111–119.
19. Voorhees, W.B.; Johson, J.F.; Randall, G.W.; Nelson, W.W. Corn growth and yield as affected by surface and subsoil compaction. *Agron. J.* **1989**, *81*, 294–303.
20. Stalham, M.A.; Allen, E.J.; Rosenfeld, A.B.; Herry, F.X. Effects of soil compaction in potato (*Solanum tuberosum*) crops. *J. Agric. Sci.* **2007**, *145*, 295–312.
21. Lal, R.; Tanaka, H. Simulated harvest traffic effects on corn, oat, and soybean yields in western Ohio. *Soil Till. Res.* **1992**, *24*, 65–78.
22. Steward, G.A.; Vyn, T.J. Influence of high axle load and tillage systems on soil properties and grain corn yield. *Soil Till. Res.* **1994**, *29*, 229–235.
23. Kayombo, B.; Lal, R. Effect of soil compaction by rolling on soil structure and development of maize in no till and disking system on a tropical alfisol. *Soil Till. Res.* **1986**, *7*, 117–134.
24. Dalal, R.C.; Mayer, R.J. Long-term trends in fertility of soils under continuous cultivation and cereal cropping in southern Queensland: 1. Overall changes in soil properties and trends in winter cereal yields. *Aust. J. Soil Sci.* **1986**, *24*, 265–279.
25. Stanely, J.; Hunter, H.M.; Thomas, G.A.; Blight, G.W.; Webb, A.A. Tillage and crop residue management affect vertisol properties and grain sorghum growth over seven years in the semi-arid subtropics: 2. Changes in soil properties. *Soil Till. Res.* **1990**, *18*, 367–388.
26. Ortolani, A.F.; Coam, O.; Salles, H.C. Influence of soil compaction on the development of soybean (*Glycine max* L). *Eng. Agric.* **1982**, *6*, 35–42.
27. Ishaq, M.; Hassan, A.; Saeed, M.; Ibrahim, M.; Lal, R. Subsoil compaction effects on crops in Punjab, Pakistan: 1. Soil physical properties and crop yield. *Soil Till. Res.* **2001**, *59*, 57–65.
28. Saeed, M.; Rashid, M.; Mustafa, G. Compaction effect on the physical properties of soil and crop (maize fodder) yield. *Pak. J. Soil Sci.* **1993**, *8*, 19–22.
29. Ahmad, N.; Hassan, F.U.; Belford, R.K. Effect of soil compaction in the sub-humid cropping environment in Pakistan on uptake of NPK and grain yield in wheat (*Triticum aestivum*): I. Compaction. *Field Crop. Res.* **2009**, *110*, 54–60.
30. Hatibu, N.; Kayombo, B.; Simalenga, T.E. Effects of tillage methods on soil physical conditions and yield of beans in a tropical sandy loam soil. In *Proceedings of the 12th Conference of International Soil Tillage Research Organization*, IITA Annual Report, Ibadan, Nigeria, July 8–12, 1991; 129–140.
31. Maurya, P.R. Performance of zero tillage in wheat and maize production under different soil and climatic conditions in Nigeria. In *Proceedings of the 11th Conference of International Soil Tillage Research Organization*, Edinburgh, 1988; Vol. 2, 769–774.
32. Gill, K.S.; Aulakh, B.S. Wheat yield and soil bulk density response to some tillage systems on an oxisol soil. *Soil Till. Res.* **1990**, *18*, 37–45.
33. Fawcett, R.G.; Maynard, F.R.; Pederson, N.R.; Hannay, J.N. The effect of traffic and other factors on wheat yields in South Australia. In *Proceedings of the 11th Conference of International Soil Tillage Research Organization*, Edinburgh, 1988; Vol. 1, 257–262.
34. Sial, J.K.; Sheikh, G.S.; Afzal, M. Emergence of wheat seedlings as affected by soil compaction. *Pak. J. Agric. Sci.* **1987**, *24*, 225–230.
35. Ahmad, M.; Sabir, M.S.; Aslam, M. Effect of soil compaction and organic matter on growth of wheat crop. *Pak. J. Soil Sci.* **1992**, *71*, 39–41.
36. Ishaq, M.; Ibrahim, M.; Lal, R. Persistence of subsoil compaction effects on soil properties and growth of wheat and cotton in Pakistan. *Exp. Agr.* **2003**, *39*, 341–348.
37. Hamza, M.A.; Anderson, W.K. Soil compaction in cropping systems: A review of nature, causes and possible solutions. *Soil Till. Res.* **2005**, *82*, 121–145.
38. Hamza, M.A.; Anderson, W.K. Combination of ripping depth and tine spacing for compacted sandy and clayey soils. *Soil Till. Res.* **2008**, *99*, 213–220.
39. Olesen, J.E.; Munkholm, L.J. Subsoil loosening in a crop rotation for organic farming eliminated plough pan with mixed effects on crop yield. *Soil Till. Res.* **2007**, *94*, 376–385.
40. Bennie, A.T.P.; Botha, F.J.P. Effect of deep tillage and controlled traffic on root growth, water-use efficiency, and yield of irrigated maize and wheat. *Soil Till. Res.* **1986**, *7*, 85–95.
41. Orellana, M.; Barber, R.G.; Diaz, O. Effects of deep tillage and fertilization on the population, growth and yield of soya during an exceptionally wet season on a compacted sandy loam, Santa Cruz, Bolivia. *Soil Till. Res.* **1990**, *17*, 47–61.
42. Lal, R. A soil suitability guide for different tillage systems in the tropics. *Soil Till. Res.* **1985**, *5*, 179–196.
43. Willis, T.M.; Hall, D.J.M.; McKenzie, D.C.; Barchia, I. Soybean yield as affected by crop rotations, deep tillage and irrigation layout on a hard-setting alfisol. *Soil Till. Res.* **1997**, *44*, 151–164.
44. Oussible, M.; Crookston, R.K. Effect of subsoiling a compacted clay loam soil on growth, yield and yield components of wheat. *Agron. J.* **1987**, *79*, 882–886.
45. Sojka, R.E.; Horn, D.J.; Ross, C.W.; Baker, C.J. Subsoiling and surface tillage effects on soil physical properties and forage oat stand and yield. *Soil Till. Res.* **1997**, *40*, 125–144.
46. Garman, C.; Juo, A.S.R. *Long-Term Tillage Studies*. IITA Annual Report; IITA: Ibadan, 1983; 188–189.

47. Ahmad, N.; Hassan, F.U.; Belford, R.K. Effect of soil compaction in the sub-humid cropping environment in Pakistan on uptake of NPK and grain yield in wheat (*Triticum aestivum*): II. Alleviation. *Field Crop. Res.* **2009**, *110*, 61–68.
48. Omer, M.A.; Elamin, E.M. Effect of tillage and contour diking on sorghum establishment and yield on sandy clay soil in Sudan. *Soil Till. Res.* **1997**, *43*, 231–242.
49. Vepraskas, M.J.; Waggoner, M.G. Corn root distribution and yield response to subsoiling for paleudults having different aggregate sizes. *Soil Sci. Soc. Am. J.* **1990**, *54*, 849–854.
50. Diaz-Zorita, M. Effect of deep tillage and nitrogen fertilization interactions on dry land corn (*Zea mays* L.) productivity. *Soil Till. Res.* **2000**, *54*, 11–19.
51. Cassel, D.K.; Edwards, E.C. Effect of subsoiling and irrigation on corn production. *Soil Sci. Soc. Am. J.* **1985**, *49*, 996–1001.
52. Eck, H.V. Profile modification and irrigation effects on yield and water use of wheat. *Soil Sci. Soc. Am. J.* **1986**, *50*, 724–729.
53. Eck, H.V.; Winter, S.R. Soil profile modification effects on corn and sugar beet grown with limited water. *Soil Sci. Soc. Am. J.* **1992**, *56*, 1298–1304.
54. Ewing, R.P.; Waggoner, M.G.; Denton, H.P. Tillage and cover crop management effects on soil water and corn yield. *Soil Sci. Soc. Am. J.* **1991**, *55*, 1081–1085.
55. Mullins, G.L.; Burmester, C.H.; Reeves, D.W. Cotton response to in-row subsoiling and potassium fertilizer placement in Alabama. *Soil Till. Res.* **1997**, *40*, 145–154.
56. Kamprath, E.J.; Cassel, D.K.; Gross, H.D.; Dibb, D.W. Tillage effects on biomass production and moisture utilization by soybeans on coastal plain soils. *Agron. J.* **1979**, *71*, 1001–1005.
57. Ley, G.J.; Mullins, C.E.; Lal, R. Hard-setting behaviour of some structurally weak tropical soils. *Soil Till. Res.* **1989**, *13*, 365–381.
58. Radford, B.J.; Yule, D.F.; McGarry, D.; Playford, C. Amelioration of soil compaction can take 5 years on a vertisol under no till in the semi-arid subtropics. *Soil Till. Res.* **2007**, *97*, 249–255.
59. Wong, M.T.F.; Asseng, S. Yield and environmental benefits of ameliorating subsoil constraints under variable rainfall in a Mediterranean environment. *Plant Soil.* **2007**, *297*, 29–42.
60. Hulugalle, N.R.; Lal, R. Root growth of maize in a compacted gravelly tropical alfisol as affected by rotation with a woody perennial. *Field Crop. Res.* **1986**, *13*, 33–44.
61. Lal, R. Conservation tillage for sustainable agriculture: Tropics versus temperate environments. *Adv. Agron.* **1989**, *42*, 85–197.
62. Kang, B.T.; Reynolds, L.; Atta-Krah, A.N. Alley farming. *Adv. Agron.* **1990**, *43*, 315–359.
63. Lal, R.; Wilson, G.F.; Okigbo, B.N. Changes in properties of an alfisol produced by various crop covers. *Soil Sci.* **1979**, *127*, 337–382.
64. Lal, R.; Wilson, G.F.; Okigbo, B.N. No-till farming after various grasses and leguminous cover crops in tropical alfisol: 1. Crop performance. *Field Crop. Res.* **1978**, *1*, 71–84.
65. Kamper, B.; Derpsch, R. Results of studies made in 1978 and 1979 to control erosion by cover crops and no-tillage techniques in Parana, Brazil. *Soil Till. Res.* **1981**, *1*, 253–267.
66. Derpsch, R.; Sidiras, N.; Roth, C.H. Results of studies made in 1977 to 1984 to control erosion by cover crops and no-tillage techniques in Parana, Brazil. *Soil Till. Res.* **1986**, *8*, 253–263.
67. Hermawan, B.; Bomke, A.A. Effects of winter cover crops and successive spring tillage on soil aggregation. *Soil Till. Res.* **1997**, *44*, 109–120.
68. Miransari, M.; Bahrami, H.A.; Rejali, F.; Malakouti, M.J.; Torabi, H. Using arbuscular mycorrhiza to reduce the stressful effects of soil compaction on corn (*Zea mays* L.) growth. *Soil Biol. Biochem.* **2007**, *39*, 2014–2026.
69. Miransari, M.; Bahrami, H.A.; Rejali, F.; Malakouti, M.J. Effect of soil compaction and arbuscular mycorrhiza on corn (*Zea mays* L.) nutrient uptake. *Soil Till. Res.* **2009**, *103*, 282–290.
70. Hulugalle, N.R.; Lal, R.; Ter Kuile, C.H.H. Amelioration of soil physical properties by mucuna after mechanized land clearing of a tropical rain forest. *Soil Sci.* **1986**, *141*, 219–224.
71. Salako, F.K.; Hauser, S.; Babalola, O.; Tian, G. Improvement of the physical fertility of a degraded alfisol with planted and natural fallows under humid tropical conditions. *Soil Use Manage.* **2001**, *17*, 41–47.
72. Bowen, H.D. Alleviating mechanical impedance. In *Modifying the Root Environment to Reduce Crop Stress*; Monograph 4; Arkin, A.F., Taylor, H.M., Eds.; ASAE: St. Joseph, 1981; 21–57.
73. Adem, H.H.; Tisdall, J.M. Management tillage and crop residues for double cropping in fragile soils of southeastern Australia. *Soil Till. Res.* **1984**, *4*, 577–589.
74. Mason, W.K.; Small, D.R.; Pritchard, K.E. Effects of irrigation and soil management for fodder crops on root zone conditions in a red-brown earth. *Aust. J. Soil Res.* **1984**, *22*, 207–218.

Croplands

B.A. Stewart

*Department of Agricultural Sciences, West Texas A&M University,
Canyon, Texas, U.S.A.*

Abstract

Production of food and fiber depends is focused on croplands. With the exception of land devoted to perennial crops, cropland is commonly referred to as arable land. Worldwide, cropland increased from about 365 Mha in 1700 to around 1.2 Bha in 1950. By 1990, it reached 1.4 Bha but there has been little or no increase since. Even though cropland expansion since 1950 has only been about 15%, production of cereals increased 2.9-fold and they occupy about 50% of the cropland. Therefore, most of the increased production between 1950 and today has resulted from higher yields rather than from expanded croplands that were the dominant factor prior to 1950. Irrigated area more than doubled since 1961, and consumption of synthetic nitrogen fertilizer increased more than tenfold. With the world population increasing and becoming more prosperous, projected food and fiber needs are expected to increase by 70% by 2050. Although there is potential for some increase in cropland area, particularly Africa and Latin America, increased production will remain focused on sustainable intensification of existing cropland.

INTRODUCTION

Croplands are used to produce food, fiber, and other products. Depending upon one's definition, the term "croplands" can include harvested croplands, land with perennial crops such as orchards and vineyards, pasture, land with failed crops, and areas idled in a particular year either for production reasons or through government programs. Cropland is often considered synonymous with arable land, but arable land statistics may exclude cropland devoted to permanent crops such as orchards. Statistics used in this report were obtained from Food and Agriculture Organization (FAO) data reported as arable land (FAOSTAT). Cropland devoted to pasture is intensively managed land that supports forage such as improved grasses or a mixture of grasses and legumes. Pasture land is very different from rangeland, which infers to unplowed areas where native grasses, forbs, shrubs, and trees are used for forage. Rangeland is included in agricultural land but not in cropland.

CROPLAND IN THE UNITED STATES

Cropland occupies about 155.1 Mha or 17% of the nation's land area (Table 1). This is a decrease from 19.7% in 1961. Government programs have often had the most significant effect on how much cropland is actively used and, in some cases, how it is used. Since the mid-1950's, area of cropland idled by U.S. Government programs has ranged from 0% to a high of nearly 20% with a high of 31 Mha in 1988. As part of the 1985 Food Security Act, the U.S. Congress established the Conservation Reserve Program (CRP) that

pays producers to take highly erodible land out of production. These areas are placed under 10- to 15-year leases and seeded to grass or trees. By 1993, almost 14 Mha of cropland had been taken out of production. Although some land enrolled in the program has been put back into production, other land has been placed into the program and about 12 Mha was in the program in 2012. Prior to the CRP, government programs that idled cropland were explicitly designed to control or stabilize the amount of crops produced. Urbanization has also significantly reduced cropland area and losses continue at the rate of about 400,000 ha annually.^[1]

The amount of crops and livestock produced in the United States more than doubled between 1948 and 1997,^[2] even though the amount of cropland was declining slightly (Table 1). The average agricultural productivity during that period increased 1.9% per year. The rate of growth has been somewhat less since 1997 but still increasing.^[3] The productivity increase was the result of farm inputs such as fertilizers, improved breeding, pesticides, and other capital investment. Irrigation also played a major role. Only about 15% of the cropland is irrigated but accounts for approximately 40% of the total value of crops produced.^[4] Irrigated areas dominate the production of several major crops, including rice with 100% irrigation, orchards 79%, and vegetables 66%.^[5]

Irrigation in the United States increased by more than 40% between 1961 and 1980 but has increased little since. It has actually decreased slightly since 1990 (Table 1). Perhaps more striking is the shift in irrigated area from western states to eastern states. Between 1982 and 2012, irrigated area in the western 17 states decreased from 84%

Table 1 Land area and use in the United States (FAOSTAT data).

	1961	1970	1980	1990	2000	2012
	Millions of hectares					
Land area	915.9	915.9	915.9	915.9	916.2	914.7
Agricultural land	447.5	434.4	428.2	426.9	414.4	408.7
Arable land	180.6	188.7	188.8	185.7	175.4	155.1
Irrigated land	19.0	21.7	25.0	26.9	27.0	26.4

of the total irrigated area to 72%. Irrigated area in the 17 western states actually decreased from 16.7 to 16.4 Mha.^[6]

CROPLAND IN THE WORLD

According to the FAO of the United Nations, there are about 1.4 Bha of arable land in the world (Table 2). It was

earlier estimated that the global extent of cropland increased from about 365 Mha in 1700 to around 1.2 Bha in 1950.^[7] Since 1961, however, the increase in arable land area has been relatively. From 1961 to 1990, arable land area in the world increased less than 10%, and there has been no increase since (Table 2). During the same time period, cereal production increased 2.9-fold. More than 50% of the arable land is dedicated to the production of cereals. Wheat (*Triticum aestivum*), maize (*Zea mays*), and rice (*Oryza sativa*) are the dominant cereals and occupied 15.5%, 12.8%, and 11.6% of the arable land in 2012. Cereals are of extreme importance because humans get about 48% of their calories from grains.^[8] In 2013, worldwide production of cereals totaled 2780 Mt of which 716 Mt was wheat, 741 Mt was rice, and 1018 Mt was maize.^[9] Historically, wheat was produced in the largest quantity followed by rice and then maize. Maize has become the dominant cereal grain because increasing prosperity in developing countries is resulting in changing diets that

Table 2 Arable land, irrigated land, cereal production, and population of the world and continents from 1961 to 2012 (FAOSTAT data).

	1961	1970	1980	1990	2000	2012
Arable land (million hectares)						
World	1,277	1,331	1,339	1,400	1,381	1,396
Africa	152	169	163	180	202	237
Americas	307	338	353	361	362	369
Asia	411	419	423	459	498	467
Europe	373	359	353	349	288	275
Oceania	33	45	47	51	49	33
Irrigated land (million hectares)						
World	161	184	221	258	288	324
Africa	7.4	8.4	9.3	11.1	13.2	14.3
Americas	28	32	40	44	47	52
Asia	106	119	137	160	198	229
Europe	19	23	33	40	27	26
Oceania	1.1	1.6	1.7	2.1	2.8	3.4
Cereals (million tonnes)						
World	877	1,193	1,550	1,952	2,061	2,563
Africa	46	60	73	93	112	174
Americas	228	287	340	468	532	611
Asia	330	481	632	873	996	1,316
Europe	264	351	428	494	351	421
Oceania	10	1.4	1.7	2.4	3.5	4.5
Population (billions)						
World	3.08	3.69	4.45	5.32	6.13	7.08
Africa	0.29	0.37	0.48	0.62	0.81	1.08
Americas	0.43	0.52	0.62	0.73	0.84	0.96
Asia	1.69	2.08	2.58	3.15	3.72	4.25
Europe	0.65	0.70	0.75	0.79	0.73	0.74
Oceania	0.02	0.02	0.02	0.03	0.03	0.04

include more animal-based products. About 66% of the global maize production is used for animal feed. The figure is higher for developed countries, which in 2000 used about 76% of the maize for animal feed, and lower (about 56%) for developing countries. Rice is consumed almost exclusively by people and Asia contains a staggering 89% of the world's harvested rice area.^[10]

CROPPING INTENSITY

A simple measure of how intensive cropland is used is the cropping intensity index. This is defined as the annual harvested area expressed as a proportion of the total cropland area (land in use plus fallow). For example, some cropping systems require a portion of the production area to be placed in fallow every year resulting in a cropping intensity index less than 1. In comparison, some irrigated areas can produce up to three crops per year from the same physical area and have a cropping intensity of 3. An extensive study of the increasing global crop harvest frequencies was conducted.^[11] It is generally assumed that crop production is increased either by expanding the area of standing cropland or by increasing intensification on existing croplands through the use of increased fertilizer, irrigation, mechanization, and improved seed varieties. The study^[11] focused on a third way of using the existing standing cropland, where appropriate, more frequently each year through multiple cropping, leaving less land fallow, and having fewer crop failures. The analysis showed that between 1961 and 2007, this third way contributed to 9% increase in global crop production, and the ratio of annually harvested land to standing cropland increased from 0.78 to 0.89 between 1961 and 2011. The study concluded that the 28% increase in global crop production between 1985 and 2005 was due to a combination of increasing harvested area about 7% and increasing average yields about 20%. In summary, the study attributed roughly a quarter of crop production increases to changing harvested area and the remaining three quarters to increasing yields.

IRRIGATION AND FERTILIZERS

As already stated, most of the increased crop production that has occurred since 1961 has been due to increasing crop yields. Although many factors contributed to higher yield, increasing irrigation area and fertilizer use have been of extreme importance. Irrigation area doubled worldwide between 1961 and 2012 and even more than doubled in Asia (Table 2). This removed water as the major constraint in many areas and made chemical fertilizers particularly effective. Clearly, nitrogen (N) and phosphorus (P) fertilizer uses became dominant factors in increasing cereal production, and China and India, the two most populous nations, were able to produce grain at rates faster than the

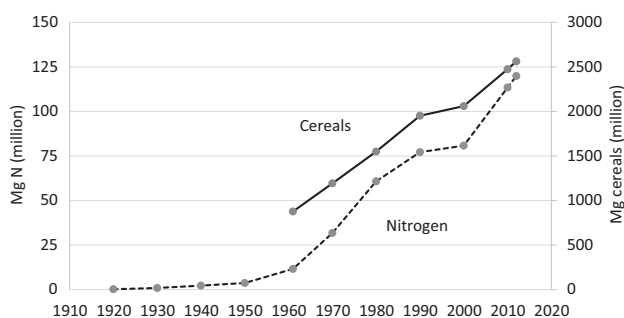


Fig. 1 Worldwide use of synthetic N fertilizer and production of cereal grains.

Source: Data from Smil.^[12] ©1999 and FAOSTAT.

population growth rate (Table 2). In 1961, these countries used less than 8% of the N fertilizer and less than 2% of the P. In 2012, they used 52% of the N fertilizer and 50% of the P fertilizer. China alone used 38% of the N and 36% of the P fertilizers.^[9] Synthetic N fertilizer production by the Haber–Bosch process became readily available following World War II that ended in 1945, and N fertilizer coupled with expanding irrigation area dramatically increased grain production as shown in Fig. 1. During the period from 1961 to 2012, cereal production increased threefold worldwide but in Asia where 60% of the population resides, cereal production increased fourfold (Table 2). Expansion of irrigated area is becoming increasingly difficult and costly so enhanced intensification of production on existing areas will be a primary focus.

SOIL DEGRADATION

With the focus on increasing intensification of croplands, there must also be enhanced attention on guarding against soil degradation. A 2010 report stated that of the 77% (837 Mha out of 1094 Mha) of the world's cropland area that is affected by water erosion, 83% (457 Mha out of 548 Mha) of the land prone to wind erosion, 97% (132 Mha out of 136 Mha) of that subject to nutrient mining, 94% (72 Mha out of 77 Mha) affected by salinization, and 83% (5 Mha out of 6 Mha) of that prone to acidification, occur in developing countries.^[13] These are also the regions where 99% of the projected increase in world population is expected to occur. Furthermore, farming communities in developing countries consist of resource-poor and small-size (<2 ha) landholders who use extractive farming practices involving litter or on off-farm input. Maintaining the amount and quality of cropland will require the use of sustainable management systems such as conservation agriculture and restoration practices to restore degraded lands. The principles of conservation agriculture are direct planting of seeds, permanent soil cover by crop residues and cover crops, and crop diversity.

CROPLANDS EXPANSION

The area of arable land increased slowly between 1961 and 1990 but has shown little change since 1990 (Table 2). Although there is some potential for expanding cropland, this is partially offset by losses due to urbanization and soil degradation. At the same time, the world population is expected to reach 9.1 billion by 2050 and food demand expected to increase by 70% more than in 2010 because of increasing prosperity in developing countries resulting in changing diets that include more animal products.^[9] The FAO states that despite the fact that 90% of the growth in crop production is projected to come from higher yields and increased cropping intensity, arable land will have to expand by around 120 Mha in developing countries, mainly in sub-Saharan Africa and Latin America. Arable land in use in developed countries is expected to decline by some 50 Mha, although this could be changed by the demand for biofuels. Sustainable cropping systems that not only control soil and water erosion but also improve soil quality and protect the environment must be accepted and practiced worldwide.

REFERENCES

1. Environmental Protection Agency. *Land Use Overview, Ag 101*; Environmental Protection Agency: Washington, DC. Available at <http://www.epa.gov/agriculture/ag101/landuse.html> (accessed April 25, 2016).
2. The Heinz Center. *Designing a Report on the State of the Nation's Ecosystems: Selected Measurements for Croplands, Forests and Coasts and Oceans*; The H. John Heinz III Center: Washington, DC, 1997.
3. USDA Economic Research Service. *Agricultural Productivity in the U.S.*; USDA Economic Research Service: Washington, DC. Available at <http://www.ers.usda.gov/data-products/agricultural-productivity-in-the-us.aspx> (accessed April 25, 2016).
4. USDA. *Agricultural Resources and Environmental Indicators, 1996–1997*, Agricultural Handbook Number 712; Economic Research Service, U.S. Department of Agriculture: Washington, DC, 1997.
5. U.S. Agricultural Census. U.S. Department of Commerce: Washington, DC.
6. Walton, B. *U.S. Irrigation Pushed Eastward by Drought and Financial Risks*. Circle of Blue, 2014. Available at <http://www.circleofblue.org/waternews/2014/world/u-s-irrigation-pushed-eastward-drought-financial-risks/> (accessed April 25, 2016).
7. Turner, B.I., II; Clark, W.C.; Kates, R.W.; Richards, J.F.; Mathews, J.T.; Meyer, W.B., Eds. *The Earth as Transformed by Human Action: Global and Regional Change in the Biosphere Over the Past 300 Years*; Cambridge University Press: Cambridge, 1990.
8. Worldwatch Institute. *Grain Harvest Sets Record, but Supplies Still Tight*, 2015. Available at <http://www.worldwatch.org/node/5539> (accessed April 25, 2016).
9. FAOSTAT. *FAO Statistical Database*; Food and Agriculture Organization of the United Nations: Rome. Available at <http://faostat.fao.org/> (accessed April 25, 2016).
10. Wood, S.; Sebastian, K.; Scherr, S.J. *Pilot Analysis of Global Ecosystems*; World Resources Institute: Washington, DC.
11. Ray, D.K.; Foley, J.A. Increasing global crop harvest frequency: Recent trends and future directions. *Environ. Res. Lett.* **2013**, *8* (4), 04404. doi:10.1088/1748-9326/8/4/044041.
12. Smil, V. Detonator of the population explosion. *Nature* **1999**, *400*, 415.
13. Lal, R.; Stewart, B.A. *Introduction: Food Security and Soil Quality, Advances Soil Science*; CRC Press: Boca Raton, 2010; 1–3.

Crops: Flooding Tolerance

Tara T. VanToai

Soil Drainage Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Columbus, Ohio, U.S.A.

Getachew Boru

Soil Drainage Research Unit and Department of Horticulture and Crop Science, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Columbus, Ohio, U.S.A.

Jianhuan Zhang

Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Periodic flooding during the growing season adversely affects crop growth and productivity, with the exception of flooded rice, in many areas of the United States and the rest of the world. Tolerance of anoxia and hypoxia has been used synonymously with the tolerance of flooding stress. Tolerance of field flooding appears to be much more complex than tolerance of artificially induced hypoxia and anoxia. Contrary to the injury seen in flooded fields, soybeans can thrive in stagnant water. Soybean, therefore, is much more tolerant to excessive water and a lack of oxygen than previously expected. The reasons underlying the dramatic differences between responses to flooding in the greenhouse and flooding in the field are not known.

INTRODUCTION

Periodic flooding during the growing season adversely affects crop growth and productivity, with the exception of flooded rice, in many areas of the United States and the rest of the world. Soil can become flooded when it is poorly drained or when rainfall or irrigation is excessive. Other terms, such as soil saturation, waterlogging, anoxia, and hypoxia, are also commonly used to describe flooding conditions. Flooding causes premature senescence, which results in leaf chlorosis, necrosis, defoliation, reduced nitrogen fixation, the cessation of growth, and reduced yield. The severity of the flooding stress is affected by many factors, including flooding duration, crop variety, growth stage, soil type, fertility levels, pathogens, and flooding conditions.^[1] In general, stream flooding, characterized by the overflow of rivers or creeks into a flood plain, is more damaging than the lowland flooding, characterized by inadequate surface drainage and slow soil permeability of depressional areas. Sediments carried by stream flooding, when deposited on the leaves of flooded plants, can cause severe wilting and plant death within 24 hours of the stress. Flooding can be further divided into either waterlogging, where only the roots are flooded, or complete submergence, where the entire plants are under water. While plants develop adaptive mechanisms to allow them to survive long-term waterlogging, most plants die within 1 or 2 days of submergence.^[1]

The lack of oxygen has been proposed as the main problem associated with flooding.^[2] Indeed, tolerance of anoxia and hypoxia has been used synonymously with tolerance of flooding stress. During the last decades, a great deal of information has been accumulated from research on the molecular, biochemical, and physiological responses of plants to the lack of oxygen rather than to flooding per se.^[3–5] However, tolerance of field flooding appears to be much more complex than tolerance of artificially induced hypoxia and anoxia. Contrary to the injury seen in flooded fields, soybeans can thrive in stagnant water in the greenhouse, and soybeans grown in hydroponic medium continuously bubbled with nitrogen gas, where the dissolved oxygen level was not detectable showed no symptoms of stress.^[6] Soybean, therefore, is much more tolerant to excessive water and a lack of oxygen than previously expected. The reasons underlying the dramatic differences between responses to flooding in the greenhouse and flooding in the field are not known. However, growth reduction and yield loss in flooded fields could arise from root rot diseases, nitrogen deficiency, nutrient imbalance, and/or the accumulation of toxic levels of carbon dioxide (CO₂) in the root zone. Indeed, the levels of CO₂ commonly found in flooded soil (30%) severely caused leaf chlorosis and reduced plant biomass of soybean, a flood-susceptible crop, but not of rice, a flooding-tolerant crop (VanToai, unreported data).

MORPHOLOGICAL AND ANATOMICAL ADAPTATION TO FLOODING STRESS

One important morphological change associated with flooded roots is the formation of aerenchyma tissue, which contains continuous gas-filled channels connecting the root with the shoot. Other morphological changes, including hypertrophy and the formation of lenticels, adventitious roots, and pneumatophores, have also been observed in many plant species.^[7] Flooding can also change the direction of root growth. Roots of tomato and sunflower plants become disgeotropic or negatively geotropic under flooding conditions instead of positively geotropic.^[7] The changes in orientation of roots under flooding conditions enable them to escape stress from the reduced oxygen availability by growing closer to the better aerated soil surface.

PHYSIOLOGICAL, BIOCHEMICAL, AND MOLECULAR ADAPTATION TO FLOODING STRESS

Rice cultivars that showed rapid leaf and sheath growth during submergence did not survive as well as cultivars that did not elongate. Tolerant cultivars appeared to conserve carbohydrates in the shoots and roots during submergence.^[8] Upon removal of the stress, tolerant rice cultivars are able to recover more rapidly and suffer less plant mortality.^[9] An adequate supply of sugar is needed for corn root tips to survive anoxic and hypoxic stresses.^[10] The lack of oxygen induces a set of anaerobic proteins in roots to allow plants to cope with the stress.^[11] These stress proteins are enzymes of either glycolysis, glucose metabolism, or fermentation.^[10,12]

GENETIC VARIABILITY IN FLOODING TOLERANCE

Flooding tolerance is usually defined as minimal or no yield loss. According to VanToai et al.,^[13] waterlogging for 4 weeks during the early flowering stage reduced the average grain yield of 84 U.S. soybean cultivars by 25%. Yield reduction, however, varied from 9% in the most flooding tolerant cultivar to 75% in the most flooding susceptible cultivar. Flooding tolerance can also be defined as high yield under flooding stress. According to this definition, the most flooding tolerant variety in this study produced 3.7 Mg ha⁻¹, while the least produced 1.27 Mg ha⁻¹. When the cultivars were ranked for flooding tolerance based on both definitions, 7 of 10 most flooding tolerant cultivars were the same, and 7 of 10 least flooding tolerant cultivars were also the same. Thus, the two definitions of flooding tolerance, either high yield under flooding or minimal yield difference between non-flooded and flooded conditions, appear to be compatible. Flooding tolerance is independent from non-flooded yield indicating that genetic

variability for flooding tolerance exists and could be improved through plant breeding and selection.

Studies of submergence tolerance of rice showed that the unimproved landraces FR13A, Janki, and FR43B had survival values ranging from 41% to 51% after 10 days of submergence, while only 2–4% of elite cultivars (IR74, IR48, and IR68) survived.^[9]

IMPROVING FLOODING TOLERANCE BY TRADITIONAL PLANT BREEDING

Tolerance of flooding in wheat (*Triticum aestivum* L.) and rice (*Oryza sativa* L.) is a quantitative trait controlled by a small number of genes.^[9,14] Using the submergence tolerant landraces FR13A, Janki, and FR43B as donor parents, rice breeders at the International Rice Research Institute at Los Baños in the Philippines have developed an experimental rice line (IR49830-7-1-2-2) from crosses with the short stature, high-yield IR lines, which produced as much as 4880 kg ha⁻¹.^[9] The result showed that submergence tolerance can be incorporated into improved, high-yielding cultivars to raise the productivity in submergence-prone areas.

IDENTIFYING FLOODING TOLERANCE LOCI AND IMPROVING FLOODING TOLERANCE BY MOLECULAR PLANT BREEDING

Xu and Mackill^[15] identified a single submergence tolerance locus, *Sub1*, that controls about 50% of the variation in submergence tolerance of rice. During the last few years, molecular marker aided selection has been used successfully for the breeding of crops with improved quantitative traits.

VanToai et al.^[16] identified a single DNA marker that was associated with improved plant growth (from 11% to 18%) and grain yields (from 47% to 180%) of soybean in waterlogged environments. The identified marker was uniquely associated with waterlogging tolerance and was not associated with maturity, normal plant height, or grain yield. Near isogenic lines with and without the flooding tolerant marker have been developed and are being field tested under waterlogging conditions to confirm the association of the marker with the tolerance of soybean to waterlogging stress.

IMPROVING FLOODING TOLERANCE BY GENETIC TRANSFORMATION

Flooding induces or accelerates plant senescence in tobacco, tomato, sunflower, carrot, barley, peas, wheat, maize, and soybean. The most obvious visual symptom of flooded plants under stress is the yellowing of leaves followed by necrosis due to premature senescence. Within 1 day of flooding, the concentration of the antiaging

hormone, cytokinin, in sunflower xylem sap declined sharply to a very low level.^[17] In order to test if enhanced endogenous cytokinin production could improve flooding tolerance, Zhang et al.^[18] generated transgenic plants containing a gene coding for cytokinin biosynthesis. Four transgenic *Arabidopsis* lines were chosen for cytokinin and flooding tolerance determinations. The levels of cytokinin were similar between wild-type and transgenic plants in the unflooded treatment. After 5 days of waterlogging, the cytokinin increased 3 to 10 times in transgenic plants as compared to wild-type plants. In three independent experiments, all four transgenic lines were consistently more tolerant to soil waterlogging and complete submergence than wild-type plants. The results indicated that endogenously produced cytokinin can regulate senescence caused by flooding stress, thereby increasing plant tolerance of flooding. This study provides a novel mechanism to improve flooding tolerance in plants.^[18]

In summary, while the lack of oxygen has been used interchangeably with flooding stress, tolerance of field flooding is more complex than tolerance of anoxia and hypoxia. The use of molecular plant breeding and genomic transformation to improve flooding tolerance in crops is promising to be successful.

REFERENCES

1. Sullivan, M.; VanToai, T.; Fausey, N.; Beuerlein, J.; Parkinson, R.; Soboyejo, A. Evaluating on-farm flooding impacts on soybean. *Crop Sci.* **2001**, *41*, 1–8.
2. Kozlowski, T.T., Ed. Extent, causes, and impacts of flooding. In *Flooding and Plant Growth*; Academic Press Inc.: Orlando, 1984; 1–7.
3. Kennedy, R.A.; Rumpho, M.E.; Fox, T.C. Anaerobic metabolism in plants. *Plant Physiol.* **1992**, *84*, 1204–1209.
4. Perata, V.M.; Alpi, A. Plant responses to anaerobiosis. *Plant Sci.* **1993**, *93*, 1–17.
5. Ricard, B.; Couee, I.; Raymond, P.; Saglio, P.H.; Saint-Ges, V.; Pradet, A. Plant metabolism under hypoxia and anoxia. *Plant Biochem.* **1994**, *32*, 1–10.
6. Boru, G.; VanToai, T.; Alves, J.D. Flooding injuries in soybean are caused by elevated carbon dioxide levels in the root zone. Fifth Natl. Symp. Stand Establish. **1997**, 205–209.
7. Hook, D.D. Adaptations to flooding with fresh water. In *Flooding and Plant Growth*; Kolowski, T.T., Ed.; Academic Press: Orlando, 1984; 265–294.
8. Jackson, M.B.; Waters, I.; Setter, T.; Greenway, H. Injury to rice plants caused by complete submergence: A contribution by ethylene (ethene). *J. Exp. Bot.* **1987**, *38*, 1826–1838.
9. Mackill, D.J.; Amante, M.M.; Vergara, B.S.; Sarkarung, S. Improved semidwarf rice lines with tolerance to submergence of seedlings. *Crop Sci.* **1993**, *33*, 749–753.
10. Ricard, B.; Saglio, P.; VanToai, T.; Chourey, P. Evidence for the critical role of sucrose synthase for anoxic tolerance of maize roots using a double mutant. *Plant Physiol.* **1998**, *116*, 1323–1331.
11. Sachs, M.M.; Freeling, M.; Okimoto, R. The anaerobic proteins of maize. *Cell* **1980**, *20*, 761–767.
12. Sachs, M.M.; Subbaiah, C.C.; Sabb, I.N. Anaerobic gene expression and flooding tolerance in maize. *J. Exp. Bot.* **1996**, *47*, 1–15.
13. VanToai, T.; Beuerlein, J.E.; Schmitthenner, A.F.; St. Martin, S.K. Genetic variability for flooding tolerance in soybeans. *Crop Sci.* **1994**, *34*, 1112–1115.
14. Boru, G.; Ginkel, V.M.; Kronstad, W.E.; Boersma, L. Expression and inheritance of tolerance to waterlogging stress in wheat. *Euphytica* **2001**, *117*, 91–98.
15. Xu, K.; Mackill, D.J. A major locus for submergence tolerance mapped on rice chromosome 9. *Mol. Breed.* **1996**, *2*, 219–224.
16. VanToai, T.T.; St. Martin, S.K.; Chase, K.; Boru, G.; Schnipke, V.; Schmitthenner, A.F.; Lark, K.G. Identification of a QTL associated with tolerance of soybean to soil waterlogging. *Crop Sci.* **2001**, *41*, 1247–1252.
17. Burrows, W.J.; Carr, D.J. Effects of flooding the root system of sunflower plants on the cytokinin content of the xylem sap. *Physiol. Plant.* **1969**, *22*, 1105–1112.
18. Zhang, J.; VanToai, T.; Huynh, L.; Preisner, J. Development of flooding-tolerant *Arabidopsis Thaliana* by autoregulated cytokinin production. *Mol. Breed.* **2000**, *6*, 135–144.

Cultivation: Raised-Bed

Kenneth D. Sayre

*International Maize and Wheat Improvement Center (CIMMYT),
Mexico City, Mexico*

Abstract

The raised-bed system of cultivation has been used since time immemorial and continues to play an important role in modern agriculture in both dry land and irrigated production systems. Raised beds have provided a means to help ameliorate the adverse effects when excess water leads to waterlogged conditions that hamper growth and development of most crops. They also provide a mechanism for the efficient application of irrigation water by furrow irrigation when properly managed. It is obvious that raised-bed cultivation systems can play an even more important role if appropriate crop management practices are developed that allow both dramatic reductions in tillage as well as opportunities to retain crop residues on the soil surface and if the needed implements are developed that allow farmers to more easily apply the technology. It is of particular importance that this technology be made readily accessible to small-scale farmers in developing countries.

INTRODUCTION

Indigenous farmers have used raised-bed cultivation systems for centuries. These include the chinapas or floating gardens used in the Valley of Mexico long before the Spanish conquest, the waru warus or caballones, large, raised embankments (in some cases more than 4 m wide by 20 m long), used at high elevations near lake Titicaca in Bolivia and Peru and which closely resemble similar structures used in Ethiopia on poorly drained Vertisols, the “deep ditch-high bed systems” found in the Pearl River basin in China and the mound planting systems used by numerous traditional farmers in Africa, Asia, many Pacific Islands, and the Americas together with numerous other raised-bed examples.^[1] There are several features common to traditional, raised-bed planting systems. They have been widely used to modify wetlands allowing agricultural production, but they have also been used to provide drainage in areas with either persistent or sporadic periods of excess rain that lead to waterlogged conditions. In addition, age-old raised-bed or mound-planting systems have been widely used for many root and tuber crops such as cassava (*Manihot esculenta*, Crantz), yams (*Dioscorea* sp.), sweet potatoes (*Ipomea batatas*, Lam.), taro (*Alocasia* sp.), and potatoes (*Solanum tuberosum*, L.), which is likely related to associated soil conditions that are less favorable for development of many soil-borne diseases associated with poor drainage.^[2]

Raised-bed planting systems, however, have also been used in water-limited situations. The Kofyar people of the Jos Plateau in Nigeria, e.g., enclosed ridges by “tying the ridges” with small dykes to trap water so that

after a rain their fields appeared to be checkered with pools of retained water for later crop use instead of being lost as run off.^[3] A more widely used application of raised beds in many semiarid/arid areas is planting crops on beds/ridges that are formed between furrows that are then used to supply irrigation water. This system has been used in dry regions such as Central Asia and China where irrigation systems were developed in conjunction with the production of row-planted crops such as cotton (*Gossypium* sp.) and has been widely used in the arid, Western United States.

NEED AND ECOLOGICAL NICHE FOR RAISED-BED CULTIVATION SYSTEMS

Therefore, the origin and use of raised-bed cultivation systems have traditionally been associated with water management issues either by providing opportunities to reduce the adverse impact of excess water on crop production through direct drainage, by offering a “raised platform” that allows crops to maintain their roots in unsaturated soil above the level of standing water, or, in contrast, by providing efficient irrigation water delivery to the fields for crop production in semiarid/arid regions. In most cases, the traditional use of raised-bed cultivation systems has involved considerable use of tillage operations (by hand, draft animals, or machinery) as a part of normal preparation to make the beds before planting the next crop. However, there were exceptions to these practices such as the chinampas in Mexico and some of the systems used in the Pearl River Valley in China, which used essentially no tillage combined with

Table 1 Suitability of raised bed cultivation systems for different agro-ecological niches.

Agro-ecological niche	Conventional-till beds with residue removal or incorporation	Permanent beds with surface residue retention
Gravity, furrow irrigated systems	Highly suitable	Highly suitable
High rainfall, dry land systems with water-logging/poor drainage	Suitable	Highly suitable
Low rainfall, dry land systems with season rainfall	Suitable with tied ridges	Highly suitable
Low rainfall, dryland systems using stored soil moisture	Possibly not suitable	Probably suitable

mulching of crop residues on the bed surface. For systems such as the waru warus used in the highlands of Peru/Bolivia, the form and structure of these large, raised beds were maintained as permanent beds with only minimal, superficial tillage.^[1] Table 1 provides a simple classification of different agro-ecological niches where bed planting may have relevance, especially when considering the possible use of permanent beds with surface-retained crop residues.

MODERN APPLICATIONS OF RAISED-BED CULTIVATION

As most crops are very intolerant to waterlogged soil conditions, there are many examples in modern agriculture that have applied raised-bed cultivation to ameliorate problems created by excess water for both dry land and irrigated conditions. In dry land production systems in Australia, e.g., the use of wide, raised beds (1.5–3 m wide depending on machinery) has been found to alleviate serious water logging following episodes of excess rainfall in Victoria, whereas in Western Australia wide, raised beds have proven useful where water logging occurs with even nominal rainfall on the prevalent, duplex soils that are characterized by extremely poor water infiltration. Yields for a wide variety of crops including wheat (*Triticum aestivum*, L.), barley (*Hordeum vulgare*, L.), canola (*Brassica* sp.), lupins (*Lupinus* sp.), faba beans (*Vicia faba*, L.), and oats (*Avena sativa*, L.) have been markedly increased compared to conventional flat planting as a result of 1) prevention of water logging that allows better root aeration; 2) an increase in the amount of plant available water because of more robust root systems; 3) more optimum planting time provided by more timely field access through better drainage and new opportunities to reduce crop turnaround time by reuse of the same bed; and 4) reduction in soil compaction through restricting all machine wheel traffic to furrows between the beds, which provide a permanent guide for controlled traffic patterns.^[4] Where excess water is a problem and raised beds contribute to facilitating drainage, the width of the bed can be quite variable depending upon the methods

used to form and maintain the beds (machinery, draft animal, and or human labor) and on soil type to some extent.

At the other extreme of the moisture/rainfall spectrum is the use of raised beds in mainly semiarid/arid regions to provide irrigation water by furrow irrigation. Farmers located in the Yaqui Valley in the state of Sonora in northwest Mexico provide an exceptional example of the application of raised-bed cultivation with furrow irrigation over about 255,000 ha of irrigated land. During the past, more than 95% of the farmers have changed from the conventional technology of planting most of their crops on the flat with flood and basin irrigation to planting all crops including wheat, the most widely grown crop, on beds. Bed width ranges from 60–100 cm, center bed to center bed, and this narrower bed width facilitates the efficiency of furrow irrigation for most soil types.^[5]

A single row is planted on top of each bed for row crops such as maize (*Zea mays*, L.), soybean (*Glycine max*, L.), cotton, sorghum (*Sorghum vulgare*, L.), safflower (*Carthamus tinctorius*, L.), and dry bean (*Phaseolus vulgaris*, L.), one to two rows per bed are planted for crops such as chickpea (*Cicer arietinum*, L.) and canola, but two to four defined rows, spaced 15–30 cm apart depending on bed width, are used for wheat and other small grains. The farmers who grow wheat on beds obtain about 8% higher yields, use approximately 25% less irrigation water, and encounter at least 25% less operational costs as compared to those still planting conventional tilled wheat on the flat, using flood irrigation.^[5] Surveys that have taken place to track farmer adoption of bed planting in the Yaqui Valley have indicated numerous reasons for this change in crop management and are presented in Table 2.

Most farmers in the Yaqui Valley who have adopted raised beds continue to practice conventional tillage, destroying the beds after the harvest of each crop followed by tillage operations before new beds are formed for the succeeding crop, often accompanied by burning crop residues although some maize and wheat straw are used for fodder.^[5] However, there has been intense farmer interest in the development of new production technologies that will allow marked tillage reductions

Table 2 Farmer survey results concerning reasons for farmer adoption of raised bed planting systems in the Yaqui Valley, Sonora, Mexico.

1. Improves irrigation water management and irrigation water use efficiency by 25–30% as compared to flood irrigation.
2. Facilitates preceding irrigation, which provides an opportunity for initial weed control prior to planting and allows better crop stand establishment especially on soils prone to crust formation following irrigation.
3. Provides a more uniform drainage system that reduces standing water in parts of the field where prolonged waterlogged conditions can occur.
4. Establishes a defined, controlled traffic system by restricting machinery wheels and animals to the furrows between the beds eliminating compaction in the seeded area on top of the bed.
5. Facilitates field access allowing:
 - i. New opportunities both mechanical weed control and more efficient hand weeding for all crops but especially for wheat, thereby reducing herbicide dependence.
 - ii. Better prospects to band apply nutrients (especially nitrogen fertilizers) when and where they can be most efficiently used.
6. Usually allows use of lower seeding rates for small grain crops compared to conventional planting systems.
7. Usually reduces crop lodging, especially for small grains when compared to conventional, flat planting systems.

Source: From Sayre & Moreno Ramos.^[6]

combined with retention of crop residues that may lead to potential reductions in production costs, improved water/input-use efficiency, and more sustainable soil management while allowing continued use of the furrow irrigation system.

RESEARCH AND DEVELOPMENT CHALLENGES

As bed planting provides a natural opportunity to reduce compaction by confining traffic to the furrow bottoms, the next step needed to reduce tillage and manage crop residues on the surface is to only reshape the beds as needed between each crop cycle following even distribution of the previous crop's residues. The seeding of two to four defined rows on top of the bed as opposed to the normal, solid seeding pattern has made it possible to include wheat and other small grains in the system (Fig. 1).

Obviously, there has been and continues to be an important implement modification and development activity associated with the evolution of permanent bed planting technologies in addition to the development of other agronomic management aspects. The furrow makers that farmers already use to make beds with tillage can be easily modified to reshape the permanent beds by simply attaching cutting disk coulters ahead of the furrow openers for residue management. Similarly, existing zero till planters for row crops such as maize and soybean can be easily adapted for planting on permanent beds into crop residues. However, no commercial zero till wheat planter is readily available that can plant two to four rows on wheat on top of 60–100 cm wide beds without tillage and into high levels of crop residues. More effort is needed to develop suitable permanent bed, small grain seeders. And a clear priority needs to focus on the development of appropriate implements, especially for small grains, so that these new



Fig. 1 Wheat and soybean planted on permanent beds under furrow irrigation with retention of crop residues.

permanent bed technologies can be extended to small-scale farmers.

CONCLUSION

The raised-bed system of cultivation has been used since time immemorial and continues to play an important role in modern agriculture in both dry land and irrigated production systems. Raised beds have provided a means to help ameliorate the adverse effects when excess water leads to waterlogged conditions that hamper growth and development of most crops. They also provide a mechanism for the efficient application of irrigation water by furrow irrigation when properly managed. It is obvious that raised-bed cultivation systems can play an even more important role if appropriate crop management practices are developed that allow both dramatic reductions in tillage as well as opportunities to retain crop residues on the soil surface and if the needed implements are developed that allow farmers to more easily apply the technology. It is of particular importance that

this technology be made readily accessible to small-scale farmers in developing countries.

REFERENCES

1. Thurston, H.D. *Sustainable Practices for Plant Disease Management in Traditional Farming Systems*; Westview Press: Boulder, 1992.
2. Barta, A.L.; Schmittthener, A.F. Interaction between flooding stress and Phytophthora root rot. *Plant Dis.* **1986**, *70*, 310–313.
3. Netting, R.M. *Hill Farmers of Nigeria. Cultural Ecology of the Kofyar of the Jos Plateau*; University of Washington Press: Seattle, 1968.
4. Hamilton, G. Raised bed farming newsletter. *Agr. West. Aust.* **1998**, *1* (1), 1–4.
5. Aquino, P. *The Adoption of Bed Planting of Wheat in the Yaqui Valley, Sonora, Mexico*; Wheat Special Report; CIM-MYT: Mexico, 1998; Vol. 17a.
6. Sayre, K.D.; Moreno Ramos, O.H. *Applications of Raised-Bed Planting System Wheat*; Wheat Special Report; CIM-MYT: Mexico, 1997; Vol. 31.

Culture: Soil-less

Nazim Gruda

Federal Office for Agriculture and Food and University of Bonn, Bonn, Germany

Giorgio Prosdocimi Gianquinto

Department Agricultural Sciences, University of Bologna, Bologna, Italy

Yuksel Tuzel

Department of Horticulture, Ege University, Bornova, Turkey

Dimitrios Savvas

Laboratory of Vegetable Production, Agricultural University of Athens, Athens, Greece

Abstract

Soilless systems allow farmers to grow plants without soil. This can eliminate or reduce many problems associated with traditional cultivation on soil *in situ*, such as soil-borne diseases and pests, the related use of agrochemicals, and decline of soil structure and fertility due to continual cultivation of the same or related crop species. Although often associated with vegetables, soilless systems allow cultivation of many other plants, including flowers and ornamentals. In soilless cultivation, plant roots may grow either in porous media (substrates), which are frequently irrigated with a nutrient solution, or directly in the nutrient solution without any solid material. Soilless systems can be either open-loop or closed-loop cultivation systems. The latter, which involves reusing any drainage solution, can substantially reduce potential pollution of water resources by nitrates and phosphates, while contributing to an appreciable reduction in water and fertilizer consumption. Other general advantages of the soilless systems are as follows: 1) plant nutrition can be better controlled than in crops cultivated in the soil, 2) soil tillage and preparation are avoided, thereby increasing the potential length of the cultivation time, and 3) yields are increased. The major disadvantages of soilless culture are the high costs that are required for the initial installation, as well as the increased technical skills that are needed.

INTRODUCTION

Several civilizations have utilized soilless culture. Egyptian hieroglyphic records dating back to several hundred years B.C. describe the growing of plants in water without soil. Somewhere between mythical and historical fact, we find the hanging gardens of Babylon built around 590 B.C. by King Nabucodonosor II, as a gift to his wife Queen Amitis.

In the 11th century, in Mesoamerica, the nomadic Mexica tribe, one of the native tribes who later formed the Aztec empire, built the City of Tenochtitlan on the shores of Lake Texcoco. Artificial islands called chinampas were created by fencing in the shallow lake bed with wattle. In the Andes, the ancient Aymaras built their villages on floating islands, made of totora reeds (*Schoenoplectus californicus* ssp. *tatora*), over the Titicaca lake at 4000 m a.s.l. People lived out of fishing and crops grown on these islands.

On the other face of the earth, in Myanmar (Burma), vegetables and fruits were grown in large gardens that float

on the surface of the Inle lake. Following an ancient tradition, the beds of the floating gardens were made by creating rafts out of rushes and reeds (to make floating beds) and anchoring them with bamboo poles. People dredge up soil from the shallow bottom of the lake and pile it onto the rafts.

In this entry, we define “Culture: Soil-less” as the cultivation of plants in systems other than soil *in situ*, including hydroponics and other growing media or substrates. In soilless culture, either a liquid or aggregate medium is used. The term “hydroponics” described the growing of crops with their roots only in a liquid medium. Growing media are presented in another separate entry (p. 533).

OPPORTUNITY AND CHALLENGES OF SOILLESS CULTURE

Soilless culture offers significant advantages compared with cultivation in soil. Crop production becomes independent from soil characteristics and is possible even in areas with

poor soil structure or high salinity. The main reason for using soilless culture, however, is the reduction of soil-borne pathogens and the improved control over water and nutrient supplies.^[1] Other advantages, such as better environmental protection with closed systems and improved product quality through precise dosage of nutrients, are becoming more and more important.^[2]

The main characteristic of SCS is the restricted volume of rooting medium in comparison to soil-grown crops.^[1] Systems may be open or closed loop. In an open system, any excess irrigation is allowed to go to waste and is not recycled. In a closed system, any drainage is captured and recycled. Most hydroponic systems are inherently closed, but many systems based on solid materials were open. The installations are closed systems, which will likely become mandatory as the nutrient management planning is implemented in more countries. The nutrients used in these systems are applied as a complete nutrient solution.^[1] A closed-system demands improved water quality to prevent the buildup of unwanted ions. It also means that the composition of the nutrient top-up solution has to closely match the nutrient ratios absorbed by the crop. The nutrient top-up solution has been performed based on electrical conductivity measurement. This may be replaced by the use of specific ion sensors, if they are robust enough.^[3] Closed systems also increase the risk of spreading root diseases through the system, and consideration must be given to the treatment of the captured drainage water before the recycling process.^[4]

SYSTEMS WITHOUT A SOLID MEDIUM

Savvas et al.^[5] classified soilless culture and systems into water culture, which includes deep water culture, float hydroponics, nutrient film technique (NFT), deep flow technique, aeroponics, and substrate/aggregate culture (Fig. 1). The first deep water culture systems described by Gericke^[6] were not widely adopted, due to problems in adequately aerating the solution.^[7] Cooper^[8] devised a system in which the plants were grown in a recirculating nutrient solution in polyethylene gullies set on a sloped greenhouse floor. The plant roots developed in the shallow stream of solution in the gullies, hence the name NFT. The shallow stream was devised to overcome the aeration problem inherent in deep water systems.

Some leafy vegetables, such as lettuce, rocket, lamb's lettuce, and spinach, are successfully grown in floating hydroponics (Fig. 1d). In this system, plants are grown in polystyrene plates or other containers that float on the nutrient solution. The main advantages of this system are that 1) installation costs are low; 2) plants have constant access to water and the nutrient solution and a large buffer capacity for pH and nutrients; and 3) crops can be moved easily into the greenhouse; thus results in fast growth and an early harvest with more production cycles during the year. This

novel hydroponic system has been increasingly utilized commercially. In aeroponics, the roots are periodically sprayed with a fine mist of nutrient solution (Fig. 1a). The advantage is optimal aeration, but production costs are high. In addition to these systems, systems with a solid medium are used as well (Fig. 1c and e). Different irrigation systems, such as a row or a drip irrigation system, are used in this case (Fig. 2). For more information concerning growing media, refer to p. 533.

SOILLESS CULTURE AND YIELD

In soilless culture, higher yields can be achieved compared to soil cultivation due to an optimization of nutrients and water availability to the plants. In addition, the soilless cultivation enhances the early yield in crops planted during the cold season of the year, due to higher temperatures that can be maintained in the root zone during the day. For instance, Selma et al.^[9] showed that a growth period of 102 days was needed for fresh-cut lettuces (*Lactuca sativa*) to reach the same maturity stage in soil compared to 63 days in soilless culture. Also, soilless systems may increase water use efficiency over soil-based systems, i.e., the yield per unit of water applied.

However, crop yields can be affected by many factors such as a soilless culture system (i.e., type, substrate, and open or closed), water quality (i.e., sodium chloride content), composition of nutrient solution [i.e., electricity conductivity (EC), pH, and nutrient ratios], fertilization and irrigation strategies, climate conditions, and length of the growing season and management. EC of a nutrient solution, which is a measure of the total ionic content, is an important property of the nutrient solution and is commonly used to control the day-to-day supply of nutrients backed up by periodic analysis of the nutrient solution for levels of individual nutrients. The EC can be manipulated to influence yield and quality, and the optimum level may vary widely among different species.^[5]

SOILLESS CULTURE AND PRODUCT QUALITY

According to Gruda,^[2] soilless culture does not automatically guarantee high-quality products. However, some consumers are rather mistrustful with the vegetables produced in soilless cultivations. This attitude is mainly based on the assumption that the soilless cultivation of plants is based on the extensive use of “chemicals,” in contrast to plants grown in soil that acquire “natural substances.” However, this belief is not based on scientific knowledge. It is well known that higher plants need inorganic substances, mainly in ionic form, to satisfy their nutritional requirements. Thus, for instance, plants take up nitrogen (N) as NO_3^- and NH_4^+ . Any possible uptake of N in organic forms, e.g., amino acids, has a minor contribution to plant N nutrition. Organic N is generally to be converted into inorganic form



(a) Tomatoes grown in an aeroponics system



(b) Commercial NFT cultivation of lettuce



(c) Production of tomatoes in rockwool



(d) Lettuce grown in a commercially-operating floating hydroponic system



(e) Commercial tomato crop grown on rockwool

Fig. 1 Different soilless culture systems: (A) aeroponics; (B) NFT; (C) and (D) solid material (rockwool); and (E) floating hydroponic system.

Source: (A, B, D, and E) Photos by D. Savvas, private collection. (C) Photo by Gruda, private collection.

before it can be absorbed. Consequently, with respect to the quality of edible vegetable products, it is irrelevant whether the N contained in the plant tissues stems from organic

substances of the soil or from inorganic fertilizers. The only main factor influencing vegetable quality is the quantity of absorbed N and the way in which it is utilized in the plant



Fig. 2 Pot cultures of Cyclamen using different irrigation systems.
Source: (A) and (B) Photos by Gruda, private collection.

metabolism, which has an impact on the nitrate (NO₃)–N concentration in edible plant tissues. Both these factors are better managed in soilless cultures, because the small volume of the rooting medium applied in soilless cultures enables a more efficient control of the nutrient supply through the composition of the nutrient solution. Thus, reducing the NO₃–N concentration in the nutrient solution supplied to lettuce or other leafy vegetables for some days prior to harvesting may lower the NO₃[–] content in the leaves of the plants considerably, without a significant yield loss. Moreover, the pressure from root zone diseases is much weaker than in soil grown crops.^[5] Several studies confirm that soilless culture enables growers to produce vegetables without quality loss compared to soil cultivation. As a result, the need to use soil disinfecting chemicals is considerably reduced or eliminated in soilless culture. This carries obvious advantages for the quality of the produced vegetables. Last but not least, the taste of some fruits such as tomato, melon, etc. may be substantially improved in soilless culture by manipulating the total salt and nutrient concentration in the supplied nutrient solution. To obtain optimum results, cultural management must always be adapted to the specific production system being used.

Soilless culture offers more opportunities to control the yield and quality of the product than do conventional soil systems.

ECONOMIC CONSIDERATIONS

Soilless culture has to be economically viable, in order to be adopted. The data in Table 1 show that under greenhouse conditions in Northern Europe, yields and returns tend to be higher in soilless culture systems compared to traditional soil-based systems. This offsets higher costs and results in increased margins.

There is also evidence that the removal of soil sterilants such as methyl bromide will improve the adoption of soilless culture systems under a wider range of conditions.^[11]

CONCLUSION

A major advantage of soilless culture is that it provides a practically sterile root environment, free of pathogens, and independence from soil characteristics. As a result, yield is increased and product quality is improved. This has been

Table 1 An economic comparison between soil systems and soilless cultivation systems for different vegetable crops.

Vegetable crop System	Bell pepper		Tomato		Eggplant		Runner bean		Cucumber	
	Soil	SCS ^a	Soil	SCS	Soil	SCS	Soil	SCS	Soil	SCS
Duration (weeks)	1–46	1–44	50–32	50–32	9–42	52–42	2–26	2–26	15–40	13–38
Yield (kg m ^{–2})	20.0	21.6	25.4	29.0	17.7	31.6	5.0	7.5	25.9	43.2
Receipts (€ m ^{–2})	30.22	33.47	22.25	26.24	17.66	36.50	13.69	21.10	9.74	16.33
Variable costs	9.96	10.14	9.09	8.58	7.31	10.54	4.83	4.88	3.37	5.68
Labor costs	6.92	8.21	6.02	7.15	5.29	8.95	5.63	7.48	3.15	4.33
Margin	13.34	15.12	7.14	10.51	5.06	17.01	3.23	8.74	3.22	6.32
System difference	—	1.78	—	3.37	—	11.95	—	5.51	—	3.10

^aSoilless culture system.
Source: From Göhler, Molitor, et al.^[10]

the case particularly when combined with the use of sophisticated control systems for nutrient supply and irrigation. The disadvantage is that soilless systems are generally expensive and require high qualified personal. Therefore, they have mainly found applications in the production of high-value pharmaceutical crops, fresh vegetables, ornamentals, and in few cases of some fruits.

ACKNOWLEDGMENTS

The authors would like to thank the coauthors of the 1st and 2nd edition of this entry, Munoo Prasad and Michael Maher from Ireland. They also thank Michael Maher for language revision and valuable recommendations.

REFERENCES

1. Gruda, N.; Tanny, J. Protected crops. In *Horticulture – Plants for People and Places, Production Horticulture*; Dixon, G.R., Aldous, D.E., Eds.; Springer Science + Business Media: Dordrecht, 2014; Vol. 1, 327–405.
2. Gruda, N. Do soilless culture systems have an influence on product quality of vegetables? *J. Appl. Bot. Food Qual.* **2009**, *82*, 141–147.
3. Inden, H.; Kubota, Y.; Awaya, S. Automatic ion control of nutrient solutions. *Acta Hort.* **1999**, *481*, 707–712.
4. Wohanka, W. Slow sand filtration and UV radiation, low-cost techniques for disinfection of recirculating nutrient solution or surface water. In *Proceedings of the Eighth International Congress on Soilless Culture, Hunter's Rest, South Africa*, Oct 2–9, 1992. Wageningen, The Netherlands, 1993; 497–511.
5. Savvas, D.; Gianquinto, G.; Tüzel, Y.; Gruda, N. Soilless culture. In *Good Agricultural Practices for Greenhouse Vegetable Crops – Principles for Mediterranean Climate Areas*; Baudoin, W., Nono-Womdim, R., Litaladio, N., Hodder, A., Castilla, N., Leonardi, C., De Pascale, S., Qaryouti, M., Eds.; Plant Production and Protection Paper 217; Food and Agriculture Organization of the United Nations (FAO): Rome, 2013; 303–354.
6. Gericke, W.F. Hydroponics—Crop production in liquid culture media. *Science* **1937**, *85*, 177–178.
7. Stoughton, R.H. *Soilless Cultivation and Its Application to Commercial Horticultural Crop Production*; United Nations Food and Agriculture Organisation: Rome, 1969.
8. Cooper, A.J. Crop production in recirculating nutrient solution. *Sci. Hort.* **1975**, *3*, 251–258.
9. Selma, M.V.; Luna, M.C.; Martínez-Sánchez, A.; Tudela, J.A.; Beltrán, D.; Baixauli, C.; Gil, M.I. Sensory quality, bioactive constituents and microbiological quality of green and red fresh-cut lettuces (*Lactuca sativa* L.) are influenced by soil and soilless agricultural production systems. *Postharvest Biol. Tec.* **2012**, *63*, 16–24.
10. Göhler, F.; Molitor, H.D.; Roth, K.; Wohanka, W. *Erdelose Kulturverfahren im Gartenbau*; Ulmer GmbH & Co.: Stuttgart, 2002; 267 pp.
11. Engindeniz, S. Economic analysis of growing greenhouse cucumber with soil-less culture system: The case of Turkey. *J. Sustain. Agr.* **2004**, *23* (3), 5–19.

Cycling: Carbon and Nitrogen (C/N)

Sylvie M. Brouder

Ronald F. Turco

Department of Agronomy, Purdue University, West Lafayette, Indiana, U.S.A.

Abstract

The demands on modern agriculture to provide ample food, feed, and energy coupled with anthropogenic (human)-driven global climate change have greatly increased the importance of understanding and managing the dynamics of soil carbon (C) and nitrogen (N) cycling (C/N cycling). The world's C and N supplies exist in a continuous and dynamic cycle, and key processes are constantly active in the soil beneath our feet. The amount of C in the top one meter of soil (the pedologic pool) is approximately 3.5-fold greater than that in the atmosphere, and this C helps soil to retain its structure and function including crop production and ecosystem stability. In contrast to C, the great majority of the earth's N is not in soil but in the atmosphere. However, most of the forms useful to plants are found in the soil. Plant productivity in natural systems is often N limited; in agriculture, farmers routinely supplement soil N with fertilizers and manure and, thus, perturb C/N cycles. The cycles of C and N in soil are linked and are controlled by microorganisms. As a result, microorganisms and their activity have a significant impact on some of our greatest challenges including sustaining food production and water quality and reducing soil emission of the greenhouse gases that drive climate change. This entry provides a new overview of the forms and locations of C and N in soil, emphasizes the biological factors controlling pedologic C/N cycling, and addresses how anthropogenic activities stress the soil ecosystem, potentially impacting its stability.

INTRODUCTION

Carbon (C) and nitrogen (N) exist in soils in both inorganic and organic forms. Globally, only 780 petagrams (Pg; 10^{15} g) of C are in the atmosphere while the top one meter of soil is estimated to contain approximately 2700 Pg of C, 57% of which is soil organic matter (SOM).^[1] In contrast, N in soil is less than one-hundredth of 1% of the N content of the atmosphere but, as with soil C, most soil N is in organic versus inorganic forms.^[2] The inorganic C and N forms in soils include the gases carbon dioxide (CO_2) and dinitrogen (N_2) in the air space of soil pores and soluble forms in soil water such as carbonates from the chemical breakdown or weathering of rocks and minerals (e.g., silicates and limestones) and nitrate (NO_3^-) and ammonium (NH_4^+) from the breakdown or mineralization of SOM. C and N occur in all living tissues: C in sugars, starches, lipids, and a staggering array of complex molecules and N in nucleic acids (DNA and RNA) and in amino acids that are assembled into proteins. Both C and N are an integral component of SOM, which is formulated from plant and animal residues in various stages of decomposition including particulates of various size fractions, dissolved OM in soil water, and humus or stable OM as well as the living biomass of soil microorganisms (e.g., bacteria and fungi).^[3] To a large degree, microorganisms control or drive global C and N (C/N) cycling. Even under relatively extreme environmental conditions, soil

C and N are in a state of constant flux where microbiology transforms C and N among an array of molecules and drives C and N movement between the soil and the surrounding air and water. The goal of this entry is to summarize the state-of-the-art knowledge on the microbiology of soil C/N cycling and discuss how human activities including agriculture are causing large perturbations in soil C/N cycles for which outcomes are uncertain but anticipated to continue to contribute to global climate change.

BACTERIA AND FUNGI CONTROL SOIL C/N CYCLES

The proportion of soil C and N in the biomass of soil microorganisms is surprisingly small (1.2% and 2.6%^[4]), given the importance of soil microbiology in C/N cycling. Soil bacteria are typically 1 μm or less in length and live in tiny colonies (only 100 or so cells per colony) found inside of soil aggregates and are typically attached to clay and silt fractions. However, while the actual bacteria are small in physical size, soils are rich in cell number, microbial diversity, and, consequently, metabolic abilities. A surface soil can contain greater than 4000 different microorganism communities (i.e., genomes) and can support as many as 10^9 (one billion) bacteria in one gram.^[5,6] Subsurface soils will tend to have lower population levels but will exceed 10^7 cells g^{-1} .^[7] Because bacteria are small but soil surface

area large, bacteria are found on less than 0.17% of the total surface area of SOM and less than 0.02% of the soil's mineral surface area.^[8] Further, soil bacteria are largely sessile (not freely moving) and prefer to adhere to surfaces using a form of electrostatic interaction (London-van der Waals forces) followed by a hydrophobic interaction, where the cells secrete a polymer that binds them to the surface.^[9] This binding reaction attaches bacteria and soil particles together, helping to create the soil structure. Water is the critical connecting force in soil, as it moves nutrients and oxygen from their sources to the sessile resident population [Fig. 1, process (P) 1]. The exact locations within soil colonized by bacteria reflect the availability of nutrients (typically embedded organic C) and a preference by bacteria for the small soil pores (between 0.25 and 6 μm diameter) associated with the microaggregate fractions.^[10,11] Analysis has shown that greater than 80% of soil bacteria are located within the soil's microaggregate (2 and 53 μm in size) fraction.^[12] Small pores offer an ideal location for bacteria as they allow for the passage of water, which carries nutrients and oxygen but blocks predators such as protozoa.

Fungi also contribute to soil C/N cycling, but fungi differ from bacteria in several important attributes. Soil fungi are resident on the outside of aggregates and are associated with larger organic materials and the coarse sand

fractions.^[13] More importantly, fungi are made mobile by extension of their hyphae and in doing so are the early colonizers and decomposers of fresh organic residues added to soil via natural and anthropogenic activities (Fig. 1, P 3). Fungi function by secreting a wide array of enzymes, which start many important soil processes. The combined abilities of residue colonization and enzyme secretion give fungi a critical role in initiating the C/N cycling processes in fresh residue additions to soil.

Continuous Soil C/N Cycling Requires Diversity in Organisms and Substrates

Soils and fresh additions of plant and animal materials present the resident microorganisms with a compositionally diverse array of organic substrates as potential energy and nutrient sources; genetic and associated functional diversity of the soil microbiology ensures ongoing soil C/N and other nutrient cycling in the ever-changing physical and chemical environments.^[14] Simplistically, nutrient cycling reflects two sets of processes by which microorganisms meet their need for (1) energy and (2) essential nutrients [e.g., C, N, phosphorus (P), and sulfur] to maintain or increase their biomass (cell numbers). Because both bacteria and fungi must assimilate nutrients across a membrane, organic

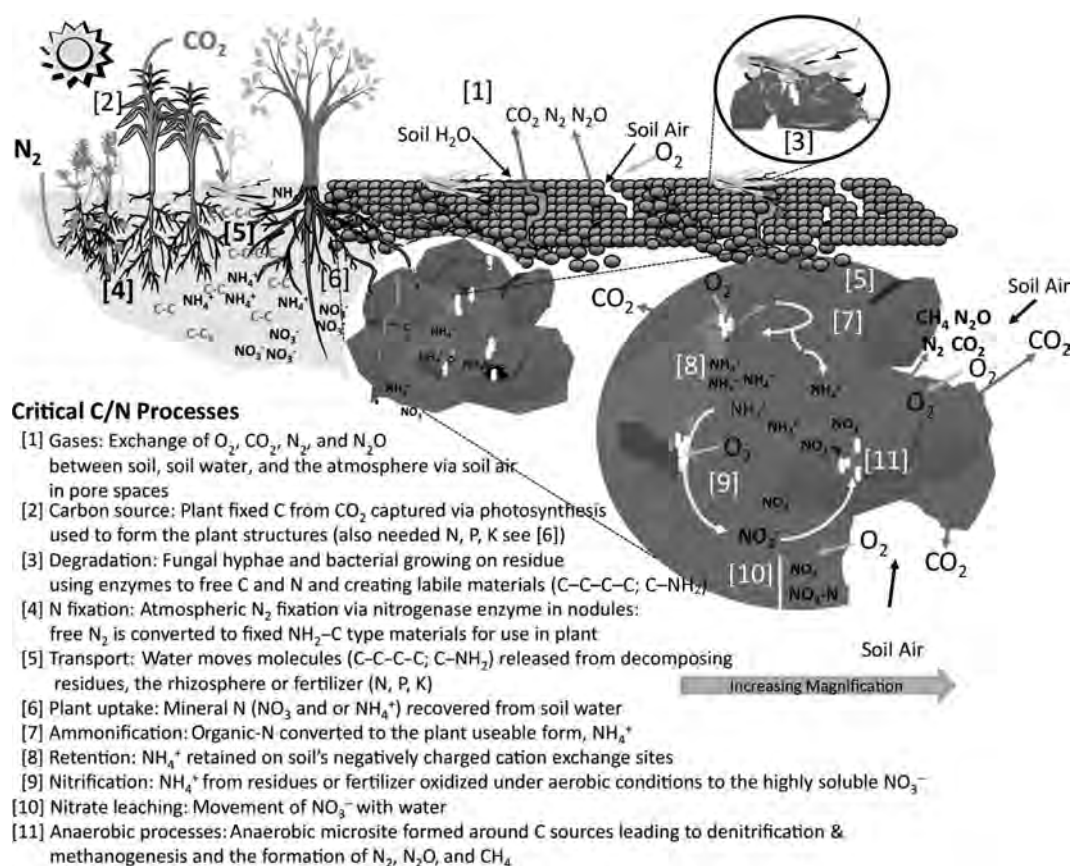


Fig. 1 Diagrammatic representation of the major C/N cycle soil processes. Left side shows C/N cycling at the terrestrial scale and right side is magnified to highlight cycling within the soil matrix.

substrates must first be solubilized into physically smaller fractions. In organic residues of plant origin, the cell walls composed of lignin and cellulose form a recalcitrant physical barrier to degradation. Fungi are one of a few kinds of organisms that can secrete the enzymes necessary to break down cellulose to glucose and the only known organism that can completely degrade lignin via in-place oxidation. Thus, initial decomposition typically involves hyphal extension and enzyme excretion into newly introduced organic material to release labile substances (Fig. 1, P 3). This release process is general and creates a pool of soluble materials including physically smaller C and N molecules such as simple sugars, amino acids, and proteins that can be accessed by other bacteria, fungi, and growing plants. Labile materials not immediately assimilated by fungi may be washed out of decomposing tissue by rainfall; once these substrates reach the soil, they are transported into the soil matrix in soil water and diffuse to the resident microorganisms inside and outside of the soil aggregates (Fig. 1, P 5).

Under most conditions, once soluble organic N molecules are transported into the soil matrix, they rapidly enter a multistep process that sequentially creates the inorganic N forms of NH_4^+ and NO_3^- . Proteins and other organics rich in N are acted upon by bacteria that produce ammonification enzymes that free ammonia (NH_3) from C; in water, NH_3 dissolves to form NH_4^+ (Fig. 1, P 7). The $\text{NH}_4\text{-N}$ can be assimilated by soil bacteria or plant roots (Fig. 1, P 6). When soil water concentrations of NH_4^+ are high, excess NH_4^+ may adsorb onto negatively charged soil surfaces where it is held until soil water NH_4^+ concentrations dissipate and the NH_4^+ diffuses back into solution (Fig. 1, P 8). Additionally, most soils are rich in chemoautotrophic bacteria. Whereas heterotrophic bacteria and fungi power their assimilation of organic C by the oxidation of organics, chemoautotrophic bacteria rely on the oxidation of inorganics (e.g., NH_3) and free CO_2 as their energy and C nutrition sources, respectively. Under aerobic conditions, chemoautotrophic (e.g., nitrifying) bacteria oxidize NH_4^+ , consume CO_2 , producing NO_3^- instead of CO_2 as an end product (Fig. 1, P 9). Nitrate is both water soluble and non-reactive with the negatively charged soil surface; thus, once in the $\text{NO}_3\text{-N}$ form, N is not only available for assimilation by plants and microorganisms but also highly susceptible to loss from the soil system via the physical process of leaching with excess rainfall (Fig. 1, P 10).

It is important to note that different organisms and processes dominate when water from rainfall or flooding causes soils to become anoxic, and these anaerobic processes have significant impacts on soil emission of greenhouse gases. When soil water content is high, macropores in soil are filled with water and O_2 consumption by aerobic bacteria, and plant root respiration (energy derived by electron transfer to O_2) exceeds the O_2 replacement rate from the soil atmosphere. As soil O_2 is depleted, the activity of anaerobic microorganisms that can use electron acceptors other than O_2 will increase. Provided adequate C substrate,

many soil bacteria can use $\text{NO}_3\text{-N}$ instead of O_2 as a terminal electron acceptor in the energy formation process with N_2O and N_2 as end products, a process called denitrification (Fig. 1, P 11). Alternatively, in fermentation, bacteria use the organic substrate itself to replace O_2 as the oxidizing agent; an important fermentation end product is methane (CH_4). Not only is denitrification an important loss pathway for soil N but products of both denitrification and fermentation have also been identified as potent greenhouse gases. The global-warming potentials on the individual molecule basis are 321- and 12-fold that of CO_2 for N_2O and CH_4 , respectively (100-year timeframe).^[15]

An additional subset of microorganisms is critical to the soil C/N cycle. While most soil microorganisms and plants cannot use the wealth of N_2 in the soil atmosphere, some have the ability to convert or “fix” the inert N_2 gas into ammonia NH_3 . This biological N fixation (BNF) has three forms: it can be independent (free living), associative, or symbiotic with respect to higher plants. Free-living N_2 -fixing bacteria are widely distributed and ubiquitous in the soil, but their contributions to terrestrial N is comparatively minor ($<1 \text{ kg ha}^{-1} \text{ yr}^{-1}$) as these organisms lack easy access to C substrates for energy to drive the extremely energy-intensive fixation process. These bacteria fix N for their own metabolic needs but, when they die and decompose, the fixed N becomes available to plants. Some of these free-living bacteria live on the soil surface and can photosynthesize and have been estimated to add $10\text{--}38 \text{ kg N ha}^{-1} \text{ yr}^{-1}$.^[16] Much larger contributions come from soil microorganisms that can colonize root surfaces and invade plant tissues to gain more direct access to energy sources provided by plants. The largest contributions occur when soil microorganisms form true symbiotic relationships with plants such as occurs in legumes. Here, molecular dialogue between the soil microorganism and the plant root leads to the development of a specialized structure or nodule in the plant root to house the N_2 fixation process (Fig. 1, P 4). In the resulting symbiosis, plants and microorganisms achieve a relatively direct exchange of C for N. Symbiotic N fixation can contribute 24 to 250 kg N ha^{-1} in a growing season^[17] and, when plant tissues die, this organic N is returned to the soil. Estimates of total annual contributions of BNF to fixed N in terrestrial ecosystems range from $107^{[18]}$ to $195^{[19]}$ teragrams (Tg; 10^{12} g) N yr^{-1} .

NATURAL VERSUS ANTHROPOGENIC CONTROLS OF C/N CYCLES

Among all the nutrient elements essential for soil microorganisms to grow, C and N are required in the largest quantities and are typically the most limiting in soils of natural systems; consequently, the general state of soil microbiological communities in their natural condition is one of

repressed activity. Higher plants capture their C from atmospheric CO₂ with photosynthesis (Fig. 1, P 2) but, as with soil microorganisms, N must be acquired from soils (Fig. 1, P 6) and is required in higher quantities than any other soil-derived nutrient. Thus, N also constrains plant productivity in unmanaged ecosystems. Over the past several decades, human activities have increasingly perturbed pedologic C/N cycles by physically disturbing soils and/or by increasing quantities of C and/or N in labile form in the soil matrix thereby enhancing the overall activity of microorganisms and increasing opportunities for C and N loss from soils. Chief among anthropogenic activities known to disrupt previously stable cycles are: (1) conversion of natural to managed systems with the goal of growing selected plant species (e.g., crops) with enhanced net primary productivity and (2) the common agricultural practices of soil tillage and N fertilization.

In natural ecosystems, equilibrium exists between what happens when fresh organic residues are added and the genesis and decay of the resident soil humus. The rates of C/N cycling are modulated both by the physical environment and by the composition of added organic materials. Whereas labile substances such as simple sugars, amino acids, and proteins decompose rapidly, subsequent, slower rates of decomposition involve microorganisms attacking increasingly recalcitrant materials. Soil humus, the stable components of SOM that, by definition, bear no resemblance to the floral, faunal, or waste material of origin, is relatively resistant to biodegradation and includes materials derived from lignins, tannins, cutins, and so on.^[20] In the absence of fresh residues, SOM serves as the major nutrient source for resident flora. However, as SOM is difficult to degrade and is N limited, the function of microorganisms is slow when they are constrained to this nutrient source. Without fresh additions, soil microorganisms are “starved” and waiting for the arrival of nutrient (C, N, and P)-rich materials; as a general rule, new additions from an N-rich residue such as from an annual legume will decompose faster as compared to an equivalent mass of lignin-rich residues from tree branches and trunks.

Major physical determinants of cycling include the soil temperature and moisture regimes. Soil temperatures fluctuate daily and seasonally, especially at the soil surface. Cold or freezing temperatures can drastically slow or halt activity of microorganisms, often without killing them, while temperatures above 35°C not only halt activity but may also kill heat-sensitive organisms. Likewise, drought and low soil water status can restrict microbiology and desiccate microorganisms causing dormancy while soil flooding and anoxia slowdown the growth of organisms that prefer aerobic conditions. However, niche differentiation is the hallmark of successful microbiological communities: heat-tolerant will proliferate at the expense of heat-sensitive soil microorganisms under high soil temperatures and, as discussed

in the preceding text, anaerobic will take over from aerobic organisms with prolonged soil flooding.

Anthropogenic Factors Disrupting C/N Cycles: Land Conversion, Tillage, and N Fertilizers

Long-term stability in soil function is a direct result of centuries of buildup of SOM by soil microorganisms. However, the impacts of human activities related to agriculture can result in SOM degradation at rates that far outpace the comparatively slow but constant rates of SOM accumulation in natural systems. Ruddiman^[21,22] dates anthropogenic soil losses of C as CO₂ to 8000–10,000 years ago with the beginnings of settled agriculture; soil C losses as CH₄ date to 5000 years ago and the introduction of paddy rice production. Estimates are that net anthropogenic emissions to the atmosphere total approximately 9.1 Pg C yr⁻¹, 82% of which comes from fossil fuel emissions; the remaining 1.6 Pg C yr⁻¹ has been sourced to land use conversion (deforestation and biomass burning), agricultural cultivation, and associated erosion.^[1] A meta-analysis of impacts of land use change on soil C showed that C stocks in the top 60 cm of soil decreased an average of 50% when soils were converted from native forests to row crop agriculture.^[23] In contrast, converting forest to pasture did not negatively impact soil C stocks, but converting pastures into crops was as detrimental as direct conversion to crops from native forests. Conversion of cropland back to pasture or secondary forest restored some or all soil C, respectively.

Tillage, the mechanical disturbance of soil, is an agricultural practice that dates from the first settled agriculture.^[24] It is a proven technology for conditioning the soil for agricultural crops including breaking up soil layers that are too hard and impede root exploration, mechanically controlling weeds that compete with crops for water and nutrients, incorporating nutrients, manure, and residue, controlling residue-borne pests and pathogens, and aerating the soil. Incorporation promotes rates of decomposition of residues of the soil surface by mechanically breaking down residues into small pieces, mixing residue fragments with soil, and enhancing the surface area of the residue directly in contact with the decomposer microorganisms in the soil. However, potential negative impacts are equally numerous.^[25] Tilled soils that are not covered with a vigorously growing crop or even a thick mat of fallen residue are highly susceptible to erosion by wind and rainfall. Further, pulverization and compaction of the surface and underlying soil layers, respectively, can lead to reduced water infiltration and poor root development that reduce yields. The implementation of tillage on previously pastured and forested land typically results in dramatic losses of soil C.^[23] The implementation of surface residue conservation and no-tillage crop management can restore some soil C but not under all conditions,^[26] and the value of these practices as a universal solution is uncertain.^[27]

Unlike tillage, the application of large amounts of fertilizer N has strong linkages to global climate change. The objective of fertilization in agriculture is to ensure crop productivity and is not limited by competition with soil microorganisms over inadequate mineral N supplies. Early agriculture could rely on only modest amounts of additional N from manures or legumes. Dramatic accelerations of soil C/N cycling in intensively managed agricultural systems are linked to the 1908 discovery of the Haber–Bosch process for high temperature and pressure conversion of N_2 to NH_3 . Subsequent commercialization of the process revolutionized the production and thus availability of N fertilizers. Estimates of global N inputs into agricultural land are 213 Mt (10^6 t) $N\ yr^{-1}$ of which 47% is fertilizer.^[28] In contrast, livestock manures and BNF by leguminous crops account for only 16% and 28%, respectively, of annual global N inputs; cereals, primarily wheat, rice and maize, account for 55 Mt $N\ yr^{-1}$ of fertilizer N. These fertilizer applications are associated with large increases in crop CO_2 capture but also with the acceleration of rates of C/N cycling including losses from soil. The negative impacts of N fertilizer on global N cycles are known to be substantial and range from eutrophication of aquatic systems to acid rain and ozone depletion.^[29] However, implications of an altered N cycle for other biogeochemical cycles are much less certain and include a complex array of climate-relevant feedbacks.^[30] Gruber and Galloway^[30] identify reducing uncertainty in our understanding of coupled C/N cycle perturbations as essential to improving global climate change projections.

CONCLUSION

With reference to ecological function, Becking and Beijerinck concluded more than a century ago “everything is everywhere, but the environment selects.”^[31] Anthropogenic activities change the environment, and any form of agriculture should be viewed as a selective pressure on soil that will cause the equilibrium of the natural C/N cycle to shift. Soil microbiology, with its great diversity of organisms, will respond with strategies optimizing the use of both quantities and forms of C and N introduced by soil management. For agriculture, the challenge for the production of major staple crops that underpin global food security remains the design of management systems that appropriately balance the trade-offs between yield objectives, soil biology, and long-term soil stability and sustainability of ecosystem function. The challenge is particularly complex, as the nature and magnitude of trade-offs are site and soil specific. As we are managing soils to maximize crop yield, the soil biology is constantly responding to our interventions. For example, when reduced forms of N are applied, a subset of soil microorganisms will increase cell numbers and activity to oxidize the material and create the soluble forms such as NO_3^- -N. These soluble forms can be either

leached out of the soil causing water pollution or transported to reduced locations where denitrifying microorganisms consume them and emit greenhouse gases. Conversely, when residues high in C but low in N are applied, soil microorganisms will tie up and use all forms of available nutrients, limiting the amount of N and P available to growing plants. Food security goals that seek to double agricultural yields on all arable lands will require enhanced quantities of plant-available nutrients. More efficient nutrient management strategies to increase crop production while minimizing impacts on air and water quality and global climate change have been frequently identified as critical to achieving global food security. Much of our research investment has gone toward exploring genetic aspects of crop yield and resource (e.g., N and water)-use efficiencies. Research on functional roles of soil microorganisms and their responses to management has lagged and must be elevated to the prominence of plant genetics, as the soil’s resident microorganisms are the systems-level control point for many key processes governing use efficiency of soil resources.

REFERENCES

1. Lal, R. Sequestration of atmospheric CO_2 in global carbon pools. *Energy Environ. Sci.* **2008**, *1*, 86–100.
2. Tamm, C.O. *Nitrogen in Terrestrial Ecosystems: Questions of Productivity, Vegetational Changes, and Ecosystem Stability*; Springer-Verlag: New York, 1991.
3. Stevenson, F.J. *Humus Chemistry: Genesis, Composition, Reactions*, 2nd Ed.; John Wiley & Sons: New York, 1994.
4. Xu, X.; Thornton, P.E.; Post, W.M. A global analysis of soil microbial biomass carbon, nitrogen and phosphorus in terrestrial ecosystems. *Global Ecol. Biogeogr.* **2013**, *22*, 737–749.
5. Torsvik, V.; Goksoy, J.; Daee, F.L. High diversity in DNA of soil bacteria. *Appl. Environ. Microb.* **1990**, *56*, 782–787.
6. Torsvik, V.; Salte, K.; Sorheim, R.; Goksoy, J. Comparison of phenotypic diversity and DNA heterogeneity in a population of soil bacteria. *Appl. Environ. Microb.* **1990**, *56*, 776–781.
7. Konopka, A.E.; Turco, R.F. Biodegradation of organic compounds in vadose zone and aquifer sediments. *Appl. Environ. Microb.* **1991**, *57*, 2260–2268.
8. Hissett, R.; Gray, T.R.G. Microsites and time changes in soil microbial ecology. In *The Role of Terrestrial and Aquatic Organisms in Decomposition Processes*; Anderson, J.M., MacFadyen, A., Eds.; Blackwell: Oxford, 1976; 23–39.
9. van Loosdrecht, M.C.M.; Lyklema, J.; Norde, W.; Zehnder, A.J.B. Influence of interfaces on microbial activity. *Microbiol. Rev.* **1990**, *54*, 75–87.
10. Hattori, T.; Hattori, R. The physical environment in soil microbiology: An attempt to extend the principles of microbiology to soil microbiology. *Crit. Rev. Microbiol.* **1976**, *4*, 423–461.
11. Killham, K.; Amato, M.; Ladd, J.N. Effect of substrate location in soil and soil pore-water regime on carbon turnover. *Soil Biol. Biochem.* **1993**, *25*, 57–62.

12. Ranjard, L.; Poly, F.; Combrisson, J.; Richaume, A.; Gourbière, F.; Thioulouse, J.; Nazaret, S. Heterogeneous cell density and genetic structure of bacterial pools associated with various soil microenvironments as determined by enumeration and DNA fingerprinting approach (RISA). *Microbial. Ecol.* **2000**, *39*, 263–272.
13. Kandeler, E.; Tscherko, D.; Bruce, K.D.; Stemmer, M.; Hobbs, P.J.; Bardgett, R.D.; Amelung, W. Structure and function of the soil microbial community in microhabitats of a heavy metal polluted soil. *Biol. Fert. Soils* **2000**, *32*, 390–400.
14. Zak, D.R.; Tilman, D.; Paramenter, R.; Fischer, F.M.; Rice, C.; Vose, J.; Milchunas, D.G.; Martin, C.W. Plant production and soil microorganisms in late successional ecosystems: A continental-scale study. *Ecology* **1994**, *75*, 2333–2347.
15. Intergovernmental Panel on Climate Change. *Climate Change: The Physical Science Basis*; Cambridge University Press: Cambridge, 2007; 31–35.
16. Witty, J.F.; Keay, P.J.; Frogatt, P.J.; Dart, P.J. Algal nitrogen fixation on temperate arable fields: The Broadbalk experiment. *Plant Soil* **1979**, *52*, 151–164.
17. Werner, D. Production and biological nitrogen fixation of tropical legumes. In *Nitrogen Fixation in Agriculture, Forestry, Ecology, and the Environment*; Werner, D., Newton, W.E., Eds.; Springer: Dordrecht, 2005; 1–13.
18. Galloway, J.N.; Dentener, F.J.; Capone, D.G.; Boyer, E.W.; Howarth, R.W.; Seitzinger, S.P.; Asner, G.P.; Cleveland, C.C.; Green, P.A.; Holland, E.A.; Karl, D.M.; Michaels, A.F.; Porter, J.H.; Townsend, A.R.; Vorosmarty, C.J. Nitrogen cycles: Past, present and future. *Biogeochemistry* **2004**, *70*, 153–226.
19. Cleveland, C.C.; Townsend, A.R.; Fisher, H.; Howarth, R.W.; Hedin, L.O.; Perakis, S.S.; Latty, E.F.; Von Fischer, J.C.; Elseroad, A.; Wasson, M.F. Global patterns of terrestrial biological nitrogen (N₂) fixation in natural ecosystems. *Global Biogeochem. Cy.* **1999**, *13*, 623–645.
20. Derenne, S.; Largeau, C. A review of some important families of refractory macromolecules: Composition, origin, and fate in soils and sediments. *Soil Sci.* **2001**, *166*, 833–847.
21. Ruddiman, W.F. *Plows, Plagues and Petroleum: How Humans Took Control of Climate*; Princeton University Press: Princeton, 2003.
22. Ruddiman, W.F. The anthropogenic greenhouse gas era began thousands of years ago. *Clim. Change* **2005**, *61*, 262–292.
23. Guo, L.B.; Gifford, R.M. Soil carbon stocks and land use change: A meta analysis. *Glob. Change Biol.* **2002**, *8*, 345–360.
24. Lal, R. The plow and agricultural sustainability. *J. Sust. Agric.* **2009**, *33*, 66–84.
25. Hobbs, P.R.; Sayre, K.; Gupta, R. The role of conservation agriculture in sustainable agriculture. *Phil. Trans. R. Soc. B* **2008**, *363*, 543–555.
26. Six, J.; Ogle, S.M.; Breidt, F.J.; Conant, R.T.; Mosier, A.R.; Paustian, K. The potential to mitigate global warming with no-tillage management is only realized when practiced in the long term. *Glob. Change Biol.* **2004**, *10*, 155–160.
27. Giller, K.E.; Witter, E.; Corbeels, M.; Tittonell, P. Conservation agriculture and smallholder farming in Africa: The heretics' view. *Field Crops Res.* **2009**, *114*, 23–34.
28. Connor, D.J.; Loomis, R.S.; Cassman, K.G. *Crop Ecology: Productivity and Management in Agricultural Systems*, 2nd Ed.; Cambridge University Press: New York, 2011; 195–261.
29. Galloway, J.N.; Aber, J.D.; Erisman, J.W.; Seitzinger, S.P.; Howarth, R.W.; Cowling, E.B.; Cosby, B.J. The nitrogen cascade. *Bioscience* **2003**, *53*, 341–356.
30. Gruber, N.; Galloway, J.N. An earth-system perspective of the global nitrogen cycle. *Nature* **2008**, *451* (17), 293–296.
31. De Wit, R.; Bouvier, T. Everything is everywhere, but, the environment selects: What did Baas Becking and Beijerinck really say? *Environ. Microbiol.* **2006**, *8*, 755–758.

Debris Flow

Jau-Yau Lu

Department of Civil Engineering, National Chung Hsing University, Taichung, Taiwan

Abstract

Debris flow is a rapidly moving two-phase gravity flow with a high content of wide-gradation solid particles. Debris flow hazard mitigation is very important because the flow often causes heavy loss of life and property. This entry discusses the classification, initiation, movement and deposition, and hazards mitigation of the debris flow. Based on materials contained in the flow, debris flow can be classified into the following three types: viscous debris flow, non-viscous debris flow and mud flow. In general, physical processes of debris flow consist of initiation, transportation, and deposition of solid particles. Conditions relating to the initiation of debris flow include 1) availability of loose solid material; 2) steep slope gradient; and 3) heavy rainfall. The first two are the potential factors, and the third one is the triggering factor. Debris flow usually occurs near the upland areas with steep slope gradients, moves along gullies or channels, and decelerates rapidly once it reaches a gully or channel outlet where the slope abruptly levels off. The depositional area below the channel outlet is usually called “debris cone” or “debris fan.” One of the interesting characteristics during the transportation of debris flow is the accumulation of large boulders at the front of the flow. Absolute control over debris flows is rarely feasible either physically or economically. The most commonly accepted measures for reducing debris flow damage can be divided into two categories, i.e., structural and non-structural measures.

INTRODUCTION

Debris flow is a rapidly moving two-phase gravity flow with a high content of wide-gradation solid particles. It usually occurs in gullies or on sloping land near the upstream areas. Debris flow has a strong erosive force that erodes the gully bed or banks as it moves downstream. Debris flow hazard mitigation is very important because the flow often causes heavy losses of lives and properties.

The debris flow problem has been recognized as a very critical problem worldwide. Fig. 1 is an example showing the disaster caused by the debris flow of April 29, 2000, at Chia-Yi County in central Taiwan. This debris flow event was induced by a devastating earthquake (100-year return period), 7.3 on the Richter scale, in central Taiwan on September 21, 1999. The earthquake caused the exposure of a large amount of loose soil on the sloping land. With a dry season from October through February, debris flow did not occur until the wet season the following spring. The total amount of rainfall for the storm (April 28–29, 2000) that triggered this particular debris flow event was 152 mm.

Literature review related to the mechanics, predictions, and hazards mitigation of the debris flow can be found in Chien and Wan,^[1] Jan and Shen,^[2] Chen,^[3] Takahashi,^[4] Chen,^[5] Wiczorek and Naeser,^[6] Rickenmann and Chen,^[7] Chen and Major,^[8] Proceedings of the Fifth International Conference on Debris-Flow Hazards Mitigation: Mechanics, Prediction, and Assessment.^[9]

CLASSIFICATION OF DEBRIS FLOW

Debris flow can be classified according to the composition or the causes and processes of occurrence. The causes of occurrence include rainfall, snowmelt, earthquake, volcanic eruption, and destruction of lake.

Based on materials contained in the flow, debris flow can be classified into the following three types:^[1,10] 1) viscous debris flow; 2) non-viscous (stony, waterborne) debris flow; and 3) mud flow (muddy debris flow).

Viscous debris flow, which carries materials ranging from fine clay to large boulders, is the most common type of debris flow. The debris flow shown in Fig. 1 is an example of the viscous debris flow. Both non-viscous (stony) debris flow and the mud flow can be considered as two extreme cases of the viscous debris flow. Non-viscous (stony) debris flow has very few fine particles. It usually runs smoothly as a single and continuous event. On the contrary, viscous debris flow is characterized by the intermittent pulsing. A single event may include several dozen pulses.

Mud flow mainly carries sediment finer than coarse sand (2 mm). However, the flow sometimes also carries a small amount of rock fragment. After an intense thunderstorm on August 25, 1967, a mud flow of more than 50,000 cubic yards occurred in the 1.5-square mile Second Creek Basin in Nevada. It damaged properties and roadways and polluted Lake Tahoe.^[11]

Non-viscous debris flow and mud flow can be considered as the extreme cases of viscous debris flow. Table 1



Fig. 1 Disaster caused by the debris flow of April 29, 2000, at Chia-Yi County in central Taiwan.

Source: Photo by I.Y. Wu.

compares the material and physical characteristics for different types of debris flow.^[1,10]

INITIATION OF DEBRIS FLOW

In general, physical processes of debris flow consist of initiation, transportation, and deposition of solid particles. Conditions that trigger the initiation of debris flow include 1) availability of loose solid material; 2) steep slope gradient; and 3) heavy rainfall.^[1] The first two are the potential factors, and the third one is the triggering factor for the formation of a debris flow.

Table 1 Comparison of the material and physical characteristics for different types of debris flow.

Type	Material	Physical characteristics
Viscous debris flow	Carries fine clay to large boulders, wide gradation	Intermittent pulsing, an event may include several dozen pulses Usually laminar flow
Non-viscous debris flow (stony or waterborne debris flow)	Large boulders carried by water or slurry	Single and continuous event Usually turbulent flow Accumulation of large boulders at the head due to particle collision
Mud flow	Finer than coarse sand (<2 mm), with limited amount of rock fragment	Violent turbulent mixing May be caused by the volcanic eruption



Fig. 2 View looking upstream of the 1996 debris flow near Hsin-Yi bridge.

Source: Photo by S.H. Tseng.

The loose solid material may be due to the prior land-falls, landslides, typhoons, earthquakes (e.g., Fig. 1), glaciers, or rock weathering. The production of loose material is also related to the geological conditions. The gravitational force associated with steep slopes is an important driving force for the initiation of debris flow. An analysis of 150 debris flow gullies in Tibet showed that the debris flow occurred in gullies with slopes of 10–30%. In Japan, most of the debris flow occurred in gullies with slopes greater than 15% or 27%. These observations have been incorporated into a model that predicts the critical slope for the initiation of debris flow.^[12]

An intense rainstorm is the triggering factor for the initiation of debris flow as illustrated in Fig. 1. Fig. 2 shows the debris flow of July 31, 1996, near Hsin-Yi bridge (180 m long), Nantou County, Taiwan. The debris flow was caused by a 24-hour rainfall of 745 mm from Typhoon Herb, which broke the local 200-year maximum daily rainfall record. The amount of deposited material was so large that the surface of the deposited material almost reached the lower surface of the bridge deck (5–6 m high from the original channel bottom).

Essentially, three types of debris flow initiation predominate.^[10] The first two types are due to erosion of channel bed and landslide. The third type is the destruction of a natural dam by the overtopping of river water or by the collapse of the dam body itself under the effects of the seepage water and the hydraulic pressure.

Several landslides occurred near the Tsao-Ling area in central Taiwan, which is approximately 40 km south of the epicenter for the earthquake (7.3 on the Richter scale) of September 21, 1999. An upstream impoundment of water, with an estimated total storage volume of about 43,000,000 m³, was produced. In fact, this location has a long history of landslides induced by either earthquake or heavy rainfall, resulting in the impoundment of water and subsequent breach outflow causing downstream damage and loss of life. For example, landslides in December 1941

and August 1942 produced a total impoundment about 170 m deep with a volume of about 120 million m³ by October 1942 (after heavy rainfall), which eventually failed during overtopping (4 m depth) in May 1951, resulting in 437 deaths and associated damage.

MOVEMENT AND DEPOSITION OF DEBRIS FLOW

Debris flow usually occurs near the upland areas (upper reach of a watershed) with steep slope gradients, moves along gullies or channels (middle reach of a watershed), and decelerates rapidly once it reaches a gully or channel outlet (mouth) where the slope abruptly levels off. The depositional area below the channel outlet is usually called “debris cone” or “debris fan.”

Fig. 3 shows an aerophoto of the debris flow disaster caused by Typhoon Toraji at Shang-An Village, Shueili Township, Nantou County, Taiwan, in 2001. One can clearly see the “debris fan” at the channel mouth.

One of the interesting characteristics during the transportation of debris flow is the accumulation of large boulders at the front of the flow. The phenomenon can be explained by Bagnold's^[13] dispersive pressure concept; i.e., the dispersive force is proportional to the square of particle diameter. With this vertical sorting mechanism, the coarse particles gradually move toward the water surface. Since the maximum flow velocity occurs near the water surface in an open channel flow, the large boulders proceed faster than the smaller particles and gradually accumulate at the front of debris flow.

The mechanics of debris flow are very complex. Models that account for the rheological properties of the debris flow material and processes have been developed by

researchers.^[2,3,14] The debris flow has been modeled as a Bingham substance, as a viscoplastic fluid, or as a dilatant fluid. The flow velocities of a steady 2-D debris flow can be obtained using various models with the assumption of a uniform mixture (continuum).

Mathematical models have also been developed to predict the snout profiles of constantly advancing debris flows on hillslopes.^[4,15] The deposition process of debris flow can be modeled based on the theory of mass and momentum conservation.^[4,16,17] The proper prediction of the deposition areas (e.g., evolution of debris cones) and deposition depths is very useful for disaster and damage prevention.

HAZARDS MITIGATION

Absolute control over debris flows is rarely feasible either physically or economically. There are many different methods for debris flow hazard mitigation. The most commonly accepted measures for reducing debris flow damage can be divided into two categories, i.e., structural and non-structural measures.^[5–9,18]

The structural measures include different types of dams (e.g., consolidation dams, dams with holes such as grid-type dam and slit dam), retarding basins, and diverting and deflecting walls.^[19] Selection of a proper structure depends on the type of debris flow (e.g., stony debris flow vs. mud flow) and the area to be protected (i.e., upper reach, middle reach, or debris cone of a watershed).

The watershed management practices, such as revegetation of barren areas and reforestation, are non-structural debris flow countermeasures. Other non-structural methods include hazard zoning, relocation of endangered properties, and temporary evacuation of people in the debris flow–threatened areas on the basis of warning systems.

Debris flow mitigation projects often utilize a combination of several measures based on both physical and economical considerations. For example, many landslides occurred on the hillslopes and mountainous areas in central Taiwan due to the devastating earthquake of September 21, 1999 (7.3 on the Richter scale). As it was a dry season from October 1999 through next February, the revegetation and reforestation were not immediately effective after the earthquake. The cost and the construction time also prohibited the selection of structural measures. Therefore, hazard zoning, relocation, and emergency evacuation were the most suitable means of debris flow mitigation until other long-term measures became effective or could be implemented.

Improvement of the numerical models is needed to accurately predict the movement and deposition of the debris flow. Collection of more field data for the debris flows is also extremely important for calibrating the numerical models and to increase the reliability of the real-time warning systems.



Fig. 3 Aerophoto of the debris flow disaster caused by Typhoon Toraji at Shang-An Village, Nantou County in 2001.

Source: Photo by Soil and Water Conservation Bureau (SWCB), Council of Agriculture, Taiwan.

CONCLUSION

Absolute control over debris flows is rarely feasible either physically or economically. The most commonly accepted measures for reducing debris flow damage can be divided into two categories, i.e., structural and non-structural measures. Debris flow mitigation projects often utilize a combination of several measures based on both physical and economical considerations. Improvement of the numerical models is needed to accurately predict the movement and deposition of the debris flow. Collection of more field data for the debris flows is also extremely important for calibrating the numerical models and to increase the reliability of the real-time warning systems.

REFERENCES

- Chien, N.; Wan, Z. *Mechanics of Sediment Transport*; ASCE Press: Reston, VA, 1999; 913 pp.
- Jan, C.D.; Shen, H.W. Review dynamic modeling of debris flows. *Lect. Notes Earth Sci.* **1997**, *64*, 93–116. doi:10.1007/BFb0117764.
- Chen, C.L. Comprehensive review of debris flow modeling concepts in Japan. Debris flows/avalanches: Process recognition, and mitigations. *Rev. Eng. Geo., Geo. Soc. Am.* **1987**, *7*, 13–29.
- Takahashi, T. *Debris Flow: Mechanics, Prediction and Countermeasures*, 2nd Ed.; CRC Press: London, 2014; 572 pp.
- Chen, C.L. *Proceedings of the First International Conference on Debris-Flow Hazards Mitigation: Mechanics, Prediction, and Assessment*; American Society of Civil Engineers: New York, 1997; 817 pp.
- Wieczorek, G.F.; Naeser, N.D. *Proceedings of the Second International Conference on Debris-Flow Hazards Mitigation: Mechanics, Prediction, and Assessment*; Balkema: Rotterdam, 2000; 608 pp.
- Rickenmann, D.; Chen, C.L. *Proceedings of the Third International Conference on Debris-Flow Hazards Mitigation: Mechanics, Prediction, and Assessment*; Millpress: Rotterdam, 2003; 1355 pp.
- Chen, C.L.; Major, J.J. *Proceedings of the Fourth International Conference on Debris-Flow Hazards Mitigation: Mechanics, Prediction, and Assessment*; Millpress: Rotterdam, 2007; 748 pp.
- Proceedings of the Fifth International Conference on Debris-Flow Hazards Mitigation: Mechanics, Prediction, and Assessment*; University La Sapienza: Padova, 2011; 1118 pp.
- Takahashi, T. Initiation and flow of various types of debris-flow. In *Proceedings of the Second International Conference on Debris-Flow Hazards Mitigation*, Taipei, Taiwan, Aug 16–18, 2000; Wieczorek, G.F., Naeser, N.D., Eds.; Balkema: Rotterdam, 2000; 15–25.
- Glancy, P.A. A mudflow in the second creek drainage, Lake Tahoe Basin, Nevada, and its relation to sedimentation and urbanization. USDI. Geol. Surv. Prof. Paper. **1969**, 650-C, C195–C200.
- Takahashi, T. Mechanical characteristics of debris-flow. *J. Hydraul. Eng.-ASCE* **1978**, *104* (8), 1153–1169.
- Bagnold, R.A. Experiments on a gravity free dispersion of large solid sphere in a Newtonian fluid under shear. *Proc. R. Soc. Lon. Ser-A* **1954**, *225*, 49–63.
- Chen, C.L. Generalized viscoplastic modeling of debris flow. *J. Hydraul. Eng.* **1988**, *114* (3), 237–258. doi:10.1061/(ASCE)0733-9429(1988)114:3(237).
- Chen, C.L.; Ling, C.H. Fully developed snout profiles of noncohesive debris flows with internal friction. In *Proceedings of the Second International Conference on Debris-Flow Hazards Mitigation*, Taipei, Taiwan, Aug 16–18, 2000; Wieczorek, G.F., Naeser, N.D., Eds.; Balkema: Rotterdam, 2000; 335–344.
- Takahashi, T.; Nakagawa, H.; Harada, T.; Yamashiki, Y. Routing debris flows with particle segregation. *J. Hydraul. Eng.* **1992**, *118* (11), 1490–1507. doi:10.1061/(ASCE)0733-9429(1992)118:11(1490).
- Hsieh, C.L. *On the Analysis and Prediction of Hazardous Zone of Debris Flows*. Research Report NSC80-0410-E006-29; National Science Council: Taipei, 1991; 1–71.
- Scott, K.M. Precipitation-triggered debris-flow at Casita Volcano, Nicaragua: Implications for mitigation strategies in volcanic and tectonically active steeplands. In *Proceedings of the Second International Conference on Debris-Flow Hazards Mitigation*, Taipei, Taiwan, Aug 16–18, 2000; Wieczorek, G.F., Naeser, N.D., Eds.; Balkema: Rotterdam, 2000; 3–13.
- Heumader, J. Technical debris-flow countermeasures in Austria—A review. In *Proceedings of the Second International Conference on Debris-Flow Hazards Mitigation*, Taipei, Taiwan, Aug 16–18, 2000; Wieczorek, G.F., Naeser, N.D., Eds.; Balkema: Rotterdam, 2000; 553–564.

Degradation

Rosa M. Poch Claret

José A. Martínez-Casasnovas

University of Lleida, Lleida, Spain

Abstract

Soil degradation is defined as the loss of the soil's capacity to develop its functions; e.g., support for plant growth, hydrological regulation of watersheds, environmental filtering, and support for buildings, among others. The best known of the soil degradation processes are soil erosion, compaction, alkalization, salinization, pollution, acidification, nutrient depletion, and organic matter loss. The entry reviews the concept of soil degradation and its historical evolution, listing the processes and causes of soil degradation. In addition, the concept of soil conservation is also introduced, as the measures to maintain the soil quality or the techniques to maintain land productivity, reducing soil degradation in the broad sense. Other terms, often considered synonymous to soil degradation, such as desertification of land degradation, are also discussed.

INTRODUCTION

Soil degradation is defined as the loss of the soil's capacity to develop its functions; e.g., support for plant growth, hydrological regulation of watersheds, environmental filtering, and support for buildings, among others. Former approaches to soil degradation, from a productivity or agronomic point of view, have evolved to an environmental one in the last three decades. A more comprehensive approach considers the soil to be a part of the land and part of a socioeconomic, historical, and cultural reality. The environmental point of view enables other degradation processes to be accepted—not only those processes affecting soil's intrinsic characteristics (changes in the physical, chemical, biological soil properties or agricultural use) but also those processes due to externalities. The best known of these degradation processes are soil erosion, compaction, alkalization, salinization, pollution, acidification, nutrient depletion, and organic matter loss. There are other, less common factors that limit soil use, such as the lack of accessibility, natural disasters, and war and even cultural, socioeconomic, and historical limitations to proper use of the land. The increasing interest in sustainable land management is a result of the scarcity of land resources, together with the probable effects of global climatic changes. The soil in a sustainable system must keep its qualities by means of adequate use.

SOIL DEGRADATION: CONCEPTS AND EVOLUTION

Historical Perspective

The earlier definitions of soil degradation were based on the zonality concept,^[1–3] later, they are replaced by definitions

related to land use. Degradation means the loss of the soil's agricultural productivity, which according to Charman^[4] includes soil erosion, salinization, loss of chemical fertility, acidification, and structural degradation as the main soil degradation processes. Aveyard and Charman^[5] also stress the importance of the off-site effects and human impact on the degradation processes, which are the result of the mismanagement of the soil. These processes interact, as soil compaction due to traffic may result in the decline of ecological functions.^[6] Within the framework of land evaluation, degradation is then defined as *the decline in soil quality caused through its improper use by humans*.^[7] The Food Agricultural Organization (FAO)^[8] also points out the importance of the rate at which the process occurs: Degradation may not necessarily be continuous in time but can occur during a short period between two stages of ecological equilibrium. Six types of soil degradation that follow this concept are presented, together with the units of measurement that assess the rate of the process: 1) water erosion ($\text{Mg ha}^{-1} \text{yr}^{-1}$); 2) wind erosion ($\text{Mg ha}^{-1} \text{yr}^{-1}$); 3) excess of ions: salinization (electrical conductivities) and alkalization (exchangeable sodium percentage); 4) chemical degradation: acidification (ΔVyr^{-1}) and toxicity ($\Delta \text{ppm yr}^{-1}$); 5) physical degradation (Δpyr^{-1} ; ΔKsyr^{-1}); and 6) biological degradation (ΔOMyr^{-1}).

At the end of the 1970s, society's general awareness of the importance of the environment permitted an expansion of the concept of soil degradation, which is finally defined as *the loss of the soil's capacity to develop its functions in the environment*. These functions are based on soil being the natural substrate for plant growth, the regulator of the water regime of hydrological units, and the natural environmental filter.^[9] This definition allows an acceptance of processes affecting the land and its socioeconomic, historical, and cultural framework other than those related to

agricultural aspects. Degradation processes include those limiting soil uses in the short, medium, and long terms, not merely those caused by inappropriate agricultural or forestry use among others. Polluting soils, urbanization, mining activities, watershed mismanagement, flooding, global climatic change, loss of accessibility, and even minefield installation in war zones are also recognized as human activities or natural phenomena leading to land degradation. All of these processes have been better assessed with the increase of soil knowledge, the use of models to foresee its behavior, and land information management, particularly geographic information systems.

The concept of degradation has evolved to consider other aspects of the environment, and soil degradation is considered as a particular case of land degradation. For example, the Global Environment Outlook^[10] considers as forms of land degradation, soil erosion, nutrient depletion, water scarcity, salinity, and disruption of biological cycles. Other soil degradation processes, in addition to the traditional approach, have been recognized, e.g., at European Union level,^[11] as erosion, organic carbon decline, soil biodiversity decline, compaction, contamination, salinization and sodification, soil sealing, and landslides. From these, the first six processes are closely linked to agriculture.^[12]

Nevertheless, sometimes the functions developed by the soil are mutually exclusive. It is not expected that soil along a river can be used for agriculture and act as a green filter simultaneously or serve as the support for human settlement and become an area of water table recharge as well. Is a saline soil with a specific halophytic community a degraded soil? Or, must a value be given to the genetic background of the plants growing there? The application of conservation agriculture to African agroecosystems^[13] is a good example of the conflict between competing goals in land management. In fact, soil degradation processes are often assessed from an anthropocentric point of view, by the evaluation of the possible use humankind can make of it, instead of considering its impact on the environment in the long run.

Soil Conservation and Degradation

The concept of soil conservation has also evolved along with soil degradation. Although defined as the group of measures used to prevent degradation, conservation was originally used only as a synonym for antierosion measures. It was formerly defined as *measures to maintain the soil layer*^[11] or *the techniques to maintain the productivity of the land, reducing erosion*.^[14] Other management techniques, such as liming, salt control practices in irrigated land, bioremediation techniques in polluted soils, reduced tillage and no-tillage, among others, are also included when defining soil conservation in the broadest sense.

Other environmental definitions of soil conservation implicitly include the concept of sustainability and refer

more to the conservation of a land devoted to a specific use than to the intrinsic conservation of the soil itself. Soil conservation consists of the *optimal rational use of the natural resources and the environment, taking into account*

Table 1 Soil degradation processes. Interactions between them are not included.

Intrinsic processes
Degradation of the physical fertility
Compaction
Sealing
Crusting
Structural degradation
Soil loss: water and wind erosion
Land movements by civil engineering or urbanization (levelings, landfills, and landcuts)
Mining
Resource consumption; urbanization on highly productive soils
Water excess
Hydromorphism
Impermeabilization by urban use
Sodification and alkalization
Degradation of chemical fertility
Loss of nutrients: Leaching
Extraction by plants (nutrient mining)
Runoff
Immobilization
Acidification, including that due to acid rain
Salinization
Sodification and alkalization
Pollution
Degradation of biological fertility
Loss of organic matter
Extrinsic processes
Loss of accessibility: loss or damage of roads
Change of ownership regime
Loss of material or human resources
Conversion to risk area: natural disasters
Volcanic and seismic activity
Floods
Fires
Human diseases
Wars
Development of inadequate agricultural policies
Climatic fluctuations
Illiteracy
Cultural traumas

Table 2 Causes of soil degradation.

Agricultural practices
Improper agricultural practices
New cultures in former rangeland
Monocultures
Traffic of machinery
Excessive irrigation schemes and schemes without drainage systems
Inadequate or overfertilization
Inappropriate or excess of tillage
Overgrazing
Deforestation
Generation and application of residues to the soil
Natural causes
Climatic changes
Earthquakes and active volcanoes
Floods
Fires
Industrial activities
Mining
Residue applications
Buildings and civil engineering structures
Socioeconomic causes
Human pressure
Urbanization of cropland and marginal agroecosystems
Lack of education
Inadequate ownership system
Tourism
Political causes
Environmental policies
Social and cultural policies
Economical and market policies
Wars

The use of the term desertification was generalized after the Nairobi Conference in 1977, when desertification was defined as the diminution or destruction of the biological potential of the land, and can lead ultimately to desert-like conditions. It is an aspect of the widespread deterioration of ecosystems and has diminished or destroyed the biological potential, i.e., plant and animal production, for multiple-use purposes at a time when increased productivity is needed to support growing populations.^[22]

The previous definition has a clear, economical approach applicable to developing countries and assumes anthropic causes in the process, derived from a population increase causing a high demographic pressure in the system. It does not specifically refer to arid environments, although the lack of water and the lack of the environment's storage capacity are conditions similar to those of a desert.

Thus, because soil is the component of the ecosystem with the largest water storage capacity, it is evident that soil degradation is one of the main causes of desertification. In this sense, Porta et al.^[23] indicate that desertification may be better defined as the *soil degradation that goes beyond the tolerance limits, generally due to a human intervention*. The authors advise to restrict the use of term to degradation processes taking place in marginal areas near deserts. The use of the general term *land degradation* would be more appropriate for environments other than arid or semiarid.

Soil degradation problems can be successfully addressed through a driving forces–pressures–states–impacts–responses analysis.^[24] Some applications can be found in Porta and Poch^[25] and Emadodin et al.^[26]

REFERENCES

1. Plaisance, G.; Cailleux, A. *Dictionnaire des Sols*; La Maison Rustique: Paris, 1958.
2. SSSA. *Glossary of Soil Science Terms*; Soil Science Society of America: Washington, DC, 1979.
3. SSSA. *Glossary of Soil Science Terms*; Soil Science Society of America: Washington, DC, 1982.
4. Charman, P.E.V. Other forms of soil degradation. In *Soils: Their Properties and Management. A Soil Conservation Handbook for New South Wales*; Charman, P.E.V., Murphy, B.W., Eds.; Sydney University Press: Sydney, 1991; 48–53.
5. Aveyard, J.M.; Charman, P.E.V. Soil degradation and productivity: A concluding perspective. In *Soils: Their Properties and Management. A Soil Conservation Handbook for New South Wales*; Charman, P.E.V., Murphy, B.W., Eds.; Sydney University Press: Sydney, 1991; 317–321.
6. van der Ploeg, R.R.; Ehlers, W.; Horn, R. Schwerlast auf dem Acker. *Spektrum der Wissenschaft* **2006**, August, 80–88.
7. Houghton, P.D.; Charman, P.E.V. *Glossary of Terms Used in Soil Conservation*; Soil Conservation Service of New South Wales: Sydney, 1986.
8. FAO. *Método Provisional para la Evaluación de la Degradación de Suelos*; FAO: Rome, 1980.
9. Pla, I.; Ovalles, F. Efecto de los Sistemas de Labranza en la Degradación y Productividad de los Suelos. In *Memorias de la II. Reunión Bienal de la Red Latinoamericana de Labranza Conservacionista*, Guanare, Acarigua, Venezuela, Nov 14–19, 1993; 26–41.
10. UNEP (United Nations Environment Programme). *Global Environment Outlook (GEO-4): Environment for Development*; Progress Press Ltd.: Valletta, 2007.
11. Commission of the European Communities. Proposal for a Directive of the European Parliament and of the Council establishing a framework for the protection of soil and amending Directive 2004/35/EC; Brussels, COM (2006) 232 final, 2006.
12. Louwagie, G.; Gay, S.H.; Sammeth, F.; Ratering, T. The potential of European Union policies to address soil degradation in agriculture. *Land Degrad. Dev.* **2011**, 22 (1), 5–17.
13. Tittone, P.; Scopel, E.; Andrieu, N.; Posthumus, H.; Mapfumo, P.; Corbeels, M.; van Halsema, G.E.; Lahmar, R.

- Lugandu, S.; Rakotoarisoa, J.; Mtambanengwe, F.; Pound, B.; Chikowo, R.; Naudin, K.; Triomphe, B.; Mkomwa, S. Agroecology-based aggradation-conservation agriculture (ABACO): Targeting innovations to combat soil degradation and food insecurity in semi-arid Africa. *Field Crop. Res.* **2012**, *132*, 168–174.
14. Agence de Coopération Culturelle et Technique. *Dictionnaire D'Agriculture et des Sciences Annexes*; La Maison Rustique: Paris, 1997.
 15. Dalal-Clayton, D.B. *Black's Agricultural Dictionary*, 2nd Ed.; A&C Black: London, 1985.
 16. Dudal, R. An evaluation of conservation needs. In *Soil Conservation: Problems and Prospects*; Morgan, R.P.C., Ed.; Wiley: New York, 1981; 3–12.
 17. FAO. *Framework for Land Evaluation*; FAO Soils Bulletin 32; FAO: Rome, 1976.
 18. FAO. *Planning for Sustainable Use of Land Resources: Towards a New Approach*; FAO Land and Water Bulletin 2; FAO: Rome, 1995.
 19. FAO. *Land Quality Indicators and Their Use in Sustainable Agriculture and Rural Development*; FAO Land and Water Bulletin 5; FAO: Rome, 1997.
 20. Pla, I. Advances in soil conservation research: Challenges for the future. *Span. J. Soil Sci.* **2014**, *4* (3), 265–282.
 21. DeLong, C.; Cruse, R.; Wiener, J. The soil degradation paradox: Compromising our resources when we need them the most. *Sustainability* **2015**, *7* (1), 866–879.
 22. UN Conference on Desertification (UNCOD). *Roundup, Plan of Action and Resolutions*; United Nations: New York, 1978.
 23. Porta, J.; López-Acevedo, M.; Roquero, C. *Edafología Para la Agricultura y el Medio Ambiente*, 2nd Ed.; Mundi-prensa: Madrid, 1999.
 24. Görlach, B.; Landgrebe-Trinkunaite, R.; Interwies, E. *Assessing the Economic Impacts of Soil Degradation. Volume I: Literature review. Study commissioned by the European Commission, DG Environment, Study Contract ENV.B.1/ETU/2003/0024*; Ecologic Institute: Berlin, 2004.
 25. Porta, J.; Poch, R.M. DPSIR analysis of land and soil degradation in response to changes in land use. *Span. J. Soil Sci.* **2011**, *1* (1), 100–115.
 26. Emadodin, I.; Narita, D.; Bork, H.R. Soil degradation and agricultural sustainability: An overview from Iran. *Environ. Dev. Sustain.* **2012**, *14* (5), 611–625.

Degradation: Biological

Ibrahim Ortaş

Department of Soil Science and Plant Nutrition, University of Cukurova, Adana, Turkey

Abstract

The soil component of the agro system consists of plant roots, microflora, fauna, organic matter, and the abiotic geochemical matrix. Soil can be a sink for atmospheric carbon dioxide and mitigate the “greenhouse effect.” Soil is a complex living and fragile system that must be protected and managed for its long-term stability and productivity. Soil degradation has been defined as the loss of a soil’s capacity to produce crops and also as the antithesis of soil resilience and quality, which describe biological degradation as the decline in soil organic matter and biomass carbon, a decrease in diversity and activity of soil flora and fauna, or the indiscriminate use of chemicals and pollutants. The increasing loss of species in the past few decades has been a consequence of habitat destruction and direct exploitation of plant populations. As a result, most plant species that are unique to a defined geographic location (called) have disappeared simultaneously with most rhizosphere organisms, such as mycorrhizal fungi and Rhizobium bacteria, causing the decline in soil biological diversity. Mycorrhizal fungi are the largest existing symbiotic plant communities in plant roots, which are an essential part of healthy plant growth, survival, and soil biological quality, particularly in desertified landscapes, because of their essential role in sustaining the vegetation cover. In natural soils, beneficial rhizosphere microorganisms such as mycorrhizae are abundant and readily available to plants for their health and soil quality. Desertified and mismanaged ecosystems are very fragile and subject to progressive disturbance of the vegetation cover and rapid degradation of soils.

INTRODUCTION

The soil component of the agro system consists of plant roots, microflora, fauna, organic matter, and the abiotic geochemical matrix. Soil can be a sink for atmospheric carbon dioxide (CO₂) and mitigate the “greenhouse effect.”^[1] In this system so far, soil has taken second place with respect to plants. Bethlenfalvay and Schüepp^[2] stated that the importance of soil is recognized not only as an agricultural resource base but also as a complex, living, and fragile system that must be protected and managed for its long-term stability and productivity. Soil degradation has been defined as the loss of a soil’s capacity to produce crops and also as the antithesis of soil resilience and quality.^[3] Bedano, Dominguez et al.^[4] describe biological degradation as the decline in soil organic matter (SOM) and biomass carbon (C), a decrease in diversity and activity of soil flora and fauna, or the indiscriminate use of chemicals and pollutants.

The human relationship with soil started more than 15,000 years ago with the ancient civilizations in Anatolia and Mesopotamia, located in the Fertile Crescent of the Tigris and Euphrates river basins. Throughout the early history of agriculture, human use of land for farming and grazing had little effect on the depletion of soil properties. As the human population increased and an agriculture became more settled over the 19th and 20th centuries, however, cultivation practices such as tillage, intensified

conversion of native soil to agriculture, and a large amount of native microorganisms have disappeared, resulting in depletion of organic matter and decreasing soil fertility. Harvesting of residues and deforestation gradually accelerated the problem. Increasing loss of species in the past few decades has been a consequence of habitat destruction and direct exploitation of plant populations. As a result, most plant species being unique to a defined geographic location (called) have disappeared simultaneously with most rhizosphere organisms, such as mycorrhizal fungi and Rhizobium bacteria, causing the decline in soil biological diversity.

Mycorrhizal fungi are the largest existing symbiotic plant communities in plant roots, which are essential part of healthy plant growth, survival, and soil biological quality, particularly in desertified landscapes, because of their essential role in sustaining the vegetation cover. In natural soils, beneficial rhizosphere microorganisms such as mycorrhizae are abundant and readily available to plants for their health and soil quality. However, loss of mycorrhizal fungi is common in mismanaged soils with excess use of chemicals and practices of tillage and irrigation. Desertified and mismanaged ecosystems are very fragile and subject to progressive disturbance of the vegetation cover and rapid degradation of soils. High disturbance and degradation generally result in the loss or reduction of soil microorganisms such as mycorrhizal propagules. Azcon-Aguilar et al.^[5] indicated that under the high degree of

degradation of the ecosystem, diversity of arbuscular mycorrhizal fungi (AMF) present in the soil is rather low. If the mycorrhizal inoculum potential is low or ineffective, it may limit the successful reestablishment of native plants as a result of the decline of soil organic carbon (SOC) and productivity and increase in biological degradation.

FACTORS THAT AFFECT SOIL BIOLOGICAL DEGRADATION

Soil Management

Soil management practices significantly impact soil aggregation^[6] and C dynamics. Ortas, Akpınar, and Lal^[7] have shown that compared to inorganic fertilizer, organic fertilizers, including mycorrhizal inoculation, contributed considerably to soil aggregation and soil C sequestration. Aggregates encapsulate SOC and reduce the rate of its decomposition.^[8] Similarly, plant roots and AMF hyphae provide physical protection to soil C against microbial decomposers through aggregation.^[9] Oades^[10] indicated that the SOC encapsulated within soil aggregates has a lower decomposition rate than that located outside of aggregates.

Soil Tillage Accelerates Soil Biological Degradation

Agricultural machinery, mainly heavy tillage, often causes soil compaction, which increases bulk density and alters soil pore size distribution, and movement of air, water, and nutrients. Soil aggregation is one of the important soil characteristics which mediates many soil chemical, physical, and biological properties and improves soil quality and sustainability.^[11] These aggregates also encapsulate soil organic compounds and reduce the rate of their decomposition.

Heavy tillage results in the loss of SOM through mineralization of C due to breakdown of soil aggregates. Through the mineralization of SOM, more CO₂ is released to the atmosphere.^[12] On the other hand, reduced tillage has been shown to result in increased SOM^[13] and soil microbial biomass levels.^[14] Under field conditions, the soil microbial biomass C and the bacterial functional diversity as well as the microbial activity were good indicators of soil quality, which were positively correlated with SOM contents.^[15]

North American farmers have been widely using a technique known as conservation tillage or zero tillage to reduce the degradation problem. This technique employs special machinery and herbicides to plant crops with minimal disturbance to the soil surface (Fig. 1). Studies since the 1980s have indicated that soil disturbance by tillage reduced the effectiveness of mycorrhizae symbiosis.^[16] The extraradical or external soil hyphae of AM and plant roots associates frequently to produce sticky materials that cause soil particles to adhere, creating small aggregates that



Fig. 1 Heavy plowing and tilt direction tillage reduce SOM levels more than other tillage activities.

Source: Photo Ortas, 2014.

impart structural properties to soil, allowing for improved aeration and water percolation, as well as stability.^[17]

Soil disturbance can reduce the infective potential of AMF in several ways: 1) propagules may be physically damaged, spores may be crushed, and the external hyphae network and colonized root fragments may be disrupted; 2) disturbance may alter the physical, chemical, or biological environment of the soil, which in turn prevents the colonization by or germination of AM propagules; 3) soil disturbance may eliminate host plants, leading to changes in the C supply available to the fungi. C fixation through photosynthesis is the biggest contributor to increasing soil organic pool in through soil profile.

Tisdall, Smith et al.^[18] studies clearly document the role of mycorrhizal hyphae on facilitating the flow of C compounds to the bulk soil, which is an essential process for soil stability. After the development of the soil structure, the aggregates enhance C and store nutrients, thus providing microhabitats for soil organisms.

The Effect of Chemicals on Soil Biological Degradation

Long-term applications of phosphorus (P) and nitrogen (N) and micronutrient fertilizers may reduce the abundance of indigenous AMF and other soil organisms. The highest concentrations of P, zinc (Zn), and copper in maize were observed in nonreduced and reduced tillage treatments concurrently with the highest hyphal densities,^[19] preserved against destruction by tillage. The results have clearly shown that the greater rates of P fertilizers were not required in reduced tillage systems, compared to conventional causing greater soil disturbance.^[20]

High concentrations of heavy metals were determined to adversely affect the size, diversity, and activity of microbial populations in soil, and as a results, soil microbial biomass is reduced. Pesticide, herbicide, fungicides, and nematocides are used to control a wide variety of organisms on many crops; unfortunately, often the control they provide is

erratic. This erratic behavior is not always predictable and has been associated with the chemical, physical, and biological degradation of soils.

CROP MANAGEMENT

Fallowing

Crop rotation and fallow systems have been observed to significantly affect the diversity and the function of mycorrhizal fungi and the loss of root colonizing AMF that benefit the plant by increasing uptake of P in the semiarid cropping systems of Australia and the United States.^[21] Decline in AM colonization and soil propagule densities were the results of fallowing and correlated with plant P and Zn deficiency Ellis.^[21]

Crop Rotation

Humankind's knowledge of the benefit of crop rotation dates back more than 3,000 years. Available literature links its effect on yield to increasing AM colonization. The so-called green revolution inducing the trend of monoculture has been responsible for the increased use of chemical and the control of weeds, insects, and plant diseases.

It has been demonstrated that rotation not only affects mycorrhiza formation in general but also can affect SOC content and also species of AMF associated with different crops. The effect of the conventional rotation and cover crop on protection of soil biological degradation is vital. Many farmers are using cover crop to protect soil sustainability. Results revealed that under the low-input systems, cover crops tended to increase SOC in Mediterranean vineyards.^[22] Under cover crop conditions, soils have higher populations of rhizosphere organisms and mycorrhizae spores and a greater diversity of mycorrhizal fungi than did the conventional farming systems.

Stubble Burning and Grazing

Crop residues are a major source of SOM. Sometimes crop residues are disposed of by burning in an attempt to ease

tillage and also control weeds and soil-borne diseases (Fig. 2). Residue burning significantly reduces microbial activity and diversity, thereby reducing microbial immobilization of N and sulfur, resulting in short-term availability of these nutrients. Chan and Heenan^[23] indicated that under Australian conditions, crop residues burning and tillage are two of the major processes responsible for the decline of SOM content in cropped soils and the resulting soil degradation. Stubble burning causes oxidative losses of N and C. Through this management, SOM content is reduced.

Depletion of SOM is exacerbated on continually cropped arable land because of the burning of crop residues. Forest clearing followed by maize/cassava cropping in Ghana is a good example with a decrease in the C content of the soil from 2.19% to 1.50% in 8 years.^[24] Harvest residues are usually burned on the surface or used as forage, household fuel wood, or construction material before soil cultivation for second crop sowing. This, together with overgrazing, removes large quantities of organic matter from the soil, causing biological degradation by altering the turnover rate of the organic matter. Serious negative environmental impacts of harvesting and overgrazing result, inducing land degradation reflected by increased soil compaction and erosion and decreased soil fertility. This leads to the biological degradation of the soil, with decreased indigenous mycorrhizae and other beneficial soil organisms—the “must” for success for sustainable agriculture, conventionally restored by pasture development. Grazing management is necessary to maintain the ecological function in order to preserve the SOM contents. Heavy grazing strongly affected soil properties indirectly through a change in shrub composition and cultivation, resulting in a decrease in soil N and organic matter.

Depletion of Soil Organic Matter

Most of the soils of many regions of the world have low SOM ranging from 1% to 2%.^[25] Thus, SOM concentration has a strong influence on soil fertility and yield. The low SOC stock and poor soil fertility are attributed to harsh climate and soils of low inherent fertility. The stock of SOC plays a key role in the global C cycle and



Fig. 2 Stubble burning significantly reduces SOM content in soil surface.

Source: Photo Ortas, 2005.

strongly impacts soil biological fertility. The SOC stock is reduced by excessive tillage.^[26] Because of supraoptimal temperatures, high concentrations of calcium carbonate, and excessive heavy tillage, SOM decomposes rapidly and low soil fertility is rather common throughout the most of the world. The severe depletion of SOC is attributed to extractive farming practices and mismanagement of soil and crops.

Depletion of the SOM content is one of the major causes underlying the declining agronomic productivity in soils of the Mediterranean region.^[27] Degradation of organic matter is resulted in the release of large amounts of plant nutrients, particularly C and N. The accelerated soil degradation also depletes soil fertility and increases the emission of CO₂ and methane from erosion-induced transport of SOM.^[28]

SOIL RESILIENCE

Soil biological degradation can be restored by several methods: 1) recycling of organic wastes; 2) suitable soil and crop management reducing tillage; 3) grazing management; 4) increasing crop rotation; 5) better water management; 6) use of cover crops for enhancing SOM; and 7) use of beneficial soil organisms, such as mycorrhizal fungi and bacteria.

Agroforestry is being used to control soil biological degradation, for reforestation and to establish pasture. The latter, especially with deep-rooted perennial grassland, cover crops allow the sustainability of the soil to be regained by increasing the number of indigenous microorganism populations and thereby increasing C levels in the soil through organic matter.

CONCLUSION

Soil and crop management have significant effects on soil quality and protection. Protective agricultural practices are vital for sustainable life. Especially reducing tillage; less chemical fertilizer use; using cover crop, crop rotation, mulching; and using organic fertilizer are very important to increase SOC content.

The use of native and genetically engineered microorganisms holds considerable promise for increasing soil biological productivity. An understanding of the impact of agronomic practices on communities of mycorrhizal fungi and other rhizosphere organisms would help to ensure an opportunity for the utilization of the symbiosis and would contribute to the success of sustainable agriculture. It is important to appreciate that beneficial soil microorganisms are present and are an integral part of most conventional agriculture production systems. It is also important to identify how those benefits can be maximized to enhance agricultural sustainability.

REFERENCES

1. Lal, R. Soil carbon dynamics in cropland and rangeland. *Environ. Pollut.* **2002**, *116* (3), 353–362.
2. Bethlenfalvay, G.J.; Schüepp, H. Arbuscular mycorrhizae and agrosystem stability. In *Impact of Arbuscular Mycorrhizas on Sustainable Agriculture and Natural Ecosystems*; Gianinazzi, S., Schüepp, H., Eds.; Birkhauser Verlag: Basel, 1994; 117–132.
3. Carron, M.P.; Pierrat, M.; Snoeck, D.; Villenave, C.; Ribeyre, F.; Suhardi; Marichal, R.; Caliman, J.P. Temporal variability in soil quality after organic residue application in mature oil palm plantations. *Soil Res.* **2015**, *53* (2), 205–215.
4. Bedano, J.C.; Dominguez, A.; Arolfo, R. Assessment of soil biological degradation using mesofauna. *Soil Till. Res.* **2011**, *117*, 55–60.
5. Azcon-Aguilar, C.; Palenzuela, J.; Roldan, A.; Bautista, S.; Vallejo, R.; Barea, J.M. Analysis of the mycorrhizal potential in the rhizosphere of representative plant species from desertification-threatened Mediterranean shrublands. *Appl. Soil Ecol.* **2003**, *22*, 29–37.
6. Tisdall, J.M.; Oades, J.M. Landmark Papers: No. 1. Organic matter and water-stable aggregates in soils. *Eur. J. Soil Sci.* **2012**, *63*, 8–21.
7. Ortas, I.; Akpinar, C.; Lal, R. Long-term impacts of organic and inorganic fertilizers on carbon sequestration in aggregates of an entisol in Mediterranean Turkey. *Soil Sci.* **2013**, *178* (1), 12–23.
8. Lal, R. Soil carbon stocks under present and future climate with specific reference to European ecoregions. *Nutr. Cycl. Agroecosys.* **2008**, *81* (2), 113–127.
9. Jastrow, J.D.; Miller, R.M.; Lussenhop, J. Contributions of interacting biological mechanisms to soil aggregate stabilization in restored prairie. *Soil Biol. Biochem.* **1998**, *30*, 905–916.
10. Oades, J.M. Soil organic-matter and structural stability—mechanisms and implications for management. *Plant. Soil* **1984**, *76*, 319–337.
11. Moreno-de las Heras, M. Development of soil physical structure and biological functionality in mining spoils affected by soil erosion in a Mediterranean-Continental environment. *Geoderma* **2009**, *149*, 249–256.
12. Larionova, A.A.; Kurganova, I.N.; de Gerenyu, V.O.L.; Zolotareva, B.N.; Yevdokimov, I.V.; Kudeyarov, V.N. Carbon dioxide emissions from agrogray soils under climate changes. *Eurasian Soil Sci.* **2010**, *43* (2), 168–176.
13. Laudicina, V.A.; Novara, A.; Barbera, V.; Egli, M.; Badalucco, L. Long-term tillage and cropping system effects on chemical and biochemical characteristics of soil organic matter in a Mediterranean semiarid environment. *Land Degrad. Dev.* **2015**, *26* (1), 45–53.
14. Sommer, R.; Ryan, J.; Masri, S.; Singh, M.; Diekmann, J. Effect of shallow tillage, moldboard plowing, straw management and compost addition on soil organic matter and nitrogen in a dryland barley/wheat-vetch rotation. *Soil Till. Res.* **2011**, *115*, 39–46.
15. Lupwayi, N.Z.; Clayton, G.W.; O'Donovan, J.T.; Harker, K.N.; Turkington, T.K.; Rice, W.A. Soil microbiological properties during decomposition of crop residues under

- conventional and zero tillage. *Can. J. Soil Sci.* **2004**, *84* (4), 411–419.
16. Miller, M.H.; McGonigle, T.P.; Addy, H.D. Functional ecology of vesicular-arbuscular mycorrhizas as influenced by phosphate fertilization and tillage in an agricultural ecosystem. *Crit. Rev. Biotechnol.* **1995**, *15* (3–4), 241–255.
 17. Hosseini, F.; Mosaddeghi, M.R.; Hajabbasi, M.A.; Sabzalian, M.R. Influence of tall fescue endophyte infection on structural stability as quantified by high energy moisture characteristic in a range of soils. *Geoderma* **2015**, *249–250*, 87–99.
 18. Tisdall, J.M.; Smith, S.E.; Rengasamy, P. Aggregation of soil by fungal hyphae. *Aust. J. Soil Res.* **1997**, *35*, 55–60.
 19. Kabir, Z.; O'Halloran, I.P.; Hamel, C. Combined effects of soil disturbance and fallowing on plant and fungal components of mycorrhizal corn (*Zea mays* L.). *Soil Biol. Biochem.* **1999**, *31*, 307–314.
 20. Miller, M.H. Arbuscular mycorrhizal and the phosphorus nutrition of maize: A review of Guelph studies. *Can. J. Plant Sci.* **2000**, *80*, 47–52.
 21. Ellis, J.R. Post flood syndrome and vesicular-arbuscular mycorrhizal fungi. *J. Produc. Agric.* **1998**, *11*, 200–204.
 22. Peregrina, F.; Perez-Alvarez, E.P.; Garcia-Escudero, E. The short term influence of aboveground biomass cover crops on C sequestration and beta-glucosidase in a vineyard ground under semiarid conditions. *Span. J. Agric. Res.* **2014**, *12* (4), 1000–1007.
 23. Chan, K.Y.; Heenan, D.P. The effects of stubble burning and tillage on soil carbon sequestration and crop productivity in southeastern Australia. *Soil Use Manage.* **2005**, *21* (4), 427–431.
 24. Ney, P.H.; Greenland, D.J. *The Soil Under Shifting Cultivation*; Commonwealth Bureau of Soils: Herpenden, 1960.
 25. Ortas, I.; Lal, R. Climate change and food security in West Asia. In *International Conference on Adaptation to Climate Change and Food Security in West Asia and North Africa*, Kuwait City, Nov. 13–16, 2011, Kuwait Institute for Scientific Research (KISR): Kuwait, 2011.
 26. Lal, R. Conservation tillage for sustainable agriculture—tropics versus temperate environments. *Adv. Agron.* **1989**, *42*, 85–197.
 27. Khresat, S.; Al-Bakri, J.; Al-Tahhan, R. Impacts of land use/cover change on soil properties in the Mediterranean region of northwestern Jordan. *Land Degrad. Dev.* **2008**, *19* (4), 397–407.
 28. Lal, R. Global potential of soil carbon sequestration to mitigate the greenhouse effect. *Crit. Rev. Plant Sci.* **2003**, *22* (2), 151–184.

Degradation: Chemical

M. Rifat Derici

Soil Science, University of Cukurova, Adana, Turkey

Abstract

Soil chemical degradation is undesirable changes in soil chemical properties such as pH, size and composition of cation exchange complex, contents of organic matter, mineral nutrients, and soluble salts. These changes that severely deteriorate the chemical fertility of soils may result in soil acidity, alkalinity, salinity, sodicity, and depletion of soil organic matter and plant nutrients, and lead to a decrease in soil productivity.

INTRODUCTION

Soils are continually undergoing natural chemical changes as a result of weathering. The combination of the weathering process with other factors, such as parent material, climate, biota, and topography, is responsible for the evolution of soil variety. The forces and factors affecting soil formation are always operational in a manner such that a static equilibrium state is never attained. However, the balance at any given time is highly sensitive to these factors, and a new equilibrium toward soil degradation may be the outcome when there are natural or man-made changes. Soil chemical degradation can be described as an undesirable change in soil chemical properties such as pH, size and composition of cation exchange complex, contents of organic matter, mineral nutrients, and soluble salts. Changes in one or more of these properties often have direct or indirect adverse effects on the chemical fertility of soils, which can lead to a decrease in soil productivity.

The widespread types of chemical degradation in soils are an excessive decrease or increase in pH (acidity or alkalinity), an increase in soluble salt content (salinity), a decrease in organic matter content, and a loss of mineral nutrients through leaching or crop offtake. Salinity and alkalinity are closely related and thus will be discussed together.

SOIL pH AND DEVELOPMENT OF SOIL ACIDITY

Chemically fertile soils typically have a pH range of 5.5–7.5. Soil pH is determined by the mineralogical makeup (such as clay minerals, various metal oxides and hydroxides, lime, etc.), organic matter content of the soils, and dissolved carbon dioxide in the aqueous phase. In spite of small diurnal or seasonal fluctuations, soil pH is strongly buffered by these factors. Any measure of pH below 7 is defined as the active acidity, whereas the ability of the soil to maintain a low pH level is referred to as the potential acidity. The active acidity represents the concentration of H^+ ions in the soil solution. Potential acidity includes exchange and non-exchangeable

(titratable) acidities. The exchange acidity includes the H^+ ions associated with the cation exchange sites on the clay mineral and organic fractions. The exchange acidity as a portion of total acidity varies with the mineral makeup of the soil and with the percentage base saturation. There is an equilibrium between active and exchange acidities, and as the H^+ ions in soil solution are neutralized, the cation exchange phase releases new H^+ ions into the solution. The sources of soil acidity are humus, aluminosilicates, hydrous oxides, and soluble salts.^[1]

Humus or humic matter causes acidity through dissociation of H^+ ions in its carboxylic, phenolic, and similar H^+ ions yielding functional groups. The humic fraction is considered as the weak acid component of the total acidity. The cation exchange sites on aluminosilicate clay minerals are occupied by various cations present in the solution phase. The portion of the total exchange sites occupied by H^+ ions increases as the portion of basic cations such as calcium (Ca), potassium (K), magnesium (Mg), and sodium (Na) are reduced through leaching or by plant uptake. This process is accompanied by a reduction in pH, as the dominating exchangeable H^+ ions controls the solution phase. When the soil pH falls below 5.5, aluminum (Al) in octahedral sheets dissociates and is adsorbed in an exchangeable form by clays, thereby increasing the Al saturation. Exchangeable Al is the major cause of exchange acidity. When dissociated from the exchange complex, these Al^{3+} ions produce H^+ ions through hydrolysis reactions. Hydrous oxides of Al and iron (Fe) either in free crystalline and amorphous forms or as coatings or interlayers in clay minerals are also subjected to stepwise hydrolysis and thus produce H^+ ions. The strongly adsorbed Al^{3+} ions on the exchange sites reduce the effective cation exchange capacity of the soil.^[1]

Soil acidity limits the plant growth by H^+ toxicity and decreases the macronutrient base cation content. Furthermore, the solubility of Fe, manganese, and Al containing minerals is enhanced at low pH levels, and the toxicity of these elements becomes a major problem. The activities of soil organisms, including nitrifying bacteria, are severely restricted at pH levels lower than 5.5.^[2]

Soluble salts stemming from dissolution of soil minerals or from fertilizers may directly or indirectly cause acidity. When only the cation of a salt is taken up by plants, H^+ ions released from roots can contribute acidity. Human activities may accelerate or initiate acidification of soils. Fertilizers, especially those of ammonium, indirectly cause acidity by producing H^+ ions as a by-product of nitrification. A monoculture with an excessive use of fertilizers on sensitive soils often results in the depletion of bases and an increase in soil acidity. Soil acidification can also arise as a result of changes in land use pattern, mostly owing to differential rooting patterns and nutrient requirements of plants. For example, establishment of coniferous forest on former grassland in New Zealand increased exchangeable acidity in the top 20–30 cm of soil.^[3] Oxides of nitrogen and sulfur released in the atmosphere by industrial activity are washed onto soils with rainfall (acid rains). Acid rains are not confined to industrial areas but are transported for long distances by mass air movements and affect both agricultural areas and forestland. Forests are especially vulnerable to atmospheric deposition of acids.^[4]

SALINITY AND ALKALINITY

Salinity

Salinity is a common problem in arid and semiarid regions where evapotranspiration exceeds rainfall (dry land salinity). Under these conditions (salinization regime), there is not enough water to wash the soluble salts down the profile below the rooting zone. Thus, soluble salts, originating from various sources, accumulate in the soil profile at certain depths—known as the salic horizon—or at the soil surface (Fig. 1), depending on the water regime. The potential for soil salinization may be greater in irrigated agricultural areas because even good quality irrigation water contains appreciable amounts of salts. If not washed from the soil profile to a drainage system by the leaching fraction of water, the concentration of chloride, sulfate, carbonate, and bicarbonate salts of Na may increase in the soil profile in a very short period (irrigation salinity). Another cause of salinity is the absence of a drainage system or poor drainage, especially in lowlands.^[5] Land use changes may induce or accelerate soil salinization. For example, clearing of perennial species and converting the land to production of annual crop species with much lower transpiration rates have led to rising water tables across vast areas of Australia. Consequently, the upward capillary movement of water carried the dissolved salts, previously present at depth, to the rooting zone.^[5,6] The sources of soluble salts, besides irrigation water, are mineral weathering, fertilizers, salts used on frozen roads, atmospheric transfer of sea spray, and lateral movement of groundwater from salt-containing areas. The contribution from each of these sources depends on the geographical position and topography of the area. In this respect, coastal areas,



Fig. 1 Salt crusts on soil surface caused by irrigation with low quality water in Faisalabad, Pakistan.

Source: Photo courtesy of Dr. C. Kirda, University of Çukurova, Turkey.

especially delta plains and regions lacking natural water outlets, are the most vulnerable areas. It is a well-established fact that unless the salts are properly discharged to the sea by an effective drainage system, the salinity problem cannot be eliminated but will only be transferred to another location.

Salinity affects plant growth by affecting water and nutrient uptake and through specific toxicity of Na, chlorine (Cl), and boron. The dissolved salts in water increase the osmotic potential, thereby creating the so-called physiological drought. Toxicity develops when the ions taken up from soil solution accumulate in leaves. As water is lost in transpiration, the concentration of the toxic ions increases and causes damage to various degrees, depending on the sensitivity of plants. Salinity affects the mineral nutrition of plants by reducing the availability and uptake of nutrients through the interaction of Na and Cl with nutrient cations and anions and by interfering with the transport of elements within the plant.^[7]

The degree of salinity is measured as electrical conductivity of a soil saturation paste or extract and is reported as deci Siemens per meter (dS/m). Plant sensitivity to salinity is expressed as a percent decrease in yield at a given level of salinity.

Alkalinity

Addition of salts to soils increases the concentration of Na in the soil solution more than those of Ca and Mg and alters the

composition of the exchange phase in favor of Na, because the Na salts are the most soluble salts in nature. This increase in exchangeable Na (Na_x) is called sodication, and soils degraded in this manner are referred to as sodic soils. The measure of sodicity is exchangeable sodium percentage, which is the ratio of Na_x to cation exchange capacity. This parameter is sometimes expressed as the exchangeable sodium ratio, which is the ratio of Na_x to other exchangeable cations. If the soil solution contains CO_3^{2-} and HCO_3^- in excess of Ca^{2+} and Mg^{2+} , highly soluble Na salts of these anions hydrolyze and the soil pH rises above 8.5. This process is termed as alkalization. Sodic soils do not necessarily have high pH, but in well-aerated soils, alkalization often follows sodicity. Sodicity, although a chemical property, has adverse effects on soil structure. As Na_x increases, the binding effects of divalent Ca and Mg ions on clay particles are overcome by the dispersing action of Na ions. Dispersed particles move with water and quickly clog the soil pores, causing drastic reductions in water and air permeability in sodic non-saline soils.^[2]

DEPLETION OF SOIL ORGANIC MATTER

The accumulation of soil organic matter, namely humus, in soils starts with the production of biomass and approaches equilibrium dependent on the effects of factors such as climate, type of vegetation, topography, soil texture, and drainage conditions. At equilibrium, when additions by biomass and removal by mineralization are in balance, organic matter contents range from less than 1% in arid regions to over 20% in organic deposits of cool humid climates. Any change in these factors may disturb the system, and a new equilibrium toward depletion of soil organic matter results. A highly disturbing factor in this respect is cultivation. Less organic material is returned to soil at harvest in most cropping systems, and tillage accelerates decomposition of soil organic matter.^[3,8,9] Man-made changes in plant cover, such as the conversion of forests and grasslands to croplands, promote rapid decomposition of the organic matter present in soils.^[8,9] Soil erosion is another factor that causes significant reductions in soil organic matter content, first, by decreasing the overall productivity and thus the production of biomass, some of which is returned to soil, and second, by carrying away the organic matter present in the lighter fraction of the surface soils.^[8]

LOSS OF PLANT NUTRIENTS

Plant nutrient elements are continuously lost from soils by crop removal (nutrient mining), erosion, leaching, and volatilization at rates determined by the type of vegetative cover, cropping system, and climatic conditions. In natural systems such as grasslands and forests, nutrient losses from soils are somewhat compensated by returns from falling plant debris and mineral weathering. On the other hand,

in intensive agriculture, much larger amounts of nutrients are taken away from soil with little return in crop residues, in many cases, exhausting the nutrient reserves in soils.^[1] Erosion permanently removes the readily available nutrients from the topsoil, which is the most productive layer.

Leaching losses in agricultural lands can be significant for anionic nutrients, especially for nitrate, under various farming systems.^[10] Basic nutrient cations such as Ca, Mg, and K may be leached from soils acidified by atmospheric deposition^[11] or from those under shifting cultivation.^[12]

CONCLUSION

Soil chemical degradation can be started and/or accelerated by improper use. Therefore, care should be taken to consider the climatological factors and quality of all agricultural inputs, including that of irrigation water, in the management of soils.

REFERENCES

1. Tan, Z.X.; Lal, R.; Wiebe, K.D. Global soil nutrient depletion and yield reduction. *J. Sustain. Agr.* **2005**, *26* (1), 123–146.
2. Brady, N.C.; Weil, R. *Nature and Properties of Soils*; Prentice Hall: Lebanon, IN, 2007.
3. Alfredsson, H.; Condron, L.M.; Clarholm, M.; Davis, M.R. Changes in soil acidity and organic matter following the establishment of conifers on former grassland in New Zealand. *Forest. Ecol. Manag.* **1998**, *112*, 245–252.
4. Ulrich, B. Waldsterben: Forest decline in West Germany. *Environ. Sci. Technol.* **1990**, *24*, 436–441.
5. Rengasamy, P. World salinization with emphasis on Australia. *J. Exp. Bot.* **2006**, *57* (5), 1017–1023.
6. Clarke, C.J.; George, R.J.; Bell, R.W.; Hatton, T.J. Dryland salinity in south-western Australia: Its origins, remedies, and future research directions. *Aust. J. Soil Res.* **2002**, *40* (1), 93–113.
7. Grattan, S.R.; Grieve, C.M. Salinity–mineral nutrient relations in horticultural crops. *Sci. Hortic.* **1999**, *78*, 127–157.
8. Gregoricha, E.G.; Greerb, K.J.; Andersonb, D.W.; Liang, B.C. Carbon distribution and losses: Erosion and deposition effects. *Soil Till. Res.* **1998**, *47*, 291–302.
9. Riezebos, H.T.H.; Loerts, A.C. Influence of land use change and tillage practice on soil organic matter in Southern Brazil and Eastern Paraguay. *Soil Till. Res.* **1998**, *49*, 271–275.
10. Hansen, B.; Kristensen, E.S.; Grant, R.; Høgh-Jensen, H.; Simmelgaard, S.E.; Olesen, J.E. Nitrogen leaching from conventional versus organic farming systems—A system modelling approach. *Eur. J. Agron.* **2000**, *13*, 65–82.
11. Thimoier, A.; Dupouey, J.-L.; Le Tacon, F. Recent losses of base cations from soils of *Fagus Sylvatica* L. stands in Northeastern France. *AMBIO* **2000**, *29*, 314–321.
12. Brand, J.; Pfund, J.L. Site and watershed-level assessment of nutrient dynamics under shifting cultivation in Eastern Madagascar. *Agr. Ecosyst. Environ.* **1998**, *71*, 169–183.

Degradation: Critical Limits and Irreversible Degradation

Anthony R. Dexter

Institute of Soil Science and Plant Cultivation (IUNG), Pulawy, Poland

Michael A. Zebisch

German Agency for International Cooperation, Tashkent, Uzbekistan

Abstract

The major causes of soil degradation are compaction, loss of organic matter, salinization, and pollution. Soil properties that are important for crop growth or for soil management must often lie within certain critical limits. Degradation of topsoils is usually reversible, whereas degradation of subsoils is much more difficult to reverse or may even be permanent.

INTRODUCTION

The major causes of soil degradation are compaction, loss of organic matter, salinization, and pollution. Soil properties that are important for crop growth or for soil management must often lie within certain critical limits which are summarized in Table 1. Degradation of topsoils is usually reversible, whereas degradation of subsoils is much more difficult to reverse or may even be permanent.

SOIL STRUCTURE

Virgin soils have structures that took hundreds or thousands of years to form as a result of pedogenic processes. When such soils are brought under cultivation, some structural features are destroyed permanently. However, if the soil is managed carefully and appropriately, then it can be productive and can form part of a sustainable ecosystem. Soil with a good structure has a wide distribution of pore sizes, and the structure is stable.^[1] A wide distribution of pore sizes is necessary for soil function and arises when the soil primary particles are arranged into compound particles of different sizes. The structure will be stable if the clay and other colloidal particles, which glue the soil together, remain flocculated and do not disperse when in contact with water. Therefore, a low value of clay dispersibility is required for a stable soil structure. Factors associated with low clay dispersibility include organic matter and calcium. Factors that cause clay dispersion include the presence of sodium and mechanical energy inputs, e.g., from excessive tillage, especially when the soil is wetter than the lower plastic (or lower Atterberg) limit.^[2]

Soil Bulk Density

The bulk density of the soil (or mass per unit volume) is not important as such. Plant roots, for example, do not sense

soil density. What they do sense is mechanical impedance to elongation, which is positively correlated with bulk density, although the correlation is different for different soils. Effects of bulk density on root growth for soils of different clay contents are shown in Fig. 1. The values in this figure are indicative only because mechanical impedance depends on soil water potential, clay mineralogy, soil physical chemistry, and other factors. Roots may also bypass the dense parts of a soil by growing in cracks and biopores. Therefore, an exact relationship between bulk density and root growth is not to be expected. Bulk density is often used as an indicator of the soil physical condition, although it is not a measure of soil function and its use is subject to limitations.

HYDRAULIC PROPERTIES

Water Retention

Soil must be able to store useful amounts of water for crop growth. The maximum amount that can be held is known as the *field capacity*, which is the water content of a soil after at least 2 days of drainage following heavy rain or a large irrigation. In the laboratory, it may be measured as the water content of soil samples when drained from saturation to a matric water potential of -100 h Pa. The dry limit for crop use is the *permanent wilting point*, which is a plant characteristic. This is the water content of the soil below which plants lose turgor pressure and cease growing, even under conditions of zero transpiration. For many crops (e.g., cereals), it corresponds to a matric water potential of -1.5 MPa. The *available water capacity* or *plant available water* is defined as the difference between field capacity and permanent wilting point, and quantify how much water a soil can store for use by crops. The higher this quantity, the better. However, the soil must be aerobic at field capacity. Soil compaction reduces the

Table 1 Critical limits and optimum values of soil properties for most soils under typical agricultural use.

	Soil water/solution			Aeration			Mechanical	
	Matric potential (kPa)	EC (mS cm ⁻¹)	Acidity (pH)	Air-filled porosity (%)	Redox potential Eh (mV)	ODR (μg m ⁻² s ⁻¹)	Penetrometer resistance (kPa)	Friability
Maximum	-5 to -10 ^a	1.5–12 ^b	7.5 ^b	—	700	—	2000	—
Optimum	-20	<0.5	7.0	15	450	>50	1000	0.4, ^c 0.8 ^d
Minimum	-1000 to -1500 ^b	—	6.5 ^b	10	330	20–30 ^b	500	0.2, ^c 0.4 ^d

^aSoil dependent.
^bPlant dependent.
^cBy the method of size dependence.
^dBy the method of coefficient of variation.

value of the available water capacity and also the air-filled porosity through preferential destruction of the larger pores.

Infiltration and Hydraulic Conductivity

Rain or irrigation water must infiltrate into soil where it can be stored for subsequent use by crops, otherwise, runoff will occur with adverse consequences (e.g., erosion, flooding, transport of nutrients, etc.). Infiltration occurs in two main stages. At short times, infiltration is rapid because water is being “sucked” into the soil by capillary forces. At longer times, infiltration reaches a steady rate equal to the saturated hydraulic conductivity of the soil. Whereas irrigation rates can be reduced to match the ability of the soil to absorb water, no such control over rainfall is possible. To avoid runoff, the rate at which the soil can absorb water should be greater than

the highest rainfall rates that may be expected. Because of the two-stage nature of water infiltration, the onset of runoff depends not only on rainfall intensity but also on storm duration and “storm profile” (how intensity varies with time through a storm). Subsoil compaction may reduce hydraulic conductivity by factors of 10 or even 100. Therefore, once the pore space in the topsoil has been filled-up with water, runoff may occur even at low rainfall rates.

SOIL AERATION

Air-Filled Porosity

Adequate soil aeration requires a network of continuous air-filled pores throughout the soil volume. For soil to be aerobic at field capacity (see earlier), these pores need to be larger than 30 μm. Air-filled porosity is the simplest measure of soil aeration.^[4,5] The soil air should contain at least 5% oxygen.

Electrochemical Measurements

Redox potential, E_h , may be measured by the electrical potential between a saturated calomel reference electrode and a platinum (Pt) electrode.^[4] E_h ranges from +700 mV in fully aerobic conditions to -200 mV in completely anaerobic and reduced conditions. A value of E_h = +300 mV corresponds to the initiation of iron reduction, which causes the blue mottling that is characteristic of inadequately aerated soils.

The oxygen diffusion rate (ODR) is a measure of the rate at which oxygen molecules impact with a simulated root that is inserted into the soil. Typically, a root is simulated by a Pt wire of 0.5 mm diameter at an electrical potential of -650 mV relative to a saturated calomel reference electrode.^[4] When an oxygen molecule impacts with the electrode, it is reduced and four electrons flow in an electrical circuit. The electrical current in the circuit is proportional to the number of molecules per second which impact with the Pt wire electrode.

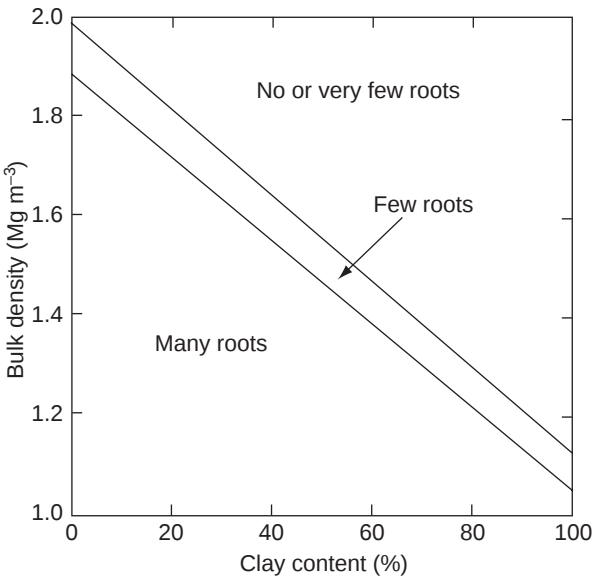


Fig. 1 Critical values of soil bulk density for root growth at a water potential of -33 kPa as a function of soil clay content.
Source: Adapted from Jones.^[3]

SOIL STRENGTH

Penetrometer Resistance

Penetrometers are used to measure soil strength in the field.^[6] Penetrometer resistance is measured by the force required to push a conical metal probe into the soil. The probe is typically of about 20 mm in diameter (or smaller in strong soils) and 30° total included angle. Penetrometer resistance is expressed as the force per unit projected area of the cone, in kPa or MPa. When penetrometer resistance >5 MPa, crop development is seriously affected. Penetrometer resistance increases sharply with decreasing soil water content.

Tillage becomes difficult when penetrometer resistance >2 MPa. There is no real upper limit of soil strength for tillage; it is a question of how much energy the farmer is prepared to use to produce the desired soil structure.

The lower limit for soil strength arises from the requirement for the soil to support traffic and from the need for the soil to be strong enough to prevent plants from falling over. It has been found that passage of a tractor across the soil was not possible when penetrometer resistance was <250 kPa, was possible when it was in the range 250–500 kPa, and was easy when it was >500 kPa.^[7] Trafficability of soil depends very much on the design of the vehicle. Similarly, the ability of the soil to hold plants upright depends not only on the soil but also on the geometry and mechanical properties of the plants.

Precompaction Stress

When soil is subjected to compressive stresses, it may compact (i.e., its volume will reduce) until the stress required for further compaction is equal to the applied stress.^[8] Compaction is an irreversible deformation, and the soil will not return to its original volume when the applied stress is removed. Further compaction will not occur until the previous maximum value of applied stress or *precompaction stress* of the soil is exceeded. Precompaction stress can be measured in the laboratory, and can then be used to determine the vehicle parameters (e.g., total weight, axle loads, and tire inflation pressures), which may be used without causing further compaction damage. It is not possible to give a maximum value for precompaction stress, as each case needs to be considered separately. For example, some soils even after light traffic may have extremely low hydraulic conductivities or infiltration rates which would then give rise to water runoff. For every soil, there is a critical value of precompaction stress which should not be exceeded and which must be determined in relation to all aspects of soil function.^[8]

Friability

Friability describes the ease with which soil can be tilled to produce a seedbed. This can be evaluated qualitatively

in the field by feeling how easily the soil crumbles when rubbed between the fingers. Friability depends on the existence of air-filled pores that provide surfaces of weakness along which the soil can break readily. Such pores are typically microcracks that are wider than 30 µm and therefore air-filled at field capacity. Friability can be quantified either from the decrease of tensile strength with increasing sample size or from the coefficient of variation (standard deviation divided by the mean) of tensile strength values.^[9] A well-structured soil has a high friability, whereas a structureless soil has a small (in the extreme case, zero) friability. Friability is greater when the content of readily dispersible clay in the soil is smaller; in non-compacted soil relative to compacted soil, and in direct-drilled (zero-tilled) soil relative to conventional (plow) tilled soil.

SALINITY AND SODICITY

The term salinity refers to the amount of salt (usually sodium chloride) in the soil, whereas sodicity refers to the amount of sodium adsorbed on the surfaces of the soil particles (most of the surface area exists on the clay particles). Increase of salinity or of other sodium salts (e.g., sodium carbonate and sodium bicarbonate) produces an increase of sodicity. Salinity of soil water is usually measured by electrical conductivity, EC, and sodicity is usually expressed as the exchangeable sodium percentage, ESP.

Salinization of soil occurs when the input of salt is greater than the output. This can occur in irrigation systems where there is some salt in the irrigation water and where there is inadequate drainage to remove these salts. It can also occur as dryland salinization in which clearance of natural deep-rooted vegetation and its replacement by crop or pasture species reduces water extraction from deep layers. The water table can then rise and can bring salt of geological origin toward or to the soil surface.

Sodicity causes soil particles to repel and to disperse in water. Therefore, sodic soils are highly unstable. They dry into extremely dense, strong blocks and their friability is essentially zero. Tillage of such soils, if possible at all, can be done only over a very narrow range of water contents. Sodic soils may exhibit a complete loss of soil function. The critical value of ESP beyond which soil physical degradation becomes apparent is 15%.

ACIDITY (PH)

The normal range of soil pH values is from 4 to 10. However, most plants prefer conditions between pH 6.5 and 7.5. Some specialized plants tolerate values outside this range. In highly acidic conditions (pH < 5), some heavy metals and other elements such as iron, copper, zinc, cobalt, and aluminum become soluble and can become toxic to plants and may be mobilized in the environment with the potential

to cause pollution. In agriculture, there is a natural tendency for pH values to drop. A typical rate of decrease of pH might be 0.4 unit for every 10 years although, of course, this depends on the soil type and on conditions. Therefore, soil pH must be adjusted periodically by additions of lime (as calcium hydroxide or calcium carbonate).

DEGRADATION: REVERSIBLE OR IRREVERSIBLE?

All types of soil degradation can be caused very easily. However, a cure may be either very slow and expensive or completely impracticable.

Symptoms of degradation of the topsoil (i.e., the arable layer in agricultural soils) include slumping, crusting, low friability, and so on. These problems can usually be alleviated by increasing the soil organic matter content and by adding calcium compounds such as gypsum (calcium sulfate) or lime if the soil is acidic. Organic matter can be increased by changing crop rotations to include a pasture phase, by keeping plant cover all the year round, by adding manures, and also by minimizing losses of soil organic matter, which can occur as a result of excessive or high-intensity tillage. Organic matter increases soil aggregation and reduces clay dispersion in water. Calcium compounds flocculate the clay, thereby reducing dispersion and increasing soil stability. Clay floccules form the building blocks for larger compound particles such as aggregates. The use of such practices over a number of years can produce a soil structure that is better for agricultural production and is more stable. Compaction damage of topsoils is usually not permanent because of the ameliorative effects of tillage, biological activity, and wetting and drying cycles.

Degradation of the subsoil often occurs through compaction.^[8] Subsoil tillage may make a temporary improvement, but often the soil will recompact to be as bad as or worse than it was before subsoiling. However, recompaction can be minimized if vehicle weights or axle loads after subsoiling are kept significantly below those that caused the original compaction. Any attempt at subsoil tillage should be done only when the whole soil layer to be tilled is drier than the plastic (lower Atterberg) limit; otherwise, the soil will be destabilized. In many soils, the effects can be considered to be permanent.

Soil salinity can be cured in principle by leaching. However, dryland salinity usually cannot be reversed but may be slowed or halted by the planting of deep-rooted plants (e.g., salt-tolerant trees) that can transpire enough water to stop further water table rise. Sodicty can be cured by displacing the sodium with calcium. However, this is difficult because sodic soils have an extremely low permeability to water.

Displacement and leaching may be accelerated by not using pure water but using water with a high enough electrolyte concentration to keep the clay flocculated and hence the permeability high.^[10,11] It is necessary to consider where the displaced sodium will go in order to avoid degradation of other areas or pollution of water courses.

Heavy metals can be removed slowly from topsoils by “hyperaccumulator plants,” which can be harvested and taken to a safe place for disposal. However, heavy metals are often best controlled by keeping them “fixed” in the soil (adsorbed on the exchange complex). This may require liming of the soil periodically for all time, otherwise the naturally occurring pH drop of the soil may eventually mobilize the heavy metals in the environment. This is the so-called “chemical time bomb” effect.

REFERENCES

1. Dexter, A.R. Advances in the characterization of soil structure. *Soil Till. Res.* **1988**, *11*, 199–238.
2. Watts, C.W.; Dexter, A.R. The influence of organic matter in reducing the destabilisation of soil by simulated tillage. *Soil Till. Res.* **1997**, *42*, 253–275.
3. Jones, C.A. Effect of soil texture on critical bulk densities for root growth. *Soil Sci. Soc. Am. J.* **1983**, *47*, 1208–1211.
4. Glinski, J.; Stepniewski, W. *Soil Aeration and Its Role for Plants*; CRC Press: Boca Raton, 1985.
5. Ball, B.C.; Smith, K.A. Gas movement and air-filled porosity. In *Soil and Environmental Analysis: Physical Methods*; Smith, K.A., Mullins, C.E., Eds.; Marcel Dekker: New York, 2000; 499–538.
6. Bengough, A.G.; Campbell, D.J.; O'Sullivan, M.F. Penetrometer techniques in relation to soil compaction and root growth. In *Soil and Environmental Analysis: Physical Methods*; Smith, K.A., Mullins, C.E., Eds.; Marcel Dekker: New York, 2000; 377–403.
7. Japanese Agriculture. Forestry and Fisheries Research Council Secretariat, Paddy Field Engineering for Mechanization of Agriculture, Bulletin of Major Research Task No. 40, Tokyo, Japan, 1969 (in Japanese).
8. Horn, R.; van den Akker, J.J.H.; Arvidsson, J. *Subsoil Compaction: Distribution, Processes and Consequences*; Catena: Reiskirchen, 2000.
9. Dexter, A.R.; Watts, C.W. Tensile strength and friability. In *Soil and Environmental Analysis: Physical Methods*; Smith, K.A., Mullins, C.E., Eds.; Marcel Dekker: New York, 2000; 405–433.
10. Quirk, J.P.; Pereira, C.; Tanton, T.W. Soil permeability in relation to sodicity and salinity. *Philos. T. R. Soc. A.* **1986**, *A316*, 297–317.
11. Reeve, R.C.; Bower, C.A. Use of high-salt waters as a flocculant and source of divalent cations for reclaiming sodic soils. *Soil Sci.* **1960**, *90*, 139–144.

Degradation: Economic Incentives and Investments

Dennis Wichelns

International Water Management Unit, Colombo, Sri Lanka

Abstract

Soil degradation reduces the productivity of soil and water resources in many areas of the world, causing reductions in crop yields, higher production costs, and lower net income from agriculture. Many researchers agree that soil degradation occurs, in part, because of inappropriate decisions regarding inputs and outputs in agriculture. Yet not all researchers agree on the size of economic damages that result from soil degradation. Researchers also disagree regarding the appropriate allocation of public funds between amelioration activities that reduce the rates of soil degradation directly and research that will generate knowledge of new production methods.

INTRODUCTION

Some authors describe the levels of soil degradation as serious, and they warn that worldwide food production will not keep pace with the increasing demand without substantial expenditures to reduce soil degradation.^[1,2] Others suggest that investments in research are needed to ensure that aggregate food production will be sufficient to meet future demands.^[3–5] Those authors also suggest that public expenditures on agricultural research will have greater impacts on efforts to produce food in the future than expenditures made explicitly to reduce soil degradation.

Both perspectives regarding soil degradation have merit. The economic impacts of soil degradation might be moderate when viewed from a global perspective, but the impacts are severe in selected areas.^[6] Policy reforms and public investments are needed in those areas to reduce soil degradation, improve rural incomes, and enhance food security.^[7] A combination of expenditures on amelioration and research is needed to achieve those goals. The authors examine the potential role of policy reforms and public investments in areas where soil degradation reduces agricultural productivity and household incomes, such as in Asia and sub-Saharan Africa.

CONCEPTUAL FRAMEWORK

Both the farm-level and off-farm impacts of soil degradation arise, in part, because of imperfections in the market structure in which households and firms operate. In most areas, land rental markets do not include provisions by which tenant farmers can gain the long-term benefits of investments in soil management programs. Hence, tenant farmers choose crops and inputs to maximize near-term net revenues, with insufficient consideration of long-term

impacts. Incentives to consider off-farm impacts also are absent in many production settings. Farmers and herders often can choose inputs to maximize net revenues, without considering the impacts of their decisions on other farmers and herders, or on environmental resources.

Market imperfections can be corrected with policy interventions that modify the prices or availability of inputs, or the prices of output generated by firms and households. Quantitative restrictions or per-unit taxes on polluting inputs will reduce the use of those inputs. Per-unit subsidies, low-interest loans, and cost-sharing programs will motivate greater use of inputs and technologies that generate less soil degradation per unit of output. Public investments in research and development also will lower the incremental cost to firms and households of implementing socially desirable production methods. The use of public funds for research and development can be viewed as an effort to address the market imperfection associated with public goods. All firms and households might benefit from new knowledge regarding production methods, but individual firms lack incentives to invest in discovering that knowledge.

In most countries, government policies influence farm-level decisions regarding production activities. Policies that do not reflect resource scarcity or the potential off-farm impacts of soil degradation contribute to the inappropriate use of inputs. Often, policies that are implemented to achieve desirable goals contribute unexpectedly to soil degradation. For example, some governments reduce the farm-level cost of water and energy to promote higher levels of aggregate crop production. The long-term impacts of such subsidies can include waterlogging, salinization, and groundwater depletion.

Macroeconomic policies also have unintended consequences on farm-level decisions. Government procurement programs, exchange rate policies, and credit availability

influence cropping patterns and the types of inputs used in production. Some governments require farmers to sell crops to public agencies at prices below market values. Procurement requirements can reduce farm-level incentives for applying irrigation water and fertilizer. Overvalued exchange rates also reduce farm-level incentives to use fertilizers by making export crops less attractive to potential buyers.

EMPIRICAL EXAMPLES

The impacts of policies on soil degradation are found in areas characterized by intensive agricultural production and where production is extensive, with little use of chemicals and other modern inputs.^[7] Policy reforms and public investments in research and development can reduce soil degradation in both areas. The authors describe recommendations for the Indo-Gangetic plains (an intensive area) and sub-Saharan Africa (an extensive area).

Intensive Agriculture in Asia

Government policies that reduce the farm-level cost of water, energy, and machinery have contributed to the intensification of agriculture in large portions of Asia.^[8–10] Farmers on the Indo-Gangetic plains were motivated to replace traditional wheat varieties with high-yielding varieties during the 1960s and 1970s. The new varieties required more irrigation water and fertilizer, and those inputs were provided to farmers at subsidized prices. The intensification of agriculture generated increasing yields of food crops throughout the 1980s.

The rate of increase in crop yields on the Indo-Gangetic plains slowed markedly in the 1990s, causing concern regarding future food production in the region.^[7,11–13] One cause of the declining growth rates is soil degradation. Large areas have become waterlogged and saline because of excessive irrigation. In some areas, groundwater volume and quality are threatened by the widespread use of tube wells by farmers who receive electricity at heavily subsidized prices.^[14] The inappropriate use of fertilizers also contributes to the declining rates of growth in crop yields.^[15] In some areas, farmers lack the funds or information needed to apply the correct combinations of nutrients.

Appropriate policy reforms include adjusting the prices of irrigation water, electricity, and fertilizers to reflect production costs and scarcity values. In areas where volumetric pricing is not feasible, allocation policies can motivate farmers to improve resource use efficiency. Public investments in research regarding soil, water, and nutrient interactions can generate information describing how these inputs can be used to maximize net revenues, while minimizing the farm-level and off-farm impacts of soil degradation. Research also is needed to determine the optimal

program for improving resource use, increasing household incomes, and enhancing food security in rural areas.

Extensive Agriculture in Sub-Saharan Africa

Food production per capita has declined in sub-Saharan Africa, as population has increased faster than agricultural production.^[16] The causes of slow growth in agricultural productivity include poor soil fertility, insufficient use of chemical and organic fertilizers, and inadequate investment in irrigation systems and other infrastructure.^[17,18] Water scarcity and the inadequate use of fertilizers have resulted in soil degradation in many areas. In addition, government policies that have taxed agriculture heavily have limited the farm-level accumulation of capital and discouraged farmers from making the investments needed to enhance soil productivity.

Appropriate policy reforms include allowing farmers to receive full market prices for their output and removing import restrictions on fertilizer. Incentives that motivate private firms to import, distribute, and deliver fertilizers to farms also will motivate greater use. Public investments in rural infrastructure, such as roads, warehouses, and distribution centers, will stimulate greater use of fertilizer, while also enhancing farm-level crop marketing opportunities. Research designed to determine optimal water and fertilizer management strategies in rain-fed areas also will be helpful in reducing soil degradation and improving rural incomes.

CONCLUSION

The economic impacts of soil degradation are severe in selected areas where reductions in soil productivity limit production opportunities and reduce rural incomes. Policy reforms and public investments are needed to reduce soil degradation and restore soil productivity in those areas. Knowledge gained from those interventions will be helpful in assessing the potential gains from additional investments in other areas where the impacts of soil degradation are less severe.

ACKNOWLEDGMENT

I appreciate the helpful comments of Ellen Burnes, Megumi Nakao, and two reviewers regarding earlier versions of this entry (contribution no. TP 03-20 of the California Water Institute).

REFERENCES

1. Pimentel, D.; Kounang, N. Ecology of soil erosion in ecosystems. *Ecosystems* **1998**, *1*, 416–426.
2. Pimentel, D. Soil erosion and the threat to food security and the environment. *Ecosyst. Health* **2000**, *6*, 221–226.

3. Crosson, P. Soil erosion estimates and costs. *Science* **1995**, *269*, 461–465.
4. Crosson, P. Will erosion threaten agricultural productivity? *Environment* **1997**, *39*, 4–9, 29–31.
5. Crosson, P. Sustainable agriculture. *Environment* **1994**, *36*, 45.
6. Faeth, P. Building the case for sustainable agriculture. *Environment* **1994**, *36*, 16–20, 34–39.
7. Rosegrant, M.W.; Paisner, M.S.; Meijer, S. The future of cereal yields and prices: Implications for research and policy. *J. Crop Prod.* **2003**, *9*, 661–690.
8. Vyas, V.S. Asian agriculture: Achievements and challenges. *Asian Dev. Rev.* **1983**, *1*, 27–44.
9. Vaidyanathan, A. India's agricultural development policy. *Econ. Polit. Weekly* **2000**, *35*, 1735–1741.
10. Vaidyanathan, A. Agricultural input subsidies. *Agr. Situat. India* **2000**, *57*, 261–265.
11. Aggarwal, P.K.; Bandyopadhyay, S.K.; Pathak, H.; Kalra, N.; Chander, S.; Kumar, S. Analysis of yield trends of the rice–wheat system in north-western India. *Outlook Agr.* **2000**, *29*, 259–268.
12. Dobermann, A.; Dawe, D.; Roetter, R.P.; Cassman, K.G. Reversal of rice yield decline in a long-term continuous cropping experiment. *Agron. J.* **2000**, *92*, 633–643.
13. Murgai, R.; Ali, M.; Byerlee, D. Productivity growth and sustainability in post-green revolution agriculture: The case of the Indian and Pakistan Punjab. *World Bank Res. Obs.* **2001**, *16*, 199–218.
14. Faruquee, R. Role of economic policies in protecting the environment: The experience of Pakistan. *Pak. Dev. Rev.* **1996**, *35*, 483–506.
15. Singh, G.B. Fertilizer use and sustainability of indian agriculture. In *Nutrient Management for Sustainable Crop Production in Asia*; Johnston, A.E., Syers, J.K., Eds.; CAB International: Wallingford, 1998; 394 pp.
16. Rukuni, M. Food crisis: The need for an African solution to an African problem. In *Food Security: New Solutions for the Twenty-First Century*; El Obeid, A.E., Johnson, S.R., Jensen, H.H., Smith, L.C., Eds.; Iowa State University Press: Ames, 1999.
17. Barbier, E.B. The economic linkages between rural poverty and land degradation: Some evidence from Africa. *Agr. Ecosyst. Environ.* **2000**, *82*, 355–370.
18. Sanchez, P.A.; Leakey, R.R.B. Land use transformation in Africa: Three determinants for balancing food security with natural resource utilization. *Eur. J. Agron.* **1997**, *7*, 15–23.

Degradation: Farm-Level Economic Implications

Dennis Wichelns

International Water Management Unit, Colombo, Sri Lanka

Ellen Burnes

MWH Americas, Inc., Sacramento, California, U.S.A.

Abstract

The role of soil and its interaction with other inputs can be described using conceptual and empirical production functions, although precise measures of the impacts of soil characteristics on crop yields are not always available. An optimization model is helpful in describing why degradation occurs and in identifying policies that might reduce the extent and severity of soil degradation. We present an economic model of farm-level agricultural production, which we use to examine farm-level decisions that lead to soil degradation and to describe policies that might be helpful in reducing soil degradation.

INTRODUCTION

Soil degradation has been defined as a reduction in soil quality, in response to natural or anthropogenic causes, in a manner that reduces current or potential productivity.^[1–4] The economic implications of degradation are described best by focusing on productivity, rather than on soil quality, which is difficult to quantify and does not account for the interaction of soil characteristics with other inputs, such as labor, capital, and management.^[5] Economists view soil degradation as a reduction in the productivity of soil, for a given set of inputs. As soil productivity declines, more units of other inputs are required to maintain the yield of crop or livestock products, and greater expenditures are required to achieve agricultural objectives.^[6]

A FARM-LEVEL OPTIMIZATION MODEL

An intertemporal optimization model is appropriate because of the following reasons: 1) farmers seek to maximize the present value of the sum of net revenues earned over time; and 2) soil is a durable resource. Production decisions in any year can affect soil productivity in ways that influence crop and livestock choices, input use, and yields in many subsequent years. If the rate of change in soil productivity is small, farmers might not be aware of the relationship between current decisions and long-term profitability. Consequently, they might not account for the cumulative, long-term impacts of soil degradation when choosing production strategies.

The objective function of the farm-level optimization model includes J crop and livestock activities and T time periods. Total revenue and total costs are functions of land, labor, capital, and other inputs:

$$\text{Maximize PVNR} \sum_{t=0}^T \sum_{j=1}^J [P_{jt} Y_{jt}(L_{jt}, B_{jt}, K_{jt}, A_{jt}) - TC_{jt}(L_{jt}, B_{jt}, K_{jt}, A_{jt})] [1 + r]^{-t}$$

where

PVNR is the present value sum of net revenue,
 P_{jt} is the price of crop or livestock product j in year t ,
 Y_{jt} represents the output of crop and livestock products,
 TC_{jt} represents the total costs of production,
 L_{jt} is the amount of land used in each production activity,
 B_{jt} is the amount of labor,
 K_{jt} is the amount of capital,
 A_{jt} represents all other inputs, and
 r is the appropriate discount rate.

We use this framework to analyze the farm-level impacts of reductions in soil productivity on crop yields, input use, and production costs. Our goals are to gain insight regarding why degradation occurs and to identify public policies for reducing the extent and rate of soil degradation.

FARM-LEVEL IMPLICATIONS

The opportunity to generate net revenue motivates farmers to maintain or enhance soil productivity. Key parameters in farm-level decisions include the prices of inputs and outputs and the available technology. The role of soil productivity is implicit in the technology of production, as described by the yield and total cost functions, Y_{jt} and TC_{jt} . Production functions describe how inputs are combined to generate output, whereas cost functions describe the expenditures required to produce a given level of output. Both production and cost functions represent existing technology, which determines how much output can be produced

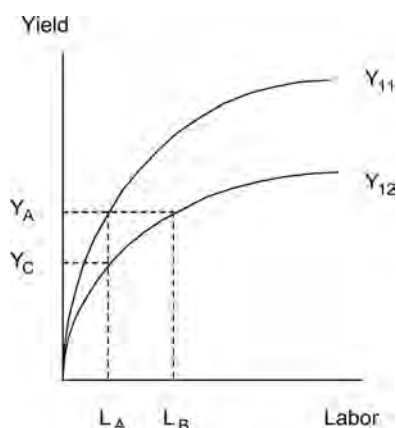


Fig. 1 The impact of soil degradation on the labor required to maintain crop yield.

with a given set of inputs and what the cost will be to generate a desired amount of output.

The production and cost functions are described using subscripts for the crop or livestock activity, j , and the time period, t . The inclusion of the time subscript allows for changes or shifts in the functions over time, due to changes in the productivity of soil and other inputs. If soil productivity declines, greater amounts of labor, fertilizer, or other inputs will be required to achieve the same level of yield that once was obtained with fewer inputs. Hence, a larger expenditure will be required to achieve a given level of output.

A decline in soil productivity causes a downward shift in a crop production function if there are no changes in technology. Fig. 1 depicts a 2-D, conceptual relationship between labor and crop yield. If soil degradation causes the production function to shift downward over time, from Y_{11} to Y_{12} , the labor required to achieve the original yield, Y_A , will increase from L_A to L_B . If labor is not increased, the yield will fall from Y_A to Y_C .

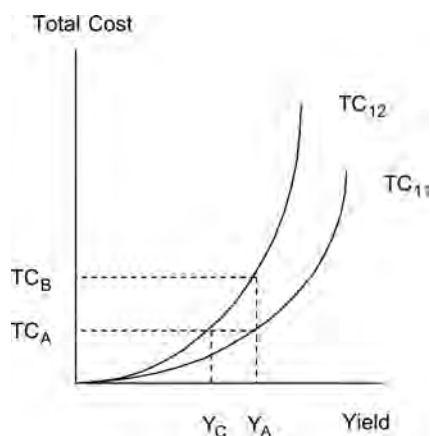


Fig. 2 The impact of soil degradation on the cost of maintaining crop yield.

This effect can be described alternatively using total cost functions (Fig. 2). If soil degradation causes the cost function to shift upward from TC_{11} to TC_{12} , the cost of maintaining production at Y_A will increase from TC_A to TC_B . If expenditures for inputs are not increased, output will decline from Y_A to Y_C .

The value of net revenue will decline with soil degradation, if output is reduced while using the same level of inputs or if inputs are increased to maintain the original level of output. The optimal response to soil degradation likely will involve both a reduction in output and an increase in the use of some inputs. Farmers lacking knowledge, capital, or access to key inputs might be unable to respond optimally to soil degradation. Those farmers will sustain larger reductions in net revenue.

Optimal farm-level strategies regarding soil management and the prevention of soil degradation can be determined by evaluating the potential long-term impacts of decisions regarding production activities and input use. Several conditions are required to ensure that long-term impacts are considered when choosing near-term production strategies. Farmers must 1) understand the potential impacts of their decisions on soil productivity; 2) perceive that they will gain the future benefits of efforts to maintain or enhance soil productivity; and 3) have the financial ability or access to credit required to invest in measures that will generate returns primarily in future.

Farmers often have limited information regarding the impacts of their decisions on soil productivity. The shifts in production and cost functions that result from declining productivity can occur very slowly, such that farmers are not aware of an accumulating problem of soil degradation. In addition, the annual variation in crop yields and input requirements caused by weather events, pest infestations, and other stochastic influences can make it difficult for farmers to observe a downward trend in average productivity because of soil degradation. Public efforts to provide farmers with better information regarding the long-term impacts of production strategies can motivate them to implement optimal soil management programs.

Owners of land have an economic incentive to consider the long-term impacts of decisions because they will retain the incremental benefits generated by their actions. Tenant farmers and users of resources for which secure, long-term property rights are not assigned will lack that incentive in most cases. Incentives for tenant farmers can be provided by including requirements pertaining to soil management practices in their lease agreements and by writing those agreements for long periods.

Many farmers, particularly in developing countries, lack the financial ability or the access to credit required for implementing an optimal soil management program.^[7]

Private owners of farmland might perceive the economic rationale for investing in a subsurface drainage system to prevent salinization or constructing terraces to reduce erosion, but they might not have the funds required to make those investments. Public policies that provide farmers with low-interest loans or cost-sharing opportunities for making desirable investments can motivate the wider implementation of optimal soil management programs.

CONCLUSION

Soil degradation can be described as an economic phenomenon that causes production functions to shift downward and cost functions to shift upward. Greater amounts of complementary inputs are required to maintain the yields of crop and livestock products when soil productivity is reduced because of degradation. An optimization model describes the role of land, labor, and other inputs in agricultural production. Soil degradation is more likely to occur when 1) farmers are not aware of gradual reductions in soil productivity; 2) property rights are neither secure nor well defined or when tenure is of limited duration; and 3) farmers do not have the financial ability to implement optimal soil management programs. Public policies that clarify and enforce the assignment of property rights to farmland and those that modify the prices of selected inputs can motivate farmers to consider the long-term impacts of soil degradation when choosing production strategies.

ACKNOWLEDGMENT

We appreciate the helpful comments of Megumi Nakao and two reviewers regarding earlier versions of this entry (contribution no. TP 03-15 of the California Water Institute).

REFERENCES

1. Lal, R.; Hall, G.F.; Miller, F.P. Soil degradation. I. Basic processes. *Land Degrad. Rehabil.* **1989**, *1*, 51–69.
2. Lal, R. Degradation and resilience of soils. *Philos. Trans. R. Soc. Lond. B.* **1997**, *352*, 997–1010.
3. Stocking, M.; Murnaghan, N. *Handbook for the Field Assessment of Land Degradation*; Earthscan Publications: London, 2001.
4. Lal, R. Cropping systems and soil quality. *J. Crop Prod.* **2003**, *8*, 33–52.
5. Letey, J.; Sojka, R.E.; Upchurch, D.R.; Cassel, D.K.; Olson, K.R.; Payne, W.A.; Petrie, S.E.; Price, G.H.; Reginato, R.J.; Scott, H.D.; Smethurst, P.J.; Triplett, G.B. Deficiencies in the soil quality concept and its application. *J. Soil Water Conserv.* **2003**, *58*, 180–187.
6. Pimentel, D.; Harvey, C.; Resosudarmo, P.; Sinclair, K.; Kurz, D.; McNair, M.; Crist, S.; Shpritz, L.; Fitton, L.; Saffouri, R.; Blair, R. Environmental and economic costs of soil erosion and conservation benefits. *Science* **1995**, *267*, 117–1123.
7. Scherr, S.J. A downward spiral? Research evidence on the relationship between poverty and natural resource degradation. *Food Policy* **2000**, *25*, 479–498.

Degradation: Food Aid Needs in Low-Income Countries

Shahla Shapouri

Markets and Trade Economics Division, U.S. Department of Agriculture—Economic Research Service (USDA-ERS), Washington, District of Columbia, U.S.A.

Stacey Rosen

U.S. Department of Agriculture—Economic Research Service (USDA-ERS), Washington, District of Columbia, U.S.A.

Abstract

Global food production has grown faster than the world's population in the past years. As a result, enough food is available worldwide, so that countries with production shortfalls can import food as long as they have adequate financial resources. Yet, many poor countries and millions of poor people do not have a share in this abundance. They suffer from food insecurity and hunger, which are the conditions associated with low-quality resources and inadequate purchasing power. For these countries, demand for food aid is high and is likely to increase unless steps are taken to implement improved policies and raise productivity. Gains in domestic production will likely stem from improved yields that are, in part, dependent on soil quality.

INTRODUCTION

The Economic Research Service (ERS) estimates that almost 33%—roughly 800 million—of the developing world's population suffer from insufficient food intake.^[1] For 67 lower-income countries, the amount of food needed to provide nutritionally adequate diets was estimated to be 17.1 million tons in 2000 and will increase by 70% to more than 22 million tons by 2010. Sub-Saharan Africa is the most vulnerable region and is projected to account for almost 80% of this “nutritional” gap (of the 67 study countries) in 2010 (Table 1).

The donor countries have showed an interest in providing food assistance to low-income countries; however, there are budget pressures that limit these transfers. The question at hand is what are the factors that are leading to food insecurity in these countries and what kind of changes can be initiated to compensate for the decline in the quality of resources without reducing the benefits of increased productivity.

IMPROVING AGRICULTURAL PRODUCTIVITY

A common characteristic among the food-deficit countries is the large contribution of domestic food production to food consumption, with only a very small share of domestic consumption provided by imports. Agricultural products are also very important sources of foreign exchange earnings to support the growing import needs. Agricultural production growth depends on the availability and quality of

resources. In low-income, food-deficit countries, this means land and labor because of limited use of new technologies.

Constraints in Expanding Agricultural Areas

In many low-income countries, most increases in agricultural output have stemmed from area expansion (Fig. 1).^[2,3] In sub-Saharan Africa, area expansion accounted for more than 80% of the region's grain output growth between 1980 and 1999. This means that yield growth contributed to less than 20% of the growth. In Latin America, area expansion accounted for 68% of the growth in grain production. In Asia, however, the reverse was true; area expansion accounted for less than 5% of the growth in grain output. In other words, gains in yields contributed to nearly all of the growth experienced in the region's grain output.

The long-term prospects for acreage expansion are not bright because, in most countries, a large part of land that could be used for farming is unfit to cultivate without major investment. In Latin America and sub-Saharan Africa, continued expansion of cropland means converting rangeland and forestland to crop production, a process with high economic and environmental costs. According to the estimates of Food and Agriculture Organization, about half of the land that could be used to produce food in sub-Saharan Africa has poor quality soil.^[3] Sub-Saharan Africa has a vast and diverse land area, but the region faces a number of resource constraints (such as

Table 1 Food gaps by region.

	2000 (million tons)	2010 (million tons)
Sub-Saharan Africa	10.70	17.50
Latin America	0.63	0.99
Asia	1.70	3.63
Other	0.17	0.18
Total (67 countries)	13.20	22.30

lack of water) to sustainable agricultural growth.^[4] Some countries, such as Sudan and the Democratic Republic of the Congo, have vast areas of rain-fed land with crop potential, while other countries such as Kenya and Madagascar have already exhausted their high-potential land. Demographic changes are placing increased pressure on land in the region. More than 20% of all vegetative land is degraded because of human causes; however, water and wind erosion account for a majority of the affected hectares. Much of this degraded area is in the Sahel, Sudan, Ethiopia, Somalia, Kenya, and Southern Africa. Historically, farmers adjusted to resource constraints by following several years of planting with several fallow years. However, population pressures have reduced the practice of these sustainable agricultural techniques, leading to declines in land productivity.

Given the constraints to area expansion, changes were made in the assumptions for area growth relative to the base scenario for sub-Saharan Africa, Latin America, and Asia using the structural framework of the ERS food security model. For sub-Saharan Africa and Latin America, we cut the projected area growth rates used for the baseline scenario into half for each country in the two regions. In Asia, we assumed the area to remain constant during the entire projection period under the new scenario, as population pressures will intensify competition for land. In sub-Saharan Africa, production in the baseline scenario was projected to grow at a rate of 2.1% per year; under the reduced area growth scenario, this rate is projected to fall to 1.7%. As a result of the slower production growth,

Table 2 Food gaps under alternative scenarios.

	Baseline (million tons)	Reduced area growth
Sub-Saharan Africa	17.5	25.6
Latin America	1.0	1.6
Asia	3.6	4.7

the region’s nutritional food gap in 2010 jumps more than 40% to more than 25 million tons (Table 2). In Latin America, the decline in production growth is much less significant—from 2.3% to 2.2%. The impact is less in this region because gains in yields are expected to be a more significant contributor to production growth than area expansion. In sub-Saharan Africa, both area and yield were estimated to contribute almost evenly to production gains; therefore, a cut in area growth is going to have a greater impact. This small change in production growth will still have strong implications for the region’s nutrition gap that is projected to rise 40% under this new scenario. In Asia, production growth is projected to slow from 1.7% to 1.6% per year, when area is held constant. This slowdown translates into a 30% increase in the nutrition gap for 2010.

Limited Use of New Technologies

The only option to sustain production growth is by increasing yields. Within the developing world, average regional grain yields are the highest in North Africa and the lowest in sub-Saharan Africa. Yields are highly dependent on the use of improved inputs, particularly fertilizer use. However, yields do not increase commensurately with fertilizer use. A cross-country estimate for the developing countries showed that a 1% increase in fertilizer use results in only a 0.1–0.3% increase in yields.^[4,5] The principal factor that limits yield response to fertilizer use is the inadequate supply of water during the growing season. Although water availability varies considerably across regions, it has been a serious problem in many countries. According to a study by World Bank, the depletion and degradation of water resources have become major problems many low-income countries facing.^[5,6] Within 10 years, if population grows at projected rates, per capita water availability will decrease by an average of 20% in developing countries and by 34% in African countries. The agricultural sector consumes over half the annual freshwater withdrawals in most of the countries and could face greater competing demands from household and industrial uses in the future.

The sparse rainfall that characterizes much of sub-Saharan Africa affects response to and demand for fertilizer.^[7,8] Farmers are reluctant to risk fertilizer use until rain begins to fall because inadequate moisture will fail to dissolve the nutrients in fertilizer (especially nitrogen)

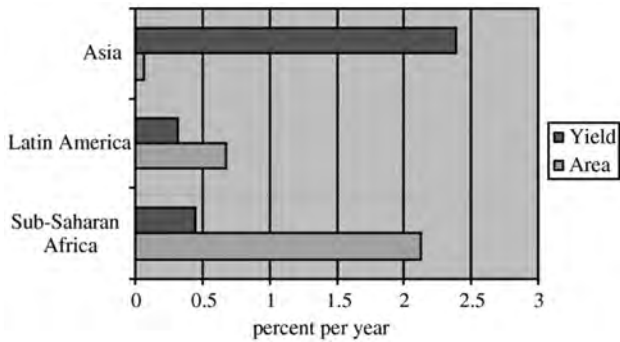


Fig. 1 Sources of grain production growth, 1980–1999.

Table 3 Land use and agricultural productivity.

	Irrigated land (% of crop land)	Arable land (ha per capita)	Fertilizer use (kg per hectare arable land)		Agricultural productivity (agr. value added per agr. worker (1995 dollars))	
	1995–1997	1995–1997	1979–1981	1996–1998	1979–1981	1996–1998
North Africa	13.2	0.24	91	117	1255	1982
Sub-Sahara Africa	4.2	0.25	12	10	244	128
Asia	36 ^a	0.14 ^a	54 ^b	135 ^b	521 ^a	618 ^a
Latin America	13.2	0.17	56	99	1528 ^c	1779 ^c

^aRegional average without Afghanistan, North Korea, and Vietnam.^bRegional average without North Korea.^cRegional average without Bolivia.

and crops can “burn.” Irrigation can make the use of fertilizer profitable and can increase agricultural output. However, in sub-Saharan Africa, only 4.2% of arable land is irrigated (Table 3). This is low even when compared with other developing regions. In Latin America, 13% of arable land is irrigated, and 36% is irrigated in Asia.^[9] The world average is 19%. There is potential for expanding irrigated area in sub-Saharan Africa, but it is costly and requires investment. Increasing the use of fertilizer raises production costs. In many low-income countries, particularly those in sub-Saharan Africa and Latin America, almost all fertilizers are imported, and the lack of adequate foreign exchange constrains the availability.

CONCLUSION

Uneven distribution of the world’s resources means that the poor, low-resource countries are vulnerable to food insecurity. Sub-Saharan Africa, the most food-insecure region, faces the challenge of achieving the following multiple goals: meeting basic human needs, stabilizing population, encouraging economic development, and conserving natural resources for future growth. In most low-income, food-deficit countries, gains in production have been achieved because of area expansion. However, population pressures and poor farming practices that have contributed to soil erosion and nutrient-deficient soils have already pushed farmers to marginal lands. Therefore, the only option for sustainable production growth is to increase yields. So far, most countries in the region are unsuccessful in adopting new technologies to raise crop yields and increase productivity.

REFERENCES

1. Shapouri, S.; Rosen, S. Global food security: Overview. In *Food Security Assessment, International Agricultural and Trade Reports, Situation and Outlook Series*; United States Department of Agriculture, Economic Research Service, 2000; 3–4.
2. Brown, M.; Goldin, I. *The Future of Agriculture: Developing Country Implications*; Development Center of the Organization for Economic Cooperation and Development: Paris, 1992.
3. United Nations Food and Agriculture organization. *Agriculture: Towards 2010*; FAO: Rome, 1993.
4. Ingram, K.; Frisvold, G. Productivity in African agriculture: Sources of growth and stagnation. In *International Agricultural and Trade Reports, Africa and Middle East Situation and Outlook Series WRS-94-3*; United States Department of Agriculture, Economic Research Service: Washington, 1994; 27–31.
5. Cleaver, K.; Schreiber, G. *Reversing the Spiral: The Population, Agricultural, and Environment Nexus in Sub-Saharan Africa*; The World Bank: Washington, 1994; 189–191.
6. Crosson, P.; Anderson, J. *Resources and Global Food Prospects: Supply and Demand for Cereals*, World Bank Technical Paper No. 184; The World Bank: Washington, 1992.
7. Harold, C.; Larson, B.; Scott, L. Fertilizer consumption remains low. In *International Agricultural and Trade Reports, Africa and Middle East Situation and Outlook Series WRS-94-3*; United States Department of Agriculture, Economic Research Service: Washington, 1994; 32–36.
8. Seckler, D.; Gollin, D.; Antoine, P. Agricultural potential of “Mid-Africa.” A technological assessment. In *Agricultural Technology in Sub-Saharan Africa, World Bank Discussion Papers 126*; The World Bank: Washington, 1991.
9. United Nations, Food and Agriculture Organization. Available at <http://apps.fao.org/cgi-bin/nph-dh.pl>.

Degradation: Food Security and Poverty

Pierre Crosson

Senior Fellow and Resident Consultant, Resources for the Future, Frederick, Maryland, U.S.A.

Abstract

While soil degradation does not appear to be a serious threat to world food security, this does not mean that measures to protect the soil are unimportant. On the contrary, they are very important. Continued protection of the soil against the inroads of degradation is essential to maintaining and increasing the global capacity to produce food. Using that capacity to achieve global food security requires an increase in the incomes of the poor worldwide. If that is done on an adequate scale, the problem of global food insecurity will be solved.

INTRODUCTION

Food-insecure people are defined here as those who are considered undernourished by the Food and Agriculture Organization (FAO) of the United Nations. These are people who do not consume enough calories over the course of a year to maintain normal body weight and support light physical activity. According to the FAO, there were approximately 780 million people who, by this definition, were food insecure in 1990.^[1] Almost all were from the less developed countries (LDCs) of Asia, Africa, and Latin America.

POVERTY AND FOOD SECURITY

The consensus among students of world agricultural development is that, where people are food insecure, it is mainly because of poverty. People are too poor to purchase the food needed for adequate nourishment. This means that the food-insecurity problem lies primarily on the demand side of world food markets. Soil degradation reduces the capacity of the land to produce food, which means that soil degradation is on the supply side of world food markets. In principle, it could also be on the demand side because degradation-induced production losses reduce farmers' incomes, and thus their ability to buy food.

This implies that reducing soil degradation would help solve the problem of global food insecurity to the extent that degradation significantly reduces farm income. The investigation of information on soil degradation and its productivity consequences suggests, with some caveats, that the effect of soil degradation on farm income is not significant on a global scale.

FAO data on the number of undernourished people in developing countries show that the 780 million in this category (in 1990) represented 20% of the total population of these countries. This was a marked decline from the 36% of

the LDC population that was food insecure in the late 1960s. It must be noted, however, that this improvement occurred in only Asia and Latin America. In Africa, the number of food-insecure people increased over the 20-year period, both as a percentage of the population and in absolute number.

In the 1990s, per-capita food production and per-capita income increased in Asia and Latin America, so it is likely that the population percentage and the absolute amount of food-insecure people in those areas declined. In Africa, however, the trends in food production and per-capita income continued to be unfavorable in that time frame, suggesting that, in percentage and absolute number, food-insecure people increased further.

These numbers indicate that despite the relatively good performance in Asia and Latin America, the global community continues to confront a serious problem of food insecurity in the LDCs, especially in Africa.

The consensus mentioned previously that poverty is the main reason for food insecurity includes the view that if food-insecure people had sufficient income and the global food trading system were no less robust farmers worldwide could increase food production enough to meet the higher demand at prices only moderately higher than those prevailing. It also was mentioned that because soil degradation is a supply factor in agricultural production, reducing soil degradation would not significantly contribute toward world food security. However, this assertion cannot be accepted on its face; it must be investigated. Specifically, we must review estimates of the extent and productivity consequences of soil degradation worldwide.

SOIL DEGRADATION

It should be noted that most soil scientists and the general public have viewed soil degradation as a serious threat to agricultural production capacity. In 1928, Hugh Hammond

Bennett—regarded as the father of the soil conservation movement in the United States—and a colleague published *Soil Erosion: A National Menace*.^[2] In the United States, this view of erosion was widely shared and persists among many soil scientists and others. The same view was more expressed, with respect both to the United States, and the world as a whole, by Brown and Wolf,^[3] and Pimentel et al.^[4]

The view has held despite the lack of strong scientifically reliable information about the amounts and extent of soil erosion or its productivity impacts, not only in the United States but around the world. In the United States, it was not until the U.S. Soil Conservation Service (an agency of the Department of Agriculture) conducted a comprehensive nationwide survey in 1977 that such information became available. On a global scale, no such data existed, even as late as 1988. Nelson^[5] expressed the consensus after conducting a comprehensive literature review and described worldwide evidence regarding the rate, extent, and severity of soil degradation as “extraordinarily skimpy.” Other students of the problem came to the same conclusion as well.^[6,7]

MEASUREMENTS OF SOIL DEGRADATION

In 1982, 1987, 1992, and 1997, the U.S. Soil Conservation Service (the Natural Resources Conservation Service) conducted nationwide surveys of erosion amounts comparable with those done in 1977. The surveys showed that over the period covered the amounts declined, both in total and per hectare.

In addition, studies done of the productivity consequences of erosion in the United States indicate that in the past they were, and in the future would continue to be, small. (In these studies, productivity is measured as crop output per hectare.) Crosson^[8] found that from 1950 to 1980, erosion in more than 600 counties in the Corn Belt and Great Plains reduced increases in corn and soybean yields by a cumulative 1.5–2%. That is, in 1980, yields of those crops were smaller by those percentages than they would have been without erosion. Crosson also found that erosion in those counties had no statistically significant effect on wheat yields.

Soil scientists at the University of Minnesota developed a Productivity Index model to estimate the future effects of soil erosion on crop yields. They applied the model to 39 million hectares of cropland in the Corn Belt and found that if 1977 rates of erosion were to continue for 100 years, corn yields at the end of the period would be approximately 4% less than without erosion.^[9]

In yet another study, a team of soil scientists, agronomists, and economists at the Department of Agriculture facility in Temple, Texas, developed the Erosion Productivity Impact Calculator, a model designed to estimate the effects of erosion on crop yields.^[10] The team used the

model to estimate the erosion-induced loss of yield in the nation as a whole that would occur if the 1982 crop production and management practices were to continue for 100 years. They found that at the end of that time period yields would be 2–3% lower than without erosion.

In a comprehensive review of 90 microscale studies of erosion-induced productivity losses in the United States and Canada, den Biggelaar et al.^[11] came to a conclusion consistent with that of the three previous studies.

Two studies done in the early 1990s provide the only reasonably reliable global scale estimates of the extent and severity of soil degradation on agricultural land. One of the studies, by a team headed by Roel Oldeman^[12] at the Agricultural University at Wageningen, the Netherlands, was based on a survey of more than 200 worldwide soil specialists. The specialists were asked to classify as degraded or not degraded the soils in their areas that were in permanent or annual crops, permanent pasture, or in forest and woodland. They were then asked to categorize each type of land into four degrees of degradation: light, moderate, strong, and extreme. To translate these estimates to the global scale, Oldeman et al. used FAO data showing that some 8735 million hectares were in the four land uses. Responses from the specialists were used to allocate these hectares into degraded or not degraded; if degraded, they were placed into each of the four degrees of degradation. The results showed that 1965 million of the 8735 million hectares (22%) were degraded. Of the 1965 degraded hectares, 33% were lightly so, 46% moderately, 20% strongly, and 1% severely.

Another global study of productivity effects of soil degradation was done by Dregne and Chou,^[13] specialists in dryland agriculture at Texas Tech University. They estimated the degradation-induced percentage losses of yield on irrigated cropland, rain-fed cropland, and range in dry regions of the world. (Dry regions are those in arid, semi-arid, and dry subhumid agroclimatic zones.) Using FAO data, Dregne and Chou found approximately 5100 million global hectares in the three categories of land use. Drawing on the published literature and other available evidence, as well as their experience in dryland agriculture, Dregne and Chou classified land in irrigated and rain-fed crop production into four categories by severity of degradation-induced losses of productivity. Slightly degraded land was estimated to have lost 0–10% of its undegraded level of productivity. The estimated loss for moderately degraded land was 10–25%, for severely degraded land the loss was 25–50%, and very severely degraded land was estimated to have lost more than 50%. For rangeland, the estimated losses of productivity in each of the four categories were assumed to be somewhat higher (e.g., slightly degraded rangeland was estimated to have lost 0–25% of its undegraded productivity). For each of the three land uses, Dregne and Chou divided the amount in each among the four degree-of-degradation categories.

Crosson^[14] used the data in the Dregne and Chou study to estimate that the weighted average cumulative global loss of soil productivity on the three classes of land use in dry areas was 12%. This is the loss over some period, which Dregne and Chou did not specify, but it must have been averaged over approximately three to four decades. Over three decades, the average annual rate of productivity loss would be 0.4%.

Crosson^[14] combined parts of the Dregne and Chou study with results of the Oldeman et al. study^[12] and calculated the global scale cumulative loss of productivity on land in all agroclimatic zones to be 5%. Oldeman et al. specifically note that their estimate is for the period from 1945 to 1990. Over that 45-year period, the average annual rate of loss would be 0.1%.

In the latter calculation, Crosson assumed that the degradation-induced loss of productivity on the undegraded land in the Oldeman et al. study (78% of the total land) was zero. With so much of the land in an undegraded state, the extent of degradation-induced losses of productivity on *all* the land necessarily would be small.

These results suggest that over the past several decades, the impact of soil degradation on United States and global agricultural productivity has been small. Results that estimate future erosion-induced losses of productivity in the United States suggest that with current rates of erosion, the losses would continue to be small.

CONCLUSION

The conclusion for both the United States and global scales is subject to five important caveats. First, other students of the United States and the global soil erosion issue do not agree with the conclusion;^[3,4] however, it must be noted that neither of these authors has directly confronted the argument made here.

Another caveat is that the data used in the Dregne and Chou and Oldeman et al. studies are subject to wide margins of error. Both sets of authors recognize this and warn against careless use of their data. If and when more reliable data become available, they may contradict these initial findings.

The third caveat, regarding global scale estimates, is that the estimates obscure the fact that in many areas, particularly in the LDCs, there are "hot spots" where soil degradation has been severe—with significant impact on the food security of people in those areas.^[15] In parts of sub-Saharan Africa, e.g., the productivity effects of soil degradation may be particularly severe on a local scale because of shallow depths of topsoil and a high concentration of nutrients in the surface soil layer. In these areas, the erosion-induced nutrient loss is particularly serious because supplies of inorganic fertilizers are either not available or are very expensive. Thus, to say that soil degradation on a global scale does not seem to be a

significant threat to food security does not mean the problem is not serious for the millions of people in particular locations affected by it.

Fourth, with respect to the global estimates, the estimates reflect experience over the past several decades. While soil degradation has not greatly reduced agricultural productivity over this period, this does not guarantee that, in the future, degradation could not increase enough to significantly threaten world food security.

Much will depend on whether land tenure systems in the LDCs move in the direction of providing farmers more secure property rights in the land. The evidence around the world is clear that where these rights are protected, farmers have incentive to invest in soil conservation because they can be confident that they will reap the gains of the investments. Several studies in Asia and Africa suggest that land tenure systems in those areas are slowly shifting toward more secure property rights in the land.^[16–18] If this trend continues, it will have much impact on the size of the future threat of soil degradation to world food security.

Finally, while soil degradation does not presently appear to be a serious threat to world food security, this does not mean that measures to protect the soil are unimportant. On the contrary, they are very important. For most farmers, land is by far their most important asset. The apparently low cumulative and present rates for erosion-induced losses of soil productivity are strong indirect evidence that farmers worldwide recognize this and have taken measures to protect their land against erosion damages. If this practice continues, soil degradation will not likely threaten future global food security.

Continued protection of the soil against the inroads of degradation is essential to maintaining and increasing the global capacity to produce food. Using that capacity to achieve global food security requires an increase in the incomes of the poor worldwide. If that is done on an adequate scale, the problem of global food insecurity will be solved.

REFERENCES

1. FAO. *World Agriculture: Toward 2000*; Food and Agriculture Organization: Rome, 1995.
2. Bennett, H.; Chapline, W. *Soil Erosion: A National Menace*; Circular Number 13; Soil Conservation Service, U.S. Department of Agriculture, Government Printing Office: Washington, 1928.
3. Brown, L.; Wolf, E. *Soil Erosion: Quiet Crisis in the World Economy*; Worldwatch Paper 60; Worldwatch Institute: Washington, 1984.
4. Pimentel, D.; Harvey, C.; Resosudarmo, P.; Sinclair, K.; Kurz, D.; McNair, M.; Crist, S.; Shpritz, L.; Fitton, L.; Saffouri, R.; Blair, R. Environmental and economic costs of soil erosion and conservation benefits. *Science* **1995**, *267*, 1117–1123.

5. Nelson, R. *Dryland Management: The Soil Erosion Problem*; Environment Department Working Paper No. 8; The World Bank: Washington, 1988.
6. Dregne, H. Desertification of drylands. In *Challenges in Dryland Agriculture: A Global Perspective*, Proceedings of the International Conference on Dryland Agriculture; Unger, P., Sneed, T., Jordan, W., Jensen, R., Eds.; Texas Agricultural Experiment Station: Bushland/Amarillo, 1988; 678–680.
7. El-Swaify, S.; Dangler, E.; Armstrong, C. *Soil Erosion by Water in the Tropics*; University of Hawaii Press: Honolulu, 1982.
8. Crosson, P. *Productivity Consequences of Cropland Erosion in the United States*; Resources for the Future: Washington, 1983.
9. Pierce, F.; Dowdy, R.; Larson, W.; Graham, W. Soil erosion in the corn belt: An assessment of erosion's long-term effect. *J. Soil Water Conserv.* **1984**, *39*, 131–136.
10. U.S. Department of Agriculture. *The Second RCA Appraisal: Soil, Water, and Related Resources on Non-federal Land in the United States*; U.S. Department of Agriculture: Washington, 1989.
11. den Biggelaar, C.; Lal, R.; Weibe, K.; Breneman, V. Impact of soil erosion on crop yields in North America. *Adv. Agron.* **2001**, *72*, 1–52.
12. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *World Map of the Status of Human-Induced Soil Degradation: An Explanatory Note*; International Soil Information and Reference Center: Wageningen; United Nations Environment Programme: Nairobi, 1990.
13. Dregne, H.; Chou, N.T. Global desertification dimensions and costs. In *Degradation and Restoration of Arid Lands*; Dregne, H., Ed.; Texas Tech University: Lubbock, 1992; 249–382.
14. Crosson, P. *Soil Erosion and Its On-Farm Productivity Consequences: What Do We Know?* Resources for the Future: Washington, 1995.
15. Sheer, S.; Yadev, S. *Land Degradation in the Developing World: Implications for Food and the Environment*; International Food Policy Research Institute: Washington, 1996.
16. English, J.; Tiffen, M.; Mortimore, M. *Land Resource Management in Machakos District, Kenya*; World Bank: Washington, 1994.
17. Pingali, P. Institutional and environmental constraints to agricultural intensification. *Popul. Dev. Rev.* **1989**, *15*, 243–260.
18. Migot-Adholla, S.; Hazell, P.; Blarel, B.; Place, F. Indigenous land rights systems in sub-Saharan Africa: A constraint on productivity? *World Bank Econ. Rev.* **1991**, *5* (1), 155–175.

Degradation: Greenhouse Effect

Eddy De Pauw

International Center for Agricultural Research in the Dry Areas (ICARDA), Aleppo, Syria

Michael A. Zoebis

German Agency for International Cooperation, Tashkent, Uzbekistan

Abstract

Soil degradation refers to a decline of soil quality that reduces the soil's productive capacity, sustainability, resistance to degradation, or the biodiversity of the supported natural ecosystems. Soil degradation is a global phenomenon, and its most important expressions are water and wind erosion, fertility decline, salinization, and waterlogging, or lowering of the groundwater table. Being both a cause and an effect of rural poverty, soil degradation is one of the most serious threats to the prosperity of rural populations worldwide and is accelerated by non-sustainable land management induced by rural poverty.

INTRODUCTION

The greenhouse effect refers to the trapping of heat in the lower levels of the atmosphere. The atmosphere allows a large percentage of the rays of visible light from the sun to reach the earth's surface and heat it. Part of this energy is reradiated by the earth's surface in the form of long-wave infrared radiation, much of which is absorbed in the atmosphere by water vapor and greenhouse gases, in particular carbon dioxide (CO₂), but also other trace gases, notably methane (CH₄), nitrous oxide (N₂O), and chlorofluorocarbons. This process of heat-trapping causes the earth's surface and lower atmospheric layers to warm to higher temperatures than would otherwise be the case. In fact, this natural greenhouse effect makes life on earth possible. Atmospheric concentrations of greenhouse gases have increased because of human activities, primarily the combustion of fossil fuels (coal, oil, and gas), deforestation, and agricultural practices. Since the beginning of the pre-industrial era around 1750, CO₂ has risen by nearly 30%, CH₄ by more than a factor of two, and N₂O by about 15%. Their concentrations are higher than at any time during the last 420,000 years. The human-induced greenhouse gas emissions are responsible for the rise of average global temperature of about 0.7°C over the last years, with the most rapid rise occurring since the mid-1970s. If the greenhouse gas emission trends continue, an increase in global mean surface temperature in the range of 1.5–6°C is expected by 2100^[1] (<http://www.ipcc.ch/press/sp-cop6.htm>). In addition, there is evidence that precipitation patterns are changing, sea level is increasing, glaciers are retreating worldwide, Arctic Sea ice is thinning, and the incidence of extreme weather events is increasing in some parts of the world.

The overwhelming majority of scientific experts, while recognizing that scientific uncertainties exist, believe

that global warming is at least partially linked to the increased concentration of greenhouse gases by human activities and that climate change is inevitable. This will have potentially major repercussions on water resources, agriculture, forestry, fisheries and human settlements, ecological systems, and human health with the developing countries being the most vulnerable^[1] (<http://www.ipcc.ch/press/sp-cop6.htm>).

LINKAGES BETWEEN CARBON, GREENHOUSE EFFECT, AND SOIL DEGRADATION

Soil degradation and the greenhouse effect are both linked to human activities and are likely to interact in the context of global climate change. The expected shift in climate patterns, which will involve changes in both temperature and rainfall patterns, has the potential either to accelerate degradational trends in regions where they already prevail or to initiate certain types of degradation in new areas where they do not exist. On the other hand, current or future soil degradation is likely to enhance the greenhouse effect through a reduced ability of the soil to retain carbon.

The global carbon cycle, showing the major reservoirs and flows of carbon in gigatonnes of carbon (GtC; 10¹² kg) per year, is outlined in Fig. 1. The main stores or “sinks” of carbon are the oceans, the lithosphere with geological deposits containing fossil fuels, the terrestrial ecosystems, composed of living biomass and soil, and the atmosphere. Of these, the oceanic carbon reservoir is the largest but least reactive, and the atmospheric reservoir, although the smallest, is the most critical one since it drives global warming. With the exception of the fossil fuel reservoir, each reservoir at times behaves as either a source or a sink of carbon, and it is the rate of exchange of carbon between different

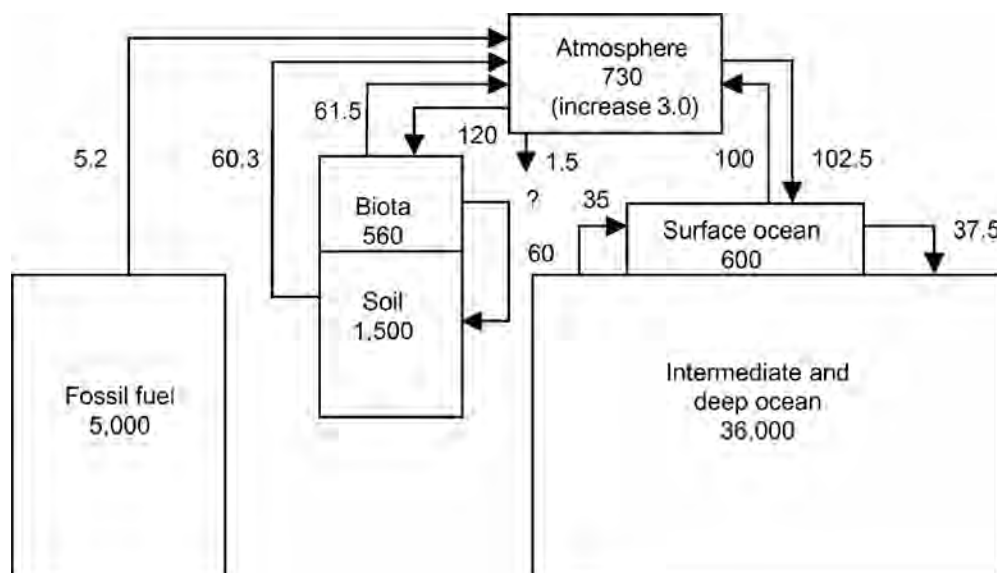


Fig. 1 The global carbon cycle in about 1980.

Source: From Rosenzweig & Hillel.^[2]

reservoirs that determines whether they act as net sinks or sources of carbon. Fig. 1 represents an estimate and refers to the situation of fossil fuel use in about 1980. An estimate of carbon flows and reservoir sizes around 1995, based on the data provided by the Intergovernmental Panel on Climate Change (IPCC), is given in Table 1 and compared with the estimates of 1980.

Table 1 shows that the terrestrial ecosystems have both an important carbon source and sink role. They act as a source of atmospheric carbon through land use change and tropical deforestation, at the same time they act as an important sink through the regrowth of forests in temperate regions; 1.5–1.7 GtCyr⁻¹ is not accounted for. The “missing” carbon is probably going into the terrestrial biosphere in the northern hemisphere where it contributes to an increasing capture of carbon in terrestrial ecosystems through the CO₂ fertilization effect.^[3]

Within the terrestrial ecosystems, soils constitute the largest reservoir of carbon, which is retained in the form of soil organic matter. All soil types of the world have an

inherent capacity to store organic carbon, which is determined by soil temperature and moisture regime, internal drainage, textural composition, and clay mineralogy. World soils show great variation in soil organic matter content, with values of 4–6% (weight basis) in the most fertile to less than 0.5% in others.^[5]

Fig. 1 shows that soils are a net source of carbon to the atmosphere. This is attributed to the massive land use change resulting from the conversion of forests and other natural ecosystems into farmland. From a terrestrial carbon balance model, Esser^[6] estimated for the period 1860–1980 the net decline of the carbon pool in plant biomass at 22 Gt and in the soil at 5 Gt. If an adjustment was made for increased carbon mineralization resulting from the global temperature increase, the loss of carbon in the soil reservoir would have increased to 12 Gt.

Apart from land use change, soil degradation has been considered a major contributing factor to the loss of soil carbon to the atmosphere and therefore to an increase in the greenhouse effect. The main reason is that soil degradation, through the processes of erosion, salinization, and fertility decline affects the productive capacity of land, and as a result, less carbon is returned to the soil. In fact, due to degradation many soils throughout the world do not achieve their potential carbon-carrying capacity. This is particularly the case in most arid and semiarid regions of the developing world, where overgrazing, fuel wood collection, and opportunity cropping have seriously depleted the limited stock of organic matter. The resulting erosion, soil structural degradation, and fertility loss have often all but prevented the reestablishment of the natural vegetation, and in order to regain the original productive capacity, expensive reclamation or conversion into protected lands is required in huge parts of the dry areas.

Also in the (sub)humid tropics, particularly in Africa, soil carbon stocks have been much depleted due to a

Table 1 Global carbon flux budget.

Carbon flows	1980	1995
Annual atmospheric increase of CO ₂	3.0	3.4
<i>Sources</i>		
Terrestrial ecosystems	1.8	2.7
Fossil fuels	5.2	6.4
<i>Sinks</i>		
Terrestrial ecosystems		2.0
Oceans	2.5	2.0
“Missing”	1.5	1.7

Note: Units are GtC (gigatonne carbon, 10¹² kg).

Source: From CAST^[3] and IPCC.^[4]

generally negative nutrient balance resulting from inadequate fertilizer use and reductions of fallow periods in shifting cultivation systems.

The scale of soil degradation worldwide is certainly sufficient to have impact on the global terrestrial carbon stocks. Oldeman et al.^[7] estimated that 38% of the agricultural land, 21% of the permanent pastures, and 18% of the forests and woodlands worldwide are affected by one form or another form of soil degradation. However, an accurate assessment of the contribution of soil degradation to the greenhouse effect is difficult because available inventories of the actual extent of different soil types, their carbon-carrying capacities in different ecoregions, the extent of various types of land degradation, and the carbon contents of degraded soils are inadequate.

In addition, some of the degradation observed does not necessarily lead to carbon loss. In the case of erosion, carbon is transported from one place to another without involving an increase in mineralization. There is also evidence from the geological past that an increased atmospheric CO₂, coupled with warmer than current temperatures, higher rainfall, and increased biological activity—the scenario predicted by global change—could lead to higher concentrations of reactive bicarbonate (HCO₃⁻) in soil solutions, producing more deeply weathered soil profiles rather than atmospheric carbon.^[8]

Global warming is also likely to increase the risk of soil degradation. In its assessment of regional vulnerability to climate change,^[9] the IPCC anticipates an increase of global mean precipitation and the frequency of intensive rainfall, entailing an increased risk of soil erosion by water and of soil acidification in humid areas. However, arid and semiarid regions in Southern and Northern Africa, Southern Europe, the Middle East, and parts of Latin America and Australia are expected to become even drier. Such areas would then become affected by more frequent drought and associated wind erosion, salinization, and carbon mineralization. Most regions of the world are expected to experience an increase in floods because of the projected increase in heavy precipitation events. Whereas the IPCC study outlines likely global trends and can be used for policy guidance at regional level, a quantified assessment of the impact of global warming at national or subnational level is highly speculative owing to major uncertainties about the geographical location and magnitude of the expected changes.

Whether the soil ultimately serves as a source or a sink of carbon very much depends on the way it is managed. Even agricultural land is capable of storing more carbon if appropriate land management practices are adopted and thus to reclaim some or all of the carbon lost in the transformation from natural ecosystems. In this respect, the potential to increase the carbon sink is particularly high in the dry areas, because their carbon content is far below their potential and the size of desertified and degraded lands worldwide, estimated at some 2 billion hectares.^[3] Soil management principles advocated to increase carbon

sequestration include the increase of the total and subsoil organic matter content, soil microaggregation, and soil biodiversity.^[10] Among the technological options proposed are conservation tillage practices, crop residue management, shelterbelts and windbreaks, intercropping, optimal crop rotations, winter cover crops, fertility improvement through green manuring, fallowing, precision farming, and improvement in the soil moisture regime through irrigation and salinity management.^[2,10]

Generally speaking, land use practices designed to prevent or remedy soil degradation will contribute to carbon sequestration. Given the high spatial variability of soils, which will persist under global warming, it is essential that the proposed principles are fine-tuned for specific soils and ecoregions, and the practices adapted to local conditions.

REFERENCES

1. Watson, R.T. *Presentation of the Chair Intergovernmental Panel on Climate Change at the 6th Conference of Parties to the United Nations Framework Convention on Climate Change*. Available at: <http://www.ipcc.ch/press/spcop6.htm> (accessed November 2000).
2. Rosenzweig, C.; Hillel, D. Soils and global climate change: Challenges and opportunities. *Soil Sci.* **2000**, *165* (1), 47–56.
3. CAST. *Storing Carbon in Agricultural Soils to Help Mitigate Global Warming*; Issue Paper 14, Apr 2000; Council for Agricultural Science and Technology: Ames, 2000.
4. IPCC. *Climate Change 1995: The Science of Climate Change*; Working Group 1; Cambridge University Press: New York, 1996.
5. Lal, R.; Logan, T.J. Agricultural activities and greenhouse gas emissions from soils of the tropics. In *Soil Management and Greenhouse Effect. Advances in Soil Science*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1995; 293–307.
6. Esser, G. Modelling global terrestrial sources and sinks of CO₂ with special reference to soil organic matter. In *Soils and the Greenhouse Effect*; Bouwman, A.F., Ed.; Wiley: Chichester, 1990; 247–262.
7. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *World Map of The Status of Human-Induced Soil Degradation: An Explanatory Note*; 3 maps. ISRIC/UNEP: Wageningen, 1990; 27 pp.
8. Bird, M.; Fyfe, B.; Chivas, A.; Longstaffe, F. Deep weathering at extra-tropical latitudes: a response to increased atmospheric CO₂. In *Soils and the Greenhouse Effect*; Bouwman, A.F., Ed.; Wiley: Chichester, 1990; 383–389.
9. Watson, R.T.; Zinyawera, M.C.; Moss, R.H.; Dokken, D. *The Regional Impacts of Climate Change. An Assessment of Vulnerability*; Special report of IPCC Working Group II. Intergovernmental Panel on Climate Change, ISBN 92-9169-110-0, 1997.
10. Lal, R.; Kimble, J.; Stewart, B.A. Towards soil management for mitigating the greenhouse effect. In *Soil Management and Greenhouse Effect*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; Advances in Soil Science; CRC Press: Boca Raton, 1995; 373–381.

Degradation: Mapping by Normalized Difference Vegetation Index (NDVI)

Zhanquo Bai

World Soil Information, International Soil Reference and Information Centre (ISRIC), Wageningen, the Netherlands

David Dent

Center for Sustainability Studies, Lund University, Lund, Sweden

Abstract

The title of an article in *Advances in Space Science*, *The exciting and totally unexpected success of AVHRR in applications for which it was never intended*, published in 2001 says it all. The advanced, very high resolution radiometer carried by National Oceanic and Atmospheric Administration satellites is actually of low resolution, but its large field of vision and daily global coverage make it ideal for global monitoring. And, by chance, the ratio of red to near-infrared radiation [normalized difference vegetation index (NDVI)] measured using the radiometer is a good indicator of vegetation dynamics. The NDVI trends were used in 2008 for the first global assessment of land degradation to be based on consistent physical measurements. The results contradicted received wisdom and met with the usual public reaction to a new truth: “It’s not true”; then, “It’s against scripture.” The final accolade, “We knew it all along,” has been a long time in coming because NDVI can only be a proxy for land degradation and it is hard to confirm in the field. But no one has come up with an alternative. More sensors, e.g., the Moderate Resolution Imaging Spectroradiometer operating since the year of 2000, yield more detailed, more direct measures of net primary productivity but cannot match the 30 years of NDVI data that reveal significant changes in the trends of land degradation.

INTRODUCTION

The land provides 95% of our food and clothing as well as timber, biofuel, freshwater, and environmental services that underpin our economy and society, and it is taken for granted. Between 1965 and 1980, the *green revolution* increased crop yields threefold; for a generation, global food production outpaced population growth, and political attention turned elsewhere. But the end of the era of cheap food and fuel has concentrated minds, again, on food security. So, policy makers face all the old challenges writ large:

- *Burgeoning demand:* By 2050, 70% additional food will be required than that at present, which is double in the developing countries.
- *We are not making any more land:* The area under cereals peaked in 1981, arable in 1991, while the period 1981–2011 has witnessed the degradation of one-quarter of the land surface, and tracts of the best land are being lost to urban development and infrastructure.
- *Food system is unsustainable:* The green revolution depended on cheap fuel, fertilizer, and irrigation applied to new, responsive crop varieties. Fuel and fertilizer are no longer cheap, water resources are overcommitted, and yields have leveled off—in some places, they are declining and there is compelling evidence that present land

use is driving land degradation, and climate change is leading to more extreme weather and a rising sea level that threatens prime farmland.

A U.K. All-Party Parliamentary Inquiry in 2010 predicted “a perfect storm” from the collision of ever-growing demand for food, energy, and freshwater; the stresses of climate change and land degradation; the destabilization of governments that cannot provide their people’s basic needs; and the migration from poor countries to those better endowed. And there are no charts, i.e., from 1981 to 2011, knowledge of the land and capacity to face land resource issues have atrophied.^[1]

Estimates of the extent and severity of land degradation, its costs, and its impact on food security and the environment are uncertain, so it has rarely been at the top of national policy agendas. The exception was the United States’ response to the Dust Bowl in the 1920s and 1930s; the United States continues to reap the benefits of that investment, but, in general, the investments required to arrest land degradation are not being made—they are not even known. Uncertainty stems from the lack of a well-accepted definition of land degradation, the kinds of degradation accounted for, uncertainties in the underlying data, and questionable science. Whereas some signs of land degradation are obvious—landslides, gullies cutting through

farmland, dust storms, silted-up rivers, and salinity—others, like loss of soil organic matter and biodiversity, are invisible, and it is hard to measure all these effects consistently. Questions that need to be answered in a scientifically justifiable way include the following: is land degradation a global issue or a collection of local problems? Which are the hardest hit regions? Is it mainly a problem of drylands? Is it associated mainly with farming? Is it related to population pressure or poverty?

THE NDVI DATA AND APPLICATIONS

Surveys of natural resources have always made good use of new technology originally developed for other purposes. The title of an article in *Advances in Space Science* says it all, *The exciting and totally unexpected success of AVHRR in applications for which it was never intended*.^[2] The advanced very high resolution radiometer (AVHRR) carried by U.S. National Oceanic and Atmospheric Administration (NOAA) satellites is actually of low resolution (4 km) compared with the Landsat satellite data already being collected at the time, but its large field of vision and daily global coverage make it ideal for global monitoring. And by chance, the ratio of red (RED) to near-infrared (NIR) radiation measured by the radiometer, the normalized difference vegetation index (NDVI) [$NDVI = (NIR - RED)/(NIR + RED)$], is a good indicator of vegetation dynamics, reflecting leaf area index, the fraction of photosynthetically active radiation (PAR) absorbed by vegetation,^[3,4] and net primary productivity (NPP).^[5] It has been used to estimate vegetation change as an index^[6] or as one input to dynamic vegetation models.^[7,8] These applications are made possible by consistent time series data at spatial resolutions from 20 m to 8 km. The Global Inventory Modeling and Mapping Studies' (GIMMS) data set now comprises more than 30 years' data corrected for instrument calibration, variations in solar angle and view zenith angle, and volcanic dust. Cloud and haze effects are minimized by taking the highest fortnightly value within the composite 8 km² blocks of pixels.^[9,10] Newer sensors that provide subkilometer resolution include Satellite Pour l'Observation de la Terre Végétation, operating since 1998 with subkilometer definition,^[11] and Moderate Resolution Imaging Spectroradiometer (MODIS) carried by NOAA satellites, Terra and Aqua satellites since 2000 and 2002, respectively, which has even higher resolution and independent measurement of the fraction of PAR absorbed (f_{PAR})—a more direct measure of NPP than the estimation via NDVI.^[12] Based on GIMMS data, Fig. 1 depicts the mean global greenness that has increased by 3.8% over a 30-year period ($P < 0.001$; Fig. 2), thanks to atmospheric fertilization.

Land degradation may be defined as a long-term loss of ecosystem function and productivity caused by disturbances from which the land cannot recover unaided. This may be assessed in terms of NDVI, but a decreasing trend does not necessarily mean land degradation; biomass

depends on climate, land use and management, large-scale ecosystem disturbances such as fires, and the global increase in nitrate deposition and atmospheric carbon dioxide. To interpret NDVI trends in terms of land degradation, we have to eliminate false alarms. This can be done for climate, for which a century's consistent data are available, but there are no global time series for land use, which must be examined case by case. And there is an issue of credibility. Translating satellite measurements of reflected solar radiation into the information that policy makers want (loss of production and environmental services, action required to arrest these losses, and the economic and political payback for this action) requires leaps of deduction—some might say imagination.

Global Assessment of Land Degradation

Under the Food and Agriculture Organization (FAO) of the United Nations program Land Degradation in Drylands, the Global Assessment of Land Degradation and Improvement (GLADA) assessed global land degradation by the analysis of trends of climate-adjusted NDVI making use of GIMMS data up to 2003.^[13] Drought effects are screened using *rain use efficiency* (RUE) estimated from the ratio of the annual sum NDVI and annual rainfall. Areas where biomass productivity depends on rainfall variability were identified as those with a positive relationship between NDVI and rainfall. In these areas, the below-normal rainfall is reflected in below-normal NDVI and, usually, increased RUE; when there is decreasing NDVI but steady or increasing RUE, loss of productivity is attributed to drought and these areas are masked. When both NDVI and RUE decline, something else is happening, and these areas are included in the next stage of analysis together with the areas where production is not limited by rainfall. Similarly, *energy use efficiency*, the ratio of NDVI, and accumulated temperature are used to screen trends driven by rising temperatures. All data are expressed at 90% probability; to provide a more tangible measure of land degradation, NDVI was translated into NPP by correlation with MODIS NPP data for the overlapping period from 2000.

The picture revealed was against the received wisdom that reckoned degradation to be worst in the Sahel, the Amazon rain forest, and more generally in drylands; in fact, most of the usual suspects showed improvement. Fig. 3 shows an updated analysis using the GIMMS 3G data set for 1981–2012. The picture differs from the earlier GLADA analysis—not just because of the longer run of data but because of the changes in GIMMS processing to avoid stationarity bias in between-instrument calibration of AVHRR measurements.

The new analysis provides unambiguous answers to most of our policy questions:

- Land degradation is a global issue, with 22% of the land degrading over a 30-year period, representing an NPP

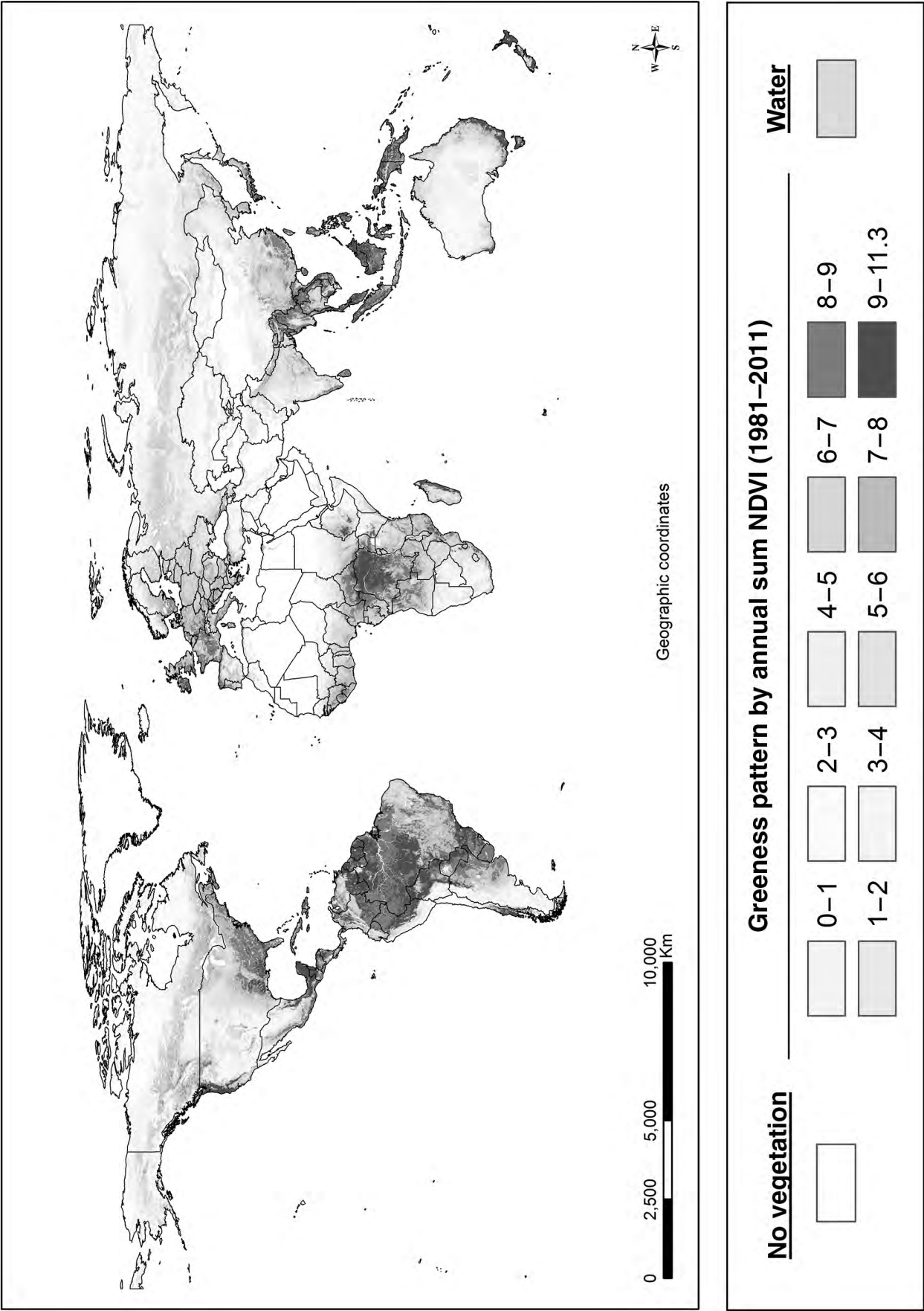


Fig. 1 Global greenness by annual sum NDVI (1981–2011).

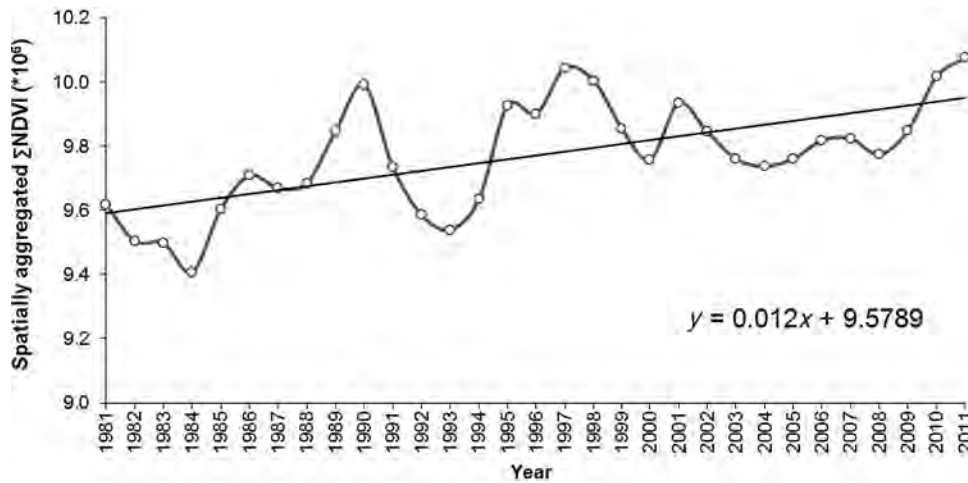


Fig. 2 Trend of global aggregated annual sum NDVI (1981–2011).

loss of about 150 million tons but a loss of soil organic carbon orders of magnitude more. The hardest hit areas are Africa, south of the equator, with an arm of degradation extending northward to the Ethiopian highlands and the Nile provinces of Sudan; the Gran Chaco, Pampas, and Patagonia; southeast Asia; the grain belt from Ukraine eastward through south Russia to Kazakhstan; the Russian far east and Northeast China; swaths of high-latitude forest; and the western slopes of the Great Dividing Range in Australia.

- All kinds of land use are afflicted. Cropland comprises 12.6% of the global land area but makes up 15.1% of the total degrading land; grassland occupies 28.8% of the land but constitutes 41.5% of degraded land; and forest is also overrepresented, comprising 23.4% of the land area, but 37% of the degrading area. At the same time, 14% of the land registered increasing productivity.
- The comparison of rural population density with land degradation shows no simple pattern, taking infant mortality and the percentage of young children who are underweight as proxies for poverty, and there is some correlation but a more rigorous analysis is called for.

(77.4% showed no change or was barren). For the country as a whole, drought explains 15% of negative trends; increased rainfall explains 20% and increased temperature explains 38% of the positive trends; what remains may be attributed in some measure to land use and management. But linear trends analysis is a blunt instrument. [Fig. 5](#) illustrates the NDVI trends for five southern provinces that exhibit general improvement, i.e., at the beginning of the time series, the trends showed a cyclical decline that was reversed after about 1995 by the grain-for-green initiative.^[15]

Making allowance for terrain, soil, and land use

We might expect resilience against land degradation to depend on terrain, soil, land use, and management. The Shuttle Radar Topography Mission digital elevation model provides almost global topography at a horizontal resolution of 30 m, but there is no comparable soil information. Regional data are provided by soil and terrain (SOTER) compilations of various surveys in a common format. [Fig. 6](#) depicts the 30-year trend in China with regard to climate-adjusted NDVI by 1:1 million-scale SOTER units. If SOTER units were homogeneous, this procedure would eliminate SOTER effects, but the mapping units are by no means pure—SOTER are normalized only to the extent that differences within mapping units are less than those between units. SOTER residuals ([Fig. 7](#)) reveal the relative departure of each pixel from the trend of its mapping unit, revealing patterns at the landscape scale—where most management decisions are taken. Differences in response to management relate directly to local conditions of SOTER and highlight black spots where production has been declining relative to the local norm and, as a corollary, bright spots that are doing better.

The analysis of land degradation in terms of land use and management is more problematic because no two consecutive land use surveys have used the same classification. In China, comparing the extent of urban areas with areas of

Changing trends

The extended data set reveals some dramatic turnabouts. For instance, in China^[14] ([Fig. 4A and B](#)), there has been persistent and expanding degradation in Tibet and the southwestern provinces, a dramatic increase in degradation across the northeast, and a loss of impetus in many intensively farmed areas despite the increasing application of synthetic fertilizer (from 7 million tons in 1977 to more than 58 million tons in 2012). Over the period 1981–1996, degradation afflicted 1.8% of the country, but 17.5% was improving (80.7% showed no significant change or was barren); between 1996 and 2011, 12.6% of the land was degrading and only 10% showed improvement

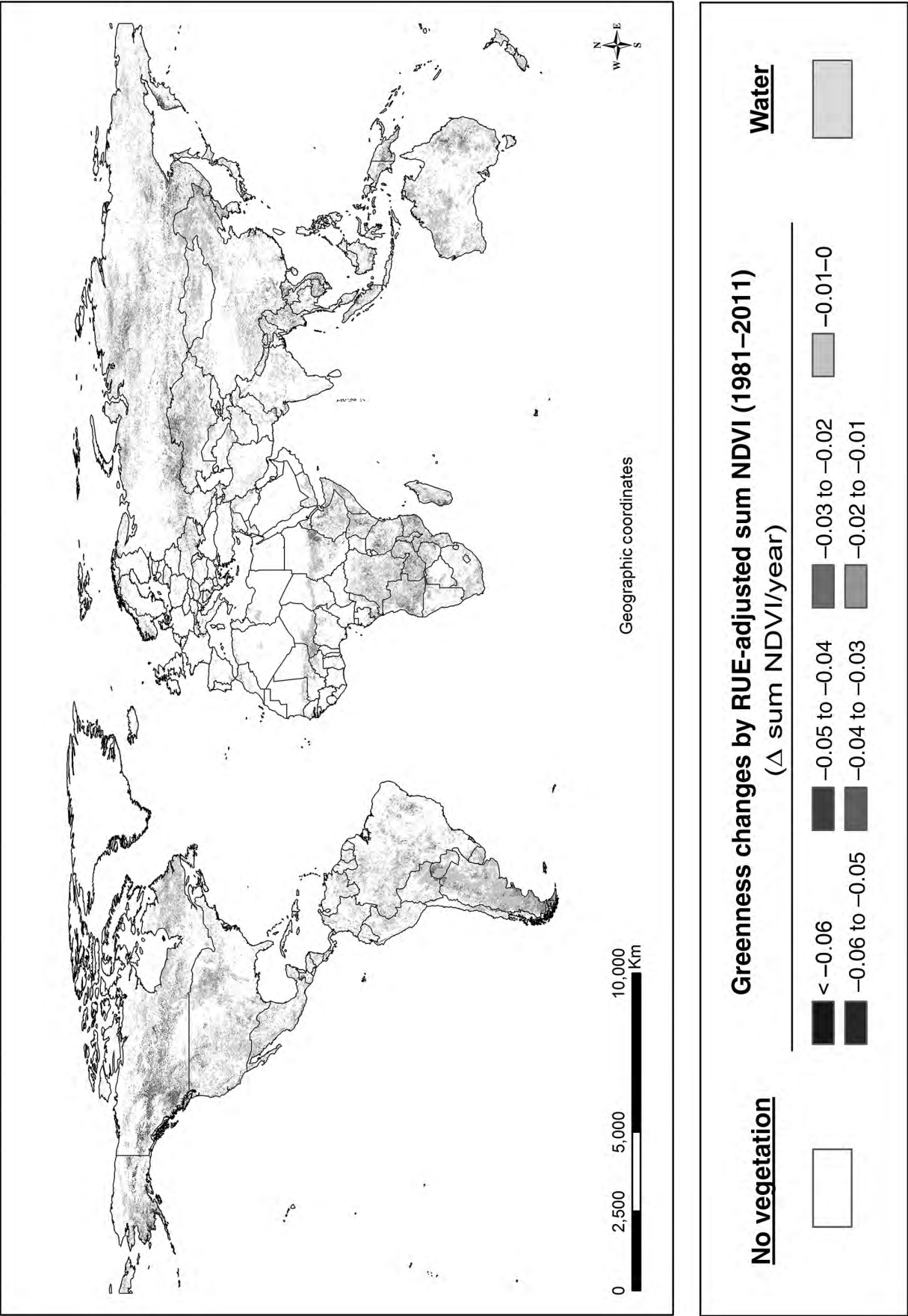
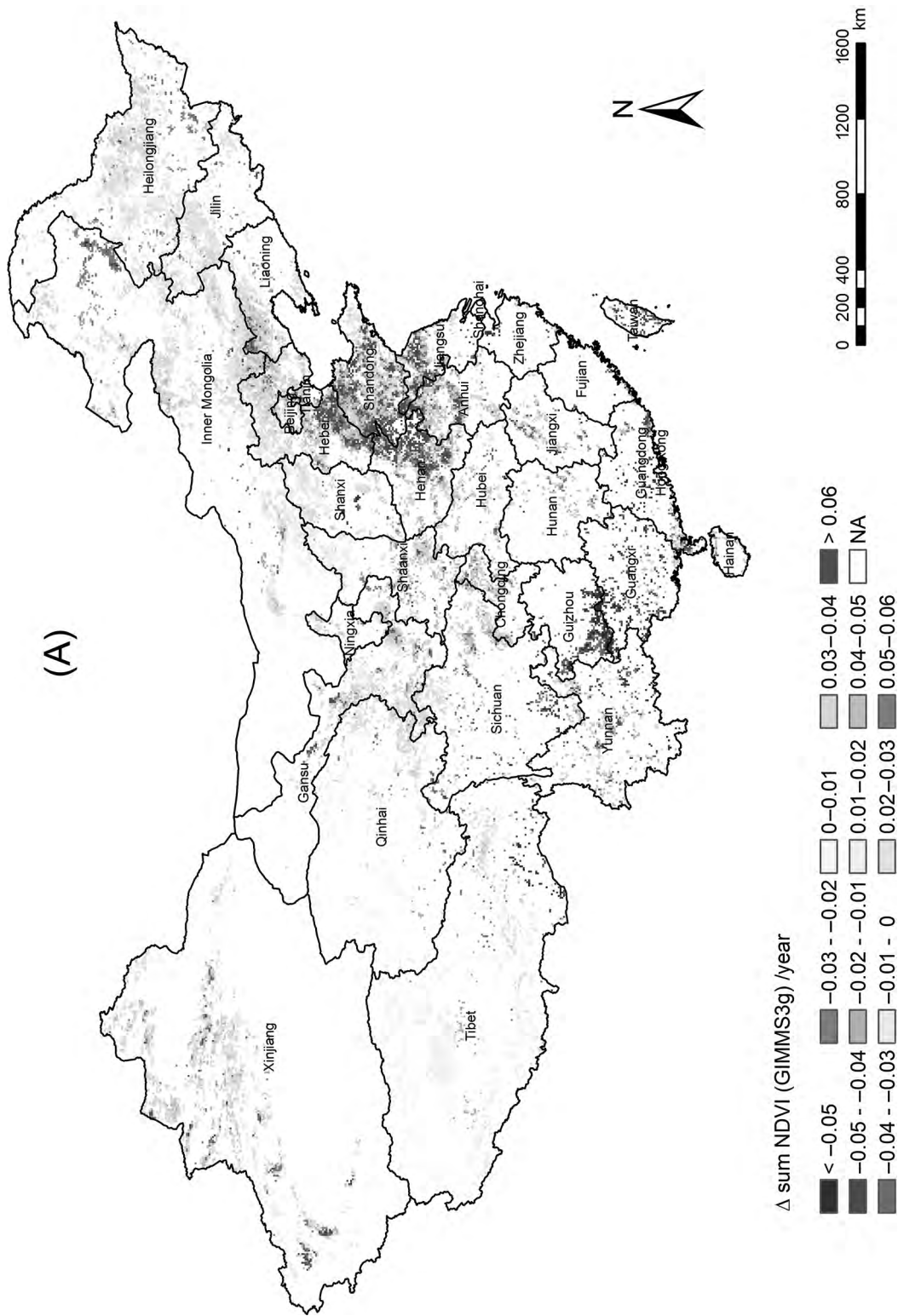


Fig. 3 Global negative trend in RUE-adjusted NDVI (1981–2011).



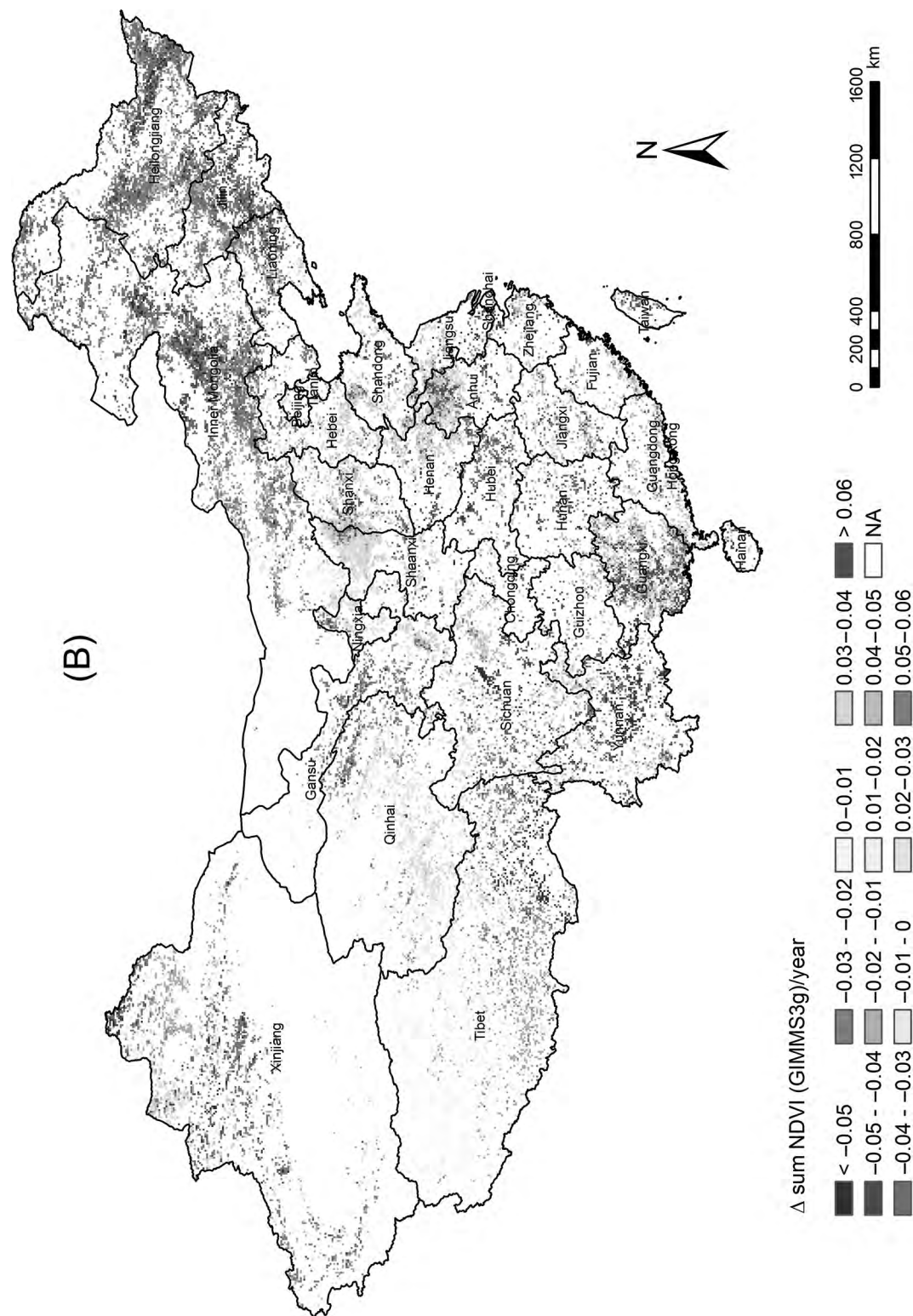


Fig. 4 China: changes in annual sum NDVI – (A) 1981–1996 and (B) 1996–2011.

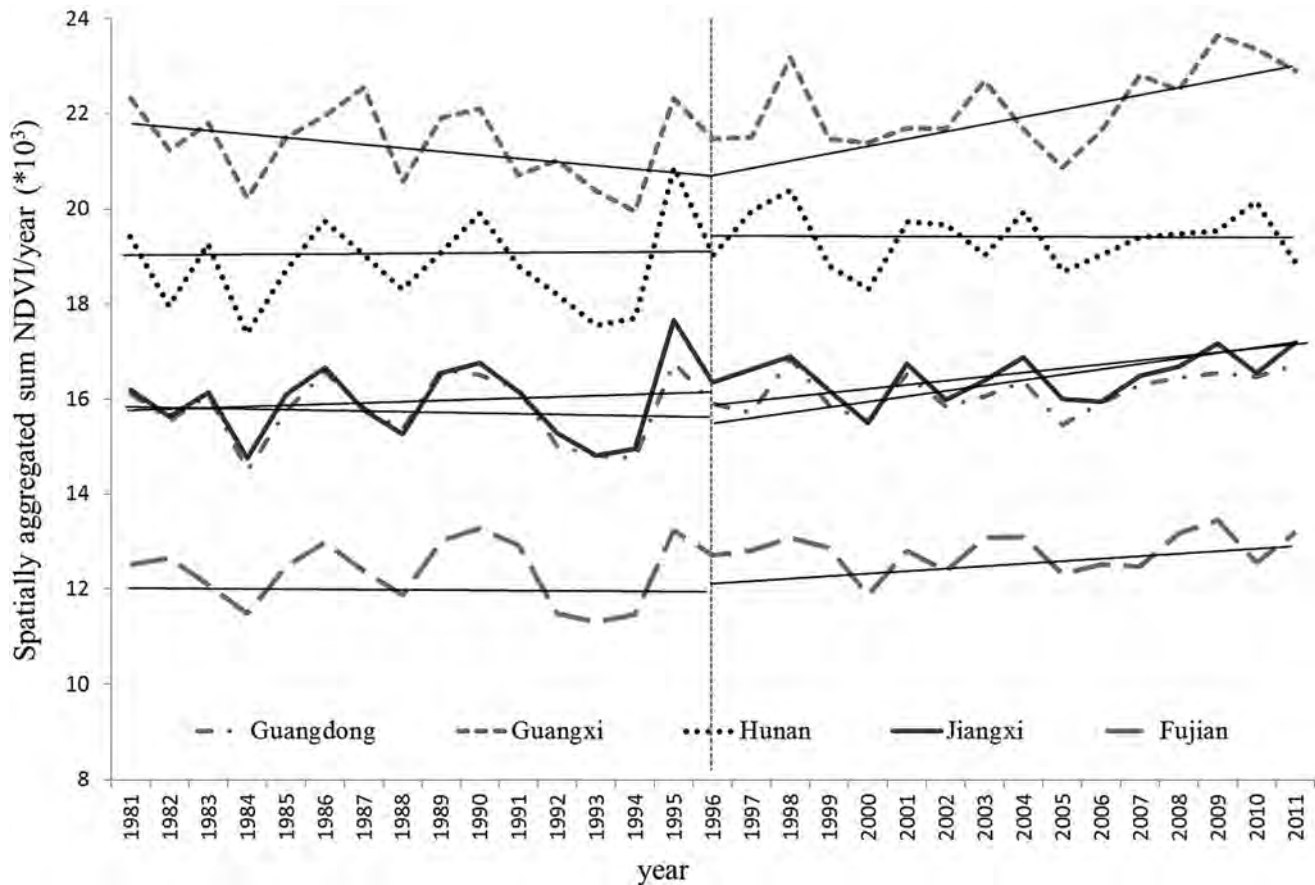


Fig. 5 China: NDVI trends in five southern provinces (1981–2011).

declining NDVI, it is obvious that, in rapidly industrializing areas, urbanization and linking infrastructure account for much of the decline of NDVI. Using Landsat imagery to assess the influence of land use change, Bai et al.^[14] plotted climate-adjusted NDVI and SOTER residuals against the numerical change in NDVI ascribed to land use. They found a clear relationship between land use change and land degradation—and apparently contradictory trends that might be explained by changes in management, without change in the kind of land use, and by forest fires and outbreaks of pests and disease.

CONCLUSIONS

What NDVI Can and Cannot Do

The aims of GLADA were to provide a globally consistent yardstick for measuring land degradation and to highlight areas where significant biological change is happening—so that they can be investigated and attended to. But follow-up has been disappointing; the results were greeted by disbelief, some published and rebutted but, mostly, behind the scenes and directed to the sponsoring FAO program, which cobbled together a pick and mix of assessments in support

of the old beliefs.^[16] The GLADA, itself, flagged some caveats—most of which have been cleared:

- *The NDVI signal can be saturated at closed vegetation canopy*—so it is more sensitive for cropland and range-land than for forest. But NDVI is useful for forests, and many forests do not have a closed canopy.
- *Maximum composite values may underestimate greenness under cloud and snow cover.* Removal of cloud effects is done by harmonic analysis of the GIMMS data and comparison of global NDVI trends using the reconstructed data, with the original showing no measurable difference—so GLADA is unaffected by cloud cover.
- *Soil moisture affects the NDVI signal in areas of sparse vegetation.* Areas of NDVI less than 0.05 (arid) are not considered.
- *Autocorrelation nullifies trend analysis.* Autocorrelation in the data (whereby any individual value is influenced by the preceding values) is avoided by using annual sum NDVI rather than the fortnightly GIMMS values. But this entails a loss of information; for example, we cannot analyze subtly changing seasonal responses of the NDVI signal that may indicate the nature of any degradation. de Jong et al.^[17] applied the

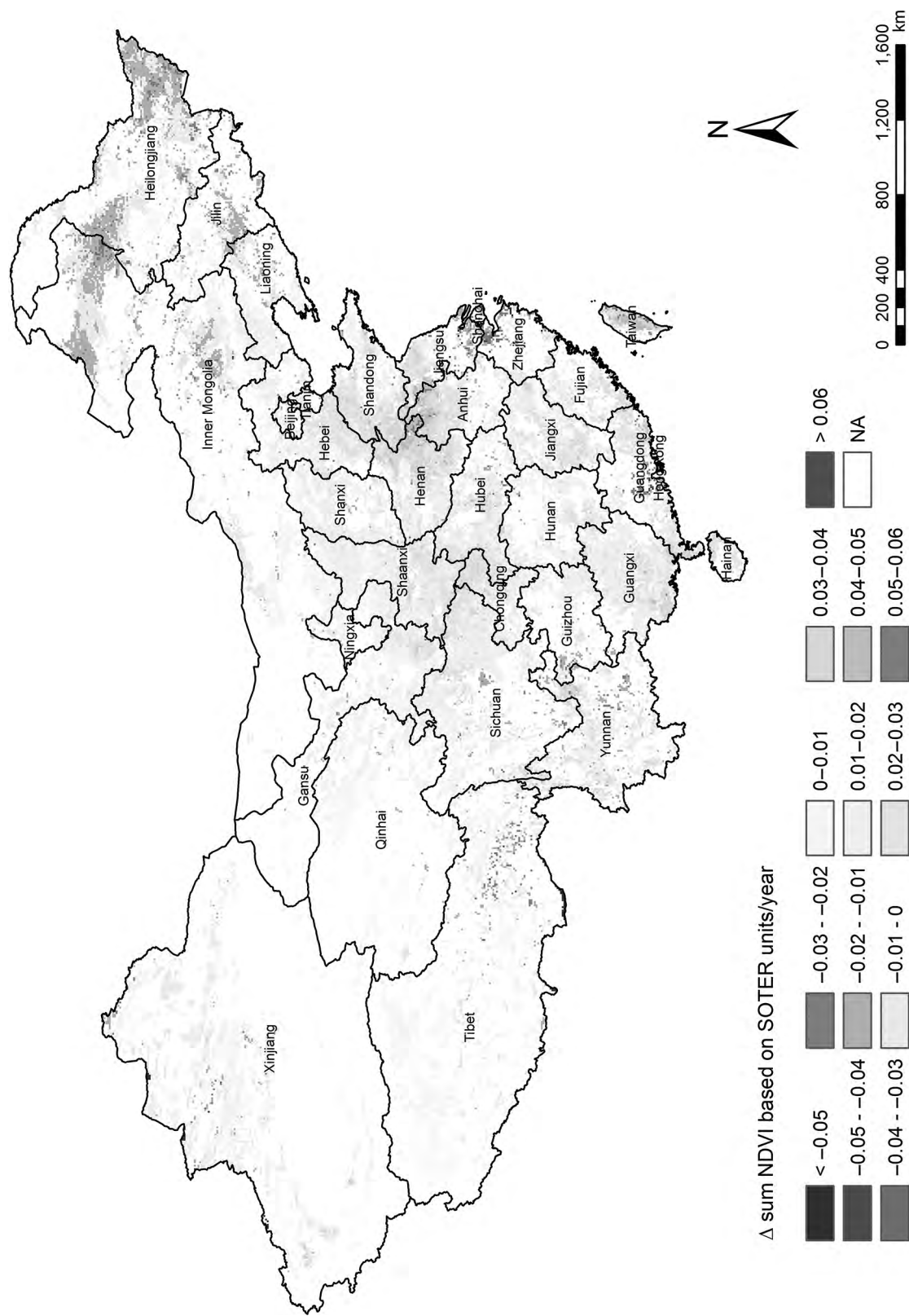


Fig. 6 Change in sum NDVI by SOTER units (1981–2011).

Cultivation
Desertification
— noncultivation

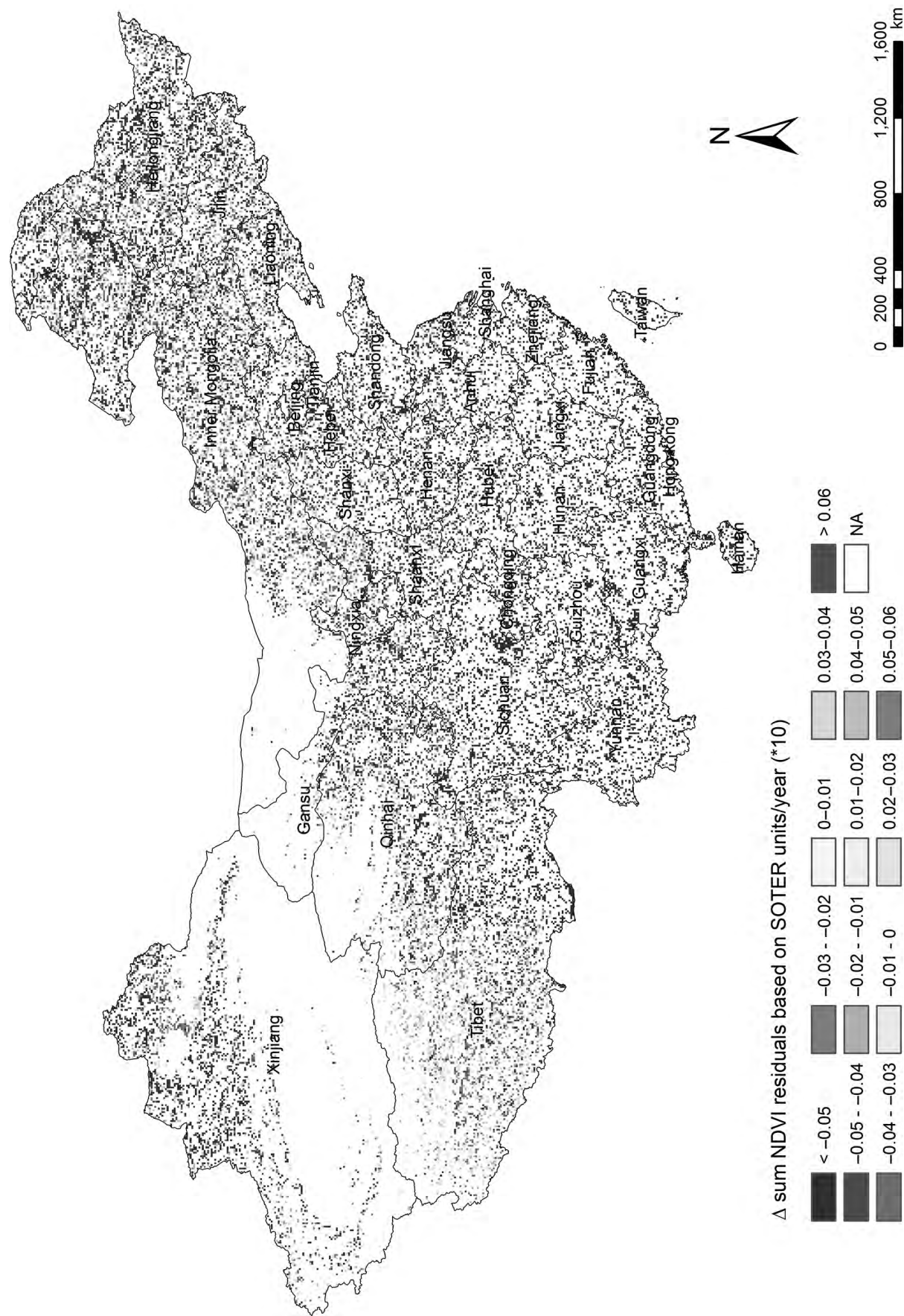


Fig. 7 Trends of NDVI residuals from SOTER units (1981–2011).

non-parametric Mann–Kendall model that is unaffected by autocorrelation to GIMMS NDVI data and normalized the data for seasonal variations in phenology rather than calendar years (which should be better in the southern hemisphere where growing seasons do not fall neatly within the calendar year). Linear regression and the Mann–Kendal procedure measure different factors: the first measures the annual accumulated photosynthetic activity, whereas the second measures the photosynthetic intensity of the growing season. Each has its own tale to tell, but the close similarity of the patterns of greening and browning revealed by the two models suggests that both are robust.

- *It is hard to separate the effects of climate from the effects of land degradation.* Wessels et al.^[18] used the trend of residuals (RESTREND) to distinguish the human-induced land degradation from the effects of rainfall variability, but this measure is not amenable to economic analysis. Empirical screening of GLADA using RUE produces much the same patterns as RESTREND but loses data. Conijn et al.^[19] disentangled climatic effects and other factors affecting NDVI by modeling biomass production independently according to crop characteristics and global data for climate, soil, and land use. Combining NDVI trends and modeled changes in biomass yields four scenarios, namely *positive $\sum$ NDVI and positive biomass change*—greening might be explained by improving weather; *positive $\sum$ NDVI and negative biomass*—despite worsening weather, greenness has increased, thanks to management or atmospheric fertilization; *negative $\sum$ NDVI and positive biomass*—greenness declines against a trend of expected increase, so land degradation or land use change has outweighed favorable weather; and *negative $\sum$ NDVI and negative biomass*—declining greenness may be explained by worsening weather. Whether the benefits of clearer separation of the climatic drivers are worth the substantial effort required is moot. The spatial variability of rainfall in drylands makes interpolation of the sparse point measurements problematic; biomass modeling using just a few vegetation types and predefined management has its own limitations. It is not easier to verify changes in the calculated biomass using independent data than to verify changes in NDVI.
- *Measurement of land degradation NDVI trends underestimates the problem.* Farming everywhere is running down stocks of soil organic matter that supplies plant nutrients; maintains infiltration, available water capacity, and resilience against erosion; and fuels soil biodiversity. Over the 20th century, 60% of soil and biomass carbon has been lost through land use change;^[20] Chernozem, the best arable soils in the world, has lost 30–40% of its organic carbon^[21]—yet they yield abundantly till a tipping point is reached and then the system flips—like the Dust Bowl. The NDVI data show that

heavy use of fertilizer across much of China, the Indo-Gangetic Plain, Europe, the Midwestern United States, and Southern Brazil is no longer accompanied by increasing production but may be concealing soil degradation.

Policy Applications

The NDVI data demonstrate that land degradation is a global issue and, by and large, present policies are failing to arrest it. Arresting land degradation, not to mention remediation, requires long-term investment. This may yield high social returns, but budgetary constraints demand prioritization—so decision-makers need information about the location, extent, and severity of degradation, and an early warning system so that timely interventions may be made in specific areas where land and water resources are overloaded. Thirty years of NDVI data show natural cyclical changes and, also, changes in these trends beyond individual drought cycles. The NDVI trends translated in terms of NPP provide a robust measure of land degradation that is amenable to economic evaluation. Deviation from the local or regional trend can also provide an early warning system: for instance, residuals from SOTER units that, to some extent, normalize the data for local soils, terrain, and climate. But greening and browning cannot be other than a proxy for land improvement and degradation: they identify places where significant biological change is happening but tell us little about the nature of these changes and nothing about the drivers.

The resolution of GIMMS AVHRR data is a limitation in that an 8-km pixel integrates the signal from a wider surrounding area. Many symptoms of severe degradation, such as gullies, rarely extend over such a large area: they must be severe indeed to be seen against the signal of surrounding unaffected areas. Annually aggregated data also lose much of the detail within the data set, but the use of the harmonic analysis of NDVI time series algorithm to eliminate the effects of haze, snow, and cloud cover and integration with auxiliary data sources such as climate and land use information open the door to more forensic analysis of the wealth of information within the data set. And newer sensors offer subkilometer resolution and, in the case of MODIS, a more direct measure of NPP.

REFERENCES

1. Dent, D.; Dalal-Clayton, B. *Securing Land Resources: Information Needs Today and Tomorrow*; IIED: London, 2014; 115 pp. Environmental Governance Series 8.
2. Cracknell, A. The exciting and totally unexpected success of AVHRR in applications for which it was never intended. *Adv. Space Res.* **2001**, *28*, 233–240.
3. Myneni, R.B.; Ramakrishna, R.; Nemani, R.; Running, S. Estimation of global leaf area index and absorbed PAR using

- radiative transfer models. *IEEE T. Geosci. Remote* **1997**, *35*, 1380–1393.
4. Sellers, P.J.; Los, S.O.; Tucker, C.J.; Justice, C.O.; Dazlich, D.A.; Collatz, G.J.; Randall, D.A. A revised land surface parameterization (SiB2) for atmospheric GCMs—part II: The generation of global fields of terrestrial biophysical parameters from satellite data. *J. Clim.* **1996**, *9*, 706–737.
 5. Rasmussen, M.S. Developing simple, operational, consistent NDVI-vegetation models by applying environmental and climatic information: Part I: Assessment of net primary production, Part II: Crop yield assessment. *Int. J. Remote Sens.* **1998**, *19*, 97–117.
 6. Anyamba, A.; Tucker, C.J. Analysis of Sahelian vegetation dynamics using NOAA-AHRR NDVI data from 1981–2003. *J. Arid Environ.* **2005**, *63*, 596–614.
 7. Nemani, R.R.; Keeling, C.D.; Hashimoto, H.; Jolly, W.M.; Piper, S.C.; Tucker, C.J.; Myneni, R.B.; Running, S.W. Climate-driven increases in global terrestrial net primary production from 1982 to 1999. *Science* **2003**, *300*, 1560–1563.
 8. Seaquist, J.W.; Olsson, L.; Ardö, J. A remote sensing-based primary production model from grassland biomes. *Ecol. Model.* **2003**, *169*, 131–155.
 9. Tucker, C.J.; Pinzon, J.E.; Brown, M.E. *Global Inventory Modeling and Mapping Studies (GIMMS) Satellite Drift Corrected and NOAA-16 Incorporated Normalized Difference Vegetation Index (NDVI), Monthly 1981–2002*; University of Maryland: College Park, 2004.
 10. Pinzon, J.E.; Tucker, C.J. A non-stationary 1981–2012 AVHRR NDVI_{3g} time series. *Remote Sens.* **2014**, *6*, 6929–6960.
 11. Maisongrande, P.; Duchemin, B.; Dedieu, G. VEGETATION/SPOT. An operational mission for Earth monitoring: Presentation of new standard products. *Int. J. Remote Sens.* **2004**, *25*, 9–14.
 12. Zhao, M.; Heinsch, F.A.; Nemani, R.R.; Running, S.W. Improvements of the MODIS terrestrial gross and net primary production global data set. *Remote Sens. Environ.* **2005**, *95*, 164–176.
 13. Bai, Z.G.; Dent, D.L.; Olsson, L.; Schaepman, M.E. Proxy global assessment of land degradation. *Soil Use Manage* **2008**, *24*, 223–234.
 14. Bai, Z.G.; Wu, Y.J.; Dent, D.L.; Zhang, G.L.; Dijkshoorn, J.A.; van Engelen, V.W.P.; van Lynden, G.W.J. *Assessment of Land Degradation and Improvement in China. 2. Accounting for Soils, Terrain and Land Use Change*, ISRIC Report 2010/05; ISRIC-World Soil Information: Wageningen, 2010.
 15. Deng, L.; Liu, G.; Shanguan, Z. Land use conversion and changing soil carbon stocks in China's "Grain-for-Green" program: A synthesis. *Glob. Chang. Biol.* **2014**, *20*, 3544–3556.
 16. FAO. *SOLAW—Land Degradation*, 2009. Available at http://www.fao.org/fileadmin/templates/solaw/files/thematic_reports/SOLAW_thematic_report_3_land_degradation.pdf (accessed June 2014).
 17. de Jong, R.; de Bruin, S.; de Wit, A.; Schaepman, M.E.; Dent, D.L. Analysis of monotonic greening and browning trends from global NDVI time-series. *Remote Sens. Environ.* **2011**, *115*, 692–702.
 18. Wessels, K.J.; Prince, S.D.; Malherbe, J.; Small, J.; Frost, P.E.; VanZyl, D. Can human-induced land degradation be distinguished from the effects of rainfall variability? A case study in South Africa. *J. Arid Environ.* **2007**, *68*, 271–297.
 19. Conijn, J.G.; Bai, Z.G.; Bindraban, P.S.; Rutgers, B. *Global Changes in Net Primary Productivity Affected by Climate and Abrupt Land Use Change Since 1981*, Report 2013/01; Plant Research International/ISRIC-World Soil Information: Wageningen, 2013.
 20. Lal, R. Anthropogenic influences on world soils and implications to global food security. *Adv. Agron.* **2007**, *93*, 69–93.
 21. Krupenikov, I.A.; Boincean, B.P.; Dent, D. *The Black Earth*; Springer: Dordrecht, 2012.

Degradation: Off-Farm Economic Implications

Dennis Wichelns

International Water Management Unit, Colombo, Sri Lanka

Ellen Burnes

MWH Americas, Inc., Sacramento, California, U.S.A.

Abstract

We present an economic model of the off-farm impacts of soil degradation. Our goals are to describe why off-farm impacts occur and to identify policies that might be helpful in motivating reductions in negative impacts. We also examine policies that can motivate farmers to adopt technologies that generate positive off-farm impacts. The appropriate framework for this analysis is a social optimization model that includes the net revenues generated by agriculture and any costs and benefits due to off-farm impacts.

INTRODUCTION

Farm-level decisions regarding inputs and outputs often have impacts on other farmers and on natural and environmental resources. Some of the impacts involve soil degradation, either directly or through impacts on complementary inputs that enhance soil productivity. For example, excessive irrigation on one farm can generate surface runoff that carries sediment into irrigation channels used by other farmers. Excessive withdrawals of water from a canal or aquifer can reduce the volume of water available to other farmers in a region. Soil productivity is reduced on those farms if the water supply becomes inadequate to support crop production. Some farm-level decisions have positive impacts on soil productivity and environmental quality. Farmers installing sub-surface drainage systems might relieve the pressure from a shallow water table for several neighbors. Farmers who use sprinkler or drip irrigation systems might reduce their withdrawal of water from regional sources, while also reducing surface runoff and deep percolation.

A SOCIAL OPTIMIZATION MODEL

The net social benefits of agricultural production include the net revenues earned by farmers and the costs or benefits of any off-farm impacts. The appropriate framework is intertemporal because 1) farmers seek to maximize the present value of the sum of net revenues earned over time and 2) soil is a durable resource. Production decisions in any year can affect soil productivity in ways that influence crop and livestock choices, input use, and yields in many subsequent years. Decisions can also have long-term impacts on soil and water resources used by other farmers

and on environmental resources that generate public benefits. Pesticides and salt that enter streams or aquifers via surface runoff or deep percolation can have long-lasting impacts on water quality. The waterlogging and salinization that result from excessive irrigation or inadequate drainage can reduce soil productivity for many years.

The objective function of the social optimization model includes a set of J crop and livestock activities and T time periods. Total revenue, total costs, and off-farm impacts are shown as functions of land, labor, capital, and other inputs. This model differs from the farm-level optimization model by the inclusion of off-farm benefits and damages. In addition, the objective function involves the maximization of the value of net benefits rather than net revenue

$$\begin{aligned} \text{PVNB} \sum_{t=0}^T \sum_{j=1}^J [& P_{jt} Y_{jt}(L_{jt}, B_{jt}, K_{jt}, A_{jt}) \\ & - TC_{jt}(L_{jt}, B_{jt}, K_{jt}, A_{jt}) + OFB_{jt}(L_{jt}, B_{jt}, K_{jt}, A_{jt}) \\ & - OFD_{jt}(L_{jt}, B_{jt}, K_{jt}, A_{jt})] [1 + r]^{-t} \end{aligned} \quad (1)$$

where PVNB is the value sum of net benefits, P_{jt} is the price of crop or livestock product j in year t , Y_{jt} represents the output of crop and livestock products, TC_{jt} represents the total costs of production, OFB_{jt} represents the off-farm benefits generated by producing product j in year t , OFD_{jt} represents the off-farm damages caused by production, L_{jt} is the amount of land used in each production activity, B_{jt} is the amount of labor, K_{jt} is the amount of capital, A_{jt} represents all other inputs, and r is the appropriate discount rate.

We use this model to analyze the off-farm impacts of farm-level decisions, such as soil erosion, waterlogging, and salinization. Our goals are to examine the causes of

off-farm impacts and to describe public policies that might reduce those impacts in some areas.

OFF-FARM BENEFITS AND DAMAGES

Farmers have an economic incentive to maximize the present value of net returns from crop production described by the first two terms in the optimization model ($P_{jt}Y_{jt}$ minus TC_{jt}). In the absence of appropriate policies, farmers often have little incentive to consider the off-farm benefits or damages generated by their activities (OFB_{jt} and OFD_{jt}).

Farmers seeking to maximize net revenue will equate the marginal cost of their decisions with the expected marginal benefits, as viewed from their perspective. The optimizing criteria that guide farm-level decisions will not include the positive or negative impacts on other farmers or environmental resources. As a result, farm-level choices will not be consistent with those that would maximize the present value of net social benefits. From the public's perspective, farmers will use too few inputs that generate off-farm benefits and too many inputs that generate off-farm damages.

The divergence between farm-level and socially optimal input levels can be described using diagrams that depict incremental costs and benefits. For example, the farm-level incremental cost (also called the marginal cost) of obtaining groundwater for irrigation might be considered constant within a season, whereas the marginal benefit declines with the amount of water applied per hectare. The farm-level profit-maximizing water application (W_F) is found by equating the marginal benefit (MB_F) and marginal cost (MC_F), as shown in Fig. 1. In many areas, groundwater extraction and irrigation generate off-farm impacts, such as a reduction in water available for other farmers and surface runoff that carries sediment. The incremental off-farm impacts must be added to the farm-level marginal costs to determine the socially optimal water application, W_S

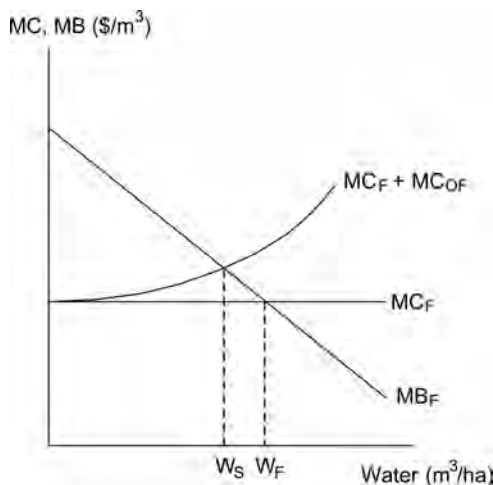


Fig. 1 The marginal costs and benefits of water applied for irrigation.

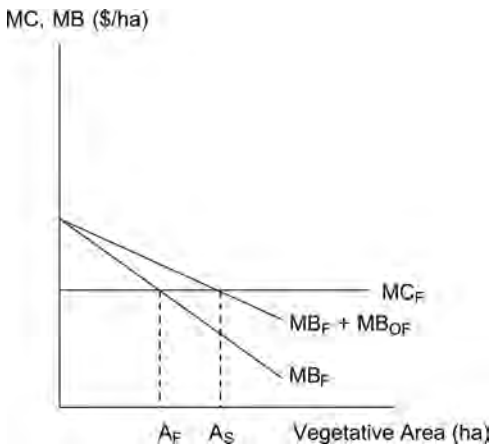


Fig. 2 The marginal costs and benefits of vegetative filter strips.

(Fig. 1). In the absence of appropriate policies, farm-level input use will exceed the socially optimal level when off-farm marginal costs (MC_{OF}) are non-zero.

Farmers can use vegetative filter strips to reduce the sediment transport from farms. Farm-level benefits include the retention of topsoil and nutrients, whereas off-farm benefits include lower costs of removing sediment and nutrients from waterways. The incremental cost of installing vegetative filter strips might be constant within a season, whereas the farm-level incremental benefits likely decline with the area planted in filter strips. The optimal area from the farmer's perspective (A_F) is found by equating the marginal benefit (MB_F) and marginal cost (MC_F), as shown in Fig. 2. The incremental off-farm benefits must be added to the farm-level marginal benefits to determine the socially optimal area, A_S (Fig. 2). In the absence of appropriate policies, the area planted in vegetative filter strips will be smaller than the socially optimal area when off-farm marginal benefits (MB_{OF}) are non-zero.

POLICY IMPLICATIONS

Economic incentives and other policies can be implemented to encourage farmers to reduce their use of inputs that generate off-farm damages and to increase their use of inputs that generate off-farm benefits. Per-unit taxes and subsidies on inputs will raise or lower the farm-level marginal costs of inputs. Per-unit taxes on irrigation water will motivate improvements in water management, whereas per-unit subsidies for vegetative filter strips will encourage greater planting activity. Low-interest loans and cost-share programs can encourage greater farm-level investment in production methods that reduce off-farm damages or increase off-farm benefits.^[1]

Public policies designed to reduce soil degradation must be chosen carefully, as farm-level responses will vary among regions because of differences in existing institutions and cultural perceptions.^[1-4] Farmers must consider

many components of public policies when selecting crop production activities, technologies, and variable inputs. Commodity price support programs motivate higher levels of production for some crops, whereas conservation reserve programs encourage farmers to retire land from agricultural production. The appropriate empirical values of taxes, subsidies, and other incentives will vary among regions according to differences in the incremental costs and damages of agricultural production and with differences in other policies that influence farm-level decisions.

The justification for policy intervention to reduce off-farm damages will also vary among regions. Some authors suggest that the magnitude of off-farm damages from soil degradation worldwide is quite large,^[5,6] whereas others suggest the magnitude is reasonable.^[7,8] Most authors probably would agree that off-farm impacts are severe in some areas^[9] and that policy intervention is needed to increase social net benefits in those areas. Policies that motivate advances in knowledge are needed to address local issues regarding soil degradation and to ensure future improvements in aggregate productivity.^[8,10]

CONCLUSION

Market prices for inputs and outputs often do not reflect the off-farm impacts of production activities, such as those arising from soil degradation. Public policies can motivate farmers to consider off-farm benefits and damages, but policy intervention is not always appropriate. The need for policy intervention and the empirical values of parameters that will motivate optimal input use will vary among regions, and over time, with differences in existing institutions and in the off-farm impacts of farming activities.

ACKNOWLEDGMENT

We appreciate the helpful comments of Megumi Nakao and two reviewers regarding earlier versions of this entry (contribution no. TP 03-19 of the California Water Institute).

REFERENCES

1. Sanders, D.; Cahill, D. Where incentives fit in soil conservation programs. In *Using Incentives for Soil Conservation: From Theory to Practice*; Sanders, D.W., Huszar, P.C., Sombatpanit, S., Enters, T., Eds.; Science Publishers: Enfield, 1999; 11–24.
2. Pagiola, S. Economic analysis of incentives for soil conservation. In *Using Incentives for Soil Conservation: From Theory to Practice*; Sanders, D.W., Huszar, P.C., Sombatpanit, S., Enters, T., Eds.; Science Publishers: Enfield, 1999; 41–56.
3. Barbier, E.B. The economic linkages between rural poverty and land degradation: Some evidence from Africa. *Agr. Ecosyst. Environ.* **2000**, *68*, 355–370.
4. Ananda, J.; Herath, G. Soil erosion in developing countries: A socio-economic appraisal. *J. Environ. Manag.* **2003**, *68*, 343–353.
5. Pimentel, D.; Kounang, N. Ecology of soil erosion in ecosystems. *Ecosystems* **1998**, *1*, 416–426.
6. Pimentel, D. Soil erosion and the threat to food security and the environment. *Ecosyst. Health* **2000**, *6*, 221–226.
7. Crosson, P. Soil erosion estimates and costs. *Science* **1995**, *269*, 461–465.
8. Crosson, P. Will erosion threaten agricultural productivity? *Environment* **1997**, *39* (4–9), 29–31.
9. Faeth, P. Building the case for sustainable agriculture. *Environment* **1994**, *36* (16–20), 34–39.
10. Crosson, P. Sustainable agriculture. *Environment* **1994**, *36*, 45.

Degradation: Physical

Michael A. Zebisch

German Agency for International Cooperation, Tashkent, Uzbekistan

Anthony R. Dexter

Institute of Soil Science and Plant Cultivation (IUNG), Pulawy, Poland

Abstract

Different forms of soil physical degradation go along with each other—they are interlinked and not always clearly distinguishable from each other. Different types of soil physical degradation cannot be distinguished uniquely in terms of simple cause-and-effect relationships. The causes and effects are often the same for several types of degradation. The main types of soil physical degradation are characterized by the different sets of processes involved and responsible for the degradation. Deterioration of soil structure is the overriding and more general form of soil physical degradation. Compaction, sealing and crusting, and hardsetting are important special types of soil structure breakdown with important implications for soil management and land use.

INTRODUCTION

There is no accepted standard terminology concerning the definitions of terms used to describe and classify soil degradation. It is difficult to draw clear boundaries between the different types of soil degradation because, as with any complex natural system, there are a lot of interrelationships and interdependencies between chemical, biological, and physical degradation processes in the soil. Different forms of soil physical degradation go along with each other—they are interlinked and not always clearly distinguishable from each other. The International Union of Soil Science^[1] has proposed a comprehensive terminology for soil physical processes and characteristics, but the use of terms is not coordinated and sometimes confused. The classification of the different types of soil physical degradation is, therefore, not clearly differentiated. For example, it may be argued that waterlogging is not a degradation type on its own. It is rather a consequence of other physical degradation processes and caused by a reduction in permeability due to soil structure breakdown or compaction. Because of impeded water entry and movement, waterlogging can also lead to other types of degradation, such as salinization. These examples illustrate the fuzziness of boundaries between soil physical degradation and other forms of degradation. Different types of soil physical degradation cannot be distinguished uniquely in terms of simple cause-and-effect relationships. The causes and effects are often the same for several types of degradation. Fig. 1 depicts the cause-and-effect system typical for soil physical degradation.

The main types of soil physical degradation described in this entry are characterized by the different sets of processes involved and responsible for the degradation. Deterioration

of soil structure is the overriding and more general form of soil physical degradation. Compaction, sealing and crusting, and hardsetting are important special types of soil structure breakdown with important implications for soil management and land use.

LOSS OF SOIL STRUCTURE

There are a number of definitions of soil structure in the literature. Many of these describe the arrangement of primary particles into aggregates of different sizes and shapes and the associated pore spaces between them. Therefore, a structured soil is heterogeneous, whereas a degraded, structureless soil is homogeneous. A more general definition is “the spatial heterogeneity of the different components or properties of soil.”^[2] Soil structure significantly influences all processes that take place in the soil. It influences water infiltration (and hence runoff), the movement of water within the soil, and the amount of water that can be stored in the soil. Therefore, it has a direct influence on soil erosion. Soil structure also determines aeration levels in the soil, which are essential for the oxygen supply to roots, soil fauna, and for aerobic microbial activity. High soil strength impedes seedling emergence and root growth and also makes tillage more difficult and more energy demanding. These functions demonstrate the role and importance of the size and continuity of pores in the soil for the movement of soil and water into and within the soil.^[3] Not only the soil structure but also the stability of the structure is of major importance. Structural stability determines the ability of a soil to withstand imposed stresses without changes in its geometric structure and functions. These stresses may be

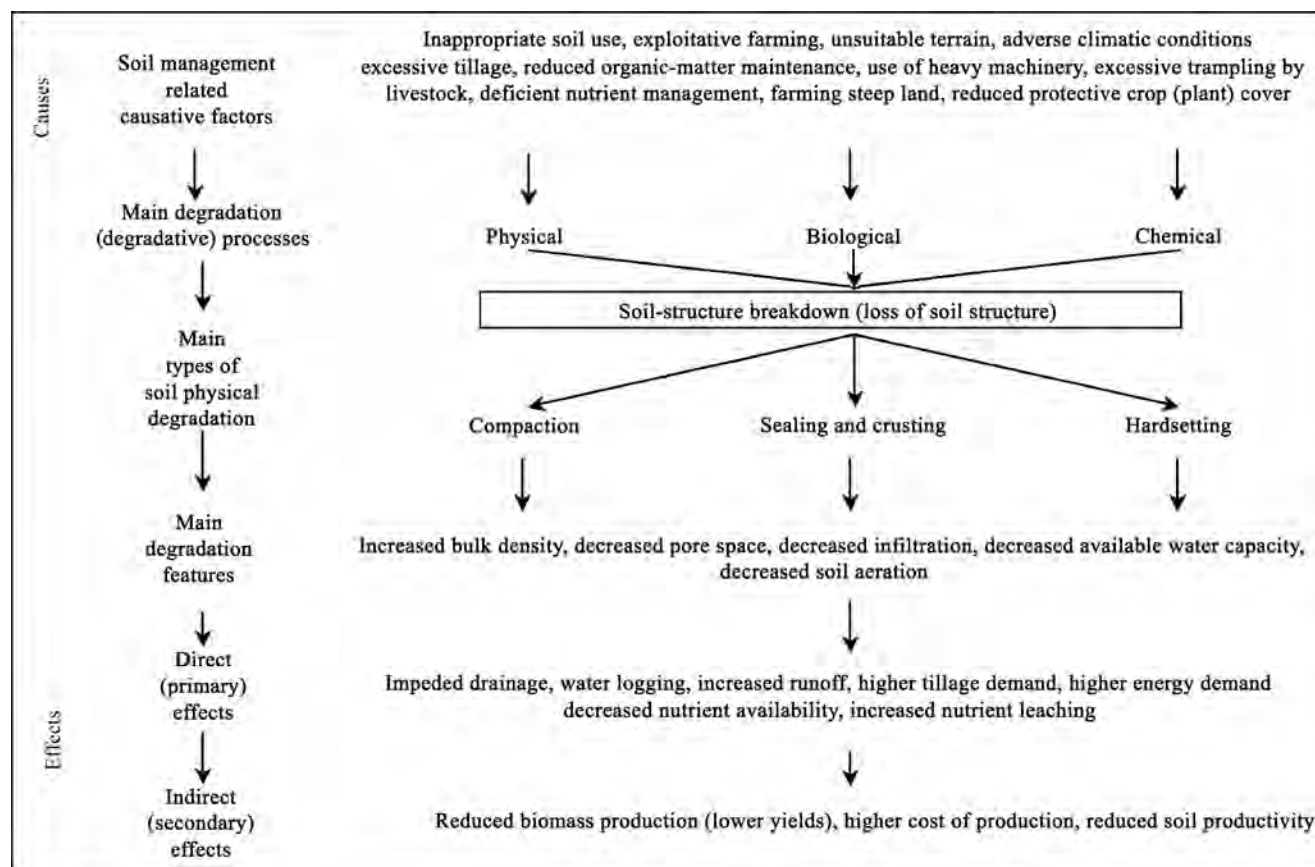


Fig. 1 Simplified cause-and-effect system of soil physical degradation.

Source: Adapted from Dexter^[2] ©1988, Shaxson^[3] ©1999, Gabriels, Horn et al.^[4] ©1998, Lal & Ahmadi^[5] ©2000, and Arvidsson^[6] ©2000.

due to rapid wetting, raindrop impact, wheel traffic, and excessive tillage.

Soil physical degradation reflects soil structure breakdown or homogenization, when soil aggregates are destroyed by internal or external forces.^[4] Internal forces are applied when entrapped air breaks out of soil aggregates upon flooding. External forces appear in the form of rain impact or pressure and shearing forces exerted by animal trampling, wheel traffic, and tillage implements. Depending on the water content, this results in either pulverization or compaction of the soil. Soil physical degradation, however, depends not only on the degrading forces and stresses but also on the stability of a soil to withstand these stresses and its resilience to recover from different levels of short-term and long-term degradation.

Not all activities that break down soil are necessarily degrading. Appropriate use and management of the soil also include measures to maintain—and improve—the resilience of soils. Appropriate soil tillage aims at producing soil structural conditions that improve plant establishment and growth. However, excessive tillage with high-energy inputs can result in losses of soil organic matter and can increase the amount of readily dispersible clay. Both of

these consequences can result in decreased structural stability and hence structure breakdown. Such a degraded soil will be weaker when wet but stronger when dry and will be more difficult to manage under all conditions.

Compaction

Compaction describes the state, or the increase, of “compactness,” i.e., bulk density, of a soil. Compared with its undisturbed condition, a compacted soil exhibits reduced total pore space, especially because of a drastic reduction of the macropores and a pronounced discontinuity of the pore system within the profile.^[3] This affects the conductive properties of the soil and reduces its ability to retain air and water. Hence, plants growing under high evaporative demand suffer more from compaction than plants growing under low evaporative demand.^[7] Compaction also inhibits root penetration and development, thus affecting nutrient uptake and, consequently, plant growth. [Table 1](#) shows examples of the effects of soil compaction on the yields of different crops, due to tillage and heavy machinery.

The most important cause of compaction is off-road wheel traffic and the use of heavy machinery in mechanized

Table 1 Crop yield decline due to soil compaction.

Crop	Soil type	Selected control parameter	Yield trend	Country	Source
Wheat	Sandy loam	Penetration resistance increased from 0.83 to 1.29 MPa; bulk density increased from 1.68 to 1.78 Mg m ⁻³ after two crops	Decrease between 12% and 38%	Pakistan	[8]
Sorghum (fodder)	Sandy loam	As above, but after one crop	Decrease between 14% and 22%		
Spring barley Oilseed rape Winter wheat	Loam and sandy loam	Saturated hydraulic conductivity reduced by about 90% after one crop	Average overall decrease of only 1%, non-significant	Sweden	[9]
Corn (grain)	Silt loam	Bulk density was correlated with tillage type, i.e., lowest under chisel plowing (1.25 Mg m ⁻³) and highest under moldboard plowing (1.31 Mg m ⁻³) after 11 cropping seasons	Under regular machine traffic, yield decline was 5% for 7.5 Mg axle load and 10% for 7.5 Mg axle load over 11 cropping seasons	Ohio, U.S.A.	[5]
Barley	Clay loam	Bulk density increased from 1.15 to 1.50 g cm ⁻³ after one crop depending on depth, type of tillage, and equipment used	Yield declined from around 1300–1000 kg ha ⁻¹	Jordan	[6]

agriculture.^[7,10] In the upper part of the soil, mainly the contact pressure of moving vehicles and the shearing effects of the wheels determine compaction. At a greater soil depth, the axle load is more important. The bulk density of the soil increases with the weight, i.e., pressure, exerted by vehicles and machinery, and also increases with increasing soil water content and the number of passes of the vehicles and machinery over the soil surface. Soils with high clay contents and well-developed pore systems are generally more compressible than sandy soils. Tillage practices also influence the degree of topsoil compaction. Within the annual cropping cycle, the topsoil undergoes changes in its bulk density. To a large extent, plowing alleviates the effects of topsoil compaction produced by the crop husbandry measures and harvesting activities carried out during and after the cropping season. However, seedbed quality may be severely reduced if the topsoil was severely compacted in the previous year. A cumulative effect of compaction caused by heavy machinery is often observed after several years. Soil compaction increases the demand for tillage. It has been shown that compaction reduces the range of water content over which tillage can satisfactorily be done and therefore reduces the opportunities for tillage.^[11]

For practical purposes, two main types of soil compaction can be distinguished, namely the surface layer compaction and subsoil compaction.

Surface layer compaction describes the compaction in the upper part of the soil profile, i.e., in arable soils usually the plow layer. Compaction in the surface layer is dynamic and changes significantly over the cropping season, increasing with increasing machinery passes over the field and decreasing again with primary tillage for seedbed preparation for the following season. In reduced tillage systems, the effects of soil compaction

build up and tend to be more lasting than in plow systems.^[5] These effects seem to be more pronounced in sandy soils than in clay soils, because of their lower capacity to maintain a continuous macropore system.^[2] Adequate tillage effectively reduces soil compaction in the surface layer and its effects.

Subsoil compaction affects soils beyond the surface layer at depths >30 cm. It is caused by heavy machinery, especially those with high axle loads.^[5,10] Swelling–shrinking, freezing–thawing, and biological activities can alleviate compaction to a certain extent. However, these processes are not usually effective beyond a depth of 35 cm. Subsoiling using specially designed equipment can in some cases alleviate subsoil compaction, but it is very energy demanding. Subsoiling should be attempted only when the whole soil profile is drier than the plastic limit (or lower Atterberg limit) of the soil, otherwise more harm than good may be done. Soil at these depths can be dried sufficiently only by plant roots. However, the compaction may prevent the necessary root growth. In many situations after subsoiling, the soil recompacts readily to be denser than it was before sub-soiling because of the destabilization of the soil caused by the mechanical energy input from the subsoiling operation. Compaction should be considered to be an irreversible, permanent form of degradation.

Sealing and Crusting

Seals and crusts are consequences of rain and flooding on unprotected soil surfaces. Under the impact of raindrops and the “soaking” effect of water, the bonds that hold the particles together become weak, and the soil aggregates tend to fall apart. Individual particles become separated. These particles become rearranged, and the finer particles

tend to be washed into the cavities of the surface. There they form a very thin (1–5 mm) and dense layer that clogs the soil pores and seals the surface. This process is also referred to as capping. These seals are usually very elastic. Typical characteristics of soil seals are that they do not crack and cannot be removed from the surface.

Soil crusts are formed by the same processes that form seals. They are much thicker than seals (usually 5–20 mm), they can be separated easily from the soil surface, and they crack upon drying. Crusts are typically formed on soils with high contents of non-swelling clay susceptible to dispersion. Seals that become hard upon drying are frequently also termed crusts.^[12] Soils with a high content of fine or very fine sand or silt are especially prone to sealing and crusting. The presence of exchangeable sodium in the soil can enhance clay dispersion and thus contribute to seal and crust formation.

Depending on the dominant processes, three types of seals or crusts are distinguished.^[4,13] *Structural crusts* are a result of physical forces, i.e., raindrop impact, animal trampling, wheel traffic, and flooding. *Slaking crusts* are formed as a result of the disintegration of the soil structure when in contact with water. This can take the form of chemical dispersion of the clay due to, e.g., the presence of exchangeable sodium, mainly in sodic soils, or of the slaking of the soil into microaggregates. *Depositional crusts* are formed by the translocation and deposition of fine particles, e.g., by erosion processes and on irrigated fields.

The main direct effects of seals and crusts are a decrease in infiltration and hence increase in surface runoff and soil erosion and an obstruction to seedling emergence.

Hardsetting

Hardsetting is a result of structural breakdown and reconsolidation of soil under repeated wetting and drying cycles.^[12] The result is a hard, homogeneous mass difficult—if not impossible—to till and therefore very restrictive for soil use and management. Hardsetting takes place in two stages. First, the soil structure breaks down under wetting. This process is, in principle, the same for all forms of soil physical degradation and is influenced, as described before, by a complex set of interacting physical, chemical, and biological conditions and processes. The second stage is characterized by the drying of the soil—without a significant regain of structure—and hardening of the soil mass. During this second stage, effective stresses due to the menisci of water films and the water matric potential contribute to the consolidation process. The main effect of hardsetting—an increase in soil tensile strength—increases with each wetting and drying cycle. Soils that tend to hardset usually are low in non-swelling clays and have moderate levels of silt and fine sand.^[12] In soils that release soluble salts when moist, the effects of hardsetting upon drying are usually increased. Hardsetting

can affect soils to depths of 20–30 cm. Because of their extremely high tensile strength, hardsetting soils are very difficult to till when dry. To offset the effects of drying, tillage has to be timed after rainfall or irrigation before the hardening of the soil occurs. This restricts the time available for effective soil tillage.

Hardsetting processes affect many soils, but this does not necessarily make them hardsetting soils. More sandy soils with a weaker structure do not usually become too hard to till upon drying, although they also undergo hardsetting processes. These soils develop features and problems related to hardsetting, but they can still be tilled without major limitations and constraints. Typical effects of hardsetting are the obstruction to seedling emergence and increased resistance to root growth.

CAUSES AND EFFECTS OF SOIL PHYSICAL DEGRADATION

The main causes of soil physical degradation, i.e., the breakdown of the soil structure, are inappropriate land-use and soil management practices. All “exploitative” practices will ultimately lead to degradation and hence reduced soil productivity. The basic detrimental effects of certain practices—or of their neglect—on different soil properties are known. But their specific contribution to the breakdown of the soil structure in a specific location also depends on the other prevalent practices and factors. The same set of circumstances applies to the effects that soil physical degradation has on soil productivity and the environment.

In highly mechanized systems, heavy machinery and excessive tillage are important factors of degradation. In the developing world, land not suitable for cultivation, such as drylands or steep terrain, is increasingly being cropped. The cultivation and husbandry practices associated with these land use systems are largely responsible for degradation. The systems themselves are, in the long term, also largely affected by them. Fig. 1 depicts the causative factors and effects that lead to soil physical degradation. It shows that inappropriate soil and land management leads to an array of different types of soil physical degradation. Ultimately, soil physical degradation leads to reduced plant growth, crop yields, and soil productivity.

The soil environment is very dynamic, and changes in one property may affect the response of another property to different degrading stresses. Also, climatic conditions, which play an important role in degradation, vary from season to season.^[5]

MEASURES TO MITIGATE AND REDUCE SOIL PHYSICAL DEGRADATION

The best way to prevent and control soil degradation is by using the land and soil according to their capability and

observing natural constraints, such as terrain and climatic conditions. This is best done by what is commonly referred to as “appropriate” soil and land use. However, the conditions determining the suitability of soil use are always location specific. Therefore, there is no single universally valid system of appropriate farming. All measures that lead to an optimization of plant cover and plant population, and maintain a stable structure, are suited to prevent soil physical degradation.

ASSESSMENT AND MEASUREMENT OF SOIL PHYSICAL DEGRADATION

Because of the high degree of interaction between the different physical, biological, and chemical factors and processes involved, the quantification of soil physical

degradation in its totality and its effects on plant growth are extremely difficult. There are no standard, objective methods to assess soil physical degradation in its entirety.^[2,4,7,13] In practice in most cases, it is usually a subjective assessment by specialists, based on their individual experience. A well-known example of this is the methodology used by the Global Assessment of Human-Induced Soil Degradation project.^[14]

However, many basic soil properties that are altered by physical degradation can be measured objectively and quite easily. Changes over time in these characteristics can be used to estimate the influence of a single or a set of properties on plant growth and soil productivity. Table 2 lists a number of simple field and laboratory methods that can be used to measure and characterize these properties.

One of the two common basic concepts is the assessment of the severity of past and ongoing degradation.

Table 2 Examples of simple (indicative) tests to assess soil characteristics important for the evaluation of soil physical degradation.

Soil characteristic (indicator)	Field assessment (test)	Laboratory assessment (test)
Compaction	Bulk density (e.g., sand-replacement method) Penetration resistance (cone penetrometer) Shear vane test	Bulk density (using soil cores) Precompaction stress
Total porosity	From bulk density (above) assuming a particle density of 2.65 g cm ⁻³	Pycnometer test
Pore space distribution	N.P.	Calculation from bulk density Water-retention at various potentials
Clay dispersion	“Emerson (1967) test” ^[11]	Gravimetric method (Pipette test) Turbidimetry
Aggregate stability	“Emerson (1967) test” ^[11]	Wet sieving
Aggregate strength	Feel (crushing between thumb and first finger)	Tensile strength test (crushing force)
Infiltration	Double-ring infiltrometer Tension (disk) infiltrometer	
Permeability	Constant-head permeameter	Constant-head permeameter
Hydraulic conductivity	Tension (disk) infiltrometer	Falling-head permeameter
Runoff generation	Rainfall simulation	Rainfall simulation
Soil organic matter content	N.P.	Loss on combustion
Walkley Black method		
Texture (particle-size distribution)	By feel	Sedimentation test
Crust strength	Cone penetrometer test Crop-emergence count	Cone penetrometer test Modulus of rupture test
Water-holding capacity	Neutron probe or TDR 2 days after heavy rain	Soil cores
Aeration status	Air permeability Redox potential Oxygen diffusion rate Oxygen diffusion rate	Air-filled porosity Air permeability Redox potential
Soil-particle detachment	Rainfall simulation and sediment sampling	Rainfall simulation and sediment sampling

Note: N.P., not possible.

Source: Adapted from ISSS,^[1] Gabriels, Horn, et al.,^[4] Lal & Ahmadi,^[5] Arvidsson,^[6] Håkansson & Voorhees,^[7] and Abu-Hamdeh & Al-Widyan.^[9]

This is usually a comprehensive assessment of actual field conditions. The other concept is the assessment of the susceptibility of a certain soil to degradation under a variety of soil-use and soil management practice scenarios. Often, treatments are simulated in the laboratory. Another aspect of concern is the impact of climate change on the sensitivity of soils to physical degradation.

REFERENCES

1. ISSS. *Terminology for Soil Erosion and Conservation*; International Society of Soil Science Publication; International Soil Reference and Information Centre (ISRIC): Wageningen, 1998.
2. Dexter, A.R. Advances in the characterization of soil structure. *Soil Till. Res.* **1988**, *11*, 199–238.
3. Shaxson, F. *New Concepts and Approaches to Land Management in the Tropics with Emphasis on Steep-lands*; FAO Soils Bulletin 75; Food and Agriculture Organization of the United Nations: Rome, 1999.
4. Gabriels, D.; Horn, R.; Villagra, M.M.; Hartmann, R. Assessment, prevention, and rehabilitation of soil structure caused by soil-surface sealing, crusting, and compaction. In *Methods for the Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentin, C., Stewart, B.A., Eds.; CRC Press: New York, 1998; 129–165.
5. Lal, R.; Ahmadi, M. Axle load and tillage effects on crop yield for two soils in central Ohio. *Soil Till. Res.* **2000**, *54*, 111–119.
6. Arvidsson, J. Subsoil compaction caused by heavy sugar-beet harvesters in southern Sweden. I. Soil physical properties and crop yield in six field experiments. *Soil Till. Res.* **2001**, *60*, 67–78.
7. Håkansson, I.; Voorhees, W.B. Soil compaction. In *Methods for the Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentin, C., Stewart, B.A., Eds.; CRC Press: New York, 1998; 167–179.
8. Ishaq, M.; Hassan, A.; Saeed, M.; Ibrahim, M.; Lal, R. Subsoil compaction effects on crops in Punjab, Pakistan. I. Soil physical properties and crop yield. *Soil Till. Res.* **2001**, *59*, 57–65.
9. Abu-Hamdeh, N.H.; Al-Widyan, M.I. Effect of axle load, tire pressure, and tillage system on soil physical properties and crop yield of a Jordanian soil. *T. ASABE* **2000**, *43* (1), 13–21.
10. Horn, R.; van den Akker, J.J.H.; Arvidsson, J. *Subsoil Compaction: Distribution, Processes and Consequences*; Catena Verlag: Reiskirchen, 2000.
11. Dexter, A.R.; Bird, N.R.A. Methods for predicting the optimum and the range of soil water contents for tillage based on the water retention curve. *Soil Till. Res.* **2001**, *57*, 203–212.
12. Mullins, C.E. Hardsetting. In *Methods for the Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentin, C., Stewart, B.A., Eds.; CRC Press: New York, 1998; 109–127.
13. Valentin, C.; Bresson, L.M. Soil crusting. In *Methods for the Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentin, C., Stewart, B.A., Eds.; CRC Press: New York, 1998; 89–107.
14. ISRIC-UNEP. *World Map of the Status of Human-Induced Soil Degradation*; Explanatory Note; International Soil Reference and Information Centre (ISRIC): Wageningen, 1990.

Degradation: Quality and

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Understanding principal processes, factors, and causes of soil degradation is an important prerequisite to our ability to reverse the degradative trends and to preserve a healthy state of global environment. Being a complex problem, it requires a systematic and coordinated effort to effectively address the major issues involved. These issues include: 1) obtaining reliable estimates of the land area affected by different soil degradative processes; 2) standardizing definitions and methods of assessment of soil degradation; 3) evaluating on-site and off-site impacts of soil degradation; 4) developing appropriate measures of soil restoration; and 5) assessing economic and environmental impacts of restoring degraded soils and ecosystems.

INTRODUCTION

Soil degradation is defined as diminution of soil quality in terms of its existing and potential productivity and/or reduction in its ability to be a multipurpose resource by both natural and anthropogenic causes.^[1,2] A decrease in soil quality implies adverse changes in soil properties and processes with a negative effect on the soil's life-support systems.^[2–4] Soil degradation has both agronomic and ecological consequences. Agronomically, soil degradation leads to loss of production and renders an ecosystem unsustainable under the specific land use.^[2] Ecologically, soil degradation leads to pollution of natural waters, emission of radiatively active gases into the atmosphere, decrease in soil biodiversity, etc. Soil degradation is the result of interactive effects of processes, factors, and causes or management practices.

PROCESSES OF SOIL DEGRADATION

The processes of soil degradation involve mechanisms, actions, and interactions that affect the soil's: 1) resilience or capacity of self-regulation; 2) life-support processes or productivity; and 3) ability to moderate environments. The principal processes of soil degradation are briefly shown in [Fig. 1](#) and [Table 1](#).

Physical Processes

These processes lead to adverse changes in soil's physical, mechanical, or hydrological properties. Principal among these processes is the decline in soil structure, which manifests in crusting, compaction, impeded surface and subsoil drainage, and the poor or limited aeration. A reduction in aggregation and the dispersion of clay lead to a loss of clay

through erosion and eluviation. Accelerated soil erosion is a major soil degradative process, which has potentially severe ecological and economic consequences, and has local, regional, or global impacts.

Chemical Processes

These processes lead to undesirable changes in soil chemical (reaction) and nutritional (fertility) properties. Soil chemical degradation involves adverse changes in: 1) properties and processes that regulate nutrient capacity and intensity; 2) soil's ability to inactivate or denature toxic substances; 3) the soil's chemical balance, leading to elemental toxification [Al, manganese (Mn), and salts] and nutrient deficiencies; and 4) the quality of natural water, atmosphere, and the general environments.

Biological Processes

These processes affect soil biodiversity. The principal effect of biological degradation is a decline in soil organic matter content, a reduction in biomass carbon, and a decrease in the activity and species diversity of soil fauna and flora.

NATURAL VS. ANTHROPOGENIC PROCESSES

Soil degradative processes may be natural or anthropogenic. Natural processes are generally slow, soil forming or pedological mechanisms, and their effect is felt over the geological time-span. In contrast, anthropogenic processes are rapid, highly destructive, and their effect is felt over human time-span. Some important natural and anthropogenic processes of soil degradation are listed in [Table 2](#) and are briefly described as follows.

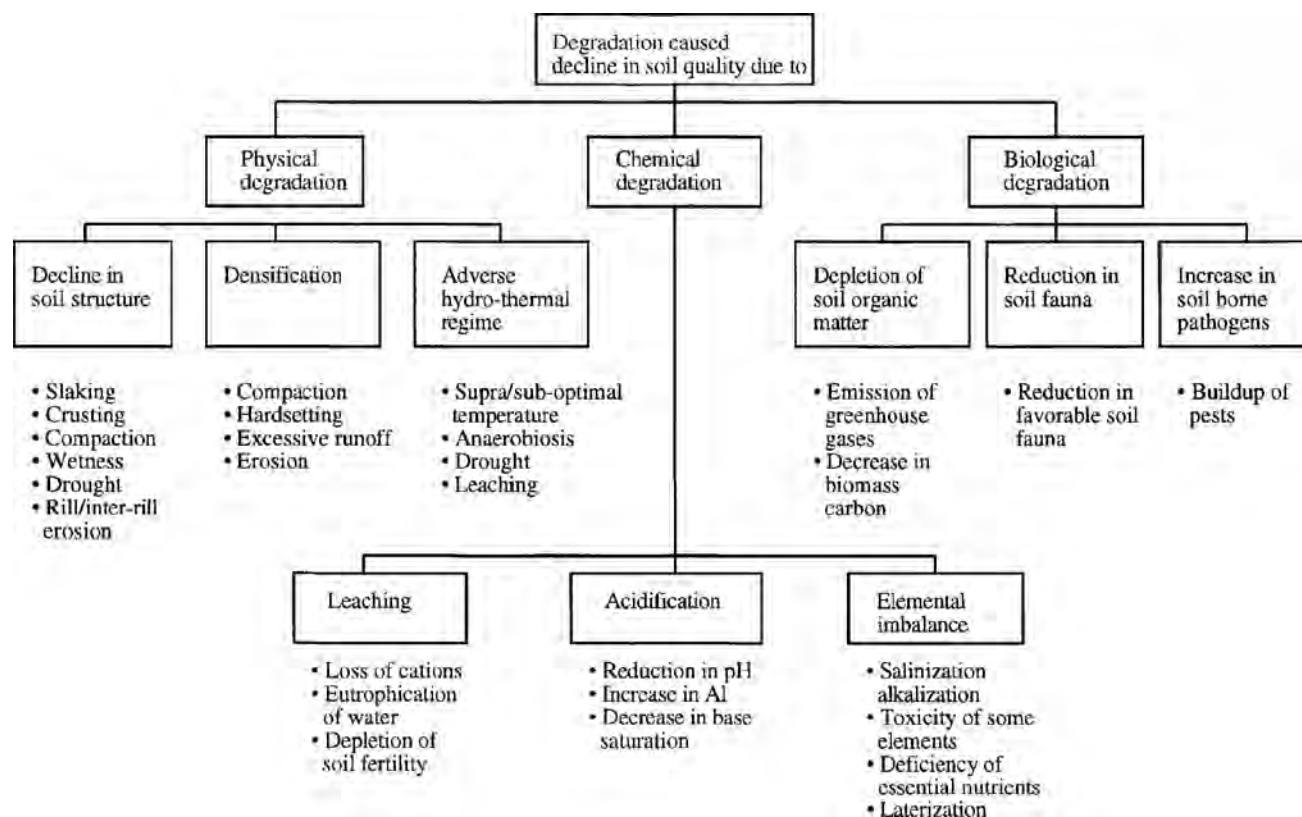


Fig. 1 Principal processes of soil degradation.

Source: From Lal & Stewart.^[5]

Natural Processes

One of the natural degradative processes is “laterization,” which is a process of iron accumulation in soils. At its extreme, the process involves intense weathering and leaching. Intense weathering leads to the breakdown

of all minerals except quartz. Intense leaching of soil removes all the soluble salts and some of the iron (Fe) and aluminum (Al). These processes lead to a predominance of kaolinite and a depletion of plant-available nutrients. As a result, high concentrations of Fe and Al cause complex fertility management problems.

Hard-setting is another natural process. Hard-setting soils are characterized by weakly cemented surface horizons in deeply weathered soils of low inherent fertility and low soil organic matter content.^[6] Such soils have massive, single-grain or weak pedality of the surface. These soils set hard on drying and have a narrow range of moisture content at which they can be managed. Hard-setting

Table 1 Processes of soil degradation.

Physical processes	Chemical processes	Biological processes
Decline in soil structure, e.g., compaction, crusting, etc.	Leaching and acidification	Decline in soil biodiversity
Accelerated erosion by water, wind, gravity, and glaciation	Nutrient depletion	Decrease in soil organic matter content
Decrease in effective rooting depth	Nutrient imbalance: i) toxicity, e.g., Al, Mn, Fe ii) deficiency, e.g., P	Reduction in soil biomass carbon
Anaerobiosis or excessive wetness	Sodication	Increase in pests and pathogens
Excessive drainage or droughtiness	Laterization	

Table 2 Natural and anthropogenic processes of soil degradation.

Natural processes	Anthropogenic processes
Set-in-motion by natural causes	Set-in-motion by human intervention
Laterization	Accelerated erosion
Fragipan formation	Impeded surface drainage
Clay pan formation	Soil compaction
Hard-setting	Fertility depletion

characteristics are accentuated by physical and biological processes of degradation.

Pan formation within the soil profile is another natural degradative process. There are two principal types of pans. Fragipans are pedogenic subsurface horizons characterized by their medium texture; a high content of silt, very fine sand, and low-to-moderate clay content; high bulk density; low organic matter content; and brittle when moist and hard when dry.^[1] These pans affect rooting depth and plant-available water reserves. In contrast to Fragipans, clay pans are not brittle and may not have high density. *In situ* formation of a clay pan in the B horizon happens due to translocation of fine clay to the subsoil. The physical barrier formed by the clay pan is root-restrictive and can also lead to impeded drainage.

Anthropogenic Processes

Human activities have created the formation of “anthropic soils.” Anthropogenic soils may be due to urbanization, waste disposal, or other drastic perturbation.

FACTORS AND CAUSES OF SOIL DEGRADATION

Factors of soil degradation are ecosystem characteristics. Notable among these are climate (e.g., rainfall and temperature), terrain characteristics, and vegetation (Fig. 2). Factors are agents and catalysts and accentuate the rate of different processes of soil degradation. The causes of soil degradation are anthropogenic activities, which grossly alter the influence of processes and factors. Important causes of soil degradation include agricultural activities, industrial activities, and urbanization (Fig. 3 and Table 3). The causes of degradation are driven by demographic pressure, human needs, and other issues related to the human dimension. Consequently, increase in demographic pressure aggravates the effects of processes and factors of soil degradation.

EFFECTS OF SOIL DEGRADATION

Soil degradative processes are influenced strongly by biophysical factors and socioeconomic causes (Fig. 4). The

effects of physical processes of soil degradation are exacerbated by anthropogenic activities in ecologically sensitive ecoregions. The soil erosion problem is extremely severe when sloping lands and shallow soils are cultivated for row crop production in ecologically sensitive ecoregions such as the Himalayan–Tibetan ecosystem, Andean region, the Caribbean, and the West African Sahel. Deforestation in ecologically sensitive regions can cause severe degradation of soil and water resources, e.g., tropical rainforest ecosystem.

Degradation of soil has both economic and ecologic consequences. The most direct and economic effect of degradation of soil resources is the loss of agricultural productivity. The adverse effects of soil degradation on productivity are accentuated by a progressive decline in the per capita arable land area at national and global scales. The per capita arable land is rapidly decreasing due to an increase in population and the degradation of soil resources. Global per capita arable land area of 0.3 ha in 1985 declined to 0.23 ha in 2000, 0.15 ha by 2050, and 0.14 ha by 2100. Countries with per capita arable land area of less than 0.15 ha were 5 in 1960, 23 in 1990, and will be 50 in 2025.^[7] Mismanagement, neglect, and exploitation can deplete or even ruin the fragile soil resources.

While the per capita arable land area is decreasing, its productivity is also being reduced by several processes of soil degradation, e.g., erosion, desertification, leaching, etc. The effects of soil degradation on productivity are not easily understood because they are often cumulative and may be observed long after the degradation has occurred. Furthermore, productivity is the net result of all factors of production, including climate, and the effects are masked by management and the prevailing weather conditions. In addition to the direct effects of degradation on productivity, there are many indirect or hidden effects due to alterations in soil properties. In fact, there are no reliable estimates of worldwide loss in productivity due to different processes of soil degradation.^[8,9]

Ecologic consequences of soil degradation are major environmental concerns. The annual discharge of chemicals to world rivers is phenomenal and rapidly increasing. Chemical discharge into natural waters from agricultural land may be from fertilizers and pesticides, feedlots, and

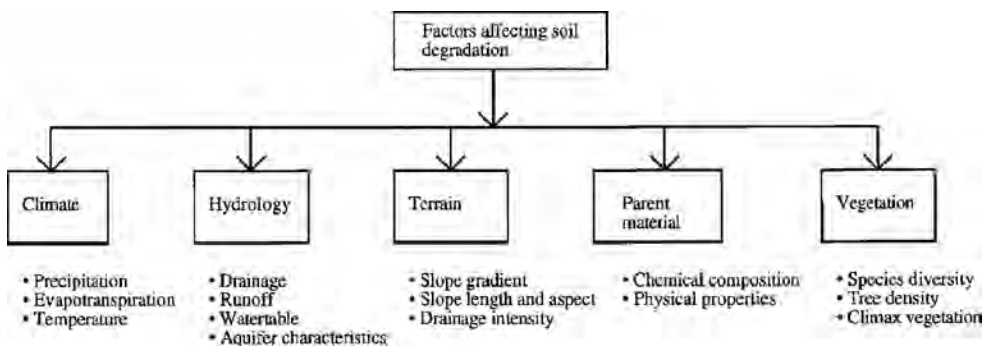


Fig. 2 Major natural and anthropogenic factors affecting soil degradation.

Source: From Lal & Stewart.^[5]

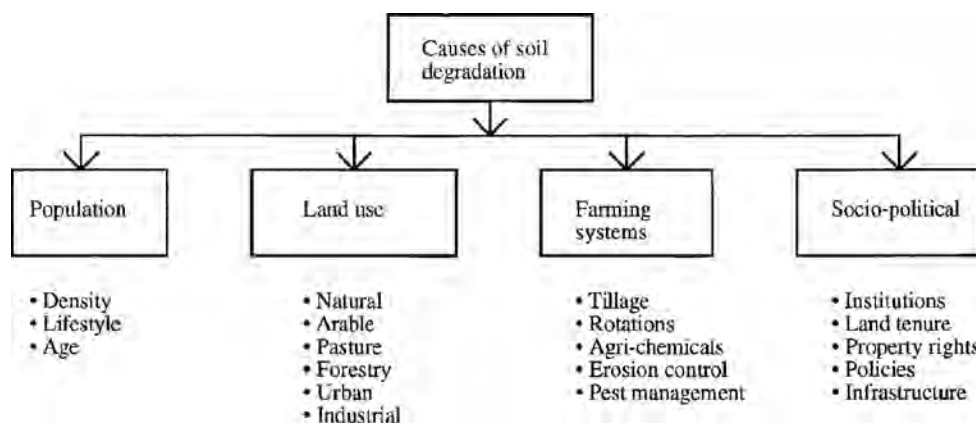


Fig. 3 Principal causes of soil degradation.

Source: From Lal & Stewart.^[5]

from sediment. Risks of agrochemical contamination of natural waters are extremely high in both developing and developed economies.

Emission of radiatively active gases from soil-related processes is another principal ecologic consequence of soil degradation. The emission of CO₂ and other greenhouse gases (e.g., CH₄, N₂O, and NO_x) by soil erosion can be enormous.^[10] Yet, world soils can be a major source or sink for radiatively active gases depending on the land use and management system adopted.^[11]

GLOBAL EXTENT OF SOIL DEGRADATION

The problem of soil degradation has been accentuated drastically by changes in land use witnessed since the 18th century. Despite the widespread awareness about the problem, exact estimates of the extent and severity of soil degradation are not known. According to some estimates, about 2.0 billion hectares of land is degraded to some degree out of the world's total land areas of 13.4 Bha.^[12,13] Furthermore, Asia and Africa combined account for a total of 1.24 Bha of degraded land. The most predominant degradative processes are the

accelerated erosion by wind and water, chemical degradation and soil fertility depletion, and physical degradation resulting from a decline in soil structure (Table 4). Estimates of soil degradation in arid climates, due to wind erosion and desertification, are equally alarming (Table 5).

DATA ACCURACY AND RELIABILITY

Statistics, such as those shown in Table 5, play an important role in creating awareness about the hazard, and in formulating a global strategy in addressing it. Indeed, if these statistics are correct, the challenge they present to the human race is one of the greatest because soil resources are finite and non-renewable over the human time-span.^[14,15] Even though the statistics can be approximately correct, it is a matter of greatest urgency for decision makers to do something about it. However, careful analyses of the data may reveal several problems:

1. There may be a problem with the definition. The term soil degradation is vague and highly subjective. To avoid ambiguity, soil degradation needs to be defined

Table 3 Causes of soil degradation.

Anthropogenic causes		Natural causes
Agricultural activities	Industrial activities and urban development	Climate and land
Deforestation: felling trees and biomass burning	Mining: surface or strip mining and subsurface mining	Rainfall: intensity and amount energy load
Cultivation: plowing, leveling, and land forming	Waste disposal: industrial and urban wastes and by-products	Wind: velocity direction
Farming/cropping systems: cropping intensity, grain crops vs. tree crops, pasture, and stocking rate	Acid rain: intensity and chemical agents	Temperature: maximum, minimum seasonal and diurnal fluctuations length aspect
Agricultural chemicals: fertilizers and pesticides	Nonagricultural land uses: urban development civil structures and mining	Terrain: slope gradient and length aspect Parent material: geological formation Vegetation: age and species composition

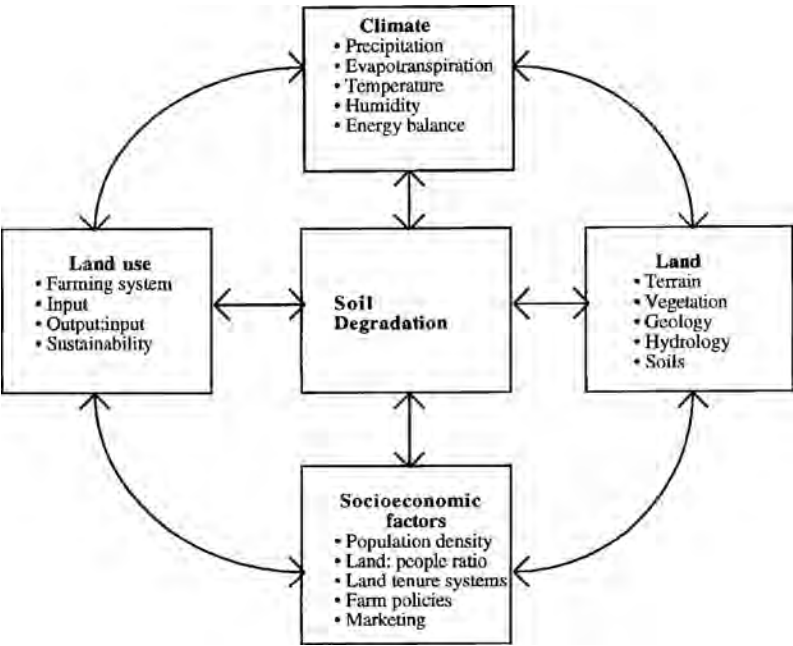


Fig. 4 Interdependence of soil degradation on biophysical and socioeconomic factors.
Source: From Lal & Stewart.^[5]

quantitatively.^[2] To do so is to delineate threshold values or critical limits of soil properties beyond which soil’s life-support processes are severely jeopardized. Important soil properties whose critical limits need to be defined are rooting depth, plant-available water capacity, soil organic matter content, soil structural attributes, capacity and intensity factors, and limiting levels of principal nutrients. These limits are not known and vary among soils, climates, farming systems, land uses, and management.

- 2. There is a problem with the methodology. There have been only few attempts to standardize the methodology and validate the estimates by ground truthing representative soils and ecoregions. Consequently, estimates of soil degradation by the same process vary widely because of the differences in methodology and criteria used.
- 3. Soil degradation is estimated on the basis of visual observations, reconnaissance surveys, and opinions

of regional/local organizations. It is rarely related to actual and potential productivity.

- 4. There is a problem with double accounting. The same soil prone to more than one degradative process may be counted as many times.
- 5. Productivity is affected by land use, inputs, improved technology, and management. Estimating productivity without considering of important technology and management processes leads to erroneous data.

CONCLUSION

Understanding principal processes, factors, and causes of soil degradation is an important prerequisite to our ability to reverse the degradative trends and to preserve a healthy state of global environment. Being a complex problem, it requires a systematic and coordinated effort to effectively address the major issues involved. These issues include: 1) obtaining reliable estimates of the land area affected by different soil degradative processes; 2) standardizing definitions and methods of assessment

Table 4 Global estimates of soil degradation by different processes.

Process of soil degradation	World’s total degraded area (10 ⁶ ha)	Regional degraded area (10 ⁶ ha)	
		Asia	Africa
Water erosion	1100.0	433.2	227.3
Wind erosion	550.0	224.1	187.3
Chemical degradation	235.8	74.7	59.3
Physical degradation	78.6	19.8	15.0
Total	1964.4	747.0	494.2

Source: From WRI^[12] and Oldman.^[13]

Table 5 Global estimates of desertification in arid, semiarid, and subhumid regions.

Land use	Total area (10 ⁶)	Land affected by desertification	
		Area (10 ⁶ ha)	% of total
Rangeland	3700	3100	80
Rainfed cropland	570	335	60
Irrigated land	131	40	30
Total	4409	3475	70

Source: From UNEP.^[16]

of soil degradation; 3) evaluating on-site and off-site impacts of soil degradation; 4) developing appropriate measures of soil restoration; and 5) assessing economic and environmental impacts of restoring degraded soils and ecosystems.

REFERENCES

1. Lal, R.; Hall, G.F.; Miller, F.P. Soil degradation. I. Basic processes. *Land Degrad. Rehab.* **1989**, *1*, 51–69.
2. Lal, R. Degradation and resilience of soils. *Phil. Trans. R. Soc. Lond. B.* **1997**, *352*, 997–1010.
3. Lal, R. Soil quality and agricultural sustainability. In *Soil Quality and Agricultural Sustainability*; Lal, R., Ed.; Ann Arbor Press: Chelsea, 1998; 3–12.
4. Lal, R. Soil quality and food security. In *Soil Quality and Food Security: The Global Perspective*; Lal, R., Ed.; CRC Press: Boca Raton, 1999; 3–16.
5. Lal, R.; Stewart, B.A. Need for action: research and development priorities. *Adv. Soil Sci.* **1990**, *11*, 331–336.
6. Mullins, C.E.; MacLeod, D.A.; Northcote, K.H.; Tisdall, J.M.; Young, T.M. Hard-setting soil's behavior, occurrence and management. *Adv. Soil Sci.* **1990**, *11*, 37–108.
7. Engehnar, R.; LeRoy, P. *Conserving Land: Population and Sustainable Food Production*; Population and Environment Program, Population Action International: Washington, 1995; 48 pp.
8. Lal, R. Soil erosion impact on agronomic productivity and environment quality. *Crit. Rev. Plant Sci.* **1998**, *17*, 319–464.
9. Oldeman, L.R. *Soil Degradation: A Threat to Food Security*; ISRIC Report 98/01; ISRIC: Wageningen, 1998.
10. Lal, R. Global soil erosion by water and carbon dynamics. In *Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1995; 131–142.
11. Lal, R. Soil management and restoration for C sequestration to mitigate the accelerated greenhouse effect. *Prog. Environ. Sci.* **1999**, *1*, 307–326.
12. WRI. World resources institute, towards sustainable development. In *A Guide to the Global Environment*; WRI: Washington, 1992; 385 pp.
13. Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 99–118.
14. Lal, R. Land degradation and its impact on food and other resources. In *Food and Natural Resources*; Pimentel, Ed.; Academic Press: New York, 1989; 85–140.
15. Pierce, F.J.; Lal, R. Soil management in the 21st century. In *Soil Management for Sustainability*; Lal, R., Pierce, F.J., Eds.; Soil and Water Conservation Society: Ankeny, 1991; 175–179.
16. UNEP. *Status of Desertification, and Implementation of United Nations Plan of Action to Combat Desertification*; UNEP: Nairobi, 1991; 77 pp.

Desertification: Biological Productivity Decline

Uriel N. Safriel

*Department of Ecology, Evolution and Behavior, Hebrew University of Jerusalem,
and Center of Environmental Conventions, Jacob Blaustein Institutes for Desert
Research, Jerusalem, Israel*

Abstract

Desertification is a persistent reduction of land's biological productivity of economic value—a terminal state of the land degradation process. Land degradation and desertification (LDD) can occur anywhere but the focus of this entry is on drylands, whose natural low productivity is water limited. LDD is a syndrome of interactive biophysical processes whose direct drivers are land resource overexploitation by land users. Underlying causes of LDD are economic, demographic, social, cultural, and political, operating across scales. About a quarter of global land is degraded and active LDD is ongoing. LDD is of concern—it is believed to contribute to poverty, migrations, refugees, and conflicts; it undermines global food security and the emissions from degraded lands exacerbate global climate change, which is projected to increase the contribution of drought to LDD and has already increased drying and a geographical expansion of some drylands. Some dryland communities demonstrate resilience to the underlying causes and take measures for avoiding LDD and the United Nations Convention to Combat Desertification is instrumental in assisting developing countries addressing their LDD.

INTRODUCTION

“Desertification” is a human-driven process, which impinges on a significant part of earth's natural capital, land, by reducing its productivity. Desertification is probably as old as the pastoral and farming livelihoods but has attracted attention only in the 20th century, such that as it turned into the new millennium, desertification has been ranked among the greatest contemporary environmental problems.^[1] Although literally the term links with deserts, desertification hardly occurs in deserts. Rather, desertification affects non-desert areas, all around the globe, and is of major concern, as it is a driver of poverty, mainly in the rural areas of the developing world, but at the same time puts at-risk food security at the global scale. This entry explains what desertification is and describes how it occurs, where, by whom, and why. It then elaborates on its implications to people across scales, from local to global. This is followed by scanning through the measures taken by people and the international community to address this threat to human survival and global sustainability. The entry closes with the effect of global climate change on desertification, now and in the future.

WHAT IS DESERTIFICATION?

Deserts—areas of poor vegetation cover and of the driest climate on earth—were molded by climatic processes, such as scarce rainfall and intense evaporation. These jointly constrain plant productivity such that much of the desert

appears lifeless. As the world's deserts were formed by natural climatic processes over long periods during which they have grown and shrunk, the term “desertification” can grammatically describe desert formation. However, the term was apparently coined first in a 1949 publication,^[2] not for describing a climate-driven outward spreading of deserts, but for a human-driven decline of land productivity to levels reminiscent to those of deserts, while climate remained a non-desert one. The term came into widespread international use only as of the 1970s, then being defined and redefined ever since. All definitions, however, related to a land attribute critical to all people, everywhere and always—the marketed biological products, food and fiber, which are derived from the land's biological productivity. Thus, desertification is best expressed by a persistent reduction, relative to the potential, of the land's biological productivity of an economic value.^[1,3]

DESERTIFICATION AND LAND DEGRADATION, DRYLANDS AND NON-DRYLANDS

The decline in biological productivity of economic value is driven by biophysical processes of “land degradation,” which can operate on lands used by people anywhere on earth but mostly affect lands whose productivity is constrained by water. Water is one of the raw materials required for the biological production process, but atmospheric carbon dioxide (CO₂) and soil minerals, as well as light. Each of these can limit productivity, even if all the others are available in abundance. Land degradation occurs in areas

of low water and high evaporation, thus making soil water the limiting factor. This water scarcity exists in all lands exposed to a potential annual evaporative loss that is at least about one and one-half times greater than the mean annual gain through rainfall. These are “drylands,” depending on their degree of aridity, they are classified into desert drylands (hyperarid and arid drylands) and non-desert ones (semiarid and dry subhumid drylands; Fig. 1).

The United Nations Convention to Combat Desertification (UNCCD) confined its mandate (to “combat desertification”) to degradation of drylands only, excluding the most arid ones, the hyperarid desert drylands. Also, the stakeholders of the UNCCD habitually use the two terms, “land degradation” and “desertification” interchangeably or jointly. However, there is an increasing tendency to address “land degradation” as a process, while regarding “desertification” as a terminal, extreme state of this process and to use both terms for processes impinging on biological productivity in non-dryland areas too.^[3,4] These tendencies have driven an increasing usage of an acronym for Land Degradation and Desertification, namely, LDD.

HOW LDD IS CAUSED?

Biological productivity depends on the land’s soil, which provides trees, forage and crop plants physical anchor, and raw materials for the production process. The loose and aerated texture of the soil is due to its organic matter that

binds the soil’s mineral particles together. The binding creates aggregates with air spaces between them that retain water in which the plant mineral nutrients are dissolved. This particulate texture of soil, however, makes it vulnerable to the mechanical force of wind, which blows away the soil’s fine components what often leaves sandy areas and dunes, and of rainwater, which carries soils away. These soil erosion processes reduce fertility as most nutrients are confined to the uppermost soil layers, which are directly exposed to the eroding forces. As the rate of soil formation is extremely slow, for all practical purposes soil is non-renewable and hence topsoil loss is one of the major causes of persistent productivity decline. It is plant cover that provides soil protection, by mitigating wind force and reducing the impact of raindrops that would otherwise destroy the delicate soil texture. Breakage of the aggregates results in soil crusting, leading to land surface sealing that reduces soil water absorption. Rather than infiltrating and enriching soil moisture, rainwater turns into surface runoff that erodes the soil or leaching away dissolved soil minerals and its organic compounds. These losses directly reduce fertility and indirectly reduce the water-holding capacity of the soil, what also slows down the soil microbial activity essential in nutrient recycling. Thus, due to soil erosion plants may lose their physical support as well as their production resources. Another major driver of declining productivity is soil salinization. Almost all types of water, including rainwater, contain some dissolved salts, which are left in the top soils after soil moisture evaporates or is transpired by the plants. This accumulated salinity reduces plants’ ability to suck moisture from the soil, and the resulting soil sodicity (predominance of sodium ion in soil moisture) limits water absorption by soils. To conclude, while climatically driven water scarcity makes dryland productivity low compared to most other lands,^[1] drylands are vulnerable to soil erosion and salinization (Fig. 2), the major physical and chemical processes, respectively, leading to LDD, whose ultimate expression is productivity loss.

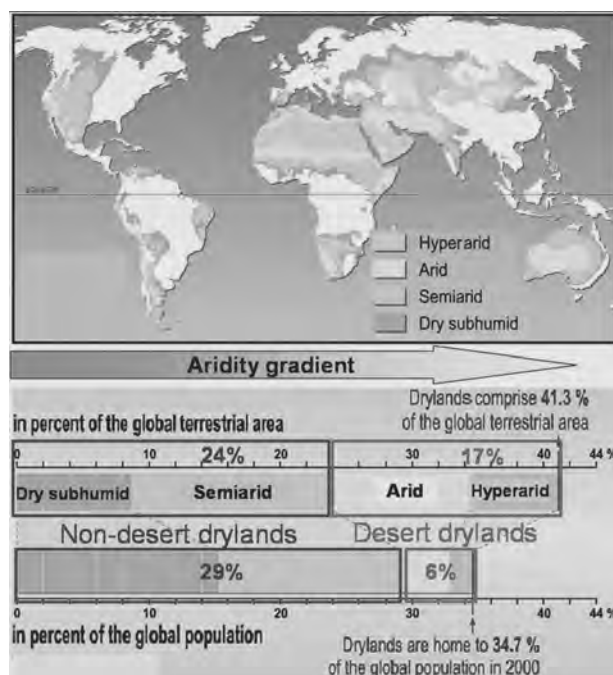


Fig. 1 The global drylands. Geographic layout of the four sections of the aridity gradient, their percent share of global land, their human populations, and their percent share of global population.

Source: Adapted from Adeel, Safriel, et al.^[1]

WHO CAUSES LDD?

Depending on whether the biological products of economic value are derived directly from wild plants (e.g., woody plants used for timber and firewood and other edible wild species) or indirectly from free-ranging livestock feeding on wild plants or from cultivated crop plants; lands and their productivity are managed by their users through interventions expressed in natural resource exploitation rates, livestock stocking rates, and soil fertility levels. These interventions sustain people, societies, and their prosperity, yet they often pose potential threats to the land’s productivity, which could ultimately impinge on the land users themselves and on others. This is because while all productivity is derived from plants, their exploitative removal, either through grazing or harvesting, exposes the land to

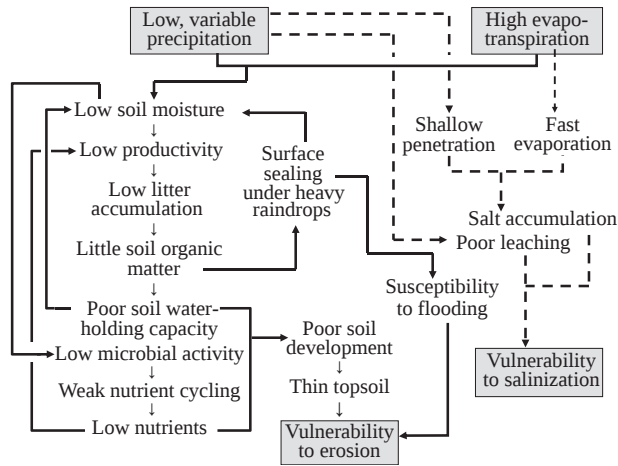


Fig. 2 The dryland syndrome—climate-driven natural processes driving soil biophysical processes leading to inherent low productivity, to erosion vulnerability (solid arrows, two negative feedback loops), and to salinization vulnerability (broken arrows).

Source: Adapted from Safriel.^[5]

degradation processes once a threshold of plant cover is removed and/or one of regeneration relative to exploitation rate is crossed.

Regarding the pastoral livelihood, high livestock densities maintained throughout a long period reduce the overall soil protective cover, exploit vegetation faster than it can regenerate, and export nutrients (through selling livestock and their products) faster than their natural renewability. These encourage non-forage plants to dominate the range, promote encroachment of bushes that obstruct livestock foraging movement, and reduce soil physical resilience to livestock trampling, which compacts soil surface, reduces rainfall infiltration thus increasing risks of powerful surface runoff that erodes the soil and changes the local topography. Individually or combined, these processes lead to a persistent rangeland productivity loss (Fig. 3). As to the farming livelihood, long fallow periods expose the croplands to erosion forces, but in short fallow periods, a high number of crops cultivated each year and a large amount of nutrients exported in the harvested agricultural products, individually or combined, are of a potential to cross the threshold of land degradation. This is through nutrient loss that is faster than its renewability and a decline in soil organic matter, which reduces soil permeability, and the water-holding capacity leading to increased erosion vulnerability. Irrigation, used mainly in drylands for increasing the productivity of otherwise rainfed agriculture, has dramatically increased not only the economic productivity but also the soil salinity. Dryland water is scarce and hence not used for leaching soil salinity and when used, nutrients are leached too. Furthermore, much of dryland water is brackish, and irrigation systems devoid of drainage cause water logging, leading to further soil salinization. The irrigation-driven increased economic productivity amplifies nutrient

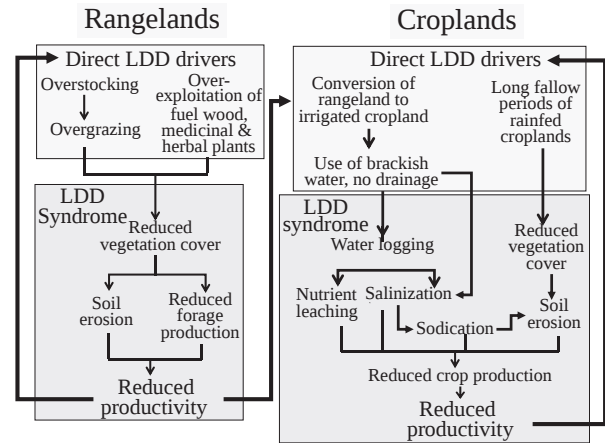


Fig. 3 Actions of land users that constitute the proximate, direct drivers (top boxes) of the LDD syndrome (bottom boxes) and their feedbacks and linkage (thick arrows), in each of the two prevailing dryland livelihoods.

export through harvest, which needs to be balanced by fertilizers. This combination of irrigation and fertilization could generate immediate economic gains, often succeeded by LDD (Fig. 3).

Practitioners of both livelihoods, but more often of the pastoral one, also exploit the woody plants of natural or planted woodlands and of rangelands, for timber and fuel wood, or remove all vegetation cover of woodlands or rangelands, to be replaced by expanding cultivation. These processes, too, expose large swaths of land to erosion processes and eventual LDD. To conclude, human land uses directed at increasing the inherently low dryland productivity of economic value are of a potential that is often materialized to bring about LDD. The persistent decline in productivity, rather than its aspired increase, attests to a preceding shift from grazing, cropping, and exploitation to overgrazing, overcropping, and overexploitation—activities of land users that constitute the direct, biophysical drivers of the LDD processes.^[6]

WHY LAND USERS CAUSE LDD?

It can be expected that land users are well adapted to their local environment and hence possess the knowledge for using their lands sustainably, i.e., without degrading it. As land degradation does occur in areas used by experienced users, factors beyond their control may be involved. Indeed, indirect drivers of change, or underlying causes of LDD that affect the direct, proximate causes that change non-degraded to a degraded land can be invoked (e.g., Fig. 3). These can be population (or demographic), economic, technological, policy and institutional, cultural, and political drivers.^[7] Each of these (Fig. 4) can affect the direct LDD biophysical drivers alone, jointly, linearly, or interactively, across different temporal and spatial scales.^[8]

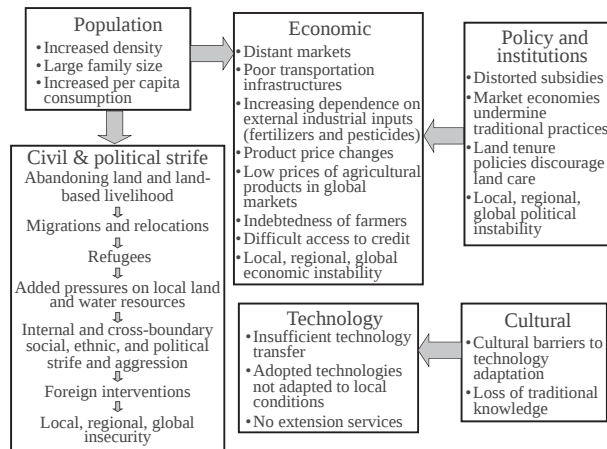


Fig. 4 Underlying socioeconomic and policy indirect LDD drivers and their interactions.

Source: Adapted from Reynolds, Grainger, et al.^[8]

However, in spite of an increasing research and political interest in these LDD drivers, there is not enough field evidence, hence also agreement, regarding the role and relative significance of each of the variables that structure the network of pathways to LDD, and therefore the LDD “syndrome” is illustrated by paradigms that can be perceived as structured programs for further research. An “LDD paradigm” (Fig. 5) integrates elements of the Desertification Paradigm,^[9] the Dryland Livelihood Paradigm,^[10] and the Dryland Development Paradigm.^[11] Under this paradigm, LDD is driven by mutually interactive biophysical processes within the land’s “Ecological System” and by mutually interactive socioeconomic and policy processes within a “Human System,” integrated into an interactive “Human–Environment System” that oscillates between land’s sustainable use and LDD. The LDD paradigm attends the effect of LDD on human well-being at different spatial scales—effects that often further exacerbate rather than mitigate the LDD state. These are mainly poverty at the local scale, and migrations, refugees, conflicts, and foreign interventions at the regional and even global scales. In short, land users are driven to overexploit land, mostly by economic factors that in themselves can be driven by demographic variables, social and cultural factors, and by economic and social policies and political processes molded at diverse regional and global processes on which they have no control.

WHERE AND HOW MUCH LDD THERE IS?

As biological productivity expressed in market-traded products of economic value is regularly monitored a detected decline in the production of crops or in livestock-derived products can be used to delineate LDD-affected areas and quantify their spatial extent. However, the quantities of

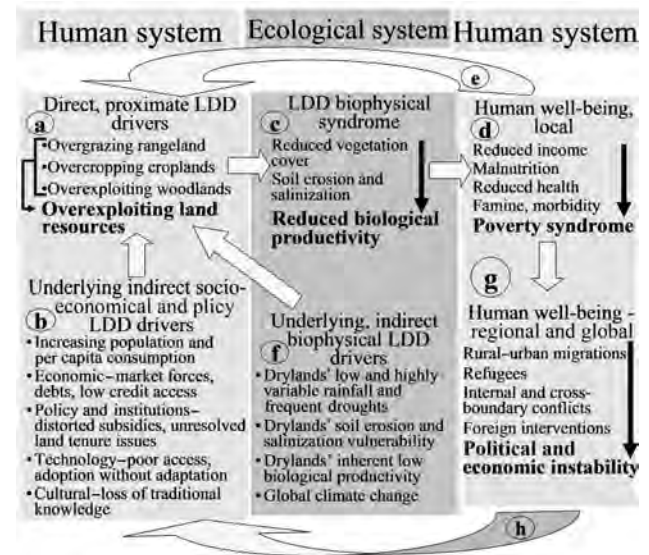


Fig. 5 The LDD paradigm—interactions and feedback loops: Direct proximate LDD drivers (A) (detailed in Fig. 3), driven by underlying socioeconomic and policy drivers (B) (detailed in Fig. 4) generate the LDD syndrome (C) (detailed in Fig. 3); the reduced human well-being impinges on the direct drivers thus further exacerbating LDD (E); underlying indirect biophysical drivers also drive people to further exacerbate LDD (F) (detailed in Fig. 2); the locally affected human well-being (D) also has regional and global repercussions (G); and human well-being at all levels impinges on the underlying LDD drivers (H). Vertical arrows indicate the linear effects of variables from top to bottom of lists, while bullets indicate the independent effects of each variable.

marketable agricultural products per unit area do not necessarily reflect land productivity, as these much depend on the land users’ management decisions, driven mainly by economic considerations. Therefore, instead of monitoring marketed land products scientists turned to monitoring land and soil attributes serving as indicators of productivity. As different studies use different indicators and measurement methods, it is not surprising that five global assessments during the last decades resulted in LDD estimates ranging from 15% to 63% of global land and 4% to 74% of its subset of global drylands.^[12] A critical review of these studies suggested (with “medium certainty”) that 10–20% of the global drylands have been degraded by year 2000,^[9] while an earlier study qualified 12% of non-dryland as degraded.^[13] The latter study qualified 5.7% and 3.8% of Asian and African lands as degraded, respectively, and 1.9%, 1.7%, 1.2%, and 0.8% of lands in South America, Europe, North America, and Australasia as degraded, respectively, which combined to classify 15.1% of global land as degraded.^[14]

The development of frequent and accessible time series of earth monitoring from space enabled assessing not just the amount of degraded land but the rates of ongoing LDD. Satellite-born sensors of radiation reflected and emitted from the earth’s surface, and especially from its vegetation

cover, are used for following the course of biological productivity changes within given time periods. Most studies attend selected regions but one^[15] addresses the dynamics of biological productivity of vegetation cover at a global scale. This 23-year (1981–2003) analysis of remote sensing images revealed persistently declining overall productivity (i.e., not just the subset of direct economic value) at 24% of the global land, on which 1.5 billion people reside. It mainly occurred in Africa south of the equator, southeast Asia and South China, north-central Australia, the Argentinean Pampas, and swaths of the Siberian and North American taiga. The study also found that land degradation in China is more severe in forests than in croplands. However, measures taken to control the productivity–rainfall interaction need to be refined in making such assessment fully reliable.^[16] Nevertheless, the available studies demonstrate that as much as a quarter of the global land is already degraded, and active LDD is in force, which points at the vulnerability of non-degraded lands around the globe, especially in drylands.

WHY LDD IS OF CONCERN?

LDD reduces the capacity of earth to provide food; hence, it is of concern not only to the direct land users but also to humanity at large. Furthermore, even the LDD's local effect on the well-being of the direct land users is implicated with generating externalities of a global concern, as the local LDD-driven poverty leads to within-country, cross-boundary, and overseas migrations of refugees, which often trigger conflicts of variable severity and foreign and international interventions (Figs. 4 and 5). This component of the LDD paradigm is widely alluded to, though hard data on the causation and the dimension of each of these processes at the global scale are scarce. Regarding poverty, countries' statistics of poor rural land users do not distinguish between LDD-driven poverty and poverty not driven by LDD; it adds the poverty of land users with that of rural people who do not live off the land, as well as that of the country's dryland and non-dryland inhabitants. Thus, only localized statistics (e.g., 68–84% poverty in the drylands in northern Kenya that is higher than in other, non-dryland parts of the country) suggests a prevalence of poverty in drylands of developing countries and also has led to a rough estimate of about nearly one billion people in poor rural drylands.^[17] A direct, quantifiable linkage between poverty and LDD at the global scale is so far provided only by the observation that the poverty indicators—infant mortality and malnutrition of under-five-year-old children track the trajectory of desertification severity across the global aridity gradient (Fig. 6). Similar to the LDD–poverty nexus, there is no hard data on the linkage among poverty, migrations, refugees, and conflicts, and evidence for LDD being instrumental is circumstantial. Most cited are migrations following severe droughts, while only a few cases present a direct linkage, e.g., the Tambacuanda region of Senegal's

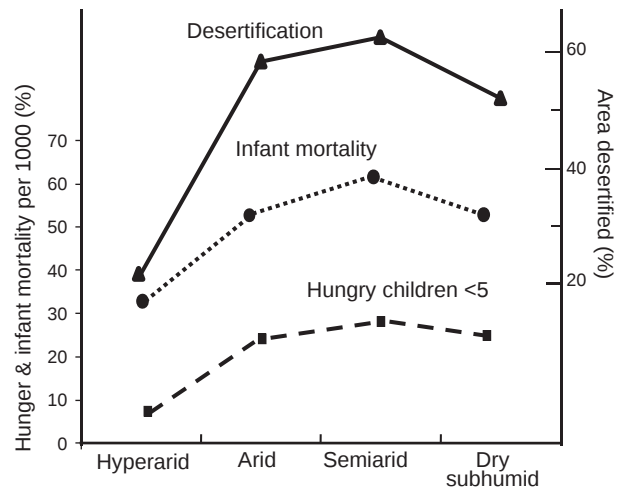


Fig. 6 Trends of LDD and poverty indicators across the aridity gradient of the global drylands.

Source: Adapted from Safriel.^[18]

1960s migrations driven by soil erosion-triggered decline of crop production—to the capital, other cities, other African countries, and Europe.^[19] Social scientists also disagree on the role of environmental degradation (LDD included) in generating conflicts and violence and concluded that empirical evidence is insufficient to demonstrate a tangible effect.^[20] However, comprehensive research of the Darfur conflict^[21] asserted that LDD in both croplands and rangelands has been one of the major factors in that crisis but it has been also intertwined with a range of other socioeconomic and political issues. Even if LDD does not always generate local poverty of regional and global repercussions, any local loss of land productivity undermines food security not only locally, but also at the global scale. It has been estimated that for feeding the global population by year 2050, 8.9 million km² of good cultivable land would be required, on top of the 15 million km² already under cultivation.^[22] However, only 4.4 million km² of cultivable land will be available.^[23] Therefore, LDD which practically converts cultivable to non-cultivable land could dramatically aggravate the already fragile global food security. LDD is of concern not only due to loss of crop and forage production but also due to loss of other benefits that people derive from the land, such as water regulation and purification, local climate amelioration, pollination, biodiversity conservation, and others. These benefits, called “ecosystem services,” are an expression of the land’s functionality as an ecological system. Adopting a holistic approach, LDD is increasingly described as a persistent, substantial reduction in a bundle of services provided to humans by cultivated, range, or natural ecosystems.^[1,24] The persistent decline in the provision of one of these dryland ecosystem services, carbon sequestration (the uptake of atmospheric CO₂ by plants and transforming it to plant biomass and litter) is of global concern, too. All dryland vegetation and all dryland soils store 14% and 20% of the

global organic carbon, respectively.^[9] LDD-associated soil erosion and loss of plant cover and plant productivity deplete this land's storage of organic carbon. This locally removed organic carbon is eventually oxidized and emitted back to the atmosphere, thus increasing atmospheric CO₂ concentration and exacerbating climate change, globally. Indeed, drylands' LDD-driven emissions are estimated to comprise about 4% of the total global emissions from all sources combined (in year 2000),^[11] and added LDD would further amplify global climate change.

Land degradation and desertification (LDD) data are of GLASOD,^[13] with the "low degradation" category excluded; data of infant mortality per 1000 births and percentage of hungry children under the age of 5, drylands of OECD countries excluded, are from the Millennium Ecosystem Assessment.^[9]

WHAT HAS BEEN DONE AND WHAT CAN BE DONE?

That most lands used in drylands are not yet affected by LDD suggests that drylands can be used sustainably. Many cases have been documented, mostly in East and West African drylands, in which rural communities adapted to rapid population growth, globalization, market development, and technological change and avoided LDD. These apparently detrimental changes triggered innovative actions: investments in more intensive yet sustainable land use, which helped reaping benefits of the enlarged market opportunity; exploitation of local bioclimatic and social comparative advantages; access to technologies that increased land and labor productivity faster than population growth; and improved access to growing markets.^[25] Avoiding LDD can be also achieved by adding income-generating "alternative" land-independent livelihoods like tourism and aquaculture,^[26] or generating supplementary income from seasonal migration to urban centers.^[19]

Nevertheless, it is developing countries that share most of the global drylands,^[9] as well as probably the most rural poverty and LDD risks. Noting these facts and being aware of the global threats of dryland LDD, the international community negotiated and adopted the UNCCD, with the intention of coordinating and promoting assistance of industrial (developed) to developing countries in addressing their LDD problems. Entering into force in 1996, the 195 country parties of the UNCCD together with the Secretariat and the subsidiary bodies of the Convention succeeded in raising awareness, encouraging knowledge and research, and promoting capacity-building and partnerships in projects on the ground, as well as installing mechanisms for assessing success. A major thrust of the international community is to drive the political process of setting a sustainable development goal—striving for Land Degradation Neutral World—and its associated target of achieving Zero Net Land Degradation (ZNLD) by 2030, initiated at the United

Nations Conference on Sustainable Development convened in June 2012 in Rio De Janeiro. The rationale behind this target is that completely halting further decline of the productive, non-degraded land within a relatively short time is unattainable. However, the amount of non-degraded land that becomes degraded within a time period can be offset by a similar amount of already degraded land, whose productivity is restored within the same time period. Once this offset is attained, the net rate of LDD is zero. Setting such goals and targets can help shape expectations and create the conditions for all stakeholders to assess progress and take appropriate action in addressing LDD.

As to reducing LDD for addressing the ZNLD target, "sustainable land management" (SLM) is widely prescribed. In practice, any land that supports the user does not lead LDD to qualify as SLM. More specifically, practices of land use need to adaptively maintain the resilience of the used ecosystems to all LDD drivers, such that the thresholds in ecosystem functions will not be crossed and the service provision will be sustained.^[27] Avoiding degradation may be easier and cheaper than restoring already degraded land but, as land is often degraded inadvertently while restoration is planned, its success prospects are relatively high. The first stage is removal of the LDD causes, to be followed by selecting from the rich literature of ecological restoration,^[28] the toolbox appropriate for the local state and the set restoration target.

DESERTIFICATION AND CLIMATE, AND CLIMATE CHANGE

Drylands are not only exposed to climate-driven water scarcity but also to high between-year variability. This includes relatively short periods (<5 years) during which rainfall and hence productivity remains far below the long-term average. These periods are termed "droughts," and since their associated productivity decline is reversible, this productivity decline does not qualify as desertification. It can be expected that a drought-driven productivity decline would persist after the drought terminate, provided that it has been severe and exceptionally long. Such events, if ever occurred, have not been reliably documented. On the other hand, drylands in which human-induced LDD processes have been already initiated are likely to be pushed by a drought to cross the desertification threshold, such that productivity decline would persist after the drought terminates. Such cases are rare, too. Even the reduced productivity during the Sahel droughts of 1968–1973 and 1982–1984 in the 20th century, which took a heavy toll on properties and lives and aroused the world's concern that culminated in adopting the UNCCD, was restored once the droughts terminated.^[29] Nevertheless, although droughts alone do not cause LDD, they do exacerbate LDD and its effect on human well-being, and hence droughts are listed among the underlying, indirect biophysical LDD drivers (Fig. 5). The role of drought in

desertification is laid out in the full name of the UNCCD—a convention “to combat desertification, in those countries experiencing serious droughts ...” and in its objective, that is not only to “combat desertification” but also to “mitigate the effects of droughts.”^[30] In spite of the difference between “combating” and “mitigating,” an emerging and widely used acronym for the UNCCD’s subject matter is Desertification, Land Degradation, and Drought, or DLDD.

Coining the DLDD acronym is partly justified given that the frequency of droughts and their severity is projected to increase, driven by the global warming and climate change. Indeed, climate change (and poverty) is projected to intensify desertification by 2050 in all the four Millennium Ecosystem Assessment’s scenarios of global governance.^[1] Climate change’s effect on global aridity is already evident. Comparison of the climatic conditions, precipitation, and temperatures that prevailed in 1931–1960 and used for setting the drylands’ boundaries, with those prevailing in the period of 1961–1990, demonstrated that the spatial extent of the African hyperarid and arid drylands increased by 51 and 3 million hectares, respectively, which comprise 1.7% and 0.1% of their area extent in 1931–1960, respectively. More significant is the “loss” of 25 million hectares of African non-drylands of 1931–1960 that turned into dry subhumid drylands during 1961–1990.^[31] Furthermore, rainfall decreases and evapotranspiration increases are projected to intensify this trend in many drylands by mid-century (IPCC 2008).^[32] To conclude, although LDD is driven by humans at the local scale, climate change, apparently human-driven, too, also leads to local productivity persistent decline, that is LDD, but one in which the local land users are not directly involved.

CONCLUSION

Desertification is a persistent reduction in land’s biological productivity of an economic value, as well as in other services provided by the land’s ecosystems. It is an extreme, terminal state of the land degradation process, and LDD can occur anywhere but the focus is on drylands; 40% of global land whose inherent low productivity is constrained by water. LDD comprises a syndrome of dynamically interactive biophysical processes, directly driven by land management practices of land users, themselves driven by underlying socioeconomic and policy interlinked processes that cross temporal and spatial scales, as well as by the inherent vulnerability of dryland soils to LDD, and by the effects of droughts.

Although the cumulative area of degraded land is only about a quarter of the globe’s productive land, the process is ongoing apparently at an increasing rate. LDD is therefore of global concern—the poverty it inflicts on the local land users is a source of migrations, refugees, and conflicts across boundaries and scales; the local loss of productive land risks the already fragile global food security. In addition, LDD and global climate changes are interlinked; climate

change increases dryness and hence LDD vulnerability, and LDD reduces the land’s carbon storage locally thus increasing emissions and warming, globally. The good news is that the UNCCD succeeded in raising awareness to the LDD plight, such that the international community seems ready to commit itself to significantly reducing further degradation and investing in restoring already degraded lands. Knowledge, institutions, and instruments are available, and the political will and resources need to be secured.

REFERENCES

1. Adeel, Z.; Safriel, U.; Niemeijer, D.; White, R. *Ecosystems and Human Well-Being: Desertification Synthesis*; The Millennium Ecosystem Assessment; World Resources Institute: Washington, 2005.
2. Aubreville, A. *Climats, Forest, et Desertification de l'Afrique Tropicale*. Societe de Editions 3/Geographiques; Maritim et Coloniales: Paris, 1949.
3. Vogt, J.V.; Safriel, U.; Von Maltitz, G.; Sokona, Y.; Zougmore, R.; Bastin, G.; Hill, J. Monitoring and assessment of land degradation and desertification: Towards new conceptual and integrated approaches. *Land Degrad. Dev.* **2011**, *22*, 150–165.
4. Safriel, U.N. Global land degradation: State, risks and prospects under global change. In *Soil, Society and Global Change*; Bigas, H., Gudbransson, G.I., Monanarella, L., Arnalds, O., Eds.; European Communities: Italy, 2009; 46–52.
5. Safriel, U.N. Dryland development, desertification and security in the Mediterranean. In *Desertification in the Mediterranean Region. A Security Issue*; Kepner, W.G., Rubio, J.L., Mouat, D.A., Pedrazzini, F., Eds.; NATO Security through Science Series; Springer Publishers: Germany, 2006; Vol. 3, 227–250.
6. *White Paper of DSD Working Group 1, August 2009, Integrated Methods for Monitoring and Assessing Desertification/Land Degradation Processes and Drivers*. Available at http://dsd-consortium.jrc.ec.europa.eu/documents/WG1_White-Paper_Draft-2_20090818.pdf (accessed March 2012).
7. Geist, H.J.; Lambin, E.F. Dynamic causal patterns of desertification. *Bioscience* **2004**, *54*, 817–829.
8. Reynolds, J.F.; Grainger, A.; Stafford Smith, D.M.; Bastin, G.; Garcia-Barrios, L.; Fernandez, R.J.; Janssen, M.A.; Jurgens, N.; Scholes, R.J.; Veldkamp, A.; Verstraete, M.M.; Von Maltitz, G.; Zdruli, P. Scientific concepts for an integrated analysis of desertification. *Land Degrad. Dev.* **2011**, *22*, 166–183.
9. Safriel, U.N.; Adeel, Z. Dryland Systems. In *Ecosystems and Human Well-Being: Current State and Trends*; Hassan, R., Scholes, R., Ash, N., Eds.; Island Press: Washington, 2005; 623–662.
10. Safriel, U.N. Alternative livelihoods for attaining sustainability and security in drylands, Vol 5, Hexagon Series on Human and Environmental Security and Peace. In *Coping with Global Environmental Change, Disasters and Security*; Brauch, H.G., Oswald Spring, U., Mesjasz, C., Grin, J., Kameri-Mbote, P., Chourou, B., Dunay, P., Birkmann, J., Eds.; Springer: Berlin, 2011; 835–852.

11. Reynolds, J.F.; Stafford Smith, D.M.; Lambin, E.F.; Turner, B.L., II; Mortimore, M.; Batterbury, S.P.J.; Downing, T.E.; Dowlatabadi, H.; Fernandez, R.J.; Herrick, J.E.; Huber-Sannwald, E.; Jiang, H.; Leemans, R.; Lynam, T.; Maestre, F.; Ayarza, M.; Walker, B. Global desertification: Building a science for dryland development. *Science* **2007**, *316*, 847–851.
12. Safriel, U.N. The assessment of global trends in land degradation. In *Climate and Land Degradation*; Sivakumar, M.V.K., Ndiagui, N., Eds.; Springer: Berlin, 2007; 1–38.
13. GLASOD. *Global Assessment of Soil Degradation*; International Soil Reference and Information Centre, United Nations Environment Programme, Wageningen, Netherlands, Nairobi, Kenya, 1990. Available at <http://www.isric.org/projects/global-assessment-human-induced-soil-degradation-glasod> (accessed March 2012).
14. Middleton, N.; Thomas, D., Eds. *World Atlas of Desertification*; Arnold: London, 1997; 182.
15. Bai, Z.G.; Dent, D.L.; Olsson, L.; Schaepman, M.E. Proxy global assessment of land degradation. *Soil Use Manag.* **2008**, *24*, 223–234.
16. Wessels, K. Comments on 'proxy global assessment of land degradation,' by Bai et al. 2008. *Soil Use Manag.* **2009**, *25*, 91–92.
17. Dobie, P. *Poverty and the drylands*. UNDP, Nairobi September, 2001. Available at <http://arabstates.undp.org/contents/file/Poverty-and-the-Drylands-Challenge-Paper.pdf> (accessed March 2012).
18. Safriel, U.N. Deserts and desertification: Challenges but also opportunities. *Land Degrad. Dev.* **2009**, *20*, 353–366.
19. Leighton, M. Desertification and migration. In *Governing Global Desertification: Linking Environmental Degradation, Poverty and Participation*; Johnson, P.-M., Mayrand, K., Paquin, M., Eds.; Ashgate Publishing: Aldershot, 2006; 43–58.
20. Safriel, U.N.; Adeel, Z. Development paths of drylands—is sustainability achievable? *Sustain. Sci. J.* **2008**, *3* (1), 117–123.
21. UNEP. *Sudan Post-Conflict Environmental Assessment*; United Nations Environment Programme: Nairobi, Kenya, 2007. Available at http://postconflict.unep.ch/publications/UNEP_Sudan.pdf (accessed March 2012).
22. Fischer, G.; Shah, M.; van Velthuizen, H.; Nachtergaele, F.O. *Global agro-ecological assessment for agriculture in the 21st century*, 2001. Available at <http://www.iiasa.ac.at/Research/LUC/SAEZ/index.html> (accessed March 2012).
23. Tilman, D.; Fargione, J.; Wolff, B.; D'Antonio, C.; Dobson, A.; Howarth, R.; Schindler, D.; Schlesinger, W.H.; Simberloff, D.; Swackhamer, D. Forecasting agriculturally driven global environmental change. *Science* **2001**, *292*, 281–284.
24. Verstraete, M.M.; Scholes, R.J.; Stafford Smith, M. Climate and desertification: Looking at an old problem through new lenses. *Front. Ecol. Environ.* **2007**, *7* (8), 421–428.
25. Nkonya, E.; Winslow, M.; Reed, M.S.; Mortimore, M.; Mirzabaev, A. Monitoring and assessing the influence of social, economic and policy factors on sustainable land management in drylands. *Land Degrad. Dev.* **2011**, *22*, 240–247.
26. Adeel, Z.; Safriel, U. Achieving sustainability by introducing alternative livelihoods. *Sustain. Sci. J.* **2008**, *3* (1), 125–133.
27. Cowie, A.L.; Penman, T.D.; Gorissen, L.; Winslow, M.D.; Lehmann, J.; Tyrrell, T.D.; Twomlow, S.; Wilkes, A.; Lal, R.; Jones, J.W.; Paulsch, A.; Kellner, K.; Akhtar-Schuster, M. Towards sustainable land management in the drylands: Scientific connections in monitoring and assessing dryland degradation, climate change and biodiversity. *Land Degrad. Dev.* **2011**, *22*, 248–260.
28. Bainbridge, D.A. *A Guide for Desert and Dryland Restoration*; Island Press: Washington, 2007.
29. Prince, S.D.; Brown de Colstoun, E.; Kravitz, L.L. Evidence from rain use efficiencies does not indicate extensive Sahelian desertification. *Glob. Change Biol.* **1988**, *4*, 359–458.
30. United Nations Convention to Combat Desertification (UNCCD). Available at <http://www.unccd.int>.
31. Hulme, M.; Marsh, R.; Jones, P.D. Global changes in a humidity index between 1931–1960 and 1961–1990. *Clim. Res.* **1992**, *2*, 1–22.
32. Parry, M.L.; Canziani, O.F.; Palutikof, J.P.; van der Linden, P.J.; Hanson, C.E., Eds. *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*; Cambridge University Press: Cambridge, 2007.

BIBLIOGRAPHY

1. Allington, G.R.H.; Valone, T.J. Reversal of desertification: the role of physical and chemical soil properties. *J. Arid. Environ.* **2010**, *74*, 973–977.
2. Briassoulis, H. *Policy Integration for Complex Environmental Problems. The Example of Mediterranean Desertification*; Ashgate: Burlington, 2005.
3. Corell, E. *The Negotiable Desert. Expert Knowledge in the Negotiations of the Convention to Combat Desertification*; Linköping University, Linköping: Sweden, 1999; 1–286.
4. Fantechi, R.; Peter, D.; Balabanis, P.; Rubio, J.L. *Desertification in a European Context: Physical and Socio-Economic Aspects*; European Commission: Luxembourg, 1955.
5. Grainger, A. *The Threatening Desert: Controlling Desertification*; Earthscan and UNEP: London, 1990.
6. Safriel, U.N. The concept of sustainability in dryland ecosystems. In *Arid Lands Management – Toward Ecological Sustainability*; Hoekstra, T.W., Shachak, M., Eds.; University of Illinois Press: Urbana, 1999; 117–140.
7. Schlesinger, W.H.; Reynolds, J.F.; Cunningham, G.L.; Huenneke, L.F.; Jarrell, W.M.; Virginia, R.A.; Whitford, W.G. Biological feedbacks in global desertification. *Science* **1990**, *247*, 1043–1048.
8. Swift, J.; Leach, M.; Mearns, R. *Desertification. Narratives, Winners & Losers*; James Currey: Oxford, 1996.
9. Tiffen, M.; Mortimore, M.; Gichuki, F. *More People, Less Erosion: Environmental Recovery in Kenya*; J. Wiley: Chichester, England, 1994.
10. Thomas, D.S.G.; Middleton, N.J. *Desertification: Exploring the Myth*; J. Wiley: Chichester, 1994.
11. Williams, M.A.J.; Balling, R.C., Jr. *Interactions of Desertification and Climate*; Arnold: London, 1996; 1–372.
12. Xu, D.Y.; Kang, X.W.; Zhuang, D.F.; Pan, J.J. Multi-scale quantitative assessment of the relative roles of climate change and human activities in desertification – A case study of the Ordos Plateau. *China. J. Arid Environ* **2010**, *74*, 498–507.

Desertification: Extent

Victor R. Squires

Adelaide University, Adelaide, South Australia, Australia

Abstract

Desertification is a global problem. The United Nations Environment Program estimates that about 35% of the earth's land surface and about 20% of its population are affected by it. Clearly, desertification is a major environmental and social problem. The problem is worse for heavily populated countries on the desert margins. The global overview of desertification provides a summary but recognition must be given to regional differences and impacts. A global summary of the kind presented in this entry is a useful first step to take when looking at a large-scale issue.

INTRODUCTION

Desertification, as a *concept*, is one of the most complex to consider objectively. The word desertification unfortunately conjures up images of advancing sand dunes, which is only part of the problem. There is a general acceptance of the term *desertification* to encompass a process of land degradation in drylands (those regions with a growing season of about 75–120 days per year) induced by climatic factors and human use. Fig. 1 shows the broad distribution of the world's drylands.

Whatever the definition, it is clear that large areas of the world's drylands are affected by land degradation and that this adversely affects the lives of about 1 billion people on almost every continent. To assess the global extent of desertification implies that there is universal acceptance of the term and that the monitoring and evaluation are somewhat uniform. Neither supposition is true. Until the decision by the United Nations Conference on Desertification,^[1] there was only broad agreement as to what was meant by it. It is only as signatories to the convention to combat desertification (CCD) have been preparing and implementing their respective action plans that are more accurate assessments which have been made of the status of desertification in each country. Prior to this, the maps and other documents produced by the United Nations through its various agencies were the product of expert panels who developed maps of the areas of potential desertification.^[2] Every inhabited continent has a problem with desertification, but in some, it is quite acute. Land degradation induced by human activities (anthropogenic) is believed to be lowering the productivity of at least one-fifth of the world's agricultural lands. Soil erosion from water and wind is a principal but by no means the only soil degrading process. Subhumid and semiarid areas in crops or pastures are particularly vulnerable to erosion.

Land degradation in such areas leads to desertification by definition. Anthropogenic land degradation, however, also affects large areas of arable land and pasture in humid regions as well.

Depending on how it is measured exactly,^[3] up to 75% of some continents are affected. The impacts are greatest in those regions with high population density and where the level of economic development is low; e.g., Australia is reported as having more than 75% of its land surface affected to a greater or lesser extent by desertification. Its low population density and the low level of dependence on these drier regions for its national prosperity mean that desertification is perceived as a lesser threat in Australia than in regions of comparable climate in the Indian subcontinent or in Africa.

ASSESSING THE EXTENT OF THE DESERTIFICATION PROBLEM

Desertification comprises two main types of degradation: vegetation degradation and soil degradation.

These can occur anywhere in dry areas and not just on desert fringes. Vegetation degradation involves a temporary or permanent reduction in the density, structure, species composition, or productivity of vegetation cover. It has been argued that degradation of vegetation should be given less weight in the assessment of desertification because there is clear evidence that most change in vegetation is reversible over reasonable timescales, whereas soil degradation is unlikely to occur within a human life span.

Desertification might be thought of as a situation where a landscape is stressed beyond its resilience. Most scientists would agree that, despite their apparent fragility, dryland ecological systems are quite resilient. The spatial and temporal variation in conditions means that

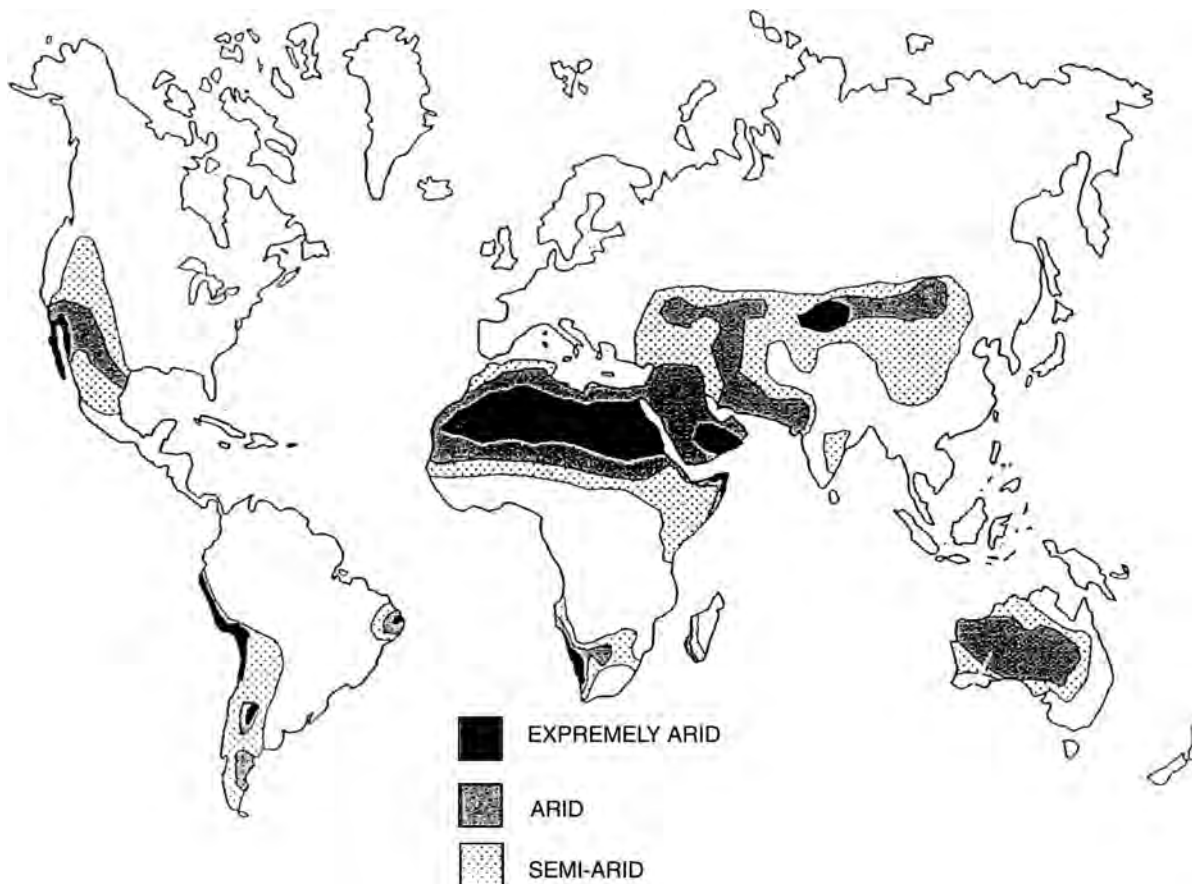


Fig. 1 Distribution of world's drylands.

coping mechanisms have developed to allow the system to continue in the face of adversity. Under the combined force of human-induced pressure and climatic forces, such as recurrent drought, major changes in soils and vegetation can occur. This is what has come to be recognized as desertification. This is what the CCD is designed to combat.

Desertification is a product of complex interactions between the social and economic systems (disease, poverty, hunger, and unreliable economy) and natural factors (drought, water erosion, soil salinization, degradation, and loss of vegetation cover). It is not surprising then that the criteria used to objectively assess the areal extent vary between regions. It is clear that a definition based purely on climatic indices does not give a true picture of the full extent of the problem.

No one has been able so far to monitor and document desertification and the resulting land cover change through reliable, verifiable, and repeated observation on a global, continental, regional, or even national scale. There is not full agreement on a single indicator of dryland degradation or an approach to assess and study desertification. Dryland degradation has many faces (driving forces as well as symptoms). It can only be assessed and understood through

an interdisciplinary study of the changing characteristics and integrated trends of a variety of biological, agricultural, physical, and socioeconomic indicators over a long time period and at a variety of spatial scales.

Because of the scale of mapping and the fact that most desertification occurs as a mosaic of small, often-isolated, areas, mapping defies the use of synoptic techniques such as remote sensing from Landsat and other satellites. Land cover/land use maps are being prepared by a number of countries, and on a global scale, land cover is being assessed by the analysis of satellite data. These may help to clarify and quantify the extent of desertification, especially where local input helps in verification and ground truthing.

The use of the low-resolution NOAA/Advanced Very High Resolution Radiometer satellite to generate a greenness index and to develop the normalized difference vegetation index might, on a broad scale, assist in monitoring change in plant cover and hence be an early warning of incipient change. But such change may only be a reflection of variability in space and time that characterizes the world's dryland regions. It is likely to be more useful in studies of drought and vegetation mapping. Its potential for use in desertification monitoring may be limited.^[3]

Table 1 Some common manifestations of desertification.

Economic manifestations	Ecological manifestations	Social manifestations
Economic loss in cash	Loss of diversity in terms of wildlife, plants and ecosystems	Migration of population off affected areas
Decreased crop yield	Loss of inland lakes	Rural poverty
Loss of farmland due to desertification	Loss of topsoil in terms of organic matter, N, P, and K plant nutrients	Influx of ecological refugees into urban areas
Loss of rangeland due to desertification	Decreased ground water level, increasing salinity of water	
Decreased grazing capacity in terms of the number of livestock	Increased frequency of sand storms and associated losses of human lives and livestock	
Abandoned farmland		
Abandoned rangeland		
Drifting sand affects railway lines and highways		
Increase in suspended load raises river height and increases flood problems		

The most immediately detectable changes can be quite misleading.

AREAL EXTENT OF DESERTIFICATION

The area affected by desertification in many developing countries is on the increase although the lack of sound baseline data makes any attempt at quantification difficult. Because a lot of desertification is characterized by a more dispersed, patch-like process of degradation, it is not amenable to remote sensing. It has long been recognized that

operational monitoring of patchy soil and vegetation degradation in dry areas is inherently difficult, even with medium-resolution satellite technology like Landsat MSS and those improved techniques are needed to make reliable monitoring and trend detection viable.

What is clear is that most of the more than 170 signatories to the CCD report growing concern about loss of productivity of their land in the face of rapid and continuing land degradation. The impacts are reflected in:

- loss of land resource through erosion and transportation by wind and water
- loss of productivity through land degradation
- eco-environmental change impacting on living conditions and health
- increased poverty and social instability in the rural hinterland
- flow-on impacts on cities and on industry
- effects on infrastructure and economic development

The principal manifestations of severe desertification are shown in [Table 1](#).

The United Nations Environment Program (UNEP)'s 1991^[4] estimate of the area of desertified land is 3.6 billion hectare, including 1 billion hectare of dryland suffering from soil degradation and another 2.6 billion hectare of rangeland with degraded vegetation.^[5] But an external review claimed that its accuracy was limited by considerable subjectivity (so observations were not repeatable); lack of resolution (so comparisons through time were not possible); and the use of point assessments which were unrepresentative of larger areas.^[6]

As indicated earlier, there are few objective measures of the areas of land affected by desertification. The most reliable maps are those published by UNEP in the revised edition of the *World Atlas of Desertification*.^[2]

Dr. Harold Dregne of Texas Tech University calculated the area of potential desertification ([Table 2](#)). Dregne determined the desertification status according to three factors taken as indicators, namely:

- changes in the composition of the vegetation
- extent of erosion; and
- the presence of soil salinity

Table 2 Distribution of the area of land susceptible to desertification (by bioclimatic zones).

Susceptibility to desertification	Bioclimatic zone (000's km ² %)						
	Arid	Semiarid	Subhumid				
Moderately	1144.5	6.6	12713.8	68.4	3345.6	23.4	
Highly	14585.8	82.5	2686.5	14.2		589.8	4.0
Very highly	1040.3	6.4	2157.5	12.4		173.5	1.3
Total	16770.6		17557.8		4108.9		

Source: Data from UNEP and other sources. Used with permission.

Table 3 Area of desertified lands by continent/region.

Degree of desertification	Africa	Asia	Australia	North America	South America	Europe	World total
Slight	12430	7980	2330	440	1340	—	24520
Moderate	1870	4480	3510	2720	1050	140	13770
Severe	3030	3210	520	1200	680	60	8700
Very severe	—	—	—	67	8	—	73
Total	17330	15670	6360	4427	3076	200	47063

Source: Data from UNEP, CCD, and other sources. Used with permission.

He singled out the following four degrees of desertification: slight, moderate, strong, and very strong. Strong desertification is an irreversible process when it is impossible to restore the land. This classification comprises mobile sand dunes, heavily salt-affected land, and badlands. In Dregne's opinion, such areas are not extensive, and their area is about 50,000 km².^[7]

The areas subject to severe desertification (Table 3) are in Africa, Asia, and South America and are mainly occupied by the developing countries. There are a number of drivers of change. These include rapid population growth, the availability of modern technologies for land conversion, as well as loss of traditional land use controls. Political and economic policy decisions, market and trade arrangements, the lack of environmental awareness, and the lack of capacity to combat the problem also contribute.

CONCLUSION

The evidence from many sources is that desertification is a global problem. UNEP estimates that about 35% of the earth's land surface and about 20% of its population are affected by it. Clearly, according to these figures and the data presented here, desertification is a major environmental and social problem. The problem is worse for heavily populated countries on the desert margins. When the extent of desertification is broken down according to major types of land use, we see that grazing land and rainfed cropping are most severely affected, each being desertified on over three-quarters of its area. Irrigated land suffers less so, with about one-fifth of the irrigated area in drylands affected by desertification (salinization and waterlogging).

The global overview of desertification provides a summary, but recognition must be given to regional differences and impacts. A global summary of the kind presented in this entry is a useful first step to take when looking at a large-scale issue. However, when working out a plan of action to combat desertification at the regional or local level, more detailed information needs to be gathered on the way desertification is happening and on its root causes. Such information can only be gathered by long-term monitoring of a particular area.

REFERENCES

1. UNCOD. International convention to combat desertification in those countries experiencing serious drought and/or desertification, particularly in Africa, I.L.M. content summary. *Int. Legal Mater.* **1995**, 33, 1328–1382.
2. Middleton, N.; Thomas, D. *World Atlas of Desertification*, 2nd Ed.; UNEP/Edward Arnold: London, 1997.
3. Hellden, U. Desertification monitoring—Is the desert encroaching? *Desertification Contr. Bull.* **1998**, 17, 8–12.
4. UNEP. *Status of Desertification and Implementation of the United Nations Plan of Action to Combat Desertification*; UNEP: Nairobi, 1991.
5. Dregne, H.M.; Kassas, M.; Rosanov, B. A new assessment of the world status of desertification. *Desertification Contr. Bull.* **1991**, 20, 6–18.
6. UNEP. *Draft Report of the Expert Panel Meeting on Development of Guidelines for Assessment and Mapping of Desertification/Land Degradation in Asia/Pacific*; UNEP: Nairobi, 1994.
7. Kassas, M. Desertification: A general review. *J. Arid Environ.* **1995**, 30, 15.

Desertification: Greenhouse Effect

Craig Idso

Center for the Study of Carbon Dioxide and Global Change, Tempe, Arizona, U.S.A.

Sherwood E. Idso

U.S. Water Conservation Laboratory, Phoenix, Arizona, U.S.A.

Abstract

What is the impact of the rise in the air's carbon dioxide (CO₂) concentration on the ecological stability of the world's deserts and the shrubs and grasslands that surround them? This question weighs heavily on the minds of many, as the nations of the earth debate the pros and cons of the prodigious CO₂ emissions produced by the burning of fossil fuels. On the downside, there is concern that more CO₂ in the air will exacerbate the atmosphere's natural greenhouse effect, producing changes in climate that lead to desertification. On the upside, the aerial fertilization effect of additional atmospheric CO₂ may enhance plant prowess, increasing plant water use efficiency and enabling vegetation to reclaim great tracts of desert. The challenge, therefore, is to determine the relative merits of these competing phenomena.

THE WORLD IN TRANSITION

Climatical Changes

Questions surrounding the climatical effects of the rise in the air's carbon dioxide (CO₂) content are contentious. There is evidence the earth has warmed significantly over the course of what is deemed the "Age of Fossil Fuels." Most of this warming occurred well before the largest increases in the air's CO₂ content were recorded, peaking in the 1930s. Thereafter, the atmospheric CO₂ concentration rose at a much greater rate than it had previously, while temperatures stagnated before staging a hotly contested comeback in the 1980s and 1990s—a comeback more virtual than real. It is also possible that the atmospheric warming in the late 19th and early 20th centuries was nothing more than a natural recovery from the global chill of the Little Ice Age. Hence, although many scientists believe there has been a discernable human influence on the global climate of the 20th century, other scientists take serious issue with that contention.

Biological Changes

Climatical effects in the biological arena are also complicated, but not so much that certain facts cannot be used to draw some broad conclusions. CO₂, e.g., is one of two main raw materials (the other being water) used by plants to produce the organic matter from which they construct their tissues. Thus, increasing the air's CO₂ content typically enables plants to grow better, as has been demonstrated in literally thousands of laboratory and field experiments.^[1]

In addition, the higher concentrations of atmospheric CO₂ cause many plants to reduce the apertures of the small pores in their leaves, through which water vapor escapes to the air. Consequently, with more biomass production per unit of water lost, plant water use efficiency is significantly enhanced. In fact, it approximately doubles with a doubling of the air's CO₂ content,^[2] allowing many species of plants to grow and reproduce where it had been too dry for them previously. Furthermore, the enhanced degree of ground cover resulting from this phenomenon reduces the magnitude of soil erosion caused by the ravages of wind and rain. And with greater plant growth, both aboveground and belowground, there are significant increases in the amounts of organic matter that enter the soil. That matter enhances the soil's ability to sustain the more productive shrub and grassland ecosystems that come into being via this process of reverse desertification.

Although supported by a plethora of scientific studies, this scenario of vegetative transformation has been challenged on the assumption that resource limitations and environmental stresses encountered in nature might overpower the ability of atmospheric CO₂ enrichment to significantly enhance the vitality of plants. However, in a massive literature review designed to investigate this question, it was found that the percentage increase in plant growth produced by an increase in atmospheric CO₂ is generally not reduced by less-than-optimal levels of light, water, or nutrients or by high temperatures, salinity, or gaseous air pollution.^[3] In fact, the data demonstrate that the relative growth-enhancing effects of atmospheric CO₂ enrichment are typically the greatest when resource limitations and environmental stresses are most severe.

Greening of the Earth Hypothesis

In light of these observations, there is reason to believe that the historical trend in the air's CO₂ concentration, which rose from a value of 280 parts per million (ppm) at the start of the 19th century to 370 ppm at the turn of the millennium, may already be producing an ubiquitous "greening" of the earth. This is especially true for bushes, shrubs, and trees since woody plants are typically more positively affected by increases in the air's CO₂ content than herbaceous plants.^[4] Consequently, the most readily documented aspect of this biological transformation should be seen in the spreading of woody vegetation onto grasslands.

WOODY PLANT RANGE EXPANSIONS

From Prehistory to Industrial Revolution

The savannas, grasslands, and deserts of the American Southwest and the Southern Great Plains got their start approximately 25 million years ago. About that time, the climate began to dry, and it continued to become more arid, particularly over the past 15 millennia. But when the engines of the Industrial Revolution began to pump CO₂ into the air at rates that exceeded natural geological processes, things began to change. As early as 1844, in fact, a trader from Santa Fe, New Mexico wrote in his memoirs^[5] "there are parts of the southwest now thickly set with trees of good size, that, within the remembrance of the oldest inhabitants, were as naked as the prairie plains." He summarized the situation by saying "we are now witnessing the encroachment of timber upon the prairies." And in surveying the land a century later, Malin^[6] would verify that trader's prescience by referring to the ecosystem it supported as a "tangled jungle."

Modern Studies

An especially good history of the vegetative transformation of the American Southwest was developed by Blackburn and Tueller^[7] from a study of the growth rings of juniper and pinyon communities in east-central Nevada, where they determined that juniper began expanding its coverage of the land well before 1800, with more rapid increases in the densities and sizes of both species occurring subsequent to that time. But just as the rise in the air's CO₂ content developed most dramatically in the 20th century, the most remarkable increases in the presence and abundance of trees at these sites manifest themselves after 1920.

A parallel example involving other woody species has been documented on the Jornada Experimental Range in New Mexico.^[8] When surveyed by the U.S. Land Office in 1858, no shrubs were evident on 60% of the land, and only 5% of the range contained what

could be described as dense stands of brush. However, 105 years later, none of the area was free of woody plants, and 73% of it was dominated by thick stands of mesquite and creosote. Texas, too, had been thus transformed. By 1963, 88 million acres of former grasslands had been replaced by brush and shrubs. By 1982, the figure had risen to 105 million acres (Figs. 1 and 2).

In a comprehensive review of this subject, Idso^[8] assembled many examples of woody-plant range expansions onto grasslands, citing studies conducted in California, Idaho, Kansas, Missouri, Montana, Nebraska, North Dakota, Oklahoma, Oregon, South Dakota, and a number of New England states. He also cited examples of the same phenomenon in South America, Europe, Asia, Africa, Australia, and New Zealand, augmenting this evidence with equally numerous and widespread reports of ever-accelerating woody-plant growth-rate increases.

Although a number of different hypotheses have been proposed to account for this vegetative transformation of the planet, many of which play significant roles in specific locations and circumstances, the ubiquitous nature of the phenomenon argues strongly for a single worldwide

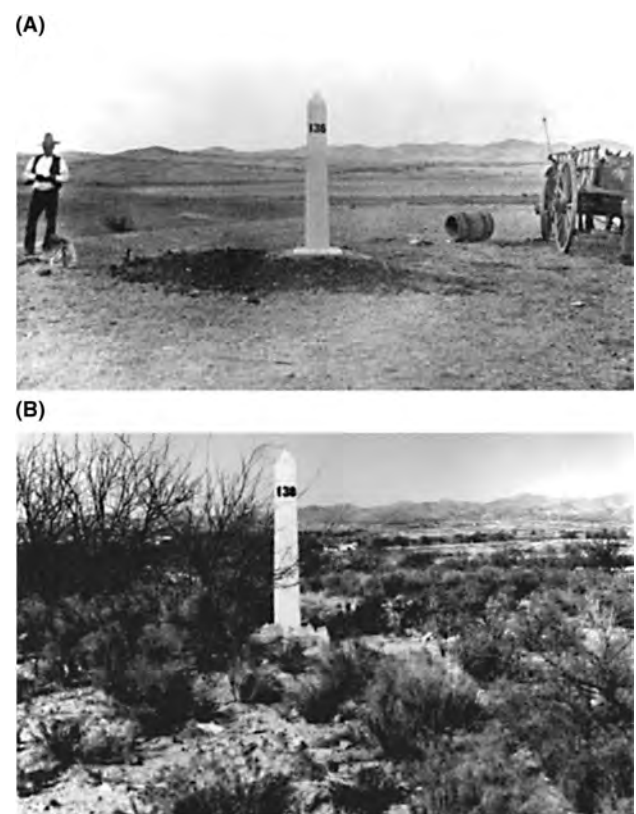


Fig. 1 Photographs of the U.S.-Mexico border just east of Sasabe, Arizona. (A) Taken in 1893 by D.R. Payne: the area was devoid of shrubs. (B) Same areas, taken in 1984 by R.R. Humphrey, were dominated by velvet mesquite, ocotillo, velvet-pod mimosa, snakeweed, and burroweed.

Source: From Humphrey.^[9]

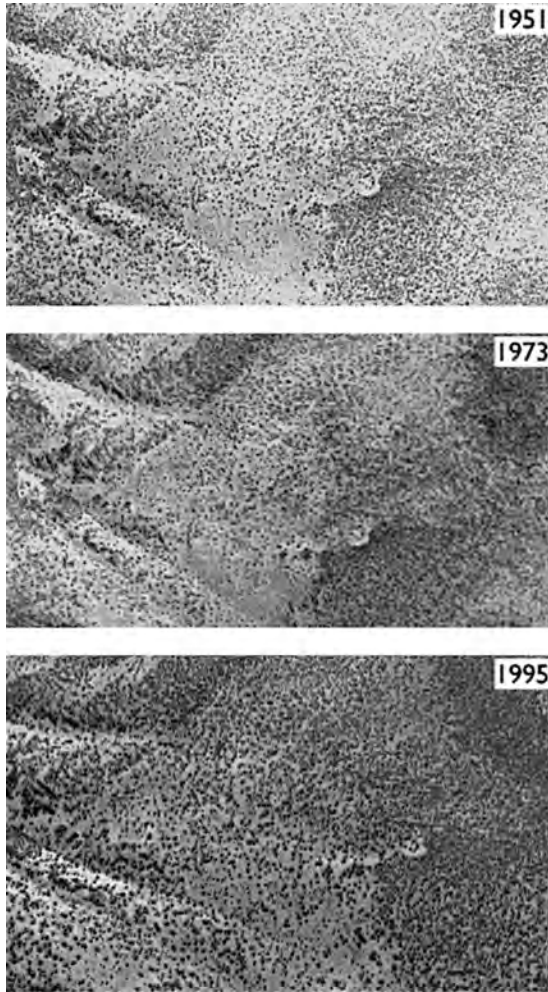


Fig. 2 Aerial photographs of the Horse Ridge Research Natural Area in Central Oregon depicting an increase in western juniper cover and density between 1951 and 1995.

Source: From Knapp & Soule.^[10]

forcing factor that dominates the effects of most other influences. The rise in the air's CO₂ concentration is the only phenomenon that would appear to meet this global-scale criterion.

CONCLUSION

What does the future hold? It will probably be more of the same. There are no signs that humanity's appetite for fossil-fueled energy will abate any time soon. Neither are there any indications the nations of the earth will be able to reverse this trend by regulatory or legislative fiat. Therefore, if we do not completely cover the globe with concrete and asphalt before the Age of Fossil Fuels ends, we will probably see woody plants continue their invasion of

grasslands, while grasslands intensify their assault upon the world's deserts.

With respect to global warming, it's anybody's guess. Even if it does occur, CO₂-induced increases in plant water use efficiency (a doubling for a doubling of atmospheric CO₂) and plant thermal tolerance (a 5°C increase in optimum growing temperature for a doubling of CO₂) should enable earth's ecosystems to keep on responding as they have over the 19th and 20th centuries. Thus, the greening of the earth should continue increasing vegetative productivity, the biomass of higher food-chain trophic levels, and ecosystem biodiversity concomitantly.^[11]

REFERENCES

1. Idso, K.E. *Plant Responses to Rising Levels of Atmospheric Carbon Dioxide: A Compilation and Analysis of the Results of a Decade of International Research into the Direct Biological Effects of Atmospheric CO₂ Enrichment*; Climatological Publications Scientific Paper No. 23; Office of Climatology; Arizona State University: Tempe, 1992; 1–186.
2. Idso, S.B. *Carbon Dioxide and Global Change: Earth in Transition*; IBR Press: Tempe, 1989; 1–292.
3. Idso, K.E.; Idso, S.B. Plant responses to atmospheric CO₂ enrichment in the face of environmental constraints: A review of the past 10 years' research. *Agr. For. Meteorol.* **1994**, *69*, 153–203.
4. Idso, S.B.; Kimball, B.A. Tree growth in carbon dioxide enriched air and its implications for global carbon cycling and maximum levels of atmospheric CO₂. *Global Biogeochem. Cy.* **1993**, *7*, 537–555.
5. Gregg, J. *Commerce of the Prairies: Or, the Journal of a Santa Fe Trader, during Eight Expeditions across the Great Western Prairies, and a Residence of Nearly Nine Years in Northern Mexico*; Henry, C., Ed.; Langley: New York, 1844; Vol. 2, 202.
6. Malin, J.C. Soil, animal, and plant relations of the grassland, historically reconsidered. *Sci. Mon.* **1953**, *76*, 207–220.
7. Blackburn, W.H.; Tueller, P.T. Pinyon and Juniper invasion in Black Sagebrush communities in East-central Nevada. *Ecology* **1970**, *51*, 841–848.
8. Idso, S.B. *CO₂ and the Biosphere: The Incredible Legacy of the Industrial Revolution*; Department of Soil Water & Climate, University of Minnesota: St. Paul, 1995; 1–60.
9. Humphrey, R.R. *90 Years and 535 Miles: Vegetation Changes Along the Mexican Border*; University of New Mexico Press: Albuquerque, 1987.
10. Knapp, P.A.; Soule, P.T. Recent *Juniperus occidentalis* (Western Juniper) expansion on a protected site in central Oregon. *Global Change Biol.* **1998**, *4*, 347–357.
11. Idso, K.E.; Idso, S.B.; Idso, C.D. Atmospheric CO₂ enrichment: Implications for ecosystem biodiversity. *Technology* **2000**, *7S*, 57–69.

Desertification: Humid Environment in Iceland

Andres Arnalds

Soil Conservation Service, Hella, Iceland

Abstract

The term “desert” can have many different meanings. The scale of ecosystem disturbance in Iceland, where barren deserts have replaced vegetation and thick soils in many areas regardless of ample precipitation, demonstrates the global nature of this severe environmental problem. The world’s forests and woodlands are being reduced at an alarming rate in many parts of the world, and large areas are being overgrazed. The weakening of the vegetative cover can lead to a chain of ecosystem disturbances reducing further the resilience of the ecosystems toward further degradation. If prolonged, this can lead to desertification in a wide range of moisture regimes, as demonstrated by the Icelandic experience described in this entry.

INTRODUCTION

Desertification has wide-ranging consequences on a global scale. However, this term has generally been associated with or even confined to “land degradation in arid, semi-arid, and dry subhumid areas” (United Nations Convention to Combat Desertification, Article 1). Such definitions ignore the effects of natural or anthropogenic disturbances leading to ecosystem disturbances of functional equivalence in more humid environments or even high rainfall areas.^[1] Narrow definitions of “desertification” exclude other cases of severe degradation and may hinder progress in research, management, and policy development for mitigation of this immense environmental problem.^[2]

OVERVIEW

The history of vegetation and soils in Iceland since settlement of the island around 874 A.D. demonstrates clearly how the processes of desertification can render fertile ecosystems into barren wastelands, i.e., deserts, in a humid environment. During this time, Iceland may have lost about 96% of the original tree cover and about 50% of the vegetation. Land use interacting with natural disturbances led to catastrophic soil erosion that has devastated large parts of the Icelandic ecosystems.

More than 37% of Iceland is classified as barren deserts.^[3] In addition are other eroded areas and large damaged areas with limited plant production. The desert soils are infertile and generally have 0.5–5% vegetation cover, and plant succession may be considered primary. Botanical composition of remaining vegetation in many areas is characterized by unpalatable plant species that heavy grazing has given competitive advantage. Desertification continues to be a major threat to Iceland’s natural resources.

Such large-scale ecosystem disturbances greatly affect hydrological attributes such as infiltration rates, soil water storage capacity, precipitation characteristics, and rain use efficiency.^[4] Growing conditions for vegetation thus can change from mesic to xeric and from fertile to infertile regardless of precipitation, as evidenced by desertification of the humid environment in Iceland. Here, the perspective of van der Leeuw^[5] will be followed, namely, “to view desertification as a special case of very heavy and large-scale degradation, and look at the wider range of processes rather than focus on desertification.”

ICELAND—A WARM ISLAND WITH A COLD NAME

The Viking settlers who came to Iceland 1100 years ago (about 874 A.D.) saw a fertile land. Vegetation may have covered more than 60% of the country, and woodlands, mainly birch (*Betula pubescens*), covered at least 25% of the land area^[6] or even 40%.^[7] Deserts covered only 5000–15,000 km².^[2]

This vegetation had evolved under a relative warm regime, which continued through the first two centuries of settlement. The “icy” part of the name Iceland derives from icebergs from the north that Vikings saw drifting in one of the western fjords.^[1]

Climate and Soils

Iceland is in the subarctic belt, touching the Arctic Circle in the north. A branch of the warm Gulf Stream nearly encircles this 103,000 km² large island. The climate is maritime cold temperate in the lowlands to subarctic in the highlands. The winds are frequent and strong, mainly from the southeast and north. Annual rainfall is highly

variable, from 400 mm in the rain shadow north of the Vatnajökull Glacier to 3500 mm at the south coast.

Iceland is a young country in geologic terms, and there may be, on the average, one eruption every 5 years. The soils in most parts of Iceland are Andosols that draw their major characteristics from their volcanic parent material.^[8,9] Icelandic soils are also influenced by steady flux of Aeolian materials originating from desert wind erosion areas.

A Green Island—How Do We Know?

The extent and composition of vegetation in Iceland bear little resemblance to what it was historically. The ecosystem degeneration and desertification following settlement have been so extensive and spectacular. Iceland is not as fertile and hospitable to its inhabitants as it could be.

Arnalds^[10,11] summarizes some of the various sources that can be used to reconstruct the vegetation of the past and trace some of the major changes in cover and composition through the centuries. To mention just a few, the settlement time Sagas, written about 200 years after the episodes, refer in many cases to the woodlands of the past. Pollen studies, linked to the use of ash layers for dating, reveal sharp changes in botanical composition after settlement, especially the disappearance of birch from many lowland areas.

Historical records are a good source for comparison of past and existing land condition. A description of all farms in Iceland from beginning of the 18th century indicates fertile land in many areas that are severely degraded or barren deserts. Many old site or farm names, such as “Sko-gar” (woods), indicate lush vegetation in desertified areas, and the woodland word “holt” means a barren or infertile hill. Remnants of pits where charcoal was made out of birch, and other land use indicators, also tell a sad story of land degradation (Fig. 1).

Such information about changes in land health is important as one of the tools for setting goals for restoration of damaged land in Iceland. It is also a sharp reminder on the potential consequences of long-term unsustainable land use in a sensitive environment.

DESERTIFICATION IN ICELAND

The vegetative cover provided good protection for fragile volcanic soils. With the settlement, a delicate balance between forces of nature and vulnerable vegetation was disrupted.^[10,12] Subsequent soil erosion has devastated large parts of the ecosystem, reducing vegetative cover from about 60% to 30%. Trees cover only 1% of the land area.

Causes and Consequences

The initial causes of soil erosion in Iceland vary from place to place. However, the interaction of livestock grazing with weak soil structure, harsh climate, and volcanic eruptions is considered to be the main reason for the great ecosystem disturbance. Climatic fluctuations have also sped up this process, reducing further the ability of the vegetation to meet the relentless pressure of man and livestock. The importance of timescale and rare events, such as cold spells, volcanic eruptions, and periods of excessive tree clearing or overgrazing, as triggers of disturbance and destruction should be stressed in the interpretation of the degradation processes and history.

The initial impact of settlement was both immense and had long-lasting consequences. There are indications that the settlers brought with them cultivation techniques from their countries, mainly Norway. Vast areas of woodlands may initially have been burned to make space for pastures and cropland or to make nutrients stored in the woods available for crop and pasture plants. The birch and later shrubs were also extensively cut for fuel, and wildfires damaged many areas. Livestock grazing further changed botanical composition, reduced stability of the ecosystems, and interfered with vegetation recovery after weakening by other land uses and natural disturbances (Fig. 2).

A Vicious Cycle

While the birch woodlands were extensive, they sheltered sensitive soils from the natural forces. Woodlands have relatively strong resilience toward disturbance and recover quickly after limited disturbance, such as landslides, if grazing does not hamper regeneration.^[13] The woodlands

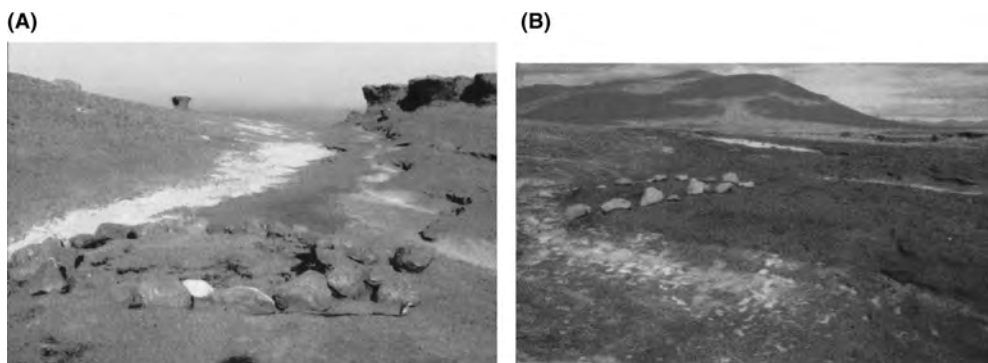


Fig. 1 Desertification in Iceland, leaving the naked glacial pavement behind. (A) Encircled by stones in the foreground, and visible for a short time while the soil is disappearing, is the bottom of a pit where charcoal was made out of birch trees. (B) Same place 16 years later.



Fig. 2 Unsustainable grazing is one of the interacting factors causing desertification in Iceland.

had a key role in stabilizing soils on hill slopes, which are generally in very poor condition. Active and severe soil erosion is on about 6000 km² of hill-type land. Many other slopes are devoid of vegetation, including stony scree slopes on about 5000 km².^[14]

Farms extended to relatively high elevations during the first centuries of settlement, reflecting good condition of the inland areas. The highlands were also used for sheep grazing from lower elevation farms. Many of highland farms had been abandoned around year 1100; critical thresholds in land health had been exceeded, and the climate was gradually getting colder. A vicious cycle had been initiated. Ecosystem stability had been reduced and both cold spells and ash from volcanic eruptions had ever-increasing impact on remaining vegetation and soils.

Following the initial impact of settlement, woodland destruction, livestock grazing, and environmental events continued to interact leading to damaged ecosystems or total loss of vegetation cover in large parts of Iceland. The rate of erosion may have been highest in the late 19th century and the first decade of the 20th century. The climate was unfavorable, with frequent cold spells. Iceland, then a Danish colony, gained independence of trade in 1885, and high grazing intensities following new markets for sheep wool and meat led to a period of massive ecosystem degeneration in many areas. Gísladóttir^[15] gives details of such marked related environmental changes in an area in south-west Iceland during the last part of the 19th century. The rapid ecosystem destruction was widespread, and in North and South Iceland, many kilometer-wide sand fronts were destroying vegetated areas, often at a rate of 200–400 m in a year, until the mid-20th century.

Pressure on dwindling sources for fuel wood kept rising until the late 19th century. Especially, there was a high demand for charcoal from birch to whet the sieges used for cutting hay, which had to be done by heating. A new type of siege brought initially to Iceland from Scotland in 1876 reduced demand for charcoal and may have saved many of the woodland remnants from total destruction.

With access to machinery and fertilizers in the 1950s, Icelandic farmers could produce more hay for winterfeeding, and livestock numbers rose rapidly. This led to a new period of overgrazing, until market forces and reduced

(A)



(B)



(C)



Fig. 3 The process of desertification in Iceland. (A) Birch (*Betula pubescens*) and willows protected sensitive soils. (B) With the destruction of the woodland cover, bare ground patches began to form, destabilizing the ecosystems and increasingly subjecting the sites to wind and water erosion. (C) Deserts characterized by glacial till, lava, and sandy surfaces have replaced fertile ecosystems in large parts of Iceland. This area was birch woodland about 400 years ago.

governmental subsidies in the early 1990s eventually led to 50% reduction in sheep numbers.

Stages of Desertification

The ecosystem changes and consequent desertification of Icelandic landscapes are likely to have followed stepwise degradation and desertification pathways as described by a conceptual model outlined by Aradottir et al.^[16] and further adapted by Archer and Stokes.^[1] Continuous cover of birch, willows, grasses, and forbs (State I) shifts to less-productive heath land (State II). Spots of erosion begin to form with continued interaction of anthropogenic and human disturbances (State III) gradually increasing the area of exposed soil. Erosion fronts (“rofabards”) form^[17] and soil and sand begin to encroach upon and weaken remaining vegetation (States III and IV). Remnants of vegetation and soils gradually disappear leaving glacial till or sandy surfaces behind (States V and VI). This desertification process is further shown in Fig. 3.

The cost of restoration associated with degradation increases rapidly within Stages III–V and becomes almost prohibitive in Stage VI (Fig. 3C), where nutrients, seed sources, etc. have been completely lost.^[16] For further elaboration, see also the work of Tongway and Hindley.^[18]

THE NATURE AND EXTENT OF SOIL EROSION AND DESERTIFICATION

Desertification continues to be a major threat to Iceland’s natural resources. A national assessment of soil erosion and desertification at a scale 1:100,000 was conducted over the period of 1991–1996.^[14] It is based on classification of

Table 1 Division of land in Iceland according to erosion classes.

Erosion class	Area (km ²)	Percentage of whole
0 No erosion	4,148	4
1 Little erosion	7,466	7.3
2 Slight erosion	26,698	26
3 Considerable erosion	23,106	22.5
4 Severe erosion	11,322	11
5 Extremely severe erosion	6,375	6.2
Mountains	9,794	9.5
Glaciers	11,361	11.1
Rivers and lakes	1,436	1.4
Unmapped	1,010	1
Total	102,716	100

Source: From Arnalds, Thorarinsdottir, et al.^[14]

erosion forms that can be identified on the landscape. The survey indicates that 40% of Iceland is experiencing serious soil erosion (Table 1; Fig. 4). Areas with high mountains, glaciers, rivers, and lakes were excluded from the overall assessment of land.

Soil erosion in Iceland is characterized by two overall mechanisms. The first is sand encroachment, where sand buries vegetated areas resulting in bare sandy deserts. The second involves mainly water and wind, which results in the removal of rich Andosol soils.

Sandy deserts with active Aeolian processes characterize nearly 22,000 km² or about 21% of the total area of Iceland.^[3] In large parts of the Icelandic highlands (about 13,000 km²), sand has drifted over the old gravel surface left by the Pleistocene glacier. A large part of

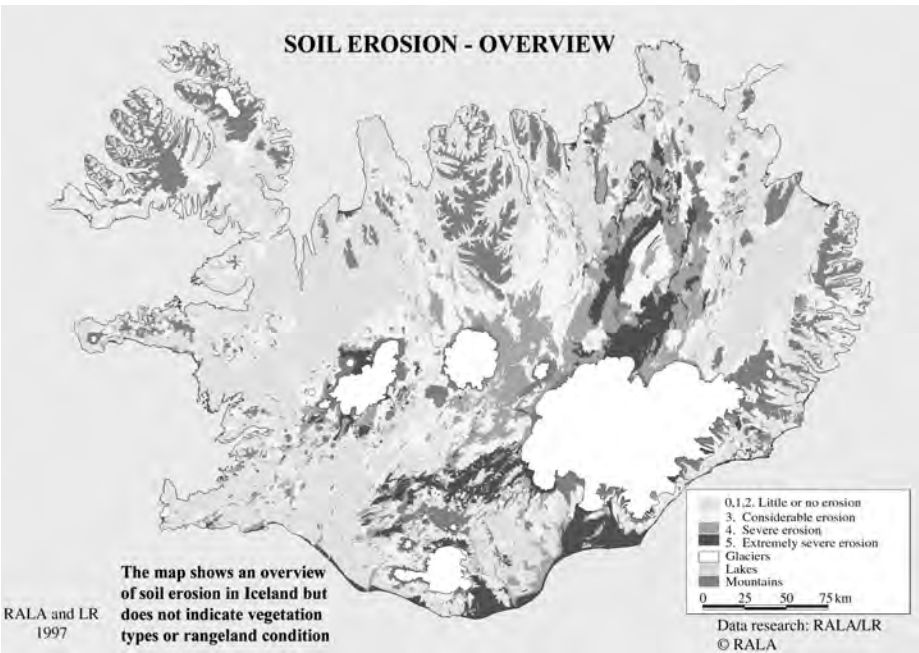


Fig. 4 Serious soil erosion characterizes 40% of Iceland, indicated here by the yellow (considerable), brown (severe), and red (extremely severe erosion) colors.

Source: From Arnalds, Thorarinsdottir, et al.^[14]

these areas were previously covered with 50–200 cm thick Andosols.

Erosion spots, openings in the vegetation with active erosion, are common on 27% of Iceland. Such erosion is mostly found in areas with low intensity erosion, Classes 1 and 2 (Table 1), but commonly leads to more severe erosion such as the formation of erosion fronts, the “rofabard.”

The “rofabard” is one of the most striking erosion features in Iceland. These are erosion escarpments where vegetated Andosols are being truncated from the surface and barren desert is left behind (Fig. 1). This form of erosion is prominent on about 20% of the area of Iceland and is a major environmental problem. Other forms of soil erosion are also active in destroying the Icelandic ecosystems.^[14]

Other common erosion forms are solifluction, landslides, gullies, soil movement on gravel flats (eroded land), and scree (for further, see the work of Arnalds et al.).^[14]

CONCLUSION

The term “desert” can have many different meanings. The term desertification thus may be regarded as a vague concept, as discussed by Arnalds^[2] in his appeal for a broader perspective on this global issue.

The scale of ecosystem disturbance in Iceland, where barren deserts have replaced vegetation and thick soils in many areas regardless of ample precipitation, demonstrates the global nature of this severe environmental problem.^[19] In Icelandic, the word “eyðimörk” is commonly used to describe the barren areas of the arid parts of the world, the “deserts.” It has two parts: “eyði,” which means empty or deserted and “mörk,” which is an old term for woodland. This word thus has its roots in the most severe environmental problem of Iceland, the conversion of woodlands to barren landscapes. The word “auðn,” which means empty or deserted, is commonly used to describe large barren landscapes in Iceland.

The world’s forests and woodlands are being reduced at an alarming rate in many parts of the world, and large areas are being overgrazed. The weakening of the vegetative cover can lead to a chain of ecosystem disturbances reducing further the resilience of the ecosystems toward further degradation. If prolonged, this can lead to desertification in a wide range of moisture regimes, as demonstrated by the Icelandic experience described here.

REFERENCES

1. Archer, S.; Stokes, C. Stress, disturbance and change in rangeland ecosystems. In *Rangeland Desertification*; Arnalds, O., Archer, S., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 5–16.
2. Arnalds, O. Desertification: An appeal for a broader perspective. In *Rangeland Desertification*; Arnalds, O., Archer, S., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 5–16.
3. Arnalds, O.; Gísladóttir, F.O.; Sigurjonsson, H. Sandy deserts of Iceland: An overview. *J. Arid Environ.* **2001**, *47*, 359–371.
4. Thurow, T.L. Hydrologic effects on rangeland degradation and restoration processes. In *Rangeland Desertification*; Arnalds, O., Archer, S., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 53–66.
5. van der Leeuw, S.E., Ed. *The Archaeomedes Project—Understanding the Natural and Anthropogenic Causes of Land Degradation and Desertification in the Mediterranean Basin*; Office for Official Publications of the European Communities: Luxembourg, 1998; Vol. 5, 440 pp.
6. Sigurdsson, S. Birki á Íslandi [Birch in Iceland]. In *Skógar-mál*; Gudmundsson, H., Ragnarsson, H., Thorsteinsson, I., Jonsson, J., Sigurdsson, S., Blöndal, S., Eds.; 1977; 146–172. Edda Print, Reykjavík. (In Icelandic).
7. Bjarnason, H. Comments on the history of Icelanders in relation to woodland destruction. *Yearb. Icel. For. Soc.* **1974**, *30*–43. (In Icelandic, with English summary).
8. Arnalds, O. Soils and soil erosion in Iceland. In *Geochemistry of the Earth’s Surface*; Armannsson, H., Ed.; A.A. Balkema: Rotterdam, 1999; 135–138.
9. Arnalds, O.; Kimble, J. Andosols of deserts in Iceland. *Soil Sci. Soc. Am. J.* **2001**, *65*, 1778–1786.
10. Arnalds, A. Ecosystem disturbance in Iceland. *Arct. Alp. Res.* **1987**, *19*, 508–513.
11. Arnalds, A. Landgæði á Íslandi fyrr og nú [Land resources in Iceland, past and present]. In *Icelandic Yearbook of Soil Conservation*; Arnalds, A., Ed.; Soil Conservation Service: Gunnarsholt, 1988; Vol. 1, 13–31. (In Icelandic).
12. Thorsteinsson, I. The effect of grazing on stability and development of northern rangelands: A case study of Iceland. In *Grazing Research at Northern Latitudes*; Gudmundsson, O., Ed.; Plenum Press: New York, 1986; 37–43.
13. Aradóttir, A.L.; Arnalds, O. Ecosystem degradation and restoration of birch woodlands in Iceland. In *Nordic Mountain Birch Ecosystems*; Wielgolaski, F.E., Ed.; UNESCO: Paris, 2001; Vol. 27, 295–308.
14. Arnalds, O.; Thorarindóttir, E.F.; Metusalemsson, S.; Jonsson, A.; Gretarsson, E.; Arnason, A. *Soil Erosion in Iceland*; The Soil Conservation Service and Agricultural Research Institute: Reykjavík, 2001; 1–121.
15. Gísladóttir, G. Environmental characterisation and change in south-western Iceland. *Dep. Phys. Geogr. Stockh. Univ. Diss. Ser.* **1998**, *10*, 1–188.
16. Aradóttir, A.; Arnalds, O.; Archer, S. Hnignun groðurs og jarðvegs [Degradation of vegetation and soils]. In *Icelandic Yearbook of Soil Conservation*; Arnalds, A., Ed.; Soil Conservation Service: Gunnarsholt, 1992; 73–82.
17. Arnalds, O. The Icelandic ‘rofabard’ soil erosion features. *Earth Surf. Processes Landf.* **2000**, *25*, 17–28.
18. Tongway, D.; Hindley, N. Assessing and monitoring desertification with soil indicators. In *Rangeland Desertification*; Arnalds, O., Archer, S., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 89–98.
19. Arnalds, O.; Aradóttir, A.L.; Thorsteinsson, I. The nature and restoration of denuded areas in Iceland. *Arct. Alp. Res.* **1987**, *19*, 518–525.

Desertification: Impact

David A. Mouat

Division of Earth and Ecosystem Sciences, Desert Research Institute, Reno, Nevada, U.S.A.

Judith Lancaster

Desert Research Institute, Reno, Nevada, U.S.A.

Abstract

Desertification's impacts on landscapes, biodiversity, and human populations are moderately well documented, and there are numerous suites of indicators in use for assessment and monitoring. Studies of the actual processes involved are valuable, particularly in areas that are marginal for desertification susceptibility, and are contributing to our understanding and therefore to our ability to meaningfully intervene to reduce or mitigate the effects of desertification on both the environment and human populations. Desertification is a phenomenon that will not disappear as a result of human intervention but may be reduced in severity or extent by a strategy including increasing the understanding of processes involved, monitoring, and investigating alternative management options.

INTRODUCTION

The United Nations (UN) defines desertification as follows: "Land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities."^[1] These dryland regions comprise 41% of the global land area, or 6150 million ha,^[2] and are present in every continent. Hyperarid regions are almost rainless, natural deserts, and not susceptible to desertification processes. Areas of concern are those that are desertified, that are at risk for increases or acceleration of desertification processes, and that are on the threshold of change but where sustainable human life is still possible.

Economic and political factors have changed the patterns and strategies of human occupation of dryland regions, where previous lifestyles of transhumance or nomadism and minimal involvement in a market economy permitted human occupation of arid and semiarid areas even in times of drought, without causing ecosystem stress.

MANIFESTATIONS OF THE DESERTIFICATION PROCESS

Dryland ecosystems are fragile, highly vulnerable to natural climatic fluctuations, and susceptible to desertification. Desertification impacts not only human populations and their livelihood but also the landscape—namely vegetation, soils, and hydrology—as well as insect, animal, and bird populations and biodiversity as a whole. Dryland soils are particularly at risk—their low levels of biological activity, organic matter, and aggregate stability can easily result in a

breakdown or decline of soil structure, accelerated soil erosion, reduction in moisture retention, and increase in surface runoff.^[3] Reduction in plant cover will exacerbate these processes, leading to a change of scale in the spatial distribution of soil resources,^[4] an increasingly "patchy" landscape, and a decline in sustainability that is difficult or impossible to reverse.^[5]

Dryland vegetation communities include grassland, shrubland, woodland, savanna, and steppe (Fig. 1) and vary according to climatic, edaphic, hydrologic, and anthropogenic factors.^[3] Change in vegetation community structure and reduction in cover may be a catalyst for desertification processes.^[6] A shift from grassland-dominated to shrub-dominated rangeland seems to be ubiquitous^[7,8] and is generally associated with increased soil runoff. The main causes of this vegetation change seem to be grazing, fencing, and alterations to the natural fire regime, in association with climatic variability.^[9]

The *World Atlas of Desertification*^[10] suggests that overgrazing, agricultural practices, the overexploitation of vegetation, and deforestation are the four most significant causes of soil degradation although bioindustry plays a locally important role in some countries. Soil degradation may result from displacement (by wind and water erosion) or internal deterioration by chemical or physical variables.^[11] Erosion of soil by water results in loss of topsoil as well as changes to the landscape such as gully and rilling. Loss of topsoil has an impact on both natural vegetation and cultivated crops^[12] and is the dominant erosion process in Africa.^[10] Wind erosion is most severe in areas that have been disturbed by human activity—e.g., by grazing pressure that reduces plant cover. Chemical

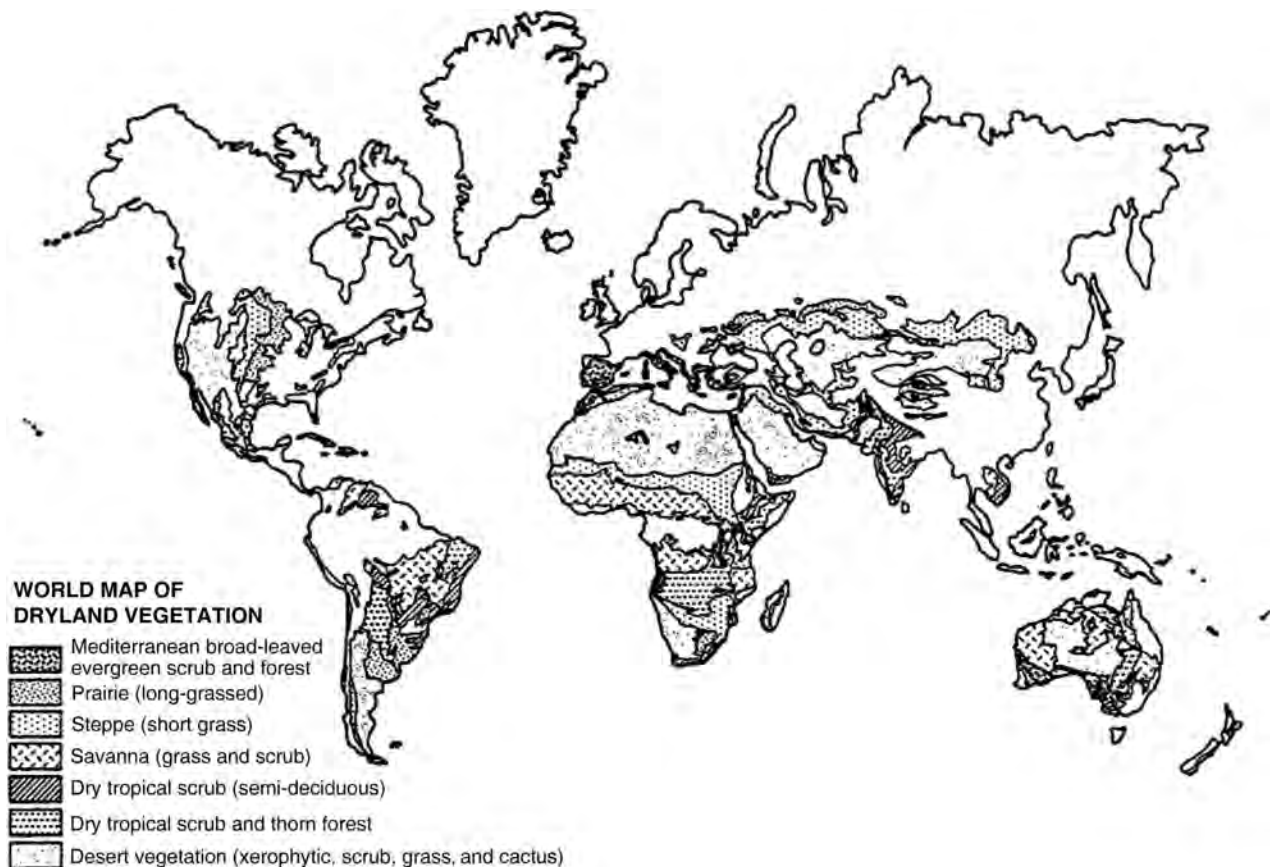


Fig. 1 Global distribution of vegetation communities.

Source: From Williams & Balling.^[3]

deterioration can be divided into the following three classes: a loss of nutrients, salinization, and acidification.^[10] A change in soil nutrients and a patchy distribution of nutrients at the landscape scale is a consequence of desertification^[5] and in some cases may be partially due to erosion of fine particles by wind and water.^[13] Physical changes in soil structure result from compaction, crusting, and waterlogging and—like salinization and acidification—are exacerbated by pressure of human land use and widespread irrigation.^[10]

Desertification is linked to poverty^[12,14] and tends to result in out-migration, initially of young people and men. This changes family structure and places burdens and constraints on the women, children, and old people remaining in the settlements—which in turn changes the patterns of land use and may exacerbate desertification.^[12] Loss of human dignity is perhaps the most insidious consequence of the desertification process—it is difficult to measure and difficult to reverse.

Measurement of the extent and severity of desertification has been an ongoing process since the 1930s and is well documented in many countries. However, in order to comply with UN treaties, management plans must be developed and implemented, education programs initiated, and local communities involved in decision-making. To do this

effectively, it is necessary to identify the areas that will most benefit from intervention and where desertification processes may be stabilized or reversed. International conferences addressing these issues were held in Tucson, Arizona, U.S.A., in 1994 and 1997^[15,16] at which indicators of desertification were much discussed.

INDICATORS OF DESERTIFICATION

Although some indicators may be relevant universally,^[17] a standardized list of indicators is not possible as the climatic, geographic, and anthropomorphic conditions of desertification are variable, as are technology, human, and financial resources. Most indicators tend to be of a “snapshot” nature that provide measures of condition and trend and a measure of what is “normal” only when compared in time and space. Process-based studies in semiarid and arid ecosystems that appear to be structurally and functionally intact^[18] may provide baseline data on the expected and inherent variability of systems, to permit the establishment of thresholds indicative of irreversible change.^[19]

Reporting on environmental indicators for the land, to meet Australia’s UN obligations, Hamblin^[20] selected

indicators that would distinguish between anthropogenic interventions and natural causes and recommended “key indicators” of condition, pressure, and response that would describe trend and impact on the environment. The indicators are grouped by process in relation to change in erosion, habitat, hydrology, biota, nutrients, and pollution.

The 1994 International Desertification Conference in Arizona identified nutrient availability, water budget, energy balance, and biological diversity as key ecosystem processes affected by desertification which apply across international boundaries and again recommended indicators that were grouped by process—in this case, nutrient availability, water budget, energy balance, and biological diversity. Of the total 28 recommended indicators, the following may be the most significant:

1. Infiltration, both under and between plants;
2. Cover type and distribution;
3. Erosion as indicated by rills, gullies, pedestalling, litter movement, flow patterns, and fetch length;
4. Depth to water table;
5. Normalized Difference Vegetation Index as a measure of greenness;
6. Surface temperature; and
7. Ratio of native to exotic species for flora and fauna.

There is a paucity of demographic or socioeconomic indicators in this list. Of the numerous indicators of desertification in general use, those concerning socioeconomic factors are the least developed, and there is a disconnect in many areas between the communities affected by desertification processes, the scientists conducting research, and policy-makers and managers.^[21] The disconnect between communities, researchers, and policy-makers has been a subject of concern in South Africa and Namibia for some time and has been addressed by several projects.^[22–24] The Karoo veld (rangeland) assessment handbook produced by Milton and Dean^[25] is a successful example of orienting scientific data and their interpretation to a specific non-science audience.

Soil is variable and complex, and the methods for assessing its condition are slow, tedious, and expensive; to compound the issue, differences in soil type may be mistaken for differences in soil condition.^[26] Also targeted to ranchers and managers, Tongway’s rangeland soil condition assessment handbook^[27] proposes an approach to assess the soil condition based on recognizing and classifying the processes related to erosion, infiltration, and nutrient cycling.

The world’s drylands provide critical habitats for migratory birds, and other wildlife,^[12] and the maintenance of biodiversity in arid and semiarid areas is a major concern. Measures of floral, faunal, and avian diversity and reproductive success must be included in the list of indicators used to assess and monitor the status and extent of desertification. However, like demographic indicators, those

involving biodiversity can be difficult to define, measure, and analyze and are frequently site specific. They may, however, provide the elusive “early warning” of a susceptibility for desertification that researchers and managers are seeking.

TECHNOLOGY, MEASUREMENT, AND ANALYSIS

The field data needed for desertification assessment and monitoring, such as soils and vegetation measurements, are typically collected at the plot scale, and many researchers, managers, and policy-makers have questioned the use of remote sensing because it provides a generalized measurement at scales of tens of meters to kilometers. However, there are numerous remote-sensing systems, operating at different spectral, spatial, temporal, and radiometric scales. The latest in the Earth Observing System series of missions launched in December 1999 carried new sensors, two of which are the Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) and the Moderate Resolution Imaging Spectrometer. The ASTER mission objectives focus on understanding the earth as a system and the construction of models of earth’s global dynamics—including desertification.^[28]

The long-term perspective critical to desertification assessment is also an essential component of the evaluation of future management options and the development of remediation, land use, and management plans. A newly developed technique, i.e., alternative futures assessment, models potential changes to the landscape in a geographic information system (GIS) over several timescales, including projections for population growth. Models for basic environmental processes are then operated on the results of the change scenarios to show cumulative effects on the system as a whole.^[29] Workshops and questionnaires are used to ascertain current and future attributes of the area that are considered by stakeholders the most desirable, to identify plausible change scenarios, and to evaluate the results.

Long-term ecological studies at plot, regional and local scales, remote-sensing data, GIS, and alternative futures assessments are components of an integrated assessment, monitoring, and management strategy that is evolving in response to the need for greater interaction and communication between scientists, managers, and communities. This is a global concern, and the following case study for Southeastern Arizona discusses research in the context of a strategy that involves local communities, landowners, and managers in the decision-making process.

CASE STUDY—THE SAN PEDRO RIVER BASIN

The San Pedro River rises in Mexico and flows north through the semiarid shrub steppe of Southeastern Arizona. The Bureau of Land Management (U.S. Department of

Interior) manages the San Pedro Riparian National Conservation Area immediately bordering the river, where there is one of the highest animal biodiversity totals in North America. Vegetation change, especially the deterioration of grasslands due in part to grazing and the suppression of fire, has been a concern in the area for some years.^[30] This change was quantified using aerial photography, remote sensing, and GIS to compare the extent and change over time between the eight vegetation classes in the region.^[9] The most striking change between 1974 and 1987 was the considerable fragmentation of vegetation classes. This patchiness of the landscape is indicative of loss of biodiversity and could be an early warning of desertification in the area.^[9]

A multiagency, multinational, global change initiative to investigate the consequences of natural and human-induced environmental change in semiarid regions was started in 1995 in the San Pedro River Basin. Several multidisciplinary projects have been initiated, and ASTER imagery is being acquired for the region. Among the studies being conducted in the San Pedro River Basin is an alternative futures assessment. This latter study has the following two objectives: to develop an array of plausible alternative patterns of land use and to assess their impacts on biodiversity, vegetation dynamics, fire regimes, hydrology, and aesthetics; and as a pilot study to develop a methodology applicable to other areas.^[31]

CONCLUSION

Desertification impacts on landscapes, biodiversity, and human populations are moderately well documented, and there are numerous suites of indicators in use for assessment and monitoring.^[20] Studies of the actual processes involved are valuable, particularly in areas that are marginal for desertification susceptibility, and are contributing to our understanding and therefore to our ability to meaningfully intervene to reduce or mitigate the effects of desertification on both the environment and human populations.^[18]

In the long term, we do not know what "desertification" really means, nor what the impacts to humans, flora, fauna, and the landscape will be. Desertification is a phenomenon that will not disappear as a result of human intervention but may be reduced in severity or extent by a strategy including increasing the understanding of processes involved, monitoring, and investigating alternative management options. In particular, the societal impacts of desertification may be mitigated by involving local communities in the policy and decision-making process.

REFERENCES

1. *Report of the United Nations Conference on Environment and Development*; United Nations Conference on Environment and Development (UNCED): United Nations, New York, 1992; Vol. 1; 486 pp.
2. Kassas, M. Desertification: A general review. *J. Arid Environ.* **1995**, *30*, 115–128.
3. Williams, M.A.J.; Balling, R.C., Jr. *Interactions of Desertification and Climate*; Arnold: London, 1996; 270 pp.
4. de Soyza, A.G.; Whitford, W.G.; Herrick, J.E.; Van Zee, J.W.; Havstad, K.M. Early warning indicators of desertification: Examples of tests in the Chihuahuan desert. *J. Arid Environ.* **1998**, *39* (2), 101–112.
5. Schlesinger, W.H.; Reynolds, J.F.; Cunningham, G.L.; Huenneke, L.F.; Jarrell, W.M.; Virginia, R.A.; Whitford, W.G. Biological feedbacks in global desertification. *Science* **1990**, *247*, 1043–1048.
6. Le Houérou, H.N. Climate change, drought and desertification. *J. Arid Environ.* **1996**, *34*, 133–185.
7. Ludwig, J.A.; Tongway, D.J. Desertification in Australia: An eye to grass roots and landscapes. In *Desertification in Developed Countries*; Mouat, D.A., Hutchinson, C.F., Eds.; Kluwer Academic Publishers: Dordrecht, 1995; 231–237.
8. Milton, S.J.; Dean, W.R.J. South Africa's arid and semiarid rangelands: Why are they changing and can they be restored? In *Desertification in Developed Countries*; Mouat, D.A., Hutchinson, C.F., Eds.; Kluwer Academic Publishers: Dordrecht, 1995; 245–264.
9. Mouat, D.A.; Lancaster, J. Use of remote sensing and GIS to identify vegetation change in the upper San Pedro River Watershed, Arizona. *Geocarto Int.* **1996**, *11* (2), 55–67.
10. Middleton, N.; Thomas, D. *World Atlas of Desertification*, 2nd Ed.; Middleton, N., Thomas, D., Eds.; Arnold: London, 1997; 182 pp.
11. Oldeman, L.R. *Guidelines for General Assessment of the Status of Human-Induced Soil Degradation*; International Soil Reference and Information Center: Wageningen, 1988.
12. United Nations Convention to Combat Desertification. <http://www.unccd.int/main.php> (accessed December 2000).
13. Schlesinger, W.H.; Ward, T.J.; Anderson, J. Nutrient losses in runoff from grassland and shrubland habitats in Southern New Mexico: II. Field Plots. *Biogeochemistry* **2000**, *49*, 69–86.
14. Glantz, M.H. *Drought Follows the Plow: Cultivating Marginal Areas*; Glantz, M.H., Ed.; Cambridge University Press: Cambridge, 1994; 197 pp.
15. Mouat, D.A.; Hutchinson, C.F. *Desertification in Developed Countries*; Mouat, D.A., Hutchinson, C.F., Eds.; Kluwer Academic Publishers: Dordrecht, 1995; 363 pp.
16. Mouat, D.A.; McGinty, H.K., Eds. *Combating Desertification: Connecting Science with Community Action*; *J. Arid Environ.* (Special Issue); Academic Press: London, 1998; 340 pp.
17. Mouat, D.A.; Lancaster, J.; Wade, T.; Wickham, J.; Fox, C.; Kepner, W.; Ball, T. Desertification evaluated using an integrated environmental assessment model. *Environ. Monit. Assess.* **1997**, *48*, 139–156.
18. Dean, W.R.J.; Milton, S.J. *The Karoo: Ecological Patterns and Processes*; Dean, W.R.J., Milton, S.J., Eds.; Cambridge University Press: Cambridge, 1999; 374 pp.
19. Tausch, R.J.; Wigand, P.E.; Burkhardt, J.W. Plant community thresholds, multiple steady states, and multiple successional pathways: Legacy of the quaternary? *J. Range Manag.* **1993**, *46* (5), 439–447.

20. Hamblin, A. *Environmental Indicators for National State of the Environment Reporting: The Land*; Department of the Environment: Canberra, 1998; 124 pp.
21. Mouat, D.A.; McGinty, H.K.; McClure, B.C. Introduction. In *Combating Desertification: Connecting Science with Community Action*; Mouat, D.A., McGinty, H.K., Eds.; Special Issue: J. Arid Environ.; Academic Press: London, 1998; 97–99.
22. Seely, M.K. Can Science and community action connect to combat desertification? J. Arid Environ. **1998**, *39* (2), 267–278.
23. van Rooyen, A.F. Combating desertification in the Southern Kalahari: Connecting science with community action in South Africa. J. Arid Environ. **1998**, *39* (2), 285–297.
24. Milton, S.J.; Dean, W.R.J.; Ellis, P.R. Rangeland health assessment: A practical guide for ranchers in arid karoo shrublands. J. Arid Environ. **1998**, *39* (2), 253–266.
25. Milton, S.J.; Dean, W.R.J. *Karoo Veld: Ecology and Management*; Agricultural Research Council Range and Forage Institute: Lynn East, 1996; 94 pp.
26. Tongway, D. Monitoring soil productive potential. J. Arid Environ. **1998**, *39* (2), 303–318.
27. Tongway, D.J. *Rangeland Soil Condition Assessment Manual*; Commonwealth Scientific and Industrial Research Organization: Melbourne, 1993; 69 pp.
28. Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER). <http://asterweb.jpl.nasa.gov> (accessed December 2000).
29. Steinitz, C.; Binford, M.; Cote, P.; Edwards, T.; Ervin, S.; Forman, R.T.T.; Johnson, C.; Kiester, R.; Mouat, D.; Olson, D.; Shearer, A.; Toth, R.; Wills, R. *Biodiversity and Landscape Planning: Alternative Futures for the Region of Camp Pendleton, California*; Harvard University, Graduate School of Design: Cambridge, 1996; 142.
30. Hastings, J.R.; Turner, R.M. *The Changing Mile: An Ecological Study of Vegetation Change with Time in the Lower Mile of an Arid and Semi-Arid Region*; University of Arizona Press: Tucson, 1965.
31. <http://www.gsd.harvard.edu/faculty/steinitz/sanpedro-river.html> (accessed December 2000).

Desertification: Mapping Constraints and Challenges

Pandi Zdruli

Land and Water Resources Management Department, International Center for Advanced Mediterranean Agronomic Studies (CIHEAM) of Bari, Valenzano, Italy

Michael Cherlet

European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy

Claudio Zucca

Integrated Water and Land Management (IWLMP) Research Program, International Center for Agricultural Research in the Dry Areas (ICARDA), Amman, Jordan

Abstract

Mapping desertification has proven to be a challenging task. Difficulties derive from its ambiguous definition and the comprehensive integration of various biophysical and socioeconomic indicators that need to be considered in the evaluation process. In the early 1990s, assessments and mapping were based primarily on expert opinions that introduced uncertainties and obvious shortcomings. Later on, with the development of the remote sensing technology and the advancement of the geographic information systems, global mapping became commonplace. Results showed that temporal and spatial scales are crucial components of the mapping and assessment process, but great difficulties arose comparing maps that were developed using different methodologies. Global maps could depict only general trends in desertification caused by human-induced land use changes or climatic variations but proved to be of limited value at local level. On the other side, local studies have problems of extrapolation. This necessitates the performance of mapping at various scales, but only after a methodological approach has been developed that accounts for all the components of the desertification process, allowing upscaling from global to local level and vice versa. Desertification mapping is under way by the World Atlas of Desertification (WAD) 3rd edition. The WAD places particular importance on case studies that document local realities affected by desertification as well as mitigation actions. Finally, concerned efforts must be made to develop and implement sustainable land use planning and land management techniques that arrest and reverse the negative consequences of desertification.

INTRODUCTION

What is Desertification?

Desertification is a term widely used by the scientific community and many other stakeholders associated to the United Nations Convention to Combat Desertification (UNCCD), but non-experts often confuse it with naturally occurring deserts.^[1] It was first Lavauden^[2] who used the term in Tunisia describing it as low rangeland productivity due to inadequate management. However, many researchers credit the French ecologist A. Aubréville^[3] who in 1949 applied the term desertification not to drylands, but to the tropical forests of Africa that receive much more rainfall than what conforms the official UNCCD definition. Aubréville noted that the influence of human activities, i.e., cultivation, deforestation, and accelerated erosion, contributed to the process of transformation of tropical forests into savannas and finally into desert-like environments.

It took more than two decades after Aubréville's warnings of desertification before the topic reached the political agenda. This was largely influenced by the impact of extended drought and moving deserts in the West African Sahel in the early 1970s^[4] that brought about the United Nations Conference on Desertification in Nairobi in 1977. Two decades later, the UNCCD entered into force reaffirming a previous United Nations Environment Programme's (UNEP)^[5] definition of desertification with very slight modifications: *desertification is land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities* (UNCCD, Article 1). The UNCCD uses the aridity index, calculated as the ratio between mean annual precipitation to mean annual potential evapotranspiration, to identify the drylands of concern. These areas are characterized by large water deficits, because potential evapotranspiration is much greater than precipitation. The aridity indices in the range of 0.05–0.65 (excluding polar and subpolar regions) identify the arid, semiarid, and dry subhumid zones, likely to be affected by desertification.

However, criticism has been raised, for instance, about the exclusion of hyperarid areas.^[6] On the other hand, even Iceland^[7] claims to have desertification problems! No matter the definition, the final outcome of this type of degradation has typically been considered to be either a reduction or a loss of both biological and economic productivity and of environmental quality derived by the complexity of interaction of various biophysical and socioeconomic factors.

The UNCCD definition of desertification has been widely debated over the years. This became even more critical after almost every country in the world signed the Convention, even those not located in the drylands. To overcome this situation and accommodate all the parties of the Convention, a new terminology was introduced by the UNCCD, namely desertification, land degradation, and drought (DLDD), reflecting the subjects the Convention wishes to be responsible for. For instance, Article 2 of Annex V of the UNCCD attempts to broaden the definition of desertification referring to specific conditions in Eastern Europe, mostly covering human-induced soil and land degradation, occurring under climatic regimes that fall outside arid, semiarid, and dry subhumid areas. Land resources in this context include soil, local water resources, and vegetation (or crops), and degradation is an ecological and economic reduction in resource potential, e.g., due to water erosion, wind erosion, a long-term reduction in the diversity of natural vegetation, and salinization and/or sodification. Therefore, in the broadest terms, desertification includes the degradation of land, soil, water, vegetation, and other resources.^[8] No matter the definition debate, the main goal of the Convention remains the same, as defined by its Article 2 that emphasizes effective actions as all levels to combat desertification particularly in Africa.^[9]

Desertification terminology was greatly enriched after the comprehensive analyses of dryland degradation made by Reynolds, Stafford Smith, et al.^[10] and Reynolds, Bastin, et al.^[11] and the appearance of dryland development paradigm (DDP) that shed new light on desertification and placed it into a new perspective. The DDP integrated biophysical and human-induced factors in a complex approach based on five principles used to assess the stability or disturbances and malfunctions of the dryland ecosystems. This is particularly relevant for these areas that despite being characterized generally by lower soil fertility and problematic climate (e.g., low rainfall and high evaporation rate) are home to more than one billion people who depend on them for their very survival.^[12]

The terminology used to describe resource base degradation creates often confusion, especially for the decision-makers, since no clear indication is made about the differences between soil degradation, vegetation degradation, land degradation, and desertification. However, this last is seldom considered as the final product of degradation and *includes all* of the abovementioned forms despite being used mainly to characterize dryland degradation. Problems

are further exacerbated when desertification, mostly a human-induced process, is combined with naturally occurring drought. They both globally cause great losses every year to agricultural production and contribute thus to food insecurity, famine, and poverty and can give rise to social, economic, and political tensions that can cause conflicts, forced migration, and further impoverishment of the local people. UNCCD^[9] places special importance to resource base degradation in terms of physical, chemical, and biological deterioration of the land system and its implications to the well-being of affected desertification areas.

Long before, in 1979, United Nations Food and Agriculture Organization (FAO)^[13] defined degradation as a process that lowers the current and/or future *capability of soils to produce* (quantitatively and/or qualitatively). Historically, this reflects the main school of thought at that time when *soil degradation* was seen purely as a process identified from the biophysical indicators; it was the domain of soil scientists, and few recognized that *land degradation comprises* many components of the land system, of which soil is only a part.

A further development in the terminology of resource base degradation came after the publication of the Millennium Ecosystem Assessment (MEA)^[14] that defined land degradation as *the reduction in the capacity of the land to perform ecosystem goods, functions and services that support the society and development*. MEA deserves credit for the fundamental change in the philosophy of natural resources management that was based since then on ecosystem's functions and services and not on a single component of it, such as the soil.

The Land Degradation Assessment in Drylands (LADA) project funded by the Global Environment Facility, implemented by UNEP, and executed by the United Nations FAO made an important contribution to the study of land degradation in the drylands. LADA operated from 2006 until 2010, and its main objective was to *develop a set of tools and methods to assess and quantify the nature, extent, severity, and impacts of land degradation on dryland ecosystems at a range of spatial and temporal scales*. To achieve this, the project developed its own definition of land degradation that was inspired by the MEA but was slightly changed to reflect the temporal aspect of the process and draw attention to the fact that the value of goods and services are dependent on the stakeholders concerned, and it is therefore a subjective concept.^[15] Full definition of land degradation according to LADA^[16] is as follows: *The reduction in the capacity of the land to produce ecosystem goods and services and assure its functions over a period of time for its beneficiaries*. Further on, LADA performed also a Global Assessment of Land Degradation (GLADA)^[17] and also developed an online Global Land Degradation Information System (GLADIS; http://www.fao.org/nr/lada/index.php?option=com_content&view=article&id=161&Itemid=113&lang=en) that is under improvement.^[16] GLADIS based its assessment on the status and trends of

biomass, soil health, water quality, biodiversity, economic benefit, and social/cultural benefits showing graphically the results in the so-called radar diagrams.

Following the above considerations, desertification includes all forms of resource base degradation deriving from human-induced activities and climatic adversaries and is expressed by the inefficiency of the dryland ecosystems to maintain economic and ecological functions and to provide goods and services.

METHODS AND APPROACHES

Is It Possible to Map Desertification?

Initial mapping efforts

The first assessment of the state of human-induced Global Assessment of Soil Degradation (GLASOD) was published in 1991.^[18] It surprised both the scientific community and many top environmental decision-makers around the world by estimating that 17% of the 11.5 billion hectares of vegetated land on earth was degraded, largely by erosion, and that 1 in 6 hectares of this land could no longer be cultivated.^[19] GLASOD has been debated over the years for its shortcoming, being based on expert opinion and on a mostly qualitative assessment. It produced disputable results and offered poor relationships between soil degradation assessment and policy development and implementation.^[20] However, GLASOD deserves credit because it contributed greatly to raising the profile of soil degradation at the highest levels of decision-making and remains the only reference for soil degradation at a global scale. The GLASOD methodology was applied also regionally in Asia,^[21] Russia,^[22] and Central and Eastern Europe.^[23] In 2003, Lal^[24] modified GLASOD data and based on information from Oldeman^[25] and FAO^[26] estimated the extent of various types of soil degradation globally as well as their distribution on continental level (Table 1).

Attempts to map desertification have been made by various authors.^[27–32] For instance, Eswaran et al.^[33]

developed Fig. 1 estimating the vulnerability to desertification based on a reclassification of global climate and soil maps at a 2-minute grid cell resolution. The same authors also correlated soil data with population density and identified that the Indian subcontinent was at a high risk of being affected by desertification^[34] due to increased human pressure and unfavorable biophysical indicators that could worsen due to climate change.

The advent of remote sensing (RS) and geographical information systems (GIS)

After the GLASOD experience, all other global assessments of desertification have been mainly based on RS technology and GIS applications. GLADA integrated a range of indicators obtained from RS data with existing global data sets and models to identify degrading areas or regions where degradation may have been reversed. Otherwise, this methodology is known as “hot spot” and “bright spot”. The main indicators used in the assessments were net primary productivity (NPP), rainfall use efficiency (RUE), aridity index, rainfall variability, and erosion risk. Furthermore, these indicators were interpreted on the basis of the global land use systems (Fig. 2) and another series of specific indicators typical for a given area such as land cover, urban and protected areas, livestock pressure, irrigation, crops, temperature and thermal regime, rainfall regime, dominant soils and terrain slope, population density, and poverty level.

The remotely sensed normalized difference vegetation index (NDVI) is used as a proxy, and its deviation from the norm serves as an indicator of land degradation and improvement if other factors that may be responsible (climate, soil, terrain, and land use) are accounted for. Rainfall effects may be accounted for by RUE (i.e., NDVI per unit of rainfall) and residual trends of NDVI such as temperature effects by energy use efficiency (derived from annual accumulated temperature). Land degradation is indicated by a declining trend of climate-adjusted NPP and land improvement by an increasing trend. Translation of NDVI to NPP could enable the economic appraisal of land degradation,^[14] and one of

Table 1 Estimates of global extent of soil degradation by different processes.

Region	Total land area (M ha)	Total degraded area (M ha)	Total degraded area (%)	Water erosion (M ha)	Wind erosion (M ha)	Physical degradation (M ha)	Chemical degradation (M ha)
Africa	2,964	494	17	227	186	19	62
Asia	3,085	749	24	441	222	12	74
South America	1,753	243	14	123	42	8	70
Central America	108	63	58	46	5	5	7
North America	2,029	96	5	60	35	1	—
Europe	2,260	218	10	114	42	36	26
Oceania	849	102	12	83	16	2	1
World	13,048	1,965	15	1,094	548	83	240

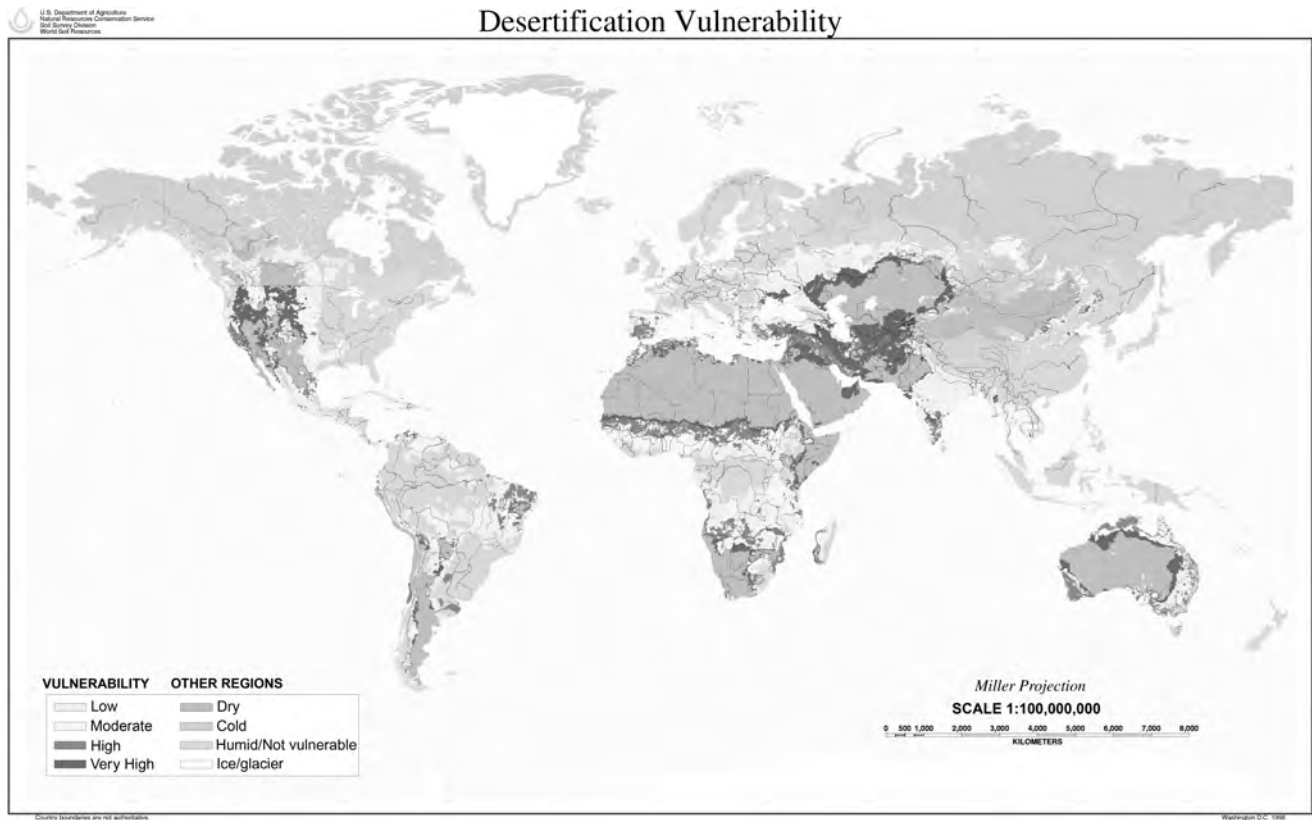


Fig. 1 Desertification vulnerability based on soil and soil climate data.

such results is shown in Fig. 3. Based on this methodological approach and data provided for the period 1981–2006 deriving from the Global Inventory Modeling and Mapping Studies NDVI database, Bai et al.^[35] reported that 23.54% of the earth is degraded and 1.5 billion people were affected over this period. As the focus in studies such as these and, e.g., GLASOD is different, it is very hard to spatially compare these to deduct what happened to land degradation at global scale within a given period.

Fig. 4 is a most challenging one produced by LADA. It describes the overall status in provision of biophysical ecosystem services and the processes of declining these services by considering the combined value of each biophysical axis (biomass, soil, water, and biodiversity). Data show that 32% of land globally is in areas with high provision of biophysical goods and services status, but with medium to strong degradation processes, while a large part of the global population (27%) lives in areas with a low status and a medium to strong degradation trend.^[36]

The new world atlas of desertification (WAD)

A new initiative to develop, publish, and establish an online digital new WAD^[37] is coordinated by the European Commission (EC) and UNEP. The WAD endorses a new approach that relies on up-to-date information on the combined biophysical and socioeconomic

situation in order to provide a convergence of evidence of potential on-going land degradation processes. Cause–effect analysis of possible land degradation sources also identifies trade-offs and hence offers routes for solutions, such as sustainable land management (SLM) options. WAD intends to be a global updatable reference on where land degradation processes are ongoing and what could be done to reverse these.

WAD is based on the scientific findings that suggest that the manifold reported causes of land degradation can be clustered into well-defined groups or main desertification issues^[38,39] and that these interplay differently at various scales. Using the most up-to-date, and, at global scale, available information on these issues, e.g., on drought, human population changes, and inappropriate land use, it analyses the cause–effect relations with observed land system productivity dynamics as derived from satellite time series and regarding these as coupled processes. Consequently, areas where a single or multitude of land stress processes, such as an apparent reduced land system productivity, drought events, and agriculture intensification or expansion, are simultaneously ongoing are identified as areas probably prone to land degradation. In this way, trade-offs and solution pathways for land use and land use strategies can be better identified.

Addressing one of the major problems in mapping land degradation at various scales, this WAD concept is scale independent. WAD, starting at global level, illustrates how

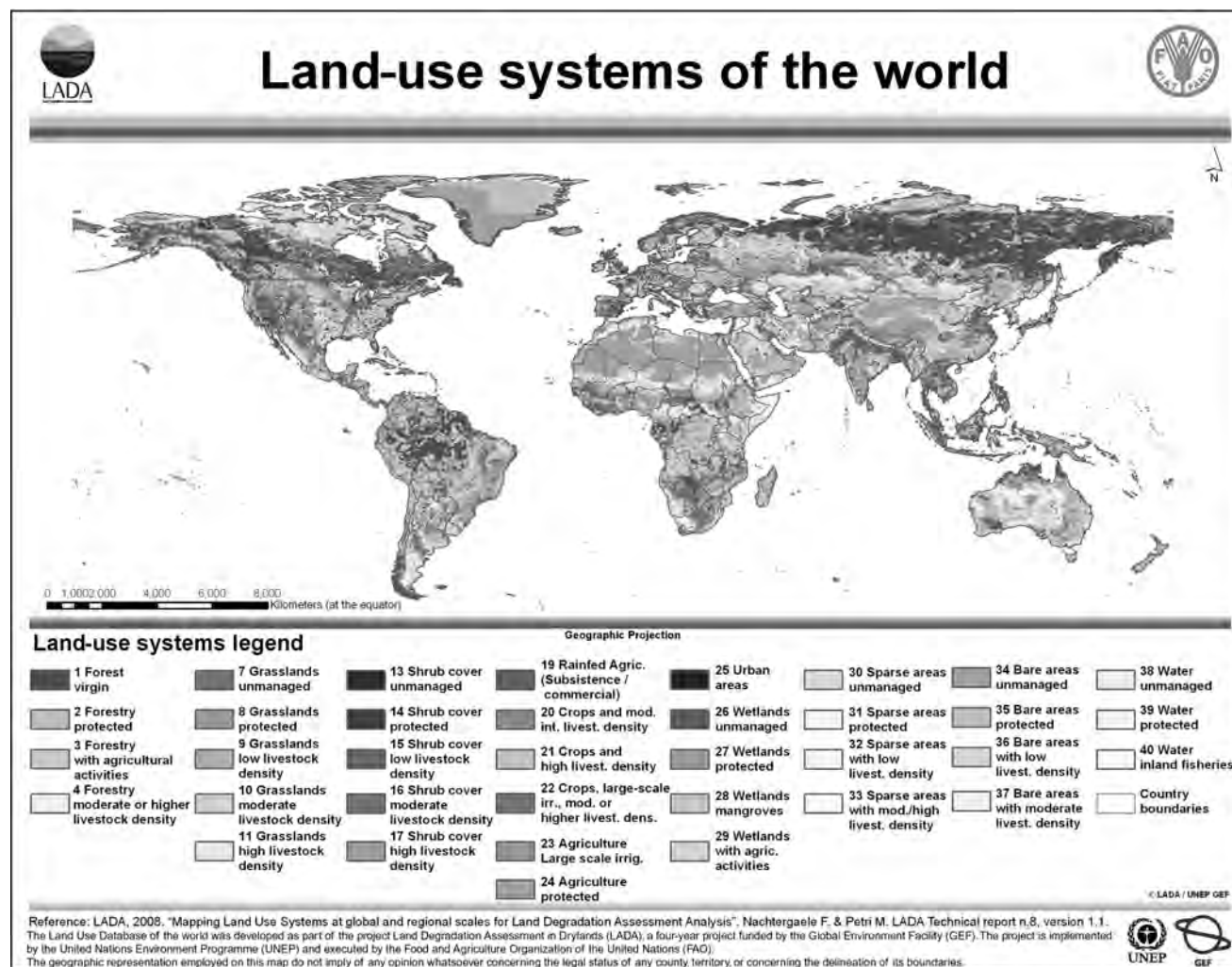


Fig. 2 Land use systems of the world.

an assessment can be taken down to the local conditions by adding more detailed contextual information while keeping thematic compatibility at all scales. Spatially inventorying such knowledge offers a basis for designing and economically evaluating options to mitigate desertification, in particular through implementation of SLM practices. WAD therefore wants to promote a change from a negative focus on land degradation, e.g., mapping merely problems, to a more useful positive consideration in that areas are marked to a degree that improvements in land management are needed jointly with identification of the causes on which mitigation strategies need to concentrate.

The Joint Research Centre of the EC has developed the land system productivity dynamics data layer and a tool to compile this from RS time series. Figs. 5 and 6 represent an analysis of 15 years of SPOT VGT-based phenology and productive variables, for the period of 1999–2013 at 1-km resolution. It combines long-term undercurrents with performance of productive seasonal biomass developments of the land system. This is not a land degradation map in itself but an important layer for its evaluation.

CONSTRAINTS AND CHALLENGES

If a comparison was to be made between Figs. 1 and 4, very few areas are in common and this, caused by differences in methods and time periods considered, is the crucial handicap in global desertification mapping that often creates confusion for the end users, mostly decision-makers. Fig. 1, for instance, puts almost the whole of China into humid and not vulnerable areas, while the LADA map provides better details on land degradation classes. Same holds true for much of the Western Europe and eastern parts of the United States and Canada.

Other problems related to desertification assessment and mapping are linked to spatial and temporal scales. Such global maps provide only approximate assessments on the distribution of degradation but are insufficiently detailed to be useful at local level. Conversely, the temporal aspect is often a contradictory constraint because if a comparison has to be made between the present status of degradation with an earlier stage, the comparison period has somehow to be established arbitrarily. GLASOD, e.g., used a period of

Global loss of annual net primary productivity between 1981 and 2006

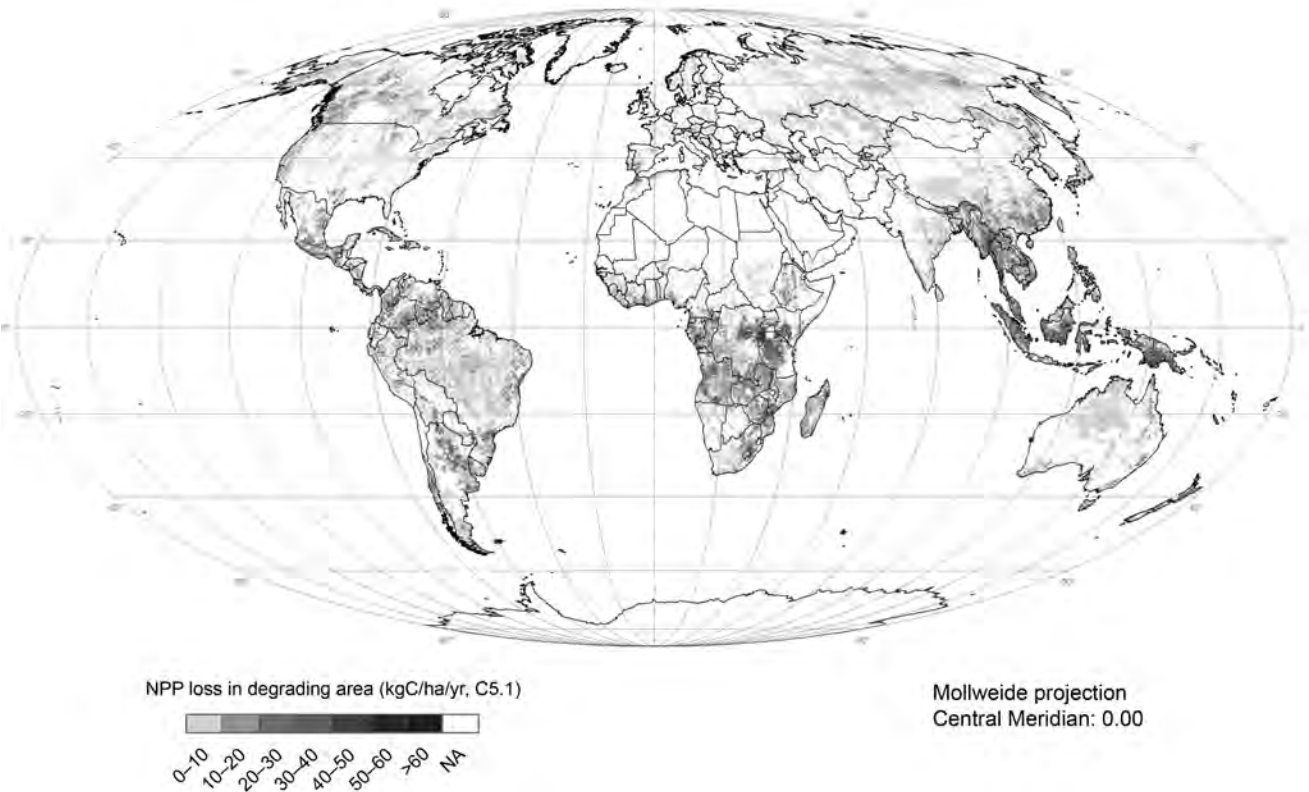


Fig. 3 Global loss of net primary productivity between 1981 and 2006.

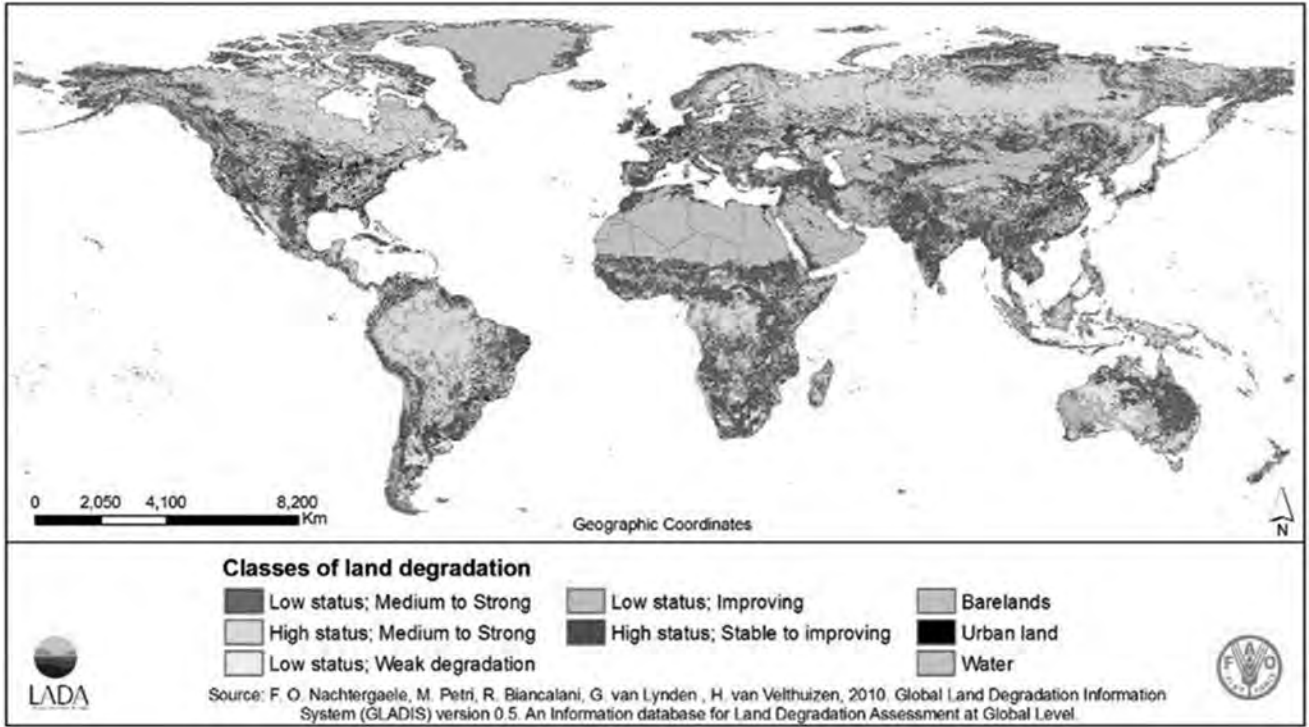


Fig. 4 Classes of land degradation.

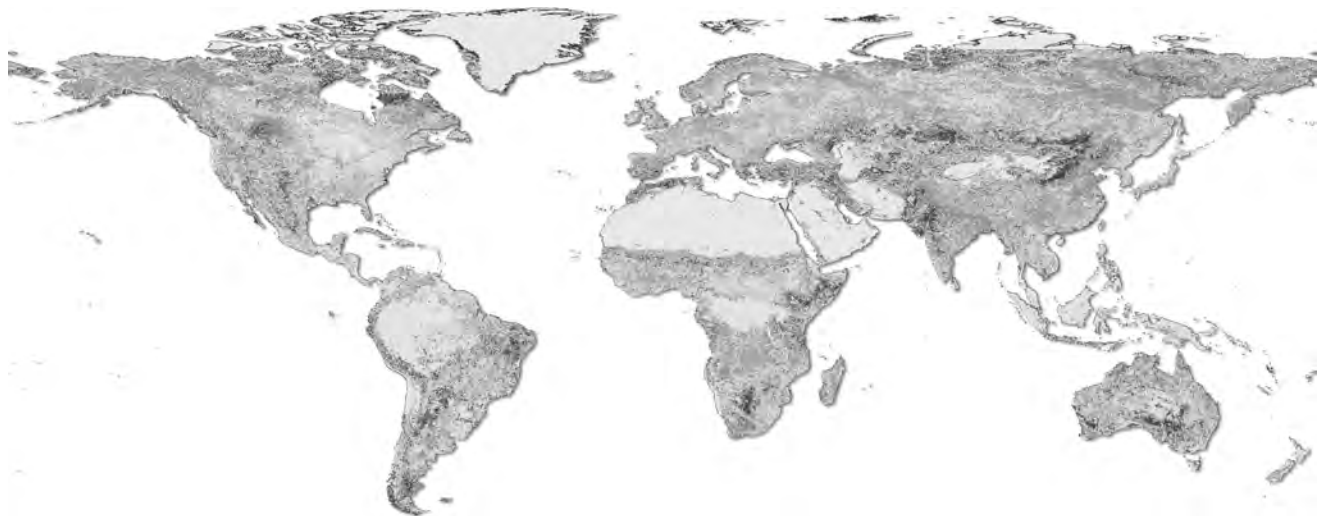


Fig. 5 Consistent decline in ecosystem productivity.

25 years, but it is often difficult to obtain evidence that conditions were better (or worse) in the past than at present and that the presumed degradation was due to present land use changes.^[15]

Mapping desertification is complicated also by the diversity of methodologies used, the selection of indicators included in the assessments, and the intrinsic complexity of the process itself.^[40] A major constraint though is the perceptive aspect of deciding what land degradation really is. This can only be defined in view of the needed or planned optimal land use combination. For example, a land use change (i.e., infrastructure development, housing, and increased economic activity) might provide better income sources for the local population in the short term, but in the long term, ecosystem's stability could be disturbed and perhaps will never return to its original status. Is this degradation and how to map it? LADA indicated indeed that there are trade-offs between socioeconomic gains and environmental losses and vice versa, and this cannot be captured simply in a single index. Related to this

is that these gains and losses are subjective and depend on the interests of different stakeholders which are often completely different. This is true at all levels, be it local or global. The UNCCD is struggling for years to establish a minimum set of indicators to be used in land degradation and desertification assessments even though yet final agreement is far away.^[41]

However, a global assessment can be undertaken, bringing together the best of the available databases supplemented with proxies, and probably the best option could be to develop a map of pressures of land degradation based on conditions of ecosystems and land use systems.^[42]

Mitigating Desertification and Its Consequences

Until 2013, the UNCCD has held 11 conference of parties (COP) as the highest governing body. At COP8 held in Madrid, Spain, in 2008, the UNCCD adopted the 10-year strategy (2008–2018) that aims *to forge a global partnership to reverse and prevent desertification/land degradation and to mitigate the effects of drought in affected areas in order to support poverty reduction and environmental sustainability*. These goals are under scrutiny of the COP and of the civil society organizations that strongly endorse the bottom-up approach.

Despite criticism on the achievements of the UNCCD, data show that there are many positive results worldwide, demonstrating that when local stakeholders are both managers of natural resources and implementers of the right policies, it is possible to make a change.^[43–46] Surely these results are not attributed solely to UNCCD activities; however, implementing SLM practices could make a difference. The most common SLM techniques include soil and water management (i.e., terracing, contour planting, living barriers, low or no tillage, mulches, cover crops including biological nitrogen-fixing legumes, grazing corridors,

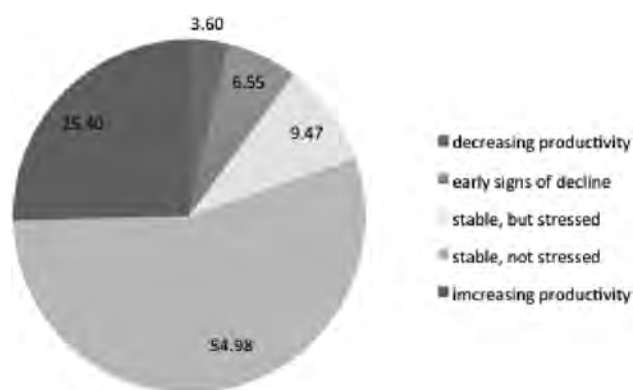


Fig. 6 Global distribution of productivity dynamics.

and water harvesting) and soil fertility management (i.e., manure, compost, biochar, biomass transfer, agroforestry including nitrogen-fixing trees like *Faidherbia albida*, and overall integrated good soil management).

CONCLUSION

The outcomes of assessments of desertification are often debated as they use different methods, measure different variables, and operate at different scales (both temporal and spatial), with some of them focusing on particular bounded systems such as the drylands. This means that the results from different assessments cannot be usefully compared. Despite these issues, in the context of other sustainable development, challenges such as population growth, food security, climate change, and biodiversity loss, nevertheless desertification assessments demonstrate that this is a key issue and that it is likely to worsen unless timely action is taken and appropriate mitigation practices are implemented.

Land degradation is also a natural process. Without soil erosion, neither the fertile Nile floodplain in Egypt, the floodplains of many other valleys around the world, nor most of the Netherlands would exist. The most visible human land degradation impact is probably soil sealing, a “modern” form of desertification present in Europe, in North America, and in the fast-growing economies of China, India, Brazil, South Africa, and Russia and threatening fertile valleys around the world.

Therefore, efforts to map desertification should be accompanied by more emphasis on mapping also sustainable land use practices and by identifying areas that must be protected from unsustainable human activities as the WAD is proposing. The concept of *endangered soil species* should be fostered, especially for the highly productive Mollisols of U.S. soil taxonomy^[47] or the Phaeozems, Kastanozems and Chernozems of the World Reference Base (WRB) for soil resources.^[48] These soils cover only 3% of the global land surface but produce 40% of the global food and 90% of the world's cereals. The United Nations proclaimed 2015 as the International Year of Soils. This is an excellent opportunity to raise awareness on land protection, how to assess it, and to engage in concrete actions to combat desertification and conserve our natural land resources for the future generations.

ACKNOWLEDGMENTS

Authors are grateful to Dr. Freddy Naghtergaele (retired from United Nations Food and Agriculture Organization) for his careful review, comments, and suggestions and to Dr. Robert Jones, retired principal research scientist at Cranfield University, School of Applied Sciences, United Kingdom, for fine-tune editing and adding important information to this entry.

REFERENCES

1. Zdruli, P. Desertification in the Mediterranean region. In *Mediterranean Yearbook 2011*; IEMed, Ed.; European Institute of the Mediterranean: Girona, 2012; 250–255.
2. Lavauden, L. Les forêts du Sahara. *Rev. Eaux et Forêts* **1927**, 7 (65), 329–341.
3. Aubréville, A. *Climats, Forêts et Désertification de l'Afrique Tropicale*; Société d'Éditions Géographiques, Maritimes et Coloniales: Paris, 1949.
4. UNEP. *Sahel Atlas of Changing Landscapes: Tracing Trends and Variations in Vegetation Cover and Soil Condition*; United Nations Environment Programme: Nairobi, 2012.
5. UNEP. *United Nations Convention to Combat Desertification*; United Nations Environmental Programme: Geneva, 1994.
6. Safriel, U. Personal communication, 2013.
7. Arnalds, O. Desertification in Iceland. *Desertification Contr. Bull.* **1997**, 32, 22–24.
8. Martinez-Fernandez, J.; Esteve, M.A. A critical view of the desertification debate in southeastern Spain. *Land Degrad. Dev.* **2005**, 6, 529–539.
9. UNCCD. Text of the Convention including all annexes. Available at <http://www.unccd.int/en/about-the-convention/Pages/Text-overview.aspx> (accessed July 2014).
10. Reynolds, J.; Stafford Smith, M.; Lambin, E.; Turner, B.; Mortimore, M.; Batterbury, S.; Downing, T.; Dowlatabadi, H.; Fernandez, R.; Herrick, J.; Huber-Sannwald, E.; Jiang, H.; Leemans, R.; Lynam, T.; Maestre, F.; Ayarza, M.; Walker, B. Global desertification: Building a science for dryland development. *Science* **2007**, 316 (5826), 847–851.
11. Reynolds, J.; Bastin, G.; Garcia-Barrios, L.; Grainger, A.; Janssen, M.; Jürgens, N.; Scholes, B.; Stafford Smith, M.; Veldkamp, T.; Verstraete, M.; Von, M.; Zdruli, P. Scientific concepts for an integrated analysis of desertification processes. *Land Degrad. Dev.* **2011**, 22, 166–183.
12. Bisaro, A.; Kirk, M.; Zdruli, P.; Zimmermann, W. Global drivers setting desertification research priorities: Insights from a stakeholder consultation forum. *Land Degrad. Dev.* **2013**, 25 (5), 5–16. doi:10.1002/ldr.2220.
13. FAO, UNEP, UNESCO. *A Provisional Methodology for Soil Degradation Assessment*; FAO: Rome, 1979; 84 pp.
14. Millennium Ecosystem Assessment (MEA). *Ecosystems and Human Well-being: Desertification Synthesis*; Island Press: Washington DC, 2005.
15. Zucca, C.; Biancalani, R.; Kapur, S.; Akça, E.; Zdruli, P.; Montanarella, L.; Nachtergaele, F. The role of soil information in land degradation and desertification mapping. A review. In *Soil Security for Ecosystem Management: Mediterranean Ecosystems*; Kapur, S., Ersahin, S., Eds.; The Springer Briefs in Environment, Security, Development and Peace: Heildeberg, 2014; 31–59. doi:10.1007/978-3-319-00699-4_3.
16. Land Degradation Assessment in Drylands. FAO. Available at: <http://www.fao.org/nr/lada> (accessed July 2014).
17. Bai, Z.G.; de Jong, R.; van Lynden, G.W.J. *An Update of GLADA – Global Assessment of Land Degradation and Improvement*. ISRIC Report 2010/08; ISRIC, World Soil Information: Wageningen, 2010.

18. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *World Map of the Status of Human-induced Soil Degradation*, 2nd Ed.; ISRIC: Wageningen, 1991.
19. Kaiser, J. Wounding earth's fragile skin. *Science* **2004**, *304* (5677), 1614–1615. doi:10.1126/science.304.5677.1614.
20. Sonneveld, B.G.J.S.; Dent, D. How good is GLASOD? *J. Environ. Manage.* **2007**, *90*, 184–197. doi:10.1016/j.jenvman.2007.09.008.
21. Van Lynden, G.W.J.; Oldeman, L.R. *Assessment of the Status of Human Induced Soil Degradation in South and South East Asia*; International Soil Reference and Information Centre: Wageningen, 1997; 35 pp. Available at <http://lime.isric.nl/Docs/ASSODEndReport.pdf> (accessed July 2014).
22. Stolbovoy, V.; Fischer, G. *A New Digital Georeferenced Database of Soil Degradation in Russia*. Interim Report IR-97-084/November, 1997; International Institute for Applied Systems Analyses. 14 pp. Available at <http://www.iiasa.ac.at/Publications/Documents/IR-97-084.pdf> (accessed July 2014).
23. Van Lynden, G.W.J. *Soil Degradation in Central and Eastern Europe. The Assessment of the Status of Human-induced Soil Degradation*. Report 2000/5; FAO and ISRIC: Rome, Wageningen, 2000; 15 pp.
24. Lal, R. Soil degradation and global food security: A soil science perspective. In *Land Quality, Agricultural Productivity, and Food Security: Biophysical Processes and Economic Choices at Local, Regional, and Global Levels*; Wiebe, K., Ed.; Edward Elgar Publishing Ltd.: Northampton, MA, 2003; 16–35.
25. Oldeman, L.R. The extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 99–188.
26. FAO. *Land Degradation in South Asia: Its Severity, Causes and Effects Upon People*, World Soil Resources Report, 78; FAO: Rome, 1994.
27. Dregne, H.E. Map of the status of desertification in the hot arid regions. Document. In *A/CONF.74/31. In United Nations Conference on Desertification*, Nairobi, Kenya, 1977.
28. Dregne, H. *Desertification of Arid Lands*; Harwood Academic Publisher: London, 1983.
29. FAO/UNEP. *Provisional Methodology for Assessment and Mapping of Desertification*; FAO: Rome, 1984.
30. UNEP. *World Atlas of Desertification*; Thomas, D.S.G., Middleton, N.J., Eds.; Arnold: London, 1992.
31. Eswaran, H.; Reich, P.; Beinroth, F. Global desertification tension zones. In *Sustaining the Global Farm*; Stott, D.E., Mohtar, R.H., Stenhardt, G.C., Eds.; International Soil Conservation Organization in cooperation with US Department of Agriculture and Purdue University: West Lafayette, 2001; 24–28.
32. Mairota, P.; Thornes, J.B.; Geeson, N., Eds. *Atlas of Mediterranean Environments in Europe: The Desertification Context*; MEDALUS project publication; John Wiley & Sons: Chichester, 1998; 205 pp.
33. EEA. *Mapping Sensitivity to Desertification in Northern Mediterranean*. Final report, version 2; European Topic Centre on Land Use and Spatial Information and European Environment Agency: Copenhagen, 2008.
34. Eswaran, H.; Reich, P.; Vearasilp, T. Perspectives on desertification during the anthropocene. In *Proceedings of the Fourth International Conference on Land Degradation, Cartagena, Spain, Sept 12–17, 2004*; Angel, F., Roque, O., Gregorio, G., Eds.; 2004.
35. Bai, Z.G.; Dent, D.L.; Olsson, L.; Schaepman, M.E. Proxy global assessment of land degradation. *Soil Use Manage.* **2008**, *24*, 223–234.
36. Nachtergaele, F.O.; Petri, M.; Biancalani, R.; van Lynden, G.; van Velthuizen, H.; Bloise, M. *Global Land Degradation Information System (GLADIS), Version 1.0. An Information Database for Land Degradation Assessment at Global Level*, LADA Technical report No. 17; FAO: Rome, 2011.
37. *World Atlas of Desertification*, 3rd Ed.; Joint Research Centre, European Commission. Available at <http://wad.jrc.ec.europa.eu> (accessed July 2014).
38. Geist, H.J.; Lambin, E.F. Dynamic causal patterns of desertification. *Bioscience* **2004**, *54* (9), 817–829.
39. Geist, H.J. *The Causes and Progression of Desertification*, ISBN 0-7546-4323-9; Ashgate, 2005.
40. Zucca, C.; Della Peruta, R.; Salvia, R.; Sommer, S.; Cherlet, M. Towards a world desertification Atlas. Relating and selecting indicators and datasets to represent complex issues. *Ecol. Indic.* **2012**, *15*, 157–170. doi:10.1016/j.ecoind.2011.09.012. ISSN: 1470-160X. 2012.
41. Orr, B.J. *Scientific Review of the UNCCD Provisionally Accepted Set of Impact Indicators to Measure the Implementation of Strategic Objectives 1, 2 and 3*. White Paper Version 1; UNCCD: Bonn. Available at http://www.unccd.int/science/docs/Microsoft%20Word%20-%20White%20paper_Scientific%20review%20set%20of%20indicators_Ver1_31011%E2%80%A6.pdf (accessed July 2014).
42. Nachtergaele, F.O.; Biancalani, R.; Petri, M.; van Lynden, G.; Sonneveld, B.; Zucca, C.; van Lynden, G. Mapping land degradation at global scale: A reflection. In *20th World Congress of Soil Science*, Jeju, Korea, Jul 8–13, IUSS, 2014.
43. Zdruli, P.; Paglai, M.; Kapur, S.; Faz Cano, A. What we know about the saga of land degradation and how to deal with it? In *Land Degradation and Desertification: Assessment Mitigation and Remediation*; Zdruli, P., Paglai, M., Kapur, S., Faz Cano, A., Eds.; Springer Dordrecht Heidelberg: London and New York, 2010; 3–15. doi:10.100/978-90-481-8657-0_1.
44. Thomas, R.; Stewart, N.; Schaaf, T. *Drylands: Sustaining Livelihoods and Conserving Ecosystem Services. A Policy Brief Based on Sustainable Management of Marginal Drylands (SUMAMAD) Project*; The United Nations University: Hamilton, 2014.
45. Schwilch, G.; Bachmann, F.; Valente, S.; Coelho, C.; Moreira, J.; Laouina, A.; Chaker, M.; Aderghal, M.; Santos, P.; Reed, M. A structured multi-stakeholder learning process for Sustainable Land Management. *J. Environ. Manage.* **2012**, *107*, 52–63.
46. Liniger, H.P.; Mekdaschi Studer, R.; Hauert, C.; Gurtner, M. *Sustainable Land Management in Practice: Guidelines for Best Practices for Sub-Saharan Africa*; Terrafrica, World Overview of Conservation Approaches and Technologies (WOCAT) and Food and Agriculture Organization of the United Nations: Rome, 2011.
47. Soil Survey Staff. *Keys to Soil Taxonomy*, 11th Ed.; USDA-Natural Resources Conservation Service: Washington, DC, 2010.
48. IUSS Working Group WRB. *World Reference Base for Soil Resources 2014. International Soil Classification System for Naming and Creating Legends for Soil Maps*. World Soil Resources Report No. 106; FAO: Rome, 2014.

Desertification: Prevention and Restoration

Claudio Zucca

Integrated Water and Land Management (IWLMP) Research Program, International Center for Agricultural Research in the Dry Areas (ICARDA), Amman, Jordan

Susana Bautista

Department of Ecology, University of Alicante, Alicante, Spain

Franco Previtalli

Department of Environmental Science, University of Milan—Bicocca, Milan, Italy

Abstract

Approaches and strategies to prevent and reverse desertification can be broadly grouped as prevention, mitigation, and restoration. Desertification is framed within multiscale, coupled human–environmental dynamics, and thus must be the approaches for prevention and reversal. The development of integrated analytical methods for evaluating and monitoring interventions is crucial. Approaches focus on indicators that relate to ecosystem integrity and services and to human well-being. Lessons learned in the field from implemented projects are discussed.

INTRODUCTION

This entry provides a brief overview of the most significant approaches to prevent and reverse land degradation in drylands, focusing on conceptual frameworks and on methods to monitor and evaluate their impacts. The implementation of mitigation and restoration programs is specifically addressed, with particular reference to the constraints and risk factors.

DESERTIFICATION PREVENTION AND REVERSAL

Desertification is defined here as land degradation in drylands. It results from a long-term unbalance between demand for and supply of ecosystem services, leading to a persistent reduction of the biological and economic productivity and ecological complexity.

The diverse actions that aim at preventing or reversing desertification can be grouped under three major classes: prevention, mitigation, and restoration. Their boundaries are vague and they rather form a continuum of potential actions to be adapted to the particular sites and dynamics through adaptive management approaches (Fig. 1).

Prevention and mitigation actions can be considered as avoidance approaches, either through proactive management efforts where ecological functions are relatively intact or through changes in land use and management, leading to desertification. However, vast areas of drylands are already severely degraded, with reduced plant cover and species

diversity, depleted or eroded soils, and very low potential for spontaneous recovery of ecosystem functions, even when degradation forces are not acting any more. Restoration of these areas is the only option possible to recover ecosystem services that have been lost.

Examples of prevention and mitigation actions include measures to improve water management and agricultural practices. For example, long-term crop rotations with cereals/legumes; more efficient use of fertilizers; improvements in water use efficiency; conservation-minded tillage methods; traditional water-harvesting techniques, water storage; measures that protect soils from erosion, salinization, and other forms of soil degradation; improved crop-livestock integration, combining livestock rearing and cropping; and *in situ* conservation of genetic resources. To create viable livelihood alternatives can also help reduce pressures on drylands. For desertified lands, rehabilitation and restoration approaches often involve the increase and improvement of vegetative cover through, e.g., the establishment of seed banks, reintroduction of selected species, control of invasive species, and reforestation programs. In general, to develop the appropriate engagement between scientific and local environmental knowledge is critically important.

Desertification is driven by a combination of proximate causes and underlying forces, including their interactions and feedbacks, which vary from region to region and change over time.^[1] It is framed within multiscale, coupled human–environmental dynamics,^[2] and thus must be the approaches for desertification prevention and reversal. The relationships between land degradation and its causal

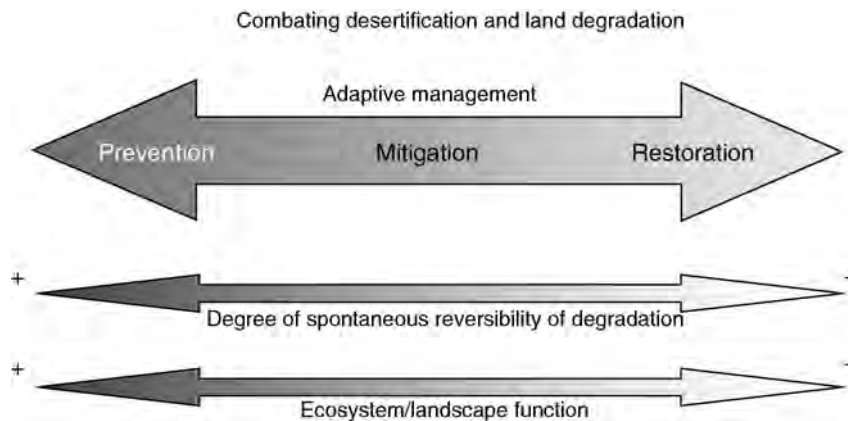


Fig. 1 Continuum of actions to combat desertification and land degradation.

agents are nonlinear and complex. Degradative and aggradative trajectories commonly exhibit hysteresis dynamics, thresholds, and rapid shifts. These relationships are critical in the design of strategies to combat desertification.

Finally, there is growing evidence that land degradation in desertification-prone areas is likely to increase with climate change.^[3] Desertification is linked to biodiversity loss and global climate change through the regulation of water and carbon fluxes, and this perspective requires the development of multifunctional mitigation strategies.

INTEGRATED EVALUATION OF PREVENTION AND RESTORATION ACTIONS

The development of biophysical and socioeconomic integrated analytical methods for evaluating progress and success, along with a framework for knowledge sharing and transfer, is crucial to combating desertification.^[2,4] Furthermore, monitoring and evaluation are needed to demonstrate the benefits of sustainable drylands management, to establish cost-effective thresholds for the various management alternatives, and to identify priority areas where actions could be most effective.

There have been a number of initiatives to develop common and comprehensive methodologies for assessing and evaluating the effectiveness of management and restoration programs.^[5,6] These approaches focus on indicators that relate to ecosystem integrity and services and to human well-being (socioeconomic and cultural variables). Because of the large spatial and temporal variability of dryland systems, it is critical to focus on “slow variables”;^[7] hence the assessment is not confused by short-term variations in land and socioeconomic conditions.

Evaluation frameworks must take account of the cross-scale and social–ecological interactions affecting the response of degraded lands to mitigation/restoration actions.^[8] A multiscale approach is always advisable. Farm- and project-scale assessments focus on local resources and productivity, and private economic valuation perspective (marked prized goods and services), while

landscape- and program-level indicators address environmental benefits and public/social welfare considerations.

IMPLEMENTATION OF MITIGATION AND RESTORATION PROGRAMS

Despite the availability of technological, institutional, and even financial resources, efforts to combat desertification often fail because of a poor implementation. The following cases summarize major constraint or risk factor that can hinder the successful implementation of well-designed projects and highlight the necessary improvements and lessons learned

- Inability to cope with natural crises: especially in projects aimed at increasing plant cover, a poor or delayed wet season or recurrent droughts may cause the loss of the year investment or even of the whole project. Emergency/contingency funds, quick diagnostic and intervention mechanisms, and flexibility in project duration would be needed.
- Lack of longtime planning: very often the longtime dynamics (and its ecological implications) of the newly established systems are not fully considered. This is of particular concern in the case of introducing alien species.
- Passive community participation: community participation should be strongly based on responsible awareness and sharing of project objectives. Sometimes they in fact implement sectoral administration goals, which require farmer action but do not represent the key problem for people.
- Uncertain community commitment: the representativity of the community can be a problem. Who is committing on behalf of whom? The commitment should include economic participation, be linked to final results and based on ownership and responsibility after the project ends, more than related to single tasks.
- Institutional commitment: complex projects, in which the implementation is based on the support of local

administration staff, need a formal and clear institutional commitment. This may take time and very rarely can be established before project approval: projects should have an inception phase to set agreements.

- Schematic approach: sometimes big programs are extensively implemented, often through private sub-contracting, by adopting schematic approaches, which are not able to adapt to specific local land features and people needs, or are too easily funded in areas where they are not necessary or even have negative side impacts. Implementation protocols with multiple technical solutions are needed.
- Lack of assessment: projects are often not adequately monitored and assessed, so lessons are learned too late. Suitable indicators and methods are needed.
- Socioeconomic and demographic dynamics: some projects have experienced lack of man power due to young people migration. Furthermore, the return of people on the land previously “closed for restoration” may cause unsustainable pressure. Mitigation of poverty and migration in the short term should precede/accompany the benefits of medium-term land conservation.
- Market drivers: the dynamics of international and local market prices may completely revert in a few years the achievements produced by years of conservation programmes.
- Contrasting policies: farming incentives may generate negative impacts contrasting mitigation programmes, if not accompanied by adequate guidelines.
- Short time projects: very often project duration imposed by donors (3–5 years) is too short to allow for effective “inception or learning phases,” to deal with the necessity to adapt to changing conditions and needs, to cope with unforeseen events and delays, and most of all to monitor and assess project impacts and sustainability.

CONCLUSION

The brief analysis performed underlined a range of key methodological issues and knowledge gaps, such as threshold conditions and the impact of mitigation and restoration actions on carbon and water cycles and on biodiversity. These should become the basis to develop multifunctional mitigation strategies and formulate inter-linked policies to address desertification, climate change, and biodiversity.

Lessons learned in the field highlighted several issues threatening the success of prevention/reversal actions. The adoption of extensively schematic approaches with poor ex-ante evaluation and scarce integration of sectoral

strategies are among the more critical aspects. Community participation and commitment must receive great attention.^[9] In particular, the development of participatory approaches in the assessment of interventions, including capacity building and information circulation, is critical to improve the adoption of cost-effective, environmentally sound mitigation and restoration measures. For more information, see the entries *Desertification: Reversal* (p. 654), *Land Restoration* (p. 1304), *Restoration Ecology* (p. 1932), and *Restoration: Success and Completion Criteria* (p. 1935).

REFERENCES

1. Geist, H.J.; Lambin, E.F. Dynamic causal patterns of desertification. *BioScience* **2004**, *54* (9), 817–829.
2. Reynolds, J.F.; Stafford Smith, D.M.; Lambin, E.F.; Turner, B.L. II; Mortimore, M.; Batterbury, S.P.J.; Downing, T.E.; Dowlatabadi, H.; Fernández, R.J.; Herrick, J.E.; Huber-Sannwald, E.; Leemans, R.; Lynam, T.; Maestre, F.T.; Ayarza, M.; Walker, B. Global desertification: Building a science for dryland development. *Science* **2007**, *316*, 847–851.
3. IPCC. *Climate Change 2007. Impacts, Adaptation and Vulnerability. 4th Assessment Report. Summary for Policy-makers*; IPCC Secretariat: Geneva, 2007.
4. Zucca, C.; Zdruli, P.; Montanarella, L. Integrated monitoring and transnational coordination to support LDM strategies: Ideas for new joint euro-mediterranean initiatives. In *Managing Natural Resources through Implementation of Sustainable Policies*; Zdruli, P., Trisorio Liuzzi, G., Eds.; IAM Bari: Italy, 2007; 135–150.
5. Bautista, S.; Alloza, J.A.; Vallejo, V.R. Conceptual framework, criteria, and methodology for the evaluation of restoration projects. In *The REACTION Approach. CEAM Foundation*; 2004; 1–7. Available at <http://www.ceam.es/reaction/documents.htm> (accessed September 2007).
6. Adeel, Z.; King, C. Development of an assessment methodology for sustainable development of marginal drylands. In *Proceedings of the Third Project Workshop for Sustainable Management of Marginal Drylands (SUMAMAD)*, Djerba, Tunisia, Dec 11–15, 2004, UNESCO-MAB: Paris, 2005; 13–22.
7. Carpenter, S.R.; Turner, M.G. Hares and tortoises: Interactions of fast and slow variables in ecosystems. *Ecosystems* **2000**, *3*, 495–497.
8. MEA, Millennium Ecosystem Assessment. *Ecosystems and their Services. Ecosystems and Human Well-Being: A Framework for Assessment*; Island Press: Washington, 2003.
9. Zucca, C.; Lubino, M.; Previtali, F.; Enne, G. The Euro-Mediterranean partnership: A participatory demonstration project to fight desertification in Morocco and Tunisia. In *Determining an Income-Product Generating Approach for Soil Conservation Management*; Zdruli, P., Trisorio Liuzzi, G., Eds.; IAM Bari: Italy, 2005; 317–325.

Desertification: Productivity

Kris M. Havstad

Jornada Experimental Range, U.S. Department of Agriculture (USDA), Las Cruces, New Mexico, U.S.A.

Abstract

Despite the increase in desertified lands during the 20th century, the world's populations of cattle, sheep, and goats have collectively increased from 1% to 1.5% annually since World War II. Effects of desertification on primary or secondary productivity of rangelands will be related to resulting species alterations and/or changes on water and nutrient economics.

INTRODUCTION

The United Nations convention to combat desertification (UN-CCD) defined desertification as the degradation of land in arid, semiarid, and dry subhumid areas resulting from various factors, including climatic variations and human activities.^[1] These susceptible drylands cover approximately 40% of the world's land surface.^[2] Desertification is not a new problem, but one that traces back to 2200 B.C. and the degradation of Mesopotamia.^[3] However, the estimated extent of degradation of the world's dry lands at the end of the second millennium A.D. is substantial (Table 1).^[2] Although estimates of the magnitude of degradation vary widely, it is documented that soils of at least 12% of the world's 5169 m ha of drylands are at least moderately desertified.^[2] It should also be noted that these estimates often do not consider lands outside the world's subtropics.^[4] The UN-CCD^[1] estimates that over 250 million people are affected by desertification with about one-sixth of the world's population at risk.^[1] For some continents, the numbers of people at risk are much greater. In Africa, Asia, and South America, between 30% and 40% of the populations live within these susceptible drylands.^[2] Generally, the effects of this land degradation are reductions in desirable plant production, alterations in biomass, lowered carrying capacities for livestock, increased soil erosion, and an overall increase in environmental deterioration.^[5] It is usually assumed that desertification has direct and measurable effects on either primary or secondary production from these lands. In very severely desertified situations, such as Southern Kazakhstan, productivity has been dramatically reduced.^[6] For other regions of the world where desertification is certainly serious but not severe, the relationships between degradation and productivity are less obvious and direct. These relationships need to be understood so that measures other than simply productivity

reductions are employed to accurately assess rates, extent, and threats of degradation to these land resources.

PRIMARY PRODUCTIVITY

Primary productivity is defined as the fixation of carbon dioxide into organic molecules by photosynthetic organisms. Typically, agriculturally and ecologically based estimates of primary productivity focus on some aspect of annual above ground net primary productivity (ANPP). For crop lands, this may be annual grain yields, and reductions in yield of 25% or more have been reported for dry land and irrigated crops.^[7] Often, these losses in productivity are coupled to increased soil erosion because of degradation. For rangelands, primary production is often expressed as biomass production on a dry matter basis per some unit area. Plant processes affected by degradation include reproduction, germination, establishment, survival, and competition, and adverse effects on these processes can result in reductions in ANPP. As a result, there are a number of documented reports from different continents where these shifts have resulted in large declines in primary productivity.^[8] The initial effects of chronic disturbances on these processes may result in structural changes in vegetation such as a shift from herbaceous to woody dominated plant communities, and these structural changes do not always result in reductions in primary productivity.^[9,10] For example, in the northern Chihuahuan Desert of North America, ANPP does not differ among major vegetation types, including shrub land types that are regarded as degraded from the historic desert grasslands.^[11] These types do differ in other vegetation attributes including species richness and in their seasonality of production. Irrespective, seasonal measures of primary production in heterogeneous environments are extremely variable both temporally and spatially.^[12] Thus, it is difficult to consistently and directly equate reductions

Table 1 Soil degradation (from light to severe) by region of the drylands of the world (arid, semiarid, and dry subhumid regions).

Continent	m ha	%
Africa	319.4	25
Asia	370.3	22
Australia and New Zealand	87.5	13
Europe	99.4	33
North and Central America	79.5	11
South America	79.1	15

Source: From Middleton & Thomas.^[2]

in primary productivity in all rangeland environments with desertification.

SECONDARY PRODUCTIVITY

Secondary productivity is the transfer of energy from primary producers to consumers and is often expressed as animal production. For the world's rangelands, secondary production is often measured in terms of livestock productivity. There are specific instances where desertification has resulted in a documented decline in livestock stocking rates, such as in the Western Cape of South Africa.^[13] However, in many instances, losses of stocking rates have not been directly linked to desertification, and loss of secondary productivity can be an ineffective indicator of degradation.^[7] For example, looking more broadly at South Africa during the last half of the 20th century, livestock numbers have been fairly stable.^[14] A more effective indicator of desertification may not be the total number of livestock, but the shift in livestock species used as vegetation types typically changes from herbaceous to more woody species with degradation.^[15]

CONCLUSION

Despite the increase in desertified lands during the 20th century, the world's populations of cattle, sheep, and goats have collectively increased from 1% to 1.5% annually since World War II.^[16,17] This is an increase of 1.8 billion head of livestock over the 1940s, and the increase has been spatially heterogeneous. For example, cattle, sheep, and goat numbers in Africa and Asia have increased over 80% since World War II and comprise 65% of the world's total population. Sixty-five percent of the global increase in sheep numbers has occurred in Asia. These continents have reported extensive desertification during the 20th century. Declines in numbers of livestock may be misleading as an indicator of desertification at regional, national, and continental scales.

Hoffman^[18] has described five phases of vegetation change during the degradation of the Karoo in South

Africa. These were: 1) primary degradation with a loss of cover, primarily palatable perennial grasses and shrubs; 2) denudation with further loss of palatable plants, an increase in unpalatable species, and accelerated rates of erosion; 3) revegetation with an increase in unpalatable species, particularly woody plants; 4) secondary degradation with a relatively stable community of undesirable plants; and 5) desertification with loss of most plant species other than a few hardy shrubs, a soil surface widely exposed to accelerated erosion, and invasion by exotics. Similar vegetation dynamics have been described for other dry land environments in response to overgrazing and prolonged drought. Throughout these phases primary and secondary productivity may or may not be reduced. In general, effects of desertification on primary or secondary productivity of rangelands will be related to resulting species alterations and/or changes on water and nutrient economics.^[19]

REFERENCES

1. Secretariat for the Convention to Combat Desertification. *United Nations Convention to Combat Desertification: An Explanatory Leaflet*. Available at <http://www.unccd.int/convention/text/leaflet.php> (accessed February 2001).
2. Middleton, N.; Thomas, D. *World Atlas of Desertification*, 2nd Ed.; Edward Arnold Publishing Agencies: London, 1997.
3. Burns, W.C. The international convention to combat desertification: Drawing a line in the sand? *Mich. J. Int. Law* **1995**, *16*, 831–882.
4. Arnalds, O. Desertification: An appeal for a broader perspective. In *Rangeland Desertification, Advances in Vegetation Science 19*; Arnalds, O., Archer, S., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 5–15.
5. Dregne, H.E. Desertification: Man's abuse of the land. *J. Soil Water Conserv.* **1978**, *33* (1), 11–15.
6. Shaikhmanov, S.S. The influence of anthropogenic desertification on the productivity of pasture lands. *Probl. Desert Dev.* **1996**, *2*, 81–83.
7. Dregne, H.E.; Chou, N. Global desertification dimensions and costs. In *Degradation and Restoration of Arid Lands*; Dregne, H.E., Ed.; International Center for Arid and Semiarid Land Studies, Texas Tech University: Lubbock, 1992; 249–282.
8. Milchunas, D.G.; Lauenroth, W.K. Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecol. Monogr.* **1993**, *63*, 327–366.
9. Whitford, W.G.; Reynolds, J.F.; Cunningham, G.L. How desertification affects nitrogen limitations of primary production on Chihuahuan desert watersheds. In *Strategies for Classification and Management of Native Vegetation for Food Production in Arid Zones, U.S. For. Serv. Gen. Tech. Rep. RM-150*; Aldon, E.F., Gonzales-Vincente, C.E., Moir, W.H., Eds.; United States Department of Agriculture: Fort Collins, 1987; 143–153.
10. Grover, H.D.; Musick, H.B. Shrub land encroachment in Southern New Mexico, U.S.A.: An analysis of desertification processes in the American Southwest. *Clim. Change* **1990**, *17*, 305–330.

11. Huenneke, L.F. Shrublands and grasslands on the Jornada long-term ecological research site: Desertification and plant community structure in the Northern Chihuahuan Desert. In *Proceedings: Shrub Land Ecosystem Dynamics in a Changing Environment*, U.S. Dept. Agric., Forest Service Gen. Tech. Rep. INT-GTR-338; Barrow, J.R., McCarthur, E.D., Sosebee, R.R., Tausch, R.J., Eds.; United States Department of Agriculture: Ogden, 1996; 48–50.
12. Huenneke, L.F.; Clason, D.; Muldavin, E. Spatial heterogeneity in Chihuahuan desert vegetation: Implications for sampling methods in semi-arid ecosystems. *J. Arid Environ.* **2001**, *47*, 257–270.
13. Dean, W.R.J.; MacDonald, I.A.W. Historical changes in stocking rates of domestic livestock as a measure of semi-arid and arid rangeland degradation in the Cape Province, South Africa. *J. Arid Environ.* **1994**, *26*, 281–298.
14. Milton, S.J.; Dean, R.J. South Africa's arid and semiarid rangelands: Why are they changing and can they be restored? *Environ. Monit. Assess.* **1995**, *37*, 245–264.
15. Babiker, M.M. Nomad settlements and deterioration of rangelands in Sudan. *Environ. Conserv.* **1985**, *12*, 356–358.
16. FAO. *Yearbook of Food and Agricultural Statistics, Production*; Food and Agricultural Organization of the United Nations: Rome, 1948; Vol. 1.
17. FAO. *Production Yearbook*; Food and Agricultural Organization of the United Nations: Rome, 2001; Vol. 55.
18. Hoffman, M.T. Agricultural and ecological perspectives of vegetation dynamics and desertification. In *Rangeland Desertification, Advances in Vegetation Science 19*; Arnalds, O., Archer, S., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 115–130.
19. Illius, A.W.; O'Connor, T.G. When is grazing a major determinant of rangeland condition and productivity? In *People and Rangelands*, Proceedings of the VIth International Rangeland Congress; Eldridge, D., Freudenberger, D., Eds.; VIth International Rangeland Congress, Inc.: Queensland, 1999; Vol. 1, 419–424.

Desertification: Restoration

Alon Tal

Mitrani Department of Desert Ecology, Jacob Blaustein Institute for Desert Research, Ben Gurion University, Beer-Sheva, Israel

Abstract

The impacts of land degradation are limited not solely to soil loss through erosion but also to their subsequent role in reducing the biological and economic productivity of drylands. Reversing desertification, therefore, requires the restoration of the ecosystem's state, function, and services that have suffered or even disappeared in degraded drylands. Because of the soil's reduced water holding capacity and poor productivity, restoration tends to be even slower in desertified areas than in non-dryland regions experiencing land degradation. Two stages of activities are necessary for restoring ecosystems in desertified areas: removing disturbances to the lands and putting forth proactive efforts to restore the flora and fauna to degraded regions. In semiarid and arid regions, special importance is also given to managing watershed regimes and, when possible, reinstating natural flood patterns.

INTRODUCTION: RESTORATION ECOLOGY AND THE DRYLANDS

Activities designed to restore degraded drylands should draw from the fundamental principles of the relatively new field of restoration ecology. Ecological restoration is typically defined as “an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity, and sustainability.”^[1] The original ecosystems that suffer desertification can be woodland, rangeland, or cropland, each with its specific functions and associated strategies/nuances for successful restoration. Yet, typically two types or stages of activities are necessary for restoring ecosystems in desertified areas: removing disturbances to the lands and putting forth proactive efforts to restore flora and fauna to degraded regions.

The climatic and the biotic context that distinguishes desertification from general soil degradation also informs about restoration strategies. The United Nations' legal definition characterizes desertification as “land degradation in arid, semiarid and dry subhumid areas resulting from various factors, including climatic variations and human activities.”^[2] The impacts of land degradation are limited not solely to soil loss through erosion but also to their subsequent role in reducing the biological or economic productivity of drylands. Reversing desertification, therefore, requires the restoration of ecosystem state, function, and services that have suffered or even disappeared in degraded drylands.^[3] As natural communities are primarily composed of plants, restoration ecology for the drylands tends to rely on botanical science.^[4] In semiarid and arid regions, special importance is also given to managing watershed

regimes and, when possible, reinstating natural flood patterns which affect taxa richness in general and micro-invertebrate communities in particular.

Restoration policies must clearly articulate which ecosystem services are to be restored. For instance, farmers may not be inclined to convert eroded farmlands back to original grasslands but may still seek to increase the sustainability of their agricultural practices. Indeed, while restoration efforts aspire to recover or reconstruct the original functioning communities, this typically does not imply a return to the pristine conditions which existed prior to human disturbance. Such an ambitious objective may not be realistic or even desirable.

Degraded drylands may be in such an advanced or enduring state that while it may be possible to assess the original indigenous communities and roughly estimate an appropriate distribution of species, given the nonlinear function associated with biodiversity loss, the damage may be impossible to reverse. In the cases where abiotic conditions have changed dramatically owing to land contamination (e.g., salination), or diversion of natural drainage, restoration efforts will only be able to approximate original conditions. These more modest efforts have come to be called “rehabilitation.” Similarly, others endorse creative efforts to restore ecosystems, even in areas of relatively intense human settlements, calling for a “reconciliation ecology.”^[5] This entry will include both of these approaches under the general title of ecological “restoration.” Yet, restoration, rehabilitation, and reconciliation activities should all seek to return key components and interactions of the original ecosystem and assist in the damaged ecosystem's recovery.

Returning lands to previous levels of biological productivity and diversity is a protracted process. Ecological restoration tends to be even slower in desertified areas than in non-dryland regions experiencing land degradation. This happens because of the soil's reduced water holding capacity and poor productivity or in the cases of soil salination, where water availability is critical for leaching of salts. Accordingly, to attain reasonable progress in highly degraded areas, more aggressive interventions to expedite the increase in ecosystem services and soil productivity are generally necessary. The temporal concerns that drive restoration strategies need to be informed by spatial consideration. A basic principle of restoration ecology recognizes the close association between the scale of an ecosystem's disturbance and the amount of time required for its restoration. When desertification takes place in a relatively small area, restoration can be attained far more quickly than in larger areas.^[6]

REMOVAL OF DISTURBANCES

If the causes of the desertification process are not addressed, then proactive restoration efforts that focus on the symptoms will be unsuccessful. The reversal of desertification, therefore, begins with the removal of the direct drivers that were the proximate cause of the desertification and ecosystem functioning loss in the first place. In agricultural areas, this can mean the cessation of unsustainable cultivation practices—from plowing and planting on excessively steep gradients to excessive pesticide application—alongside the adoption of traditional soil conservation practices, such as terracing or no-till plowing that can prevent erosion processes. If soil degradation is a result of deforestation and loss of land cover, then rules must be enacted to enjoin wood clearing. In drylands, unsustainable water management decisions can be among the primary drivers of desertification, and restoration strategies must ensure that they include prescriptive and enforcement efforts throughout the watershed.

Ecosystem restoration often also requires the consideration of relevant zoological factors, e.g., removal of problematic exotic species/herbivores. For instance, in a semiarid region of Chile, removal of herbivores (in particular European rabbits) increased sapling survival by as much as 60% in a controlled restoration experiment.^[7] In degraded areas which are tourist destinations, restoration programs may need to enact limitations on off-road vehicle or human access. Moreover, a sustainable restoration strategy must also address the indirect human forces that influence the biophysical processes.^[8] This can involve the introduction of social policies ranging from the reduction of population pressures or enhancing the status of women (who are frequently excluded from soil conservation decisions, notwithstanding their key role as farmers) to the modifying market incentives that encourage poor land

stewardship. Given the prevalence of pastoral activities in the drylands, controlling overgrazing is among the most common measures that must be taken to create the potential for the restoration of desertified ecosystems.

Sustainable Grazing

Historically, the seasonal migrations of pastoralists from China to the Sahel indigenous and their nomadic herds did not necessarily exceed ecosystem carrying capacity and in many cases may actually have contributed to long-term sustainability and biodiversity.^[9] Yet the steady shrinking of available rangelands, population increase among pastoral communities, and their associated activities along with the steady improvement of veterinary practices in the modern period have led to increasing pressure on rangelands worldwide. In the drylands, where soils are generally nutrient poor and water constrains rangeland resilience, overgrazing has led to excessive soil compaction, loss of soil cover, soil erosion, and in many cases the conversion of grasslands to shrublands—leading to increased runoff, soil heterogeneity, and state change. In areas where grazing has led to the decimation of land cover, stocking limits that prescribe maximum animal densities are relatively easy regulations to enforce and constitute a normative prerequisite to ecosystem restoration efforts.^[10]

When managed appropriately, grazing produces net ecological benefits. Removal of biomass reduces fodder available for bushfires. Livestock can facilitate the spreading of seeds, with manure providing limited available fertilization on nutrient poor lands. When optimized, grazing can be a key element in restoring ecosystems, assisting the growth of grasses, breaking detrimental soil crusts, and enhancing biodiversity, by controlling plants that would otherwise become dominant. For example, in Israel, afforestation has proven to be an effective mechanism for reducing dryland soil erosion. In these woodlands, grazing by indigenous bedouin has proven to be so important in local strategies to prevent forest fires that in some areas grazing fees on public lands have been reversed, and foresters are paying herders to bring their sheep in a controlled grazing regime into the forests to clear out grasses and shrubs. (Ironically, in areas with greater rainfall and tree densities, initiating fires has become a key land management tool.)

Accordingly, ecological restoration efforts should characterize the level of land resilience which varies according to soil type, rainfall patterns, etc. Subsequently, they should permit grazing that does not exceed the carrying capacity. The impacts of reduced grazing are clearly manifested along the Israeli–Egyptian border. When Egypt reclaimed the Sinai Peninsula, within a few years, satellite images identified a geopolitical border. On the Israeli side, a biogenic crust remained intact, reflecting the continuation of grazing regulations, as opposed to the dramatic loss of land cover in Egypt where grazing restrictions were relaxed.^[11]

Seemingly degraded lands will often return to their pre-grazing state following the cessation or reduction of grazing, with natural seed banks containing sufficiently viable seeds to reestablish high-diverse, indigenous plant cover. In a retrospective study of five sites where degraded sites were fenced off from grazing animals in Northern China, 15 years after the intervention, the vegetative species restoration in the protected sites was deemed to be more successful and effective than comparable areas where aerial seeding alone was conducted.^[12] The ultimate test of success of such interventions, however, can only be measured by the ability of restored sites to once again withstand a controlled grazing presence.

Controlled grazing need not imply fencing or the truncation of traditional grazing routes. Rangeland carrying capacity can be maintained by migration and rotation. Indeed, “reaggregation policies” can offer a more sustainable alternative; seasonal movement by pastoral communities across private and communally owned lands is facilitated, as nomadic pastoralists continue to rotate their flocks.^[13] Calculating precise carrying capacity for grazing herds tends to be site specific and requires monitoring to assess the validity of initial estimates.

RESTORING VEGETATION, WILDLIFE, AND DRAINAGE PATTERNS

While research in ecosystem succession suggests that following the removal of disturbances, dryland ecosystems will recover naturally, ongoing soil loss may be sufficient or so far advanced as to justify proactive remedial measures. Desertification is frequently a direct result of soil exposure following the loss of vegetative cover. Hence, restoring trees and grasses cover is generally a key component of ecological restoration. Among the range of practices that have been successfully employed are the replenishment of organic materials and nutrients in the soil, aerial seeding, and artificial restocking of natural seed banks, reforestation, and planting of grasslands, seed inoculation with organisms that support plant growth, and species reintroduction along with soil conservation activities such as gulley controls or road replanning.

Restoration of Vegetation

Vegetation and reforestation programs depend on local climatic and soil conditions as well as the original, indigenous vegetation types. The introduction of plant species that are known to have existed on the degraded land prior to human disturbances is a typical strategy, especially when agriculture or human settlement has erased original flora. An underlying assumption behind such an approach is that as abiotic conditions remain unchanged, the ultimate mix will approximately come to resemble the original plant combinations. The introduction of shrubs and grasses is thought

to be more cost-effective as a means for reducing erosion than are trees in dryland restoration efforts although dryland forestry has proven to be successful in a variety of regions.

In general, restoration strategies that run counter to botanical succession patterns will tend to be less successful than those that emulate them. For instance, the “Great Green Wall,” initiated in 1978 in Northern China, may be the most ambitious land restoration project undertaken in a desertified region. The results have been disappointing, not withstanding considerable government investment. Local researchers suggest that the prevailing 350-mm rainfall and high evaporation levels are unable to support the type of woodlands introduced.^[12] Afforestation efforts in Israel have established relatively dense canopies as well as new habitats and recreational venues in semiarid climates with as little as 250 mm of annual rainfall. While in Mediterranean regions of the country, 2nd generation forests have emerged; returning original stands of oak-pistacia woodlands, the low rate of natural regeneration in semiarid conifer forests raises questions about the long-term sustainability.

Because moisture is generally the greatest controlling factor in dryland ecosystem restoration, attention should be given to the availability of water and homogeneity of runoff. In areas where there has been desertification of productive grasslands, the resulting heterogeneity of runoff can produce a loss of soil and nutrients that can ultimately lead to a change of state, with new invasive shrub communities emerging.^[14] Returning grasslands to such semiarid and dry subhumid lands through proactive seeding will improve rainfall infiltration and begin to reverse runoff and erosion intensity.^[15]

In areas of low precipitation, vegetation in the drylands is often naturally sparse, with vegetative patches surrounded by exposed soils with biogenic crusts. Vegetation is present as steppes that often take on a spotted or banded structural formation. Indeed, Chinese land managers generally see 30% vegetative cover of soils as sufficient to prevent wind erosion.^[16] Accordingly, the rejuvenation of grasslands and woodlands need not aspire to reach full coverage. Studies in New Mexico showed that net primary productivity in shrublands was comparable to the native grasslands, even as the economic utility of grasslands is considered to be greater, given their higher levels of support for grazing.^[14] Drylands probably select for shrub species that are better able to exploit the moisture available in the deeper soil layers after intermittent, intensive rainfall events as well as for crust species that enable shrubs to harvest the deeper waters.

The ecological interactions in the patchy, so-called “resource islands” are generally characterized by facilitation between plant species, a dynamic that can and should be utilized in restoration activities in the drylands. Existing vegetation can be utilized to provide microsites for the introduction of desirable species. Seedling survival and

growth in drylands are generally far higher when planted under the canopy of shrubs or trees than in open areas. For example, in Spain, shrubs were more successfully reestablished near existing tussocks than in open semiarid areas that served as controls;^[15] in Kenya, planting aloe improved the effectiveness of grass reseeding on degraded rangelands.^[17] In a restored Israeli degraded site, artificially created pits served as sinks that reduced leakages of water, soil, and nutrients. The result was a significant increase in total biomass production as well as annual plant species diversity.^[18]

The patchiness of vegetation in drylands makes natural seed limitation a key obstacle in restoration efforts with the result often being incomplete species assemblies. Direct seeding via the air dispersal or less costly conventional methods offer the most common method for overcoming the loss of deep-rooted perennial plants and accelerating the vegetative change. As ecological restoration typically seeks to reinforce indigenous species, the collection of seeds from native species constitutes a considerable logistical and economic challenge. It is important, therefore, to create conditions that ensure the maximum success of seeding efforts. Survival rates for seedlings are usually increased by moderate soil disturbance (e.g., harrowing or raking) prior to the spreading of seeds. Ensuring the availability of nitrogen, phosphorus, and potassium is critical, as the runoff produced by degraded shrublands often leads to a poor key elements land. Microbial symbionts are also an important supplement in seeding efforts.

The Role of Soil Microbes

Loss of vegetative cover is often preceded by damage to basic soil properties such as nutrient concentrations, soil structure, organic matter content, and microbial activities. Microbial symbionts, the microorganisms that inhabit upper layers of the soil, play a key role in governing nutrient cycles in plants, maintaining symbiotic relationships with root systems. While agriculture and forestry have focused on these interactions for sometime, only lately has the potential role of mutualistic soil organisms in reestablishing native plant communities in desertified regions been demonstrated.^[19]

Of particular importance are the mycorrhizal fungi that strengthen the ability of plants to cope with stress and rhizobia, which facilitate the fixing of nitrogen and allow plants to grow even when available nitrogen in soils is low. In essence, their presence extends surface area of roots several hundred times, allowing plants to overcome heat and drought as they more efficaciously absorb water and nutrients. Because these organisms are dependent on the presence of native host plants, especially legumes, frequently a depletion of microbial symbionts will accompany loss of vegetative cover. In addition, arbuscular mycorrhizal fungi have been demonstrated to increase phosphorous uptake.

When ecological restoration involves the planting of legumes, the inoculation of seeds with rhizobia has proven to be effective. This holds true for trees as well. In an Australian study, inoculated native acacia seedlings grew 10–58% faster than uninoculated controls.^[20] Research in Mediterranean ecosystems demonstrated that plants introduced after inoculation with mycorrhizal fungi and rhizobial ecosymbionts showed significant advantages in five-year survival rates and size. Furthermore, measurable improvements found in the soil's physiochemical properties included higher nitrogen concentrations, organic matter, and soil aggregation after inoculation.^[19]

Restoration of Animals to Desertified Ecosystems

Loss of wildlife and changes in fauna formations in desertified systems may be affected by causes of biodiversity loss that are not directly associated with land degradation (hunting, habitat loss, etc.). Concern has also been expressed for the impact of climate change on population dynamics in drylands due to the anticipated increase in interannual variance in arid environments.^[21] Hence, restoration programs should consider general conservation measures that can be applied in conjunction with proactive animal repatriation on desertified lands efforts. When vegetation returns to its natural state, assuming that there have not been local extinctions, wildlife should eventually return to its original diversity and levels. Organisms which have evolved to accommodate the extreme climatic conditions of the drylands may be well equipped for relatively harsh physiological and behavioral adaption imposed by moderate changes in soil. Yet, the reduction in vegetative productivity will ultimately be felt along the food chain, and its rejuvenation holds the key to successful repatriation of native wildlife. For instance, after the restoration of vegetation to a hyperarid, dry salt marsh north of the Gulf of Aqaba (which had been converted to a garbage tip), bird populations returned during their migrations from Africa to Europe.^[22]

The reintroductions of animal species that have disappeared have been successful in such dryland countries as the United States, Spain, Oman, and Israel. These are generally popular with the public although they are typically expensive,^[23] and failure is more common than success. The International Union for Conservation of Nature (IUCN) has formed a "Reintroduction Specialist Group" that has, for some time, drafted animal-specific release protocols, which take into consideration the climatic conditions in drylands. Given the economic role of grazing in drylands, finding the optimal balance between domestic and wild animals in restored areas may not be self-evident and should be clarified when establishing restoration objectives. Frequently, restoration strategies will have to weigh competing priorities (e.g., fencing versus preservation of migration routes). Moreover, restoring top predators populations is often the best practice but will cause

tension between conservationists and domestic herd owners. One should also take into consideration that restoring desertified land can change the local species communities.

REINSTATING DRAINAGE AND FLOODING PATTERNS

As limited water resources are taken for irrigation and other human uses, the natural flooding patterns in drylands are often disturbed, affecting the natural ecosystems. As ground water tables drop and runoff no longer reaches open spaces, vegetative response is often dramatic. In areas where water has been tapped for human endeavors, meaningful ecological restoration cannot take place without reinstating some meaningful portion of runoff and approximating original drainage patterns. Especially in semiarid and arid zones, periodic floods also allow for colonization by micro-invertebrates, insects, and autotrophic communities, which in turn attract avian populations. Of particular importance to ecological restoration efforts in drylands is the shortening of the duration of protracted dry periods by ensuring reasonable frequency of floods that preserve micro-invertebrate and macrophyte populations in their resistant stages. For example, using emergence of micro-invertebrates from resting eggs as an indicator, an Australian study, along the arid floodplains, showed that increased drying significantly reduced taxa richness.^[24]

Water extraction is not the only way that hydrology can affect ecosystems. Often, land use activities in the upper reaches of a watershed are the primary cause of soil loss. For instance, the narrowing of gullies or poorly designed roads can speed water flow and exacerbate the erosive effects of storms. Identifying causes as opposed to mere symptoms implies that restoration activities in the lower sections of such a watershed will be undermined by new, erosive flow patterns unless interventions are made to reduce flow velocity. Increasing the meandering pathway of ephemeral streams or building terraces and dikes has attenuated floods and helped approximate original, basin-wide flows.

CONCLUSION

Severe desertification reflects a change in state, where key thresholds have been crossed.^[16] It is invariably easier to remedy a situation prior to a shift into a new, degraded state, then to restore a decimated ecosystem. Ongoing monitoring of soil and ecosystem conditions to identify gradual degradation is, of course, advisable. Notwithstanding lack of international progress in combating desertification, land degradation trends need not be destiny. International experience demonstrates that the restoration of desertified ecosystems is possible. Restoration efforts require patience and prolonged attention from

government the decision-makers, land managers, rangers, and general public. After removing the proximate and indirect drivers of the land degradation, proactive efforts can be cost-effective. In addition to ensuring the general application of soil conservation, activities in a watershed should include the thoughtful selection of indigenous plant and wildlife and their introduction with appropriate technologies (e.g., reestablishment of microbial symbionts). When such measures are supported by community involvement and appropriate policies, it is possible to reestablish healthy and productive ecosystems in desertified lands.

ACKNOWLEDGMENT

The author is grateful to the helpful comments and suggestions of Mr. Yizhak Moshe, Professor Uriel Safriel, Dr. Uri Shanas, and Professor Moshe Shachak in the preparation of this entry.

REFERENCES

1. SER. *The SER International Primer on Ecological Restoration*; Society for Ecological Restoration International, Science & Policy Working Group: Tucson, Arizona, 2004. Available at www.ser.org.
2. United Nations Convention to Combat Desertification in Those Countries Experiencing Serious Drought and/or Desertification, Particularly in Africa; June 17, 1994. Article 9, 33 I.L.M. 1328 Article 1.
3. Adeel, Z.; Safriel, U.; Niemeijer, D.; White, R. *Millennium Ecosystem Assessment. Ecosystems and Human Well-Being: Desertification Synthesis*; World Resources Institute: Washington, 2005; 1–3.
4. Young, T.P.; Petersen, D.A.; Clary, J.J. The ecology of restoration: Historic links, emerging issues and unexplored realms. *Ecol. Lett.* **2005**, *8*, 662–673.
5. Rosenzweig, M. *Win Win Ecology, How the Earth's Species Can Survive in the Midst of Human Enterprise*; Oxford University Press: Oxford, 2003.
6. Krebs, C.J. *Ecology, the Experimental Analysis of Distribution and Abundance*; Benjamin Cummings: San Francisco, 2001; 480–481.
7. Gutierrez, J.R.; Holmgren, M.; Manrique, R.; Squeo, F.A. Reduced herbivore pressure under rainy ENSO conditions could facilitate dryland reforestation. *J. Arid Environ.* **2007**, *68*, 322–330.
8. Reynolds, J.F.; Stafford Smith, D.M.; Lambin, E.F.; Turner, B.L.; Mortimore, M.; Batterbury, S.P.J.; Downing, T.E.; Dowlatabadi, H.; Fernández, R.J.; Herrick, J.E.; Huber-Sannwald, E.; Jiang, H.; Leemans, R.; Lynam, T.; Maestre, F.T.; Ayarza, M.; Walker, B. Global desertification: Building a science for dryland development. *Science* **2007**, *316* (5826), 847–851.
9. Perevolotsky, A.; Seligman, N. Role of grazing in Mediterranean range land ecosystems. *Bioscience* **1998**, *48* (12), 1007–1017.

10. Tal, A.; Cohen, J. Adding “Top Down” to “Bottom Up”: A new role for environmental legislation in combating desertification. *Harvard J. Environ. Law* **2007**, *31* (1), 163–219.
11. Karnieli, A.; Tsoar, H. Spectral reflectance of biogenic crust developed on desert dune sand along the Israel-Egypt border. *Int. J. Remote Sens.* **1995**, *16* (2), 369–374.
12. Liu, M.; Jiang, G.; Yu, S.; Li, Y.; Li, G. The role of soil seed banks in natural restoration of the Degraded Hunshandak Sandlands, Northern China. *Restor. Ecol.* **2009**, *17* (1), 127–136.
13. Burnsilver, S.; Mwangi, E. *Beyond Group Ranch Subdivision: Collective Action for Livestock Mobility, Ecological Viability and Livelihood*; CGIAR: Washington, 2007.
14. Schlesinger, W.H.; Reynolds, J.F.; Cunningham, G.L.; Huenneke, L.F.; Jarrell, W.M.; Virginia, R.A.; Whitford, W.G. Biological feedbacks in global desertification. *Science* **1990**, *247*, 1043–1048.
15. Maestre, F.T.; Bautista, S.; Corina, J.; Belot, J. Potential for using facilitation by grasses to establish shrubs on a semiarid degraded steppe. *Ecol. Appl.* **2001**, *11* (16), 1641–1655.
16. Shengyue, F.; Lihua, Z. Desertification control in China: Possible solutions. *J. Hum. Environ.* **2001**, *6* (30), 384–385.
17. King, E.G.; Stanton, M.L. Facilitative effects of Aloe Shrubs on grass establishment, growth, and reproduction in degraded Kenyan Rangelands: Implications for restoration. *Restor. Ecol.* **2008**, *16* (3), 464–474.
18. Shachak, M.; Sachs, M.; Moshe, I. Ecosystem management of desertified shrublands in Israel. *Ecosystems* **1998**, *1*, 475–483.
19. Requena, N.; Perez-Solis, E.; Azcon-Auilar, C.; Jeffries, P.; Barea, J.G. Management of indigenous plant–microbe symbioses aids restoration of desertified ecosystems. *Appl. Environ. Microbiol.* **2001**, *67* (2), 495–498.
20. Thrall, P.; Millsom, D.A.; Jeavons, A.C.; Waayers, M.; Harvey, G.R.; Bagnall, D.J.; Brockwell, J. Seed inoculation with effective root nodule bacteria enhances revegetation success. *J. Appl. Ecol.* **2005**, *42*, 740–751.
21. Saltz, D.; Rubenstein, D.I.; Whittie, G.C. The impact of increased environmental stochasticity due to climate change on the dynamics of Asiatic wild ass. *Conserv. Biol.* **2006**, *20* (5), 1402–1409.
22. Yosef, R.; Chernetsov, N. Stopover ecology of migratory Sedge Warblers (*Acrocephalus schoenobaenus*) at Eilat, Israel. *Ostrich* **2004**, *75* (1–2), 52–56.
23. Beck, B.B.; Rapaport, L.G.; Stanley-Price, M.R.; Wilson, A.C. Reintroduction of captive born animals. In *Creative Conservation: Interactive Management of Wild and Captive Animals*; Olney, P., Mace, G., Feistner, A., Eds.; Chapman and Hall: London, 1994; 265–286.
24. Jenkins, K.M.; Boulton, A.J. Detecting impacts and setting restoration targets in arid-zone rivers: Aquatic micro-invertebrate responses to reduced floodplain inundation. *J. Appl. Ecol.* **2007**, *44*, 823–832.

Desertification: Reversal

David Tongway

Ecosystem Sciences, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia

John Ludwig

Ecosystem Sciences, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Winnellie, Northern Territory, Australia

Abstract

This entry describes an ecologically-based set of data-gathering processes by which desertification can be assessed and combated using a framework that characterizes the biogeochemical status of the system. This can be simply expressed as “assessing the regulation of vital resources in space and time.” The procedure is able to deliver biogeochemical solutions appropriate to the problem, rather than a set of recipes with varying degrees of potential success. The information-gathering procedure can be adapted to a wide range of bioclimatic situations, because it deals with the basic processes controlling the availability of vital resources to biota.

INTRODUCTION

The term “desertification” was coined to graphically represent the state of the Sahelian lands in the 1970s, when major drought accompanied by major increases in the human population served to cause the desert margins to apparently move into formerly more productive land.^[1,2] The image of an encroaching desert is powerful and evocative and resulted in major international efforts to understand and deal with the problem. Since that time, the notion of desertification has been reworked in light of additional information to the extent that the desert is no longer seen as inexorably increasing in size nor restricted to the Sahel.^[3–5] Most rangeland areas in the world have suffered some sort of degradation, and reviews^[6] have shown the process to be not at all restricted to hot deserts or areas of high population density. This is not to deny, however, the major effects on the human populations using these lands, and no doubt, much human hardship has been endured.

biogeochemical systems, with logical sequences of processes and feedback loops.

Vital resources such as water, nutrients, and topsoil are distributed, utilized, and cycled in space and time by processes such as runoff/run-on, erosion/deposition, and decomposition.^[8,10,11] Landscapes are said to be “functional” or non-desertified, if resources are retained within the system and utilization and cycling processes are efficient. “Dysfunctional” or desertified landscapes are characterized by the depletion of the stock of some vital resources and the continued flow of these resources out of the system. This mindset emphasizes the system attributes of processes acting in space, over time, in relation to applied stress and disturbance, rather than just the change in the lists of species or yields of commodities. The role of vegetation as a significant regulator of energy and resources is integral to this approach.^[12] Desertification should be viewed as a continuum ranging from slight to severe, rather than as a simple step function (Fig. 1).

DESERTIFICATION REDEFINED

This entry focuses on the biophysical aspects of desertification. Traditionally, easy-to-measure plant structural attributes, such as species composition and productivity, were the means by which desertification was assessed. These methods served to show the effect of desertification, but did not provide a predictive understanding of how to combat it. However, advances in landscape and restoration ecology^[7–9] have proposed generic approaches to study the basic nature and reversibility of desertification. Principally, this involves treating the affected landscapes as

ASSESSING THE DEGREE OF DESERTIFICATION

If field sites were characterized according to “resource regulation” capacity, not only could the degree of desertification be assessed, but also the critical pathway of resource loss might be identified. Tongway and Hindley^[13,14] have designed and implemented monitoring programs to quickly provide information about biophysical processes and edaphic properties related to plant habitat favorability at both landscape and plot scales. The data gathered need an interpretational framework to facilitate generic application. Graetz and Ludwig^[15] proposed that system response to

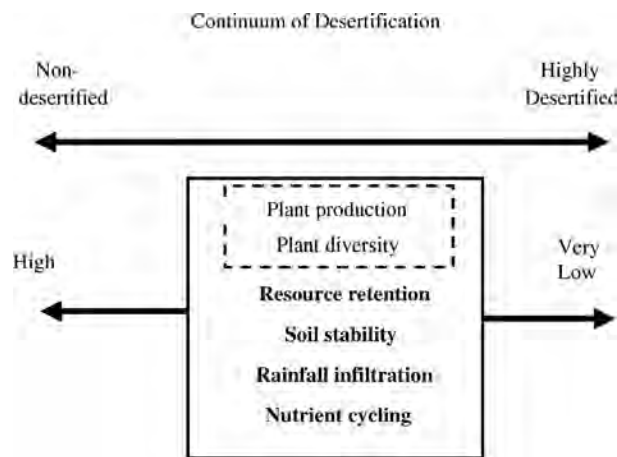


Fig. 1 Desertification as a continuum. The four new biophysical parameters (bold) are added to the two existing desertification descriptors (dotted box) to locate any given landscape on the continuum.

desertification be represented by a four-parameter sigmoidal curve. Their field data analysis suggested that this shape was appropriate to relate landscape function to the degree of stress and disturbance, causing the dysfunction. The curve form acknowledges upper and lower asymptotic plateaus at the non-desertified ends and highly desertified ends of the spectrum, respectively, and a transition between those plateaus. This approach permits questions about whether a system was “fragile”—easily made, dysfunctional, with low restoration potential—or “robust”—able to withstand stress and disturbance with only small biogeochemical attenuation (Fig. 2) to be asked. Importantly, this curve type enables critical thresholds to be detected and quantified using field indicators.^[16]

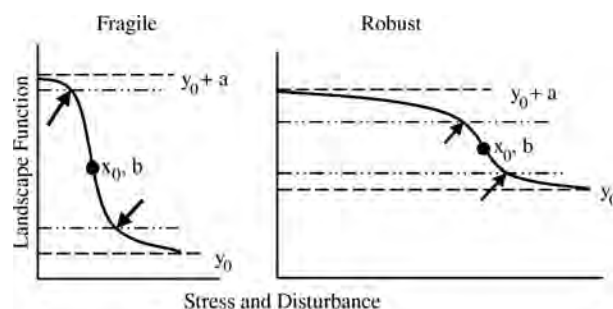


Fig. 2 Examples of response curves for fragile and robust landscapes. The initial response of landscape function to stress and/or disturbance is markedly different. The fragile landscape deteriorates with low applied stress and has a much lower base value (y_0) than the more robust landscape. Four-parameter sigmoid curves of the form $y = (y_0 + a)/(1 + e^{-(x-x_0)/b})$ provide four practical values reflecting the nature of the landscape. Critical thresholds (arrows) for each index of desertification can be determined by curve analysis.

PROCEDURES TO REVERSE DESERTIFICATION

Rehabilitation of desertified landscapes, under the biogeochemical system mindset, requires that processes that accumulate resources be reintroduced, thus providing a rational procedure in restoration of desertified landscapes. The approach explicitly seeks to retain resources by ecological processes.^[9,17,18] Once the analysis of resource regulation system has been completed and the most affected process have been identified, remedial efforts can begin; e.g., Rhodes and Ringrose-Voase^[19] deduced that ponding water on swelling clay soils would eventually result in open, friable soils with high infiltration and water store through swelling and shrinking processes. Tongway and Ludwig^[17] used piles of branches arranged on the contour line to trap water, sediment, and organic matter to effect substantial improvements in a range of soil properties, permitting perennial grasses to self-establish. Both these procedures were effective because each had identified the critical biophysical process to be ameliorated in their respective landscapes. Attempting to revegetate desertified areas by simply reseeding without understanding both the existing and the required edaphic properties needed for the desired vegetation mix frequently results in unexplained failure. In some instances, where the system function is close to the lower asymptote (Fig. 2), simple treatments such as exclusion from grazing will be too slow or ineffective, and active intervention may be needed to improve one or more functional processes.

MONITORING REHABILITATION

It will be important to monitor the progress of processes set in train by the rehabilitation activities. Essentially, the degradation curves such as those in Fig. 2 need to be driven “in reverse.” The procedure proposed by Tongway and Hindley^[20,21] can be used to follow the trajectory of the system. The procedures use simple, easily observed indicators of processes of resource regulation. Biota establishment and development will also need to be monitored and interpreted in terms of the response of the whole system to restoration. It is important that the monitoring process should provide accurate information quickly and at low cost. Remotely sensed data related to landscape function is a cost-effective procedure.^[22] Also, the effect of rare stochastic events such as fire may need to be assessed to see whether the resultant stress and disturbance have set the system back beyond a critical threshold or not.

CONCLUSION

We have described an ecologically based set of data-gathering processes by which desertification can be assessed and combated using a framework that characterizes the biogeochemical status of the system. This can

be simply expressed as “assessing the regulation of vital resources in space and time.” The procedure is able to deliver biogeochemical solutions appropriate to the problem, rather than a set of recipes with varying degrees of potential success. The information-gathering procedure can be adapted to a wide range of bioclimatic situations, because it deals with the basic processes controlling the availability of vital resources to biota.

REFERENCES

1. UNEP. *United Nations Conference on Desertification: An Overview*; United Nations Environment Program: Nairobi, 1977.
2. UNEP. *General Assessment of Progress in the Implementation of the Plan of Action to Combat Desertification 1978–84*; United Nations Environment Program: Nairobi, 1984.
3. Arnalds, O. Desertification: An appeal for a broader perspective. In *Rangeland Desertification*; Arnalds, O., Archer, S., Eds.; Kluwer Academic: Dordrecht, 2000; 5–15.
4. Chen, Z.; Li, X. Degradation of grassland ecosystems in China and its biological processes. In *People and Rangelands: Building the Future*, Proceedings of the VI International Rangeland Congress, Townsville, Australia, Jul 19–23, 1999; Eldridge, D., Freudenberger, D., Eds.; VI International Rangeland Congress: Townsville, 1999; 105–107.
5. Tongway, D.; Whitford, W. How does desertification affect soil processes? In *People and Rangelands: Building the Future*, Proceedings of the VI International Rangeland Congress, Townsville, Australia, Jul 19–23, 1999; Eldridge, D., Freudenberger, D., Eds.; VI International Rangeland Congress: Townsville, 1999; 89–142.
6. Archer, S.; Stokes, C. Stress, disturbance and change in Rangeland ecosystems. In *Rangeland Desertification*; Arnalds, O., Archer, S., Eds.; Kluwer Academic: Dordrecht, 2000; 17–38.
7. Breedlow, O.A.; Voris, P.V.; Rogers, L.E. Theoretical perspective on ecosystem disturbance and recovery. In *Shrub-Steppe: Balance and Change in a Semi-Arid Terrestrial Ecosystem*; Rickard, W.H., Rogers, L.E., Vaughn, B.E., Liebetrau, S.F., Eds.; Elsevier: New York, 1988; 257–269.
8. Ludwig, J.A.; Tongway, D.J. A landscape approach to rangeland ecology. In *Landscape Ecology Function and Management: Principles from Australia's Rangelands*; Ludwig, J., Tongway, D., Freudenberger, D., Noble, J., Hodgkinson, K., Eds.; CSIRO: Melbourne, 1997; 1–12.
9. Whisenant, S.G. *Repairing Damaged Wildlands: A Process-Oriented, Landscape-Scale Approach*; Cambridge University Press: Cambridge, 1999; 312 pp.
10. Jorgenson, S.E.; Mitsch, W.J. Ecological engineering principles. In *Ecological Engineering: An Introduction to Eotechnology*; Mitsch, W.J., Jorgensen, S.E., Eds.; John Wiley and Sons: New York, 1989; 21–37.
11. Schlesinger, W.H.; Reynolds, J.F.; Cunningham, G.L.; Hueneke, L.F.; Jarrel, W.M.; Virginia, R.A.; Whitford, W.G. Biological feedbacks in global desertification. *Science* **1990**, *247*, 1043–1048.
12. Farrell, J. The influence of trees in selected agroecosystems in Mexico. In *Agroecology: Researching the Ecological Basis for Sustainable Agriculture*; Gliessman, S.R., Ed.; Springer-Verlag: New York, 1990; 167–183.
13. Tongway, D.J. *Rangeland Soil Condition Assessment Manual*; CSIRO: Melbourne, 1994; 69 pp.
14. Tongway, D.; Hindley, N. *Assessment of Soil Condition of Tropical Grasslands*; CSIRO Division of Wildlife and Ecology: Canberra, 1995; 60 pp.
15. Graetz, R.D.; Ludwig, J.A. A method for the analysis of piosphere data applicable to range assessment. *Aust. Rangeland J.* **1978**, *1*, 126–136.
16. Tongway, D.; Hindley, N. *Ecosystem Function Analysis of Rangeland Monitoring Data*, 2000. Available at <http://www.nlwra.gov.au/atlas>.
17. Tongway, D.J.; Ludwig, J.A. Rehabilitation of semi-arid landscapes in Australia. I. Restoring productive soil patches. *Restor. Ecol.* **1996**, *4*, 388–397.
18. Ludwig, J.A.; Tongway, D.J. Rehabilitation of semi-arid landscapes in Australia. II. Restoring vegetation patches. *Restor. Ecol.* **1996**, *4*, 398–406.
19. Rhodes, D.W.; Ringrose-Voase, A.J. Changes in soil properties during scald reclamation. *J. Soil Conserv. Service NSW* **1987**, *43*, 84–90.
20. Tongway, D.; Hindley, N. Assessing and monitoring desertification with soil indicators. In *Rangeland Desertification*; Arnalds, O., Archer, S., Eds.; Kluwer Academic: Dordrecht, 2000; 89–98.
21. Tongway, D.J.; Hindley, N.L. *Landscape Function Analysis Manual Version 3.1*; CSIRO Sustainable Ecosystems: Brisbane, 2004.
22. Kinloch, J.E.; Bastin, G.N.; Tongway, D.J. Measuring landscape function in chenopod shrublands using aerial videography. In *Proceedings of the 10th Australasian Remote Sensing and Photogrammetry Conference*, Aug 21–25, 2000; Causal Productions: Adelaide, 2000; 480–491.

Desertization: Irreversible Extension of Desert Land

Henry Noel LeHouérou

Functional and Evolutionary Ecology Center (CEFE), National Center for Scientific Research (CNRS), Montpellier, France

Abstract

Combating desertization first requires the discontinuation, or at least the serious mitigation of previous destructive activities that brought the situation about. Biological recovery may not be feasible anymore whenever degradation is extreme; artificial human-induced intervention may then be deemed necessary. On the other hand, in less extreme cases, artificial rehabilitation and restoration may be desirable to hasten the desired goal. Biological recovery may be achieved via centuries-old techniques of soil and water conservation, water harvesting, runoff farming, agroforestry, appropriate farming, stocking, range, and forest management. But such activities often necessitate a deliberate policy and adequate strategies of conservative land use; this, in turn, may require political and legal actions encouraging responsible land use management to achieve sustainability, i.e., a profound land reform. This has actually been achieved by various centuries-old peasant civilizations in various parts of the world arid lands.

INTRODUCTION

Desertization is the irreversible extension of desert-like landscapes and landforms to areas where they did not occur in the past.^[1,2] Desertization results, to a large extent, from land abuse by humans and livestock; it is not a consequence of climate fluctuation, albeit drought may worsen and accelerate a phenomenon that could have been triggered during high-rainfall periods (there are many examples of such seemingly paradoxical situations). It is estimated^[3–6] that over 20% of the 42 million km² of dry lands are submitted to a greater or lesser extent to desertization processes.

The causes of desertization are both direct and indirect. Direct causes^[3,7] are those that have a strong immediate impact on site soil characteristics and properties. The major one is the reduction of perennial plant cover and biomass that contributes to soil organic matter depletion. This reduction of soil organic matter content triggers a chain of actions and reactions affecting both soil and microclimate, leading to lowered biological activity, compacted soils with high runoff, and fast erosion processes, by both water and wind; soil may thus ultimately be turned sterile or almost so in a few years of abuse.

The indirect causes are those that bring about the depletion of natural resources, such as long-standing overcultivation, overstocking, poor farming practices, deforestation, destructive collection of fuel wood, and other mismanagement practices,^[1,2] and all these result from excessive and permanent pressure on the land by exceedingly numerous human and livestock populations: continuous high densities and stocking rates.

DEFINITION

Deserts are defined as regions where rainfed agriculture is not feasible because of excessive aridity.^[8] Unlike the word “desertification,” which has at least four distinct meanings, the term “desertization” is clearly defined. It implies irreversible extension of desert landforms and conditions to regions where they did not occur in the past.^[1,2] The desertization is primarily due to destructive human and livestock activities in arid environments. In spite of the United Nations Environment Programme (UNEP) Plan to Combat Desertification (as a follow-up to the United Nations Conference on Desertification (UNCOD) held in Nairobi in 1977, the Earth Summit of Rio de Janeiro in 1992, and the Convention of Paris in 1994), little has been done on a large scale, so desertization is likely to continue over the next years as it has in the past years.^[9]

FACTS AND FALLACIES

Origin of Deserts

Deserts are regions where, because of excessive aridity, no agriculture is feasible without supplemental irrigation.^[8,10] On the basis of their origin, there are five types of deserts.^[10,11] True climatic deserts result from general atmospheric circulation whereby the adiabatic subsidence of dried-up air masses occurs on the limits between the Ferrel and Hadley cells along the 30°N and 30°S parallels.^[11] The origin of true deserts varies with local conditions (e.g., tropical and subtropical deserts). Rain shadow deserts

(a particular case of climatic deserts) are formed due to orographic obstacles cutting the trajectory of rain-bearing frontal depressions (e.g., the Death Valley and Colorado River Delta east of the Sierra Nevada of California, Patagonia). Coastal deserts are formed due to the atmospheric stability resulting from the upwellings of coastal cold currents, such as along the Western continental shores of America and Africa (e.g., the Humboldt Current in Chile and Peru, the Benguela Current of Southwest Africa). Edaphic deserts result from geological and soil conditions that are inappropriate for agricultural activities and the development of natural vegetation, such as unweathered hard or toxic rocks, salinity, and alkalinity. Man-made deserts result from anthropogenic activities and destructive land abuse. This entry is concerned with the formation of the human-induced desert.

Fluctuations and Trends of Climatic Variables in Historical Times

Rainfall

There is a consensus among scientists on the absence of trends in long-term rainfall since the beginning of history.^[3,12–14] There is, however, evidence of considerable changes in the course of the Pleistocene and Holocene periods beginning 3000 and 10,000 years B.P., respectively, with some 10 alternating periods of dry and humid climates in the Sahara and Arabia.^[9,12,15,16] This evidence comes essentially from tree ring studies, pollen analysis, and lake-level fluctuations. Minor fluctuations took place, however, in the course of history such as the 25-year period of drought in the African Sahel (1968–1983) and East Africa from 1975 to 1984,^[15,17] in Chile from ca. 1900 to date, and in Argentina between 1900 and 1950.

Temperature

Variations in temperature are more complex to analyze. There is a consensus on a global increase of 0.5°C over the past 100 years.^[13,14] But this includes the effect of urbanization that may globally account for up to 0.2°C.^[18] Fluctuations during the historical period are fairly well documented. The causes of these “random” fluctuations are not known with any certainty. The magnitude of increase over the past 100 years is thus of the same dimension or smaller than previous random fluctuations.

Air Pollution

There is general agreement on the fact that the atmospheric content of carbon dioxide (CO₂) and other pollutants is increasing. The CO₂ content of the atmosphere was 280–300ppmv (parts per million in volume) at the beginning of the industrial era some 150 years ago.^[19] It is 370ppm, essentially due to the burning of fossil fuel.^[14] Similarly, the content of methane (CH₄) has doubled from 800 to

1750 ppbv (parts per billion in volume) over the same period.^[14] The main sources of CH₄ are swamps, rice fields, ruminants, forest fires, garbage disposal, oil industries, and termites.

But the situation is different in as much as it is the increasing content of CO₂ and CH₄ in the atmosphere that is causing a temperature rise, while in the past 160,000 years, and presumably all the quaternary, the situation was the opposite. It is the increase in temperature from orbital forcing that resulted in a rise of CO₂ and CH₄, by the oxidation of soil organic matter content, the melting of tundra permafrost soils, and the release of CH₄ from the thaw of frozen swamps. The undisputed 0.5°C/100 years of global temperature increase thus cannot, strictly speaking, be attributed to CO₂ or other atmospheric pollutants.

Land Surface Albedo

As vegetation is increasingly depleted and perennial plant cover recedes, the proportion of bare ground increases, enhancing its reflectivity, or the surface albedo.^[20] Higher albedo reduces temperature, decreases evaporation, and creates a thermic depression, impeding cloud formation and possibly decreasing rainfall. This so-called surface albedo feedback theory is, however, highly debated—actual facts have not, so far, proved it; rather the opposite.^[21,22]

DESERTIZATION

Definition

Desertization is the irreversible degradation of arid land resulting in desert-like landforms in areas where they did not occur in the past.^[1,2] This definition differs from the concept of desertification as accepted by UNEP in the 1977 UNCOD held in Nairobi.^[4,23] It was then defined as “land degradation under arid, semiarid, and dry subhumid climates that may ultimately lead to desert-like conditions.” The term “desertification” is used for two distinct meanings. It implies the *irreversible* degradation of arid lands, leading to the formation of man-made deserts. In Western Europe and European Economic Community, desertification is synonymous with land abandonment, resulting in natural reforestation and expansion of woodlands.^[24,25] Land abandonment may be referred to as “land desertion” because the rural population migrates to towns.^[25]

Causes of Desertization: Degradation Processes

The causes of desertization are both direct and indirect.

Direct causes

The direct causes pertain essentially to the destruction of the perennial plant cover and biomass and its consequences on soil characteristics through erosion, sedimentation,

waterlogging, and salinization.^[1,2,16,26–28] The destruction of perennial biomass and cover results in the following degradative chain reaction:

1. Reduced litter production and return to the soil particles decrease soil organic matter content.
2. Reduced soil organic matter weakens or destroys soil structure and aggregate stability.
3. Unstable soil aggregates increase surface soil sealing from raindrop splash on silty or loamy soils and biological crusting on coarse sandy soils.^[29]
4. Sealed soil surfaces increase runoff, decrease soil water intake and storage, and create a drier environment.
5. Increased runoff causes flooding; waterlogging; and mass die-off of shrubs, trees, and crops downstream and in closed depressions.
6. High wind speed on the soil surface, a consequence of reduced perennial plant cover, leads to higher albedo, increased potential evapotranspiration, increased water and wind erosion, and therefore drier microclimatic conditions.
7. Higher soil-surface daily maximum temperatures and therefore high evapotranspiration, due to the lack of shading, enhances the oxidation of soil organic matter, a rapid depletion of water reserves, and a shorter growing season. Lower soil-surface night temperature causes a larger diurnal and seasonal temperature range (thermal amplitude), leading to less favorable conditions for germination, emergence, and establishment of seedlings and to the restoration of the natural vegetation.
8. Decreases in the number, biomass, and activity of soil microflora (e.g., bacteria, actinomycetes, fungi, algae, and rhizospheric symbionts) as a result of decreasing organic matter content in the soil slow down nutrient turnover and reduce the availability of major and trace geobiogenic elements, leading to lower fertility and productivity.
9. Decreased productivity reduces the number, biomass, and activity of soil microfauna, mesofauna, macrofauna, and megafauna.
10. Wind erosion (creeping, saltation, suspension, and corrosion) results in the development of desert pavements (regs, serirs, hamadas, and gobis), rock mushrooms, and yardangs.
11. Wind deposition leads to the formation of sand clay dunes (lunettes), loess, nebkas, ramblas, shamo, ergs, edeyen, nefuds, sand sheets, and sand veils. Gravity erosion leads to dry colluvium deposits.
12. Water erosion (sheet, rill, gully, pipe, landslides, etc.) leads to flooding, sedimentation, and alluvium deposition.
13. Rising water tables and salinity result from higher runoff and lesser pumping by shrubs and trees.
14. Rising salinity and soil sterilization are caused by poor drainage and other mismanagements.

Indirect causes

The indirect causes of desertization are those that exacerbate the development of the direct causes.

1. Rapid human population growth and quick increase in population density lead to high human and livestock pressure on the land. The human population in the arid lands of the developing countries has grown at an annual rate of 3.2% since 1950.^[2,3,16,30]
2. Livestock populations in Africa and Southwest Asia have increased at an average annual rate of 2.2% since 1950.^[3,12,16]
3. Soil degradation is accentuated by overcultivation, inappropriate farming methods, poor land management (e.g., cultivation of land that is too dry, too steep, and too shallow), inappropriate tillage (up and downhill plowing), quasi-exclusive utilization of disc plows, inappropriate crop rotation patterns, inadequate field size, careless harvesting, postharvesting land management, etc.
4. Deforestation is exacerbated by excessive collection of firewood (1.5 kg per person per day). As arid land steppe vegetation of dwarf shrubs bears some 300–600 kg dry woody matter per hectare, each person depending on this type of fuel destroys about 1 ha of steppe vegetation per annum.^[8]
5. Overcollection of medicinal and other useful plants contributes to the decline of plant cover and biomass.
6. Wildfires are particularly harmful in the tropical savannas. As they occur in the dry season, they leave a soil surface that remains bare and therefore vulnerable to erosion during the next rainy season.^[17,31]
7. Vegetation arcs are particular patterns of distribution, whereby plants are concentrated on contour strips alternating with bare and sealed interstrips. Seen from above, as on air photos or satellite images, the land looks like a tiger skin, hence the expression of “tiger bush.”^[17,24,27] This is a case of edaphic aridity when, owing to the thinning of vegetation from woodcutting and/or overbrowsing and the consecutive runoff and erosion, the soil moisture regime is too low for the vegetation. Thus, vegetation is confined in strips that permit higher soil moisture due to run-on.
8. Inadequate legislation or lack of enforcement of legislation on land use and erosion control measures.
9. Inappropriate land-ownership systems under which land, grazing, and water are free communal resources while livestock are privately owned. Thus, it is in each stockman’s interest to draw the maximum from the common resource without investing anything in it, the “tragedy of the commons.”
10. Inappropriate marketing facilities and inadequate incentive to de-stock and cull animals.
11. Poverty and lack of education and inefficient extension services.

Table 1 Desertization risk assessment from perennial plant canopy cover evaluation.

MAR nmm	Perennial plant canopy projection cover in percent of ground surface					
	0–1	1–5	5–15	15–25	25–50	>50
50–100	5	5	4	3	2	1
100–200	5	4	3	2	1	0
200–300	4	3	2	1	0	0
300–400	3	2	1	0	0	0
400–500	2	1	0	0	0	0
500–600	1	0	0	0	0	0

Note: 0, present hazard non-existent or very low; 1, immediate risk low; 2, immediate hazard moderate; 3, immediate risk serious; 4, immediate hazard severe.

Source: From Le Houérou.^[33]

Diagnosis and evaluation of desertization

Desertization may be evaluated via various criteria, namely measuring soil erosion and sedimentation, assessing perennial plant canopy cover and biomass or the basal cover of perennial species, and evaluating the rain-use efficiency (RUE = kg DM/ha/yr/mm) and computing the production to rain variability ratio (PRVR = CV Ann. Prod./CV Ann. rainf.).^[32] The evaluation of RUE is a particularly easy, quick, and clean method.

Low perennial plant cover is the principal factor of soil surface erosion by water and wind. In the Mediterranean steppelands, there is virtually no soil surface erosion when perennial plant canopy cover reaches 25% of soil surface. In such cases, erosion is compensated by deposition and sedimentation.^[26] In the North African steppes, for instance, each 1% of perennial canopy cover corresponds to a biomass

of 30 to 60 kg DM/ha.^[33] It is then easy to determine the stage of degradation and desertization from the evaluation of perennial plant canopy cover (Table 1). The world regional distribution of arid lands is shown in Table 2.

Severity and Extent of Desertization

Can desertization occur without the impact of drought?

True climatic deserts cannot exist without permanent and extreme climatic aridity. But such is not the case with the occurrence and spread of man-made deserts. Contracted vegetation occurs in all world deserts with a mean annual rainfall (MAR) isohyets of between 50 and 150 mm depending on the substrate, the anthropogenic impact, and other local factors [e.g., high atmospheric humidity and occult precipitation, or, conversely, continentality, low RH, and high precipitation to potential evapotranspiration ratio (PET)]. Desertization is often caused by droughts, but the primary causes have often been operating long before drought occurred. The effects may have been hidden by a period of better than average climatic conditions. This was the case in the West African Sahel.

The analyses of available information lead to the following conclusions:

1. Desertization may be triggered during non-drought periods.
2. Desertization cannot result from drought alone.
3. The main causative factor appears to be land mismanagement from excessive anthropogenic pressure on the land (land abuse), beyond the ecosystems' resilience.

Table 2 Regional distribution of world arid lands.

Geographic area	Continental distribution (10 ³ km ²)					% ^a	
	Eremitic	Hyperarid	Arid	Semiarid	Total		
MAR (mm)	<50	50–100	100–400	400–600			
100 P/PET (mm) ^b	<3	3–5	5–28	28–45			
Budyko's aridity index ^c	>50	10–50	3–10	2–3			
World	130,737	7,500	7,059	14,330	12,651	41,440	100
Africa	30,312	6,232	3,017	3,570	2,951	15,770	52
North America	21,322	10	90	1,025	1,935	3,060	14
South America	17,818	275	105	972	1,274	2,626	14
Asia	43,770	1,595	3,225	5,415	4,817	15,042	34
Europe (Spain)	500	NA	NA	100	300	400	80

Note: NA, not applicable.

^aPercent of geographic area.

^b100 P/PET = precipitation to potential evapotranspiration ratio, × 100 for easier handling. PET being evaluated via Penman equation.

^cBudyko's index is the ratio between net energy budget and actual annual rainfall; in other words, it evaluates how many times the energy budget could evaporate the MAR (which does not take into consideration of the aerodynamic parameter of Penman's equation).

Source: From Le Houérou.^[3]

Table 3 Extent and severity of arid and semiarid land degradation.

Bioclimatic zones	Degradation intensity (10 ³ km ² and %)									
	Light		Moderate		Strong		Very strong		Total	
	S	%	S	%	S	%	S	%	S	% ^a
Arid and semiarid zones	3601	42	3964	45	1096	12	62	1	8723	20
Water erosion	1475	37	1757	45	666	17	40	1	3938	9
Wind erosion	1,662	46	1815	50	152	4	15	0	3644	8
Chemical degradation	373	44	266	31	203	24	7	1	849	2
Physical degradation	91	31	126	43	75	26	0	0	292	1

^aPercent of dry lands (SA + A + HA.Er).

Source: From Oldeman, Hakkeling, et al.^[5] and Middleton & Thomas.^[6]

Severity and extent

The global extent of land degradation has been assessed by International Soil Reference and Information Centre/UNEP.^[4–6,23] The conclusions are in good agreement with other independent studies.^[3] Tables 3, 4, and 5 show that overstocking and overgrazing are the major causes of arid land degradation worldwide, followed by overcultivation and poor farming methods. The regional and local causes, however, may considerably alter this pattern. Overcultivation and shrubland clearing, for instance, are by far the prime causes in Northern Africa, Southwestern Asia, and Northwestern China, while overgrazing is the foremost cause in the Sahel, East Africa, North and South America, and Australia (Table 5). Salinization affects 1.5 million hectares annually out of a world irrigated land area of 2 million km². The total area lost to secondary salinity over the 20th century is about 25 million hectares.^[3,32]

Natural Amelioration

Natural amelioration of land management to restore or rehabilitate ecosystems and environment implies the elimination or substantial reduction of the causes that leads to degradation. This goal can be achieved by reducing stocking rates to carrying capacity; restricting farming to appropriate areas, such as depressions with deep soils benefiting from runoff or having a water table; and improving grazing systems through the control of livestock, fencing, rotational/controlled grazing, and installing temporary exclosures. These activities are usually cheap and efficient tools for restoring and rehabilitating arid lands.^[34–36]

Managed Restoration

Human-assisted rehabilitation includes a large array of techniques from the simplest and cheapest to the most sophisticated and costly but not necessarily efficient.

Surface roughness

Techniques to create surface roughness break the surficial crust (e.g., a silt seal or a biological crust), increase permeability to air and rainwater, and facilitate seed germination and seedling emergence. A simple technique for breaking crust involves spreading branches from spiny trees and shrubs on the soil surface (*Acacia* spp., *Balanites* spp., *Ziziphus* spp., *Prosopis* spp., *Rhus* spp., *Parkinsonia* spp., *Commiphora* spp., etc.). Soil surface roughening may also be achieved with the plow harrow, hoe, cultivator, subsoiler, ripper, basin lister, etc.^[17]

Contour lining

This may be achieved by using stones or pieces of hardpan or duricrusts (e.g., gypsic pans, calcrete, iron pans, or lateritic pans). Light contour lining aims at reducing sheet erosion and improving water intake. It may also be achieved via mechanical equipment used for terracing and building contour banks, contour benches, or ditches.

Ripping and subsoiling

These techniques use heavy equipment to break deep, hard lime, gypsum, silicium (silcrete), or iron pans or to loosen hard compacted soils to improve water intake and storage

Table 4 Causes of degradation in dry lands (10³ km²).

	Deforestation	Overgrazing	Overcultivation	Other	Total degradation	Non-degradation	Total dry lands
S	1,700	4,490	2,450	83	8,723	34,832	43,554
%	20	51	28	1	100/20	80	100

Source: From Le Houérou,^[3] Le Houérou,^[12] UNEP,^[23] and Le Houérou.^[37]

Table 5 Causes of desertization and percentage of desertized land.

Regions	Overcultivation	Overstocking	Fuel wood collection	Salinization	Urbanization	Other	Total
Northwest China	45	18	18	2	3	14	100
North Africa and Southwest Asia	50	26	21	2	1	—	100
Sahel and East Africa	25	65	10	—	—	—	100
Middle Asia	10	62	—	9	10	9	100
Australia	20	75	—	5	?	—	100
The United States	22	73	—	5	1	—	100

Source: From Le Hou  rou^[7] and Le Hou  rou.^[30]

and deepen the root zone of protective plants, particularly shrubs and trees.

Termite-assisted hand-pitting across a surficial pan

The southern Sahelian technique of the “Zai” is a cheap and efficient practice. It involves digging pits (40 length × 40 width × 15 depth) about 80 cm apart at a density of 2500 pits/ha. These pits are dug a few months prior to the onset of the rainy season, and 1000–1500 kg/ha of organic matter (e.g., litter, straw, and stovers) is placed at the bottom of these pits. The organic matter attracts termites that dig deep galleries below the pan. The microenvironment thus created is then manured at a rate of 1000 kg/ha of dry corral dung. Pearl millet or sorghum planted in these pits may reach maturity even in a dry year, owing to the better soil water regime. Each pit is a green island amid a devastated environment landscape, and the beneficial effect may last up to 30 years. The cost of making Zai is 150–300 human-days per hectare for a pit density of 2500/ha, 1000 kg of organic matter, and 1000 kg of corral manure.^[38]

Table 6 World areas under agroforestry management.

Fodder cacti	1,200,000 ha	North Africa, South Africa, North and South America
Saltbushes	1,000,000 ha	North Africa, South Africa, Southwest Asia, North and South America, Australia
Acacias spp.	600,000 ha	North Africa, Sahel, South Africa, Australia
Prosopis spp.	50,000 ha	North and South America, India
Agave americana	100,000 ha	South Africa
Miscellaneous (Saxaouls, Tamarix, Bohemia olive, and poplars)	250,000 ha	Africa, Australia, North and South America, Europe, Central Asia

Range reseeding

Range reseeding and establishing improved pasture are also feasible in the semiarid zone, but not so in the arid zone.^[7,28]

Agroforestry

Agroforestry is practiced on 3.2 million hectares in world arid lands. Agroforestry, using either native or exotic species, is a very potent and efficient tool for biological recovery, allowing productivity 3 to 10 times higher than that of pristine vegetation under the same ecological conditions. These lands are distributed as shown in Table 6.^[35,36]

Wildlife management, wildlife husbandry, and tourism

Wildlife management and husbandry are actively pursued in an increasing number of arid land countries: namely the United States, Australia, East Africa, Southern Africa, New Zealand, Argentina, and Western Europe. Privately owned game ranches or game farms are quickly developing in Africa.

Adaptive arid land strategy and sustainable development policies

Mitigating desertization is a problem of sustainable arid land development. Sustainable development policies include many facets (e.g., technical, social, economic, political, legislative, organizational, and educational). An important step toward sustainable development is reforming the land tenure system. This would require a fundamental land reform in order to shift from communal or tribal land ownership to secure private systems, whereby long-term investment becomes possible and long-term planning can be implemented.

REFERENCES

1. Le Hou  rou, H.N. La D  sertisation Du Sahara Septentrional Et Des Steppes Limitrophes. Ann. Alg  r. de G  ogr. 1968, 6, 2–27.

2. Le Houérou, H.N. The nature and causes of desertization. *Arid Lands Newslett.* **1976**, 3, 1–7.
3. Le Houérou, H.N. An overview of vegetation and land degradation in world arid lands. In *Degradation and Restoration of Arid Lands*; Dregne, H.E., Ed.; Texas Technical University: Lubbock, 1992; 127–163.
4. UNCOD. *United Nations Conference on Desertification*; UNEP: Nairobi & Pergamon Press: New York, 1977.
5. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *World Map of Status of Human-Induced Soil Degradation: An Explanatory Note*; ISRIC: Wageningen & UNEP: Nairobi, 1990.
6. Middleton, N.; Thomas, D. Eds. *World Atlas of Desertification*, 2nd Ed.; Arnold, Hodder Headline, PLC: London, 1997.
7. Le Houérou, H.N. Drought-tolerant and water-efficient trees and shrubs for the rehabilitation of tropical and subtropical arid lands. *J. Land Husbandry* **1996**, 1 (1–2), 43–64.
8. Meigs, P. *World Distribution of Arid and Semiarid Homoclimates*; 5 Maps, Reviews of Research on Arid Zone Hydrology; UNESCO: Paris, 1952.
9. Le Houérou, H.N. *Climate Change Drought and Desertification*; Report to IPCC; IPCC: Washington, 1995.
10. McGinnies, W.G.; Goldman, B.G.; Paylore, P. *Deserts of the World: An Appraisal of Research into Physical and Biological Environments*; University of Arizona Press: Tucson, 1968.
11. Monod, T. *Les Déserts*; Horizons de France, Publ.: Paris, 1973.
12. Le Houérou, H.N. Vegetation and land-use in the Mediterranean basin by the year 2050: A prospective study. In *Climatic Change and the Mediterranean*; Jeftic, L., Milliman, J.D., Sestini, G., Eds.; Edward Arnold: London, 1992; 175–232.
13. Houghton, J.T.; Callander, B.A.; Varney, S.K. *Climate Change, 1992: The Supplementary Report to IPCC Scientific Assessment*; Cambridge University Press: London, 1992.
14. Houghton, J.T.; Jenkins, G.J.; Ephraums, J.J. *Climate Change: The IPCC Scientific Assessment*; Cambridge University Press: London, 1990.
15. Le Houérou, H.N. Changements Climatiques Et désertisation. *Sècheresse* **1993**, 4, 95–111.
16. Le Houérou, H.N. La Méditerranée En 2050: Impacts Respectifs D'une éventuelle évolution Climatique et de la démographie Sur La Végétation, Les écosystèmes Et L'utilisation Des Terres. Etude Prospective. *La Météorologie*, **1991**, VII Série. No. 36, 4–37.
17. Le Houérou, H.N. *The Grazing Land Ecosystems of the African Sahel*, Ecological Studies no. 75; Springer Verlag: Heidelberg, 1989.
18. Duplessy, J-C. Quand l'Océan Se fâche. In *Histoire Naturelle du Climat*; Éditions Odile Jacob: Paris, 1996.
19. Keeling, C.D.; Bacastow, R.B.; Bainbridge, A.E.; Ekdahl, C.A.; Guenther, P.R.; Waterman, L.S. Carbon dioxide variation at mauna loa observatory Hawaii. *Tellus* **1976**, 28, 28.
20. Charney, J. Dynamics of deserts and drought in the Sahel. *Quat. J. Roy. Meteor. Soc.* **1975**, 101, 193–202.
21. Williams, M.A.J.; Balling, R.C., Jr. *Interactions of Desertification and Climate*; WMO: Geneva & UNEP: Nairobi, 1994.
22. Courel, M.F. *Etude de L'évolution Recente Des Milieux Sahéliens à Partir Des Mesures Fournies Par Les Satellites*; IBM: Paris, 1985.
23. UNEP. *World Atlas of Desertification*; UNEP: Nairobi & Edward Arnold: London, 1992.
24. Le Houérou, H.N. Impact of man and his animals on Mediterranean vegetation. In *Mediterranean-Type Shrublands, Ecosystems of the World*; Di Castri, F., Goodall, D.W., Specht, R.L., Eds.; Elsevier: Amsterdam, 1981; Vol. 11, 479–521.
25. Le Houérou, H.N. Land degradation in Mediterranean Europe: Can agroforestry be a part of the solution? A prospective review. *Agroforestry Syst.* **1993**, 21, 43–61.
26. Le Houérou, H.N. Recherches écologiques et floristiques sur la végétation de la Tunisie méridionale. *Mém. H.S. no. 6*; Institut. Rech. Sahar; Univ. d'Alger Algeria, 1959.
27. Le Houérou, H.N. The Sahara from the bioclimatic view point. Definition and limits. *Ann. Arid Zone* **1995**, 34 (1), 1–16.
28. Le Houérou, H.N. Bioclimatologie Et Biogéographie Des Steppes Arides Du Nord De l'Afrique. In *Options Méditerranéennes*. Sér. B (10): 1–396; CIHEAM: Paris, 1995.
29. Verrecchia, E.; Yaïr, A.; Kidron, G.J.; Verrecchia, K. Physical properties of the psammophile cryptogamic crust and their consequences to the water regime of sandy soils, North-Western Negev Desert, Israel. *J. Arid Environ.* **1995**, 29 (4), 427–438.
30. Le Houérou, H.N. Dégénération, Régénération Et Mise En Valeur Des Terres Sèches D'Afrique. In *L'homme Peut-il Refaire ce Qu'il a Défait?*; Pontanier, R., M'Hiri, A., Aronson, J., Akrimi, N., Le Floch, E., Eds.; John Libbey Eurotext: Montrouge, 1995; 65–104.
31. Goldammer, J.G., Ed. *Fire in Tropical Biota*; Ecological Studies No. 84; Springer Verlag: Heidelberg, 1990.
32. Le Houérou, H.N. A Probabilistic approach to assessing arid rangelands productivity, carrying capacity and stocking rates. In *Drylands: Sustainable Use of Rangelands into the Twenty-First Century*; Squires, V.R., Sidahmed, A., Eds.; Chapter 12; IFAD: Rome, 1998; 159–172.
33. Le Houérou, H.N. *Aspects météorologiques De la Croissance et du Développement Végétal Dans Les Déserts et Les Zones Menacées De Désertification*; Report No WMO/TD-No 194; UNEP: Nairobi & WMO: Geneva, 1987.
34. Le Houérou, H.N. The role of cacti (*Opuntia* Spp.) in land rehabilitation in the Mediterranean basin. *J. Arid Environ.* **1996**, 32, 1–25.
35. Le Houérou, H.N. Utilization of fodder trees and shrubs for the arid and semiarid zones of West Asia and North Africa. *Arid Soil Res. Rehabil.* **2000**, 14, 101–135.
36. Le Houérou, H.N. Restoration and rehabilitation of arid and semiarid Mediterranean ecosystems in North Africa and West Asia. A review. *Arid Soil Res. Rehabil.* **2000**, 14, 3–14.
37. Le Houérou, H.N. Man-made deserts: Desertization process and threats. *Arid Land Res. Manag.* **2000**, 16, 1–36.
38. Roose, E. *Introduction à la Gestion Conservatoire de L'eau, de la Biomasse et de la fertilité Des Sols (GCES)*; Bulletin Pédologique No. 70; FAO: Rome, 1994.

Developing Countries: Economics of Soil Management

Stefano Pagiola

Environment Department, World Bank, Washington, District of Columbia, U.S.A.

Abstract

Land degradation problems seem unlikely to cause the catastrophic agricultural declines as some fear, but they can have acute consequences at the local level, particularly for poor farmers. Farmers have strong incentives to respond to the on-site effects of soil degradation, and the evidence shows they often do so. The changing understanding of farmers' incentives to undertake appropriate soil management practices has shifted the emphasis of interventions away from top-down projects focusing on technical solutions. Overall policy reform is seen as a necessary precondition for any intervention. Where on-site effects dominate, participative projects, in which responses appropriate to local conditions are developed in collaboration with the affected land users, have come to play an important role. Where off-site effects dominate, efforts have turned to the development of systems of payments for environmental services.

INTRODUCTION

Agriculture plays a vital role in the economies of developing countries and in the welfare of their populations, including many of their poorest members. Inappropriate soil management is thought to threaten the sustainability of agricultural production by undermining the soil resource base on which agriculture depends.^[1–4] Developing country farmers often have limited capacity to substitute fertilizers and other inputs for soil fertility, making the problem particularly acute. Addressing such problems is not easy, however, because decisions about soil management are made not by governments, but by individual land users according to their own priorities and constraints. Economic analysis of soil management problems in developing countries has focused increasingly on understanding why farmers use soils in the way they do, and in particular on understanding their reasons for not using them sustainably.

ASSESSING THE PROBLEM

Soil has value because of its role in crop production. Cultivation, however, can damage the soil. For example, clearing vegetation cover exposes soil to water and wind erosion; repeated tillage weakens soil structure; crop production removes nutrients; and use of machinery leads to soil compaction. The economic impacts of land degradation can be divided into on-site and off-site effects. On-site degradation can result in lower yields (and, hence, lower revenues) or in the need for higher input levels (and, hence, higher costs) to maintain yields.^[5] Off-site degradation can cause problems such as sedimentation and changes in hydrological flows.^[6,7] Adverse off-site effects can also occur in the absence of on-site effects (e.g., if

agrochemicals contaminate water supplies). Land degradation has also been seen as contributing to global problems, such as climate change, by reducing soil's ability to store carbon.^[8,9] On-site problems have usually been thought to be most important in developing countries,^[10] although attention has been focused on off-site effects, particularly in Asia and Latin America.^[11]

Assessing the absolute and relative magnitude of soil degradation and its consequences is difficult. Valuing on-site effects is straightforward in principle, but it has proved empirically difficult in practice, because of the lack of appropriate data, particularly on the yield impact of degradation. The site specificity of soil management relationships limits the applicability of data collected in one location to analysis of problems at another. Efforts to value off-site effects are also constrained by insufficient data, exacerbated by unclear cause-and-effect relationships between events that are widely separated in both space and time.^[12] In sub-Saharan Africa, estimated annual on-site losses from degradation range from under 1% of agricultural gross domestic product (GDP) in Madagascar, Mali, and South Africa, to as much as 8% in Zimbabwe.^[13] Three different studies of Ethiopia, however, showed estimated annual losses of less than 1%,^[14] 4%,^[15] and 5%^[16] of agricultural GDP, demonstrating the weakness of the data and the dependence on the assumptions made.

Although data weaknesses make it difficult to arrive at strong conclusions, fears of catastrophic damage from soil degradation do not appear to have been realized. A review of 26 global and regional studies and 54 national studies concludes that degradation is not likely to threaten aggregate global food supply by 2020, although degradation problems can be acute at the local level.^[4] An analysis of degradation problems in El Salvador, e.g., concludes that, while the oft-cited estimate that “75% of the country is

degraded” is almost certainly exaggerated, degradation does affect about one-third of farm households—a significant social problem by any standard.^[17]

UNDERSTANDING FARMER BEHAVIOR

Because farmers adopt soil management practices that degrade the soil, many researchers have concluded that farmers are ignorant or tradition bound. Beginning with the work of Schultz,^[18] however, it has been increasingly recognized that developing country farmers are, in fact, rational decision makers.^[19,20] Research demonstrates that they are quite aware of the properties and behavior of their soils and of the problems that inappropriate soil management can cause.^[21–25]

Farmers experience the effects of any on-site problems directly, so they generally have a direct incentive to respond to them. But even though sustainable practices may bring long-term benefits, they can also be costly, both directly, in terms of investment requirements and maintenance costs, and indirectly, in terms of forgone production. The critical question farmers face is whether the long-term benefits of sustainable practices make these costs worth bearing. In many cases, the answer is clearly yes. Almost all farmers in the Kitui/Machakos area of Kenya, e.g., have adopted some form of conservation practice,^[26,27] while small-scale farmers in El Salvador have done so on over one-half the fields on moderate and steep slopes.^[17] In some instances, however, the costs of adopting particular conservation techniques exceed the benefits. Analysis of conservation measures in six Central American and Caribbean countries, e.g., found that many measures promoted by conservation projects had negative returns from the farmers’ perspective.^[28]

Many factors can affect the profitability of conservation measures from the farmers’ perspective. These include the rate and severity of damage to the soil, the consequent effects on productivity, the value of lost production (for sale or subsistence consumption), the extent to which fertilizers and other inputs can cost-effectively substitute for lost fertility, the effectiveness of available conservation measures and their costs, including initial investment and maintenance, the relative riskiness of production, with and without conservation, and the farmers’ own preferences, including—crucially—their preference for present as opposed to future consumption (i.e., their discount rate).^[27–30] Because all of these factors can vary substantially, even in small areas, it should not be surprising that the extent of adoption of conservation measures also varies substantially. Adoption can also vary over time: When initial productivity losses are small, it makes sense to delay adopting conservation practices until later.^[31,32]

Poverty is often thought to play a significant role in conservation adoption decisions of developing country farmers, by causing them to emphasize short-term over long-term benefits and limiting their ability to undertake

investments.^[33,34] Conversely, it has been argued that poorer farmers may have greater incentives to conserve soil than better-off farmers, because they face greater penalties for failing to do so.^[35] Empirically, one can also observe that poor farmers do often undertake a variety of long-term investments—in livestock, tree crops, the education of their children, and other assets with long-term returns. Thus, in the case of Kitui/Machakos, even expensive conservation measures such as terraces have been widely adopted by poor farmers, who lack access to formal credit markets.^[26,27] Like many other aspects of the soil-conservation debate, the exact role played by poverty is likely to vary from case to case.

Government policies can often affect the perceived incentives to adopt conservation practices. Many developing countries’ policies discriminated heavily against agriculture,^[20,36] making investments in agriculture, including conservation investments, less attractive. The policy environment has been identified as the main cause of degradation, e.g., in Colombia.^[37] The relationship between government policies and conservation is complex, however, and could go in either direction depending on the details of the policy and the specific conditions encountered at any given site.^[38,39] By keeping prices low, e.g., Kenyan maize price policy tended to reduce returns to conservation in Kitui/Machakos, and thus discourage its adoption, but even in that area the opposite effect was also observed.^[39]

Institutions often play a critical role in conservation decisions. A key institutional factor is the security of rights to land. Given the long-term nature of conservation investments, incentives to undertake them will be lower, if farmers fear losing their land.^[40,41] Lack of titles is not, however, always synonymous to tenure insecurity. A substantial literature demonstrates that many traditional African land tenure systems do not result in tenure insecurity, even in the absence of titles.^[42] Institutions can also play other roles. In Kitui/Machakos, women’s labor exchange groups (mwethya) substituted for missing credit markets and allowed farmers to undertake costly conservation measures.^[27]

When the effects of soil degradation are felt primarily off-site, the situation is very different. Because these problems affect others rather than themselves, farmers have no incentive to address them. Many soil conservation projects have been motivated by the desire to protect downstream infrastructure, such as dams, from the effects of upstream degradation. Because the practices promoted by these projects are not necessarily profitable from the farmers’ perspective, their success rate has been very low.^[43] Subsidies and/or compulsion can sometimes lead to temporary adoption of the measures being promoted, but they tend to be rapidly abandoned once the project is completed.^[28,44]

CONCLUSION

Land degradation problems seem unlikely to cause the catastrophic agricultural declines as some fear, but they can

have acute consequences at the local level, particularly for poor farmers. Farmers have strong incentives to respond to the on-site effects of soil degradation, and the evidence shows they often do so. The changing understanding of farmers' incentives to undertake appropriate soil-management practices has shifted the emphasis of interventions away from top-down projects focusing on technical solutions. Overall policy reform is seen as a necessary precondition for any intervention. Where on-site effects dominate, participative projects, in which responses appropriate to local conditions are developed in collaboration with the affected land users, have come to play an important role. Where off-site effects dominate, efforts have turned to the development of systems of payments for environmental services.

REFERENCES

1. Eckholm, E. *Losing Ground: Environmental Stress and World Food Prospects*; W.W. Norton: New York, 1976; 223.
2. Brown, L.R.; Wolf, E.C. *Soil Erosion: Quiet Crisis in the World Economy*; Worldwatch Paper No. 60; Worldwatch Institute: Washington, 1984.
3. Pimentel, D.; Harvey, C.; Resosudarmo, P.; Sinclair, K.; Kurz, D.; McNair, M.; Crist, S.; Shripter, L.; Fitton, L.; Saffouri, R.; Blair, R. Environmental and economic costs of soil erosion and conservation benefits. *Science* **1995**, *267*, 1117–1123.
4. Scherr, S.J. *Soil Degradation: A Threat to Developing Country Food Security by 2020*; Food, Agriculture and Environment Discussion Paper No. 27; IFPRI: Washington, 1999.
5. Lal, R.C. Effects of erosion on soil productivity. *CRC Crit. Rev. Plant Sci.* **1987**, *5* (4), 303–367.
6. Hamilton, L.S.; King, P.N. *Tropical Forest Watersheds: Hydrologic and Soils Response to Major Uses and Conversions*; Westview Press: Boulder, 1983.
7. Chomitz, K.; Kumari, K. The domestic benefits of tropical forests: A critical review. *World Bank Res. Observer* **1998**, *13* (1), 13–35.
8. Pagiola, S. *The Global Environmental Benefits of Land Degradation Control on Agricultural Land*; Environment Paper No. 16; World Bank: Washington, 1999.
9. Lal, R.; Logan, T.J. Agricultural activities and greenhouse gas emissions from soils in the tropics. In *Soil Management and Greenhouse Effect: Advances in Soil Science*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1995; 293–307.
10. Magrath, W.B.; Arens, P. *The Cost of Soil Erosion on Java: A Natural Resource Accounting Approach*; Environment Department Working Paper No. 18; World Bank: Washington, 1989.
11. Asian Productivity Organization (APO). *Soil Conservation and Watershed Protection in Asia and the Pacific*; Asian Productivity Organization: Tokyo, 1995.
12. Walling, D.E. Measuring sediment yield from river basins. In *Soil Erosion Research Methods*; Lal, R., Ed.; Soil and Water Conservation Society: Ankeny, 1988; 39–73.
13. Bojő, J. The costs of land degradation in sub-Saharan Africa. *Ecol. Econ.* **1996**, *16* (2), 161–173.
14. Food and Agriculture Organization (FAO). *Highlands Reclamation Study: Ethiopia*; FAO: Rome, 1986.
15. Bojő, J.; Cassells, D. *Land Degradation and Rehabilitation in Ethiopia: A Reassessment*; AFTES Working Paper No. 17; World Bank: Washington, 1995.
16. Sutcliffe, J.P. *Economic Assessment of Land Degradation in the Ethiopian Highlands: A Case Study*; National Conservation Strategy Secretariat, Ministry of Planning and Economic Development: Addis Ababa, 1993.
17. Pagiola, S.; Dixon, J.A. Land degradation problems in El Salvador. In *El Salvador: Rural Development Study*; Report No. 16253-ES; World Bank: Washington, 1997.
18. Schultz, T.W. *Transforming Traditional Agriculture*; University of Chicago Press: Chicago, 1964.
19. Popkin, S.L. *The Rational Peasant*; University of California Press: Berkeley, 1979.
20. Timmer, C.P.; Falcon, W.P.; Pearson, S.R. *Food Policy Analysis*; Johns Hopkins University Press for the World Bank: Baltimore, 1983.
21. Forsyth, T. The use of cesium-137 measurements of soil erosion and farmers' perceptions to indicate land degradation amongst shifting cultivators in Northern Thailand. *Mountain Res. Dev.* **1994**, *14* (3), 229–244.
22. Kiome, R.M.; Stocking, M. Rationality of farmer perception of soil erosion. *Glob. Environ. Change* **1995**, *5* (4), 281–295.
23. Pender, J.; Kerr, J. *Determinants of Farmers' Indigenous Soil and Water Conservation Investments in the India's Semi-Arid Tropics*; EPTD Discussion Paper No. 17; IFPRI: Washington, 1996.
24. Reij, C. Building on tradition: the improvement of indigenous soil and water conservation techniques in the West African Sahel. In *Adopting Conservation on the Farm*; Napier, T.L., Camboni, S.M., El-Swaify, S.A., Eds.; Soil and Water Conservation Society: Ankeny, 1994; 143–156.
25. Ryder, R. Farmer perception of soils in the mountains of the Dominican Republic. *Mountain Res. Develop.* **1996**, *14* (3), 261–266.
26. Tiffen, M.; Mortimore, M.; Gichuki, F. *More People, Less Erosion: Environmental Recovery in Kenya*; John Wiley: Chichester, 1994.
27. Pagiola, S. Soil conservation in a semi-arid region of Kenya: Rates of return and adoption by farmers. In *Adopting Conservation on the Farm*; Napier, T.L., Camboni, S.M., El-Swaify, S.A., Eds.; Soil and Water Conservation Society: Ankeny, 1994; 171–188.
28. Lutz, E.; Pagiola, S.; Reiche, C. Cost-benefit analysis of soil conservation: The farmers' viewpoint. *The World Bank Res. Observer* **1994**, *9* (2), 273–295.
29. Bojő, J. *The Economics of Land Degradation: Theory and Applications to Lesotho*; Stockholm School of Economics: Stockholm, 1991.
30. Shively, G.E. Consumption risk, farm characteristics, and soil conservation adoption among low-income farmers in the Philippines. *Agr. Econ.* **1997**, *17*, 165–177.
31. Walker, D.J. A damage function to evaluate erosion control economics. *Am. J. Agr. Econ.* **1982**, *64* (4), 690–698.
32. Pagiola, S.; Bendaoud, M. *Long-Run Economic Effects of Erosion on Wheat Production in a Semi-Arid Region of*

- Morocco: A Simulation Analysis*; Agricultural Economics Staff Paper No. AE 95-12; Washington State University: Pullman, 1994.
33. Holden, S.T.; Shiferaw, B.; Wik, M. Poverty, market imperfections and time preferences: Of relevance for environmental policy? *Environ. Develop. Econ.* **1998**, *3* (1), 105–130.
 34. Reardon, T.; Vosti, S.A. Poverty-environment links in rural areas of developing countries. In *Sustainability, Growth, and Poverty Alleviation: A Policy and Agroecological Perspective*; Vosti, S.A., Reardon, T., Eds.; Johns Hopkins University Press for IFPRI: Baltimore, 1997; 135–145.
 35. Pagiola, S. *The Effect of Subsistence Requirements on Sustainable Land Use Practices*, Annual Meetings of the American Agricultural Economics Association, Indianapolis, IN, Aug 6–9, 1995.
 36. Schiff, M.; Valdès, A. *The Political Economy of Agricultural Pricing Policy*; Johns Hopkins University Press: Baltimore, 1992.
 37. Heath, J.; Binswanger, H. Natural resource degradation effects of poverty and population growth are largely policy-induced: the case of Colombia. *Environ. Develop. Econ.* **1996**, *1* (1), 65–83.
 38. LaFrance, J.T. Do increased commodity prices lead to more or less soil degradation? *Aust. J. Agr. Econ.* **1992**, *36*, 57–82.
 39. Pagiola, S. Price policy and returns to soil conservation in semi-arid Kenya. *Environ. Resource Econ.* **1996**, *8*, 255–271.
 40. Ervin, D.E. Constraints to practicing soil conservation: Land tenure relationships. In *Conserving Soil: Insights from Socioeconomic Research*; Lovejoy, S.B., Napier, T.L., Eds.; Soil and Water Conservation Society: Ankeny, 1986; 36–62.
 41. Wachter, D. Land titling: possible contributions to farmland conservation in Central America. In *Economic and Institutional Analyses of Soil Conservation Projects in Central America and the Caribbean*; Lutz, E., Pagiola, S., Reiche, C., Eds.; Environment Paper No. 8; World Bank: Washington, 1994; 150–157.
 42. Place, F.; Hazell, P. Productivity effects of indigenous land tenure systems in sub-Saharan Africa. *Am. J. Agr. Econ.* **1993**, *75* (1), 10–19.
 43. Pagiola, S. Economic analysis of incentives for soil conservation. In *Using Incentives for Soil Conservation: From Theory to Practice*; Sanders, D.W., Huszar, P.C., Sombatpanit, S., Enters, T., Eds.; Science Publishers: Enfield, 1999; 41–56.
 44. Enters, T. The token line: Adoption and non-adoption of soil conservation practices in the highlands of Northern Thailand. In *Soil Conservation Extension: From Concepts to Adoption*; Sombatpanit, S., Zöbisch, M.A., Sanders, D.W., Cook, M.G., Eds.; Science Publishers: Enfield, 1997; 417–427.

Diagnostic Horizons

E.M. Bridges

*International Soil Reference and Information Centre (ISRIC), Wageningen,
the Netherlands*

Abstract

The use of diagnostic horizons, focused on observable and measurable soil morphology, rather than ill-defined genetic processes, helped greatly in reaching agreement for the World Reference Base as a single international system of soil classification. In both theory and practice, it has been one of the most useful developments in soil classification.

INTRODUCTION

In layman's terms, the concept of topsoil and subsoil is widely understood, but pedologists have built upon this idea and given it a scientific basis, using the term soil horizons for these layers of soil. A soil horizon may be defined as a layer of soil lying approximately parallel to the earth's surface, having pedological characteristics. A soil profile includes the collection of soil horizons revealed in a section of the soil downward from the surface. As soil horizons develop by means of the processes of soil formation acting upon weathered geological materials, they are often referred to as genetic horizons. The definition of these genetic horizons was effectively qualitative in approach, reflecting a perceived means of genesis. Toward the end of the 19th century and throughout the first part of the 20th century, various systems of nomenclature arose in different countries allocating letters O, A, B, C, etc. to genetic horizons, but their use was not uniformly applied and so they became discredited in the eyes of some authorities. Such shorthand designations are "an interpretive symbol," based upon horizon morphology and implied genesis that are used to identify and label a soil horizon.^[1] In the United States, as demonstrated by Soil Survey Staff,^[2,3] certain organic O horizons and mineral A, B, and C horizons were designated master horizons and were in vogue for the description of soil profiles. A similar system was developed in the then Soviet Union as described in *Pochvennaya Syemka*,^[4] and many other countries followed their example. Although the ABC system of labeling worked reasonably well in the areas where it developed, e.g., Russia and the United States, soil scientists working in tropical areas, where soils had a longer and more complex history of development, found it difficult to use,^[5,6] who strived to develop systems of their own. Prior to the 1960s, the presence of certain horizons was essential for the classification of soils; Podzols, e.g., had to possess a bleached horizon and below it a horizon of accumulation of organic matter, iron, and aluminum, and gley soils had to have the features of hydromorphism. As

the definitions of soil horizons in the early 20th century were cast in general, rather than in specific, quantitative terms, there were overlaps and gaps that caused confusion in the recognition of taxonomic units employed in soil classification systems.

The idea of diagnostic horizons first appeared in the 6th Approximation of 1957,^[7] when the earlier concept of soil horizons was taken and developed into a basis for the quantitative identification of horizons and their unambiguous incorporation into systems of soil classification.

DIAGNOSTIC HORIZONS IN THE U.S. DEPARTMENT OF AGRICULTURE SOIL TAXONOMY

From 1951, soil scientists in the United States began to develop a new approach to soil classification and after a number of preliminary drafts, the 7th Approximation was published as a discussion document. In this approach to soil classification, named diagnostic horizons were selected as a key to the classification system. Originally, 6 surface diagnostic horizons, called epipedons, and 12 subsurface diagnostic horizons were proposed. Although the conceptual basis of what was a diagnostic horizon or how it was chosen were not defined or described, it amounted to a soil horizon or a group of closely related horizons that possess a set of quantitatively defined properties. By fulfilling their criteria, and with the use of a key, soils may be placed in their correct position in one of the 10 orders of the system.

The quantitative definition of these diagnostic horizons is such that they cannot be related in all cases to the master horizons of other systems of genetic horizon designation. In discussion about the development of the Soil Taxonomy system, Smith^[7] explained that the concept of master horizons and the ABC system was not used in Soil Taxonomy because of a lack of agreement about its use among pedologists. Therefore, he adopted the diagnostic horizon concept, which did not carry with it any inherited controversy.

Table 1 Diagnostic horizons of the USDA 7th Approximation.

Epipedons	Subsurface horizons	Other horizons
Mollic	Agric, albic, anthropic, and argillic	Calcic
Umbric	Cambic	Gypsic
Histic	Natric	Salic
Ochric	Oxic	Pans: duripan and fragipan
Plaggen	Spodic	

Source: From Soil Survey Staff.^[8]

The diagnostic horizons proposed in the 7th Approximation are presented in [Table 1](#).

Since its publication in 1975, Soil Taxonomy has been continually updated and improved through correlation meetings and advice received from soil scientists around the world. As a result, two new epipedons have been introduced: a melanic epipedon to provide for the surface horizons of soils on tephra and volcanic glass and a folistic epipedon to provide for organic soil materials, saturated for more than 30 days and consisting of more than 75% sphagnum. Six of the subsurface diagnostic horizons that first appeared in the 7th Approximation remain with improved definition in Soil Taxonomy. These are the agric, argillic, cambic, natric, oxic, and spodic horizons, together with horizons referred to as pans: duripan and fragipan. “Other horizons” that have persisted throughout the development of Soil Taxonomy are the calcic, gypsic, and salic horizons. New horizons that have been introduced are the glossic (to provide for degraded argillic), kandic, and natric horizons; the kandic has been introduced to provide for a more clayey subsurface horizon with an apparent cation exchangeable capacity of 16 cmol or less per kg of clay; orstein has been separated from the spodic horizon, and placic has been introduced for thin iron pans. Petrocalcic and petrogypsic have been introduced for carbonate and gypsic-cemented horizons, respectively. Sombric caters for dark-colored, freely drained subsurface horizons with illuvial humus. Additionally, there is a sulfuric horizon with a low pH and containing jarosite. Other diagnostic soil characteristics, e.g., slickensides and lamellae, and soil conditions, e.g., andic soil properties, are strictly defined. Full definitions of all these horizons and characteristics are given in Soil Taxonomy,^[9] and summaries are provided by Tiurin et al.^[10] and in various textbooks. Since the development of Soil Taxonomy, the concept of diagnostic horizons has been widely used as the basis of several national systems of soil classification (e.g., Brazil and South Africa) and in the legend for Food and Agriculture Organization (FAO).^[11]

REFERENCE HORIZONS OF FITZPATRICK

In parallel with developments in the United States, FitzPatrick,^[12–14] independently proposed a system of

Table 2 Reference horizons proposed by FitzPatrick.

Alkalon	Fermenton	Kastanon	Proxon
Alton	Ferron	Krasnon	Pseudofibron
Anmorphon	Fibron	Kuron	Rosson
Andon	Flambon	Limon	Rubon
Anmooron	Flavon	Lithon	Rufon
Arenon	Fragon	Litter	Sapron
Argillon	Gelon	Luton	Seron
Buron	Gleyson	Luvon	Sesquon
Calcon	Glosson	Marblon	Sideron
Candon	Gluton	Minon	Solon
Celon	Gypson	Modon	Sulphon
Cerulon	Gytjon	Mullon	Tannon
Chernon	Halon	Oron	Thion
Chloron	Hamadon	Pallon	Verton
Clamon	Hudepon	Pelon	Veson
Crumon	Humifon	Pesson	Zhelton
Cryon	Husesquon	Placon	Zolon
Cumulon	Hydromoron	Planon	
Dermion	Ison	Primon	

Source: From FitzPatrick.^[12]

classification with a greater number of defined horizons, called reference horizons. The soil is viewed as a 3-D continuum, whereas a soil profile is in reality only a 2-D section through the soil. As the soil reflects the conditions in which it was formed, most importantly the effects of climate, parent material, and relief, a pattern of soils occurs over the landscape that is 3-D. The boundaries between soils are rarely abrupt, but are zones of gradual change. Thus, it is necessary to consider soils as 3-D entities that vary in space and time. Within this 3-D system, both conceptual space and observed field characteristics are accommodated in reference segments where there is a single unique dominating property or combination of properties formed principally by a single set of processes. Intergrade segments contain properties that gradually change between two reference segments. Within each segment, it is possible to recognize reference horizons, intergrade horizons, compound horizons, and composite horizons. The reference horizons proposed are presented in [Table 2](#).

THE FAO–UNITED NATIONS EDUCATIONAL, SCIENTIFIC AND CULTURAL ORGANIZATION (UNESCO) SOIL MAP OF THE WORLD LEGEND

The FAO–UNESCO Soil Map of the World was completed during the decade 1971–1981^[11,15] with the aim of making a first scientific appraisal of the soil resources of the world, providing a basis for transfer of knowledge, and promoting a generally accepted system of soil classification, especially

Table 3 Diagnostic horizons used in the legends of the Soil Map of the World.

Histic H horizon	Albic E horizon	Calcic horizon
Mollic A horizon	Argillic B horizon	Gypsic horizon
Umbric A horizon	Cambic B horizon	Sulfuric horizon
Ochric A horizon	Natric B horizon	
Spodic B horizon		
Oxic B horizon		

Source: From FAO.^[11]

for educational, research, and development activities. Following the use of diagnostic horizons in Soil Taxonomy, a similar limited number of horizons with measurable and observable properties were employed by the editors of the 1974 legend of the Soil Map of the World (Table 3).

Reflecting the increasing impact of human activity in modifying, in particular, the A horizon of soils, a fimic A horizon was added in the 1988 revision to provide for horizons with a man-made surface layer with a thickness of 50 cm or more, produced by long continued manuring with earthy mixtures. It commonly contains artifacts such as brick and pottery and includes the plaggen epipedon and the anthropic epipedon of the Soil Taxonomy system. The difficulties, which had been experienced in the field, especially in tropical regions, where oxic and argillic horizons were difficult to separate, resulted in a redefinition into the argic and the ferrallic B horizons, respectively. The petrocalcic and petrogypsic phases became diagnostic horizons and other definitions were amended, but calcic, gypsic, histic A, mollic A, ochric A, spodic B, sulfuric, and umbric A horizon definitions were unchanged. As with the system of Soil Taxonomy, a number of diagnostic properties, which are not considered as horizons, are also used in the definitions of the diagnostic horizons. These refer to andic soil material and ferralitic, ferric, fluvic, hydromorphic, nitric, and vertic properties. With their sets of quantitatively defined properties, produced by the soil-forming processes, the use of diagnostic horizons has made it possible to base a classification on general principles of soil genesis but objectivity was retained because the processes themselves are not used as criteria, only their effects expressed in terms of morphological properties, which have measurable values.

THE WORLD REFERENCE BASE (WRB) OF THE INTERNATIONAL UNION OF SOIL SCIENCES

Within the international community, it has long been appreciated that there has not been a generally accepted system of soil classification. In 1982, the International Society of Soil Science initiated the program to develop an international reference base for soils, but it languished until 1992. In that year, the decision was taken to make a WRB by

developing the revised (1988) FAO–UNESCO legend for the Soil Map of the World.^[16] In this way, advantage could be taken of the international correlation work that had taken place since the Soil Map of the World had been published.

The objectives of WRB were to develop the FAO–UNESCO map legend into a comprehensive, internationally acceptable system for delineating soil resources and to provide a sound scientific basis to support it. The WRB is based upon morphological characterization of soils and is supported by laboratory analyses where necessary. Its authors claim that it provides an easy means of communication about soils, including the transfer of data and technology relevant to soils both for soil scientists and for related environmental fields. As with other soil classification systems, it purports to show the relationships within and between the different soils of the world.

WRB adopted the use of diagnostic horizons as reflecting the results of widely recognized processes of soil development and improved the FAO definitions in terms of morphological characteristics and/or analytical criteria. Of the 16 diagnostic horizons of the 1988 FAO–UNESCO revised legend, the fimic A horizon has been discarded and replaced by hortic, plaggic, and terric horizons, respectively. The histic horizon has been broadened as its minimum thickness has been reduced to 10 cm and has been used to define Histosols. Experience with Chinese soils has shown that the former P₂O₅ requirement for the mollic and umbric A horizons of the revised FAO–UNESCO system is untenable as a criterion. The unique properties of Chernozems were not effectively reflected in the definition of the mollic A horizon, and so a chernic horizon has been introduced for the deep, blackish, porous surface horizons typical of Chernozems. Increasing the percentage of clay skins from 1% to 5% on both horizontal and vertical ped faces and pores is expected to give an improved correlation between field and micromorphological evidence.

The cambic horizon has always suffered from being a negative concept, without many pedological criteria. Its positive points such as depth, texture, structure, color, and the presence of weatherable minerals have given it a more positive image. In the case of the spodic horizon, it has been supplemented with the 1996 developments in Soil Taxonomy. In addition to the diagnostic horizons mentioned, additional 19 new horizons have been proposed. Full list is given in Table 4 and in the WRB for Soil Resources.^[17,18]

The adoption of the concept of diagnostic horizons by U.S. Soil Survey Staff has led to their acceptance in systems of soil classification in many countries of the world. Subsequent refinements and work for the WRB indicate the necessity for a greater number of diagnostic horizons to cover the nature and range of soil morphological features.

Table 4 Diagnostic horizons of the WRB for Soil Resources.

Albic	Ferric	Mollic	Salic
Andic	Folic	Natric	Spodic
Anthraquic	Fragic	Nitric	Sulfuric
Argic	Fluvic	Ochric	Takyric
Calcic	Gypsic	Petrocalcic	Terric
Cambic	Histic	Petroduric	Umbric
Chernic	Hydragric	Petrogypsic	Vertic
Cryic	Hortic	Petroplinthic	Vitric
Duric	Irragric	Plaggic	Yermic
Ferralic	Melanic	Plinthic	

Source: From FAO^[17] and ISSS Working Group RB.^[18]

The use of diagnostic horizons, focused upon observable and measurable soil morphology, rather than ill-defined genetic processes, helped greatly in reaching agreement for the WRB as a single international system of soil classification. In the revised legend for the Soil Map of the World,^[16] it is stated that the use of diagnostic horizons has proved to be most appropriate. In both theory and practice, it has been one of the most useful developments in soil classification.

REFERENCES

1. Bridges, E.M. *Soil Horizon Designations*; Technical Paper 19; International Soil Reference and Information Centre: Wageningen, 1990.
2. Soil Survey Staff. *Soil Survey Manual*; Agriculture Handbook 18; USDA: Washington, 1951.
3. Soil Survey Staff. *Supplement to Soil Survey Manual*; Agriculture Handbook 18; USDA: Washington, 1962; 173–188.
4. Tiurin, L.V.; Gerasimov, I.P.; Ivanova, E.N.; Nosin, V.A. *Pochvennaya Syemka*; Akademia Nauk: Moscow, 1959.
5. Nye, P.H. Some soil-forming processes in the humid tropics. *J. Soil Sci.* **1954**, *5*, 7–21.
6. Watson, J.P. A soil catena on granite in southern Rhodesia. 1. Field observations. *J. Soil Sci.* **1964**, *15*, 238–250.
7. Smith, G.D. *The Guy Smith Interviews: Rationale for Concepts in Soil Taxonomy*; Technical Monograph No. 11; Soil Management Support Services, Soil Conservation Service, USDA/Agronomy Department, Cornell University: Washington, 1986.
8. Soil Survey Staff. *Soil Classification: A Comprehensive System (7th Approximation)*; Soil Conservation Service; USDA: Washington, 1960.
9. Soil Survey Staff. *Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; Agriculture Handbook 436; USDA–Natural Resources Conservation Service: Washington, 1999.
10. Soil Survey Staff. *Keys to Soil Taxonomy*; USDA–Natural Resources Conservation Service: Washington, 1998.
11. FAO. *FAO–UNESCO Soil Map of the World, 1:5,000,000, Legend*; UNESCO: Paris, 1974; Vol. I.
12. FitzPatrick, E.A. Soil nomenclature and classification. *Geoderma* **1967**, *1*, 91–105.
13. Fitzpatrick, E.A. *Soil Horizon Designation and Classification*; Technical Paper 17; International Soil Reference and Information Centre: Wageningen, 1988.
14. FitzPatrick, E.A. *Soils: Their Formation, Classification and Distribution*; Longman: London and New York, 1980.
15. FAO. *FAO–UNESCO Soil Map of the World, 1:5,000,000*; UNESCO: Paris, 1971–1981; Vols. 2–10.
16. FAO. *FAO–UNESCO Soil Map of the World, 1:5,000,000, Revised Legend*; World Soil Resources Report No. 60; FAO: Rome, 1988.
17. FAO. *World Reference Base for Soil Resources*; World Soil Resources Report No. 84; FAO: Rome, 1998.
18. ISSS Working Group RB. *World Reference Base for Soil Resources*; Vol. 1. Introduction (Deckers, J.A., Nachtergaele, F.O., Spaargaren, O.C. Eds.); Vol. 2. Atlas (Bridges, E.M., Batjes, N.H., Nachtergaele, F.O., Eds.). ISRIC/FAO, ACCO: Leuven, 1998.

Diffusivity: Measurement

Toru Nakajima

Department of Agriculture, Meiji University, Kawasaki, Japan

Abstract

Soil gas diffusion is important for understanding many processes in the interchange of gases between soil and the atmosphere. The Currie method, using a soil core sample, is the most widely used method for determining the soil gas diffusion coefficient (D_p/D_0). The choice of the method depends on available apparatus, cost, precision, and advantages of the methods. Modeling transport of gas movement in soil requires knowledge of the gaseous diffusion coefficient. Numerous models of soil gas diffusion using soil properties have been proposed. Empirical models vary for types of soil materials and ranges of soil air content used in their development.

INTRODUCTION

Diffusion of gases [e.g., oxygen (O_2), carbon dioxide, methane, and nitrous oxide] through the soil has far-reaching implications on soil biota and gaseous emissions. For instance, plant roots and soil microbes need O_2 for respiration which can be quickly depleted in the soil and must be replenished with O_2 from the atmosphere. Gaseous movement in soils is mainly driven by the diffusion processes associated with differences in gas concentration gradients,^[1] soil physical properties (e.g., soil texture, clay mineral, macro and micro pores, bulk density, total porosity, aggregation, and pore size distribution), and moisture conditions.^[2] Further, gaseous diffusion depends on the continuity of soil porous media through the gas exchange process, more directly than any other soil physical properties.^[3] Therefore, the principle of gaseous movement in soils must be well understood to optimize the potential of agricultural practices and to reduce emissions of greenhouse gases. This entry describes a simple laboratory method using a soil core sample for measuring soil gas diffusion coefficients.

PRINCIPLES AND EQUATIONS

Molecular gas diffusion in soils can be described by Fick's law for steady state:^[1]

$$F_g = -D_p \frac{\partial^2 C_g}{\partial x^2} \quad (1)$$

where F_g is the gas flux density ($g\ m^{-2}\ s^{-1}$), D_p is the soil gas diffusion coefficient ($m^2\ s^{-1}$), C_g is the gas concentration in the gaseous phase ($g\ m^{-3}$), and x is the distance (m soil). For non-steady-state condition, gas diffusion in soil can be described by combining the continuity equation (conservation of mass) with Fick's first law (Eq. 1).^[1]

$$\epsilon \frac{\partial C_g}{\partial t} = D_p \frac{\partial^2 C_g}{\partial x^2} \quad (2)$$

where ϵ is the soil air content ($m^3\ m^{-3}$) and t is the time (s). Because D_p is significantly affected by temperature and atmospheric pressure, the relative gas diffusion coefficient (D_p/D_0) defined as the ratio of the soil gas diffusion coefficient to that in free air is used. The D_0 is the gas diffusion coefficient in free air.

MEASUREMENT OF SOIL GAS DIFFUSION COEFFICIENT

Gaseous diffusion in soil under laboratory conditions is measured by the Currie method^[4,5] (Fig. 1). The device for this method consists of two main parts: the diffusion chamber and soil sample holder. The diffusion chamber has an O_2 concentration sensor and inlet and outlet ports for injecting nitrogen (N_2) gas in and venting the chamber gas out. The O_2 concentration is measured by the O_2 sensor (e.g., Model KE50; Figaro USA Inc., Illinois, U.S.A.) and is recorded with a data logger (e.g., Model DI-710; DATAQ Instruments Inc., Ohio, U.S.A.) for duration of the soil gas diffusion measurement. The soil sample holder is located above the diffusion chamber with the bottom of the soil sample facing the diffusion chamber. In order to achieve an airtight seal between the sample and the diffusion chamber, O-ring rubber and silicon oil are used. The diffusion chamber is initially flushed with 100% N_2 , and the O_2 concentration is measured in the diffusion chamber in order to monitor the intake rate of atmospheric O_2 through the soil sample. The time rate of O_2 change of concentration in the chamber is determined and related to the soil gas diffusion coefficient. In general, the tracer gases for soil gas diffusion measurements are N_2 and O_2 because of ease to obtain and treat, and the possibility to be easily detected by sensor. However, when O_2 is used for experiment, the experimental time should be

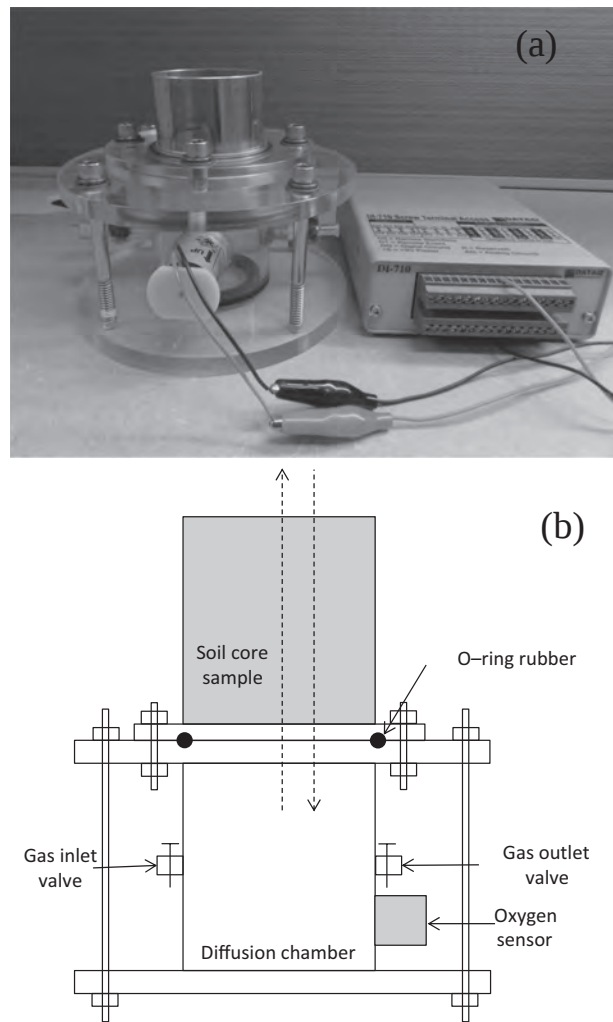


Fig. 1 (a) Photograph and schematic diagram of the device and (b) design used for measuring the soil gas diffusion coefficient of a soil core sample.

preferably short to minimize O₂ consumption by soil respiration. The choice of the method must depend on available apparatus, cost, precision, and advantages of the methods related to the goal of the study.

PREDICTIVE EQUATIONS FOR SOIL GAS DIFFUSION COEFFICIENT

Since measurement methods are time consuming, costly, and often not available, many gas diffusion empirical models have been developed for predicting gaseous diffusion. To predict soil gas diffusion coefficient from soil properties (e.g., soil air content, total porosity, and water retention parameter), several models are proposed (Table 1). One of the earliest attempts to describe the soil gas diffusion was by Buckingham.^[6] Numerous studies were conducted on model development during the first half of the 20th century,^[5,7,8] and a considerable progress was made.^[11,14,15] Models outlined in Table 1 vary in their approach and for use on type of soil materials and range of soil air content used for their development.^[16] Several models listed in Table 1 are widely used in numerical simulations; however, they are sensitive to error at different levels in the input parameter. In addition, there is no obvious *a priori* best relationship for a soil. Comparison of data obtained by different models is shown in Fig. 2.

CONCLUSION

This entry describes a Currie method using a soil core sample for measuring soil gas diffusion coefficients (D_p/D_0). The choices of the method depend on available apparatus, cost, precision, and advantages of the methods. Numerous models of soil gas diffusion using soil properties have been proposed. In this entry, the comparisons of relative soil gas diffusion coefficient by different models are

Table 1 Predictive equations for the soil gas diffusion coefficient.

Model	References	Comments
$\frac{D_p}{D_0} = \epsilon^2$	Buckingham ^[6]	
$\frac{D_p}{D_0} = 0.66\epsilon$	Penman ^[7]	
$\frac{D_p}{D_0} = \alpha\epsilon^\beta$	Currie ^[5]	α, β refer to pore shape, diameter, continuity, and tortuosity.
$\frac{D_p}{D_0} = \frac{\epsilon^2}{\phi^{2/3}}$	Millington & Quirk ^[8] and Jin & Jury ^[9]	Good for different textures of repacked soil sample
$\frac{D_p}{D_0} = \phi^2 \left(\frac{\epsilon}{\phi} \right)^{2+3/b}$	Campbell ^[10]	b is the Campbell pore-size distribution parameter
$\frac{D_p}{D_0} = (2\epsilon_{100}^3 + 0.04\epsilon_{100}) \left(\frac{\epsilon}{\epsilon_{100}} \right)^{2+3/b}$	Moldrup et al., ^[11] Clapp & Hornberger, ^[12] and Olesen et al. ^[13]	b can be described by following equation: $b = 13.6 \times (\text{clay fraction}) + 3.5$
$\frac{D_p}{D_0} = \epsilon^{1.5} \left(\frac{\epsilon}{\phi} \right)$	Moldrup et al. ^[11]	It is good for sieved and repacked soil sample.

Note: ϵ = the soil air content ($\text{m}^3 \text{ m}^{-3}$); ϕ = the total porosity ($\text{m}^3 \text{ m}^{-3}$).

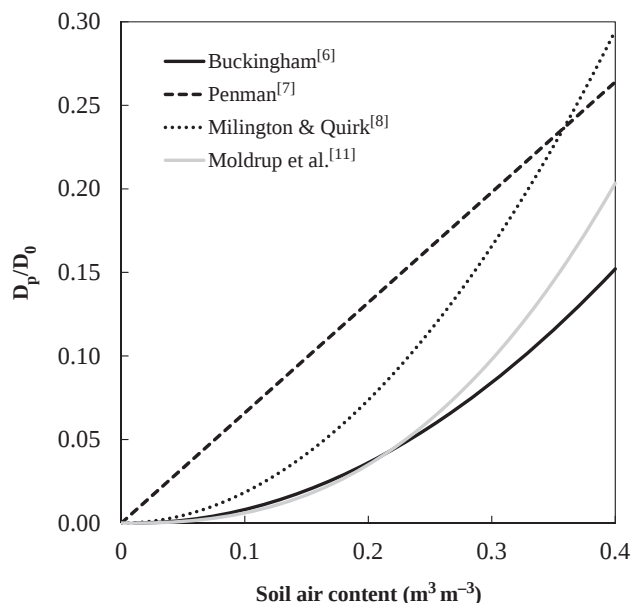


Fig. 2 Comparison of relative soil gas diffusion coefficient by different models. The total porosity is $0.4 \text{ (m}^3 \text{ m}^{-3}\text{)}$, b (pore size distribution parameter) is 5.54, and ϵ_{100} is $0.1 \text{ (m}^3 \text{ m}^{-3}\text{)}$.

shown. Empirical models vary for type of soil materials and range of soil air content.

REFERENCES

1. Rolston, D.E.; Moldrup, P. 4.3 Gas diffusivity. In *Methods of Soil Analysis: Part 4 Physical Methods*; Dane, J.H., Topp, G., Eds.; Soil Science Society of America: Madison, 2002; 1113–1139. DOI:10.2136/sssabookser5.4.c45.
2. Moldrup, P.; Olesen, T.; Komatsu, T.; Schjønning, P.; Rolston, D.E. Tortuosity, diffusivity, and permeability in the soil liquid and gaseous phases. *Soil Sci. Soc. Am. J.* **2001**, *65* (3), 613–623.
3. Ball, B.C.; Dobbie, K.E.; Parker, J.P.; Smith, K.A. The influence of gas transport and porosity on methane oxidation in soils. *J. Geophys. Res.-Atmos.* **1997**, *102* (D19), 23301–23308. DOI:10.1029/97jd00870.
4. Currie, J.A. Gaseous diffusion in porous media Part 1—A non-steady state method. *Brit. J. Appl. Phys.* **1960**, *11* (8), 314.
5. Currie, J.A. Gaseous diffusion in porous media. Part 2—Dry granular materials. *Brit. J. Appl. Phys.* **1960**, *11* (8), 318.
6. Buckingham, E. *Contributions to Our Knowledge of the Aeration of Soils*; U.S. Dept. of Agriculture, Bureau of Soils: Washington, DC, 1904; 60 pp.
7. Penman, H.L. Gas and vapour movements in the soil: I. The diffusion of vapours through porous solids. *J. Agr. Sci.* **1940**, *30* (3), 437–462. DOI:10.1017/S0021859600048164.
8. Millington, R.J.; Quirk, J.P. Permeability of porous solids. *T. Faraday Soc.* **1961**, *57*, 1200–1207. DOI:10.1039/TF9615701200.
9. Jin, Y.; Jury, W.A. Characterizing the dependence of gas diffusion coefficient on soil properties. *Soil Sci. Soc. Am. J.* **1996**, *60* (1), 66–71.
10. Campbell, G.S. A Simple method for determining unsaturated conductivity from moisture retention data. *Soil Sci.* **1974**, *117* (6), 311–314.
11. Moldrup, P.; Olesen, T.; Gamst, J.; Schjønning, P.; Yamaguchi, T.; Rolston, D.E. Predicting the gas diffusion coefficient in repacked soil water-induced linear reduction model. *Soil Sci. Soc. Am. J.* **2000**, *64* (5), 1588–1594. DOI: 10.2136/sssaj2000.6451588x.
12. Clapp, R.B.; Hornberger, G.H. Empirical equations for some soil hydraulic properties. *Water Resour. Res.* **1978**, *14* (4), 601–604. DOI:10.1029/WR014i004p00601.
13. Olesen, T.; Moldrup, P.; Henriksen, K.; Petersen, L.W. Modeling diffusion and reaction in soils: IV. New models for predicting ion diffusivity. *Soil Sci.* **1996**, *161* (10), 633–645.
14. Moldrup, P.; Olesen, T.; Yoshikawa, S.; Komatsu, T.; Rolston, D.E. Three-porosity model for predicting the gas diffusion coefficient in undisturbed soil. *Soil Sci. Soc. Am. J.* **2004**, *68* (3), 750–759.
15. Hamamoto, S.; Moldrup, P.; Kawamoto, K.; Komatsu, T. Organic matter fraction dependent model for predicting the gas diffusion coefficient in variably saturated soils. *Vadose Zone J.* **2012**, *11* (1), DOI:10.2136/vzj2011.0065.
16. Allaire, S.E.; Lafond, J.A.; Cabral, A.R.; Lange, S.F. Measurement of gas diffusion through soils: Comparison of laboratory methods. *J. Environ. Monitor.* **2008**, *10* (11), 1326–1336. DOI:10.1039/b809461f.

Digital Soil Mapping

Budiman Minasny

Alex B. McBratney

*Department of Environmental Sciences, University of Sydney, Sydney,
New South Wales, Australia*

Florence Carré

*Land Management and Natural Hazards Unit, Institute for Environment
and Sustainability, Ispra, Italy*

Abstract

Digital soil mapping is the creation of a spatial soil information system using field and laboratory observation methods coupled with quantitative spatial prediction techniques. Digital soil mapping follows the advancement in soil and environmental observations using proximal and remote sensing. It also utilizes contemporary mathematical and statistical techniques that allow better prediction of soil properties in areas with little or no information as well as indicating the uncertainty of such predictions.

INTRODUCTION

Digital soil mapping is defined as the creation and the population of geographically referenced soil databases generated at a given spatial resolution using field and laboratory observation methods coupled with environmental data through quantitative relationships.^[1,2] Associated with this definition, three components can be specified: the input (soil data), the process (soil inference system), and the output (spatial information system) (see Fig. 1).

Input Data

The input data required to produce digital soil maps are geo-referenced soil observations coupled with environmental variables. Soil and environmental observations can be captured using proximal and remote sensors that can cover variation over large spatial extent. Although proximal and remote sensors do not provide accurate soil details as field surveys and laboratory analysis, they allow for mapping larger extents at higher temporal frequencies (Fig. 2).

Soil Inference Systems

The soil inference systems refer to the quantitative prediction or inference of the soil attributes. This can be spatial inference system to predict soil attributes based on environmental variables, usually using spatial interpolation techniques under the constraints of the environmental

data. Non-spatial inference systems predict functional soil properties from basic soil properties, usually using pedo-transfer functions.

Spatial Information System

The soil information system is formed by the digital soil maps and soil data, where the maps are stored together with field and laboratory observations of soil properties. This includes spatial coordinates of the soil observations and also raster information of the predicted soil maps and environmental data. This information system is the first necessary component for dynamically producing geographically referenced databases created at a given spatial resolution.

Fig. 1 summarizes the process of digital soil mapping, where geo-referenced soil observations coupled with environmental variables form the input data. Under a spatial soil inference system, soil properties over the whole area can be predicted and mapped using spatial soil prediction functions (such as regression, kriging, or a combination of both, see pedometrics). This prediction is based on correlations among the environmental variables and soil attributes as well as the spatial autocorrelation of the attributes themselves. These spatially inferred soil properties can be used to predict more difficult-to-measure functional soil properties, such as field capacity, available water capacity, using pedotransfer functions under soil inference systems. All of the predicted soil properties can be used to evaluate soil functions (soil biomass production, buffering capacities, potential evaporation, etc).

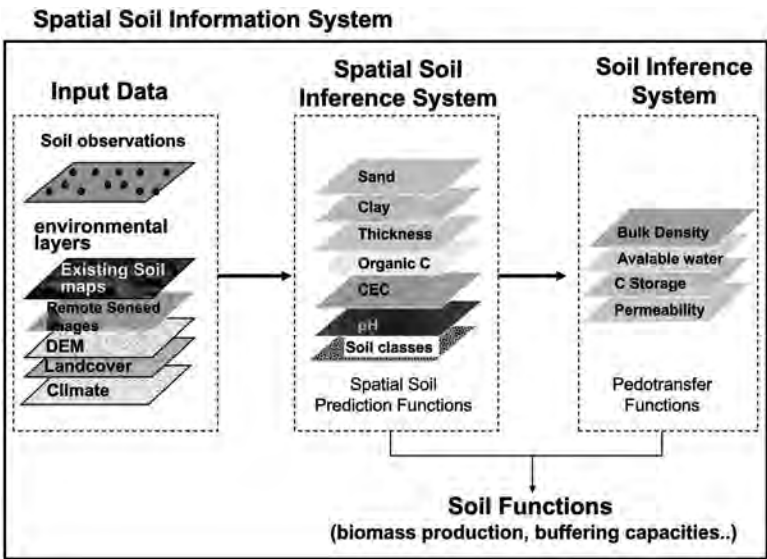


Fig. 1 A general model for digital soil mapping.

Thus, digital soil mapping refers to the production of soil maps digitally ab initio, based on soil observations combined with environmental data through quantitative statistical relationships. In that sense, digital soil mapping is not simply digitizing the existing soil maps and more than just producing paper maps. There are other terminologies such as “predictive soil mapping,”^[3] which refers to the production of digital soil maps, and “environmental correlation,”^[4] which is an aspect of the spatial soil inference systems. The topic of digital soil mapping is extensively reviewed by McBratney et al. and Lagacherie et al.^[1,2]

RATIONALE

The creation of soil maps digitally started in the 1980s, when its emergence corresponded with the development

of geographical information systems (GIS) and associated data, especially digital elevation models. Digital soil mapping serves as an answer to the global needs of soil spatial information for environmental modeling. The demand is to provide soil information with reasonable accuracy, in a reasonable time, and usually with limited resources. Available soil information is neither exhaustive nor precise enough for such a system. This is because conventional soil survey and mapping techniques are slow and expensive.

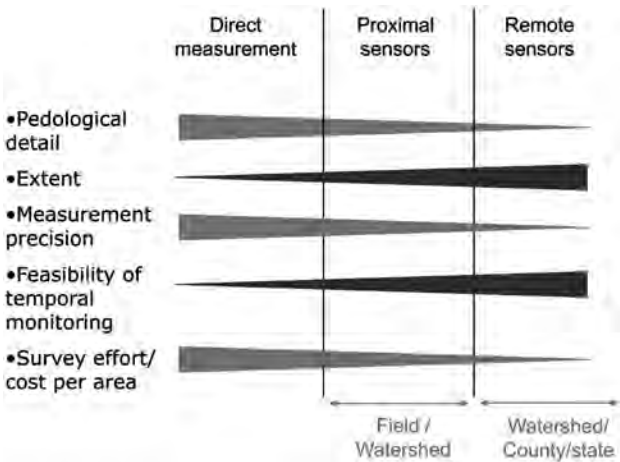
Digital soil mapping has moved from the research phase^[5] to production of maps for regions, catchments, whole countries, and continent.^[6,7] Digital soil mapping is recognized as a body of science with the first review on this subject.^[1] In the review, 70% of studies have successfully mapped soil properties: including physical properties (particle size distribution and available water capacity) and chemical constituents (salt content, carbon content, and storage, nitrogen content, cation exchange capacity, etc.). Thirty percentage of the studies mapped soil classes, such as drainage, type of parent material, and soil units. Soil attributes have been found to have a good relationship with to landform and environmental variables such as geology, climate, and land cover.^[1,8] The soil properties also exhibit strong spatial dependence or trend, and with sufficient field samples can be mapped successfully by a variety of techniques.

A MODEL FOR DIGITAL SOIL MAPPING

A prediction model for spatial inference for digital soil mapping is proposed by McBratney et al.^[1] as the *scorpan* model, which can be written as:

$$S = f(s, c, o, r, p, a, n)$$
 (1)

Fig. 2 A comparison of direct measurement, proximal and remote sensors for obtaining soil information.



where S is either soil classes or soil attributes. This formulation is not for explanation but for empirical quantitative descriptions of relationships between soil and other spatially referenced factors with a view to using these as soil spatial prediction functions. The seven factors are:

1. s : soil, other properties of the soil at a point;
2. c : climate, climatic properties of the environment at a point;
3. o : organisms, vegetation or fauna or human activity;
4. r : topography, landscape attributes;
5. p : parent material, lithology;
6. a : age, the time factor; and
7. n : space, spatial position.

These environmental or the so-called *scorpan* factors can be derived from various sources (digital elevation models, remote sensing images, available soil maps, etc.), available in digital form and cover the whole area of interest. They are used to generate soil information in the form of a database where the information consists of predictions that are statistically optimal.

The general spatial soil prediction model for digital soil mapping can be written as:

$$S = f(Q) + e \quad (2)$$

where Q is the predictor variables (or *scorpan* factors) and e is an error term. The general approach is to take m number of observations S in the field at known locations and fit a

function f with a set of pedologically meaningful predictor variables Q , which will be generally raster data layers of size M in a GIS. Once the model is fitted at the m observation points, the prediction can be extended to the M points or cells in the raster, thereby giving a digital map. The efficiency of the method relies on the fact that m is much smaller than M , because S is much more difficult and expensive to measure than the Q .

The main use of the *scorpan*-spatial soil prediction functions approach is to replace the polygon-based soil maps of the past with digital maps of soil properties and classes and their associated uncertainties for areas previously mapped, or for new areas. These maps will be stored and manipulated in digital form in a GIS creating the possibility of vast arrays of data for analysis and interpretation.

EXAMPLE

The following example shows the production of a digital soil class map in Western France (around La Rochelle).^[9] The soil information system is composed of (Fig. 3):

1. Soil point observations: 4984 geo-referenced soil observations, all described by horizons and each horizon being described by depth, texture class, and Munsell color, carbonate content, salt content, gravel and stone (amount and size), organic matter content,

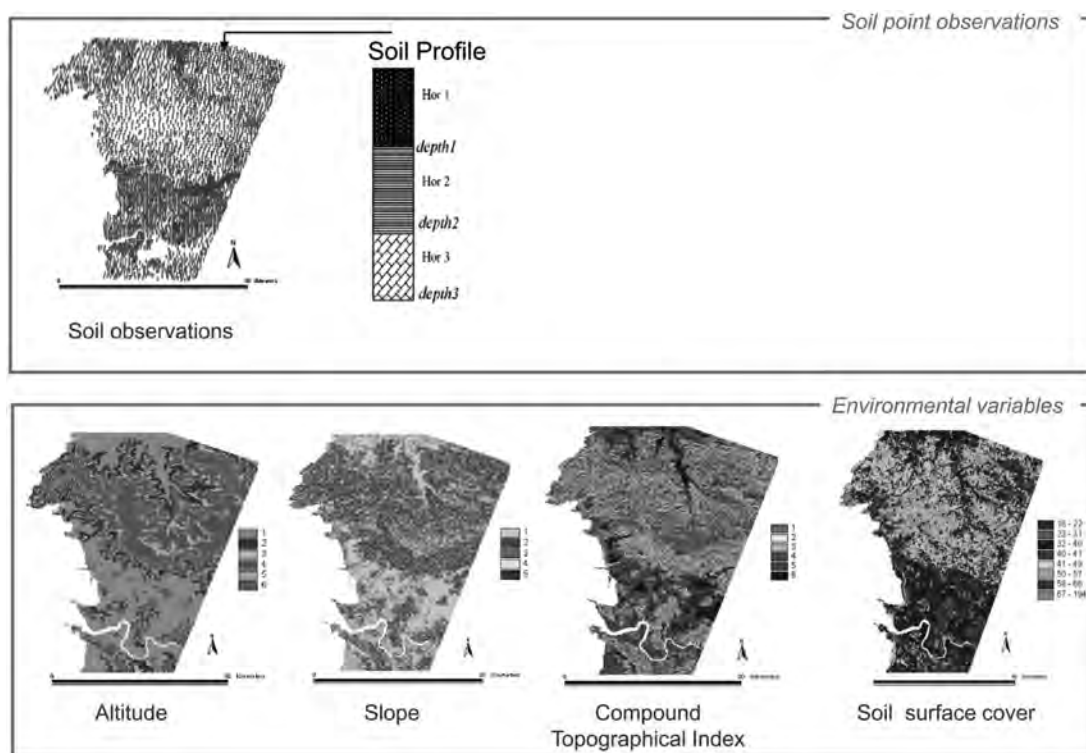


Fig. 3 Soil information system of La Rochelle surroundings area (France).

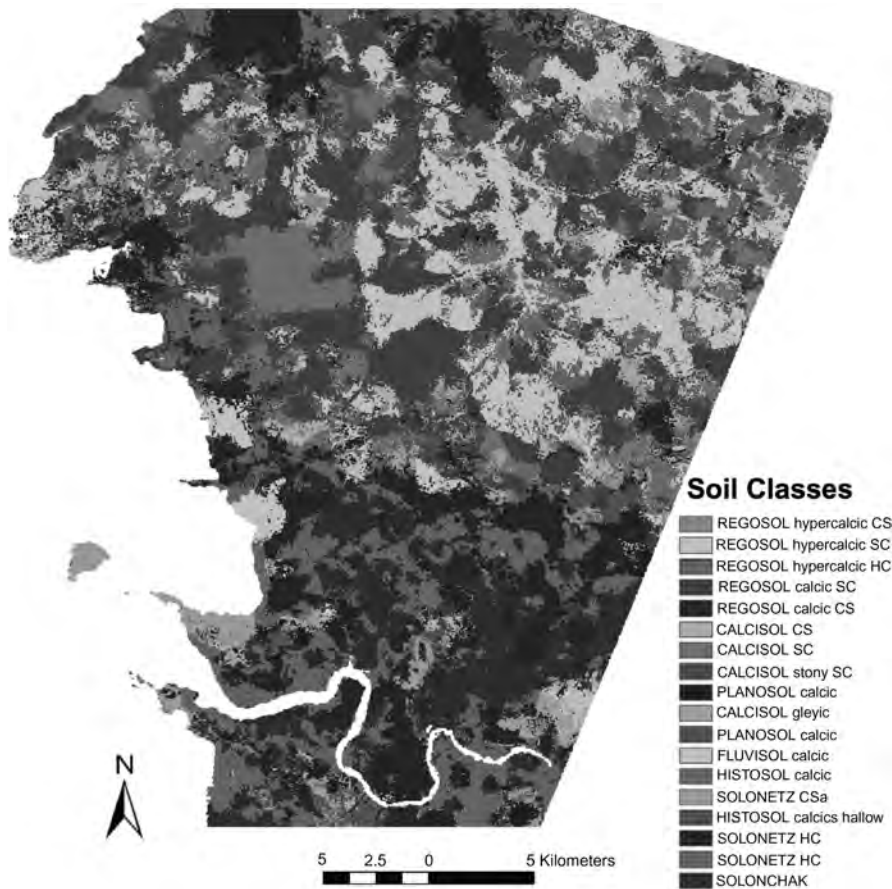


Fig. 4 Digital soil map of the La Rochelle area.

degree of hydromorphism, and lithology of underlying material. The database came from the available soil observations (also called “legacy data”).

2. Environmental (*scorpan*) variables represented as raster with a resolution of $25\text{ m} \times 25\text{ m}$ covering the whole area. The variables include digital elevation model and its derivatives (slope, curvatures, and compound topographical index) and soil surface cover from satellite images.

The spatial soil inference system was produced as follows:

1. Transformation of the soil observations into numerical soil classes. Soil variables are first coded into quantitative variables. First, 18 centroidal (or modal) soil profiles (one for each of the soil classes) are selected for the whole area. Then, taxonomic distances between the representative soil horizons and the observed soil horizons are calculated, and these distances are then aggregated for each profile.
2. Spatial interpolation of the numerical soil classes using regression kriging from the environmental soil factors (Fig. 3).

For each soil class, the distances between each observation i and soil class k (D_{ik}) are first modeled using a multiple linear regression with environmental factors Q as predictors:

$$D_{ik} = \alpha_{0k} + \sum_{j=1}^J \alpha_{jk} Q_{ij} + \epsilon_{ik} \quad (3)$$

The spatially correlated residuals (ϵ_{it}) of the regression are then predicted using kriging. The map of the regression prediction ($\hat{D}$) and the residuals (ϵ) is then summed to obtain the 18 soil class distance maps. For each pixel, the soil class is represented as the class with the smallest distance because the pixel is the most similar to that soil class.

The final soil map (Fig. 4) can be used for environmental modeling, and the uncertainties produced from digital soil mapping can be considered as a representation of the uncertainty or the soil mappers’ confidence.

CONCLUSION

Digital soil mapping uses a range of technologies allowing for more accurate and efficient prediction of

soil properties through optimal sampling strategies, rapid analysis of soil properties, and rapid acquisition of environmental variables over large extent. Combined with pedometric techniques, it can provide the best prediction of soil properties at the required resolution with associated uncertainties. Digital soil mapping can be thought of as a means for modernizing and systematizing traditional soil survey. A global consortium has been formed to generate a global soil map with a resolution of $90\text{ m} \times 90\text{ m}$ using this new technology (see www.globalsoilmap.net).

REFERENCES

1. McBratney, A.B.; Mendonça-Santos, M.L.; Minasny, B. On digital soil mapping. *Geoderma* **2003**, *117*, 3–52.
2. Lagacherie, P.; McBratney, A.B.; Voltz, M., Eds. *Digital Soil Mapping: An Introductory Perspective*; Elsevier: Amsterdam, 2007.
3. Scull, P.; Franklin, J.; Chadwick, O.A.; McArthur, D. Predictive soil mapping: A review. *Prog. Phys. Geog.* **2003**, *27*, 171–197.
4. McKenzie, N.J.; Ryan, P.J. Spatial prediction of soil properties using environmental correlation. *Geoderma* **1999**, *89*, 67–94.
5. Moore, I.D.; Gessler, P.E.; Nielsen, G.A.; Peterson, G.A. Soil attribute prediction using terrain analysis. *Soil Sci. Soc. Am. J.* **1993**, *57*, 443–452.
6. Henderson, B.L.; Bui, E.N.; Moran, C.J.; Simon, D.A.P. Australia-wide predictions of soil properties using decision trees. *Geoderma* **2005**, *124*, 383–398.
7. Hengl, T.; Toomanian, N.; Reuter, H.I.; Malakouti, M.J. Methods to interpolate soil categorical variables from profile observations: Lessons from Iran. *Geoderma* **2007**, *140*, 417–427.
8. Jenny, H. *Factors of Soil Formation. A System of Quantitative Pedology*; McGraw Hill: New York, 1941.
9. Carré, F.; Girard, M.C. Quantitative mapping of soil types based on regression-kriging of taxonomic distances with landform and land-cover attributes. *Geoderma* **2002**, *110*, 241–263.

Drainage: Aeration and Trafficability

Norman R. Fausey

Soil Drainage Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Columbus, Ohio, U.S.A.

Abstract

Soil drainage is a natural process by which water moves through and out of the soil as a result of the force of gravity. This natural process provides water that supports seeps, springs, stream base flow, and aquifer recharge. As water leaves the soil, air moves into the space previously occupied by the water. This process is called aeration. Soil aeration is vital for plant roots and for many beneficial organisms that live in the soil and require oxygen for respiration. As the proportion of water and air in the soil changes as a result of drainage, the ability of the soil to provide support and traction for animals and vehicles (trafficability) is altered, because the strength of the soil changes with water content. The natural drainage of the soil can be accelerated by the use of surface and subsurface drainage practices. Surface drainage diverts excess water from the soil surface directly to streams, thereby reducing the amount of water that will move into and possibly through the soil. Subsurface drainage, provided by ditches and drain pipes, collects and diverts water from within the soil directly to streams.

DRAINAGE (ACCELERATED DRAINAGE)

A complete modern reference for drainage is provided by the American Society of Agronomy.^[1] This monograph provides a comprehensive treatment of the need for, and consequences of, drainage as well as the methods and materials used for drainage, and the design of drainage systems.

Soil comprises 50–70% solid material. The solid phase of the soil is predominately mineral particles in varying proportions of sand, silt, and clay sizes. A small amount of the solid phase is organic material, and this amount decreases with depth. While the amount of the soil organic matter is quite small on a weight basis, its impact on drainage, aeration, and trafficability is substantial.^[2] This leaves 30–50% of the soil volume that comprises void space that can be filled with either air or water. The relative amounts of air and water that are present in the soil determine the ability of the soil to bear the weight of machines and animals, to provide traction, to sustain the biological life in the soil, and to maintain productive and healthy plants. Drainage plays an important role in managing the relative amounts of air and water that are present in the soil.

FACTORS THAT AFFECT SOIL–AIR–WATER RELATIONSHIPS

Bulk Density

The greater the proportion of mineral particles to void spaces, the higher the bulk density. Rocks, boulders,

stones, sand grains, silt particles, and even clay particles typically have a specific weight (density), compared to water, in the range of 2.6–2.7 Mg/m³, because they are all made up of mineral particles. If you were to pour dry soil into a bucket, with the individual soil particles having a specific density of 2.65 Mg/m³, the overall or bulk density of the soil in the bucket would typically be about 1.4 Mg/m³, because there are void spaces between the soil particles. The bulk density of the soil varies depending on the parent material from which it formed, the organic matter content, the depth below the soil surface, and the soil management. Certain processes and activities can decrease the bulk density and thereby create more void space in the soil, while other processes and activities can increase the bulk density and cause loss of void space. The growth of ice crystals in the soil during freezing can result in soil heaving and a decrease in bulk density. Cultivation loosens soil, resulting in more voids and a lower bulk density. Heavy machine traffic compacts the soil, resulting in fewer voids and a higher bulk density.

Soil Texture

Soil texture refers to the relative amounts of sand, silt, and clay size mineral particles in the soil. Sand particles are the largest, ranging from 0.05 to 2.0 mm equivalent diameter; silt particles are smaller, ranging from 0.002 to 0.05 mm equivalent diameter; and clay particles are smallest, less than 0.002 mm equivalent diameter. Equivalent diameter is used to characterize the size, because soil particles are not spherical, but rough and

random in shape. The clay size mineral particles have unique and important characteristics including an electrical charge and tremendous surface area per unit volume. Most of the electrical charges on clay size particles are negative, causing the particles to repel each other like similar magnetic poles and establishing void spaces between the particles. When considering the sand and silt particles, a useful illustration is to consider how many basketballs (representing sand particles) can fit into a bathtub. Between the basketballs, there are large void spaces. Then, consider how many marbles (silt particles) would fit into the same bathtub. The void spaces between the marbles are smaller, but the total volume of the void space is similar in both cases. By putting small particles into the large voids between the large particles, it is possible to end up with more solids and less voids. This happens in soils where the small silt and smaller clay size particles fill the spaces between the large sand size particles. Even when smaller particles occupy the spaces between larger particles, some void space will remain—usually 30–40% of the total space will remain unfilled. However, all the void spaces will be small.

Sandy soils tend to have uniformly distributed large void spaces, resulting in rapid movement of fluids through the soil. Silty soils tend to have uniformly distributed small void spaces and slow movement of fluids. Clay soils tend to have only very small void spaces and very slow movement of fluids.

Soil Structure

Mixtures of the sand, silt, and clay size particles, and the organic matter in the soil become bonded together—a result of the electrical charges on the clays and sticky organic materials in the soil—forming larger units that have a visible structure and are referred to as aggregates. Because they are large units, these aggregates can be thought of again as larger size spheres having larger voids between them. Within the aggregates, there are small voids; between the aggregates, the voids are larger. Aggregates are dynamic rather than permanent arrangements of particles. Accelerated drainage tends to reduce the general size of aggregates, but it increases their stability. Any soil tillage activities tend to disrupt aggregates, leading to smaller voids. Water moves more rapidly through soils with durable aggregates or structural units, and these soils will also typically retain more water that can be utilized by crop plants.

Pore Size Distribution

The interconnected void spaces in soil are typically referred to as pores. Most of these pores are small and can be visualized as capillary tubes, while some pores are as large as drinking straws. Pores are not usually straight and

uniform in diameter, however, being more tortuous as they twist and turn around particles and through or between aggregates. The flow and retention of fluids (air and water) in pores is ultimately what controls the drainage, aeration, and trafficability of the soil. Fluids move more easily through larger pores than through small and more tortuous pores. Burrowing insects or earthworms usually create the largest pores, called macropores. Well-aggregated soils and sandy soils also have large, well-connected pores that transmit air and water easily. Silt and clay soils with many small pores experience slow movement of fluids and entrapment of air within pores during rewetting events. Soils with a range of pores sizes, some macropores, some large pores between aggregates, and some smaller pores within aggregates are usually better suited for agricultural use and respond economically to accelerated drainage installation.

AERATION

Aerobic respiration of plant root cells and microorganisms in the soil is dependent on an adequate supply of oxygen. This oxygen is supplied to a great extent by air moving into the soil through the soil pores. When it rains, some of the rainwater enters the soil through the soil pores and displaces the air from the pores. If enough rainwater enters the soil, all the pores become filled with water and the soil becomes saturated. In the saturated pores, air movement is blocked, and the oxygen needed for respiration, by plant roots and microorganisms, cannot be supplied through the soil pores.

Under natural drainage conditions, some of the water that saturates the soil may slowly percolate downward through the soil to great depths or may leave the soil by evaporation. In some soils, the natural drainage of water occurs rapidly enough to prevent the death of plant roots and microorganisms from the lack of oxygen. In other soils, accelerated drainage is needed to provide a means to get the water out of the pores (and get oxygen in) quickly enough, so that the plant roots and microorganisms can survive.

TRAFFICABILITY

The ability of the soil to support the weight of machines and animals and provide traction for moving about on the soil is called trafficability. Trafficability depends on both bearing strength and slippage. The surfaces of soil particles are typically not smooth, and friction develops between soil particles when force or weight is applied, such as when a machine drives across the surface. Organic matter also increases the resistance of the soil to deformation and increases elasticity.^[2] It is this friction that gives the soil (bearing) strength to support the weight of the machine and (shear) strength to move the machine

forward as the wheels turn. Water acts as a lubricant between soil particles, reducing the friction and allowing the soil particles to move past one another when force or weight is applied. This results in a loss of strength and poor trafficability. With the loss of strength, sinkage and slippage occur in response to the weight of the machine and the rotational force on the tires. Similar sinkage and slippage occur under the feet of animals and humans. Drainage accelerates the removal of water from the soil pores between the soil particles and allows friction between soil particles to occur, so that the bearing and shear strength can support trafficability.

Trafficability can also be impeded by stickiness whereby clay particles stick to the tires of machines or the feet of animals and humans. This phenomenon occurs in very wet but not saturated soils with a high proportion of clay size particles.

CONCLUSION

Accelerated drainage, using ditches or drain pipes to collect the water that is free to be moved by gravity and to transport this water out of the soil, promotes the desired states of aeration and trafficability that are important for efficient and economical management of soil for agriculture, highways, recreation, and many other uses.

REFERENCES

1. Skaggs, R.W.; van Schilfgaarde, J. *Agricultural Drainage*; American Society of Agronomy: Madison, 1999.
2. Soane, B.D. The role of organic matter in soil compactibility: A review of some practical aspects. *Soil Till. Res.* **1990**, *16*, 179–201.

Drainage: Anaerobiosis

Robert O. Evans

Department of Biological and Agricultural Engineering, North Carolina State University, Raleigh, North Carolina, U.S.A.

Abstract

Excess water in the root zone displaces oxygen (O_2) and leads to a condition of anaerobiosis, also known as O_2 stress. O_2 stress leads to many adverse effects on plants including discoloration of leaves (chlorosis), accumulation of toxic substances in the root zone, yield loss, and, in extreme cases, plant death. Some plants can tolerate longer periods of excess soil water or waterlogging than others. The mechanisms of tolerance are varied and may include the ability to respire anaerobically, slowed growth or self-imposed dormancy, the development of adventitious roots, and the development of aerenchyma cells. Drainage can, as it has for centuries, be used to remove excess water from land and reduce the occurrence of waterlogging and subsequent anaerobiosis.

INTRODUCTION

Agricultural crops are obligate aerobes. As such, most plant metabolic processes require oxygen (O_2); in particular, aerobic respiration provides energy from the degradation of glucose (derived from sugars and starches in the tissue) to pyruvic acid, which is further oxidized to carbon dioxide (CO_2) and water. Anaerobiosis implies life in the absence of O_2 . Anaerobiosis is most commonly caused by waterlogging of the root zone. Under anaerobic conditions (lack of O_2), pyruvic acid produced from glucose in the root cells of higher plants is converted to CO_2 and ethyl alcohol. Ethyl alcohol tends to accumulate within root cells and, if anaerobic conditions persist, may accumulate at concentrations that can become toxic. Anaerobiosis thus creates an environment that impairs the growth rate of most plants that require O_2 for respiration. Drainage is the practice of removing excess water from land to minimize the occurrence of waterlogging. It has been an important management practice for centuries,^[1] and the consequences of poor drainage have been studied extensively in the 21st century.^[2–8]

SOIL AERATION

Either air or water occupies the pore volume of the soil. Excess water displaces air and affects the root zone environment and subsequent plant survival, growth, and yield. Processes affected by excess soil water include soil O_2 and CO_2 exchange with the atmosphere; oxidation reduction reactions in the soil and subsequent diffusion of gaseous metabolites; nutrient availability and uptake; and accumulation of CO_2 , ethylene, and other

toxic substances (Fig. 1). These conditions can impact crop production through the processes such as suppressing seed germination, restricting root growth, death of root cells, shoot wilting, chlorosis (yellowing of leaves because of the failure of leaves to produce normal amounts of chlorophyll), and slowed growth and dry matter accumulation.

The most basic consequence of excessive soil water is restriction of the supply of O_2 to roots by displacing air in the pore space and by reducing the exchange of soil air with the atmosphere. These processes are often lumped together and referred to as soil aeration.^[10] Soil aeration status has often been reported on a percentage basis of water-free pore space. Air porosities in the range of 5–20% were reported in earlier literature to be adequate for crop growth. A minimum value of 10% was reported in an early study,^[11] although the optimum air filled porosity may range from 4% to 50%, depending on soil texture, type and age of crop, and other growing conditions.^[7]

Soil aeration can also be expressed as the O_2 content of air samples extracted from the soil.^[12] In well-aerated soil, the composition of soil air closely approximates that of the atmosphere, namely, 20% O_2 and 0.03% CO_2 . O_2 contents in the range of 8–14% have been reported to be minimum for unrestricted root respiration; however, critical values are affected by many factors including soil physical properties such as texture and structure, depth in the soil, type and growth stage of crop, soil water content (SWC), organic matter present, and, of course, temperature. Numerous studies have documented declines in O_2 content to less than 2% within 24–48 hours under waterlogged conditions (Fig. 2). The rate of decline of O_2 is closely correlated to metabolic activity of microbes and roots and to soil temperature.^[13,14]

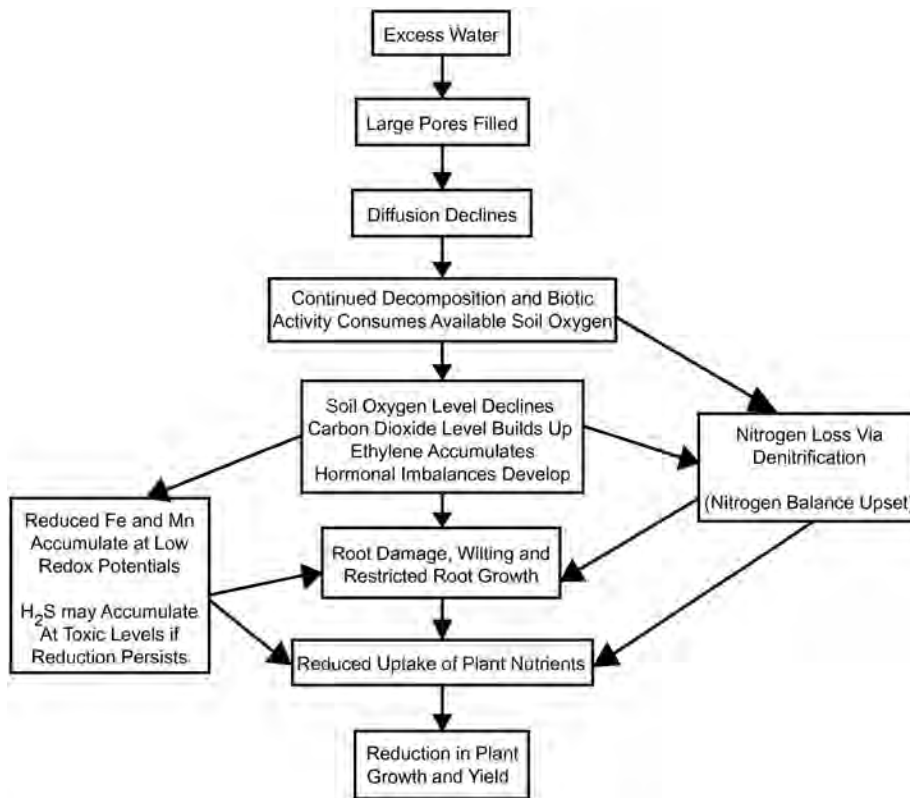


Fig. 1 Processes causing reduction in plant growth and yields owing to excessive soil water and anaerobiosis. **Source:** From Patwardhan, Nieber, et al.^[7] and Ravelo.^[9]

The oxygen diffusion rate (ODR) is the rate of exchange of soil air with the atmosphere. At the soil surface, the composition of soil air is in equilibrium or nearly so with that of the atmosphere. The O_2 content of soil air generally declines and CO_2 content resulting from the production of CO_2 during aerobic respiration increases with soil depth. At any depth, the rate of exchange of soil air with the atmosphere decreases as SWC increases (Fig. 3). Under waterlogged conditions, the O_2 content declines to less than 1% and ODR typically approaches zero.^[12] The exchange of soil air with the atmosphere occurs by two mechanisms: convection (often referred to as mass flow) and diffusion. Early work^[16] reported that O_2 transport into the soil by mass flow (convective flow) was insignificant compared

with that by diffusion. The diffusion rate of O_2 is roughly 10,000 times less through water than through contiguous, air filled soil pores. Convective transport has been considered to be important under some conditions such as shallow depths or in soils with large pores.^[7,10] Theoretical relationships for describing airflow processes in soil have been reported.^[7,12]

ODRs required for normal plant growth have been reported for several crops. In early work, plants such as sunflowers, cotton, and snapdragons stopped growing when the ODR in the soil dropped below $3.3 \times 10^{-8} \text{ kg/m}^2/\text{sec}$.^[17] Plant functions such as photosynthesis may be suppressed when the ODR drops below $2 \times 10^{-8} \text{ kg/m}^2/\text{sec}$.^[18] A value of $6.8\text{--}5 \times 10^{-8} \text{ kg/m}^2/\text{sec}$ was found for soybean^[19] with a

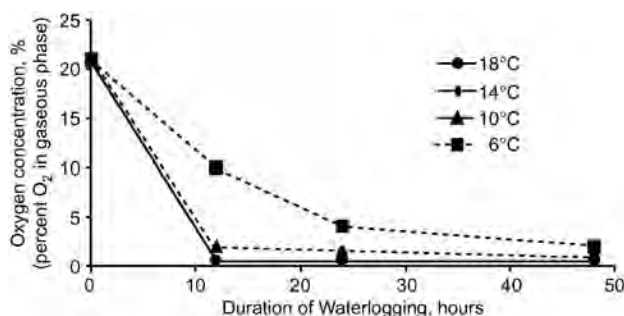


Fig. 2 O_2 content of soil air as influenced by duration of waterlogging at four soil temperatures. **Source:** From Trought & Drew.^[13]

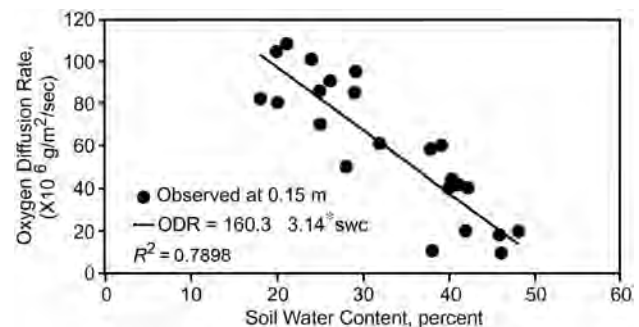


Fig. 3 Relationship between ODR and SWC. **Source:** From Bornstein, Benoit, et al.^[15]

critical value of $3.3 \times 10^{-8} \text{ kg/m}^2/\text{sec}$. The required rate of diffusion is constantly changing because of the constant changes in growing conditions.^[12]

PLANT RESPONSES TO POOR AERATION

As noted earlier, there are many crop responses to poor aeration that results from waterlogged or excessive soil water conditions. A prolonged lack of O_2 suppresses and, in many cases, prevents the germination of most seeds. While seeds of many species can tolerate short periods of limited aeration, cell division and growth rate are reduced when the supply of O_2 is cutoff, even for a few hours.^[13,20,21] Fausey and McDonald^[21] observed that emergence suppression was much greater at warmer temperatures.

Low O_2 diffusion limits root respiration and the exchange of gaseous metabolites such as CO_2 and ethylene. There is extensive evidence that the rapid decline in O_2 content is the primary mechanism of injury to roots.^[2,22–24] Although ethylene accumulation was once thought to be a direct cause of root injury,^[25] Cannell and Jackson,^[2] citing the works of others, concluded that ethanol accumulation in roots likely did little more than modestly depressing the root growth. Root injury tends to be in the form of slow or terminated growth during waterlogged conditions because of a lack of energy normally obtained through aerobic respiration. O_2 content within the root zone often drops below critical levels within hours of waterlogging.^[26] Again, growth retardation responses can be rapid, especially at higher temperatures.^[27] In studies involving static water tables, investigators have reported little or no root penetration below the water table.^[28] McDaniel^[29] observed that corn roots responded quickly to saturation, with all growth ceasing within 24 hours. He observed that three days of waterlogging significantly affected root mass, maximum root depth, and consumptive water use for the remainder of the growing season. Like roots, plant shoots can respond quickly to flooded conditions. Frequently cited symptoms of above-ground plant parts to waterlogging include shoot wilting,^[30] increased resistance to water flow,^[31–35] elevated plant temperature,^[36] and chlorosis that leads to slowed growth and dry matter accumulation.^[12,23,24,31,36–38]

In soils with a high water table (water table less than 2m from the soil surface during the growing season), yield of most crops can be related to the water table depth. The yield response is generally not a direct response to the position of the water table but rather to the indirect effects the water table has on aeration status, nutrient availability, and soil moisture that is available to the plant. At very shallow water table positions, most soil pores are filled with water and the crop experiences aeration or O_2 stress (anaerobiosis), which in turn suppresses yield. A common method for quantifying the stress imposed on a crop by excess soil water or lack of

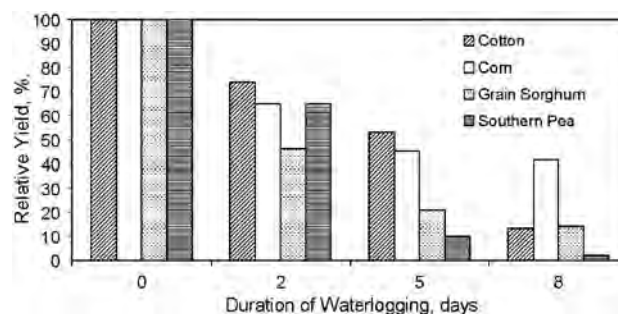


Fig. 4 Relationship between crop yield and flooding duration. Source: From Howell & Hiler^[39] and Hiler.^[40]

O_2 relies on experimental determination of crop yield, in response to an elevated water table. The most frequently applied approach has been to grow crops in lysimeters and examine the effect of inundation or waterlogging on the crop yield. This approach does not directly account for the stress caused by lack of O_2 , excess CO_2 , or accumulation of toxins within the soil profile; but it does provide one method of relating crop yield to a condition of obvious stress.^[7]

Several general conclusions can be developed from inundation studies. Crop-yield reductions increase as the flooding duration increases, as shown in Fig. 4. This conclusion is consistently supported by most studies for most crops. Many crops can tolerate short-term periods of waterlogging, but the tolerance duration depends on many factors. One important factor affecting crop-yield response to excess water and poor aeration is the crop's stage of development. Most crops studied tend to be more susceptible during earlier stages of development (preflowering).^[26,41–43] Another important factor is the accumulation of toxic substances found in the soil associated with anaerobic respiration. Nitrite is the first potentially toxic product of anaerobic respiration. At low pH, incomplete conversion of nitrate to N_2 gas can favor the formation of nitrous acid.^[44] Under extended reducing conditions where E_h values drop below 100 mV, iron and manganese can be reduced and accumulate at toxic concentrations. If anaerobic conditions persist, hydrogen sulfide can be produced that is reported to kill roots at concentrations greater than 2.5 ppm.^[45]

DRAINAGE TO REDUCE ANAEROBIOSIS

A primary goal of agricultural drainage in humid regions is to lower the water content of the root zone to provide adequate aeration following excessive rainfall or irrigation such that it leaves a soil condition that is favorable for crop growth.^[46] Several techniques are available that improve drainage and reduce excess water-related stress. Surface drainage^[47] can be achieved by a system of open ditches,

typically installed on 100–200 m intervals parallel to each other in the general direction of the prevailing slope. To encourage surface runoff and reduce surface ponding, fields are often graded and sometimes crowned into a turtleback shape. Field ditches are typically 0.5–1.5 m deep and discharge into collector canals typically laid out on 1 or 2 km grids. Irregularly spaced ditches are often used to drain depressional areas. Bedded rows parallel to the field ditches are sometimes used to elevate the seedbed and help reduce water stress while the plants are small. Surface drainage systems encourage aeration of the root zone by quickly removing surface water. Once surface ponding is eliminated, the water table declines rapidly during periods with moderate to high potential evapotranspiration, which is usually the case during the growing season. Thus, a surface drainage system encourages reaeration of the root zone by reducing the volume of water entering the soil profile, while the water table is actually lowered by some means other than direct drainage, i.e., evapotranspiration. As the water table declines, air replaces the water that is removed.

Excess water can also be removed by a system of subsurface drains^[48–50] comprising clay tile, perforated plastic tubing, or unlined mole drains. In general, 100–150 mm diameter tile or tubing is buried 1–1.5 m deep at intervals of 10–50 m. The subsurface drainage pipes generally flow into to an open ditch or stream. In United States, subsurface drainage systems have typically been designed to remove 125 mm (0.5 in.) of water per day, for grain crops. Higher design rates up to 250 mm/day have been used for vegetable crops grown in organic soils. Subsurface drainage systems directly improve root zone aeration by lowering the water table following heavy rainfall. In many cases, crop protection is also provided as a result of the water table being lowered before the rainfall. In this case, subsurface drainage provides adequate soil storage to prevent the water table from rising into the root zone and as a result aerobic conditions may persist throughout the rainfall/drainage event. As a general rule, aeration and O₂ diffusion into the root zone are associated with increased water table depth and greater drained porosity.^[51,52] No degree of site drainage will provide the “optimal” soil water environment in all situations. Drainage requirements depend on many inter-related factors including rainfall duration and intensity, soil physical properties, soil and air temperature, type of crop being grown, and the growth stage of the crop.

CONCLUSION

Excess water in the root zone displaces O₂ and leads to a condition of anaerobiosis, also known as O₂ stress. O₂ stress leads to many adverse effects on plants including discoloration of leaves (chlorosis), accumulation of

toxic substances in the root zone, yield loss, and, in extreme cases, plant death. Some plants can tolerate longer periods of excess soil water or waterlogging than others. The mechanisms of tolerance are varied and may include the ability to respire anaerobically, slowed growth or self-imposed dormancy, the development of adventitious roots, and the development of aerenchyma cells. Work using molecular genetics techniques has shown that specific genes are responsible for flooding tolerance^[53] and provides future promise as a means of improving the flooding tolerance of major agronomic crops. Drainage can, as it has for centuries, be used to remove excess water from land and reduce the occurrence of waterlogging and subsequent anaerobiosis. Improved knowledge of the physiological mechanisms of plant tolerance may lead to alternatives for overcoming the effects of poor soil aeration.^[2] However, drainage continues to be the most viable alternative for reducing the frequency and duration of anaerobiosis resulting from excess soil water.

REFERENCES

1. Beaucamp, K.H. A history of drainage and drainage methods. In *Farm Drainage in the United States: History, Status, and Prospects*; Pavelis, G.A., Ed.; Economics Research Service, U.S. Department of Agriculture: Washington, 1987; 13–29.
2. Cannell, R.Q.; Jackson, M.B. Alleviating aeration stresses. In *Modifying the Root Environment to Reduce Crop Stress*; Arkin, G.F., Taylors, H.M., Eds.; ASAE: St Joseph, 1981; 141–192.
3. Kozlowski, T.T., Ed. *Flooding and Plant Growth*; Academic Press: Orlando, 1984; 356 pp.
4. Glinski, J.; Stepniewski, W. *Soil Aeration and Its Role for Plants*; CRC Press: Boca Raton, 1985; 229 pp.
5. Heritage, A.D. Temporary waterlogging, poor soil aeration and root susceptibility to fungal infection—A review. In *Root Zone Limitations to Crop Production on Clay Soils*; Muirhead, W.A., Humphreys, E., Eds.; CSIRO: Melbourne, 1985; 117–125.
6. Kowalik, P.J. *Influence of Land Improvement on Soil Oxidation*; Report 42; Swedish University of Agricultural Science: Uppsala, 1985; 145 pp.
7. Patwardhan, A.S.; Nieber, J.L.; Moore, I.D. Oxygen, carbon dioxide, and water transfer in soils: Mechanisms and crop response. *Trans. ASAE* **1988**, *31* (5), 1383–1395.
8. Evans, R.O.; Fausey, N.R. Effects of inadequate drainage on crop growth and yield. In *Agricultural Drainage*; Skaggs, R.W., van Schilfgaarde, J., Eds.; Agronomy Monograph 38; American Society of Agronomy, Inc.: Madison, 1999; 13–54.
9. Ravelo, C.J. A rational approach for incorporating crop needs into drainage system design. Ph.D. dissertation, Texas A&M University, College Station, 1978; 159 pp.
10. Hillel, D. *Fundamentals of Soil Physics*; Academic Press: New York, 1980; 413 pp.

11. Wesseling, J.; van Wijk, W.R. Soil physical conditions in relation to drain depth. In *Drainage of Agricultural Lands*; Luthin, J.N., Ed.; Monograph No. 7; ASA: Madison, 1957; 461–504.
12. Wesseling, J. Crop growth and wet soils. In *Drainage for Agriculture*; van Schilfhaarde, J., Ed.; Monograph No. 17; ASA: Madison, 1974; Vol. Vol. Chapter 3, 39–90.
13. Trought, M.C.T.; Drew, M.C. Effects of waterlogging on young wheat plants (*Triticum Aestivum* L.) and on soil solutes at different soil temperatures. *Plant Soil* **1982**, *69*, 311–326.
14. Belford, R.K.; Canell, R.Q.; Thomson, R.J. Effects of single and multiple waterlogging on the growth and yield of winter wheat on a clay soil. *J. Sci. Food Agr.* **1985**, *36*, 142–156.
15. Bornstein, J.; Benoit, G.R.; Scott, F.R.; Hepler, P.R.; Hedstrom, W.E. Alfalfa growth and soil oxygen diffusion as influenced by depth to water table. *Soil Sci. Soc. Am. J.* **1984**, *48*, 1165–1169.
16. Buckingham, E. *Contributions to Our Knowledge of the Aeration of Soils*; Bulletin 38; U.S. Bureau Soils: Washington, 1904.
17. Stolzy, L.H.; Letey, J. Characterizing soil oxygen conditions with a platinum microelectrode. *Adv. Agron.* **1964**, *16*, 249–279.
18. Sojka, R.E.; Stolzy, L.H. Soil oxygen effects on stomatal response. *Soil Sci.* **1980**, *130*, 350–358.
19. VanToai, T.T.; Desmond, E.D.; Fausey, N.R. Lysimeter study of soybean responses to excess water. In *Drainage Design and Management, Proceedings of the Fifth National Drainage Symposium*; Johnston, W.R., Ed.; ASAE: St Joseph, 1987; 213–216.
20. Cannell, R.Q.; Belford, R.K. Crop growth after transient waterlogging. In *Advances in Drainage, Proceedings of the Fourth National Drainage Symposium*; Kriz, G.J., Ed.; ASAE: St Joseph, 1982; 163–170.
21. Fausey, N.R.; McDonald, M.B. Emergence of inbred and hybrid corn following flooding. *Agron. J.* **1985**, *77*, 51–56.
22. Box, J.E., Jr. The effects of waterlogging on rooting of soft red winter wheat. *Dev. Agr. Manage. For. Ecol.* **1991**, *24*, 418–430.
23. Huang, B.; Johnson, J.W. Root respiration and carbohydrate status of two wheat genotypes in response to hypoxia. *Ann. Bot.* **1995**, *75* (4), 427–432.
24. Huang, B.; NeSmith, D.S.; Bridges, D.C.; Johnson, J.W. Responses of squash to salinity, water logging, and subsequent drainage. II. Root and shoot growth. *J. Plant Nutr.* **1995**, *18* (1), 141–152.
25. Crawford, R.M.M. Tolerance of anoxia and ethanol metabolism in germinating seeds. *New Phytol.* **1977**, *79*, 511–517.
26. Mukhtar, S.; Baker, J.L.; Kanwar, R.S. Corn growth as affected by excess soil water. *Trans. ASAE* **1990**, *33* (2), 437–442.
27. del Castillo Torres, D. Soybean root growth and recovery under temporary anaerobic soil conditions. Ph.D. Dissertation, Mississippi State University, 1983.
28. Baser, D.K.; Jaggi, I.K.; Sinha, S.B. *Root and Shoot Growth and Transpiration from Maize Plant as Influenced by Various Depths of Water Table*; Jawaharlal Nehru Agric. Univ.: Jabalpur, 1981; 390–399.
29. McDaniel, V. Effects of shallow water tables on corn roots, crop yield and hydrology. Ph.D. Dissertation, North Carolina State University, 1995; 166 pp.
30. Parsons, L.R.; Kramer, P.J. Diurnal cycling in root resistance to water movement. *Physiol. Plant* **1974**, *30*, 19–23.
31. Galloway, R.; Davidson, N.J. The response of *Atriplex amnicola* to the interactive effects of salinity and hypoxia. *J. Exp. Bot.* **1993**, *44* (260), 653–663.
32. Timsina, J.; Garrity, D.P.; Pandey, R.K. Water table gradient effects on the performance of diverse cowpea cultivars. *Agron. J.* **1993**, *85* (2), 359–367.
33. Huang, B.; Johnson, J.W.; NeSmith, D.S.; Bridges, D.C. Growth, physiological and anatomical responses of two wheat genotypes to waterlogging and nutrient supply. *J. Exp. Bot.* **1994**, *45* (271), 193–202.
34. Huang, B.; Johnson, J.W.; NeSmith, D.S.; Bridges, D.C. Root and shoot growth of wheat genotypes in response to hypoxia and subsequent resumption of aeration. *Crop Sci.* **1994**, *34* (6), 1538–1544.
35. Pezeshki, S.R.; Pardue, J.H.; DeLaune, R.D. Leaf gas exchange and growth of flood-tolerant and flood-sensitive tree species under low soil redox conditions. *Tree Physiol.* **1996**, *16* (4), 453–458.
36. Timsina, J.; Garrity, D.P.; Pandey, R.K. Cowpea water relations growth response on a toposequence water table gradient. *Agron. J.* **1993**, *85* (2), 368–378.
37. Cannell, R.Q.; Belford, R.K.; Blackwell, P.S.; Govi, G.; Thomson, R.J. Effects of waterlogging on soil aeration and on root and shoot growth and yield of winter oats (*Avena Sativa* L.). *Plant Soil* **1985**, *85*, 361–373.
38. Ahmad, N.; Kanwar, R.S.; Kaspar, T.C.; Bailey, T.B. Effect of soil surface submergence and a water table on vegetative growth and nutrient uptake of corn. *Trans. ASAE* **1992**, *35* (4), 1173–1177.
39. Howell, T.A.; Hiler, E.A. Effects of inundation period on seedling growth. *Trans. ASAE* **1974**, *17* (2), 286–288.
40. Hiler, E.A. Drainage requirements of crops. In *Proceedings of the Third National Drainage Symposium*; Hoffman, G.J., Ed.; ASAE: St Joseph, 1977; 127–129.
41. Griffin, J.L.; Saxton, A.M. Response of solid-seeded soybean to flood irrigation. II. Flood duration. *Agron. J.* **1988**, *80* (6), 885–888.
42. Scott, H.D.; DeAngulo, J.; Daniels, M.B.; Wood, L.S. Flood duration effects on soybean growth and yield. *Agron. J.* **1989**, *81* (4), 631–636.
43. Evans, R.O.; Skaggs, R.W.; Sneed, R.E. Normalized crop susceptibility factors for corn and soybean to excess water stress. *Trans. ASAE* **1990**, *33* (4), 1153–1161.
44. Lee, R.B. The effect of nitrite on root growth of barley and maize. *New Phytol.* **1979**, *83*, 615–622.
45. Culbert, D.L.; Ford, H.W. The use of a multi-celled apparatus for anaerobic studies of flooded root systems. *Hort. Sci.* **1972**, *71*, 29–31.
46. Fausey, N.R.; Doering, E.J.; Palmer, M.L. Purposes and benefits of drainage. In *Farm Drainage in the United States: History, Status, and Prospects*; Pavelis, G.A., Ed.; Economics Research Service, U.S. Department of Agriculture: Washington, 1987; 48–51.
47. Carter, C.E. Surface drainage. In *Agricultural Drainage*; Skaggs, R.W., van Schilfhaarde, J., Eds.; Agronomy

- Monograph 38; American Society of Agronomy, Inc.: Madison, 1999; 1023–1050.
48. Schwab, G.O.; Fouss, J.L. Drainage materials. In *Agricultural Drainage*; Skaggs, R.W., vanSchilfgaarde, J., Eds.; Agronomy Monograph 38; American Society of Agronomy, Inc.: Madison, 1999; 911–926.
 49. Broughton, R.S.; Fouss, J.L. Subsurface drainage installation machinery and methods. In *Agricultural Drainage*; Skaggs, R.W., van Schilfgaarde, J., Eds.; Agronomy Monograph 38; American Society of Agronomy, Inc.: Madison, 1999; 967–1004.
 50. Spoor, G.; Leeds-Harrison, P. Nature of heavy soils and potential drainage problems. In *Agricultural Drainage*; Skaggs, R.W., van Schilfgaarde, J., Eds.; Agronomy Monograph 38; American Society of Agronomy, Inc.: Madison, 1999; 1051–1182.
 51. Skaggs, R.W.; Wells, L.G.; Ghate, S.R. Predicted and measured drainable porosities for field soils. *Trans. ASAE* **1978**, *21* (3), 522–528.
 52. Silins, U.; Rothwell, R.L. Spatial patterns of aerobic limit depth and oxygen diffusion rate at two peatlands drained for forestry in Alberta. *Natl. Res. Council Can.* **1999**, *29* (1), 53–61.
 53. VanToai, T.T.; Beuerlein, J.E.; Schmitthenner, A.F.; St. Martin, S.K. Genetic variability for flooding tolerance in soybeans. *Crop Sci.* **1994**, *34* (4), 1112–1115.

Drainage: Water Quality

James L. Baker

Department of Agricultural and Biosystems Engineering, Iowa State University, Ames, Iowa, U.S.A.

Abstract

Artificial subsurface drainage is required on many agricultural lands to remove excess precipitation and/or irrigation water in order to provide a suitable soil environment for plant growth and a soil surface capable of physically supporting necessary traffic, e.g., for tillage, planting, and harvesting. Although this drainage makes otherwise wet soils very productive, subsurface drainage alters the time and route by which excess water reaches surface waters and can carry nutrients, pesticides, bacteria, and suspended solids to surface waters and cause non-point source pollution problems. The net water quality impact of subsurface drainage considered here is determined by a comparison to the same cropping system not having subsurface drainage (not on the fact that the existence of adequate drainage will affect land use). The degree of pollutant transport with surface runoff and subsurface drainage is determined by the product of the volumes of water and the pollutant concentrations in the water. These are both influenced by environmental conditions, pollutant properties, and management factors, and their interactions with subsurface drainage are discussed.

ENVIRONMENTAL CONDITIONS

Soils/Hydrology

Whether water added to the soil surface as precipitation or irrigation infiltrates or becomes surface runoff is critical to water quality. Topography/slope, soil moisture content, texture, and soil structure, including the existence of preferential flow paths or “macropores,” can affect both the rate and route of water infiltration (as shown in Fig. 1). While different drain spacings are used to provide a desired “drainage coefficient” (e.g., 0.5 in./day) based on the internal drainage characteristics of the subsoils, the conditions of the surface soil will determine what percentage of applied water (rain or irrigation) will infiltrate. In general, the existence of subsurface drainage increases the volume of infiltration and thus decreases the volume of surface runoff (and pollutant loss with that runoff), increases shallow percolation, and lowers water tables. This effect will be more pronounced for “lighter” soils that have higher infiltration rates and lower water holding capacities. These changes affect the final quality of cropland drainage because of differences in time and type of soil–water–chemical interactions. Surface runoff allows water to come in contact only with the surface soil and materials such as crop residue and surface-applied fertilizers, manures, and pesticides present on it for short periods of time. On the other hand, water that infiltrates and percolates through the soil comes in intimate contact with all the soils in the profile to at least the depth of the

tile drain, and generally, for much longer periods of time. The soil residence time is shorter for that portion of infiltrated water intercepted by the tile drains than for water that must flow through the underground strata to appear as base flow.

Climate/Precipitation

The timing, amount, and intensity of water inputs, including precipitation and irrigation, relative to evapotranspiration (ET) determine the timing and amount of excess water that will leave agricultural land as surface runoff and subsurface drainage. Inputs at low intensities generally will totally infiltrate, but as intensities increase past the rate of infiltration, surface runoff begins. As just discussed, decreased antecedent soil moisture contents (resulting from the existence of artificial subsurface drainage) increase infiltration rates, and the volume of surface runoff generally is decreased when subsurface drainage exists. This effect will be more pronounced for areas where input intensities often exceed the rates of infiltration.

For a given crop and climatic region, the amount of ET is relatively constant, so the volume of subsurface drainage is very much dependent on the amount of input water above that value. As an example, in a 4-year Iowa tile drainage experiment,^[1] during a low rainfall year, only a trace of subsurface flow occurred; in a wet year (116 cm of precipitation vs. the average of 80 cm), there was 29 cm of flow; and the overall average flow for the 4 years was 15 cm.

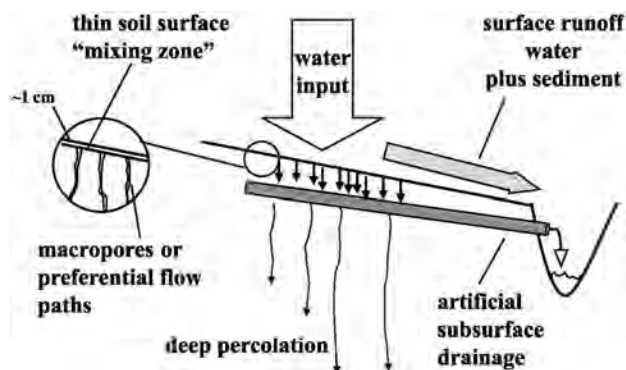


Fig. 1 Schematic of transport processes.

POLLUTANT PROPERTIES

Persistence

Pollutants that have a limited existence in agricultural lands because of plant uptake/chemical transformation (i.e., nutrients), degradation (i.e., pesticides), and die-off (i.e., microorganisms) have different potentials for off-site transport with water based on their persistence. Because surface runoff is much more of an immediate process than subsurface drainage, concentrations of non-persistent pollutants in subsurface drainage are usually lower than in surface runoff. This effect will be more pronounced, the lesser the persistence is. However, the “route” of infiltration, where in some cases the drainage water moves quickly through the soil profile because of macropores (see Fig. 1), plays a role and can somewhat negate the expected dissipation effect.

Adsorption/Filtration

Based on their chemical properties and/or their physical size, some pollutants transported down into and through the soil with subsurface drainage are removed from the flow stream by the soil. In particular, pollutants that are positively charged [e.g., ammonium–nitrogen ($\text{NH}_4\text{-N}$)] and larger, less soluble, organic compounds (e.g., some pesticides) can be attenuated by adsorption to soil clay and organic matter. Inorganic P ions can be removed by complexation or precipitation with soil cations. Microorganisms and sediment can be filtered out by small soil pores. Thus, in general, with the exception of soluble salts of nitrate–nitrogen ($\text{NO}_3\text{-N}$), sulfate, and chloride anions, pollutant concentrations in subsurface drainage are lower than in surface runoff. This effect will be more pronounced, the greater the interaction between the pollutant and soil is.^[2]

For sediment itself, because soil erosion is dependent on the erosive ability and transport capacity of surface runoff, reducing the runoff flow rate and volume with subsurface drainage reduces sediment loss. For example, in a 6-year

study in the lower Mississippi Valley, surface runoff volumes from plots on a clay loam soil with subsurface drainage were 34% less than for plots without subsurface drainage, and the corresponding soil loss of 3500 kg/ha/yr represented a decrease of 30%. For the plots with subsurface drainage, both sediment concentrations and losses in subsurface drainage were about one-tenth of those in surface runoff.^[3]

For nitrogen (N), loss from poorly drained soils is usually much less than that from soils with improved drainage systems.^[4] While significant N can be transported with sediment (often sediment has at least 1000 ppm N), land needing subsurface drainage is usually not highly susceptible to erosion, and N loss is dominated by soluble inorganic N loss. In a three-year tile-drained watershed study in northeast Iowa,^[5] $\text{NO}_3\text{-N}$ losses in solution represented over 85% of the total N losses, including $\text{NH}_4\text{-N}$, organic N in solution, and N associated with sediment. In a five-year study in East Central Iowa,^[6,7] where nutrient concentrations in both surface runoff and subsurface drainage from cropland were monitored, $\text{NO}_3\text{-N}$ concentrations in subsurface drainage averaged about 12 mg/L and 2–3 times those in surface runoff. While $\text{NH}_4\text{-N}$ concentrations were usually 2–10 times higher in surface runoff, on an absolute scale, the concentrations were overall much lower than those for $\text{NO}_3\text{-N}$ and constituted only a fraction of the total N loss.

For phosphorus (P), transport is primarily in surface runoff with sediment (often sediment has at least 500 ppm P) and dissolved in surface runoff. For conventionally tilled cropland, about 75–90% of P transported in surface runoff is with sediment. In areas where soil erosion is minimal, soluble P in surface runoff water can dominate transport. The soluble P in surface runoff (and subsurface drainage) is regulated by adsorption/desorption characteristics of soil. Therefore, with P in surface soils generally much higher than in subsoils, subsurface drainage usually has much lower soluble P concentrations than in surface runoff.

For pesticides, because of their adsorption characteristics, like with P, concentrations in surface runoff water are usually much greater than in subsurface drainage. However, this effect is even more dramatic for pesticides because unlike P, pesticides have a limited persistence, which decreases their potential for movement with subsurface flows with long travel times. In a series of studies on herbicides in surface and subsurface drainage,^[8–10] atrazine concentrations in May (the period of application) were 75 $\mu\text{g/L}$ in surface runoff for plots with surface drainage only and were 51 $\mu\text{g/L}$ and 1 $\mu\text{g/L}$ in surface runoff and subsurface drainage, respectively, for plots that also had subsurface drainage. Not only were there much lower atrazine concentrations in subsurface drainage water but the presence of the subsurface drains delayed and reduced surface runoff such that concentrations in surface runoff from the plots with subsurface drainage were reduced about one-third.

For bacteria, a review by Crane et al.^[11] showed that fecal coliform counts in surface runoff from manured lands often were greater than 10,000/100 ml. In a comparison of surface runoff and subsurface drainage, Culley and Phillips^[12] found similar counts (>10,000/100 ml) for fecal coliform in surface runoff from both manured and fertilized plots, but with much lower counts (<5/100 ml) in subsurface drainage.

MANAGEMENT PRACTICES

Tillage Systems

The hydrologic interactions between tillage systems and subsurface drainage that affects water quality involve how tillage affects the timing, route, and volume of infiltration (and hence the relative volumes of subsurface drainage and surface runoff). In a review of the hydrologic effects of conservation tillage, Baker^[13] noted that changing the relative volumes of subsurface drainage and surface runoff can also affect chemical concentrations in those carriers. In general, increasing infiltration increases the time for beginning of surface runoff, which in turn reduces the concentrations of chemicals at the soil surface (shown as a thin mixing zone in Fig. 1) and therefore in surface runoff. However, the effect of conservation tillage on infiltration is time dependent. For the first storm after any tillage there is usually less runoff from the tilled soil, although on an annual basis, conservation (or less) tillage often results in lower total surface runoff volumes.

Cropping

As with tillage, the hydrologic interactions between cropping systems and subsurface drainage that affects water quality involve how cropping affects the timing, route, and volume of infiltration (and hence the relative volumes of subsurface drainage and surface runoff). A major difference between perennial crops such as forages, and row crops such as corn and soybeans, is the volume and timing of ET demands. In general, with higher, more consistent ET, perennial crops would have lower total drainage volumes. A bigger effect of cropping would be the effect of needed chemical applications and their potential losses. For example, the large amounts of N needed (added, recycled, and/or fixed) in a continuous corn or corn-soybean rotation mean there are usually high NO₃-N concentrations in the soil profile, and hence in subsurface drainage when it occurs. However, for grasses and alfalfa, NO₃-N concentrations in subsurface drainage are much lower.^[14,15]

Controlled Drainage

In areas where subsurface drainage exists, controlling the timing of outflows has been suggested as one method to

reduce chemical losses. This controlled drainage could reduce losses by reducing both subsurface volumes and chemical concentrations. The potential for reduced concentrations is probably the greatest for NO₃-N, where the process of denitrification would reduce NO₃ to N gases in the soil profile where high water tables and the presence of organic matter drives the system anaerobic. The results summarized from 125 site years of data from North Carolina^[16] showed that controlled drainage reduced subsurface drainage volumes on an average of 30% compared to uncontrolled drainage systems. Reductions in N and P lost with subsurface drainage were 45% and 35%, respectively. While almost all the reduction of P loss was due to decrease in drainage volume; for N, reductions in NO₃-N concentrations also contributed to the reduction.

CONCLUSION

The total effect of surface drainage on surface water resources receiving drainage from agricultural lands involves the relative volumes of surface runoff and subsurface drainage and the relative concentrations of sediment, nutrients, pesticides, and bacteria. In general, the existence of subsurface drainage increases infiltration rates, which delays and reduces the volume of surface runoff. For pollutants lost mostly with surface runoff, which include sediment, NH₄-N, P, pesticides, and bacteria, not only is the volume of the carrier reduced but also the concentrations. This is because delayed runoff and more water moving through the surface-mixing zone reduce the amounts of contaminants at the soil surface available to interact with added water and overland flow. Thus the only real water quality negative to subsurface drainage is the increased volume of water moving thorough the soil profile carrying the soluble unadsorbed NO₃-N anion. The use of improved in-field N management in the way of rate, method, and timing of N applications has some potential to reduce this problem.^[17] However, other practices such as controlled drainage or construction/reconstruction of wetlands may be needed to provide some NO₃-N reduction treatment. Alternatively, reducing the amount of row crops grown on subsurface-drained lands could have a large impact although the economics of doing that would be quite negative.

REFERENCES

1. Baker, J.L.; Campbell, K.L.; Johnson, H.P.; Hanway, J.J. Nitrate, phosphorus and sulfate in subsurface drainage water. *J. Environ. Qual.* **1975**, *4*, 406-412.
2. Gilliam, J.W.; Baker, J.L.; Reddy, K.R. *Water Quality Effects of Drainage in Humid Regions*; Drainage Monograph; Am. Soc. Agron.: Madison, 1999.
3. Bengston, R.L.; Carter, C.E.; Morris, H.F.; Bartkiewicz, S.A. The influence of subsurface drainage practices on

- nitrogen and phosphorus losses in a warm, humid climate. Trans. ASAE **1988**, *31*, 729–733.
4. Gambrell, R.P.; Gilliam, J.W.; Weed, S.B. Nitrogen losses from soils of the North Carolina coastal plain. J. Environ. Qual. **1975**, *4*, 317–323.
 5. Baker, J.L.; Melvin, S.W.; Agua, M.M.; Rodecap, J. *Collection of Water Quality Data for Modeling the Upper Maquoketa River Watershed*; Paper No. 99-2222; ASAE: St. Joseph, 1999.
 6. Johnson, H.P.; Baker, J.L. *Field-to-Stream Transport of Agricultural Chemicals and Sediment in an Iowa Watershed: Part I. Data Base for Model Testing (1976–1978)*; Rep. No. EPA-600/S3-82-032; USEPA Environ. Res. Lab.: Athens, 1982.
 7. Johnson, H.P.; Baker, J.L. *Field-to-Stream Transport of Agricultural Chemicals and Sediment in an Iowa Watershed: Part II. Data Base for Model Testing (1979–1980)*; Rep. No. EPA-600/S3-84-055; USEPA Environ. Res. Lab.: Athens, 1984.
 8. Southwick, L.M.; Willis, G.H.; Bengston, R.L.; Lormand, T.J. Effect of subsurface drainage on runoff losses of atrazine and metolachlor in Southern Louisiana. Bull. Environ. Contam. Tox. **1990**, *45*, 113–119.
 9. Southwick, L.M.; Willis, G.A.; Bengston, R.L.; Lormand, T.J. Atrazine and metolachlor in subsurface drain water in Louisiana. J. Irrig. Drain. Eng. **1990**, *116*, 16–23.
 10. Bengston, R.L.; Southwick, J.M.; Willis, G.H.; Carter, C.E. The influence of subsurface drainage practices on herbicide losses. Trans. ASAE **1990**, *33*, 415–418.
 11. Crane, S.R.; Moore, J.A.; Grismer, M.E.; Miner, J.R. Bacterial pollution from agricultural source: A review. Trans. ASAE **1983**, *26*, 858–866.
 12. Culley, J.L.B.; Phillips, P.A. Bacteriological quality of surface and subsurface runoff from manured sandy clay loam soil. J. Environ. Qual. **1982**, *11*, 155–158.
 13. Baker, J.L. Hydrologic effects of conservation tillage and their importance relative to water quality. In *Effects of Conservation Tillage and Groundwater Quality: Nitrates and Pesticides*; Logan, T.J., Davidson, J.M., Baker, J.L., Overcash, M.R., Eds.; Lewis Publishers, Inc.: Chelsea, 1987; 113–124.
 14. Baker, J.L.; Melvin, S.W. Chemical management, status, and findings. In *Agricultural Drainage Well Research and Demonstration Project—Annual Rep. and Project Summary*; Iowa Dep. of Agric. and Land Stewardship and Iowa State University: Ames, 1994; 27–60.
 15. Weed, D.A.J.; Kanwar, R.S. Nitrate and water present in and flowing from root-zone soil. J. Environ. Qual. **1996**, *25*, 709–719.
 16. Evans, R.O.; Skaggs, R.W.; Gilliam, J.W. Management practice effects on water quality. In Proc. Irrig. Drain. 1990 Natl. Conf. ASCE Irrig. Div., Durango, 1990; 182–191.
 17. Baker, J.L. Limitations of improved nitrogen management to reduce nitrate leaching and increase use efficiency. Optimizing nitrogen management in food and energy production and environmental protection: Proceedings of the 2nd international nitrogen conference on science and policy. Sci. World **2001**, *1* (S2), 10–16.

Drought: Drought Resistance

Graeme C. Wright

Department of Primary Industries, Queensland Department of Primary Industries, Kingaroy, Queensland, Australia

Nageswararao C. Rachaputi

Queensland Department of Primary Industries, Kingaroy, Queensland, Australia

Abstract

This entry concentrates on crop production, and in that context drought is a term used to define circumstances in which growth or yield of the crop is reduced because of insufficient water supply to meet the crop's water demand. During the 1960s to 1980s, most of the drought research was dedicated to understanding the mechanisms of survival and growth under drought conditions. It is only after that attention has been given to recognizing the complex nature of drought and to separating the productivity of crop plants under drought from survival mechanisms. Drought resistance in modern agriculture requires sustainable and economically viable crop production, despite stress. However, plant survival can be a critical factor in subsistence agriculture, where the ability of a crop to survive drought and produce some yield is of critical importance.

INTRODUCTION

Drought can be considered as a set of climate pressures that can result from a combination of heat, aerial, or soil water deficits as well as salinity. The diversity of drought created from these phenomena has led to the selection of numerous types of resistance mechanisms that operate at different levels of life organization (molecule, cell, organ, plant, and crop). Decades of research have been dedicated to the understanding of these mechanisms, with a premise that the improved understanding would contribute to the long-term improvement of plant and crop production under drought conditions.^[1]

DISCUSSION

Hence, in the context of agricultural production, drought resistance in a crop can be best defined in terms of the optimization of crop yield in relation to a limiting water supply.^[2] Multitudes of options exist for farmers to alleviate the effects of drought on crop yield, depending on the probability of drought. These can be categorized into management and genetic options, although they can be integrated into a package to manage drought in the target environment. The basis of most management technologies adopted by farmers revolves around optimizing water conservation and its subsequent utilization by the crop. Examples include the use of deep tillage to increase rainfall infiltration, stubble retention to minimize soil evaporation,^[3] and intercropping.^[4] Genetic options include the use of the best locally adapted varieties or landraces as well

as relaying and intercropping varieties with varying phenology to exploit differences in timing and severity of drought patterns.^[5]

Advances in crop drought resistance and associated improvements in productivity under drought are most likely to come from genetic improvement programs that can apply the wealth of knowledge created over the 20th century. As Richards^[6] states, however, it will never be possible to *overcome* the effects of drought, any progress is likely to be slow, and the gains will only be small. The following sections will therefore concentrate on opportunities and emerging technologies for the improvement of drought resistance in crop plants, using genetic enhancement.

DROUGHT RESISTANCE TRAITS

Levitt^[7] has proposed a terminology for drought resistance and its subdivision into different categories based on different mechanisms. These three categories of drought resistance have been widely accepted, and they continue in a slightly modified form to provide a framework for evaluating potential traits for use in crop breeding programs.^[8–10] They are drought escape, dehydration postponement, and dehydration tolerance.

Drought Escape

Matching the phenology to the expected water supply in a given target environment has been an important strategy for improving productivity in water-limited environments.^[11] In most crop species, there is large genetic variability in

phenological traits, and these traits are highly heritable and amenable to selection in large-scale breeding programs.^[12] Matching phenology has proven to be a highly successful approach in environments that have a high probability of end-of-season drought stress pattern.^[13]

Dehydration Postponement

Crops use a variety of mechanisms to maintain turgor in leaves and reproductive structures, despite declining water availability. They can effectively regulate water loss from leaves via stomatal control, with large varietal differences in stomatal conductance in response to leaf water potential recorded in cereals^[14] and grain legumes.^[15]

Production of abscisic acid (ABA) has been implicated as a mechanism behind stomatal control.^[16] Other benefits of ABA, including maintenance of turgor in wheat spikelets and subsequent grain set, have also been reported.^[17] Subsequent stimulation of research activity into the use of the ABA trait in crop breeding programs has followed. However, Blum^[18] concluded that while ABA is undoubtedly involved in plant response to drought stress and even perhaps in desiccation tolerance, its value in the context of drought resistance breeding is questionable.

Osmotic adjustment (OA) has been reported as an important drought-adaptive mechanism in crop plants where solutes accumulate in response to increasing water deficits, thereby maintaining tissue turgor despite decreases in plant water potential.^[19] OA has been shown to maintain stomatal conductance, photosynthesis, and leaf expansion at low water potential,^[20] as well as reducing flower abortion^[21] and improving soil-water extraction despite declining water availability.^[22,23] More research has confirmed that OA is directly associated with grain yield (GY) in a number of crops, including wheat,^[24,25] sorghum,^[26] and chickpea.^[27]

Dehydration Tolerance

The ability of cells to continue metabolism at low-leaf water potential is known as dehydration tolerance. Membrane stability and the associated leakage of solutes from the cell^[28] provide one measure of the ability of crop/genotypes to withstand dehydration. Sinclair and Ludlow^[29] have suggested that the lethal water potential is a key measure of dehydration tolerance, with significant variation among crops and cultivars observed. Some stages of the plant's life cycle are less susceptible to large reductions in water content. For instance, prior to establishment of a large root system, seedlings may often survive large reductions in water content.^[30] Protection against lethal damage in seedlings and seeds is correlated with the accumulation of sugars and proteins.^[31] Proline has also been implicated in cellular survival of water deficits and has also been involved in OA. Its role as a selection trait for enhanced drought resistance has been questioned.^[32] Although the

work on drought resistance mechanisms has produced a few promising leads, their application in practical breeding programs has been limited.

HEIRARCHY OF DROUGHT RESISTANCE TRAITS

Richards^[6] suggests there are two major principles to consider when identifying critical traits to use in breeding programs aimed at improving productivity under drought, namely, the influence of the trait in relation to time scale and to level of organization.

Time Scale

Traits that influence drought resistance in crops can span a wide range of time scales. Short-term responses to water deficit, e.g., include many of the processes covered earlier (heat-shock proteins, stomatal closure, OA, and ABA), which Passioura^[2] suggests are often primarily concerned with "metabolic housekeeping." Although these processes are important, they tend to be associated with crop survival rather than with events that influence crop productivity. At the other end of the time scale are longer-acting processes such as control of leaf area development, which can be modulated by the crop to adjust water supply to prevailing demand.^[33] It is not always clear which processes are operating to control these balances, but presumably hormonal signals are involved. Passioura^[2] argues that researchers need to distinguish between traits linked to short-term responses that might be important for overall drought resistance and those that are unimportant when integrated over longer time scales.

Level of Organization

The capacity of the trait to influence yield is related to the level of organization (molecule–cell–organ–plant–crop) in which the trait is likely to be expressed.^[34] Richards^[6] cites an example that despite the doubling of crop yields since 1900, the rate of leaf photosynthesis, which is expressed at the cellular-organ level, has remained the same or decreased. Increases in leaf area during this period have been largely responsible for the yield increases. It is concluded that the closer the trait is to the level of organization of the crop the more influence it will have on productivity (Fig. 1).

DROUGHT RESISTANCE TRAITS IN TERMS OF THE PASSIOURA WATER MODEL

Passioura^[2,34,35] argues that there are no traits that confer global drought resistance. Also, given the earlier discussion which clearly suggests that short-term responses to drought stress operating at the cellular level may have no bearing on

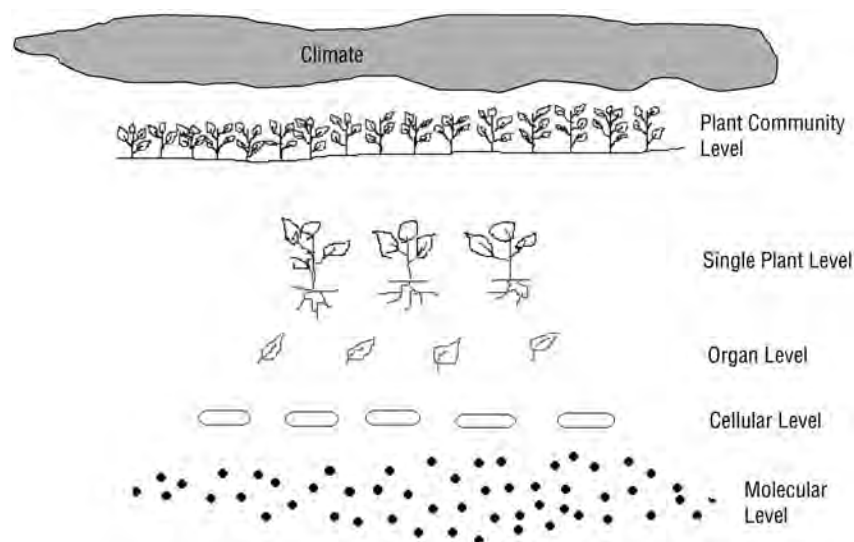


Fig. 1 The hierarchy of processes leading to crop yield.

the yield of water-limited crops, crop productivity is best analyzed from the top down. Here the thinking has shifted from understanding defense mechanisms for survival to applying this knowledge to optimize economic productivity under a given water-limited condition.

Passioura proposed that when water is the major limit, GY is a function of the amount of water transpired by the crop (T), the efficiency of use of this water in biomass production (WUE), and the proportion of biomass that is partitioned into grain, or harvest index (HI), thus:

$$GY = T \times WUE \times HI \quad (1)$$

It was argued that individual traits could be assessed in terms of their contribution to each of these functional yield components and thereby increase yield under drought. The identity has provided a framework to more critically identify and evaluate important drought resistance traits. The following examples demonstrate the utility of this approach.

In a number of crops, improvement in the WUE trait has been achieved via selection for carbon isotope discrimination,^[36] or correlated surrogate measures.^[37–39] Readers are referred to several reviews for more details on this subject.^[6,40,41]

In maize, increased partitioning to the grain (or higher HI) has been brought about by using an ideotype selection index that focused on a reduction in anthesis to silking interval.^[42] GY of the final selections increased by 108 kg/ha/cycle and came about by an increase in HI, with no change in final biomass relative to the parents.

Wright et al.^[43] proposed that estimates of each of the water model components for a peanut crop could be derived from simple and low-cost measurements of total and pod biomass at harvest, from estimates of WUE from correlated measures of specific leaf area, and by reverse engineering the TDM component of the water model, such

that $T = TDM/TE$. A selection study is being conducted on four peanut populations, in which a selection index approach utilizing T, WUE, and HI is assessing the value of these traits in a large-scale breeding program in India and Australia.^[44]

CONCLUSION

Although drought is commonly referred to as “prolonged water deficit” periods during a crop’s life, it is indeed a complex syndrome with various climate pressures operating together in infinite combinations, resulting in significant reductions in crop performance. Historically, options for managing agricultural drought have revolved around management techniques such as deep tillage, mulching, intercropping, etc. However, options for drought management will increasingly be based on the genetic improvement of crops for targeted environments. With genetic options, matching phenology to water availability in target environments has proven to be highly a successful approach. Although our understanding of drought resistance mechanisms has improved significantly, it is essential to distinguish the traits linked to short term responses from those which are important when integrated over longer time scales.

REFERENCES

1. Schulze, E.D. Adaptation mechanisms of non-cultivated arid-zone plants: Useful lessons for Agriculture. In *Drought Research Priorities for the Dryland Tropics*; Bidinger, F.R., Johansen, C., Eds.; ICIRSAT: Patancheru, 1988; 159–177.
2. Passioura, J.B. Drought and drought tolerance. In *Drought Tolerance in Higher Plants: Genetical, Physiological, and*

- Molecular Biological Analysis*; Belhassen, E., Ed.; Kluwer Academic Publishers: The Netherlands, 1996; 1–6.
3. Unger, W.P.; Jones, O.R.; Steiner, J.L. Principles of crop and soil management procedures for maximising production per unit Rainfall. In *Drought Research Priorities for the Dry Land Tropics*; Bidinger, F.R., Johansen, C., Eds.; ICIRSAT: Patancheru, 1988; 97–112.
 4. Natarajan, M.; Willey, R.W. The effects of water stress on yield advantages of intercropping systems. *Field Crops Res.* **1986**, *13*, 117–131.
 5. Nageswara Rao, R.C.; Wadia, K.D.R.; Williams, J.H. Intercropping of short and long duration groundnut genotypes to increase productivity in environments prone to end-of-season drought. *Exp. Agr.* **1990**, *26*, 63–72.
 6. Richards, R.A. Defining selection criteria to improve yield under drought. In *Drought Tolerance in Higher Plants: Genetical, Physiological, and Molecular Biological Analysis*; Belhassen, E., Ed.; Kluwer Academic Publishers: The Netherlands, 1996; 71–78.
 7. Levitt, J. *Response of Plants to Environmental Stresses*; Academic Press: New York, 1972.
 8. Turner, N.C. Crop water deficits: A decade of progress. *Adv. Agron.* **1986**, *39*, 1–51.
 9. Turner, N.C.; Wright, G.C.; Siddique, K.H.M. Adaptation of grain legumes (pulses) to water-limited environments. *Adv. Agron.* **2001**, *71*, 193–231.
 10. Turner, N.C. Drought resistance: A comparison of two frameworks. In *Management of Agricultural Drought: Agronomic and Genetic Options*; Saxena, N.P., Johansen, C., Chauhan, Y.S., Rao, R.C.N., Eds.; Oxford and IBH: New Delhi, 2000.
 11. Subbarao, G.V.; Johansen, C.; Slinkard, A.E.; Rao, R.C.N.; Saxena, N.P.; Chauhan, Y.S. Strategies for improving drought resistance in grain legumes. *Crit. Rev. Plant Sci.* **1995**, *14*, 469–523.
 12. Jackson, P.; Robertson, M.; Cooper, M.; Hammer, G. The role of physiological understanding in plant breeding; From a breeding perspective. *Field Crops Res.* **1996**, *49*, 11–37.
 13. Siddique, K.H.M.; Loss, S.P.; Regan, S.P.; Jettner, R. Adaptation of cool season grain legumes in Mediterranean-type environments of south-western Australia. *Aust. J. Agr. Res.* **1999**, *50*, 375–387.
 14. Wright, G.C.; Smith, R.C.G.; Morgan, J.M. Differences between two grin sorghum genotypes in adaptation to drought stress. III. Physiological responses. *Aust. J. Agr. Res.* **1983**, *34*, 637–651.
 15. Flower, D.J.; Ludlow, M.M. Contribution of osmotic adjustment to the dehydration tolerance of water-stressed pigeonpea (*Cajanus cajan* (L.) millsp.) leaves. *Plant Cell Environ.* **1986**, *9*, 33–40.
 16. Davies, W.J.; Tardieu, F.; Trejo, C.L. How do chemical signals work in plants that grow in drying soil? *Plant Physiol.* **1994**, *104*, 309–314.
 17. Morgan, J.M. Possible role of abscisic acid in reducing seed set in water-stressed wheat plants. *Nature (Lond.)* **1980**, *285*, 655–657.
 18. Blum, A. Crop responses to drought and interpretation of adaptation. In *Drought Tolerance in Higher Plants: Genetical, Physiological, and Molecular Biological Analysis*; Belhassen, E., Ed.; Kluwer Academic Publishers: The Netherlands, 1996; 57–70.
 19. Morgan, J.M. Osmoregulation and water stress in higher plants. *Annu. Rev. Plant Physiol.* **1984**, *35*, 299–319.
 20. Jones, M.M.; Rawson, H.M. Influence of the rate of development of leaf water deficits upon photosynthesis, leaf conductance, water use efficiency, and osmotic potential in sorghum. *Physiol. Plant.* **1979**, *45*, 103–111.
 21. Morgan, J.M.; King, R.W. Association between loss of leaf turgor, abscisic acid levels and seed set in two wheat cultivars. *Aust. J. Plant Physiol.* **1984**, *11*, 143–150.
 22. Morgan, J.M.; Condon, A.G. Water use, grain yield and osmoregulation in wheat. *Aust. J. Plant Physiol.* **1986**, *13*, 523–532.
 23. Wright, G.C.; Smith, R.C.G. Differences between two grin sorghum genotypes in adaptation to drought stress. II. Root water uptake and water use. *Aust. J. Agr. Res.* **1983**, *34*, 627–636.
 24. Morgan, J.M. Osmoregulation as a selection criterion for drought tolerance in wheat. *Aust. J. Agr. Res.* **1983**, *34*, 607–614.
 25. Blum, A.; Mayer, J.; Gozlan, G. Associations between plant production and some physiological components of drought resistance in wheat. *Plant Cell Environ.* **1983**, *6*, 219–225.
 26. Santamaria, J.M.; Ludlow, M.M.; Fukai, S. Contributions of osmotic adjustment to grain yield in *Sorghum bicolor* (L.) Moench under water-limited conditions. I. Water stress before anthesis. *Aust. J. Agr. Res.* **1990**, *41*, 51–65.
 27. Morgan, J.M.; Rodriguez-Maribona, B.; Knights, E.J. Adaptation to water-deficit in chickpea breeding lines by osmoregulation: Relationship to grain-yields in the field. *Field Crops Res.* **1991**, *27*, 61–70.
 28. Blum, A. *Plant Breeding for Stress Environments*; CRC Press: Boca Raton, 1988.
 29. Sinclair, T.R.; Ludlow, M.M. Influence of soil water supply on the plant water balance of four tropical grain legumes. *Aust. J. Plant Physiol.* **1986**, *13*, 329–341.
 30. Chandler, P.M.; Munns, R.; Robertson, M. Regulation of dehydrin expression. In *Plant Responses to Cellular Dehydration During Environmental Stress*; Close, T.J., Bray, E.A., Eds.; American Society of Plant Physiologists: Rockville, 1993; 159–166.
 31. Close, T.J.; Fenton, R.D.; Yang, A.; Asghar, R.; DeMason, D.A.; Crone, D.E.; Meyer, N.C.; Moonan, F. Dehydrin: The protein. In *Plant Responses to Cellular Dehydration During Environmental Stress*; Close, T.J., Bray, E.A., Eds.; American Society of Plant Physiology: Rockville, 1993; 104–118.
 32. Hanson, A.D.; Hitz, W.D. Metabolic responses of mesophytes to plant water deficits. *Annu. Rev. Plant Physiol.* **1982**, *33*, 163–203.
 33. Mathews, R.B.; Harris, D.; Williams, J.H.; Nageswara Rao, R.C. The physiological basis for yield differences between four groundnut genotypes in response to drought. II. Solar radiation interception and leaf movement. *Exp. Agr.* **1988**, *24*, 203–213.
 34. Passioura, J.B. The interaction between physiology and the breeding of wheat. In *Wheat Science—Today and*

- Tomorrow*; Evans, L.T., Peacock, W.J., Eds.; Cambridge University Press: Cambridge, 1981; 191–201.
35. Passioura, J.B. Grain yield, harvest index and water use of wheat. *J. Aust. Inst. Agr. Sci.* **1977**, *43*, 117–120.
 36. Farquhar, G.D.; Richards, R.A. Isotopic composition of plant carbon correlates with water-use efficiency in wheat genotypes. *Aust. J. Plant Physiol.* **1984**, *11*, 539–552.
 37. Nageswara Rao, R.C.; Wright, G.C. Stability of the relationship between specific leaf area and carbon isotope discrimination across environments in peanuts. *Crop Sci.* **1994**, *34*, 98–103.
 38. Masle, J.; Farquhar, G.D.; Wong, S.C. Transpiration ratio and plant mineral content are related among genotypes of a range of species. *Aust. J. Plant Physiol.* **1992**, *19*, 709–721.
 39. Clark, D.H.; Johnson, D.A.; Kephart, K.D.; Jackson, N.A. Near infrared reflectance spectroscopy estimation of ^{13}C discrimination in forages. *J. Range Manag.* **1995**, *48*, 132–136.
 40. Subbarao, G.V.; Johansen, C.; Rao, R.C.N.; Wright, G.C. Transpiration efficiency: Avenues for genetic improvement. In *Handbook of Plant and Crop Physiology*; Pessarakli, M., Ed.; Marcel Dekker: New York, 1994; 785–806.
 41. Wright, G.C.; Rachaputi, N.C. Transpiration efficiency. In *Water and Plants (Biology) for the Encyclopedia of Water Science*; Stewart, B.A., Howell, T., Eds.; Marcel Dekker, Inc.: New York, 2003; 982–988.
 42. Fischer, K.S.; Johnson, E.C.; Edmeades, G.O. Breeding and selection for drought resistance in tropical maize. In *Drought Resistance in Crops with Emphasis on Rice*; International Rice Research Institute: Los Banos, 1983; 377–399.
 43. Wright, G.C.; Rao, R.C.N.; Basu, M.S. A physiological approach to the understanding of genotype by environment interactions—A case study on improvement of drought adaptation in groundnut. In *Plant Adaptation and Crop Improvement*; Cooper, M., Hammer, G.L., Eds.; CAB International: Wallingford, 1996; 365–381.
 44. Nigam, S.N.; Nageswara Rao, N.C.; Wright, G.C. Breeding for increased water-use efficiency in groundnut. In *New Millennium International Groundnut Workshop*, Shandong Peanut Research Institute, Qingdao, China, Sept 4–7; 2001; 1–2.

Drought: Precipitation, Evapotranspiration, and Soil Moisture

Joshua B. Fisher

Konstantinos M. Andreadis

Jet Propulsion Laboratory, California Institute of Technology, Pasadena, California, U.S.A.

Abstract

Drought is major hydrological disaster for human society and encompasses multiple hydrological components. There are numerous definitions of drought, many of which do not include transpiration or evaporation [evapotranspiration (ET)]. In this entry, we describe the role of ET in drought, and why ET is important to consider when assessing drought. We emphasize the dynamic roles of ET, precipitation, and soil moisture, with a focus on vegetation response during water stress. Finally, we conclude with a case study example of drought in Amazonia.

INTRODUCTION

Drought ranks as one of the most expensive natural disasters in terms of human welfare and food security. For example, in the United States, the annual cost of drought relief measures has been estimated between US\$6 billion and US\$8 billion. Droughts can cover very large areas and last for several years with the so-called megadroughts documented from medieval times.^[1] In general terms, drought is caused by extremes within the natural variability of climate but can be exacerbated by human activity (e.g., deforestation). The literature on drought is extensive, with definitions categorically ranging from meteorological (climatological, or atmospheric), agricultural, hydrologic, and socioeconomic (e.g., management based),^[2–5] but we focus here on the vegetation transpiration and evaporation [actual evapotranspiration (ET)] component of drought.^[6] From a vegetation perspective in general, physical drought is the drying of soil such that the overlying vegetation experiences physiological water stress manifested in a reduction of productivity, loss of leaves/needles, and, ultimately, mortality. As such, a given soil moisture content (SMC) would correspond to different classes of drought, depending on the ability of vegetation to adapt to decreased soil moisture. For instance, some species or stages of succession may have plants with deep roots that can tap deep sources of soil moisture, even the groundwater table, so the ability of these plants to withstand what would otherwise be considered a drought may, in fact, not be considered a drought for some time.^[7] Other species, however, may be poorly adapted to low levels of soil moisture through sparse root distribution, low water use efficiency, or high temperature sensitivity and thus may enter into a drought much more quickly than other, better adapted, species.

SMC is inherently coupled directly to precipitation (PPT) and ET, with PPT as the moisture input and ET as the moisture withdrawal; soil water-holding capacity acts as the intermediate “bucket” size if considering a bucket model of SMC change. Among these three variables (i.e., SMC, PPT, and ET), the end-members of SMC can be outlined as follows: 1) for two areas with equivalent PPT and ET, SMC may be different because of different SMC retention properties (e.g., sandy soils may hold less water than soil with more clay); 2) for two areas with equivalent soils and PPT, SMC may be different because of different ET; and 3) for two areas with equivalent soils and ET, SMC may be different because of different PPT. The same exercise can be applied for areas to have similar SMC (e.g., two areas with equivalent soils: one with high PPT and high ET and the other with low PPT and low ET). By this definition, and with reference to the title of this entry, ET can vary under drought or non-drought situations, although ET will go to zero under persistent and intense drought. Between the ET components, transpiration will go to zero as drought persists (assuming no deep water sources) because stomata is close to avoid water loss (assuming no leaky stomata),^[8] and plants will maintain respiration through carbon stores; evaporation from the soil surface will also go to zero as the soil dries out (assuming no hydraulic lift/redistribution from deep water sources).^[9] The role of ET in drought is particularly pertinent in already water-limited environments where increasing temperatures over time accelerates ET, which leads to greater drought severities.^[10]

Plants are typically able to withstand relatively short periods of SMC decline, so a given day with no PPT and high ET may not be considered a drought. Although there are different metrics of drought that take ET into account such as the widely used Palmer Drought Severity Index,^[2] which

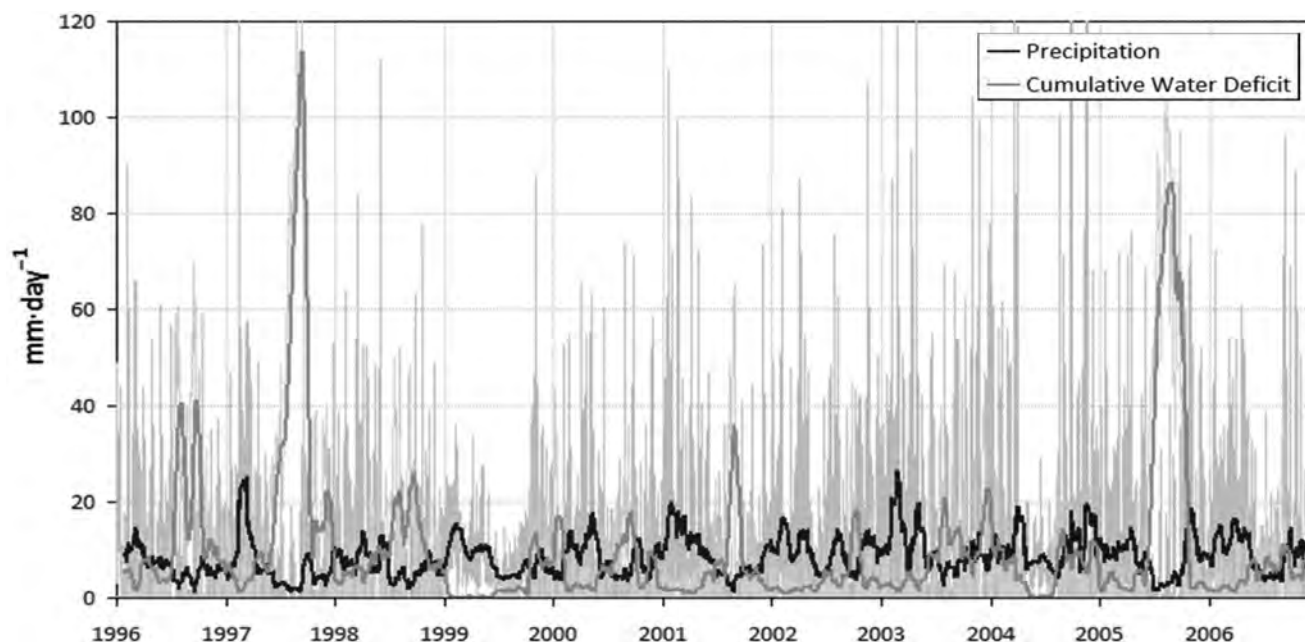


Fig. 1 PPT (black) and CWD (gray) over 10 years for a site in Amazonia, described in Phillips et al. The light colors are the daily values, and the dark colors are the 30-day moving averages.

Source: From Phillips, Aragão, et al.^[11]

uses the potential ET, it is the *cumulative water deficit* (CWD) that plants respond to—i.e., the summation of days in which the soil water deficit (SWD) is below a critical water stress threshold. SWD may be calculated as follows:

$$\begin{aligned} \text{If } \text{PPT} - \text{ET} > 0, \text{ Then } \text{SWD}_1 &= 0, \\ \text{Else } \text{SWD}_1 &= \text{P} - \text{ET} + \text{SWD}_0 \end{aligned}$$

where the subscripts indicate adjacent time steps. The maximum CWD (MCWD) reached during the time period of interest relative to the time-averaged climatological MCWD may be considered “drought.”^[11] Both the length of the CWD and the MCWD in a given period must surpass the long-term means of the two for that period to be considered a drought although a corresponding vegetation water stress response is also necessary.

An example of MCWD drought and vegetation response is shown in the *Science* paper by Phillips et al.,^[11] *Drought sensitivity of the Amazon rainforest*. Here, we describe how the measurements of anomalously low PPT indicated the possibility of an intense drought over Amazonia—the lowest PPT at the time, in fact, in the past 100 years. With few soil moisture measurements available, SWD was constructed from measured PPT, estimated ET using meteorological measurements,^[12,13] and measurements of the soil water-holding capacity. We calculated MCWD at 136 sites where we also observed vegetation response and determined that sites experiencing the greatest hydrologic drought as defined by MCWD also had the greatest vegetation response, specifically mortality and biomass loss. It can be seen at one of the sites, e.g., Fig. 1, that PPT varies seasonally, as does CWD, but in 2005 (also 1997 and 2001)

CWD spikes well beyond the mean CWD for the 10-year record. In this analysis, the length and peak (MCWD) of the CWD spike vary by site and are proportional to the vegetation response (e.g., mortality).

CONCLUSION

Evaporation and transpiration are critical components to drought although many traditional definitions of drought ignore ET. Vegetative drought inherently implies a response from vegetation, and this response must be calculated from a CWD, as $\text{ET} > \text{PPT}$ adds up over time, drying out the soil. In the absence of soil moisture measurements, soil moisture may be calculated from PPT and ET. Even with soil moisture measurements, the understanding of the bioclimatic variables controlling ET helps elucidate and predict how drought will change given changes in the controlling factors.

ACKNOWLEDGMENT

This entry was written by the Jet Propulsion Laboratory, California Institute of Technology, under a contract with the National Aeronautics and Space Administration.

REFERENCES

1. Cook, E.R.; Woodhouse, C.A.; Eakin, C.M.; Meko, D.M.; Stahle, D.W. Long-term aridity changes in the western United States. *Science* **2004**, *306* (5698), 1015–1018.

2. Palmer, W.C. *Meteorological Drought Rep.*; U.S. Department of Commerce: Washington, 1965; 1–58.
3. Dracup, J.A.; Lee, K.S.; Paulson, E.G., Jr. On the definition of droughts. *Water Resour. Res.* **1980**, *16* (2), 297–302.
4. Wilhite, D.A.; Glantz, M.H. Understanding the drought phenomenon: The role of definitions. *Water Int.* **1985**, *10*, 111–120.
5. McKee, T.B.; Doesken, N.J.; Kleist, J. The relationship of drought frequency and duration to time scales. In *Eight Conference on Applied Climatology*; American Meteorological Society: Anaheim, 1993; 179–184.
6. Fisher, J.B.; Whittaker, R.H.; Malhi, Y. ET Come Home: A critical evaluation of the use of evapotranspiration in geographical ecology. *Global Ecol. Biogeogr.* **2012**, *20*, 1–18.
7. Fisher, R.A.; Williams, M.; Da Costa, A.L.; Malhi, Y.; Da Costa, R.F.; Almeida, S.; Meir, P. The response of an Eastern Amazonian rain forest to drought stress: Results and modelling analyses from a throughfall exclusion experiment. *Global Change Biol.* **2007**, *13* (11), 2361–2378.
8. Fisher, J.B.; Baldocchi, D.D.; Misson, L.; Dawson, T.E.; Goldstein, A.H. What the towers don't see at night: Nocturnal sap flow in trees and shrubs at two AmeriFlux sites in California. *Tree Physiol.* **2007**, *27* (4), 597–610.
9. Dawson, T.E. Hydraulic lift and water use by plants: Implications for water balance, performance and plant-plant interactions. *Oecologia* **1993**, *95* (4), 565–574.
10. Andreadis, K.M.; Lettenmaier, D.P. Trends in 20th century drought over the continental United States. *Geophys. Res. Lett.* **2006**, *33*, L10403.
11. Phillips, O.L.; Aragão, L.E.O.C.; Lewis, S.L.; Fisher, J.B.; Lloyd, J.; López-González, G.; Malhi, Y.; Monteagudo, A.; Peacock, J.; Quesada, C.; van der Heijden, G.; Almeida, S.; Amaral, I.; Arroyo, L.; Aymard, G.; Baker, T.R.; Bánki, O.; Blanc, L.; Bonal, D.; Brando, P.; Chave, J.; Oliveira, A.C.A.; Cardozo, N.D.; Czimczik, C.I.; Espejo, J.; Feldpausch, T.; Freitas, M.A.; Higuchi, N.; Jiménez, E.; Lloyd, G.; Meir, P.; Mendoza, C.; Morel, A.; Neill, D.; Nepstad, D.; Patiño, S.; Peñuela, M.C.; Prieto, A.; Ramírez, F.; Schwarz, M.; Silveira, M.; Thomas, A.S.; ter Steege, H.; Stropp, J.; Vásquez, R.; Zela-zowski, P.; Dávila, E.A.; Andelman, S.; Andrade, A.; Chao, K.-J.; Erwin, T.; Di Fiore, A.; Honorio, E.; Keeling, H.; Killeen, T.; Laurance, W.; Cruz, A.P.; Pitman, N.; Vargas, P.N.; Ramírez-Angulo, H.; Rudas, A.; Salamá, R.; Silva, N.; Terborgh, J.; Torres-Lezama, A. Drought sensitivity of the Amazon rainforest. *Science* **2009**, *323* (5919), 1344–1347.
12. Fisher, J.B.; Tu, K.; Baldocchi, D.D. Global estimates of the land-atmosphere water flux based on monthly AVHRR and ISLSCP-II data, validated at 16 FLUXNET sites. *Remote Sens. Environ.* **2008**, *112* (3), 901–919.
13. Fisher, J.B.; Malhi, Y.; de Araújo, A.C.; Bonal, D.; Gamo, M.; Goulden, M.L.; Hirano, T.; Huete, A.; Kondo, H.; Kumagai, T.; Loescher, H.W.; Miller, S.; Nobre, A.D.; Nouvellon, Y.; Oberbauer, S.F.; Panuthai, S.; von Randow, C.; da Rocha, H.R.; Rouspard, O.; Saleska, S.; Tanaka, K.; Tanaka, N.; Tu, K.P. The land-atmosphere water flux in the tropics. *Global Change Biol.* **2009**, *15*, 2694–2714.

Earthworms

Patrick J. Bohlen

MacArthur Agro-Ecology Research Center, Lake Placid, Florida, U.S.A.

Abstract

Earthworms are arguably the most important soil animals due to their well-recognized role in organic matter breakdown, increased nutrient turnover, and soil structural development. Assessing the role of earthworms in soil requires an understanding of the characteristics of their different ecological groups and the way in which they interact with soil characteristics, such as soil texture and organic matter content, over different scales of time and space. Earthworms can process large amounts of litter and soil in many productive ecosystems, and their activity is often associated with faster nutrient turnover rates. Their effects on nutrient cycling and organic matter turnover are mediated by their interaction with soil microorganisms. Their effects on soil structure are usually beneficial, such as improving soil aggregation, but in some cases are detrimental, such as increasing soil compaction and bulk density. The majority of studies examining the effects of earthworms on plant growth have reported that earthworms enhance plant growth. There are instances in which earthworms might be considered pests, but in general, they are considered to greatly benefit soil fertility. Agricultural practices influence earthworm populations, which generally decrease in response to increasing tillage or decreasing organic matter. Earthworms exhibit varying sensitivity to pesticides, heavy metals, and other soil pollutants and have potential for use in land reclamation and processing of organic wastes.

INTRODUCTION

Earthworms are perhaps the most important soil animals in terms of their influence on organic matter breakdown, soil structural development, and nutrient cycling, especially in productive ecosystems. Aristotle called them the “intestines of the earth” and the eminent 19th century biologist, Charles Darwin, spent many years observing their major influence on humus formation and soil transport.^[1] A vast body of scientific literature on earthworms has been published over the past several decades, which has greatly expanded our understanding of their role in ecological systems and has identified gaps in knowledge and suggested promising areas for further research. Despite this extensive literature, we lack knowledge of many aspects of earthworm biology and ecology and their long-term effects on soils and ecosystems.^[2,3]

EARTHWORM BIOLOGY AND CLASSIFICATION

Earthworms are terrestrial annelids with bilateral symmetry and corresponding external and internal segmentation. They have a thinly pigmented cuticle bearing setae on all segments except for the first two. All earthworms are hermaphroditic. Their gonads are situated in specific segments, which vary among different taxa. When sexually mature, they develop a swollen area of the epidermis called a clitellum. This region produces a cocoon in which one or more eggs are deposited, and then, the cocoon is passed

over the anterior segments and deposited in or on the soil. The young ones develop within the cocoon, and the newly hatched worms resemble the adults. The time to hatching and reproductive maturity vary widely among different earthworm species and are influenced by environmental factors.

Earthworms are classified within the phylum Annelida and the class Oligochaeta, which consists of as many as 36 families worldwide. About two-thirds of Oligochaete families comprise aquatic or semiaquatic worms, and the remaining families comprise mostly or exclusively terrestrial worms or earthworms. There are over 3500 known earthworm species, and it is estimated that the global total may be twice that number. Distinct taxonomic groups have arisen on every continent except Antarctica, and some groups are distributed throughout the world.

EARTHWORM ECOLOGY

Ecological Groups

Earthworm species can be grouped according to behavioral, morphological, or physiological adaptations that enable different species within earthworm communities to partition available resources. The three main ecological groups are termed epigeic, anecic, and endogeic.^[2,3] Epigeic worms feed on plant litter, dwell on the soil surface or within the litter layer, tend to be heavily pigmented, and are small to medium sized. Anecic worms feed on plant litter and soil,

live in permanent nearly vertical burrows, and are dorsally pigmented and large. Endogeic species are soil feeders, are not heavily pigmented, form extensive horizontal burrow systems, and range in size from small to large. Endogeic worms have been further divided into polyhumic, mesohumic, and oligohumic groups, which are separated, respectively, by the decreasing importance of organic matter in their diet and increasing size. Earthworm species do not always fall neatly into these three main categories and may even exhibit traits of different groups at different life stages or under different environmental conditions.

Earthworm Communities

Earthworm communities generally consist of one to six species. The relative abundance and species composition of earthworm communities depend on soil type, topography, and vegetation and are also influenced by land use history and earthworm biogeography. Earthworms account for the majority of animal biomass in soil in a wide range of productive ecosystems, from temperate grasslands, pastures, and forests to tropical pastures, savannas, and rain-forests, and many temperate and tropical agroecosystems. They generally do not occur in deserts and arid grasslands or in extreme alpine or boreal habitats. Earthworms are often absent from strongly acidic forest soils with poor litter quality, such as some northern coniferous forests.

Many earthworm communities consist of invasive exotic species. In North America, where approximately 100 native earthworm species have been described, at least 45 exotic species have been introduced. Lumbricid earthworms of European origin dominate many North American agroecosystems. A worldwide survey of earthworms in tropical regions reported that a total of 51 exotic and 151 native species commonly occur in tropical agroecosystems.^[4]

EFFECTS OF EARTHWORMS ON SOIL PROPERTIES

The influence of a given earthworm species on soil properties depends on the ecological group to which that species

belongs to. For example, the large vertical burrows of anecic earthworms, such as the common nightcrawler *Lumbricus terrestris*, can facilitate preferential flow of water through the soil profile and thereby increasing the transport of water, nutrients, and agricultural chemicals into deeper soil layers.^[5,6] Epigeic species facilitate the breakdown and mineralization of surface litter and can completely transform organic soil layers into a layer of earthworm casts. Anecic species incorporate surface litter deeper into the soil profile and can also bring soil from deeper soil horizons to the surface, which over a long period of time can change the mineralogy of surface soil. Endogeic species can facilitate a more thorough mixing of surface organic matter with mineral soil, and in some instances, in tropical soils the species can form compact casts that increase soil bulk density and decrease water infiltration.

Effects on Organic Matter Breakdown and Nutrient Cycling

Earthworm activity accelerates the decomposition of plant litter, increases rates of nutrient transformation and plant nutrient uptake, improves soil aggregation and porosity, and enhances water infiltration and solute transport. Earthworms can consume and incorporate large amounts of organic matter into soil (Table 1). Such mixing is largely responsible for the formation of mull soils in which surface organic horizons are thoroughly mixed with underlying mineral soil.

Earthworms have a major influence on nutrient cycling in many ecosystems. Although they generally increase the mineralization of soil carbon, earthworms can also decrease mineralization of carbon by contributing to the formation of stable soil aggregates in which carbon is protected from further breakdown. Direct fluxes of nutrients through earthworm biomass can be considerable. For example, the turnover of nitrogen (N) through earthworm tissues can be up to 150 kg N/ha/yr.^[3] Relative to surrounding soil, earthworm casts contain elevated amounts of nutrients including inorganic N. As a consequence, earthworms can greatly enhance the mineralization of N and can stimulate other N transformations such as denitrification.^[14] Much of the influence of earthworms on nutrient cycling can be

Table 1 Amount of organic matter ingested or incorporated into soil by earthworm populations in different environments.

Ecosystem	Location	Type of organic matter	Amount consumed or incorporated (kg/ha)	References
Maize field	United States	Maize residues	840	[7]
Orchard	England	Apple leaves	2000	[8]
Mixed forest	England	Canopy tree leaves	3000	[9]
Oak forest	Japan	Oak leaves	1071	[10]
Alfalfa field	United States	Alfalfa residues	1220	[11]
Tallgrass prairie	United States	Total organic matter	740–8980	[12]
Savanna	Ivory Coast	Total organic matter	1300	[13]

attributed to their interaction with soil microorganisms. Ecosystems with large earthworm populations tend to be characterized by faster cycling and bacterially dominated communities as compared to slower cycling and fungally dominated systems without earthworms.

Earthworms can also influence hydrologic fluxes of nutrients. By increasing bypass flow of infiltrating water, earthworm burrows can increase the amount of N and other nutrients leaching from the soil profile.^[5] Alternatively, earthworms can reduce the amount of nutrients lost in surface runoff by increasing the rates of water infiltration into the soil.^[15]

Effects on Physical Properties of Soil

The effect of earthworms on soil structure results from the net outcome of their feeding and burrowing activities. Earthworms ingest soil particles and organic matter, mixing these two fractions together and egesting them as surface or subsurface casts. Estimates of annual rates of production of earthworm casts range from less than 5 to over 250 Mg/ha in various ecosystems (Table 2). Once egested, soil in casts can be eroded due to the impact of rainfall or can form stable soil aggregates through a variety of stabilizing mechanisms.^[16]

Earthworms generally improve the aeration and porosity of soil through formation of burrows and by increasing the proportion of large aggregates in the soil, and their effects are especially important in poorly structured or reclaimed soil. By increasing the rates of water infiltration, earthworms can reduce the amount of surface runoff. Alternatively, earthworms can increase erosion by removing the protective cover of surface litter, increasing surface sealing,

and depositing surface casts, which can be carried down-slope during heavy rains. Some tropical species actually increase soil bulk density and decrease infiltration by producing compact soil casts. In general, the effects of earthworms on soil structure are considered to benefit soil fertility.

Effects on Plant Growth

The majority of studies examining the influence of earthworms on plant growth have reported that earthworms stimulate plant growth, although some studies have reported no effect or even a negative effect of earthworms on plant growth.^[2,17] Earthworms have been shown to increase the production of shoots and grain in a variety of field trials and greenhouse experiments. Introduction of earthworms into reclaimed polders in the Netherlands and in pastures in New Zealand resulted in large increases in forage quantity and quality.^[18,19] Beneficial effects of earthworms on plant growth may be due to increased nutrient and water availability, improved soil structure, and stimulation of microorganisms or formation of microbial products that enhance plant growth or possibly through direct production of plant growth promoting substances.^[17]

Undesirable Effects of Earthworms

Despite the documented and purported beneficial effects of earthworms on nutrient dynamics, soil structure, and fertility, some aspects of earthworm activities are considered undesirable.^[2] These include removing and burying surface residues that would otherwise protect soil surfaces from erosion; producing fresh casts that increase erosion and surface sealing; depositing castings on the surface of lawns and golf greens or irrigation ditches where they are a nuisance or in pastures where they interfere with haying operations; dispersing weed seeds in gardens and agricultural fields; transmitting plant or animal pathogens; increasing losses of soil N through leaching and denitrification; and increasing soil carbon loss through enhanced microbial respiration. Studies of earthworm invasions of north temperate forests in North America indicate that such invasion may have deleterious effects of native plant communities and contribute to loss of soil carbon.^[18] Clearly, there are instances in which earthworms can have undesirable effects, despite the fact that they are generally considered to benefit soil fertility.

Table 2 The amount of earthworm casts produced in different environments and various locations around the world.

Environment	Locations	Earthworm casts (Mg/ha)
Arable	Germany	92
Arable	Switzerland	18–81
Arable	Nigeria	50
Arable/floodplain	Egypt	268
Pasture	England	19–40
Pasture	England	28
Pasture	Germany	91
Pasture	Switzerland	22–42
Tallgrass prairie	United States	24–94
Grassland	India	4–78
Savanna	Columbia	10–50
Tropical forest	Ivory Coast	32–50
Temperate forest	Germany	7–60

Source: From Edwards & Bohlen^[2] and Lee.^[3]

EARTHWORMS AND SOIL MANAGEMENT

Influence of Agricultural Practices on Earthworm Populations

Earthworms can be affected favorably or adversely by different agricultural practices. Whereas heavy cultivation is

detrimental to earthworm populations, reduced and no-tillage agricultural practices promote their growth. Organic amendments, such as animal or green manures, also stimulate the growth of earthworm populations. Inorganic fertilizers may benefit earthworm populations by increasing the production of crop residues, but their effects are not as great as the effects of organic fertilizers, and the long-term application of inorganic fertilizers may adversely affect earthworm populations because of soil acidification or other changes in the soil environment. Liming can benefit earthworm populations in some instances.

Earthworms and Soil Reclamation

Introduction of appropriate earthworm species or encouraging natural population through the addition of suitable amendments can increase the rate of soil improvement and genesis of soil structure of reclaimed land. The introduction of European earthworms into pastures in New Zealand and Australia, as well as in Dutch polders, greatly facilitated improvements in soil structure and plant productivity.^[19,20] There has also been some success in introducing earthworms into reclaimed mining sites and in reclaimed peat with beneficial effects on soil structural development, nutrient cycling, and productivity.^[21]

Effects of Pesticides and Soil Pollutants on Earthworms

Pesticides have widely varying effects on earthworm populations, with many chemicals having little or no toxicity toward earthworms and others exhibiting acute toxicity.^[22]

In general, the carbamate insecticides and soil fumigants are very or highly toxic to earthworms. Herbicides, in general, show low toxicity toward earthworms, although there are some exceptions. Organochlorine and organophosphorus insecticides have varying levels of toxicity.

Soil contamination with organic pollutants, heavy metals, and acid precipitation can be detrimental to earthworm populations. Most toxic organic soil pollutants are highly toxic to earthworms. Heavy metals vary in their toxicity toward earthworms and can have acute as well as sublethal effects that include declines in earthworm growth and reproduction. This is also probably true of other soil pollutants but has not been well studied. Heavy metals can accumulate in earthworm tissues, increasing the risk of transfer of heavy metals to higher trophic levels, because of the wide array of animals that prey upon earthworms. Earthworm species vary in their tolerance of acid soil conditions, but some reports have noted a decline in earthworm populations in response to acid deposition.

Earthworms and Waste Management

Because they can consume large amounts of organic matter, there is an increasing interest in using earthworms to

process or manage organic wastes in a process known as “vermicomposting.”^[23] There are a limited number of large-scale commercial vermicomposting facilities operating at various locations throughout the world. The resulting compost material has been shown to have remarkably beneficial effects on plant growth in horticultural mixes.

CONCLUSION

The importance of earthworms to soil formation and organic matter breakdown has been recognized for a long time, and a global perspective has emerged that recognizes the complexity of interactions between earthworms and soil physical, chemical, and biological properties under different environmental, climatic, and edaphic conditions. The majority of research on earthworms has taken place in agricultural systems, and there is a need for more research in forests, natural grasslands, and other wildland ecosystems. With increasing interest in biodiversity and ecosystem function, and concern over biological invasions and global change, it will be important to evaluate how the introduction and spread of exotic earthworm species influence the conservation of native flora and fauna and modify valued ecosystem services, such as carbon storage, over time.

REFERENCES

1. Darwin, C.R. *The Formation of Vegetable Mould, Through the Action of Worms, with Observations on Their Habits*; John Murray: London, 1881.
2. Edwards, C.A.; Bohlen, P.J. *Biology and Ecology of Earthworms*, 3rd Ed.; Chapman and Hall: London, 1996.
3. Lee, K.E. *Earthworms: Their Ecology and Relationships with Soils and Land Use*; Academic Press: New York, 1985.
4. Fragoso, C.; Kanyonyo, J.; Moreno, A.; Senapati, B.K.; Blanchart, E.; Rodriguez, C. A survey of tropical earthworms: Taxonomy, biogeography and environmental plasticity. In *Earthworm Management in Tropical Agroecosystems*; Lavelle, P., Brussaard, L., Hendrix, P., Eds.; CABI Publishing: New York, 1999; 1–26.
5. Domínguez, J.; Bohlen, P.J.; Parmelee, R.W. Earthworms increase nitrogen leaching to greater soil depths in row crop agroecosystems. *Ecosystems* **2004**, *7*, 672–685.
6. Edwards, W.M.; Shipitalo, M.J.; Traina, S.J.; Edwards, C.A.; Owens, L.B. Role of *Lumbricus terrestris* (L.) burrows on quality of infiltrating water. *Soil Biol. Biochem.* **1992**, *24*, 1555–1561.
7. Bohlen, P.J.; Parmelee, R.W.; McCartney, D.A.; Edwards, C.A. Effects of earthworms (*Lumbricus terrestris*) on the carbon and nitrogen dynamics of decomposing surface litter in corn agroecosystems. *Ecol. Appl.* **1997**, *7*, 1341–1349.
8. Raw, F. Studies on earthworm populations in orchards. I. Leaf burial in apple orchards. *Ann. Appl. Biol.* **1962**, *50*, 389–404.
9. Satchell, J.E. Lumbricidae. In *Soil Biology*; Burgess, A., Raw, F., Eds.; Academic Press: London, 1967; 259–322.

10. Sugi, Y.; Tanaka, M. Population study of an earthworm. In *Pheretima Sieboldi*. In *Biological Production in a Warm-Temperate Evergreen Oak Forest of Japan*; Kira, T., Ono, Y.H., Hosokawa, T., Eds.; University of Tokyo Press: Tokyo, 1978; 163–171.
11. Gallagher, A.V.; Wollenhaupt, N.C. Surface alfalfa residue removal by earthworms *Lumbricus terrestris* L. in a no-till agroecosystem. *Soil Biol. Biochem.* **1997**, *29*, 477–480.
12. James, S.W. Soil, nitrogen, phosphorus, and organic matter processing by earthworms in tallgrass prairie. *Ecology* **1991**, *72*, 2101–2109.
13. Lavelle, P. Consommation annuelle d'une population naturelle de vers de terre (*Millsonia anomala* omodes, acanthodrilidae: oligochaetes) dans la savanne de lamto (côte d'ivoire). In *Progress in Soil Zoology*; Vanek, J., Ed.; Academia Publishing House: Prague, 1975; 299–304.
14. Elliot, P.W.; Knight, E.; Anderson, J.M. Denitrification in earthworm casts and soil from pastures under different fertilizer and drainage regimes. *Soil Biol. Biochem.* **1990**, *22*, 601–605.
15. Sharpley, A.N.; Syers, J.K.; Springett, J.A. Effect of surface-casting earthworms on the transport of phosphorus and nitrogen in surface runoff from pasture. *Soil Biol. Biochem.* **1979**, *11*, 459–462.
16. Tomlin, A.D.; Shipitalo, M.J.; Edwards, W.M.; Protz, R. Earthworms and their influence on soil structure and infiltration. In *Earthworm Ecology and Biogeography in North America*; Hendrix, P.F., Ed.; Lewis Publishers: Chelsea, 1995; 159–184.
17. Brown, C.; Edwards, C.A.; Brussaard, L. How earthworms affect plant growth: Burrowing into the mechanisms. In *Earthworm Ecology*, 2nd Ed.; Edwards, C.A., Ed.; CRC Press: Boca Raton, 2004; 13–50.
18. Bohlen, P.J.; Scheu, S.; Hale, C.M.; McLean, M.A.; Migge, S.; Groffman, P.M.; Parkinson, D. Invasive earthworms as agents of change in north temperate forests. *Front. Ecol. Environ.* **2004**, *2*, 427–435.
19. Hoogerkamp, M.; Rogaar, H.; Eijssackers, H.J.P. Effect of earthworms on grassland on recently reclaimed polder soils in the Netherlands. In *Earthworm Ecology: From Darwin to Vermiculture*; Satchell, J.E., Ed.; Chapman and Hall: London, 1983; 85–106.
20. Stockdill, S.M.J. Effect of introduced earthworms on the productivity of New Zealand pastures. *Pedobiologia* **1982**, *24*, 29–35.
21. Scullion, J.; Malik, A. Earthworm activity affecting organic matter, aggregation and microbial activity in soils restored after opencast mining for coal. *Soil Biol. Biochem.* **2000**, *32*, 119–126.
22. Edwards, C.A.; Bohlen, P.J. The effects of toxic chemicals on earthworms. *Rev. Environ. Contam. T.* **1992**, *125*, 23–99.
23. Edwards, C.A.; Arancon, N.Q. The use of earthworms in the breakdown of organic wastes. In *Earthworm Ecology*, 2nd Ed.; Edwards, C.A., Ed.; CRS Press: Boca Raton, 2004; 345–380.

Eco-Agriculture

Norman Uphoff

Cornell University, Ithaca, New York, U.S.A.

Janice E. Thies

Department of Crop and Soil Sciences, Cornell University, Ithaca, New York, U.S.A.

Abstract

A number of related terms and practices get grouped under the rubric of “eco-agriculture” and its alter ego “agroecology.” While each approach has some distinctive characteristics and claims, all share a concern with the promotion and maintenance of *soil health* and with the *sustainability* of agricultural systems. Their holistic perspective emphasizes *connectedness* among elements within and between systems. Still, the processes and practices that these approaches favor are amenable to analytical treatment and for the most part to evaluation by accepted scientific methodologies. Trends in the 21st century—e.g., less arable land available per capita, lesser reliable water supplies for agriculture, continuing degradation of soil systems, and climate change—appear likely to make agricultural systems that were successful in the 20th century more amenable to, or even needing, revision in agroecological directions. Examples are given for profitable large-scale farming operations in Brazil, South Africa, and Pakistan that are moving agricultural production toward contemporary eco-agriculture.

INTRODUCTION

The designation “eco-agriculture” covers a number of related systems of agricultural practice, all of which are concerned with improving and/or maintaining the status and productivity of soil systems, while at the same time reducing negative environmental impacts. This term encompasses a dozen or more approaches to agricultural production that not only have some common elements but also some differences in emphasis or degree for certain practices.

A central concern in eco-agricultural approaches is the promotion and maintenance of *soil health*, sometimes characterized as “the life in the soil.” The health and fertility of soil systems are regarded as both a condition for and a result of eco-agricultural management. Another shared concern of all these approaches is *sustainability*.^[1] These approaches began focusing on sustainability challenges some time before they became a concern of policy makers and the general public.

A third concern in common is with the *connectedness* of agriculture, both with other sectors and of the many elements within agriculture. The sector is not regarded as isolated or autonomous. Reductionist modes of thinking do not dominate analysis and prescription. Rather, connections with the *natural environment*—to its soil, water, and biotic elements—and with *human beings*—their households, communities, and society—are identified and valued. The interdisciplinary subject of *soil ecology*^[2,3] is basic to all approaches to eco-agriculture.

Explicit consideration is given to the ways in which agricultural practice derives *benefits* from both the environment and society, while the *constraints* that emanate from each of these are also considered. The positive contributions that agriculture can make to natural ecosystems and to people’s well-being—when due attention is paid to ecological principles and opportunities—are highlighted. The *costs* that certain methods of agricultural management can impose on both the environment and humankind when the connectedness among natural and human factors is ignored or undermined are also considered.

The diverse but mostly compatible approaches within the domain of eco-agriculture are summarized in the following sections.

ORGANIC AGRICULTURE

Sir Albert Howard^[4,5] is generally regarded as the founder of the organic agriculture “movement,” based on years of research and experimentation in India. His propositions were reinforced by the writings of Lady Eve Balfour,^[6] which gave impetus to organic farming in the United Kingdom. In the United States, J.I. Rodale founded The Rodale Institute in 1930 to evaluate and demonstrate organic farming practices, launching the magazine *Organic Farming and Gardening* in 1942.

While this movement was often operationally defined by what is *not* done—no use of synthetic or inorganic fertilizers to increase the supply of plant nutrients in the soil, or

of agrochemical compounds to control pests and diseases—its main message was for farmers to provide organic matter to agricultural soils to enhance their functioning and fertility and to use beneficial insects and microbes and cultural control measures to manage pests and diseases. Organic agriculture is sometimes equated with eco-agriculture, but the latter is a more encompassing term.

AGROECOSYSTEMS/AGROECOLOGY

These perspectives have emphasized the operation of ecological principles and dynamics within agricultural production, parallel to and influenced by what occurs within the larger ecosystems within which agricultural activities take place.^[7] Both agroecosystems and agroecological analysis were proposed in the 1950s and 1960s, but by the 1990s, “agroecology” had become the more common rubric, in part because it came to cover a variety of approaches, some more biophysical and others more focused on social and economic concerns. Formative works on this subject include Altieri,^[8] Conway,^[9] and Gliessman.^[10]

ECO-AGRICULTURE

This term has been used to characterize the whole set of approaches considered in this entry; however, a specific use of the word has been introduced by McNeely and Scherr.^[11] These authors had respective institutional bases in the International Union for the Conservation of Nature and the Consultative Group for International Agricultural Research and have complementary disciplinary bases in conservation biology and agricultural economics. They have identified ways in which the respective aims of agricultural production and biodiversity conservation, as well as livelihood creation to benefit especially the poor, could be jointly achieved. This was seen as preferable to pitting these objectives against each other as has often occurred in both the conservation and agricultural development communities. A version of this known as *wildlife-friendly farming* in the United Kingdom has also been taken up in some other countries (<http://www.oursouthwest.com/farming/farming1.htm>).

SPECIFIC ECO-AGRICULTURAL SYSTEMS

A number of methodologies have combined certain practices that fall under the above-mentioned approaches. Each has encountered some controversy within academic and policy circles, but many practitioners have reported both yield and environmental advantages from these systems of practice:

- *Biodynamic agriculture* derives from the work of Rudolf Steiner, who, in a series of lectures in 1924 in what was then Germany (now Poland), proposed

organic preparations to enhance crop production along with green manures and cover crops, crop rotations, and other measures such as attention to astrological calendars. The association of biodynamic agriculture with Steiner’s philosophical and pedagogical beliefs gave this approach an aura of unscientific status, which limited its acceptance. A number of agronomic assessments have not supported claims of the methods’ superiority; however, there are national associations of biodynamic farmers in some 10 countries.

- *Biointensive agriculture* has been promoted by John Jeavons and others to attain maximum yields from a minimum area of land, seeking self-sufficiency and sustainability from given land areas.^[12] Much of the system derives from traditional agricultural knowledge and practices. More labor is required for double digging (deep digging) of the soil, intercropping, recycling organic materials, and making raised beds and compost. However, resulting yields can be very high. Biointensive agriculture can also include animals within the farming system although some versions focus solely on vegetable cropping. The effort to practice agriculture within a closed system, drawing on solar energy as the main or only external input, has made biodynamic farming controversial with many academic agronomists.
- *Permaculture* is an alternative agricultural system that establishes permanent agricultural areas along with permanent human settlements, seeking to capitalize on synergies among crops and animals. It utilizes polyculture and designs zonation and layers (stories) to organize productive relationships among crops, seeking to mirror naturally evolved ecological dynamics. The work of Smith^[13] was seminal for permaculture theory and practice, with the most widely used presentation being that by Mollison and Holmgren.^[14]
- *Natural farming* is the term describing a no-till, integrated cropping strategy developed and promoted by Masanobu Fukuoka in his influential book *One-Straw Revolution*.^[15] With the motto of “no machines, no fertilizer, and no chemicals,” Fukuoka was able to match or surpass average yields achieved with these purchased inputs although not the very highest yields obtained with them. This methodology was similar to what was called *nature farming* by Mokichi Okada in 1935, which also emphasized deep respect for nature. These practitioners have informed organic agriculture in Japan.
- *Holistic management* is similar to most of the above approaches but focuses on restoring degraded grasslands, managing land in concert with natural resources to reverse desertification and thereby achieve economic, environmental, and social benefits (<http://www.holisticmanagement.org/>). A summation of its practices was presented by Allan Savory.^[16] This strategy of intensifying and managing livestock to build up soil fertility is counterintuitive, but it has demonstrated empirical support.

TILLAGE-FOCUSED ECO-AGRICULTURE

The above-mentioned approaches have emphasized optimization in the mixes and management of *crops* (and/or live-stock), with attention to improved soil health and nutrient status. Other related approaches have focused on *reducing or eliminating tillage* as the basic feature of eco-agriculture. There is no inherent conflict between a focus on land preparation rather than on agronomic practices. However, this does lead to different presentations and somewhat different prescriptions:

- *Reduced till/zero till/no-till agriculture* represents an evolution of practice since the 1970s that has increasingly put into practice what soil scientists and farmers have known for a long time: that mechanical tillage has negative effects on soil health, structure, and function, disturbing and altering the soil biota.^[17] While there can be some short-term benefits of tillage—the most immediate being easier crop establishment and then weed control—there are adverse impacts on soil fertility and productivity over time: losses of soil carbon, shifts in the populations of soil organisms (numbers, diversity, and activity), and soil erosion. Consequently, increasing number of farmers have moved away from tillage, and there are more than 115 million hectares of agricultural land that are farmed without tilling.^[18] The movement to eliminate tillage has evolved into the following more comprehensive strategy of eco-agriculture.
- *Conservation agriculture (CA)* is a three-pronged strategy that starts with a *cessation of tillage*. It may be productive to break up subsurface hard pans by ripping where these have developed due to many years of intensive tillage, but this is not considered tillage, which mechanically changes soil structure repeatedly to create or prepare a seed bed. CA includes (in addition to minimized or no tillage) the *maintenance of permanent organic mulch cover on the soil* using crop residues and cover crops (including deep-rooted legumes) that halt erosion, improve water infiltration, and enhance soil nutrient contents and the *diversification of crop species grown in rotation or associations of crops* to enhance soil biodiversity and phytosanitation as well as to improve soil structure and function. The justification, sustainability, and spread of CA are presented by Kassam et al.,^[19] which show CA being practiced in all continents and ecosystems by large and small farmers alike. The Food and Agriculture Organization maintains a website on this beneficial soil management strategy (www.fao.org/ag/ca/).
- *Direct-seeded mulch-based cropping systems* is a version of CA developed by scientists with the *Centre Internationale de Recherche Agronomique pour le Développement* in Montpellier, France, from their work in various tropical countries although it is applied in

temperate ecosystems as well. Its methods for plant, soil, water, and nutrient management mimic the processes found in natural ecosystems for both productivity and sustainability.^[20]

SUSTAINABLE LAND USE

This is another focus of eco-agricultural theory and practice, usually driven more by practice than theory, particularly in response to crises of soil erosion or loss of soil fertility:

- *Landcare* is a programmatic approach to eco-agriculture, started in Australia in the late 1980s and formally recognized by the government in 1989, becoming a national movement with community, business, and government cooperation to control land degradation and deterioration of the hydrological cycle (http://en.wikipedia.org/wiki/Landcare_Australia). Better land use practices are expected to improve the “triple bottom line” of financial, environmental, and social gains. Landcare has spread to New Zealand, South Africa, the Philippines, the United States, Canada, and elsewhere. This is programmatic eco-agriculture with broad participation.
- *Agricultural stewardship* is a concept associated with the writings of Wes Jackson and Wendell Berry, having a broad philosophical as well as a pragmatic management orientation.^[21] Under this rubric, which originated in the United States but which is also gaining support elsewhere, most of the practices noted in the approaches summarized above can be promoted. Such an approach sustains often strong personal and organizational support such as seen in the Land Institute in Kansas (<http://www.landinstitute.org/>), the Prairie Plains Resource Institute in Nebraska (<http://www.prairieplains.org/>), and the Agricultural Stewardship Foundation in New York State (<http://www.agstewardship.org>). While there are scientific underpinnings for many of the practices recommended, this approach is driven more by observations and values than by scientific research.

Sustainable intensification, described as “low-input intensification” in a study commissioned by the European Parliament, is a restatement of eco-agricultural principles and strategies that has received endorsement from the scientific community and government circles in Europe.^[22,23] Previously, “intensification” was used to refer to *increased* use of external inputs to raise agricultural production. There is a growing understanding of how more intensive (vs. extensive) and ecologically informed management of available land, water, and other resources can increase farmers’ productivity in cost-effective ways.

THE FUTURE OF ECO-AGRICULTURE

There have been many conflicts surfacing between agriculture and the environment, with agriculture regarded as the greatest single threat to the conservation of biodiversity and with environmental protection seen as making agriculture less profitable and less viable. The different versions of eco-agriculture summarized here, however, see agriculture as part of the environment and as best practiced in consonance with ecological concepts and principles. Rather than accepting zero-sum (or negative sum) relations between the two domains, eco-agriculture undertakes to make this relationship a *positive sum* one, with aspirations of sustainability and equity alongside higher productivity.

During much of the 20th century, eco-agricultural approaches were contrasted unfavorably with “mainstream,” “modern,” or “conventional” agriculture, and most were regarded as marginal or alternative enterprises. Conditions in the 21st century, however, are likely to give more parity or even precedence to eco-agriculture. The continuing decline of arable land per capita, to one-third in 2050 compared to what it was in 1950, will reduce the comparative advantage of large-scale, mechanized agriculture, and higher energy costs will make energy-intensive production methods less competitive economically. Higher transportation costs will make long-distance trade in agricultural products less profitable.

Climate change together with growing scarcity and unreliability of rainfall will further disadvantage some of the more “thirsty,” monocropping technologies. Increasing concern with soil and water quality and with environmental health will make heavy reliance on inorganic fertilizers and agrochemical means of crop protection less tenable. The technologies and factor proportions that have prevailed in the 20th century are likely to warrant reconsideration in the future.

There is no longer justification for the stereotype of eco-agricultural methods as old-fashioned and suitable only for small-scale production. The following examples from three continents show how it can be innovative and forward-looking.

Brazil

The Balbo family's sugar enterprise began converting to organic production in 1986. By 1997, 14,000 ha of its 20,000 ha plantation were certified as “organic.” Three hundred smaller producers supply cane to the factory from another 24,000 ha. Together they grow and process 6.1 million tons of cane, producing 300,000 tons of crystal sugar and 320,000 m³ of fuel ethanol annually. The factory operation has become entirely self-sufficient for energy, and the farm operation will soon be meeting all of its energy needs, so that the whole enterprise is carbon neutral.^[24]

Both efficiency and profitability are achieved through recycling of organic matter, improvement of soil health,

integrated pest management, and biological control for insects and weeds. Sugar plantations are usually some of the least biodiverse areas to be found. Yet a five-year evaluation by researchers from the Brazilian national agricultural research authority EMBRAPA documented on the plantation's 79 km² over 300 species: 230 bird species, 26 kinds of amphibians, 17 species of reptiles, and 39 different mammals, including jaguars and pumas at the top of the food chain. Many of these are rare, and none were stocked; all returned naturally once the plantation was redeveloped to contain diverse habitats compatible with efficient sugarcane production. The Balbo enterprise supplies one-third of the world's consumption of organic sugar, and its products are exported to 60 countries (<http://www.nati vealimentos.com.br/pt-br/>).

South Africa

ZZ2, a multisystem farming conglomerate located primarily in the Limpopo Province, has grown from a subsistence farm to the largest producer of tomatoes in sub-Saharan Africa. It registered as a private company in 1966 and produces tomatoes, onions, avocados, apples, pears, and beef, primarily, exporting some of these commodities to Europe, the Middle East, Canada, and neighboring African countries. The bulk of its production is consumed domestically, however, as ZZ2 supplies 30% of the domestic market's tomatoes. The enterprise manages 60,000 ha of land, of which 10,000 ha are used to produce food. The remaining area is used for conservation through private game conservancies, to ensure water resources, and to fallow (rest) productive lands between crops as unimproved pasture.

In the late 1990s, ZZ2 began to experience declining yields despite increased use of fertilizers and pesticides and operating a seven-year pasture rotation between tomato crops. Various consultants helped the farm managers understand that their fundamental problem was degraded soil. In 2002, to restore and then maintain soil fertility, the firm embarked on a “nature farming” approach (Natuurboerdery[®]), applying more organic inputs and fewer agrochemicals. ZZ2 has moved away from industrial agricultural practices and toward the use of compost, compost teas, fermented plant extracts (FPEs), and effective microorganisms as alternative fertilizers and/or pesticides. It relies primarily on green technologies, using biocontrol agents such as *Trichoderma*, arbuscular mycorrhizal fungi, FPEs, and compost teas. Average tomato yields per hectare, which were declining steadily year by year, have stabilized at 77 tons ha⁻¹ in 2009; although when initially converting to Natuurboerdery, ZZ2's tomato production was reduced in the early years of the transition. The existing system is demonstrating both profitability and sustainability (http://www.zz2.biz/index.php?option=com_content&view=article&id=19&Itemid=23).

Pakistan

FarmAll Technology Ltd. has been one of the leading companies for agricultural mechanization in this country, importing and distributing tractors and other equipment from major foreign producers over since 1980s, as well as manufacturing Ford and Ursus tractors. FarmAll was the first operation to laser-level agricultural fields to reduce water requirements. It has developed machinery for the otherwise labor-intensive system of rice intensification (SRI), reducing both labor and water requirements while getting higher yield. Its 2009 trials on an 8 ha plot averaged 13 tons ha⁻¹ with 70% less water and labor. The transplanting and weeding machinery used for rice has been adapted to other crops (wheat, potatoes, and a wide range of vegetables) with similar cost savings and higher yield.

FarmAll is synthesizing SRI methods (younger seedlings, reduced plant populations, no flooding of the fields, and active soil aeration) with CA (permanent raised beds with soil cover) and with organic agriculture (precision application of organic matter, sometimes supplemented with small amounts of synthetic fertilizer). FarmAll's general manager Asif Sharif calls this "paradoxical agriculture" because the combination of practices enables the company to produce more agricultural output with reduced inputs by utilizing ecologically based principles (<http://www.farmalltechnology.com/>).

These enterprises utilize modern science and technology within a framework of theory and practice that utilizes ecological principles and perspectives with the goal of making agriculture both more productive and more sustainable. These complex farming systems represent a synthesis of agricultural and ecological knowledge that can revise and mitigate the widespread tensions between them.

REFERENCES

1. Pretty, J.N. Agricultural sustainability: Concepts, principles and evidence. *Philos. T. R. Soc. B* **2008**, *363*, 447–465.
2. Coleman, D.C.; Crossley, D.A.; Hendrix, P.F. *Fundamentals of Soil Ecology*; Amsterdam: Elsevier, 2004.
3. Whalen, J.K.; Sampedro, L. *Soil Ecology and Management*; CABI: Wallingford, 2010.
4. Howard, A. *An Agricultural Testament*; Oxford University Press: London, 1940.
5. Howard, A. *The Soil and Health: A Study of Organic Agriculture*; Devin-Adair: New York, 1947.
6. Balfour, E. *The Living Soil*; Faber & Faber: London, 1943.
7. Francis, C.; Lieblein, G.; Gliessman, S.; Breland, T.A.; Creamer, N.; Harwood, R.; Salomonsson, L.; Helenius, J.; Rickerl, D.; Salvador, R.; Wiedenhoeft, M.; Simmons, S.; Allen, P.; Altieri, M.; Flora, C.; Poincelot, R. Agroecology: The ecology of food systems. *J. Sustain. Agr.* **2003**, *22*, 99–118.
8. Altieri, M.A. *Agroecology: The Scientific Bases for Alternative Agriculture*; Westview Press: Boulder, 1985.
9. Conway, G. Agroecosystems analysis. *Agr. Admin.* **1985**, *20*, 31–55.
10. Gliessman, S.R. *Agroecology: Researching the Ecological Basis for Sustainable Agriculture*, Ecological Series No. 78; Springer-Verlag: New York, 1990.
11. McNeely, J.A.; Scherr, S.J. *Eco-Agriculture: Strategies to Feed the World and Save Wild Biodiversity*; Island Press: Washington, 2003.
12. Cox, C.; Jeavons, J. *The Sustainable Vegetable Garden: A Backyard Guide to Healthy Soil and Higher Yields*; Ten-Speed Press: Berkeley, 1999.
13. Smith, J.R. *Tree Crops: A Permanent Agriculture*; Harcourt: New York, 1919.
14. Mollison, B.; Holmgren, D. *Permaculture One*; Transworld Publishers: Milsons Point, 1978.
15. Fukuoka, M. *One-Straw Revolution*; Rodale Press: Emmaus, 1978.
16. Savory, A.; Butterfield, J. *Holistic Management: A New Framework for Decision-Making*; Island Press: Washington, 1999.
17. Faulkner, E.H. *Plowman's Folly*; Grosset & Dunlap: New York, 1943.
18. Friedrich, T.; Kassam, A.H.; Shaxson, F. Conservation agriculture. In *Agriculture for Developing Countries, Annex 2. Science and Technology Options Assessment (STOA) Project*; European Technology Assessment Group: Karlsruhe, 2009, Available at http://www.europarl.europa.eu/stoa/publications/studies/stoa2008-02_annex_2_en.pdf (accessed March 2011).
19. Kassam, A.H.; Friedrich, T.; Shaxson, F.; Pretty, J. The spread of conservation agriculture: Justification, sustainability and uptake. *Int. J. Agric. Sustain.* **2009**, *7*, 292–320.
20. Séguin, L.; Bouzinac, S.; Husson, O. Direct-seeded tropical soil systems with permanent soil cover. In *Biological Approaches to Sustainable Soil Systems*; Uphoff, N., Ball, A., Fernandes, E.C.M., Herren, H., Husson, O., Laing, M., Palm, C., Pretty, J., Sanchez, P.A., Sanginga, N., Thies, J.E., Eds.; CRC Press: Boca Raton, 2006; 323–342.
21. Jackson, W.; Berry, W.; Colman, B. *Meeting the Expectations of the Land: Essays in Sustainable Agriculture and Stewardship*; North Point Press: Berkeley, 1984.
22. The Royal Society. *Reaping the Benefits: Science and the Sustainable Intensification of Global Agriculture*, Report of Commission chaired by Sir D. Baulcombe; The Royal Society: London, 2009.
23. Meyer, R. *Agricultural Technologies for Developing Countries*. European Parliament: Brussels, 2009. Available at <http://www.ital.fzk.de/deu/lit/2009/meye09a.pdf> (accessed March 2011).
24. McMahon, C.; Martin, A. Opportunity, cubed, for foreign growers who go organic: A sugar mill in Brazil found dividends beyond its dreams after making the switch. The reason? U.S. food makers can hardly meet demand. *Los Angeles Times*, 2006; A.29.

Ecology: Cycling of Carbon and Nitrogen

Alan J. Franzluebbers

U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
Watkinsville, Georgia, U.S.A.

Abstract

Carbon (C) is found throughout nature in a wide variety of forms and particularly in soil. Nitrogen (N) is an essential element of plants, animals, and microorganisms, which is a part of chlorophyll, enzymes, amino acids, and proteins, which are necessary for the growth and development of organisms. C and N occur in various forms and undergo transformations from one form to another. This entry reviews the environmental influences on these cycles.

INTRODUCTION

Carbon (C) and nitrogen (N) are two of the most important elements that affect the soil's productivity and environmental quality.^[1] C is found throughout nature in a wide variety of forms and particularly in soil as 1) complex organic compounds (e.g., $C_xH_{2x}O_x$) derived from living organisms; 2) carbonate minerals such as calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$); and 3) carbon dioxide (CO_2) and methane (CH_4) as decomposition end products. N is an essential element of plants, animals, and microorganisms, which is a part of chlorophyll, enzymes, amino acids, and proteins, which are necessary for growth and development of organisms. In soil, the quantity of N in organic matter and as clay-fixed ammonium (NH_4^+) far exceeds quantities in plant-available forms of nitrate (NO_3^-) and NH_4^+ .

C AND N CYCLES

C and N occur in various forms and undergo transformations from one form to another, primarily through biochemical manipulations involving enzymes.^[2,3] Enzymes are catalysts of very specific reactions that function either 1) intracellularly within plants, microorganism, or soil animals or 2) extracellularly in soil solution or attached to soil colloids.

The forms and fluxes of an element are commonly illustrated in a cycle following the principles of conservation of mass (i.e., elements are transferred from one molecule to another). The C and N cycles have global dimensions with terrestrial, aquatic, and atmospheric components of major significance.^[4,5] The sun initiates a chain of energy reactions, which drive elemental cycles. The elemental cycles of C and N interact closely with the water cycle, as water is a fundamental internal component of life and a major transport mechanism of nutrients.

Autotrophic fixation of atmospheric CO_2 by plants captures the energy of the sun within organic compounds via the process of photosynthesis (Fig. 1). Inorganic N is taken up by plant roots and synthesized into amino acids and proteins during plant development. Plants are eventually consumed by animals or microorganisms, transferring portions of this stored energy through biochemical processes into various cellular components. Once in soil, the C cycle is dominated by the heterotrophic process of decomposition, i.e., the breakdown of complex organic compounds into simple organic constituents. Mineralization is the complete decomposition of organic compounds into mineral constituents:



Immobilization of N occurs simultaneously with N mineralization when soil organisms consume inorganic N to meet the demands for new body tissue. Net N mineralization occurs when gross N mineralization exceeds that of N immobilization.

ENVIRONMENTAL INFLUENCES ON SOIL MICROBIAL ACTIVITY

The dominant organisms responsible for the decomposition of organic matter and associated mineralization of C and N are soil microorganisms, composed of bacteria, actinomycetes, fungi, and protozoa.^[6,7] Soil fauna also indirectly affect C and N cycling by 1) comminuting plant residues and exposing a greater surface area to soil microorganisms; 2) transporting plant and animal residues to new locations in the soil to facilitate decomposition, interaction with soil nutrients, or isolation from environmental conditions; 3) inoculating partially digested organic substrates with specific bacteria and enzymes; and 4) altering physical

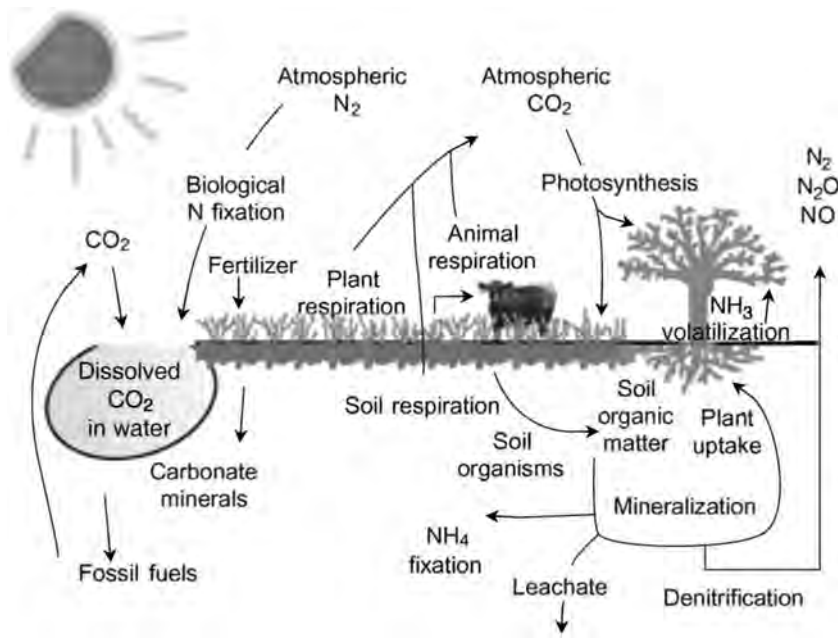


Fig. 1 Generalized diagram of the C and N cycles in soil.

characteristics of soil by creating burrows, fecal pellets, and distribution of soil particles that influence water, air, nutrient, and energy retention and transport. With suitable environmental conditions, soil microorganisms grow rapidly in response to the availability of organic substrates rich in C and N.

Soil Temperature

Temperature controls both plant and soil microbial activity, although not at the same level (Fig. 2). Plant and soil microbial activity are limited by low temperature, resulting in low photosynthetic potential, as well as low decomposition potential. For many plants, net photosynthetic activity is optimized between 20°C and 30°C, because at higher temperatures plant respiration consumes energy for maintenance. In many temperate soils, microbial activity is maximum near 30°C and decreases at higher temperatures. An intermediate temperature is often ideal for maximizing C retention in soil, where optimum plant activity outdoes soil microbial activity.

Soil Water Content

The greatest diversity of soil microorganisms is found under aerobic conditions, where maximum energy is obtained. However, there are a number of soil bacteria that thrive under anaerobic conditions, in which alcohols, acetic acid, lactic acid, and CH₄ become C end products via fermentation, and NO₃⁻ is converted to N gases (e.g., N₂, N₂O, and NO) via the process of denitrification. Soil C mineralization and net N mineralization are maximized at an

optimum balance between soil moisture and oxygen availability (Fig. 3). Significant denitrification occurs at water-filled pore space > 70%, resulting in what appears as reduced net N mineralization.

Soil Texture

Soil texture can influence the quantity of both C and N accumulation in soil and their potential mineralization. Potential C mineralization is often greater in coarse-textured than in fine-textured soils, which may be due to both increased microbial predation by soil fauna and greater accessibility of organic substrates in coarse-textured soils. Organic C and N can also be protected from decomposition when bound within soil aggregates, which are a coherent assemblage of primary soil particles (i.e., sand, silt, and clay) cemented through natural forces and substances derived from root exudates and soil microbial activity.

Spatial Distribution of Organic Substrates

Distribution of organic substrates in soil has a major impact on potential C and N mineralization. Potential C mineralization is often several fold greater in the rhizosphere (i.e., 0–5 mm zone surrounding roots) than in bulk soil. However, because of the high demand for N by plant roots and the stimulated soil microflora, net N mineralization is often lower in the rhizosphere because of immobilization of N.

The soil surface often contains greater quantities of organic matter than at lower depths due to surface deposition of plant residues, as well as highest plant root

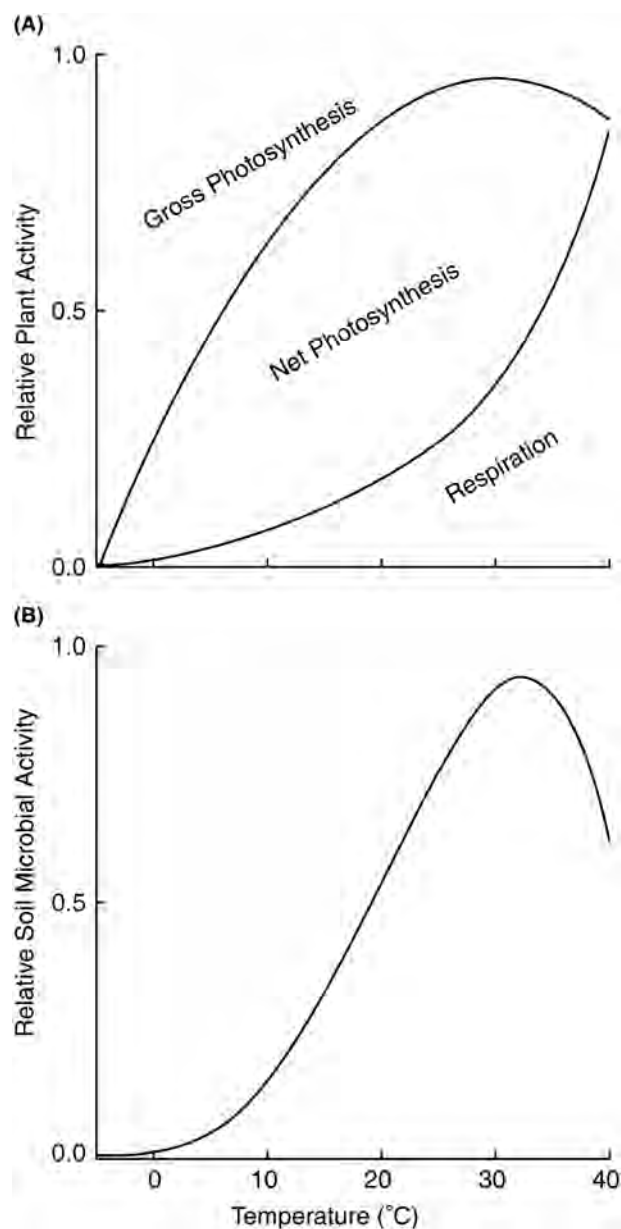


Fig. 2 Typical responses of plant and soil microbial activities to temperature.

activity. The soil surface usually undergoes the most extreme drying/wetting cycles and has the greatest exchange of gases, both of which contribute to enhanced soil microbial biomass and activity. Tillage of soil with traditional agriculture redistributes the organic substrates uniformly within the tillage layer, often resulting in stimulated soil microbial activity from the disruption of organic substrates protected within stable soil aggregates. Minimum soil disturbance with conservation tillage practices can reduce oxidation of soil organic matter and preserve more C within soil, which can have implications for potentially mitigating the greenhouse effect.^[8]

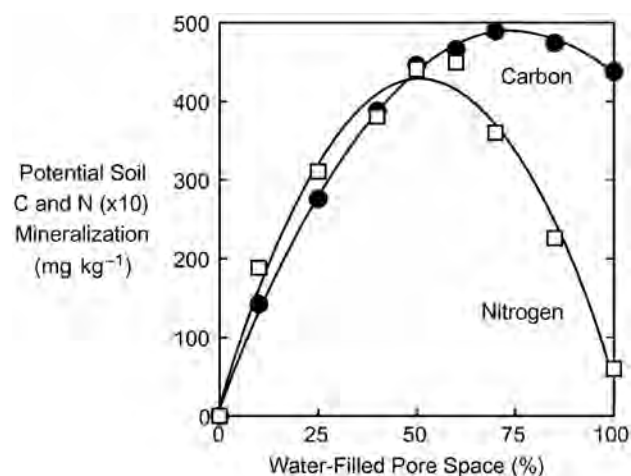


Fig. 3 Responses of potential soil C and N mineralization to water-filled pore space in Typic Kanhapludults in Georgia, the United States. Air-filled pore space is 100 – (water-filled pore space).

ORGANIC SUBSTRATE QUALITY

The quality of organic substrates has a major influence on the rate of decomposition and the transformations that occur in soil. Plant residues do not vary greatly in total C concentration on a dry weight basis (e.g., 37–47 mg g⁻¹), but do vary in the suite of C compounds, which determine its quality or conversely its resistance to degradation. The diversity of organic compounds attacked by soil microorganisms is extensive (e.g., organic acids, polysaccharides, lignins, aromatic and aliphatic hydrocarbons, sugars, alcohols, amino acids, purines, pyrimidines, proteins, lipids, and nucleic acids). Almost all naturally occurring organic compounds, and even most synthetic organic compounds, are susceptible to decomposition, given the appropriate environment, microbial community, and time.^[9,10] Generally, the primary components of plants can be categorized according to the relative rate of decomposition: rapid (sugars, starches, fats, and proteins), intermediate (cellulose and hemicellulose), and slow (lignin and lignocellulose). Young plants are of high quality and low resistance to decomposition, whereas with aging, lignin and polyphenolic concentrations increase, resulting in greater resistance to decomposition. Low N concentration of organic amendments usually results in temporary net N immobilization into microbial biomass, which grows rapidly in response to the availability of organic C. Soil microbial biomass maintains a C/N ratio of 10±5. Following a proliferation of microbial biomass that depletes the source of readily decomposable organic C/N in excess of microbial demands becomes mineralized and available for plant uptake (Fig. 4). In general, plant residues with C/N ratio > 40 will result in longer periods of net N immobilization.

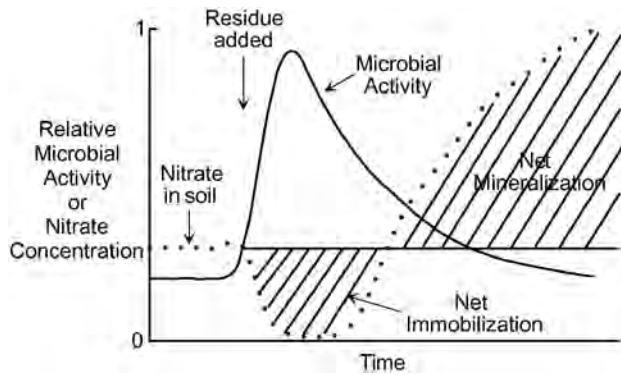


Fig. 4 Typical responses in soil microbial activity and soil NO_3^- concentration with the addition of plant residue relatively low in N concentration.

Soil Organic Matter

Soil organic matter is composed of a large variety of organic compounds that can be characterized in many ways. A useful separation of soil organic matter for modeling is based on the turnover times, whereby at least three pools can be defined: 1) active (composed of microbial biomass and light fraction material with a turnover time of < 1 year); 2) passive (composed of macro-organic matter and protected organic matter with a turnover time of 3–10 years); and 3) slow (stable humus fraction with a turnover time of > 100 years).

LOSSES OF N FROM SOIL

N cycling in soil is different from that of C because of the more numerous transformations that can occur upon mineralization to an inorganic form.^[11] Mineralization of N from organic matter results in NH_4^+ released into soil solution. In the presence of nitrifying bacteria, NH_4^+ is converted to NO_3^- , a process called nitrification. The fate of NO_3^- in soil depends on environmental conditions. Active plant growth in natural and agricultural systems would provide enough demand to re-immobilize N into organic forms. However, NO_3^- can be used as an electron acceptor in place of O_2 under anaerobic conditions with a supply of soluble organic C, resulting in gaseous loss of N to the atmosphere via denitrification. In temperate soils with a net negative charge on colloidal surfaces, NO_3^- can readily leach into the vadose zone and contaminate groundwater. In tropical soils with a net positive charge, NO_3^- can be retained on anion exchange sites. Contradicting behavior of the cations, NH_4^+ and NO_3^- , occurs with respect to clay mineralogy.

Humans, as well as roving animals, impose great demands on the C and N cycles. Management of agricultural and forest land for food and fiber removes nutrients from soil for consumption and utilization elsewhere. Return

of these nutrients to soil is possible when municipal and agricultural solid wastes and wastewater are applied to land. Losses of C and N from managed lands also occur through soil erosion, which transports nutrients via 1) water from overland flow into streams, lakes, and oceans; and 2) air from exposed land.

Volatilization of ammonia (NH_3) to the atmosphere is possible when NH_4^+ is exposed to alkaline soil conditions. Significant NH_3 volatilization can occur with surface application of urea fertilizer to non-acidic soils, from animal manures, and from green plant materials not incorporated into the soil.

FIXATION OF C AND N

Both C and N are biologically fixed from inorganic atmospheric forms to organically bound plant and microbial forms. As mentioned earlier, photosynthesis converts CO_2 from the atmosphere into organically bound forms in plants, algae, and cyanobacteria. Biological N_2 fixation is a unique transformation carried out by a number of bacteria, which convert N_2 gas into NH_3 for biological uptake. These bacteria are most prevalent in symbiotic relationships with plants, such as *Rhizobium* that forms nodules on the roots of clovers where the nitrogenase enzyme catalyzes the reaction. Fertilizer manufacturing converts N_2 gas into NH_3 in a similar manner without an enzyme, but rather large quantities of energy necessary to create the pressure required for the transformation.

Under certain conditions, both inorganic C and N can be chemically fixed in the subsoil. CO_2 forms carbonic acid in water, which can precipitate with the basic cations, Ca^{2+} , Mg^{2+} , and Na^+ , to form pedogenic carbonates. Inorganic C is most abundant in soils of the semiarid and arid regions. NH_4^+ can be fixed as non-exchangeable components of the lattice structure of 2:1-type clay minerals, which are especially prevalent in the subsoil of many younger soils.

REFERENCES

1. Follett, R.F.; Stewart, J.W.B.; Cole, C.V., Eds. *Soil Fertility and Organic Matter as Critical Components of Production Systems*; Spec. Publ. No. 19; Soil Science Society of America: Madison, 1987; 166 pp.
2. Burns, R.G. *Soil Enzymes*; Academic Press: London, 1978; 380 pp.
3. Tabatabai, M.A. Soil enzymes. In *Methods of Soil Analysis, Part 2. Microbiological and Biochemical Properties*; Weaver, R.W., Angle, J.S., Bottomley, P.S., Eds.; Book Series No. 5; Soil Science Society of America: Madison, 1994; 775–833.
4. Stevenson, F.J.; Cole, M.A. *Cycles of Soil: Carbon, Nitrogen, Phosphorus, Sulfur, Micronutrients*, 2nd Ed.; Wiley: New York, 1999; 427 pp.

5. Schlesinger, W.H. *Biogeochemistry: An Analysis of Global Change*, 2nd Ed.; Academic Press: San Diego, 1997; 588 pp.
6. Alexander, M.A. *Introduction to Soil Microbiology*, 2nd Ed.; Krieger: Malabar, 1991; 467 pp.
7. Sylvia, D.M.; Fuhrmann, J.J.; Hartel, P.G.; Zuberer, D.A., Eds. *Principles and Applications of Soil Microbiology*; Prentice-Hall: Upper Saddle River, 1998; 550 pp.
8. Lal, R.; Kumble, J.M.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Ann Arbor Press: Chelsea, 1998; 128 pp.
9. Tate, R.L., III. *Soil Organic Matter: Biological and Ecological Effects*; Wiley: New York, 1987; 291 pp.
10. Alexander, M. *Biodegradation and Bioremediation*; Academic Press: San Diego, 1994; 302 pp.
11. Stevenson, F.J. *Nitrogen in Agricultural Soils*; American Society of Agronomy: Madison, 1982; 940 pp.

Economics of Soil Management: U.S.

Jeffrey W. Hopkins

Resource Economics Division, U.S. Department of Agriculture—Economic Research Service (USDA-ERS), Washington, District of Columbia, U.S.A.

Abstract

Economists have a long and continuing tradition of evaluating soil management practices as they affect soil fertility and soil degradation. But economists realize that, in practice, soil management techniques are applied only to the point where the farmer (or society) is better off economically. Because economics considers all inputs and outputs of soil management, including off-site impacts, the environmental impacts of soil management demand constant study as well. Therefore, an economic assessment of soil management must extend beyond the farm field to consider off-site effects on water, air quality, and wildlife habitat.

INTRODUCTION

The economics of soil management differs from the physical science of soil management—it looks at maximizing long-run economic return, rather than maximizing soil performance and quality. With regard to maximizing soil performance, the prescription from soil scientists is clear and can be summarized as follows: add organic matter, avoid most tillage and other activities that might erode or compact the soil, apply chemicals on an as-needed basis, and keep the ground covered between growing seasons; increase crop diversity across a landscape; and systematically monitor soil chemical and physical properties.^[1]

ECONOMIC EVALUATION OF SOIL MANAGEMENT PRACTICES

Soil Fertility Management

Starting in the 1950s, economists used static production functions to formulate soil fertility recommendations, relying heavily on data generated by soil scientists and agronomists on crop yield response to soil management practices.^[2–4] Economists have since sought improvement over the single-period crop soil management recommendation by searching for solutions that increase both existing and future profits. In cases in which a dynamic biological process is considered, such as fertilizer carryover,^[5] or crop response to management is highly variable,^[6] multi-period analysis has proven to be particularly important. In the 1990s, with advances in computer, positioning systems, and application technology, many producers, as well as scientists, have become interested in managing spatial variability within a field.^[7] Economic management of site-specific soil conditions requires significant knowledge about the variability of soil properties and how these impact

crop response. While the relative profitability of site-specific soil management is unproven, some studies have shown that modest improvements in economic return can be achieved.^[8,9]

Soil Degradation Management

Soil fertility research finds ways to increase yields; soil degradation research looks for soil management techniques to keep crop yield growth from leveling off. In the United States, water and wind erosion and soil compaction remain the primary soil degradation concerns^[10] despite a 38% decrease in cropland wind and water erosion from 1982 to 1997.^[11] Soil erosion decreases potential crop yield by depleting soil nutrients and topsoil depth. Some have argued that it is the latter effect that poses the greatest threat to future food security.^[12] Because the rate of soil genesis is so low, topsoil losses are often considered irreversible and, therefore, of particular concern. This argument has apparently been particularly persuasive in the United States, where conservation policy abandoned an earlier recommendation to limit degradation to “tolerable” nutrient losses and, instead, focused exclusively on limiting degradation to “tolerable” losses of soil depth.^[13] However, U.S. irreversible topsoil losses have been shown to be slight in aggregate,^[14] especially when viewed in relation to past yield increases from improvements in technology and management.

Off-Site Impacts of Soil Management

Except for doubts regarding the ability to continue to achieve productivity gains that doubled world grain harvests since 1960,^[15] conventional thinking among economists is that the downstream costs of poor soil management (borne by users of downstream surface water) are larger

than the combined costs of on-site productivity losses and future food security.^[16] To decrease the likelihood of both social and private costs from non-optimal soil management, U.S. conservation policy has promoted farmer adoption of soil-enhancing practices. These *best management practices* hold the promise of a win-win solution for farmers and society at large. However, farmer adoption is far from universal for any single practice.

ADOPTION OF SOIL MANAGEMENT PRACTICES IN THE UNITED STATES

The adoption of most agricultural innovations has been shown to depend on both *farm* and *farmer* characteristics. The experience of U.S. hybrid corn adoption is instructive in understanding why best management practice adoption may not be universal. While hybrid corn was especially quick to catch on in Iowa, the adoption of open-pollinated corn elsewhere was in part limited by the availability (different hybrids appeared in different parts of the country) and acceptability (some farmers were reluctant initially) of the technology.^[17] The adoption of soil-based best management practices is likewise limited to factors related to the farm (climate and soils) and the farmer (firm management and household characteristics). The following material examines soil performance characteristics and on-farm adoption patterns of three best management practices—namely conservation tillage, monitoring soil properties, and crop rotations.

Conservation Tillage

Conservation tillage is a system of crop production that leaves additional crop residue remaining on the field after harvest to decompose naturally. Conservation tillage can increase soil organic matter and, therefore, increase soil productivity by increasing moisture retention and improving soil tilth. It can also drastically decrease soil erosion levels compared to conventional tillage methods and similarly reduce some rates of nutrient loss, particularly phosphorus. While the adoption of practice (defined strictly as 30% or more of the soil surface covered with flat and standing residue) grew steadily through the 1970s until the mid-1990s and is growing for some crops, overall adoption has hovered at 35–37% of U.S. planted cropland, according to field surveys.^[18,19]

The adoption rate of conservation tillage follows a regional distribution, with the highest rate (about 50%) among corn and soybeans producers in the Corn Belt and lower adoption rates (about 10%) by cotton and small grain producers in the West.^[20] While the adoption of conservation tillage can result in stable or increased crop yields compared to conventional systems, as well as decreased input costs (primarily machinery, chemical, fuel, and labor costs), these savings are not always seen in practice.^[21]

Any absolute economic advantage is likely derived only after several years of use of the system and may still be slight unless multiplied over many acres. In 1995, conservation tillage adoption rates on farms with sales greater than \$100,000 were more than 50% higher than on smaller farms.^[22]

Monitoring Soil Properties

Monitoring soil chemical and physical properties through soil testing can provide valuable information to farmers, allowing them to more closely match soil conditions with soil requirements for the growing crop. Optimal use of monitoring technology allows farmers to match soil conditions with crop-growing requirements both temporally (such as splitting nutrient applications into two smaller doses rather than one large dose) and spatially (dividing a field into different management units depending on soil conditions). The adoption of soil monitoring varies by the soil property monitored (20% of all cropland is tested for nutrient levels) and is the greatest in high-value crops, particularly fruit, vegetables, and cotton production.^[20] Site-specific soil management systems include spatially monitoring crop yield and soil quality as well as spatially applying inputs. The on-farm adoption of site-specific management has been less widespread than the soil performance gains suggest, with less than 1% adoption across all farms in the United States. The adoption of some components, such as intensive soil sampling, has appealed to certain types of farms, however, with corn and soybean producers (11% adoption) and high sales farms (18% adoption for farms with sales greater than \$500,000) among the early adopters.^[23]

Crop Rotations

Crop rotations allow producers to increase soil organic matter and reduce soil erosion, although these soil performance improvements are most likely to occur when row crops are rotated with hay, meadow, or pasture. In 1997, while 80% of cropland was grown in some sort of rotation, only 3% of acreage followed the hay, meadow, or pasture rotation reported to maximize soil performance.^[20] The economic role of crop rotations may be different from the physical role. For corn producers, crop rotations are particularly popular with both very large farms (with more than \$250,000 in sales) and with part-time farmers.^[22] For large farms, rotations could play a role in managing busy planting and harvesting periods, while for part-time farmers the role may be in managing on-farm and off-farm employment. Low use of hay rotations for row crop producers may be an indication that few producers are willing to forego any row crop sales in favor of greater soil performance. Cover crops, which do not occupy a full growing season, can effectively increase soil organic matter and decrease erosion, while still allowing farmers to produce the crop giving highest economic return. Cover crops, however, are

used rarely outside of the South, where the practice is common with soybean production. In this system, some cash gain can be achieved in double-cropping winter wheat or rye.^[20]

IMPLEMENTING SOIL MANAGEMENT PROGRAMS IN THE UNITED STATES

In cases in which the off-site costs of suboptimal soil management are great, such as where soil and nutrient runoff cause water quality problems, additional economic incentives have been used to increase the adoption of a best management practice. For example, the Environmental Quality Incentive Program, administered by the U.S. Department of Agriculture, provides technical assistance and extension education, as well as cash payments to producers adopting certain best management practices. The program is unlike earlier soil management initiatives because it targets participation to maximize the environmental benefit per dollar spent on the program. In most cases, encouraging adoption first in those areas where off-site damages can be decreased cheaply is an improvement over programs that solely attempt to increase adoption in general. No matter how programs are targeted; however, increasing the existing size of the pool of adopters has come at increasing cost.

Because improved practices have costs as well as benefits, economists advise a farmer who invests in soil performance to do so selectively, with an eye toward balancing the costs of improving soil performance with the long-term economic opportunities afforded by the practice. In general, however, the cost to producers of an improvement in soil performance is usually more clear than the benefits, which are realized in the future. To address the uncertainty of benefits relative to costs, producers will continue to need information on how to profitably improve soil management.

REFERENCES

1. University of Minnesota Extension Service. *Soil Management*; BU-7390-GO, 2000.
2. Swanson, E.R. The static theory of the firm and three laws of plant growth. *Soil Science* **1963**, 95, 338–343.
3. Heady, E.O.; Dillon, J.L. *Agricultural Production Functions*; The Iowa State University Press: Ames, 1961.
4. Hall, H.H. Economic evaluation of crop response to lime. *Am. J. Agric. Econ.* **1983**, 65, 811–817.
5. Kennedy, J.O.S.; Whan, I.F.; Jackson, R.; Dillon, J.L. Optimal fertilizer carryover and crop recycling policies for a tropical grain crop. *Aust. J. Agric. Econ.* **1973**, 17, 104–113.
6. Antle, J.M. Sequential decision making in production models. *Am. J. Agric. Econ.* **1983**, 65, 282–290.
7. Sawyer, J.E. Concepts of variable rate technology with considerations for fertilizer application. *J. Prod. Agric.* **1994**, 7, 195–201.
8. Schnitkey, G.D.; Hopkins, J.W.; Tweeten, L.G. An economic evaluation of precision fertilizer applications on corn-soybean fields. In *Proceedings of the Third International Conference on Precision Agriculture*, Minneapolis, Minnesota, Jun 23–26, 1996; Robert, P.C., Rust, H.R., Larson, W.E., Eds.; American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America: Madison, 1996; 977–987.
9. Babcock, B.A.; Pautsch, G.R. Moving from uniform to variable fertilizer rate on Iowa corn: Effects on rates and returns. *J. Agric. Res. Econ.* **1998**, 23 (2), 385–400.
10. Syers, J.K. Managing soils for long-term productivity. In *Land Resources on the Edge of the Malthusian Precipice*; Greenland, D.J., Gregory, P.J., Nye, P.H., Eds.; CAB International: New York, 1998; 151–160.
11. U.S. Department of Agriculture. *Summary Report: 1997 National Resources Inventory; Natural Resources Conservation Services*; U.S. Department of Agriculture: Washington, 1999.
12. Carter, V.G.; Dale, T. *Topsoil and Civilization*; University of Oklahoma Press: Norman, 1974.
13. Hall, G.F.; Logan, T.J.; Young, K.K. Criteria for determining tolerable erosion rates. In *Soil Erosion and Crop Productivity*; Follett, R.F., Stewart, B.A., Eds.; ASA-CSSA-SSSA: Madison, 1985; 173–188.
14. Crosson, P. Future supplies of land and water for world agriculture. In *Population and Food in the Early Twenty-First Century: Meeting Future Food Demand of an Increasing Population*; Islam, N., Ed.; International Food Policy Research Institute: Washington, 1995; 143–159.
15. Mann, C. Crop scientists seek a new revolution. *Science* **1999**, 283, 310–314.
16. Trimble, S.W.; Crosson, P. U.S. soil erosion rates—Myth and reality. *Science* **2000**, 289, 248–250.
17. Griliches, Z. Hybrid corn: An exploration in the economics of technological change. *Econometrica* **1957**, 25 (4), 501–522.
18. Lal, R.; Kimble, J.M.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Sleeping Bear Press: Chelsea, 1998.
19. Padgett, M.; Newton, D.; Penn, R.; Sandretto, C. *Production Practices for Major Crops in U.S. Agriculture, 1990–1997*; Agricultural Statistical Bulletin No. 969; June Economic Research Service, USDA: Washington, 2000.
20. CTIC. *Crop Residue Management Executive Summary*; Conservation Technology Information Center: West Lafayette, 1998.
21. Day, J.C.; Sandretto, C.L.; McBride, W.D.; Breneman, V.E. *Conservation Tillage in U.S. Corn Production: An Economic Appraisal*; American Agricultural Economics Association Annual Meeting, Salt Lake City, Utah, August 2–5, 1998.
22. Soule, M. Soil management and the farm typology: Do small family farms manage soil and nutrient resources differently than large family farms? *Agric. Res. Econ. Rev.* **2001**, 30 (2), 179–188.
23. Daberkow, S.; McBride, W. Adoption of precision agriculture technologies by U.S. farmers. In *5th International Conference on Precision Agriculture*, Minneapolis, Minnesota, Jul 16–19, 2000.

Ecosystems: Life Attributes and Environment

Dennis R. Linden

Soil Water and Climate Department, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), St. Paul, Minnesota, U.S.A.

Abstract

Soil is the home for plant roots and organisms of a wide variety of types and sizes, and soil ecology is the science of linkages between life attributes and the life-supporting environment. Life attributes involve an individual organism or group of organisms, their interaction with other organisms or groups, and direct or indirect functions or reactions. Soil life is supported in a porous lattice of mineral matter, living and dead organic material, water and its dissolved and suspended constituents, and soil air. The spatial scale of concern for soil ecology might be the microscopic environment of a few unicellular organisms or the expansive reaches of a particular soil or vegetation type. The temporal scale of concern might be the microsecond of a chemical or biological reaction to the eons of soil-profile development. It is thus visualized as a wide-ranging discipline made even more complex by the intricacies of the dynamic soil environment.

COMPOSITION AND NATURE OF SOIL

Soils are composed of minerals, living and dead organic material, air, and water arranged in a complex matrix. The minerals and organic material form a semirigid lattice, which leaves voids that are occupied by water and air. Air and water are usually free to exchange so that the void space that is not occupied by water is occupied by air. Water is a bipolar molecule and is held, at strengths that decrease with distance, to the exposed surfaces of the lattice. Under a given temperature regime, it is the balance between air and water and the composition of the solution and the air that exerts the most control of the life-supporting soil environment.

LIFE ATTRIBUTES AND THE SOIL ENVIRONMENT

Some of the terms that describe life and life functions (life attributes), and terms that describe the controlling environment are listed in [Table 1](#). Attributes ranging from the presence of a single organism to a collection of organisms and reactions such as N fixation are shown as examples of the attribute side of ecology. Water content, pH, CO₂ concentration, and temperature are examples of the environmental limits of ecology.

An important example will clarify the definition of soil ecology. The process known as denitrification results in the conversion of nitrogen (N) from bound forms to nitrous oxides that are gaseous and easily lost from the soil environment. The lost N may have been beneficial to many organisms, especially plant roots, and

its loss may contribute to the emission of a greenhouse gas. This process, however, is a major source of energy for a particular group of soil organisms. Energy is released when hydrogen atoms bonded to N are replaced with oxygen, and when oxygen is removed to form gaseous nitrous oxide or N gas. This process requires both an aerobic step where oxygen from the air replaces the hydrogen bonded to N atoms and an anaerobic reaction where oxygen from nitrate is stripped away from the N atom in the absence of oxygen in the air. This complex sequence of microbial-driven biochemical reactions thus supports both an aerobic and an anaerobic group of organisms that may occupy only slightly different niches within short distances in a soil. Soil ecology may only be interested in the step-by-step process or in the overall outcome of the system.

THE DYNAMICS OF WATER AND AIR

Soil–water content is the most important soil environmental factor of the soil ecosystem. Water has a direct effect on most life attributes, and water content is often indirectly related to properties of the air and the semirigid matrix. The dynamic nature of soil water and its importance to life attributes are imbedded in the concept of “least limiting water content.” As soils become drier by evaporation or by root uptake and translocation, the water remaining within the soil is sorbed by roots or other organisms with more difficulty. Eventually, soils will become so dry that most life ceases or becomes dormant. Drier soils also strongly resist deformation that is necessary for root growth and displacement by burrowing animals. When soils are

Table 1 A partial list of terms and phrases defining life attributes, soil–environment conditions, and their linkages considered by the discipline of soil ecology.

Life attributes	Soil–environment
Organism	Water and air
Individual/species/genus/family	Water content/air content
Fungi/bacteria/algae/actinomycetes/protozoa	Water-filled pores
Insects/invertebrates	Wet/dry/moist/damp/arid/poorly drained/well drained/saturated/flooded
Nematodes/earthworms	Field capacity/water-holding capacity/wilting point/available water
Functional grouping	Tension/head/suction
Anaerobic/aerobic	Oxygen and carbon dioxide
Terrestrial/aquatic	Anaerobic/aerobic/aerated
Indigenous/invasive/native/peregrine	Enriched/depleted
Active/dormant	Soil
Dead/living	Class/landscape position/climate
Fauna/flora	Terrestrial/mineral soil/peat/bog Arid/humid
Interactions between organisms or groups	Wetland/swamp/upland
Diversity/hierarchical Parasitic/saprophytic	Agricultural/forest/grassland
Predator/prey	Urban/rural
Symbiotic/antagonistic	Heavy/light/tight/loose
Life functions	Sand/silt/clay
Consume/egest/death	Shrink/swell
Growth/cell division	Structure/aggregated/massive
Reproduction/mating	Tropical/temperate/tundra
Enlarge/expand/shrink/swell	Thermal
Burrow/crawl/fly	Temperature/frozen/thawed
Reaction (function performed)	Frigid/mesic/thermic/hyperthermic
Enzyme/enzyme activity	Chemical (solution)
Biochemical decomposition/mineralization/immobilization/fixation/ dissolution/sorption/adsorption/desorption	pH/basic/acidic
Respiration	Ionic strength/salty/sodic
	N/P/K/C and others
	Nutrient cycling
	Reduction/oxidation
	Perturbation (event)
	Fire/flood/tillage/traffic/clear cut/harvest
	Energy and food source
	Carbon/N/crop/animal waste/detritus/soil/minerals

rewetted, the cycle of life is renewed. Prolonged wet soils restrict the movement of air and often become oxygen depleted. Intermittently, wet soils are common and usually do not result in a serious depletion of oxygen. Soils that remain wet because they are continually supplied either with water such as under a pond or from a shallow water table, or that has an internal layer, which restricts the outflow of water may also become oxygen depleted. These conditions support only those life attributes that do not require oxygen from the atmosphere. Soils vary in their ability to absorb and hold water at both the dry and wet ends of the water-holding spectrum. The size and packed arrangement of particles control the relative

spaces available to transport or retain water. Clays and organic material are largely responsible for a soil’s ability to hold water, while sands and aggregated particles provide for the spatial voids for liquid and gas transport. A given soil with a known composition of sand, silt, and clay can be subjected to considerable packing variability depending on its history of perturbations. A loose soil that allows water entry and free drainage, while retaining some water to support life may become compact, nearly impermeable, or water logged due to compacting traffic. The reader is referred to those sections describing soil water for a more detailed discussion of this important dynamic aspect of soil.

SOIL TEMPERATURE

Soils transmit and store heat. Thus, soil temperature will rise and fall depending primarily on the temperature and radiation conditions of the atmosphere directly above the soil. As temperatures rise and fall on either daily or seasonal cycles, the activities of soil organisms also rise and fall. Activity is influenced by both the extremes of the short-term cycles and the long-term patterns of soil temperature. Most soil activities exhibit a response to temperature that would begin low, rise to optimum, and then decline as temperature increases. The range and optimum soil temperatures for a life activity are uniquely characteristic of different organisms. Either cold or high temperature may be lethal to an organism. Soil temperature varies most at the soil surface, where daily swings might exceed 25°C, and varies less with increasing soil depth. As the soil is home to a multitude of organisms, the capacity to respond to changes in soil temperature is another unique aspect of the soil ecosystem. Cold soils in the winter have little activity, but it rises as temperatures increase. High summer temperature might lower the activity of many organisms, but their activity will increase again as temperatures decline in the fall. Soil-temperature fluctuations are highly linked to soil–water conditions because soil–water provides a large proportion of the soil’s capacity to store heat and because soils cool as water evaporation removes heat. There is usually a web of organisms somewhat unique to a soil’s temperature regime. For example, warm tropical soils support a much different group of organisms than temperate or arctic soils.

BUFFERED AND PROTECTED ENVIRONMENT

The soil is an ecologically rich environment because of its high capacity to protect organisms and to buffer them against changes in environmental conditions. Soils absorb, store, and slowly release water, heat, nutrients, and organic materials that provide the life-sustaining needs of organisms. Organisms within the soil are protected from the sudden and sometimes catastrophic changes occurring aboveground.

AGRICULTURAL CHEMICALS

One of the most important management interactions with the soil ecosystem comes from chemicals that are used to enhance or inhibit specific life functions. Chemicals used in agriculture and forestry fall into two categories. Essential plant growth nutrients that may be supplemented through chemical additions. This is a very common practice and often has little direct effect on organisms or their function. However, these nutrients may strongly influence the biochemical

reactions within soil organisms. The second class of agricultural chemicals is pesticides used to inhibit the activity of specific plant or animal pests. Weed and insect control are common agricultural practices. Pesticides come from a variety of chemical compounds and their action ranges from inhibiting a specific organism to a general biocide.

Other chemicals that influence life functions may occasionally occur in soils. Petroleum spills and heavy metals are two examples of these occasional chemicals. In general, chemicals of this nature are not beneficial to soil life and may be directly harmful. An emerging field of science is matching organisms of specific genetic capability to degrade and thus remove these contaminants from soil.

TILLAGE

The state of a soil may be drastically changed by mechanical disturbance. Tillage of some form is a common practice in agriculture in order to improve the seedbed, control weeds, bury crop residues, or incorporate chemicals. Tillage thus suddenly produces a new arrangement of solid, liquid, and organic materials, which results in a shift of soil activity. Tillage has an effect on most of the soil properties that control the water, oxygen, and temperature dynamics within the soil. Tillage also will often break open aggregates or clods exposing the interior soil, which had been somewhat isolated, to a fresh oxygen-rich atmosphere. Two-known consequences of this breakage are flushes of carbon dioxide released to the atmosphere and N released into the soil solution as organic matter is degraded by soil organisms.

THE IMPORTANCE OF SOIL ECOLOGY

Soil ecology is important to the expected performance of a soil in many respects because of the varied forms of life within the soil. A list of organisms, however, does not fully identify this importance as we are primarily interested in the processes performed rather than the organisms. Carbon and N cycles that are driven by organisms within the soil are important for production of crops and forests, safeguarding the environment, maintaining soil quality, and to global warming. Nutrient cycling from decaying matter into new life is essential to maintaining that life and is an important function studied under soil ecology. Many diseases and pests are strongly influenced by the soil environment and they are often a major economic factor of production. Plant roots are only part of a living organism, but their function in nutrient and water uptake is critical to support life. The soil environment and the

organisms near plant roots are an especially important region of soil, known as the rhizosphere. Because of its importance, the rhizosphere is the subject of as much work in soil ecology as all other subjects combined. The reader is referred to sections of this encyclopedia for additional detail concerning these important aspects of soils.

BIBLIOGRAPHY

1. Brady, N.C.; Weil, R.R. *Elements of the Nature and Properties of Soils*; Prentice Hall: Upper Saddle River, 2000; 559 pp.
2. Brussard, L.; Ferrera-Cerrato, R. *Soil Ecology in Sustainable Agricultural Systems*; CRC/Lewis Publishers: Boca Raton, 1997; 168 pp.
3. Cooley, J.H. *Soil Ecology and Management*; International Association for Ecology: Athens, 1985; 132 pp.
4. Dindal, D.D. *Soil Biology Guide*; Wiley: New York, 1990; 1349 pp.
5. Hatfield, J.H.; Stewart, B.A. *Soil Biology: Effects on Soil Quality*; Lewis Publishers: Boca Raton, 1994; 169 pp.
6. Killham, K. *Soil Ecology*; Cambridge University Press: New York, 1994; 242 pp.
7. Volobuev, V.R. *Ecology of Soils*. Translated by A. Gourevich; Program for Scientific Translations: Jerusalem, 1964; 260 pp.

Enchytraeidae

María Jesús Iglesias Briones

Department of Animal Ecology and Biology, University of Vigo, Vigo, Spain

Abstract

Enchytraeid worms constitute a little known group of soil mesofauna because of their small size and the difficulties associated with their identification. They are commonly the most abundant invertebrates in high organic soils and frequently comprise over 70% of the soil faunal biomass. Their populations usually concentrate in the upper soil layers where they feed on a mixture of dead organic matter and microorganisms. They are known to be ecosystem architects that increase soil aeration by burrowing and have important effects on nutrient cycling.

INTRODUCTION

Enchytraeidae (potworms) constitute a little known family of aquatic (both marine and freshwater) and terrestrial worms within the class Oligochaeta, phylum Annelida. They are less conspicuous in size than other soil Annelids like earthworms; their length varies in the range from 6 to 50 mm (Fig. 1). They are regarded as “mesofauna,” together with Acari and Collembola, as they occupy pore spaces with a diameter of less than 2 mm. As the rest of Oligochaetes, enchytraeids are hermaphroditic with amphimictic reproduction, although there are some species that can reproduce parthenogenetically and asexually by fragmentation and also by self-fertilization.

IDENTIFICATION

The lack of taxonomic information about enchytraeids is because of their small size and the confusing identification criteria. Only after the revisions published by Nielsen and Christensen^[1–3] is it possible to resolve the taxonomy of this group by providing a standard set of criteria for identification of genera and species. More than 600 species have been recognized, although the list is increasing.

Taxonomical criteria include external characteristics such as color (although the majority of the species are gray or whitish and usually gut contents, lymphocytes, chloragogen cells, and blood vessels are responsible for other colorations), size, number of segments, cutaneous glands (size, shape, and arrangement), setae (shape and arrangement) or setae follicles (except in the genus *Achaeta* where they are absent), dorsal pores (presence and position) and clitellum (position), and internal ones such as the number and location of oesophageal appendages, salivary glands, and septal glands, the origin of the dorsal vessel and the color of the blood, the shape of the

nephridia, and other characteristics of several reproductive organs (e.g., size and shape of seminal vesicles, length and shape of the sperm funnel, size and shape of the spermatheca, or the number of eggs). The analysis of these features requires a careful study of living, mature specimens because fixative liquids turn the tegument opaque, making it impossible to see the internal organs. Dózsa-Farkas^[4] provided a list of 20 characteristics necessary for the identification and description of enchytraeids; this emphasizes the difficulty of assigning specimens to the species level with certainty.

Furthermore, identification solely based on morphological aspects is problematic because of the high variability of these external and internal characteristics, the close similarity of related species, and the difficulty in identifying immature specimens.^[5] Therefore, the combined use of morphological and modern tools including spermatozoal ultrastructure, chemical composition of lipids, protein analysis and isoenzymes, immunological methods, and restriction fragmentation patterns is recommended.^[5,6]

SAMPLING

Sampling of field populations of enchytraeids is performed by taking soil cores of 5–6 cm diameter, which gives an area of 20 cm² for accurate counting of the worms and easy calculation of the numbers per square meter. The depth of the core depends on the soil type; because enchytraeids are mainly concentrated in the surface organic horizons, sampling depth has to be determined from the preliminary trials in the studied area. In relation to the number of sampled cores, O'Connor^[7] established that a minimum number of 10 units would ensure a representative estimate of the population numbers in a given ecosystem. The extraction of enchytraeids from the soil cores is based on the movement of the worms in response to a

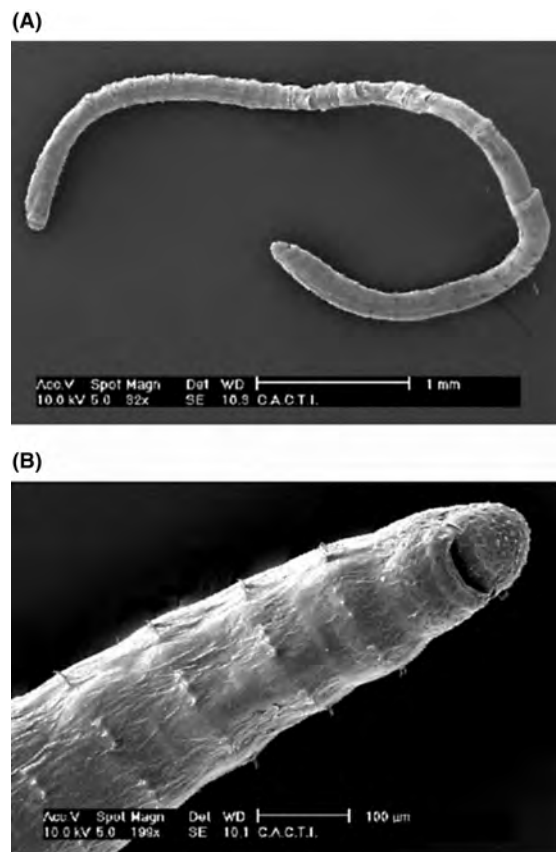


Fig. 1 SEM of *C. sphagnetorum*: (A) whole worm and (B) ventral view of the head region showing the prostomium, the mouth, and the setae follicles.

gradient of light and temperature. Although several methods can be employed, the most widely used is “wet extraction” (Fig. 2), which has been shown to extract more than 95% of the total population of organic soils in three hours.^[8] A modified wet extractor has been proposed^[9] in which heating is omitted and the length of the extraction time is increased to several days. This method proved to be the best in terms of efficiency, but the extremely long extraction time could be inconvenient when processing a large set of samples. Further, a less laborious and faster method has been developed,^[10] which involves flooding the soil sample with Ludox (colloidal silica), followed by gentle mixing and finally counting the animals when they float to the surface of the solution. However, this method seems to be more efficient for recovering enchytraeids from defaunated soils.

SPATIAL DISTRIBUTION AND POPULATION DYNAMICS

Terrestrial enchytraeids are distributed globally, although the majority of the studies on their distribution have been carried out in Europe. Climatic events, competition

with other species, linkage with certain vegetation or soil types, and historical events are thought to be the main controlling factors of enchytraeid distribution.^[11]

Knowledge of the distribution of enchytraeids is highly variable because most studies are based on a low number of samples from few localities, and that these samples are taken over a short period of time with different core sizes, sampling depths, and extraction methods. For these reasons, it is very difficult to give accurate estimates of enchytraeid population sizes in different habitats. This results in the observed variability of abundance, with the numbers ranging from 0 to an average of 350,000 individuals per square meter. Didden^[12] reviewed the available literature and concluded that highest population densities are found in cold to temperate moist habitats, such as moorland soils, coniferous forests, and grasslands; however, low numbers have been recorded in these habitats when other soil characteristics are unsuitable (e.g., soil type).

Besides the methods employed to obtain the mean density values of enchytraeids in various habitats, seasonal climatic fluctuations play an important role in controlling the population dynamics. Temperature and/or moisture content, as well as other biotic activities, seems to be the main factors responsible for the observed variation in enchytraeid population sizes. Summer droughts can drastically affect total numbers as these organisms cannot survive if the soil moisture content is less than 10% of field capacity, and their reproduction rate can be reduced at water contents of 15%.^[13] Severe cold winters can also profoundly affect reproduction with frequent frost periods, leading to higher enchytraeid mortality.

Briones et al.^[14] carried out a transplant experiment to determine whether the responses to climate were species dependent or whether the community as a whole was responding to changes in environmental conditions. The data suggested that temperature was the main controlling factor determining population sizes and that response was species specific: some species were positively influenced by the warmer temperatures and responded by increasing their reproduction rate (*Cognettia sphagnetorum*), while other species were resistant to higher temperature (*Achaeta eiseni*) and/or dramatically reduced in numbers as result of the higher temperature regimes (*Cernosvitiella atrata*). Differences in reproduction strategies, including asexual reproduction by fragmentation by the former species (Fig. 3), and physiological adaptations such as a thicker cuticle in several species of the genus *Achaeta* could have conferred more resistance to the adverse conditions.

Enchytraeids mainly concentrate on the upper soil layers where organic matter is accumulated, although different species show varying vertical distributions. It is also known that some enchytraeid species can also migrate to deeper soil layers in response to moisture changes,^[14,15] but numbers in deeper layers depend greatly on other factors such as

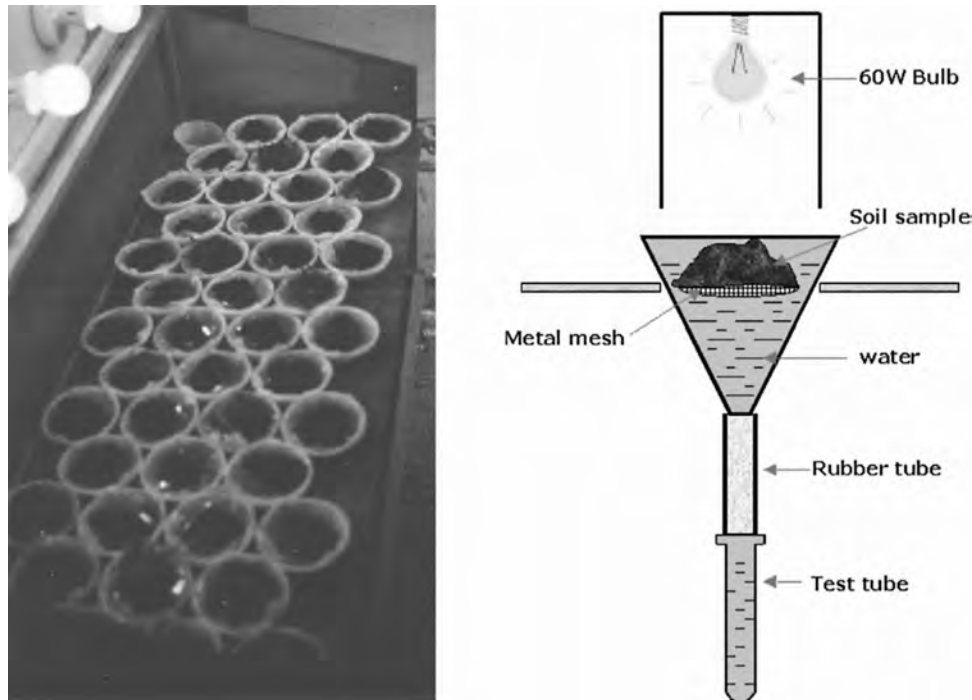


Fig. 2 Wet extractor: the soil sample placed on 1 mm stainless metal mesh is submerged in a water-filled funnel. The temperature is gradually increased to a maximum of 40°C; after three hours, the worms are collected alive in the test tube.

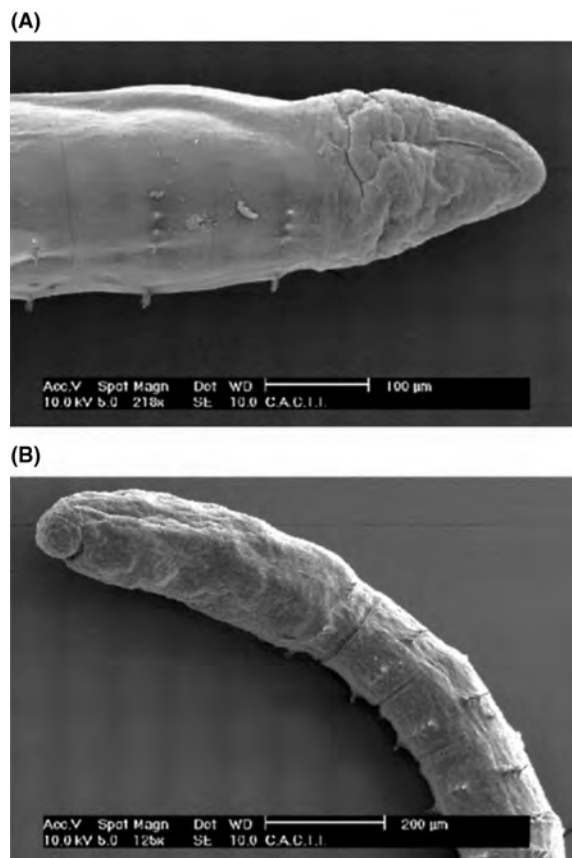


Fig. 3 SEM of a tail fragment of *C. sphagnetorum* regenerating its head: (A) early stage and (B) later stage.

stage of decomposition and differences in food supply.^[16] Vertical migration seems to be the short-term strategy to overcome adverse temporal climatic conditions,^[14,15] at least for some species, whereas others are unable to burrow deeper.^[14]

FEEDING BEHAVIOR

Culture methods suggested that enchytraeids are fungivorous rather than bacteriovorous^[17] and ultrastructural studies of the intestinal content of several species of enchytraeids show that they are saprovores consuming plant debris.^[18] Moreover, some studies suggest that enchytraeids ingest mineral particles, fungi, bacteria, oats and yeasts, algae, and even dead bodies of lumbricids and arthropods. However, evidence suggests that microbivory has been underestimated,^[12] and microorganisms, and especially fungal mycelia, are important food sources for enchytraeid worms.^[19,20] Furthermore, studies using ¹⁴C carbon dating have shown that enchytraeids are also assimilating organic matter that consists predominantly of material that is on average 5–10 years old.^[21]

ECOLOGICAL IMPORTANCE

The contribution of the enchytraeids to energy and nutrient fluxes in soil ecosystems is the result of their consumption and assimilation efficiency, their influence on

soil properties, and feedbacks to other trophic levels.^[12] This contribution is difficult to assess because it depends on their activity in relation to other decomposer invertebrates and microorganisms, and on the amount and quality of the organic matter available. The assimilation efficiency of enchytraeid populations is very low; as a result, they have to ingest large quantities of organic material, and considerable amounts of undigested material are produced, creating a perfect substrate for microbial growth. The high production of fecal pellets, which later fuse to form a soil matrix, and the tunneling and burrowing activities are key influences of enchytraeids on soil structure^[22] and the morphological characteristics of organic horizons.^[23]

The feeding activities of enchytraeids also have strong effects on soil microflora. Hedlund and Augustsson^[24] demonstrated that the grazing activities of *C. sphagnetorum* increased the hyphal length and respiration of the fungus *Mortierella isabellina* and this could have important implications for organic matter turnover. Similarly, Cole et al.^[25] showed that enchytraeids enhanced microbial activity by 35% in the surface horizon of blanket peat.

The contribution of enchytraeids to total soil respiration is generally between 2% and 40%.^[12,26] However, warmer temperature regimes (e.g., a 5°C increase above the mean ambient temperature) induce biomass increases of their populations that could result in a near twofold increase in soil carbon dioxide release.^[27]

Enchytraeidae can also influence strongly the immobilization or excretion of certain cations, such as Ca²⁺.^[28] In addition, leaching of nutrients, particularly of N and P,^[29–31] is also enhanced by the presence of these organisms, but only when these nutrients are not limited.^[25,30] Studies have also demonstrated the effect of these organisms on carbon mineralization measured as dissolved organic carbon (DOC) released into soil solution.^[20,25,32] The positive effect of enchytraeids on the release of DOC is thought to be controlled by the effect of climate on their total abundance and vertical distribution.^[32]

Despite all the information that is available, the functional role of enchytraeids in terrestrial ecosystems has not been fully described. It is clear that in organic soils they are key organisms in terms of their effect on carbon and nitrogen dynamics. However, unlike with other soil fauna groups such as earthworms^[33] and nematodes,^[34] there is not a functional classification of enchytraeids (functional group = a grouping of organisms which affect a process in a similar way) with respect to the different functional roles of enchytraeid species in ecosystems. An attempt at an ecological classification of these organisms has been performed by Graefe and Schmelz^[35] in which species are classified in relation to pH, soil moisture, salinity, reproductive strategy, stress tolerance, and their occurrence in the gradient of humus forms.

CONCLUSION

Enchytraeid worms constitute a little known group of soil mesofauna because of their small size and the difficulties associated to their identification. They are commonly the most abundant invertebrates in high organic soils and frequently comprise over 70% of the soil faunal biomass. Their populations usually concentrate in the upper soil layers where they feed on a mixture of dead organic matter and microorganisms. They are known to be ecosystem architects that increase soil aeration by burrowing and have important effects on nutrient cycling. However, more information is needed to fully understand their functional role in terrestrial ecosystems as changes in environmental conditions can alter their population densities and their vertical distribution with important consequences for organic matter turnover.

REFERENCES

1. Nielsen, C.O.; Christensen, B. The enchytraeidae, critical revision and taxonomy of European species. *Nat. Jutl.* **1959**, *8–9*, 7–160.
2. Nielsen, C.O.; Christensen, B. The enchytraeidae, critical revision and taxonomy of European species (suppl. 1). *Nat. Jutl.* **1961**, *10*, 1–23.
3. Nielsen, C.O.; Christensen, B. The enchytraeidae, critical revision and taxonomy of European species (suppl. 2). *Nat. Jutl.* **1963**, *10*, 1–19.
4. Dózsa-Farkas, K. List of parameters useful for identification of enchytraeids. *Newslett. Enchytraeidae* **1992**, *3*, 49–50.
5. Schmelz, R.M. Species separation and identification in the Enchytraeidae (Oligochaeta, Annelida): Combining morphology with general protein data. *Hydrobiologia* **1996**, *334*, 31–36.
6. Schmelz, R.M. *Taxonomy of Fridericia (Oligochaeta, Enchytraeidae). Revision of Species with Morphological and Biochemical Methods*; Abhandlungen des Naturwissenschaftlichen Vereins: Hamburg, 2003; Vol. 38, 488 pp.
7. O'Connor, F.B. The enchytraeids. In *Methods of Study in Quantitative Soil Ecology*; Philipson, J., Ed.; Blackwell: Oxford, 1971; 83–106.
8. O'Connor, F.B. Extraction of enchytraeid worms from a coniferous forest soil. *Nature* **1955**, *175*, 815–816.
9. Graefe, U. Eine einfache methode der extraktion von enchytraeiden aus bodenproben. In *Protokoll des Workshops zu Methoden der Mesofaunaerfassung (Moderation H. Koehler) und zu PCP-Wirkungen auf Collembolen und Andere Mesofauna-Gruppen*, Beck: Bermen, 1984; Vol. 17.
10. Phillips, C.T.; Kuperman, R.G.; Checkai, R.T. A rapid and highly-efficient method for extracting enchytraeids from soil. *Pedobiologia* **1999**, *43*, 523–527.
11. Römbke, J. Contribution to the biogeography of some species of terrestrial Enchytraeidae (Oligochaeta, Annelida). *Soil Biol. Biochem.* **1992**, *24* (12), 1283–1290.

12. Didden, V.A.M. Ecology of terrestrial Enchytraeidae. *Pedobiologia* **1993**, *37*, 2–29.
13. Beylich, A.; Achazi, R.K. Influence of low soil moisture on enchytraeids. *Newslett. Enchytraeidae* **1999**, *6*, 49–58.
14. Briones, M.J.I.; Ineson, P.; Pearce, T.G. Effects of climate change on soil fauna; responses of enchytraeids, diptera larvae and tardigrades in a transplant experiment. *Appl. Soil Ecol.* **1997**, *6*, 117–134.
15. Springett, J.A.; Brittain, J.E.; Springett, B.P. Vertical movement of Enchytraeidae (Oligochaeta) in moorland soils. *Oikos* **1970**, *21*, 16–21.
16. Didden, V.A.M.; Fluiter, R. Dynamics and stratification of enchytraeids in the organic layer of a scots pine forest. *Biol. Fertil. Soils* **1988**, *26*, 305–312.
17. Standen, V.; Latter, P.M. Distribution of a population of *Cognettia sphagnetorum* (Enchytraeidae) in relation to microhabitats in a blanket bog. *J. Anim. Ecol.* **1977**, *46*, 213–229.
18. Toutain, F.; Villemin, G.; Albrecht, A.; Reisinger, O. Etude ultrastructurale des processus de biodégradation. II. Modèle enchytraeides-litière de feuillus. *Pedobiologia* **1982**, *23*, 145–156.
19. Krištufek, V.; Nováková, A.; Pižl, V. Coprophilous streptomycetes and fungi—Food sources for enchytraeid worms (Enchytraeidae). *Folia Microbiol.* **2001**, *46*, 555–558.
20. Cole, L.; Bardgett, R.D.; Ineson, P.; Hobbs, P.J. Enchytraeid worm (Oligochaeta) influences on microbial community structure, nutrient dynamics and plant growth in blanket peat subjected to warming. *Soil Biol. Biochem.* **2002**, *34*, 83–92.
21. Briones, M.J.I.; Ineson, P. Use of ^{14}C carbon dating to determine feeding behaviour of enchytraeids. *Soil Biol. Biochem.* **2002**, *34*, 881–884.
22. Dawood, V.; FitzPatrick, E.A. Some population sizes and effects of the enchytraeidae (Oligochaeta) on soil structure in a selection of Scottish soils. *Geoderma* **1993**, *56*, 173–178.
23. Davidson, D.A.; Bruneau, P.C.M.; Grieve, I.C.; Young, I.M. Impacts of fauna on an upland grassland soil as determined by micromorphological analysis. *Appl. Soil Ecol.* **2002**, *20*, 133–143.
24. Hedlund, K.; Augustsson, A. Effects of enchytraeid grazing on fungal growth and respiration. *Soil Biol. Biochem.* **1995**, *27* (7), 905–909.
25. Cole, L.; Bardgett, R.D.; Ineson, P. Enchytraeid worms (Oligochaeta) enhance mineralization of carbon in organic upland soils. *Eur. J. Soil Sci.* **2000**, *51*, 185–192.
26. Satchell, J.E. Feasibility study of an energy budget for Meathop wood. In *Productivity of Forest Ecosystems*; DuVigneaus, P., Ed.; UNESCO: Paris, 1971; 619–630.
27. Briones, M.J.I.; Poskitt, J.; Ostle, N. Influence of warming and enchytraeid activities on soil CO_2 and CH_4 fluxes. *Soil Biol. Biochem.* **2004**, *36*, 1851–1859.
28. Anderson, J.M.; Ineson, P.; Huish, S.A. The effects of animal feeding activities on element release from deciduous forest litter and soil organic matter. In *New Trends in Soil Biology*; Lebrun, P., André, H.M., deMedts, A., Grégoire-Wibo, C., Wauthy, G., Eds.; Université Catholique de Louvain: Louvain-La-Neuve, 1983; 87–100.
29. van Vliet, P.C.J.; Beare, M.H.; Coleman, D.C.; Hendrix, P.F. Effects of enchytraeids (Annelida: Oligochaeta) on soil carbon and nitrogen dynamics in laboratory incubations. *Appl. Soil Ecol.* **2004**, *25*, 147–160.
30. Parfitt, R.L.; Yeates, G.W.; Ross, D.J.; Mackay, A.D.; Budson, P.J. Relationships between soil biota, nitrogen and phosphorous availability, and pasture growth under organic and conventional management. *Appl. Soil Ecol.* **2005**, *28*, 1–13.
31. Briones, M.J.I.; Carreira, J.; Ineson, P. *Cognettia sphagnetorum* (Enchytraeidae) and nutrient cycling in organic soils: A microcosm experiment. *Appl. Soil Ecol.* **1998**, *9*, 289–294.
32. Briones, M.J.I.; Ineson, P.; Poskitt, J. Climate change and *Cognettia sphagnetorum*: Effects on carbon dynamics in organic soils. *Funct. Ecol.* **1998**, *12*, 528–535.
33. Bouché, M.B. *Lombriciens de France. Écologie et Systématique*; Institut de la Recherche Agronomique: Paris, 1972; 671 pp.
34. Yeates, G.W.; Bongers, T.; de Goede, R.G.M.; Freckman, D.W.; Georgieva, S.S. Feeding habits in soil nematode families and genera—An outline for soil ecologists. *J. Nematol.* **1993**, *25*, 315–331.
35. Graefe, U.; Schmelz, R.M. Indicator values, strategy types and life forms of terrestrial Enchytraeidae and other microannelids. *Newslett. Enchytraeidae* **1999**, *6*, 59–67.

Entisols

Larry T. West

National Soil Survey Center, U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS) (Retired), Fayetteville, Arkansas, U.S.A.

Abstract

Entisols are soils with minimal pedogenic development and no diagnostic horizons other than an ochric epipedon. Entisols are the most abundant soil order comprising about 18% of the earth's land area. Because these soils are little affected by pedogenesis, their properties mostly reflect those of their parent material which vary widely. Most Entisols are on unstable geomorphic surfaces such as floodplains of rivers and streams, sand dunes, steep mountains, and mined or otherwise disturbed lands where erosion or deposition rate exceeds the rate of pedogenesis. The five Entisols suborders are (1) Aquepts—wet Entisols; (2) Fluvents—Entisols on stream floodplains; (3) Psamments—sandy Entisols; (4) Orthents—Entisols on steep slopes, in deserts, and/or that have resulted from soil disturbance; and (5) Wassepts—subaqueous Entisols. Entisols often have properties that limit their use, including flooding, steep slopes, sandy textures, poor drainage, and stoniness. Entisols have been and continue to be closely associated with human activities, however. Those on river floodplains are often agriculturally productive. In arid areas and mountainous areas, they provide watershed protection, recreation, and wildlife habitat. Many cities and towns on coasts or near rivers are built on Entisols, and the soil disturbance from urban development is continually creating Entisols. The environments in which many Entisols occur, such as floodplains, mountains, and deserts, make them easy to degrade. Thus, proper management is critical to preserve this component of the soil resource for the future.

INTRODUCTION

Entisols are soils that have minimal pedogenic development and no diagnostic horizons other than an ochric epipedon. Most Entisols occur on unstable geomorphic surfaces that are subject to frequent deposition, high rates of geologic erosion and truncation, or drastic disturbance by humans. Common Entisol landscapes include floodplains of rivers and streams, sand dunes, steep mountains, and mined or otherwise disturbed lands. Entisols may also be found on older, more stable landscapes if parent material or other factors retard pedogenesis. Because of limited pedogenesis, subsoil characteristics are typically those of the parent material.

Because of flooding, steep slopes, arid conditions, and other environmental factors, Entisols often have use limitations and can be easily degraded. Even so, Entisols have been and continue to be important for food production and other human activities. Entisols along river floodplains are often intensively farmed and are some of the most agriculturally productive soils in the world. Many towns and cities are located where rivers or oceans provide transportation, and these landscapes have abundant Entisols. In addition, human land disturbance from mining, intensive agricultural use, and in and near urban centers have created Entisols from the predisturbance soil resource.

SUBORDERS

Entisols are the most extensive soil order comprising about 18% of the earth's ice-free land area. About 60% of the Entisols occur in desert regions,^[1] and the largest area of Entisols occurs in Africa where they comprise about 27% of the continent's land area (Table 1). Entisols are divided into five suborders: Aquepts, Fluvents, Psamments, Orthents, and Wassepts.^[3]

Aquepts, the wet Entisols, are not extensive worldwide covering less than 0.1% of the ice-free land surface (Table 1). However, they are widely distributed in landscape segments that accumulate water and receive sediment additions, including salt marshes of coastal tidal zones, freshwater marshes and swamps, floodplain backswamps, and outwash plains. Except for seasonal saturation with water and reducing (anoxic) conditions at or near the surface, Aquept subsoil physical and chemical properties are widely variable and depend on properties of the parent material. Redoximorphic features resulting from reduction and oxidation of iron (Fe) and manganese are common in upper horizons. Aquepts in coastal marshes may contain sulfide minerals (Sulfaquepts) and/or have low bearing capacity (Hydraquepts). Aquepts seasonally saturated at or above the soil surface for long periods may accumulate sufficient organic matter to develop O horizons. Soils with O horizons thicker than 20 cm (histic epipedon), however,

Table 1 Land area of Entisols suborders by continent.

Suborder	North America	South America	Europe	Africa	Asia	Australia/Oceania	Global
Aquepts	— ^a	1	—	43	65	—	109
Fluvents	123	620	343	666	1,284	9	3,056
Orthents	1,121	631	299	7,896	4,634	761	15,834
Psamments	—	787	22	2,271	122	1,251	4,447
Wassents	—	—	—	—	—	—	—
Total	1,243	2,536	664	10,876	6,105	2,022	23,446
Total ice-free land area	21,671	17,681	9,599	29,778	43,924	8,000	130,681

^aThe suborder occurs on all continents but individual areas are too small to show and be tallied from global scale inventories.

Source: Data from Eswaran, Reich, et al.^[2] ©2011 CRC Press. Land area (km² × 10³).

are excluded from Entisols unless sulfidic materials are present within 50 cm of the surface (Sulfaquepts).^[4]

Fluvents are Entisols formed in recentwater-deposited sediments and are commonly found on floodplains, fans, and deltas of rivers and streams where the sediment deposition rate exceeds the rate of pedogenesis. About 2.3% of soils worldwide are Fluvents (Table 1). Fluvents are defined as having an irregular decrease in organic carbon (C) with depth or as having more than 0.2% organic C at a depth of 1.25 m.^[4] Because clayey and loamy strata commonly have more organic C than sandy strata, organic C decreases irregularly in stratified sediments. If the sediment has uniform texture, organic C content in lower subsoil horizons of Fluvents is more than 0.2% because of limited time for organic C decomposition. Physical and chemical properties of Fluvents are variable and depend largely on the source of the alluvial sediment, primarily material eroded from soil upslope in the watershed, serving as parent material. Fluvents landscapes are commonly subject to frequent flooding, which limits their suitability for many uses. Many Fluvents are fertile and productive, however, and are used extensively for crop production if flood cycles do not coincide with cropping periods.

Orthents are primarily the Entisols on recent erosional surfaces. The erosion may have been geologic or human induced by cultivation, mining, or other soil disturbing activities. Orthents are common in deserts and mountainous areas. They are the most extensive Entisols suborder and comprise about 12.1% of the earth's land surface (Table 1). Pedogenesis is limited in deserts because of the lack of water to drive pedogenic processes. In mountainous landscapes, erosion rates are high, and the rate at which weathered material is removed exceeds the rate of pedogenesis. In colluvial landscapes at the base of steep slopes, the rate of sediment deposition may exceed the soil development rate, which will also promote the occurrence of Orthents. Soils that have been formed or drastically disturbed by mining, road building, and agricultural activities such as deep subsoiling, urban/suburban development, and other human activities also classify as Orthents. Extremely slow weathering of resistant parent materials such as quartzite-rich bedrock retards horizon formation and results in

landscapes dominated by Orthents.^[5] Physical and chemical properties of Orthents are inherited from the parent material and vary as widely as the properties of the unconsolidated sediments in which the soils occur.

Psamments are the Entisols with sandy parent materials and cover about 3.4% of the earth's surface (Table 1). Specifically, Psamments must have texture of loamy fine sand or coarser throughout the control section (25–100 cm for soils deep to rock), and many Psamments have sandy textures to depths of 2 m or more.^[5] Common Psamment landscapes include sand dunes, natural levees, sand sheets, and sandy outwash plains. These landscapes can be young or ancient. On ancient landscapes, the materials in which Psamments occur are commonly quartzitic sands that do not readily form horizons because of resistance to weathering. Psamments are by definition sandy and thus are commonly low in plant nutrients and water-holding capacity and high in hydraulic conductivity. These characteristics limit crop production without irrigation and may impose limitations for other uses. Psamments are subject to wind erosion if bare, but with irrigation and proper management are often used for citrus, melon, and vegetable production.

Wassents are subaqueous Entisols formed in shallow water environments. These soils are permanently flooded and must have a positive water pressure at the soil surface for at least 21 hours of each day in all years.^[4] Depth of water overlying the Wassents surface must be such that rooted plants will grow, and this depth is generally considered to be about 2.5 m. Although they have been recognized as soils, Wassents and Wassists (subaqueous Histosols) have horizons that have formed as a result of the same generalized processes as subaerial soils—additions, losses, transfers, and transformations.^[6] Wassents occur in both salt and freshwater environments, and parent materials are commonly Holocene sediments. Physical and chemical properties of soils in this suborder vary widely and reflect the properties of the parent material, landform on which they occur, rate at which parent sediments were deposited, and chemistry of the aquatic environment.^[7] Properties and distribution of Wassents are important aids for design and management of systems for coastal and marine ecosystem management, conservation, and enhancement.^[7]

GENESIS

Although the state factor equation for soil formation,

$$S = f(Cl, PM, R, B, T) \quad (1)$$

where S = soil, Cl = climate, PM = parent material, R = relief, B = the biotic factor, and T = time,^[8] has many shortcomings, it is a convenient way to discuss the influence of various environmental factors on the genesis of soils, including Entisols. Limited time for soil development because of unstable landscapes is the most common and most easily understood reason for occurrence of Entisols. Entisols may occur, however, on relatively stable landscapes on parent materials that have been in place for a relatively long time. Conditions that lead to the occurrence of Entisols on more stable landscapes include arid climates that limit water movement through the soil, parent materials resistant to weathering, parent materials toxic to plant growth, and permanent or almost permanent, saturation with water.

Climate

Entisols occur in all climates, and there are no climatic restrictions on Entisol classification. Because low rainfall limits rates of leaching and other pedogenic processes, formation of diagnostic horizons, and evolution of Entisols to other orders is slower in arid climates than in semiarid and humid areas.^[5] In humid climates, mollic epipedons and Mollisols have been reported to form in as little as 100 years,^[9] and cambic horizons and Inceptisols may develop within 500–600 years or less.^[10,11] To form cambic or calcic horizons and Aridisols in arid environments, however, may take more than 1000 and possibly as much as 4000 years. Retardation of soil development by limited leaching partially explains the high proportion of Entisols that occur in arid regions. In addition, sparse vegetation and intense rainfall events enhance erosion and fluvial deposition, which limits soil development and contributes to occurrence of Entisols in desert areas.^[12,13]

Parent Material

Entisols are found on all types and compositions of parent material. Entisols are often associated with alluvial parent materials because many of these soils are found in floodplains with active fluvial deposition. Extremely resistant bedrock weathers slowly, which tends to maintain Entisol landscapes. Sediments dominated by resistant minerals, such as quartzitic sand, do not have or produce clay, Fe, or other mobile components needed to form diagnostic horizons that would result in the evolution of Entisols to other orders.^[14]

Relief

Entisols are often associated with two differing types of relief: steep eroding landscapes and nearly level to gently

sloping depositional landscapes. On steep convex slopes, erosion may result in soils that are rapidly truncated, which destroys diagnostic horizons that may be forming. On low-relief depositional areas, such as stream floodplains, deltas, fans, and colluvial footslopes, the rate of deposition may be such that the soil accretes faster than diagnostic horizons are formed. Entisols also occur in areas where the local relief results in soils that are permanently saturated, such as coastal marshes, freshwater swamps, and subaqueous environments. Permanent saturation inhibits leaching, translocation, and reduction–oxidation cycles that form diagnostic horizons.

Biota

Entisols can be found associated with any type of vegetation. Thus, little can be said about vegetation effects on genesis of Entisols. If humans are included, however, biota may have a profound impact on development and properties of Entisols. earth moving activities and/or deep tillage often destroys existing soils and creates Entisols.

Time

Time or age of the soil, specifically limited time over which soil development processes have proceeded, is the environmental factor most often associated with Entisol landscapes. If the landscape is “young” because of continual erosion or deposition, time for soil development will be limited, and Entisols will dominate. If parent materials are inert, however, Entisols may occur in old landscapes if weathering does not produce mobile components that are necessary for formation of diagnostic horizons.

CONCLUSIONS

Entisols are soils that are minimally developed and lack diagnostic horizons other than an ochric epipedon. Thus, properties of Entisols are those of the parent material and vary as widely as the properties of the materials in which they occur. Most Entisols occur on unstable geomorphic surfaces that are subject to frequent deposition, high rates of geologic erosion and truncation, or drastic disturbance by humans, although Entisols may occur in old stable landscapes if environmental or parent material characteristics retard pedogenesis.

Although Entisols often have properties that limit their use, they are intimately linked with human activities. Entisols on river floodplains are often highly productive and thus are used extensively for agricultural production. In mountainous and arid areas, Entisols are important for watershed protection, recreation, and wildlife habitat. Soil disturbance in contemporary urban landscapes as well as mining and other soil disturbing activities is continually creating Entisols. The environments in which many

Entisols occur, such as floodplains, mountains, and deserts, make them easy to degrade. Thus, proper management is critical to preserve this component of the soil resource for the future.

REFERENCES

1. West, L.T.; Wilding, L.P. Introduction: General characteristics of soil orders and global distributions. In *Handbook of Soil Science*, 2nd Ed.; Huang, P.M., Li, Y., Sumner, M.E., Eds.; CRC Press: Boca Raton, 2012; 33-3–33-8.
2. Eswaran, H.; Reich, P.; Padmanabhan, E. World soil resources: Opportunities and challenges. In *Advances in Soil Science: World Soil Resources and Food Security*; Lal, R., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2011; 29–52.
3. Soil Survey Staff. *Illustrated Guide to Soil Taxonomy*. U.S. Department of Agriculture, Natural Resources Conservation Service, National Soil Survey Center: Lincoln, Nebraska, 2014. Available at http://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/class/?cid=nrcs142p2_053580#illustrated (accessed April 2015).
4. Soil Survey Staff. *Keys to Soil Taxonomy*, 12th Ed.; U.S. Department of Agriculture Natural Resources Conservation Service: Washington, DC, 2014.
5. Nordt, L.C.; Collins, M.E.; Fanning, D.S.; Monger, H.C. Entisols. In *Handbook of Soil Science*, 2nd Ed.; Huang, P.M., Li, Y., Sumner, M.E., Eds.; CRC Press: Boca Raton, 2012; 33-49–33-63.
6. Demas, G.P.; Rabenhorst, M.C. Subaqueous soils: Pedogenesis in a submerged environment. *Soil Sci. Soc. Am. J.* **1999**, *63*, 1250–1257.
7. Stolt, M.H.; Rabenhorst, M.C. Subaqueous soils. In *Handbook of Soil Science*, 2nd Ed.; Huang, P.M., Li, Y., Sumner, M.E., Eds.; CRC Press: Boca Raton, 2012; 36-1–36-14.
8. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
9. Ruhe, R.V.; Fenton, T.E.; Ledesma, L.L. *Missouri River History, Flood Plain Construction, and Soil Formation in Southwestern Iowa*, Research Bulletin 580; Iowa Agricultural and Home Economics Experiment Station: Ames, 1975.
10. Bilzi, A.F.; Ciolkosz, E.J. Time as a factor in the genesis of four soils developed in recent alluvium in Pennsylvania. *Soil Sci. Soc. Am. J.* **1977**, *41*, 122–127.
11. Scully, R.W.; Arnold, R.W. Holocene alluvial stratigraphy in the upper Susquehanna river basin, New York. *Quaternary Res.* **1981**, *15*, 327–344.
12. Gile, L.H. Holocene soils and soil-geomorphic relations in an arid region of southern New Mexico. *Quaternary Res.* **1975**, *5*, 321–360.
13. Gile, L.H.; Grossman, R.B. *The Desert Project Soil Monograph*; U.S. Department of Agriculture Natural Resources Conservation Service: Washington, DC, 1979.
14. Cabrera-Martinez, F.; Harris, W.G.; Carlisle, V.W.; Collins, M.E. Evidence for clay translocation in coastal plain soils with sandy/loamy boundaries. *Soil Sci. Soc. Am. J.* **1989**, *53*, 1108–1114.

Environmentally Friendly Indigenous Technologies

Selim Kapur

Departments of Soil Science, Landscape Architecture, and Agricultural Engineering, University of Cukurova, Adana, Turkey

Erhan Akça

Center of Remote Sensing, Adiyaman University, Adiyaman, Turkey

Abstract

Indigenous technical knowledge (ITK) is quite known to be the critical factor for sustainable development. Empowerment of local communities is a prerequisite for the integration of ITK into the development process as well as the conservation and improvement of biodiversity. Basic economic factors such as efficiency, effectiveness, and sustainable development have been enhanced by appropriate ITK systems integrated into many development programs.

INTRODUCTION

Indigenous technical knowledge (ITK), as any other knowledge, calls for the consistent use, challenge, and adoption of the ever-evolving local contexts and support networks of traditional practitioners and communities.^[1] These help disseminate useful ITK to participatory actions within the development process with innovative mechanism incorporated at substantial magnitude.^[2]

ITK AND SUSTAINABLE LAND MANAGEMENT (SLM)

The primary reason for the ancient misuse of land that was sustainable and environmentally friendly at the beginning, as was the case with the Tikal situation, the Mayan site in Guatemala, and later gradually converted to a degradation process, has been due to the increase in population. Thus, the pressure of the number of underfed versus overfed people seeking higher standards of living has increased. This leads to the intensive and extensive use of limited land resources with the ultimate disaster of soil degradation. The misuse of the coastal environments of the Mediterranean encountered a similar fate of degradation due to the intense overgrazing and deforestation dating back to the period of King Solomon (ca. 3000 years B.P.) who supplied 80,000 lumberjacks to work in the Levantine forests with 70,000 more to skid the logs to the sea from a forest that is estimated to have been about 5000 km². The mountain ranges of ancient Lebanon during Phoenician times were covered with the famous cedar forests which were frequently transported to Egypt by ships laden with timber before 2900 B.C. as deciphered at an inscription at Karnak.^[3]

The degradation of the environment which mainly started by overgrazing since the Greek period and even earlier by deforestation created a new shrub ecosystem of maquis that is typical of the Mediterranean coastal and epicoastal areas. The beginning of the Eastern Mediterranean coastal settlements of Late Rome and Byzantine Empire (ca. 2000 B.P.) was also the cause of the subtle degradation process, which went on for ca. 4000 to 3000 years (Fig. 1). However, despite all the abovementioned degradation processes together with the mass migrations of people due to warfare, a well-established and consistent Mediterranean ecosystem evolved via an appropriate SLM. This system was followed throughout the mid-late Roman and Byzantine periods and was practiced in Southern Europe, North Africa, and the Middle East.^[3] The remaining forests during Roman and Byzantine periods were protected by strong legislations, as evidenced by the inscriptions in many late Roman monuments such as the one demarcating the forest boundary of Emperor Hadrian Augustus in the Lebanese mountains.^[3] This legislation, with high regard to an environmentally friendly SLM system, was partially followed in the Mediterranean Basin during different imperial periods including the Ottomans. Unfortunately, the overexploitation of natural vegetation by the locals, earlier during the Umayyad and Ottoman periods, caused further degradation of the well-established shrub environment particularly in North Africa and the Greek islands and parts of the Levant.^[5]

THE ANCIENT RATIONAL SLM OF THE MEDITERRANEAN

Ancient SLM was attained within the Mediterranean coast of Anatolia especially in Kizkalesi (Korykos) and Ayas

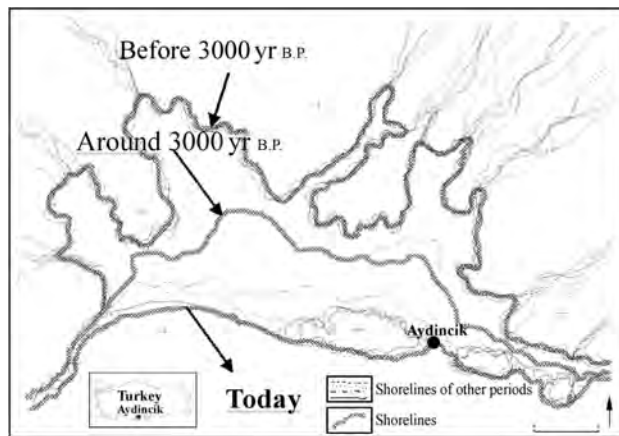


Fig. 1 The shoreline changes of the Aydıncık (Kelenderis) delta (S. Turkey).

Source: From Bal, Kelling, et al.^[4]

(Elaiussa Sebaste) during the late Roman–Byzantine periods (Fig. 2). The historical name of the latter means Emperor Sebastian's olive grove on the olive ecosystem of the late Roman Empire. This conserved the soil, harvested the scarce water sources, and protected the forests as well as the shrub ecosystem (the shrub agroecosystem or the olive ecosystem as coined by Akça et al.^[6] comprising olives as the main tree crop with carobs, vines, and figs) of fodder plants and cash crops of the highlands together with nonindigenous citrus (the sacred tree crop of the ancient Israelites and the Kibbat of the Arabs) adapted to the lower parts of the coastal regions of Southeast Asia around 1300 B.C.^[7]

Polygonal stone-walled terraces 1–2 m high and 0.5–5 m wide (Fig. 2), similar to the ones in Lebanon, South France, and Tunisia, were constructed as part of the ancient ultrastructure in the area consisting of

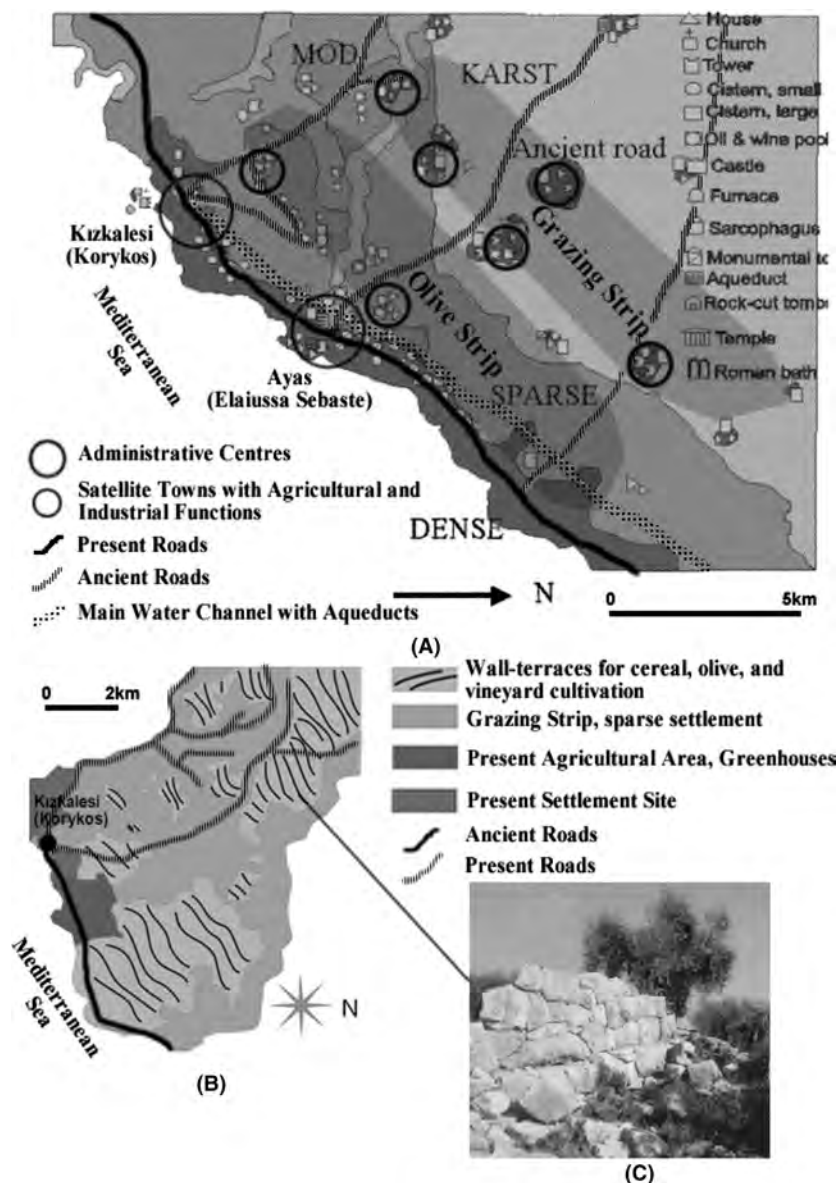


Fig. 2 Korykos and Elaiussa Sebaste settlement pattern (A), ancient and present land use map of the Korykos area (B), and polygonal stone wall terraces (C).

Source: From Akça, Kelling, et al.^[6]

functional land strips (from several hundred meters to kilometers long). These strips were used for grazing by goats and sheep as barn or semidomesticated flocks. Widespread grazing of goats in the late Roman period in Southern Anatolia dating back to late antiquity is also documented by the use of two types of textiles, namely the linen and a coarse cloth made from goat hair giving rise to the ancient Greek name of the region, “Cilicia” (Cilicium in Latin^[8]). Seminomadic people (Yuruks) of the Cilician highlands (local yaylas) live in tents made from goat hair during summer grazing. Together with native olive orchards, semidomestication of grazing large land fragments/pastures has been practiced for a long time in the Turkish Aegean island of Gokceada (Imbros) and the Greek islands. The Lebanese (Beirut) stone-walled terraces of the Beiteddine mountain are estimated to have cost U.S.\$2000–5000/acre, assuming the cost of labor at 40 cents/hr. These immense structures show the extent of the people’s concern for protecting their soils on marginal and scarce lands of the Mediterranean Basin.^[3,6]

THE PLOW IMPLEMENT OF THE MILLENNIA

The weed management practices of the farmers in the Usambara mountains in Tanzania^[9] and the highly efficient Ethiopian plow, the “Maresha,” are the symbols of the ingenuity of African ITK. This plow includes animal-drawn drainage equipment, the broad bed maker, used along with improved crop varieties, fertilizers, and other agricultural practices. The broad bed maker is a modified traditional Ethiopian plow used for the construction of broad beds and furrows, allowing drainage of excess water during heavy rains. The Anatolian plow dating back to the pre-Hittites (ca. 4000 B.P.) was used for cultivating the indigenous wheat species of *Triticum monococcum* and *dicoccum*, with their crossbred varieties being cultivated using similar plows in marginal areas. These plows are used to till the soil to a depth of 10–15 cm with minimal soil disturbance (Fig. 3).^[10,11]

This implement has been extremely efficient especially in olive and pistachio groves. These groves are

traditionally located on calcrete (caliché) surfaces in Anatolia and the Middle East (especially the Levant) and date back to the birth of the olive tree.^[12,13] Thus, the proper tillage of the shallow soils causes the mixing of the stone fragments of the underlying soil-derived calcrete parent material, which contains sufficient amounts of palygorskite clay minerals forming a porous microstructural pattern with calcite minerals. These systems help retain plant-available water throughout the year in the ancient rainfed olive (the Mediterranean coast and epicoast) and pistachio (semiarid and mild-arid Southeast Anatolia and the Middle East) groves. The high terrace (TH, mudflow terraces of Late Pleistocene and Early Holocene) and low terrace (TL, Holocene river terraces) sequences of the indigenous olive, carob, and vineyard surfaces on geomorphological–pedological sections represent similar sites in the Mediterranean Basin (Fig. 4). Fig. 4 illustrates the layout of typical olive, carob, and vineyard sequences on limestone, mudflows, and lacustrine deposits with calcretes underlying the widespread fertile shallow soils (Alfisols; ca. 30 cm) of South and Southeast Anatolia as well as the sustainably cultivated lands of Southern Europe and North Africa by the ancient Greeks and Romans.^[14]

APPLICATIONS OF ITK TO SLM

The momentum for developing a philosophical concept for improving particular technologies on land management was gained by ITK in India. A proper example of this is the ITK on the use of shifting sand dunes—the theri soil, which has been passed from one generation to the next. The theri soil—formed by wind erosion—and widely distributed in Tamil Nadu (the coastal belt of the Bay of Bengal) is red to dark red in color, sandy, single-grained, and neutral.^[15] A study conducted in Sivapur in 1993 revealed the use of drought-resistant indigenous crops of palmyra (*Borassus flabellifer*), odai (*Acacia planifrons*), tamarind (*Tamarindus indica*), and neem (*Azadirachta indica*), which are highly suitable for the theri ecosystem but less profitable with irrigation. Farmers have been growing rice, groundnuts, coconuts, and bananas as well as cashew and other indigenous species when no irrigation is available. Farmers have been reclaiming these sandy soils by the addition of tank silt (ca. 100 tons/ha) with the use of farmyard manure and regionally developed irrigation systems based on ITK. The experiences of Sivapur village demonstrate that when farmers of India, as elsewhere in the world, are under pressure, they will develop new techniques for cultivation based on ITK.^[16] Similar technologies have been developed in the Indian subcontinent based on ITK in Nepal for fodder tree management for livestock.^[17] ITK has a proven scientific validity for growth and survivability of the seedlings as well as foliage production from Bedulo (*Ficus clavata*),



Fig. 3 Anatolian plow with rural family of late 19th century. **Source:** Drawing Courtesy of A. Başçetinçelik, University of Çukurova, Turkey.

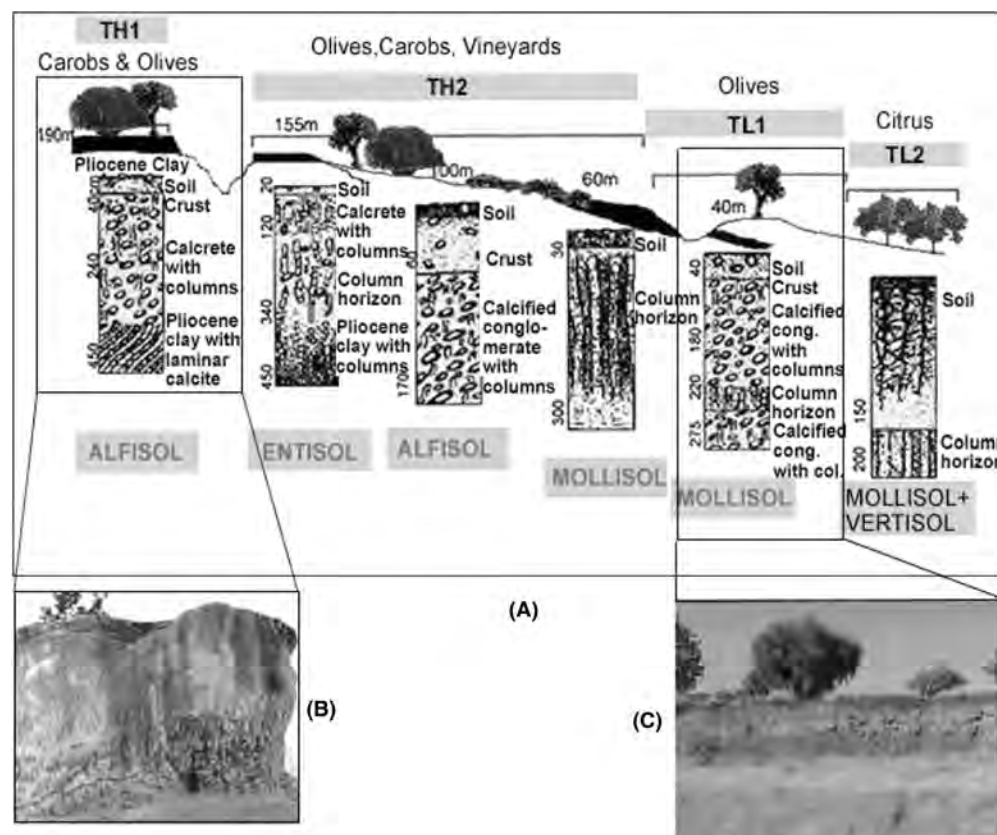


Fig. 4 Pedological section (A) of TH (B) and TL (C) surfaces.

Source: From Kapur, Cavusgil, et al.^[18]

Ginderi (*Premna integrifolia*), Khanayo (*Ficus semicordata*), Kabro (*Ficus lacor*), and Pakhuri (*Ficus glaberrima*).

The ultimate level of ITK based on the participatory approach by farmers in the Indian subcontinent is the “Vertisol Technology.” This technology was developed to manage problematic soils (e.g., those that develop wide cracks in dry periods) with specially developed agronomic practices. These practices not only increase the production but also reduce erosion during the periods of heavy rains. The International Crops Research Institute for the Semi-Arid Tropics’ (ICRISAT’s) conceptual approach on implementation of soil technologies has been used in Africa and elsewhere in Asia, namely for chickpea production in Kenya and Uganda. The ITK used included replacement of traditional pigeon pea varieties by improved cultivars and utilization of the available indigenous system. Pest control was achieved by boiling local plants, weeds, or tree components (leaves, roots, and bark) in water. The mixture is sprayed in the fields to avoid pest infestations. The efficiency of these traditional systems has yet to be surpassed and serves as viable alternatives to costly chemicals. Moreover, the indigenous pest control does not lead to pest resistance as in the case with chemicals.^[17]

Finally, identifying problems and creating solutions in SLM, for small-scale farmers within the concept of

indigenous agroecological zones, may help define research agendas for international and national agricultural research centers that seek to alleviate poverty, as in the case of ICRISAT’s ITK studies.^[17]

REFERENCES

1. Warren, D.M. Indigenous knowledge, biodiversity conservation and development. In *Keynote Address at the International Conference on Conservation of Biodiversity in Africa: Local Initiatives and Institutional Roles*, Nairobi, Kenya, Aug 30–Sept 3, 1992; 1992.
2. Gorjestani, N. *Indigenous Knowledge for Development, Opportunities and Challenges*. Proceedings of UNCTAD Conference on Traditional Knowledge, Geneva, Nov 1, 2000. Available at www.worldbank.org/afr/ik/ikpaper_0102.pdf.
3. Lowdermilk, W.C. *Conquest of the Land Through Seven Thousands Years*. USDA, Soil Conservation Service, No. 99, US Government Printing Office: Washington, 1994; 1–30.
4. Bal, Y.; Kelling, G.; Kapur, S.; Akça, E.; Çetin, H.; Erol, O. An improved method for determination of Holocene coastline changes around two ancient settlements in Southern Anatolia: A geoarchaeological approach to historical land degradation studies. *Land Degrad. Dev.* **2003**, *14*, 363–376.
5. Di Castri, F.; Mooney, H.A., Eds. *Mediterranean Type Ecosystems: Structure and Function*; Springer-Verlag: Berlin, 1973.

6. Akca, E.; Kelling, G.; Kapur, S.; Bal, Y.; Gultekin, E.; Everest, A.; Yetis, C.; Senol, S.; Darici, S. Preserving our heritage: A case study from southern Anatolia. In *Conference Proceedings of International Urban Fellow Association*; Publication of the Mersin University Faculty of Architecture, 2002; 68–76.
7. Davies, F.S.; Albrigo, L.G. *Citrus*; CAB International: Wallingford, 1994.
8. Pleket, H.W. Models and inscriptions: Export of textiles in the Roman Empire. *Epigraphica Anatolica* **1998**, *30*, 117–128.
9. Jabbar, M. *New Technology for Poorly Drained Soils*; Asareca-Agriforum Publication, Graphic Systems (U) Ltd.: Kampala, 2002; Vol. 5.
10. Demirci, R.; Özçelik, A. *History of Agriculture*; Pub. of the Univ. of Ankara, Fac. of Agr.: Ankara, 1990; Vol. 1186.
11. Bascetincelik, A. From the plough to the satellite: Agriculture in the Cukurova region. In *Adana, the Bridge to Ancient Civilization*; Artun, E., Koz, M.S., Eds.; YKY Publishing: Istanbul, 2000; Vol. 1392, 83–589.
12. Hartmann, H.T.; Bougas, P.C. Olive production. *J. Econ. Bot.* **1970**, *24*, 443.
13. Loukas, M.; Krimbas, C.B. History of olive cultivars based on their genetic distances. *J. Hortic. Sci.* **1983**, *58*, 121.
14. Juo, A.S.R.; Wilding, L.P. Land and civilisation: An historical perspective. In *Response to Land Degradation*; Bridges, E.M., Hannam, I.D., Oldeman, L.R., de Vries, F.W.T.P., Scherr, S.J., Sombatpanit, S., Eds.; Oxford & IBH Pub. Co. Pvt. Ltd.: New Delhi, 2001; 13–19.
15. Monaharan, M.; Kombairaju, S. *ITK Suits Transported Sandy Soils*; 2002. Available at www.nuffic.nl/ciran/ikdm/3-1/articles/manoharan.html.
16. Pokharel, K.R. *Indigenous Technical Knowledge (ITK) of People on Fodder Tree Management*; 2002. Available at http://www.panasia.org.sg/nepalnet/forestry/itk_paper.htm.
17. ICRISAT. *ITK—Indigenous Technical Knowledge*; ICRISAT: Patancheru, 2002; 324, 502. Available at www.icrisat.org/text/research/nrmp/researchbriefs/vertisol.asp.
18. Kapur, S.; Cavusgil, V.S.; Senol, M.; Gurel, N.; FitzPatrick, E.A. Geomorphology and pedogenic evolution of quaternary calcretes in the northern Adana basin. *Z. Geomorphol.* **1990**, *34*, 45–59.

Enzymes

Veronica Acosta-Martinez

Cropping Systems Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Lubbock, Texas, U.S.A.

Susanne Klose

University of California—Davis, Davis, California, U.S.A.

Abstract

The functionality and resilience of natural and managed ecosystems mainly rely on the metabolic abilities of microbial communities, which are the main sources of enzymes in soils. Enzyme-mediated reactions are critical in the decomposition of organic matter, cycling of nutrients, and in the breakdown of herbicides and pollutants. This entry describes the groups, locations, and role of enzymes in soils and the relationship between soil enzyme activities, namely, soil properties, soil management, and pollution.

INTRODUCTION

Enzymes are protein catalysts in biochemical reactions and an integral part in nutrient cycling and energy transformation in soil. Enzymes (E) catalyze reactions without undergoing permanent alteration by reducing the activation energy via the formation of an enzyme–substrate complex (ES); thus, the product (P) of the reaction is released at faster rates than if the enzymes were not present.



Plants, animals, and microorganisms are sources of enzymes in soils, but the microbial component, composed of protozoa, fungi, actinomycetes, and bacteria, is their main source.^[1] Despite the short life cycle of microorganisms, most enzymes contribute to the degradation potential of soil: 1) intracellularly inside an active cell and 2) extracellularly in the soil solution or attached to soil surfaces. Enzyme-mediated reactions are essential in the decomposition of organic residues, humification, release of plant-available nutrients, transformation of nitrogen (N) compounds including N₂ fixation, nitrification, and denitrification, stabilization of soil structure, and degradation of xenobiotic (foreign or strange) compounds.

SOIL ENZYME CHARACTERISTICS

Grouping of Soil Enzymes

Enzymes are categorized according to the mechanism of the reaction they catalyze, and four major enzyme groups have been studied in soil:

1. Oxidoreductases catalyze the electron transfer from one molecule to another. Examples are catalases that participate in the oxygen release from hydrogen peroxide and dehydrogenases that catalyze organic matter oxidation. Dehydrogenases are strictly intracellular enzymes and thus reflect the total oxidative activities of the soil microflora.
2. Transferases are involved in transferring groups [i.e., one-carbon (C), glycosyl, hydroxymethyl, formyl, and carboxyl] from one molecule to another. Examples are dextran sucrose transferase that catalyzes the transfer of glucosyl residues from sucrose to dextran polymer releasing glucose and fructose, and thiosulfate S-transferase (rhodanese) that catalyzes the transfer of cyanide to thiosulfate releasing thiocyanate and sulfite, which is an intermediate step in elemental sulfur (S) oxidation.
3. Lyases are enzymes involved in the cleavage of chemical bonds other than that by hydrolysis or oxidation. Selected N-cycling enzymes belong to this group, such as: 1) glutamate decarboxylase, which is involved in the decarboxylation of glutamate; 2) tyrosine decarboxylase, which catalyzes the removal of the carboxyl group from tyrosine; and 3) L-histidine ammonia lyase, which catalyzes the deamination of L-histidine producing urocanate and ammonia.
4. Hydrolases catalyze the hydrolysis of various chemical bonds (i.e., ester, glucosyl, ether, and peptide bonds) by reacting with water. Most commonly assayed soil enzyme activities belong to this group, such as phosphatases, urease, L-asparaginase, cellulase, xylanase, β -glucosidase, protease, and arylsulfatase.

Location of Soil Enzymes

The total activity of any enzyme in soil is a composite of activities associated to 10 different locations^[2] (Fig. 1).

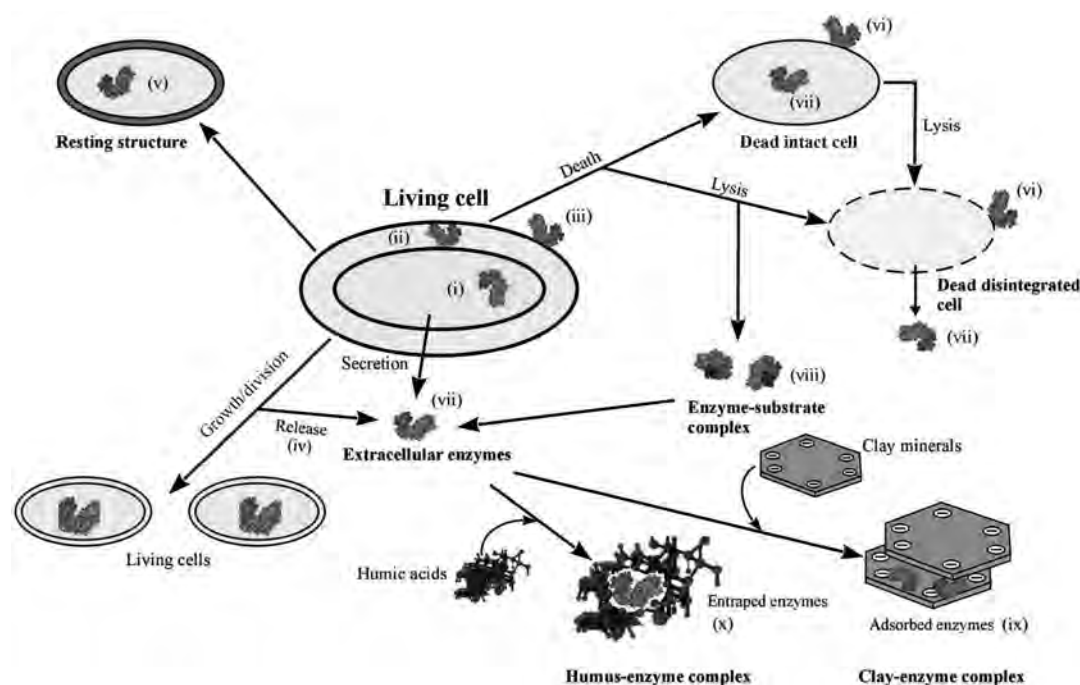


Fig. 1 Locations of enzymes in soil: (i) enzymes functioning within the cytoplasm of proliferating microbial, animal, and plant cells; (ii) enzymes restricted to the periplasmic space of proliferating gram-negative bacteria; (iii) enzymes attached to the outer surface of viable cells with active sites extending into the soil environment; (iv) enzymes released into the soil solution by living cells during cell growth and division; (v) enzymes within non-proliferating cells such as fungal spores, protozoa cysts, plant seeds, and bacteria endospores; (vi) enzymes attached to an entire dead cell and cell debris; (vii) enzymes leaking from intact cells or released from lysed cells, originally located on the cell membrane or within the cell, which may survive for a short period in the soil solution; (viii) enzymes temporarily associated with soluble or insoluble enzyme-substrate complexes; (ix) enzymes absorbed to the external or internal (i.e., within the lattices of 2:1 layer silicates) surfaces of clay minerals; and (x) enzymes complexed with humic colloids by absorption, entrapment, or copolymerization during humification.

Approaches to separate extracellular enzymatic activities from those associated with soil microorganisms have included exposure of soils to: 1) elevated temperatures; 2) fumigants; 3) plasmolytic and antiseptic agents; and 4) irradiation with gamma rays or electron beams. Research with several soil enzymes showed that chloroform fumigation of soil led to an increase in the activities of urease and aryl-sulfatase. This increase in enzyme activities was believed to be due to the release of intracellular microbial enzymes and thus may provide an estimate of the intracellular fraction of these enzymes in soil.^[3] All approaches are limited by the potential of enzyme denaturation by these treatments or degradation by proteases. Distinguishing the contribution of accumulated enzymes and enzymes of active microbial communities to total enzyme activities is as a challenge in soil enzyme research because of the importance of enzymatic reactions in various processes linked to soil metabolic functioning.

Enzyme Protein Concentrations in Soil

Enzyme protein concentrations (mg enzyme protein kg⁻¹ soil) can be calculated by comparing the activity of a

reference enzyme protein and the corresponding enzyme in soils, assuming that the chemical structure of the reference protein is similar to that of the soil enzyme.^[3] Enzyme protein concentrations range widely between 0.014 and 3.67 mg protein kg⁻¹ soil for glycosidases, between 0.73 and 3.38 mg protein kg⁻¹ soil for amidohydrolases, and between 2.1 and 22.5 mg protein kg⁻¹ soil for phosphatases in agricultural surface soils with varying pH values (5.5–7.9), organic C (12–44 g kg⁻¹), and clay content (170–360 g kg⁻¹).

ROLE OF SOIL ENZYMES (HYDROLASES) IN BIOGEOCHEMICAL CYCLING

C-Cycling (Glycosidases)

C occurs in soil in simple or complex organic and various inorganic forms (carbonates and CO₂), which undergo biochemical transformations involving enzyme reactions. Glycosidases play a key role in the breakdown of low-molecular-weight carbohydrates (i.e., disaccharides) producing sugars, which are important energy sources for soil microorganisms. Major soil glycosidases are as follows: 1)

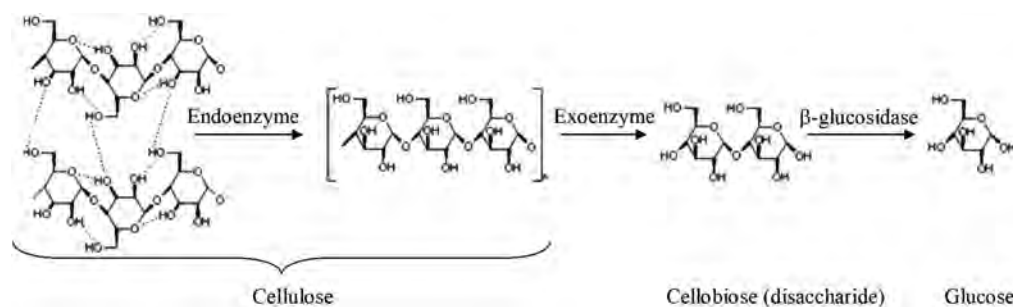
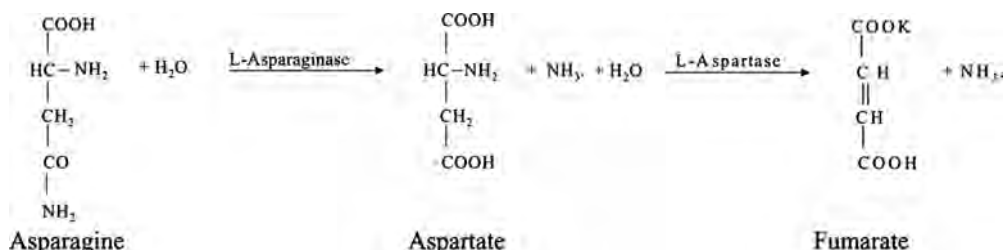


Fig. 2 Enzymatic cellulose degradation.

α -glucosidase (maltase) catalyzing the hydrolysis of α -D-glucopyranosides (maltose); 2) α -galactosidase (melibiase) catalyzing the hydrolysis of α -D-galactopyranosides; 3) β -galactosidase (lactase) catalyzing the hydrolysis of β -D-galactopyranosides; and 4) β -glucosidase (gentibiase or cellobiase) catalyzing the hydrolysis of β -D-glucopyranosides (cellobiose). The β -glucosidase activity catalyzes the final step of cellulose degradation, which is the most abundant organic compound in the biosphere (Fig. 2).

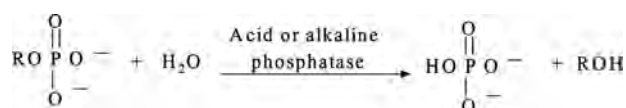
N-Cycling (Amidohydrolases)

N is the most commonly limiting plant nutrient in soils because more than 95% of the total N occurs in organic forms, unavailable to plants. Amidohydrolases including L-asparaginase, L-aspartase, L-glutaminase, and amidase cleave C–N bonds other than peptide bonds in linear amides and release plant-available NH_3 . L-Asparaginase catalyzes the hydrolysis of L-asparagine to aspartate, which is further hydrolyzed by L-aspartase to fumarate:



Phosphorus (P) Cycling and S-Cycling (Phosphatases and Arylsulfatase)

P is present in soil in a variety of organic and inorganic forms. P is essential for plant development but is the least mobile of the major plant nutrients. Phosphatases are crucial in organic and inorganic P transformation processes and plant nutrition. Phosphatases belong to a broad group of enzymes catalyzing the hydrolysis of both esters and anhydrides of H_3PO_4 . Most common phosphatases studied in soils are phosphomonoesterases and phosphodiesterases.^[1] Phosphomonoesters are hydrolyzed by acid or alkaline phosphatases (phosphomonoesterases) depending on the soil pH:



S occurs in organic forms in most surface soils, and mineralization of these organic fractions is critical for

plant nutrition. Arylsulfatase is believed to be partially responsible for S-cycling in soils by catalyzing the hydrolysis of organic sulfate esters following the reaction:



ENZYME ACTIVITIES AS INDICATORS OF SOIL METABOLIC FUNCTIONING

Enzyme Activities and Soil Properties Relationship

Stabilization and protection of soil enzymes by humic and clay particles may increase the resistance of enzymes to changes in soil temperature and pH. Enzyme activities increase with increasing soil temperatures and are generally inactivated at temperatures between 60°C

Table 1 Enzyme activities (mg product kg⁻¹ soil h⁻¹) as affected by soil texture, organic C content, and soil management at 0–5 cm depth (Past = pasture; Rot = rotation; Mono = monoculture) as summarized from research conducted in semiarid soils.

Texture	Organic C (g kg ⁻¹)	Soil classification	C-cycling			C- and N-cycling			P-cycling			S-cycling		
			β-Glucosidase			β-Glucosaminidase			Alkaline phosphatase			Arylsulfatase		
			Past	Rot	Mono	Past	Rot	Mono	Past	Rot	Mono	Past	Rot	Mono
Clay	8.5–13.3	Torretic Paleustolls	240	185	125	58	35	18	305	329	150	40	38	23
Sandy clay loam	3.8–18.7	Aridic Paleustolls	198	63	31	34	8	4	308	119	38	58	23	11
Loam	4.9–13.1		186	89	60	58	34	30	149	27	31	18	5	2
Sandy loam	3.1–14.0	Aridic Paleustalfs	124	78	23	40	13	4	190	147	45	34	13	9
Loamy sand	2.8–4.0		49	46	22	9	7	5	120	46	28	25	9	2
Sand	1.4–1.8		49	66	15	9	7	2	63	46	15	24	19	3

Source: From Acosta-Martínez, Mikha, et al.^[4] and Acosta-Martínez, Klose, et al.^[5]

and 70°C. Each enzyme has a specific pH value at which the reaction rate is optimal, and at each side of this pH optimum the rate is lower. Phosphomonoesterases, e.g., are classified as acid and alkaline phosphatases because they show optimum activities in soils under acid and alkaline ranges, respectively. As presented in Table 1, enzyme activities are higher in clay and loam soils than in sandy soils following the clay and organic C contents.^[4,5] Enzyme activities generally decrease with increasing soil depth following the amounts of organic C and N along the soil profile and are higher in the rhizosphere when compared with the bulk soil.

Enzyme Activities and Soil Management Relationship

Crop rotation, fertilization, tillage, and soil amendment affect enzyme activities by altering soil structure, bulk density, soil pH, and amounts and distribution of organic matter and nutrients in soil. Soils under crop rotations generally show higher enzyme activities compared with monocropping systems owing to diversified organic inputs, improved soil structure, nearly year-round rhizosphere and plant cover, and higher root density. Changes in enzyme activities may even anticipate the changes in organic matter of soil as affected by soil management. However, the changes in enzyme activities due to crop rotations can depend on the soil type and the quantity and quality of plant residues and nutrients entering the soil. Inorganic fertilizer applications can affect enzyme activities through higher plant yields, crop residue amounts, and changes in soil pH and soil solution chemistry. However, the addition of enzyme reaction products by inorganic fertilizers can also suppress enzyme synthesis.^[6] Liming of agricultural soils can stimulate most enzyme activities because of increases in soil pH, except for acid phosphatase activity, which is decreased by increasing soil pH. Organic soil amendments can lead to an initial increase

in enzyme activities, but subsequent additions may fail to sustain high enzyme activities.^[7] Intensive tillage generally decreases enzyme activities in the upper 10 cm soil layer.^[8]

Enzyme Activities and Soil Pollutants Relationship

Organic contaminants, heavy metals, and pesticides are pollutants of major environmental concern because of their persistence and broad biocidal activity against key soil processes affecting ecosystem stability and resilience. Each soil enzyme reacts differently to soil pollution depending on the type of pollution, pollutant concentration, and soil type.^[9] Soil amendments with urban and industrial wastes, releasing various organic and inorganic pollutants, waste or high-salinity water, or atmospheric depositions may affect soil microorganisms and enzyme activities in various ways. For example, the addition of organic waste materials generally enhances soil enzyme activities owing to the increase in soil organic matter and nutrient contents, and improvement of soil physical properties unless these organic wastes also contain high levels of heavy metals and other toxic compounds. The degree of enzyme activity inhibition by organic wastes depends on the enzyme, soil organic matter, and clay contents, as well as the concentration and form of heavy metals in these waste materials. Heavy metals can react with active groups of enzymes resulting in inhibition or inactivation or decreasing enzyme proliferation by changing the microbial community.^[10] Pesticide effects on soil enzyme activities are difficult to summarize because pesticides are subjected to different microbial processes, including biodegradation, cometabolism, polymerization or conjugation, and accumulation in microorganisms.^[6] Short-term studies have reported a small initial stimulatory effect on dehydrogenase activity and slight increases or no significant effect on the activities of ureases and phosphatases when pesticides are applied at recommended field rates.^[6]

If pesticides are applied to soils at very high concentrations simulated by accidental spills, enzyme activities are significantly reduced.^[6] The toxicity of pesticides, in particular soil fumigants, is related to: 1) their interference with respiratory enzymes; 2) their ability to chelate metal cations; 3) inhibition by the unchelated ion; and 4) toxic degradation products.^[11] Soil fumigation with six different fumigant pesticides reduced the activities of arylsulfatase, dehydrogenase, acid phosphatase, and β -glucosidase between 6% and 62% over a three-month period. Fumigated agricultural soils showed longer-term decreases in the activities of β -glucosidase, acid phosphatase, and dehydrogenase. Changes in enzyme activities were fumigant specific and varied between different locations, supporting the hypothesis of a significant natural variability within and between soils, and the influence of the soil type on the response of enzyme activities to biocides and soil pollutants.

CONCLUSION

The information obtained from different enzyme activities should be interpreted with caution because enzymes are involved in very specific biochemical reactions, and the total enzyme activity measured in a soil is a reflection of the potential activity only. Despite the complexity in the behavior of biochemical reactions toward various pollutants and chemicals including fertilizers, enzyme activities are considered as a sensitive indicator of soil disturbance, environmental stress, and changes in soil metabolic functionality.

REFERENCES

1. Tabatabai, M.A. Soil enzymes. In *Methods of Soil Analysis: Chemical and Microbiological Properties*. Part 2. Book Series No. 5; Weaver, R.W., Angle, J.S., Bottomley, P.S., Eds.; Soil Science Society of America: Madison, 1994; 775–833.
2. Burns, R.G. Enzyme activity in soil: Location and a possible role in microbial ecology. *Soil Biol. Biochem.* **1982**, *14*, 423–427.
3. Klose, S.; Tabatabai, M.A. Response of glycosidases in soils to chloroform fumigation. *Biol. Fert. Soils* **2002**, *35*, 262–269.
4. Acosta-Martínez, V.; Mikha, M.M.; Vigil, M.F. Microbial communities and enzyme activities in soils under alternative crop rotations compared to wheat-fallow for the Central Great Plains. *Appl. Soil Ecol.* **2007**, *37*, 41–52.
5. Acosta-Martínez, V.; Klose, S.; Zobeck, T.M. Enzyme activities in semiarid soils under conservation reserve program, native rangeland, and cropland. *J. Plant Nutr. Soil Sci.* **2003**, *166*, 699–707.
6. Dick, R.P. Soil enzyme activities as integrative indicator of soil health. In *Biological indicators of soil health*; Panthurst, C.E., Doube, B.M., Gupta, V.V.S.R., Eds.; Cab International: New York, 1997; 121–156.
7. Martens, D.A.; Johanson, J.B.; Frankenberger, W.T. Production and persistence of soil enzymes with repeated addition of organic residues. *Soil Sci.* **1992**, *153*, 53–61.
8. Kandeler, E.; Tscherko, D.; Spiegel, H. Long-term monitoring of microbial biomass, N mineralization and enzyme activities of a Chernozem under different tillage management. *Biol. Fert. Soils* **1999**, *28*, 343–351.
9. Nannipieri, P.; Kandeler, E.; Ruggiero, P. Enzyme activities and microbiological and biochemical processes in soil. In *Enzymes in the Environment. Activity, Ecology and Applications*; Burns, R.G., Dick, R.P., Eds.; Marcel Dekker: New York, 2002; 1–33.
10. Kandeler, E.; Kampichler, C.; Horak, O. Influence of heavy metals on the functional diversity of soil microbial communities. *Biol. Fert. Soils* **1996**, *23*, 299–306.
11. Klose, S.; Ajwa, H. Enzyme activities in agricultural soils fumigated with methyl bromide alternatives. *Soil Biol. Biochem.* **2004**, *36*, 1625–1635.

Erosion

Dennis C. Flanagan

*U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
West Lafayette, Indiana, U.S.A.*

Abstract

Erosion is the detachment of soil particles and transportation to another location. Wind erosion occurs when wind speed exceeds a critical threshold level, and loose soil particles or soil particles removed by abrasion then move in one of three ways: creep, saltation, and suspension. Erosion by water is typically by raindrops or by concentrated water flows on a soil surface. Raindrops vary in size from about 0.5 mm in diameter to 4 mm or larger and can reach terminal velocities of 10 m/s. The momentum of large drops at high speeds can result in soil compaction and ejection of water and sediment sprays (splash) at the point of impact. Water on a soil surface that does not infiltrate will flow downhill and typically coalesce into small channels or “rills.” When enough water is flowing in a rill, flow shear forces will act to detach soil from the channel bottom and sides. Rills are the major pathway for both water and sediment detached by raindrops and flow shear stress to move downslope and ultimately off of a piece of land. In larger channels, similar detachment due to flow shear forces can also occur that can produce different types of gullies, as well as additional soil loss from other processes including bank sidewall sluffing and headcutting. A number of practices are recommended to reduce or prevent erosion, including use of mulch or plant residues to protect the soil surface, terraces to reduce slope length and steepness, and grade control structures to safely transport water flow to lower elevations.

INTRODUCTION

Soil erosion is the detachment or breaking away of soil particles from a land surface by some erosive agent, most commonly water or wind, and subsequent transportation of the detached particles to another location. Usually, erosion occurs when a fluid (air or water) moves into and/or across a soil surface. Fluid and sediment particle impact forces, shear forces, and turbulence act to detach and lift soil into the fluid flow that then transports the particles away (Fig. 1). The force of gravity moves detached soil particles downward, while cohesive forces between soil particles resist detachment and transport. Physical and chemical dispersion can disrupt cohesion and break soil aggregates into smaller and more easily transported particles. At some time and location away from the initial point of detachment, the sediment particles will eventually move back down to a state of rest on a soil surface, in a process known as deposition or sedimentation.

Erosion is a natural process and is a critical factor in soil formation from rock parent material. However, once productive agricultural soils have been formed over periods of thousands or millions of years, the erosion of soil material is then usually very low or negligible because of the effects of protective natural plant and residue cover. Human activities are responsible for greatly accelerating erosion rates, usually by reducing or eliminating plant and residue cover. This exposes the soil to wind and water erosive forces,

weakening the soil cohesive forces by tillage disturbance and increasing the erosive agents, particularly by activities that increase surface runoff.

EROSION BY WIND

Erosion by wind occurs when wind speed exceeds a certain critical or threshold value. Soil particles can be detached and moved through suspension, saltation, or creep (Fig. 1). Suspension usually lifts the smallest soil particles (clays, silts, and organic matter) so high into the air mass that they are easily kept in motion and can travel for long distances. Soil particles that move by creep are larger sand grains and aggregates that stay in contact with the soil surface at almost all times—their motion is often through rolling and bouncing. Saltating soil particles are usually moderate in size and, once detached, move in trajectories up into the air and then back down to the soil surface. Saltating particles often cause further detachment through abrasion, by striking the soil surface with sufficient momentum to dislodge additional soil particles from the *in situ* soil mass.

EROSION BY WATER

The most common types of soil erosion by water are sheet and rill erosion on upland areas, channel and gully erosion in small watersheds, and stream channel and bank erosion

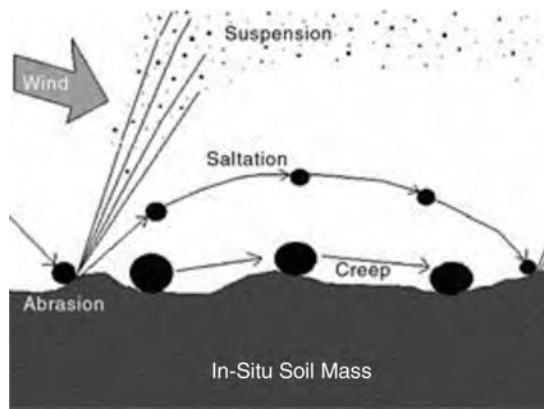


Fig. 1 Soil erosion by wind, showing the three modes of movement (creep, saltation, and suspension).

in larger catchments. Sheet erosion is caused by the action of raindrops (Fig. 2) and shallow overland flows that remove a relatively uniform depth (or sheet) of soil. Raindrops vary in size from about 0.5 mm in diameter to 4 mm or larger and can reach terminal velocities of 10 m/s. The momentum of large drops at high speeds can result in soil compaction and ejection of water and sediment sprays (splash) at the point of impact. Raindrop impacts on shallow surface water flows also keep sediments previously detached by splash in motion. Water on a soil surface that does not infiltrate will flow downhill and typically coalesce into small channels or “rills” (Fig. 3).

Because of the uniform nature of the soil loss, it is often difficult to detect and gauge the extent of damage caused by sheet erosion. On the contrary, rill erosion occurs in well-defined and visible flow concentrations (Fig. 3). Soil detachment in rills is largely because of flow shear stress forces acting on the wetted perimeter of the rill channel (Fig. 4). Once detached, larger sediment particles move as bedload, rolling and bouncing downslope with the flow, and are almost always in contact with the soil (or bed)

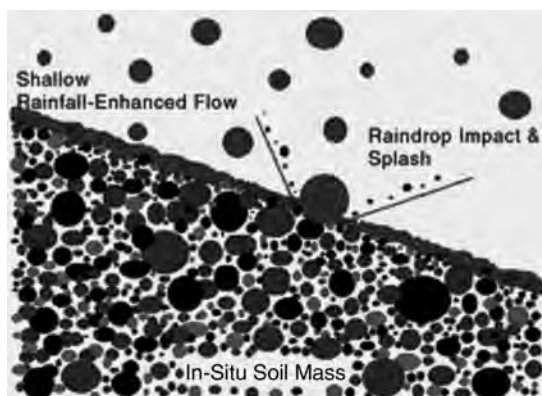


Fig. 2 Soil detachment by raindrop impact and shallow flow transport.



Fig. 3 Rill erosion, which is caused by the concentration of flowing water, forms easily recognizable regions of detachment on a soil surface.

Source: Photo courtesy of M. Huhnke, Oklahoma State University.

surface. Smaller sediment particles (silts and clays) are much easier to transport and travel in the rill channels as suspended load. Rills are also the major pathways for transporting away sediment that is detached by sheet erosion (also known as interrill detachment).

By definition, rill channels are small enough to be obliterated by tillage and will not reform in exactly the same location. As one moves from smaller hillslopes to larger fields and watersheds, additional erosion processes come into play, because of the increasing amounts of runoff water. Gullies are incised erosion channels that are larger than rills and form in regions of large runoff flow concentration. Ephemeral gullies are a common type of erosion feature in many fields (Fig. 5). They are small enough to be tilled over but will re-form in the same location owing to the convergent topography in small catchments. Runoff flows from large events can erode down through tilled soil layers, until a non-erodible layer is reached; then, the ephemeral gully channel will widen, and soil detachment will decrease. Classical gullies are larger erosion features that cannot normally be tilled across (Fig. 6). The physical

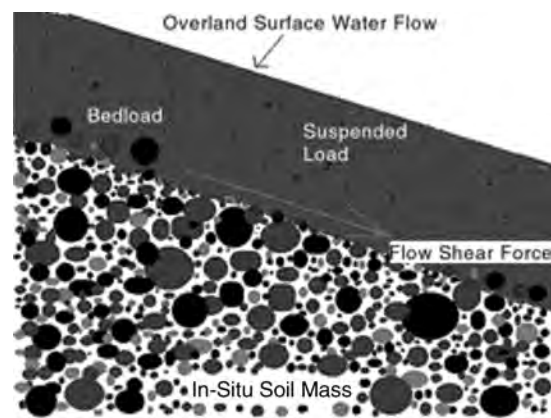


Fig. 4 Soil detachment and transport in rills are largely because of flow shear forces.



Fig. 5 Typical ephemeral gully, located in a soybean field in Indiana.

processes in classical gullies include other factors such as headcutting, seepage, sidewall sluffing, and cleanout of fallen sidewall materials.

As the size of watersheds increases further and streams increase in size and become perennial (because of subsurface water flows from springs and aquifers), the erosion processes in play change as well. Stream and channel erosion at these larger scales can include scouring of the channel beds as well as sediment contributions from the channel banks. Areas in streams may be in states of degradation, in which active detachment is lowering the level of the channel bed, or they may be in states of aggradation, in which sediment deposition is raising the bed level.

GRAVITY-INDUCED EROSION

There are also less frequent but more extreme forms of gravity-induced erosion on steep slopes from saturated soils that can be exacerbated by events such as earthquakes. Large masses of land can slowly or rapidly slide down hills, when the cohesive forces holding them in place fail (land creep, landslide, debris flow, etc.). These types of erosion events typically occur when large rainfall or snowmelt water depths saturate soil profiles and weaken their resistance to slip. Human activities such as removal of



Fig. 6 Classical gullies in the Loess Plateau of China. Terrace farming is being used to stabilize some of the hillslopes.

deep-rooted trees by logging can increase the risks of occurrence of erosion events on susceptible hill slopes.

EROSION ASSESSMENT

Erosion is a serious problem within the United States and throughout the world. In 2007, the U.S. Department of Agriculture–Natural Resources Conservation Service estimated that about 1.7 billion tons of soils are lost each year from non-federal rural lands because of sheet and rill erosions by water and erosion by wind.^[1] Also, 28% of cropland in the United States is eroding at excessive rates. These estimates are on the low side because erosion of other types (e.g., gully) and at other locations (e.g., urban lands and federal lands) were not included in this inventory. Throughout the world, the Food and Agriculture Organization has estimated that 16% of the total land area (21,960,000 km² of 134,907,000 km²) is subject to significant risk of soil erosion.^[2] In Asia, South America, and Africa, soil erosion rates are highest at an estimated average of 30–40 t/ha/yr, while in Europe and North America, average rates are somewhat lower at about 17 tons/ha/yr.^[3,4] A sustainable rate of soil loss (rate of soil loss is equal to rate of soil formation) is thought to be about 1 t/ha/yr.

Erosion assessment can be a difficult task to perform in the field, and monitoring of soil lost and transported by wind or water can be expensive and prone to measurement errors. Gullies are easy to recognize, while soil lost to sheet and rill erosion is hard to gauge. Sheet and rill erosion may be occurring on hill slopes that are adjacent to a gully and may actually contribute more sediment to runoff water than the gully itself. Visual assessment of rates of wind erosion losses can be even more difficult to perform. Mathematical equations or sets of equations have been developed and used since the mid-1900s to estimate the rates of soil loss caused by wind (“Wind Erosion Equation”^[5]) or water (“Universal Soil Loss Equation”^[6,7]). More recently, computer models are being applied to simulate soil erosion processes and to estimate detachment, transport, and deposition of sediment.^[8,9]

EROSION IMPACTS

Erosion has a range of impacts, both on-site and off-site. Soil loss removes fertile topsoil, organic matter, and nutrients, thus decreasing the tilth, water-holding capacity, and general productivity of a soil for on-site agricultural production. Regions of detachment can expand to dislodge and remove small crop seedlings, while regions of deposition can bury and kill small plants. In the case of wind erosion, the erosion process can damage fragile young seedlings through abrasion of plant tissue. When excessive detachment occurs, such as is the case with gully erosion, whole sections of fields may be destroyed or may become inaccessible to farmers and their equipment. Eroded sediment

can cause a number of off-site problems, including deposition along windbreaks, ditches, and waterways. The deposited sediment may require costly dredging and removal operations. Nutrients and pesticides associated with sediments can also contaminate air and water bodies. Erosion by wind can cause massive dust storms that blind drivers and cause highway accidents, and sand particles can abrade and damage painted surfaces on buildings and vehicles. Some estimates are that the cost of combined on-site and off-site effects from soil erosion in the United States is as high as \$44 billion (1992 dollars) per year.^[3]

EROSION CONTROL AND SOIL CONSERVATION

Many nations have created government agencies or organizations to specifically deal with soil erosion problems and to interact with landowners to get conservation practices implemented on the landscape. In the United States, the Natural Resources Conservation Service assists in the implementation of soil conservation practices on agricultural lands, the Forest Service manages sediment delivery from forests and timber harvest roads, and the Bureau of Land Management manages soil loss on range and grazing lands. The Department of Defense is responsible for managing erosion and off-site sediment delivery from lands that it uses for military training activities. However, in some countries, efforts to address and minimize erosion problems are non-existent or severely limited because of poor economic conditions, failure to recognize the erosion threat and/or the extreme magnitude of the soil erosion.

A variety of soil conservation methods is available that can be applied on a landscape to minimize erosion problems caused by wind or water. Wind erosion can be controlled through the use of windbreaks, crop residues, and tillage to induce significant surface roughness. Control procedures for erosion by water need to be determined based on the types of active erosion processes. For example, if sheet and rill erosion are a major problem, then some type of conservation tillage practice that leaves large amounts of crop residues intact on the soil surface may be appropriate. However, if the water erosion problem is because of large amounts of surface runoff water concentrating in a field and forming an ephemeral gully, then crop residues may not be adequate; instead, permanent vegetative cover in a grass waterway may be necessary, along with appropriate engineering structures (e.g., drop box and terraces). Erosion prediction models can be used to assist in selecting and designing appropriate conservation practices.

Through the use of proper conservation planning and application of appropriate soil conservation methods, most erosion problems can be minimized or eliminated. This is critically important if the soil resource is to be preserved for continuous use in food and fiber production for current and future generations.

CONCLUSION

Erosion is a natural process of soil detachment and removal that can be greatly influenced through human activities (agriculture, construction, timber-harvesting, etc.). Use of proper erosion prediction technology and appropriate erosion control methodologies is critical if we are to sustain the soil resource for use by future generations.

REFERENCES

1. United States Department of Agriculture – Natural Resources Conservation Service. *National Resource Inventory*; USDA-NRCS: Washington, DC, 2007. Available at http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs143_012269.pdf (accessed August 2015).
2. Food and Agricultural Organization of the United Nations. *World Soil Resources Report 90*; FAO Land and Water Development Division: Rome, 2000. Available at <ftp://ftp.fao.org/agl/agll/docs/wsr.pdf>.
3. Pimental, D.; Harvey, C.; Resosudarmo, P.; Sinclair, K.; Kurz, D.; McNair, M.; Crist, S.; Shpritz, L.; Fitton, L.; Saffouri, R.; Blair, R. Environmental and economic costs of soil erosion and conservation benefits. *Science* **1995**, *267*, 1117–1122.
4. Barrow, C.J. *Land Degradation*; Cambridge University Press: Cambridge, 1991.
5. Woodruff, N.P.; Siddoway, F.H. A wind erosion equation. *Proc. Soil Sci. Soc. Am.* **1965**, *29*, 602–608.
6. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses—A Guide to Conservation Planning*; Agriculture Handbook No. 537; USDA: Washington, DC, 1978.
7. Renard, K.G.; Foster, G.R.; Weesies, G.A.; McCool, D.K.; Yoder, D.C. Comp. *Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE)*; Agriculture Handbook No. 703; USDA: Washington, DC, 1997; 384 pp.
8. Flanagan, D.C.; Nearing, M.A., Eds. *USDA-Water Erosion Prediction Project (WEPP) Hillslope Profile and Watershed Model Documentation*; NSERL Report No. 10; National Soil Erosion Research Laboratory, USDA-Agricultural Research Service: West Lafayette, 1995; 298 pp. Available at <http://www.ars.usda.gov/Research/docs.htm?docid=18073> (accessed August 2015).
9. Hagen, L.J. A wind erosion prediction system to meet user needs. *J. Soil Water Conserv.* **1991**, *46* (2), 106–111.

Erosion Management: Hill Soils

Narendra Kumar Lenka

Indian Institute of Soil Science, Bhopal, India

B. Krishna Rao

Research Center, Indian Institute of Soil and Water Conservation, Vasad, India

Abstract

Soil erosion is one of the major constraints to sustainable food production in hilly ecosystems. The magnitude gets amplified with improper agricultural management practices and poor adoption of soil conservation practices. The entry describes major biological and mechanical erosion control measures and their suitability. The major biological erosion control measures are contour cultivation, cover cropping, vegetative filter strips, hedge row barriers and selection of appropriate cropping systems having good canopy cover and erosion resistance crops. The mechanical measures normally adopted in the arable sloping lands are contour bunding, graded bunding and terracing. In non-arable lands, drainage line treatments or gully control structures are used, which may be temporary or permanent in nature. Drop spillway, drop inlet spillway and chute spillway are the common permanent gully control structures. Despite the importance of erosion control measures, a holistic approach involving integrated watershed management with participation of the stakeholders is crucial for ensuring sustainable development of hilly regions.

INTRODUCTION

The predominant form of soil erosion in hill soils is water erosion. Because of the land terrain and steep slopes, hilly regions are highly prone to water erosion. Further, in many parts of the world, shifting cultivation or slash-and-burn agriculture is practiced, which aggravates the problems of soil erosion. Without the adoption of management or control practices, the erosion effects get intensified with adverse on-site and off-site impacts. Erosion management principles are based on two basic mechanisms: 1) through establishment/provision of cover on soil surface so as to protect the soil from falling raindrops; and 2) through establishment/provision of barriers so as to reduce the runoff velocity and thus decreasing the erosive power of the runoff water. A number of biological and mechanical measures are available for control and management of soil erosion. However, their use is limited by the technology cost and suitability to the particular soil, climate, and location. Important measures commonly adopted are briefly presented below.

BIOLOGICAL CONTROL MEASURES

Cropping Systems

Cropping systems with inclusion of cover crops help protect the soil surface through spread of quick canopy cover. Thus, the soil exposure to raindrops is reduced in the early crop growth stages. Canopy cover is a measure of the fraction of the soil surface covered by vegetation. Plant

canopy acts as a physical barrier and intercepts the falling raindrops and thus reduces the erosion due to splash and also the kinetic energy of the raindrops is decreased.^[1] Intercropping and strip cropping with erosion-resistant crops such as legumes (except tall and slow-growing crops like Pigeon Pea, *Cajanus cajan* L.) also help in conserving soil from erosion.

Contour Cultivation

In hilly regions, planting crops across the slope in the line of equal elevation (called contour line) is highly essential as it reduces the runoff velocity through the barrier of crop rows and the series of contour furrows that act as mini barriers. All the cultural operations are done across the slope. The effectiveness of contour cultivation depends on the soil type, slope, and crop cover, with maximum effectiveness on medium slope and deep permeable soils.^[3] The relative effectiveness decreases with increase in the degree of slope as evident from [Table 1](#).

Cover Crops

Cover crops are being promoted as an important conservation practice. Cover crops are close growing in nature with fast growth habit and quickly spread the canopy to cover the exposed soil surface. Apart from having soil conservation value, cover crops are also helpful in suppression of weeds, soil health improvement, nutrient recycling, nitrogen fixation, and increasing soil organic matter. Cover crops are mainly grown between two crop seasons. They

Table 1 Effectiveness of contour cultivation on different land slope groups.

Slope groups (% slope)	Ratio of soil loss from contour cultivation and from up and down cultivation
<1%	0.6
2–7%	0.5
7–12%	0.6
18–24%	0.9

Source: From Smith & Wischmeier.^[2] ©1962 Academic Press Inc.

can also be grown as rotational crops and companions to main crops. Because of the fast growth habit, they are able to capture the nutrients from the previous main crops and thus reducing nutrients from leaching losses. In addition to recovery of nutrients from previous crops, they also provide additional nutrients to the succeeding crops. For instance, legume cover crops supply 50–300 kg N ha⁻¹ to the following crops. Use of cover crops also increases biomass return to the soil and thereby increases the microbial activity and improves soil productivity.^[4]

Vegetative Filter Strips

Vegetative filter strips are an area of grass or other permanent vegetation planted between agricultural fields and streams for reducing sediment, nutrients, and other pollutants in water runoff to improve downstream water quality. These buffers are commonly used in the United States and in some parts of Europe.^[2] The filter strips are a useful conservation practice to reduce water pollution with sediment, nutrients, heavy metals, and pesticides from agricultural fields. Under sheet flow, as much as 90% of sediment is reduced by 9-m-wide filter strips. These structures are commonly followed in land slopes up to 8%.

Vegetative Grass Barriers

Vegetative grass barriers are usually two rows of grasses planted by either normal or staggered planting methods, across the slope for erosion control in arable lands. These are narrow strips of thick-stemmed, dense, and close-growing perennial grasses (Fig. 1). The grasses usually selected for the purpose are adapted to local soil type and climate. A common vegetative barrier grass species adopted across countries is the Vetiver (*Vetiver zizanioides*) grass. Normally, vegetative barriers are established at a vertical interval (VI) of 2.0 m between barriers. With vegetative barriers spaced at optimum interval, they reduce the degree of slope and gradually make the land leveled. Normally, they are effective in mild slopes with land slope of 2–6%.

Hedge Row Barriers

These biological measures are effective in arable as well as in non-arable sloping lands, with land slope up to 8%. These



Fig. 1 Vegetative barrier of grass species on arable lands.

are similar to grass barriers, but the plant species used are perennial shrubs (Fig. 2). The species should have strong rooting pattern with good canopy cover, particularly from near-ground level. Some leguminous shrub species like *Indigofera teysmannii* and *Gliricidia sepium* are very suitable for this purpose. Foliage of these species is rich in nitrogen and thus contributes to soil fertility buildup through leaf litter. To cover up the gap between two hedge plants at the ground or near-ground level, sometimes grass species are planted as grass filters, either between the two hedge plants or at staggered points in front of the hedge rows. Studies from eastern India indicate provision of grass filter strips to the hedge row barriers significantly reduces the runoff and soil loss and also contributes to additional soil moisture storage.^[5]

MECHANICAL MEASURES

Mechanical measures involve construction of mechanical barriers across the direction of flow of runoff water to retard the flow velocity. The basic principles behind planning of mechanical measures are 1) to increase the time of concentration of runoff and thus allowing more water to be



Fig. 2 Hedge row barrier of *Indigofera teysmannii* in Eastern Ghat highland zone of India.

absorbed into the soil; and 2) to break long slopes into short ones so as to maintain less than a critical velocity of the runoff water. The common mechanical measures include contour bunding, graded bunding, terracing, contour trenches, and staggered contour trenches.

Contour Bunds

Contour bunds are mechanical barriers such as earthen bunds, constructed across the slope. These are narrow-based trapezoidal embankments on contours to impound runoff water behind them so that all the impounded water is absorbed slowly into the soil profile. They are recommended for low-rainfall areas (<600 mm annual rainfall) and for permeable soils to serve as both a water and a soil conservation measure. They are effective in the 2–6% land slope range.^[4] In the events of high rainfall, they are useful by providing safe diversion of excess runoff and thus allowing deposition of suspended sediments along the bunds. Spacing between two contour bunds is fixed by the vertical interval (VI), which is directly proportional to the land slope.

Graded Bunds

Graded bunds, though similar to contour bunds, are recommended in high rainfall and poorly drained soils. In areas

with high rainfall (>600 mm annual rainfall) and in soils with poor drainage characteristics, there are chances of impoundment of runoff water and flow over the top of the bunds. Hence, a gradient is given to the accompanying channels constructed across the slope, which are thus known as graded bunds.^[4] They are recommended up to 10% land slope. The grade of the channels is decided by the soil type, and spacing between two graded bunds is determined by the VI.

Bench Terraces

Bench terracing involves converting the original ground into level step like fields constructed by half cutting and half filling and is commonly practiced in hilly regions. The factors such as soil depth, uniformity in the spread of top soil, slope of land, rainfall amount and distribution, farming practices, and the proposed crops to be grown have direct bearing on design of bench terraces. The width of the bench terraces depends upon the degree of land slope. They are of three different types: table top, outward sloping, and inward sloping (Fig. 3). In general, bench terraces are recommended for slopes up to 33%, but due to location-specific topographic features and socioeconomic compulsions, this practice is being adopted even up to 100% land slopes (Fig. 4). Although terracing makes farm operation easier, it is costly and in the initial years

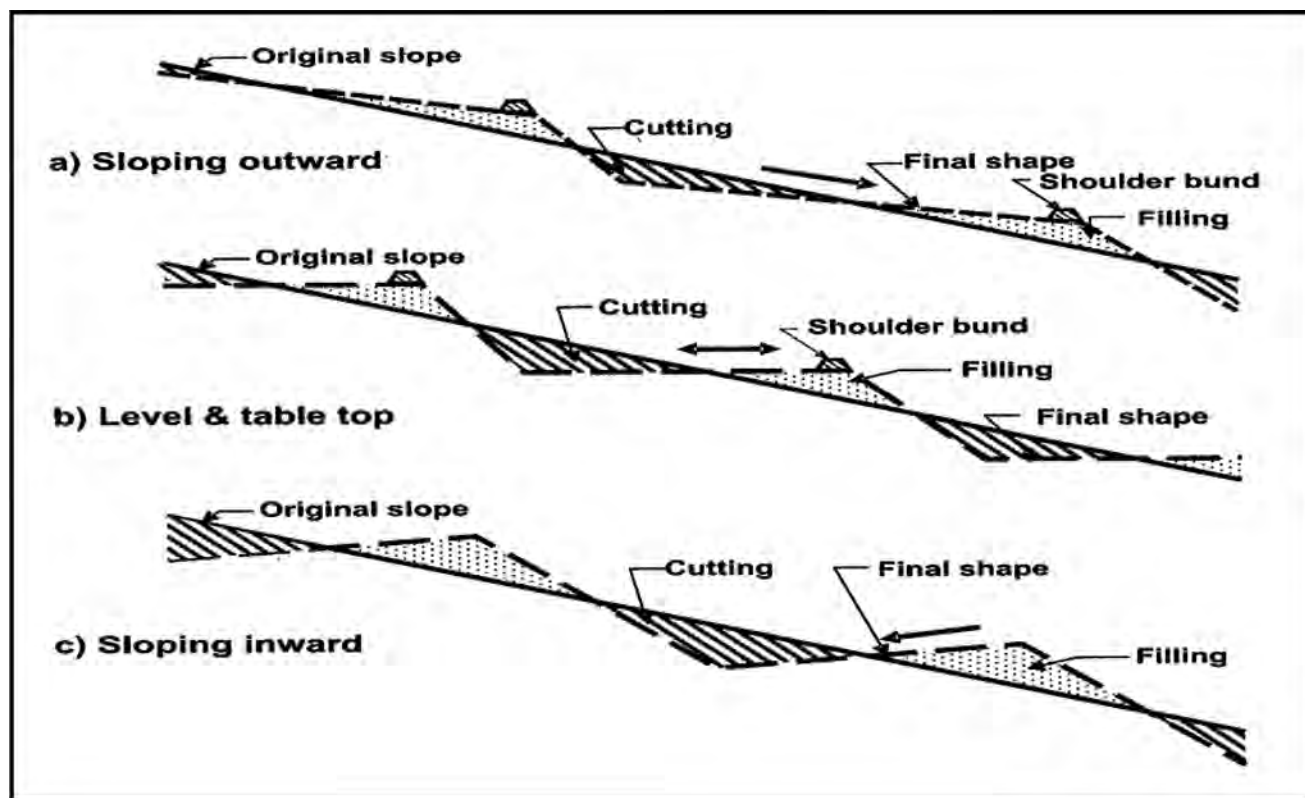


Fig. 3 Schematic diagram showing types of bench terraces.



Fig. 4 Bench terracing in the northeastern hill region of India.

results in the loss of productive soil depth, leading to reduction in crop yield. Hence, in these situations, Puer-torican types of terraces are recommended in which mechanical or vegetative barriers are placed on the hill slopes so that terraces are gradually formed over a period of about 5 years. Sometimes this is also referred to as bioterracing (Fig. 5).

Adaptability of types of bench terraces

Level bench terraces (table top). These are used for paddy cultivation for providing uniform impounding and can be adopted even on mild slopes receiving medium rainfall and having highly permeable deep soils.

Inwardly sloping bench terraces. Some crops are quite susceptible to water stagnation. Hence in order to minimize losses, the terrace is provided with a water disposal channel



Fig. 5 Natural terraces formed between two rows of hedge barriers.

toward the riser of next higher terrace (in the original ground rather than on the fill portion). The inner water disposal channel has a longitudinal grade to safely dispose off excess water into the natural waterway. It is usually adopted in high-rainfall regions in deep soils with good permeability for crops that cannot withstand waterlogging such as potato and maize. These terraces help in quick and safe disposal of runoff through the drain provided on the inner side.

Outwardly sloping bench terraces. These terraces are constructed under low rainfall conditions when soil depth is a limiting factor. The farmers make the benches gradually over a long period of time with shallow permeable soils. Under this situation, water disposal channels or shoulder bunds should be constructed to allow runoff to pass safely from one terrace to another, thus preventing excess runoff from damaging the riser. Where soil depth or rainfall is not limiting, it is considered as the intermediate step for construction of either the level bench or inwardly sloping terraces. In the middle Himalayan region, 70% of the bench terraces are constructed between land slope of 50% and 70% with average outward and longitudinal slopes of 10% and 8%, respectively.

Computation of width and earthwork for terracing

Terrace spacing is usually expressed in terms of VI at which the terraces are constructed. The VI affects the depth of cutting and the height of the bund and thereby the total vertical drops. The width of the terraces should be such that it enables convenient and economic agricultural operations.

The width of terrace (W) can be computed for a given slope (s) by Eq. 1 (Fig. 6):

$$W = \frac{200d}{s} \quad (1)$$

where d is the maximum depth of productive soil range in meters, and s is the land slope in percent.

The earthwork in bench terracing can be computed as follows:

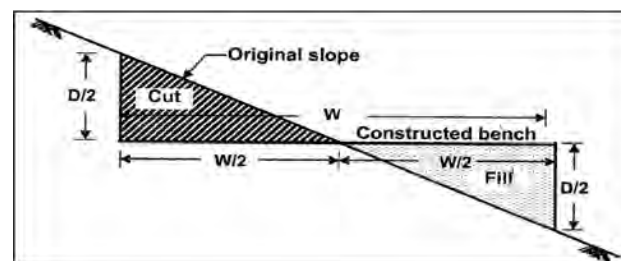


Fig. 6 Schematic diagram showing "cut and fill" in terracing.

$$E = \frac{100Ws}{8} \quad (2)$$

where E is the earthwork (m³), W is the width of terrace (m), and s is the land slope (%).

Grassed Waterways

Grassed waterways serve as safe channels for the exit of the runoff water from a watershed and are designed along the prevailing slope. The most ideal location for grassed waterways is a natural depression or drainage line. In case grassed waterways can't be located in natural courses, they are artificially constructed along the fence lines to avoid inconvenience to farm operations.

Diversion Drains

Diversion drains are provided at the top of arable area to intercept the uncontrolled flow of runoff water from the upper catchment area and to take the water safely to a watercourse. The construction of diversion drains is important since all the conservation measures in the arable area are planned with the assumption that water from the upper nonarable catchment is safely diverted. Diversion drains are provided across the slope on slight gradient, and the capacity is designed for a 10-year recurrence interval storm.

Gully Control Drainage Line Treatment Structures

Gully control structures are provided for establishment of gullies and to prevent the widening of gullies further. They help in reducing the runoff velocity and thus facilitate the establishment of vegetation so that runoff is intercepted at the vegetated locations and thus velocity is significantly lowered in the downstream flow regions. Loose boulder check dams are common type of gully control structures which are less costly and affordable by small farmers. The gully control structures are classified as temporary, semi-permanent, and permanent gully control structures.

Temporary gully control structures

These structures provide support to the vegetation during the initial period of gully establishment. The vegetation once established takes care of the gully. Temporary check dams made of locally available materials like brushwood and logwood are used in small gullies, mostly in the upper reaches where runoff is less. Brushwood check dams, loose boulder structures, gunny bag structures, and earthen gully plugs are the common temporary gully control structures.

Gunny bag structures (boribunds). These types of embankment structures are made from “gunny bags (*bori*)” using available local materials for blocking active and



Fig. 7 A view of *Boribund* structures in gullied lands.

erosion-prone first-order drains. These are made up of empty cement bags filled with sand, clay, and other locally available materials and are placed at the course of the stream to reduce the runoff velocity (Fig. 7). They are used across first- or second-order drains, particularly in places where the runoff velocity is low. The height of the boribund is usually kept at 0.9–1.5 m, and length is kept as equal to the channel width. The spacing depends upon the gradient of the channel bed.

Loose boulder/dry stone check dams

These are most widely used temporary structures in areas where stones are available in plenty (Fig. 8). As far as possible, the location of the check dam should be in the middle of a straight reach of at least 10–15 m. The check dam should be at a right angle to the straight reaches. The loose boulder check dams should have a maximum height of 1 m only. The VI between two structures should be about 1.0 m. The structures should be anchored in the sides of gully up to a length of 0.5–1 m on both



Fig. 8 Loose boulder structures.

sides depending upon site and compaction of soil of gully sides. The level of the structure should be higher at both the sides than the center portion to avoid undercutting on sides.

Semipermanent structures

These structures are placed in the middle reaches, and the life of the structures is up to 15 years. The common semipermanent structures are gabion check dams and logwood dams.

Gabion check dams. Stone wire crate or gabion check dams are commonly used for drainage line treatment in second- and third-order (main) gullies/channels.^[3] The structures are made of stones put in structures woven in galvanized iron (GI) wires (Fig. 9). They are useful for retention of debris and soil accumulation without ponding. The debris-carrying capacity in the gullies between two gabion structures reduces with time due to reduction in channel gradient. The check dam encourages good plant cover not only along the bank but also in the bed of the stream due to increased moisture regime.

Permanent gully control structures

These structures are used where vegetative or temporary structures are not adequate. The major types of permanent structures are drop spillways, drop inlet spillways, and chute spillways.^[3]

Drop spillway. It is a weir structure, and the runoff water passes through the weir opening and drops to an approximate level apron or stilling basin and then passes to the downstream channel (Fig. 10). These structures are effective to control relatively low heads and normally up to 3.0 m. They are constructed of reinforced concrete, plain concrete, rock masonry, and concrete blocks with or without reinforcement or gabions.



Fig. 9 Gabion structures as low cost gully control measures.



Fig. 10 Drop spillway as a drainage line treatment.

Drop inlet spillways. A drop inlet spillway is a closed conduit that carries water under pressure from above an embankment to a lower elevation. The usual function of a drop inlet spillway is to convey a portion of the runoff through or under an embankment without erosion. It is a very efficient structure for controlling relatively high gully heads usually above 3.0 m.

Chute spillways. A chute spillway is an open channel with a steep slope, in which flow is carried at a supercritical velocity. It consists of an inlet, vertical curve section, steep-sloped channel, and an outlet (Fig. 11). Reinforced concrete is widely used to construct chute spillways.

RAINFALL ANALYSIS: RETURN PERIOD/RECURRENCE INTERVAL

For design of soil conservation structures, the important rainfall parameters are amount, duration, intensity, and frequency (recurrence interval). The first three parameters do not need much elaboration; however, it is important to



Fig. 11 A chute structure in place across the drainage line.

Table 2 Recommended maximum design frequencies for various types of structures.

S. No.	Type of structure	Frequency (years)
1	Storage and diversion dams having permanent spillways	50–100
2	Earth fill dams storages having natural spillways	25–50
3	Stock water dams	25
4	Small permanent masonry gully control structures	10–15
5	Terrace outlets and vegetated waterways	10
6	Field diversions	15

understand the concept of return period/recurrence interval of rainfall storms or events.

The design of soil conservation structures requires information about the return period/recurrence interval of rainfall events. Return period/recurrence interval refers to the period in years during which a storm of given duration and intensity is likely to equal or exceed a given amount of rainfall intensity. It does not mean that a given rainfall/storm intensity will repeat exactly after the return period. For example, if the rainfall of 10 cm day⁻¹ at a particular place has a recurrence interval of 5 years, it means that the chances of rainfall are such that once in 5 years, rain is likely to equal or exceed 10 cm. This does not mean that a rain equal to 10 cm or more will occur after every 5 years. It may occur twice in one set of 5 years and may not occur for consecutive 8 years or so.

Recurrence interval plays a crucial role in the economic and safe design of soil conservation structures. It is essentially a compromise between the cost and risk associated with the structure. Depending upon the nature and importance of the structure and risk involved in case of its failure, a life period is assigned, and the structure is designed for the largest rainfall event that is expected to occur during that period. For example, an earthen bund may be designed with a return period of 10 years as its failure does not involve high risk. A water harvesting dam with permanent spillway, on the other hand, needs to be designed for a higher design frequency (50–100 years) storm. In cases where human life is likely to be endangered, the structure is designed for highest ever storm which might have occurred in the available record of rainfall data lasting not less than 50 years. The recommended maximum design frequencies for various types of structures are given in Table 2.

WATERSHED APPROACH

Hydrologically, watershed is a land unit from which the runoff flows to a single outlet. It can be as big as a river catchment and may be small in the size of a microwatershed

covering one or two villages. However, watershed is not only a hydrological unit but also a sociopolitical and ecological entity.

Watershed management refers to implementing land and water management methods to protect soil, water, and other natural resources within a watershed area as well as additional benefits to the downstream areas. Mere construction of soil conservation structures has failed to address the issue of integrated natural resource management in the erosion-prone hilly and rainfed regions. Thus, the conservation of natural resources is being attempted through integrated and participatory watershed approach.^[6] The soil and water conservation measures are only one component of the watershed management program. The other components include entry point activities, capacity-building program of the stakeholders, income-generating activities for landless and marginal farmers, and land-based interventions for crop and livestock development. Major objectives of the watershed programs are as follows:^[7]

- Development and efficient use of land as per the land capability;
- Efficient use of natural resources for livelihood improvement;
- Soil and water conservation and their optimum utilization;
- *In situ* conservation of rain water and water harvesting;
- Flood control and protecting water storage structures and reservoirs from siltation; and
- Maintaining ecological balance.

REFERENCES

1. Blanco, H.; Lal, R. *Principles of Soil Conservation and Management*; Springer: New York, 2008; 557 pp.
2. Smith, D.D.; Wischmeier, W.H. Rainfall erosion. *Adv. Agron.* **1962**, *14*, 109–148.
3. Dhruva Narayana, V.V.; Sastry, G.; Patnaik, U.S. *Watershed Management*; Indian Council of Agricultural Research (ICAR): New Delhi, 1997; 175 pp.
4. Samra, J.S.; Sharma, U.C. *Soil Erosion and Conservation, Fundamentals of Soil Science*; Indian Society of Soil Science: New Delhi, 2002; 15–170.
5. Lenka, N.K.; Dass, A.; Sudhishri, S.; Patnaik, U.S. Soil carbon sequestration and erosion control potential of hedge rows and grass filter strips in sloping agricultural lands of eastern India. *Agr. Ecosyst. Environ.* **2012**, *158*, 31–40.
6. Wani, S.P.; Joshi, P.K.; Ramakrishna, Y.S.; Sreedevi, T.K.; Singh, P.; Pathak, P. A new paradigm in watershed management: A must for development of rainfed areas for inclusive growth. In *Conservation Farming: Enhancing Productivity and Profitability of Rainfed Areas*; Swarup, A., Bhan, S., Bali, J.S., Eds.; Soil Conservation Society of India: New Delhi, 2007; 163–178.
7. Wani, S.P.; Sreedevi, T.K.; Reddy, T.S.V.; Venkateswarlu, B.; Prasad, C.S. Community watersheds for improved livelihoods through consortium approach in drought prone rainfed areas. *J. Hydrol. Res. Dev.* **2008**, *23*, 55–77.

Erosion Tolerance

David L. Schertz

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Washington, District of Columbia, U.S.A.

Mark A. Nearing

Watershed Research Center, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Tucson, Arizona, U.S.A.

Abstract

Soil loss tolerance (T) values have proven tremendously beneficial in developing and evaluating conservation systems that maintain or improve the soil resource. Without them, soil conservationists, farmers, and ranchers would have little basis for evaluating the adequacy of a particular conservation system. With the additional scientific knowledge existing, we have the basis to refine T values to make them more responsive to site-specific conditions.

INTRODUCTION

Scientists first defined the *Soil Loss Tolerance* (T) value in the 1940s as “the amount of soil that could be lost without a decline in fertility, thereby maintaining crop productivity indefinitely.”^[1–3] The Natural Resources Conservation Service [formerly Soil Conservation Service (SCS)] has used T values in conservation planning since the mid-1960s. T value is most commonly defined as “the maximum rate of annual soil erosion that may occur and still permit a high level of crop productivity to be obtained economically and indefinitely.”^[4]

Although not a component of erosion prediction technologies, the T value is used as a basis for comparing estimated soil losses by prediction models to determine whether or not a particular cropping and management system is sustainable and therefore acceptable; e.g., if a particular soil under a specific cropping and management system is estimated to be eroding at 22.4 tonnes/ha/yr, and the T value for that particular soil is 9.0 tonnes/ha/yr, soil erosion would be considered excessive, and additional conservation treatment would be recommended.

Because tremendous technological advances have occurred in the area of soil fertility and amendments, on which the original definition was based, the T value concept may now be outdated. In other words, if any loss in soil fertility because of soil erosion can be replenished with commercial fertilizer, is there cause for concern about soil erosion? The answer to this question is obviously yes, especially if maintenance of our soil resource, as well as other resource concerns, is considered.

BASIS FOR T VALUES

A primary reason for setting T values was to maintain soil fertility and productivity levels. Organic matter was used to indicate the level of soil fertility present. Other soil characteristics considered included soil loss limits to prevent the formation of gullies, variation in soil type, and whether soils contained restrictive layers (e.g., claypan or heavy infertile subsoil).^[5] Early work^[3] assigned a T value of 6.7 tonnes/ha/yr for Putnam soil, and 9.0 tonnes/ha/yr for Marshall and Shelby soils, taking into account that if soil loss exceeded these rates, soil fertility levels would decline. Using available research on a limited set of soils, T values were then assigned to other soils based on interpolation.

The rate of soil formation for each soil type must also be considered in establishing a soil erosion limit. There are even less scientific data available to address this area. Estimates suggest that under natural conditions soil forms at a rate of about 2.5 cm in 300–1000 years.^[6] Under cultivation, however, soil may form at a rate of about 2.5 cm in 100 years. A research suggests that the formation of the A horizon may occur at a rate of 2.5 cm in 30 years for medium to moderately coarse-textured soils but would be slower in finer-textured soils.^[7] However, increasing the depth of a favorable rooting zone through weathering takes much longer than the development of the A horizon. One estimate indicates that we might expect to develop 1.1 tonnes/ha/yr of soil for unconsolidated parent material and much longer for consolidated parent material.^[8]

Assuming a soil density of 1320 kg/m, if the A horizon renews itself at the rate of 2.5 cm every 30 years, an

average erosion rate of 11.2 tonnes/ha/yr would maintain the A horizon. However, assuming that rooting depth is renewed at the rate of 1.1 tonnes/ha/yr from unconsolidated parent material, total rooting depth will continue to decrease even with soil erosion occurring at 11.2 tonnes/ha/yr.^[5] Therefore, if soils erode for several centuries at T levels, considerable loss in soil productivity may result.^[9] However, reducing the allowable soil loss to 1.1 tonnes/ha/yr (the level estimated to be the rate of rooting depth development for unconsolidated parent material) would likely cause significant production constraints. Hugh Hammond Bennett, the first Chief of the Soil Erosion Service, stated in 1939, “As nearly as can be ascertained, it takes nature, under most favorable conditions, including a good cover of trees, grass, or other protective vegetation anywhere from 300 to 1000 years or more to build a single inch of topsoil. When seven inches of topsoil is allowed to wash away, therefore at least 2000–7000 years of nature's work goes to waste.”^[10]

A 1956 joint conference involving the U.S. Department of Agriculture, Agricultural Research Service, and the SCS addressed the issues surrounding the establishment of T values.^[11] As a result of this conference, a committee established the following specific factors as being important in establishing T values:

- 1. Maintenance of adequate soil depth
- 2. Value of nutrients lost
- 3. Maintenance of water control structures and control of floodplain sedimentation
- 4. Prevention of gullies

- 5. Yield reduction per inch of topsoil lost
- 6. Water losses
- 7. Seeding losses

This committee also set the caveat that T values should not exceed 11.2 tonnes/ha/yr. The SCS held six regional workshops in 1961 and 1962 that resulted in T values from 2.2 tonnes/ha/yr to 11.2 tonnes/ha/yr being established in each region.^[8] Table 1 shows the percent distribution of T values in 12 soil orders occurring in the United States. Although some adjustments have been made to the criteria for establishing T values, the basic structure is in use.

EXISTING CONCERNS

Since the late 1950s and early 1960s, soil erosion prediction technology has significantly improved. The Universal Soil Loss Equation was replaced by the Revised Universal Soil Loss Equation in 1995. In addition, considerable work has been completed on process-based models for both wind and water erosion. As erosion prediction models become more sophisticated and better represent site-specific conditions, questions arise as to whether traditionally defined T values are outdated.

Because T values were developed primarily for use on cropland soils, there is a concern that they may not be applicable to permanent pasture or rangeland. For the very fragile landscapes on which rangeland often occurs, tolerable soil losses might need to be

Table 1 T values expressed in percent of soils mapped for each soil order in the United States.

Soil order	Order code	T Value (t/ha)					Total area (ha) ^a
		2.2	4.5	6.7	9.0	11.2	
Alfisol	A	1	6	26	16	51	103,829,046
Andisol	C	2	18	33	15	31	9,307,531
Aridisol	D	21	19	12	3	44	53,041,017
Entisol	E	10	15	5	2	68	78,667,763
Gelisol	G	23	61	9	1	7	141,601
Histosol	H	16	32	30	4	19	7,711,917
Inceptisol	I	8	19	37	5	32	54,910,918
Mollisol	M	10	9	11	6	64	166,850,655
Oxisol	O	0	2	2	8	89	143,521
Spodosol	S	12	15	17	9	48	20,028,476
Ultisol	U	1	5	24	24	46	71,305,466
Vertisol	V	0	3	19	8	70	12,546,385

Note: ha, hectares; t, tonnes; T value, soil loss tolerance value.

^aTotal hectares shown represent approximately 68% of the total land area in the United States stored in the Soil Survey Database.

Source: Data from Soil Survey Staff.^[12]

significantly lower than those established for similar, more fertile soils that are cropped. For example, where a maximum of 11.2 tonnes/ha/yr is recommended for soils that are cropped, a limit of 5–7 tonnes/ha/yr might be more appropriate for rangeland. In addition, sheet and rill erosion occurs less often and concentrated flow erosion is often of greater concern on rangeland than on cropland. T values, which address only sheet and rill erosion, are not applicable to concentrated flow or gully erosion. Therefore, consideration should be given to establishing T values based on the specific use of the land. For example, it is not appropriate to use a T value of 11.2 tonnes/ha/yr for a range site that would likely be devastated if more than 5 tonnes/ha of soil erosion occurred on an annual basis. Productivity of the site, potential loss of desired vegetation, and soil rooting depth must therefore be taken into consideration.

In contrast to on-site effects caused by sheet and rill erosion, off-site impacts of soil erosion—such as sediment damage, water and air quality degradation, and emission of greenhouse gases—are of greater concern. Society's increasing urban nature and environmental consciousness have made off-site concerns more important than they were at the time that soil tolerance values were established. Formally incorporating the societal costs of off-site damages into the concept of soil loss tolerance is a major challenge for the future. Some of those costs are direct and relatively easy to quantify, such as the cost of dredging navigable waterways, ditches, and reservoirs. Other costs, such as those associated with the degradation of water quality in lakes and streams, impact their use by humans, fish, and wildlife and are therefore less easily quantified.

The sediment discharge and the amount of deposition in sensitive areas resulting from the eroding fields depend greatly on the distance and the transport conditions between both areas. For example, excessive erosion on a field of several kilometers from a sensitive area is less of a potential concern than erosion on a field adjacent to that same sensitive area. Therefore, proximity to sensitive areas, as well as transport conditions, may need to be considered when establishing or revising T values. In addition, fish and other wildlife vary in their ability to tolerate sediment and associated pollutants. Some species are very sensitive, while others are not, indicating that a single tolerance value is inadequate when a broader set of concerns is evaluated.

If site-specific T values become an option, they must consider these concerns in order to be effective in protecting and enhancing all of our natural resources. In addition, a more modern approach to traditional productivity losses from erosion is needed. The short- and long-term cost and benefit trade-offs of soil conservation

can and should be addressed using state-of-the-art methods of establishing T values.

CONCLUSION

T values have proven tremendously beneficial in developing and evaluating conservation systems that maintain or improve the soil resource. Without them, soil conservationists, farmers, and ranchers would have little basis for evaluating the adequacy of a particular conservation system. Several national programs, including Conservation Compliance, Conservation Reserve Program, and the Environmental Quality Incentives program, have been implemented based, in part, on the T value concept.

Because of its widespread use in program activities, it is time to reevaluate the T value concept. Perhaps, a T value assigned to an individual soil should be viewed as a generic T value or basic starting point. Tables, equations, or process-based models could be developed to adjust the generic T value based on various conditions, such as land use, proximity to sensitive areas, susceptibility of wildlife species to sediment and associated contaminants, and other off-site concerns. For example, an assigned T value for a cropped soil may need to be reduced to maintain the ecological integrity of a rangeland site. This same T value may need to be reduced further, depending on off-site considerations, or a T value for a cropped site may need to be adjusted downward if excessive sedimentation would be detrimental to the proper functioning of an adjacent wetland.

With the additional scientific knowledge existing, we have the basis to refine T values to make them more responsive to site-specific conditions.

REFERENCES

1. Browning, G.M.; Parish, G.L.; Glass, J. A method for determining the use and limitation of rotation and conservation practices in the control of soil erosion in Iowa. *J. Am. Soc. Agron.* **1947**, *39*, 65–73.
2. Smith, D.D. Interpretation of soil conservation data for field use. *Agr. Eng.* **1941**, *22*, 173–175.
3. Smith, D.D.; Whitt, D.M. Evaluating soil losses from field areas. *Agr. Eng.* **1948**, *29*, 394–398.
4. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses—A Guide to Conservation Planning*; Science and Education Administration Agriculture Hand book No. 537; U.S. Department of Agriculture: Washington, 1978; 1–58.
5. Schertz, D.L. The basis for soil loss tolerances. *J. Soil Water Conserv.* **1983**, *38*, 10–14.
6. Pimentel, D.; Terhune, E.C.; Dyson-Hudson, R.; Rochereau, S.; Samis, R.; Smith, E.A.; Denman, D.; Reifschneider, D.; Shepard, M. Land degradation: Effects on food and energy resources. *Science* **1976**, *194*, 149–155.

7. Hall, G.F.; Daniels, R.B.; Foss, J.E. Rate of soil formation and renewal in the USA. In *Determinants of Soil Loss Tolerance*; ASA Publication No. 45; American Society of Agronomy: Madison, 1982.
8. McCormack, D.E.; Young, K.K.; Kimberlin, L.W.- Current criteria for determining soil loss tolerance. In *Determinants of Soil Loss Tolerance*; ASA Publication No. 45; American Society of Agronomy: Madison, 1982.
9. McCormack, D.E.; Young, K.K. Technical and societal implications of soil loss tolerance. In *Soil Conservation, Problems and Prospects*; Morgan, R.P.C., Ed.; John Wiley & Sons: Chichester, 1981; 365–376.
10. Bennett, H.H. *Soil Conservation*; McGraw-Hill: New York, 1939; 993 pp.
11. Paschall, A.H.; Klingebiel, A.A.; Allaway, W.H.; Bender, W.H.; Carpenter, W.W.; Glymph, L.M. *Permissible Soil Loss and Relative Erodibility of Different Soils*; Committee Report; U.S. Department of Agriculture, Agricultural Research; Service and Soil Conservation Service: Washington, 1956.
12. Soil Survey Staff. *National Map Unit Interpretation Record (MUIR)*; U.S. Department of Agriculture, Natural Resources Conservation Service; National Soil Survey Center: Lincoln and Iowa State University Statistical Laboratory: Ames, 1997.

Erosion: Aggregate Breakdown Mechanisms

Yves Le Bissonnais

Orleans Research Center, Science of the Sun Unit, National Institute of Agronomic Research (INRA), Orleans, France

Abstract

Crusting and interrill erosion on cultivated soils result mainly from aggregate breakdown and detachment of fragments by raindrops. The measurement of fragment size distribution following breakdown and the analysis of aggregate breakdown mechanism allow the assessment of the soil's susceptibility to crusting and erosion. It will help to replace the actual empirical erodibility parameters in erosion models by physically based soil-specific characteristics. The knowledge of the different aggregate breakdown mechanisms will also allow to establish the relationship between soil properties and erosion processes.

INTRODUCTION

Aggregate breakdown influences several aspects of soil's physical behavior, such as infiltration, crusting, and erosion.^[1,2] Soil susceptibility to breakdown into finer, more transportable, particles and microaggregates is of major importance in erosion processes. Aggregate breakdown measurement is often considered as a good mean for soil erodibility assessment.^[3,4]

Various processes are responsible for aggregate breakdown. The type of breakdown process affects the intensity of aggregate breakdown and the size distribution of soil fragments available for detachment and transport. Soil particles detachability and importance of runoff volume during interrill erosion are governed by the formation of soil crust. Hydrodynamic properties and formation characteristics of soil crust are related to size distribution of breakdown products.^[5–7]

Soil loss can be described as a net balance between the detachment of *in situ* soil fragments by splash and overland flow, transport of the previously detached sediment, and deposition. In the case of interrill erosion, the detachment of soil fragments is due to the impact of raindrops (splash), and transport is principally performed by overland flow.^[8,9] These erosion subprocesses depend on the size distribution of available soil fragments provided by previous soil breakdown.

THE MECHANISMS OF AGGREGATE BREAKDOWN

Aggregate breakdown by water may result from a variety of physico-chemical and physical mechanisms and may involve different scales of soil structure from clay particle interactions to the macroscopic behavior of aggregates.^[4,10–12] Four main mechanisms can be identified from

the various reviews already available: 1) physico-chemical dispersion due to osmotic stress; 2) slaking, i.e., breakdown caused by the compression of entrapped air during wetting; 3) breakdown by raindrop impact; and 4) breakdown by differential swelling.

These mechanisms differ in many ways: in the nature of interparticle bonds and the energy involved in their disruption, in the physical and chemical conditions required for disaggregation, in the kinetics of the breakdown process, in the type of soil properties influencing the mechanism, and in the nature and size distribution of the breakdown products.

Physico-Chemical Dispersion

Physico-chemical dispersion results from the reduction of the attractive forces between colloidal particles while wetting. Stability or dispersion depends on cation size and valence. Polyvalent cations cause flocculation, whereas the monovalent cations cause dispersion. Dispersion is affected by the electrolyte concentration (EC) of the soil solution, the EC of applied water, and by the mechanical disturbance from raindrop impact or shaking.

Dispersion depends mainly on the exchangeable sodium percentage (ESP) of the soil. The defining characteristic of dispersion is the production of elementary particles rather than microaggregates. Therefore dispersion is the most effective process of aggregate breakdown, and it greatly increases the effect of the others. Dispersion generally induces very fast crusting, slow infiltrability, and great mobility of particles in water.

Breakdown by Compression of Trapped Air: Slaking

Slaking is caused by the compression of air entrapped inside aggregates during wetting. It occurs when dry

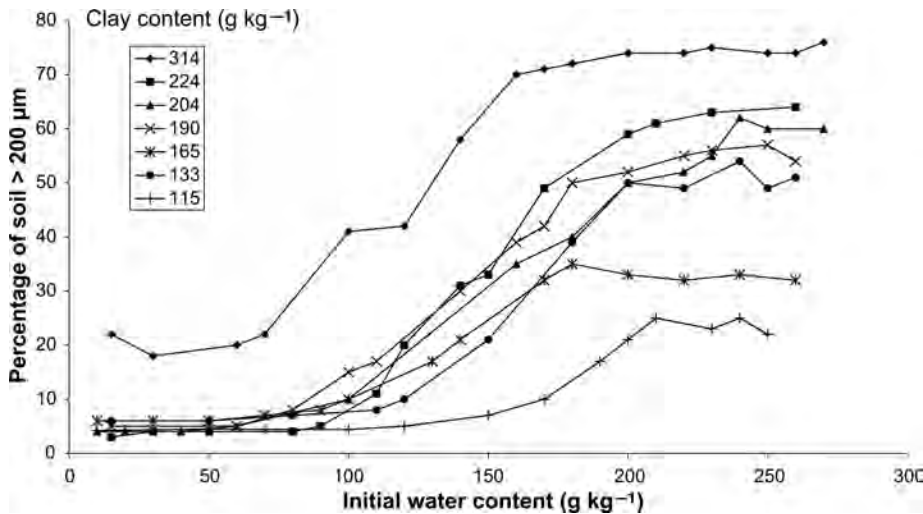


Fig. 1 Aggregate stability (percentage of fragments $>200\ \mu\text{m}$ after immersion of 3–5 mm aggregates in water) as a function of initial water content and clay percentage, for various cultivated soils from northern France.

Source: From Le Bissonnais.^[4]

aggregates are immersed in water or rapidly wetted. The effect of trapped air depends on the volume of air inside the aggregates, on the rate of wetting, and on the shear strength of wet aggregates. Slaking occurs even without any shaking of soil in water, although shaking increases the effect of slaking because of additional mechanical breakdown or dispersion. Le Bissonnais^[4] found that slaking decreases as the initial moisture content increases until saturation is reached (Fig. 1). This is due to the reduction of the volume of air that is entrapped during wetting and also to the reduction of gradients of matric potential. Fig. 1 also shows that, in the range of 100 to 300 g kg^{-1} of clay content, breakdown decreases when clay content increases.

The fragments resulting from slaking are mainly microaggregates, and they increase in size with increasing clay content. This can be explained according to the model of textural porosity.^[13] The clay volume and the resistance of clay bonds between skeleton grains increase with clay content, so the elementary assemblages of primary particles are bigger.

Mechanical Breakdown by Raindrop Impact

Mechanical breakdown of aggregates by raindrop impact usually occurs in combination with the other mechanisms if the kinetic energy of raindrops is great enough. The importance of this effect is clearly demonstrated by the role of vegetation cover or mulch, which protects the soil surface against such impact. Mechanical breakdown by raindrop impact plays a dominant role in wet soils because the aggregates are weaker when the soil is wetter. Also, under “undrained” conditions, the compressive stress of the raindrop impact is transformed into lateral shear stress that causes the fragments to detach and project. Therefore raindrop impact not only detaches but also displaces previously detached fragments.^[2,5] This mechanism is called the splash effect. Because very stable macroaggregates can be moved by splash, a discrepancy between aggregate

stability and splash measurement under rainfall may be expected. The fragments resulting from raindrop detachment are generally small, being either elementary particles or small microaggregates ($<100\ \mu\text{m}$).

Breakdown by Differential Swelling

Differential swelling and shrinkage during wetting and drying of clay soil result in a microcracking of aggregates. It depends on the same properties as slaking, including the rate of wetting. It produces microaggregates that are about the same size and type as microaggregates resulting from slaking of initially wet soils. However, we make the distinction between the two mechanisms because physical processes are different, although some authors use the word “slaking” for both mechanisms. Breakdown by slaking decreases as clay content increases, whereas breakdown by differential swelling increases with increasing clay content. The consequences of breakdown by differential swelling (microcracking) on infiltration are less severe than those of slaking or dispersion because of the larger size of the resulting fragments. However, microcracking may accelerate the formation of structural crusts by reducing the mean aggregate size.

Soil Properties Influencing Aggregate Breakdown

The main soil properties influencing aggregate breakdown and erosion are soil texture, clay mineralogy, organic matter content, type and concentration of cations, sesquioxide content, and calcium carbonate content, with multiple interactions between these properties that can modify their individual influence.^[10,11] ESP affects particle dispersion, iron and aluminum oxides, and oxyhydroxide cement aggregates, particularly for tropical and lateritic soils. Organic matter protects the surface against raindrop impact; it is a bonding agent between mineral soil particles and it may impart hydrophobic characteristics that reduce slaking.

THE RELATIONS BETWEEN AGGREGATE BREAKDOWN AND ERODIBILITY

Soil erodibility may be defined as the inherent susceptibility of soil to detachment and transport by rain and runoff. Relations exist between aggregate breakdown, crusting, and erosion.^[5,6]

Crust Formation

For most cultivated lands, erosion results from runoff caused by the decrease of infiltration rate due to surface sealing and crusting by rain. The infiltration rate may decrease to as little as 1 mm hr⁻¹. Another effect of sealing is the reduction of surface roughness that results in smaller depressional storage and in more uniform flow over the surface.

Two main morphological types of crust can be distinguished. Structural crusts are formed by a reorganization with little displacement of fragments and without sorting and sedimentation. They result from the gradual packing and coalescing of particles and small aggregates, which are mainly produced by slaking and microcracking. Depositional or sedimentary crusts result from fragment and particle displacement and sorting under ponding conditions. The mechanisms are mainly dispersion and mechanical breakdown by raindrop impact. These two crust morphologies correspond to two successive stages in a general pattern of seal and crust development.^[14] Structural crusts are generally formed during the preponding phase, whereas depositional crusts and erosion are co-incident. A distinction between the various mechanisms of breakdown may help to explain the dynamics of surface crusting and erosion.

Whether they are primary particles or microaggregates, fragments <100 µm in size packed together result in pores smaller than 15 µm. It can be shown from Poiseuille's law, which describes flow discharge in a

cylinder and the measurements of raindrop pressure, that for pores <15 µm in size, frictional effects reduce water penetration:

$$\text{Poiseuille's law: } Q = \pi R^4 P / 8 \eta L$$

With Q = flow discharge (m³ sec⁻¹), R = radius, P = pressure, η = viscosity, and L = length.

Image analysis and mercury porosimetry support this theoretical approach and show that if all the pores are <15 µm in size then infiltration almost ceases.

Soil Fragments Release

In addition to being the primary cause of crusting, aggregate breakdown is responsible for the production of microaggregates and particles, which are easily transported by splash and runoff. This effect of breakdown is particularly important in interrill erosion, where the flow detachment capacity is limited (Fig. 2). The conceptual models for soil erosion in interrill areas divide soil erosion into subprocesses: detachment, transport, and sedimentation. In the process-based model developed by Hairsine and Rose,^[9] it has been supposed that soil detachment is due to raindrop impacts and that the only role of overland flow is to transport downslope the detached aggregates. This model calculates the fragment size distribution leaving a uniform area through the concept of settling velocity classes. It describes the rate of rainfall detachment and uses an empirical detachability parameter. We hypothesize that aggregate breakdown measurement can be used to model the size distribution of fragments resulting from raindrop effect.

Moreover, the fragment size distributions of splash and overland flow sediment have been observed to be finer than the original soil, as a result of selective removal of silt and clay-sized particles or preferential deposition of coarse fragments. However, the size selectivity of wash and splash subprocesses has generally been studied in comparison

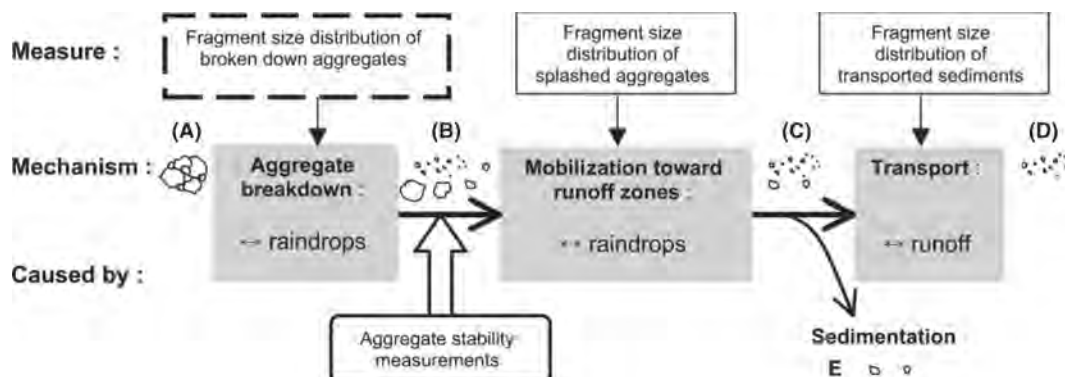


Fig. 2 Conceptual modeling of interrill erosion in subprocesses: breakdown, mobilization, and transport/sedimentation. (A) Initial aggregates; (B) breakdown into fragments; (C) raindrops mobilize fine fragments; (D) available for transport by overland flow for the finest; (E) or to sediment for the coarsest.

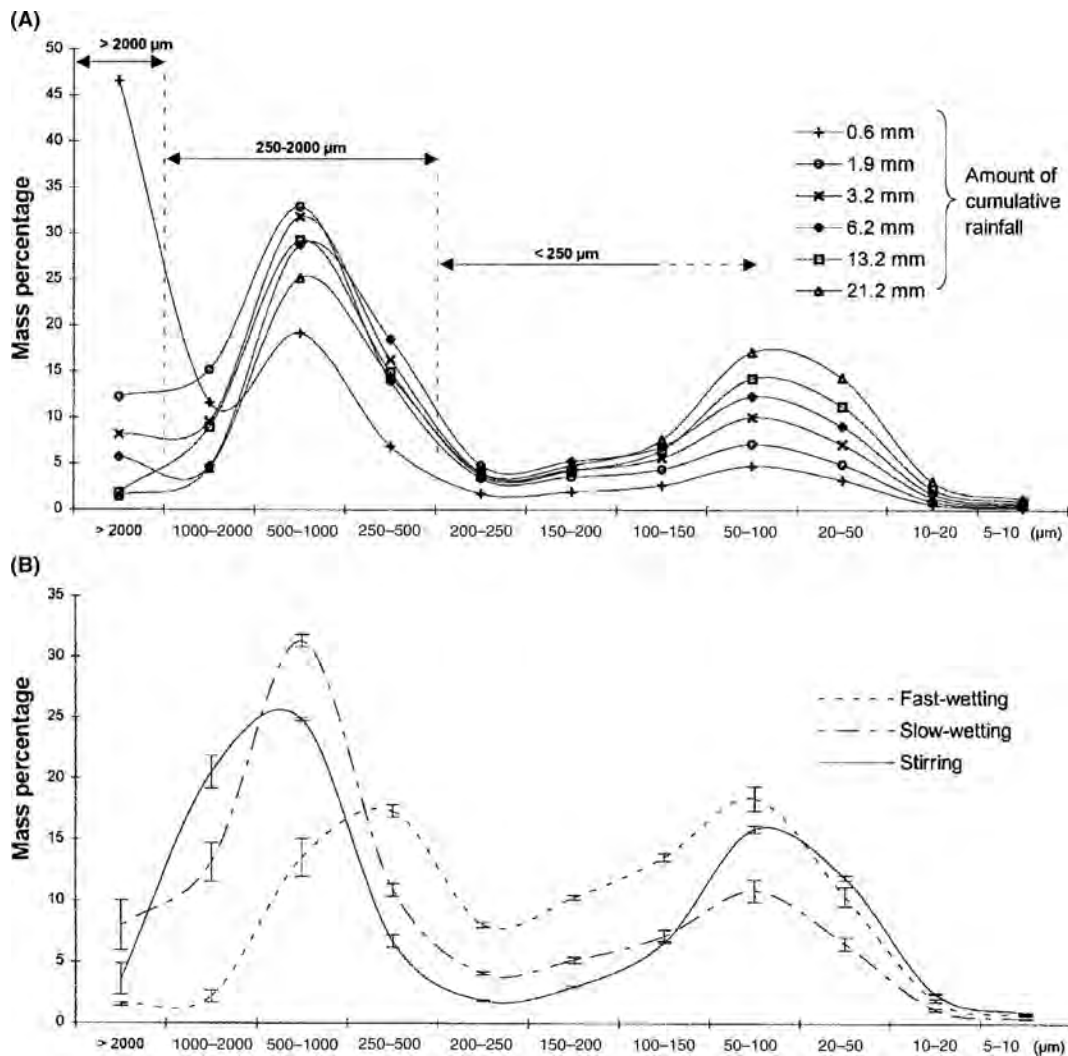


Fig. 3 Fragment size distribution resulting from breakdown of 3–5 mm aggregates for a silt loam. (A) Comparison between size distribution corresponding to different heights of cumulative rainfall and (B) the three aggregate stability treatments. Three granulometric size ranges with specific dynamics can be distinguished in (A). Standard errors are represented for each aggregate stability treatment in (B).

with the soil matrix without considering the aggregate breakdown. Knowing the dynamic of size distribution of breakdown products helps understand both the sealing processes and the selectivity of splash and overland flow

(Fig. 3). Table 1 gives the various classes of values for aggregate stability expressed as mean weight diameter (MWD)^[4] and the relationship with crusting and interrill erodibility.

Table 1 Classes of aggregate stability, crustability, and interrill erodibility according to MWD values measured with the three treatments offered.

Class	Examples of soil	MWD value (mm)	Aggregate stability	Crustability and interrill erodibility
1	Sodic soil, silty soil...	<0.4	Very unstable	Systematic crust formation very high erodibility
2	Silty clay loam with low organic matter content...	0.4–0.8	Unstable	Crusting frequent high erodibility
3	Silty clay loam with high organic matter content...	0.8–1.3	Medium	Moderate crusting moderate erodibility
4	Clay loam...	1.3–2.0	Stable	Crusting rare low erodibility
5	Heavy clay or organic soil...	>2.0	Very stable	No crusting very low erodibility

Note: The differences in MWD values for the three treatments for a given soil correspond to the differences in soil behavior, which varies with initial water content and rainfall characteristics.

CONCLUSION

Crusting and interrill erosion on cultivated soils result mainly from aggregate breakdown and detachment of fragments by raindrops. The measurement of fragment size distribution following breakdown and the analysis of aggregate breakdown mechanism allow the assessment of the soil's susceptibility to crusting and erosion. It will help to replace the actual empirical erodibility parameters in erosion models by physically based soil-specific characteristics. The knowledge of the different aggregate breakdown mechanisms will also allow to establish the relationship between soil properties and erosion processes.

REFERENCES

1. De Ploey, J.; Poesen, J. Aggregate stability, runoff generation and interrill erosion. In *Geomorphology and Soils*; Richards, K.S., Ed.; Allen and Unwin: London, 1985; 99–120.
2. Farres, P.J. The dynamics of rainsplash erosion and the role of soil aggregate stability. *Catena* **1987**, *14*, 119–130.
3. Bryan, R.B. The development, use and efficiency of indices of soil erodibility. *Geoderma* **1968**, *2*, 5–26.
4. Le Bissonnais, Y. Aggregate stability and assessment of soil crustability and erodibility: I. Theory and methodology. *Eur. J. Soil Sci.* **1996**, *47*, 425–437.
5. Bradford, J.M.; Ferris, J.E.; Remley, P.A. Interrill soil erosion processes: I. Effects of surface sealing on infiltration, runoff, and soil splash detachment. *Soil Sci. Soc. Am. J.* **1987**, *51*, 1566–1571.
6. Fox, D.; Le Bissonnais, Y. Process-based analysis of aggregate stability effects on sealing, infiltration, and interrill erosion. *Soil Sci. Soc. Am. J.* **1998**, *62* (3), 717–724.
7. Loch, R.J.; Foley, J.L. Measurement of aggregate breakdown under rain: Comparison with tests of water stability and relationships with field measurements of infiltration. *Aust. J. Soil Res.* **1994**, *32*, 701–720.
8. Kinnell, P.I.A. The mechanics of raindrop-induced flow transport. *Aust. J. Soil Res.* **1990**, *28*, 497–516.
9. Hairsine, P.B.; Rose, C.W. Rainfall detachment and deposition: Sediment transport in the absence of flow-driven processes. *Soil Sci. Soc. Am. J.* **1991**, *55*, 320–324.
10. Emerson, W.W.; Greenland, D.J. Soil aggregates—Formation and stability. In *Soil Colloids and Their Associations in Aggregates*; De Boodt, M., Hayes, M., Herbillon, A., Eds.; Plenum Press: New York, 1990; 485–511.
11. Amézketa, E. Soil aggregate stability: A review. *J. Sustain. Agr.* **1999**, *14* (2/3), 83–151.
12. Oades, J.M.; Waters, A.G. Aggregate hierarchy in soils. *Aust. J. Soil Res.* **1991**, *29*, 815–828.
13. Fies, J.-C.; Stengel, P. Densité texturale de sols naturels: II. Eléments d'interprétation. *Agronomie* **1981**, *1*, 659–666.
14. Bresson, L.-M.; Boiffin, J. Morphological characterization of soil crust development stages on an experimental field. *Geoderma* **1990**, *47*, 301–325.

Erosion: Agronomic Practices

Glenn A. Weesies

National Soil Erosion Research Laboratory, U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), West Lafayette, Indiana, U.S.A.

David L. Schertz

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Washington, District of Columbia, U.S.A.

William F. Kuenstler

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Fort Worth, Texas, U.S.A.

Abstract

Soil erosion by wind and water degrades our soils by removing material from the fertile topsoil. Windstorms and rainstorms detach and transport soil particles high in nutrients and organic matter. The result is a soil surface with reduced plant-available nutrients, high in bulk density, and low in porosity and capacity for water intake.

SOIL EROSION AND ITS CONTROL

In regions with adequate rainfall, nature's way of protecting the soil is for vegetation to grow during periods of adequate rainfall and temperature, die during periods of low rainfall or temperature, and emerge again to start the next cycle. Either growing plants or plant residue would cover the soil to protect it from erosion. During the process of soil formation, the cycle of growing plants and decaying residue would provide protection for the soil while new soil developed.^[1]

Mechanization has, over the 20th century, allowed farmers to till the soil and plant row crops continuously under intensive cultivation. This has seriously altered nature's system of producing topsoil and protecting the soil from erosion. Under intensive, conventional tillage, soils are left unprotected and exposed to wind, rainfall, and runoff. Erosion has reduced the depth of topsoil, altered its natural tilth and structure, and reduced its productivity. Further advances in technology have made it possible to employ techniques similar to nature's methods of protecting the soil during crop production by leaving the previous crop's residue on the soil surface, adding nutrients to the soil, and maintaining its tilth and structure.^[1] Of all the tools that technology has made available to farmers to control erosion, the most efficient and economical are agronomic practices—those that involve management practices that protect the soil during production of crops.

AGRONOMIC PRACTICES THAT CONTROL EROSION

Types of Agronomic Practices

The more common agronomic practices used to control erosion on cropland are described as follows:

Residue management

Residue management is managing the amount, orientation, and distribution of crop and other plant residue on the soil surface year-round while growing crops.^[2] Crops are grown in narrow slots, in residue-free strips in previously untilled fields, or in fields where the entire surface is tilled prior to planting. It includes limiting or reducing tillage to leave more residues on the surface. An increase in ground cover of just 10% can reduce soil loss by as much as 20%.

Conservation tillage

Conservation tillage is any tillage and planting system that covers 30% or more of the soil surface with crop residue, after planting, to reduce soil erosion by water. Where soil erosion by wind is the primary concern, any system that maintains the equivalent of at least 1120 kg/ha (1000 lb/acre) of flat, small grain residue on the soil surface throughout the critical wind erosion period qualifies as conservation tillage.^[3] Types of conservation tillage

include no-till, strip-till, ridge-till, and mulch-till. With this agronomic practice, tillage tools are selected and operated to maximize the amount of residues left on the surface. Weed control is accomplished with a combination of herbicides and cultivation.

Contour farming

Contour farming is tillage, planting, and other farming operations performed on or near the contour of the field slope.^[2] Crops are planted around the hill nearly on the level rather than up and down the hill. This makes many short slopes out of one long slope. Contouring can reduce erosion by 50% in many instances.

Contour strip cropping

This is growing row crops, forages, small grains, or fallow in a systematic arrangement of equal width strips on or near the contour of the field slope.^[2] This practice involves a combination of crop rotation and contouring. Contour strip cropping can reduce soil loss by 75% in many instances (Fig. 1).

Contour buffer strips

These are narrow strips of permanent, herbaceous vegetative cover established across the slope. These strips are alternated down the slope with parallel, wider cropped strips.^[2] The permanent vegetation in the buffers is typically grass, legumes, or a grass-legume mix. Contour buffer strips can reduce erosion by 60% in many instances.



Fig. 1 A field with a contour strip cropping system. Strips of corn are alternated with strips of a hay crop. Planting is on the contour.

Source: Photo courtesy of USDA, Natural Resources Conservation Service.

Filter strips

A strip or area of herbaceous vegetation situated between cropland, grazing land, or disturbed land (including forest land) and environmentally sensitive areas.^[2] Typically, a filter strip is placed at the base of a slope to filter sediment and other contaminants from runoff water before they reach offsite areas or bodies of water.

Crop rotation

This involves growing crops in a recurring sequence on the same field. Erosion control is achieved by changing the crops year by year to alternate high residue-producing crops with lower residue producers, close-growing crops with row crops or perennial crops with annual crops. Some rotations can reduce erosion by 60%.

Cover crops

Grasses, legumes, forbs, or other herbaceous plants established for seasonal cover and conservation purposes are known as cover crops.^[2] Cover crops temporarily protect the soil during seasons when major crops are not growing (Fig. 2).

Grassed waterways

These are natural or constructed channels that are shaped or graded to required dimensions and established with suitable vegetation.^[2] This practice is used where water usually concentrates from terraces, diversions, concentrated flow channels, or natural waterways as it runs off a field. Grassed waterways will slowdown water and



Fig. 2 Corn growing in a no till system. The crop was planted in residue from a cover crop that has been chemically killed prior to planting corn. The cover crop and its residue protect the soil until the corn becomes established.

Source: Photo courtesy of USDA, Natural Resources Conservation Service.

guide it off the field, significantly reducing gully erosion.

Erosion Control with Surface Cover

Each of the agronomic practices listed previously utilizes surface cover to control erosion. Surface cover is the single most important agronomic factor in determining soil loss. Surface cover affects erosion by water by reducing the transport capacity of runoff water, by causing deposition in ponded areas, and by decreasing the surface area susceptible to raindrop impact. It also protects the soil surface from erosive winds. Surface cover includes the live plant canopy as well as plant residue.

CROP RESIDUE MANAGEMENT WITH CONSERVATION TILLAGE

The remaining section discusses in greater detail the two most important agronomic practices for controlling erosion—crop residue management and conservation tillage. Crop residue management is an “umbrella” term encompassing several conservation tillage systems. In the 1960s, the terms *minimum tillage* and *reduced tillage* were used to refer to less extensive tillage operations during the production of agronomic crops. In the 1980s, the term *conservation tillage* was used to refer to any tillage and planting system that left at least 30% of the soil surface covered by plant residue. The term *crop residue management* has evolved to address the various methods and benefits of surface residue in reducing erosion.

Crop residue management is any tillage and planting system that uses conservation tillage (preferably no-till) or another system designed to retain all or a portion of the previous crop's residue on the soil surface.

Conservation tillage, defined earlier, is any tillage and planting system that leaves more crop residue on the soil surface after planting to reduce soil erosion. Types of conservation tillage include:

- No-till/strip-till—the soil is left undisturbed from harvest to planting except for strips up to one-third of the row width (strips may involve only residue disturbance or may include soil disturbance). Planting or drilling is accomplished using disc openers, coulters (s), row cleaners, in-row chisels, or rotary tillers. Weed control is accomplished primarily with herbicides. Cultivation may be used for emergency weed control.^[4] Other common terms used to describe no-till include direct seeding, slot planting, zero-till, row-till, and slot-till.
- Ridge-till—the soil is left undisturbed from harvest to planting except for strips up to one-third of the row width. Planting is completed on the ridge and usually involves the removal of the top of the ridge. Planting is

completed with sweeps, disk openers, coulters, or row cleaners. Residue is left on the surface between ridges. Weed control is accomplished with herbicides (frequently banded) and/or cultivation.^[4] Ridges are rebuilt during row cultivation (Fig. 3).

- Mulch-till—full-width tillage involving one or more tillage trips which disturbs all of the soil surface and is done prior to and/or during planting. Tillage tools such as chisels, field cultivators, disks, sweeps, or blades are used. Weed control is accomplished with herbicides and/or cultivation.^[4]

Table 1 shows average amounts of residue cover left after various tillage operations in corn and soybeans.^[5] The residue left after harvesting soybeans was 83%, and that left after corn was 93%. Both levels gave very good erosion control compared with bare soil. The loss of erosion control because of the reduction in residue cover is evident, especially for fragile residues like soybean residue.

Crop residue management utilizes agronomic practices for the entire cropping year. A crop rotation sequence is chosen to include high residue producing crops. A crop is planted that will provide residue to meet a specified goal, such as reducing erosion to a specified level. Planting in narrower rows and at higher plant populations will create more residue. Cover crops can be used when residue from the major crop is insufficient. When no-till is not utilized, tillage operations are delayed until near planting time instead of tilling soon after harvest. With mulch-till, tillage is performed at shallower depths, with fewer passes across the field. Tillage tools are used that bury less residue compared to conventional tillage. At harvest, residue is distributed



Fig. 3 Corn growing in a ridge till system. During planting, the ridgetop was cleared and the seed planted. Residue is left on the surface between ridges. The ridges will be rebuilt during cultivation as the crop grows.

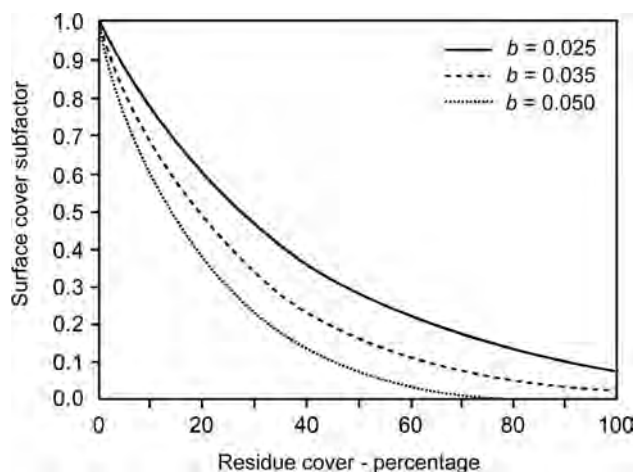
Source: Photo courtesy of USDA, Natural Resources Conservation Service.

Table 1 Percentage of soybean and corn residue cover remaining after harvest in different tillage operations in Iowa.

Crop and type of operation	Average % residue cover remaining	Erosion reduction% (compared to fall moldboard plowing)
Harvest		
Soybeans	83	
Corn	93	
Fall moldboard plowing		
Soybeans	2	
Corn	9	
Fall chisel plowing		
Soybeans	15	37
Corn	49	75
Spring disking		
Soybeans	18	43
Corn	56	81
No till		
Soybeans	58	86
Corn	86	93

uniformly over the soil surface, and baling, grazing, burning, and other operations that remove residue are eliminated.

The effectiveness of residue cover in reducing erosion varies, as shown in Fig. 4. However, as this figure illustrates, even at relatively low residue cover levels, residue cover is highly effective. In Fig. 4, residue effectiveness is expressed as a b value. Fields easily eroded by flow, such as thawing soils where rill erosion

**Fig. 4** Effect of residue cover and b values on the surface cover subfactor for smooth field surfaces in the Revised Universal Soil Loss Equation.

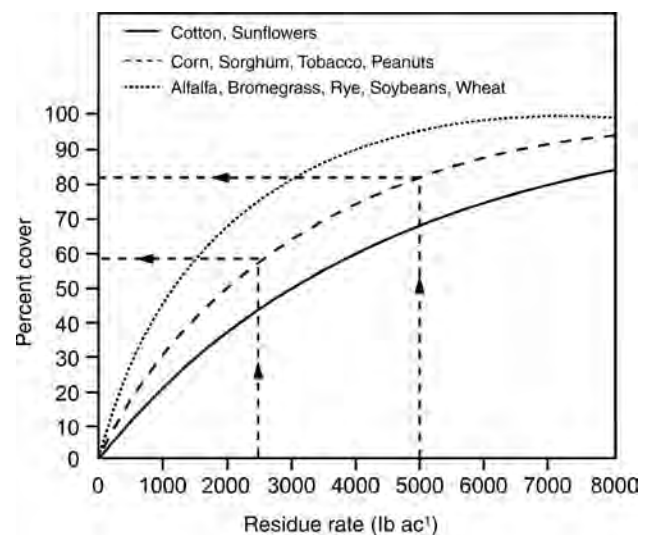
Source: From Renard, Foster, et al.^[6]

is the primary mechanism of soil loss, have a b value of 0.05. Fields with highly cohesive soils, such as with long-term no-till where interrill erosion is the primary mechanism of soil loss, have a b value of 0.025. Fields that are regularly tilled have an intermediate b value of 0.035.^[6]

Residues vary in their effectiveness, depending on their size and weight, as shown in Fig. 5.^[6] For a given weight, different kinds of residue provide varying amounts of surface cover. For example, 2690 kg/ha (2400 lb/acre) of cotton residue provides 45% surface cover. The same weight of corn provides 60% surface cover, and that weight of soybean residue provides nearly 85% cover.

Leaving crop residue on the soil surface has a number of advantages over tillage that leaves the soil surface nearly bare. The most significant advantage is the greatly reduced erosion. Other advantages include the potential for increased yield due to water conserved by surface residue, lower soil temperatures in warmer climates, improved soil quality over time due to increased organic matter levels, and reduced input of time, labor, and fuel.^[7]

Success of agronomic practices, such as crop residue management and conservation tillage to control erosion, has been proven by the increasing rate of acceptance and use by farm operators. Success is due in part to the improved effectiveness and reduced cost of herbicides. Success is also due to improvements in tillage tools that leave more residue on the surface and planting equipment that works well in high-residue conditions. Finally, farm operators measure success through economics. The reduction in the number of tillage operations, the size and

**Fig. 5** Relationship of residue amount to percent residue cover for various crops.

Source: From Renard, Foster, et al.^[6]

number of tractors, fuel used, and maintenance costs are important considerations in making the decision to use crop residue management practices.

REFERENCES

1. Agronomic Development Center. *No-Till User's Manual*; BASF Corporation: Research Triangle Park, 1990.
2. *Field Office Technical Guide, Section IV*; USDA Natural Resources Conservation Service.
3. Conservation Technology Information Center (CTIC). *Core4/Conservation Tillage*, 2000. Available at <http://www.ctic.purdue.edu/Core4/CT/Definitions.html> (accessed October 5, 2000).
4. USDA–Soil Conservation Service. *Conservation Catalog for the 1990s*; USDA–Soil Conservation Service: Des Moines, 1991.
5. Colvin, T.S.; Laflen, J.M.; Erbach, D.C. A review of residue reduction by individual tillage implements. In *Crop Production with Conservation in the 80s*; Siemens, J.C., Ed.; American Society of Agricultural Engineers Publication: St. Joseph, 1980; 7–81.
6. Renard, K.G.; Foster, G.R.; Weesies, G.A.; McCool, D.K.; Yoder, D.C. Predicting soil erosion by water: A guide to conservation planning with the revised universal soil loss equation (RUSLE). In *Agriculture Handbook*; U.S. Department of Agriculture: Washington, 1997; Vol. 703.
7. USDA–Soil Conservation Service. *Crop Residue Systems for Conservation and Profit*; USDA Soil Conservation Service: Washington, 1992.

Erosion: Assessment

John Boardman

Department of Geographical and Environmental Science, University of Cape Town, Cape Town, South Africa and Environmental Change Institute, University of Oxford, Oxford, U.K.

Abstract

Soil erosion is the loss of soil from the surface of the earth. It is a two-stage process with detachment of soil particles preceding transport by the agency of water or wind. It is a natural process occurring at relatively low rates but which may be accelerated by human actions. Erosion is driven by both socioeconomic (“ultimate factors”) and physical factors such as slope, soil, and rainfall (“proximal factors”). Most erosion is associated with the formation of rills and gullies. Erosion has serious impacts on the fertility of soils, on water quality, and on reservoir storage capacity. Methods of assessing erosion include experimental plots, sediment yield in rivers, field monitoring, remote sensing, Cesium-137, historical analysis, expert opinion, and modeling. Combinations of these have often been used. Assessment of actual and potential erosion is necessary so that measures can be taken to prevent the loss of soil and limit the off-site impacts of erosion. A range of conservation measures have been used with varying degrees of success.

INTRODUCTION

Soil erosion is the loss of soil from the surface of the earth (Fig. 1). It is a two-stage process with detachment of soil particles preceding transport by the agency of water or wind. It is a natural process occurring at relatively low rates, which may be accelerated by human actions. Most erosion is associated with the formation of rills and gullies. Erosion has serious impacts on the fertility of soils, on water quality, and on reservoir storage capacity. Assessment of actual and potential erosion risk is necessary so that measures can be taken to prevent the loss of soil.

Agents of erosion are wind, water, and gravity (causing mass movement). Movement of people and animals may contribute, and soils will be more susceptible if weakened by weathering (wetting and drying, frost, etc.).

Erosion is an important contributor to global soil degradation. It is also a major component of desertification. Under semiarid conditions, vegetation damage typically precedes erosion. If climate variability is a major influence, then degradation may be cyclic, interspersed with periods of improved land quality. Frequently, because of population pressure, degraded land does not recover.

The major impact of soil erosion is on agricultural land, both arable and grazing. Other areas of concern are construction sites and protected natural areas (e.g., footpath erosion in National Parks).

This entry is mainly concerned with erosion by water that globally appears to affect twice the area of wind erosion.^[1] Overgrazing as a cause of erosion is very widespread but is difficult to classify as it often leads to erosion by water and, in some areas, by wind.

The causes of erosion may be regarded as: 1) proximal and 2) ultimate. Proximal causes relate to physical factors such as rainfall, runoff, wind velocity, gradient, and soil type. Ultimate causes relate to socioeconomic conditions that influence farmer decision making and therefore land-use and farming practices. Important farmer decisions concern what crops to plant, where to plant them, and how to plant them. Such decisions are strongly influenced by farmer attitudes and traditions, community views and, especially in the developed world, by government support for agriculture. Some such support may be regarded as a “perverse subsidy” in that it leads to unwanted and unforeseen environmental impacts including erosion and pollution; examples include European Union support for almonds in Spain,^[2] land leveling in Norway,^[3] and olive oil production in Spain.^[4] A major socioeconomic impact on eastern European landscapes was the collectivization of agriculture under Communism.^[5,6]

The relationship between population growth, food production and, exploitation of the soil is suggested by Montgomery (p. 84):^[7]

A fundamental model of agricultural development in which prosperity increases the capacity of the land to support people, allowing the population to expand to use the available land. Then, having eroded soils from marginal land, the population contracts rapidly before soil rebuilds in a period of low population density.

Major texts on erosion, its causes and consequences include those by Bennett,^[8] Blaikie,^[9] Morgan,^[10] Boardman and Poesen,^[11] and Montgomery.^[7]



Fig. 1 Severely eroded winter wheat fields, South Downs, United Kingdom, October 1987.

SCALE AND EXTENT OF EROSION

Rates of erosion on grassed or wooded areas are usually below 1 t/ha/yr. Rates of soil formation are very low—similar to natural rates of erosion. The oft-quoted USDA figure is 1 inch per 500 years which is about 0.6t/ha/yr.^[7] Erosion in excess of this figure means the loss of, or decline in, a finite resource. Erosion rates on farmland in temperate areas occasionally exceed 100 t/ha/yr. On the Loess Plateau of China, 7.2 Mha was reported to have erosion rates in excess of 100 t/ha/yr.^[12]

Hotspots of global erosion are difficult to define and rank due to lack of or unreliable data. However, we may suggest the following:^[13]

1. Loess plateau, the Yangtze basin, and the southern hill country of China
2. Ethiopia
3. Swaziland and Lesotho
4. The Andes
5. South and East Asia
6. Iceland
7. Madagascar
8. The Himalayas
9. The West African Sahel
10. Caribbean and Central America

There is very limited data for some of these areas, particularly the Andes and Madagascar. If areas that have been severely affected in the past and to some extent are more vulnerable are to be included, then the list would be extended:

1. The Dust Bowl in the 1930s in the United States (parts of Colorado, Kansas, New Mexico, Texas, and Oklahoma)^[14] (Fig. 2)
2. The Virgin Lands Project of the former Soviet Union^[15]
3. The Mediterranean basin^[16]



Fig. 2 Dust storm approaching Stratford, Texas Dust Bowl surveying in Texas, April 18, 1935.

Source: Image ID: theb1365, Historic C&GS Collection, NOAA George E. Marsh Album.

IMPACT OF EROSION

The following are the impacts of erosion:

1. On-site, which includes loss of crop, seeds, and fertilizer and thinning of the soil layer thus directly affecting present and future productivity. The amount of yield reduction is disputed with work suggesting a figure of 4% per 10 cm of soil loss.^[17] In extreme cases, land may become unworkable or unprofitable because of soil loss and nutrient depletion, as in the case of badlands (Fig. 3). Montgomery discusses many cases where land is abandoned and farmers move on, e.g., 18th century tobacco growing in Virginia, Georgia, and the Carolinas or the Amazon lowlands (p. 118).^[7] This process may result in exploitation of less suitable land or land at greater risk of erosion e.g., Rome (p. 62).^[7]
2. Off-site, where people, water/air, or property not directly responsible for erosion are affected. Runoff from agricultural land generally transports sediment



Fig. 3 Badlands, northern Spain.

particles and attached phosphorous and pesticides. Impacts include dam sedimentation,^[18] pollution of water courses,^[19] and property damage by muddy floods.^[20]

Costs of erosion are underresearched. This is partly because it is difficult to put a value on soil. As soils thin crop yields decline or they are artificially enhanced by the use of fertilizers. This is a cost which less developed societies cannot afford. Serious soil depletion results in abandonment and migration (see above). Costs resulting from off-site impacts are easier to estimate; e.g., water cleaning costs in England and Wales are estimated at several hundred million pounds per year for the removal of nitrates, phosphates, and pesticides, some of which are associated with runoff and erosion events.^[21] The costs of muddy floods in central Belgium range from 16 to 172 million euros per year.^[22] Fine-particle sediments (mainly of agricultural origin) affect fish spawning in gravel-bedded rivers.^[23,24] Sedimentation of ports is recorded from the Mediterranean and from the eastern states of the United States (p. 139).^[7]

Impacts may also be seen as either immediate or long term. The former includes the formation of gullies, the burial of crops, and the flooding of houses, and the latter refers to the gradual thinning of soils or sedimentation of reservoirs. Unfortunately the long-term effects are easy to overlook. Societies in the Middle East, Rome, Greece, MesoAmerica, etc. have all been seriously affected by soil loss.^[7]

EROSION ASSESSMENT

Erosion assessment may describe the “actual” risk of erosion: the Global Assessment of Soil Degradation project (GLASOD) is an example of this type of assessment applied globally.^[1] Similarly, Canada has published maps of the risk of wind and water erosion at the provincial level.^[25] New Zealand has combined, on single maps at 1:250,000 scale, the actual and “potential” risk of erosion.^[26] Potential in this case refers to a change of land use to pastoral or arable. Maps from the CORINE project show actual and potential soil erosion in Southern Europe, and the latter is defined as the worst possible situation that might be reached.^[27]

Erosion assessment may focus on changes through time in the same area. Trimble^[28] has measured and modeled changes in erosion and deposition rates in the Coon Creek catchment in Wisconsin since the beginning of European settler farming in the 1850s. This study relates erosion to land use change, management, and conservation techniques, as well as to the capacity of rivers to transport eroded materials. It therefore links on-site and downstream effects through time and space. Erosion rates since Neolithic clearance have been modeled on the loessic soils of the South Downs, United Kingdom. This

assessment is based on changing land use, climate, soils, and farming practices (Fig. 4).^[29] Assessment of erosion and its impacts may be combined in complex scenarios of change involving population pressure, e.g., Hurni for Ethiopia.^[30]

Modeling constitutes an important approach to erosion assessment. This is for several reasons: many areas lack the data that allow direct estimates of erosion; modeling may be based on scenarios of future land use or climate and therefore estimates may be made of erosion under assumed conditions; and application of a model may be far less expensive and time consuming than field surveys.

The spatial scale of modeling may vary from a few square meter to thousands of square kilometer. The global change and terrestrial ecosystems model evaluation exercise divides models into field, catchment, and landscape, based on spatial scale.^[31] Resolution of erosion models is determined by grid size, i.e., km² grid size used in CORINE for Southern Europe and the 250-m grid used in the maps of seasonal erosion risk in France.

Assessments are frequently expressed in map form.^[32] Landscape features indicative of erosion may be mapped as a record of an actual event or series of events. In arable landscapes, features may be temporary; thus rills, ephemeral gullies, and fans are usually plowed out annually, e.g., erosional features produced as a result of a snowmelt event in May 1993 in Slovakia.^[5] In semiarid, grazed landscapes erosional features may become permanent, e.g., gullies and badlands. Assessments should distinguish features active at the present time from those that are inactive and represent erosion under former conditions.

Several methods of erosion assessment are commonly used and Table 1^[33] and will be reviewed briefly.

1. Small experimental plots in the laboratory or in the field, and these may operate with natural or simulated rainfall^[34] (Figs. 5 and 6). They are assumed to represent larger parts of the landscape. Results are frequently extrapolated to fields or catchments and are used as input to models. Extrapolation from small plots to regional, national, or continental size areas is not acceptable.^[35]
2. Sediment yield of rivers represents the net loss of soil from a catchment.^[36] Erosion rates on fields may be very different owing to storage of soils between the field and the river. Highest rates of sediment yields from a large river are from the Yellow River, China, with an average silt content of 38 kg m⁻³.^[12] Rivers of east and south Asia deliver about 67% of the sediment reaching the world's oceans, and rates are highest in this region owing to tectonic uplift and steep slopes, high rainfall, deforestation, and population pressure resulting in

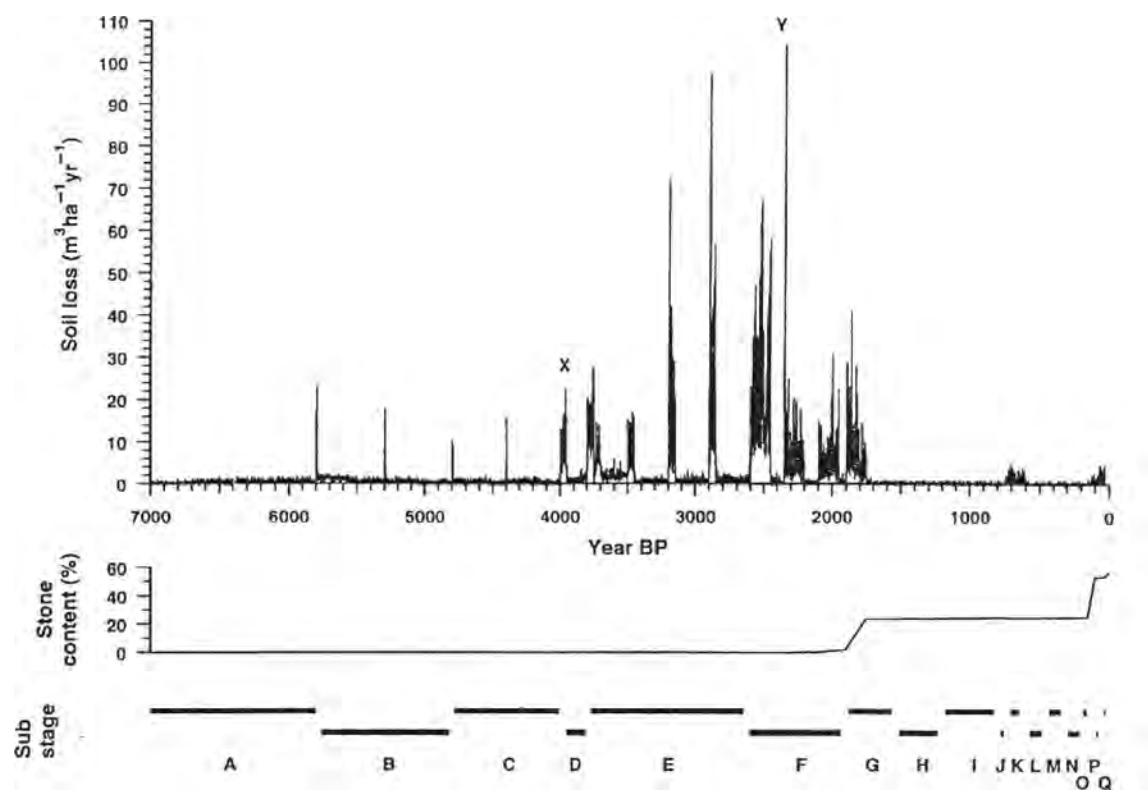


Fig. 4 Simulated annual erosion rates and surface stone content for a “thin” soil profile (1.22 m depth to chalk) on the South Downs, Southern England.
Source: Adapted from Favis-Mortlock, Boardman, et al.^[29]

intensive farming activities.^[37] Alternatively, the sediment yield from river basins may be estimated by measurement of the volume of sediment trapped in a still water body such as a lake, reservoir, or pond. A date for the construction of a reservoir are often known. Sediment including bedload will be

trapped. Losses in water passing through the dam may be estimated (trap efficiency). The advantage of this method is the possibility of medium-term records of >100 years.^[38] Sediment yield to three small dams in the Karoo, South Africa, is estimated at 115, 357, and 654 t/km²/yr for a period of

Table 1 Assessment methods: their merits and limitations.

Assessment method	Merits	Limitations
Experimental plots in laboratory or field	Control over erosion factors; ease of measurement of runoff and soil	Small size; difficult to extrapolate to larger areas
Sediment yield of rivers	Ease of measurement; may cover >100 years if reservoir sediments used	Measures total soil lost from watershed, not from specific areas
Field monitoring	Records soil lost from rills and gullies; covers large areas; inexpensive	Gives very approximate amounts of soil lost
Remote sensing	Covers large areas and may cover long time-span	Limited availability for many areas; may not be suitable for small scale features
Cesium-137	Gives 55-year average	Many unknowns; unreliable without careful calibration; expensive and time-consuming
Stratigraphy and pedology	Deposited soil may contain artifacts; suitable for long time period studies	Scarcity of suitable sites; low precision of some dating methods
Expert opinion	May reveal unrecorded information	Subjectivity: difficult to compare different areas
Models	Objectivity; ease of use even by “non-experts”; numerical output	Availability of data; complexity of process descriptions; frequent lack of validation



Fig. 5 Experimental plot (flume), University of Toronto.

65 years. These rates are adjusted to take into account trap efficiency, and the contrasts are due to differences in land use within the catchments.^[39]

Similar approaches have been used in Australia^[40] and in Ethiopia.^[41]

3. Field monitoring is defined as “field-based measurement of erosional and/or depositional forms over a significant area (e.g., >10 km²) and for a period of >2 years.”^[42] Regular measurements of volumes of soil loss from rills and gullies or deposited in fans are made. Notable examples are the monitoring of 17 localities in England and Wales during 1982–1986,^[43] a 10-year program on the South Downs, England during 1982–1991,^[44] monitoring of ephemeral gullies in Belgium during 1982–1993,^[45] and a long-term monitoring scheme in the Swiss midlands.^[46,47]
4. Simple techniques of measurement and assessment of erosion suitable for Less Developed Countries include pedestal height, depth of armor layer, plant/tree root and fence post exposure, tree mounds, and build up of sediment behind barriers and in drains.^[48] Many



Fig. 6 Experimental plots, South-Limbourg, the Netherlands.



Fig. 7 Erosion pin site in the Karoo, South Africa.

studies have used erosion pins to assess rates of erosion over time (Fig. 7). These are particularly suited to bare degraded land such as badlands. Rates may be related to topographic position, presence of vegetation, weathering, and rainfall/wind factors^[49] (Fig. 8).

5. Remote sensing is used increasingly in erosion assessment. For rill and gully erosion, it may be used as a locational technique to allow detailed field measurements to be made at selected sites. Gullies and badlands may be identified on black and white air photographs of 1:20,000 scale, and in the Karoo of South Africa, comparisons have been made between the extent of gullying in 1945 and 1980.^[50] Also in South Africa, in a pioneering study, Talbot mapped gullying in the Swartberg resulting from land conversion to wheat farming^[51] (Fig. 9). The extent of Icelandic land degradation has been assessed using Landsat imagery.^[52]
6. Cesium-137, derived from weapons testing since 1954, is used as a tracer to assess amounts of erosion and deposition. Annual average rates for the last ca. 55 years are estimated by measurement of the amount and distribution of cesium in comparison with an uneroded reference site.^[53] Studies have been carried out in Australia, Canada, China, the Netherlands, Poland, Thailand, United Kingdom, and United States. However, results are controversial with particular issues regarding the original assumed regular distribution of cesium across the landscape via the assumed homogenous spatial distribution of rainfall, and the assumption that cesium “sticks” to fine soil particles in a predictable way. These assumptions have been strongly challenged, and erosion rates obtained using this technique appear unreliable.^[54]
7. Total amounts of soil loss since a historical baseline such as woodland clearance or the beginning of European settlement have been assessed by comparison of existing soil profiles with uneroded ones. Evans estimates historical losses of topsoil of 150–250 mm in

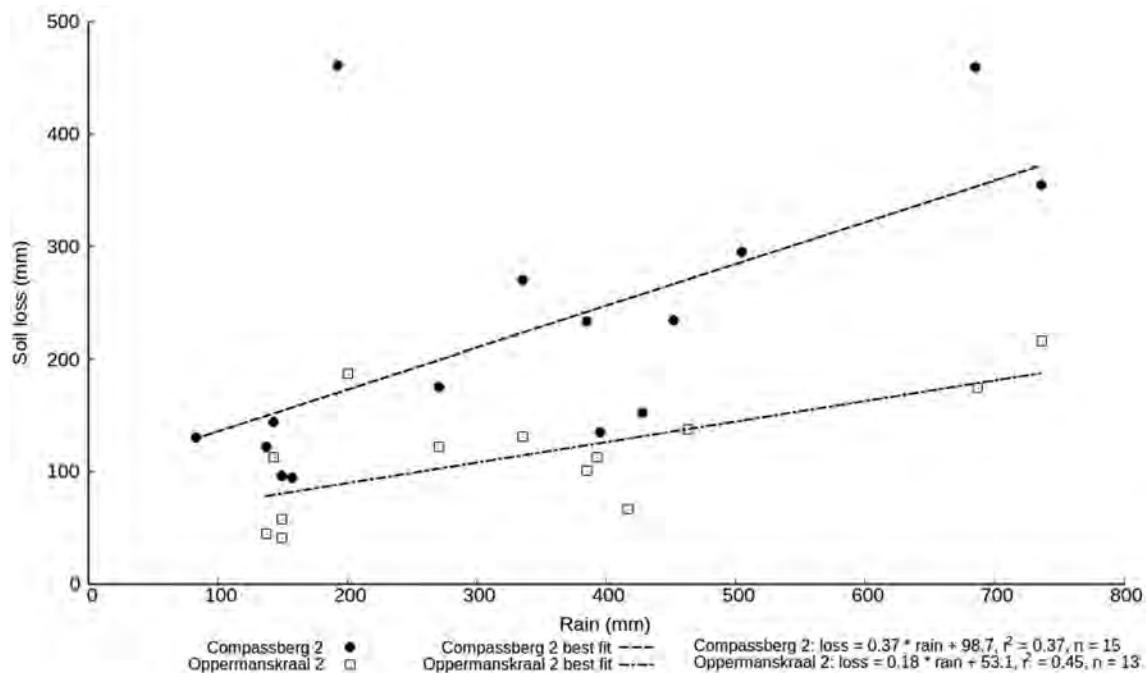


Fig. 8 Average rates of erosion for eroding pins at two sites in the Karoo, South Africa, plotted against rainfall amount for each measurement period.

lowland England and Wales, much of which is stored as alluvium and colluvium.^[55] Sedimentation above marker horizons in reservoirs may be used to compute sediment yield for periods of time.^[39] Pedological and stratigraphic approaches have also been used to



Fig. 9 Severely gullied hill country ca. 8 km northwest of Durbanville, South Africa. Gullies are on ploughed land. Photography March 1938; the land had been abandoned by 1944.

Source: Adapted from Talbot.^[51]

assess the effect of past extreme events, e.g., the major storms and floods in Germany in the early 14th century.^[56]

8. Expert opinion has been used to assess erosion on a regional or global scale. GLASOD assesses erosion at a global scale (10 M).^[1] A total of 12 degradation types are recognized under the broad headings of water erosion, wind erosion, chemical deterioration, and physical deterioration. The “degree” of degradation is assessed for each mapping unit, as is its “relative extent.” A combination of degree and relative extent gives an assessment of “severity” which is expressed in “severity classes” and mapped in four colors. However, there is evidence that GLASOD is a very imperfect predictive tool. Van-Camp et al.^[57] are particularly critical and suggest that the GLASOD approach should be abandoned: comparison with maps and data for Spain show that assessment of water erosion was poor.^[58] GLASOD has also been assessed by other authors.^[59] At a more local and detailed level, expert opinion appears more reliable, e.g., a South African assessment.^[60]
9. Assessment of erosion has frequently been based on the application of models. The Universal Soil Loss Equation (USLE) is most widely used because of its simplicity and low-level data requirements.^[61] Further developments in modeling have concentrated on simulation of erosional processes and computerization. Models may be used to estimate average, long-term erosion rates under specified conditions (USLE), or to assess the effect of a particular rainfall/runoff event,

e.g., EUROSEM.^[62] Erosion models have also been used to predict wind erosion, and to assess chemical losses in solution, or attached to soil particles (for reviews, see Morgan).^[63]

CONTROLLING SOIL EROSION AND OFF-SITE IMPACTS

Controlling soil erosion is based on two principles, protecting the soil from runoff or wind by providing vegetative protection, or reducing the slope and thereby decreasing water velocity. Erosion generally takes place on bare soil (Fig. 1), and therefore the protection of a crop or crop debris at vulnerable times of the year is important. Most radical solutions such as minimum tillage and conservation tillage aim to have some degree of vegetation cover throughout the year, and therefore drill the new crop through the remnants of the old thus avoiding the bare ground inherent in tillage systems involving the plough.

Large areas of the same crop give rise to the risk of runoff at the same time. Thus the breakup of large fields into strips of different crops can reduce the risk. Valley-bottom zones of concentrated flow may be grassed (grass waterways) to prevent erosion. Buffer strips of grass are widely used around arable fields in order to reduce runoff volumes and loss of soil from fields. Evans and Boardman show how return to grass of a small area interrupts valley-bottom runoff from large areas of winter cereals thus preventing flooding of houses.^[64]

The desire to reduce slope gradients has traditionally been addressed by the construction of terraces. An added advantage is that a terrace allows for cultivation of slopes that were too steep for agriculture. Terrace systems are known from the Middle Bronze Age on Crete^[65] and from the Incas of Peru. Many systems around the Mediterranean are abandoned and erosion associated with their post-abandonment state has been much researched.^[66] Maintenance of terraces is therefore vital if they are to prevent erosion (Fig. 10).

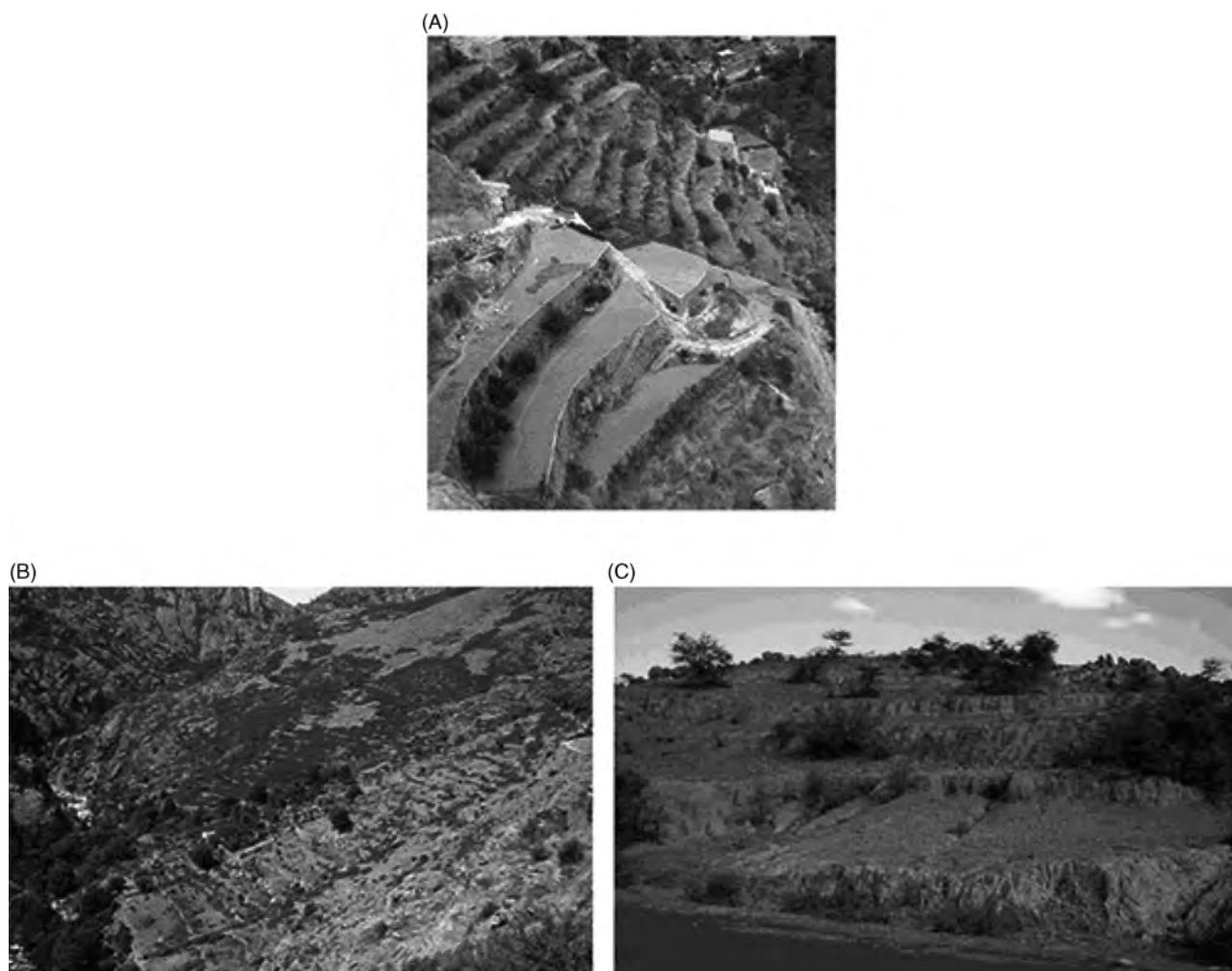


Fig. 10 (A) Old abandoned terraces and rebuilt terraces, Mallorca, Spain; (B) abandoned terraces, central Corsica, France; and (C) abandoned and degraded terraces, Eritrea.



Fig. 11 Mitigation measures: grass buffers along field edges, the Melsterbeek catchment, Flanders, Belgium (Karel Vandaele, Samenwerking Land & Water).

It is doubtful if any one conservation technique will be effective on its own. Most successful schemes involve several approaches. For example, the Melsterbeek Catchment soil and water conservation scheme in Flanders use grass waterways, buffer strips, and retention ponds (Fig. 11). Successful schemes, such as this, must involve the local population of land owners and managers which generally requires financial support and technical assistance.^[67]

In the Palouse wheat growing area of Washington State, high erosion rates have persisted for over a 100 years and are particularly associated with the practice of summer fallow. The costs of reducing erosion by conservation practices are well illustrated:

Applying various measures of conservation treatment to the land will affect the economy of the basin. But erosion rates can be reduced by 40% in the low and high precipitation zones and 60% in the intermediate precipitation zone—without decreasing farm income. The erosion rate can be reduced 80% through maximum levels of conservation practices and retirement of 35,000 acres of steep, erodible land. Achieving this level of reduction would cost more than \$29 million in reduced productivity and increased operating costs.^[68]

Soil erosion is often best addressed in a wide framework of natural resource improvements. The issue of sustaining the livelihoods of the farmers as well as introducing conservation measures is critical. Examples of successful approaches from West Africa, New Zealand, Southern Brazil, northwest India, northern Cameroon, and Kenya are discussed by El Swaify.^[69] There is also the well-known case of the Machakos District, Kenya.^[70] Hudson discusses reasons for failure of soil conservation projects.^[71]

Previously quoted extremely high erosion rates for the Loess Plateau, China, and associated high sediment yields

for the Yellow River, appear to have been substantially reduced. Water–soil conservation practices are credited with 40% reduction in sediment loads, sediment trapping by major reservoirs with 30% and most of the remaining is due to precipitation decrease.^[72]

CONCLUSION

The assessment of erosion is necessary in order to understand the scale and extent of the problem. Many assessment methods have been used including experimental plots, sediment yield in rivers, field monitoring, remote sensing, Cesium-137, historical analysis, expert opinion, and modeling. All have advantages and disadvantages and may not be appropriate for particular circumstances. Assessment will include an evaluation of the type of erosion processes and their relative importance. The design of mitigation measures will depend on the outcome of the assessment. Mitigation measures must take into account the driving forces of erosion particularly its socioeconomic context and the need to support local peoples in recommending appropriate solutions.

REFERENCES

1. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G., Eds. *World Map of the Status of Human-Induced Soil Degradation: An Explanatory Note*; ISCRIC: Wageningen and UNEP: Nairobi, 1990; 1–32.
2. Faulkner, H. Gully erosion associated with the expansion of terraced almond cultivation in the coastal Sierra de Lujar, S. Spain. *Land Degrad. Rehabil.* **1995**, *6* (1), 179–200.
3. Lundekvam, H.E.; Romstad, E.; Oygarden, L. Agricultural policies in Norway and effects on soil erosion. *Environ. Sci. Policy* **2003**, *6* (1), 57–67.
4. de Graaff, J.; Eppink, L.A.A.J. Olive oil production and soil conservation in southern Spain, in relation to EU subsidy policies. *Land Use Policy* **1999**, *16*, 259–267.
5. Stankoviansky, M. Geomorphic effect of surface runoff in the Myjava Hills, Slovakia. *Ziatschrift fur Geomorphologie* **1997**, *110*, 207–217.
6. Boardman, J.; Poesen, J.; Evans, R. Socio-economic factors in soil erosion and conservation. *Environ. Sci. Policy* **2003**, *6* (1), 1–6.
7. Montgomery, D.R. *Dirt: The Erosion of Civilizations*; University of California Press: Berkeley, 2007.
8. Bennett, H.H. *Soil Conservation*; McGraw Hill: New York, 1939.
9. Blaikie, P. *The Political Economy of Soil Erosion in Developing Countries*; Longman: London, 1985.
10. Morgan, R.P.C. *Soil Erosion and Conservation*, 3rd Ed.; Blackwell: Oxford, 2005.
11. Boardman, J.; Poesen, J., Eds. *Soil Erosion in Europe*; John Wiley: Chichester, 2006.
12. Wen, D. Soil erosion and conservation in China. In *World Soil Erosion and Conservation*; Pimentel, D., Ed.; Cambridge University Press: Cambridge, 1993; 63–85.

13. Boardman, J. Soil erosion science: Reflections on the limitations of current approaches. *Catena* **2006**, 68, 73–86.
14. Worster, D. *Dust Bowl: The Southern Plains in the 1930s*; Oxford University Press: New York, 1979.
15. McCauley, M. *Khrushchev and the Development of Soviet Agriculture*; The Virgin Lands Program 1953–1964. Holmes & Meier Publishers, Inc.: New York, 1976.
16. Faulkner, H.; Hill, A. Forests, soils and the threat of desertification. In *The Mediterranean: Environment and Society*; King, R., Proudfoot, L., Smith, B., Eds.; Arnold: London, 1997; 252–272.
17. Bakker, M.M.; Govers, G.; Rounsevell, M.D.A. The crop productivity-erosion relationship: An analysis based on experimental work. *Catena* **2004**, 57, 55–76.
18. Renwick, W.H. Continental-scale reservoir sedimentation patterns in the United States. In *Erosion and Sediment Yield: Global and Regional Perspectives*, Proceedings of the Exeter Symposium, July 1996; Walling, D.E., Webb, B.W., Eds.; IAHS Publication No. 236; IAHS: Wallingford, 1996; 513–522.
19. Evans, R. Pesticide run off into English rivers – A big problem for farmers. *Pestic. News* **2009**, 85, 12–15.
20. Boardman, J.; Verstraeten, G.; Bielders, C. Muddy floods. In *Soil Erosion in Europe*; Boardman, J., Poesen, J., Eds.; John Wiley: Chichester, 2006; 743–755.
21. Evans, R. *Soil Erosion and Land Use: Towards a Sustainable Policy*; Cambridge Environmental Initiative, Professional Seminar Series No. 7; White Horse Press: Cambridge, 1995; 14–26.
22. Evrard, O.; Bielders, C.; Vandaele, K.; van Wesemael, B. Effectiveness of erosion mitigation measures to prevent muddy floods in central Belgium, off-site impacts and potential control measures. *Catena* **2007**, 70, 443–454.
23. Theurer, F.D.; Harrod, T.R.; Theurer, M. *Sedimentation and Salmonids in England and Wales*; Technical Report; Environment Agency: England and Wales, 1998.
24. Birt, K. Runoff from potato farms blamed for fish kills on Canadian Island. *J. Soil Water Conserv.* **2007**, 62 (6), 136A.
25. Manitoba. *Water Erosion Risk Map 1:1,000,000, Agriculture Canada*; Publication 5259/B; Ottawa, 1988.
26. Prickett, R.C. Sheet 23 Oamaru: Erosion Map of New Zealand, 1:250,000. In *National Water and Soil Organisation*; Wellington: New Zealand, 1984.
27. CORINE. *Soil Erosion Risk and Important Land Resources in the Southern Regions of the European Community*; EUR 1323; Luxembourg, 1992.
28. Trimble, S.W. Decreased rates of alluvial sediment storage in the Coon Creek Basin, Wisconsin, 1975–1993. *Science* **1999**, 285, 1244–1246.
29. Favis-Mortlock, D.T.; Boardman, J.; Bell, M. Modelling long-term anthropogenic erosion of a loess cover, South Downs, UK. *Holocene* **1997**, 7 (1), 79–89.
30. Humi, H. Land degradation, famine, and land resource scenarios in Ethiopia. In *World Soil Erosion and Conservation*; Pimentel, D., Ed.; Cambridge University Press: Cambridge, 1993; 27–61.
31. Boardman, J.; Favis-Mortlock, D.T., Eds. *Modelling Soil Erosion by Water*; NATO ASI Series; Springer: Berlin, 1998; 3–6.
32. Morgan, R.P.C. Erosion hazard assessment. In *Soil Erosion and Conservation*, 3rd Ed.; Blackwell: Oxford, 2005.
33. Boardman, J. Soil erosion: The challenge of assessing variation through space and time. In *Geomorphological Variations*; Goudie, A.S., Kalvoda, J., Eds.; Nakladatelsti P3K: Prague, 2007; 205–220.
34. Mutchler, C.K.; Murphree, C.E.; McGregor, K.C. Laboratory and field plots for erosion studies. In *Soil Erosion Research Methods*; Lal, R., Ed.; Soil and Water Conservation Society: Ankeny, 1988; 9–36.
35. Boardman, J. An average soil erosion rate for Europe: Myth or reality? *J. Soil Water Conserv.* **1998**, 53 (1), 46–50.
36. Walling, D.E. Measuring sediment yield from river basins. In *Soil Erosion Research Methods*; Lal, R., Ed.; Soil and Water Conservation Society: Ankeny, 1988; 39–73.
37. Milliman, J.D.; Meade, R.H. World-Wide delivery of river sediment to the oceans. *J. Geol.* **1983**, 91, 1–21.
38. Verstraeten, G.; Bazzoffi, P.; Lajczak, A.; Radoane, M.; Rey, F.; Poesen, J.; de Vente, J. Reservoir and pond sedimentation in Europe. In *Soil Erosion in Europe*; Boardman, J., Poesen, J., Eds.; John Wiley: Chichester, 2006; 759–774.
39. Boardman, J.; Foster, I.; Rowntree, K.; Mighall, T.; Gates, J. Environmental stress and landscape recovery in a semi-arid area, the Karoo, South Africa. *Scott. Geogr. J.* **2010**, 126 (2), 64–75.
40. Erskine, W.D.; Mahmoudzadeh, A.; Myers, C. Land use effects on sediment yields and soil loss rates in small basins of Triassic sandstone near Sydney, NSW, Australia. *Catena* **2002**, 49 (4), 271–287.
41. Haregeweyn, N.; Poesen, J.; Nyssen, J.; De Wit, J.; Haile, M.; Govers, G.; Deckers, S. Reservoirs in Tigray (northern Ethiopia): Characteristics and sediment deposition problems. *Land Degrad. Dev.* **2006**, 17, 211–230.
42. Boardman, J. Soil erosion by water: Problems and prospects for research. In *Advances in Hillslope Processes*; Anderson, M.G., Brooks, S.M., Eds.; Wiley: Chichester, 1996; Vol. 1, 489–505.
43. Evans, R. Extent, frequency and rates of rilling of arable land in localities in England and Wales. In *Farm Land Erosion: In Temperate Plains Environment and Hills*; Wicherek, S., Ed.; Elsevier: Amsterdam, 1993; 177–190.
44. Boardman, J. Soil erosion and flooding on the eastern South Downs, southern England, 1976–2001. *Trans. Inst. Br. Geogr.* **2003**, 28, 176–196.
45. Poesen, J. Gully typology and gully control measures in the European loess belt. In *Farm Land Erosion: In Temperate Plains Environment and Hills*; Wicherek, S., Ed.; Elsevier: Amsterdam, 1993; 221–239.
46. Prasuhn, V. Soil erosion in the Swiss midlands: results of a 10-year field survey. *Geomorphology* **2011**, 126, 32–41.
47. Prasuhn, V. *On-farm Effects of Tillage and Crops on Soil Erosion Measured over 10 Years in Switzerland*; Elsevier/International Soil Tillage Research Organization (ISTRO): 2012; Vol. 120, 137–146.
48. Stocking, M.A.; Murnaghan, N. *Handbook for the Field Assessment of Land Degradation*; Earthscan: London, 2001.
49. Haigh, M.J. The use of erosion pins in the study of slope evolution. *British Geomorphol. Res. Group., Tech. Bull.* **1977**, 18, 31–49.
50. Keay-Bright, J.; Boardman, J. Evidence from field based studies of rates of erosion on degraded land in the

- central Karoo, South Africa. *Geomorphology* **2009**, *103*, 455–465.
51. Talbot, W.J. *Swartland and Sandveld*; Oxford University Press: Cape Town, 1947.
 52. Arnalds, O.; Thorarinsdottir, E.F.; Metusalemsson, S.; Jons-son, A.; Gretarsson, E.; Arnason, A. *Soil Erosion in Iceland*; Soil Conservation Service and the Agricultural Research Institute: Iceland, 2001.
 53. Quine, T.A.; Walling, D.E. Assessing recent rates of soil loss from areas of arable cultivation in the UK. In *Farm Land Erosion: In Temperate Plains Environment and Hills*; Wicherek, S., Ed.; Elsevier: Amsterdam, 1993; 357–371.
 54. Parsons, A.J.; Foster, I.D.L. What can we learn about soil erosion from the use of ^{137}Cs ? *Earth-Sci. Rev.* **2011**, *108*, 101–113.
 55. Evans, R. Soil erosion: Its impact on the English and Welsh landscape since woodland clearance. In *Soil Erosion on Agricultural Land*; Boardman, J., Foster, I.D.L., Dearing, J.A., Eds.; Wiley: Chichester, 1990; 231–254.
 56. Bork, H.-R. Soil erosion during the past millennium in Central Europe and its significance within the geomorphodynamics of the Holocene. *Catena* **1989**, *15*, 121–131.
 57. Van-Camp, L.; Bujarrabal, B.; Gentile, A.-R.; Jones, R.J.A.; Montarnarella, L.; Olazabal, C.; Selvaradjou, S.-K. *Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection*. EUR 21319 EN/2; Office for Official Publications of the European Communities: Luxembourg, 2004.
 58. Sanchez, J.; Recatala, L.; Colomer, J.C.; Ano, C. Assessment of soil erosion at national level: A comparative analysis for Spain using several existing maps. In *Ecosystems and Sustainable Development III*; Villacampa, Y., Brevia, C.A., Uso, J.L., Eds.; Advances in Ecological Sciences 10, WITT Press: Southampton, 2001; 249–258.
 59. Sonneveld, B.G.J.S.; Dent, D.L. How good is GLASOD? *J. Environ. Manag.* **2009**, *90*, 274–283.
 60. Hoffman, T.; Ashwell, A. *Nature Divided: Land Degradation in South Africa*; University of Cape Town Press: Cape Town, 2001.
 61. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses, Agricultural Research Service Handbook 537*; U.S. Department of Agriculture: Washington, 1978.
 62. Morgan, R.P.C.; Quinton, J.N.; Smith, R.E.; Govers, G.; Poesen, J.W.A.; Auerswald, K.; Chischi, G.; Torri, D.; Styczen, M.E. The European Soil Erosion Model (EUROSEM): A dynamic approach for predicting sediment transport from fields and small catchments. *Earth Surf. Proc. Land.* **1998**, *23*, 527–544.
 63. Morgan, R.P.C. Modelling soil erosion. In *Soil Erosion and Conservation*, 3rd Ed.; Blackwell: Oxford, 2005; 116–151.
 64. Evans, R.; Boardman, J. The curtailment of muddy floods in the Sompting catchment, South Downs, West Sussex, southern England. *Soil Use Manage.* **2003**, *19*, 223–231.
 65. Grove, A.T.; Rackham, O. Cultivation terraces. In *The Nature of Mediterranean Europe: An Ecological History*; Yale University Press: New Haven, 2001.
 66. Lesschen, J.P.; Cammeraat, L.H.; Nieman, T. Erosion and terrace failure due to agricultural land abandonment in a semi-arid environment. *Earth Surf. Proc. Land.* **2008**, *33*, 1574–1584.
 67. Boardman, J.; Vandaele, K. Soil erosion, muddy floods and the need for institutional memory. *Area* **2010**, *42*, 502–513.
 68. USDA. *Erosion in the Palouse: A Summary of the Palouse River Basin Study*; United States Department of Agriculture, Soil Conservation Service, Forest Service, Economics, Statistics, and Cooperative Service: Washington, 1979.
 69. El-Swaify, S.A. with contributors. *Sustaining the Global Farm – Strategic Issues, Principles, and Approaches*. International Soil Conservation Organisation (ISCO) and the Department of Agronomy and Soil Science; University of Hawaii at Manoa: Honolulu, 1999.
 70. Tiffin, M.; Mortimore, M.; Gichuki, F. *More People, Less Erosion: Environmental Recovery in Kenya*; Wiley: Chichester, 1994.
 71. Hudson, N.W. *A Study of the Reasons for Success or Failure of Soil Conservation Projects*, *FAO Soil Bull.*; FAO: Rome, 1991; 1–64.
 72. Peng, P.; Chen, S.; Dong, P. Temporal variation of sediment load in the Yellow River basin, China, and its impacts on the lower reaches and the river delta. *Catena* **2010**, *83*, 135–147.

Erosion: Carbon Dioxide

Pierre A. Jacinthe

Indiana University—Purdue University Indianapolis, Indianapolis, Indiana, U.S.A.

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Sizable amounts of carbon dioxide can be produced as a result of soil organic carbon mobilization by water erosion. Water erosion could elicit profound disturbances in the plant–soil–atmosphere segment of the global carbon cycle through a cascading process involving nutrient depletion, decreased biomass production, and the alteration of radiative and thermal properties of soils.

INTRODUCTION

The projected warming of global climate and the degradation of land resources by erosion are among the most pressing environmental challenges of the modern era. The warming trend in the world's climate is linked to the accumulation of radiatively active gases [carbon dioxide (CO_2), nitrous oxide, and methane (CH_4)] in the atmosphere.^[1] Atmospheric CO_2 concentration has been increasing at a rate of 3.2 Pg C yr^{-1} ($1 \text{ Pg} = 10^{15} \text{ g}$),^[2] which, if maintained, could profoundly alter the chemical composition and the energy balance of the earth's atmosphere. Although fossil fuel burning is the primary contributor to the atmospheric CO_2 buildup, it is estimated that agriculture and land use change contributed 25% of total anthropogenic emissions of CO_2 .^[3] In soils, the evolution of CO_2 occurs through soil respiration which includes root respiration and the microbial decomposition of crop residues and soil humus.

The soil organic carbon (SOC) content is positively related to structural stability and negatively related to the susceptibility of soils to erosion.^[4] Cultivation reduces the stability of aggregates,^[4] thereby increasing the erodibility of soils. The effect of SOC on susceptibility to erosion is best exemplified by the use of SOC content in determining the soil erodibility factor (K) in the universal soil loss equation.^[5] Worldwide, an estimated 1.11×10^9 ha of land are affected by water erosion,^[6] resulting in the annual transport of $20\text{--}25 \times 10^9$ Mg of sediment to rivers and oceans.^[7] While much is known about the protective role of SOC against soil erosion, information is limited regarding the fate of SOC mobilized by erosion. In this entry, we deliberate that water erosion, as a mechanism for SOC removal, has a profound effect on the global CO_2 budget, and we present an approach to assess erosion-induced CO_2 emission into the atmosphere.

ERODED SOC: STATE OF THE KNOWLEDGE

Water erosion involves the detachment of soil aggregates, the release of aggregate-protected SOC, and the translocation and deposition of detached soil particles. Globally, the estimates of SOC displaced in terrestrial ecosystems by water erosion are in the order of 57 Pg C yr^{-1} .^[8] There is considerable uncertainty regarding the fate of the eroded SOC, however. The Intergovernmental Panel on Climate Change echoed this uncertainty in its handbook for conducting national greenhouse gas inventory by stating that the net effect of erosion on CO_2 evolution is unclear at the time.^[2]

The fate of SOC mobilized by erosion can include redistribution, long-term storage in aquatic and terrestrial systems, and mineralization during transport and deposition leading to emission of CO_2 (the production of CH_4 is possible depending on the oxidation status). During transport, detached soil particles are hydrodynamically sorted and redistributed over the eroding landscape as determined by flow condition, surface roughness, particle size, and geomorphology. Between 80% and 90% of the total soil mass mobilized by water erosion remains near the site of erosion.^[9] A fraction of the eroded SOC is exported to streams, rivers, and lakes. Several publications have documented the magnitude of these exports which are in the order of $0.6\text{--}0.8 \text{ Pg C yr}^{-1}$.^[10,11]

Eroded SOC in runoff is also retained in the low-lying portions of the eroding landscape, but the fate of SOC entrapped in these terrestrial deposits is also not fully understood. On the premise that the decomposition process may be oxygen constrained, it has been suggested that entrapment of eroded SOC in terrestrial deposits can be a mechanism for carbon (C) sequestration.^[12,13] It is estimated that a sequestration of $0.6\text{--}1.5 \text{ Pg C yr}^{-1}$ may be possible via this mechanism.^[13]

Eroded soil materials typically contain twice as much (or more) C than the average topsoil.^[12] In addition, these

materials are enriched in labile C.^[14] Given the propensity of these SOC fractions to be associated with the lighter and most erosion-prone soil particles, the mineralization of eroded C during transport and deposition is a reasonable expectation. A study comparing CPMAS ¹³C-nuclear magnetic resonance spectrometry data of eroded and deposited soil samples indicated that 70% of SOC in eroded soil could be decomposed during transport and deposition.^[15] A review suggests a lower (20%) mineralization rate.^[8]

The abovementioned information clearly underscores the need for better quantitative assessments of the various fates of eroded SOC. Of particular interest is an assessment of water erosion's contribution to atmospheric CO₂. At that time, there is almost no experimental data to support such an evaluation. However, an indirect determination of CO₂ production during transport and deposition of eroded SOC is possible using literature data pertaining to the other pathways identified above.

ASSESSMENT OF EROSION-INDUCED CO₂ FLUXES FROM AVAILABLE DATA

Conceptual Framework and Description of the Mass Balance Approach

In order to estimate CO₂ production during transport and deposition of eroded SOC, a mass balance approach,

described in Fig. 1, is proposed. This approach assumes a steady-state equilibrium with respect to SOC content. Augmentation of the initial pool of SOC via humification of crop residues compensates for SOC losses, which include soil respiration, leaching, and SOC translocation with runoff waters. It follows that:

$$C_0 + h.C_A = C_t + S_R + E \quad (1)$$

$$S_R = k.C_t \quad (2)$$

$$E = L_C + S_W + E_W + S_L + E_L \quad (3)$$

$$E_L = (C_0 - C_t)/t + h.C_A - k.C_t - (L_C + S_W + E_W + S_L) \quad (4)$$

where C_0 is the initial SOC stocks, h is the humification rate, C_A is the annual residue addition, C_t is the SOC stocks after t years, S_R is the annual SOC mineralization, k is the coefficient of SOC mineralization, L_C is the leaching loss of SOC, S_W is the terrestrial C delivered to lakes and oceans, E_W is CO₂ evasion from aquatic systems (lakes, rivers, and streams), S_L is the C storage in terrestrial deposits, and E_L is the mineralization and CO₂ evolution during transport and deposition of eroded SOC. The coefficients h and k are in units of yr^{-1} , C_0 and C_t are in g C m^{-2} , and the other parameters are in units of $\text{g C m}^{-2} \text{yr}^{-1}$.

Using Eq. 4, CO₂ evolution during transport and deposition of eroded SOC (E_L) can be computed. Values

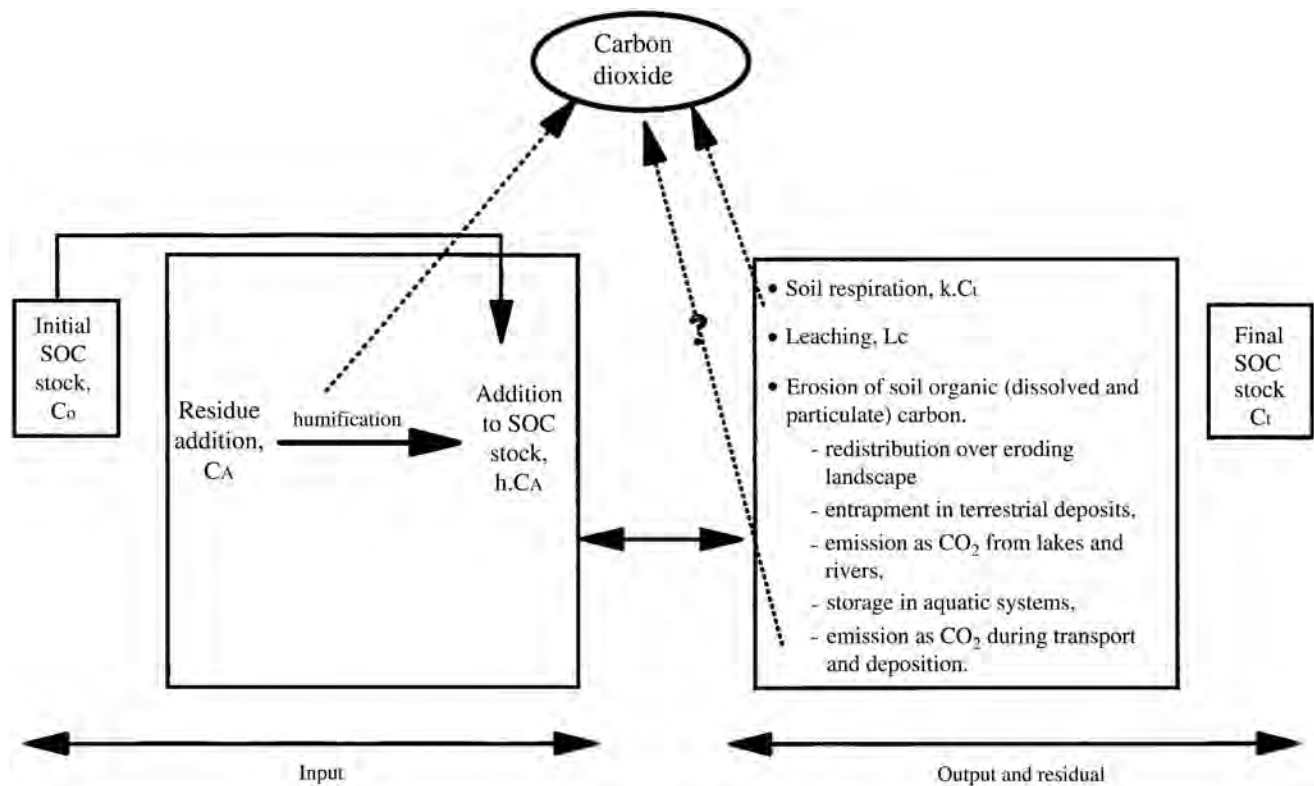


Fig. 1 A conceptual mass balance model of SOC evolution in soils.

SOC might go unnoticed for some time. Removal of SOC via sheet erosion could be particularly difficult to detect in the short-term, hence a possible, dismissal of water erosion as an important mechanism for SOC loss.^[26] However, field study data support the view that erosion is a significant contributor to SOC decline and sometimes contributes more to SOC depletion than soil respiration.^[27,28]

If the data in Table 2 are extrapolated to the global cropland (1.5×10^9 ha),^[29] erosion-induced global CO₂ fluxes to the atmosphere ranging between 0.12 and 0.55 PgCyr⁻¹ can be computed. These fluxes could represent up to one-third of the annual CO₂ emission (1.6 PgCyr^{-1}) due to deforestation and land use change in the tropics.^[8]

CONCLUSION

The information presented in this entry supports our contention that sizable amounts of CO₂ can be produced as a result of SOC mobilization by water erosion. Moreover, water erosion could elicit profound disturbances in the plant–soil–atmosphere segment of the global C cycle through a cascading process involving nutrient depletion, decreased biomass production, and the alteration of radiative and thermal properties of soils. The strong control of temperature on CO₂ emission^[30] and the higher temperature usually recorded in eroded soils^[31] also indicate a shortened residence time of photosynthetically fixed C in the soil system. Although many of the effects are to be evaluated experimentally, available information suggests a positive feedback of water erosion on atmospheric CO₂ and climate warming.

ACKNOWLEDGMENT

The authors gratefully acknowledge the technical support provided by Dr. J.M. Kimble of the NSSC of NRCS at Lincoln, Nebraska.

REFERENCES

1. IPCC. *Land Use, Land Use Change and Forestry*. Special Report; Intergovernmental Panel on Climate Change; Cambridge University Press: Cambridge, 2000.
2. IPCC. *Revised IPCC Guidelines for National Greenhouse Gas Inventories*. Reference Manual. Intergovernmental Panel on Climate Change, 1996; Vol. 3.
3. Duxbury, J.M. The significance of agricultural sources of greenhouse gases. *Fert. Res.* **1994**, *38* (2), 151–163.
4. Cook, G.D.; So, H.B.; Dalal, R.C. Structural degradation of 2 vertisols under continuous cultivation. *Soil. Till. Res.* **1992**, *24* (1), 47–64.
5. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses: A Guide to Conservation Planning*; Agriculture

- Handbook No. 537; United States Department of Agriculture: Washington, 1978; 58 pp.
6. Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 99–118.
7. Milliman, J.D.; Syvitski, J.P.M. Geomorphic/tectonic control of sediment discharge to the ocean: The importance of small mountainous rivers. *J. Geol.* **1992**, *100* (5), 325–344.
8. Lal, R. Global soil erosion by water and carbon dynamics. In *Soil and Global Change*; Lal, R., Kimble, J.M., Levine, E., Stewart, B.A., Eds.; CRC/Lewis: Boca Raton, 1995; 131–142.
9. Walling, D.E. The sediment delivery ratio problem. *J. Hydrol.* **1983**, *65*, 209–237.
10. Hope, D.; Billet, M.F.; Cresser, M.S. A review of the export of carbon in river water: Fluxes and processes. *Environ. Pollut.* **1994**, *84* (3), 301–324.
11. Ludwig, W.; Amiotte-Suchet, P.; Probst, J.L. River discharges of carbon to the world's oceans: Determining local inputs of alkalinity and of dissolved and particulate organic carbon. *Comptes Rendus de l'Académie des Sciences, Série II., Sciences de la Terre et des Planètes* **1996**, *323* (12), 1007–1014.
12. van Noordwijk, M.; Cerri, C.; Woormer, P.L.; Nugroho, K.; Bernoux, M. Soil carbon dynamics in the humid tropical forest zone. *Geoderma* **1997**, *79* (1–4), 187–225.
13. Stallard, R.F. Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial. *Global Biogeochem. Cy.* **1998**, *12* (2), 231–257.
14. Tiessen, H.; Stewart, J.W.B. Particle-size fractions and their uses in studies of soil organic matter: II. Cultivation effects on organic matter composition in size fractions. *Soil Sci. Soc. Am. J.* **1983**, *47*, 509–514.
15. Beyer, L.; Frund, R.; Schleuss, U.; Wachendorf, C. Colluviosols under cultivation in schleswig-holstein. 2. Carbon distribution and soil organic matter composition. *J. Plant Nutr. Soil Sci.* **1993**, *156* (3), 213–217.
16. Guggenberger, G.; Kaiser, K. Significance of DOM in the translocation of cations and acidity in acid forest soils. *J. Plant Nutr. Soil Sci.* **1998**, *161* (2), 95–99.
17. Mulholland, P.J.; Elwood, J.W. The role of lake and reservoir sediments as sinks in the perturbed global carbon cycle. *Tellus* **1982**, *34* (5), 490–499.
18. Spitzy, A.; Leenheer, J. Dissolved organic carbon in rivers. In *Biogeochemistry of Major World Rivers*; Degens, E.T., Kempe, S., Richey, J.E., Eds.; SCOPE 42; John Wiley and Sons: Chichester, 1991; 214–232.
19. Meybeck, M. Concentrations des Eaux Fluviales en Eléments Majeurs et Apports en Solution aux Océans. *Revue de Géologie Dynamique et de Géographie Physique* **1979**, *21*, 215–246.
20. Buyanovsky, G.A.; Kucera, C.L.; Wagner, G.H. Comparative analyses of carbon dynamics in native and cultivated ecosystems. *Ecology* **1987**, *68* (6), 2023–2031.
21. Paul, E.A.; Harris, D.; Collins, H.P.; Schulthess, U.; Robertson, G.P. Evolution of CO₂ and soil carbon dynamics in biologically managed, row crop agroecosystems. *Appl. Soil Ecol.* **1999**, *11* (1), 53–65.

22. Rasmussen, P.E.; Albrecht, S.L.; Smiley, R.W. Soil C and N changes under tillage and cropping systems in semiarid pacific northwest agriculture. *Soil Till. Res.* **1998**, *47* (3–4), 197–205.
23. Dumanski, J.; Desjardins, R.L.; Tarnocai, C.; Monreal, C.; Gregorich, E.G.; Kirkwood, V.; Campbell, C.A. Possibilities for future carbon sequestration in canadian agriculture in relation to land use changes. *Climatic Change* **1998**, *40* (1), 81–103.
24. Lal, R. Soil quality changes under continuous cropping for seventeen seasons of an alfisol in Western Nigeria. *Land Degr. Dev.* **1998**, *9* (3), 259–274.
25. Moeller, J.R.; Minshall, G.W.; Cummins, K.W.; Petersen, R.C.; Cushing, C.E.; Sedell, J.R.; Larson, R.A.; Vannote, R. Transport of dissolved organic carbon in streams of differing physiographic characteristics. *Org. Geochem.* **1979**, *1*, 139–150.
26. Schlesinger, W.H. Changes in soil carbon storage and associated properties with disturbance and recovery. In *The Changing Carbon Cycle: A Global Analysis*; Trabalka, J.R., Reichle, D.E., Eds.; Springer-Verlag: New York, 1986; 194–220.
27. Slater, C.S.; Carleton, E.A. The effect of erosion on losses of soil organic matter. *Soil Sci. Soc. Am. Pro.* **1938**, *3*, 123–128.
28. Gregorich, E.G.; Anderson, D.W. Effects of cultivation and erosion on soils on four toposequences in the canadian prairies. *Geoderma* **1985**, *36* (3–4), 343–354.
29. FAO. *Production Yearbook*; Food and Agricultural Organization of the United Nations: Rome, 1998; Vol. 52, 380 pp.
30. Fortin, M.C.; Rochette, P.; Pattey, E. Soil carbon dioxide fluxes from conventional and no-tillage smallgrain cropping systems. *Soil Sci. Soc. Am. J.* **1996**, *60* (5), 1541–1547.
31. Sutherland, R.A.; Menard, T.; Perry, J.L.; Penn, D.C. The influence of rolled erosion control systems on soil temperature and surface Albedo: Part I. A greenhouse experiment. *Land Degrad. Dev.* **1998**, *9* (2), 159–178.

Erosion: Controlling Irrigation-Induced

Robert E. Sojka

Northwest Irrigation and Soils Research Lab, U.S. Department of Agriculture (USDA), Kimberly, Idaho, U.S.A.

David L. Bjorneberg

Northwest Irrigation and Soil Research, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Kimberly, Idaho, U.S.A.

Abstract

Erosion is the greatest threat to agricultural sustainability. Irrigation-induced erosion and rainfall-induced erosion result from the same physical and chemical processes. However, the processes come together and interact very differently in each case. This entry presents methods of combatting irrigation-induced erosion.

INTRODUCTION

Erosion is the greatest threat to agricultural sustainability. Most irrigation is on fragile arid soils that have enormous crop yield potential when irrigated. However, the yield potential is easily lost if the thin veneer of “topsoil” is eroded.^[1] Erosion prevention on irrigated land is arguably more important than on rain-fed land. Yields from irrigated land are more than double than those from non-irrigated land, with nearly triple the crop value per hectare.^[2] In addition, runoff and irrigation return flows (necessary in many surface irrigation schemes) deliver sediment; human, animal, and plant pathogens; and nutrients and pesticides to downstream fields and riparian waters. These pollutants accumulate in runoff primarily as a result of erosion.

IRRIGATION'S UNIQUE EROSION CHARACTERISTICS

Irrigation-induced erosion and rainfall-induced erosion result from the same physical and chemical processes. However, the processes come together and interact very differently in each case.^[2–4] The magnitude of the differences depends on the type of irrigation system and on soil and water properties. (See the entry *Irrigation: Erosion*, p. 1261.) Briefly, the most important differences stem from soil and water chemistry, wetting rate, water application and infiltration patterns, and, for surface irrigation, the absence of water drop impact. These factors are the basis for many erosion control practices unique to irrigation.^[4–7] Since 1990, advances in irrigation erosion control have resulted from the improved understanding of water quality and antecedent soil condition effects on erosion and from the development of polyacrylamide (PAM) use.

The key to controlling erosion is controlling runoff. Runoff is controlled in two ways. It is minimized by scheduling irrigation to meet, but not exceed, crop water and salt leaching requirements (i.e., avoid over-irrigation), and it is managed by using application rates during each scheduled irrigation that minimize runoff and erosion for that event. Systems should be designed and operated to minimize over-irrigating of some areas in order to adequately irrigate others. In furrow irrigation, this is accomplished by reducing the length of furrows, managing inflow rates and advance times, and where possible, cutting back inflow rates once runoff begins or through use of surge irrigation (surge irrigation sometimes erodes near the inlet because of higher flows during initial pulsing of water). Sprinkler systems can reduce runoff with variable rate emitters that match application rates to soil infiltration rates at specific field locations.

Erosion reduction from improved scheduling and application management is usually proportional to runoff reduction. Reducing overapplication also reduces pumping costs and losses of applied nutrients and agrichemicals. In surface irrigation systems, where 20–40% runoff is often required to achieve field application uniformity, erosion remediation can be integrated into water supply enhancement by pumping sediment-laden drain water back onto fields. This does not prevent erosion but does replace most of the eroded soil along with the saved water for the pump-back cost.

METHODS OF CONTROL

Conversion to Sprinklers

One effective way to prevent irrigation-induced erosion is conversion from surface to sprinkler irrigation.

Again, the soil conservation benefit from conversion to sprinklers derives from and is proportional to the reduction of runoff. Sprinkler irrigation has higher technical, capital, energy, and infrastructure requirements than surface irrigation. Therefore, sprinklers are used only on a small fraction of global irrigated area, whereas nearly 60% of U.S. irrigated land uses sprinklers. Properly designed and managed sprinkler systems eliminate 100% of off-site sediment losses. However, with sprinklers, there is a tendency to extend irrigation to steeper slopes or otherwise more erosive lands. On steep land, when sprinkler systems are poorly designed or managed, erosion can occur.

Center pivots can cause erosion problems because of water running in wheel ruts, down steep slopes, or due to high application rates at outer reaches of the pivot,^[8] especially when using extendable booms and high volume end guns to reach corners. Erosion from high application areas, or where runoff concentrates, can be reduced using tillage, pitting, and mulching between rows to increase surface roughness storage and reduce runoff.^[9–11]

Soil Protection and Tillage

Many approaches developed to control rainfall-induced erosion can prevent irrigation-induced erosion, particularly under sprinklers, e.g., no-till and conservation tillage, which rely on crop residue to protect the soil surface. Despite typical erosion reductions >90%, often with increased yields,^[12] no-till and conservation tillage are rarely practiced by surface irrigators. Floating residue often migrates along and clogs irrigation furrows, washing out adjacent beds and furrows, while under-irrigating the blocked furrow. In basin flood irrigation, floating debris can interfere with water spreading, sometimes concentrating initial flows, eroding some areas, and burying emerging plants with sediment or debris. No-till farming with furrow irrigation is further complicated by crop rotations that require different row (and furrow) spacings each season.

Sojka et al.^[13] demonstrated 60% reduction in field sediment loss from furrow-irrigated potatoes that were paratilled (subsoiled) following planting. Slight yield increases and significant tuber grade improvements raised profitability under both furrow and sprinkler irrigation with paratilling.^[14] Because irrigation assures crop water availability, yield benefits from improved root development are not consistently seen with subsoiling in irrigated crops.^[15] Subsoiling is commonly practiced with sprinkler irrigation to enhance infiltration and decrease runoff, thereby reducing erosion. However, farmers are cautious about subsoiling furrow-irrigated crops because of the potential for irregular water flows in subsurface cracks to interfere with irrigation uniformity. Field preparation or land forming practices that

reduce water application uniformity, or increase runoff, are avoided by irrigators.

Placing mulch or growing sod in irrigated furrows reduces erosion. Sod nearly eliminated runoff sediment.^[16] Straw mulching reduced sediment loss from 52% to 71%.^[17–20] Drawbacks of these techniques relate to the management of sodded furrows, the added operations and equipment needed to place straw, and debris migrating along and clogging mulched furrows.

Site Modification

Various “engineering” approaches have been used to reduce field sediment losses from surface irrigation. The most common is use of settling basins. Large quiescent pools to facilitate particulate settling from runoff collected from fields up to 20 hectares are fairly typical. Settling pond size depends upon the area served, rate and volume of runoff, sediment concentrations expected, and particle-size distribution. Small settling basins along the bottom of surface-irrigated fields, serving a few rows per basin, are sometimes easier to manage at season’s end, when trapped sediment spread back onto the field using farm equipment. Big ponds require large-scale equipment for construction, cleaning, and soil redistribution. For medium-textured soils, about 60% of suspended mass entering settling ponds is retained. The non-retained soil is of the clay size range.^[21] Since clay carries most of sediment’s adsorbed nutrient and chemical load, failure of ponds to retain clay impedes the retention of agricultural chemical pollutants despite the high percentage of sediment mass captured. Furthermore, effectiveness declines as ponds fill with sediment, reducing water residence time. Another variation on ponds is the installation of buried drains and standpipes to regulate water level in tail ditches.^[22] The standpipes force ponding and prevent gradual concaving of field tail ends. They do not, however, prevent loss into the drain of sediment entrained in runoff from upper field reaches.

Altering canopy configuration can reduce erosion. Sojka et al.^[23] halved field sediment loss using narrow or twin row plantings. Water ran between closely placed furrows, reducing irrigation duration (and runoff) and allowing root systems and canopy debris to reduce soil detachment in the furrow. Filter strip crops drilled at right angles into the final 3–6 m of furrow-irrigated row crops also remove entrained sediments from runoff^[6] but do not prevent sediment migration from field inlet to tail end. Because filter strip management is a compromise between two crops, yield from the strips is typically half that expected for either crop alone.

Water Properties

Both the physical and chemical properties of irrigation water affect erosion. Erosion is greatly reduced by reducing

sprinkler droplet size or energy^[24,25] or by reducing stream flow in furrows.^[26] These physical changes require adjustments in application timing, furrow lengths, and irrigation durations to properly match water application constraints with crop water needs.

Water electrolyte chemistry greatly affects the erosiveness of irrigation water.^[27–30] High sodium adsorption ratio (SAR) and low electrical conductivity (EC) contribute to soil aggregate detachment, disruption, and dispersion of fine primary soil particles in runoff. The effect of low EC and high SAR is synergistic. Increasing electrolyte concentration with a calcium source lowers SAR, shrinks the ionic diffuse double layer around charged soil particles, and prevents dispersion, thereby maintaining aggregate stability and resisting erosion. The conjunctive use of waters from different sources or the addition of calcium can raise EC and/or lower SAR to reduce erosion potential and improve infiltration by stabilizing surface soil structure.

Adding large polymeric compounds to irrigation water is an effective erosion prevention technology.^[31–33] These compounds, when delivered in dilute concentrations (typically 1–10 ppm) by the irrigation stream, increase aggregate stability and interaggregate cohesion as water infiltrates. The erosion reduction of 95% is typical for the application of 1–2 kg/ha per treated irrigation. Adoption has been the greatest for furrow irrigation erosion reduction, but interest in extending the technology to sprinklers is growing as much to improve infiltration uniformity as to reduce erosion.^[34–36] The most successful class of polymers has been anionic PAM, allowing safe, easy, and effective erosion prevention for seasonal application rates of 3–5 kg/ha^{−1}, or under \$35/ha^{−1} per season.^[37]

REFERENCES

1. Carter, D.L.; Berg, R.D.; Sanders, B.J. The effect of furrow irrigated erosion on crop productivity. *Soil Sci. Soc. Am. J.* **1985**, *49* (1), 207–211.
2. Bjorneberg, D.L.; Kincaid, D.C.; Lentz, R.D.; Sojka, R.E.; Trout, T.J. Unique aspects of modeling irrigation-induced soil erosion. *Int. J. Sediment. Res.* **2000**, *15* (2), 245–252.
3. Trout, T.J.; Neibling, W.H. Erosion and sedimentation processes on irrigated fields. *J. Irrig. Drain. E-ASCE* **1993**, *119* (6), 947–963.
4. Sojka, R.E. Understanding and managing irrigation-induced erosion. In *Advances in Soil and Water Conservation*; Pierce, F.J., Frye, W.W., Eds.; Ann Arbor Press: Chelsea, 1998; 21–37.
5. Carter, D.L. Soil erosion on irrigated lands. In *Irrigation of Agricultural Crops*; Stewart, B.A., Nielson, D.R., Eds.; Agronomy Monograph No. 30; American Society of Agronomy: Madison, 1990; 1143–1171.
6. Carter, D.L.; Brockway, C.E.; Tanji, K.K. Controlling erosion and sediment loss from furrow-irrigated cropland. *J. Irrig. Drain. E-ASCE* **1993**, *119* (6), 975–988.
7. Sojka, R.E.; Carter, D.L. Constraints on conservation tillage under dryland and irrigated agriculture in the United States Pacific Northwest. In *Conservation Tillage in Temperate Agroecosystems*; Carter, M.R., Ed.; Lewis Publishers: Boca Raton, 1994; 285–304.
8. Gilley, J.R.; Mielke, L.N. Conserving energy with low-pressure center pivots. *J. Irrig. Drain. Div. ASCE* **1980**, *106* (1), 49–59.
9. Aarstad, J.S.; Miller, D.E. Soil management to reduce runoff under center-pivot sprinkler systems. *J. Soil Water Conserv.* **1973**, *28* (1), 171–173.
10. Kranz, W.L.; Eisenhauer, D.E. Sprinkler irrigation runoff and erosion control using interrow tillage techniques. *Appl. Eng. Agr.* **1990**, *6* (6), 739–744.
11. Oliveira, C.A.S.; Hanks, R.J.; Shani, U. Infiltration and runoff as affected by pitting, mulching and sprinkler irrigation. *Irrigation Sci.* **1987**, *8* (1), 49–64.
12. Carter, D.L.; Berg, R.D. Crop sequences and conservation tillage to control irrigation furrow erosion and increase farmer income. *J. Soil Water Conserv.* **1991**, *46* (12), 139–142.
13. Sojka, R.E.; Westermann, D.T.; Brown, M.J.; Meek, B.D. Zone-subsoiling effects on infiltration, runoff, erosion, and yields of furrow-irrigated potatoes. *Soil Till. Res.* **1993**, *25* (4), 351–368.
14. Sojka, R.E.; Westermann, D.T.; Kincaid, D.C.; McCann, I.R.; Halderson, J.L.; Thornton, M. Zone-subsoiling effects on potato yield and grade. *Am. Potato J.* **1993**, *70* (6), 475–484.
15. Aase, J.K.; Bjorneberg, D.L.; Sojka, R.E. Zone-subsoiling relationships to bulk density, cone index, infiltration, runoff and erosion on a furrow irrigated soil. *Trans. ASAE* **2001**, *44* (3), 577–583.
16. Cary, J.W. Irrigating row crops from sod furrows to reduce erosion. *Soil Sci. Soc. Am. J.* **1986**, *50* (5), 1299–1302.
17. Aarstad, J.S.; Miller, D.E. Effects of small amounts of residue on furrow erosion. *Soil Sci. Soc. Am. J.* **1981**, *45* (1), 116–118.
18. Brown, M.J. Effect of grain straw and furrow irrigation stream size on soil erosion and infiltration. *J. Soil Water Conserv.* **1985**, *40* (4), 389–391.
19. Miller, D.E.; Aarstad, J.S.; Evans, R.G. Control of furrow erosion with crop residues and surge flow irrigation. *Soil Sci. Soc. Am. J.* **1987**, *51* (2), 421–425.
20. Brown, M.J.; Kemper, W.D. Using straw in steep furrows to reduce soil erosion and increase dry bean yields. *J. Soil Water Conserv.* **1987**, *42* (4), 187–191.
21. Brown, M.J.; Bondurant, J.A.; Brockway, C.E. Ponding surface drainage water for sediment and phosphorus removal. *Trans. ASAE* **1981**, *24* (6), 1478–1481.
22. Carter, D.L.; Berg, R.D. A buried pipe system for controlling erosion and sediment loss on irrigated land. *Soil Sci. Soc. Am. J.* **1983**, *47* (4), 749–752.
23. Sojka, R.E.; Brown, M.J.; Kennedy-Ketcheson, E.C. Reducing erosion from surface irrigation using furrow spacing and plant position. *Agron. J.* **1992**, *84* (4), 668–675.
24. Bubenzer, G.D.; Jones, B.A. Drop size and impact velocity effects on the detachment of soils under simulated rainfall. *Trans. ASAE* **1971**, *14* (4), 625–628.

25. Kinnell, P.I.A. Laboratory studies on the effect of drop size on splash erosion. *J. Agr. Eng. Res.* **1982**, *27* (3), 431–439.
26. Trout, T.J. Furrow irrigation erosion and sedimentation: On-field distribution. *Trans. ASAE* **1996**, *39* (5), 1717–1723.
27. Le Bissonais, Y.; Singer, M.J. Seal formation, runoff, and interrill erosion from seventeen California soils. *Soil Sci. Soc. Am. J.* **1993**, *57* (1), 224–229.
28. Lentz, R.D.; Sojka, R.E.; Carter, D.L. Furrow irrigation water quality effects on soil loss and infiltration. *Soil Sci. Soc. Am. J.* **1996**, *60* (1), 238–245.
29. Levy, G.J.; Levin, J.; Shainberg, I. Seal formation and interrill soil erosion. *Soil Sci. Soc. Am. J.* **1994**, *58* (1), 203–209.
30. Shainberg, I.; Laflen, J.M.; Bradford, J.M.; Norton, L.D. Hydraulic flow and water quality characteristics in rill erosion. *Soil Sci. Soc. Am. J.* **1994**, *58* (4), 1007–1012.
31. Brown, M.J.; Robbins, C.W.; Freeborn, L.L. Combining cottage cheese whey and straw reduces erosion while increasing infiltration in furrow irrigation. *J. Soil Water Conserv.* **1998**, *53* (2), 152–156.
32. Lentz, R.D.; Sojka, R.E. Field results using polyacrylamide to manage furrow erosion and infiltration. *Soil Sci.* **1994**, *158* (4), 274–282.
33. Orts, W.J.; Sojka, R.E.; Glenn, G.M. Biopolymer additives to reduce erosion-induced soil losses during irrigation. *Ind. Crop. Prod.* **2000**, *11* (1), 19–29.
34. Aase, J.K.; Bjorneberg, D.L.; Sojka, R.E. Sprinkler irrigation runoff and erosion control with polyacrylamide—laboratory tests. *Soil Sci. Soc. Am. J.* **1998**, *62* (6), 1681–1687.
35. Bjorneberg, D.L.; Aase, J.K. Multiple polyacrylamide applications for controlling sprinkler irrigation runoff and erosion. *Appl. Eng. Agr.* **2000**, *16* (5), 501–504.
36. Sojka, R.E.; Lentz, R.D.; Ross, C.W.; Trout, T.J.; Bjorneberg, D.L.; Aase, J.K. Polyacrylamide effects on infiltration in irrigated agriculture. *J. Soil Water Conserv.* **1998**, *53* (4), 325–331.
37. Sojka, R.E.; Lentz, R.D. Reducing furrow irrigation erosion with polyacrylamide (PAM). *J. Prod. Agr.* **1997**, *10* (1), 47–52.

Erosion: Crop Yield

Michael Stocking

School of Developmental Studies, University of East Anglia, Norwich, U.K.

Abstract

Erosion-induced loss in soil productivity is one of the major threats to global food and economic security, especially to resource-poor farmers in the tropics. It not only diminishes the quality of soil resources but also makes gaining a livelihood from the land increasingly difficult. By reducing the productivity of the soil, it affects outputs such as crop yields derived from the renewable natural resource systems of the biosphere. By affecting the net primary production of vegetation, erosion is the driver for a number of critical environmental, economic, and social issues in the developing and developed countries alike.

INTRODUCTION

In tropical and subtropical developing countries, the majority of people are dependent on low external input agriculture. Directly they rely on the inherent quality of soil resources. Because of high-intensity (erosive) rainfall, susceptible (erodible) soils, and sensitive ecological balances, the processes of erosion are significantly accelerated in the tropics compared to temperate areas. Their impacts, in terms of both yields lost and livelihoods jeopardized, are considerably magnified. In the developed countries, usually temperate to subtropical, soil erosion is also a critical concern. However, fewer people are directly dependent on the soil, and the processes can be addressed more easily through the application of technology and economic resources. Impacts may be mitigated through, e.g., the application of fertilizer or using irrigation. Off-site effects such as siltation of reservoirs can be offset by available, albeit costly, remedial measures. Nevertheless, many commentators (e.g., see Pimentel et al.^[1]) find the prevalence of soil erosion and associated losses in crop yield totally unacceptable and unsustainable, and they predict environmental catastrophe. It should be noted, however, that the evidence for erosion and its widespread impact is by no means unchallenged.^[2,3]

Soil quality and soil resilience, soil fertility and plant nutrients, conservation and rehabilitation, soil management, yield output and environmental risk, and farming livelihoods are just a few of the problematic implications of soil erosion and declining crop yields. Arguably, erosion affects not only the economics of production, but also the stability, security, and life support of society.

“HOT SPOTS” FOR SOIL EROSION AND ITS IMPACT

Largely to identify those parts of the world with especially severe impacts of soil erosion, the concept of hot spots of land degradation was devised in 1995 by an expert meeting at the International Food Policy Research Institute in Washington, D.C.^[4] It is a useful concept in that erosion can then be related to processes such as deforestation, inappropriate agricultural mechanization, and building of engineering structures, to identify not only the severe status of soil erosion but also the primary causative agents and the effects on human society. An example of hot spots for Africa is given in [Table 1](#). These are places where expert opinion has identified soil erosion as posing a significant threat to food security for large numbers of people. Policy interventions should be targeted at such situations in order to address the developmental needs of the poorest people in the most vulnerable biophysical environments.

SOIL EROSION-PRODUCTIVITY RELATIONSHIPS

Unambiguously ascribing declining yields to the effects of the processes of soil erosion is difficult. Yields may decline for various reasons, such as excessive off-take of nutrients in crops without replenishment, pests and diseases, weed infestations, and increasing prevalence of drought because of global climate change. Soil factors, some associated with erosion, may also be responsible, such as reduction in effective rooting depth; decrease in available water capacity; decline in soil organic carbon; salinity and sodicity; and other chemical changes causing

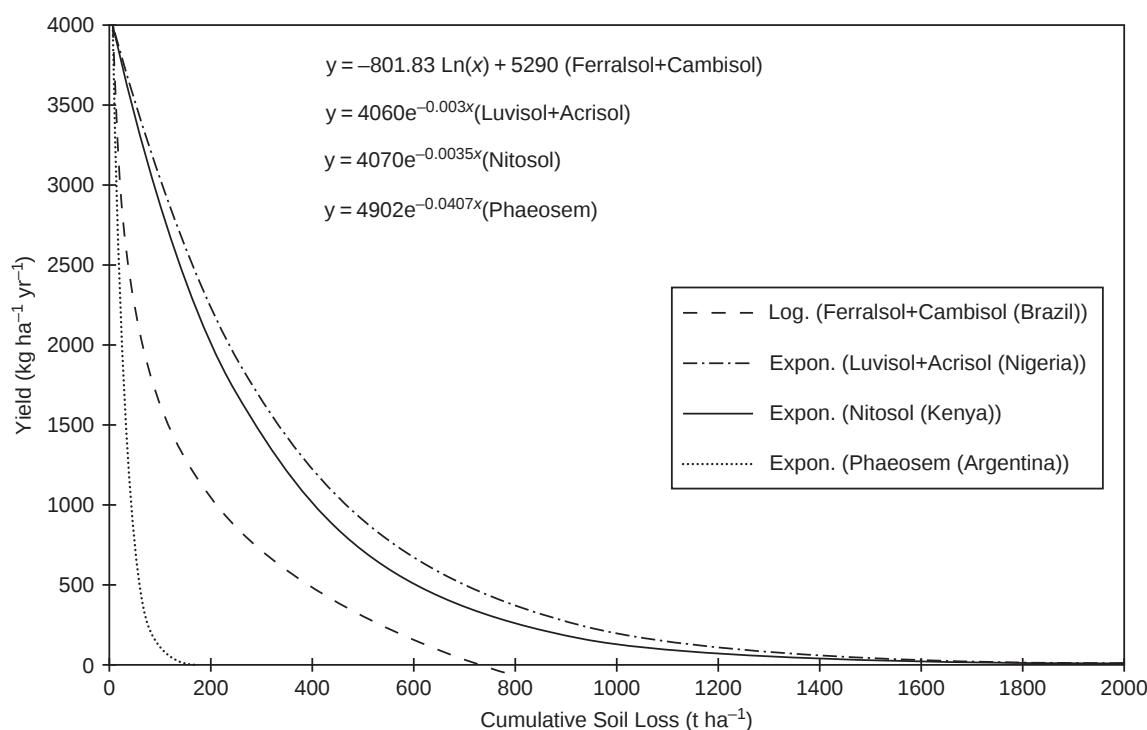
Table 1 Hot spots for soil erosion, land degradation, and crop yields in Africa.

Nutrient depletion	Water and wind erosion	Vegetation degradation	Constraints to yield increase
Semiarid croplands of Burkina Faso and Senegal (leading to outmigration)	Subhumid SE Nigeria on sandy soils Wind erosion in Sahel	Arid and semiarid rangeland devegetation (e.g., Ciskei), particularly near water sources	Lack of suitable technology for crops grown in areas less than 300-mm rainfall in North Africa
Large areas under transition to short fallow or permanent cropping	Mechanization in North Africa causing water and wind erosion	Devegetation due to intensive collection of wood fuel (around cities)	Unsustainability of annual crops in humid lowlands of West Africa
Reduction of silt deposits in Nile Delta following construction of Aswan High Dam	Mechanization with inappropriate plowing techniques (e.g., transition zone of West Africa)	Devegetation due to overstocking (e.g., Morocco and Tunisia) Reduced yields due to <i>Imperata</i> and <i>Chromolaena</i> infestations in degraded soils	Densely populated highlands in Rwanda, Burundi, and Kenya—no obvious source of production increase

Source: From Scherr & Yadav^[4] and Tengberg & Stocking.^[5]

toxicity by aluminum, heavy metals, or acidification generally. Behind all these, as an associated factor, is one form of soil degradation or another, including the most common type, soil erosion by water. Indeed, such is the complexity of interrelated factors that soil erosion and declining biomass production will accompany almost any negative environmental change. Thus, at the least serious end of the scale of impact, the conversion of dryland savannah to continuous soybean will necessarily provoke accelerated soil erosion because of the soil

disturbance and reduced vegetation cover. The legume may contribute some replenishment of soil nitrogen, and the remaining roots and stover may contribute some organic matter. But eventually that erosion must have a net deleterious effect on production because of reduced topsoil thickness, loss of colloids, and a lowered capacity to hold plant-available water. However, at a cost, sustainable agricultural production systems can accommodate these changes, and production can effectively be continued indefinitely through the application of sufficient

**Fig. 1** Erosion–yield relationships for a selection of tropical soils.

Source: From FAO.^[6]

Table 2 Years taken for different soil orders to reach a critical yield level of 1000 kg ha⁻¹ yr⁻¹ with continued erosion.

Management level	Ferralsol	Acrisol	Luvisol	Phaeozem	Cambisol	Nitisol
Good	3	4	93	200	210	950
Moderate	2	3	23	65	42	19
Poor	1	2	9	7	23	4
Bare soil	1	1	5	3	9	3

Source: From Tengberg & Stocking.^[7]

resources and skills. At the most serious end of the scale, “badlands” (the combinations of extremely serious erosion, with gullies, rills, pipes, and other phenomena typical of abused semiarid sites) have no vegetation whatsoever. In both cases, is it soil erosion that causes declining vegetation?—yes and no. A diminished soil quality brought about by erosion must affect plant processes of production and reproduction though the effect may be masked through technology and inputs. Equally, reduced vegetation brings about more soil erosion. However, completely denuded landscapes are rare. The eroded soil is deposited elsewhere, bringing with it nutrients, organic matter, and water, as well as enhanced production opportunities for local people. South Asia has many examples of “sediment harvesting,” such as the *nullah plugs* in India. So, in the discussion of erosion–productivity relationships, it must always be recognized that: 1) the relations are complex and direct cause–effect conclusions should only be entertained cautiously; and 2) a deleterious effect in one part of the landscape may bring benefits elsewhere.

With these caveats in mind, it is possible to define the relationships between erosion and yield. This is usually done experimentally on runoff plots, where measured soil losses are related not only to the factors of erosion but also to the existing and future yields of crops (see Olson et al.^[6] for other methods to study the erosion–productivity relationship). In the mid-1980s, the United Nations’ Food and Agriculture Organization (FAO) developed an experimental design to investigate erosion–productivity relationships.^[8] By attempting to hold all factors constant other than prior degree of erosion and crop yields, the small plots were allowed to erode by natural rainfall to different degrees and then planted to a standard crop to measure the impact on plant production. At the same time, the explanation for the relationships between erosion and productivity was determined through analyzing the change in *in situ* soil variables assumed to have been caused by the erosion. The database has been used to construct scenarios of food insecurity if erosion were allowed to continue (e.g., see Tengberg and Stocking^[5]).

There is good evidence that yield decline with erosion follows a curvilinear, negatively exponential form (Fig. 1). After a sharp initial decline from a status of high productivity, there follow successive stages of decreasing further impact. While this general form of the relationship may not

hold true for all conditions, the implications for attempts to maintain food security are clear. It is exceedingly difficult to bring back severely degraded land into useful, or even economic production. Slightly degraded land is relatively easily brought back into use, and the returns on an investment in conservation are always better if erosion has not progressed beyond a 50% drop in yield levels. Erosion–yield–time scenarios have been addressed by the FAO, and Table 2 summarizes overall results in the form of years to reach a threshold yield level for maize of 1000 kg ha⁻¹ for some typical soil orders (in the FAO classification). The differences between a resilient soil such as a Nitisol (clay-rich, common in basic rocks of highlands) and extremely sensitive soils such as Ferralsols and Acrisols (common in acid humid tropical rainforests) are stark.

CONCLUSION

Both soil erosion and its effect on crop yield vary in their extent and seriousness. Tropical, developing countries have the most critical conditions, through a combination of environmental, social, and economic factors. Scientists are recognizing that some of the crisis statements of the past owed more to rhetoric than to reality. However, policies are urgently needed to combat the hot spots of soil erosion, so that local communities may continue to rely on their soil resources in the future. International initiatives, such as those through the Convention to Combat Desertification, are starting to address the complex issues involved. But much more needs to be done in matching technologies to local conditions, addressing the link between erosion and yields, showing how local and global objectives such as carbon sequestration are related to soil erosion, and understanding the associated economic and social impacts of allowing erosion to continue. The challenge is immense.

REFERENCES

1. Pimentel, D.; Harvey, C.; Resosudarmo, P.; Sinclair, K.; Kurz, D.; McNair, M.; Crist, S.; Shpritz, L.; Fitton, L.; Saffouri, R.; Blair, R. Environmental and economic costs of soil erosion and conservation benefits. *Science* **1995**, 267 (5201), 1117–1123.

2. Lomborg, B. *The Skeptical Environmentalist*; Cambridge University Press: Cambridge, 2001.
3. Stocking, M.A. Soil erosion. In *The Physical Geography of Africa*; Adams, W.M., Goudie, A.S., Orme, A.R., Eds.; Oxford University Press: Oxford, 1996; 326–341.
4. Scherr, S.J.; Yadav, S. *Land Degradation in the Developing World: Implications for Food, Agriculture and the Environment to 2020*; Food, Agriculture and the Environment Discussion Paper 14; International Food Policy Research Institute: Washington, 1996.
5. Tengberg, A.; Stocking, M. Land degradation, food security and agro-biodiversity—examining an old problem in a new way. In *Response to Land Degradation*; Bridges, E.M., Hannam, I.D., Oldeman, L.R., Penning de Vries, F.W.T., Scherr, S.J., Sombatpanit, S., Eds.; Oxford & IBH Publishing: New Delhi and Science Publishers: Enfield, 2001; 171–185.
6. Olson, K.R.; Lal, R.; Norton, L.D. Evaluation of methods to study soil erosion–productivity relationships. *J. Soil Water Conserv.* **1994**, *49* (6), 586–590.
7. Tengberg, A.; Stocking, M. *Erosion-Induced Loss in Soil Productivity and Its Impacts on Agricultural Production and Food Security*; FAO/AGRITEX Expert Consultation, Harare, Zimbabwe, Dec 8–12, 1997; Land and Water Development Division, UN Food and Agriculture Organization: Rome, 1997.
8. FAO. *Erosion-Induced Loss in Soil Productivity: A Research Design*; Consultants' Working Paper 2; Soil Conservation Programme; United Nations Food and Agriculture Organization: Rome, 1985.

Erosion: Disturbed Lands

William J. Elliot

U.S. Department of Agriculture—Forestry Service (USDA-FS), Moscow, Idaho, U.S.A.

Abstract

Disturbed lands include construction sites, mine sites, roads, forests that have been disturbed by fire or other operations, and military maneuver sites. Erosion rates on disturbed lands are highly dependent upon factors such as climate conditions immediately following the disturbance, the type and degree of disturbance, local topography, and the natural vegetation regeneration or mitigation measures. Soil properties also influence erosion rates but tend to be overshadowed by the other factors. A high degree of variability is associated with each of these factors; thus, the erosion rates resulting from disturbances are highly variable.

CLIMATE AND HYDROLOGY

Vegetation regeneration on disturbed sites depends upon precipitation. Too little precipitation means that the site takes longer to regenerate, whereas too much precipitation will likely lead to excessive soil erosion that may also hinder revegetation. However, high levels of low-intensity precipitation will generally result in rapid regeneration and minimal erosion risk.

DISTURBANCES

In the absence of disturbances, land surfaces are often covered in forests or rangelands. Disturbances can accelerate erosion rates as much as one thousand times more quickly than on undisturbed sites. Disturbances may be natural, such as major storm or flood events and fire (Fig. 1), or human caused, such as roads (Fig. 1), construction (Fig. 2), forestry, or mining (Fig. 3). Erosion rates of undisturbed rangelands and forests are low (Table 1). However, occasional natural disturbances are part of the ecosystem, and any increased erosion rates from human disturbances should be compared with both the undisturbed and naturally disturbed erosion rates.

Fire and Water Repellency

Fire may be natural or intentional. Generally, natural or “wild” fires tend to be more severe because the forest or range must be in a drier condition for a wildfire to initiate. Intentional or “prescribed” fires are applied in early summer or after autumn rains have begun, to minimize the amount of litter that is burned, while reducing the amount of aboveground woody debris. Managers use fire to provide improved grazing for livestock and wildlife and to try to improve forest or range health. Fires are

commonly used in forests following a harvesting operation to reduce the amount of remaining woody material or to improve the health of the forest and the success of regeneration. Such practices—when used with a suitable undisturbed buffer between the fire and a stream channel—can minimize upland erosion and sediment delivery risks.

Fire may cause soils to become water repellent. Water repellency is caused by the condensation of hydrocarbons on soil particles and aggregates when organic material is combusted. It can reduce the hydraulic conductivity of soil from 30–80 to 15–40 mm hr⁻¹ in forests.^[6] Rangelands tend to be less susceptible to repellent conditions following fire, although some range shrub communities as well as some forest soils can be naturally repellent. The reduced hydraulic conductivity values associated with repellency may be greater than rainfall intensities or snowmelt rates in the year following the fire. The chemicals causing repellency are water soluble, and repellent soils tend to recover to prefire conditions within 3 or 4 years, if there is sufficient precipitation. If there is a major storm or runoff event that occurs when soils are highly repellent, erosion rates can be large (20–100 Mg ha⁻¹), but if storms are of low to average intensity, erosion rates are likely to be moderate, typically under 10 Mg ha⁻¹. Water repellency can be reduced through tillage, but burying remaining surface residue tends to offset any benefits of reduced repellency.

Anthropogenic Soil Disturbances

Anthropogenic or human-caused disturbances can cause minor changes in the landscape, such as a road (Fig. 1), a footpath, a logging skid trail, or off-road vehicle tracks from recreational or military uses. Human activities can



Fig. 1 Two disturbances that may cause increased erosion are wildfire, and roads. This photograph was taken several weeks after forest fires ravaged Western Montana in 2000.

also cause major landscape disturbances, such as a construction site (Fig. 2) or a surface mine spoil heap (Fig. 3). The same general erosion principles, however, can be applied to all disturbances. The main impacts of these disturbances are reduced live vegetation, loss of surface cover by decaying vegetation, and soil compaction. In addition, roads and mining may expose soils low in organic matter, and high in rock content or undesirable trace elements. These conditions are not conducive to vegetation regeneration, further aggravating long-term erosion risks.

Runoff rates are much higher on most disturbed areas than undisturbed areas. On roads, for example, the hydraulic conductivity can be reduced to near zero (under 0.1 mm hr^{-1}).^[7] Bulk densities are generally increased from under 1.0 Mg m^{-3} in forests and range to over 1.4 Mg m^{-3} on roads. Compaction reduces the water-holding capacity of the soil, which increases runoff and erosion. The increased runoff reduces the amount of water held in storage for the summer, subsequently reducing vegetation regeneration rates. Large



Fig. 2 A construction site exposing large areas to soil erosion until vegetation can be re-established. Silt fences and straw bales have been installed to reduce sediment delivery.



Fig. 3 A surface mine spoil heap many years after mining in northern Idaho. The lack of vegetation increases susceptibility to erosion, but the large rock content reduces actual erosion risks.

rocks (Fig. 3) and acidic soils are common to many mining sites. The rocky surface tends to reduce runoff and erosion but increases the difficulty in establishing vegetation.

In many cases, it is not possible to prevent erosion on a road, a highly disturbed construction site, or mining site. In these cases, managers frequently incorporate practices to reduce off-site erosion using various sediment control techniques such as silt fences, straw bales (Fig. 2), and sediment basins. Vegetated buffer strips are

Table 1 Typical erosion rates in forests, rangelands, and disturbed sites.

Condition	Estimated area in the United States (millions of hectares)	Typical erosion rate (Mg ha^{-1})
Undisturbed forests	200	0.1
Undisturbed rangelands	200	1
Harvested forests	4	1
Conservation agriculture	80	5
Forests after low-severity wildfire	1	10
Overgrazed rangelands	50	10
Rangelands after fire	10	10
Forest roads	3	20
Intensive agriculture	80	20
Surface mines in need of reclamation	0.02	50
Forests after high-severity wildfire	1	100
Construction sites	—	100

Source: From USDA Forest Service,^[1] USDA Natural Resource Conservation Service,^[2] USDI Bureau of Land Management,^[3] USDI National Park Service,^[4] and USDI Office of Surface Mining.^[5]

also commonly used to reduce the amount of runoff and sediment leaving such sites.

Reclamation of highly disturbed sites generally involves techniques such as subsoiling, mulching, and fertilizing to improve soil productivity. Vegetation is then established that will likely provide rapid surface cover until natural cover returns. Once the site is revegetated, erosion risk is minimal.

VARIABILITY

Soil erosion varies greatly with time and in space on disturbed sites. Soil properties and subsurface geologic properties vary along all hillslopes. Roads, skid trails, and other trails can be quite different than the surrounding areas in hydrology and erodibility. Spatial variation is very high following both prescribed and wildfire, leading to major variability in the erosion risk that may follow.^[8] Seasonal variability in climate can have major impacts on erosion rates. Large soil erosion events tend to occur only from major runoff events, or when large runoff events occur on areas that have been disturbed. "Average annual erosion" has little meaning under conditions dominated by temporal variability.

Following disturbance, hydrologic recovery also varies greatly. North-facing slopes may recover faster than south-facing slopes. Topsoils recover faster than subsoils or mine spoil. A dry year following a disturbance may result in little water repellent recovery, a moderate year significant recovery, but a wet year may cause catastrophic flooding and erosion. Large events could disturb a watershed for decades, while stream channels and upland surfaces return to hydrologic stability. Under these conditions, it may be more appropriate to consider the probability of a given amount of erosion occurring rather than an average value.

Because of the spatial variability following disturbances, a given landscape will have sites of net runoff and sediment generation and sites of net infiltration and deposition. The distribution and occurrence of these sites will determine the net landscape response to an erosion event. On landscapes with a few scattered areas of severe disturbance, there may be no net runoff and erosion. This is particularly true with roads, where sediment delivery is minimal if the roads are located a reasonable distance from streams. Only severe storm events that exceed the infiltration capacity of the entire hillslope will cause any net off-site sedimentation.

SURFACE COVER

The amount and distribution of surface cover will determine the erosion rate following most disturbances. Surface cover is generally decomposing vegetation but can also include animal waste, gravel, rocks, and

fabricated materials like erosion control blankets. As surface cover increases from zero to 100%, erosion rates will drop by 99% or more. With fires and grazing, natural spatial variability is extremely high, whereas with military activities, forest harvesting, and roads, the distribution of disturbance on the landscape may be more predictable and more easily managed. Increasing surface cover with vegetation regeneration, mulch, or erosion control fabrics is usually the most effective mitigation method for disturbed sites, although costs can be high.

EROSION PREDICTION

Erosion on disturbed sites can be predicted with both empirical models [such as the Universal Soil Loss Equation (USLE)] and physically based models [such as the Water Erosion Prediction Project (WEPP) model].^[9] Factors have been developed for construction sites for the USLE,^[10] and that technology has been incorporated into many predictive software programs. Interfaces for predicting erosion following fire and forest roads have been developed for the WEPP model,^[11] as have input soil files for construction sites.^[12] When using any of these technologies, model users must carefully consider the period of time that the site is disturbed, the chosen mitigation practices, and how fast the site is revegetated.

REFERENCES

1. USDA Forest Service. *National and Regional Areas Summary*. Available at <http://www.fs.fed.US.2001> (accessed January 2001).
2. USDA Natural Resource Conservation Service. *Land Cover/Use of Nonfederal Rural Land, by State and Year, 2001*. Available at <http://www.nrcs.usda.gov> (accessed January 2001).
3. USDI Bureau of Land Management. *Bureau of Land Management Facts, 2001*. Available at <http://www.blm.gov/nhp/index.htm> (accessed January 2001).
4. USDI National Park Service. *The National Park System Acreage, 2001*. Available at <http://www.nps.gov> (accessed January 2001).
5. USDI Office of Surface Mining. *Unreclaimed Public Health and Safety Coal Related Problems, 2001*. Available at <http://www.osmre.gov> (accessed January 2001).
6. Robichaud, P.R. Fire effects on infiltration rates after prescribed fire in northern rocky mountain forests, USA. *J. Hydrol.* **2000**, 231–232, 220–229.
7. Elliot, W.J.; Hall, D.E. *Water Erosion Prediction Project (WEPP) Forest Applications*; General Technical Report INT-GTR-365; USDA Forest Service, Rocky Mountain Research Station: Ogden, Utah, 1997. Available at <http://forest.moscowfs1.wsu.edu/4702/forestap/forestap.pdf> (accessed October 2001).

8. Robichaud, P.R.; Miller, S.M. Spatial interpolation and simulation of post-burn duff thickness after prescribed fire. *Int. J. Wildland Fire* **2001**, *9* (2), 137–143.
9. Flanagan, D.C.; Nearing, M.A., Eds. *USDA-Water Erosion Prediction Project (WEPP) Hillslope Profile and Watershed Model Documentation*; NSERL Report No. 10. National Soil Erosion Research Laboratory, USDA–Agricultural Research Service: West Lafayette, 1995; 298 pp. Available at <http://topsoil.nserl.purdue.edu/nserlweb/weppmain/docs/readme.htm> (accessed October 2001).
10. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses—A Guide to Conservation Planning*; USDA Agriculture Handbook No. 537; U.S. Department of Agriculture: Washington, 1978.
11. Elliot, W.J.; Hall, D.E.; Scheele, D.L. *FS WEPP Forest Service Interfaces for the Water Erosion Prediction Project Computer Model*; USDA Rocky Mountain Research Station: Fort Collins, 1999. Available at <http://forest.moscowfsl.wsu.edu/fswepp/> (accessed January 2001).
12. Laflen, J.M. *Application of WEPP to Construction Sites*. In *Proceedings of the International Symposium for Soil Erosion Research for the 21st Century*, Honolulu, Hawaii, Jan 3–5, 2001; American Society of Agricultural Engineers: St. Joseph, 2001; 135–138.

Erosion: Effects on Human Life

Neroli Pedro Cogo

Renato Levien

Federal University of Rio Grande do Sul, Porto Alegre, Brazil

Abstract

Soil erosion is a problem associated with human life on earth. It threatens soil productivity, food production, the environment, and quality of life. Fortunately, erosion-induced problems have been well recognized and sound control measures have been developed. To maintain the natural resources and environment for a sustainable civilization, soil erosion from agricultural lands must be controlled.

INTRODUCTION

Soil is, and long will be, a vital resource and an essential medium for life on earth. Plants growing in the soil provide food for human and animal needs. However, because the rate of human-induced water and wind erosion can greatly exceed natural rates of soil formation and erosion,^[1,2] the soil may become depleted and severely affect crop production. Therefore, the soil must be considered a non-renewable natural resource to be protected and preserved for future generations. Maintaining soil productivity for sustainable food production is a challenge for farmers, agricultural and environmental professionals, and governments worldwide.

HISTORICAL EVIDENCE OF EROSIONAL EFFECTS

Decline of civilizations as a result of soil degradation (particularly soil erosion) is well documented.^[3–7] Soil degradation and civilization interact in both directions, meaning erosion-induced land degradation affects human lives, and human activities often enhance soil degradation.^[6]

Historically, three major human-induced erosion periods can be identified.^[7] The first period, between 1000 and 3000 years ago, occurred because of the excessive timber cutting and expanded cultivation. The second period, during the 19th and early 20th centuries, occurred in countries colonized by European immigrants. Through the establishment of an exportation type of agriculture, the immigrants forced the native population to move and to explore the more erodible, marginal lands, while holding the better lands for a marketing economy. The third period, beginning in the early 1920s and extending to the present day, occurred mainly in developing countries with a growing population pressure for more land and food production. This population growth forces farmers to develop new

lands, some good and some only marginal for agricultural production, and to adopt intensive, non-conserving cropping and management systems which cause spectacular levels of soil erosion.

The situation varies throughout the world. In many regions, the seriousness of the erosion problem has been recognized and effective erosion control measures adopted. In other regions, erosion is a major crop production impairment and continuously causes extensive sedimentation and water pollution problems.

HOW SOIL EROSION AFFECTS CROP PRODUCTIVITY AND HUMAN LIFE

The primary effect of soil erosion on crop productivity is a reduced yield (less kg/ha of grain, dry matter biomass, tubers, and fruits) that results from the removal of fertile topsoil and the contained nutrients. In a review paper, Langdale and Schrader^[8] prepared a table showing soil erosion–crop productivity relationships for soils in the Southeastern and Midwestern United States; some of their findings are detailed in [Table 1](#). Because it reduces animal feeds, erosion also results in a reduced livestock yield (less kg/ha of meat, milk, and wool). This means erosion affects both vegetal- and animal-originated food productions. In addition, products of erosion (e.g., soil particles, water-runoff, dissolved chemicals in the runoff-water, and chemicals adsorbed to the soil particles) may cause serious damage to the environment. Decreased food production and increased sedimentation and water pollution caused by soil erosion mean a diminished quality of life for humans.^[8–13]

Erosion affects crop yield through degradations in the soil's physical, chemical, and biological properties. Physical degradations caused by soil erosion are: 1) loss of plant-available water reserve and water storage capacity; 2) reduced rooting depth; 3) increased soil crusting; 4) reduced aggregate stability and soil tilth; 5) formation

Table 1 Examples of how soil erosion affected crop productivity.

Degree of erosion	Crop yield (kg/ha)				
	Corn	Soybeans	Cotton	Small grains	Forage
Southeastern United States					
Grenada silt loam (Glossic Fragiudalfs), 0–5% slope					
None	6,000	2,700	840	3,600	7,200
Eroded	5,300	2,000	784	3,100	6,700
Severe	4,400	1,600	672	2,700	6,000
Cecil sandy clay (Typic Hapludults), 2–10% slope					
Deposition (local alluvium)	6,200	—	—	—	—
Eroded	5,800	2,100–3,100	1,389	2,400	17,400
Severe	1,900	1,500–2,400	866	1,600	13,700
Midwestern United States					
Marshall silty clay loam (Typic Hapludolls), 2.5–6.0% slope					
None	—	—	—	—	—
Slight	6,700	2,800	—	2,200	9,000
Moderate	6,200	2,600	—	2,000	8,500
Ida silt loam (Typic Udorthents), 6.0–9.0% slope					
None	—	—	—	—	—
Moderate	5,200	2,200	—	2,100	6,900
Severe	4,300	1,700	—	1,700	5,800

Source: From Langdale & Schrader.^[8]

of rills and gullies that fragmentize the land; and 6) exposure of a subsurface soil layer that is inadequate for plant growth. Chemical degradations from erosion include: 1) reduced plant nutrients; 2) reduced cation-exchange capacity from the removal of fine soil particles and associated organic matter; and 3) increased toxicity of aluminum and manganese, and soil acidity owing to the exposure of a less-favorable subsurface soil layer. Biological factors affected by erosion include loss of carbon in the biomass and reduced micro- and macrofauna activity.

Production costs on degraded lands are also elevated because of increased fertilizer needs to offset soil nutrient loss, machine power to till and plant more compacted subsoil, and additional seed cost in replanting stand loss from soil crusting and seeds washing off. These effects also cause an increase in machine wear, fuel consumption, time, and labor, and, consequently, increased production cost.

The effects of soil erosion on crop productivity and the environment may be strongly felt by society as a whole, but it essentially affects the human's quality of life (Fig. 1). In addition to rural society stricken by reduced crop yield and



Fig. 1 A family from Rio Grande do Sul State, Brazil, whose farm suffers from severe soil erosion.

increased production costs, urban society also suffers from higher cost for food and reduced environmental quality. There is evidence of this in developing countries with extreme cases of erosion. In these instances, farmers cannot maintain a basic subsistence level, so they abandon their land, move to the cities, and cause what is known as rural exodus. The society also bears the costs of restoring reservoir and river capacities and cleaning chemically polluted surface waters caused by eroded sediments.

AWARENESS OF SOIL EROSION: CROP PRODUCTIVITY RELATIONSHIP

Despite historical evidence that human-induced erosion has occurred worldwide wherever land has been cultivated, awareness of the erosion-induced problem developed slowly. The study of soil erosion as a science started only approximately 100 years ago, between 1877 and 1895, by German soil scientist Ewald Wollny.^[14]

It is difficult to assess the cost of erosion-induced damages to society in terms of loss of food production, water storage and quality, and quality of life. This may partly explain why soil erosion, even though universally recognized as the greatest threat to food security, environmental quality, and human well-being, is continuously uncontrolled in many parts of the world. If soil erosion and crop productivity relationships were better understood, people might be more willing to accept erosion control measures. The society could determine whether erosion control regulations should be imposed for land use purposes.^[15] Perhaps a change must be made to the customary form of reporting soil erosion loss, from mass of soil loss per hectare per year to an economic loss expressed in monetary values and associated with production and natural resource losses.^[11,15]

The effects of erosion on crop productivity are difficult to evaluate independently because of various interactions.^[8–12,15,16] Interactions occur among soil properties, crop characteristics, and climate, which should be accounted for when relating soil erosion and crop yields. Much of the soil's water and nutrient supply potential may be lost before the productivity loss is realized, because erosion effects are gradual, cumulative, not felt from year to year, and significant only over a long time period.

Furthermore, the combined effect on crop yield varies with the intrinsic characteristics of the soil and the specific soil management systems used. This is especially true because improved technology can offset the removal of plant nutrients and soil water storage capacity because of erosion and greatly mask erosion effects on crop yields, especially in the short term. Krauss and Allmaras^[16] stated that “sustained productivity of an eroding soil cannot be determined unless yield increases from technology advances are separated from soil productivity changes due to erosion.” Even with great obstacles, efforts to quantify

erosion's effects on crop productivity must continue. This information is fundamental in developing soil conservation plans and effective land use policies.

TECHNIQUES FOR EVALUATING EROSION IMPACT ON CROP PRODUCTIVITY

Several techniques have been proposed to evaluate the cause-and-effect relationship between crop yield and soil erosion.^[8,11] The direct method involves field experiments of short and long duration in which the soil of the test plot is naturally eroded by rain or artificially eroded by mechanical removal of the topsoil layer. With the direct method, productivity losses can be measured under field conditions, where crop performance, over a period of time, is related to soil loss or erosion-induced alterations in soil properties.

Overall, the direct method is more accurate than the indirect method but requires more capital, time, and labor to accomplish. The indirect method relates measured crop yields with estimated erosion losses from changes in physical and chemical soil properties occurring in the field. The crop productivity–soil erosion relationship is evaluated through mathematical, geomorphological, and crop productivity models. These models are less accurate than the direct method but require less time and labor and are cheaper to execute. Although satisfactory techniques are available to monitor soil erosion effects on crop productivity (i.e., the on-site damages), there is a lack of effective procedures to quantitatively evaluate sedimentation and water pollution impacts on the environment and human life (i.e., the off-site damages).

CONCLUSION

Soil erosion is a problem associated with human lives on earth. It threatens soil productivity, food production, the environment, and quality of life. Fortunately, erosion-induced problems have been well recognized, and sound control measures have been developed. To maintain the natural resources and environment for a sustainable civilization, soil erosion from agricultural lands must be controlled.

REFERENCES

1. Shumm, S.A.; Harvey, M.D. Natural erosion in the USA. In *Determinants of Soil Loss Tolerance*; Kral, M.K., Ed.; American Society of Agronomy, Soil Science Society of America: Madison, 1982; 15–22.
2. Hall, G.F.; Daniels, R.B.; Foss, J.E. Rate of soil formation and renewal in the USA. In *Determinants of Soil Loss Tolerance*; Kral, M.K., Ed.; American Society of Agronomy, Soil Science Society of America: Madison, 1982; 23–29.
3. Bennett, H.H. *Soil Erosion*; McGraw-Hill: New York, 1939.

4. Lowdermilk, W.C. *Conquest of the Land Through Seven Thousand Years*; Agricultural Information Bulletin 99; United States Department of Agriculture: Washington, 1953; 1–12.
5. Stallings, J.H. *Soil Conservation*; Prentice Hall: New Jersey, 1957.
6. Hudson, N.W. *Soil Degradation and Civilization*; Occasional Paper 9; Cranfield Institute of Technology: Cranfield, 1980; 1–13.
7. Dregne, H.E. Historical perspective of accelerated erosion and effect on world civilization. In *Determinants of Soil Loss Tolerance*; Kral, M.K., Ed.; American Society of Agronomy, Soil Science Society of America: Madison, 1982; 14 pp.
8. Langdale, G.W.; Schrader, W.D. Soil erosion effects on soil productivity of cultivated cropland. In *Determinants of Soil Loss Tolerance*; Kral, M.K., Ed.; American Society of Agronomy, Soil Science Society of America: Madison, 1982; 41–51.
9. Stocking, M. *Erosion and Soil Productivity—A Review*; Food and Agriculture Organization: Rome, 1984.
10. Crosson, P. Impact of erosion on land productivity and water quality in the United States. In *Soil Erosion and Conservation*; El-Swaify, S.A., Moldenhauer, W.C., Lo, A., Eds.; Soil Conservation Society of America: Ankeny, 1985; 217–236.
11. Lal, R. Soil erosion and its relation to productivity in tropical soils. In *Soil Erosion and Conservation*; El-Swaify, S.A., Moldenhauer, W.C., Lo, A., Eds.; Soil Conservation Society of America: Ankeny, 1985; 237–247.
12. Lal, R. Monitoring of soil erosion's impacts on crop productivity. In *Soil Erosion Research Methods*; Lal, R., Ed.; Soil and Water Conservation Society: Ankeny, 1988; 187–200.
13. Young, K.K. The impact of erosion on the productivity of soils in the United States. In *Assessment of Erosion*; DeBoodt, M., Gabriels, D., Eds.; John Wiley & Sons, Ltd.: London, 1980; 295–303.
14. Hudson, N. *Soil Conservation*, 3rd Ed.; Cornell University Press: New York, 1995.
15. Moldenhauer, W.C. Soil erosion—A global problem. In *Assessment of Erosion*; DeBoodt, M., Gabriels, D., Eds.; John Wiley & Sons, Ltd.: London, 1980; 3–8.
16. Krauss, H.A.; Allmaras, R.R. Technology masks the effects of soil erosion on wheat yields—A case study in Whiteman County, Washington. In *Determinants of Soil Loss Tolerance*; Kral, M.K., Ed.; American Society of Agronomy, Soil Science Society of America: Madison, 1982; 75–86.

Erosion: Fly-Ash Spheres as Time Marker

Kenneth R. Olson

Department of Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.

Alexander N. Gennadiyev

Faculty of Geography, Moscow State University, Moscow, Russia

Abstract

Fly ash, the particulate matter resulting from high temperature combustion of coal, was historically dispersed into the atmosphere. Fly-ash spheres that settled on the surface soil were from a variety of boilers, including those of steam locomotives and steam-powered farm machinery. In the Central United States and western Russia, fly-ash spheres provide a time marker starting between the 1850s and 1910s and ending between the 1950s and 1960s. At many locations, fly-ash generation coincided with the development of railroads, cultivation, and settlements. These fly-ash spheres are stable and remain in the soils for more than a hundred years. Tillage and erosion can redistribute and reduce the depth of occurrence and the amount of fly ash present on upland cultivated side slopes. It appears that fly-ash spheres were initially deposited uniformly within the local landscape even when there were slight variations in aspect, elevation, slope gradient, and distance from the source. Erosion phases of soils on upland landscape positions can be determined based on the amount of fly ash remaining in soil surface layers. Accelerated erosion of cultivated side slopes results in deposition of fly-ash rich sediment on the adjacent foot or toe slopes. The proposed fly-ash method provides a tool to assess the extent of soil translocation from a cultivated upland landscape and subsequent deposition.

INTRODUCTION

In the Central United States and western Russia, fly-ash spheres provide a time marker starting between the 1850s and 1910s and ending between the 1950s and 1960s. Fly ash, the particulate matter resulting from high temperature combustion of coal, was historically dispersed into the atmosphere. Fly-ash spheres that settled on the surface soil were from a variety of boilers, including those of steam locomotives and steam-powered farm machinery. At many locations, fly-ash generation coincided with the development of railroads, cultivation, and settlement. These fly-ash spheres are stable and remain in the soils for more than a hundred years. Tillage and erosion can redistribute and reduce the depth of occurrence and the amount of fly ash present on upland cultivated side slopes. It appears that fly-ash spheres were initially deposited uniformly within the local landscape even when there were slight variations in aspect, elevation, slope gradient, and distance from the source. Erosion phases of soils on upland landscape positions can be determined based on the amount of fly ash remaining in soil surface layers. Accelerated erosion of cultivated side slopes results in deposition of fly-ash rich sediment on the adjacent foot or toe slopes. The proposed fly-ash method provides a tool

to assess the extent of soil translocation from a cultivated upland landscape and subsequent deposition.

TEXT

Fly Ash

Fly ash is a particulate matter resulting from high temperature combustion of coal. It is produced in a variety of boilers, including those in steam locomotives and steam-powered farm machinery (Fig. 1). The principal minerals in the coal of the Central United States and western Russia are feldspars, pyrite, siderite, quartz, calcite, and mostly kaolinite.^[1] The minerals are vitrified above 1473 K, and a common product is spherules composed of glass, quartz, mullite, wustite, and magnetite.^[2] Until technologies allowed removal of fly ash from stack gases emanating from boilers, the ash was a component of smoke and was deposited over a wide area around the sources, especially near coal-fired power plants. Further, sphere occurrence and abundance have been used to identify sediments laid down during the industrial epoch.^[3] The occurrence of ferromagnetic minerals, usually embedded in the glassy phase, offers a convenient

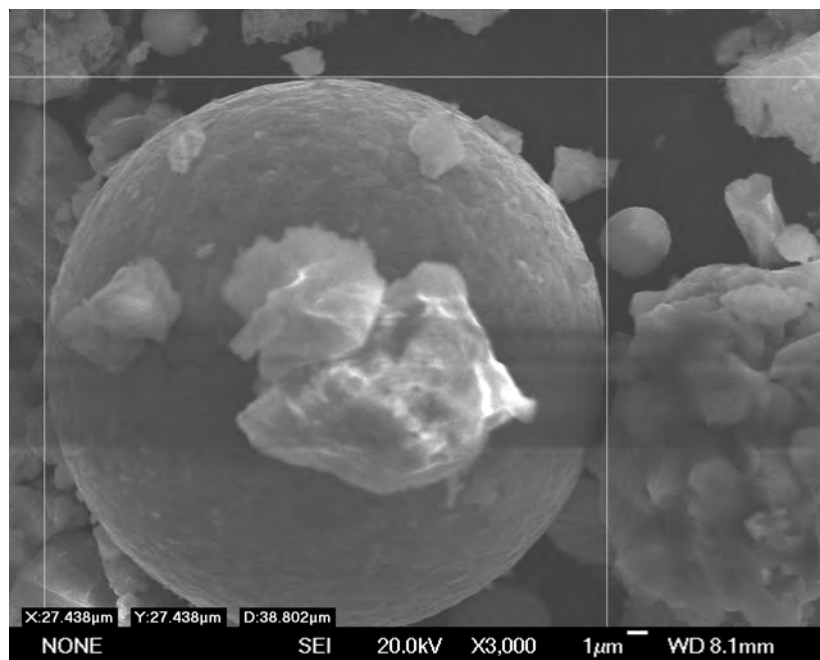


Fig. 1 Enlarged fly-ash sphere visible using a scanning electron microscope.

way to separate the spheres for identification and analysis. The use of fly ash and magnetic provides a time marker extending back between the 1850s and 1910s and is present in soils. The date coincides with Central United States and western Russia railroad development and initial period of American and Russian cultivation. Fly ash related to steam locomotive use was widespread by the 1920s in both countries. Some fly-ash spheres may have been derived from steam-powered farm equipment. Threshers and self-propelled tractors, the latter coming into greatest use from 1880 to about 1920,^[4] could have produced magnetic fly ash if their boilers were fed by soft (bituminous) coal. For economic and supply reasons, however, many steam boilers on farms were fired with straw or wood that probably could not produce highly siliceous, glass-phase fly-ash spheres.

Jones and Olson^[5] reported that fly-ash spheres were present in large amounts in soils and sediments near urban areas and in smaller quantities in rural locations within Illinois. Between the 1850s and present, every Illinois county had a railroad source of fly ash within or adjacent to the county. Fly ash had the potential of being an easily identified time marker in sedimentation studies. These airborne fly-ash spheres were first deposited on soils and sediments between the 1850s and 1910s, the time of settlement and the use of high-temperature coal-fired boilers, which are also nearly contemporaneous with initial American cultivation in Central United States. Fly-ash spheres were generated until the 1950s or 1960s when coal-fired locomotives were replaced. Fly ash present in soils can be used to identify sediments accumulated since Euro-American settlement and to interpret the stratigraphy of sediments and their relationships to the underlying soils.

The incidence of fly ash in soil profiles^[6–8] was aided by a study of carefully chosen sites near sources of the fly ash, e.g., undisturbed areas on Mississippi mounds at archaeological sites that were adjacent to and downwind from railroads. As some of the fly-ash particles should be in the silt- and sand-size range and not prone to eluviation, their distribution in the soil provides valuable information regarding biotic and abiotic mixing and pedoturbation of surface horizons since settlement. Knowledge of mixing characteristics helps interpret evidence of erosion or accretion in soil profiles as well as the time frame for these events.

Analytical Methods

The fly-ash sphere content was determined using procedures developed by Jones and Olson.^[5] A soil sample weighing 10 g was oxidized with sodium hypochlorite. Magnetic particles were separated by suspending this sample in distilled water and circulating it over a magnet wrapped in plastic film. The magnetic minerals were collected by removing the film from the magnet and washing the adherent particles into an evaporating dish. Extraction was repeated until no more particles were removed. This extracted sample was dried and weighted. The coefficient of variation for three duplicates was 6.6%.

The magnetic particle sample was then wetted with a minimum of water and mixed; a drop was transferred to a cover glass, dried, and carefully mounted with Canada balsam for petrographic microscope study (with strong oblique lighting). This type of mount facilitates scanning by presenting a flat field of magnetic particles on the cover glass bottom. The spheres were easily discerned

among the anhedral (or occasionally subhedral) opaque minerals, which were mostly magnetite, ilmenite, and leucoxene.

Fly ash was determined microscopically by choosing a random field using $\times 400$ – 1500 magnification (Fig. 2). A digital image of the field was made, and this image was analyzed for spherical particles using ImagePro image analysis software (Media Cybernetics, Silver Spring, Maryland, U.S.A.). Each spherical particle was identified by the microscopist and counted as fly ash. The metric reported an area of the fly ash as a percentage of the total area of opaque particles that were assumed to be ferrimagnetic. At least six fields were counted. The abundance of fly-ash spheres in the magnetic fraction was estimated, and each sample was placed in a category of <1.1 – 5% , 6 – 25% , 26 – 75% , or $>75\%$.

In order to compare sites and predict soil loss and gain from soil profiles, fly-ash sphere content needs to be determined and expressed on a volume basis. The fly-ash content (g m^{-2}) remaining on cultivated landscape positions can then be compared with the fly-ash content of the similar uncultivated landscape positions. The percent loss of the fly-ash content becomes an indicator of the percent reduction in the previous 20-cm soil surface layer of the cultivated and eroded landscape positions. The weight of the uneroded 20-cm thick surface layer on 1 ha can then be determined using measured bulk density and multiplied by the percent loss of the fly-ash content to determine the weight of the surface layer removed by erosion. Total weight of soil surface removed (Mg ha^{-1}) was divided by the number of cultivated landscape to determine erosion rate ($\text{Mg ha}^{-1} \text{yr}^{-1}$).

Cesium-137 activities were obtained by measuring the activity at 661.66 keV, using high-resolution spectrometric equipment, based on hyperpure germanium coaxial y-ray detector BDEG-30.195 with energy resolution 1.95 keV at energy line of 1332 keV and 30% efficiency. Count times were not less than 12 hours and were dictated by the appearance of well-shaped cesium-137 peaks.

Use of Fly Ash as Time Marker in Erosion Studies

In Southern Illinois, Hussain et al.^[9] observed higher amounts of fly ash in the A horizon for all landscape positions at an uncultivated site than at a paired cultivated site. These finding indicated the removal by cultivation and erosion of approximately 47% or 10.6 cm of the original surface layer in 142 years. Fly ash represented a better indicator of soil erosion than organic carbon (C), magnetic minerals, or magnetic susceptibility. Olson et al.^[8] used content of fly ash as an indicator of soil erosion at a site near Moscow, Russia. Replicated transects in cultivated and reforested sites were sampled and analyzed. The entire cultivated landscape contained 12% lower fly-ash content and magnetic susceptibility than the reforested site. These results suggested the loss of 2.4 cm of original A horizon and placed the soil in the slightly eroded phase.

The use of the fly-ash method made it possible to estimate the rates of erosion of plowed soils on southern and northern slopes. The uneven intensity of snow melting on southern and northern slopes induces uneven snowmelt runoff, so that the rate of erosion on the relatively warm south and west facing slopes was higher than that on the north facing slopes. Soil thawing on the slopes of southern

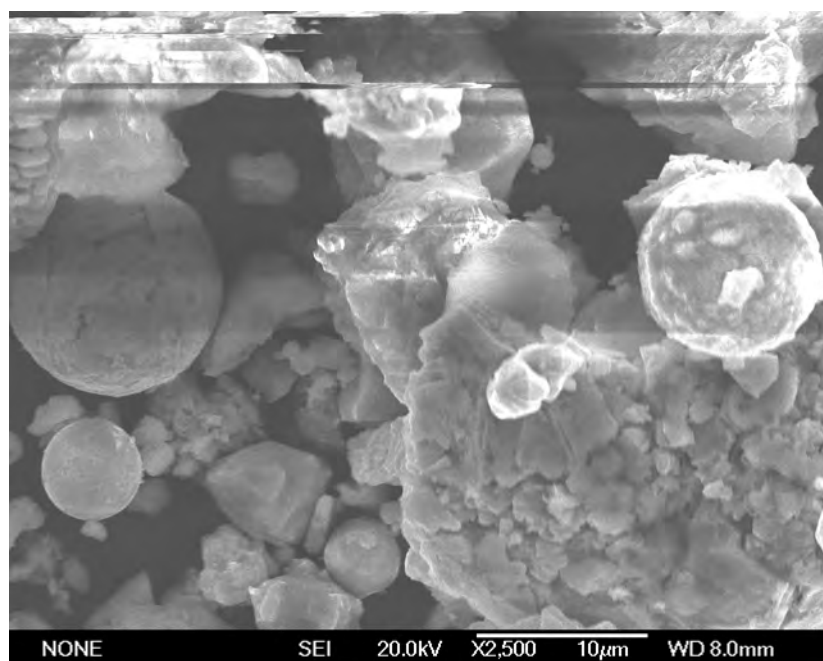


Fig. 2 Fly-ash field visible using a scanning electron microscope.

aspects is more active, and the duration of the period when the thawed soils are unprotected from erosion is larger on south facing slopes. On the basis of fly-ash method, the intensity of erosional processes on the slopes of different aspects at three key plots of Central Russia has been studied.^[10] The maximum steepness of the slopes at these plots was about 6°, 5°, and 3°, respectively. For leached and podzolized Chernozems (Argiudolls) of the Aleksandrovka plot, they reached 11 Mg ha⁻¹ yr⁻¹ on the southern slope and 2 Mg ha⁻¹ yr⁻¹ on the northern slope; for typical and leached Chernozems (Hapludolls) of the Gracheva Loshchina plot, 10 and 4 Mg ha⁻¹ yr⁻¹, respectively; and for typical and leached Chernozems (Hapludolls) of the Tolmachi plot (with the minimum slope), 6 and 4 Mg ha⁻¹ yr⁻¹, respectively. Thus, on all the investigated plots, soil erosion on the south facing slopes was more intense than that on the north facing slopes. In general, the difference between the rates of erosional processes on the slopes of northern and southern aspects for plowed soils was well pronounced on all the plots.

The fly-ash method was used for the quantification of the lateral migration of the solid-phase products of the pedogenesis within the catchment basins.^[11] This method allowed separating the spatial dispersion and accumulation zones of the soil material, which is genetically conjugated with one another and represents a component of the migration structure of the catchment basin. It was revealed that the manifestations of the dispersion and accumulation zones in the catchment area are of multifactor nature: the zones can be located on the upper, middle, and lower parts of the slopes; on the near watershed (near ridge) surfaces; and on the slopes of different exposures. The second component of the migration structure of the catchment basin included the formalized system of soil catenas within the territory. They can be classified with consideration for: 1) the transport rate of the soil material within the catenas; 2) the degree of their openness (i.e., the assessment of the portion of material removed beyond the catena limits); and 3) the localization of the zones of the soil material's accumulation on the slope.

Impact of Land Use, Erosion, and Landscape Position on Soil Organic Carbon (SOC) Stocks

Olson et al.^[12] found that the dynamics of SOC are affected by many factors including land use, management history, soil type, climate, and soil landscape processes. They compared the storage of SOC on sloping woodland and cropland landscapes of Northwestern Illinois. Soil erosion and sediment transport resulted in some SOC being redeposited in lower landscape positions but retained in the landscape. Results suggested that the cropland landscape retained 52% of the total SOC on a volumetric basis during cultivation, soil erosion, and agricultural use from 1860 to 2010. The other 48% of the SOC was either deposited in the water or released to the atmosphere.

Olson et al.^[13] determined the SOC level in a native timberland of west Central Illinois and then compared with the SOC levels in adjacent cropland and pastureland after agricultural use from 1860 to 2010. Results suggested that the cropland landscape retained 59.7 Mg C ha⁻¹ (87%) of the total SOC in the native timberland on a volumetric basis during cultivation and cropland use from 1860 to 2010. Soil erosion and sediment transport measured using the fly-ash method resulted in SOC being deposited in lower landscape positions but retained in landscape. The other 13% net loss of the SOC storage was either deposited in the water or released to atmosphere. The conversion of timberland to cropland increased soil erosion and reduced SOC storage, and any conversion to pastureland increased SOC storage and retention.

Use of Fly Ash and Cesium-137 as Time Markers in Transport Studies

Olson et al.^[14] used a radio-caesium (cesium-137) and a technogenic magnetic tracer (magnetic fly-ash spheroids) methods for the quantitative assessment of erosion, transport, and deposition processes. Both cultivated and uncultivated sites located near Springfield, Illinois, were compared. The fly-ash and cesium-137 methods did provide consistent results for six of eight cultivated and uncultivated landscape positions when significant deposition or erosion did not occur before the 1960s. The lower side slope site with a woodland/pastureland land use had significant erosion before the 1960s (reflected in fly-ash data) and eroded after cultivation in the 1960s. The relevant time period varied for both the fly-ash and cesium-137 methods. The fly-ash method assessed the soil stability, soil erosion, transport, and sediment deposition situation from 1860 to 2010, whereas the cesium-137 method assessed the soil stability, soil erosion, transport, and sediment deposition situation only from 1960 to 2010.

Olson et al.^[15] used both the fly-ash and cesium-137 methods to determine the amount and annual rate of soil erosion from 1960 to 2010 and from 1910 to 2010 time periods at a Knoxville, Illinois, site. Initially, the fly ash assessed the soil stability, soil erosion, and sediment deposition situation from 1910 to 2010 (based on time of coal-fired locomotives), whereas the cesium-137 method only assessed the soil stability, soil erosion, and sediment deposition situation from 1960 to 2010. After 1910, fly ash from coal-fired locomotives was deposited on the cropland. Using the 1910–2010 time period for the fly-ash annual erosion rate calculation and the 1960–2010 time period for cesium-137, the predicted annual erosion rates would be similar. However, the total amount of erosion was higher for the fly-ash method due to longer time period. The fly-ash and cesium-137 methods did provide consistent annual soil erosion rate results for five of five landscape positions on the cropland landscape positions even if the cesium-137 method uses the 1960–

2010 time period and the fly-ash time period was from 1910 to 2010. These combined method annual erosion rates ($0.7\text{--}4.0\text{ Mg ha}^{-1}\text{ yr}^{-1}$) for either from 1960 to 2010 or from 1910 to 2010 time periods are below the tolerable soil loss rates for the slightly eroded Rozetta and Hickory soils ($11\text{ Mg ha}^{-1}\text{ yr}^{-1}$).

The combined use of the fly-ash method and the radio-cesium method allowed assessing the spatial-temporal pattern of the sediment redistribution on typical slopes in different parts of the forest steppe zone of the East European Plain.^[16] A clear trend of the decreasing of soil loss rate in this zone from 1985 or 1990 to 2010 was revealed compared with the average rate 1860 or 1870 to 2010 period of plowing. The absolute values of the soil loss well agree with the data for the soil erosion rate in the forest steppe zone obtained using the conventional methods and approaches for assessing the intensity of the soil erosion. The obtained results confirmed the continuous nature of the soil erosion and accumulation during the transport of the sediments along the slope, which results in the alternation of the erosion zones and redeposition zones on the slopes.

Use of Fly Ash as a Time Marker in Depositional Studies

Kreznor et al.^[17,18] quantified the postsettlement deposition in a 2.49-ha sediment basin with a single outlet in a 10.54-ha watershed and estimated the sediment delivery to a first-order stream in Northwestern Illinois. Buried A horizons (dated using fly ash as a time marker) identify the presettlement (1854) surface, which was overlain by as much as 116 cm of sediment. Volume of the sediment within the basin was calculated at 34 m^3 with a mass of 16,480 Mg. The modern soils of the sediment basin were characterized and classified, and the spatial variability of the sedimentation process was examined. Based on representative measurements of postsettlement sediment delivery obtained from research of drainage basins having similar size or soil characteristics, it was inferred that 20,975 Mg of sediment was delivered to the stream and with a total of 37,455 Mg of sediment being removed from the watershed hillslopes as a result of accelerated soil erosion. The measured rate of post settlement sediment accumulation was approximately 0.34 cm yr^{-1} .

Olson et al.^[19] studied a nearly level plot area near Monmouth, Illinois, with Muscatune and Sable soils. The fly-ash method was used to determine the extent of erosion as a result of cultivation between 1880 or 1910 to 2010. The fly-ash rich sediment being transported to and deposited on the adjacent Sable soil in the sample plot area was measured. For the 0–50 cm layer, the Sable soil fly-ash content is 19% higher than the Muscatune soils. The erosion rate, using the fly-ash method, was $4.35\text{ Mg ha}^{-1}\text{ yr}^{-1}$. Approximately, 6 cm of the total difference of 21 cm between Sable and Muscatune A horizons thicknesses was

a result of erosion of the Muscatune soil and deposition on the Sable soils.

CONCLUSION

Fly-ash sphere method can be used as a time marker in erosion, transport, and depositional studies. The soil erosion rates and depositional rates can be measured for from 1910 to 2010 or 1860 to 2010. Erosion phases of soils can be determined on upland landscape positions based on amount of fly ash remaining in soil surface layers. The fly-ash method has been used to determine the effects of aspect on soil erosion rates and to quantify lateral migration of solid-phase products within catchment basins. The sediment depositional rate can be measured using the presence or absence of fly-ash spheres in buried soil layers. The effect of land use change, erosion, transport, and deposition on SOC stock can be determined by landscape position and weighted for the whole landscape in each land use.

REFERENCES

1. Harvey, R.D.; Ruch, R.R. Mineral matter in Illinois and other U.S. coals. In *Mineral Matter and Ash in Coals*; Vorres, K.S., Ed.; American Chemical Society: Philadelphia, 1984; 10–40.
2. Huffman, G.P.; Huggins, F.E. Reactions and transformations of coal mineral matter at elevated temperature. In *Mineral Matter and Ash in Coals*; Vorre, K.S., Ed.; American Chemical Society: Philadelphia, 1984; 100–113.
3. Locke, G.; Bertine, K.K. Magnetite in sediments as an indicator of coal combustion. *Appl. Geochem.* **1986**, *50*, 345–356.
4. Wik, R.M. *Steam Power on the American Farm*; University of Pennsylvania Press: Philadelphia, 1953.
5. Olson, K.R.; Jones, R.L. Use of fly ash as a time marker in sedimentation studies. *Soil Sci. Soc. Am. J.* **1990**, *54*, 855–859.
6. Olson, K.R.; Gennadiyev, A.N.; Jones, R.L.; Chernyanskii, S. Erosion patterns on cultivated and forested hillslopes in Moscow Region, Russia. *Soil Sci. Soc. Am. J.* **2002**, *66*, 193–201.
7. Olson, K.R.; Jones, R.L.; Gennadiyev, A.N.; Chernyanskii, S.; Woods, W.I.; Lang, J.M. Accelerated soil erosion of a Mississippian mound at Cahokia site in Illinois. *Soil Sci. Soc. Am. J.* **2002**, *66*, 1911–1921.
8. Olson, K.R.; Jones, R.L.; Gennadiyev, A.N.; Chernyanskii, S.; Woods, W.I.; Lang, J.M. Soil catena formation and erosion of two Mississippian mounds at Cahokia archaeological site, Illinois. *Soil Sci.* **2003**, *168*, 812–824.
9. Hussain, I.; Olson, K.R.; Jones, R.L. Fly ash and erosion patterns on cultivated and uncultivated hillslopes in Southern Illinois. *Soil Sci.* **1998**, *163*, 726–728.
10. Gennadiyev, A.N.; Zhidkin, A.P.; Olson, K.R.; Kachinskii, V.L. Soil erosion under different land uses: Assessment by the magnetic tracer method. *Eurasian Soil Sci.* **2010**, *43* (9), 1047–1054.

11. Gennadiyev, A.N.; Koshovskii, T.S.; Zhidkin, A.P.; Kovach, R.G. Lateral migration of soil solid-phase material within a landscape-geochemical arena detected using the magnetic tracer method. *Eurasian Soil Sci.* **2013**, *46* (10), 983–993.
12. Olson, K.R.; Gennadiyev, A.N.; Zhidkin, A.P.; Markelov, M.V. Impact of land use change and soil erosion in Upper Mississippi River Valley on soil organic carbon retention and greenhouse gas emissions. *Soil Sci.* **2011**, *176* (9), 449–458.
13. Olson, K.R.; Gennadiyev, A.N.; Zhidkin, A.P.; Markelov, M.V. Impact of land use change, slope and erosion on soil organic carbon retention and storage USA. *Soil Sci.* **2012**, *177* (4), 269–278.
14. Olson, K.R.; Gennadiyev, A.N.; Golosov, V.N. Comparison of radioactive and magnetic tracer methods to assess soil erosion and deposition in Illinois landscapes. *Soil Sci.* **2008**, *173*, 75–586.
15. Olson, K.R.; Gennadiyev, A.N.; Zhidkin, A.P.; Markelov, M.V.; Golosov, V.N.; Lang, J.M. Use of magnetic tracer and radio-caesium methods to determine past cropland soil erosion amounts and rates. *Catena* **2013**, *104*, 103–110. Available at <http://dx.doi.org/10.1016/j.catena.2012.10.015> (accessed December 2013).
16. Golosov, V.N.; Gennadiyev, A.N.; Olson, K.R.; Markelov, M.V.; Zhidkin, A.P.; Chendev, Yu.G.; Kovach, R.G. Spatial and temporal features of soil erosion in the forest steppe zone of the East European Plain. *Eurasian Soil Sci.* **2011**, *44* (7), 794–801.
17. Kreznor, W.R.; Olson, K.R.; Jones, R.L.; Johnson, D.L. Quantification of postsettlement deposition in a Northwestern Illinois sediment basin. *Soil Sci. Soc. Am. J.* **1990**, *54*, 1393–1401.
18. Kreznor, W.R.; Olson, K.R.; Johnson, D.L. Field evaluation of methods to estimate soil erosion. *Soil Sci.* **1992**, *153*, 69–81.
19. Olson, K.R.; Gennadiyev, A.N.; Kovach, R.G.; Lang, J.M. The use of fly ash to determine the extent of sediment transport on nearly level western Illinois landscapes. *Soil Sci.* **2013**, *178* (1), 24–28.

Erosion: Global Change

Xingxiang Wang

Taolin Zhang

Longxi Cao

Yin Liang

Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China

Abstract

China is one of the nations that is most severely affected by soil erosion in the world. The land area affected by accelerated soil erosion was estimated to be 295 Mha in 2010, and the average annual soil loss due to soil erosion is estimated to be 5000 million tons. On a global scale, the total land area affected by soil erosion is 1643 Mha, and the annual soil loss due to accelerated soil erosion is estimated to be 75 billion tons. As the most serious and widespread form of soil degradation and an important form of global change, soil erosion severely threatens the quality of soil, air, and water resources and hence menaces agricultural development, ecological civilization, and human well-being. Soil erosion and global climate change are related by bidirectional interaction, and soil erosion is likely to significantly increase under global climate change unless effective amelioration measures are taken in time. Therefore, it is critical to take scientific and effective actions against soil erosion.

INTRODUCTION

Soil erosion is the movement and transport of soil by various agents, particularly by water and wind, which lead to a loss of soil. Usually, soil erosion takes place very slowly in natural ecosystems, but it can occur at accelerated rates in areas where landscapes are disturbed improperly by human activities. Global change has increasingly become a great concern since it was put forward by the International Council for Science in 1986. Global change, as a result of physical, chemical, and biological processes in the earth's system—particularly of interactions between humans and the environment—refers to the changes in global environmental components that are vital to human survival, such as climate, land, water, and air, in the forms of greenhouse effect/global warming, diminution of forest, loss of biodiversity, degradation of land, and deficiency in water resources. These global changes represent the major global challenges the humankind are facing. Soil erosion is an important form of global change and severely threatens the quality of soil, air, and water resources, hence undermining agricultural development, ecological civilization, and human well-being. Soil erosion and other forms of global changes are related by bidirectional interactions, and notably, global climate change may exacerbate soil erosion. Although it is a daunting task, soil conservation has been greatly advanced through effective soil and water conservation measures in China and other regions of the world.

THE CONTRIBUTION OF SOIL EROSION TO MAJOR GLOBAL CHANGES/CHALLENGES

Extent of Soil Erosion in China and in the World

China is one of the nations that is most severely affected by soil erosion in the world. The land area affected by accelerated soil erosion was estimated to be 295 Mha in 2010, including 129 Mha by water erosion and 166 Mha by wind erosion (Table 1).^[1] The average annual soil loss due to soil erosion is estimated at 5000 million tons, including 2350 million tons from the watershed of Yangtze River and 1581 million tons from the watershed of Yellow River.^[2] The region most severely affected by soil erosion in China is the watershed of Yellow River (Fig. 1), with an erosion area of 40.2 Mha and an erosion modulus of $4840 \text{ t km}^{-2} \text{ yr}^{-1}$, especially in the Loess Plateau with an erosion modulus as high as $30,000 \text{ t km}^{-2} \text{ yr}^{-1}$ and an annual soil loss of 1.63 billion tons.^[2] Fortunately, about 24% of erosion area in the Loess Plateau has been controlled since the 1980s.^[3]

On a global scale, total land area affected by water erosion is 1094 Mha, of which 751 Mha is severely affected, and that by wind erosion is 549 Mha, of which 296 Mha is severely affected (Table 2).^[4–6] Worldwide, the annual soil loss due to accelerated soil erosion is estimated at 75 billion tons.^[7] Most of the eroded area is distributed in Africa and Asia, accounting for 64.7% of water erosion area and 63.5% of wind erosion area in the world (Table 2).

Table 1 Area affected by accelerated soil erosion in China.

Erosion grade	Water erosion (Mha)	Wind erosion (Mha)
Slight	66.76	71.60
Moderate	35.14	21.74
Severe	16.87	21.82
Very severe	7.63	22.04
Extremely severe	2.92	28.39

Soil Erosion Threatens Global Food Security through Degrading Soil Quality

Soil is one of the most important natural resources and a major factor in global food production. Soil erosion is widely considered as the most serious form of soil degradation.^[8] When soil erosion rate exceeds soil development rate, it would lead to soil degradation and pose a significant threat to world's food production capacity and global food security. Approximately, one-third of the world's agricultural land has been affected by soil degradation in the past years, and most of this damage is attributed to water and wind erosion. It is reported that the soil erosion rate is 1–2 magnitude orders greater than the soil development rate on conventional plow-based agricultural lands.^[9] The average rate of soil loss in agricultural areas has been estimated at $17 \text{ t ha}^{-1} \text{ yr}^{-1}$ in the United States and Europe and $30\text{--}40 \text{ t ha}^{-1} \text{ yr}^{-1}$ in Asia, Africa, and the Southern United States, mainly due to inappropriate land use and management.^[7] It is estimated that the annual global topsoil loss from agricultural lands due to water erosion is 20–30 billion tons.^[10]

Soil erosion reduces area of arable land and leads to depletion of nutrients. Increases in the vulnerability of soils

Table 2 Area affected by accelerated soil erosion in the world.

Region	Area affected by severe erosion (moderate + level; Mha)	
	Water erosion	Wind erosion
Africa	169	98
Asia	317	90
Europe	93	39
Central America	45	5
North America	46	32
Oceania	4	16
South America	77	16
World	751	296

to erosion suggest that more organic matter and mineral nutrients will be lost from soils, leading to a decline in ecosystem-sustaining characteristics such as water-holding capacity, plant productivity, and biodiversity.^[9] The process has exerted a profound effect on the balance of world food supply, although the on-site effects of soil erosion on crop productivity are easily minimized through additional external agricultural inputs and improved agricultural technologies. Soil erosion and its effects on crop production have been extensively investigated since the turn of the 20th century. The tolerable erosion rates have been quantified to describe the maximum rate of soil erosion that can permit crop productivity to be sustained economically and maintained indefinitely. The negative effect of erosion on crop productivity can be assessed using field runoff plots or paired watersheds. The actual loss may depend on weather conditions during growing season, farming systems, soil management, and soil ameliorative inputs. Simulation models have been developed to predict the effects of erosion on soil properties and crop

**Fig. 1** Landscape of severe soil erosion in the Loess Plateau of China.

responses.^[6] By linking with geographic information system tools, these models can be expanded to the regional, national, and global scales, and the dynamics of agricultural management can be studied.^[11,12] According to the loss rate, 9933 km² of surface black soil (mollic epipedon) on agriculture land will be lost, and crop yield will decrease by 40% in the northeast China in 50 years if no effective countermeasures are implemented.^[13] Loss estimates of partial global food suggest that annually farmers lose about 2.2 million tons of maize, 0.07 million tons of millet, 1.1 million tons of potatoes, 0.1 million tons of soybeans, and 1.1 million tons of wheat, which may be more severe in Asia, sub-Saharan Africa, and other tropics.^[8] Furthermore, as a driver of land use change, soil erosion will induce abandonment of arable land due to declining productivity^[14] and exacerbate poverty and hunger since the poverty and soil erosion are related by bidirectional interaction; generally, the poorer areas are also the areas affected by severe soil erosion.^[13]

Soil Erosion Deteriorates Global Environment and Ecological Civilization

Soil erosion deteriorates the global environment through flood disasters, environmental pollution, desertification, etc. It aggravates flood disasters through raising riverbeds, silting lakes, and shrinking water bodies. On a global scale, about 0.5–1.0% of water storage is lost annually as a result of sedimentation.^[15] The economic cost of reservoir storage losses is reported more than \$13 billion, excluding other environmental and social costs incurred from additional dam development.^[16] In China, flood disasters increased from once per 20 years 1300 years ago to once per 1.6 years in the 1990s in the Yangtze River Basin due to soil erosion. Riverbed in lower reaches of Yellow River was raised by 8–10 cm per year due to soil erosion.^[1]

Soil erosion is also an important non-point source of pollution. With enrichment of nutrients, heavy metals, and pesticides in sediments and a large amount of rapidly available nutrients in surface runoff, water erosion can cause serious damage to the water quality and aquatic ecosystems in receiving waters. In contrast, wind erosion can have an adverse effect on transportation, communication, and human health through increasing dust in the air, being most adverse in the case of dust storm, which is the most intensive and powerful form or consequence of wind erosion and recognized as a disastrous weather. The most well-known dust storm are those that swept over the southern areas of the United States in the 1930s and that have afflicted Northern China including Beijing, the capital of China. The off-site damage from wind erosion in the United States is estimated to be nearly \$10 billion each year.^[7] It is reported that about 20 human infectious disease microorganisms, such as anthrax and tuberculosis, be present in the soil particles transported by wind.^[17] In addition, soil erosion by water or wind can be a driving force to soil desertification; e.g. the area of rocky desertification

resulted from soil loss due to water erosion is estimated to be doubled in the karst region of southwest China in 35 years, if no scientific and effective actions are taken against the soil erosion in the region.^[13]

Soil erosion influences global climate change by changing the cycles of carbon, nitrogen, and water. The global soil carbon pool is about 2500 Pg, 3.3 times the size of the atmospheric pool and 4.5 times the size of the biotic pool.^[18]

Small changes in the pools of carbon and nitrogen in soils could have large effects on atmospheric concentrations of carbon dioxide (CO₂), methane, and nitrous oxide. Soil erosion has a profound impact on the quality and quantity of soil organic matter (SOM) and can be a significant factor in losses and redistribution of local carbon reservoirs. It is estimated that 0.8–1.2 Pg C emission is added to the atmosphere annually due to global soil erosion,^[6] which contributes to the global warming problem. The effect of water erosion would become more important in the global carbon budget, because it will be strongly influenced by climatic change and land use in many regions.^[19] However, it is uncertain whether the increased sediment yield results in a net carbon sink into passive carbon pools or a net source into the atmosphere as a result of mineralization of labile carbon.^[6]

Soil erosion may cause a shortage of freshwater through deteriorating soil physical structure, increasing surface runoff while less water entering the soil, and changing global water cycling. Compared with uneroded soils, moderately eroded soils retain 10–300 mm less water per hectare per year from rainfalls. This represents a decrease of 7–44% in available water for plant growth and most likely results in water shortage for crops in farmlands.^[20] In China, the annual average of drought-affected areas has reached 22.3 Mha during the period of 2000–2012, most of which are located in the areas severely eroded.^[1,21]

As mentioned above, soil erosion leads to a decrease in overall biomass and productivity by degrading soil quality. Ultimately, this has a profound effect on the diversity of plants, animals, and microbes in the entire ecosystems based on eroded soils. In areas where erosion rates are 10–20 times higher than soil formation rates, the diversity and abundance of soil organisms are significantly decreased. On the contrary, when biomass is increased due to soil conservation remedies, the number of species can be increased appreciably. It is reported that a 10-fold increase in bird diversity was seen with a 100-fold increase in plant biomass production.^[20] In addition, soil erosion has indirect effects on ecosystems as damaging the effects of decreasing plant biomass productivity. For example, a grassland with reduced plant richness due to soil erosion is more susceptible to drought than the one with an abundance of plant species.^[22] Sometimes, soil erosion may cause the loss of a keystone species and then lead to cascading effects on the survival of a wide range of other species within the ecosystem.

Through degrading soil quality, soil erosion may cause changes to landscapes and ecosystems on local, regional, and global scales, altering the delivery of ecosystem

services, microclimates, and even the global climatic pattern and ultimately jeopardizing the construction of ecological civilization.

Global Climate Change May Accelerate Soil Erosion

The main factors controlling soil erosion are the erosivity of eroding agents (rainfall and wind), the erodibility of soil, the slope of land (slope steepness and length), and the nature of plant covers. Erosion processes and their influential factors vary according to the studied scale. Therefore, soil erosion forecasting is associated with time and space scale, from the field scale at which it affects the individual farmer, up to national or global scales where it can make an important contribution to decision or policy makers. Since the 1950s, significant advances have been made in predicting erosion risks under different conditions, particularly with the development of modeling technology.^[23] However, the response of soil erosion to global climate change is complex temporally and spatially due to uncertainties in general circulation models and the complex relationships of soil erosion processes with climate, land use, and land cover.

Changes in greenhouse gases concentration and climate will change the hydrologic cycle and hence affect the soil–plant–water interactions, which in turn affect soil erosion. Detachment and transport of soil by water depend on the amount and intensity of rainfall, which is directly affected by changes in climate. Results from modeling show that rainfall erosivity is projected to increase significantly with the projected climate change.^[12] The responses obtained by models such as LISEM, MEFIDIS, RUSLE, STREAM, KINEROS, SWAT, and WEEP to rainfall change indicate that soil erosion is likely to increase significantly under climate change unless amelioration measures are taken.^[24] Average annual runoff rates are expected to increase by 30% to 40% in high latitudes and decrease by 10% to 30% in arid and semiarid regions prone to drought stress.^[25] Increases in air temperature may lead to faster rates of snow melting, which can result in higher snowmelt runoff. The predicted climate change is expected to increase the risks of soil erosion, and the magnitude of this expected increase in water and wind erosion risks will most likely depend on local and regional conditions.^[26,27] In some regions, any increase in rainfall (e.g., amount, intensity, and frequency) as a result of climate change may directly exacerbate erosion. On the contrary, in other regions, a decrease in rainfall may be expected. In this case, a decline in soil moisture because of the enhanced evaporation will occur. Drier soil conditions and less vegetation can make soils more vulnerable to wind erosion. Moreover, soil erosion tends to be dominated by extreme events. An increase in the possibility of extreme rainfall events, which is suggested by several climatic change models, may accelerate soil erosion.

Global climate change may influence soil erosion through changing temporal and spatial evolution of global

soil cover, such as changes in soil-forming processes and soil properties associated with soil erodibility. The response of soil to the erosion process is complex and closely related to soil properties such as texture, structural stability, organic matter content, clay mineralogy, and chemical constituents. All these factors are likely to change with climate and/or land use. The increasing air temperature may accelerate the SOM decomposition rate, severely deplete the SOM reserves, and increase soil erodibility.^[28] A decline in SOM level would cause a decrease in soil aggregate stability, lower infiltration rates, increase in runoff, and then increase in the likelihood of soil erosion. The increased atmospheric CO₂ concentrations will enhance plant growth. However, the beneficial effects of high CO₂ concentrations on biomass production may be offset by increasing air temperature and reduced precipitation in dry regions.^[12] The change in the magnitude of soil erosion risk with the reduction in SOM is a clear manifestation of the impact of global warming on soil erosion risk. Fortunately, the impacts of SOM on climate change and food security have been concerned, and some strategies such as less till and no-till farming systems to increase SOM have been proposed.^[18]

Major changes in wind erosion rate driven by climate change would be associated with local or regional changes in soil moisture and vegetation. It is reported that wind erosion will increase on grazing lands of the Southwestern United States because of increased aridity and associated reductions of vegetation cover.^[29] Increased wind erosion is also likely to increase the incidence of wildfires, which in turn will increase wind erosion rates as well as water erosion rates due to the drastic reductions in ground cover after wildfires.^[30] Model projections predict the trends of decreasing wind speed in some regions, likely leading to a decrease in evapotranspiration in cropping regions and the potential for wind erosion.^[31]

Sea level change is also an important item in global change as a result of global warming. A research indicates that more coastal erosion is triggered by sea level rise,^[32] e.g., annual beach sediments loss was $1.67 \times 10^4 \text{ m}^3$ due to sea level rising in the southern Shandong sandy coast in China.^[33]

Soil and Water Conservation in China under Global Climate Change

The severe consequences and grim projections of soil erosion' impact on global food security and environmental and ecological sustainability as described earlier emphasize the need to implement soil conservation strategies and practices. The basic principle underlying all these strategies and practices is to keep the land protected from wind and water energy by constructing some form of vegetative cover on the land. An array of soil conservation techniques has been developed for various erosion types under different natural and economic conditions. In China, the national strategies for soil erosion control are as follows: 1) prioritizing



Fig. 2 Severe soil erosion in the sparse Masson pine forest in subtropical China.

prevention and conservation measures; 2) promoting integrated small watershed management with focus on sloping croplands and gully-eroded areas; 3) ensuring closed/forbidden areas from disturbance for ecological recovery on their own; 4) planning goals and measures specific for individual target zones according to local conditions; and 5) strengthening leadership by establishing comprehensive management mechanisms. Since 1978, China has launched some of the world's most large-scale conservation programs, including the Three Norths Shelter Forest Project, Grain for Green Project, and the Natural Forest Conservation Program, which together cover almost entire mainland China except five eastern coastal provinces.^[34] About 30 years after these programs were implemented, the environment and society in project areas have been improved greatly as indicated by more acreage of green lands and less population in poverty, avoiding the vicious cycle in which poverty leads to environmental damage, which in turn deepens poverty.^[34] According to the meteorological observation in Beijing, the capital of China, dusty weather event including dust storm has shown a trend of decrease in frequency and intensity since 2000, which is coincided with the progress in vegetation restoration associated with these ecological programs within Beijing and its neighboring areas. However, strong dust storm still outbreaks now and then in northern areas of China including Beijing. Nearly 290,000 km² of sandy lands in the northern areas of China remain to be greened up, some of which is the dust source of dust storm outbreaks in Beijing.

The land area affected by soil erosion decreased by 4900 km² annually in the humid South China due to afforestation and soil and water conservation activities. However, unreasonable vegetation construction cannot effectively control soil erosion; e.g., severe soil erosion was observed in the sparse Masson pine forest in subtropical China (Fig. 2).^[35]

In agricultural areas, where soil erosion is usually a severe problem, if production systems are integrated with appropriate conservation management systems, the adverse effects of climate changes such as increased precipitation amounts and intensities on soil erosion can be alleviated. The major conservation practices or technologies include the use of crop rotation, less till and no-till, biomass mulches, cover crop, contour cropping, shelterbelts, grass strips, and various combinations of these techniques. The adoption of these techniques can also improve soil quality by increasing SOM content and forming good soil structure, both of which may improve the water-holding capacity of soils and hence could strengthen the adaptability of water management during drought.^[36]

CONCLUSION

Soil erosion is an important form of global change; severely threatens the qualities of soil, air, and water resources; and

hence menaces agricultural development, environmental quality, ecological civilization, and ultimately human well-being. Soil erosion and global climate change are related by bidirectional interaction, and soil erosion is likely to increase significantly under global climate change unless amelioration measures are taken. Therefore, it is critical to take scientific and effective actions against soil erosion. The potential for global climate change to increase the risk of soil erosion is undoubtful, but the actual damage is not clear. More work should be done to refine the modeling of soil erosion under the scenarios of global climate changes in different regions. During the combating of its serious soil erosion problem, China has accumulated a wealth of knowledge and experience in soil and water conservation practices, which could be useful for other countries and regions that face the soil erosion issue alike.

REFERENCES

1. Ministry of Water Resources. *Bulletin of First National Water Census for Soil and Water Conservation*; 2013. Available at http://www.mwr.gov.cn/slx/slyw/201305/t20130517_435841.html (accessed May 2013).
2. Ministry of Water Resources. *Soil and Water Loss and Ecological Security in China – Soil and Water Loss Data*; Science Press: Beijing, 2010.
3. Hu, X.B.; Li, Z.B.; Hao, M.D.; Tang, K.L.; Zheng, F.L. Down-scale analysis for water scarcity in response to soil-water conservation on Loess Plateau of China. *Agr. Ecosyst. Environ.* **2003**, *94*, 355–361.
4. Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 99–118.
5. Scherr, S. *Soil Degradation: A Threat to Developing Country Food Security by 2020*, In IFRI Food, Agriculture and Environment Discussion Paper 27; International Food Policy Research Institute: Washington, DC, 1999; 63 pp.
6. Lal, R. Soil erosion and the global carbon budget. *Environ. Int.* **2003**, *29*, 437–450.
7. Pimentel, D.; Harvey, C.; Resosudarmo, P.; Sinclair, K.; Kurz, D.; McNair, M.; Crist, S.; Shpritz, L.; Fitton, L.; Saffouri, R.; Blair, R. Environmental and economic costs of soil erosion and conservation benefits. *Science* **1995**, *267*, 1117–1123.
8. den Biggelaar, C.; Lal, R.; Wiebe, K.; Eswaran, H.; Brene-man, V.; Reich, P. The global impact of soil erosion on productivity II: Effects on crop yields and production over time. *Adv. Agron.* **2004**, *81*, 49–95.
9. Montgomery, D.R. Soil erosion and agricultural sustainability. *P. Natl. Acad. Sci. USA* **2007**, *104* (33), 13268–13272.
10. Govers, G.; Van Oost, K.; Wang, Z. Scratching the critical zone: The global footprint of agricultural soil erosion. *Procedia Earth Planet. Sc.* **2014**, *10*, 313–318.
11. Liu, J.; Williams, J.R.; Zehnder, A.J.B.; Yang, H. GEPIC – Modelling wheat yield and crop water productivity with high resolution on a global scale. *Agr. Syst.* **2007**, *94*, 478–493.

12. Blanco, H.; Lal, R. *Principles of Soil Conservation and Management*; Springer: Berlin, 2010.
13. E, J.P. The summary report of integrated survey for soil and water loss and ecological security in China. *Soil Water Conserv.* **2008**, *12*, 3–6.
14. Bakker, M.M.; Govers, G.; Kosmas, C.; Vanacker, V.; van Oost, K.; Rounsevell, M. Soil erosion as a driver of land-use change. *Agr. Ecosyst. Environ.* **2005**, *105*, 467–481.
15. White, W.R. *Evacuation of Sediments from Reservoirs*; Thomas Telford: London, 2001.
16. Palmieri, A.; Shah, F.; Annandale, G.W.; Dinar, A. *Reservoir Conservation*, Vol. I; The RESCON Approach; World Bank: Washington, DC, 2003.
17. Griffin, D.W.; Kellogg, C.A.; Shinn, E.A. Dust in the wind: Long range transport of dust in the atmosphere and its implications for global public and ecosystem health. *Glob. Change Human Health* **2001**, *2* (1), 20–33.
18. Lal, R. Soil carbon sequestration impacts on global climate change and food security. *Science* **2004**, *304* (5677), 1623–1627.
19. Ito, A. Simulated impacts of climate and land-cover change on soil erosion and implication for the carbon cycle, 1901 to 2100. *Geophys. Res. Lett.* **2007**, *34* (9), L09403, DOI: 10.1029/2007GL029342.
20. Pimentel, D. Soil erosion: A food and environmental threat. *Environ. Dev. Sustain.* **2006**, *8*, 119–137.
21. Ministry of Water Resources. *Bulletin of Flood and Drought Disasters in China*; China Water & Power Press: Beijing, 2013.
22. Tilman, D.; Downing, J.A. Biodiversity and stability in grasslands. *Nature* **1994**, *367*, 363–365.
23. Ingram, J.; Lee, J.; Valentin, C. The GCTE soil erosion network: A multi-participatory research program. *J. Soil Water Conserv.* **1996**, *51* (5), 377–380.
24. Nearing, M.A.; Jetten, V.B.; Baffaut, C.; Cerdan, O.; Couturier, A.; Hernandez, M.; Le Bissonnais, Y.; Nichols, M.H.; Nunes, J.P.; Renschler, C.S.; Souchere, V.; Van Oost, K. Modeling response of soil erosion and runoff to changes in precipitation and cover. *Catena* **2005**, *61*, 131–154.
25. IPCC. *Climate Change 2007: The Physical Science Basis. Summary for Policymakers. Contribution of Working Group I to the Fourth Assessment Report to the Intergovernmental Panel on Climate Change (IPCC)*; IPCC: Paris, 2007.
26. Lal, R. Influence of soil erosion on carbon dynamics in the world. In *Soil Erosion and Carbon Dynamics*; Roose, E.J., Lal, R., Feller, C., Barthes, B., Stewarts, B.A., Eds.; Advances in Soil Science; CRC; Taylor & Francis: Boca Raton, 2006; 23–25.
27. O'Neal, M.R.; Nearing, M.A.; Vining, R.C.; Southworth, J.; Pfeifer, R.A. Climate change impacts on soil erosion in Midwest United States with changes in crop management. *Catena* **2005**, *61*, 165–184.
28. Mengistu, D.; Bewket, W.; Lal, R. Soil erosion hazard under the current and potential climate change induced loss of soil organic matter in the upper Blue Nile (Abay) river basin, Ethiopia. In *Sustainable Intensification to Advance Food Security and Enhance Climate Resilience in Africa*; Lal, R., Singh, B.R., Mwaseba, D.L., Kraybill, D., Hansen, D.O., Eik, L.O., Eds.; Springer: Cham, Heidelberg, New York, Dordrecht, London, 2015; 137–163.
29. Munson, S.M.; Belnap, J.; Okin, G.S. Responses of wind erosion to climate-induced vegetation changes on the Colorado Plateau. *P. Natl. Acad. Sci.* **2011**, *108* (10), 3854–3859.
30. Sankey, J.B.; Germino, M.J.; Benner, S.J.; Glenn, N.F.; Hoover, A.N. Transport of biologically important nutrients by wind in an eroding cold desert. *Aeolian Res.* **2012**, *7*, 17–27.
31. Segal, M.; Pan, Z.; Arritt, R.W.; Takle, E.S. On the potential change in wind power over the U.S. due to increases of atmospheric greenhouse gases. *Renew. Energ.* **2001**, *24*, 235–243.
32. Ren, M.E. Research advances in sea level research. *J. Nanjing Univ.* **2000**, *36* (3), 269–278.
33. Zhuang, Z.Y.; Ying, P.; Wu, J.Z.; Zhuang, L.H. Coastal erosion and its influence on southern Shandong sandy coast. *Mar. Geol. Qual. Geol.* **2000**, *20* (3), 15–21.
34. Cao, S. Impact of China's large-scale ecological restoration program on the environment and society in arid and semiarid areas of China: Achievements, problems, synthesis and applications. *Crit. Rev. Environ. Sci. Technol.* **2011**, *41*, 317–335.
35. Zhao, Q.G. Some considerations for present soil and water conservation and ecology security of south China. *Bull. Soil Water Conserv.* **2006**, *26* (2), 1–8.
36. Walthall, C.L.; Hatfield, J.; Backlund, P.; Lengnick, L.; Marshall, E.; Walsh, M.; Adkins, S.; Aillery, M.; Ainsworth, E.A.; Ammann, C.; Anderson, C.J.; Bartomeus, I.; Baumgard, L.H.; Booker, F.; Bradley, B.; Blumenthal, D.M.; Bunce, J.; Burke, K.; Dabney, S.M.; Delgado, J.A.; Dukes, J.; Funk, A.; Garrett, K.; Glenn, M.; Grantz, D.A.; Goodrich, D.; Hu, S.; Izaurralde, R.C.; Jones, R.A.C.; Kim, S.-H.; Leaky, A.D.B.; Lewers, K.; Mader, T.L.; McClung, A.; Morgan, J.; Muth, D.J.; Nearing, M.; Oosterhuis, D.M.; Ort, D.; Parmesan, C.; Pettigrew, W.T.; Polley, W.; Rader, R.; Rice, C.; Rivington, M.; Roskopf, E.; Salas, W.A.; Sollenberger, L.E.; Srygley, R.; Stöckle, C.; Takle, E.S.; Timlin, D.; White, J.W.; Winfree, R.; Wright-Morton, L.; Ziska, L.H. *Climate Change and Agriculture in the United States: Effects and Adaptation*, USDA Technical Bulletin 1935; USDA: Washington, DC, 2013; 186.

Erosion: On-Site and Off-Site Impacts

Jerry L. Hatfield

National Laboratory for Agriculture and the Environment, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.

Abstract

Erosion, either by wind or by water, moves large amounts of soil each year. Erosion is the process of moving soil by the action of water or wind from one location to another in the landscape and often over large distances as off-site impacts. Redistribution of soil within the field is often overlooked as a major impact of erosion processes compared to the more visible off-site movement that is often seen as alluvial fans in streams and rivers or the dunes of sand in highly erodible arid regions. Continued loss of topsoil removes the most productive portion from the soil profile, and although nutrients and water can be resupplied to the soil, the productivity levels remain affected. The major on-site impact of erosion is the loss of crop productivity, and this is mirrored in reductions in crop yield and water use efficiency and nutrient use efficiency. In regions of the world with variable rainfall, the loss of topsoil increases the vulnerability of human food supplies.

INTRODUCTION

Erosion, either by wind or by water, moves large amounts of soil each year. It has been estimated that over 80% of the world's agricultural lands are affected to some degree by erosion.^[1,2] Continued loss of topsoil removes the most productive portion from the soil profile, and although nutrients and water can be resupplied to the soil, the productivity levels remain affected. The major on-site impact of erosion is the loss of crop productivity, and this is mirrored in reductions in crop yield and water use efficiency and nutrient use efficiency. In regions of the world with variable rainfall, the loss of topsoil increases the vulnerability of human food supplies. Off-site impacts of erosion relate to the economic and ecological costs of sediment, nutrients, or agricultural chemicals being deposited in streams, rivers, and lakes. Adoption of soil management practices to reduce erosion can have profound effects on future world food supplies.

Erosion is the process of moving soil by the action of water or wind from one location to another in the landscape and often over large distances as off-site impacts. Redistribution of soil within the field is often overlooked as a major impact of erosion processes compared to the more visible off-site movement that is often seen as alluvial fans in streams and rivers or the dunes of sand in highly erodible arid regions. Erosion processes are responsible for the shaping of the canyons, creation of the Palouse areas in Washington, and the loess deposits in both China and the United States. Erosion is a powerful force at work in the agricultural lands of the world. It has been estimated that almost 56% of the world's agricultural lands have been degraded by water erosion, while 28% have been degraded by wind erosion. Continued degradation of world agricultural lands by erosion threatens the stability of food production.^[1–3]

ON-SITE IMPACTS

Topsoil in agricultural fields has been removed since humans began farming. Evidence of this effect is seen by the level of agricultural fields compared to adjacent fence rows or native pastures. Removal of this topsoil has a large impact on the ability of the soil to produce food, feed, or fiber. On-site impacts of erosion are evident in the productivity levels of the soil. Erosion removes the topsoil that is highest in organic matter content, has the most stable soil structure, and offers the most optimal seedbed for germinating and emerging plants. The upper layers of soil are those that reflect the influence of anthropogenic activities. Removal of the topsoil reduces the water-holding capacity of the soil and further reduces the available rooting depth of the crop. All of these changes have a negative impact on soil productivity. Soil productivity levels represent the potential capability of the soil to produce a crop under optimal meteorological conditions. Therefore, any removal of soil would reduce this potential (Fig. 1).

There is a large variation among soils in their response to erosion. Temperate soils are less affected by erosion than tropical soils; however, in both soils, productivity is negatively affected. In tropical soils, the on-site impacts vary among crops. Cereal crops are more affected than root crops.^[3] Similar responses are seen for other areas of the world. Continued erosion will cause a shift in the types of plants that can be grown on a given soil. Erosion effects can be modified by the use of inputs (nutrients, tillage, and supplemental supplies of water). However, adding soil nutrients is not often a replacement for the natural productivity of the soil. Rooting depth of the crop is restricted more in eroded than in non-eroded soils. Problems with

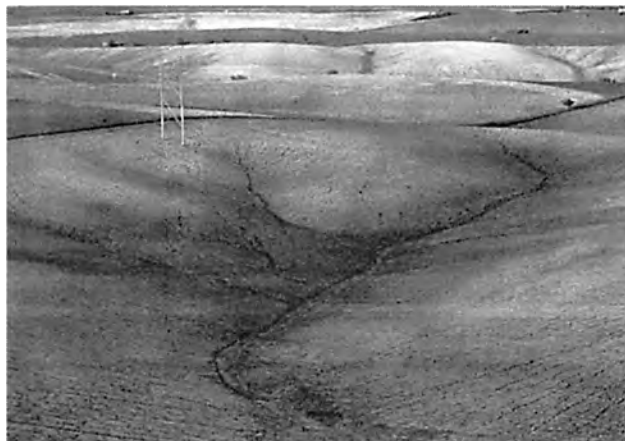


Fig. 1 Erosion dissects productive cropland with water flow and sediment movement through fields.

Source: Photo courtesy of U.S. Department of Agriculture–Agricultural Research Service.

rooting depth and extensive root development reduce the efficiency of water and nutrient use by the crop.^[4–6]

On-site impacts of soil erosion are difficult to quantify. Crop production levels (yields) are often the most common measure of soil performance. However, the measurement of water use efficiency or nutrient use efficiency provides a more quantitative method of being able to compare among locations or crop species. Water use efficiency and nutrient use efficiency (expressed as the amount of grain or total biomass produced relative to the amount of water used and nutrient applied, respectively) provide a measure of the capability of the soil to produce a crop. In wheat, removal of the upper 0.18 m of the soil profile reduced the water use efficiency by 20%, but did not affect the amount of water used by the crop.^[7,8] Nutrient use efficiency in the wheat was affected, and it was found that additions of either nitrogen or phosphorus fertilizers to eroded soils did not restore productivity to the non-eroded levels.^[4] In tropical soils, erosion reduced the water use efficiency by 60% with 0.35 m of erosion and reduced nutrient use efficiency of the non-eroded soil by over 500%.^[2] These examples are typical of the responses that have often been observed when crop production efficiency has been studied. Efficiency of crop production diminishes with decreasing topsoil depth.

In Africa—where much of the crop production is by subsistence agriculture—the loss of 0.4 m of topsoil could reduce crop yields by over 75%.^[3] A conceptual diagram of the potential changes in productivity and required production inputs is shown in Fig. 2. This level of reduction would increase the potential vulnerability of food supplies to weather variation within the growing season or climate changes across years. Erosion would have a larger impact on productivity in arid and semiarid environments—where water is a limiting component in crop or forage production—than in the temperate or tropical climates.

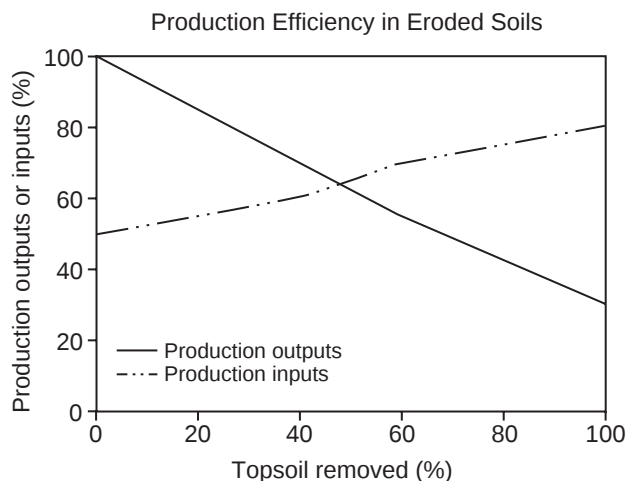


Fig. 2 Conceptual diagram of the effect of topsoil removal on the input and output changes in agricultural systems.

In addition to the removal of the topsoil from the field, an on-site impact is often the disruption of cultural operations because of gullies or rills in the field. The generation of eroded areas within the field can cause problems with all cultural operations that result in damage to machinery or increased labor costs to repair these problems.^[2] Deposition of soil within a field can create problems by burying emerging crops, and often the moving water transports crop residues from previous crops to low-lying areas of the field. Another on-site impact that occurs is the deposition of soil that contains agricultural chemicals onto adjacent fields. These chemicals can cause problems with crop growth, particularly if the crop in the adjacent field is sensitive to the particular chemical. This effect is seen in localized sites, but the impact is often large if there are high-value crops being grown (Fig. 3).



Fig. 3 Surface runoff occurs in fields and removes large amounts of soil and crop material.

Source: Photo courtesy of U.S. Department of Agriculture–Agricultural Research Service.

OFF-SITE IMPACTS

Off-site impacts are those linked with the transport and deposition of soil from production fields or paddocks onto adjacent or downstream areas. These impacts can be classified into two major components: economic and environmental. Economic impacts of either water or wind erosion are difficult to quantify. Costs associated with the removal of eroded soil from roads after floods or windstorms are billions of dollars each year.^[1] This amount does not include the amount of money spent on repairing damage caused by sediment deposited in homes after floods or the costs associated with temporary housing. These off-site impacts are often the ones that are most noticeable due to water and wind erosion. Other economic impacts of erosion are the removal of sediment from lakes and rivers to maintain navigation, recreation, or flow.

Environmental impacts from erosion may take a longer time to be noticeable. Sediment deposition into streams, rivers, and lakes impacts the biological health of the ecosystem. The presence of excessive sediment in the water column disrupts the ecological balance, and it is difficult to place a monetary value on this impact. Sediment that is moved off-site may contain nutrients, in particular phosphorus, agricultural chemicals, or pathogens. Each of these components has an impact on the ecosystem. Excess phosphorus can cause eutrophication in water bodies that leads to diminished productive capability and water quality problems.^[9] Phosphorus movement is mainly through attachment to sediment, and movement of soil from fields often transports large amounts of sediment-bound materials.^[10] Likewise, many agricultural chemicals are tightly bound to sediments and only move off-site when soil is transported. Transport of agricultural chemicals (herbicides or insecticides) poses health hazards to both fish and humans if the concentration in the water supply exceeds toxic levels. Treatment of drinking water supplies to remove agricultural chemicals varies across regions; however, annual costs that exceed \$500,000 in water treatment facilities have been reported (L.D. McMullen, Des Moines Water Works, Personnel Communication, 2000). Environmental quality aspects of erosion are more evident in the effect of materials bound to the moving sediment than sediment itself.

Transport of pathogens from agricultural lands to adjacent waterbodies has become one of the most important topics in agricultural watersheds. Pathogens may impact fish and human populations, and the tragedy in Milwaukee, Wisconsin, with the deaths of over 20 people from waterborne pathogens, shows the potential of off-site movement to affect human life. Movement of sediment and nutrients has been considered to be the largest off-site impact and the most costly.^[9,10] However, transport of pathogenic material from fields or stream banks into water supplies is emerging as one of the critical problems in watershed management, and scientific papers to evaluate the cause and effect relationships will emerge from these observations.

Off-site impacts from wind erosion are often characterized as nuisances because of the problems with deposition of windblown materials. The cost of removing these deposits is less compared to the costs of removing sediment. There is also a nuisance factor caused by reduced visibility; however, there are direct human health concerns associated with increased particulate loads and soilborne pathogens in the air.

CONCLUSION

Erosion has a major impact on both on-site productivity and efficiency of production. It is difficult to develop an exact cost figure for erosion. However, continual degradation of the soil through erosion threatens the stability of future food supplies. Off-site impacts of erosion affect both water and air quality. Increasing concern about the potential impacts of off-site movement of nutrients, chemicals, and pathogens on water quality will continue to be a major concern in how we view the soil resource and implement practices to protect soil. Decreasing erosion will have positive on-site and off-site impacts.

REFERENCES

1. El-Swaify, S.A. State of the art for assessing soil and water conservation needs and technologies. In *Adopting Conservation on the Farm*; Napier, T.L., Camboni, S.M., El-Swaify, S.A., Eds.; Soil and Water Conservation Society: Ankeny, 1994; 13–27.
2. Larson, W.E.; Pierce, F.J.; Dowdy, R.H. The threat of soil erosion to long-term crop production. *Science* **1983**, *219*, 458–465.
3. Lal, R. Erosion–crop productivity relationships for soils of Africa. *Soil Sci. Soc. Am. J.* **1995**, *59*, 661–667.
4. Massee, T.W.; Waggoner, H.O. Productivity losses from soil erosion on dry cropland in the intermountain area. *J. Soil Water Conserv.* **1985**, *40*, 447–450.
5. Martin, C.K.; Cassel, D.K. Soil loss and silage yield for three tillage management systems. *J. Prod. Agric.* **1992**, *5*, 581–586.
6. McGregor, K.C.; Cullum, R.F.; Mutchler, C.K. Long-term management effects on runoff, erosion, and crop production. *T. ASABE* **1999**, *42*, 99–105.
7. Tanaka, D.L. Topsoil removal influences on spring wheat water-use efficiency and nutrient concentration and content. *T. ASABE* **1990**, *33*, 1518–1524.
8. Tanaka, D.L.; Aase, J.K. Influence of topsoil removal and fertilizer application on spring wheat yields. *Soil Sci. Soc. Am. J.* **1989**, *53*, 228–232.
9. Chambers, B.J.; Garwood, T.W.D.; Unwin, R.J. Controlling soil water erosion and phosphorus losses from arable land in England and Wales. *J. Environ. Qual.* **2000**, *29*, 145–150.
10. Sauer, T.J.; Daniel, T.C.; Nichols, D.J.; West, C.P.; Moore, P.J., Jr.; Wheeler, G.C. Runoff water quality from poultry litter treated pasture and forest sites. *J. Environ. Qual.* **2000**, *29*, 515–521.

Erosion: Overland Research and Measurement Techniques

Chia Chun Joseph Wu

National Pingtung University of Science and Technology, Neipu, Taiwan

Abstract

Water erosion initiates from uplands and continues its action through transportation and deposition. It normally produces 0.03 ~ 64 t/ha of soil loss annually. However, it can greatly accelerate to 3 ~ 750 t/ha of sediment annually when cultivation is involved. Measurement techniques are required to understand as well as quantify the erosion processes. Progress in technology has advanced the measurement instrumentation from manual to automatic oriented. Nevertheless, researchers need to identify their studies' objective, the environment in which their studies being conducted, and the characteristics of sediment before making selection of the suitable measurement techniques. Water erosion measurement techniques associated with overland erosion processes are addressed in the entry. Examples of Universal Soil Loss Equation standard runoff plots and small plots are illustrated through figures. Volumetric sediment collection and micro-topography measurement devices are also addressed. Cautions addressed in the entry may require additional attention while selecting water erosion measurement techniques.

INTRODUCTION

In the early days, erosion controls were mostly implemented on a trial-and-error basis with little assurance of the outcome. Loss of lives and properties as well as overdesign of counter-measures had finally caught researchers' attention to quantify the soil loss caused by water erosion. The first effort to quantify water erosion processes was initiated in 1914. The Dust Bowl in the early 1930s attracted public attention in soil erosion. A joint federal-state committee was appointed, and soil erosion experiment stations were established at different geographic locations representing a wide range of soils and climatic regions throughout the United States.

Built upon over 10,000 plot-years of data, a soil loss estimation tool, known as Universal Soil Loss Equation (USLE)—involving the multiplication of six key important factors—was introduced to both academic and field application arenas. These key factors include rainfall erosivity, soil erodibility, slope length, slope steepness, cover and management, and support practice factors.

Soil loss estimation techniques have gone through a gradual transformation since the 1970s from solely empirical-based models to process-based models. Despite the complexity of process-based models, USLE serves as an important guideline for the advances in water erosion research, and water erosion measurement techniques developed throughout the chronicle of water erosion research are commonly employed in the research community.

WATER EROSION MEASUREMENT TECHNIQUES

Soil loss and non-point source pollution estimation has gained greater interest in the last decades for the

purposes of conservation planning and policy making, which has shifted federal funding from fundamental researches to modeling. For instance, ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation),^[1] CREAMS (Chemicals, Runoff and Erosion from Agricultural Management Systems),^[2] EUROSEM (European Soil Erosion Model),^[3] and WEPP (Water Erosion Prediction Project)^[4] have extended soil loss estimation from hillslope to watershed. Nevertheless, every mathematic module embedded in a simulation model requires scientific sound support obtained from fundamental researches. Hence, field as well as laboratory erosion research requires appropriate measurement techniques and adequate experiment design. Therefore, the following sections address the common measurement techniques in water erosion.

Rainfall Erosivity

Rainfall erosivity factor describes the potential of rainstorm in water erosion. It is formulated as the product of rainfall kinetic energy (E) and maximum 30-minute rainfall intensity (I_{30}).^[5,6] The maximum 30-minute rainfall intensity can be easily extracted from rainfall records. However, rainfall kinetic energy is less straightforward. Drop size distribution representing the local climatic condition must be measured prior to the development of rainfall erosivity factor.

Several techniques have been developed for drop size measurement. They include the flour pellet method,^[7] stain method,^[8] oil method, high-speed photography, laser method, and momentum method. The high-speed photography, laser method, and momentum method require sophisticated equipment to collect raindrop data, and the instrumentation can be costly. The flour pellet method

requires a longer time for preparation prior to implementation. It also requires consideration of the raindrop-induced pellets postprocess. It is greatly affected by the humidity of the working environment, which may cause variability in the drop size measurement. The oil method, on the other hand, is less difficult to prepare but more difficult to transport for outdoor measurement. The stain method, with the assistance of an image scanner,^[9] is probably the most cost-efficient and labor-efficient technique for raindrop size measurement.

By combining the drop size distribution, rainfall intensity, and terminal velocity, the rainfall kinetic energy can be determined. Due to geographic variability in rainfall characteristics, rainfall erosivity factor is the most needed parameter that requires local attention. It also requires long-term measurements to obtain a representative index with a wide spectrum of intensities for local climate.^[10]

Soil Erodibility

Soil erodibility describes the potential of a soil in resisting soil erosion. The higher the erodibility a soil possesses, the less resistant the soil is to erosion. Soil erodibility can be determined from the soil loss measurement on a USLE standard runoff plot if it is accessible. Otherwise, it can be determined by measuring aggregate stability, soil properties related to water transmission or to water retention.^[11]

USLE Standard Runoff Plot

USLE standard runoff plot has a slope length of 22.1 m with various plot widths designed to meet the research objective. It is installed on a 9% uniform slope. The plot has to be in fallow condition with up-and-down plowing for two consecutive years. Use of USLE standard plot is highly recommended for those regions with a limited soil erodibility data.

Soil erodibility can be determined by dividing annual soil loss by annual rainfall erosivity. Reliable soil erodibility data may require a monitoring period much longer than 2 years, especially when cropping factors are considered. Using rainfall simulators on small plots provides an alternative for soil erodibility measurement.

Small Plot

Moldenhauer^[12] once used a rainfall simulator on small plots to study the effects of soil characteristics on soil erodibility. Small plots are much easier to prepare; however, the edge effect from plot boundaries may affect research results. The edge effects associated with small plots include the splash in and out of the plot perimeter as well as the drainage of the soil water from the bottom of the plot.^[13]

For small plot erosion study, it is necessary to retain a border area around the perimeter of the plot so that the splashed soil in and out of the plot can be compensated. As far as the bottom boundary of the small plot is

concerned, an open bottom made of screen is often used to provide a free drainage condition.

Careful arrangement of the storm sequence is necessary in simulated rainfall studies to replicate the effect of a natural storm pattern and antecedent soil–water contents on soil erodibility. Therefore, it is always a good practice to have field-measured soil erodibility data to correlate with simulated rainfall erodibility results.

Rainfall Simulators

A rainfall simulator is a common tool used for water erosion research. It greatly reduces the time span of erosion study, and it is more controllable than natural rain. It also provides researchers the means to repeat the study anywhere and anytime they wish.

The basic requirement for a rainfall simulator is to produce rainstorms with drop size distribution, impact velocity, and intensity similar or close to those of natural storms.^[14] Numerous types of rainfall simulators have been developed for erosion study. Rainfall simulator can be mobile for field study, and it can also be housed in a building. Two types of raindrop-forming techniques are commonly used, namely nozzle-type and free-fall-type.

The nozzle-type rainfall simulator generates simulated rainfall by forcing water through a spray nozzle to increase the drop velocity, so that as simulated rainfall reaches erosion plot, the impact velocity can reach that of the natural rain. On the other hand, the free-fall-type simulator generates rainfall by forming waterdrops from a matrix of hypodermic needles. Therefore, it requires greater fall height for drops to reach terminal velocity. The raindrop generator shown in Fig. 1 is housed in a 15-m-tall rainfall simulator building with an effective fall height of 11 m. This rainfall simulator has an effective working area of 3 m × 4 m, as shown in Fig. 2, with a manual and automatic control for desired rainfall intensities. Users can feed the rainfall



Fig. 1 Free-fall-type raindrop generator.



Fig. 2 Adjustable working platform for erosion research.

simulator with a rainstorm histogram so that the history of a storm can be simulated.

Extending the results generated from rainfall simulator studies to erosion under natural rains can be difficult. Water erosion at field conditions is affected not only by rain characteristics and soil's dynamic behavior but also by cropping and land management practices that from time to time cannot be effectively quantified under rainfall simulation environment. Hence, one needs to recognize the differences between simulated and natural environments.

Plot Design and Layout

Erosion plots are often laid out with different widths and lengths to meet various research objectives. Rill erosion seldom occurs on slope length less than 5 m with gentle slope steepness;^[13] however, rilling has been commonly observed on steep slopes with slope length less than 5 m, such as newly constructed embankments along the roadways. Therefore, researchers use different plot layouts to meet their specific needs.

Interrill Erosion

Interrill erosion is mainly caused by raindrop impact. It is less noticeable than other forms of erosion. Researchers have been using small plots 0.5–1 m long along with rainfall simulators to quantify interrill erosion. The measurement technique adapted for interrill erosion study is quite simple—by sampling sediment concentration or sediment load and runoff volume at the downstream end of a small plot.

Rill Erosion

Rill erosion is a common erosion process on longer slope lengths and/or steeper slopes. It often produces greater amount of sediment; therefore, H-flume as well as auto-sampler may suffer the sedimentation problem during sample



Fig. 3 Coshocton wheel sampler.

collection. An H-flume^[15] with a calibrated stage recorder can be attached at the outlet of runoff plot to obtain the runoff hydrograph, whereas an auto-sampler like Coshocton wheel sampler (Fig. 3) can be used to acquire sediment concentration histogram. However, due to high sediment concentration, H-flume as well as auto-sampler may suffer the sedimentation problem during sample collection.

There are more options of plot design to choose from to study the effects of slope length and slope steepness. One option is to set up runoff plots at various slope lengths and slope steepness. The research site shown in Fig. 4 was located on a 60% steepness slope. Due to land limitation, runoff plots were divided into segments along the slope using grass buffer and drainage ditches to complement the desired slope lengths.

Another option to quantify the slope steepness effect is to use soil boxes with adjustable slopes (Fig. 5). At the downstream end of a soil box, collection trough and sampling buckets are often used to collect lumped samples. H-flumes, stage recorders, and auto-samplers can be used to obtain continuous samples.

The added-inflow technique^[16] is an alternative to simulate long slope length condition. The technique basically consists of inflow pipe, flow rate regulator, and a damping



Fig. 4 Sixty percent steepness erosion research site.



Fig. 5 Erosion research conducted using soil boxes on a slope-adjustable platform.

device. The purpose of the damping device is to prevent inflow from scouring the soil bed at the flow entrance, which may constitute a greater portion of sediment in the sample. The amount of inflow added to small runoff plots depends on the slope length that a researcher wants to simulate. It requires runoff hydrographs generated from short slope lengths at otherwise identical experimental conditions.

For those only interested in total runoff and soil loss, a set of stilling tanks with sample splitters installed at the rims of tank walls is probably the cheapest way to achieve the objective (Fig. 6). As the main sediment stilling tank is filled with sediment and runoff, the splitter can split the excess runoff in a desired portion into the second tank and so forth.

Cover/Management and Support Practices Factors

The effects of cover/management and support practices on soil erosion require erosion study on larger scales because all these controlling factors affect runoff routing, surface roughness, and sediment delivery, which are difficult to



Fig. 6 Stilling tanks and runoff splitters.

capture on small plots. Hence, rainfall simulator studies resulting from small plots may not unveil the actual physical processes occurring at large scales. In such case, a pair of gauged watersheds is probably the most suitable technique to choose. One watershed is treated with selected management or practices, and the other serves the sole purpose of comparison. In the case of watershed study, collection trough and stage recorder are usually used.

As far as the measurement of sediment outflow is concerned, auto-samplers, Coshocton wheel samplers, or a set of stilling tanks will serve the purpose. However, Coshocton wheel samplers as well as auto-samplers require undivided attention to ensure that devices are operational. A motor may be needed to drive Coshocton wheel sampler to guarantee free rotation of the sampler especially during low flow. For sites that having high potential of yielding coarse sediment, clogging at the sample vane of Coshocton sampler as well as at siphon tube intake may cause problem.

OTHER MEASUREMENT TECHNIQUES ASSOCIATED WITH WATER EROSION ASSESSMENT

Aerial photographs taken at different times can be used to assess the occurrence of erosion, progress of gully erosion processes as well as mass movement. It can also assess the effectiveness of erosion control qualitatively. Using aerial photographs with geographic information system helps identify erosion hot spots on large-scale areas that may not be easily visualized from ground survey. Nevertheless, thorough ground survey is needed to quantify the volume of gully erosion.

If the progress of rilling is the main subject of the study, a rill meter (Fig. 7) or laser profiler^[17] is a convenient tool. Geometry of the rills can be photoed manually or scanned automatically transaction by transaction and analyzed digitally. However, natural consolidation due to soil wetting and compaction caused by raindrop impact may affect the surface profile readings. Therefore, quantification of soil

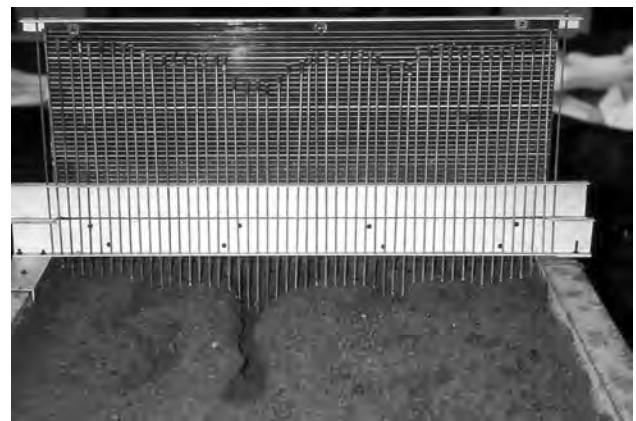


Fig. 7 Rill meter.

loss volume using profile scanning may require additional adjustment.

Collecting sediment concentration samples from a stream or river channel is another option for qualitative erosion assessment. One needs to be cautioned on the interpretation of stream or channel sediment data. Sediment samples collected from a river channel are actually the product from upland erosion, sediment delivery along the basin, and erosion initiated from stream banks. There is no certain means to distinguish the source of sediment.

CONCLUSION

Erosion measurement techniques vary widely in scale, time, and cost involved, and there is no standard procedure to follow. Different methodologies produce different results. Researchers need to make their own judgments to select the most feasible methodology to ensure that the research objectives are addressed. A proper understanding of the quality of data associated with specific erosion measurement technique will result in useful information for the erosion research.

REFERENCES

1. Beasley, D.B.; Huggins, L.F.; Monke, E.J. ANSWERS: A model for watershed planning. *T. ASAE*. **1980**, *23*, 938–944.
2. Knisel, W.G. CREAMS: A field scale model for chemicals, runoff and erosion from agricultural management system. USDA Conservation Research Report 26, USDA: Washington, DC, 1980.
3. Morgan, R.P.C.; Quinton, J.N.; Rickson, R.J. Modelling methodology for soil erosion assessment and soil conservation design: The EUROSEM approach. *Outlook Agr.* **1994**, *23* (1), 5–9.
4. Laflen, J.M.; Elliot, W.J.; Flanagan, D.C.; Meyer, C.R.; Nearing, M.A. WEPP predicting water erosion using a process-based model. *J. Soil Water Conserv.* **1997**, *52* (2), 96–102.
5. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses—A Guide To Conservation Planning*. UDSA-SEA Agricultural Handbook 537; U.S. Department of Agriculture: Washington, DC, 1978; Vol. 58.
6. Renard, K.G.; Foster, G.R.; Weesies, G.A.; McCool, D.K.; Yoder, D.C. *Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE)*; U.S. Department of Agriculture: Washington, DC, 1997; 404 pp.
7. Hudson, N.W. *The Flour Pellet Method for Measuring the Size of Raindrops*. Res. Bull. No. 4; Department of Conservation: Salisbury, 1964.
8. Hall, M.J. Use of the stain method in determining the drop size distributions of coarse liquid sprays. *T. ASAE*. **1970**, *13* (1), 33–37.
9. Wu, C.-C.; Wang, A.-B. Drop size characteristics and erosive kinetic energy of natural rainstorms in Pingtung Laopi area. *J. Chinese Soil Water Conserv.* **1996**, *27* (2), 151–165.
10. Hudson, N.W. *Field measurement of soil erosion and runoff*; FAO Soils Bulletin: Rome, 1993; Vol. 68, 109–127.
11. Lal, R.; Elliot, W. Erodibility and erosivity. In *Soil Erosion Research Methods*, 2nd Ed.; Lal, R., Ed.; Soil and Water Conservation Society: Ankeny, 1994. Chapter 8, 181–210.
12. Moldenhauer, W.C. Procedure for studying soil characteristics using disturbed samples and simulated rainfall. *T. ASAE*. **1965**, *8* (1), 74–75.
13. Mutchler, C.K.; Murphree, C.E.; McGregor, K.C. Laboratory and field plots for erosion research. In *Soil Erosion Research Methods*, 2nd Ed.; Lal, R., Ed.; Soil and Water Conservation Society: Ankeny, 1994; Vol. Chapter 2, 11–37.
14. Meyer, L.D. Rainfall simulators for soil erosion research. In *Soil Erosion Research Methods*, 2nd Ed.; Lal, R., Ed.; Soil and Water Conservation Society: Ankeny, 1994; Vol. Chapter 4, 83–103.
15. United States Department of Agriculture. *Field Manual for Research in Agricultural Hydrology; Agricultural Handbook No. 224*; U.S. Department of Agriculture: Washington, DC, 1979; 88–91. 248–250.
16. Meyer, L.D.; Harmon, W.C. Sediment losses from cropland furrow of different gradients. *T. ASAE*. **1985**, *28* (2), 448–453. 461.
17. Huang, C.-H.; Bradford, J.M. Applications of a laser scanner to quantify soil microtopography. *Soil Sci. Soc. Am. J.* **1992**, *56*, 14–21.

Erosion: Perennial Crop Plantations

Alfred E. Hartemink

International Soil Reference and Information Centre (ISRIC), Wageningen, the Netherlands

Abstract

It is generally assumed that a perennial tree cover protects the soil better against erosion than do annual crops. But tree crops may require several years to close their canopy, whereas most annual crops provide adequate cover within 6 weeks after planting. During the immature phase of tree crops, there may be insufficient soil cover. The effects of tree crops on soil erosion have been fairly well documented. Soil erosion and sediment transport from catchments with natural forests are minimal (<1 Mg/ha/yr), but the levels of soil erosion tend to increase when the natural forest is changed to tree crop plantations. Considerable differences have been found between crops and sites, and much depends on the soil, site factors (slope, rainfall, etc.), and management practices. In this entry, the main findings for oil palm, coffee, cocoa, and tea plantations in tropical regions are reviewed.

INTRODUCTION

Plantation agriculture is an important form of land use in the tropics. Large areas of natural and regenerated forest have been cleared for growing oil palm, rubber, cocoa, coffee, and other perennial tree crops. These crops grown both on large-scale plantations and by smallholders are an important source of income for many farmers in tropical regions.

EROSION UNDER OIL PALM

Several soil erosion studies under oil palm have been conducted in Malaysia (Table 1). Soil erosion from Oxisols ranged from 13 to 78 Mg/ha/yr and depended on the slope of site. Soil erosion on Ultisols ranged from 1 to 28 Mg/ha/yr, and the erosion was higher in harvesting paths. In mature oil palm plantations, soil erosion losses chiefly depend on the slope of site and soil management practices. Under young oil palm, soil erosion is usually limited because of the cover crop protecting the soil, and the limited erosion is not attributable to the palms. This was also reported from rubber plantations (Fig. 1).^[1] As the cover crop disappears after the closure of the palm canopy, harvest paths become exposed and compacted which enhances runoff and soil erosion. Therefore, soil erosion may not necessarily decrease when the palms get older and the canopy is closed. Chew, Kee, and Goh^[2] reviewed and published soil erosion data for moist forest and tree crop systems. Erosion under mature oil palm ranged from 7 to 21 Mg/ha/yr. These values are lower than those reported in Table 1, but the review showed that soil

erosion could be considerable at oil palm plantations in Malaysia.

The effects of erosion under oil palm are that the soil is removed from between the tertiary and quaternary feeding roots near the soil surface, in particular in the weeded circle. Exposed roots dry up and die, so that the water and nutrient uptake capacity of the root system is reduced. Although no experimental evidence is available, it is obvious that oil palms growing under these conditions undergo water deficits and nutritional deficiencies.^[3] Moreover, the nutrient use efficiency of applied fertilizers is reduced because of the lower uptake capacity of the roots.

EROSION UNDER COFFEE AND COCOA

Soil erosion losses can be considerable in coffee plantations that have no adequate shade or a low planting density with little natural mulch formed by litter. This is especially important for coffee grown in highlands on steep slopes and in new coffee plantations. Research in Colombia showed that annual soil nitrogen losses from unprotected areas exceeded the amount extracted by a good crop of coffee, but on well-developed coffee plantations that are adequately shaded or with a high planting density, erosion can be reduced to less than 2% of the losses that occur on unprotected plots.^[4]

In Venezuela, where since the mid-1970s the government has actively promoted the removal of shade trees from coffee plantations, low erosion losses were found.^[5] Under shaded coffee, total erosion losses were very low and less than 2 Mg soil/ha/yr, whereas under coffee

Table 1 Soil erosion losses under oil palm in Malaysia.

Soil order	Palm age (years)	Slope (%)	Condition	Soil erosion (Mg/ha/yr)
Trophectic Hapludox (Oxisols)	2–4	2	With legume cover crop	18.8
		5	With legume cover crop	24.0
		9	With legume cover crop	35.4
		15	With legume cover crop	50.0
Typic Hapludox (Oxisols)	2–4	12	<5 Uncovered	12.5
		2	With legume cover crop	23.5
		5	With legume cover crop	38.8
		9	With legume cover crop	57.1
Orthoxic Tropudult (Ultisols)	11	5	With legume cover crop	77.6
			Harvesting path	14.9
			Palm row	7.4
Typic Paleudult (Ultisols)	12–16	3–5	Beneath row	1.1
			Uncovered	28.0
			Plots with fronds cut	19.7
			Plots with extra fronds cut	16.3

Source: Adapted from PORIM^[12] and Hartemink.^[13]

without shade, erosion losses were 7 Mg soil/ha in the first year after the shade was removed. Erosion losses of unshaded coffee after 2 years were comparable to that of shaded coffee, whereas in general runoff and soil loss are lower in shaded than in unshaded plantations.^[6] The research in Venezuela showed that soil erosion correlated positively with agricultural activities (i.e., harvesting, pruning, and weeding).

Under monocropping cocoa in Malaysia, soil erosion losses were 11 Mg/ha/yr, but losses were considerably lower when cover crops such as *Indigofera spicata* were planted.^[7] When the cocoa was intercropped with banana, and clean weeding with herbicide was practiced, soil losses up to 70 Mg soil/ha/yr were measured, which are high losses based on a general rating of tolerable soil erosion losses.^[8,9]

EROSION UNDER TEA

In mature tea plantations, erosion is negligible (Fig. 2) because of the complete soil cover. Some soil erosion may occur directly after pruning and when the prunings



Fig. 1 Young rubber trees with cover crop in North Sumatra.

are removed. Othieno^[10] reported from Kericho (Kenya) erosion losses up to 168 Mg soil/ha in the first year after the establishment of a tea plantation. In the second year, soil losses were up to 81 Mg/ha, whereas in the third year losses were less than 7 Mg soil/ha. Three-quarter of the total erosion over the three-year period of the experiment occurred between planting and the time when the canopy had developed to 30%. Soil erosion can also be a problem when plantations run down. This was found to occur in Sri Lanka where tea plantations have been neglected since the mid-1970s, causing serious soil erosion of vacant patches.^[11]



Fig. 2 Mature tea plantation in the Papua New Guinea highlands.

Table 2 Soil chemical properties in eroded and non-eroded Oxisols under coffee and in virgin Oxisols near Lake Victoria, Tanzania.

Land use	Sampling depth (m)	pH	Organic C (g/kg)	CEC and exchangeable cations (mmol/kg)				
				Available P (mg/kg)	CEC	Ca	Mg	K
Virgin	0–0.15	5.2	25.2	12	259	61	40	1.8
	0.15–0.30	4.2	14.5	3	249	22	8	1.0
Coffee (non-eroded)	0–0.15	5.2	25.9	33	160	52	21	3.2
	0.15–0.30	4.8	12.2	3	128	23	18	1.8
Coffee (eroded)	0–0.15	4.1	19.0	5	256	23	14	1.9
	0.15–0.30	3.9	13.1	<2	259	8	3	1.6

Note: CEC, cation exchange capacity; C, carbon; P, phosphorus; Ca, calcium; Mg, magnesium; and K, potassium.

Source: Adapted from Moberg^[14] and Hartemink.^[13]

THE EFFECTS OF SOIL EROSION ON SOIL CHEMICAL PROPERTIES

The on-site and off-site effects of soil erosion are well documented. In tropical regions where many soils suffer from an inherent low fertility that is mostly concentrated in the topsoil, loss of topsoil by soil erosion means a serious reduction in soil chemical fertility.^[15–17] Moberg^[14] compared soil fertility properties of Oxisols developed from sandstone in eroded and non-eroded coffee plots and in virgin land near Lake Victoria, Tanzania. Coffee gardens where erosion occurred were more acid and had lower levels of soil fertility than non-eroded soils with coffee, the levels of which were comparable to virgin soils (Table 2).

EROSION CONTROL

Soil erosion is likely to occur during land preparation and when the tree crop is immature, but several well-established measures exist and are being used. On oil palm plantations, soil erosion is commonly checked by early cover crop establishment, strategic placement and treatment of pruned fronds, and old palm trunks with felling, terracing, the construction of silt pits, and mulching with empty fruit bunches.^[12] Research in Kenya indicated that soil erosion in fields with young tea can be effectively controlled by either mulching or interrow planting of oats although oats may compete with young tea for water and nutrients.^[10] In general, cover crop establishment and terracing sufficiently control erosion in tree crop plantations in tropical regions.

CONCLUSION

Soil erosion under perennial crops is relatively low, provided the crops are well managed.^[13] Measured data show that erosion could be high in oil palm,^[12] cocoa,^[7] and tea^[10]—particularly during establishment or when plantations are neglected. Agroforestry research has

accumulated considerable evidence confirming lower erosion in land use systems with tree crops than with perennial plantation crops.^[18,19] The available data confirm that soil erosion is lower in land use systems with perennial crops than under annual cropping: land use systems with perennial crops form better protection against soil erosion than annual crops simply because they cover the soil throughout the whole year. The risks of soil erosion under plantation crops differ among crop species depending on their canopy characteristics and soil management system.^[20,21] As soil erosion affects the soil chemical fertility, the fertility of the soil should be taken into account when interpreting erosion losses. In general, tolerable losses from acid and poor fertility soils should be appraised much lower than losses from inherently fertile soils.

REFERENCES

1. Morgan, R.P.C. *Soil Erosion and Conservation*; Longman: Harlow, 1995.
2. Chew, P.S.; Kee, K.K.; Goh, K.J. Cultural practices and their impact. In *Oil Palm and the Environment—A Malaysian Perspective*; Singh, G., Huan, L.K., Leng, T., Kow, D.L., Eds.; Malaysian Oil Palm Growers' Council: Kuala Lumpur, 1999; 55–81.
3. Ferwerda, J.D. Oil palm. In *Ecophysiology of Tropical Crops*; Alwim, T.P., Kowlowski, T.T., Eds.; Academic Publisher: New York, 1977; 351–382.
4. Bornemisza, E. Nitrogen cycling in coffee plantations. *Plant Soil* **1982**, *67*, 214–246.
5. Ataroff, M.; Monasterio, M. Soil erosion under different management of coffee plantations in the Venezuelan Andes. *Soil Technol.* **1997**, *11*, 95–108.
6. Beer, J.; Muschler, R.; Kass, D.; Somarriba, E. Shade management in coffee and cacao plantations. *Agroforest. Syst.* **1998**, *38*, 139–164.
7. Hashim, G.M.; Ciesiolka, C.A.A.; Yusoff, W.A.; Nafis, A.W.; Mispan, M.R.; Rose, C.W.; Coughlan, K.J. Soil erosion processes in sloping land in the east coast of Peninsular Malaysia. *Soil Technol.* **1995**, *8*, 215–233.
8. Hudson, N. *Soil Conservation*; B T Batsford Limited: London, 1986.

9. Hartemink, A.E. Nutrient stocks, nutrient cycling, and soil changes in cocoa ecosystems—A review. *Adv. Agrono.* **2005**, *86*, 227–253.
10. Othieno, C.O. Surface run-off and soil erosion on fields of young tea. *Trop. Agr.* **1975**, *52*, 299–308.
11. Botschek, J.; Neu, A.; Skowronek, A.; Jayakody, A.N. Agricultural suitability of degraded Acrisols and Lixisols of former tea lands in Sri Lanka. *Zeitschrift für Pflanzenernährung und Bodenkunde* **1998**, *161*, 627–632.
12. PORIM. In *Environmental Impacts of Oil Palm Plantations in Malaysia*; Palm Oil Research Institute of Malaysia: Malaysia, 1994.
13. Hartemink, A.E. *Soil Fertility Decline in the Tropics—with Case Studies on Plantations*; ISRIC-CABI Publishing: Wallingford, 2003.
14. Moberg, J.P. Some soil fertility problems in the west lake region of Tanzania, including the effects of different forms of cultivation on the fertility of some Ferralsols. *E. Afr. Agr., For. J.* **1972**, 35–46.
15. Lal, R. Soil degradative effects of slope length and tillage method on Alfisols in Western Nigeria. 2. Soil chemical properties, plant nutrient loss and water quality. *Land Degrad. Dev.* **1997**, *8*, 221–244.
16. Ruppenthal, M.; Leihner, D.E.; Steinmuller, N.; Elsharkawy, M.A. Losses of organic matter and nutrients by water erosion in cassava-based cropping systems. *Exp. Agric.* **1997**, *33*, 487–498.
17. Zobisch, M.A.; Richter, C.; Heiligt, B.; Schlott, R. Nutrient losses from cropland in the central highlands of Kenya due to surface runoff and soil erosion. *Soil Till. Res.* **1995**, *33*, 109–116.
18. Young, A. *Agroforestry for Soil Management*, 2nd Ed.; CAB International: Wallingford, 1997.
19. Sanchez, P.A. Science in agroforestry. *Agroforest. Syst.* **1995**, *30*, 5–55.
20. Lal, R. *Soil Erosion in the Tropics—Principles and Management*; McGraw-Hill: New York, 1990.
21. Lal, R. A brief review of erosion research in the humid tropics of south east Asia. In *Soil Conservation and Management in the Humid Tropics*; Greenland, D.J., Lal, R., Eds.; John Wiley: Chichester, 1979; 203–212.

Erosion: Sedimentation Control Amendment Techniques

X.-C. (John) Zhang

Grazinglands Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), El Reno, Oklahoma, U.S.A.

Abstract

Any material that is added to soil to benefit the soil or plant is called a soil amendment. A closely related term to soil amendment is soil conditioner. Soil amendments are either incorporated into the soil or applied on the surfaces. Soil amendments, such as animal waste, farmyard manure, green manure, crop residues, and composts, have been used for many centuries to recycle plant nutrients and to improve soil structure and tilth. Crop residues, preserved by no-till and conservation tillage, have been widely used to reduce soil erosion. Due to increasing landfill costs, applications of municipal and industrial wastes and by-products to agricultural lands have increased dramatically in past decades. Further, interests in using synthetic polymers to reduce water runoff and soil erosion have been revived. All these soil amendments have beneficial effects on the physical, chemical, and biological properties of soil and, therefore, on plant growth. Most amendments are effective in reducing soil erosion by water, although only a few are effective in controlling erosion by wind.

INTRODUCTION

Any material that is added to soil to benefit the soil or plant is called a soil amendment.^[1] For example, lime, gypsum, animal manure, green manure, sewage sludge, compost, crop residue, woodchips, leaves, sawdust, synthetic polymers, and industrial waste such as paper sludge are all soil amendments. Soil amendments, though arbitrary, may be grouped by their origins into five different categories (Table 1). Technically, chemical fertilizers are soil amendments, but conventionally they are not referred to as such. A closely related term to soil amendment is soil conditioner. A soil conditioner is a substance that has the ability to improve the physical condition of a soil for better aeration, water-holding capacity and infiltration, porosity, structural stability, erodibility (susceptibility to erosion), and tilth. Actually, soil conditioners are also soil amendments that are used to create favorable physical environments for plant growth.

Soil amendments are either incorporated into the soil or applied on the surfaces. Incorporated materials ameliorate soil properties, while surface-applied materials mostly affect the atmosphere–soil interface. Soil amendments, such as animal waste, farmyard manure, green manure, crop residues, and composts, have been used for many centuries to recycle plant nutrients and to improve soil structure and tilth. Crop residues, preserved by no-till and conservation tillage, have been widely used to reduce soil erosion.^[4] Due to increasing landfill costs, applications of municipal and industrial wastes and by-products such as sewage sludge, paper mill sludge, wood products, and gypsum to agricultural lands have increased dramatically in past decades.^[5] Interests in using synthetic polymers to

reduce water runoff and soil erosion have been revived. All these soil amendments have beneficial effects on the physical, chemical, and biological properties of soil and, therefore, on plant growth.^[1,4,5] In relation to erosion control, most soil amendments are effective in increasing soil aggregation and soil structural stability,^[6] decreasing erosive forces of raindrop impact and flowing water, reducing surface crusting (a thin, dense surface layer with very low water permeability), and improving water infiltration. These beneficial effects not only reduce soil erodibility but also rainfall-runoff erosivity. Thus, most amendments are effective in reducing soil erosion by water, although only a few (e.g., standing residue cover) are effective in controlling erosion by wind.

TYPES OF SOIL AMENDMENTS

Lime

Liming materials such as quicklime (CaO) and limestones are the most widely used mined or mineral soil amendments. Liming soil has a long history in the United States. Early settlers started to lime soil from the beginning of the 19th century. By the early 20th century, liming had become a routine agricultural practice to raise soil pH and increase crop productivity. Liming soil generally improves the physical and chemical properties of soil^[1,7] by neutralizing soil acidity, reducing phyto-toxicity of aluminum (Al), manganese, and hydrogen ions, increasing plant availability of many nutrients, promoting microbial activity, and increasing clay structural stability. However, over-liming can result in clay dispersion and aggregation disintegration in

Table 1 Type and example of commonly used soil amendments.

Type	Example
Mined or mineral	Lime, gypsum
Synthetic polymer	PAM
Manure and compost	Animal manure, green manure
Mulches	Crop residue, woodchips
Municipal and industrial wastes	Sewage sludge, paper mill sludge

Source: From Zhang & Miller.^[2,3]

highly weathered, high Al^{3+} soils. Beneficial effects of liming on soil erosion result from reduction of soil erodibility because of improved soil aggregation and structural stability and increase of surface cover due to enhanced plant growth. Lime has also been successfully applied as a stabilizer and cementing agent to control side sloughing of canal embankments and pipe erosion on earthen dams, which are composed of highly dispersive and expansible clays.^[1,8]

Gypsum

Gypsum (mined or industrial by-products) has been widely used for many decades to ameliorate unstable, high sodium (Na) soils (e.g., sodic soils). In years, research has demonstrated that soils with low Na and electrolyte concentration (EC) can also be dispersive, and gypsum addition is effective in reducing surface sealing and crusting, increasing water infiltration, and reducing soil erosion by water.^[9,10] The beneficial effects of gypsum addition on soil erosion reduction result from increased soil aggregation and structural stability due to elevated EC in soil solution and the predomination of calcium ions on the exchange sites. Cation valence and EC are two major factors influencing clay behavior, i.e., clay flocculation or dispersion.^[11] When two clay particles approach each other, their electric double layers (a negatively charged clay surface surrounded by a compensating cation cloud) begin to overlap or repel each other. The cation cloud is pushed toward the particle surface when EC is increased or when a divalent cation replaces a monovalent cation. The reduction in the cloud thickness results in a decrease in this repulsive force. When a critical EC is reached, at which the van der Waals attractive force overcomes the repulsive force, rapid coagulation takes place. This concentration is termed critical flocculation concentration (CFC). It should be pointed out that cation valence has a dominant effect on clay flocculation. This is the reason for using Al salts, such as Alum, to flocculate suspended particles in many water-treatment plants. Among the common soil cations (e.g., Al^{3+} , Ca^{2+} , Mg^{2+} , K^+ , Na^+ , and H^+), Ca^{2+} has the greatest flocculation power excluding Al^{3+} (which is only significant in acid soils), while

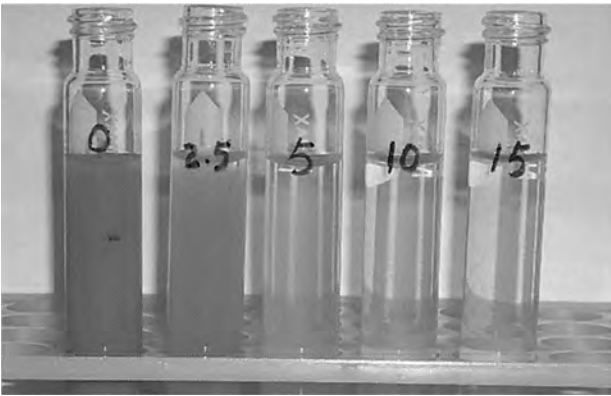


Fig. 1 Flocculation test series after 6-hour settling using Cecil soil clay. (From left: 0, 2.5, 5, 10, and 15 mmol_e gypsum/L).

Na^+ has the least. This is why sodic soils have poor soil structure and gypsum addition helps to improve it. On the basis of this theory, clay particles flocculate when EC is greater than CFC, otherwise dispersion takes place. Clay flocculation is clearly shown in Fig. 1. After 6-hour settling, more dispersed clay particles have coagulated into microaggregates and settled at the bottom as gypsum concentrations increase from left to right. Clay flocculation promotes soil aggregation and structural stability, while clay dispersion often leads to deterioration of soil structure, partially because clay is one of the primary cementing agents for soil aggregation. Increased soil aggregate stability decreases soil erodibility, improves water infiltration, and reduces soil erosion by water.^[1,9,10]

Synthetic Polymer

Synthetic polymers with various charge and chemical properties have been used to improve soil physical properties. In early studies, polymers were typically



Fig. 2 Soil aggregates of a Miami silt loam soil both without (left) and with (right) an anionic PAM treatment after exposure to a simulated rainfall event.

Table 2 Effects of gypsum, anionic PAM, and canopy cover on water infiltration and soil loss in a Cecil sandy loam soil during an initial 31-minute simulated rain at an intensity of 88/mm/hr.

Treatment	Total infiltration (mm)	Percent runoff (%)	Erosion rate (kg/m ²)
Untreated	20.5	54.7	1.63
Canopy cover	27.2	39.9	0.67
Gypsum	26.5	41.4	1.13
PAM	43.2	4.5	0.01

incorporated into the entire plow layer to promote structural stability and water permeability. The enthusiasm reached its peak in the 1950s upon reports that polymers were effective in reducing erosion, improving water infiltration, and increasing crop yields. However, the enthusiasm was subdued by high cost and large quantities needed to amend deep soil layers. In years, interest has been revived by new polymer products and new application strategies. Only a small amount of polymer, when sprayed on the soil surface or dissolved in irrigation water, can reduce surface seal and crust formation and, therefore, reduce runoff and soil erosion.^[1] Surface sealing, formed by disintegration of soil aggregates and dispersion of clay particles, causes extremely low water permeability. The threadlike large polymer molecules or interwoven polymer nets are strongly (often irreversibly) adsorbed by soil aggregates and clay particles. The adsorbed polymer nets that surround soil aggregates physically protect them from disintegration by raindrop impact or shear force imparted by flowing water. This is shown in Fig. 2, where soil aggregates, which have been treated with an anionic polyacrylamide (PAM), are more stable than untreated aggregates after exposure to a 65-mm/hr rain for 10 minutes. Polymers can also chain clay particles together to promote flocculation. Polymers have been used to flocculate clay

particles in sedimentation ponds. For anionic polymers, the effectiveness is usually improved by adding an electrolyte, such as gypsum, because cations are basic ingredients for building cation bridges between negatively charged clay platelets and polymer molecules. Because of the ability to stabilize soil aggregates and improve soil structure, polymers are effective in controlling surface seal and crust formation, increasing water infiltration, and reducing soil erosion. Polymers have also been successfully used to control soil erosion on steep slopes on construction sites and road banks.

Manure, Compost, and Organic Sludge

Animal, farmyard, and green manures have been used as organic soil amendments to improve soil productivity for many centuries. In years, application of municipal biosolids (sewage sludge) and industrial waste compost such as paper sludge to agricultural lands has increased dramatically. These materials have the ability to increase aggregate stability, porosity, and water-holding capacity, reduce runoff and erosion, improve soil tilth, increase plant nutrient availability and soil cation exchange capacity, and elevate soil microbial activities.^[9] Decomposed organic substances such as humus along with soil clays are primary binding agents for soil aggregation. An increase in soil organic matter content often leads to increased soil aggregation and structural stability. Thus, the addition of organic materials such as manure and compost can reduce soil erosion by water by improving soil aggregation and reducing runoff volumes. In addition, improved soil conditions enhance rooting and plant growth, which increases plant cover and root biomass that further reduce soil erosion. It should be pointed out that substantial rates of application over time are needed to considerably increase soil humus levels.

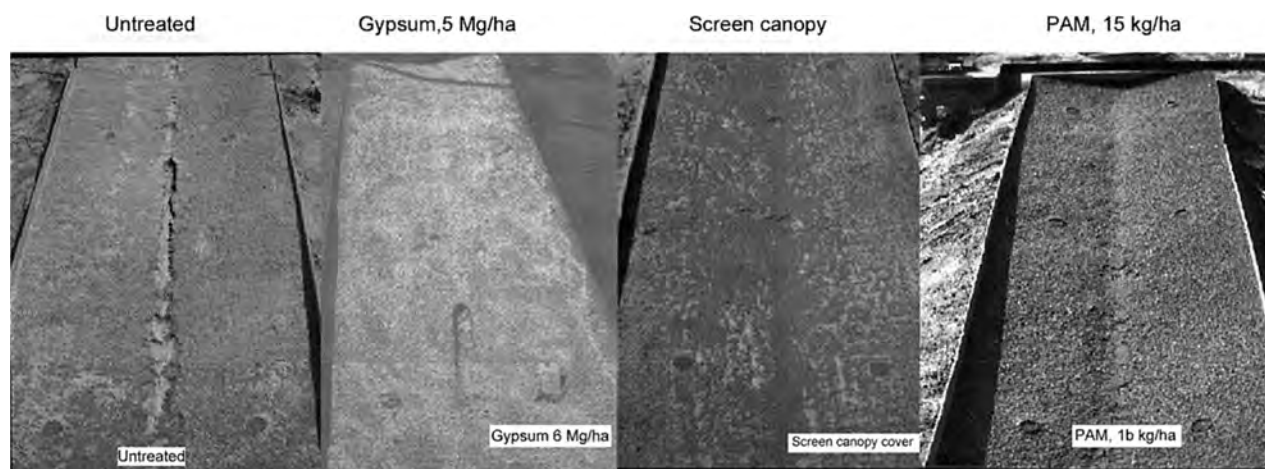


Fig. 3 Rill development and surface conditions of four surface treatments (from left: untreated, 5/Mg/ha gypsum, screen canopy cover, and 15/kg/ha PAM) after exposed to an 88-mm/hr rain for 31 minutes (soil: Cecil, sandy loam; plot size: 3.5 by 0.9/m; slope: 11%).

Mulches

Most mulch materials used for runoff and erosion control are naturally occurring soil amendments. Both inorganic and organic mulches have been used. Inorganic mulches include stone, gravel, sand, and many others, while organic mulches are normally undecomposed materials derived from living matters such as crop residue, hay, leaves, woodchips, and sawdust. These materials dissipate raindrop impact energy, protect surface soil aggregates from disintegration by raindrop impact, prevent surface seal formation, reduce raindrop-impact-induced turbulence, obstruct overland flow, and enhance water infiltration. These processes not only reduce soil detachment by raindrop impact but also suppress sediment transport by thin overland flow. Therefore, surface mulches are very effective in reducing surface runoff and soil erosion by water. Many studies have shown that both runoff volumes and soil loss rates decrease rapidly (often exponentially) with increase of surface mulches.^[4,12,13] Buried or incorporated residues are also effective in reducing soil erosion, especially rill erosion (erosion by concentrated overland flow in small channels). However, the reduction is generally less as compared to surface residues. Moreover, organic mulches eventually decompose into humus and impart to the soil all the beneficial qualities derived from traditional organic matter additions as discussed earlier.

CONCLUSION

An example from field studies^[2,3] is summarized in Table 2 to illustrate the general effects of soil amendments on erosion control. Compared with an untreated plot, the canopy cover, gypsum, and PAM treatments were all effective in reducing runoff and erosion. In this example, PAM was the most effective treatment followed by the cover treatment. This was also illustrated by the surface conditions in Fig. 3. A rill was fully developed in the untreated plot during a 31-minute rain; however, only a small rill was initiated in the gypsum plot, and no rill incision occurred in the cover and PAM plots. Also, the resultant surface conditions differed among different treatments. A smoother surface, on the untreated and the cover

plots, indicates disintegration of soil aggregates and seal/crust formation. In contrast, a well-aggregated, rough surface was exhibited on the PAM plot. This explains why almost all rainfall water has infiltrated into the soil on this plot. In general, the effectiveness of these treatments declined with time and the treatment longevity varied from weeks to months. To maintain the effect, PAM and gypsum must be reapplied at some intervals, which depend largely upon soil and environmental conditions.

REFERENCES

1. Wallace, A.; Terry, R.E., Eds. *Handbook of Soil Conditioners: Substances That Enhance the Physical Properties of Soil*; Marcel Dekker: New York, 1998.
2. Zhang, X.C.; Miller, W.P. Physical and chemical crusting processes affecting runoff and erosion in furrows. *Soil Sci. Soc. Am. J.* **1996**, *60* (3), 860.
3. Zhang, X.C.; Miller, W.P. Polyacrylamide effect on infiltration and erosion in furrows. *Soil Sci. Soc. Am. J.* **1996**, *60* (3), 866.
4. Oschwald, W.R., Ed. *Crop Residue Management System*; ASA Spec. Publ. No. 31; ASA, CSSA, and SSSA: Madison, 1978.
5. Rechcigl, J.E.; MacKinnon, H.C., Eds. *Agricultural Uses of By-Products and Wastes*; ACS Symp. Series 668; ACS: Washington, 1997.
6. Emerson, W.W.; Bond, R.D.; Dexter, A.R., Eds. *Modification of Soil Structure*; John Wiley & Sons: New York, 1978.
7. Adams, F., Ed. *Soil Acidity and Liming*; Agron. Monogr. 12; ASA, CSSA, and SSSA: Madison, 1984.
8. Gutschick, K.A., Ed. *Lime for Environmental Uses*; ASTM: Philadelphia, 1985.
9. Sumner, M.E.; Stewart, B.A., Eds. *Soil Crusting: Chemical and Physical Processes*; Lewis: Boca Raton, 1992.
10. Shainberg, I.; Sumner, M.E.; Miller, W.P.; Farina, M.P.W.; Pavan, M.A.; Fey, M.V. Use of gypsum on soils: A review. *Adv. Soil Sci.* **1989**, *9*, 1.
11. Van Olphen, H. *An Introduction to Clay Colloid Chemistry*; John Wiley & Sons: New York, 1977.
12. Unger, P.W., Ed. *Managing Agricultural Residues*; Lewis: Boca Raton, 1994.
13. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses: A Guide to Conservation Planning*; USDA Agriculture Handbook 537; U.S. Government Printing Office: Washington, 1978.

Erosion: Sedimentation Control Vegetative Techniques

Samson D. Angima

Oregon State University, Corvallis, Oregon, U.S.A.

Abstract

Soil detachment and erosion occur when soil is disturbed by either human activity or natural conditions such as extreme weather. As a result of this, the soil is moved from one point to another by either wind or water. Practices such as road construction, suburban and industrial developments, stream channel and other types of construction on sloping lands, inadequate drainage facilities, poor grading practices, deforestation, cultivation on sloping lands, and general lack of adequate planning by land users cause soil erosion. Consequences of soil erosion include water and air pollution, reduced land productivity from loss of topsoil and nutrients, and degradation of the environment. Soil erosion by water is controlled mainly by earthworks and engineered constructions such as terracing aimed at reducing the slope and at collecting and storing moisture while reducing runoff to acceptable limits. Vegetative barriers are grass, shrubs, and small trees grown in close rows that can be used to control both water and wind erosion by providing protection from soil and dislodging sources such as rainfall and offering a semipermeable barrier to erosion agents resulting in soil deposition. The resultant vegetation also shields the soil surface from overland flow and decreases the erosive capacity of water flow by reducing its velocity.

VEGETATIVE MATERIALS FOR SOIL EROSION AND SEDIMENTATION CONTROL

Vegetative cover is essential for the design and stabilization of many structural erosion-control devices (Fig. 1). Proper vegetative cover provides excellent erosion protection and sedimentation control. Vegetative barriers are planted in close rows along contours on slopes to intercept water runoff or are planted perpendicular to the direction of wind to retard wind movement, resulting in soil deposition. Plant roots and lateral stems provide a structure to hold soil particles in place. These features also improve the soil's physical properties and increase infiltration rate, thereby decreasing runoff. Plant transpiration reduces soil moisture levels, which increases soil absorption capabilities. Not every plant material can serve as a soil erosion-control agent. Those plants that possess some bioengineering characteristics in both the root and the shoot systems that encompass both living plants and organic materials as construction elements for erosion control. Such properties include noncompetition with adjacent crops or fruit trees for moisture or nutrients, a rooting system that reaches deep down to anchor the plant and also extract leached nutrients from the subsoil and shade tolerance. The plants must be perennials with high seed vigor, have the ability to increase soil organic matter and to reduce surrounding soil bulk density to allow faster water infiltration, and have strong woody stalks to withstand pressure from erosive agents.

Vegetative materials used for soil erosion control include grass species, legumes, trees, shrubs, vetches, and sods. Common grass species that bunch during growth

include Tall fescue, Perennial ryegrass, Orchardgrass, Timothy, Switchgrass, Weeping lovegrass, Deertongue, and Big bluestem. Bunching legumes include birdsfoot trefoil and Sericea lespedeza. Cereals such as winter wheat, winter rye, spring oats, Sudangrass, and Japanese millet are also used. Sod-forming species include grasses such as redbud, fine fescue, Kentucky bluegrass, smooth brome-grass, and legumes such as crownvetch and flatpea.^[1] Shrubs and small trees that are used as barriers in erosion control include *Calliandra calothyrsus*, *Sesbania sesban*, *Leucaena leucocephala*, *Gliricidia sepium*, *Cassia siamea*, *Eucalyptus* spp., *Casuarina* spp., *Acacia* spp., *Azadirachta indica*, and *Grevillea robusta*.^[2] Tree species selection depends on local land use and climatic conditions that favor establishment.

MASS PLANTING OF VEGETATIVE COVER

Mass planting of vegetative cover is done where land that is susceptible to erosion is converted from cropland to vegetative cover such as in the conservation reserve program. In this program, trees as well as sods are planted in strategic places, where they will provide maximum protection to the soil resource, and the landowner agrees to leave the areas under vegetation for a given period of time for land stabilization to occur. The vegetative cover also reduces water pollutants such as fertilizer nutrients, pesticides, and herbicides in the runoff water, increases oxygen levels, reduces greenhouse gases and evaporation rates, and provides shade and buffers against high winds.^[2]



Fig. 1 Vegetative barriers/strips hold the terraces intact for agricultural production in Kabale, Uganda.

Source: Photo courtesy of Samson Angima.

VEGETATIVE STRIPS

Vegetative strips constitute different types of filter strips that reduce runoff velocity and provide differing degrees of filtering action depending on the species used. Larger soil particles tend to settle out readily, leaving only clay particles suspended and thereby reducing pollutant transfer to ponds, rivers, and lakes. Filter strips can be classified as grass strips, vegetated waterways, filter strip terraces, buffer and riparian strips, and settling basins.^[3]

Grass Strips

Grasses are by far the most important vegetative materials used to control erosion. Bands of grass about 1–2 m in width are planted along the contours and spaced every 10–100 m depending on the slope (Fig. 2). Prior to planting, the land is roughened by disking, harrowing, or raking and then limed, fertilized, and seeded with mixtures of adapted grasses and legumes to enhance a good stand.^[4] Vetiver grass (*Vetiveria zizanioides*), Napier grass (*Pennisetum purpureum*), and Stiff grass (*Miscanthus sinensis*) are the most commonly used species.^[2] These are upright, tufted, deep-rooted, and very dense grasses that are by far the most important in erosion control, particularly in tropical countries. Vetiver is a bunch grass with very rapid growth in warm and moist conditions. It grows to more than 2 m in height and has a remarkably dense and vertical rooting system, growing deep (3–5 m). Napier grass is a tall perennial reaching over 3 m high. It is resistant to drought and grows at altitudes up to 2400 m. Stiff grass has dense roots and coarse stems that withstand erosive agents. These grass hedges present a virtually impenetrable barrier through which soil can hardly pass and only water, with much reduced velocity, passes.^[5] As silt builds up behind the grass, other grass shoots arise from the nodes above the

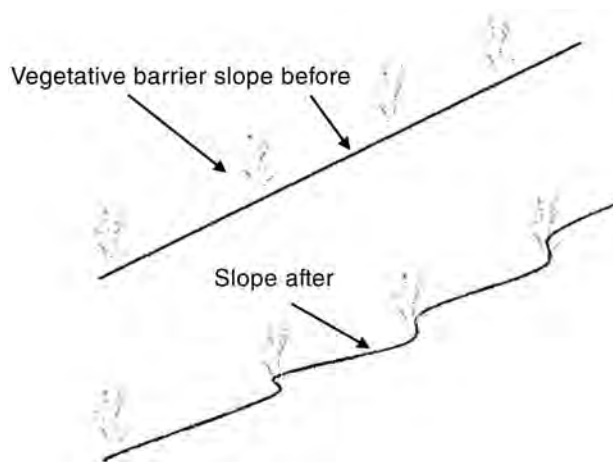


Fig. 2 Schematic sketch of vegetative barriers illustrating expected change in land slope over time resulting from tillage and erosion deposition process.

deposited silt to form a natural terrace. Weeds and undesirable foreign grasses are unable to penetrate through a well-established grass hedge. Studies carried out in Mississippi, using stiff grass (*M. sinensis*) on no-till and conventional tillage,^[6] showed that grass strips help reduce sediment losses by up to 88% on conventional tillage and 57% on no-till cotton plots (Table 1). Other grass species include Buffalo grass (*Buchloe dactyloides*), grama or mesquite grasses (genus *Bouteloua*), Switchgrass (*Panicum virgatum*), and *Phalaris aquatica*.

Vegetated Waterways

Vegetated waterways are channels or waterways that transfer runoff from a higher to a lower elevation over a short distance without allowing erosion to occur (Fig. 3). In addition to dissipating flow energy, some structures also act to retain soil. They are especially effective in arresting gully development, a situation that might need both structure(s) and some vegetated channels. They have many uses in comprehensive conservation plans, but they primarily collect and concentrate flows and then safely transport the water to major drainage systems. Dense vegetation is used to minimize the area required, i.e., the protective action of the vegetation permits higher flow velocities and thus smaller waterway cross sections. Most vegetated waterways run directly down a slope; however, they can also be constructed somewhat across the slope as diversions or sometimes just to reduce channel slope. Vegetated channels are not used where continuous flow occurs because the vegetation will die out. Tillage near a waterway is accomplished in a direction across the waterway. Sod-forming, cool-season grasses such as smooth brome or Western wheatgrass are used in grass waterways. Grass waterways are usually designed to carry runoff of a 24-hour storm of the intensity that happens once every 10 years. In areas with

Table 1 Sediment loss with and without grass strip (*M. sinensis*) under no-till and conventional tillage cotton at 5% slope.

Period	Sediment loss (t/ha)				
	No-till with grass strip	No-till without grass strip	Conventional tillage with grass strip	Conventional tillage without grass strip	No-till with winter wheat cover
1992	2.6	4.5	12.3	60.3	2.9
1993	0.9	1.6	5.4	21.9	1.2
1994	3.2	9.6	18.6	63.2	1.9
Average	2.2	5.2	12.1	48.5	2.0

Source: From McGregor, Dabney, et al.^[6]

prolonged water flows, high water tables, or seepage problems, a rock-lined center is added.^[1]

Filter Strip Terraces

Filter strip terraces are strips of grass sod, legumes, and other vegetation on the contour that surface water runoff crosses as it runs downhill. They serve as an alternative to earthen terraces but do not have a channel to conduct water along the contour, as earthen terraces do. Filter strip terraces are excellent removers of sediment, pesticides, organic matter, and other pollutants. They are better than grass waterways because water enters the strip uniformly and over a wide area.^[3] The width and type of vegetation established in the filter area are determined by site conditions including soil type, land slope, and the type of runoff entering the filter.

Small trees and shrubs are also used as conservation hedges in erosion control. These are planted close together and are periodically pruned to maintain height while the cut branches are inserted upslope of the hedge to trap more sediment. Trees that have been used successfully in erosion control, especially in agroforestry systems in the tropics, include *C. calothyrsus*, *S. sesban*, *L. leucocephala*, *G. sepium*, and *C. siamea*. Trees used for windbreaks

include *Eucalyptus* spp., *Casuarina* spp., *Acacia* spp., Neem tree (*A. indica*), and *G. robusta*.^[7] In the temperate zone, thorny hedge plants include barberry, Osage orange, buckthorn, and hawthorn. Evergreen hedge plants are box, privet, azalea, yew, arborvitae, rhododendron, mountain laurel, and holly.^[8] Decorative deciduous shrubs often used are lilac, forsythia, mock orange, Spiraea, euonymus, and viburnum. *Rosa rugosa* can be planted along the highway embankments, and rows of poplars, hemlocks, and other trees can be used as shelterbelts. Vegetative or biological measures may include log bundles anchored to the stream bank or the planting of herbaceous or woody plants, which can withstand high velocity flow, while the roots form a protective net for the soil.

Buffer Strips and Riparian Strips

Buffer strips at lower elevations of fields (Fig. 4) and riparian strips along stream banks, ponds, and lakes (Fig. 5) intercept surface runoff water from crop fields. Buffer strips may constitute ordinary grassed fencerows or strips of grasses, shrubs, and trees lining hillsides or banks of rivers. Runoff water must flow in a shallow, even layer across the buffer strip to remove sediments. Most common grasses used in buffer strips are Bluestem and Indiangrass. Riparian

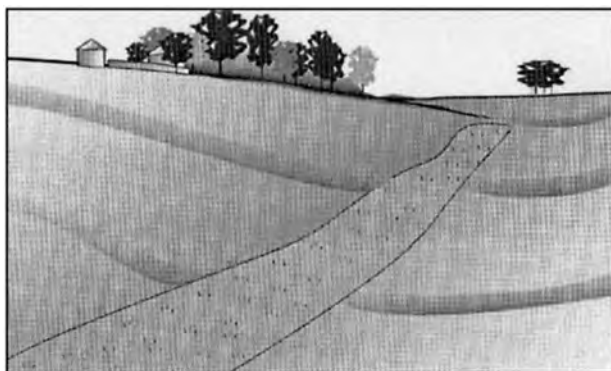


Fig. 3 Grass waterways are the most common type of vegetative filter strip.

Source: From Regehr, Devlin, et al.^[3]

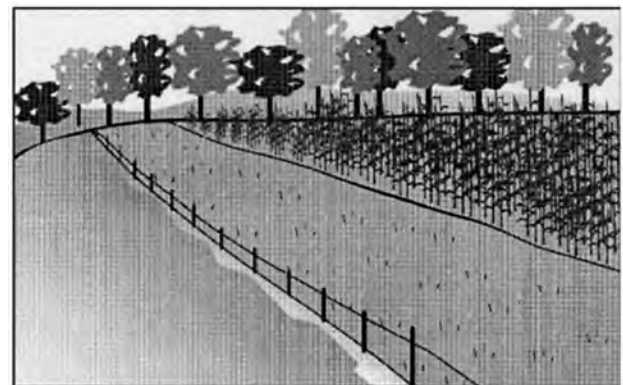


Fig. 4 Buffer strips at lower elevations of fields intercept surface runoff water from crop fields.

Source: From Regehr, Devlin, et al.^[3]

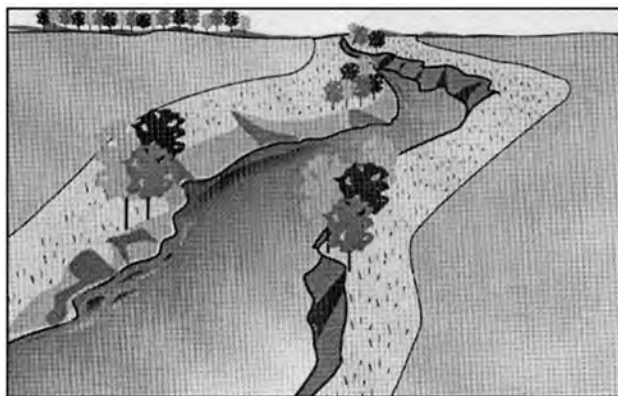


Fig. 5 Riparian strips along stream banks intercept surface runoff water from crop fields.

Source: From Regehr, Devlin, et al.^[3]

strips are planted so that surface and subsurface runoff must filter through them before it reaches a pond, lake, or stream.^[1] The body of water can be permanent or temporary. Riparian strips can also be placed next to wetlands, such as marshy or swampy areas, and additional vegetation can be placed uphill if excessive amounts of sediments enter the waters.

Settling Basins

Settling basins are constructed around inlets to tile-outlet terraces. These are important in reducing soil sediment loads and also act as setback zones for herbicides such as Atrazine. The basins are designed to retain water for up to 24 hours, giving most of the larger sediments time to settle out.^[3] These are common where tile lines have been installed.

CONCLUSION

Vegetative techniques for erosion and sedimentation control offer a wide range of tools for use by a wide range

of land users especially those that cannot afford engineered or earthen structures. However, for these tools to work well in erosion prevention, they have to be regularly maintained and landowners need to understand that they require time to effectively control enough sediment that can gradually change slope. Research is adding more species of grass and/or shrubs that can be multipurposely used to control erosion and at the same time provide other uses beneficial to the landowner. When used well in conjunction with best management practices, vegetative techniques can substantially contribute to overall water and wind erosion control.

REFERENCES

1. Landschoot, P. *Erosion Control and Conservation Plantings on Non Cropland*; Penn State College of Agricultural Sciences Extension Publication, 1997.
2. Young, A. *Agroforestry for Soil Conservation*; CAB International: Wallingford, 1989; 197–229.
3. Regehr, D.L.; Devlin, D.L.; Barnes, P. *Using Vegetative Strips in Crop Fields*; Kansas State University, 1996. Available at <http://www.oznet.ksu.edu>
4. Dabney, S.M.; Liu, Z.; Lane, M.; Douglas, J.; Zhu, J.; Flanagan, D.C. Landscape benching from tillage erosion between grass hedges. *Soil Till. Res.* **1999**, *51*, 219–231.
5. Dabney, S.M.; McGregor, K.C.; Meyer, L.D.; Grissinger, E.H.; Foster, G.R. Vegetative barriers for runoff and sediment control. In *Integrated Resources Management and Landscape Modification for Environmental Protection*; Mitchell, J.K., Ed.; American Society of Agriculture Engineers: St. Josephs, 1993; 60–70.
6. McGregor, K.C.; Dabney, S.M.; Johnson, J.R. Runoff and soil loss from plots with and without stiff-grass hedges. *Trans. Am. Soc. Agr. Eng.* **1999**, *42* (2), 361–368.
7. Angima, S.D.; O'Neill, M.K.; Omwega, A.; Stott, D.E. Use of tree/grass hedges for soil erosion control in the Central Kenyan highlands. *J. Soil Water Conserv.* **2000**, *55* (4), 478–482.
8. Nair, P.K.N. *An Introduction to Agroforestry*; Kluwer Academic Publishers: Boston, 1993; 325–343.

Erosion: Snowmelt

Donald K. McCool

U.S. Department of Agriculture (USDA), Pullman, Washington, U.S.A.

Abstract

This entry reviews winter processes including a range of conditions, from those where the soil freezes and thaws diurnally and is subjected mainly to rainfall, to those where the soil remains frozen for several months and precipitation occurs as snow or in other solid form during all or part of the winter season. In some areas, the soil freezes in the autumn and stays frozen until spring, and the ground is blanketed with snow for the entire winter; erosion is then confined to a period of spring snowmelt. Some areas are subjected to several runoff and erosion events each winter because the winter temperatures and precipitation patterns lead to multiple soil freezing and thawing occurrences with accompanying rain or snowmelt events.

INTRODUCTION

In many areas of the world, winter hydrology is an important part of the annual erosion process; in some regions, it is the primary cause of erosion. Sharratt et al.^[1] indicate that about half of the earth's land surface is frozen at some time during the year. In this topic, we will consider winter processes to include a range of conditions from those where the soil freezes and thaws diurnally and is subjected mainly to rainfall to those where the soil remains frozen for several months and precipitation occurs as snow or in other solid form during all or part of the winter season. In areas where daily minimum temperatures are rarely below 0°C, winter erosion processes are of relatively minor importance as compared to spring, summer, and autumn erosion. At the other end of the spectrum are areas where the soil freezes in the autumn and stays frozen until spring and the ground is blanketed with snow for the entire winter. Erosion is then confined to a period of spring snowmelt. Some areas such as the Palouse region of eastern Washington, northern Idaho, and northeastern Oregon are subjected to several runoff and erosion events each winter because the winter temperatures and precipitation patterns lead to multiple soil freezing and thawing occurrences with accompanying rain or snowmelt events. The result of a particularly severe rain and snowmelt event when the soil had thawed at the surface on a fall-seeded field is shown in [Fig. 1](#).

PRECIPITATION

Precipitation in areas where winter processes are important can occur as rain, or in solid form as sleet or snow. Rain can cause erosion due to splash and runoff detachment, whereas sleet and snow cause detachment by runoff as they melt. In general, winter rainfall intensities are lower than summer rainfall intensities because of the lack of thunderstorm

activity in the winter. However, the frequency and duration of precipitation are commonly greater. As water loss from the soil is considerably lower during the winter due to lower mean temperatures and the absence of actively growing crops, even low intensity rainfall can increase the moisture content of soil rather quickly with a resultant dramatic increase in its susceptibility to erosion. The kinetic energy and intensity associated with rainstorms are not a direct factor in snowmelt erosion. Snowmelt erosion is more closely related to volume and peak rate of runoff, which can be strongly influenced by rainfall when it occurs concurrently with the snowmelt.

SOIL

The phase of the water in the surface layer of the soil is important in the winter erosion process. When soil water is frozen, erodibility, the susceptibility of the soil to erosion, is very low and erosion rates under sheet flow conditions are generally quite low. However, when runoff concentrates in small channels, the flow may cut into and through the frozen layer, leaving gullies and deeply incised channels. Soils that have been frozen and are thawing from the surface are weakened due to water expansion during the freezing process. Lee^[2] cited a number of studies that indicate an increase in water content as water moves to the freezing front from deeper in the soil because the area where the soil is freezing is at very high water tension. The soil will regain strength after the frost has thawed and the soil reconsolidates as water drains from the soil.^[3]

Water content of the soil at the time of freezing is important to the permeability of the soil and the opportunity for infiltration and deep percolation. Lee^[2] found a linear relationship with a negative slope between the ratio of frozen soil infiltration hydraulic conductivity to the unfrozen soil infiltration hydraulic conductivity. Soil frozen at water



Fig. 1 Erosion on fall-seeded winter wheat in eastern Washington caused by rain and snowmelt on thawing soil. Measured rill erosion was 200 metric tons per hectare.

content near saturation is frequently impervious and runoff can be nearly 100% of the rainfall or snowmelt,^[2] thus leading to severe concentrated flow and gully erosion (Fig. 2).

EFFECT OF SNOW AND FREEZING CONDITIONS

During the non-winter period or in climatic regions where winter erosion processes are not a factor, the only type of water erosion that occurs is rainfall on unfrozen soil. Winter erosion is more complex because there are seven basic types of erosion events:

1. Rain on frozen soil;
2. Rain on thawing soil;
3. Rain on unfrozen soil;
4. Snowmelt on frozen soil;
5. Snowmelt on unfrozen soil;
6. Rain on snowmelt on frozen soil; and
7. Rain on snowmelt on unfrozen soil.



Fig. 2 Gully erosion from snowmelt runoff in northern Idaho.

Snowmelt on thawing soil and rain on snowmelt on thawing soil are not listed because, as discussed later, these conditions generally do not occur due to the insulating properties of snow. Further complicating these various types of events is that they typically do not stand alone but occur in some combination and how they combine will dictate the severity of any resultant erosion.

A blanket of snow on the soil surface can have positive as well as negative effects with regard to soil erodibility. Snow is an excellent insulator; the deeper the snow the better the insulation. Thus, a snow layer prior to or concurrent with freezing conditions can retard or prevent frozen soil. However, this insulating property also works extremely well at preventing frozen soil from thawing from the surface down (although thawing will occur in an upward direction due to deeper, warmer unfrozen layers if aboveground temperatures are favorable). Snow cover on frozen soil typically results in the soil surface remaining frozen until bare soil begins to appear as the snow melts with warming temperatures, at which time the exposed soil begins to thaw from the surface down (Fig. 3). The potential for severe erosion can be quite high with the presence of snow and depends on how all the contributing factors come together. If rainfall accompanies the warming temperatures required for snowmelt, runoff can be initiated over the frozen soil beneath the snowpack. If these conditions persist as patches of soil begin to be exposed and surface thawing follows, soil erosion will begin. The nature and severity of the erosion will be determined largely by the intensity and duration of the rainfall and to a lesser degree by temperature that affects the rapidity of the thawing process.

RUNOFF EVENTS

It is difficult to ascribe a single set of characteristics to describe winter hydrology events on hillslopes or small watersheds. Events from rainfall on frost-impacted soil can



Fig. 3 Melting snow exposes bare soil to thaw and increases erodibility.

be of very short duration, whereas events resulting from extended periods of snowmelt can be of much longer duration. Events involving rainfall without snowmelt are usually brief although erosion rates can be high if the soil is thawing from the surface. Events involving snowmelt alone are extended in nature; runoff may fluctuate diurnally as air temperature dips below freezing at night. Erosion resulting from snowmelt alone is generally not severe. The highest runoff rates and flooding result from a combination of rainfall and snowmelt with high air temperatures. As long as the soil is frozen, sheet and rill erosion rates may be quite small, although concentrated flow erosion may be significant. When substantial quantities of bare soil appear, and the soil frost starts to thaw from the surface, erosion rates may increase dramatically. Sediment concentration graphs may lead or lag hydrographs under winter conditions. When there is no snow and the soil has thawed at the surface, a small amount of rainfall may lead to very high sediment concentration that occurs early in the event. When snow melts and the soil is not frozen, the sediment concentration and hydrographs may coincide. When snow melts over frozen soil, there may be little sediment until a portion of the soil is bare and melts. Then the runoff from the melting snow will detach and transport sediment, leading to a significant lag of peak sediment concentration.

MODELING

Erosion models are used to estimate location specific soil losses under various management systems within different land uses. They are also used to estimate expected amounts of sediment and chemicals that may be associated with those sediments. Models may be empirical in nature, such as the universal soil loss equation (USLE)^[4] or the revised universal soil loss equation (RUSLE),^[5] or they may be process based such as WEPP.^[6] The inclusion of winter erosion processes even in empirical models such as USLE or RUSLE complicates the models. Erosion relationships based on runoff characteristics rather than rainfall characteristics may be more appropriate, creating the need to estimate runoff. Process-based models require inclusion of soil freezing and thawing, snowfall and snowmelt, and movement of water into frost impacted soil. Simple soil frost models are driven by air temperature, but more complex models consider radiation and other methods of energy

transfer. It is difficult to strike a balance between excessive data requirement and adequate performance for applied use.

AMELIORATION

Preventing erosion damages from winter processes is similar to preventing erosion damages in other seasons. Cropland and rangeland are best protected by crop and surface cover. Snow collected in standing residue can prevent soil freezing and the creation of an impermeable condition. Surface cover protects from splash detachment, insulates the soil, and slows runoff. No-till practices produce root channels and other pores that enable infiltration when the soil is frozen. Likewise, for areas where frost depth seldom exceeds 30–40 cm, deep ripping, chiseling, or slot mulching to expected frost depth can prevent formation of a continuous frost layer and lead to increased infiltration.

REFERENCES

1. Sharratt, B.S.; Radke, J.K.; Hinzman, L.D.; Iskandar, I.K.; Gruenevelt, P.H. Physics, chemistry, and ecology of frozen soils in managed ecosystems: An introduction. In *International Symposium Physics, Chemistry, and Ecology of Seasonally Frozen Soils*; Iskandar, I.K., Ed.; U.S. Army Cold Regions Research and Engineering Laboratory Spec. Rep. 97–10; Hanover, 1997; 1–7.
2. Lee, H.W. *Determination of Infiltration Characteristics of a Frozen Palouse Silt Loam Soil Under Simulated Rainfall*; Ph.D. Dissertation, Doctor of Philosophy in Agricultural Engineering, University of Idaho, Moscow, 1983.
3. Formanek, G.E.; McCool, D.K.; Papendick, R.I. Freeze-thaw and consolidation effects on strength of a wet silt loam. *Trans. ASAE* **1984**, 26 (6), 1749–1752.
4. Wischmeier, W.H.; Smith, D.D. *Predicting Soil Erosion by Water—A Guide to Conservation Planning*; USDA Agriculture Handbook No. 537, 1978; 58 pp.
5. Renard, K.G.; Foster, G.R.; Weesies, G.A.; McCool, D.K.; Yoder, D.C. *Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE)*; USDA Agriculture Handbook No. 703, 1997; 404 pp.
6. Flanagan, D.C.; Nearing, M.A. *USDA–Water Prediction Project: Hillslope Profile and Watershed Model Documentation*; NSERL Report No. 10; USDA-ARS National Soil Erosion Research Laboratory: West Lafayette, 1995.

Erosion: Soil Conservation

Eric T. Craswell

*Fenner School of Environment and Society, College of Medicine, Biology,
and Environment, Australian National University, Canberra, Australian Capital
Territory, Australia*

Abstract

The prospects for better control of soil erosion have improved in years. Our knowledge of the human and biophysical factors affecting soil erosion has been improved through the use of new methods such as Geographic Information Systems (GIS)-based multiagent modeling; in particular, by locating and targeting erosion hotspots in the landscape, the GIS methods can greatly improve the effectiveness of soil conservation programs. Furthermore, in some developed countries, the elaboration of the concept of multifunctionality of agricultural land will foster approaches and practices that take into account important ecosystem services such as biodiversity protection, landscape integrity, carbon sequestration, and watershed protection. For the developing countries, there have been moves to devise schemes to subsidize the ecosystem services provided by land users in upper watershed. Thus it may be prevention of the off-site impacts of soil erosion that motivates an expansion in the use of soil conservation measures. Whatever the motivation, an expanded effort is needed urgently, particularly in the tropics.

INTRODUCTION

The conservation of soil and water resources is a major challenge to humankind because these resources are under stress to produce food and fiber to meet the needs of a burgeoning population. Consequently, poor land management associated with land clearing and agricultural intensification has caused widespread land degradation. Most natural ecosystems with climax vegetation display a dynamic equilibrium between soil formation and erosion. This equilibrium is displaced when vegetation is cleared or overgrazed, and when fire and mechanical cultivation expose soil to water and/or wind erosion. Apart from agricultural intensification, construction of roads and buildings causes serious, but commonly localized, problems of soil erosion. Loss of the soil-surface layer, which contains the highest level of nutrients and organic matter, reduces soil fertility and hence vegetative growth and yields. The broad scale of human-induced land degradation due to soil loss on agricultural lands is indicated by the estimates of Oldeman et al.^[1] Globally, the total area affected by moderate to serious soil erosion is 1028 million hectares, of which 748 million hectares is because of water erosion and the rest by wind erosion. Asia and Africa together make up 673 million hectares of the total affected area. Oldeman also estimates that 186 million hectares are affected by chemical and physical degradation, which reduce vegetative cover and exacerbate soil losses by erosion.

SOIL CONSERVATION— THE EVOLVING PARADIGM

Soil conservation may be defined as the protection of soil against physical loss by erosion or against chemical deterioration. This entry emphasizes soil erosion and its control. Human efforts to combat soil erosion on steepland areas date back to the dawn of recorded history. Indigenous technologies, documented,^[2,3] include a wide range of methods that comprise terraces, minimum tillage, handmade ridges, natural vegetation strips, stone bunds, logs, etc. (Table 1). Some of these technologies, conceived over the centuries in tune with the diverse belief systems of the people, are similar to those based on modern science, which will be discussed later. Soil erosion affects not only the productivity of the individual farm, but also the long-term capacity of a society to produce food for its population. The importance of the social dimensions of soil erosion is reinforced by the off-site impacts of transported soil or sediment in the landscape in the case of water erosion and the environmental impacts on air quality and dust deposition of airborne soil particles in the case of wind erosion. These off-site impacts may affect urban population at long distances from the site of the erosion, and in some spectacular cases—such as the Dust Bowl in the United States in the 1930s—have drawn the attention of policy makers to the need for more investment in soil conservation in rural areas. Models for the most effective community and the government's involvement in soil conservation continue to evolve.

Table 1 Some examples of indigenous technologies for soil conservation and erosion control.

Country	Technology	Description	Source (reference)
China	Ditches along hillsides	Grassed contour ditches between rows of fruit trees	[2]
Ethiopia	Stone bunds	Stones collected and placed in lines on contours	a
India	Kana bandi	Windbreaks of dead wood	[2]
Kenya	Fanya-juu terraces	Soil thrown uphill to form contour bunds	a
Laos	Lao Theung systems	Sustainable shifting cultivation on mid-altitude steeplands	[2]
Nepal	Terraces	Manually built on steep slopes; terrace risers sliced regularly	[2]

Source: ^a<http://www.wocat.net/materials/qtqalist.pdf>.^[3]

In the 20th century, governments in countries with mechanized modern agriculture, such as the United States, recognized the serious problems created by erosion and established institutions to promote and subsidize soil conservation. A land classification scheme was developed to identify high-risk erosion-prone land that was in some cases set aside as natural rangeland. On other classes of vulnerable land, agronomic practices, and/or conservation structures such as contour banks, were introduced on an individual farm basis with the help of government agents. Some developing countries introduced similar schemes. However, these efforts have not been successful in many countries. In the United States, despite massive investments in soil conservation since the 1930s, some experts believe that the rates of soil erosion have not declined.^[4] Many practitioners consider that traditional top-down soil conservation programs should be replaced by systems utilizing community-based approaches. Movements such as the landcare program in Australia^[5] and conservation districts in the United States^[6] are based on a bottom-up approach in which members of rural communities work together to conserve not only the soil and water resources but also, on the basis of a more holistic approach, the biodiversity and landscape integrity.

The limited successes of soil conservation schemes contributed to concerns about global environmental security that in the mid-1980s led to widespread acceptance of the concept of sustainability. The World Commission on Environment and Development defined sustainable development as meeting the needs of the present without compromising the ability of future generations to meet their own needs.^[7] This new thinking also recognized that technological innovation is a necessary but not a sufficient requirement for successful soil and water conservation. A new paradigm that shifted from an emphasis on soil and water conservation to the more holistic goal of sustainable land management was developed. The shift entailed a change from a technology-based approach to an approach designed to preserve and enhance the quality and productivity of global natural and environmental resources, with a view to the dynamic and constantly changing needs of all of the

earth's inhabitants.^[8] The scales of intervention ranged from the household through the community to the international level, involving treaties and conventions.

At the farm and field level, a mechanism is needed for assessing sustainable land management. The framework for evaluating sustainable land management (FESLM) was designed for this purpose.^[9] The FESLM sets the goal of a system that combines technologies, policies, and activities aimed at integrating socioeconomic principles with environmental concerns so as to simultaneously maintain or enhance production/service; reduce the level of production risk; protect the potential of the resource base; be economically viable; and socially acceptable. These objectives—productivity, security, protection, viability, and acceptability—constitute the five pillars of the FESLM.

This broad view of sustainable land management, involving social and economical as well as biophysical dimensions, requires a new community-based approach to institutional support. Farmer participation is the key to success in both developed and developing countries although the most appropriate frameworks for community, institution, and government involvement vary. In developing countries, there is an increased adoption of participatory technology development that involves farmers in the planning, collection, selection, testing, and dissemination of technologies. The landcare approach in Australia is an example of the new approach in the context of a developed country. Landcare was initiated to address the failure of the conventional government soil conservation approach to halt the disturbing rate of natural resource degradation in Australia; in 1996, there were 3000 landcare groups.^[10] The model is based on government support of these landcare groups consisting of a collection of farmers and interested citizens joined together to reverse natural resource degradation. The roles of communities and governments at different levels in underpinning the landcare movement are evolving and are critically important to the ultimate success of these soil conservation efforts. The policy interventions needed in developing countries must take account of credit, land tenure, and pricing policies that strongly influence the actions of farmers in marginal areas.^[11]

Table 2 Technologies for soil erosion control on agricultural land.

Vegetative	Mechanical
Cultivated crop land	Cultivated crop, forest, and grazing land
Mulching	
Crop management	Conservation tillage
Dense planting	Contour tillage
Multiple cropping/agroforestry (contour strips or rotations)	Ridging and ridge tying
Cover cropping	Minimum till and no till
Forest land	
Tree planting	Terracing
Agri silviculture	Waterways
Grazing land	Structures
Plant grasses	
Plant shrubs	
Wasteland	
Plant trees, shrubs, grasses	
Biological engineering	
Wattling and staking	
Brush matting	
Hydro seeding	

Source: From El-Swaify, Dangler, et al.^[12]

EROSION CONTROL

The control of soil erosion can be achieved using a wide range of technologies, both vegetative and mechanical (Table 2). The principles underlying these control measures have been summarized in models such as the USLE, RUSLE, WEPP^[13] and GUEST.^[14] Rainfall impact is a key factor in dispersing and detaching the surface soil, hence soil cover that protects the surface is an important factor in selecting technologies to control erosion. However, because surface runoff plays an overriding and critical role in most erosion, control measures should generally be selected to increase infiltration and thereby reduce runoff, or to slow the surface runoff rate by reducing the degree of slope or slope length. In tropical steepplands with high rainfall rates, high surface runoff rates create high stream power leading to unacceptable erosion rates higher than 100 t/ha/yr. Some soils with initially high infiltration rates have no runoff until the profile is saturated by tropical storms, then upwelling and runoff occur leading to high soil losses.^[14] For erosion control, the choice of technologies from lists such as those in Table 2 depends on the biophysical, social, and economical characteristics of the site. Use of schemes such as the FESLM discussed earlier helps ensure that the technology(ies) chosen lead to sustainable land management.

Lal^[15] presented the rationale to consider the watershed as the basic unit for sustainable land and water management. Because erosion processes of detachment, transport, and deposition operate in the landscape, effective erosion control requires an understanding of those processes, and the hydrological processes and their impact in the watershed scale. The trend toward integrated watershed management as the basis for natural resource management is consistent and adds weight to the shift toward community-based land management discussed in the previous section on soil conservation.

Control of wind erosion presents a difficult challenge, particularly in the developing world. The underlying causes are overgrazing and removal of crop residues in cultivated fields in dry areas. The first line of defense is to reduce stocking rates or retain residues in the field. However, these techniques, and long-term solutions such as the planting of windbreaks, are not within the realm of economic possibilities for poor farmers in regions such as West Africa. Nevertheless, Stirk^[16] showed that farmer-managed regeneration of woody species was a promising and effective approach to reduce wind erosion without reducing income or requiring too much investment of labor or capital. Once again, this illustrates the principle that all the dimensions of sustainable land management must be considered to achieve successful erosion control.

CONCLUSION

The prospects for better control of soil erosion have improved in years. Our knowledge of the human and biophysical factors affecting soil erosion has been improved through the use of new methods such as GIS-based multiaгент modeling; in particular, by locating and targeting erosion hotspots in the landscape, the GIS methods can greatly improve the effectiveness of soil conservation programs. Furthermore, in some developed countries, the elaboration of the concept of multifunctionality of agricultural land will foster approaches and practices that take into account important ecosystem services such as biodiversity protection, landscape integrity, carbon sequestration, and watershed protection. For the developing countries, there have been moves to devise schemes to subsidize the ecosystem services provided by land users in upper watershed. Thus it may be prevention of the off-site impacts of soil erosion that motivates an expansion in the use of soil conservation measures. Whatever the motivation, an expanded effort is needed urgently, particularly in the tropics.

REFERENCES

1. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *World Map of the Status of Human-Induced Soil Degradation: An Explanatory Note*; International Soil Reference and Information Centre: Wageningen, 1991; 1–34.

2. Pongsapich, A. *Indigenous Technical Knowledge for Land Management in Asia*; Issues in Sustainable Land Management No. 3; International Board for Soil Research and Management: Bangkok, 1998.
3. <http://www.wocat.net/materials/qtqalist.pdf> (accessed June 2005).
4. Trimble, S.W.; Crosson, P. U.S. soil erosion rates—Myth and reality. *Science* **2000**, *289* (5477), 248–250.
5. www.landcareaustralia.com.au/ (accessed June 2005).
6. www.nacdnet.org/ (accessed June 2005).
7. World Commission on Environment and Development. *Our Common Future*; Oxford University Press: Oxford, 1987; 1–400.
8. Humi, H. *Precious Earth: From Soil and Water Conservation to Sustainable Land Management*; International Soil Conservation Organization and Centre for Development and Environment: Berne, 1996; 1–89.
9. Smyth, A.J.; Dumanski, J. *FESLM: An International Framework for Evaluating Sustainable Land Management*; Soil Bulletin No. 32; FAO: Rome, 1993; 1–74.
10. Prior, J.C. From technology transfer to community development: The policy implications of the Australian landcare movement. In *Soil Conservation Extension: From Concepts to Adoption*; Sombatpanit, S., Zebisch, M.A., Sanders, D.W., Cook, M.G., Eds.; Oxford and IBH Publishing: New Delhi, 1997; 77–86.
11. Anderson, J.R.; Thampapillai, J. *Soil Conservation in Developing Countries: Project and Policy Intervention*; Policy and Research Series No. 8; World Bank: Washington, 1990; 1–45.
12. El-Swaify, S.A.; Dangler, E.W.; Armstrong, C.L. *Soil Erosion by Water in the Tropics*; Research Extension Series No. 24; College of Tropical Agriculture and Human Resources, University of Hawaii: Honolulu, 1982; 1–173.
13. <http://topsoil.nserl.purdue.edu/nserlweb/> (accessed June 2005).
14. Coughlan, K.J.; Rose, C.W. *A New Soil Conservation Methodology and Application to Cropping Systems in Tropical Steeplands*; Technical Report No. 40; Australian Centre for International Agricultural Research: Canberra, 1997; 1–147.
15. Lal, R. Rationale for watershed as a basis for sustainable management of soil and water resources. In *Integrated Watershed Management in the Global Ecosystem*; Lal, R., Ed.; CRC Press: Boca Raton, 2000; 3–16.
16. Stirk, G. *Wind Erosion in the Sahelian Zone of Niger: Processes, Models, and Control Techniques*; Tropical Resource Management Papers, No. 15; Wageningen Agricultural University: Wageningen, 1997; 1–151.

Erosion: Water Quality

Andreas Klik

Department of Water, Atmosphere and Environment, University of Natural Resources and Applied Life Sciences, Vienna, Austria

Abstract

Soil and water quality problems caused by agricultural production practices are receiving increased attention and are partly perceived by society as important environmental problems. Agricultural land use has a great impact on soil erosion and sediment deposition as well as on water quality. Sediment input is the most polluting source of aquatic ecosystems worldwide. Severe soil degradation from soil erosion by wind and water can destroy the productive capacity of the soil. Sediments as well as nutrients, pesticides, salts, other trace elements, pathogens, and toxic substances (which are either attached to sediments or dissolved in surface runoff) lead to contamination of surface and groundwater bodies and diminish water quality.

INTRODUCTION

The adjustments and modernization of agriculture has led to considerable changes of the agricultural landscape. To increase economic efficiency, changes were made mainly in land use, crop production, crop rotation, fertilizer management, and pest control. These human impacts affected the stability of agricultural ecosystems. Soil erosion by wind and water increased in many parts of the world, resulting in negative changes of soil and water resources.

EFFECTS OF SEDIMENTS

Within a watershed, various soil erosion processes including detachment, transport, and deposition occur. Their magnitude mainly depends on slope, soil, and topographic and land use conditions and can greatly vary within an area. Although high erosion rates can be reached in a watershed, not all sediments may reach surface water bodies because of deposition nearby.

According to estimates, on a global scale sediments carried into the oceans increased from 9 billion tons per year before introduction of intensive agriculture, grazing, and other activities to between 23 and 45 billion tons thereafter.^[1] Of the total 0.9 billion tons of sediments carried by rivers from the continental United States, about 60% is estimated to come from agricultural lands.^[2] In the Chinese Loess Plateau, total soil loss is estimated to be about 2200 million tons annually. Three-quarters of the total soil loss is transported to the lower reaches of the Yellow river.^[3] Therefore, average sediment concentration of Yellow river is 38 kg/m, 20 times higher than that of the Nile River in Egypt and 38 times higher than that of the Mississippi

River. During periods of floods, silt content in the Yellow River can rise to more than 650 kg/m.

Different erosion processes produce different sediment qualities. Sheet or inter-rill erosion usually produces fine-textured sediment from topsoil layers. These layers contain the bulk of agriculturally applied chemicals that attach to and move with the eroded soil. Channel erosion produces sediment from all soil layers incised by this erosion process. Stream banks erode into previously deposited alluvial sediments that normally do not contain significant amounts of agrochemicals.

High concentration of suspended sediments in streams diminishes their recreational uses because pathogens and toxic substances commonly associated with suspended sediments are threats to public health. High sediment concentrations reduce water clarity and the esthetic appeal of streams. Suspended sediment is also harmful to stream biota; it inhibits respiration, diminishes the transmission of light needed for plant photosynthesis, and promotes infections. Sediment depositions can lead to suffocation of benthic organisms.

EFFECTS OF NUTRIENTS

For growth of plants and animals, macronutrients, such as nitrogen (N), phosphorus (P), potassium, sodium, sulfur (S), magnesium (Mg), and calcium (Ca), and micronutrients, such as boron, molybdenum, chlorine, iron (Fe), manganese, and others, are essential. When input in surface water bodies, N and P especially can increase their biological productivity. Although N and carbon (C) are essential to the growth of aquatic biota, the focus of attention was on P inputs, because of the difficulty in controlling the exchange of N and C between the atmosphere and water,

Table 1 Dissolved load from different continents to the world's oceans.

Continent	Dissolved load (106 Mg/yr)
Africa	201
Asia	1592
Europe	425
North and Central Europe	758
Oceania/Pacific Islands	293
South America	603
Total	3872

Source: From Walling.^[4]

and fixation of atmospheric N by some blue-green algae. Thus, P is often the limiting element and its control is of prime importance in reducing the accelerated eutrophication of fresh waters. Walling^[4] calculated that globally the transport of a dissolved P (DP) load from different continents to the oceans is about 4 billion tons per year (Table 1).^[5]

In undisturbed rivers, total P concentrations are generally lower than 25 µg P/L. Concentrations higher than 50 µg P/L result from human activities. Investigations show that considerably more than half of all European river stations exceed that level. In Europe, the highest levels are found in a band stretching from the northwest across the middle part of the continent, reflecting the intensity of agriculture and livestock production in these regions.^[6] From 1980 to 1989, average P concentrations in U.S. freshwater ecosystems were 0.1 mg/L or greater. Concentrations greater than 0.5 mg/L were especially common in the central and south-central regions, where extensive agricultural use of P and highly erodible soils combines to create large non-point source loadings.^[7] Nationwide measurements since 1991 showed that annual amounts of total P and total N measured in agricultural streams were equivalent to less than 20% of P and less than 50% of the N that was applied to the land.^[8] In more than one-half of sampled streams and in three-fourths of agricultural and urban streams, average annual concentrations of total P exceeded the U.S. Environmental Protection Agency desired goal for prevention of nuisance plant growth. The highest total N and total P concentrations were found in small streams draining watersheds with large proportions of agricultural or urban land.

Nitrogen

Most of the N in the soil is stored in soil organic matter. This N is transformed through mineralization into ammonium ions (NH₄) and released into the soil. Ammonium adsorbs to clay minerals and organic matter and can be transported to surface water attached to sediment or suspended matter. Nitrification transforms ammonium ions to

nitrite (NO₂) and nitrate (NO₃). Nitrate is totally soluble and moves freely in solution.

The dominant factors in the loss of N in runoff are the amount and timing of rainfall and soil properties. Much of the N that enters the rivers is associated with eroded sediments and eroding soil organic matter or it is dissolved in surface runoff. Runoff during rainfall or snowmelt may have a high concentration of organic N attached to sediments but is typically low in NO₃ concentration. Most rains have a significant wetting period before runoff starts. Soluble fertilizer is carried down into the soil during these first few minutes, so little is lost in the runoff. Exceptional losses can occur if intense rain strikes very suddenly or if rain cannot infiltrate because of frozen or impermeable soil, or if water exfiltrates at the foot of the hillslope because of near-surface hydraulic gradient.^[9]

High NO₃ levels in water are a critical concern for human beings because ingestion of such water has the potential to reduce the oxygen-carrying capacity of the blood. When NO₃ is reduced to NO₂⁻, it oxidizes the Fe of the hemoglobin in the blood and methemoglobin is formed, which cannot carry oxygen.^[10] Infants are more endangered to suffer these ill effects than are adults.

Phosphorus

Soil P exists in inorganic and organic forms. In most agricultural soils, 50–75% of P is inorganic, although this fraction can vary from 10% to 90%. Inorganic P forms are dominated by hydrous sesquioxides and amorphous and crystalline Al and Fe compounds in acidic and non-calcareous soils and by Ca compounds in alkaline and calcareous soils. Organic forms include relatively labile phospholipids, nucleic acids, inositols, and fulvic acids, while more resistant forms comprise humic acids. The liability of these forms of P is based on the extent to which extractants of increasing acidity or alkalinity, which are applied sequentially, can dissolve soil P.^[11]

During erosion processes, P can be transported by surface runoff in DP and particulate P (PP) forms. PP includes P attached to eroded soil particles and absorbed by organic matter. PP constitutes the major fraction of P transported from cultivated land with values between 60% and 90%.^[12,13] Runoff from grass or forest land carries little sediment and is therefore generally dominated by DP. While DP is immediately available for biological uptake, PP can provide a long-term source of P for aquatic biota.^[14] Not all P transported by sediments is bioavailable.^[15] Sediment transported P that is not presently bioavailable may become bioavailable,^[16] with the release of P depending on the nature of the sediment and surrounding environment. Under regular pH and aerobic conditions in aquatic systems, P is present as secondary or tertiary phosphate and combines with Fe as a heavily soluble substance that is deposited in the sediment. Under anaerobic conditions (i.e., oxygen shortage), Fe has a higher affinity to S and

combines with it under production of Fe sulfide; P is then released and again available as nutrient for aquatic biota.^[17]

As P is usually the limiting factor in surface water bodies, each increase of P concentration leads to increased growth of undesirable algae and aquatic weeds and oxygen shortages, and subsequently to problems with fisheries and water used for recreation, industry, and drinking. Massive surface blooms of cyanobacteria (blue-green algae) lead to

fish kills, make drinking water unpalatable, and contribute to the formation of trihalomethane during water chlorination.^[18] The consumption of algal blooms or of the water-soluble neurotoxins and hepatotoxins released when the algae die can kill livestock and may pose a serious health hazard to humans.^[19]

Most of the annual nutrient losses measured in surface runoff are associated with sediment losses. Data in [Table 2](#)

Table 2 N and P losses associated with surface runoff (R) and soil loss (S) as well as total losses (T).

Study description	Land use/crop	Fertilization (kg/ha)	Runoff (mm/yr)	Soil loss (Mg/ha/yr)	N losses associated with (kg/ha)			P losses associated with (kg/ha)			References
					R	S	T	R	S	T	
Minnesota field plot study	Fallow	Recommended fertilizer application	12.72	37.00	3.43	146.86	150.29	0.19	33.15	33.34	[20]
	Continuous corn		8.61	16.47	2.42	75.56	77.98	0.41	18.19	18.60	
	Corn rotation		4.74	7.54	1.18	34.77	35.95	0.23	8.43	8.66	
	Hay rotation		13.21	3.96	4.01	0.09	4.10	0.66	0.02	0.68	
England field plot study	Contour	No information	7.4	2.43	0.14	5.85	5.99	0.09	3.32	3.41	[21]
Potatoes–winter wheat–winter barley	Up/down slope		9.0	3.51	0.21	5.09	5.30	0.11	4.30	4.41	
Nigeria field plot study	Bare fallow		504.1	232.6	11.5	310	321.5	3.7	20	23.7	[22]
	Maize–maize (mulch)		29.3	0.2	0.6	T	0.6	1.0	t	1.0	
	Maize–maize (plow till)		88.4	7.2	1.6	14	15.6	0.6	1	1.6	
Alabama watershed study	Cotton conventional tillage	67 N, 36 P	182	2.98	4.95	n.a.		0.75	0.24	0.96	[23]
Cotton conservation tillage	78 N, 25 P		273	1.31	6.39	n.a.		2.62	0.30	2.92	
Mississippi watershed study	Soybeans conventional tillage	45 N at beginning of experiment	532	n.a.	3.47	11.60	15.04	0.48	12.21	12.69	[24]
Soybeans no till			392	n.a.	3.33	1.54	4.87	2.06	0.92	2.98	
North Carolina field plot study	Conventional till	120 N and 60 P	313.5	6.84	8.49	22.76	31.25	2.51	0.17	2.68	[25]
	Strip till		348.5	3.59	12.81	12.27	25.08	4.63	0.06	4.69	
	No till		284.5	2.04	11.47	8.77	20.24	4.2	0.03	4.23	
Southern China field	Peanut	309 N and 197 P at beginning of experiment	160	9.0	17.2	84.0	101.2	3.1	63.6	66.7	[26]
Plot study	Corn–soybean–buckwheat–rape		160.5	6.0	24.6	75.4	100.0	3.4	63.5	66.0	
	Conventional corn–soybean–buckwheat–rape min. till + mulch		72.2	1.16	9.6	13.2	22.8	1.4	10.1	11.5	

Note: t, traces; n.a., not available.

show that losses of soluble N and P in runoff water are usually much less than the losses of these nutrients transported attached to sediments. Soil erosion protection measures like contouring or no-till management are able to diminish nutrient losses by reducing soil loss.

EFFECTS OF PESTICIDES

Pesticides are used to protect food and fiber from damage by weeds, insects, diseases, nematodes, and rodents. When used properly, modern pesticides can perform their

functions without causing significant hazards to humans or to the environment. The most serious long-term effects of pesticides on humans are cancer, genetic defects, and birth defects. Effects on non-target organisms may lead to a decreasing biodiversity.

The application of pesticides in the European Community fell between 1985 and 1995, but this does not necessarily indicate a decrease in environmental impact because the range of pesticides in use has changed.^[6] In United States, the potential for pesticide runoff loss is greatest for watersheds in the Midwest and the Mississippi Embayment.^[27]

Table 3 Pesticide losses associated with runoff.

Study description	Compound	Application rate (kg/ha/yr)	Location	Crop/treatment	Total seasonal losses by runoff in % of application	References
Comparison between field and watershed scale 1997–1998			France	Vineyards		[28]
	Diuron	2.0		No-till field	3.28	
				Tilled field	0.92	
				Watershed	0.52	
	Simazine	1.0		No-till field	2.96	
				Tilled field	0.54	
				Watershed	0.24	
Field study 1993–1996	Endosulfan	2.25–3.0	Australia	Irrigated cotton	1.5–1.9	[29]
Field study natural rainfall	Atrazine	1.45–4.03	Georgia	Corn	0.2–1.9	[30]
	Paraquat	1.52–15.3		Corn	3.4–10.9	
	Trifluralin	1.12		Soybean	0.1–0.3	
	Cyanazine	1.35–1.61		Corn	0.07–1.0	
	Diphenamid	2.31–3.52		Soybean	0.1–7.2	
Field study, natural rainfall	Alachlor	2.24	Iowa	Corn	0.96 (average)	[31]
	Atrazine	2.24		Corn	2.1 (average)	
	Cyanazine	2.24		Corn	2.1 (average)	
	Fonofos	1.12		Corn	0.36 (average)	
Field study, natural rainfall 1994–1999	Rimsulfuron	0.014	Austria	Corn–winter wheat–sugar beet–summer barley–sunflower rotation		[32]
	Bromoxynil	0.225		Conventional tillage	5.63 (average)	
	Metamitron	0.49		Conservation tillage	1.74 (average)	
	Pendimethalin	1.40		No till	2.58 (average)	
Field study, rainfall simulation	Alachlor	3.36	Illinois	Up/down slope	6.2	[33]
				Contour	2.4	
	Tebufos	1.12		Up/down slope	0.5	
				Contour	0.4	
Field study, rainfall simulation	Atrazine	0.56	Oklahoma	Continuous winter wheat		[34]
				No till	7.9 (average)	
				Chisel tillage	10.5 (average)	
				Disk tillage	7.5 (average)	

Table 4 Pesticides in stream and water bodies resulting from agricultural applications.

Watershed stream system	Location	Pesticide residues	Concentration	Comments	References
Rivers and agricultural drainage	Japan	CNP	0.01–16.67	Highest levels found above 4 weeks after rice planting and when floodwaters released from paddies	[35]
Parena River, 600 km upstream from month	Argentina	Lindane	0.009 (average)	Sediment transported pesticides were positively correlated with discharge as was sediment concentrations	[42,36]
		Parathion	0.022 (average)		
		ABHC	0.009 (average)		
Rhein River	Germany	Six main herbicides (bentazon, chloridazon, chlortoluron, isoproturon, MCPP, and terbutylazin)	10.9 ton/yr total load	2.7 g/ha/yr from non-point source = 0.15% of applied amount; main part from point sources	[37]

The persistence of a pesticide in the environment and three physical properties of pesticides—soil sorption propensity, solubility, and vapor pressure—determine the tendency of pesticides to move from the application site.^[38,39]

Persistence is a pesticide’s resistance to decomposition through chemical, photochemical, and microbial action. It is expressed in terms of half-life (in days). It varies from a few hours to months depending on the pesticide and the local environment. Pesticides that persist are more likely to move off-site than less persistent ones. Most pesticides decompose into less toxic chemical forms, but sometimes the metabolites retain pestical properties.

Sorption is the binding of the chemical to the soil. Some pesticides are preferentially transported and adsorbed to entrained sediments. Wauchope^[40] found that those pesticides with solubility below 2 mg/L are transported primarily by sediment. As soil organic matter is mainly responsible for adsorption of non-ionic organic compounds, sorption is commonly evaluated by using a sorption-partitioning coefficient (Koc, expressed in ml/g) based on the organic carbon content of the soil.

Water solubility (expressed in mg/L) determines how easily it goes into solution with water. Simply being water soluble does not mean that a pesticide will leach into groundwater or runoff into surface water. However, solubility means that if a soluble pesticide somehow gets into water, it will probably stay there and go where the water flows. Although the relation between soil sorptivity and water solubility is complex, it is generally true that given a particular soil adsorption level, the greater the solubility, the more potential there is for losses from the field when it rains.

Vapor pressure (expressed in mmHg) is the measure of a pesticide’s tendency to evaporate. Wind speed, air temperature, soil temperature, humidity, and equipment determine the amount of losses to the atmosphere.

Soil, pesticide, and rainfall characteristics influence the timing and amount of pesticide loss^[41,42] and the dominant transporting agent of that pesticide. Pesticide persistence and sorption properties influence the time they stay near

their application site and whether they will be adsorbed to sediment or remain in solution phase. Pesticide persistence and sorption determine, in part, the probability of loss by runoff, and in what form the loss will occur (in solution/runoff water or adsorbed to sediment). Percentage of pesticide in runoff and on sediment will depend not only on sorption properties but also on the time elapsed since application^[43] and on the processes controlling runoff and sediment production during rainfall.^[44,45] Those pesticides that are strongly sorbed to soil clays and organic matter may be subject to removal by surface runoff or wind. Other pesticides that are weakly sorbed and have high water solubility may be lost in the dissolved state.

In Table 3, the results from field experiments with natural and simulated rainfall about losses of different pesticides associated with runoff are summarized. In many studies, the bulk of measured pesticide losses occurred in a single storm soon after pesticide application. Therefore, pesticide runoff losses from simulated rainfall may be higher than that from natural rainfall studies. Examples of reports of pesticides in surface waters are shown in Table 4.

CONCLUSION

Further, it will be difficult to prevent contamination of water bodies by sediments, nutrients, pesticides, salts, and other pollutants if soil degradation is not controlled. Worldwide, there already exist various effective soil erosion control practices and measures. They are able to reduce soil erosion by wind and/or water and therefore to prevent negative on-site and off-site impacts. Additionally, predictive tools are available to identify problem areas and to perform and evaluate environmental sound watershed management to achieve sustainable land and soil use. We have to keep in mind that soil and water quality are inherently linked; conserving or enhancing soil quality is a fundamental step to improve the quality of our surface and groundwater bodies. Improvement of the quality of both resources (soil and water) will be essential for future generations.

REFERENCES

- Judson, S. What's happening to our continents? In *Use and Misuse of Earth's Surface*; Skinner, B.J., Ed.; William Kaufmann: Los Altos, 1981; 12–139.
- National Research Council. *Productive Agriculture and a Quality Environment*; National Academic Press: Washington, 1974.
- Wen, D. Soil erosion and conservation in China. In *World Soil Erosion and Conservation*; Pimentel, D., Ed.; Cambridge University Press: Cambridge, 1993; 63–85.
- Walling, D.E. Rainfall, runoff, and erosion of the land. A global review. In *Energetics of Physical Environment*; Gregory, K.J., Ed.; Wiley: Chichester, 1977; 89–117.
- Lal, R. Global overview of soil erosion. In *Soil and Water Science: Key to Understanding Our Global Environment*; Baker, R.S., Gee, G.W., Rosenzweig, C., Eds.; SSSA Special Publication Number 41; Soil Science Society of America: Madison, 1994; 39–51.
- European Environment Agency (EEA). *Report of the EEA: Europe's Environment—The Second Assessment*. Available at <http://www.mem.dk/aarhus-conference/issues/dobris+3/second.htm> (accessed October 2000).
- Smith, R.A.; Alexander, R.B.; Lanfear, K.J. Stream water quality in the conterminous United States: Status and trends of selected indicators during the 1980s. In *National Water Summary 1990–91: Hydrologic Events and Stream Water Quality*; Water Supply Pap. 2400; Paulson, R.W., Chase, E.B., Williams, J.S., Moody, D.W., Compilers, Eds.; U.S. Geological Survey: Washington, 1993; 111–140.
- U.S. Geological Survey. *The Quality of Our Nation's Waters—Nutrients and Pesticides, Circular 1225*; U.S. Geological Survey: Washington, 1999; 82 pp.
- Zheng, F.; Huang, C.; Norton, L.D. How near-surface moisture gradients affect phosphorus and nitrate losses. In *Soil Erosion Research for the 21st Century*, Proceedings of the International Symposium, Honolulu, Hawaii, Jan 3–5, 2001; American Society of Agricultural Engineers: St. Joseph, 2001; 649–652.
- Keeney, D.R. Nitrogen management for maximum efficiency and minimum pollution. In *Nitrogen in Agricultural Soils*; Stevenson, F.J., Ed.; American Society of Agronomy: Madison, 1982; Vol. 22, 605–649.
- Sharpley, A.N.; Halverson, A.D. The management of soil phosphorus availability and its impact on surface water quality. In *Soil Processes and Water Quality, Advances in Soil Science*; Lal, R., Stewart, B.A., Eds.; Lewis: Boca Raton, 1994; 7–90.
- Sharpley, A.N.; Smith, S.J.; Jones, O.R.; Berg, W.A.; Coleman, G.A. The transport of bioavailable phosphorus in agricultural runoff. *J. Environ. Qual.* **1992**, *21*, 30–35.
- Sharpley, A.N.; Rekolainen, S. Phosphorus in agriculture and its environmental implications. In *Phosphorus Losses from Soil to Water*; Tunney, H., Carton, O.T., Brookes, P.C., Johnston, A.E., Eds.; CAB International: Wallingford, 1997; 1–54.
- Sharpley, A.N.; Menzel, R.G. The impact of soil and fertilizer phosphorus on the environment. *Adv. Agron.* **1987**, *41*, 297–324.
- Sonzogni, W.C.; Chapra, S.C.; Armstrong, D.E.; Logan, T.J. Bioavailability of phosphorus inputs to lakes. *J. Environ. Qual.* **1982**, *11*, 555–563.
- Wildung, R.E.; Schmidt, R.L.; Gahler, A.R. The phosphorus status of eutrophic lake sediments as related to changes in limnological conditions—Total, inorganic and organic phosphorus. *J. Environ. Qual.* **1974**, *3*, 133–138.
- Hartmann, L. *Oekologie und Technik Analyse, Bewertung und Nutzung von Ökosystemen*; Springer: Berlin, 1992; 269 pp.
- Kotak, B.G.; Kenefick, S.L.; Fritz, D.L.; Rousseau, C.C.; Prepas, E.E.; Hrudey, S.E. Blue-green algal toxins in drinking water supplies. *Research in Alberta. Lake Line* **1994**, *14*, 37–40.
- Martin, A.; Cooke, G.D. Health risks in eutrophic water supplies. *Lake Line* **1994**, *14*, 24–26.
- Burwell, R.E.; Timmons, D.R.; Holt, R.F. Nutrient transport in surface runoff as influenced by soil cover and seasonal periods. *Soil Sci. Soc. Am. Proc.* **1975**, *39*, 523–528.
- Catt, J.A.; Quinton, J.N.; Rickson, R.J.; Styles, P. Nutrient losses and crop yields in the wooburn erosion reference experiment. In *Conserving Soil Resources. European Perspectives*; Rickson, R.J., Ed.; CAB International: Wallingford, 1994; 94–104.
- Lal, R. Soil erosion and conservation in West Africa. In *World Soil Erosion and Conservation*; Pimentel, D., Ed.; Cambridge University Press: Cambridge, 1993; 7–25.
- Soileau, J.M.; Touchton, J.T.; Hajek, B.F.; Yoo, K.H. Sediment, nitrogen, and phosphorus runoff with conventional- and conservation-tillage cotton in a small watershed. *J. Soil Water Conserv.* **1994**, *49* (1), 82–89.
- Schreiber, J.D.; Cullen, R.F. Tillage effects on surface and groundwater quality in loessial upland soybean watersheds. *Trans. ASAE* **1998**, *41* (3), 607–614.
- Reddy, G.B.; Raczowski, C.W.; Reyes, M.R.; Gayle, G.A. Surface losses of N, P, and herbicides from a long-term tillage study at North Carolina A & T State University. In *Soil Erosion Research for the 21st Century*, Proceedings of the International Symposium, Honolulu, Hawaii, Jan 3–5, 2001; American Society of Agricultural Engineers: St. Joseph, 2001; 665–668.
- Zhang, T.; Wang, X.; Zhang, B. Soil erosion and nutrient cycling on sloping upland ecosystems of red soil in southern China. In *Soil Erosion and Dryland Farming*; Laflen, J.M., Tian, J., Huang, C., Eds.; CRC Press: Boca Raton, 2000; 203–216.
- United States Department of Agriculture-National Resources Conservation Services. *Water Quality and Agriculture. Status, Conditions, and Trends*; Working Paper No. 16; USDA-NRCS: Washington, 1997; Vol. 20013, 125 pp.
- Louchart, X.; Voltz, M.; Andrieux, P.; Moussa, R. Herbicide transport to surface waters at field and watershed scales in a Mediterranean vineyard area. *J. Environ. Qual.* **2001**, *30*, 982–991.
- Kennedy, I.R.; Sánchez-Bayo, F.; Kimber, S.W.; Hugo, L.; Ahmad, N. Off-site movement of endosulfan from irrigated cotton in New South Wales. *J. Environ. Qual.* **2001**, *30*, 683–696.
- Leonard, R.A.; Langdale, G.W.; Fleming, W.G. Herbicide runoff from upland piedmont watersheds—Data and

- implications for modeling pesticide transport. *J. Environ. Qual.* **1979**, *8*, 223–229.
31. Baker, J.L.; Johnson, H.P. The effect of tillage systems on pesticides in runoff from small watersheds. *Trans. ASAE* **1979**, *22*, 554–559.
 32. Klik, A.; Zartl, A.S.; Rosner, J. Tillage effects on soil erosion, nutrient, and pesticide transport. In *Soil Erosion Research for the 21st Century*, Proceeding of the International Symposium, Honolulu, Hawaii, Jan 3–5, 2001; American Society of Agricultural Engineers: St. Joseph, 2001; 71–74.
 33. Kenimer, A.L.; Mitchell, J.K.; Felsot, A.S.; Hirschi, M.C. Pesticide formulation and application technique effects on surface pesticide losses. *Trans. ASAE* **1997**, *40* (6), 1617–1622.
 34. Basta, N.T.; Huhnke, R.L.; Stiegler, J.H. Atrazine runoff from conservation tillage systems: A simulated rainfall study. *J. Soil Water Conserv.* **1997**, *52* (1), 44–48.
 35. Suzuki, M.; Yamato, Y.; Akiyama, T. Fate of herbicide 2,4,6-trichlorophenyl-4-nitrophenyl ether in rivers and agricultural drainages. *Water Resour.* **1978**, *12*, 777–782.
 36. Lenardon, A.M.; Hevia, M.I.M.; Fuse, J.A.; DeNochetto, C.B.; Depetris, P.J. Organochlorine and organophosphorus pesticides in Paran River, Argentina. *Sci. Total Environ.* **1984**, *34*, 289–298.
 37. Bach, M.; Huber, A.; Frede, H.-G.; Mohaupt, V.; Zullei-Seibert, N. *Schaetzung der Eintraege von Pflanzenschutzmitteln aus der Landwirtschaft in die Oberflaechengewasser Deutschlands*. Berichte/Umweltbundesamt; Erich Schmidt: Berlin, 2000; 273 pp.
 38. Hornsby, A.G.; Wauchope, R.D.; Herner, A.E. *Pesticide Properties in the Environment*; Springer: New York, 1996; 227 pp.
 39. Linders, J.B.H.J.; Jansma, J.W.; Mensink, B.J.W.G.; Otermann, K. *Pesticides: Benefaction or Pandora's Box?* Report No. 679101014; National Institute of Public Health and Environmental Protection: Bilthoven, 1994; 204 pp.
 40. Wauchope, R.D. The pesticide content of surface water draining from agricultural fields: A review. *J. Environ. Qual.* **1978**, *7*, 459–472.
 41. Leonard, R.A. Herbicides in surface waters. In *Environmental Chemistry of Herbicides*; Grover, R., Ed.; CRC Press: Boca Raton, 1988; Vol. 1, 45–87.
 42. Leonard, R.A. Movement of pesticides into surface waters. In *Pesticides in the Soil Environment: Processes, Impact and Modeling*; Cheng, H.H., Ed.; Soil Sci. Soc. Am. Book Ser. 2; Soil Science Society of America: Madison, 1990; 303–350.
 43. Auerswald, K. Seasonal variation of erosive rains in southern Germany. *Zeitschrift Fuer Kulturtechnik und Landentwicklung* **1996**, *37* (2), 81–84.
 44. Zhang, X.C.; Norton, L.D.; Hickman, M. Rain pattern and soil moisture content effects on atrazine and mato-lachlor losses in runoff. *J. Environ. Qual.* **1997**, *26*, 1539–1547.
 45. Truman, C.C.; Steinberger, P.; Leonard, R.A.; Klik, A. Laboratory determination of water and pesticide partitioning. *Soil Sci.* **1998**, *163* (7), 556–569.

Erosivity and Erodibility

Peter I.A. Kinnell

*School of Resource, Environmental and Heritage Sciences, University of Canberra,
Holt, Australian Capital Territory, Australia*

Abstract

In water erosion, erosivity and erodibility are the terms used to identify rain-associated factors driving erosion and the susceptibility of the soil to erosion. Their application in modeling erosion in empirical and process-based models will be described and discussed.

THE UNIVERSAL SOIL LOSS EQUATION (USLE)

Historically, the terms “erosivity” and “erodibility” were originally associated with the R and K factors in the following USLE:

$$A = RKLSCP \quad (1)$$

where A is the annual average soil loss per unit area, R is the rainfall-runoff (erosivity) factor, K is the soil (erodibility) factor, L is the slope length factor, S is the slope gradient factor, C is the crop and crop management factor, and P is the support practice factor.^[1,2] Fundamental to the USLE is the concept of the unit plot—a bare fallow plot having a slope length of 72.6 ft (22.1 m), a slope gradient of 9%, and cultivation up and down the slope. For the unit plots, the following expressions are used:

$$C = L = S = P = 1.0$$

and

$$A = RK \quad (2)$$

In the USLE, R is defined as the average annual value of the product of the total storm kinetic energy and the maximum 30-minute intensity for rainfall events producing at least 0.5 in. (12.5 mm) of rain or at least 0.25 in. (6.35 mm) of rain in 15 minutes. In the Revised USLE (RUSLE),^[3] all rainfall events were considered in calculating R in the western part of the United States. Customary U.S. units for R are $10^2 \text{ ft T}_f \text{ in./A/hr/yr}$ (hundreds of foot-Tonf-inch per Acre-hour-year) and K are $\text{TA hr} \times 10^{-2} / \text{ft T}_f \text{ in.}$ (Ton-Acre-hour per hundreds of foot-Tonf-inch). Customary metric units are MJ mm/ha/hr/yr (megajoule-millimeter per hectare-hour-year) and t hr/MJ/mm (metric ton-hour per megajoule-millimeter).^[3]

K is essentially the regression coefficient for the direct relationship between R and A under unit plot conditions. Experimental plot data were used originally to determine the values of K. Subsequently, equations have been

developed to predict K from soil properties in various geographic locations.^[2–4]

The USLE model works reasonably well in some locations (Fig. 1A) but does not in others (Fig. 1B). The USLE is less effective when the soil has a high capacity to accept rainfall.^[5] In the latter situation, there is a tendency for the USLE to overestimate low annual soil losses. Including runoff in the storm erosivity factor by multiplying the EI_{30} index by the runoff coefficient (Q_R , runoff amount divided by rainfall amount) provides a model known as the USLE-M^[5] which, as can be seen by comparing Fig. 2 with Fig. 1, is better able to deal with the effect of differences in soil infiltration characteristics. It should be noted that, as indicated by the different regression coefficients shown in Figs. 1B and 2, any change in the event erosivity index from the product of event energy and the maximum 30-minute intensity means that the USLE K factor values cannot be used.

MORE PROCESS-BASED APPROACHES

The USLE is designed to predict erosion from sheet- and rill-eroded areas. Modern understanding of erosion processes recognizes that the energy required for soil detachment associated with rill erosion comes from flow energy, while detachment in sheet and inter-rill erosion is dominated by energy derived from raindrop impact. The USLE does not take these different energy sources into account. So more process-based models have been developed to overcome this limitation. Rill and inter-rill erosion equations have been developed within the U.S. Department of Agriculture's Water Erosion Prediction Project (WEPP).^[6] In the rill equation,

$$D_r = K_r(\tau - \tau_0)(1 - q_s T_{cf}^{-1}) \quad (3)$$

D_r is rill detachment, K_r is the rill erodibility, τ is the flow shear stress, τ_0 is the critical shear stress, q_s is the sediment load, and T_{cf} is the sediment load when the transport

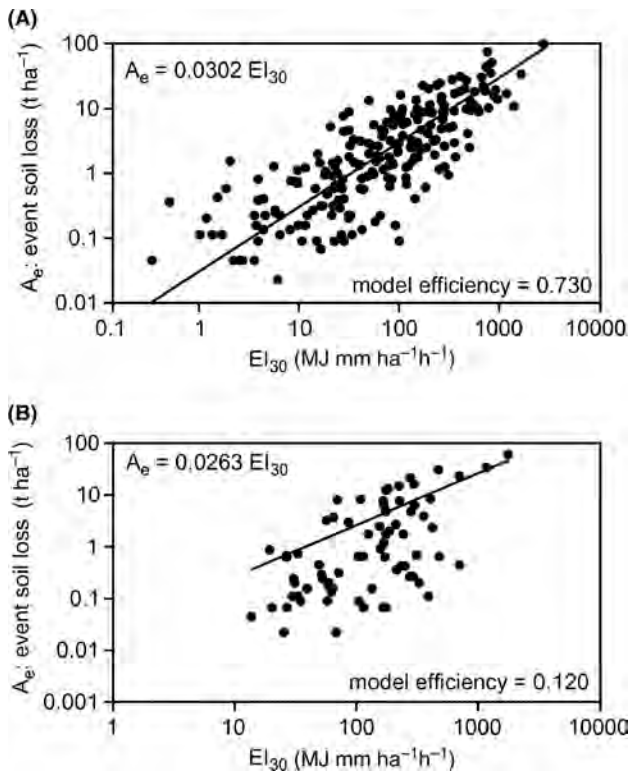


Fig. 1 Relationships between event soil losses from bare fallow and the EI_{30} index on (A) plot 5 at Holly Springs, MS, during 1961–1968 and (B) plot 13 at Morris, MN, during 1961–1971. The logarithmic form of the Nash–Sutcliffe efficiency factor provides the measure of model efficiency.

Source: From Kinnell & Risse.^[5]

capacity is reached. τ_0 is a soil-dependent parameter. However, it is not considered to be an erodibility factor. In the basic inter-rill equation,

$$D_i = K_i I_e I_x S_f \quad (4)$$

D_i is inter-rill detachment, K_i is the inter-rill erodibility, I_e is the effective rainfall rate during the period of excess rainfall, I_x is the excess rainfall rate, and S_f is a function of

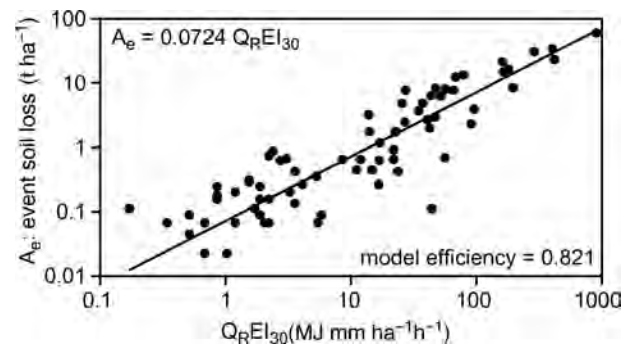


Fig. 2 Relationships between event soil losses from bare fallow and $Q_R EI_{30}$ index on plot 13 at Morris, MN, during 1961–1971.

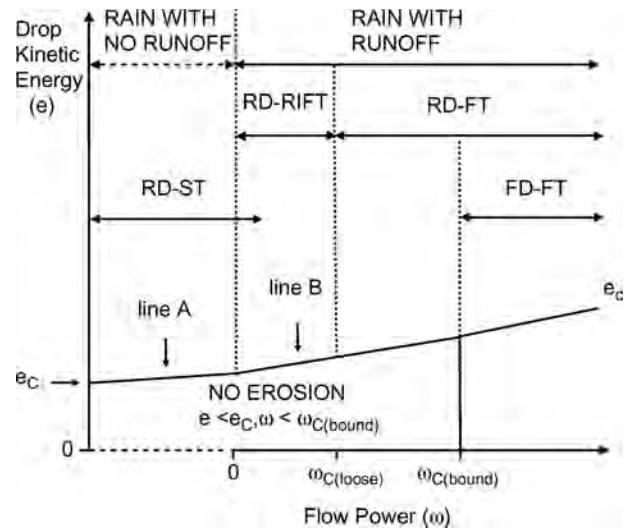


Fig. 3 Detachment and transport processes associated with variations in raindrop and flow energies. e_c , critical raindrop energy to cause erosion. (Line A) e_c prior to flow (increasing through crust development). (Line B) e_c when flow occurs (increasing drop energy used to penetrate flow). $\tau_{c(\text{loose})}$: critical shear stress for transporting loose material; $\tau_{c(\text{bound})}$: critical shear stress for detaching soil from surface of soil matrix; RD–ST: raindrop detachment–splash transport; RD–RIFT: raindrop detachment–raindrop induced flow transport; RD–FT: raindrop detachment–flow transport; and FD–FT, flow detachment–flow transport.

Source: Adapted from Kinnell.^[7]

inter-rill slope. Base values for K_r and K_i have been determined from experiments undertaken on a number of soils in the United States.^[6] In practice, the base values for K_r and K_i are adjusted to account for various effects including roots, sealing, and crusting to predict soil loss. The surface conditions of freshly tilled soils impacted by raindrops change over time, and this causes the susceptibility of the soil to erosion to change with time. The base values for K_i were determined from data usually collected after rain at 62 mm/hr had been applied to short (250 mm long slopes) inclined slopes for 30 minute or more to ensure reasonably stable surface conditions.

Although inter-rill erosion is dominated by detachment caused by raindrop impact, there are a number of detachment and transport processes that can occur during a rainfall event. At the onset of rain on a dry surface, material detached by raindrop impact is splashed down slope. This is shown as raindrop detachment–splash transport (RD–ST) in Fig. 3. The RD–ST system controls what is commonly known as splash erosion. Before detachment can occur, a raindrop must possess sufficient energy to detach soil material from the cohesive forces that hold that material within the surface layer. This critical energy is indicated by e_c . e_c tends to increase during rain as, e.g., a surface crust develops, and this contributes to a decrease in soil erodibility associated with splash erosion. As runoff develops, raindrops penetrating the flow lift soil material into the flow,

and this material moves downstream in the flow. This detachment–transport system is called the raindrop detachment–raindrop induced flow transport (RD–RIFT) system, and it requires both raindrop impact and flow to operate. On short slopes, if the slope gradient is sufficiently high, flow can entrain loose material on the surface without the aid of raindrop impact. This produces a raindrop detachment–flow transport (RD–FT) system. The flow detachment–flow transport (FD–FT) system is normally associated with rills but can also occur on short high-gradient slopes. In principle, each detachment–transport system can be represented by an equation that has an erosivity term and associated erodibility term. As all the detachment–transport systems may operate to various degrees in any given area depending on the prevailing conditions, inter-rill erodibilities are less predictable than rill erodibilities.

In the USLE, soil erodibility (K) was kept constant during the calculation of the annual soil loss, while interaction between short-term variations in erosivity and crop factors was considered in determining C values. In RUSLE, K can be considered to vary during the year. At the process model level, changes in the soil surface conditions associated with, e.g., the development of soil crusts have been noted already as having an effect on erodibility. However, erosion is limited by either detachment or transport processes. Crusts influence detachment, but transport-limiting conditions need to be considered in modeling erosion. In the WEPP rill model,^[6] this is achieved by the term $1 - q_s T_{CF}^{-1}$. The RD–ST and RD–RIFT are inherently transport-limited systems because material detached by drop impact tends to fall back to the soil surface and is remobilized by numerous drop impacts before being discharged from the eroding area. The material on the surface, near the end of the eroding area, contributes directly to the material being discharged and also protects the underlying surface to some degree. This influences erodibility.^[8] For the RD–RIFT systems, erodibility is given by the following equation:

$$K_i = H_{PD} K_{iPD} + (1 - H_{PD}) K_{iM} \quad (5)$$

where H_{PD} is the degree of protection provided by the previously detached material, K_{iPD} is the erodibility when the previously detached material fully protects the underlying surface, and K_{iM} is the erodibility of the soil surface when there is no previously detached material upon it. However, because H_{PD} and the composition of the previously detached material vary with flow conditions and particle characteristics, H_{PD} and K_{iPD} vary in time and space. Relatively complex mathematical models that consider particle characteristics^[9] have been developed with the aim of dealing with this variation.

SIMPLE CONCEPT FOR COMPLEX SYSTEMS

Originally, the concept of erosivity and erodibility as considered in the USLE was quite simple because the model

was designed to predict long-term average annual soil loss based on experimental data. Equations developed to predict K and C enabled the model to be applied to areas and situations not covered in the original experiments. However, the USLE and the RUSLE do not capture the effects of some of the fundamental erosion processes well in some cases. To overcome this deficiency, more process-based models have been developed. However, because the susceptibility of the soil to erosion can vary in space and time as a result of an interaction between the soil surface and the detachment and transport mechanisms, erodibility can vary in time and space in a complex manner. Complex process-based mathematical models may be needed to account for this variation, but less complex models may produce sufficiently accurate results to make reasonable decisions on land management options. Runoff is an important factor in rainfall erosion. Models that include it as part of the erosivity factor have the potential to account for erosion better than those that do not.

REFERENCES

1. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall–Erosion Losses from Cropland East of the Rocky Mountains: Guide for Selection of Practices for Soil and Water Conservation*; Agriculture Handbook 282; U.S. Department of Agriculture: Washington, 1965.
2. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses: A Guide to Conservation Planning*; Agriculture Handbook 537; U.S. Department of Agriculture: Washington, 1978.
3. Renard, K.G.; Foster, G.R.; Weesies, D.K.; McCool, D.K.; Yoder, D.C. *Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE)*; Agriculture Handbook 703; U.S. Department of Agriculture: Washington, 1997.
4. Loch, R.J.; Slater, B.K.; Devoil, C. Soil erodibility values (K_m) for some Australian soils. *Aust. J. Soil Res.* **1998**, *36*, 1045–1055.
5. Kinnell, P.I.A.; Risse, L.M. USLE-M: Empirical modelling rainfall erosion through runoff and sediment concentration. *Soil Sci. Soc. Am. J.* **1998**, *62*, 1667–1672.
6. Flanagan, D.C.; Nearing, M.A. *USDA—Water Erosion Prediction Project: Technical Documentation*; NSERL Report No. 10; National Soil Erosion Research Laboratory: West Lafayette, 1995.
7. Kinnell, P.I.A. *A Discourse on Rainfall Erosion Processes and Modelling on Hillslopes*; Occasional Paper No. 6; Centre for Australian Regolith Studies, University of Canberra: Australia, 2000.
8. Kinnell, P.I.A. The effect of slope length on sediment concentrations associated with side-slope erosion. *Soil Sci. Soc. Am. J.* **2000**, *64*, 1004–1008.
9. Hairsine, P.B.; Sander, G.C.; Rose, C.W.; Parlange, J.-Y.; Hogarth, W.L.; Lisle, I.; Rouhipour, H. Unsteady soil erosion due to raindrop impact: A model of sediment sorting on a hillslope. *J. Hydrol.* **1999**, *220*, 115–128.

Encyclopedia of Soil Science

Third Edition

Volume II

Ethics through Pollution
Pages 848–1781

Ethics –
Food Security

Forest –
Gypsic

Hard-Setting –
Infiltration

Inorganic –
Land Use

Landfill –
Metal

Microbial –
Nitrogen-15

Nitrous Oxide –
Particle

Peatlands –
Pollution

Ethics

John Ronald Engel

Center for Humans and Nature, Beverly Shores, Indiana, U.S.A.

Abstract

This entry presents the following six arguments as to how soil ethics can contribute to the making of a new global covenant committed to universal principles of sustainability, justice, and peace: 1) Land care is exemplary of the necessity of addressing the environmental and social issues of our planet together; 2) soil integrity is the sine qua non of ecological and biospheric integrity; 3) soil “stewardship” is an essential component of the covenants we make between the generations; 4) the need for precaution in the ways we treat soil, given the limits of our scientific and practical understanding of how to preserve and restore it, demonstrates the importance of the precautionary principle; 5) soil conservation requires a “common but differentiated” approach to our cultural as well as economic and technological capacities for land care; and 6) the evolutionary and spiritual solidarity we experience with the earth through soil is a strong naturalistic foundation for our ultimate commitment to sustaining the life of the planet.

I am the Living Earth, I am the softened tissue of rocks, baked by the sun, split by ice, carried by water, and winnowed by the wind. I am interwoven by myriads of tiny plants and animals that pulse and breathe. I am the invisible universe of sparkling molecules in the infinity of living soils that bless the mantle of the globe.

—*Len Webb*

That land is a community is the basic concept of ecology, but that land is to be loved and respected is an extension of ethics.

—*Aldo Leopold*

MAKING THE GLOBAL COVENANT

Calls for a New Global Covenant

Over the course of the last several decades, calls for a new earth covenant have come from all quarters of the globe.^[1] Sometimes the call is for a “new global compact,” a term used by United Nations Director Generals Javier Perez de Cuellar and Kofi Annan. But more prophetically and accurately in the words of Liberation Theologian Leonardo Boff and Political Philosopher David Held, it is a call for “a new covenant with Earth.” These calls presume that we are citizens of the world as well as of nations, that we live in an interdependent planetary community, and that if we are to overcome the tragedy of the global commons, we must reach agreement on the fundamental moral principles of global governance.

Covenants are open, unconditional commitments to be faithful to others regarding our most fundamental

values and behaviors, and they have historically served as the spiritual and moral authority for national constitutions and international treaties. “Covenant” and “compact” are often used interchangeably to describe public agreements of unlimited duration that require mutual consent to be abrogated, but covenant typically places primary emphasis on the moral dimension, while compact on the legal. Both need to be distinguished from contracts, which are limited agreements for the sake of a utility of direct benefit to the consenting parties and which are dissolved once that utility is obtained. Calls for a new global covenant or compact recognize that to address issues as daunting as climate change we need not only new and better contracts but a transformation in our personal and collective loyalties.

The Earth Charter and the Proposed Soil Charter

Officially launched in The Hague in 2000, the Earth Charter, which may be accessed at www.earthcharter.org, is the most complete public statement we have of the moral and spiritual vision of a new earth covenant. As a people’s treaty circulated throughout the world, with endorsements by hundreds of nongovernmental organizations, municipalities, national governments, and international agencies, it is the most legitimate as well. The Earth Charter may be viewed as a landmark in the series of repeated attempts since the drafting of the United Nations Charter and the Universal Declaration of Human Rights, including the long line of soft law declarations on environment and development running from the 1972 Stockholm Declaration on the Human Environment to the

2000 Millennium Development Goals, to rewrite the global compact along more comprehensive covenantal lines. Like its precursor, *Caring for the Earth*, the Earth Charter affirms “respect and care for the community of life”—a time-honored covenantal principle—as the overarching principle of moral obligation.^[2]

One of the most significant contributions the Earth Charter can make at this juncture in world history is to serve as a framework and catalyst for continuing the dialogue on global ethics and moving the international covenant-making process forward.

Soil health is one of the most critical, if neglected, components in the well-being of the biosphere and the human civilizations that depend on it. Our readiness to assume responsibility for it must be considered an essential part of the movement to create a new global covenant committed to universal principles of sustainability, justice, and peace.

Reading the Earth Charter with soil and soil responsibilities in mind, we can see that it is not a complete or fully adequate document. “Soil” is mentioned only three times and those in passing. Nonetheless, the Charter is an invaluable resource for understanding the basic moral approach we need to take soil and the moral guidelines we need to follow for promoting better land care throughout the world, while an explicit elaboration of soil ethics can substantially improve the Charter’s conceptual foundations and practical effectiveness. On this basis, we may greet the prospects of a dialogue between the Earth Charter and a revised World Soil Charter and World Soils Policy, or a new Global Charter for Land Care and a “new, binding, international instrument concerning the protection and sustainable use of soils,” as suggested in the Program of Action that emerged from the 2007 Selfoss Forum on Soils, Society, and Global Change, as a positive step toward advancing both the conservation of soil and the earth covenant.^[3]

THE CONTRIBUTIONS OF SOIL ETHICS TO THE NEW GLOBAL COVENANT

Eco-Justice

The most significant development that may be traced through the long line of international declarations that paved the way to the Earth Charter is the recognition, beginning at Stockholm, that the most urgent environmental issues of the world are inseparable from the most urgent issues of social justice. Growing awareness of these relationships fueled the discussion of “sustainable development” in the 1980s and “just and sustainable communities” in the 1990s and has subsequently expanded to include security and peace. The Earth Charter expresses this evolution in global ethics in its explicit adoption of what has come to be called an “eco-justice”

ethic, an integrative moral approach in which ecological and social (including economic, political, and cultural) well-being are considered as both dependent and independent variables. The 71 principles of the Earth Charter spell out our obligations for “Ecological Integrity,” “Social and Economic Justice,” and “Democracy, Non-violence, and Peace,” not only by attending to the specific obligations we have for each of these basic moral concerns but also by spelling out the relationships between them. Principle 7, e.g., expresses the imperative of integrating a wide range of interdependent ecological and social values: “Adopt patterns of production, consumption, and reproduction that safeguard earth’s regenerative capacities, human rights, and community well-being.”

In past years, out of a laudable zeal to bring attention to the devastating degradation of soils across the world, advocates of soil conservation often isolated the values of land care from other great questions of global ethics such as poverty, human rights, and military conflict. There were tendencies to interpret soil as primarily a resource commodity and to restrict the moral concern for soil to its role in agriculture, thus neglecting the many different and critical roles soils play in the evolution and ecology of the planet.

The 2007 Selfoss Forum on Soils, Society, and Global Change took major strides toward breaking this single-minded vision. The presentations forthrightly addressed the intimate connections of soil health to other matters of global concern such as climate change, poverty, biodiversity loss, and unrestrained economic and population growth. It made clear why we cannot hope for significant change in soil conservation values and practices without addressing these other issues and putting into practice principles of social and economic justice, human rights, democratic participation, nonviolence, and international law, nor can we hope to make significant headway on these global social issues without paying more attention to the functions soil performs in sustaining the ecosphere and human civilization. The Forum thus not only lifted up the crucial importance of soil for the future of humankind and the planet but also showed how soil demonstrates the validity of the strong eco-justice approach of the emerging earth covenant.

Ecological Integrity

A distinguishing feature of the Earth Charter is the prominent place it gives to ecological integrity as an ethical as well as a scientific principle. The second of the four main sections of the Charter is entitled “Ecological Integrity.” Principle 5 states “Protect and restore the integrity of earth’s ecological systems, with special concern for biological diversity and the natural processes that sustain life.”

Brendan Mackey, professor at Australian National University and the member of the Earth Charter drafting

committee most responsible for its emphasis on ecological integrity, writes that “ecological integrity can be understood in terms of the capacity of earth’s ecosystems to continue flourishing so that the environmental services are maintained upon which the well-being of humans and all life depend.”^[4] A major contribution that soil ethics can make to the earth covenant is to emphasize the fact that soil integrity is both a critical component of ecological integrity and its *sine qua non*—without which there will not be ecological integrity.

An influential figure in the field of soil conservation and environmental ethics who can help make that argument is Aldo Leopold (1887–1948), an American forester and wildlife, ecologist whose book, *A Sand County Almanac*, recounts his experiences restoring a tract of land on the Wisconsin River and climaxes in a proposal for a “land ethic” that makes ecological integrity a first principle of environmental ethics.

The primary motivation of Leopold’s career was what he called “the oldest task in human history to live on a piece of land without spoiling it.”^[5] He came to that understanding as a result of his firsthand experiences in soil erosion and degradation, first in the rangelands in the American southwest and then in the agricultural areas of the midwest. In 1921, he wrote:

With enough time and money, a neglected farm can be put back on its feet—if the soil is still there. With enough patience and scientific knowledge, an overgrazed range can be restored—if the soil is still there. By expensive replanting and a generation or two of waiting, a ruined forest can again be made productive—if the soil is still there. With infinitely expensive works, a ruined watershed may again fill our ditches or turn our mills—if the soil is still there. But if the soil is gone, the loss is absolute and irrevocable. Three years later he wrote: “Soil is the fundamental resource, and its loss the most serious of all losses.”^[6]

Leopold used “land” to describe soil, water, plants, animals, and people collectively, and he often spoke of land as a “community.” He preferred land over terms such as ecosystem because it communicated to the ordinary citizen our primary dependence on fertile soil, and his choice of community was influenced by the new field of ecology often defined as the “science of communities.” But Leopold was also well versed in the writings of the Hebrew prophets. His ideas of “land community” and land ethic may be interpreted as contemporary science-based translations of their understanding of an aboriginal covenant between God, people, and the earth. Leopold’s proposal for a land ethic may be set forth in the form of a syllogism.^[7]

1. “All ethics rest upon a single premise: that the individual is a member of a community of interdependent

parts. [It is] a limitation on freedom of action in the struggle for existence—[a way of evolving] modes of cooperation.”

2. We are members of the land community.
3. Therefore, we need to exercise the same constraints on our relations to the other members of the land community—soils, waters, plants, and animals—as we do in our relations to people.
4. Thus, the land ethic, “A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise.”

From which we may extrapolate, “A thing is right when it tends to preserve the integrity, stability, and beauty of the soil community. It is wrong when it tends otherwise.”

By “stability” Leopold meant the capacity of the land to cycle nutrients efficiently and continuously because its biotic pyramid was intact and its food circuits were open; by “integrity” he meant that the land possessed all the parts needed to maintain its stability and their successful coordination through competition/cooperation; by “beauty” he meant the “pleasing appearance to the eye, ear, and soul” of land that possesses stability and integrity.^[8] Since the mid-20th century, new scientific notions of nature as dynamic, ever changing, even chaotic, and species and individual interactions as selfish, rather than symbiotic, have challenged earlier ecological notions of nature as composed of “integrated communities” in dynamic equilibrium, with a diverse array of species in positive interactions with one another. In light of these developments, environmental ethicists such as Baird Callicott have sought to dynamize the concept of land ethic and incorporate the crucial norms of scale in evaluating anthropogenic changes in nature.^[9] Regardless, it is unlikely that ecological integrity as such defined as “the capacity of Earth’s ecosystems to continue flourishing so that the environmental services are maintained upon which the well-being of humans and all life depend” and its crucial component, soil integrity, will ever be dislodged as the primary focus of our covenantal responsibilities to the earth.^[4]

Stewardship

Global ethics must also address the covenantal dimensions of the earth community’s passage through time, and the responsibilities we bear, as individuals and societies, for passing on a healthy biosphere, including the responsibilities we bear to past, present, and future generations of humans for the sustainable use and just distribution of the resources of nature, all of which may be summed up in the ethical imperative to serve the common good.

To assume an inclusive, long-term responsibility for the good of nature and the resources humans make of nature are often identified with the idea of “stewardship.”

It is surprising that the term stewardship does not appear in the Earth Charter. Two reasons are offered for this omission. One is that no word or its equivalent could be found in some important world cultures—Russia is an example. A second is that for indigenous people the word implies a separation from nature or a paternalistic relationship, evidence, unfortunately, for the ways in which the term has taken on meanings alien to its origin. Nonetheless the principle of stewardship is included in the Charter's Preamble: "The protection of Earth's vitality, diversity, and beauty is a sacred trust" and in Principle 4: "Secure Earth's bounty and beauty for present and future generations."

Stewardship was a concept with great motivational power at the beginning of the conservation movement in the early 20th century, and it has reemerged in the ethics of "sustainable development." But its origin lies in the common property arrangements of hunter-gatherer and premodern agrarian and nomadic societies. In these societies, individual family units agreed to share land as a common property arrangement that gave each family the right of use but not ownership.^[10] This was the social context in which the biblical understanding of stewardship arose—evident, e.g., in the story of Joseph who was entrusted with the position of guardian (namely, custodian, manager, overseer, caretaker, and trustee) of the land of Egypt and who by careful oversight and foresight so conserved the produce of the land that famine was avoided and the common good secured. Patrick Dobel writes, "The stewardship imperative assumes that the moral and ecological constraints are respected, and it adds the obligation to distribute the benefits justly. The steward must 'give them their portion of the food at the proper time.'"^[11]

The "stewardship ethic" therefore has three elements: 1) long-term responsible care for the common good of nature; 2) sustainable ecological and economic use of these goods; and 3) just sharing of these goods—and these responsibilities—among the population at large.

It is difficult to imagine a more powerful example of what stewardship means in this original moral sense than stewardship of soil. It is the common property of the human and natural community, we must use it sustainably, and its care and benefits should be shared by all members of the human community.

Prevailing land use practices substitute short-term gains for wise stewardship. The addition of synthetic fertilizers may, for a time, produce yields equal to or even better than soils whose structure, moisture retention, and nutrient levels have been enriched by crop rotation and manure. Monopoly of land ownership and control can increase efficiency and help maximize soil productivity in the short run. But long-term stewardship requires soils whose inherent fertility has been retained by sound land use practices, equitable land tenure, and fair resource distribution.

In 1938, Walter Loudermilk, a close colleague of Aldo Leopold, published the first extensive documentation on the consequences of soil exhaustion in the fall of civilizations. Soon afterward, he composed what Moses might have given as the Eleventh Commandment, had he foreseen such consequences (*italics added*):

Thou shalt inherit the Holy Earth as a *faithful steward*, conserving its resources and productivity from generation to generation. Thou shalt safeguard thy fields from soil erosion, thy living waters from drying up, thy forests from desolation, and protect thy hills from overgrazing by thy herds, that thy descendants may have abundance forever. If any shall fail in this stewardship of the land thy fruitful fields shall become sterile stony ground and wasting gullies, and thy descendants shall decrease and live in poverty or perish from off the face of the earth.^[12]

The Precautionary Principle

Scientific uncertainty has frequently been used as a reason to avoid taking action to protect the environment. The precautionary principle recognizes that lack of certainty regarding the threat of environmental harm is to be expected and that this should not be an excuse for not taking action, rather it is only a reason for more positive moral approaches to the environment such as preserving ecological integrity and stewardship. The Earth Charter states the precautionary principle in those terms:

6. Prevent harm as the best method of environmental protection and, when knowledge is limited, apply a precautionary approach.

- a. Take action to avoid the possibility of serious or irreversible environmental harm even when scientific knowledge is incomplete or inconclusive.
- b. Place the burden of proof on those who argue that a proposed activity will not cause significant harm and make the responsible parties liable for environmental harm.
- c. Ensure that decision making addresses the cumulative, long-term, indirect, long-distance, and global consequences of human activities.
- d. Prevent pollution of any part of the environment and allow no buildup of radioactive, toxic, or other hazardous substances.
- e. Avoid military activities damaging to the environment.

At its core, the precautionary principle is an acknowledgment of the limits of our knowledge of the natural world and an admonition to be wary of all forms of technological and economic boosterism that promise

quick fixes to environmental problems. Accepting our finitude and ignorance requires a moral stance of humility and prudence, one that must be explicitly woven into the covenant we make with one another and the earth. The 2007 revised International Union for Conservation of Nature “Guidelines for Applying the Precautionary Principle to Biodiversity Conservation and Natural Resource Management” make this point clear: “Implementing the Precautionary principle entails ... humility and restraint, acknowledging human fallibility in the quest for certainty, the limits of science, and the tendency to over-reach in the quest for human security and well-being.”^[13]

Since our knowledge of soil and our understanding of how effectively to conserve and restore it under contemporary ecological and social conditions are limited, the precautionary principle must be an important part of soil ethics. Soil science and conservation both confirm and benefit from the Earth Charter’s strong endorsement of the precautionary principle, and the covenantal virtues of humility, restraint, and prudence that inform it.

Common but Differentiated Responsibilities

One indication that the nations of the world are groping toward a covenantal—and therefore federalist—understanding of global governance is the fact that the principle of “common but differentiated responsibilities” has become an increasingly prominent component of international law.^[14] Although the Earth Charter does not explicitly reference the principle of “common but differentiated responsibilities,” it does affirm a federal model of “partnership” for global governance in its declaration: “We are at once citizens of different nations and of one world in which the local and global are linked” and its admonition that with “increased freedom, knowledge, and power comes increased responsibility to promote the common good.”

Informed by formal principles of global equity and justice, international agreements such as the 1992 United Nations Framework Convention on Climate Change have focused on the implications of the principle for economic, political, and technological differences, and these are undoubtedly important considerations for understanding the respective obligations of different nations to address global soil issues, but the principle can also illumine another aspect of our understanding of the interdependence of soil, society, and global change. It can help us to grasp the importance of the diverse ways in which cultural attitudes and belief systems impact our relationships to the land. How we understand and discharge our responsibilities for soil are rooted in the distinct biogeocultural narratives of our several societies. As indicated in the Program for Action of this forum “[e]xperiences of soil stewardship and restoration efforts in communities around the world are diverse and location-specific.”^[15] It follows that our

differing biogeocultural interpretations of the meaning and significance of soil need to be brought into dialogue with one another, and their respective strengths and limitations identified, if we are to build a rich covenantal culture of international land care.

Aldo Leopold vigorously pursued a scientific and ethical critique of what he considered to be the biblically inspired narrative of “Abrahamic land conquest.” It was in counterpoint to this narrative that he generated his alternative narrative of the land ethic and sought *A Sand County Almanac* to communicate the new vision to his fellow citizens.

“We are the landscape,” Iceland’s former president, Vigdis Finnbogadóttir, declared during the Selfoss Forum. The power of the Iceland biogeocultural narrative was apparent in the wood carvings that graced the walls of the Soil Conservation Service headquarters at Gunnarsholt, telling the story of the original promise of the Icelandic Commonwealth 930–1262, the erosion of its land, its prosperity, and its democracy in the subsequent centuries, and the rebirth of the Republic and the efforts to restore the land in the 20th century. It was also apparent in the narrative of the Soil Conservation Service itself, which for a century has worked to realize Hannes Hafstein’s vision, that a time will come when the “land’s wounds are healed.” The Soil Conservation Service of Iceland modeled the principle of “common but differentiated responsibilities” when it decided to convene an International Forum in 2007, where it could contribute the inspiration—and the lessons—of its own history to others working on behalf of soil conservation in many different cultural and environmental contexts and to the common task of finding a way forward for the world as a whole.

Earth Spirituality

Some sympathetic yet critical voices, such as environmental philosopher Strachan Donnelley, have raised the question of whether the existing conceptions of the earth covenant, as reflected in the text of the Earth Charter, have “taken the Earth seriously enough or bound humanity sufficiently to the Earth’s well-being.”^[16]

What Donnelley and others are saying is that to take the earth’s evolution seriously requires accepting the realities of deep time, the dynamics of evolutionary change, basic continuity rather than radical disjunction between “living” and “nonliving,” a geocentric rather than an anthropocentric worldview, the need for protecting a majority of the earth’s surface from human interference if evolution is to continue unimpaired, and a profound personal solidarity and intimacy with the natural world. They discern reluctance in existing discussions of global ethics to forthrightly acknowledge that we are not the reference point of earth’s evolution but mortal creatures who like all creatures are born and die of earthly processes, are absolutely dependent on them for our

sustenance, and are completely bound with them in a common destiny.

Soil is our most direct link to the evolutionary origins of life in the waters of the planet, and the greatest repository we have of the early history of earth's evolution. We—and all other forms of terrestrial life—are absolutely dependent on it. It may well be that a reluctance to appreciate the importance of soil in our lives is due to a reluctance to accept the full spiritual implications of our participation in this reality. If so, then a contribution that soil ethics can make to the earth covenant is to bring this reality to our consciousness and encourage the emergence of an evolutionary spirituality that can celebrate, rather than deplore, our earthly identity.

Many of the great religious traditions of the world have acknowledged this reality. We speak in theology of ultimate reality as the “ground of being.” *Human*, *humility*, and *humus* come from the same ancient Indo-European root meaning “soil.” *Adamah* means soil in Hebrew, and Adam means the “son of soil, formed from the dust of the Earth”; Eve is “the mother of all living” and therefore Adam and Eve literally mean “soil and life.” The Indian Upanishads bear similar understandings. Moreover, as the widely read Agrarian Essayist Wendell Berry reminds us, “soil is the place where death becomes life again.”^[6]

Soil makes clear the meaning of the covenant of being, that all creatures are fellow voyagers in the great ecological and evolutionary odyssey of “dust unto dust.” Soil reveals some of the most important conditions laid down by the creativity in which we participate and upon which all life depends.

CONCLUSION

A new global covenant requires a transformation of our individual and communal commitments so that they are in keeping with truly universal principles of global justice, peace, and sustainability. The post–World War II history of international accords leading to the Earth Charter and the Millennium Development Goals indicates that such a process has begun. But it will take much additional ethical reflection, opportunity for transcultural dialogue, and determined effort by “conscience constituencies” and political and business leaders across the world if it is to be carried forward. Making a true global covenant ultimately requires the participation of every person and society on the planet and the critical reform of every ecological, economic, social, and spiritual relationship of human life.

Increased awareness of our ethical responsibilities to and for the soils of the planet is an essential component in moving this process forward. Not only is responsibility for soil health by every person and community a biophysical precondition of our future existence on the planet and therefore any opportunity we may have to exercise every other form of covenantal responsibility,

but by understanding the special character of our ethical responsibilities for soil we are able to better understand the meaning of all our obligations to one another and the rest of nature. Soil ethics has become a paradigmatic case of what is universally required of us. It is no coincidence that in the English language we employ the same word, “earth,” as a name both for soil and for the planet.

We have examined six ways in which soil ethics lifts into clearer view both our special ethical responsibilities for soil health and our general covenantal responsibilities for sustaining the community of life. We have argued that 1) successful soil policy requires that ecological and social imperatives be addressed together, as a matter of eco-justice; 2) the need for soil integrity demonstrates why ecological integrity must be the *first principle* of our commitment to sustainability; 3) soil stewardship is grounded in intra and intergenerational covenants; 4) the need for precaution in the ways we treat soil reveals the limits of our scientific and practical understanding of how to conserve all the planet's resources; 5) a “common but differentiated” approach to our cultural as well as economic and technological capacities for land care is a foundational principle for global ethics; and 6) our destiny as *humans* and *humus* reveals the evolutionary and spiritual solidarity we experience with all creatures on earth.

We have also suggested several ways in which the Earth Charter's principles may be drawn upon to better understand the content of soil ethics, and conversely, how attention to soil ethics has the potential to enhance the conceptual foundations and practical effectiveness of the Earth Charter. Not least important, in respect to the latter, is the need to address the omission of soil as a major subject in its own right in the text of the Earth Charter and to explicitly recognize our responsibilities for soil health in the global ethics dialogue.

A new covenant with soil is essential for making a new covenant with earth. It is also a test of whether we will be able to succeed in answering the great calling of our age.

REFERENCES

1. Engel, J.R. A covenant model of global ethics. *World V. Environ. Cult. Relig.* **2004**, 8 (1), 29–46.
2. Engel, J.R. The Earth Charter as a new covenant for democracy. In *Just Ecological Integrity: The Ethics of Maintaining Planetary Life*; Miller, P., Westra, L., Eds.; Rowman and Littlefield: Lanham, 2002; 37–52.
3. Hannam, I.; Boer, B. *Legal and Institutional Frameworks for Sustainable Soils: A Preliminary Report*; IUCN: Gland and Cambridge, 2002; 88 pp.
4. Mackey, B.G. The Earth Charter and ecological integrity—some policy implications. *World V. Environ. Cult. Relig.* **2004**, 8 (1), 72–92.
5. Leopold, A. *The River of the Mother of God and Other Essays*; Flader, S.L., Callicott, J.B., Eds.; The University of Wisconsin Press: Madison, 1991.

6. Johnson, P.W. Traveling in the right direction. In *The Essential Aldo Leopold: Quotations and Commentaries*; Meine, K., Knight, R.L., Eds.; The University of Wisconsin Press: Madison, 1999; 72–86.
7. Callicott, J.B. Into terra incognita. In *The Essential Aldo Leopold: Quotations and Commentaries*; Meine, K., Knight, R.L., Eds.; The University of Wisconsin Press: Madison, 1999; 296–313.
8. Newton, J.L. *Aldo Leopold's Odyssey*; Island Press: Washington, 2006.
9. Callicott, J.B. From the balance of nature to the flux of nature: The land ethic in a time of change. In *Aldo Leopold and the Ecological Conscience*; Knight, R.L., Reidel, S., Eds.; Oxford University Press: New York, 2002; 90–105.
10. Northcott, M.S. Soil, stewardship and spirit in the era of chemical agriculture. In *Environmental Stewardship: Critical Perspectives—Past and Present*; Berry, R.J., Ed.; T & T Clark: London, 2006; 213–219.
11. Dobel, P. The Judeo-Christian stewardship attitude to nature. In *Environmental Ethics: Readings in Theory and Application*; Pojman, L.J., Ed.; Wadsworth Publishing: Belmont, 1998; 26–29.
12. Loudermilk, W.C. *Conquest of the Land through 7,000 Years*, Rev Ed.; U.S. Department of Agriculture Information Bulletin, No. 99: Washington, 1938.
13. IUCN. *Guidelines for Applying the Precautionary Principle to Biodiversity Conservation and Natural Resource Management*. Available at http://cmsdata.iucn.org/downloads/ln250507_ppguidelines.pdf.
14. Stone, C.D. Common but differentiated responsibilities in international law. *Am. J. Int. Law* **2004**, *98*, 276–301.
15. Programme for Action. *Soils, Society and Global Change*. In Proceedings of an International Forum, Selfoss, Iceland, Aug. 31–Sept. 4, 2007; Bigas, H., Gudbrandsson, G.I., Arnalds, A., Eds.; United Nations University: Tokyo, 2008.
16. Donnelley, S. Chartering the earth for life's Odyssey. *World V. Environ. Cult. Relig.* **2004**, *8* (1), 93–100.

Eutrophication

Thomas G. Franti

Department of Biological Systems Engineering, University of Nebraska, Lincoln, Nebraska, U.S.A.

Kyle D. Hoagland

School of Natural Resources, University of Nebraska, Lincoln, Nebraska, U.S.A.

Abstract

Eutrophication is the nutrient enrichment of surface water and the subsequent impacts on water quality and the aquatic ecosystem. Cultural eutrophication is the result of excess nutrients—primarily nitrogen (N) and phosphorus (P)—delivered to rivers, lakes, and estuaries by the activities of humans. The rate of eutrophication is controlled by the rate at which N and P are delivered to a body of water. There are various biological effects of eutrophication. The best way to reduce or reverse eutrophication is to reduce nutrient loading; that is, targeting the source of the problem.

INTRODUCTION

Eutrophication is the nutrient enrichment of surface water and the subsequent impacts on water quality and the aquatic ecosystem. The overabundance of plant nutrients, usually nitrogen and phosphorus—but sometimes silicon, potassium, calcium, iron, or manganese—creates the conditions for excessive plant growth.^[1] Algal blooms are an example of excessive growth caused by the over-supply of nutrients. A body of water is classified by its trophic state based on the amount of nutrients supplied to it. An oligotrophic state is low in nutrients, a mesotrophic state is intermediate, and a eutrophic or hypereutrophic state is high in nutrients.^[2] Eutrophication is a natural process. However, as a natural process, it takes generations, or even thousands of years, for eutrophication to cause significant changes. It is the acceleration of the process, known as cultural eutrophication, which is of the greatest concern to water quality and the health of aquatic systems.

CULTURAL EUTROPHICATION

Cultural eutrophication is the result of excess nutrients—primarily nitrogen and phosphorus—delivered to rivers, lakes, and estuaries by the activities of humans. Nutrient loading can come from both point sources and diffuse sources. Point sources of nutrients are generally urban sewage treatment or industrial water treatment. Diffuse sources (or non-point sources) of nutrients include agriculture, deforestation, and urban lawn runoff. The use of commercial fertilizer in crop production

can lead to losses of nutrients to surface water. Concentrated animal feeding operations are also a potential source of excess nutrients.^[2,3] Finally, deforestation and subsequent soil erosion can deliver excess nutrients, sediment, and organic matter to surface water, creating the conditions for eutrophication.

The natural aging process of lakes in some climates can lead from an oligotrophic state, with clean open water supporting usually cold water fish species, to a eutrophic state of warmer, shallow water supporting different species of plants and fish and, ultimately, to a filled lake closed over by a bog or a fen. This process usually takes thousands of years. In contrast, cultural eutrophication can occur in a matter of one or two generations.

The last 50 years of the 20th century saw the greatest impact of eutrophication on the world's waters. Cultural eutrophication has accelerated the lake aging process, reduced water quality, and impacted aquatic plant and animal populations, at economic costs. These impacts are being felt worldwide.^[1]

Modern society's use of phosphorus additives in detergents has led to excessive phosphorus loading to surface waters.^[1] Modern agriculture's use of commercial fertilizer and fertilizer's relative abundance, low cost, and over use have led to excessive nitrogen loading to surface water. Agriculture also contributes excessive phosphorus from fertilizer and from intensive livestock operations and land application of manure.^[3,4] Urban sources, such as sewage treatment plants, storm water runoff, industrial water treatment facilities, and food processing, contribute excess nutrients. Sewage treatment loading to waters, including treated gray water, is generally considered the greater source of phosphorus.

The greater contributor of nitrogen is agricultural crop production. Other significant agricultural sources are animal confinement operations, forestry, and atmospheric inputs.

RATE OF EUTROPHICATION

The rate of eutrophication is controlled by the rate at which nitrogen and phosphorus are delivered to a body of water. It is generally accepted that phosphorus is the limiting nutrient for algal growth in lakes.^[1,5] Another limiting factor for algal growth is light. As excess phosphorus enters a lake, it can trigger high levels of algal growth. Excessive growth, or blooms, can reduce water clarity. Aquatic macrophytes may also thrive under these conditions. This process can increase the productivity of a lake to an extent. When it becomes excessive, the process of cultural eutrophication has begun.

The nutrients supplied to an aquatic system is the most important factor that determines the species and amount of plant material, which in turn controls available oxygen and the animal species that thrive.^[1] As plants and macrophytes grow and die, organic matter accumulates on the lake bottom. Decomposition of excessive amounts of organic matter can consume available oxygen and create anoxic conditions. An anoxic condition drives away plant and animal species dependent on oxygen. Slowly, a lake's original community changes.

This anoxic state creates another problem if a lake becomes stratified. Temperature differences in the water at the lake's surface and at depth can create thermal stratification. In this condition, the cold, low-oxygen water in the hypolimnion, or lower layer, does not mix with the epilimnion, the warmer surface water. In this condition, the anoxic conditions in the hypolimnion create a condition where the sediment releases phosphorus, which is transported upward to the epilimnion and contributes to increased algal growth. This internal cycling of phosphorus continues the eutrophication process even if no additional phosphorus enters the lake.^[5,6]

EFFECTS OF EUTROPHICATION

There are various biological effects of eutrophication. Besides the problems of algal blooms and anoxic conditions previously described, there are changes in water temperature, reduction in water clarity, increased macrophyte production, population shifts in both plants and animals, and accelerated aging of lakes. Fish kills and the coastal "red tides" and "brown tides" are also potential environmental impacts from eutrophication.^[5,7]

There are many impacts on human use of water from eutrophication. Potable water—water used for human consumption—can be significantly degraded by eutrophication. Besides the excess nutrients themselves, there is

excess algae and other plant growth, which can contribute to unwanted odors and tastes. Thus, additional water treatment is needed to make the water drinkable.^[1,4]

The decline of commercial and recreational fisheries is also an indirect result of eutrophication. Fish populations may move away or over time shift to less desirable fish species as water temperature, clarity, and quality change.^[1]

There are also negative impacts on recreational use of surface waters such as boating and swimming as a result of floating plant growth, smell, and the overall loss of aesthetic quality.^[4] Finally, the general decline of an aquatic ecosystem and the loss of biodiversity are impacts of eutrophication.^[1,3,7]

Some of these impacts can cause significant economic losses as well. Water treatment costs rise as water quality decreases. Algal blooms can contribute to shutting down water treatment plants. Advanced water treatment of municipal water to remove nutrients (such as alum addition)—hence removing a contributing factor to eutrophication—adds another expense to water treatment. Loss of commercial fisheries is an economic impact to a region. Finally, loss of recreation income from tourists, boaters, and recreational fisherman can have significant impacts on a local economy.^[4,7]

REDUCTION AND MANAGEMENT

Two approaches can be taken to the reduction and management of eutrophication: 1) reduction of nutrient loads; and 2) managing the high-nutrient state.^[1] Reduction of loading is clearly the more robust approach. Reduced loads are necessary if long-term improvement is expected. However, because of the ecosystem changes brought on by eutrophication and the potential problem of phosphorus cycling previously described, water quality and ecosystem improvements may not respond quickly to reduced loading.^[3] A combination of reduced loads and in-lake controls may be needed.

Reducing nutrient loads requires reduction of both point and diffuse sources. To reduce point sources, sewage and industrial water treatment plants need advanced water treatment to remove nutrients.^[1,7] Advanced water treatment is expensive, but is the only way to reduce these point source nutrients. Most diffuse sources contributing nutrients to surface water come from agricultural operations. Loading is dependent on the type of crops grown, soil type, climate, cultural practices, fertilizer use, and whether animal waste management practices are sufficient to reduce loading. Nutrient reduction from these sources will require improved management of fertilizer and animal waste and continued reduction of soil erosion throughout the watershed.^[1]

There are several methods used to manage nutrient levels in lakes. These fall into two categories: those that 1) remove nutrients or 2) manage nutrient levels without

nutrient removal. Methods to remove nutrients include the following: 1) lake flushing; 2) hypolimnetic water withdrawal; 3) sediment removal (e.g., dredging); and 4) nutrient inactivation by precipitation (e.g., alum treatment). Methods to manage nutrients without removing them include the following: 1) artificial mixing and/or aeration; 2) dilution by the addition of water low in nutrients; 3) bottom sealing to prevent internal nutrient cycling; 4) manipulation of lake biological communities (e.g., selective fish harvesting or the introduction of fish predators such as largemouth bass, walleye, or brown trout); 5) introduction of biological controls for unwanted macrophyte growth (e.g., weevils or grass carp); and 6) herbicide or other chemical treatments (e.g., copper sulfate) of excessive algal or macrophyte growth.^[1,6]

Clearly, the best way to reduce or reverse eutrophication is to reduce nutrient loading, i.e., targeting the source of the problem.^[1] This long-term solution involves participation and management by people throughout the entire watershed. In-lake nutrient management can be done, but may require annual inputs and regular management.

REFERENCES

1. Harper, D. *Eutrophication of Freshwaters: Principles, Problems and Restoration*; Chapman & Hall: London, 1992.
2. Smith, V.H.; Tilman, G.D.; Nekola, J.C. Eutrophication: Impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. *Environ. Pollut.* **1999**, *100*, 179–196.
3. Carpenter, S.R.; Caraco, N.F.; Correll, D.L.; Howarth, R.W.; Sharpley, A.N.; Smith, V.H. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecol. Appl.* **1998**, *8* (3), 559–568.
4. Daniel, T.C.; Sharpley, A.N.; Lemunyon, J.L. Agricultural phosphorus and eutrophication: A symposium overview. *J. Environ. Qual.* **1998**, *27* (2), 251–257.
5. Correll, D.L. The role of phosphorus in the eutrophication of receiving waters: A review. *J. Environ. Qual.* **1998**, *27* (2), 261–266.
6. Cooke, G.D.; Welch, E.B.; Peterson, S.A.; Newroth, P.R. *Lake and Reservoir Restoration*; Butterworths Publishers: Stoneham, 1986.
7. Klapper, H. *Control of Eutrophication in Inland Waters*; Ellis Horwood Limited: Chichester, 1991.

Evaporation

William P. Kustas

Hydrology and Remote Sensing Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Beltsville, Maryland, U.S.A.

Abstract

Soil evaporation can significantly influence energy flux partitioning of partially vegetated surfaces, ultimately affecting plant transpiration. While important, quantification of soil evaporation, separately from canopy transpiration, is challenging. Techniques for measuring soil evaporation exist and continually improve. The large variability in soil water content requires that there be careful thought to the design of soil evaporation measurements in the field. Numerical models for simulating soil evaporation have been developed and are shown to be fairly robust. However, the required inputs for defining model parameters often limit their application. For many operational applications where detailed soils and ancillary weather data are unavailable or where daily evaporation values are only needed, some of the analytical models described may provide the necessary level of accuracy. Moreover, in the application of weather forecast and hydrologic models, the use of simplified approaches is necessitated by the computational requirements or the lack of adequate data or both for defining more complex numerical model inputs. For large area estimation, the use of remotely sensed soil moisture and surface temperature offer the greatest potential for operational applications.

INTRODUCTION

Soil evaporation not only determines partitioning of available energy between sensible and latent heat flux for bare soil surfaces but can also significantly influence energy flux partitioning of partially vegetated surfaces. This latter effect occurs via the impact of soil evaporation on the resulting surface soil moisture and temperature. These, in turn, strongly influence the microclimate in partially vegetated canopies, indirectly affecting plant transpiration.^[1] Over a growing season, soil evaporation can be a significant fraction of total water loss for agricultural crops.^[2] On a seasonal basis in semiarid and arid regions, soil evaporation can significantly alter the relative fraction of runoff to rainfall, which in turn has a major impact on the available water for plants.^[3] In deserts, in spite of its small magnitude, soil evaporation can introduce significant errors in meteorological forecasting if neglected.^[4]

The measurement of soil evaporation at field scale is typically obtained using standard micrometeorological techniques, namely Bowen ratio and eddy covariance methods. Traditionally, due to fetch and measurement requirements, under partial canopy cover conditions, these techniques are not able to partition the total evapotranspiration into its soil evaporation and plant transpiration components. A novel procedure for partitioning evapotranspiration through utilizing the measured high-frequency time series of carbon dioxide and water vapor concentrations has been developed and tested.^[5] This approach relies upon the simple assumption that

contributions to the time series of carbon dioxide and water vapor concentrations derived from stomatal processes (i.e., photosynthesis and transpiration), and non-stomatal processes (i.e., respiration and direct evaporation) separately conform to flux-variance similarity. Vegetation water use efficiency is the only parameter needed to perform the partitioning. Further work is needed to evaluate the utility of this technique with eddy covariance data collected over a variety of land cover and climate conditions.

Soil evaporation in partial canopy cover conditions varies spatially depending primarily on soil water distribution, canopy shading, and under-canopy wind patterns. These effects are magnified in row crops and under various irrigation techniques (e.g., drip irrigation). Soil evaporation can be measured using microlysimeters,^[6] chambers,^[7] time-domain reflectometers (TDRs),^[8] a combination of microlysimetry and TDR,^[9] micro-Bowen ratio systems,^[3,10] or heat pulse probes.^[11] Given the high spatial variability in the driving forces under partial canopy cover conditions, these point-based measurements are difficult to extrapolate to the field scale. Therefore, models have been developed to estimate the contribution of soil evaporation to the total evapotranspiration process.

Measurement methods are described, and models of varying degrees of complexity are reviewed, focusing primarily on relatively simple analytical models, some of which provide daily estimates and can be implemented operationally. The potential application of models using remote sensing data for large-scale estimation is also briefly discussed.

METHODOLOGIES

Measurement Methods

Microlysimeters

Microlysimeters have been widely used to measure evaporation from the soil surface of irrigated crops.^[8,12,13] Typically, an undisturbed soil sample (a representative vertical section of the soil profile) is inserted into a small cylinder open at the top. The microlysimeter is inserted back into the soil with its upper edge level with the soil surface and weighed either periodically or continuously. Changes in weight reflect an evaporative flux. To eliminate vertical heat conduction through the microlysimeter cylinder and minimize horizontal heat flux in the deeper layers of the sample, poly(vinyl chloride) (PVC) has been found to be the most suitable material. The microlysimeter's dimensions are typically a diameter of ~8 cm and a depth of 7–10 cm.^[14–16] Theoretically, the microlysimeters provide absolute reference for soil evaporation, as long as their soil and the heat balance are similar to the surrounding area.

Chambers

Chambers are used to directly measure the flux of gases between the soil surface and the atmosphere by enclosing a volume and measuring all flux into and out of the volume.^[17] Reviews of chamber designs and calculations of fluxes based on chamber methods can be found in Livingston and Hutchinson^[18] and Hutchinson and Livingston.^[19]

With infrared gas analyzers (IRGAs) becoming increasingly common, they are widely considered to be the method of choice for chamber-based soil respiration and evaporation measurements.^[20] Chambers can be used in either of two modes to calculate fluxes:^[18] 1) in steady-state mode, the flux is calculated from the concentration difference between the air flowing at a known rate through the chamber inlet and outlet after the chamber headspace air has come to equilibrium concentration of carbon dioxide and 2) in the non-steady-state mode, the flux is calculated from the rate of increasing concentration in the chamber headspace of known volume shortly after the chamber is put over the soil.

In both modes, air is circulated between a small chamber that is placed on the soil and an IRGA. Typically, a soil chamber of ca. 1 L volume is placed on a PVC collar of about 80 cm² area. This collar is inserted about 2 cm into the soil and secured to prevent movement when the chamber is placed on it. When the chamber is placed on the collar, circulation of air between the chamber and the external IRGA is induced by a pump, and the water vapor concentration is measured.^[21]

Soil water balance

Soil evaporation (E) can be extracted from the water balance equation, provided that all other components are known

$$E = I + P - R + F - \Delta S \quad (1)$$

where I is irrigation, P is precipitation, R is runoff or runoff, F is deep soil water flux (percolation), and ΔS is change in soil water storage. For an experimental field site, irrigation and precipitation can be easily monitored, and runoff and runoff may be controlled to near-zero amounts by diking. Deep soil water flux errors can be reliably estimated in several ways, with the most important being the monitoring or measuring of the soil water content well below the root zone. The change in soil water storage can be determined fairly accurately with profile measurements of soil water content over multiple depths at the beginning and end of a defined time period.

There are many soil water content sensors, all of which work by measuring a surrogate property that is empirically or theoretically related to the soil water content. A comparative review by Evett et al.^[22] concluded that soil water content is best determined using the neutron probe, gravimetric sampling, and conventional TDR^[23] methods as compared to bore hole capacitance methods. Of the three optimal methods in the study, TDR is the only methodology capable of providing automated continuous measurements. However, other continuous measurement sensors such as frequency-domain reflectometers (FDRs),^[24] and time-domain transmission^[25] sensors are also emerging as options for long-term installations, given appropriate calibration.

Micro-Bowen ratio systems

The Bowen ratio energy balance (BREB) approach is one of the simplest and most practical methods of estimating water vapor flux^[10,26] and has thus been used extensively under a wide range of conditions providing robust estimates.^[27] Use of the BREB concept^[28] enables solving the energy balance equation by measuring simple gradients of air temperature and vapor pressure in the near-surface layer above the evaporating surface. The BREB equation is

$$LE = \frac{R_N - G}{1 + B_o} \quad (2)$$

in which LE is the latent heat flux, R_N is the net radiation, G is the soil heat flux, and B_o the Bowen ratio, which is found from measurements of temperature and vapor pressure at two heights within the constant flux layer.^[29] Assuming equal transfer coefficients for heat and vapor, the Bowen ratio is defined as

$$B_o = \frac{H}{LE} = \gamma \frac{\partial T}{\partial e} \quad (3)$$

where H is the sensible heat flux, γ is the psychrometric constant, T is the air temperature, and e is the water vapor pressure.

Application of the Bowen ratio concept to measure bare soil surface evaporation was suggested^[3,10] by measuring temperature and vapor pressure close to the soil surface (e.g., at 1 and 6 cm). Compared to microlysimeters, the micro-Bowen ratio (MBR) system yielded good results over bare soil.^[10] The potential of the MBR approach was demonstrated by successfully measuring soil evaporation within a maize field.^[30] The testing of the MBR is very limited, and further research is required to examine the performance of the technique under various environmental and agronomic conditions.

Heat pulse probes

A novel approach for measuring soil evaporation has been proposed, based on the soil sensible heat balance.^[11,31] In this approach, a sensible heat balance is used to determine the amount of latent heat involved in the vaporization of soil water following Gardner and Hanks.^[32]

$$LE = (H_0 - H_1) - \Delta S \quad (4)$$

where H_0 and H_1 are soil sensible heat fluxes at depths 0 and 1, respectively; ΔS is the change in soil sensible heat storage between depths 0 and 1; L is the latent heat of vaporization; and E is evaporation.

Typically, three-needle sensors like those described by Ren et al.^[33] are used, which are spaced 6 mm apart, in parallel. Temperature is measured by all three needles, and the central needle also contains a resistance heater for producing a slight pulse of heat required for the heat pulse method. At a given time interval (2–4 hours), a heat pulse is executed, and the corresponding rise in temperature at the outer sensor needles is recorded. Soil volumetric heat capacity and thermal diffusivity are then determined from the heat input and temperature response following the procedures described by Knight and Kluitenberg^[34] and Bristow et al.,^[35] respectively. Having measurements of soil temperature, thermal conductivity, and volumetric heat, evaporation is estimated from Eq. 4.

Numerical Models

Numerous mechanistic/numerical models of heat and mass flows exist and are primarily based on the theory of Philip and de Vries.^[36] However, they continue to be refined through improved parameterization of the moisture and heat transport through the soil profile.^[37–39] Some of these mechanistic models have been used to explore the utility of bulk transfer approaches used in weather forecasting models^[40] and in soil–vegetation–atmosphere models.^[41]

computing field to regional-scale fluxes. These bulk transport approaches are commonly called the “alpha” and “beta” methods defined by the following expressions

$$LE_s = \frac{\rho C_p}{\gamma} \left(\frac{ae_*(T_s) - e_A}{R_A} \right) \quad (5)$$

and

$$LE_s = \frac{\rho C_p}{\gamma} \beta \left(\frac{e_*(T_s) - e_A}{R_A} \right) \quad (6)$$

where r is the air density ($\sim 1 \text{ kg/m}^3$), C_p the heat capacity of air ($\sim 1000 \text{ J/kg/K}$), γ the psychrometric constant ($\sim 65 \text{ Pa}$), $e_*(T_s)$ the saturated vapor pressure (Pa) at soil temperature T_s (K), e_A the vapor pressure (Pa) at some reference level in the atmosphere, and R_A is the resistance (s/m) to vapor transport from the surface usually defined from surface layer similarity theory.^[42] For a, several different formulations exist^[43,44] with one of the first relating a to the thermodynamic relationship for relative humidity in the soil pore space,^[45]

$$h_R = \exp \left(\frac{\psi g}{R_V T_s} \right) \quad (7)$$

where ψ is the soil matric potential (m), g the acceleration of gravity (9.8 m/s^2), and R_V the gas constant for water vapor (461.5 J/kg/K). From Eq. 6, β can be defined as a ratio of aerodynamic and soil resistance to vapor transport from the soil layer to the surface, R_s , namely

$$\beta = \frac{R_A}{R_A + R_s} \quad (8)$$

Both modeling^[44] and observational results^[46,47] indicate that more reliable results are obtained using the beta method. In fact, Ye and Pielke^[44] formulate an expression similar to Camillo and Gurney,^[48] which combines Eqs. 5 and 6 and provides more reliable evaporation rates for a wider range of conditions,

$$LE_s = \frac{\rho C_p}{\gamma} \beta \left(\frac{ae_*(T_s) - e_A}{R_A} \right) \quad (9)$$

Unfortunately, there is no consensus concerning the depth in the soil profile to consider in defining the a and b terms. In particular, studies evaluating the soil resistance term R_s use a range of soil moisture depths: 0–1/2 cm,^[48] 0–1 cm,^[7] 0–2 cm,^[39,46] and 0–5 cm.^[39] The study conducted by Chanzy and Bruckler^[38] appears to be one of the few studies that attempts to determine the most useful soil moisture depth for modeling soil evaporation by considering the penetration depth of passive microwave sensors of varying wavelengths. Using field data and numerical simulations with a mechanistic model, they find that the 0–5 cm depth, which can be provided by the L-band microwave frequency, appears to be the most adequate frequency for evaluating soil evaporation.

Table 1 Bulk soil resistance formulations, R_S , from previous studies.

R_S formula (s/m)	Value of coefficients	Soil type	Depth (cm)	References
$R_S = a \left(\frac{\theta_s}{\theta} \right)^n + b$	$a = 3.5$ $b = 33.5$ $n = 2.3$	Loam ^a	0–1/2	[21]
$R_S = a(\theta_s - \theta) + b$	$a = 4,140$ $b = -805$	Loam ^b	0–1/2	[48]
$R_S = R_{S\text{MIN}} \exp[a(\theta_{\text{MIN}} - \theta)]$	$R_{S\text{MIN}} = 10 \text{ s/m}$ $\theta_{\text{MIN}} = 15\%$ $a = 0.3563$	Fine sandy loam	0–1	[7]
$R_S = \frac{a(\theta_s - \theta)^n}{2.3 \times 10^{-4} (T_s/273.16)^{1.75}}$	$a = 2.16 \times 10^2$ $n = 10$ $\theta_s = 0.49$	Loam	0–2	[46]
	$a = 8.32 \times 10^5$ $n = 16.16$ $\theta_s = 0.392$	Sand	0–2	[46]
$R_S = a\theta + b$	$a = -73,420 - 51,650$ $b = 1,940 - 3,900$	Sand	0–2	[39]
$R_S = \exp\left(b - a\left(\frac{\theta}{\theta_s}\right)\right)$	$a = 4.3$ $b = 8.2$	Silty clay loam ^a	0–5	[41]
	$a = 5.9$ $b = 8.5$	Gravelly sandy loam	0–5	[66]

^aSoil type for the data from Jackson et al.^[49] was determined from texture-dependent soil hydraulic conductivity and matric potential equations of Clapp and Hornberger evaluated by Camillo & Gurney.^[48]

^bSoil type was determined from texture-dependent soil hydraulic conductivity and matric potential equations of Clapp and Hornberger evaluated by Sellers et al.^[41]

Besides the depth of the soil layer to consider, the equations relating soil moisture to R_S have ranged from linear to exponential (Table 1). Furthermore, observations and numerical models have shown that it varies significantly throughout the day and that its magnitude is also affected by climatic conditions.^[38,39]

These are not the only complicating factors that make the use of such a bulk resistance approach somewhat tenuous. From detailed observations of soil moisture changes and water movement, Jackson et al.^[49] found the soil water flux in the 0–9 cm depth to be very dynamic with fluxes at all depths continually changing in magnitude and sometimes direction over the course of a day. These phenomena observed by Jackson et al.^[49] are owing in part to a process that occurs during soil drying where dry surface soil layer forms, significantly affecting the vapor transport through the profile.^[50] Yamanaka et al.^[51] developed and verified, using wind tunnel data, a simple energy balance model in which the soil moisture available for evaporation is defined using the depth of the evaporating/drying front in the soil. This approach removes the ambiguity of defining the thickness of the soil layer and resulting moisture available for evaporation. However, the depth of the evaporating surface is not generally known *a priori* nor can it be measured in field conditions; hence, this approach is

limited to exploring the effects of evaporating front on R_S type formulations.

Analytical Models

To reduce the effect of temporal varying, R_S , Chanzy and Bruckler^[38] developed a simple analytically based daily LE_S (E_D) model using simulations from their mechanistic model for different soil texture, moisture, and climatic conditions as quantified by potential evaporation (E_{PD}), as given by Penman.^[52] The analytical daily model requires midday 0–5 cm soil moisture θ , daily potential evaporation, and daily average wind speed (U_D). The simple model has the following form:

$$\frac{E_D}{E_{PD}} = \left[\frac{\exp(A\theta + B)}{1 + \exp(A\theta + B)} \right] C + (1 - C) \quad (10)$$

$$A = a + 5\max(3 - E_{PD}, 0) \quad (11)$$

$$B = b - 5(-0.025b - 0.05) \max(3 - E_{PD}, 0) + \alpha(U_D - 3) \quad (12)$$

$$C = 0.90 - 0.05c(U_D - 3) \quad (13)$$

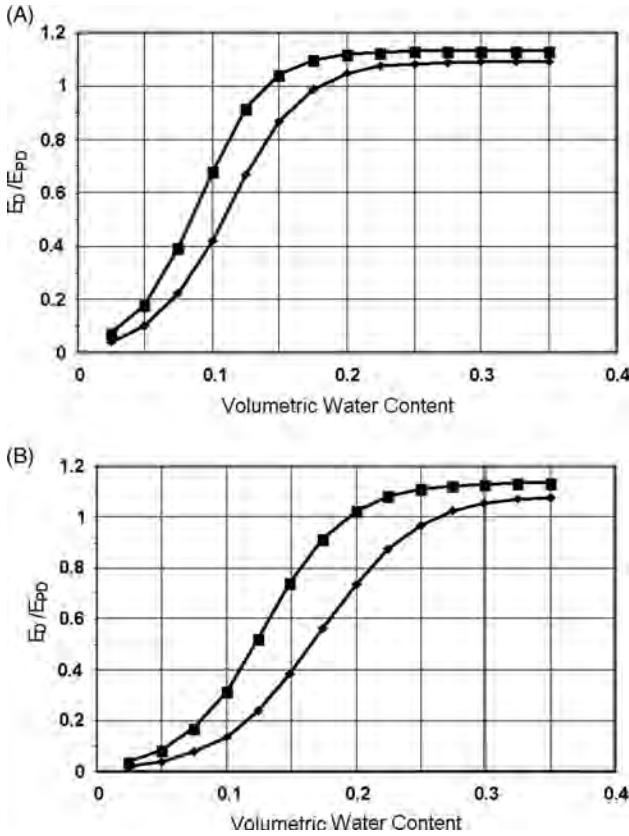


Fig. 1 A plot of E^D/E^{PD} , estimated using Eq. 10 vs. volumetric water content for (A) loam and (B) silty clay loam soil under two evaporative demand conditions: $U^D = 1$ m/sec and $E^{PD} = 2$ mm/day (squares) and $U^D = 5$ m/sec and $E^{PD} = 10$ mm/day (diamonds).

Source: From Chanzy & Bluckler.^[38]

where the coefficients a , b , and c depend on soil texture (Table 1) and were derived from their detailed mechanistic model simulations for loam, silty clay loam, and clay soils.^[38] In Fig. 1, a plot of Eq. 10 is given for two soil types, loam (Fig. 1A) and silty clay loam (Fig. 1B) under two climatic conditions, namely, a relatively low evaporative demand condition with $U_D = 1$ m/s and $E_{PD} = 2$ mm/day and high demand $U_D = 5$ m/s and $E_{PD} = 10$ mm/day. Notice the transition from $E_D/E_{PD} \sim 1$ to $E_D/E_{PD} < 1$ as a function of θ varies not only with the soil texture but also with the evaporative demand. The simplicity of such a scheme outlined in Eq. 10 needs further testing for different soil textures and under a wider range of climatic conditions.

The ratio E_D/E_{PD} as a function of θ illustrated in Fig. 1 also depicts the effect of the two “drying stages” typically used to describe soil evaporation.^[42] The “first stage” (S₁) of drying is under the condition where water is available in the near-surface soil to meet atmospheric demand, i.e., $E_D/E_{PD} \sim 1$. In the “second stage” (S₂) of drying, the water availability or θ falls below a certain threshold where the soil evaporation is no longer controlled by the evaporative demand, namely, $E_D/E_{PD} < 1$. Under S₂, several studies

find that a simple formulation can be derived by assuming that the time change in θ is governed by desorption, namely, as isothermal diffusion with negligible gravity effects from a semi-infinite uniform medium. This leads to the rate of evaporation for S₂ being approximated by^[42,53]

$$E_D = 0.5D_E t^{-1/2} \quad (14)$$

where the desorptivity D_E (mm/day^{1/2}) is assumed to be a constant for a particular soil type and t is the time (in days) from the start of S₂. Although both numerical models and observations indicate that the soil evaporation is certainly a more complicated process than the simple analytical expression given by Eq. 14, a number of field studies^[3,54–58] have shown that for S₂ conditions, reliable daily values can be obtained using Eq. 14. In many of these studies for determining D_E , the integral of Eq. 14 is used, which yields the cumulative evaporation as a function of $t^{1/2}$

$$\sum E_D = D_E(t - t_0)^{1/2} \quad (15)$$

where t_0 is the number of days where $E_D/E_{PD} \sim 1$ or is in S₁. In practice, observations of $\sum E_D$ are plotted vs. $(t - t_0)^{1/2}$, and in many cases the choice of the starting point of S₂ is $t_0 \approx$ or immediately after the soil is saturated. As shown by Campbell and Norman,^[58] the course of evaporation rate for three drying experiments (see Fig. 9. 6 in Campbell and Norman^[58]) indicates that for a loam soil, t_0 depends on the evaporative demand or E_{PD} with $t_0 \sim 2$ days when E_{PD} is high vs. $t_0 \sim 5$ days when E_{PD} is low. On the other hand, for a sandy soil, there is almost an immediate change from S₁ to S₂ conditions with $t_0 \approx 1$ day. As suggested by the analysis of Jackson et al.^[56] and as stated more explicitly by Brutsaert and Chen,^[59] the value of t_0 can significantly influence the value computed for D_E . Jackson et al.^[56] also show that for the same soil type, the value of D_E has a seasonal dependency (ranging from 4 to 8 mm/day^{1/2}) most likely related to the evaporative demand, which they correlate to daily average soil temperature (see Fig. 2 in Jackson et al.^[56]). Values of D_E from the various studies are listed in Table 2. Brutsaert and Chen^[59] modified Eq. 14 for deriving D_E by rewriting in terms of a “time-shifted” variable $T = t - t_0$ and expressing it in the form

$$E_D^{-2} = \left(\frac{2}{D_E}\right)^2 T \quad (16)$$

where D_E and t_0 will come from the slope and intercept (see Fig. 1 in Brutsaert and Chen^[59]). It follows that $\sum E_D$ under S₂ will start at $T = T_0$ and not at $T = 0$, so that Eq. 15 is rewritten as

$$\sum E_D = D_E(T^{1/2} - T_0^{1/2}) \quad (17)$$

They evaluated the effect on the derived D_E using this technique with the data from Black et al.^[54] The value of

Table 2 Values of desorptivity, D_E , evaluated from various experimental sites.

Desorptivity D_E (mm/day ^{1/2})	Soil type	References
4.96–4.30	Sand	[54]
5.08	Clay loam	[55]
4.04	Loam	[55]
3.5	Clay	[55]
~4 to ~8 ^a	Loam ^b	[56]
5.8	Clay loam	[57]
4.95 ^c	Silty clay loam ^d	[59]
2.11	Gravelly sandy loam	[3]

^aThe magnitude of D_E was found to have a seasonal dependency.

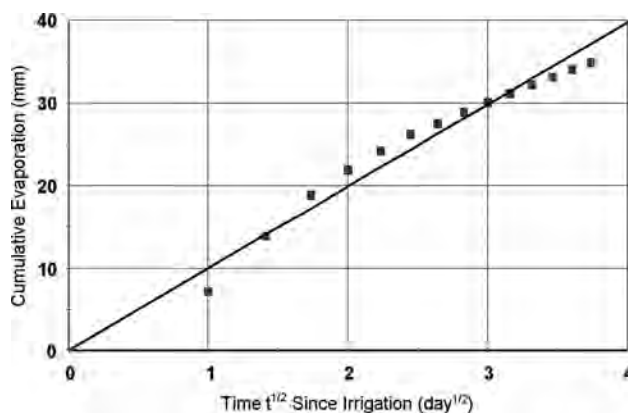
^bSoil type was determined from texture-dependent soil hydraulic conductivity and matric potential equations of Clapp and Hornberger evaluated by Camillo and Gurney.^[48]

^cThis value was evaluated for a vegetated surface.

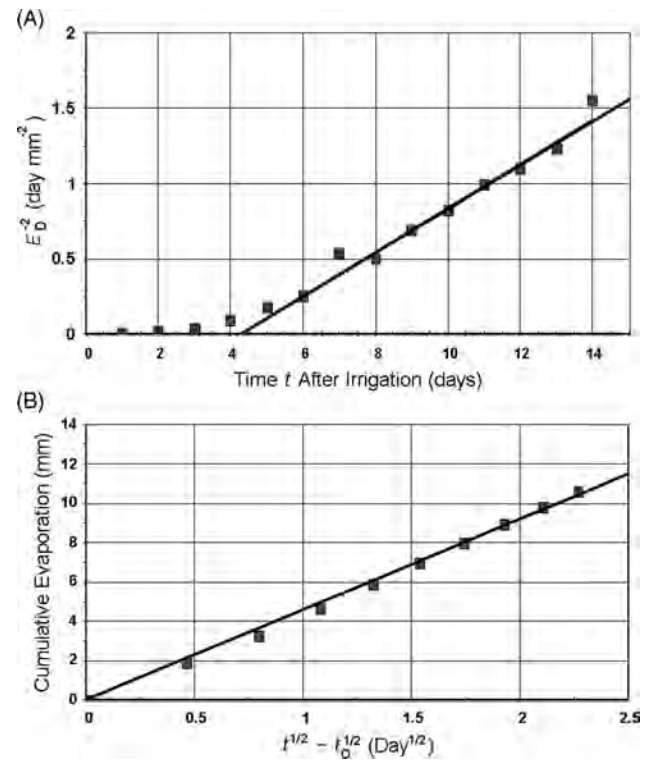
^dSoil type was determined from texture-dependent soil hydraulic conductivity and matric potential equations of Clapp and Hornberger evaluated by Sellers et al.^[41]

D_E using Eqs. 16 and 17 was estimated to be approximately 3.3 mm/day^{1/2}, which is smaller than D_E values reported by Black et al.,^[54] namely, 4.3–5 mm/day^{1/2}. However, Brutsaert and Chen^[59] show that this technique yields a better linear fit to the data points that were actually under S₂ conditions.

Equations 16 and 17 were used with the September 1973 data set from Jackson et al.^[56] and compared to using Eq. 15 with $t_O = 0$. The plot of Eq. 15 with the regression line in Fig. 2 yields $D_E \sim 10$ mm/day^{1/2}, which is significantly larger than any previous estimates (Table 2). Moreover, it is obvious from the figure that Eq. 15 should not be applied with $t_O = 0$, as this relationship is not linear over the whole drying processes. With Eq. 16, applied to the data, t_O is estimated to be approximately 4.3 days, and thus a linear

**Fig. 2** The desorptivity D_E (mm/day^{1/2}) estimated from a least squares linear fit to the data assuming $t_O = 0$ (i.e., stage-two drying occurs immediately after irrigation/precipitation).

Source: From Jackson, Idso, et al.^[56]

**Fig. 3** Estimation of (A) D_E and t_O with the data from Fig. 2 using Eq. 16 and (B) the resulting cumulative evaporation ΣE_D curve under second stage drying using Eq. 17.

Source: From Brutsaert & Chen.^[59]

relationship should start at the shifted time scale $T = t - 4.3$; this means ΣE_D should start on day 5 or $T_O \approx 5 - 4.3$ (Fig. 3A). With Eq. 17, a more realistic $D_E \approx 4.6$ mm/day^{1/2} is estimated for the linear portion of daily evaporation following the S₂ condition (Fig. 3B).

While this approach is relatively easy to implement operationally, D_E will likely depend on climatic factors as well as soil textural properties. However, it might be feasible to describe the main climate/seasonal effect on D_E from soil temperature observations.^[56] These might come from weather station observations or possibly from multitemporal remote sensing observations of surface temperature.

The difficulty in developing a formulation for R_S , which correctly describes the water vapor transfer process in the soil, was recognized much earlier by Fuchs and Tanner^[60] and Tanner and Fuchs.^[61] They proposed a combination method instead that involves atmospheric surface layer observations and remotely sensed surface temperature, T_{RS} . Starting with the energy balance equation

$$R_N = H + G + LE \quad (18)$$

where R_N is the net radiation, H the sensible heat flux, LE the latent heat flux, and G the soil heat flux all in W/m², and assuming the resistance to heat and water vapor transfer are similar yielding,

$$H = \rho C_p \left(\frac{T_{RS} - T_A}{R_A} \right) \quad (19)$$

$$LE = \frac{\rho C_p}{\gamma} \left(\frac{e_{RS} - e_A}{R_A} \right) \quad (20)$$

an equation of the following form can be derived

$$LE = \left(\frac{\gamma + \Delta}{\Delta} \right) LE_p - \frac{\rho C_p}{\Delta} \left(\frac{e_*(T_{RS}) - e_A}{R_A} \right) \quad (21)$$

where

$$LE_p = \rho C_p \left(\frac{e_*(T_A) - e_A}{R_A(\Delta + \gamma)} \right) + \Delta \left(\frac{R_N - G}{\Delta + \gamma} \right) \quad (22)$$

The difference $e_*(T_A) - e_A$ is commonly called the saturation vapor pressure deficit, and the value of soil surface vapor pressure e_{RS} is equal to $h_{RS}e_*(T_{RS})$ where h_{RS} is the soil surface relative humidity. Substituting Eq. 21 into Eq. 22 yields

$$LE = R_N - G - \frac{\rho C_p}{\Delta} \left(\frac{e_*(T_{RS}) - e_*(T_A)}{R_A} \right) \quad (23)$$

This equation has the advantage over the above bulk resistance formulations using R_S in that there are no assumptions made concerning the saturation deficit at or near the soil surface or how to define h_{RS} . Instead, this effect is accounted for by T_{RS} because as the soil dries, T_{RS} increases and hence $e_*(T_{RS})$, which generally results in the last term on the right-hand side of Eq. 23 to increase, thus causing LE to decrease. In a related approach,^[62] the magnitude of LE is simply computed as a residual in the energy balance equation, Eq. 21, namely,

$$LE = R_N - G - \rho C_p \left(\frac{T_{RS} - T_A}{R_A} \right) \quad (24)$$

Particularly crucial in the application of either Eqs. 23 or 24 is a reliable estimate of R_A and T_{RS} . Issues involved in correcting radiometric temperature observations for surface emissivity, viewing angle effects, and other factors are summarized in Norman and Becker.^[63] The aerodynamic resistance R_A is typically expressed in terms of Monin–Obukhov similarity theory^[42]

$$R_A = \frac{\left\{ \ln \left[\frac{z - d_0}{z_{OM}} \right] - \psi_M \right\} \left\{ \ln \left[\frac{z - d_0}{z_{OS}} \right] - \psi_S \right\}}{k^2 U} \quad (25)$$

where z is the observation height in the surface layer (typically 2–10 m), d_0 the displacement height, z_{OM} the momentum roughness length, z_{OS} the roughness length for scalars (i.e., heat and water vapor), k (~ 0.4) von Karman's constant, ψ_M the stability correction function for momentum, and ψ_S the stability correction function for scalars. Both d_0 and z_{OM} are dependent on the height and density of the roughness obstacles and can be considered a constant

for a given surface, while the magnitude of z_{OS} can vary for a given bare soil surface as it is also a function of the surface friction velocity.^[42] Experimental evidence suggests that existing theory with possible modification to some of the “constants” can still be used to determine z_{OS} providing acceptable estimates of H for bare soil surfaces. However, application of Eq. 24 in partial canopy cover conditions has not been successful in general because z_{OS} is not well defined in Eq. 25, exhibiting large scatter with the existing theory.^[64]

For this reason, estimating soil evaporation from partially vegetated surfaces using T_{RS} invariably has to involve “two-source” approaches, whereby the energy exchanges from the soil and vegetated components are explicitly treated.^[65] Similarly, when using remotely sensed soil moisture for vegetated surfaces, a two-source modeling framework needs to be applied.^[66] In these two-source approaches, there is the added complication of determining aerodynamic resistances between soil and vegetated surfaces and the canopy air space. Schematically, the resistance network and corresponding flux components for two-source models are shown in Fig. 4. An advantage with the two-source formulation of Norman et al.^[65] is that R_S is not actually needed for computing LE_s as it is solved as a residual. Nevertheless, the formulations in such parameterizations that are used (such as the aerodynamic resistance formulations) are likely to strongly influence LE_s values.^[1]

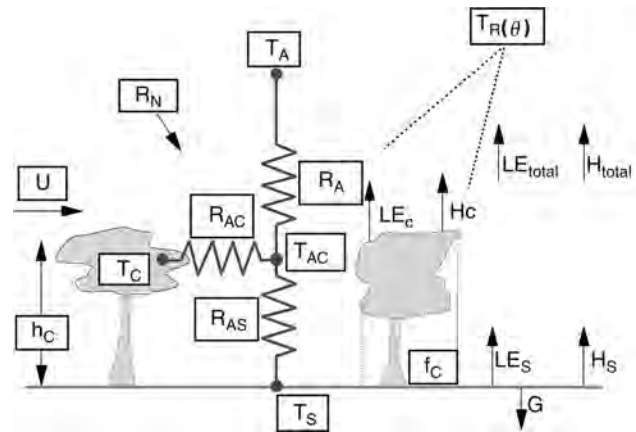


Fig. 4 A schematic diagram illustrating the resistance network for the two-source approach where the subscript c refers to the vegetated canopy and s refers to the soil surface. The symbol $T_R(\theta)$ refers to a radiometric temperature observation at a viewing angle θ , T_{AC} is the model-derived within-canopy air space temperature, h_C is the canopy height, f_C is the fractional vegetation/canopy cover, and R_{AC} and R_{AS} are the aerodynamic resistances to sensible heat flux from the canopy and soil surface, respectively. The main meteorological inputs required for the model are also illustrated, namely wind speed, U , air temperature, T_A , and the net radiation, R_N .

Source: Adapted from Norman, Kustas, et al.^[65]

Yet this two-source formulation is found to be fairly robust in separating soil and canopy contributions to evapotranspiration.^[67]

CONCLUSION

Techniques for measuring soil evaporation, separately from plant transpiration, exist and continually improve. Given the high variability in soil water content often exhibited in the field, the exact position of the instrumentation may have a large effect on the measurement. This is magnified in row crops, where the variability is structured, rather than random. In row crops, soil evaporation is not only dependent on the soil properties, but also local climatic conditions, and the timing of irrigation/precipitation. Under low-to-moderate wind speed conditions (~ 1 to 4 m sec^{-1}), row orientation relative to the wind direction also affects the amount of soil evaporation.^[68] This large variability requires that there be careful thought to the design of soil evaporation measurements in the field. The methodology proposed by Scanlon and Kustas^[5] using eddy covariance measurements suffers less from the sampling issue but requires *a priori* knowledge of the vegetation water use efficiency, which will vary with vegetation type and condition.

For many landscapes having partial vegetative cover, the contribution of soil evaporation to the total evapotranspiration flux cannot be ignored, particularly with regard to the influence of surface soil moisture and temperature on the microclimate in the canopy air space (Fig. 4). Numerical models for simulating soil evaporation have been developed and are shown to be fairly robust. However, the required inputs for defining model parameters often limit their application to field sites having detailed soil profile information and that are well instrumented with ancillary weather data.

For many operational applications, where detailed soil and ancillary weather data are unavailable or where daily evaporation values are only needed, some of the analytical models described may provide the necessary level of accuracy. Moreover, in the application of weather forecast and hydrologic models, the use of simplified approaches is necessitated by the computational requirements or the lack of adequate data or both for defining more complex numerical model parameters and variables.

For large area estimation, the use of remotely sensed soil moisture and surface temperature offers the greatest potential for operational applications. Development of modeling schemes that can incorporate this remote sensing information and readily apply it on a regional scale basis has been proposed and shows promise.^[66,69–71]

REFERENCES

1. Kustas, W.P.; Norman, J.M. Evaluation of soil and vegetation heat flux predictions using a simple two-source model with radiometric temperatures for partial canopy cover. *Agric. Forest Meteorol.* **1999**, *94*, 13–25.
2. Tanner, C.B.; Jury, W.A. Estimating evaporation and transpiration from a row crop during incomplete cover. *Agron. J.* **1976**, *68*, 239–242.
3. Wallace, J.S.; Holwill, C.J. Soil evaporation from tiger-bush in south-west Niger. *J. Hydrol.* **1997**, *188–189*, 426–442.
4. Agam, N.; Berliner, P.R.; Zangvil, A.; Ben-Dor, E. Soil water evaporation during the dry season in an arid zone. *J. Geophys. Res. Atmos.* **2004**, *109*, 1–D16103. DOI:16110.11029/12004JD004802.
5. Scanlon, T.M.; Kustas, W.P. Partitioning carbon dioxide and water vapor fluxes using correlation analysis. *Agric. Forest Meteorol.* **2010**, *150*, 89–99.
6. Allen, S.J. Measurement and estimation of evaporation from soil under sparse barley crops in northern Syria. *Agric. Forest Meteorol.* **1990**, *49*, 291–309.
7. Van de griend, A.A.; Owe, M. Bare soil surface resistance to evaporation by vapor diffusion under semiarid conditions. *Water Resour. Res.* **1994**, *30*, 181–188.
8. Plauborg, F. Evaporation from bare soil in a temperate humid climate – Measurement using micro-lysimeters and time domain reflectometry. *Agric. Forest Meteorol.* **1995**, *76*, 1–17.
9. Baker, J.M.; Spaans, E.J. Measuring water exchange between soil and atmosphere with TDR-Microlysimetry. *Soil Sci.* **1994**, *158*, 22–30.
10. Ashktorab, H.; Pruitt, W.O.; Pawu, K.T.; George, W.V. Energy-balance determinations close to the soil surface using a micro-Bowen ratio system. *Agric. Forest Meteorol.* **1989**, *46*, 259–274.
11. Heitman, J.L.; Horton, R.; Sauer, T.J.; Desutter, T.M. Sensible heat observations reveal soil-water evaporation dynamics. *J. Hydrometeorol.* **2008a**, *9*, 165–171.
12. Shawcroft, R.W.; Gardner, H.R. Direct evaporation from soil under a row crop canopy. *Agric. Meteorol.* **1983**, *28*, 229–238.
13. Lascano, R.J.; Van bavel, C.H.M. Simulation and measurement of evaporation from a bare soil. *Soil Sci. Soc. Am. J.* **1986**, *50*, 1127–1133.
14. Boast, C.W.; Robertson, T.M. A “micro-lysimeter” method for determining evaporation from bare-soil: Description and laboratory evaluation. *Soil Sci. Soc. Am. J.* **1982**, *46*, 689–696.
15. Walker, G.K. Measurement of evaporation from soil beneath crop canopies. *Can. J. Soil Sci.* **1983**, *63*, 137–141.
16. Evett, S.R.; Warrick, A.W.; Matthias, A.D. Wall material and capping effects on microlysimeter temperatures and evaporation. *Soil Sci. Soc. Am. J.* **1995**, *59*, 329–336.
17. Denmead, O.T.; Dunin, F.X.; Wong, S.C.; Greenwood, E.A. N. Measuring water use efficiency of Eucalyptus trees with chambers and micrometeorological techniques. *J. Hydrol.* **1993**, *150*, 649–664.
18. Livingston, G.P.; Hutchinson, G.L. Enclosure-based measurement of trace gas exchange: Applications and sources of error. In *Biogenic Trace Gases: Measuring Emissions from Soil and Water*; Matson, P.A., Harriss, R.C., Eds.; Blackwell Scientific Publications: Oxford, 1995; 14–51.
19. Hutchinson, G.L.; Livingston, G.P. Gas flux. In *Methods of Soil Analysis: Part 1, Physical Methods*; Dane, J.H.,

- Topp, G.C., Eds.; Soil Science Society of America: Madison, 2002.
20. Davidson, E.A.; Savage, K.; Verchot, L.V.; Navarro, R. Minimizing artifacts and biases in chamber-based measurements of soil respiration. *Agric. Forest Meteorol.* **2002**, *113*, 21–37.
 21. Norman, J.M.; Kucharik, C.J.; Gower, S.T.; Baldocchi, D.D.; Crill, P.M.; Rayment, M.; Savage, K.; Striegl, R.G. A comparison of six methods for measuring soil-surface carbon dioxide fluxes. *J. Geophys. Res.-Atm.* **1997**, *102*, 28771–28777.
 22. Evett, S.R.; Schwartz, R.C.; Casanova, J.J.; Heng, L.K. Soil water sensing for water balance, ET and WUE. *Agric. Water Manag.* **2012**, *104*, 1–9.
 23. Topp, G.C.; Davis, H.L.; Annan, A.P. Electromagnetic determination of soil water content: Measurements in coaxial transmission lines. *Water Resour. Res.* **1980**, *16*, 574–582.
 24. Logsdon, S.D.; Green, T.R.; Seyfried, M.S.; Evett, S.R.; Bonta, J. Hydra probe and twelve-wire probe comparisons in fluids and soil cores. *Soil Sci. Soc. Am. J.* **2010**, *74*, 5–12.
 25. Miralles-Crespo, J.; Van Lersel, M.W. A calibrated time domain transmissometry soil moisture sensor can be used for precise automated irrigation of container-grown plants. *Hort. Science* **2011**, *46*, 889–894.
 26. Allen, R.G.; Pereira, L.S.; Howell, T.A.; Jensen, M.E. Evapotranspiration information reporting: I. Factors governing measurement accuracy. *Agric. Water Manag.* **2011**, *98*, 899–920.
 27. Farahani, H.J.; Howell, T.A.; Shuttleworth, W.J.; Bausch, W.C. Evapotranspiration: Progress in measurement and modeling in agriculture. *Trans. Am. Soc. Agric. Biol. Engineers* **2007**, *50*, 1627–1638.
 28. Bowen, I.S. The ratio of heat losses by conduction and by evaporation from any water surface. *Phys. Rev.* **1926**, *27*, 779–787.
 29. Monteith, J.L.; Unsworth, M.H. *Principles of Environmental Physics*, 3rd Ed.; Elsevier: Amsterdam, 2008.
 30. Zeggaf, A.T.; Takeuchi, S.; Dehghanisani, H.; Anyoji, H.; Yano, T. A Bowen ratio technique for partitioning energy fluxes between maize transpiration and soil surface evaporation. *Agronomy J.* **2008**, *100*, 988–996.
 31. Heitman, J.L.; Xiao, X.; Horton, R.; Sauer, T.J. Sensible heat measurements indicating depth and magnitude of subsurface soil water evaporation. *Water Resour. Res.* **2008b**, 1–44.
 32. Gardner, H.R.; Hanks, R.J. Evaluation of the evaporation zone in soil by measurement of heat flux. *Soil Sci. Soc. Am. Proc.* **1966**, *32*, 326–328.
 33. Ren, T.S.; Ochsner, T.E.; Horton, R. Development of thermo-time domain reflectometry for vadose zone measurements. *Vadose Zone J.* **2003**, *2*, 544–551.
 34. Knight, J.H.; Kluitenberg, G.J. Simplified computational approach for dual-probe heat-pulse method. *Soil Sci. Soc. Am. J.* **68**, 447–449.
 35. Bristow, K.L.; Kluitenberg, G.J.; Horton, R. Measurement of soil thermal-properties with a dual-probe heat-pulse technique. *Soil Sci. Soc. Am. J.* **1994**, *58*, 1288–1294.
 36. Philip, J.R.; de Vries, D.A. Moisture movement in porous materials under temperature gradients. *Trans. Am. Geophys. Union* **1957**, *38* (2), 222–232.
 37. Camillo, P.J.; Gurney, R.J.; Schugge, T.J. A soil and atmospheric boundary layer model for evapotranspiration and soil moisture studies. *Water Resour. Res.* **1983**, *19*, 371–380.
 38. Chanzy, A.; Bruckler, L. Significance of soil surface moisture with respect to daily bare soil evaporation. *Water Resour. Res.* **1993**, *29*, 1113–1125.
 39. Daamen, C.C.; Simmonds, L.P. Measurement of evaporation from bare soil and its estimation using surface resistance. *Water Resour. Res.* **1996**, *32*, 1393–1402.
 40. Noilhan, J.; Planton, S. A simple parameterization of land surface processes for meteorological models. *Mon. Weather Rev.* **1989**, *117*, 536–549.
 41. Sellers, P.J.; Heiser, M.D.; Hall, F.G. Relations between surface conductance and spectral vegetation indices at intermediate (100 m² to 15 km²) length scales. *J. Geophys. Res.* **1992**, *97*, 19033–19059.
 42. Brutsaert, W. *Evaporation into the Atmosphere: Theory, History and Applications*; D. Reidel Publishing Co.: Boston, 1982.
 43. Mahfouf, J.F.; Noilhan, J. Comparative study of various formulations of evaporation from bare soil using *in situ* data. *J. Appl. Meteorol.* **1991**, *30*, 1354–1365.
 44. Ye, Z.; Pielke, R.A. Atmospheric parameterization of evaporation from non-plant-covered surfaces. *J. Appl. Meteorol.* **1993**, *32*, 1248–1258.
 45. Philip, J.R. 1957, Evaporation, and moisture and heat fields in the soil. *Journal of Meteorology*. *14*, 354–366.
 46. Kondo, J.; Saigusa, N.; Sato, T. A parameterization of evaporation from bare soil surfaces. *J. Appl. Meteorol.* **1990**, *29*, 385–389.
 47. Cahill, A.T.; Parlange, M.B.; Jackson, T.J.; O'Neill, P.; Schugge, T.J. Evaporation from nonvegetated surfaces: Surface aridity methods and passive microwave remote sensing. *J. Appl. Meteorol.* **1999**, 1–38.
 48. Camillo, P.J.; Gurney, R.J. A resistance parameter for bare-soil evaporation models. *Soil Sci.* **1986**, *141*, 95–104.
 49. Jackson, R.D.; Kimball, B.A.; Reginato, R.J.; Nakayama, F.S. Diurnal soil water evaporation: Time-depth-flux patterns. *Soil Sci. Soc. Am. Proc.* **1973**, *37*, 505–509.
 50. Hillel, D. *Applications of Soil Physics*; Academic Press: San Diego, 1980.
 51. Yamanaka, T.; Takeda, A.; Sugita, F. A modified surface-resistance approach for representing bare-soil evaporation: Wind tunnel experiments under various atmospheric conditions. *Water Resour. Res.* **1997**, *33*, 2117–2128.
 52. Penman, H.L. Natural evaporation from open water, bare soil and grass. *Proc. R. Soc. Lond.* **1948**, *A193*, 120–146.
 53. Gardner, W.R. Solution of the flow equation for the drying of soils and other porous media. *Soil Sci. Soc. Am. Proc.* **1959**, *23*, 183–187.
 54. Black, R.A.; Gardner, W.R.; Thurtell, G.W. The prediction of evaporation, drainage, and soil water storage for a bare soil. *Soil Sci. Soc. Am. Proc.* **1969**, *33*, 655–660.
 55. Ritchie, J.T. Model for predicting evaporation from a row crop with incomplete cover. *Water Resour. Res.* **1972**, *8*, 1204–1213.
 56. Jackson, R.D.; Idso, S.B.; Reginato, R.J. Calculation of evaporation rates during the transition from energy-limiting to soil-limiting phases using albedo data. *Water Resour. Res.* **1976**, *12*, 23–26.

57. Parlange, M.B.; Katul, G.G.; Cuenca, R.H.; Kavvas, M.L.; Nielsen, D.R.; Mata, M. Physical basis for a time series model of soil water content. *Water Resour. Res.* **1992**, *28*, 2437–2446.
58. Campbell, G.S.; Norman, J.M. *An Introduction to Environmental Biophysics*, 2nd Ed.; Springer-Verlag: New York, 1998.
59. Brutsaert, W.; Chen, D. Desorption and the two stages of drying of natural tallgrass prairie. *Water Resour. Res.* **1995**, *31*, 1305–1313.
60. Fuchs, M.; Tanner, C.B. Evaporation from a drying soil. *J. Appl. Meteorol.* **1967**, *6*, 852–857.
61. Tanner, C.B.; Fuchs, M. Evaporation from unsaturated surfaces: A generalized combination method. *J. Geophys. Res.* **1968**, *73*, 1299–1304.
62. Jackson, R.D.; Reginato, R.J.; Idso, S.B. Wheat canopy temperature: A practical tool for evaluating water requirements. *Water Resour. Res.* **1977**, *13*, 651–656.
63. Norman, J.M.; Becker, F. Terminology in thermal infrared remote sensing of natural surfaces. *Agric. Forest Meteorol.* **1995**, *77*, 153–166.
64. Verhoef, A.; De bruin, H.A.R.; Van den hurk, B.J.J.M. Some practical notes on the parameter kB-1 for sparse vegetation. *J. Appl. Meteorol.* **1997**, *36*, 560–572.
65. Norman, J.M.; Kustas, W.P.; Humes, K.S. A two-source approach for estimating soil and vegetation energy fluxes in observations of directional radiometric surface temperature. *Agric. Forest Meteorol.* **1995**, *77*, 263–293.
66. Kustas, W.P.; Zhan, X.; Schmugge, T.J. Combining optical and microwave remote sensing for mapping energy fluxes in a semiarid watershed. *Rem. Sen. Environ.* **1998**, *64*, 116–131.
67. Kustas, W.P.; Anderson, M.C. Advances in thermal infrared remote sensing for land surface modeling. *Agric. Forest Meteorol.* **2009**, *149*, 2071–2081.
68. Agam, N.; Evett, S.R.; Tolk, J.A.; Kustas, W.P.; Colaizzi, P.D.; Alfieri, J.G.; Mckee, L.G.; Copeland, K.S.; Howell, T.A.; Chávez, J.L. Evaporative loss from irrigated interrows in a highly advective semi-arid agricultural area. *Adv. Water Resour.* **2012**, DOI:10.1016/j.advwatres.2012.07.010.
69. Mecikalski, J.R.; Diak, G.R.; Anderson, M.C.; Norman, J.M. Estimating fluxes on continental scales using remotely sensed data in an atmospheric-land exchange model. *J. Appl. Meteorol.* **1999**, *38*, 1352–1369.
70. Kustas, W.P.; Jackson, T.J.; French, A.N.; Macpherson, J.I. Verification of patch- and regional-scale energy balance estimates derived from microwave and optical remote sensing during SGP97. *J. Hydrometeorol.* **2001**, *2*, 254–273.
71. Anderson, M.C.; Kustas, W.P.; Norman, J.M.; Hain, C.R.; Mecikalski, J.R.; Schultz, L.; Gonzalez-Dugo, M.P.; Cammalleri, C.; D'urso, G.; Pimstein, A.; Gao, F. Mapping daily evapotranspiration at field to continental scales using geostationary and polar orbiting satellite imagery. *Hydrol. Earth Syst. Sci.* **2011**, *15*, 223–239.

Evolution: Soil Seed Bank

Paul B. Cavers

Plant Sciences, University of Western Ontario, London, Ontario, Canada

David Susko

Environmental Science, Department of Natural Sciences, University of Michigan at Dearborn, Dearborn, Michigan, U.S.A.

Abstract

The number and composition of plant seeds found within the soil in a given unit of area comprises the soil seed bank. Seed banks can be incredibly large, often consisting of hundreds to hundreds of thousands of seeds per square meter in a variety of different habitats and communities around the world. These seeds can be short- or long-lived. Seed banks provide a “memory” of prevailing site conditions, as well as climatic and edaphic conditions present many years or decades earlier. Hence, much of the genetic diversity of plant populations is preserved in seed banks. Furthermore, as seeds age in the soil, mutations accumulate and novel variation is introduced into a population. Soil seed banks play a pivotal role in the formation, maintenance, and evolution of plant communities.

INTRODUCTION

The development of the concept of the soil seed bank is generally attributed to Harper.^[1] He separated the typical “bank” into a “deposit account,” consisting of seeds in a dormant state and a “current account,” consisting of non-germinated seeds “in which the only hindrance to immediate germination is a shortage of water and a favorable temperature.” The first use of the term “seed bank” is attributed to Sarukhán.^[2] Over the 20th century, there have been hundreds of studies of seed banks made in a variety of terrestrial and aquatic habitats. Seed banks can be incredibly complex; often with hundreds of different species distributed heterogeneously, with few to hundreds of thousands of seeds per species in each square meter of soil and with great variability in dormancy and longevity, even among the seeds of a single species. Our understanding of this complexity is far from complete.^[3]

SIZES OF SEED BANKS IN DIFFERENT HABITATS AND REGIONS

The numbers of seeds in seed banks can be determined by 1) extraction of seeds followed by identification and counting or 2) identification and counting of seedlings arising from soil samples put under controlled conditions in greenhouses or growth rooms. There are problems with both approaches.^[4] Seed extraction is laborious and may miss small or cryptically colored seeds; additionally, some viable seeds can die before subsequent germination tests. Counts of seedlings, the preferred method in most studies,^[5] will

miss seeds that are alive but unable to germinate under the conditions provided. For the most abundant species, there is usually a good correlation between the two procedures.^[4]

Fenner^[6] summarized the reports on the numbers of seeds found in a wide range of communities; 10^2 – 10^3 m⁻² for forest soils, 10^3 – 10^6 m⁻² for grasslands, and 10^3 – 10^5 m⁻² for arable soils. Arable and disturbed soils often have huge seed banks,^[5] but seed numbers in arable soils have declined as use of chemical herbicides has increased.^[7] Wet sites and marshlands can have enormous seed banks as well.^[8] Baskin and Baskin^[9] presented data on seed banks for 78 disparate communities. In general, their values fell into the ranges described by Fenner, but they cited several forests and grasslands with seed banks having <10 seeds m⁻² vs. up to 646,000 seeds m⁻² in arable land.

SEED LONGEVITY IN THE SOIL

Two well-known experiments, one started by Beal in 1879^[10] and the second by Duvel in 1905,^[11] have shown that the seeds of some plant species can remain dormant for decades, and that crop seeds have the shortest longevity, on average, and weed seeds the greatest. Since then, there have been numerous reviews of seed longevity in seed banks. Baskin and Baskin^[9] separated seed bank studies into 1) those with freshly collected seeds that are buried for specified periods before testing and 2) those that infer the age of buried seeds by dating material in the soil around them.

Studies in which fresh seeds have been buried for various lengths of time are numerous but very few of them have extended beyond 10 years.^[9] Beal’s experiment is still

continuing,^[10] but numbers of seeds per sample are small, and only one set of storage conditions was used. None of the other studies exceeds 40 years, and only two continuing studies are designed to reach 50 years.^[9] There is a great need for large, long-term, comprehensive studies of this type.

In studies of known-age seeds, the decrease in seed numbers over time generally follows the pattern of an exponential decay curve; a constant percentage being lost each year through death and germination.^[5] Seed dormancy overcomes and germination is initiated by many factors. Among the most important are light (including quality, quantity, and daylength), diurnally alternating temperatures, wet stratification at chilling temperatures (0–6°C), and soil nutrients, especially nitrogen (nitrate) levels.^[9]

VARIABILITY IN SEED BANKS IN TIME AND SPACE

Seed banks are more complex than communities of growing plants, primarily because they contain species that are not present as growing plants and because species found as growing plants have additional genotypes present in the seed bank. In addition, seed banks are variable over time and space.

Variability in Time

When viable seeds are retrieved from the seed bank, they can be classified as either: 1) able to germinate when exposed to favorable conditions; or 2) remaining dormant when exposed to a favorable germination regime.^[12] Many fresh seeds fall into the latter category, but this dormancy is often lost after a few weeks or months.^[12] Furthermore, unfavorable environmental conditions, such as high summer temperatures or drought, can return seeds from the former to the latter category.

Thompson and Grime^[12] made an important distinction by classifying the seed banks of individual species as either: 1) “transient,” lasting for less than 1 year and consequently with periods each year when no viable seeds are available to replace or reinforce the plant population; or 2) “persistent” lasting for more than 1 year with remaining seeds overlapping fresh inputs each year. Thompson and Grime^[12] produced four models of seed banks, two transient and two persistent, which have been used as the basis for comparison in many subsequent articles.

Variability in Space

One of the greatest problems facing an investigator of seed banks is how to estimate accurately their sizes, given the unknown and notoriously heterogeneous distribution of seeds, both laterally and at different depths in the soil.^[13,14] For example, the number of seeds per unit area declines

rapidly with increasing depth in the soil.^[9,15] Further, seeds buried at greater depths tend to remain dormant longer, but fewer of them can germinate and emerge successfully to become established seedlings.^[15]

SEED BANK GENETICS

The offspring of plant genotypes that existed may be preserved together in the soil, since seed banks can retain non-germinating but viable seeds for many years or even centuries. Hence, the majority of genetic variation present in a plant population (i.e., the gene pool) is often contained within the seed bank, particularly in short-lived species with long-lived persistent seed banks, while only a fraction of this total variation is present in surface plants at any given date.

Levin^[16] summarized the genetic consequences of a persistent seed bank on a population's gene pool. First, a seed bank maintains the genetic composition of a population and buffers it from extinction by increasing the effective population size, even when plant population size fluctuates dramatically over time.^[17] For instance, the genotypes of surface plants that fail to reproduce before they die may still be present in the seed bank. Second, the seed bank slows the rate of response of seedling or adult plant characters to selection, since gene loci not directly related to seed dormancy are constrained as much as those that influence dormancy. The degree of evolutionary retardation is dependent on seed longevity in the seed bank; retardation is greater in species with long-lived seeds than in those with short-term seed viability, because of the increased genetic load in seed banks containing long-lived seeds.^[18] Third, selection is biased toward traits associated with years of high seed production. As a consequence, the seed bank can serve as an evolutionary filter, where seeds from years of large seed crops may be favored by selection over seeds from years of small seed crops.^[19] Lastly, seed banks may be sources of novel genetic variation. Levin^[16] noted that as seeds age, seed viability declines, whereas chromosome damage and the frequency of mutations within seeds increase. If such mutations are not lethal and can be transmitted to future generations, then the potential for plant populations to evolve may be increased due to the presence of a persistent seed bank.

Empirical studies have compared the amount and distribution of genetic variation in seed banks and surface plants. Typically, such comparisons among seeds of different ages within the seed bank, or among seeds in the soil vs. surface plants are made either 1) by examining morphological or physiological traits of genotypes when grown in common and multiple environment experiments^[20] or 2) by analyzing polymorphic allozyme and isozyme loci using starch gel electrophoresis.^[21–23] From the former technique, consistent phenotypic differences were found among seedlings arising from young (recently buried) and old seeds in the

seed bank when the plants were grown at different densities, temperatures, and nutrient levels.^[20] These results indicated genetic differences between seeds of different ages, but our understanding of the causal mechanisms of such differences is incomplete.

Studies using electrophoretic techniques have shown, in general, that allele frequencies in seed banks differ significantly from those in surface plants,^[21–23] but one study found little differentiation.^[17] Furthermore, many of these studies have found greater homozygosity in the seed bank than in later stages of the life cycle.^[21–23] These results suggest that the genotypes of surface plants constitute only a genetic subpopulation of the gene pool present in the soil seed bank. Hence, seed banks may be particularly important for the persistence and/or recovery of rare and endangered plant species. Presumably, susceptibility to genetic drift and inbreeding depression in small, scarce populations could be mediated by the presence of a homogeneously distributed, genetically diverse reserve of seeds. Examples of such mediation are few, but a study on the rare annual *Clarkia springvillensis* by McCue and Holtsford^[23] supports this argument.

CONCLUSION

The number and composition of plant seeds found within the soil in a given unit of area comprises the soil seed bank. Seed banks can be incredibly large, often consisting of hundreds to hundreds of thousands of seeds per square meter in a variety of different habitats and communities around the world. Also, these seeds can be short- or long-lived. Seed banks provide a “memory” of prevailing site conditions, as well as climatic and edaphic conditions present many years or decades earlier. Hence, much of the genetic diversity of plant populations is preserved in seed banks. Furthermore, as seeds age in the soil, mutations accumulate and novel variation is introduced into a population. It is clear from the accumulated information in this entry that soil seed banks play a pivotal role in the formation, maintenance, and evolution of plant communities.

REFERENCES

1. Harper, J.L. *Population Biology of Plants*; Academic Press: New York, 1977.
2. Sarukhán, J. Studies on plant demography: *Ranunculus repens* L., *R. bulbosus* L. and *R. acris* L. II. reproductive strategies and seed population dynamics. *J. Ecol.* **1974**, *62* (1), 151–177.
3. Vázquez-Yanes, C.; Orozco-Segovia, A.; Sánchez-Coronado, M.E.; Rojas-Arèchiga, M.; Batis, A.I. Seed ecology at the northern limit of the tropical rain forest in America. In *Seed Biology: Advances and Applications*; Black, M., Bradford, K.J., Vázquez-Ramos, J., Eds.; CAB International: Wallingford, 2000; 375–388.

4. Gross, K.L. A comparison of methods for estimating seed numbers in the soil. *J. Ecol.* **1990**, *78* (4), 1079–1093.
5. Roberts, H.A. Seed banks in soil. *Adv. Appl. Biol.* **1981**, *6*, 1–55.
6. Fenner, M. *Seed Ecology*; Chapman and Hall: London, 1985.
7. Cavers, P.B.; Benoit, D.L. Seed banks in arable land. In *Ecology of Soil Seed Banks*; Leck, M.A., Parker, V.T., Simpson, R.L., Eds.; Academic Press: San Diego, 1989; 309–328.
8. Staniforth, R.J.; Griller, N.; Lajzerowicz, C. Soil seed banks from coastal subarctic ecosystems of bird cove, Hudson Bay. *Ecoscience* **1998**, *5* (2), 241–249.
9. Baskin, C.C.; Baskin, J.M. *Seeds: Ecology, Biogeography and Evolution of Dormancy and Germination*; Academic Press: San Diego, 1998.
10. Kivilaan, A.; Bandurski, R.S. The one hundred-year period for Dr. Beal's seed viability experiment. *Am. J. Bot.* **1981**, *68* (9), 1290–1291.
11. Toole, E.H.; Brown, E. Final results of the duvel buried seed experiment. *J. Agr. Res.* **1946**, *72* (6), 201–210.
12. Thompson, K.; Grime, J.P. Seasonal variation in the seed banks of herbaceous species in ten contrasting habitats. *J. Ecol.* **1979**, *67* (3), 893–921.
13. Benoit, D.L.; Derksen, D.A.; Panneton, B. Innovative approaches to seed bank studies. *Weed Sci.* **1992**, *40* (4), 660–669.
14. Dessaint, F.; Barralis, G.; Caixinhas, M.L.; Mayor, J.-P.; Recasens, J.; Zanin, G. Precision of soil seedbank sampling: how many soil cores? *Weed Res.* **1996**, *36* (2), 143–151.
15. Cavers, P.B. Seed demography. *Can. J. Botany* **1983**, *61* (12), 3578–3590.
16. Levin, D.A. The seed bank as a source of genetic novelty in plants. *Am. Nat.* **1990**, *135* (4), 563–572.
17. Mahy, G.; Vekemans, X.; Jacquemart, A.-L. Patterns of allozymic variation within *Calluna vulgaris* populations at seed bank and adult stages. *Heredity* **1999**, *82* (4), 432–440.
18. Templeton, A.R.; Levin, D.A. Evolutionary consequences of seed pools. *Am. Nat.* **1979**, *114* (2), 232–249.
19. Venable, D.L. Modelling the evolutionary ecology of seed banks. In *Ecology of Soil Seed Banks*; Leck, M.A., Parker, V.T., Simpson, R.L., Eds.; Academic Press: San Diego, 1989; 67–87.
20. Vavrek, M.C.; McGraw, J.B.; Bennington, C.C. Ecological genetic variation in seed banks. III. Phenotypic and genetic differences between young and old seed populations of *Carex bigelowii*. *J. Ecol.* **1991**, *79* (3), 645–662.
21. Tonsor, S.J.; Kalisz, S.; Fisher, J.; Holtsford, T.P. A life-history based study of population genetic structure: Seed bank to adults in *Plantago lanceolata*. *Evolution* **1993**, *47* (3), 833–843.
22. Cabin, R.J.; Mitchell, R.J.; Marshall, D.L. Do surface plant and soil seed bank populations differ genetically? a multipopulation study of the desert mustard *Lesquerella fendleri* (brassicaceae). *Am. J. Bot.* **1998**, *85* (8), 1098–1109.
23. McCue, K.A.; Holtsford, T.P. Seed bank influences on genetic diversity in the rare annual *Clarkia Springvillensis* (Onagraceae). *Am. J. Bot.* **1998**, *85* (1), 30–36.

Fauna

Mary C. Savin

Department of Crop, Soil, and Environmental Sciences, University of Arkansas, Fayetteville, Arkansas, U.S.A.

Abstract

Soil fauna are diverse, abundant, and critical to ecosystem functions. Soil fauna account for almost one-quarter of living species and facilitate important ecological interactions that contribute to decomposition, nutrient cycling, and stabilization of terrestrial ecosystems. Soil fauna are composed of diverse groups of organisms that range across orders of magnitude in size, with many less than a few centimeters or even a few millimeters. These organisms function under different abiotic conditions and, despite their small size, contribute to characteristics and processes observable at field and landscape scales. Research efforts that build upon past discoveries should enhance the understanding of how soil fauna connect aboveground and belowground ecology for proper management and sustainability of ecosystems and ecosystem services.

INTRODUCTION

Soil fauna are diverse, abundant, and critical to ecosystem functions. Soil fauna account for almost 25% of living species.^[1] Although they compose a group of organisms that has been ignored historically when it comes to land management, there is growing recognition and understanding of the roles soil fauna play in ecosystems. Appreciation for which organisms are present in soil, their habitats, community diversity, and contributions to ecosystem functions, stability, and sustainability is increasing with accumulating evidence from scientific research. Further development of our understanding of soil ecology is essential because soil fauna contribute directly and indirectly to properties and processes at the scale of the organism and its habitat with outcomes observed up to field and landscape levels. Future research efforts that build on past discoveries should enhance our understanding of how soil fauna connect aboveground and belowground ecology for the proper management and sustainability of ecosystems and ecosystem services.^[2]

SOIL FAUNA—ORGANISMS

Soil fauna, protists, and animals include organisms that span orders of magnitude in size (from micrometer to centimeter to meter). Generally, soil fauna are eukaryotic, aerobic chemoheterotrophs, but they occupy many niches in soil food webs.^[3] Organisms are commonly distributed into groupings that can range from low to high taxonomic resolution. One common differentiation is classification by size. Sized-based groups are microfauna, mesofauna, macrofauna, and megafauna. (See soil microbiology and

ecology textbooks for more detailed descriptions of the groups.^[4–7])

Because categorization based on size is an empirical separation, the upper size cutoff value may differ depending on the reference in the literature that one is searching. Microfauna may be considered up to 100–200 μm in length and include protozoa and nematodes. Nematodes (phylum Nematoda) may also be designated as mesofauna in the literature. Protozoa (kingdom Protista) are often grouped into flagellates, amoeba, and ciliates. Flagellates and amoeba tend to be smaller and numerous, while ciliates are relatively larger and less numerous. Protozoa consume bacteria, protozoa, and other organisms. Abundances range from 10 to 10^5 g^{-1} soil.^[8] (The diverse range of numbers for soil faunal abundances is related to the heterogeneity of soil conditions.) Nematodes, roundworms, are often grouped by into trophic groups: bacterivores, fungivores, omnivores, predators, and plant parasites. Microfauna, and also the mesofauna enchytraeid, are considered to reside in water films and water-filled pores inside aggregates.^[9,10]

Mesofauna range from 100 (or 200) μm to 2000 (sometimes considered up to 10,000) μm . At this larger size range, mesofauna are considered to be in the interaggregate pore space, which is likely to be air-filled, rather than the intra-aggregate pore space expected to be occupied by microfauna.^[11] This category includes micro-arthropods such as springtails (subclass Collembola) and mites (subclass Acari), and the enchytraeid (family Enchytraeidae; potworms). Micro-arthropod abundances may range from 10 to 10^5 m^{-2} (reviewed in Coleman et al.)^[5]. Enchytraeida are particularly important in temperate and boreal forests (reviewed in Huhta et al.)^[12] at abundances ranging from 10^3 m^{-2} to greater than 10^5 m^{-2} . Enchytraeida consume fine plant litter covered in fungi and bacteria. Collembola

are likely omnivorous, although they are often considered fungivorous, and mites feed on fungi, plant detritus, or are predaceous.^[5] Mesofauna are generally regarded as important consumers in litter transformation, especially important given that about 90% of terrestrial organic inputs are processed through detrital food webs.^[5]

Macrofauna are generally considered to be greater than 2 mm, up to a few centimeters, and some species of earthworms are meters long. Macrofauna are often cited as including ants (widely distributed, family Formicidae), earthworms (phylum Annelida; class Oligochaeta), termites (tropical and some temperate; Termitidae in the Order Isoptera), and macro-arthropods (phylum Arthropoda). Earthworm abundances range from less than 10 m⁻² to greater than 1000 m⁻² in temperate and tropical ecosystems.^[13] When present, ants and termites can be so abundant that researchers often do not enumerate abundances. Megafauna sometimes include earthworms and include vertebrates such as moles, gophers, and prairie dogs that can be important in many ecosystems.^[14] Macrofauna and megafauna can be ecosystem engineers. These organisms are too large to exist within the existing pores; in soil, they move soil particles, modify soil pore structure, and create structures such as aggregates.^[15,16] In general, ecosystem engineers modify their physical environment and resources for others.^[14–17] Further, they alter the environment distinctively from abiotic processes.^[14,17]

Much of the biological activity in soil occurs in a small volume relative to whole that poses logistical challenges in representative sampling. However, these “hotspots” or small volumes of soil where conditions are conducive for high activity can be placed in important functional domains.^[16] These domains, or spheres of influence regulating microbial activity, are often associated with the activity of larger organisms, such as plant roots and macrofauna, and include examples such as rhizosphere (zone of influence around a root), drilosphere (zone of influence around an earthworm burrow), and termitosphere (zone of influence under termite activity). Macrofauna create conditions conducive for high rates of activity by directly changing the “architecture” of soil at scales relevant to smaller organisms,^[18] and the accumulation of their direct and indirect activities changes properties and processes in fields. Macrofauna may reside in the surface litter layer (e.g., epigeic earthworms) and soil (e.g., endogeic earthworms) and/or move across soil–litter boundaries (e.g., anecic earthworms). Macrofauna are also involved in important symbiotic relationships in soil, such as protozoan gut communities in termites, fungus gardens of termites, and microbes in the external rumen of earthworms.^[7]

SOIL FAUNA—DIVERSITY AND BIOMASS

The enormous diversity of fauna means that not all groups have been identified and described. Fauna grouped by size

may be placed into broad taxonomic categories that use trophic status for differentiation. Beyond taxonomic classifications, soil fauna are often placed into functional groups. Characterizing communities based on functional groups or higher-level taxonomic groups may provide stronger statistical power^[19] than trying to identify organisms at finer resolution. However, species may feed across trophic levels,^[20] and thus taxonomic trophic groups may not be the most appropriate level of resolution for understanding the functional importance of soil fauna. Salamon et al.,^[21] e.g., found that different species of Collembola respond to increases in fungal compared to bacterial biomass. Although there is one food web aboveground, there are both bacterial-based and fungal-based food webs operating belowground.

Despite the difficulty of determining the appropriate level of resolution to understand soil faunal diversity, diversity is an important consideration for processes, ecosystem functions, and productivity. There is much that is not well known about the diversity of belowground communities. Regardless, it is safe to state that both local and global diversity of soil fauna are huge (reviewed in Coleman).^[3] Estimates for numbers of species can span hundreds in a few hundred square meters.^[22] However, relatively few taxa may dominate soil communities within a land use. In the Southeastern United States, rank–abundance relationships suggest that a few taxa dominated in four land uses, with most abundant taxa belonging to different types of beetles in cultivated and pasture sites, centipedes in hardwood forest, and millipedes in pine stand forest.^[23] A large portion of biomass in tundra, boreal forest, and temperate forest soil (high latitude ecosystems) was accounted for in enchytraeid, while earthworms dominated in tropical forests and temperate grassland soils.^[24] Mites, Collembola, and Enchytraeidae are abundant in acidic forests with thick organic layers.^[20] Despite the diversity of organisms, there are a few predominant organisms in ecosystems in many studies, i.e., many rare, and few cosmopolitan taxa.

A modern survey of soil animals (i.e., not including protozoa) in 11 locations around the globe using molecular analysis of 18S rDNA sequences resulted in a dominance of nematodes and micro-arthropods.^[25] Almost 96% of operational taxonomic units (surrogate for species defined by 99% sequence similarity) were detected in one location only, indicating high endemism and a lack of cosmopolitan species.^[25] Predominance of nematodes (grasslands) and micro-arthropods (forests) was related to pH with nematodes positively correlated and micro-arthropods negatively correlated with pH. In fact, several environmental variables were related to abundances of different groups; micro-arthropods predominate in forests with lower pH, root biomass, mean annual temperature, inorganic nitrogen (N), and higher carbon (C)/N ratio, litter, and moisture.^[25] Further, on a global scale, the soils from ecosystems with higher aboveground plant diversity

had lower belowground faunal diversity.^[25] Diversity matrices have been less informative than multivariate analyses.^[25,26] Thus, it may be that diversity per se is not the best approach to assessing soil faunal communities, whereas multivariate approaches relating abundance and functional structure to environmental variables may be informative.

While it is generally postulated that high biodiversity levels are positive for ecosystem health, stability, and function, there may be much functional redundancy among soil fauna. Impacts of fauna depend in part on characteristics of the ecosystem. In a field experiment investigating the contributions of soil fauna to litter decomposition, fauna of all size classes increased decomposition of litter in its home environment when the plant community was in late succession and contained more recalcitrant compounds.^[27] In early succession, when litter contained easily degraded compounds, the presence of an indigenous community was less important. Often there are changes in taxonomic groups within a system but little change in diversity. Thus, it may be that functional structure of soil fauna is more important than taxonomic structure in ecosystems. Responses to elevated carbon dioxide concentrations were related to trophic structure among soil biota in a meta-analysis.^[28] Abundances of detritivores increased, while there were no changes at higher trophic levels. So, change in the functional characteristics of species composition is important. In a microcosm study, Cole et al.^[29] found that an increasing density of micro-arthropods increased nitrate concentrations but increased richness decreased N mineralization and nitrification. The authors suggest that increased predation pressure with increased richness decreased the microbivore grazing of remaining micro-arthropods. Thus, they concluded it was not a change in richness per se that was important but a change that altered functional group status that was important for functioning.

The study by Cole et al.^[29] also brings to light another question in understanding the impacts of soil fauna on ecosystems. In addition to the extent of diversity, be it functional or taxonomic, among ecosystems, there is also a question of whether total abundance of soil fauna or diversity is more important. In another global meta-analysis of seven biomes, the authors estimated biomass of five major faunal groups. Despite the challenges in computing estimates for biomass that resulted in low to medium confidence for several of the estimates from several biomes, faunal biomass was found to be 40–80% of animal biomass in different biomes except desert.^[24] However, soil faunal biomass (with the exception of deserts at <0.02% of microbial biomass) averaged 2% (1.5–3.6%) of microbial biomass.^[24] Microbial biomass is 2–5% soil organic C, which is usually 1–5% of soil by volume. Despite composing a minor amount of the biomass in soil, soil fauna are major players in many important functions in terrestrial ecosystems.

SOIL FAUNA—ROLES IN ECOSYSTEM FUNCTIONS, STABILITY, AND SUSTAINABILITY

Lavelle et al.^[2] argue that invertebrates are a “resource that needs to be properly managed to enhance ecosystem services” since fauna mediate soil formation, nutrient cycling, primary production, and regulation services. Soil organisms alter soil aggregation through the production of particular compounds, the physical breakdown of organic materials, and the movement of soil particles. Changing soil pore and aggregate structure impacts water infiltration or run-off, and distribution and water storage within the soil profile. Water content affects the habitat, diffusion of gases and compounds, and activity and access of soil organisms, which has implications for the rates and extent of nutrient cycling. Grazing consumes microbes, changes community structure and activity, and releases nutrients. Soil fauna may consume microorganisms, other fauna, and plant detritus. Thus, soil fauna impact nutrient cycling directly through the incorporation and comminution of plant detrital materials and indirectly through the consumption, movement, selection, and activation of microbial communities. These processes impact sequestration of C (and other nutrients) in soil and gaseous releases that affect climate. The activities of soil fauna impact root production and enhanced nutrient cycling can increase aboveground plant biomass.^[30]

While the negative impacts of pests and pathogens are well studied and appreciated because of the economic damage they can cause, the positive impacts of the activities of soil fauna on plant growth and other ecosystem functions are often less appreciated and intentionally utilized. Ingham et al.^[30] demonstrated the positive influence of free-living nematodes on plant biomass by inoculating soil with bacteria and bacterial-feeding nematodes. About 30% N taken up by plants can come from mineralization as a result of grazing of fauna on soil microorganisms.^[31] The presence of earthworms has been shown to decrease plant parasitic nematode infection of rice by more than 80%, allowing plants to maintain biomass production similar to uninfected plants.^[32] In a modern study as part of the biodiversity experiment in Jena, Germany, use of the insecticide chlorpyrifos decreased the biomass of two functional plant groups (forbs and grass), indicating that negative effects from reductions in Collembola abundance, whose grazing of microorganisms contributes to nutrient cycling, outweighed benefits that might have been gained from reductions in herbivore populations.^[33]

Soil fauna are an integral component of belowground ecology that can alter aboveground ecology, but these fauna also respond to aboveground management. Fauna may respond to or emit plant signaling compounds (reviewed in Kardol and Wardle).^[34] Reestablishing mixed forests in coniferous stands changed *Collembola* species richness and diversity, highlighting how aboveground management can affect belowground ecology.^[35] In a study comparing conventional practices to organic wheat farming, organic

management promoted both aboveground and belowground interactions with positive effects on soil fauna compared to a conventional system using mineral fertilizers and herbicides.^[36] Organic management improved soil quality by enhancing generalist predators (top-down control); however, the system was also enhanced through the addition of resources (bottom-up control).^[36]

Subject to much debate is whether terrestrial systems and soil food webs are controlled by bottom-up or top-down forces. Invertebrate numbers and populations respond to or covary with soil properties, such as soil organic matter (OM), temperature, and electrical conductivity.^[26] However, there are conflicting data about which properties and the degree of importance of properties control soil fauna. Mites correlated positively with high soil OM.^[27] Soil pH is an important controlling variable. For example, enchytraeid are abundant in low pH soil.^[20] Soils are often considered to be controlled through bottom-up controls. However, research results do not necessarily clarify the debate. Although microorganisms are the primary agents of decomposition in soil, microbes do not always respond to resource inputs. This may be in part because fauna are consuming biomass, and thus preventing measurable increases in microbial production. In a temperate deciduous forest in Germany, while particular organisms responded to resource input, the structure of the food web was not clearly related to bottom-up control.^[21] Earthworms changed habitat for other organisms, reducing effects of resource inputs.^[21] Increased predation pressure reduces N mineralization, presumably through cascading effects that reduce grazing of microorganisms.^[29,36] These results indicate that top-down control of food web dynamics is also important in understanding how functions emerge in systems. Taken together, there is many gaps in the data for many groups of fauna to adequately assess what conditions lead to the greatest diversity. It is often hypothesized that with little disturbance and much competition, diversity will be limited, and under too much stress, diversity will be limited. However, the heterogeneity of soil, patchy community structures, and the activities of ecosystem engineers may all complicate these relationships and lead to high diversity of fauna in soil ecosystems.^[37]

In addition to diverse community composition, inherent spatial and temporal variability in soil properties and activities and interactions of organisms poses challenges in accurate assessments. In a study investigating the literature from 1940 to 1992, spatial variability of mesofauna and macrofauna was analyzed.^[19] While common species showed lower variation than rare species, the coefficient of variation for common species was often 100% with much of the variability attributed to random error.^[19] An important step in making cross-study comparisons is to standardize field collection methods to assess soil faunal communities.^[38] Abundances and diversity are often greatest at the soil surface. While organisms can be found at

depths below 30 cm, Callaham et al.^[23] discontinued sampling in four land uses in the Southeastern United States at the 15–30 cm depths for their faunal community assessments because fauna were found at the 0–15 cm soil depth. Abundances often need to be transformed to fit parametric statistical analyses. Distributions can be difficult to sample representatively because they are patchy at best within an ecosystem and variable in response to soil properties and to disturbances.^[39,40] Further, distributions and activity are temporally variable. For example, enchytraeid activity is closely related to rainfall.^[41]

Soil fauna are responsive to changes in their environment. This responsiveness, which occurs in part because the soil is their habitat, makes soil fauna useful indicator organisms of disturbance, recovery, and soil quality.^[42] Nematodes, e.g., have been used as indicators of ecosystem recovery and function, maturity, and the soil food web.^[43,44] Earthworms are utilized as indicators of pollution, including heavy metals. However, their diversity in niche separation may make it difficult to interpret responses. We remain limited not only by scale of resolution in sampling and identification but also by what we do not know about the biology and ecology of unidentified and even identified organisms. Barbercheck et al.^[26] suggested separating Collembola by habitat type for effectiveness of using Collembola as indicators. However, relying on one indicator or group of indicators to describe effects of disturbance among ecosystems is not advised.^[19,26]

In restoration ecology, it has been proposed that to restore ecosystems, one must integrate the belowground with the aboveground.^[34] Organisms are useful as indicators because they integrate cumulative effects of chemistry and physics. However, there are different types of disturbances that occur at different frequencies, intensities, durations, and extent of spatial coverage. Greater diversity of fauna was measured in the less disturbed hardwood forest site, followed by the pine stand, followed by pastures, and the least diversity occurred in cultivated soil in four land uses in Southeastern United States.^[23] Native earthworms were present in fields and forests but most commonly collected from forests.^[23] Introduced earthworms were most often collected from cultivated fields and pastures. Introduced species are well known for drastically altering characteristics and functions of ecosystems, especially if they become invasive. The introduction of non-native earthworms has resulted in change in the location and cycling of OM in forests, such that ecosystems are no longer sustainable (northern U.S. forests).^[45] It is frequently expected that introduced organisms will be more common in disturbed areas and native species more common in undisturbed locales.^[46] However, although reduced in abundance, native species are not always eliminated from an ecosystem as a result of a disturbance. To what extent native and introduced organisms co-occur and alter ecosystem function needs research.

CONCLUSION

Although there is growing recognition of their integral effects on soil functions, soil fauna have been largely ignored in the development of land management plans. There is an accumulating body of knowledge of the importance of these organisms in driving ecosystem functions. Beyond the small size of many organisms, soil fauna may not have received deserved levels of research attention because there is general agreement that most important drivers of nutrient cycling are microorganisms. However, soil fauna interact with microbes, influence community structure and dispersal, change the physical environment, and facilitate decomposition and nutrient cycling. Soil fauna may be the link that is necessary to connect microorganisms to fields and landscapes, i.e., the scale at which processes are observed.

REFERENCES

1. Decaëns, T.; Jiménez, J.J.; Gioia, C.; Measey, G.J.; Lavelle, P. The values of soil animals for conservation biology. *Eur. J. Soil Biol.* **2006**, *42*, S23–S38.
2. Lavelle, P.; Decaëns, T.; Aubert, M.; Barot, S.; Blouin, M.; Bureau, F.; Margerie, P.; Mora, P.; Rossi, J.P. Soil invertebrates and ecosystem services. *Eur. J. Soil Biol.* **2006**, *42*, S3–S15.
3. Coleman, D.C. From peds to paradoxes: Linkages between soil biota and their influences on ecological processes. *Soil Biol. Biochem.* **2008**, *40*, 271–289.
4. Amador, J.A.; Görres, J.H. Chapter 8: Fauna. In *Principles and Applications of Soil Microbiology*, 2nd Ed.; Sylvia, D.M., Fuhrmann, J.J., Hartel, P.G., Zuberer, D.A., Eds.; Pearson Education Inc.: Upper Saddle River, 2005; 181–200.
5. Coleman, D.C.; Crossley, D.A., Jr.; Hendrix, P.F. *Fundamentals of Soil Ecology*, 2nd Ed.; Elsevier: San Diego, 2004.
6. Coleman, D.C.; Wall, D.H. Fauna: The engine for microbial activity and transport. In *Soil Microbiology, Ecology, and Biochemistry*, 3rd Ed.; Paul, E.A., Ed.; Elsevier Academic Press: San Diego, 2007; 163–191.
7. Lavelle, P.; Spain, A.V. *Soil Ecology*; Kluwer Academic Publishers: Dordrecht, 2001.
8. Clarholm, M. Protozoan grazing of bacteria in soil – Impact and importance. *Microb. Ecol.* **1981**, *7*, 343–350.
9. Bouwman, L.A.; Zwart, K.B. The ecology of bacterivorous protozoans and nematodes in arable soil. *Agric. Ecosyst. Environ.* **1994**, *51*, 145–160.
10. Görres, J.H.; Savin, M.C.; Neher, D.A.; Weicht, T.R.; Amador, J.A. Grazing in a porous environment: 1. The effect of pore structure on C and N mineralization. *Plant Soil* **1999**, *212*, 75–83.
11. Lavelle, P. Faunal activities and soil processes: Adaptive strategies that determine ecosystem function. *Adv. Ecol. Res.* **1997**, *27*, 93–132.
12. Huhta, V.; Persson, T.; Setälä, H. Functional implications of soil fauna diversity in boreal forests. *Appl. Soil Ecol.* **1998**, *10*, 277–288.
13. Curry, J.P. Factors affecting earthworm abundance in soils. In *Earthworm Ecology*; Edwards, C.A., Ed.; CRC Press: Boca Raton, 2000; 37–64.
14. Reichman, O.J.; Seabloom, E.W. The role of pocket gophers as subterranean ecosystem engineers. *Trends Ecol. Evol.* **2002**, *17*, 44–49.
15. Lavelle, P.; Barois, I.; Martin, A.; Zaidi, Z.; Schaefer, R. Management of earthworm populations in agroecosystems: A possible way to maintain soil quality? In *Ecology of Arable Land*; Clarholm, M., Bergström, L., Eds.; Kluwer Academic Publishers: Dordrecht, 1989; 109–122.
16. Brown, G.G.; Barois, I.; Lavelle, P. Regulation of soil organic matter dynamics and microbial activity in the drilosphere and the role of interactions with other edaphic functional domains. *Eur. J. Soil Biol.* **2000**, *36*, 177–198.
17. Jones, C.G.; Lawton, J.H.; Shachak, M. Organisms as ecosystem engineers. *Oikos* **1994**, *69*, 373–386.
18. Görres, J.H.; Savin, M.C.; Amador, J.A. Soil micropore structure and carbon mineralization in burrows and casts of an anecic earthworm (*Lumbricus terrestris*). *Soil Biol. Biochem.* **2001**, *33*, 1881–1887.
19. Ekschmitt, K. Population assessments of soil fauna: General criteria for the planning of sampling schemes. *Appl. Soil Ecol.* **1998**, *9*, 439–445.
20. Scheu, S.; Falca, M. The soil food web of two beech forests (*Fagus sylvatica*) of contrasting humus type: Stable isotope analysis of a macro- and a mesofauna-dominated community. *Oecologia* **2000**, *123*, 285–286.
21. Salamon, J.A.; Alphei, J.; Ruf, A.; Schaefer, M.; Scheu, S.; Schnieder, K.; Sühlig, A.; Maraun, M. Transitory dynamic effects in the soil invertebrate community in a temperate deciduous forest: Effects of resource quality. *Soil Biol. Biochem.* **2006**, *38*, 209–221.
22. Hansen, R.A. Effects of habitat complexity and composition on a diverse litter microarthropod assemblage. *Ecology* **2000**, *81*, 1120–1132.
23. Callahan, M.A., Jr.; Richter, D.D., Jr.; Coleman, D.C.; Hofmockel, M. Long-term land-use effects on soil invertebrate communities in Southern Piedmont soils, USA. *Eur. J. Soil Biol.* **2006**, *42*, S150–S156.
24. Fierer, N.; Strickland, M.S.; Liptzin, D.; Bradford, M.A.; Cleveland, C.C. Global patterns in belowground communities. *Ecol. Lett.* **2009**, *12*, 1238–1249.
25. Wu, T.; Ayres, E.; Bardgett, R.D.; Wall, D.H.; Garey, J.R. Molecular study of worldwide distribution and diversity of soil animals. *Proc. Natl. Acad. Sci. U.S.A.* **2011**, *108*, 17720–17725. DOI: 10.1073/pnas.1103824108.
26. Barbercheck, M.E.; Neher, D.A.; Anas, O.; El-Allaf, S.M.; Weicht, T.R. Response of soil invertebrates to disturbance across three resource regions in North Carolina. *Environ. Monit. Assess.* **2009**, *152*, 283–298.
27. Milcu, A.; Manning, P. All size classes of soil fauna and litter quality control the acceleration of litter decay in its home environment. *Oikos* **2011**, *120*, 1366–1370.
28. Blankinship, J.C.; Niklaus, P.A.; Hungate, B.A. A meta-analysis of responses of soil biota to global change. *Oecologia* **2011**, *165*, 553–565.
29. Cole, L.; Dromph, K.M.; Boaglio, V.; Bardgett, R.D. Effect of density and species richness of soil mesofauna on nutrient mineralisation and plant growth. *Biol. Fertil. Soil* **2004**, *39*, 337–343.

30. Ingham, R.E.; Trofymow, J.A.; Ingham, E.R.; Coleman, D.C. Interactions of bacteria, fungi, and their nematode grazers: Effects on nutrient cycling and plant growth. *Ecol. Monogr.* **1985**, *55*, 119–140.
31. Verhoef, H.A.; Brussaard, L. Decomposition and nitrogen mineralization in natural and agro-ecosystems: The contribution of soil animals. *Biogeochemistry* **1990**, *11*, 175–211.
32. Blouin, M.; Zuily-Fodil, Y.; Pham-Thi, A.-T.; Laffray, D.; Reversat, G.; Pando, A.; Tondoh, J.; Lavelle, P. Below-ground organism activities affect plant aboveground phenotype, inducing plant tolerance to parasites. *Ecol. Lett.* **2005**, *8*, 202–208.
33. Eisenhauer, N.; Sabais, A.; Schonert, F.; Scheu, S. Soil arthropods beneficially rather than detrimentally impact plant performance in experimental grassland systems of different diversity. *Soil Biol. Biochem.* **2010**, *42*, 1418–1424.
34. Kardol, P.; Wardle, D.A. How understanding aboveground-belowground linkages can assist restoration ecology. *Trends Ecol. Evol.* **2010**, *25*, 670–679.
35. Chauvet, M.; Titsch, D.; Zaytsev, A.S.; Wolters, V. Changes in soil faunal assemblages during conversion from pure to mixed forest stands. *For. Ecol. Manag.* **2011**, *262*, 317–324.
36. Birkhofer, K.; Bezemer, T.M.; Bloem, J.; Bonkowski, M.; Christensen, S.; Dubois, D.; Ekelund, F.; Fließbach, A.; Gunst, L.; Hedlund, K.; Mäder, P.; Mikola, J.; Robin, C.; Setälä, H.; Tatin-Froux, F.; Van der Putten, W.H.; Scheu, S. Long-term organic farming fosters below and aboveground biota: Implications for soil quality, biological control and productivity. *Soil Biol. Biochem.* **2008**, *40*, 2297–2308.
37. Decaëns, T. Macroecological patterns in soil communities. *Global Ecol. Biogeograph.* **2010**, *19*, 287–302.
38. Römcke, J.; Sousa, J.P.; Schouten, T.; Riepert, F. Monitoring of soil organisms: A set of standardized field methods proposed by ISO. *Eur. J. Soil Biol.* **2006**, *42*, S61–S64.
39. Jiménez, J.-J.; Decaëns, T.; Amézquita, E.; Rao, I.; Thomas, R.J.; Lavelle, P. Short-range spatial variability of soil physico-chemical variables related to earthworm clustering in a neotropical gallery forest. *Soil Biol. Biochem.* **2011**, *43*, 1071–1080.
40. Rossi, J.P.; Huerta, E.; Fragoso, C.; Lavelle, P. Soil properties inside earthworm patches and gaps in a tropical grassland (la Mancha, Veracruz, Mexico). *Eur. J. Soil Biol.* **2006**, *42*, S284–S288.
41. Nieminen, J.K. Enchytraeid population dynamics: Resource limitation and size-dependent mortality. *Ecol. Monit.* **2009**, *220*, 1425–1430.
42. Wardle, D.A.; Yeates, G.W.; Watson, R.N.; Nicholson, K.S. The detritus food-web and the diversity of soil fauna as indicators of disturbance regimes in agro-ecosystems. *Plant Soil* **1995**, *170*, 35–43.
43. Bongers, T. The maturity index: An ecological measure of environmental disturbance based on nematode species composition. *Oecologia* **1990**, *83*, 14–19.
44. Neher, D.A. Role of nematodes in soil health and their use as indicators. *J. Nematol.* **2001**, *33*, 161–168.
45. Bohlen, P.J.; Groffman, P.M.; Fahey, T.J.; Fisk, M.C.; Suarez, E.; Pelletier, D.M.; Fahey, R.T. Ecosystem consequences of exotic earthworm invasion of north temperate forests. *Ecosystems* **2004**, *7*, 1–12.
46. Hendrix, P.F.; Baker, G.H.; Callaham, M.A.Jr.; Damoff, G.A.; Fragoso, C.; Gonzalez, G.; James, S.W.; Lachnicht, S.L.; Winsome, T.; Zou, X. Invasion of exotic earthworms into ecosystems inhabited by native earthworms. *Biol. Invasions* **2006**, *8*, 1287–1300.

Fauna and Microflora: Macrofauna

Judy Tisdall

Department of Agricultural Sciences, LaTrobe University, Bundoora,
Victoria, Australia

Abstract

This entry discusses earthworms, termites, and ants and their effect on soil structure. One or more of these groups generally dominate the soil fauna.

INTRODUCTION

Soil macrofauna (>2 mm) live all or part of their life below ground (Table 1). Through turnover (or bioturbation) of soil, the macrofauna contribute significantly to the structure, texture, and nutrient cycling of soil. Tillage destroys macrofauna habitat and food, hence the number of macrofauna is usually higher in undisturbed soils than in tilled soils.

EARTHWORMS

About 3000 species in 20 families of earthworms have been identified, and their distribution and number of species vary with climate and soil management. Earthworms affect soil structure by producing casts and burrows.

Casts

Earthworms daily ingest and mix in the gut about half their own mass of soil plus large volumes of organic matter and deposit the material as casts. Each year, earthworms deposit 0.3–27.0 kg m⁻² of casts on the surface of the soil, with more below the surface.^[1] Earthworms remould the soil in the gut^[2] so that fresh casts can be up to 15% more dispersive, i.e., less stable,^[3] than the surrounding soil and can easily be eroded during rain. As they age and dry, the casts often become more stable and stronger than the surrounding soil, due to age hardening, cohesive forces of water and cements, clay-bound carbohydrates, and microorganisms.^[4–6] By preventing the formation of crusts, stable casts on the surface of soil often increase infiltration and seedling emergence. Physical properties of casts may depend on the properties (content of clay, water or organic matter, or compaction) of the original soil.^[2,3,7] Some casts produced in very wet soil are so dense and hard that the movement of water and air, mineralization of carbon, and root growth is decreased.^[4,8]

Burrows

The main types of burrows are as follows: 1) semipermanent vertical burrows from the surface to at least 3 m depth; 2) almost horizontal and some vertical burrows to the surface; and 3) deep temporary burrows.^[1] Extensive networks of horizontal and vertical burrows develop in soils with several species of earthworms, but may vary seasonally, being filled with casts or destroyed by tillage or mammals.^[9]

Earthworms can produce burrows in soil at least as hard as 3 MPa, whereas roots are often limited in soil at 2 MPa.^[10] Many burrows are stable and may persist long after the earthworms have left, possibly because the burrows are lined with orientated clay or humic materials, carbonates and iron oxides.^[1] The burrows can be up to at least 10 mm wide depending on the size of the earthworm. The burrows affect the continuity and tortuosity of pores and often increase the macroporosity and infiltration in the soil.^[9] The matrix of the soil may be wetted laterally from open burrows rather than from the surface of the soil.^[11] Water that is absorbed into the matrix may later move back to burrows. Although water runs down the burrows during rainfall in temperate climates, the burrows always contain air and do not need to be linked for better aeration of the matrix.^[9]

Roots can grow along the earthworm burrows,^[12] possibly through hard, dry, waterlogged, or acidic soil to reach nutrients and water in soft, moist, drained, or neutral soil. However, water, nutrients, and pesticides may follow burrows below the root zone and pollute the groundwater.^[4]

TERMITES

Termites consist of several thousand species, with several castes in a colony.^[13] Termites turn over 0.02–470 t soil ha⁻¹ yr⁻¹, affecting over 0.5–20% of the soil surface.^[14] Termites can increase erosion by removing protective litter from the surface and bringing dispersive particles to the surface.

Table 1 Climatic zone, feeding habit and food of invertebrate macrofauna in soil.

Fauna		Climatic zone	Feeding habit (food)
Oligochaeta			
Earthworms		All but driest and coldest	Saprovores, geovores (organic litter, soil)
Macroarthropods			
Isopods		Various including desert	Saprovores, predators (organic litter, roots and leaves of seedlings)
Millipedes (Diplopoda)		Temperate, tropical, arid, semiarid	Saprovores (leaf litter, dung)
Centipedes (Chilopoda)		Forest to desert	Saprovores, predators
Scorpions		Mainly desert; also warm dry tropics, temperate	Predators, cannibals
Spiders (Araneae)		All but Arctic/Antarctic	Carnivores, predators (arthropods, small vertebrates)
Termites (Isoptera)		Mainly tropics, subtropics; also temperate and deserts	Saprovores (wood, plants, lichens, humus, fungi)
Ants (Hymenoptera)		Arctic to tropics	Saprovores, carnivores, predators (organic litter, seeds, secretions of plants and aphids, plants)
Wasps (Hymenoptera)		Temperate, tropics	Saprovores (nectar, sap)
Beetles (Coleoptera)		Temperate	Saprovores, predators, carnivores (dung, animal carcasses, seeds, roots)

Termites build a wide range of nests (mounds), in which they live and store food, and sometimes runways and subterranean galleries (i.e., the nests contain large pores). The nests and runways consist of soil, excreta, plant residues, and sometimes saliva, often with higher concentrations of clay, exchangeable cations and microbial activity, than the surrounding soil.^[14,15]

Mounds are usually up to at least 30 cm wide and 9 m high,^[13,14] with 200–1000 small mounds ha⁻¹, or 2–10 large mounds ha⁻¹; there are few mounds on sands, self-mulching clays, and shallow soils. Mounds last from 8 to 80 years, surviving while the mound is active. Once inactive, mounds are quickly eroded, especially when dug by mammals.^[13] The surface becomes covered with fine particles of subsoil, adding 25–1000 mm depth of soil every 1000 years. The runways are quickly eroded by rain and rebuilt just as quickly.

The few reports available suggest that termites have less overall effect on soil structure than do earthworms. Some termites slightly increase the bulk density of soil, probably because they repack the soil particles and select fine particles. Reports on the infiltration of water in termite galleries differ from that of the surrounding soil, although water-holding capacity is usually higher in the mounds.^[14]

ANTS

There are at least 15,000 species of ants, all of which are social insects living in colonies with up to 500,000 individuals, with a biomass of 0.1–0.8 g m⁻².^[14] Many ants burrow extensively, producing nests ranging from simple excavations on the surface or under stones to craters and domes.^[16] The mounds are either Type I, which are small and ephemeral, or Type II, which are large and dense and

can last for more than 100 years.^[17] Ants move between 3–11,360 kg of soil ha⁻¹ yr⁻¹ (agricultural land) and 21–8410 kg ha⁻¹ yr⁻¹ (natural vegetation) depending on the species.^[18] Ants mix soil and litter, or decompose organic litter or the surface of soil, deposit subsoil on the surface,^[19] and appear eventually to homogenize the texture to a depth of at least 260 mm.^[20]

The effect of ants on soil structure has not been greatly studied. However, compared with the surrounding soil, the mounds may have a lower plastic limit,^[19] be more stable and denser^[14] (at least at the top of the mound), or less dense,^[19,21] with higher macroporosity due to the galleries.^[22] The infiltration rate into an ant nest was over three times that of the surrounding soil,^[14] probably through the galleries. Compared with the surrounding soil, infiltration into ant mounds was increased as the liquid moved to a depth of 2 m through vertical and lateral galleries.^[22] However, ants expose bare soil around their burrows, which could impede infiltration and encourage soil erosion.

REFERENCES

1. Lee, K.E. *Earthworms—Their Ecology and Relationships with Soils and Land Use*; Academic Press: Sydney, 1985.
2. Schrader, H.; Zhang, H. Earthworm casting: Stabilization or destabilization of soil structure. *Soil Biol. Biochem.* **1997**, *29*, 469–475.
3. Hindell, R.P.; McKenzie, B.M.; Tisdall, J.M. Influence of drying and ageing on the stabilization of earthworm (*Lumbricidae*) casts. *Biol. Fertil. Soils* **1997**, *25*, 119–126.
4. Edwards, W.M.; Shipitalo, M.J. Consequences of earthworms in agricultural soils: Aggregation and porosity. In *Earthworm Ecology*; Edwards, C.A., Ed.; CRC Press: Boca Raton, 1998; 147–161.

5. Hindell, R.P.; McKenzie, B.M.; Tisdall, J.M.; Silvapulle, M.J. Relationships between casts of geophagous earthworms (*Lumbricidae*, *Oligochaeta*) and matric potential. *Biol. Fertil. Soils* **1994**, *18*, 119–126.
6. McKenzie, B.M.; Dexter, A.R. Physical properties of casts of the earthworm *Aporrectodea rosea*. *Biol. Fertil. Soils* **1987**, *5*, 152–157.
7. Buck, C.; Langmaack, M.; Schrader, S. Influence of mulch and soil compaction on earthworm cast properties. *Appl. Soil Ecol.* **2000**, *14*, 223–229.
8. Cockroft, B.; Olsson, K.A. Degradation of soil structure due to coalescence of aggregates in no-till, no-traffic beds in irrigated crops. *Aust. J. Soil Res.* **2000**, *38*, 61–70.
9. Kretzschmar, A. Earthworm interactions with soil organization. In *Earthworm Ecology*; Edwards, C.A., Ed.; CRC Press: Boca Raton, 1998; 163–176.
10. Dexter, A.R. Tunnelling in soil by earthworms. *Soil Biol. Biochem.* **1978**, *10*, 447–449.
11. Smettem, K.R.J. Analysis of water flow from cylindrical macropores. *Soil Sci. Soc. Am. J.* **1986**, *50*, 1139–1142.
12. Hirth, J.R. *Interactions between Plant Roots (Lolium Perenne L.) and Earthworm Burrows (Aporrectodea Species)*. Ph.D. Thesis, La Trobe University, 1996.
13. Goudie, A.S. The morphological role of termites and earthworms in the tropics. In *Biogeomorphology*; Viles, H., Ed.; Basil Blackwell: Oxford, 1988; 166–192.
14. Lobry de Bruyn, L.A.; Conacher, A.J. The role of termites and ants in soil modification: A review. *Aust. J. Soil Res.* **1990**, *28*, 55–93.
15. Holt, J. Microbial activity in the mounds of some Australian termites. *Appl. Soil Ecol.* **1998**, *9*, 183–187.
16. Mitchell, P. The influence of vegetation. In *Animals and Micro-Organisms on Soil Processes: Biogeomorphology*; Viles, H., Ed.; Basil Blackwell: Oxford, 1988; 43–82.
17. Paton, T.R.; Humphries, G.S.; Mitchell, P.B. *Soils: A New Global View*; Yale University Press: New Haven, 1995.
18. Lobry de Bruyn, L.A. Ants as bioindicators of soil function in rural environments. In *Invertebrate Biodiversity as Bioindicators of Sustainable Landscapes*; Paoletti, M.G., Ed.; Elsevier: Amsterdam, 1999; 425–441.
19. Hulugalle, N.R. Effects of ant hills on soil physical properties of a vertisol. *Pedobiologia* **1995**, *39*, 34–41.
20. Lobry de Bruyn, L.A.; Conacher, A.J. The bioturbation activity of ants in agricultural and naturally vegetated habitats in semi-arid environments. *Aust. J. Soil Res.* **1994**, *32*, 555–570.
21. Cowan, J.A.; Humphreys, G.S.; Mitchell, P.B.; Murphy, C.L. An assessment of pedoturbation by two species of mound-building ants. *Camponotus intrepidus* (Kirby) and *Iridomyrmex purpureus* (F. smith). *Aust. J. Soil Res.* **1985**, *22*, 95–107.
22. Nkem, J.N.; de Bruyn, L.A.L.; Grant, C.D.; Hulugalle, N.R. The impact of ant bioturbation and foraging activities on surrounding soil properties. *Pedobiologia* **2000**, *44*, 609–621.

Fertility Decline: Definitions and Assessment

Alfred E. Hartemink

International Soil Reference and Information Centre (ISRIC), Wageningen, the Netherlands

Abstract

Assessing soil fertility decline is difficult because most soil chemical properties either change very slowly or have large seasonal fluctuations; in both cases, it requires long-term research commitment. There are several other confounding factors that make the assessment of soil fertility decline complicated (e.g., spatial and temporal variation and soil analytical methods) and, for those reasons, other techniques have been used to estimate the rates and changes in soil fertility decline. The methods to assess soil fertility decline are described in this entry.

INTRODUCTION

In permanent agricultural systems, soil fertility is maintained through applications of manure, other organic materials, inorganic fertilizers, lime, the inclusion of legumes in the cropping systems, or a combination of these. In many parts of the world, the availability, use, and profitability of inorganic fertilizers have been low whereas there has been an intensification of land use and an expansion of crop cultivation onto marginal soils. As a result, soil fertility has declined and it is perceived to be widespread, particularly in sub-Saharan Africa.^[1–3] Soil fertility decline is considered as an important cause for low productivity of many soils.^[4,5] It has not received the same amount of research attention as soil erosion; possibly as soil fertility decline is less visible and less spectacular and more difficult to assess.

DEFINITIONS

Growing agricultural crops implies that nutrients (nitrogen, phosphorous, potassium, etc.) are removed from the soil through the agricultural produce (food, fiber, and wood) and crop residues. Nutrient removal may result in a decline of the soil fertility, if replenishment with inorganic or organic nutrient inputs is inadequate. Soil fertility is defined as “the quality of a soil that enables it to provide nutrients in adequate amounts and in proper balance for the growth of specified plants or crops.”^[6]

Although it omits the importance of soil physical and biological conditions for crop productivity, it is a useful simplification. A decline in soil fertility implies a decline in the quality of the soil, and soil fertility decline is defined as “the decline in chemical soil fertility, or a decrease in the levels of soil organic matter, pH, cation exchange capacity (CEC), and plant nutrients.”

Soil fertility decline thus includes:

1. Nutrient depletion or nutrient decline (larger removal than addition of nutrients).
2. Nutrient mining (large removal of nutrients and no inputs).
3. Acidification (decline in pH and/or an increase in exchangeable Al).
4. The loss of organic matter.
5. An increase in toxic elements (e.g., aluminum and magnesium).

ASSESSMENT OF SOIL FERTILITY DECLINE

Different data types are used to assess soil fertility changes: 1) expert knowledge; 2) monitoring of soil chemical properties over time or at different sites; and 3) nutrient balances. Some of these data can be relatively easily collected, whereas some other data require long-term commitment and involve high cost. Each data type has specific advantages and disadvantages, and the type of data collected is determined by the research plan, the financial conditions, and objectives of the study (Table 1).

Expert Knowledge

Farmers and other users of the land have expert knowledge about their soils; this is mostly empirical knowledge, which is not soil process or data oriented, but yield or management oriented.^[7] Yield decline as observed by a farmer could, however, be caused by a variety of factors including soil fertility decline, adverse weather conditions, invasion of weeds, soil physical deterioration, or a combination of factors. It is difficult to distinguish soil fertility decline from other factors,

Table 1 Data types in soil fertility decline studies and their advantages and disadvantages.

Data type	Short description	Advantages	Disadvantages
Expert knowledge	Combination of field observations with general knowledge	Easy to obtain, rapid assessment, useful for small-scale studies	Subjective, not quantitative
Measured values Type I Chronosequential	Monitoring soil properties over time	Accurate, hard data, using existing data	Slow, expensive, contamination of monitoring sites, spatial and temporal variability, sample storage
Measured values Type II Biosequential	Comparing soil properties under different land use	Easy to obtain, rapid, relatively hard	Soils at sampling sites may differ, unknown land use history of sites, spatial and temporal variability
Nutrient balances and budgets	Combination of existing data with pedotransfer functions or models	Using existing data, fairly rapid, indicative, appealing outcome	Not very hard, require computer power

Source: From Lal.^[4]

and farmers' knowledge on soil fertility decline must be substantiated by other types of data, e.g., crop yield, weather conditions or pests, and disease information. Farmers' knowledge can be instrumental in selection of sampling sites or for additional information and various examples in the literature exist where such information was used to investigate long-term changes in soil fertility.^[8,9] Another form of expert knowledge is that of soil scientists and agronomists who can make some estimation on the soil fertility status and its changes based on vegetation or crop growth. As with farmers' knowledge, such information is important but not very quantitative.

MEASURED CHANGE IN SOIL CHEMICAL PROPERTIES

Two different approaches have been used to monitor soil chemical properties. First, soil dynamics can be monitored over time at the same site, which is called chronosequential sampling^[10] or Type I data.^[11] Type I data show changes in a soil chemical property under a particular type of land use over time. Usually the original level is taken as the reference level to investigate the trend in such changes. It is most useful if trends are also followed under other land use systems, e.g., under cultivation, secondary regrowth, and natural forest over the same period. Type I data have been used for quantifying soil contamination by comparing soil samples collected before the intensive industrialization period with samples taken from the same locations.^[12] Type I data are also useful to assess the sustainability of land management practices in the tropics,^[13] but limited data sets exist as they require long-term research commitment and detailed recordings of soil management and crop husbandry practices.^[14]

In the second approach, soils under adjacent different land use systems are sampled at the same time and

compared. This is called biosequential sampling,^[10] Type II data,^[11] "sampling from paired sites" in the soil science literature from Australia,^[15,16] the "space-for-time" method,^[17] and the "inferential method."^[18] The underlying assumption is that the soils of the cultivated and uncultivated land are the same soil series, but that differences in soil properties can be attributed to the differences in land use. Obviously, this is not always the case and the uncultivated land may have been of inferior quality and therefore not planted. Also, spatial variability may be confounded with changes over time when, e.g., tree crops of different age are sampled at the same time, and soil properties are often confounded with genetic improvement and silvicultural practices.^[11] Other confounding factors are differences in clay content, soil depth, or unknown history of land use, and these largely affect the usefulness of Type II data for the assessment of soil fertility decline. In ecology, Type II data studies have often proved to be misleading as functional parameters like nutrient availability and plant-animal interactions have been conspicuously underrepresented.^[17] When carefully taken, however, biosequential soil samples can provide useful information, and this sampling strategy has been followed in a considerable number of studies.^[14]

Monitoring of soil chemical properties gives information on how soils respond to agricultural activities and whether soil fertility decline takes place. Spatial and temporal variation requires a cautious selection of sampling sites, an appropriate number of replicated observations, and a careful interpretation of the results. Provided these are met, it may be difficult to extrapolate the information or to derive maps of the patterns observed in single pedons. Coupling the soil information to soil maps in a geographical information system (GIS) provides opportunities to map soil fertility decline in different areas. A large amount of data of sufficient quality is needed before such maps can be derived, and there are few examples where measured soil fertility decline was

coupled to a GIS.^[19] More studies exist in which the nutrient balance has been used to map soil fertility decline.

Nutrient Budgets and Balances

A third way of studying soil fertility decline embraces a semiquantitative approach using partial nutrient balances and budgets. Such studies operate at a much coarser (smaller) scale, namely, national or supranational scale, and available soil data are combined with pedotransfer functions (regression analysis between some soil or other parameters) into a GIS to estimate the decline in soil fertility at a given location.^[20] This is essentially a mechanistic modeling approach in which expert knowledge is also important. It is generally perceived that such studies are not to replace soil-monitoring efforts but must be seen as the best possible way of getting the most out of available data.^[14] The outcome of such studies provides qualitative, comparative, and spatial information on the decline in soil fertility.

Nutrient balances provide a convenient and biologically meaningful context within which to organize what is known about a system's biogeochemical cycles, put nutrient pools and fluxes into perspective, and can lead to considerable insight into processes that regulate nutrient cycling. Nutrient balances help to guide system management decisions and direct the course of future research.^[21] The use of nutrient balances has been encouraged through a systems approach to both improve food crop production and maintain the soil resource base.^[22]

CONCLUSION

Evaluating soil fertility decline can be done with different types of data. There are data from measured soil chemical properties, and such data can be from the same plot that is permanently cultivated (Type I data) or from plots under different land use (Type II data). Soil fertility decline can also be assessed in a more semiquantitative way using nutrient balances. Each data type has its merits and drawbacks, and data are either quickly collected and indicative for what is going on or the collection is more tedious which usually implies that the data are harder and more meaningful. Boundary conditions need to be properly set and the study must indicate whether soil fertility decline is assessed for a pedon, watershed, region, country, etc. At the watershed level, soil fertility may decline in one pedon but it may increase in a lower pedon, which illustrates the need for the delineation of spatial boundaries. Soil fertility decline studies must also have temporal boundaries and in general long-term observations yield better results.

An important aspect in soil fertility decline studies is the spatial and temporal variation in soil properties. Soil

spatial variation has been sufficiently tackled by research and various methods exist to quantify the variation. Temporal variation is a more difficult issue, and fewer studies are available. As with spatial variation, it requires a sufficient amount of subsamples and samples before rigid conclusions can be drawn. Soil fertility decline studies are largely dependent on soil chemical analysis, which include soil sampling, soil analysis, and interpretation of the results. Errors are possible in all three steps, although most errors are generally being made during soil sampling. The choice of the analytical technique in relation to the soil property or soil type is another potential source of errors.

REFERENCES

1. Pieri, C. *Fertilité des Terres de Savanes*; Ministère de la Coopération et CIRAD-IRAT: Paris, 1989.
2. Henao, J.; Baanante, C. *Estimating Rates of Nutrient Depletion in Soils of Agricultural Lands of Africa*; IFDC: Muscle Shoals, 1999.
3. Smaling, E.M.A. *An Agro-ecological Framework for Integrated Nutrient Management with Special Reference to Kenya*; Agricultural University: Wageningen, 1993; 250 pp.
4. Lal, R. Land degradation and its impact on food and other resources. In *Food and Natural Resources*; Pimentel, D., Hall, C.W., Eds.; Academic Press: San Diego, 1989; 85–140.
5. Sanchez, P.A. Soil fertility and hunger in Africa. *Science* **2002**, *295* (5562), 2019–2020.
6. SSSA. *Glossary of Soil Science Terms*; SSSA: Madison, 1997.
7. Bouma, J. Soil behavior under field conditions—Differences in perception and their effects on research. *Geoderma* **1993**, *60* (1–4), 1–14.
8. Sillitoe, P.; Shiel, R.S. Soil fertility under shifting and semi-continuous cultivation in the Southern Highlands of Papua New Guinea. *Soil Use Manage.* **1999**, *15* (1), 49–55.
9. Peters, J.B. Gambian soil fertility trends, 1991–1998. *Commun. Soil Sci. Plant Anal.* **2000**, *31* (11–14), 2201–2210.
10. Tan, K.H. *Soil Sampling, Preparation, and Analysis*; Marcel Dekker: New York, 1996; 408 pp.
11. Sanchez, P.A.; Palm, C.A.; Davey, C.B.; Szott, L.T.; Russel, C.E. Tree crops as soil improvers in the humid tropics? In *Attributes of Trees as Crop Plants*; Cannell, M.G.R., Jackson, J.E., Eds.; Institute of Terrestrial Ecology: Huntingdon, 1985; 79–124.
12. Lapenis, A.G.; Torn, M.S.; Harden, J.W.; Hollocker, K.; Babikov, B.V.; Timofeev, A.I.; Hornberger, M.I.; Nattis, R. Scientists unearth clues to soil contamination by comparing old and new soil samples. *EOS* **2000**, *81*, 55–57.
13. Greenland, D.J. Soil science and sustainable land management. In *Soil Science and Sustainable Land Management in the Tropics*; Syers, J.K., Rimmer, D.L., Eds.; CAB International: Wallingford, 1994; 1–15.

14. Hartemink, A.E. *Soil Fertility Decline in the Tropics—With Case Studies on Plantations*; ISRIC-CABI Publishing: Wallingford, 2003.
15. Bramley, R.G.V.; Ellis, N.; Nable, R.O.; Garside, A.L. Changes in soil chemical properties under long-term sugar cane monoculture and their possible role in sugar yield decline. *Aust. J. Soil Res.* **1996**, *34* (6), 967–984.
16. Garside, A.L.; Bramley, R.G.V.; Brislow, K.L.; Holt, J.A.; Magarey, R.G.; Nable, R.O.; Pankhurst, C.E.; Skjemstad, J.O. Comparisons between paired old and new land sites for sugarcane growth and yield and soil chemical, physical, and biological properties. *Proc. Aust. Soc. Sugarcane Technol.* **1997**, *19*, 60–66.
17. Pickett, S.T.A. Long-term studies: Past experience and recommendations for the future. In *Long-term Ecological Research—An International Perspective*. SCOPE 47; Risser, P.G., Ed.; John Wiley: Chichester, 1991; 71–88.
18. Ekanade, O. The nutrient status of soils under peasant cocoa farms of varying ages in southwestern Nigeria. *Biol. Agric. Hortic.* **1988**, *5*, 155–167.
19. Stoorvogel, J.J. Optimizing land use distribution to minimize nutrient depletion: A case study for the Atlantic zone of Costa Rica. *Geoderma* **1993**, *60*, 277–292.
20. Stoorvogel, J.J.; Smaling, E.M.A. *Assessment of Soil Nutrient Decline in Sub-Saharan Africa, 1983–2000*; Winand Staring Centre-DLO: Wageningen, 1990.
21. Robertson, G.P. Regional nitrogen budgets: Approaches and problems. *Plant Soil* **1982**, *67*, 73–79.
22. Johnston, A.E.; Syers, J.K. Working group reports and conference summary. In *Nutrient Management for Sustainable Crop Production in Asia*; Johnston, A.E., Syers, J.K., Eds.; CAB International: Wallingford, 1998; 375–382.

Fertility: Environmentally Compatible Management

Bal Ram Singh

Department of Environmental Sciences, Norwegian University of Life Sciences, Aas, Norway

Abstract

Soil fertility integrates the basic principles of soil biology, chemistry, and physics to develop the practices needed to manage nutrients in a profitable and environmentally sound manner. Maintenance and management of soil fertility is central to the development of sustainable food production systems. Sustainability is dependent to a large degree on recycling the inputs into a production system, thereby increasing efficiency of output per unit of resource input. The essential plant nutrients may be divided into macronutrients (nitrogen, phosphorus, sulfur, potassium, magnesium, and calcium), micronutrients (iron, manganese, zinc, copper, boron, molybdenum, chlorine, and nickel), and beneficial elements (sodium, silicon, and cobalt). The modern soil fertility management in addition to involving soil productivity must also include the protection of the environment, and, therefore, issues related to soil quality, water quality, air quality, heavy metals, food quality, and safety are discussed. Nutrient mining and loss of organic matter lead to deterioration of soil quality and accelerated eutrophication of surface waters and nitrate content of drinking water are the main concerns for water quality. The nitrous oxide produced during nitrification and denitrification contributes to global warming and stratospheric ozone depletion and thus poses threats to air quality. Anthropogenic inputs of metals through fertilizers, amendments, sewage sludge, pesticides, mining, and industry lead to soil contamination, creating concern for food safety. A number of management practices to reduce metal transfer to food chain are tested with varying success.

INTRODUCTION

Soil Fertility in the Past

In writings, dating back to 2500 B.C., the fertility of land is mentioned. Herodotus, the Greek historian, traveling through Mesopotamia some 2000 years later reported the phenomenal yields obtained by the inhabitants of this land. Later Theophrastus (372–287 B.C.) recommended the abundant manuring of thin soils but suggested that rich soils be manured sparingly. During the 17th and 18th centuries, agricultural writings reflected that plants consisted of one substance, and most of the workers were searching for this principle of vegetation during this period. It was not until the latter half of 19th and the beginning of the 20th century that some progress was made to understand the subject of plant nutrition and crop fertilization. It was Justus von Liebig (1803–1873), a German chemist, who effectively deposed the humus myth and eventually developed the law of minimum. The law says that the growth of plants is limited by the plant–nutrient element present in the smallest quantity, all others being present in adequate amounts. These developments led to a rapid increase in chemical fertilizer's use.

Soil Fertility in Modern Times

With advances in our knowledge with regard to various processes affecting the nutrient dynamics in soils, the

definition of soil fertility is also refined. Soil fertility integrates the basic principles of soil biology, chemistry, and physics to develop the practices needed to manage nutrients in a profitable and environmentally sound manner. The focus of soil fertility is to manage nutrient status in soils to create optimum conditions for plant growth. Two fundamental principles underlay the study of soil fertility. First is the recognition that optimum nutrient status alone will not ensure soil productivity. Other factors, such as soil moisture and temperature, soil physical conditions, soil acidity and salinity, and biotic stress, can reduce the productivity of even more fertile soils. Second is the realization that modern soil fertility practice must stress soil productivity and environmental protection.^[1] Taking the second realization in perspective, it is imperative that soil fertility in relation to agricultural sustainability and environmental protection will be the focus of this entry.

SOIL FERTILITY AND AGRICULTURAL SUSTAINABILITY

A new dimension to soil fertility in relation to agricultural production has been added. It is the concept of “sustainability,” which has been defined and interpreted differently by different workers. Okigbo^[2] after analyzing the various definitions of sustainable agriculture by different workers defined “a sustainable agricultural production system as one that maintains an acceptable and increasing level of

productivity that satisfies prevailing needs and is continuously adapted to meet the future needs for increasing the carrying capacity of the resource base and other worthwhile human needs.”^[2]

The sustainability of a production system is location specific and is determined to a greater degree by an interaction among several production factors, namely, physiochemical (soil, climate, radiation, etc.), biological (crop species, weeds and pests, etc.), management, and socioeconomic elements. Maintenance and management of soil fertility is central to the development of sustainable food production systems. Sustainability is dependent to a large degree on recycling the inputs into a production system, thereby increasing the efficiency of output per unit of resource input. Soil fertility management is concerned with the essential plant nutrients, their amounts, availability to crop plants, chemical reactions in soil, loss mechanisms, processes making them unavailable or less available to crop plants, and ways and means of replenishing them in these soils.^[3]

ESSENTIAL NUTRIENTS

Because soil fertility involves management of nutrients required for plant production, it is important to describe briefly the elements, which are considered essential for plant growth. The essential nutrients required by higher plant are exclusively of inorganic nature, and Arnon and Stout^[4] proposed the term essential nutrient (element). The essential element must meet three criteria to be considered essential: 1) a given plant must be unable to complete its life cycle without the presence of the mineral element in question; 2) the function of the element cannot be replaced by another element; and 3) the element is directly involved in the nutrition of the plant—for example, as a component of an essential plant constituent such as enzyme—or must be required for a distinct metabolic step such as an enzyme reaction.

Some elements, which either compensate for the toxic effect of other elements or simply replace mineral elements in some of their less specific functions, such as maintenance of osmotic pressure, can be described as beneficial elements. Out of a large number of elements found in plants, 14 mineral elements are recognized as essential, whereas the requirement of chlorine (Cl) and nickel (Ni) is restricted to a limited number of plant species. The plant nutrients may be divided into: 1) macronutrients [nitrogen (N), phosphorus (P), sulfur (S), potassium (K), magnesium (Mg), and calcium (Ca)]; 2) micronutrients [iron (Fe), manganese (Mn), zinc (Zn), copper (Cu), boron (B), molybdenum (Mo), Cl, and Ni]; and 3) beneficial elements (sodium (Na), silicon (Si), and cobalt (Co)]. The last three elements have been found essential only for some plant species, for example, Na for plants with C4 photosynthetic pathway, Si for rice, and Co for fixation of atmospheric N by rhizobia

and blue green algae, but they have not been included in the list of essential nutrients.^[3]

MANAGEMENT OF SOIL FERTILITY AND THE ENVIRONMENT

As pointed out by Sims^[1] that modern soil fertility in addition to involving soil productivity must also include protection of the environment, and, thus, the environmental aspects related to soil fertility are described under this section. Long-term use of organic and inorganic fertilizers to agricultural soils has led to slow build up of nutrient reserves, especially under temperate conditions. The same nutrients, which are considered essential for plant growth and crop production, if lost from the system, can create a concern for the environment. Increased use of fertilizers for meeting the world food demand and recycling of on-farm organic wastes (manures and slurries) and off-farm organic wastes (municipal and industrial sludges) on agricultural lands in the past decades has resulted in some undesirable effects on the environment in intensively cultivated areas. This has created greater awareness on environmental issues not only among scientists and policy makers but also among general public. Although there are a number of issues related to soil fertility and the environment, emphasis is placed on four main concerns of environmental protection, namely, soil quality, water quality, air quality, heavy metals, and food concerns.

Soil Quality

The export of agricultural produce takes with it large amounts of nutrients (especially N, P, K, and S), and if these nutrients are not replenished, the soil organic matter (SOM) and the fertility of the soil will decline, and the soil will degrade. A number of such examples from the subsistence farming of the tropics are available, where “nutrient mining” of soils has been responsible for decline in soil fertility and soil quality leading to loss of productivity.^[5] The rapid loss of SOM of the topsoil after clearing of the natural vegetation or under continuous cultivation is a common phenomenon in Africa. In Zambia, it was found that the SOM content dropped from 59 to 32 Mg ha⁻¹ after 17 years of cultivation of a newly cleared land, which represented an average loss of 1.6 Mg ha⁻¹ yr⁻¹.^[6]

On the other hand, long-term studies carried out in temperate regions have shown that organic matter of soil can be maintained or raised modestly by proper fertilization, especially by organic manures and crop residue management.^[7] Any management practice that results in an increase in the crop residues returning to soil will have a positive influence on SOM,^[7] which, in turn, will affect soil aggregation and related properties such as soil structure, erodibility, workability, and water infiltration. Growing of legumes is

an effective way for promoting good soil aggregation and for reversing degradative trends.

Water Quality

The major concerns with regard to water quality are accelerated eutrophication of surface waters and nitrate (NO_3) content of drinking water. Eutrophication, the rapid growth and decay of aquatic vegetation, is most often limited by P and sometimes by N concentration in water. A NO_3 concentration of 10 mg L^{-1} in drinking water is considered safe, but higher concentration can cause methaemoglobinaemia (reduced carrying capacity of blood for oxygen) in children.

Both overfertilization and underfertilization can lead to N losses. Losses occur either as runoff to surface water or as leaching to underground waters. Under normal conditions, runoff losses in watersheds are low (e.g., 10 mg L^{-1}). Much of the N transported in runoff is particulate N associated with the sediments, and it can range from 0 to 7 kg N ha^{-1} .^[8] These losses can be reduced by incorporation of fertilizer and providing a good vegetation cover. Much of N losses occur through leaching, but there is a large variation in the quantities of N lost in leaching depending on the amount of N fertilizer applied, soil type, crop grown, and climatic conditions. Letey et al.^[9] reviewed NO_3 concentration in tile effluents from 55 sites of California; the concentration ranged from 1 mg L^{-1} to 196 mg L^{-1} , with only one-quarter of the sites averaging $<10 \text{ mg L}^{-1}$. High nitrate levels in drinking water have also been observed where animal or poultry manures are applied. In Sussex county, Delaware, it was found that 32% of the water wells had high ($>10 \text{ mg L}^{-1}$) nitrate levels.^[10,11]

From surface water viewpoint, P is the element of primary concern, because it is considered limiting for eutrophication. Excessive application of organic and inorganic fertilizers can result in building of P near the soil surface, which is subjected to soil erosion leading to P losses to surface waters. In aquatic ecosystems of southwestern Australia, an excess of nutrients has caused serious eutrophication, which was manifested by excessive growth and accumulation of green and blue green algae.^[12] P is generally the limiting nutrient for algae growth, and phosphatic fertilizers applied to nutrient deficient, leaching, and sandy soils were the main source of P in these ecosystems of Australia.

Air Quality

The primary goal of good nutrient management and especially N should be to maximize N uptake at critical growth stages and to minimize transformation processes, which lead to the formation of "greenhouse gases." The main N-containing greenhouse gas is nitrous oxide (N_2O). The N_2O produced during nitrification and denitrification

contributes to global warming and stratospheric ozone depletion. Annual losses of N as N_2O from fertilized field soil can be as high as 40 kg h^{-1} compared to $<2 \text{ kg h}^{-1}$ from unfertilized soils.^[13] Based on a summary of 104 field experiments conducted in temperate climate, it was reported that almost all experiments showed evidence of fertilizer derived N_2O emissions.^[14] Legumes, such as fertilizers, have been shown to contribute to increased N_2O emissions^[7] caused by enhanced denitrification due to the addition of easily decomposable organic matter. On the other hand, it was found that gaseous losses via denitrification and N_2O emissions from irrigated corn in Shelton, Nebraska, United States, represented only 1–5% of the N applied as fertilizer, and, thus, it was concluded that under good irrigation and good N management practices gaseous N losses from denitrification and N_2O emissions do not pose additional threat to air quality.^[15]

Under tropical climate, N loss to air takes place both through denitrification and volatilization of ammonia (NH_3). High losses of NH_3 in rice fields have been attributed to high concentration of ammonium-N following urea application, high pH, high temperature, and high wind speed. NH_3 loss of applied urea-N ranged from 7% to 54% in Philippines and the loss depended on time and the method of urea-N application. For example, the incorporation of urea before transplanting with no standing water reduced urea-N loss as NH_3 from 42% to 11% at IRRI, Philippines.^[16] Direct field measurement of denitrification has been made difficult by the lack of suitable methodology. Buresh and Austin^[17] used a floating chamber technique to directly collect $(\text{N}_2 + \text{N}_2\text{O})^{-15}\text{N}$ from labeled urea applied to flooded soil and found that the total N loss as unaccounted for ^{15}N by the ^{15}N balance study ranged from 26% to 46% of the applied N.

HEAVY METALS AND FOOD CONCERNS

Contamination Sources

The major sources of anthropogenic inputs of metals to soils are either primary sources as fertilizers and amendments (e.g., lime), sewage sludge, and pesticides or secondary source, where metals are added to the soil as a result of nearby activities such as mining or industry or Aerosol deposition from more distant sources. However, there are also rural areas, where the content of metal in soils is many fold higher due to natural processes and geological formation of soils. A good example is alum shale soils (developed from sulfide-bearing rocks formed in anoxic environment) in southeastern Norway.^[18]

It is well documented that the use of commercial P fertilizers is one of the main sources of metals and especially of cadmium (Cd) addition to agricultural soils.^[19–21]

Besides fertilizers, certain animal waste products may also contain high amount of metals. Pig manure contained

more than twice the amount of Zn (158 mg/kg) and Cu (15 mg/kg) than cow manure because of supplementation of these metals in their diets.^[22] Disposal of biosolids (e.g., sewage sludge) is a potential source to contaminate soils. In many developed countries, soil metal loading through sewage sludge is regulated through “maximum permitted concentration (MPC).”

Soil Contamination with Metals and Guidelines for Soil Quality

That long-term use of fertilizers can lead to increased content of metals in soils is well documented,^[21,23,24] and especially Cd enrichment of soils after long-term use of P fertilizers is reported in many countries.^[20,23,25,26] Many studies based on long-term fertility experiments in United States and Western Europe have shown a net accumulation of Cd in soils.^[20,25,26] The balance calculations based on 70 years data indicated that Cd accumulation in the soil was $<1 \text{ g Cd ha}^{-1} \text{ yr}^{-1}$ but increasing the doses of commercial fertilizers or farm yard manure would likely result in increased accumulation of this element.^[20] In some countries, guidelines to protect soil quality have been developed. A best example of such values is found in The Netherlands. They have developed “reference or target values,” which represent soils with negligible risks and “intervention values,” which represent soils with the maximum possible risk levels (Table 1). In Australia and New Zealand, the environmental soil quality guidelines to protect plant production, animal health, and food quality have been prepared. These soil quality guidelines for Cd, Cu, Pb, Ni, and Zn are 3, 60, 300, 60, and 200 mg/kg, respectively.^[21]

Phytouptake and Phytotoxicity

Soil conditions related directly or indirectly to soil fertility such as soil pH, organic matter and clay content, metal content, moisture content, and soil aeration are determining factors for metal uptake. For these reasons, some workers have found increased plant uptake of Cd added through commercial fertilizers,^[19,24,26,27] whereas others did not

Table 1 Dutch reference values and interventional values (mg kg^{-1}) in soils based on ecotoxicity and human health.

Metal	Reference value for standard soil ^a	Ecotoxicity intervention value	Human health intervention value
Cd	0.8	12	680
Cu	36	190	16,000
Pb	85	290	300
Ni	35	210	6,600
Zn	140	720	56,000

^aStandard soil with 25% clay and 10% organic matter.

Source: From McLaughlin, Hamon, et al.^[21] ©2000.

Table 2 Cd concentration (mg kg^{-1}) in different crop species as affected by varying levels of Cd in P fertilizer applied.

Cd applied in fertilizer ($\mu\text{g kg}^{-1}$)	Oat grain	Ryegrass	Carrot root	Spinach
0.0	0.06	0.21	0.14	—
2.7	0.07	0.21	0.15	1.52
12.5	0.07	0.24	0.18	1.80

Source: From He & Singh.^[23] ©1993.

find any increase.^[25] The application of P fertilizer containing increasing amount of Cd resulted in increased concentration of Cd in different crops, but the magnitude of Cd uptake differed considerably among crops. For example, spinach contained almost 25 times more Cd than oat grain (Table 2).^[27] Furthermore, increasing disposal of sludges and other organic wastes to agricultural lands has also resulted in increased uptake of metal in plants.^[28,29]

The uptake of Zn, Cu, and Cd in field pea increased significantly in sludge or pig manure treated plots, but liming of these plots reduced the concentration of all metals significantly (Table 3).^[29]

Metals in Foods and Their Regulations

Among the metals, Cd in food stuff is considered a greater risk and hence regulatory limits for food, the so-called “MPC,” have been worked out in several countries (e.g., The Netherlands, Germany, Denmark, Sweden, Australia, and New Zealand). These values do not pose health risks, but they are taken as target concentrations for initiating corrective measures to minimize Cd entry into the food chain. Cd concentration in leafy vegetable is much higher than seed grains and fruits (Table 3). For metals such as lead and mercury, their entry through roots is rather restricted, resulting in their low concentration in food crops.

Table 3 Zn, Cu, and Cd concentration in field pea (*Pisum arvense*) in sludge and pig manure treated soils with and without liming.

Treatment	Zn (mg/kg)	Cu (mg/kg)	Cd ($\mu\text{g/kg}$)
Unlimed			
Control	73.3cd	9.2c	63.2d
Sludge	77.9bc	7.8d	81.8dc
Pig manure	75.3cd	7.4bd	60.9d
Limed			
Control	49.5e	7.0be	37.3e
Sludge	67.5	7.1b	36.7e
Pig manure	61.1f	7.0be	33.5e

Note: Mean values within the same column followed by the same letter are not significantly different at $P = 0.05$.

Source: From Krebs, Gupta, et al.^[29] ©1998.

Management Practices for Minimizing Metal Transfer to Food Crops

Elevated levels of metals in soils and food crops have created a need to search for management practices, which can reduce metal transfer from soils to food crops. A number of practices have been tested with varying rates of success. However, the mechanisms involved are far from fully understood. One practice, which has been more often used, is liming, but the results obtained have shown both decrease and increase in metal uptake and especially that of Cd.^[27,30] Liming raises soil pH and thus reduces metal concentration in soil, but it also raises the Ca concentration in soil solution. Increased concentration of Ca further complicates the effect of liming. The addition of Zn is shown to reduce Cd uptake by crops and especially under Zn deficiency conditions,^[31] but the mechanism is not known. In metal-rich alum shale soils, increasing rates of cow and pig manures decreased the exchangeable fractions of Cd and Ni.^[22] Other practices, such as crop rotation showed marked effect, where wheat grown after lupins had higher concentration of Cd than wheat grown after a cereal crop.^[32]

CONCLUSION

Soil fertility management is essential for sustainable development of food production systems through recycling of the inputs, thereby increasing resource input efficiency. All essential elements, although very important for plant growth and productivity, can lead to soil contamination and deterioration of water and air quality if applied in excessive and imbalanced ratio. Increased use of fertilizers for meeting the world food demand and recycling of on-farm (manures and slurries) and off-farm organic wastes (municipal and industrial sludge) on agricultural lands in the past decades has resulted in some undesirable effects on the environment in intensively cultivated areas.

Many agronomic management practices have been tested to reduce the undesirable consequences of poor soil fertility management with varying degree of success. However, the mechanisms involved are far from fully understood. Research is needed to explore the ways to reduce nutrient losses to the environment while sustaining soil fertility and improving the food quality with respect to nutritive values and avoidance of harmful elements (toxic metals such as Cd).

REFERENCES

1. Sims, T. Soil fertility evaluation. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 1999; D113–D153.
2. Okigbo, B.N. *Development of Sustainable Agricultural Production Systems in Africa*; International Institute of Tropical Agriculture: Ibadan, 1991; 66 pp.
3. Prasad, R.; Power, J.F. *Soil Fertility Management for Sustainable Agriculture*; Lewis Publishers: Boca Raton, 1995; 1–4.
4. Arnon, D.I.; Stout, P.R. The essentiality of certain elements in minute quantity for plant with special reference to copper. *Plant Physiol.* **1939**, *14* (2), 371–375.
5. Stoorvogel, J.J.; Smaling, E.M.A.; Janssen, B.H. Calculating soil nutrient balances in Africa at different scales: I. Supranational scale. *Fert. Res.* **1993**, *35*, 227–235.
6. Singh, B.R.; Goma, H.C. Long-term soil fertility management in eastern Africa. In *Soil Management-Experimental Basis for Sustainability and Environmental Quality*; Lal, R., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 1995; 347–382.
7. Campbell, C.A.; Myers, R.J.K.; Curti, D. Managing nitrogen for sustainable crop production. *Fert. Res.* **1995**, *42*, 277–296.
8. Smith, S.J.; Schepers, J.S.; Porter, L.K. Assessing and managing agricultural nitrogen losses to the environment. *Adv. Agron.* **1990**, *14*, 1–43.
9. Letey, J.; Blair, J.W.; Devitt, D.; Lund, L.J.; Nash, P. Nitrate-nitrogen in effluent from agricultural tile drains in California. *Hilgardia* **1977**, *45*, 289–319.
10. Ritter, W.E.; Chirnside, A.E.M. *Ground Water Quality in Selected Areas of Kent and Sussex Counties, Delaware*; Project Completion Report; Delaware Agriculture Experiment Station.
11. Moore, P.A. Jr.; Daniel, T.C.; Sharpley, A.N.; Wood, C.W. Poultry manure management: Environmentally sound options. *J. Soil Water Conserv.* **1995**, *50* (3), 321–327.
12. Hodgkin, E.P.; Hamilton, B.H. Fertilizers and eutrophication in Southwestern Australia: Setting the scene. *Fert. Res.* **1993**, *36*, 95–103.
13. Sahrawat, K.L.; Keeny, D.R. Nitrous oxide emission from soils. *Adv. Soil Sci.* **1986**, *4*, 103–148.
14. Eichner, M.J. Nitrous oxide emissions from fertilized soils: Summary of available data. *J. Environ. Qual.* **1990**, *19*, 272–280.
15. Qian, J.H.; Doran, J.W.; Weier, K.L.; Mosier, A.R.; Peterson, T.A.; Power, J.F. Soil denitrification and nitrous oxide losses under corn irrigated with high-nitrate ground water. *J. Environ. Qual.* **1997**, *26*, 348–360.
16. De Datta, S.K.; Buresh, R.J. Integrated nitrogen management in irrigated rice. *Adv. Agron.* **1989**, *10*, 143–169.
17. Buresh, R.J.; Austin, E.R. Direct measurement of dinitrogen and nitrous oxide flux in flooded rice fields. *Soil Sci. Soc. Am. J.* **1988**, *52*, 681–688.
18. Mehlum, H.K.; Arnesen, A.K.M.; Singh, B.R. Extractability and plant uptake of heavy metals in alum shale soils. *Commun. Soil Sci. Plant Anal.* **1998**, *29*, 1183–1198.
19. Singh, B.R. Trace element availability to plants in agricultural soils, with special emphasis on fertilizer inputs. *Environ. Rev.* **1994**, *2*, 133–146.
20. Jeng, A.J.; Singh, B.R. Cadmium status of soils and plants from a long-term fertility experiment in southeast Norway. *Plant Soil* **1995**, *175*, 67–74.
21. McLaughlin, M.J.; Hamon, R.E.; McLaren, R.G.; Speir, T.W.; Roger, S.L. Review: A bioavailability-based rationale

- for controlling metal and metalloid contamination of agricultural land in Australia and New Zealand. *Aust. J. Soil Res.* **2000**, *38*, 1037–1086.
22. Narwal, R.P.; Singh, B.R. Effect of organic materials on partitioning, extractability and plant uptake of metals in an alum shale soil. *Water Air Soil Poll.* **1998**, *103*, 405–421.
 23. He, Q.B.; Singh, B.R. Plant availability of Cd in soils: I. Extractability of Cd in newly- and long-term cultivated soils. *Acta. Agr. Scand.* **1993**, *43*, 134–141.
 24. McLaughlin, M.; Tiller, K.; Naidu, R.; Stevens, D. Review: The behaviour and environmental impact of contaminants in fertilizers. *Aust. J. Soil Res.* **1996**, *34*, 1–54.
 25. Mordvedt, J.J. Cadmium levels in soils and plants from some long-term soil fertility experiments in the USA. *J. Environ. Qual.* **1987**, *16*, 137–142.
 26. Jones, K.C.; Johnston, A.E. Cadmium in cereal grain and herbage from long-term experimental plots at Rothamstead. U.K. *Environ. Pollut.* **1989**, *57*, 199–216.
 27. He, Q.B.; Singh, B.R. Crop uptake of Cd from phosphate fertilizers: I yield and Cd content. *Water Air Soil Poll.* **1994**, *74*, 251–265.
 28. Brown, S.L.; Chaney, R.L.; Angle, J.S.; Ryan, J.A. The phytoavailability of cadmium to lettuce in long-term biosolids-amended soils. *J. Environ. Qual.* **1998**, *27*, 1071–1078.
 29. Krebs, R.; Gupta, S.K.; Furrer, G.; Schulin, R. Solubility and plant uptake of metals with and without liming of sludge-amended soils. *J. Environ. Qual.* **1998**, *27*, 18–23.
 30. Maier, N.A.; McLaughlin, M.J.; Heap, M.; Butt, M.; Smart, M.K.; William, C.M.J. Effect of current-season application of calcitic lime on soil pH, yield and Cd concentration in potato (*Solanum tuberosum* L.) tubers. *Nutr. Cycl. Agroecosys.* **1996**, *47*, 29–40.
 31. Oliver, D.P.; Hamon, R.; Tiller, K.G.; Wilhelm, N.S.; Merry, R.H.; Cozens, G.D. The effect of zinc fertilization on cadmium concentration in wheat grain. *J. Environ. Qual.* **1994**, *23*, 705–711.
 32. Oliver, D.P.; Schultz, J.E.; Tiller, K.G.; Merry, R.H. The effect of crop rotations and tillage practices on Cd concentration in wheat grain. *Aust. J. Agric. Res.* **1993**, *44*, 1221–1234.

Fertility: Evaluation Systems

Ram C. Dalal

Department of Science, Information Technology, and Innovation, Queensland Government, Dutton Park, Queensland, Australia

A. Subba Rao

Indian Institute of Soil Science, Bhopal, India

Abstract

The soil fertility evaluation is aimed at providing an adequate supply of essential plant nutrients to ensure optimum plant productivity while maximizing economic benefits and minimizing environmental degradation. The essential plant nutrients include nitrogen, phosphorus, potassium, sulfur, calcium, magnesium, sodium, copper, zinc, manganese, iron, molybdenum, boron, and chloride. The soil fertility evaluation systems include biological assessment, plant nutrient analysis, soil testing (chemical, biological, and physical), and spectral sensing for site-specific soil fertility.

BIOLOGICAL ASSESSMENT

The significance of the supply of an essential plant nutrient for crop production was contained in the Liebig's law of the minimum,^[1] which states that by the deficiency or absence of one necessary constituent, all others being present, the soil is rendered barren for all those crops to the life of which that one constituent is indispensable. When crop response to the quantity of an essential nutrient present in soil or added to it increases in direct proportion, it can be expressed by the following linear equation:

$$y = a + mx \quad (1)$$

where y is the crop yield, x is the quantity of nutrient added, a is the intercept, and m is the slope. Eq. 1 applies to low soil fertility conditions and for nutrient uptake. In such cases, the value of a , the intercept, is equivalent to the plant-available quantity of nutrient in soil.

However, as the quantity of a nutrient increases, the unit increment in crop yield becomes smaller, and eventually, the crop yield reaches a maximum value. The crop response curve is then represented by the Mitscherlich equation:^[2]

$$y = A(1 - e^{-Cx_t})$$

or

$$y = A(1 - e^{-C(x+b)}) \quad (2)$$

where x_t or $x + b$ is the total nutrient, b is the plant-available quantity of nutrient in soil (and seed, considered if present in significant quantity), A is the maximum yield attainable when $x + b$ is present in sufficient quantity, and C is the efficiency factor.

When quantity of a nutrient present in, as well as added to, soil is in excess of crop requirements, the nutrient becomes toxic to plants or other factors become yield limiting, and then the crop yield response curve obeys polynomial equations such as the quadratic equation as follows:

$$y = a_1 + b_1x + c_1x^2 \quad (3)$$

where a_1 , b_1 , and c_1 are constants. The maximum yield is given by the value of $-b_1/2c_1$. When y is expressed as the nutrient uptake, a_1 estimates the quantity of available nutrient in soil.

Because maximum yield, A in Eq. 2 and $-b_1/2c_1$ in Eq. 3, varies across crops, soils, seasons, and years, it is usually preferable to express plant productivity as relative yield in a field experiment. The relative yield at a given nutrient supply is $100 \times y_n/A$ from Eq. 2 and $100 \times y_n/(-b_1/2c_1)$ from Eq. 3, where y_n is the crop yield at a given nutrient supply in soil.

This biological assessment of soil fertility evaluation is shown in Fig. 1.

Although this biological assessment in the field is the most appropriate system for soil fertility evaluation, it is expensive and time-consuming and subject to the limitations and constraints imposed on crop growth because of other factors such as pests, diseases, water stress, and spatial variability. It is primarily used for benchmarking other soil fertility evaluation systems such as plant analysis and tissue testing, soil testing, and remote sensing.

PLANT ANALYSIS

When nutrient deficiency is severe enough, plants exhibit visual symptoms,^[3] and this provides a rapid but qualitative

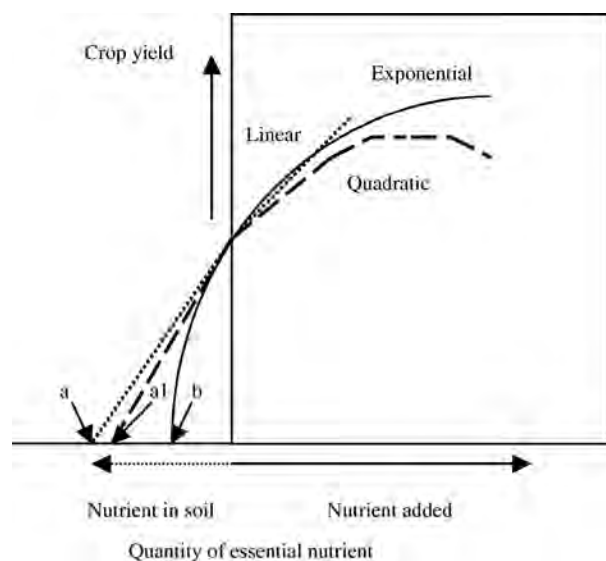


Fig. 1 Crop yield response curve with quantities of a nutrient added in soil (and seed, if contributing significant quantity); the quantity in soil is estimated from the values of a in Eq. 1, b in Eq. 2, and a_1 in Eq. 3.

assessment of soil fertility. Conversely, nutrient toxicity in the presence of excess nutrients and heavy metals in soil can be readily assessed from visual symptoms on plants in the field.^[3] However, it is often too late to take corrective measures to avert the decline in crop yields. Other limitations include disease, undefined symptoms, environmental factors, and multinutritional disorders.

Because critical nutrient concentration range ($\approx 90\%$ maximum yield) in plants at which crop yield is affected is relatively narrow, plant analysis for critical nutrient concentration has been used to evaluate soil fertility and recommend nutrient applications to correct nutrient deficiency.^[3]

The further application of plant analysis for critical nutrient concentration is in tissue testing of a crop growing in the field. This involves a rapid analysis of a nutrient, a form of the nutrient such as nitrate for nitrogen (N), enzyme activity such as nitrate reductase activity for nitrate, ribonuclease activity for zinc, and greenness index for chlorophyll associated with critical N concentration in the leaf. It usually provides only a qualitative or semiquantitative assessment of soil fertility because the critical concentration differs among crop cultivars, crop types, incidence of disease, time of sampling, nature of nutrient interactions, water supply, temperature, dry matter yield level, physiological maturity of the leaf or plant part sampled, and other undefined factors.

The balance of nutrients within a plant is often more important than the concentration of any individual nutrient. The diagnosis and recommendation integrated system (DRIS) focuses on nutrient balance as an alternative to the critical nutrient concentration range approach to plant

analysis interpretation. One advantage of the DRIS is that nutrient ratios in plants tend to be less variable throughout the growing season than individual nutrient concentrations, especially for mobile nutrients such as N, phosphorus (P), and sulfur, but there is no universal benchmark nutrient ratio for plants.^[4]

SOIL TESTING

Soil testing provides a rapid chemical and/or biological analysis to assess the quantity of plant-available nutrients in soil and, hence, for the evaluation of soil fertility.^[4-6] This is usually done by analyzing an extract of the soil obtained by adding water, chemical extractants, ion-exchange resin and membranes, and electro-ultrafiltration. Short-term incubation, especially for assessing N mineralization, fungal growth such as *Aspergillus niger*, or even short-term plant growth in the greenhouse, is employed as biological methods for soil testing. However, biological methods are time-consuming and are rarely used in soil testing except for short-term anaerobic N mineralization.^[4]

Unfortunately, most values of plant-available nutrients are expressed on a soil's mass basis rather than on the volume basis; the latter requires soil's bulk density and depth of sampling. Furthermore, for most plant-available nutrients, soil from only the top 10-cm or 30-cm depth is sampled although crop roots access nutrients and water from much deeper layers, often deeper than 100 cm in many soils.

SOIL TEST-CROP RESPONSE CORRELATION

Soil test values are used for evaluating soil fertility of a group of soils based on correlation with crop response such as dry matter yield, crop yields, nutrient yield, or nutrient concentration. It is essential that the soil tests be calibrated against crop response to applied nutrients in field experiments conducted over a wide range of soils, crops, and climate. A properly calibrated soil test correctly identifies the degree of deficiency or sufficiency of a nutrient. This is shown in Fig. 2.

Most soil tests are based on theoretical consideration of some aspects of plant nutrient availability (i.e., intensity, quantity, buffer power, and diffusion of nutrients in soil). Isotopic dilution techniques are used to label the plant-available nutrient in the soil, such as P in greenhouse or field, and then to trace its availability through crops, thus avoiding the artifact effects of soil extractants.^[7] In some soil test-crop response correlation procedures, soils are grouped according to clay content, cation exchange capacity, or soil pH, and separate calibrations are established for each group of soils. In fact, the fertility capability soil classification system groups soils according to surface soil texture, subsoil texture, and 15 other soil characteristics.^[8]

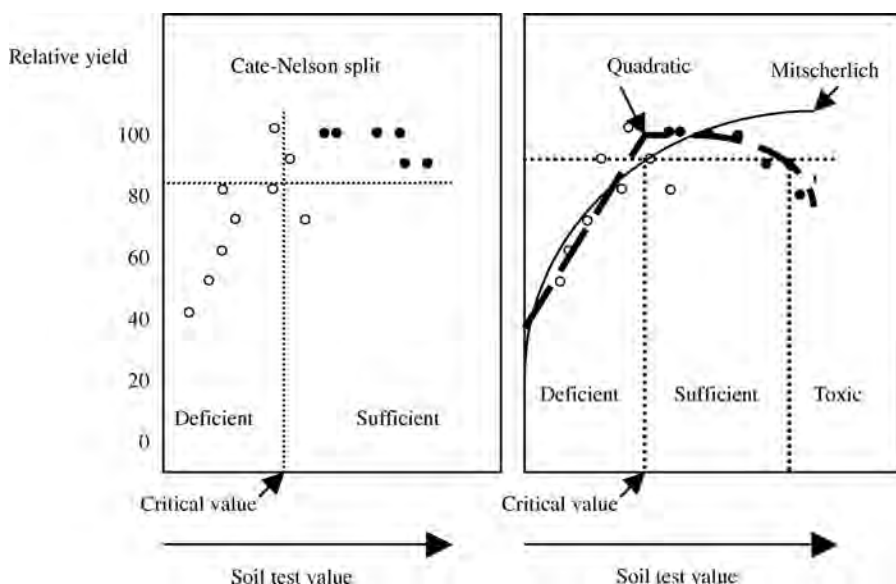


Fig. 2 A diagrammatic representation of soil test-crop response correlation to evaluate soil fertility of a group of soils. In the deficient range, soil fertility status is further classified as very low (<50% relative yield), low (50–75% relative yield), medium (75–90% relative yield), and high (>90% relative yield). Toxic range is of special interest for environmental considerations of soil fertility.

SPECTRAL SENSING AND SITE-SPECIFIC SOIL FERTILITY EVALUATION

Soil fertility variability in conventional soil testing, which aims at obtaining an average value for a field or experimental site, is not evaluated to make site-specific management decisions. The two outstanding benefits of site-specific soil fertility evaluations are 1) increased economic efficiency of nutrient application, i.e., maximum profits; and 2) consequent reduced environmental nutrient pollution. Spectral sensing techniques (physical soil testing) such as static and portable near- and mid-infrared, gamma ray, and satellites using hyperspectral sensors^[9] are increasingly being used to map soil fertility for multiple nutrients and crop yields at <1 m² scale to >1 km² scale, while using precision technologies for site-specific nutrient applications.

ENVIRONMENTAL IMPACT

Usually, maximum economic yield is obtained with soil nutrient supply just enough for achieving about 90–95% of relative yield when nutrients are fully taken up by plants. In crops when quality of the product attracts a premium such as protein concentration of wheat >11.5% protein, N supply at the higher level of sufficiency and luxury range is economically preferred. Yet high soil nitrate-N fertility may lead to excess of nitrate-N in soil and, consequently, to denitrification (greenhouse effect) and leaching. Excess nitrate-N in surface water bodies causes algal blooms and in drinking water causes methemoglobinemia in infants. Also, adverse environmental and health impacts occur when soil nutrient fertility is in toxic range (Fig. 2), especially for heavy metals, because of human consumption of product or on ecosystems.

PERSPECTIVES

Soil fertility evaluation has evolved in the last two centuries, from a major concern for enhancing crop production to that of maintaining environmental integrity. With increasing population, crop production will be the main objective of managing the soil fertility. However, with the advances in multielement extractants and analysis and remote sensing and precision technologies for site-specific soil fertility evaluation, it is possible to meet the twin goals of economic optimum yields and minimal environmental pollution. By managing soil fertility this way, sustainable use of land, water, and nutrients is also ensured, resulting in enhanced soil and water quality for the existing and future generations. In spite of the advances in sensor and analytical technologies, however, the challenge lies ahead in developing soil nutrient tests that closely mimic nutrient uptake by a crop.

CONCLUSION

Significant progress has been made in soil fertility evaluation systems in last decades because of valuable contributions made by ecologists and geographers, besides the continuing interest from agronomists and soil scientists. Consequently, a broad spectrum of technologies is being applied to soil fertility evaluation for sustainable natural resource management, food and fiber production, and ecologically sustainable practices for the benefit of both rural and urban communities. Further refinements in soil fertility evaluation systems would come from the integrative use of both classical and spectral technologies and spatial and computational capacities for their appropriate applications to ecosystems and landscapes.

REFERENCES

1. Russell, E.W. *Soil Conditions and Plant Growth*, 10th Ed.; Longman: London, 1973; 49–50.
2. Mitscherlich, E.A. Das Gesetz des Minimums und das Gesetz des Abnehmenden Bodenertrages. *Landwirtschaftliche Jahrbücher* **1909**, 38, 537–552.
3. Robinson, D.J.; Reuter, J.B. *Plant Analysis: An Interpretation Manual*, 2nd Ed.; CSIRO Publishing: Melbourne, 1997; 1–572.
4. Black, C.A. *Soil Fertility Evaluation and Control*; Lewis Publishers: Boca Raton, 1993; 1–746.
5. Havlin, J.L.; Beaton, J.D.; Tisdale, S.L.; Nelson, W.L. *Soil Fertility and Fertilizers*, 6th Ed.; Prentice Hall: Upper Saddle River, 1999; 1–499.
6. Peverill, K.I.; Sparrow, L.A.; Reuter, D.J., Eds. *Soil Analysis: An Interpretation Manual*; CSIRO Publishing: Melbourne, 1999; 1–369.
7. Dalal, R.C.; Hallsworth, E.G. Evaluation of the parameters of soil phosphorus availability factors in predicting yield response and phosphorus uptake. *Soil Sci. Soc. Am. J.* **1976**, 40, 541–546.
8. Sanchez, C.A.; Couto, W.; Buol, S.W. The fertility capability soil classification system: Interpretation, application and modification. *Geoderma* **1982**, 27, 282–309.
9. Sudduth, K.A.; Hummel, J.W.; Birrel, S.J. Sensors for site-specific management. In *The State of Site-Specific Management for Agriculture*; Pierce, F.J., Sadler, E.J., Eds.; ASA/CSSA/SSSA: Madison, 1997; 183–210.

Fertility: Low Input and Traditional Maintenance Methods

Miguel Altieri

*Department of Environmental Science, Policy and Management,
University of California—Berkeley, Berkeley, California, U.S.A.*

Abstract

Throughout the world, more and more farmers are learning that the combination of reduced tillage, cover crops, and better rotations can have a dramatic effect on their soil and the health of crops. They found that, by combining practices, they were able to reduce pest damage, improve soil tilth, vastly reduce runoff and erosion, increase soil organic matter, and produce better crop growth.

INTRODUCTION

There are thousands of farmers throughout the world (mostly organic farmers and resource-poor farmers) who, due to economic constraints or environmental reasons, cannot or do not want to use chemical fertilizers to enhance or maintain soil fertility.^[1] These farmers who can neither afford nor rely on a regular supply of inorganic fertilizers must find alternative sources of nutrients. These sources are often cheaper, more efficient than inorganic compounds and their use focuses on recycling of nutrients rather than on supplying nutrients on a regular basis.^[2] The techniques that these farmers use to maintain soil fertility and to improve biological, chemical, and physical soil properties tend to enhance organic matter content and increase the efficiency of nutrient use by closing the nutrient cycles (by returning exported nutrients to cropland) and by minimizing nutrient loss from the agroecosystem.^[3]

PRINCIPLES OF LOW-INPUT SOIL FERTILITY MANAGEMENT

Although many farmers have developed alternative fertilizing techniques through trial and error, research showed that there are several agroecological principles underlying such low-input methods of maintaining soil fertility.^[4,5] Such principles can be summarized as follows.

Securing Arable Soil Conditions for Plant Growth

In order to grow healthy and productive plants, farmers must create and/or maintain the following soil conditions:^[6] 1) timely availability of water, air, and nutrients in balanced and buffered quantities; 2) soil structure that enhances root growth, exchange of gaseous elements, water availability, and storage capacity; 3) soil temperature that

enhances soil biology and plant growth; 4) absence of toxic elements such as pesticide residues; 5) increased addition of organic residues by using varied sources of organic materials.

Optimizing Nutrient Availability and Cycling

Nutrient availability depends greatly on the general soil condition, soil life, and organic matter content. However, deliberate attention must also be given to providing the nutrients required for plant growth. There is a constant flow of nutrients through the farm. Some of the nutrients are lost or exported, e.g., by export of products, erosion, leaching, and volatilization. These nutrients have to be replaced. Nutrient losses can be limited by:^[7] 1) recycling organic wastes, e.g., manure, night soil, and crop residues by returning them to the field, either directly or treated (composted, fermented, etc.); 2) handling organic fertilizers in such a way that nutrients are not leached by excessive rain or volatilized by high temperature or solar radiation; 3) reducing runoff and soil erosion, which removes nutrients and organic matter; 4) reducing burning of vegetation when farming is intensified, as this leads to losses of organic matter; 5) reducing leaching by maintaining a high humus content in the soil and soil cover and intercropping plant species with different rooting depths; and 6) pumping up partly leached nutrients from deeper soil layers and bringing them back to the topsoil by using litter from trees or other deep-rooting plants.

Clearly, nutrient export from the farm to the market cannot be completely avoided nor can all nutrient losses resulting from erosion and leaching. However, soil cover and efficient organic management can minimize losses to acceptable levels. In fact, nutrients can be captured on the farm by:^[8] 1) fixing nitrogen by microorganisms living in symbiosis with leguminous trees, shrubs, or cover crops; 2) harvesting nutrients by capturing wind or water sediments

from outside the farm; this can be done with special soil conservation practices; and 3) integrating livestock into the farming system.

TECHNIQUES FOR SUSTAINABLE SOIL FERTILITY MANAGEMENT

The above principles are universal in nature but when applied by different farmers under a diversity of environmental and socioeconomic conditions, principles take various technical forms which can be grouped as follows.^[9]

Soil Conservation

The most effective way to improve the productivity of small, resource-poor farmers is to conserve and utilize the resources that they already possess. Soil conservation thus aims to maintain soil fertility by preserving soil structure, organic matter, and nutrients. These practices include techniques that decrease rates of leaching, erosion, and organic matter destruction. Also important are methods that capture and conserve water, especially in low-rainfall areas, for water and soil fertility act synergistically on crop production.^[3]

Conservation techniques are often advantageous because they require little outlay of cash and they can usually be undertaken during the fallow season, when farmers have less work. As a result, they do not put the farmer in debt risk and they do not interrupt or interfere with traditional work schedules. These techniques include the use of windbreaks, terraces, and deep-rooted plants. Windbreaks are usually composed of tree hedgerows planted at regular intervals in a crop field. They reduce erosion by reducing wind velocity. Terraces can be used to prevent soil erosion on cultivated steep lands. Although large-scale terracing programs may be expensive and require much labor, there are several techniques for low-resources terrace building, in which the terraces actually build themselves.^[10] Deep-rooted plants, such as perennial shrubs and trees, can be used to recycle nutrients that have leached to deeper layers in the subsoil, beyond the reach of the roots of most crop species. This strategy is particularly important in certain forms of agroforestry such as alley farming.^[11]

Nutrient Recycling

The nutrients that are removed from agricultural fields as crops eventually become wastes and thus can be reused in the field. Some wastes, such as manure, can be used directly on cropland, but most organic materials must first be “prepared” before they are used as a fertilizer. The variety of materials that have been used to maintain soil fertility include animal wastes, bones,

blood, seaweed, seashells, peat, ashes, sawdust, leaves, sewage, sludge, and so on.^[2]

In temperate regions, one of the most common recycling practices is the use of “mixed” farming, where livestock or chickens are the chief product of the farm. On these farms, grains and hay are grown and fed to the livestock. Their manure is used to fertilize the soil. Legumes are grown in rotation with grain crops in order to resupply the soil with nitrogen. Opportunities are endless, but the main approaches include the following.

Cover crops and green manures

Some crops are grown solely for the purpose of enhancing soil fertility. These crops are not harvested, but rather they are plowed into the soil, while they are still growing, which is why they are referred to as green manures. Typically, green manures or cover crops are grown in rotation or as “improved fallow” and are mixed into the soil before seeding the subsequent crop, allowing enough time for the decomposition of residues and the mineralization of nutrients. Green manures serve many different purposes in soil fertility. They can improve the soil physical structure by adding organic matter. They can be used to prevent erosion, as when they are used as a fallow season cover crop. They can improve soil nutrient status when they are used as “nutrient pumps” to recycle nutrients that have leached to deeper horizons.^[12] When a legume is used as a green manure crop, large amounts of soil nitrogen may be added.

One of the most remarkable cover crops is the velvet-bean (*Mucuna pruriens*). This has been widely promoted as part of the work of World Neighbors in Central America, though its effectiveness is attested by its spontaneous spread from village to village without outside intervention. Establishing *Mucuna* requires very little investment in labor. It grows rapidly, is palatable to animals and people, fixes large amounts of nitrogen, and can produce as much as 60 ton/ha of organic matter. It can grow on most soils, and its spreading habit suppresses weed growth. Incorporating such green manures into cropping systems can increase yields substantially. Honduran farmers are able to harvest some 2.5–3.2 ton/ha of maize when grown after velvetbean. This compares with an average for the country of just 0.6 ton/ha.^[13]

Azolla and anabaena

Blue-green algae are another important source of nitrogen, the most widely exploited being the alga *Anabaena azollae* which fixes atmospheric nitrogen while living in cavities in the leaves of a small fern, *Azolla*, which grows in the water of rice fields in both tropical and temperate regions. *Azolla* quickly covers the water surface in the ricefield but does not interfere with the normal cultivation of the rice crop.

Very high nitrogen production is possible following *Azolla* inoculation in rice fields. In the Philippines, 57 tons of fresh weight *Azolla* can be harvested after 100 days yielding more than 120 kg/ha of nitrogen. Over the whole year, *Azolla* can fix more than 400 kg N/ha, a rate in excess of most tropical and subtropical legumes. This nitrogen is only available to the rice crop after *Azolla* has decomposed and so exploitation consists of incorporating the ferns into the soil while wet as a green manure or removing them for drying and then reapplying them to the ricefields. The results of at least 1500 studies in China, Philippines, Vietnam, India, Thailand, and the United States have shown that when *Azolla* is grown in paddy fields, rice yields increase by on average 700 kg/ha, with a range of 400–1500 kg/ha.^[11]

Mulches

Soil fertility can often be enhanced simply by covering the soil's surface with a layer of organic material (mainly straw), a process that is known as mulching. In the humid tropics, where cultivation of any sort is often deleterious to soil fertility and crop productivity, mulching is a highly effective method to maintain soil health. Mulches are particularly useful in preventing soil damage because they reduce the destructive effect of raindrop impact on soil aggregation and prevent soil erosion. Mulches also improve soil water status by increasing infiltration rates. Mulches also stimulate the growth of soil biota.^[9]

Agroforestry and intercropping

There is a huge diversity of agroforestry systems throughout the world, in which the incorporation of bushes and trees result in many benefits. Trees with nitrogen-fixing capacity have beneficial effects on plants growing beneath them. In many arid parts of Africa, millet is planted under *Acacia albida* trees. The acacias, which conveniently drop their leaves at the start of the growing season, provide nutrients for the growing crop; millet yields under the tree's canopy are 56–64% higher than yields obtained away from the tree.^[14] Most positive effects of trees are the result of the fixed nitrogen, but significant quantities of N can also be supplied in the leaf litter or from deliberate pruning. In Africa, leguminous shrubs (*Leucaena*) are grown as hedgerows 2–4 m apart, and their prunings added to crops grown in the alleys. This proved a remarkably efficient agroforestry technique to stabilize maize yields at about 2 ton/ha without the use of fertilizers.^[15]

Trees also improve the microclimate by acting as wind-breaks, by improving the water-holding capacity of the soil, and by acting as shade trees for livestock—so focusing the deposition of manure.^[16]

Intercropping, the process of growing more than one crop simultaneously in the same plot, is used throughout the world and has many advantages over monoculture, including a more efficient use of space and nutrients, higher total yields, and less vulnerability to crop failure due to pests.^[17] As in the case of maize–bean polycultures, legumes can be used in the intercrop to add nitrogen and thereby reduce competition between the intercropped species. In fact, bean-fixed N is made directly available to the maize through mycorrhizal fungi connections between root systems. Intercropping also helps prevent erosion and soil surface damage (i.e., degradation of surface pores) by keeping the soil covered. Due to these and other effects, polycultures generally overyield corresponding monocultures.^[18]

Integration of livestock

Livestock are a critical component of sustainable agricultural systems. The nutrient value of manures largely depends on how it is handled, stored, and applied. Losses of nitrogen tend to be highest when liquid systems of storage are used and when the manure is broadcast without incorporation. Livestock manures from cattle, pigs, and chickens are important, as they positively affect soil structure and water retention, and benefit soil organisms. Manure itself is an excellent fertilizer, and soils usually require 5–20 ton/ha of fresh manure to remain in continuous productivity.^[19]

It is becoming more common for small farmers to keep their animals permanently penned in zero-grazing or stall-feeding units rather than permit livestock to graze freely. In many rural areas, zero-grazing units are a central part of efforts to improve soil and water conservation. Fodder grown on the farm in the form of improved grasses, tree fodder, and the residues of cultivated crops are cut and carried to the animals. Because of the proximity to the crops, manures can be returned directly to the land, so improving nutrient supply and soil structure.^[20]

Reduced tillage

The use of reduced tillage or no till systems can maintain or enhance soil organic matter more than the conventional moldboard plow and disk system. The decreased soil disturbance under reduced tillage slows the rate of organic matter decomposition and helps maintain a soil structure that allows rainfall to infiltrate rapidly. Leaving residue on the surface encourages the development of earthworm populations, which also improves soil structure. Compared with conventional tillage, soil erosion is greatly reduced under minimum tillage systems, which helps keep the organic matter and rich topsoil in place. The only drawback of these systems is their high dependence on herbicides.

Composting

Organic wastes can also be concentrated through the process of composting, in which microorganisms attack the waste before it is used as fertilizer. There are many methods of composting, each combining the use of animal manures, green material, and household wastes in different ways and proportions. The materials are heaped or placed in a pit in such a fashion that anaerobic decomposition occurs. Harmful substances and toxic products of metabolism are broken down, while pathogens and the seeds and roots of weeds are destroyed by the heat generated within the compost heap. Compost created by a variety of methods has proven to be an effective fertilizer in many agricultural settings.

Inoculating the soil with microorganisms

Soil microbes play many beneficial roles in maintaining soil fertility. In some situations, however, these beneficial microbes might be present in very low numbers. In such situations, the soil can be “inoculated” with beneficial microorganisms thus enhancing crop productivity. These introductions may be done with any species of microbe that is known to benefit plant productivity. Those species that fix nitrogen, enhance phosphorus availability, increase mineralization of soil organic matter, or prevent diseases have attracted the attention of researchers. There are many studies that have examined the use of improved strains of the nitrogen-fixing bacterium *Rhizobium*, the nitrogen-fixing soil bacteria *Azotobacter* and *Azospirillum*, and Mycorrhizal fungi that enhance plant phosphorus nutrition.^[21]

INTEGRATED APPROACHES TO SOIL FERTILITY MAINTENANCE

As discussed earlier, there are many options for making soils more healthy and fertile. Combining the various options is challenging but usually implies the following.^[7]

1. Using rotations that utilize grass, legume, or a combination of grass and legume sod crops: or crops with large amounts of residue as important parts of the system.
2. Using of cover crops when soils would otherwise be bare to add organic matter, capture residual plant nutrients, and reduce erosion. Cover crops also help maintain soil organic matter in resource-scarce regions that lack possible substitutes to using crop residues for fuel or building materials.
3. Leaving residues from annual crops in the field or, if they were removed for use as bedding for animal, returning them to the soil as manure or compost.

4. Raising animals or having access to animal wastes from nearby farms gives farmers wider choices of economically sound rotations. Rotations that include perennial forages make hay or pasture available for use by dairy and beef cows, sheep, and goats. In addition, on mixed crop-and-livestock farms, animal manures can be applied on cropland. It is easier to maintain organic material on a diversified crop-and-livestock farm, where sod crops are fed to animal and manures returned to the soil. However, growing crops with high quantities of residues plus frequent use of green manures and composts from vegetable residues helps maintain soil organic matter even without animals.

Throughout the world, more and more farmers are learning that the combination of reduced tillage, cover crops, and better rotations can have a dramatic effect on their soil and the health of crops. They found that, by combining practices, they were able to reduce pest damage, improve soil tilth, vastly reduce runoff and erosion, increase soil organic matter, and produce better crop growth.^[22] Among resource-poor farmers in developing countries, adoption of sustainable soil fertility management practices can have important impacts on crop yields. An analysis of 208 agroecological projects showed clear increases in food production over 29 million hectares, with nearly 9 million households benefiting.^[11] In North America and Europe, organic farmers who adopt low-input soil fertility practices typically obtain yields 6–11% lower than conventional farmers who rely on chemical fertilizers. Such lower yields are, however, offset by the economic and environmental performance of organic agriculture.^[2]

REFERENCES

1. Conway, G.R. *The Doubly Green Revolution*; Penguin Books: London, 1997.
2. Lampkin, N. *Organic Farming*; Farming Press: Ipswich, 1990.
3. Lal, R.; Pierce, F.J., Eds. *Soil Management for Sustainability*; Soil and Water Conservation Society: Ankeny, 1991.
4. Altieri, M.A. *Agroecology: The Science of Sustainable Agriculture*; Westview Press: Boulder, 1995.
5. Gliessman, S.R. *Agroecology: Ecological Processes in Agriculture*; Ann Arbor Press: Ann Arbor, 1998.
6. Reinjntjes, C.; Haverkort, B.; Waters-Bayer, A. *Farming in the Future*; McMillan Press: London, 1992.
7. Magdoff, F.; van Es, H. *Building Soils for Better Crops*; Sustainable Agriculture Network: Beltsville, 2000.
8. Kotschi, J. *Ecofarming Practices for Tropical Smallholdings*; Masgraf: Weikersheim, 1990.
9. Gershuny, G.; Smillie, J. *The Soul of Soil: A Guide to Ecological Soil Management*; GAIA Services: St. Johnsburg, 1986.

10. Dover, M.; Talbot, L.M. *To Feed the Earth: Agroecology for Sustainable Development*; World Resources Institute: Washington, 1987.
11. Pretty, J.N. *Regenerating Agriculture*; Earthscan: London, 1995.
12. Sarrontino, M. *Northeast Cover Crop Handbook*; Soil Health Series, Rodale Institute: Kutztown, 1997.
13. Buckles, D.; Triomphe, B.; Sain, G. *Cover Crops in Hillside Agriculture*; International Development Research Center: Ottawa, 1998.
14. Poschen, P. An evaluation of the *Acacia albida*-based agroforestry practices in the hararghe highlands of eastern Ethiopia. *Agrofor. Syst.* **1986**, *4*, 129–143.
15. Kang, B.T.; Wilson, G.F.; Lawson, T.L. *Alley Cropping: A Stable Alternative to Shifting Cultivation*; International Institute of Tropical Agriculture: Ibadan, 1986; 22 pp.
16. Nair, P.K. *Agroforestry Systems in the Tropics*; Kluwer Academic Publication: Dordrecht, 1989.
17. Francis, C.A. *Multiple Cropping Systems*; Wiley and Sons: New York, 1986.
18. VanderMeer, J. The ecological basis of alternative agriculture. *Annu. Rev. Ecol. Syst.* **1995**, *5*, 201–224.
19. Follet, R.F. *Soil Fertility and Organic Matter as Initial Components of Production Systems*; SSSA special Pub. No. 19; Soil Science Society of America: Madison, 1987.
20. Pearson, C.J.; Ison, R.L. *Agronomy of Grassland Systems*; Cambridge University Press: Cambridge, 1995.
21. McGuinness, H. *Living Soils: Sustainable Alternatives to Chemical Fertilizers for Developing Countries*; Consumers Policy Institute: New York, 1993.
22. Edwards, C.A. *Sustainable Agricultural Systems*; SWCS: Amkeng, 1990.

Fertility: North China Plains

Xiangbin Kong

*College of Resources and Environmental Science, China Agricultural University,
and Key Laboratory of Farmland Quality, Monitoring and Control, National
Ministry of Land Resources, Beijing, China*

Rattan Lal

*Carbon Management and Sequestration Center, Ohio State University, Columbus,
Ohio, U.S.A.*

Abstract

Sustainable use and management of soil quality in Huang–Huai–Hai plain are critical to China's food security because it produces almost 35% of China's maize and 70% of wheat. Historically, soil quality of cropland was low due to salinization by topographic, hydrologic, and hydrogeologic conditions. Thus, a series of reclamation techniques for salt-affected soils were implemented. The widespread adoption of recommended techniques of soil reclamation improved soil quality. Consequently, soil organic carbon (SOC) was improved between the 1980s and 2010s, and SOC concentration is 12.34 g kg⁻¹ in Quzhou and 6.0–13.4 g kg⁻¹ in Yucheng. However, sustainable improvement in soil quality can also face major challenges of groundwater depletion, greenhouse gas emissions, excessive agricultural intensification, and the potential adverse effects on salt balance affected by increase in temperature. Strategies such as sustainable intensification, increase in SOC, and decrease in salt content must be the research priority for improving soil quality.

INTRODUCTION

Soil quality is the key factor affecting crop production and the quantity of input including chemical fertilizers, pesticides, groundwater, and the labor. Thus, soil quality is the basis of cropland production and a critical indicator of the environment.^[1] However, soil quality is affected by many factors such as climate, topography, parent materials, land use change, and intensification.^[2] While soil quality differs widely over time and space, management is critical to improving soil quality at plot, farm, region, and even global level.^[3–5] Therefore, management and sustainable use of soil quality are the research priority.

Soil quality is high in the United States that supports high crop production with less input;^[6] on the contrary, high crop production in China has been realized by higher input because of low inherent soil quality.^[7] Consequently, farmers in China have to apply high rates of nitrogen (N), phosphorus (P), and potassium to sustain high yields. Therefore, China is considered the global hot spot to ensure food security while reducing greenhouse gas (GHG) emissions by agricultural intensification.^[8] China, India, and the United States together account for 64–66% use of N and P. Furthermore, rice, wheat, and maize are alone responsible for 58–60% use of N and P.^[8] Thus, achieving food security in contemporary China faces major challenges including increase in crop production by sustainable systems of improving soil quality of cropland while reducing the negative environmental effects. Therefore, sustainable

management of soil quality of cropland helps China not only to ensure its food security but also to reduce GHG emissions. This entry aimed at considering the Huang–Huai–Hai (HHH) plain as an example to demonstrate the techniques of sustainable management of soil quality in China.

SOIL QUALITY MANAGEMENT IN THE HHH PLAIN

The HHH plain is formed by alluvial sediments deposited by the following three rivers: Huang He or Yellow, Huai, and Hai rivers. It is the largest plain (~350,000 km²) in China consisting of five provinces, i.e., Hebei, Henan, Shandong, Anhui, Jiangsu, and two municipalities i.e., Beijing and Tianjin. It is home to 35% of China's total population.^[5]

The region has a semihumid climate with an annual precipitation of 500–900 mm. About 60–75% of the annual precipitation is concentrated in summer monsoon period, and the other seasons either are dry or receive only a little rainfall. Topographic, hydrologic, and hydrogeologic conditions make the soils of this region prone to salinization. In addition, the alluvial plain is relatively flat in topography and has impeded surface drainage. Thus, climate, coupled with shallow groundwater of high salinity, leads to the widespread salinization in the HHH plain.

The entire HHH plain is divided into three subregions on the basis of soil and watershed flow, groundwater distribution, temperature, and precipitation. Subregion I comprises

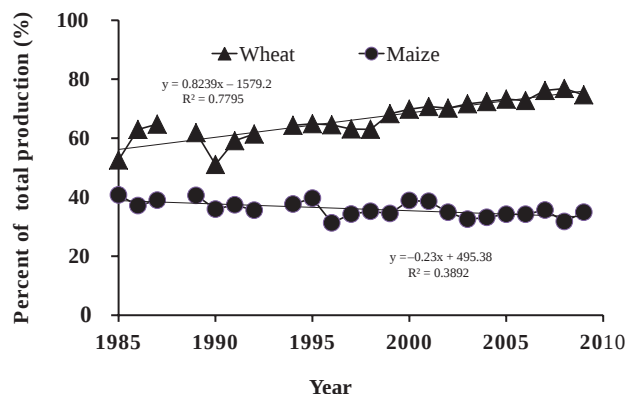


Fig. 1 Change in crop production during 1985–2010.

the piedmont plain of Taihang and Yan Mountains. Principal soils of the subregion I include argillic fluvo-udic and fluvo-aquic Inceptisols. The groundwater table is deep, and salt concentration is low ($<0.1\%$ in soil profile); subregion II comprises the flood and coastal plains of Huang He and Hai rivers. Principal soil types of this region are light, medium, and heavy saline fluvo-aquic Inceptisols. This subregion has saline phreatic water reserves ($2\text{--}10\text{ g L}^{-1}$), which may exacerbate the risks of salinization,^[9] subregion III comprises the flood plains of Huang He and Huai rivers. Principal soil types of this region are yellow fluvo-aquic and light saline fluvo-aquic Inceptisols, and relatively high rainfall leaches salts into the subsoil. Prior to the 1980s, there were 2.33 million hectares (Mha) of salt-affected soils and an additional 2.67 Mha of soils with high risks of salinization.^[9]

Soils of the HHH region have low inherent quality affected by the factors such as salinization, drought, and waterlogging due to the climate, topographic, hydrologic, and hydrogeologic conditions of this region. Therefore, the region suffers from both frequent droughts and waterlogging, and the soil salinization was a historically widespread problem.

Therefore, a series of reclamation techniques of salt-affected soils were implemented. Reclamation techniques included regionalization for comprehensive evaluation and monitoring, predicting, and regulating the regional water and salt movement. The widespread adoption of

recommended techniques of reclamation coupled with the soil quality improvement practices decreased salt concentration in soils and enhanced soil organic carbon (SOC) concentration across the HHH region.

Thus, HHH plain is an important region of arable land use in China with double-cropping system of winter wheat (*Triticum aestivum*) and summer maize (*Zea mays*). The HHH produces almost 35% of China's maize and 70% of wheat (Fig. 1),^[5] and the average yield of grain crops (kg ha^{-1}) has increased from 1582 to 5860 for wheat (Fig. 2) and 4492 to 5610 for maize (Fig. 3) between 1985 and 2010. The yields of HHH region are 1.25 and 1.07 times the China's national average yields of wheat and maize, respectively. However, agricultural intensification has also increased SOC concentration between the 1980s and 2010s, which was 12.34 g kg^{-1} in Quzhou, $6.0\text{--}13.4\text{ g kg}^{-1}$ in Yucheng, and 10.4 g kg^{-1} in Huantai.^[3]

CHALLENGES FOR SUSTAINABLE MANAGEMENT OF SOIL QUALITY IN THE HHH PLAIN

Managing soil quality of cropland is facing serve challenges in the HHH region from groundwater depletion, climate change, excessive agricultural intensification, and the scarcity of data on soil quality. All these factors impact soil quality and sustainable use. Thus, the management of soil quality should be done according to the changing trends in soil quality and the potential adverse effects.

The highest risks of adverse changes in soil quality come from the widespread groundwater depletion for both deep and shallow aquifers. The HHH accounts for 16% of the China's national average arable land but has access to only 7.6% of the China's national average renewable water resources compared with the highly endowed southern region.^[10,11] During the 1970s, farmers installed irrigation wells at 5–10 m depth.^[12,13] During the 2010s, the irrigation wells are installed at 300–400 m depth in eastern parts of the HHH, and many farmers have abandoned winter wheat in the northern parts of the HHH because of insufficient groundwater. Thus, the northern parts of the HHH are

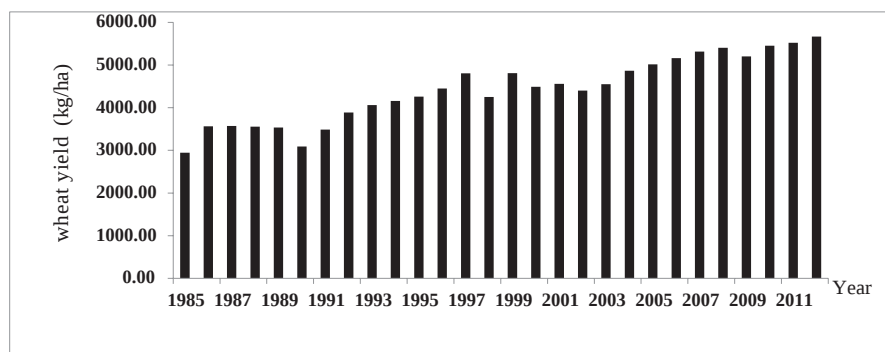


Fig. 2 Change in wheat yield during 1985–2011.

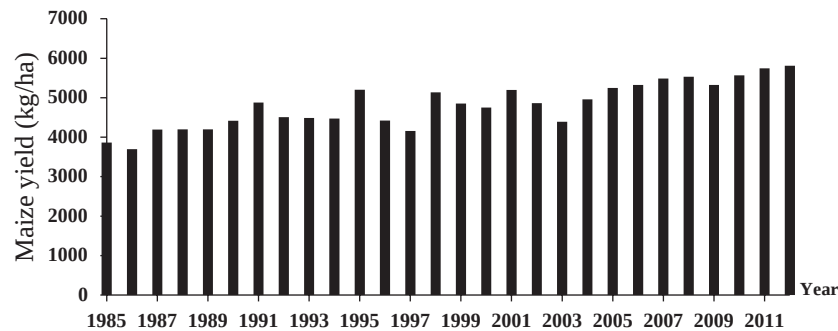


Fig. 3 Change in corn yield during 1985–2011.

no longer suitable for cultivation because the aquifer would be exhausted within 40 years.^[11] The seasonal and interannual deficit of groundwater due to intensive cropping of wheat and maize perturbs the hydrologic balance in the HHH. It is the excessive pumping of groundwater for intensive wheat production that causes a rapid decline in groundwater year after year.

Climate change is another potential risk factor to sustainable management of soil quality of cropland. The rate of increase of average temperature ranged from 0.4°C/decade at Nanyang to 0.61°C/decade at Zhengzhou and 0.71°C/decade at Luancheng, from south to north of HHH plains.^[14] Sunshine hours over the past 25 years (since 1985) decreased remarkably at both Zhengzhou and Luancheng but with a slightly declining trend at Nanyang. Annual mean precipitation in the HHH had a slightly increasing trend but with a slight decrease in Beijing. Climatic shifts in temperature and precipitation have a major influence on the decomposition and amount on SOC stored within a soil ecosystem.^[15] The SOC dynamics such as moisture, temperature, pH, and salt concentration in the topsoil will be affected by increase in temperature, precipitation, and groundwater depletion.

Groundwater depletion coupled with increase in temperature may lead to higher soil water transpiration, which will lead to pumping of more groundwater for irrigation. Furthermore, some soils of the cropland in the northern HHH region become rainfed cropland because of groundwater depletion. Consequently, salt in deep soil will move to surface layer by the higher transpiration if there is not enough groundwater for irrigation to leaching the salt into the subsoil. Therefore, the increase in salt concentration in topsoil will decrease SOC sink capacity and jeopardize soil quality.

The third challenge is from the agricultural intensification in HHH plain. SOC in the HHH has increased through agricultural intensification.^[5] Therefore, HHH has been treated as the leverage point to reduce the emission of GHG such as nitrous oxide and to reduce groundwater depletion.^[8] Thus, excessive intensification has to be shifted to sustainable or green intensification.^[16] As a result, managing soil quality of cropland in the HHH plain is becoming an arduous task for agronomist and soil scientists in China.

MANAGING SOIL QUALITY OF CROPLAND IN HHH PLAIN

Improve Soil Quality of Cropland by Reclamation

The improvement in soil quality in the HHH plain is affected by inherent soil characteristics and land use. Low-productivity soils (e.g., sandy and light soil and clayey soil) are distributed along the Huai, Huang He, and Hai rivers. These soil types are marginal croplands. Therefore, biocarbon practices may be used to alter soil texture, increase soil moisture reserves, reduce soil evaporation, and improve soil fertility.

Increase Soil Quality by Adopting Ecological Practices

A moderate level of cropland intensification can enhance SOC stock and reduce the depletion of groundwater. Thus, understanding principles and techniques to increase SOC stock and improve water use efficiency is a research priority. The adoption of best management practices for the HHH region [e.g., no-tillage (NT) farming, integrated nutrients management (INM) with the combined use of organic manure and balanced fertilizer, organic manure, improved grazing, and harvesting] is critical. The goal is to enhance the eco-efficiency of production systems and to improve the soil quality of cropland in the HHH region.

Measures to Improve SOC Stock

Adopting measures of increasing SOC stock and improving soil quality has numerous cobenefits (i.e., mitigating climate change and improving water quality). Therefore, incentivizing the farm household through carbon (C) trading and payments for ecosystem services is needed to improve soil quality. Well-planned measures may involve a series of steps as follows:

1. Cropland planning for increasing the SOC stock must be done on the basis of temperature, precipitation, soil subgroups, and the household factors and socioeconomic conditions. Detailed maps must be prepared for

the SOC stock at subregion and plot scales. Appropriate cropping systems must be outlined for subregional climate, water, and soil type. The rate of built-up of SOC stock must be evaluated, and the information was made available to policy-makers and the farm households.

2. Site-specific practices of improving SOC stock must be identified, such as the conversion of plow tillage to NT farming with mulching and use of cover crops, creation of positive C and nutrient budgets, reduction of nutrient losses by leaching and volatilization, minimization of soil disturbance, provision of a continuous ground cover by mulching and cover cropping, and use of INM strategies for applying manure and biofertilizers along with judicious use of chemical fertilizers.^[17]
3. Modus operandi must be developed for increasing SOC contents at the farm and community levels. The SOC stock is not only an indicator of soil quality but also a large C sink.^[18] Therefore, it is necessary to develop market-based mechanisms to trade soil C in the HHH region.

CONCLUSION

Soil quality of cropland in the HHH is critical to China's food security and environmental quality. Improvement in soil quality has built strong foundation for the increased yield of wheat and maize since the 1950s by a series of program funded by China central government. However, sustainable management of soil quality meets the major challenges from groundwater depletion, GHG emissions, excessive agricultural intensification, the potential adverse effects on salt, and moisture regimes affected by increase in temperature. Therefore, managing soil quality of cropland should take sustainable intensification, increase in SOC, and decrease in salt content as the researchable priority.

ACKNOWLEDGMENT

This work was supported by Chinese Social Science Fund (2014-14AZD031), China Agricultural University Fund (2014RW014), and China Scholarship Council Fund (201406355013).

REFERENCES

1. Lal, R. Soil carbon sequestration impacts on global climate change and food security. *Science* **2004**, *304*, 1623–1627.
2. Shi, Y.C. Comprehensive reclamation of salt-affected soils in China's Huang-Huai-Hai Plain. *J. Crop. Prod.* **2003**, *7* (1–2), 163–179.
3. Kong, X.B.; Zhang, F.R.; Qi, W. Influence of land use change on soil nutrients in an intensive agricultural region of North China. *Soil Till. Res.* **2006**, *88*, 85–94.
4. Xu, Y.; Wu, K.N.; Cheng, X.J.; Liu, P.J. Spatial variation in cultivated land production capacity and analysis of main impact factors for promoting production capacity in North-east China. *Res. Sci.* **2011**, *33* (11), 2030–2040.
5. Foley, J.A.; Ramankutty, N.; Brauman, K.A.; Cassidy, E.S.; Gerber, J.S.; Johnston, M.; Mueller, N.D.; O'Connell, C.; Ray, D.K.; West, P.C.; Balzer, C.; Bennett, E.M.; Carpenter, S.R.; Hill, J.; Monfreda, C.; Polasky, S.; Rockström, J.; Sheehan, J.; Siebert, S.; Tilman, D.; Zaks, D.P.M. Solutions for a cultivated planet. *Nature* **2011**, *478* (7369), 337–342.
6. Kong, X.; Lal, R.; Li, B.; Liu, H.; Li, K.; Feng, G.; Zhang, Q.; Zhang, B. Chapter Four—Fertilizer intensification and its impacts in China's HHH plains. *Adv. Agron.* **2014**, *125*, 135–169.
7. Kong, X. China must protect high-quality arable land. *Nature* **2014**, *506*, 7.
8. West, P.C.; Gerber, J.S.; Engstrom, P.M.; Mueller, N.D.; Brauman, K.A.; Carlson, K.M.; Cassidy, E.S.; Johnston, M.; MacDonald, G.K.; Ray, D.K.; Siebert, S. Leverage points for improving global food security and the environment. *Science* **2014**, *345* (6194), 325–328.
9. Shi, Y.C. Comprehensive reclamation of salt-affected soils in China's Huang-Huai-Hai Plain. *J. Crop Prod.* **2003**, *7* (1–2), 163–179 (in Chinese).
10. Liu, C.; Xin, J. Water problems and hydrological research in the Yellow River and the Huai and Hai River basins of China. *Hydrol. Process.* **2004**, *18*, 2197–2210.
11. The World Bank. *Addressing China's Water Scarcity: Recommendations for Selected Water Resource Management Issues*; International Bank for Reconstruction and Development: Washington, DC, 2009.
12. Moiwu, J.P.; Yang, Y.; Li, H.; Han, S.; Yang, Y. Impact of water resource exploitation on the hydrology and water storage in Baiyangdian Lake. *Hydrol. Process.* **2010**, *24*, 3026–3039.
13. Liu, Z.P. Impact of agricultural activities on region groundwater variation—A case study in Shijiazhuang (in Chinese) Doctoral dissertation, Beijing. *China Acad. Geogr. Sci.* **2010**, 29–35.
14. Liu, Y.; Wang, E.; Yang, X.; Wang, J. Contributions of climatic and crop varietal changes to crop production in the North China Plain, since 1980s. *Glob. Change Biol.* **2010**, *16*, 2287–2299.
15. Grace, P.R.; Colunga-Garcia, M.; Gage, S.H.; Robertson, G.P.; Safir, G.R. The potential impact of agricultural management and climate change on soil organic carbon of the North Central Region of the United States. *Ecosystems* **2006**, *9*, 816–827.
16. Garnett, T.; Appleby, M.C.; Balmford, A.; Bateman, I.J.; Benton, T.G.; Bloomer, P.; Burlingame, B.; Dawkins, M.; Dolan, L.; Fraser, D.; Herrero, M.; Hoffman, I.; Smith, P.; Thornton, P.K.; Toulmin, C.; Vermeulen, S.J.; Godfray, H.C. J. Sustainable intensification in agriculture: Premises and policies. *Science* **2013**, *341*, 33–34.
17. Lal, R. Beyond Copenhagen: Mitigating climate change and achieving food security through soil carbon sequestration. *Food Sec.* **2010**, *2*, 169–177.
18. Lal, R. Enhancing crop yields in the developing countries through restoration of the soil organic carbon pool in agricultural lands. *Land Degrad. Dev.* **2006**, *17*, 197–209.

Fertilizers: Application Methods

John Ryan

*International Center for Agricultural Research in the Dry Areas (ICARDA),
Aleppo, Syria*

Abstract

As crop yields increase, even more fertilizer than used at present will be needed. The growth in demand will come particularly from developing countries where fertilizer usage is low and where populations are expanding rapidly. The twin concerns—efficient economic use and mitigating environmental damage, as well as maintaining nutritional quality of crops—underscore the continued importance of precise, accurate, and efficient application of chemical fertilizers.

INTRODUCTION

As the beginning of civilization, soil fertility has been a dominant factor influencing human destiny. It took millennia before humans could unravel the mysteries of soil nutrients in relation to crops. In the 19th century, the invention of superphosphate laid the basis for the chemical fertilizer industry, an essential component of modern agriculture. Developments in fertilizer technology have provided an array of commercial fertilizer types and formulations, including solids, liquids, gases, and suspensions; single elements to complex multinutrients; soluble to slow-release materials; pelleted and powder forms; and bagged and bulk materials.^[1] Soil chemistry–plant nutrition research has provided the technical basis for fertilizer application methods. Both research and legislation have evolved to address changing fertilizer technology and cropping systems. The chemical fertilizer industry, a complex global one, is the cornerstone of agricultural production and world food supplies through the provision of crop nutrients.^[2] The broad goals of nutrient management are in the following: 1) cost-effective production of quality crops; 2) efficient use of fertilizers; 3) maintenance of soil quality; and 4) protection of the environment. This brief review highlights fertilizer application methods within such a context.

FERTILIZER APPLICATION: BASIC CONSIDERATIONS

Decisions about sustainable and economical fertilizer use must consider many crop, soil, and environmental factors. Soil testing is a well-established practice for identifying the fertility status of the soil, nutrient constraints, and the need for fertilization, especially in the west,^[3] however, progress has also been made in developing countries in promoting soil testing, correlation, and calibration.^[4] It involves

representative field sampling; analyzing the sample in the laboratory using appropriate tests; interpreting the test data using established norms; and making fertilizer recommendations. Criteria vary with the nutrient in question, soil type, or management. Apart from nutrients, crop growth is indirectly influenced by soil factors such as acidity, salinity, sodicity, toxic levels of elements, e.g., boron, and by high nutrient “fixation” capacity. Soil physical constraints such as compaction and increased bulk density must also be considered.

Crop characteristics such as yield goal or predicted yield and specific nutrient uptake are considered in fertilizer recommendations. Thus, fertilizer nutrient needs vary widely between short- and long-season crops and within crops; the higher the uptake or demand at maximum yield, the more fertilizer is needed. The architecture and extent of the crop’s root system, including mycorrhizal colonization, dictate the relative use of soil and fertilizer nutrients and fertilizer application methods.

Environmental factors govern fertilizer-use efficiency. For example, frozen soil leads to poor nutrient penetration and runoff loss; rain after application promotes leaching losses; high ambient temperature promotes both crop growth and nutrient uptake; warm, windy conditions can increase ammonia (NH₃) volatilization; and increasing water use, from rainfall or irrigation, enhances the uptake of fertilizer nutrients. Strategies to adapt to such conditions include splitting the fertilizer rates to coincide with crop growth demand, with due consideration of rainfall, and soil incorporation of nitrogen (N) fertilizers to minimize loss.

FERTILIZER APPLICATION METHODS

Given the variability of chemical fertilizers (physical state, type of nutrient, solubility, release rates) and range of crops (field, greenhouse, annual, perennial), fertilizer management conditions (rainfed, irrigated), and economic

considerations, fertilizer application methods have correspondingly changed.^[1] The application methods used should be accurate, efficient in terms of nutrient use, and economic in terms of comparative investment. Fertilizers can be applied during soil preparation and before planting the crop, i.e., the more traditional approach, and/or after crop emergence (Table 1). With increased fertilizer use and greater emphasis on efficiency, fertilizers can be applied by broadcasting, banding, or with water, i.e., foliar sprays and in irrigation water.

Table 1 Fertilizer application methods.

Application time/method	Attribute (+/-)	Remarks
1. Preplant		
Broadcast	+	Traditional, suitable for solids/liquids
	-	Less efficient for P, could cause NH ₃ loss
Injection	+	Reduces NH ₃ loss, suitable for immobile nutrients
	-	More expensive, elaborate equipment needed
2. Planting time		
With seed	+	Efficient, economic
	-	Needs machines, possible damage to germination, especially with N
Band placement	+	More efficient for P in high-fixing soils
	+	Can use liquids and solids
3. Postemergence		
Side dressing	+	Suitable for P and K
	-	Needs care with N
Topdressing	+	Suitable for cereals, perennial crops
	+	Solids and liquids can be used
4. Fertigation		
	+	High water and N-use efficiency
	-	High initial costs and maintenance
	-	Need soluble nutrient sources
	-	Inline precipitation, poor soil mobility with P
5. Foliar sprays		
	+	Suitable, effective where soil application is not possible
	+	Good for micronutrients, especially in perennial crops
	-	May not supply full N, P, K requirement
	-	Several sprayings needed
	-	Limited N concentration (1–2%), also biuret

Broadcasting

The essence of broadcasting fertilizer involves spreading the material evenly on the bare or cropped soil. Prior to the age of mechanization, solid-form fertilizer was spread by hand and is done so in many developing countries. Mechanized application represented a major improvement in speed, accuracy, and uniformity of spreading. The range of such equipment varies from small to large, and from pull-type drop-spreaders to self-propelled machines with power-driven, disc spreaders. Generally, the application rate of fertilizer in such machines is determined by their ground speed. Large, bulk fertilizer spreaders have flotation types to mitigate soil compaction.

Topdressing is essentially broadcasting over the established crop; it is the only feasible means for pasture and perennial forage crops and flooded rice. Fertilizer-use efficiency and reduced adverse environmental effects are promoted with splitting the fertilizer application between planting and a later crop growth stage. Thus, depending on the standing crop, its nutrient needs, and growing conditions, application of fertilizer to the crop stand is a common practice. Topdressing may be done several times with long-season crops. Plant analysis can help in determining the need for such applications.

The use of liquid fertilizers is a new development, mainly in the United States for N fertilizers; liquids can be sprayed or dribbled on the soil surface in bands between rows with gravity feed. Where spraying is involved, there is need for a tank, pump, pressure gauge, nozzles, fittings, and a spreading boom; it can have variable traction mechanisms, many tractor mounted. In the age of precision agriculture, sophisticated and expensive machinery is used to variably apply the fertilizer, liquid or solid, across a given field having different fertility levels, as determined by soil analyses, whose spatial variability is determined by global positioning systems.

Placement/Banding

The main approach to fertilizer application at planting time is with the seed or by banding. When applied with the seed, application rates must be limited for avoiding seedling damage and poor emergence, especially with N fertilizers and those with a high “salt index.” Banding implies application of fertilizer a few centimeters to the side and below the seed row. Fertilizer and seed are usually in the same drilling machine, but in parallel hoppers; both are delivered by metered gravity feed through tubes behind the shoe or disc. Banding reduces P fixation, and hence increases fertilizer-use efficiency. The efficiency of N fertilizer, alone or with P, is also increased.

The placing of the fertilizer, usually liquids, at some depth in the soil, via tool bar-mounted knives in shank openers, is referred to as injection or “knifing in.” It helps

reduce or eliminate volatile loss of N fertilizers such as urea and ammonium compounds, and, to a lesser extent, it places immobile nutrients like phosphorus (P) closer to the root zone and also reduces nutrient losses due to erosion and runoff. This can be done before planting or after crop emergence.

As with preplant application, liquids such as anhydrous NH_3 can be injected under pressure into the soil between or close to the seed rows; use of solid materials is less common by this method. Placement can be combined with row cultivation. The type of fertilizer must be considered; both P and potassium (K) should be close to the roots as they are immobile. Plant damage due to high localized NH_3 concentrations, as well as disruption of plant roots, must be avoided.

Fertigation

The practice of applying water and fertilizers together, fertigation, involves technical and economic advantages, e.g., savings in labor, time, and equipment and fuel costs.^[5] Open fertigation systems involve the metering of soluble solid or liquid fertilizer, mainly N, into irrigation ditches. However, volatile NH_3 gas loss can occur when the N source is urea or anhydrous NH_3 . Simultaneous application of sulfuric acid (H_2SO_4) can counteract the alkalinity and rise in pH following NH_3 application;^[6] it also prevents calcite precipitation,^[7] thus reducing the water's sodium absorption ratio and increasing infiltration. P is less suitable for open-channel application because it can be readily precipitated and thus be unevenly spread within the field. Where soils are sodic and have poor hydraulic characteristics, amendments such as polysulfides, gypsum, and H_2SO_4 can be readily metered into the irrigation system.

Closed fertigation systems involve fertilizer applied to irrigation water in pipes under pressure with pumping, e.g., sprinklers and trickle or drip irrigation, with many variations of the latter system; fertilizer solubility and subsequent nutrient movement in the soil are considered. As most N fertilizers are water soluble, N is easily used in both systems; K sources are soluble but relatively immobile and remain in the top few centimeters of soil or at the point of dripper. The primary concern with P fertilizers is poor solubility, precipitation, and limited mobility at the dripper site. With calcium (Ca)-rich water, soluble P forms precipitates in the tubing that can block the tiny orifices, thus rendering the system ineffective. Flushing with acids such as urea phosphate and phosphoric acid can dissolve such precipitates. Acidic solutions promote greater P mobility in the wetted soil volume and by dissolving insoluble soil P sources, namely $\text{Ca}_3(\text{PO}_4)_2$.^[8] Acidic P solutions may also enhance micronutrient availability in calcareous soils.

While close-system irrigation is a sophisticated technology, there are many obstacles to using fertilizers in such

systems. However, these obstacles are well understood by commercial farmers and irrigation company personnel, and workable solutions are readily available. As a consequence, fertigation is part of high-intensity production systems in field and under greenhouse conditions.

Foliar Feeding

The concept of applying nutrients as foliar sprays is relatively new and is a practice adapted in situations where soil application of fertilizers is ineffective and where a quick growth response is required.^[9] Aerial application of nutrients in sprays is used for broadacre crops where the equipment has problems entering or crossing the field and where large areas need to be sprayed or where nutrient uptake is limited by cold. Such nutrients can be normally combined with pesticides using ordinary pesticide sprayers; however, compatibility and stability of the ingredients must be considered.

However, as with any technology, there are pitfalls and obstacles. Fertilizer solution is a salt solution and as such can injure growing plants. Thus, nutrient concentration in the sprayed solution should be no greater than 1–2%. While urea is suitable for foliar sprays, the concentration of biuret, a toxic component, should be limited. Other practical considerations associated with foliar application involve varying the nutrient application relative to the irrigation spray and applying nutrient sprays frequently where the permissible concentrations of nutrients are low with respect to the growing crop needs, especially for major nutrients. However, one or two sprayings with micronutrients are usually sufficient to alleviate the deficiency.

CONCLUSION

As crop yields increase, even more fertilizer than used at present will be needed. The growth in demand will come particularly from developing countries where fertilizer usage is low and where populations are expanding rapidly. The twin concerns—efficient economic use and mitigating environmental damage, as well as maintaining nutritional quality of crops—underscore the continued importance of precise, accurate, and efficient application of chemical fertilizers. A major part of world food production is attributable to chemical fertilizers. This will continue—there is no alternative. But the industry must be sensitive to societal concerns. While the emphasis will continue to be on the environment in the developed world and on production in the developing countries, efficient fertilizer application technology can accommodate both concerns. Fertilizer application will, in future, be part of strategies of “positive soil fertility management,” where the broader perspective will apply, i.e., consideration of the whole cropping system—not just 1 year's

crop—and the residual or carryover value of the added fertilizer. Indeed, its implications for the entire food chain cannot be ignored. While research will continue to achieve even greater efficiencies of application, the greatest challenge will be to transfer what is known to those who need it most, especially in the developing world.

REFERENCES

1. UNIDO/IFDC. *Fertilizer Manual*; United Nations Industrial Development Organization, and International Fertilizer Development Center Kluwer: Dordrecht, 1998.
2. IFA/UNEP. *The Fertilizer Industry, World Food Supplies and the Environment*; International Fertilizer Industry Association and United Nations Environment Programme: Paris, 1998.
3. Matar, A.; Torrent, J.; Ryan, J. Soil and fertilizer phosphorus and crop responses in the dryland Mediterranean zone. *Adv. Soil Sci.* **1992**, *18*, 82–146.
4. Ryan, J., Ed. Accomplishments and future challenges in dryland soil fertility research in the Mediterranean area. In *Proceedings of the International Soil Fertility Workshop*, Nov 19–23; ICARDA: Aleppo, 1997.
5. Ryan, J., Ed. *Plant Nutrient Management under Pressurized Irrigation Systems in the Mediterranean Region*; World Phosphorus Institute, Casablanca, Morocco, and International Center for Agricultural Research in the Dry Areas: Aleppo, 2000.
6. Miyamoto, S.; Ryan, J.; Stroehlein, J.L. Sulfuric acid for the treatment of ammoniated irrigation water. I. Reducing ammonia volatilization. *Soil Sci. Soc. Am. Proc.* **1975**, *30*, 544–548.
7. Miyamoto, S.; Ryan, J. Sulfuric acid for the treatment of ammoniated irrigation water. II. Reducing calcite precipitation and sodium hazard. *Soil Sci. Soc. Am. J.* **1976**, *40*, 305–309.
8. Ryan, J.; Eisa, M.A.; Tabbara, H.; Baasiri, M. Phosphorus movement following water-applied urea phosphate and phosphoric acid in a calcareous clay soil. *J. Fert. Issues* **1998**, *5* (3), 89–96.
9. Soil Improvement Committee, California Fertilizer Association. *Western Fertilizer Handbook*, 7th Ed.; The Interstate Printers and Publishers: Danville, 1985.

Fertilizers: Leaching Losses

Paul G. Saffigna

Tropical Forestry, University of Queensland, Gatton, Queensland, Australia

Ian R. Phillips

School of Environmental Engineering, Griffith University, Nathan, Queensland, Australia

Abstract

Leaching is the downward movement of fertilizer or waste in soil with the drainage water and is a critical process in global nutrient cycling. To be available for plant uptake, fertilizer/waste (solute) must dissolve in the soil water. The combination of soil water and dissolved solute is the soil solution. The soil solution and soil air are present in the pores between soil particles. If water fills all the pores and no air is present, the soil is saturated and leaching is rapid. If both water and air are present, the soil is unsaturated and leaching is slower.

INTRODUCTION

Efficient nutrient use occurs when the plant demand coincides with the presence of solute in the root zone. When solute leaches below the root zone, it is unavailable for plant uptake and is lost from the soil–plant system. Depending on the amount of water draining below the root zone, the leached solute may simply accumulate at depth in the soil or it may pollute the underlying groundwater.^[1,2] Pollution of groundwater by fertilizers and waste is a major environmental problem worldwide.^[1] Although substantial progress has been made in understanding the movement of fertilizers and wastes in soil, the words of Leonardo da Vinci are equally applicable today as they were 500 years ago: “We know more about the movement of celestial bodies than about the soil underfoot.”

THE MECHANISMS AND PATTERN OF LEACHING

Water movement due to gravity causes leaching.^[3] When there is no mixing between the solute and the draining water, the solute is pushed ahead of the advancing water like a piston and remains in a narrow zone or band. This somewhat idealized leaching pattern is referred to as *piston flow* (Fig. 1B). Commonly, mechanisms such as dispersion, ion exchange, and ion exclusion modify this leaching pattern. *Dispersion* is a result of solute mixing with the drainage water by molecular diffusion^[3] and/or hydrodynamic dispersion^[3,4] and causes solute to be more “spread out” in the soil than if only piston flow occurred (Fig. 1C). The process of *cation exchange* with negatively charged soil colloids^[5] slows the rate of cation leaching in soil, reduces the number of cations in the soil solution, and skews the fertilizer leaching pattern (Fig. 1D). *Exclusion* of negatively

charged solute ions such as nitrate and chloride (Cl) by the negatively charged soil colloids^[3] confines them to the middle of the soil pores. Here water drains faster due to reduced friction, causing solute to leach more quickly than if piston flow alone was occurring (compare Fig. 1E with Fig. 1B).

MEASUREMENT AND MODELING OF LEACHING

Although the pattern of leaching is well represented by the examples shown in Fig. 1, this does not provide sufficient information to measure leaching losses. By multiplying the concentration of solute by the volume of drainage water, the amount of solute leached from the soil can be quantified. Representative samples of soil solution are critical for quantifying solute leaching.^[6] Column displacement and centrifugation are commonly employed to extract soil solution in the laboratory, while tension lysimeters are commonly used in the field. The volume of water draining through soil can be measured using repacked soil cores, intact (undisturbed) soil cores, and field lysimeters (undisturbed or refilled). These methods can be used to quantify the key mechanisms controlling the rate and magnitude of solute leaching in the laboratory and field.^[7]

Modeling solute leaching in and below the plant root zone is important for predicting soil water quality and for developing fertilizer and waste management practices which minimize adverse environmental impacts. Numerous analytical and numerical models have been developed which include key physical and chemical processes that affect solute leaching.^[8] Water flow models, solute transport models, and aqueous chemical equilibrium models were often developed independently, and solute leaching models mostly considered only a

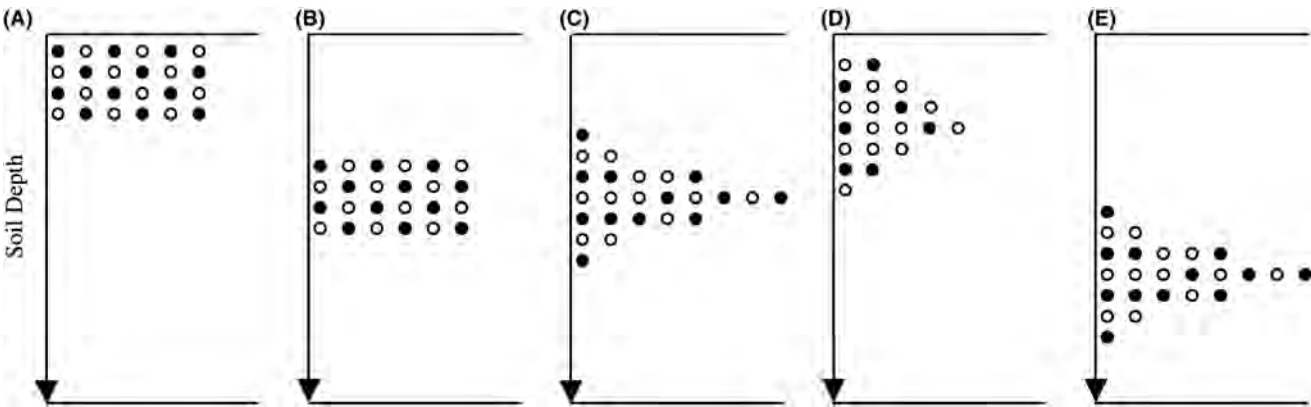


Fig. 1 Schematic representation of solute leaching in soil. (A) Solute distribution pattern prior to leaching—note that the solute band contains 12 cations (•) and 12 anions (○) (24 ions); (B) Solute distribution pattern after leaching involving piston flow only—note that the solute band still contains 24 ions and has only been moved deeper in the soil; (C) Solute distribution pattern after leaching involving piston flow and dispersion—note that the solute band has become more spread out but still contains 24 ions; (D) Solute distribution pattern after leaching involving piston flow, dispersion, and exchange—note that the solute band only contains seven cations, is skewed in shape, and is at a shallower depth than (B); (E) Solute distribution pattern after leaching involving piston flow, dispersion, and exclusion—note that the solute band still contains 24 ions but has moved deeper in the soil than (B).

single ion and simplified chemical processes. There has been a conscious effort to couple leaching models for water flow and solute transport with chemical equilibrium models.^[9]

MAGNITUDE OF LEACHING

Not all nutrient elements are susceptible to leaching, and the amount of a nutrient lost by leaching varies widely between climates, land use, soils, and nutrients (Table 1).^[10]

Generally, fertilizer-leaching losses are greatest in sandy soils, moderate in loamy soils, and least in clay soils for two

reasons. Firstly, water drains faster in sandy soils, which have a low capacity to retain cations. Secondly, anions tend to leach faster than cations because soils carry mainly negative electrical surface charge. For example, after adding 50, 200, or 1000 kg/ha of potassium chloride to a clay soil, about 7, 8, and 6 kg/ha of potassium (K) was leached below the root zone, whereas in a sandy soil receiving the same fertilizer rates, 7, 20, and 240 kg/ha of K was leached.^[14] In both of these soils, nearly all of the added Cl leached below the root zone. Nitrate (200 kg/ha of nitrate–nitrogen) and Cl leaching to depths >5 m and into the underlying groundwater have been found in sandy soils within 1 year of fertilizer application.^[1,11]

Table 1 Leaching losses from agriculture under different climates and land uses.

Country	Land use	Nutrient	Form	Rate applied (kg/ha/yr)	Amount leached (kg/ha/yr)	Reference
United Kingdom	Grazing	N	Fertilizer	250	4–184	[10]
United Kingdom	Grazing	N	Fertilizer	500	88–590	[10]
Europe	Cropped	N	Fertilizer	80	47–74	[10]
United States	Potatoes	N	Fertilizer	265	200	[11]
United States	Potatoes	Cl	Fertilizer	360	300	[11]
Australia	Bare	P	Wastewater	50	2–6	[12]
Australia	Bare	N	Wastewater	180–1500	113–315	[12]
Australia	Bare	K	Wastewater	300–2040	1–280	[12]
Australia	Cropped	Ca	Wastewater	185–530	127–330	[12]
New Zealand	Grazing	N	Wastewater	200–400	6–10	[13]
New Zealand	Grazing	N	Fertilizer	200–400	8–17	[13]
New Zealand	Bare	K	Fertilizer	50	1–8	[14]
New Zealand	Bare	K	Fertilizer	200	6–22	[14]
New Zealand	Bare	K	Wastewater	1000	20–240	[14]
New Zealand	Bare	Mg	Fertilizer	62	0.6–17	[14]

Although there are many benefits from using organic wastes as nutrient sources, their long-term environmental impact is a major concern. Sewage sludge can contain heavy metals (copper, lead, cadmium, and zinc) which can contaminate soils, while the use of wastewater can introduce very high concentrations of unwanted nutrients (nitrogen, Cl, and K), which can increase the likelihood of groundwater pollution. Generally, leaching of inorganic (fertilizer) phosphorus (P) is negligible; however, P leaching (ca. 10% of applied) has been observed in a wide range of soils treated with wastewater. The major source of wastewater P is dissolved organic P, and organic P compounds are relatively stable and mobile in soils. Consequently, the sustainability of long-term wastewater application to soil is a major environmental concern for many countries.

MITIGATION AND MANAGEMENT OF FERTILIZER NUTRIENT LOSSES

The amount of solute leached from soil is highly dependent on dissolved nutrient type (cation or anion), concentration and form (ionic or complexed), chemical, physical, and biological properties of the soil, timing and placement of fertilizer application, rainfall, and irrigation practice. For example, large leaching losses of fertilizer nitrogen occur if: it is present as nitrate; there is limited plant uptake; and the soil is permeable (well drained) and located in a high rainfall environment. Mitigation of fertilizer leaching is site specific and highly dependent on site management practices. Recommended management strategies include: 1) synchronization of fertilizer supply with crop demand and 2) employing irrigation strategies to minimize the drainage of soil water and loss of nutrients from the plant root zone. Other strategies include the use of nutrient release inhibitors, slow release fertilizers (maintains low-nutrient concentrations in the soil solution), and incorporation of amendments (zeolite, iron oxides, gypsum, and fly ash) into soil that remove fertilizer nutrients from the soil solution. There is an urgent need to develop best management practices for land disposal of organic wastes as these nutrient sources are likely to have a greater effect on soil and water pollution than fertilizers.

REFERENCES

1. Smil, V. Nitrogen in crop production: An account of global flows. *Global Biogeochem. Cy.* **1999**, *13*, 647–662.
2. Saffigna, P.G.; Keeney, D.R. Nitrate and chloride in groundwater under irrigated agriculture in Central Wisconsin. *Groundwater* **1977**, *15*, 170–177.
3. Biggar, J.W.; Nielsen, D.R. Miscible displacement and leaching phenomenon. In *Irrigation of Agricultural Lands*; *Am. Soc. Agron Monos. 11*; Hagen, R.M., Ed.; Am. Soc. Agron.: Madison, 1967; 254–274.
4. Fried, J.J.; Combarnous, M.A. Dispersion in porous media. *Adv. Hydrosol.* **1971**, *7*, 169–282.
5. Thomas, G.W. Historical developments in soil chemistry: Ion exchange. *Soil Sci. Soc. Am. J.* **1977**, *41*, 230–238.
6. Wolt, J. *Soil Solution Chemistry. Applications to Environmental Science and Agriculture*; Wiley: New York, 1994; 345 pp.
7. Phillips, I.R. Collection and automation of large undisturbed soil cores for laboratory leaching studies. *Commun. Soil Sci. Plant Anal.* **2001**, *32* (5-6), 843–862.
8. Ghadiri, H.; Rose, C.W. *Modelling Chemical Transport in Soils. Natural and Applied Contaminants*; Lewis Publishers: London, 1992; 217 pp.
9. Suarez, D.L.; Simunek, J. UNSATCHEM: Unsaturated water and solute transport model with equilibrium and kinetic chemistry. *Soil Sci. Soc. Am. J.* **1997**, *61*, 1633–1646.
10. Wild, A.; Cameron, K.C. Soil nitrogen and nitrate leaching. In *Soils In Agriculture*; Tinker, P.B., Ed.; Blackwell: Oxford, 1980; 36–70.
11. Saffigna, P.G.; Keeney, D.R.; Tanner, C.B. Nitrogen, chloride, and water balance with irrigated russet burbank potatoes in a sandy soil. *Agron. J.* **1977**, *69*, 251–257.
12. Phillips, I.R. Nutrient leaching losses from undisturbed soil cores following application of piggery wastewater. *Aust. J. Soil Res.* **2002**, *40*, 515–532.
13. Silva, R.G.; Cameron, K.C.; Di, H.J.; Hendry, T. A lysimeter study of the impact of cow urine, dairy shed effluent, and nitrogen fertilizer on nitrate leaching. *Aust. J. Soil Res.* **1999**, *37*, 357–369.
14. Phillips, I.R.; Black, A.S.; Cameron, K.C. Effect of cation exchange on the distribution and movement of cations in soils with variable charge. II—Effect of lime or phosphate on potassium and magnesium leaching. *Fert. Res.* **1988**, *17*, 31–46.

Fertilizers: Mineral

L.L. Hammond

*Economic and Policy Development Program, Market Development Division,
International Fertilizer Development Center (IFDC), Muscle Shoals, Alabama, U.S.A.*

Balu L. Bumb

International Fertilizer Development Center (IFDC), Muscle Shoals, Alabama, U.S.A.

Abstract

Mineral fertilizers, also known as chemical fertilizers (hereinafter mineral fertilizers are also referred to as fertilizers), refer to the products manufactured by the chemical fertilizer industry through chemical reactions of different elements or products. Nitrogen (N), phosphate, and potash fertilizers are primary mineral fertilizers produced by the fertilizer industry. N fertilizer products are mostly manufactured from ammonia (which is derived from chemical reaction of N and hydrogen). Phosphate fertilizers are derived by reacting phosphate rock with sulfuric or phosphoric acid, and potash fertilizers are refined from mined potash ores or sea brines.

INTRODUCTION

Harvested crops remove nutrients. Unless the removed nutrients are replenished, soil fertility declines and the capacity of the soils to produce additional crops degrades. To maintain soil health and productivity, nutrients must be continuously replaced. Because natural processes can replenish only limited quantities of the nutrients removed, these nutrients must be supplied from external sources including organic materials, biological fixation, and mineral fertilizers. Although organic and biological sources are important in supplying nutrients, these are not sufficient to meet the nutrient requirements of the food and fiber production needed for an ever-growing world population. World population increased from less than two billion at the turn of the 20th century to over six billion at the beginning of the 21st century and may reach over nine billion by 2050.^[1–3] Since the mid-1960s, mineral fertilizers have played an important role in promoting cereal production, especially in developing countries (Fig. 1).

MAIN FERTILIZERS—NUTRIENTS AND PRODUCTS

There are more than 100 chemical elements in nature but only 16 are considered essential for plant growth. If any of these 16 elements are lacking, plants cannot complete their vegetative or reproductive cycles. Thirteen of these are “mineral” nutrients (Table 1). Some of these nutrients combine to form compounds that compose cells, enzymes, etc. Others must be present in order for certain chemical and physiological processes of the plant to occur. A productive

soil should contain all the essential plant nutrients in sufficient quantity and in balanced proportions. Soils differ widely in their ability to provide the essential nutrients for plant growth, depending on the total amount of nutrient native to the soil and the mobilization or fixation reactions in the soil that affect the chemical form and resulting availability of the nutrients. When the nutrient supply from the soil is insufficient for crop needs, additional nutrient can be supplied by fertilizer to compensate for the difference. The type and the amount of fertilizer required depend on the crop to be grown and the soil characteristics.

The nutrients taken up by plants from mineral fertilizer products are in the same form as the nutrients taken up from organic sources, but many different products have been developed to meet specific needs for crop production.

Simple Fertilizers vs. Complex Fertilizers

Simple fertilizers contain only one primary plant nutrient. In contrast, complex fertilizers contain two or more plant nutrients. Complex fertilizers are produced by the processes involving chemical reaction between the constituents containing the primary plant nutrients (ammonium phosphates, nitrophosphate, NPK fertilizer, etc.). Fertilizers produced by granulating together single-nutrient materials or mixing granules of single-nutrient fertilizers are referred to as compound or mixed fertilizers. The nutrient content of fertilizer products, or “grade” expressed as a percentage, may refer to either the total or available nutrient content, and may be expressed traditionally for some nutrients in the oxide form [phosphorus pentoxide (P₂O₅) and potassium oxide (K₂O)].

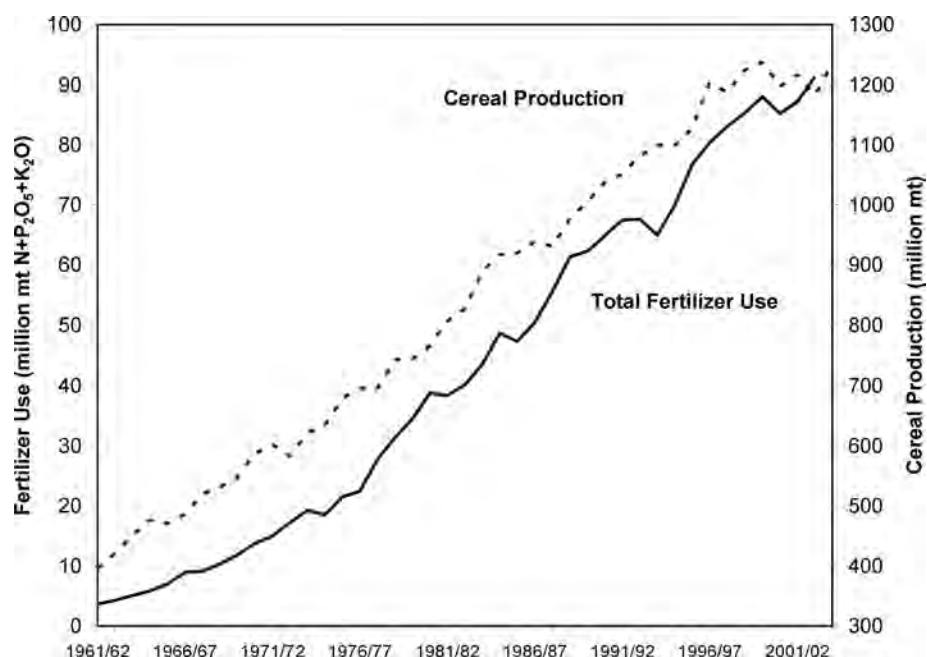


Fig. 1 Fertilizer use and cereal production in developing countries 1961/62–2002/03.

Source: Data from Food and Agriculture Organization of the United Nations (FAO).^[4]

Conversion factors from oxides to elements are as follows: $P = 0.436 P_2O_5$, $K = 0.830 K_2O$. Thus, 100 kg of 12:6:6 fertilizer contains 12 kg of N, 6 kg of P_2O_5 , and 6 kg of K_2O , respectively.

Table 1 Essential mineral nutrients and their chemical symbols.

	Principal chemical form taken up by plants	Nutrient uptake by wheat (5 mt/ha) (kg nutrient/ha)
Primary nutrients		
Nitrogen (N)	NH_4^+ , NO_3^-	105
Phosphorus (P)	$H_2PO_4^-$	18
Potassium (K)	K^+	15
Secondary nutrients		
Sulfur (S)	SO_4^{2-}	8
Magnesium (Mg)	Mg^{2+}	6
Calcium (Ca)	Ca^{2+}	2
Micronutrients		
Chlorine (Cl)	Cl^-	3
Iron (Fe)	Fe^{2+}	0.2
Manganese (Mn)	Mn^{2+}	0.2
Zinc (Zn)	Zn^{2+}	0.2
Copper (Cu)	Cu^{2+}	0.03
Boron (B)	H_3BO_3	0.02
Molybdenum (Mo)	MoO_4^{2-}	
Essential non-mineral nutrients		
Carbon (C)		
Hydrogen (H)		
Oxygen (O)		

N Fertilizers

N fertilizers can broadly be classified into four groups depending on the chemical form in which the N is present in them: ammonium fertilizers, nitrate fertilizers, combined ammonium and nitrate fertilizers, and amide fertilizers (Table 2). Also, there are NPK complex fertilizers, mixed compounds such as granular compound fertilizers, and bulk blends.

N fertilizers are completely water-soluble, and most sources are essentially equal in effectiveness for crops if incorporated into well-drained soils. If the N source applied is in the nitrate (NO_3^-) form, it is immediately available for plant uptake. If it is applied in the ammonium form, NH_4^+ is initially absorbed by the soil organic matter and clay particles. Under soil conditions suitable for good plant growth, however, the ammonium form is rapidly converted by nitrifying bacteria to the nitrate form (nitrification). Because the nitrate is not absorbed strongly by soil particles, it moves freely in soil water. For maximum uptake of N fertilizer, plants must be in a vigorous growth stage soon after the N is applied. Otherwise, the nitrate may move below the root zone.

Phosphate Fertilizers

Phosphate fertilizers vary widely in solubility. Phosphate rock is essentially insoluble in water and is only slowly available to plants if applied directly to the soil. Phosphate rock, however, is the basic raw material from which the most typically used phosphate fertilizers are produced. The main deposits and production units of phosphate rock are located in United States, Morocco, Togo, Senegal, Tunisia, Russia, Jordan, China, and Oceania.

Table 2 Fertilizer materials.

Fertilizer products	Grade (%)		
	N	P ₂ O ₅	K ₂ O
N fertilizers			
Ammonium fertilizers			
Anhydrous ammonia	82		
Ammonium sulfate (AS)	21		
Ammonium chloride	26		
Nitrate fertilizers			
Calcium nitrate	16		
Sodium nitrate	16		
Ammonium nitrate fertilizers			
Ammonium nitrate (AN)	34		
Calcium ammonium nitrate (CAN)	21		
Ammonium nitrate-sulfate	26		
Amide fertilizers			
Urea	46		
S-coated urea	35		
Calcium cyanamide	22		
Multinutrient fertilizers			
Nitrophosphate (NP)	21	21	
Mono-ammonium phosphate (MAP)	11	52	
Diammonium phosphate (DAP)	18	46	
Liquid ammonium polyphosphate	12	40	
Urea ammonium phosphate	28–32	30	
Phosphate fertilizers			
Water-soluble types			
Single superphosphate (SSP)		18	
Triple superphosphate (TSP)		46	
Partly water-soluble types			
Partially acidulated phosphate rock		(23–26)	
Slow-acting types			
Dicalcium phosphate (citrate soluble)		40	
Basic slag		9	
Calcium metaphosphate		62	
Very slow-acting types			
Rock phosphates		<32	
K fertilizers			
Muriate of potash (potassium chloride) (MOP)			60
Sulfate of potash (potassium sulfate) (SOP)			48
K-Mag/sulpomag (K, Mg sulfate)			23
Nitrate of potash (potassium nitrate)	13		44
Potassium polyphosphate		59	39

The rock can be solubilized by direct acidulation with sulfuric acid to form SSP or to produce phosphoric acid. The phosphoric acid can then be used to directly acidulate more phosphate rock to form TSP or the acid can be reacted with ammonia to form ammonium phosphates. The availability of the phosphorus to the plant is typically expressed as the sum of the water- and citrate-soluble P in the fertilizer source. Water-soluble P is easily available to plants, while citrate-soluble P is more slowly available to plants. The soluble P, however, can react with other minerals in the soil, thus reducing the immediate availability to plants. This transformation can be retarded somewhat through granulation and placement of the fertilizer. The P products formed in the soil, however, limit the movement of P in the soil and result in a longer term “residual value” not observed with N. Generally, the agronomic performance of phosphate fertilizers containing 60% or more water-soluble P is essentially equal to that of fertilizers containing 100% water-soluble P.^[5]

Potash Fertilizers

Canada has the world's largest known deposit of K. Potash is also mined in France, Germany, Italy, Spain, England, Israel, Jordan, Russia, Belarus, Ukraine, Brazil, and China. Potassium chloride (KCl) accounts for more than 90% of the K sold in North America. As with most of the common potash fertilizers, KCl is water soluble, whereas the concentration of K in KCl is high 60% and the concentration of chloride in KCl is about 45% and, therefore, potassium sulfate is often used for chloride-sensitive crops such as tobacco and to supply S. Sulfate of potash-magnesia contains not only potash but also about 11% Mg and 22% S. It is a good source of water-soluble K and Mg, and is important where Mg and/or S is deficient, or with Cl-sensitive crops. Potassium nitrate contains little or no Cl or S and is completely water soluble. It is widely used in foliar spray applications for fruit, vegetables, and cotton.

After application, most of the K from mineral fertilizer is absorbed near the application point in the soil by clay minerals and organic matter like ammonium N. It thus becomes “exchangeable” K. However, unlike ammonium N, which converts to a mobile form, K continues to be held in the soil and is relatively immobile.

Secondary and Micronutrients

While the nomenclature of secondary nutrients (Ca, Mg, and S) and micronutrients (Cl, Fe, Mn, Zn, Cu, B, and Mo) may indicate a lower degree of importance relative to N, P, and K, these nutrients are just as important to plant nutrition as the primary nutrients. They are referred to in this manner

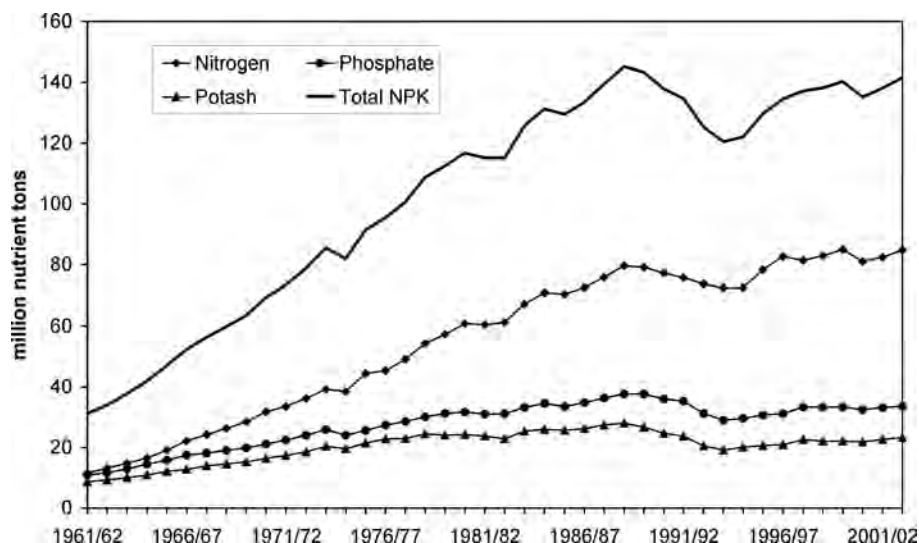


Fig. 2 World: N, phosphate, potash, and total NPK consumption, 1961/1962–2002/2003.

Source: Data from Food and Agriculture Organization of the United Nations (FAO).^[4]

because plants do not usually require as great a quantity of them. Calcium is generally supplied in the limestone (calcitic or dolomitic) used to reduce soil acidity or in basic slag from the steel industry or gypsum. The most common source of magnesium is dolomitic limestone. Other sources include potassium–magnesium sulfate, magnesium sulfate, magnesium oxide, and basic slag. Most fertilizer S sources are sulfates (ammonium, K, or magnesium) that are moderately to highly water soluble and are quickly available to plants. Elemental S (>85% S) can also be used as a fertilizer, but it results in a slower crop response because it is water insoluble. If well incorporated into the soil in advance of planting, however, elemental S is an agronomically effective and economically efficient S source.

While there are many fertilizer sources available to correct micronutrient deficiencies, some of the more common are borax (11% B), copper sulfate (23% Cu), iron sulfate (19–23% Fe), iron oxide (69–73% Fe), manganese sulfate (26–30% Mn), manganese oxide (41–68% Mn), ammonium molybdate (54% Mo), sodium molybdate (40% Mo), zinc sulfate (23–36% Zn), and zinc oxide (78% Zn). Cu-, Fe-, Mn-, and Zn-chelates are also common fertilizer sources.

FERTILIZER USE: HISTORY AND TRENDS

Although the first use of mineral fertilizers was introduced during the early 20th century, large-scale use of mineral fertilizers started only after World War II—initially as a result of the use of surplus ammonia capacity remaining from the production of explosives during the war and then as a result of the introduction of fertilizer-responsive improved crop varieties during the Green Revolution era.

At the turn of the 20th century, fertilizer use was insignificant. By 1950, it increased to 14 million nutrient tons, and by 1960, total nutrient use was about 28 million tons.^[6,7]

After the mid-1960s, because of the adoption of the Green Revolution technologies, fertilizer use increased rapidly—reaching 85 million tons in 1973/1974 and 146 million nutrient tons in 1988/1989 (Fig. 2). Thereafter, global fertilizer use decreased because of a steep decline in fertilizer use in the transitional markets (Eastern Europe and Eurasia). These markets are in-transition from centrally planned economies to free market economies. Policy reforms in Africa and Latin America and agricultural policy changes in the developed markets also contributed to this process (Fig. 3). After declining to a low of 120 million tons in 1993/1994, global fertilizer use recovered slowly to reach 142 million tons in 2002/2003. Yet, it was lower than the peak of 146 million nutrient tons that was reached in 1988/1989.

During the 1990s, while fertilizer use was declining in the transitional markets and was nearly stagnant in the developed markets, it continued to grow in the developing markets—mostly in Asian economies. Consequently, the developing markets' share of global fertilizer use increased from 34% in 1980/1981 to 66% in 2002/2003. China, United States, India, Brazil, and France are five main users of mineral fertilizers in the world. These countries account for 60% of the global fertilizer use.

Fertilizer Use by Nutrients and Products

In 2002/2003, global use of N, P₂O₅, and K₂O was estimated to be 84.7, 33.6, and 23.3 million nutrient tons, respectively. Nutrient use by different regions is shown in Table 3. East Asia, South-West Asia, and North America are dominant users of N and P₂O₅. In K₂O use, Latin America dominates South-West Asia. Transitional markets account for less than 5% of global fertilizer use.

Urea, DAP/MAP, and MOP are the main fertilizer products manufactured by the fertilizer industry. These products accounted for over 50% of global fertilizer use

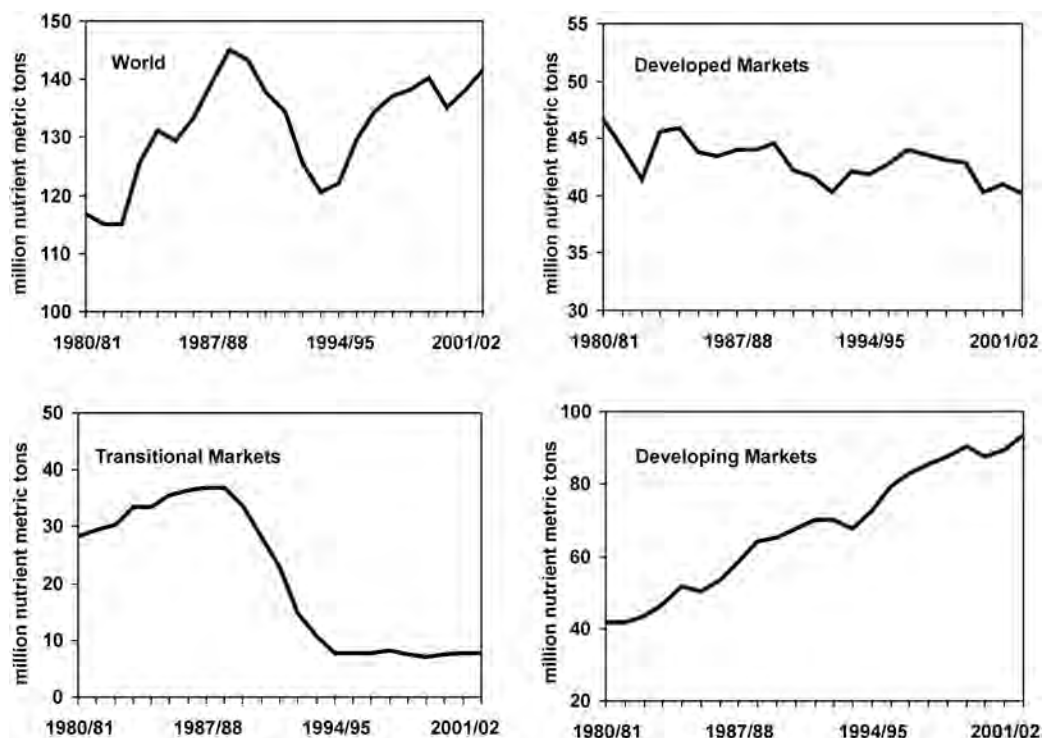


Fig. 3 Total fertilizer use by markets, 1980/1981–2002/2003.

Source: Data from Food and Agriculture Organization of the United Nations (FAO).^[4]

in 2002/2003 (Table 4). Other commonly used products are AS, AN/CAN, SSP, TSP, and NPKs.

Rapid growth in fertilizer use during the 1970s and 1980s also induced growth in fertilizer production and trade.

Table 3 Fertilizer use by regions and markets, 2002/2003.

Market/region	N (million nutrient tons)	P ₂ O ₅ (million nutrient tons)	K ₂ O (million nutrient tons)	Total (million nutrient tons)
Developed markets	22.9	8.9	8.5	40.2
North America	12.5	4.5	4.9	21.9
Western Europe	9.1	2.9	3.2	15.1
Oceania	1.3	1.5	0.4	3.2
Transitional markets	5.1	1.2	1.4	7.7
Eastern Europe	2.4	0.6	0.6	3.7
Eurasia	2.7	0.6	0.8	4.1
Developing markets	56.8	23.4	13.4	93.6
Africa	2.7	1.0	0.6	4.3
East Asia	31.6	12.2	6.7	50.5
South and West Asia	17.4	6.1	2.1	25.6
Latin America	5.0	4.2	4.0	13.2
World	84.7	33.6	23.3	141.6

Note: Totals may not add because of rounding.

Source: Data from Food and Agriculture Organization of the United Nations (FAO).^[4]

Total fertilizer production increased from 28 million nutrient tons in 1959/1960 to 90 million nutrient tons in 1973/1974 and to 158.5 million tons in 1988/1989 and then decreased until 1994/1995 and recovered slowly thereafter.^[9–11] In 2002/2003, 147.0 million tons of nutrient production consisted of 87.2 million tons of N, 33.9 million tons of P₂O₅, and 25.9 million tons of K₂O (Table 5). North America, Eurasia, East Asia, and

Table 4 Global fertilizer use by products, 2002/2003.

Product	N (million nutrient tons)	P ₂ O ₅ (million nutrient tons)	K ₂ O (million nutrient tons)	Total (million nutrient tons)
Urea	43.4			43.4
As	2.4			2.4
AN/CAN	8.8			8.8
TSP		1.8		1.8
SSP		6.1		6.1
DAP/MAP	4.9	14.5		19.4
MOP			14.8	14.8
SOP			0.8	0.8
NP/NPKs	9.3	9.1	7.0	25.4
Others	16.3	2.0	0.7	19.0
Total	85.2	33.4	23.4	142.0

Source: From International Fertilizer Industry Association.^[8]

Table 5 Global fertilizer production by markets and nutrients, 2002/2003.

Market/region	N (million nutrient tons)	P ₂ O ₅ (million nutrient tons)	K ₂ O (million nutrient tons)	Total (million nutrient tons)
Developed markets	21.2	10.7	13.3	45.2
North America	13.3	8.3	8.7	30.3
Western Europe	7.5	1.4	4.5	13.4
Oceania	0.4	1.0	0	1.5
Transitional markets	13.7	3.9	8.2	25.8
Eastern Europe	3.3	0.7	0	4.0
Eurasia	10.4	3.2	8.2	21.8
Developing markets	52.3	19.3	4.4	76.0
Africa	3.1	2.7	0	5.8
East Asia	28.5	9.4	0.5	38.4
South and West Asia	18.1	5.5	3.1	26.7
Latin America	2.7	1.6	0.8	5.1
World	87.2	33.9	25.9	147.0

Note: Totals may not add because of rounding.

Source: Data from Food and Agriculture Organization of the United Nations (FAO).^[4]

South/West Asia are dominant producers of N and P₂O₅, whereas K₂O production is mostly concentrated in Eurasia and North America.

Fertilizer Use Efficiency

Because of rapid growth in fertilizer use in the past, little attention was paid to the efficiency of fertilizer use. Overall, fertilizer efficiency may not exceed 40%. N use efficiency may vary from 30% to 50% for rice, maize, and sorghum in developing countries.^[12] Phosphorus and K use efficiencies are also low. Improving fertilizer use efficiency by proper placement and timing of fertilizer products and soil- and crop-specific recommendations could lead to a win-win situation by increasing crop productivity and reducing nutrient losses to the atmosphere thereby protecting the environment. New technologies of precision agriculture may prove useful in improving nutrient use efficiency by targeting fertilizer applications more accurately.

CONCLUSION

At the turn of the 21st century, the world faces many challenges. First, food production must be increased to eliminate hunger and malnutrition and ensure food security for all. Second, natural resources, such as rain forests, wildlife habitats, and biodiversity, must be protected. Third, soil degradation and nutrient depletion must be halted. Fourth, global warming and pollution must be reduced.

In confronting all these challenges, increased and environmentally sound use of mineral fertilizers will be essential for enhancing the productivity of cultivable lands, so that more food could be produced while protecting the environment by sparing forests and marginal lands, replenishing the removed nutrients, and sequestering carbon in the biomass production and the soils.

Because of uncertainties associated with events, it is difficult to make precise projections. However, one study estimated that annual fertilizer use should increase to 263 million tons to satisfy the food production requirements in 2020.^[13] This means that fertilizer use should be nearly doubled. Given the successes of the past (over five-fold increase in fertilizer use during the 1960–1990 period), this may not be a difficult target to achieve but would require a strong commitment from all stakeholders including donors and policymakers.

REFERENCES

1. United Nations (UN). *The State of World Population, 1988*; UN: New York, 1988.
2. UN. *Long Range World Population Projections: Two Centuries of Population Growth, 1950–2150*; UN: New York, 1992; 32 pp.
3. UN. *World Population Prospects: The 2000 Revision—Highlights, Draft*; UN: New York, 2001.
4. Food and Agriculture Organization of the United Nations (FAO). Available at <http://faostat.fao.org>.
5. Engelstad, O.P.; Hellums, D.T. *Water Solubility of Phosphate Fertilizers: Agronomic Aspects—A Literature Review, Paper Series P-17*; IFDC: Muscle Shoals, 1993; 17 pp.
6. Food and Agriculture Organization of the United Nations (FAO). *Fertilizers: An Annual Review of World Production, Consumption and Trade*; FAO: Rome, 1961.
7. FAO. *Fertilizers: A World Report on Production and Consumption*; FAO: Rome, 1952.
8. International Fertilizer Industry Association. *IFADATA Statistics; 1973/74 to 2003/03*, CD-ROM: Paris, 2004.
9. Bumb, B.L. *Global Fertilizer Perspective, 1960–95: The Dynamics of Growth and Structural Change*; Technical Bulletin T-34, Executive Summary T-35; IFDC: Muscle Shoals, 1989.
10. Bumb, B.L. *Global Fertilizer Perspective, 1980–2000: The Challenges in Structural Transformation*; Technical Bulletin T-42; IFDC: Muscle Shoals, 1995.
11. Bumb, B.L.; Berry, J. *Global and Regional Data on Fertilizer Production and Consumption, 1961/62–2002/03*. IFDC: Muscle Shoals, 2004.
12. Peoples, M.B.; Freney, J.R.; Mosier, A.R. Minimizing gaseous losses of nitrogen. In *Nitrogen Fertilization in the Environment*; Bacon, P.E., Ed.; Marcel Dekker, Inc.: New York, 1995; 565–602.
13. Bumb, B.L.; Baanante, C.A. *The Role of Fertilizer in Sustaining Food Security and Protecting the Environment to 2020, Food, Agriculture, and the Environment Discussion Paper 17*; International Food Policy Research Institute (IFPRI): Washington, 1995.

Fertilizers: Organic

Gary J. Gascho

Crop and Soil Sciences Department, University of Georgia, Tifton, Georgia, U.S.A.

Abstract

By one definition, organic fertilizers may be nutrient sources that can be used in organic farming. Organic fertilizers have been used for thousands of years. Ancient people principally used animal wastes to enhance the growth of plants, usually with great success. Guano, a commercial organic material from the excreta of seabirds and bats, was an important fertilizer in the Southern United States in the early 1900s. The application of organic fertilizers has become much more complicated, as modern municipal and industrial products, by-products, and wastes must be considered.

INTRODUCTION

By one definition, organic fertilizers may be nutrient sources that can be used in organic farming. With such a definition, inorganic materials that are deemed to be “natural,” such as mined gypsum, untreated salts, unprocessed lime, etc. can be included.^[1] For this entry, we define organic fertilizers as those that are organic, from a chemical standpoint. Only an overview of the subject can be provided in this brief contribution. More comprehensive publications are available.^[1,2]

ANCIENT APPLICATIONS OF ORGANIC FERTILIZER MATERIALS

The first fertilizers were organic. They have been used for thousands of years, and the literature is voluminous. Ancient people principally used animal wastes to enhance the growth of plants, usually with great success. Guano, a commercial organic material from the excreta of seabirds and bats, was an important fertilizer in the Southern United States in the early 1900s. The application of organic fertilizers has become much more complicated, as modern municipal and industrial products, by-products, and wastes must be considered.

GENERAL ADVANTAGES AND DISADVANTAGES OF ORGANIC FERTILIZERS

Agriculture occupies a great proportion of the land mass in comparison to industrial operations. It is quite natural that a disposal philosophy exists. Unwanted wastes are often promoted as beneficial for farm crops, sometimes only to remove the wastes from manufacturing sites or to save costly landfilling. Agriculture should make careful assessments of all fertilizer materials applied but especially of industrial and municipal wastes. That assessment should

include not only the nutrient value in relation to need but also the concentration of elements and compounds that could lead to crop and/or soil toxicity. The concentrations of heavy metals are a primary concern in many cases. When applying wastes containing the acceptable concentrations of heavy metals, careful maintenance of soil pH at >6.0 can help limit their release in the soil and availability to plants.

A general advantage of organic fertilizers can be the cost of nutrient, but that is not always the case. It largely depends on the proximity of the organic fertilizer to the site of application and the value of the organic. In cases where a material must be placed in a landfill, the material may have a negative value due to the cost and maintenance of that landfill. If so, a material may be obtained at little or no cost, or even at a negative cost. As organic fertilizers generally have lesser nutrient values than modern manufactured inorganic fertilizers, the cost of transportation per unit of nutrient is often the controlling factor in value of the fertilizer. A second potential advantage of organic fertilizers is that they may provide nutrient release over an extended period of time; in particular, nitrogen (N) can be mineralized over a season to eliminate the need for repeated applications. Soil organic matter can be increased when organic fertilizers are applied at sufficient rates. The value of soil organic matter is well recognized, but the ability of added organic matter to substantially add to the pool of soil organic matter is debatable. It greatly depends on many factors, especially climatic conditions.

General disadvantages of organics are their fertilizer ratios that are not likely to be matched to plant needs, their non-uniform composition that can prohibit precise uniform applications, and their unpredictable release of nutrients that may not coincide with the time a particular nutrient is most required by a particular plant. Some organic fertilizers may also contain unwanted, harmful, and/or toxic elements or compounds making them unsatisfactory for utilization. Some may be objectionable due to strong smells or irritants to humans. This latter disadvantage is important

to the quality of life in residential developments near the sites of application.

ANIMAL RESIDUES

Manures

Animal manures were the first fertilizers used in agriculture. Feces and urine from large animals and feces from poultry are the main sources. Most are applied, together with the bedding material, directly to the soil and usually in near proximity to the confined animals. Some are composted and/or pelleted to provide a material that is more agreeable for handling and transport. Nutrient composition can vary widely depending on the animal, its feed, the bedding, and the proportion of manure to bedding, and handling or storage prior to the application (Table 1).

Because of the importance of animal agriculture, manures will continue to be important sources of fertilizers and pose problems for agriculture and society. As farmers move toward precision applications (that provide improved nutrition without overapplication that can result in pollution of soil or water), it becomes increasingly important to analyze manures and apply them in accordance with plant needs. Analyses provided indicate that phosphorus (P) composition is highly relative to N for most plant needs. Therefore, application according to N needs provides too much P that may be bioavailable and add to the *eutrophication* of water via runoff. In some areas with high soil P, the recommendation is to apply manures on the basis of P needs. Such a philosophy will greatly limit applications in fields that are close to confined animals. It will also require N application and possibly other fertilizer nutrients from commercial fertilizers for balanced fertilization of most crops.

One attempt to provide more uniform applications and products that are more environmentally friendly to transport is to compost or pelletize manures. Such operations concentrate the manure and reduce transportation costs. Many composting processes have been developed. The economics are favorable for some specialty plants but not for broad-scale crop production.

Table 1 The concentrations of N, P, and K on a dry weight basis in commonly applied wastes.

Waste	N	P	K	References
Livestock manures	1–3%	0.4–2.0%	1–2.5%	[1,2]
Poultry manures	3–5%	1–3%	1–2%	[1,2]
Plant residues	1–7%	0.1–1.7%	0.1–9%	[1]
Municipal biosolids	2–9%	1.5–5%	0.2–0.8%	[3]
Municipal effluents	1.6–2.7 mg L ⁻¹	0.2–1.2 mg L ⁻¹	1.1–1.7 mg L ⁻¹	[4,5]

Other Animal Residues

Bonemeal, dried blood, and other animal processing wastes are important fertilizers but account for little of the total fertilizer used in the United States. Bonemeal is a good source of P, often used by the homeowner but seldom used in commercial agriculture. Dried blood is a rapid releasing organic N source. Other animal wastes are variable in composition and are often considered as disposal problems rather than desirable fertilizers. Applications may be difficult due to poor and non-uniform physical characteristics. Chemical analysis should be known before application.

PLANT RESIDUES

Leaving plant residues on the soil surface provides both nutrients for a following crop and soil protection from wind and rain erosion. Plant residue fertilization may also be obtained from the application of wastes or by-products from many agricultural processing plants. Trash from cotton gins, filter mud from sugarcane, and processing wastes from fruits and vegetables are but a few of the plant residues utilized. Soil conservationists have long recommended keeping the soil protected by maintaining cover crops during the seasons when the main crops are not planted and returning those plant residues to the soil. The modern emphasis on conservation tillage includes planting directly into cover crops. If the cover crop is a legume, N may be supplied to the planted crop due to the nitrogen gas fixation from the legume. A good stand of alfalfa, clover, bird's-foot trefoil, or vetch plowed into the soil may replace a fertilizer N application of 80–125 kg N ha⁻¹ for a following crop of corn.^[6,7] It is believed that such cover crops used in conservation tillage and not incorporated would supply nearly as much N to the following crop.

MUNICIPAL WASTES

Sewage treatment plants produce fertilizers in the forms of effluent (the liquid portion) and as biosolids that are settled from the effluent. In some cases, biosolids are placed in a landfill, but costs of maintaining and monitoring seepage from landfills is making the economics favor transportation and land application. Proper handling in the treatment plant removes most human and animal health concerns, but not the stigma of such applications.

Effluents

Ammonium and nitrate N should be monitored to avoid overapplication of effluents, as N leaching can be a problem in sandy soils. When application is made year-around, it is necessary to keep the soil covered with an actively

growing crop to minimize leaching.^[8] P fixation capacity of the soil is an important determinant of the amount of effluent that should be applied without promoting a soluble-P problem. High fixing soils will handle greater amounts of P than soils with little fixing capacity.^[9] Heavy metals are not a problem, as they are concentrated in the solids. N, P, and potassium (K) concentrations of effluents vary quite widely (Table 1).

Biosolids

Because of advances in technology, treatment plants are able to remove nearly all heavy metals and bacteria, allowing the nutrient-rich organic material by-products or biosolids to be recycled and applied as fertilizer. Elemental concentrations in biosolids from different sources have extremely wide variances (Table 1). They are generally good sources of both N and P for crops, but as for most manures, P application will be too great if enough is applied to satisfy N requirements. Application according to P requirements will allow little or no use in many cases.

OTHER BY-PRODUCTS AND WASTES

Many other organic by-products and wastes may be suitable and even valuable as fertilizers. Included would be paper and pulp, vegetable and fruit processing by-products, food wastes, etc.—the list is too extensive to be included here. All wastes should be analyzed for nutrient elements and for potential pollutants prior to application.

REFERENCES

1. Huntley, E.E.; Barker, A.V.; Stratton, M.L. Composition and uses of organic fertilizers. In *Agricultural Uses of Byproducts and Wastes*; Rechcigl, J.E., MacKinnon, H.C., Eds.; American Chemical Society: Washington, 1997; 120–139.
2. Miller, D.M.; Miller, W.P. Land application of wastes. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; G-217–G-242.
3. Bruulsema, T.W.; Christie, B.R. Nitrogen contribution to succeeding corn from alfalfa and red clover. *Agron. J.* **1987**, *79*, 96–100.
4. Fox, R.H.; Piekielek, W.P. Fertilizer nitrogen equivalence of alfalfa, birdsfoot trefoil, and red clover for succeeding corn crops. *J. Prod. Agr.* **1988**, *1*, 313–317.
5. Hook, J.E. Comparison of the crop management strategies developed from studies at Pennsylvania state university. University of Minnesota, and the Muskegon county land treatment system. In *Land Treatment of Municipal Wastewater*; D'Itri, F., Ed.; Ann Arbor Science: Ann Arbor, 1982; 65–78.
6. Sumner, M.E. Beneficial use of effluents, wastes, and biosolids. *Commun. Soil Sci. Plant* **2000**, *31* (11–14), 1701–1715.
7. Ellis, B.G.; Erickson, A.E.; Jacobs, L.W.; Knezek, B.D. Crop management studies at the Muskegon county Michigan land treatment system. In *Land Treatment of Municipal Wastewater*; D'Itri, F., Ed.; Ann Arbor Science: Ann Arbor, 1982; 49–63.
8. Dowdy, R.H.; Clapp, C.E.; Marten, G.C.; Linden, D.R.; Larsen, W.E. Wastewater crop management studies in Minnesota. In *Land Treatment of Municipal Wastewater*; D'Itri, F., Ed.; Ann Arbor Science: Ann Arbor, 1982; 35–47.
9. Forste, J.B. Biosolids processing, products and uses. In *Agricultural Uses of Byproducts and Wastes*; Rechcigl, J.E., MacKinnon, H.C., Eds.; American Chemical Society: Washington, 1997; 50–62.

Fertilizers: Types and Formulations

Douglas Robert McGuffog

McGuffog & Co Pty Ltd, Noosa Heads, Queensland, Australia

Abstract

Fertilizer products are applied in crops, pastures, horticulture, ornamental plants, and turf to provide nutrients that are essential for plant growth. The choice of product will depend upon a number of factors including the nutrient status of the soil and the plant's requirements. A range of basic fertilizer types are produced as solids, liquids, or gas in large-scale chemical manufacturing plants or from mining processes. Urea, containing 46% nitrogen (N); diammonium phosphate, containing 18% N and 20% phosphorus (P); and potassium chloride, containing 50% potassium (K), are the most widely used products and are usually applied in solid granular form. The use of fluid forms of fertilizers is increasing in large scale intensely farmed areas in a number of countries. Ammonium polyphosphates, containing 10–11% N and 15–16% P; urea ammonium nitrate, containing 28–32% N; and potassium chloride are the most widely used components in the production of fluid fertilizers. Both solid and fluid fertilizers may be blended or mixed, and in the case of solid products compounded or coated, to provide a wide range of formulations to suit particular nutrient requirements and method of application. Secondary nutrients (calcium, magnesium, and sulfur) and micronutrients (boron, chlorine, copper, iron, manganese, molybdenum, and zinc) are usually incorporated with the primary nutrients (N, P, and K) in blends, mixtures, or compounds. Animal manures are an important source of plant nutrients in a number of countries.

INTRODUCTION

Mineral fertilizers provide essential plant nutrients for application to crops, pastures, horticulture, ornamental plants, and turf. There is a wide range of fertilizer types and formulations available that enable users to match a fertilizer to suit specific soil and plant requirements and method of application.

Fertilizers are classified by their plant nutrient content. Nitrogen (N), phosphorus (P), and potassium (K) are referred to as primary nutrients; calcium (Ca), magnesium (Mg), and sulfur (S) as secondary nutrients; and boron, chlorine, copper, iron, manganese, molybdenum, and zinc as micronutrients. Selenium is a micronutrient that is not required by plants but is important for grazing animals.

Table 1 shows the primary and secondary nutrient contents of the most commonly used fertilizers.

Animal manures are also an important source of plant nutrients in many countries and have the benefit of returning organic matter to the soil which in turn improves moisture retention.

The nutrient requirements for crop or pasture production will depend upon a number of factors such as the fertility status of the soil, the crop requirements, and the nutrient uptake pattern of the plant. This entry provides a brief overview of the major mineral fertilizer types and how they are formulated as mixtures, blends, and compounds to suit the particular application need.

TYPES OF FERTILIZER PRODUCTS

Fertilizers are produced as solids, liquids, and gas.

Most fertilizers are applied as a solid material, usually in a granular form. Fluid fertilizers (liquids) are applied as suspensions or solutions in a number of countries. In some countries, ammonia is applied directly to the soil.

Government regulations require the labeling of fertilizers with their plant nutrient content usually known as the guaranteed analysis.^[1] The N, secondary nutrient, and micronutrient contents are normally expressed as a percentage by weight of the mineral element. The P and K contents are expressed either as percentage of the element or of the oxide form, depending on individual country regulations. The typical analyses of commonly used fertilizers, expressed as a percentage by weight in both the elemental and oxide forms for the P and K contents, are shown in Table 1.^[1,2]

Gaseous Fertilizers

Anhydrous ammonia is a liquefied gas that is transported, stored, and applied under pressure. It is mostly used as a preplant fertilizer during crop fallows and is drilled directly into the soil.

Fluid Fertilizers

These are major forms of fertilizer usage in the United States, particularly for applications of N, and their use is

Table 1 Typical analyses of commonly used fertilizers.

Fertilizer product	Nitrogen	Phosphorus		Potassium		Sulfur	Magnesium	Calcium
	% N	% P	% P ₂ O ₅	% K	% K ₂ O	% S	% Mg	% Ca
Anhydrous ammonia	82	—	—	—	—	—	—	—
Urea	46	—	—	—	—	—	—	—
Ammonium nitrate	34	—	—	—	—	—	—	—
UAN	28–32	—	—	—	—	—	—	—
Ammonium sulfate	20–21	—	—	—	—	24	—	—
CAN	21–27	—	—	—	—	—	—	8–14
MAP	10–12	22–24	50–55	—	—	1–2	—	—
DAP	18	20	46	—	—	1–3	—	—
Triple superphosphate	—	20	46	—	—	1–4	—	15–16
Ammonium polyphosphate	10–11	15–16	34–37	—	—	—	—	—
Single superphosphate	—	8–9	18–21	—	—	11	—	18–20
Potassium sulfate (muriate of potash)	—	—	—	48–51	58–62	—	—	—
Potassium sulfate (sulfate of potash)	—	—	—	40–42	48–50	16	—	—
Potassium nitrate	13	—	—	37–38	44–45	—	—	—
Potassium magnesium sulfate	—	—	—	18	22	22	11	—

growing in other parts of the world.^[1] There are two types: solutions and suspensions.

Solutions

All the ingredients are completely dissolved in water.

Urea ammonium nitrate (UAN) is the most commonly used N solution. It is manufactured by mixing hot solutions of urea and ammonium nitrate.^[1]

Ammonium polyphosphates are the major source of P for solutions. They are manufactured by reacting wet process superphosphoric acid containing 30–31% P (68–70% phosphorus pentoxide (P₂O₅)), ammonia, and water.^[1]

Solutions can be applied on their own or as mixtures and with added K, normally as potassium chloride. Fertilizer solutions may also be applied in low concentrations through irrigation water or as foliar sprays.^[2] Potassium nitrate is commonly used in foliar fertilizer sprays.

Suspensions

More concentrated fluid fertilizers can be prepared if insoluble or less soluble components can be held in suspension. This is particularly important for mixtures containing K and when micronutrients are required in the mixture. Insecticides, fungicides, and herbicides can also be incorporated as suspensions.

Ammonium polyphosphates are widely used in suspensions, but cheaper phosphate fertilizers such as monoammonium phosphate (MAP), diammonium phosphate (DAP), and merchant grade phosphoric acid containing 24% P (54% P₂O₅) may also be used as sources of P.^[1]

The N content of a suspension may be sourced from ammonium polyphosphates, UAN solutions, and anhydrous ammonia. The K source is normally potassium chloride. Micronutrients can be dispersed with greater uniformity in suspensions compared to dry blends.

Suspensions are typically formulated in blending vessels fitted with high-speed agitators to disperse finely divided, insoluble or partially soluble fertilizers in a liquid fertilizer solution. Suspending agents such as clays or xanthan gums are used to assist in maintaining the solids in suspension.^[3]

Suspensions will settle out if not agitated and must be used soon after formulation. Their use is largely confined to markets where there is a high demand in a relatively concentrated cropping area.

Solid Fertilizers

The majority of fertilizers are purchased and used as solid products. Products referred to as a “straight” fertilizer contain only one primary nutrient. A “compound” fertilizer contains two or more primary nutrients obtained chemically or by blending or both. Fertilizers containing two or more primary nutrients may be referred to as a blended fertilizer; complex fertilizer; mixed fertilizer; and N, P, and K fertilizer.^[1]

N Fertilizers

Urea is the world’s most commonly traded and used N fertilizer.^[4] It is manufactured by reacting ammonia and carbon dioxide at high temperatures and pressures^[5] and is produced as either a granule or a prill. It has good storage

and handling characteristics and is compatible in blends with most other commonly used fertilizers but is not compatible with superphosphates or in dry blends with ammonium nitrate.^[6]

Ammonium nitrate is the most popular form of N fertilizer in most European countries.^[1] It is manufactured by reacting nitric acid and ammonia.^[5] It is a strong oxidizing agent and may explode under certain conditions. Because of its hazardous properties, it is subject to a range of transport and storage regulations and its application as a straight fertilizer is prohibited in some countries.^[1]

Calcium ammonium nitrate (CAN) is used as a substitute for ammonium nitrate. It is manufactured by mixing a concentrated ammonium nitrate solution with a Ca source such as limestone.^[1]

Ammonium sulfate (sulfate of ammonia) is mainly sourced as a by-product from industry. It is also produced in granulation plants from sulfuric acid and ammonia.^[1] In addition to N, it contains 24% S in the sulfate form, which is readily available for plant root uptake.

P Fertilizers

DAP is the most commonly traded and used P fertilizer.^[4]

MAP is often used in preference to DAP in crops where the planting fertilizer and seed are placed together in the soil, through the same delivery hose behind a tine. Seedling injury is seldom observed with MAP.^[7]

DAP and MAP are produced in granulation plants by reacting ammonia and phosphoric acid.^[5]

Triple superphosphate is a commonly used P fertilizer in crops. It is produced by reacting phosphoric acid with phosphate rock.^[5]

Single superphosphate is used mainly as a pasture fertilizer. It contains 11–12% S in the sulfate form. It is produced by reacting sulfuric acid with phosphate rock.^[5]

K Fertilizers

Potassium chloride (muriate of potash) is the most commonly used K fertilizer.^[4] It is mined mainly from deposits of sylvinite and sylvite and then refined to fertilizer by crystallization or floatation processes.^[1] The other source is from brines in lakes or inland seas. It cannot be used on certain chloride-sensitive crops.^[2]

Potassium sulfate (sulfate of potash) is used in place of potassium chloride on chloride-sensitive crops. It is usually produced from brines.^[1]

Potassium nitrate is produced from an ore containing sodium nitrate, potassium nitrate, chlorides, and sulfates or from the reaction of potassium chloride with nitric acid.^[1] It is usually the preferred source of K in foliar sprays.

Potassium magnesium sulfate (sulfate of potash magnesia) is used where both Mg and S are required and for use on chloride-sensitive crops. It is produced from the mineral langbeinite.^[2]

Solid fertilizer compounds and blends

For many users, a manufactured compound fertilizer or a blend of two or more fertilizer products is the most cost-effective means of applying a combination of nutrients in ratios matched to the requirements of the soil and crop.

Compound Fertilizer

These are produced by one of the three major processes.

Chemical reaction—This is based on the granulation of solid raw materials such as ammonium sulfate, ammonium phosphate, and potash with liquid phase materials involving a chemical reaction between the components. The liquid phase is typically introduced from the reaction of ammonia with phosphoric acid in a process prior to introduction to the granulator. Granules are formed by agglomeration and the granules are subsequently dried.^[1]

Steam/water granulation—Solid and relatively finely divided raw materials are premixed and introduced into a granulator where steam and water are added. This results in the agglomeration of the various components into granules which are subsequently dried.^[1]

Compaction—Finely divided raw materials are preblended and mechanically agglomerated by high pressure into thin sheets of dried material on a revolving drum. The sheets are subsequently crushed to the desired particle size.^[1]

Secondary Nutrients and Micronutrients

The incorporation of secondary and micronutrients can be achieved by incorporating the nutrient sources in the solid phase prior to granulation or by coating them on the surface of a granular fertilizer. Secondary nutrients can also be introduced through choice of raw materials containing both primary and secondary nutrients, e.g., using ammonium sulfate as a source of both N and S.

A number of coating technologies are in commercial use that enable micronutrients and other additives to be evenly dispersed over the surface of granular fertilizers.

Slow and Controlled Release Fertilizers

The principal process for manufacture of controlled release fertilizer is by coating conventional soluble granules or prills with a protective coating or encapsulation.^[8] They are mainly used in high-value horticulture, in ornamentals, and on sporting ground turf.

Blended Fertilizers

These are a mixture or blend of fertilizers prepared without a chemical reaction from components that retain their physical form. They are usually prepared shortly before use and

supplied to users in bulk. DAP and potassium chloride are widely used in blends.

To achieve a uniform distribution of nutrients throughout the blend, it is essential for the components to be compatible and to be closely matched in size range and in size distribution.^[1]

Fertilizers absorb moisture causing caking and agglomeration of the particles. A mixture of two products will absorb moisture more readily than either of the individual products alone. This phenomenon and other aspects of the physical quality of a fertilizer such as moisture content and the presence of impurities and fine particles (fines) may limit the compatibility of blending materials.^[6] Absorption of moisture increases if the product components have a high content of fines.

Bulk dry blends will tend to segregate during transfer and handling operations. The degree of segregation will depend on the size and shape of the individual blending ingredients. The smaller particles will tend to move downward through the product while being transferred by conveyors or as a result of vibration when transported in bulk or bags.

Secondary nutrients such as Ca, Mg, and S can be introduced into the blend by choosing a blending ingredient containing the nutrient, e.g., Mg and S can be supplied by choosing potassium magnesium sulfate as the K source.

It is much more difficult to achieve an even distribution of micronutrients in dry blended fertilizers because of the much smaller amounts required. To overcome these limitations, it is often necessary to use micronutrients that have been bulked up with fillers and granulated.

CONCLUSION

The availability of the major fertilizer types has changed little in the past 50–60 years. Most of the innovative changes have occurred in production techniques, in the development of fluid forms of fertilizer, and in the range of different fertilizer formulations.

Reducing the environmental impact of fertilizers by the precise application of nutrients required and reduction of

nutrient losses from the point of application will become increasingly important as agricultural production intensifies. These factors will significantly influence the development of fertilizer types and formulations.

ACKNOWLEDGMENT

The author thanks Mr. G.M. Kuhn, Incitec Pivot Fertilizers Ltd for providing valuable suggestions for the structure and content of this entry.

REFERENCES

1. United Nations Industrial Development Organisation (UNIDO), International Fertilizer Development Center (IFDC). *Fertilizer Manual*, 3rd Ed.; Kluwer Academic Publishers: Dordrecht, 1998.
2. Fertilizer Industry Federation of Australia, Inc. *Australian Soil Fertility Manual*, 3rd Ed.; CSIRO Publishing: Melbourne, 2006.
3. Wolf, B.; Fleming, J.; Batchelor, J. *Fluid Fertilizer Manual*; National Fertilizer Solutions Association: Illinois, 1985; Vol. 2, Chapters 8 and 9.
4. International Fertilizer Industry Association. *Fertilizer Indicators May 2013*, 3rd Ed.; IFA: Paris, 2013.
5. United Nations Environment Programme. *Industry and Environment. The Fertilizer Industry's Manufacturing Processes and Environmental Issues*, 1st Ed.; UNEP: Paris, 1998.
6. European Fertilizer Manufacturers Association. *Guidance for the Compatibility of Fertilizer Blending Materials*; EFMA: Brussels, 2006.
7. Havlin, J.L.; Beaton, J.D.; Tisdale, S.L.; Nelson, W.L. *Soil Fertility and Fertilizers: An Introduction to Nutrient Management*, 6th Ed.; Prentice Hall: Upper Saddle River, 1999.
8. Trenkel, M.E. *Slow- and Controlled-Release and Stabilized Fertilizers: An Option for Enhancing Nutrient Efficiency in Agriculture*, 2nd Ed.; International Fertilizer Industry Association (IFA): Paris, France, Oct 2010.

Fertilizers: Urban Waste and Sludges

Rory O. Maguire

Virginia Polytechnic Institute and State University, Blacksburg, Virginia, U.S.A.

James Thomas Sims

*Institute of Soil and Environmental Quality and Delaware Water Resources Center,
University of Delaware, Newark, Delaware, U.S.A.*

Abstract

Environmental quality is a major issue in many parts of the world and includes important topics such as sustainability and environmental protection. Sustainability and environmental protection often overlap and cover issues such as soil conservation, careful use of finite resources such as inorganic fertilizers, and the responsible recycling of urban wastes and animal manures by agriculture. The beneficial reuse of urban wastes and by-products, e.g., sewage sludge (biosolids), has become an increasingly important issue to attain sustainability as the world's population increases. This entry describes the uses of urban wastes and biosolids as fertilizers primarily for agricultural settings and the major trends, issues, options, and regulations involved. The main focus is on biosolids, as they constitute the largest portion of urban wastes that are land applied.

TYPES OF URBAN WASTES THAT ARE USED AS FERTILIZERS

Legislation in most countries requires treatment of wastewater from combined residential, commercial, and industrial sources. Treatment of wastewater produces a semisolid by-product that is commonly known as “sewage sludge” or “biosolids.” There are several disposal pathways for biosolids including incineration, landfilling, and application to land as a fertilizer. Use of biosolids in agriculture is a well-established and regulated process in many parts of the world, including the United States and Europe. To a lesser extent, other urban and industrial by-products, e.g., paper waste, and municipal solid waste (MSW), such as leaves, grass clippings, or other organic materials, are also applied to agricultural land, often following composting. Sometimes biosolids can be cocomposted with industrial by-products or MSW.

PRODUCTION OF URBAN WASTES USED AS FERTILIZER

Biosolids Production and Quality

There are several treatment processes that are routinely carried out in wastewater treatment plants. These determine the quality and properties of the biosolids produced, which in turn affects the options for land application. Following initial screening, wastewater can undergo primary, secondary, and tertiary treatments (Table 1). The treatment

used at any particular wastewater treatment plant depends on the effluent discharge limits [e.g., biological oxygen demand (BOD) or nutrient content of the treated wastewater to be discharged into surface waters] and the proposed use for the biosolids and resources available. In addition to these wastewater treatment processes, de-watering of biosolids is frequently used to reduce biosolids volume, which can in turn reduce transportation and disposal costs.

Composting of Biosolids and MSW

Composting (often called cocomposting when two or more materials are composted together) is the decomposition of organic matter by microorganisms in a controlled environment that has optimum moisture and oxygen contents. The increase in temperature during composting can destroy most pathogens. Composting of biosolids or the organic fraction of MSW may cause offensive odors and odor control systems, such as scrubbers and biofilters, are usually required in populated areas. Composting of biosolids involves mixing de-watered biosolids with a bulking agent, such as MSW, wood chips, or straw, followed by aerobic decomposition. Only a small proportion of biosolids and MSW are composted worldwide, but in certain areas composting accounts for a large proportion of the urban wastes generated. For example, Edmonton, Canada, is able to divert 70% of its residential waste from landfill using recycling and cocomposting.^[1] Where the organic fraction of MSW is composted, it can either be source separated by the

Table 1 Wastewater treatment processes.

Treatment name	Treatment process
Screening and grit removal	Generally, a combination of sedimentation and passing through a large screen to remove stones, grit, and any large debris, such as branches, that may have entered the sewage system. The product of this process is nearly always regarded as a solid waste rather than biosolids and is usually landfilled
Primary	Follows screening and grit removal and usually involves sedimentation to remove suspended solids prior to secondary treatment
Secondary	Normally, a biological treatment in which micro-organisms are used to reduce suspended solids and BOD, thereby eliminating fish kills when the wastewater is discharged. This is the minimum wastewater treatment process required in the United States
Tertiary	Used where higher standards are required for effluent quality or to improve biosolids quality for land application. Examples include addition of lime for pathogen and odor control, iron or aluminum salts for precipitation of P, and polymers for removal of suspended sediments

resident or screened and separated from regular residential garbage, and composted alone or mixed with biosolids before being composted.

REGULATIONS COVERING LAND APPLICATION OF URBAN WASTES

Land application of biosolids normally depends on the wastewater treatment process used to produce the biosolids. As specific rules vary between countries, it is impossible to describe about all of them in this entry.^[2,3] However, it is possible to make some generalizations about land application of biosolids. For example, secondary and/or tertiary treatment, mainly to control pathogens and odors, is normally required before land application is permitted.

In United States, biosolids applications are governed by Title 40 of the Code of Federal Regulations, Part 503, commonly referred to as the “503 rule,” which sets limits for toxic metals in biosolids and for both annual and cumulative loadings to land (Table 2). Limits for chromium and molybdenum are under consideration. The 503 rule also classifies biosolids as either “Class A” or “Class B” for pathogen control. The content of polychlorinated biphenyls in biosolids to be land applied is limited to a maximum of 50 mg/kg, under Title 40 of the Code of Federal Regulations, Part 761. Similar rules are in effect in the European Community, under Council Directive 86/278/EEC, implemented in 1986 and under revision.

Table 2 Pollutant limits set by the 503 rule for toxic metals in biosolids applications to land.

Pollutant	Ceiling concentration limits for all biosolids applied to land ^a (mg/kg)	Cumulative pollutant loading rate limits (kg/ha)	Annual pollutant loading rate limits (kg/ha/yr)
Arsenic	75	41	2
Cadmium	85	39	1.9
Copper	4300	1500	75
Lead	840	300	15
Mercury	57	17	0.85
Nickel	420	420	21
Selenium	100	100	5
Zinc	7500	2800	140

^aMaximum concentration of pollutant permitted in any biosolids to be land applied.

AGRICULTURAL MANAGEMENT OF URBAN WASTES AS FERTILIZERS

The benefits of biosolids and compost applications to soil quality are many and well documented. Biosolids and composts are good sources of nitrogen (N), phosphorus (P), and potassium (K). Typical biosolids contain 4.0%, 2.0%, and 0.4% of N, P, and K, respectively, on a dry weight basis, while the equivalent values for MSWs are 0.7%, 0.2%, and 0.3%, respectively. Composted biosolids contain less total N than uncomposted biosolids, owing to the addition of other materials to aid composting and loss of ammonia during the composting process. The N in composted biosolids is released more slowly, which decreases nitrate leaching. Thus, the N is available to plants over a longer period, which is more consistent with crop N uptake patterns.^[2] Biosolids and composts can also be good sources of Ca, Mg, and S, and of the micronutrients such as Fe, Cu, B, Zn, Mn, and Mo. Biosolids and composts can promote beneficial microbial activity and diversity that suppress plant diseases and the need for costly pesticides. Biosolids and composts can also improve water infiltration, water retention, and soil structure, which in turn increases resistance to wind and water erosion. In United States, in 1998, land application of biosolids accounted for 41% of the total produced, while advanced treatments such as composting accounted for 12%.^[2] The annual production of biosolids (7 million Mg/yr) is small in comparison with animal manure production (174 million Mg/yr) in United States. However, the land application of biosolids constitutes a significant economic saving for many biosolids producers and saves landfill space, which is an increasingly expensive, finite resource in many areas.

POTENTIAL PROBLEMS ASSOCIATED WITH LAND APPLICATION OF BIOSOLIDS

Legitimate concerns about the need to prevent toxic metals and pathogens from affecting human and ecosystem health have been addressed through regulation and record keeping of applications of biosolids to land. However, there is a debate as to whether the limits set for toxic metals and pathogens are strict enough. Some scientists think that metal bioavailability will increase as the organic matter added with the biosolids is mineralized, while others argue that the evidence to support this hypothesis is inconclusive.

Biosolids and composts have a low N/P ratio compared to crop requirements. Biosolids applications are generally carried out according to N-based nutrient management plans that over-apply P and can lead to a buildup of P in agricultural soils.^[4] Buildup of soil P in many areas has been linked to increase in losses of P from agriculture to surface waters, with a corresponding decrease in water quality. Biosolids may have to be applied according to P-based nutrient management plans. This will decrease the amount of biosolids that can be applied per unit area of land, introduce the necessity for inorganic N fertilizers, and increase other associated costs. Nuisance issues such as odors and attraction of pests are of public concern when application sites are close to residential areas.^[5]

CONCLUSION

Efforts to recycle municipal wastes will likely continue, driven by human population growth, the continuing rise in cost of landfill, and the goal of sustainability in agriculture and the environment as a whole. However, the land-application programs for biosolids are uncertain. For

example, the USEPA has forecast an increase in the beneficial use of biosolids in land-application programs because of increasing costs associated with landfill and an increase in biosolids quality owing to stricter regulations covering biosolids production. Despite increasing biosolids quality, the argument over the long-term environmental impact of pollutants in biosolids will likely continue. Further regulation of the application of P to agricultural land may also increase costs associated with land-application programs for biosolids and municipal wastes. However, if the potential negative side effects associated with the land application of biosolids and composted waste are properly managed, then the beneficial use of these by-products will continue.

REFERENCES

1. City of Edmonton, Alberta, Canada. Available at www.gov.edmonton.ab.ca.
2. U.S. Environmental Protection Agency. *Biosolids Generation, Use, and Disposal in the United States*; EPA530-R-99-009; EPA Municipal and Industrial Solid Waste Division: Washington, 1999.
3. Commission of the European Communities. Council directive on the protection of the environment, and in particular soil, when sewage sludge is used in agriculture (86/278/EEC). *Off. J. Eur. Comm.* **1986**, *181*, 6–12.
4. Maguire, R.O.; Sims, J.T.; Coale, F.J. Phosphorus solubility in biosolids-amended farm soils in the mid-Atlantic region of the U.S. *J. Environ. Qual.* **2000**, *29* (4), 1225–1233.
5. Sims, J.T.; Pierzynski, G.M. Assessing the impacts of agricultural, municipal, and industrial by-products on soil quality. In *Land Application of Agricultural, Industrial, and Municipal By-Products*; Power, J.F., Warren, A.D., Eds.; Soil Science Society of America: Madison, 2000; 237–261.

Fire and Soils

Partap K. Khanna

*Commonwealth Scientific and Industrial Research Organisation (CSIRO) (Retired),
Canberra, Australian Capital Territory, Australia*

John Raison

Mullion Group, Canberra, Australian Capital Territory, Australia

Abstract

Natural or man-made vegetation fires lead to major changes in soil properties and nutrient cycling processes. Through their direct and indirect effects, fires can significantly impact on land productivity, plant diversity, and greenhouse gas emissions. The level of fire impact is highly dependent on the fire regime (intensity, frequency, and season of burn), soil type (especially erodibility), and postfire weather conditions. Fire impacts nutrient cycling in ecosystems via the effects of soil heating, ash additions, altered microclimate, and changed vegetation dynamics. Key mechanisms are: 1) direct or indirect losses of nutrients during and after fires; 2) transformation of some nutrients from organic into inorganic forms; 3) heating of the soil which initiates a number of chemical and biotic processes affecting decomposer activity and nutrient uptake by vegetation; 4) changed vegetation composition, structure, and growth rate, and thus the uptake and turnover of nutrients; 5) changed chemical environment of plant roots (pH, exchangeable cations, soil solution composition, solubility, and composition of minerals); 6) changed hydrology and temperature regimes which affect other soil processes; and 7) changes in vegetation succession (e.g., stimulation of nitrogen-fixing understory vegetation).

INTRODUCTION

Fires have played an integral role in the evolution and ecology of many ecosystems of the world. Natural and managed fires burn extensive areas annually and shape the landscape through effects on vegetation, soil, water, biodiversity, and socioeconomics. Fire has been actively managed for human benefit for millennia. In the tropics, slash-and-burn agriculture uses fire to release many of the nutrients accumulated in vegetation during a fallow period. Similarly, fire is widely used to remove forest and agricultural residues prior to establishment of a new crop or to enhance grazing.

Fire is a dominant driver of the historical and vegetation dynamics in forests, woodlands, shrublands, and grasslands in many parts of the world. Fire management practices (suppression, prescribed burning, etc.) remain controversial in many parts of the world because they can affect soil and a range of other factors with environmental values, including greenhouse gas (GHG) emissions [carbon dioxide (CO₂), carbon monoxide (CO), methane (CH₄), and nitrogen dioxide (NO₂)] and air quality.

FIRE REGIMES AND FIRE IMPACTS ON SOILS

Fire can affect soil properties and processes both directly, as a result of combustion (via nutrient transfers to the

atmosphere, ash and char inputs, and soil heating), and subsequently as a result of a myriad of changes to ecosystem processes such as changed mineralization rates of soil organic matter and litter, erosion of ash and nutrient-rich surface soil, vegetation succession, and changes to nitrogen (N)-fixing systems.^[1–3] The scale of change ranges from minutes (soil heating) to many decades (vegetation succession), with the postfire impacts on soils often being the most dominant.

Intensity and frequency of fires are major factors affecting the change in soils. The type and quantity of fuel consumed, and thus the duration of soil heating can be used to categorize fires in terms of impacts on soil and ecosystem processes (Table 1).

ATMOSPHERIC LOSSES OF ELEMENTS

During vegetation fires, atmospheric losses of elements occur in non-particulate [e.g., carbon (C), N, sulfur (S), and phosphorus (P)] and particulate forms [calcium (Ca), magnesium (Mg), potassium (K), and P], and postfire residues (partially burnt fuels, ash, and char components) are deposited on the soil. Depending upon the degree of combustion, a range of GHG gases are formed, the most important ones being CO₂, CO, nitrous oxide (N₂O), and CH₄.

Elements undergoing non-particulate (gaseous) losses would undergo a long-range transport in the atmosphere

Table 1 Fire categories as defined by fuel type, fuel quantity, and the duration of soil heating.

Fire category	Fuel types	Fuel consumption (t/ha)	Maximum temperature (°C) at 2-cm depth	Time (hr) for which soil is heated above 80% of the maximum temperature	Frequency of occurrence (yr)	Relative impact on soils
Windrows	Mostly trunks and slash	100–400+	500	>24	30–100+	High
Forest regeneration/slash	Tree crowns, some logs, shrubs, and litter	50–300+	200	0.5–2	30–100+	High
Forest wildfire	Litter, shrubs, and crowns	20–60	100	0.1–1	20–100+	Medium to high
Shrubland	Litter and crowns	15–30	80	0.1–0.5	20+	Medium to low
Grassland, crop residue	Litter, crop residues	0.5–5.0	60	0.1–0.3	1+	Low

Note: Values are a guide, with considerable variation observed.

Source: From Walker, Raison, et al.^[6]

(considered to be a complete loss to ecosystems), whereas particulates may be carried for only shorter distances. Raison, Khanna, and Woods^[4] used the conservation of Ca to distinguish between particulate and non-particulate losses of different elements and reported that a direct correlation existed, providing the possibility of estimating accurately the loss of N from N content of the fuel and the amount of dry matter burnt. Up to 60% of P contained in the fuel may be lost, depending on factors such as the temperature, forms of P in the fuel, cation content of the ash, and the amount of ash transport. The non-particulate transfer of P may range from 30% (low combustion) to 50% (high combustion). When combustion is relatively complete (gray ash is produced), non-particulate losses of many elements may account for 60–80% of the total atmospheric transfer.^[4]

EFFECTS OF SOIL HEATING

Less than 10% of the heat produced during a vegetation fire is radiated downward, yet this heating is responsible for much of the direct changes in soil properties caused by forest fire.^[5] The direct effects of heating on organic matter and soils range from mild sterilization and denaturing of protein at 50–60°C to changes to clay minerals at 950°C.^[6] Depending upon the degree of oxidation, carbonized products are produced, which range from mineral gray ash containing very little residual C to black residue with a high amount of C and charred substances. Moderate heating can render soil organic matter more prone to microbial respiration under postfire conditions.^[6] Soil heating may be as important as the addition of ash in slash-and-burn systems and can increase mineral N and P fractions.^[7]

EFFECTS OF ASH ADDITIONS

After fire, postfire residues deposit highly aromatic (char) C. About 1–5% of burnt fuel is converted to char, and this is

considered to be relatively inert and likely to be long-lived in soils^[8] and sediments.^[3]

Elements, except C and N, are enriched when ash is formed, with the level of their enrichment dependent on the degree of combustion and initial fuel characteristics. In comparison with unburnt eucalypt fuels, concentrations of Ca, Mg, and P increased by 10- to 50-fold, 10- to 35-fold, and 10-fold, respectively,^[9] with high values in gray ash. As ash is prone to be transported by wind and water, any loss of ash can cause major losses of elements from ecosystems. Ash is highly alkaline and will increase the pH of the soil and its capacity to buffer protons. The salt content of ash may initially inhibit seedling growth, but plants growing around an ashbed benefit from the nutrients made available in the short and long term, causing the so-called “ashbed effect.” Ash can also stimulate soil biological activity^[10] and interact positively with the effects of soil heating.^[2,6]

CHANGES IN SOIL PHYSICAL PROCESSES

Fire affects water penetration and erosion susceptibility by creating hydrophobicity^[11] in the surface soil. Erosion has major effects on soil fertility. Blackening of the soil surface and removal of vegetation and litter can increase soil temperature. Where fire reduces the vegetation cover, frost damage may increase, plant water uptake is reduced, thereby increasing the soil wetness, and, sometimes, rates of mineralization of soil organic matter and leaching of nutrients.

CHANGES IN SOIL CHEMICAL PROPERTIES AND PROCESSES

Large quantities of cations (Ca, Mg, K, and ammonium (NH₄)), anions (chlorine (Cl) and sulfate (SO₄)), and soluble silica are mobilized in surface soils by vegetation fire, especially under ashbeds^[12] where Ca remained the main

cation in the solution phase of surface soil during a 3-year study period. Increase in exchange capacity (because of the pH change in acid soils) and the exchangeable base cations occurs after fire and may remain so for many years after intense fire. Nitrification rates and hence the potential for leaching of nitrate and cations may be increased after intense fire.^[1,3,13] A small fraction (about 10%) of total P in ash may be soluble in water.^[10] Protons are needed to mobilize P deposited in ash and, therefore, mixing of ash with acid soil is required before P can be used by postburn vegetation. The P-sorption capacity of soils usually increases after intense forest fire.

CHANGES IN BIOLOGICAL PROPERTIES OF SOILS

The effects of fire on biological processes are complex and can be very site dependent. Fire often enhances the decomposition of organic matter and the mineralization of organically bound elements. This can be because of the changed microbial populations, altered substrate quality (additional soluble C), and more favorable environmental factors (temperature, moisture, and pH). For example, in laboratory experiments where ash was added to different soils, enhanced respiration rates, especially in soils with high organic matter content, were observed during 6 weeks of incubation^[10] until the easily mineralizable soil C was respired.^[13] N mineralization rates and the amount of nitrate produced usually increase following a fire, and in some cases may lead to loss of N by denitrification.^[10] A change in microbial population (autotrophic nitrifiers replacing heterotrophic nitrifiers) after fire has been proposed by Bauhus, Khanna, and Raison.^[13] Study of denitrification rates after fire deserves greater research attention.

CHANGES IN NUTRIENT CYCLING

Selection of appropriate soil parameters is needed for successful study of the impacts of fire on nutrient cycling. Often the selected soil parameters are either insensitive to small changes which may be caused by fire or studies are not of sufficient duration. The objective in most studies should be to ensure that the active biological sites and labile nutrient fractions in surface soils are adequately sampled. Care must be taken to avoid treatments to field soils prior to analysis that could mask the effects of fire that are occurring in the field. For example, disturbed (mixed or dried) soil samples are moistened to a fixed level and incubated at a fixed temperature regime to study the effects of fire on N mineralization, the results may be very different to those taking place *in situ*.^[14]

For forest ecosystems, fire may pose a significant risk to the maintenance of nutrient cycles in the following circumstances:^[15]

1. Where forests are already nutrient limited and where natural inputs of nutrients, especially of N and P, are small.
2. Where a high proportion of site nutrient pools are held in combustible vegetation or dead organic matter.
3. Where fuel mass (and fuel nutrient content) reaccumulates rapidly after burning, and there is frequent fire. Where fire intensity and the mass of fuel combusted are high.
4. Where fire induces increases in erosion of ash and/or surface soil (typically when there are erodible soils, steep slopes, and high rain intensities).
5. Where fire reduces rates of N fixation (e.g., by changing the composition of understory vegetation).
6. Where the frequency of fire is increased compared to historical rates due to climate change, population pressure, or changed fire management policies.

MANAGEMENT OF SOILS FOLLOWING FIRE

One of the immediate concerns following vegetation fires is minimizing soil erosion and replenishing nutrients that are lost by atmospheric transfer or erosion. Losses of N can be large (several hundred kg/ha) in forests that are subjected to wildfires or slash burns. In natural systems, given sufficient time, N-fixing processes can maintain N balance. Replenishment of lost P and cations may take decades or centuries, so that plant productivity can be reduced if such fires occur too frequently. In managed systems, fertilizer inputs may be needed to maintain productivity.

REFERENCES

1. Raison, R.J.; O'Connell, A.M.; Khanna, P.K.; Keith, H. Effects of repeated fires on nitrogen and phosphorus budgets and cycling processes in forest ecosystems. In *Fire in Mediterranean Ecosystems, Report No. 5, Ecosystem Research Series*; Trabaud, L., Prodon, R., Eds.; Commission of the European Communities: Brussels, 1993; 347–363.
2. Raison, R.J. Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: A review. *Plant Soil*. **1979**, *51*, 73–108.
3. Forbes, M.S.; Raison, R.J.; Skjemstad, J.O. Formation, transformation and transport of black carbon (charcoal) in terrestrial and aquatic ecosystems. *Sci. Total Environ*. **2006**, *370* (1), 190–206.
4. Raison, R.J.; Khanna, P.K.; Woods, P.V. Mechanisms of element transfer to the atmosphere during vegetation fires. *Can. J. For. Res.* **1985**, *15* (1), 132–140.
5. Neary, D.G.; Klopatek, C.C.; DeBano, L.F.; Folliott, P.F. Fire effects on belowground sustainability: A review and synthesis. *Forest Ecol. Manage.* **1999**, *122* (1–2), 51–71.
6. Walker, J.; Raison, R.J.; Khanna, P.K. Fire. In *Australian Soils—The Human Impact*; Russell, J.S., Isbell, R.F., Eds.; Queensland University Press: Brisbane, 1986; 186–216.

7. Giardina, C.P.; Sanford, R.L.; Dockersmith, I.C. Changes in soil phosphorus and nitrogen during slash-and-burn clearing of a dry tropical forests. *Soil Sci. Soc. Am. J.* **2000**, *64*, 399–405.
8. Skjemstad, J.O.; Taylor, J.A.; Smernik, R.J. Estimation of charcoal (char) in soil. *Commun. Soil Sci. Plan.* **1999**, *30* (15–16), 2283–2298.
9. Raison, R.J.; Khanna, P.K.; Woods, P.V. Transfer of elements to the atmosphere during low-intensity prescribed fires in three Australian subalpine eucalypt forests. *Can. J. Forest Res.* **1985**, *15* (4), 657–664.
10. Khanna, P.K.; Raison, R.J.; Falkner, R.A. Chemical properties of ash components derived from Eucalyptus litter and its effects on forest soils. *For. Ecol. Manage.* **1994**, *66*, 107–125.
11. Doerr, S.H.; Shakesby, R.A.; MacDonald, L.H. Soil water repellency: A key factor in post-fire erosion. In *Fire Effects on Soils and Restoration Strategies*; Cerda, A., Robichaud, P.R., Eds.; Science Publishers: Enfield, 2009; 589.
12. Khanna, P.K.; Raison, R.J. Effects of fire intensity on solution chemistry of surface soil under a Eucalyptus pauciflor-aforest. *Aust. J. Soil Res.* **1986**, *24*, 423–434.
13. Bauhus, J.; Khanna, P.K.; Raison, R.J. The effect of fire on carbon and nitrogen mineralization and nitrification in an Australian forest soil. *Aust. J. Soil Res.* **1993**, *31*, 621–639.
14. Khanna, P.K.; Raison, R.J. *In situ* coring techniques for estimating soil mineral-N fluxes: Re-evaluation based on 25 years of application and experience. *Soil Biol. Biochem.* **2013**, *64*, 203–210.
15. Raison, R.J.; Khanna, P.K.; Romanya, J.; Serrasolses, I. Effect of fire on forest nutrient cycles. In *Fire Effects on Soils and Restoration Strategies*; Cerda, A., Robichaud, P., Eds.; Science Publishers: Enfield, 2009; 225–256.

Fluid Flow: Modeling

John L. Nieber

*Department of Biosystems and Agricultural Engineering, University of Minnesota,
St. Paul, Minnesota, U.S.A.*

Abstract

The unsaturated zone (vadose zone) of the earth's crust is an important interface for both the underlying groundwater and the overlying surface water resources and atmosphere. Quantifying fluid flow, mass transport, and energy transport processes in the unsaturated zone have become a focus for researchers, government agencies, and consultants because it is found that the outcomes of these processes have an impact on the sustainability of modern social structures. While sophisticated sensors and instrumentation seem to have been developed to provide data from the field on a real-time basis, these cannot be used to directly predict the possible outcomes. Instead, this prediction requires the use of mathematical models representing the flow and transport processes. The following describes some of the applications of models for flow in the unsaturated zone, past challenges and achievements in improving modeling methods, and future challenges to modeling flow.

APPLICATION OF MODELS

Models for simulating flow processes in soil and groundwater find applications in many environmentally oriented disciplines including geography, soil science, agricultural engineering, civil engineering, geoengineering, hydrogeology, and meteorology. These models are generally based on numerical solutions of governing equations and require the use of digital computers to complete the calculation task. There are many potential applications for such models, but a short list of the common applications includes the following: 1) the estimation of groundwater recharge volume and contaminant loading; 2) the design of measures to remediate contaminated soils and groundwater; 3) the design of efficient drainage and irrigation systems for efficient crop production; 4) the estimation of runoff production from land in response to rainfall and/or snowmelt; and 5) the assessment of the impact of global climate change on surface and subsurface water resources.

Models developed for these applications need to satisfy two criteria to be put to use by a practitioner. They have to be fairly easy to use and dependable. The first criterion, ease of use, only requires a good team of programmers and does not pose a challenge for flow modeling. The second criterion stipulates that the model provides accurate results in a timely manner. This is a direct challenge to flow modeling. The following sections present information on the past and future challenges associated with the development of dependable models.

PAST CHALLENGES AND ACHIEVEMENTS

There has been substantial progress in the development of numerical models for simulating relevant unsaturated

zone processes. During this time, predominant attention in modeling flow processes was given to the Richards' equation.^[1] In the past, many of the difficulties in model development involved the determination of the best ways to solve this equation.

Due to the highly non-linear character of the Richards' equation, analytical solutions^[2] to the equation were possible only for simplified conditions, and therefore, numerical solution methods were necessary to treat realistic field conditions. Numerical methods such as the finite difference method and the finite element method were adopted from other engineering and science applications and applied to discretize the Richards' equation into the systems of non-linear algebraic equations. Two major numerical solution problems were faced by researchers in solving this equation. One was the highly non-linear nature of the equation and associated boundary conditions, leading to problems of slow convergence or even non-convergence of the solution methods. This problem was handled by applying non-linear equation solvers classified within the broad class of Newton's methods and Picard's methods.^[3]

A second problem was the need to be able to solve large systems of algebraic equations, which in earlier years involved hundreds or perhaps thousands of equations. Due to the relatively small memory available on early computers, it was necessary to use iterative methods to solve even moderate-sized problems. Conventional iterative methods such as Jacobi, Gauss-Seidel, and successive over-relaxation methods^[4] were used to solve these problems. While these methods were satisfactory for relatively homogeneous systems, for strongly heterogeneous systems, it was found that these methods led to poor convergence or even non-convergence. Computer memory did increase substantially during the 1980s and 1990s, and this allowed the use of direct equation solvers for the types of problems

solved in the earlier years. However, with the increase of computer memory storage, the problems tackled have also increased memory requirements (tens of thousands to millions of equations need to be solved), and iterative methods are once again back in vogue. Fortunately, the efficiency of iterative methods has also increased substantially with the development of conjugate gradient methods^[5] and multi-grid methods.^[6]

There has been an effort to find the recipe for a numerical method or set of methods that will be robust for solving the Richards' equation.^[3,4,7-12] The desire has been to develop computationally efficient methods applicable to a broad range of practical problems, especially for large-scale, 3-D, heterogeneous flow systems. Associated problems involved assuring that the solutions were mass conservative and that the iterative methods used to solve the non-linear algebraic equations would converge even for conditions where the soil is very dry and highly heterogeneous. The detailed analysis of how to assure a mass conservative solution has been given in the study by Rathfelder and Abriola.^[8] Assurance of non-linear iteration convergence has been found to be more problematic, and improvements have been made using techniques involving primary variable switching,^[9,10] higher-order time integration,^[11] and variable transformation.^[12]

Aside from the obvious problems associated with solving the Richards' equation, there has been the need to assign (spatially) the equation parameters for field-scale applications of the numerical solutions. For instance, the catchment scale model of Paniconi and Wood,^[13] as the model of Freeze,^[4] was developed to facilitate the simulation of 3-D variably saturated flow over an entire catchment. While the solution of the large system of algebraic equations for such a problem offers a significant challenge to modeling, an equal if not larger challenge is the problem of assigning parameters to the cells in the numerical grid.

The problem of parameter assignment is twofold. There is the problem of determining the spatial distribution of the parameters at the scale of the numerical grid cell. Involved with this problem is the uncertainty in predicting these values, given limited field data. Geostatistical methods^[14] were developed to facilitate such prediction and to estimate the limits of uncertainty. A second problem arises because small-scale features, and even small-scale processes, occurring below the sampling scale of the field-scale grid cell can significantly influence the actual outcome at the field scale but are not directly predictable by the governing equation(s) discretized at the field-scale grid. There has been some success^[15] in developing techniques that provide parameters that account for these subgrid scale features and processes. These parameters, called *effective* parameters, are those that yield *effectively* the same outcome as would occur if the more detailed parameter distribution were used.

FUTURE CHALLENGES TO MODELING

Two of the greatest future challenges to modeling include the need to more completely describe the flow and transport processes and the need to incorporate multiscale phenomena into the modeling analysis. The first challenge involves the expansion of the governing equations to include coupled physical and chemical processes. The second challenge involves the assignment of equation parameters and the incorporation of subgrid features and processes. To assure the success of future modeling, the second challenge is the most critical to address.

Governing Equations

As awareness of environmental problems has increased, and environmental regulations have become more stringent, the scope for modeling has expanded from modeling the flow of water alone to modeling coupled multiphase fluid flows, mass (solute) transport, and energy (thermal) transport. The coupled equations for isothermal multiphase fluid flow have been reviewed in detail.^[16] Numerical solutions of coupled two-phase flow equations for applications to environmental problems have advanced considerably. One of the earlier multiphase flow solutions is given in the study by Kaluarachchi and Parker,^[17] while methods representing the advances in computational efficiency are given in the study by Bastian and Helmig.^[18]

Equations for non-isothermal conditions, which have received much less attention, have been presented in the study by Nassar and Horton.^[19] The solutions of these equations offer new challenges to those developing numerical solutions.^[20,21] Techniques such as those mentioned earlier for the solution of the Richards' equation and the coupled multiphase flow equations should be tested to improve the efficiency of such solutions.

Process Scale

Numerical solutions of flow and transport processes for field-scale applications are generally performed on relatively large numerical grids. The reason for this is twofold. First, the computational effort to simulate field-scale problems can easily exceed the capabilities of the modern computers (using modern numerical methods); if too fine, a grid is used. Second, one generally does not know what values of parameters to assign to very fine grids due to inadequate spatial resolution in field data. As a result, it is necessary to assign effective parameters to the field-scale grid cells. One approach is to use stochastic methods^[15] to derive effective parameters. Renormalization methods,^[22] which rely on numerical methods, are another means to derive effective parameters.

In some instances, the governing equations may behave differently at the small scale than at the large scale. For instance, in the case of finger flow^[23,24] or funnel

flow,^[25] the flow occurs on a scale of about 10 cm. To simulate these flow features directly, it is necessary to use relatively small grid cells. Using large grid cells without considering these small-scale processes leads to a diffuse solution, and the small-scale features are missed. Effectively capturing the dynamics of these small-scale processes into a field-scale grid requires an appropriate procedure for upscaling the governing equations from the small scale to the field-grid scale. Such a procedure has been demonstrated^[26] for viscous fingering in multiphase flow and is an area of active research.

The discussion about process scale poses the question about whether the governing equations maintain a constant form in the progression from the small scale to the large scale. Addressing this question involves the principles developed in the field of multiscale science.^[27] While these developments originated in the fields of solid mechanics and fluid mechanics, they are receiving much attention in hydrology, soil physics, and hydrogeology.

REFERENCES

- Richards, L.A. Capillary conduction of liquids through porous mediums. *J. Appl. Phys.* **1931**, *1*, 318–333.
- Philip, J.R. Steady infiltration from spheroidal cavities in isotropic and anisotropic soils. *Water Resour. Res.* **1986**, *22*, 1874–1880.
- Paniconi, C.; Putti, M. A comparison of Picard and Newton iteration in the numerical solution of multidimensional variably saturated flow problems. *Water Resour. Res.* **1994**, *30*, 3357–3374.
- Freeze, R.A. Three-dimensional, transient, saturated-unsaturated flow in a groundwater basin. *Water Resour. Res.* **1971**, *7*, 347–366.
- Saad, Y. *Iterative Methods for Sparse Linear Systems*; PWS Publ. Co.: Boston, 1996.
- Brandt, A. Multi-level adaptive solutions to boundary value problems. *Math. Comput.* **1977**, *31*, 333–390.
- Huyakorn, P.S.; Thomas, S.D.; Thompson, B.M. Techniques for making finite elements competitive in modeling flow in variably saturated media. *Water Resour. Res.* **1986**, *20*, 1099–1115.
- Rathfelder, K.; Abriola, L.M. Mass conservative numerical solutions of the head-based Richards equation. *Water Resour. Res.* **1994**, *30*, 2579–2586.
- Forsyth, P.A.; Wu, Y.S.; Pruess, K. Robust numerical methods for saturated-unsaturated flow with dry initial conditions in heterogeneous media. *Adv. Water Resour.* **1995**, *18*, 844–856.
- Diersch, H.J.G.; Perrochet, P. On the primary variable switching technique for simulating unsaturated-saturated flows. *Adv. Water Resour.* **1999**, *23*, 271–301.
- Tocci, M.D.; Kelley, C.T.; Miller, C.T. Accurate and economical solution of the pressure-head form of Richards' equation by the method of lines. *Adv. Water Resour.* **1997**, *20*, 1–14.
- Williams, G.A.; Miller, C.T. An evaluation of temporally adaptive transformation approaches for solving Richards' equation. *Adv. Water Resour.* **1999**, *22*, 831–840.
- Paniconi, C.; Wood, E.F. A detailed model for simulation of catchment scale subsurface hydrologic processes. *Water Resour. Res.* **1993**, *29*, 1601–1620.
- Webster, R. Quantitative spatial analysis of soil in the field. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer Verlag: New York, 1985; Vol. 3, 1–70.
- Mantoglou, A.; Gelhar, L.W. Stochastic modeling of large-scale transient unsaturated flow systems. *Water Resour. Res.* **1987**, *23*, 37–46.
- Miller, C.T.; Christakos, G.; Imhoff, P.T.; McBride, J.F.; Pedit, J.; Trangenstein, J.A. Multiphase flow and transport modeling in heterogeneous porous media: Challenges and approaches. *Adv. Water Resour.* **1999**, *21*, 77–120.
- Kaluvarachchi, J.J.; Parker, J.C. An efficient finite element method for modeling multiphase flow. *Water Resour. Res.* **1989**, *25*, 43–54.
- Bastian, P.; Helmig, R. Efficient fully-coupled solution techniques for two-phase flow in porous media; parallel multi-grid solution and large scale computations. *Adv. Water Resour.* **1999**, *23*, 199–216.
- Nassar, I.N.; Horton, R. Heat, water and solute transfer in unsaturated porous media: I. Theory development and transport coefficient evaluation. *Transport. Porous Med.* **1997**, *27*, 17–38.
- Thomas, H.R.; Missoum, H. Three-dimensional coupled heat, moisture, and air transfer in a deformable unsaturated soil. *Int. J. Numer. Meth. Eng.* **1999**, *44*, 919–943.
- Chounet, L.M.; Hilhorst, D.; Jouron, C.; Kelanemer, Y.; Nicolas, P. Simulation of water flow and heat transfer in soils by means of a mixed finite element method. *Adv. Water Resour.* **1999**, *22*, 445–460.
- King, P.R. The use of renormalization for calculating effective permeability. *Transport. Porous Med.* **1989**, *4*, 37–58.
- Glass, R.J.; Steenhuis, T.S.; Parlange, J.-Y. Wetting front instability as a rapid and far-reaching hydrologic process in the vadose zone. *J. Contam. Hydrol.* **1988**, *3*, 207–226.
- Nieber, J.L. Modeling finger development and persistence in initially dry porous media. *Geoderma* **1996**, *70*, 209–229.
- Ju, S.-H.; Kung, K.-J.S. Steady-state funnel flow: Its characteristics and impact on modeling. *Soil Sci. Soc. Am. J.* **1997**, *61*, 416–427.
- Blunt, M.; Christie, M. How to predict viscous fingering in three-component flow. *Transport. Porous Med.* **1993**, *12*, 207–236.
- Glimm, J.; Sharp, D.H. Multiscale science: A challenge for the twenty-first century. *SIAM News* **1997**, *30*, 4, 17, 19.

Food Crops: Zinc and Iron Biofortification

Yashbir Singh Shivay

Rajendra Prasad

Division of Agronomy, Indian Council of Agricultural Research (ICAR), New Delhi, India

Abstract

Zinc (Zn) and iron (Fe) are essential for the normal, healthy growth and reproduction of all higher plants, animals, and humans and are therefore called “essential trace elements” or “micronutrients” in plants and “micro mineral nutrients” in animals and humans. Zn and Fe deficiencies are well-documented public health issues and an important soil constraint to crop production. Generally, there is a close geographical overlap between soil deficiency and human deficiency of Zn and Fe, suggesting the need for increasing concentrations of these micronutrients in food crops. Breeding new plant genotypes for high grain concentrations of Fe and Zn (genetic biofortification) is an effective strategy to address the problem; but this strategy is a long-term process. A rapid and complementary approach is therefore required for biofortification of food crops with Zn and Fe in the short term. Agronomic biofortification/ferti-fortification using micronutrient fertilizers represents such a fast and effective strategy. Agronomic biofortification of food crops has received considerable attention globally.

INTRODUCTION

Micronutrient deficiencies in soils, plants, animals, and humans are becoming increasingly important globally. Intensive cultivation of high-yielding cultivars with heavy applications of nitrogen (N), phosphorous (P), and potassium (K) fertilizers and year after year heavy harvests without application of micronutrient fertilizers have led to micronutrient deficiencies in soils of many countries especially in high-populated Asian countries, such as India and China. The most micronutrient deficiencies reported worldwide are those of zinc (Zn) and iron (Fe). Zn as well as Fe deficiencies occur mostly in alkaline calcareous soils where chemical availability of Zn and Fe to plant roots is extremely low. It is estimated that Fe deficiency occurs in about 30% of the cultivated soils on the world and results in large decreases in crop production and quality. Zn deficiency is also widespread throughout the world and occurs in nearly all countries. Based on analysis of 298 soil samples from different countries, Zn deficiency has been found to be the most widespread micronutrient deficiency worldwide. Nearly 50% of the soils cultivated for cereal production globally have low levels of plant available Zn. In Turkey, Zn deficiency is the most widespread; the half of all cultivated soils (28 million hectares) has Zn deficiency. Zn and Fe deficiencies are prevalent, particularly in the developing world.

In many parts of the world, micronutrient deficiencies are a growing public health and socioeconomic issue and are a more widespread problem than poor dietary quality and low-energy intake, and about 20% of deaths in children under 5 years can be attributed to vitamin A, Zn, Fe, and/or

iodine deficiencies. Dietary deficiency of essential micronutrients such as Zn and Fe affects more than two billion people worldwide, especially pregnant women and children below the age of five who suffer from severe acute malnutrition.

Fe deficiency is the most common nutritional disorder in the world, and almost 1.6 billion people are suffering from its deficiency. Fe deficiency anemia results in impaired physical growth, mental development, and learning capacity. Zn deficiency is equally serious and is ranked as the 5th leading risk factor for diseases in the developing world. Numerous health problems link Zn deficiency to retarded growth, skeletal abnormalities, delayed wound healing, increased abortion risk, and diarrhea. Approximately one-third of the world's population is suffering from Zn deficiency. The situation is even more adverse in developing countries where more than half of the children and pregnant women are suffering from Fe and Zn deficiencies.

In countries with a high incidence of micronutrient deficiencies, cereal-based foods represent the largest proportion of the daily diet. The HarvestPlus initiative of the Consultative Group for International Agricultural Research (CGIAR) consortium (www.harvestplus.org) is working with national and international partners to alleviate deficiencies of these mineral nutrients by biofortifying staple food crops with essential minerals and vitamins; an approach considered to be the most economical solution to human micronutrient deficiency. The biofortification program is focusing on three micronutrients that are widely recognized by the World Health Organization as limiting. These are Fe, Zn, and vitamin A. Full-time breeding programs are under way for six staple food crops viz., rice

(*Oryza sativa*), wheat (*Triticum aestivum*), maize (*Zea mays*), cassava (*Manihot esculenta*), sweet potatoes (*Ipomoea batatas*), and common beans (*Phaseolus vulgaris*). Prebreeding feasibility studies are proposed for eleven additional staples: bananas, barley, cowpeas, groundnuts, lentils, millet, pigeon peas, plantains, potatoes, sorghum, and yams. Although plant breeding is the most sustainable solution to the problem, developing new micronutrient-rich plant genotypes is a protracted process and its effectiveness can be limited by the low amount of readily available pools of micronutrients in soil solution.

Cereal crops are inherently very low in grain Zn and Fe concentrations, and growing them on potentially Zn- and Fe-deficient soils further reduces Fe and Zn concentrations in grain. Application of Zn- and Fe-containing fertilizers (i.e., agronomic biofortification) is a rapid solution and represents a complementary approach to breeding, which needs to be taken on priority at the global level to overcome these two essential micronutrients deficiencies in the food chain.

AGRONOMIC BIOFORTIFICATION OF CEREAL GRAINS

Essentiality of Fe in plants was reported by Sachs in 1860, while that of Zn was established by Maze in 1916.^[1] Zn deficiency was later reported in citrus in the United States. A number of reviews on Zn in crop nutrition are available.^[2] In India, Zn deficiency was first reported in rice by Nene^[3] and was followed by that in wheat in Punjab. Research on Zn in relation to crop production in India has been thoroughly reviewed.^[4] However, most work on Zn fertilization was done from the viewpoint of increasing crop yield. Work on agronomic biofortification of wheat was started in Turkey by Cakmak,^[5] whereas in India, it was initiated on rice by the authors of this entry.^[6–11] Most information on biofortification of cereal grains with Zn is available on rice and wheat and is briefly reviewed.

Rice

Method of application

Zn can be applied to soil or foliage. The seed priming with Zn fertilizers and dipping of rice seedlings in Zn fertilizer solutions have also been tested and recommended for increased yield, but no data are available on their effect on biofortification of rice grains. Shivay and Prasad^[6] from New Delhi showed that on Zn-deficient soils, application of Zn as zinc sulfate heptahydrate (ZnSHH) significantly increased grain yield of rice as well as Zn concentration in rice grain.

Foliar application of only 1.2 kg Zn ha⁻¹ as compared with 5.3 kg Zn ha⁻¹ as soil application gave similar grain yield of rice, but higher Zn concentration in grain.^[12,13] Agronomic efficiency of Zn with foliar application was about 4 times of that for soil application and rate of Zn application was much lower when applied on foliage. A study from Ludhiana, India, showed that averaged on five rice cultivars foliar-applied Zn (three sprays of 0.5% ZnSHH solution) recorded a Zn concentration of 47.0 mg kg⁻¹ grain in brown rice as compared to 33.8 mg kg⁻¹ grain in no Zn check.^[14] In a multilocation study in China, India, Lao PDR, Thailand, and Turkey, Zn concentration in unhusked rice grain was about 69% higher with foliar application than with soil application; at some locations, it was almost twice that of with soil application.^[15] This study also provided data on relative Zn concentration in unhusked (whole grain with husk, known as paddy in India; most of the data on biofortification of rice are on unhusked rice), brown rice (whole caryopsis with husk removed by hand), and white rice (outer layer of pericarps is including pericarp, testa, mucella, and part of aleurone layer along with embryo removed by polishing for 30 seconds in a standard laboratory mill). White rice is also known as polished rice (the form in which rice is mostly consumed). When Zn was foliar applied, only 53–54% of that in unhusked rice was found in polished or white rice as compared with 84.8%, when Zn was soil applied (Table 1).

Table 1 Grain yield and relative zinc concentration in unhusked, brown, and white (polished) rice (averaged over nine site years in China, India, Lao PDR, Thailand, and Turkey).

Characteristic	Control (no Zn)	Soil Zn	Foliar Zn	Soil + Foliar Zn	Significance
Grain yield (t ha ⁻¹)	6.7	7.0	6.9	7.0	NS
Zn in unhusked rice (mg kg ⁻¹)	18.7	19.1	32.3	34.7	P < 0.01
Zn in brown rice (mg kg ⁻¹)	19.1	20.8	24.4	25.5	P < 0.01
	(102.1) ^a	(108.9)	(75.5)	(73.5)	
Zn in polished rice (mg kg ⁻¹)	16.1	16.2	17.7	18.4	P < 0.01
	(18.1) ^b	(84.8)	(54.8)	(53.0)	
	(84.2) ^c	(77.9)	(72.5)	(72.1)	

^aZn in brown rice expressed as percentage of unhusked rice.

^bZn in polished rice expressed as percentage of brown rice.

^cZn in polished rice expressed as percentage of unhusked rice.

Source: From Phattarakul, Rerkasem, et al.^[15] ©2012 Springer.

However, when Zn is soil applied, brown rice may contain a little more than in unhusked rice. Thus, a greater portion of foliar-applied Zn remained in husk. These data support that in rice, Zn absorbed from the root plays a major role, whereas mobilization from the leaves plays a minor role.^[16] Using the data of Phattarakul et al.^[15] as the base, it worked out that although unhusked rice contained 52.6% Zn, the polished rice from it is likely to contain only 28.8% Zn, when Zn was foliar applied in the study of Shivay and Prasad.^[6] As a contrast, when Zn was soil applied in sufficient quantity, Zn concentration in unhusked rice was 47.5%, whereas it was 40.3% in polished rice. Total Zn uptake by polished rice was also higher with soil-applied Zn; of course, much more Zn was applied to soil (25 kg ha⁻¹) as compared with that on foliage (1.2 kg ha⁻¹). Biofortification recovery efficiency ($BRE_{Zn} = 100\{[Zn \text{ concentration in polished rice (mg kg}^{-1}) \times \text{polished rice yield (t ha}^{-1}) \text{ in plus (+) Zn plots}] - [Zn \text{ concentration in polished rice (mg kg}^{-1}) - \text{polished rice yield (t ha}^{-1}) \text{ in minus (-) Zn plots}]\} \div \text{Zn applied in kg ha}^{-1}\}$), the term suggested by Impa and Johnson-Beebout,^[17] with foliar application was about eight times of that obtained with soil application. A study from Thailand reported that the decrease in Zn concentration on milling of rice ranged from 16.2% to 48.2% in rice genotypes, being more in long and slender grain types. The range of Zn (mg kg⁻¹) in polished rice was 9.6–40.2 (mean 20.6) when compared with 17.3–59.2 (mean 28.7) in brown rice.^[18]

In eastern India and in some other Asian countries, rice is parboiled before milling. Parboiling is a hydrothermal process to which unhusked rice is subjected before milling. It involves soaking in water, steaming, and drying; the degree of soaking and steaming differs considerably.^[19] As parboiled rice gives much better head rice recovery and most of the millers practiced parboiling to some degree to avoid the breakage losses. On soaking unhusked rice, nutrients from the outer layer of endosperm (pericarp, seed coat, nucellus, and aleurone layer) move into endosperm and the parboiled white rice is much richer in vitamins and minerals and has a better storage quality. Thus, the data on biofortification of rice grains depend very much on the milling process adopted. A study reported that when rice grains were soaked in Fe-ethylenediaminetetraacetic acid (EDTA) + zinc sulfate (ZnSO₄) solutions, Fe and Zn penetrated across the husk and aleurone layer into endosperm, and when parboiled, the polished rice retained 70–80.5% of Fe and Zn.^[20] Also, Fe and Zn polished rice was highly bioaccessible.

Sources of Zn

Results from New Delhi, India, reported that ZnSHH-coated urea was significantly superior to zinc oxide (ZnO)-coated urea in increasing Zn concentration in unhusked rice^[8,10–12] (also in polished rice when calculated on the basis of Phattarakul's data). The superiority of

ZnSHH was also recorded in succeeding wheat.^[8,10] Zn was applied to rice only. Water solubility of Zn sources is considered as an important criterion for Zn availability. A study from the United States reported that the effectiveness of six granulated Zn fertilizers decreased as the percent of water-soluble Zn decreased in them and calculated that at least 50% water-soluble Zn was considered desirable.^[21] In the United States, Zn fertilizer manufacturers are producing mixture of ZnSO₄ and ZnO, which are known as Zn-oxysulfates. However, from the manufacturer's point of view, ZnO is easier to coat because it forms a good emulsion with oil. Kiekens^[22] suggested that ZnO, zinc hydroxide, and zinc carbonate are about 105 times more soluble than soil Zn and these materials could be used as fertilizers.

A field study was carried out to compare ZnSHH and Zn-EDTA for rice at Pakyong, Sikkim. ZnSHH was applied at 10 and 20 kg ha⁻¹ as basal or in two equal splits (half basal and the rest half at grand tillering stage). Zn-EDTA was applied at 0.5 or 1.0 kg ha⁻¹ in single application as basal; 1 kg ha⁻¹ was also applied in two equal splits. Zn concentration in rice grain was significantly more (30.3 mg kg⁻¹) with 0.5 kg ha⁻¹ Zn-EDTA than with 10 kg ha⁻¹ ZnSHH (25.5 mg kg⁻¹). Split application was better than a single application in ZnSHH but not in Zn-EDTA. Zn-EDTA was better than ZnSHH, but more expensive.^[23]

Wheat

Soil Zn deficiency in major wheat growing areas leads to inherently low grain Zn concentration and is considered as a major factor in low human Zn intake.^[24] Soil Zn applications are less effective in increasing grain Zn concentration, while foliar Zn applications result in remarkable increases in grain Zn concentration in wheat.^[25,26] By optimizing the timing and the solute concentration of foliar Zn application, wheat grain Zn concentration could be further increased, not only in whole grains but also in the endosperm.^[26,27] Various methods of Zn application may differentially influence yield and grain Zn concentration. Knowledge of the different forms of Zn fertilizer and timing of foliar Zn application are crucial for enhancing grain Zn. The most effective method for increasing grain Zn is the soil + foliar application method, which may result in an about threefold increase in grain Zn concentration.^[25] When a high concentration of grain Zn is targeted, in addition to a high grain yield, combined soil and foliar application are recommended. Alternatively, using seeds with high Zn concentrations, together with foliar application of Zn, is also an effective way to improve both grain yield and grain Zn concentration. Applying Zn during the grain development stage contributes to increased grain Zn concentration^[27] as foliarly applied Zn can be absorbed by the leaf epidermis and then transported to other plant parts via the xylem and phloem. Field experiment results reported from Rothamsted, U.K., showed that sewage sludge application

to soil can increase Zn concentration in wheat grain in non-calcareous soils, but not in a calcareous soil for at least 2–8 years after application and was similar in effectiveness to Zn carbonate.^[28]

The timing of foliar Zn application is an important factor determining its effectiveness in increasing grain Zn concentration; large grain Zn increases are most likely when foliar Zn fertilizers are applied to plants at a late growth stage. A field study in Turkey was carried out to see the changes in grain Zn concentration in wheat during the reproductive stage and found that the highest concentration of grain Zn occurs during the milk stage of grain development. Foliar application of Zn during reproductive growth seems to be more effective in increasing grain Zn concentration than spraying of Zn at earlier growth stages. In addition to increasing the concentration of Zn in the whole grain, foliar application also increased the concentration in the starchy endosperm. Late season foliar application of Zn increased the Zn concentration in the starchy endosperm by up to threefold.^[29] Since the concentration of phytate in the starchy endosperm (i.e., white flour) of wheat is very low, or even not measurable, such an increase in Zn implies a positive effect on the use of the grain for human nutrition. The increased Zn in the starchy endosperm resulting from foliar application should also be highly bioavailable due to the low phytate content.

Among the different forms of Zn fertilizer that were tested, application of Zn as ZnSHH was most effective in increasing grain Zn, compared to other forms of Zn. The Harvest Zinc (www.harvestzinc.org) initiative has been investigating different fertilizer strategies and the most efficient Zn application method for promoting Zn uptake and maximizing grain Zn accumulation. Increasing grain Zn by soil and/or foliar applications also provides additional positive impacts in terms of seed vitality and seedling vigor. Priming seeds in Zn-containing solutions is an alternative way to increase seed Zn prior to sowing. High-seed Zn concentrations ensure good root growth and contribute to better protection against soil-borne pathogens.^[30] Preliminary studies showed that ZnSHH could be mixed with some wheat herbicides, insecticides, and fungicides without affecting the effectiveness of foliar application for increasing grain Zn concentration. This would increase the possibility that farmers may be willing to apply ZnSHH in their fields by reducing the cost and time of application (Cakmak and Ortiz-Monasterio, unpublished results). Cakmak et al.^[26] and Cakmak,^[30] using laser ablation inductively coupled plasma mass spectrometry, have reported increase in Zn concentration in endosperm by Zn sprays.

Maize

Much information on Zn and Fe agronomic biofortification in maize through fertilizers is not available in the literature. Small holder farmers in South Africa, Zimbabwe, and other

African countries use very little amounts of chemical fertilizers and use of Zn fertilizers is negligible. A study in Zimbabwe showed that the application of cattle manure (supplying 113 g Zn ha⁻¹) + NPK and leaf litter (supplying 430 g Zn ha⁻¹) + NPK significantly increased Zn concentration in corn grain over NPK.^[31] Results from Indian Agricultural Research Institute, New Delhi, India, reported that for Zn biofortification of corn grain and stover, foliar application of 1 kg ZnSO₄ ha⁻¹ (in two sprays at tasseling and initiation of flowering) or application of Zn-coated urea is better than soil application of ZnSO₄. However, highest Zn concentration of 49.2 mg kg⁻¹ corn was recorded with 5 kg Zn to soil + 1 kg Zn as foliar, which was 22.4% higher than control (no Zn).^[32] The expected increase in Zn and Fe from plant breeding is likely to be lesser in corn than in rice and wheat.

Oats (*Avena sativa*)

In a study, Indian Agricultural Research Institute, New Delhi, India, reported that coating Zn as ZnO or ZnSO₄ onto oats seeds at 2 kg Zn per 100 kg seed (required for sowing 1 ha) gave a Zn concentration of about 32 mg kg⁻¹ as compared with about 25 mg kg⁻¹ obtained with soil application. For soil application, ZnSO₄ was better than ZnO.^[33]

Chickpea (*Cicer arietinum*)

Application of Zn to soil or foliage as ZnSHH or Zn-EDTA increased Zn concentration in grain and straw of chickpea.^[34] In the case of grain, three sprays of ZnSHH or, Zn-EDTA recorded significantly more Zn in grain than soil application or one or two sprays (Figs. 1 and 2). The two sources of Zn differed significantly, when two or three sprays were made; Zn-EDTA recorded significantly higher Zn concentration in grain than ZnSHH in both the years of study. With both the sources of Zn different methods of application were in the following order: three foliar

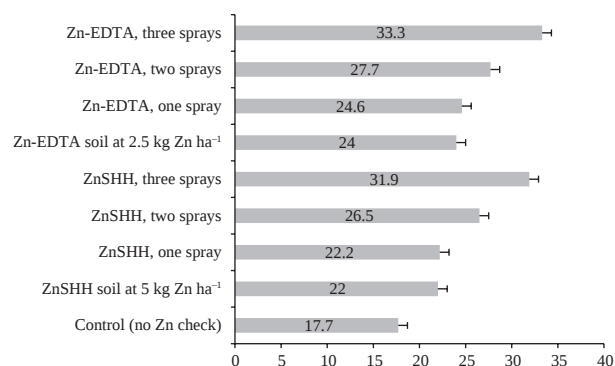


Fig. 1 Zn concentration [mg kg⁻¹ dry matter (DM)] in chickpea straw (mean of 2 years). Error bar indicates the critical difference (CD) value at 5% level of significance which is 2.66.

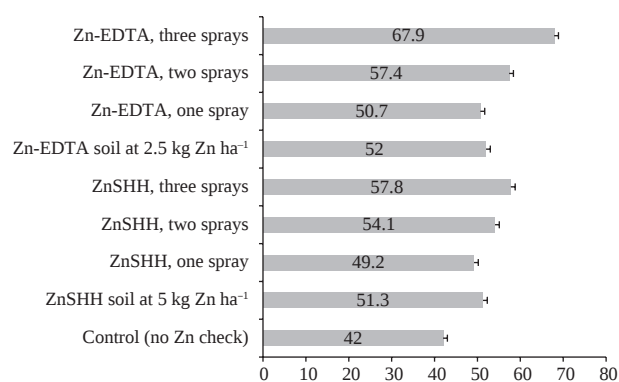


Fig. 2 Zn concentration (mg kg⁻¹ DM) in chickpea grain (mean of 2 years). Error bar indicates the CD value at 5% level of significance which is 3.32.

sprays > two foliar sprays > one foliar sprays or soil application; three foliar applications recorded the highest Zn concentration in straw. When soil applied or a single foliar application was made, Zn-EDTA recorded significantly more Zn in chickpea straw than ZnSHH straw.

CONCLUSION

Agronomic biofortification is the easiest and fastest way for biofortification of cereal grains or food crops with Fe, Zn, or other micro mineral nutrients in developing Asian and African countries, where cereals are the staple food. Agronomic biofortification is the only way to reach the poorest of the poor rural masses, who hardly have money to buy mineral supplements nor can afford to improve the components of their diet by incorporating more animal products. From the biofortification viewpoint, foliar application is better and requires lesser amount of Fe and Zn fertilizers than their soil application. Zn fertilization also increases grain Zn concentration in pulses, which are the major protein source in a vegetarian diet. This is important for developing countries especially like India, where a majority of people are vegetarian. When cultivars or genetically modified crops with grains denser in Fe and Zn are developed, adequate Fe and Zn fertilization will be necessary. The genetic and agronomic approaches are therefore complementary to each other and should progress in tandem.

REFERENCES

- Bell, D.W.; Dell, B. *Micronutrients for Sustainable Food, Feed, Fibre and Bio-energy Products*; International Fertilizer Industry Association: Paris, 2008; 175 pp.
- Alloway, B.J. *Zinc in Soils and Crop Nutrition*, 2nd Ed.; International Zinc Association: Brussels, 2008.
- Nene, Y.L. Symptoms, causes and control of khaira disease. *Bull. Indian Phytopathol. Soc.* **1966**, *3*, 99–101.
- Shukla, A.K.; Behera, S.K.; Shivay, Y.S.; Singh, P.; Singh, A.K. Micronutrients and field crop production in India: A review. *Indian J. Agron.* **2012**, *57*, S123–S130.
- Cakmak, I. *Identification and Correction of Wide-Spread Zinc Deficiency in Turkey—A Success Story*; The International Fertilizer Society: London, 2004.
- Shivay, Y.S.; Prasad, R. Zinc coated urea improves productivity and quality of basmati rice (*Oryza sativa*) under zinc stress condition. *J. Plant Nutr.* **2012**, *35*, 928–951.
- Shivay, Y.S.; Kumar, D.; Ahlawat, I.P.S.; Prasad, R. Relative efficiency of zinc oxide and zinc sulphate coated urea for rice. *Indian J. Fert.* **2007**, *3* (2), 51–56.
- Shivay, Y.S.; Kumar, D.; Prasad, R. Effect of zinc enriched urea on productivity, zinc uptake and efficiency of an aromatic rice-wheat cropping system. *Nutr. Cycl. Agroecosys* **2008**, *81*, 229–243.
- Shivay, Y.S.; Prasad, R.; Rahal, A. Relative efficiency of zinc oxide and zinc sulphate enriched urea for spring wheat. *Nutr. Cycl. Agroecosys.* **2008**, *82*, 259–264.
- Shivay, Y.S.; Kumar, D.; Prasad, R. Relative efficiency of zinc sulphate and zinc oxide coated urea in rice-wheat cropping system. *Commun. Soil Sci. Plant Anal.* **2008**, *39*, 1154–1167.
- Shivay, Y.S.; Kumar, D.; Prasad, R.; Ahlawat, I.P.S. Relative yield and zinc uptake by rice from zinc sulphate and zinc oxide coatings on to urea. *Nutr. Cycl. Agroecosys.* **2008**, *81*, 181–188.
- Shivay, Y.S.; Prasad, R.; Rahal, A. Genotypic variation for productivity zinc utilization efficiencies, and kernel quality in aromatic rice under low available zinc conditions. *J. Plant Nutr.* **2010**, *33*, 1835–1848.
- Shivay, Y.S.; Prasad, R.; Rahal, A. Studies on some nutritional quality parameters of organically or conventionally grown wheat. *Cereal Res. Commun.* **2010**, *38*, 342–352.
- Dhaliwal, S.S.; Sadana, U.S.; Khurana, M.P.S.; Dhadli, H.S.; Manchanda, J.S. Enrichment of rice grains through ferti-fortification. *Indian J. Fert.* **2010**, *6* (7), 28–35.
- Phattarakul, N.; Rerkasem, B.; Li, L.J.; Wu, L.H.; Zou, C.Q.; Ram, H.; Sohu, V.S.; Kang, B.S.; Surek, H.; Kalayci, M.; Yazici, A.; Zhang, F.S.; Cakmak, I. Biofortification of rice grain with zinc through zinc fertilization in different countries. *Plant Soil* **2012**, *361*, 131–141.
- Jiang, W.; Strik, P.C.; Lingna, J.; van Keulen, H.; Ming, Z.; Stomph, T.J. Uptake and distribution of root-applied or foliar applied ⁶⁵Zn after flowering in aerobic rice. *Ann. Appl. Biol.* **2007**, *150*, 383–391.
- Impa, S.M.; Johnson-Beebout, S.E. Mitigating zinc deficiency and achieving high grain Zn in rice through integration of soil chemistry and plant physiology research. *Plant Soil* **2012**, *361*, 3–41.
- Saenchai, C.; Prom-u-thai, C.; Jamjod, S.; Bernard, D.; Rerkasem, B. Genotypic variation in milling depression of iron and zinc concentration in rice grain. *Plant Soil* **2012**, *361*, 271–278.
- Singh, A. Storage and processing of rice. In *A Text Book of Rice Agronomy*; Prasad, R., Ed.; Jain Brothers: New Delhi, 1999; 203–210.
- Prom-u-thai, C.; Humg, L.; Cakmak, I.; Rerkasem, B. Simultaneous fortification of iron and zinc in parboiled rice. *Sci. Asia* **2011**, *37*, 296–302.
- Westfall, D.; Gangloff, B. Zinc fertilizers-is there a difference. *Agron. News.* **2001**, *21* (6), 8.

22. Kiekens, L. Zinc. In *Heavy Metals in Soils*; Alloway, B.J., Ed.; Blackie Academic and Professional: London, 1995; 284–305.
23. Naik, S.K.; Das, D.K. Relative performance of chelated Zn and zinc sulphate for lowland rice (*Oryza sativa* L). *Nutr. Cycl. Agroecosys.* **2008**, *81*, 219–227.
24. Alloway, B.J. Soil factors associated with zinc deficiency in crops and humans. *Environ. Geochem. Health.* **2009**, *31*, 537–548.
25. Cakmak, I.; Pfeiffer, W.H.; McClafferty, B. Biofortification of durum wheat with zinc and iron. *Cereal Chem.* **2010**, *87*, 10–20.
26. Cakmak, I.; Kalayaci, M.; Kaya, Y.; Torun, A.A.; Aydin, N.; Wang, Y.; Arisoy, Z.; Erdem, H.; Yazici, A.; Gokmen, O.; Ozturk, L.; Horst, W.J. Biofortification and localization of zinc in wheat grain. *J. Agr. Food Chem.* **2010**, *58*, 9092–9102.
27. Zhang, Y.Q.; Shi, R.L.; Karim, M.R.; Zhang, F.S.; Zou, C.Q. Iron and zinc concentrations in grain and flour of winter wheat as affected by foliar application. *J. Agr. Food Chem.* **2010**, *58*, 12268–12274.
28. McGarth, S.P.; Chamber, B.J.; Taylor, M.J.; Carlton-Smith, C.H. Biofortification in wheat grain by the application of sewage sludge. *Plant Soil* **2012**, *361*, 97–108.
29. Ozturk, L.; Yazici, M.A.; Yucel, C.; Torun, A.; Cekic, C.; Bagci, A.; Ozkan, H.; Braun, H.J.; Sayers, Z.; Cakmak, I. Concentration and localization of zinc during seed development and germination in wheat. *Physiol. Plantarum* **2006**, *128*, 144–152.
30. Cakmak, I. HarvestPlus zinc fertilizer project: Harvest Zinc. *Better Crops* **2012**, *96*, 17–19.
31. Manzeke, G.M.; Mapfumo, P.; Mtambanengwe, F.; Chikovo, R.; Tendayi, T.A.; Cakmak, I. Soil fertility management effects on maize productivity and grain Zn content in small holder farming system of Zimbabwe. *Plant Soil* **2012**, *361*, 57–69.
32. Shivay, Y.S.; Prasad, R. Effect of source and methods of zinc application on corn productivity, nitrogen and zinc concentrations and uptake by high quality protein corn (*Zea mays*). *Egypt. J. Biol.* **2014**, *16* (1), 72–78.
33. Shivay, Y.S.; Prasad, R.; Pal, M. Zinc fortification of oats grain and straw through zinc fertilization. *Agr. Res.* **2013**, *2*, 375–381.
34. Shivay, Y.S.; Prasad, R.; Pal, M. Effect of source, level and method of zinc application on yield, zinc biofortification of grain, zinc uptake and zinc use efficiency in (*Cicer arietinum* L.). *Commun. Soil Sci. Plant Anal.* **2015**, *46*, 2191–2200.

Food Security and Soil Water Management

Mark W. Rosegrant

Ephraim Nkonya

Rowena A. Valmonte-Santos

Environment and Production Technology Division, International Food Policy Research Institute (IFPRI), Washington, District of Columbia, U.S.A.

Abstract

Agricultural soil water management is a critical variable in food production growth to meet food demand and improve food security. This entry reviews the influence of soil water management on crop production in both low-input and high-input agricultural environments. Technological and managerial options are available to significantly reduce soil water constraints and improve crop productivity growth and must be tailored to specific agroclimatic environments. The options must be supported with policy reforms to provide appropriate economic incentives for soil water management and, particularly in Africa, investment in infrastructure, communications, and markets.

INTRODUCTION

With growing incomes and world population being projected to reach eight billion in 2025, there remains great pressure on the agricultural sector to increase food production to ensure food availability and feed the growing populace. Effective soil water management to improve soil and water productivity is an essential element for food security. This entry reviews the evidence on the impact of soil degradation on crop productivity, assesses the constraints to productivity growth arising from ineffective soil water management, and proposes solutions to enhance productivity growth and food security.

IMPACT OF LAND DEGRADATION ON FOOD SECURITY

Soil degradation, in significant part, due to poor soil water management, has direct on-farm production costs (in the form of lower yields or greater production costs as more fertilizers and other inputs are applied to offset yield losses) and off-site costs (in the form of siltation of dams and irrigation systems and damage to mangrove swamps and coral reefs). Many studies have assessed the rates of land degradation for countries and regions, with a wide range of results. According to a widely cited global study, 14% of all agricultural land, permanent pasture, forests, and woodland suffered from serious soil degradation between 1945 and 1990.^[1] The study estimated degradation to be generally more prevalent in the developing countries: 31% of land in Central America, 19% in Africa, and 16% in Asia were considered to be seriously degraded. The share of degraded area was estimated to be highest on agricultural

land (38%), especially in the developing countries. The most common forms of soil degradation were soil erosion by water and wind, followed by soil nutrient depletion, salinization, compaction, and sealing and crusting. The economic costs of this land degradation are difficult to assess because it is hard to quantify the impact of land degradation on crop productivity due to complex soil fertility and crop yield relationships. However, based on the physical soil degradation assessed in that study, a global study estimated the 1945–1990 cumulative crop productivity loss from land degradation worldwide of 5–13% depending on the underlying assumptions.^[2] While another study estimated losses of 5–9%, considering cropland and pasture productivity.^[3] These cumulative losses are equivalent to a productivity drop of 0.10–0.20% per year. This decline represents a significant drag on cereal yield, which during the period 1970–2004 grew globally by 1.19% per year, and only 0.94% per year in the developing countries. Moreover, a review of the literature showed cumulative crop yield losses due to past erosion of 2–40% across all African countries, with a mean of 8.2% for the entire continent and 6.2% for sub-Saharan Africa (SSA).^[4] These national-level estimates confirm that land degradation can be devastating in some countries, especially in fragile environments.

SOIL WATER CONSTRAINTS ON AGRICULTURAL PRODUCTION IN LOW-INPUT AND INTENSIVE AGRICULTURE

In this section, we discuss soil water management constraints to agricultural production in two polar cases, the low-input system in Africa and intensive irrigated agriculture in Asia. Accounting for one-third of total GDP, 70%

full-time employment, and 40% total export earnings, in the early 2000s, agriculture is crucial to African economic development and food security. But population growth rates have exceeded the rate of production growth in many areas, leading to declining per capita food production. Crop yields in SSA are low (799 kg/ha in 1967) and have only grown slowly (980 kg/ha in 2005) compared with other regions. The green revolution that brought huge improvements to food security in Asia largely bypassed Africa, which is characterized by predominantly rainfed rather than irrigated agriculture, poor soil fertility and soil water management, less crop diversification, underinvestment in agricultural research and development and infrastructure, lack of competitive markets and conducive enabling environments, large and growing impact of human health on agriculture, low-labor productivity, minimal mechanization, limited access to agricultural inputs, credit, and poverty that limit farmer ability to buy inputs.^[5]

Inadequate soil water management practices in Africa have contributed to slow growth in production through soil fertility depletion, including limited adoption of organic and inorganic fertilizer replenishment strategies; inadequate adoption of soil and water conservation measures; declining use and length of fallow periods; expansion of agricultural production into marginal and fragile areas, such as cultivation on steep slopes or in arid areas without proper anti-erosion measures; use of animal dung and crop residues as fuel rather than as matter for soil enrichment; and the removal of vegetation through overgrazing, logging, development, and domestic use.^[6]

In intensive Asian cropping systems, soil water management is also an important factor in promoting—or undermining—food security. Growth in crop yields, especially rice and wheat, was very rapid in much of Asia from the introduction of modern high-yielding varieties in the mid-1960s through the late 1980s. But environmental and resource constraints adding to soil water management have also contributed significantly to the evident slowdown in yield growth. Increased land use intensity has led to higher input requirements to sustain yield gains. Particularly in irrigated rice systems, declining yield growth trends can be directly associated with the ecological consequences of intensive rice monoculture systems, plus buildup of salinity and waterlogging, use of poor-quality groundwater, nutrient depletion and mining, increased soil toxicities, and increased pest buildup, especially soil pests.

Inappropriate management of irrigation has contributed to environmental problems such as excessive water depletion, water quality reduction, waterlogging, and salinization. Poor irrigation practices accompanied by inadequate drainage have often damaged soils through oversaturation and salt buildup. However, intensification per se is not the root cause of lowland resource-based degradation. A policy environment that encourages monoculture systems and excessive or unbalanced input use is significantly to blame. Trade policies, output price policies, and input subsidies,

particularly for water and fertilizer, have driven excessive use of these inputs. The successful drive for food self-sufficiency in Asian countries with an exhausted land frontier came at a high ecological and environmental cost. Appropriate policy reforms at the macro and sector level will go a long way toward arresting degradation trends, but the degree of degradation in many regions will pose severe policy challenges.

SOLUTIONS

Solving food security challenges in the developing countries requires all the contributing factors to be addressed, but soil water management is a critical area where improvement in policy and management would improve food security.

Soil Quality, Fertilizer Use, and Nutrient Management

A number of approaches have been adopted to deal with soil infertility in SSA. These approaches consist of organic farming, high external input agriculture, low external input sustainable agriculture, and integrated soil fertility management. No single approach is likely to succeed because of the diverse agroclimatic contexts of the region.^[7] Integrated nutrient management (INM), which combines the application of chemical fertilizer and organic matter and use of other organic soil fertility practices such as agroforestry, relay cropping, soil erosion control, and tillage methods, will be essential.^[8,9] INM is often more sustainable than sole use of either chemical fertilizer or organic matter. Chemical fertilizer provides the macronutrients (nitrogen, phosphorus, and potassium) but does not provide the organic matter which is essential to soil fertility. On the other hand, organic matter such as manure is poor in phosphorus and potassium and contains only a small share of nitrogen.^[10]

A substantial increase in the use of inorganic fertilizer is therefore also essential to agricultural growth in Africa. Fertilizer use can be promoted through subsidies and vouchers, public investment in soil fertility enhancement technologies, locally tailored fertilizer recommendations, and coordinated service provision, including output markets and the use of public-private partnerships.^[11] The affordability of fertilizers is one of the major challenges in Africa. Subsidies can compensate farmers in remote areas for high-transport costs involved in supplying their inputs and purchasing their outputs. But many of the benefits of subsidies are captured by richer farmers who have the most effective demand for fertilizer. A more appropriate role for the government in enhancing fertilizer use would be to provide public goods, including agricultural market and transportation and communication infrastructure, and extension services, to advise farmers on the appropriate quantity, quality, and timing of fertilizer applications.

Sustainable and Efficient Water Use

Investment in small dams, groundwater development, and water harvesting offers a major opportunity for promoting agricultural production and improving livelihoods in some African regions. Simple and affordable innovations to tap shallow groundwater, such as treadle pump and small motor pump technology, have the potential to dramatically improve poor people's access to groundwater, as has been achieved in West African countries like Nigeria, Niger, and Chad. Shallow aquifers are also accessible using indigenous methods of well construction and low-cost techniques, such as hand dug wells and tube wells.^[12] However, aquifers should be utilized in ways that avoid undermining their sustainability. Investments and policies focused on innovative water management approaches for rainfed agriculture, like rainfall harvesting and moisture conservation-based farming, can also significantly improve Africa's future water resources scenario.

In Asia, overuse of water that has led to soil salinization needs to be addressed in a number of ways such as giving increasing priority to water efficiency investments within the irrigation system and on-farm production; watershed management, flood control, drought management, and drainage; the institutional and incentive framework for surface and groundwater management and conservation. The policy framework to establish incentives may be the most important. Water and power subsidies should be reduced or eliminated because they promote overuse and therefore distort user incentives toward conservation of water and cost-effective practices. The command-and-control approach is often the least cost-effective approach because of poor governance and complex property rights issues. A crucial policy reform would be the establishment of clear and secure water rights. Water rights empower users by requiring their consent to any reallocation of water and by receiving compensation for transferred water. Security of water rights tenure based on well-defined rights encourages investment in water-saving technology.

To further address the salinity and water-logging problems that have affected the irrigated and some rainfed areas in Asia, there is a need to improve development and dissemination of conservation tillage and water-harvesting technologies. This will also increase the water productivity and help to mitigate the effects of drought in the dry areas and increase the crop yield response to fertilizer and other inputs.^[9]

CONCLUSION

Inadequate soil water management constrains crop productivity growth and food security in both low-input African cropping systems and intensive Asian systems. Technological and management options are available to significantly reduce these constraints and must be tailored to specific agroclimatic environments. But these solutions will not be adopted without policy reforms to provide appropriate

economic incentives for soil water management and, particularly in Africa, investment in infrastructure, communications, and markets.

REFERENCES

1. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *World Map of the Status Of Human-induced Soil Degradation: An Explanatory Note*; International Soil Reference and Information Centre and United Nations Environment Program: Wageningen and Nairobi, 1991.
2. Crosson, P.R. *Soil Erosion and Its On-farm Productivity Consequences: What Do We Know*; Discussion Paper 95-29; Resources for the Future: Washington, 1995.
3. Oldeman, L.R. *Soil Degradation: A Threat to Food Security*; Report 98/01; International Soil Reference and Information Centre: Wageningen, 1998.
4. Scherr, S.; Yadav, S. *Land Degradation in the Developing World: Implications for Food, Agriculture, and the Environment to 2020*; 2020 Vision for Food, Agriculture, and the Environment Discussion Paper No. 14; International Food Policy Research Institute: Washington, 1996.
5. InterAcademy Council (IAC). *Realizing the Promise and Potential of African Agriculture: Science and Technology Strategies for Improving Agricultural Productivity and Food Security in Africa*; Royal Netherlands Academy of Arts and Sciences: Amsterdam, 2004; 266 pp.
6. Pender, J.; Place, F.; Ehui, S. Strategies for sustainable agricultural development in the East African highlands. *Proceedings of the International Conference Strategies for Poverty Alleviation and Sustainable Resource Management in the Fragile Lands of Sub-Saharan Africa*, Entebbe, Uganda, May 25-29, 1998; Knox McCullough, A., Babu, S., Hazell, P., Eds.; Feldafig, Germany, 1999.
7. Pender, J.; Mertz, O. Soil fertility depletion in sub-Saharan Africa: What is the role of organic agriculture? *Global Development of Organic Agriculture: Challenges and Promises*; Halberg, N., Alrøe, H.F., Knudsen, M.T., Kristensen, E.S., Eds.; CABI Publishing: Wallingford, 2005; 215-240.
8. Vanlauwe, B. Integrated soil fertility management research at TSBF: The framework, the principles, and their application. In *Managing Nutrient Cycles to Sustain Soil Fertility in Sub-Saharan Africa*; Bationo, A., Ed.; Academy of Science Publishers: Nairobi, 2004; 25-42.
9. Bationo, A.; Waswa, B.; Kihara, J.; Kimetu, J. *Advances in Integrated Soil Fertility Management in Sub Saharan Africa: Challenges and Opportunities*, Nutrient Cycling in Agroecosystems; Springer: New York, 2007; 1091 pp.
10. Palm, C.A.; Myers, R.J.K.; Nandwa, S.N. Combined use of organic and inorganic nutrient sources for soil fertility maintenance and replenishment. In *Replenishing Soil Fertility in Africa*; Buresh, R.J., Sanchez, P.A., Calhoun, F., Eds.; SSSA, ASA: Madison, 1997; 193-217.
11. Poulton, C.; Kydd, J.; Dorward, A. *Increasing Fertilizer Use in Africa: What Have We Learned?* ARD DP 25; World Bank; World Bank: Washington, 2006.
12. Inocencio, A.; Sally, H.; Merrey, D.J. *Innovative Approaches to Agricultural Water Use for Improving Food Security in Sub-Saharan Africa*, Working Paper 55; International Water Management Institute: Colombo, 2003.

Food Security: Agricultural Productivity and Resources

Keith D. Wiebe

*Resource Economics Division, U.S. Department of Agriculture (USDA),
Washington, District of Columbia, U.S.A.*

Abstract

Soil resources and agricultural productivity affect food security both through their impact on food supply and through their impact on the incomes of that half of the world's people whose livelihoods depend directly on agriculture. Agricultural productivity depends in turn on a variety of factors, including the quality of soil and other natural resources. Resource quality has often received insufficient attention owing to the scarcity of appropriate data, but improvements in data and methods allow better understanding of the impact of differences in the quality of land, labor, institutions, and infrastructure on agricultural productivity, and thus on food security.

WHAT IS FOOD SECURITY?

Food security has been defined and used in many ways, but the World Bank's 1986 definition, "access by all people at all times to sufficient food for an active, healthy life," is typical of most.^[1] Access to food derives from opportunities to produce food directly or to exchange other commodities or services for food.^[2] These opportunities are based in turn on resources, including soil, as well as on production technologies, environmental and market conditions, and other factors.^[3]

To ensure food security, access to food must be sufficient under all possible circumstances in any particular period of time, because all sources of access are subject to variation. Food production varies with weather and other environmental factors, e.g., access to food via exchange depends on market factors such as wages and food prices. Access to sufficient food must also be sustainable over the long term. A household (or nation) can hardly be considered food secure in the long run if it is able to meet its nutritional requirements only by depleting its endowment of resources. Appropriate use and conservation of soils are thus central to the concept of food security.

TRENDS IN GLOBAL FOOD SECURITY

The world's population approached 6.4 billion in 2004 and continues to grow. Food production has grown even more rapidly in decades, increasing in per capita terms by 0.9% annually on a global scale, and even faster in China and other populous developing countries.^[4] Production growth notwithstanding, over 800 million people, most of them in developing countries, suffer from chronic food insecurity because they cannot produce or purchase enough food.^[5] The Economic Research Service projects that the gap

between what the poorest countries can produce or purchase and what they need to meet their nutritional requirements will grow by over 50% over the decade, owing in part to depletion and degradation of soil and water resources.^[6]

Population growth has slowed in years, but even so the world's population is projected to reach 7.5 billion by 2020. Almost all of this growth will occur in the developing world. Average incomes also continue to rise, and by 2005 over half of the world's population is expected to live in urban areas, with associated changes in dietary preferences.^[7] As a result of these factors, the International Food Policy Research Institute projects an increase of about 40% in world demand for grain over the period 1993–2020, primarily in the developing countries.^[8]

Such growth in food demand, representing an annual increase of 1.3%, is well within the range of growth in production since the mid-20th century.^[9] Yet historic rates of growth in both cultivated area and yields have slowed in years. The Food and Agriculture Organization (FAO) reports that only a third of the estimated 2.8 billion hectares of land with potential for crop production in the developing countries is in cultivation, but the costs of expanded cultivation are often high in economic and environmental terms.^[10] As a result of these costs, the FAO projects that area expansion will contribute only about 20% of production growth in 30 years. Increased multiple cropping and shorter fallow periods will contribute another 11%, while the majority of production increases (69%) is projected to come from increased yields. Global average cereal yield growth slowed from 3.0%/yr during the 1960s to 1.1%/yr during the 1990s,^[11] owing to reduced growth in input use, low cereal prices, and low levels of investment in agricultural research and technology. Poorly functioning markets and a lack of appropriate infrastructure and credit also contribute.^[8] Changes in soil quality may also play a role.

SOIL QUALITY AND AGRICULTURAL PRODUCTIVITY

Soils play a central role in food security, both in terms of food production and in terms of the livelihoods of farmers who produce crops other than food. Given limits on the expansion of agricultural land, increases in both food production and farmers' incomes depend critically on growth in agricultural output per unit of land, labor, and other inputs, i.e., agricultural productivity. While it has long been recognized that agricultural productivity depends directly on soil fertility, water-holding capacity, and other aspects of the quality of soil resources, these relationships have been difficult to quantify on global and regional scales owing to limitations of data and spatial variations in climate, topography, management practices, and other factors.^[12] Improvements in spatially referenced data and analytical methods have allowed better understanding of the relationship between soil quality and agricultural productivity.

To isolate and control the effects of differences between countries in land quality, Wiebe et al.^[13] used spatially referenced soil and climate data in combination with new high-resolution land-cover data to measure the share of each country's cropland that is not limited by major soil or climate constraints to agricultural production. Econometric analysis of this measure, controlling for levels of labor, fertilizer, and other inputs to agricultural production, indicates that in most regions of the world over the period 1961–1997, agricultural productivity was 20–30% higher in countries with above-average soils and climates than it was in countries with below-average soils and climates. Soil quality also influences the impact of other inputs on agricultural productivity. For example, fertilizer response in sub-Saharan Africa is significant and positive both in countries that have good soils and climate and in countries that do not, but the magnitude of the response is about twice as large in the latter countries.^[14] While these productivity impacts are estimated from differences in soil quality across countries, they suggest the importance of examining more closely the productivity impacts of changes in soil quality over time, i.e., soil degradation.

SOIL DEGRADATION AND FOOD SECURITY

Although data are limited, it is estimated that about 22% of the area under agriculture, permanent pasture, forest, or woodland worldwide has been degraded to some degree by physical, chemical, or biological processes since the mid-20th century.^[15] Numerous studies have documented the impacts of such processes on crop yields in specific locations, but extrapolation to larger scales is difficult. Rough estimates suggest that soil degradation has reduced global average agricultural productivity by about 0.1% annually.^[16] Such losses have historically been masked by increased use of inputs and improvements in production technology, and as

a result, it is generally agreed that they do not pose a threat to global food security. Degradation-induced losses may become more apparent, however, or more costly to mitigate, if yield growth continues to slow down.

Yield effects may also be much more severe in particular areas. In an analysis of over 300 original field studies around the world, Den Biggelaar et al.^[17] found that the impact of soil erosion on yields varied widely by crop, soil, and region. For example, wheat yields in Europe declined by an average of only 0.04%/yr as a result of soil erosion, while maize yields in Central and South America declined by an average of 0.94%/yr. Lal^[12] and Scherr^[18] report similar variation in impacts across crops, soils, and regions elsewhere in the world, with corresponding variation in the potential impact of soil degradation on food security.

Actual impacts in any particular location depend not only on crop and soil but also on climate, topography, and a host of other factors, including management practices. Understanding the impact of soil degradation on agricultural productivity and food security thus requires an understanding of the incentives farmers face when making choices about management practices.

INCENTIVES FOR SOIL USE AND CONSERVATION

Some forms of soil degradation—such as nutrient depletion—can generally be feasibly reversed given technologies and market conditions, but others—such as topsoil loss owing to erosion—are effectively irreversible. How soils are actually managed to avoid, minimize, or reverse degradation depends on a variety of factors influencing farmer decisions, including the prices of agricultural inputs and outputs, farmer characteristics, and land tenure.^[19] Where markets function well and property rights are well defined, farmers generally have an incentive to manage soils to protect their long-term productive potential.^[20] Where land tenure is uncertain or access to credit is limited, on the other hand, this incentive may be weak and farmers may not find it optimal to adopt costly soil conservation measures.^[21]

In addition to its effects on productivity, soil degradation may also affect water quality, siltation in reservoirs, flood frequency, and other outcomes.^[22] As these impacts are felt off-site, farmers generally need additional incentives to address such concerns sufficiently. These concerns motivate policy measures to encourage increased adoption of conservation measures by farmers in many countries, including farmers in the United States.

CONCLUSION

Growth in global population, incomes, and food demand presents continuing challenges for agriculture in the 21st century. New technologies offer considerable potential

to maintain and improve soil quality, agricultural productivity, and food security, and evidence suggests that soil degradation has had a relatively small impact on global agricultural productivity. Actual resource management strategies and outcomes vary widely with circumstances, however, and depend critically on the incentives and institutions faced by farmers. The greater the spatial scale of potential costs and benefits, and the farther they are into the future, the greater is the challenge for policymakers to structure appropriate incentives and institutions, and the greater is the importance of research and improved understanding of the alternatives available.

REFERENCES

1. World Bank. *Poverty and Hunger: Issues and Options for Food Security in Developing Countries*; The World Bank: Washington, 1986.
2. Sen, A. *Poverty and Famines*; Clarendon Press: Oxford, 1981; 257 pp.
3. Maxwell, D.; Wiebe, K. Land tenure and food security: Exploring dynamic linkages. *Dev. Change* **1999**, *30* (4), 825–849.
4. World Bank. *World Development Indicators*; The World Bank: Washington, 2004.
5. Food and Agriculture Organization (FAO) of the United Nations. *The State of Food Insecurity in the World 2004*; FAO: Rome, 2004; 6 pp.
6. Shapouri, S.; Rosen, S. *Food Security Assessment: Why Countries Are at Risk*; Agriculture Information Bulletin 754; U.S. Department of Agriculture Economic Research Service: Washington, 1999; 21 pp.
7. Food and Agriculture Organization (FAO) of the United Nations. *The State of Food and Agriculture 1998*; FAO: Rome, 1998; 371 pp.
8. Pinstrup-Andersen, P.; Pandya-Lorch, R.; Rosegrant, M.W. *World Food Prospects: Critical Issues for the Early Twenty-First Century*; International Food Policy Research Institute: Washington, 1999; 32 pp.
9. Byerlee, D.; Heisey, P.; Pingali, P. Realizing yield gains for food staples in developing countries in the early twenty-first century: Prospects and challenges. *Presented at the Study Week on Food Needs of the Developing World in the Early Twenty-First Century*, The Vatican, Jan 27–30, 1999.
10. Food and Agriculture Organization (FAO) of the United Nations. Crop production and natural resource use. In *World Agriculture: Towards 2015/30: An FAO Perspective*; Jelle, B., Ed.; Earthscan Publications: London, 2003; 128.
11. Food and Agriculture Organization (FAO) of the United Nations. *FAOSTAT Database*; FAO: Rome. Available at <http://faostat3.fao.org/home/E> (accessed August 21, 2015).
12. Lal, R. Soil erosion impact on agronomic productivity and environment quality. *Crit. Rev. Plant Sci.* **1998**, *17* (4), 319–464.
13. Wiebe, K.; Soule, M.; Narrod, C.; Breneman, V. Resource quality and agricultural productivity: A multi-country comparison. In *Presented at the Annual Meeting of the American Agricultural Economics Association*, Tampa, Florida, Jul 31, 2000.
14. Wiebe, K.; Soule, M.; Narrod, C.; Breneman, V. Resource quality and agricultural productivity: A multi-country comparison. In *Land Quality, Agricultural Productivity, and Food Security: Biophysical Processes and Economic Choices at Local, Regional, and Global Levels*; Wiebe, K., Ed.; Edward Elgar: Northampton, 2003; 147–165.
15. Oldeman, L.R. *An International Methodology for an Assessment of Soil Degradation and Georeferenced Soils and Terrain Database*; Working Paper and Preprint 93/06; International Soil Reference Information Centre: Wageningen, 1993; 22 pp.
16. Crosson, P. *Soil Erosion and Its On-Farm Productivity Consequences: What Do We Know?* Discussion Paper 95–29; Resources for the Future: Washington, 1995; 18 pp.
17. Den Biggelaar, C.; Lal, R.; Wiebe, K.; Breneman, V. The global impact of soil erosion on productivity (II): Effects on crop yields and production over time. *Adv. Agron.* **2004**, *81*, 49–95.
18. Scherr, S. *Soil Degradation: A Threat to Developing-Country Food Security by 2020?* Food, Agriculture, and Environment Discussion Paper 27; International Food Policy Research Institute: Washington, 1999; 63 pp.
19. Soule, M.J.; Tegene, A.; Wiebe, K.D. Land tenure and the adoption of conservation practices. *Am. J. Agr. Econ.* **2000**, *82* (4), 993–1005.
20. Hopkins, J.W.; Schnitkey, G.D.; Sohngen, B.L.; Miranda, M.J.; Tweeten, L.G. Optimal cropland degradation with reversible and irreversible components. In *Presented at the Annual Meeting of the American Agricultural Economics Association*, Salt Lake City, Utah, Aug 2–5, 1998.
21. Pagiola, S. Economic analysis of incentives for soil conservation. In *Using Incentives for Soil Conservation: From Theory to Practice*; Sanders, D.W., Huszar, S., Sombatpanit, S., Enters, T., Eds.; Science Publishers, Inc.: Enfield, 1999; 41–56.
22. Crosson, P. Future supplies of land and water for world agriculture. In *Population and Food in the Early Twenty-First Century*; Islam, N., Ed.; International Food Policy Research Institute: Washington, 1995; 143–159.

Food Security: Soil Degradation, Global Scale

Michael A. Zoebisch

German Agency for International Cooperation, Tashkent, Uzbekistan

Eddy De Pauw

International Center for Agricultural Research in the Dry Areas (ICARDA), Aleppo, Syria

Abstract

Growing populations lead to an increasing demand for food. With decreasing per-capita areas of arable land, this leads to additional pressure and stresses on the limited land resources, especially the soils. As a consequence, large parts of the world are affected by soil degradation, and this has a direct effect on food security.

INTRODUCTION

Food security is commonly defined as the access by all people at all times to enough food for an active, healthy life. Extended concepts include food quality aspects, cultural acceptability of the food, equitable distribution among the different social groups, and gender balance.^[1–4] These concepts imply that, in addition to producing enough food, people must also have access to it. This opens the arena for the socioeconomic, cultural, policy, and political dimensions of food production and supply. Thus, food security—and consequently food insecurity—are multidimensional. It is not possible to separate clearly the biophysical and socioeconomic dimensions of food security, because they are closely interrelated and interdependent.^[2] However, the most limiting natural resources needed for food production are soil and water. If these resources are depleting, land productivity—the key factor for crop production—declines, and the land is no longer capable of producing the biomass needed for direct and indirect human consumption. If soil degrades, its plant-life-supporting functions are lost, together with its role in the hydrological cycle. Soil is a limited resource and, because of the long duration of soil-forming processes, can be considered as non-renewable. The production of food and feed is directly linked to the productive capacity of the soil. The degree and extent of soil degradation and, consequently, its effect on food production are dependent on how the land is used.

UNDERNOURISHMENT—A MAJOR INDICATOR OF FOOD INSECURITY

The proportion of chronically undernourished people is a manifestation of the degree of food insecurity. Worldwide, more than 830 million people are undernourished; 800 million of these people live in the developing countries,

i.e., about 18% of the population of these countries.^[1] Table 1 shows that the most seriously affected region is sub-Saharan Africa, with 33% of the population chronically undernourished. In Asia and the Pacific, chronic undernourishment has decreased significantly over the past decades.

Yields of the major food crops in many countries of these regions have risen remarkably. But in most cases the yield increases have not been able to keep pace with the needs of growing populations. Subregional trends, especially from Africa, China, South Asia, and Central America, indicate large yield declines due to soil degradation.^[2,6,7] Countries of the less degraded temperate regions can substitute for the losses in other regions. However, in the affected areas, food prices will increase and hence the incidence of malnutrition. Densely populated countries and regions that depend solely on agriculture will be most affected. As input requirements—and their costs—grow with increasing degradation, these countries will probably not be able to maintain adequate levels of soil productivity without special efforts to stabilize the system.

VULNERABLE AREAS

Only about one-tenth of the world's arable areas are not endangered by degradation. Most soils are vulnerable to degradation and have low resilience to fully recover from stresses. The soils in the tropics and subtropics are more vulnerable to degradation than those in temperate areas, mainly due to the climatic conditions.^[8] The total land area available for agricultural production is limited. Different types of land have different capabilities and limitations. Not all land is suitable for cultivation. The major natural limiting factors for soil productivity are steep slopes, shallow soils, low levels of natural fertility, poor soil drainage, sandy or stony soil, salinity, and sodicity.

Different agroecosystems are typically susceptible to different types of soil degradation.^[7–9] In mountainous

Table 1 Undernourishment in the developing countries.

Region/subregion	Undernourished population		Number of people affected 1995/1997 (millions)
	Proportion of total population 1979/1981 (%)	Proportion of total population 1995/1997 (%)	
Total developing world	29	18	791.5
Asia and Pacific	32	17	525.5
East Asia	29	14	176.8
Oceania	31	24	1.1
Southeast Asia	27	13	63.7
South Asia	38	23	283.9
Latin America and Caribbean	13	11	53.4
North America	5	6	5.1
Caribbean	19	31	9.3
Central America	20	17	5.6
South America	14	10	33.3
Near East and North Africa	9	9	32.9
Near East	10	12	27.5
North Africa	8	4	5.4
Sub-Saharan Africa	37	33	179.6
Central Africa	36	48	35.6
East Africa	35	42	77.9
Southern Africa	32	44	35
West Africa	40	16	31.1

Source: From Lal & Singh.^[5]

areas and steep lands, water erosion is the dominant form of soil degradation. For arid areas, both water and wind erosion and the loss of soil organic matter are typical. In humid areas, soil acidification and fertility decline are of importance. In irrigated areas, the hazards of soil salinization and waterlogging are of special significance. Knowledge of these hazards can help understand the limitations of the land and introduce appropriate land use practices that reduce soil degradation and loss of soil productivity.

UNDERLYING CAUSES

Misuse of Land

The main causative factors of human-induced soil degradation, in a broad sense, are overuse and inappropriate management of agricultural land, deforestation and the removal of natural vegetation, and overgrazing as well as industrial activities that pollute the soil.^[7] Worldwide, soil erosion by wind and water is the most important form of

soil degradation, accounting for between 70% and 90% of the total area affected by soil degradation.^[6,7] High population pressure and the resultant increased need for higher land productivity are the main factors leading to inappropriate and exploitative land use.

In high-input systems, loss of soil quality due to degradation can be compensated to a certain extent by inputs, such as fertilizers and irrigation water. In low-input systems—usually practiced by resource-poor farmers (in poor countries)—soil quality contributes relatively more to agricultural productivity.^[10] Therefore, soil degradation in these systems has more drastic and immediate effects on soil productivity and, hence, food production.

Pressure on the Land

When the land is not enough, i.e., when land scarcity becomes a limiting factor for food production, people usually resort to intensification of their land use and opening up new, often unsuitable, land for cultivation.^[4,10] Both pathways will most certainly lead to soil degradation, if unsuitable land use technologies are used. The majority of land users in the developing world do not have adequate access to appropriate land use technologies and the financial means to invest in their land.^[1] The area available for crop production therefore puts a definite limit to sustainable food production in many parts of the developing world. Alternative, i.e., non-agricultural means of income generation would enable the people to supplement their food requirements on the market and could relieve immediate pressure on the land. This would save soil and land resources.

Losses and Gains of Agricultural Land

Estimates show that more than 10 million hectares of agricultural land worldwide is lost per year, most of it due to soil degradation and also due to urbanization (i.e., between 2 and 4 million hectares).^[4] Annual deforestation and clearing of savanna land accounts for about 20 million hectares, of which 16 million hectares is converted to cropland. Thus, there is an annual net gain of cropland of about 5–6 million hectares. Most of these converted lands can be considered marginal, i.e., less suitable for cropping. Loss of traditional sources of grazing and firewood, together with the ecological functions of forests, i.e., their role in regulating watershed hydrology, is the factor inducing the degradation of soil.

MINIMUM PER-CAPITA CROPLAND REQUIREMENT

On a global scale, it is estimated that the minimum per-capita area of cropland required to produce an adequate quantity of food is around 0.1 ha.^[2,6] This rough estimate is dependent on many different location-specific factors and circumstances, such as soil, terrain, climate, farming

system, and land use technology. However, it permits a suitable comparison at the overall global scale. Estimates predict that a large number of countries will have reached the 0.1 ha limit of per-capita cropland by the year 2025.^[1,2,8] In the absence of more land for cultivation, future food needs of the people in these countries will have to depend on increased soil productivity through intensified land use. Resource-poor farmers with no option to expand their cultivated area usually cannot invest in necessary soil conservation and soil fertility maintenance measures. Over time, this leads to nutrient depletion and other forms of soil degradation that affect soil productivity.^[4,5,10]

Although some developing countries have indeed increased their yields on a per-unit area basis significantly, these increases cannot compensate for the increased demands by growing populations. Table 2 illustrates past and future trends in per-capita available cropland in selected countries with high population growth and gives examples of yield level trends. The table shows that, to a large extent, the gains in yields are offset by the decrease in available cropland to grow the required total quantities of crops to feed the population. In these countries, food insecurity can directly be related to the pressure on the land or land scarcity.

The growing pressure on the land is a reflection of not only population growth and the direct needs for food, but also changed diets and increasing financial needs (by the

land users) that have to be met from land cultivation, putting additional stress on the soil resources.

ESTIMATING SOIL DEGRADATION ON A GLOBAL SCALE

Limitations of Methodology

The scientific relationships between different soil degradation processes and soil productivity have been established mainly on field and plot level.^[5,12] Although reliable data are available for a wide range of different agroecologies and farming systems, they cannot give a precise picture of degradation-dependent soil productivity for larger land units. The increasing complexity of the determining factors and processes on larger land units, the endless array of location-specific conditions, and the lack of adequate data make upscaling extremely difficult for country and regional levels. Estimates of the extent and severity of soil degradation—and their effects on soil productivity—on a country, regional, continental, and global levels therefore bear a substantial degree of uncertainty. Most estimates use the methodology developed by the Global Assessment of Human-Induced Soil Degradation (GLASOD) project.^[6,7] The GLASOD estimates are based on expert assessment and are therefore largely subjective. However, the

Table 2 Trends of available per-capita cropland and average yield levels of main staple crops for selected countries.

Country	Per-capita cropland (ha)					Crop	Average yield (kg/ha)			
	1961	1980	1990	2000	2015 ^a		1961	1980	1990	2000 ^b
Syria	1.40	0.62	0.46	0.34	0.17	Wheat	575	1,536	1,544	1,882
						Barley	461	1,312	310	381
Kenya	0.21	0.15	0.10	0.08	0.04	Wheat	1,090	2,156	1,864	1,400
						Maize	1,253	1,200	1,580	1,321
Pakistan	0.34	0.25	0.17	0.13	0.07	Wheat	822	1,568	1,825	2,492
						Rice	1,391	2,423	2,315	2,866
Bangladesh	0.17	0.10	0.09	0.07	0.05	Wheat	573	1,899	1,504	2,258
						Rice	1,700	2,019	2,566	2,851
Nepal	0.19	0.17	0.14	0.11	0.07	Wheat	1,227	1,199	1,415	1,820
						Rice	1,937	1,932	2,407	2,600
Zambia	1.52	0.92	0.65	0.57	0.28	Wheat	1,600	4,068	4,399	5,454
						Maize	882	1,688	1,432	1,431
Côte d'Ivoire	0.69	0.44	0.31	0.24	0.10	Rice	757	1,166	1,155	1,548
						Sorghum	667	583	575	352
Sudan	0.97	0.62	0.52	0.43	0.22	Rice	1,409	634	1,250	563
						Sorghum	970	712	428	634
Peru	0.20	0.20	0.17	0.14	0.10	Wheat	1,001	939	1,085	1,289
						Potato	5,287	7,196	7,881	10,818

^aUN Projection, medium level.

^bFAO estimates.

Source: From Engelman & LeRoy^[2] and FAO.^[11]

Table 3 Estimates of degradation and losses in soil productivity for different land use types in the major regions.

Region	Degraded areas								Loss in productivity ^a	
	Agricultural land		Pasture land		Forests		Total degraded area			
	(Mha)	(%)	(Mha)	(%)	(Mha)	(%)	(Mha)	(%)	Cropland (%)	Pasture (%)
Africa	121	65	243	31	130	19	494	30	25	6.6
Asia	206	38	197	20	344	27	747	27	12.8	3.6
South America	64	45	68	14	112	13	244	16	13.9	2.2
Central America	28	74	10	11	25	38	63	32	36.8	3.3
North America	63	26	29	11	4	1	96	9	8.8	1.8
Europe	72	25	54	35	92	26	218	27	7.9	5.6
Oceania	8	16	84	19	12	8	104	17	3.2	1.1
World	562	38	685	21	719	18	1,966	23	12.7	3.8

^aCumulative loss in productivity since 1945. Adjusted figures according to type and degree of degradation.

Source: From Scherr,^[6] van Lynden & Oldeman,^[7] and Oldeman.^[13]

methodology has been widely adopted, and GLASOD estimates of soil degradation are accepted as the best estimates available. Global estimates are, therefore, only rough approximations and should not be taken literally.

Effects on Soil Productivity and Food Production

Table 3 shows estimates of degradation and cumulative losses in soil productivity on a continental level for different land use types. The figures show that, overall, Africa and Central America are the most severely affected continents, in terms of both soil degradation and reduction of productivity; 65% of African and 75% of Central American cropland are affected by soil degradation. The overall loss in productivity in these two regions over the last years is estimated between 25% and 37%. On a continental basis, Africa has land reserves, but large areas of the continent are marginal for crop production, and their food production is therefore most seriously affected by degradation. Europe also shows relatively high trends in degradation, but mainly of pastures and forests. Productivity losses generally are low, as for North America and Oceania (i.e., mainly Australia), and these can most probably be compensated for by improvements in technology and input supply.

A more detailed picture of trends in food productivity for the main food cereals in selected countries with high population growth rates and agriculture-based economies is given in Fig. 1. The overall past trends in the different countries show a consolidation of per-capita food production. However, the projections until 2025 indicate clear decreases. This suggests that long-term food security in these countries is at stake.

SOIL DEGRADATION AND DECLINE IN PRODUCTIVITY

Over the past decades, the cumulative loss of productivity of cropland has been estimated at about 13%. However,

aggregate global food security does not appear to be under a significant threat.^[2,6] There is no conclusive evidence that soil degradation has in the past affected global food security. We cannot conclude to such relationship because

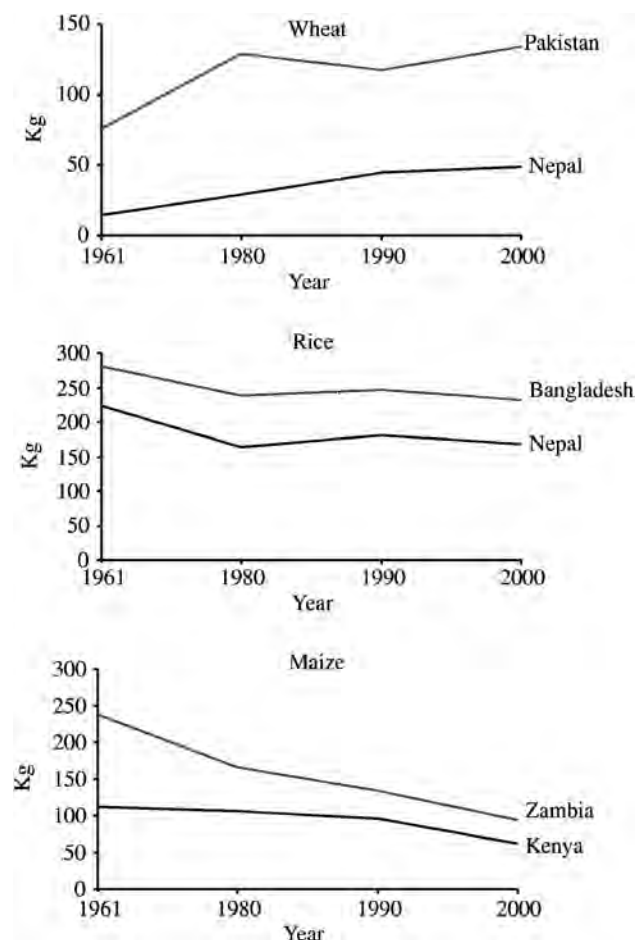


Fig. 1 Per-capita production of selected crops in countries with high population pressure.

Source: From Engelman & LeRoy^[2] and FAO.^[11]

1) the database on the extent and magnitude of various types of soil degradation is inadequate; and 2) the impact of soil degradation on land productivity is very site specific, often anecdotal and difficult to quantify at regional and global levels. In fact, globally, per-capita food production has increased by about 25% since 1961 when the first production surveys were conducted by Food and Agriculture Organization of the United Nations (UN).^[1,11] The yields per unit area of the major cereals (i.e., wheat, rice, and maize) have steadily increased and are rising.^[1,11] This, however, does not imply that sufficient food is—and will be—available in all countries and regions. Food is not necessarily produced where it is most needed. A top priority should therefore be to improve inventories of land degradation at regional, national, and subnational levels.

However, the evidence is conclusive that soil degradation affects food security at the subnational and national levels in the developing countries by the gradual decline of the land's productive capacity and that this trend will continue in the future. We therefore have to assume that, eventually, land degradation will constitute a serious threat to global food security by its particular impact on the developing countries. According to most scenarios, these countries are most vulnerable to degradation induced by increasing pressure on their land resources, the effects of climate change, and their inability to finance programs to rehabilitate affected areas and prevent further degradation and decline of productivity.

REFERENCES

1. FAO. *The State of Food Insecurity in the World*; Food and Agriculture Organization of the United Nations (FAO): Rome, 1999.
2. Engelman, R.; LeRoy, P. *Conserving Land: Population and Sustainable Food Production*; Population and Environment Program; Population Action International: Washington, 1995.
3. Saad, M.B. *Food Security for the Food-Insecure: New Challenges and Renewed Commitments*; Position Paper: Sustainable Agriculture; Centre for Development Studies, University College Dublin: Dublin, 1999.
4. Kindall, H.W.; Pimentel, D. Constraints on the expansion of the global food supply. *Ambio* **1994**, *23* (3), 198–205.
5. Lal, R.; Singh, B.R. Effects of soil degradation on crop productivity in East Africa. *J. Sustain. Agr.* **1998**, *13* (1), 15–36.
6. Scherr, S.J. *Soil Degradation—A Threat to Developing-Country Food Security by 2020?* Discussion Paper 27; International Food Policy Research Institute: Washington, 1999.
7. van Lynden, G.W.J.; Oldeman, L.R. *The Assessment of the Status of Human-Induced Soil Degradation in South and Southeast Asia*; International Soil Reference and Information Centre: Wageningen, 1997.
8. Greenland, D.; Bowen, G.; Eswaran, H.; Rhoades, R.; Valentin, C. *Soil, Water, and Nutrient Management Research—A New Agenda*; IBSRAM Position Paper; International Board for Soil Research and Management: Bangkok, 1994.
9. Lal, R. Soil management in the developing countries. *Soil Sci.* **2000**, *165* (1), 57–72.
10. Stocking, M.; Murnaghan, N. *Land Degradation—Guidelines for Field Assessment*; UNU/UNEP/PLEC Working Paper; Overseas Development Group; University of East Anglia: Norwich, 2000.
11. FAO. *FAOSTAT. Statistical Database of the Food and Agriculture Organization of the United Nations*; 2000. Available at <http://www.fao.org> (accessed January 2001).
12. Stocking, M.; Benites, J. *Erosion-Induced Loss in Soil Productivity: Preparatory Papers and Country Report Analyses*; Food and Agriculture Organization of the United Nations: Rome, 1996.
13. Oldeman, L.R. *Soil Degradation: A Threat to Food Security?* Report 98-01; International Soil Reference and Information Centre (ISRIC): Wageningen, 1998.

Food Security: Soil Degradation, Local and Eco-Regional Scale

Sara J. Scherr

Forest Trends, Washington, District of Columbia, U.S.A.

Abstract

Soil degradation is widespread, and its pace has certainly accelerated in developing countries. Current and future soil degradation hot spots include areas with degradation-prone soils, inadequately managed irrigation, and rapidly intensifying production without the technology or the economic incentives for good resource husbandry. Thus, while posing particular problems for the poor, soil degradation is likely to have far-reaching impacts on economic development in many agriculture-dependent countries.

INTRODUCTION

An estimated 16% of all agricultural land in developing countries, and a much higher proportion of cropland, has experienced significant soil degradation since the middle of the 20th century. Deterioration in the physical or chemical attributes of these soils has reduced agricultural yields and increased production costs. Large land areas—an estimated 5–7 million hectares (ha) per year—have gone out of production entirely.^[1,2] Degradation appears to have had little impact on food availability or prices in international markets because of the global capacity for substitution and the dominance of temperate producers.^[2] However, international trade plays a minor role in food supplies in most poor countries, and especially in most poor rural areas, largely because of a lack of income to purchase imported food and, in some cases, poor transport and market infrastructure. Thus, many areas undergoing serious soil degradation are experiencing reduced food supply and farm income, in some cases on a scale large enough to slow economic development. Degradation especially threatens the livelihoods of poor producers, who have limited access to capital inputs and for whom the underlying quality of agricultural soil is the principal determinant of farm productivity.

The period since World War II has seen remarkable growth in rural population, agricultural land area, and agricultural productivity in the developing world. Although the annual rural population growth rate declined from 2.2% between 1960 and 1965 to 1% between 1990 and 1995, the absolute number of rural dwellers grew by almost 40%, from 2.0 to 2.8 billion. Total growth rates of agricultural production in developing countries have rivaled or surpassed historical growth rates in industrialized countries, though not on a per capita basis. This growth came in part from extensive land clearing but

primarily through yield increases. It is not surprising that rural population increase, area expansion, and widescale intensification were associated with widespread soil resource degradation.^[2]

NATURE OF SOIL DEGRADATION

The nature of soil degradation has varied according to the particular pathways of agricultural intensification, reflecting different pressures and resource endowments. *Irrigated lands* have expanded and become highly productive as beneficiaries of Green Revolution technologies. However, poor water management has led to widespread salinization and waterlogging, while subtle nutrient management problems associated with multiple cropping have slowed yield increases. In *high-quality rainfed lands*, with naturally fertile, less weathered soils and lower climatic risks, cropping intensity also increased greatly. But inappropriate machinery use has sometimes led to soil compaction, and poor vegetation management has exposed soils to erosion. Substitution of organic inputs with chemical fertilizers has led to declining organic matter, acidification of vulnerable soils, and changes in soil fauna. In *densely populated, marginal areas*, low-quality and degradation-prone soils, which were traditionally managed through moderate- to long-fallow systems, are under continuous cultivation. Erosion from poor soil cover and nutrient depletion from inadequate management of organic matter and fertilizers are common results. In *extensively managed, marginal lands*, degradation has been caused principally by the land-clearing process itself, by nutrient depletion due to declining fallow periods, by widespread burning to control weeds and pests and provide ash for plant nutrition, and in rangelands by overgrazing. The importance of *urban agriculture* accelerated dramatically in

the 1980s, contributing to food security, income generation, and recycling of urban wastes but also to some environmental problems. Soil contamination with urban pollutants may pose a health hazard to consumers and reduce production, while insecure land access and tenure discourage sustainable grazing and soil management practices.

The concerns of policymakers regarding agricultural soil degradation relate not to its extent or severity but to its demonstrated impact on the following policy objectives:

- aggregate supply, stability, or price of agricultural output (i.e., if lands with degrading soils are a significant source of market supply for national consumers or export markets and alternative sources of supply not available or economic);
- agricultural income or economic growth (i.e., if soil degradation reduces agricultural income and its economic multiplier effects, through lower production or higher costs, and alternative sources of economic growth are limited or expensive to develop);
- consumption by poor farm households (i.e., if lands with degrading soils are a critical source of food security for subsistence or semisubsistence producers with few alternative livelihood options); or
- national wealth (i.e., if degradation reduces the long-term productive capacity of soil resources deemed to be of future economic or environmental significance, threatening the resource base and food security of future generations).^[2]

Environmental effects of soil degradation—including off-site damages from sedimentation and diminished watershed function as well as habitat alteration and reduced carbon sequestration—are also of interest to policymakers.

Considerable evidence of the economic and food security impacts of soil degradation has been produced since the late 1980s, although results must be interpreted cautiously.^[2] Weaknesses include uneven geographic coverage and numerous methodological and data problems.^[3] For many soil types, there is little information about the productivity impacts of degradation, the costs of rehabilitation, and the thresholds for soil quality below which investment in restoration will be uneconomic.^[4] Most studies use simplistic models of degradation–yield–income relationships, ignoring the dynamics of markets and farmer response that have been widely observed in field studies.^[5,6] Only a few studies explicitly document the impacts of soil degradation on consumption by the rural poor. Still, the weight of the evidence (summarized in the following section) suggests that the magnitude of agricultural supply and economic growth impacts of soil degradation justify more active policy engagement.

AGRICULTURAL SUPPLY

Asia

In Asia during the 1980s, it was estimated that over one-third of irrigated areas and over one-half of dry rainfed areas had experienced at least a 10% loss in productive potential, and 8% of irrigated and 10% of rainfed dry areas had suffered more than a 25% loss. Soil productivity losses of at least 20% from human-induced water erosion was well confirmed in eight countries, and presumptive evidence of such losses was documented in five other countries. Wind erosion was found to have little effect.^[7]

The 1995 expert survey found that soil degradation had moderate or worse impacts on agricultural productivity on one-tenth of all lands in South and Southeast Asia, with serious fertility decline or salinization affecting 15% of arable land.^[8]

Field-based evidence confirms supply effects. A synthesis of survey and experimental data in India concluded that 48% of soils had suffered a productivity loss of over 33%.^[9] Salinization and waterlogging in four villages of Uttar Pradesh, India, reduced paddy yield by 61% and wheat yield by 68% over 10 years.^[10]

In Bangladesh, deteriorating nutrient balance and organic matter decline led to an annual 0.36-ton decrease in total factor productivity in an intensive cropping system.^[11] In irrigated districts of the Pakistani Punjab, soil degradation lowered aggregate economic growth from 1966 to 1997 in four farm production systems by 0.58% per year; in the wheat–rice (*Triticum sativum*–*Oryza sativa*) system, resource degradation more than cancelled the effects of technological change.^[12] In China, grain yields would have been 30% higher between 1983 and 1989 in the absence of erosion, salinity, and degradation due to increased cropping intensity.^[13]

Africa

Soil experts estimated that in the 1980s over one-half of dry rainfed areas in Africa had experienced at least a 10% loss in productive potential, with irreversible productivity losses of at least 20% due to erosion in large parts of 11 countries.^[14] Cropland productivity loss from all types of soil degradation between the mid-20th century and 1990 was estimated to be 25%.^[15] In 1989, crop yield losses resulting from past erosion were calculated to range from 2% to 40%, with a mean of 6.2% for sub-Saharan Africa.^[16]

Evidence from field studies and cropping system models is variable. A Malawi study found crop-specific soil erosion impacts ranging from 4% to 11% per year yield losses.^[16] Models of maize production in four highland farming systems predicted significant yield declines, even with fertilizer use, although cotton

(*Gossypium* spp.) and coffee (*Coffea* spp.) yields were more stable. Fertilizer-induced soil acidification reduced highland maize yield to zero in 20 years; even with lime, yields were projected to drop by half in 30 years.^[17] National- and district-level regressions for maize (*Zea mays*) and sorghum (*Sorghum vulgare*) yields in Lesotho and for maize yields in Zimbabwe found no statistically significant relationship with erosion, possibly due to the dominant effect of climatic variation.^[18,19]

Central America

The cumulative loss of agricultural productivity due to soil degradation was estimated by soil experts to be especially high—nearly 37%—in Central American croplands.^[15] Most cost-benefit models in the region predict large yield that declines without soil conservation measures.^[20] Bioeconomic-model simulations predicted that between 1991 and 2000, soil erosion in Nicaragua would result in 16–21% reductions in output for sesame (*Sesamum indicum*), beans (*Phaseolus* spp.), maize, cattle, and cotton; large declines in coffee, cotton, and sesame exports; and price increases of over 25% for maize, beans, and cattle.^[21] Because they find ways to manage their soils, farmers generally report lower estimates of production impact when surveyed. For example, a national sample of smallholders in hilly El Salvador reported erosion-related productivity problems in less than one-fifth of sloping farm fields.^[22]

ECONOMIC GROWTH

Asia

The annual cost of soil degradation in South Asia in the early 1990s was estimated to be \$9.8–\$11.0 billion, the equivalent of 7% of aggregate agricultural gross domestic product (AGDP). Water and wind erosion accounted for over two-thirds of the loss, salinization and waterlogging for one-fifth, and soil fertility decline the rest.^[23] Other studies of China, India, Java, and Pakistan calculated impacts ranging from less than 1% to 5% of AGDP.^[2] In Java, one-year costs of erosion in 1984 equaled 4% of the annual value of rainfed farm output—the same order of magnitude as annual recorded growth in uplands production.^[24] In the irrigated wheat-rice system of the Pakistani Punjab, degradation also entirely offset productivity gains from other sources.^[12] The economic loss from degradation in China in the late 1980s was equal to China's entire budget for rural infrastructure investment, though less than 1% of AGDP.^[25] An econometric analysis of soil erosion and salinization using district-level data in China showed more serious impacts in poorer and more densely populated areas.^[26] Yet despite nutrient

depletion, the economic value of topsoil rose by nearly 8% between the 1950s and 1980s as a result of shifts to soil-preserving products, yield increases, and actions to reduce soil alkalinity.^[27]

Africa

Bojö^[28] evaluated evidence on the economic losses due to soil erosion from 12 studies completed in eight countries in sub-Saharan Africa. He found low gross annual immediate losses for hill agriculture in Madagascar, the Ethiopian highlands, Mali, and South Africa. Losses were significantly greater in Ghana, in another Ethiopian study, and in Zimbabwe for nutrient replacement alone. The gross discounted future loss from degradation varied from under 1% to 18%, and the gross discounted cumulative loss (which assumed a continued process of degradation over time) for five countries ranged from under 1% to 44%, of AGDP. Further estimates,^[29] based on replacement cost of depleted nutrients (calculated from nutrient balances predicted for 2000), are much higher than others based on productivity loss. Farm monitoring in Kenya found that the average cost of replacing depleted soil nutrients equaled 32% of average net farm income.^[30] A model was developed for Ghana to calculate the impacts of soil erosion, acidification, and nutrient depletion on different crops and agroecozones and trace their effects through the national economy. Comparing the baseline scenario over 8 years, with and without soil degradation, showed agricultural productivity losses of 2.9% per year. Even with greater fertilizer use, such losses would lower economic growth by one percentage point. In some scenarios, real GDP is reduced as much as 4.8% in the eighth year.^[31]

Central America

The few estimates of agricultural income effects in Central America are high. The annual cost of replacing nutrients lost through soil erosion in Costa Rica was estimated at 5.3–13.3% of annual value-added in agriculture that year.^[32] The average cost of soil erosion in areas planted to maize in Mexico represented 2.7% of AGDP, reaching 12.3% in the areas with highest erosion effects. Economic losses were nine times higher in the highlands and semi-arid regions than in the lowland tropics and four times higher without than with soil conservation.^[33] A model for Nicaragua projected that annual productivity loss due to erosion in 1991 would reduce GDP, imports, exports, and consumption in 2000 by 4–7% from the baseline scenario without erosion and investment by 9%. Erosion would raise the producer and domestic price indexes by 1.7% and 2.1%, respectively, compared with the baseline, while consumer price indexes would rise 4.0–5.8% for different social groups.^[2]

CONCLUSION

Soil degradation is widespread, and its pace has certainly accelerated in developing countries. Major productivity declines have resulted in some parts of Africa, Latin America, and Asia. The principal impacts appear to be on food consumption by the rural poor and agricultural income in areas where alternative livelihood options, sources of food supply, or non-agricultural development potential have been limited. Current and future soil degradation “hot spots” include areas with degradation-prone soils (particularly in sub-Saharan Africa), inadequately managed irrigation (particularly in South Asia), and rapidly intensifying production without the technology (many densely populated marginal lands) or the economic incentives (extensively managed marginal lands) for good resource husbandry.^[34] Thus, while posing particular problems for the poor, soil degradation is likely to have far-reaching impacts on economic development in many agriculture-dependent countries.

REFERENCES

- Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; Commonwealth Agricultural Bureau International: Wallingford, 1994; 99–118.
- Scherr, S.J. *Soil Degradation: A Threat to Developing Country Food Security in 2020?* Food, Agriculture and the Environment Discussion Paper No. 27; International Food Policy Research Institute: Washington, 1999.
- Scoones, I.; Toulmin, C. *Policies for Soil Fertility Management in Africa*; A Report Prepared for the Department for International Development; International Institute for Environment and Development, Drylands Programme and the Institute of Development Studies; The Environment Group: London and Brighton, 1999.
- Tengberg, A.; Stocking, M. Land degradation, Food security and agro-biodiversity—examining an old problem in a new way. In *Response to Soil Degradation*; Bridges, E.M., Hannam, I.D., Oldeman, L.R., Penning de Vries, F.W.T., Scherr, S.J., Sombatpanit, S., Eds.; Oxford & IBH Publishing Co. Pvt. Ltd: New Delhi, 2001; 171–185.
- Templeton, S.; Scherr, S.J. The effects of demographic and related microeconomic change on land quality in hills and mountains of developing countries. *World Dev.* **1999**, *27* (6), 903–918.
- Tiffen, M.; Mortimore, M.; Gichuki, F. *More People, Less Erosion: Environmental Recovery in Kenya*; John Wiley and Sons: Chichester, 1994; 311 pp.
- Dregne, H.E. Erosion and soil productivity in Asia. *J. Soil Water Conserv.* **1992**, *47* (1), 8–13.
- Van Lynden, G.; Oldeman, R. *Soil Degradation in South and Southeast Asia*; International Soil Reference and Information Centre for the United Nations Environment Programme: Wageningen, 1997.
- Seghal, J.; Abrol, I.P. *Soil Degradation in India: Status and Impact*; Oxford and IBH: New Delhi, 1994.
- Joshi, P.K.; Jha, D. *Farm-Level Effects of Soil Degradation in Sharda Sahayak Irrigation Project*. Working Papers on Future Growth in Indian Agriculture No. 1, Central Soil Salinity Research Institute, ICAR, and International Food Policy Research Institute, 1991.
- Pagiola, S. *Environmental and Natural Resource Degradation in Intensive Agriculture in Bangladesh*; Environmental Economics Series Paper 15, Environment Department; The World Bank: Washington, 1995.
- Ali, M.; Byerlee, D. Productivity growth and resource degradation in Pakistan's Punjab. In *Response to Land Degradation*; Bridges, E.M., Hannam, I.D., Oldeman, L.R., Penning de Vries, F., Scherr, S.J., Sombatpanit, S., Eds.; Oxford & IBH Publishing Co. Pvt. Ltd: New Delhi, 2001; 171–185.
- Huang, J.; Rozelle, S. Environmental stress and grain yields in China. *Am. J. Agr. Econ.* **1994**, *77*, 246–256.
- Dregne, H.E. Erosion and soil productivity in Africa. *J. Soil Water Conserv.* **1990**, *45* (4), 432–436.
- Oldeman, L.R. *Soil Degradation: A Threat to Food Security?* Report 98/01; International Soil Reference and Information Centre: Wageningen, 1998.
- Lal, R. Erosion-crop productivity relationships for soil of Africa. *Soil Sci. Soc. Am. J.* **1994**, *59* (3), 661–667.
- Aune, J.B.; Kullaya, I.K.; Kilasara, M.; Kaihura, F.S.B.; Singh, B.R.; Lal, R. Consequences of soil erosion on productivity and productivity restoration by soil management in Tanzania. In *Soil Quality and Sustainable Agriculture*; Lal, R., Ed.; Ann Arbor Press: Chelsea, 1997; 197–212.
- Böjör, J. *The Economics of Land Degradation: Theory and Applications to Lesotho*; The Stockholm School of Economics: Stockholm, 1991.
- Grohs, F. *Economics of Soil Degradation, Erosion and Conservation: A Case Study of Zimbabwe*; Wissenschaftsverlag Vauk Kiel KG: Kiel, 1994.
- Lutz, E.; Pagiola, S.; Reiche, C. *Economic and Institutional Analyses of Soil Conservation Projects in Central America and the Caribbean*; A CATIE-World Bank Project, World Bank Environment Paper No. 8; The World Bank: Washington, 1994.
- Alfsen, K.M.; Defranco, M.A.; Glomsrod, S.; Johnsen, T. The cost of soil erosion in Nicaragua. *Ecol. Econ.* **1996**, *16*, 129–145.
- Pagiola, S.; Dixon, J. *Land Degradation Problems in El Salvador*; Annex 7, El Salvador Rural Development Study Report 16253 - ES; World Bank: Washington, 1997.
- Young, A. *Land Degradation in South Asia: Its Severity, Causes and Effects upon the People*; Final Report for the Economic and Social Council of the United Nations; Food and Agriculture Organization of the United Nations, United Nations Development Programme, and United Nations Environment Programme: Rome, 1993.
- Repetto, R.; Magrath, W.; Welk, M.; Beer, C.; Rossini, F. *Wasting Assets*; World Resources Institute: Washington, 1989.

25. Rozelle, S.; Veeck, G.; Huang, J. The impact of environmental degradation on grain production in china's provinces. *Econ. Geogr.* **1997**, *73* (June), 44–66.
26. Rozelle, S.; Huang, J.; Zhang, L. Poverty, population, and environmental degradation in China. *Food Policy* **1997**, *22* (3), 229–251.
27. Lindert, P. *The Bad Earth? China's Agricultural Soils Since the 1930s*; Working Paper Series No. 83; Agricultural History Center, University of California: Davis, 1996.
28. Bojö, J. The costs of land degradation in sub-Saharan Africa. *Ecol. Econ.* **1996**, *16*, 161–173.
29. Drechsel, P.; Gyiele, L.A. *The Economic Assessment of Soil Nutrient Depletion: Analytical Issues for Framework Development*; International Board for Soil Resource Management: Bangkok, 2000; 80 pp.
30. Shepherd, K.D.; Soule, M.J. Soil fertility management in west Kenya. *Agr. Ecosyst. Environ.* **1998**, *71* (1–3), 133–147.
31. Alfsen, K.M.; Bye, T.; Glomsrod, S.; Wiig, H. Soil degradation and economic development in Ghana. *Environ. Dev. Econ.* **1997**, *2*, 119–143.
32. Solorzano, R.; De Camino, R.; Woodward, R. *Accounts Overdue: Natural Resource Depreciation in Costa Rica*; World Resources Institute: Washington, 1991.
33. McIntire, J. A review of the soil conservation sector in mexico. In *Economic and Institutional Analyses of Soil Conservation Projects in Central America and the Caribbean*; Lutz, E., Pagiola, S., Reiche, C., Eds.; A CATIE-World Bank Project, World Bank Environment Paper No. 8; The World Bank: Washington, 1994; 107–130.
34. Scherr, S.J.; Yadav, S. *Land Degradation in the Developing World: Implications for Food, Agriculture, and the Environment to 2020*; Food, Agriculture and Environment Discussion Paper 14; International Food Policy Research Institute: Washington, 1995.

Forest Soils

Nicholas Comerford

North Florida Research and Education Center, University of Florida, Quincy, Florida, U.S.A.

Thomas Fox

Forest Resources and Environmental Conservation, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, U.S.A.

Abstract

Forest Soils is the science of dealing with soils as a natural resource supporting forest ecosystems. It is a soil that has developed under, and is currently supporting, a forest cover, yet these soils may have a past history having once been cropland or pasture and then converted back to forest. Management of these soils and the forests they support encompass a range of intensities – from very intensive to a hands-off approach. The most significant aspect of the modal forest soil is its closed nutrient cycle with the forest. This is what often separates the concept of a forest soil from cropland soils. Other unique aspects of Forest Soils are the long time frame between harvests, the low erosion rates, the low rates of fertilization under intensive management, the presence of a forest floor and the potential to use very deep soil horizons for sustenance. The history of research of Forest Soils covers soil/site relationships, nutrient cycling, long-term productivity decline, soil fertility management, and off-site effects of management, among others. As management intensity increases and rotation age decreases the differences between forest soils and cropland soils converge.

INTRODUCTION

Forest soil science is a well-established field of study, having developed as a specific subdiscipline of soil science in Europe in the late 1800s. This is illustrated by both the number of textbooks and the dates of the texts published in this area (Table 1). Other soil scientists have asked: “If there is a subdiscipline called forest soils, why not also study corn or tomato soils?” The justification for forest soil science is twofold. First, a long-lived forest cover imposes a unique set of characteristics on the soil in which it grows. Second, and probably of equal importance, is that the soil characteristics and processes important to the forest soil scientist are often not of great concern to cropland soil scientists for whom the biological and economic time frames of food production are relatively short (Table 2). In order to understand “Why study forest soils?” it is useful to first define the science and then discuss the soil/ecosystem characteristics that are unique to forest soil. The Soil Science Society of America defines soil science as “the science dealing with soils as a natural resource on the surface of the earth including soil formation, classification and mapping, physical, chemical, biological, and fertility properties of soils per se; and these properties in relation to the use and management of soils.”^[1]

In this context, the following definition is offered. Forest soil science is “the science dealing with soils as a natural resource on the earth’s surface which supports a forest

ecosystem. It includes how the forest influences, and is influenced by, soil formation and soil physical, chemical, and biological properties. It addresses how soil properties affect the sustainable use of the soil for the production of multipurpose forest outputs including, but not limited to, wood products, wildlife, water, and the maintenance of ecosystem services.”

WHY FOREST SOILS?

A forest soil is one that developed under, and is currently supporting, a forest cover. Forest soils include soils that have developed under continuous forest cover, soils that are supporting a forest cover different from the native vegetation, and soils that were once in cropland or pasture and have been converted back to a forest. Therefore, the chemical and physical properties of a forest soil are determined by the conditions under which it was formed as well as its more recent land use history. At one time, forests covered approximately 50% of the terrestrial surface of the earth. That has been reduced to around 30% as forests were cleared for other land uses such as pasture and row crop agriculture. The remaining forests are often located on steep slopes or on soils that are rocky, infertile, and with excessive or limited moisture. Forests can be remote from markets and urban areas limiting development pressure, which, combined with land ownership patterns and environmental policy, are reasons why agriculture has not

Table 1 Textbooks in forest soils.

1893	Ramann, E. <i>Forstliche Bodenkunde und Standortlehre</i> ; Julius Springer: Berlin.
1908	Henry, E. <i>Les Sols Forestiers</i> ; Berger-Levrault & Co.: Paris.
1946	Lutz, H.J.; Chandler, R.F., Jr. <i>Forest Soils</i> ; John Wiley & Sons: New York.
1946	Wilde, S.A. <i>Forest Soils and Forest Growth</i> ; Chronica Botanica Co.
1950	Tamm, Olof. <i>Northern Coniferous Forest Soils</i> ; The Scrivener Press: Oxford.
1951	Hartmann, F. <i>Der Waldboden: Humus-, Boden- und Wurzeltypen als Standortsanzeiger</i> . Sterreichisches Produktivitäts-Zentrum: Wein, Germany.
1958	Wilde, S.A. <i>Forest Soils: Their Properties and Relation to Silviculture</i> ; Ronald Press: New York.
1969	Remezov, N.P.; Pogrebnyak, P.S. <i>Forest Soil Science</i> ; Israel Program for Scientific Translations: Jerusalem.
1971	Stefanovits, P. <i>Brown Forest Soils of Hungary</i> ; Akadémiai Kiad; Budapest, Hungary.
1977	Armson, K.A. <i>Forest Soils: Properties and Processes</i> ; University of Toronto: Toronto.
1979	Pritchett, W.L. <i>Properties and Management of Forest Soils</i> ; Wiley: New York.
1986	Binkley, D. <i>Forest Nutrition Management</i> ; Wiley: New York.
1987	Pritchett, W.L.; Fisher, R.F. <i>Properties and Management of Forest Soils</i> , 2nd Ed.; Wiley: New York.
1987	Attiwill, P.M. <i>Forest Soils and Nutrient Cycles</i> ; Melbourne University Press: Carlton, Victoria.
1987	Salas, Gonzalo de las. <i>Suelos y Ecosistemas Forestales: Con Enfasis en America Tropical</i> ; Instituto Interamericano de Cooperacion para la Agricultura: San Jose, Costa Rica.
2000	Fisher, R.F.; Binkley, D.; Pritchett, W.L. <i>Ecology and Management of Forest Soils</i> ; Wiley and Sons: New York.
2013	Osman, K.T. <i>Forest Soils: Properties and Management</i> ; Springer: New York.
2013	Binkley, D.; Fisher, R.F. <i>Ecology and Management of Forest Soils</i> , 4th Ed.; Wiley-Blackwell: New York.

claimed these soils. Forest soils can be highly variable but what they all have in common is that they support a forest ecosystem, managed or unmanaged.

Historically, forest soil management encompassed a range of intensities depending on the product to be

produced and the economic value of that product^[2] (Fig. 1). The intensity of management represents a significant difference in perspective between forest soil scientists and cropland soil scientists. In the most intensively managed forests, the time required to grow a forest to a useful size is about 7 years. In the least, the forest is preserved in perpetuity with no intent to harvest wood products. The relatively long management time frame means that soil processes and properties, which have minimal importance in cropland systems, can be controlling factors in forest soils (Table 2).

A closed nutrient cycle is one of the most significant processes that separates a forest soil from an agricultural soil. The time between management interventions, if any, allows organic soil horizons (forest floor) to form above the mineral soil surface. The degree of organic matter incorporation into the surface mineral soil will produce a range of

Table 2 Contrast of soil properties, processes, and management strategies between forest soils and cropland soils (management contrasts are for intensive forest management vs. intensive cropland management).

Property/process/ management	Forest soil	Cropland soil
Time between harvests	Seven years to indefinite	Annual to biannual
Nutrient cycling	Closed	Open
Erosion	Very low	High
Topography	Level to very steep	Level to moderate
Fertilization practices	Low amounts applied one to several times per rotation	Annual high amounts
Irrigation	Not employed	Employed
Soil temperature	Buffered by a forest canopy	More variable
Utilized soil depth	Deeper	Shallower
Stoniness	None to high	Low
Remoteness from markets	Close to remote	Close
Surface soil organic horizons	Present	Absent

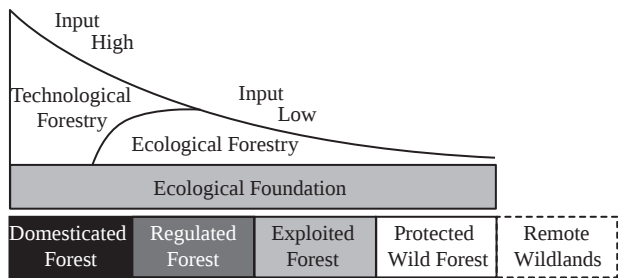


Fig. 1 The range of management intensity for a forest soil and forest stand.

Source: Adapted from Stone.^[2] ©1975 Laval University Press.

forest floor types. These organic horizons are unique to forest soils and strongly influence carbon and nutrient cycling in forest ecosystems. Cycling of some nutrients such as potassium is rapid while other nutrients such as nitrogen and phosphorus may be immobilized and only slowly released through time.

Due to the perennial nature of forest cover, and the infrequent management interventions during stand development, roots are capable of growing, surviving, and functioning at depth, allowing the forest stand to define a greater soil depth and to exploit water resources that short-lived agricultural crops might not find. The concept of the solum that includes the A, E, and B horizons and is thought of as the zone of pedogenic processes driven by biological activity does not always apply in forest soils where tree roots can extend throughout the C horizon and into the R horizons. Where cropland is most commonly defined by monocultures and where fertilization controls soil fertility, forest soils generally support a diverse plant community where groups of species may have their own strategy for using the soil's resources.

Wood products generally have low economic value compared to agricultural crops. Therefore, low-intensity management can be required in order to gain an economic return on the investment. In some circumstances, wood products are not the primary goal of management. That goal might be water yield and quality, wildlife, or maintenance of biodiversity. The latter objectives require an advanced understanding of how the entire forest ecosystem functions as well as of the multitude of consequences (on-site and off-site) that might come from human intervention. In this sense, Ecological Economics and the valuation of ecosystem services have been a part of forest management concepts for generations.

Consequently, forest soil scientists are often more interested with the O horizon, deep rooting, nutrient cycling, a tree's ability to thrive under native soil fertility, soil properties that influence the long-term productive potential of a forest stand that experiences periodic disturbances such as forest harvesting, climatic shifts, and anthropogenic inputs. Many of these are of little concern to cropland soil scientists.

In some intensively managed forests, such as eucalyptus in South America, forestry takes an agronomic approach where soil properties and processes are actively manipulated to optimize productivity. As forest management becomes more intensive, forest soil management moves toward the timescale of other cropland ventures, and the differences between forest soil science and other areas of soil science become less clear.

CONTRIBUTIONS OF FOREST SOILS TO SOIL SCIENCE

Like many scientific fields, the discipline of forest soils has evolved through time in order to better meet changing societal needs. Yet the accomplishments in forest soil science

during the 1900s testify to its importance in ecology, forest management, and forest conservation. The following is not meant to be a compilation of research in the area but is offered to show this field's breadth and depth.

FOREST ECOSYSTEM NUTRIENT CYCLING AND LONG-TERM PRODUCTIVITY DECLINE

Evaluating and developing forest management strategies based on nutrient cycling have been a collaborative effort of ecologists, silviculturalists, tree physiologists, and forest soil scientists. Nutrient cycling is often the basis for both sustainable soil management and forest harvesting schemes. A problem that constantly haunts forest managers is whether their harvesting regimes allow for sustainable forest productivity.^[3] Defining the soil's role in nutrient cycling as related to mineralization, exchange reactions, water regime, and root depth has been crucial in defining a site's ability to maintain sustainable forest growth.

SOIL DEVELOPMENT AND SOIL-SITE RELATIONSHIPS

One of the five soil-forming factors is the influence of vegetation. The influence of different species on soil formation is well known, as is the influence of the soil on forest ecosystems. This is best seen in the surface 20–40 cm of mineral soil. The study of forest soils has defined soil changes that occur under different types of forest vegetation. The research to describe and classify the different types of forest floor (mor vs. mull)^[4] is indicative of this type of influence. The ability of low-nutrient-demanding conifers to ameliorate a degraded soil and make it more productive for nutrient-demanding hardwood species has been well established.^[5] Podzolization has been linked with certain types of conifers that provide the acidic litter and soluble organic compounds that are responsible for sesquioxide movement in soils as well as soil acidification.

As forest ecosystems are related to the soils on which they grow, species have been matched to sites that meet their autecological requirements. Likewise, the productivity of a species will change with changes in soil characteristics. Soil-site studies have demonstrated broad general relationships between soil properties and forest ecosystem properties. However, we have to resolve the underlying causal relationships, most likely due to the highly empirical nature of the studies.^[6] Such work provides an appreciation of the range of site characteristics of economically important forest species and provides clues as to why their growth patterns vary on different soils.

FERTILITY OF FOREST SOILS

Soil fertility is probably the most studied topic in forest soil science. Evaluation of nutrient deficiencies and testing of

fertilization practices to correct deficiencies have provided productive forest management strategies throughout the world. Given the fact that many soils are not in cropland because of their inherent low fertility, commercially productive forests require nutrient management strategies that will allow large growth responses to minimal fertilizer inputs.

OFF-SITE EFFECTS OF FOREST MANAGEMENT

Probably, the most controversial environmental debates of the late 1900s in which forest soils played a role concerned the off-site effects of forest harvesting on water quality. Studies at the Hubbard Brook Experimental Forest forced scientists to evaluate whether harvesting could be responsible for serious nutrient losses and detrimental off-site effects. These articles set off a flurry of soil research on mineralization rates, water flow in forest soils, leaching losses through forest soils, and soil recovery from degradation. The stimulation was healthy in that it produced a compendium of information, from a wide range of landscapes, showing how soil treatments influence water quality.

The influence of acid deposition, both in the United States and in Europe, stimulated another round of research on forest soils. Soil's natural buffering against acidification, leaching of aluminum from soils, nitrate loading of soils by pollution, and amelioration practices to combat the effect of acid deposition were all topics of interest that stimulated research and provided a better understanding of the soil resource.

CONCLUSION

In conclusion, forest soil science is a crucial discipline in the management of our natural forest resources. Tall trees use the soil for physical support, as well as a source of water and nutrients. We will need to understand the relevant processes. As our information becomes more mechanistic, we should find it harder to separate the study of forest soil science from cropland systems because the difference is the dominance of different sets of processes working in a different management timescale. As our forests are considered a source of wood and pulp products as well as water, wildlife, and recreation, forest soil science will continue to contribute to both the knowledge base and the management of these natural renewable resources.

REFERENCES

1. Soil Science Society of America. *Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 1996.
2. Stone, E.L. Soils and man's use of forest land. In *Forest Soils and Forest Land Management*; Bernier, B., Winget, C.H., Eds.; Laval University Press: Quebec, 1975; 1–9.
3. Powers, R.F. On the sustainability of planted forests. *New Forest*. **1999**, *17*, 263–306.
4. Tamm, O. *Northern Coniferous Forest Soils*; The Scrivener Press: Oxford, 1950.
5. Stone, E.L. Effects of species on nutrient cycles and soil change. *Philos. T. R. Soc Lond.* **1975**, *271*, 149–162.
6. Carmean, W.H. Forest site quality evaluation in the United States. *Adv. Agron.* **1975**, *27*, 209–269.

Forest Soils: Calcium Depletion

Tom G. Huntington

U.S. Geological Survey (USGS), Augusta, Maine, U.S.A.

Abstract

Calcium (Ca) depletion in forest soils is a natural pedogenic process that can be accelerated by harvest removals and acidic deposition. Several lines of evidence support the fact that Ca depletion is an ongoing process in many forest soils. Direct remeasurements have shown Ca depletion in the Eastern United States and Europe. Mass balance studies strongly suggest that outputs exceed inputs in many intensively studied forest catchments. Indirect evidence from strontium isotope analysis, stem wood chemistry, fertilization experiments, and other studies is also consistent with ongoing Ca depletion. Ca depletion is a threat to forest ecosystems because decreases in Ca in soils can ultimately adversely influence health and vigor of sensitive trees, fish, snails, birds, and mollusks. Depletion is also a threat because it can delay forest and associated aquatic ecosystem recovery following reductions in emission and deposition of acidic compounds.

INTRODUCTION

Calcium (Ca) depletion in forest soils is the process by which Ca stored in the soil in exchangeable, organic, and mineral-bound forms is gradually lost in response to natural and anthropogenic mechanisms. The primary natural mechanism is the weathering of soil minerals and leaching of dissolved Ca to ground and surface water. The primary anthropogenic mechanisms are removal of Ca in forest products and the acceleration of leaching losses by elevated atmospheric acidic deposition.^[1,2] When the rate of Ca losses through leaching and harvest removals exceeds the rate of replenishment through atmospheric deposition and the weathering of primary minerals, soil Ca is depleted. These input and output processes are shown schematically in Fig. 1. Ca depletion results in decreased Ca availability to plants and lower Ca concentrations in soil, surface, and ground water.

WHAT IS THE EVIDENCE FOR CA DEPLETION?

There is a growing body of evidence that supports the hypothesis that forest soils in the Eastern United States and Europe are experiencing Ca depletion. The evidence can be grouped into three major categories: 1) direct remeasurements over time; 2) mass balance studies; and 3) indirect evidence that is consistent with decreasing Ca. Several studies in Europe^[3–5] and the United States^[6–9] have found significant decreases in exchangeable Ca in forest soil following decades of acidic deposition and uptake by aggrading forests. These remeasurement studies are the most direct evidence that forest soils in many regions are experiencing net Ca depletion.

Many studies have measured the input and output fluxes of Ca in forest ecosystems and concluded that the rates of

loss substantially exceed inputs.^[10–13] Examples of pools and fluxes of Ca determined in biogeochemical studies of this kind are summarized in Table 1. In these studies, investigators have attempted to measure atmospheric inputs, vegetation uptake, and soil leaching losses over a period of several years to determine average annual rates. The rate of weathering is usually estimated indirectly. There is more uncertainty in the weathering estimate than in other estimates, but it can frequently be constrained using silica export of strontium isotope ratios. The balance of evidence from the majority of intensively studied sites indicates that Ca outputs exceed inputs. The potential for weathering resupply is an important uncertainty, but the weight of evidence suggests that declines in Ca reserves in forest ecosystems are ongoing, arguing that weathering rates are too slow to replenish Ca.^[13,19]

There have also been a substantial number of studies that have provided insight into the status of forest soil Ca using indirect evidence. Strontium isotope ratio analysis has been used to infer ongoing Ca depletion in the Northeastern United States.^[20,21] Trends in stem wood chemistry have also been shown to be consistent with Ca depletion in red spruce dominated forests of the Northeastern United States.^[22,23] Studies have also reported that the failure of stream water to recover from acidification despite large reductions in sulfur deposition is indicative of Ca depletion.^[24]

Long-term trends in stream water chemistry and sulfate deposition at several forested watersheds at the Coweeta Hydrologic Laboratory in North Carolina and in the Shenandoah National Park in Virginia support the hypothesis that soil retention of atmospherically derived sulfate has decreased in years^[25,26] supporting the mechanism for ongoing Ca depletion.^[1,2] Many streams in the Eastern United States have experienced statistically significant decreases in Ca concentrations that may be related to soil

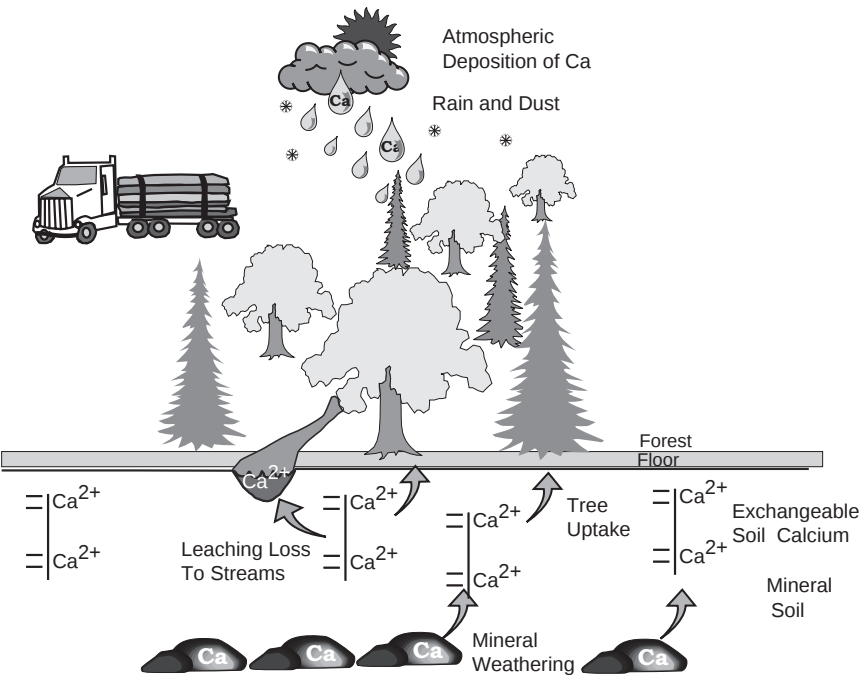


Fig. 1 Ca cycling in forest ecosystems. Inputs to the available pool of exchangeable soil Ca result from atmospheric deposition in precipitation and dust and the weathering of primary minerals. Outputs from this pool result from plant uptake and removal of wood products and leaching from the soil to streams.

Table 1 Soil Ca pools, ecosystem fluxes, and net depletion, from selected forest sites.

Location	References ^a	Predominant species	Soil exchange pool (kg ha ⁻¹)	Total soil pool (kg ha ⁻¹)	Net wood increment ^b (kg ha ⁻¹ yr ⁻¹)	Total atmospheric deposition (kg ha ⁻¹ yr ⁻¹)	Soil leaching (kg ha ⁻¹ yr ⁻¹)	Net depletion (kg ha ⁻¹ yr ⁻¹)
Stewart Co., Georgia	[14]	<i>Pinus taeda</i> L.	840	840	6	3.2	1.1	3.9
Duke Forest, North Carolina	[13]	<i>Pinus taeda</i> L.	2,130	4,900	14	8.1	8.1	13.6
Panola Mountain, Georgia	[15]	<i>Carya</i> , <i>Quercus</i> , <i>Liriodendron tulipifera</i> , <i>Pinus taeda</i> L.	2,200	9,500	10	2.3	2.9	10.7
Clemson, South Carolina	[16]	<i>Pinus taeda</i> L.	1,450	ND ^c	3.4	2.9	3.9	4.4
Oak Ridge, Tennessee	[13]	<i>Pinus taeda</i> L.	6,900	8,600	5.5	5.4	19.2	19.3
Huntington Forest, New York	[13]	<i>Acer saccharum</i> Marsh., <i>Fagus grandifolia</i> , Ehrh., <i>Acer rubrum</i> L., <i>Betula lutea</i> Michx. F., <i>Picea rubens</i> Sarg	606	120,000	2.7	5.1	15	12.6
Thompson Forest, Washington	[13]	<i>Pseudotsuga menziesii</i> , <i>Alnus rubra</i>	1,093	46,000	1.5	8.8	13	5.7
Howland, Maine	[13]	<i>Picea rubens</i> Sarg., <i>Abies balsamea</i>	288	59,000	5.1	4.6	8.0	8.5
Sabah, Malaysia	[17]	<i>Acacia mangium</i>	502	727	47 ^d	<4.0	ND	ND
Nordmoen, Norway	[13]	<i>Picea abies</i> L. Karst	194	29,000	6.1	5.4	9.0	9.8

(Continued)

Table 1 Soil Ca pools, ecosystem fluxes, and net depletion, from selected forest sites. (Continued)

Location	References ^a	Predominant species	Soil exchange pool (kg ha ⁻¹)	Total soil pool (kg ha ⁻¹)	Net wood increment ^b (kg ha ⁻¹ yr ⁻¹)	Total atmospheric deposition (kg ha ⁻¹ yr ⁻¹)	Soil leaching (kg ha ⁻¹ yr ⁻¹)	Net depletion (kg ha ⁻¹ yr ⁻¹)
Turkey lakes, Ontario	[13]	<i>Acer saccharum</i> Marsh., <i>Betula aleghaniensis</i> Britton, <i>Ostrya virginiana</i> Mill., <i>Acer rubrum</i> L.	671	59,000	1.3	9.3	88	80
St.-Hippolyte Quebec	[18]	<i>Acer saccharum</i> Marsh.	ND	ND	ND	3.0	13.3	ND

^aReference numbers correspond to reference list.

^bNet wood increment assumes that merchantable (stemwood) will be harvested and removed from the site. Whole-tree harvesting would result in greater removals.

^cND, not determined.

^dEstimated from growth at 45 months, 398 kg ha⁻¹ was reportedly removed during harvest.

Source: From Huntington.^[15]

depletion.^[10] Other studies have demonstrated positive responses in soil and tree health and vigor following fertilization with Ca suggesting that Ca was a limiting nutrient at these forest sites.^[27–29]

WHY IS CA DEPLETION A THREAT TO FOREST ECOSYSTEMS?

Ca depletion has been implicated in the failure of stream water alkalinity to recover in spite of significant decreases in acidic deposition.^[19,30,31] Decreases in Ca availability have been suggested as factors influencing health and vigor of fish,^[32] snails,^[33] birds,^[34] and mollusks.^[35] Ca depletion is of concern for a variety of reasons related to the many critical roles that Ca plays in tree physiology and the likelihood that Ca limitation will adversely influence many aspects of forest function.^[36] Ca limitation in forest ecosystems is thought to adversely influence disease resistance, wound repair, frost hardiness, and lignin synthesis in trees.^[36] Ca depletion is considered to be an especially serious threat to tropical forests growing on highly weathered soils in the tropics.^[17]

CONCLUSION

Ca depletion in forest soils is a natural pedogenic process that can be accelerated by harvest removals and acidic deposition. Several lines of evidence support the fact that Ca depletion is an ongoing process in many forest soils. Direct remeasurements have shown Ca depletion in the Eastern United States and Europe. Mass balance studies strongly suggest outputs exceed inputs in many intensively studied forest catchments. Indirect evidence from strontium isotope analysis, stem wood chemistry, fertilization experiments, and other studies is also consistent with ongoing Ca depletion. Ca depletion is a threat to forest ecosystems because decreases in Ca in soils can ultimately adversely influence

health and vigor of sensitive trees, fish, snails, birds, and mollusks. Ca depletion is also a threat because it can delay forest and associated aquatic ecosystem recovery following reductions in emission and deposition of acidic compounds.

ACKNOWLEDGMENT

This analysis was supported by the U.S. Geological Survey's (USGS) Water Energy and Biogeochemical Budgets, Atmospheric Deposition, and Hydrologic Benchmark Network Programs.

REFERENCES

1. Reuss, J.O.; Johnson, D.W. *Deposition and Acidification of Soils and Waters*; Ecological Studies 59; Springer-Verlag: New York, 1986.
2. Robarge, W.P.; Johnson, D.W. The effects of acidic deposition on forested soils. *Adv. Agron.* **1992**, *47*, 1–83.
3. Bergkvist, B.; Folkesson, L. The influence of tree species on acid deposition, proton budgets, and element fluxes in south Swedish forest ecosystems. *Ecol. Bull.* **1995**, *44*, 90–99.
4. Falkengren-Grerup, U.; Eriksson, H. Changes since soil, vegetation, and forest yield between 1947 and 1988 in beech and oak sites in southern Sweden deciduous forest soils. *For. Ecol. Manag.* **1990**, *38*, 37–53.
5. Wesselink, L.G.; Meiwes, K.J.; Matzner, E.; Stein, A. Long term changes in water and soil chemistry in spruce and beech forests, Solling, Germany. *Environ. Sci. Technol.* **1995**, *29*, 51–58.
6. Trettin, C.A.; Johnson, D.W.; Todd, D.E.J. Forest nutrient and carbon pools: A 21-year assessment. *Soil Sci. Soc. Am. J.* **1999**, *63*, 1436–1448.
7. Knoepp, J.D.; Swank, W.T. Long-term soil chemistry changes in aggrading forest ecosystems. *Soil Sci. Soc. Am. J.* **1994**, *58*, 325–331.
8. Richter, D.; Markewitz, D.; Wells, C.G.; Allen, H.L.; April, R.; Heine, P.R.; Urrego, B. Soil chemical change during

- three decades in an old-field loblolly pine (*Pinus taeda* L.) ecosystem. *Ecology* **1994**, *75*, 1463–1473.
9. Johnson, A.H.; Anderson, S.B.; Siccama, T.G. Acid rain and soils of the Adirondacks: I. Changes in pH and available calcium. *Can. J. For. Res.* **1994**, *24*, 193–198.
 10. Huntington, T.G.; Hooper, R.P.; Johnson, C.E.; Aulenbach, B.T.; Cappellato, R.; Blum, A.E. Calcium depletion in a southeastern United States forest ecosystem. *Soil Sci. Soc. Am. J.* **2000**, *64*, 1845–1858.
 11. Adams, M.B.; Burger, J.A.; Jenkins, A.B.; Zelazny, L. Impact of harvesting and atmospheric pollution on nutrient depletion of eastern hardwood forests. *For. Ecol. Manag.* **2000**, *138*, 301–319.
 12. Federer, C.A.; Hornbeck, J.W.; Tritton, L.M.; Martin, W.C.; Pierce, R.S.; Smith, C.T. Long-term depletion of calcium and other nutrients in eastern US forests. *Environ. Manag.* **1989**, *13*, 593–601.
 13. Johnson, D.W.; Lindberg, S.E. *Atmospheric Deposition and Forest Nutrient Cycling*; Springer-Verlag: New York, 1992.
 14. Huntington, T.G. Assessment of the potential role of atmospheric acidic deposition in the pattern of southern pine beetle infestation in the northwest Coastal Plain Province of Georgia, 1992–1995. In *U.S. Geological Survey Water Resources Investigation Report 96-4131*; U.S. Geological Survey: Reston, 1996; 75.
 15. Huntington, T.G. The potential for calcium depletion in forest ecosystems of southeastern United States: Review and analysis. *Global Biogeochem. Cycles* **2000**, *14*, 623–638.
 16. Johnson, D.W.; Kelly, J.M.; Swank, W.T.; Cole, D.W.; Van Miegrot, H.; Hornbeck, J.W.; Pierce, R.S.; Van Lear, D. The effects of leaching and whole tree harvesting on cation budgets of several forests. *J. Environ. Qual.* **1988**, *17*, 418–424.
 17. Nykvist, N. Tropical soils can suffer from a serious deficiency of calcium after logging. *Ambio* **2000**, *29*, 310–313.
 18. Hendershot, W.H.; Courchesne, F. Effect of base cation addition on soil chemistry in a sugar maple forest of the lower Laurentians Quebec. *Can. J. For. Res.* **1994**, *24*, 609–617.
 19. Turner, R.S.; Cook, R.B.; Van Miegrot, H.; Johnson, D.W.; Elwood, J.W.; Bricker, O.P.; Lindberg, S.E.; Hornberger, G.M. Watershed and lake processes affecting surface water acid–base chemistry. In *Acid Deposition: State of Science and Technology*, Rep. 10, Natl. Acid Precip., Assess. Program, Washington, DC, 1990; 1–167.
 20. Miller, E.K.; Blum, J.E.; Friedland, A.J. Determination of soil exchangeable-cation loss and weathering rates using Sr isotopes. *Nature* **1993**, *362*, 438–441.
 21. Bailey, S.W.; Hornbeck, J.W.; Driscoll, C.T.; Gaudett, H.E. Calcium inputs and transport in a base poor forest ecosystem as interpreted by Sr isotopes. *Water Resour. Res.* **1996**, *32*, 707–719.
 22. Bondietti, E.A.; Momoshima, N.; Shortle, W.C.; Smith, K.T. A historical perspective on divalent cation trends in red spruce stemwood and the hypothetical relationship to acidic deposition. *Can. J. For. Res.* **1990**, *20*, 1850–1858.
 23. Shortle, W.C.; Smith, K.T.; Minocha, R.; Larwence, G.B.; David, M.B. Acidic deposition, cation mobilization, and biochemical indicators of stress in health red spruce. *J. Environ. Qual.* **1997**, *26*, 871–876.
 24. Stoddard, J.L.; Jeffries, D.S.; Lükewille, A.; Clair, T.A.; Dillon, P.J.; Driscoll, C.T.; Forsius, M.; Johannessen, M.; Kahl, J.S.; Kellogg, J.H.; Kemp, A.; Mannio, J.; Monteith, D.T.; Murdoch, P.S.; Patrick, S.; Rebsdorf, A.; Skjelkvåle, B.L.; Stainton, M.P.; Traaen, T.; Van Dam, H.; Webster, K.E.; Wieting, J.; Wilander, A. Regional trends in aquatic recovery from acidification in North America and Europe. *Nature* **1999**, *401*, 575–578.
 25. Johnson, D.W.; Swank, W.T.; Vose, J.M. Simulated effects of atmospheric sulfur deposition on nutrient cycling in a mixed deciduous forest. *Biogeochemistry* **1993**, *23*, 169–196.
 26. Ryan, P.F.; Hornberger, G.M.; Cosby, B.J.; Galloway, J.N.; Webb, J.R.; Rastetter, E.B. Changes in the chemical composition of stream water in two catchments in the Shenandoah National Park, Virginia, in response to atmospheric deposition of sulfur. *Water Resour. Res.* **1989**, *25*, 2091–2099.
 27. Moore, J.-D.; Camire, C.; Ouimet, R. Effects of liming on the nutrition, vigour, and growth of sugar maple at the Lake Claire Watershed, Quebec, Canada. *Can. J. For. Res.* **2000**, *30*, 725–732.
 28. Long, R.P.; Horsley, S.B.; Lilja, P.R. Impact of forest liming on growth and crown vigor of sugar maple and associated hardwoods. *Can. J. For. Res.* **1997**, *27*, 1560–1573.
 29. Nilsson, L.W.; Wiklund, K. Nutrient balance and P, K, Ca, Mg, S and B accumulation in a Norway spruce stand following ammonium sulphate application, fertigation, irrigation drought and N-free-fertilisation. *Plant Soil* **1995**, *168*, 437–446.
 30. Kirchner, J.W.; Lydersen, E. Base cation depletion and potential long-term acidification of Norwegian catchments. *Environ. Sci. Technol.* **1995**, *29*, 1953–1960.
 31. Lawrence, G.B.; David, M.B.; Lovett, G.M.; Murdoch, P.S.; Burns, D.A.; Stoddard, J.L.; Baldigo, B.P.; Porter, J.H.; Thompson, A.W. Soil calcium status and the response of stream chemistry to changing acidic deposition rates in the Catskill Mountains of New York. *Ecol. Appl.* **1999**, *9*, 1059–1072.
 32. Bulger, A.J.; Cosby, B.J.; Webb, J.R. Current, reconstructed past, and projected future status of brook trout (*Salvelinus fontinalis*) streams in Virginia. *Can. J. Fish. Aquat. Sci.* **2000**, *57*, 1515–1523.
 33. Graveland, J. Avian eggshell formation in calcium-rich and calcium-poor habitats: Importance of snail shells and anthropogenic calcium sources. *Can. J. Zool.* **1996**, *74*, 1035–1044.
 34. Graveland, J.; van der Wal, R.; van Balen, J.H.; van Noordwijk, A.J. Poor reproduction in forest passerines from decline of snail abundance on acidified soils. *Nature* **1994**, *368*, 446–448.
 35. Wareborn, I. Changes in land mollusc fauna and soil chemistry in an inland district in southern Sweden. *Ecography* **1992**, *15*, 62–69.
 36. McLaughlin, S.B.; Wimmer, R. Tansley review no. 104. Calcium physiology and its role in terrestrial ecosystem processes. *New Phytol.* **1999**, *142*, 373–417.

BIBLIOGRAPHY

1. Johnson, D.W.; Todd, D.E. Nutrient cycling in forests of Walker Branch Watershed, Tennessee: Roles of uptake and leaching in causing soil changes. *J. Environ. Qual.* **1990**, *19*, 97–104.

Forest Soils: Major

Ken Van Rees

Department of Soil Science, University of Saskatchewan, Saskatoon, Saskatchewan, Canada

Abstract

Soils vary around the world and reflect the types of vegetation and other soil-forming factors and processes that have been acting upon them. Soils are much younger and less developed in boreal forests compared to those soils associated with tropical forests. Regardless of the degree of soil development, forest soils will continue to play an important role in ecosystem function.

INTRODUCTION

Soil is defined as the unconsolidated mineral or organic material that occurs at the earth's surface and is capable of supporting plant growth. Forest soils are those soils that have developed underneath forest vegetation.^[1] Forests cover approximately 29% of the world's land surface and play a key role in most ecosystems except for the tundra, deserts, some grasslands, and wetlands. Forests and forest soils contain about 60% of the carbon contained in the earth's land surface and thus are an integral part in the global carbon cycle.^[2] Soil development or genesis beneath forest ecosystems is influenced by a number of different factors. These factors include the type and nature of the forest vegetation, parent material and topography, climate, human or organism influence, and the amount of time that these factors have been influencing soil development.^[3] All these factors combined together can result in soils with different physical, chemical, and biological properties that are unique to those conditions and provide the framework for how soils are classified. Generally, there are two main classification systems used in the world: the U.S. soil taxonomic^[4] and the Food and Organization systems.^[5]

Soils types will vary around the world, but some generalizations can be made based on the major forest biomes that exist. The major forest biomes include boreal and coniferous forests, temperate deciduous/mixed forests, scrub and woodlands, temperate rain forest, tropical rain forests, and tropical monsoon/deciduous forests. A generalized vegetation map of the world showing most of these forest biomes is provided in [Fig. 1](#). Typical soil types associated with these forests are summarized in [Table 1](#) and discussed later for each major forest biome.

BOREAL AND OTHER CONIFEROUS FORESTS

Coniferous forests, otherwise known as boreal forests, are comprised mainly of spruce and pine trees with birch

and poplar appearing at the southern range. These forests, located in the northern hemisphere, extend southward from the tundra tree line and occur across Canada, Europe, and Asia. The soils for these forests are generally acidic and form on sandy deposits from glaciation and range in thickness from >1 m to very shallow (<1 m) underlain by bedrock (Precambrian shield). These forests receive about 90 cm/yr of precipitation or less and have low rates of evapotranspiration, which results in water moving through the soil profile.^[1] Because these forests have adapted to cooler climates, the decomposition of litter material is slow, resulting in thick forest floors compared to other forests as well as the accumulation of mosses and lichens. Soils of the boreal forest are relatively young compared to tropical soils because of glaciation; consequently, there are soils that have little to no development or no distinct soil horizons and are known as Entisols or Regosols. Sandy soils that have undergone more soil development, such as the development of a reddish colored B horizon, are called Inceptisols and are associated with pine forests growing on outwash plains. More soil development can result when organic acids that are released from slowly decomposing litter material bind with iron (Fe) and aluminum (Al) in the soil causing them to be leached deeper into the soil profile. This process is called podzolization and results in the development of a dark-colored B or spodic horizon (the accumulation of Fe, Al, and organic matter), which belongs to the Spodosol or Podzol soil groups. Often these soil profiles have a bleached-looking (light gray) surface horizon (E or Ae horizon) underlain by a darker colored (reddish brown) spodic horizon. Boreal forests also have extensive low-lying areas in the landscape that have high water tables resulting in organic soils known as Histisols or organic soils. These organic layers can be thicker than 40 cm and in some cases are few meters thick. These soils are generally unproductive for tree growth but are still important for wildlife and water quality.

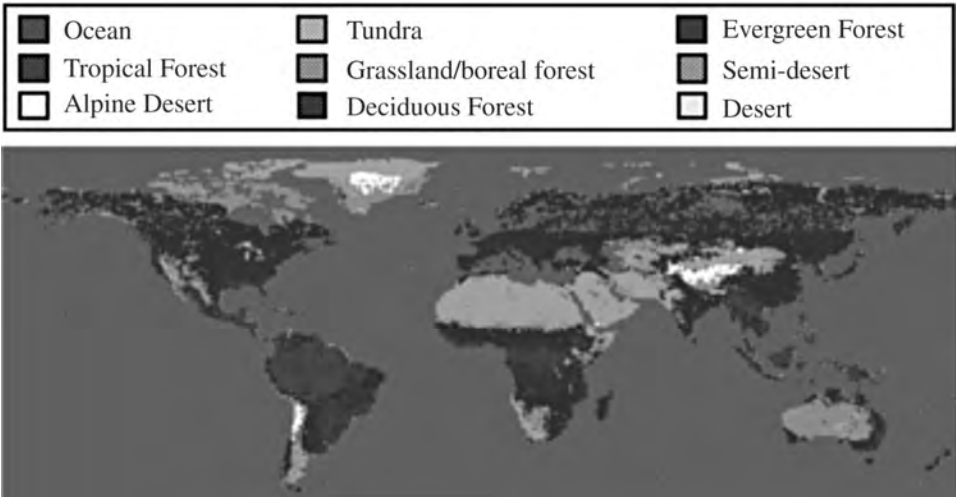


Fig. 1 World map of major vegetation types.
Source: From Murai, Honda, et al.^[6]

TEMPERATE DECIDUOUS FORESTS

Temperate deciduous forests are found in North America, Europe, and China and are comprised of oaks, beech, ash, maple, basswood, and yellow-poplar trees. These forests occur on a wide diversity of parent materials, but the soils are generally mildly acidic. Spodosols are common under oak/birch forests in Western Europe. In Central Europe and China, forests have developed on more neutral, clayey Mollisols or chernozemic soils, which are characterized by a surface mineral horizon containing high amounts of well-decomposed organic matter. This surface horizon enriched with organic matter makes these soils very productive for tree growth, and the accumulation of organic matter is believed to have originated from grassland conditions. In North America, deciduous forests have

developed on Alfisols in the north and on Ultisols in the south. Alfisols develop in cool to warm humid climates and are characterized by an accumulation of clay in the subsurface. This clay was eluviated or transported from upper horizons and deposited or illuviated in the B horizon to form an argillic horizon. These argillic horizons have more than 35% of their cation exchange sites saturated with bases such as calcium (Ca) and magnesium.^[3] The degree of soil development for Alfisols is greater than that for Inceptisols but less than that for Spodosols. Ultisols are similar to Alfisols in that they have developed an argillic horizon; however, Ultisols develop in more humid tropical areas with greater precipitation, resulting in more weathering and thus <35% base cations with more acidic conditions compared to Alfisols. Subsurface horizons in Ultisols tend to be redder in color, indicating an accumulation of Fe oxides, and are less productive than Alfisols.

Mixed forests are transition forests between boreal and deciduous forests. Mixed forests of aspen/spruce or birch, maple, and oaks growing with conifers occur in the northern hemisphere. In the southern hemisphere in Chile, Southern Brazil, South Africa, and New Zealand, one can also find mixed forests of broad-leaved evergreens and conifers. These forests will occur on Mollisols, Inceptisols, and Alfisols.

Table 1 Soil types associated with the major forest biomes of the world.

Major forest biomes	Area (ha × 10 ⁸)	Soil orders
Boreal and other coniferous forests	18.1	Spodosols, Histisols, Inceptisols, Entisols, and Gelisols
Temperate deciduous forests	11	Spodosols, Alfisols, Mollisols, and Ultisols
Temperate rain forests	3.2	Spodosols, Alfisols, Inceptisols, and Ultisols
Shrub and woodland	8.5	Aridisols, Mollisols, and Alfisols
Tropical rain forests	17.4	Ultisols, Inceptisols, and Oxisols
Tropical monsoon/deciduous forests	4.6	Ultisols and Oxisols

Source: From Landsberg & Gower,^[2] Pritchett & Fisher,^[7] and Waring & Schlesinger.^[8]

TEMPERATE RAIN FORESTS

Temperate rain forests consisting of conifers are located on the northwest coast of North America and broad-leaved evergreens in southern New Zealand, Chile, and southeastern Australia. These areas have milder, marine climates with relatively higher rainfalls. Spruce, cedar, hemlock, Douglas fir, and redwoods are the dominant species in North America, whereas beech and eucalyptus are found in the southern hemisphere. These forests have thick forest

floors, and soil types range from Spodosols and Inceptisols to Alfisols and Ultisols. These forests are highly productive and produce rather tall and large trees.

SHRUB AND WOODLAND

These forests are limited by the amount of rainfall and are less productive than most forests in the world. They range from sclerophyllous communities with Mediterranean-type climates to xerophytic shrublands where droughts can persist throughout the year.^[8] The sclerophyllous communities consisting of evergreen oaks or eucalyptus species are located in southwestern North America, Australia, Chile, South Africa, and Mediterranean regions. The xerophytic shrublands (oaks, pinyon, and ponderosa pines) are located in the interior of the Southwestern United States, Central Australia, Eurasia, southern Sahara, and the western regions of the Andes in central and southern South America. Soils that have developed under these conditions range from Mollisols and Alfisols to Aridisols. Aridisols, also known as dry soils, are common when evapotranspiration exceeds precipitation inputs, resulting in the accumulations of Ca carbonate, gypsum, soluble salts, or exchangeable sodium in the soil profile. The accumulation of soluble salts makes these soils more alkaline which are termed saline soils; however, if there are high concentrations of sodium on the exchange sites, these soils are classified as sodic soils, which can disperse organic matter and clay resulting in a black surface crust.^[3]

TROPICAL RAIN FORESTS

There are three distinct tropical rain forests in the world—the American, African, and Indo-Malaysian.^[1] These evergreen forests have high air temperatures and humidity, resulting in rapid decomposition of the forest floor, which is the opposite of that seen for boreal forests. Higher precipitation in these areas results in highly weathered soils, with soil development occurring deeper in the soil profile. The range of soil types is quite variable under tropical forests, and they have been studied less than temperate forests. Soils range from Inceptisols and Ultisols to Oxisols and Andisols. Oxisols are the oldest and hence most weathered soils, with intense leaching that removes a large part of the silica from silicate materials, resulting in more 1:1 type silicate clays or non-expanding clays. The sub-surface horizon characteristic of Oxisols is the oxic

horizon, which is very high in clay particles dominated with oxides of Fe and Al.^[3]

TROPICAL MONSOON/DECIDUOUS FORESTS

Tropical monsoon forests experience significant seasonal droughts (5 mo with <9 cm of rain) and are dominated by seasonal deciduous trees such as teak. These forests are found in Central and South America, Indo-Malaysia, and Africa, with the largest area of tropical deciduous forests located in Southeastern Asia. The soils are not as highly weathered as tropical rain forests but consist of soils from the Ultisols and Oxisols orders.

CONCLUSION

Soils vary around the world and reflect the types of vegetation and other soil-forming factors/processes that have been acting upon them. Soils are much younger and less developed in boreal forests compared to the soils associated with tropical forests. Regardless of the degree of soil development, forest soils will continue to play an important role in ecosystem function.

REFERENCES

1. Fisher, R.F.; Binkley, D. *Ecology and Management of Forest Soils*; John Wiley & Sons, Inc.: New York, 2000; 489 pp.
2. Landsberg, J.J.; Gower, S.T. *Applications of Physiological Ecology to Forest Management*; Academic Press: San Diego, 1997; 354 pp.
3. Brady, N.C.; Weil, R.R. *The Nature and Properties of Soils*; Prentice Hall: Upper Saddle River, 1996; 740 pp.
4. United States Department of Agriculture. *Keys to Soil Taxonomy*, 8th Ed.; Natural Resources Conservation Service; U. S. Government Printing Office: Washington, 1998; 326 pp.
5. Food and Agriculture Organization of the United Nations. *FAO-UNESCO Soil Map of the World*; Legend; UNESCO: Paris, 1974; Vol. 1, 59 pp.
6. Murai, S.; Honda, Y.; Asakura, K.; Goto, S. *An Analysis of Global Environment by Satellite Remote Sensing*; Institute of Industrial Science; University of Tokyo: Tokyo, 1990. Available at <http://www-cger.nies.go.jp/griddata/grid22.gif> (accessed May 2005).
7. Pritchett, W.L.; Fisher, R.F. *Properties and Management of Forest Soils*; John Wiley & Sons, Inc.: New York, 1987; 494 pp.
8. Waring, R.H.; Schlesinger, W.H. *Forest Ecosystems. Concepts and Management*; Academic Press: San Diego, 1985; 340 pp.

Forest Soils: Nutrient Cycling

Neil W. Foster

Natural Resources Canada, Sault Ste. Marie, Ontario, Canada

Jagtar S. Bhatti

Canadian Forest Service, Northern Forestry Centre, Edmonton, Alberta, Canada

Abstract

Nutrient cycling in forest ecosystems is controlled primarily by the following four key factors: climate, site, abiotic properties (topography and parent material), and biotic communities. The role of each factor in ecosystem nutrient dynamics is discussed and illustrated with selected examples from boreal, temperate, and tropical zones. The importance of ecosystem disturbance to nutrient cycling is examined briefly, since some nutrients are added or lost from forest ecosystems through natural (e.g., fire, erosion, and leaching) or human activity (harvesting and fertilization).

INTRODUCTION

Nutrients are elements or compounds that are essential for the growth and survival of plants. Plants require large amounts of nutrients such as nitrogen (N), phosphorus (P), carbon (C), hydrogen, oxygen, potassium (K), calcium, and magnesium (Mg) but only small amounts of others such as boron, manganese, iron (Fe), copper, zinc, and chlorine (micronutrients). Forest nutrient cycling is defined as the exchange of elements between the living and non-living components of an ecosystem.^[1] The processes of the forest nutrient cycle include nutrient uptake and storage in vegetation perennial tissues, litter production, litter decomposition, nutrient transformations by soil fauna and flora, nutrient inputs from the atmosphere and the weathering of primary minerals, and nutrient export from the soil by leaching and gaseous transfers.

Each nutrient element is characterized by a unique biogeochemical cycle. Some of the key features of the major nutrients are presented in [Table 1](#). Forest trees make less demand on the soil for nutrients than annual crops because a large proportion of absorbed nutrients are returned annually to the soil in leaf and fine root litter and are reabsorbed after biological breakdown of litter materials. Also, a large portion of nutrient requirement of trees are met through internal cycling as compared with agricultural crops.

INFLUENCE OF CLIMATE

Large-scale patterns in terrestrial primary productivity have been explained by climatic variables. In aboveground vegetation, nutrient storage generally increases in the following order: boreal < temperate < tropical forests ([Table 2](#)). In contrast, forest floor nutrient content and residence time

increase from tropical to boreal forests, as a result of slower decomposition in the cold conditions of higher latitudes.

In subarctic woodland soils and Alaskan taiga forests, nutrient cycling rates are low because of extreme environmental conditions.^[2] Arctic and subarctic forest ecosystems have lower rates of nutrient turnover and primary production because of low soil temperature, a short growing season, low net actual evapotranspiration (AET), and the occurrence of permafrost. Low temperature reduces microbial activity, litter decomposition rates, and nutrient availability and increases C accumulation in soil.

In contrast with high latitudes, conditions in a tropical forest favor microbial activity throughout the year, which generally results in faster decomposition except in situations with periodic flooding, soil desiccation, and low litter quality.^[3] Rates of plant material decay are an order of magnitude higher in tropical soils than in subarctic woodland soils. The low storage of C and high amount of litter production in highly productive tropical forests contrast with the high C storage and low litter production in boreal forests ([Table 2](#)).

INFLUENCE OF BIOTIC FACTORS

Nutrient cycles are modified substantially by tree species-specific controls over resource use efficiency (nutrient use per unit net primary production). Species vary widely in their inherent nutrient requirements and use.^[4] These effects can be split into the following two categories: the accumulation into living phytomass and the production of various types of nutrient-containing dead phytomass. Rapid accumulation of phytomass is associated with a net movement of nutrients from soil to vegetation. More than half of the annual nutrient uptake by a forest is typically returned

Table 1 Features of the major nutrient cycles.

Element	Uptake by the trees	Major sources for tree uptake	Limiting situations
Carbon	Atmosphere	Atmosphere	Atmospheric concentration may limit growth
Oxygen	Atmosphere	Atmosphere	Waterlogged soils
Hydrogen	Atmosphere	Atmosphere	Extremely acidic and alkaline conditions
Nitrogen	Soluble NO ₃ and NH ₄ ; N ₂ for N-fixing species	Soil organic matter; atmospheric N ₂ for N-fixing species	Most temperate forests, many boreal forests, and some tropical forests
Phosphorus	Soluble P	Soil organic matter; adsorbed phosphate and mineral P	Old soils high in Fe and aluminum, common in subtropical and tropical environment
Potassium, calcium, and magnesium	Soluble K ⁺ /soluble Ca ²⁺ /soluble Mg ²⁺	Soil organic matter; exchange complex and minerals	Miscellaneous situations and some old soils

to forest floor (litterfall) and soil (fine root turnover). The subsequent recycling of these nutrients is a major source of available nutrients for forest vegetation. The mean annual litterfall from aboveground vegetation increases from boreal regions to the tropics following the gradient of productivity (Table 2).

Table 2 Nutrient distribution in different forest ecosystems.

	Vegetation (Mg ha ⁻¹)	Forest floor (Mg ha ⁻¹)	Soil (Mg ha ⁻¹)	Residence time (year)
Carbon				
Boreal coniferous	78–93	37–113	41–207	800
Temperate deciduous	103–367	42–105	185–223	200
Tropical rain forest	332–359	7–72	2–188	120
Nitrogen				
Boreal coniferous	0.3–0.5	0.6–1.1	0.7–2.87	200
Temperate deciduous	0.1–1.2	0.2–1.0	2.0–9.45	6
Tropical rain forest	1.0–4.0	0.03–0.05	5.0–19.2	0.6
Phosphorus				
Boreal coniferous	0.033–0.060	0.075–0.15	0.04–1.06	300
Temperate deciduous	0.06–0.08	0.20–0.10	0.91–1.68	6
Tropical rain forest	0.2–0.3	0.001–0.005	0.06–7.2	0.6
Potassium				
Boreal coniferous	0.15–0.35	0.3–0.75	0.07–0.8	100
Temperate deciduous	0.3–0.6	0.050–0.15	0.01–38	1
Tropical rain forest	2.0–3.5	0.020–0.040	0.05–7.1	0.2

Nutrient availability is strongly influenced by the quantity and quality of litter produced in a forest. A high proportion of the variation in foliar N concentrations at the continental scale has been explained by differences between forest types, which in turn have large impact on litter quality and the nutrient content of forest floors. In many temperate and boreal forest ecosystems, microbial requirement for N increases or decreases with labile supplies of soil C. Increased microbial demand for N may temporarily decrease the N availability to trees during the initial decomposition of forest residues with a wide C/N ratio. Microbes immobilize N from the surrounding soil, relatively rapid for readily decomposable organic matter (needle litter) and more slowly for recalcitrant material (branches and boles).

Rates of net N mineralization are higher, and the retention of foliar N is lower in temperate and tropical than in boreal forest soils. The N limitation of productivity, therefore, is weak in tropical forests and increases from temperate to boreal and tundra forest systems. Trees may obtain organic N and P from the soil via mycorrhizae or by relocation from older foliage prior to abscission and, thereby, partly reduce their dependence on soil as a source of inorganic nutrients. Increased understanding of the fundamental relationships between soil properties and plant nutrients requirements will most likely come from the examination of plant–fauna–microbe interactions at root surfaces (rhizosphere), rather than in the bulk soil.

INFLUENCE OF ABIOTIC FACTORS

Forests have distinctive physiographic, floristic, and edaphic characteristics that vary predictably across the landscape within a climatically homogeneous region. Differences in the elemental content of parent material influence the tree species composition between and within a landscape unit. For example, wind deposited soils, which support hardwood or mixed wood forest, are likely to be fine textured with high nutrient supplying capacity. In contrast, outwash sands that often support pine forests are coarse textured and infertile.

Heterogeneity within the landscape results in sites differing in microclimatic conditions and physical and chemical properties, which produce different geochemical reaction rates and pools of available nutrients in soil. Soil type and topographic-microclimate interactions are important feedback that influence biological processes, such as the rate of N mineralization in soil. Low P availability is a characteristic of geomorphically old, highly weathered tropical, subtropical, and warm temperate soils.^[3] The type and age of parent material from which the soil is derived can influence the base status and nutrient levels in soil. Soils in glaciated regions are relatively young and rich in weatherable minerals. Mineral weathering is an important source of most nutrients for plant uptake, with the exception of N. Nutrient availability is regulated by the balance between weathering of soil minerals and precipitation, adsorption, and fixation reactions in soil.

Edaphic conditions can exert a strong influence on forest productivity and produce considerable variation in nutrient cycling processes. Soils with low N, P, or pH support trees with low litter quality (high in lignin and tannins that bind N) that decomposes slowly. Edaphic limitations on growth may be compensated by an increase in rooting density and depth. Some late-succession or tolerant species have a shallower root distribution relative to intolerant pioneer species and are adapted to sites where nutrients and moisture are concentrated at the soil surface. In contrast, nutrient uptake from subsoil horizons is more important in highly weathered warm temperate soils where nutrient depletion takes place deeper in the soil.

ROLE OF DISTURBANCE

Disturbances such as fire, harvesting, hurricanes, or pests affect nutrient cycling long after the event. In fire-dominated ecosystems, intensive wildfire results in a horizontal and vertical redistribution of ecosystem nutrients. Redistribution results from the combined effects of the following processes: 1) the oxidation and volatilization of live and decomposing plant material; 2) the convection of ash particles in fire generated winds; 3) the water erosion of surface soils; and 4) the percolation of solutes through and out of the soil. The relative importance of these processes varies with each nutrient and is modified by differences in fire intensity, soil characteristics, topography, and climatic patterns. Expressed as a percentage of the amount present in vegetation and litter before fire, the changes often follow the given order:

$$N > K > Mg > Ca > P$$

Harvesting removes nutrients from the site and interrupts nutrient cycling temporarily. The recovery of the nutrient

cycle from harvest disturbance is dependent partly on the rate of re-establishment of trees and competing vegetation. Re-vegetation may occur within months in the tropics, 2–5 years in temperate regions, and longer in boreal and tundra regions.^[5] Recovery assumes that the soil's ability to supply nutrients to plant roots has not been altered by disturbance. If nutrients cannot be supplied by the soil at rates sufficient to at least maintain the rate of growth of the previous forest, then fertilization may be necessary to maintain site productivity.

Nutrient cycling and the impacts of disturbance on nutrient cycling have been examined thoroughly in many representative world forests. The impact of natural disturbances and management practices on nutrient cycling processes is generally characterized of the stand or occasionally on a watershed basis. There is a growing demand from policy makers and forest managers for spatial estimates on nutrient cycling at local, regional, and national scales.

The availability of N, P, and K in soil largely determines the leaf area, photosynthetic rate, and net primary production of forest ecosystems. Forest management practices that produce physical and chemical changes in the soil that accentuate the cycle of nutrients between soil and trees may increase forest productivity. Clear-cut harvesting and site preparation practices (mechanical disturbance and slash burning) remove nutrients from soil in tree components and by increased surface runoff, soil erosion, and off-site movement of nutrients in dissolved form or in sediment transport. In the tropics, potential negative impacts associated with complete forest removal and slash burning are the greatest because a larger proportion of site nutrients are contained in the living biomass. Environmental impacts associated with clear-cutting and forest management in general are confounded by climatic, topographic, soil, and vegetation diversity associated with the world's forests. Best forest management practices can be utilized to control negative impacts on nutrient cycling.

REFERENCES

1. SSSA. *Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 1996; 134 pp.
2. Van Cleve, K.; Chapin, F.S., III; Dyrness, C.T.; Viereck, L.A. Element cycling in taiga forests: State factor control. *Bioscience* **1991**, *41*, 78–88.
3. Vitousek, P.M.; Stanford, R.L., Jr. Nutrient cycling in moist tropical forest. *Ann. Rev. Ecol. Syst.* **1986**, *17*, 137–167.
4. Cole, D.W.; Rapp, M. Elemental cycling in forest ecosystems. In *Dynamic Properties of Forest Ecosystems*; Reichle, E.D., Ed.; Cambridge University Press: London, 1981; 341–409.
5. Keenan, R.J.; Kimmins, J.P. The ecological effects of clear-cutting. *Environ. Rev.* **1993**, *1*, 121–144.

Forest Soils: Properties and Site Productivity

Paul A. Arp

Faculty of Forestry and Environmental Management, University of New Brunswick, Fredericton, New Brunswick, Canada

Helmut H. Krause

University of New Brunswick, Fredericton, New Brunswick, Canada

Abstract

Forest growth restrictions occur on soils with nutrient-poor parent materials, inefficient water and nutrient storage and retrieval capacities, unfavorable pH, low soil organic matter contents, and extreme texture (sand and clay). Some of these soil restrictions can—in part—be corrected by silvicultural means such as change of forest cover type, control of stand density, fertilization, irrigation, and artificial drainage.

INTRODUCTION

Site productivity is a key variable in forest management and yield: it depends on site quality—the inherent capability of the site to produce—and on management input.^[1] Site quality—in turn—is largely determined by soil, topography, and climate. Maximum production occurs when macronutrients [e.g., nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), and sulfur] and micronutrients [e.g., zinc (Zn), boron (B), and copper (Cu)] are consumed in certain amounts and proportions to each other for every unit of biomass produced, soil moisture and nutrients are available in sufficient quantities, and an extensive root system exists for efficient water and nutrient uptake. Related soil and site research throughout the world^[2] have, inter alia, produced simple guidelines for interpretations of soil survey data and complex site classification systems based on climate, geology, and soil properties.^[3]

In practice, the site index (SI = average height of dominant trees in a stand at a certain reference age, e.g., 50 years) is a local measure of site quality. This index is presumed to integrate the effects of climate, topography, and soil. Unfortunately, SI also varies with stand history, stocking level, and species composition. Soil properties that affect site quality and, therefore, forest growth and ecosystem functioning are highlighted in [Table 1](#). Examples of soils with contrasting productivity are shown in [Fig. 1](#).

INFLUENCE OF SOIL PHYSICAL PROPERTIES

Highest site productivity generally occurs on deep, friable, and organically enriched soils with well-functioning root systems. In other soils, root growth is often restricted by, e.g., mechanical impedance, lack of aeration, low

temperature, and/or the presence of toxic substances. Aside from shallow bedrock, roots encounter mechanical impedance in soils with high soil strength. High soil strength arises naturally in hardpans and fragipans, beneath the plough layer on old fields and on soils that may have been compacted by logging equipment. In such cases, tree growth generally decreases with decreasing depth of root-permeable soil. In mineral soils, roots usually encounter mechanical impedance at bulk density values of 1.3–1.75 Mg m⁻³, depending mainly on texture.

SOIL MOISTURE

A crucial requirement to maximize site utilization is the soil's ability to store water to compensate for precipitation deficits during periods of high evapotranspiration.^[4] For example, annual basal area increments of southern pines were shown to increase directly with number of days of adequate soil water quality. Available water is customarily estimated as the difference between the field capacity and the permanent wilting point of the root-accessible portion of each soil. These parameters—in turn—are primarily determined by soil texture and organic matter.

SOIL AERATION

Good air supply is essential for good root growth.^[5] Insufficient aeration occurs in soils with high water tables and in fine-textured, structureless soils that are water-saturated throughout most of the growing season. Insufficient aeration is usually indicated by gleyed and mottled soil layers. Most tree roots do not penetrate such soil layers. However, some species are tolerant of low soil aeration. For example, Douglas fir does not tolerate low soil

Table 1 Soil properties related to site quality.

Soil property	Relation to site quality	Optimal conditions
Mineralogical composition of parent material	Affects K, Mg, and Ca supply	Abundance of dark (mafic) minerals
pH	Affects nutrient availability	5–7
SOM content	Source of N and P, promotes structure	Presence of forest floor, accumulations in A and B horizons
Texture	Affects drainage, water, and nutrient storage	Sandy loam, loam, silt loam ^a
Structure	Promotes aeration, counteracts compaction, and high soil strength	Granular ^a
Consistency	Affects soil strength	Friable ^a
Drainage	Determines soil moisture regime and aeration status	Well and moderately well drained ^a
Depth of A horizon	Location of high biological activity and nutrient availability	>10 cm
Stone content	Improves thermal regime, limits soil volume for rooting	<15%
Depth to least permeable horizon	Delimits rooting space, improves moisture regime in coarse soils	> 80 cm
Depth to mottling	Reveals presence of excess water and lack of aeration	> 75 cm
Thickness of B horizon	Clay, SOM, and nutrient enriched zone	> 30 cm

^aClasses as defined in Agriculture Canada Expert Committee on Soil Survey, 1987. The Canadian System of Soil Classification.

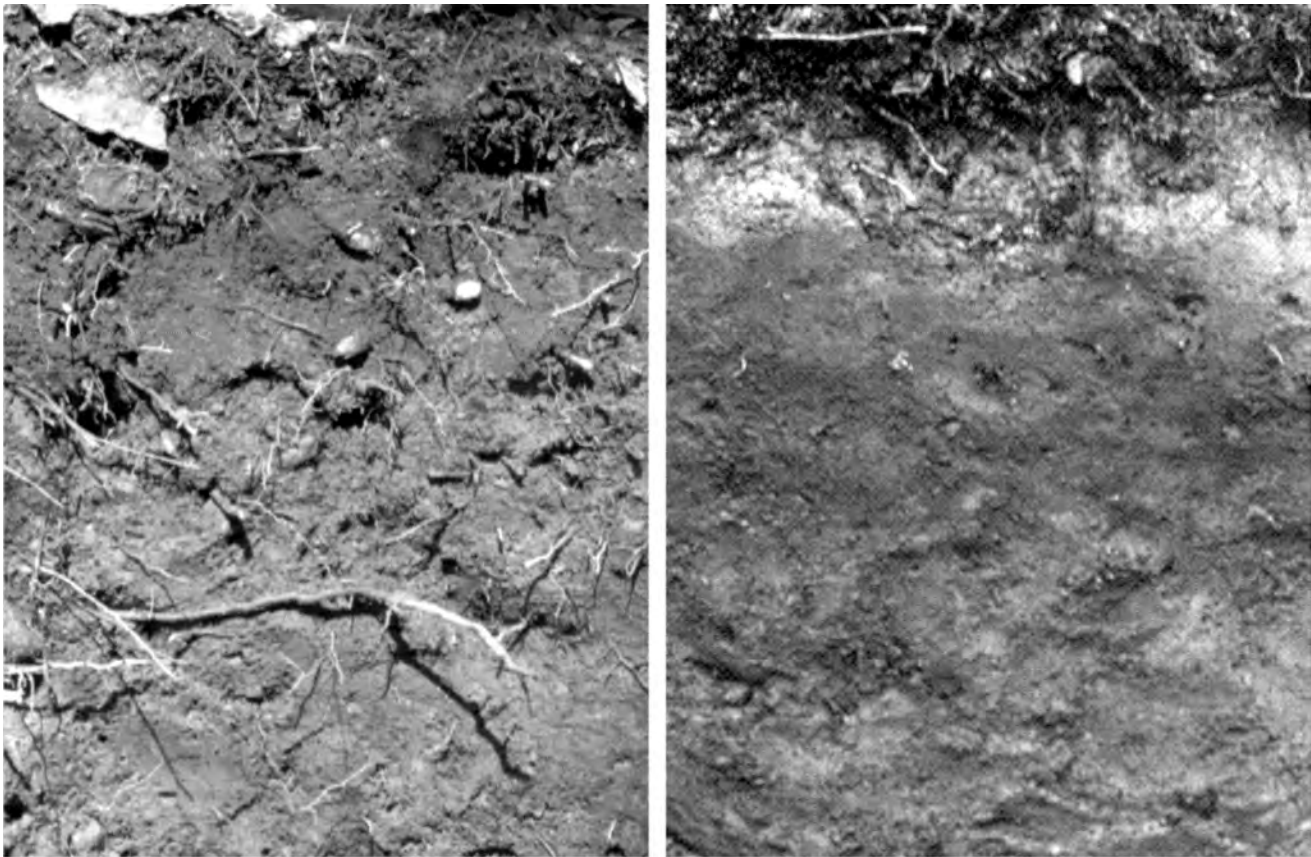


Fig. 1 Soils with contrasting productivity in the Atlantic Region of Northeastern North America. Left: brunisolic Podsol supporting vigorous forest of northern tolerant hardwoods. Right: gleyed Podsol supporting a pure black spruce stand. Note uniformly brown color, structure (granular to subangular blocky), and abundance of roots throughout.

aeration, but red alder, western cedar, sitka spruce, and western hemlock are adapted to shallow water tables.^[1] Controlled drainage to remove excess water often improves productivity and postdisturbance tree regeneration, especially when the roots continue to be constantly supplied with water by way of capillary rise from the water table below, as is often the case for sandy soils.^[6] On fine-textured soils, however, this benefit might not occur because the same capillary rise may reduce soil aeration.

SOIL TEMPERATURE

Soil temperature is an important parameter because it influences root growth and microbial activities, which—in turn—influence soil organic matter (SOM) decomposition and nutrient availability. Literature values for minimal (0–7°C) and optimal (10–25°C) soil temperatures vary over wide ranges, thereby reflecting the climatic adaptation of species. Soil temperatures generally decrease with increasing elevation and latitude and are generally lower under coniferous forests than deciduous forests: dense conifer canopies intercept much of the incoming solar radiation, thereby reducing the solar heating of the soils underneath, especially of wet soils. Delayed soil heating often implies delayed root growth, thereby shortening the effective growing season and hence forest growth. Fine roots are also sensitive to soil acidification:^[7] slowly soluble soil aluminum (Al) may convert to toxicologically active Al as the forest soil acidifies either naturally during the course of their development or on account of acid deposition.

INFLUENCE OF NUTRIENT SUPPLIES AND AVAILABILITIES

Matters dealing with nutrient supplies, retention, and availability are of particular importance to sustainable forest production as follows:

1. Long-term nutrient supplies of forest ecosystems need to be sustainable in principle. Additions of nutrients to each site come from external sources (atmospheric accretions, liming, and fertilizer) and from internal sources (weathering of primary soil minerals, biochemical cycling). Under natural conditions, origin and type of soil parent material are strong determinants of rate of weathering and related nutrient supplies.
2. Nutrient retention and release involve numerous reactions, including ion sorption and desorption on mineral and organic colloids; precipitation by metal oxides; complexation by humic substances; immobilization of nutrient ions by microorganisms; and SOM mineralization. All these processes are influenced by soil texture, SOM content, cation exchange capacity

(CEC), pH, base saturation, aeration, and prevailing soil temperature and moisture regimes.

3. Nutrient availability results from the net cumulative difference between supplies, immobilization, leaching, mineralization, and past uptake and is also controlled by chemical equilibria. Potential fertility of a soil can provisionally be appraised from simple field observations. For example, favorable contents and distribution of SOM and clay imply improved N, P, and base cation supply, and the presence of black minerals in the parent material may indicate K and Mg sufficiency. To be reliable, such observations must be supplemented by appropriate soil analysis.

The effect of soil parent material, via the continuing weathering of primary minerals, is often reflected in the species composition of natural forests. For example, in central regions of eastern North America, soils derived from calcareous shales often support, e.g., basswood, white ash, yellow-poplar, and hickory, while soils derived from acid sedimentary rocks support high percentages of beech, yellow birch, red maple, and certain oaks. As well, white cedar and eastern red cedar grow better on calcareous soils than on acid soils. In Sweden, site productivity is grouped by Ca content of the soil parent material: soils derived from Ca-poor substrates commonly support Scots pine at low productivity; soils with intermediate Ca availability support pine and mixed conifer forests with high productivity; and soils derived from basic igneous sedimentary substrates support productive stands of Norway spruce and hardwoods.

CA, MG, K, P, AND TRACE ELEMENTS

In areas where soils are very old and deeply weathered (e.g., tropical and subtropical regions in Africa, Asia, Australia, and Central and South America) or where soils have been heavily cropped, forest productivity is often limited because of limited supplies of Ca, Mg, K, P, and/or micronutrients such as Zn, Cu, and B.^[8] Such soils often consist of abundant accumulation of Al and iron (Fe) sesquioxides, which typically produce soils with high anion exchange capacities (AEC) but low CEC. As a result, nutrient elements such as K, Mg, and Ca are generally in short supply and severe losses of forest productivity are likely to occur when such soils are stripped of their natural forest vegetation. Such soils, furthermore, may lose their originally friable consistency with gradual SOM loss and become cementation when exposed to air and allowed to dry.

Areas that have been covered by glaciers and have a temperate to boreal climate can support productive forests set within the limits set by climate and soil depth. This can be attributed to the continuing release of Ca, Mg, K, and P through natural weathering of the ground-up bedrock (till). Soils in these areas generally have low accumulations of Fe

and Al sesquioxides, except in areas of high soil acidity where forest productivity may decrease as a result of advanced podsolization, i.e., the formation of Fe and organic matter cemented hardpans within the B horizon. Except for N, mineral deficiencies are infrequently encountered in these areas.

N AVAILABILITY

In boreal and subboreal climates, forest productivity is commonly restricted by low N availability, which is the result of slow SOM turnover rates. Under these conditions, the forest floor is the principal source of available N: the A horizon is—more often than not—strongly leached, poorly developed or absent, and N in the B layers is essentially unavailable. In these soils, N availability is further adversely affected by high soil acidity, impeded drainage, and restricted soil aeration. In contrast, soils with mesic or warmer temperature regimes have SOM-enriched A layers with high levels of available N. As shown in many studies,^[2] site quality on these soils rises with increasing depth of the A layer.

CONCLUSION

Forest growth restrictions occur on soils with nutrient-poor parent materials, inefficient water and nutrient storage and retrieval capacities, unfavorable pH, low SOM contents, and extreme texture (sand and clay). Some of these soil restrictions can—in part—be corrected by silvicultural means such as change of forest cover type, control of stand density, fertilization, irrigation, and artificial drainage. All of these growth restrictions have been considered in efforts to develop soil quality indices for monitoring sustainability of forest management

practices.^[9] Additional efforts are being made to quantify the relationships between soil and forest production by way of modeling.^[10,11]

REFERENCES

1. Pritchett, L.W.; Fisher, F.R. *Properties and Management of Forest Soils*, 2nd Ed.; Wiley: New York, 1987; 494 pp.
2. Carmean, W.H. Forest site quality evaluation in the United States. *Adv. Agron.* **1975**, *27*, 209–269.
3. Klinka, K.; van der Horst, W.D.; Nuszdorfer, F.C.; Harding, R.G. An ecosystematic approach to forest planning. *For. Chron.* **1980**, *56*, 97–103.
4. Zahner, R. Water deficit and growth of trees. In *Water Deficits and Plant Growth*; Kozlowski, T.T., Ed.; Academic Press: New York, 1968; 191–254.
5. Drew, M.C. Soil aeration and plant root metabolism. *Soil Sci.* **1992**, *154*, 259–268.
6. Wilde, S.A.; Iyer, J.G.; Tanzer, C.; Trautman, W.L.; Watterston, K.G. *Growth of Wisconsin Coniferous Plantations*; Research Bulletin 262; University of Wisconsin: Madison, 1965; 80 pp.
7. Reuss, J.O.; Walthall, P.M.; Roswall, E.C.; Hopper, R.W.E. Aluminum solubility, calcium–aluminum exchange, and pH in acid forest soils. *Soil Sci. Soc. Am. J.* **1990**, *54*, 374–380.
8. Stone, E.L. Microelement nutrition of forest trees: A review. In *Forest Fertilization—Theory and Practice*, Symposium on Forest Fertilization; Tennessee Valley Authority: Knoxville, 1968; 132–175.
9. Burger, J.A.; Kelting, D.L. Using soil quality indicators to assess forest stand management. *Forest. Ecol. Manag.* **1999**, *122*, 155–166.
10. Kimmins, J.P.; Scoullar, K.A. *FORCYTE 10*; Faculty of Forestry, University of British Columbia: Vancouver, 1983; 112 pp.
11. Bhatti, J.S.; Foster, N.W.; Oja, T.; Moayeri, M.H.; Arp, P.A. Modelling potentially sustainable biomass productivity in jack pine forest stands. *Can. J. Soil. Sci.* **1998**, *78*, 105–113.

Freeze and Thaw: Diurnal

Joseph L. Pikul

Northern Grain Insect Research Lab (NGIRL), U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Brookings, South Dakota, U.S.A.

Abstract

Diurnal freezing and thawing of soils are a worldwide phenomenon impacting many biological and physical aspects of agricultural soils. Available management options to reduce soil-freezing events may be limited to manipulation of crop residues. Surface cover creates a unique microclimate, affording thermal protection to soil and thereby reducing the incidence and severity of frozen soil. The reduction of soil-freezing events could be expected to reduce soil water evaporation, soil surface crusting, and undesirable movement of agricultural chemicals.

INTRODUCTION

In northern climates throughout the world, periodic freezing and thawing of soil are common. Sharratt et al.^[1] estimated that approximately 30% of the earth's landmass is periodically frozen, and that an additional 20% of the landmass is underlain by permafrost. Some of the world's most productive soil lies within regions where soil may be seasonally or diurnally frozen. Freezing and thawing of soil exert a profound effect on soil water distribution, solute movement, soil physical and chemical condition, biological processes, and hydrology.^[1–4] There are few land-management options available to lessen the effect of freezing weather on the soil thermal environment. However, maintaining surface cover can reduce the incidence and penetration depth of frozen soil^[5] on croplands. This entry discusses the diurnal soil freeze–thaw and soil–water movement associated with night-time radiative freezing of the soil surface.

SOIL FREEZING AND WATER MOVEMENT

Nocturnal soil freezing and daytime thawing cycles are numerous in late autumn and early spring.^[6] About 100 diurnal freezing and thawing cycles, recorded at a 1-cm depth, have been observed in a year in mountain regions.^[7]

Night-time freezing, termed *radiation frost*,^[8] is a consequence of rapid loss of long-wave radiation from the earth to the atmosphere on cloudless nights.

When moist soil begins to freeze, as during a radiation frost, water migrates upward to the freezing front (surface) and is held there as ice. During subsequent daytime thaw, near-saturated soil conditions may develop at the surface.^[9] Differences in soil water

potential within the soil profile are responsible for water movement within soil. Water moves from regions of high water potential (wet soil) to low water potential (dry soil). How fast water moves within soil depends on the water potential gradient (driving force) and the ability of soil to transmit water (hydraulic conductivity). In the case of soil freezing, very large water potential gradients develop as a consequence of rather small temperature gradients.

As the surface cools and soil temperature dips below 0°C, a fraction of the water freezes within the soil pores.^[10] Ice particles remain separated from solid particles by a film of water; the thickness of which depends upon temperature, pore size distribution, solutes in pore water, and freezing and thawing history.^[11] Ice formation in soil pores effectually dries the soil and results in a reduction of water potential. A water potential gradient is then established across the freezing front, creating the conditions whereby water flows from unfrozen soil toward the freezing front. Water flows toward the region of low potential.

Repeated saturation and subsequent drying of the soil surface, as occurs during diurnal freeze and thaw cycles, may accelerate surface crusting. However, the identification of direct consequences of freeze–thaw on soil structural stability is confounded due to soil type, soil water content at freezing, and characteristics of cyclic freezing and thawing.^[12]

FIELD STUDIES

Microenvironment created by the presence or absence of surface residue strongly affects surface energy exchange and, consequently, soil thermal environment. Pikul and Allmaras^[9] revealed effects of crop residue cover on soil

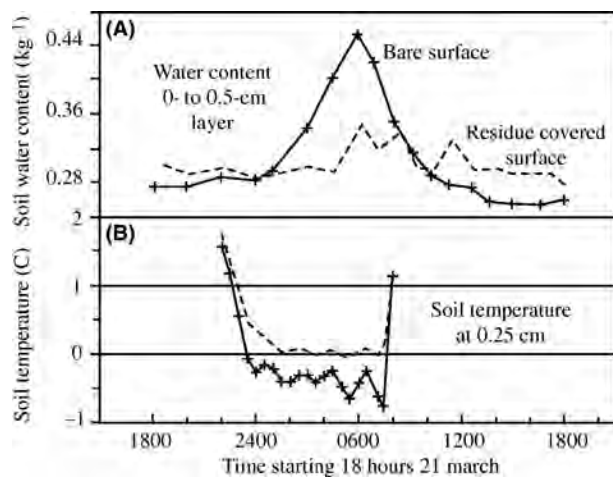


Fig. 1 Water redistribution during one freeze-thaw cycle: (A) soil water content (ice and liquid) of the surface to 0.5-cm depth and (B) soil temperature at 0.25 cm. Soil froze to a depth of 1.5 cm on the bare surface treatment but did not freeze under residue cover.

water movement during consecutive freezing and thawing cycles. Their field studies included both residue-covered and bare plots. Water redistribution during one freeze-thaw cycle described by Pikul and Allmaras^[9] is shown in Fig. 1. The soil did not freeze on plots with residue cover but froze to about 1.5 cm on plots having a bare surface. Soil water content under residue cover remained essentially unchanged throughout the night. On bare plots, soil water content of the surface, 0.5-cm layer (Fig. 1), increased from 0.28 kg/kg at 12:00 P.M. to 0.45 kg/kg at 6:00 A.M. Soil water content includes both liquid and ice. The quantity of water redistributed, as a consequence of soil freezing, depends on soil water content at the onset of soil freezing.^[13] In this example, both the bare surface and residue-covered plots entered the freezing cycle at about the same water content (Fig. 1). Therefore, the sharp increase in soil water on the bare surface plot was a result of steep water potential gradients that developed as a consequence of soil water freezing.

Concurrent with the decrease in soil temperature to below freezing was an increase in soil water content in the bare surface plot (Fig. 1). The onset of soil freezing in the bare surface plot was at 11:30 P.M., and significant water redistribution began shortly thereafter. Minimum soil temperature at 0.25 cm was -0.8°C . Soil temperature in residue-covered plots hovered near 0.0°C during the night (Fig. 1).

SIMULATION STUDIES

Heat and water flow in freezing soil have been modeled using several approaches with varying degrees of complexity. All models require detailed knowledge of

physical, chemical, hydrologic, and thermal properties of the soil. Generally, the most complex models include provisions for frost heave and overburden pressures.^[14] However, heat and water flow during daily freeze and thaw cycles have been successfully simulated by excluding frost heave.^[15] Cary^[16] provided an example of a simulation model that included solute transport and frost heave. The Simultaneous Heat and Water model,^[17] one of the more detailed models of snowmelt and soil freezing, has been used to simulate soil freezing, thawing, snowmelt, and water runoff.

CONCLUSION

Diurnal freezing and thawing of soils are a worldwide phenomenon impacting many biological and physical aspects of agricultural soils. Available management options to reduce soil-freezing events may be limited to manipulation of crop residues. Surface cover creates a unique microclimate, affording thermal protection to soil and thereby reducing the incidence and severity of frozen soil. The reduction of soil-freezing events could be expected to reduce soil water evaporation, soil surface crusting, and undesirable movement of agricultural chemicals.

REFERENCES

1. Sharratt, B.S.; Radke, J.K.; Hinzman, L.D.; Iskandar, I.K.; Groenevelt, P.H. Physics, chemistry, and ecology of frozen soils in managed ecosystems: An introduction. In *Proceedings of the International Symposium on Physics, Chemistry, and Ecology of Seasonally Frozen Soils*, Fairbanks, Alaska, Jun 10–12, 1997; CRREL Special Report 97–10; Iskandar, I.K., Wright, E.A., Radke, J.K., Sharratt, B.S., Groenevelt, P.H., Hinzman, L.D., Eds.; U.S. Army Cold Regions Research and Engineering Lab: Hanover, 1997; 1–7.
2. Williams, P.J. The seasonally frozen layer—Geotechnical significance and needed research. In *Proceedings of the International Symposium on Physics, Chemistry, and Ecology of Seasonally Frozen Soils*, Fairbanks, Alaska, Jun 10–12, 1997; CREEL Special Report 97–10; Iskandar, I.K., Wright, E.A., Radke, J.K., Sharratt, B.S., Groenevelt, P.H., Hinzman, L.D., Eds.; U.S. Army Cold Regions Research and Engineering Lab: Hanover, 1997; 9–15.
3. Ellsbury, M.M.; Pikul, J.L., Jr.; Woodson, W.D. A review of insect survival in frozen soils with particular reference to soil dwelling stages of corn rootworms. *Cold Reg. Sci. Technol.* **1998**, 27, 49–56.
4. Kane, D.L. Snowmelt infiltration into seasonally frozen soils. *Cold Reg. Sci. Technol.* **1980**, 3, 153–161.
5. Pikul, J.L., Jr.; Zuzel, J.F.; Greenwalt, R.N. Formation of soil frost as influenced by tillage and residue management. *J. Soil Water Conserv.* **1986**, 41, 196–199.

6. Hershfield, D.M. The frequency of freeze-thaw cycles. *J. Appl. Meteorol.* **1974**, *13*, 348–354.
7. Fahey, B.D. An analysis of diurnal freeze-thaw and frost heave cycles in the Indian peaks region of the Colorado front range. *Arctic Alpine Res.* **1973**, *5*, 269–281.
8. Oke, T.R. *Boundary Layer Climates*; Methuen and Co.: London, 1978.
9. Pikul, J.L., Jr.; Allmaras, R.R. Hydraulic potential in unfrozen soil in response to diurnal freezing and thawing of the soil surface. *T. ASAE* **1985**, *28*, 164–168.
10. Farouki, O.T. *Thermal Properties of Soils*; CRREL Monograph 81-1; U.S. Army Cold Regions Research and Engineering Lab: Hanover, 1981.
11. Anderson, D.M.; Tice, A.R. Predicting unfrozen water contents in frozen soils from surface area measurements. *Highw. Res. Rec.* **1972**, *393*, 12–18.
12. Dagesse, D.F.; Groenevelt, P.H.; Kay, B.D. The effect of freezing cycles on water stability of soil aggregates. In *Proceedings of the International Symposium on Physics, Chemistry, and Ecology of Seasonally Frozen Soils*, Fairbanks, Alaska, Jun 10–12, 1997; CREEL Special Report 97-10; Iskandar, I.K., Wright, E.A., Radke, J.K., Sharratt, B.S., Groenevelt, P.H., Hinzman, L.D., Eds.; U.S. Army Cold Regions Research and Engineering Lab: Hanover, 1997; 537–543.
13. Cary, J.W.; Papendick, R.I.; Campbell, G.S. Water and salt movement in unsaturated frozen soil: Principles and field observations. *Soil Sci. Soc. Am. J.* **1979**, *43*, 3–8.
14. O'Neill, K. The physics of mathematical frost heave models: A review. *Cold Reg. Sci. Technol.* **1983**, *6*, 275–291.
15. Pikul, J.L., Jr.; Boersma, L.; Rickman, R.W. Temperature and water profiles during diurnal soil freezing and thawing: Field measurements and simulation. *Soil Sci. Soc. Am. J.* **1989**, *53*, 3–10.
16. Cary, J.W. A new method for calculating frost heave including solute effects. *Water Resour. Res.* **1987**, *23*, 1620–1624.
17. Flerchinger, G.N.; Seyfried, M.S. Modeling soil freezing and thawing, and frozen soil runoff with the SHAW model. In *Proceedings of the International Symposium on Physics, Chemistry, and Ecology of Seasonally Frozen Soils*, Fairbanks, Alaska, Jun 10–12, 1997; CREEL Special Report 97-10; Iskandar, I.K., Wright, E.A., Radke, J.K., Sharratt, B.S., Groenevelt, P.H., Hinzman, L.D., Eds.; U.S. Army Cold Regions Research and Engineering Lab: Hanover, 1997; 537–543.

Frozen Soils: Water Relations

John M. Baker

U.S. Department of Agriculture—Agriculture Research Service (USDA-ARS),
St. Paul, Minnesota, U.S.A.

Abstract

Heat and water relations in frozen soil are difficult to study. Researchers are hampered by the inability to separately measure water and ice contents at similar spatial scales and by the complications arising from simultaneous, coupled transport of heat and water in a deformable medium. Formal theory describing phase equilibrium in ideal, uniform porous medium is well established but application of that theory to transient phenomena in structured, plastic, soils with complex mineralogy is elusive.

INTRODUCTION

Generally, soils freeze from the surface downward, due to radiative and convective heat loss at the surface. Water contained in soil does not freeze at a single temperature; rather, it freezes gradually over a temperature range. Within this range, water and ice coexist in thermodynamic equilibrium, with the relative proportions of each dependent on temperature, solute content of the water, and retention properties of the medium. As the temperature decreases and more ice is formed, the water potential of the remaining liquid decreases as well. The decline in water potential accompanying freezing creates a gradient favoring water flow toward the freezing front. The extent of freezing-induced water redistribution depends on the balance between heat flow and water flow. If the delivery of latent heat (the product of the water flow rate and the latent heat of fusion) to the freezing front matches the (sensible) heat flow rate away from it, the downward movement of the freezing front will stall as ice accumulates, filling available pore space. Under proper circumstances, ice can continue to form even after all pore space is filled, resulting in the formation of lenses of pure ice and displacement of the soil above, a process known as frost heave.^[1]

RELATIONSHIPS

Once ice nucleation has occurred in a freezing soil, the pressure in the liquid phase is related to the temperature through the Clapeyron equation:

$$\frac{p_w - \pi}{\rho_w} - \frac{p_i}{\rho_i} = L_f \frac{T - T_0}{T_0}$$

where p_i and p_w are the gauge pressures within the ice and water phases, ρ_i and ρ_w are the densities of the respective

phases, and π is the osmotic pressure of the soil solution. L_f is the latent heat of fusion (334 kJ/kg), T is the temperature, and T_0 is the temperature at which bulk water freezes, both in K.

Because water also exists as a vapor, the relationship between phase pressures and temperature can also be expressed psychrometrically:

$$\frac{p_w - \pi}{\rho_w} - \frac{p_i}{\rho_i} = \frac{RT}{M} \ln \frac{e_i(T)}{e_w(T)}$$

The saturation vapor pressures over ice and water, e_i and e_w , are tabled functions of temperature, while R is the gas constant (8.314 J/mol/K) and M is the partial molar mass of water (0.018 kg/mol). Evaluation of either equation shows that the quantity on the left side has a temperature dependence of approximately 1.2 kJ/kg/K. The osmotic pressure depends on the solute concentration of the soil water, but in any event, its temperature dependence is quite small, on the order of π/T . In many cases, and particularly in unsaturated soil, the gauge pressure within the ice phase, p_i , should be negligible. Hence the change in p_w (more commonly known as matric potential) with respect to T in a freezing soil will be about 1.2 MPa/K. The relationship between the temperature of a frozen soil and its liquid water content is graphically expressed in a freezing characteristic curve, analogous to the moisture characteristic curve that describes water retention in unfrozen soil.^[2,3]

APPLICATIONS

Can this information be used to estimate the liquid water content of a frozen soil? Yes, but with some important restrictions and caveats.^[2] It has been shown that the moisture characteristic and the freezing characteristic are

superimposable for porous media that are completely colloidal, i.e., clay suspensions, where surface tension effects are negligible. For such materials, the liquid water content corresponding to a specific pore water pressure (matric potential) should be the same whether the cause of the gauge pressure is drying or freezing (ignoring the issue of hysteresis). For media that are devoid of colloids, i.e., pure sands and silts, the rules for similarity are also clear, but different. Here, the ratio of the surface tensions of an air–water interface (σ_{aw}) and an ice–water interface (σ_{iw}) must be taken into account. For such materials, it has been experimentally shown for a given soil that for similar water contents during drying and freezing the pore water pressure will be more negative in the drying soil by a factor of 2.2, the ratio of σ_{aw} to σ_{iw} . In other words, at a specific pore water pressure, there will be less liquid water in the frozen soil than in the drying soil. Unfortunately, most soils contain both colloidal and noncolloidal particles, so *a priori* scaling of a freezing characteristic curve from known moisture characteristic data is not possible. Developments in instrumentation and methodology offer means to directly measure soil freezing characteristic curves.^[3,4]

Internal water movement within a frozen soil is a difficult process to study. Because thermodynamic similarity exists between freezing and drying, i.e., both are functions of pore size, it is expected that for similar liquid water contents, the hydraulic conductivity of a frozen soil and an unfrozen soil will also be similar, but experimental verification data are scant. Coarse-textured sandy soils, when unfrozen, generally have high saturated hydraulic conductivities but because their pores drain at gauge pressures close to zero, their conductivities dramatically decrease with desaturation. For the reasons mentioned above, the decline in liquid water content (as a function of p_w) upon freezing is even steeper, so such soils rapidly lose their ability to conduct water as they freeze. Finer textured soils generally have lower saturated hydraulic conductivities but because the decrease in water content upon freezing is more gradual, their conductivities decrease more slowly, so that they can often sustain more water movement in the frozen state than sandy soils. For this reason, they are more prone to frost heave. Frost heave can cause tremendous structural damage to buildings and roadways and can also harm plants and trees.

Despite the substantial decrease in water potential accompanying freezing, often there is minimal movement of water as the freezing front penetrates. Unless there is a ready supply of water close to the plane of freezing, the soil beneath will soon become desiccated, causing a sharp decrease in hydraulic conductivity, to the point that the delivery of latent heat cannot match the rate of sensible heat loss, so the freezing front moves downward. Thus initially dry soils may freeze with little or no redistribution of moisture. Consistent with the

Clapeyron equation, the largest water-filled pores freeze first, at temperatures closest to 0°C, and as the temperature decreases, the water in progressively smaller pores freezes. Even in relatively moist soils, the hydraulic conductivity is often insufficient to support anything more than local redistribution of moisture. This is manifested in ice crystal formation in large pores and cracks, without significant change in profile water distribution at a scale detectable by traditional methods of soil moisture measurement.

The infiltration of water into frozen soil has received considerable attention, due to the catastrophic flooding that can occur sometimes following snowmelt or rainfall on frozen soil. It is commonly accepted that freezing dramatically lowers the infiltration capacity of a soil. This is generally true, for reasons alluded to earlier. In wet soils, particularly those with water tables near the surface, water movement during freezing fills large pores and in extreme cases creates lenses of pure ice. Also, just as the largest water-filled pores are the first to freeze, they are also the last to melt, at temperatures closest to 0°C. These are the pores that are most important in infiltration; consequently, infiltration rates are much lower if they are ice filled. Even in well-drained unsaturated soils, local redistribution during freezing is often sufficient to fill large pores at the soil surface with ice, retarding subsequent infiltration. However, in drier soils, particularly if they freeze rapidly, the large pores can remain air filled, and infiltration rates may approach those measured under unfrozen conditions.^[5] Although initial infiltration rates into frozen soil are usually quite low, often resulting in ponding, there have been observations of abrupt increases in infiltration prior to the complete disappearance of frozen soil below.^[6,7] A number of possible explanations have been suggested but none have been experimentally demonstrated.

CONCLUSION

Heat and water relations in frozen soil are difficult to study. Researchers are hampered by the inability to separately measure water and ice contents at similar spatial scales and by the complications arising from simultaneous, coupled transport of heat and water in a deformable medium. Formal theory describing phase equilibrium in ideal, uniform porous medium is well established, but application of that theory to transient phenomena in structured, yet plastic, soils with complex mineralogy is elusive.

REFERENCES

1. Miller, R.D. Freezing phenomena in soils. In *Applications of Soil Physics*; Hillel, D., Ed.; Academic Press: New York, 1980; 254–299.

2. Koopmans, R.W.R.; Miller, R.D. Soil freezing and soil water characteristic curves. *Soil Sci. Soc. Am. Proc.* **1966**, *30*, 680–685.
3. Spaans, E.J.A.; Baker, J.M. The soil freezing characteristic: Its measurement and similarity to the soil moisture characteristic. *Soil Sci. Soc. Am. J.* **1996**, *60*, 13–19.
4. Spaans, E.J.A.; Baker, J.M. Examining the use of time domain reflectometry for liquid water content measurement in frozen soil. *Water Resour. Res.* **1995**, *31*, 2917–2925.
5. Granger, R.J.; Gray, D.M.; Dyck, G.E. Snowmelt infiltration to frozen prairie soils. *Can. J. Earth Sci.* **1984**, *21*, 669–677.
6. Baker, J.M.; Spaans, E.J.A. Mechanisms of meltwater movement above and within frozen soil. In *Proceedings of the International Symposium on Physics, Chemistry, and Ecology of Seasonally Frozen Soils*, Fairbanks, Alaska, Jun 10–12, 1997; Iskandar, I.K., Wright, E.A., Radke, J.K., Sharratt, B.S., Groenevelt, P.H., Hinzman, L.D., Eds.; Spec. Rept. 97-10; U.S. Army Cold Regions Research and Engineering Laboratory: Hanover, 1997; 31–36.
7. Stahli, M.; Lundin, L.C. Soil moisture redistribution and infiltration in frozen sandy soils. *Water Resour. Res.* **1999**, *35*, 95–103.

Fungi

Serita Frey

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Soil fungi are a highly diverse group of microscopic, eukaryotic organisms. They are indeterminate organisms, growing and extending indefinitely as long as sufficient resources are available to sustain their growth. The mycelial growth form is well suited to the heterogeneous soil environment where nutrient resources are spatially separated over long distances at the microbial scale. Fungi are heterotrophs, obtaining carbon, nutrients, and energy through the extracellular degradation and absorption of organic matter from their external environment. Oxygen is generally required for growth. Fungi are an integral part of the soil biotic community.

INTRODUCTION

Soil fungi are a highly diverse group of microscopic, eukaryotic organisms belonging to the Kingdom Fungi (Mycota). While some are single-celled fungi (e.g., yeast), most are multicellular with a mycelial morphology comprising a network of tube-like strands (hyphae). They are indeterminate organisms, growing and extending indefinitely as long as sufficient resources are available to sustain their growth.^[1] The mycelial growth form is well suited to the heterogeneous soil environment where nutrient resources are spatially separated over long distances at the microbial scale. Fungi are heterotrophs, obtaining carbon, nutrients, and energy through the extracellular degradation and absorption of organic matter from their external environment. Oxygen is generally required for growth. Fungi are an integral part of the soil biotic community, significantly contributing to the decomposition of organic matter, the release and turnover of nutrients, the formation and maintenance of soil structure, the extension of plant root systems via the formation of mycorrhizal networks, and the promotion and suppression of plant disease.^[2]

FUNGAL CLASSIFICATION

Fungi obtained kingdom status in 1969 when Whittaker^[3] proposed a five-kingdom classification system that recognized the following five distinct organism groups: Plantae, Animalia, Fungi, Monera (bacteria and cyanobacteria), and Protocista (algae, slime molds, and protozoa). This classification scheme has been much debated, largely due to advances in molecular biology which allow the direct analysis of genetic material (i.e., RNA and DNA). These molecular analyses have greatly enhanced biologists' ability to discern evolutionary relationships and have provided extraordinary insight into the wealth of biological diversity

present on earth. Several new classification schemes containing a greater number of kingdoms have been proposed to adequately represent this diversity.^[4,5] Fungi are assigned to the following four subdivisions within Kingdom Fungi: Basidiomycotina, Ascomycotina, Zygomycotina, and Mastigomycotina.^[6]

Classification within these groups is based on the characteristics of both vegetative and reproductive structures. The oomycetes, an important group of organisms that include several highly destructive plant pathogens (e.g., *Pythium* and *Phytophthora* species), were traditionally classified as fungi within the Mastigomycotina, but are now generally considered to represent a separate evolutionary line from the "true fungi" and are placed with the brown algae in the Kingdom Chromista.^[7]

GROWTH AND REPRODUCTION

Fungi—often incorrectly labeled as plants—superficially resemble plants in that they possess cell walls, are generally non-motile, and reproduce by microscopic, seed-like structures (i.e., spores). However, fungi differ from plants in several key features, including being heterotrophic and having cell walls composed mainly of chitin rather than cellulose. Heterotrophy results in fungi obtaining nutrients and energy by producing and releasing extracellular enzymes which break complex organic compounds down into simpler constituents (e.g., sugars and amino acids) that are then absorbed back through the fungal cell wall and plasmalemma. Molecular evidence suggests that fungi are, in fact, more closely related to animals than to plants.^[7,8]

The vegetative structure of most fungi consists of a filamentous network of hyphae that extends outward by growing at the hyphal tips. Interconnected hyphae are collectively referred to as the fungal mycelium or thallus. Hyphae may be branched or unbranched and may or may

not contain septa, the rigid but porous cross-walls that provide structural support and regulate the flow of cytoplasm. Fungi reproduce either sexually or asexually by producing microscopic propagules called spores.^[2] Spores may be produced singly or in visible fruiting bodies (e.g., mushrooms). In addition to serving as dispersal agents, spores also provide an important survival mechanism, often resistant to low temperatures, low nutrient availability, desiccation, ultraviolet (UV) irradiation, and other adverse environmental conditions.^[2,6] Dormant spores may remain viable for long periods of time, germinating when conditions again become favorable for vegetative growth.^[2]

ABUNDANCE AND DIVERSITY

Fungi, with bacteria, comprise the majority of the total living biomass of soil biota.^[9,10] In some soils, fungal biomass equals or exceeds that of plant root biomass.^[11] The hyphae of filamentous fungi are usually 2 to 10 μm in diameter, but can reach extensive lengths and cover large areas. A gram of soil typically contains several hundred meters of fungal hyphae and may easily contain several hundred different fungal species. Despite their microscopic size, a single fungal mycelium can cover an area of several hectares.^[12,13] Approximately, 70,000 fungal species have been described worldwide, but an estimated 1.5 million species exist, indicating that 95% of all extant species have yet to be identified.^[14] Limitations to the adequate characterization of fungal diversity include the methodological constraints historically associated with the identification of microscopic organisms, the lack of trained fungal taxonomists, and the economic cost associated with the large research efforts required for this type of work.^[15]

FUNGI AND ECOSYSTEM FUNCTION

Soil fungi can be grouped into three general functional groups (Table 1). While plant pathogenic fungi, such as *Fusarium* and *Rhizoctonia*, cause significant losses of agricultural crops each year, most soil fungi are beneficial and perform a number of critically important ecological

functions. A primary role of fungi in any soil is the decomposition of plant residues. Saprophytic fungi produce a suite of extracellular enzymes capable of depolymerizing plant cell constituents such as cellulose, hemicellulose, and lignin. During the decomposition process, fungi immobilize (i.e., retain in their biomass in organic form) and mineralize (i.e., release to the environment in inorganic form) nutrients simultaneously, with the balance between these two processes determining the plant availability of nutrients such as nitrogen, phosphorus, potassium, and sulfur. In addition to nutrient immobilization, fungi are known to potentially accumulate toxic substances within the mycelium, including radionuclides and heavy metals.^[9] The branching of fungal hyphae around soil particles, combined with the production of extracellular polysaccharides that serve as binding agents, promotes the formation of stable soil aggregates.^[15] This fungal-mediated process modifies air and water relations by altering soil permeability and may be an important mechanism for the physical protection of soil organic matter.^[16] Many saprophytic fungi promote the suppression of plant disease, either by producing antibiotics inhibitory to disease-causing organisms or by out-competing pathogens for available resources,^[11] e.g., the fungus *Chaetomium globosum* produces an antibiotic suppressive to *Fusarium*, the agent of seedling blight on maize.^[11]

Mutualistic associations involving fungi include lichens, endophytes, and mycorrhizas. In all cases, a fungus establishes a mutually beneficial relationship with an autotrophic organism. Lichens are fungus–alga or fungus–cyanobacterium associations in which the alga or cyanobacterium partner captures energy through photosynthesis, and the fungal partner provides structural support, supplies mineral nutrients, and helps regulate water relations.^[11] Endophytic fungi grow within living plants, obtaining nutrients from the plant without causing noticeable damage and, in at least some cases, providing protection from insects and plant pathogens;^[2] e.g., some grasses (e.g., ryegrass) can be infected with a fungus that produces toxins thought to inhibit insect attack.^[2]

Mycorrhizal fungi, which form an intimate association with plant roots, enhance the uptake and transfer of mineral nutrients to the plant host in exchange for carbon. “Indeed, mycorrhizas, not roots, are the chief organs of nutrient uptake by land plants.”^[17] Phosphorus is often the nutrient of primary importance, but nitrogen, zinc, and sulfur have also been shown to be absorbed and transported by mycorrhizal fungi.^[11] Some mycorrhizal species participate in water as well as nutrient absorption and may also provide protection against root pathogens.^[11] Mycorrhizas are currently grouped into seven types, namely arbuscular mycorrhizas, ectomycorrhizas, ectendomycorrhizas, arbutoid mycorrhizas, monotropoid mycorrhizas, ericoid mycorrhizas, and orchid mycorrhizas.^[17] Arbuscular mycorrhizas are the most common and are formed by fungi in the order Glomales (Zygomycetes) in association with roots of a

Table 1 Functional groups of soil fungi.

Group	Function
Saprotrophs	Decomposition of organic matter
	Nutrient immobilization and release
	Accumulation of toxic materials
	Aggregate formation and stabilization
	Suppression of pathogens
Mutualists	Transport of nutrients and water to plant roots
	Protection from root pathogens and heavy metals
Pathogens	Cause plant and animal disease

wide range of plants.^[17] Fossil evidence suggests that mycorrhizal-like structures, resembling modern arbuscular mycorrhizas formed by fungi in the genus *Glomus*, were present 460 million years ago and may have played an important role in the successful establishment of early terrestrial plants.^[18,19] Ecological studies indicate that mycorrhizal species diversity in soil can influence the diversity and productivity of plant communities.^[20]

Fungi are sensitive to disturbance, and human-induced modifications of the soil environment often have negative impacts on the soil fungal community. Tillage of agricultural land disrupts the establishment and development of hyphal networks, thereby significantly reducing fungal hyphal lengths and biomass within the plow layer.^[21,22] Increased nitrogen inputs, through fertilization^[23,24] and atmospheric deposition,^[25] have been implicated in the decline of both fungal biomass and fungal species diversity. Exposure to UV-B radiation, which has been increasing due to thinning of the protective stratospheric ozone layer, has been shown to influence fungal growth and the relative competitive abilities of different fungal species.^[26] Reductions in fungal biomass, activity, and diversity could have significant implications for the functioning of terrestrial ecosystems.

REFERENCES

- Andrews, J.H. Fungal life-history strategies. In *The Fungal Community: Its Organization and Role in the Ecosystem*; Carroll, G.C., Wicklow, D.T., Eds.; Marcel Dekker: New York, 1992; 119–145.
- Deacon, J.W. *Modern Mycology*, 3rd Ed.; Blackwell Science: Oxford, 1997.
- Whittaker, R.H. New concepts of kingdoms of organisms. *Science* **1969**, *163*, 150–160.
- Cavalier-Smith, T. Eukaryote kingdoms: Seven or nine? *BioSystems* **1981**, *14*, 461–481.
- Woese, C.R.; Kandler, O.; Wheelis, M.L. Towards a natural system of organisms: Proposal for the domains Archaea, Bacteria, and Eucarya. *Proc. Natl. Acad. Sci. USA* **1990**, *87*, 4576–4579.
- Cooke, R.C.; Whipps, J.M. *Ecophysiology of Fungi*; Blackwell Scientific Publications: London, 1993; 337 pp.
- Baldauf, S.L.; Palmer, J.D. Animals and fungi are each other's closest relatives: Congruent evidence from multiple proteins. *Proc. Natl. Acad. Sci. USA* **1993**, *90*, 11558–11562.
- Baldauf, S.L. A search for the origins of animals and fungi: Comparing and combining molecular data. *Am. Nat.* **1999**, *154*, S178–S188.
- Killham, K. *Soil Ecology*; Cambridge University Press: Cambridge, 1994.
- Beare, M.H. Fungal and bacterial pathways of organic matter decomposition and nitrogen mineralization in arable soils. In *Soil Ecology in Sustainable Agricultural Systems*; Brussaard, L., Ferrera-Cerrato, R., Eds.; Lewis Publishers: Boca Raton, 1997; 37–70.
- Paul, E.A.; Clark, F.E. *Soil Microbiology and Biochemistry*, 2nd Ed.; Academic Press: San Diego, 1996.
- Smith, M.L.; Bruhn, J.N.; Anderson, J.B. The fungus *Armillaria bulbosa* is among the largest and oldest living organisms. *Nature* **1992**, *356* (6368), 428–431.
- Bonello, P.; Bruns, T.D.; Gardes, M. Genetic structure of a natural population of the ectomycorrhizal fungus *Suillus pungens*. *New Phytol.* **1998**, *138*, 533–542.
- Hawksworth, D.L. The fungal dimension of biodiversity: Magnitude, significance, and conservation. *Mycol. Res.* **1991**, *95* (6), 641–655.
- Cannon, P.F. Strategies for rapid assessment of fungal diversity. *Biodivers. Conserv.* **1997**, *6*, 669–680.
- Beare, M.H.; Hu, S.; Coleman, D.C.; Hendrix, P.F. Influences of mycelial fungi on soil aggregation and organic matter storage in conventional and no-tillage soils. *Appl. Soil Ecol.* **1997**, *5*, 211–219.
- Smith, S.E.; Read, D.J. *Mycorrhizal Symbiosis*, 2nd Ed.; Academic Press: San Diego, 1997.
- Blackwell, M. Terrestrial life–fungal from the start? *Science* **2000**, *289*, 1884–1885.
- Redecker, D.; Kodner, R.; Graham, L.E. Glomalean fungi from the Ordovician. *Science* **2000**, *289*, 1920–1921.
- van der Heijden, M.G.A.; Klironomos, J.N.; Ursic, M.; Moutoglou, P.; Streitwolf-Engel, R.; Boller, T.; Wiemken, A.; Sanders, I.R. Mycorrhizal fungal diversity determines plant biodiversity, ecosystem variability and productivity. *Nature* **1998**, *396*, 69–72.
- Beare, M.H.; Parmelee, R.W.; Hendrix, P.F.; Cheng, W.; Coleman, D.C.; Crossley, D.A. Microbial and faunal interactions and effects on litter nitrogen and decomposition in agroecosystems. *Ecol. Monogr.* **1992**, *62* (4), 569–591.
- Frey, S.D.; Elliott, E.T.; Paustian, K. Bacterial and fungal abundance and biomass in conventional and no-tillage agroecosystems along two climatic gradients. *Soil Biol. Biochem.* **1999**, *31*, 573–585.
- Bardgett, R.D.; Hobbs, P.J.; Frostegard, A. Changes in soil fungal: bacterial biomass ratios following reductions in the intensity of management of an upland grassland. *Biol. Fert. Soils* **1996**, *22*, 261–264.
- Bardgett, R.D.; Lovell, R.D.; Hobbs, P.J.; Jarvis, S.C. Seasonal changes in soil microbial communities along a fertility gradient of temperate grasslands. *Soil Biol. Biochem.* **1999**, *31*, 1021–1030.
- Arnolds, E.J.M. The changing macromycete flora in the Netherlands. *T. Brit. Mycol. Soc.* **1988**, *90*, 391–406.
- Duguay, K.J.; Klironomos, J.N. Direct and indirect effects of enhanced UV-B radiation on the decomposing and competitive abilities of saprobic fungi. *Appl. Soil Ecol.* **2000**, *14*, 157–164.

Future of Soil Science

Eric C. Brevik

Department of Natural Sciences, Dickinson State University, Dickinson, North Dakota, U.S.A.

Samuel J. Indorante

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Du Quoin, Illinois, U.S.A.

Dylan E. Beaudette

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Sonora, California, U.S.A.

Richard W. Arnold

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Washington, District of Columbia, U.S.A.

Abstract

Human population, resource consumption, and waste generation have increased rapidly throughout the Anthropocene. Feeding this growing population, supplying needed resources, and dealing with waste products require expanding our knowledge of and the sustainable use of soil resources. At the same time, anthropogenic activity has created major changes in ecosystems, including the soil system. These changes may alter the functions of soils that mankind depends on to sustain both life and modern standards of living. To ensure productivity, conservation, and general respect of soils, humans need to overcome the stigma of “dirt” or “filth” often associated with soil and instead treat soil as a vital natural resource. Doing so will require expanded use of modern technologies/advances including proximal and remote sensing, geographic information systems, and spatial statistics. However, it is equally important to recognize that the models and predictions generated by these new techniques are limited by the quality and quantity of field-based observations. Therefore, more traditional field investigations, including soil landform studies, continue to be important to soil science heading into the future.

INTRODUCTION

Studies indicate that during the Anthropocene, humanity has experienced exponential growth of consumption, generation of waste, and population. The human economy depends on the planet’s natural capital that provides all ecological services and natural resources, and since about 1980, human demands have exceeded the capacity of the earth to sustain such use.

Cultural attitudes determine the role of soils in the world as our fragmented global community struggles to resolve the global issues of food security, environmental protection, and overall sustainability. A variety of world views influence the search for a sustainable, socially acceptable balance among soil functions that provide for viable economic growth and development, safe healthy environments, and intergenerational equity.

Linking entire social systems in a web of production, distribution, and consumption, agriculture often fore-shadows the degree of economic well-being. Because

agriculture operates simultaneously in the realms of ecology and economics, each of which marks time by different clocks, decisions affecting food security and environmental protection have become increasingly complex and variable over time and space.

THE PEDOSPHERE

Soils are a critical interface between society and natural resources. Thus, the basic principles of the organization and functioning of the earth’s soil cover, the pedosphere, can provide a scientific basis for addressing global environmental, social, economic, agronomic, and human health issues throughout the 21st century.^[1]

Natural soils result from the interaction of processes taking place over time on the earth’s surface. Most involve gases and liquids that transform the solid phase of the surficial materials into features recognized as soils. These processes are influenced by soil-forming factors—namely,

climate, biota, topography, parent material, and time—leading to great heterogeneity in the world's soil cover. Geomorphic processes that alter the landscape by erosion, transport, and deposition of rocks and sediments and the interaction of biological systems with soils are all subject to major modifications associated with global and regional climate changes. The result is an intricate patchwork of soils, ranging in size from a few square meters to thousands of square kilometers.

For millennia, naturally evolving soils were dominant in the world, and determining the properties and distributions of major kinds of soils and their local geographic associates enabled societies to effectively tap these resources to satisfy their needs. Soils provided habitats for plants, animals, and microorganisms. Soils possessed the capacity for fertility and potential productivity because of water, physical support, and biological interactions that provided nutrients.^[2]

Mapping the pedosphere and deciphering the sequence of events and processes causing such complexity have revealed a multidimensional hierarchy that is meaningful for assessing many kinds of soils and predicting soil-related behavior.

ALTERED ECOSYSTEMS

For more than a century, scientific studies of soils as natural independent entities on the earth's surface have contributed to a better understanding of the interconnectivity of the earth's systems.^[3] As societies introduced more and more invasive procedures, they drastically altered ecosystem processes and biogeochemical cycles. Although some ecosystems have been enhanced, more have been degraded and are being used unsustainably.^[4,5] It has become relevant to monitor, predict, and mediate the behavior and responses of both natural and artificial (anthropogenic) soil environments and landscapes. It is believed that many changes being made in ecosystems are increasing the likelihood of detrimental, nonlinear changes.^[6]

With increases in the size of the human population and its increasing rate of consumption, the available natural resources are stressed, some beyond their limits of resilience.^[1] One estimate is that human resource use in 2000 was about 20% above the global carrying capacity.^[7] When soils are so stressed, they are unable to return to their former productive states without massive external inputs. Thus, sustainable reconciliation of societal desires and natural resource capabilities is commonly jeopardized.

FUNCTIONS OF SOILS

Functions of soils commonly include: 1) promoting biomass transformations; 2) serving as the earth's geomembrane to filter and buffer; 3) providing biological habitats; 4) providing usable materials; and 5) serving as sources and repositories of our cultures.^[8]

Soils are primarily used by most people for the production of food, feed, fuel, and fiber. The capacity to store and release water and the ability to renew, store, and release plant nutrients have dominated agronomic and forestry research and practical experimentation for many years, going back as far as the 16th century. Biomass transformations are highly dependent on the microbiological populations that inhabit soils and facilitate the formation and use of beneficial compounds.

The pedosphere is a sensitive geomembrane which mediates the transfer of air, water, and energy into, out of, and among the biosphere, atmosphere, hydrosphere, and lithosphere (Fig. 1). Temperatures are moderated with depth, and water and associated compounds are filtered, retained, stored, and transferred in ways that contribute to clean, healthy environments.

Soils are the habitat for millions of organisms, ranging from cellular bacteria to burrowing animals. Communities of microorganisms decompose organic materials, facilitate the release of mineral elements, and produce the chemical and biological compounds essential for life on earth.^[2]

Many soils are directly used as raw materials for constructing dams and foundations; others are source materials for landscaping industrial and urban sites; and some are ingredients for bricks, aluminum, ceramic products, and medications. Most transportation networks and urban communities rest on soils. In addition, soils are excavated to create disposal sites for society's numerous wastes. Soils that are disturbed or removed from their natural environments are commonly called "dirt." As such, displaced soil materials are generally considered a nuisance in our daily activities and need to be washed out, removed, and personal contact with them minimized. Historically, stigma was often attached to those associated with the "filth and dirt" of using and managing soils.^[9]

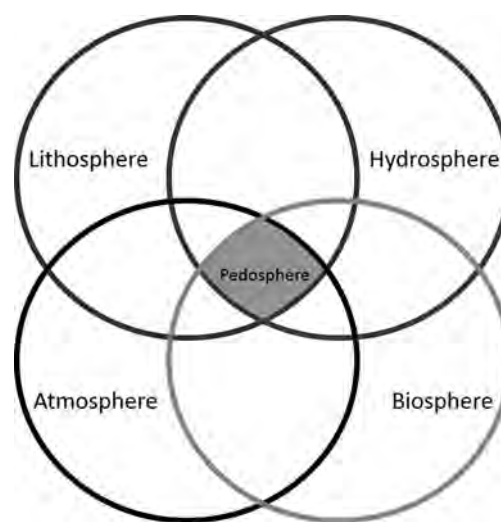


Fig. 1 The pedosphere results from the interaction throughout space and time of the atmosphere, biosphere, and hydrosphere with the lithosphere. Once developed, the pedosphere plays a significant role in supporting and maintaining life on the planet.

In the stories of most indigenous cultures, the earth is sanctified and revered as a vital element of nature, thereby reinforcing humankind's link with it. This sacredness remains in the use of soils as cemeteries and as places where the spirits of ancestors reside. Archeological investigations often involve deciphering the memories of times and events recorded in soil properties.^[10]

CONTINUING NEEDS

It is obvious that soils continue to be used to maintain and improve productivity for food, feed, fuel, and fiber for a long time because the supply of elements and minerals necessary for life is derived, directly or indirectly, from soils. Healthy ecosystems and environments depend on soil-hosted microorganisms that facilitate filtering and purifying. There is much to be learned about how soils behave in maintaining healthy anthropogenic landscapes. Unprecedented demands to safely handle wastes and provide major increases in food and feed are enormous challenges.^[7,11]

Perhaps less appreciated is the need of the human psyche for renewal through contact with nature. It appears that psychological well-being is, in part, related to our communion with the beauty, tranquility, and mysterious forces of nature.^[12] Although we are connected to the land and soils in ways that most of us do not readily comprehend, we do recognize the sense of belonging and the feeling of renewal associated with our contacts with the natural world. Is it a "dust thou art, and to dust thou shall return" syndrome in which our life cycle is subsumed in other natural life cycles? Whatever the explanation, the human-to-nature relationship is important, and soil is vital to our survival and growth.

CHANGING DEMOGRAPHICS

It is estimated that during the 21st century, more people will be living in cities than currently live on earth.^[13] As cities evolve into megacities through the use of adjacent lands, the unique culture of urban dwellers, including their estrangement from rural landscapes and ecosystems, also develops.^[1]

Ecological functions of soils are common in extensive rural areas of the earth, where the productive capacity of soils is easily recognized. In urban landscapes, which are very different in appearance, structure, and composition, the role of soils is commonly not observed, or even imagined, except in parks and residential lawns and gardens. Sewers and trash trucks remove numerous waste products, smoke stacks and vehicles exude particulates into the atmosphere, and streams and rivers wash away other debris. In rural landscapes, many of these functions are provided by the pedosphere—the covering of soils that seems to be ubiquitous and so common that it is taken for granted.

How much relatively undisturbed land would be required to provide the energy and products consumed in

an urban environment, and to adequately handle its wastes? Such a measure could be thought of as a city's environmental footprint.^[11] We do not measure and monitor the enormous fluxes that occur in urban environments, yet they are becoming major stressors on our global habitat.

INFLUENCING THE ROLE

Is there a possibility of simply stretching ecological theory to encompass urban ecosystems? It can be argued that our understanding of dynamics and processes of populations and soils can be extended to homeowners' associations and pavement.^[11] If people act as other organisms do, guided by individual self-interest, there is no basis for a moral or aesthetic call to environmental stewardship. Is there, perhaps, a spiritual or moral dimension that defies explanations offered by evolution or natural selection?

The challenge of understanding urban ecosystems requires specialists from many different disciplines but it also requires that at least some individuals think in interdisciplinary and transdisciplinary ways—a task that may be difficult to accomplish.

The single most important force of landscape change in urban areas is land conversion driven by institutional decisions, population growth, and economic forces.^[11] A city's footprint, which indicates the dependence of an urban ecosystem on other ecosystems, may be tens or hundreds of times larger than the city itself.

Soil ecosystems are being altered, and even created, to meet the expectations of urban communities. Consequently, the kinds and patterns of adjacent soil landscapes are important to the monitoring and assessment of environmental health and sustainability.^[4,7]

THE FUTURE

There are many challenges that need to utilize soil knowledge if they are to be adequately addressed. A global consensus must develop that will clearly define a minimum set of common norms and international standards for sustainable uses of various kinds of soils. Knowledge about the limitations of specific kinds of soils for particular uses is essential for informed decisions and agricultural policies (Table 1). For every conceivable use of soil, there is a hypothetical ideal soil with the right set of properties and supporting processes to achieve a satisfactory level of success. Comparing local soils with a specified "ideal" soil can lead to recommended measures that minimize limitations and contribute toward attaining the expected behavior of such an ideal soil. When local economics of managing individual soils are considered, it is possible to develop rankings of suitability and economic feasibility.

Since the 1980s, the amount of field work done in soil science has significantly declined due to shifts in priorities leading to cuts in funding and diminished field training at

Table 1 Soil attributes that can become constraints to achieving desired balances of soil functions if excessive demands are placed on soil resources

Soil attribute	Constraint
Resilience	Recovery from disturbance
Productivity	Capability for plant growth and yield
Responsiveness	Capacity for external enhancement
Sustainability	Dynamic equilibrium of interactions
Resistance	Stability to maintain current condition
Flexibility	Multiplicity of uses related to properties
Pedoclimate	Location and extent of suitable climate
Residence time	Capacity to store and release compounds
Geography	Availability due to location or intricacy of pattern

colleges and universities. New techniques utilizing proximal and remotely sensed information gathered with geographic positioning system technologies and analyzed in geographic information systems, often incorporating geostatistical models, have grown in popularity.^[14] This popularity is due in part to the high data density that proximal and remote sensing can provide and quantitative models linking these measurements to specific soil properties. The popularity is also partly because proximal and remote sensing techniques are less expensive than traditional field work. Providing soil information that can address the challenges of the future will require the high data density that proximal and remote sensing can provide evaluated with geostatistical and other advanced mathematical techniques. However, it is very important that field work continues to be supported as well.^[14] The models and predictions generated from proximal and remote sensing techniques are limited by the quality and quantity of field-based observations and measurements of soil and landscape parameters. Therefore, more traditional field investigations, including soil land-form studies, need to be supported so that information modeled from proximal and remotely sensed data can be verified as being meaningful or improved when field checks show the model to be flawed. The soil scientist of the future will need to be able to combine competence in the field with competence in quantitative methods.

CONCLUSIONS

Humanity surpassed the earth's capacity to support our ever-increasing demands on natural resources several decades ago. There is great disparity among nations in their ecological footprints, thus the challenges of implementing practices to attain sustainability are many. Supplying quantitative soils information at scales suitable to address needs is a challenge facing the soil science community. Soils are vital to supplying food, feed, fiber, and fuel to support the development of future generations; consequently, improved understanding of

the functions and limitations of local soils is relevant to helping meet the desires for resources and the handling of wastes in a sustainable global habitat for all humanity.

REFERENCES

1. Brevik, E.C.; Cerdà, A.; Mataix-Solera, J.; Pereg, L.; Quinton, J.N.; Six, J.; Van Oost, K. The interdisciplinary nature of soil. *Soil* **2015**, *1*, 117–129. DOI:10.5194/soil-1-117-2015.
2. Kovda, V.A. *The Role and Functions of the Soil Cover in the Earth's Biosphere*; Scientific Center of Biological Research, Academy of Science, USSR: Pushchino, 1985; 1–12.
3. Targulian, V.O.; Rode, A.A.; Dmitriev, N.A.; Armand, A.D. Soil as a component of natural ecosystems and the study of its history, modern dynamics and anthropogenic changes. In *Selection, Management and Utilization of Biosphere Reserves*. General Technical Report PNW 82, Proceedings of the United States-Union of Soviet Socialist Republics Symposium on Biosphere Reserves, Moscow, USSR, May, 1976; Franklin, J.F., Krugman, S.L., Eds.; USDA Pacific Northwest Forest and Range Experiment Station: Corvallis, Oregon, 1979; 186–197.
4. Millennium Ecosystem Assessment. Main findings. Available at <http://www.millenniumassessment.org/en/About.html> (accessed April 30, 2015).
5. Yaalon, D.H.; Yaron, B. Framework for man-made soil changes – an outline of metapedogenesis. *Soil Sci.* **1966**, *102* (4), 272–277.
6. Meadows, D.H.; Randers, J.; Meadows, D.L. *Limits to Growth: The 30-Year Update*; Chelsea Green Publ. Co.: White River Junction, 2004.
7. Wackernagel, M.; Schulz, N.B.; Deumling, D.; Linares, A.C.; Jenkins, M.; Kapos, V.; Monfreda, C.; Loh, J.; Myers, N.; Norgaard, R.; Randers, J. Tracking the ecological overshoot of the human economy. *P. Natl. A. Sci. (U.S.)* **2002**, *99* (14), 9266–9271.
8. German Advisory Council on Global Change. *World in Transition: The Threat To Soils*, Vol. 1994; Annual Report; Economica Verlag: Bonn, 1995.
9. Yaalon, D.H.; Arnold, R.W. Attitudes toward soils and their societal relevance; then and now. *Soil Sci.* **2000**, *165* (1), 5–12.
10. Holliday, V.T. *Soils in Archaeological Research*; Oxford University Press: New York, 2004.
11. Collins, J.P.; Kinzig, A.; Grimm, N.B.; Fagan, W.F.; Hope, D.; Wu, J.; Borer, E.T. A new urban ecology. *Am. Sci.* **2000**, *88* (5), 416–425.
12. Heckman, J.R. Human contact with plants and soils for health and well-being. In *Soils and Human Health*; Brevik, E.C., Burgess, L.C., Eds.; CRC Press: Boca Raton, 2012; 227–240.
13. Brown, L.R.; Gardner, G.; Halweil, B. *Beyond Malthus: Sixteen Dimensions of the Population Problems*. Worldwatch Paper 143; Worldwatch Institute: Washington, DC, 1998.
14. Brevik, E.C.; Calzolari, C.; Miller, B.A.; Pereira, P.; Kabala, C.; Baumgarten, A.; Jordán, A. Soil mapping, classification, and modeling: History and future directions. *Geoderma* **2016**, *264*, 256–274.

Gas and Vapor Phase Transport

Dennis E. Rolston

Land, Air, and Water Resources, University of California—Davis, Davis, California, U.S.A.

Abstract

Diffusion is the principal mechanism in the interchange of gases between the soil and the atmosphere. Fick's law of diffusion can describe the diffusion of gases in soils as long as the restrictive conditions for the applicability of the equation apply. The soil-gas diffusion coefficient, D_p , is the fundamental property that must be known to use Fick's law to calculate gas transport in soils. Values of D_p change with soil-air content and tortuosity. Techniques for accurately measuring D_p in soil cores are well established. However, because measured data for D_p are often not available, equations to predict D_p from easily obtainable soil-physical parameters have been developed. It is well established that different equations are needed to estimate D_p for undisturbed and for disturbed soil samples. Equations for undisturbed soil must include parameters that take into account the effects of soil type, such as texture, structure, and pore-size distribution, on D_p . Soil-type effects on D_p in sieved and repacked soils appear to be minor.

INTRODUCTION

Diffusion is the principal mechanism in the interchange of gases between the soil and the atmosphere. The interchange results from concentration gradients established within soil by respiration of microorganisms and plant roots; by production of gases associated with biological reactions such as fermentation and nitrogen transformations; and by soil incorporation of materials such as fumigants, anhydrous ammonia, pesticides, and various volatile organic chemicals in toxic waste sites. The diffusion of water vapor within the soil also occurs due to differences in vapor pressure gradients induced by temperature differences or by evaporative conditions at the soil surface.

TRANSPORT PHYSICS AND EQUATIONS

The diffusion velocities of gas mixtures in porous media are related to each other in a complex manner dependent on the mole fraction of each gas, the molar fluxes of each gas, and the binary diffusion coefficient of each gas pair. If gravity effects are ignored or diffusion occurs only horizontally, the well-known Stefan–Maxwell equations provide the theoretical framework for diffusion of gases in soils. Fick's law for diffusion is a restrictive case of the Stefan–Maxwell equations and is generally applicable for only a few special cases.^[1] One of these cases is for the diffusion of a trace gas in a binary mixture, meaning that the mole fraction of the tracer gas is small. The second special case is for diffusion of two gases in a closed system (total pressure remains constant). In this case, neither gas needs to be in trace amounts. A third case where Fick's law is applicable is for a three-component system where one of the gases

exists in trace amounts and the binary diffusion coefficients of the other two pairs do not differ much from one another (basically the first case). Examples of this case would be for the gas pairs of nitrogen (N_2)–oxygen and N_2 –argon.

Assuming that the special case conditions are met, Fick's law is given by as follows:

$$\frac{M_g}{At} = f_g = -D_p \frac{dC_g}{dx} \quad (1)$$

where M_g is the amount of gas diffusing (g gas), A is the cross-sectional area of the soil (m^2 soil), t is time (sec), f_g is the gas flux density (g gas/ m^2 soil/sec), C_g is concentration in the gaseous phase (g gas/ m^3 soil air), x is distance (m soil), and D_p is the soil-gas diffusion coefficient (m^3 soil air/m soil/sec).

MEASUREMENT OF THE SOIL-GAS DIFFUSION COEFFICIENT

The soil-gas diffusion coefficient, D_p , is the fundamental property that must be known to use Eq. 1 to calculate gas transport in soils. Values of D_p change with soil-air content and tortuosity (crookedness of the diffusion path).

The standard laboratory method for measuring the soil-gas diffusion coefficient is based on establishing gas concentration, C_o , within a chamber (Fig. 1). One end of a soil core of concentration C_s is placed in contact with the gas within the chamber.^[2] The other end of the soil core is maintained at concentration C_s . The gas of interest diffuses either into or out of the chamber depending upon the concentration C_o compared to that outside the core. Obviously, the other gases making up the atmosphere will diffuse in an opposite direction to that of the gas of interest. The time

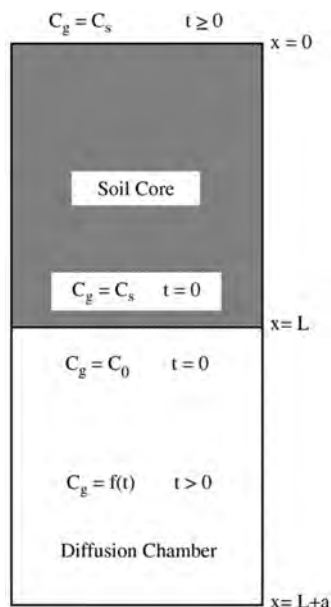


Fig. 1 Schematic diagram of the apparatus design used for measuring the soil-gas diffusion coefficient of a soil core.

rate of change of concentration in the chamber is related to the soil-gas diffusion coefficient and can be described by equations for unsteady diffusion of gas. Several investigators^[2] have used similar procedures.

The unsteady diffusion of a gas, which is non-reactive (physically, biologically, and chemically), is described by the combination of Fick's first law (Eq. 1) and the continuity equation (conservation of mass)

$$\epsilon \frac{\partial C_g}{\partial t} = D_p \frac{\partial^2 C_g}{\partial x^2} \quad (2)$$

where ϵ is the soil-air content ($\text{m}^3 \text{ air}/\text{m}^3 \text{ soil}$).

In developing Eq. 2, it is assumed that the soil is uniform with respect to the diffusion coefficient and that ϵ is constant in space and time. A simple solution of Eq. 2 that allows for the determination of D_p from laboratory measurements is available.^[2,3]

PREDICTIVE EQUATIONS FOR THE SOIL-GAS DIFFUSION COEFFICIENT

Gas transport and fate simulation studies often depend on accurately estimating gas diffusivity (D_p/D_0) as a function of soil-air content (ϵ) from easily obtainable physical parameters of the soil because measured data of $D_p(\epsilon)$ are often not available. Many empirical models for predicting the gas diffusivity have been proposed, such as the well-known Millington–Quirk equation,^[4] with varying degrees of prediction accuracy.^[5] A series of papers^[5–9] have offered new equations that appear to greatly increase the predictive accuracy. Separate equations are needed for undisturbed

soils compared to sieved and repacked soils. To accurately predict diffusivity in undisturbed soils, the effects of texture and structure on the diffusivity must be considered.

Undisturbed Soil

It has been shown that the Campbell^[10] pore-size distribution (water retention) parameter, b , is an effective parameter to describe effects of soil type (soil texture and structure) on $D_p(\epsilon)$ in undisturbed soils^[5–8] to give

$$\frac{D_p}{D_0} = \Phi^2 \left(\frac{\epsilon}{\Phi} \right)^{2+3/b} \quad (3)$$

where D_0 is the gas diffusion coefficient in air (without soil; $\text{m}^2 \text{ air}/\text{sec}$), Φ is soil total porosity ($\text{m}^3 \text{ voids}/\text{m}^3 \text{ soil}$), and b is the Campbell pore-size distribution parameter, corresponding to the slope of a plot of the log of the soil-water pressure potential vs. the log of the volumetric soil-water content.

It has also been shown that there is a significant effect of macroporosity on D_p .^[8] In this respect, macroporosity is defined as the air-filled porosity at a soil-water pressure head of $-100 \text{ cm H}_2\text{O}$ (-10 kPa), corresponding to the volumetric content of soil pores with an equivalent pore diameter larger than $30 \mu\text{m}$. The macroporosity (ϵ_{100}) is found by subtracting the volumetric soil-water content measured at a soil-water pressure head of -100 cm from the soil total porosity. Using this concept results in an additional equation as follows:

$$\frac{D_p}{D_0} = (2\epsilon_{100}^3 + 0.04\epsilon_{100}) \left(\frac{\epsilon}{\epsilon_{100}} \right)^{2+3/b} \quad (4)$$

To summarize, Eq. 4 appears slightly more accurate and is recommended if both ϵ_{100} and b are known, whereas Eq. 3 is recommended if only b is known.

If the Campbell pore-size distribution parameter, b , is not measured, clay fraction is a good indicator of b ^[11,12] and is given by as follows:

$$b = 13.6 \text{ CF} + 3.5 \quad (5)$$

Thus, Eq. 5 can be used to determine b to be subsequently used in either Eq. 3 or 4.

Sieved and Repacked Soil

Soil-type effects, such as texture and structure (manifested through pore-size distribution), on gas diffusivity in sieved and repacked soils appear to be minor and can likely be neglected.^[9,13,14] A simple, physically based model^[9] for $D_p(\epsilon)/D_0$ in sieved and repacked soils is given by as follows:

$$D_p/D_0 = \epsilon^{1.5} \left(\frac{\epsilon}{\Phi} \right) \quad (6)$$

The reduction term (ϵ/Φ) describes the increased tortuosity in a wet soil, compared to a dry soil at the same soil-air content, because of interconnected water films. This equation has been shown to give accurate predictions of the soil-gas diffusion coefficient in sieved and repacked soil samples.^[9]

CONCLUSION

Although techniques for accurately measuring D_p in soil cores are well established, measured data for D_p are often not available and predictive equations are required. It is well established that equations needed to estimate D_p for undisturbed and for disturbed soil samples are indeed different. Equations for undisturbed soil must include soil-physical parameters that take into account the effects of soil type, such as texture, structure, and pore-size distribution, on D_p . Soil-type effects on D_p in sieved and repacked soils appear to be minor.

REFERENCES

1. Amali, S.; Rolston, D.E. Theoretical investigation of multi-component volatile organic vapor diffusion: Steady-state fluxes. *J. Environ. Qual.* **1993**, *22*, 825–831.
2. Rolston, D.E.; Moldrup, P. Gas diffusivity. In *Methods of Soil Analysis, Part 4. Physical Methods*; Dane, J.H., Topp, G.C., Eds.; Agronomy Monograph No. 9; Soil Science Society of America: Madison, 2002; 1297–1306.
3. Currie, J.A. Gaseous diffusion in porous media. Part 1. A non-steady state method. *Brit. J. Appl. Phys.* **1960**, *11*, 314–317.
4. Millington, R.J.; Quirk, J.M. Permeability of porous solids. *Trans. Faraday Soc.* **1961**, *57*, 1200–1207.
5. Moldrup, P.; Kruse, C.W.; Rolston, D.E.; Yamaguchi, T. Modeling diffusion and reaction in soils. III. Predicting gas diffusivity from the Campbell soil-water retention model. *Soil Sci.* **1996**, *161*, 366–375.
6. Moldrup, P.; Olesen, T.; Rolston, D.E.; Yamaguchi, T. Modeling diffusion and reaction in soils. VII. Predicting gas and ion diffusivity in undisturbed and sieved soils. *Soil Sci.* **1997**, *162*, 632–640.
7. Moldrup, P.; Olesen, T.; Yamaguchi, T.; Schjønning, P.; Rolston, D.E. Modeling diffusion and reaction in soils. IX. The Buckingham–Burdine–Campbell equation for gas diffusivity in undisturbed soil. *Soil Sci.* **1999**, *164*, 542–551.
8. Moldrup, P.; Olesen, T.; Schjønning, P.; Yamaguchi, T.; Rolston, D.E. Predicting the gas diffusion coefficient in undisturbed soil from soil water characteristics. *Soil Sci. Soc. Am. J.* **2000**, *64*, 94–100.
9. Moldrup, P.; Olesen, T.; Gamst, J.; Schjønning, P.; Yamaguchi, T.; Rolston, D.E. Predicting the gas diffusion coefficient in repacked soil: Water-induced linear reduction model. *Soil Sci. Soc. Am. J.* **2000**, *64*, 1588–1594.
10. Campbell, G.S. A simple method for determining unsaturated conductivity from moisture retention data. *Soil Sci.* **1974**, *117*, 311–314.
11. Clapp, R.B.; Hornberger, G.M. Empirical equations for some soil hydraulic properties. *Water Resour. Res.* **1978**, *14*, 601–604.
12. Olesen, T.; Moldrup, P.; Henriksen, K.; Petersen, L.W. Modeling diffusion and reaction in soils. IV. New models for predicting ion diffusivity. *Soil Sci.* **1996**, *161*, 633–645.
13. Xu, X.; Nieber, J.L.; Gupta, S.C. Compaction effects on the gas diffusion coefficient in soils. *Soil Sci. Soc. Am. J.* **1992**, *56*, 1743–1750.
14. Jin, Y.; Jury, W.A. Characterizing the dependency of gas diffusion coefficient on soil properties. *Soil Sci. Soc. Am. J.* **1996**, *60*, 66–71.

Gelisols

James G. Bockheim

University of Wisconsin, Madison, Wisconsin, U.S.A.

Abstract

Gelisols occur in the polar and in some alpine regions and are soils containing permafrost within 100 cm of the soil surface or gelic materials within 100 cm and permafrost within 200 cm of the surface. Gelisols originate from cryopedogenic processes that include cryoturbation, freezing and thawing, cryogenic sorting, ice segregation, and cryodesiccation. The low temperatures in Gelisols give rise to cryostatic pressure development and migration of water with resultant ice buildup. Therefore, Gelisols are differentiated from other soils primarily on the basis of thermal characteristics and physical properties that are readily observed in the field. The three main types of Gelisols are Histels (organic soils with permafrost), Turbels (mineral soils with cryoturbation), and Orthels (other mineral soils with permafrost in the upper 100 cm). The pedon concept is especially important for sampling Gelisols and is determined by the size of the repeating units of patterned ground features, e.g., <2, 2–7, or >7.

INTRODUCTION

Gelisols, the permafrost-affected soils, comprise 13.8 million km² or about 10% of earth's land surface. They occur primarily in the Arctic (83%) and high-mountain regions (17%) (Fig. 1). Because only 0.36% of Antarctica is ice-free, it contains 0.1% of the world Gelisols. Gelisols are of global concern because they contain many protected areas, support indigenous people who depend on the land and surrounding oceans for sustenance, may be subject to considerable impacts from human development (oil, coal, gas, and gas hydrate exploitation), and are already experiencing global warming. Gelisols have been used in relative dating of soils, correlation of glacial deposits, understanding glacier dynamics, and as extraterrestrial analogs.^[1]

Gelisols have permafrost within 100 cm of the soil surface or gelic materials within 100 cm of the surface and permafrost within 200 cm of the surface.^[2] Permafrost is a thermal condition in which a material remains below 0°C for two or more years in succession; it may be ice-cemented or, in the case of insufficient interstitial water, dry. The zone above permafrost that is subject to seasonal thawing is known as the active layer. A transition layer reflecting long-term changes in the active layer often separates the active layer and the underlying permafrost. Gelic materials are defined as seasonally or perennially frozen mineral, or organic soil materials that show cryoturbation (frost churning), ice segregation, or cracking from cryodesiccation.

THE PEDON AS A BASIC SOIL UNIT FOR GELISOLS

The pedon is the basic soil unit for sampling in *Soil Taxonomy*^[2] and is especially important for describing,

classifying, and sampling Gelisols, which often occur in areas with patterned ground. Patterned ground is a general term for any ground surface that is ordered into polygons, nets, circles, or stripes as a result of freezing and thawing processes. Fig. 2 shows mudboils in Antarctica resulting from such processes. Each mudboil is 1–1.5 m across. The pedon is defined so as to encompass the full cycle of patterned ground with a 2-m linear interval or a half cycle with a 2–7-m cycle. This interval is suitable for most patterned ground features. In the case of large-scale (>7 m) patterned ground, such as ice-rich low-center polygons on the North Slope of Alaska, two pedons are established—one within the center of the polygon and the other within the ice wedge. Therefore, each mudboil in the figure represents a single pedon for descriptive and sampling purposes.

CLASSIFICATION OF GELISOLS

There are three suborders within the Gelisol order—Histels, Turbels, and Orthels; they are distinguished on the basis of

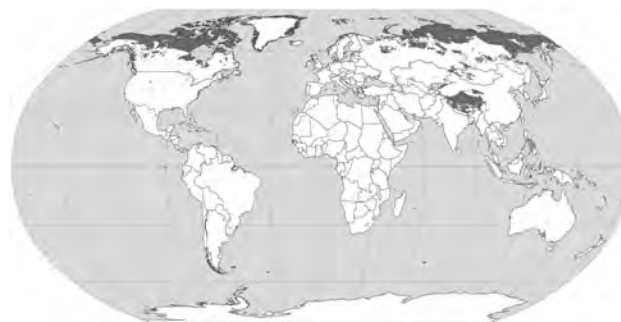


Fig. 1 Distribution of Gelisols in the Northern Hemisphere. **Source:** Natural Resources Conservation Service.

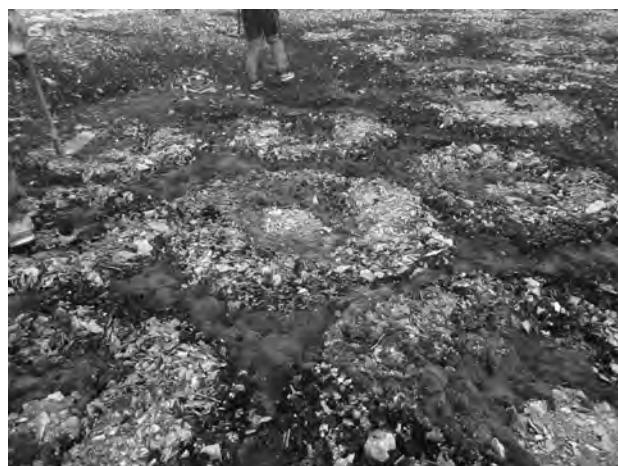


Fig. 2 Mudboils on Nelson Island in the South Shetland Group of Antarctica. Each mudboil is 1–1.5 m across.

organic matter content, and for mineral soils, whether or not there is evidence for cryoturbation.^[2]

Histels have 80% or more organic materials by volume within the upper 50 cm or to a restricting layer. They are subdivided into five great groups based on the nature of the underlying material and the degree of decomposition—Glacistels, Folistels, Fibristels, Hemistels, and Sapristels.^[2]

The Fibristel illustrated in Fig. 3 contains organic soil materials extending into permafrost at 90 cm. Disturbance to the surface-insulating layer may cause ice layers, such as ice wedges, to thaw and the soil to collapse, a condition known as thermokarst (Fig. 4). The key properties of Histels are abundant organic materials ranging from woody to highly amorphous materials, a high moisture-holding capacity, and a pH, that for an array of Histels, may range from as low as 2.5 to above 7.0. Histels comprise 9% of the Gelisols globally, especially in North America and Russia.

Turbels represent a second suborder of the Gelisols; they are mineral soils subject to cryoturbation. Turbels comprise about 56% of the Gelisols on a global basis. Cryoturbation is evidenced by irregular or broken horizons, involutions, organic matter accumulated on the surface of the permafrost, oriented rock fragments, and silt coatings and silt-enriched subsoil horizons that result from freezing and thawing, frost heaving, and cryogenic

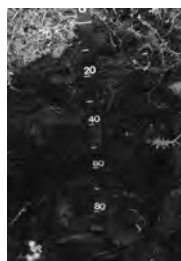


Fig. 3 A Fibristel in northern Alaska. The organic soil materials extend from the surface into the permafrost at 90 cm.



Fig. 4 A landform in Northern Canada containing ice-wedge polygons that have thawed following disturbance of the surface organic materials, resulting in a condition known as thermokarst.

sorting. Cryoturbation is caused primarily by differential frost heave, but its action can be enhanced by cryostatic pressure, differential swelling, and load casting on poorly drained sites.

Fig. 5 shows a Haploturbel in Northern Canada. This pedon shows intense cryoturbation. Although the underlying concept of Turbels is that they are subject to cryoturbation, they may contain diagnostic horizons that are common to soils not having permafrost. However, these horizons are recognized at the great group level. There are seven great groups within the Turbels (Histoturbels, Aquiturbels, Anhyturbels, Molliturbels, Umbrturbels, Psammiturbels, and Haploturbels) that link them with other soils not containing permafrost.^[2]

It should be emphasized that cryopedogenic processes such as cryoturbation, thermal cracking, and ice segregation are soil-forming processes characteristic of soils with



Fig. 5 A Haploturbel in Northern Canada. This pedon showing intense cryoturbation.

Source: From C. Tarnocai.



Fig. 6 A Spodic Psammoturbel in northern Russia. The soil is excessively drained, developed in sandy floodplain deposits, and contains weakly developed and cryoturbated spodic materials in the upper part. Permafrost occurs at 140 cm.

permafrost. They should not be viewed as operating against the other soil-forming processes in low-latitude soils; rather, they are distinctive processes producing horizons and properties that are uncommon to other soil orders. Processes common to the other soil orders operate in Gelisols but at a lesser magnitude because of the dominance of cryopedogenic processes.

The third suborder within the Gelisols is the Orthels, which are other mineral soils containing permafrost within the upper 100 cm. These soils comprise 35% of the Gelisols globally. The Orthels are subdivided into eight great groups (Historthels, Aquorthels, Anhyorthels, Mollorthels, Umbrorthels, Argiorthels, Psammorthels, and Haplorthels) that more or less are parallel to those in the Turbels and that link them to soils not containing permafrost. Fig. 6 shows a Spodic Psammoturbel in northern Russia. The soil is derived from sandy floodplain deposits and contains weakly developed spodic materials.

GELISOLS AND LAND USE

Gelisols offer special challenges to land management for interpretation and practices, including construction (structures and pipelines), mining, forestry, and agriculture. For



Fig. 7 An apartment complex in the Chersky in the Russian Far East that is built on stilts to avoid heat transfer, melting of permafrost, and subsidence.

example, structures must be built on refrigerated pilings or aboveground (Fig. 7) so that heat from the structure does not melt the permafrost and cause subsidence. A main concern with Gelisols is that they are large carbon (C) sinks. The concern is that warming in the arctic could increase the thickness of the seasonal thaw layer and enhance heterotrophic respiration, releasing additional carbon dioxide to the atmosphere. In this case, Gelisols would become a C source. In contrast, warming in Antarctica, especially along the Antarctic Peninsula, is causing recession of glaciers and increases in land area and plant cover, in which case Gelisols would become a C sink.

REFERENCES

1. Bockheim, J.G. *Cryopedology*; Springer: New York, 2014.
2. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed., USDA Natural Resources Conservation Service. Agric. Handbook No. 436; U.S. Government Printing Office: Washington, DC, 1999.

Geologic Carbon Capture and Storage

Sean I. Plasynski

National Energy Technology Laboratory, U.S. Department of Energy, Pittsburgh, Pennsylvania, U.S.A.

John T. Litynski

National Energy Technology Laboratory, U.S. Department of Energy, Morgantown, West Virginia, U.S.A.

Timothy R. Carr

Department of Geology and Geography, West Virginia University, Morgantown, West Virginia, U.S.A.

Howard G. McIlvried

Rameshwar D. Srivastava

National Energy Technology Laboratory, Science Applications International Corporation, Pittsburgh, Pennsylvania, U.S.A.

Abstract

An option that has been proposed to combat the buildup of greenhouse gases in the atmosphere is the capture of carbon dioxide (CO₂) at large stationary sources, such as power plants, and injection into a geologic formation for permanent storage—generally referred to as carbon capture and storage. This entry provides an overview of the major geologic targets in sedimentary basins for long-term storage of CO₂. These targets are oil and gas reservoirs, deep coal beds, and deep saline formations saturated with brackish water or brine.

INTRODUCTION

Atmospheric levels of carbon dioxide (CO₂) have risen significantly from preindustrial levels of 280 parts per million (ppm) to present levels of 384 ppm.^[1] Evidence suggests that the observed rise in atmospheric CO₂ levels is the result of expanded use of fossil fuels for energy. The predictions of increased global energy use indicate a continued increase in carbon emissions^[2] and rising concentrations of CO₂ in the atmosphere unless major changes are made in the way we produce and use energy—in particular, how we manage carbon.^[3,4] One approach to managing carbon is to use energy more efficiently to reduce our reliance on the major carbon emissions source—fossil fuel combustion. Another option is to increase the use of low-carbon and carbon-free fuels and technologies (nuclear power and renewable sources, such as solar energy, wind power, and biomass fuels). The third strategy is to manage carbon through geologic storage (GS), sometimes referred to as geologic sequestration. GS is part of the process of carbon capture and storage (CCS), which involves separation and capture of CO₂ at the point of emission followed by injection and storage in a deep underground geologic formation (Fig. 1).^[5]

CO₂ sinks are a natural part of the carbon cycle; however, due to the concerns about global climate change related to greenhouse gas emissions, efforts are underway to better utilize CO₂ sinks as a form of carbon management to offset emissions derived from fossil fuel combustion and other human activities. The concept of carbon sinks has become more widely known because the Kyoto Protocol^[6] under the United Nations Framework Convention on Climate Change^[7] allows the use of engineered CO₂ sinks as a form of carbon offset. Natural sinks include the oceans, plants and other photosynthetic organisms, and geologic formations.

In 2006, about 28 billion metric tons of CO₂ was emitted globally into the atmosphere, in that the United States' share being about 5.6 billion metric tons.^[8] About half of this came from large stationary sources, primarily coal-fired power plants, which produce flue gases with a low CO₂ concentration (<20%).

GS of CO₂ to limit atmospheric emissions has been underway for more than a decade with projects that have provided significant data and experience, such as Sleipner in Norway,^[9] In Salah in Algeria,^[10] and a joint U.S.–Canadian effort at the Dakota Gasification facility and Weyburn field.^[11] Numerous field projects in saline

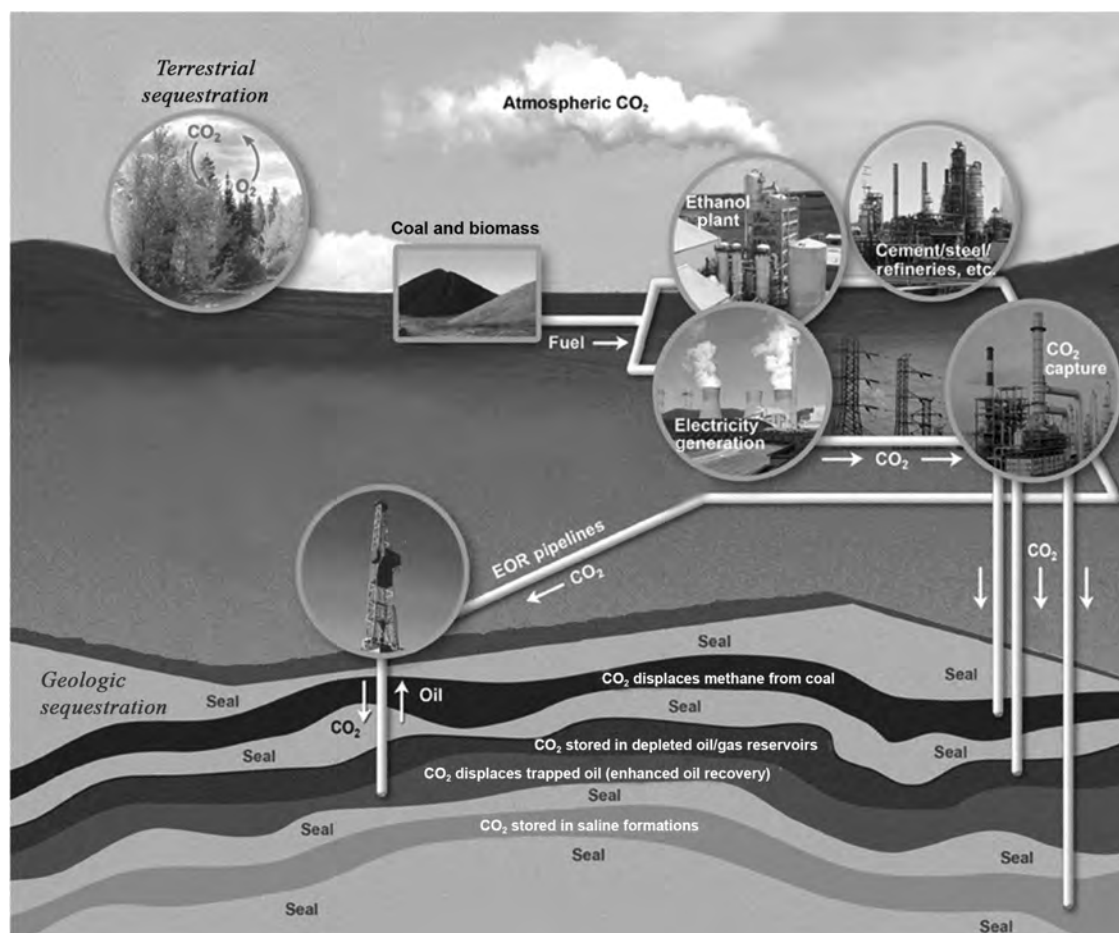


Fig. 1 The illustration of CCS process showing major pathways for geologic and terrestrial storage.

Source: National Energy Technology Laboratory, U.S. Department of Energy.^[5]

formations, oil reservoirs, and coal seams are being developed in the United States and Canada through the U.S. Department of Energy's Regional Carbon Sequestration Partnerships initiative.^[12]

It is expected that large numbers of new power plants and processing facilities based on coal and other nonliquid hydrocarbons (i.e., coal-to-liquid fuels and coal-to-synthetic natural gas plants) will be built in the coming decades, in both the developing world and some areas of the developed world. These new facilities have the potential for being appropriately fitted, during design and construction, with efficient CCS technologies.^[5] Existing plants can be retrofitted for CO₂ capture. If we are to effectively manage the CO₂ emissions from all these plants, the use of geologic carbon sinks will be required.

GS is most efficient at depths greater than 2600 feet (800 m) because CO₂ increases in density and becomes a supercritical fluid at pressures that naturally exist at such depths (Fig. 2). Supercritical fluids take up much less volume and diffuse better through the pore spaces in storage formations than gases or ordinary liquids.

GS TARGETS

GS requires a reservoir and seal (caprock) to trap CO₂ for long periods of time. GS can take place in the following three major geologic targets in sedimentary basins^[13]: deep saline formations, saturated with brackish water or brine; oil and gas reservoirs; and deep coal beds (Fig. 1).

Basalt formations and organic-rich shales have been proposed as potential GS options, but their utility and suitability need to be demonstrated.^[5,14] Keeping the CO₂ in the subsurface requires geologic conditions that create a trap consisting of one or more layers of impermeable rock (caprocks), which are layers of sediments (shale or evaporates) that impede CO₂ movement and act to trap the CO₂ for millennia.

Saline Formations

Saline formations are layers of porous rock, such as sandstone and limestone, which are saturated with brine. They are much more extensive than coal seams or oil- and

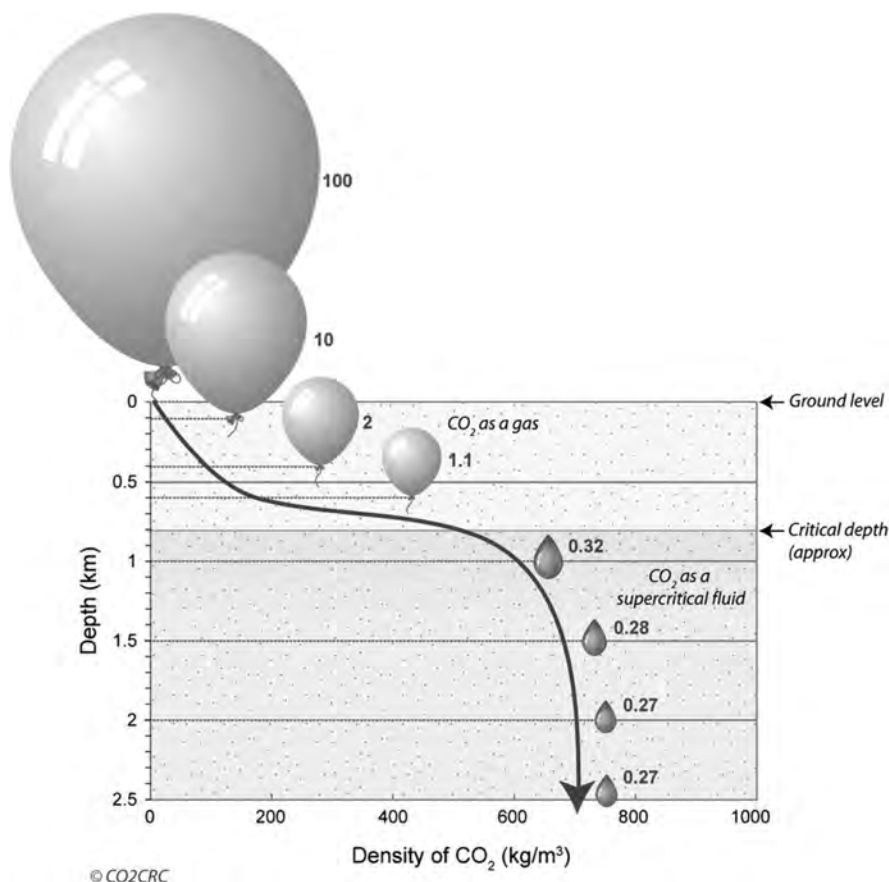


Fig. 2 CO₂ injected at depths below 2600 feet (800 m) increases in density with depth and becomes a supercritical fluid. Supercritical fluids take up much less space and diffuse better than either gases or ordinary liquids through the tiny pore spaces in storage rocks. The blue numbers show the volume of CO₂ at each depth compared to a volume of 100 at the surface.

Source: Data from CO₂CRC (<http://www.co2crc.com.au/imagelibrary/>).

gas-bearing reservoirs and represent an enormous potential for CO₂ storage. However, much less is known about saline formations because of a lack of characterization data, such as that acquired through resource recovery from oil and gas reservoirs and coal seams. Therefore, there is greater uncertainty regarding the volume and suitability of saline formations for CO₂ storage.^[14]

The suitability of deep saline formations for the storage of CO₂ is the result of various physical and chemical mechanisms that act on different timescales, such as dissolution and mineral precipitation. In stratigraphic and structural traps, CO₂ storage capacity can be estimated by following procedures similar to those for oil and gas reservoirs but still involves an understanding of temperature, pressure, and salinity constraints.

In the large areas that do not have well-defined traps, a general rough estimate of capacity can be achieved through time-discounted estimates of dissolution potential. With time, increasingly secure physical and chemical trapping mechanisms come into play, and the overall security of storage in saline formations increases (Fig. 3). However, it is critical to understand the displacement and migration of saline water and the injected CO₂ within these deep formations. The GS potential of saline formations is very large, but accurate capacity estimates require significant local refinement and modeling. The GS potential for saline

formations in portions of the United States has been estimated to range from 900 billion metric tons to more than 3300 billion metric tons.^[5]

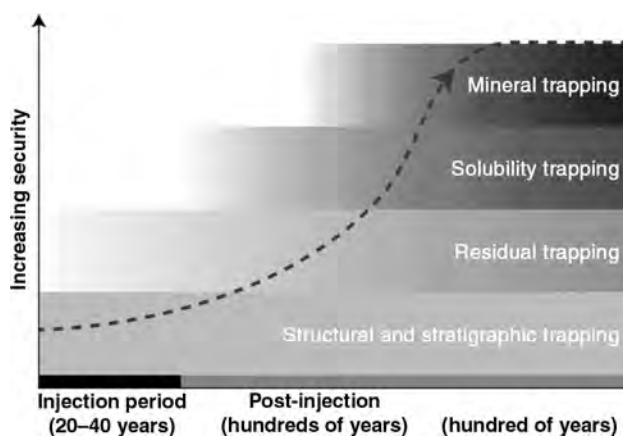


Fig. 3 As time passes, increasingly secure trapping mechanisms come into play, and overall storage security increases. This is especially important in saline formations where relatively well-understood structural and stratigraphic trapping operates over short time frames.

Source: Data from CO₂CRC (<http://www.co2crc.com.au/imagelibrary/>).

Oil and Gas Reservoirs

Mature oil and gas reservoirs have held crude oil and natural gas over millions of years. They consist of a layer of permeable rock with a layer of non-permeable rock (cap-rock) above, such that the non-permeable layer forms a trap that holds the hydrocarbons in place. Oil and gas fields have many characteristics that make them excellent target locations for GS of CO₂. The geologic conditions that trap oil and gas are also the conditions that are conducive to CO₂ storage.

As a value-added benefit, CO₂ injected into a mature oil reservoir can enable incremental oil recovery. A small amount of CO₂ will dissolve in the oil, increasing the bulk volume and decreasing the viscosity, thereby facilitating flow to the wellbore. Typically, primary oil recovery and secondary recovery via water flooding produce 30–40% of a reservoir's original oil in place (OOIP). A CO₂ flood allows the recovery of an additional 10–15% of the OOIP. Oil and gas reservoirs are the best understood of the potential GS options as a result of exploration for, and production of, hydrocarbons. The volume of CO₂ that can be stored in an oil and gas reservoir can be estimated based primarily on the reservoir's depth and size and the volume of produced fluids.^[14]

Enhanced oil recovery (EOR) using CO₂ flooding has proven to be valuable technology in areas with natural CO₂ supplies, as miscible and immiscible CO₂ flooding can revitalize mature oil fields. Limited availability of CO₂ in most regions constrains production from CO₂-based EOR. However, carbon capture applied to advanced electric power generation facilities and other industrial processes could produce large quantities of CO₂, which could be sold to oil and gas producers for use in enhanced oil and gas recovery while at the same time storing CO₂ in reservoirs deep beneath the earth's surface. An initial estimate of GS potential of oil and gas fields in the United State exceeds 83 billion metric tons.^[5]

Coal Seams

Unminable coal seams are seams that are too deep or too thin to be economically mined. All coals have varying amounts of methane adsorbed onto pore surfaces, and wells can be drilled into unminable coal beds to recover this coal bed methane (CBM). Initial CBM recovery methods, such as dewatering and depressurization, leave a considerable amount of methane in the formation. Additional recovery can be achieved by sweeping the coal bed with CO₂. Depending on coal rank and properties, two or more molecules of CO₂ are adsorbed for each molecule of methane released, thereby providing an excellent storage site for CO₂ with the additional benefit of enhanced coal bed methane (ECBM) recovery. Similar to maturing oil reservoirs, unminable coal beds are good value-added candidates for CO₂ storage.

The suitability of a coal seam for CO₂ GS can be evaluated on technical, economic, and regulatory (resource protection) criteria.^[14,15] However, limited operating experience with ECBM technology and poor understanding of CO₂/coal interactions under reservoir conditions have limited storage applications. A number of pilot projects in coal seams are underway or planned to address gaps in knowledge. An initial estimate of GS potential of coal seams in the United State is 150–180 billion metric tons.^[5]

MONITORING, VERIFICATION, AND ACCOUNTING (MVA)

GS of CO₂ requires MVA activities at the storage site, as well as risk assessment and development of mitigation strategies that can be implemented should a problem arise. Effective application of MVA technologies ensures the safety of sequestration projects, with respect to both human health and the environment, and provides the basis for establishing carbon credits on trading markets for sequestered CO₂. Monitoring methods, such as subsurface pressure monitoring, geochemical monitoring, well logging, and surface monitoring, will help demonstrate that the CO₂ is contained in the storage formation and not leaking. Other techniques, such as seismic imaging and electromagnetic imaging, provide an indication of where the CO₂ plume might migrate in the formation so that the monitoring strategies can be modified during the project's life. Risk assessment focuses on identifying and quantifying potential risks to humans and the environment associated with geologic CO₂ storage and helps

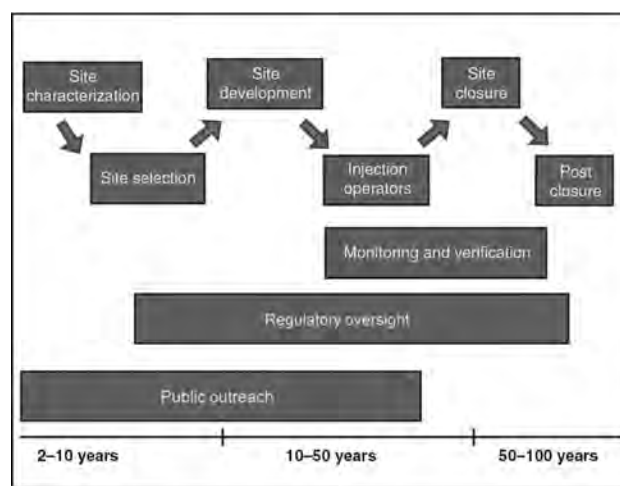


Fig. 4 GS will be a long-term process stretching over decades of active operations and possibly hundreds of years of custodial care. GS of CO₂ will include developing an understanding of storage sites and long-term CO₂ behavior in the subsurface, implementing monitoring and verification techniques, maintaining regulatory oversight, establishing long-term custody of the stored CO₂, and keeping the public informed.

to ensure that these risks remain low. For well-selected, designed, and managed GS sites, risks are estimated to be comparable to those associated with existing hydrocarbon activities.^[13]

CONCLUSION

GS is one of the several strategies for reducing CO₂ emissions to the atmosphere. Several projects injecting CO₂ into geologic formations have been underway for more than a decade, and more than 30 years of industrial experience has demonstrated that CO₂ can be safely transported and injected into the deep subsurface. There are a number of challenges to the widespread implementation of the GS of CO₂, including reducing capital and operating costs, improving understanding of long-term CO₂ behavior in the subsurface, improving monitoring methods, developing a satisfactory regulatory environment, establishing long-term custody of the stored CO₂, and keeping the public informed and accepting of the technology (Fig. 4). Numerous pilot and large-scale field projects are underway or planned throughout the United States^[12] and the world^[13] over the next few years. Information from these projects should go a long way toward facilitating the development of geologic CO₂ storage.

REFERENCES

1. Tans, P. *Trends in Atmospheric Carbon Dioxide—Mauna Loa*; National Oceanic and Atmospheric Administration, U.S. Department of Commerce: Boulder.
2. Energy Information Administration, U.S. Department of Energy. *International Energy Outlook 2007*; Energy Information Administration: Washington, DC.
3. Socolow, R.; Hotinski, R.; Greenblatt, J.B.; Pacala, S. Solving the climate problem—Technologies available to curb CO₂ emissions. *Environment* **2004**, *46* (10), 8–19.
4. Greenblatt, J.B.; Sarmiento, J.L. Variability and climate feedback mechanisms in ocean uptake of CO₂. In *The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World*; Field, C.B., Raupach, M.R., Eds.; Island Press: Washington, DC, 2004; 257–275.
5. National Energy Technology Laboratory, U.S. Department of Energy. *Carbon Sequestration Atlas of the United States and Canada*, 2nd Ed., 2008.
6. United Nations. Third Conference of the Parties, Kyoto, Japan, Dec 1–10, 1997.
7. United Nations. *United Nations Framework Convention on Climate Change*, 1992.
8. Energy Information Administration, U.S. Department of Energy. *Energy-related Emissions and Environmental Analyses*, 2007.
9. Korbol, R.; Kaddour, A. Sleipner vest CO₂ disposal: Injection of removed CO₂ into the Utsira Formation. *Energ. Convers. Manage.* **1995**, *36* (6–9), 509–512.
10. Riddiford, F.; Wight, I.; Bishop, C.; Tourqui, T.A. Monitoring geological storage in the In Salah gas CO₂ storage project. In *Proceedings of the 7th International Conference on Greenhouse Gas Control Technologies (GHGT-7)*, Vancouver, Sept 5–9, 2004.
11. Moberg, R.; Stewart, D.; Stachniak, D. The IEA Weyburn CO₂ monitoring project. In *Proceedings of the Sixth International Conference on Greenhouse Gas Control Technologies*, Kyoto, Japan, October 1–4; Gale, J., Kaya, J., Eds.; 2002; 219–224.
12. National Energy Technology Laboratory, U.S. Department of Energy. *Regional Carbon Sequestration Partnerships*. Available at <http://energy.gov/fe/science-innovation/carbon-capture-and-storage-research/regional-partnerships> (accessed August 20, 2015).
13. Benson, S.M.; Cook, P.; Anderson, J.; Bachu, S.; Nimir, H.B.; Basu, B.; Bradshaw, J.; Deguchi, G.; Gale, J.; von Goerne, G.; Heidug, W.; Holloway, S.; Kamal, R.; Keith, D.; Lloyd, P.; Rocha, P.; Senior, B.; Thomson, J.; Torp, T.; Wildenborg, T.; Wilson, M.; Zarlenga, F.; Zhou, D.; Celia, S.M.; Gunter, B.; Ennis King, J.; Lindegerg, E.; Lombardi, S.; Oldenburg, C.; Pruess, K.; Rigg, A.; Stevens, S.; Wilson, E.; Whittaker, S. Underground geological storage. In *Special Report on Carbon Dioxide Capture and Storage*, IPCC—Intergovernmental Panel on Climate Change; Metz, B., Davidson, O., de Coninck, H.C., Loos, M., Meyer, L.A., Eds.; Cambridge University Press: Cambridge and New York, 2005; 195–276.
14. Bachu, S.; Bonijoly, D.; Bradshaw, J.; Burrussd, R.; Holloway, S.; Christensenf, N.P.; Mathiassen, O.M. CO₂ storage capacity estimation: Methodology and gaps. *Int. J. Greenh. Gas Con.* **2007**, *4*, 430–443.
15. Oudinot, A.; Schepers, K.; Reeves, S. Gas injection and breakthrough trends as observed in ECBM sequestration pilot projects and field demonstrations. In *International Coalbed Methane Symposium*, Tuscaloosa, AL, May 21–25; 2007, Paper No. 0714.

Geology and Soil

Kenneth R. Olson

Department of Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.

Abstract

Geology is the study of earth, its internal structure, its materials, its chemical and physical processes, and its physical and biological history. One of the most important geological discoveries was that rocks could form by crystallization of molten material. Rocks do change from one kind to another. As rocks get exposed to the surface weather, particles move downstream, eventually to be deposited as sediments that lithify into sedimentary rocks. By tracing distributions of rock materials, early geologists were able to link sediments to highly consolidated, mineralogically distinct metamorphic rocks.

ATMOSPHERE, HYDROSPHERE, AND LITHOSPHERE

The crust, mantle, and core account for 99.97% of the mass of the earth.^[1] The lithosphere includes the earth's crust and upper mantle. The term has previously been used to describe the entire portion of the earth that is composed of rocks. The remaining 0.03% comprises the atmosphere and the hydrosphere. The hydrosphere includes the portion of the earth's surface that is covered by water. The hydrosphere behaves as an intermediate reservoir for carbonates, silicates, and other mineral groups leached from the rock of the continents and carried to the oceans by rivers.^[1] Both silicates and calcium carbonate follow involved paths from the time they are weathered from continental rock, until they are deposited on the sea floor.

The composition of the atmosphere is strongly influenced by the cycling of water from the oceans to the continents and by its return to the oceans in rivers and subsurface flow. The amount of free oxygen in the atmosphere seems to remain essentially constant. Volcanic gases contain only traces of molecular oxygen but eject large quantities of molecular hydrogen, carbon monoxide, and sulfur dioxide. These gases react with atmospheric oxygen to produce carbon dioxide, water vapor, and sulfur dioxide. The lithosphere, which consumes free oxygen through weathering of rock, and the biosphere, which produces oxygen through photosynthesis, maintain this equilibrium. The control over this equilibrium may be a feedback mechanism involving the organisms whose by-products and remains become constituents of sedimentary rock.

CYCLING OF ELEMENTS

Oxygen, silicon, aluminum, and iron contribute 96% by volume of the elemental composition of the earth's crust.^[2]

The other elements only occupy the remaining 4%. Individual atoms or molecules of such elements such as carbon, nitrogen, oxygen, and sodium change form countless times as they cycle through the atmosphere, biosphere, and lithosphere.^[1] Oxygen, for instance, may be converted from a neutral atom to dissolved gas, from combined molecule to ionized particle, to protoplasmic components, and, perhaps, back to the ionic state as a constituent of rock.^[1] Other atoms, such as sodium or silicon, may be more constrained in the variety of chemical forms they may assume. These and other representative routes are indicated in a revised and modified scheme shown in [Fig. 1](#).

CHEMICAL WEATHERING PROCESSES

"Weathering" is a collective term for the combined effects of all the physical and chemical processes that break down and transform pre-existing rock materials near the earth's surface to products that are more stable under the physical and chemical conditions at the surface. Weathering constitutes the response of rock materials to several forms of energy as a function of time.^[1] The products of weathering include solids (i.e., sediments and soils) and liquids (including the solutions of salts present in rivers and the ocean). Each physical and chemical factor of weathering affects the outcome of the weathering process. These factors and related variables of the rock cycle, including erosion, transportation, and sedimentation, combine to operate as a complex chemical sorting system which distributes products of different composition to various sites of deposition.

Soils are the result of leaching, oxidation, and dissolution of surface materials by the percolation of groundwater and by humic acids derived from oxidized organic materials. New minerals, such as clays, are formed by the chemical alteration of the original mineral of the bedrock.^[1]

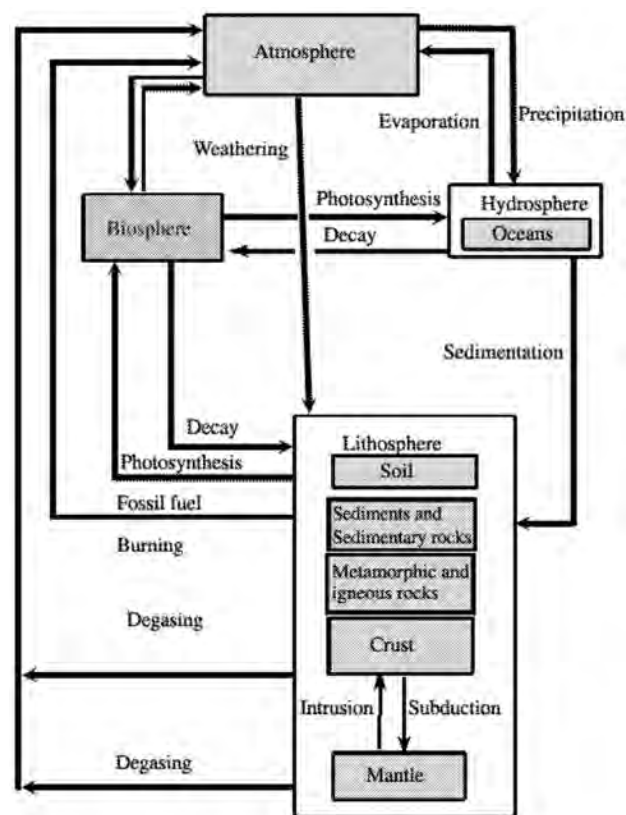


Fig. 1 Cycling of elements.

Source: From Jackson.^[1]

The chemical weathering process develops a soil profile ranging from heavily weathered surface materials down to unaltered rock. The soil zone is the zone of transition between solid rock and the atmosphere. The solid rock, or bedrock, usually has numerous tiny cracks or joints. Chemical weathering caused by water, which fills the cracks, attacks the joints' surfaces and enlarges the cracks.

PARENT MATERIAL

Parent material is the initial or starting material from which a soil develops. This initial material can include either consolidated rock or unconsolidated material deposited by gravity, wind, or water and consists of specific minerals of different sizes, or even plant materials of various plant types.^[2] Mineral matter inherited from rocks is referred to as soil parent material because it is the principal ingredient from which soils are formed.^[3] However, the principal parent materials of organic soils are decomposing plants.

In many cases, relief prior to and during soil formation is related to the nature of the initial soil material. In broad river deltas, crests of natural levees near the stream channels have more coarse material than the areas beyond the levees that are nearly level and have the finer textured

initial material.^[4] In steeper topography, where the valleys below the mountain ranges are characterized by broad alluvial-colluvial fans, the initial material near the mountain range contains more coarse and angular material than areas farther away from the mountain range.^[5]

After unconsolidated parent material is deposited on a stable landscape, or after bedrock has been exposed at the earth's surface, soil formation begins and continues over time. The rate of soil formation depends on the climate, including temperature and rainfall. It also depends on vegetation and the activity of other organisms, which live on or in the parent material. These organisms help convert parent materials to soil.

Russian pedologists^[6] identified parent material as one of the five significant soil-forming factors. Early approaches to soil survey and classification were based on the geology and composition of the soil-forming material.^[7,8] The geologic origin and composition of the initial material were identified by the terms "Agranite soils" or "Aglacial soils." Soils that originate from a parent material inherit the mineral types found in them. Over time, these original minerals are weathered (dissolved) and new minerals form and accumulate in the soils.^[2]

Russian soil scientists^[9] showed the controlling effects of parent material on soil properties. Jenny^[10] conducted a systematic analysis of the relationship between soil properties and parent materials from which the soils developed. Jenny proposed parent materials as an independent soil-forming factor, defining parent material as "the state of the soil system at time zero of soil formation." The physical body of soil and its associated mineralogical and chemical properties are the starting point for the interaction of other soil-forming factors. Previous weathered rock—even a previous soil—could be considered as parent material.

The properties of modern soil are the result of the composition of the surficial layer, which existed when the other factors started to impact, and the alterations resulting from the effect of these factors over time. Properties of younger soils are greatly influenced by parent material. Weathering and pedogenic processes result in characteristics of the original parent material being eliminated. Extremely resistant initial material, such as quartz or sand, may still exist in old, weathered soils. It can be difficult to separate the nature (or characteristics) of the initial material and its influence on soil, the kind of "preweathering" of the initial material before becoming parent material for the soil, and the effects of the other soil-forming factors on the parent material of this soil. The complex environmental history (climatic and vegetation changes in the geologic past) makes it difficult to separate parent material influences on soil properties from other factor effects.

Rock types influence the soil properties of the modern soil. The impact of rock type on soil properties was organized by Buol, Hole, and Mc Craken^[11] into the following subdivisions: sedimentary, igneous, and combinations of mineralogically similar metamorphic and igneous rocks.

Sedimentary rocks include unconsolidated glacial deposits, loessial deposits, unconsolidated coastal plain sediments, and consolidated rock, such as limestone and dolomites, sandstones, and shales.

Siliceous crystalline rocks include more “acidic” quartzose, igneous, and metamorphic rocks including granites, granite gneiss, and schists. Dark-colored ferromagnesian rocks include andosites, diorites, basalt, and hornblende gneiss. Volcanic ash parent materials are composed of non-crystalline materials, any glass fragments, bits of the easily weatherable feldspars, ferromagnesian minerals, and varying amounts of quartz.

Mineral components of many soils are inherited almost exclusively from parent materials, while others are developed mostly *in situ* during the course of weathering and pedogenesis. Primary minerals are formed at high temperature in igneous and metamorphic rocks. Secondary minerals are formed at lower temperature in sedimentary rocks and soils.^[12]

CONCLUSION

The basic model of soil implies that soils are dynamic and geographical. In soil systems, the processes or driving forces are best described as dynamic rather than static. Morphological properties of soil are the result of processes acting on parent materials. In addressing the influence of parent materials in soil genesis, Chesworth^[13,14] stated that “time has the result of modifying the parent material effects so that only in young or relatively immature soils will the parent material exert its strongest influence on the soil-forming process. That influence will be an inverse function of time.”

REFERENCES

1. Jackson, C. *Geology Today*; CRM Books, Communications Research Machines: Del Mar, 1973.
2. Singer, M.J.; Munns, D.N. *Soils, An Introduction*, 3rd Ed.; Prentice Hall: Upper Saddle River, 1996.
3. Troeh, F.R.; Thompson, L.M. *Soils and Soil Fertility*, 5th Ed.; Oxford University Press: New York, 1993.
4. Russell, R.J. *River Plains and Sea Coasts*; University of California Press: Berkeley, 1967.
5. Bitor, P. *The Cycle of Erosion in Different Climates*; Translated by Jackson, C.I.; Clayton, K.M., University of California Press: Berkeley, 1968.
6. Dokuchaev, V.V. Russian Chernozem (Russkii Chernozem). In *Collected Writings (Sochineniya)*; Academy of Science: Moscow, 1883; Vol. 3.
7. von Richthofen, F.F. *Fuhrer fur Forschungsreisende*; (cited in Joffe, J.S., 1936; Pedology, Rutgers University press: New Brunswick, 1886.
8. Thaer, A.D. *Grundsätze der Rationaellen Landwirtschaft*; (cited in Joffe, J.S., 1949; Pedology Publ.: New Brunswick, 1809, 1810, 1812; Vol. 1–4.
9. Polynov, B.B. Das Muttergestein als Faktor de Bodenbildung und als Kriterium fur die Bodenklassifikation. *Soil Res.* **1930**, 2, 165–180.
10. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
11. Buol, S.W.; Hole, F.D.; Mc Cracken, R.J. *Soil Genesis and Classification*, 2nd Ed.; The Iowa State University Press: Ames, 1980.
12. Jackson, M.L. Chemical composition of soils. In *Chemistry of Soil*; Bear, F.E., Ed.; 2nd Ed.; Reinhold: New York, 1964.
13. Chesworth, W. The parent rock effect in the genesis of soil. *Geoderma* **1973**, 10, 215–225.
14. Chesworth, W. Conceptual models in pedogenesis: A rejoinder. *Geoderma* **1976**, 16, 257–260.

Geomorphology

Robin N. Thwaites

*School of Natural Resource Sciences, Queensland University of Technology,
Brisbane, Queensland, Australia*

Abstract

The principles, concepts, and models of soil geomorphology are most effectively applied in surveys and mapping of soil landscapes. Many types and methods of survey have been, employed, but modern technology and computer modeling have advanced the application of soil geomorphological principles and techniques. There are many other applications of soil geomorphological techniques—primarily, research into landscape evolution and climate change, quaternary studies, landscape ecology and paleoecology, archaeology, paleontology, environmental engineering, and mineral exploration. Technology for remote sensing of the land surface and soil materials has also aided soil geomorphological investigation, particularly the use of airborne and satellite environmental sensors, as well as ground-based sensing. The greatest impact of all techniques in soil geomorphology, however, has been that of computer spatial modeling, particularly the use of digital terrain models as land surface analogies for analyzing the dynamics and effects of hillslope and drainage basin processes.

INTRODUCTION

The soil, the landscape in which it forms, the materials from which it forms, and the processes that form it all constitute the 3-D systems through time that are part of the science of geomorphology. Soil geomorphology (or pedogeomorphology) is the study of the evolution (temporal aspects) and distribution (spatial aspects) of soil and soil materials and the landscapes in which they are formed and altered.^[1] It is a science that relies strongly on geological and geomorphological principles and techniques.^[2] These have unique application to soil that augments the science of pedology to express the concepts of soil landscapes and soil landscape systems. As such, soil geomorphology provides: 1) principles for the understanding of the soil landscape and 2) a process by which models of soil material variation and distribution at all scales can be derived. It is an assessment of the genetic relationships between soils and landforms and the resulting landscapes.^[3]

SOIL GEOMORPHOLOGICAL CONCEPTS

The roots of the soil geomorphological approach can be traced to the more applied aspects of soil science in the 20th century from the need for mapping soil distribution. Thus, there is the fundamental concept of the soil catena from Milne^[4,5] and the procedure of Integrated Survey in Australia in the 1940s and 1950s to produce land systems^[6] that express the application of what are now termed soil geomorphological principles and the concepts laid out by

Ruhe^[7–9] in the United States and by Butler,^[10] among others, in Australia. From this basis, the more theoretical and specific expressions of soil geomorphology have been developed by Conacher and Dalrymple,^[11] who coined the term pedogeomorphology from their work in New Zealand and Australia; Birkeland,^[12,13] Daniels,^[2,14] and others in the United States; Gerrard^[3] and Huggett^[15] in the United Kingdom; Dan Yaalon in the arid lands; and Paton^[16] and Pain and Ollier^[17] among others in Australia. The incorporation of geological and geomorphological principles^[18,19] into pedology allows the investigation and better understanding of soil materials and processes at different spatial and temporal scales as well as understanding soil as earth materials in the geological landscape.

Geomorphic and pedological processes act together on the hillslope, which can be viewed as the fundamental 2-D unit of study. This is the basis of the catena concept and the expression of the “toposequence” that arises from referral to the state factor model of soil formation.^[20] Later, soil geomorphic studies also focused on temporal development, investigating “chronosequences” for pedogenetic and landscape evolution, which, again, had developed as a function of the state factor model. However, from a more spatial geomorphological viewpoint, hillslope processes together form a *soil geomorphological system* from which the fundamental 3-D unit, a soil landscape unit,^[15] is derived. One of the important principles of soil geomorphology is the development and expression of the concepts of “soil landscapes” and “soil landscape systems.”^[15,17]

The physical focus of soil science, through its basis of soil–plant interactions and land use management, has

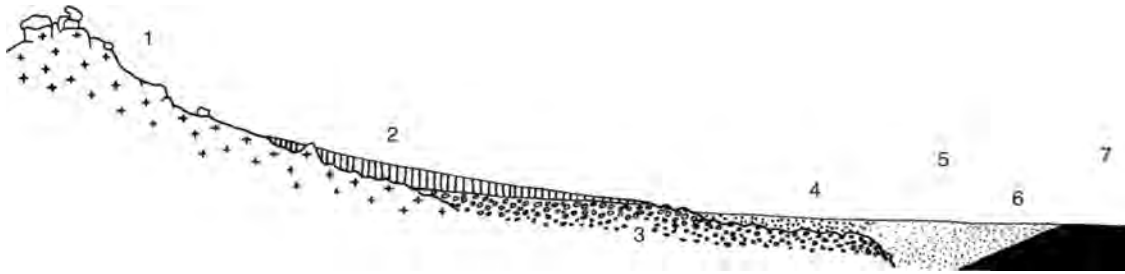


Fig. 1 Milne's original catena on a uniform parent material (granite): (1) shallow skeletal dark gray loam, (2) deeper red earth, (3) coarse granite grit in ferruginous cement, (4) washed sand, (5) silty sand, (6) clayey sand, and (7) clay floor.

Source: From Redrawn from Milne.^[4]

traditionally been more on the solum (the A and B soil horizons) rather than on the whole regolith (the solum plus the C/D soil horizons, saprolite, and other non-bedrock materials), though there have been exceptions to this, largely concerning the nature and influence of parent material, particularly with glacial materials in the mid-United States. Therefore, another major principle for soil geomorphology is to include the nature and dynamics of the regolith and geologic materials in models of soil evolution and distribution as part of soil landscape systems.

SOIL GEOMORPHOLOGICAL MODELS

The fundamental basis for models of soil geomorphology is that of surface and subsurface hillslope processes as an open system. The original model in this respect is the

catena, expressed by Milne^[4,5] and expanded to other environmental scenarios by Gerrard,^[3,21] whereby natural surface processes operating over distance along hillslopes are invoked to explain characteristic pattern of soil material variation (Fig. 1).

This coupled with the acknowledgment of the influences of topography and parent material over time from the state factor model^[20] of pedogenesis [itself a reemphasis of Dokuchaev's (1883) pedological model], and the explicit relationships between geomorphology and pedogenesis made by Wooldridge^[22] laid the foundation for the soil geomorphological conceptual models of the 1950s and 1960s. The revision of the pedogenetic paradigm by Simonson^[23] to introduce the concept of it as an open, dynamic system of accessions, removals, transfers, and transformations then provided the basis for the 3-D soil geomorphological system models that are accepted. The introduction of geomorphological principles and the

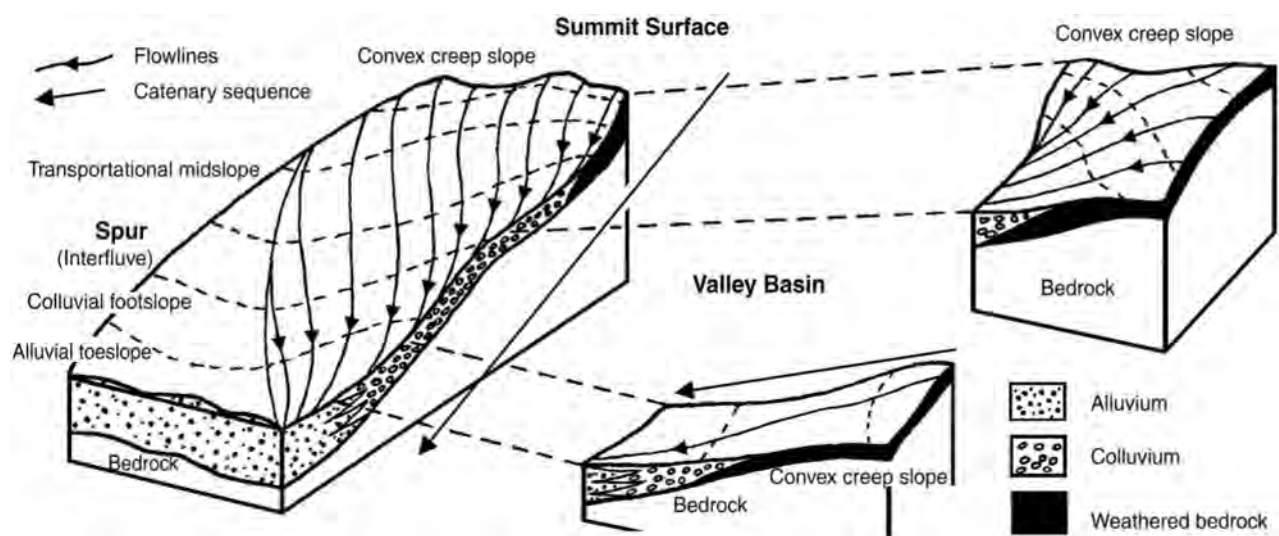


Fig. 2 The 3-D, dynamic soil landscape system as a primary soil geomorphological model. Catenary sequences in multiple directions follow surface and subsurface water flow lines within a drainage basin landform.

Source: Adapted from Huggett.^[15]

hydrological processes operating within the hillslope system over varying depths of time were the tenets of the first real soil geomorphological models: by Ruhe^[8,9] in the United States, by Butler^[10] as cycles of stability and instability on the hillslope in Australia, then by Dalrymple, Blong, and Conacher^[24] as the dynamic, process-based “9-unit land surface” model as the basis for *pedogeomorphic* (soil geomorphological) research.^[11] They believe that there are four main aspects to soil geomorphological research that distinguishes it from either pedology or geomorphology: the conceptualization, the classification, the modeling, and the explanation of soil–slope relationships at all scales.

More functional and mechanistic 3-D soil geomorphic system models were developed in the 1970s and 1980s, the most outstanding of which is that presented by Huggett^[15] of the soil landscape system (Fig. 2), which expresses meso- and macroscale dynamics of soil materials and water over drainage basins and treats natural erosion and deposition as variable but continuing processes over time and space. The advent of more powerful computer modeling facilities has allowed for complex process simulation and analysis of these system models through digital terrain models (DTMs) as mathematical representations of the land surface.

CONCLUSION

The principles, concepts, and models of soil geomorphology are most effectively applied in surveys and mapping of soil landscapes. Many types and methods of survey have been employed, but modern technology and computer modeling have advanced the application of soil geomorphological principles and techniques. There are many other applications of soil geomorphological techniques: primarily, research into landscape evolution and climate change, quaternary studies, landscape ecology and paleoecology, archaeology, paleontology, environmental engineering, and mineral exploration. Many of these techniques are outlined by Daniels and Hammer^[2] and Gerrard,^[3] such as soil stratigraphical interpretation, tephrochronology in volcanic regions, radioisotope dating of materials and relative dating of land surfaces, and the use of environmental and paleoenvironmental indicators. Technology for remote sensing of the land surface and soil materials has also aided soil geomorphological investigation, particularly the use of airborne and satellite environmental sensors [e.g., airborne gamma ray spectrometry (or “radiometrics”), airborne electromagnetic (EM) induction, geomagnetism, and satellite multichannel infrared and hyperspectral sensors] as well as ground-based

sensing (e.g., ground penetrating radar, EM sensors, and radio spectrometry). The greatest impact of all techniques in soil geomorphology, however, has been that of computer spatial modeling, particularly the use of DTM as land surface analogies for analyzing the dynamics and effects of hillslope and drainage basin processes. Such advances are outlined and discussed well by Lane, Richards, and Chandler^[25] and Wilson and Gallant.^[26] This has allowed much complex theoretical and functional modeling of soil landscape processes (via the geomorphic effects of topography) over space and time to aid the task of soil landscape survey and mapping as well as soil geomorphic process simulation and prediction for environmental problem solving.

REFERENCES

1. Wysocki, D.A.; Schoeneberger, P.J.; LaGarry, H.E. Geomorphology of soil landscapes. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; E1–E40.
2. Daniels, R.B.; Hammer, R.D. *Soil Geomorphology*; John Wiley & Sons: New York, 1993.
3. Gerrard, J. *Soil Geomorphology: An Integration of Pedology and Geomorphology*; Chapman & Hall: London, 1992.
4. Milne, G. Some suggested units of classification and mapping, particularly for East African soils. *Soil Res.* **1935**, *4*, 183–198.
5. Milne, G. Normal erosion as a factor in soil profile development. *Nature* **1936**, *138*, 548–549.
6. Christian, C.S.; Stewart, G.A. Methodology in integrated surveys. In *Aerial Surveys and Integrated Studies*, Proceedings of the Toulouse Conference; Sept 21–28, 1964; Natural Resources Research No. 6. UNESCO: Paris, 1968; 233–280.
7. Ruhe, R.V. Geomorphic surfaces and the nature of soils. *Soil Sci.* **1956**, *82*, 441–455.
8. Ruhe, R.V. Elements of the soil landscape. In *Transactions 7th International Congress of Soil Science*; Madison, 1960; Vol. 4, 165–170.
9. Ruhe, R.V.; Daniels, R.B.; Cady, J.G. *Landscape Evolution and Soil Formation in Southwestern Iowa*. USDA Tech. Bull. 1349, **1967**.
10. Butler, B.E. *Periodic Phenomena in Landscapes as a Basis for Soil Studies*; CSIRO Australian Soil Publication No. 14. Commonwealth Scientific and Industrial Research Organization: Melbourne, 1959.
11. Conacher, A.J.; Dalrymple, J.B. The nine-unit land surface model: An approach to pedogeomorphic research. *Geoderma* **1977**, *18* (1–2), 1–154.
12. Birkeland, P.W. *Soils and Geomorphology*; Oxford University Press: New York, 1984.
13. Birkeland, P.W. Soil-geomorphic research: A selective overview. *Geomorphology* **1990**, *3*, 207–224.

14. Daniels, R.B.; Gamble, E.E.; Cady, J.G. The relation between geomorphology and soil morphology and genesis. *Adv. Agron.* **1971**, *23*, 51–88.
15. Huggett, R.J. Soil landscape systems: A model of soil genesis. *Geoderma* **1975**, *13* (1), 1–22.
16. Paton, T.R. *The Formation of Soil Material*; George Allen & Unwin: London, 1978.
17. Pain, C.; Ollier, C. *Regolith, Soils and Landforms*; John Wiley: New York, 1996.
18. Holmes, A. *Principles of Physical Geology*; Thos. Nelson & Sons: London, 1965; 1288 pp.
19. Thornbury, W.D. *Principles of Geomorphology*, 2nd Ed.; Wiley: New York, 1969.
20. Jenny, H. *Factors of Soil Formation. A System of Quantitative Pedology*; McGraw-Hill: New York, 1941.
21. Gerrard, A.J. *Soils and Landforms*; George Allen & Unwin: London, 1981.
22. Wooldridge, S.W. Geomorphology and soil science. *J. Soil Sci.* **1950**, *1* (1), 31–34.
23. Simonson, R.W. Outline of a generalised theory of soil genesis. *Proc. Soil Sci. Soc. Am.* **1959**, *23*, 152–156.
24. Dalrymple, J.B.; Blong, R.J.; Conacher, A.J. An hypothetical nine unit land surface model. *Z. Geomorph.* **1968**, *12*, 60–76.
25. Lane, S.N.; Richards, K.S.; Chandler, J.H., Eds. *Landform Monitoring, Modelling and Analysis*; Wiley: New York, 1998; 219–240.
26. Wilson, J.P.; Gallant, J.C., Eds. *Terrain Analysis. Principles and Applications*; John Wiley: New York, 2000.

Geophysical Methods

Barry J. Allred

Soil Drainage Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Columbus, Ohio, U.S.A.

Viacheslav I. Adamchuk

Bioresource Engineering, McGill University, Ste-Anne-de-Bellevue, Quebec, Canada

Raphael A. Viscarra Rossel

Bruce E. Butler Laboratory, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia

Jim Doolittle

National Soil Survey Center, U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Newtown Square, Pennsylvania, U.S.A.

Robert S. Freeland

Biosystems Engineering and Soil Science Department, University of Tennessee, Knoxville, Tennessee, U.S.A.

Katherine R. Grote

Geosciences and Geological and Petroleum Engineering Department, Missouri University of Science and Technology, Rolla, Missouri, U.S.A.

Dennis L. Corwin

United States Soil Salinity Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Riverside, California, U.S.A.

Abstract

Near-surface geophysical methods have become an important tool for agriculture. Geophysical investigations for agriculture are most often focused on the first 2 meters directly beneath the ground surface, which includes the crop root zone and all, or at least most of the soil profile. Resistivity, electromagnetic induction, and ground-penetrating radar are the three geophysical methods most commonly employed for agricultural soil investigations; however, optical reflectance and γ -ray spectroscopy are increasingly becoming more widely utilized. Temporal and spatial variation of conditions and properties in the soil profile are important considerations when conducting a geophysical survey within an agricultural setting. Geophysical methods have been applied to soil surveys, precision farming, soil water content measurement, and soil salinity monitoring; with new agricultural geophysics applications continuing to evolve. Future development of multi-sensor platforms and the use of unmanned aerial vehicles will dramatically improve geophysical soil investigation capabilities with respect to field accessibility and data interpretation.

INTRODUCTION

Agricultural geophysics as described in this entry involves the application of physical quantity measurement techniques to provide information on conditions or features within the soil environment. With the exception of borehole geophysical methods and soil probes with electrical conductivity, optical, and penetration resistance sensors, these techniques are generally non-invasive with physical quantities determined from measurements made mostly on or

directly above the ground. Agricultural geophysics tends to be heavily focused on a 2 m zone directly beneath the ground surface, which includes the crop root zone and all, or at least most, of the soil profile.^[1] Complexities encountered with agricultural geophysics include transient soil temperature and moisture conditions, which can appreciably alter, over a period of days or even hours, the values of measured soil physical quantities (especially electrical conductivity and dielectric constant (K)). Additionally, physical quantities measured in the soil environment with

geophysical methods often exhibit substantial variability over very short horizontal and vertical distances. The three geophysical methods most commonly used for agricultural purposes are resistivity, electromagnetic induction (EMI), and ground-penetrating radar (GPR). However, optical reflectance and gamma (γ)-ray spectroscopy are gaining more widespread utilization. All five of these employed agricultural geophysics methods are summarized in the next section of this entry.

GEOPHYSICAL METHODS

Resistivity

The resistivity method, employed in its most conventional form, uses an external power source to supply electrical current between two “current” electrodes inserted at the ground surface. The propagation of current in the subsurface is three-dimensional, and so too is the associated electric field. Information on the electric field is obtained by measuring the voltage between a second pair of “potential” electrodes also inserted at the ground surface. The two current and two potential electrodes together comprise a single four-electrode array. The magnitude of the current applied and the measured voltage are then used in conjunction with data on electrode spacing and arrangement to determine an apparent soil electrical conductivity (EC_a), for a bulk volume of soil. The current and potential electrodes are often arranged inline, and the depth of investigation increases with increased length of the electrode array.

Continuous measurement galvanic contact resistivity systems integrated with global positioning system (GPS) receivers have been developed in the United States and Europe specifically for farm field soil investigations (Fig. 1). These resistivity systems can have more than one four-electrode array providing shallow investigation depths of 0.3–2 m, with short time or distance intervals between the continuously collected discrete EC_a measurements. The location for each EC_a measurement is determined accurately by GPS. For the system shown in Fig. 1, steel coulters (disks) that cut through the soil



Fig. 2 EMI DUALEM-21 S GCM (DUALEM, Milton, Ontario, Canada) measuring soil electrical conductivity at a golf course green.

surface are utilized as current and potential electrodes. Maps of EC_a spatial patterns across a farm field are generated with resistivity survey data to gain insight on lateral variations in soil properties/conditions, such as water content, soil texture, organic matter content, and so on.

Electromagnetic Induction

Electromagnetic induction (EMI) methods also measure the EC_a for a bulk volume of soil directly beneath the surface. An instrument called a ground conductivity meter (GCM) is commonly employed for relatively shallow EMI investigations (Fig. 2). In operation, an alternating electrical current is passed through one of two small electric wire coils spaced a set distance apart and housed within the GCM, which itself is positioned at, or a short distance above, the ground surface. The applied current produces an electromagnetic field around the “transmitting” coil, with a portion of the electromagnetic field extending into the subsurface. This electromagnetic field, called the primary field, induces an alternating



Fig. 1 Example of a continuous measurement galvanic contact resistivity system, Veris 3100 Soil EC Mapping System (Veris Technologies, Salina, Kansas, United States; left) and close-up of steel coulters used for current and potential electrodes by the Veris 3100 Soil EC Mapping System (right).

electrical current within the ground, in turn producing a secondary electromagnetic field. This secondary field extends back to the surface and the air above. The second wire coil acts as a receiver measuring the resultant amplitude and phase components of both the primary and secondary fields. The amplitude and phase differences between the primary and secondary fields are then used, along with the intercoil spacing, to calculate an “apparent” value of EC_a for a bulk volume of soil. Provided low induction number conditions are satisfied, the EMI depth of investigation is dependent on the spacing distance between the electric wire coils within the GCM and the orientation (vertical, horizontal, perpendicular) of the electric wire coils.^[2] As with resistivity methods, maps of EC_a spatial patterns across a farm field are generated with EMI data to gain insight on lateral variations in soil properties/conditions, such as water content, soil texture, organic matter content, etc.

Ground-Penetrating Radar

A ground-penetrating radar (GPR) system directs an electromagnetic radio energy (radar) pulse into the subsurface, followed by measurement of the elapsed time taken by the radar signal, as it travels downward from the transmitting antenna, partially reflects off subsurface boundaries or objects, and is eventually returned to the surface, where it is picked up by a receiving antenna. Reflections from different depths produce a signal trace, which is a function of radar wave amplitude vs. time. Radar waves that travel along direct and refracted paths through both air and ground from the transmitting antenna to the receiving antenna are also included as part of the signal trace. Antenna frequency, soil moisture conditions, clay content,

salinity, and the amount of iron oxide present all have a substantial influence on the distance beneath the surface to which the radar signal penetrates.^[3] The K of a material governs the velocity for the radar signal traveling through that material. Differences in K across a subsurface discontinuity feature control the amount of reflected radar energy, and hence radar wave amplitude, returning to the surface. GPS technologies are often integrated with GPR systems to obtain accurate and precise geographic positions of GPR measurements and to improve the efficiency of GPR field surveys (Fig. 3). As an end product, radar signal amplitude (energy) data are displayed as two-dimensional depth sections or aerial maps to gain insight on belowground conditions or to provide information on the positions and character of subsurface features. The GPR data collected can be used to measure soil volumetric water content (VWC), determine soil horizon depths/thicknesses, and map buried agricultural drainage pipe systems.

Optical Reflectance (UV/VIS/NIR/MIR)

Optical sensors are used to determine the soil’s ability to reflect light in different parts of the electromagnetic spectrum. The light can be from the sun or from an artificial source. Proximal optical sensors are fundamentally the same as optical remote sensors; the advantage of proximal sensors is that they can provide measurements at either the soil surface or within the soil. These proximal optical sensors can be used for on-the-spot or on-the-go soil measurements (Fig. 4). Optical sensing systems cover the ultraviolet (UV; 100–400 nm), visible (VIS; 400–700 nm), near infrared (NIR; 700–2500 nm), and mid-infrared (MIR; 2500–25,000 nm) portions of the electromagnetic spectrum. Typically, instruments used for soil measurements include their own light source (e.g., halogen light bulb or light-emitting diode). Photodiodes or array detectors are used to estimate the intensity of reflected light and relate this measure to the light reflected from a given set of standards. Both source and reflected light can be transmitted through the air, via fiber optics, or when feasible, through a contact window fabricated from highly resistive material, such as sapphire or quartz.

Measurements obtained using optical sensors can be related to a number of soil attributes, such as soil mineral composition, clay content, soil color, moisture, organic carbon, pH, and cation exchange capacity (CEC). Measurements can be viewed as direct, when relationships are based on a physical phenomenon that affects light reflectance in a specific part of the spectrum (e.g., predicting soil mineralogy or water content using characteristic water absorption bands); or indirect, when the relationships are deterministic for a finite domain and the combined effects of several soil attributes can be related to a given soil characteristic (e.g., predicting soil organic matter). Sensor calibration strategies



Fig. 3 GPR SIR 3000 system (Geophysical Survey Systems, Inc., Nashua, New Hampshire, U.S.A) integrated with Topcon HiPer XT Real Time Kinematic—GPS (RTK-GPS) receiver (Topcon Positioning Systems, Inc., Livermore, California, U.S.A).

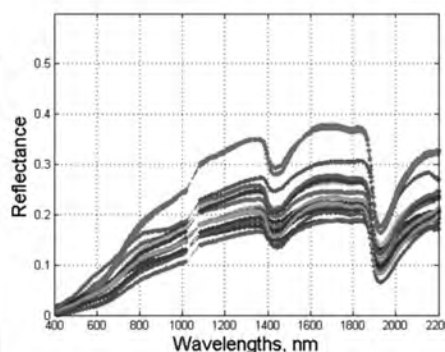


Fig. 4 Veris® P4000 (Veris Technologies, Inc.) probe equipped with two spectrometers covering VIS and NIR parts of the spectrum as well as electrical conductivity and insertion load sensors operated by McGill University (Ste-Anne-de-Bellevue, Quebec, Canada; left), and examples of soil spectra obtained by this instrument at different depths or locations (right).

ranged from a simple linear regression to multivariate methods, chemometrics, and data mining. Although some of these models may be applied to large geographic areas, most are linked to a specific range of soil and environments. UV radiation has also been used in combination with visible and infrared spectra to characterize iron oxides and organic matter.

γ -Ray Spectroscopy

γ rays contain a very large amount of energy and are the most penetrating radiation from natural or artificial sources. γ -ray spectrometers measure the distribution of the intensity of γ radiation versus the energy of each photon. Sensors may be either active or passive. Active γ -ray sensors use a radioactive source (e.g., ^{137}Cs) to emit photons of energy that can then be detected using a γ -ray spectrometer. Passive γ -ray sensors measure the energy of photons emitted from naturally occurring radioactive isotopes of the element from which they originate. As shown in Fig. 5, on-the-go measurement of soil elemental isotopes can be performed by installing a γ -ray sensor on a vehicle. Data interpretation may include analysis of measures related to the isotopes of potassium, thorium, and uranium, the total isotope count, or the entire energy spectrum. While shown to be a useful tool for predicting soil properties in different soil landscapes, a significant amount of preprocessing is often required to reveal relationships between the γ -ray spectra and the soil data. The γ -ray sensor signal can be

related to soil mineralogy, particle size distribution, and the effects of attenuating materials such as water and bulk density.

Inelastic neutron scattering (INS) spectroscopy relies on the detection of γ rays that are emitted following the capture and reemission of fast neutrons, as the sample is bombarded with neutrons from a pulsed neutron generator. The emitted γ rays are characteristic of the excited nuclide, and the γ -ray intensity is directly related to the elemental content of the sample. A thorough review of these methods and other proximal soil sensing methods can be found in the work of Viscarra Rossel et al.^[4]

APPLICATIONS TO SOIL SCIENCE

U.S. Department of Agriculture (USDA)/ Natural Resources Conservation Center (NRCS) Soil Surveys

Since the 1970s, GPR and EMI have been used by the National Cooperative Soil Survey (NCSS) as quality control tools to investigate, map, and interpret soils. GPR has been used principally, as shown in Fig. 6, to document the presence, depth, lateral extent, and variability of diagnostic subsurface horizons that are used to classify soils, and to name, characterize, and improve the interpretations of soil map units.^[5] GPR has also been used as a research tool to assess spatial and

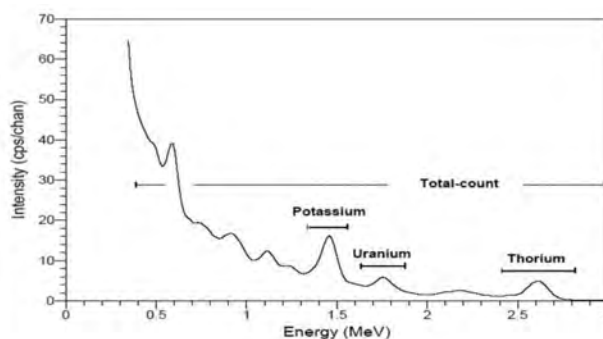


Fig. 5 The Mole γ -ray sensor (The Soil Company, Groningen, Netherlands) operated by Practical Precision, Inc. (Tavistock, Ontario, Canada; left), and an energy/intensity relationship used to define reported measurements (right).

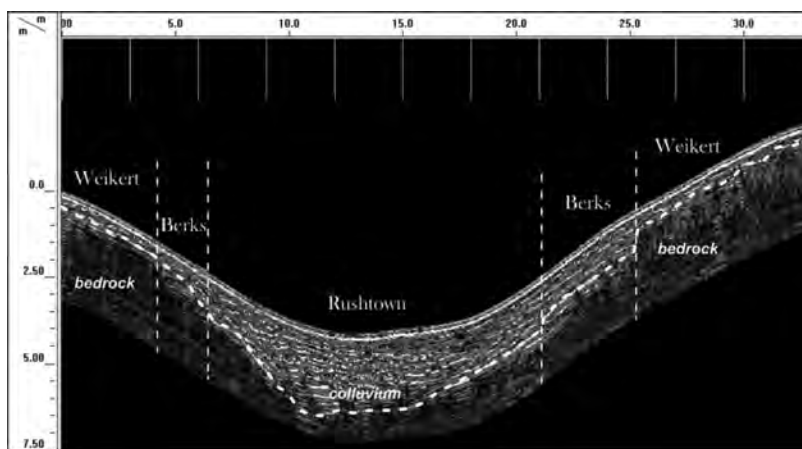


Fig. 6 Variations in the depth to shale bedrock are used to distinguish areas of shallow Weikert, moderately deep Berks and very deep Rushtown soils along this surface normalized GPR profile from the Northern Appalachian Ridges and Valley Region of central Pennsylvania.

temporal variations in soil properties. With the near completion of soil mapping in the United States, GPR is being increasingly used to support USDA conservation programs that help sustain agricultural productivity and environmental quality, address issues of soil health and soil functions, and improve the utility of soil survey data for modeling dynamic soil-hydrologic conditions and functionality at different scales.

The NCSS has used EMI to indirectly measure and map the spatial and temporal variability of soil properties at field scales. Initially used to assess soil salinity, EMI is used to map soil types; refine soil boundaries; identify contrasting soil components within soil map unit delineations; characterize soil water content and flow patterns; assess variations in soil texture, CEC, ionic composition, calcium carbonate content, organic carbon content, plant available nutrients, pH, and bulk density; and determine the depth to subsurface horizons, stratigraphic layers or bedrock, among other uses.^[6] As a tool for high-intensity soil surveys, EMI has been used to assist site-specific management, characterize

variability in soil physiochemical properties, and direct soil sampling.

Soil surveys require periodic maintenance and updating, as land use change and new demands require additional soil properties to be identified and interpreted. It is anticipated that, as soil surveys evolve, a greater understanding of the complexity and interaction of soil structures, processes, and functions will be needed at increasingly higher levels of resolution. In fulfilling this need, both EMI and GPR should find greater use in field-scaled research projects on representative soils.

Precision Farming

The agricultural production practice, called precision farming, employs spatially mapped field variables that provide assistance to producers for cropping decisions. For example, on-board computers can reference digital maps, directing automated field machinery to vary spatially the

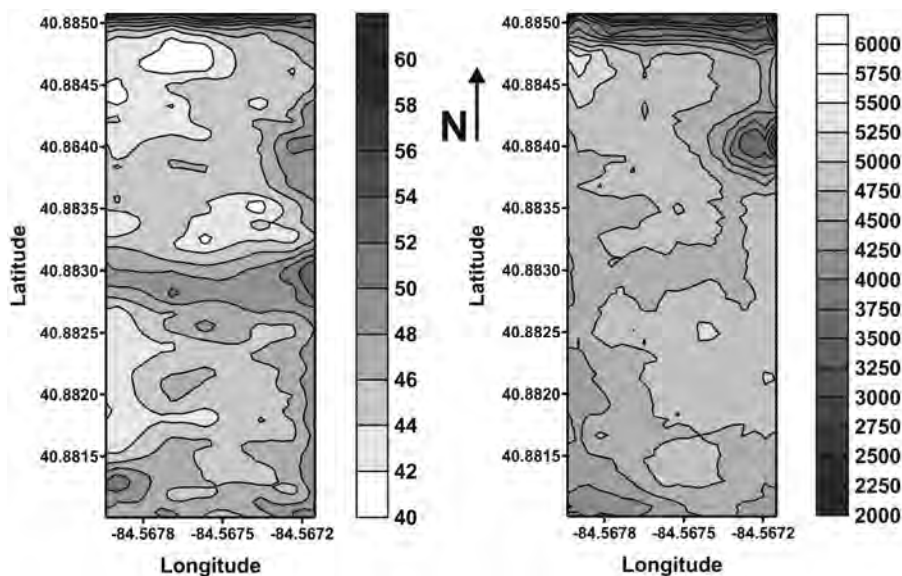


Fig. 7 Soil EC_a and crop yield comparison for a 3 hectare field in northwest Ohio. The EC_a map (left) was obtained with a Veris 3100 Soil EC Mapping System, and values are in millisiemens/meter. The soybean yield map (right) has values in kilograms/hectare. The spatial correlation coefficient (r) between EC_a and soybean yield is -0.51 .

application rates of fertilizers, pesticides, and seeds. As precision agriculture technologies evolve, higher spatial resolution surveys of the near subsurface will be required because some soil properties, such as soil moisture and organic matter, continually change.

GPSs were a major technology trigger for precision farming during the first decade of the 21st century. GPS is a common farm tool, and it provides accurate positioning for automated field-machine guidance and speed control and supplies the real-time positioning required for the precise placement of seeds, nutrients, and pesticides, along with the geospatial mapping of harvested yields. Global positioning integrated with massive, yet inexpensive, data storage and computational power allows for large acreages to be geospatially mapped by farmers at very high resolutions.

The relatively low-sampling data density of both on-the-go resistivity and EMI instruments, when coupled with GPS positioning, supplies an effective solution for spatially mapping any soil characteristic that can be delineated by relative resistivity/conductivity boundary shifts. For these sensors, the low-data density characteristics are their greatest advantage, as this easily allows for the generation of detailed spatial field maps for very large acreages with little visual interpretation requirements. This simplicity was a trigger for their rapid adoption by farmers and farm consultants. Field maps of EC_a from resistivity or EMI surveys, often show significant correlation with crop yield maps (Fig. 7). Consequently, EC_a maps can be employed to separate the field into different management zones and thereby increase crop production.

GPR is another geophysical tool showing great promise in precision agriculture. At the small plot scale, researchers have shown GPR's abilities to spatially map numerous soil properties essential to agricultural production. Almost any near-surface soil characteristic boundary that can be spatially delineated by sharp shifts in K can be mapped. However, the extreme amount of data generated and its variability when deployed over large acreages are too time-consuming and challenging for visual interpretations. The data computational requirements and survey data storage capacity of GPR surveys were once significant limiting factors, but they are no longer. However, efficient automated processing of the GPR data sets is lacking for large watershed-scale surveys having naturally occurring soils. In contrast with other geophysical sensors having lower data quantity requirements, this geophysical technology has not gained a following by producers using precision agriculture technologies.

Unmanned aerial vehicles (UAVs) coupled with remote geophysical sensors are projected by some experts to revolutionize precision agriculture. Rapid delivery of essential, time-critical information at very low cost is the advantage of the UAV over ground-

based geophysical surveys. Aerial field surveys will become automated, inexpensive, and on-demand. For example, miniaturized and lightweight EMI sensors are being introduced for UAV surveys over large areas. The low-flying UAV will closely examine crops, pastures, and timberlands. As they should be able to detect infestations at the very outset, farmers will be able to focus precisely on containment. Cattlemen will be able to remotely monitor livestock health and pinpoint both strays and their predators while they "ride herd" overhead using UAV thermal imaging. Georeferenced nutrient and yield maps will become commonplace, advancing precision agriculture using geophysical applications even further.

Soil Water Content Measurement

GPR can be used to estimate the VWC of soil or rock, as electromagnetic velocity (v) primarily depends on the volume of water in the pore space of earthen materials. Velocity is only slightly affected by mineralogy, grain size, or temperature, so measurements of v can be converted easily to VWC. Velocity is usually converted to K before calculating VWC; several petrophysical relationships based either on empirical data or volumetric mixing models are available in the literature to convert K to VWC.^[7]

The three most common techniques for estimating VWC from GPR data are reflected waves, groundwaves, and air-launched methods.^[8] For reflected waves, the travel time to

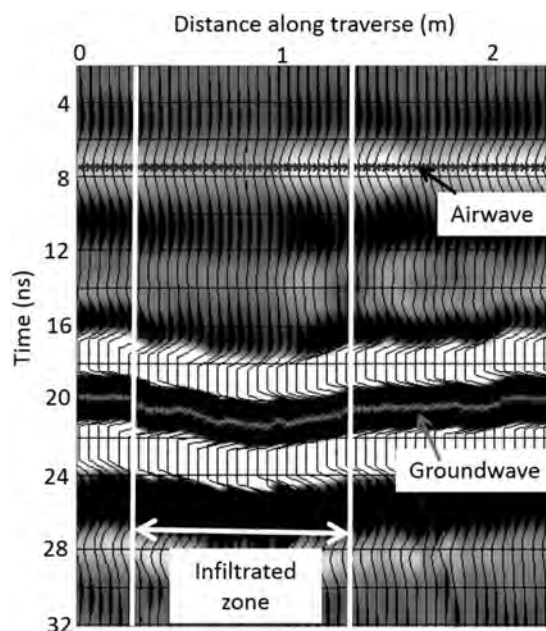


Fig. 8 GPR groundwave data acquired over an infiltrated area surrounded by dryer soil. Wetter soils have lower velocities and longer travel times, as observed over the infiltrated zone.

a subsurface interface can be measured while the GPR antennas are pulled in the common-offset mode along a traverse. If the depth to the interface is known, the travel time is used to estimate v , which can then be related to VWC. If the depth to the interface is not known, v can be obtained by performing a variable-offset survey, which provides information on both v and depth to an interface at one location. Reflections from a continuous subsurface interface can be used to measure VWC to greater depths (i.e., the root zone) and to understand VWC variability across a field. If a continuous interface at a known depth is not available, reflections from isolated subsurface objects or buried pipes that create a reflection hyperbola in the GPR record can be used to estimate v using reflection hyperbola analysis.

GPR groundwave techniques can be used to estimate VWC by measuring the travel time of this wave. Groundwaves travel in the shallow subsurface (0 cm to ~30 cm) directly between the transmitting and receiving antennas; by noting the antenna separation and the time needed for energy to travel between antennas, v can be calculated (Fig. 8). Groundwaves do not require a reflective interface, and so can be used in many near-surface soil environments to provide continuous VWC measurements.

Air-launched GPR techniques use the magnitude of the reflection from the ground surface to estimate K . Air-launched data can be acquired and processed quickly, but have a sampling depth of less than 5 cm, and the accuracy of the data greatly diminished by vegetation, uneven soil surfaces, and vertical variations in water content, this technique can be limited in its applications.

Soil Salinity Monitoring

Soil salinity refers to the concentration of dissolved inorganic solutes in the soil solution, consisting of four major cations (Na^+ , K^+ , Mg^{2+} , and Ca^{2+}) and five major anions (Cl^- , HCO_3^- , NO_3^- , SO_4^{2-} , and CO_3^{2-}). Soil salinity reduces plant growth and, in severe cases, causes crop failure by limiting plant water uptake due to an osmotic effect making it more difficult for the plant to extract water, by specific-ion toxicity, or by upsetting the nutritional balance of plants.

In the laboratory, soil salinity is most commonly determined from the measurement of the electrical conductivity of the solution extract of a saturated soil paste (EC_e), which is proportional to the concentration of ions in the solution. However, because salinity is a highly spatially and temporally variable soil property, the use of EC_e to measure salinity at field scales is impractical due to the need for hundreds or even thousands of soil samples. The use of EC_e to measure salinity at field scales is only practical when soil sampling is directed using correlated spatial information. Geospatial measurements of EC_a using geophysical techniques (i.e., electrical resistivity, EMI, or time domain reflectometry) are sources of spatial information used to direct soil sampling to characterize field-scale soil salinity variation.

EC_a measures the conductance through not only the soil solution but also through the solid soil particles and via exchangeable cations that exist at the solid–liquid interface of clay minerals. The interpretation of EC_a measurements is not trivial due to the complexity of current flow in the bulk soil through the three conductance pathways—liquid, solid, and solid–liquid interface. Because of the three pathways of

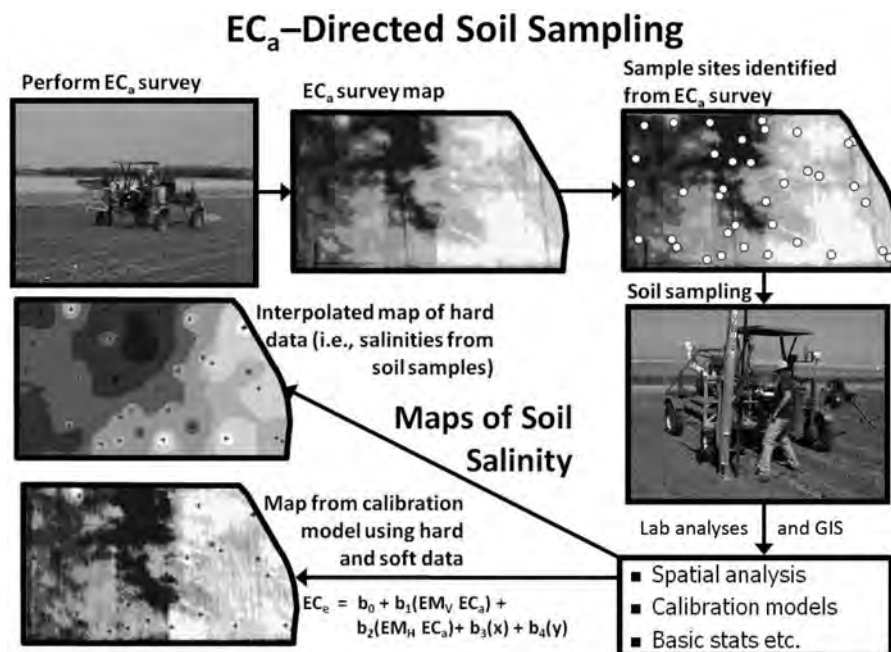


Fig. 9 Schematic of EC_a -directed soil sampling for mapping the spatial variability of soil salinity.

conductance, the EC_a measurement is influenced by several soil properties—soil salinity, texture, water content, bulk density, and temperature. Detailed protocols for conducting an EC_a survey to characterize the spatial variability of soil salinity with EC_a -directed soil sampling are provided by Corwin and Lesch.^[9,10] A schematic of EC_a -directed soil sampling is shown in Fig. 9.

CONCLUSION

Near-surface geophysical methods have become increasingly important tools for soil investigations in agricultural settings. The methods predominantly employed are resistivity, EMI, and GPR; however, optical reflectance and γ -ray spectroscopy are beginning to find more widespread utilization. Furthermore, research indicates that other geophysical methods, such as cosmic-ray neutron probes, seismic, self-potential, magnetometry, nuclear magnetic resonance, etc., all exhibit potential for measurement of soil properties or conditions. The application to soil surveys, precision farming, soil water content measurement, and soil salinity monitoring has been described in this entry, but geophysical methods have also been used for soil investigations in forested areas, confined animal feeding facilities, and golf courses, with new applications continuing to evolve. Development of multisensor platforms and sensors mounted on UAVs will dramatically improve geophysical soil investigation capabilities with respect to field accessibility and data interpretation. Consequently, the future of near-surface geophysics in soil science appears very promising.

REFERENCES

1. Allred, B.J.; Ehsani, M.R.; Daniels, J.J. General considerations for geophysical methods applied to agriculture. In *Handbook of Agricultural Geophysics*; Allred, B.J., Daniels, J.J., Ehsani, M.R., Eds.; CRC Press LLC: Boca Raton, 2008; 3–16.
2. McNeill, J.D. *Electromagnetic Terrain Conductivity Measurement at Low Induction Numbers – Technical Note TN-6*; Geonics Limited: Mississauga, 1980; 15 pp.
3. Conyers, L.B. *Ground-Penetrating Radar*; AltaMira Press: Walnut Creek, 2004; 201 pp.
4. Viscarra Rossel, R.A.; Adamchuk, V.; Sudduth, K.; McKenzie, N.; Lobsey, C. Proximal soil sensing: An effective approach for soil measurements in space and time. In *Advances in Agronomy 113*; Sparks, D.L., Ed.; Academic Press: San Diego, 2011; 237–282.
5. Doolittle, J.A.; Butnor, J.R. Soils, peatlands, and biomonitoring. In *Ground Penetrating Radar: Theory and Applications*; Jol, H.M., Ed.; Elsevier: Amsterdam, 2008; 179–202.
6. Doolittle, J.A.; Brevik, E.C. The use of electromagnetic induction techniques in soils studies. *Geoderma* **2014**, 223–225, 33–45.
7. Jol, H.M. *Ground Penetrating Radar: Theory and Applications*; Elsevier Science: Amsterdam, 2009; 524 pp.
8. Huisman, J.A.; Hubbard, S.S.; Redman, J.D.; Annan, A.P. Measuring soil water content with ground-penetrating radar: A review. *Vadose Zone J.* **2003**, 2, 476–491.
9. Corwin, D.L.; Lesch, S.M. Characterizing soil spatial variability with apparent soil electrical conductivity: I. Survey protocols. *Comput. Electron. Agr.* **2005**, 46 (1–3), 103–133.
10. Corwin, D.L.; Lesch, S.M. Protocols and guidelines for field-scale measurement of soil salinity distribution with EC_a -directed soil sampling. *J. Environ. Eng. Geoph.* **2013**, 18 (1), 1–25.

Geostatistics

Anja Gassner

Institute of Science and Technology, Malaysia Sabah University, Kota Kinabalu, Malaysia

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

Scientific knowledge is based on the interaction between observations and the hypothesis that corresponds to those observations. Soil science is increasingly asked to answer questions like how do soils function in environment and how will they function and respond to a changing environment. The classical description of soil does not provide the answer. Geostatistics offers a range of tools that allow the modeling of soil functions and the interpretations of the outcome. For a range of applications such as optimal interpolation of soil properties, optimizing sampling schemes, and uncertainty analysis, to name only a few, geostatistics is essential to obtain the maximum information contained in observations and thus should be an integrated part of modern soil science.

INTRODUCTION

In the last few decades, research in soil science saw a drastic shift from an overriding importance of purely empirical examination and description of soils to a balanced approach including theory development and modeling. After the work of Jenny,^[1] soil properties are no longer seen as randomly distributed events, but rather as the outcome of an environmental process. Thus, the spatial distribution of most soil properties consists of a structural, presented by the mean or a constant trend, a spatial correlated, observations closer together are more alike than observations further apart and a random component. Geostatistics, introduced into soil science since 1980s,^[2] enables us to analyze these structures and allows us not only to predict soil classes, characteristics, and behavior, but also to assess the uncertainty of these predictions. An understanding of the spatial structure of the landscape and how it affects the correlation between soil properties is the prerequisite to develop economically feasible sampling strategies and to utilize auxiliary data to model the attribute of interest.

BASIC CONCEPTS

Theory of the Regionalized Variable

The fundamental difference between classical statistics and geostatistics is that the latter is based on the theory of regionalized variables.^[3,4] Whereas in classical statistics,

every variable is purely *random*, meaning that it has a variety of values in accordance with a particular probability distribution,^[4] in geostatistics the variable is seen as distributed in space and/or time and thus “regionalized.” So is the soil phosphorus (P) concentration [regionalized variable, $z(x)$] depending on the location $x = (x_1, x_2)^T = (\text{northing, easting})^T$. In this sense, geostatistics pays a tribute to the intuitive sense that observations (P concentration in the soil) that are closer together are more related (more similar) than observations that are further apart. The underlying assumption is that the phenomenon studied is one possible outcome of an unknown environmental process (random function). It is the modeling of the random function, which is the core of geostatistics.

Modeling of the Random Function

The random function can be modeled by recognizing explicitly the spatial correlation between the data. The classical geostatistical approach is to use the semivariance:

$$\gamma(h) = \frac{1}{2} E \left[\left\{ Z(x) - Z(x+h) \right\}^2 \right] \quad (1)$$

where E signifies expectation, and h is the vector, the so-called “lag,” separating the two points x and $x+h$.

The semivariance, which is only dependent on the distance h and not on the positions x and $x+h$, is a measure of the average degree of dissimilarity between two observations, e.g., the P concentration at point x and the P concentration at point $x+h$.

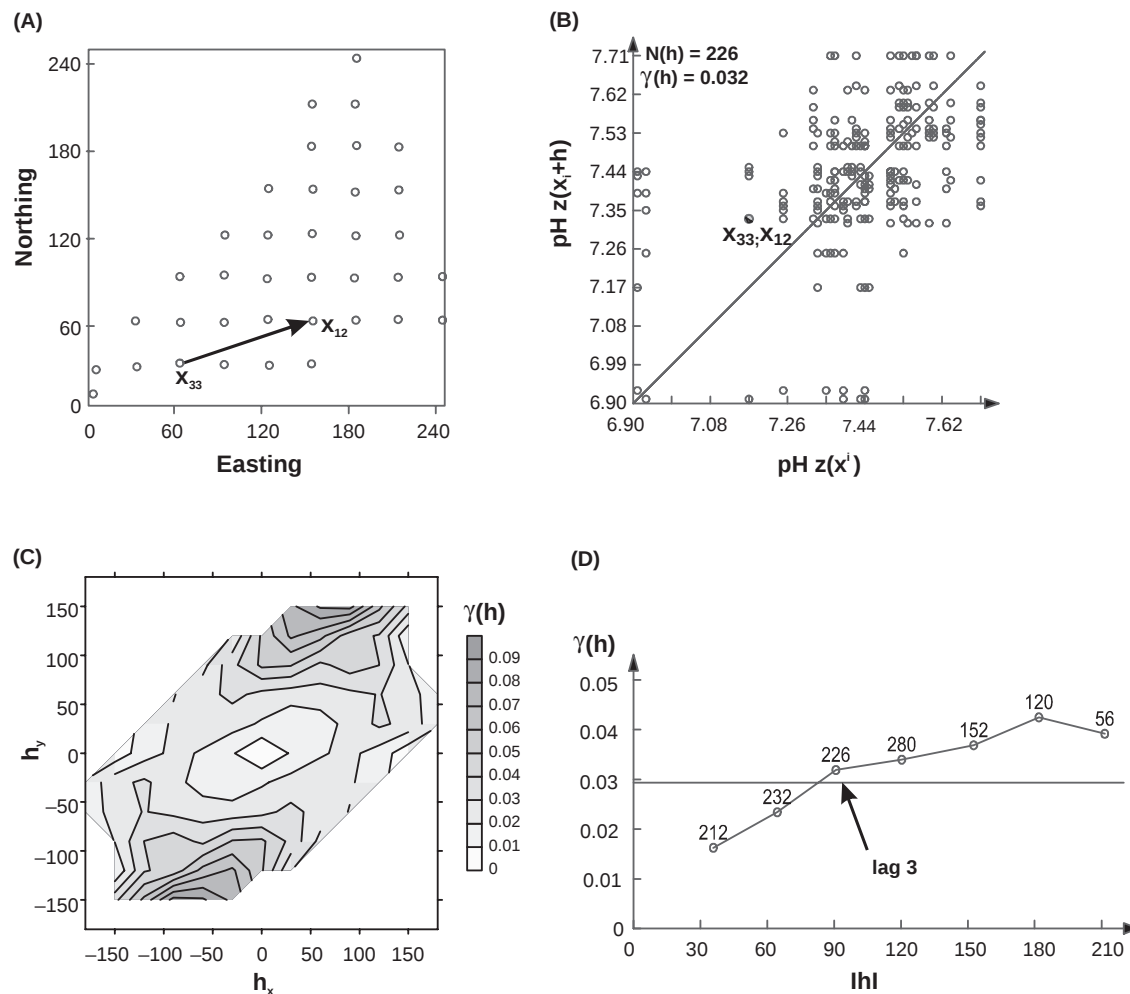


Fig. 1 Various views of spatial continuity and their relationships. The “sample map” (A) can be used to identify errors in the coordinates and data clustering. An “h-scatterplot” (B) is the bivariate equivalent of the histogram. The value of one variable at position x is plotted against the value of the same variable at position $x+h$. “The variogram map” (C) is a 2-D plot of the experimental semivariogram values in the system of coordinates (h_x, h_y) . When the variation is isotropic, the increase is fairly similar in every direction; hence, the map shows concentric contour lines. Conversely, geometric anisotropy appears as elliptical contour lines whose major axis indicates the direction of maximum continuity. “The experimental semivariogram” (D) that forms the basis for the modeling of the random function.

Source: From Pannatier.^[5]

By plotting the semivariance against the distance h , one obtains the experimental semivariogram, often just referred to as the variogram (Fig. 1). Because the underlying processes that govern the spatial distribution of data often have preferred orientations, data may change more quickly in one direction than in another. Thus, the variogram is a 3-D function, consisting of two independent variables, namely the direction and the distance h , and one dependent variable, the observation $z(x_i)$. Distributions that have preferred orientations are called anisotropic, whereas otherwise they are referred to as isotropic. A useful tool to assess the orientation of the soil property of interest is the so-called “variogram surface.”

To detect the random function that caused the spatial dependency of the observations, it is necessary to fit a model through the experimental variogram. There are

certain restrictions on the functions that can be used to model the experimental variogram.^[6] The most common permissible models are illustrated in Fig. 2. The spherical model is probably the most commonly used model. It has a simple polynomial expression, and the shape matches well with what is often observed: an almost linear increase in the semivariance up to a certain distance and then stabilizing. The exponential model rises more rapidly initially and reaches the so-called “sill” asymptotically. The Gaussian model represents an extremely continuous phenomenon, such as soil water content. In contrast to the above-described models, the power model, with the linear model being a special case, is not bounded^[6] and indicates an underlying trend or drift in the data.

The nugget variance C_0 represents the random variation of the data set that arises first from measurement error and

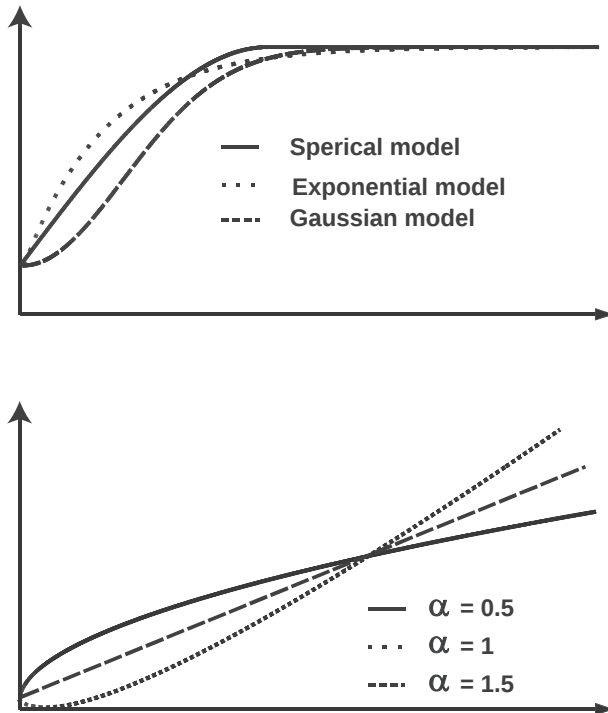


Fig. 2 Permissible variogram models. Bounded models with the same practical range (top graph) and power models for different values of the parameter α (bottom graph).

Source: From Goovaerts.^[6]

second from a spatial variability at distances smaller than the shortest sampling interval. The structural semivariance C_1 represents the component of total variation that is contributed by systematic sources. The sill is the plateau every bounded model will reach. The sill gives an estimation of the total variance (random + structural) of the data.^[7] The distance at which the model reaches the sill is called the “range.” All data within this range are assumed to be spatially autocorrelated (Fig. 3).

A quantification of the contribution of the random variation to the semivariance observed in a data set can be estimated from the ratio of nugget semivariance to sill semivariance:^[8]

$$NR = \frac{C_0}{C_0 + C_1} \times 100 \quad (2)$$

This ratio has been used to describe the strength of the spatial dependence within a field attribute^[9] where:

NR ≤ 25%: indicates a strong spatial dependency

25 < NR < 75%: indicates a moderate spatial dependency

NR ≥ 75%: indicates a weak spatial dependency

(3)

The variogram parameters, nugget, sill, and range form the basis for all further geostatistical processes, such as kriging

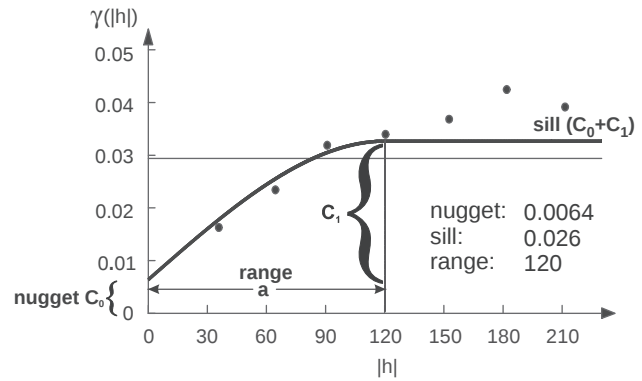


Fig. 3 Spherical model with nugget effect fitted to the experimental variogram for soil pH in Warburg, Germany (E10°54', N52°10').

or simulations. Thus, the correct modeling and interpretation of the variogram are extremely important. As an excellent guide to variogram modeling, the reader is referred to Gringarten and Deutsch.^[10] Although the calculation of experimental variograms and subsequent fitting of appropriate variogram models is the most common method to model spatial patterns, there is no general agreement on how to evaluate the goodness of the chosen model. There is a tendency within the geostatistic community to justify the choice of a particular model using statistical criteria such as the weighted least squares criteria,^[11] cross-validation,^[12,13] and the indicative goodness of fit.^[5] The disadvantage of these statistical criteria is that they purely rely on the experimental variogram without taking any ancillary information into account. As with any model, the outcome of the variogram model has to be an approximation toward the truth,^[14] and thus, it has to reflect physical knowledge of the area and of the phenomenon under study. Ancillary data that can be utilized to improve the model are topography, distribution of different soil texture classes, and geological formations as well as management practices.^[15]

APPLICATION IN SOIL SCIENCE

Optimal Interpolation and Isarithmic Mapping of Soil Properties

Where measured data are sparse, as they often happened to be, it is necessary to predict observations at unsampled locations. Several interpolation techniques have been proposed, but kriging is the only technique that not only minimizes the error of interpolation but also evaluates it.^[2] The kriging principle consists of determining the weights of the variable values in neighboring points to estimate the variable value in the target point or spatial domain. The weight of each point is determined from the variogram. The weights are selected to obtain an unbiased estimate

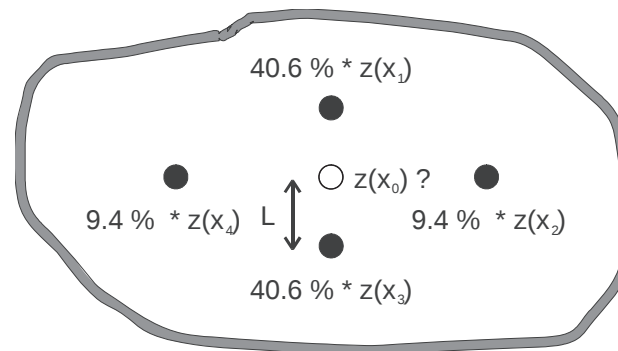


Fig. 4 With a spherical model of range $a/L=2$, the remote samples get lower weights. Z at the unsample location x_0 will be estimated according to $\hat{Z}_0 = \sum_{i=1}^N w_i Z_i \Leftrightarrow \hat{Z}_0 = 40.6\% \times (z(x_1) + z(x_3)) + 9.4\% \times (z(x_2) + z(x_4))$.

Source: From Wackernagel.^[16]

(all weights sum up to 1) of the target value and the minimum variance of the estimate:

$$\sigma_E^2 = E\left\{\left[Z_0 - \hat{Z}_0\right]^2\right\} \quad (4)$$

The closer the specific testing site to the point or domain for which the parameter should be determined, the more substantial is its contribution to the target value (Fig. 4).

Optimizing Sampling Schemes

For most soil surveys, the limiting factors for obtaining sufficient data are the costs and manpower involved in sampling and subsequent analysis of samples. Prior knowledge of the within-field variability of the soil property of interest is essential to optimize full-scale sampling. Several papers suggested determining the variogram for the property in a prior reconnaissance stage of a survey. Once the maximum permissible variance for the survey has been determined, the optimal sampling interval is easily obtained from the variogram.^[17,18] As the amount of samples needed to estimate a variogram in advance of full-scale sampling can be a “serious obstacle,” McBratney and Pringle^[19] proposed the use of proportional variograms, whereby the variogram can be predicted from a mean value.

Uncertainty Analysis

The quality of spatial soil data modeling is subjected to two kinds of uncertainty. The first type, local uncertainty,^[6] addresses the quantification of the precision of interpolated estimates, whereas the second type, spatial uncertainty,^[6] is a measure of the joint uncertainty about the outcome of a soil property analysis at several locations taken together. Local uncertainty is often estimated based on the kriging variance.^[6,20,21] However, it can only be used as a measure of reliability of the kriging estimate whether the variance of the errors is independent of the actual data values and depends only on the data configuration, a situation referred

to as “homoscedasticity.” As this requirement is hardly ever met in soil data, a more popular approach is to use indicator kriging.^[22–24] To give credit to the inherent local variability of the data and the spatial uncertainty, a process known as stochastic simulation has become more popular than kriging.^[25] The principle is to perform variogram modeling and subsequent kriging of the data at all grid locations to obtain a “base” map. In simulation, the objective is to generate multiple realizations of this base map each of which would honor the actual data (i.e., a conditional map), and approximately the same variogram and distribution. The key is to generate a high number of realizations (repetitions) such that the simulation results are truly representative of the stochastic nature of spatial heterogeneity and uncertainty (refer Heuvelink and Burrough^[26] also).

CONCLUSION

Scientific knowledge is based on the interaction between observations and the hypothesis that corresponds to those observations. Soil science is increasingly asked to answer questions like how do soils function in environment and how will they function and respond to a changing environment. The classical description of soil does not provide the answer. Geostatistics offers a range of tools that allow the modeling of soil functions and the interpretations of the outcome. For a range of applications such as optimal interpolation of soil properties, optimizing sampling schemes, and uncertainty analysis, to name only a few, geostatistics is essential to obtain the maximum information contained in observations and thus should be an integrated part of modern soil science.

REFERENCES

1. Jenny, H. *Factors of Soil Formation: A System of Quantitative Pedology*; Dover Publications: New York, 1941.

2. Burgess, T.; Webster, R. Optimal interpolation and isarithmic mapping of soil properties. I. The semivariogram and punctual kriging. *J. Soil Sci.* **1980**, *31*, 315–331.
3. Matheron, G. The theory of regionalized variables and its applications. *Les Cahiers du Centre Morphologie Mathématique de Fontainebleau* **1971**, *5*, 1–211.
4. Journel, A.; Huijbregts, G. *Mining Geostatistics*; Academic Press: New York, 1997.
5. Pannatier, Y. *VARIOWIN: Software for Spatial Data Analysis in 2D*; Springer-Verlag: New York, 1996.
6. Goovaerts, P. *Geostatistics for Natural Resources Evaluation*; Oxford University Press: New York, 1997.
7. Barnes, R.J. The variogram sill and the sample variance. *Math. Geol.* **1991**, *23*, 673–678.
8. Trangmar, B.B.; Yost, R.S.; Uehara, G. Application of geostatistics to spatial studies of soil properties. *Adv. Agron.* **1985**, *38*, 45–94.
9. Cambardella, C.A.; Moorman, T.B.; Ovak, J.M.; Parkin, T.B.; Karlen, D.L.; Turco, R.F.; Konopka, A.E. Field-scale variability of soil properties in central Iowa soils. *Soil Sci. Soc. Am. J.* **1994**, *58*, 1501–1511.
10. Gringarten, E.; Deutsch, C.V. Variogram Interpretation and Modeling. *Math. Geol.* **2001**, *33* (4), 507–534.
11. Cressie, N.A. Fitting variogram models by weighted least squares. *Math. Geol.* **1985**, *17* (5), 563–86.
12. Davis, B.M. Use and abuse of cross-validation in geostatistics. *Math. Geol.* **1987**, *19* (3), 241–248.
13. Journel, A.G. *Fundamentals of Geostatistics in Five Lessons*; American Geophysical Union: Washington, 1989.
14. Hoosbeek, M.R.; Bryant, R.B. Towards the quantitative modelling of pedogenesis—A review. *Geoderma* **1992**, *55*, 183–210.
15. Gassner, A. *Factors Controlling the Spatial Speciation of Phosphorus in Agricultural Soils*; Landbauforschung Völk-enrode SH 244: Braunschweig, 2003.
16. Wackernagel, H. *Multivariate Geostatistics*; Springer-Verlag: Berlin, 1998.
17. Burgess, T.M.; Webster, R.; McBratney, A. Optimal interpolation and isarithmic mapping of soil properties. IV. Sampling strategy. *J. Soil Sci.* **1981**, *32*, 643–659.
18. Webster, R.; Burgess, T.M. Sampling and bulking strategies for estimating soil properties in small regions. *J. Soil Sci.* **1984**, *35*, 127–140.
19. McBratney, A.B.; Pringle, M.J. Estimating average and proportional variograms of soil properties and their potential use in precision agriculture. *Precis. Agric.* **1999**, *1*, 219–236.
20. Journel, A.G. Geostatistics: Roadblocks and challenges. In *Geostatistics Troia '92*; Soares, A., Ed.; Kluwer Academic Publishers: Dordrecht, 1993; 213–224.
21. Armstrong, M. Is research in mining geostats as dead as a dodo? In *Geostatistics for The Next Century*; Dimitrakopoulos, R., Ed.; Kluwer Academic Publishers: Dordrecht, 1994; 303–312.
22. Isaaks, E.H.; Srivastava, R.M. *An Introduction to Applied Geostatistics*; Oxford University Press: New York, 1989; 561.
23. Bierkens, M.F.P.; Burrough, P.A. The indicator approach to categorical data. Parts I and II. *J. Soil Sci.* **1993**, *44*, 361–381.
24. Goovaerts, P. Comparative performance of indicator algorithm for modeling conditional probability distribution functions. *Math. Geol.* **1994**, *26* (3), 389–412.
25. Heuvelink, G.B.M.; Webster, R. Modelling soil variation: Past, present, and future. *Geoderma* **2001**, *100*, 269–301.
26. Heuvelink, G.B.M.; Burrough, P.A. Developments in statistical approaches to spatial uncertainty and its propagation. *Int. J. GIS* **2002**, *16*, 111–113.

GIS Integration: Remote Sensing

Egide Nizeyimana

*Department of Agronomy and Environmental Resources Research Institute,
Pennsylvania State University, University Park, Pennsylvania, U.S.A.*

Abstract

Remote sensing (RS) is often viewed as technology for data acquisition on the earth's surface. This entry provides ways in which RS, including aerial photography, is integrated to a geographic information system (GIS) for environmental assessments needed in land use planning and management. RS is compatible with GIS because the data generated are geo-referenced to a known coordinate system and are spatial in nature. These data are generally stored and queried as grid, a data structure supported by the most popular commercial GIS software. Later, they are transferred to a GIS platform after processing for integration with other spatial land attributes from spatial data sources such as soil databases and digital elevation models. This RS/GIS integration and associated data sets can also be part of GIS-based systems such as GIS/model interfaces and spatial decision support systems.

COMPATIBILITY ISSUES BETWEEN REMOTE SENSING (RS) AND GIS

A geographic information system (GIS) can be defined as a set of computer tools for capturing, storing, analyzing, and displaying spatially referenced data. The data within a GIS consist of two elements: spatial entities represented by points (e.g., well locations), lines (e.g., streams, road networks), and polygons (e.g., soil delineations) and attribute data or information that describes characteristics of the spatial features. The spatial entity is referenced to a geographic coordinate system and is stored in either a vector or raster model. GIS is primarily a platform that integrates spatial information from variable data sources and provides tools to overlay and analyze it.

RS is compatible with GIS because the information acquired by imaging sensors carried aboard satellites and airplanes is geographic in nature, referenced to a known coordinate system, and in a grid layout, a raster data model commonly found in most popular GIS software packages. In a raster model environment, the analysis is performed pixel by pixel. The commercial GIS-based software packages have vector and raster capability analyses and allow data conversion from one data model to the other. Non-imaging systems are also important to RS/GIS integration because they provide point data, which at geo-referenced locations are often combined with other environmental data for site-specific assessments.

LINKING RS AND GIS

RS as a Source of Spatial Data

RS is one of the most important sources of land use/cover information used in GIS analyses. It provides information

on the location and spatial and temporal distribution of land cover on the earth's surface. Land use/cover distributions may be derived from aerial photographs after they are corrected for relief displacement and distortions caused by camera angle and registered to a coordinate system. Delineations of different land use/cover types on photographs are made from visual interpretations of characteristics (e.g., tone, texture, and color) aided by optical devices. Digital maps of these classes are created by digitizing and processing boundaries between land uses or by scanning photographs covering the area of interest and screen digitizing their boundaries.

Land use distributions on the earth's surface may also be obtained using RS imaging systems. In this case, RS provides opportunities in the area of land use planning which would not be otherwise available. While aerial photographs are effective and appropriate for analyses of small areas, RS offers tremendous advantages when planning for large areas such as river basins and regions. The fact that the land surface is observed from reflected/emitted energy for large areas and over a wide range of wavelengths allows for easy differentiation of land uses (e.g., wetlands and degraded landscapes).^[1] Earth observation satellites also offer a repetitive coverage of the land, thus providing the possibility of monitoring land use pattern changes over time. Gathering information on land uses at several time intervals is particularly important in the monitoring and land evaluation stages of the land use planning process. Factors such as plant stress, crop growth and yields that serve as measures of agricultural productivity can be rapidly estimated following digital processing and analysis of RS imagery. RS has been used in many instances to monitor land surface conditions such as soil degradation and soil salinity.^[2] It is true that RS cannot replace field mapping of land use/cover. However, RS supplements provide

information that would not otherwise be available for land use planners and managers. There is no doubt that the development of sensors of high spatial resolution (1 m and higher) in orbits would increase the use of RS in land use planning and management.

As indicated before, RS and GIS technologies are highly compatible primarily because of the nature of RS as a source of spatial land use/land cover used in various environmental applications. Remote sensed imagery is a grid cell layer, a data format easily handled by GISs. In the RS/GIS integration, GIS appears as a platform that stores and integrates spatial data from different sources including RS and output information needed for environmental analyses depending on the type of analysis sought. Land use/cover distributions derived from RS are input to GIS, GIS-based systems (spatial decision support systems), and models (e.g., hydrologic/water quality, crop yields, primary productivity) commonly used in land evaluation for land use planning and management. GIS has been proven to be a valuable tool in integrating RS-derived data with climate and land surface parameters (soils, terrain, etc.) to generate digital maps of ecosystem productivity or vulnerability to environmental factors.^[3] The parameterization of the Terrestrial Ecology Model and a forest ecosystem model using GIS- and RS-derived data at regional scales has also been accomplished.^[4,5] The RS/GIS approach has been adopted by many state and federal agencies involved in environmental assessments. For example, in the GAP Analysis Program, the U.S. Geological Survey (USGS) and collaborative institutions map and/or model potential natural habitats of native vertebrate species from remotely sensed data across the country and use GIS map overlay tools to determine the degree of richness of habitats in these species.^[6]

Finally, GIS and RS applications have promoted the development of other high-resolution spatial technologies that enhance the land use assessment and planning. Some of these are the global positioning system (GPS) and digital orthophoto quadrangles (DOQ). GPS allows the user to record accurately and rapidly geographic coordinates of any location in the field with precisions ranging from several meters to a centimeter. Soil, terrain, and land use attributes observed or measured can then be input to a GIS along with their precise locations and extent. GPS coupled with a GIS can improve the accuracy of land quality mapping by increasing the spatial variability of soil and landscape attributes. DOQs, on the other hand, are digital images of aerial photographs that were corrected to remove relief displacement and distortion caused by the camera angle. The USGS distributes single-band, 256-scale, gray-scaled DOQs at 1-m grid resolution. Although DOQs have not been used extensively in land use planning, their high resolution and photograph-like characteristics make them a potential source of data in this area. Furthermore, repetitive DOQs acquired at different times can be used to monitor the magnitude of changes in land use over time. The relationship between GIS, RS, and other data sources is shown in Fig. 1.

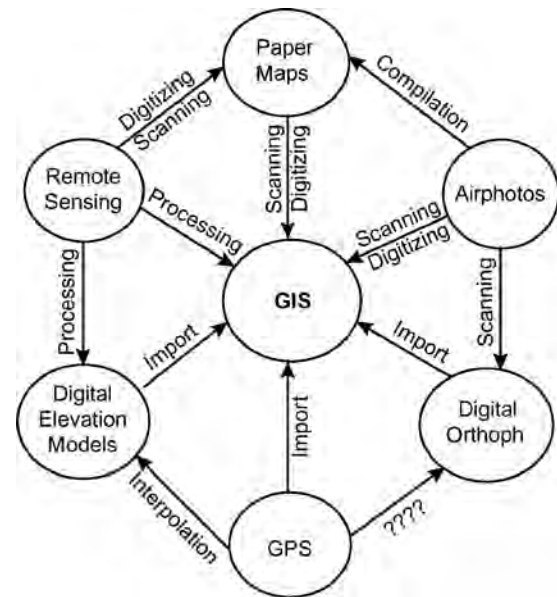


Fig. 1 Relationships between RS and GIS.

Source: Adapted from Nizeyimana & Petersen.^[7]

Applications of RS/GIS in Soil Science

The need for geospatial data for use in various spatial database development and analyses in industry, government, and universities have increased the demand for remotely sensed data. In the area of soil science, the RS technology is used to acquire data and digitally process and analyze it. Subsequent analyses may involve error assessment, and data conversion from raster to vector format and vice versa before the data is merged with other data sets in GIS for specific analyses. Table 1 summarizes RS techniques used in various aspects of soil science and potential limitations for use in GIS and advantages of each. These are laboratory approaches, field methods, and aircraft-/satellite-based methods. The latter involves interpretation and analyses of digital images, color composites, or radiances.

RS/GIS Integration for Site-Specific Farming (SSF)

The goal of SSF or precision farming is to optimize the profitability of a farm practicing variable management according to soil conditions found at each site. The concept is based primarily on the fact that soil chemical and physical properties that affect crop production (e.g., pH, nutrients, available water, and impeding layers) vary spatially across agricultural landscapes. Management practices such as fertilizer and pesticide applications, irrigation water, and crop varieties should, therefore, be prescribed according to this soil variability. The profitability associated with variable rate applications of SSTs should avoid or reduce waste and the risk for environmental pollution because these

Table 1 Documented uses of RS in various areas of soil science.

Application	Advantages	Potential limitations	References
Laboratory methods	Provides accurate measurements of reflectance values	Provide point data rather than areal extent of soil properties	[8]
Field methods	Easily related to on-site conditions	Provide limited data coverage; measurements are affected by soil conditions (soil roughness and moisture), sun angle, etc.	[9]
Airborn-/satellite-based methods	Provides areal extent and temporal coverage of soil properties	Radiance measurements affected by atmospheric and soil surface conditions	(–)
a) Interpretation of digital images	Good results when data are well analyzed and interpreted	Image analysis can be costly; requires experienced technician for good results; field verification or prior knowledge of area required for best results	[10]
b) Interpretation of color composites	Easy to use and rapid interpretations	Intensive field verification is needed for good results due to the fact that interpretation is based on differences in tone and physical characteristics of objects; requires experienced interpreter for good results	[11]
c) Interpretation of radiances	Relatively easy, cheap, quantitative; data can be normalized to remove environmental effects	Results often unreliable	[12]

Source: Adapted from Nizeyimana & Petersen.^[13]

agrochemicals are applied to the field only in amounts needed for optimal crop growth.

In an SST, the real-time detection of continuous soil variable is made possible by mobile devices including sensors and differential GPS units, while agrochemicals are applied using variable rate field applicators mounted on farm equipments. These sensors are based on the same principles as RS and have been developed to detect, directly or indirectly, soil properties such as soil moisture, soil texture, nitrates, etc. The data acquired, along with their respective locations, are often integrated with spatial information from other sources such as soil databases, climate, landscape, and satellite/airborne RS to delineate meaningful management zones within the farm.

REFERENCES

- Petersen, G.W.; Nizeyimana, E.; Evans, B.M. Applications of geographic information systems in soil degradation assessments. In *Methods for Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentine, C., Rose, B.A., Eds.; Advances in Soil Science; CRC Press: Bacon Raton, 1997; 377–391.
- Raina, P.; Joshi, D.C.; Kolarkar, A.S. Mapping of soil degradation by remote sensing on Alluvial plan, Rajasthan, India. *Arid Soil Res. Rehab.* **1993**, *7*, 145–161.
- Parrish, D.A.; Townsend, L.; Saunders, J.; Carney, G.; Langston, C. USEPA. In *Region 6 Comparative Risk Project. Evaluating Ecological Risk*. EPA Unpublished Report, 1993.
- Pan, Y.; McGuire, A.D.; Kicklighter, D.W.; Melillo, J.M. The importance of climate and soils for estimates of net primary production: A sensitivity analysis with the terrestrial ecosystem model. *Glob. Change Biol.* **1996**, *2*, 5–23.
- Lathrop, R.G., Jr.; Abler, J.D., Jr.; Bognar, J.A., Jr. Spatial variability of digital soil maps and its impact on regional ecosystem modeling. *Ecol. Model.* **1995**, *82*, 1–10.
- Scott, J.M.; Davis, F.; Csuti, B.; Noss, R.; Butterfield, B.; Groves, C.; Anderson, H.; Caicco, S.; D'Erchia, F.; Edwards, T.C., Jr.; Ulliman, J.; Wright, R.G. GAP Analysis: A geographic approach to protection of biological diversity. *Wildlife Monogr.* **1993**, *123*, 1–41.
- Nizeyimana, E.; Petersen, G.W. Land use planning and environmental impact assessment using GIS. In *Environmental Modeling Using GIS and Remote Sensing*; Skidmore, A., Ed.; Taylor&Francis: London, 2002; 227–251.
- Stoner, E.R.; Baumgardner, M.F. Characteristic variations in reflectance surface of soils. *Soil Sci. Soc. Am. J.* **1981**, *45*, 1161–1165.
- Gausman, H.W.; Leamer, R.W.; Noriega, J.R.; Rodriguez, R.R.; Wiegand, C.L. Field-measured spectrometric reflectance of disked and non-disked soil with and without wheat straw. *Soil Sci. Soc. Am. J.* **1977**, *41*, 493–496.
- Connors, K.F.; Gardner, T.W.; Petersen, G.W. Digital analysis of the hydrologic components of watersheds using simulated SPOT imagery. In *Proceedings of Workshop on Hydrologic Applications of Space Technology*; IAHS: Cocoa Beach, 1985; Vol. 160, 355–365.
- Bocco, G.; Palacio, J.; Valenzuela, C.R. Gully erosion modeling using GIS and geomorphologic knowledge. *ITC J.* **1990**, *3*, 253–261.
- Pickup, G.; Nelson, D.J. Use of landsat radiance parameters to distinguish soil erosion, stability, and deposition in Arid Central Australia. *Remote Sens. Environ.* **1984**, *16*, 195–209.
- Nizeyimana, E.; Petersen, G.W. Remote sensing applications to soil degradation assessments. In *Methods for Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentine, C., Rose, B.A., Eds.; CRC Press: Bacon Raton, 1997; 393–405.

Global Resources

Paul Reich

*U.S. Department of Agriculture—Natural Resources Conservation Service
(USDA-NRCS), Washington, District of Columbia, U.S.A.*

Hari Eswaran

*U.S. Department of Agriculture—Natural Resources Conservation Service
(USDA-NRCS) (Retired), Washington, District of Columbia, U.S.A.*

Abstract

For most purposes, the term biome is used to identify the natural habitat conditions around the world. Depending on the purpose, the global ecosystem is divided into units each characterized by a specific combination of climatic factors. Two major determinants of biome type are precipitation (total and its distribution) and air temperature. These two elements of climate have been commonly used to define the major biomes of the world. A third variable that affects the habitat type is the soil. This entry presents a general overview of the soils characterizing the major biomes of the world.

INTRODUCTION

A biome is defined as “a community of organisms interacting with one another and with the chemical and physical factors making up their environments.”^[1]

Detailed maps showing the biomes of the world are not available due to conceptual differences of definitions and reliable global databases. There are many excellent and detailed studies of specific habitats around the world, and using these and the global soils and climate database of the world,^[2] a map showing the distribution of the major biomes was drawn. The terms used to describe the biomes are common in use, but their subdivisions are based on important differentiating factors.

MAJOR BIOMES

The five major biomes of the world and their subdivisions are listed in [Table 1](#) and their distribution is shown in [Fig. 1](#). Some geographers have used elevation as a differentiating factor and recognized a “montane biome.” This biome’s small extent precludes its description in this section. Few geographers recognize the Mediterranean biome as presented here as a subdivision of the Temperate biome. Within each biome, some major distinction is made. In the Polar biome, an attempt is made to differentiate areas with permafrost and warmer areas with intermittent permafrost. The latter is termed “interfrost.” The Boreal Forest biome has a humid and a semiarid counterpart similar to the temperate and tropical zones. The semiarid vegetation is generally grass or shrubs. The desert areas are divided into hot, cool, and cold equivalents.

Deserts occupy about 30% of the global ice-free landmass. Potential evapotranspiration exceeds precipitation

during most days of the year, and the biome shows the greatest contrast in temperature, both diurnal and seasonal. Desert vegetation is very sparse with a few isolated shrubs dominated by succulents and annuals. Grasses and forbs become more dominant at the margins. The tropics occupy about 27% of the landmass and are characterized by only a small variation in temperature during the year. The soil moisture conditions range from semiarid to humid. The availability of soil moisture determines the biota. The temperate grassland and forest biome occupies about 15%. The cold biomes, the boreal and polar, are mainly in the northern latitudes with small areas in South America. The low temperatures control the flora and fauna, and vegetation is scarce to non-existent in the extreme cold polar regions. Each of these areas has specific kinds of soils that have developed as a response to the specific soil moisture and temperature conditions. The diversity in vegetation has its equivalent in fauna with animal species adapting to the specific bioclimatic conditions. Within each of the biomes, there are specific conditions that promote unique flora and fauna. Examples of such localized systems are volcanic hot springs, wetlands, and oases.

Soils of the Major Biomes

Soil temperature and moisture, with their seasonal and annual variations, are an integral part of the soil classification system called Soil Taxonomy^[3] which is used here. As [Fig. 1](#) and [Table 1](#) are based on soil moisture and temperature conditions, there is an implied link between the biomes and their soil resource endowments. The purpose of this introductory section is to present the general geographic distribution of the biomes and the major soils characterizing each. The major soil orders are listed in [Table 2](#).

Table 1 Major global biomes with subdivisions and their areas

Global land area					
Biome	Subdivision	Remarks	In thousand km ²	In%	In%
Polar	Permafrost	Mosses and lichens	10,547	8.06	15.46
	Interfrost	Shrubs and stunted trees	9,685	7.40	
Boreal	Semi arid	Shrub	3,552	2.72	9.94
	Humid	Forest	9,446	7.22	
Temperate	Semiarid	Grassland	7,348	5.62	15.14
	Humid	Forest	12,453	9.52	
	Mediterranean warm	Shrubs and forest	3,624	2.77	3.37
	Mediterranean cold	Forbs and shrubs	791	0.60	
Desert	Hot	Barren	4,418	3.38	29.47
	Cool	Barren	28,521	21.81	
	Cold	Barren	5,599	4.28	
Tropical	Semiarid	Grassland, savanna	20,299	15.52	26.62
	Humid	Forests	14,514	11.10	
Total			130,796	100.00	

The subsequent sections elaborate on the soil resources of each of the six biomes listed in [Table 1](#).

Polar Biome

Characterized largely by a mean annual soil temperature of less than 0°C, the Polar biome occupies a large land mass

adjacent to the Arctic circle to about 60°N latitude. This is generally the ice-free land of the northern latitudes. The sub-zero soil temperatures that prevail during most of the year are conducive to the formation of permafrost. The dominant soils of the region are Gelisols, occupying 59% of the Polar biome. Turbels freeze and thaw once or more during a year which triggers cryoturbation or physical

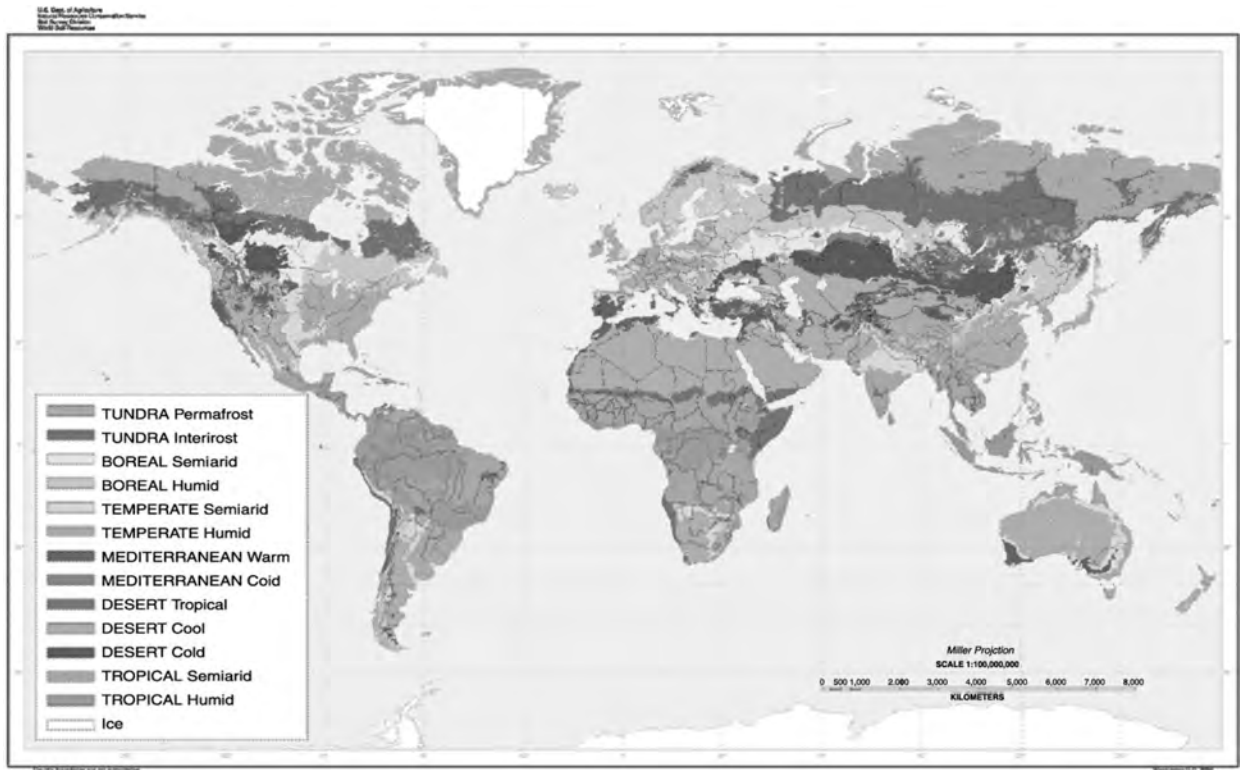


Fig. 1 Major biomes.

Table 2 Dominant soils in the major biomes (area in thousand km²).

Soil Order	Polar			Boreal			Temperate				Desert			Tropics	
	Ice-free land	Permafrost		Interfrost	Semi-arid	Humid	Semi-arid	Humid	Mediterranean		Warm	Cool	Cold	Semi-arid	Humid
									warm	cold					
Gelisols	11,869	10,476	—	1,393	—	—	—	—	—	—	—	—	—	—	—
Histosols	1,526	—	—	—	299	686	17	82	1	8	—	23	93	37	281
Spodosols	4,596	55	—	1,058	227	2,566	105	450	39	6	—	—	28	—	61
Andisols	975	6	—	56	47	196	26	135	32	—	17	10	14	198	235
Oxisols	9,811	—	—	—	—	—	9	184	—	—	27	4	—	3,387	6,199
Vertisols	3,160	—	—	—	2	15	595	303	99	0	239	651	0	1,170	87
Aridisols	15,629	1	—	—	60	76	353	18	203	23	1,824	11,014	2,013	44	1
Ultisols	11,053	—	—	—	1	14	266	3,122	20	18	—	—	—	4,371	3,240
Mollisols	9,161	9	147	—	1,059	511	1,280	1,232	874	323	49	1,110	2,255	136	176
Alfisols	12,621	—	—	—	1,002	1,816	1,867	2,149	860	125	—	—	—	4,165	638
Inceptisols	21,805	—	—	6,751	167	3,333	1,636	3,078	683	89	—	—	—	3,566	2,502
Entisols	21,467	—	—	280	172	232	1,173	1,697	807	45	1,708	10,693	342	3,225	1,093
Miscellaneous	7,122	—	—	—	516	1	20	4	6	153	553	5,016	853	—	—
Total	130,796	10,547	9,685	3,552	9,446	7,348	12,453	3,624	791	4,418	28,521	5,599	20,299	14,514	

mixing of the soil material. The orthels are generally shallower or drier soils and have little or no cryoturbation. The low temperatures and periodic moisture saturation promote the accumulation of organic matter. The histels are characterized by the high organic matter and occupy about 5% of the Polar biome. These organic soils form the largest contiguous extent of such soils in the world and serve as an important sink of carbon dioxide. Other soils that occur in the Tundra zone are Inceptisols 33% and Spodosols 6%, while Mollisols and Entisols are each 1% of the area.^[4]

Boreal Forest Biome

Bordering the southern flank of the Polar biome is the Boreal Forest biome in the northern hemisphere. The high volcanic area between Chile and Argentina has similar climatic regimes but the nature of the soil and physiography may result in a different set of habitat conditions. There are about 243,000 km² of Andisols or volcanic ash soils characterizing the Boreal Forest biome in the southern hemisphere. Such soils only occur as a small area in the Kamchatka Peninsula in the Northern Hemisphere. In Northern Europe and Siberia, the Boreal Forest biome has a humid part and is bordered in its southern periphery by a semiarid to arid part. In Canada, the semiarid part is in the middle of the continent. A range of soils are present, and the most extensive are the Spodosols, which occupy 2.7 million km² (20.6%). The Spodosols are mostly in the humid part of the Boreal Forest biome and form under acid vegetation. Histosols are present in the depressions in this cold region.

Temperate Grassland and Forest Biome

The Temperate Grassland and Forest biome extends from about 25 to 55°N with counterparts in the southern hemisphere. Large areas in this belt are also deserts. Due to the favorable climate and soil endowments of this biome, much of the land is used for agriculture and native habitats are local and sporadic. About 50% of the zone (12.1 million km²) is occupied by Mollisols and Alfisols (grasslands), and Ultisols in the forests. Their general good fertility and tilth have made them in great demand for grain production. Due to the long history of civilization in the Temperate Grassland and Forest biome of Europe and China, pristine ecosystems are rare. In large areas, there have been successive replacements and changes in the floral and faunal composition. Within the Temperate biome, are areas characterized by moist winters and dry summers. These Mediterranean conditions are conducive to unique ecosystems. These areas occur around the Mediterranean Sea and small areas in the Western United States and Southern Australia.

Desert Biome

Deserts occupy about 38.6 million km² and may be distinguished as warm (or tropical), cool (or temperate), and cold (or boreal) kinds of habitats. Although the biome is characterized by lack of moisture for normal vegetative growth of most plants, the ambient temperature conditions further distinguish the habitat conditions. About 38.6% of the Desert biome is occupied by Aridisols, which by definition have some kind of subsurface horizon. Shifting sands or moving dunes occupy about 13.7% and Entisols, which are deposits, occupying about 33% of the Desert biome. The harshness of the environment has resulted in plants and animals with special adaptive features. Many of the Aridisols are increasingly used for agriculture when irrigation facilities are made available. In this fragile ecosystem, both soil and habitat conditions are drastically altered when irrigation is introduced.

Tropics Biome

An absence of winter and summer temperature extremes characterizes the Tropics biome, which occupy about 34.8 million km². Availability of moisture separates the semiarid from the humid tropics. As a corollary to the desert, in the humid tropics there are the perhumid areas where the potential evapotranspiration never exceeds the precipitation during any month of the year. This is a biomic condition that deserves much greater detailed studies. The Tropics biome is the home to the Oxisols (9.6 million km²), which are unique to this biome. These are highly weathered soils where the original vegetation is closed or open forests. Most of the plant nutrients are concentrated in the top 5 cm of the soil and recycled through plant uptake and leaf fall. Ultisols also occur in this biome. Apart from these soils, there are small areas of most of the other soil orders. The wetlands of the tropics present a separate and unique habitat condition and those along the coast have some special soils.

REFERENCES

1. Tootil, E. *Dictionary of Biology*; Intercontinental Kook Productions, Ltd.: Maidenhead, Berkshire, 1980.
2. Eswaran, H.; Beinroth, F.H.; Kimble, J.; Cook, T. Soil diversity in the tropics: Implications for agricultural development. In *Myths and Science of Soils of the Tropics*; Lal, R., Sanchez, P.A., Eds.; Soil Sci. Soc. Am. Spec. Publ.; Soil Science Society of America: Madison, 1992; Vol. 29, 1–16.
3. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; Natural Resources Conservation Service. US Department of Agriculture, Handbook 436; US Government Printing Office: Washington, 1999.
4. Soil Survey Staff. *Keys to Soil Taxonomy*, 9th Ed.; US Government Printing Office: Washington, 2003.

Grass Strip Hydrology

Hossein Ghadiri

School of Australian Environmental Studies, Griffith University, Nathan Campus, Nathan, Queensland, Australia

Calvin W. Rose

Faculty of Environmental Sciences, Griffith University, Nathan Campus, Nathan, Queensland, Australia

Abstract

The zone of hydraulic adjustment or backwater is a zone of considerable deposition from sediment-laden flow. The efficiency of the grass strips in slowing down the flow and unloading its sediment in the backwater region appears to be largely independent of the width of strips in the flow direction. The layer of deposited sediment formed in the backwater region tends to be richer in larger particles than the eroding sediment. Sediment size distribution appears to be a dominant factor governing the efficiency of the buffer strip in trapping sediment. Soil-sorbed nutrients and other agricultural chemicals are mainly attached to finer soil particles, which are more likely to pass through the strips largely unchanged. Thus the sediment that emerges from a buffer strip can be enriched in such chemicals relative to the eroding soil. Hence although buffer strips can be very efficient in forcing the deposition of suspended sediment in the backwater region on certain slopes, they are helpful, but less effective, in preventing chemical pollutants from entering surface water bodies.

INTRODUCTION

The use of vegetated strips is one of the simplest and most widely employed methods of reducing the flux of eroding soil and associated chemicals across slopes (Fig. 1) and into streams. Grass strips of various type, width, spacing, density, rigidity, and strength have been used for soil erosion control on sloping lands and for water quality control in riparian zones with varying degrees of success.^[1–3] It was widely believed, with some support from field studies, that the effectiveness of buffer strips in sediment trapping was largely because of a filter-type action, with strips filtering out the suspended sediment as runoff passes through them, leaving the emerging runoff significantly cleaner in terms of sediment, nutrients, and soil-sorbed contaminants. Because of such perceived mode of operation, buffer strips have been commonly referred to as “filter strips.”^[4,5] However, studies have shown that a filtering action is not a major contributing process to buffer strip effectiveness. Considerable experimental evidence in the literature suggests little or no net deposition within the buffer strips.^[6–9] Some have reported the occurrence of erosion, rather than deposition, inside the grass strips.^[4,8]

Studies have sought to understand the mechanics of flow through grass strips, seeking the real reason for the inconsistencies observed in their effectiveness.^[2,7–11] It has been recognized that a flow-resistive element such as a cross-slope strip of vegetation modifies the hydrology of overland flow, and that this modification has implications for

the transport and deposition of sediment and associated nutrients in and around the strips.^[7,8,11,12] It is the hydraulic consequence for overland flow when it meets and flows through a resistive element, which first needs to be understood to ascertain the effectiveness, or otherwise, of grass strips in reducing erosion, enhancing deposition, and reducing the transportation of pollutants into surface water bodies. Such understanding is also necessary for the development of models and the prediction of buffer strip effectiveness under different conditions. This entry covers the progress made in this area.

EFFECT OF BUFFER STRIPS ON FLOW HYDRAULICS

One of the main hydrologic features of flow through grass strips is the formation of a pronounced backwater,^[8] or a zone of hydraulic adjustment.^[9,13] The length of this zone appears to be dependent on land slope, strip density, and flow rate (Fig. 2). At any constant flow rate, the length of backwater *B* has been shown to be linearly related to the ratio of strip density over slope:

$$B = k \frac{D}{S} \quad (1)$$

where *k* is a proportionality constant, *D* is percent grass coverage in the strip, and *S* is slope. Eq. 1 shows that decreasing density and increasing slope both have the effect



Fig. 1 Strips of Vetiver grass used as a soil erosion control technique in Queensland, Australia.

Source: Photo courtesy of Mr. Cyril Ciesiolka of DPI, Toowoomba, Queensland, Australia.

of decreasing backwater length^[8] (Fig. 3). The starting point of the backwater region is a hydraulic jump, which moves closer to the strip as the slope increases or as the grass density decreases.

The front edge of the backwater is usually sharp and clear at high slopes (Figs. 2 and 3), getting less distinct as it moves away from the strips with decreasing slope or increasing grass density. At low slopes of around 1%, the front edge of the backwater breaks into a number of waves. The width of the strips does not appear to play an important role in backwater formation or its characteristics.^[8]

Theoretical interpretations of flow through grass strips are very few. However, there is a long history of investigation of hydraulic jumps in the hydraulic literature. Chow^[13] developed the hydraulic jump theory for deep rectangular

channel flow encountering solid barriers such as dams and weirs. This interpretation does not apply to our condition of shallow unconfined flow being partially blocked by porous barriers such as grass or nail strips. Rose et al.^[9] have shown that the change in hydraulics because of the presence of a buffer strip can be understood in terms of the conservation of momentum theory.

EFFECT OF BUFFER STRIPS ON SEDIMENT TRANSPORT

Runoff sediment concentration results from the balance of two quite rapid processes, such as, entrainment (or re-entrainment) by the overland flow and deposition. Even a



Fig. 2 Illustrating flow and sediment deposition in the backwater region of a barrier strip.

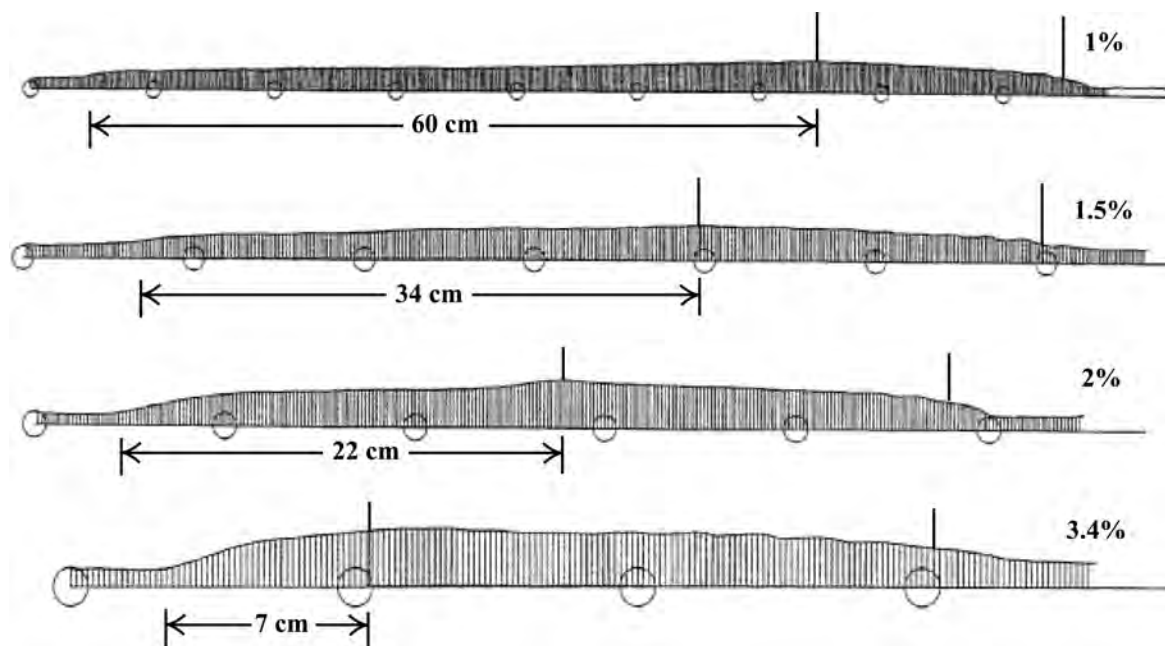


Fig. 3 Effect of slope on the length of backwater (flow height recorded digitally; flow is from left to right and the two long vertical lines on each trace show the beginning and the end of the grass strip).

modest reduction in flow velocity in the backwater region produced by the vegetation buffer strips reduces the entrainment rate, resulting in net deposition in this region. Thus the effect of grass strips on erosion/sedimentation is mainly through their impact on surface hydrology.

Net deposition of coarse sediment load commences at the starting edge of the backwater (or at the hydraulic jump) (Fig. 2). This site of initial net deposition approaches the strip with an increase in slope, or a decrease in strip density, eventually burying and entering the strip. Sediment deposition generally follows the hydrologic pattern established at the early stages of the flow, as predicted by Eq. 1.

Little, if any, deposition appears to take place inside the strips prior to the point of grass collapse.^[6,8,14] Sediment that deposits in the backwater moves into the strip zone only after burying the first few rows of the vegetation.^[2] The unburied rows then become the new front edge of the strip. However, once the first rows of the strips give way, the process usually continues until the entire width of the strip collapses at one point or more (Fig. 4).

Many researchers have reported that the width of the grass strip has little effect on the efficiency of the strips in slowing down the flow, or unloading its sediment content.^[2,15] However, there are others who have reported

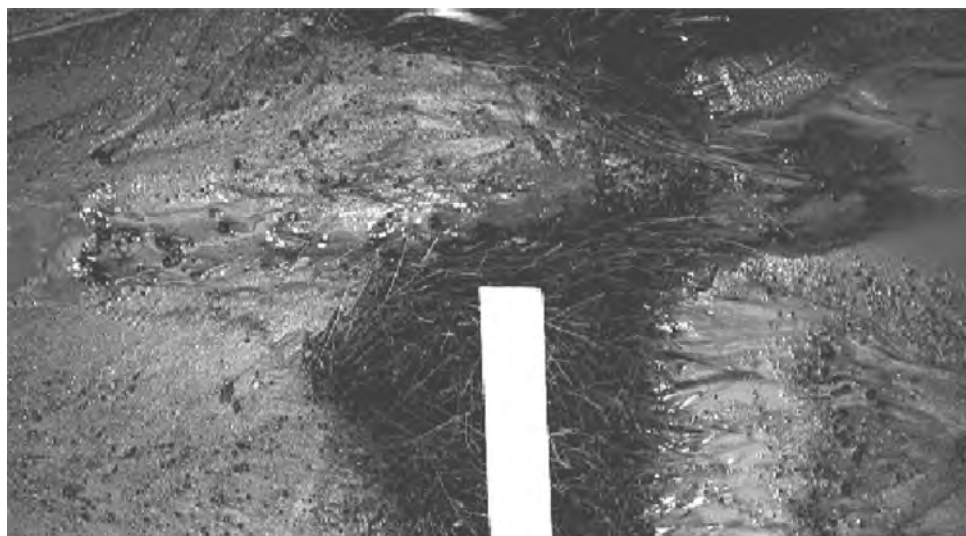


Fig. 4 Collapsed grass strip on 8% slope (flow from left to right; total collapse happened after the first few rows of grass were buried under the deposited sediment; vertical white bar is a 15-cm ruler).

the opposite.^[1,3] The disagreement between the two groups is yet to be resolved.

THEORETICAL INTERPRETATION OF SEDIMENT TRANSPORT THROUGH BUFFER STRIPS

Mass conservation of deposited sediment was used by several researchers^[16,17] to describe the resulting change in the shape of the deposited layer. However, their models do not represent the hydrology and net deposition upslope of the grass strips (Fig. 2), and only deal with single-size particles. A more appropriate theoretical explanation of flow through porous barriers is given by Rose et al.,^[12] who showed that the appropriate equation to solve (under steady flow conditions) is as follows:

$$\frac{\delta}{\delta t} M_{di} = -q \frac{dc_i}{dx} = d_i - r_{ri} \quad (2)$$

where t is time, M_{di} is the mass per unit area of sediment of size class i in the region of net deposition, q is volumetric water flux, c_i is sediment concentration of size class i , x is downslope distance, d_i the rate of deposition per unit area, and r_{ri} is the rate of re-entrainment of newly deposited sediment of size class i .

EFFECT OF BUFFER STRIPS ON CHEMICAL TRANSPORT

Soil chemicals are preferentially sorbed to the finer fractions of soil.^[18,19] Buffer strip influences the transport of sorbed chemicals mainly through sediment sorting processes, which take place in the backwater. Sediment passing through the strips is significantly finer than that initially dislodged by the flow. It was shown by Ghadiri and Rose^[19] that the concentrations of organic matter, sorbed nutrients, and agricultural chemicals can be significantly higher on the finer particles. The preferential deposition of larger aggregates and particles in the backwater leaves the sediment, which emerges from the buffer strip, relatively enriched in fine particles and thus in soil-sorbed chemicals. Such transmitted fine particles, together with their chemical load, may stay in suspension until entry into receiving waters. Therefore, grass strips are less effective in reducing overland transport of solutes or solids-associated chemical pollutants than they are in reducing sediment load.

CONCLUSION

Buffer strips behave like porous barriers against flow, creating backwater regions upslope of the strips with reduced flow velocity and increased depth in a length that varies with slope, strip density, and flow rate. For a constant flow rate, there is a linear relationship between

backwater length and the ratio of strip density over slope. Momentum theory appears to provide a reasonably good prediction, both of the shape of the water profile within the strips, and of the slope and extent of backwater region.

The zone of hydraulic adjustment or backwater is a zone of considerable deposition from sediment-laden flow. The efficiency of the grass strips in slowing down the flow and unloading its sediment in the backwater region appears to be largely independent of the width of strips in the flow direction. The layer of deposited sediment formed in the backwater region tends to be richer in larger particles than the eroding sediment. Sediment size distribution appears to be a dominant factor governing the efficiency of the buffer strip in trapping sediment. Soil-sorbed nutrients and other agricultural chemicals are mainly attached to finer soil particles, which are more likely to pass through the strips largely unchanged. Thus the sediment that emerges from a buffer strip can be enriched in such chemicals relative to the eroding soil. Hence although buffer strips can be very efficient in forcing the deposition of suspended sediment in the backwater region on certain slopes, they are helpful, but less effective, in preventing chemical pollutants from entering surface water bodies.

REFERENCES

1. Dickey, E.C.; Vanderholm, D.H. Vegetation filter treatment of livestock feedlot runoff. *J. Environ. Qual.* **1981**, *10*, 279–284.
2. Dillaha, T.A.; Reneau, R.B.; Mostaghimi, S.; Lee, D. Vegetation filter strips for agricultural nonpoint source pollution control. *Trans. ASAE* **1989**, *32*, 513–519.
3. Van Dijk, P.M.; Kwaad, F.J.P.; Klapwijk, M. Retention of water and sediment by grass strips. *Hydrol. Process.* **1996**, *10*, 1069–1080.
4. Loch, L.J.; Espigares, T.; Costantini, A.; Garthe, R.; Bubb, K. Vegetative filter strips to control sediment movement in forest plantations: Validation of a simple model using field data. *Aust. J. Soil Res.* **1999**, *37*, 929–946.
5. Jin, C.X.; Romkens, M.J.M. Experimental studies of factors in determining sediment trapping in vegetative filter strips. *Trans. ASAE* **2001**, *44*, 277–288.
6. Lidgi, E.; Morgan, R.P.C. Contour grass strips: A laboratory simulation of their role in soil erosion control. *Soil Technol.* **1995**, *8*, 109–117.
7. Ghadiri, H.; Hogarth, W.; Rose, C.W. The effectiveness of grass strips for the control of sediment and associated pollutant transport of runoff. In *The Role of Erosion and Sediment Transport in Nutrient and Contaminant Transfer*; Stone, M., Ed.; International Association of Hydrologic Sciences (IAHS) Publication No. 263; Wallingford, 2000; 83–91.
8. Ghadiri, H.; Rose, C.W.; Hogarth, W.L. The influence of grass and porous barrier strips on runoff hydrology and sediment transport. *Trans. ASAE* **2001**, *44*, 259–268.

9. Rose, C.W.; Hogarth, W.; Ghadiri, H.; Parlange, J.-Y.; Okom, A. Overland flow to and through a segment of uniform resistance. *J. Hydrol.* **2002**, *255*, 134–150.
10. Hairsine, P.B. Comparing grass filter strips and near-natural riparian forests for buffering intense hillslope, sediment sources. In *Proceedings of the 1st National Conference on Stream Management in Australia, CRC for Catchment Hydrology*, Monash University, Melbourne, 1996.
11. Dabney, S.M.; Meyer, L.D.; Harmon, W.C.; Alonso, C.V.; Foster, G.R. Depositional patterns of sediment trapped by grass hedges. *Trans. ASAE* **1995**, *38*, 1719–1729.
12. Rose, C.W.; Yu, B.; Hogarth, W.; Okom, E.A.; Ghadiri, H. Sediment deposition from flow at low gradients into a buffer strip—A critical test of re-entrainment theory. *J. Hydrol.* **2003**, *280* (1–4), 33–51.
13. Chow, V.T. *Open Channel Hydraulics*; McGraw-Hill Book Company: Singapore, 1959.
14. Boubakari, M.; Morgan, R.P.C. Contour grass strips for soil erosion control on steep lands: A laboratory evaluation. *Soil Use Manage.* **1999**, *15*, 21–26.
15. Smith, C.M. Riparian afforestation effects on water yields and water quality in pasture catchments. *J. Environ. Qual.* **1992**, *21*, 237–245.
16. Munoz-Carpena, R.; Pearson, J.E.; Gilliam, J.W. Modelling hydrology and sediment transport in vegetative filter strips. *J. Hydrol.* **1999**, *214*, 111–129.
17. Deletic, A. Modelling of water and sediment transport over grasses areas. *J. Hydrol.* **2001**, *248*, 168–182.
18. Ghadiri, H.; Rose, C.W. Sorbed chemical transport in overland flow: Part 1. An enrichment mechanism for sorbed nutrients and pesticides. *J. Environ. Qual.* **1991**, *20*, 628–633.
19. Ghadiri, H.; Rose, C.W. Sorbed chemical transport in overland flow: Part 2. Enrichment ratio of sorbed chemicals and its variation with time, particle size and erosion process. *J. Environ. Qual.* **1991**, *20*, 634–641.

Grasslands Soils

Douglas D. Malo

Biostress Center of Excellence, Brookings, South Dakota, U.S.A.

Abstract

Soils and plants interact and evolve together forming different ecosystems. These soil–plant relationships are often most strongly expressed when native vegetation is present. The soils in grasslands, a major ecosystem, are very different from other soils due to this interaction.

INTRODUCTION

Soil science developed in response to problems. In Europe during the 1800s, food shortages, social upheavals, and declining soil productivity brought about the need to study soil to improve and increase its productivity. At the same time, in Russia, a need arose to administer and manage geographically diverse soil resources. Russia had large areas of fertile, productive soils unlike those in Europe. As a result, Russian scientists developed an inventory of agricultural resources and determined the factors causing soils to vary across Russia. Soils were found to have relationships with climatic and vegetative zones.^[1] It is from these concepts that the living soil individual was developed. Soil is a natural body and not a geologic formation. With time, soil develops from the parent material under the influence of the climate, vegetation, and topography (relief). These five factors of soil formation are interdependent and not independent. Changing one soil-forming factor often changes other soil-forming factors. Changing any one, some, or all of the soil-forming factors causes differences in soils and soil profiles.^[2]

IMPORTANCE, LOCATION, AND EXTENT OF GRASSLAND SOILS

Grassland soils have thick, dark-colored, humus-rich, surface horizons, or layers and are some of the most productive soils in the world. These soils occupy about 7% of the world's land area (Table 1). The majority of grassland soils occur in temperate (3.4%) and boreal (3.2%) regions.^[3] These soils are the dominant food- and fiber-producing soils in subhumid and semiarid regions because of favorable soil properties and climatic conditions. The world's breadbaskets (e.g., U.S. Midwest and Great Plains prairies; Canadian prairie provinces of Manitoba and Saskatchewan; the pampas of Argentina, Paraguay, and Uruguay; and the steppe regions of Europe, Russia, Mongolia, and Northern China) are dominated by grassland soils (Fig. 1). The largest grassland area in the world is found in Kazakhstan,

Russia, and the Ukraine. Generally, small grains and grain sorghum are raised in the drier grassland regions. The warmer, humid grasslands are better suited for row crops such as maize (corn) and soybeans. Where slopes are too great for cultivation or the climate is not favorable, grassland soils are used for pasture and rangeland. Native grassland types include: desert grasslands, <15 cm high; short-grass prairie, 15–30 cm high; mid-grass prairie, 30–100 cm high; and tall-grass prairie, 1–3 m high.^[7]

Grassland soils (e.g., Mollisols) are very extensive in the United States (Fig. 2), especially the Midwest and Great Plains. They occupy about 21.5% of the U.S. land area,^[3] more than any other soil ecosystem.

Grasslands tend to occur in subhumid to semiarid regions, where average annual precipitation ranges from 20 to 90 cm and average annual temperatures range from 0°C to 32°C.^[9] Grasslands tend to be located in areas between deserts (e.g., Aridisols) and forested (e.g., Alfisols) areas.^[3] Different climates with similar effective annual precipitation levels (e.g., prairies of California and western Kansas) give rise to grassland soils with similar properties.^[10] Soil texture and topography do modify the boundaries between prairie, forest, and desert areas (e.g., sandy, south-facing slopes are drier than loamy, north-facing slopes).^[11]

PROPERTIES OF GRASSLAND SOILS

Soils that form under grass vegetation have unique soil properties and characteristics (Table 2). Grassland soils (e.g., Houdek soil—fine-loamy, mixed, superactive, mesic Typic Argiustoll) have thick, soft, dark-colored, moderately high organic carbon-containing A horizons (e.g., 0.5–10% organic carbon) with developed structure (e.g., mollic surface). Granular structure is most common in surface horizons with blocky (humid areas) and prismatic (semiarid areas) structure present in B horizons. Cultivation often causes granular surface soil structure to change to subangular blocky.^[13] Grassland soils have sufficient strength to support grazing livestock and normal

Table 1 Global grassland soil distribution (based on ice-free land).

Continent/region	Area (km2)	Area (% of region)	Area (% of world's grasslands)	Area (% of world's total area)
Africa	76,911	0.26	0.84	0.06
Asia	4,041,017	8.45	43.90	3.10
Australia/Oceania	103,569	1.31	1.13	0.08
Central America	32,302	4.64	0.35	0.02
Europe	724,280	12.62	7.87	0.56
North America	3,230,829	15.57	35.10	2.48
South America	994,431	5.67	10.81	0.76
Total	9,203,339	100.00	7.06	

Source: From Soil Survey Staff,^[3] Superintendent of Documents, U.S. Government Printing Office and U.S. Department of Agriculture-Natural Resources Conservation Service,^[5] and unpublished data from P. Reich, USDA–NRCS, World Soil Resources Division.

cultivational activity. The dark-colored A horizon slowly changes into a B horizon in most prairie-derived soils (Fig. 3).

The Houdek Bt horizon has developed structure and evidence of clay illuviation (Table 2). Clay illuviation is much less than that found in comparable forested soils. Base saturation and soil pH are high in all layers of most grassland soils. Divalent cations, especially Ca^{2+} , dominate the soil-cation exchange sites. Most grassland soils have smectite (2:1 lattice silicate clay minerals) present in the clay fraction, and textures are usually finer than loamy fine sand.^[14] The average cation-exchange capacity of the soil solum is at least $15 \text{ cmol}_c \text{ kg}^{-1}$ but averages $25\text{--}35 \text{ cmol}_c \text{ kg}^{-1}$. Grassland soils can have many types of B horizons. Soils of the steppes tend to have horizons of lime accumulation (Bk in Fig. 3), whereas the grassland soils in humid areas tend to lack horizons of lime accumulation but have argillic, Bt, horizons.

Some grassland soils have pans (e.g., duripans, petrocalcic layers), and some have strongly developed eluvial, E, horizons (e.g., albic). Still other grassland soils have only weakly developed B horizons (Bw and Bg) or no B horizon at all.

GENESIS OF GRASSLAND SOILS

The genesis of grassland soils is strongly dependent on the vegetation. Grasses, a source of organic matter and nutrient cycling, provide these two key components to the soil body. The principal process in grassland genesis is the accumulation of high base-content humus in the soil from the dense grass root systems near the soil surface.^[2] Through the microbial decay of plant roots and plant tissue, organic matter is added to the soil surface and the soil profile in

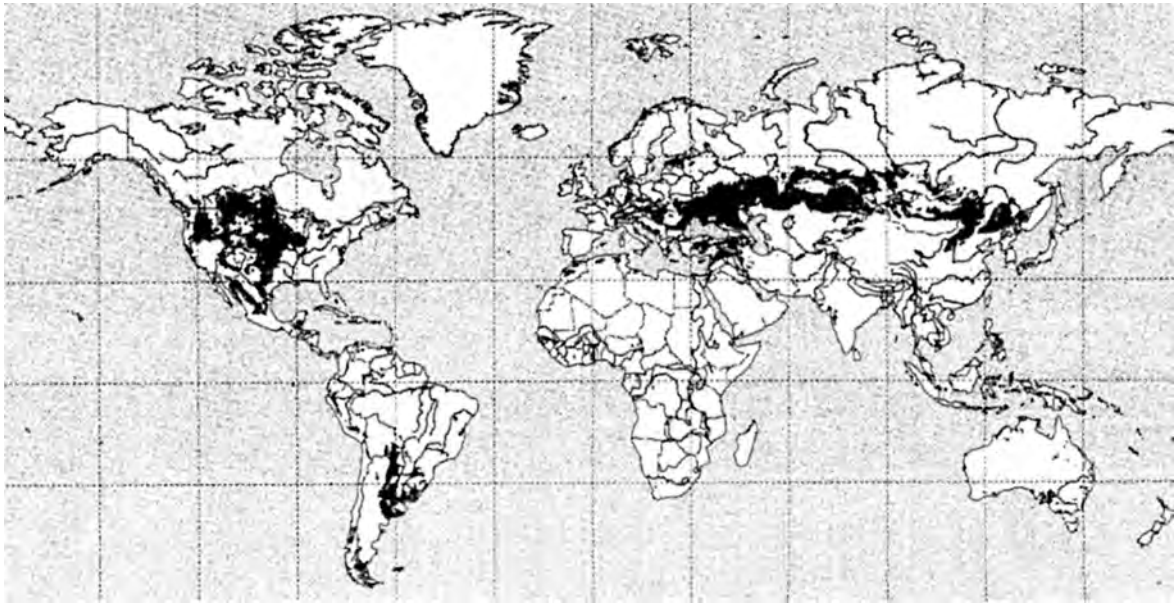


Fig. 1 Distribution of grassland soils (Mollisols) in the world.

Source: From Soil Survey Staff^[3] and U.S. Department of Agriculture-Natural Resources Conservation Service.^[5]

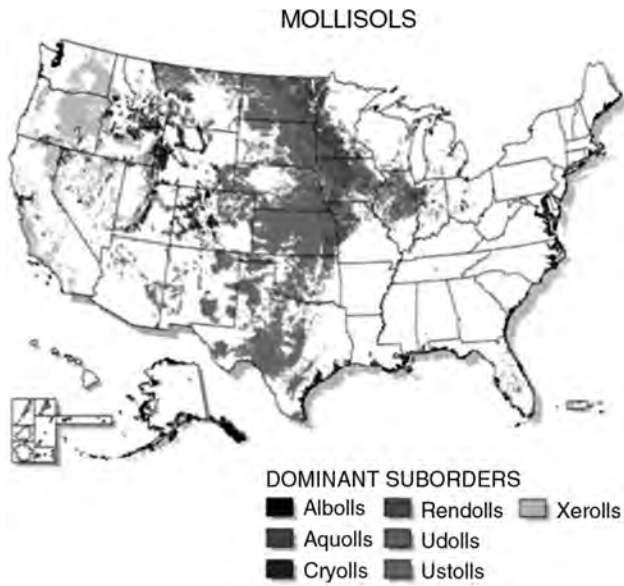


Fig. 2 Distribution of grassland soils (Mollisols) in the United States.
Source: From U.S. Department of Agriculture-Natural Resources Conservation Service.^[8]

grasslands. Soil macrofauna (e.g., earthworms) incorporate the aboveground biomass tissue into the soil surface and help increase surface organic-carbon levels. Most grass roots are found in the top 30 cm of grassland soils.^[15] Each year more than 50% of the biomass produced by unharvested grasses (nearly all of the aboveground biomass and about 30% of the below-surface biomass) is added to the soil.^[16] The amount of aboveground grassland-biomass production can range from 1500 to 3500 kg ha⁻¹ yr⁻¹, dry weight.^[2] The amount of air-dry organic matter added per year can vary depending on climate conditions and vegetation type (e.g., 1250 kg ha⁻¹ for humid prairie areas to 600 kg ha⁻¹ for short- to mid-grass prairies).^[17]

The distribution of organic carbon in grassland soils shows a gradual decline with increasing soil depth due to the gradual decline of fibrous grass roots and microbial activities with increasing soil depth (Table 2). This pattern

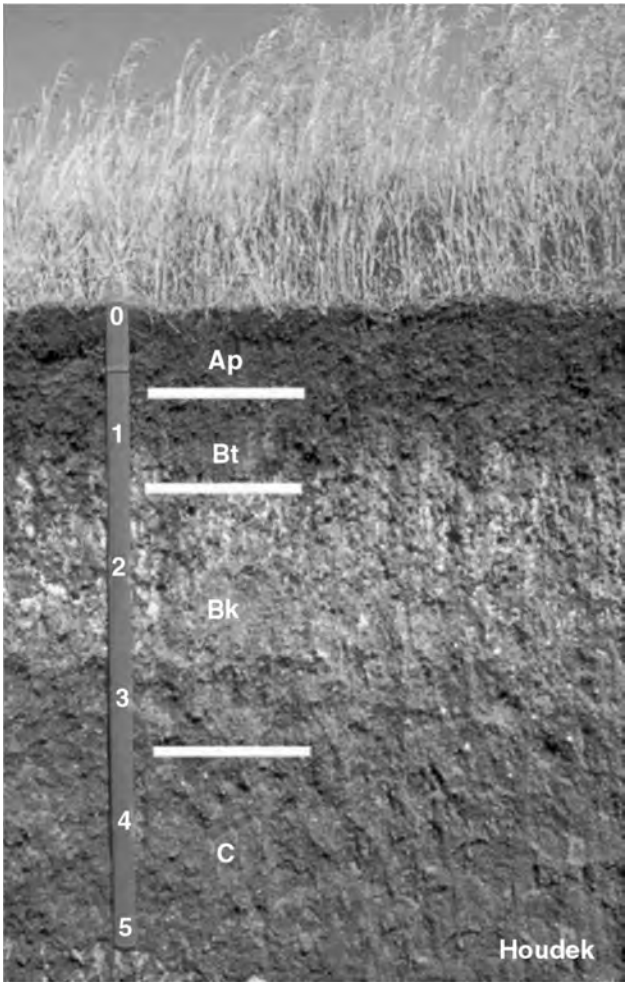


Fig. 3 Typical Great Plains grassland soil (Houdek—fine-loamy, mixed, superactive, mesic Typic Argiustoll). Scale is in feet.

of organic-carbon distribution is strikingly different from that in forest-derived soils where the organic carbon is high in a thin surface horizon and low in the rest of the soil. As a result, the structural stability of grassland soil peds tends to be stronger than non-grassland soil peds. Humus is not very water soluble, so it tends to stay in one location unless there

Table 2 Selected soil properties of typical grassland (Houdek) soil.

Horizon	Depth (in.)	Clay (%)	pH	Org. C (%)	Base saturation (%)	Structure	
						Grade	Shape
Houdek (grassland derived)							
Ap	0–6	22.0	6.2	4.6	84	Strong	Granular
Bt1	6–10	29.2	6.4	1.8	86	Strong	Subangular blocky
Bt2	10–18	26.5	7.0	1.1	95	Moderate	Prismatic
Bk1	18–28	25.9	8.2	0.5	100	Moderate	Prismatic
Bk2	28–40	24.8	8.4	0.3	100	Weak	Prismatic
C	40–80	23.2	8.4	0.2	100	Structureless	Massive

Source: From Natural Resources Conservation Service.^[12] and unpublished data from D.D. Malo, South Dakota State University.

is suspension or some mechanism for mechanical movement. The organic-carbon decline continues until the native rooting depth of the grasses is reached, and there is little or no organic carbon.

Grasses are large users of bases, especially calcium (Ca^{2+}). As a result, when grass residue is added to the soil surface, large amounts of cations (Ca^{2+} , Mg^{2+} , and K^{1+}) are brought to the surface, replacing cations lost by leaching and weathering activities. This results in higher soil pH values and base saturation throughout the profile (Table 1). The high concentration of bases helps, along with the high content of shrinking–swelling silicate clay minerals and high humus levels, to form the granular structure that is common in grassland soils. In general, leaching under grass vegetation is minimal when compared to other types of vegetation (e.g., forest). Deep percolation of precipitation to the water table is not common, and deep leaching is not usually evident in most grassland soils. Grasses tend to utilize the water in the soil, preventing the deep leaching of nutrients. The grass plants keep recycling bases to the soil surface and this limits soil acidity, clay illuviation, and base-saturation reductions.

The development of textural horizons and silicate clay illuviation in grassland soils tends to happen in three steps:^[18]

1. Removal of free carbonates: as long as free lime is present, the soil remains flocculated and little or no silicate clay will translocate.
2. Silicate clay formation and alteration: silicate clays are formed and altered as a result of weathering and soil genesis.
3. Silicate clay eluviation and illuviation: the fine silicate clays ($<0.2\ \mu\text{m}$) move and precipitate out lower in the soil profile. The place where silicate clay illuviation occurs depends on pH, lime content, microbial activity, and moisture levels. Often, joint illuviation of both silicate clays and humus in grassland soils forms organo-argillans.

Another key indicator of soil genesis in grassland soils is phosphorus. Significant differences in phosphorus species have been noted in different grassland soils. Semiarid grasslands (Ustolls or Chestnut soils) had a larger percentage of calcium phosphates when compared to humid grasslands (Udolls or Chernozems), whereas humid grasslands had a higher percentage of iron phosphate.^[19] Phosphorus distributions (total, organic, and inorganic) within the soil profile depend on horizon, lime illuviation, vegetation, and drainage.^[18]

With cultivation in grassland soils, significant changes occur in the soil. Significant organic-carbon losses ($>20\%$ reduction in organic C levels after 20 years of cultivation) and soil quality reductions occur.^[7] The losses are most dramatic in the soil surface and diminish with increasing soil depth.^[18]

Table 3 Soil classification of grassland soils.

Soil or environmental conditions	Soil taxonomy	1938/1949 equivalents	FAO equivalents
Albic horizon	Alboll	Planosol	Planosol
Wet, hydric conditions	Aquoll	Gley	Gleysol
Very cold	Cryoll	Chernozem	Greyzem
Highly calcareous parent materials	Rendolla	Rendzina	Rendzina
Humid	Udoll	Brunizem	Phaeozem
Moist spring/dry summer	Ustoll	Chernozem, Chestnut	Chernozem, Kastanozem
Dry summer/moist winter	Xeroll	Chestnut, Brown	Chernozem, Kastanozem

^aMostly formed under forest vegetation.

Source: From Soil Survey Staff,^[3] Baldwin, Kellogg, et al.,^[4] Thorp & Smith,^[20] and Food and Agriculture Organization (FAO) of the United Nations Land and Plant Nutrition Management Service.^[21]

CLASSIFICATION OF GRASSLAND SOILS

The classification of grassland soils with various soil-classification systems is given in Table 3.

CONCLUSION

Throughout the world where grasslands are located, developed cultures exist. These soils are extremely productive and have allowed societies to flourish. About 40% of the grasslands are tall-grass prairies and 60% are short- and mid-grass steppes.^[7] Grasslands have higher organism activity than most other soils.^[9] Carbon sequestration in grasslands is critical to the global warming process. Grasslands are not only important for food and fiber production, but they have the potential to help solve one of the major problems facing human survival, global warming.^[6] Grasslands are one of the most important ecosystems in the world.

REFERENCES

1. Kellogg, C.E. Soil in society. In *Soils and Men, Yearbook of Agriculture*; Wallace, E.A., Ed.; U.S. Department of Agriculture, U.S. Government Printing Office: Washington, 1938; 863–886.
2. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
3. Soil Survey Staff. *Soil Taxonomy*, 2nd Ed.; Agriculture Handbook 436; U.S. Department of Agriculture, Natural Resources Conservation Service, Superintendent of Documents, U.S. Government Printing Office: Washington, 1999.
4. U.S. Department of Agriculture-Natural Resources Conservation Service. *Global Distribution of Mollisols*; Soil Survey Division, World Soil Resources: Washington, 2000.

5. Buol, S.W.; Hole, F.D.; McCracken, R.J.; Southard, R.J. *Soil Genesis and Classification*, 4th Ed.; Iowa State University Press: Ames, 1997; 317–329.
6. U.S. Department of Agriculture-Natural Resources Conservation Service. *Percent of Land Area in Mollisols, Map m4034*; U.S. Department of Agriculture, Natural Resources Conservation Service, National Soil Survey Center: Lincoln, 1999.
7. Brady, N.C.; Weil, R.R. *The Nature and Properties of Soils*, 12th Ed.; Prentice Hall: Upper Saddle River, 1999; 29–116.
8. Barshad, I. Chemistry of soil development. In *Chemistry of the Soil*, 2nd Ed.; Bear, F.E., Ed.; American Chemical Society Monograph 160; Reinhold: New York, 1964; 1–70.
9. Jenny, H. *The Soil Resource*. Ecological Studies 37; Springer: New York, 1980; 337–360.
10. Natural Resources Conservation Service. *Houdek Series, Official Series Description Sheet*; U.S. Department of Agriculture, Natural Resources Conservation Service: Ames, 1997.
11. Soil Survey Division Staff. *Soil Survey Manual*. U.S. Department of Agriculture Handbook 18; U.S. Department of Agriculture, Superintendent of Documents; U.S. Government Printing Office: Washington, 1993; 59–196.
12. Kononova, M.M. Humus of virgin and cultivated soils. In *Soil Components*; Gieseking, J.E., Ed.; Springer: New York, 1975; Vol. 1, 475–526.
13. Douglas, C.L.; Fehrenbacher, J.B.; Ray, B.W. The lower boundary of selected mollisols. *Soil Sci. Soc. Am. Proc.* **1967**, *31*, 795–800.
14. Kononova, M.M. *Soil Organic Matter*; Pergamon Press: Oxford, 1966.
15. Thorp, J. How soils develop under grass. In *Yearbook of Agriculture*; U.S. Department of Agriculture, U.S. Government Printing Office: Washington, 1948; 55–66.
16. Fenton, T.E. Mollisols. In *Pedogenesis and Soil Taxonomy, Part II, The Soil Orders*. Development in Soil Science 11B; Wilding, L.P., Smeck, N.E., Hall, G.F., Eds.; Elsevier: Amsterdam, 1983; 125–163.
17. Westin, F.C.; Buntley, G.J. Soil phosphorus in South Dakota. III. Phosphorus fractions of some borolls and ustolls. *Soil Sci. Soc. Am. Proc.* **1967**, *31*, 521–528.
18. Baldwin, M.; Kellogg, C.E.; Thorp, J. Soil classification. In *Soils and Men: Yearbook of Agriculture*; U.S. Department of Agriculture, U.S. Government Printing Office: Washington, 1938; 979–1001.
19. Thorp, J.; Smith, G.D. Higher categories of soil classification. *Soil Sci.* **1949**, *67*, 117–126.
20. Food and Agriculture Organization (FAO) of the United Nations Land and Plant Nutrition Management Service. *Key to the FAO Soil Units in the FAO/UNESCO Soil Map of the World*; FAO: Rome, 2000.
21. Lal, R.; Kimble, J.M.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Sleeping Bear Press: Chelsea, MI, 1998; 128 pp.

Grazing Lands: Gaseous Emissions

Lipismita Samal

*College of Agriculture, Odisha University of Agriculture and Technology,
Bhawanipatna, India*

Veerasamy Sejian

*Animal Physiology Division, National Institute of Animal Nutrition and Physiology,
Bangalore, India*

M. Bagath

R.U. Suganthi

Raghavendra Bhatta

National Institute of Animal Nutrition and Physiology, Bangalore, India

Rattan Lal

*Carbon Management and Sequestration Center, Ohio State University, Columbus,
Ohio, U.S.A.*

Abstract

Livestock production is responsible for most of the agricultural emissions, with about 22% from enteric fermentation, 10% from managed waste, and 18% from grazed lands. Grazing lands can be both sources and sinks of greenhouse gases (GHGs) as a consequence of vegetation and soil biological processes resulting in the transformation of carbon (C) and nitrogen (N) compounds into GHGs. This entry addresses in detail the factors influencing gaseous emissions, the types of gases produced, and the different management practices to decrease these emissions from grazing lands. Of the GHG emissions from grazed lands, 98% is in the form of nitrous oxide and the remaining 2% is methane. Although grazing lands are considered to be carbon dioxide (CO₂) neutral, they have the potential to store over 100 Tg CO₂ yr⁻¹. Gaseous production from grazing land is sporadic in both time and space, and so it is a challenge to scale up the measurements of gaseous emissions from a given location and time to regional and national levels. The exact approaches also depend on local conditions and therefore may vary from region to region.

INTRODUCTION

Grazing lands can be both sources and sinks of greenhouse gases (GHGs) as a consequence of vegetation and soil biological processes resulting in the transformation of carbon (C) and nitrogen (N) compounds into GHGs. In grazing systems, C and N can be exchanged in many forms among atmosphere, plant, soil, and animal components. Since the 19th century, carbon dioxide (CO₂), nitrous oxide (N₂O), and methane (CH₄) emissions have increased by 36%, 18%, and 148%, respectively.^[1] Although varied among years, the atmospheric concentrations of N₂O and CH₄ have increased by 4.9% and 14.9%, respectively, between 1990 and 2009.^[1] Globally, grazing lands and pasture cover about 3.6 billion ha or 8.9 billion acres, representing 27% of the world land area (13 billion ha) and 70% of the world agricultural area (5.2 billion ha).^[2] Broadly, grazing lands can be of different types, e.g., pasture lands, meadow, and rangelands (namely grasslands, shrublands, and savannas). Grasslands are particularly important sources of N₂O

emissions, which are greater in grazed (pastures) grasslands than in mowed (ungrazed) grasslands. Livestock grazing can increase C and N cycling by consuming crop and weed residues and returning C and N inputs into the soil through manure (feces and urine).

Livestock production is responsible for most of the agricultural emissions, with about 22% from enteric fermentation, 10% from managed waste, and 18% from grazed lands.^[3] Livestock manure is unmanaged when it is deposited directly on grazed lands. Alternatively, livestock manure can be managed in storage and treatment systems. Of the emissions from grazed lands, 98% is in the form of N₂O. The remaining 2% of GHG emissions from grazed lands is CH₄.^[3] N₂O, whose emissions into the atmosphere are influenced by the management of croplands and grazing lands, has a lifetime of approximately 120 years and contributes 6.24% to the overall global radiative forcing.^[4] There exists a positive correlation between CO₂ and N₂O, i.e., similarity in the rates of gas accumulation and diffusive flux due to soil moisture.^[5] By employing proper

management techniques, grazing lands can both sequester C and reduce emissions of CO_2 , CH_4 , and N_2O , thereby reducing their GHG footprint. This entry addresses the types of gases produced from grazing lands, the factors influencing these gaseous emissions, and the different mitigation strategies to reduce these losses.

DIFFERENT GASES PRODUCED FROM GRAZING LANDS

Nitrous Oxide

Grazed lands emit N_2O due to enhanced N cycling. Globally, livestock grazing is estimated to contribute about 1.55 Tg of N_2O -N flux yr^{-1} .^[5] N loss through volatilization during cattle grazing is ~ 20 Gg N. Unlike the case of CO_2 and CH_4 , there are no significant biological sinks of atmospheric N_2O . N_2O emission is predominantly from grazing lands amended with N-rich inputs such as synthetic and organic fertilizers, sewage sludge, compost, unharvested crop residues (both aboveground and below-ground), and manure.^[4] Grazing in grasslands reduces N uptake by plants, leading to the temporary accumulation of mineral forms of N in the soil. Increase in mineralization of soil organic matter (SOM) may release inorganic N in the soil, thereby stimulating microbial activity that

produces N_2O .^[5] A portion of the N cycled within the plant–animal–soil system volatilizes to the atmosphere in various gaseous forms and is eventually redeposited onto the soils where it can contribute to indirect N_2O emissions. N_2O gas is produced near the soil surface and is accompanied by a net increase in soil nitrate (NO_3^-). Some N in the form of NO_3^- can leach into groundwater and surface runoff, undergo denitrification, and also contribute to indirect emissions.^[3]

Mechanism of N_2O production

N_2O is produced in soil by the microbial action during autotrophic aerobic nitrification and subsequent heterotrophic anaerobic denitrification processes, which are affected by soil and climatic factors such as composition and availability of organic matter (OM) as a source of energy, decomposable organic C, N substrate supply or soil NO_3^- concentration, partial pressure of oxygen (O_2), O_2 supply, temperature, soil pH, soil enzyme activities, rainfall/irrigation, water-filled pore space (WFPS), and salinity. N_2O is a by-product of the first step of nitrification, whereas it is an intermediate or end product in denitrification. Although N_2O production by nitrification is possible, the N_2O emission peaks in grazed pastures are generally attributed to the denitrification process. Fig. 1 shows the N cycle for grazing lands.

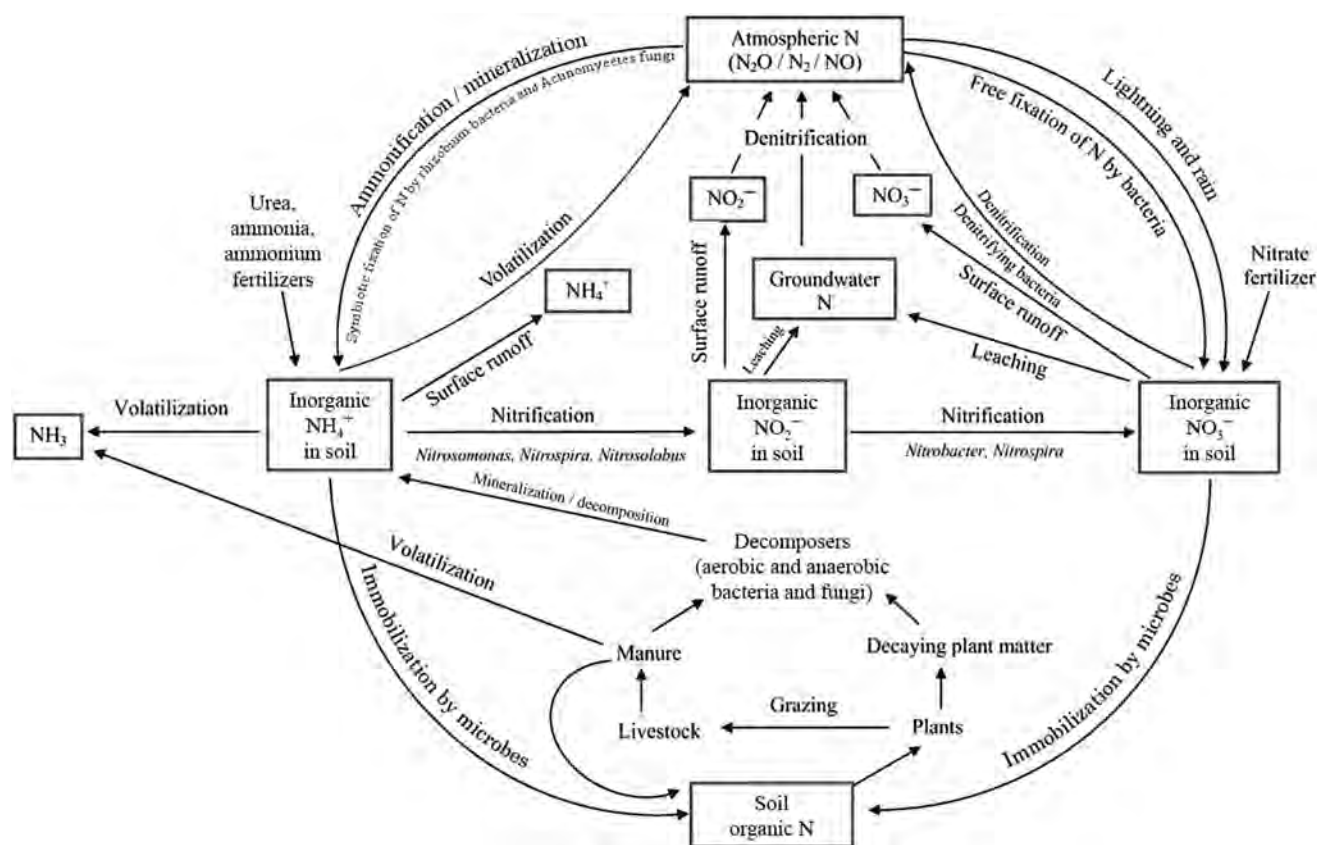


Fig. 1 N cycle for grazing land.

Nitrification. It is a two-step process in which ammonium ion (NH_4^+) is oxidized to nitrite ion (NO_2^-) by two groups of nitrifiers, i.e., ammonia (NH_3)-oxidizing bacteria (*Nitrosomonas* spp., *Nitrospira* spp., and *Nitrosolobus* spp.) and NH_3 -oxidizing archaea (*Nitrosopumilus maritimus* and *Nitrososphaera viennensis*). During the second step, NO_2^- is oxidized by nitrite-oxidizing bacteria (*Nitrobacter* spp. and *Nitrospira* spp.) to NO_3^- .

Denitrification. It represents sequential dissimilative microbial reduction of NO_3^- via NO_2^- to gaseous nitric oxide (NO), N_2O , and finally molecular dinitrogen (N_2) via four enzymatic complexes. It is the last step in the global N cycle by which fixed N is returned back to the atmosphere. Although denitrifying organisms normally respire and metabolize aerobically when O_2 is present, a lower partial pressure of O_2 causes them to use oxidized forms of N as alternative terminal electron acceptor in their respiration chain. Denitrification is the only biological process of N_2O consumption; however, this process usually produces more N_2O than it consumes because the denitrification sequence is usually incomplete, i.e., not all of the NO_3^- is reduced all the way to N_2 , resulting in two main denitrification products: N_2O and N_2 . This process is usually carried out by heterotrophic denitrifiers, e.g., Thermoproteaceae, Cytophagaceae, Corynebacteriaceae, Streptomycineae, Bacillaceae, Rhodospirillaceae, Rhodobacteraceae, Rhizobiaceae, Burkholderiaceae, Nitrosomonadaceae, Neisseriaceae, and Pseudomonaceae, which can reduce NO_2^- into N_2O or N_2 . Denitrification can also be contributed by several autotrophic (*Nitrosomonas europaea*) and heterotrophic (*Pseudomonas denitrificans*) nitrifiers.

Other Nitrogenous Emissions

After the addition of N to farms as animal excreta, fertilizer, crop residues, or biological fixation, it can be lost by gaseous emissions to the atmosphere as NH_3 , NO, N_2 , and other NO_x besides N_2O , by runoff or leaching of NO_3^- and by soil erosion. In well-aerated soils, NO emissions are several times larger than N_2O emissions. The emission of NO is the principal gaseous N loss mechanism during nitrification in soil.

Methane

Manure deposited on grazed lands (i.e., unmanaged manure) produces and emits CH_4 . CH_4 emissions from this source are relatively small, less than 3% of total grazed land GHG emissions, because grazed lands do not often experience the anaerobic conditions required for CH_4 production to exceed CH_4 uptake.^[3]

Carbon Dioxide

Although grazing lands are considered to be CO_2 neutral, with a small uptake of 0.2 Tg CO_2 eq. through sequestration

of CO_2 in soil organic carbon (SOC),^[3] these lands can be emission sources or net sinks for atmospheric CO_2 depending on whether C inputs to the soil from plant residues and manure exceed C losses from the decomposition of SOM. Grazed lands have the potential to store over 100 Tg CO_2 yr^{-1} .^[6] Permanent grasslands used as pastures, rangelands, and hayfields can maintain large soil C stocks due to several characteristics. Perennial grasses allocate a high proportion of photosynthetically fixed C belowground, which is sequestered in the soil as OM after the residue is returned to soil, maintain plant cover year-round, and promote the formation of stable soil aggregates. CO_2 is emitted from soils to the atmosphere through the decomposition of dead OM and respiration by roots and mycorrhizal fungi. Soil C level is determined by the balance between the amount of plant residue C added to the soil and the rate of C mineralized as CO_2 emissions in unmanured soil. Plant residues also vary in their inherent decomposability due to differences in their chemical characteristics. Alterations in plant residues also alter the soil microbial community, e.g., an increase in plant lignin content favors soil fungi and thus leads to enhanced soil C storage. Fig. 2 shows the C cycle for grazing land.

FACTORS AFFECTING GHG EMISSIONS FROM GRAZING LAND

Abiotic Factors

These include soil temperature, water content, pH, texture, weather patterns, and various farm management practices. Soil fluxes of GHGs are also affected by other factors including air temperature, precipitation, net primary productivity, soil respiration rates, substrate quality, soil physical and chemical properties, and rates of decomposition of OM.

Soil temperature

It is the main environmental variable controlling CO_2 emissions, while CH_4 is controlled primarily by soil wetness and anaerobiosis. N_2O emissions respond to both changes in soil temperature and wetness. The temperature effect is normally expressed as Q_{10} value, which corresponds to the increase of reaction rate due to the increase in temperature by 10°C . For N_2O emission, values vary between 2 and 10. Soil temperatures in rangelands vary considerably not only between seasons but also diurnally. The N_2O emissions show a diurnal cycle, varying with the temperature of surface soil. Peak N_2O emission rates occur in the afternoon and are minimum near sunrise. Temperature influences enzyme kinetics and metabolic turnover rates of nitrifiers and denitrifiers with optimum temperature at $30\text{--}35^\circ\text{C}$. Microbial activity, OM mineralization, urea hydrolysis, and nitrification are higher at higher temperature and higher soil

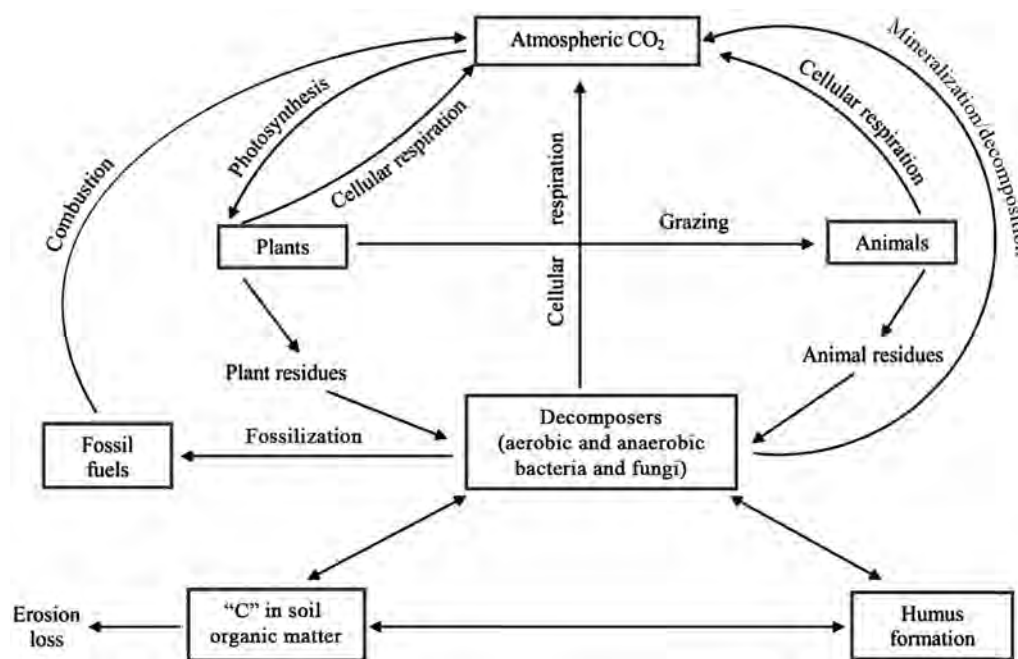


Fig. 2 C cycle for grazing land.

water content, thereby resulting in higher CO₂ and N₂O emissions.

Soil moisture

Soil moisture or WFPS effect is mostly indirect, as it influences the rate of O₂ diffusion into the soil and controls the intensity of aeration, thus determining the degree of anaerobiosis. The interaction between moisture and aeration in the soil matrix influences NO₃⁻ formation and its stability. The nitrification process occurs with WFPS at 35–60%, and NH₄⁺ should be available in the soil. At levels higher than 60% of WFPS, denitrification is the predominant source of N₂O emissions, due to N mineralization and hindered diffusion of O₂ into the soil, favoring the formation of anaerobic environments. N₂O emissions often increase with increasing aeration during drainage of anaerobic soils. With the soil moist to less than field capacity (60% WFPS), average daily emission can range from 6 to 25 g N ha⁻¹ day⁻¹.^[7] The magnitude of N₂O emissions may increase about 12-fold from pasture soils when WFPS increases from 60% to 80%.^[8]

Soil pH

Soil pH influences the rate of denitrification and the ratio of denitrification products, N₂O and N₂, and subsequent N gas emissions influence not only directly through the kinetics of denitrification reductases but also indirectly through its impact on composition and abundance of the denitrifier community.^[9] Under acidic pH, the activity of N₂O reductase is

lowered, and the synthesis of new N₂O reductase is inhibited, resulting in an increased accumulation of N₂O.^[9] In soils with higher pH values, the activity and synthesis of N₂O reductase are supported, and the N₂O/(N₂O + N₂) ratio is decreased and, therefore, more N₂ is produced.^[9]

Soil texture

CO₂ emission may increase with decreasing clay content due to increased SOM mineralization. Soil texture also impacts CH₄ and N₂O fluxes. Soil structure and texture affect hydrological properties and thus the leaching behavior of soils and the mineralization rate.

Weather patterns

Peak losses of N may occur in early spring and autumn in the Mediterranean environment and lower N losses in summer due to low soil water contents. However, in subtropical and tropical environments, large N losses may also occur in summer. Substantial N₂O emissions may occur in soils after significant rainfall events, snow melts, and also during freeze-thaw or spring-thaw events. Even though the soil temperature may be near 0°C, the N₂O emission in cold soils is caused by microbial activity, and the production of N₂O exceeds its reduction to gaseous N₂ at low temperatures.^[10] The strategic application of dairy effluent during dry summer and autumn can significantly reduce N₂O emissions from grazed pastures, and delaying effluent irrigation after a grazing event can reduce emissions by decreasing the levels of surplus mineral N.

Farm management practices

Management practices also indirectly affect GHG emissions by altering soil temperature and moisture regimes. In the short term, drainage and fertilization accelerate GHG emissions significantly although their long-term effects are likely moderated by C accumulation in the aboveground and belowground biomass.

Drainage. Machinery used for drainage may also affect GHG fluxes by compacting the soil and thereby altering properties such as bulk density, air-filled pore space, and hydraulic conductivity. Drainage increases soil temperature, improves aeration by lowering water table depth, and increases the production of highly decomposable fine roots and exudates, thereby stimulating soil microbial activity and enhancing OM decomposition, which increases CO_2 and N_2O and decreases CH_4 fluxes.^[11] In nutrient-rich peat soils, large fluxes of N_2O occur after drainage when increased O_2 availability stimulates nitrification.^[11] The low fluxes of CH_4 from the drained peatlands may be attributed to decreased methanogenesis and/or increased oxidation of CH_4 by oxidizing microbes in the aerobic part of the peat, with part of the CH_4 oxidized in the aerobic peat layers and converted to atmospheric CO_2 . The effect of drainage may persist and may not decelerate the rate of SOM decomposition for a long time.

Irrigation. It moderates soil water content and reduces soil temperature fluctuations compared to no irrigation, but N fertilization can reduce these parameters compared to no N fertilization by increasing canopy cover and water uptake through increased biomass production. Higher N losses are expected from flood-irrigated pastures.^[7] The addition of, as little as, 5 mm of water to a grass sward can increase N_2O emission rates markedly.^[7]

Mounding. Mounding or raised beds may modify the soil microbial processes and increase both CO_2 and CH_4 fluxes.^[11] Mounding creates three subsites (mounds, hollows, and undisturbed ground) with a different microclimate and OM distribution. Mounds increase soil temperature, decrease soil moisture content, and accentuate OM decomposition. N_2O fluxes may increase in mounded hollows and the top of mounds than in undisturbed grounds, probably because soil moisture conditions, temperature, and N availability favor nitrification and denitrification. The two processes may occur simultaneously at shallow depths and result in the gaseous loss of N_2O and N_2 from the soil mineral N pool.^[11] N_2O consumption may take place in waterlogged conditions when denitrifiers reduce N_2O to N_2 . N_2O fluxes may be reduced in both the mounds and the hollows.^[11]

Fertilization. N fertilization typically has a stimulatory effect on N_2O and CH_4 emissions but a variable effect on CO_2 emissions. The extensive application of N fertilizer also increases leaching of N. Both fertilized and unfertilized soils emit N_2O . While fertilizer N is a source of N_2O in case of fertilized soils, the mineralization of SOM contributes to the production of N_2O from unfertilized soils. Perennial grassland generally maintains higher levels of SOM compared to those rotated cropland.^[12] N_2O escapes from soils especially after the applications of inorganic N fertilizer (e.g., urea and mono-ammonium phosphate) for crops and pastures of mixed farming operations. Unlike crops, most pastures have an active extensive root system throughout the growing season. N_2O emission after the application of anhydrous NH_3 can be 2 to 4 times higher than surface applying urea, ammonium NO_3^- , or broadcasting urea. The surface application of fertilizer and manure is subject to greater volatilization losses than injected fertilizer, predominantly as NH_3 , which is eventually deposited downwind in environments where it may result in N_2O emissions. More than 50% of the fertilizer N can be lost as N_2O through the process of denitrification or nitrification (depending on soil conditions), to produce soluble NO_3^- , which may be removed through leaching and runoff.^[13] Fertilizing with ammonium fertilizers, such as urea, increases the potential for NH_3 emission, but under anaerobic flooded soil, it could minimize gaseous N emissions through denitrification.^[14] Fertilization may stimulate the N-limited decomposing microorganisms. In addition, it also increases the aboveground standing plant biomass, root biomass, which increases rhizospheric deposition and autotrophic respiration. It is the dominant practice affecting GHG emissions in the short term, and its effects appear short lived and of small magnitude. The effect of N fertilizer on increase in CH_4 emissions has been attributed to NH_4^+ ions inhibiting CH_4 oxidation by competitive inhibition of the mono-oxygenase enzyme and also inhibitory effects of N on soil methanotrophs.^[11]

Biotic Factors

Species

In the United States, beef cattle are responsible for the highest proportion of direct N_2O emissions from grazed lands because the vast majority of grazed lands are used for beef production.^[3] In general, emissions from managed grazed land are about twice those of managed manure.^[3] This is due to large numbers of beef cattle on grazing land (more than 80% of all cattle) compared to feedlots, which are a source of managed waste.^[3] The distribution of manure by animals during grazing at the soil surface is uneven; however, distribution can be more uniform with sheep than with cattle grazing.^[15] Grazing sheep and cattle together increases soil bulk density, OM, and grass yields compared with grazing sheep or cattle alone.

Manure

The deposition of animal excreta on pastures, including manure and effluents in grazing lands, results in a patchy distribution of N and C and consequently enhanced biogenic transformations of the nutrients.

Uneven deposition of excretal N by grazing animals can result in hot spots equivalent to an application of 400–2000 kg N ha⁻¹ yr⁻¹ in the small affected area leading to wide spatial and magnitude variations in N₂O emissions. These hot spots correspond to 14.5% ha⁻¹ yr⁻¹, considering a patch area of 0.4 m² per urine deposition.^[16] Emission factor may be different for urine and fecal depositions. Majority of CH₄ comes from feces, and on average 0.96 and 0.03 g CH₄ day⁻¹ cow⁻¹ are emitted from feces and urine, respectively, in grazing cattle,^[17] whereas N₂O emissions are more from the urine, which contains higher concentrations of N and also a more readily available mineral-N form than the feces. The amount of N lost from urine patches depends on the N load deposited, the time of year when urine is applied, and the soil type. N₂O emissions after urine application are higher in soils with high moisture and high temperature. The addition of urine causes the release of SOC by hydrolyzing OM, leading to increased N₂O production when O₂ diffusion is restricted. Cattle urine N concentration is typically 6–7 g N/L when the cattle are grazing pasture while urine-affected, compacted winter forage soils can result in relatively large N₂O fluxes. Winter grazing of *Brassica* crops such as swedes and kale during high soil moisture conditions leads to urine being excreted onto wet, compacted soils, which is likely to result in significant N₂O emissions. N₂O emission factor for cattle urine is 0.20% of the applied urine N in Brazil and 0.66% in the United Kingdom.^[16] Intergovernmental Panel on Climate Change lists a default N₂O emission factor from cattle urine of 2%, i.e., 2% of the urine N is assumed to be emitted as N₂O.^[1] In general, 0.1–3.8% of urine N and 0.1–0.7% of fecal N are emitted to the atmosphere as N₂O.^[16]

Grazing type and intensity

The quantity and form of N lost from grazing lands are mainly a result of grazing intensity, which affects the spatial distribution of urine patches. The intensity and timing of grazing can also influence the growth, C allocation, and plant species of grasslands, thereby affecting the amount of soil C. Increased soil C on optimally grazed lands is often greater than on ungrazed or overgrazed lands. Overgrazing and continuous trampling decrease the efficiency and viability of the pasture systems. There is a significant difference in CH₄ emission rates depending on whether pastures are rotationally or permanently grazed, and a 9% lower CH₄ production ha⁻¹ day⁻¹ was measured in rotationally grazed pastures compared to continuous grazing.^[18] Furthermore, the use of pasture diminishes emissions from manure storage (because there is less

manure in storage systems) but increases N₂O emissions from deposition by grazing animals.

Rangelands

Although the rates of N₂O emissions from most rangelands may be low, considering the extent of total rangeland area, it is important to measure N₂O emissions in the field over an extended period, as small amount of rainfall may trigger significant nitrification activity and hence N₂O production following a prolonged dry period. The N₂O emissions from the urine patches may be significant in rangelands. Usually animal feces are voided in large areas, thus avoiding anaerobic N mineralization from animal waste.^[7] In extensive grazing system, large areas of rangelands are used for grazing purpose. In view of increasing the pasture availability, tree clearing is commonly practiced that may result in increased soluble C even in deeper layers of the rangelands.^[7] These are associated with CO₂ concentration due to microbial respiration and hence intense O₂ demand, resulting in partial anaerobiosis.

Leguminous pastures

In intensive grazing system, most of the N₂ supply for dairy pastures is derived from pasture legumes, which are typically seeded in heavily grazed pastures. Forage legumes on grazed lands contribute to N₂O emissions, because legumes fix N₂ from the atmosphere into forms that can be used by plants and soil microbes, N₂ can become mineralized in the soil and contribute to nitrification and denitrification. Thus, more N₂ added to soil yields more N₂O released to the atmosphere. N₂O emissions from legume-based pastures may be in the order of 1–2 kg N ha⁻¹.^[7] There is a greater N₂O emission with legumes than with non legumes due to the presence of rhizobium bacteria related to root nodules. Because of lower C/N ratio and higher N concentration, legumes decompose rapidly compared to non legumes, thereby increasing N₂O emissions.^[15]

Burning of savannas

In tropical savannas, where fire is frequently used as a pasture management tool, the N₂O and NO production rates are much higher. In such a system, where N is mineralized rapidly after fire, N₂O emission rate could be higher, especially if fire is followed by moderate rainfall events.^[7] GHG emissions can be significantly reduced following a savanna fire early in the dry season because the moisture content of plant material is still relatively high resulting in low-intensity fires that can reduce fuel loads without killing trees.^[19]

MITIGATION STRATEGIES

A mitigation practice may affect more than one gas, by more than one mechanism, e.g., strategies intended to

sequester C in soils may impact the fluxes of N_2O and CH_4 .^[3] Furthermore, the temporal pattern of influence may vary among practices or among gases for a specified practice. One strategy that may be feasible for more intensely managed pastures is the use of nitrification inhibitors (NIs).

Nitrification Inhibitors

The application of NI to the soil can minimize N_2O emission by 30–80%.^[20] NI can reduce NO_3^- leaching and also promote better N utilization in the soil. It temporarily delays the bacterial oxidation of NH_4^+ to NO_2^- by inhibiting *Nitrosomonas* spp. bacteria.^[16] An ideal NI should specifically block NH_4^+ oxidation to NO_2^- but not NO_2^- oxidation to NO_3^- . However, a NI does not inhibit nitrification indefinitely, and its effect depends on its rate of degradation and persistence in soils. Examples of NI are dicyandiamide, piadin, acetylene, nitrapyrin (2-chloro-6-(trichloromethyl)-pyridine), and 3,4-dimethylpyrazole phosphate. However, it is difficult to apply NI on commercial pastoral farms because there is no technology available to precisely apply only on urine patches and the application over the whole farm is not economically efficient.^[16]

Biochar/Agrichar

It is a promising technology to boost the SOC besides producing bioenergy. The process involves pyrolysis of forest or agricultural biomass to form a charcoal product known as biochar. Under proper conditions, 30–50% of the biomass C is retained in the porous biochar structure. The biochar is highly recalcitrant in soils and can sequester C for thousands of years. The application of biochar also reduces N_2O emissions from urine patches.^[12]

Grazing Management Practices

Grazing intensity

Any practice that increases the photosynthetic input of C or slows the return of stored C through respiration will increase stored C, thereby enhancing SOC or building C sinks.^[14] So, light grazing should be practiced instead of heavy grazing. Optimal grazing management can enhance the accrual of soil C.

Controlled grazing

Grazing management may affect N_2O emissions from pasture, e.g., in Inner Mongolia, increasing stocking rates of sheep reduced N_2O emissions compared with those from ungrazed pasture. Reducing the sheep stock number and spring lamb production appears to be an option for reducing N_2O emissions from Australian grazing lands.^[7] Increasing the number of grazing animals and grazing in the late

fall/winter and in the early spring will ensure that the manure will be spread over the field. Another method of grazing called swath grazing may also be followed. This practice spreads manure around the field and will also decrease the feeding costs.

Rotational grazing

Grassland practices such as rotational grazing can reduce C losses and promote C sequestration in grazing lands.^[3] By subdividing pastures and rotating cattle, producers can manage stocking densities and grazing duration to allow the vegetation time for rest and recovery.

Soil Conservation Practices

These practices may sequester soil C and enhance CH_4 consumption. The largest potential is decreasing soil erosion and restoring eroded and degraded soils, so that they become net C sinks. Grassland systems that have been degraded in the past or maintained under suboptimal management conditions are most conducive to sequestering additional C. Soils that have been historically cropped using conventional tillage are often depleted of C because tillage disturbs soil aggregation and warms soil, whereas both increase decomposition rates. C-depleted soils can act as CO_2 sinks upon conversion to grazing because grazed soils are typically not ploughed. Although grazed mineral soils are a net sink of CO_2 , grazed organic soils are a net source. If half of the grazed organic soils were converted back to wetlands, CO_2 emissions from this source could be reduced from approximately 4.6–2.3 Tg $\text{CO}_2\text{e yr}^{-1}$.^[3]

CONCLUSION

The production of GHGs from grazing land is sporadic in both time and space, and so it is a challenge to scale up the measurements of GHG emissions from a given location and time to regional and national levels. The exact approaches also depend on local conditions and, therefore, vary from region to region. A national management strategy is needed to reflect region-specific soil types and climatic conditions. Regionally appropriate conservation agricultural practices should be identified to build up SOM, promote SOC, and mitigate GHG emissions. However, there is large uncertainty in the emission factor, and different countries and regions must have different specific gaseous emission factors due to different soil, temperature, rainfall, and grazing systems. It is also difficult to obtain definitive gas flux values because of the analytical techniques, large spatial variability, and seasonal variability. Simulation modeling should complement these measurements over seasons and locations.

FUTURE PERSPECTIVES

Although GHG emissions derived from soil have been researched for 50–60 years, there are geographic regions and grazing land systems that have not been well characterized. Forage quality should be improved for grazing animals on smaller livestock operations through better pasture management. The effectiveness of NI should be assessed under different conditions and for different soil types. Additional incentives and educational programs are needed for land managers to adopt improved grazing management. More research is needed on the potential use of biochar to reduce GHG emissions from grazing systems. Improved methodologies should be developed to improve the accuracy of determining changes in SOC and GHG emissions from grazing lands. Comprehensive research program along with a whole system analysis is needed to account for the interactions to arrive at cost-effective and efficient GHG mitigation management, policy, and legislation options for sustainable management of grazing lands.

REFERENCES

- Intergovernmental Panel on Climate Change (IPCC). Climate Change 2007: The physical science basis. Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change. In *IPCC Fourth Assessment Report (AR4)*; Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L., Eds.; Cambridge University Press: Cambridge, 2007; 996 pp.
- International Academy of Ecology and Environmental Sciences (IAEES). *Land Uses*, 2015. Available at <http://www.iaees.org/enviromdata/enframe.asp?xuhao=2,2009-2015> (accessed August 2008).
- U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990–2005. *Global Change Program Office, Office of the Chief Economist, U.S. Department of Agriculture. Technical Bulletin No. 1921*, 2008; 161. Available at http://www.usda.gov/oce/climate_change/AFGG_Inventory/USDA_GHG_Inventory.pdf.
- Intergovernmental Panel on Climate Change (IPCC). Climate change 2013: The physical science basis. In *Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*; Stocker, T.F., Qin, D., Plattner, G., Tignor, M.M.B., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M., Eds.; Cambridge University Press: Cambridge, 2013.
- Sey, B.K.; Whalen, J.K.; Gregorich, E.G.; Rochette, P.; Cue, R.I. Carbon dioxide and nitrous oxide content in soils under corn and soybean. *Soil Sci. Soc. Am. J.* **2008**, *72*, 931–938.
- Follett, R.L.; Kimble, J.M.; Lal, R. The potential of U.S. grazing lands to sequester soil carbon. In *The Potential of U.S. Grazing Lands to Sequester Soil Carbon and Mitigate the Greenhouse Effect*; Follett, R.L., Kimble, J.M., Eds.; CRC Press: Boca Raton, 2001; 1–442.
- Dalal, R.C.; Wang, W.; Robertson, G.P.; Parton, W.J. Nitrous oxide emission from Australian agricultural land and mitigation options: A review. *Aust. J. Soil Res.* **2003**, *41*, 165–195.
- Maljanen, M.; Komulainen, V.-M.; Hytönen, J.; Martikainen, P.J.; Laine, J. Carbon dioxide, nitrous oxide and methane dynamics in boreal organic agricultural soils with different soil characteristics. *Soil Biol. Biochem.* **2004**, *36*, 1801–1808.
- Čuhel, M.J. *The Linkage between Denitrification Activity, N Gas Emissions, and The Size of The Denitrifier Community in Pasture Soils*; Ph.D. Thesis, Series No. 1.; Faculty of Science, University of South Bohemia: České Budějovice, 2011; 78 pp.
- Holtan-Hartwig, L.; Dorsch, P.; Bakken, L.R. Low temperature control of soil denitrifying communities: Kinetics of N₂O production and reduction. *Soil Biol. Biochem.* **2002**, *34*, 1797–1806.
- Mojeremane, W.; Rees, R.M.; Mencuccini, M. The effects of site preparation practices on carbon dioxide, methane and nitrous oxide fluxes from a peaty gley soil. *Forestry* **2012**, *85*, 1–15.
- Lal, R.; Delgado, J.A.; Groffman, P.M.; Millar, N.; Dell, C.; Rotz, A. Management to mitigate and adapt to climate change. *J. Soil Water Conserv.* **2011**, *66*, 276–285.
- Ffoulkes, D. *Factsheet 2: Sources of Greenhouse Gas Emissions from NT Pastoral Industry, Northern Territory Government*, 2008; 1–2. Available at http://www.nt.gov.au/d/Content/File/p/Climate_Change/CCFS02.pdf.
- Adhya, T.K.; Sharma, P.D.; Gogoi, A.K. Mitigating greenhouse gas emission from agriculture. In *Climate Change and Crops, Environmental Science and Engineering*, Chapter 15; Singh, S.N., Ed.; Springer-Verlag: Berlin, Heidelberg, 2009; 329–344.
- Barsotti, J.L. *Soil Carbon and Nitrogen and Greenhouse Gas Emissions Affected by Sheep Grazing under Dryland Cropping Systems*; M.Sc. Thesis; Montana State University: Bozeman, 2012; 74 pp.
- Barneze, A.S. *N₂O Emission from Soil Due to Urine Deposition by Grazing Cattle and Potential Mitigation*; M.S. Dissertation, Centro de Energia Nuclear na Agricultura, Universidade de Sao Paulo: Piracicaba, 2013; 87 pp.
- Kebreab, E.; Clark, K.; Wagner-Riddle, C.; France, J. Methane and nitrous oxide emissions from Canadian animal agriculture: A review. *Can. J. Anim. Sci.* **2006**, *86*, 135–157.
- Lengers, B. *Up to Date Relevant GHG Abatement Options in German Agricultural Dairy Production Systems*, Technical Paper 2012, Institute for Food and Resource Economics, University of Bonn, 2012; 1–23.
- Ffoulkes, D. *Factsheet 3: Reducing Greenhouse Gas Emissions from the NT Pastoral Industry*, Northern Territory Government, 2008; 1–2. Available at http://www.nt.gov.au/d/Content/File/p/Climate_Change/CCFS03.pdf.
- Akiyama, H.; Yan, X.; Yagi, K. Evaluation of effectiveness of enhanced-efficiency fertilizers as mitigation options for N₂O and NO emissions from agricultural soils: Meta analysis. *Glob. Change Biol.* **2010**, *16*, 1837–1846.

Green Revolution: China North Eastern Plains

Xiangbin Kong

College of Resources and Environmental Science, China Agricultural University
and Key Laboratory of Farmland Quality, Monitoring and Control, National
Ministry of Land Resources, Beijing, China

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus,
Ohio, U.S.A.

Abstract

Major challenges in sustainable land use ecosystems have emerged since the 1980s in the China's North Eastern Plain (CNEP) because of increased competition for land, groundwater, labor, and other natural resources driven by growing food demand, urbanization, and industrialization. Food security has been ensured at the cost of weakening or loss of other land use ecosystem functions and services such as carbon (C) sequestration, groundwater purification, land use diversity, local tradition and land use knowledge, open space, and so on. Land expansion and land use intensification have led to high rates of nitrogen (N), phosphorus (P), and potassium (K) use since the 1960s averaging 1.66, 4.43, and 1.46 times the national average rates for China, respectively. Further, intensive irrigation has caused rapid groundwater depletion. Consequently, CNEP is China's bread basket accounting for almost 35% of maize (*Zea mays*) and 70% of wheat (*Triticum aestivum*) production. For achieving sustainable land use systems and restoring multi-function services, a series of green revolutions must be realized. These green revolutions must limit the conversion of natural land into managed land use systems, restrict the urban encroachment, restore soil quality of intensively used soils, map the permanent prime land and ecological hot spots, and narrow the yield gap on arable land. Living Common Community Management Units (LCCMUs) including arable land, wetland, forest land, grass land, and traditional land use and cropping knowledge should be built through an integrated application of top-down method but with the bottom-up approach. Closing potential productivity gap while reducing resources-intensive use should be ensured by improved arable land quality management based on the LCCMU. Sustainable intensification including agronomic yield, with due consideration of spatial and temporal variability at a regional level, can be achieved by application of precision agricultural technology and investment policy in rural land.

CHALLENGES

China's North Eastern Plain (CNEP) is an important food-producing region. It involves the double-cropping system of winter wheat (*Triticum aestivum*) and summer maize (*Zea mays*). The CNEP produces 35% of China's maize and 70% of wheat. The average yield of grain crops (kg ha^{-1}) has increased from 1582 to 5860 for wheat and 4492 to 5610 for maize between 1985 and 2010.^[1] The yields are 1.25 and 1.07 times those of China's national average yields of wheat and maize, respectively. However, CNEP accounts for about 16% of China's arable land but has access to only 7.6% of China's renewable water resources compared with the highly endowed southern plains. Therefore, the dramatic increase in crop yields and production in the CNEP has been achieved by the expansion of arable land, intensive irrigation, and high rates of nitrogen (N), phosphorus (P), and potassium (K) use since

1960 averaging 1.66, 4.43, and 1.46 times those of the national average rates, respectively. The conversion of arable land to built-up land and of natural ecosystems to arable land, as well as the intensive use of chemical fertilizers, has made CNEP a major source of greenhouse gases (GHGs). On the other hand, the utilization of approximately 900 mm of water for annual production of wheat and maize^[2] has caused severe groundwater depletion. Both west–east and south–north transects chosen to conduct the buffer analysis reveal the existence of four large cones of shallow groundwater depression in Mancheng ($\sim 1.50 \text{ m yr}^{-1}$), Shijiazhuang ($\sim 2.00 \text{ m yr}^{-1}$), Xingtai ($\sim 1.75 \text{ m yr}^{-1}$) in Hebei Province, and Anyang ($\sim 1.25 \text{ m yr}^{-1}$) in Henan Province. Further, three large cones of deep groundwater depression also exist in Shijiazhuang ($\sim 2.25 \text{ m yr}^{-1}$) and Hengshui ($\sim 2.00 \text{ m yr}^{-1}$) in Hebei Province and Dezhou ($\sim 3.00 \text{ m yr}^{-1}$) in Shandong Province.

There is unknown solution to sustainable land use ecosystems, except through the realization of soil-based green revolutions in this region. There exists high competition driven by land, water, labor, energy, and other natural resources for continuous increase in food demand, rapid urbanization and industrialization, and multifunction services by local communities. Thus, soil-based green revolutions are needed to increase crop production from available cultivated land in ways that do not jeopardize the environment nor undermine the productive capacity of land and water resources.

DECREASE IN ECOSYSTEMS MULTIFUNCTION AND SERVICES SINCE THE 1980S

The land ecosystems provide essential ecosystem functions and services such as the net primary productivity (NPP), renewal of freshwater, regulation of climate and biogeochemical cycles, maintenance of soil fertility, preservation of habitat for biological diversity, and well-being of household (Fig. 1). However, it is the continuous increase in agricultural intensification since the 1980s that has advanced crop production quickly and has transferred the historic grain production over centuries from the south to the north. Unfortunately, agricultural intensification in conjunction with the high rate of application of chemical fertilizers and the irrigation from groundwater have created the vicious cycle of higher wheat yields, more water depletion, and more severe deficit of groundwater.

Simultaneously, the continuous conversion of natural land systems (wetlands, coastal deposits, grass lands, etc.) to arable land for growing crops and arable land to build-up land for city, rural settlements, and factories have greatly altered ecosystem functions and services. The loss of natural ecosystems has exacerbated GHG emissions, decreased land use and crop biodiversity, reduced surface water, and increased outward migration of rural households.

Both arable land expansion and agricultural intensification have advanced crop production. However, the land ecosystem multifunctions have been weakened, destroyed, or lost at an accelerated rate. Thus, the land ecosystem functions and services should be restored into land use zoning on the basis of the spatial land ecosystem services. The marginal arable land converted from the natural ecosystems should be restored, prime farmland should be marked permanent, and urban encroachment should be zoned. The comprehensive and multifunction services should be rehabilitated by means of top-down method, but with the bottom-up approach based on Living Common Community Management Unit (LCCMU) of arable lands, wetlands, forest lands, grass lands, traditional lands, and cropping knowledge. The strategy is to close the arable land potential productivity gap and reduce input of chemical fertilizers. The goal is to improve arable land quality through sustainable intensification of agroecosystems.

THE WAY FORWARD

Restoration of Land Ecosystems Multifunction Services

There is a trilemma of food security, ecosystems multifunctions, and city development in the CNEP (Fig. 2). The city growth boundary, the permanent prime farmland, and the ecological hotspot must be zoned through the strategy of preserving land ecosystem multifunctions and services.

Restore Excessive Marginal Arable Land Expansion

Sustaining land ecosystem multifunctions and services must be a high priority. Natural ecosystems (e.g., rivers, ponds, grasslands, wetlands, and other natural land use systems) were in abundance prior to the 1980s, which

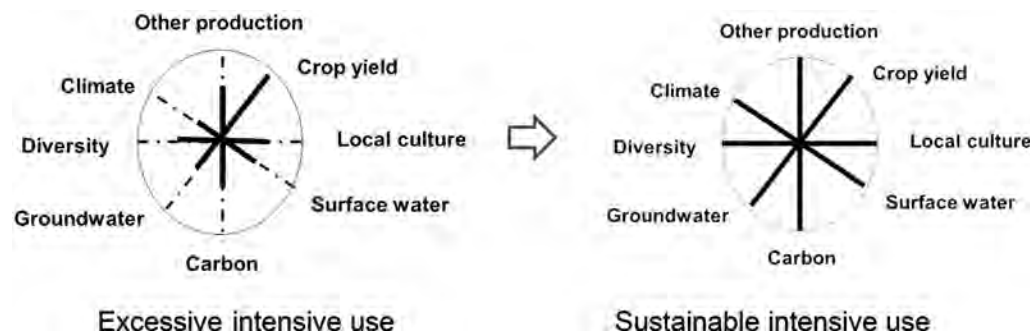


Fig. 1 Conceptual framework for intensifying land use and ecosystem multifunction in the China north plains. The provisioning of multifunction under different land use regimes can be illustrated with these diagrams, in which the condition of each ecosystem service is indicated along each axis. For purposes of illustration, we compare two hypothetical landscapes: an excessive intensively land use system (left) and a sustainable intensively land use system (right). The intensively managed land use system, however, is able to produce high crop production at the cost of diminishing other land use multifunction. Multifunction should be restored (right).

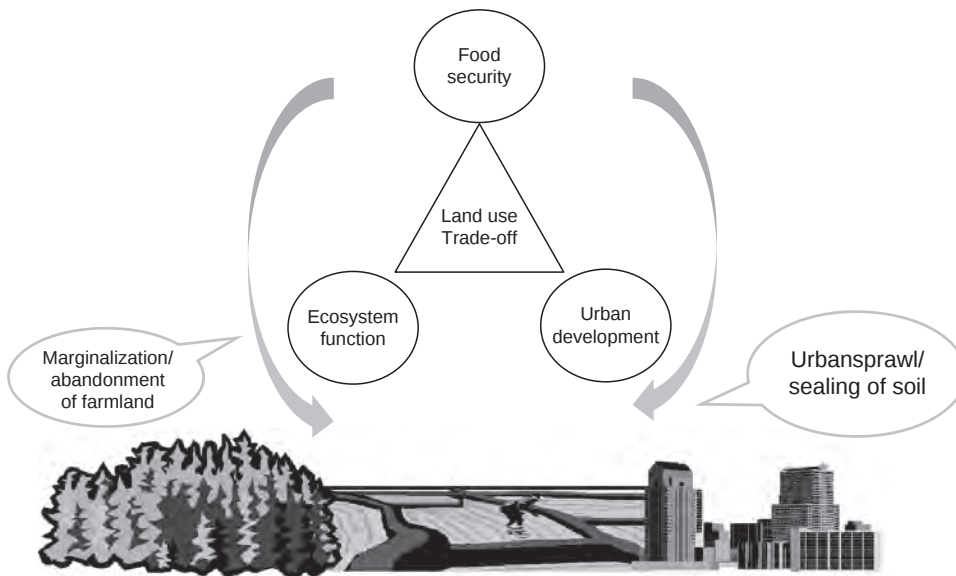


Fig. 2 Concept framework for trilemma for food security in China north eastern plains. Grand challenges between food security, ecosystem multifunction, and urban growth are to be solved within the highest densely populated and scarce land resources in China. The urban growth boundary, the permanent prime farmland, and the ecological redline should be zoned by introduction the ecosystem multifunction.

strengthened the ecological network. However, the continuous expansion of arable land has strongly altered the land use and land cover. Thus, there is a strong need to restore ecosystem multifunctions and services network, which must be zoned and built.

Groundwater has been severely depleted in the CNEP plains by intensive land use, especially in the northern plains, which has not only led to irreversible and adverse ecological effects but has also exacerbated water scarcity for 430 million people. Thus, sustainable use of groundwater is essential, which must be protected in several critical regions (e.g., Hengshui and Cangzhou), and not used for irrigation of vegetables and winter wheat.

Mapping Permanent Prime Cultivated Land Zone

The CNEP has experienced a rapid urbanization since the 1980s. Consequently, large tracts of arable lands have been converted into built-up land, especially around the capital of Beijing and other provincial capitals in the region. The loss of high-quality arable land is the driving force leading to conversion of natural ecosystems to arable land and agricultural intensification.

Appropriation of an adequate amount of area to the arable land use in the region is necessary to advance food security and economic development. Thus, prime arable land must be protected around large cities (e.g., Beijing, Tianjin, Shijiazhuang, and Tanshan).

The permanent farmland zone should include not only cultivated lands but also forest lands, grass lands, wetlands, and village settlements. The main function for the permanent farmland zone is to enable households to achieve diverse land use patterns, grow diverse crops, advance food security, and generate income by growing some cash crops. In this zone, land should be reallocated according to tenure rights and property ownership to improve production

efficiency. Urban encroachment must be prohibited on prime farmland of a high quality.

Mapping the City Growth Boundary

The local government provides incentives to convert arable land into built-up land because of the high return from city encroachment and factory production. However, the city sprawl has led to severe and negative environmental effects.^[3] The concentration of transportation and industry in urban centers has exacerbated the point sources of carbon dioxide and other GHGs, which affected the region's climate. In addition, several trace gases (e.g., nitric oxide, nitrogen dioxide, ozone, sulfur dioxide, and nitric acid), and some organic acids, and urban environments have altered the energy budget leading to the formation of urban heat island (UHI) effects. These UHI effects not only affect local and regional climate but also pollute water resources and air quality, jeopardize human health, and adversely impact biodiversity and ecosystem functioning.

Therefore, smart growth city should be the future priority strategy. An innovative strategy of urbanization proposed by the central government must focus on the effective use of the already built-up land in both urban and rural centers and ban any new urban encroachment in the region. Thus, city growth boundary should be mapped based on the arable land quantity and quality, groundwater depletion, and land ecosystem functions and services.

Restore Local LCCMU

The comprehensive and integrated LCCMU system should be restored within the CNEP on the basis of the top-down method but with a bottom-up approach (Fig. 3). The top-down method fully considers the change of food security, land demand, agricultural development, rural household

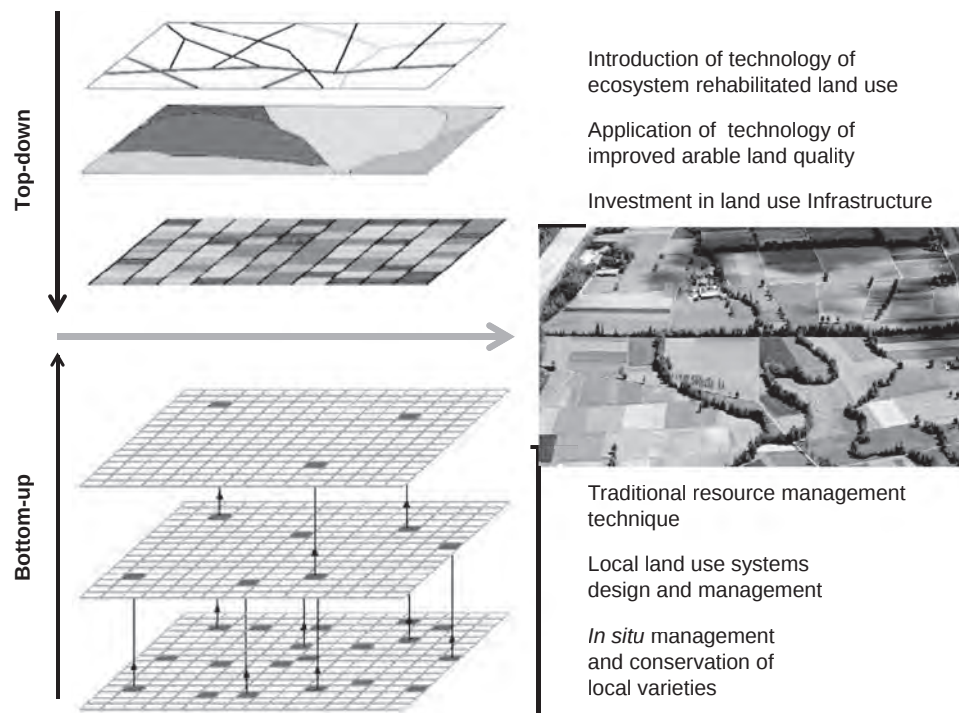


Fig. 3 Concept of the land rehabilitation for multifunction on the basis of the merge of top-down method and bottom-up approach.

shift, infrastructure, and road development plans at the macro level.

However, there are a lot of farmers who live in the rural villages. They have the similar traditional ecosystem knowledge, traditional land use practices from similar soil quality, groundwater distribution, and climate. Thus, traditional land use knowledge should be introduced into the LCCMU on the basis of the bottom-up approach.

Indigenous peoples' knowledge of ecosystems usually results in diverse agricultural landscapes managed for multiple uses, resulting in local food self-sufficiency and ecosystem multifunctions.^[4] Traditional methods are particularly instructive because they provide a long-term perspective on successful land use management. Traditional agroecosystems and associated land diversity are the results of a complex coevolutionary process between natural and social systems, resulting in strategies for ecosystem appropriation.

Thus, a few key principles seem to underlie the sustainability of LCCMU systems—the land use diversity, the soil organic matter (SOM) accumulation, the enhanced recycling of biomass and nutrients, the minimization of resource losses through soil cover and water harvesting, and the maintenance of high levels of functional biodiversity.

Close the Arable Land Potential Productivity Gap

Understanding the arable land potential productivity gap for each plot of arable land (Fig. 4) based on the crop biophysical limits, and possibility of attainable maximum crop yield is critical. The realization of the potential productivity gap is a high priority for research, investment, and policy intervention.

Raise Yield to the Crop Biophysical Limits in Different Climate–Soil–Groundwater Zones

China comprises 12 agroecological zones, 49 agroecological subzones, and 105 agroecological regions.^[5] However, drastic changes have and are occurring in climate, soil quality, groundwater, dominant crops, and cropping

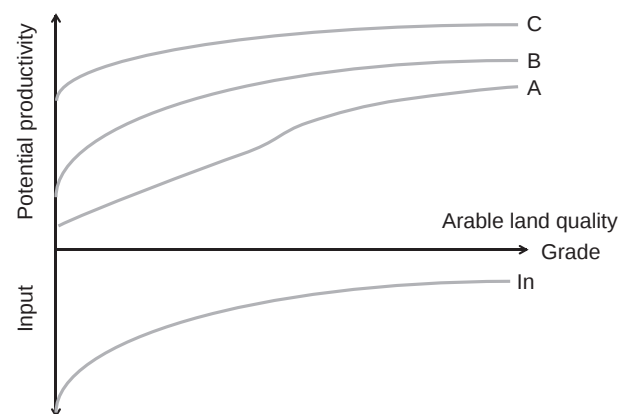


Fig. 4 Concept model for increase ceiling of crop yield potential productivity and sustainable closing productivity gap while reducing input by improved arable land quality management. (A) The scenario of maximum attainable crop yield by household with current managed soil-crop practices upon the improved arable land quality. (B) The scenario of maximum crop yield with existing technology of soil-crop management upon improved arable land quality. (C) The scenario of crop yield with the existing crop varieties upon improved arable land quality. (In) The scenario of input by household for crop yield upon improved arable land quality management.

systems. Thus, new and emerging agroecological zones should be rezoned on the basis of some limiting factors and land's capacity to produce goods and services. Improving yield potential of crop varieties through advancement in crop breeding technology will be a critical component for future crop production.^[6] However, crop productivity depends on continued innovations to control weeds, diseases, insects, and other pests as they evolve resistance to different control measures, or as new species emerge, or are dispersed into new regions. Thus, continued efforts are needed to raise the productivity and crop yield to biophysical limits.

Attaining Yields to the Biophysical Limits by Introduction of Improved Crop–Soil Management

Whether the highest yield can be attained depends on the improvements in crop–soil management and the corresponding arable land quality. Improvements in attainable crop yields to the climate–soil limits can be realized by precision management of crop–soil system while reducing GHG emissions. An “integrated soil–crop systems of management for maize” has been introduced in China.^[7] This involves the use of the hybrid-maize simulation model to maximize the use of solar radiation and the exploitation of temperature changes. It also uses a root-zone, in-season N management strategy to synchronize the supply of N from soil and fertilizer and with crop demand.^[8] The experimental success has provided the data basis on crop biophysical limits for the specific agroecosystem zone, but important issues to be addressed include scaling up of agroecological approaches for improving success in the experimental sites of the region.

Advancing the Maximum Attainable Yield in Different Regions by Improving Arable Land Quality

A high quality of soil under arable land use in the United States supports high crop yield even with less input. On the contrary, high crop production in the CNEP region has been realized by higher input because of low inherent soil quality. Consequently, farmers in the CNEP region have to apply high rates of N, P, and K to obtain high yields. Therefore, the production system in the CNEP is not sustainable. Strategically, the maximum attainable yield in different regions must be realized by improving soil quality of arable land use (Fig. 4). High crop yields to the upper limits could be realized by restoring soil quality^[9] and improving convenient infrastructure (e.g., rural roads, irrigation, and drainage systems). The goal is to reduce inputs of chemical fertilizers, water, labor, and so on by reducing losses and improving use efficiency. Innovative strategies include improvements in irrigation and drainage systems, field leveling, and improving rural roads linking the field to village and market.

Furthermore, improvement in soil quality involves increasing SOM content,^[10] improving structure and tilth, and reducing salinity and sodicity. In addition, adoption of best management practices (BMPs) (e.g., Northern Territory farming, integrated nutrients management (INM) with combined use of organic manure and balanced fertilizer, organic manure, improved grazing, and harvesting) is necessary to improving SOM contents.

Realizing the Maximum Attainable Crop Yields by Providing Subsidy Incentive and Prompting Rural Development

There are several options available to help farmers realize the maximum attainable crop yield in different agroecosystem zones through improvement of the deteriorated or non-existent rural infrastructure.

Investments in rural roads can increase land production by intensifying land use, as well as consolidating the links between agricultural and non-agricultural activities within rural areas and between rural and urban communities.^[11]

Investments are also needed for developing new water management infrastructure. While the economic cost of irrigation may be unprofitable, productivity can be enhanced by soil–water conservation, by increasing efficiency of the systems, and by increasing crop productivity per unit of water used. Among strategies of building, the crop yield is the need for a collective action at the local level, as well as the participation of government and non-governmental organizations at the community level.

Sustainable Intensification

One of the biggest opportunities for restoration of land use multifunctions and services is the development and implementation of a new pattern of sustainable of agriculture.^[12] This new pattern should be based on the combination of local traditional knowledge and the modern agricultural managements.

Local rural farmers use a wide range of techniques that are knowledge intensive rather than input intensive. Traditional knowledge constitutes intercropping, SOM management, enhanced recycling of biomass and nutrients, minimization of resource losses through soil cover and water harvesting, and maintenance of high levels of functional biodiversity. Traditional agriculture is a critical source of genetic material and regenerative farming techniques.

Indeed, traditional knowledge is the foundation of a sustainable intensification of managed rural land and is critical to introducing agricultural modernization among resource-poor farmers. The latter may include prudential use of monocultures, new varieties, agrochemical packages, all of which are critical to increasing yields, labor

efficiency, and farmers' income. The foundation of sustainable intensification must be grounded in traditional knowledge for site-specific situation.

Therefore, sustainable intensification could succeed in the context of combining the traditional and modern agriculture, which can meet the growing demands of crop production for food, conservation, and enhancement of soil C stocks. Sustainable intensification opportunities exist in three key areas:

1. *Yield intensification* implies the increase in crop production per unit area and inputs. It begins with closing the yield gap at the regional level and refers to the yield difference between lower yielding production systems and the highest yields that can be achieved with BMPs for similar climate–soil conditions.
2. *Temporal intensification* implies an increase in the number of different crops grown per year, through local and traditional crop diversification including crops for food, and green manure for improving soil quality.
3. *Spatial intensification* implies land optimization and the implementation of MBPs for addressing trade-off among multiple competing land uses on the basis of soil quality, groundwater, and traditional cropping systems. It requires an integrated and comprehensive approach to managing land systems covering all production and conservation uses.

In general, CNEP is the important food production region of China. It is the continuous increase in agricultural intensification since the 1980s, which has transferred the historic grain production over centuries from the south to the north. However, agricultural intensification related to high rate of application of chemical fertilizers and the irrigation from groundwater have created the vicious cycle of higher wheat yields, more water depletion, and more severe deficit of groundwater. The regional consequences of arable land expansion and the excessive intensification have led to severe loss of land ecosystem functions and services. There is no silver bullet to sustainable land use, along with realization of the soil-based green revolutions in this region. Addressing the following challenges requires action through green revolutions in the CNEP plains: (a) restoration of the land use ecosystem multifunction services should be a high priority; (b) crop production should be continuously increased; and (c) shift of the unsustainable intensification to green intensification should be the logical way to solve the dilemma of increasing crop production, restoring groundwater depletion, and reducing the GHG emissions.

ACKNOWLEDGMENTS

This work was supported by Chinese Social Science Fund (2014-14AZD031), China Agricultural University Fund (2014RW014), and China Scholarship Council Fund (201406355013).

REFERENCES

1. Kong, X.; Lal, R.; Li, B.; Liu, H.; Li, K.; Feng, G.; Zhang, Q.; Zhang, B. Fertilizer intensification and its impacts in China's HHH plains. *Adv. Agron.* **2014**, *125*, 135–169.
2. Liu, Y.; Wang, L.; Ni, G.; Cong, Z. Spatial distribution characteristics of irrigation water requirement for main crops in China (in Chinese). *Trans. CSAE.* **2009**, *25*, 6–12.
3. Grimm, N.B.; Faeth, S.H.; Golubiewski, N.E.; Redman, C.L.; Wu, J.; Bai, X.; Brigg, J.M. Global change and the ecology of cities. *Science* **2008**, *319*, 756–760.
4. Altieri, M.A. Linking ecologists and traditional farmers in the search for sustainable agriculture. *Front Ecol. Environ.* **2004**, *2* (1), 35–42.
5. Zhang, J.K.; Zhang, F.R.; Zhang, D.; He, D.X.; Zhang, L.; Wu, C.G.; Kong, X.B. The grain potential of cultivated lands in Mainland China in 2004. *Land Use Policy* **2008**, *26*, 68–76.
6. Godfray, H.C.J.; Beddington, J.R.; Crute, I.R.; Haddad, L.; Lawrence, D.; Muir, J.F.; Pretty, J.; Robinson, S.; Thomas, S.M.; Toulmin, C. Food security: The challenge of feeding 9 billion people. *Science* **2010**, *327*, 812–818.
7. Fan, M.S.; Shen, J.B.; Yuan, L.X.; Jiang, R.F.; Chen, X.P.; Davies, W.J.; Zhang, F.S. Improving crop productivity and resource use efficiency to ensure food security and environmental quality in China. *J. Exp. Bot.* **2012**, *63* (1), 13–24.
8. Chen, X.-P.; Cui, Z.-L.; Vitousek, P.M.; Cassman, K.G.; Matson, P.A.; Bai, J.-S.; Meng, Q.-F.; Hou, P.; Yue, S.-C.; Römhild, V.; Zhang, F.-S. Integrated soil–crop management system for food security. *P. Natl. A. Sci.* **2011**, *108*, 6399–6404.
9. Kong, X. China must protect high-quality arable land. *Nature* **2014**, *506*, 7.
10. Lal, R. Soil carbon sequestration impacts on global climate change and food security. *Science* **2004**, *304*, 1623–1627.
11. Rosegrant, M.W.; Cline, S.A. Global food security: Challenges and policies. *Science* **2003**, *302*, 1916–1917.
12. Garnett, T.; Appleby, M.C.; Balmford, A.; Bateman, I.J.; Benton, T.G.; Bloomer, P.; Burlingame, B.; Dawkins, M.; Dolan, L.; Fraser, D.; Herrero, M.; Hoffmann, I.; Smith, P.; Thornton, P.K.; Toulmin, C.; Vermeulen, S.J.; Godfray, H.C. J. Sustainable intensification in agriculture: Premises and policies. *Science* **2013**, *341*, 33–34.

Greenhouse Effect

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

World soils constitute a major terrestrial carbon (C) pool, and even a small change in the C pool of world soils can lead to a substantial change in the atmospheric C pool. A considerable increase in the atmospheric C pool, since the industrial revolution, has come from the soil and biotic pools. Agricultural activities that lead to emissions of carbon dioxide and methane from soil to the atmosphere include deforestation, biomass burning, and plowing. Gaseous emissions are accentuated by soil degradative processes. Conversion to an appropriate land use and adoption of judicious management can reverse the trend by making world soils a major sink for C. The soil organic carbon content of cropland soils can be maintained or enhanced by use of conservation tillage, application of crop residue mulch, and judicious use of chemical fertilizers and organic amendments that enhance soil quality.

INTRODUCTION

Earth's vast reservoir of carbon (C) is contained in four distinct pools (Table 1). The soil C pool comprises both organic and inorganic forms. Soil organic carbon (SOC) is estimated at 1550 Pg (1 Pg = petagram = 10^{15} g = 1 billion metric tons) and soil inorganic carbon (SIC) at 950 Pg. The atmospheric pool contains 760 Pg of C, and the vegetation for the biotic pool amounts to about 610 Pg of C. The pedologic or soil C pool, with a total reserve of about 2500 Pg in both organic and inorganic form, is about 3.3 times the atmospheric pool and 4.1 times the biotic pool.^[1–10]

In addition to being a pool, world soils also play a major role in the global C cycle (Fig. 1). The SOC pool is highly dynamic, concentrated near the soil surface, and comprises material of plant and animal origin at various stages of decay and decomposition. A large part of the SOC pool is held in the organic soils of tundra and boreal forest ecosystems. It is the dynamic SOC pool that is intimately connected with the projected global warming or the accelerated greenhouse effect. Change in soil C pool by 1 Pg is equivalent to 0.47 ppm change in atmospheric concentration of carbon dioxide (CO₂).

These global pools are interconnected through fluxes of C (Fig. 1). The atmospheric pool has been increasing at the expense of other pools. Some known sources and sinks are listed in Table 2. The two principal sources are fossil-fuel combustion (6.8 Pg/yr) and deforestation and soil disturbance (1.6 Pg/yr). The two proven sinks are atmosphere (3.5 Pg/yr) and ocean (2.0 Pg/yr). The unknown sink for the remaining 2.7 Pg/yr is most likely the world soils and other terrestrial ecosystems.^[11] The exchanges occur between: 1) world biota and the atmosphere with an annual flux of 120 Pg photosynthesized by plants and

returned to the atmosphere in equal amounts through plant and soil respiration and 2) oceans and the atmosphere amounting to an annual flux of 107 Pg from atmosphere to the ocean and return of 105 Pg making the ocean a net sink of about 2 Pg/yr.

There has been a steady increase in the atmospheric concentration of greenhouse gases [GHGs, CO₂, methane (CH₄), and nitrous oxide (N₂O)] since the beginning of agriculture,^[12] but more strongly with the onset of the industrial revolution in the 1850s. This increase is primarily due to human activities, e.g., industry-based fossil-fuel consumption, and agriculture-based deforestation and conversion to cultivable land involving intensive use of fertilizers and manures. The CO₂ concentration in the atmosphere has increased from 280 ppmv in the 1850s to about 370 ppmv in 2000^[4] and is increasing at the rate of 0.5% per year (Table 2).

AGRICULTURAL ACTIVITIES AND GHG EMISSIONS

GHGs are emitted by numerous agricultural activities (Fig. 2). Agricultural sources of CO₂ are deforestation, land use conversion, biomass burning, soil plowing and other mechanical disturbances of the topsoil, and fossil-fuel consumption. About 50% of the total increase in atmospheric CO₂ is attributed to agricultural activities. Among principal sources of CH₄ are enteric fermentation in ruminant animals, cultivation of rice paddies, anaerobiosis in poorly drained soils, and biomass burning. Use of nitrogenous fertilizers and manures is the principal source of N₂O emission. Soil management plays a major role in the type and amount of emission of GHGs.

Table 1 Global pools of C.

Pool	Amount (Pg)
Ocean	
Intermediate and deep ocean	38,100
Surface ocean	1,020
Dissolved organic carbon	700
Surface sediment	150
Marine biota	3
Soil (1-m depth)	
SOC	1,550
SIC	950
Vegetation	610
Atmosphere	760

Source: From Kennel.^[4]

The rate of oxidation or mineralization of the SOC pool depends on its composition and numerous other factors. Factors that increase oxidation are soil disturbance, high soil temperature, lack of protective vegetal cover, and risks of soil erosion and degradation. In contrast, factors that decrease oxidation are minimal soil disturbance, low soil temperature, minimal soil disturbance, presence of protective vegetal cover, and high soil resilience. The annual flux of CO₂ from soil to the atmosphere, about 60 Pg, is due to soil respiration and its rate increases with increase in soil temperature. The mean resident time of SOC in soil is about 32 years, but some resistance humic substances can stay in soil for thousands of years. Losses of SOC pool by oxidation are accentuated by soil erosion of water and wind^[13–15] and other degradative processes.

Introduction of mechanized agriculture (e.g., tractor based plowing and other farm operations causing soil disturbance) have accentuated losses of SOC pool from cultivated soils. Some soils lose as much as 25% of their SOC pool within 1 year after deforestation and cultivation.^[16] The rate of loss of SOC content is more in the tropics than in the temperate climate. Some tropical soils, cultivated for millennia in Asia, have extremely low levels of remaining SOC content (as low as 0.2%). These soils have probably lost 75–80% of their initial SOC content. It is likely that as much as 66–90 Pg of C in the atmospheric pool has been contributed by world soils.^[17]

In addition to being a source of atmospheric C (as CO₂, CH₄, or CO), world soils are also a major source of N₂O. The relative contribution of N₂O has also increased because of the intensive use of nitrogenous fertilizers since World War II. The magnitude of N₂O emission varies among soils, type of nitrogenous fertilizer used (formulation), climate, land use, and soil and crop management.^[3]

POTENTIAL OF WORLD SOILS AS C SINK

The SOC in most agricultural soils is below their potential capacity. As a considerable part of atmospheric CO₂ has come from world soils, it is possible to reverse the trend and store C back into the soil. Increase in SOC content, through change in land use and farming practices, could slow the rate of increase of atmospheric CO₂ and provide a negative feedback to the greenhouse effect. Because of their vast pool, it is very likely that world soils are the unknown sink for 2.7 Pg of C/yr (Table 2).

Principal mechanisms by which soils can be used as an effective sink for some of the atmospheric CO₂ through

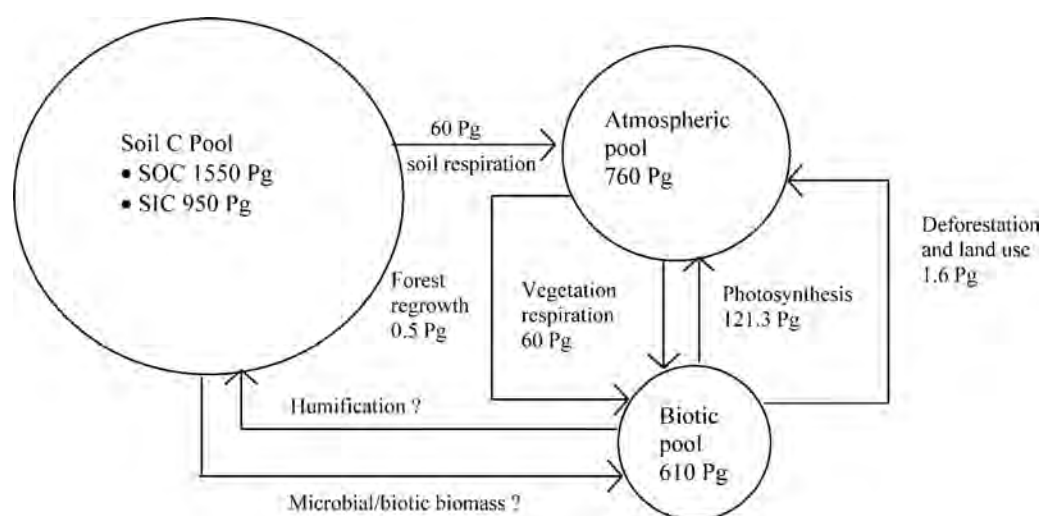


Fig. 1 Interactive effects among the three C pools (soils, biota, and atmosphere) have led to a steady increase in the atmospheric pool at the expense of the soil and biotic pools. Values in the arrow indicate direction and magnitude of the known fluxes, while the question mark indicates the lack of information.

Source: From Schlesinger,^[1] Kennel,^[4] and Batjes.^[10]

Table 2 Sources and sinks of atmospheric CO₂.

Source/sink	Flux (Pg)
Source	
Fossil fuel combustion	6.8
Deforestation, land use, and world soils	1.6
Total	8.4
Sinks	
Increase in atmospheric concentration	3.5
Ocean uptake	2.0
Total	5.7
Balance of unknown sink most likely to be world soils	2.7

Source: From Schlesinger,^[1] Kennel,^[4] and Bouwman.^[5]

increase in SOC content include: 1) humification or conversion of biomass into resistant humic substances with long turnover time; 2) aggregation or formation of stable organo-mineral complexes that render SOC inaccessible to microbes; and 3) deep incorporation of SOC below the plow layer through deep root system development and translocation by leaching. These mechanisms can be set-in-motion through adoption of proper land use, choice of appropriate farming/cropping systems, and use of judicious soil and crop management practices.^[18–22]

The potential of C sequestration in world soils depends on the land use, present level of SOC content, the initial SOC content prior to soil disturbance by deforestation and conversion to other land uses, climatic or eco-regional characteristics, and farming systems. Among all land uses, the largest potential increase in SOC content is in arable lands. If the SOC content of the world's arable land can be increased at the low rate of 0.01% per year to 1 m depth, which may be difficult to achieve for soils of arid and semiarid regions, the C sequestration potential of such a possibility is about 2 Pg/yr (Eq. 1). These calculations are based on the assumption that the mean bulk density of soils is 1.33 Mg/m³.

C sequestration potential of 1500 × 10⁶ ha of world arable land

$$= (1500 \times 10^6 \text{ ha})(10^4 \text{ m}^2/\text{ha})(1 \text{ m}) \times (1.33 \text{ Mg/m}^3)(1 \times 10^{-4}/\text{yr}) = 2 \text{ Pg/yr} \quad (1)$$

MANAGING WORLD SOILS FOR MITIGATION OF THE GREENHOUSE EFFECT

There are several soil management practices that can be used to increase SOC content and reduce the risk of accelerated greenhouse effect (Fig. 3). These practices can be grouped under two broad categories: 1) those

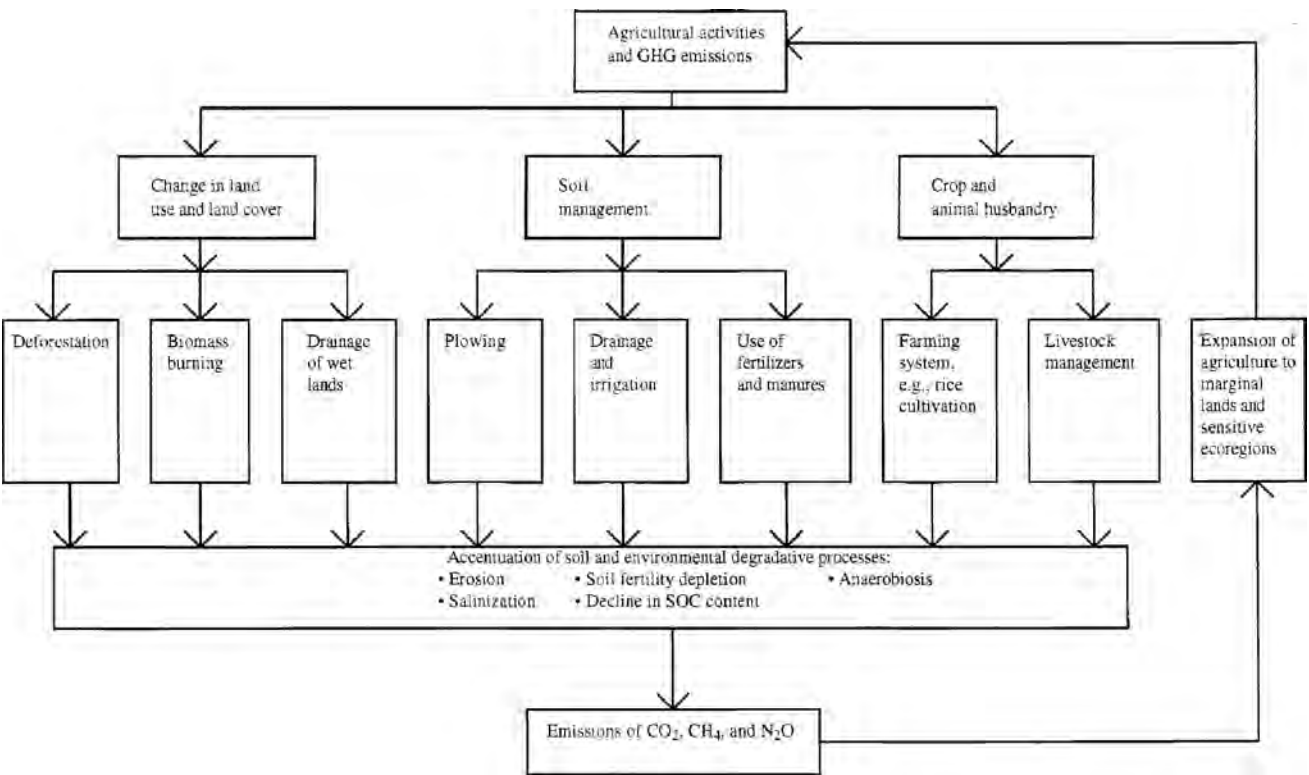


Fig. 2 Agricultural activities and emission of GHG from soil to the atmosphere.

that decrease losses due to soil degradative processes and 2) those that increase SOC content through soil restorative processes (Table 3). Principal strategy is of agricultural intensification through adoption of appropriate land use and recommended agricultural practices.

A tremendous potential of mitigation of the greenhouse effect exists through restoration of degraded and drastically disturbed soils. It is estimated that the area of the degraded soils of the world is about 2.0 billion hectares.^[23] These soils include the cultivated land area, pastoral land areas, and other managed and natural ecosystems. Restoration to improve soil quality can lead to an increase in ground biomass production (both above and below the ground), increase in SOC content, and strengthening of a negative feedback to the greenhouse effect. The C sequestration potential of restoration of degraded soils is about 3 Pg/yr (Eq. 2):

$$\begin{aligned} \text{C sequestration potential of restoration of degraded} \\ \text{soils} &= (2 \times 10^9 \text{ ha}) \times (10^4 \text{ m}^2/\text{ha}) \times (1\text{m})(1.5\text{Mg}/\text{m}^3) \\ &\quad \times (10^{-4}/\text{yr}) = 3 \text{ Pg/yr} \end{aligned} \quad (2)$$

These calculations are based on the assumption that mean soil bulk density to 1 m depth is 1.5 Mg/m³. The potential of

Table 3 Strategies for increasing SOC content for mitigation of the greenhouse effect.

Practices that decrease loss of SOC content	Practices that enhance SOC content
Soil erosion control techniques	Improved soil fertility and integrated nutrient management
Conservation tillage	Precision farming
Residue management	Nutrient cycling
Terraces and contour barriers	Application of biosolids and biological nitrogen fixation
Improved cropping systems	Restoration of degraded soils
Afforestation	Eroded soils
Decreasing losses of dissolved organic carbon through leaching	Mineland and drastically disturbed soils
	Salt-affected soils
	Water table management (drainage and subirrigation)

3 Pg/yr is almost equivalent to the rate of annual increase in atmospheric C at 3.5 Pg/yr (Table 2) and can be realized over 25–50 years. Realization of this potential requires development of strict land care ethics and global soil policy that must be strictly adhered to. While attempts at decreasing the emissions due to fossil-fuel combustion must be

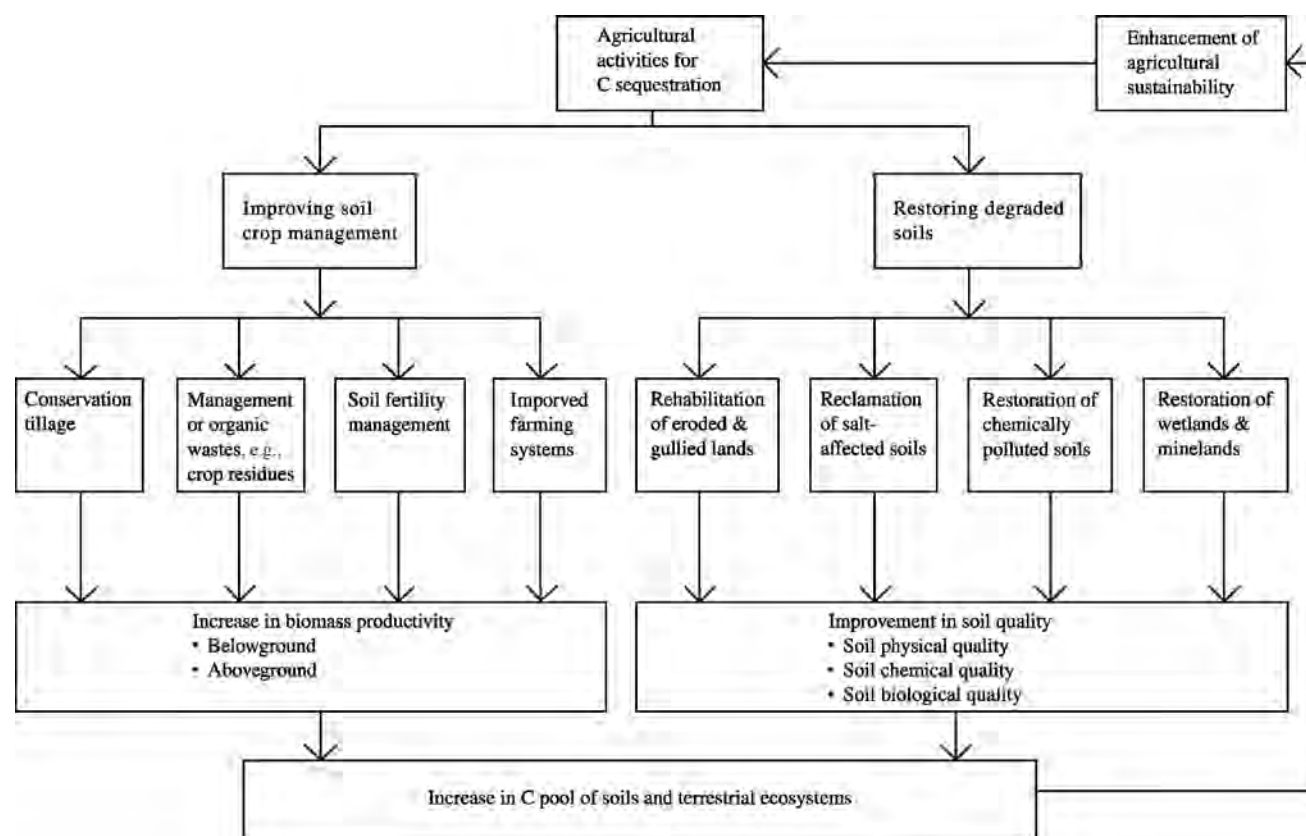


Fig. 3 Agricultural activities with a potential of sequester C in world soils that decrease the risks of the greenhouse effect.

continued, it is important that the vast potential of world soils in C sequestration is realizable through adoption of known and proven technologies. The attainable potential of world soils for C sequestration is 0.6–1.2 Pg C/yr.^[19]

CONCLUSION

World soils constitute a major terrestrial C pool, and even a small change in the C pool of world soils can lead to a substantial change in the atmospheric C pool. A considerable increase in the atmospheric C pool, since the industrial revolution, has come from the soil and biotic pools. Agricultural activities that lead to emissions of CO₂ and CH₄ from soil to the atmosphere include deforestation, biomass burning, and plowing. Gaseous emissions are accentuated by soil degradative processes including accelerated erosion, structural decline, etc.

Conversion to an appropriate land use and adoption of judicious management can reverse the trend by making world soils a major sink for C. The SOC content of cropland soils can be maintained or enhanced by use of conservation tillage, application of crop residue mulch, and judicious use of chemical fertilizers and organic amendments that enhance soil quality. Restoration of the world's degraded soils and desertification control has a potential to sequester C at the rate of 3.0 Pg/yr.

REFERENCES

- Schlesinger, W.H. An overview of the carbon cycle. In *Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1995; 9–25.
- Schlesinger, W.H. *Biogeochemistry: Analysis of Global Change*; Academic Press: San Diego, 1991.
- Lal, R.; Kimble, J.; Levine, E.; Whitman, C. World soils and greenhouse effect: An overview. In *Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1995; 1–7.
- Kennel, C. *UC Reville Program on Climate, Science and Policy*; Scripps Institution of Oceanography: La Jolla, 2000.
- Bouwman, A.F. *Soils and the Greenhouse Effect*; Wiley: London, 1990; 575.
- Post, W.M.; Emmanuel, W.R.; Zinke, P.J.; Stangenberger, A.G. Soil carbon pool in world life zones. *Nature* **1982**, *298*, 156–159.
- Post, W.M.; Peng, T.H.; Emmanuel, W.R.; King, A.W.; Dale, V.H.; De Angelis, D.L. The global C cycle. *Am. Sci.* **1990**, *78*, 310–326.
- Sandquist, E.T. The global CO₂ budget. *Science* **1993**, *259*, 934–941.
- Eswaran, H.; Vandenberg, E.; Reich, P.; Kimble, J. Global soil carbon resources. In *Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1993; 27–43.
- Batjes, N.H. Total carbon and nitrogen pools in soils of the world. *Eur. J. Soil Sci.* **1996**, *47*, 151–163.
- Tans, P.P.; Fuang, I.Y.; Takahashi, T. Observational constraints on the global atmospheric CO₂ budget. *Science* **1990**, *247*, 1431–1438.
- Ruddiman, W.F. The anthropogenic greenhouse era began thousands of years ago. *Climatic Change* **2003**, *61*, 261–283.
- Lal, R. Global soil erosion by water and carbon dynamics. In *Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1995; 131–141.
- Lal, R. Soil erosion and the global carbon budget. *Env. Intl.* **2003**, *29*, 437–450.
- Lal, R. Soil erosion and carbon dynamics. *Soil Tillage Res.* **2005**, *81*, 137–142.
- Lal, R. Deforestation and land use effects on soil degradation and rehabilitation in Western Nigeria. III. Runoff, soil erosion and nutrient loss. *Land Degrad. Dev.* **1996**, *7*, 99–119.
- Lal, R. Soil management and restoration for C sequestration to mitigate the accelerated greenhouse effect. *Prog. Env. Sci.* **1999**, *1*, 307–326.
- Lal, R. Agricultural activities and the global carbon cycle. *Nut. Cycling Agroecosystems* **2005**, *70*, 103–116.
- Lal, R. Soil carbon sequestration impacts on global climate change and food security. *Science* **2004**, *304*, 1623–1627.
- Lal, R. Global potential of soil carbon sequestration to mitigate the greenhouse effect. *Crit. Rev. Plant Sci.* **2003**, *22*, 151–184.
- Lal, R. Off-setting global CO₂ emissions by restoration of degraded soils and intensification of world agriculture and forestry. *Land Degrad. Dev.* **2003**, *14*, 309–322.
- Jarecki, M.; Lal, R. Crop management for soil carbon sequestration. *Crit. Rev. Plant Sci.* **2003**, *22*, 1–32.
- Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 99–118.

Growing Media

Nazim Gruda

Federal Office for Agriculture and Food and University of Bonn, Bonn, Germany

Jean Caron

Soil Science and Agrifood Engineering Department, Laval University, Ste-Foy, Quebec, Canada

Munoo Prasad

Bord na Móna Horticulture, Co., Kildare, Ireland

Michael Maher

Kinsealy Research Centre, Teagasc, Dublin, Ireland

Abstract

The use of growing media, as opposed to soil *in situ*, has become widespread especially for the production of high-value greenhouse crops as well as in plant propagation and the production of plants in small containers. Growing media provide a root environment that is free of plant pathogens and can have physical and chemical properties that ensure adequate aeration for the root system and optimize the supply of water and nutrients to the shoot. The result is the reliable production of high-quality plant material and the achievement of increased yields compared with conventional soil culture. A wide range of inorganic and organic materials are used as components of growing media. Peat moss has been the most important material in many countries. There is increased interest in the use of organic waste materials such as composted crop residues and biowaste and in the forms of charcoal such as biochar. Systems based on inorganic materials such as rock wool, perlite, and polyurethane foam have been developed in combination with fertigation systems to successfully produce many crops. Standard methods for measurement of the physical, chemical, and biological properties of these materials have been developed and are widely used in the analysis of growing media. Given their success and the need to develop intensive crop production systems, it seems likely that the use of growing media will continue to expand.

INTRODUCTION

The effort to develop growing media on a scientific basis began with the work of Lawrence and Newell at the John Innes Horticultural Institution in the United Kingdom in the 1930s. They developed media based on the use of loam, peat, and sand with added fertilizers. In the 1950s, Baker at the University of California (UC) developed the UC loamless composts based on mixes of peat and sand, as suitable loam was difficult to obtain. Puustjarvi, in Finland, subsequently developed growing media based exclusively on peat. In many countries, especially in Europe, over a number of decades, growing media were based almost exclusively on peat in the 1970s. Fears about the sustainability of the continued use of peat have led to increased interest in alternative materials and a wide range of inorganic and organic, particularly, waste materials are being used as growing media or as components of growing media.

Growing media have been used traditionally mostly for plant propagation, bedding, and pot plant production, but this range of use has expanded to include the total production of many food and ornamental crops especially high-value ones grown under protection in greenhouses. These systems are called soilless culture systems. *Culture: Soil-less* is presented in another separate entry (p. 533).

Because the plants are grown in much restricted volume compared with soil *in situ*, the requirements for the properties of the growing medium, i.e., physical, chemical, and biological, become more stringent.

PHYSICAL PROPERTIES

Because of a broad range of applications much different from growing conditions in agricultural fields, the range of values for physical properties of organic media are

quite different from those in mineral soils. Relative to an agricultural field, the fact that they are potted results in a direct effect of container geometry on the resulting prevailing physical properties.^[1] Growing media have therefore specific norms to guide their manufacturing and for diagnosis of plant growth problems.^[1] Some media are relatively friable, and their properties are affected by their preparation, processing, and handling prior to potting. Moreover, once potted, their properties continue to evolve due to substrate subsidence and decomposition during use. Consequently, their properties must be measured with methods adapted to their specific context of use (see the section “Testing of Growing Media” for testing). The concept of particle size distribution, extensively used in mineral soils to relate texture to physical properties like saturated hydraulic conductivity, available water, and bulk density among others, has limited use as a quality criterion with organic growing media. Its success has been essentially limited to detecting severe aeration problems, but particle size distribution has remained a poor predictor of other properties and growth performances, as the shape of coarse fragments (>10 mm diameter and larger) can result in aeration limitations compared to intermediate-sized fragments and because of the interactive effect of particle shape on mixes. The most commonly used physical properties are related to aeration and available water in growing media, as they are related to plant performances. Specifically, such properties are related to the storage and exchange of air (e.g., air-filled porosity, gas diffusivity), the drainage properties (e.g., hydraulic conductivity), the availability of the water for plant growth (e.g., available water), and the degree of compaction as per its bulk density, among others. Air-filled porosity is the most often referred index of aeration processes, with recommended values being in the range of 0.20–0.30 cm³ cm⁻³ at -1 kPa. Studies on aeration processes confirmed through the clear advantage of measuring gas diffusivity to relate the quality of the substrate to plant growth. Moreover, the very high respiration needs of some organic growing media should also be included, as competition for oxygen may limit crop performance. An excellent substrate should also have an equivalent proportion (0.20–0.30 cm³ cm⁻³) of water available (defined as the proportion of water between container capacity, typically -1 kPa, and the beginning of stress for plant growth set at -10 kPa). The lower limit of water availability is reached at -10 kPa, as typically water in peat and peat-soil mixes becomes less available to plants (first signs of stress) at a higher range of water potentials (-5 to -15 kPa), with permanent wilting in the range of -30 kPa, compared to that in mineral soils (-10 to -60 kPa) for first sign of wilting, and then to -1500 kPa for permanent wilting, respectively. Hence, available water is and should typically be calculated using -10 kPa as the lower limit. When performing

these evaluations, the water desorption curve should be described with the van Genuchten–Durner approach, incorporating wetting angle changes during the desorption and rewetting process and taking into account the hysteresis phenomenon, which being sometimes extraordinarily large in organic media.^[1]

CHEMICAL AND BIOLOGICAL PROPERTIES

For the evaluation of chemical properties of a growing medium, the most important criteria are pH and the nutrient content of macroelements and microelements. Significant variations in pH can occur for some substrates, depending on their provenance. Therefore, corrections are often necessary. The pH plays an important role in plant substrates, determining the availability of various nutrients. Although plant pH requirements differ, for most plants optimal nutrient availability occurs when the pH value of a nutrient solution is between 5.5 and 6.0. Higher values nearly always reduce the solubility of phosphates, iron, and most micronutrients. Moreover, high pH values (>7.5) in the irrigation water are undesirable, given the probable precipitation of calcium and magnesium carbonates, as well as orthophosphates, which can clog the drippers.^[2] It is a standard practice to adjust the pH of an acidic growing medium such as peat moss by the addition of limestone. Since the pH of some materials can change during the storage process, it is recommended to analyze materials shortly before using and to adjust the pH value according to plant requirements. This is valid for nutrient levels as well.

Cation exchange capacity (CEC) gives information about the sorption force and buffering ability of a substrate for nutrients. Substrates with high CEC can store more nutrients, and plants are fertilized more intensively. In addition, such substrates buffer the fertilizer or mineral materials better when hard water is used. From a practical point of view, considering the small volumes of growing media used for plant production, continuous fertigation of substrate-grown crops enables the use of different substrates with different CEC. High CEC growing media may lead to nutrient imbalance if not adequately adjusted; however, frequent fertigation can mitigate the negative effects.^[2]

A good growing media must be free from pests and pathogens, be biologically stable, and not be toxic. The use of forestry products as well as immature compost container substrates can involve problems of phytotoxicity. Some of these materials have a protection function and defend, e.g., bark against insects or infections, and can be toxic to other organisms, such as greenhouse plants cultivated in substrates originating from these materials. Different methods such as composting, aging, leaching, washing, mixing, and fertilization are used to reduce or eliminate phytotoxicity properties of growing

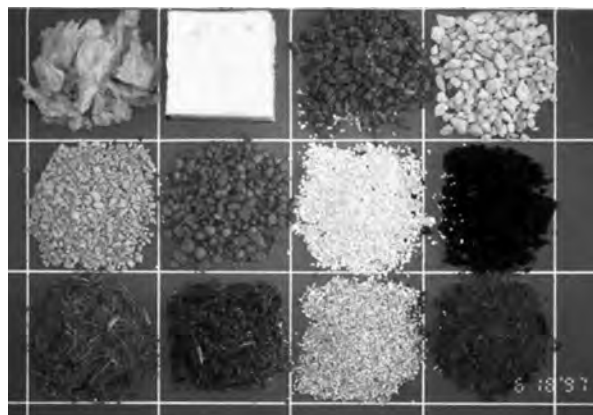


Fig. 1 Some materials used for soilless culture and growing media. From left to right: Rock wool, polyurethane foam, expanded shale, volcanic material, open porous clay granulate, expanded clay, perlite, black peat, coarse wood fiber, fine wood fiber, vermiculite, and light peat.

Source: Photo courtesy of N. Gruda.

media materials. Some of these methods are also used to reduce the nitrogen (N) immobilization of wood fiber products.^[2]

MATERIALS USED AND GROWING MEDIA CHOICE

An important difference between soil and substrate culture is the limited volume of substrate that can be used, which means a much reduced volume for the root system. As a

consequence, adaptation in cultivation methods, mainly in irrigation, is required.

Some of the growing media used are illustrated in Fig. 1. Table 1 provides an overview of some characteristics for different substrates. A brief presentation of the most used substrates is shown as follows.

A look at the diversity of the existing growing media teaches us that the success of cultivating seedling and transplants in them could be achieved in different ways, and since technical and financial implications are involved, there is no unequivocal scheme for the choice of growing media. A mixture of growing media constituents and additives is generally used in the horticultural industry. Growing media constituents include combinations of different materials, mainly peat and other organic or inorganic materials. Growing media additives include fertilizers, liming materials, and biocontrol or wetting agents. The choice of which material to be used as a growing medium or a component depends on the type of crop to be grown; for instance, some of the materials with high pH, e.g., composted green waste, would not be suitable for ericaceous crops. In addition, seed germination and growth of some sensitive plants are particularly susceptible to high electrical conductivity (EC), and some of the materials, particularly some composted organic wastes, would not be suitable. Crops grown over a long term need stable growing media, as collapse of the growing medium in the pot could cause aeration problems.

INORGANIC SUBSTRATES

Gravel and sand were used in older installations. They are used as components to improve the aeration and give

Table 1 Advantages and disadvantages of materials used as growing medium or as growing media constituents.

Material	Advantages	Disadvantages
Sand	Relatively inexpensive, good drainage ability	Low nutrient- and water-holding capacity
Rock wool	Lightweight and ease of handling, totally inert, nutrition can be carefully controlled	Disposal problems, energy consumed during manufacture
Vermiculite	Lightweight, high nutrient-holding ability, good water-holding ability, good pH buffering capacity, good aeration	Compacts when too wet, energy-consuming product, expensive
Perlite	Lightweight, sterile, neutral in pH, no decay	Low nutrient capacity, energy-consuming product, expensive
Peat	Physical stability, good air and water-holding capacity, low microbial activity, lightweight, low and easily to adjusted pH, low nutrient content	Finite resource, environmental concerns and contribution to carbon dioxide release, increasing cost due to energy crisis, may be strongly acidic
Coconut coir	Physical stability, good air content- and water-holding capacity, low and easily to adjusted pH, lightweight	May contain high salt levels, energy consumption during transport
Bark (well-aged)	Good air content and water-holding capacity	Increasing cost since used as an alternative to fuel and in landscaping
Green compost	Good source of potassium and micronutrients, suppression of diseases, good moisture-holding capacity	Variable in composition, may contain excess salt, high CEC, becomes easily waterlogged
Biochar and hydrochar	Production is energy neutral, helps with carbon sequestration, biologically very stable; wet material can be used for hydrochar; hydrochar has low EC	High production costs; biochar often has high pH, can be dusty; properties vary dependent on feedstock, particularly for biochar

weight to the container in nursery, avoiding plants' fall during strong wind events. The use of polythene-wrapped rock wool, originally produced as insulation in the construction industry, aided by its lightweight and ease of handling, has become the dominant soilless culture system for greenhouse vegetables and is used throughout the world. A complete nutrient solution is supplied through the irrigation system. Besides rock wool, different inorganic substrates such as perlite, tuff, volcanic porous rock, expanded clay granules, vermiculite, as well as synthetic materials have been used as growing media. In addition, organic substrates can be used.

ORGANIC SUBSTRATES

Peat, formed by partial decomposition of sphagnum, other mosses, and sedges, has long been used as a component of standardized growing media, but work from the 1950s and later showed that it could be used as a growing medium in its own right both for container plants and for vegetable and cut flower production. The main advantage of peat as a growing medium is its relative consistency, low nutrient content, and low pH, so that pH and nutrient levels can easily be adjusted to desirable levels. The other advantages of peat are its lightweight, high water-holding capacity, and high air space.

Wood fiber, wood chips, and sawdust are renewable resources from the woodworking industry. All these products are characterized by low water retention, good air content, and high saturated hydraulic conductivity. Wood chips and sawdust are usually used in the proportion of 20–30% (volume basis) in mixtures with other substrate components. However, with an adequate supply of complete nutrient solution, it is possible to use wood fiber alone as a growing medium. A reduction in particle size, an increase in volume weight, and an increase in irrigation frequency are recommended.^[3] Furthermore, a compression of wood fiber substrates in pots is recommended to minimize substrate loss. Additional N fertilization from the beginning of plant cultivation is recommended to overcome N immobilization.^[4]

Composted materials such as green waste and biowaste can be used as a component of a growing medium up to 50%, but not on its own.^[5] The limiting factor regarding the use of composted green waste is its high EC and potassium (K) concentration. There can also be a problem of plant pathogens, human pathogens, and weed contamination if the composting process is not properly conducted, i.e., if the temperature time exposure is not sufficient.

Bark from conifers (soft wood) and deciduous trees (hardwood) can be used as a component of growing medium. Bark has to be composted to eliminate phytotoxins, and N must be added during composting to overcome N immobilization.^[6] The air and water-holding capacity of

the bark can be adjusted by varying the percentage of fine materials (less than 1–2 mm).

Coir is a by-product of the coconut industry and comes from the fibrous husk of coconuts. The powdery material can be used as a growing medium. Most of the coir dust comes from Sri Lanka, India, and Southeast Asia. It can contain high levels of sodium, K, and chlorine, which may need to be leached. Due to its high water-holding capacity and concomitant high levels of aeration due to the cellular nature of coir, rooting of plants is very good.^[7]

Due to environmental concerns and thus pressure on reduced peat usage, there is a strong sentiment on total peat replacement or dilution. Materials that are being used for this are mainly coir, wood fiber, bark, and composted green waste. The addition of these materials to peat can lead to increase in certain nutrient levels, e.g., K in case of composted green waste. The addition of composted green waste can also lead to increase in pH and exchangeable calcium and micronutrients.^[8] In these cases, there can be savings on the nutrients and limestone to be added. However, the addition of some of these materials, e.g., wood fiber, sawdust, and bark can lead to N immobilization.^[4] These materials are generally not as stable as peat. Therefore, stability^[9] and N immobilization measurements should be made before they are added to peats for use as growing media.

NEW DEVELOPMENTS

Despite the high value of peat as a growing substrate, there is a concern with the fact that peat comes from peatland ecosystems, which are important ecosystems for a wide range of wildlife habitats, for water quality and cycle and carbon sequestration. Consequently, a “wise use” approach of peat usage has been adopted, and extensive research is carried out on sphagnum farming and peat bog rehabilitation to balance conflicting demands on peat land use. As a partial or complete replacement for peat, composted materials such as biowaste and bark, coir, and wood fibers will be increasingly used.^[10] The overall increasing use of growing media has resulted in the world demand for peat continuing to grow, despite substantial efforts to find alternatives, and hence, this “wise use” approach is appropriate and much needed. There has been an increasing interest in biochar and hydrochar. Both are lightweight materials and can be produced from many organic feedstocks. Biochar is a form of charcoal, which is manufactured by pyrolysis and can be used as a growing medium or a growing media component. It has a high stability and can have low EC, which is essential for its use as a growing medium. One potential innovative technology is hydrothermal carbonization (HTC). The final product is similar in structure to biochar produced by pyrolysis, however, in contrast to biochar production; the HTC process requires just moderate

temperatures and pressures and is usually used for materials with high water content, e.g., plant materials and waste.^[10] With HTC, heavy metals and EC in the waste and the solid fraction are reduced during processing,^[11] but the presence of volatile fatty acids can render them toxic to seedlings and young plants, which be overcome by simple pretreatment.

TESTING OF GROWING MEDIA

As seen in the previous section, numerous raw or processed products have been offered as a growing medium or as a component of growing media. Often they are waste products from the agricultural and forestry industry, e.g., coir, green waste, wood fiber, and sawdust. Some of these materials have to be processed further by composting (green waste) or leaching (coir). Laboratory tests are necessary to check the suitability of materials for quality control and for fertilization advice. The tests used are chemical, physical, and biological. There has been much advancement in the testing of growing media, and new methods have been developed, e.g., pH, EC, available nutrients, stability, and phytotoxicity, some of which differ from those used in conventional soil analysis. Many of these have been developed under the auspices of the European Committee for Standardization involving cooperative testing among a number of European laboratories. A good example is the development of stability and phytotoxicity tests for growing media and soil improvers.^[9] For physical characteristics, methods limiting disturbance are also prescribed to limit any effect on the properties.^[1] Methods that compact samples prior to analysis by the use of a standard weight for compaction or that apply a suction prior to measuring the characteristics of potted substrates are in use. The many steps following the processing and mixing of the substrate, which involves loosening, filling, and planting in various containers, may affect the resulting physical properties, and even if methods exist that compact samples to a bulk density similar to that found in the container, preferable approaches include putting instrumentation such as tensiometers, and water content probes directly into the container or weighing of potted substrates should be preferred. Permeameters should also be used on potted substrates. Driving cores into the substrates is problematic and extracting material from a container and trying to repack cores is unlikely to achieve a representative sample. Hydraulic processes linked to water storage and exchange are partly a function of container capacity (the equivalent “field capacity” for substrate in container), itself related to container geometry as described elsewhere.^[1] Hence, drainage will depend very much on container configuration and in practice, substrate water pressure will typically be in the range of -0.5 to -1.5 kPa after saturation and drainage,

clearly different from matric potential values encountered in agricultural fields (-10 to -33 kPa). As a result, characterization of volumetric water content at container capacity, and of air-filled porosity, should be performed directly on the pot equilibrated at container capacity. This also applies for unsaturated hydraulic conductivity and gas diffusivity determination. Such measurements must be repeated at specific intervals in the container itself inasmuch as possible to take into account the dynamic changes of physical properties that are occurring in growing media, the rooting effect and the effect of container configuration.^[1]

CONCLUSION

A major advantage of growing media is that it provides a root environment relatively free of pathogens at the start of the crop cycle or which can contain beneficial micro-organism. They are made of organic or inorganic components, of variable price, quality, and availability. A high proportion of them are made of peat or coir but important successful progress have been made to increase the proportion of compost in them. When the growing medium has the correct physical properties and nutrient supply is close to optimum, the net results are large increases in yield and improved product quality. This has especially been the case when combined with improved growing methods, carbon dioxide supplementation, and computer environment control. However, cultivation systems are expensive to install and need a high degree of technical expertise on the grower side. Therefore, they have mainly found applications in the production of high-value crops of fresh vegetables or ornamentals under protection, and also in outdoor containerized nursery stock production. They expand to new fields of application like strawberry and raspberry production.

REFERENCES

1. Caron, J.; Price, J.S.; Rochefort, L. Physical properties of organic soil: Adapting mineral soil concepts to horticultural growing media and histosols characterization. *Vadose Zone J.* **2015**, *14* (6).
2. Gruda, N.; Qaryouti, M.M.; Leonardi, C. Growing media. In *Good Agricultural Practices for Greenhouse Vegetable Crops – Principles for Mediterranean Climate Areas*, Plant Production and Protection Paper 217; Food and Agriculture Organization of the United Nations (FAO): Rome, 2013; 271–302.
3. Gruda, N.; Schnitzler, W.H. Suitability of wood fiber substrates for production of vegetable transplants: I. Physical properties of wood fiber substrates. *Sci. Hortic.* **2004**, *100* (1–4), 309–322.
4. Gruda, N.; Tucher, S.V.; Schnitzler, W.H. N-immobilization of wood fiber substrates in the production of tomato

transplants (*Lycopersicon lycopersicum* (L.) Karst. Ex. Farw.) (La: Ger.). J. Appl. Bot. **2000**, 74 (1–2), 32–37.

5. Maher, M.J.; Prasad, M. The effect of N source on the composting of green waste and its properties as a component of a peat growing medium. In *Proceedings of the International Conference Orbit 2001 on Biological Processing of Waste*, Seville, May 9–12, 2001; Spanish Waste Club, Orbit Association: Seville, 2001; 299–306.
6. Solbraa, K. Composting of bark: Potential growth reducing compounds and elements in bark. Medd. Norsk I. Skogfor. **1979**, 34, 443–508.
7. Dyke, M.J. The development of lignocell coir as a propagating medium. Comb. Proc. Intl. Plant Prop. Soc. **1994**, 44, 150–153.
8. Prasad, M.; Maher, M.J. Botanic composted material and spent mushroom compost as a micronutrient source for peat diluted growing media. In *Proceedings of the 4th International Conference ORBIT*, Weimar, Germany, Sept. 13–15, 2006, European Compost Network ECN e.V.; 441–446.
9. Prasad, M.; Baumgarten, A.; Grunstaedl, U.; Ni Chualain, D. Recent developments in the measurement of stability and

maturity of compost in Europe. In *Conference Orbit 2010, 7th International Conference*, Heraklion, Greece, 2012; 894–901.

10. Gruda, N. Current and future perspective of growing media in Europe. Acta Hortic. **2012**, 960, 37–44.
11. Reza Taufiq, M.; Andert, J.; Wirth, B.; Busch, D.; Pielert, J.; Lynam, J.G.; Mumme, J. Hydrothermal carbonization of biomass for energy and crop production. Appl. Bioenerg. **2014**, 1, 11–29.

BIBLIOGRAPHY

1. Baker, K.F. The U.C. system of producing healthy container grown plants. Calif. AES Bull. **1957**, 23, 331.
2. Lawrence, W.J.C.; Newell, J. *Seed and Potting Composts*; Allen and Unwin Ltd: London, 1952; 163 pp.
3. Puustjarvi, V. *Peat and Its Use in Horticulture*; Turveteollisuusliitto Ry, Publication 3: Helsinki, 1973.
4. Raviv, M.; Lieth, J.H., Eds. *Soilless Culture, Theory and Practice*; Elsevier: Amsterdam, 2008; 565 pp.

Gulley Erosion

Fenli Zheng

*Institute of Soil and Water Conservation, Northwest A&F University,
Yangling, China*

Chihua Huang

*U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
West Lafayette, Indiana, U.S.A.*

Ximeng Xu

*Institute of Soil and Water Conservation, Northwest A&F University,
Yangling, China*

Abstract

Gully erosion is prevalent throughout the world and is defined as erosion in channels where runoff accumulates and removes soil from this channel area. Gullies can be divided into ephemeral gully and classical gully depending on the obvious differences in scale. New kinds of techniques have been developed for the dynamic gully erosion process research for generating high resolution topography. There is still a long way to go to predict gully erosion before successfully quantifying the gully dynamic evolution process and validating and calibrating suitable parameters for calculating gully erosion rate. Reasonable gully control practices can well prevent gully erosion in various environments.

INTRODUCTION

Gully erosion is defined as erosion in channels where runoff water accumulated and removes soil from this channel area.^[1] Gully erosion is prevalent throughout the world. In 1939, Bennett^[2] estimated that 200 million active gullies were in existence in the United States. These gullies range in depths from 0.3 to 0.6 m to as much as 50 to 100 m. The severe erosion in 1930s prompted the U.S. government to take actions to control erosion, and successful results are well known all over the world. Gullies remove portions of land from production completely and fragmentize the landscape making it difficult for vehicles or farm machinery to move across. Gully erosion significantly reduces land quality and value and also contributes to downstream sedimentation problems.^[3] Despite the fact that a gully can be viewed as a part of the natural drainage system, it is different from a stream channel due to its intermittent flow triggered only by rainfall events.

Gullies often develop from intense erosion caused by flow over a steep overfall at the top of the gully. This overfall, called a headcut, moves upstream in a natural drainage way, and it can be initiated off-site and move into a field. Gullies can also be enlarged by lateral erosion, sloughing of their sidewalls, and cleaning out of debris by flow in the gullies. Subsurface flow through the gully walls can significantly reduce soil strength and accelerate gully erosion.^[3]

Depending on the scale, gullies are divided into two types—ephemeral gully (Figs. 1 and 2) and classical gully (Figs. 3 and 4). Ephemeral gullies are wider and deeper than rills, but they can be tilled across and filled in partially or completely. Although the ephemeral gully can be filled in or obliterated by tillage operations, it tends to reappear later at the same location because the depression it has formed on the landscape will concentrate the runoff. Classical gullies are eroded channels too large to cross and obliterate with tillage equipment. Table 1 shows the comparative characteristics of rill erosion, ephemeral gully erosion, and classical gully erosion.^[4]

EPHEMERAL GULLY EROSION

Ephemeral gully erosion is referred to as concentrated-flow erosion, mega-rill erosion^[4] or shallow gully erosion.^[5] The ephemeral gully erosion is the process of overland flow converging into a few major natural waterways, detaching soil particles from the waterway boundary, and transporting sediments downstream. Ephemeral gullies are the main drainage systems for a field, and most runoff and sediment are discharged from fields through these channels. The contribution of ephemeral gully erosion varies geographically. The U.S. Department of Agriculture–Natural Resources Conservation Service (USDA-NRCS) estimates from 19 U.S. states (Table 2) showed that ephemeral gully



Fig. 1 Ephemeral gully in the United States.

erosion contributed to 17% of total soil loss in New York state and to 73% in Washington state.^[6,7] Ephemeral gully erosion also affects the loess belt and Mediterranean region of Western Europe and the Loess Plateau of China. Ephemeral gully erosion contributes to at least 10% of the total soil loss in loess area in Europe^[8] and 35% of the total soil loss in the Loess Plateau of China. Most erosion in ephemeral gully channels is from rainstorms soon after seedbed preparation. As time progresses through the growing season, soil consolidates and these channels become less erodible. Many ephemeral gullies begin at a third to halfway down the slope, and then extend and grow in both the directions.

The following factors have been identified for the development of ephemeral gullies: 1) critical slope length and slope gradient that are dependent on slope characteristic and crop row direction and vegetation cover; 2) occurrence and depth of a fragipan; 3) agricultural practices, principal row direction, and timing of cultivation; and 4) timing and total amount of precipitation.^[7,9]

CLASSICAL GULLY EROSION

Classical gullies are those that are too large to be filled in or obliterated through tillage operations. The qualities of sediment moved from gully system in loessial regions are highly variable. Gully erosion contributed 10–30% of

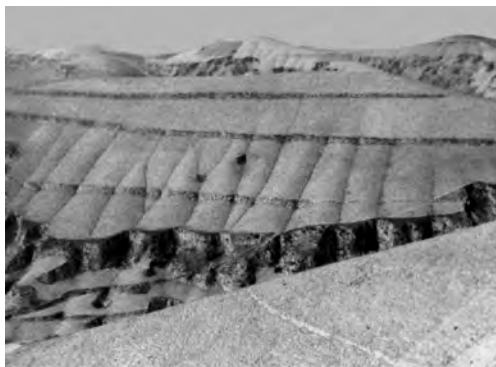


Fig. 2 Ephemeral gully in China.



Fig. 3 Classical gully in China.

annual sediment yield in the deep loess of western Nebraska from 1937 to 1952.^[10] At the Loess Plateau of China, gully erosion contributes about 60% of total sediment yield in the hilly gully region and 90% of total sediment yield in the high plateau with deep cutting and large gully region (Fig. 3). The contribution of gully erosion to total sediment yield also changes with year, dependent upon annual precipitation and the timing of storms. Gully channel formation is very rapid during the period of gully initiation, when morphological characteristics of a gully (such as length, width, area, and volume) are far from stable. This period is relatively short, about 5% of a gully's lifetime. For the most part of a gully's lifetime, its size is nearly stable, except near the headcut area.

Main factors affecting gully development in loessial regions are watershed size, runoff, the distance from the top of the gully to the watershed boundary, gully channel gradient, surface material, substratum moisture, and vegetation cover.

TECHNIQUES OF MEASUREMENT

Aerial photography is widely used for making accurate and repetitive measurements of gully development.^[11] Sequential stereo photographs taken at critical intervals provide excellent information for assessing gully development. Field survey is also used to quantify gully geometry. Nevertheless, measuring gully geometry, especially at different



Fig. 4 Classical gully in the United States.

Table 1 Comparative characteristic of rill erosion, ephemeral gully erosion, and classical gully erosion.^[4]

Rill erosion	Ephemeral gully erosion	Classical gully erosion
Rills are normally erased by tillage, do not usually reoccur in the same place.	Ephemeral gully are temporary features, usually obscured by tillage; recur in the same location.	Gullies are not obscured by normal tillage operations.
May be of any size but are usually smaller than ephemeral gullies.	May be of any size but are larger than rills and smaller than the permanent gullies.	Usually larger than ephemeral gullies.
Cross-section tends to be narrow relative to depth.	Cross-section tends to be wide relative to depth. Sidewalls are frequently not well defined. Headcuts are usually not readily visible and are not prominent because of tillage.	Cross section of many gullies tends to be narrow relative to depth. Sidewalls are steep. Headcuts are usually prominent.
Flow pattern develops as many small disconnected parallel channels ending at ephemeral gullies, terrace channels, or where deposition occurs. They are generally spaced and sized.	Usually forms a dendritical pattern along depression watercourses, beginning where overland flow including rills converge. Flow patterns may be influenced by tillage, crop rows, terraces, or other man-related features.	Tend to form a dendritical pattern along natural watercourses. Non-dendritical patterns may occur in road ditches, terrace, or diversion channels.
Occurs on smooth side slopes above drainage ways.	Occurs along shallow drainage ways upstream from incised channels or gullies.	Generally occurs in well-defined drainage ways.
Soils is removed in shallow channels, but annual tillage causes the soil profile to become thinner over the entire slope.	Soil is removed along a narrow flow path, typically to the depth of tillage layer where the untilled layer is resistant. Soil is moved into the voided area from adjacent land by mechanical action (tillage) and sheet and rill erosion, damaging area wider than the eroded channels.	Soils may be eroded to depth of the profile and can erode into soft bedrock.

Table 2 Assessment of ephemeral gully erosion in selected areas of the United States.

Location	Estimated sheet and rill erosion (tons ha ⁻¹ yr ⁻¹)	Measured ephemeral gully erosion (EGE ^a) (tons ha ⁻¹ yr ⁻¹)	EGE as a percentage of total (sheet, rill, and gully erosion) (%)
Alabama	5.73	3.42	37
Delaware	0.38	0.93	71
Illinois	2.61	1.91	42
Iowa	3.53	1.10	24
Kansas	8.07	2.94	27
Louisiana	6.54	2.22	25
Maine	4.12	1.89	31
Maryland	1.95	1.47	43
Michigan	1.72	0.45	21
Mississippi	6.46	2.75	30
New Jersey	2.46	1.91	44
New York	8.73	1.85	17
North Dakota	2.77	1.30	32
Pennsylvania	0.93	0.65	41
Rhode Island	3.31	1.36	29
Vermont	1.65	2.24	58
Virginia	4.77	4.70	50
Washington	0.25	0.69	73
Wisconsin	2.89	1.54	35

^aEGE is ephemeral gully erosion.**Source:** From USDA-NRCS.^[6] ©1977 USDA-NRCS and Robinson, Bennett, et al.^[7] ©1998 American Society of Agricultural Engineering.

times to monitor the gully development, is laborious. From 2005, new remote-sensing techniques have been developed for gully erosion research for generating high-resolution topography. They are the unmanned aerial vehicle (UAV),^[12] the time-of-flight (ToF) cameras (or range cameras),^[13] total station and Real Time Kinematic (RTK) survey, overlapping photographs (stereo pair),^[14] airborne and terrestrial light detection and ranging (LiDAR).^[15]

The UAV, commonly known as a drone, is an aircraft without a human pilot on board. It has integrated autopilot technology, which gives it semi or fully autonomous navigation, flight control, and image acquisition capabilities.^[12] The ToF cameras are a new generation of active sensors, which allow the acquisition of three-dimensional (3D) point clouds without any scanning mechanism and from just one point of view at video frame rates. The working principle is the measurement of the ToF of a signal emitted by the device toward the object to be observed, with the advantage of simultaneously measuring the distance for each pixel of the camera sensor.^[13] A total station is an electronic/optical theodolite integrated with a distance meter whereby coordinates of an unknown point relative to a known coordinate can be determined as long as a direct line of sight can be established between the two points. The basic concept of RTK is to have a base station receiver set on a point with known coordinates in the project site. Overlapping photographs (stereo pair) of a certain area of interest are acquired. Control targets within the stereo pair are surveyed with an electronic total station. Bearings (horizontal and vertical angles) to the targets from multiple stations are converted to 3D coordinates using intersection and

triangulation methods.^[14] Ground-based LiDAR systems provide detailed multitemporal analysis of microtopography. With LiDAR, it is possible to obtain digital terrain models, estimate soil roughness, perform volume calculations, and terrain topography changes.^[15]

PREDICTION OF GULLY EROSION

To separately predict ephemeral gully erosion and gully erosion, topographic threshold models were first given all around the world. Critical topographic conditions, expressed by local slope gradient and drainage area, can well describe the difference between ephemeral gully erosion and gully erosion.^[16]

The ephemeral gully erosion model, a computer program to provide estimates of the average annual soil loss removed from a single ephemeral gully, was developed by the USDA-ARS and USDA Soil Conservation Service.^[17] However, it showed poor adaptability around the world,^[18–20] and seeking and developing suitable physical process-based model and topographical index-based model in various environment needs more research.^[21]

Two types of models have been developed to predict gully development: 1) Dynamic models to predict rapid changes of gully morphology at the initial period of gully development with active headcut and channel expansion, which are based on the solution of the equations of mass

conservation and gully bed deformation; and 2) Static models to estimate final geometric parameters of stable gullies, which are based on the assumption of final morphological equilibrium of a gully.^[22]

The dynamic and static models were verified on the data of gully morphology and dynamics from Yamal Peninsula, Russia, and New South Wales, Australia. However, shortcomings like several parameters of gully morphology needed to be input before running the model, makes it hard to be widely used in different erosion environments. There is still a long way to go to predict gully erosion before successfully quantifying the gully dynamic evolution process and validating and calibrating suitable parameters for calculating gully erosion rate.

CONTROL OF GULLY EROSION

There are different practices to prevent or mitigate ephemeral gully formation and erosion. Conservation tillage is an excellent soil conservation practice for preventing ephemeral gully formation in agricultural fields. In areas where ephemeral gullies have formed, vegetation control, such as grassed waterways and vegetation barriers are often used to reduce their further development. Sometimes, a retention structure is built within gully channels to trap sediment.^[7]

Because many classical gullies have become a permanent landscape feature, stabilizing the channel banks and



Fig. 5 Classical gully control.

headcut area is the primary goal. Establishment of vegetation along the channel banks and headcut area is most effective in preventing classical gully growth. For large gullies such as those at the Loess Plateau of China, terracing on the gully slope and building retention structures (like check dam) in the gully channels are widely used for controlling classical gully erosion (Fig. 5).

REFERENCES

- Hutchinson, D.E.; Pritchard, H.W. Resource conservation glossary. *J. Soil Water Conserv.* **1976**, *31* (4), 1–63.
- Bennett, H.H. *Soil Conservation*; McGraw-Hall: New York, 1939; 993 pp.
- Piest, R.F.; Bradford, J.M.; Wyatt, G.M. Soil erosion and sediment transport from gullies. *J. Hydr. Div.* **1975**, *101* (HY1), 65–80.
- Foster, G.R.; Lane, L.J.; Mildner, W.F. Seasonally ephemeral cropland gully erosion. In *Proceedings of Natural Resources Modeling Symposium*, Pingree Park, Oct 16–21, 1983; 463–465.
- Zhu, X. Soil erosion classification at the loessial region. *Acta Pedolog. Sin.* **1956**, *4* (2), 99–116.
- USDA-NRCS. *America's Private Land, a Geography of Hope*; USDA-NRCS: Washington, DC, 1977.
- Robinson, K.M.; Bennett, S.J.; Casali, J. Headcut dynamics and ephemeral gully erosion. In *ASAE Annual International Meeting*, Orlando, FL, July 12–16, 1998. American Society of Agricultural Engineering: St Joseph, 1998.
- Wijdenes, D.J.O.; Poesen, J.; Vandekerckhove, L.; Ghesquiere, M. Spatial distribution of gully head activity and sediment supply along an ephemeral channel in a Mediterranean environment. *Catena* **2000**, *39* (3), 147–167.
- Smith, L.M. *Investigation of Ephemeral Gullies in Loessial Soils in Mississippi*; U.S. Technical Report GL-93-11 Army Corps of Engineers; Geotechnical Laboratory, Engineer Research and Development Center, Vicksburg, 1990.
- Piest, R.F.; Spomer, R.C. Sheet and gully erosion in the missouri valley loessial region. *Trans. ASAE.* **1968**, *11* (6), 850–853.
- Thomas, A.W.; Welch, R. Measurement of ephemeral gully erosion. *Trans. ASAE.* **1988**, *31* (6), 1723–1728.
- Hugenholtz, C.H.; Whitehead, K.; Brown, O.W.; Barchyn, T.E.; Moorman, B.J.; LeClair, A.; Riddell, K.; Hamilton, T. Geomorphological mapping with a small unmanned aircraft system (sUAS): Feature detection and accuracy assessment of a photogrammetrically-derived digital terrain model. *Geomorphology* **2013**, *194*, 16–24.
- Piatti, D.; Rinaudo, F. SR-4000 and CamCube3.0 Time of Flight (ToF) cameras: Tests and comparison. *Remote Sens.* **2012**, *4* (4), 1069–1089.
- Wells, R.R.; Momm, H.G.; Rigby, J.R.; Bennett, S.J.; Bingner, R.L.; Dabney, S.M. An empirical investigation of gully widening rates in upland concentrated flows. *Catena* **2013**, *101*, 114–121.
- Perroy, R.L.; Bookhagen, B.; Asner, G.P.; Chadwick, O.A. Comparison of gully erosion estimates using airborne and ground-based LiDAR on Santa Cruz Island, California. *Geomorphology* **2010**, *118* (3–4), 288–300.
- Torri, D.; Poesen, J. A review of topographic threshold conditions for gully head development in different environments. *Earth Sci. Rev.* **2014**, *130*, 73–85.
- Woodward, D.E. Method to predict cropland ephemeral gully erosion. Special issue: Soil erosion modeling at the catchment scale. *Catena* **1999**, *37* (3–4), 393–399.
- Nachtergaele, J.; Poesen, J.; Vandekerckhove, L.; Wijdenes, D.O.; Roxo, M. Testing the ephemeral gully erosion model (EGEM) for two Mediterranean environments. *Earth Surf. Proc. Land.* **2001**, *26* (1), 17–30.
- Valcárcel, M.; Taboada, M.T.; Paz, A.; Dafonte, J. Ephemeral gully erosion in northwestern Spain. *Catena* **2003**, *50* (2–4), 199–216.
- Capra, A.; Mazzara, L.M.; Scicolone, B. Application of the EGEM model to predict ephemeral gully erosion in Sicily, Italy. *Catena* **2005**, *59* (2), 133–146.
- Daggupati, P.; Sheshukov, A.Y.; Douglas-Mankin, K.R. Evaluating ephemeral gullies with a process-based topographic index model. *Catena* **2014**, *113*, 177–186.
- Sidorchuk, A. Dynamic and static models of gully erosion. Special issue: Soil erosion modeling at the catchment scale. *Catena* **1999**, *37* (3–4), 401–414.

Gypsic Soils

Juan Herrero

Department of Soils and Irrigation, Laboratory for Agronomy and the Environment (DGA-CSIC), Zaragoza, Spain

Jaime Boixadera

Department of Agriculture, Livestock, and Fisheries, Government of Catalunya, Lleida, Spain

Abstract

Gypsic is applied in a broad sense to those soils whose behavior and appearance are dependent on their gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) content either in the entire profile or in a particular horizon. High amounts of gypsum in soils are more likely to occur in dry climates.

GYPsic SOILS IN THE WORLD

The world distribution of gyprock outcrops and the solubility of gypsum restrict the extensive occurrence of gypsic soils to the dry regions (Figs. 1 and 2). These soils are distinct and recognized by local populations, as well as by early pedologists, in areas such as Northern Africa, the Middle East, and Spain. Most soil classification systems have the formative element *Gyps*. The extent of soils with gypsic or petrogypsic diagnostic horizons^[1] in Africa, Asia, the Near East, Europe, and the United States has been estimated at 207 million ha,^[2] and broad gypseous areas also occur in Australia, Mexico, and South America. These soils and other gypsiferous surface formations interact in global cycles through calcium and the bicarbonate equilibria with the atmosphere and water,^[3,4] processes also involving respiration. Gypsic soils deserve an environmental valuation due to the pedodiversity protection and plant endemisms produced by selective forces not yet established.^[5]

OCCURRENCE OF GYPSUM IN SOILS AND THEIR RELATED PROPERTIES

The semisolubility of gypsum in water (~2.6 g/L) controls the occurrence of gypsum in soils. The soil solution or the water extracts of gypsic soils have an electrical conductivity of 2.2 dS/m or more if soluble salts with no common ions are present. The leaching of gypsum out of the profile is common in wet climates. The low solubility of gypsum enables its transport from gyprock or other primary sources, its redistribution in the soil by dissolution/precipitation, and its permanent presence as a significant soil component in dry environments or in specific geomorphic positions. Moreover, the size of some pedogenic or sedimentary

gypsum crystals allows their transport by wind from bare ground surfaces. Gypsum in soils can also result from other natural or artificial materials containing sulfur, e.g., rain, industrial products, or mine spoils, but in these cases gypsum is a minor soil component.

Gypsum, even in small quantities, can prevent clay dispersion and soil degradation. The effect is due to the displacement of sodium ion from the cation exchange complex of the soil by the calcium ions (Ca^{2+}) released by the gypsum. This is one of the reasons for the use of gypsum as a soil amendment not only for sodic soils but also to avoid soil sealing and crusting under irrigation with sodic or with low electrical conductivity waters.^[6] Gypsum is also a soil acidity ameliorant.^[7,8] Plants are not stressed by the osmotic potential generated by gypsum, and this fact together with the beneficial effect on crops with high calcium or sulfate requirements must be taken into account when land evaluation deals with gypsiferous soils.^[9] In gypsic soils, a constant osmotic potential generated by the gypsum exists over all the range of soil moisture contents; it may reach up to -80 kPa and it has a mild effect on plants quite different from the osmotic effect in saline soils because the last one is several times higher even for the very slightly saline soils (electrical conductivity in the saturation extract = 2–4 dS/m at 25°C).

HYPERGYPSIC SOILS

The name of hypergypsic was coined for a subsurface diagnostic soil horizon with 60% or more gypsum,^[2] and later reused with genetic meaning.^[10] Those soils having gypsum as a major component can also be named hypergypsic.^[11] These types of soils appear on gyprock outcrops or close to some other gypsum source, commonly in arid climates.^[3] In these conditions, gypsum is not transitory and

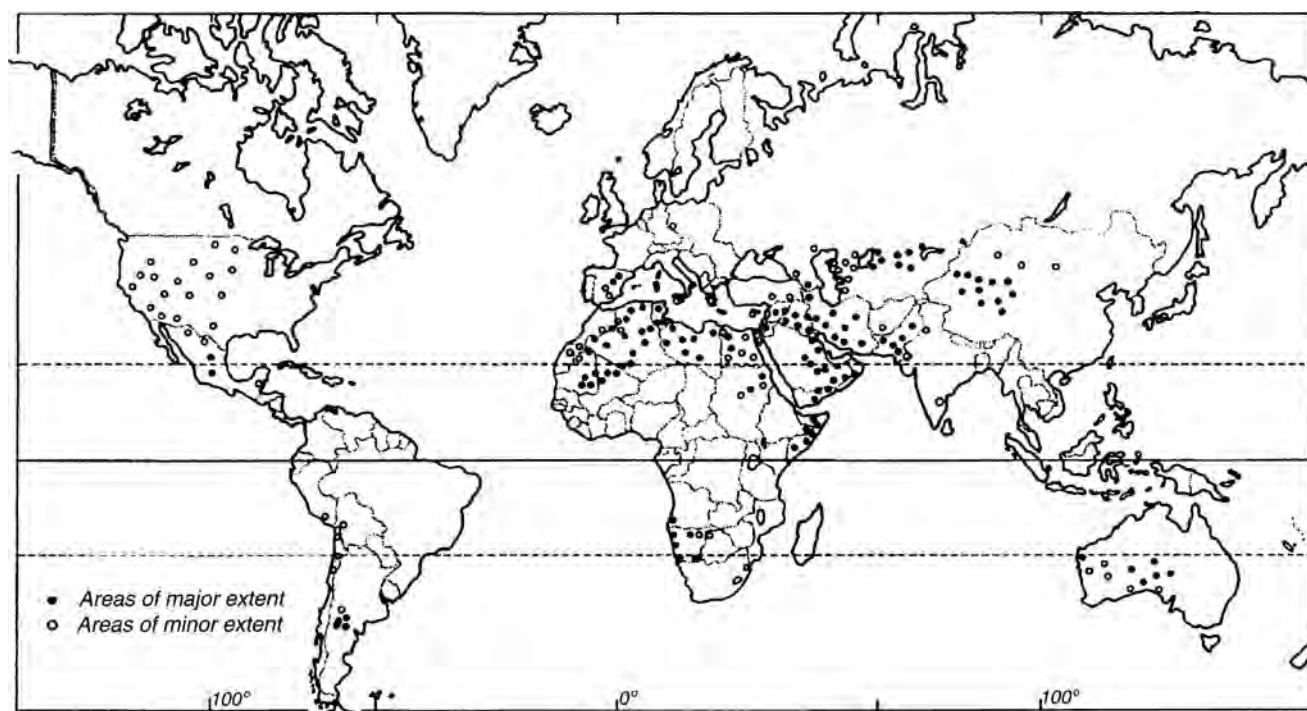


Fig. 1 World distribution of gypsic soils.

Source: Prepared from data contained by Eswaran & Zi-Tong,^[2] Alphen & Ríos,^[12] Boyadgiev & Verheye,^[13] Bridges & Batjes,^[14] FAO,^[15] Fierotti, Dazzi, et al.,^[16] Heinze & Fiedler,^[17] Jacob,^[18] Laya,^[19] Macau & Riba,^[20] Mashali,^[21] Mees,^[22] Nettleton et al.,^[23] Watson,^[24] many of the documents quoted by Herrero & Porta,^[3] and from authors' studies.

can constitute the soil groundmass, supporting pedological processes such as the redistribution of silty or clay-size carbonatic materials in the profile.^[25] The growth of gypsum crystals is displacive, resulting in a mixing of the soil

components: This can be illustrated by the disturbance and comminution of lutitic geological materials leading to C horizons^[26] or by the mechanical weathering of the parent gyprock.^[27] Many hypergypsic horizons show an isles



Fig. 2 Aerial view of the valleys carved with a fingered pattern in the gyprock surrounding Zaragoza (Spain). These valleys are flat-bottomed because of their filling during the Quaternary with materials from the soils on the slopes. Transversal dry-stone walls were built to control the erosion and to capture water in the bottoms. Deep gypsic soils there only permit poor barley yields even in the favorable years, because of the aridic soil moisture regime. The hills have shallow discontinuous gypsic soils, with shrub cover in the shaded slopes.

Source: Photo by O. Artieda, 1997.

fabric under the microscope, i.e., masses of non-gypsic fine material embedded in a gypsic pedofeature or the gypsic pedofeature can even be continuous producing a horizon with gypsic groundmass.^[11] In the field, these horizons feel either “gritty,” “flour-like,” or like “hard-bread crumbs” composed of sandy or coarser gypsum lentils, microcrystalline gypsum, or travertine gypsum, respectively.^[28,29] Some of these horizons can qualify for petrogypsic^[1,10] if a certain degree of cementation is attained. Commonly gypsic soils occur in Holocene surfaces, but the ones with cemented horizons have been found in Pleistocene surfaces.

For sand-sized or smaller grains, the distinction between pedogenic or geologic gypsum is often unfeasible even under the microscope.^[30] Moreover, this distinction lacks agricultural or environmental interest, at least in the gypsic soils in areas with ubiquitous gypsum. This makes it possible to argue against the incorporation of the distinction between primary and secondary gypsum into the concept of hypergypsic^[10] and even in the definitions of the gypsic and petrogypsic diagnostic horizons.^[1,10]

CONSTRAINTS TO THE USE OF GYPSIC SOILS

The presence of gypsum must not be confused with the osmotic effect of soil salinity, even though gypsic soils can also be saline, as is the case in playa- or sabkha-like^[31] environments or in some alluvial plains with incomplete drainage. In fact, gypsum is often used as an amendment in sodic soils to allow salt leaching.

Physical and Engineering Constraints

A large amount of gypsum in soils implies minor contents of clay minerals and other components having high water and nutrient retention capacity. Water is the main limitant to plants' life, thus in arid climates the soil water-holding capacity is decisive for life. In gypsic soils, this capacity is controlled by the gypsum grain size. Moreover, the soil exploration by roots can be hampered by massive gypsic layers.^[2] Drought is severe for plants in gypsic soils grazed or sown with barley or other annuals, except where irrigation or rain is frequent during the growing season.

Sparse shrubs or trees, also ants or other animals, often take advantage of cracks, or shallow karstic cavities in gyprock where lower temperatures and some moisture are available in arid environments.

Dissolved gypsum attacks concrete and iron, hindering, together with karstic phenomena, the construction of irrigation canals and tunnels. These problems plus the sinkholes by dissolution make flood irrigation of gypsic soils difficult or impossible;^[12] although sprinkler or drip irrigation techniques and the use of plastics in agriculture have overcome some of the obstacles to the irrigation of these soils. The water return flows from irrigated gypsic soils

are saturated with calcium and sulfate, limiting the urban or industrial reusability of these waters.

Fertility Constraints

Gypsic soils support forests or shrubs under appropriate climatic and management conditions and have been grazed and cultivated from times immemorial in a sustainable manner, but fertility is a matter of concern for intensive agriculture in gypsic soils. One favorable point is that, in contrast with other anions, no luxury consumption of sulfate by plants occurs, as was earlier reported^[32] from pot experiments. Sulfur is an essential element, and sulfate its main source for plants, but many different sulfur-based anions can occur in the soil depending on local or temporal conditions of pH, redox, microbial action, etc.^[33] making the assessment of the role of gypsum in soil fertility difficult. The relationship between sulfate and phosphate adsorption^[34] must be remembered, but probably the main fertility problems in gypsic soils are related to their low nutrient retention capacity and to the high amounts of Ca^{2+} in the soil solution (gypsum is over 100 times more soluble than calcite), reducing the availability of P for plants or inducing chlorosis in sensitive fruit trees. These and other constraints related to nitrogen and other nutrients in gypsiferous soils have been reviewed by FAO;^[2] this work also gives an overview on the effects of gypsum on several crops. Many knowledge gaps in this subject are due to the limited research conducted about fertility in gypsic soils and to the failure of some experiments to meet the analytical procedures^[35] required for gypsiferous soils.

REFERENCES

1. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; USDA Agriculture Handbook 436; USDA: Washington, 1999; 869 pp.
2. Eswaran, H.; Zi-Tong, G. Properties, genesis, classification and distribution of soils with gypsum. In *Properties, Characteristics and Genesis of Carbonate, Gypsum and Silica Accumulations in Soils*; Nettleton, W.D., Ed.; SSSA Spec. Publ. 26, SSSA: Madison, 1991; 89–119.
3. Herrero, J.; Porta, J. The terminology and the concepts of gypsum-rich soils. *Geoderma* **2000**, *96*, 47–61.
4. FAO. *Management of Gypsiferous Soils*. FAO Soils Bulletin 62; FAO: Rome, 1990; 81 pp.
5. Escudero, A.; Carnes, L.F.; Pérez-García, F. Seed germination of gypsophytes and gypsosvags in semi-arid central Spain. *J. Arid Environ.* **1997**, *36*, 487–497.
6. Amézketa, E. Soil aggregate stability: A review. *J. Sustain. Agr.* **1999**, *14*, 83–151.
7. Sumner, M.E. Gypsum and acid soils: The world scene. *Adv. Agron.* **1993**, *51*, 1–32.
8. Toma, M.; Sumner, M.E.; Weeks, G.; Saigusa, M. Long-term effects of gypsum on crop yield and subsoil chemical properties. *Soil Sci. Soc. Am. J.* **1999**, *63*, 891–895.

9. Laya, D.; Van Ranst, E.; Herrero, J. A modified parametric index to estimate yield potentials for irrigated alfalfa on soils with gypsum in quinto (Aragón, Spain). *Geoderma* **1998**, *87* (1–2), 111–122.
10. FAO. *World Reference Base for Soil Resources*, World Soil Resource Report 84; FAO: Rome, 1998; 88 pp.
11. Herrero, J.; Porta, J.; Fédoroff, N. Hypergypsic soil micromorphology and landscape relationships in N.E. Spain. *Soil Sci. Soc. Am. J.* **1992**, *56*, 1188–1194.
12. Alphen, J.G.; Ríos, F. *Gypsiferous Soils Notes on Their Characteristics and Management*; ILRI: Wageningen, 1971; 44 pp.
13. Boyadgiev, T.G.; Verheye, W.H. Contribution to a utilitarian classification of gypsiferous soils. *Geoderma* **1996**, *74*, 321–338.
14. Bridges, E.M.; Batjes, N.H. *Nachtergaele World Reference Base for Soil Resource: Atlas*; Acco: Leuven, 1998; 79 pp.
15. FAO. *World Soil Resources: An Explanatory Note on the FAO World Soil Resources Map at 1/25.000.000 Scale*; FAO: Rome, 1993; 64 pp.
16. Fierotti, G.; Dazzi, C.; Raimondi, S. Isuoli della serie gessoso solfifera: Bosco di mustigarufi. Guida alla escursione pedologica. *Boll. Società Sci. Suolo, Nuova Serie* **1993**, *4*, 63–131.
17. Heinze, M.; Fiedler, H.J. Chemische eigenschoaften von gips-rendzinen und begleitbodenformen des kyffhäuiser-gebirges (DDR). *Chemie Erde* **1984**, *43*, 65–75.
18. Jacob, J.S. Archaeological pedology in the maya low lands. In *Pedological Perspectives in Archaeological Research*; Collins, M.E., Carter, B.J., Gladfelter, B.G., Southard, R.J., Eds.; SSSA Sp. Publ. 44, SSSA: Madison, 1995; 51–80.
19. Laya, H. *Génesis y propiedades tecnológicas y posibilidades de utilización de los suelos salinos de la cuenca del duero y del tajo con énfasis en los suelos yesíferos*. PhD Thesis, Universidad Politécnica de Madrid: Madrid, 1989; 350 pp.
20. Macau, F.; Riba, O. Situación, características y extensión de los terrenos yesíferos en España. In *Primer Coloquio Internacional Obras Públicas en Terrenos Yes?íferos*; Servicio Geológico de Obras Públicas: Madrid, 1965; 28 pp.
21. Mashali, A.M. Soil management practices for gypsiferous soils. In *Proceedings of the International Symposium on Soils with Gypsum*. Catalonia, Spain, September 15–21, 1996. Poch, R., Ed.; Lleida, 1996; 34–52.
22. Mees, F. Distribution patterns of gypsum and kalistronite in a dry lake basin of the southwestern Kalahari (Omongwa Pan Namibia). *Earth Surf. Proc. Land.* **1999**, *24*, 731–744.
23. Nettleton, W.D.; Nelson, R.E.; Brasher, B.R.; Derr, P.S. *Gypsiferous Soils in the Western United States*; Kittrick, J.A., Fanning, D.S., Hossner, L.R., Eds.; SSSA Spec. Publ. 10, SSSA: Madison, 1982; 147–167.
24. Watson, A. Structure, chemistry and origin of gypsum crusts in southern Tunisia and the central Namib desert. *Sedimentology* **1985**, *32*, 855–875.
25. Herrero, J. *Morfología y Génesis de Suelos Sobre Yesos*, Monografía 77; I.N.I.A.: Madrid, 1991; 447 pp.
26. Artieda, O.; Herrero, J. Micromorphological features of lutitic Cr horizons in gypsiferous soils. In *Extended Abstracts of 6th International Meeting on Soils with Mediterranean Type of Climate*; Bech, J., Ed.; University of Barcelona: Barcelona, 1999; 527–529.
27. Artieda, O. Decimetric weathering features in the gyprock surface (Ebro Basin, Spain). In *Extended Abstracts of 6th International Meeting on Soils with Mediterranean Type of Climate*; Bech, J., Ed.; University of Barcelona: Barcelona, 1999; 524–526.
28. Artieda, O. *Génesis y Distribución de Suelos en un Medio Semiárido. Quinto (Zaragoza)*; Ministerio de Agricultura, Pesca y Alimentación: Madrid, 1996; 223 + map.
29. Herrero, J.; Artieda, O. Gypsum-rich horizons as seen in the field and under the microscope. In *Annual Meeting Abstracts*, A.S.A.-C.S.S.A.-S.S.S.A., Salt Lake City, UT, Oct. 31–Nov 4. Soil Science Society of America: Madison, 1999; 269 pp.
30. Stoops, G.; Poch, R.M. Micromorphological classification of gypsiferous soil materials. In *Soil Micromorphology: Studies in Management and Genesis*, Development in Soil Science, 22; Ringrose-Voase, A.J., Humphreys, G.S., Eds.; Elsevier: Amsterdam, 1994; 327–332.
31. Briere, P.R. Playa, playa lake, sabkha: Proposed definitions for old terms. *J. Arid Environ.* **2000**, *45*, 1–7.
32. Barbier, G. Action des chlorures et sulfates sur la nutrition minérale des plantes. *Comptes Rendues Acad. Agri.* **1937**, *123*, 699–706.
33. Pédro, G.; Robert, M. Les minéraux contenant du soufre, nature, cristalochimie, stabilité. In *International Symposium on Sulphur in Agriculture*, Versailles, Dec 1970; Institut National de la Recherche Agronomique: Paris, 1972; 137–180.
34. Kamprath, E.J.; Nelson, W.L.; Fitts, J.W. Influence of pH, sulfate and phosphate concentrations on the adsorption of sulfate by soils. *Soil Sci. Soc. Am. Proc.* **1956**, *20* (4), 463–468.
35. Porta, J. Methodologies for the analysis and characterization of gypsum in soils: A review. *Geoderma* **1998**, *87* (1–2), 31–46.

Gypsic Soils: Gypsum Formation

Ahmet R. Mermut

Department of Soil Science, University of Saskatchewan, Saskatoon, Saskatchewan, Canada

Hossein Khademi

Isfahan University of Technology, Isfahan, Iran

Abstract

Gypsiferous soils occur commonly in arid and semiarid regions with less than 400-mm annual precipitation. Over 200 million hectares of gypsiferous soils occur in the world. Gypsum and calcite often occur together in soil. They have a common ion (calcium), and the solubility of each of calcite and gypsum is determined by the presence of either mineral. The origin of sulfate ions in the soil solution can be different, in some circumstances, because of the presence of S-rich minerals such as sulfides in the parent materials. Gypsiferous soils are severely prone to soil erosion and the hydraulic conductivity of the soil is generally low, and gypsic horizon can also act as a barrier for root penetration. High amounts of gypsum also affects its soil fertility.

INTRODUCTION

Soils containing sufficient amount of gypsum to interfere with plant growth are generally defined as gypsiferous soils.^[1] These soils occur commonly in arid and semiarid regions with less than 400-mm annual precipitation.^[2] Many gypsiferous soils are found in densely populated areas, where irrigated agriculture is necessary for food production (Table 1).

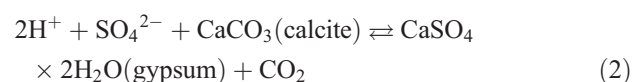
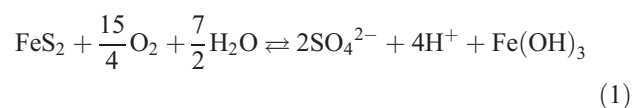
GENESIS AND ORIGIN OF SULFATE

Gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) is the dominant mineral in gypsiferous soils. Anhydrite (CaSO_4) is mainly associated with marine evaporites and is rapidly converted to gypsum when it is exposed to normal soil environment. Bassanite or hemihydrate ($\text{CaSO}_4 \cdot 0.5\text{H}_2\text{O}$) is an intermediate form between gypsum and anhydrite. These two mineral phases have been identified at soil surfaces as dehydration products of gypsum under highly dry conditions.^[3] Both pedogenic and inherited gypsum may be found in the soil.

Carter and Inskeep^[4] reported that pedogenic gypsum accumulation is high in soils with high content of total gypsum and relatively long period of soil formation. Gypsum is more soluble than calcite and normally occurs in: 1) a dry moisture regime; 2) regions with marked seasonal moisture fluctuations; and 3) a relatively short timescale, compared with that of calcite. Upward movement of water by capillary rise and subsequent evaporation are important factors responsible for the generation and accumulation of gypsum near the soil surface.^[5] Downward movement of

gypsum may occur following the incomplete wetting of the soil.^[6]

The origin of sulfate ions in the soil solution is, in some circumstances, a result of the presence of S-rich minerals such as sulfides in the parent materials. By weathering and oxidation, the sulfur in these minerals is transformed into sulfuric acid, which, in calcareous soils, reacts with calcite or dolomite to form gypsum.^[7] As shown in Eqs. 1 and 2:



The pedogenic gypsum of gravely gypsic horizons mostly occurs as pendants below pebbles (Fig. 1). In nongravely materials, it occurs as whitish, powdery or crystalline, soft masses, or diffused in the soil matrix at a certain depth (Fig. 2). The topographic setting strongly influences the quantity and the location of gypsum in the soil.^[8]

Geological formation is the origin of most gypsiferous soils particularly in Central Asia, Middle East, North Africa, the Southwestern United States, southern Spain, Northern Mexico, and Australia.^[9] Inherited gypsum can be redeposited by wind or water in new locations.^[3,5] Eolian deposits of gypsum are well known in the Southwestern United States, North Africa, and southeast Australia.^[5]

Watson^[8,10] uses the term *gypsum crust* to describe gypsum accumulation at or within about 10 m of the land

Table 1 Global distribution of gypsiferous soils.

Region	Gypsiferous soils (area × 1000 ha)
Africa	50,248
Asia	67,538
Australia	23
Europe	3,896
Middle East	76,721
North America	2,500
Total	200,926

Source: FAO.^[11]

surface from 0.1 to 5.0 m thick containing more than 15 wt. % gypsum and at least 5 wt.% more gypsum than the underlying bedrock. He also states that desert gypsum crusts are of three main types: 1) shallow-water evaporites; 2) subsurface crusts; and 3) surface crusts.

Gypsum and calcite often occur together in soil. They have a common ion (calcium), and the solubility of each of calcite and gypsum is determined by the presence of either mineral. As the partial pressure of carbon dioxide ($p\text{CO}_2$) affects directly the solubility of calcite, this can also indirectly affect the solubility of gypsum. Decreased acidity and SO_4^{2-} activity favor the formation of calcite at the expense of gypsum. An increased CO_2 partial pressure results in a shift of the calcite stability field into the gypsum field. Gypsum is too soluble to persist in soils, unless the SO_4^{2-} concentration is more than 10^{-2} M.^[12]

During evaporation of a solution containing Ca^{2+} , CO_3^{2-} , and SO_4^{2-} , calcium carbonate precipitates until saturation with respect to gypsum is reached. After that, both calcium carbonate and gypsum precipitate simultaneously and the composition of the solution remains unchanged. Coprecipitation of gypsum and calcite is unavoidable if a solution is evaporated to dryness, although the initial precipitate is a single mineral.^[13,14] As a result of saline groundwater migrating upward, less-soluble mineral (calcite) precipitates first and then coprecipitation of both



Fig. 1 Gypsum accumulation as pendants below gravels, from Central Iran.

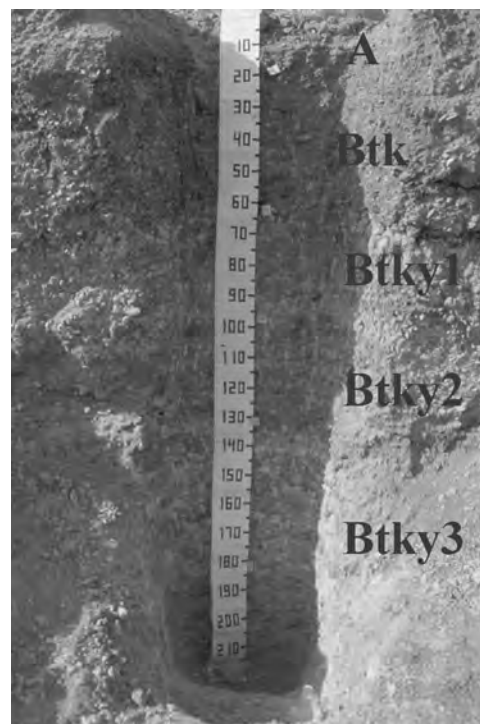


Fig. 2 Gypsum occurring as whitish, powdery or crystalline, soft masses in a calcic argigypsic soil from Central Iran.

minerals occurs. Because gypsum is more soluble, it can move a longer distance in soil solution and precipitates near the soil surface or inside the profile. The reverse sequence (descending mode) may also occur^[4] and is attributed to leaching. However, it seems that under arid conditions, leaching may not occur, except in some closed depressions.

GLOBAL DISTRIBUTION OF GYPSIFEROUS SOILS

Gypsiferous soils are found in arid and semiarid areas of the world. [Table 1](#) shows the worldwide distribution of gypsiferous soils in the world. The total amount in this table is more than double that was estimated in several previous publications. Over 100–150 million hectares areas of gypsiferous soils occur only in the Middle East and Central Asia. Iran alone has more than over 25 million hectares of gypsiferous soils; therefore the estimates in [Table 1](#) may still be an underestimate.

PROPERTIES OF GYPSIFEROUS SOILS

Where gypsum particles are present in the surface layer, their types, amount, and the degree of crystallization, the depth of gypsum layer, and its degree of induration have a profound impact on the physical and physicochemical properties of the soil. Gypsiferous soils are weakly aggregated because the cohesive forces attracting single soil



Fig. 3 Severely eroded gypsiferous soils from Central Iran.

particles are very weak.^[4] Consequently, gypsiferous soils are severely prone to erosion (Fig. 3).

In the presence of petrogypsic horizon, the hydraulic conductivity of the soil is generally low and this horizon can also act as a barrier for root penetration. Keren et al.^[15] reported that the reduction in hydraulic conductivity of the soils that contain gypsum is mainly because of mechanical plugging of soil pores by the small gypsum particles. When pedogenic gypsum accumulates in the subsurface horizons to the extent that the soil matrix becomes plugged, the growing gypsum crystals tend to interlock and indurate the horizon, causing a serious impediment to root growth. Abrukova and Isayev^[16] attribute the difficult workability in the soil, i.e., digging trenches, to the hardness of the gypsic horizon when it is dry. Gypsum reduces the total water reserve of the soil because gypsum itself has low water holding capacity.^[17]

High amount of gypsum in soils also affects its fertility. In addition, the availability of some of macronutrients such as Phosphorous (P), potassium (K), and magnesium (Mg) and also micronutrients such as molybdenum (Mo) is affected by the presence of gypsum. Gypsiferous soils require large quantities of nitrogen (N), P, and K for crop production.^[18] P is one of the micronutrients that is immobilized by gypsum in soil through the formation of different calcium phosphate compounds such as octacalcium phosphate and hydroxy apatite.^[9] The increase in Ca concentration in solution usually leads to a decline in K and Mg uptake by plants.^[5] Different plants have different ability to tolerate high levels of gypsum in the soil. Alfalfa (*Medicago sativa*), e.g., usually maintains an optimal ionic composition regardless of the gypsum content. In contrast, plants such as corn (*Zea mays*) are highly sensitive to gypsum.

Olsen and Watanabe^[19] reported that the added gypsum to sodic soils decreases Mo concentration and increases that of iron, manganese, and zinc in sorghum (*Sorghum bicola*). Singh and Taneja^[20] reported that

addition of gypsum to reclaim the salt-affected soils in India released $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ than control. Beneficial effects of gypsum on N mineralization in the soil may be a result of stimulation of microorganisms and/or neutralization of toxic salts.

CONCLUSION

Gypsiferous soils occur commonly in arid and semiarid regions with less than 400-mm annual precipitation. Over 200 million hectares of gypsiferous soils occur in the world. Gypsum and calcite often occur together in soil. They have a common ion (calcium), and the solubility of each of calcite and gypsum is determined by the presence of either mineral. The origin of sulfate ions in the soil solution can be different, in some circumstances, because of the presence of S-rich minerals such as sulfides in the parent materials. Gypsum is too soluble to persist in soils, unless the SO_4^{2-} concentration is more than 10^{-2} M.^[12] Gypsiferous soils are severely prone to soil erosion and the hydraulic conductivity of the soil is generally low, and gypsic horizon can also act as a barrier for root penetration. High amount of gypsum also affects its soil fertility. These soils require large quantities of N, P, and K. Under irrigation, variety of crops can be grown in gypsiferous soils.

REFERENCES

1. Sayegh, A.H.; Khan, N.A.; Khan, P.; Ryan, J. Factors affecting gypsum and cation exchange capacity determinations in gypsiferous soils. *Soil Sci.* **1978**, *125* (5), 294–300.
2. Nettleton, W.D. *Occurrence, Characteristics, and Genesis of Carbonate, Gypsum, and Silica Accumulations in Soils*; Soil Science Society of America (SSSA) Special Publication: Madison, 1991; Vol. 26, 149 pp.
3. Doner, H.E.; Lynn, C.L. Carbonate, sulfate, and sulfide minerals. In *Minerals in Soil Environments*; Dixon, J.B., Weeds, S.B., Eds.; SSSA Book Series; SSSA: Madison, 1989; Vol. 1, 279–330.
4. Carter, B.J.; Inskeep, W.P. Accumulation of pedogenic gypsum in western Oklahoma soils. *Soil Sci. Soc. Am. J.* **1988**, *52*, 1107–1113.
5. FAO. *Management of Gypsiferous Soils*; FAO Soils Bulletin; FAO: Rome, 1992; 81 pp.
6. Jafarzadeh, A.A.; Burnham, C.P. Gypsum crystals in soils. *J. Soil Sci.* **1992**, *43*, 409–421.
7. Mermut, A.R.; Arshad, M.A. Significance of sulfide oxidation in soil salinization in southern Saskatchewan, Canada. *Soil Sci. Soc. Am. J.* **1987**, *51*, 247–251.
8. Watson, A. Structure, chemistry and origins of gypsum crusts in southern Tunisia and the central Namib desert. *Sedimentology* **1985**, *32*, 855–875.
9. Muhammad, H.J.; Jones, K.C. Phosphorous in gypsiferous soils: The influence of soil properties on P fractionation. *Geoderma* **1992**, *53*, 97–104.

10. Watson, A. Desert gypsum crusts as palaeoenvironmental indicators: A micropetrographic study of crusts from southern Tunisia and the central Namib desert. *J. Arid Environ.* **1988**, *15*, 19–42.
11. FAO. *Management of Gypsiferous Soils*; Soils Bulletin; Food and Agricultural Organization of the United Nation FAO-UN: Rome, 1990; Vol. 62.
12. Lindsay, W.L. *Chemical Equilibria in Soils*; John Wiley & Sons: New York, 1979.
13. Wigley, T.M.L. Chemical evolution of the system calcite–gypsum–water. *Can. J. Earth Sci.* **1973**, *10*, 306–315.
14. Timpson, M.E.; Richardson, J.L.; Keller, L.P.; McCarthy, G.J. Evaporite mineralogy associated with saline seeps in southwestern North Dakota. *Soil Sci. Soc. Am. J.* **1986**, *50*, 490–493.
15. Keren, R.; Kreit, J.F.; Shainberg, I. Influence of size of gypsum particles on the hydraulic conductivity of soils. *Soil Sci.* **1980**, *130* (3), 113–117.
16. Abrukova, L.P.; Isayev, V.A. Structural–mechanical properties of gray-brown gypsiferous soils. *Sov. Soil Sci.* **1983**, *15* (6), 90–100.
17. Minashina, N.G.; Khamrayev, T.R.; Yallayev, S. Effect of gypsum in soils on cotton quality and yield. *Sov. Soil Sci.* **1983**, *15* (1), 34–40.
18. Eswaran, H.; Zi-Tong, G. Properties, genesis, classification, and distribution of soils with gypsum. In *Occurrence, Characteristics, and Genesis of Carbonate, Gypsum, and Silica Accumulations in Soils*; Nettleton, W.D., Ed.; SSSA Special Publication; SSSA: Madison, 1991; Vol. 26, 89–119.
19. Olsen, S.R.; Watanabe, F.S. Interaction of added gypsum in alkaline soils with uptake of iron, molybdenum, manganese, and zinc by sorghum. *Soil Sci. Soc. Am. J.* **1979**, *43*, 125–130.
20. Singh, B.R.; Taneja, S.N. Effects of gypsum on mineral nitrogen status in alkaline soils. *Plant Soil* **1977**, *48*, 315–321.

Hard-Setting Soils

Richard S.B. Greene

*Fenner School of Environment and Society, Australian National University,
Canberra, Australian Capital Territory, Australia*

Abstract

The problem of hard-setting soils is increasingly being recognized as a widespread form of land degradation in Australia and overseas. Hard setting is a characteristic of soil horizons, usually cultivated seedbeds, which contain unstable soil aggregates. The instability of surface horizons in Australia is usually related to low levels of soil organic matter and/or high levels of exchangeable sodium (often remaining from when the profile was affected by salinity). Under rapid wetting by either rainfall or irrigation, the aggregates collapse, the seedbed slumps, and upon drying, a hard, structureless mass of soil results. In surface horizons, it causes problems with infiltration, aeration, and seedling emergence. Lower in the profile, cohesive horizons, which are a form of hard setting, can also occur and cause problems of root elongation and water uptake. This entry discusses the occurrence of hard-setting soils, and the soil properties and processes that lead to the phenomena of hard setting. Methods for the assessment and management of hard-setting soils are also presented.

INTRODUCTION

Hard setting is a characteristic of soil horizons, usually cultivated seedbeds, which contain unstable soil aggregates. Under rapid wetting by either rainfall or irrigation, the aggregates collapse, the seedbed slumps, and upon drying, a hard, structureless mass of soil results. The change of structure is because of the breakdown of the soil aggregates under rapid wetting into microaggregates <250 μm diameter by a process called slaking.^[1] Hard-setting soils may also have poor physical properties because of the breakdown of their soil aggregates by the process of clay dispersion.^[2] Hard setting is a widely occurring problem, as most soils (apart from well-structured clays, organic soils, and sands) can undergo hard setting. Even though it is usually associated with the A horizon, it can occur in any horizon, thus causing a range of management problems. Mullins et al.^[3] and Blackwell^[4] have reviewed the topic of hard setting. This entry discusses the occurrence, formation, assessment, and management of hard-setting soils.

OCCURRENCE AND PROCESSES OF FORMATION

Generally, the term hard setting has been used only in Australia and was first used by Northcote^[5] in his soil-classification system. It is a widespread phenomena in Australian soils—e.g., in the new Australian classification developed by Isbell,^[6] of the 14 orders recognized, only the Organosols (soils with organic horizons), Podisols (deep

sands), and probably the Calcarosols (calcareous soils) can be generally assumed to be non-hard-setting soils. Internationally, the conditions also exist for hard-setting soils, but due to the lack of a common definition, they are harder to recognize. In Brazil, dense cohesive subsoil horizons occur at a range of depths from 0.2 to 1.42 m and may be considered to be similar to the hard-setting surface horizons described in Australia.^[7]

Even though the soil properties that predispose a soil to surface crusting and hard setting may be similar and involve the same processes (such as slaking and dispersion), the phenomena apply to different thicknesses of the seedbed. Crusting occurs when surface aggregates breakdown on wetting (usually under droplet impact) to form a hard, dense, structureless mass at the soil surface, often only a few millimeters thick. Hard setting, however, involves a much greater thickness of material, which commonly includes not only the A1 or Ap horizon but also the A2 (E) horizons, which may be bleached. Therefore, in some soils, the depth affected by hard setting may be up to 0.3 m or more.^[8] Thus, hard setting can cause different problems at different depths. For example, when it occurs at the surface horizon, infiltration and aeration are reduced, and seedling emergence is inhibited (Fig. 1). When it occurs lower in the profile in the form of cohesive horizons, it causes problems of root elongation and water uptake.

The processes required for the formation of hard-setting surface horizons are: 1) slaking of aggregates under rapid wetting; and/or 2) dispersion of the clay fraction. Although the worst possible situation results when both slaking and dispersion occur, slaking or dispersion alone will result in a



Fig. 1 A silty loam surface soil that has hard set following irrigation. Note the problems of seedling emergence.

soil developing hard-setting properties. The soil mass consisting of these breakdown products then increases in strength as the soil dries. For the genesis of cohesive sub-soil horizons, evidence by Bezerra et al.^[7] suggests a combined action of poorly sorted sands and fine clay argilluviation is required. The role of siliceous and silico-aluminous compounds as temporary cementing agents in cohesive and hard-setting horizons also needs to be considered.^[9] These cementing agents further exacerbate the problem of hard setting. As they are important in the formation of hard-setting surface soils, the processes of slaking and dispersion are discussed here in detail. The interparticle bonding in hard-setting soils is also outlined.

Slaking Under Rapid Wetting by Irrigation or Rainfall

Under the forces involved in rapid wetting, soil aggregates may disintegrate into smaller particles called microaggregates.^[1] The extent of breakdown depends on how the strength of the aggregate can withstand the stresses due to rapid wetting. The stresses include swelling of the clays, compression of entrapped air, release of the heat of wetting, and the mechanical action of the wetting.^[10,11] Slaking is therefore affected by the amount and type of clay minerals, the organic matter content, the initial moisture content, and the rate of wetting.^[12] For example, silty surface horizons that are frequently cultivated are commonly low in organic matter. Under rapid wetting, they undergo severe slaking and result in hard-setting horizons. The hard-setting soil in Fig. 1 is typical of such a soil. It has a silty clay loam texture, organic carbon content of approximately 1.5%, and is used for row cropping.

Dispersion

Dispersion is the process whereby the microaggregates formed by slaking undergo further breakdown into

individual clay particles or packets of clay particles known as quasicrystals.^[13] The amount of dispersion depends on the exchangeable sodium percentage (ESP) of the soil, the total electrolyte concentration of the soil solution, and the amount of mechanical agitation the soil receives during wetting.

Interparticle Bonding in Hard-Setting Soils

The material produced by slaking of hard-setting soils consists of a mixture of sand grains and fragments of aggregated material as well as clay particles or packets of clay particles.^[12] In soils that have undergone both slaking and dispersion, the released clays bridge between the sand grains and fragments of disrupted material to form a dense, structureless mass. Greene, Eggleton, and Rengasamy^[14] demonstrated that when soils were sodium (Na^+) saturated, the extent of hard setting increased as the smectite content increased. However, when calcium (Ca^{2+}) saturated, hard setting decreased as the smectite content increased. The difference was because of the formation of quasicrystals in the Ca^{2+} system, which formed an open, less-dense structure with less hard-setting properties than a Na^+ system.^[13] With cohesive horizons, there is a significant reduction in void sizes and interconnecting pores compared to non-cohesive horizons. This poorly connected void network leads to higher values of bulk density and penetrometer resistance.^[7]

ASSESSMENT OF HARD-SETTING SOILS

It is important to be able to assess the likelihood of a soil to set hard in the field and to characterize the severity of that hard setting. Some of the measurements used are: 1) modulus of rupture (MOR); 2) tensile strength; 3) extent of slaking into microaggregates; and 4) amount of clay dispersion. The first two measurements involve strength measurements on reconstituted soil discs, while the latter two involve subjecting natural soil aggregates to wetting and characterizing the amount of structural disintegration into microaggregates and clay particles, respectively. Details of the four measurements are as follows.

MOR

Aylmore and Sills^[15] used a combination of MOR and hydraulic conductivity measurements to classify hard-setting sandy loam soils of the western Australian wheat belt. Soils were classified as hard setting if their MOR had a value of ≥ 60 kPa. Hard-setting soils also exhibited a more rapid increase in MOR with increasing ESP (sodium sensitivity) compared with non-hard-setting soils.

Tensile Strength

Tensile strength measurements on remoulded cores of soil are usually carried out after saturating the exchange complex of the soil with Na^+ . Greene, Eggleton, and Rengasamy^[14] used measurements of tensile strength to characterize the effect of clay content, cation exchange capacity, and clay type on the hard-setting properties of a range of hard-setting soils used for horticulture in the Carnarvon district of Western Australia.

Extent of Slaking (Wet Sieving Measurements)

Wet sieving techniques are used to measure the portion of soil aggregates that disintegrate on wetting due to slaking into microaggregates $<250\text{ }\mu\text{m}$ in size. These microaggregates are important in clogging interaggregate porosity and initiating problems of hard setting and crusting. As the extent of slaking depends upon conditions such as the initial aggregate size, moisture content, and rate of wetting, it is important to standardize these parameters in the test. It is also important to relate the size range of the products of aggregate disintegration formed by laboratory wet sieving procedures to actual processes of structural decline and soil strength development in the field.

Measurement of Soil Dispersion

Rengasamy et al.^[2] used measurements of clay dispersion to predict if a soil was likely to undergo surface crusting processes in the field. The test can also be used to predict the likelihood of a soil undergoing hard setting. Because clay dispersion was measured under different conditions, i.e., spontaneous and mechanical situations, it was also possible to use this test to predict the effects of different management procedures, e.g., tillage techniques and types of rotation, on soil behavior. The test was particularly useful in predicting the likely response of the soil to ameliorants such as gypsum and/or lime and to irrigation with different qualities of effluent.

MANAGEMENT

As rapid and prolonged wetting of air-dried cultivated soils leads to severe slaking and formation of hard-setting soils, any management techniques that avoid these practices will minimize structural breakdown and hence the extent of hard setting. Therefore, the use of surface mulches, low-application rate microirrigation, and raised beds, all of which slow the rate of wetting,^[12] is important.

Soil organic matter is critical in controlling the structural stability of a soil and hence its tendency to hard set. Tillage practices such as intensive, continuous

cropping lead to low soil organic matter concentrations and concomitant low structural stability. Therefore, pasture phases, which increase soil organic matter, are critical for the structural improvement of hard-setting soils.

Sodic soils, i.e., those containing a level of exchangeable $\text{Na} \geq 6\%$, are prone to dispersion and formation of crusting surfaces and hard-setting properties in the profile. The application of gypsum to dispersive, sodic soils, is a very efficient way of preventing dispersion and thus the problems of hard setting.

CONCLUSION

Hard setting is shown to be one of the most serious forms of soil structural decline that can occur. Its effects on a range of soil processes such as infiltration, emergence, air–gas exchange, and waterlogging are critical in determining plant performance and productivity. These potential limitations to productivity can be avoided only by correct assessment, followed by appropriate amelioration and management practices. Further work is urgently needed to develop an international system for classifying hard-setting soils.

REFERENCES

1. Oades, J.M. Soil organic matter and structural stability: Mechanisms and implications for management. *Plant Soil* **1984**, 76, 319–337.
2. Rengasamy, P.; Greene, R.S.B.; Ford, G.W.; Mehanni, A.H. Identification of dispersive behaviour and the management of red–brown earths. *Aust. J. Soil Res.* **1984**, 22, 413–431.
3. Mullins, C.E.; MacLeod, D.A.; Northcote, K.H.; Tisdall, J.M.; Young, I.M. Hard-setting soils: Behaviour, occurrence, and management. In *Soil Degradation*; Lal, R., Stewart, B.A., Eds.; Adv. Soil Sci. 11; Springer-Verlag: New York, 1990; 37–108.
4. Blackwell, P.S. Slaking and hardsetting soils: Some research and management aspects. *Soil Till. Res.* **1992**, 25, 111–261.
5. Northcote, K.H. *A Factual Key for the Recognition of Australian Soils*; Rellim Technical Publications Pty Ltd: Coffs Harbour, 1960; 124 pp.
6. Isbell, R.F. *The Australian Soil Classification*; CSIRO Publishing: Melbourne, 1996; 144.
7. Bezerra, C.E.E.; Ferreira, T.O.; Romero, R.E.; Mota, J.C. A.; Vieira, J.M.; Duarte, L.R.S.; Cooper, M. Genesis of cohesive soil horizons from north-east Brazil: Role of argilluviation and sorting of sand. *Soil Res.* **2015**, 53, 43–55.
8. Isbell, R.F. Sealing, crusting and hardsetting conditions in Australian soils. In *Proceedings of the Second International Symposium on Sealing, Crusting and Hardsetting Soils: Productivity and Conservation*; So, H.B., Smith, G.D., Raine, S.R., Schafer, B.M., Loch, R.J., Eds.; ASSSI: Queensland, 1995; 15–30.

9. Chartres, C.J.; Kirby, J.M.; Raupach, M. Poorly ordered silica and aluminosilicates as temporary cementing agents in setting soils. *Soil Sci. Am. J.* **1990**, *54*, 1060–1067.
10. Emerson, W.W. Physical properties and structure. In *Soil Factors in Crop Production in a Semi-Arid Environment*; Russell, J.S., Greacen, E.L., Eds.; Queensland University Press: Brisbane, 1977; 78–104.
11. Collis-George, N.; Greene, R.S.B. The effect of aggregate size on the infiltration behaviour of a slaking soil and its relevance to ponded irrigation. *Aust. J. Soil Res.* **1979**, *17*, 65–73.
12. Chan, K.Y.; Mullins, C.E. Slaking characteristics of some Australian and British soils. *Eur. J. Soil Sci.* **1994**, *45*, 273–283.
13. Greene, R.S.B.; Posner, A.M.; Quirk, J.P. A study of the coagulation of montmorillonite and illite suspensions by calcium chloride using the electron microscope. In *Modification of Soil Structure*; Emerson, W.W., Bond, R.D., Dexter, A.R., Eds.; Wiley: New York, 1978; 35–40.
14. Greene, R.S.B.; Eggleton, R.A.; Rengasamy, P. Relationships between clay mineralogy and hardsetting properties of soils in the Carnarvon horticultural district of western Australia. *Appl. Clay Sci.* **2002**, *20* (4–5), 211–223.
15. Aylmore, L.A.G.; Sills, I.D. Characterization of soil structure and stability using modulus of rupture-exchangeable sodium percentage relationships. *Aust. J. Soil Res.* **1982**, *20*, 213–224.

Health

Robin F. Harris

Department of Soil Science, University of Wisconsin, Madison, Wisconsin, U.S.A.

D.E. Romig

Energy, Minerals and Natural Resources Department, State of New Mexico, Santa Fe, New Mexico, U.S.A.

Abstract

Historically, the “healthy soil–healthy people” concept made an important grassroots contribution to our reawakened interest in the key role that soil plays in the earth’s life-support system. Soil health may be viewed as the component of soil quality and ecosystem health that reflects the properties of soil as a living system. Although a healthy soil is an integral part of a healthy ecosystem, soil health plays a variable and sometimes negative role in soil quality, depending on the specific homocentric use of soil under consideration. The terms soil quality and soil health are converged but are not conceptually identical when the specific use of soil is to support ecosystem health.

SOIL HEALTH AND SOIL QUALITY

The terms soil health and soil quality have been used synonymously, at least from the standpoint that objective differences have not been clearly established.^[1–5] Soil quality is used almost exclusively by the scientific community, in part for brevity and in part because soil quality, concisely defined as “the degree of fitness of soil for a specific use” or “capacity of soil to function,”^[5–7] is felt to be less subjective, more quantifiable, and thus more defensible and less controversial than soil health (e.g., see Karlen et al.^[7]). In addition, integrating the concise and broader definitions of soil quality^[7,8] and the six landscape functions of soil^[9] identifies that soil quality is broader than soil health. Accordingly, for the three ecosystem functions, soil quality becomes the degree of fitness of soil to sustainably: 1) support biomass production and diversity; 2) protect environmental water and air quality by filtering, buffering, and transforming; and 3) maintain the earth’s genetic heritage; and, for the three “non-ecosystem” functions, soil quality is the degree of fitness of soil to act as sustainably as possible as a: 1) physical medium for infrastructures; 2) source of mining materials; and 3) keeper of the earth’s paleontological and our archeological heritage. The concept of soil health has little or no relevance to the three “non-ecosystem” functions, unless to rationalize a negative relationship between a healthy biologically active soil and the quality of soil to support these functions.

Despite the shift away from soil health to soil quality, there is general agreement that soil health appropriately conveys a more “soil as a living system” image as compared to the animate and/or inanimate “soil as a resource

for human development” focus of soil quality.^[2–7,9] Rather than force synonymy between soil health and soil quality, we suggest that soil health should be considered a component of soil quality that reflects the properties of soil as a living system and, within the context of ecosystem functions, can be defined separately as the degree of fitness of soil as a living system, within its natural means, to sustainably: 1) support biological production and promote plant and animal health; 2) maintain the earth’s genomic heritage; and 3) act as a living filter protector of water and air quality. The “within its natural means” qualification recognizes that the fitness of a soil to perform its ecosystem functions may be limited because of its inherent makeup, yet the soil may still be a resilient living system as long as it is managed within its capabilities. In practice, this definition of soil health converges with, but is not conceptually identical to, an exclusively ecosystem-based definition of soil quality.

SOIL HEALTH AND SOIL QUALITY WITHIN THE CONTEXT OF ECOSYSTEM HEALTH

Conflicts over the pros and cons of health as compared to quality concepts exist for managing rivers and watersheds as well as soil. In this regard, Karr and Chu^[10] persuasively rationalize that the concept of “ecological health,” as difficult as it is to define and measure, is an important policy goal with a greater opportunity to engage public support and interest than “ecological quality” because people more readily extend the idea of their own health to that of a watershed or ecosystem. As a component of ecosystem

health,^[11] soil health may be defined as the fitness of soil as a living system to sustainably help: 1) prevent ecosystem distress syndrome; 2) support ecosystem sustainability; and 3) protect the environmental quality of interfacing ecosystems. Ecosystem distress syndrome comprises a group of signs by which ecosystem breakdown is generally recognized, and for terrestrial ecosystems, it includes leaching of soil nutrients, reduced species diversity, shifts in species composition to opportunistic species, reduced productivity, and increased pest and disease loads.^[11] Fig. 1, with its

focus on the key role of soil as a biological reactor,^[13] illustrates the role of soil health in supporting biological production, biogeochemical cycling, energy flux, and environmental protection in an agroecosystem. For example, for a healthy soil: 1) agroecosystem distress syndrome is prevented by optimized biogeochemical cycling and a stable soil tilth restricting leaching and surface runoff loss; 2) agroecosystem sustainability is promoted by minimal external input of chemicals to the soil; and 3) the contamination of interfacing ecosystems is attenuated by the presence of

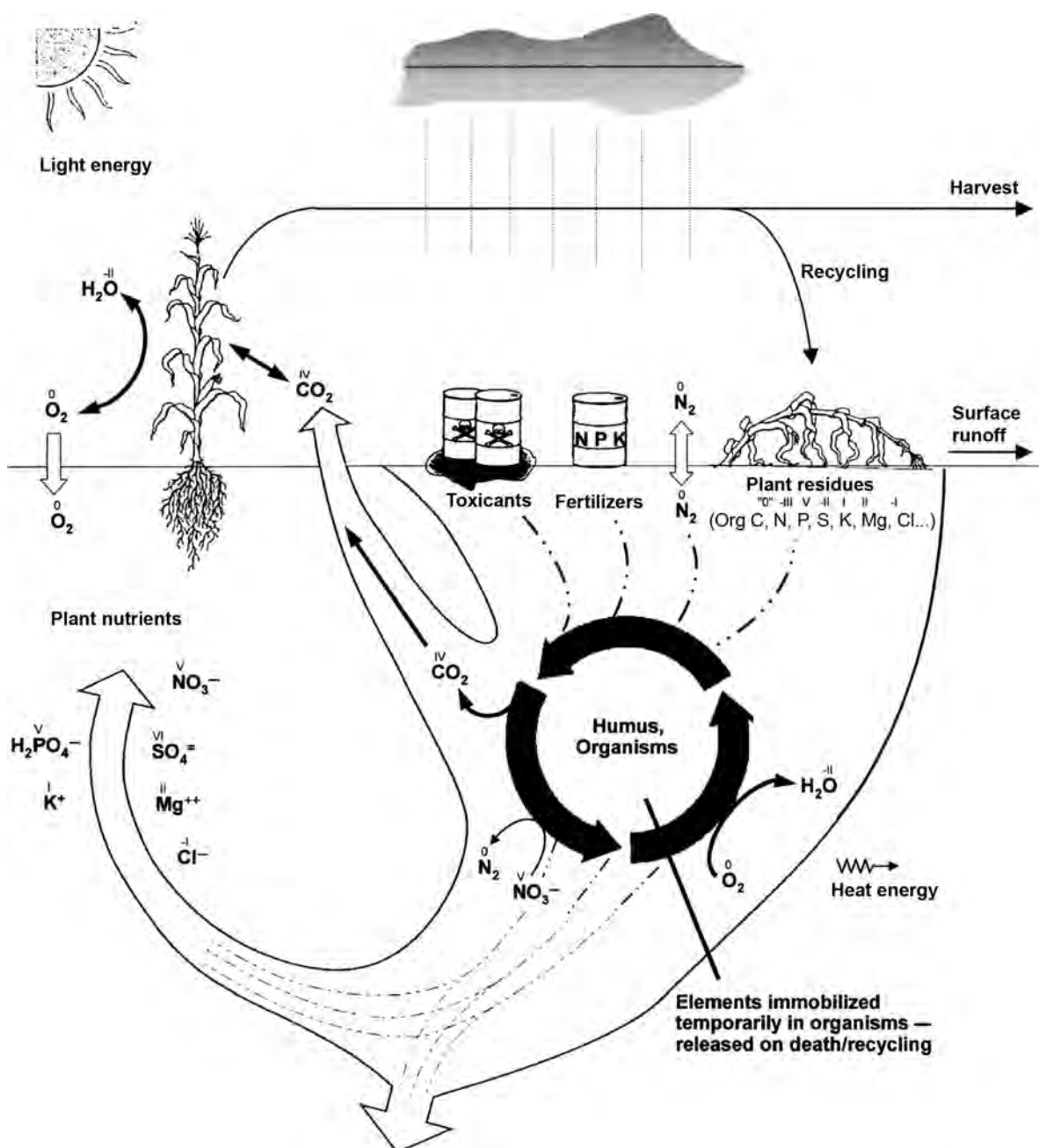


Fig. 1 Role of soil health in biomass production, biogeochemical cycling, energy flux, and environmental protection in an agricultural ecosystem. A Roman numeral above an element is the oxidation number of the element.

Source: Adapted from Romig, Garlynd, et al.^[12]

an effective living filter of soil and non-leakage of nutrients (such as nitrate) into groundwater by tightly coupled mineralization and plant root uptake. Similarly, contaminants entering the soil, either as a function of agronomic use or by accident, would ideally be integrated into the natural nutrient cycles and converted to harmless by-products for metabolism by soil organisms and higher plants (Fig. 1). Finally, it should be recognized that as a component of ecosystem health, soil health implicitly has built-in checks and balances. For example, because of its contribution to ecosystem distress syndrome and environmental contamination, rapid organic turnover and nitrogen mineralization resulting in soil organic matter depletion and excessive nitrate loss to the groundwater are not the characteristics of a healthy soil from an ecosystem health standpoint.

HOW DO YOU RECOGNIZE A HEALTHY SOIL?

As for other subsets of soil quality, the institutional approach to recognizing or assessing soil health involves describing each function on which soil health is to be based, selecting characteristics or properties that influence the capacity of the soil to carry out each function, choosing indicators of these characteristics, and using methods that can accurately measure these indicators.^[5]

Many of the results of the application of this approach to soil quality^[1-8] also apply to soil health and are not reviewed here.

Alternatively, building on tacit knowledge of how farmers recognize a healthy soil,^[1] a soil health scorecard was developed from structured interviews with Wisconsin farmers.^[12,14] The Wisconsin Soil Health Scorecard uses an ordinal, qualitative scoring mechanism for 43 soil health indicator properties to evaluate and monitor a soil's ability to support crop production. In addition to soil properties per se, farmers used plant, animal, human health, and to a lesser extent water properties to judge the health of their soils and relied heavily on sensory, descriptive criteria expressing how a healthy soil looked, felt, and smelled. Farmers commonly associated a sense of well-being and harmony with nature, with a healthy soil characterized by dark, rich earthy smelling humus, and friable, crumb structured soil tilth. However, it should be noted that the top 10 choices of farmers were more scientifically acceptable indicators of soil health, specifically, soil organic matter content, crop appearance, erosion, earthworms, drainage, tillage ease, soil structure, pH, soil nutrient test, and crop yield.^[12,14] The usefulness of integrating farmer-based, descriptive soil health indicators with analytical indicators in assessment of agroecosystem health remains to be seen.

REFERENCES

1. Harris, R.F.; Bezdicheck, D.F. Descriptive aspects of soil quality/health. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezedick, D.F., Stewart, B.A., Eds.; Special Pub. No. 35; American Society of Agronomy: Madison, 1994; 23-35.
2. Acton, D.F.; Gregorich, L.J. Understanding soil health. In *The Health of Our Soils Towards Sustainable Agriculture in Canada*; Acton, D.F., Gregorich, L.J., Eds.; Center for Land and Biological Resources Research, Research Branch, Agriculture and Agr-Food Canada: Ottawa, 1995; 5-10.
3. Doran, J.W.; Sarrantonio, M.; Leibig, M.A. Soil health and sustainability. *Adv. Agron.* **1996**, *56*, 1-54.
4. Pankhurst, C.E.; Doube, B.M.; Gupta, V.V.S.R. Biological indicators of soil health: Synthesis. In *Biological Indicators of Soil Health*; Pankhurst, C.E., Doube, B.M., Gupta, V.V.S.R., Eds.; CAB International: Wallingford, 1997; 419-435.
5. Carter, M.R.; Gregorich, E.G.; Anderson, D.W.; Doran, J.W.; Janzen, H.H.; Pierce, F.J. Concepts of soil quality and their significance. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 1-9.
6. Singer, M.J.; Ewing, S. Soil quality. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; G271-G298.
7. Karlen, D.L.; Mausbach, M.J.; Doran, J.W.; Cline, R.G.; Harris, R.F.; Schuman, G.E. Soil quality: A concept, definition, and framework for evaluation. *Soil Sci. Soc. Am. J.* **1997**, *61*, 4-10.
8. Harris, R.F.; Karlen, D.L.; Mulla, D.J. A conceptual framework for assessment and management of soil quality and health. In *Handbook of Methods for Assessment of Soil Quality*; Doran, J.W., Jones, A.J., Eds.; Special Pub. No. 49; American Society Agronomy: Madison, 1996; 61-82.
9. Blum, W.E.H.; Santelises, A.A. A concept of sustainability and resilience based on soil functions. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 535-542.
10. Karr, J.R.; Chu, E.W. *Restoring Life in Running Waters*; Island Press: Washington, 1998.
11. Rapport, D.J.; McCullum, J.; Miller, M.H. Soil health: Its relationship to ecosystem health. In *Biological Indicators of Soil Health*; Pankhurst, C.E., Doube, B.M., Gupta, V.V.S.R., Eds.; CAB International: Wallingford, 1997; 29-47.
12. Romig, D.E.; Garlynd, M.J.; Harris, R.F. Farmer based assessment of soil quality: A soil health scorecard. In *Handbook of Methods for Assessment of Soil Quality*; Doran, J.W., Jones, A.J., Eds.; Special Pub. No. 49; American Society Agronomy: Madison, 1996; 39-60.
13. Harris, R.F.; Arnold, S.M. Redox and energy aspects of soil bioremediation. In *Bioremediation: Science and Applications*; Skipper, H.D., Turco, R.F., Eds.; Special Pub. No. 43; American Society Agronomy: Madison, 1995; 55-87.
14. Romig, D.E.; Garlynd, M.J.; Harris, R.F.; McSweeney, K. How farmers assess soil health and quality. *J. Soil Water Conserv.* **1995**, *50*, 229-236.

Heat and Water Movement

Gerald N. Flerchinger

Northwest Watershed Research Center, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Boise, Idaho, U.S.A.

Abstract

The interrelation between heat and water flow in soil is complex. Temperature gradients can induce vapor and liquid water transfer within the soil; in turn, water movement carries heat with it, thus altering the thermal regime of soil. These interactions are often neglected to simplify analysis of thermal and/or moisture regimes of soil. However, many situations and processes require that soil heat and water flow be analyzed simultaneously. Soil freezing during cold season processes and water vapor movement in conjunction with soil surface evaporation are two examples where thermal and moisture processes are tightly linked and require simultaneous evaluation. These and other less obvious interrelations between soil heat and water are discussed.

THEORY OF SOIL HEAT AND WATER MOVEMENT

Temperature distribution in soil with provisions for water movement on heat transfer is given by the following equation:

$$C_s \frac{\partial T}{\partial t} - \rho_i L_f \frac{\partial \theta_i}{\partial t} = \frac{\partial}{\partial z} \left[k_s \frac{\partial T}{\partial z} \right] - \rho_l c_l \frac{\partial q_l T}{\partial z} - L_v \left(\frac{\partial q_v}{\partial z} + \frac{\partial \rho_v}{\partial t} \right) \quad (1)$$

The terms ($W m^{-3}$) represent, specific heat for change in energy stored due to a temperature increase; latent heat required to freeze water; net thermal conduction into a layer; net thermal advection into layer due to water flow; and net latent heat of evaporation within the soil layer, respectively. In Eq. 1, C_s and T are volumetric heat capacity ($J kg^{-1} C^{-1}$) and temperature (C) of the soil, respectively, t is time (s), ρ_i is density of ice ($kg m^{-3}$), L_f is latent heat of fusion required to freeze water ($J kg^{-1}$), θ_i is volumetric ice content ($m^3 m^{-3}$), z is depth within the soil (m), k_s is soil thermal conductivity ($W m^{-1} C^{-1}$), ρ_l is density of water ($kg m^{-3}$), c_l is specific heat capacity of water ($J kg^{-1} C^{-1}$), q_l is liquid water flow ($m s^{-1}$), L_v is latent heat of vaporization required to evaporate water ($J kg^{-1}$), q_v is water vapor transfer ($kg m^{-2} s^{-1}$), and ρ_v is vapor density ($kg m^{-3}$) within the soil. Three specific processes addressed by Eq. 1 in which heat and water movement are most closely coupled are soil freezing, water vapor movement, and thermal advection.

Soil Freezing

Due to negative water potentials, soil water exists in equilibrium with ice at temperatures below the normal freezing

point of bulk water and over the entire range of soil freezing temperatures normally encountered. When ice is present, soil water potential is a function of temperature.^[1] This relation is expressed as follows:

$$\phi = \pi + \psi = \frac{L_f}{g} \left(\frac{T}{T_K} \right) \quad (2)$$

where ϕ is total water potential (m), π is soil water osmotic potential (m), ψ is soil matric potential, g is acceleration of gravity ($m s^{-2}$), and T_K is absolute temperature (K). As temperature at the freezing front decreases, more and more water freezes, water potential becomes more negative, and liquid water content continues to drop, creating a gradient in water potential and liquid water content. This drop in liquid water content at the freezing front has a similar effect as drying of the soil, and water will migrate from moist regions to the freezing front. This often results in elevated ice contents, ice lenses, and frost heave.

Water Vapor Movement

Water vapor transfer can have a significant effect on the soil thermal environment due to the large latent heat of vaporization. Evaporation and subsequent vapor loss from the soil are very effective at cooling the soil. Vapor transfer in soil can be computed as the sum of flow due to a gradient in vapor density, q_{vp} , and that due to a temperature gradient, q_{vT} ,^[2] where:

$$q_v = q_{vp} + q_{vT} = -D_v \rho_v \frac{dh_r}{dz} - \zeta D_v h_r s \frac{dT}{dz} \quad (3)$$

Here D_v is vapor diffusivity ($m^2 s^{-1}$) in soil, h_r is relative humidity within the soil based on water potential, s is the slope of the saturated vapor pressure curve ($kg m^{-3} C^{-1}$),

and ζ is an enhancement factor. Observed vapor transfer in response to a temperature gradient exceeds that predicted by Eq. 3; therefore, an enhancement factor is included, which can vary from around 10 at high water contents to unity under very dry conditions.^[3]

Thermal Advection

As water moves through soil, it carries thermal energy with it. Liquid water moving from warm soil to relatively cooler soil will tend to increase the temperature of the cooler soil. The heat transported is based on heat capacity of water and is proportional to the liquid water flow, q_l , and temperature gradient, as expressed in Eq. 1. Under most conditions, water movement through soil is sufficiently slow that this term can be ignored.

Thermally Induced Liquid Flow

Variation in water surface tension with temperature and the heat of wetting can cause liquid flow in response to thermal gradients. Dependence of surface tension (or water potential) on temperature and the effect of thermal gradients on liquid water movement must be considered for non-isothermal conditions.^[4] As a result, the Richards' equation for water flow can become complex when generalized for non-isothermal conditions. Studies comparing simulations for surface soil evaporation with and without thermal effects show that liquid flow in response to thermal gradients accounted for only 1% of the evaporation.^[5] Thus, thermally induced liquid flow can usually be ignored.

EXAMPLES OF COUPLED HEAT AND WATER FLOW

Soil heat transfer due to soil freezing, water vapor transfer, and thermal advection is described by Eq. 1. The effects of these processes are illustrated using data collected in southwestern Idaho, U.S.A., and simulated by the Simultaneous Heat and Water model.^[6,7] The site had the following two soil types: a loamy sand and a silt loam.

Soil Freezing

The migration of water due to soil freezing is controlled primarily by the rate of freezing front advance in relation to the unsaturated hydraulic conductivity. With rapid soil freezing, soil water is essentially frozen in place, and there is little opportunity for water to migrate to the freezing front. Similarly, if the unsaturated conductivity is low, water migration to the freezing front will be slow. Thus, dry and/or coarse-textured soils exhibit much less frost-related water movement than moist, fine-textured soils.

Soil-water dynamics during soil freezing are illustrated for the silt loam soil in Fig. 1. For comparison, simulated

water content without considering freeze/thaw processes is also plotted. Elevated total water content due to water migration to the freezing front can be observed for all three depths, while the agreement between simulated and measured liquid water content was reasonable. Prior to day 344, simulated water content above 20 cm was decreasing due to drainage. After the initiation of soil freezing on day 344, the direction of flow reversed, and simulated water flow above 10 cm was upward toward the freezing front. As the 5-cm depth began to freeze on day 344, total water content began to increase, while simulated liquid water content continued to decrease. As the frost front advanced, the 5-cm total water content began to level off, and the 10-cm total water content began to level off, and the 10-cm total water content increased upon freezing on day 345. Subsequently, the 20-cm depth began to freeze on day 347.

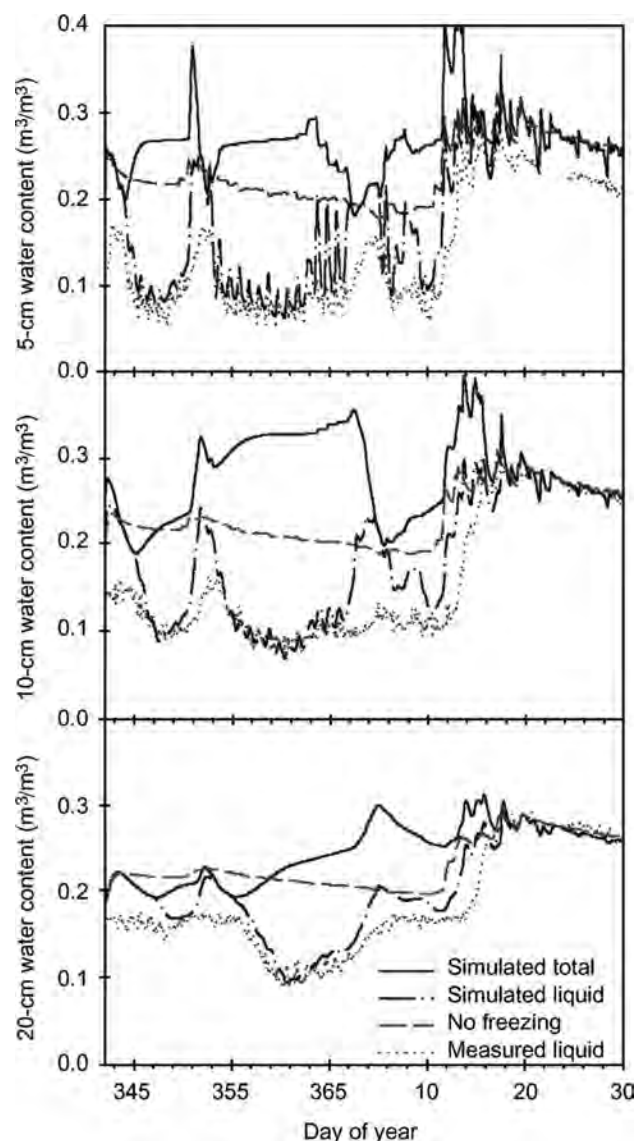


Fig. 1 Simulated total water content and simulated and measured liquid water content for a silt loam soil for the 5-, 10-, and 20-cm depths. Also plotted is simulated water content without considering freezing dynamics.

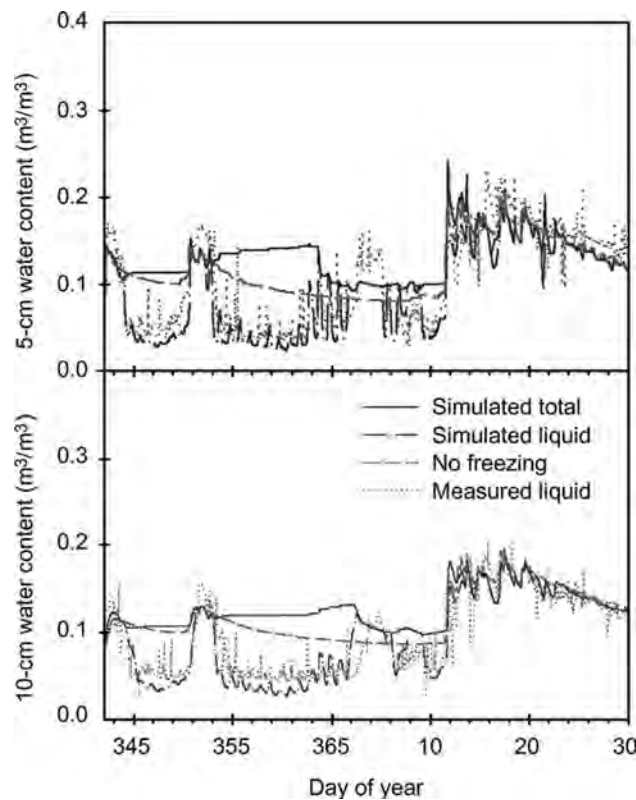


Fig. 2 Simulated total water content and simulated and measured liquid water content for a loamy sand soil for the 5- and 10-cm depths. Also plotted is simulated water content without considering freezing dynamics.

Soil–water dynamics for the loamy sand were considerably less responsive to freeze/thaw processes than the silt loam (Fig. 2). Due to the low unsaturated conductivity of the loamy sand, the simulated increase in total water content was much smaller compared to the silt loam. Even so, total water content in the frozen layer increased gradually, while water content of the simulation which ignored freezing processes continued to drain.

Soil–water dynamics during freezing and thawing are critical to predicting frozen-soil-related runoff. Although not measured, simulated runoff for the silt loam and loamy sand were 16 and 13 mm, respectively; no runoff was simulated when freezing was not considered.

Ignoring freeze/thaw processes had a minor, but not insignificant, effect on simulated soil temperature. Compared to the full freeze/thaw simulation, ignoring effects of soil freezing resulted in simulated soil temperatures differing by typically 2.2°C for the 5- and 10-cm depths. Differences between the loamy sand simulations were somewhat less because there was less water to freeze.

Water Vapor Movement

Under moist conditions, vapor transfer within soil is usually negligible due to the lack of air-filled pore space.

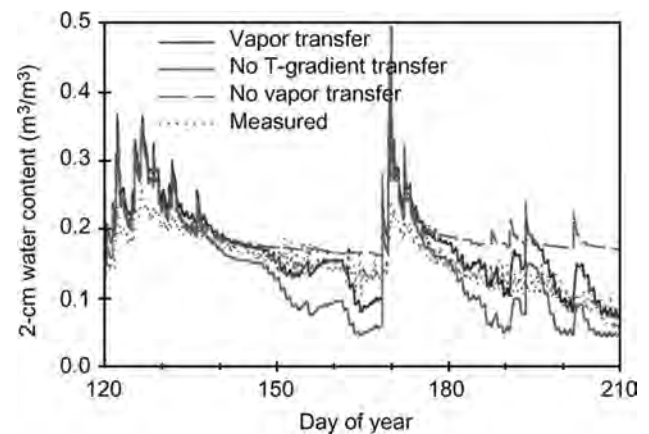


Fig. 3 Measured 2-cm water content for a silt loam soil and simulated 2-cm water contents: 1) considering water vapor transfer; 2) assuming no water vapor transfer; and 3) neglecting the effect of thermal gradients on water vapor transfer.

However, under drier conditions, vapor transfer can significantly contribute to soil water movement and has a large effect on soil thermal conditions. Water vapor movement is especially important for a situation where the surface soil is dry and underlain by moist soil.

Measured 2-cm soil water content is shown in Fig. 3 along with simulated water content for three simulations as follows: vapor transfer included, vapor transfer ignored, and vapor transfer due to thermal gradients (q_{vT}) ignored. The simulation that ignored all water vapor transfer overestimated water content at the 2-cm depth because liquid flow through the dry ($<0.05 \text{ m}^3 \text{ m}^{-3}$) soil above 2 cm was insufficient to dry the soil beyond 1 cm. Conversely, the simulation which ignored thermal gradient vapor transfer, q_{vT} , overestimated water vapor transfer and therefore resulted in excessive water loss. Typically, daytime temperature gradients inhibit vapor movement toward the surface and reduce evaporation. Simulated evaporation for the 90-day period was 111 mm for the silt loam soil when vapor transfer was considered, 105 mm with no vapor transfer, and 117 mm when q_{vT} was ignored. Simulated evaporation for the loamy sand soil was 101, 96, and 105 mm, respectively.

Differences in evaporation between simulations had a moderate effect on simulated soil temperature. On average, the simulation without water vapor transfer had soil temperatures approximately 0.5°C warmer due to reduced evaporation. However, the absolute difference between simulations was as much as 1.2°C because ignoring vapor transfer caused higher temperatures during the day and cooler temperatures at night.

Thermal Advection

Heat transfer by soil–liquid water flow is usually quite small compared to thermal conduction. Average absolute

thermal gradient near the soil surface simulated for the bare soil plots was 1.6 C cm^{-1} . For typical values of thermal conductivity, it would require 1.3 mm hr^{-1} of liquid water flow (q_l) to transfer an equivalent amount of heat within a 1-cm layer. Thus, the effect of thermal advection of heat becomes important only for rapid infiltration of water. Indeed, one-year simulations neglecting the advection term during infiltration resulted in near-surface soil temperatures differing by typically $0.3\text{--}0.4^\circ\text{C}$ compared to simulations in which heat of advection during infiltration was considered. Maximum hourly differences of over 10°C occurred due to summer rainstorms on a hot soil surface (40°C). When advective heat transfer was considered during infiltration but ignored during redistribution, the temperature was typically within 0.03°C of the simulation considering advection. Thus, advective heat transfer can be significant during infiltration, but not generally important during the redistribution of soil water.

CONCLUSION

Interrelation between water and heat flow within the soil can influence the soil temperature regime. This influence is most pronounced in the case of soil freezing and thawing, and water vapor movement during surface evaporation. With the exception of rapid infiltration of water, heat carried by liquid water flow can usually be ignored. Thermally induced liquid water flow is usually insignificant.

From a practical standpoint, although water vapor movement influences evaporation from the soil and in turn affects water availability for plant growth, ignoring

this process usually provides evaporation estimates within 10% of actual values. Soil freezing, however, can significantly influence infiltration and runoff, causing severe flooding from relatively mild rainfall or snowmelt events.

REFERENCES

1. Fuchs, M.; Campbell, G.S.; Papendick, R.I. An analysis of sensible and latent heat flow in a partially frozen unsaturated soil. *Soil Sci. Soc. Am. J.* **1978**, *42* (3), 379–385.
2. Campbell, G.S. *Soil Physics with BASIC: Transport Models for Soilplant Systems*; Elsevier: Amsterdam, 1985; 150 pp.
3. Cass, A.; Campbell, G.S.; Jones, T.L. Enhancement of thermal water vapor diffusion in soil. *Soil Sci. Soc. Am. J.* **1984**, *48*, 25–32.
4. Milly, P.C.D. Moisture and heat transport in hysteretic, inhomogeneous porous media: A matric head-based formulation and a numerical model. *Water Resour. Res.* **1982**, *18* (3), 489–498.
5. Milly, P.C.D. A simulation analysis of thermal effects on evaporation from soil. *Water Resour. Res.* **1984**, *20* (8), 1087–1098.
6. Flerchinger, G.N.; Saxton, K.E. Simultaneous heat and water model of a freezing snow-residue-soil system I. Theory and development. *Trans. Am. Soc. Agric. Engr.* **1989**, *32* (2), 565–571.
7. Flerchinger, G.N. *Simultaneous Heat and Water Model: Technical Documentation*; NWRC Technical Report 2000–09; USDA—Agricultural Research Service, Northwest Watershed Research Center: Boise, Idaho, 2000; 38. Available at <http://www.ars.usda.gov/SP2UserFiles/Place/20520000/ShawDocumentation.pdf>.

Heat Capacity

Surinder Kumar Jalota

B.S. Ghuman

Department of Soils, Punjab Agricultural University, Ludhiana, India

Abstract

Soil temperature is an important edaphic factor. It plays a significant role in influencing soil microclimate which, in turn, affects seed germination, seedling emergence, and subsequent crop development. Changes in soil temperature are governed by its thermal properties, viz. conductivity, diffusivity, and heat capacity, in addition to solar radiation. Of these properties, heat capacity of soil is the one that controls temperature fluctuations to a larger extent and refers to the amount of heat existing in it. It is estimated from the mass or volume fractions and respective specific heats of different soil constituents such as minerals, organic matter, water content, and air. Soil and crop residue management practices such as tillage, compaction, puddling, and straw retention or removal from the field are commonly used for modifying heat capacity of the soil and thus its temperature.

HEAT CAPACITY

Heat capacity, also called thermal capacity, is not measured per se. It is measured by its effect on soil temperature. When heat is supplied, soil temperature increases, and when the heat is released, the temperature decreases. The ratio of heat supplied to a soil to its corresponding temperature rise is called the heat capacity, C , and is given by

$$C = \frac{Q_h}{\Delta T} \quad (1)$$

where Q_h is the quantity of heat supplied to soil in calories, and ΔT is an increase in its temperature in $^{\circ}\text{C}$.

Heat capacity of a soil can be expressed on mass basis (C_m) or volume basis (C_v). Heat capacity per unit mass of a substance, also called specific heat, is defined as the quantity of heat required to raise the temperature of 1 g of the substance by 1°C and is given by

$$C_m = \frac{\text{Heat capacity}}{\text{Mass}} = \frac{Q_h/\Delta T}{m} = \frac{Q_h}{m\Delta T} \quad (2)$$

$$Q_h = mC_m\Delta T \quad (3)$$

where C_m is the specific heat of the substance in $\text{cal g}^{-1}^{\circ}\text{C}^{-1}$, and m is the mass in g. Eq. 3 implies that the quantity of heat required to produce a given temperature change depends on the mass and the specific heat of the substance being heated. The heat capacities on mass basis or specific heats of soils, minerals, and some other materials are given in Table 1.

Specific heat affects soil temperature to a greater extent. Everything else being equal, a soil with a high mass heat capacity exhibits lesser temperature change than does one

having a low heat capacity. The presence of water in soil increases its heat capacity as heat capacity of water is 1.0 cal g^{-1} and that of many soil-forming minerals is nearly 0.2 cal g^{-1} . Therefore, moisture tends to buffer the soil against a very rapid change in temperature; e.g., heat capacity of dry soil is 0.2 cal g^{-1} , and if its moisture content is increased to 20%, the heat capacity of this soil would become 0.33 cal g^{-1} , and so on. It implies that more heat is required to warm a moist than a dry soil. Heat capacity on mass basis can be estimated from the following equation:

$$C_m = X_s C_s + X_w C_w + X_a C_a \quad (4)$$

where X_s , X_w , and X_a are mass fractions of soil solids, moisture, and air, respectively, and C_s , C_w , and C_a are their respective heat capacities.

In the soil physics literature, heat capacity on volume basis is a preferred term over mass basis. This is often known as volumetric heat capacity (C_v) and is defined as the quantity of heat required to raise the temperature of 1 cm^3 of soil by 1°C , and thus, its unit is $\text{cal cm}^{-3}^{\circ}\text{C}^{-1}$. For a dry soil, it is given by

$$C_v = D_b C_m \quad (5)$$

where D_b is the bulk density of soil in g cm^{-3} .

On volumetric basis, the heat capacity of a soil is expressed by De Vries^[10] as follows:

$$C_v = Y_s C_s + Y_w C_w + Y_a C_a \quad (6)$$

where Y_s , Y_w , and Y_a are the volume fractions of solid material, water, and air, respectively, and C_s , C_w , and C_a are their respective heat capacities. The contribution of the third term on the right-hand side of Eq. 6 to C_v is negligible

Table 1 Specific heat of some soils, minerals, and other materials.

Material	Specific heat (cal g ⁻¹ °C ⁻¹)	References
Calcareous sandy soil	0.249	Lang ^[1]
Humus calcareous sandy soil	0.257	Lang ^[1]
Coarse quartz sand	0.190–0.198	Bowers and Hanks, ^[2] Kersten, ^[3] Lang, ^[1] Ulrich, ^[4] and White ^[5]
Fine quartz sand	0.192–0.197	Kersten, ^[3] Lang, ^[1] and Ulrich ^[4]
Darby loamy sand	0.210	Bowers and Hanks ^[2]
Kharagpur sandy loam	0.231	Ghildyal and Tripathi ^[6]
Sandy loam	0.260	Ghuman and Lal ^[7]
Sarpy fine sandy loam	0.221	Bowers and Hanks ^[2]
Cass loam	0.210	Bowers and Hanks ^[2]
Haldi loam	0.210	Tripathi and Ghildyal ^[8]
Loam	0.370	Ghuman and Lal ^[7]
Silt loam	0.164–0.194	Kersten ^[3]
Sandy clay loam	0.341	Ghuman and Lal ^[7]
Clay	0.322	Ghuman and Lal ^[7]
Solomon clay	0.270	Bowers and Hanks ^[2]
Garden soil	0.267	Lang ^[1]
Al ₂ O ₃	0.217	Lang ^[1]
Albite	0.195–0.210	Bowers and Hanks ^[2] and White ^[5]
Apatite	0.183–0.210	Bowers and Hanks ^[2] and Ulrich ^[4]
CaCO ₃	0.206–0.208	Lang ^[1] and Ulrich ^[4]
Corundum	0.193	Bowers and Hanks ^[2]
Dolomite	0.222–0.230	Bowers and Hanks ^[2] and Ulrich ^[4]
Fe ₂ O ₃	0.163–0.165	Lang ^[1] and Ulrich ^[4]
Fe(OH) ₃	0.226	Ulrich ^[4]
Feldspar	0.190–0.220	Bowers and Hanks ^[2] and Kersten ^[3]
Gypsum	0.195	Ulrich ^[4]
Hematite	0.161	Bowers and Hanks ^[2]
Hornblende	0.195	Ulrich ^[4]
Kaolin	0.233–0.244	Lang ^[1] and Ulrich ^[4]
Magnesia mica	0.206	Ulrich ^[4]
Microcline	0.187–0.210	Bowers and Hanks ^[2] and White ^[5]
MgCO ₃	0.246–0.260	Lang ^[1] and Ulrich ^[4]
Oligoclase	0.205	Ulrich ^[4]
Orthoclase	0.194	Ulrich ^[4]
Potash mica	0.208	Ulrich ^[4]
Quartz powder	0.189–0.209	Lang ^[1] and Ulrich ^[4]
Talc	0.209	Ulrich ^[4]
Humus	0.443–0.477	Lang ^[1] and Ulrich ^[4]
Ice	0.489	Hillel ^[9]
Organic matter	0.461	Hillel ^[9]
Water (liquid)	1.000	Hillel ^[9]

as compared to the other two terms and, therefore, can be neglected safely. The soil solid material is generally composed of soil minerals and organic matter. If the volume fractions of the soil minerals and organic matter are denoted

by Y_m and Y_o , respectively, and substitute the average value of heat capacity 0.46 for mineral constituents, 0.60 for organic matter, and 1.0 for water in Eq. 6, the heat capacity per unit volume of soil is given by

$$C_v = 0.46Y_m + 0.60Y_o + Y_w \quad (7)$$

In S.I. units, Eq. 7 can be written as follows:

$$C_v = 1.92Y_m + 2.51Y_o + 4.18Y_w \quad (8)$$

The unit of C_v becomes $\text{MJ m}^{-3} \text{K}^{-1}$.

By knowing the volume fractions of mineral matter, organic matter, and water content, one can obtain the heat capacity of soil from Eq. 7 or 8.

In the laboratory, heat capacity of soil can be determined by calorimeter method.^[11]

MANAGEMENT FACTORS AFFECTING HEAT CAPACITY OF SOIL

Heat capacity of soils is influenced by tillage, compaction, puddling, and mulching with crop residues. Higher volumetric heat capacity by $0.61 \text{ MJ m}^{-3} \text{K}^{-1}$ was reported in the top 15 cm layer of silt loam soil under no tillage than that under conventional tillage system ($2.05 \text{ MJ m}^{-3} \text{K}^{-1}$).^[12] Higher value of C_v in the no-till system relative to tilled was caused by increased aggregate stability, organic matter, and water storage capacity in the former system. Compaction improved the heat capacity of soils because of increased bulk density that resulted in more mass per cm^3 of soil.^[13] Mulching of soil with crop residue enhanced its heat capacity^[14] due to increased soil moisture content and organic matter. Puddling increased volumetric heat capacity of clay loam soil^[15] owing to increased moisture content though there was a decrease in bulk density.

CONCLUSION

In conclusion, heat capacity of soil, defined as the amount of heat existing in it, is affected by soil makeup and different soil and crop management practices. It is an important edaphic factor that affects seed germination, seedling establishment, plant growth, and yield. Value of heat capacity for a soil can be estimated from its mineral matter, organic matter, and water content. Heat capacity of a soil can also be measured in the laboratory.

REFERENCES

1. Lang, C. Uber warmecapacitat der bodenconstituenten. *Forsch. Gebiete Agr. Phys.* **1878**, *1*, 109–147.
2. Bowers, S.A.; Hanks, R.J. Specific heat capacity of soils and minerals as determined with a radiation calorimeter. *Soil Sci.* **1962**, *94* (6), 392–396.
3. Kersten, M.S. *Thermal Properties of Soils*; Technol. Bull. 28; University of Minnesota: Minneapolis, 1949.
4. Ulrich, R. Untersuchungen uber die warmecapacitat der bodenconstituenten. *Forsch. Gebiete Agr. Phys.* **1894**, *17* (1), 1–31.
5. White, W.P. Silicate specific heat. *Am. J. Sci.* **1919**, *47* (1), 1–43.
6. Ghildyal, B.P.; Tripathi, R.P. Effect of varying bulk densities on the thermal characteristics of lateritic sandy clay loam soil. *J. Indian Soc. Soil Sci.* **1971**, *19* (1), 5–10.
7. Ghuman, B.S.; Lal, R. Thermal conductivity, thermal diffusivity and thermal capacity of some Nigerian soils. *Soil Sci.* **1985**, *139* (1), 74–80.
8. Tripathi, R.P.; Ghildyal, B.P. Thermal properties of fine textured mollisols under varying moisture conditions. *J. Indian Soc. Soil Sci.* **1974**, *22* (2), 103–108.
9. Hillel, D. Soil temperature and heat flow. In *Fundamentals of Soil Physics*; Academic Press: New York, 1980; 294 pp.
10. De Vries, D.A. Thermal properties of soils. In *Physics of Plant Environment*; vanWijk, W.R., Ed.; North Holland Publisher: Amsterdam, 1963; 210–235.
11. Taylor, S.A.; Jackson, R.D. Heat capacity and specific heat. In *Methods of Soil Analysis*; Black, C.A., Ed.; Part 1; American Society of Agronomy: Madison, 1965; 345–348.
12. Arshad, M.A.; Azooz, R.H. Tillage effects on soil thermal properties in a semiarid cold region. *Soil Sci. Soc. Am. J.* **1996**, *60* (2), 561–567.
13. Yadav, M.R.; Saxena, G.S. Effect of compaction and moisture content on specific heat and thermal capacity of soils. *J. Indian Soc. Soil Sci.* **1973**, *21* (2), 129–132.
14. Van Doren, D.M.; Allamaras, R.R. Effect of residue practices on the soil physical environment, microclimate and plant growth. In *Crop Residue Management Systems*; Oschwald, W.R., Ed.; ASA Spec. Pub. 31; ASA, CSSA, and SSSA: Madison, 1978; 49–83.
15. Sharma, P.K.; De Datta, S.K. Thermal characterization of a tropical rice soil in relation to puddling, field water depth and percolation rate. *Soil Technol.* **1991**, *4*, 167–175.

Heat Flux

Thomas J. Sauer

National Soil Tilth Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.

Abstract

The range of climatic conditions existing on the earth's land surface is related to physical position (elevation and latitude) and large-scale meteorological forces (such as ocean and air currents). Soil plays an integral role in influencing the climate near the ground. Properties of the soil surface affect the amount of sunlight energy reflected back to the atmosphere and the amount used to evaporate water, grow plants, warm the air above the ground, or warm the soil. The amount of thermal energy that moves through an area of soil in a length of time is referred to as the soil heat flux or heat flux density. *Soil heat flux* is a measure of the amount of energy moving into or out of the soil, which determines soil temperature and the rate of daily and seasonal temperature change. Soil temperature is one of the key factors in determining the rate of chemical and biological processes in the soil.

SOIL HEAT FLUX AND SURFACE ENERGY BALANCE

Early research on soil thermal regimes focused on soil temperature and its effects on crop growth.^[1,2] Soils subjected to different tillage practices or with contrasting physical properties were observed to warm at different rates in the spring. However, the processes controlling heat flow in soils were not adequately understood. Patten^[3] provided the first quantitative, comprehensive treatment of heat-transfer processes in soil by measuring the thermal properties of several soils under controlled laboratory conditions. By the mid-20th century, meteorologists became interested in measuring soil heat flux as a component of the surface energy balance:

$$R_n - G = LE + H \quad (1)$$

where R_n is the net radiation, G is the soil heat flux, LE is the latent heat flux, and H is the sensible heat flux (all in watts per square meter, $W\ m^{-2}$). *Net radiation* represents the net difference between incoming shortwave and longwave (thermal) radiation and reflected shortwave and terrestrial longwave radiation. *Latent heat flux* is the amount of energy consumed by evaporating water or released during dew formation. *Sensible heat flux* refers to the energy involved in heating or cooling the air layer near the soil surface.

The magnitude of G can vary, from near zero on a daily basis in temperate regions with dense crop cover, to 50% of hourly R_n in semiarid regions with no vegetation. Energy balance investigations^[4,5] have found complex interactions between atmospheric transport processes and soil thermal properties because both affect the dynamics of energy flow

in surface soil layers. Fig. 1 presents an example of diurnal patterns of energy balance terms at a pasture site in northwestern Arkansas. These data are for two days in autumn (October) when the soil was moist and the grass canopy was 0.35 m tall. The sum of G for both days was negative (i.e., the soil was cooling) and equivalent to less than 1% of the magnitude of the daily R_n .

FACTORS AFFECTING SOIL HEAT FLUX

The partitioning of R_n at the soil surface and the soil thermal properties influences diurnal and annual patterns of soil heat flux and determines the ability of the soil to transfer energy to and from the surface. The amount of reflected shortwave radiation, which depends on soil color, soil wetness, surface cover, and surface roughness, is one component of the partitioning of R_n at the surface. In general, plants or plant residues reflect more shortwave radiation than do bare soils. Smooth surfaces with lighter colored or dry soils also tend to reflect more shortwave radiation than do uneven surfaces with dark-colored or wet soils. Surface roughness and soil wetness also affect LE and H in that roughness enhances turbulent mixing of the air near the surface and the availability of soil water affects evaporation.

Soil heat capacity and thermal conductivity are the key thermal properties affecting soil heat flux.^[6] The heat capacity of a soil is the amount of energy (MJ) required to change the temperature of a volume of soil (m^3) $1^\circ K$. A soil's thermal conductivity is the constant of proportionality relating the temperature gradient across the soil to the soil heat flux, as defined by Fourier's equation:

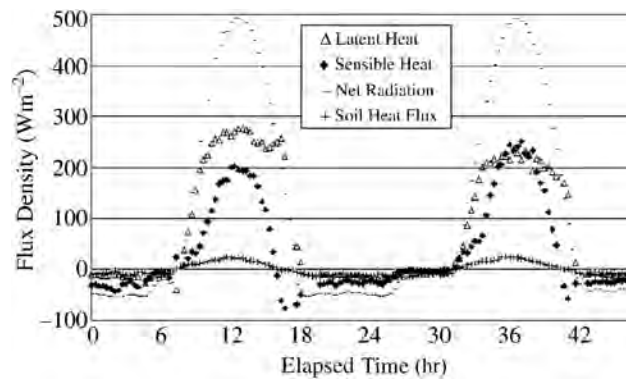


Fig. 1 Energy balance components for a pasture site in north-western Arkansas for two days in October 1997.

$$G = -k \, dT/dz \quad (2)$$

where k is the thermal conductivity ($\text{W m}^{-1} \text{K}^{-1}$), T is the soil temperature (K), and z is the distance between the soil temperature measurements (m). Table 1 lists typical heat capacity and thermal conductivity values for different soils and soil constituents. Because of the much greater (approximately 3500 times) heat capacity of water as compared with air (4.18 vs. $0.0012 \text{ MJ m}^{-3} \text{K}^{-1}$), a moist soil has a much higher heat capacity than a dry one. However, the thermal conductivity of water is approximately 23 times greater than air (0.57 vs. $0.025 \text{ W m}^{-1} \text{K}^{-1}$). Thus, a moist soil conducts heat more efficiently than a dry one, but it takes much more energy to develop the same temperature gradient in a moist soil as compared with a dry one.

In field settings, G can be limited by the ability of the soil to conduct heat or by the amount of available energy at the surface. Soil heat flux measured over a 24-hour period at a depth of 0.05 m in a no-till corn field in central Iowa in fall, winter, and spring is presented in Fig. 2. In fall, the fresh corn residue effectively insulated the dry

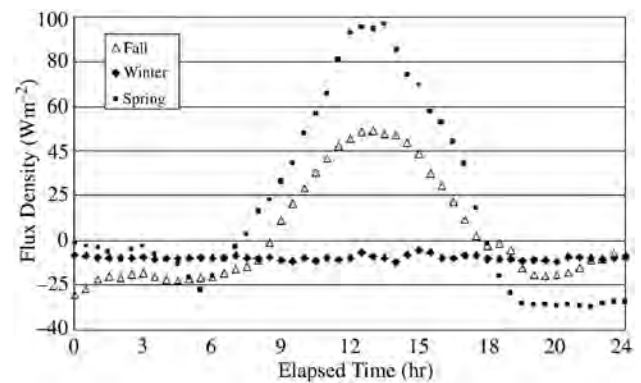


Fig. 2 Seasonal variation in soil heat flux at 0.05 m in a no-till corn field in central Iowa.

soil from solar radiation inputs. In winter, a layer of snow 0.23 m thick covered the wet residue and soil. Because of the insulating effect of the snow layer, the soil heat flux remained negative (i.e., the soil was slowly losing heat to the atmosphere) and showed little diurnal trend. By spring, the increase in decayed residue exposed more of the wet soil to incoming radiation, causing the peak soil heat flux to nearly double that of the peak measured during the fall.

MEASUREMENT TECHNIQUES

Because soil heat flux is such an important parameter in agrometeorology and soil science research, several measurement methods have been developed.^[9,10] The choice of a specific method depends on the intended use of the data, the resources available, and the degree of accuracy required. One category of techniques uses measured and/or estimated soil thermal properties combined with soil heat flow theory to estimate G . These techniques, which include calorimetric, gradient, and combination, generally require numerous and accurate measurements of soil temperature, heat capacity, and thermal conductivity at various depths in the soil. Depending on the accuracy desired, some or all of these parameters must be measured at several locations, at frequent intervals, and for each depth throughout the measurement period. Such requirements make their use relatively tedious and time consuming. The second category uses a sensor, called a heat flux plate or heat flow transducer, designed specifically to measure soil heat flux. Soil heat flux plates are small (often <50 mm per side), thin (<5 mm thick), rigid sensors placed horizontally in the soil. Most employ a thermopile located inside a waterproof housing to measure the temperature difference between the top and bottom of the plate. Temperature difference across the plate is used to estimate heat flux through the soil. Although this technique is much simpler to use, the plate itself can disturb soil heat and water flow, thereby reducing accuracy in some situations. Corrections

Table 1 Observed thermal properties of various moist soils and soil components at ambient temperatures.

Material	Heat capacity ($\text{MJ m}^{-3} \text{K}^{-1}$)	Thermal conductivity ($\text{W m}^{-1} \text{K}^{-1}$)
Sandy soil	2.09	1.8
Silt loam soil	1.02	1.2
Peat soil	3.14	0.29
Quartz	2.13	8.8
Other soil minerals	2.39	2.9
Organic matter	2.50	0.25
Water	4.18	0.57
Air	0.0012	0.025

Note: K, Kelvin; m, meters; MJ, megajoules; W, watts.

Source: From de Vries,^[6] van Wijk & de Vries,^[7] and Al Nakshabandi & Kohnke.^[8]

must be made for heat storage in the soil above the plate in order to calculate G at the soil surface accurately.

CONCLUSION

The rates of chemical and biological processes in soil are strongly affected by temperature. Soil heat flux, the amount of heat moving into or out of the soil, can be altered to optimize the soil thermal environment. Irrigation, drainage, tillage, and crop residue management can all be used to affect energy transfer at the soil surface by modifying the surface characteristics and soil thermal properties. Knowledge of the factors affecting soil heat flux is essential for optimizing plant growth through modification of the soil thermal regime and microclimate near the ground.

REFERENCES

1. King, F.H. *The Soil*; Macmillan: New York, 1906.
2. Keen, B.A. *Physical Properties of the Soil*; Longmans, Green and Co.: London, 1931.
3. Patten, H.E. *Heat Transference in Soils*; Bureau of Soils Bulletin No. 59; U.S. Department of Agriculture: Washington, 1909.
4. Staley, R.C.; Gerhardt, J.R. Soil heat flux measurements. In *Exploring the Atmosphere's First Mile*; Lettau, H., Davidson, B., Eds.; Pergamon Press: New York, 1957; Vol. 1, 58–80.
5. Sellers, W.D. *Physical Climatology*; The University of Chicago Press: Chicago, 1965.
6. de Vries, D.A. Thermal properties of soils. In *Physics of Plant Environment*; van Wijk, W.R., Ed.; 2nd Ed.; North-Holland Publishing Co.: Amsterdam, 1966; 210–235.
7. van Wijk, W.R.; de Vries, D.A. Periodic temperature variations in a homogeneous soil. In *Physics of Plant Environment*; van Wijk, W.R., Ed.; 2nd Ed.; North-Holland Publishing Co.: Amsterdam, 1966; 102–140.
8. Al Nakshabandi, G.; Kohnke, H. Thermal conductivity and diffusivity of soils as related to moisture tension and other physical properties. *Agr. Meteorol.* **1965**, 2, 271–279.
9. Fuchs, M. Heat flux. In *Methods of Soil Analysis: I. Physical and Mineralogical Methods*; Klute, A., Ed.; 2nd Ed.; Soil Science Society of America: Madison, 1986; 957–968.
10. Kimball, B.A.; Jackson, R.D. Soil heat flux. In *Modification of the Aerial Environment of Plants*; Barfield, B.J., Gerber, J.F., Eds.; American Society of Agricultural Engineers: St. Joseph, 1979; 211–229.

Heavy Metals

Mike J. McLaughlin

Land and Water, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Glen Osmond, South Australia, Australia

Abstract

Some heavy metals are essential for either plant or animal survival on land, while others are non-essential and toxic at low concentrations. This entry reviews heavy metals, their abundance, and their biological effects.

INTRODUCTION

It is generally accepted that metals having a specific gravity (weight per unit volume) greater than 5 Mg m^{-3} are termed heavy metals. In soils, these elements include cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), mercury (Hg), manganese (Mn), molybdenum (Mo), nickel (Ni), lead (Pb), and zinc (Zn).^[1] The term “heavy metals” is often used synonymously with the term “trace elements,” but this is incorrect as trace elements are generally defined as those elements normally occurring in soil at concentrations less than 100 mg kg^{-1} ,^[2] which precludes several heavy metals, e.g., Cr, Fe, and Mn. Arsenic (As) is often included in the group “heavy metals” but is more correctly classified as a metalloid.

GENERAL CHEMISTRY

Some heavy metals are essential for either plant or animal survival on land (Co, Cr, Cu, Fe, Mn, Mo, Ni, and Zn), while others are non-essential and toxic at low concentrations (Cd, Hg, and Pb). All the heavy metals, except Pb, are transition elements, belonging to the d-block in the periodic table. Many of these elements differ from the alkaline earth metals (e.g., Na, Ca, and Mg) in that they can exist in several valence states in soil (Table 1). In particular, Cr, Fe, and Mn are markedly affected by soil redox potential and undergo both oxidation and reduction depending on soil conditions. This has important implications for the availability and toxicity of many heavy metals. As both Fe and Mn are major structural metals in soil minerals, reduction of the Fe^{3+} and Mn^{4+} ions may lead to a change in soil mineral surfaces important for retention of many elements, including other metals. For Cr, oxidation converts the non-toxic Cr^{3+} ion to the toxic and carcinogenic Cr^{6+} ion. This reaction has even more significance in soils as a strongly sorbed or precipitated cation (Cr^{3+}) is converted into a poorly sorbed or soluble anion (CrO_4^{2-}).

However, even in aerobic soils, Cr^{3+} is the thermodynamically stable state, so added Cr^{6+} ion is rapidly converted to Cr^{3+} in most soils.

Solubility and availability/toxicity to organisms of heavy metal cations (Cd^{2+} , Cr^{3+} , Fe^{n+} , Pb^{2+} , Mn^{n+} , Hg^{2+} , Ni^{2+} , and Zn^{2+}) decrease as soil pH increases. This is due to the increase in negative charge on variable charge surfaces in soil^[4] and the propensity for these metals to precipitate as sparingly soluble compounds (phosphates, carbonates, and hydroxides), as soil pH increases.^[5] On the other hand, solubility and availability/toxicity to organisms of anionic heavy metals (CrO_4^{2-} and MoO_4^{2-}) may increase as soil pH increases, again due to increases in surface negative charge on soil particles affecting sorption. Under reducing conditions in soil, many of the metals form insoluble metal sulfides, therefore reducing availability to plants and animals.

Heavy metals are also subjected to complexation reactions in soil with both inorganic and organic ligands, which may markedly increase mobility in soil. This may be used to good effect by plants in scavenging essential metals from soil, e.g., phytochelatins (see below). However, for non-essential metals, complexation and increased mobility also may lead to increased environmental risks through leaching and plant uptake, e.g., chloride-induced uptake of Cd by food crops in saline soils.^[6]

ABUNDANCE IN ROCKS AND SOILS

Heavy metals become incorporated into primary minerals in igneous rocks through isomorphous substitution. In sedimentary rocks, heavy metals are incorporated as constituents of minerals, or are removed from the water column and trapped in sediments by adsorption or precipitation processes.^[7] Apart from Cr, Fe, and Mn, heavy metals are generally present at trace concentrations ($<100 \text{ mg kg}^{-1}$) in most soils, with the exception of soils developed over mineralized parent materials or soils developed through biological enrichment.^[8] Background concentrations in soil

Table 1 Physical and chemical properties of the heavy metals.

Element	Symbol	Atomic no.	Molecular weight	Valence	Natural isotopes	Density (Mg m ⁻³)	Melting point (°C)	First ionization potential (eV)	Dominant species in soil	Dominant species in soil solution	
										pH 3.5–6.0	pH 6.0–8.5
Cadmium	Cd	48	112.41	2	8	8.65	321	8.96	Cd ²⁺	Cd ²⁺ , CdCl ⁺ , CdSO ₄ ⁰	Cd ²⁺ , CdCl ⁺ , CdSO ₄ ⁰
Chromium	Cr	24	52.01	2, 3, 6	4	7.19	1875	6.76	Cr ³⁺ , CrO ₄ ²⁻	Cr ³⁺ , CrOH ²⁺	Cr(OH) ₄ ⁻
Cobalt	Co	27	58.94	2, 3	1	8.90	1493	7.86	Co ²⁺	—	—
Copper	Cu	29	63.54	1, 2	2	8.94	1083	7.73	Cu ²⁺	Cu ²⁺ , Cu-org.	Cu-hydroxy species, CuCO ₃ ⁰ , Cu-org.
Iron	Fe	26	55.85	2, 3	4	7.87	1536	7.87	Fe ²⁺ , Fe ³⁺	Fe-hydroxyspecies, Fe-org.	Fe-hydroxy species, Fe-org.
Lead	Pb	82	207.19	2, 4	4	11.35	327	7.42	Pb ²⁺	Pb ²⁺ , PbSO ₄ ⁰ , Pb-org.	Pb-hydroxy and carbonate species, Pb-org.
Manganese	Mn	25	54.94	2, 3, 4, 7	1	7.44	1244	7.44	Mn ²⁺ , Mn ⁴⁺	Mn ²⁺ , MnSO ₄ ⁰ , Mn-org.	Mn ²⁺ , MnSO ₄ ⁰ , MnCO ₃ ⁰
Mercury	Hg	80	200.61	1, 2	7	13.54	−39	10.44	Hg ²⁺ , (CH ₃) ₂ Hg	—	—
Molybdenum	Mo	42	95.94	6	7	10.22	2610	7.10	MoO ₄ ²⁻	—	—
Nickel	Ni	28	58.71	2, 3	5	8.91	1453	7.64	Ni ²⁺	Ni ²⁺ , NiSO ₄ ⁰ , Ni-org.	Ni ²⁺ , NiHCO ₃ ⁺ , NiCO ₃
Zinc	Zn	30	65.37	2	5	7.14	420	9.39	Zn ²⁺	Zn ²⁺ , ZnSO ₄ ⁰ , Zn-org.	Zn ²⁺ , Zn-hydroxy and carbonate species, Zn-org.

Source: From Logan^[2] and Ritchie & Sposito.^[3]

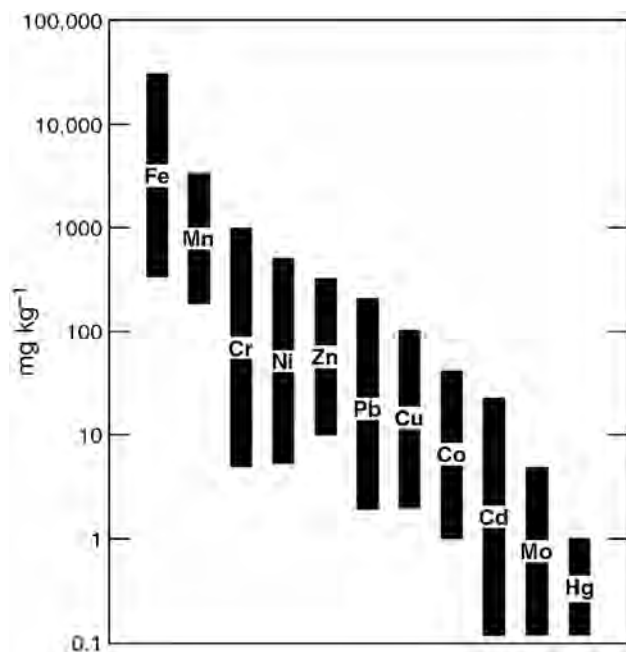


Fig. 1 Range of heavy metal concentrations of soil. Bars represent commonly found values.

Source: Adapted from Logan,^[2] Alloway,^[7] Barry & Rayment,^[8] Lantzy & Mackenzie,^[9] Kabata-Pendias & Pendias,^[10] Swaine,^[11] and Ferguson.^[12]

usually reflect the composition of the parent rock material. Background concentrations of heavy metals are usually determined where no known anthropogenic inputs have occurred, but this assessment is often problematic due to the global spread of anthropogenic emissions through atmospheric transport processes.^[9] The typical range in abundances of heavy metals in unpolluted soils is shown in Fig. 1.

Concentrations of heavy metals in soil can be significantly increased through human activity. A number of primary and secondary sources have been identified as contributing to enhanced concentrations of heavy metals in soil^[12] (Table 2). Many countries have introduced legislation to minimize the amount of metals accumulating in soil and set ceiling concentrations above which further metal additions must stop.

BIOLOGICAL EFFECTS

Co, Cr, Cu, Fe, Mn, Mo, Ni, and Zn are all essential for either healthy plant or animal functioning in soil and all these metals, except Ni, are used in fertilizers or animal stock supplements to ensure efficient agricultural production on soils deficient in essential metals. Due to the low solubility of essential heavy metals in neutral and alkaline soils, many plants have developed strategies to mobilize solid phase forms to facilitate uptake by roots (e.g., Fe, Mn, and Zn). Non-proteinogenic amino acids,

Table 2 Sources of heavy metal contamination in soils.

Source	Main heavy metals
Primary sources	
Fertilizers	Cd, Cu, Mo, Pb, Zn
Irrigation water	Cd, Fe
Manures and composts	Cd, Cr, Cu, Fe, Hg, Mn, Mo, Ni, Pb, Zn
Pesticides	Cu, Hg, Pb, Zn
Sewage biosolids (sludges)	Cd, Cr, Cu, Fe, Hg, Mn, Mo, Ni, Pb, Zn
Soil amendments (lime, gypsum, etc.)	Cu, Mn, Pb, Zn
Secondary sources	
Automobile Aerosols	Pb
Coal combustion	Pb
Mine waste and effluents	Cd, Cu, Fe, Hg, Mn, Ni, Pb, Zn
Nonferrous smelter waste	Cd, Cu, Hg, Mn, Ni, Pb, Zn
Paint dispersal	Cd, Pb
Tire wear	Cd, Zn
Waste combustion	Cd, Pb

Source: Adapted from Barry & Rayment.^[8]

or phytochelatins, secreted by actively growing roots are important in acquisition of Cu, Fe, Zn, and possibly Mn from deficient soils.^[13] At high concentrations in soil, all the essential elements may pose risks to microorganisms, plants, animals, or humans (Fig. 2). Non-essential heavy metals in soil (Cd, Hg, and Pb) have no beneficial effects at low concentrations (Fig. 2) and may be toxic at even trace concentrations (e.g., Cd and Hg).

Critical exposure pathways for expression of heavy metal toxicity in soil have been examined as part of regulation governing reuse of biosolids (sewage sludge) on soil^[14] but act as a general risk analysis template for all types of pollution of soil by heavy metals.^[15] For some elements, e.g., Cd and Co, food chain transfer is the main risk pathway, as these elements are easily accumulated by plants in edible tissues. For other elements, sorption in soil

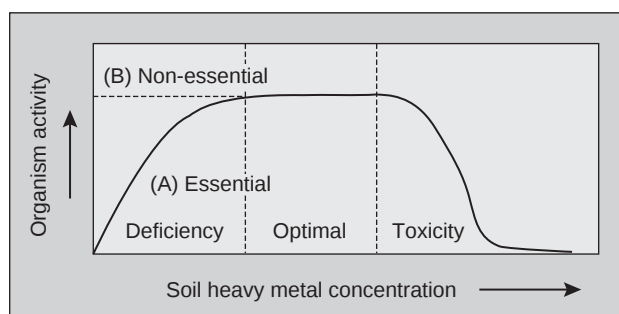


Fig. 2 Typical concentration–response relationship for (A) essential and (B) non-essential heavy metals in soil.

Table 3 Critical risk pathway assessments for heavy metal pollution of soils.

Metal	Dominant risk pathway	Secondary risk pathway	Most important predictor required
Cd	Food chain transfer	Phyto- and ecotoxicity	Soil–plant uptake
Co	Food chain transfer	Phyto- and ecotoxicity	Soil–plant uptake
Cr	Phyto- and ecotoxicity	Metal leaching	Toxic threshold definition, speciation
Cu	Phyto- and ecotoxicity	Soil ingestion by animals/humans	Toxic threshold definition
Fe	Phyto- and ecotoxicity ^a	None	Toxic threshold definition
Hg	Soil ingestion by animals/humans	Metal leaching	Toxic threshold definition, speciation
Mn	Phyto- and ecotoxicity	Soil ingestion by animals/humans	Toxic threshold definition
Ni	Phyto- and ecotoxicity	Soil ingestion by animals/humans	Toxic threshold definition
Pb	Soil ingestion by animals/humans	Phyto- and ecotoxicity	Oral bioavailability assessment
Zn	Phyto- and ecotoxicity	Food chain transfer	Soil–plant uptake

^aOnly in acid soils under reducing condition.

Source: Adapted from McLaughlin & Zarcinas.^[15]

is strong, bioaccumulation by microorganisms low, and plant uptake and translocation so low that the dominant risk pathway is (for higher animals and humans) through direct ingestion of soil, e.g., Hg and Pb. For the other heavy metals, behavior in soil and bioaccumulation characteristics result in toxicity to plants and microorganisms (phyto- and ecotoxicity) being the dominant risk pathway at high concentrations, e.g., Cr, Cu, Mn, Ni, and Zn. Thus, prediction of risks from heavy metal pollution of soils through soil testing requires a different emphasis depending on the metal considered (Table 3).

REFERENCES

1. Soil Science Society of America. *Internet Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 2001. Available at <http://www.soils.org/sssagloss/> (accessed February 2001).
2. Logan, T.J. Soils and environmental quality. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; G155–G163.
3. Ritchie, G.S.P.; Sposito, G. Speciation in soils. In *Chemical Speciation in the Environment*; Ure, A.M.E., Davidson, C.M., Eds.; Blackie Academic and Professional: London, 1995; 201–233.
4. Barrow, N.J. *Reactions with Variable Charge Soils*; Martinus Nijhoff: Dordrecht, 1997; 191 pp.
5. Lindsay, W.L. *Chemical Equilibria in Soils*; Wiley: New York, 1979; 449 pp.
6. McLaughlin, M.J.; Tiller, K.G.; Smart, M.K. Speciation of cadmium in soil solution of saline/sodic soils and relationship with cadmium concentrations in potato tubers. *Aust. J. Soil Res.* **1997**, *35*, 1–17.
7. Alloway, B.J. The origins of heavy metals in soils. In *Heavy Metals in Soils*; Alloway, B.J., Ed.; 2nd Ed.; Blackie Academic and Professional: London, 1995; 38–57.
8. Barry, G.; Rayment, G.E. Heavy metals and nutrients in soils and sediments of Raine island, great barrier reef. *Land Contam. Reclam.* **1997**, *5* (4), 281–285.
9. Lantzy, R.J.; Mackenzie, F.T. Atmospheric trace metals: Global cycles and assessment of man's impact. *Geochim. Cosmochim. Acta.* **1979**, *43*, 511–525.
10. Kabata-Pendias, A.; Pendias, H. *Trace Elements in Soils and Plants*, 2nd Ed.; CRC Press: Boca Raton, 1992; 365 pp.
11. Swaine, D.J. *The Trace Element Content of Soils*. Commonwealth Bureau of Soil Science Technical Communication No. 48; Commonwealth Agricultural Bureaux: Farnham Royal, 1955; 157 pp.
12. Ferguson, J.E. *The Heavy Elements: Chemistry, Environmental Impact and Health Effects*; Pergamon Press: Oxford, 1990; 614 pp.
13. Römheld, V. Different strategies for iron acquisition in plants. *Physiol. Plant.* **1987**, *70*, 231–234.
14. USEPA. *Part 503—Standards for the Use and Disposal of Sewage Sludge*; Federal Register 58; U.S. Government Publishing Office: Washington, 1993; 9387–9404.
15. McLaughlin, M.J.; Zarcinas, B.A.; Stevens, D.P.; Cook, N. Soil testing for heavy metals. *Commun. Soil Sci. Plant Anal.* **2000**, *31* (11–14), 1661–1700.

History of Soil Science

Eric C. Brevik

Department of Natural Sciences, Dickinson State University, Dickinson, North Dakota, U.S.A.

Artemi Cerdà

Department of Geography, University of Valencia, València, Spain

Abstract

Human knowledge of soil has come a long way since agriculture began about 9,000 B.C.E., when finding the best soils to grow crops in was largely based on a trial and error approach. Many innovations to manage and conserve soil, such as the plow, irrigation techniques, terraces, contour tillage, and even the engineering of artificial soils, were developed between 9,000 B.C.E. and 1,500 C.E. Scientific methods began to be employed in the study of soils during the Renaissance and many famous scientists addressed soil issues but soil science did not evolve into an independent scientific field of study until the 1880s. In the early days of the study of soil as a science, soil survey activities provided one of the major means of advancing the field. As the 20th century progressed, advances in soil biology, chemistry, genesis, management, and physics allowed the use of soil information to expand beyond agriculture to environmental issues, human health, land use planning, and many other areas. The development of soil history as a subfield of the discipline in the latter part of the 20th century has promise to help advance soil science through a better understanding of how we have arrived at the major theories that shape the modern study of soil science.

INTRODUCTION

It is important for soil scientists to study the history of our field. A true understanding of the hypotheses and theories that shape the way we approach the modern study of soils can only be appreciated if we also understand where they came from and what other possibilities have been explored. This facilitates understanding the strengths and weaknesses of our knowledge base and in developing a core understanding of our discipline. Studying history can also assist in directing research away from routes that are likely to be unfruitful as well as point us in directions that have not been well explored previously and may have promise for additional research. Understanding soil science history from an international perspective can help improve communication between countries, avoid wasted effort through the exploration of ideas already established elsewhere, and perhaps more importantly, can assist in forming a common language among soil scientists. Knowing our history can make us more efficient and effective as scientists.

Soil science history has become a fairly well-established subfield in modern soil science. Professional societies including the European Geosciences Union (EGU), International Union of Soil Sciences (IUSS), and Soil Science Society of America (SSSA) have established subdivisions or committees within their structures that promote the study of soil science history. In addition, many leading soil science journals will accept soil science history submissions,

and several internationally known soil scientists and historians have contributed to our understanding of the field's history.

ANCIENT SOIL SCIENCE (9000 B.C.E. TO 1500 C.E.)

Through much of history soil knowledge has been linked to agriculture. The earliest evidence of agricultural activity, including manipulation of soil to promote crop growth, comes from a site near Jarmo, Iraq.^[1] However, agriculture also developed in areas beyond the Middle East. Africa, China, and Peru and Mexico in the Americas saw the development of societies that cropped their own cereals, tubers, or vegetables. Animals were domesticated, which changed soils due to grazing and the manuring of exhausted soils, reducing the use of shifting agricultural practices and encouraging sedentary societies and increasingly intensive use of soil resources.^[2] The development of agriculture independently in different parts of the world explains the high biodiversity of agricultural crops, and human reliance on agriculture transformed the soils of the world in a few millennia.^[2] Except for Antarctica, agriculture, including grazing, is widespread over all the continents of the world. A number of management advances were made over the next several thousand years, including the development of early plows, terracing, and contour tillage,^[1] all designed to improve soil conditions for the purpose of crop production.



Fig. 1 Intercropped maize and squash with ground cover in a traditional Wampanoag garden at Plimoth Plantation, Massachusetts. Note in particular the small mounds at the base of each maize plant, showing the planting of the maize in individual hills.
Source: Photograph by Eric Brevik.

During this time, humans also began to recognize spatial patterns in soil and settlement patterns in many regions began to correlate to soil types.^[3] By 2000 B.C.E., the Chinese had developed a soil classification system, as did the Greeks by about 300 B.C.E.^[1] Advances were also made in the Americas. The Aztec, Inca, and Maya in Central and South America constructed artificial soils to improve crop production and developed bench terraces; these civilizations were among the most successful in human history at minimizing soil erosion and creating sustainable agricultural systems.^[1] In a sense they originated the concept of soil conservation. Native Americans in North America were cultivating crops by 5000 B.C.E. and maintained soil fertility using intercropping of diverse crop mixes, adding ash from burned weeds and trees, and incorporating fallow periods.^[4] Crops were typically planted in small mounds (Fig. 1), which were more resistant to erosion than the rows that were traditional in Europe.^[4]

SCIENCE IS APPLIED TO THE STUDY OF SOIL (1500 C.E. TO 1880 C.E.)

The Renaissance began in the western world about 1500 C.E., and scientific practice in western societies was revolutionized. Scientific methodology began to be applied to the investigation of a number of phenomena, and scientists who are well known for their contributions to other fields applied scientific principles to the study of soil. For example, Charles Darwin, famous for the theory of evolution, worked on the concept of soil profiles and through his study of the influence of earthworms on soil formation became a leading figure in establishing soil biology^[1] while Leonardo da Vinci, Francis Bacon, and Robert Boyle, widely regarded as a founder of modern chemistry,

conducted early studies in nutrient cycling in and plant nutrition from soil.^[5] By the late 1820s, C. Sprengel proposed a theory on mineral nutrition of plants and the law of the minimum that would become the basis of J. von Liebig's "Mineral Theory" of plant nutrition, which is well known to soil scientists.^[6]

During this period, governments in Europe became interested in land valuation as a basis for taxation, which led to creation of the first soil maps in the early 1700s. The Chinese had also used soil attributes as a basis for taxation as early as the 1100s but did not create soil maps to assist in this.^[5] As more accurate and detailed base maps became widely available in many parts of Europe in the 1800s, thematic maps such as soil maps could be made much more efficiently and accurately.^[7] This led to a significant expansion of soil mapping efforts in Europe and the United States.^[1,5]

Despite the application of scientific principles to soil science issues during this time, the study of soil science as a scientific field had not developed. From 1500 C.E. to 1880 C.E., soils were being investigated as components of chemical, geological, or biological systems but were not viewed as an independent field of study in and of themselves. Events that occurred in the 1880s would change that view.

SOIL SCIENCE DEVELOPS AS AN INDEPENDENT SCIENTIFIC FIELD (1880 C.E. TO 2000 C.E.)

The early 1880s were marked by the 1883 publication of "Russian Chernozem" by Vasilii V. Dokuchaev, an event that is widely recognized as the birth of modern soil science.^[1,5,8] In this work and others, Dokuchaev: 1) recognized soil as an independent natural body worthy of study in its own right; 2) established the five soil forming factors that are widely used and developed one of the first models of soil genesis; and 3) broadly introduced the concept of A, B, and C horizons as they are used today.^[1,8,9] Although Dokuchaev certainly did not do all of this on his own, and aspects of his ideas had been discussed previously by other scientists, Dokuchaev was able to pull all the ideas together and present them in a logical and convincing fashion to a scientific world that was largely ready to accept them,^[5] even if it would take decades before some parts of the world would do so.^[1]

The end of the 1800s and beginning of the 1900s also saw the beginning of detailed national soil survey programs in many countries (Table 1).^[8] These surveys became significant drivers in the development of soil science knowledge, as soil surveyors sought to understand the genesis and function of the soils they discovered as well as develop classification systems to organize this new knowledge. Initially, these surveys tended to be focused on soil knowledge as needed for agriculture, but over time the role of these surveys expanded to include information

Table 1 The beginning date for detailed nationally organized soil survey for select countries.

Country	Date	Country	Date
United States	1899	China	1931
Russia	1908	Poland	1935
Canada	1914	The Netherlands	1945
Australia	1920s	Ghana	1946
Great Britain	1920s	Belgium	1947
Mexico	1926	Malaysia	1955
Sri Lanka	1930		

Source: Information from Brevik & Hartemink^[1] ©2010 and Brevik, Calzolari et al.^[8] ©2016.

relevant to geomorphology studies, land use planning, paleoenvironmental reconstructions, archaeology, human health, soil and water conservation, and other environmental issues.^[8] The use of soil information to address environmental issues was probably the fastest growing non-agricultural area of soil science in the second half of the 20th century.^[11] Early soil surveyors used paper maps and then aerial photographs as base maps, but by the end of the 20th century, remote and proximal sensed imagery managed with geographic information systems (GIS) was available that significantly advanced the information available to map soils and investigate spatial variability. These new data sources and tools also challenged traditional concepts of map scale.^[7]

By the early 1900s, there was considerable interest in the links between soils and human health. In Europe, Robert McCarrison, the County Palatine of Chester Local Medical and Panel Committee, Lady Eve Balfour, and André Voisin were all early voices promoting soil and human health links, while in the U.S. individuals such as Charles E. Kellogg, Selman Waksman (who was awarded the 1952 Nobel Prize in Physiology or Medicine for the isolation of antibiotics from soil microorganisms), William Albrecht, and J. I. Rodale investigated the issue.^[10] Sir Albert Howard, well-known worldwide for his work promoting organic agriculture, was a major proponent of the idea that soils were intimately linked to human health. In the latter part of the 20th century, a wide array of soil and human health topics were studied, including exposure to heavy metals, organic chemicals, and pathogens, the filtration capacity of soils, nutrient supply through the soil–plant system, and the supply of medications (about 40% of all prescription drugs have a soil origin). By the end of the 20th century, much had been learned about links between soils and human health, but there was a need for well-designed scientific studies.^[10]

Selected individuals, such as M.E. Wöllny in Germany^[1] and William John McGee and Edward Elway Free in the United States,^[4] studied soil erosion in the late 1800s and early 1900s, but few people viewed soil erosion as a serious problem until the Dust Bowl hit the United States in the

1930s.^[11] A combination of the Dust Bowl and persistent preaching of the erosion problem by Hugh H. Bennett led to the formation of the Soil Erosion Service (SES) by President F.D. Roosevelt in 1933, which was transformed into the Soil Conservation Service (SCS; now the Natural Resources Conservation Service) by an act of Congress in 1935.^[1] Both the SES and the SCS were under Bennett's direction. The SES/SCS undertook a number of soil erosion projects to test and demonstrate soil conservation and erosion control measures; one of the major results of these efforts was the development of conservation tillage.^[4] These projects were also interdisciplinary and transdisciplinary, including agronomists, anthropologists, biologists, engineers, economists, and soil scientists.^[4] The idea that interdisciplinary and transdisciplinary studies are critical to the advancement of soil science is quite popular,^[8,11] but the SES/SCS projects show that this is not a new idea. Despite the fact that soil erosion studies received a lot of attention in the United States beginning in the 1930s, by the end of the 20th century the United States was the only country in the world that had long-term soil erosion data collected using standardized methods for large expanses of the country. Other countries either did not have standardized data collection in place or waited until much later in the 20th century to begin national programs of erosion data collection and assessment.^[12] While erosion data were available for much of the rest of the world, coverage was more sporadic than in the United States and methods were not as standardized.^[12]

In the latter half of the 20th century, soil science history began to develop as a subfield of the discipline. Most who work in this subfield have their primary specialization in one of the scientific subfields of soil science or in a subfield of history but devote some of their time to soil science history because they feel it is important. Many well-respected soil scientists, including but certainly not limited to Ronald Amundson, Winfried Blum, James Bockheim, Johan Bouma, Marlin Cline, Christian Feller, Walter H. Gardner, Alfred Hartemink, Daniel Hillel, Hans Jenny, Rattan Lal, Alex McBratney, Daniel deB. Richter, Roy Simonson, Johan Six, Donald Sparks, and Dan Yaalon have published on various aspects of soil science history, and there has been a concerted effort by the soil scientists in the soil history community to reach out to like-minded historians for collaborations. Historians who have engaged in these collaborations include Lloyd Ackert, Benjamin Cohen, John McNeill, Martin Melosi, Laura Sayre, Richard Unger, and Verena Winiwarter.^[13] The study of soil science history received a major boost in 1982 when IUSS Commission 4.5—History, philosophy, and sociology of soil science was created, largely due to the efforts of Dan Yaalon.^[13] SSSA formed the Council on History, Philosophy, and Sociology of Soil Science in 1990 to coordinate with IUSS activities in this area with leadership from John Tandarich, Walter Gardner, Chris Johannsen, and Roy Simonson.^[13] The most recent professional scientific society to add soil science history to regular society activities was EGU. This

began in 2011, when Artemi Cerdà, the president of the Soil Systems Sciences (SSS) Division of EGU at the time, contacted Eric Brevik and Alfred Hartemink with a request to organize a soil science history session for the 2012 EGU General Assembly (GA). At the 2012 EGU-GA, the History, Education, and Society of Soil Science subdivision was created during a reorganization of the SSS.^[13]

CURRENT STATUS (2000 C.E. TO 2015 C.E.)

Through history soil science has made great strides as a scientific field, particularly during the latter parts of the 19th century and the 20th century. Information learned and soil maps generated have proven highly valuable in addressing many modern issues from agriculture to the environment, human health, and beyond. However, there were plenty of challenges for soil scientists to address as we transitioned from the 20th century to the 21st century. Soil survey coverage is very good in some countries but is largely lacking in others,^[12] and even countries that have nearly complete soil survey coverage lack the quantitative, georeferenced soil information that is needed as input into modern high tech computer models.^[8] There are many different soil classification systems, which make it difficult to communicate soil information internationally, so a common language would help.^[8] The new high-powered computers, global positioning systems, GIS, proximal and remote sensing information, and advanced spatial statistical and numerical analysis techniques available to soil scientists make a wealth of information available, but better models are needed to tie all the available information together and take full advantage of it.^[8,12] There is also a need for interdisciplinary and transdisciplinary collaborations that will bring new ability and insights into the work we do, enhancing the value of the final products produced for all end users.^[8,11] These challenges should not be viewed as problems or weaknesses but rather as challenges that continue to make the field of soil science vibrant, exciting, and needed in the modern world.

CONCLUSION

Studying the history of our field provides insights into how we arrived at modern understandings and the strengths and weaknesses inherent in that understanding. It also has the ability to point out potentially promising new areas of study by identifying areas that have not received adequate attention, and it has the ability to identify past directions of study that were not fruitful. This entry is only a very brief summary of major points in soil science history, and there is much more to be explored. Publications presented in the references and bibliography can provide a large amount of additional information that this entry did not have the space to address.

REFERENCES

1. Brevik, E.C.; Hartemink, A.E. Early soil knowledge and the birth and development of soil science. *Catena* **2010**, *83*, 23–33.
2. Diamond, J. *Guns, Germs, and Steel: The Fates of Human Societies*; W.W. Norton & Company: New York, 2005.
3. Miller, B.A.; Schaetzl, R.J. History of soil geography in the context of scale. *Geoderma* **2016**, *264*, 284–300.
4. Brevik, E.C.; Fenton, T.E.; Homburg, J.A. Historical highlights in American soil science—Prehistory to the 1970s. *Catena* **2015**, DOI:10.1016/j.catena.2015.10.003.
5. Krupenikov, I.A. *History of Soil Science From Its Inception to the Present*; Oxonian Press: New Delhi, India, 1992.
6. Feller, C.; Thuriès, L.J.-M.; Manlay, R.J.; Robin, P.; Frossard, E. “The principles of rational agriculture” by Albrecht Daniel Thaer (1752–1828). An approach to the sustainability of cropping systems at the beginning of the 19th century. *J. Plant Nutr. Soil Sc.* **2003**, *166*, 687–698.
7. Miller, B.A.; Schaetzl, R.J. The historical role of base maps in soil geography. *Geoderma* **2014**, *230–231*, 329–339.
8. Brevik, E.C.; Calzolari, C.; Miller, B.A.; Pereira, P.; Kabala, C.; Baumgarten, A.; Jordán, A. Soil mapping, classification, and modeling: History and future directions. *Geoderma* **2016**, *264*, 256–274.
9. Tandarich, J.P.; Darmody, R.G.; Follmer, L.R.; Johnson, D.L. Historical development of soil and weathering profile concepts from Europe to the United States of America. *Soil Sci. Soc. Am. J.* **2002**, *66*, 335–346.
10. Brevik, E.C.; Sauer, T.J. The past, present, and future of soils and human health studies. *Soil* **2015**, *1*, 35–46. DOI:10.5194/soil-1-35-2015.
11. Brevik, E.C.; Cerdà, A.; Mataix-Solera, J.; Pereg, L.; Quinton, J.N.; Six, J.; Van Oost, K. The interdisciplinary nature of soil. *Soil* **2015**, *1*, 117–129. DOI:10.5194/soil-1-117-2015.
12. Brevik, E.C.; Pereira, P.; Muñoz-Rojas, M.; Miller, B.A.; Cerdà, A.; Parras-Alcántara, L.; Lozano-García, B. Historical perspectives on soil mapping and process modeling for sustainable land use management. In *Soil Mapping and Process Modelling for Sustainable Land Use Management*; Pereira, P., Brevik, E., Muñoz-Rojas, M., Miller, B., Eds.; Elsevier: Philadelphia, in press.
13. Landa, E.R.; Brevik, E.C. Soil science and its interface with the history of geology community. *Earth Sci. Hist.* **2015**, *34* (2), 296–309.

BIBLIOGRAPHY

The number of references that could be used for this entry were very limited. However, there are many additional soil history publications available, some of which are given here.

1. Blume, H.P. Some aspects of the history of German soil science. *J. Plant Nutr. Soil Sci. Z. Pflanz. Bodenkunde.* **2002**, *165*, 377–381.
2. Bockheim, J.G.; Gennadiyev, A.N.; Hammer, R.D.; Tandarich, J.P. Historical development of key concepts in pedology. *Geoderma* **2005**, *124*, 23–36.

3. Boulaïne, J. *Histoire Des Pédologues et de la Science Des Sols*; INRA: Paris, France, 1989.
4. Gardner, W.H. Early soil physics into the mid-20th century. *Adv. Soil S.* **1986**, *4*, 1–101.
5. Gong, Z.; Zhang, X.; Chen, J.; Zhang, G. Origin and development of soil science in ancient China. *Geoderma* **2003**, *115*, 3–13.
6. Gonzalez, J.G.; Ventura, E., Jr.; Castellanos, J.Z.; Brevik, E.C. Soil science in Mexico: History, challenges, and future. *Soil Surv. Horiz.* **2010**, *51*, 63–71.
7. Helms, D.H.; Effland, A.B.W.; Durana, P.J., Eds. *Profiles in the History of the U.S. Soil Survey*; Iowa State Press: Ames, 2002.
8. Hillel, D. *Out of the Earth: Civilization and the Life of the Soil*; University of California Press: Berkeley, The Netherlands, 1991.
9. Jenny, H.E.W. *Hilgard and the Birth of Modern Soil Science*; Collana della Revisa Agrochemical: Pisa, Italy, 1961.
10. Lal, R. Evolution of the plow over 10,000 years and the rationale for no-till farming. *Soil Till. Res.* **2007**, *93*, 1–12.
11. Minasny, B.; McBratney, A.B. Digital soil mapping: A brief history and some lessons. *Geoderma* **2016**, *264*, 301–311.
12. Montgomery, D.R. *Dirt: The Erosion of Civilizations*; University of California Press: Berkeley, 2007.
13. Simonson, R.W. Historical highlights of soil survey and soil classification with emphasis on the United States, 1899–1970. International Soil Reference and Information Centre Technical Paper 18, Wageningen, The Netherlands, 1989.
14. Wagner, L.E. A history of wind erosion prediction models in the United States department of agriculture: The Wind Erosion Prediction System (WEPS). *Aeolian Res.* **2013**, *10*, 9–24.
15. Warkentin, B.P., Ed. *Footprints in the Soil: People and Ideas in Soil History*; Elsevier: Amsterdam, The Netherlands, 2006.
16. Yaalon, D.H.; Berkowicz, S., Eds. *History of Soil Science: International Perspectives*; Catena Verlag: Reiskirchen, Germany, 1997.
17. Young, A. *Thin on the Ground: Land Resource Survey in British Overseas Territories*; The Memoir Club: Plymouth, UK, 2007.

History of Soil Science: 1927–2000

Fred P. Miller

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

In 1927, the First International Congress of Soil Science was convened. By this time, soil science fragmented, multidisciplinary origins had begun to coalesce through national and international conferences the formation of scientific societies and institutional organization of its subject matter. Soil science is positioned to attain both recognition and legitimacy within the scientific community as a natural science discipline. Soil plays a major role in accommodating and driving fundamental ecological processes. It is the planetary medium in which our human sustenance is literally rooted, upon which much of civilization's activities take place, and through which most of society's waste loads are disposed of and recycled.

INTRODUCTION

The year 1927 serves as a logical divide between the early history of soil science and its 20th century counterpart. That year, the First International Congress of Soil Science was convened in Washington, D.C. By this time, soil science fragmented, multidisciplinary origins had begun to coalesce through national and international conferences the formation of scientific societies and institutional organization of its subject matter; e.g., in 1924, the International Society of Soil Science had been founded in Rome, linking the major subdisciplines of soil science (soil physics; soil chemistry; soil biology; and soil genesis, classification, and cartography, collectively known as pedology) in a single, structured international organization.^[1] Pedology was seen as an independent subdiscipline of soil science no longer tethered to its more narrowly focused mother discipline of geology.^[1–8]

While chemistry (including mineralogy), physics, and biology (microbiology) constitute major pillars upon which the discipline of soil science was founded and rests, the segment of soil science known as pedology consists of a hard core of science unique to soil science and that has no place in other more basic sciences.^[7] Pedology, therefore, constitutes the fourth pillar of the discipline of soil science—the study of the origin of soils (soil genesis), their morphology, and the relationship of soils to landscapes and ecosystems.

DEVELOPMENT OF SOIL SCIENCE IN THE UNITED STATES

The First Half of the 20th Century

In another article, John Tandarich has set forth the early history of soil science, including the development of soil

science in the United States up to 1927. During the first half of the 20th century, soil science and its literature in the United States were heavily influenced by the soil survey program, eventually called the National Cooperative Soil Survey of the United States.^[9–11]

In 1920, those working in the soil survey program organized themselves into the American Association of Soil Survey Workers, later known as the American Soil Survey Association in 1923. It was the American Soil Survey Association that sponsored the First International Congress of Soil Science in 1927 in Washington, D.C. In 1936, this association merged with the soil section of the American Society of Agronomy, which was founded in 1907, to form the Soil Science Society of America (SSSA). Although this merger was not without detractors in the soil survey program, it did unite the various sub disciplines of soil science [soil physics, soil chemistry (including mineralogy and fertility), soil biology–microbiology, and pedology] under a single professional–scientific society. As recorded in an increasing literature about soil science, much progress had been made in soil physics, chemistry, mineralogy, and microbiology by this time.^[5,6,10]

After 1929, the Great Depression coupled with the Dust Bowl of the 1930s stimulated demand for soil information in general and soil survey information in particular. Soil erosion had long been the Achilles heel of American agriculture. More information about the problem was needed by farmers and land managers to manage land better and minimize the ecological scourges of land and soil degradation. Economic conditions coupled with drought and erosion in America's heartland during the early 1930s prompted the Roosevelt administration to form the Soil Erosion Service in 1933, later restructured into the Soil Conservation Service in 1935. After a rocky fiscal start, the soil survey program was a major component underpinning these agencies' missions.^[9]

In 15 years preceding his death in 1935, Curtis F. Marbut, the American pioneer of pedology, devoted himself to developing a comprehensive natural system for classifying all known soils.^[9] Marbut's final soil classification system (1935) was revised in 1938 by his successor, Dr Charles E. Kellogg, and published in the 1938 Yearbook of Agriculture.^[12] This classification system was again revised in 1949 and published in the journal *Soil Science* as a special issue.^[13]

At the same time, the other subdivisions of soil science were making advances. Likewise, refinements in soil classification continued, and soil mapping was greatly enhanced by the introduction of aerial photography. Although the feasibility of mapping on air photos was well established by 1929, aerial photo base maps did not come into general use in soil surveys until several years later.^[9] In 1937, Kellogg issued the first Soil Survey Manual,^[14] setting forth guidelines, standards, and techniques for soil surveys.

Midway through the 20th century, the major paradigms of soil science had been established. These included:

- The pedogenesis concepts of Dokuchaev–Hilgard (vs. the earlier geology-based concepts of soil formation), refined by Marbut, Jenny, Cline, Simonson, and others.^[1,2,5,6,9,11,15]
- The chemical basis of plant nutrition, begun by Sprengel and von Liebig in Germany in the mid-1800s (vs. earlier focus on humus as the essence of soil)^[1,6] and essential concepts such as the chemistry of ion exchange, solute transport, and colloidal chemical phenomena.^[5,6]
- The theoretical basis of soil physics, such as soil hydraulic conductivity (Darcy in 1856) and soil mechanics (Terzaghi in the 1920s and the 1930s).^[1,5]
- The understanding and characterization of soil mineralogy, particularly clay mineralogy (with the advent of X-ray techniques and thermic analysis) and its roles in soil chemical phenomena and the physical behavior of soils.^[6]
- The basis of soil microbiology and the role microorganisms play in nitrification and the cycling of elements and compounds, the practical relationship of soil microbiology to plant growth, the decomposition of plant residues, the character and dynamics of humus, and an inventory of microbial species in the soil and the study of their roles.^[1,5,6]

As Boulaine^[6] noted, the end of World War II and the close of the first half of the 20th century brought much attention to the need for soil science, including “the expansion of irrigation, the conquest of newly cultivated land, attempts at rehabilitating salty soils, the drainage of marshes, and the fight against erosion requiring technical inventories, periodic surveillance, and predictions concerning the eventual evolution of the land. These needs had repercussions on

research programs and on ways of defining pedological variables.” Boulaine concluded that “a remarkable flowering of soil science” marked the next half century to follow.^[6]

The Second Half of the 20th Century

McDonald^[16] noted that the year 1950 is a logical starting point for an analysis of modern trends in soil science and its literature. In 1949, the first *Advances in Agronomy* had been issued under the auspices of the American Society of Agronomy and included a number of articles concerning soils. In 1951, the revised and expanded *Soil Survey Manual* was published.^[16,17] This book refined the standardization of soil survey procedures and guidelines as well as established the primary functions of the United States soil survey program.^[9] Research on soils was taking place in many areas; e.g., in 1952, Selman A. Waksman, a microbiologist–biochemist at Rutgers University, received a Nobel Prize for his discovery and isolation of the soil microbe streptomycin.^[1,5,6]

The year 1950 also marked the beginning of an effort to develop an entirely new natural system for classifying the known soils of the world. Since 1938 major revision of the U.S. classification system^[12] and its less intensive revision in 1949,^[13] expanded knowledge of soils, more intensive demands on the use of soils, and new and broader applications of soil surveys could no longer be reconciled with the established soil classification schemes of 1938 and 1949.^[9,18] Guy D. Smith, coauthor of the 1949 revision^[13] and a soil scientist in the United States Division of Soil Survey, assumed leadership of this international effort to develop a new comprehensive system of soil classification.^[9]

This project resulted in the publication of *Soil Taxonomy* in 1975.^[18,19] Subsequently, international committees were established by the U.S. National Cooperative Soil Survey to work on problems and concerns arising from the use of *Soil Taxonomy*.^[8] Revisions of this work resulted in further refinement of this new classification system with the world's soil resources classified into 12 major global soil orders. The lowest level in the classification system, with the most narrowly prescribed ranges of soil properties, is known as the soil series. In the United States, there are more than 14,000 established soil series, a 24-fold increase in the eighty-year period between 1912 and 1992. By the end of 1992, virtually all arable land in the United States had been classified and mapped under the mandates of the Food Security Act of 1985.^[18]

Advances in technology including isotopic tracers, computers, satellites, fractal mathematics, geo-statistics, and biotechnology have rapidly been applied to many areas of soil science. Computers and sophisticated software have allowed complex soil system behavior to be modeled. The remote sensing of landscapes via satellites and rapidly processed information contributed to the development of

sophisticated geographic information systems that provide soil scientists with powerful tools to investigate spatial soil relationships as well as predict soil behavior under various ecosystem manipulations. Biotechnology provides tools for assessing, understanding, and manipulating the soil microbes that drive many soil and ecological processes. These and many other opportunities for soil science research have been reviewed.^[20]

In the 1960s and 1970s, agricultural interests emphasized economic and resource efficiency in production. During the same period, the environmental movement became more influential. Soil scientists and their colleagues in agronomy, engineering, forestry, range management, and other scientific areas began to research not only the cropping system responses to production inputs and ecosystem manipulations, but also their environmental impacts. Both interest in and demand for such research, coupled with available funding, drove the research agenda, as reflected in the literature published during this time in the mainline soil science journals (*Journal of Environmental Quality*, *Soil Science Society of America Journal*, *Agronomy Journal*, and *Soil Science*).

The discipline of soil science, particularly pedology, has also had significant influence on American assessments of land and water resources inventories during the 20th century. After Bennett's first comprehensive soil erosion inventory in 1934,^[21] a soil-and-water conservation needs inventory was conducted and published in 1945.^[22] Later inventories of the nation's soil and water resources were conducted using scientific and statistically reliable protocols. A series of conservation needs inventories were carried out between 1958 and 1967.^[23,24] Subsequent inventories of the condition of the nation's land and water resources, known as the National Resource Inventories (NRI), were completed in response to the requirements of the Rural Development Act of 1972 and the Soil and Water Resources Conservation Act of 1977 (RCA). The first RCA appraisal^[25] was conducted in 1977. The second and more comprehensive RCA–NRI appraisal^[26] reported conditions for the nation's land for 1982. The 1987, 1992, and 1997 (revised in 2000) NRI updates have been completed^[27–29] and provide information to guide policy formulation in managing and sustaining the nation's natural resources.

During the 1990s, the discipline of soil science focused on a variety of agricultural and non-agricultural problems and issues. An assessment of the literature of soil science as it pertains to these issues can be found in McDonald.^[30] Soil scientists in the 1990s assessed soils as ecosystem components and concentrated on processes that drive ecosystem functions. Several references for this entry provide examples of the research initiatives of the 1990s, including isotopic signatures in soil development, computer-assisted tomography, thermo-mechanical processes in soils, fractal mathematics and chaotic flow analytical methodologies, aqueous and surface chemical speciation, microbial habitat assessment and genetic characterization of soil microbes,

and extraterrestrial pedology.^[20,31–35] The number of issues and problems addressed by American soil scientists is also illustrated by the topics presented at the 2000 annual meeting of the SSSA. These topics and issues contrast sharply with the subjects addressed by soil scientists years ago because of scientific advances during this time coupled with society's demands for stewarding the nation's natural resources. These topics include:

- ecosystem manipulation and response to mitigation strategies;
- capacity of soils and the environment for food production;
- carbon sequestration potential of managed soil-cropping systems;
- remediation of contaminated soils and landscapes;
- indicators of soil and environmental quality;
- sustaining ecological integrity under high-output agriculture;
- computer modeling and computer-assisted decision making; and
- riparian buffer management as a water quality protection practice.

THE STATUS OF SOIL SCIENCE AS A DISCIPLINE IN 2000

Born from the basic and natural science disciplines, but nurtured throughout the 20th century by agricultural institutions (U.S. land grant colleges and the U.S. Department of Agriculture agencies, such as the Soil Conservation Service/Natural Resources Conservation Service and the Agricultural Research Service), soil science seeks professional–scientific parity with its earth science and natural science counterparts.^[36,37] Soil science is perceived by many to be an agriculturally oriented discipline. But since World War II, and particularly since the 1970s and 1980s, soil science research and its applications have been broadened well beyond the boundaries of agriculture. Evidence of the discipline's growth as a bona fide earth science and natural science discipline includes:

- greater identity of the SSSA from its long professional partnership with the American Society of Agronomy and the Crop Science Society of America;
- SSSA's membership in the American Geological Institute;
- publication of soil science research in international science journals such as *Science*, *Nature*, and others;
- research addressing global, national, regional, and local issues in both agricultural and non-agricultural domains;
- soil scientists becoming members of the U.S. National Academy of Science and other prestigious science-oriented affiliations and awards;

- collaboration in interdisciplinary research;
- successfully competing for research grants;
- membership on political boards, advisory panels, and other political organizations guiding public policy;
- long history of international exposure and participation; and
- contributions to state-of-the-art research on a natural resource critical to the sustenance of humankind and its well-being.

CONCLUSION

At the threshold of the 21st century, soil science is positioned to attain both recognition and legitimacy within the scientific community as a natural science discipline. Soil plays a major role in accommodating and driving fundamental ecological processes. It is the planetary medium in which our human sustenance is literally rooted, upon which much of civilization's activities take place, and through which most of society's waste loads are disposed of and recycled. To paraphrase Boulaine's observation in the mid-20th century,^[6] the "flowering of soil science" should continue into the 21st century.

REFERENCES

1. Yaalon, D.H. History of soil science in context: International perspective. *Adv. GeoEcol.* **1997**, 29, 1–13.
2. Tandarich, J.P.; Sprecher, S.W. The intellectual background for the factors of soil formation. In *Factors of Soil Formation: A Fiftieth Anniversary Retrospective*; Amundson, R., Harden, J., Singer, M., Eds.; Special Publication No. 33; Soil Science Society of America: Madison, 1994; 1–13.
3. Tandarich, J.P. Pedology: disciplinary history. In *Sciences of the Earth, An Encyclopedia of Events, People, and Phenomena*; Good, G.A., Ed.; Garland Publishing Inc.: New York, 1998; Vol. 1, 666–670.
4. Tandarich, J.P. Agricultural geology: Disciplinary history. In *Sciences of the Earth, an Encyclopedia of Events, People, and Phenomena*; Good, G.A., Ed.; Garland Publishing Inc.: New York, 1998; Vol. 1, 23–29.
5. Warkentin, B.P. Trends and developments in soil science. In *Literature of Soil Science*; McDonald, P., Ed.; Cornell University Press: Ithaca, 1994; 1–19.
6. Boulaine, J. Early soil science and trends in the literature. In *The Literature of Soil Science*; McDonald, P., Ed.; Cornell University Press: Ithaca, 1994; 20–42.
7. Simonson, R.W. Soil classification in the United States. *Science* **1962**, 137, 1027–1034.
8. McCracken, R.J.; Helms, D. Soil surveys and maps. In *The Literature of Soil Science*; McDonald, P., Ed.; Cornell University Press: Ithaca, 1994; 275–311.
9. Gardner, D.R. *The National Cooperative Soil Survey of the United States*. Historical Notes No. 7; U.S. Department of Agriculture, Natural Resources Conservation Service, Soil Survey Division: Washington, 1998; 1–270.
10. Simonson, R.W.; McDonald, P. Historical soil science literature of the United States. In *The Literature of Soil Science*; McDonald, P., Ed.; Cornell University Press: Ithaca, 1994; 379–434.
11. Simonson, R.W. *Historical Highlights of Soil Survey and Soil Classification with Emphasis on the United States, 1899–1970*; Technical Paper No. 18; International Soil Reference and Information Center: Wageningen, 1989; 1–83.
12. Baldwin, M.; Charles, E.K.; Thorp, J. Soil classification. In *Soils and Men*; U.S. Department of Agriculture, U.S. Government Printing Office: Washington, 1938; 979–1001.
13. Thorp, J.; Smith, G.D. Higher categories of soil classification: Order, suborder, and great soil groups. *Soil Sci.* **1949**, 67, 117–126.
14. Kellogg, C.E. *Soil Survey Manual*; Misc. Publication No. 274; U.S. Department of Agriculture: Washington, 1937; 1–116.
15. Jenny, H. *Factors of Soil Formation: A System of Quantitative Pedology*; McGraw-Hill: New York, 1941; 1–281.
16. McDonald, P. Characteristics of soil science literature. In *The Literature of Soil Science*; McDonald, P., Ed.; Cornell University Press: Ithaca, 1994; 43–72.
17. Soil Survey Staff. *Soil Survey Manual*; Agricultural Handbook No. 18; U.S. Department of Agriculture, U.S. Government Printing Office: Washington, 1951; 1–503.
18. Simonson, R. Evolution of soil series and type concepts in the United States. *Adv. GeoEcol.* **1997**, 29, 79–108.
19. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*; USDA Handbook No. 436; U.S. Department of Agriculture, U.S. Government Printing Office: Washington, 1975; 1–754.
20. Sposito, G.R. Jr.; Frankenberger, W.T. Jr.; Sims, R.C. Jr. *Opportunities in Basic Soil Science Research*; Soil Science Society of America, Inc.: Madison, 1992; 1–109.
21. Bennett, H.H. Mimeograph. In *Report of the Chief of the Soil Conservation Service*; U.S. Department of Agriculture, Soil Conservation Service: Washington, 1935 (to the Secretary of Agriculture).
22. U.S. Department of Agriculture. *Soil and Water Conservation Needs Estimates for the U.S. by States*; Soil Conservation Service, U.S. Government Printing Office: Washington, 1945; 1–109.
23. U.S. Department of Agriculture. *Basic Statistics of the National Inventory of Soil and Water Conservation Needs*; Soil Conservation Service, U.S. Government Printing Office: Washington, 1962; 1–164.
24. U.S. Department of Agriculture. *National Inventory of Soil and Water Conservation Needs, 1967*; Statistical Bulletin No. 461; Soil Conservation Service, U.S. Government Printing Office: Washington, 1971; 1–211.
25. U.S. Department of Agriculture. *Basic Statistics: 1977 National Resources Inventory*; Statistical Bulletin No. 686; Soil Conservation Service, U.S. Government Printing Office: Washington, DC, 1982; 1–267.
26. U.S. Department of Agriculture. *Basic Statistics: 1982 National Resources Inventory*; Statistical Bulletin No. 756; Soil Conservation Service, U.S. Government Printing Office: Washington, 1987; 1–153.
27. U.S. Department of Agriculture. *Summary Report 1987 National Resources Inventory*; Statistical Bulletin No. 790;

- Soil Conservation Service, U.S. Government Printing Office: Washington, 1989; 1–37.
28. U.S. Department of Agriculture. *Summary Report: 1992 National Resources Inventory*; Soil Conservation Service, U.S. Government Printing Office: Washington, 1994; 1–54.
 29. U.S. Department of Agriculture. *1997 National Resources Inventory*; U.S. Government Printing Office: Washington. Available at www.nhq.nrcs.usda.gov/NRI/1999. (Revised December 2000).
 30. McDonald, P. *The Literature of Soil Science*; Cornell University Press: Ithaca, 1994; 1–448.
 31. Boersma, L.L. *Future Developments in Soil Science Research*; Soil Science Society of America, Inc.: Madison, 1987; 1–537.
 32. Doran, J.W.; Coleman, D.C.; Bezdicek, D.F.; Stewart, B.A. *Defining Soil Quality for a Sustainable Environment*; SSSA Special Publication No. 35; Soil Science Society of America, Inc., and the American Society of Agronomy, Inc.: Madison, 1994; 1–244.
 33. Wagenet, R.J.; Bouma, J. *The Role of Soil Science in Interdisciplinary Research*; SSSA Special Publication No. 45; American Society of Agronomy, Inc. and the Soil Science Society of America, Inc.: Madison, 1996; 123–143.
 34. Huang, P.M. *Soil Chemistry and Ecosystem Health*; SSSA Special Publication No. 52; Soil Science Society of America, Inc.: Madison, 1998; 1–386.
 35. Adriano, D.C.; Bollag, J.M.; Frankenberger, W.T. Jr.; Sims, R.C. Jr. *Bioremediation of Contaminated Soils*; Agronomy Monograph No. 37; American Society of Agronomy, Inc., Crop Science Society of America, Inc., and Soil Science Society of America, Inc.: Madison, 1999; 1–772.
 36. Simonson, R.W. Soil science—goals for the next 75 years. *Soil Sci.* **1991**, *151*, 7–18.
 37. Miller, F.P. Soil science: A scope broader than its identity. *Soil Sci. Soc. Am. J.* **1993**, *57*, 299–564.

History of Soil Science: Early to Mid-20th Century

John P. Tandarich

Hey and Associates, Inc., Chicago, Illinois, U.S.A.

Abstract

This entry briefly traces the development of soil science in the western world from its origins in Greek thought, evolving as a separate agricultural interest of classical chemistry and classical mineralogy. The agricultural interests crystallized as agricultural chemistry and agricultural geology. The science evolved from the mid-19th century from the interactions with other disciplines, principally, chemistry and geology.

INTRODUCTION

Soil science is the scientific study of soil as a unique independent phenomenon of nature, in all its complex interrelationships. The interrelationships of soil are those interactions of physical, chemical, biological, geological, geomorphological, climatological, and temporal factors that are combined to create the soil and are responsible for the geographical arrangement of soil on the shallow water-scapes and landscapes of the earth.

The historiography of soil science has been increasing since the late-20th century. Some of the works are by Boulaine,^[1,2] Jenny,^[3] Krupenikov,^[4] Simonson,^[5] Strzemeski,^[6] Tandarich,^[7–11] Warkentin,^[12] and Yaalon.^[13]

This entry ends at 1927 when the First International Congress of Soil Science was held. Miller treats soil science history subsequent to this point.

AGRICULTURAL CHEMISTRY

In the Greek world, it was Empedocles of Agrigentum who proposed four elements as the basis for all existence, namely: earth, air, fire, and water. These four elements of Empedocles, developed and refined by Aristotle, became the accepted “Classical Model” for almost 2000 years.^[14] The Classical Model of the four fundamental elements was the basis for understanding the physical world from the 8th through the 14th centuries, when knowledge centers were within monastery walls and courts of royalty. The rediscovery of the ancient Greek and Roman philosophers by Renaissance Scholars in monasteries and newly established universities revitalized scientific inquisitiveness and allowed progress in knowledge to move forward in the 15th and 16th centuries.^[10]

By the late 17th century, the Royal Society of London, and particularly Robert Boyle, recognized the agricultural value of the knowledge of soil types, their properties, and condition.^[15] Simultaneously, northern European chemists

were also analyzing and classifying soils; prominent among these were Johann Becher and his student Georg Stahl and Johann Wallerius and his student Count Gustavus Adolphus Gyllenborg.^[14] By the late 18th century, Scottish chemist Joseph Black had analyzed and classified calcareous marls.

With the rise in France of the new experimental chemistry of Antoine Lavoisier in the late 18th and early 19th centuries, scientists in many countries became involved in soil analyses, particularly focused on the organic matter component; that for them was the soil. Some of these individuals were Jean Chaptal and Jean-Baptiste Boussingault in France; Humphry Davy in England; and Albrecht Thaer, Heinrich Einhof, and their student Carl Sprengel; Jons Berzelius and his students Heinrich Rose, Eilhard Mitscherlich, and Gerardus Mulder; and Gustav Schubler in Northern Europe.^[10,14]

Scientists in Northern Europe were particularly active. In 1837, Sprengel^[16] first used the term *Bodenkunde* (soil knowledge) in his text. At the same time, Justus von Liebig, his colleagues, and his students initiated an academic tradition that continued to the early 20th century and produced many agricultural chemists.^[14] Many of these scientists were also outstanding teachers, such as Pieter Muller, Emil Ramann, Hermann Stremme, and Georg Wiegner and his student Hans Jenny in Europe; Joseph H. Gilbert in England; Alexandr Khodnev, Pavel Ilienkov, Alexandr Voskresenskii, and his student Dmitri Mendeleev, and Nikolai Zinin and his student Alexandr Butlerov in Russia; and Eben Horsford, Eugene Hilgard, John Norton, William H. Brewer, Samuel W. Johnson, and George Caldwell in the United States.^[10,14,17,18]

AGRICULTURAL GEOLOGY

Simultaneously with the development of agricultural chemistry, knowledge of soils was accumulating, expressed within what would be called geology and geography. In the

late 17th and early 18th centuries, Royal Society of London members Martin Lister, John Aubrey, and William Stukeley proposed that soils be analyzed and mapped. The 1684 work by Lister^[19] included the first soil classification scheme. In 1743, Christopher Packe^[20] published a large-scale physiographic and soils map of East Kent, England. Pioneer soil mapping of the late 18th and early 19th centuries in Great Britain and Ireland culminated in a series of county-based agricultural surveys, called “General Views of Agriculture” made for the British Board of Agriculture under the leadership of Arthur Young.^[21]

By the end of the 18th century, a renewed interest in soils developed from a redefinition of classical mineralogy by German classical mineralogist Abraham Werner into Geognosie (geognosy—eventually replaced by the name geology), Oryctognosie (modern mineralogy), and Mineral Geography.^[22,23] Geognosie was defined as “the abstract systematic knowledge of the solid earth.”^[22] That part of geognosy regarding the earth’s surface, and as applied to agriculture, was spread through Werner’s students. Werner’s student Alexander von Humboldt and his student and colleague Carl Ritter spread new geographic and cartographic methods and techniques through Europe and Russia. They influenced their academic colleagues and successors and government officials to adopt these new methods and ideas. Some of the inspired in Northern Europe were Friedrich Fallou, who coined the name Pedologie in 1862 to represent the true science of the soil elevated from Sprengel’s soil knowledge;^[24] and Albert Orth,^[25,26] who developed the soil profile concept; Alexandre Brongniart, Georges Cuvier, and his Swiss student Louis Agassiz in France; and members of the Imperial Free Economic Society of St. Petersburg in Russia such as Vasilii Dokuchaev.

In Scotland, Robert Jameson’s interpretation of that part of geognosy concerned with the earth’s surface influenced William Maclure. In the early 19th century, Maclure brought this knowledge, much of which would become known as agricultural geology, to Benjamin Silliman and his students and academic successors in the United States; some of whom were Amos Eaton, Edward Hitchcock, Ebenezer Emmons, David Dale Owen, George H. Cook, Nathaniel Shaler, William Morris Davis, Alexander Winchell, Eugene Hilgard, Thomas Chamberlin, W.J. McGee, Frank Leverett, and Curtis F. Marbut.^[7–9,11] The agricultural geologists’ concept of soil focused on soil as separate formation in its own right deserving of study.^[27] This discipline became concerned with processes leading to soil formation, such as the weathering of rocks and minerals, and how these processes influenced soil fertility.

Practicing agricultural geologists (or agrogeologists as they called themselves by the turn of the 20th century) began meeting in 1909.^[28] Van Baren^[29] summarized the state of the science of agrogeology in an introduction to a comprehensive bibliographic work on the subject by Wulff.^[30] The agricultural geology, or agrogeology, period lasted until the 1920s.^[11,13]

THE EMERGENCE OF SOIL SCIENCE

When did modern soil science emerge from agricultural chemistry and agricultural geology? It appears that the emergence took place during the period from 1837 to 1927, and particularly after 1860, as shown by the indicators that Greene^[31] proposed: 1) the establishment of major organizations representing soil science; 2) review articles and bibliographies published; and 3) textbooks published on the subject.

Rossiter^[17] forcefully argued that agricultural chemistry was already accepted, taught, and practiced in the United States by the mid-19th century. Simultaneously, agricultural geology was already establishing itself in Europe and the United States with its own literature.^[11,30] Modern soil science developed from agricultural chemistry and agricultural geology through a complex series of interactions among individual scholars; academic and governmental institutions; and scientific societies of England and Scotland, France, Denmark, Germany, Hungary, Romania, Sweden, Russia, and the United States.

Many of the major organizations representing soil science were created during the early 20th century: the American Society of Agronomy (ASA), Soils Section in 1908; the International Conferences on Agrogeology in 1909 that became the International Society of Soil Science (ISSS) in 1924;^[28] and the American Association of Soil Survey Workers in 1920 that became the American Soil Survey Association in 1922. Between 1924 and 1927, the ISSS created an organizational structure that recognized commissions (soil science subdisciplines) of soil physics, soil chemistry, soil bacteriology (biology), nomenclature, classification and cartography (what is known as pedology), soil fertility, and soil technology.^[13]

Greene^[31] noted that the appearance of historical review articles of disciplines indicates that the particular discipline is in a transitional or significant developmental state. In Russia, Dokuchaev^[32] reviewed the literature on soils in 1879. There were review articles and bibliographies on soils published immediately after the formation of the ASA,^[33,34] prior to the establishment of the ISSS,^[29,30] and in conjunction with the First International Congress of Soil Science.^[35]

Textbooks on soil science began to appear in Europe and Russia beginning in the early to mid-19th century.^[2,12,14,17,36] Briefly, some of the principal works were by Sprengel,^[16] Fallou,^[24] Orth,^[25,26] and Dokuchaev,^[32,37,38] and Dokuchaev’s student Konstantin Glinka^[39] brought the Russian ideas to the West with the assistance of Hermann Stremme and Marbut.^[40]

In England, significant publications included those by John Bennett Lawes and Joseph Henry Gilbert, who pioneered in soil fertility research at the Rothamsted Experiment Station, Lawes established in 1843,^[41,42] and by Arthur Hall^[43] and Edward J. Russell.^[44] In Germany, the

books of Emil Ramann,^[45,46] who originated forest soil science, and Hermann Stremme^[47] built upon the work of Fallou and Orth.

Significant books on soils in the United States began with Nathaniel Shaler's^[48,49] in the early 1890s. Others that followed included those by Franklin King^[50] in soil physics, Hilgard,^[51] Milton Whitney,^[52] Thomas L. Lyon, Elmer Fippin, and Harry Buckman,^[53] George Coffey,^[54] Marbut et al.,^[55] Lyon,^[56] and Lyon and Buckman^[57] that has gone through 11 editions.

The First International Congress of Soil Science was held in the United States in Washington, D.C., in 1927 and brought together scientists from the Eastern and Western worlds and from all subdisciplines of soil science for meetings and a transcontinental field trip.^[58] This meeting effectively launched modern soil science. Miller treats the history of soil science from this turning point.

REFERENCES

1. Boulaine, J. *Histoire des Pedologues et de la Science du Sol*; INRA: Paris, 1989.
2. Boulaine, J. Early soil science and trends in the literature. In *The Literature of Soil Science*; McDonald, P., Ed.; Cornell University Press: Ithaca, 1994; 20–42.
3. Jenny, H. *E.W. Hilgard and the Birth of Modern Soil Science*; Collana Della Rivista Agrochimica No. 3; CDRA: Pisa, 1961.
4. Krupenikov, I.A. *History of Soil Science from Its Inception to the Present*; Academy of Sciences of USSR, translated from the Russian Amerind Publishing: New Delhi, 1992.
5. Simonson, R.W. *Soil Classifications in the Past—Roots and Philosophies*. Annual Report 1984; International Soil Reference and Information Centre: Wageningen, 1985; 6–18.
6. Strzemiński, M. *Ideas Underlying Soil Systematics*; Pub. TT 73-54013 (Translated from Polish) Foreign Scientific Publications Department of the National Center for Scientific, Technical and Economic Information: Warsaw, 1975.
7. Tandarich, J.P.; Darmody, R.G.; Follmer, L.R. The development of pedological thought: Some people involved. *Phys. Geog.* **1988**, *9*, 162–174.
8. Tandarich, J.P.; Darmody, R.G.; Follmer, L.R. Some international and interdisciplinary connections in the development of pedology. In *Transactions of the 14th International Congress of Soil Science*, Kyoto, Japan, Aug 8–15, 1990; International Society of Soil Science: Kyoto, 1990; Vol. 5, 191–196.
9. Tandarich, J.P.; Sprecher, S.W. The intellectual background for the factors of soil formation. In *Factors of Soil Formation: A Fiftieth Anniversary Retrospective*; Amundson, R., Harden, J., Singer, M., Eds.; Soil Sci. Soc. Am. Spec. Pub. No. 33; Soil Science Society of America: Madison, 1994; 1–13.
10. Tandarich, J.P. Agricultural chemistry—disciplinary history. In *Sciences of the Earth: An Encyclopedia of Events, People, and Phenomena*; Good, G., Ed.; Garland Publishing: New York, 1998; Vol. 1, 19–23.
11. Tandarich, J.P. Agricultural geology—Disciplinary history. In *Sciences of the Earth: An Encyclopedia of Events, People, and Phenomena*; Good, G., Ed.; Garland Publishing: New York, 1998; Vol. 1, 23–29.
12. Warkentin, B.P. Trends and developments in soil science. In *The Literature of Soil Science*; McDonald, P., Ed.; Cornell University Press: Ithaca, 1994; 1–19.
13. Yaalon, D.H.; Berkowicz, S. History of soil science in context: International perspective. In *History of Soil Science: International Perspectives*; Yaalon, D.H., Berkowicz, S., Eds.; Catena Verlag GMBH: Reiskirchen, 1997; 15–46.
14. Browne, C.A. A source book of agricultural chemistry. *Chron. Bot.* **1944**, *8* (1), 1–144.
15. Geological committee. Enquiries concerning agriculture. *Philos. T. R. Soc. Lond.* **1665**, *1*, 91–96.
16. Sprengel, C.S. *Die Bodenkunde oder die Lehre vom Boden Nebst Einer Vollständigen Anleitung zur Chemischen Analyse der Ackererden*; Leipzig, 1837.
17. Rossiter, M.W. *The Emergence of Agricultural Science: Justus Liebig and the Americans, 1840–1880*; Yale University Press: New Haven, 1975.
18. Vucinich, A. *Science in Russian Culture 1861–1917*; Stanford University Press: Stanford, 1970.
19. Lister, M. An ingenious proposal for a new sort of maps of countrys, together with tables of sands and clays, such chiefly as are found in the north parts of England, drawn up about 10 years since, and delivered to the royal society mar. 12, 1683. *Philos. T. R. Soc. Lond.* **1684**, *14*, 739–746.
20. Packe, C. *Ankographia; Sive, Convallium Descriptio. In Which are Briefly but Fully Expounded the Origine, Course and Insertion; Extent Elevation and Congruity of all the Valleys and Hills, Brooks and Rivers (As an Explanation of a New Philosophico-Choreographical Chart) of East Kent*; J. Abre: Canterbury, 1743.
21. Young, A. *General View of the Agriculture of the County of Sussex*; McMillan: London, 1793.
22. Ospovat, A.M. *Short Classification and Description of the Various Rocks by Abraham Gottlob Werner*; Translator Hafner Publishing Company: New York, 1971.
23. Laudan, R. *From Mineralogy to Geology: The Foundations of a Science, 1650–1830*; The University of Chicago Press: Chicago, 1987.
24. Fallou, F.A. *Pedologie oder Allgemeine und Besondere Bodenkunde*; Verlag Schonfeld: Dresden, 1862.
25. Orth, A. *Das Geologische Bodenprofil Nach Seiner Bedautung für den Bodenwert und die Landeskultur*; Berlin, 1875.
26. Orth, A. *Die Geognostisch-Agronomische Kartierung*; Verlag von Ernst & Korn: Berlin, 1875.
27. Emmons, E.; Prime, A.J. Agricultural geology. *Am. Q. J. Agr. Sci.* **1845**, *2* (1), 1–13.
28. Szabolcs, I. The 1st international conference on agrogeology. In *History of Soil Science: International Perspectives*; Apr 14–24, 1909, Budapest, Hungary. Yaalon, D.H., Berkowicz, S., Eds.; Catena Verlag GMBH: Reiskirchen, 1997; 67–78.
29. Van Baren, J. Agrogeology as a science. *Mededeelingen van de Landbouwhoogeschool* **1921**, *20*, 5–9.
30. Wulff, A. *Bibliographia agrogeologia. Mededeelingen van de Landbouwhoogeschool* **1921**, *20*, 1–285.
31. Greene, M.T. History of geology. *Osiris* (2nd series) **1985**, *1*, 97–116.

32. Dokuchaev, V.V. *Cartography of Russian Soils*; Russian Imperial University of St. Petersburg: St. Petersburg, 1879.
33. Coffey, G.N. The development of soil survey work in the United States with a brief reference to foreign countries. *Proc. Am. Soc. Agron.* **1912**, 3, 115–129.
34. Fippin, E.O. The practical classification of soils. *Proc. Am. Soc. Agron.* **1912**, 3, 76–89.
35. McDonald, P., Ed. *A Classified List of Soil Publications of the United States and Canada*; Bibliographical Contributions No. 13; U.S. Department of Agriculture Library: Washington, 1927.
36. McDonald, P., Ed.; *The Literature of Soil Science*; Cornell University Press: Ithaca, 1994.
37. Dokuchaev, V.V. *Tchernozeme (Terre Noire) de la Russie D'Europe*; Societe Imperiale Libre Economique Imprimeric Trenke & Fusnot: St. Petersburg, 1879.
38. Dokuchaev, V.V. *Russian Chernozem*; Russian Imperial University of St. Petersburg: St. Petersburg, 1883.
39. Glinka, K.D. *Die Typen der Bodenbildung, Ihre Klassifikation und Geographische Verbreitung*; Gebrueder Borntraeger: Berlin, 1914.
40. Marbut, C.F. *The Great Soil Groups of the World and Their Development By K.D. Glinka*; Edwards Brothers: Ann Arbor, 1927.
41. Russell, E.J. *A History of Agricultural Science in Great Britain: 1620–1954*; George Allen & Unwin: London, 1966.
42. Catt, J.A.; Henderson, I.F. Rothamsted experiment station—150 years of agricultural research/the longest continuous scientific experiment? *Interdiscipl. Sci. Rev.* **1993**, 18 (4), 365–378.
43. Hall, A.D. *The Soil*; E. P. Dutton: New York, 1907.
44. Russell, E.J. *Soil Conditions and Plant Growth*, 1st Ed.; Longmans, Green and Co.: New York, 1912.
45. Ramann, E. *Forstliche Bodenkunde und Standortslehre*; Verlag von Julius Springer: Berlin, 1893.
46. Ramann, E. *Bodenkunde*, 3rd Ed.; Verlag von Julius Springer: Berlin, 1911.
47. Stremme, Hermann. *Grundzuge der Praktischen Bodenkunde*; Verlag von Gebruder Borntraeger: Berlin, 1926.
48. Shaler, N.S. *Aspects of the Earth*; Charles Scribner's Sons: New York, 1890.
49. Shaler, N.S. The origin and nature of soils. In *Twelfth Annual Report of the United States Geological Survey 1890–91*; U.S. Government Printing Office: Washington, 1891; 219–345.
50. King, F.H. *The Physics of Agriculture*; F.H. King: Madison, 1901.
51. Hilgard, E.W. *Soils*; Macmillan: New York, 1906; 593 pp.
52. Whitney, M. *Soils of the United States*; U.S. Dept. Agr. Bur. Soils Bull. 55; U.S. Government Printing Office: Washington, 1909.
53. Lyon, T.L.; Fippin, E.O.; Buckman, H.O. *Soils: Their Properties and Management*; Macmillan: New York, 1916.
54. Coffey, G.N. *A Study of Soils of the United States*; U.S. Dept. Agr. Bur. Soils Bull. 85; U.S. Government Printing Office: Washington, 1913.
55. Marbut, C.F.; Bennett, H.H.; Lapham, J.E.; Lapham, M.H. *Soils of the United States*; U.S. Dept. Agr. Bur. Soils Bull. 96; U.S. Government Printing Office: Washington, 1913.
56. Lyon, T.L. *Soils and Fertilizers*; Macmillan: New York, 1917.
57. Lyon, T.L.; Buckman, H.O. *The Nature and Properties of Soils*; Macmillan: New York, 1922.
58. Tedrow, J.C.F.; Simonson, R.W. The transcontinental excursion of the first international congress of soil science. *Bull. Intl. Soc. Soil Sci.* **1997**, 91, 75–78.

Histosols

David L. Lindbo

*V. James Research and Extension Center, North Carolina State University,
Plymouth, North Carolina, U.S.A.*

Deborah A. Kozlowski

Edenton, North Carolina, U.S.A.

Abstract

Histosols differ from other soils in that organic materials dominate. They form in wet anaerobic environments from the bottom-up rather than from the top-down. The utility of Histosols extends beyond agriculture and horticulture to a source of renewable fuel and waste treatment. The potential hazards of fire, subsidence, and building/construction difficulties can be overcome with proper management. Histosols in their natural state may be home to rare plants and animals and provide a unique environment for all.

INTRODUCTION

Definition

Soil Taxonomy^[1] defines Histosols as soils composed primarily of organic matter (OM). The amount of OM [in terms of organic carbon (OC)] needed to classify any soil as organic varies depending on the clay content of the sample (Fig. 1). It should be noted that even at high (60%) clay contents, the amount of OC needed to classify material as organic is only 18% by weight; thus, 82% of the remaining matter will be inorganic mineral material. Therefore, one should consider a Histosol not only in terms of the actual amount of organic material, but also in terms of how the organic materials dominate the soil properties.

The name Histosol has its roots in Greek *histos* for tissue and Latin *solum* for soil; thus, Histosol is “tissue soil.” The source of the tissue is the deposition and subsequent decomposition by products of both plants and animals. The deposited tissues are the parent materials. For the formation of Histosol, the rate of decomposition must be slower than the rate of deposition, resulting in an accumulation of organic material (Fig. 2).

Distribution

Histosols can be found worldwide in areas where OM is accumulating (Table 1).^[2,3] These generally are areas where the OM is deposited in saturated and anaerobic environments such as wetlands. They may occur in cold climates where decomposition is slowed down due to lower temperature and biological activity or wherever decomposition is slow.

Histosols can be divided into three groups.^[2,4] First are those that form due to landscape position in depressional areas, seeps, marshes, or where surface flow causes saturated and anaerobic conditions. These can occur in any climatic setting. Second are those formed on flat topography where annual precipitation exceeds annual evapotranspiration. Pocosins, blanket peats, and raised bogs are typical of this group.^[5–7] Third are the Histosols on mountains where extremely shallow soils are composed only of plant material. These may not remain saturated and anaerobic for extended periods, but accumulate OM due to reduced biological activity.^[2–4]

Description

Describing Histosols involves methods beyond those used for describing mineral soils. Organic horizons are described for color (wet and dry if possible), structure, consistency, reaction, roots, pores, additional features, and boundary just as mineral soils. Fiber content (excluding living roots and material <2 mm in size) is also estimated, first by observing a freshly broken face, then by rubbing the sample 10 times between thumb and forefinger and by determining the amount of fibers remaining. Both rubbed and unrubbed colors are determined.^[1,4]

All horizons considered OM based on OC content are designated as O horizons. Further divisions are based on the degree of decomposition represented in the horizon: Oi (fibric or peat) the least decomposed; Oe (hemic or mucky peat) of intermediate decomposition; and Oa (sapric or muck) the most decomposed. Peat material has > 75% fibers, muck peat has between 75% and 17% fibers, and muck has < 17% fibers remaining after rubbing.^[1]

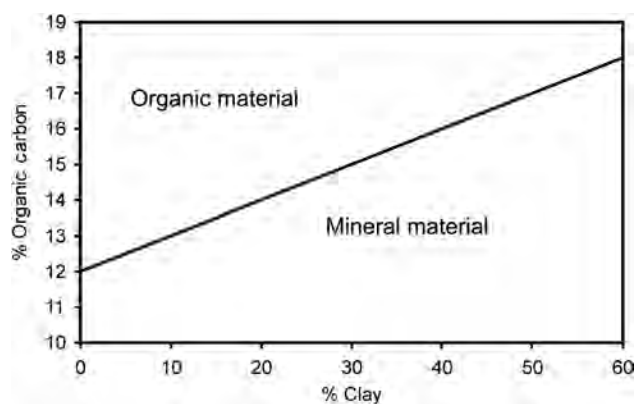


Fig. 1 Relationship between OC and clay content in order for a horizon to be considered organic.

FORMATION

In mineral soils, the least weathered or pedogenically altered material is the C horizon material, which is farthest from the surface. Organic soils are just the opposite. In order to understand this, consider that the freshly deposited litter is C horizon or parent material from which the Histosol

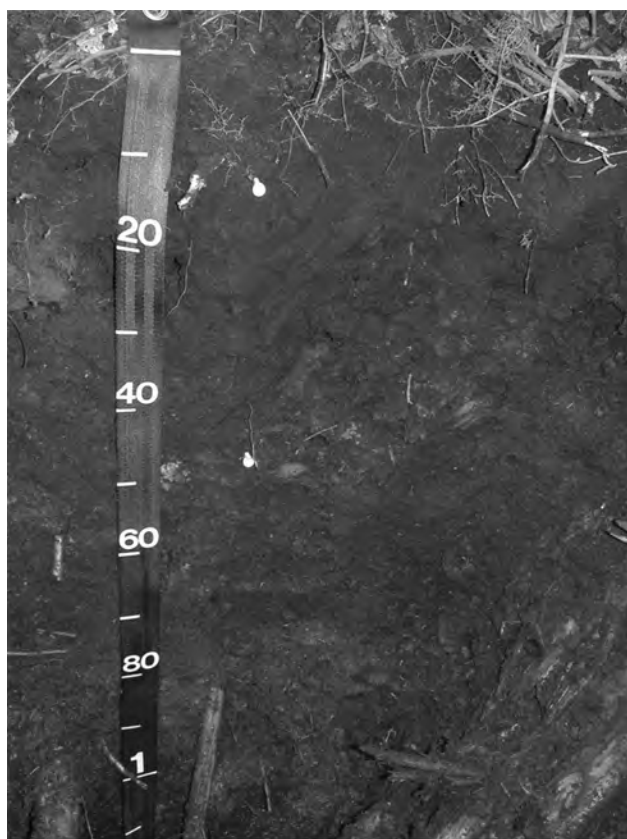


Fig. 2 Photograph of the Pungo series. This is an undrained profile located in Washington Co., North Carolina, U.S.A. Note the visible fibers near the surface and the decrease in their amount with depth. Also note the large wood fragment preserved in the profile. The scale is metric.

Table 1 Worldwide distribution of Histosols (peatlands, in 10^3 km^2).

Region	Area
Russia and Former	
Soviet Union	1537
Canada	1338
SE Asia	242
Europe	203
Alaska	131
Continental United States	102
Other	48
Total	3601

Source: From Bridgman & Ping.^[3]

developed. The litter falls on the surface of the soil, buries the decomposing OM from previous years, and begins to decompose. The bottom consists of the most decomposed material, and the top is the least decomposed.

Factors governing decomposition are temperature, moisture content, type of OM, pH, microbial activity, and time.^[4] The decomposition process must be slow enough to allow the net accumulation of OM. Saturation is one way to assure this, as high OM contents rapidly lead to anaerobic conditions. Anaerobic microbial decay is significantly slower than aerobic decay, allowing OM to accumulate. Decomposition rates also slow down as temperature drops; however, lower temperatures alone may not result in OM accumulation; in most cases, anaerobic conditions must occur still.

Ripening or aerobic breakdown may occur when organic soils are drained (Fig. 3). Ripening results in increased decomposition and overall loss in volume of the Histosol.^[8,9] Limiting the depth of drainage can control the degree of this subsidence, but if accumulation rates drop below decomposition rate, a volume decrease is inevitable. Ripening may result in the formation of strong structural units in the OM, greatly improving the hydraulic conductivity.^[7,10]

PROPERTIES

The physical and chemical properties of Histosols are related to OM content.^[4] Histosols have bulk densities below 1.0 Mg m^{-3} , but vary with the type of OM involved and mineral content. As the OM content increases, bulk density generally decreases, and as the degree of decomposition increases, bulk density increases.^[3,11] Histosols also have high *in situ* moisture contents and moisture-holding capacity because they are formed under wet conditions and have a low bulk density. The high water-holding capacity does not necessarily result in a high hydraulic conductivity as some have K_{sat} values less than those in clay soils, while others are too rapid to measure.^[7,12] The high moisture



Fig. 3 Photograph of the Scuppernong series. This is a drained profile located in Hyde Co., North Carolina, U.S.A. Note the darker surface caused by the ripening of the surface due to drainage. The scale is in inches.

Source: Photo by J.A. Gagnon, Jr., USDA-NRCS.

content results in significant shrinkage upon drying. The low bulk density and high moisture content give Histosols low bearing capacity.

As the high OM content accounts for low bulk density, it also accounts for a high cation exchange capacity (CEC), which can easily exceed $100 \text{ cmol } 100 \text{ g}^{-1}$.^[13] As the OM decomposes, more functional groups associated with the organic compounds are exposed; thus, CEC will increase with greater decomposition.

USES AND PROBLEMS

Histosols have many uses including crop production, horticultural products, fuel, and waste treatment. In their natural state, Histosols are utilized as wildlife refuges and are recognized for their aesthetic charm. They serve as a repository for paleoclimatic and anthropologic information.

Crop production is perhaps the most common commercial use of Histosols. Low bulk density makes the soils ideal

for root crops but presents problems with management.^[7,13] For equipment to traverse the low bulk density material, the soils must be drained. Once drained, the rate of decomposition increases. Management of the water table must achieve a balance among subsidence, trafficability, and plant growth. Drainage is also needed to allow proper rooting depths of crops and assure proper surface drainage. The result may be a complex management plan that includes ditches to lower the water table and improve surface drainage. The drained Histosol decomposes, releasing nutrients, but the amount released will not sustain a viable agronomic crop. Soil testing of organic soils is critical to meet the macronutrient and micronutrient needs. The lack of micronutrients in organic soils, copper in particular, was not fully understood.^[7] Once adequate levels of micronutrients were added, productivity on some organic soils exceeded that of mineral soils. Another challenge with organic soils is maintaining a proper pH. Target pH on acidic organic soils is typically lower than that on mineral soils.^[7]

Histosols are the major source of organic soil amendments such as peat moss. The material may be extracted by vacuuming the dry material off the surface or by scraping a thin surface layer off and stockpiling it until dry. Some peat resources, if managed appropriately, will heal; thus, they are considered a renewable resource.^[14]

The idea of organic soils as a renewable resource is appealing when considering its use as a fuel. Russia and Ireland have vast peat reserves and use them as fuel for heating and generating electricity. On a dry weight basis in British thermal units, organic soil falls between wood and lignite.

The high surface area and CEC of Histosols make them ideal for waste treatment.^[15] Organic material from Histosols can be used as an air filter. Sewage can be treated using peat materials both *in situ* and in specially constructed peat filters. These filters have been used to treat wastewater generated from single-family homes commonly removing 95% of the fecal coliform, 90–95% of the biochemical oxygen demand (BOD), and 95% of the total suspended solid (TSS) (Table 2, Fig. 4).^[16] Working with Histosols

Table 2 Average treatment results from four peat biofilter systems in North Carolina.

	Untreated waste water effluent	Peat biofilter effluent (treated effluent)	% change
Fecal Coliform (cfu/100 ml)	2×10^5	1.1×10^3	–99
BOD ₅ (mg L ^{–1})	143	5	–96
TSS (mg L ^{–1})	116	6	–95
Total Kjeldahl nitrogen	27.5	3	–90
NH ₄ –N	20.3	0.9	–96
NO ₃ –N	0.4	20.1	+98

Source: From Lindbo & MacConnell.^[16]



Fig. 4 Photograph of a modular peat biofilter system for wastewater treatment being installed at a single-family home in New Hanover Co., North Carolina, U.S.A. After installation, the cover would be closed and soil would be placed in a mound around the system.

does present problems such as increased fire hazard, subsidence, building and construction difficulties, and the possibility of corrosion to metal and concrete. Fire hazards are greatly increased when the soils are drained or during extreme dry periods.^[9] Once fire starts, it is extremely difficult to extinguish it and may burn deeply into the soil for months or longer until the water table rises to douse the fire.^[7]

Subsidence begins as soon as decomposition exceeds accumulation. Typically, this occurs with drainage. In cropped areas, the surfaces fall gradually. Subsidence can cause roadways to buckle or washboard if support techniques or removal of the organic material is not done appropriately. Buildings may end up “floating” precariously as the ground surface drops.

Beyond subsidence, in construction, the low strength of the material also poses a challenge. Vehicles working on Histosols need to have low bearing weights in order to avoid sinking in the muck. Prior to the use of large equipment on Histosols, plow animals may be equipped with special bog shoes to prevent them from sinking. The low pH of Histosols may cause corrosion; therefore, metal or concrete requires prophylactic treatment.

CONCLUSION

Histosols differ from other soils in that organic materials dominate. They form in wet anaerobic environments from the bottom-up rather than from the top-down. As long as enough water is present to slow the decomposition rate below the rate of accumulation, Histosols may form. The high OM content in Histosols accounts for

their low bulk density, low strength, high surface area, and high CEC. These properties not only lead to high crop productivity, but also lead to increased management concerns involving drainage, macro- and micronutrient needs, and trafficability. The utility of Histosols extends beyond agriculture and horticulture to a source of renewable fuel and waste treatment. The potential hazards of fire, subsidence, and building/construction difficulties can be overcome with proper management. Finally, Histosols in their natural state may be home to rare plants and animals and provide a unique environment for all.

REFERENCES

1. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*; Agricultural Handbook No. 436; Soil Conservation Service, United States Department of Agriculture: Washington, 1975; 754 pp.
2. Buol, S.W.; Hole, R.J.; McCracken, R.J.; Southard, R.J. *Soil Genesis and Classification*, 4th Ed.; Iowa State Press: Ames, 1997; 527 pp.
3. Bridgham, S.D.; Ping, C.-L.; Richardson, J.L.; Updegraff, K. Soils of northern peatlands: Histosols and gelisols. In *Wetland Soils*; Richardson, J.L., Vepraskas, M.J., Eds.; Lewis Publishers: Boca Raton, 2001; 343–370.
4. Collins, M.E.; Kuehl, R.J. Organic matter accumulation and organic soils. In *Wetland Soils*; Richardson, J.L., Vepraskas, M.J., Eds.; Lewis Publishers: Boca Raton, 2001; 137–162.
5. Daniels, R.B.; Gamble, E.E.; Wheeler, W.H.; Holzey, C.S. The stratigraphy and geomorphology of the Hoffman Forest Pocosin, North Carolina. *Soil Sci. Soc. Am. J.* **1977**, *41*, 117–180.
6. Inavov, K.E. *Water Movement in Mirelands*; Academic Press: London, 1981; 349 pp.
7. Lily, J.P. *The Blackland Soils of North Carolina: Their Characteristics and Management for Agriculture*; Technical Bulletin 270; North Carolina Agricultural Research Service: Raleigh, 1981; 70 pp.
8. Everett, K.R. Histosols. In *Pedogenesis and Soil Taxonomy, II. The Soil Orders*; Wilding, L.P., Smeck, N.E., Hall, G.F., Eds.; Elsevier: Amsterdam, 1983; 1–53.
9. Lucas, R.E. *Organic Soils (Histosols): Formation, Distribution, Physical and Chemical Properties and Management for Crop Production*; Michigan State University, Agricultural Experiment Station: East Lansing, 1982; 77 pp.
10. Boulter, D.H. Hydraulic conductivity of peats. *Soil Sci.* **1965**, *100*, 227–231.
11. Boulter, D.H. Physical properties of peats as related to degree of decomposition. *Soil Sci. Soc. Am. Proc.* **1969**, *33*, 606–609.
12. Skaggs, R.W.; Gilliam, J.W.; Sheets, T.J.; Barnes, J.S. *Effect of Agricultural Land Development on Drainage Water in the North Carolina Tidewater Region*; Report 159, Water

- Resources Institute, University of North Carolina: Raleigh, 1980; 164 pp.
13. Dolman, J.D.; Buol, S.W. *A Study of Organic Soils (Histosols) in the Tidewater Region of North Carolina*; Technical Bulletin 181; North Carolina Agricultural Research Service: Raleigh, 1967; 52 pp.
 14. Northlands Associates Ltd. *Research on Bog Preparation and Peat Sod Drying. Techniques in Newfoundland (1986–1989)*; Northlands Associates Ltd: St. Johns, 1989.
 15. Farnham, R.S. Use of organic soils for wastewater filtration. In *Histosols: Their Characteristics, Classification and Use*; Aandahl, A.R., Ed.; Special Publication No. 6; Soil Science Society of America: Madison, 1974; 111–118.
 16. Lindbo, D.L.; MacConnell, V.L. Evaluation of a peat biofilter treatment system. In *On-Site Wastewater Treatment*, Proceedings of the Ninth National Symposium on Individual and Small Community Sewage Systems; Fort Worth, TX, Mar 2001, Mancel, K., Ed.; ASAE: St. Joseph, 2001; 225–234.

Human Culture and Soils

Daniel Hillel

Center for Climate Systems Research, Columbia University, New York,
New York, U.S.A.

Abstract

For at least 90% of its existence as a species, the human animal functioned merely as one member of a community of numerous species who shared the same environment. Early humans were adapted to subsist within the bounds defined by the natural ecosystem. By and large, our ancestors led a nomadic life, roaming in small bands, foraging wherever they could find food. They were gatherers, scavengers, and opportunistic hunters. Unlike their primate cousins who remained primarily vegetarian, humans diversified their diet to include the flesh of whichever animals they could find or catch, as well as a variety of plant products such as nuts, berries and other fruits, seeds, some succulent leaves, bulbs, tubers, and fleshy roots.

INTRODUCTION

Our *species*' birthplace was evidently in the continent of Africa, and the original habitat was probably the sub-tropical savannas that constitute the transition zone of sparsely wooded grasslands lying between the zone of the humid and dense tropical forests and the zone of the arid steppes. We can infer the warm climate of our place of origin from the fact that we are naturally so scantily clad, or furless, and we can infer the open landscape from the way we are conditioned to walk, run, and gaze over long distances.

EARLY DEVELOPMENT

Some hundreds of thousands of years ago, members of the *Homo* genus learned to fashion stone tools as well as to set and use fire, probably at first only to cook and soften food. Later, they used fire to clear woody vegetation, to flush out game, and to encourage the subsequent grazing of ungulates, which they could thereby hunt more readily. Eventually, that practice had a great effect on the environment, as brush fires apparently set by humans modified entire ecosystems on an increasing scale. The control of fire became even more important when humans moved out of the tropics into colder climes, where bonfires and hearths were needed to warm their shelters in winter as well as to cook their food. In time, the clearing of woodlands and shrublands by repeated firings also set the stage for the advent of agriculture. However, as the vegetative cover is affected by fire, so is the underlying soil. Following repeated denudations of the land, soil organic matter and nutrients were gradually depleted, while soil erosion took place,

resulting in the increased transport of silt by streams and its deposition in river valleys and estuaries.

By the later stages of the so-called Paleolithic period, nearly all the regions of human habitation had experienced some anthropogenic modification of the floral and faunal communities. The gradual increase in human population density required the development of more intensive methods of land use, aimed at inducing an area to yield a greater supply of human needs. The selective eradication of less desirable plant and animal species and the encouragement of desirable ones (producing edible and, preferably, storable products) led eventually to the domestication and propagation of crops and to purposeful soil management aimed at creating favorable conditions for production. That is to say that these activities culminated in the development of agriculture and the sedentary (rather than nomadic) way of life in regular villages.

THE AGRICULTURAL TRANSFORMATION

Concurrently with the domestication of plants, humans also learned to select and domesticate animals. The earliest domesticates in the ancient Near East were sheep, goats, and bovine cattle, which could be herded and pastured in the semiarid rangelands. Cattle (oxen) were also used as beasts of burden to pull carts and plow the soil. Donkeys, horses, and eventually camels were also harnessed in the same region to convey people and goods.

The agricultural transformation, also called the Neolithic Revolution, was very probably the most momentous turn in the progress of humankind, and many believe it to be the real beginning of civilization. It first took place in the Near East approximately 10 millennia ago, initially in the

relatively humid subregions where precipitation was sufficiently abundant and regular to permit rain-fed farming. Agriculture was later extended to river valleys in the drier zones by means of irrigation, which was based on the diversion of water from rivers onto adjacent flat lands. At the same time, humans discovered that, by kneading and shaping lumps of clayey soil and then baking them in ovens, they could fashion vessels for storing water as well as for cooking food. Bricks made of molded clay served for the construction of homes and grain storages, while tablets of clay could be imprinted with various signs to form permanent records in an early form of writing called cuneiform. Later still, people learned to extract, alloy, and mold metals such as copper and iron and to fashion them into implements.

Rain-fed and/or irrigated farming practices, along with ceramics and metallurgy, later spread to, or were developed independently in, most of the other human-inhabited regions around the world.

The ability to raise crops and livestock, while resulting in a larger and more secure supply of food than was possible previously, definitely required attachment to controllable sections of land and hence brought about the growth of permanent settlements and of larger coordinated communities. The economic and physical security so gained accelerated the process of population growth and necessitated further expansion and intensification of production.

LAND DEGRADATION

The evolution of agriculture left a strong imprint on the land in many regions. The vegetation, animal populations, slopes, valleys, and soil cover of land units were radically altered. The practices of tillage and fallowing, terracing, and irrigation and drainage, as well as grazing of flocks, further accelerated soil erosion in sloping lands and sedimentation in lowlands. Soil lost from denuded cultivated slopes could not be regenerated unless the land was allowed to revert to its vegetative cover for many decades or centuries. Moreover, soils that were irrigated in poorly drained river valleys tended to become waterlogged and saline, with the result that the practice of farming could not be sustained there in the long run.

The same anthropogenic processes, which began in the early history of civilization, have continued ever since on a more extensive scale. Especially vulnerable are cultivated and overgrazed soils in semiarid regions, where droughts occur periodically, accompanied by strong winds that deflate the bare and pulverized soil surface, and where the intermittent onset of rains causes further damage. Consequently, once-productive lands may become so degraded as to acquire desert-like characteristics. Hence, the process of land degradation in semiarid regions has been termed “desertification.”

Examples of large-scale land degradation can be cited in many parts of the world. A case in point is the famous

“Dust Bowl” that took place in the Southern Great Plains of the United States in the 1930s and that caused the uprooting and migration of entire farming communities. A similar process has been occurring on a vaster scale in the African Sahel which is the continent-wide semiarid savanna belt that lies between the Sahara desert in the north and the tropical forests in the south. A particularly disastrous example of waterlogging and salination because of irrigation is seen in the Aral Sea Basin of Central Asia (Uzbekistan, Turkmenistan, and Kazakhstan). The same processes are occurring in highly developed parts of the world, such as the Murray–Darling Basin of Australia or San Joaquin Valley of California. Even more consequential is the large-scale destruction of rainforests and the loss of biodiversity resulting from the expansion of agriculture in the tropical zones of South America, Central Africa, and Southeast Asia, where the warm temperatures and high humidity contribute to the rapid decomposition of the soil’s original organic matter content and to the deterioration of the soil’s structure as well as its nutrient content.

THE IMPERATIVE TO IMPROVE SOIL MANAGEMENT

In addition to its uses for agricultural production, the soil also serves human communities as a depository of waste products—domestic, agricultural, and industrial. In principle, the soil is endowed with inherent capacities to absorb, retain, and decompose many waste materials and pathogenic agents that might otherwise accumulate to poison the environment. However, those capacities are limited, and when exceeded by the application of larger than normal quantities of potentially toxic wastes (such as pesticides, heavy metals, hydrocarbons, and non-biodegradable synthetic compounds), the soil may merely be a way station, detaining but not retaining harmful agents, which it eventually releases to the biosphere.

Prospects for improving the management of soils in the future will likely be affected by the threat of global warming because of the increasing concentrations of radiatively active gases in the atmosphere. In a warmer world, all weather phenomena may be intensified, with more frequent and stronger rainstorms punctuating longer and more severe droughts. Such a trend will require far reaching and expensive adjustments in the zonation and methodology of agricultural production as well as in water storage and supply facilities.

CONCLUSION

Humanity depends on the soil more crucially than ever, not only because population has grown but also because available soil and water resources have diminished and deteriorated. Our needs and those of future generations require us to seek better ways to manage those precious resources

judiciously and sustainably. What past generations despoiled inadvertently owing to lack of comprehensive scientific knowledge or social-environmental consciousness, we must begin to rectify deliberately. Developing, disseminating, and applying knowledge of the soil and its processes are essential tasks if we are to ensure the lasting welfare of humanity as well as the terrestrial ecosystem as a whole.

BIBLIOGRAPHY

1. Boyden, S. *Biohistory: The Interplay between Human Society and the Biosphere*; UNESCO: Paris, 1992.
2. Goudie, H. *The Human Impact on the Natural Environment*; MIT Press: Cambridge, 1986.
3. Hillel, D. *Out of the Earth: Civilization and the Life of the Soil*; University of California Press: Berkeley, 1992.

Human Dimensions: Soil Sustainability

Mary Seely

Patrik Klintenberg

Claus Hager

Desert Research Foundation of Namibia, Windhoek, Namibia

Abstract

Human dimensions of soil sustainability are manifold. Soil sustainability depends on a variety of human actions that influence the soil inadvertently or with deliberate intention to do so. Inadvertent influences may be brought about by a diversity of socioeconomic changes including, inter alia, the global, national, or local economy; political regimes; changing global markets; urban expansion; labor availability; farming systems; residential patterns; or local development. These changes may be beyond the control of the individual farmer, cooperative, or industry or be based on the farmers' selection of alternative socioeconomic priorities unrelated to soil productivity. Most deliberate influences on the soil focus on enhancing its productivity or its long-term stewardship, particularly in developed economies. This entry provides examples from a developing country of human dimensions indirectly or directly related to soil sustainability.

INTRODUCTION

The goal of this entry is to use the example of central northern Namibia to probe some of the human dimensions indirectly or directly related to soil sustainability. Although the examples are specific, the principles elaborated are broadly applicable. Central northern Namibia is the most densely populated area of the country, where a dual farming system of crop production, primarily pearl millet, and livestock management is pursued under low rainfall on Kalahari sands in an ephemeral wetland.^[1] The population has grown eightfold in the 20th century during which time human immunodeficiency virus (HIV) and acquired immunodeficiency syndrome (AIDS) appeared and an imperfect market economy became established, all leading to major socioeconomic changes.

People and their livestock settled in the Cuvelai inland delta approximately 400 years ago.^[1] Settlement began by clearing and cultivating the stabilized Kalahari dunes lying among ephemeral watercourses (oshanas) and shallow lakes. Early farms were laid out perpendicular to the oshanas sometimes encompassing as many as 8 or 10 of the 26 different recognized indigenous land units.^[2] Rainfall is the primary factor determining the time of planting, and in poor rainfall years, planting is commonly staggered to ensure at least some harvest. In a well-established farm, the most fertile soil types are chosen for early planting. Originally, the low population allowed a system of livestock management that brought cattle close to the crop farms for part of the year where they were enclosed and manure could accumulate. They were allowed into the fields after the millet

harvest to graze on crop residues and deposit manure directly. With an increasing population and smaller, more closely situated farms, livestock remain at cattle posts throughout the year, while millet stalks are used, inter alia, for household construction and fencing.^[3] In 2005, crop productivity of this area is extremely low at 250 kg/ha,^[4] less than a third of the global average yield.^[5]

SOCIOECONOMIC DETERMINANTS OF SOIL SUSTAINABILITY

Changes in labor availability have a major impact on farming practices and consequently on use, fertility, and sustainability of soil. In the past, men were responsible for the preparation of the fields for planting, which included distributing manure from the livestock enclosures.^[3] In the past, development of migrant labor meant that most men worked away from home. In some instances, remittances were available and used to support additional family members in the homestead providing alternative labor. The cash economy, fuelled by globalization, and preference for white-collar work, particularly with government, draw the men and some women to distant towns for jobs and young men to the capital in search of employment. Many of the men remaining at the homestead are old-age pensioners. As a consequence, smaller fields may be cultivated, often by older women, while the absence of livestock in the vicinity means little or no application of manure. Confounding this situation is the requirement for children to be in school, often in boarding schools far from the homestead, so this

source of part-time labor is also unavailable. In 2003, the impact on farming labor and soil management is the advent of HIV and AIDS.^[6] This has reduced the life expectancy of people in the rural areas by approximately 20 years and has impacted primarily on the productive population directly removing people from the labor pool, necessitating time and effort for care of those affected and reducing remittances. For wealthier farmers, hiring labor to fill the gap is an option but one that is only used for livestock herders, not for addressing crop farming or the soil quality that makes this profitable.

Changing approaches to farm and homestead management have also had an impact on soil management and sustainability. With the increase in population, shifting cultivation and leaving fields to fallow is no longer an option. New fields are opened up, but these are on deeper sandy soils or soils with shallow hardpans away from the central ephemeral wetlands. At the same time, transport of manure from cattle posts, often at distant grazing areas, is too expensive for most. Alternative fertilizer sources are also considered too expensive and are hardly used, while available cash is spent on livestock for its cultural and status values.^[1]

Another development affecting soils is the construction of permanent brick houses. Previously households were constructed of wood with thatched roofs, which were moved short distances at approximately decadal intervals. Deforestation and lack of suitable building materials, combined with the desire for a modern home, have promoted this shift. The site of the previous household was considered more fertile than the surrounding soil and usually planted for the first rains.^[2] The shade trees, which had been protected at the previous homestead, continued to provide shade and reduced drying of the soil.^[7]

Mainly, the presence of women at the homestead to prepare the fields and manage the crop has led to greater dependency on tractors or draught animals. This broad-scale approach has reduced the ability to selectively plant on particular soil types, often only several square meters in size. Interest in and knowledge of different soil characteristics are decreasing. The introduction of government-subsidized chemical fertilizers has also contributed to this shift. Moreover, the change from polygamy to monogamy in household composition has reduced the availability of womanpower to prepare fields by hand in the absence of men to do this traditional job.

MANAGEMENT OF SOIL QUALITY AND VARIABILITY

Direct management of soil character and its variability to increase productivity is practiced according to the options available. Farmer selection of farms to span several soil types, elevations, and distances, ranging from low dune crests to oshanas, was an approach used before the increased population had occupied all available,

high-quality land.^[8] Livestock were managed to provide manure directly to the crop fields when low human population pressure allowed livestock to move between the grazing area and the homestead with its crop fields.

When the population density was low and before the advent of tractors, trees were grown within fields providing shade and contributing to soil moisture retention. This tradition may not have represented direct management of soil moisture, but the value of maintaining selected trees within the crop fields is recognized.^[9]

Knowledge of soil types is used extensively by older farmers,^[8] although maintenance of this information base is questioned by researchers.^[9] Further studies have indicated that the main local indicators of soil types are reduced to crop yield and soil color with more detailed knowledge restricted to older persons. Awareness of the presence of a hard pan, which influences the selection of crops, e.g. millet vs. sorghum, remains.

Application of manure and crop residues through management of livestock was a primary tool to manage soil fertility. However, manual application of crop residues and ashes from crop residues, as well as application of household waste, is being used.^[9] In general, the cropping system on Kalahari sands in the oshana area is considered to be one of the low-input and low-output systems providing some basic sustenance. Nevertheless, it receives little focused attention in a culture entirely based on livestock for its social and status value.

Recommendations for soil and productivity improvement have included using legumes interplanted among millet and other cereal crops. Traditionally, some legumes were planted with millet, and this type of soil improvement continues.^[3]

The concept of conservation tillage has been introduced and is being used in a few pilot sites. Conservation tillage concentrates on the use of scarce resources (labor, seeds, fertilizer and organic matter, and water) on selected areas, chosen on the basis of indigenous knowledge of soil types and microtopography. Furrows are intensively prepared throughout the year and repeatedly over the seasons, with the advantage of spreading labor requirements and building up soil structure and water retention capacity in the plant root zone.^[10]

CONCLUSION

The human dimensions of soil sustainability are varied. Most are inadvertent relationships brought about by socioeconomic developments at local, national, or global scale. These developments include family structure, age, migration, labor availability, gender, health, farm management, and residential and urban expansion as well as the changing environmental conditions and strong cattle culture. Some relationships are based on direct interventions by people living on the soil or by external sources such as government directives or global markets. In the developed countries,

economic aspects drive the relationship between soil productivity and sustainability. In the developing countries, rapidly changing social and local to national economic conditions tend to be most important.

REFERENCES

1. Mendelsohn, J.M.; el Obeid, S.; Roberts, C.S. *A Profile of North-central Namibia*; Gamsberg Macmillan: Windhoek, 2000.
2. Verlinden, A.; Dayot, B. A comparison between indigenous environmental knowledge and a conventional vegetation analysis in north central Namibia. *J. Arid Environ.* **2005**, *62*, 143–175.
3. Cunningham, T.; Hubbard, D.; Kinahan, J.; Kreike, E.; Seely, M.; Stuart-Williams, V.; Marsh, A. *Oshanas, Sustaining People, Environment and Development in Central Owambo; Namibia*; Desert Research Foundation of Namibia: Windhoek, 1992; 52 pp.
4. RoN (Republic of Namibia). *Agricultural Statistics Bulletin*; Ministry of Agriculture, Water and Forestry: Windhoek, 2005; 32 pp.
5. ICRISAT. *Pearl Millet*. Available at <http://www.icrisat.org/crop-pearlmillet-genebank.htm> (accessed August 20, 2015).
6. NPC. *Population and Housing Census, 2001: National Report, Basic Analysis with Highlights*. Central Bureau of Statistics and National Planning Commission: Windhoek, 2003.
7. Marsh, A. *Trees—Threatened Lifelines of Namibia: The People's Perspective*; Gamsberg, Macmillan: Windhoek, 1994; 72 pp.
8. Verlinden, A.; Seely, M.K.; Hillyer, A.E.M. Settlement, trees and termites in Central North Namibia: A case of indigenous resource management. *J. Arid Environ.* **2006**, *66*, 307–335.
9. Palmer, M.W.; Movirongo, C. Evaluating an indigenous knowledge land classification system in North Central Namibia, Submitted to Agriculture; Ecosystems and Environment, 2008.
10. Misika, P.; Mwenya, E. Conservation tillage with animal traction for soil-water management and environmental sustainability in Namibia. In *Conservation Tillage and Animal Traction*; Kaumbotho, P.G., Simalenga, T.E., Eds.; Animal Traction Network for Eastern and Southern Africa (ATNESA): Harare, 1999; 22–26.

Human Health and Soil

Charles F. Sing

David B. Sing

Department of Human Genetics, University of Michigan, Ann Arbor, Michigan, U.S.A.

Abstract

Soil is a complex system of minerals, organic matter, and biota that covers the terrestrial earth in a layer above the underlying bedrock. Soil is the ecological support structure for all terrestrial life, including humans. Human health is an optimal state of general physiological and psychological well-being associated with a lack of disease.

Health is a dynamic emergent property that unfolds over the lifetime of the individual as a consequence of the complex interactions of an inherited biological predisposition with exposures to environmental agents. The relationship of soil with human health is a subset of an ecological whole, which includes the entire ecosystem of the planet. Agriculture is a human activity designed to produce domesticated plants and animals and involves the modification of landscapes and soils and the breeding of plants and animals to maximize the production of commodities to be processed into food. Modern agriculture is organized into farms. Farms are human-operated land management systems in which farming activities replace normal ecosystem feedback controls. The relationships between soil, food, and human health are first derivatives of local and national agricultural practices and policies that are influenced by the economic and cultural norms of society. The management of agricultural soil is a proven force in the relationship of soil and human well-being. While the ecology of soils is poorly understood, research in the fields of epigenetics and nutrigenomics suggests a deeply embedded crucial relationship between human health and our genetic potential for responding to exposures to nutrients in food and to contact with the soil. Epigenetics is the study of the influences of environmental exposures on the expression and imprinting of genes. The field of nutrigenomics studies the role of nutrients on gene expression. The study of the role of exposures to soil constituents and components of food in determining gene function is an exciting field of research, which provides the possibility to understand the biological mechanisms that are responsible for the observed associations between soil biology and human phenotypes of health and disease.

INTRODUCTION

Soil is the ecological support structure for all terrestrial life, including humans. As the advent of agriculture, our species has developed a strong relationship with soils, as our nutrient and energy requirements are met by the cultivation and processing of plants and animals grown in and on the soil. Civilizations have risen and fallen on the quality and management of their agricultural soils. Biological connections have been discovered between human health and the nutritive elements and biota found indigenously in soils. There is a great potential for discovery, both in the ecology and biology of soils and in their relationships with human health. Although the ecology of soils is poorly understood, research in the fields of epigenetics and nutrigenomics suggests a deeply embedded crucial relationship between human health and our genetic potential for responding to exposures to nutrients in food and to contact with the soil. In this entry, we introduce the fundamental relationships between soil and human health and call attention to the potential for understanding those relationships that are emerging as a consequence of advances in genetic biology.

Soil is a complex system of air, water, minerals, organic matter, and biota that covers the terrestrial earth in a layer above the underlying bedrock. The multitude of interactions among the myriad constituents of soil has been interpreted to infer that soil is a highly dynamic, ecologically complex, and diverse living entity (see the entry *Human Society and Soil*, p. 1123). A healthy agricultural soil is characterized by a diverse compilation of soil components, including a mixture of biota and organic materials,^[1] that provide the condition of optimal fertility for growing healthy plants for animal feed and for processing into human food. Food is consumable material derived from plants and animals that have been selected by humans for their nutritive and energy values and for their ease of domestication, commodification, and processing. Human health is an optimal state of general physiological and psychological well-being associated with a lack of disease. Health is a dynamic emergent property that unfolds over the lifetime of the individual as a consequence of the complex interactions of an inherited biological predisposition with exposures to environmental agents, which include direct contact with soil and dietary components that are

derived from the soil through plants and animals.^[2] Agriculture is a human activity designed to produce domesticated plants and animals. Modern agriculture involves the modification of landscapes and soils and the breeding of plants and animals to optimize the production of commodities to be processed into food. Modern agriculture is organized into farms. Farms are human-operated land management systems in which farming activities replace normal ecosystem feedback controls.^[3] Taken as a whole, the relationship of soil with human health is a subset of an ecological whole, which includes the entire ecosystem of the planet. This relationship is influenced by the cultural practices of the people who grow and eat the food.

Despite an age-old consensual cultural agreement that there is a connection between the health of the soil and human health, convincing evidence for direct connections is scant. Our limited understanding of the efficacy of soil in regard to human health has been mainly concerned with food and food products. As a nascent understanding of soil ecology becomes more developed, it is becoming apparent that human health is also influenced by a direct contact with the soil biota and with the organic and inorganic components of the soil.^[4,5] For approximately 10,000 years since the beginning of agriculture, human culture has heavily influenced the management of soils and how plants and animals are selected, grown, and processed into food. Later, we briefly review the direct effects of soil on human health, the effects of soil on health translated through food, and the effects of cultural practices that impact the soil health–human health relationship.

SOIL CONTACT AND HUMAN HEALTH

Throughout our evolutionary past, human health has been influenced by a contact with the soil, either directly by dietary ingestion of soil (geophagy) or indirectly by breathing airborne soil constituents or by ingesting soil constituents in drinking water. Soil contact can expose humans to soil elements and biota that are beneficial to health and to toxic materials and pathogenic organisms that contribute to the risk of disease. Geophagy developed in preagricultural humans as a method of ingesting nutritive elements otherwise not found in available foods and as a way to detoxify certain plant products to make them edible.^[6] Ingestion of certain clays has been an important human dietary behavior since the origin of our species, providing nutritionally significant amounts of nutritive elements in regions where diets are otherwise lacking in these constituents. It has been suggested by Johns and others that the earliest cultivation of certain poisonous plants, such as potato, depended on human geophagic knowledge of the detoxifying properties of certain soils.

Geophagy persists as a developmental behavior of young children who consume soil and other non-food materials as a means of exploring and assessing their

environment.^[7] In urban and suburban settings, such geophagic activity can expose individuals to toxins that have accumulated in the soil, such as lead and other heavy metals, and pathogenic organisms, such as clostridium tetani (causing tetanus) and helminth nematodes (hookworm, etc.).^[7–9] Conversely, there are also soil biota that may be ingested that are beneficial to the development of the human immune system. It has been suggested that the urbanization of human populations and a cultural preference for an antiseptic microenvironment have, for many people, created a physical disconnect with soils that may be responsible for a marked increase in asthma and other atopic diseases in Western cultures.^[10]

From an epidemiological perspective, the discussion of the relationship between contact with the soil and human health has focused on specific environmental exposures that lead to disease or nutritional toxicity.^[4,5] Potential adverse exposures include artificially introduced toxic materials, e.g., contained in fertilizers and pesticides, biologic pathogens, and certain naturally occurring elements in toxic concentrations. Unhealthy levels of some elements may accumulate in livestock, fruits, and vegetables. Conversely, consuming foods grown in soils with low levels of certain essential elements can lead to nutritive deficiencies in humans. Damage to produce during production can allow for toxins and pathogens existing in the soil to enter and accumulate in the damaged fruits and vegetables. Humans are then exposed to these toxins and pathogens when they consume damaged or improperly cleansed fruits and vegetables.^[4,5,6]

Leaching of soil nutrients, such as nitrates, from agricultural soils into groundwater is a common process that connects soil with human health. Nitrogen is an essential component in the growth and metabolism of all living systems and exists environmentally in a number of nitrogenous compounds, including ammonia (NH₄), nitrates (NO₃), and nitrites (NO₂). As the advent of nitrogen-rich fertilizers in the early 20th century, food production has greatly increased, as has the amount of nitrates introduced into agricultural soils. These excess nitrates accumulate in the soil and are leached into groundwater. Nitrate-consuming bacteria in the soil and groundwater can convert the excess nitrates into nitrites, which are toxic and have serious implications for human health. Intrauterine exposure to high levels of maternal dietary nitrites in the drinking water can lead to methemoglobinemia (blue-baby syndrome), a potentially fatal disease.^[11] People living in agricultural areas where high-nitrogen fertilizers are applied to the soil and whose source of drinking water is groundwater wells are at the greatest risk of exposure to toxic levels of nitrites.

HEALTHY SOIL, HEALTHY FOOD, AND HUMAN HEALTH

The connection between soil health and human health through foods is largely unappreciated where food is

abundant and the quantity of fertile soils is not a limiting factor. Most foods consist of processed plant and animal products that provide energy and nutrients for developing and maintaining the body. The dominant American food culture is characterized by food products that are produced in high quantity and are highly processed amalgams of different commodities from various, and often geographically disparate, locations. Processed foods high in refined carbohydrates (sugar) and certain types of fat have been shown to be detrimental to human health.^[12,13] In North America over the past several decades, a marked increase in diet-related human illnesses and chronic morbid conditions, including obesity, cardiovascular diseases, diabetes, and certain cancers of the digestive and endocrine systems, have been associated with increased availability of fatty and sugary food products.^[12,13] In general, North American agriculture supports a food industry that produces inexpensive, convenient, high-calorie, nutrient-poor food products that can be mass processed and distributed to a public that expects their foods to be inexpensive, convenient, rich in flavor, and high in caloric potential and that perceives more nutritive, less-processed foods as expensive, and more time-consuming to prepare and consume. An emerging awareness of the health consequences of this food culture is reflected in a surge of consumer interest in organic and locally grown foods in North America and Europe.^[14,15]

In North America, the soil where most food is grown is of relatively high quality, assuring any number of nutritious food choices to be produced at high levels. In regions of the world where agricultural soils are marginal or degraded, overall crop production is often inadequate, and the nutritive quality of the foods available is limited by the poor quality of the soils that remain arable. Consequently, the people of these regions often suffer from malnutrition and famine and have higher rates of disease and shorter life spans.

There is abundant evidence of the deleterious nutritional effects of eating foods grown in soils that have low levels of certain naturally occurring trace elements.^[4,5,16,17] Low levels of dietary iodine can lead to an array of iodine deficiency disorders, including goiter, hyperthyroidism, and childhood physical and mental retardation. Soils that are low in iodine are found throughout the world. In response to the nearly global effects of iodine deficiency disorders, various international groups, including United Nations Children's Fund and World Health Organization, created the International Council of Control of Iodine Deficiency Disorders (ICCIDD) in 1985, with the goal of eradicating iodine deficiency. The most effective result of ICCIDD has been the effort toward the universal iodization of salt. Iodized salt is available in most of the world, and as a result, iodine deficiency disorders have dropped dramatically.^[16] Selenium is another globally distributed soil trace element that has been found to be essential to human metabolism and health. Selenium deficiency has been linked to a number of diseases and disorders, including viral infections, immune system disorders, cancer, cardiovascular diseases, low sperm motility,

higher rates of miscarriage, and mood disorders, such as depression. Dietary selenium is crucially involved with selenocysteine, the 21st amino acid, which promotes the production of selenoproteins. Most of the selenoproteins that have been identified are involved in a number of important metabolic functions, revealing a direct relationship between the levels of dietary selenium and human health.^[17] Other such relationships between soil trace elements, the nutritional composition of food processed from plants grown on that soil, and human health have been well documented.^[4]

It has been suggested that organically grown food may be beneficial to human health.^[18–20] Public interest in organic agriculture, whereby foods are produced without the use of synthetic pesticides and herbicides, soluble fertilizers, non-therapeutic antibiotics, and growth hormones, has popularized the concept of healthy food derived from healthy soil. Organic foods are often selected by consumers for their lack of these agricultural inputs and for the subsequent reduction in the negative impacts of farming on the ecology of soil and water. However, there is a lack of credible research to support anecdotal claims that organic methods produce foods nutritionally “healthier” than foods grown using conventional agricultural practices. Further work describing nutritional benefits of organic vs. conventionally produced food has been criticized for the general lack of definitive data.^[3,21,22] Regardless of this, the further rise in public interest in organic foods reflects an increased cultural concern for a healthier lifestyle that is associated with healthier foods. Although more organic products are available in the marketplace, they remain relatively rare, specialized, and expensive when compared with the majority of food products, which are produced and processed conventionally.^[3]

Epigenetics is the study of the impact of exposures to environmental agents on the expression of genes and how this relationship influences the development and, in particular, the emergence of measures of health and disease with aging.^[23] The field of nutrigenomics studies the impact of nutrients on gene expression.^[24] Research involving the role of folic acid in the diet suggests an epigenetic relationship between nutrition of the mother and the functions of the offspring's genetic make up during development, which can manifest phenotypically later in life.^[25] The direct exposures to soil constituents and components of food in determining gene function are an exciting field of research, which provide the possibility for a better understanding of the biological mechanisms that are responsible for the observed associations between soil biology and human phenotypes of health and disease.

THE IMPACT OF CULTURAL PRACTICES ON SOIL, HUMAN HEALTH, AND THE SURVIVAL OF CIVILIZATIONS

The relationships between soil, food, and human health are the first derivatives of local and national agricultural

practices and policies that are influenced by the economic and cultural norms of society.^[26,27] The management of agricultural soil is a proven force in the relationship between soil and human well-being. Modern agriculture policies and market infrastructures encourage large monocultures of crops such as corn and soybeans, which are grown as inputs for processed foods and biofuel production. When farmed using conventional methods, monocultures can contribute to loss of soil through erosion and degradation of soil health, which can have a major impact of the sustainability of agriculture, with subsequent effects on the nutritional health of the societies involved.

High-tech monoculture farming operations in North America and Asia have contributed to increased rates of soil degradation. Lal suggests that cultures that develop a spiritual relationship with the soil, whereby the soil is understood to be a living entity worthy of respect, encourage agricultural methods that functionally resist the degrading effects of technology on the soil. By integrating social and spiritual beliefs about soil into agricultural practice, such cultures are more likely to persist.^[5,28] The benefits of traditional agricultural methods, such as crop rotation, low- and no-tillage methods, composting of organic materials, the use of green and brown manures, and the use of other traditional methods that foster a sustainable agriculture, are well documented.^[15,20] When compared with the conventional methods alone, these traditional agricultural methods decrease tillage and fuel, fertilizer, and pesticide inputs and have been shown to keep production levels high, decrease soil loss, and improve soil health.^[7,15,20]

A further increase in public discussion of the environmental and health benefits of organically produced foods has increased cultural concern for the potential non-sustainability of conventional farming methods that degrade soil. Modern organic methods of production and processing follow ethical guidelines that include sustainable soil management, reduced fertilizer, herbicide and pesticide inputs, and an acknowledgment of complex interrelationships between the soil, the food commodities being produced, the humans that produce and consume the food, and the environment.^[15] In the United States, the number of U.S. Department of Agriculture–certified organic acres of agricultural land has increased from just less than 1 million acres in 1992 to more than 4 million acres in 2005. This represents less than 1% of the total acreage dedicated to agriculture in the United States.^[29,30] Newer agricultural practices that blend aspects of both conventional and traditional methods have been shown to keep production levels high while decreasing fuel, fertilizer, herbicide, and pesticide inputs and decreasing soil degradation (when compared with conventional methods alone).^[3,15,20] Kirschenmann summarizes such a new sustainable food system “that meets the challenges of our post-industrial era ... that is more resilient, more secure, more energy efficient, and provides us with healthier food and more pleasurable eating than the industrial food system

currently offers us.”^[31] Hopefully, the increased public awareness of sustainable and organic methods of food production and the integration of sustainable and conventional practices has started a trend toward better soil management.

THE FUTURE

Regardless of our knowledge of food production and awareness of the relationship between food and health, there is a void of academic work directly connecting the ecology and biology of the soil with human health. The role of soil ecology in determining human health is the least understood component of the agricultural process. Our methods for understanding soil ecology give only an incomplete picture of the soil-microbe community.^[32,33] In particular, the relationship between soil-microbe ecosystems and the observed health benefits of organically grown food has to be fully appreciated and is emerging from research in progress.^[21,22,26,34–36] Insights offered by research on the epigenetic effects of direct and indirect exposures to soil constituents and constituents of food hold promise for establishing the role of the human genome in connecting soil with human health.

CONCLUSION

Finally, it is relevant to call attention to that the historic decline of past civilizations has been linked directly to their failure to properly manage their agricultural soils. The past 10 millennia have seen the rise and fall of many human civilizations, and one recurring theme through most of the failures has been the depletion of the agricultural soils due to poor soil management.^[28,37] The parasitic, forceful, and unsustainable occupation of soils by humans and human activities reveals modern humans as a pathogenic agent, negatively “afflicting” the soil with unsustainable agricultural methods until both the host (soil) and parasite (humans) are exhausted. The future well-being of our species depends, in no small part, on how we manage the production of food while remaining mindful of the relationship between soil and human health.

ACKNOWLEDGMENT

The authors thank Heidi Ciofani for her editorial help and Fred Kirschenmann for critical comments on early drafts of this entry.

REFERENCES

1. Howard, A. *The Soil and Health*; University Press of Kentucky: Lexington, 2006.
2. Sing, C.F.; Stengard, J.H.; Kardia, S.L.R. Dynamic relationships between the genome and exposures to environments as

- causes of common human diseases. In *Nutrigenetics and Nutrigenomics, World Review Nutrition and Dietetics*; Simopoulos, A.P., Ordovas, J.M., Eds.; Karger: Basel, 2004; Vol. 93, 77–91.
3. Trewavas, A. Urban myths of organic farming. *Nature* **2001**, *410* (6827), 409–410.
 4. Oliver, M.A. Soil and human health: A review. *Eur. J. Soil Sci.* **1997**, *48* (4), 573–592.
 5. Abrahams, P.W. Soils: Their implications to human health. *Sci. Total Environ.* **2002**, *291* (1–3), 1–32.
 6. Johns, T. Detoxification function of geophagy and domestication of the potato. *J. Chem. Ecol.* **1986**, *12* (3), 635–646.
 7. Calabrese, E.J.; Stanek, E.J.; James, R.C.; Roberts, S.M. Soil ingestion: A concern for acute toxicity in children. *Envir. Health Perspect.* **1997**, *105* (12), 1354–1358.
 8. Bethony, J.; Brooker, S.; Albonico, M.; Geiger, S.M.; Loukas, A.; Diemert, D.; Hoetz, P.J. Soil-transmitted helminth infections: Ascariasis, trichuriasis, and hookworm. *Lancet* **2006**, *367*, 1521–1532.
 9. Hawley, J.K. Assessment of health risk from exposure to contaminated soil. *Risk Anal.* **1985**, *5* (4), 289–302.
 10. von hertzen, L.; Haahtela, T. Disconnection of man and the soil: Reason for the asthma and atopy epidemic? *J. Allergy Clin. Immun.* **2006**, *117* (2), 334–344.
 11. Mesinga, T.J.; Speijers, G.J.A.; Meulenbelt, J. Health implications of exposure to environmental nitrogenous compounds. *Toxicol. Rev.* **2003**, *22* (1), 41–51.
 12. Bray, G.A.; Nielsen, S.J.; Popkin, B.M. Consumption of high-fructose corn syrup in beverages may play a role in the epidemic of obesity. *Am. J. Clin. Nutr.* **2004**, *79* (4), 537–543.
 13. Getz, G.S.; Reardon, C.A. Nutrition and cardiovascular disease. *Arterioscl. Throm. Vas.* **2007**, *27*, 2499–2506.
 14. Siderer, Y.; Maquet, A.; Anklam, E. Need for research to support consumer confidence in the growing organic food market. *Trends Food Sci. Tech.* **2005**, *16* (8), 332–343.
 15. Rigby, D.; Cáceres, D. Organic farming and the sustainability of agricultural systems. *Agric. Syst.* **2001**, *68*, 21–40.
 16. Delange, F.; de Benoist, B.; Pretell, E.; Dunn, J.T. Iodine deficiency in the world: Where do we stand at the turn of the century? *Thyroid* **2001**, *11* (5), 437–447.
 17. Rayman, M.P. The importance of selenium to human health. *Lancet* **2000**, *356*, 233–241.
 18. Magnusson, M.K.; Arvola, A.; Hursti, U.-K.K.; Åberg, L.; Sjöden, P.-O. Choice of organic foods is related to perceived consequences for human health and to environmentally friendly behavior. *Appetite* **2003**, *40* (2), 109–117.
 19. Worthington, V. Nutritional quality of organic versus conventional fruits, vegetables, and grains. *J. Altern. Complem. Med.* **2001**, *7* (2), 161–173.
 20. Tilman, D. The greening of the green revolution. *Nature* **1998**, *396*, 211–212.
 21. Worthington, V. Effect of agricultural methods on nutritional quality: A comparison of organic with conventional crops. *Altern. Ther. Health Med.* **1998**, *4* (1), 58–69.
 22. Mathews, R.A. Response to Worthington on nutritional quality of organic versus conventional fruits, vegetables, and grains. *J. Altern. Complem. Med.* **2002**, *8* (6), 695–696.
 23. Hatchwell, E.; Greal, J.M. The potential role of epigenomic dysregulation in complex human disease. *Trends Genet.* **2007**, *23* (11), 588–595.
 24. Simopoulos, A.P. Preface. In *Nutrigenetics and Nutrigenomics, World Review of Nutrition and Dietetics*; Simopoulos, A.P., Ordovas, J.M., Eds.; Karger: Basel, 2004; Vol. 93, VII–XII.
 25. Jirtle, R.L.; Skinner, M.K. Environmental epigenomics and disease susceptibility. *Nat. Rev. Genet.* **2007**, *8* (4), 253–262.
 26. Waltner-Toews, D.; Lang, T. A new conceptual base for food and agricultural policy: The emerging model of links between agriculture, food, health, environment and society. *Global Chang. Hum. Health* **2000**, *1* (2), 116–130.
 27. Faergeman, O. *Coronary Artery Disease: Genes, Drugs, and the Agricultural Connection*; Elsevier: Amsterdam, 2003.
 28. Diamond, J. *Collapse*; Viking Books (Penguin Group): New York, 2005.
 29. Greene, C. U.S organic farm sector continues to expand. *Amber Waves* **2006**, *4* (2), 13.
 30. Greene, C. *U.S. Organic Agriculture. Agricultural Indicators and Environmental Indicators*; EIB-19, Economic Research Service/USDA: Washington, 2006; Chapter 4.9.
 31. Kirschenmann, F. Food as relationship. *J. Hunger Env. Nutr.* **2008**, *3* (2–3), 106–121.
 32. Harris, J.A. Measurements of the soil microbial community for estimating the success of restoration. *Eur. J. Soil Sci.* **2003**, *54* (4), 801–808.
 33. Hill, G.T.; Mitkowski, N.A.; Aldrich-Wolfe, L.; Emele, L.R.; Jurkonie, D.D.; Ficke, A.; Maldonado-Ramirez, S.; Lynch, S.T.; Nelson, E.B. Methods for assessing the composition and diversity of soil microbial communities. *Appl. Soil Ecol.* **2000**, *15* (1), 25–36.
 34. Raskin, I.; Ribnicky, D.M.; Komarnytsky, S.; Ilic, N.; Poulev, A.; Borisjuk, N.; Brinker, A.; Moreno, D.A.; Ripoll, C.; Yakoby, N.; O'Neal, J.M.; Cornwell, T.; Pastor, I.; Fridlender, B. Plants and human health in the twenty-first century. *Trends Biotechnol.* **2002**, *20* (12), 522–531.
 35. Young, I.M.; Crawford, J.W. Interactions and self-organization in the soil-microbe complex. *Science* **2004**, *304* (5677), 1634–1637.
 36. Allison, V.J.; Yermakov, Z.; Miller, R.M.; Jastrow, J.D.; Matamala, R. Assessing soil microbial community composition across landscapes: Do surface soils reveal patterns? *Soil Sci. Soc. Am. J.* **2007**, *71* (3), 730–734.
 37. Lowdermilk, W.C. *Conquest of the Land through Seven Thousand Years*, S.C.S. MP-32; USDA Soil Conservation Service: Washington, 1948.

Human Society and Soil

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Soil for modern societies is even more important than ever before. The increase in importance lies in the fact that soil resources of the world are: 1) finite; 2) non-renewable over the human time frame; 3) liable to misuse and mismanagement; 4) unevenly distributed among biomes and ecoregions and 5) highly variable over time and space. Area under prime soils, with few or no limitations to intensive use for agricultural production, is rapidly shrinking because of soil degradation, conversion to non-agricultural uses, and urbanization. This entry describes the dependence of human society on soil in both ancient and modern eras, and the need for maintaining and enhancing soil quality for achieving food security and improving environment quality.

INTRODUCTION

Humans are the most highly evolved animals who, in addition to flesh and skeleton, have intellectual capacity to perceive, rationalize, and develop technological innovations to overcome nature's constraints to meet their needs. There is a strong interdependence between soil and humans. Soil is the 3-D layer of earth's crust that, through numerous biophysical and chemical interactive processes, is capable of supporting plant and animal life and moderating environment (air and water) quality. Rather than a mere combination of inorganic and organic substances to varying proportions represented among different soil types, soil is a living entity. It is full of life, highly dynamic, and ceases to produce biomass and purify water and air when life in it ceases to exist due to natural or human-induced causes. Thus, there exists a strong interrelationship between soil and humans. Ever since the evolution of humans on earth, the quality of life and number of humans in any ecoregion or biome have depended on the quality and resilience characteristics of the soils of that region. While human societies rose and vanished, along with the quality of soil that supported them, they also changed soil quality in a manner that no other species of the animal kingdom has ever done throughout earth's history. The fate of humans and the soil that supports its society is so closely intertwined that it is difficult to understand the nature of one without knowing the properties and dynamics of the other. Humans have manipulated soils to meet the nutritional needs of their body and the spiritual needs of their soul. In that process, they have left an indelible mark on soil quality. As Lowdermilk^[1] wrote, "Individual nations and civilizations write their records on the land—a record that is easy to read by those who

understand the simple language of the land." The extent and severity of change in soil quality due to human-induced perturbation depend on the demands based on human needs and expectations.

Soil is the most basic of all natural resources, because most processes governing terrestrial life depend on it. Important among these processes are: 1) biomass productivity, 2) purification of water, 3) detoxification of pollutants, 4) recycling of elements, and 5) resilience and restoration of ecosystems. Hugh Hammond Bennett, father of the soil conservation movement in the United States, wrote "Out of the long list of nature's gifts to man, none is perhaps so utterly essential to human life as soil." The human history is intimately linked to soil and its quality. That is why "soil" or "Mother Earth" has been held sacred and worshipped by many ancient cultures. Ancient Hindus and Buddhists worshipped earth as Dharra, Vasundharra, Buhumi, and Prithvi as sacred. The word Dharra has the same roots as the Latin word Terra. The ancient Greeks worshipped the earth as Gaea or the Mother Earth. Water, another important constituent of the land and essential for life, is also held sacred in many cultures. The Koran, sacred to the followers of Islam, states that "with water we have made all living things." Similar beliefs exist in Hebrew, Christian, and other cultures.^[2]

SOIL AND ANCIENT SOCIETIES

The history of human civilization is based on the relation between humans and soil, humans' influence on soil quality, and humans' perception of how to manage soil resources. Hyams^[3] outlined three types of interactions

between soil and ancient societies, depending on the renewability of the soil and the managerial skills of the evolving human society.

Soil Parasitism

Parasitism implies the life of an organism through exploitation (and in extreme cases, eventual debilitation) of its host. Humans exist as parasites on soils because of the dawn of civilization and evolution of structured society. Some alluvial soils, due to their natural rejuvenation and renewability, have supported human societies for millennia. Examples of such constantly renewed alluvial soils include those in the valleys of the Nile, the Euphrates–Tigris, the Indus, the Yangtze, and the Hwang-Ho. Similar to alluvium, some loess (wind-borne) soils also have the renewability and capacity to withstand human exploitation for a long time. In arid and semiarid regions where loess is formed, soils must have adequate supply of good quality water to support and sustain human society for an extended period of time. Because of their natural renewability, caused by annual floods, societies living off the alluvial soils did not have to discover ameliorative tools (e.g., plow, manure, and other amendments).

Soil Occupation or Militaries

In contrast to the deltaic soils of the Nile, those of the Euphrates–Tigris river system were easily eroded and degraded because of undulating terrain. For example, the soils of ancient Babylon were desolated by misuse and abuse over an extended period of time. Sustaining production on these soils implied input of energy, nutrients, and water. Therefore, it was in this region where innovations such as the plow, irrigation, and manuring were introduced to alleviate soil-related constraints to crop production. The “ard” or plow evolved from a planting stick about 10,000–12,000 years ago,^[4] probably in the Middle East.^[5] Yet irrigation and soil erosion cannot be taken for granted. After all, it was the neglect of ancient canals, which created waterlogging and salinity on the one hand, and erosion in the catchment, which led to siltation of waterways and reservoirs on the other. Agriculture that was thriving about 10,000 B.C. in Mesopotamia, present-day Iraq, was converted to desert and shifting sand dunes by deforestation and erosion of the surrounding hills.^[1,6] There is a lot of similarity in ecosystems of Euphrates–Tigris and Indus valleys. Salinity, erosion (by water and wind), and siltation also played havoc with human-occupied soils of the Indus. Widespread deforestation for wood for brick-making in the Indus Valley^[3] might have caused severe problems of erosion and degradation. The collapse of the 1700-year old Mayan civilization in Guatemala around 900 A.D. is also attributed to accelerated soil erosion.^[7] Increase in population caused soil depletion, and the Mayan culture paid the ultimate price.^[8] Military

occupation of any form, on a soil or society, cannot survive/sustain for an extended period.

Humans as Soil Pathogens

There are many examples where humans have been the “afflicting” and the soil, the “afflicted” species in functions rather than in form. A disease is caused by imbalance between the activities of afflicting (parasite) over that of the afflicted (host) species, leading to eventual collapse of both host and parasite. The extinction of many ancient societies (e.g., Mesopotamia in Iraq, ancient kingdoms of Sardis and Lydia in Turkey, the Mayan kingdom in Guatemala, etc.) are but a few examples of the severe cases of humans afflicting soil, leading to an eventual collapse of both. In contrast to the vanquished, ancient societies that realized that soil is to be respected and worshipped as a living entity have thrived and flourished.

THE VANISHING SOIL AND MODERN SOCIETY

The per capita arable land area is rapidly decreasing due to an ever-increasing human population. In 1995, the per capita land area was 0.23 ha worldwide, 0.23 ha in Africa, 0.20 ha in Latin America, 0.89 ha in North America, 0.12 ha in Asia, 0.17 ha in Europe, and 1.8 ha in Oceania. By 2050, even if there is no soil degradation and conversion to non-agricultural uses, the per capita land area will change to 0.14 ha worldwide, 0.08 ha in Africa, 0.11 ha in Latin America, 0.69 ha in North America, 0.07 ha in Asia, 0.19 ha in Europe, and 1.1 ha in Oceania. Similar to the arable land area, world grain harvested area per person declined from 0.23 ha in 1950 to 0.11 ha in 1998^[9] and will decrease to 0.07 ha by 2050.^[10] These vanishing soil resources imply that the basic necessities of life (e.g., food, feed, fuel, and fiber) will have to be met from low per capita land area and likely with adverse moisture/rainfall and temperature regimes due to the projected greenhouse effect. The cereal production of 2081 million Mg in 1998 will have to be increased by 62% in 2025 and 121% in 2050 to meet the food demands.^[11] The percentage increase in cereal production will be much greater in the developing countries of Asia and Africa.^[11]

The importance of soil resources on global food security cannot be overemphasized. Despite impressive gains in food production brought about by the Green Revolution technology, the problem of food insecurity persists in several less-endowed and densely populated regions of the world. The food insecure population of the world was 960, 938, 831, and 791 millions in 1970, 1980, 1990, and 1996, respectively, and is projected to be 680 million in 2010.^[12,13] There is also a widespread problem of malnutrition,^[14] which is accentuated by food grown on nutrient-deficient and degraded soils.

The threat of pending famine was voiced by Thomas Malthus in 1798 when the world population was merely 940 million. Since then, the world population has multiplied geometrically, as Malthus predicted, to 1.26, 1.65, 2.52, and 6.06 billions in 1850, 1900, 1950, and 2000, respectively.^[15–17] In 1000 years, from year 1000 to 2000, the human race has multiplied 20-fold and is increasing at the rate of 73 million or 1.3%/yr. The population is projected to be 7.5 billion by 2020 and 9.4 billion by 2050. The welfare of the population depends on soil quality and resilience under an intensive agricultural use. Several global challenges facing human society at the onset of the 21st century (Table 1) will have to be met through sustainable use of the world's soil resources.

SOIL–HUMAN SYMBIOSIS

The incidence of undernourishment in developing countries will persist during the first half of the 21st century. The number of undernourished people in developing countries will be 610 million by 2015 and 443 million by 2030.^[18] The challenge lies in improving the quality of life while diminishing the ecological foot prints of natural resources. Thus, soil resources must be managed, improved, and restored, if the challenges of the 21st century (Table 1) are to be successfully met. The large human population (6 billion and 9.4 billion by 2050) cannot survive by parasitism, militaries, or pathogenic strategies. Humans have to live in harmony or symbiosis with soil. Humans, through their inquisitive nature and innovative instincts, have the capacity to meet the challenges of the growing population.

Table 1 Challenges posed by human society at the threshold of the 21st century that world soils must meet.

Challenge	Status	Rate of change
World population	6 billion	+1.3%/yr
Food insecure population	790 million	–1.0%/yr
Soil degradation	1966 Mha	+5–10 Mha/yr
Desertification	1016 Mha	+5.8 Mha/yr
Global irrigated area per person	0.045 ha	–1.3%/yr
World grain harvested area per person	0.11 ha	–0.55%/yr
Forested area per capita	0.59 ha	–0.78%/yr
Atmospheric concentration of GHGs		
1) CO ₂	370 ppmv	+0.5%/yr
2) CH ₄	1.74 ppmv	+0.75%/yr
3) N ₂ O	311 ppbv	+0.25%/yr

Source: From Lal^[9] and Brown, Gardner, et al.^[16]

As Stevenson wrote, “Nature is neutral. Man has wrested from nature the power to make the world a desert or to make the desert bloom.” The power to make the desert bloom lies in the realization of maintaining and enhancing soil quality. The concept of soil quality in one form or another has been used by agricultural philosophers and writers for some 2500 years. An agriculturist of Moorish Spain, *Ibn-Al-Awam* wrote several volumes during the 12th century on agricultural issues. The book *Kitab al-Felhah* or *Book of Agriculture* was translated into Spanish in 1802 and was brought to public attention in the *Encyclopedia of Islam* (1760–1777). In this book, the author writes, “The first step in the science of the agriculture is the recognition of soils and of how to distinguish that which is of good quality and that which is of inferior quality” (vol. 1, p. 23).^[1] The concept of soil quality, well developed and understood for its management and enhancement, has to be used in a “symbiotic” manner for meeting the challenge of the modern era. These challenges have to be met through: 1) restoration of degraded soils, 2) enhancement of soil organic matter content and soil quality, 3) intensification of agriculture on prime soils, 4) minimizing risks of erosion and soil degradation by other processes, and 5) enhancing efficiency of all inputs (water, fertilizers, and energy). Humans have to learn to live in symbiosis with soil for their survival.

CONCLUSION

In the old Roman Empire, all roads led to Rome. In the long-term survival of human society, all roads lead back to the soil.^[13] Human society must respect soil, the media that feeds the world and purifies the environment. World soils have the capacity to feed the population, provided that soils are used, improved, and restored. Judicious management of soil resources is a solution to the environmental issues (water quality and the greenhouse effect) and also to achieving global food security.

REFERENCES

1. Lowdermilk, W.C. *Conquest of the Land Through Seven Thousand Years*; Agric. Inform. Bull. 99; SCS: Washington, 1953.
2. Hillel, D.J. *Out of the Earth: Civilization and the Life of the Soil*; The Free Press: New York, 1991; 321 pp.
3. Hyams, E. *Soil and Civilization*; Thames and Hudson: London, 1952; 312 pp.
4. White, K.D. *Agricultural Implements of the Roman World*; Cambridge Univ. Press: Cambridge, 1967.
5. Lal, R. Historical development of no-till farming. In *Sustainable Agriculture and the International Rice-Wheat System*; Lal, R., Hobbs, P., Uphoff, N., Hansen, D.O., Eds.; Marcel Dekker: New York, 2204; 55–82.

6. Beasley, R.P. *Erosion and Sediment Pollution Control*, 1st Ed.; Iowa State University Press: Ames, 1972.
7. Olson, G.W. Archaeology: Lessons on future soil use. *J. Soil Water Conserv.* **1981**, *36*, 261–264.
8. Hardin, G. The tragedy of the commons. *Science* **1968**, *162*, 1243–1248.
9. Lal, R. *Controlling Greenhouse Gases and Feeding the Globe Through Soil Management*; University Distinguished Lecture, Feb 17; The Ohio State University: Columbus, 2000.
10. Brown, L.R. *Outgrowing the Earth: The Food Security Challenge in an Age of Falling Water Tables and Rising Temperatures*; W. W. Norton and Company: New York, 2004; 239 pp.
11. Wild, A. Soils, land and food. In *Managing the Land During the 21st Century*; Cambridge University Press: Cambridge, 2003; 246 pp.
12. FAO. The state of food and agriculture. In *Thirtieth Session of the FAO Conference*, Rome, Italy, Nov 12–23, 1999.
13. FAO. *Assessment of the World Food Security Situation*. In Report #CFS: 99/2 of the 25th Session of the Committee on World Food Security, Rome, Italy, May 31–Jun 2, 1999.
14. Mesham, A.R.; Chatterjee, M. *Wasting Away: The Crisis of Malnutrition in India*; The World Bank: Washington, 1999; 78 pp.
15. Anon. Like herrings in a barrel. *The Economist* Dec. 23, **1999**, 13–14.
16. Brown, L.R.; Gardner, G.; Halweil, B. *Beyond Malthus: Sixteen Dimensions of the Population Problem*; Worldwatch Paper 143; Worldwatch Institute: Washington, 1998; 87 pp.
17. Hambridge, G. Soils and men: A summary. In *Soils and Men*; Year book of Agriculture; USDA: Washington, 1938; 1–44.
18. Bruinsma, J. *World Agriculture Towards 2015/2030. An FAO Perspective*. Earthscan/FAO: Rome, 2003; 432 pp.

Human Society and the Planet: Soil as a Heritage

Fred P. Miller

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Both nature and humans have left (and are still leaving) imprints on the earth's soil resources. Soil is the earth's equivalent to anatomical derma—a planetary vellum with the “signature” of pedogenic processes, such as climate, acting upon a variety of soil parent materials (e.g., sedimentary, igneous, or metamorphic bedrock; alluvium; glacial deposits; wind-blown silts or loess; and volcanic ash and lava). The morphology and character of the soils originating from these interactions are further modified by the topographic configuration of landscapes, the flora and fauna adapted to the resulting ecological niches, and the duration of these active processes. The spectrum of soils and ecosystems born of these pedogenic and ecological processes constitutes the natural resource heritage from which humankind and its civilizations are sustained. Agriculturally productive soils, occupying only a limited portion of the planetary soil resource pool, were precursors for the genesis of civilization. They are primary requisites for generating the necessary biomass to sustain more than 6 billion humans occupying this planet, with another 2 to 3+ billion projected to be here in the near future.

INTRODUCTION

The early Latin derivation of *Homo* (Latin for man) was *hemo*, meaning the earthly one; akin to the Latin word for earth or soil, namely, *humas*, and the Latin word for humans, namely, *humanus*.^[1] The history of humankind, and our ancestral *Homo* species, is the history of our relationship to the earth and its environment.^[2] Archeological evidence in many parts of the world reveals our penchant to alter the environment,^[3–5] be it to our boon or bane.^[6] Soil is a major component of the natural resource trinity (soil–sun–climate) from which humankind's sustenance is rooted. Soil is the common earthen parchment upon which humanity's cultural signatures are imprinted. Speech, tools, and fire (allowing for clustered habitation in inclement climates) formed the tripod of culture for Paleolithic man and his later counterparts.^[6] The sophistication and capability of modern tools, the development of multiple energy sources, and the increase in population have resulted in the capacity for modern civilization to alter the environment in ways, and at scales, unprecedented in human history. As early as the mid-19th century, human activity was recognized as a significant force in altering the environment.^[6–10] Even at the turn of the 20th century, geologists were calling humans the dominant geological force of the planet.^[9] Still, progress was measured in terms of our increasing control over nature, marching through the stages of our cultural evolution as if the earth was simply the stage upon which this human drama was acted out.^[9]

The impact of progressive stages of civilization on the environment and the tapping of the earth's stocks of natural

resources to sustain these civilizations were largely ignored.^[9] There were those thinkers and writers who recognized that civilization was dependent upon soil and other natural resources for its preservation, and that humankind had failed in its role as steward of the soil and natural resources heritage that sustains it and civilization itself.^[7,8,11]

HISTORICAL PERSPECTIVE

The dawn of civilization was rooted in the soil. As Bradley^[12] noted, “the fabric of human life has been woven on earthen looms. It everywhere smells of the clay.” An old saying, attributed to the Chinese, states that, “Man—despite his artistic pretensions, his sophistication, and his many accomplishments—owes his existence to a six-inch layer of topsoil and the fact that it rains.”

The domestication of plants and animals, i.e., the beginning of agriculture, occurred about 10,000 years ago^[2,5,13–17] and released the human species from its bondage to a hunting and gathering existence since its origin. Evidence of plant and animal domestication in the early Holocene has been documented on several continents.^[5,15–17] The effect of this profound and transforming revolution in human history was to increase the carrying capacity^[18] of a region and its ecosystem. Archeological evidence within the Fertile Crescent and areas around the Mediterranean suggests that their inhabitants were domesticators of animals and cultivators of the earliest founder cereals, such as einkorn wheat, emmer wheat, and barley, and founder legumes such as lentil, chick pea, pea, and bitter

vetch.^[14] This domestication of nature's provisions allowed the accumulation of surplus food supplies that underwrote major population expansion, the division of labor, which released many from the task of gathering—producing food, and the rise of cities and urban states.^[13,19]

Diamond^[20] posits that the development of agriculture was “the worst mistake in the history of the human race.” Most historians, anthropologists, and others, however, see this agricultural transformation of 10,000 years ago as “the greatest single step forward in the history of mankind”—the most momentous turn in the progress of humankind.^[13,21] In the words of Thomas Hobbes, life before agriculture was “nasty, brutish, and short.”^[21]

An axiom of agricultural geography holds that cultivation-based agriculture is predominantly located on soils derived from relatively young geologic parent materials such as alluvium, glacial deposits, loess, and volcanic ash. Soil provides the medium through which nutrient flows and energy conversions take place. Thus, it is not coincidental that early agriculture, and the civilizations that sprang from it, originated in broad alluvial flood plains and adjacent foothills such as the Tigris and Euphrates rivers (the Fertile Crescent), the Jordan Valley, the Nile River, the Indus River, the major rivers of Asia, and intermontane (alluvial) valleys of the Middle East. Similarly, the other early cultures were nurtured by agriculture rooted in soils derived from loess and other relatively young soil parent materials.^[5,9,16,22] Contemporary agricultural equivalents include the North American Corn Belt (glacial deposits and loess), the Central Valley of California (alluvium), the wheat region of the Northwestern United States (volcanic deposits), the rice cultures of Asia (alluvium), and the Chernozem-Black Soils of the Russian grain belt (glacial and loess materials).

The advent of the agricultural revolution was accompanied by increasing ecological manipulation. The development of the ox-drawn hoe, followed by the plow, occurred approximately 5000–6000 years ago throughout Mesopotamia, Egypt, and China.^[13,23,24] Then came plows of increasing sophistication and improved design, including seeder plows that simultaneously allowed the opened furrow to be planted, which is used in parts of the Middle East.^[13] These developments ratcheted up the capacity to manipulate the environment and commonly resulted in deleterious environmental impacts, with soil erosion being the Achilles heel of cultivation-based agriculture. Hillel^[13] has opined that, contrary to the prophet Isaiah, the plowshare became more destructive than the sword. Similarly, animal herding and overgrazing of hillsides exacerbated the environmental impacts of agriculture.

Another early technological innovation accompanied the advance of cultivation-based agriculture: irrigation. Early societies, cultures, and civilizations that developed in the arid and semiarid regions of the Middle East

needed to manipulate the hydrologic cycle for crop production. The earliest evidence of irrigated farming was found in the Jordan River Valley, within which, lay the ruins of the ancient city Jericho, dating back perhaps eight millennia.^[23] It is not surprising that irrigation was an early achievement, given the juxtaposition of the alluvial soils and the watercourses from which the alluvium was derived.

Canal systems were built throughout much of the Middle East to intercept portions of the adjacent river flows and distribute the water to crop fields. These elaborate irrigation networks relied on gravity flow, although there were gates and other features to accommodate the rise and fall of the rivers' seasonal discharges. Water-lifting devices, such as Archimedes' screw (tambour), and the animal-powered water wheel (sagia) were designed to lift water from irrigation canals and rivers to crop fields.^[25]

Irrigation came to be relied on to support civilizations throughout the history of Mesopotamia and much of the semiarid Middle East. These cultures are characterized as hydraulic civilizations,^[26] riverine,^[27,28] or irrigation-based civilizations.^[25] But civilizations that try to sustain themselves in these rainfall-deficient environments are vulnerable to two soil-related problems: silt and salt. As these civilizations' populations expanded, and agriculture and grazing moved further up-slope into the watersheds supplying the rivers and irrigation systems, accelerated soil erosion began its insidious gnawing away at the soil resources of the uplands. The silts and sediments carried downstream eventually flowed into the irrigation canals and networks, clogging them and reducing their effectiveness. Likewise, the river channels themselves became silt-laden, raising the riverbed, rendering the river unstable and prone to flooding the adjacent fields. Thus, in addition to tending to the agriculture and irrigation systems, intensive diking and levee building were required—the classic example being China's Yellow River.^[24] Removing this silt and diking river courses became labor intensive. Some cultures resorted to enslaving others for labor, capturing them as the spoils of conflict between nation states.

Without drainage systems and intensive water management in these water-deficient environments, irrigation caused water tables to slowly rise, exacerbated by silt clogging river channels causing the riverbed to rise. These rising water tables brought with them salts that eventually wicked to the soil surface, rendering the soils inhospitable to crop production with many soil areas becoming sterile. Coupled with invasions from other peoples, competition for water, reduced agricultural production, and internal conflicts and weaknesses, the collapse of civilizations throughout Mesopotamia and the Middle East reads like an historical casualty list in the “Graveyard of Empires,” including once-mighty Babylon itself.^[24]

The scarred and soil-denuded landscapes of this region and many areas around the Mediterranean attest to the

ravages of soil erosion caused by cropping or overgrazing the hillsides. Many structures and entire cities of these early civilizations are entombed in the sediments and salts unleashed by mismanagement of the land. It is ironic that the soil resources that were the heritage of these civilizations eventually became the materials contributing largely to the collapse and burial of these empires. Indeed, humankind had deeply etched its early history into the soil that became the heritage of subsequent civilizations.

The rocky and barren skeletal remnants of the upland landscapes in these regions stand in stark contrast to the original soil resources and vegetative cover. The lush cedars of Lebanon, once covering more than one half million hectares, were clear-cut for the ships of Phoenicia, for King Solomon's temple, to make way for agriculture, and other uses. Only four small (few hectares) remnant groves remain. Standing on Mount Nebo overlooking the Jordan Valley, the site where Moses once saw the lush land of "milk and honey," one sees a denuded and decimated land, capable of producing only a small fraction of its original potential.^[24,29] But within these degraded ecosystems is a lesson that is still valid. Those soils that were protected from erosion by terraces, and other measures are being cultivated as they had been for two or more millennia.

The Nile Valley has nurtured and sustained more than five uninterrupted millennia of civilizations.^[13,30,31] The duration of the Nile River civilizations and multiple collapses of Mesopotamian civilizations are because of the different soil and water regimes of the two riverine ecosystems. The scourges of siltation and salinization were not as severe along the Nile during its annual pulses as they were in the Tigris–Euphrates plain. Thus, the land of Egypt could remain perennially cultivated and productive, while the land of Mesopotamia suffered degradation.^[13] The building of the Aswan High Dam in the 1960s may be problematic for Egypt's agriculture as water tables are rising, requiring the country to invest in artificial drainage systems^[13] to stem the scourge of salinity to which it was so long immune.

The influence of soil on civilizations was not lost on the Greek or Roman empires. Plato offered the reason that Attica, a region in southeast Greece, in former times could support a soldiery exempt from the toil of farming: its soils, as is proved by the remnants now left, surpassed all others in fertility.^[9] The Greek poet Hesiod and Roman writers such as Virgil, Pliny, Varro, and Columella recognized the human impacts on nature and the importance of soil quality, particularly soil fertility and its conservation, to sustain civilization and its cultures.^[9]

The Book of Genesis in the Old Testament had a profound impact on the formation of conceptions about the relationship of humans to the earth. The commandment was to be fruitful and multiply and take possession of the earth. In much of western culture, humankind, by divine authority, assumed a powerful control over nature that led to

seeing its place in manipulating nature, which had been so wonderfully designed by the Creator.^[9] By the 17th century, the classical notion of a design in nature, the Old Testament ideas, and the new passion for science coalesced and provided the stimulus for the study of nature. The concept of civilization cooperating with nature began to unfold. By the 19th century, especially in the works of Marsh,^[7] it is recognized that humankind's unsteward-like manipulations of nature were upsetting its balance and harmony.^[9]

Population growth, the progressing of civilization, and increased pressure on the environment do not necessarily have to lead to accelerated environmental damage, although most of human history has proven otherwise. As Butzer^[32] noted, we can learn much about the environmental successes and failures through the study of human history and the lessons of our human heritage encoded in the settlement and land use histories.

CONTEMPORARY PERSPECTIVE

More than two centuries ago, Malthus^[33] recognized the decreased "power of the land" resulting from human-induced land degradation. During the 18th century, European powers were vying for dominance, control, and settlement of the North American continent. Settlement of the continent was accompanied by the same exploitive behavior that occurred throughout the history. It was during this time that Marsh^[7] and others^[6,8–10] recognized the significance of human impacts on the environment and the historical arrogance toward nature.^[34–37] White^[34] argued that this attitude is dominant in western traditions stemming from westerners' religious beliefs. But Tuan^[37] argued that the tendency toward environmental degradation and the desire to maximize one's well-being characterize all human existence. White^[34] stated that "the emergence in widespread practice of the Baconian creed that scientific knowledge means technological power over nature can scarcely be dated before about 1850. Its acceptance as a normal pattern of action may mark the greatest event in human history since the invention of agriculture, and perhaps in nonhuman terrestrial history as well."

The early American settlements became the catalyst for mass migrations of Europeans to this new land, migrations that lasted for three centuries. The vast and resource-rich continent that lay before these settlers, and the speed with which it was populated, are unprecedented in history.^[29] As a testimonial to the resources that these settlers inherited, the dense forests were considered an obstacle to agriculture even though they were utilized for fuel and timber. The volume of this timber resource was so large that the center of the commercial lumbering industry did not move beyond western New York until after 1850.^[36] Between 1850 and 1910, American farmers cleared more forest than in the previous 250 years—about 77 million hectares

(190 million acres), equivalent to clearing 35 square kilometers (13.6 square miles) every day for 60 years.^[38]

During these westward migrations, the vastness of the resource base, and the open and cheap land areas available to the west, became the settler's talisman, and worked against a psychology of permanence.^[36] The knowledge that these western lands were still available tended to salve the anxiety of failure and was not conducive to fostering a conservation ethic or to promoting a sense of stability. So plentiful was the North American natural resource heritage that its vastness betrayed its vulnerability.

To illustrate this impermanence syndrome and the environmental impact of its perpetrators, Trimble^[39] cited one wit of the late 1830s, who summarized the situation in the southern Piedmont by noting that "the scratching farmer's cares and anxieties are only relieved by his land soon washing away. As that goes down the rivers, he goes over the mountains." Gray^[40] also pointed out the tendency to deplete land and then migrate west by stating that, "Over the upland soils from Virginia to Texas the wave of migration passed like a devastating scourge. Especially in the rolling piedmont lands, the planting of corn and cotton in hill and drill hastened erosion, leaving the hillsides gullied and bare." An 1853 appraisal of Laurens County, South Carolina^[39] was written in apocalyptic prose, "The destroying angel has visited these once fair forests and limpid streams. The farms, the fields are washed and worn into unsightly gullies and barren slopes—everything everywhere betrays improvident and reckless management."

As northern and southern migrations into the semiarid portion of the new nation continued, the exploitation of the soil resources under the onslaught of these migrations was exacerbated by another human imprint on the soil—the great Dust Bowl era. The vast expanse of prairie encountered by the western migration was unparalleled in the history of human settlements. The European background of the encroaching settlers, and any experience that may have been acquired in the humid east, left unprepared for those who first entered the tallgrass prairie and then the semiarid shortgrass region. The latter ecosystem proved the undoing of those who broke the sod. Tempted by their tradition, lured by the bait of immediate profits, and encouraged by financial and industrial interests,^[36] the settlers' breaking of the sod in this region, coupled with the vagaries of its climate, triggered one of the great ecological disasters of human history. On May 12, 1934, the droughts common to this marginal-cropping region resulted in the first of many dust storms. It was a tragic event in American history that scarred both the land and its people. The impacts of this Dust Bowl were felt from Texas to the Dakotas, damaging some 60–80 million hectares of land.^[41]

The commentary of various witnesses to the erosive demise of many landscapes provides a unique description

of ecological devastation as seen through the eyes of those who experienced it personally. Such accounts echo a common concern, if not resignation, among those who wrote about it. Recorded in these accounts is an intensity of feeling and despair reserved only to those who become personally engulfed in an ecological catastrophe where its shock and impact have not been tempered by either time, healing of the landscape, or by the muffled accounts of historians.

Sears^[36] quotes the owner of a large tract in the short-grass country that had been plowed for wheat in the 1930s, "We're through. It's worse than the papers say. Our fences are buried, the house is hidden to the eaves, and our pasture which was kept from blowing by the grass, has been buried and is worthless now. We see what a mistake it was to plow up all that land, but it's too late to do anything about it." Others were not so articulate about the problem. They simply packed their few belongings and headed west, much as they and their predecessors had done when the water-induced erosion to the east had rendered the land scarred and unyielding. These hardy but tragic people became the human pulp for Steinbeck's *The Grapes of Wrath*. They left behind the scars of a squandered heritage: a damaged ecosystem totaling many tens of millions of hectares.^[42]

In response to this ecological disaster, a new federal agency was formed in 1933, the Soil Erosion Service, which became the Soil Conservation Service in 1935. Soil conservation became a national policy, although its implementation was based on voluntary participation of land owner-operators baited with the incentives of technical and financial assistance. As the necessity to manage ecosystems more holistically became understood, and the concept of ecological sustainability entered the global lexicon, agencies, and institutions took a more comprehensive view of ecosystem management. During the 1990s, the Soil Conservation Service was renamed the Natural Resources Conservation Service, and many federal and state agencies adopted sustainable ecosystem management as their mantra.

CONCLUSION

Is modern society, with all of its technological sophistication, as bound by the limits of its environmental heritage as were previous societies and cultures? Or do science and technology allow humankind to pursue its prerogatives, unhindered from any obligation for stewardship and sustainable management of its natural resources and soil heritage? The answer to these questions is a resounding "no." The laws of nature and thermodynamics preclude any shortcuts to sustaining civilizations without the sustainable management of the ecosystems and ecological processes that undergird our sustenance and well-being. Even though humankind has

opted out of the tyranny of natural selection,^[43] the inextricable interconnectedness of the environment and human well-being requires our understanding of the global environment. The sustainability of the biosphere is now seen to be inseparably bound to issues of economic development, social equity, and international peace and security.^[44] One of the major challenges is to continue to ratchet up the carrying capacity of our soil heritage while sustaining its inherent productive qualities and minimizing leakages of production inputs. Humans have already expanded the carrying capacity of their agricultural ecosystems a 1000-fold^[18] through such innovations as genetic manipulation of plants and animals and the development of synthetic nitrogen. There are only certain areas of our soil heritage that can accommodate the intensity of these ecological manipulations productively and sustainably. Our soil heritage is as vital to modern civilization as it was to our ancestral hunter-gatherer kin. As Smil^[18] has noted, “our ‘postmodern’ civilization would do quite well without computers and software, without ATMs and the WWW but it would disintegrate in a matter of years without synthetic nitrogen fertilizers, and it would collapse in a matter of months without thriving (soil) bacteria. Our first duty is to take care of these true essentials.” Modern civilization is not less dependent upon the soil and ecological heritage than its foraging and pastoral fore-bearers. The causes of the earth’s major environmental problems are rooted in human behavior. Thus, our greatest challenge is to understand the limits and vulnerability of our soil and natural resources heritage and behave accordingly.

REFERENCES

1. Thomas, W.L. Jr., Ed. *The Random House College Dictionary*; Random House: New York, 1988.
2. Hymas, E. *Soil and Civilization*; Harper and Row: New York, 1976.
3. Thomas, W.L. Jr., Ed. *Man's Role in Changing the Face of the Earth*; The University of Chicago Press: Chicago, 1956; 1–1152.
4. Redman, C.L. *Human Impact on Ancient Environments*; Univ. of Arizona Press: Tucson, 1999; 1–255.
5. Roberts, N. *The Holocene: An Environmental History*, 2nd Ed.; Blackwell: Oxford, 1998.
6. Sauer, C.O. The agency of man on the earth. In *Man's Role in Changing the Face of the Earth*; Thomas William, L. Jr., Ed.; The University of Chicago Press: Chicago, 1956; 49–69.
7. Marsh, G.P. *Man and Nature*; Charles Scribner and Co.: New York, 1864; 1–577.
8. Buffon, G.L.L.C. *A Natural History, General and Particular: Containing the History and Theory of the Earth, General History of Man, the Brute Creation, Vegetables, Minerals, etc*; Thomas Kelly and Co.: London, 1866; Vol. II, 151–186. Translated by William Smellie.
9. Glacken, C.J. Changing ideas of the habitable world. In *Man's Role in Changing the Face of the Earth*; Thomas William, L. Jr., Sauer, C.O., Bates, M., Mumford, L., Eds.; The University of Chicago Press: Chicago, 1956; 70–92.
10. Humboldt, A.V. *Aspects of Nature in Different Lands and Different Climates; with Scientific Elucidations*; Lea and Blanchard: Philadelphia, 1849; 475 pp. Translated from the French by Mrs. Sabine.
11. Shaler, N.S. The origin and nature of soils. In *In Twelfth Annual Report of the United States Geological Survey, 1890–1891; Part I: Geology*; U.S. Government Printing Office: Washington, 1891; 213–345.
12. Bradley, J.H. *Autobiography of Earth*; Coward-McCann, Inc.: New York, 1935; 331 pp.
13. Hillel, D.J. *Out of the Earth*; The Free Press, A Division of MacMillan: New York, 1991; 3–174.
14. Lev-Yadun, S.; Gopher, A.; Abbo, S. The cradle of agriculture. *Science* **2000**, *288*, 1602–1603.
15. Vasey, D.E. *An Ecological History of Agriculture, 10,000 B.C.–A.D. 10,000*; Iowa State University Press: Ames, 1992; 23–43.
16. Harlan, J.R. Agricultural origins: centers and non-centers. *Science* **1971**, *174*, 468–474.
17. Brown, K. New trips through the back alleys of agriculture. *Science* **2001**, *292*, 631–633.
18. Smil, V. *Feeding the World: A Challenge for the Twenty-First Century*; The MIT Press: Cambridge, 2000; ix–52.
19. Higham, C.F.W. Prehistoric rice cultivation in Southeast Asia. *Sci. Am.* **1984**, *250*, 138–146.
20. Diamond, J. The worst mistake in the history of the human race. *Discovery* **1987**, *8* (5), 64–66.
21. Russell, E.W.B. *People and the Land Through Time; Linking Ecology and History*; Yale Univ. Press: New Haven, 1997; 111–130.
22. Narr, K.J. Early food-producing populations. In *Man's Role in Changing the Face of the Earth*; Thomas William, L. Jr., Ed.; The University of Chicago Press: Chicago, 1956; 134–151.
23. Ehrlich, P.R.; Ehrlich Ann, H.; Holdren, J.P. *Ecoscience: Population, Resources, Environment*; Freeman: San Francisco, 1977.
24. Lowdermilk, W.C. *Conquest of the Land Through Seven Thousand Years*; Agri. Information Bill. No. 99; Department of Agriculture, Soil Conservation Service: Washington, 1953; 1–30.
25. Hillel, D.J. *Rivers of Eden; The Struggle for Water and the Quest for Peace in the Middle East*; Oxford Univ. Press: New York, 1994; 1–231.
26. Wittfogel, K.A. The hydraulic civilizations. In *Man's Role in Changing the Face of the Earth*; Thomas, W.R., Ed.; University of Chicago Press: Chicago, 1956; 152–164.
27. Simmons, I. *Environmental History: A Concise Introduction*; Blackwell: Oxford, 1993; 2–3.
28. Goudie, A. *The Human Impact on the Natural Environment*, 5th Ed.; The MIT Press: Cambridge, 2000; 152 pp.
29. Carter, V.G.; Dale, T. *Topsoil and Civilization*, Revised Ed.; University of Oklahoma Press: Norman, 1974; 3–62.
30. Butzer, K.W. The human role in environmental history. *Science* **2000**, *287*, 2427–2428.

31. James, T.G.H. *An Introduction to Ancient Egypt*; Farrar Straus Giroux: New York, 1979; 17–96.
32. Butzer, K.W. *Early Hydraulic Civilization in Egypt: A Study in Cultural Ecology*; Univ. of Chicago Press: Chicago, 1976; 1–134.
33. Malthus, T.R. *An Essay on the Principle of Population, as it Affects the Future Improvement of Society*; J. Johnson: London, 1798.
34. White, L. Jr. The historical roots of our ecological crisis. *Science* **1967**, *155*, 1203–1207.
35. Bouillenne, R. Man the destroying biotype. *Science* **1962**, *135*, 706–712.
36. Sears, P.B. *Deserts on the March*, 4th Ed.; Oklahoma University Press: Norman, 1980.
37. Tuan, Y. Man and Nature. *Landscape* **1966**, *15* (3), 30–36.
38. Powell, D.S.; Faulkner, J.L.; Darr, D.R.; Zhu, Z.; MacCleery, D.W. *Forest Resources of the United States, 1992*; General Technical Report Rm 234; U.S. Department of Agriculture, Forest Service: Washington, 1993; 1–20.
39. Trimble, S.W. *Man-Induced Soil Erosion on the Southern Piedmont, 1700–1970*; Soil Conservation Society of America: Ankeny, 1974.
40. Gray, L.C. *History of Agriculture in the Southern United States to 1860*; Publication 430; Carnegie Institution of Washington D.C. Waverly Press, Inc.: Baltimore, 1933.
41. Bennett, H.H. *Soil Conservation*; McGraw-Hill Book Co: New York, 1939.
42. Miller, F.P.; Wayne, D.R.; Donald Meyer, L. Historical perspective of soil erosion in the United States. In *Soil Erosion and Crop Productivity*; Follett Ronald, F., Stewart Bobby, A., Eds.; American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America: Madison, 1985; 23–48.
43. Maddox, J. Positioning the goalposts. *Nature* **2000**, *403*, 139.
44. Jasanoff, S.; Colwell, R.; Dressolhaus, M.S.; Goldman, R.D.; Greenwood, M.R.C.; Huange, A.S.; Lester, W.; Levin, S.A.; Linn, M.C.; Lubchenco, J.; Novacek, M.J.; Roosevelt, A.C.; Taylor, J.E.; Wexler, N. Conversations with the community. *Science* **1997**, *278*, 2066–2067.

Hydraulic Conductivity

Jacob H. Dane

Department of Agronomy and Soils, Auburn University, Auburn, Alabama, U.S.A.

Marc Jalbert

Auburn University, Auburn, Alabama, U.S.A.

Jan W. Hopmans

Department of Land, Air and Water Resources, Hydrology, University of California—Davis, Davis, California, U.S.A.

Abstract

Transport of chemicals and flow of water in soil depend on, among other things, the soil's hydraulic properties. One of these hydraulic properties is the hydraulic conductivity function. In general terms, one could say that the hydraulic conductivity (m/sec) is a measure of the ability of a soil to conduct water.

BACKGROUND INFORMATION

The flow of liquid water through soil is generally viscous and laminar, i.e., non-turbulent. This is because of the small sizes of the pores in which the water movement takes place. Under these conditions, the water flux density q (m/sec, vector quantity) is proportional to the driving force, which is equal to the negative of the hydraulic head gradient ∇H (dimensionless, vector quantity). The magnitude of q is defined as the volume of water V (m³) passing through a cross-sectional area of soil A (m²) during time t (sec). The hydraulic head H (m, scalar) is the potential energy (J) of water on a weight (N) basis. It incorporates the influences of forces such as gravity, pressure, and capillarity. The proportionality factor relating the flux density to the driving force is called the hydraulic conductivity K (m/sec, tensor). In mathematical form, the flux density can be expressed as

$$q = -K \cdot \nabla H \quad (1)$$

K varies from soil to soil and even within the given soil, as it depends on soil properties such as fineness, clay content, solid particle orientation, organic matter content, and water content. Eq. 1 was formulated by the French engineer Darcy^[1] and is often referred to as Darcy's law. As the hydraulic head H (m) has units of length, the hydraulic gradient, ∇H , is dimensionless. Thus, the units for the water flux density and the hydraulic conductivity are the same, i.e., meters per second. Other, less frequently used units for the hydraulic potential result in different units of the hydraulic conductivity. The term hydraulic conductivity should not be confused with permeability (m²) as the former includes

liquid and porous medium properties, while the latter should only reflect porous medium properties.

In general, the proportionality relation, Eq. 1, between the two vectors q and ∇H is tensorial, which means that K needs to be expressed as a matrix. This is necessary when the soil's structure forces water to flow in a direction different from that of the hydraulic head gradient. These soils, called anisotropic, occur frequently in nature. Most soils resulting from sedimentary processes exhibit preferential flow pathways, often perpendicular to the deposition direction that occurred during soil formation. Also, charged clay particles often arrange themselves in sheet-like configurations that create anisotropy. Although important, consideration of the soil's anisotropy involves complications that are beyond the scope of this entry, and the reader is referred to the literature for further information.^[2]

If the soil possesses the same conductive properties in all directions, it is called isotropic. In this case, the water flow takes place parallel to and in the opposite direction of the hydraulic head gradient, and K can be considered a scalar. For a simple application of Darcy's law (Eq. 1) to an isotropic soil, such as a well-packed sand, let us consider a horizontal soil column of constant cross-sectional area subjected to a hydraulic head gradient that is established by two constant-level water reservoirs (Fig. 1). The column is located beneath the water levels, and the soil's entire pore volume is filled with water. The hydraulic head at each end of the column is then simply equal to the water levels H_1 and H_2 in the water reservoirs, as measured with respect to an arbitrary elevation reference. The hydraulic head gradient, uniform throughout the soil column, is equal to the difference between the hydraulic head values H_1 and H_2 divided by the length of the column L , i.e.:

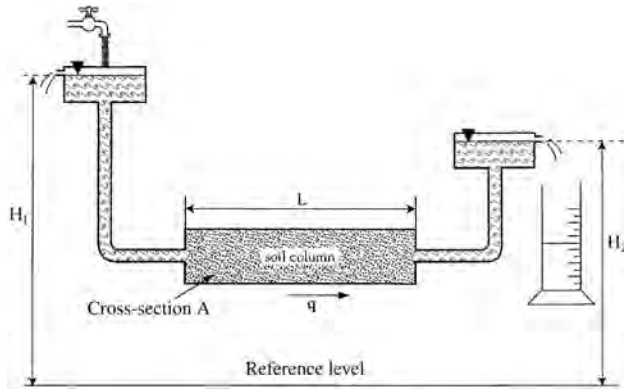


Fig. 1 Schematic diagram for application of Darcy's law.

$$\nabla H = \frac{H_2 - H_1}{L} \quad (2)$$

which is negative. This implies that water flows from reservoir 1 to reservoir 2, i.e., in the direction of decreasing hydraulic head values. For a soil with hydraulic conductivity K , the magnitude of the water flux density (q , scalar) can then be calculated from

$$q = \frac{V}{At} = K \left| \frac{H_2 - H_1}{L} \right| \quad (3)$$

As an example calculation, consider a 1 m long column ($L = 1$ m) with a cross-sectional area $A = 10^{-2}$ m², containing a water-saturated sand with a hydraulic conductivity value $K = 10^{-4}$ m/sec. The water level in the left reservoir is 0.06 m higher than in the right reservoir. This results in a water flux density of 6×10^{-6} m/sec, oriented from left to right, and hence a discharge rate, V/t , of 6×10^{-8} m³/sec, i.e., 60 μ l of water crosses any section of the column in 1 second. The value for the water flux density, which is expressed as a velocity and is sometimes referred to as the Darcy flux or Darcy velocity, should not be mistaken for the average velocity of the water in the pores, known as the seepage or pore water velocity v (m/sec, vector quantity). The difference between the water flux density and the pore water velocity is because of the fact that water occupies only a limited fraction of the soil's total volume. In the case of a water-saturated soil, the soil's volumetric water content, θ , is equal to the porosity n of the soil, i.e., the ratio of the pore volume and the soil's bulk volume. Consequently, water flows only through the fraction nA of a cross-section with area A . The true water velocity, or the pore water velocity, is therefore^[3]

$$v = \frac{q}{n} \quad (4)$$

Assuming a porosity $n = 0.3$, the average pore water velocity or seepage velocity of the water in the column is 2×10^{-5} m/sec or 7.2 cm/hr.

Darcy's law can also be applied to unsaturated soils, i.e., soils containing water and air.^[4] In this case, the soil's hydraulic conductivity is a function of its volumetric water content θ (defined as the volume of water per unit volume of bulk soil), and the seepage velocity is expressed as

$$v = \frac{q}{\theta} \quad (5)$$

Hydraulic conductivity is difficult to measure *in situ*, especially when the influence of θ needs to be quantified. Consequently, a great deal of research has been conducted to estimate K from more easily available soil data. Most of these approaches rely on assuming a certain geometry for the soil pore space, the simplest of them being a bundle of circular capillary tubes, in which the well-defined Hagen–Poiseuille flow occurs. Pioneering work in this area was conducted by Kozeny^[5] and later by Carman,^[6] whose work led to a formula for estimating K of saturated soils. By applying the Kozeny–Carman concepts, it can be shown that saturated hydraulic conductivity is approximately proportional to the square of a mean soil particle diameter. Hence, if the mean particle diameter for a given soil is twice as small as for another soil, its saturated K value will be four times smaller. Notably, clayey soils are often very restrictive to water flow and have saturated K values ranging from 10^{-6} to 10^{-9} m/sec. Sandy soils, on the other hand, offer much less resistance to water flow and have saturated K values in the range of 10^{-4} to 10^{-5} m/sec. Saturated K values (K_{sat}) for various soils are listed based on soil texture in Table 1. Although the Kozeny–Carman theory provides good results for sandy, fully saturated porous media, its application to soil science is somewhat limited, because the predictions for fine-textured or structured soils are usually not very good and most soils contain entrapped air when

Table 1 Saturated hydraulic conductivity values (K_{sat}) for various soils by soil texture.

Soil texture	Siltfraction	Clayfraction	$K_{\text{sat}} \times 10^5$ (m/sec)
Sand	0.05	0.05	1.4 (0.70–7) ^a
Loamy sand	0.10	0.07	1.0 ^b or 1.9 ^c
Sandy loam	0.25	0.10	0.7 (0.4–2) ^a
Silt loam	0.65	0.15	0.07 ^b or 0.14 ^c
Loam	0.40	0.18	0.4 (0.2–0.6) ^a
Sandy clay	0.13	0.27	0.36 ^b or 0.42 ^c
Silty clay loam	0.55	0.34	0.05 ^c or 0.06 ^c
Clay loam	0.35	0.34	0.2 (0.1–0.4) ^a
Silty clay loam	0.47	0.47	0.07 (0.01–0.1) ^a

^aNumbers in parentheses indicate the range.

^bCalculated according to equation 6.11 in Campbell.

^cCalculated according to equation 6.12 in Campbell.

Source: From ^aIsraelsen and Hansen^[7] and ^{b&c}Campbell.^[8]

“saturated.” For this reason, soil scientists sometimes use the term satiated θ , with its corresponding satiated K . Under unsaturated conditions, in the presence of a continuous air phase, capillary forces cause the water to occupy the smaller soil pores, while air invades the larger pores. Because the ability to conduct water decreases with decreasing pore size, the unsaturated hydraulic conductivity decreases rapidly with decreasing θ . Corey^[9] observed that in sedimentary rocks, K is often approximately proportional to the fourth power of θ . A twofold reduction in θ will therefore result in a 16-fold decrease in K . In fact, K values easily decrease five orders of magnitude when θ decreases from its satiated to its residual value. Besides the Brooks–Corey type of empirical relationship between K and θ , more elaborate techniques are often used to estimate unsaturated K values from water retention data. Most of these techniques assume a certain form for the soil water retention curve and use the parameters associated with the water retention relationship to express the hydraulic conductivity as a function of water content or matric head. The most commonly used models are those of Brooks and Corey^[10] and van Genuchten,^[11] using the capillary models of Burdine^[12] and Mualem,^[13] which are extensions of the Kozeny–Carman theory. Based on the pore size distribution factor of their water retention function, λ , Brooks and Corey^[10] derived

$$K_{\text{rel}} = \frac{K}{K_{\text{sat}}} = \left(\frac{h_d}{h_m} \right)^\eta \quad (6)$$

where K_{rel} and K_{sat} are the relative and saturated hydraulic conductivity values, respectively, h_m (m) is the matric head, and h_d (m) is the displacement head, i.e., the value for h_m at which water in a saturated soil is first displaced by air, and

$$\eta = 2 + 3\lambda$$

Based on the dimensionless constant m , appearing in his functional relationship of the water retention curve, van Genuchten^[11] derived

$$K_{\text{rel}} = S_e^{1/2} \left[1 - (1 - S_e^{1/m}) \right]^2 \quad (7)$$

where the effective saturation was

$$S_e = (\theta - \theta_r) / (\theta_s - \theta_r)$$

The subscripts s and r refer to saturation and residual, respectively. It was found that the applicability of these methods, based on retention information, is greatly increased when at least one value for K at a θ different from saturation (mostly satiated) is measured. This is because of the fact that it is difficult to extrapolate the soil’s conductive behavior from the situation near water saturation, where most water is conducted through macropores or fissures, which barely influence the soil’s water retention relationship. In addition, more detailed information regarding the

above and other relationships can be found in Kosugi et al.^[14] Techniques for direct or indirect determinations of saturated and/or unsaturated hydraulic conductivity values are described in Reynolds and Elrick [constant head on soil cores, falling head on soil cores, measurement techniques above the water table, pressure (single-ring) infiltrometer, and constant head well permeameter]; Booltink and Bouma (steady flow/piezometers and suction crust infiltrometer); Youngs (double/multiple ring infiltrometer); Amoozegar (auger hole and piezometer); Scotter and Clothier (one-dimensional infiltration equations, horizontal absorption, 3-D infiltration); Corey (long column); Nimmo et al. (steady-state centrifuge); Arya (wind and hot air methods, plane of zero flux); Vachaud and Dane (instantaneous profile); Parkin and Kachanoski (constant flux vertical TDR); and Hopmans et al. (inverse methods).^[14]

Complementary estimation of hydraulic conductivity can be obtained through statistical, empirical analysis (Nimmo^[14]). When numerous samples of known hydraulic conductivity have been taken over an area presenting some general characteristics, the hydraulic conductivity of a new sample can be roughly estimated from more easily available data. Particle-size data are often chosen^[15,16] because they are routinely obtained by sedimentation analysis, and because hydraulic conductivity is very much influenced by the amounts of clay contained in the soil.

Because hydraulic conductivity is very much dependent on soil structure, i.e., on the occurrence of aggregates and/or cracks, either because of natural causes (soil erosion, shrink/swell, and biological activity) or human factors (tillage and compaction), its value can vary greatly within short distances in the field. Microbial and fungal activity can also have quantitative effects on the hydraulic conductivity by pore clogging, as does temperature by its influence on the water viscosity.^[17] Finally, soil water chemistry influences the hydraulic conductivity of soils containing shrink/swell clay minerals.^[18] If the soil water does not contain sufficient quantities of mainly polyvalent cations, it can cause the clay to swell or even destroy the soil structure. This can greatly reduce the soil’s conductive properties to water. As a result of these chemical, biological, and human factors, the hydraulic conductivity of a soil is likely to vary greatly not only over distance but also over time.

CONCLUSION

One of the governing properties determining flow rates of water is the hydraulic conductivity, the other one being the hydraulic head gradient. For soils saturated with water the hydraulic conductivity, referred to as saturated hydraulic conductivity, reaches its maximum value. The hydraulic conductivity value decreases rapidly with decreasing water content. The relationship between hydraulic conductivity and water content varies from soil to soil and may be even different for the same soil at different locations.

REFERENCES

1. Darcy, H. Détermination des Lois d'Écoulement de l'Eau à Travers le Sable. In *Les Fontaines Publiques de la Ville de Dijon*; Victor Dalmont: Paris, 1856.
2. Bear, J. *Dynamics of Fluids in Porous Media*; Elsevier: New York, 1972; 764 pp.
3. Dupuit, J. *Études Théoriques et Pratiques sur le Mouvement des Eaux dans les Canaux Découverts et à Travers les Terrains Perméables*; Dunod: Paris, 1863.
4. Buckingham, E. *Studies on the Movement of Soil Moisture*; U.S. Dept. Agric. Bur. Soils Bull. 38; U.S. Government Printing Office: Washington, 1907; 29–61.
5. Kozeny, J. Über Kapillare Leitung des Wassers im Boden, Sitzungsber. Akad. Wiss. Wien **1927**, 136, 271–306.
6. Carman, P.C. Fluid flow through a granular bed. Trans. Inst. Chem. Eng. **1937**, 15, 150–166.
7. Israelsen, O.W.; Hansen, V.E. *Irrigation Principles and Practices*; Wiley: New York, 1962.
8. Campbell, G.S. *Soil Physics with Basic*; Elsevier: New York, 1985; 150 pp.
9. Corey, A.T. The interrelation between gas and oil relative permeabilities. Producer's Monthly **1954**, 19 (1), 38–44.
10. Brooks, R.H.; Corey, A.T. Properties of porous media affecting fluid flow. J. Irrig. Drain. Div. Proc. ASCE **1966**, 92, 61–87.
11. van Genuchten, M.T. A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. Soil Sci. Soc. Am. J. **1980**, 44, 892–898.
12. Burdine, N.T. Relative permeability calculations from pore-size distribution data. Trans. AIME **1953**, 198, 71–77.
13. Mualem, Y. A new model for predicting the hydraulic conductivity of unsaturated porous media. Water Resour. Res. **1976**, 12, 513–522.
14. Dane, J.H.; Topp, G.C. *Methods of Soil Analysis, Part 4*, SSSA Book Series No. 5; SSSA: Madison, 2002.
15. Puckett, W.E.; Dane, J.H.; Hajek, B.F. Physical and mineralogical data to determine soil hydraulic properties. Soil Sci. Soc. Am. J. **1985**, 49, 831–836.
16. Dane, J.H.; Puckett, W.E. Field soil hydraulic properties based on physical and mineralogical information. In *Indirect Methods for Estimating the Hydraulic Properties of Unsaturated Soils*; van Genuchten, M.Th., Leij, F.J., Lund, L.J., Eds.; University of California: Riverside, 1992; 389–404.
17. Hopmans, J.W.; Dane, J.H. Temperature dependence of soil hydraulic properties. Soil Sci. Soc. Am. J. **1986**, 50, 4–9.
18. Dane, J.H.; Klute, A. Salt effects on the hydraulic properties of a swelling soil. Soil Sci. Soc. Am. J. **1977**, 41, 1043–1049.

Hydrologic Cycle

Keith R.J. Smettem

*Soil Science and Plant Nutrition, University of Western Australia, Nedlands,
Western Australia, Australia*

Abstract

The role of soils in the hydrological cycle becomes progressively more complex, as time and space scales are reduced. The requirement for detailed soils information is essential for understanding runoff generation and preferential flow processes at local scale. However, for understanding annual or seasonal hydrological trends at large catchment or regional scales, information on soil texture and soil depth can provide sufficient information for estimating the soil water storage capacity.

INTRODUCTION

Climate, soil, vegetation, and topography all play important roles in maintaining the water balance of catchments.^[1] Changes in even one of these factors will affect the hydrological response to rainfall, in terms of both water quality and quantity. At a broad regional scale, Milly^[2] showed that the annual water balance is controlled primarily by climate (as measured by a simple climatic index of dryness, given by the ratio of annual potential evaporation to annual precipitation) and to a lesser extent by the soil storage capacity of the landscape. Reggiani et al.^[3] also show that at catchment scale, total annual evaporation and runoff are controlled by climate (through a dryness index) and soil type. Topographic or landscape factors only become significant when consideration is given to partitioning total runoff into surface and subsurface contributions.^[3]

A REGIONAL SCALE PERSPECTIVE FOR LARGE SPACE AND TIME SCALES

At large space and annual time scales, the role of the soil in the hydrological cycle is, therefore, primarily as a transient store for water. At continental scales, Manabe^[4] simulated the water balance using a simple bucket representing soil water storage and an evaporative threshold. Milly and Dunne^[5] presented similar results showing that at continental and regional scales, simple lumped storage representation of surface hydrology proved to be sufficient to capture the salient features of catchment behavior.

A LOCAL PERSPECTIVE AT SMALLER SPACE AND TIME SCALES

Soils and Runoff Generation Processes

When spatial and/or temporal scales of interest are reduced, consideration must be given to a more detailed

understanding of processes leading to runoff generation and groundwater response. In particular, water flux rates over and through the soil assume increasing importance. Surface runoff may be generated as a “top-down” or “bottom-up” process. The “top-down” process is referred to as infiltration excess runoff and occurs when the intensity of rainfall exceeds the rate of infiltration at the soil surface. The “bottom-up” process is referred to as “saturation excess” and occurs when initially unsaturated soil water stores are filled to overflowing so that additional rainfall simply runs off the saturated area.

Dynamics of Runoff Source Areas

The area of soil contributing to saturation excess runoff from a catchment is temporally dynamic, increasing as a storm progresses and diminishing thereafter. This “variable source area”^[6] for runoff generation is dependent on storm characteristics (duration and volume), soil type, and position in the landscape. Topographic indices of wetness have long been used to identify likely runoff producing areas^[7] because the general tendency of water to flow downhill is amenable to macroscopic conceptualization. Soils tend to remain wetter as the upslope contributing area increases, particularly in areas of topographic convergence,^[8] so it is perhaps not surprising to find that soils close to water-courses tend to saturate first during rainfall. The concept of an upslope contributing area for saturation-excess flow is most readily applied to soils that are shallow, relative to the hillslope scale, under conditions that are wet enough to generate lateral downslope saturated flow through the soil. Lateral downslope saturated flow can of course also be generated over impeding layers in the soil profile.

Predicting Infiltration Excess Runoff

Prediction of infiltration excess runoff requires knowledge of particular soil hydraulic properties that characterize infiltration behavior and the rainfall rate must be specified at

short timesteps (typically minutes). One typical quasi-analytic equation for predicting the time to incipient ponding (the time at which a free water film first appears on the soil surface) is^[9]

$$t_p = M(S^2/K_s R_p) \ln\{R(t_p)/[R(t_p) - K_s]\} \quad (1)$$

where S is the sorptivity ($LT^{-0.5}$), K_s is the saturated hydraulic conductivity (LT^{-1}), $R(t_p)$ is the rainfall rate at incipient ponding (LT^{-1}), R_p is the mean rainfall rate up to ponding, and M is a constant, generally set to 0.55.

The sorptivity^[10] is a characteristic of the soil's capillarity properties and varies with soil texture, structure, and initial water content. The saturated hydraulic conductivity is dependent upon soil texture and can be significantly affected by the degree of soil structure development. Both hydraulic properties can exhibit a high degree of spatial variability, and frequency distributions of saturated hydraulic conductivity are often log-normal.^[11] Rawls et al.^[12] provide summary data on saturated hydraulic conductivity for each of the major United States Department Agency textural classes and point out that soil structure effects can override any differences between texture classes.

Factors Affecting the Time of Runoff Generation

The time at which surface runoff commences after incipient ponding depends on a number of complex interacting factors. Surface microtopography is particularly important because it controls the amount of surface (detention) storage that can occur. Flow concentration into rills can have a major effect on reducing effective wetted areas subject to infiltration after rainfall ceases. If the saturated hydraulic conductivity increases downslope then this in turn can have a major impact on runoff peaks, volumes, and the time to peak runoff.^[13]

Over time scales that are typically of interest in process hydrology (1 year or less), soil texture can be regarded as a fixed property in space and therefore patterns of textural distribution within catchments are amenable to mapping. Maps of soil hydraulic properties can then be derived from this basic information using indirect functional relations.^[14,15] In contrast, soil structure arising from biotic or pedogenic activity can be highly variable temporally, particularly if the land surface is subjected to agronomic manipulation. Agronomic practices that expose soil to the beating action of raindrops often result in the destruction of surface soil structure and the consequent formation of surface skins or seals with low hydraulic conductivity. Physically, within the seal layer, both porosity and pore size distribution are altered. Large conducting pores are destroyed, and the total porosity is reduced. These changes lead to greatly enhanced surface runoff and possible removal of soil by water erosion.

The effects of soil structure on hydrologic response are often incorporated within descriptions of land use and

treatment. Although land use and treatment are not primary physical variables, they are easy to map and have a long history of use as classificatory discriminators of runoff behavior. For example, the SCS curve number procedure^[16] represents the soil-cover complex and a five-day antecedent rainfall that is referenced to dry, normal, or wet conditions. This approach is also to be found in well known hydrological models such as CREAMS.^[17]

Partial Wetting and Other Preferential Flow Processes

Soils that exhibit hydrophobic (nonwetting) behavior can yield considerably more surface runoff under dry antecedent conditions than would be predicted from a consideration of soil texture alone. Non-wetting behavior can also generate "finger flow," which is a form of physical non-equilibrium leading to preferential wetting of only a small fraction of the total soil volume. Preferential flow can also occur through soil macropores that are connected to a source of free water or by funnel-type flow due to layering structures in sandy soils.^[18]

Identification of soils and field conditions that result in preferential flow is the subject of considerable research effort because of the potential contamination risk to groundwater systems posed by the rapid movement of dissolved chemicals through the unsaturated zone.

CONCLUSION

The role of soils in the hydrological cycle becomes progressively more complex, as time and space scales are reduced. The requirement for detailed soils information is essential for understanding runoff generation and preferential flow processes at local scale. However, for understanding annual or seasonal hydrological trends at large catchment or regional scales, information on soil texture and soil depth can provide sufficient information for estimating the soil water storage capacity.

REFERENCES

1. Eagleson, P.S. Climate soil, and vegetation 1. Introduction to water balance dynamics. *Water Resour. Res.* **1978**, *14*, 705–712.
2. Milly, P.C. Climate, inter-seasonal storage of soil water and the annual water balance. *Water Resour. Res.* **1994**, *17*, 19–24.
3. Reggiani, P.; Sivapalan, M.; Hassanizadeh, S.M. Conservation equations governing hillslope responses: Exploring the physical basis of water balance. *Water Resour. Res.* **2000**, *36*, 1845–1863.
4. Manabe, S. Climate and ocean circulation. 1. The atmospheric circulation and the hydrology of the earth's surface. *Mon. Weather Rev.* **1969**, *97*, 739–774.

5. Milly, P.C.; Dunne, K.A. Sensitivity of the global water cycle to the water holding capacity of land. *J. Climate* **1994**, *7*, 506–526.
6. Dunne, T.; Black, R.D. Partial area contributions to storm runoff in a small new England watershed. *Water. Resour. Res.* **1970**, *6*, 1296–1311.
7. Beven, K.J.; Kirkby, M.J. A physically-based variable contributing area model of basin hydrology. *Hydrol. Sci. Bull.* **1979**, *24*, 43–69.
8. O'Loughlin, E.M. Saturation regions in catchments and their relation to soil and topographic properties. *J. Hydrol.* **1981**, *53*, 229–246.
9. White, I.; Sully, M.J.; Melville, M.D. Use and hydrological robustness of time-to-incipient ponding. *Soil Sci. Soc. Am. J.* **1989**, *29*, 1343–1346.
10. Philip, J.R. The theory of infiltration. 1. The infiltration equation and its solution. *Soil Sci.* **1957**, *83*, 345–357.
11. Nielsen, D.R.; Biggar, J.W.; Erh, R.T. Spatial variability of field-measured soil-water properties. *Hilgardia* **1973**, *42*, 215–259.
12. Rawls, W.J.; Brakensiek, D.L.; Logsdon, S.D. Predicting saturated hydraulic conductivity utilizing fractal principles. *Soil Sci. Soc. Am. J.* **1993**, *57*, 1193–1197.
13. Woolhiser, D.A.; Smith, R.E.; Giraldez, J.-V. Effects of spatial variability of saturated hydraulic conductivity on hortonian overland flow. *Water Resour. Res.* **1989**, *32*, 671–678.
14. Rawls, W.J.; Brakensiek, D.L.; Saxton, K.E. Estimation of soil water properties. *Trans. ASAE* **1982**, *25*, 1316–1320, 1328 pp.
15. Smettem, K.R.J.; Oliver, Y.M.; Heng, L.K.; Bristow, K.L.; Ford, E.J. Obtaining soil hydraulic properties for water balance and leaching models from survey data. 1. Water retention. *Aust. J. Agric. Res.* **1999**, *50*, 283–289.
16. USDA. Soil conservation service. In *National Engineering Handbook, Hydrology*; U.S. Government Printing Office: Washington, 1972; 548 pp.
17. Kniesel, W.G. *CREAMS: A Field-Scale Model for Chemicals, Runoff, and Erosion from Agricultural Management Systems*; Conservation Research Report No. 26; USDA, Science and Education Administration: Washington, 1980; 643 pp.
18. Ju, S.-H.; Kung, K.-J.S. Finite element simulation of funnel flow and overall flow property induced by multiple soil layers. *J. Environ. Qual.* **1993**, *22*, 432–442.

Hydropedology

Henry Lin

Department of Crop and Soil Sciences, Pennsylvania State University, University Park, Pennsylvania, U.S.A.

Abstract

Hydropedology is an intertwined branch of soil science and hydrology that embraces a systems and multiscale approach to the study of soil–water–landscape relationships across space and time. This emerging interdisciplinary science seeks to understand the complex and dynamic behavior and the distribution of soil–water interactions in contact with mineral and biological materials in the near-surface terrestrial environment. This entry provides a brief overview of its basic concept, characteristics, fundamental issues, and diverse applications.

INTRODUCTION

Hydropedology integrates pedology, hydrology, geomorphology, and other related bio- and geosciences to study soil–water–landscape relationships across space and time, emphasizing the connection between the pedon and the landscape paradigms (Fig. 1).^[1] It aims to understand pedological controls on hydrologic processes and properties as well as hydrologic impacts on soil formation and functions.^[2–4] Two fundamental questions of hydropedology are the following^[5]:

How do soil architecture and the distribution of soils over the landscape exert a first-order control on hydrologic processes (and associated biogeochemical and ecological dynamics) across spatiotemporal scales?

How does landscape water (and the associated transport of energy, sediment, chemicals, and biomaterials by flowing water) influence soil genesis, evolution, variability, and functions?

Landscape water encompasses the source, storage, availability, flux, pathway, residence time, and distribution of water in the near-surface terrestrial environment. Although source, storage, availability, and flux of water in the soil have been studied extensively in the past, attention to flow pathways (especially flow networks), residence times (ages of water), and spatiotemporal patterns of flow dynamics (and its underlying organizing principles) has been much limited.^[4,6]

Water flux into and through the soil and over the landscape is the essence of life, which resembles the way blood circulates in a human body.^[7] The interactions between soil and water are so intimate and complex that they cannot be effectively studied in a piecemeal manner; rather, they require a systems and multiscale approach. It is, in this spirit, hydropedology has emerged.^[2–5]

CHARACTERISTICS OF HYDROPEDOLOGY

Two general characteristics of hydropedology can be highlighted as follows:

1. Hydropedology emphasizes *in situ* soils in the landscape, where distinct pedogenic features (e.g., structure, horizonation, and redox feature), soil–landscape relationships (e.g., catena, soil pattern, and soil map), environmental variables (e.g., climate, landforms, and organisms), and anthropogenic impacts (e.g., land use and management) prevail and interact, leading to a complex and dynamic control on the hydrologic cycle and ecosystem functions. Hydropedology attempts to link the past of soils (i.e., soil formation and evolution as a record of the past environment) to the present and future of soils (i.e., soil changes under existing land use and future climate). Hydrology offers an attractive potential to quantify soil-forming processes, because all natural soil-forming factors affect and are affected by hydrology. Water is also key to quantify soil functions; once water regime is characterized, soil physical, chemical, and biological functions can be added as they strongly depend on the water regime and on interactive processes with the soil.
2. Hydropedology promotes a unification of soil forms (i.e., soil architectures and maps) and soil functions rather than mapping soils without considering specific soil functions or modeling soil functions without incorporating soil architectures and soil–landscape patterns. A new era of soil research must be based on soil architecture (i.e., the entirety of how the soil is structured, which encompasses solid components, pore space, and their interfaces). This will improve the prediction of flow (and reaction) pathways, patterns, and residence times but require innovative techniques for improved

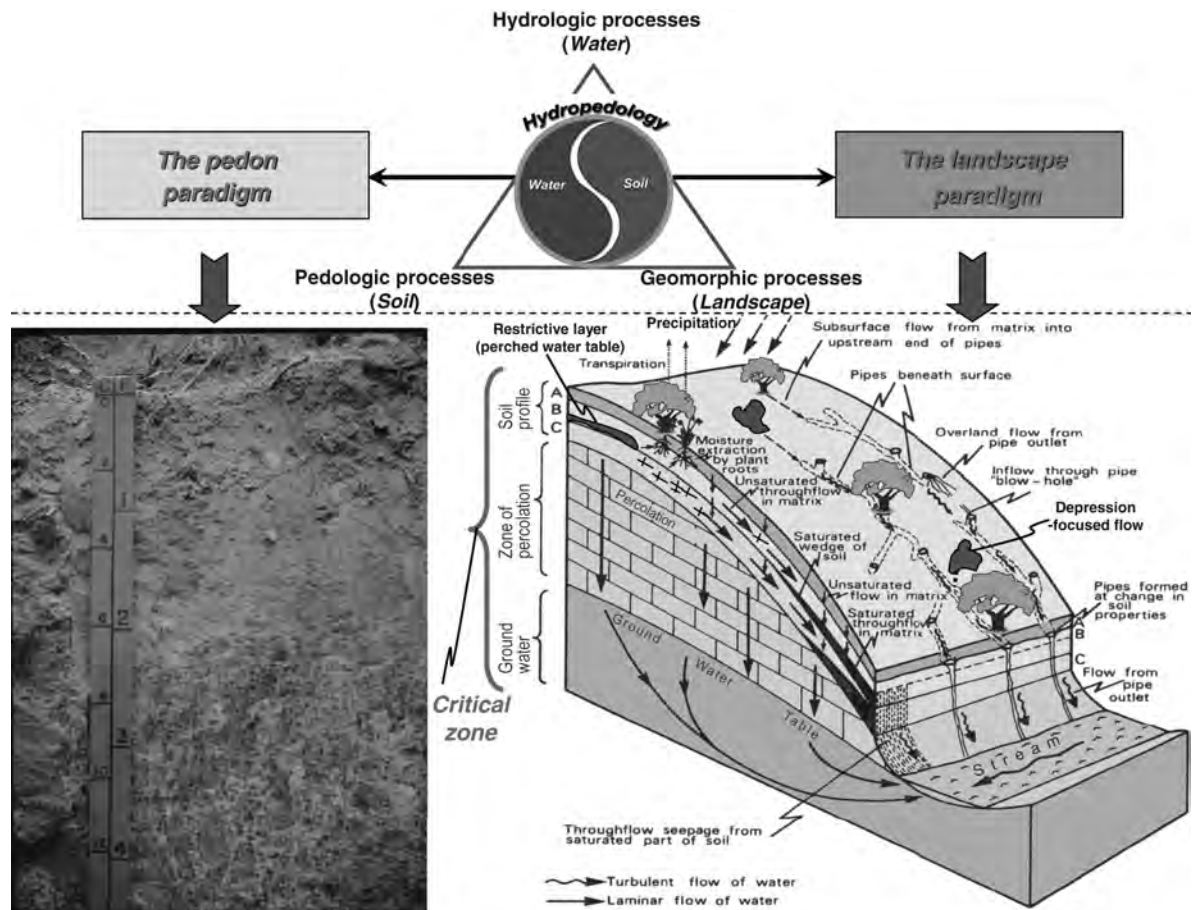


Fig. 1 Hydropedology connects the pedon and the landscape paradigms through linking phenomena occurring at the microscopic (e.g., pores and aggregates) to mesoscopic (e.g., pedons and catenae), macroscopic (e.g., watersheds and regional), and megascopic (e.g., continental and global) scales. There are strong links between ubiquitous heterogeneity and diverse preferential flow in natural soils and landscapes across scales.

Source: The lower right diagram of a hillslope is modified from Atkinson.^[1]

quantification of soil architecture and related flow dynamics across scales (especially *in situ* non-invasively and with high spatial and temporal resolution). New ways of characterizing and mapping soils in the landscape are also needed, including the identification and delineation of soil–landscape units with similar pedological and hydrologic functions.^[8]

FUNDAMENTAL ISSUES

Fundamental scientific issues of hydropedology, at this stage of its development, are centered on the following four categories^[3,5]:

1. *Soil structure and horizonation vs. in situ water flow and chemical transport*: hydropedology emphasizes quantitative soil architecture of field soils and its links to preferential flow across scales.
2. *Soil catena and distribution pattern vs. water movement over the landscape*: hydropedology focuses on

quantitative relationships between the distribution of field soils and their surrounding landscape and the impacts of such relationships on hydrologic (and related biogeochemical and ecological) processes.

3. *Soil morphology and pedogenesis vs. soil hydrology and soil change*: hydropedology utilizes quantitative soil hydromorphology as a signature of soil hydrology and uses the soil as valuable records of past environmental change over time.
4. *Soil functions and maps vs. carriers of soil quality and soil–landscape heterogeneity*: hydropedology promotes quantitative delineation of functional soil units in the landscape as well as precision soil–landscape mapping for various applications.

APPLICATIONS AND LINKS TO OTHER DISCIPLINES

Hydropedology contributes to a variety of environmental, ecological, agricultural, and natural resource issues of

societal importance, including water quality, hydrologic fluxes, soil quality, nutrient cycling, landscape processes, watershed management, contaminant fate, waste disposal, precision agriculture, climate change, and ecosystem functions. In the integrated studies of the earth's critical zone that ranges from the vegetation top to the aquifer bottom,^[5,9] hydropedology is linked to a number of related disciplines to address critical interactions and feedback mechanisms, as highlighted in the following (Fig. 2):

- There is a clear linkage between aboveground ecohydrology and belowground hydropedology in various ecosystems. For example, Li et al.^[10] demonstrated important connection between aboveground stemflow and belowground preferential flow in desert shrubs, revealing the importance of the interrelationships between hydropedology and ecohydrology and how soil moisture and plant growth influence each other.
- The integration of hydropedology and hydrogeology provides a more holistic way of predicting subsurface flow and transport from the ground surface all the way down to the aquifer. The bridging of pedon-scale observations and landscape-scale phenomena, with appropriate incorporation of soil structure into preferential flow modeling, could expedite solutions to many problems of non-point source pollution and groundwater contamination.
- Coupling hydropedology with (hydro)geophysics is an area that has not been well explored. Many questions remain regarding how best to transfer geophysical signals measured to hydrologic properties of interest. One way to get around this inversion problem is to insert pedological insights (e.g., soil types, layers, and structural phases) to help constrain or zone the geophysical inversion. Time-lapse geophysical imaging can also be used to reveal subsurface heterogeneity and complex preferential flow dynamics.
- Linking hydropedology with hydrogeomorphology provides an integrated research and application in the areas of landscape and hillslope evolution, spatial and temporal patterns of water and sediment dynamics in complex terrain, and the assessment of distributed land management practices on soil and water resources and their interactions with natural hazards. Land management decisions concerning where and when specific operations can take place are strongly influenced by spatiotemporal, hydrogeomorphic, and hydropedologic relations, as are sediment and landslide hazards.
- The linkage between hydropedology and hydrometeorology includes issues related to soil moisture and global climate change, soil carbon sequestration, greenhouse gas emission from soils, and remote sensing of soil climate.^[11]
- Coupling hydropedology and biogeochemistry can help the identification of hot or cold spots and hot or cold

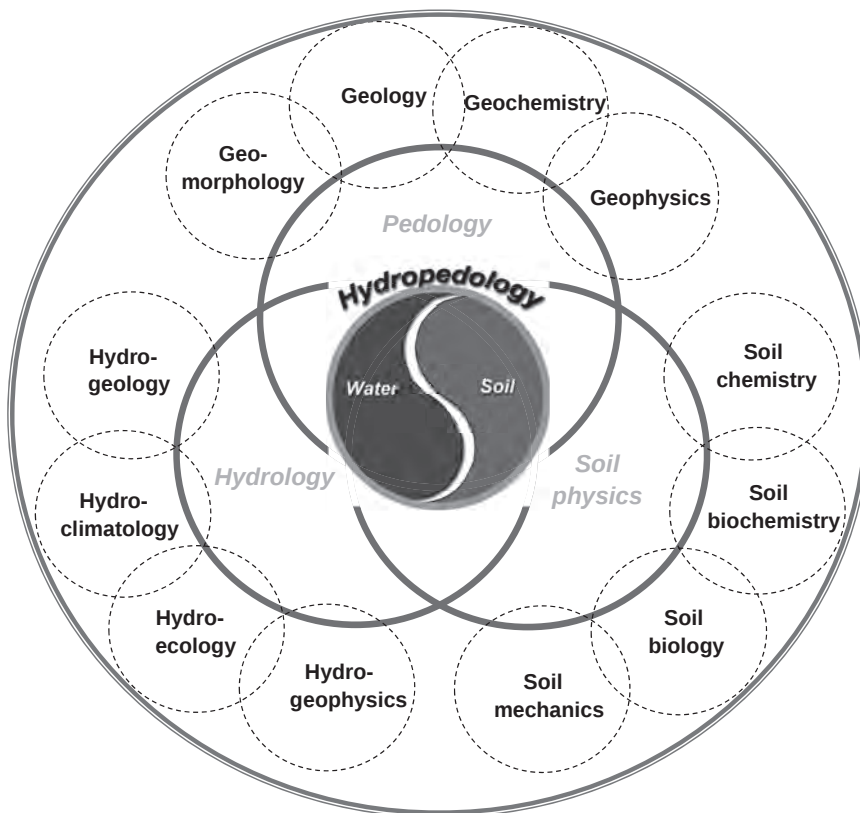


Fig. 2 Relationship of hydropedology with other related disciplines, where pedology, hydrology, and soil physics serve as the three cornerstones for hydropedology.

- moments of biogeochemical cycles in different ecosystems, which are often triggered by soil hydrologic conditions.^[12,13] This is important to subsurface microbial activities, contaminant remediation, ecological health and diversity, and environmental quality in general.
- The study of paleohydropedology (Paleosols and paleohydrology) has demonstrated valuable historical records of the past environment and ancient landscape–soil–water–ecosystem–climate relationships.^[14]
 - The connection between hydropedology and land use planning is linked to environmental regulation and policy, because how natural soils “throb” upon precipitation inputs under various climates offers clues as to “what” best can be done and “where” has the lowest risks and the greatest opportunities for land use and management.^[7]
 - Hydropedology contributes important knowledge to sustainable development using “soft,” ecological engineering that aims at utilizing natural systems to provide needed services, including stormwater management, urban drainage, and hazard mitigation. In the meantime, communications with engineers and the sociopolitical process are critical for realizing these goals.
 - The emerging critical zone science^[5,9] provides a new interdisciplinary platform for the holistic studies of water with soil, rock, air, and biotic resources in the near-surface terrestrial environment. The critical zone is the most heterogeneous and complex region of the earth that requires contributions from many disciplines including hydropedology.^[5]
 - Knowledge of the soil and its forming processes has unique contribution to extraterrestrial explorations in search of liquid water^[15] and life and to developing advanced life support systems used in space exploration. For example, how the weathering engine on Mars has transformed the protolith into various soils could shed light on the role of water (and other soil-forming factors such as climate) in the genesis and evolution of Martian soils.
- ## REFERENCES
1. Atkinson, T.C. Techniques for measuring subsurface flow on hillslopes. In *Hillslope Hydrology*; Kirkby, M.J., Ed.; John Wiley & Sons: Chichester, 1978; 1–42.
 2. Lin, H.S. Hydropedology: Bridging disciplines, scales, and data. *Vadose Zone J.* **2003**, *2*, 1–11.
 3. Lin, H.S.; Bouma, J.; Wilding, L.; Richardson, J.; Kutilek, M.; Nielsen, D. Advances in hydropedology. *Adv. Agron.* **2005**, *85*, 1–89.
 4. Lin, H.S.; Bouma, J.; Pachepsky, Y.; Western, A.; Thompson, J.; van Genuchten, M.T.; Vogel, H.; Lilly, A. Hydropedology: Synergistic integration of pedology and hydrology. *Water Resour. Res.* **2006**, *42*, W05301.
 5. Lin, H.S. Earth’s critical zone and hydropedology: Concepts, characteristics, and advances. *Hydrol. Earth Syst. Sci.* **2010**, *14*, 25–45.
 6. McDonnell, J.J.; Sivapalan, M.; Vache, K.; Dunn, S.; Grant, G.; Haggerty, R.; Hinz, C.; Kirchner, J.; Roderick, M.L.; Selker, J.; Weiler, M. Moving beyond heterogeneity and process complexity: A new vision for watershed hydrology. *Water Resour. Res.* **2007**, *43*, W07301.
 7. Bouma, J. Hydropedology as a powerful tool for environmental policy research. *Geoderma* **2006**, *131*, 275–286.
 8. Lin, H.S.; Brook, E.; McDaniel, P.; Boll, J. Hydropedology and surface/subsurface runoff processes. In *Encyclopedia of Hydrologic Sciences*; Anderson, M.G., Ed.; John Wiley & Sons, Ltd.: Chichester, 2008.
 9. National Research Council (NRC). *Basic Research Opportunities in Earth Science*; National Academy Press: Washington, 2001.
 10. Li, X.Y.; Yang, Z.P.; Zhang, X.Y.; Lin, H.S. Connecting ecohydrology and hydropedology in desert shrubs: Stem-flow as a source of preferential water flow in soils. *Hydrol. Earth Syst. Sci.* **2009**, *13*, 1133–1144.
 11. Lam, A.; Bierkens, M.F.P.; van den Hurk, B.J.J.M. Global patterns of relations between soil moisture and rainfall occurrence in ERA-40. *J. Geophys. Res.* **2007**, *112*, D17116.
 12. McClain, M.E.; Boyer, E.W.; Dent, C.L.; Gergel, S.E.; Grimm, N.B.; Groffman, P.M.; Hart, S.C.; Harvey, J.W.; Johnston, C.A.; Mayorga, E.; McDowell, W.H.; Pinay, G. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. *Ecosystems* **2003**, *6*, 301–312.
 13. Lohse, K.A.; Paul, D.B.; McIntosh, J.C.; Meixner, T.; Huxman, T.E. Interactions between biogeochemistry and hydrologic systems. *Annu. Rev. Env. Resour.* **2009**, *34*, 65–96.
 14. Ashley, G.M.; Driese, S.G. Paleopedology and paleohydrology of a volcanoclastic paleosol interval: Implications for early pleistocene stratigraphy and paleoclimate record, Olduvai Gorge, Tanzania. *J. Sediment Res.* **2000**, *70*, 1065–1080.
 15. Squyres, S.W.; Grotzinger, J.P.; Arvidson, R.E.; Bell, J.F.; Calvin, W.; Christensen, P.R.; Clark, B.C.; Crisp, J.A.; Farrand, W.H.; Herkenhoff, K.E.; Johnson, J.R.; Klingelhofer, G.; Knoll, A.H.; McLennan, S.M.; McSween, H.Y.; Morris, R.V.; Rice, J.W.; Rieder, R.; Soderblom, L.A. *In situ* evidence for an ancient aqueous environment at Meridiani Planum, Mars. *Science* **2008**, *306*, 1709–1714.

Hydrophobic and Hydrophilic Compounds

Tomas Simon

Department of Plant Nutrition, Crop Research Institute, Prague, Czech Republic

Abstract

Soil organic matter (SOM) plays a key role in the soil quality. Besides carbon and nitrogen content, SOM composition can also be defined as the amount of functional groups (i.e., hydrophobic and hydrophilic groups) and their spatial arrangement in SOM at the molecular scale. The SOM functional groups are responsible for the chemical reactivity and sorptivity of SOM. The importance of SOM hydrophobic and hydrophilic compounds for soil hydrophobicity and related spectroscopic techniques used for the determination of these SOM constituents is described in this entry. Fourier transform infrared (FTIR) spectroscopy of humic substances or bulk soil samples can distinguish hydrophobic and hydrophilic functional groups in FTIR spectra and may be useful for analyzing SOM quality and to improve interpretations of its physicochemical properties.

INTRODUCTION

Soil organic matter (SOM) is a complex material consisting of plant biopolymers, animal and microbial residues, and humic substances. It plays a key role in soil quality through moderating biological, physical, and chemical properties of soils. It contributes to the stability of soil aggregates and provides energy and body building constituents for the soil biota. The SOM also affects many other soil characteristics such as soil porosity, water-holding capacity, hydraulic conductivity and detoxication of pollutants, erosion, etc.

The SOM composition depends on soil type and land management. In arable soils, both the amount and composition of the SOM depend on the organic matter input from organic manuring, crop, and root biomass.^[1] The SOM is also related to crop rotation including cultivation of leguminous crops.

While the SOM quantity is mostly determined as the total organic carbon (C) or nitrogen (N) content in the top soil, soil quality is characterized using various methods especially determining labile components of the SOM (light fractions, water extractable fractions, dissolved organic matter, microbial biomass C, etc.).^[2] As large fraction of the SOM in many arable soils is assumed to be relatively inert, analysis of total SOM is not sensitive to assessing the management effect. On the contrary, determination of labile water soluble soil organic fractions with relatively fast turnover rates may reflect short-term effects of crop fertilization and rotation.

However, SOM contains not only C but also a number of heteroatoms, which characterize SOM composition and interaction with mineral surface. The SOM composition,

therefore, can be defined as the amount of functional groups (i.e., carboxylic and hydroxylic groups) and their spatial arrangement in SOM at the molecular scale. The spatial arrangement of the hydrophobic components within SOM strongly affects wettability, which is dependent on the SOM–mineral interactions. The SOM functional groups are responsible for the chemical reactivity and sorptivity of the SOM (e.g., hydrophobicity or cation exchange capacity).^[3]

The hydrophobicity of SOM is caused, essentially, by aliphatic C–H units present in methyl, methylene, and methine groups. These substances may be released from a range of plants, decaying SOM, soil fauna, and microorganisms. The number of aliphatic C–H units control the water affinity that influences resistance to microbial degradation, the rate of wetting, and adsorption processes. Hydrophobic organic constituents may be a primary cause of soil repellency; hydrophilic constituents affect soil wettability.^[4] Soil wettability depends, therefore, on composition and spatial arrangement of different hydrophobic constituents.

It may also be assumed for SOM that the ratio between hydrophilic and hydrophobic functional groups (e.g., C=O/CH ratio) could be used as an index soil's wettability. Most soils are neither completely hydrophilic nor hydrophobic. However, they exhibit a subcritical level of water repellence.^[5] The degree of water repellence decreases with increase in moisture content and temperature. Agricultural management strongly influences the hydrophobicity of SOM. Decline in soil organic C (SOC) pool by cultivation leads to decrease of hydrophobicity as well as of microbial activity and aggregate stability.^[3]

Several approaches using modern and effective analytical techniques are available for the characterization of SOM

and humic substances associated with bulk soils. These techniques involve the use of chemolytic methods, pyrolysis, thermochemolysis, compound specific-stable isotope, and radiation analysis. Besides these techniques, spectroscopic technique such as Fourier transform infrared (FTIR) spectroscopy is used for the analyses of SOM composition. The FTIR spectra of humic macromolecules contain a variety of bands that are diagnostic and could serve as a valuable tool to characterize the principal classes of chemical groups of which SOM is comprised. A synchrotron-based FTIR technique has been applied to determine composition of SOM extracted from the silt and clay fractions of soils.^[6] The FTIR technique is utilized to characterize SOM functional groups depending on land use and management practices. Based on FTIR analyses and evaluation of the intensities of oxygen-containing functional groups to the aliphatic and aromatic (referred to as recalcitrant) groups, the highest ratio is observed in the humic acids from the vetch/rye system with fertilizer N, compared with either rye alone or no cover crops.^[7] Isolation and extraction of humic substances are a time-consuming process, and extraction of SOM by various extraction agents is often incomplete due to the bonding with the mineral particles. Besides analyses of extracted humic substances, FTIR technique can therefore be directly used for the study of SOM composition in bulk soil samples. While using bulk soil samples, the FTIR spectra can be analyzed at absorption bands that indicate hydrophobic (CH-groups) and hydrophilic (CO-groups) functional groups. For hydrophobic methyl and methylene groups, the CH bands occur at 3020–2800 cm^{-1} ,^[3] and for hydrophilic amide and carboxylic acid or ketone groups, the C=O bands occur at 1740–1700 and 1640–1600 cm^{-1} . Focusing on analyzing of these ranges of spectra excludes any overlapping by absorption bands originating from other functional groups. Furthermore, hydrophobic/hydrophilic functional group ratio can be calculated in order to assess soil wettability and soil repellency.^[8] Based on FTIR spectroscopy, Capriel et al.^[3] defined a hydrophobicity index as the aliphatic C/total organic C ratio to characterize the degree of hydrophobicity of the SOM. Capriel and colleagues observed a close relationship between hydrophobicity of organic matter and soil texture. The organic matter of sandy soils contained more aliphatic C–H units than that in the clayey soils.

It has been demonstrated that management practices that increase SOC content may increase water repellency and decrease wettability. Urbanek et al.^[9] studied the spatial distribution of C and groupings of C–H and C=O compounds in three horizons in a silty loam Anthroisol, under long-term grassland pastures and also short-term conservation tillage wheat, and conventional tillage maize. They observed that total SOC content within individual aggregates and size classes varied considerably and were influenced by soil depths and management practices. More hydrophilic C=O groups were observed in maize and grassland biomes, where the plant biomass was greater than in

wheat. Total SOC content was strongly correlated with hydrophobic (C–H) groups, but not with hydrophilic groups. Similarly, a highly significant positive correlation was observed between SOC content and hydrophobic functional groups in soil samples taken from a long-term fallow experiment.

A good soil structure is important to sustaining long-term crop production on agricultural soils. One of the measures to form and maintain soil structure is the stability of soil aggregates in water, and this is influenced mostly by both the quality and the quantity of SOM pool. Several studies focused on the use of organic materials to improve soil structural stability have observed positive effects of total organic matter, whereas others have indicated that it is the humified fractions rather than the total amount per se that are responsible for aggregate stabilization. The practical implication of these observations is that soil amendment with organic materials containing substantial amounts of hydrophobic compounds can stabilize soil aggregates more than those with predominantly hydrophilic compounds.^[10]

CONCLUSION

There are many direct and indirect criteria for the assessment of the SOM quality. Most frequently, extraction and fractionation of humic substances mainly, humic acids and fulvic acids and calculation of so-called color quotient $Q_{4:6}$, are used. The main problems with these classical methods are difficulties with the interpretation of results and high demand of time and labor. Techniques involving determining the labile components of SOM are used for SOM quality evaluation.

Besides these techniques, non-destructive FTIR spectroscopy distinguishing hydrophobic and hydrophilic functional groups in SOM is rapid, and a less time-consuming method that may be useful for further analyzing SOM quality and improving interpretations of its physicochemical properties. It can also increase the knowledge of the effects of organic amendments on the turnover of SOM and to identify soil-amending practices based on the management of organic by-products.

ACKNOWLEDGMENT

This work was supported by research project No. MZE0002700604 from the Ministry of Agriculture of the Czech Republic.

REFERENCES

1. Kaiser, M.; Ellerbrock, R.H.; Gerke, H.H. Long-term effect of crop rotation and fertilization on soil organic matter composition. *Eur. J. Soil Sci.* **2007**, *58*, 1460–1470.

2. Ghani, A.; Dexter, M.; Perrot, K.W. Hot-water extractable carbon in soils: A sensitive measurement for determining impacts of fertilisation, grazing and cultivation. *Soil Biol. Biochem.* **2003**, *35*, 1231–1243.
3. Capriel, P.; Beck, T.; Borchert, H.; Gronholz, J.; Zachmann, G. Hydrophobicity of the organic matter in arable soils. *Soil Biol. Biochem.* **1995**, *27*, 1453–1458.
4. McKissock, I.; Gilkes, R.J.; van Bronswijk, W. The relationship of water repellency to aliphatic C and kaolin measured using DRIFT. *Aust. J. Soil Res.* **2003**, *41*, 251–265.
5. Hallett, P.D.; Baumgartl, T.; Young, I.M. Subcritical water repellency of aggregates from a range of soil management practices. *Soil Sci. Soc. Am. J.* **2001**, *65*, 184–190.
6. Solomon, D.; Lehmann, J.; Kinyangi, J.; Liang, B.; Schäfer, T. Carbon K-Edge NEXAFS and FTIR-ATR spectroscopic investigation of organic carbon speciation in soils. *Soil Sci. Soc. Am. J.* **2005**, *69*, 107–119.
7. Ding, G.; Liu, X.; Herbert, S.; Novak, J.; Amarasiriwanrdena, S.; Xing, B. Effect of cover crop management on soil organic matter. *Geoderma* **2006**, *130*, 229–239.
8. Ellerbrock, R.H.; Gerke, H.H.; Bachmann, J.; Goebel, M.-O. Composition of organic matter fractions for explaining wettability of three forest soils. *Soil Sci. Soc. Am. J.* **2005**, *69*, 57–66.
9. Urbanek, E.; Hallet, P.; Feeney, D.; Horn, R. Water repellency and distribution of hydrophilic and hydrophobic compounds in soil aggregates from different tillage system. *Geoderma* **2007**, *140*, 147–155.
10. Piccolo, A.; Mbagwu, J.S.C. Role of hydrophobic components of soil organic matter in soil aggregate stability. *Soil Sci. Soc. Am. J.* **1999**, *63*, 1801–1810.

Hydrophobicity

Joerg Bachmann

Institute for Soil Study, University of Hannover, Hannover, Germany

Abraham Marmur

Department of Chemical Engineering, Technion-Israel Institute of Technology, Haifa, Israel

Markus Deurer

Institute of Soil Science, University of Hannover, Hannover, Germany

Abstract

The most important effect of hydrophobicity is the controlling of soil water dynamics. Water repellency affects a wide range of physical, chemical, and biological properties and processes of soil and may have a significant effect on field capacity, solute transport in the unsaturated zone, soil water balance, aggregate stability, soil erosion, soil fertility, and the protection of soil organic matter against microbial degradation. A quantitative understanding of important mechanisms associated with hydrophobicity, e.g., the temporal dynamics of hydrophobicity and the spacial moisture pattern in soil, is needed.

INTRODUCTION

Many processes such as particle wetting, imbibition into porous media, adsorption, flocculation, or dispersion depend on interfacial interactions between soil solids and soil liquid. Hydrophobicity (“fear of water”) and hydrophilicity (“affinity to water”) are qualitative terms that describe the extent of interaction of surfaces with water. Hydrophobic surfaces generally repel water and tend not to absorb water, mix with it, or adsorb it. Hydrophilic surfaces show the opposite behavior.

Water repellency has been observed in sand, loam, clay, peat, and volcanic ash soils, all over the world. Research suggests that under certain conditions most soils display water repellency to some degree. Particularly sandy soils, with a comparatively smaller specific surface, are prone to hydrophobicity. Hydrophobicity, or water repellency, is caused by natural soil organic matter (SOM). SOM either covers the mineral grains as thin coatings or exists as particular organic matter, reducing potentially in both cases the wettability. The origin of SOM is mostly plant-derived like roots or plant tissues, plant-derived waxes or organic acids, fungal hyphae or microbial organic acids, and polysaccharides. For more information, see the entry *Water-Repellent Soils* (p. 2546).

HYDROPHOBICITY, WETTABILITY, AND CAPILLARITY

Interfacial interaction is mainly controlled by the interfacial tension, defined as the surface energy per unit area

(or sometimes as force per unit length) of the interface between two phases. Each phase by itself is characterized by its surface tension, which actually is the interfacial tension between this phase and a gas (the identity of which is unimportant because of its very low density). The interfacial tension can be estimated from the individual surface tensions by a variety of possible correlations.^[1] In soil, water with a high surface tension of about 72 mN/m (force per unit length) plays a dominant role. Surface tensions of soil may range from 20 to 60 mN/m.^[2] Surface tensions of solids range from 500 mN/m for metals and minerals (high-energy surfaces), depending on their hardness and melting point, down to 10 mN/m for organic surfaces.

Hydrophobicity/hydrophilicity of a surface can be quantitatively measured by the equilibrium contact angle (c.a.) that water makes with this surface in air. The c.a. is defined as the angle (°) formed between the tangent to the solid–air interface and the tangent to the liquid–air interface at the three-phase contact line, measured on the water side (Fig. 1). The c.a., θ , on ideal solid surfaces (i.e., smooth, rigid, chemically homogeneous, insoluble, and non-reactive) is related to the interfacial tensions by the Young equation,^[3] which can be given as

$$\sigma_{LV}\cos\theta = \sigma_{SV} - \sigma_{SL} \quad (1)$$

where σ_{SV} is the surface tension of the solid–gas interface, σ_{LV} the surface tension of the liquid–gas interface, and σ_{SL} the solid–liquid interfacial tension. This c.a. is

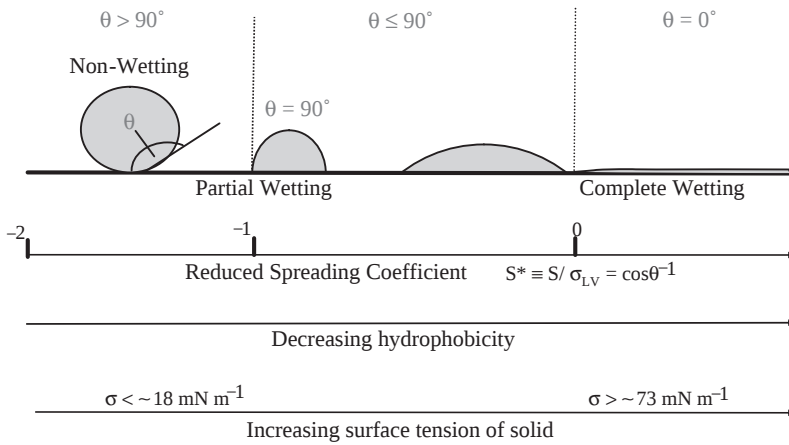


Fig. 1 Schematic representation of the various degrees of hydrophobicity as well as the corresponding solid surface tension and the reduced spreading coefficient.

referred to as the ideal or Young c.a. Although c.a. values may vary continuously depending on the surface tension of the solid in question, it is convenient to think in terms of three different wetting situations (Fig. 1): Complete wetting, for which the ideal c.a. is zero and the liquid forms a very thin film, partial wetting with $0^\circ < \theta \leq 90^\circ$, and partial wetting with $\theta > 90^\circ$, which is sometimes referred to as non-wetting. Wetting systems can also be characterized by the spreading coefficient, S (mN/m), defined by

$$S = \sigma_{SV} - (\sigma_{SL} + \sigma_{LV}) = \sigma_{LV}(\cos \theta - 1) \quad (2)$$

This is a measure of the energy per unit area needed to replace a solid–air interface by a solid–liquid or liquid–air interface ($S \geq 0$ for complete wetting and < 0 for partial wetting).

The c.a. that a liquid forms with a solid surface controls not only the spreading of the liquid on the surface but also its imbibition into a porous medium.^[4] When a liquid is brought into contact with an unsaturated porous medium like soil, it attempts to form the appropriate equilibrium c.a. with the inside surface of the soil pores. This is associated, in many situations, with the formation of a curved liquid–air interface. This curvature, in turn, implies a pressure difference across the liquid–air interface, which is given by the Young–Laplace equation.^[3] In a cylindrical, straight capillary, this pressure difference, ψ (Pa), is given by

$$\psi = p^g - p^l = 2\sigma_{LV} \frac{\cos \theta}{r} \quad (3)$$

where p^g (Pa) is the pressure in the gas phase, p^l (Pa) the pressure in the liquid, and r (m) the radius of the capillary. If this pressure difference is positive (i.e., $\theta < 90^\circ$), the liquid spontaneously penetrates into the porous medium. Hydrophobic surfaces ($\theta > 90^\circ$) feature capillary depression, unless the liquid comes in the form of small droplets. In the latter case, infiltration into porous material is possible even for a c.a. $> 90^\circ$.^[5]

ASSESSMENT OF HYDROPHOBICITY

Numerous effects such as surface roughness, chemical heterogeneity, wetting dynamics, particle shape, and gas adsorption give rise to c.a. hysteresis, i.e., larger c.a.s for an advancing wetting front and smaller c.a.s for a receding wetting front, as compared with the intrinsic equilibrium c.a.^[6–9] The stochastic combination of heterogeneous surface properties and pore topologies encountered in natural porous media emphasizes that observations relate, at best, to the apparent c.a., which does not directly represent the ideal c.a. at interfaces within the medium.^[10]

For soil, indirect assessment methods have to be applied.^[11] The most frequently applied methods include the water drop penetration time test (WDPTT) and the molar ethanol droplet test (MEDT). In MEDT, the molarity of water and ethanol mixtures is varied repeatedly, until droplets of the mixture infiltrate into the soil in a pregiven length of time. Roy and McGill^[12] suggested a standardized procedure for MEDT to improve its reproducibility. Data obtained, e.g., with WDPTT indicate that water repellency is time dependent. To measure the initial c.a. of a dry powder, Siebold et al.^[13] applied the capillary rise method for wettable or slightly repellent mineral particles. Later, Bachmann et al.^[2] proposed the Wilhelmy plate method, which allows the measurement of dynamic advancing and receding c.a. for wettable and repellent soil particles.

EFFECTS OF HYDROPHOBICITY

Reduced wettability covers a wide range of severity and is often too weak to be detected by visual diagnosis. This usually less extreme manifestation of water repellency is often referred to as subcritical repellency.^[14] Extreme soil water repellency can lead to a complete loss of infiltration capacity, enhancing runoff from hillslopes, which, in turn, may enhance soil erosion and

cause the occurrence of landslides. Enhanced ponding at the soil surface favors preferential flow in the soil.^[15] Such soil wetting patterns either force soil water into macropores or lead to persistent non-uniform soil wetting patterns, which implicate a rapid transfer of solutes into the groundwater.^[16] Compared with uniformly wetted surfaces, a surface having a pattern of dry ridges and wet furrows can potentially reduce the average evaporation rate of a field. The effect of water repellency is not restricted to dry soil. An increase of water repellency with increasing soil water content was phenomenologically observed also for moist soils.^[17] Simultaneously, a decrease of hydrophobicity with time was observed. So far, the prediction of the critical water content, at which the transition from hydrophilic to hydrophobic behavior of soil particle surfaces occurs, can be rated as an unsolved problem.

For wetting soils, Morrow^[18] presented a relation between the retention of a liquid in a porous medium and the intrinsic c.a. θ_E measured on a plane and clean surface, where θ_E is considered as a thermodynamic sound property that defines the system wettability under equilibrium conditions (Fig. 2).

A comparison of the c.a. measured on plane plates with those either assessed in roughened capillaries (advancing c.a. θ_A and receding c.a. θ_R) or derived from retention data (Φ_A and Φ_R) suggests that these operative c.a.s were

in reasonable agreement with intrinsic c.a. θ_E . So far, only few data exist for real soils to quantify the effect of the degree and the persistence of hydrophobicity on water retention.

Hydrophobicity, however, not only has negative effects. For example, hydrophobic surface layers have been used for water harvesting and for preventing evaporation of soil water in arid climates.^[15] With respect to carbon sequestration in soil, it was shown that the higher the degree of hydrophobicity of the humic material, the larger is the stabilization, i.e., protection against microbial degradation, of organic carbon because of the increased aggregate stability.^[19]

CONCLUSION

Hydrophobicity of natural soils is mainly caused by SOM of natural origin. It is becoming increasingly evident that, besides strong hydrophobicity ($\theta > 90^\circ$), subcritical repellency ($0^\circ < \theta \leq 90^\circ$) seems for many soils more the rule than the exception. Methods allow only the assessment of hydrophobicity on a small sample scale. On the field scale, it is difficult to separate experimentally the influence of hydrophobicity on the soil's hydraulic properties from that of other factors such as the pore topology and its connectivity. However, a quantitative understanding of the

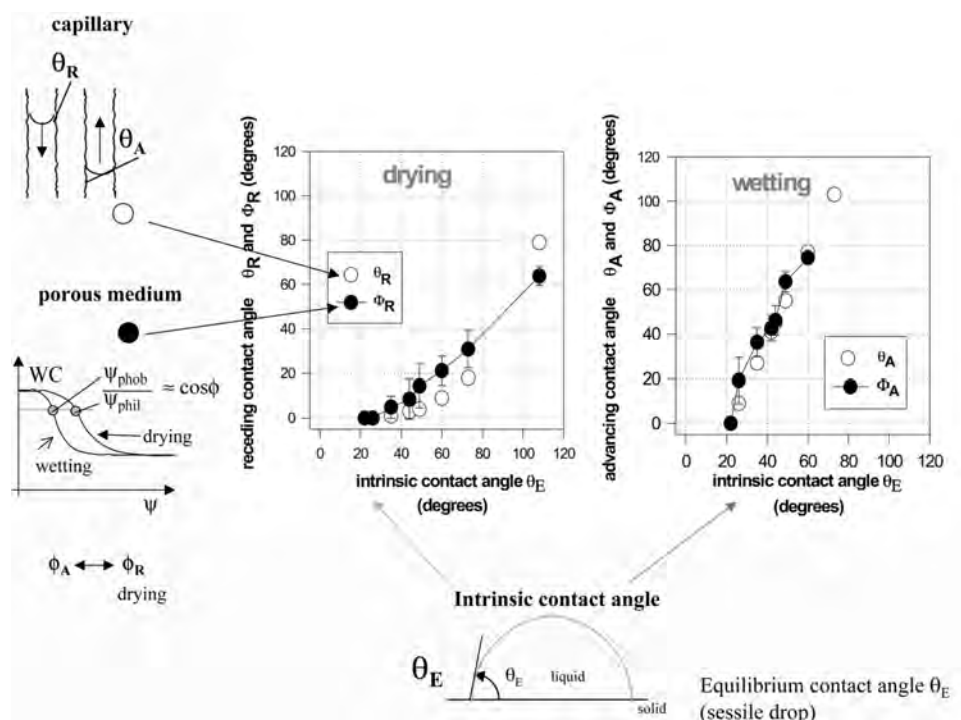


Fig. 2 Equilibrium c.a. θ_E vs. c.a. in capillaries (advancing angle θ_A and receding angle θ_R) and c.a. Φ determined from porous media for drainage Φ_R and imbibition Φ_A (the saturation of the porous medium is indicated by WC).

Source: Data from Morrow.^[18]

hydrophobic behavior of soils in time and space is a prerequisite to increase the knowledge on fundamental physical, chemical, and biological processes in terrestrial ecosystems.

REFERENCES

1. van Oss, C.J.; Chaudhury, M.K.; Good, R.J. Monopolar surfaces. *Adv. Colloid Interfac.* **1987**, *28*, 35.
2. Bachmann, J.; Woche, S.K.; Göbel, M.-O.; Kirkham, M.B.; Horton, R. Extended methodology for determining wetting properties of porous media. *Water Resour. Res.* **2003**, *39*, 1353.
3. Adamson, A.W.; Gast, A.P. *Physical Chemistry of Surfaces*, 6th Ed.; John Wiley and Sons: New York, 1997.
4. Marmur, A. Penetration and displacement in capillary systems. In *Modern Approaches to Wettability*; Schrader, M.E., Loeb, G., Eds.; Plenum Press: New York, London, 1992; Chapter 12, 327–358.
5. Marmur, A. Penetration and displacement in capillary systems of limited size. *Adv. Colloid Interfac.* **1992**, *39*, 13–33.
6. Dettre, R.H.; Johnson, R.E. Contact angle hysteresis. II. Contact angle measurement on rough surfaces. In *Contact Angle, Wettability and Adhesion, Advances in Chemistry, Series 43*; Gould, R.F., Ed.; American Chemical Society: Washington, 1964; 136–144.
7. Drehlich, J. Static contact angles for liquids at heterogeneous rigid solid surfaces. *Polish J. Chem.* **1997**, *71*, 525–549.
8. Marmur, A. Contact angle hysteresis on heterogeneous smooth surfaces: Theoretical comparison of the captive bubble and drop methods. *Colloid. Surface. A.* **1998**, *136*, 209–215.
9. Marmur, A. Wetting on hydrophobic rough surfaces: To be heterogeneous or not to be? *Langmuir* **2003**, *19*, 8343–8348.
10. Philip, J.R. Limitations on scaling by contact angle. *Soil Sci. Soc. Am. Proc.* **1971**, *35*, 507–509.
11. Wallis, M.G.; Horne, D.J. Soil water repellency. In *Advanced Soil Science*; Stewart, B.A., Ed.; Springer: New York, 1992; Vol. 20, 91–146.
12. Roy, J.L.; McGill, W.B. Assessing soil water repellency using the molarity of ethanol droplet (MED) test. *Soil Sci.* **2002**, *167*, 83–97.
13. Siebold, A.; Nardin, M.; Schultz, J.; Walliser, A.; Opplinger, M. Effect of dynamic contact angle on capillary rise phenomena. *Colloid. Surface. A.* **2000**, *161*, 81–87.
14. Tillman, R.W.; Scotter, D.R.; Wallies, M.G.; Clothier, B.E. Water-repellency and its measurement by using intrinsic sorptivity. *Aust. J. Soil Res.* **1989**, *27*, 637–644.
15. De Bano, L.F. Historical overview of soil water repellency. In *Soil Water Repellency*; Ritsema, C.J., Dekker, L.W., Eds.; Elsevier Science B.V.: Amsterdam, 2003; 3–24.
16. de Rooij, G.H.; de Vries, P. Solute leaching in a sandy soil with a water-repellent surface layer: A simulation. *Geoderma* **1996**, *70*, 253–263.
17. Doerr, S.H.; Shakesby, R.A.; Walsh, R.P.D. Soil water repellency: Its causes, characteristics and hydrogeomorphological significance. *Earth Sci. Rev.* **2000**, *51*, 33–65.
18. Morrow, N.R. Capillary pressure correlations for uniformly wetted porous media. *J. Can. Petr. Technol.* **1976**, *15*, 49–69.
19. Piccolo, A.; Spaccini, R.; Haberbauer, G.; Gerzabek, M.H. Increased sequestration of organic carbon by hydrophobic protection. *Naturwissenschaften* **1999**, *86*, 496–499.

Inceptisols

John M. Galbraith

*Crop and Soil Science, Virginia Polytechnic Institute and State University,
Blacksburg, Virginia, U.S.A.*

Kenneth Scheffe

Shawn McVey

*National Soil Survey Center, U.S. Department of Agriculture (USDA),
Lincoln, Nebraska, U.S.A.*

Abstract

Inceptisols are important soils worldwide, covering about 14.5% of the earth's ice-free surface, with the amount expected to increase as ice and frozen soil melts. *Inceptisols* are important farmlands on level areas in semiarid climates, infloodplains, and on terraced hillslopes. Inceptisols are considered an intermediate stage of development between *Entisols* and the more highly developed soil orders preceding them in the *Key to Soil Orders*, except where sandy parent material, very steep slopes, or wet, dry, or cold climate permanently limit their development. Modal *Inceptisols* have pale or thin surface horizons (*ochric epipedon*) and a minimally developed subsoil (*cambic horizon*). *Inceptisols* can form in any type of parent material except thick organic deposits, in any *soil moisture regime* other than *aridic* or *torric* (where they would be identified as *Aridisols*), and in any *soil temperature regime*. They are concentrated in four major settings—cold regions, dry regions (*ustic* and *xeric*), wetlands, and young geological surfaces (such as recent or very steep parent material deposits).

INTRODUCTION

Inceptisols

Inceptisols (terms from Soil Taxonomy^[1,2] are italicized.) are a unique soil order. They are essentially intermediate in expressed development between soil orders that precede them in *Key to Soil Orders* and soils that have no diagnostic horizons other than *anthropic*, *ochric*, or *albic* (*Entisols*). *Inceptisols* are mineral soils that have significant surface or subsoil alteration, being morphologically similar to some *Gelisols*, *Andisols*, *Spodosols*, *Vertisols*, and *Entisols*. Similar soils in *aridic* or *torric soil moisture regimes* that have *cambic* horizons classify as *Aridisols*. Their development has been or is limited by one or more of the five soil-forming factors (parent material, climate, relief, organisms, and time). For example, soils developing on steep mountain slopes from acid sedimentary rocks have limited surface (*ochric epipedon*) and subsoil horizon development (*cambic* or *fragipan* diagnostic horizon) because of steep slope and acidic parent material. If they have more than 8% clay, they must have dewatered enough following aqueous deposition to be able to support the weight of humans and most livestock (have a *n* value of more than 0.7) to avoid being *Entisols*. Note that there are also almost 300 soil series in the United States that have the morphology of *Inceptisols* but have sandy subsoils that are not allowed for a *cambic*

horizon, and are therefore considered *Entisols* because of the absence of a diagnostic subsoil horizon and limited potential for further development. *Inceptisols* can form in any type of parent material except thick organic deposits on any slope or landform from floodplains to mountaintops and are found under all vegetation types. They may occur in any *soil moisture regime* other than *aridic* or *torric* and any *soil temperature regime*. Inceptisols range in age from recent to very old, dependent mainly on the parent material and climate. Older deposits must be limited from developing by the amount of quartz sand, the climate, or the slope.

OCCURRENCE

Inceptisols are concentrated in four major settings—cold regions, dry regions (*ustic* and *xeric*), wetlands, and geologically young surfaces (such as floodplains, volcanic and glacial deposits, and steep or highly erosive slopes). The area of *Inceptisols* is expected to increase in cold regions such as Siberia, Canada, and Alaska where *permafrost* is melting. About one-fourth of the land area of Europe and Central America and less than one-tenth of the other continents consist of *Inceptisols*. Approximate areas of each suborder^[3] by continent are listed in Table 1. *Inceptisols* make up about one-quarter of the soils in Europe and

Table 1. Area (km²) of *Inceptisols* on different continents.

SUBORDER	North America	Central America	South America	Europe	Africa	Asia	Australia/Oceania	Global
<i>Aquepts</i>	348,880	45,966	413,983	194,742	369,402	1,394,880	2,878	2,770,731
<i>Gelepts</i>	1,190,995	—	1,901	98,592	—	4,751,601	—	6,043,089
<i>Cryepts</i>	482,133	—	260,776	350,803	371	1,495,418	8,672	2,598,174
<i>Ustepts</i>	56,786	255,907	180,768	369,529	763,748	589,332	14,902	2,230,971
<i>Xerepts</i>	9,326	—	8,286	320,890	167,253	176,546	261	682,562
<i>Udepts</i>	499,145	93,503	1,159,532	767,871	401,114	1,376,360	258,200	4,555,724
Total	2,587,266	395,375	2,025,245	2,102,427	1,701,888	9,784,137	284,913	18,881,251

Central America. Altogether, *Inceptisols* occur on about 14.5% of the ice-free land surface of the earth. *Inceptisol* distribution is shown in Fig. 1.

CORRELATION

The major categories (suborders) of *Inceptisols* are *Aquepts* (aquic conditions), *Gelepts* (frozen), *Cryepts* (cold, short growing season), *Ustepts* (alternating moist and dry with moist summers or no defined winter season), *Xerepts* (alternating moist and dry with dry summers and moist winters), and *Udepts* (moist in all seasons). Table 2 correlates Soil Taxonomy suborders^[1,2] with World Reference Base reference groups.^[4]

Suborders have been established because of major differences in climatic factors that affect soil genesis.

AQUEPTS

The *Aquepts* are the suborder of *Inceptisols* that formed with near-surface redoximorphic features (*aquic conditions*) under extended saturated conditions during biologically active seasons. Some have a histic epipedon and some have accumulated iron sulfate or sodium in the upper 50 cm. *Aquepts* occur in low-lying coastal areas, glacial lake plains, floodplains, playas, and in other nearly level areas where shallow groundwater or perched water exists. Major areas are sub-Arctic areas, humid areas that

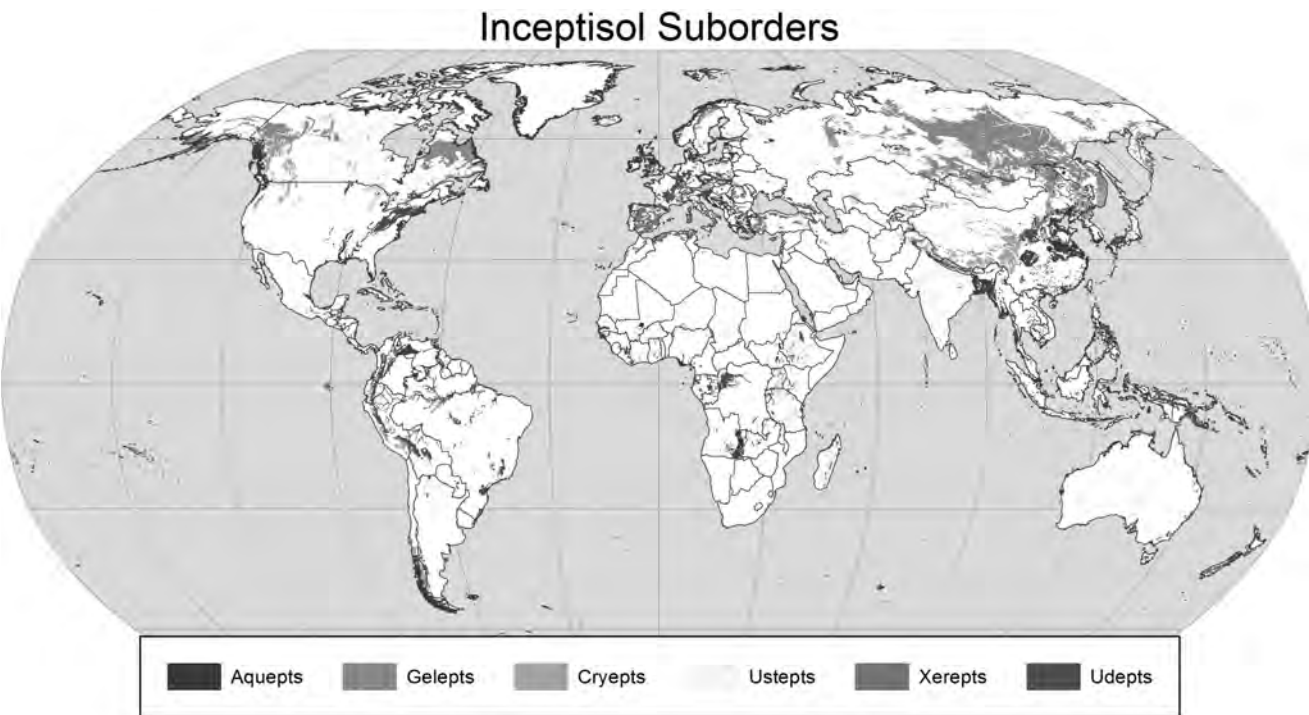


Fig. 1 Distribution of *Inceptisols* (lower 48 states) (2015).
Source: From USDA-Natural Resources Conservation Service, Soil Survey Division, National Soil Survey Center, National Geospatial Data Center, and World Soil Resources.^[3]

Table 2 Correlation of *Inceptisol* suborders with WRB reference groups.

WRB reference group	Inceptisol suborders
Gleysols, Stagnosols	<i>Aquepts</i>
Gleysols, Stagnosols, Cambisols, Umbrisols	<i>Gelepts</i>
Gleysols, Stagnosols, Cambisols, Umbrisols	<i>Cryepts</i>
Cambisols	<i>Ustepts</i>
Cambisols, Durisols, Calcisols, Gypsisols	<i>Xerepts</i>
Cambisols, Umbrisols	<i>Udepts</i>

Source: From Soil Survey Staff.^[1] ©1999 U.S. Govt. Printing Office and IUSS Working Group WRB.^[4] ©2014 FAO.

were covered with the most recent glaciation, wet floodplains and lake plains such as eastern Angola and western Venezuela, irrigated paddies of Eastern China, and wet, floodplains and flooded terraces of major rivers including the Mississippi and Ganges–Brahmaputra.^[1]

GELEPTS

The *Gelepts* are the *Inceptisols* that have a mean annual soil temperature at or below 0°C at 50 cm depth (*gelic soil temperature regime*). They occur on any slope and landform. *Gelepts* also either do not have *permafrost* within 100 cm or do not have both *permafrost* within 200 cm and evidence of cryoturbation (*gelic materials*) within 100 cm. They occur where *permafrost* has thawed or where relief and cold temperatures prevent advanced soil development. These soils are found in central Alaska to west central Canada, east central Canada, in high elevations of Scandinavia, in central, eastern, and southeastern Siberia, northern Mongolia, and in Patagonia.^[1]

CRYEPTS

Cryepts are the *Inceptisols* with persistently cold climates' mean annual soil temperature between 0°C and 8°C at 50 cm depth (*cryic soil temperature regime*), but they do not have *permafrost* within 1 m. They occur on any landform in northern latitudes and in high mountains around the globe where soil development is limited by temperature, parent material or relief, and the soil temperature regimes are *cryic* but not as cold as *gelic*.^[1]

USTEPTS

The *Ustepts* are the suborder of *Inceptisols* that receive most of their precipitation in the summer when

evapotranspiration is highest or they do not have a defined winter (*ustic soil moisture regime*). Native vegetation is typically grassland, shrubland, or savanna. They occur on any freely drained landform and that are not persistently cold. Major areas of *Ustepts* occur on major river floodplains, terraces, glacial till plains, hills, and foothills.^[1] *Ustepts* form where precipitation is sufficient to cause weathering and leaching but is insufficient to completely leach the by-products. The effectiveness of the precipitation is severely limited by either slope or high evapotranspiration or both. Cemented layers or bedrock prevents deep leaching in some of these soils. They occur largely in Central America, parts of the Andes, Europe, East Africa, and India.

XEREPTS

The *Xerepts* are the suborder of *Inceptisols* that are moist in the winter and dry in the summer (*xeric soil moisture regime*). They receive most of their precipitation in the winter season when evapotranspiration is lowest, but experience a summer moisture deficit in most years when evapotranspiration is highest (Mediterranean climate). Native vegetation ranges from grassland or chaparral to forest. They occur on any freely drained landform if the temperature is not persistently cold or hot. Major areas of *Xerepts* occur in mid-latitude coastal areas and nearby zones with Mediterranean climate.^[1] *Xerepts* occur in areas where precipitation is sufficient to cause weathering and leaching, but is insufficient to completely leach the by-products in some areas. The effectiveness of the precipitation is enhanced by low evapotranspiration rates in winter. These soils are dry in summer when evapotranspiration rates are highest. They occur largely in northwest America and Africa and Southern Europe to Turkey.

UDEPTS

The *Udepts* are the suborder of *Inceptisols* that do not experience more than short periods of moisture deficit in the growing season for many years (*udic soil moisture regime*). The moisture is distributed evenly enough to keep up with evapotranspiration losses most of the time. Native vegetation is typically forest. *Udepts* occur on freely drained landforms in climates where the temperature is not persistently cold. Fig. 2 shows an acidic *Udept* from the humid mountains of the Eastern United States. *Udepts* occur in areas where precipitation is sufficient to cause abundant weathering and leaching, unless steep slopes decrease the effectiveness of the moisture or some cemented or dense subsurface horizon or feature restricts deep leaching of the weathering by-products. They occur in humid area floodplains and in steep areas in humid hills and mountains in the Eastern



Fig. 2 An *Inceptisol* profile as shown on the poster of the 12 soil orders of Soil Taxonomy.

Source: From Soil Survey Staff.^[5]

and Northeastern United States, Central Europe, Andes, Himalayas, Southeastern Asia, and the islands of New Guinea and New Zealand.^[1]

SOIL GENESIS

Organic matter accumulates in the surface if the water tables stay very near the surface round the year, and mineralization of organic matter is restricted, forming organic or humus-rich mineral surfaces (*histic*, *mollic*, or *umbric epipedons*). Some cool, high-elevation *Inceptisols* have thick surface deposits of organic litter that is not saturated (*folistic epipedon*). Some *Inceptisols* in marine estuaries or dry climates have accumulated sodium and other soluble salts in the surface (*salic horizon*) through upward movement of an element-rich or saline water table. Evapotranspiration removes water but leaves precipitated salts behind as the water table recedes. In dry climates, incomplete leaching leaves subsoil water enriched in soluble

compounds that precipitate and cement the soil fabric with silica (*duripans*), gypsum (*gypsic* or *petrogypsic horizons*), or lime (*calcic* or *petrocalcic horizons*). Sandy parent materials may accumulate a significant number of very thin bands (*lamellae*) of layered clay in the subsoil where the soil water resides and dries. In cool and humid areas, compounds with organic carbon and either aluminum or iron may accumulate in the acidic subsoil (*spodic materials*), and thin (<10 cm) *spodic* horizons may occur. *Cambic horizons* form from incomplete accumulation of these materials in textures finer than very fine sands and loamy very fine sands.

Inceptisols with *aquic conditions* are saturated in the upper soil for more than a few consecutive days during seasons of biological activity. Water fills all the pores in a horizon when rainfall exceeds evapotranspiration, and either some subsoil horizon or layer restricts downward movement, groundwater rises in the soil, or porewater moves very slowly through a horizon. In tidal marshes or areas where reduced parent materials high in sulfides (*sulfidic materials*) are exposed to oxygen by mining or artificial drainage, a *sulfuric horizon* quickly forms and pH drops to very low levels. Water- and root-restricting subsoil horizons form in moist climates when restricted leaching leaves subsoil water enriched in soluble iron compounds that precipitate and cement the soil fabric to form continuous *plinthite* or *petroferrie (ironstone) horizons*. In moist climates, non-cemented, root-restricting subsoils (*fragipans*) may form through internal desiccation or compaction or partial cementation. The persistently cold (*cryic*) soils in northern latitudes or high elevations have a slower rate of soil-forming processes than do warmer soils. On gentle slopes, volume changes from alternating freezing and thawing may cause churning (*cryoturbation*). *Cryic* soils on steep slopes have shallow profiles because of reduced effectiveness of precipitation, increased runoff, and accelerated losses from erosion and mass movement. Volcanic materials ejected into the air, and some other physically and chemically similar materials may weather to form amorphous clays instead of layered clays. Soils rich in amorphous clays have *andic soil properties* such as high surface area, high retention of charged elements and compounds, high water-holding capacity, and very low bulk density. Deep, wide cracks and large volume changes occur in soils that wet and dry often and contain a high amount of layered clay that has high shrink-swell potential. *Inceptisols* form on major floodplains and low terraces that rarely flood or are very young deposits. Some of these soils still contain evidence of sediment deposition from flooding in the subsoil that has not been completely destroyed by soil-forming processes. *Cambic horizons* form from incomplete action of the alteration processes mentioned before.

Extremely old, highly weathered and continuously leached *Inceptisols* have *cambic horizons* that are dominated by low chemical and physical activity clays and

oxides. *Cambic horizons* may also form from removal of gypsum and lime by percolating soil water over time. Often the leached horizon develops soil structure and becomes lighter or brighter in color or accumulates red colors from the release of oxidized iron compounds. *Inceptisols* have formed on some shoulder positions from more developed soils in intensively farmed areas that have had long-term accelerated erosion. These soils may have had diagnostic horizons removed by accelerated erosion because of farming practices and their convex landscape position.

USE AND MANAGEMENT

Gently sloping, deep *Inceptisols* are well suited to use as cropland. Moderately sloping areas that are not terraced are better suited for biofuels, forages, grazing, forest products, or agroforestry because of the erosion hazards. Areas that are very steep or shallow can be used for forest products, rangeland, wildlife habitat, or recreational uses.

Udepts used for cropping or pasture seldom need irrigation to supplement growing season soil moisture. Native areas are used for forest products, recreation, and wildlife habitat. However, irrigation is often needed to grow annual crops on *Ustepts*. Without irrigation, moisture conservation farming practices and terracing are needed for growing forages and drought-tolerant crops. Perennial crops such as biofuels, root crops, and trees are better able to take advantage of stored moisture. *Xerepts* are important soils for growing small grains and other winter annual crops. Summer crops such as grapes, peas, lentils, and mustards are able to use soil moisture stored during the previous winter season. Native areas are used for timber production, grazing, and wildlife habitat. *Ustepts* and *Xerepts* with a shallow cemented root-limiting layer can be mechanically ripped to allow deeper moisture percolation and increase the rooting zone. Note that if the cemented layer is mechanically ripped for agriculture, these soils become *Entisols*. However, ripping is not financially practical unless irrigated specialty crops can be grown.

Acidic *Inceptisols* with low quantity of nutrients (low base saturation) are separated from high base saturation soils in all but the *Aquepts* suborder. These soils occur on uplands and on minor floodplains that drain mostly acidic rocks and soils. They may be used to grow native crops but must be limed to raise the pH to grow most introduced crops, especially where free aluminum is present in the soil solution and aluminum toxicity may occur. These acidic soils are inherently infertile and must be fertilized to sustain crops or forages, provided the soil mineralogy has sufficient capacity to adsorb the added nutrients. Agroforestry and perennial biofuel cropping may be used in combination with organic

fertilizers and lime on some of these soils. Extremely weathered and leached rainforest soils have most of their nutrients tied up in the above- and belowground organic matter and cannot sustain continuous cropping without the addition of fertilizer and other soil amendments.

Inceptisols on major floodplains typically have high quantity of nutrients (high base saturation). These soils are some of the most important cropland soils in the world. They may be used to grow almost any type of vegetation, depending on their texture, climate, and available moisture.

Aquepts and *Udepts* with *sulfuric horizons* have extremely low pH and are not suitable for agriculture. They can be remediated only if the remaining sulfide-bearing materials are prevented from oxidizing, either by plugging drainage ditches or by deeply burying the layer with better soil. *Aquepts* in warm climates have value for raising rice or water-tolerant specialty crops. Those that have been drained can be used for many types of forages, agricultural crops, biofuels, or forest products (timber or pulpwood). Saline and sodic *Inceptisols* must be drained to prevent the saline water table from rising near the surface if they are used for growing crops. Rainfall or non-saline irrigation water can then leach the excess sodium and salts from the surface. These soils are then suitable for growing a wide variety of crops, depending on their climate and texture.

Cryepts are not suited to many crops but may be used for small grains, berries, potatoes, forest products, wildlife habitat, and recreation. The *Cryepts* that are in high latitudes are more suited to cropland because of the long day-lengths than those at high elevations. The *Cryepts* that occur at high elevations are important sources of streams and fresh water aquifers because they typically have high runoff from rain and melting snowfall and decrease evapotranspiration.

Plaggen surfaces have many uses for agriculture because they are fertile and have high organic carbon and water-holding capacity in the surface. *Anthropic* surfaces are chemically suitable for agricultural fields or gardens but may need irrigation. Soils with high accumulation of organic carbon in the surface have better structure and higher water nutrient-holding capacity than other surface soils lower in carbon. Natural fertility is variable.

CONCLUSION

Inceptisols share similarities to all other soil orders. Soil development of *Inceptisols* has been or is limited by one or more of the five soil-forming factors (parent material, climate, relief, organisms, and time). They are mineral soils more developed than *Entisols*, can support the weight of humans and most livestock, and have diagnostic surface or subsurface horizons that are not too sandy to have continued development potential. However, *Inceptisols* either do not have the diagnostic horizons, materials,

permafrost, or other features of the more developed soil orders that precede them in *The Keys to Soil Taxonomy*. *Inceptisols* can form in any type of parent material except thick organic deposits. *Inceptisols* may occur in any *soil moisture regime* other than *aridic* or *torric* and in any *soil temperature regime*. *Inceptisols* are concentrated in four major settings—cold regions, dry regions (*ustic* and *xeric*), wetlands, and young geological surfaces (such as floodplains, volcanic and glacial deposits, and steep or highly erosive slopes). The area of *Inceptisols* is expected to increase in cold regions where *permafrost* is melting. *Inceptisols* are common in Europe and Central America. The major categories (suborders) of *Inceptisols* are *Aquepts* (aquic conditions), *Gelepts* (frozen), *Cryepts* (cold), *Ustepts* (alternating moist and dry with moist summers or no defined winter season), *Xerepts* (alternating moist and dry with dry summers and moist winters), and *Udepts* (moist in all seasons). They are considered important farming soils in level areas and highly terraced hillslopes.

REFERENCES

1. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; USDA–Natural Resources Conservation Service Agricultural Handbook 436; U.S. Govt. Printing Office: Washington, DC, 1999.
2. Soil Survey Staff. *Keys to Soil Taxonomy*, 12th Ed.; U.S. Govt. Printing Office: Washington, DC, 2014.
3. USDA–Natural Resources Conservation Service, Soil Survey Division, National Soil Survey Center, National Geospatial Data Center, and World Soil Resources. Lincoln, NE, Morgantown, WV, and Beltsville, MD, 2015.
4. IUSS Working Group WRB. *World Reference Base for Soil Resources 2014, International Soil Classification System for Naming Soils and Creating Legends for Soil Maps*. World Soil Resources Reports No. 106; FAO: Rome, 2014.
5. Soil Survey Staff. *The Twelve Soil Orders of Soil Taxonomy*. Available at http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/home/?cid=nrcs142p2_053588 (accessed June 2015).

India: Problems and Potentialities

S.K. Singh

S. Chattaraj

N.G. Patil

S.K. Ray

S. Chatterji

National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), Nagpur, India

Abstract

Soils of India have been categorized into eight groups as per traditional system. A project undertaken by ICAR-NBSS&LUP on Soil Resource Mapping (SRM) of the country adopted the physiographic heterogeneity as the basis for identifying nature of soils, their constraints, potentials, and broad order level classification. The project reported the occurrence of seven soil orders which are discussed with respect to their characteristics, constraints, potentials, and land use aspects.

INTRODUCTION

Precise knowledge of nature of soils (limitations and potentialities) and their spatial distribution is one of the most important prerequisites for proper land use planning that could ensure sustainable agricultural production.

The Indian Council of Agricultural Research (ICAR) in 1953 categorized Indian soils into the following eight major groups, namely: 1) alluvial soils; 2) black soils; 3) red soils; 4) laterite and lateritic soils; 5) forest and mountain soils; 6) arid and desert soils; 7) saline and alkaline soils; and 8) peaty and marshy soils. This has been a traditional system of classification of Indian soils widely followed in India.

The Soil Resource Mapping (SRM) project of the country undertaken by the ICAR-National Bureau of Soil Survey and Land Use Planning on 1:1 million scale (during the period 1986–1996) was based on the identification of relationship between landform (physiography) and soil.^[1]

SOIL–PHYSIOGRAPHY RELATIONSHIP

Soils of the identified physiographic units are discussed below with respect to their constraints and potentialities.

Hill Ranges (Ghats)

In hill ranges, 24.1% is moderately to very steeply sloping; 16.2% soils of the hill ghats are shallow to extremely shallow as against 48.6% deep to very deep soils; 62.6% soils are moderately well drained to well drained, and

24.8% are somewhat excessively drained to excessively drained; 43.5% soils are clayey, whereas 22% are loamy; 77.7% soils are severely to very severely eroded; 16.9% soils are affected by slight sodicity. The major soil orders are Inceptisols, Mollisols, and Ultisols.

Central Uplands

In central uplands 4.5% soils are moderately steep to very steeply sloping; 15.1% soils are shallow to extremely shallow as against 61.9% deep to very deep soils; 83.0% soils are moderately well drained to well drained, and 10.8% soils are somewhat excessively drained to excessively drained; 47.6% soils are clayey, whereas 35.4% are loamy and 2.4% are sandy; 79.4% area is severely to very severely eroded; 5.5% soils are moderately to moderately strongly saline; 74.1% soils are slightly sodic, whereas 1.1% soils are affected by strong to very severe sodicity. Inceptisols, Vertisols, and Entisols are the major soils (in this order). Workability and drainage are the main challenges in managing Vertisols.

Indo-Gangetic Plains (IGP)

In IGP, 99.2% soils of the IGP are deep to very deep soils; 67.5% soils are well drained to moderately well drained, and 28.6% soils are imperfectly to very poorly drained; 76.5% soils are loamy, whereas 14.1% are clayey; 12.3% soils are severely to very severely eroded; 14.6% soils are moderate to moderately strong in salinity; 10.2% soils are affected by moderate sodicity, and 11.8% are very severe to

strongly sodic. Dominant soils are Inceptisols, Alfisols, and Entisols. Soils under arid and semiarid parts of the IGP lack organic carbon due to high rate of decomposition. Imbalanced use of fertilizer is one of the major problems in this region. Deficiencies of nutrients occur in different combinations such as phosphorus (P) and zinc (Zn); and nitrogen and P. Introduction of irrigation without the provision of drainage led to soil salinization and alkalization within a few years in this region due to rise in the groundwater table containing high proportion of sodium. Relic salinity (legacy of the Tethys Ocean) and intensive irrigation in semiarid climate have combined to cause salinity. Excessive tillage and wet tillage (puddling), use of rotavators, and freewheeling of tractors and harvesters (combined) in the IGP have, in general, resulted in gradual compaction of soils, leading to a reduction in long-term soil productivity, especially under rice–wheat cropping system. However, subregion-specific strategies of soil management could arrest the stagnation of yields that are being reported widely. Balanced use of fertilizers alone could achieve significant increase in yield, especially in eastern IGP where fertilizer usage is very low.

The fact that the fertilizer was the key input in augmenting food grain production (after the availability of the seeds of high-yielding crop varieties) is evident from the increase in fertilizer ($N_2 + P_2O_5 + K_2O$) consumption from 1.98 Mt in 1969–1970 to 18.07 Mt by 1999–2000. Nevertheless, the average annual increase in fertilizer consumption witnessed a declining trend in the following three decades, as reflected by 16.5% during 1970s, 12.04% during 1980s, and only 5.6% during 1990s.^[2] A simple regression analysis between the food grain production and fertilizer consumption between 1960–1961 and 1999–2000 showed that the partial factor productivity of fertilizers has been continuously declining.^[2] The data available from some centers under the Indian Institute of Farming Systems Research, Modipuram, also indicate a reduction in crop response to fertilizer applications, especially when balanced fertilization is not practiced.

About 67.1% soils are moderately well drained to well drained, whereas 27.4% soils are imperfectly to poorly drained; 55.1% soils are clayey; 49.9% soils are severely eroded, whereas 48.3% are moderately eroded, 4.2% soils are moderately to strongly saline, and 2.9% soils have moderate to moderately strong salinity; 17.8% soils are slightly sodic. Dominant soils are Inceptisols and Alfisols. Acid sulfate and potential acid sulfate soils are reported to occur in the Sunderbans, 24 Parganas (South), West Bengal, and Kuttanad, the Rice Bowl of Kerala, which adversely affect crop yield.^[3,4] These soils develop as a result of the drainage of parent materials that are rich in pyrite (FeS_2). The acid sulfate soils have the problems of extreme acidity ($pH < 4.0$) and have toxic quantities of iron (Fe), aluminum (Al), and sulfur (S) which usually limit the crop choice and productivity. Potential acid sulfate soils are poorly drained soils with a high content of FeS_2 . The pH of the soils would

be neutral or slightly acidic in the field. Upon drainage, the soil becomes strongly acidic, which directly affects the growth of plants as a result of Al and Fe toxicity and indirectly decreases the availability of phosphorus and other nutrients. The other constraints are poor contents of major nutrients, low base status, and hydrogen sulfide toxicity.

The Islands

In islands, 31.3% soils are very steeply sloping; 6.9% soils of the islands are shallow to extremely shallow as against 80.9% moderately shallow to moderately deep soils and 12.1% deep to very deep soils; 67.8% soils are moderately well drained to well drained, and 24.4% are somewhat excessively drained to excessively drained; 89.8% soils are loamy, whereas 6.2% are sandy; 84.6% soils are severely to very severely eroded, whereas 15.4% soils are affected by moderate erosion; 3.8% areas are strongly saline. Dominant soils are Inceptisols and Entisols. Mollisols also have significant presence in the islands.

Himalayan Mountains and Siwaliks

In Himalayan mountains and Siwaliks, 37.9% area is steeply to moderately steep sloping, and 14.4% area is moderately to gently sloping; 12.2% soils of the Himalayan region are shallow to extremely shallow as against 22.7% deep to very deep soils; 31.5% soils are excessively to somewhat excessively drained, and 21.8% soils are well drained to moderately well drained; 25.7% soils are loamy, whereas 6.5% are sandy; 50.2% area is severe to very strongly severe, and 3.1% area is moderately eroded; 9.7% soils are slightly sodic.

Northeastern Hill Ranges and Valleys

In northeastern hill ranges and valleys, 12.3% soils of the northeastern region are moderately deep against 86.6% deep to very deep soils; 46% soils are moderately well drained to well drained, and 36% soils are somewhat excessively drained to excessively drained; 57% soils are loamy, whereas 32.8% soils are clayey; 32.2% soils are moderately steep to very steeply sloping; 73.9% soils are affected by severe to very severe erosion, whereas 20.5% soils are moderately eroded. Only 0.7% soils are moderate to moderately strong in salinity, and 0.7% soils are strongly to very strongly sodic. Dominant soils are Ultisols, Alfisols, and Inceptisols. The northeastern hill region represents 55% of acid soils in India, despite contributing only 8% of the total geographic area. Toxicity of Al, deficiency of bases [e.g., calcium, magnesium, and potassium (K)], and low availability of phosphorous are the other major problems faced in managing these soils.

Western Plains

The region is dominantly covered with high and low dunes; 23.9% soils of the Western Plains are moderately shallow to moderately deep as against 71.3% deep to very deep soils; 57.8% soils are somewhat excessively drained to excessively drained; 57% soils are sandy, whereas 39.7% are loamy; 97.6% soils are strongly to very strongly saline, 8.8% soils are moderately to moderately strongly saline, whereas 4.8% soils are affected by sodicity. Dominant soils are Aridisols and Entisols.

Deccan Plateau

In Deccan Plateau, 29.1% soils are shallow to extremely shallow as against 62.3% deep to very deep soils; 17.5% soils are somewhat excessively drained to excessively drained; 53.3% soils are clayey, whereas 28.9% soils are loamy; 84.3% soils are severely to very severely eroded, whereas 9.9% soils are affected by moderate erosion; 3.7% soils are affected by moderate to moderately strong salinity; 1.5% soils are moderately sodic. Dominant soils are Inceptisols, Vertisols, and Entisols.

Eastern Plateau

In Eastern Plateau, 66.6% area is gently to moderately sloping, whereas 28.3% is level to very gently sloping; 21.6% soils of the Eastern Plateau are moderately shallow to moderately deep, and 21.4% soils are shallow to extremely shallow as against 56.9% deep to very deep soils; 70.8% soils are moderately well drained to well drained, 15.4% soils are somewhat

excessively drained to excessively drained, and 13.8% soils are very poorly to imperfectly drained; 59% soils are loamy, whereas 33.3% are clayey; 85.7% soils are severely to very severely eroded, whereas 14.3% are moderately eroded soils; 56.6% soils are slightly to moderately sodic. Alfisols, Inceptisols, and Entisols are the dominant soils; 85.7% soils are affected by severe to very severe erosion, whereas 14.3% are moderately eroded; 56.6% soils are affected by slight salinity, whereas 56.6% soils are affected by slight to moderate sodicity. Alfisols, Inceptisols, and Entisols are the dominant soils.

Gujarat Plain

In Gujarat Plain, 28.6% area is gently to moderately sloping; 20.3% soils of Gujarat Plain are shallow to extremely shallow as against 47.5% deep to very deep soils. 81.3% soils are moderately well drained to well drained, and 10% are somewhat excessively drained to excessively drained; 51.8% soils are clayey, whereas 41.8% soils are loamy; 61.5% soils are affected by severe to very severe erosion, whereas 32.7% soils are moderately eroded; 15% soils are affected by strong to very severe salinity problem, whereas 24.8% are moderate to moderately strong in salinity; 19% soils are moderately sodic, whereas 8.3% soils are strong to very strongly sodic. Dominant soils are Inceptisols, Aridisols, and Vertisols.

The physiographic unit-wise soil classification reported the occurrence of seven soil orders, namely, Inceptisols, Entisols, Alfisols, Vertisols, Aridisols, Mollisols, and Ultisols as discussed above. Area (%) of soils under different soil orders is given in Fig. 1, their geographical distribution in the country is presented in Table 1 (across the federal

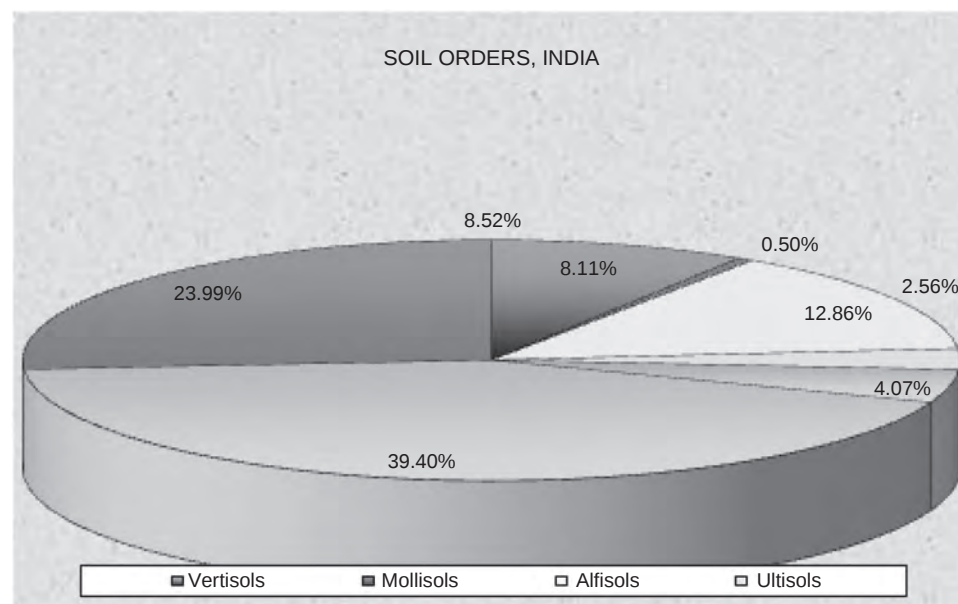


Fig. 1 Distribution of various soil orders in India.

Source: From Bhattacharyya, Sarkar, et al.^[5] ©2009 National Bureau of Soil Survey and Land Use Planning.

Table 1 Spatial distribution and extent of soil orders in federal states of India.

State	Area under soil order (ha)						
	Alfisols	Aridisols	Entisols	Inceptisols	Mollisols	Ultisols	Vertisols
Delhi	—	—	113,547	253,395	—	—	—
Haryana	20,816	899,732	2,105,378	3,501,473	—	—	—
Himachal Pradesh	—	—	2,859,942	1,250,232	34,825	—	—
Jammu & Kashmir	71,947	—	3,573,927	2,104,453	105,842	—	—
Punjab	234,379	674,460	781,479	2,501,167	—	—	—
Uttar Pradesh	1,214,576	—	1,676,391	15,945,120	86,704	—	180,203
Uttarakhand ^a	2,011	—	2,693,791	1,180,628	6,717	—	—
Andhra Pradesh	7,598,708	616,260	1,994,870	8,289,787	131,999	19,451	2,559,966
Karnataka	6,153,419	1,185,385	3,723,287	4,702,591	20,463	1,872,688	4,505,983
Kerala	91,414	—	276,369	1,358,470	70,892	2,194,052	—
Pondicherry	—	—	—	—	—	—	288
Tamil Nadu	4,027,814	—	373,411	5,725,832	157,615	5,921	1,191,201
Bihar	1,919,660	—	4,224,428	4,994,381	—	—	—
Jharkhand	3,847,712	—	1,368,035	1,444,990	—	—	—
Odisha	6,900,548	—	1,931,082	8,240,613	399,771	653,519	—
West Bengal	1,535,436	—	1,528,336	3,779,556	2,980	—	—
Sikkim	—	—	188,860	308,117	74,451	—	—
Rajasthan	61,824	6,981,022	12,036,754	7,379,462	—	10,298	521,728
Gujarat	272,364	1,527,588	2,846,549	9,809,859	—	7,120	1,733,892
Central Zone	—	—	—	—	—	—	—
Maharashtra	695,153	—	11,088,138	7,864,557	18,162	—	7,779,101
Madhya Pradesh	3,436,398	—	5,772,293	13,028,097	50,678	8,904	8,256,128
Chhattisgarh ^b	5,906,476	—	1,748,217	4,255,062	123,064	—	2,388,091
Assam	972,065	2,338,475	—	3,375,338	74,054	1,038,331	—
Arunachal Pradesh	10,492	—	3,839,505	2,436,666	—	1,700,107	—
Manipur	17,254	—	371,862	1,111,151	32,954	985,202	—
Meghalaya	114,340	—	147,484	516,865	—	588,493	—
Mizoram	73,906	—	332,112	855,013	—	539,251	—
Nagaland	4,083	—	26,649	618,747	—	107,317	—
Tripura	46,371	—	51,250	688,102	—	133,299	—
Total	45,229,165	14,222,922	67,673,945	117,519,727	1,391,171	9,863,952	29,116,581

Note: ^aPart of Uttar Pradesh earlier. ^bPart of Madhya Pradesh earlier.

Source: From Bhattacharyya, Sarkar, et al.^[5] ©2009 National Bureau of Soil Survey and Land Use Planning.

states), and Fig. 2 presents the spatial distribution of soil orders in the country. We also describe soil order-wise characteristics, problems, and potentials to have a better insight into nature of soils of India.

Inceptisols

Vertic intergrades in Madhya Pradesh (MP), Gujarat, and Andhra Pradesh (AP) support crops like soybean, cotton, and groundnut. Paddy–wheat system is dominant in loamy textured soils in Uttar Pradesh (UP) and other IGP regions of the country. More than 42% of the area under these soils

is degraded due to water erosion. Management of these soils is critical to the food safety of India because agricultural production from these soils in UP and other IGP regions has served as food granary to the nation since the Green Revolution. Higher yields could be achieved on loamy and fertile soils with the use of high-yielding varieties, higher irrigation, fertilizers, and improved soil management practices apart from socioeconomic and external factors. These are the best agricultural soils and are used for growing most agricultural crops, especially wheat, rice, maize, groundnut, and berseem. Because of the injudicious use of irrigation water, some such soils have

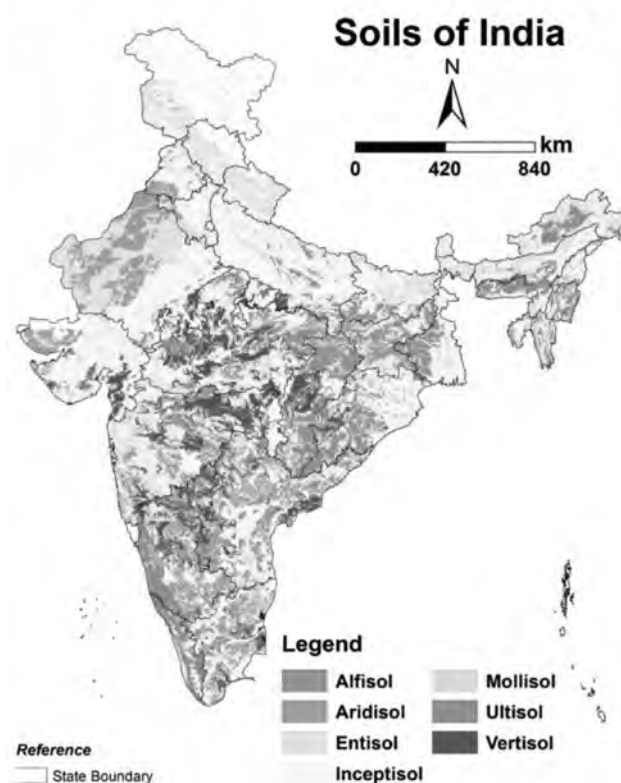


Fig. 2 Spatial distribution of various soil orders in soils of India.
Source: From ICAR-NBSS&LUP.^[1] ©2002 National Bureau of Soil Survey and Land Use Planning.

been rendered waterlogged and pose problems of soil salinity and/or sodicity mainly in Punjab, Haryana, and UP and thereby limit choice of crops.

The yields of crops have stagnated due to high degree of physical and chemical degradation of land, nutrient deficiencies and imbalances, depleting groundwater level, and pollution of soil and groundwater by nitrogen, phosphorus, and heavy metals.^[6] Buildup of salinity and/or sodicity, increased bulk density due to continuous puddling (rice crop), and excessive groundwater extraction are the major problems. In the semiarid tropics, erosion of shallow soils on the one hand and impeded drainage on the other are constraints of crop production.

Acidity is another major problem in Inceptisols occurring in all the states of the country except in Haryana, Punjab, Rajasthan, Gujarat, and Delhi. In general, soils are deficient in nitrogen, phosphorus, and humus but are well supplied with K and require judicious application of manures and fertilizers for sustainable farming. As a result of intensive cultivation, these soils exhibit second-generation problems, viz. nutrient deficiencies of S and Zn, receding and rise of groundwater table (as in central Punjab), and causing salinization in other areas adjoining Punjab and Haryana states. Inceptisols developed on coastal alluvium are found along seacoast of Karnataka, Kerala, Tamil Nadu,

AP, Odisha, West Bengal, and Goa. Stretches of coastal sands or sandy soils occur prominently in the coastal areas of Thanjavur district of Tamil Nadu, Bapatla in Guntur district of AP, and Puri district in Odisha where salinity and waterable do not pose problems and the sandy patches are put under plantations of coconut, cashew, and casuarinas.^[7]

Peaty soils (subgroup) occur in humid regions, have accumulation of high organic matter, and contain considerable amount of soluble salts. These soils are generally blue due to the presence of ferrous iron under anaerobic conditions and contain varying amounts of organic matter. Marshy soils of this type occur in coastal tract of Odisha, the Sundarban area of West Bengal, the central portion of North Bengal, and the southeast coast of Tamil Nadu.^[7]

Entisols

Entisols are also well spread (in association with Inceptisols) with representation in each federal state wherever there are depositions of alluvium, particularly along the riverbank and very shallow soils on hilltops, side slopes, plateaus, and eroded surfaces. These soils are spread over 67.67-million hectare area. Entisols, although reported from different parts of the country, are strikingly different in the Deccan basalt area compared to those in the IGP and/or coastal plains. More than 66% soils in this order are susceptible to water erosion, whereas almost 18% are estimated to be susceptible to wind erosion. The Entisols (Lithic Ustorthents) in the hilly forests developed from basalts are shallow, smectitic, and black in colour, whereas those in the IGP are very deep and often loamy to sandy/silty in texture grown mainly for paddy, wheat, and maize. Entisols in the coastal areas and deserts are Psammments containing sand throughout the soil depth. Acid sulfate soils discussed in detail under Coastal Plains belong to Entisol (Sulfa Aquent). These soils occupy second position (25%) after Inceptisols in India. Western states of Rajasthan and Maharashtra with semiarid climate have the highest coverage of this soil order. Crops like pearl millet, sorghum, and mustard are raised on these soils. Predominantly rainfed agriculture in these states leads to very low agricultural productivity. Irrigation development in many parts of these states has led to development of salinity. In high-rainfall zones, paddy is the only crop grown in these soils. Watershed management programs in this part of India have demonstrated how effectively the resources can be utilized for sustaining productivity.

Alfisols

Alfisols are observed mainly in central, eastern, and Peninsular India (area: 45.22 million hectares). It is the third largest soil order of India occupying more than 16% of area and spread well across the length and breadth of the country. Alfisols are of varied types, and those from

high-rainfall regions are acidic and poor in nutrients. Forest soils of the Himalayas,^[8] Western and Eastern Ghats, Kerala, and Mysore Plateau have high organic matter. It has been observed that the Alfisols are more acidic and have more clay and kaolin than the Mollisols. Occurrence of such soils suggests that the Alfisols (under sparse forest) are more weathered than Mollisols due to better water influx in the former. In many Alfisols of the IGP, clay increases with depth. The ratio of clay content in the clay-rich Bt horizon to that of the A horizon is >1.2 , which suggests that these soils are fairly well developed. These soils develop on stable landscapes with the concomitant and simultaneous erosion (degradation) and deposition (aggradation). There are cases where the successive horizons even in a restricted part of soil profile result from progressive alteration *in situ*, especially in subtropical regions of India.^[9]

Aridisols

Most of the Aridisols (14.22 million hectares) are located in western part of the country comprising the Great Indian Desert and the Kutch region. Aridisols are known to form under limited water availability wherein plant growth, leaching of salts, and soil formation are restricted. In India, Aridisols are largely confined to the vast contiguous Indian Desert in Rajasthan (58%) and Gujarat (12%) and to small parts of Punjab and Haryana. However, peninsular Indian states of AP and Karnataka together constitute more than 16% of area under Aridisols. The soils suffer from different kinds of degradation like wind erosion, salinity, sodicity, and pan formation such as petrocalcic and petrogypsic.

A major portion of the region consists of sand dunes and undulating sandy plains where water is a major constraint. The soils, in general, are capable of retaining only a limited amount of water and as such are droughty and prone to deep percolation losses of water received as rainfall or from canal irrigation. The fertility status is poor especially with respect to nitrogen and phosphorus due to high temperature and precipitation of applied phosphatic fertilizers as monocalcium phosphate, dicalcium phosphate, and tricalcium phosphate. Once irrigated and managed well, these soils have high potential for growing two crops in a year.

Mollisols

In contrast to the generally observed non-acidic and less weathered Mollisols in temperate semiarid and humid climate, acidic and fairly weathered Mollisols on Deccan basalt occur in hills of the Satpura Range of MP and the Western Ghats of Maharashtra under forest cover in the prevailing tropical humid climatic conditions.^[10] High rainfall and resultant water erosion degrade these soils. The formation of Mollisols in the zeolitized basaltic landscape over millions of years and their persistence in central and western India demonstrate that the quality of parent materials prevents the transformation of smectite to kaolin, helps in

the retention of adequate amount of smectite, and provides continuous supply of bases (Ca^{2+} ions) required for the formation of Mollisols even under tropical humid climate. These soils occupy the last position with less than 1% share in the country.^[11]

Ultisols

Ultisols are concentrated in the high-rainfall areas of the northeastern region and Kerala. Ultisols are found in the humid tropical areas of Kerala, southernmost tip of Tamil Nadu, Karnataka, and all the northeastern states where sub-surface kandic horizon is observed due to persistent high rainfall and temperature and a leaching environment. These soils are highly acidic in nature, poor in bases and major and minor nutrients, and prone to different forms of erosion. By virtue of being generally under forests and plantation crops, these soils are rich in organic matter. Eroded forms of these soils deposited in valleys are generally grown for paddy with the help of high-input management.^[7]

Ultisols in the humid tropical climate are dominantly clayey, and clay skins are difficult to be identified in the field in the northeastern part of India. However, the soils in the Western Ghats show evidence of translocation and accumulation of clay in the form of clay skins.^[11] Ultisols in Arunachal Pradesh show uniformity of parent materials. The clay distribution as a function of depth indicates that these soils are fairly well developed.

The soils are generally deficient in nitrogen and phosphorus but are well supplied with K. They are high in acid-soluble ferric oxide as compared to aluminum oxide. They are poor in organic matter and lime contents. These soils are cultivated intensively for a large variety of crops, including paddy, tea, coffee, cardamom, and cocoa, using heavy doses of organic manures and artificial fertilizers under irrigation conditions. Shifting agriculture is practiced in such soils.

Vertisols

Vertisols in India are a very important group of soils as the extent of their occurrence is just next to Australia and upon its proper management depends on the crop productivity in major part of semiarid tropical India. In India, the majority of the Vertisols are developed in the alluvium of weathering Deccan basalt in the states of Maharashtra, MP, Chhattisgarh, AP, Telangana, Karnataka, Gujarat, and Rajasthan. Additionally, outside the Deccan basalt region of the Peninsula, Vertisols are present in small patches in the states of Bihar, Odisha, West Bengal, and Kerala. Almost 65% of area under Vertisols is estimated to be prone to water erosion and other forms of degradation.

The arid and semiarid regions of India are dominated by Vertisols and their intergrades. About 80% of these soils form typical soils in central Peninsula. These soils are found to be spatially associated with red ferruginous soils. These

two soils occur in toposequence with red soils on crests and pediments and the black soils on foot slopes and in valleys and depressions. However, occurrence of such soils in close association under almost similar topographical situations is also reported.^[11] Vertisols are reported in the IGP.^[10]

The major constraints to crop production in Vertisols are their low workability. Sodidity is also a major productivity constraint arising out of low infiltration, poor drainage, and occasional moisture stress. These soils have high cation exchange capacity (30–50 [cmol(P+) kg⁻¹]), high base status, and high contents of exchangeable calcium and magnesium. They are, however, poor in organic matter, nitrogen, sulfur, and available phosphorus. These soils are predominantly used for growing cotton, soybean, maize, pigeon pea, millet, and sorghum in the states of MP, AP, Gujarat, and Maharashtra.^[7]

CONCLUSION

The SRM project undertaken on 1:1 million scale generated a rational and reliable soil information system of the country compatible with various soil information systems existing globally. This facilitated identifying soil characteristics including their problems and potentials. The soil information system so developed becomes a reliable component for recommending crop advisories which, if integrated with climatic and socioeconomic components, could ensure scientific land use planning.

REFERENCES

1. Staff, NBSS&LUP. *Soils of India*. NBSS Pub. 94; National Bureau of Soil Survey and Land Use Planning: Nagpur, 2002. 130+11 pp sheet maps.
2. NAAS. *Low and Declining Crop Response to Fertilizers*. Policy Paper No. 35; National Academy of Agricultural Sciences: New Delhi, 2006; 8 pp.
3. Bandyopadhyay, B.K.; Maji, B. Nature of acid soils of Sundarbans delta and suitability of classifying them as acid sulphate or potential and sulphate soils. *J. Indian Soc. Soil Sci.* **1995**, *43*, 251–255.
4. Beena, V.N.I.; Covilakom, M.T.K. Characterization of acidity in acid sulphate soils of Kerala. *J. Life Sci.* **2013**, *7* (8), 907–912.
5. Bhattacharyya, T.; Sarkar, D.; Sehgal, J.L.; Velayutham, M.; Gajbhiye, K.S.; Nagar, A.P.; Nimkhedkar, S.S. *Soil Taxonomic Database of India and the States (1:250,000 scale)*. NBSSLUP Publ. 143; National Bureau of Soil Survey and Land Use Planning: Nagpur, 2009; 266 pp.
6. Ray, S.K.; Bhattacharyya, T.; Reddy, K.R.; Pal, D.K.; Chandran, P.; Tiwary, P.; Mandal, D.K.; Mandal, C.; Prasad, J.; Sarkar, D.; Venugopalan, M.V.; Velmourougane, K.; Sidhu, G.S.; Nair, K.M.; Sahoo, A.K.; Das, T.H.; Singh, R.S.; Srivastava, R.; Sen, T.K.; Chatterji, S.; Patil, N.G.; Obireddy, G.P.; Mahapatra, S.K.; Anil Kumar, K.S.; Das, K.; Reza, S.K.; Dutta, D.; Srinivas, S.; Karthikeyan, K.; Srivastava, A.; Raychaudhuri, M.; Kundu, D.K.; Dongare, V.T.; Balbuddhe, D.; Bansod, N.G.; Wadhai, K.; Lokhande, M.; Kolhe, A.; Kuchankar, H.; Durge, S.L.; Kamble, G.K.; Gaikwad, M.S.; Nimkar, A.M.; Bobade, S.V.; Anantwar, S.G.; Patil, S.; Sahu, V.T.; Sheikh, S.; Dasgupta, D.; Telpande, B.A.; Nimje, A.M.; Likhar, C.; Thakre, S.; Mandal, K.G.; Kar, G.; Gaikwad, K.M.; Bhondwe, H.; Dohetre, S.S.; Gharami, S.; Khapekar, S.G.; Koyal, A.; Sujatha, B.; Reddy, M.N.; Sreekumar, P.; Dutta, D.P.; Gogoi, L.; Parhad, V.N.; Halder, A.S.; Basu, R.; Singh, R.; Jat, B.L.; Oad, D.L.; Ola, N.R.; Hukare, A.; Khuspure, J.; Sunitha, B.P.; Mohanty, B.; Hazarika, D.; Majumdar, S.; Garhwal, R.S.; Sahu, A.; Mahapatra, S.; Puspamitra, S.; Kumar, A.; Gautam, N. Soil and land quality indicators of the Indo-Gangetic Plains of India. *Curr. Sci. India* **2014**, *107* (9), 1470–1486.
7. Sarkar, D.; Haldar, A. *Applied Pedology Today and Tomorrow*; Printers and Publishers: New Delhi, 2014.
8. Panda, N. Acid soils of eastern India – Their chemistry and management. *J. Indian Soc. Soil Sci.* **1987**, *35*, 568–581.
9. Pal, D.K.; Bhattacharyya, T.; Sinha, R.; Srivastava, P.; Dasgupta, A.S.; Chandran, P.; Ray, S.K.; Nimje, A. Clay minerals record from late Quaternary drill cores of the Ganga Plains and their implications for provenance and climate change in the Himalayan foreland. *Palaeogeogr. Palaeoclimatol.* **2012**, *356–357*, 27–37.
10. Bhattacharyya, T.; Pal, D.K.; Mandal, C.; Chandran, P.; Ray, S.K.; Sarkar, D.; Velmourougane, K.; Srivastava, A.; Sidhu, G.S.; Singh, R.S.; Sahoo, A.K.; Dutta, D.; Nair, K.M.; Srivastava, R.; Tiwary, P.; Nagar, A.P.; Nimkhedkar, S.S. Soils of India: Historical perspective, classification and recent advances. *Curr. Sci.* **2013**, *104* (10), 1308–1323.
11. Sehgal, J. *Introductory Pedology (Soil Genesis, Survey and Classification)*, 1st Ed.; Kalyani Publishers: Ludhiana, 1996; 287 pp.

Indigenous Soil Management

Karl Herweg

Centre for Development and Environment (CDE), Institute of Geography,
University of Bern, Bern, Switzerland

Abstract

Former approaches to technology transfer tried to identify a problem and to respond to it. This not only has improved soil management but has also been accompanied by major setbacks in the sense that adoption of alien technologies often ignored socioeconomic aspects and turned out to be costly or difficult to accept at the social level. Approaches, therefore, focus on incremental adaptation of technologies rather than on adoption, i.e., understanding indigenous farming practices to guide sustainable land use development. This does not mean that indigenous knowledge always provides the solution. Rapid changes in the socioeconomic, political, and biophysical environments put indigenous knowledge systems under tremendous pressure and do not give them time to adapt themselves to the changes. But learning about local knowledge, recognizing indigenous practices and their functions, and then trying to improve soil management together with local farmers, researchers, and external advisors means making the best use of all available knowledge systems.

INTRODUCTION

“Indigenous” or “local” knowledge is unique to a given culture or society, in contrast to external knowledge that is generated within the international system of universities, research institutes, and private firms.^[1] However, virtually all indigenous knowledge systems have been exposed to external influence, be it during the eras of colonization or development cooperation, owing to migration and trading and international training of local experts, or through the mass media. What is termed as indigenous knowledge has already incorporated suitable components from external knowledge systems, or, in other words, introduced technologies have been indigenized.^[2] Thus, in this entry, the term “indigenous soil management” refers to agro-pastoral or mixed small-holder farming systems with low external agricultural inputs^[3] rather than to the rare cases that may be absolutely isolated from the outside world.

INTEGRATED SOIL MANAGEMENT

The term “indigenous soil management” indicates a typical scientific approach. In the reality confronted by land users, an equivalent term for soil management as such may not exist, because soil is always integrated in an overall farming system, e.g., to produce food and fodder. Thus, indigenous soil management should be seen in the framework of a land management system.^[4]

Indigenous knowledge systems are highly site specific.^[4,5] They undergo permanent adaptation in response to local

socioeconomic and environmental changes and thus always imply an aspect of innovation. Apart from its site specificity, indigenous soil management system may best be characterized as multifunctional, flexible, and dynamic.^[2,6,7] In [Table 1](#), examples of various indigenous soil management options are given.

Multifunctionality

Each household pursues its own livelihood strategy, and land users apply a range of land management practices to achieve their goals. Each practice may involve the management of soils besides other components such as water, crops, animals, forests, and human resources. Each practice usually serves more than one purpose, e.g., crop production and resource protection.^[6,7]

Flexibility and Dynamism

Indigenous soil and land management usually is flexible and dynamic in several aspects. Each practice responds to one or more specific problems (soil erosion, low production, pests, etc.) at a specific location (steep slope, valley bottom, communal land, etc.), as perceived by a land user with specific experience, capabilities, and options to react. In addition, the positive impact and the side effects of a practice are permanently observed, and the practice is modified, if necessary, over time. This means that, depending on the variations in these factors, a practice will always look different, from one field to another, and from one farmer to another.^[6,7]

Table 1 Examples of indigenous soil management options.

Option	Main purposes	Area (reference source)
Working arrangements (sharecropping, animal fattening)	Overcome labor shortage, optimize timing of farming operations	Ethiopia ^[6,7]
Gradual build-up of terraces	Effective use of labor	Ethiopia ^[2,6,7]
Focusing only on locations with severe erosion	Effective use of labor	Ethiopia ^[6,7]
Joint decision-making among neighbors	Avoid pests as result of individual cropping patterns	Nepal ^[8]
Fallow	Recycling of nutrients, recovering of soil fertility	Africa, ^[2,9] Ethiopia, ^[4] South America, ^[10] Nepal ^[8]
Crop rotation, selection of crops to correspond with soil potential, use of nitrogen-fixing trees	Prevent exploitation of nutrients	Africa, ^[2,9] Ethiopia, ^[4] South America, ^[10] Nepal ^[8]
Manure (from fixed pens, open grazing, compost, crop residues, household waste, leaf litter, weeds)	Improve nutrient level and physical soil properties to provide better aeration, infiltration, water-holding capacity and soil moisture	Africa, ^[2,9] Ethiopia, ^[4] Kenya, ^[5] South America, ^[10] Nepal ^[8]
Mulching	Improve soil fertility, reduce evaporation	South America ^[10]
Soil burning	Improve infiltration and availability of nutrients	Ethiopia ^[6]
Trapping eroded soil	Improve soil fertility	Mexico ^[10]
Application of fertile silt from ponds and rivers	Improve soil fertility	Africa ^[2]
Temporary structures (trash lines)	Collect and redistribute fertile accumulation	Ethiopia ^[2,6,7]
Indigenous terraces and grass and vegetation strips	Collect and redistribute fertile accumulation	Ethiopia ^[2,6,7]
Water harvesting (drainage ditches, gully reclamation)	Conserve water/moisture	Africa, semiarid areas ^[2]
Crescent-shaped or rectangular embankments	Slow down runoff, increase infiltration	Africa, semiarid areas ^[2]
Ridging and infiltration pits (Zaï, tassa)	Break hard topsoil layers, increase infiltration	Africa, semiarid areas ^[2]

INDIGENOUS SOIL CLASSIFICATIONS AND FERTILITY INDICATORS

Indigenous soil classifications reflect land users' perceptions of land and soils, their potential for a specific purpose, and their spatial variability. Such classifications are usually well categorized and based on well-identified criteria, which typically incorporate diverse properties of soil or soil-related factors, e.g., texture, color, water-holding capacity, irrigation requirements, soil softness, drainage, fertility, and crop suitability.^[10] Soils are often classified according to their suitability for growing certain crops, based on vegetative, topographic, and microclimatic indicators.

Soil fertility is considered a dynamic characteristic, not an inherent quality.^[10] In local terms, soil fertility may, for instance, be connected with manure requirements, but interestingly, it is not globally attributed to nutrient availability. For example, water-holding capacity (in semiarid areas) and permeability and drainage of the soils (in subhumid areas) may be considered more important than the overall nutrient status. Also, socioeconomic terms such as "workability" are used to describe a fertile soil, e.g., if the soil can be prepared with a few tillage operations only, if tillage is relatively easy, or if labor requirements are low. Although

indigenous soil classifications and indicators of soil fertility are highly site specific, soil texture and color are the most common factors in indigenous knowledge systems to distinguish soils of different quality.^[10]

Local knowledge of soil–plant–fauna relationships is used to select crop varieties or crop rotations suitable to a specific soil. Soil heterogeneity is acknowledged and advantageously used and not eliminated by fertilization. The choice of crops for each soil is not necessarily unanimous, but depends largely on the individual experience and interest of the land user.^[10] Local plants that indicate soil fertility are as common as the knowledge of plants that are beneficial for improving soil fertility. The range of indigenous biological indicators is wide, e.g., the appearance of mauls in the accumulation of indigenous terraces, in combination with grass production on the terraces, is known to be an indicator of improved soil fertility.^[7]

INDIGENOUS SOIL MANAGEMENT PRACTICES

Continuous farming or animal grazing affects soil in many ways; e.g., monocropping and soil erosion diminish nutrients and organic matter; soil erosion changes soil depth,

texture, structure, and the capacity to store water and nutrients; irrigation in semiarid areas increases the salt content. Indigenous soil management practices try to prevent or minimize degradation processes and focus on managing labor, soil fertility (soil nutrients and organic matter), and both the excess and shortage of water.^[8]

Managing Limited Labor

Wealthy farmers have a greater labor force, more draught power, and a greater range of soil management options compared to relatively poor land users. Limited labor and transport facilities usually result in intensively used land around the homesteads and decreasing intensity toward the most distant plots. Shortage of labor often forces farmers to rely on each other and to make sharecropping, animal fattening, and other arrangements.^[6,7] Decisions are often made together with neighbors or in the community, because individual strategies can lead to severe problems; e.g., a single farmer who decides to plant early will attract all the pests to his fields, or a single field that is left fallow will be a refuge for pests that affect all neighboring plots.^[8] Local societies have a number of working arrangements to overcome labor shortages, resolve conflicts over scarce resources, and use optimum timing for land preparation, weeding, and other farming activities.^[6,7]

Managing Soil Fertility and Soil Water

Probably the most common way of recycling nutrients and reclaiming land after a period of nutrient and water exploitation is to leave it “fallow” until its fertility has recovered. However, increasing population and animal pressure, fragmentation of the land, insecure land tenure, and other factors force land users to shorten their fallow periods and to look for alternatives,^[2,4,8–10] e.g., crop rotation (e.g., cereals and legumes), selection of crops corresponding to the potential of the soil, and the use of nitrogen-fixing trees are common methods for preventing exploitation of nutrients. A number of indigenous practices contribute to improved nutrient levels and physical soil properties that provide better aeration, infiltration, water-holding capacity, and soil moisture; e.g., the organic matter content can be improved by manure, if the herd size is sufficient, if manure can be purchased from pastoralists, or if the manure is not used for other purposes (fuel, construction, to protect seeds from diseases, etc.). Application of manure requires improved weed control. The highest concentration of manure is found around the homesteads (fixed pens). Distant plots can be improved through mobile pens and open grazing. Other sources of organic matter can include incorporation of compost, household waste, leaf litter, weeds (uprooted and burned) in the soil, and crop residues left after harvesting or disseminated as mulch. Again, availability is the limiting factor, as crop residues are frequently used as fodder. It is also common

to carry fertile silt from community ponds back to improve the cultivated fields. Soil burning—together with dung and other organic matter—may improve water infiltration and availability of nutrients on a rather short-term basis.

Particularly in arid and semiarid areas, the shortage of water is the greatest constraint on production. Effective indigenous water harvesting practices include ridging and infiltration pits (zaï, tassa) that break hard layers in the topsoil, crescent or rectangular embankments that allow slow movement of runoff and better infiltration, and mulch that reduces evaporation. Drainage ditches and gully reclamation measures help control erosion in subhumid areas and harvest water in semiarid areas. Indigenous terraces and grass and vegetation strips are used to enforce the accumulation of fertile topsoil and to prevent further decline of soil fertility.^[2] But unlike introduced soil and water conservation (SWC) structures such as soil bunds, indigenous SWC measures are built from material that can best be spared, such as stones or brushes, instead of fertile soil. In addition, indigenous SWC measures rarely cover the entire slope but concentrate on “hot spots,” where severe soil erosion is expected or where soils are particularly shallow (sedimentation management).^[10] Indigenous SWC is often a mixture of temporary structures (e.g., trash lines)—i.e., scheduled for removal to redistribute the fertile accumulation—and permanent structures (e.g., stone bunds serving at the same time as field borders or fences). Probably the greatest difference between introduced and indigenous measures is that the latter are rarely constructed in one go, but are gradually built up in the course of other farming activities such as plowing and weeding.^[2,6,7]

CONCLUSION

Former approaches to technology transfer tried to identify a problem, such as declining soil fertility, soil degradation, and pests, and to respond to it, e.g., through on-station research and dissemination of technologies. This not only has improved soil management but has also been accompanied by major setbacks in the sense that adoption of alien technologies often ignored socioeconomic aspects and turned out to be costly or difficult to accept at the social level. Approaches, therefore, focus on incremental adaptation of technologies rather than adoption, i.e., understanding indigenous farming practices to guide sustainable land use development.^[10] This does not mean that indigenous knowledge always provides the solution. Rapid changes in the socioeconomic, political, and biophysical environments put indigenous knowledge systems under tremendous pressure and do not give them time to adapt themselves to the changes. But learning about local knowledge, recognizing indigenous practices and their functions, and then trying to improve soil management together with local farmers, researchers, and external advisors means making the best use of all available knowledge systems.

REFERENCES

1. Van Marrewijk, A. *Indigenous Knowledge and Development Monitor*; Nuffic-CIRAN: The Hague, 2000; 48 pp.
2. Scoones, I.; Reij, C.; Toulmin, C. Sustaining the soil. In *Indigenous Soil and Water Conservation in Africa*; Reij, C., Scoones, I., Toulman, C., Eds.; IIED Drylands Programme: London, 1996; 260 pp.
3. Eyasu, E. Soil enrichment and depletion in Southern Ethiopia. In *Nutrients on the Move*; Hilhorst, T., Muchena, F., Eds.; IIED Drylands Programme: London, 2000; 65–82.
4. Corbeels, M.; Shiferaw, A.; Haile, M. Farmers' knowledge of soil fertility and local management strategies in Tigray, Ethiopia. *Manage. Africa's Soils* **2000**, *10*, 24.
5. Nandwa, S.M.; Ondurru, D.D.; Gachimbi, L.N. Soil fertility regeneration in Kenya. In *Nutrients on the Move*; Hilhorst, T., Muchena, F., Eds.; IIED Drylands Programme: London, 2000; 119–132.
6. Gebre Michael, Y. *The Use, Maintenance, and Development of Soil and Water Conservation Measures by Small-Scale Farming Households in Different Agro-Climatic Zones of Northern Shewa and Southern Wollo, Ethiopia*; Soil Conservation Research Project 2000; Research Report 44; Centre for Development and Environment: Bern and Addis Ababa, 1999; 188 pp.
7. Gebre Michael, Y.; Herweg, K. *Soil and Water Conservation—From Indigenous Knowledge to Participatory Technology Development*; CDE: Bern, 2000; 52 pp.
8. Tamang, D. Living in a fragile ecosystem: Indigenous soil management in the hills of Nepal. Gatekeeper Series 41; IIED: London, 1993.
9. Hilhorst, T.; Muchena, F., Eds. *Nutrients on the move: Soil Fertility Dynamics in African Farming Systems*; IIED Drylands Programme: London, 2000; 146 pp.
10. Talawar, S.; Rhoades, R.E. Scientific and local classification and management of soils. *Agr. Human Values* **1998**, *15*, 3–14.

Indo-Gangetic Plain: Agro-Ecological Regions

Jaya N. Surya

*Regional Centre Delhi, National Bureau of Soil Survey and Land Use Planning,
Indian Council of Agricultural Research (ICAR), New Delhi, India*

R.P. Yadav

*National Bureau of Soil Survey and Land Use Planning, Indian Council
of Agricultural Research (ICAR), New Delhi, India*

G.S. Sidhu

Ministry of Agriculture, Government of India, New Delhi, India

S.K. Singh

*National Bureau of Soil Survey and Land Use Planning, Indian Council
of Agricultural Research (ICAR), Nagpur, India*

Abstract

The Indo-Gangetic Plain (IGP) is largest fluvial plains in the world. In India, it occupies 13% of the total geographical area and produces nearly 50% food grain production of the country. The soils are under tremendous stress because of high demographic pressure and intensive cultivation, resulting in declining soil quality and crop productivity. To arrest this, the detailed information of soils of IGP and their characterization, constraints, and potentials were synthesized, which may be useful in sustainable land resource management. The soils of IGP varied widely in its extent, nature, and characteristics. The dominant soil belongs to orders Inceptisols (68%), Entisols (23.2%), and Alfisols (5.2%). The soil map of IGP was generated by identifying 185 subgroup associations, and the dominant subgroups are Typic Ustochrepts, Typic Ustifluvents, Fluventic Ustochrepts, Aeric Haplaquepts, Typic Ustipsamments, and Udic Ustochrepts. The Piedmont plain of IGP has deep, well-drained, loamy/sandy loam, mildly to moderately alkaline soils medium in organic carbon (OC), whereas Tarai region accounts well-drained, loamy soils rich in OC. Soils of semiarid aeolian/aeofluvial plains are sandy/coarse-loamy, saline/sodic, calcareous with low nutrient- and water-holding capacity. Soils of alluvial plains vary widely in texture, drainage, and fertility. Poor drainage, occasional flooding, and salinity/sodicity are the major constraints affecting crop productivity in upper and middle alluvial plains. Lower alluvial plains (Bengal Basin) suffer from waterlogging, frequent flooding, low nutrient status, and salinity. Appropriate land management practices and land use options are needed to ameliorate these constraints and sustain agricultural productivity.

INTRODUCTION

Soil is a limited and non-renewable resource, and its continuous cultivation even with best technologies and skills results in deterioration of its quality.^[1] About 57% of the country land area is under agriculture as against the world average of 11.5%. Major land use changes have occurred after independence when net sown area increased from 118.75 mha in 1950–1951 to 140.0 mha by 1970–1971, 143.0 mha by 1991–1992, and 140.8 mha by 2011–2012. Agricultural interstratifications and high population pressure including Green Revolution era and beyond have caused degradation of various forms. Harmonized data have put total degraded area at 121 mha—with water-induced erosion being the major form followed

by chemical degradation, wind erosion, and physical degradation.^[2]

The Indo-Gangetic Plain (IGP) region is the food bowl of the country and produces nearly half of food grains in India. Average yield of food grains (3314 kg ha^{-1}) is far better than that of the country as a whole (1860 kg ha^{-1})^[3] with high cropping intensity ranging from 133% to 188%. The vast plain is witnessing degradation of natural ecosystem, such as soil and water, and declining productivity.^[4] Sustainability of different production systems is under threat with several reports indicating fatigue of the natural resources.^[5,6]

In view of increased demands for food and fiber and limited scope for bringing new area under plough, it is imperative to have comprehensive knowledge about soil

and land resources. It will help to land management and land use options for improving crop productivity and soil and environmental quality. This entry briefly describes the distribution of soils under different physiographic and agroecological regions (AERs) in IGP with analysis of their potentials and constraints for scientific land use.^[7]

GEOGRAPHIC SETTINGS

The IGP is one of the most extensive fluvial plains of the world. It is situated between Gurdaspur district (Punjab) in the west and Jalpaiguri district (West Bengal) in the east, is extending to west Assam in the northeast, lies between 21° 45' to 31° 30' N latitude and 74° 15' to 91° 30' E longitude, and covers an area of 44 mha (Fig. 1). These are aggradational plains formed by the rivers the Ganges and the Brahmaputra and their tributaries. They include Indus plain, Rajasthan plain, Punjab plain, Ganga plain, and Brahmaputra valley. It covers the states of Punjab, Haryana, Delhi, Uttar Pradesh (UP), Uttarakhand, Bihar, West Bengal, Assam, and some parts of adjoining states.^[8]

There is a wide variation in climate, land use, and soils in the region conditions. Climate ranges from arid to semi-arid in the west to humid to perhumid in the east.^[8,9] The rice and wheat are the main crops apart from pulses, oilseeds, cash crops, and horticultural crops. Barring the area in the extreme west (Rajasthan), rice (*Oryza*

sativa L.) is grown throughout the IGP. However, in the extreme east, wheat (*Triticum estivum* L.) is not preferred due to shorter winters. Other crops grown are sugarcane (*Saccharum officinarum* L.), cotton (*Gossypium hirsutum* L.), mustard (*Brassica juncea* L.), safflower (*Carthamus tinctorius* L.), pearl millet (*Pennisetum glaucum* L.), potato (*Solanum tuberosum* L.), and groundnut (*Arachis hypogea* L.).

Geological Age of Soils

History

The IGP ranks as one of the most extensive fluvial plains of the world. The deposit of this tract represents the last chapter of earth's history. It came into existence due to collision of the Indian and Chinese plates during Middle Miocene.^[10] The Indian plate is moving at the rate of 2–5 cm yr⁻¹ toward the north and forming the world's highest mountain range on its border. The north–south compression generated throughout the plate ensures that it is continuously under stress and provides the basic resource of accumulating strain in the fractured zones.^[11] The fluvial deposits and landforms of the IGP have been influenced by the stresses directed toward north and north-east. The major rivers of the IGP have changed their courses and are flowing in southeast and easterly directions with convexity toward the southeast, which is strikingly similar to the arcuate pattern of the major thrusts bordering

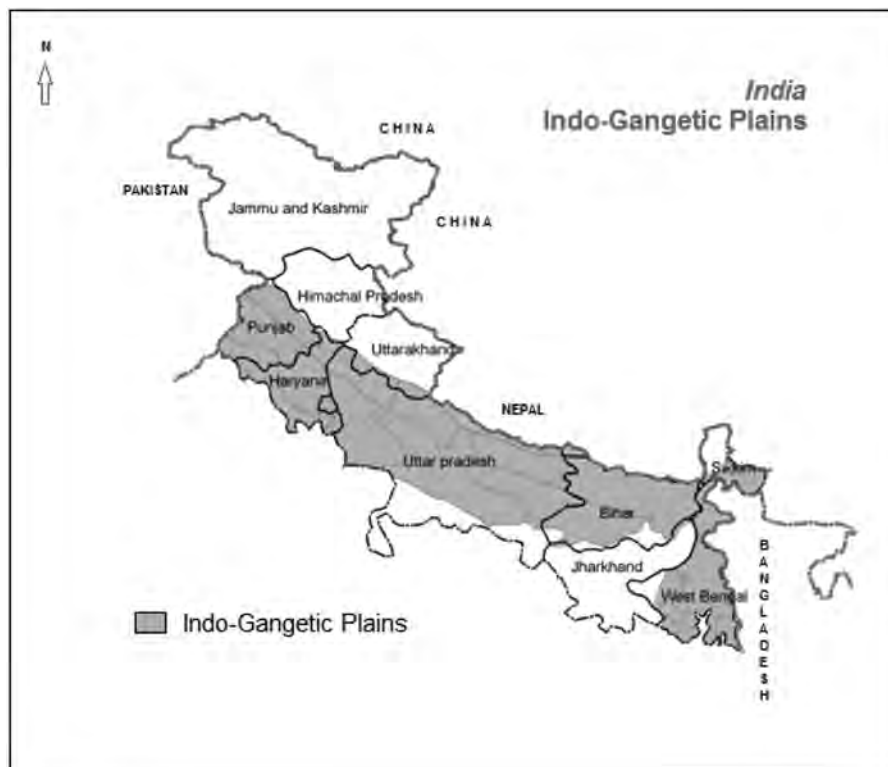


Fig. 1 Extent of IGP, India.

the IGP.^[12] Thus, the IGP shows a series of terraces, bars, and meandering scars, resulting in microhigh and microlow areas on the apparently smooth topography.^[13–15] The IGP is tectonically active, and the major sedimentation is taking place from large river systems of the IGP. The IGP is mainly developed by the alluvium of Indus, Yamuna, Ganga, Ramganga, Ghagra, Rapti, Gandak, Bhagirathi, Silai, Damodar, Ajay, and Kosi rivers. Geophysical surveys and deep drilling by the Oil and Natural Gas Commission of India^[16,17] suggest that the IGP is a vast asymmetric trough with maximum thickness of 10,000 m that thins out to the south. The nature and properties of the alluvium vary in texture from sandy to clayey, calcareous to non-calcareous, and acidic to alkaline. Though the overall topographic situation remains fairly uniform with elevations of 150 m amsl (meter above mean sea level) in the Bengal Basin and 300 m amsl in the Punjab plain, local geographic variations are significant.^[18]

PHYSIOGRAPHY AND AGROECOLOGICAL SUBREGIONS (AESRs)

The IGP is basically a riverine plain, commonly known as Indo-Gangetic alluvial Plains that are composed of featureless landform on broad scale and variations at local conditions. The elevation ranges from 150 msl to 300 msl.^[14] Broadly, the IGP is divided into six major physiographic units based on the landform analysis. Alluvial plains occupied largest area (69% of TGA) followed by transitional aeofluvial plains (19.5%), aeolian plain (5.4%), piedmont plain (4.5%), Brahmaputra Valley (0.6%), and Tarai region (0.5%).

The region reflects a close relationship between physiography–soils–climate and vegetation. Accordingly, it represents 8 AERs, 17 AESRs (Fig. 2), and 6 bioclimatic regions of the country, depending upon major physiography, climate, and length of growing period (LGP).^[7,8,19,20]

SOILS OF IGP

The soil resource information available at state level at association of subgroups^[21] was integrated in Geographic Information System (GIS) by using GIS ArcGIS-9.2 software (ArcGIS-9.2 is version) and generated the soil map of IGP (Fig. 3). The soils of the IGP varies widely in their extent, nature, and characteristics. These belong to 5 orders, 12 suborders, 18 great groups, and 36 dominant subgroups. Soils of order Inceptisols account for largest area (68.0%), followed by Entisols (23.2%), Alfisols (5.2%), Aridisols (3.0%), Mollisols (0.31%), and others (0.24%; Fig. 4). The areal extent of dominant subgroups occurring in IGP region is given in Table 1. On the basis of physiographic delineation, climate, vegetation, and parent material, the soil resources of IGP are described under the following subheads.

SOILS OF PIEDMONT PLAIN AND TARAI REGION

The soils of this region occupy an area of 2.8 mha in the states of Punjab, Haryana, UP, and southern part of Uttarakhand.^[22–26] The dominant landscape of the region is represented by gently to very gently sloping and nearly level plain. The soils mainly belong to Typic Ustifluvents, Typic Ustipsamments, Aquic/Typic Udipsamments, Typic Haplaquents, Aquic Udifluvents, Typic Udorthents, and Typic Ustorthents subgroups of Entisols; Typic/Fluventic Haplustepts, Typic/Fluventic Eutrudepts/Fluventic Haplustepts of Inceptisols; and Udic-Typic Haplustolls of Mollisols.^[21] The soils, developed on alluvium, are deep to moderately deep, loamy, and calcareous. Soils of Tarai region are high in organic carbon (OC) content.

The region pertains to AER-9. The agroclimate is characterized by hot dry subhumid to semiarid transition with dry summers and cool winters with annual rainfall ranges from 700 to 1200 mm. LGP is 150–180 days.^[7] The natural vegetation consists of northern tropical dry mixed deciduous forest. Most of the area is under rainfed agriculture. The main crops grown are maize, pigeon pea, rice, sugarcane, and cotton (in small patches) during *kharif* and wheat, barley, mustard, and lentil during *rabi* season.

Potentials and Constraints

Soils of Tarai region are suitable to grow high-value commercial crops and medicinal plants. Fine loamy soils have problems of permeability. Water erosion during rainy season through seasonal streams (Choes) is a constraint in kandi areas. Plantation of commercial trees, e.g., Poplar, *Dalbergia sissoo*, has wide scope in these areas.

SOILS OF ALLUVIAL PLAIN

These soils occupy an area of 34.2 mha in the states of Punjab, Haryana, UP, Bihar, and West Bengal.^[23,25–28] It is agriculturally important belt and popularly known as “Green Bowl” of the country. The dominant soils belong to Typic Haplustepts, Fluventic Haplustepts, Typic Psammaquents, Typic/Aeric Fluvaquents, Typic/Ustic Torripsamments, Aquic/Typic Ustipsamments, Aquic/Typic Ustifluvents, and Typic Ustorthents subgroups of Entisols; Aeric Halaquents, Typic/Vertic/Fluventic Haplustepts, Aquic/Fluventic/Typic/Vertic Haplustepts, Dystric Eutrudepts and Dystric Eutrudepts of Inceptisols; Fluventic/Typic/Vertic/Ustic Camborthids of Aridisols; and Typic Haplustalfs, Typic Natrustalfs, and Typic/Aeric/Vertic Ochraqualfs of Alfisols.^[21]

These soils are very deep, light to dark yellowish brown/brown, well-drained to moderately well-drained, loamy/fine loamy, moderately to strongly alkaline, moderate to highly calcareous, and low to medium in available nitrogen (N) and phosphorus (P) content but medium to high in



Fig. 2 Agroecological subregion (revised) map of IGP.

Source: From Mandal, Mandal, et al.^[19] ©2014.

available potassium. The deficiency of zinc (Zn), copper, and manganese has been reported in some intensively cultivated areas. Some soils of this region are reported problem of salinity.^[29] The formation of hardpan in subsurface soils has been noted in some soils of rice-wheat cropping system areas.^[30] Some low-lying areas have problem of imperfect or poor drainage and occasional flooding. Soils of northern IGP (eastern UP and Bihar state) region is, in general, deep, fine loamy/fine silty.

The low-lying tracts of Varanasi, Ghazipur, and Ballia districts of UP are of fine loamy, Aeric Ochraqualfs,^[31] while Bihar plains had Aeric Haplaquepts/Typic Haplaquepts and Ustifluvents soils that show wide variation in drainage. Occasional flooding and imperfect drainage, fine soil texture, and medium to low fertility [available P and sulfur (S)] are the problems of eastern part of the IGP. Salinity, strong alkalinity, and calcareousness induce nutrient imbalance.^[20,32] Soils of Bengal Basin (in parts of

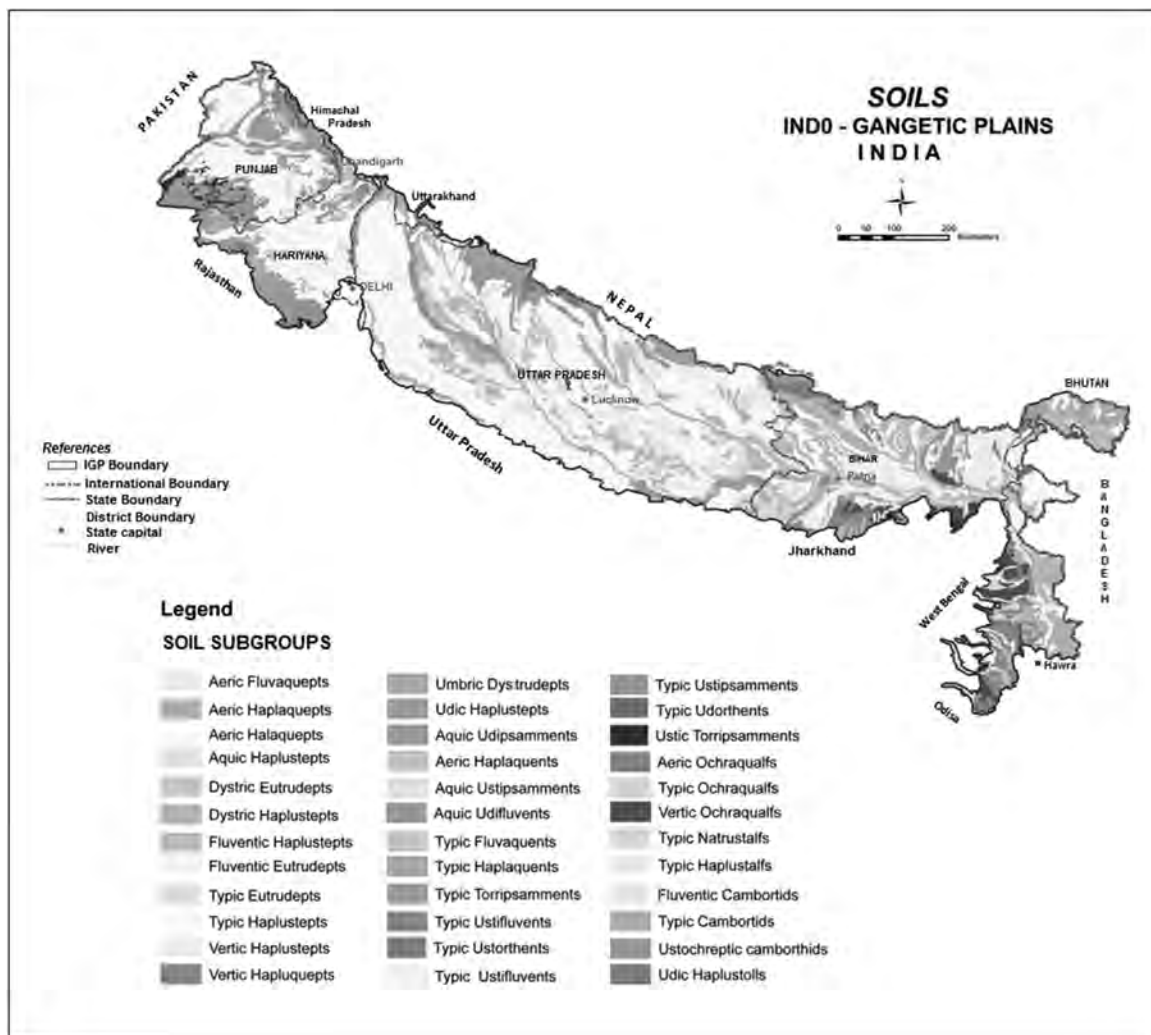


Fig. 3 The soil map of IGP, India.

Hooghly, Murshidabad, and Bardhaman districts) are very deep, fine loamy, mildly to moderately alkaline Fluvaquepts/Fluventic Eutrudepts and Vertic Ochraqualfs.^[20] It is estimated that about 69% carbon in the IGP soils is confined to upper 40 cm soil layer, where C stock ranges from 8.5 to 15.2 t C ha⁻¹.^[33] Soil OC can be enhanced following introduction of pulses in rice–wheat cropping system and organic nutrient management practices.^[32,34]

The soils represent AERs 4, 9, 13, and 15. The mean annual rainfall ranges from 500 mm (in North West) to as high as 1400 mm in eastern parts. The mean annual temperature varies from 25°C to 26°C. Accordingly the LGP varied from 90–150 days (AESRs 4.1 and 4.3) to 150–180 days (AESRs 9.1 and 9.2) and 180–>210 days (AESRs 13.1, 13.2, 15.1, and 15.3, 18.5).^[7] The natural vegetation consists of tropical dry deciduous and tropical thorn forests in west and central parts and tropical moist and deciduous forests in east. The area is mostly irrigated due to vast network of canals and tube wells. Cotton (in northwest belt), sugarcane, sunflower, beans, rice (60-day duration),

green gram, rapeseeds/mustard, etc. are grown. In AERs 13 and 15, cropping system is mainly rice-based due to high rainfall, particularly in the Bengal plains. In rabi season, rice, jute, pulses, and oilseeds are grown on stored/residual soil moisture. Wheat crop is replacing other crops in eastern parts.

Potentials and Constraints

Majority of soils are suitable for growing region-specific crops. Diversification of cropping system by introducing pulses and oilseed crops may narrow down the gap of supply and demand of these crops. Potato–sugarcane–jute in the middle and eastern region has great potentiality to enhance income of the farmers.

Among the constraints, flooding and imperfect drainage conditions, salinity, and/or sodicity throughout the north-west, central, and some parts of central, eastern parts adversely affect the crop yields. Rice soils have developed subsurface compaction. The deficiency of N, P, S, Zn, etc.

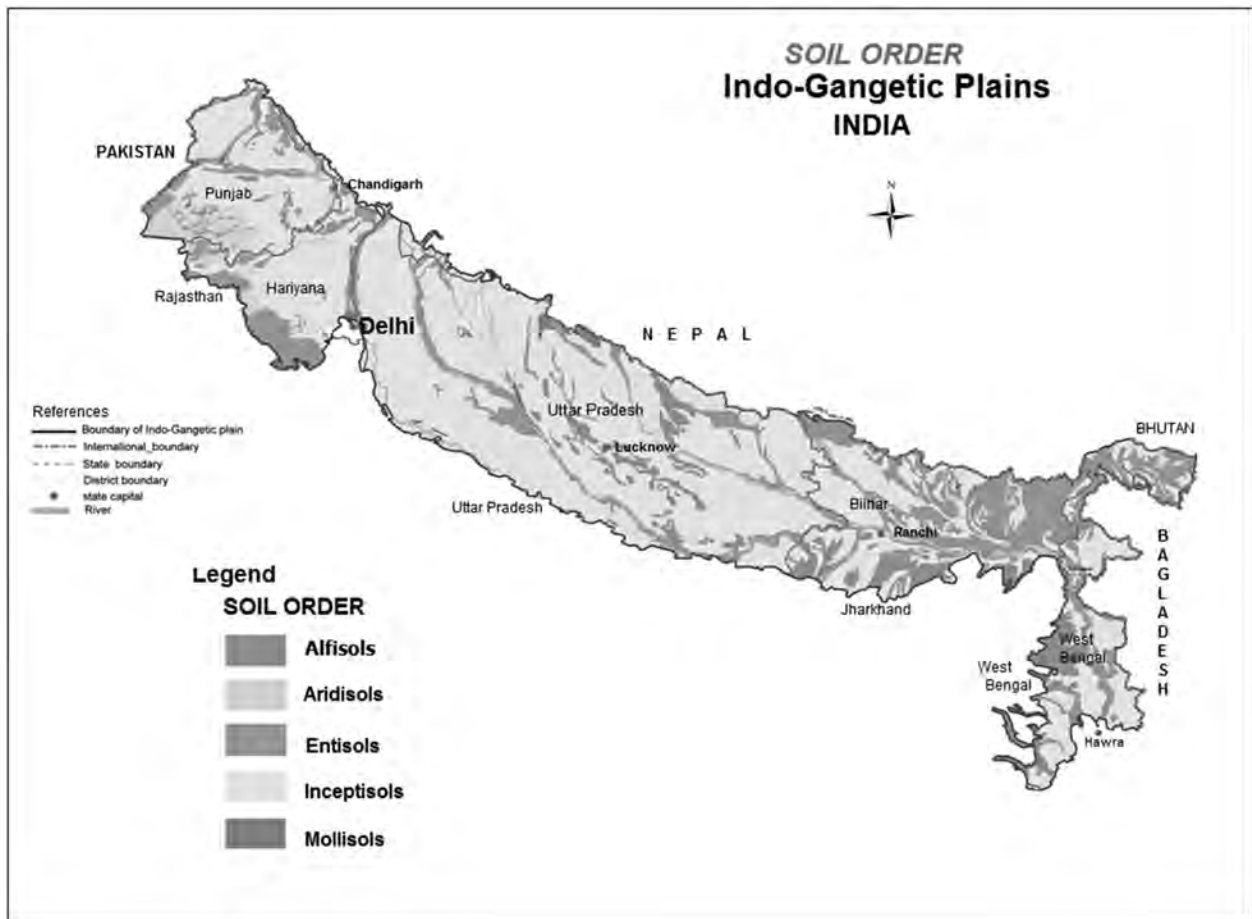


Fig. 4 Spatial distribution of soil orders in the IGP, India

are widespread in intensively cultivated areas. Sandy soils along the river and flood plains suffer due to poor soil fertility. Over exploitation of underground water for irrigation in rice–wheat growing areas has caused decline in water table at an alarming rates.

SOILS OF ARID AEOLIAN PLAIN

Southern part of Punjab, Haryana, and some part of Rajasthan states^[24,25,35] occupy an area of 2.65 mha. The dominant soils belong to Typic Torripsamments, Ustic Torripsamments/Typic Torrifluvents, and Typic Torriorrhents, Typic/Fluventic Camborthids/Typic Haplocalcids.^[21] These soils are deep, sandy to coarse loamy, low in OC, high in pH and electrical conductivity, and calcareous in nature.

These soils belong to AER 2 and AESR 2.3a (the region represents typic hot and arid climate characterized by hot and dry summers and cool and dry winters). The mean annual rainfall is <400 mm that too is concentrated in few events of high intensity. The LGP is <90 days.^[7] The natural vegetation comprises of sparse sporadic tropical thorn forests. Monocropping of drought resistant, short duration

crops such as pearl millet, sorghum (fodder) and pulses are grown under traditional farming practices. Introduction of irrigation through Indira Gandhi canal encouraged farmers to grow crops like wheat, cotton, mustard, and even sugarcane in some areas. Fruits like ber and kinnow (*Hybrid citrus*) are also grown by some farmers.

Potentials and Constraints

The soils in the ecoregion are potentially fertile because of abundant primary minerals. Severe dry hot climate coupled with strong sandy storms in the early growing stage of cotton and other kharif crops severely affects the germination and young seedlings of crops. Sandy soils suffer due to low fertility and water-holding capacity. Salinity develops due to water logging condition near canal.

SOILS OF SEMIARID TRANSITIONAL AEOFLUVIAL PLAIN

The soils of the region comprises of southern–western parts of Punjab, and northern–western parts of Haryana^[23,25]

Table 1 Areal extent of soil subgroups in IGP.

Soil subgroups	Area (mha)	% of TGA	Soil subgroups	Area (mha)	% of TGA
Inceptisols			Entisols		
Fluventic Eutrudepts	204.9	0.47	Typic Haplaquents	265.5	0.60
Typic Eutrudepts	104	0.24	Aquic Udipsamments	181.9	0.41
Umbric Dystrudepts	63.1	0.14	Aquic Udifluvents	73.3	0.17
Udic Haplustepts	1,314	2.99	Typic Ustifluvents	450	1.02
Aeric Fluvaquepts	301	0.68	Typic Udorthents	196.9	0.45
Aeric Hapluquepts	3,233.9	7.35	Typic Ustorthents	880.5	2.00
Aeric Halaquepts	47	0.11	Typic fluvaquents	330.3	0.75
Vertic Hapluquepts	756.9	1.72	Aeric Haplaquents	166.3	0.38
Aquic Haplustepts	750	1.70	Typic Torripsamments	536	1.22
Fluventic Haplustepts	3,341.5	7.59	Ustic Torripsamments	159	0.36
Typic Haplustepts	15,646	35.56	Aquic Ustipsamments	164	0.37
Dystric Haplustepts	628	1.43	Typic Ustipsamments	2,304.2	5.24
Vertic Haplustepts	654.2	1.49	Typic Ustifluvents	3,649.3	8.29
Dystric Eutrudepts	26	0.06	Total	9,357.2	21.26
Total	27,070.5	61.52			
Alfisols			Aridisols		
Aeric Ochraqualfs	359.1	0.82	Fluventic Camborthids	14	0.03
Typic Ochraqualfs	580.2	1.32	Typic Camborthids	264	0.60
Vertic Ochraqualfs	713.3	1.62	Ustochreptic Camborthids	944	2.15
Typic Natrustalfs	23	0.05	Total	1,222	2.78
Typic Haplustalfs	443.1	1.01	Mollisols		
Total	2,118.7	4.82	Udic Haplustolls	127	0.28
			Total	127	0.28
Other soil orders				98.8	0.22
Other miscellaneous land				3,605.8	9.11
Total geographical area of IGP				44,000	100

occupy an area of 9.57 mha. The dominant soils belong to Typic/Ustic Torripsamments, Typic Ustifluvents, Typic Psammaquents, Aquic Ustipsamments, Typic/Lithic Haplustepts, and Lithic/Typic/Ustic Camborthids/Camborthids.^[21] These soils are very deep and sandy/coarse loamy. These soils are underlain by salt-rich material like gypsum, calcite, and dolomite. Soils of arid and semiarid climate are prone to be calcareous and sodic due to low levels of OC.^[21]

These soils pertain to AER 4, with mean annual temperature of 25°C. The annual rainfall varies from 400 to 600 mm in different years. LGP is between 90 and 120 days.^[7,9] The natural vegetation consists of tropical dry deciduous and tropical thorn forest. These soils belong to dry region with rainfall < 600 mm, but the majority of areas are cultivated by levelling the sand dunes and irrigating with the network of canals. These areas are cultivated for wheat, cotton, gram, pulses, and oilseeds, and even rice has been introduced in some area.

Potentials and Constraints

These soils have very high potential for pulses and oilseed crops. These soils are coarse-textured poor soil fertility, nutrient-supplying capacity, and available water-holding capacity. Provision of irrigation has given rise to formation of saline and sodic problems. Use of poor quality of groundwater for irrigation further aggravates the problem of salinity in the region. Many areas face inadequate supply of irrigation water at tails of canals during peak periods of plant growth.

SOILS OF ASSAM (BRAHMAPUTRA) VALLEY

Teesta and Barak valley soils belong to Aeric/Typic Fluvaquents, Aquic/Typic Ustifluvents/Udofluvents, and Aquic/Typic Udipsamments of Entisols, Typic Halaquepts/Dystric Eutrudepts, Udic Haplustepts, Fluventic/Humic Dystrudepts of Inceptisols and Typic Hapludalfs.^[21,36,37] The soils

developed on alluvial flood plains are clayey, moderately to strongly acidic in reaction with low base saturation, but relatively rich in OC.^[36,38] The soil depth ranges from shallow (on steeply sloping hills) to moderately deep (terrace soils) to deep in valleys. Low-lying soils are deep, poorly drained, very slowly permeable, fine clayey over loamy.

These soils belong to AER 15 and AESR 15.3. Mean annual temperature is in the range of 24–25°C. The mean annual rainfall ranges from 2000 to 3200 mm. LGP is more than 300 days.^[7,20] The natural vegetation comprises of tropical moist deciduous to semievergreen, riverine Savannah and bamboo cane forests. The area is intensively cultivated under rainfed rice and jute and millets. Rice is cultivated throughout the year. The plantation crops like tea, pineapple, rubber, and banana are grown under favorable climatic conditions.

Potentials and Constraints

The acidic soils high in humus are most suitable high-value crops for tea, coffee, and rubber plantation. Shallow soil depths on hill slopes and frequent flooding in low-lying areas particularly in monsoon season are major limitation. High rainfall causes excessive leaching of bases, resulting in poor base status.

CONCLUSION

The IGP in India covers 44 mha and extends from Punjab, Haryana, Delhi, UP, Uttarakhand, Bihar, West Bengal, Assam, and northeastern states. It is an agriculturally potential area contributing about half of the food production of the country from 13% of total geographical area. Broadly, the IGP is subdivided into six major physiographic units with alluvial plains and transitional aeofluvial plains being the major ones. Although the soils of the IGP belong to five orders, Inceptisols and Entisols cover about 91% of total area under soils. Soil map of the region has 185 mapping units of subgroups in association. The soils belong to 37 dominant subgroups, but two-third area is under four soil subgroups, namely Typic Haplustepts (36%), Typic Ustifluvents (9%), Aeris Fluvaquents (7%), and Typic Ustipsamments (6%). The Piedmont plains and Tarai region represent deep, well-drained, and coarse-textured soils, while Tarai plains account loamy soils rich in OC. Major area of the IGP is covered under Alluvial plains that have medium to high productivity potentials for a variety of crops. Major constraints in alluvial plain are salinity/sodicity, drainage, and flooding, which increase from upper to lower plains. Continuous cultivation without scientific land use and management has resulted in soil fatigue and low productivity. Proper soil and crop management practices are required to ameliorate soil physical, chemical, and biological condition and improve soil fertility. Soil-site-based land use options, integrated nutrient management,

conservation agriculture, and groundwater augmentation should be adopted to address issues of sustainability in this food bowl of the country. The soil information presented in this entry provides an overview in reference to AESR, which may contribute to better land resource management and land use planning on sustainable basis.

REFERENCES

1. Sidhu, G.S.; Surya, J.N. Soils of North-Western Himalayan eco-system and their land use, constraints, productivity potentials and future strategies. *Agropedology* **2014**, *24* (1), 1–19.
2. Maji, A.K.; Reddy, G.P.O.; Sarkar, D. *Degraded and Wastelands of India – Status and Spatial Distribution*; Directorate of Information and Publication of Agriculture, Indian Council of Agricultural Research Report: New Delhi, 2010; 1–168.
3. Anonymous. *Harmonization of Wastelands/Degraded Lands Datasets of India*; National Rainfed Area Authority (NRAA) publication, Ministry of Agriculture, DPS Marg: New Delhi, 2008; 1–55.
4. Bhattacharyya, T.; Mandal, B. Soils information system of Indo-Gangetic Plains for resource management. ISSS special session on land use planning. In *Proceedings, J. Indian Soc. Soil Sci., Platinum Jubilee Symposium*, New Delhi, Dec. 23–25, 2009; 120–138.
5. Sinha, S.K.; Singh, G.B.; Rai, M. *Decline in Crop Productivity in Haryana and Punjab: Myth or Reality?*; Indian Council of Agricultural Research: New Delhi, 1998.
6. Abrol, Y.P.; Sagwan, S.; Dadhwal, V.K.; Tiwari, M. Land use/land cover in Indo-Gangetic Plains—history of changes, present concerns and future approaches. In *Land Use-Historical Perspectives-Focus on Indo-Gangetic Plains*; Abrol, Y.P., Sagwan, S., Tiwari, M., Eds.; Allied Publishers Pvt. Ltd: New Delhi, 2002; 1–28.
7. Sehgal, J.; Mandal, D.K.; Mandal, C.; Vadivelu, S. *Agro-Ecological Regions of India*. Technical Bulletin No. 24; NBSS & LUP: Nagpur, 1992; 1–130.
8. Bhattacharyya, T.; Pal, D.K.; Chandran, P.; Mandal, C.; Ray, S.K.; Gupta, R.K.; Gajbhiye, K.S. *Managing Carbon Stocks in the Indo-Gangetic Plains, India*; Rice-wheat Consortium for the Indo-Gangetic Plains: New Delhi, 2004; 1–44.
9. Velayutham, M.; Mandal, D.K.; Mandal, C.; Sehgal, J. *Agro-ecological Subregion of India for Planning and Development*. NBSS Publications 35; NBSS & LUP: Nagpur, 1999; 327 pp.
10. Parkash, B.; Kumar, S. Indo-gangetic basin. In *Sedimentary Basin of India, Tectonic Context*; Tandon, S.K., Pant, C.C., Kashyap, S.M., Eds.; Gyanodaya Prakashan: Nainital, Uttarakhand, 1991; 147–170.
11. Gaur, V.K. Evolution of seismic hazard in India towards minimizing earthquake risk. *Curr. Sci.* **1994**, *67*, 324–329.
12. Parkash, B.; Kumar, S.; Rao, M.S.; Giri, S.C.; Kumar, C.S.; Gupta, S.; Srivastava, P. Holocene tectonic movements and stress field in the Western Gangetic Plains. *Curr. Sci.* **2000**, *79*, 438–449.

13. Pal, D.K.; Srivastava, P.; Durge, S.L.; Bhattacharyya, T. Role of micro topography in formation of sodic soils in the semi-arid part of the Indo-Gangetic Plains, India. *Catena* **2003**, *51*, 3–31.
14. Pal, D.K.; Bhattacharyya, T.; Srivastava, P.; Chandran, P.; Ray, S.K. Soils of the Indo-Gangetic Plains: Their historical perspective and management. *Curr. Sci.* **2009**, *96*, 1193–1202.
15. Singh, S.; Parkash, R.; Rao, M.S.; Arora, M.; Bhosle, B. Geomorphology, pedology and sedimentology of the Deoha/Ganga–Ghaghara interfluvium, Upper Gangetic Plains (Himalayan FORELAND Basin – Extensional tectonic implications). *Catena* **2006**, *67*, 183–203.
16. Sastri, V.V.; Venkatachala, B.S.; Bhandari, L.L.; Raju, A.T.R.; Datta, A.K. Tectonic framework and subsurface stratigraphy of the Ganga Plain. *J. Geol. Soc. India* **1971**, *12*, 222–232.
17. Rao, M.B.R. The subsurface geology of the Indo-Gangetic Plains. *J. Geol. Soc. India* **1973**, *14*, 217–242.
18. Shankaranarayana, H.S. Morphology, genesis and classification of soils of the Indo-Gangetic Plains. In *Review of Soil Research in India, Part II*; 12th International Congress of Soil Science on behalf of the Indian Society of Soil Science, Division of Soil Science and Agricultural Chemistry, Indian Agricultural Research Institute. Secretary, Organising Committee: New Delhi, India, 1982.
19. Mandal, C.; Mandal, D.K.; Bhattacharyya, T.; Sarkar, D.; Pal, D.K.; Prasad, J.; Sidhu, G.S.; Nair, K.M.; Sahoo, A.K.; Das, T.H.; Singh, R.S.; Srivastava, R.; Sen, T.K.; Chatterji, S.; Chandran, P.; Ray, S.K.; Patil, N.G.; Obireddy, G.P.; Mahapatra, S.K.; Anil Kumar, K.S.; Das, K.; Singh, A.K.; Reza, S.K.; Dutta, D.; Srinivas, S.; Tiwary, P.; Karthikeyan, K.; Venugopalan, M.V.; Srivastava, A.; Raychaudhuri, M.; Kundu, D.K.; Mandal, K.G.; Kar, G.; Durge, S.L.; Kamble, G.K.; Gaikwad, M.S.; Nimkar, A.M.; Bobade, S.V.; Anantwar, S.G.; Patil, S.; Gaikwad, K.M.; Nimkar, A.M.; Bobade, S.V.; Anantwar, S.G.; Patil, S.; Gaikwad, K.M.; Sahu, V.T.; Bhondwe, H.; Dohre, S.S.; Gharami, S.; Khapekar, S.G.; Koyal, A.; Sujatha; Reddy, B.M.N.; Sreekumar, P.; Dutta, D.P.; Gogoi, L.; Parhad, V.N.; Halder, A.S.; Basu, R.; Singh, R.; Jat, B.L.; Oad, D.L.; Ola, N.R.; Wadhai, K.; Lokhande, M.; Dongare, V.T.; Hukare, A.; Bansod, N.; Kolhe, A.; Khushpure, J.; Kuchankar, H.; Balbuddhe, D.; Sheikh, S.; Sunitha, B.P.; Mohanty, B.; Hazarika, D.; Majumdar, S.; Garhwal, R.S.; Sahu, A.; Mahapatra, S.; Puspamitra, S.; Kumar, A.; Gautam, N.; Telpande, B.A.; Nimje, A.M.; Likhar, C.; Thakre, S. Revisiting agro-ecological sub-regions of India – A case study of two major food production zones. *Curr. Sci.* **2014**, *107* (9), 1519–1536.
20. Velayutham, M.; Mandal, D.K.; Bhattacharya, T.; Srivastava, P. Soils of Indo-Gangetic Plains, India – the historical perspective. In *Land Use–Historical Perspectives: Focus on Indo-Gangetic Plains*; Abrol, Y.P., Sagwan, S., Tiwary, M.K., Eds.; Allied Publishers Pvt. Ltd: New Delhi, 2002; 61–70.
21. Staff NBSS & LUP. *Soils of India*; NBSS Publ. 94, National Bureau of Soil Survey and Land Use Planning: Nagpur, 2002; 1–130.
22. Rana, K.P.C.; Walia, C.S.; Sidhu, G.S.; Singh, S.P.; Velayutham, M.; Sehgal, J. *Soils of Jammu and Kashmir for Optimising Land Use*. NBSS Publ. 62b (Soils of India Series); National Bureau of Soil Survey and Land Use Planning: Nagpur, 2000; 1–71.
23. Sidhu, G.S.; Walia, C.S.; Lal, T.; Rana, K.P.C.; Sehgal, J. *Soils of Punjab for Optimising Land Use*. NBSS Publ. 45b (Soils of India Series); National Bureau of Soil Survey and Land Use Planning: Nagpur, 1995; 1–75.
24. Sidhu, G.S.; Rana, K.P.C.; Sehgal, J.; Velayutham, M. *Soils of Himachal Pradesh for Optimising Land Use*. NBSS Publ. 57b (Soils of India Series); National Bureau of Soil Survey and Land Use Planning: Nagpur, 1997; 1–44.
25. Sachdev, C.B.; Lal, T.; Rana, K.P.C.; Sehgal, J. *Soils of Haryana for Optimising Land Use*. NBSS Publ. 44b (Soils of India Series); National Bureau of Soil Survey and Land Use Planning: Nagpur, 1995; 1–59.
26. Singh, S.P.; Jagat, R.; Walia, C.S.; Sachdev, C.B.; Dhankar, R.P.; Rana, K.P.C.; Sehgal, J.; Velayutham, M.; Gajbiye, K.S. *Soils of Uttar Pradesh for Optimising Land Use*. NBSS Publ. 68b (Soils of India Series); National Bureau of Soil Survey and Land Use Planning: Nagpur, 2004; 1–91.
27. Sidhu, G.S.; Bhattacharyya, T.; Sarkar, D.; Ray, S.K.; Chandran, P.; Pal, D.K.; Mandal, D.K.; Prasad, J.; Nair, K.M.; Sahoo, A.K.; Das, T.H.; Singh, R.S.; Mandal, C.; Srivastava, R.; Sen, T.K.; Chatterji, S.; Patil, N.G.; Obireddy, G.P.; Mahapatra, S.K.; Anil Kumar, K.S.; Das, K.; Singh, A.K.; Reza, S.K.; Dutta, D.; Srinivas, S.; Tiwary, P.; Karthikeyan, K.; Venugopalan, M.V.; Velmourougane, K.; Srivastava, A.; Raychaudhuri, M.; Kundu, D.K.; Mandal, K.G.; Kar, G.; Durge, S.L.; Kamble, G.K.; Gaikwad, M.S.; Nimkar, A.M.; Bobade, S.V.; Anantwar, S.G.; Patil, S.; Sahu, V.T.; Gaikwad, K.M.; Bhondwe, H.; Dohre, S.S.; Gharami, S.; Khapekar, S.G.; Koyal, A.; Sujatha; Reddy, B.M.N.; Sreekumar, P.; Dutta, D.P.; Gogoi, L.; Parhad, V.N.; Halder, A.S.; Basu, R.; Singh, R.; Jat, B.L.; Oad, D.L.; Ola, N.R.; Wadhai, K.; Lokhande, M.; Dongare, V.T.; Hukare, A.; Bansod, N.; Kolhe, A.; Khushpure, J.; Kuchankar, H.; Balbuddhe, D.; Sheikh, S.; Sunitha, B.P.; Mohanty, B.; Hazarika, D.; Majumdar, S.; Garhwal, R.S.; Sahu, A.; Mahapatra, S.; Puspamitra, S.; Kumar, A.; Gautam, N.; Telpande, B.A.; Nimje, A.M.; Likhar, C.; Thakre, S. Impact of management levels and land use changes on soil properties in rice-wheat cropping system of Indo-Gangetic Plains. *Curr. Sci.* **2014**, *107* (9), 1487–1501.
28. Halder, A.K.; Srivastava, R.; Thampi, C.J.; Sarkar, D.; Singh, D.S.; Sehgal, J.; Velayutham, M. *Soils of Bihar for Optimising Land Use*. NBSS Publ. 50b (Soils of India Series); National Bureau of Soil Survey and Land Use Planning: Nagpur, 1996; 1–70.
29. Mandal, A.K.; Sharma, R.C. Computerized database of salt affected soils for agro-climatic regions in the Indo-Gangetic Plains of India using GIS. *Geocarto Int.* **2006**, *21*, 47–58.
30. Chandran, P.; Tiwary, P.; Bhattacharyya, T.; Mandal, C.; Prasad, J.; Ray, S.K.; Sarkar, D.; Pal, D.K.; Mandal, D.K.; Sidhu, G.S.; Nair, K.M.; Sahoo, A.K.; Das, T.H.; Singh, R.S.; Srivastava, R.; Sen, T.K.; Chatterji, S.; Patil, N.G.; Obireddy, G.P.; Mahapatra, S.K.; Anil Kumar, K.S.; Das, K.; Singh, A.K.; Reza, S.K.; Dutta, D.; Srinivas, S.; Karthikeyan, K.; Venugopalan, M.V.; Velmourougane, K.; Srivastava, A.; Raychaudhuri, M.; Kundu, D.K.; Mandal, K.G.; Kar, G.; Dijkshoorn, J.A.; Batjes, N.H.; Bindraban, P.S.;

- Durge, S.L.; Kamble, G.K.; Gaikwad, M.S.; Nimkar, A.M.; Bobade, S.V.; Anantwar, S.G.; Patil, S.V.; Gaikwad, K.M.; Sahu, V.T.; Bhondwe, H.; Dohetre, S.S.; Gharami, S.; Khapekar, S.G.; Koyal, A.; Sujatha; Reddy, B.M.N.; Sree-kumar, P.; Dutta, D.P.; Gogoi, L.; Parhad, V.N.; Halder, A.S.; Basu, R.; Singh, R.; Jat, B.L.; Oad, D.L.; Ola, N.R.; Wadhai, K.; Lokhande, M.; Dongare, V.T.; Hukare, A.; Bansod, N.; Kolhe, A.H.; Khuspure, J.; Kuchankar, H.; Balbuddhe, D.; Sheikh, S.; Sunitha, B.P.; Mohanty, B.; Hazarika, D.; Majumdar, S.; Garhwal, R.S.; Sahu, A.; Mahapatra, S.; Puspamitra, S.; Kumar, A.; Gautam, N.; Telpande, B.A.; Nimje, A.M.; Likhari, C.; Thakre, S. Development of soil and terrain digital database for major food – growing regions of India for resource planning. *Curr. Sci.* **2014**, *107* (9), 1420–1430.
31. Murthy, R.S.; Pandey, S. *Soil Map of India (1: 6.3 million)* National Bureau of Soil Survey & land Use Planning (ICAR: Nagpur, 1983).
 32. Bhattacharya, T.; Pal, D.K. *Carbon Sequestration in Soils of Indo-Gangetic Plains. In RWC-CIMMYT. Addressing Resource Conservation Issues in Rice-wheat Systems of south Asia. A Resource Book. Rice-wheat Consortium for Indo-Gangetic Plains*; International Maize and wheat Improvement Centre: New Delhi, 2003; 68–71.
 33. Singh, H.; Pathak, P.; Kumar, M.; Raghubanshi, A.S. Carbon sequestration potentials of Indo-Gangetic agro-ecosystem soils. *Trop. Ecol.* **2011**, *52* (2), 223.
 34. Ghosh, P.K.; Venkatesh, M.S.; Hazra, K.K.; Narendra, K. Long-term effect of pulses and nutrient management on soil organic carbon dynamics and sustainability on an inceptisols of Indo-Gangetic Plains of India. *Exp. Agric.* **2012**, *48* (4), 473–487.
 35. Shyampura, R.L.; Sehgal, J. *Soils of Rajasthan for Optimising Land Use*. NBSS Publ. 51b (Soils of India Series); National Bureau of Soil Survey and Land Use Planning: Nagpur, 1995; 1–76.
 36. Sen, T.K.; Chamua, G.S.; Sehgal, J.; Velayutham, M. *Soils of Assam for Optimising Land Use*. NBSS Publ. 66b (Soils of India Series); National Bureau of Soil Survey and Land Use Planning: Nagpur, 1999; 1–51.
 37. Bhattacharya, T.; Sehgal, J.; Sarkar, D. *Soils of Tripura for Optimising Land Use: Their Kinds, Distribution and Suitability for Major Field Crops and Rubber*. NBSS Publ. 65a,c (Soils of India Series 6); National Bureau of Soil Survey and Land Use Planning: Nagpur, 1996; 1–154.
 38. Walia, C.S.; Chamuah, G.S. Flood affected soils of Brahmaputra valley and their suitability for land use planning. *J. Indian Soc. Soil Sci.* **1992**, *40*, 335–340.

Industrial Waste

Warren Dick

School of Natural Resources, Ohio State University, Wooster, Ohio, U.S.A.

Abstract

Large amounts of industrial wastes are either disposed in a planned manner or simply allowed to disperse into the environment (e.g., carbon dioxide, because of the combustion of fossil fuels). However, there are only a few places where disposal can be done on (or in) the land, dumped in water bodies such as the oceans, or incinerated. Ocean dumping has been outlawed or severely curtailed in many of the more developed countries. Incineration is also often opposed because during combustion of the waste, there is potential to create even more toxic materials such as dioxins. Therefore, the only remaining disposal area is the land. This can be in dedicated landfills. Alternatively, many industrial by-products are being viewed as having potential beneficial properties that can be captured either as a recycled material or land-applied as a soil amendment. Each of these options must be carefully researched to avoid the creation of a new problem by solving an existing problem.

INTRODUCTION

There is no universally accepted definition of industrial waste. Common sense suggests industrial wastes are any materials created or left over as a result of industrial activities to produce durable and non-durable goods. In a utopian world, we would be 100% efficient in the use of raw materials and products, which, once manufactured, would serve their purpose and then simply vanish with no need for disposal.

Unfortunately, although we are not operating in the same grossly inefficient way we did during the peak of the Industrial Revolution, we are generating millions of metric tons of industrial wastes every year. We will never achieve the utopian dream, but we can make it our goal to reduce the amount of industrial wastes and to properly dispose those materials that cannot be beneficially recycled. The first step toward achieving this goal is to identify the types of industrial activities and the wastes they create. Then processes and technologies can be designed such that they will either reduce the amounts of wastes created or recycle the wastes in ways that are environmentally safe and economically beneficial.

TYPES OF INDUSTRIAL PRODUCTS

Our modern lifestyle has generated an insatiable appetite for all types of manufactured products. In [Table 1](#), a list of industrial products is given. All these products generate industrial waste at some point in their life cycle. The type of wastes may be unique or be common to many different types of products. The major variables that affect the amounts and types of wastes for each product are the raw

materials used, the amount of processing required of the raw materials before a finished product is created, the relative toxicity of the processing materials, and the total amount of energy required during product manufacture.

TYPES OF INDUSTRIAL BY-PRODUCTS

The types of industrial by-products are as varied as there are industries to produce materials and goods for human consumption. However, there are several broad categories that can be identified and the information presented below is taken from the work of Miller et al.^[1]

Metal Refining and Processing

“Slags” are produced by iron and steel making. In 1995, 21 million metric tons were produced in United States. Slags are produced by addition of lime to furnaces to remove silicon impurities and are composed primarily of CaSiO_3 . About 60% of the slags produced are iron slags and 40% are steel slags. “Fine dusts” are fine airborne particulates collected from metal processing, refining, and fabricating industries. The most commonly encountered dusts are fines from the metal processing industry (especially from electric furnaces) and are primarily ferrous in nature with admixture of heavy metals. Approximately, five million metric tons per year are produced in United States and most are either recycled within the facility or used as landfill.

Mineral Processing

“Gypsum” ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) is produced as a by-product by a number of industries, most notably the phosphate fertilizer

Table 1 Census Bureau's industrial products overview covering manufactured durable and nondurable goods.

Industrial product category	Industrial product subcategory
Building materials	Clay construction
	Lumber and mill stocks
	Plumbing fixtures
	Refractories
Chemicals	Fertilizer materials
	Industrial gases
	Inorganic chemicals
	Paint and allied products
	Paint, varnish, and lacquer
	Pharmaceutical preparations
Consumer goods	Consumer electronics
	Electric housewares
	Household appliances
Electronics	Communication equipment
	Computers and office machines
	Electromedical equipment
	Fluorescent lamp ballasts
	Lighting fixtures
	Measurement instruments
	Semiconductors and components
	Switchgear and industrial controls
	Wiring devices
Food	Confectionery
	Fats and oils
	Flour milling
	Oilseed crushings
Glass	Flat glass
	Glass containers
	Glassware
Heavy machinery	Construction machinery
	Farm equipment
	Internal combustion engines
	Metalworking machinery
	Mining machinery
Industrial equipment	Air pollution control
	Antifriction bearings
	Fluid power
	Motors and generators
	Pumps and compressors
	Refrigeration and heating
	Steel drums and pails
	Vending machines

(Continued)

Table 1 Census Bureau's industrial products overview covering manufactured durable and nondurable goods. (Continued)

Industrial product category	Industrial product subcategory
Primary metals	Aluminum mill products
	Insulated wire and cable
	Iron and steel castings
	Non-ferrous castings
	Steel mill products
	Steel mill inventories
Textile and apparel	Apparel
	Broadwoven fabrics
	Carpets and rugs
	Cotton consumption
	Footwear (annual)
	Footwear (quarterly)
	Gloves and mittens
	Knit fabric production
	Bed and bath furnishings (sheets, pillowcases, and towels)
	Woolen consumption
Transportation	Yarn production
	Aerospace industry
	Civil aircraft and engines
	Truck trailers

Source: Adapted from U.S. Census Bureau.^[2]

industry, the pigment processing and electroplating industries, and the electricity generating industry. Large amount of gypsum are created during the processing of rock phosphate to produce phosphate fertilizer. Nearly 1 billion metric tons of "phosphogypsum" are stockpiled in the phosphate district of central Florida, where most of the commercial rock phosphate deposits are found in United States. A major concern related to phosphogypsum is the radioactivity that occurs because of the radioactive elements originating in the sedimentary phosphate deposits. "Cement kiln dust" is a fine material produced during manufacture of Portland cement, which involves heating lime and siliceous materials (clay and shale) to a molten state in rotary ovens or kilns. Dust in the exhaust is removed by filter units and is composed of fine feed material, cement, and sloughed refractory materials. In 1995, there were 118 facilities in 37 states producing 80 million metric tons of cement and approximately 3–5 million metric tons of kiln dust. "Tailings" are fines remaining after crushing and extraction of ores during mineral separation and refining. Many different separation and refining processes result in a wide range of materials such as the clayey phosphate slimes and the red muds of aluminum (Al) metal processing.

Coal Combustion By-Products

“Fly ash” is derived when coal is burned for energy and is produced in larger amounts than any other single industrial waste. In 1998, approximately 60 million metric tons of fly ash as well as 20 million tons of “bottom ash” and “boiler slag” materials and 25 millions metric tons of “flue gas desulfurization” by-products were produced in United States. All of the above-named materials are derived when coal is burned and are, in their conglomerate, referred to as coal combustion by-products. Fly ash is a fine (typically less than 0.05 mm in diameter) largely siliceous material derived from coal. It is removed from the flue gases by special collectors and about 25% is recycled as a cement additive, structural fill, or for other purposes. Fly ashes are classified as Class C or Class F and the properties of each type of fly ash are primarily because of the coal source and furnace operating conditions. Class C fly ashes are higher in alkali metals and Class F ashes have high silicon and Al contents. Bottom ashes are sand-like granular substances, which are typically used as aggregate in cement blocks, for construction backfill, and for treatment of snow or icy roads. Boiler slag is coarse textured and shiny black in appearance and resembles crushed glass. Its major applications include use as blasting grit, aggregate, granules for roofing shingles asphalt, and snow and ice abrasive. Flue gas desulfurization (FGD) by-products are created as a result of the Clean Air Acts, which mandate removal (or scrubbing) of sulfur from the flue gases when coal is burned. They are often a mixture of coal ash, gypsum, and unreactive lime, although the composition can vary tremendously based on the type of technology used to scrub the sulfur. FGD gypsum has become a major high-volume industrial by-product.

Pulp and Paper Manufacture

The production of pulp for papermaking and as an industrial feedstock by the Kraft and related processes is the major industry in several regions of United States. The total production of pulp and paper-manufacture materials in 1989 was approximately 13 million metric tons. Material properties are highly variable because of the wide range of processes used in wood digestion and in handling of the resultant wastes.

Construction Wastes

A large amount of construction waste is generated each year in the United States waste wallboard, lumber, waste fill material, and other construction materials. For example, in home construction, approximately 0.5 kg of wallboard waste is generated per square foot of floor space resulting in the annual production of about 1 million metric tons of waste. Typical construction waste of a 2000 ft² home is estimated to be 3600 kg.^[3]

LAND APPLICATION OF INDUSTRIAL BY-PRODUCTS

The physical form of industrial wastes can vary greatly, ranging from liquid to fine powders to large-sized pieces of construction wastes. Not all industrial wastes are destined to be entombed in landfills. Many have properties that make them beneficial as raw materials for subsequent manufactured products.^[4] For example, coal combustion by-products are commonly included in the preparation of high quality, high strength concrete. When industrial wastes are used in such a manner, they are really no longer considered wastes, but are valued as raw materials to create another type of industrial activity.

Many industrial wastes also have properties that make them valuable as soil amendments.^[4] Organic by-products from some food processing plants, e.g., when applied to

Table 2 Properties of soil that can be either improved or degraded by the application of by-products to soil.

Soil properties affected	Reasons these properties are important
Soil physical properties	Indicators of retention and transport of water and chemicals, soil erosion, leaching, surface and subsurface water runoff, and soil productivity
Texture	
Topsoil depth	
Rooting zone depth	
Water infiltration rate	
Soil bulk density	
Water-holding capacity	
Erodibility	
Compaction	
Soil chemical properties	Defines the fertility of the soil and the potential to either positively or negatively affect the soil's quality and the quality of the air and water that interact with the soil
Soil organic matter concentration	
pH	
Electrical conductivity	
Available plant nutrients	
Available heavy metals	Indicates the soil's microbial ability to degrade organic compounds in soil and the soil's potential to cycle nutrients and to purify the soil and water. The more microbial diversity maintained in an ecosystem, the better the ecosystem's ability to adjust to changes
Sorption capacity	
Soil biological properties	
Microbial biomass size	
Microbial biomass activity (i.e., respiration)	
Microbial diversity	
Redox potential	
Ability to biodegrade organic materials	

Note: All of the properties listed in the table are useful in models that seek to integrate soil, landscape, and geographic variability for the purpose of assessing and predicting impacts of applying by-products to soil.

Source: Adapted from Sims & Pierzynski.^[5]

soil can affect a large number of properties and lead to an increase in soil quality. A summary of the more important physical, chemical, and biological properties that can be affected by land application of by-products is provided in Table 2.

A large number of studies have been conducted to determine the best management practices for use of by-products as a soil amendment. This includes information such as that presented in Table 2 and also involves methods of application and techniques related to the application process itself.

ROLE OF REGULATORY AGENCIES REGARDING DISPOSAL AND REUSE OF INDUSTRIAL WASTES

Many states require that a plan be submitted to the state regulatory agencies, which provides information on the type of material to be land applied, the properties of the material, the rates to be applied, and how the material will be transported and applied. The by-product must be applied without creating odors, must often be mixed into the soil with farm machinery to reduce risk of runoff if a heavy rain should occur soon after application, and often cannot be applied within certain distances from wells, streams, houses, or other more sensitive areas. This is not to imply that all industrial by-products are toxic. Like medicines,

properly handled by-products can also be beneficial. Guidelines are established, however, to ensure good stewardship of our soil–water–air resources. Many states have guidelines for land application of by-products posted on the Internet.

The Resources Conservation and Recovery Act (RCRA), enacted on October 21, 1976, regulates the most hazardous of industrial wastes “from cradle to the grave.” The statute requires EPA to establish minimum acceptable requirements for all aspects of hazardous wastes for generators and transporters as well as for treatment, storage, and disposal facilities. The regulatory framework established under Subtitle C of RCRA was designed to protect human health and the environment from the effects of improper management of hazardous waste. Determining what is hazardous waste is therefore a key activity because only those wastes that meet the definition of hazardous waste are subject to Subtitle C. This is also a complex issue. In general, a material is classified as hazardous if it exhibits one of the EPA-defined characteristics of a hazardous waste (i.e., corrosive, ignitable, reactive, and toxic). A hazardous waste, however, must also meet the definition of a solid waste, and a solid waste is “any discarded material that is not specifically excluded” from a listing of solid wastes. Materials excluded by the various definitions of wastes are listed in Table 3.

Solid waste does not refer to the material’s physical state per se as it is a regulatory term only. Non-hazardous wastes,

Table 3 Exclusions from Subtitle C of RCRA.

Excluded from solid waste definition	Excluded from hazardous waste definition	Excluded materials requiring special management
Domestic sewage	Household wastes	Product storage wastes
Mixtures of domestic sewage and wastes going to publicly owned treatment works	Agricultural wastes used as fertilizers	Waste identification samples
Industrial point source discharges under Section 402CWA	Mining overburden returned to site	Treatability samples
Irrigation return flows	Discarded wood treated with arsenic	Residues remaining in empty containers
Source, special nuclear, or by-product material under AED	Specific ore and mineral beneficiating wastes	Conditionally exempt small-quantity generator wastes
		Farm wastes (pesticides)
<i>In situ</i> mining waste	Fossil fuel combustion wastes	PCB-mixed wastes ^a
Reclaimed pulling liquors from Kraft paper process	Oil and gas exploration, development and production wastes	Low-level radioactive mixed waste ^a
Spent sulfuric acid used to produce new acid	Cement kiln dust	
Secondary materials returned to the original process under certain conditions	Petroleum-contaminated media and debris from underground tank cleanup	
Spent reused wood preservatives	Spent chlorofluorocarbon refrigerant	
Certain coke by-products	Used oil filters	
Splash condenser dross residue	Still bottoms from the re-refining of used oil	
Recovered refinery oil wastes		

^aThese wastes are not explicitly excluded from RCRA but require special management.

Source: Adapted from Wagner.^[6]

as defined by RCRA, are not necessarily guaranteed to be safe for dispersal into the environment. Potential hazards could include soil degradation, surface and groundwater contamination, direct exposure during handling, or indirect exposure such as plant uptake into the food chain can all result in harmful environmental and human health effects.

MINIMIZING PRODUCTION OF INDUSTRIAL WASTES

In the late 1980s, many groups argued for a defined approach to the prevention of wastes (including industrial wastes). These efforts translated into the Pollution Prevention Act of 1990 and development of a qualitative hierarchy as a means to evaluate various approaches to pollution reduction in terms of their adherence to a pollution provision theme. A simple version of this hierarchy (from most to least desirable) is as follows: 1) source reduction, i.e., use of cleaner feed materials and processes; 2) reuse and recycle, especially in-process recycling; 3) treatment; and 4) disposal.^[7]

The Chemical Manufacturers Association^[8] has developed a five-step approach, which is applicable to the design and analysis of chemical manufacturing plants and can be used in various stages of design. This approach can be extended to many different industries that generate waste:

1. Step 1: characterize process streams.
2. Step 2: evaluate environmental impacts and issues.
3. Step 3: identify pollution prevention opportunities.
4. Step 4: analyze alternatives.
5. Step 5: document results.

CONCLUSION

Increasing human population and expanded industrial activities make it imperative that we pay attention to the

creation, recycling, and disposal of industrial wastes. Our quality of life will largely depend on how we address this issue. While progress has been made, there is a long way to go before we will achieve a truly sustainable way of life.

REFERENCES

1. Miller, D.M.; Miller, W.P.; Dudka, S.; Sumner, M.E. Characterization of industrial by-products. In *Land Application of Agricultural, Industrial, and Municipal By-Products*; Power, J.F., Dick, W.A., Eds.; SSSA Book Series 6; Soil Science Society of America: Madison, 2001; 107–126.
2. U.S. Census Bureau. *Current Industrial Reports by NAICS Subsector*. Available at <http://www.census.gov/econ/www/industry.html>.
3. Sustainable Sources. *A Sourcebook for Green and Sustainable Building—Construction Waste*. Available at <http://www.greenbuilder.com/sourcebook/ConstructionWaste.html>.
4. Power, J.F.; Dick, W.A. *Land Application of Agricultural, Industrial, and Municipal By-Products*; SSSA Book Series 6; Soil Science Society of America: Madison, 2000.
5. Sims, J.T.; Pierzynski, G.M. Assessing the impacts of agricultural, municipal, and industrial by-products on soil quality. In *Land Application of Agricultural, Industrial, and Municipal By-Products*; Power, J.F., Dick, W.A., Eds.; SSSA Book Series 6; Soil Science Society of America: Madison, 2001; 237–261.
6. Wagner, T.P. *The Complete Guide to the Hazardous Waste Regulations—RCRA, TSCA, OSHA, and Superfund*, 3rd Ed.; Wiley: New York, 1999.
7. Radecki, P.P. *Proceedings of the 51st Purdue Industrial Waste Conference*; West Lafayette, IN, May 6–8, 1996; Wukasz, R.F., Ed.; Ann Arbor Press: Chelsea, 1997; 5–22.
8. Chemical Manufacturers Association. *Designing Pollution Prevention into the Process: Research, Development and Engineering*; Chemical Manufacturers Association: Washington, 1993.

Infiltration: Management

Jutta Rogasik

Kerstin Panten

Institute of Plant Nutrients and Soil Science, Federal Agricultural Research Centre (FAL), Berlin, Germany

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Helmut Rogasik

Leibnitz-Center for Agricultural Landscape and Land Use Research (ZALF) e.V., Berlin, Germany

Abstract

Water infiltration rate is an important soil property with a strong impact on the environment. Factors such as field traffic, crop rotation, tillage operations, and fertilizer management affect water infiltration rate into the soil. Farmers need to be advised and encouraged to keep the infiltration capacity of their soils as high as possible by all means of agricultural measures.

INTRODUCTION

Infiltration rate is the velocity at which water enters into the soil. It is usually measured by the height (mm) of a water column that can penetrate the soil in 1 hour. There is a difference between 1) the initial infiltration rate, which reflects the rapid entry of water into dry soil; and 2) the equilibrium infiltration rate, which reflects the water infiltration into the soil at steady-state conditions.

High water infiltration rate or infiltration capacity of soil is an important precondition to reduce the risk of water erosion, to replenish water storage capacity, and to diminish the risk of temporal flooding of soil during extraordinary precipitation events.^[1] Soil surface sealing, soil compaction, loss of biological activity, and decrease in soil organic matter content have unfavorable impact on infiltration rates.

SOIL SURFACE DEGRADATION

Soil sealing and crusting are caused by the energy of raindrops during precipitation events. Surface seals are thin crusts ranging from a fraction of a millimeter up to several centimeters. In comparison with intact soil, the surface seal is characterized by higher bulk density along with higher shear and penetrometer resistances, low hydraulic conductivity, and reduced gas exchange processes.^[2] Sealed soils have markedly restricted infiltration

capacities,^[3] e.g., a 40–60% decrease in tilled bare systems. Surface crusting can be minimized by using mulches.^[4]

LAND USE AND FERTILIZATION

Maintaining high infiltration capacity of soil is one of the most significant achievements of agriculture and is used as a soil quality indicator. Infiltration rate and soil organic matter content are primarily functions of the prevailing land use system and the duration of cultivation (Fig. 1).^[5,6]

Conversion of natural into arable ecosystem affects the fate of stored carbon and associated properties such as bulk density, soil structure, and water infiltration rate. Soil degradation is exacerbated when perennial crops are replaced by annual row crops because of increased soil stress and disturbance of soil structure by tillage. In addition, intense rainfall may induce soil crusting, reduce macroporosity, and increase hard setting upon drying. Ley farming, a system of growing forages in rotation with agricultural crops for improving soil quality and sustaining grain and fodder production, is a viable alternative to enhance infiltration capacity.^[7]

Adopting complex crop rotations contributes to soil carbon sequestration and reduces dry bulk density through increased inputs of plant residues, longer periods of soil coverage, and less soil disturbance, resulting in enhanced infiltration rates (Fig. 2).^[6,8]

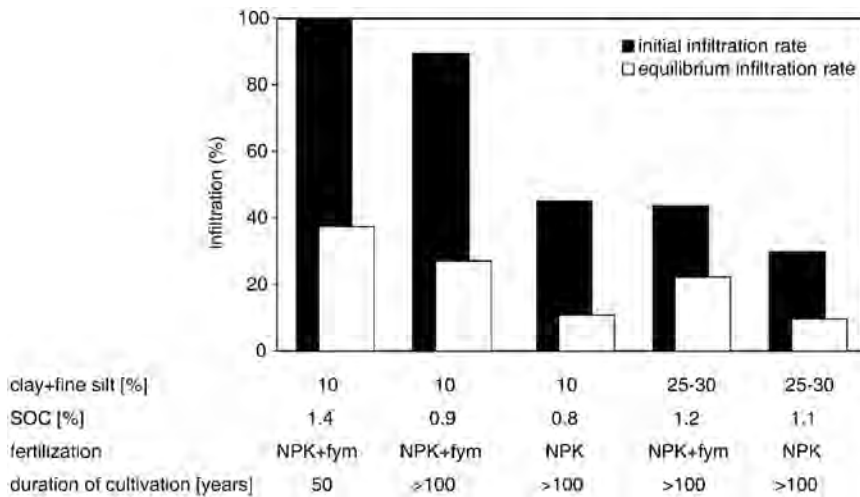


Fig. 1 Effect of management and soil properties on infiltration rates (measurement: summer 2003 with double ring infiltrometer in Germany and Romania; 100% = 332 mm/hr; fym: farm yard manure).

BASE SATURATION

Base saturation with Ca^{2+} and Mg^{2+} is an important factor influencing the stability of soil aggregates, so liming is a key measure in agriculture for the protection of soil structure. Nevertheless, a global decline of calcium balance in cultivated soils is widely observed. Liming requirement (adequate supply of Ca^{2+} and Mg^{2+} to balance leaching losses) is estimated indirectly from soil pH. However, other cations (NH_4^+ and K^+) affect soil structure and infiltration adversely (e.g., as a result of excessive liquid manure application). Soil pH alone cannot be used to adequately assess

the liming requirements to compensate for these effects. Research is urgently needed to accurately assess the lime requirement of soils.^[5]

DEGRADATION BY SALINIZATION

Soil salinization degrades soil structure, predominantly because of the replacement of Ca^{2+} and Mg^{2+} by Na^+ , resulting in decreased infiltration, increased surface hardness, and erosion by both rainfall and overland flow. Regeneration processes to increase the water infiltration

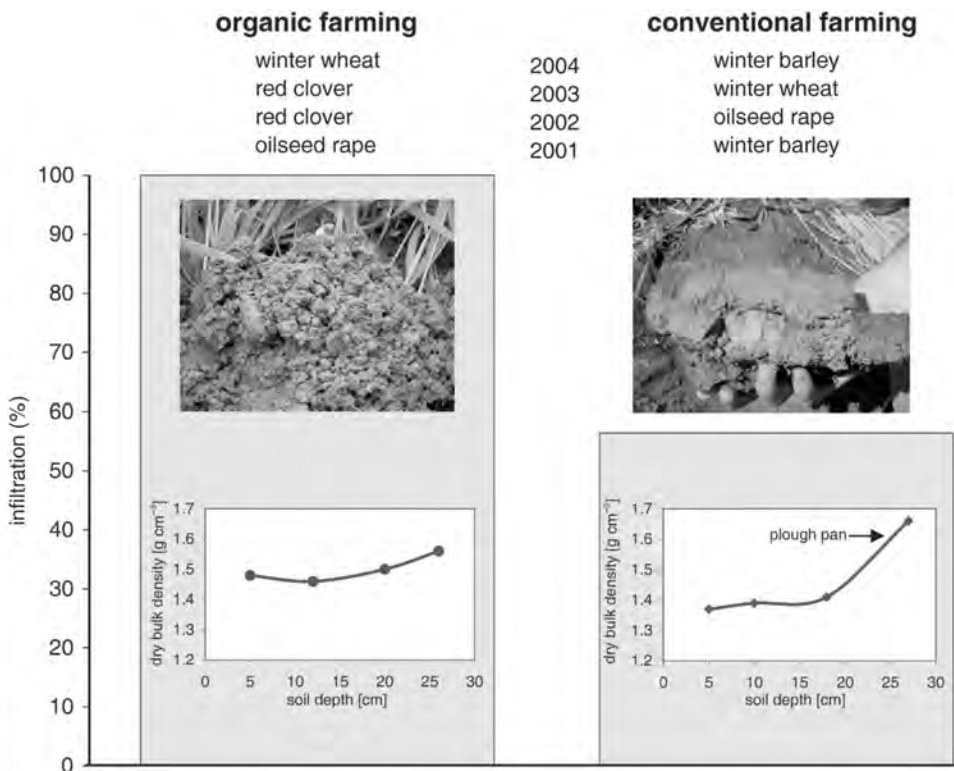


Fig. 2 Crop rotation, dry bulk density, and infiltration rate for different farming systems (measurement: spring 2004 with hood infiltrometer on the Trenthorst site, Germany; 100% = 179 mm/hr).

Source: From Schnug, Rogasik et al.^[8]

in saline soils can be initiated using gypsum together with deep ripping.^[9] Applying xenobiotic substances (e.g., acrylamides) to soils or with irrigation water can initiate the restorative processes^[10] but only when used in conjunction with gypsum.

SOIL TILLAGE SYSTEMS

Tillage methods significantly affect the soil's physical properties. Conversion from conventional to conservation tillage reduces soil disturbance and enhances soil resistance to compaction by vehicular traffic. Additionally, earthworm activity is improved by greater sources of food and undisturbed burrows.^[11,12] Earthworms create stable vertically oriented macropores with high continuity and connectivity,^[13,14] which are preferential water flow paths and thus increase infiltration rates.^[15]

Intensive soil tillage disrupts fauna and soil aggregates followed by the loss of mechanical stability, reduction in infiltration capacity, and increase in susceptibility against competitive forces (Fig. 3).

Reduced tillage intensity and the maintenance of residue cover ensure high soil structural stability and diminish surface sealing, preventing the formation of plow or traffic pans and increasing water infiltration capacity.^[17]

TRAMPLING

Changes in soil structure from cattle or sheep hoof compaction can cause severe problems of soil degradation on rangelands. Inappropriate grazing management (high stocking rates and continuous grazing) can increase the dry bulk density and tensile strength and decrease water

infiltration rates because of a reduction in macroporosity. Regeneration of soils requires the adoption of low stocking rates,^[18] controlled grazing, and no grazing during periods of high soil moisture contents, which is especially important under irrigated conditions.^[19]

SUBSOIL COMPACTION CONTROL

Plant roots, especially those of perennial species (e.g., alfalfa), can ameliorate compacted subsoil through bioturbation. Furthermore, growing cover crops can also improve soil structure.^[7] The prevention of field traffic-induced subsoil compaction includes management to enhance soil stability (reducing depth of tillage, minimum tillage, direct drilling, and minimizing soil loosening), regenerate soil structure (green manuring, providing continuous vegetal cover, supplying sufficient organic matter, and growing deep-rooted cover crops), avoid subsoil compaction by reducing in-furrow plowing, use appropriate machinery with limited wheel or axle loads, change to running gear with restricted ground contact pressures, and ensure controlled traffic.^[20]

CONCLUSION

Water infiltration rate is an important soil property with a strong impact on the environment. Factors such as field traffic, crop rotation, tillage operations, and fertilizer management (lime status and soil organic carbon) affect water infiltration rate into the soil. Farmers need to be advised and encouraged to keep the infiltration capacity of their soils as high as possible by all means of agricultural measures.

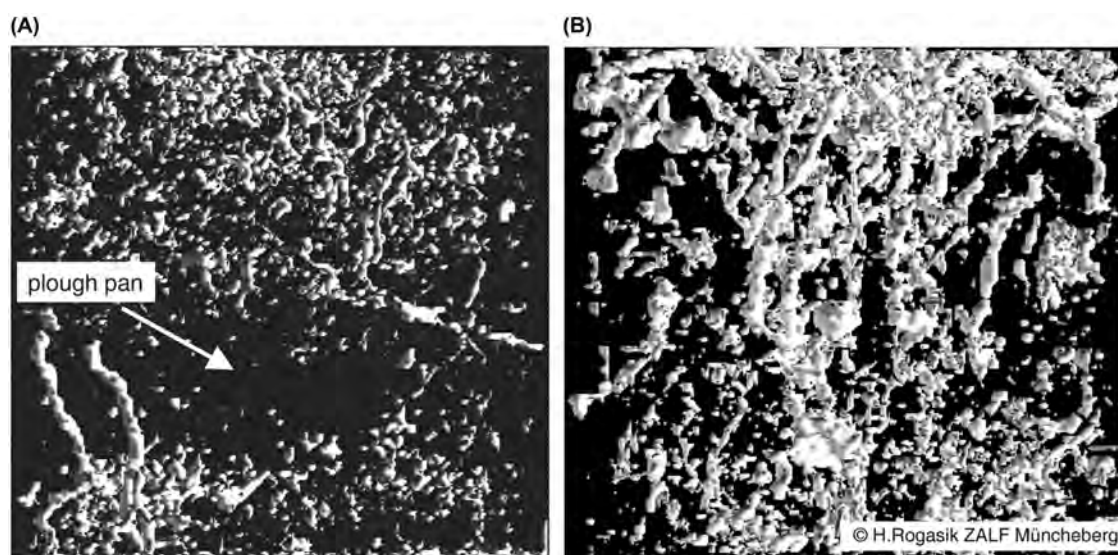


Fig. 3 The comparison of connectivity of macropores for conventional (A) and conservation tillage (B).

Source: From Rogasik, Brunotte, et al.^[16]

REFERENCES

1. Sparovek, G.; de Jong van Lier, Q.; Marcinkonis, S.; Rogasik, J.; Schnug, E. A simple model to predict river floods—A contribution to quantify the significance of soil infiltration rates. *Landbauforschung Völkenrode* **2002**, *52* (3), 187–194.
2. Mualem, Y.; Assouline, S.; Rohdenburg, H. Rainfall induced soil seal, a critical review of observations and models. *Catena* **1990**, *17*, 185–203.
3. Singer, M.J.; Le Bissonnais, Y. Importance of surface sealing in the erosion of some soils from a Mediterranean climate. *Geomorphology* **1998**, *24* (1), 79–85.
4. Dormaar, J.F.; Carefoot, J.M. Implications of crop residue management and conservation tillage on soil organic matter. *Can. J. Plant Sci.* **1996**, *76* (4), 627–634.
5. Schnug, E.; Haneklaus, S. Landwirtschaftliche Produktionstechnik und Infiltration von Böden—Beitrag des ökologischen Landbaus zum vorbeugenden Hochwasserschutz. *Landbauforschung Völkenrode* **2002**, *4*, 197–203.
6. Rogasik, J.; Schroetter, S.; Funder, U.; Schnug, E. Long-term fertilizer experiments as a data base for calculating the carbon sink potential of arable soils. *Arch. Agron. Soil Sci.* **2004**, *50* (1), 11–19.
7. Bell, M.J.; Bridge, B.J.; Harch, G.R.; Orange, D.N. Physical rehabilitation of degraded krasnozems using ley pastures. *Aust. J. Soil Res.* **1997**, *35* (5), 1093–1113.
8. Schnug, E.; Rogasik, J.; Panten, K.; Paulsen, H.M.; Haneklaus, S. Ökologischer Landbau erhöht die Versickerungsleistung von Böden. *Ökologie Landbau*. **2004**, *132* (4), 53–55.
9. Hamza, M.A.; Anderson, W.K. Responses of soil properties and grain yields to deep ripping and gypsum application in a compacted loamy sand soil contrasted with a sandy clay loam soil in Western Australia. *Aust. J. Agric. Res.* **2003**, *54* (3), 273–282.
10. Vacher, C.A.; Loch, R.J.; Raine, S.R. Effect of polyacrylamide additions on infiltration and erosion of disturbed lands. *Aust. J. Soil Res.* **2003**, *41* (8), 1509–1520.
11. Moreno, F.; Pelegrin, F.; Fernandez, J.E.; Murillo, J.M. Soil physical properties, water depletion and crop development under traditional and conservation tillage in southern Spain. *Soil Till. Res.* **1997**, *41* (1, 2), 25–42.
12. Lamande, M.; Hallaire, V.; Curmi, P.; Peres, G.; Cluzeau, D. Changes of pore morphology, infiltration and earthworm community in a loamy soil under different agricultural managements. *Catena* **2003**, *54* (3), 637–649.
13. Pitkanen, J.; Nuutinen, V. Earthworm contribution to infiltration and surface runoff after 15 years of different soil management. *Appl. Soil Ecol.* **1998**, *9* (1–3), 411–415.
14. Mooney, S.J. Three-dimensional visualization and quantification of soil macroporosity and water flow patterns using computed tomography. *Soil Use Manage.* **2002**, *18* (2), 142–151.
15. Tebrügge, F.; During, R.A. Reducing tillage intensity—a review of results from a long-term study in Germany. *Soil Till. Res.* **1999**, *53* (1), 15–28.
16. Rogasik, J.; Brunotte, J.; Rogasik, H.; Schroetter, S.; Funder, U.; Schnug, E. Effects of conservation tillage practices and different fertilization on soil fertility and yield. *Jahresbericht FAL Braunschweig*, **2002**, 17.
17. Dimanche, P.H.; Hoogmoed, W.B. Soil tillage and water infiltration in semi-arid Morocco: The role of surface and sub-surface soil conditions. *Soil Till. Res.* **2002**, *66* (1), 13–21.
18. Usman, H. Cattle trampling and soil compaction effects on soil properties of a northeastern Nigerian sandy loam. *Arid Soil Res. Rehab.* **1994**, *8* (1), 69–75.
19. Proffitt, A.P.B.; Jarvis, R.J.; Bendotti, S. The impact of sheep trampling and stocking rate on the physical properties of a red duplex soil with 2 initially different structures. *Aust. J. Agr. Res.* **1995**, *46* (4), 733–747.
20. Alakukku, L.; Weisskopf, P.; Chamen, W.C.T.; Tijink, F.G. J.; van der Linden, J.P.; Pires, S.; Sommer, C.; Spoor, G. Prevention strategies for field traffic-induced subsoil compaction: A review. Part 1. Machine/soil interactions. *Soil Till. Res.* **2003**, *73* (1–2), 145–160.

Infiltration: Properties

Walter J. Rawls

Hydrology Lab, U.S. Department of Agriculture—Agricultural Research Service
(USDA-ARS), Beltsville, Maryland, U.S.A.

Abstract

This entry reviews infiltration and standard modeling strategies. Models to characterize the infiltration for field applications usually employ simplified concepts, which predict the infiltration rate or cumulative infiltration volume and assume that surface ponding begins when the surface application rate exceeds the soil surface infiltration rate.

INTRODUCTION

Infiltration is commonly defined as the process of water entry at the land surface into a soil from a source such as rainfall, irrigation, or snowmelt. The rate of infiltration is generally controlled by the rate of soil water movement below the surface. Factors that affect infiltration have been divided into categories of soil, surface, management, and natural properties.^[1] These factors should be taken into account in the infiltration model parameters. Soil factors encompass soil particle size and morphological, chemical, and soil water properties.^[2] Surface factors are those that affect the movement of water through the air–soil interface such as soil crust. Management systems involve different types of tillage, vegetation, and surface cover.^[2,3] Natural factors include such things as precipitation, soil temperature, residue, and soil moisture, which vary with time and space, and interact with other factors in their effect on infiltration.

INFILTRATION MODELS

Models to characterize infiltration for field applications usually employ simplified concepts, which predict the infiltration rate or cumulative infiltration volume and assume that surface ponding begins when the surface application rate exceeds the soil surface infiltration rate. The evolution of infiltration modeling has taken three directions: the empirical, the approximation to the physically based models, and the physically based approach. Most of the empirical and approximate models treat the soil as a semi-infinite medium with the soil saturating from the surface down. Physically based models specify appropriate boundary conditions and normally require detailed data input. The Richards equation is the physically based infiltration equation primarily used for describing water flow in soils; however, it has not been used in operational infiltration models.

The empirical models generally relate infiltration rate or volume to elapsed time modified by certain soil properties.^[4] Parameters used in these models are commonly estimated from measured infiltration rate–time relationships for a given soil condition. The three most common empirical models were proposed by Kostiakov,^[5] Horton,^[6] and Holtan.^[7] The most common physically based models were proposed by Green and Ampt,^[8] Phillip,^[9] Morel-Seytoux and Khanji,^[10] and Smith and Parlange.^[11]

MODEL PARAMETERS

The most common ways to determine infiltration parameters are to fit models to experimental infiltration data (infiltration rate as it varies with time) or to develop relationships between readily available data such as soils and land use data and the infiltration model parameters.

Experimental infiltration measurements are normally made by applying water at a specific site to a finite area and measuring the intake rate of the soil. There are four types of infiltrometers, namely: the *ponded water ring* or *cylinder* type, the *sprinkler* type, the *tension* type, and the *furrow* type. When choosing an infiltrometer, one must choose the one that replicates the system being investigated. For example, ring infiltrometers should be used to determine infiltration rates for inundated soils such as flood irrigation or pond seepage. Sprinkler infiltrometers should be used where the effect of rainfall on surface conditions influences the infiltration rate. Tension infiltrometers are used to determine the infiltration rates of the soil matrix in the presence of macropores. Furrow infiltrometers are used when the effect of flowing water is important such as furrow irrigation.^[12]

The Green–Ampt^[8] model is typical of physically based models and has probably had the most effort directed toward estimating the parameters from readily available data. Also, as shown in [Table 1](#), the Green–Ampt parameters can

Table 1 Physically based infiltration capacity models, assuming immediate ponding.

Infiltration rate equation	Parameters	Parameter estimation	References
$i = K_s \left(1 + \frac{(\phi - \theta_i) H_f}{I} \right)$	K_s , saturated hydraulic conductivity; H_f , wetting front suction; and ϕ , porosity		
$i = St^{-1/2} + A$	S , sorptivity; and A , parameter with dimension of conductivity	$H_f = \frac{2+3\lambda}{1+3\lambda} \left(\frac{h_b}{2} \right)$	[8]
$i = \frac{\tilde{K}}{B} \left(\frac{(\phi - \theta_i)(H_0 + H_c) + 1}{I} \right)$	K , hydraulic conductivity at natural saturation; B , viscous resistance correction factor; H_c , effective capillary drive; and ϕ , porosity	$S = (2(H_0 + H_f)(\phi - \theta_i)(K_s))^{1/2}$ $A = \alpha(K_s)$ α ranges 0.33 – 1, (recommend 1)	[9]
$i = K_s \left(\frac{C}{K_{sI}} + 1 \right)$	C , sorptivity; K_s , saturated hydraulic conductivity; ϕ , porosity; and θ_i , initial water content	$\tilde{K} = K_s$ $1 \leq B \leq 1.7$ $H_c = H_f$	[10]
$i = \frac{K_s e^{(I(K_s)/C)}}{e^{(I(K_s)/C-1)}}$	i , infiltration rate (cm/hr); I , cumulative infiltration depth (cm); θ_i , initial water content (vol. fraction); H_0 , depth of ponded water (cm); λ , Brooks–Corey pore size index; and h_b , Brooks–Corey bubbling capillary pressure	$C = [2(\phi - \theta_i)H_f K_s]^{1/2}$	[11]

Source: From Rawls & Brakensiek.^[13]

be used to estimate the parameters for most of the physically based models.^[13]

The Green–Ampt^[8] model is an approximate model utilizing Darcy’s law. The original model was developed for ponded infiltration into a deep homogenous soil with a uniform initial water content. Water is assumed to infiltrate into the soil as piston flow resulting in a sharply defined wetting front that separates the wetted and unwetted zones. When neglecting the depth of ponding at the surface, the Green–Ampt rate equation is as follows

$$f = K(1 + (\phi - \theta_i)H_f/F) \quad (1)$$

where K is the effective hydraulic conductivity (cm/h), H_f is the effective suction at the wetting front (cm), ϕ is the soil porosity (cm³/cm³), θ_i is the initial water

content (cm³/cm³), F is accumulated infiltration (cm), and f is the infiltration rate (cm/h).

The Green–Ampt parameters can be estimated from the readily available data such as soils and land use data,^[14] e.g., average values for the Green–Ampt wetting front suction, S_f ; saturated hydraulic conductivity, K_s ; and porosity, ϕ , are given in Table 2 for the 11 U.S. Department of Agriculture soil textures. These values can be used as a first estimate; however, if more detailed soil properties are available, more refined estimates can be made.

Porosity

Table 2 gives the estimates of porosity for soil textures; however, the porosity, ϕ , can be estimated from measured

Table 2 Green–Ampt parameters.

Soil texture class	Porosity, ϕ	Wetting front soil suction head, S_f	Saturated hydraulic conductivity, K_s
Sand	0.437 (0.374–0.500)	4.95 (0.97–25.36)	23.56
Loamy sand	0.437 (0.363–0.506)	6.13 (1.35–27.94)	5.98
Sandy loam	0.453 (0.351–0.555)	11.01 (2.67–45.47)	2.18
Loam	0.463 (0.375–0.551)	8.89 (1.33–59.38)	1.32
Silt loam	0.501 (0.420–0.582)	16.68 (2.92–95.39)	0.68
Sandy clay loam	0.398 (0.332–0.464)	21.85 (4.42–108.0)	0.30
Clay loam	0.464 (0.409–0.519)	20.88 (4.79–91.10)	0.20
Silty clay loam	0.471 (0.418–0.524)	27.30 (5.67–131.50)	0.20
Sandy clay	0.430 (0.370–0.490)	23.90 (4.08–140.2)	0.12
Silty clay	0.479 (0.425–0.533)	29.22 (6.13–139.4)	0.10
Clay	0.475 (0.427–0.523)	31.63 (6.39–156.5)	0.06

Note: Numbers in parentheses are the 25% and 75% ranges.

Source: Adapted from Rawls, Brakensiek, et al.^[15]

bulk density or from soil particle size data (% sand and % clay), % organic matter, and cation-exchange capacity of the clay (an indicator of the shrink–swell capacity of the clay).^[16] Coarse fragments (>2 mm in size) in the soil affect the porosity, and adjustments should be made.^[17]

Initial Water Content

The initial water content (θ_i) should be measured, or it can be estimated from moisture–retention relationships.^[14] Good estimates of wet, average, and dry initial water contents are the water content held at –10, –33, and –1500 kPa, respectively.

Wetting Front Suction

The Green–Ampt wetting front suction parameter (H_f) can be estimated from the parameters of the Brooks–Corey water retention equation:^[14]

$$H_f = ((2 + 3\lambda)/(1 + 3\lambda))(h_b/2) \quad (2)$$

where H_f is the Green–Ampt wetting front suction (cm), λ is the Brooks–Corey pore size distribution index, and h_b is the Brooks–Corey bubbling pressure.

Rawls and Brakensiek further simplified the above equation by relating the Green–Ampt wetting front suction parameter to soil properties in the following equation:

$$\begin{aligned} H_f = & \exp[6.53 - 7.326(\phi) + 0.00158(C^2) \\ & + 3.809(\phi^2) + 0.000344(S)(C) \\ & - 0.04989(S)(\phi) + 0.0016(S^2)(\phi^2) \\ & + 0.0016(C^2)(\phi^2) - 0.0000136(S^2)(C) \\ & - 0.00348(C^2)(\phi) - 0.000799(S^2)(\phi)] \end{aligned} \quad (3)$$

where S is the % sand, C is the % clay, and ϕ is the porosity (vol.%).

Effective Hydraulic Conductivity

The effective hydraulic conductivity can be obtained from measurements or from saturated hydraulic conductivity prediction techniques developed from physical and hydraulic soil properties and modified for management and natural conditions. If only soil texture classes are available, the saturated hydraulic conductivity can be obtained from Table 2 and, as a rule of thumb, the effective hydraulic conductivity can be taken as half of the saturated hydraulic conductivity. There are various techniques for estimating saturated hydraulic conductivity from soil properties^[18,19] or characteristics of the water retention curve.^[15,20,21] Other techniques are given in the hydraulic conductivity section.

The effective hydraulic conductivity is determined by modifying the saturated hydraulic conductivity to incorporate the effects of such factors as coarse

fragments (>2 mm in size),^[17] soil freezing,^[14] soil crusts,^[22] and macropores.^[23]

A practical approach to handling the various conditions is to set the effective hydraulic conductivity equal to the saturated hydraulic conductivity (K_s) times a macroporosity factor.^[12] Brakensiek and Rawls^[1] developed two macroporosity factors for areas that do not undergo mechanical disturbance on a regular basis (e.g., rangeland) and one for areas that do (e.g., agricultural areas). For the bare area outside a plant canopy, the soil is assumed to be crusted, and the effective hydraulic conductivity is equal to the saturated hydraulic conductivity (K_s) times a crust factor.

CONCLUSION

Defining infiltration parameters that represent the conditions at a point are well established; however, defining how these parameters vary both temporally and spatially has only been defined in a very elementary way, and an elaborate research is needed in these areas.

REFERENCES

1. Brakensiek, D.L.; Rawls, W.J. Effects of agricultural and rangeland management systems on infiltration. In *Modeling Agricultural, Forest, and Rangeland Hydrology*, Proceedings of the 1988 International Symposium, December 12–13, 1988; American Society of Agricultural Engineers: St. Joseph, 1988; 247 pp.
2. Rawls, W.J.; Brakensiek, D.L.; Soni, B. Agricultural management effects on soil water processes: part I. Soil water retention and Green–Ampt parameters. *Trans. Am. Soc. Agr. Eng.* **1983**, *26* (6), 1747–1752.
3. Rawls, W.J.; Brakensiek, D.L.; Savabi, R. Infiltration parameters for rangeland soils. *J. Range Manage.* **1989**, *42* (2), 139–142.
4. Skaggs, R.W.; Khaleel, R. Infiltration. In *Hydrologic Modeling of Small Watersheds*; Haan, C.T., Brakensiek, D.L., Eds.; ASAE Monogr. No. 5; American Society of Agricultural Engineers: St. Joseph, 1982; 121–166.
5. Kostikov, A.N. On the dynamics of the coefficient of water percolation in soils and on the necessity for studying it from a dynamic point of view for purposes of amelioration. *Trans. Sixth Commun. Int. Soil Sci. Soc.* **1932**, 17–21.
6. Horton, R.E. An approach toward a physical interpretation of infiltration-capacity. *Soil Sci. Soc. Am. J.* **1940**, *5*, 399–417.
7. Holtan, H.N. *A Concept for Infiltration Estimates in Watershed Engineering*; U.S. Department of Agriculture: Washington, 1961; 34 pp.
8. Green, W.H.; Ampt, G.A. Studies on soil physics: 1. Flow of air and water through soils. *J. Agr. Sci.* **1911**, *4*, 1–24.
9. Philip, J.R. The theory of infiltration: 1. The infiltration equation and its solution. *Soil Sci.* **1957**, *83*, 345–357.
10. Morel-Seytoux, H.J.; Khanji, J. Derivation of an equation of infiltration. *Water Resour. Res.* **1974**, *10* (4), 795–800.

11. Smith, R.E.; Parlange, J.Y. A parameter-efficient hydrologic infiltration model. *Water Resour. Res.* **1978**, *14* (3), 533–538.
12. Rawls, W.J.; Ahuja, L.R.; Brakensiek, D.L.; Shirmohammadi, A. Infiltration and soil water movement. In *Handbook of Hydrology*; Maidment, D.R., Ed.; McGraw-Hill: New York, 1993; 5.1–5.51.
13. Rawls, W.J.; Brakensiek, D.L. Estimation of soil retention and hydraulic properties. In *Unsaturated Flow in Hydrologic Modeling*; Morel-Seytoux, H.J., Ed.; NATO ASI Series C: Mathematical and Physical Sciences; Kluwer Academic Publishers: Dordrecht, 1989; Vol. 275, 275–300.
14. Rawls, W.J.; Brakensiek, D.L. A procedure to predict Green and Ampt infiltration parameters. In *ASAE Conference on Advances in Infiltration*; American Society of Agricultural Engineers: St. Joseph, 1983; 102–112.
15. Rawls, W.J.; Brakensiek, D.L.; Logsdon, S.D. Predicting saturated hydraulic conductivity utilizing fractal principles. *Soil Sci. Soc. Am. J.* **1993**, *57*, 1193–1197.
16. Rawls, W.J. Estimating soil bulk density from particle size analysis and organic matter content. *Soil Sci.* **1983**, *135* (2), 123–125.
17. Brakensiek, D.L.; Rawls, W.J.; Stephenson, G.R. Determining the saturated hydraulic conductivity of soil containing rock fragments. *Soil Sci. Soc. Am. J.* **1986**, *50* (3), 834–835.
18. Saxton, K.E.; Rawls, W.J.; Romberger, J.S.; Papendick, R.I. Estimating generalized soil water characteristics for texture. *Soil Sci. Soc. Am. J.* **1986**, *50*, 1031–1036.
19. Rawls, W.J.; Brakensiek, D.L. Prediction of soil water properties for hydrologic modeling. In *Watershed Management in the Eighties*; Jones, E.B., Ward, T.J., Eds.; ASCE: New York, 1985; 293–299.
20. Ahuja, L.R.; Bruce, R.R.; Cassel, D.K.; Rawls, W.J. Simpler field measurement and estimation of soil hydraulic properties and their spatial variability for modeling. In *Proceedings of Modeling Agricultural Forest and Rangeland Hydrology*; ASAE Pub. 07-88; American Society of Agricultural Engineers: St. Joseph, 1988; 19–33.
21. Rawls, W.J.; Gimenez, D.; Grossman, R. Use of soil texture, bulk density, and the slope of the water retention curve to predict saturated hydraulic conductivity. *T. Am. Soc. Agr. Eng.* **1998**, *41* (4), 983–988.
22. Rawls, W.J.; Brakensiek, D.L.; Simanton, J.R.; Kohl, K.D. Development of a crust factor for the Green–Ampt model. *T. Am. Soc. Agr. Eng.* **1990**, *33*, 1224–1228.
23. Rawls, W.J.; Brakensiek, D.L.; Logsdon, S.D. Estimation of macropore properties for no till soils. *T. Am. Soc. Agr. Eng.* **1996**, *39* (1), 91–95.

Inorganic Carbon: Analysis

Richard R. Drees

C. Tom Hallmark

Department of Soil and Crop Sciences, Texas A&M University, College Station, Texas, U.S.A.

Abstract

Numerous methods for determining soil inorganic carbon exist. The nature of the soil investigation and the intended use of carbonate data govern the technique of choice. This entry presents techniques that are simple and accurate, requiring commonly available laboratory apparatus and reagents.

INTRODUCTION

Soil inorganic carbon (SIC) exists primarily as carbonates of calcium [Ca; e.g., calcite (CaCO_3)] and calcium plus magnesium [Mg; e.g., dolomite ($\text{CaMg}(\text{CO}_3)_2$)]. Carbonates may be pedogenic (i.e., formed in the soil) or lithogenic (i.e., inherited from calcareous parent materials). CaCO_3 is dominant in pedogenic environments, although $\text{CaMg}(\text{CO}_3)_2$ has been reported to occur under certain conditions.^[1] Many animal shells are also calcareous—often aragonite, which is another form of CaCO_3 . Other unique carbonate minerals may exist in soil, but regional geologic settings or unique environmental conditions govern their existence.

Soil is a major reservoir of inorganic carbon, containing more carbon than the atmosphere and biosphere combined.^[2] The estimates of SIC are important in evaluating carbon reserves, rates of carbon sequestration, or loss from the soil system. Also, the presence of carbonates has a major influence on soil chemical and physical reactions.

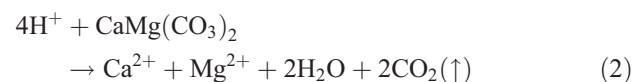
There are numerous procedures for SIC determination, and many excellent reviews exist.^[3–6] Each technique uses unique physical, chemical, or mineralogical properties of carbonates. It is not our intent to repeat the details of these procedures, but rather to present the fundamental basis of analysis and to focus on commonly used procedures in soil investigations.

CHEMICAL REACTIONS

Carbonates dissolve in strong acids. Ideal reaction equations are:



or



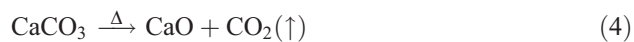
These reactions can be used to determine carbonates by measuring the weight loss due to carbonate dissolution, amount of carbon dioxide (CO_2) produced, or the amount of acid consumed in the reaction. The reaction rate of calcite 1 is much greater than $\text{CaMg}(\text{CO}_3)_2$ (Eq. 2) when particle size and surface area are standardized,^[6] which permits quantitative determination of the two minerals. For techniques that cannot differentiate CaCO_3 from $\text{CaMg}(\text{CO}_3)_2$, results are reported as CaCO_3 equivalent.

Gravimetric Techniques

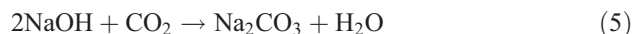
The reaction of carbonates with a strong acid releases CO_2 , resulting in a weight loss (Eqs. 1 and 2). The weight loss is a direct measure of carbonates.^[3,5]

$$\% \text{CaCO}_3 \text{ equivalent} = \frac{\text{g CO}_2 \text{ lost}}{\text{g oven-dried soil}} \times 227.3 \quad (3)$$

Weight gained by trapping evolved CO_2 gas with an absorbent is another common technique for carbonate determination.^[6] There are two methods of generating CO_2 for gravimetric techniques: 1) treating carbonates with acid (Eqs. 1 and 2) and 2) heating the sample (dry combustion) to 1000°C after organic carbon removal to release carbonate CO_2 ^[7] in the reaction:



In both instances, evolved CO_2 is passed through a series of traps to absorb water and other reaction constituents before the gas passes through a Nesbitt absorption bulb containing sodium hydroxide (NaOH) to absorb CO_2 and Mg perchlorate to retain the water generated in the reaction.^[6]



The calculations are the same as in Eq. 3, except that weight gain replaces weight loss. For the acid reaction, if the weight gain/loss is monitored over time, $\text{CaMg}(\text{CO}_3)_2$

content can be determined due to its slower reaction rate. However, it is not possible to differentiate CaCO_3 and $\text{CaMg}(\text{CO}_3)_2$ by the dry combustion method.^[7]

Volume/Pressure Techniques

Measurements of gas evolution are relatively rapid and are often used to determine soil carbonates. The common techniques used are a volumetric measure of evolved CO_2 (constant pressure) and an increase in pressure (constant volume). For the constant pressure procedure (commonly called the Chittick procedure), soil is treated with a strong acid, and the volume of released CO_2 is directly correlated with carbonate content.^[6,8] Fig. 1 illustrates a constant pressure apparatus. A finely ground sample is weighed to the nearest 0.1 mg and placed into a decomposition flask. By a stopcock (S in Fig. 1), the reaction vessel is closed to the atmosphere just before introducing the acid.

The leveling bulb and burette contain a saturated salt solution with a red indicator dye for easy reading.^[3] As the acid is added, the sample is stirred to provide a uniform

reaction, and the leveling bulb is constantly adjusted to maintain a slight negative pressure. After carbonates have reacted, the volume of solution displaced is measured. Timing of the reaction is important because CaCO_3 and $\text{CaMg}(\text{CO}_3)_2$ have different reaction rates. The volume change after 30 seconds is often used as a measure of CaCO_3 , and any volume increase between 30 seconds and 2 hours is attributed to $\text{CaMg}(\text{CO}_3)_2$. Adjustments are made for changes in temperature and atmospheric pressure during the reaction. Calculations to convert gas volume to weight percentage carbonates are given by several authors.^[5,6]

When carbonates react with acid at constant temperature and volume, the gas released causes an increase in gas pressure that is directly related to carbonate content.^[9–13] A pressure gauge, pressure transducer, or manometer is used to monitor the change in pressure over time. Pressure transducers can be connected to a personal computer for instantaneous data acquisition. Pressure change over time can be used to quantify CaCO_3 and $\text{CaMg}(\text{CO}_3)_2$ and has been shown to be highly sensitive and rapid.^[13]

The change of volume or pressure with CO_2 evolution has certain limitations. The solubility of CO_2 in acid varies with temperature, the type of acid used, acid concentration, and barometric pressure.^[3,6] Calibration with pure CaCO_3 and $\text{CaMg}(\text{CO}_3)_2$ standards is required in both the constant pressure and constant volume system for accurate results.

Titrimetric Techniques

Carbonates are the only major soil constituents that readily react with acids. The amount of acid consumed (Eqs. 1 and 2) during reaction with carbonates can be determined by titrating the unreacted acid to a common end point with a base. Normally, 50 ml of standardized 0.5 N HCl is carefully introduced to 5–25 g of soil, and the soil acid mixture is gently boiled for 5 minutes and then allowed to cool. The suspension is filtered and washed to remove all acid. The filtrate is then titrated with standardized 0.25 N NaOH to the phenolphthalein end point^[3,14] to determine the amount of acid consumed:

% CaCO_3 equivalent

$$= 100 \times \frac{(\text{Meq HCl added} - \text{Meq NaOH used})}{\text{g oven-dried soil}} \times 0.05 \text{ g CaCO}_3/\text{Meq} \quad (6)$$

X-RAY DIFFRACTION

Carbonates are crystalline minerals that yield characteristic X-ray diffraction peaks. The intensity of the diffraction peak should be proportional to the quantity of carbonate present. Accuracy is dependent on carbonate crystallinity, particle size, chemical composition, and sample orientation.^[15] Excellent results can be attained

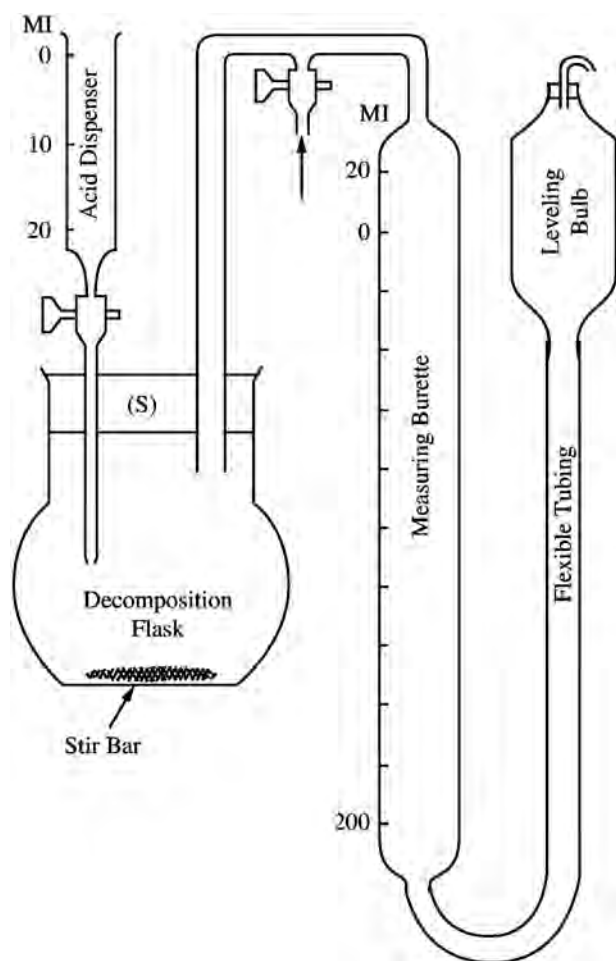


Fig. 1 Constant pressure apparatus for carbonate determination. Source: From Dreimanis.^[8]

using mineral standards [CaCO_3 or $\text{CaMg}(\text{CO}_3)_2$] and an internal reference standard.^[16] A calibration curve is constructed that plots the internal standard/mineral standard intensity ratio against the weight percentage of the mineral standard.^[16] In most instances, carbonate-containing samples and reference standards must be ground to produce a uniform, small particle size. Increased grinding time generally yields a decrease in error.^[17]

X-ray diffraction has proved to be a reliable semiquantitative method for the determination of $\text{CaMg}(\text{CO}_3)_2$ in carbonate rocks and sediments even without an internal standard.^[18] Peak intensities corrected for absorption also achieved accurate results in carbonate rocks,^[19] but it is unclear how this method would work for samples of low carbonate concentration. Due to slight differences in diffraction intensity between carbonates, it is recommended that reference carbonates be as similar as possible to those contained in the soil being analyzed.^[15]

Mg may substitute for Ca within the CaCO_3 structure up to approximately 10 mole%. X-ray diffraction can differentiate pure CaCO_3 from Mg-substituted CaCO_3 , whereas other techniques cannot. Small amounts of Mg substituted within the CaCO_3 structure result in slight shifts in diffraction peaks that have been calibrated with the amount of Mg substitution.^[20]

THIN SECTIONS

Carbonate minerals can usually be readily identified in thin sections by the high birefringence of carbonates and the microstructure of shells. Thin section techniques are time-consuming because sample preparation is lengthy and quantification relies on point count determinations,^[21] which are laborious. Accuracy depends on carbonate quantity, distribution, particle size, and how well the section is representative of the sample as a whole. As particle size decreases, positive identification becomes more difficult^[17] with a lower practical limit of detection of approximately 20 μm . For low concentrations (<10%), quantification becomes problematic. Despite limitations, microscopic examinations of soil material may differentiate pedogenic from lithogenic carbonates,^[22] which can have implications in soil genesis studies.

CaCO_3 and $\text{CaMg}(\text{CO}_3)_2$ often look similar in thin sections. $\text{CaMg}(\text{CO}_3)_2$ typically has rhombohedral habits, but this cannot be relied upon as a differentiating criterion. Chemical staining techniques have been used successfully to differentiate carbonate species.^[15,17,23–25]

X-RAY FLUORESCENCE AND MICROPROBE

X-ray fluorescence determines the chemical composition of a bulk sample, whereas microprobe is more site specific. Interpretation is based on assigning chemical composition to specific minerals. There are limitations in this

assignment, since other minerals contain Ca and Mg, and carbon approaches the detection limit of many instruments. In microprobe analyses, common stoichiometric correction factors may result in errors up to 20%.^[26] These are probably not the best techniques for carbonate determination, especially in low concentrations, for they are indirect and do not employ characteristic properties of carbonate minerals. However, if carbonate concentrations are above 10–20% and other analyses indicate that there are none to few other Ca- or Mg-bearing minerals, X-ray fluorescence and microprobe analysis of total Ca and/or Mg may be good indicators of carbonates.

Microprobe analysis is particularly useful in determining Ca/Mg ratios in $\text{CaMg}(\text{CO}_3)_2$ or Mg-substituted CaCO_3 of individual crystals, as it is site specific. However, it is not as efficient in bulk sample analysis as other techniques relying on carbonate reaction.

CONCLUSION

Numerous methods for determining SIC exist. The nature of the soil investigation and the intended use of carbonate data govern the technique of choice. Although highly sophisticated instruments can be used, the simple titrimetric technique (Eq. 6) and the change in gas volume or pressure upon reacting carbonates with acid (Fig. 1) are popular. Such techniques are simple and accurate, requiring commonly available laboratory apparatus and reagents. The change in gas pressure or volume has the advantage of differentiating between CaCO_3 and $\text{CaMg}(\text{CO}_3)_2$. The change in gas pressure technique may gain some favor because a pressure transducer can be connected to a personal computer for instantaneous results.

REFERENCES

1. Kohut, C.; Muehlenbachs, K.; Dudas, M.J. Authigenic dolomite in a saline soil in Alberta. Canada. *Soil Sci. Soc. Am. J.* **1995**, *59*, 1499–1504.
2. Drees, L.R.; Wilding, L.P.; Nordt, L.C. Reconstruction of inorganic carbon sequestration across broad geoclimatic regions. In *Soil Carbon Sequestration and the Greenhouse Effect*; Lal, R., McSweeney, K., Eds.; SSSA Special Pub. No. 57; Soil Science Society of America: Madison, 2001; 155–172.
3. Allison, L.E.; Moodie, C.D. Carbonate. In *Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties*; Black, C.A., Ed.; Soil Science Society of America: Madison, 1965; 1379–1396.
4. Nelson, R.E. Carbonate and gypsum. In *Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties*, 2nd Ed.; Page, A.L., Miller, R.H., Keeney, D.R., Eds.; Soil Science Society of America: Madison, 1982; 181–197.
5. Goh, T.B.; St. Arnaud, R.J.; Mermut, A.R. Carbonates. In *Soil Sampling and Methods of Analysis*; Carter, M.R., Ed.; Can. J. Soil Sci. Lewis Publishers: Boca Raton, 1993; 177–185.

6. Loeppert, R.H.; Suarez, D.L. Carbonate and gypsum. In *Methods of Soil Analysis. Part 3. Chemical Methods*; Sparks, D.L., Page, A.L., Eds.; Soil Science Society of America: Madison, 1996; 437–474.
7. Rabenhorst, M.C. Determination of organic and carbonate carbon in calcareous soils using dry combustion. *Soil Sci. Soc. Am. J.* **1988**, *52*, 965–969.
8. Dreimanis, A. Quantitative gasometric determination of calcite and dolomite by using chittick apparatus. *J. Sediment. Petrol.* **1962**, *32*, 520–529.
9. Presley, B.J. A simple method for determining calcium carbonate in sediment samples. *J. Sediment. Petrol.* **1975**, *45*, 745–746.
10. Schink, J.C.; Stockwell, J.H.; Ellis, R.A. An improved device for gasometric determination of carbonate in sediment. *J. Sediment. Petrol.* **1979**, *49*, 651–653.
11. Dunn, D.A. Revised techniques for quantitative calcium carbonate analysis using the Karbonat-Bombe and comparisons to other quantitative carbonate analysis methods. *J. Sediment. Petrol.* **1980**, *50*, 631–636.
12. Jones, G.A.; Kaiteris, P.A. Vacuum-gasometric technique for rapid and precise analysis of calcium carbonate in sediments and soils. *J. Sediment. Petrol.* **1983**, *53*, 655–660.
13. Evangelou, V.P.; Whittig, L.D.; Tanji, K.K. An automated monometric method for quantitative determination of calcite and dolomite. *Soil Sci. Soc. Am. J.* **1984**, *48*, 1236–1239.
14. Richards, L.A., Ed. *Diagnosis and Improvement of Saline and Alkali Soils*; U.S. Dept. Agr. Hbk. 60; U.S. Salinity Laboratory: Washington, 1954; 159 pp.
15. Arnaud, R.J.; Mermut, A.R.; Goh, T.B. Identification and measurement of carbonate minerals. In *Soil Sampling and Methods of Analysis*; Carter, M.R., Ed.; Can. J. Soil Sci. Lewis Publishers: Boca Raton, 1993; 737–743.
16. Ulas, M.; Sayin, M. Quantitative determination of calcite and dolomite in soils by X-ray diffraction. *J. Soil Sci.* **1984**, *35*, 685–691.
17. Gensmer, R.P.; Weiss, M.P. Accuracy of calcite/dolomite ratios by x-ray diffraction and comparisons with results from staining techniques. *J. Sediment. Petrol.* **1980**, *50*, 626–629.
18. Royse, C.F.; Wadell, J.S.; Petersen, L.E. X-ray determination of calcite-dolomite: An evaluation. *J. Sediment. Petrol.* **1971**, *41*, 483–488.
19. Fang, J.H.; Zevin, L. Quantitative X-ray diffractometry of carbonate rocks. *J. Sediment. Petrol.* **1985**, *55*, 611–613.
20. Hutchison, C.S. *Laboratory Handbook of Petrographic Techniques*; John Wiley & Sons: New York, 1974; 1–527.
21. Drees, L.R.; Ransom, M.D. Light microscopic techniques in quantitative soil microscopy. In *Quantitative Methods in Soil Mineralogy*; Amonette, J.E., Zelazny, L.W., Eds.; SSSA Misc. Pub; Soil Science Society of America: Madison, 1994; 137–176.
22. West, L.T.; Drees, L.R.; Wilding, L.P.; Rabenhorst, M.C. Differentiation of pedogenic and lithogenic carbonate forms in Texas. *Geoderma* **1988**, *43*, 271–287.
23. Warne, S.St.J. A quick field or laboratory staining scheme for the differentiation of the major carbonate minerals. *J. Sediment. Petrol.* **1962**, *32*, 29–38.
24. Dickson, J.A.D. A modified staining technique for carbonates in thin section. *Nature* **1965**, *205*, 587.
25. Lindholm, R.C.; Finkelman, R.B. Calcite staining: Semi-quantitative determination of ferrous iron. *J. Sediment. Petrol.* **1972**, *42*, 239–242.
26. Lane, S.J.; Dalton, J.A. Electron microprobe analysis of geologic carbonates. *Am. Mineral.* **1994**, *79*, 745–749.

Inorganic Carbon: Climate and Time

Lee C. Nordt

Department of Geology, Baylor University, Waco, Texas, U.S.A.

Abstract

Knowing the formation and distribution of soil inorganic carbon (SIC) components is important for the mapping and classification of soils, understanding carbonate equilibria reactions, better interpreting of the role of soils in the global carbon cycle, and understanding feedback mechanisms that soil may provide in a greenhouse world.

INTRODUCTION

Soil inorganic carbon (SIC) is contained in solid, solution, and gaseous phases with the regional and global distribution dependent, in large part, on the influence of climate and time.^[1] Common sources for the solid SIC phase are limestone (calcite and dolomite), carbonate-rich airborne dust, and unconsolidated calcareous parent materials such as alluvium, loess, and till. The gaseous phase consists of soil-respired carbon dioxide (CO_2) that is generated by plant and animal respiration and by the influx of atmospheric CO_2 into the upper part of the soil profile. The CO_2 phase combines with water to produce a solution enriched in carbonic acid (H_2CO_3) and bicarbonate (HCO_3^-). H_2CO_3 also enhances weathering of calcareous parent materials and calcium-bearing silicates that generates Ca^{2+} and CO_3^{2-} [or $\text{Ca}(\text{HCO}_3^-)_2$] by-products in the solution phase. Dissolution is enhanced in the upper soil profile where there is a greater flux of water, higher CO_2 production from respiration, and a lower pH.

The recombination of ions from solution into a solid SIC phase leads to the formation of pedogenic carbonate. Formation typically occurs below the zone of maximum biological activity where CO_2 production is lower and pH is higher. Further, because of evapotranspiration and drying at the terminus of the wetting front, the ionic concentration of Ca^{2+} and CO_3^{2-} may exceed the solubility product of calcium carbonate (CaCO_3). In association with a shallow groundwater table, capillary rise in soils can also contribute to the concentration of Ca^{2+} and CO_3^{2-} ions and, ultimately, to the formation of pedogenic carbonate. Also, the role of microorganisms and roots in the formation of CaCO_3 is recognized, but its importance and extent are largely unknown.^[2] Under well-drained conditions, and in soils with pHs greater than 6.5, CaCO_3 is the dominant SIC solid species.

When pedogenic forms of carbonate are visible in the field, the associated soil horizons are commonly

described as Bk (filaments, soft masses, nodules) or Bkm (indurated). Although not formally recognized, there is evidence that disseminated pedogenic carbonate accumulates in near-surface horizons not visible in the field.^[3]

SIC AND CLIMATE

The global distribution of soil CO_2 , HCO_3^- , and CaCO_3 is determined, in part, by climate, particularly in response to rainfall and temperature. The concentration of soil CO_2 is commonly 10 to 100 times greater than at the soil-atmosphere interface (365 ppmV) because of biotic respiration and organic matter decomposition.^[4] Soil CO_2 concentrations are greater in warm-humid climates where rainfall and respiration rates are higher and decrease systemically to hot- or cold-arid climates where rainfall and respiration rates are lower. Soil respiration rates and CO_2 concentrations typically increase with elevation in arid to semiarid regions in response to more rainfall, cooler temperatures, and less evapotranspiration.^[5] In areas where soil CO_2 concentrations are relatively low (arid), atmospheric CO_2 can penetrate into the soil to depths of as much as 50 cm.^[6] Increased greenhouse gases and global warming could in some areas increase soil respiration rates.^[7]

As CO_2 dissolves in soil water, the inorganic carbon species H_2CO_3 is formed, which rapidly dissociates to HCO_3^- in soils with pH above 6.5.^[8] The production of HCO_3^- will be greater in warm-humid climates, especially where there are high respiration rates or rapid weathering of carbonate-rich parent materials. The fate of HCO_3^- in leaching soils is typically to deep groundwater and to rivers.^[9] In arid to semiarid regions where evapotranspiration is high and rainfall is low, HCO_3^- may become part of the SIC pool as pedogenic carbonate. Elevated soil CO_2 levels generated in some areas by global warming could enhance HCO_3^- production from

increased interaction of water and CO_2 , leading to lower soil pH and more intense weathering of primary and other secondary minerals.^[7]

Retallack^[10] provided a robust data set quantifying the relationship between rainfall and depth to the top of Bk horizons in warm-temperate climates (Fig. 1). This relationship indicates that at less than 100 mm of mean annual precipitation, insufficient soil moisture is available for weathering and for the production of visible forms of pedogenic carbonate. In contrast, most soils are leached of carbonate within the upper 1.5–2 m when mean annual precipitation exceeds 1000 mm. Using a national soil taxonomic data base, the rainfall to depth of pedogenic carbonate relationship has been brought into question.^[11] Careful control is needed to ensure that climate is the primary soil-forming factor in using this relationship for paleoenvironmental reconstruction.

SIC AND TIME

In the initial stages of soil formation, the production and concentration of CO_2 likely increase over decades to centuries as pioneer plant communities evolve into climax plant communities. Soil CO_2 production may decrease with time in response to vegetation disturbances created by increased aridity, soil erosion, overgrazing, or plowing.^[7]

In general, soil HCO_3^- in solution will increase or decrease in proportion to the production of soil CO_2 . However, if continued weathering leads to greater concentration of H^+ , a more acidic soil environment may produce more H_2CO_3 at the expense of HCO_3^- . Another long-term

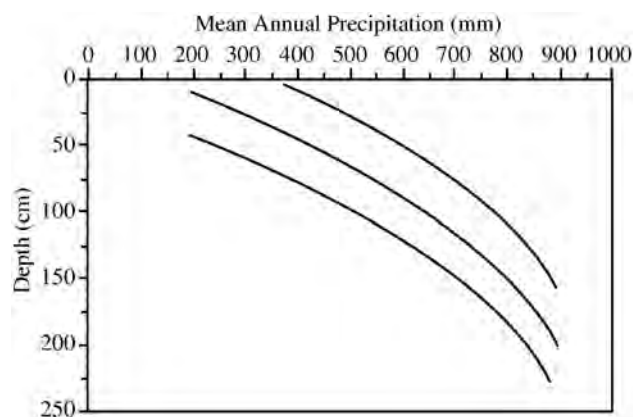


Fig. 1 Relationship between mean annual precipitation (P) in millimeters and depth to top of Bk horizon (D) in centimeters. The relationship is described by the equation $P = 139.6 - 6.388D - 0.01303D^2$, with a correlation coefficient of 0.79 and a standard error of ± 33 cm. Results derived from well-drained and loamy, calcareous soils in warm-temperate climates principally for Stages II and III carbonate morphologies.

Source: From Retallack.^[10]

mechanism for decreasing HCO_3^- in soil solution is related to the weathering and depletion of parent material carbonate.

Gile and colleagues^[12] proposed a model of carbonate formation recognized as morphological stages for desert climates. For the model, it is assumed that a sufficient supply of Ca^{2+} and HCO_3^- ions are provided to form all stages of carbonate development. In addition, minimal climatic change during the interval of pedogenesis is ideal. The rate at which carbonate development passes through the individual stages is also dependent on parent material texture.

With these caveats in mind, the carbonate stages are designated Stage I, II, III, and IV from least to most developed. In non-gravelly parent materials, it can take 25,000–75,000 years to progress from Stage I (filaments) to Stage III (weakly indurated mass) in the formation of Bk horizons (Fig. 2). Stage IV represents nearly complete plugging of macropores forming a laminar cap and Bkm horizon. In desert environments, this process can take as long as 400,000 years (Fig. 2).

Machette^[13] proposed two additional morphological stages reflecting the further passage of time and enhanced geomorphic activity. Stage V is also recognized as a Bkm horizon but with a laminar cap greater than 1 cm thick. Stage VI is characterized by multiple episodes of

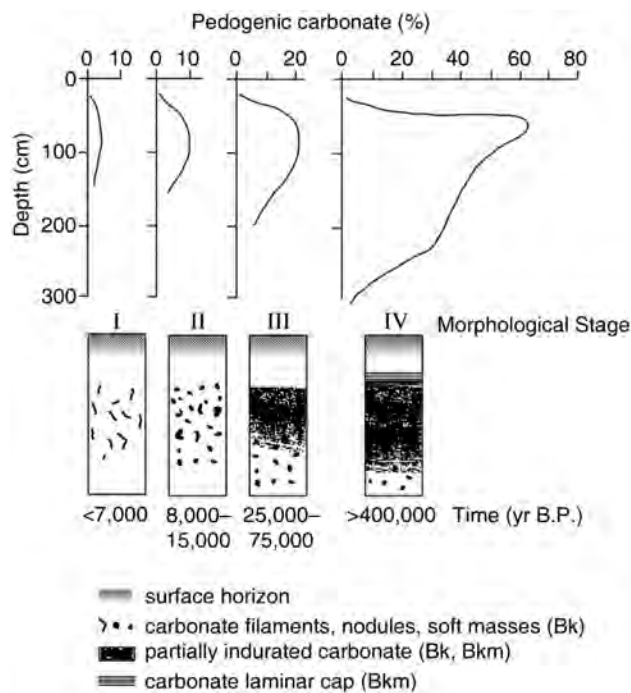


Fig. 2 Morphological stages of pedogenic carbonate with time in non-gravelly parent materials of arid to semiarid regions of the Southwestern United States. Not shown are advanced carbonate morphologies typifying Stages V and VI.

Source: From Gile, Hawley, et al.^[12]

brecciation and recementation and is thought to have formed over a period of 2 million years.

Similar stages of carbonate development form in soils with gravelly parent materials.^[12] Stage I results from the accumulation of carbonate pendants on the bottom of gravels. Stage II forms with nearly complete encasement of the gravels with a coat of carbonate. The more advanced stages have similar morphologies to the carbonate that accumulates in non-gravelly parent materials. However, because of reduced inherent porosity, the first four stages can be attained within 25,000–50,000 years of soil formation.

In soils formed from indurated and soft limestone, the upper surface tends to undergo numerous *in situ* dissolution–precipitation cycles that generate limestone porosity, unique pedogenic carbonate morphological fabrics, and eventually a laminar cap. Rabenhorst and colleagues^[14] present a model showing five stages of carbonate development in this system for semiarid to subhumid climates. Initial stages involve the accumulation of pedogenic carbonate around macrovoids either weathered into limestone or along fracture planes. Advanced stages culminate in complete plugging of both macro- and micropores along with the formation of a laminar cap.

SIC: INTEGRATION OF CLIMATE AND TIME

As shown in Fig. 3, the morphological characteristics of soil carbonate generally reflect the combined effects of climate and time. In this idealized example, it is assumed

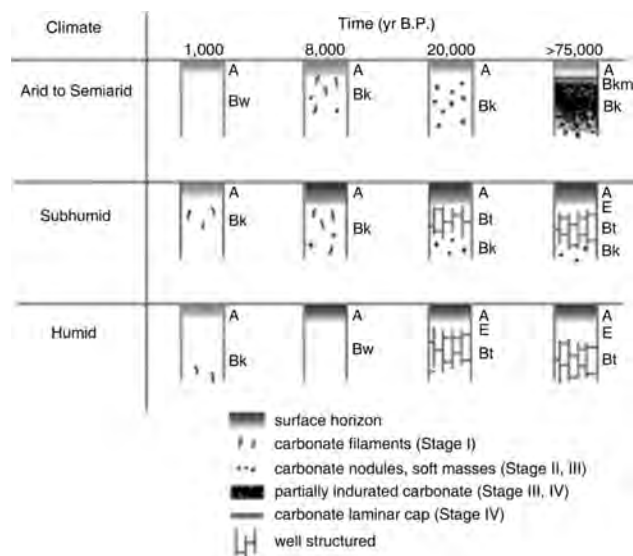


Fig. 3 Integrative effects of climate and time on horizonation and pedogenic carbonate development in soils formed from calcareous parent materials in well-drained, loamy soils.

Source: From Nordt, Wilding, et al.^[9]

that the soils are loamy, well drained, calcareous, and formed in warm-temperate climates. Horizontally, the effects of time on soil development for a constant climate can be observed. In contrast to arid to semiarid climates discussed previously, carbonate development may never pass through Stage I in humid climates before net leaching of carbonate by-products begins to intensify. Here, A-E-Bt horizon sequences quickly form at the expense of incipient Bk and Bw horizons. In transitional subhumid climates, Bk horizons may eventually evolve into Bt horizons in the upper profile with Bk horizons persisting at depth.

Observing Fig. 3 vertically across climatic regions while holding time constant provides another view of these complex relationships. At 1000 years, incipient Bk horizons form quickly in subhumid and humid climates. At the endpoint of weathering at >75,000 years, A-Bkm horizons may form and persist in arid to semiarid climates with well-developed A-E-Bt horizons forming in subhumid and humid climates.

CONCLUSION

Knowing the formation and distribution of SIC components is important for the mapping and classification of soils, understanding carbonate equilibria reactions, better interpreting of the role of soils in the global carbon cycle, and understanding feedback mechanisms that soil may provide in a greenhouse world.

REFERENCES

1. Birkeland, P.W. *Soils and Geomorphology*; Oxford University Press: New York, 1999.
2. Monger, H.C.; Daugherty, L.A.; Lindemann, W.C.; Liddell, C.M. Microbial precipitation of pedogenic calcite. *Geology* **1991**, *19*, 997–1000.
3. Nordt, L.C.; Hallmark, C.T.; Wilding, L.P.; Boutton, T.W. Quantifying pedogenic carbonate accumulations using stable carbon isotopes. *Geoderma* **1998**, *82*, 115–136.
4. Schlesinger, W.H. *Biogeochemistry: Analysis of Global Change*, 1st Ed.; Academic Press: San Diego, 1991; 1–443.
5. Amundson, R.G.; Chadwick, O.A.; Sowers, J.M. A comparison of soil climate and biological activity along an elevation gradient in the eastern Mojave desert. *Oecologia* **1989**, *80*, 395–400.
6. Cerling, T.E. The stable isotopic composition of modern soil carbonate and its relationship to climate. *Earth Planet. Sci. Lett.* **1984**, *71*, 229–240.
7. Suarez, D.L. Impact of agriculture on CO₂ as affected by changes in inorganic carbon. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2000; 257–272.

8. Schwab, A.P. The soil solution. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; B85–B120.
9. Nordt, L.C.; Wilding, L.P.; Drees, L.R. Pedogenic carbonate transformations in leaching soil systems: Implications for the global C cycle. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2000; 43–64.
10. Retallack, G.J. The environmental factor approach to the interpretation of paleosols. In *Factors of Soil Formation: A Fiftieth Anniversary Retrospective*; Amundson, R., Harden, J., Singer, M., Eds.; Special Publication Number 33; Soil Science Society of America: Madison, 1994; 31–64.
11. Royer, D.L. Depth to pedogenic carbonate as a paleoprecipitation Indicator? *Geology* **1999**, 27, 1123–1126.
12. Gile, L.H.; Hawley, J.W.; Grossman, R.B. *Soils and Geomorphology in the Basin and Range Area of Southern New Mexico-Guidebook to the Desert Project*; New Mexico Bureau of Mines and Mineral Resources Memoir 39: Socorro, 1981; 1–222.
13. Machette, M.N. Calcic soils of the Southwestern United States. In *Soils and Quaternary Geomorphology of the Southwestern United States*; Weide, D.L., Ed.; Geological Society of America: Boulder, 1985; 1–21.
14. Rabenhorst, M.C.; West, L.T.; Wilding, L.P. Genesis of calcic and petrocalcic horizons in soils over carbonate rocks. In *Occurrence, Characteristics, and Genesis of Carbonate, Gypsum, and Silica Accumulations in Soils*; Nettleton, W.D., Ed.; Soil Science Society of America: Madison, 1991; 61–74.

Inorganic Carbon: Composition and Formation

H. Curtis Monger

*Department of Agronomy and Horticulture, New Mexico State University,
Las Cruces, New Mexico, U.S.A.*

Larry P. Wilding

*Department of Soil and Crop Sciences, Texas A&M University, College Station,
Texas, U.S.A.*

Abstract

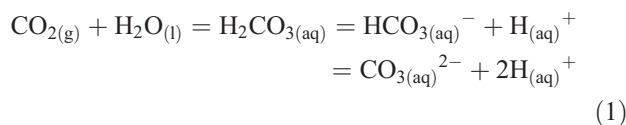
Understanding pedogenic carbonate formation has been extremely useful for understanding relative ages of geomorphic surfaces and landscape evolution. A knowledge of pedogenic carbonate formation has also been useful for soil classification. Studies of pedogenic carbonate have expanded to include questions about paleoclimate, paleoecology, paleoatmospheric composition, global carbon cycles, and the greenhouse effect.

INTRODUCTION

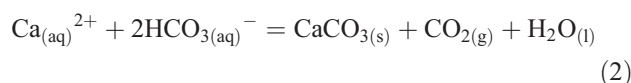
It has become increasingly common for “soil inorganic carbon” to mean soil carbonate mineral carbon, mainly calcium carbonate (CaCO_3). In the strict sense, however, inorganic carbon encompasses not only carbon in carbonate minerals but also carbon in the carbonic acid system.^[1] The carbonic acid system includes gaseous carbon dioxide [$\text{CO}_{2(g)}$], aqueous carbon dioxide [$\text{CO}_{2(aq)}$], carbonic acid ($\text{H}_2\text{CO}_{3(aq)}$), bicarbonate ion ($\text{HCO}_3(aq)^-$), and carbonate ion ($\text{CO}_3(aq)^{2-}$).

COMPOSITION

In the soil solution, as with other solutions, the interaction of these species can be represented by the following reaction:^[1]



Cations, such as Ca^{2+} , Mg^{2+} , Fe^{2+} , Mn^{2+} , and Na^+ , precipitate with HCO_3^- (which is the dominant anion between pH 6.5 and 10.5) and CO_3^{2-} (which is the dominant anion above pH 10.5) to form a variety of carbonate minerals. The reaction of $\text{Ca}_{(aq)}^{2+}$ with $\text{HCO}_3(aq)^-$ to form calcite is illustrated below:



There are about 60 carbonate minerals, which in addition to calcite, include aragonite (CaCO_3), dolomite

[$\text{CaMg}(\text{CO}_3)_2$], siderite (FeCO_3), magnesite (MgCO_3), rhodocrosite (MnCO_3), cerussite (PbCO_3), and malachite [$\text{CuCO}_3\text{Cu}(\text{OH})_2$]. In soil, the overwhelmingly abundant carbonate mineral is calcite.^[2] In unique soil environments, however, other carbonate minerals have been found, such as pedogenic siderite and dolomite.^[3]

Individual carbonate crystals of pedogenic origin are generally too small to be seen with the unaided eye. When concentrated together, their combined presence takes on a white color with a variety of macroscopic forms. These forms include carbonate filaments (also called mycelia, pseudomycelia, and threads), films, coatings, soft spheroidal segregations (white eyes), nodules, cylindroids, concretions, glaeboles, and veins. Soil fabric that is impregnated with carbonate to the point that it occurs as an essentially continuous medium has been termed “k-fabric.”^[4] Stages of morphogenetic carbonate accumulation, in which progressively greater amounts of carbonate occur in progressively older soils, are important chronologic indicators. The calcic and petrocalcic horizons are diagnostic horizons in Soil Taxonomy.^[5] Calcic horizons generally contain greater than 15% carbonate by weight. Petrocalcic horizons are indurated forms of calcic horizons. Examples of these horizons are shown in Figs. 1A and B. Examples of carbonate crystals as viewed with optical microscopy and scanning electron microscopy are shown in Figs. 1C and D.

The dissolution of carbonates in soil systems can be represented by the following reaction (Eq. 3). In humid regions, soluble products of this weathering reaction flux through the vadose zone into groundwater, or precipitate as pedogenic carbonates deep in the soil or geologic system. In arid regions, soluble products precipitate at relatively

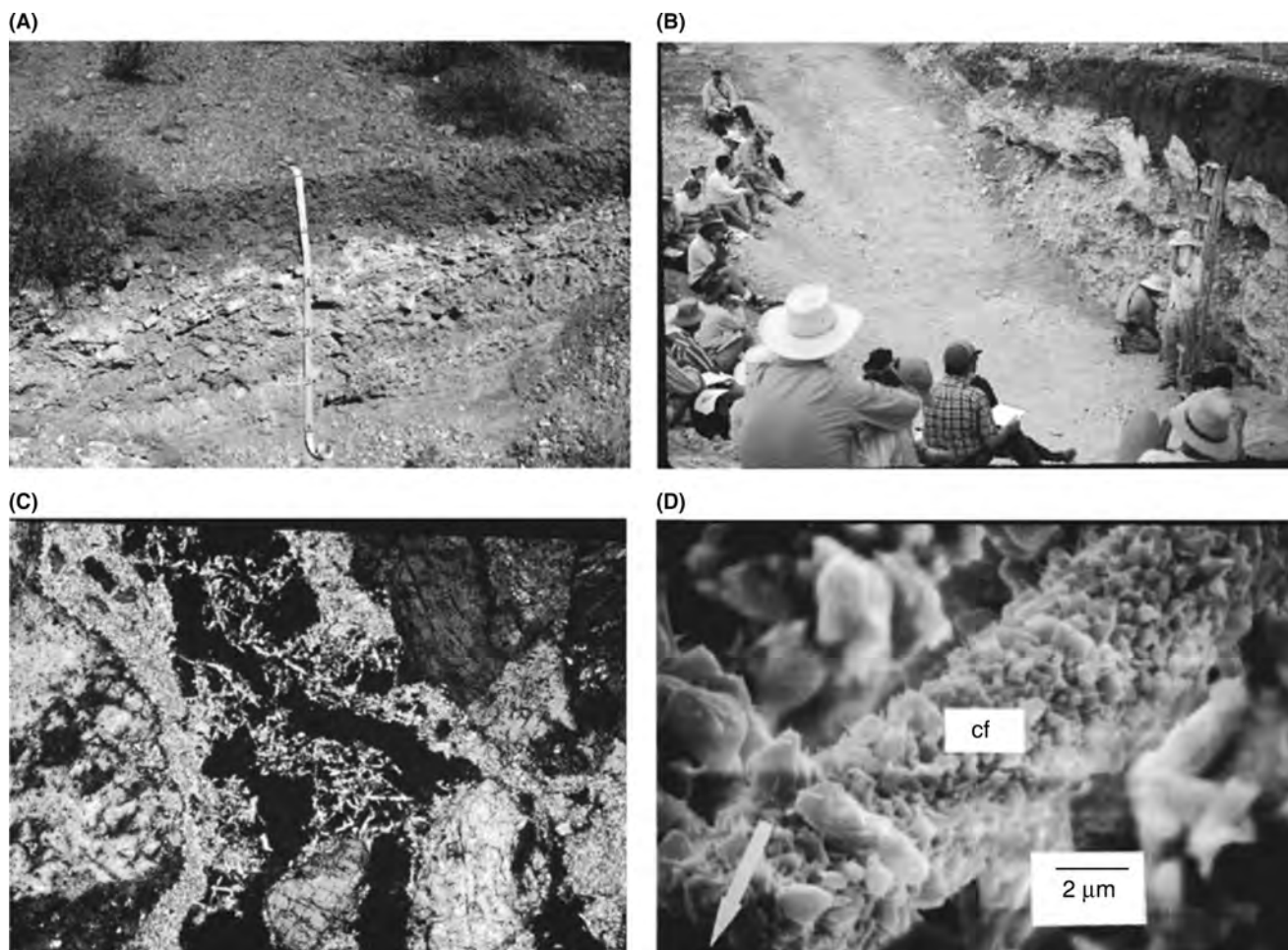
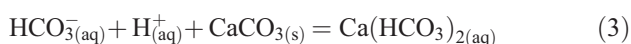


Fig. 1 Examples of inorganic carbon as it exists in the field and under the microscope. The white horizon is a calcic horizon in (A) and petrocalcic horizon in (B). The small golden crystals in (C) are calcite crystals coating sand grains; the black region is a pore space as it appears in cross-polarized light. A calcified fungal filament viewed with scanning electron microscopy is shown in (D).

shallow depths as a result of sparse rainfall and insufficient leaching.



FORMATION

Being located in an arid, semiarid, or subhumid climate is the primary factor that controls carbonate formation. In many areas, the boundary between carbonate-accumulating soil and non-carbonate-accumulating soil is about 500 mm (20 in.) mean annual rainfall.^[6] This relationship is confounded, however, by the effects of soil temperature, soil drainage, nature and properties of the parent material (e.g., soil texture, carbonate content, carbonate mineralogy, and porosity), soil drainage, land-form position, geomorphic stability, and effectiveness of precipitation (rainfall intensity and duration). Hence, there are many examples in humid and subhumid environments

where soil carbonate persists in the soil system at depths inconsistent with regional models. In humid regions, for example, inorganic carbon persists as calcite or dolomite detritus in soils derived from certain parent materials (e.g., calcareous loess, till, outwash, alluvial deposits, sedimentary and metamorphic rocks). In seasonally wet soils, carbonate can accumulate in upper subsoils from capillary rise of bicarbonates via evaporative pumping from shallow groundwater.^[7] In addition, carbonate minerals can occur in wetland soils that commonly contain soluble carbonates, bicarbonates, or carbonic acid depending on the pH of the local environment.

Pedogenic vs. Geogenic Carbonate

Many soils develop in calcareous parent materials. For these soils, it has been a challenge quantifying carbonate that formed in the soil profile vs. carbonate mechanically inherited from parent material. Carbonate formed in the soil profile has been termed “secondary,” “authigenic,” or

“pedogenic.”^[4,8] On the other hand, carbonate mechanically inherited from parent material has been termed “primary,” “geogenic,” or “lithogenic.”^[2,9] Criteria for distinguishing pedogenic from geogenic carbonates involve the scrutiny of both field and laboratory evidence. Field evidence, for example, includes differences in the presence of marine fossils, carbonate morphology (such as nodules, pendants, and laminar caps that indicate pedogenic), and distribution patterns with depth, where, e.g., a carbonate horizon of pedogenic origin is overlain and underlain by non-calcareous soil. Laboratory evidence includes comparing mineralogy, particle size, microfabric, and $^{13}\text{C}/^{12}\text{C}$ ratios of carbonate with unknown origin to those of carbonate with known geogenic origin.^[10–12]

Models of Carbonate Formation

There are several processes that cause carbonates to form in soil. Excluding geologic processes, such as lacustrine and deep groundwater cementation that preserve the original sedimentary structure, the formation pedogenic carbonate can broadly be placed into four models—per descensum, per ascensum, *in situ*, and biogenic models.

The per descensum model

The per descensum model accounts for carbonate formation resulting from downward moving meteoric water and can be subdivided into three types. First is the dissolution of preexisting carbonates in the upper profile, their vertical translocation, and their precipitation in the subsoil. This model was invoked to explain why progressively shallower carbonates occur in progressively drier climates.^[13] Later, this per descensum model was used as the basis for calculating the number of wetting fronts required to leach carbonates to a particular depth.^[14] In both cases, it was assumed that carbonate was uniformly distributed in parent material at the beginning of pedogenesis.

Second is the case in which pedogenic carbonate forms in soils with non-calcareous parent materials. Unlike the model described before, non-calcareous parent material does not have carbonate uniformly distributed throughout the profile at the beginning of pedogenesis. In southern New Mexico, e.g., prominent calcic and petrocalcic horizons occur in soils with rhyolite alluvium as parent material. This alluvium would yield low amounts of calcium if the rhyolite particles were thoroughly decomposed, which they were not.^[15] Therefore, atmospheric additions, another per descensum model, were judged to be the source of carbonates.^[15] Initially, calcareous dust was measured and considered to be the source of carbonate. Later it was realized that Ca^{2+} in rain was an additional, and even larger source of Ca^{2+} for reaction with soil HCO_3^- to form carbonates.^[15] Building on these per descensum concepts, compartmental models have been constructed that compute the

depth, amount, and distribution of pedogenic carbonate as a function of climate and time.^[16,17]

Third, in addition to vertical illuviation within a soil profile, lateral, downslope migration of the soil solution containing soluble products of carbonate is another per descensum model. In this case, carbonate is thought to precipitate after carbonate-charged waters migrate from upslope positions to lower landscape positions.^[18]

The per ascensum model

The per ascensum model accounts for carbonate formation resulting from bottom-up movement. A primary example is the capillary rise of Ca^{2+} and bicarbonate from shallow water tables by evaporative pumping, which leads to the precipitation of carbonates in the upper subsoil.^[7] Moreover, chemical studies have shown that in some environments plants promote carbonate formation by transporting Ca^{2+} upward to the land surface from subsoil, rock, and groundwater.^[19]

The *in situ* model

Third, in the *in situ* model, pedogenic carbonate is the result of in-place dissolution and reprecipitation of bedrock composed of marine carbonate.^[20] Limestone, for instance, is progressively transformed into pedogenic carbonate as a result of short-range carbonate dissolution and reprecipitation proximal to the depth of the upper contact with limestone. This is a rather unique method to form pedogenic carbonates where the total carbonate content of the zone of enriched pedogenic products is less than the carbonate content of the limestone originally. These pedogenic zones have a much higher macro- and microporosity than the limestone.

In addition to marine carbonates, the *in situ* model also includes carbonate formation resulting from in-place chemical weathering of Ca-bearing igneous rock. Upon release into the soil solution by weathering, Ca^{2+} precipitates with bicarbonate formed from the reaction of water with CO_2 generated by root and microbial respiration. In many cases, however, igneous parent material has been considered as an insufficient source of Ca^{2+} and hence external sources, such as atmospheric additions of Ca^{2+} , have been sought.^[6]

The biogenic model

Fourth, some plants, microorganisms, and termites produce CaCO_3 . Evidence that various plants play a direct role in carbonate formation comes from the presence of euhedral calcite crystals on plant roots.^[21] Moreover, several references in the Russian literature note carbonate formation by plant tissue and the downward translocation of these carbonates with wetting fronts.^[22] Evidence that some microorganisms precipitate carbonates is based on observations of calcified bacteria and fungal hyphae with electron microscopy and *in vitro* laboratory experiments.^[23,24] Evidence that

termites precipitate carbonate in certain environments is based on the studies of termite mounds in Africa and south-east Asia.^[25] Such mounds can be calcareous even though surrounding soils are non-calcareous, making the mounds attractive to native farmers who spread them over their agricultural fields.^[25]

CONCLUSION

The formation of pedogenic carbonate may be dominated by one of the models listed before or may involve several of the models working together in different magnitudes. Understanding pedogenic carbonate formation has been extremely useful for understanding relative ages of geomorphic surfaces and landscape evolution.^[15] A knowledge of pedogenic carbonate formation has also been useful for soil classification. Marbut,^[26] for instance, used the presence of carbonate as a criterion for the highest category of his soil classification—Pedocal (soils with carbonate accumulation) and Pedalfers (soil with Al and Fe accumulation). Studies of pedogenic carbonate have expanded to include questions about paleoclimate, paleoecology, paleoatmospheric composition, global carbon cycles, and the greenhouse effect.

REFERENCES

1. Morse, J.W.; Mackenzie, F.T. *Geochemistry of Sedimentary Carbonates. Developments in Sedimentology* 48; Elsevier: Amsterdam, 1990; 707.
2. Doner, H.E.; Lynn, W.C. Carbonate, halide, sulfate, and sulfide minerals. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 279–330.
3. Capo, R.C.; Whipkey, C.E.; Blachère, J.R.; Chadwick, O.A. Pedogenic origin of dolomite in a basaltic weathering profile, Kohala Peninsula, Hawaii. *Geology* **2000**, *28*, 271–274.
4. Gile, L.H.; Peterson, F.F.; Grossman, R.B. The K horizon—A master soil horizon of carbonate accumulation. *Soil Sci.* **1965**, *99*, 74–82.
5. Soil Survey Staff. *Soil Taxonomy—A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; USDA Agriculture Handbook Number 436; U.S. Govt. Printing Office: Washington, 1999.
6. Birkeland, P.W. *Soils and Geomorphology*, 3rd Ed.; Oxford University Press: New York, 1999; 430 pp.
7. Sobacki, T.M.; Wilding, L.P. Formation of calcic and argillic horizons in selected soils of the Texas coast prairie. *Soil Sci. Soc. Am. J.* **1983**, *47*, 707–715.
8. Pal, D.K.; Dasog, G.S.; Vadivelu, S.; Ahuja, R.L.; Bhattacharyya, T. Secondary calcium carbonate in soils of arid and semiarid regions of India. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: London, 2000; 149–185.
9. West, L.T.; Drees, L.R.; Wilding, L.R.; Rabenhorst, M.C. Differentiation of pedogenic and lithogenic carbonate forms in Texas. *Geoderma* **1988**, *43*, 271–287.
10. Rabenhorst, M.C.; Wilding, L.P.; West, L.T. Identification of pedogenic carbonates using stable carbon isotopes and microfabric analysis. *Soil Sci. Soc. Am. J.* **1984**, *48*, 125–132.
11. Drees, L.R.; Wilding, L.P. Micromorphic record and interpretations of carbonate forms in the rolling plains of Texas. *Geoderma* **1987**, *40*, 157–175.
12. Nordt, L.C.; Hallmark, C.T.; Wilding, L.P.; Boutton, T.W. Quantifying pedogenic carbonate accumulations using stable carbon isotopes. *Geoderma* **1998**, *82*, 115–136.
13. Jenny, H.; Leonard, C.D. Functional relationships between soil properties and rainfall. *Soil Sci.* **1934**, *38*, 363–381.
14. Arkley, R.J. Calculation of carbonate and water movement in soil from climatic data. *Soil Sci.* **1963**, *96*, 239–248.
15. Gile, L.H.; Hawley, J.W.; Grossman, R.B. *Soils and Geomorphology in the Basin and Range Area of Southern New Mexico—Guidebook to the Desert Project*; New Mexico Bureau of Mines and Mineral Resources, Memoir 39; Socorro, 1981; 222 pp.
16. McFadden, L.D.; Tinsley, J.C. Rate and depth of pedogenic-carbonate accumulation in soils: Formation and testing of a compartment model. In *Soils and Quaternary Geology of the Southwestern United States*; Weide, D.L., Ed.; Special Paper 203; Geological Society of America: Boulder, 1985; 23–41.
17. Marion, G.M.; Schlesinger, W.H. Quantitative modeling of soil forming processes in deserts: The CALDEP and CAL-GYP models. In *Quantitative Modeling of Soil-Forming Processes*; Bryant, R.B., Arnold, R.W., Eds.; Soil Sci. Soc. Am. Spec. Publ. 39; Madison, 1994; 129–145.
18. Scharpenseel, H.W.; Mtmet, A.; Freytag, J. Soil inorganic carbon and global change. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: London, 2000; 27–42.
19. Goudie, A. *Duricrusts in Tropical and Subtropical Landscapes*; Oxford University Press: London, 1973; 174 pp.
20. Rabenhorst, M.C.; Wilding, L.P. Pedogenesis on the Edwards plateau, Texas: III. A new model for the formation of petrocalcic horizons. *Soil Sci. Soc. Am. J.* **1986**, *50*, 693–699.
21. Monger, H.C.; Gallegos, R.A. Biotic and abiotic processes and rates of pedogenic carbonate accumulation in the southwestern United States—Relationship to atmospheric CO₂ sequestration. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: London, 2000; 273–289.
22. Labova, E. *Soils of the Desert Zone of the USSR*; Israel Program for Scientific Translation: Jerusalem, 1967.
23. Phillips, S.E.; Milnes, A.R.; Foster, R.C. Calcified filaments: An example of geological influences in the formation of calcretes in South Australia. *Aust. J. Soil Res.* **1987**, *25*, 405–428.
24. Monger, H.C.; Daugherty, L.A.; Lindemann, W.C.; Liddell, C.M. Microbial precipitation of pedogenic calcite. *Geology* **1991**, *19*, 997–1000.
25. Thorp, J. Effects of certain animals that live in soils. *Sci. Monthly* **1949**, *68*, 180–191.
26. Marbut, C.F. Subcommission II. Classification, nomenclature, and mapping of soils in the United States. *Soil Sci.* **1928**, *25*, 61–71.

Inorganic Carbon: Global Carbon Cycle

William H. Schlesinger

Department of Geology and Botany, Duke University, Durham, North Carolina, U.S.A.

Abstract

Changes in the quantity of carbon (C) in vegetation and soils play a major role in determining short- and long-term fluctuations in the concentration of carbon dioxide (CO₂) in earth's atmosphere. Human activities, such as the irrigation of agricultural soils in arid regions, can alter the accumulation of pedogenic carbonate in soils. For comparative purposes, biogeochemists calculate the mean residence time (MRT). Although the amount of pedogenic carbonate in world soils is quite large, this pool of C is relatively sluggish. The long MRT of the pedogenic carbonate pool ensures that it will not become a major sink or source of atmospheric CO₂ over the next several centuries.

INTRODUCTION

The “global C cycle” is defined as the exchange of carbon (C) among the atmosphere, seawater, land vegetation, and soil reservoirs (Fig. 1). Each year dead plant materials entering the soil are decomposed by soil microbes that return carbon dioxide (CO₂) to the atmosphere. If the amount of land vegetation remains the same, the amount of CO₂ removed from the atmosphere by plant growth each year is balanced by the amount of plant death and decomposition. Such a perfect balance, however, is seldom seen. Changes in the quantity of C in vegetation and soils play a major role in determining short- and long-term fluctuations in the concentration of CO₂ in earth's atmosphere. A portion of the atmospheric increase in CO₂, e.g., is because of the destruction of vegetation and the disturbance of soils by humans.

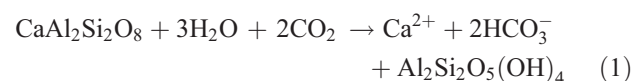
MEAN RESIDENCE TIME (MRT)

For comparative purposes, biogeochemists calculate the MRT or the amount of time C resides in each pool of the global C cycle before circulating to the others. For instance, a molecule of CO₂ spends, in average, about 5 years in the atmosphere before it enters the terrestrial biosphere or the oceans. A C atom spends, in average, about 10 years in vegetation and 35 years in soil organic matter (SOM) before it returns to the atmosphere as CO₂. In comparison, the circulation of C in the oceans is rather sluggish; the C atom spends, in average, hundreds of years in the sea, where it is found predominantly as dissolved bicarbonate (HCO₃⁻). Human activities have the greatest impact on pools with short MRTs.

WEATHERING

Most studies of soil C focus on SOM, which globally contains nearly 2300×10^{15} g C.^[1] The release of CO₂ by the microbial decomposition of SOM is one of the largest fluxes in the global C cycle (Fig. 1). However, soils also contain C in various inorganic forms—CO₂ held in the soil pore spaces, HCO₃⁻ dissolved in soil waters, and calcium carbonate (CaCO₃) as a soil mineral. CO₂ in the soil pores is largely derived from the respiration of plant roots and soil microbes, which varies as a function of soil moisture and temperature. However, even when the respiration rate is very high, the amount of C in soil gases and the soil solution is not large enough to contribute materially to the total C content of soils globally. In contrast, the amount of C held in soil carbonates is quite large, totaling about $750\text{--}1000 \times 10^{15}$ g C (Fig. 1).^[2,3] The vast majority of this carbonate is found in the world's arid and semiarid lands, where inorganic C may exceed soil organic C by a factor of 10.^[4]

In some soils, CaCO₃ and other carbonates (e.g., dolomite) are derived from the parent rocks from which the soils have formed. However, additional carbonate may form as a result of the release of calcium (Ca) from the chemical weathering of rocks and the precipitation of CaCO₃ when water is lost from the soil by evaporation and plant uptake. In the case of silicate parent minerals, such as plagioclase (calcium feldspar), the relevant reaction of weathering is as follows:



Here Ca is released from the silicate mineral by the weak solution of carbonic acid that is formed when CO₂ dissolves in water.

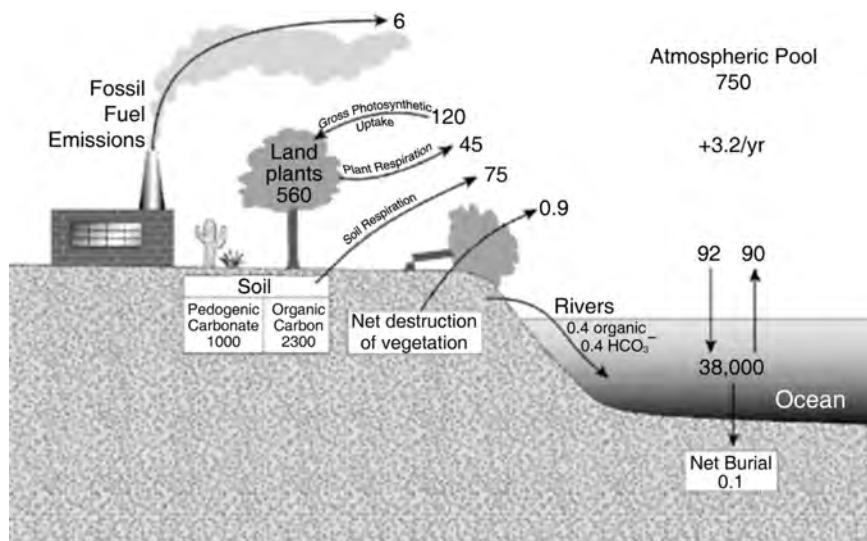
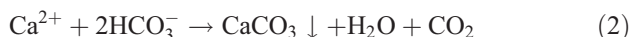


Fig. 1 The global C cycle, showing the size of reservoirs (10^{15} g C) and the annual flux between them (10^{15} g C/yr).

Source: Modified from Schlesinger.^[9]

Declining soil concentration of either CO_2 or water precipitates carbonate via the following reaction:



While the weathering process occurs most rapidly when plant activity increases the concentration of CO_2 in the soil pore space, the precipitation reaction occurs during seasonal periods of drought. Carbonate formed in the soil, known as secondary or “pedogenic” carbonate, can be distinguished from carbonate inherited from parent materials by examining thin sections and the isotopic ratio (^{13}C vs. ^{12}C) of the C in carbonate minerals.

ATMOSPHERIC DERIVATION

In most areas, the Ca content of pedogenic carbonate is derived from the atmosphere and is deposited as a constituent of rain and dust.^[5,6] However, because the Ca carried in the atmosphere is ultimately derived from the weathering of rocks in some upwind area, the two aforementioned reactions are general.^[7] Across deserts of the Southwestern United States, the formation of pedogenic carbonate is closely related to mean annual rainfall (Fig. 2).

If we know the rate of formation of pedogenic carbonate, usually expressed in grams per square meter per year ($\text{g/m}^2/\text{yr}$), and the total amount of carbonate in the soil profile, we can calculate the length of time taken for the accumulation of current quantity of soil pedogenic carbonate. In the Mojave Desert of California, radiocarbon and uranium–thorium dating show that pedogenic carbonate found in the upper 1.5 m of soils, derived from silicate materials, accumulated over a period of up to 20,000 years.^[8] During that period, pedogenic carbonate formed at rates ranging from 1.0 to 3.5 $\text{g/m}^2/\text{yr}$, with some indications of greater rates during the Pleistocene, when rainfall was greater in this region. The global MRT of soil pedogenic carbonate is about 85,000 years, making this C pool

much less dynamic than SOM, in which the global MRT for C is about 35 years (Fig. 1).

While CO_2 is sequestered from the atmosphere during the formation of pedogenic carbonate from silicate parent materials, no such net storage occurs when pedogenic carbonate forms from carbonate parent materials.^[10] Overall, the formation of soil pedogenic carbonates is less effective than the formation of SOM in the storage of atmospheric CO_2 .^[11] This is disappointing, of course, to those who view increasing the formation of pedogenic carbonate in desert soils as a means of slowing the rise of CO_2 in the earth’s atmosphere and reducing global warming.

EFFECT OF HUMAN ACTIVITIES

Human activities, such as the irrigation of agricultural soils in arid regions, can alter the accumulation of pedogenic

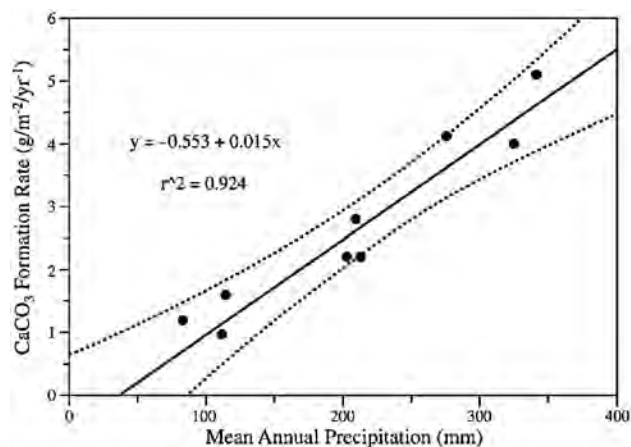
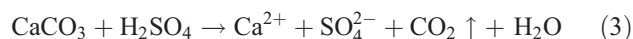


Fig. 2 The rate of formation of pedogenic carbonate in arid soils of the Southwestern United States as a function of the modern precipitation at each site.

Source: From Marion.^[12]

carbonate in soils. The groundwater used for irrigation is often extracted from subsurface environments, where CO₂ concentration is much higher than in the earth's atmosphere, and often contains high concentrations of dissolved Ca. Applying such water to arid lands precipitates dissolved Ca in the soil, thereby forming CaCO₃, and releases CO₂ to the atmosphere, via the reaction in Eq. 2. The precipitation of calcite is also favored when large amounts of gypsum (CaSO₄ · 2H₂O), a ready source of Ca, are used to remediate dryland soils. The formation of pedogenic carbonate in arid, agricultural soils, as a result of these human activities makes a small contribution to the increase in atmospheric CO₂.^[13]

Human activities leading to the formation of acid rain also affect soil carbonates. If the acidity in rainfall is derived from sulfuric acid (H₂SO₄), then CO₂ is released when the rain falls on carbonate-rich soils. The reaction is as follows:



Gypsum (CaSO₄ · 2H₂O) precipitates as the soil dries. Although most fossil fuels contribute small amounts of sulfur to the atmosphere, the global amount of CO₂ derived from this reaction is relatively small compared to the direct release of CO₂ from fossil fuel combustion (Fig. 1).

CONCLUSION

Although the amount of pedogenic carbonate in world soils is quite large, perhaps as much as 1000×10^{15} g C, this pool of C is relatively sluggish. The long MRT of the pedogenic carbonate pool ensures that it will not become a major sink or source of atmospheric CO₂ over the next several centuries.

REFERENCES

1. Jobbagy, E.G.; Jackson, R.B. The vertical distribution of soil organic matter and its relation to climate and vegetation. *Ecol. Appl.* **2000**, *10*, 423–436.
2. Eswaran, H.; Reich, P.F.; Kimble, J.M.; Beinroth, F.H.; Padmanabhan, E.; Moncharoen, P. Global carbon stocks. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2000; 15–25.
3. Batjes, N.H. Total carbon and nitrogen in soils of the world. *Eur. J. Soil Sci.* **1997**, *47*, 151–163.
4. Schlesinger, W.H. Carbon storage in the caliche of arid soils: A case study from Arizona. *Soil Sci.* **1982**, *133*, 247–255.
5. Capo, R.C.; Chadwick, O.A. Sources of strontium and calcium in desert soil and calcrete. *Earth Planet. Sc. Lett.* **1999**, *170*, 61–72.
6. Chiquet, A.; Michard, M.; Nahon, D.; Hamelin, B. Atmospheric input vs. *in situ* weathering in the genesis of calcretes: An Sr isotope study at Gálvez (Central Spain). *Geochim. Cosmochim. Acta.* **1999**, *63*, 311–323.
7. Naiman, Z.; Quade, J.; Patchett, P.J. Isotopic evidence for Eolian recycling of pedogenic carbonate and variations in carbonate dust sources throughout the southwest United States. *Geochim. Cosmochim. Acta.* **2000**, *64*, 3099–3109.
8. Schlesinger, W.H. The formation of caliche in soils of the Mojave Desert, California. *Geochim. Cosmochim. Acta.* **1985**, *49*, 57–66.
9. Schlesinger, W.H. *Biogeochemistry: An Analysis of Global Change*; Academic Press: San Diego, 1997.
10. Monger, H.C.; Martinez-Rios, J. Inorganic carbon sequestration in grazing lands. In *The Potential of U.S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect*; Follett, R.F., Kimble, J.M., Lal, R., Eds.; CRC Press: Boca Raton, 2000; 87–118.
11. Chadwick, O.A.; Kelley, E.F.; Merritts, D.M.; Amundson, R.G. Carbon dioxide consumption during soil development. *Biogeochemistry* **1994**, *24*, 115–127.
12. Marion, G.M. Correlation between long-term pedogenic CaCO₃ formation rate and modern precipitation in deserts of the American southwest. *Quaternary Res.* **1989**, *32*, 291–295.
13. Suarez, D.L. Impact of agriculture on CO₂ as affected by changes in inorganic carbon. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2000; 257–272.

Inorganic Carbon: Global Stocks

Larry P. Wilding

Department of Soil and Crop Sciences, Texas A&M University, College Station, Texas, U.S.A.

Lee C. Nordt

Department of Geology, Baylor University, Waco, Texas, U.S.A.

John M. Kimble

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Lincoln, Nebraska, U.S.A.

Abstract

There is a very large global pool of soil inorganic carbon (SIC), but its formation, quantity, and dynamics are uncertain. The major problems in determining SIC stocks are the incomplete global database and the difficulty in separating pedogenic from lithogenic carbonate sources.

INTRODUCTION

More work has focused on the role of soil organic carbon (SOC)^[1–5] than soil inorganic carbon (SIC)^[6,7] in the global carbon cycle. One of the constraints to quantifying SIC stocks is the incomplete global dataset. Thus, we can estimate global SIC stocks but with considerable uncertainty. Further, there is difficulty in differentiating lithogenic from pedogenic carbonates because most routine chemical methods determine the total quantity of carbonates in a soil sample without regard to origin. The amount of pedogenic carbonate formed in soils is typically estimated from visible segregations in the field, from chemical analysis in the laboratory under the assumption that the carbonate is pedogenic,^[8] or from stable carbon isotope analysis to differentiate lithogenic from disseminated pedogenic carbonate components.^[9] Gile and Grossman^[10] consider that much of the calcium in carbonates results from carbonate dusts or from calcium in rainwater. Chadwick and Capo^[11] believe that at least 95% of the Ca in carbonates of arid and semiarid regions is derived from dusts. Many soils also have lithogenic carbonate inherited from alluvial, till, and bedrock parent materials. It is possible that as little as 10% of the total SIC reported in soils is formed from the authigenic weathering of calcium-bearing minerals.

Perhaps even less well understood are the flux rates of SIC stocks (e.g., soluble carbonates, bicarbonates, and carbonic acid) from soil systems into the vadose zone, geologic substrata, ground water aquifers, rivers, lakes, and oceans. The flux rates of inorganic carbon stocks require knowledge of soil chronology coupled with mass balance reconstruction analyses to determine the magnitude of

gains or losses.^[12,13] This long-term transitory SIC stock is believed to have important impacts on soil carbon sequestration over geologic time periods up to 10 times longer than the SOC stocks.^[12–14] Also, the biogenesis of SIC stocks by various organisms (e.g., plant roots, blue green algae, termites, plankton, etc.), while well documented, is not as well quantified.

MORPHOLOGY OF SIC STOCKS

The major SIC stocks are in petrocalcic and calcic horizons. It is estimated that calcic horizons contain 209 Pg C (10^{15} g C), petrocalcic horizons 104 Pg C, and calcareous soils not meeting the taxonomic criteria for a named carbonate horizon about 636 Pg C.^[6] Disseminated carbonates (micritic or fine-grained carbonate particles) and segregated forms of soft masses, threads, films, or filaments are often present in soils at quantities insufficient to be recognized as calcic horizons.^[8] Such disseminated and segregated carbonates may be the most chemically reactive form of SIC.

MAGNITUDE OF SIC STOCKS

The amount of soil global inorganic carbon is large, with estimates ranging from 1576 to 695 Pg C.^[3,15,16] Eswaran et al.^[17] revised the estimates down to between 949 to 940 Pg C, which appear to be more reasonable (see [Tables 1](#) and [2](#)). Uncertainties in these numbers result because the depth of the soil represented has been arbitrarily set at 1 m, the values represent combined lithogenic and pedogenic sources, and the data for SIC stocks that are partitioned by soil taxa and global areal extent are approximate.

Table 1 SIC and TC in soil orders of soil taxonomy

Order	TC		SIC		SIC/TC % of order	SIC/TC % global
	Pg	% Global	Pg	% Global		
Alfisols	201	8.1	43	4.5	21.4	1.7
Andisols	20	0.8	0	0.0	0.0	0.0
Aridisols	515	20.9	456	48.5	88.5	18.5
Entisols	353	14.3	263	28.0	74.5	10.6
Gelisols	323	13.1	7	0.8	2.2	0.3
Histosols	180	7.2	0	0.0	0.0	0.0
Inceptisols	224	9.1	34	3.6	15.2	1.4
Mollisols	237	9.5	116	12.3	48.9	4.7
Oxisols	126	5.1	0	0.0	0.0	0.0
Spodosols	64	2.6	0	0.0	0.0	0.0
Ultisols	137	5.6	0	0.0	0.0	0.0
Vertisols	64	2.6	21	2.3	32.8	0.9
Miscellaneous	24	1.0	0	0.0	0.0	0.0
Total	2468	100.0	940	100.0	38.1	

Source: Adapted from Eswaran, Reich, et al.^[17]

Discrepancies in areal extents of soil orders/suborders, and the SIC stocks in Tables 1–4, are due to rounding errors and different sources of published information.

SIC STOCKS RELATED TO SOIL TAXONOMY ORDERS AND SUBORDERS

Areal stocks of SIC in soil orders are presented in Table 1 and partitioned by the suborders of soil taxonomy in Table 2.^[8] About 38% of the total carbon (TC) stocks are accounted for by SIC, with Aridisols, Entisols, and Mollisols contributing the most (Table 1). Nearly half of the SIC occurs in Aridisols (Table 1) with over 85% distributed among Argids, Calcids, and Cambids (Table 2). Entisols contain 28% of SIC (Table 1) with 77% associated with Orthents (Table 2). Mollisols contain the next largest amount of SIC (≈12%) with over 80% stored in Ustolls. The Alfisols contribute nearly 5% of the SIC stocks with over 95% stored in Ustalfs and Xeralfs. These distributions of SIC stocks reflect heavily on the semiarid and drier climates, coupled with carbonate-rich parent materials.

Five of the soil orders contain no SIC stocks. These orders are the Andisols, Histosols, Oxisols, Spodosols, and Ultisols. They represent soils in subhumid to humid environments where carbonates have been leached into lower vadose zones, geologic substrata, and/or ground water aquifers. In these orders one would expect the parent materials to have been either non-calcareous or low in carbonate content.

The ratios of SIC/TC as a percentage of the order and as a global percentage are given in Table 1. It is noteworthy

Table 2 Listing of soil taxonomy orders, suborders, ice-free land areas, and SIC stocks.

Soil orders	Soil suborders	Area ice-free land ^a		SIC ^b	
		10 ⁶ km ²	%	Pg	%
Alfisols		12.620	9.65	43	4.6
	Aqualfs	0.836	0.64	0	0.0
	Cryalfs	2.518	1.92	1	0.1
	Ustalfs	5.664	4.33	29	3.1
	Xeralfs	0.896	0.69	12	1.3
	Udalfs	2.706	2.07	1	0.1
Andisols		0.91	0.70	0	0.0
Aridisols		15.728	12.02	456	48.5
	Cryids	0.943	0.72	17	1.8
	Salids	0.890	0.68	21	2.2
	Gypsid	0.682	0.52	23	2.4
	Argids	5.407	4.13	135	14.4
	Calcids	4.872	3.73	165	17.6
	Cambids	2.931	2.24	95	10.1
Entisols		21.137	16.16	263	28.0
	Aquents	0.116	0.09	0	0.0
	Psamments	4.428	3.39	33	3.5
	Fluvents	2.860	2.18	27	2.9
	Orthents	13.733	10.50	203	21.6
Gelisols		11.26	8.60	7	0.8
	Histels	1.01	0.77	4	0.4
	Turbels	6.33	4.84	3	0.3

(Continued)

Table 2 Listing of soil taxonomy orders, suborders, ice-free land areas, and SIC stocks. (Continued)

Soil orders	Soil suborders	Area ice-free land ^a		SIC ^b	
		10 ⁶ km ²	%	Pg	%
	Orthels	3.91	2.99	1	0.1
Histosols		1.53	1.17	0	0.0
Inceptisols		12.829	9.81	34	3.6
	Aquepts	3.199	2.45	2	0.2
	Cryepts	0.456	0.35		
	Ustepts	4.241	3.24	18	2.0
	Xerepts	0.685	0.52		
	Undepts	4.247	3.25	14	1.4
Mollisols		9.005	6.89	116	12.3
	Albolls	0.028	0.02	0	0.0
	Aquolls	0.118	0.09	1	0.1
	Rendolls	0.266	0.20	4	0.4
	Xerolls	0.924	0.71	15	1.6
	Cryolls	1.164	0.89	3	0.3
	Ustolls	5.244	4.01	94	9.9
	Udolls	1.261	0.96	0	0.0
Oxisols		9.810	0.75	0	0.0
Spodosols		3.350	2.56	0	0.0
Ultisols		11.052	8.45	0	0.0
Vertisols		3.160	2.42	21	2.3
	Aquepts	0.054	0.00	0	0.0
	Cryerts	0.014	0.01	0	0.0
	Xererts	0.098	0.08	1	0.1
	Torrerts	0.889	0.68	12	1.3
	Usterts	1.767	1.35	8	0.9
	Uderts	0.384	0.29	0	0.0
Miscellaneous land		18.405	10.0	0	0.0
Total		130.796	100.0	940	100.0

^aSoil Survey Staff.^[8]^bEswaran, Reich, et al.^[17]

that nearly 90% of the TC is SIC in Aridisols, 75% of the TC is SIC in Entisols, and about 50% of the TC is SIC in Mollisols. This reflects environments that do not favor sequestration of SOC because the soils are youthful, well oxidized, and have primary productivity constrained by long periods of soil moisture stress.

SIC STOCKS RELATED TO SOIL MOISTURE CONDITIONS

Table 3 presents SIC and TC stocks according to global soil moisture and temperature conditions. Two soil moisture

Table 3 SIC and TC stocks according to moisture conditions.

Moisture condition	TC		SIC		SIC/TC % region	SIC/TC % global
	Pg	% Global	Pg	% Global		
Permafrost	405	16.4	18	1.9	4.4	0.7
Arid	877	35.5	732	77.8	83.5	29.7
Mediterranean	90	3.6	50	5.4	55.6	2.0
Semiarid	471	19.1	134	14.2	28.5	5.5
Humid	539	21.9	4	0.5	0.7	0.2
Perhumid	85	3.4	2	0.2	2.4	0.1
Total	2472	100.0	946	100.0	38.2	

Source: From Eswaran, Reich, et al.^[17]

conditions account for 92% of the SIC stocks, namely arid ($\approx 78\%$) and semiarid ($\approx 14\%$). This is explained by the fact that these environments exhibit little soluble transport of pedogenic and lithogenic carbon stocks. In these environments, the proportion of TC, i.e. SIC, is $\approx 84\%$ and $\approx 29\%$, respectively, indicating that accumulation of SOC is not as important in these dry climates as in humid and wetter regimes where SOC comprises $\approx 97\%$ of the TC stock (Table 3). It is interesting to note that in Mediterranean climates with winter rain, SIC contributes $\approx 56\%$ of the TC stock. Again, because of the semiarid condition the SOC pool is less important than SIC. From the above comments, it is not surprising that a little over 38% of the TC stock (2472 Pg C) is SIC (946 Pg C), with arid (30%) and semiarid (6%) regions serving as major contributors.

SIC STOCKS RELATED TO ECOLOGICAL REGIONS

Table 4 illustrates SIC and TC stocks partitioned by ecological regions. Boreal and temperate regions contribute $\approx 82\%$ of the global SIC, with tropical regions $\approx 16\%$ and tundra regions $\approx 2\%$. The SIC is much higher in the temperate region (55%) due to the larger extent of deserts. The small contributions of SIC to TC in tundra regions reflect cold temperatures favorable to SOC

Table 4 SIC and TC stocks according to ecological regions.

Ecological region	TC		SIC		SIC/TC % Region	SIC/TC % Global
	Pg	% Global	Pg	% Global		
Tundra	405	16.4	18	1.9	4.4	0.7
Boreal	632	25.6	256	27.2	40.5	10.4
Temperate	873	35.4	518	55.1	59.3	21.0
Tropical	557	22.6	149	15.9	26.7	6.0
Total	2466	100.0	940	100.0	38.1	

Source: From Eswaran, Reich, et al.^[17]

accumulation and dissolution of lithogenic SIC. Many of the tropical regions are more humid, and thus any SIC in the soil parent materials has been leached.

FLUX OF SIC STOCKS

An estimate of soluble carbonate fluxes can be made from mass balance reconstruction analysis of the soil carbonate phase. Two problems with this method are the lack of reliable estimates of soil chronology and assumptions of flux rates. The rates of soil genesis, and especially the more labile carbonate components, are known to be exponential functions, but the precise geometric rate function applicable is not known and would vary with different amounts of carbonate, carbonate mineralogy, leaching potential, soil temperature, soil permeability, and vegetative cover.

The magnitude of leaching of soluble carbonates and bicarbonates has been estimated globally from calcareous soils.^[12,13] These projections are based on a flux rate of between 7 and 8 g m⁻² yr⁻¹ from soil great groups likely to have leaching potentials sufficient to transport soluble products of carbonate weathering from the soil. Estimated results indicate that between 0.25 Pg and nearly 0.40 Pg of SIC is lost per year globally in leaching calcareous environments. The major contributors to these leachates are the Entisol, Mollisol, and Alfisol soil orders. This value probably underestimates the actual flux of SIC stocks because it does not consider the amount of carbonic acid used to weather soil minerals or the amount that is flushed from soils of high leaching potentials, such as, Oxisols, Ultisols, and Spodosols, which occupy about 20% of the ice-free earth's surface.

CONCLUSION

There is a very large global pool of SIC, but its formation, quantity, and dynamics are uncertain. The major problems in determining SIC stocks are the incomplete global database and the difficulty in separating pedogenic from lithogenic carbonate sources. To provide better estimates of SIC stocks, more sampling and analysis are needed that involve good soil chronology, stable carbon isotopes to differentiate pedogenic from lithogenic carbonates, and techniques to quantify soluble SIC stocks.

REFERENCES

1. Lal, R.; Kimble, J.M.; Eswaran, H.; Stewart, B.A., Eds. *Soil and Global Change*, Advances in Soil Science; CRC Press, Inc.: Boca Raton, 1995.
2. Lal, R.; Kimble, J.M.; Eswaran, H.; Stewart, B.A., Eds. *Soil Processes and C Cycles*; Advances in Soil Science; CRC Press, Inc.: Boca Raton, 1998.
3. Bouwman, A.F., Ed. *Soils and the Greenhouse Effect*; John Wiley and Sons: New York, 1990.
4. Wisniewski, J.; Sampson, R.N., Eds. *Terrestrial Biospheric Carbon Fluxes: Quantification of Sinks and Sources of CO₂*; Kluwer Academic Publishers: Dordrecht, 1993.
5. Lal, R.; Kimble, J.M.; Eswaran, H.; Stewart, B.A., Eds. *Soil Organic Matter in Temperate Agroecosystems*; CRC Press: Boca Raton, 1997.
6. Lal, R.; Kimble, J.M.; Eswaran, H.; Stewart, B.A., Eds. *Global Climate Change and Pedogenic Carbonates*; Lewis Publishers: Boca Raton, 2000.
7. Batjes, N.H. Total carbon and nitrogen in the soils of the world. *Eur. J. Soil Sci.* **1996**, *47*, 151–163.
8. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; USDA-NRCS Agriculture Handbook 436; U.S. Government Printing Press: Washington, 1999.
9. Nordt, L.C.; Hallmark, T.C.; Wilding, L.P.; Boutton, T.C. Quantifying pedogenic carbonate accumulations using stable carbon isotopes. *Geoderma* **1998**, *82*, 115–136.
10. Gile, L.H.; Grossman, R.B. *The Desert Project Soil Monograph*; USDA-SCS, National Technical Information Service: Springfield, 1979.
11. Chadwick, O.A.; Capo, R.C. Partitioning allogenic and authigenic sources of carbonate in New Mexico calcrete. *Agronomy Abstracts*, American Society of Agronomy, Cincinnati, Ohio, November 7–12; Madison, 1993; 295 pp.
12. Nordt, L.C.; Wilding, L.P.; Drees, L.R. Pedogenic transformations in leaching soil systems: implications for the global carbon cycle. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2000; 43–64.
13. Drees, L.R.; Wilding, L.P.; Nordt, L.C. Reconstruction of inorganic and organic carbon sequestration across broad geoclimatic regions. In *Soil Carbon Sequestration and the Greenhouse Effect*; Lal, R., Ed.; Soil Science Society of America Special Publication No 57; Madison, 2001; 155–172.
14. Chadwick, O.A.; Kelly, E.F.; Merritts, D.M.; Amundson, R.G. Carbon dioxide consumption during soil development. *Biogeochemistry* **1994**, *24*, 115–127.
15. Eswaran, H.; Van den Berg, E.; Reich, P.; Kimble, J. Global soil carbon resources. In *Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC Press, Inc.: Boca Raton, 1995; 27–44.
16. Sombroek, W.G.; Nachtergaele, F.O.; Hebel, A. Amounts, dynamics and sequestering of carbon in tropical and subtropical soils. *Ambio* **1993**, *22*, 417–426.
17. Eswaran, H.; Reich, P.F.; Kimble, J.M.; Beinroth, F.H.; Padmanabhan, E.; Moncharoen, P. Global carbon stocks. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2000; 15–26.

Inorganic Carbon: Land Use Impacts

Donald L. Suarez

U.S. Salinity Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Riverside, California, U.S.A.

Abstract

Land use affects soil inorganic carbon (C), but the changes are significant only over the long term. Both increases and decreases in C storage occur as a result of various management practices. In irrigated lands, the primary factors causing a decrease in inorganic C are high leaching and maintenance of elevated water content at or near the soil surface. These factors result in elevated carbon dioxide (CO₂) concentrations and thus increased dissolution of soil carbonates. Use of acidifying fertilizers such as ammonia and sulfur also act to reduce soil inorganic C. Practices that favor accumulation of carbonates in the soil include efficient irrigation with surface waters in arid and semiarid regions (leaching less than 30% of the applied water), irrigation with ground waters at elevated CO₂ concentrations, application of gypsum to alkaline soils, and use of nitrate fertilizer. Other factors that affect soil carbonate content include land clearing, cropping practices, and erosion.

INTRODUCTION

Soil inorganic carbon (C) constitutes a major C pool in the near surface environment. In arid regions, the inorganic C can comprise more than 90% of the total C in the soil. The major inorganic C mineral phases are calcite and dolomite. Both minerals are relatively insoluble, however, dolomite dissolution is much slower than calcite dissolution at the intermediate pH values relevant to soils. Also, dolomite does not readily precipitate under earth surface conditions. As a result, dolomite content in soils will remain constant or decrease due to dissolution, while calcium carbonate content may either increase or decrease.

LAND CLEARING AND CROPPING

Land clearing generally results in increased water runoff and soil erosion. This process or any other process such as tillage that increases erosion serves to remove the surface soil horizons. Since these horizons are generally depleted in inorganic C relative to less weathered, deeper horizons, there is an apparent increase in the inorganic C content of the surface soil as a result of erosion. In terms of carbonate dissolution, the impact of land clearing is not certain. After clearing there is increased runoff, thus decreased infiltration, favoring less dissolution of carbonates. This effect may be compensated by the decreased water consumption (lower evapotranspiration) after clearing, resulting in increased deep recharge (greater carbonate dissolution). Depending on how much biomass remains after clearing, there is likely a short-term increase in soil CO₂ followed by a longer term reduction, favoring less carbonate dissolution

in the soil. Land clearing and overgrazing in arid lands serve to increase wind erosion and to redistribute soil in the landscape. In this manner, non-calcareous soils receive inputs of carbonates. This process increases net dissolution of carbonates in the landscape.

In humid environments, inorganic C is leached from the soil. The elevated CO₂ concentrations in the soil enhance calcite solubility relative to earth surface conditions. In humid environments, carbonates are successively leached from the upper portions of the soil profile. Agricultural practices may serve to enhance or reduce the net removal of carbonates. Removal of vegetation from a site with practices such as tree harvesting or harvesting of forages serves to remove base cations and causes net acidification of the upper portions of the soil profile. If carbonates are present deeper in the soil, this acidification increases dissolution. The impact of removal of vegetation in humid environments with carbonates in the subsoil can be calculated by assuming that the net harvested alkalinity is compensated by an equal increase in carbonate dissolution in the subsurface.

FERTILIZATION

Since optimum plant growth is generally at a pH lower than that observed in untreated calcareous soils, acid fertilizers are commonly applied. Use of sulfur with subsequent oxidation to sulfate results in acid release to the soil (2 mol of protons per mole of sulfur). Application of ammonia salts, with subsequent fixation into organic matter or oxidation to nitrous oxide or elemental nitrogen, also releases protons (2 and 1 mol of protons per mole of ammonia ions,

respectively). This acidification will increase carbonate dissolution proportionately and has a significant effect, since ammonia salts are widely used.

Application of urea or ammonia gas should have no net effect on carbonate dissolution (upon oxidation to nitrous oxide or elemental nitrogen). In contrast, use of nitrate fertilizers serves to increase pH and thus reduce carbonate dissolution. Generally nitrate is not utilized on calcareous soils, so the impact on inorganic C storage in soil is slight. The quantitative impact of fertilization on changes in inorganic C is not easily calculated, as it depends on the extent of N incorporation into organic matter, mineralization, the extent to which the harvested biomass is removed from the site, and the occurrence of carbonates in the subsurface. The addition of liming products, primarily calcite, are reported as 3.7 Tg C yr^{-1} in the United States for 1978.^[1] This is a significant but temporary addition to the soil C pool as it is assumed that the majority of the material is applied to acid soils and thus it is readily dissolved.

HUMID REGION IRRIGATION

Irrigation in humid environments serves to increase the net recharge through the soils and thus should increase the removal of carbonates. These changes may be relatively difficult to detect in view of the limited amount of irrigation water added, and the fact that irrigation in humid environments, although increasing rapidly, was very limited in the past. Field studies are needed to determine the impact of irrigation on changes in inorganic C storage in humid environments.

ARID REGION SOILS

Arid zone soils usually contain at least minor amounts of carbonates, even if classified as non-calcareous. In the absence of irrigation, there may be redistribution of carbonates within the soil but little net precipitation. The majority of the pedogenic calcite is reprecipitated calcite with relatively small amounts added as a result of mineral weathering. Significant amounts of carbonates are also added to the surface of arid land soils as dust. Calcite is leached from the upper part of the soil profile by dissolution into the infiltrating rain and is mostly reprecipitated at depth after plant extraction of the available water.

Irrigation in arid and semiarid environments may result in a net increase or decrease in soil carbonate, depending on the water source and fraction of water applied that is leached (leaching fraction). There are two opposing effects. First, elevated CO_2 concentrations in the root zone relative to the atmospheric condition results in enhanced calcite solubility and dissolution. Second, plant water extraction and evaporation concentrate the salts into a smaller volume of water and enhance calcite precipitation. At low leaching fractions, the effect of concentration of salts due to plant

water extraction and evaporation is greater than the enhanced CO_2 effect, and there is net precipitation of calcite.

For a calcite-saturated surface water such as the Colorado River, it is estimated^[2] that at a leaching fraction of 0.1 there is net precipitation of $125 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of C, based on water consumption of 1.2 m yr^{-1} and a CO_2 partial pressure of 3 kPa. Model simulations indicated that net precipitation of calcite occurred as the sum of the loss of carbonates in the upper portion of the root zone and precipitation of calcite in the lower portion. At high leaching fractions, there is net dissolution of carbonates. Using a calcite equilibrium model, it is predicted that at a leaching fraction of 0.4 there will be a net dissolution of carbonates of $70 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of C, again, based on water consumption of 1.2 m yr^{-1} and CO_2 partial pressure of 3 kPa.^[3] In all instances, there is a prediction of net dissolution in the upper portion of the soil profile and net precipitation in the lower portions of the profile. Using average leaching fractions for the Western United States, it is estimated that irrigation with surface waters on 12 million hectares results in an increase in inorganic C of $1 \text{ Tg yr}^{-1[4]}$ or $80 \text{ kg ha}^{-1} \text{ yr}^{-1}$.

Irrigation with groundwater saturated with respect to calcite will result in precipitation of carbonates at almost all leaching fractions, since the irrigation water is equilibrated at the groundwater CO_2 partial pressure and is supersaturated upon degassing and application to the soil. Calcite-saturated groundwater is used for irrigation on an estimated 3.12 million hectares in the United States. It is estimated that irrigation on these soils results in a net inorganic C precipitation of 1.3 Tg yr^{-1} or $420 \text{ kg ha}^{-1} \text{ yr}^{-1}$.^[3] The above calculations are dependent on several assumptions regarding calcite equilibrium and soil CO_2 concentrations. Using a kinetic model for calcite precipitation and the measured CO_2 partial pressure in the groundwater in Palo Verde Valley, there is prediction of no net change in inorganic C at a leaching fraction of 0.5. Consistent with these predictions, the groundwater composition in Palo Verde Valley shows no evidence for net precipitation or dissolution of carbonates in the soil.

There is limited direct field evidence for the influence of irrigation on inorganic soil C, and it is difficult to be certain that differences among sites are only related to changes in management. Researchers^[4] observed a net decrease in the calcium carbonate content of three pairs of soil profiles taken from sites in Israel irrigated for approximately 40 years as compared to non-irrigated sites. The estimated input of 4.40 m of water per year at those sites is contrasted with the yearly potential evapotranspiration of 1.93 m. The observed trend is qualitatively consistent with model predictions, if we account for the input of rain and the estimated leaching fraction of 0.56. Isotopic evidence indicated that there was precipitation of pedogenic carbonate at depth, despite a net decrease in carbonate content at depth.

In a study in the San Joaquin Valley in California, researchers compared samples of a soil taken from irrigated

and native vegetation sites.^[5] They also measured a net loss of carbonates attributed to 8 years of irrigation. Net carbonate loss was estimated as $7 \times 10^3 \text{ kg ha}^{-1} \text{ yr}^{-1}$ ($800 \text{ kg C ha}^{-1} \text{ yr}^{-1}$). Leaching fractions at the site were not reported but this value corresponds to approximately 10 times greater dissolution than expected based on model simulations. However, another study by the same author^[6] found no change in total carbonate when comparing pedons with native vegetation and those irrigated for 5–25 years. In this instance, both gypsum and sulfur were applied as amendments for reclamation. Gypsum would tend to increase precipitation of carbonates, while sulfur would acidify the soil and cause net dissolution of carbonates. In an extensive study^[7] on paired soil cores (irrigated and adjacent nonirrigated sites) from the lower Colorado River basin, there were no observed changes in net inorganic C storage after 90 years of irrigation, and no isotopic C shifts indicative of recrystallization.

SODIC SOIL RECLAMATION

Reclamation of sodic soils can result in either an increase or decrease in inorganic C in the soil. Gypsum application to a sodic and alkaline soil will increase the soil carbonate content, as the increased Ca will precipitate most of the soluble bicarbonate and carbonate. Application of sulfuric acid, sulfur, or green manuring all serve to dissolve soil carbonates. Green manuring as a reclamation practice consists of incorporating plant residues into the soil and leaching with water. The high CO_2 production is combined with restricted gas transport creating very high CO_2 concentrations in the soil, dissolving large amounts of calcium carbonate. It is estimated that this process can dissolve in the order of $400\text{--}800 \text{ kg ha}^{-1}$ during a year of reclamation. Use of acid is a widespread and generally recommended practice to prevent emitter clogging in drip irrigation systems. This practice may result in total removal of carbonates within 10–20 years, for soils with less than 3% carbonate content.

IMPACT ON ATMOSPHERIC CARBON DIOXIDE (CO_2)

Dissolution of carbonates in neutral to alkaline environments results in consumption of CO_2 gas and formation of aqueous HCO_3^- , while precipitation of carbonates results in release of CO_2 . The net effect of dissolution or precipitation of soil carbonate on atmospheric CO_2 depends on the solution flow path. In regions irrigated with surface water, the dissolution of carbonates results in a net C sink. However, the high alkalinity drainage water usually flows

back to the river. The resultant degassing of carbonic acid and reprecipitation of carbonate in the river or reservoir releases CO_2 back into the atmosphere. If the water is recharged into deep aquifers, the net soil flux is preserved. In acid environments, liming of soils results in CO_2 release to the atmosphere, as there is little or no net alkalinity produced.

CONCLUSION

Land use practices have a long-term impact on soil inorganic C. Due to the large C pools in the soil, these impacts are not generally observed in short-term studies. In humid environments, the major anthropogenic impacts on inorganic C are liming of surface soils, use of ammonium (NH_4) vs. nitrate fertilizer, removal of vegetation, and erosion. In semiarid and arid environments, increased inorganic C is favored by the use of groundwater for irrigation and application of gypsum. Decreased inorganic C is favored by inefficient irrigation with surface water and application of NH_4 fertilizer. The net effect of irrigation on a global scale, neglecting the effects of fertilizer addition, is to increase soil inorganic C by 30 Tg C yr^{-1} as well as to release an equal amount of C to the atmosphere. Liming practices in humid regions throughout the world are estimated to have no net effect on inorganic soil C but release up to 85 Tg C yr^{-1} to the atmosphere.

REFERENCES

1. Voss, R.D. What constitutes an effective liming material. In *National Conference on Agricultural Limestone*; National Fertilizer Development Center: Muscle Shoals, 1980; 52–61.
2. Suarez, D.; Rhoades, J. Effect of leaching fraction on river salinity. *J. Irr. Drainage Div. ASCE* **1977**, *103* (2), 245–257.
3. Suarez, D. Impact of agriculture on CO_2 as affected by changes in inorganic carbon. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J., Eswaran, H., Stewart, B., Eds.; Lewis: Boca Raton, 1999; 257–272.
4. Magaritz, M.; Amiel, A.J. Influence of intensive cultivation and irrigation on soil properties in the Jordan valley, Israel: recrystallization of carbonate minerals. *Soil Sci. Soc. Am. J.* **1981**, *45*, 1201–1205.
5. Amundson, R.G.; Smith, V.S. Effects of irrigation on the chemical properties of a soil in the western San Joaquin valley, California. *Arid Soil Res. Rehab.* **1988**, *2*, 1–17.
6. Amundson, R.D.; Lund, L. The stable isotope chemistry of a native and irrigated typic natrargid in the San Joaquin valley of California. *Soil Sci. Soc. Am. J.* **1987**, *51*, 761–767.
7. Suarez, D.L. *Impact of Agriculture on Soil Inorganic Carbon*; Agron. Abstr.: Madison, 1998; 258–259.

Inorganic Carbon: Modeling

Leslie D. McFadden

Department of Earth and Planetary Sciences, University of New Mexico, Albuquerque, New Mexico, U.S.A.

Ronald G. Amundson

College of Natural Resources, University of California—Berkeley, Berkeley, California, U.S.A.

Abstract

Developing a numerical model of carbonate accumulation is a challenging proposition, given the remarkably complex character of the soil system. Numerical models and isotope studies have proven to be valuable tools in the study of calcic soil development. They have helped elucidate the relation of climate, vegetation, and geomorphic processes to carbonate accumulation.

INTRODUCTION

Virtually all carbon in soils of arid and semiarid regions of the world accumulates as pedogenic calcium carbonate (CaCO_3 , referred to subsequently as carbonate). The carbonate usually accumulates in layers that eventually attain the status of calcic horizons and, in much older soils, petrocalcic horizons. Numerous mechanisms for the accumulation of pedogenic carbonate in soils are recognized, but the most fundamental reason for accumulation is limited depth of soil–water movement and seasonally high evapotranspiration that favors precipitation of carbonate within the soil.^[1] Many studies of calcic soils in the past few decades demonstrate a close correspondence between the depth of pedogenic carbonate accumulation and modern annual precipitation,^[2–4] although few studies show the relationship may be more complicated.^[5] Other studies also show progressive, time-dependent accumulation in many environments;^[1] these studies have led to the generally accepted conceptual models of calcic soil development.^[6]

Development of a numerical model of carbonate accumulation, however, is a more challenging proposition, given the remarkably complex character of the soil system. Fortunately, certain aspects of calcic soils facilitate formulation of such numerical models. For example, the observed soil depth–climate relationship implies that, utilizing a sound strategy for simulation of water movement, determination of carbonate movement via solution transport is a reasonable proposition. In addition, a significant body of research shows that the majority of carbonate is derived from accumulated entrapped dust and calcium in rainwater.^[1] Finally, data pertaining to calcite geochemistry and dissolution rates in different environments are available and show that, in soils associated with typical ranges in soil carbon dioxide (CO_2), pH, and salinity, calcite is far more

soluble than virtually all silicate minerals and has more rapid dissolution rates.^[7] Consequently, a relatively simple model for carbonate movement in a soil based on relations in the $\text{CaCO}_3\text{--H}_2\text{O--CO}_2$ system can be formulated that essentially ignores the more complex chemical reactions involving aluminosilicates. Research on the nature and composition of stable and unstable isotopes in pedogenic carbonate has also helped elucidate the nature of calcic soil development and improve the design for testing the results of numerical modeling.

THE COMPARTMENT MODEL AND SIMULATIONS OF PEDOGENIC CARBONATE ACCUMULATION

The compartment-model, or “box-model,” approach to modeling of calcic soils accommodates continuously changing values among the interdependent variables that influence soil development. It enables integration of several factors that influence pedogenic carbonate accumulation and that can be explicitly considered in this model. These include soil–water movement and soil–water balance, changing soil CO_2 concentrations and temperature with depth and season, initial parent material composition, carbonate and soluble salt additions from external sources, and calcite reactant surface area. The soil profile is represented by a vertical sequence of compartments of arbitrary dimensions, with the initial characteristics of each compartment specified [i.e., texture, available water-holding capacity, and partial pressure CO_2 ($p\text{CO}_2$)]. A series of equations that enable forward modeling and simulation of evolving carbonate depth functions using the box-model approach can be derived on the basis of consideration of the factors indicated above. For example, the solubility of calcite is derived from the following equation, after Drever:^[8]

$$m^3 \text{Ca}^{2+} = \frac{(p\text{CO}_2 K_1 K_{\text{cal}} K_{\text{CO}_2})}{(4K_2 \gamma \text{Ca}^{2+} + \gamma^2 \text{HCO}_3^-)} \quad (1)$$

where K_{cal} is the calcite solubility product, K_1 , K_2 , and K_{CO_2} are dissociation constants in the carbonate system, and γCa^{2+} and γHCO_3^- are the activity coefficients of Ca^{2+} and HCO_3^- . A gravelly, permeable calcic soil probably best approximates an open system-weathering environment, in which case calcite dissolution rates are probably surface area controlled rather than diffusion controlled. Also, the dissolution rate is defined ultimately by the rate-limiting conversion of dissolved carbon dioxide (CO_2^*) to carbonic acid (H_2CO_3). At a very low solution volume to surface area ratios, and with fast, surface-controlled calcite dissolution rates, H_2CO_3 is rapidly depleted. In such circumstances, a commonly used rate equation that enables determination of dissolution rates in the CO_2 - CaCO_3 - H_2O system^[9] is

$$dC/dt = (A'k/V)(1 - C/C^*)^n \text{ mg l}^{-1} \text{ s}^{-1} \quad (2)$$

where A' is the surface area of rock in contact with water (cm^2), V the water volume (cm^3), k the reaction coefficient ($\text{mg cm l}^{-1} \text{ s}^{-1}$), n the reaction order, C the moles of calcite in solution, and C^* is the solubility of calcite. Values of n and k vary with saturation ratio, temperature, and $p\text{CO}_2$. In the model, A'/V can be specified depending on observed soil features. Eqs. 1 and 2 show that soil CO_2 content is a very important variable, but soil CO_2 contents may be highly variable.^[10] Fortunately, studies show that a depth function for $p\text{CO}_2$ that reflects prolonged seasonal respiration levels can be estimated, assuming the concentration of soil CO_2 is described by mass transport of CO_2 by gas diffusion.^[11,12] The following diffusion reaction equation, essentially Fick's Second Law for a one-dimensional case, is used in the model:

$$\partial C_s / \partial t = D_s (\partial^2 C_s / \partial z^2) + \phi_s(z) \quad (3)$$

where C_s is the concentration of CO_2 in the soil (mol cm^{-3}), t the time (s), D_s the diffusion coefficient for CO_2 in the soil ($\text{cm}^2 \text{ s}^{-1}$), z the depth in the soil (cm), and $\phi_s(z)$ is the production rate of CO_2 as a function of depth ($\text{mol cm}^{-3} \text{ s}^{-1}$). At steady state, when $\partial C_s / \partial t = 0 = D_s \partial^2 C_s / \partial z^2 + \phi_s$, the general solution to this equation to produce a simple production function is

$$C_s(z) = \phi / D_s (Lz - z^2/2) + C_0 \quad (4)$$

where C_0 is the concentration of CO_2 in the atmosphere (ppm) and L is the depth to the lower, no-flux boundary. Soil CO_2 contents with depth calculated using this method are used to calculate carbonate solubility and dissolution rates with depth.

Available water-holding capacity, infiltration, and percolation rates can be specified on the basis of laboratory soil measurements or estimated from field measurements or theoretical considerations.

Earlier versions of the simulation model included certain assumptions that simplified numerical calculations, such as simple vertical saturated flow and constant soil temperature with depth. The lack of certain types of data (e.g., variation of $p\text{CO}_2$ with depth and time of year) also constituted a limitation on utility of the model. The model did enable simulation of: 1) realistic depths and magnitudes of carbonate accumulation over thousands of years and 2) the range of effects of large climatic changes on calcic soils.^[13–16] Model results emphasized the critical roles of external Ca^{2+} influx and influence of soil CO_2 concentrations on carbonate accumulation. Model-simulated bimodal concentrations of carbonate based on theoretical, late-Pleistocene climatic conditions resembled those observed in late-Pleistocene, polygenetic soils; however, incompletely understood changes in the magnitude of climate changes, dust flux, and vegetation change in the Quaternary complicate attempts to simulate polygenetic soils.^[7,14,16]

Later versions of the model utilized important new inputs and employed routines that reflected improved understanding of key processes that strongly influence calcic soils. Studies of dust accumulation rates in the American Southwest,^[17] C, O, and Sr isotopes in carbonate and the development of more sophisticated models for unsaturated flow in calcic soils^[18] have allowed development of improved compartment models that can address new and more challenging research problems. For example, such numerical simulations demonstrate how climate changes in the Holocene might have dramatically influenced the rates and temporal patterns of soluble salt leaching and accumulation relative to pedogenic carbonate.^[19] A modeling study addressed the problem of how carbonate can occasionally accumulate at much shallower depths than those expected from the depth—annual leaching depth relationship.^[20] This study showed how carbonate can be preferentially removed from depths of a few centimeters to a few decimeters below the soil surface, while carbonate simultaneously accumulates either as collars on surface pavement clasts or in the vesicular A horizon. Model results also explain how a significant change in climate or soil erosion rates could cause the dissolution of carbonate rinds on the tops and sides of boulders and/or the tops of limestone boulders at depths of up to several decimeters, unusual features observed in some calcic soils.^[21]

ISOTOPES IN CALCIC SOILS

During weathering, parent material carbonate undergoes dissolution and reprecipitation in the soil. The carbon ($^{13}\text{C}/^{12}\text{C}$ and $^{14}\text{C}/^{12}\text{C}$) and oxygen ($^{18}\text{O}/^{16}\text{O}$) isotope ratios of pedogenic carbonate that forms from dust or parent material carbonate, or from Ca^{2+} derived from silicate weathering, are determined by isotopic composition of soil CO_2 and H_2O . These are the primary carbon and oxygen reservoirs, respectively, for the carbonate. Therefore,

pedogenic carbonate reflects only isotopic conditions of the soil and bears no memory of the isotopic composition of the rock or mineral from which it was derived.

Soil CO₂ is derived primarily from decomposition of soil organic matter and root respiration. The C isotope composition of soil CO₂ reflects: 1) the isotopic composition of these CO₂ sources; 2) the effects of the diffusion of this CO₂ toward the atmosphere; and 3) the isotopic composition of atmospheric CO₂. In the 1980s, researchers recognized that fairly simple, steady state, diffusion models could be used to reasonably explain the observed depth patterns of C isotopes in both soil CO₂ and pedogenic carbonate. The solution to the mathematical model that encompasses the a forementioned processes describes the abundance of ¹²CO₂ in soils. A related equation can be derived for ¹³CO₂, and the ratio of the two models then describes the ratio of C isotopes at any given soil depth. A similar approach can also be used to model the ¹⁴C composition of CO₂ with soil depth, with the additional complication that the two main sources of soil CO₂ (humus decomposition and root respiration) have different ¹⁴C contents, making the solution to the model slightly more complex.^[22] The C isotope diffusion model of soil CO₂ has provided the opportunity to quantitatively use pedogenic carbonates in a number of applications: 1) paleovegetation studies;^[23] 2) paleo-atmospheric CO₂ studies;^[24] and 3) radiocarbon dating of pedogenic carbonate and geomorphic surfaces.^[25,26]

The O isotopic composition of soil water is determined by the O isotope composition of precipitation and the evaporation of soil water. It has been observed that the ¹⁸O content of modern precipitation is generally correlated with mean annual temperature on a global scale,^[27] but regional differences due to storm sources can obscure these patterns.^[28] If precipitation water (once stored in the soil) is subject to evaporation, an enrichment of the remaining soil water in ¹⁸O occurs because water vapor depleted in the “heavy” isotope is preferentially removed during evaporation. Models have been made that successfully explain the key components of this process^[29] and the fact that soils subject to evaporation commonly have generally decreasing ¹⁸O contents of soil water with depth.^[30] These models have two components. The first is a vapor transport layer (describing the flow of evaporating soil water to the atmosphere through a dry soil layer), and the second is an evaporating front layer. The evaporating front layer exists below the vapor transport zone. At the evaporating front, ¹⁸O enrichment of soil water occurs as water is transferred to a vapor phase, and the remaining ¹⁸O-enriched soil water at the evaporating front then undergoes diffusional mixing with the less ¹⁸O-enriched water at greater depths. In general, these models have been more difficult to use than C isotope models due to the dynamic nature of soil water (steady state assumptions are difficult to apply) and the array of model parameters, many of them not known with certainty for most soils.

Oxygen isotopes in pedogenic carbonate have been used less extensively in paleoclimate work than C isotopes because of concern over possible evaporation of soil water that formed the carbonate. Amundson et al.^[28] demonstrated that, except for hyperarid regions, the O isotope composition of pedogenic carbonate appears to reasonably reflect that of the local precipitation. There is a growing list of studies using carbonate O isotopes in Quaternary^[28] and Tertiary^[31] paleoclimate applications.

CONCLUSION

Numerical models and isotope studies have proven to be valuable tools in the study of calcic soil development. They have helped elucidate the relation of climate, vegetation, and geomorphic processes to carbonate accumulation. The models are not able to explain the observed character of certain aspects of calcic soils, such as patterns of pedogenic carbonate development in some soil chronosequences,^[32] or the somewhat enigmatic formation of calcic soils in humid, monsoonal climates. Also, these models are not designed to simulate the evolution of very old, morphologically complex soils with petrocalcic horizons. Future models must be designed to address locally abundant calcic soils. Additional fieldwork and application of developed field and laboratory techniques will provide the basis for development of the next generation of numerical models.

REFERENCES

1. Birkeland, P.W. *Soils and Geomorphology*, 3rd Ed.; Oxford University Press: New York, 1999; 433 pp.
2. Jenny, H. *Factors in Soil Formation*; McGraw-Hill: New York, 1941; 281 pp.
3. Arkley, R.J. Calculation of carbonate and water movement in soil from climatic data. *Soil Sci.* **1963**, *96*, 239–248.
4. Retallack, G.J. The environmental factor approach to the interpretation of paleosols. In *Factors of Soil Formation: A Fiftieth Anniversary Retrospective*; Amundson, R., Harden, J., Singer, M., Eds.; Special Publication 33; SSSA: Madison, 1994; 31–64.
5. Royer, D.L. Depth to pedogenic carbonate horizon as a paleoprecipitation indicator? *Geology* **1999**, *27*, 1123–1126.
6. Machette, M.A. Calcic soils of the southwestern United States. In *Soils and Quaternary Geology of the Southwestern United States*; Weide, D.L., Faber, M.L., Eds.; Special Paper 203; Geological Society of America: Boulder, 1985; 1–21.
7. McFadden, L.D.; Tinsley, J.C. The rate and depth of accumulation of pedogenic carbonate accumulation in soils: Formation and testing of a compartment model. In *Soils and Quaternary Geology of the Southwestern United States*; Weide, D.L., Faber, M.L., Eds.; Special Paper 203; Geological Society of America: Boulder, 1985; 23–42.
8. Drever, J.I. *The Geochemistry of Natural Waters*, 2nd Ed.; Prentice-Hall: Englewood Cliffs, 1991; 437.
9. Palmer, A.N. Origin and morphology of limestone caves. *Geol. Soc. Am. Bull.* **1991**, *103*, 1–21.

10. Kiefer, R.H.; Amey, R.G. Concentrations and controls of soil carbon dioxide in sandy soil in the north Carolina coastal plain. *Catena* **1992**, *19*, 539–559.
11. Solomon, D.K.; Cerling, T.E. The annual carbon dioxide cycle in a montane soil: Observations, modeling, and implications for weathering. *Water Resour. Res.* **1987**, *23*, 2257–2265.
12. Cerling, T.E.; Quade, J.; Yang, W.; Bowman, J.R. Carbon isotopes in soils and palaeosols, and ecology and palaeoecology indicators. *Nature* **1989**, *341*, 138–139.
13. McFadden, L.D.; Amundson, R.G.; Chadwick, O.A. Numerical modeling, chemical, and isotopic studies of carbonate accumulation in soils of arid regions. *Soil Sci. Soc. Am. Spec. Publ.* **1991**, *26*, 17–35.
14. McFadden, L.D. *The Impacts of Temporal and Spatial Climatic Changes on Alluvial Soils Genesis in Southern California*; Ph.D. thesis, University of Arizona, Tucson, 1982; 430 pp.
15. Marion, G.M.; Schlesinger, W.H.; Fonteyn, P.J. Caldep: A regional model for soil CaCO_3 (caliche) deposition in southwestern deserts. *Soil Sci.* **1985**, *139*, 468–481.
16. Mayer, L.; McFadden, L.D.; Harden, J.W. Distribution of calcium carbonate in desert soils: A model. *Geology* **1988**, *16*, 303–306.
17. Reheis, M.C.; Kihl, R. Dust deposition in southern Nevada and California, 1984–1989: Relations to climate, source area, and source lithology. *J. Geophys. Res.* **1995**, *100*, 8893–8918.
18. McDonald, E.V.; Pierson, F.B.; Flerchinger, G.N.; McFadden, L.D. Application of a soil–water balance model to evaluate the influence of holocene climatic change on calcic soils, Mojave desert, California, USA. *Geoderma* **1996**, *74*, 167–192.
19. McFadden, L.D.; Crossey, L.J.; McDonald, E.V. Predicted response to calcic soil development to periods of significantly wetter climate during the late holocene (abstr.). *Geol. Soc. Am. Abstr. Progr.* **1990**, *22* (6), A252.
20. McFadden, L.; McDonald, E.; Wells, S.; Anderson, K.; Quade, J.; Forman, S. The vesicular layer and carbonate collars of desert soils and pavements: Formation, age and relation to climate change. *Geomorphology* **1998**, *24*, 101–145.
21. Treadwell-Steitz, C.; McFadden, L.D. Influence of parent material and grain size on carbonate coatings in gravelly soils, palo duro wash, New Mexico. *Geoderma* **2000**, *94*, 1–22.
22. Amundson, R.; Stern, L.; Baisden, T.; Wang, Y. The isotopic composition of soil CO_2 . *Geoderma* **1998**, *82*, 83–114.
23. Cerling, T.E. Development of grasslands and savannas in East Africa during the neogene. *Palaeogeogr. Palaeoclimatol. Palaeoecol.* **1992**, *97*, 241–247.
24. Cerling, T.E. Use of carbon isotopes in paleosols as an indicator of the $\text{P}(\text{CO}_2)$ of the paleoatmosphere. *Global Biogeochem. Cy.* **1992**, *6*, 307–314.
25. Amundson, R.; Wang, Y.; Chadwick, O.; Trumbore, S.; McFadden, L.D.; McDonald, E.; Wells, S.; DeNiro, M. Factors and processes governing the ^{14}C content of carbonate in desert soils. *Earth Planet. Sci. Lett.* **1994**, *125*, 385–405.
26. Wang, Y.; McDonald, E.; Amundson, R.; McFadden, L.D.; Chadwick, O. An isotopic study of soil in chronological sequences of alluvial deposits, providence mountains, California. *Geol. Soc. Am. Bull.* **1996**, *108*, 379–391.
27. Rozanski, K.L.; Araguás-Araguás, L.; Gonfiantini, R. Isotopic patterns in modern global precipitation. In *Climate Change in Continental Isotopic Records*; Swart, P.K., Lohmann, K.C., Mckenzie, J., Savin, S., Eds.; American Geophysical Union Monograph 78: Washington, 1998; 1–36.
28. Amundson, R.; Chadwick, O.A.; Kendall, C.; Wang, Y.; DeNiro, M. Isotopic evidence for shifts in atmospheric circulation patterns during the late quaternary in mid-north America. *Geology* **1996**, *24*, 23–26.
29. Barnes, C.J.; Allison, G.B. The distribution of deuterium and ^{18}O in dry soils. I. Theory. *J. Hydrol.* **1983**, *60*, 141–156.
30. Allison, G.B.; Hughes, M.W. The use of natural tracers as indicators of soil water movement in temperate semiarid regions. *J. Hydrol.* **1983**, *60*, 157–173.
31. Quade, J.; Cerling, T.E.; Bowman, J.R. Development of the asian monsoon revealed by marked ecological shift during the latest miocene in Northern Pakistan. *Nature* **1989**, *343*, 163–166.
32. Holliday, V.T.; McFadden, L.D.; Bettis, E.A.; Birkeland, P.W. Soil survey and soil geomorphology. In *Profiles in the History of the U.S. Soil Survey*; Helms, D., Effland, A., Dwara, P., Eds.; Iowa State University Press: Ames, 2001; 233–274.

Insect Survival: Subzero Temperature

Mike M. Ellsbury

Northern Grain Insect Research Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Brookings, South Dakota, U.S.A.

Abstract

Much of the knowledge about how soil-dwelling insects survive cold soil temperature has come from scientific studies done in arctic or alpine environments under extreme conditions. The basic concepts arising from these studies have been summarized in this entry. It is evident that insects have adapted to cold soil environments through a variety of mechanisms and that no single strategy suffices for survival in cold environments for all insects. The survival of corn rootworm eggs illustrates how some of this knowledge can be applied to an insect pest in a temperate agricultural setting.

INTRODUCTION

Insects that inhabit soil in regions where winter temperatures fall below freezing have necessarily evolved behavioral and physiological mechanisms for surviving and developing in seasonally cold environments. Reviews of literature relating to the physiological, ecological, and behavioral adaptations that allow insects to survive cold temperature appear in the work of Baust et al.,^[1] Cannon and Block,^[2] Block,^[3] and Danks.^[4] A review by Danks^[5] considers relationships between dehydration and cold hardiness in dormant insects. Danks et al.^[6] and Block^[7] have also compiled concise glossaries of scientific terms associated with cold hardiness in insects.

BEHAVIORAL ADAPTATION

Behavioral mechanisms for avoidance of low temperatures may entail movement of freezing-susceptible overwintering stages deeper into the soil or to sites protected from cold temperatures.^[8,9] Snow cover has an insulating effect that mediates winter soil temperatures, producing nearly isothermal soil temperature regimes when early snow cover occurs.^[9,10] Where snow cover is sparse or lacking soil temperature more closely follows extremes of air temperature^[11] and cold-hardiness adaptation becomes more important for overwinter survival of soil-dwelling insects. This is particularly true of soil-dwelling invertebrates of arctic regions that cannot avoid exposure to cold temperature and thus have necessarily evolved cold-hardiness traits and diapause capability that enable them to survive during exposure to subfreezing temperature. Where short warm seasons occur, modified, sometimes very prolonged, life cycles extending over more than one overwinter season are found in insects of cold environments.^[3,6,9]

PHYSIOLOGICAL ADAPTATION

Cold hardiness involves physiological and metabolic adaptations that may not always be concomitant with a depressed diapause metabolism. Diapause has been defined as hormonally controlled metabolic dormancy that occurs during a specific stage in the life cycle of an insect.^[12–14] Thus, cold hardiness may occur independent of diapause but often is an integral component of the diapause trait, and diapause expression may improve the cold-hardiness capability of the insect.^[15]

FREEZE TOLERANCE VS. FREEZE SUSCEPTIBILITY

Cold-hardy insects have been characterized into two groups: those that are tolerant to freezing and those that are susceptible to freezing.^[8,16,17] The first of these, insects that are freeze tolerant, survive ice formation in body tissues but avoid damage to intracellular components because ice formation is usually confined to extracellular fluids. Protective mechanisms for freezing resistance include elevated solute levels, presence of nucleating agents, and accumulation of cryoprotectants in body fluids. The second broad category includes insects capable of undergoing supercooling without freezing of body fluids at temperatures that are below the true freezing point or melting point.^[6] Insects that have the ability to supercool are considered freezing susceptible because they avoid freezing damage to body tissues only at temperatures above the supercooling point. Below the supercooling point, ice formation in body tissues results in death. These relationships are summarized in Fig. 1. It should, however, be noted that a strict dichotomous categorization of the cold-hardy stages of insects as either freeze tolerant or freeze susceptible may be an oversimplification.^[7] Kostal and Havelka^[18] observe that in soil

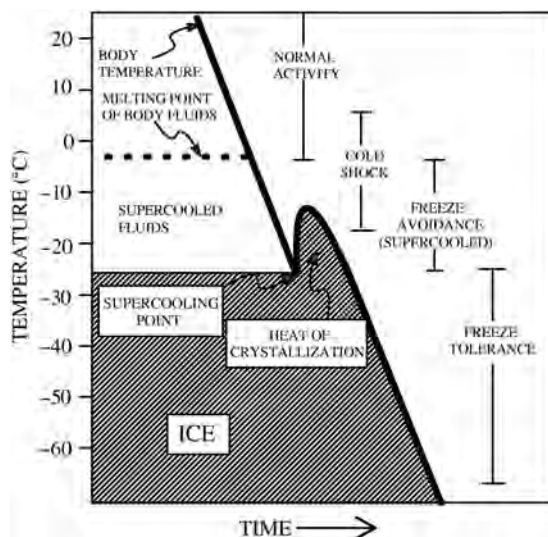


Fig. 1 Generalized diagram showing insect responses to cold temperature. Heavy line indicates insect body temperature and shaded area is a zone where ice nucleation occurs.

Source: From Danks, Kukal, et al.^[6]

environments where freezing events occur unpredictably, overwintering stages of some insects display both freeze susceptibility (supercooling) and freeze tolerance. In many insects, particularly those of more temperate environments, cold hardiness is associated with ability to survive low temperature during chilling events but is not necessarily associated with extremes of subfreezing temperature.

The capacity of insects to supercool may be enhanced by acclimation at low temperatures, still above the supercooling point, prior to exposure to subfreezing temperatures. However, where acclimation does not occur, exposure to rapidly falling temperature, even for short durations, may induce cold shock or direct chilling injury. Cold shock is defined as stress associated with short duration or rapid exposure to low, but non-freezing temperature.^[19] The rate of cooling is more significant in producing cold shock injury than are the duration or the level of low temperature attained during the chilling event. Insects subjected to cold shock sustain injury at temperatures well above the supercooling point. The mechanism of injury caused during cold shock probably involves changes in structure of lipids associated with membranes or thermoelastic stress induced by rapidly falling temperature.^[19–21]

Lee et al.,^[22] described a process for rapid cold hardening of insects during exposure to low temperature that would otherwise result in cold shock. The phenomenon has been associated with glycerol accumulation^[23] and production of stress proteins similar to those produced in response to heat shock.^[24] The rapid cold hardening phenomenon was observed in non-overwintering stages and probably provides insects with means to avoid direct chilling injury during rapid exposure to low temperatures above the supercooling point. This adaptation could be particularly

important for soil-dwelling organisms of temperate regions that may be exposed to rapid freeze–thaw events during the autumn or spring. The association of heat-shock protein production with cold hardiness suggests similarities between rapid cold hardening and insect response to heat stress by production of heat-shock proteins.

Mortality during cold exposure may result from dehydration during the freezing process, even in insects that are freeze tolerant.^[8] Indeed, Ring and Danks^[25] suggested that physiological and behavioral traits considered adaptive to cold hardiness may also be adaptive to survival in dry winter conditions. Block^[26] concluded that resistance to desiccation may be considered preadaptive to cold hardiness in terrestrial arthropods.

CORN ROOTWORMS

The ability of certain *Diabrotica* beetle species (Coleoptera: Chrysomelidae) to survive winter desiccation in tropical climates may have preadapted them for survival in more northern climates. Pest *Diabrotica*, of the corn rootworm complex, are thought to have expanded their range northward from tropical regions into areas of more temperate climate concomitantly with the agricultural development of corn, *Zea mays* L., as a primary host plant. Western corn rootworms, *Diabrotica virgifera virgifera* LeConte, considered by Krysan^[27] to be of tropical origin, survive cold winter conditions in temperate regions as a diapausing egg stage in the soil. The diapause mechanism that enabled survival of eggs during seasonal desiccation in the tropics also probably preadapted the soil-dwelling egg stage of this insect to survive in frozen soils.^[28] A closely related subspecies, the Mexican corn rootworm, *D. virgifera zea*, also overwinters as a diapausing egg stage in the soil but is adapted to survive dry winters in more southern climates.^[29] These observations are consistent with the suggestion by Block^[7] that insects tolerant of desiccation also are probably preadapted for survival of cold temperatures.

In the northern Great Plains, the *Diabrotica* pest complex consists of two species, the western corn rootworm, *D. virgifera virgifera*, and the northern corn rootworm, *D. barberi*. Because both species overwinter as eggs in the soil, the occurrence of an economic infestation during the following growing season depends in part on the overwintering survival of the egg. Chiang,^[30] Chiang et al.,^[31] Calkins and Kirk,^[32] and Gustin^[33,34] determined that winter soil temperatures reach levels low enough to cause mortality in overwintering western and northern corn rootworm eggs in the northern Great Plains. Nonetheless, a significant proportion of eggs laid each fall frequently survive winter conditions in corn-growing areas of North America to produce economic infestations of these insects each year.

Ellsbury et al.^[35,36] have shown that adult emergence patterns of both rootworm species have highly variable

spatial distributions in the field. These distributions are probably the result of behavioral and mortality factors operating on eggs in the soil at two points in the seasonal life cycle of these insects. Firstly, ovipositional patterns may vary spatially in three dimensions with adult response to corn maturity, and soil texture and moisture conditions. Secondly, distributions of eggs in the soil may change from those that exist just after the oviposition period because of differential overwinter mortality. The influences of fall tillage, soil characteristics, snow cover, and soil temperature are known to mediate egg mortality, but the interactions between these factors that produce spatial and temporal variation in surviving rootworm populations are poorly understood.

The effect of low winter soil temperature on corn rootworm eggs has been the subject of intense study. Some early research on this subject was driven by the need for laboratory rearing of large quantities of postdiapause eggs for artificial infestation and thus was focused on the biology of egg diapause, temperature limits for development, and optimal chilling conditions for storage of eggs. Much research effort has been directed toward determining developmental thresholds and diapause requirements of rootworms in cold temperature environments; little information is available about cold hardiness and the actual cold hardening mechanism of the overwintering egg stage, nor is it known whether corn rootworm eggs can be considered freeze tolerant or freeze susceptible.

CONCLUSION

Much of the knowledge about how soil-dwelling insects survive cold soil temperature has come from scientific studies done in arctic or alpine environments under extreme conditions. The basic concepts arising from these studies have been summarized in this contribution. It is evident that insects have adapted to cold soil environments through a variety of mechanisms and that no single strategy suffices for survival in cold environments for all insects. The discussion of survival of corn rootworm eggs was intended to illustrate how some of this knowledge can be applied to an insect pest in a temperate agricultural setting. There are of course other soil-dwelling insects that overwinter in various life stages in both agricultural and natural settings that, for want of space, could not be considered here. However, it can be said that these basic concepts and references provided herein should serve as a point of departure for the reader wishing to undertake more in-depth study or research on the survival of insects in frozen soil environments.

REFERENCES

1. Baust, J.G.; Lee, R.E. Jr.; Ring, R.A. The physiology and biochemistry of low temperature tolerance in insects and other terrestrial arthropods: A bibliography. *Cryo-Lett.* **1982**, *3*, 191–212.
2. Cannon, R.J.C.; Block, W. Cold tolerance of microarthropods. *Biol. Rev.* **1988**, *63*, 23–77.
3. Block, W. Cold tolerance of insects and other arthropods. *Phil. Trans. R. Soc. Lond. B.* **1990**, *326*, 613–633.
4. Danks, H.V. Winter habitats and ecological adaptations for winter survival. In *Insects at Low Temperature*; Lee, R.E. Jr., Denlinger, D.L., Eds.; Chapman and Hall: New York, 1991; 31–259.
5. Danks, H.V. Dehydration in dormant insects. *J. Insect Physiol.* **2000**, *46*, 837–852.
6. Danks, H.V.; Kukal, O.; Ring, R.A. Insect cold-hardiness: Insights from the Arctic. *Arctic* **1994**, *47* (4), 391–404.
7. Block, W. Insects and freezing. *Sci. Prog.* **1995**, *78* (4), 349–372.
8. Lee, R.E. Jr. Insect cold-hardiness: To freeze or not to freeze. *Bioscience* **1989**, *39* (5), 308–312.
9. Coulson, S.J.; Hodkinson, I.D.; Block, W.; Webb, N.R.; Worland, M.R. Low summer temperatures: A potential mortality factor for high arctic soil microarthropods. *J. Insect Physiol.* **1995**, *41*, 783–792.
10. Sharratt, B.S.; Baker, D.G.; Wall, D.B.; Skaggs, R.H.; Ruschy, D.L. Snow depth required for near steady state soil temperatures. *Agr. Forest Meteorol.* **1992**, *57*, 243–251.
11. Coulson, S.J.; Hodkinson, I.D.; Strathdee, A.T.; Block, W.; Webb, N.R.; Bale, J.S.; Worland, M.R. Thermal environments of arctic soil organisms during winter. *Arctic Alpine Res.* **1995**, *27* (4), 364–370.
12. Beck, S.D. *Insect Photoperiodism*, 2nd Ed.; Academic Press: New York, 1980; 387 pp.
13. Saunders, D.S. *Insect Clocks*, 2nd Ed.; Pergamon Press: Oxford, 1982; 288 pp.
14. Tauber, M.J.; Tauber, C.A.; Masaki, S. *Seasonal Adaptations of Insects*; Oxford University Press: New York, 1986; 411 pp.
15. Denlinger, D.L. Relationship between cold hardiness and diapause. In *Insects at Low Temperature*; Lee, R.E. Jr., Denlinger, D.L., Eds.; Chapman and Hall: New York, 1991; 174–198.
16. Salt, R.W. Principles of insect cold-hardiness. *Annu. Rev. Entomol.* **1961**, *6*, 55–74.
17. Block, W. Cold hardiness in invertebrate poikilotherms. *Comp. Biochem. Physiol.* **1982**, *73*, 581–593.
18. Kostal, V.; Havelka, J. Diapausing larvae of the midge *Aphidoletes aphidimyza* (Diptera: Cecidomyiidae) survive at subzero temperatures in a supercooled state but tolerate freezing if inoculated by external ice. *Eur. J. Entomol.* **2000**, *97* (3), 433–436.
19. Denlinger, D.L.; Joplin, K.H.; Chen, C.-P.; Lee, R.E. Jr. Cold shock and heat shock. In *Insects at Low Temperature*; Lee, R.E. Jr., Denlinger, D.L., Eds.; Chapman and Hall: New York, 1991; 131–148.
20. Quinn, P.J. A lipid-phase separation model of low-temperature damage to biological membranes. *Cryobiology* **1985**, *22*, 128–146.
21. McGrath, J.J. Cold shock: Thermoelastic stress in chilled biological membranes. In *Network Thermodynamics, Heat and Mass Transfer in Biotechnology*; Diller, K.R., Ed.; United Engineering Center: New York, 1987; 57–66.

22. Lee, R.E. Jr.; Chen, C.-P.; Denlinger, D.L. A rapid cold-hardening process in insects. *Science* **1987**, *238*, 1415–1417.
23. Chen, C.-P.; Denlinger, D.L.; Lee, R.E. Jr. Cold shock injury and rapid cold hardening in the flesh fly, *Sarcophaga crassipalpis*. *Physiol. Zool.* **1987**, *60* (3), 297–304.
24. Burdon, R.H. Heat shock and the heat shock proteins. *Biochem. J.* **1986**, *240*, 313–324.
25. Ring, R.A.; Danks, H.V. Desiccation and cryoprotection: Overlapping adaptations. *Cryo-Lett.* **1994**, *15*, 181–190.
26. Block, W. Cold or drought—The lesser of two evils for terrestrial arthropods? *Eur. J. Entomol.* **1996**, *93*, 325–339.
27. Krysan, J.L. Diapause in the nearctic species of the *virgifera* group of *Diabrotica*: Evidence for tropical origin and temperate adaptation. *Ann. Entomol. Soc. Am.* **1982**, *75* (2), 136–142.
28. Krysan, J.L. Introduction: Biology, distribution, and identification of pest *Diabrotica*. In *Methods for the Study of Pest Diabrotica*; Krysan, J.L., Miller, T.A., Eds.; Springer-Verlag: New York, 1986; 1–24.
29. Krysan, J.L.; Branson, T.F.; Castro, G.D. Diapause in *Diabrotica virgifera* (Coleoptera: Chrysomelidae): A comparison of eggs from temperate and subtropical climates. *Ent. Exp. Appl.* **1977**, *22*, 81–89.
30. Chiang, H.C. Survival of northern corn rootworm eggs through one and two winters. *J. Econ. Entomol.* **1965**, *58* (3), 470–472.
31. Chiang, H.C.; Mihm, J.A.; Windels, M.B. Temperature effects on hatching of western and northern corn rootworms. *Proc. North Cent. Br. Entomol. Soc. Am.* **1972**, *27*, 127–131.
32. Calkins, C.O.; Kirk, V.M. Effect of winter precipitation and temperature on over-wintering eggs of northern and western corn rootworms. *J. Econ. Entomol.* **1969**, *62* (3), 541–543.
33. Gustin, R.D. Soil temperature environment of overwintering western corn rootworm eggs. *Environ. Entomol.* **1981**, *10* (4), 483–487.
34. Gustin, R.D. *Diabrotica longicornis barberi* (Coleoptera: Chrysomelidae): Cold hardiness of eggs. *Environ. Entomol.* **1983**, *12* (3), 633–634.
35. Ellsbury, M.M.; Woodson, W.D.; Clay, S.A.; Carlson, C.G. Spatial characterization of corn rootworm populations in continuous and rotated corn. In *Precision Agriculture*, Proceedings of the 3rd International Symposium on Precision Agriculture, Bloomington, Minnesota, Jun 23–26, 1996; Robert, P.C., Rust, R.H., Larson, W.E., Eds.; American Society of Agronomy: Madison, 1996; 487–494.
36. Ellsbury, M.M.; Woodson, W.D.; Clay, S.A.; Malo, D.; Schumacher, J.; Clay, D.E.; Carlson, C.G. Geostatistical characterization of the spatial distribution of adult corn rootworm (Coleoptera: Chrysomelidae) emergence. *Environ. Entomol.* **1998**, *27* (4), 910–917.

Integrated Nutrient Management

Bal Ram Singh

Department of Environmental Sciences, Norwegian University of Life Sciences, Aas, Norway

Ram C. Dalal

Department of Science, Information Technology, and Innovation, Queensland Government, Dutton Park, Queensland, Australia

Abstract

Integrated nutrient management (INM) balances plant nutrient supply from both inorganic and organic sources to an optimum level for sustaining the desired crop yields as well as addresses the society's environmental concerns. The organic resources that otherwise considered waste and potential environmental polluters such as biosolids and industrial by-products are utilized as plant nutrient source complemented by mineral and synthetic fertilizers and biologically fixed nitrogen in INM system. Besides plant nutrient supply, organic component of INM also contributes to the maintenance or buildup of soil organic matter, which is critical for physical, chemical, and biological functions of soil. The INM is most relevant to agroecosystems where there is a large nutrient imbalance, and continuous depletion of organic matter and environmental concerns for disposal of organic wastes and by-products occur. Caution is required, however, to avoid accumulation of toxic elements and pesticides. In practice, INM considers the following: 1) availability of on-farm and off-farm resources; 2) yield target based on resource availability; 3) nutrient requirement to achieve desired yield; 4) integration of mineral and organic fertilizers; 5) time and method of fertilizer application; 6) efficient soil and water conservation measures; and 7) sustaining physical, chemical, and biological fertility of soils.

INTRODUCTION

Integrated nutrient management (INM) implies the maintenance or adjustment of soil fertility and of plant nutrient supply to an optimum level for sustaining the desired crop productivity on one hand and to minimize nutrient losses to the environment on the other. It is achieved through efficient management of all nutrient sources. Nutrient sources to a plant growing on a soil include soil minerals and decomposing soil organic matter (SOM), mineral and synthetic fertilizers, animal manures and composts, by-products and wastes, plant residue, and biological nitrogen fixation (BNF). Irrigation water and aerial deposition generally are minor nutrient sources. The INM is, however, not a matter of conserving soil nutrient alone, but rather it combines organic and mineral methods of soil fertility management with physical and biological measures of soil and water conservation. It integrates technologies that are site specific to agronomic and socioeconomic conditions to redress nutrient imbalances and organic matter deterioration.^[1]

The cropping system, rather than a single crop, and farming system, rather than an individual field, are the focus of an INM approach for developing the practices for the main agroecological region of a country and for the various categories of farms. The INM identifies the best associations of various types of plant nutrients in different

fields for a balanced plant nutrient and plant yield, while sustaining soil fertility and controlling nutrient losses.^[2] The farming system approach helps the farmer in the choice of the most adapted INM technologies for increasing farm production and labor productivity. It is envisaged that locally available materials of plant or animal origin as by-products of agricultural activities be used or, where such materials are not available, *in situ* production of organics, such as fast-growing leguminous shrubs or green manure crops, be practiced. The latter practice will bring atmospheric nitrogen (N) into the system, and occasionally through their deep roots, nutrients leached beyond root zones and subsoil mineralized nutrients are brought to the surface layer for use by annual crops.^[2]

The INM must essentially look into three main challenges: 1) judicious use of mineral fertilizers; 2) maximize use of organic materials; and 3) minimize negative environmental impacts. The overall strategy of INM, for determining the yield-based nutrient requirements, involves several steps, for example: 1) availability of on-farm and off-farm resources; 2) yield target based on resource availability; 3) nutrient requirement to achieve desired yield; 4) integration of mineral and organic fertilizers; 5) time and method of fertilizer application; 6) efficient soil and water conservation measures; and (7) sustaining physical, chemical, and biological fertility of soils.^[1]

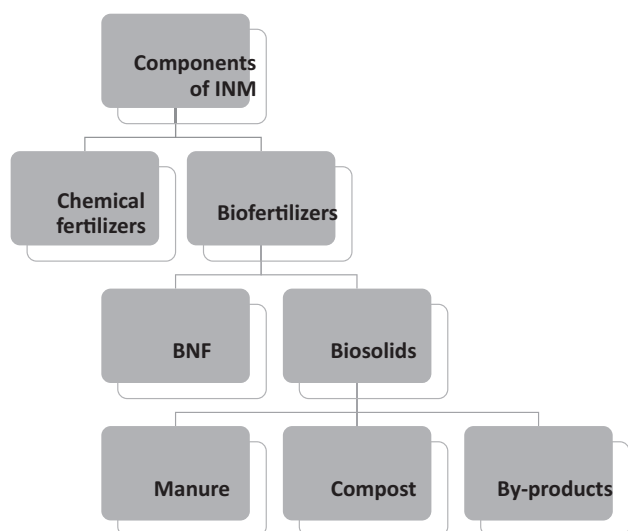


Fig. 1 Determining the yield-based nutrient requirement through a judicious combination of chemical and biofertilizers.

COMPONENTS OF INM

The main components of an INM system are shown in Fig. 1. The figure shows a judicious integration of mineral, synthetic, and biofertilizers, both organic materials and BNF.

Mineral and Synthetic Fertilizers

Fertilizers have played an important role in increasing the food supply all over the world. It is estimated that about 50% of the annual global food harvest comes from the application of mineral N fertilizer alone.^[3] In most regions of the world, increase in food production since 1980s has resulted mainly from raising yields per unit of land through increased mineral fertilizer use. The major exceptions are Africa and the Cerrado of Brazil, where most of the growth in production is caused by expansion of the cultivated area.^[3] Since 1960s, grain production in India has gone hand in hand with fertilizer consumption. The grain production increased from 52 MT in 1950/1951 to about 200 MT in 1998, while the increase in fertilizer nitrogen, phosphorus, and potassium (NPK) consumption during the same period increased from about 0.07 to 14.3 MT.^[4] It must, however, be emphasized that, along with fertilizer consumption, use of high-yielding varieties and increased irrigation facilities have played the key role in this spectacular rise in grain production.

Fertilizer consumption (N, P, and K) in the world and South Asia increased by several folds between 1975 and 2000 (Table 1). The increase in N use was only twofold on a worldwide basis, while it was about sixfold in South Asia. The corresponding increases in P and K use in South Asia were about eight- and sixfold, respectively. The average application rates of mineral fertilizers in different

Table 1 Trends in fertilizer use in the world and South Asia (in millions of metric tons).

Year	World				South Asia			
	N	P	K	Total	N	P	K	Total
1975	44	26	21	91	2.6	0.6	0.3	3.5
1985	71	35	26	133	7.2	2.5	0.8	10.5
1995	79	31	20	130	12.9	3.5	1.2	17.6
2000	87	35	23	145	14.8	4.6	1.7	21.1

Source: Adapted from Maene.^[5] ©1998 CAB International.

development regions of the world have also increased greatly, while the rate applied in Sub-Saharan African (SSA) countries has continued to be very low. For example, the weighted average of plant nutrients in East Asia and Pacific increased from 36.4 kg ha⁻¹ in 1970 to 190.3 kg ha⁻¹ in 1990. The corresponding rates in Latin America and Caribbean were 20.3 and 46.8 kg ha⁻¹, respectively, while these rates in SSA countries remained at 3.3 and 8.9 kg ha⁻¹.^[6] This has resulted in continuous decrease in per capita food production in Africa from 1980 to 1995 in contrast to sustained increase in Asia and Latin America. Assuming the base line of 1979–1981 at 100, per capita food production in Africa decreased to 93 in 1995, whereas the production during the same period in Asia and Latin America increased to 128 and 112, respectively.^[7]

Mineral fertilizers supply three major nutrients, and some management aspects of these nutrients are discussed below.

Nitrogen

Crop response to N is universal and to produce 1 t of grain, cereal crops require 15–30 kg N.^[8] Since food demands in developing countries are increasing, fertilizer N will be the most needed nutrient. N is difficult to retain in soils, and it is subjected to leaching to underground layers. Estimates of 40–50% of the mineralized N being lost under high rainfall environments of West Africa have been reported.^[9] However, leaching of mineralized NO₃-N is slower than that of NO₃-N applied as fertilizer, mainly due to time taken for the NO₃-N to diffuse to the larger pores and channels through which water drains preferentially.^[10] N is also lost to the atmosphere through denitrification and ammonium volatilization, and thus, its fertilizer use efficiency is low. The average recovery of N by rice crop in the farmer's fields in five countries was found to be only 30%,^[8] while it was 40–60% in other crops.^[4] In a long-term experiment (22 years) in India, it was found that N recovery was 17% in maize and 32% in wheat with application of N alone, but the recovery was almost doubled (33% in maize and 65% in wheat) when P and K along with N were applied.^[11] The later treatment also resulted in buildup of organic carbon (C) in the soil. Singh and Swarup^[12] also

made similar observations based on the results from several long-term experiments from different parts of India. They also found that continuous use of N alone resulted in the greatest decline in yield at all centers, indicating that other major and micronutrients were becoming limiting factors and adequate response to N could not be obtained unless the yield-limiting factors were remedied.

Phosphorus

P deficiency is considered as the main biophysical constraint for food production in humid and subhumid tropics. P dynamics in soils is complex, as it involves both chemical and biological processes and the long-term effects of sorption processes.^[13] The P supply through organic materials (crop residue and manures) is often inadequate to meet crop requirement, and thus, P fertilizer is necessary to overcome P depletion. Recovery of P is even lower than N, only 10–20%. However, unlike N, applied P is not lost from the soil except through soil erosion and gets accumulated into its reserves. In the high P-sorbing soils of Cerrado, Brazil, large application of P can replenish P, and the residual effects of P can last for 5–10 years.^[14] These corrective applications along with subsequent maintenance application and sound agronomic practices have proved very successful in this region.^[14]

High-reactive phosphate rock (PR) can be used in acid soils with similar effectiveness as superphosphate, but seldom in soils with pH above 6.2.^[15,16] High to medium reactive (>15 g citrate soluble P kg⁻¹), sedimentary PR deposits found in several African countries have been found suitable for direct application, while others are effective when partially acidulated, applied in combination with soluble P fertilizers or mixed with organic inputs.^[16] Ground PR, even on neutral or alkaline soils in India, has proved beneficial with the help of P-solubilizing organisms and organic residues.^[4] On-farm research from western Kenya shows the potential of combining organic and inorganic sources of P in moderately P-sorbing soil.^[13] They found that the application of *Tithonia* without P doubled maize yields in comparison to equivalent N rate as urea. The combination of *Tithonia* biomass with 250 kg P ha⁻¹ as Minjingu PR increased maize yields fivefold. The benefit of *Tithonia* was partially attributed to 60 kg K ha⁻¹ with the plant material. The integration of locally available organic resources with P fertilizer may be the key to increasing and sustaining levels of P on smallholder African farms.

Potassium

Unlike N and P, most K is retained in straw; thus, if crop residue is returned to the soil, the bulk of the depleted K can be replenished. However, soils such as Oxisols and Ultisols are too deficient in reserve K, and profitable yields cannot be obtained without K application. Singh and Goma^[17] reported maize response to K application on an Oxisol

3 years after the start of a long-term experiment in Zambia. With continuous mining of K with high-yielding varieties, several areas in the Indo-Gangetic alluvium have become deficient and good response to K application is reported.^[4] The results indicate that increased intensity of cropping will lead to a greater need for K input.

Biofertilizers

Although the judicious use of mineral fertilizer can play a critical role in preventing resource degradation that results from nutrient mining and from the exploitation of fragile lands or the clearing of habitat-rich forests, negative environmental impacts of mineral fertilizer application remain a matter of global concern. Therefore, mineral fertilizer application integrated with biofertilizers (organic fertilizers, crop residues, manures and wastes, and BNF) has received greater attention in sustaining crop productivity and replenishing soil fertility in many countries.

The rate of supply of nutrients from biofertilizers is generally slower than that from mineral and synthetic fertilizers, and thus, the former are essentially slow-release fertilizers. Nutrients are slowly released from biofertilizers through the microbial decomposition of organic materials. Therefore, the combined use of mineral fertilizers (fast nutrient release) with biofertilizers (slow nutrient release) in an INM system is capable of synchronizing nutrient supply with nutrient needs throughout crop growth, thereby maximizing nutrient use and minimizing nutrient loss and environmental pollution. Farmers make limited use of this concept, primarily because there are no routine techniques available to rapidly assess the nutrient availability from biofertilizers and then adjust the mineral fertilizer rates accordingly.

Biologically fixed nitrogen, crop residue, organic manures and composts, and sewage sludge as biofertilizers are discussed in the following section.

BNF, Leguminous Crop Rotations, Agroforestry, etc.

The major sources of BNF are the legume–*Rhizobium* symbiosis and Cyanophyta (blue-green algae), which fix atmospheric N into organic forms in soil. Non-symbiotic N fixers (*Azospirillum* and *Azotobacter*) and others such as *Azolla* and *Actinorrhizae* (*Frankia*) make a small, but significant, contribution to N cycling in natural ecosystems. On a global scale, it is estimated that BNF provided about 140 Mt of N compared with 108 Mt of fertilizer N in 2011, thus emphasizing the contribution of BNF to global N cycle. The legume–*Rhizobium* symbiosis fixes atmospheric N from 35 to 300 kg N ha⁻¹ in a growing season and leaves substantial amount of N (20–150 kg N ha⁻¹) for the following crop.^[18]

Agroforestry that includes legume trees provides another venue for BNF benefits, in addition to recycling

of mineral nutrients brought up from below the root zone of annual crops, organic amendments with PR to enhance P availability.^[18]

Most of the biological N fixed is present in organic form and therefore it must be mineralized before the following crop can utilize it. It thus has the potential to complement the crop nutrient needs with readily available nutrients such as from mineral and synthetic fertilizers in an INM system. Legume green manuring and incorporation of catch crops could also be used toward that purpose, although limitation of land and water for this practice and cheaper N fertilizers restricts its widespread use.

Biosolids, Manures, Composts, and By-Products (e.g., Crop Residue and Sewage Sludge)

Manures and composts provide N, P, and K (5–35 kg N t⁻¹, 1–10 kg P t⁻¹, and 5–35 kg K t⁻¹) not only to crops but also to other nutrients including micronutrients, as well as affect physical, chemical, and biological properties of soil. In an integrated crop–animal farming system, nutrient contribution from applied manures and composts becomes significant and synchronizes nutrient supply to crops when complemented by mineral and synthetic fertilizers. For example, Rao et al.^[19] showed that 120 kg N ha⁻¹ fertilizer application provided similar wheat yields to that obtained with 60 kg N ha⁻¹ fertilizer plus 4 t ha⁻¹ farmyard manure (FYM); besides, this quantity of manure also provided grain yield response equivalent of 9 kg P ha⁻¹ (Table 2). Since N release from manure occurs over a number of seasons (20–40% in the first season and 10–20% in subsequent seasons), the cumulative effect of manure application is much greater than that assessed in a single season. Furthermore, since N in compost is more stabilized than in manure, the rate of N release from the former would be slower.

Because P contained in PR is less available than that in superphosphate, its availability is enhanced when it is added in manure and organic wastes and composted together (phospho-compost) before its application to the

field. Bhardwaj^[20] reported that the grain yield increase following 25 kg P ha⁻¹ application from phospho-compost was essentially similar to that from superphosphate; this also avoids the acidifying effect of the latter to the soil.

In low-input farming systems, return of crop residues provides significant quantities of N, P, K, and calcium (Ca); in the aboveground crop biomass, the latter two exceed more than 60% of the total nutrient uptake. This becomes especially critical in some tropical Ultisols, Oxisols, and Alfisols, with low exchangeable Ca and K in these soils. For example, Sanchez and Benites^[21] reported that the retention of rice and cowpea crop residues returned to an Ultisol, 94% Ca and 89% K that were accumulated by the crops. Addition of legume residues of *Sesbania*, *Gliricidia*, and *Leucaena* and supplementation with fertilizers is an important component of INM system, especially in low-input farming systems.

Urbanization has necessitated the need for safe and economic disposal of the sewage sludge. Application of the sewage sludge to soil ensures the recycling of nutrients and organic matter and benefits plant production. Digested sewage sludge contains, on average, 7.5 kg N t⁻¹ and 3.9 kg P t⁻¹;^[22] the N and P contents of the sewage sludge lie within the range of manures and composts, almost 40% of N and 50% of P being available to crops in the first year of application to soil. Significant ammonia volatilization could occur from surface application of sludge, especially liquid sludge and animal slurries. Because the sewage sludge contains relatively more P than N required by crops, the application rates based on P requirements of the crops require supplementary N applications. Thus, it would require an INM approach to balance N and P needs of the crops.

ENVIRONMENTAL CONCERNS OF INM

Nutrient Losses

Removal of nutrients from the agricultural systems is through harvest products, soil erosion and leaching, and gaseous losses. Stoervagel and Smaling^[23] estimated the losses of 10 kg N, 1.7 kg P, and 8.3 kg K ha⁻¹ for soils in SSA and suggested that nutrient mining is responsible for declining soil fertility in this part of the world. Much of N losses occur through leaching, but the quantities of N lost in leaching depend on the amount of N fertilizer applied, soil type, crop grown, and climatic conditions. The losses can range from 0 to >100 kg N ha⁻¹ yr⁻¹ and they are more in soils with large rainfall intensities and low-storage capacity. The INM through organic and inorganic input of N can also minimize the leaching losses by improving soil chemical and physical properties. The accumulation of nitrate in soil was less under balanced use of nutrient application, and the use of FYM in conjunction with 100% NPK marginally increased the nitrate content in the surface layer.^[11] Most of P is lost through crop removal and surface erosion. P is

Table 2 Effects of manure and fertilizer applications on wheat yields.

	Wheat grain yield (t ha ⁻¹)				
	Fertilizer N (kg N ha ⁻¹)			Fertilizer P (kg P ha ⁻¹)	
Manure (t ha ⁻¹)	0	60	120	0	11
0	1.65	2.58	2.86	1.59	2.71
4	1.98	2.90	3.90	2.49	3.28
8	2.36	3.29	3.80	3.18	3.62
16	2.63	3.54	3.99	3.54	3.97
LSD (P < 0.05)	0.16	0.21		0.09	0.11

Note: LSD, least significant difference.

Source: Adapted from Rao, Singh, et al.^[19] ©1998 Indian Institute of Soil Science.

generally the limiting nutrient for algae growth and phosphate fertilizers applied to nutrient-deficient, leaching, and sandy soils were the main sources of P in these ecosystems of Australia.^[23] Neither fixed nor exchangeable K is very mobile in soils containing silicate minerals. However, Oxisols and Ultisols have limited capacity to maintain adequate K. Leaching losses in tropical soils are reported to vary from 52 to 165 kg K ha⁻¹ yr⁻¹, and they are especially higher under slash and burn.^[24] The INM with organic matter input is important in these soils to increase their cation-exchange capacity and nutrient retention capacity.

Toxic Accumulation

Commercial P fertilizers can be a major source of metal addition, especially cadmium (Cd), to agricultural soils.^[25] Also, urban sewage sludge and certain animal waste products, for example, pig manure, may also contain high amounts of metals.^[22] Repeated application of such wastes is reported to increase crop uptake of metals.^[26] Some accumulation of Cd in soils from fertilized fields, especially in Europe and North America, has been observed. The Cd content of some crops may reach to levels that could be harmful for human food and animal feed. Hence regulatory limits for food, the so-called “maximum permissible concentrations,” have been worked out in some countries. Research efforts are being made to reduce the Cd content of the fertilizer produced.

Soil Fertility Degradation/Improvement

The rapid loss of SOM of the topsoil after clearing of the natural vegetation or under continuous cultivation is a common phenomenon in the tropics and subtropics.^[27] In Zambia, the SOM content of an Oxisol decreased at the rate of 1.6 Mg ha⁻¹ yr⁻¹ in a long-term experiment.^[17] At Kabete, Kenya, the application of NP fertilizer maintained the soil organic C after 16 years of cultivation, but the content of organic C declined continuously in unfertilized plots, whereas the combined use of NP, FYM, and crop residue increased organic C and crop yields.^[28] Similarly, the long-term studies in subtropics^[11] and temperate regions^[5,29,30] have shown that organic matter of soil can be maintained or raised modestly by proper fertilization and especially by organic manures and crop residue management.

CONCLUSIONS AND RESEARCH PERSPECTIVES

Continual fertility depletion in developing countries and excess mineral nutrient applications to soils in many temperate, developed countries have led to nutrient and fertility imbalances in terms of nutrient depletion^[31] and imbalance, hence low-crop production on one hand, with excess

nutrients, soil, water, and air pollution on the other. INM is seen as the most useful approach to overcoming such problems in which adequate nutrient supply, in synchrony with crop demands, is provided within a framework of improved land and soil management practices. Implementation of INM systems could not only increase crop production per unit area of land to meet the demands of an increasing population but also improve the environment by utilizing nutrients in organic resources available to the community. Improved understanding of nutrient availability from these resources and fate of nutrients in an integrated (soil–plant–water–air) system is required as they impinge on the soil’s chemical, biological, and physical functions for sustainable production.

Increasing transport of nutrients from agricultural lands to settled and expanding urban areas through food chain has led to soil nutrient depletion and imbalance for crop productivity on one hand and excessive concentration and problematic disposal of nutrients through biosolids, food, and industrial wastes on the other hand. It is imperative that this cycle is reversed, and in this process, biofertilizers are produced and utilized. This requires research to recover nutrients from biosolids and wastes and to remove toxic elements and pesticides from biosolids and transport economically to croplands where these are used in INM system for crop productivity and sustainable intensification of agriculture.^[32]

REFERENCES

1. Singh, G.B.; Biswas, P.P. Balanced and integrated nutrient management for sustainable crop production. *Fert. News*. **2000**, *45*, 55–60.
2. Roy, R.N. Integrated plant nutrient systems—basic concepts, development and results of the trial network, initiation of project activities in AGLN, and need for cooperation. In *Integrated Plant Nutrient Systems*; Dudal, R., Roy, R.N., Eds.; FAO Fertilizer and Plant Nutrition Bulletin: Rome, 1995; Vol. 12, 49–66.
3. Dayson, T. World food demand and supply prospects. In *The Fertilizer Society Proceedings 367*, Cambridge, Dec, 6–8, 1995, Greenhill House: Peterborough, 1995.
4. Prasad, R. Nutrient management strategies for the next decade: Challenges ahead. *Fert. News*. **2000**, *45*, 13–30.
5. Maene, L.M. The use of and requirements for sustainable food production in Asia: Current review. In *Nutrient Management for Sustainable Crop Production in Asia*; Johnston, A.E., Syers, J.K., Eds.; CAB International: Wallingford, 1998; 21–33.
6. World Bank. *World Development Report 1992. Development and the Environment*; Oxford University Press: New York, 1992.
7. World Bank. *African Development Indicators*; World Bank: Washington, DC, 1996.
8. Doberman, A.; White, P.E. Strategies for nutrient management in irrigated and rainfed lowland rice systems. *Nutr. Cycl. Agroecosys.* **1998**, *53*, 1–18.

9. van der kruijs, A.C.B.M.; Wong, A.; Juo, A.S.R.; Wild, A. Recovery of ^{15}N -Labelled fertilizer in crops, drainage water and soil using monolith lysimeter in south-east Nigeria. *J. Soil Sci.* **1988**, *39*, 483–492.
10. Wild, A. Nitrate leaching under a bare fallow at a site in northern Nigeria. *J. Soil Sci.* **1972**, *23*, 315–324.
11. Bembi, D.K.; Biswas, C.R. Nitrogen balance and N recovery after 22 years of maize–wheat–cowpea cropping in a long-term experiment. *Nutr. Cycl. Agroecosys.* **1997**, *47*, 107–114.
12. Singh, G.B.; Swarup, A. Lessons from long-term fertility experiments. *Fert. News.* **2000**, *45*, 13–24.
13. Sanchez, P.A.; Shepherd, K.D.; Seale, M.J.; Place, F.M.; Buresh, R.J.; Isaz, A.N.; Mkwunye, A.U.; Kwasiga, F.R.; Ndiritu, C.G.; Woomer, P.I. Soil fertility replenishment in Africa: An investment in natural resource capital. In *Replenishing Soil Fertility in Africa*; Buresh, R.J., Sanchez, P.A., Calhoun, F., Eds.; SSSA Special Publ. 51; SSSA: Madison, 1997; 1–46.
14. Lopes, A.S. Soils under cerrado: A success story in soil management. In *IPA-PPI Regional Conf. for Latin America and the Caribbean*, Mexico City, June 25–28; 1996 IPA-PPI: Mexico City, 1996.
15. Frossard, E.; Brossard, M.; Hedley, M.J.; Metherell, A. Reactions controlling the cycling of P in soils. In *Phosphorus in the Global Environment, Transfer Cycles and Management*; Tiessen, H., Ed.; Scope 54; John Wiley and Sons: New York, 1995; 107–137.
16. Buresh, R.J.; Smithson, R.C.; Hellums, D.T. Building soil phosphorus capital in Africa. In *Replenishing Soil Fertility in Africa*; Buresh, R.J., Sanchez, P.A., Calhoun, F., Eds.; SSSA Special Publ. 51; SSSA: Madison, 1997; 97–109.
17. Singh, B.R.; Goma, H.C. Long-term soil fertility management in eastern Africa. In *Soil Management-Experimental Bias for Sustainability and Environmental Quality*; Lal, R., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 1995; 347–382.
18. Fillery, I.R.P. The fate of biologically fixed nitrogen in legume-based dryland farming systems: A review. *Aust. J. Exp. Agr.* **2001**, *41*, 361–381.
19. Rao, A.S.; Singh, M.; Reddy, D.D.; Saha, J.K.; Manna, M.C.; Singh, M.V. Integrated plant nutrient supply system to improve and sustain productivity of soybean–wheat system on typic haplusterts. In *Integrated Plant Nutrient Supply System for Sustainable Productivity*; Acharya, C.L., Tomar, K.P., Rao, A.S., Ganguly, T.K., Singh, M.V., Rao, D.L.N., Rao, C.S., Rupa, T.R., Eds.; Indian Institute of Soil Science: Bhopal, 1998; 78–91.
20. Bhardwaj, K.K.R. Role of organic wastes in soil fertility management. In *Fifty Years of Natural Resource Management Research in India*; Singh, G.B., Sharma, B.R., Eds.; Indian Council of Agricultural Research: New Delhi, 1999; 213–220.
21. Sanchez, P.A.; Benites, J.R. Low-input cropping for acid soils of the humid tropics: a transition technology between shifting and continuous cultivation. In *Land Development and Management of Acid Soils in Africa*; Latham, M., Ahn, P., Eds.; International Board for Soil Research Management: Bangkok, 1987; 85–106.
22. Smith, S.R. *Agricultural Recycling of Sewage Sludge and the Environment*; CAB International: Wallingford, 1996; 156.
23. Stoorvager, J.J.; Smaling, E.M.A. *Assessment of Soil Nutrient Depletion in Sub-Sahara Africa 1983–2000*; Winard Staring Centre: Wageningen, 1990.
24. Hodgkin, E.P.; Hamilton, B.H. Fertilizers and eutrophication in southwestern Australia: setting the scene. *Fert. Res.* **1993**, *36*, 95–103.
25. Malavolta, E. Potassium status of tropical and sub-tropical region soils. In *Potassium in Agriculture*; Munsen, R.D., Ed.; ASA: Madison, 1985; 163–200.
26. Singh, B.R. Trace element availability to plants in agricultural soils, with special emphasis on fertilizer inputs. *Environ. Rev.* **1994**, *2*, 133–146.
27. Krebs, R.; Gupta, S.K.; Furrer, G.; Schulin, R. Solubility and plant uptake of metals with and without liming of sludge-amended soils. *J. Environ. Qual.* **1998**, *27*, 18–23.
28. Dalal, R.C. On fertility degradation in vertisols. In *Ecology Today*; Gopal, B., Pathak, P.S., Saxena, K.G., Eds.; International Scientific Publications: New Delhi, 1998; 177–197.
29. Bekunda, M.A.; Bationo, A.; Sali, H. Soil fertility management in Africa: A review of selected research trials. In *Replenishing Soil Fertility in Africa*; Buresh, R.J., Sanchez, P.A., Calhoun, F., Eds.; SSSA Special Publ. 51; SSSA: Madison, 1997; 41–79.
30. Campbell, C.A.; Myers, R.J.K.; Curti, D. Managing nitrogen for sustainable crop production. *Fert. Res.* **1995**, *42*, 277–296.
31. Jones, D.L.; Cross, P.; Withers, P.J.A.; DeLuca, T.H.; Robinson, D.A.; Quilliam, R.S.; Harris, I.M.; Chadwick, D.R.; Edward-Jones, G. Nutrient stripping: The global disparity between food security and soil nutrient stocks. *J. Appl. Ecol.* **2013**, *50*, 851–862.
32. Garnett, T.; Appleby, M.C.; Balmford, A.; Bateman, I.J.; Benton, T.G.; Bloomer, P.; Burlingame, B.; Dawkins, M.; Dolan, L.; Fraser, D.; Herrero, M.; Hoffmann, I.; Smith, P.; Thornton, P.K.; Toulmin, C.; Vermeulen, S.J.; Godfray, H.C. J. Sustainable intensification in agriculture: Premises and policies. *Science* **2013**, *341*, 33–34.

Integrated Nutrient Management: Sustainability

G.S. Saroa

Ohio State University, Columbus, Ohio, U.S.A.

Rattan Lal

*Carbon Management and Sequestration Center, Ohio State University,
Columbus, Ohio, U.S.A.*

Abstract

The integrated nutrient management (INM) strategy involves meeting a crop's nutritional requirement to achieve the desired yield through judicious use of inorganic fertilizers, biofertilizers, biosolids, biological nitrogen fixation, and other nutrient cycling mechanisms that enhance nutrient use efficiency and optimize the use of the most limited or non-renewable resources. This entry outlines the basic principle of INM in the context of resource-poor farmers of Asia.

INTRODUCTION

The world population of 6 billion in 2000, increasing at the rate of 1.8%/yr, is expected to reach 8 billion by the year 2025 and 9.4 billion by 2050.^[1–3] Most of the population increase will occur in the developing countries. By 2050, the population would have increased by another 575 million in India, 300 million in China, 200 million in Nigeria, 200 million in Pakistan, and 140 million in Ethiopia.^[4] About 780 million people are already food insecure in the developing countries. The Asian farmers feed nearly 60% of the world population from only 25% of the world's land available for agriculture. The global demand for food may double over the period 1990–2030, with an increase of 2.5 to 3 times in the developing countries.

The green revolution, which started in the 1970s, was perceived by some as a near miracle preventing starvation in many Asian countries. It was brought about by increased yields of irrigated rice (*Oryza sativa*) and wheat (*Triticum aestivum*) grown on fertile soils with intensive use of fertilizers. However, gains in agricultural production are not being sustained where inputs are insufficient or imbalanced, soil fertility is declining, and other soil degradative processes are severe. Thus, food insecurity in the Asian region is once again a matter of grave concern. The threat to food security in Asia is caused by the degradation of hitherto fertile soils, especially because of the depletion of soil organic matter (SOM) content and associated plant nutrients. There is little scope for an increase in arable area, which is likely to decrease because of urbanization and soil degradation. Thus, a prerequisite for increasing food production is to develop, fine tune, and implement soil/crop management practices, which

enhance and maintain the SOM content and plant nutrient reserves at appropriate levels. Adopting a strategy of integrated nutrient management (INM) can increase nutrient use efficiencies, strengthen nutrient cycling, and increase and sustain food production.

SUSTAINABLE AGRICULTURE

Sustainable agriculture involves enhancing and sustaining agricultural production while preserving or improving soil fertility, maintaining a clean environment, and protecting or regenerating biological diversity and ecosystem health. The Technical Advisory Committee of the Consultative Group on International Agricultural Research defined sustainable agriculture as the “successful management of resources for agriculture to satisfy changing human needs while maintaining or enhancing the quality of the environment and conserving natural resources.”^[5] Traditional farming has been basically a sustainable agriculture, but at a low level of output supporting subsistence living. The transition from traditional to intensive agriculture is necessitated by the increase in global food demand and a widespread problem of soil and environmental degradation.

FERTILIZER MANAGEMENT

The use of plant nutrients per unit of net cropped area is more in Asia (129 kg/ha) as compared to that in the rest of the world (90 kg/ha). However, the ratio of nitrogen (N)/phosphorus (P)/potassium (K) use is quite balanced for the world (3.8:1.5:1), but not so for the Asian region (7.9:2.7:1).^[6] The wide ratio is not conducive to

sustained agricultural production and needs to be altered to achieve the desired ratio of about 4:2:1. The fertilizer consumption of China and India is about 70% of the total fertilizer consumption in Asia,^[6] and approximately 60% of the fertilizers consumed in Asia are used on rice.^[7] The increase in agricultural production in China during 1978–1995 was 32% by fertilizer use, 28% by irrigation, 17% by improved seed, 13% by machinery, and 6% by other factors.^[8] Furthermore, the use of chemical fertilizers will continue to play an important role in augmenting global food production. To enhance the efficiency of use and maximize the benefits, fertilizers must be applied at the right amount, placed in the root zone at the right time, and applied in the right formulation and combination. High crop yields deplete the nutrient reserves^[9] and, unless replenished judiciously and regularly, can lead to nutrient imbalance in soil. Indeed, crop yields in intensively cultivated irrigated areas of northwestern India are either stagnating or declining in the post-green-revolution period, because of the degree of nutrient imbalance, severity of soil degradation, and intensity of cropping. Nutrient removal by different crops for a definite yield target is presented in Table 1.

Nitrogen

Because of the large requirement of N for vigorous growth of the plant, crop response to N application is universal. N represents approximately 70% of the total fertilizer nutrients (N + P + K) consumed in Asia.^[10] N requirement for one metric ton (Mg) of cereal grains is 15–30 kg.^[11] As cereal grains are a staple food group

in Asia, an earnest effort has been made for increasing cereal production, thereby leading to increased demand for fertilizer N. This trend is likely to continue in future, and fertilizer N will continue to be the most wanted nutrient.

Rice crop on an average needs about 20 kg N to produce one Mg of grain yield.^[12] Thus, high yielding varieties producing 5 Mg of grain yield need 100 kg N from both soil and fertilizer sources. If grain yield without fertilizer (native soil N) is about 2.5 Mg/ha, it requires about 50 kg N/ha through fertilizer use. Assuming fertilizer N recovery of 40%, it involves applying 125 kg N/ha through fertilizer use for obtaining a target yield of 5 Mg/ha. The efficiency of applied N is usually low and ranges from 25% to 45% in rice and 50% to 70% in upland crops.^[13] In general, a 1% increase in the recovery rate of N will normally result in an economy of 0.15 million Mg of urea per year in India, which would be equivalent to one million Mg of food grains in terms of additional crop production.^[14]

Efficiency of fertilizer N is also dependent on the form of N applied and the ecosystem in which they are used. Meelu and Morris^[15] reported that amide (urea) and ammoniacal sources (ammonium sulfate) of N are superior to nitrate (calcium ammonium nitrate) for rice. But in case of hybrid rice, there are contrary reports from China. The superiority of ammoniacal source, however, is because of its stability under flooded conditions compared to nitrate, which is highly unstable and lost through leaching and denitrification. The reported superiority of nitrate for hybrid rice is attributed to the physiological/genetic differences between the inbreds and hybrids.^[16]

Phosphorus

Asia consumes about 30% of the world's P fertilizer,^[10] and China and India are the main P users in Asia. In contrast to N, only 2–5 kg P is removed per Mg of cereal produced, and P use efficiency by single crop rarely exceeds 15–20% of added P. Because of its relative immobility, applied P is fixed in the soil and may be absorbed by subsequent crops. The magnitude of the residual effect depends on the rate and kind of P fertilizer and cropping and management practices and largely on soil type. Thus, P fertilizers applied to a single crop often can meet the P requirement of subsequent crop/crops in a cropping system. Gill and Meelu^[17] reported that when P is applied to wheat, at the recommended rate, the following rice crop can be grown on residual P. However, when P is applied only to rice, grain yield of wheat is considerably lowered. De Datta, Buresh, and Mamaril^[7] also reported that P use efficiency can be greatly improved by the choice of the crop to be fertilized in a sequence. The fertilization of a single crop in a sequence can lead to net negative P balance in the soil. Regular monitoring of nutrient balance,

Table 1 Nutrient requirement of principle crops.

Crop	Scientific name	Yield (Mg/ha)	Nutrients (kg/Mg of yield)		
			N	P ₂ O ₅	K ₂ O
Wheat	<i>Triticum aestivum</i>	5.0	31	5	37
Rice	<i>Oryza sativa</i>	6.3	15	4	24
Corn	<i>Zea mays</i>	2.7	42	17	40
Sorghum	<i>Sorghum bicolor</i>	1.1	47	16	8
Millet	<i>Pennisetum typhoides</i>	0.6	50	18	170
Sugarcane	<i>Sachharum offinarum</i>	87.1	1.4	1	2.6
Potato (tubers)	<i>Solanum tuberosum</i>	17.6	5	2	8
Barley	<i>Hordeum vulgare</i>	2.5	22	12	60
Groundnut	<i>Erychus hypogea</i>	1.9	41	11	24
Mustard	<i>Brassica campestris</i>	0.7	31	16	40

Source: From *Fertilizers Statistics*.^[9]

therefore, is important for greater precision in scheduling fertilizer use.

The use efficiency of P can vary widely depending on soil type, crop, and the source of fertilizer. For short duration and winter crops water-soluble P (WSP) sources and nitrophosphates with 70% WSP have higher efficiency than water-insoluble or less water-soluble sources.^[18] For long duration and summer crops, the differences among sources with regard to P use efficiency are minimal. Method of P application plays an important role in P use efficiency, and about 50% of fertilizer can be saved by placement in row crops as compared with the broadcast method,^[19] while the reverse may be true for rice.

The dynamics of P in upland soils is markedly different from that of lowland soils. Available P increases during the flooding phase and decreases during the drying phase. In general, wetland rice is less responsive than all dryland crops grown on the same soil. Soil P continues to be an “enigma” to soil scientists, and INM strategies can help in solving some of the problems of soil P management.

Potassium

Asia consumes only 15% of the world's K fertilizer,^[10] and China and India are the main K-fertilizer-consuming countries in Asia. The K use efficiency is about 80%. The neglect of K application is evident from the highly imbalanced fertilizer consumption ratio in most the developing countries. Unlike N and P, response to K application is small on soils with illitic clay minerals. The response to K is usually observed in acidic and sandy soils during the summer season in the high rainfall areas where it is prone to leaching. The response of corn (*Zea mays*) and rice to K is more than that of wheat.^[20] Production of one Mg of rice grains requires 17–25 kg K/ha.^[21,22] Thus, continuous appraisal of the crop response to K in relation to soil test is important.

Other Nutrients

Among the secondary nutrients, sulfur deficiency is widespread with the continuous use of high analysis sulfur (S)-free fertilizers. The S deficiency usually occurs in soils of humid regions, which are highly oxidized, sandy, and low in total S and SOM contents. Similarly, zinc deficiency is another disorder limiting the yield of crops, especially of rice. Other micronutrient deficiencies are also appearing because of low use of organic manures and increased use of high analysis fertilizers.

INTEGRATED NUTRIENT MANAGEMENT

The multilocal testing by All India Co-ordinated Rice Improvement Project has shown a 50% decline in

the response of rice to fertilizer from 15–16 kg grain/kg NPK nutrients applied in the 1960s and 1970s to 7–8 kg grain/kg NPK applied in the 1990s.^[23] Thus, there is a strong need to assess both balanced fertilization and integrated plant nutrient management, which are not synonymous. The balanced fertilization implies balancing the ratio of applied nutrients to that of the nutrient requirements of the crop, and the INM implies combining organic and mineral methods of soil fertility management to reduce risks of nutrient imbalances and depletion of SOM reserves. The SOM is a biological buffer, which plays an important role in maintaining the balanced supply of available nutrients for plant growth. The higher SOM content reflects higher soil productivity.^[24] Therefore, application and management of biosolids as fertilizers are viable options to improve plant growth. Soils with low SOM content lose their buffering capacity and productivity and have low fertilizer use efficiency. In areas of intensive cropping, an increased level and better quality of SOM can contribute significantly to high productivity.^[25] Organic fertilizers vary widely in terms of their nutrient contents (Table 2), depending on the biosolids and their constituents.^[26]

The data from long-term experiments under diversified agroclimatic conditions in India showed that neither organic manures nor mineral NPK fertilizers alone can produce high yields under intensive farming, where nutrient turnover in soil–plant system is large.^[27] Only less than 40% of the nutrients present in bulky organic manures are available to the first crop following their application, but may have notable residual effect. On the other hand, residues of high-quality organic inputs such

Table 2 Nutrient analysis of some common biofertilizers.

Manures	Nutrient content (percent)		
	N	P ₂ O ₅	K ₂ O
Fresh dung			
Cattle	0.3–0.4	0.1–0.2	0.1–0.3
Horse	0.3–0.4	0.3–0.4	0.3–0.4
Sheep	0.5–0.6	0.4–0.6	0.3–1.0
Poultry manure (fresh)	1.0–1.8	1.4–1.8	0.8–0.9
Farm yard manure (dry)	0.4–1.5	4.0–5.0	0.3–1.9
Crop residues			
Rice	0.36	0.08	0.71
Corn	0.42	1.57	1.65
Wheat	0.53	0.10	1.10
Green manure			
<i>Sesbania aculeate</i>	0.62	—	—
<i>Cymosis tetragonoloba</i>	0.34	—	—
<i>Crotalaria juncea</i>	0.75	0.12	0.61

Source: From *Handbook of Agriculture*.^[26]

as green manures and legume tree pruning decompose quickly and may release 70–95% of their N within a season under tropical conditions,^[28] with a little impact on SOM content. The recovery of this released N may be 10–30% by the first crop and little, if any, for the second crop.^[29] The farmyard manure generally contains twice as much micronutrients as in *Sesbania* green manure on equal weight basis.^[30]

The crop residues contain appreciable amounts of plant nutrients and can contribute to soil quality. Prolonged removal or burning of the crop residues usually decreases soil nutrient reserves and SOM content.^[31] Increasing SOM content under tropical conditions often requires frequent and high doses of biosolids. Approximately 7 Mg/ha/yr dry matter of low-quality residues (roots and stems), or 10 Mg/ha/yr of high-quality residues (green manure leaves) may be required to maintain a 1% organic carbon in soils of the subhumid tropics.^[32]

CONCLUSION

There is a great scope for sustaining agricultural production through a judicious combination of mineral, organic, and microbial sources of plant nutrients. The dependence on chemical fertilizers can be reduced by supplementing the plant nutrients through the use of organic manures, green manures, crop residues, biofertilizers, and inclusion of legume crops in cropping sequences. This will lead to maintenance of soil productivity and improvement of soil physical conditions. There is a need for the development of simple, cost-effective, and easily adopted technologies for the production of organic manures. A judicious use of chemical fertilizers can reduce the risk of environment pollution because of reduction in use of fossil fuel for fertilizer production and low risks of contamination of natural waters. If the choice is to apply once within a rotation cycle, it may be applied to the most responsive crop in the sequence. Soil testing is an integral part of any INM program, and regular monitoring of soil health and diagnosis of nutrient availability are essential for minimizing risks of soil and environmental degradation.

REFERENCES

1. Hulse, J.H. *Science Agriculture and Food Security*; NRC Research Press: Ottawa, 1995; 7–28.
2. Fischer, G.; Heling, G.K. Population momentum and the demand on land and water resources. *Phil. Trans. R. Soc. Lond.* **1997**, 352 (Ser. B), 869–889.
3. Litvin, D. Dirt poor. *Economist* **1998**, 21 March, 3–16.
4. Daily, C.; Dasgupta, P.; Bolin, B.; Crosson, P.; Guerry, J.D.; Ehrlich, C.; Folke, A.M.; Jansson, B.O.; Jansson, N.; Kautsky, A.; Kinzig, S.; Levin, K.G.; Maler, P.; Pinstrip-Andersen, D.S.; Walker, B. Food production, population growth, and the environment. *Science* **1998**, 281, 1291–1292.
5. TAC. In *Sustainable Agricultural Production, Implications for International Agricultural Research*; CGIAR: Washington, 1989; 1–15.
6. Kumar, V. Balanced use of plant nutrients with particular reference to integrated plant nutrient systems in the Asian region. In *Nutrient Management for Sustainable Crop Production in Asia*, Proceedings of an International Conference on Nutrient Management for Sustainable Crop Production, Bali, Indonesia, Dec 9–12, 1996; Johnston, A.E., Syers, J.K., Eds.; CAB International: New York, 1998; 85–93.
7. De Datta, S.K.; Buresh, R.J.; Mamaril, C.P. Increasing nutrient use efficiency in rice with changing needs. *Fert. Res.* **1990**, 26 (1-3), 157–167.
8. Xie, J.C.; Xing, W.Y.; Zhou, M.J. Current use and requirements for nutrients for sustainable food production in China. In *Nutrient Management for Sustainable Crop Production in Asia*, Proceedings of an International Conference on Nutrient Management for Sustainable Crop Production, Bali, Indonesia, Dec 9–12, 1996; Johnston, A.E., Syers, J.K., Eds.; CAB International: New York, 1998; 267–277.
9. *Fertilizers Statistics*; Fertilizer Association of India, 1974–1975.
10. Food and Agriculture Organization. *FAO Fertilizer Yearbook*; FAO: Rome, 1989; Vol. 47, 3–16.
11. Dobermann, A.; White, P.F. Strategies for nutrient management in irrigated and rainfed lowland rice system. *Nutr. Cycl. Agroecosys.* **1998**, 53, 1–18.
12. Yoshida, S.; Oka, I.N. *Rice Research Strategies for the Future*; International Rice Research Institution: Los Banos, 1982; 51–78.
13. Hauck, R.D. *Nitrogen-15 in Soil Plant Studies*; International Atomic Energy Agency: Vienna, 1971; 65–80.
14. Abrol, I.P. Macronutrients in soils and crops. In *Proceedings of National Symposium on Macronutrients in Soils and Crops*, Meelu, O.P., Bajwa, M.S., Singh, R., Vig, A.C., Sidhu, P.S., Beri, V., Sadana, U.S., Eds.; PAU: Ludhiana, 1991; 1–15.
15. Meelu, O.P.; Morris, R.A. Green manure management in rice based cropping system. In *Green Manuring in Rice Farming*; International Rice Research Institution: Manila, 1988; 209–222.
16. Reddy, M.N.; Krishnaiah, K. Current status of crop response to fertilizers in different agro-climatic regions. *Fert. News* **1999**, 44 (4), 113–126.
17. Gill, H.S.; Meelu, O.P. Studies on the utilization of phosphorus and causes for its differential response in rice–wheat rotation. *Plant Soil* **1983**, 74, 211–222.
18. Meelu, O.P.; Rana, D.S.; Sharma, K.N.; Singh, R. Comparative efficiency of complex phosphatic fertilizers. *J. Indian Soc. Soil Sci.* **1977**, 25, 374–378.
19. Saroa, G.S.; Vig, A.C. Response of maize and wheat to nitrophosphates of varying water solubility. In *Proceedings of the Third Agricultural Science Congress, NAAS, PAU*, Mar 12–15, 1997; Vol. II, 17–18.
20. Meelu, O.P.; Bishnoi, S.R.; Randhawa, N.S.; Sharma, K.N. Response of wheat to graded doses of N and P on different P and organic carbon fertility classes under

- rainfed conditions. *J. Agric. Sci. Camb.* **1976**, *86*, 425–426.
21. De Datta, S.K. *Principles and Practices of Rice Production*; John Wiley & Sons: New York, 1981; 348–419.
 22. Velayutham, M.; Reddy, C.K.C. Balanced fertilization for increasing fertilizer use efficiency. In *Proceedings of FAI Regional Agriculture Committee Meeting*, New Delhi, India, 1987.
 23. Anonymous. In *Quinquennial Report of All India Co-ordinated Rice Improvement Project*; Fertilizer Association of India: New Delhi, 1996.
 24. Karama, A.S.; Marzuki, A.R.; Manwandan, I. Penggunaan pupuk organik pada tanaman pangan. In *Prosiding Seminar Nasional Penggunaan Pupuk V. Pusat Penelitian Tanah dan Agroklimat*, Bogor. Hlm; 1990; 395–425.
 25. Finck, A. Integrated nutrient management: an overview of principles, problems and possibilities. *Ann. Arid Zone* **1998**, *37* (1), 1–24.
 26. ICAR. *Handbook of Agriculture*; Indian Council of Agricultural Research: New Delhi, 1961; 78–114.
 27. Nambiar, K.K.M.; Abrol, I.P. Long term fertilizer experiments in India—An overview. *Fert. News* **1989**, *34* (4), 11–20.
 28. Giller, K.E.; Cadisch, G. Future benefits from biological nitrogen fixation: An ecological approach to agriculture. *Plant Soil* **1995**, *174*, 255–277.
 29. Mafongoya, P.L.; Nair, P.K.P. Multipurpose tree pruning as a source of nitrogen to maize under semi-arid conditions in Zimbabwe. Interaction of pruning quality and time and method of application on nitrogen recovery by maize in two soil types. *Agroforestry Syst.* **1997**, 1–14.
 30. Katyal, J.C.; Sharma, B.D. Role of micronutrients in crop production. *Fert. News* **1979**, *24* (7), 33–50.
 31. Hooker, M.L.; Schepers, J.S. Effect of residue removal vs. incorporation on N uptake and growth of winter wheat. In *Agronomy Abstracts*; ASA: Madison, 1984; 207.
 32. Janssen, B.H. Integrated nutrient management: The use of organic and mineral fertilizers. In *The Role of Plant Nutrients for Sustainable Food Crop Production in Sub-Saharan Africa*; Van Reuler, H., Prins, W., Eds.; Leidschendam: VKP, 1993; 89–106.

International Soil Reference and Information Centre (ISRIC)

David Dent

Center for Sustainability Studies, Lund University, Lund, Sweden

Abstract

ISRIC–World Soil Information is an independent foundation, funded by the Netherlands Government with a mandate to increase knowledge of the land, its soils in particular, and to support the sustainable use of land resources; in short, to help people understand soils. Its aims are as follows: 1) to inform and educate, through the World Soil Museum; 2) to maintain and disseminate data for the scientific community through the International Council for Science World Data Centre for Soils which is the custodian of global and regional datasets, land resources maps, and reports; for many of these, ISRIC is the sole repository; 3) to conduct applied research on land resources and their management and to support the development of national and international policy. The institute has a tradition of welcoming guest researchers; and 4) the multilingual staff has expertise in taxonomy of soils, soil survey, land evaluation and land use planning, soil and water conservation, soil fertility, and data management and interpretation—globally and especially in tropical regions.

HISTORY

On the initiative of Professor F.A. van Baren, the International Society of Soil Science promoted the establishment of an international museum of soil standards to collect, analyze, and display the soils of the world, as depicted in the FAO–UNESCO Soil Map of the World. This was adopted by the General Council of UNESCO in 1964, and support was offered by the Government of the Netherlands.

The institute was founded in 1966 with working funds provided by the Ministry of Education, Culture, and Sciences of the Netherlands and administered by the International Institute for Aerospace Survey and Earth Sciences (ITC) in Enschede. At first, facilities were provided by the University of Utrecht; in 1977, the premises—which comprise the staff quarters, laboratory, workshop, teaching and conference facilities, and the world soils exhibition—in Wageningen were provided by the Netherlands Directorate General of International Cooperation.

In line with its evolving mandate and activities, the institute was renamed International Soil Reference and Information Centre (ISRIC) in 1974; it became a foundation with its own statutes and Board of Governors in 1995, registering the logo ISRIC–World Soil Information in 2004. The business arrangement with ITC was dissolved in 2002 and a cooperative agreement concluded with Wageningen University and Research Centre. It has a memorandum of understanding with FAO, including a joint program of work; close working relations are also maintained with UNESCO, United Nations Environment

Programme, and United Nations Convention to Combat Desertification.

WORLD SOIL MUSEUM

The World Soil Museum is the focus of an active educational program. Groups and individuals are welcome to the well documented, thematic exhibition of soil monoliths, representing the major soils of the world, their landscape relationships, and management. Several monoliths are on loan to museums and universities around the world (Figs. 1–4).

A new initiative, the ISRIC World of Soils, makes available through the Internet a wide range of educational resources: programs, pictures, data, and text. A UNESCO Chair in Land Resources is proposed as an international focus for soils education, to include ISRIC courses in taxonomy, soil survey, land evaluation, and land use planning, and database management and contributions from several partner institutions in the Netherlands and Switzerland.

Publications program includes new and updated standard technical works—in 2005 new editions of the FAO Guidelines for soil profile description and the Booker Soil Manual—and ISRIC Reports, which are available both online and in paper copy.

WORLD DATA CENTRE FOR SOILS

The World Data Centre for Soils serves the scientific community under the auspices of the International



Fig. 1 Collecting a soil monolith.

Council for Science (ICSU). Its role is to collect, scrutinize, analyze, and disseminate data and information worldwide, in particular to ICSU programs in global change, climate, and the environment. Data from ICSU programs are maintained and made freely available; a major project is under way to digitize the datasets and maps to make them available through the Internet or on DVD.

The principal holdings are in the following:

1. Monolith collection of more than 900 profiles, supported by some 5000 reference samples, fully analyzed by standard methods, representing the mapping units of the FAO–UNESCO Soil Map of the World^[1,2] and being extended to encompass the groups and lower-level units of the World Reference Base for Soil Resources.^[3]
2. ISRIC soil information system (ISIS) dataset, 800 profile descriptions with complete, validated, laboratory data, available online in SQL.
3. World Inventory of Soil Emission Potentials (WISE) dataset of 4000 profiles in the public domain compiled for global climatic change studies and backed up by a working database of more than 9500 profiles.



Fig. 2 Preparation of monolith in the workshop.



Fig. 3 Teaching in the World Soil Museum.

4. The Global Soil and Terrain (SOTER) datasets for South and Central America, Central and Eastern Europe, and Southern and Eastern Africa: spatial mapping units and geo-located point data at scales from 1.1 to 1.5 million, available online.
5. Soil and water conservation database of the World Overview of Conservation Approaches and Technologies (WOCAT).

For micromorphology, the following soil collections are available:

1. Systematic collection of 3500 large thin sections from the ISIS profiles.
2. STIBOKA-Jongerius collection of 10,000 large thin sections, mainly of Dutch soils.
3. Schmidt–Lorenz collection of more than 15,000 small thin sections of soils, mainly from Europe, Africa, Asia, and Australia.

There are excellent facilities for micromorphological examination and a common catalog is in preparation to make the collections publicly accessible.

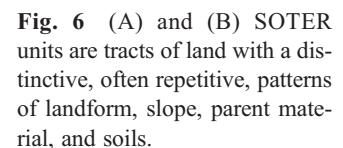


Fig. 4 Display of monoliths.



APPLIED RESEARCH

The institute is well known for underpinning the Soil Map of the World, the Global Assessment of Human-induced Soil Degradation (GLASOD)^[4] produced for the Rio Conference in 1992, the World Reference Base for Soil Resources (since 1980), the SOTER database,^[5] and for interpretations of this information for land use planning and, further, in terms of sources and sinks of greenhouse



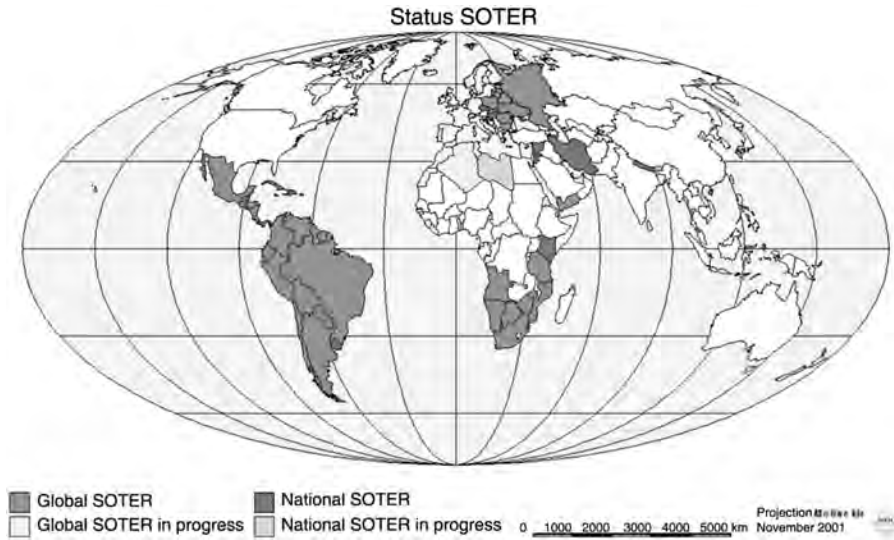


Fig. 7 Global SOTER coverage, December 2004.

gases (Figs. 5–7; Table 1). The institute is also a partner in the WOCAT (<http://www.wocat.net/>).

The trends in research over decades have been shifting away from data collection and toward putting the data to work and from solo soil science to work within multidisciplinary, international teams. An example is the study on the Pan-European Soil Erosion Assessment.^[6] Existing and future priorities include the following: a quantitative Global Assessment of Land Degradation and Improvement (GLADA), using 8 km definition satellite NDVI imagery to identify and delineate hotspots

for further characterization by 30 m definition Landsat imagery and subsequent field measurements by national teams; a policy initiative, Green Water Credits—akin to the Kyoto Protocol carbon credits mechanism—to build a global facility to pay rural land managers for water management services that are unrecognized and unrewarded, relieving poverty and enabling food and water security for everyone; and linking the SOTER and WOCAT databases with social rules to identify best-bet management practices and provide a learning network to support their application.

Table 1 Data held in ISIS and WISE datasets: attributes of the ISIS database.

No. of profiles	No. of countries	Environmental characteristics	Site characteristics	Soil morphology	Physical attributes	Chemical attributes	Mineralogical attributes	Additional information
880	81	Physiography Geology Hydrology Climate Vegetation	Surface conditions Slope Land use Human influence Crops	Horizons Color Texture Structure Consistence Cutans Nodules Coarse fragments Biological and plant activity Porosity Pans Wageningen Permeability	Particle-size distribution Water-dispersible clay Bulk density Water retention	PH Organic C Organic N CaCO ₃ CaSO ₄ Exchangeable cations Cation exchange capacity (CEC) Exchangeable Al, H Extractable Fe, Al, Si, P soluble salts Elemental composition	Clay mineralogy Sand mineralogy Heavy minerals	Thin sections Photographs Soil and soil-related maps Reports Reference soil samples Classification: local, FAO, Soil Taxonomy and World Reference Base

REFERENCES

1. FAO–UNESCO. *Soil Map of the World, 1:5M*; UNESCO: Paris, 1974–1981; Vol. I–X.
2. FAO–UNESCO–ISRIC. *Revised Legend of the Soil Map of the World. World Soil Resources Report No. 60*; FAO: Rome, 1990.
3. FAO–ISRIC–ISSS. *World Reference Base for Soil Resources. World Soil Resources Report No. 84*; FAO: Rome, 1998.
4. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *World Map of the Status of Human-Induced Soil Degradation*; ISRIC–UNEP: Wageningen, 1990.
5. van Engelen, V.W.P.; Wen, T.T. *Global and National Soils and Terrain Digital Databases (SOTER): Procedures Manual, Revised*; ISRIC: Wageningen, 1995.
6. Mantel, S.; van Lynden, G.J.; Huting, J. *Pan European Erosion Assessment (PESERA), Work Package 6: Scenario Analysis*; Final Report: Apr 2003–Sep 2003; ISRIC–World Soil Information: Wageningen, 2003.

International Union of Soil Sciences

Stephen Nortcliff

Department of Soil Science, University of Reading, Reading, U.K.

Abstract

This entry briefly describes the history and development of the International Union of Soil Sciences (IUSS) from its foundation as the International Society of Soil Science (ISSS) in 1924 through its change to the IUSS in 1998 and then up to 2016. Brief details of the changing scientific structure of ISSS and IUSS together with the officers of the society/union are provided.

INTRODUCTION

In this entry, the history of the development of the International Society of Soil Science (ISSS) and its transformation into the International Union of Soil Sciences (IUSS) is briefly described. A brief historical review of soil science in the 19th century and early 20th century provides the context for the formation of the ISSS. Formed in 1924, ISSS arose from a series of international meetings and has probably been most widely known for organizing the World Congress of Soil Science on an approximately 4-year cycle. The initial organizational structure of ISSS was established in 1924 with six commissions and was not substantially altered until its demise in 1998, the major changes being the addition of two further commissions. The officers of ISSS initially consisted of the President (normally from the country which will next host the World Congress) and the Secretary-General. Subsequently, a Vice President and Deputy Secretary-General were appointed. As ISSS, the membership was based on individuals who subscribed separately. During the 1990s, ISSS sought membership of the International Council of Science (ICSU) and was accepted in 1993. The scientific members of ICSU are based on unions who have country members rather than individual members and this was recommended for ISSS. In 1998, ISSS ceased to exist and IUSS came into existence with its membership being based on National Soil Science Societies or similar bodies. The consequence of this change was a substantial increase in membership. The change to IUSS also saw a change in scientific structure with four “divisions” each divided into a number of “commissions.” The membership of ICSU has coincided with the development of stronger ties with other scientific unions.

THE HISTORICAL CONTEXT

In comparison to many of the sciences covered by other members of ICSU, soil science is a relatively young

science, and, for a number of years, many of those who might be considered as practicing soil scientists were not identified as such and were more likely to be found as members of other broad subject areas including chemistry, agriculture, and geology. These links with other branches of science continue and many practicing soil scientists will find themselves located in institutes and university departments which are not specifically described by the term “soil science.” If the subject matter addressed by soil scientist is similarly considered, it can be found that it is equally diverse with soil scientists employing tools and approaches derived from many sciences including physics, chemistry, biology, agriculture, earth sciences, and increasingly mathematical and related sciences.

Despite being a young science, the diversity of the scientific approaches employed within soil science has provided the stimulus for many scientists to focus their attention on this fascinating material. If one looks back to the early roots of the subject in the 19th century, a diversity of approaches can be found. In part, this early diversity might be ascribed to the context within which the soil scientists were working. In much of Western Europe, the focus for soil science was principally in the chemistry laboratory, where attempts were made to increase the agricultural productivity of land, with relatively little increase in the area farmed. There were a number of key players, such as J. B. Lawes, J. H. Gilbert, and J. von Liebig, who were particularly active during this time. In contrast, in Russia and the United States, there was considerably more land available for agricultural development and the principal approach to the study of soils was to address the spatial patterns and extent of soils, in part as guidance to land development. Again there were figures, such as V. V. Dokuchaev in Russia and C. F. Marbut in the United States, who made considerable contributions to our understanding of how soils developed and their patterns in the landscape.^[1] This latter approach has been described as “agrogeological.” In the early years, these two approaches were pursued separately with little or no direct contact

between them. The coming together of these two approaches resulted, among others, in the formation of the ISSS in 1924.

THE ESTABLISHMENT OF THE ISSS—STRUCTURE AND ORGANIZATION

The establishment of ISSS took place at the Fourth International Conference on Pedology which had taken place in Rome in 1924 with over 450 participants. A meeting held in 1922 under the title of the “Third International Conference on Pedology,” set the foundations for the establishment of the ISSS, establishing a broad structure including five commissions. In the next meeting (the Fourth International Conference on Pedology), held in Rome with over 450 participants, the proposal was put forward to establish the ISSS. The aims of the society were to develop more uniform methods of soil analysis, consider a uniform nomenclature for soil classification, produce agrogeological maps of Europe at scales of 1:0.5 million and 1:2.5 million, promote the establishment of soil science in countries without such developments, and promote soil science in the educational system. At this time, six commissions were established:

1. Soil physics.
2. Soil chemistry.
3. Soil biology.
4. Nomenclature and classification of soils.

5. Soil cartography.
6. Plant physiology in relation to pedology (subsequently changed to soil technology).

In 1956, at the Paris meeting, an additional commission (VII—“soil mineralogy”) was established, and at the Montpellier meeting in 1998, another commission (VIII—“soils and the environment”) was introduced.

Following the first meeting held in Rome, subsequent meetings of the society were designated as the World Congress of Soil Science and were to be held at locations around the world.

Also at this meeting, the first office bearers of the society were elected, with J. G. Lipman as the first President (setting the precedent that the President is from the host country of the next meeting of the World Congress of Soil Science) and D. J. Hissink from The Netherlands as the first Secretary-General (and Treasurer).

This organizational structure and the format of officers continued with minor modifications for the next 74 years, throughout the life of the ISSS. A list of the Presidents of ISSS and the location of the 16 World Congresses held by ISSS, which after an initial irregular pattern of meetings established a 4-year cycle (with the exception of the 6 years between the 9th and the 10th congresses) from the 6th congress in 1960 in Paris, is given in Table 1. The activity of the ISSS during the first 50 years is summarized in a special issue of the ISSS Bulletin published in 1974.^[2] The 4-year

Table 1 Presidents of ISSS and IUSS, location and dates of the World Congresses.

President	Country	Period of presidency	Location of congress	Year of congress
J. G. Lipman	United States	1924–1927	Washington, D.C.	1927
K. Gedroiz	USSR	1927–1930	Leningrad	1930
E. J. Russell	United Kingdom	1940–1935	Oxford	1935
C. H. Edelman	The Netherlands	1950	Amsterdam	1950
R. Tavernier	Belgium	1950–1954	Leopoldville	1954
A. Oudin	France	1954–1956	Paris	1956
R. Bradfield	United States	1956–1960	Madison	1960
N. C. Cernescu	Romania	1960–1964	Bucharest	1964
E. G. Hallsworth	Australia	1964–1968	Adelaide	1968
V. A. Kovda	USSR	1968–1974	Moscow	1974
C. F. Bentley	Canada	1974–1978	Alberta	1978
J. S. Kanwar	India	1978–1982	New Delhi	1982
K. H. Hartge	Germany	1982–1986	Hamburg	1986
A. Tanaka	Japan	1986–1990	Kyoto	1990
A. A. Santelises	Mexico	1990–1994	Acapulco	1994
A. Ruellan	France	1994–1998	Montpellier	1998
S. Theerawong	Thailand	1998–2002	Bangkok	2002
D. Sparks	United States	2002–2006	Philadelphia	2006
R. S. Swift	Australia	2006–2010	Brisbane	2010
J. E. Yang	Korea	2010–2014	Jeju	2014

cycle of the World Congresses has continued with the change from ISSS to IUSS, with the 17th Congress being held in Bangkok, Thailand, in 2002; the 18th World Congress held in Philadelphia in 2006; the 19th World Congress held in Brisbane in 2010; and the 20th World Congress in Jeju, Korea, in 2014. During this time, the number of papers presented at the congresses has ranged from less than 200 in Oxford in 1935 to well over 2,000 in Bangkok in 2002, although there has been a shift from purely oral presentations to oral and poster presentations.

During the Council Meeting in Brisbane in 2010, a major change in the organization of the union was accepted. This was a move to decouple the presidency of the IUSS from the host of the World Congress of Soil Science, with a President elected from the whole membership of the union for a period of 2 years. Following the first election in 2012, R. Horn (Germany) was elected to serve as President from 2014 to December 2016. R. Lal (United States) was elected in 2014 to serve as Vice President from January 2015 to December 2016 and as President from January 2017 to December 2019. The Vice President (congress) represents the country hosting the World Congress of Soil Science on the Executive Committee and Council.

At the establishment of ISSS, it was recognized that to provide a degree of continuity to the society it was necessary to establish a bureau or secretariat which did not change with the presidency. Initially this comprised a Secretary-General/Treasurer, but subsequently the roles of secretary-general and treasurer were separated and the position of deputy Secretary-General was established. From 1924 to 2016, there have been just six holders of the position of Secretary-General (Table 2).

In 2014, the union changed its organizational structure with a move to a permanent secretariat based in Austria. This secretariat was partially funded by BIOS Science Austria which is a legal entity and a non-profit association comprising the main players in the life sciences in Vienna: The University of Natural Resources and Life Sciences, Vienna; the University of Veterinary Medicine; the Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management; the

Austrian Agency for Health and Food Safety; the Federal Research Center for Forests; the Umweltbundesamt (Environment Agency Austria); the Federal Office for Water Management; the Austrian Institute of Technology; and the Ecosocial Forum Europe. Dr. Sigbert Huber is the Secretary of IUSS in this new Structure and Dr. Moika Tulipan the Assistant Secretary.

Prior to 1998, membership of ISSS was on the basis of individual subscriptions, but with the shift to the structure of an International Union, the membership comprises national bodies, which depending upon the organizational structure in the individual countries may be the National Academies of Science or National Soil Science Societies. At the time of the Montpellier World Congress, the ISSS membership stood at a little over 7,000; under the new organizational structure where membership is through national bodies, the union represents well over 40,000 soil scientists globally.

IUSS—THE NEW SCIENTIFIC STRUCTURE

The change from ISSS to IUSS took place in 1998, and with this change, a new scientific structure was developed, with divisions as the highest level of organization. Commissions were identified as the organizational units within divisions, and “working groups” were established to address subject matter across divisional boundaries, or where topics were likely to be active over relatively short time periods.^[3] The four divisions are as follow:

1. Division 1—Soils in space and time.
2. Division 2—Soil properties and processes.
3. Division 3—Soil use and management.
4. Division 4—The role of soil in sustaining society and the environment.

Following additions in 2004 and 2010, the commissions within the divisions cover the following areas:

1. Division 1
 - C1.1 Soil morphology and micromorphology.
 - C1.2 Soil geography.
 - C1.3 Soil genesis.
 - C1.4 Soil classification.
 - C1.5 Pedometrics.
 - C1.6 Palaeopedology.
2. Division 2
 - C2.1 Soil physics.
 - C2.2 Soil chemistry.
 - C2.3 Soil biology.
 - C2.4 Soil mineralogy.
 - C2.5 Soil interfacial reactions.
3. Division 3
 - C3.1 Soil evaluation and land use planning.
 - C3.2 Soil and water conservation.
 - C3.3 Soil fertility and plant nutrition.

Table 2 Secretaries-General of ISSS/IUSS.

Secretary-General	Country	Period of office
D. J. Hissink	The Netherlands	1924–1950
F. A. van Baren	The Netherlands	1950–1974
R. Dudal	Belgium	1974–1978
W. G. Sombroek	The Netherlands	1978–1990
W. E. H. Blum	Austria	1990–2002
S. Nortcliff	United Kingdom	2002–2010
A. E. Hartemink	The Netherlands/United States	2010–2014

- C3.4 Soil engineering and technology.
- C3.5 Soil degradation control, remediation, and reclamation.
- C3.6 Salt affected soils.

4. Division 4

- C4.1 Soil and the environment.
- C4.2 Soil, food security, and human health.
- C4.3 Soil and land use change.
- C4.4 Soil education and public awareness.
- C4.5 History, philosophy, and sociology of soil science.

This structure and the 22 commissions serve to illustrate the changes and expansion of the subject matter of soil science from the original six commissions established in 1924. The subject matter of soil science has grown increasingly multidisciplinary, and many soil scientists have strong links with scientists in other scientific unions within the broad ICSU family including International Union of Geological Sciences (IUGS), International Union of Geodesy and Geophysics (IUGG) and International Geographical Union (IGU), addressing the broad aspects of geological, geophysical, and geographical sciences; International Union of Biological Sciences (IUBS) addresses biological sciences; International Union of Pure and Applied Chemistry (IUPAC) deals with pure and applied chemistry; International Union of Pure and Applied Physics (IUPAP) deals with pure and applied physics; International Union of Biochemistry and Molecular Biology (IUBMB) addresses biology and molecular biology; and International Union of Food Science and Technology (IUFOT) addresses many aspects of food science and food technology. Following the decision to become a scientific union, IUSS has been an active member of ICSU the International Council for Science.

CONCLUSIONS

IUSS is able to trace its heritage back to the early soil scientists who pioneered the analysis of soil in the laboratory and the understanding of the development and distribution of soils in the field. The move from the organizational structure of ISSS with its individual membership to that of IUSS as a member of ICSU, with membership being provided through national bodies, reflects the increasing multidisciplinary and interdisciplinary nature of the tasks facing soil scientists and the tools required to address these tasks. This new structure also enables the possibility of IUSS to provide support through national bodies for soil scientists who may have difficulties in fulfilling their scientific potential and actively participating in global soil science activities, for example, through the work of the divisions which have a small annual budget to support individual soil scientists or national bodies in pursuing activities related to the activities of the division. The change to an elected President with a supporting secretariat marks another shift in emphasis with the President taking on a broader role promoting Soil Science and Soil Scientists in Regional and International contexts.

REFERENCES

1. Yaalon, D.H.; Berkowicz, S. *History of Soil Science—International Perspectives*; Catena Verlag: Reiskirchen, 1997; 438 pp.
2. ISSS. The history of the international soil science society. *Bulletin* **1974**, *45*, 18.
3. van Baren, H.; Hartemink, A.E.; Tinker, P.B. 75 years, the international society of soil science. *Geoderma* **2000**, *96*, 1–18.

Ion Exchange

Bryon W. Bache

Cambridge University, Cambridge, U.K.

Abstract

Many substances in the soil solution exist as ions, but in a typical agricultural or garden soil most of the anions (the negatively charged particles) exist as large and insoluble macroions. Thus, the solid matrix of soil has a negative charge, and to maintain electrical neutrality, positively charged cations are adsorbed onto the surface of the insoluble anions in a diffuse-ion swarm. This scenario facilitates *cation exchange*, whereby some of the adsorbed cations are exchanged for others.

INTRODUCTION

Nature and Origin of Cation Exchange

Many common substances exist as ions. The simplest example is common salt, sodium chloride. This innocuous and life-sustaining compound is made of two dangerous and highly reactive elements, sodium and chlorine. When these two elements react together, the sodium atom loses an electron, giving it a positive charge (Na^+), while the chlorine atom gains an electron, giving it a negative charge (Cl^-). The positively charged ions are *cations*; the negative ones are *anions*. In a salt solution, the sodium and chloride ions are not physically bonded together but are free to diffuse independently in solution, attracted to each other only by electrostatic charges. Hence, the original derivation of the term *ion* (from the Greek for “wanderer”) is applied generally to any charged particle.

CATION EXCHANGE

Cation exchange is best illustrated by conducting a simple experiment. Place a layer of dry soil about 5 mm deep into a funnel, supported by a filter paper. Leach the soil slowly with demineralized water to remove the bulk of any soluble salts present and discard this leachate. Then leach with 50 ml of water, but this time collect the leachate. (Although this will take some time, mixing the soil with coarse sand will speed up the process.) Next, leach the soil with 50 ml of a solution containing 2–3 g of ammonium chloride and again collect the leachate. Test both the water leachate and the ammonium chloride leachate for calcium by adding ammonium oxalate solution. The water leachate will be free of calcium but the ammonium chloride leachate will give a white precipitate of calcium oxalate, showing that ammonium ion (NH_4^+) has displaced calcium ion (Ca^{2+}) from the soil

surfaces into solution. Ammonium has *exchanged* for calcium. A diagrammatic representation of this exchange reaction is shown in Fig. 1.

Humus and clay are the two components of the soil matrix that contain the macroions responsible for cation exchange. Humus contains proton-donating functional groups on the surface of its molecules, which are undissociated at low pH but dissociate as the pH rises, to give a negatively charged surface. The most important of these is the carboxy group:



where R is the organic core. Clay consists of fine platy crystals of aluminosilicate minerals. The lattice of these minerals has a positive-charge deficiency caused by the isomorphous substitution of aluminum ion (Al^{3+}) for Si^{4+} or magnesium ion (Mg^{2+}) for Al^{3+} in the structure of the mineral, which gives the crystal a net negative charge.

EXCHANGEABLE CATIONS, CATION EXCHANGE CAPACITY, AND BASE CATION SATURATION

In most agricultural soils, the dominant exchangeable cation is Ca^{2+} , with lesser amounts of Mg^{2+} , sodium ion (Na^+), and potassium ion (K^+). These four are often termed *base cations*—i.e., the cations of the bases $\text{Ca}(\text{OH})_2$, $\text{Mg}(\text{OH})_2$, NaOH , and KOH . These should not to be confused with *basic cations*, of which MgOH^+ is an example. In contrast, the acid cation Al^{3+} is dominant in acid mineral soils and the hydrogen ion (H_3O^+) is dominant in acid organic soils.

The total amount of cations that may be exchanged is the *cation exchange capacity* (CEC) of the soil. The sum of the positive charge on these cations equals the negative charge on the soil surface. The convention for CEC units most consistent with the International System

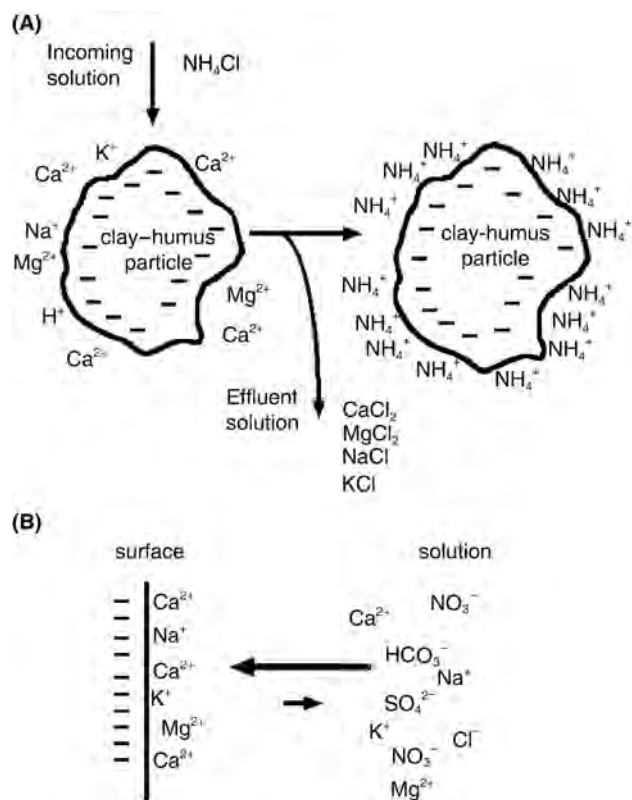


Fig. 1 Diagrammatic representations of cation exchange: (A) the experiment described in the text and (B) the equilibrium between the surface and the solution.

of Units is “millimoles of charge per kilogram of dry soil,” i.e., $\text{mmol}(+)\text{kg}^{-1}$ for the cations and $\text{mmol}(-)\text{kg}^{-1}$ for the surface charge. A variety of experimental procedures has been developed to measure exchangeable cations and CEC.^[1] CEC values in mmolkg^{-1} may vary for clay minerals from 30 for kaolinite to 1200 for smectite and for soils from 50 for Oxisols to 2000 for Histosols.

The negative charge on humus is strongly pH dependent, so that apart from the nature and amounts of clay and humus, the most important variable affecting CEC is the pH of the soil (Fig. 2). Thus, when comparing different soils, CEC is conventionally measured at a standard pH, normally at the neutral point of pH 7.0 with an ammonium acetate buffer and NH_4^+ as the displacing cation, or at pH 8.2 with a triethanolamine buffer and barium ion (Ba^{2+}) as the displacing cation.

Conventions for measuring CEC at a standard pH are appropriate for near-neutral soils dominated by permanent-charge clays. However, it is clear from Fig. 2 that they result in a highly inflated CEC value for acid soils when compared with the natural field soil, and they are even less appropriate for variable-charge soils, which acquire pH-dependent positive and negative charges (see Fig. 2 and section Anion Exchange). In these circumstances, CEC measured with unbuffered salt solutions at field pH provides a better interpretative value. This *effective* CEC

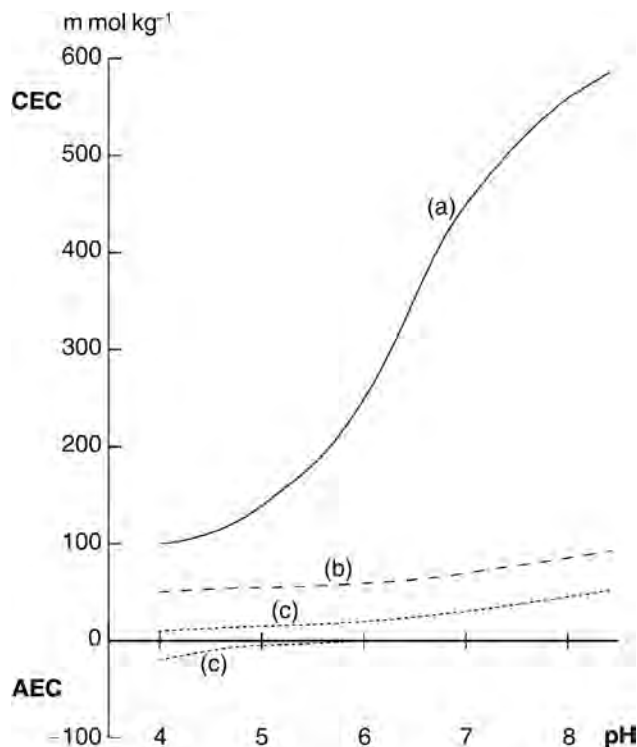


Fig. 2 The effect of pH on CEC: (a) a humic topsoil; (b) a mineral subsoil; (c) a strongly weathered Oxisol, showing both CEC and AEC.

(ECEC) equals the sum of the exchangeable cations extracted from the soil, including both base and acid cations. Thus, the pH of measurement should always be stated for CEC data.

An important concept for acid soils is the *base cation saturation* (V). This is usually expressed as a percentage and is then given by

$$V = 100\Sigma\text{M}^+/\text{CEC} \quad (2)$$

where ΣM^+ is the sum of the exchangeable base cations, ΣM^+ and CEC are measured in the same units, and CEC is at pH 7.0.

CATION EXCHANGE EQUILIBRIA AND CATION SELECTIVITY

Cation exchange was first discovered by H.S. Thompson in 1850; a review of early related work was compiled by Kelley.^[2] Good accounts are given by Mott,^[3] Spósito,^[4] and McBride.^[5] Apart from the source of cation exchange (discussed earlier), most interest has focused on the relationship between the relative concentrations of different cations in solution and the amounts of surface-adsorbed cations, and how this varies for different exchange materials (e.g., silicate clay minerals, humus, and synthetic resins).

Fig. 1B shows the equilibrium between the adsorbed cations and those in the surrounding solution, all of which are free to diffuse within the constraints of electrical neutrality. This equilibrium is established rapidly, allowing the exchange reactions to occur when the solution concentrations are altered, as in the experiment described above, or when fertilizers, irrigation water, or environmental pollutants are added to soils. Theoretical approaches to quantify this equilibrium are important to predict cation behavior in soil processes. While rigorous thermodynamic treatments are complicated,^[6] simplified empirical treatments are often more satisfactory. This author has found the most useful to be the *corrected rational selectivity coefficient*,^[7] which incorporates the activities of the solution ions and the equivalent fraction of the adsorbed ions. *Activity* has been described as effective concentration and is usually a little lower than the actual concentration measured in mole/dm³. *Equivalent fraction* is the fraction of the total negative charge occupied by the adsorbed cation. In a simple homovalent exchange, such as potassium displacing sodium,

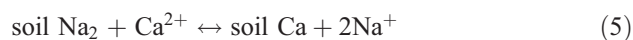


the selectivity coefficient **K** is given by as follows:

$$\mathbf{K} = \frac{X_{\text{K}} (\text{Na}^+)}{X_{\text{Na}} (\text{K}^+)} \quad (4)$$

where X is the equivalent fraction of ion on the exchanger and the parentheses indicate the activity of that ion in solution. This selectivity coefficient shows the preference of the exchanger for one ion over another. In this example, if **K** is less than 1, sodium would be preferred by the exchanger, but in practice **K** is greater than 1 and potassium is preferred.

For a heterovalent exchange reaction, such as calcium displacing sodium,



the selectivity coefficient is given by as follows:

$$\mathbf{K} = \frac{X_{\text{Ca}} (\text{Na}^+)^2}{[X_{\text{Na}}]^2 (\text{Ca}^{2+})} \quad (6)$$

using the same conventions as previously.

In general, cations with higher charge are more strongly adsorbed by exchangers (e.g., $\text{Al}^{3+} > \text{Ca}^{2+} > \text{Na}^+$) and for cations with similar charge, those with higher radius and lower hydration energy are preferred (e.g., $\text{Ba}^{2+} > \text{Ca}^{2+} > \text{Mg}^{2+}$). Structural considerations may also be involved. For example, the size of the potassium ion enables it to fit neatly into the holes on the surface of silicate clays, thereby increasing the preference of the surface for potassium.

PRACTICAL IMPORTANCE OF CATION EXCHANGE

This section outlines three of the many practical applications of cation exchange.

Plant Nutrition

Many plant nutrients exist as cations: the four base cations mentioned above, NH_4^+ , and the heavy metal cations such as iron ion (Fe^{2+}), copper ion (Cu^{2+}), zinc ion (Zn^{2+}), and manganese ion (Mn^{2+}).^[8] Plants take up their nutrients as simple ions from solution, but when the solution is depleted, *exchange desorption* allows the adsorbed cations to provide a reserve from which the solution may be replenished (Fig. 1). To maintain electrical neutrality, the desorption of one ion must be accompanied by adsorption of another, usually either Ca^{2+} (the most abundant cation) or the H_3O^+ , which plants excrete when they take up cation nutrients.

Soil Development and Acidification

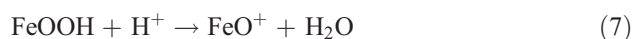
In a humid climate, the course of soil development in a freely drained environment is always toward acidification.^[5,9] The H_3O^+ is strongly adsorbed and it exchanges for the base cations on the soil surfaces, which are then leached out, so that the soil gradually becomes more acid. The main sources of hydrogen ions are the dissolution of carbon dioxide, the release of organic acids when plant residues decompose, the deposition of atmospheric pollution, and the addition of fertilizer.

Amelioration of Saline and Alkali Soils

An excess of sodium salts and/or exchangeable sodium in these soils causes problems.^[5] Amelioration is effected by adding gypsum (CaSO_4) prior to leaching the soil with excess water. Calcium dissolves from the CaSO_4 and exchanges for sodium (as in the example of heterovalent exchange 5 above), which is then leached in the drainage water.

ANION EXCHANGE

Anion exchange^[3–5] can occur on positively charged surfaces in a manner similar to cation exchange on negative surfaces (e.g., as with synthetic resins based on quaternary ammonium groups).^[7] Although there are no soil minerals that possess permanent positive charges and humus does not acquire a positive charge, strongly weathered acid soils containing amorphous hydrated iron and aluminum oxides can acquire a pH-dependent positive charge:



This allows a small *anion exchange capacity* (AEC) to develop at a pH below 5, as shown in Fig. 2C, in addition to cation exchange on the negative surfaces of these variable-charge soils.^[3,10]

The main practical importance of anion exchange is that it allows the anions, namely Cl^- and nitrate ion (NO_3^-), to be resistant to leaching from the soil. This is particularly important for nitrate, which provides a reserve for plant nutrition. Phosphate has often been associated with anion exchange in the popular soil literature, but because its bonding to both neutral and positively charged sites is by hydroxyl ligand exchange rather than by diffuse-ion swarming, it is not genuinely “exchangeable” within the meaning of this entry.

REFERENCES

1. Page, A.L.; Miller, R.H.; Keeney, D.R. *Methods of Soil Analysis, Part 2—Chemical and Microbiological Properties*, 2nd Ed.; Agronomy Series No. 9 (Part 2); American Society of Agronomy, Social Science of America: Madison, 1982.
2. Kelley, W.P. *Cation Exchange in Soils*; American Chemical Society, Monograph No. 109; Reinhold: New York, 1948.
3. Mott, C.J.B. Surface chemistry of soil particles. In *Russell's Soil Conditions and Plant Growth*; Wild, A., Ed.; 11th Ed.; Longman: Harlow, 1988; 239–281.
4. Sposito, G. *The Chemistry of Soils*; Oxford University Press: New York, 1989.
5. McBride, M.B. *Environmental Chemistry of Soils*; Oxford University Press: New York, 1994.
6. Sposito, G. *The Thermodynamics of Soil Solutions*; Clarendon Press: Oxford, 1981.
7. Helfferich, F. *Ion Exchange*; McGraw Hill: New York, 1962.
8. Mengel, K.; Kirkby, E.A. *Principles of Plant Nutrition*, 3rd Ed.; International Potash Institute: Bern, 1982.
9. Bache, B.W. Soil acidification and aluminium mobility. *Soil Use Manag.* **1985**, *1*, 10–14.
10. Gillman, G.P.; Sumpter, E.A. Surface charge characteristics and lime requirement of soils derived from basaltic, granitic, and metamorphic rocks in high-rainfall tropical Queensland. *Aust. J. Soil. Res.* **1986**, *24*, 173–192.

Iron Oxides

Nikolla P. Qafoku

James Amonette

Pacific Northwest National Laboratory, Richland, Washington, U.S.A.

Abstract

Iron (Fe) oxides are common clay-sized oxide, oxyhydroxide, and hydroxide soil minerals. They are compounds of Fe, oxygen (O), and hydrogen (H) that have structures based on close-packed arrays of O. The octahedral and tetrahedral cavities within these arrays are filled with either Fe^{3+} or Fe^{2+} to form $\text{Fe}(\text{O}/\text{OH})_6$, FeO_6 , or FeO_4 structural units. All of the naturally occurring Fe oxide minerals usually undergo some degree of isomorphous substitution of other metal ions for Fe in their structures. Relatively simple techniques may be used to identify Fe oxides in the field based on their typical colors and magnetic properties. In the laboratory, a variety of instrumental techniques can be used to confirm phase identity and to quantify amount. Of these, X-ray diffraction, infrared spectroscopy, electron microscopy, thermal analysis, and Mössbauer spectroscopy are the most commonly used techniques. As oxides, the functional groups on their surfaces may have positive, negative, or no charge depending on pH and on the concentration and nature of other ions in the contact solution. A net positive surface charge usually is observed in soils because Fe oxides have a point of zero charge in the neutral or slightly basic pHs. The functional groups on the surface form complexes with cations and anions from the aqueous phase. Their sorption and electron-buffering properties significantly affect the geochemical cycles of almost all elements having agronomic or environmental significance.

INTRODUCTION

The iron (Fe) oxides are strongly pigmented clay-sized oxide, oxyhydroxide, and hydroxide minerals that occur in almost all soils and sediments. They are particularly abundant in highly weathered soils, where, in combination with aluminum (Al) and titanium (Ti) oxides, they may account for the great majority of the soil mass^[1] and the unique physical and chemical properties of these soils (Fig. 1). The sorptive properties of Fe oxides are largely determined by the interactions with the components of the soil solution such as protons and other dissolved ions. They also buffer the rich electron-transfer chemistry of Fe^{3+} and Fe^{2+} species. As a consequence of this sorptive and electrochemical reactivity, Fe oxides play a major role in the geochemical cycles of most elements having agronomic or environmental significance.

CLASSIFICATION AND OCCURRENCE

The Fe oxides are compounds of Fe, oxygen (O), and hydrogen (H) that have structures based on close-packed arrays of O. The octahedral and tetrahedral cavities within these arrays are filled with either Fe^{3+} or Fe^{2+} to form $\text{Fe}(\text{O}/\text{OH})_6$, FeO_6 , or FeO_4 structural units. The oxides are distinguished by the ways in which these structural units are arranged in space through corner, edge, or face

sharing. Of the 15 Fe oxides recognized, only 8 are commonly found in soils and sediments. Their selected properties are presented in Table 1.^[1–4]

Isomorphous Substitution

All of the naturally occurring Fe oxide minerals will undergo some degree of substitution of other metal ions for Fe in their structures. The isomorphous substitution of Al^{3+} for Fe^{3+} is most frequent and especially common in goethite. However, other trivalent cations with similar ionic radii to Fe^{3+} (e.g., Ti^{3+} , Mn^{3+} , Co^{3+} , Cr^{3+} , and V^{3+}) may replace Fe^{3+} in the Fe oxide structures. Cations with a higher (e.g., Ti^{4+}) or lower (e.g., Cu^{2+} and Zn^{2+}) valence may also substitute for Fe^{3+} but the extent of this substitution is not greater than 0.1 mol mol^{-1} .^[5] Substitution of Mg^{2+} for Fe^{2+} also occurs in green rusts and magnetites.

METHODS OF IDENTIFICATION

Bulk Analyses

The presence and total quantity of Fe in a soil (i.e., Fe_T) can be determined by a variety of spectroscopic techniques following hydrochloric acid or hydrofluoric acid digestion or non-destructively by energy-dispersive X-ray fluorescence. In some instances, determination of Fe valence is

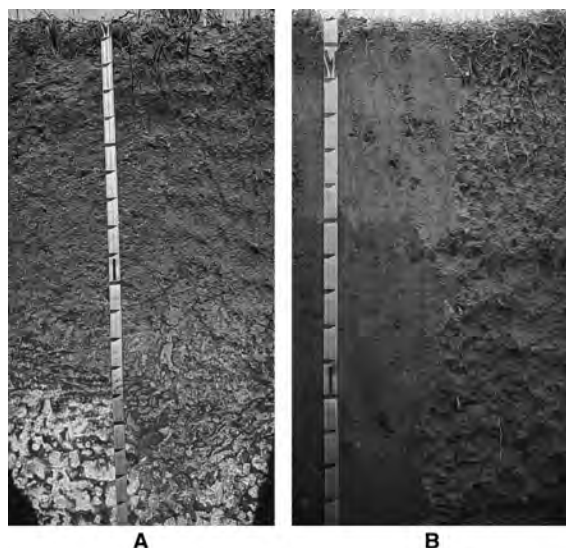


Fig. 1 Fe-rich highly weathered soils from the Southeastern United States. (A) Tifton series (fine-loamy, siliceous, thermic Plinthic Kandiodults). (B) Faceville series (clayey, kaolinitic, thermic Typic Kandiodults).

important, and this can be performed by wet chemical techniques using colorimetry (1,10-phenanthroline) or oxidimetry (vanadate, argentous ion), or non-destructively using Mössbauer spectroscopy.^[6,7]

Relatively simple techniques may be used to identify Fe oxides in the field based on their typical colors and magnetic properties. In the laboratory, a variety of instrumental techniques can be used to confirm phase identity and to quantify amount. Of these, X-ray diffraction, infrared spectroscopy, electron microscopy, thermal analysis, and Mössbauer spectroscopy are the most commonly used techniques.

Selective Extractions

Operational measurements of Fe oxide content in soils are obtained using two selective extraction techniques that discriminate between crystalline and poorly ordered oxides on the basis of solubility under different conditions. The dithionite–citrate–bicarbonate (DCB) method nominally dissolves all pedogenic Fe oxides (crystalline and poorly ordered). This Fe fraction is termed Fe_{DCB}. The ammonium oxalate extraction (conducted in the dark) essentially

Table 1 Selected properties of iron oxides commonly found in soils and sediments.

Minerals		Chemical formula	Occurrence/formation	Color and morphology	Other properties
Oxides	Hematite	$\alpha\text{-Fe}_2\text{O}_3$	Humid tropical, subtropical, and Mediterranean soils (low organic matter and warm temperatures)	Blood-red, fine-grained, hexagonal plates	High stability and crystallinity
	Magnetite	Fe_3O_4	Inherited from parent materials but also produced by bacterial activity	Black, coarsely grained cubes	Contains both Fe^{2+} and Fe^{3+} , ferrimagnetic
	Maghemite	$\gamma\text{-Fe}_2\text{O}_3$	Highly weathered soils, especially Oxisols; formed by oxidation of magnetite or heating of goethite in the presence of organic matter	Brown, coarsely grained cubes	Has magnetite structure, ferromagnetic, minor Fe^{2+}
Oxyhydroxides	Goethite	$\alpha\text{-FeOOH}$	Most common Fe oxide in soils, especially under cool and wet conditions, rich in organic matter	Yellow, fine-grained stubby crystals with little acicularity	Highly stable, Al substitution common
	Lepidocrocite	$\gamma\text{-FeOOH}$	Seasonally anaerobic, non-calcareous soils in cool areas	Distinctive orange-colored laths	Similar to goethite, but little or no Al
	Schwertmannite	$\text{Fe}_8\text{O}_8(\text{OH})_6\text{SO}_4$	Acid-sulfate soils, and waters draining mine spoils and pyritic rocks	Dark yellow-orange clusters of fibrous crystals	Sulfate anion occupies tunnel structure
Hydroxides	Green rust	Variable $(\text{Fe}^{2+}_6\text{Fe}^{3+}_2)(\text{OH})_{16}\text{CO}_3 \cdot 2\text{H}_2\text{O}$	Transitory phase formed on rapid mixing of Fe^{2+} and Fe^{3+} salts under alkaline to neutral, slightly oxic conditions; may be produced by bacterial activity.	Distinctive blue-green, very fine-grained hexagonal plates	Contains Fe^{2+} and Fe^{3+} , reacts readily with oxidants; may contain Mg and anions besides CO_3^{2-}
	Ferrihydrite	$\sim\text{Fe}_5\text{HO}_8 \cdot 4\text{H}_2\text{O}$	Organic matter-rich soils subject to a rapid change from anoxic to oxic conditions; imprecise chemical formula	Brownish-orange spheres, 2–3 nm in diameter	Very poorly ordered, high surface area

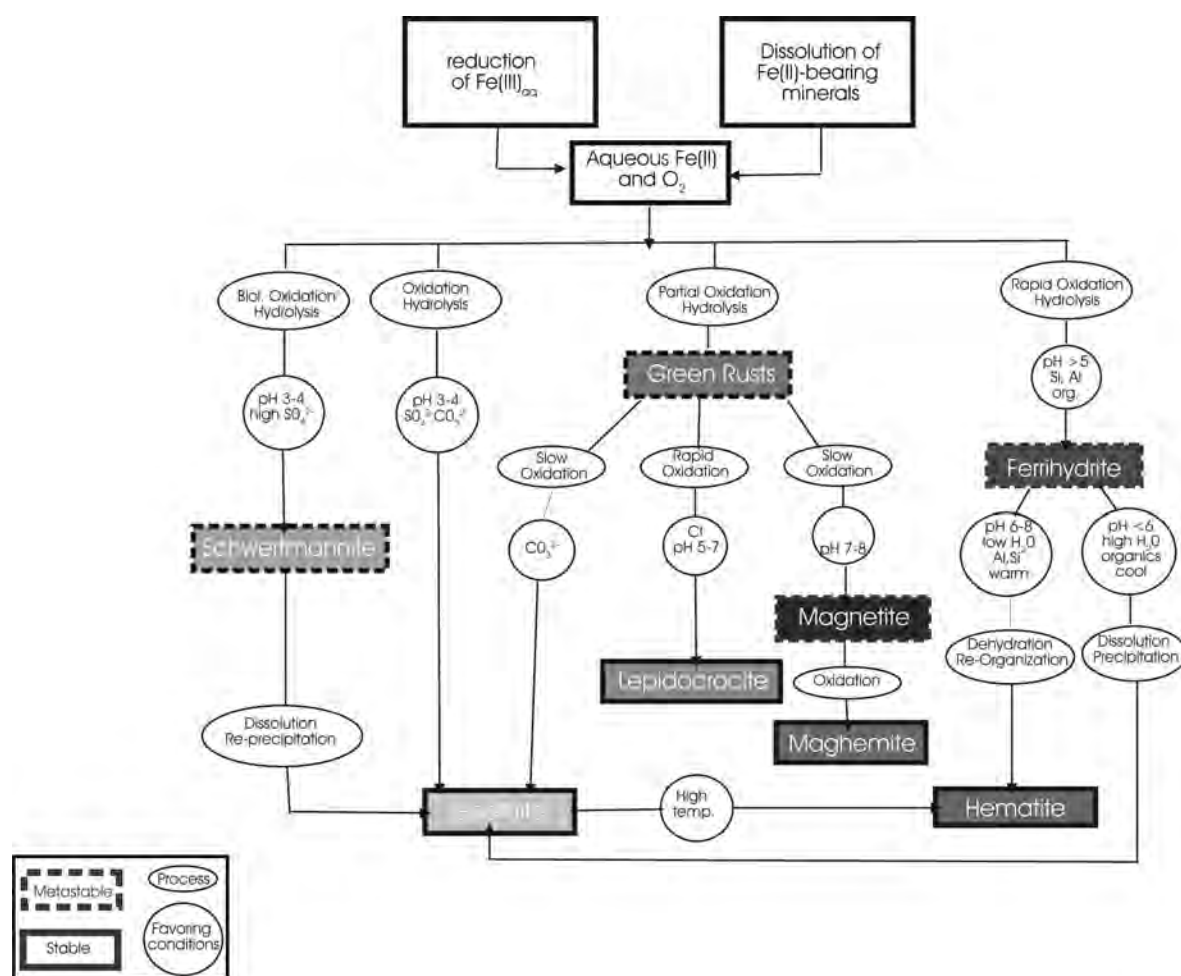


Fig. 2 Fe oxides formation and transformation in soils.

Source: From Bigham, Fitzpatrick, et al.^[2]

dissolves only the Fe fraction that is present in the poorly ordered Fe oxides (Fe_{OX}). The difference between Fe_{DCB} and Fe_{OX} is an estimate of the crystalline Fe oxide fraction (Fe_{CRYS}) present in soils. Highly weathered soils typically have high Fe_{CRYS} values.

The Fe in silicate minerals (Fe_{SIL}), which is not dissolved appreciably by the DCB method, can be calculated as the difference of Fe_{T} and Fe_{DCB} . This Fe source may be mobilized during weathering and the ratio of Fe_{SIL} to Fe_{T} is a useful parameter in estimating the stage of soil weathering. Highly weathered soils (e.g., Oxisols and Ultisols) have small $\text{Fe}_{\text{SIL}}/\text{Fe}_{\text{T}}$ values.

FORMATION AND CHEMISTRY

Formation

The process of Fe oxide formation in soils (Fig. 2) starts with the oxidation of structural Fe^{2+} or aqueous Fe^{2+} that is released during the weathering of Fe^{2+} -bearing

primary minerals. In addition, Fe^{3+} may be released upon dissolution of Fe^{3+} oxides and Fe^{3+} -bearing minerals. Ferromagnesian silicates (e.g., olivines, pyroxenes, amphiboles, and micas) and Fe-Ti oxides are the primary Fe-bearing minerals and serve as a weatherable source of Fe in soils. Usually, minerals that contain Fe^{2+} are less stable because structural Fe^{2+} may oxidize to Fe^{3+} , creating a charge imbalance that weakens the structure of the mineral and promotes dissolution.^[2] Depending on the rate and extent of oxidation, the pH, and the presence of various anions in the solution, ferrihydrite, goethite, schwertmannite, or green rust will directly precipitate from the solution. Rapid oxidation, which results in a higher Fe^{3+} activity in the soil solution, promotes the formation of ferrihydrite. Goethite forms when the soil is rich in organic matter, inorganic colloids, or biological exudates that may sorb or complex Fe^{3+} , thus decreasing the activity of the free aqueous ions. Schwertmannite forms at low pH and high SO_4^{2-} typical of acid mine drainage. Green rust forms when oxidation is very slow and incomplete and when pH is neutral or higher. Continued slow oxidation of green rust

under high CO_3^{2-} conditions and dissolution of ferrihydrite and schwertmannite eventually lead to the formation of goethite. Rapid oxidation of green rust under slightly acidic, low- CO_3^{2-} conditions forms lepidocrocite. Slow oxidation of green rust under slightly alkaline, low- CO_3^{2-} conditions forms magnetite. Hematite forms by a solid-state transformation of ferrihydrite (predominantly) or magnetite. Maghemite forms by the oxidation of lithogenic magnetite or by the transformation of other Fe oxides by heating above 300°C in the presence of organic compounds. Hematite, goethite, maghemite, and lepidocrocite are the most stable Fe oxides, although only hematite and goethite are thermodynamically stable under ambient conditions.^[8] The abundance of goethite in soils stems from the many pathways by which it can form, coupled with its high thermodynamic stability.^[2]

Chemistry

Fe oxides are one of the end products of the weathering process in soils. They form clay-sized domains as small as a few nanometers across (e.g., ferrihydrite) and have specific surfaces ranging as high as several hundreds of $\text{m}^2 \text{g}^{-1}$. Even crystalline Fe oxides (e.g., goethite) may have specific surfaces of several tens of $\text{m}^2 \text{g}^{-1}$. Aggregation and cementation of clay-sized domains lead to larger particles, and even massive deposits, in some soils. As oxides, the functional groups on their surfaces may have positive, negative, or no charge depending on the pH and on the concentration and nature of other ions in the contact solution. A net positive surface charge is usually observed in soils because Fe oxides have a point of zero charge in the neutral or slightly basic pHs. The functional groups on the surface form complexes with cations and anions from the aqueous phase. Oxyanions (e.g., phosphate) are particularly strongly bound as they form innersphere complexes in which an anion oxygen becomes part of the oxide surface structure. The positive charge on the oxide surfaces may also be electrostatically attracted to the permanent negative charge on layered aluminosilicate clay minerals. Formation of electrostatic bonds between oxide and phyllosilicate minerals promotes the formation of soil aggregates and stabilizes them, especially in highly weathered variable-charge soils (Fig. 3).

Fe oxides also buffer the oxidation–reduction activity of soils. The standard reduction potential for the aqueous $\text{Fe}^{3+}/\text{Fe}^{2+}$ couple (+ 0.77 V) is situated in the middle of the limits for aqueous chemistry (i.e., –1.1 to +1.8 V), and Fe^{3+} in the poorly ordered oxides serves as a terminal electron acceptor for microorganisms once oxygen and nitrate are consumed. Under these anoxic conditions, large quantities of soluble Fe^{2+} may be generated and become mobile. The crystalline Fe oxides are less susceptible to reductive dissolution and can adsorb and react with some of these mobile Fe^{2+} cations on their surfaces,^[9] where they become much more effective reductants for environmental

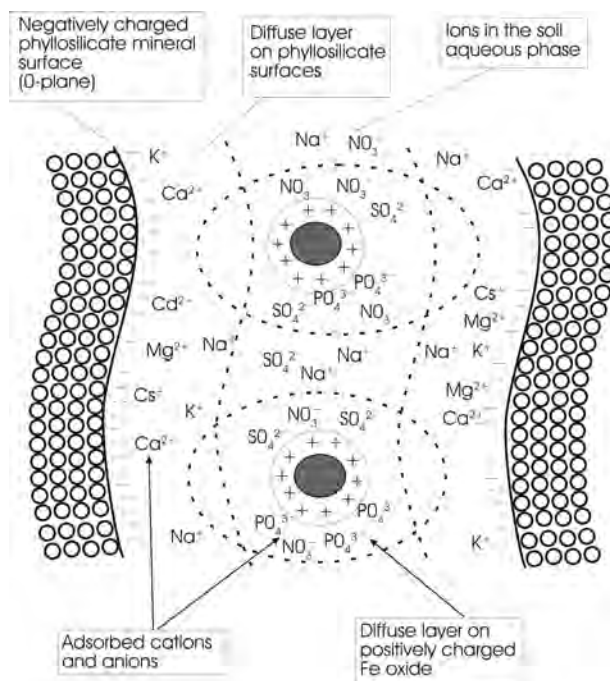


Fig. 3 Interactions among positively charged Fe oxides and negatively charged phyllosilicates in Fe-rich soils.

contaminants.^[10–12] The concentrations of aqueous Fe^{2+} in soils may be as high as $10^{-4} \text{ mol L}^{-1}$, whereas that of Fe^{3+} is typically much lower (about $10^{-12} \text{ mol L}^{-1}$) because of the low solubility of Fe^{3+} oxides at neutral and alkaline pH. Fe oxides surface chemistry and interactions with soil organic matter, plants and microbes are discussed in the following publications.^[13–16]

NANO Fe MINERALS

Soil scientists have always paid special attention to the smallest fraction in soils, or the so-called “fine-grained,” “submicron,” “ultrafine,” or “nano” fraction, because of its large surface area, percentage of surface atoms with unbalanced charge, and number of surface functional groups per unit of mass.^[17] Fe oxyhydroxides are an integral part of the soil nano fraction. Ferrihydrite, for example, exists only as a nanoparticle and larger equivalents of this soil mineral do not even exist.^[18] Despite the ubiquity of ferrihydrite in natural sediments and the fact that ferrihydrite has been subject to intensive research, the nanocrystallinity of this Fe oxyhydroxide has hampered accurate structure determination by traditional methods that rely on long-range order, and its exact structure and composition is still a matter of debate.^[19] Nano Fe oxyhydroxides in soils manifest unique size-dependent properties including point of zero charge,^[20] adsorption edge,^[21] kinetics of reductive dissolution,^[22] solubility,^[4] and electron-transfer reactivity.^[23]

FURTHER READING

For more detailed treatment of the classification, structure and chemistry of the iron oxides in soils, several excellent reviews,^[1–4,9] and two books^[5,6] are recommended. For a discussion of methods used to characterize Fe oxide and other soil minerals see the work of Amonette & Zelazny.^[7] Environmental chemistry of Fe is discussed in the following references.^[9–13] A broader discussion of the role of nanoparticles in soils and related geosystems can be found in the work of Qafoku.^[17]

CONCLUSION

Fe oxides are one of the most important mineralogical components in soils. They primarily form by the oxidation of Fe²⁺ that is dissolved during the weathering processes. Because they occur as clay-sized particles having high specific surfaces, they substantially contribute to the physical and chemical properties of the soil. Their sorption and electron-buffering properties significantly affect the geochemical cycles of almost all elements having agronomic or environmental significance.

REFERENCES

- Kämpf, N.; Scheinost, A.C.; Schulze, D.G. Oxide minerals. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; F125–F137.
- Bigham, J.M.; Fitzpatrick, R.W.; Schulze, D.G. Iron oxides. In *Soil Mineralogy with Environmental Applications*; Dixon, J.B., Schulze, D.G., Eds.; SSSA Book Series; Soil Science Society of America, Inc.: Madison, 2002; Vol. 7, 323–366.
- Schwertmann, U.; Taylor, R.M. Iron oxides. In *Minerals in Soil Environments*, 2nd Ed.; Dixon, J.B., Weed, S.B., Eds.; SSSA Book Series; Soil Science Society of America, Inc.: Madison, 1989; Vol. 1, 379–438.
- Schwertmann, U. Solubility and dissolution of iron oxides. In *Iron Nutrition and Interactions in Plants, Developments in Plant and Soil Sciences*; Chen, Y., Hadar, Y., Eds.; Kluwer Academic Publishers: Dordrecht, 1991; Vol. 43, 3–27.
- Cornell, R.M.; Schwertmann, U. *The Iron Oxides*; VCH Publ.: Weinheim, Germany, 1996.
- Stucki, J.W.; Goodman, B.A.; Schwertmann, U. *Iron in Soils and Clay Minerals*; D. Reidel Publ. Co.: Dordrecht, Holland, 1988.
- Amonette, J.E.; Zelazny, L.W. *Quantitative Methods in Soil Mineralogy*; Miscellaneous Publ.; Soil Science Society of America: Madison, 1994.
- Majzlan, J.; Grevel, K.-D.; Navrotsky, A. Thermodynamics of Fe oxides: Part II. Enthalpies of formation and relative stability of goethite (α -FeOOH), lepidocrocite (γ -FeOOH), and maghemite (γ -Fe₂O₃). *Am. Mineral.* **2003**, *88*, 855–859.
- Gorski, C.A.; Scherer, M.M. Fe²⁺ sorption at the Fe oxide-water interface: A revised conceptual framework. In *Aquatic Redox Chemistry*. ACS Symposium Series; Tratnyek, P.G., Grundl, T.J., Haderlein, S.B., Eds.; American Chemical Society: Washington, D.C, 2011; Vol. 1071, 315–343.
- Amonette, J.E. Iron redox of clays and oxides: Environmental applications. In *Electrochemical Properties of Clays*; Fitch, A., Ed.; CMS Workshop Lectures, Clay Minerals Society: Aurora, Vol. 10, 89–146, in press.
- Stumm, W.; Morgan, J.J. *Aquatic Chemistry: Chemical Equilibria and Rates in Natural Waters*, 3rd Ed.; Wiley: New York, 1996.
- Stumm, W. *Chemistry of the Solid–Water Interface*; Wiley: New York, 1992.
- Sposito, G. *The Surface Chemistry of Soils*; Oxford University Press: New York, 1984.
- Chen, C.; Dynes, J.J.; Wang, J.; Sparks, D.L. Properties of Fe-organic matter associations via coprecipitation versus adsorption. *Environ. Sci. Technol.* **2014**, *48*, 13751–13759.
- Cloy, J.M.; Wilson, C.A.; Graham, M.C. Stabilization of organic carbon via chemical interactions with Fe and Al oxides in gley soils. *Soil Sci.* **2014**, *179*, 547–560.
- Colombo, C.; Palumbo, G.; He, J.-Z.; Pinton, R.; Cesco, S. Review on iron availability in soil: Interaction of Fe minerals, plants, and microbes. *J. Soil. Sediment.* **2014**, *14*, 538–548.
- Qafoku, N. Terrestrial nanoparticles and their controls on soil/geo processes and reactions. *Adv. Agron.* **2010**, *107*, 33–91.
- Hochella, M.F. Nanogeoscience: From origins to cutting-edge applications. *Elements* **2008**, *4*, 373–379.
- Carta, D.; Casula, M.F.; Corrias, A.; Falqui, A.; Navarra, G.; Pinna, G. Structural and magnetic characterization of synthetic ferrihydrite nanoparticles. *Mater. Chem. Phys.* **2009**, *113*, 349–355.
- Vayssieres, L. On the effect of nanoparticle size on water-oxide interfacial chemistry. *J. Phys. Chem. C.* **2009**, *113*, 4733–4736.
- Madden, A.S.; Hochella, M.F.; Luxton, T.P. Insights for size-dependent reactivity of hematite nanomineral surfaces through Cu²⁺ sorption. *Geochim. Cosmochim. Acta.* **2006**, *70*, 4095–4104.
- Erbs, J.J.; Gilbert, B.; Penn, R.L. Influence of size on reductive dissolution of six-line ferrihydrite. *J. Phys. Chem. C.* **2008**, *112*, 12127–12133.
- Bose, S.; Hochella, M.F.; Gorby, Y.A.; Kennedy, D.W.; McCready, D.E.; Madden, A.S.; Lower, B.H. Bioreduction of hematite nanoparticles by the dissimilatory iron reducing bacterium *Shewanella oneidensis* MR-1. *Geochim. Cosmochim. Acta.* **2009**, *73*, 962–976.

Irrigation

Jack Keller

Keller Bliesner Eng. LLC, Logan, Utah, U.S.A.

Abstract

The basic idea behind irrigation is to supply water and store water in soils for later use by plants where or when insufficient water for the desired level of plant growth is supplied by natural precipitation. There are three basic methods used for supplying irrigation water to the surface of soils and two for applying water beneath the soil surface.

IRRIGATION METHODS

The three application methods for applying water to the surface of soils are: 1) surface irrigation, where the soil surface is used to convey the irrigation water where it will be stored in the soil; 2) sprinkle irrigation, where the water is sprinkled on the soil surface where it is to be stored; and 3) drip irrigation, where the water is directly applied by dripping or being sprayed from small holes in the pipe network where it is to be stored in the soil. The two methods for applying irrigation water beneath the soil surface are: 1) subsurface drip irrigation, where the network of pipe and outlets is buried beneath the soil surface and 2) subirrigation, where the water table is controlled and the water reaches the plant root system through capillary action.

These methods of irrigation and irrigation efficiency issues are discussed in detail in other sections of this encyclopedia.

WATER STORAGE CAPACITY OF SOILS

Understanding basic soil–water–plant relationship is central to the ability to design and manage irrigation systems. The tables related to soil water presented in this section provide useful guidelines for estimating irrigation requirements and for preliminary irrigation design purposes. However, more site-specific information should be used for designing and managing irrigation systems and evaluating irrigation system performance.

Soil Water-Holding Capacity

Soils of various textures have varying abilities to retain water. Except for required periodic leaching, any irrigation beyond the field capacity of the soil is an economic loss. [Table 1](#) gives typical ranges of available water-holding capacities (field capacity minus permanent wilting point)

of soils of different textures. [Table 1](#) should only be used as a guide for preliminary system designs and management purposes. Final designs and management decisions should be based on actual field data.

Root Depth

The total amount of soil water available for plant use in any soil is the sum of the available water-holding capacities of all soil horizons occupied by plant roots. Typical plant feeder root and total root depth are given in many references; however, the actual depths of rooting of the various crops are affected by soil conditions. Therefore, the root depth for any set of crop and site should be checked. Where local data are not available and there are no expected root restrictions, [Table 2](#) can be used as a guide to estimating the effective root depths. The values given are taken by Keller and Bliesner and are based on the author's estimate of averages selected from a large number of references. They represent the depth at which crops will obtain the major portion of the water they need when grown in a deep, well-drained soil that is adequately irrigated.

CONSUMPTIVE USE AND DESIGN

Deciding how much water an irrigation system should be able to deliver to a crop over a given period is ultimately a question of selecting a capacity that will maximize profits to the farmer. To begin to address this question of system capacity, it is necessary to know how much water a crop will use over the entire growing season and during the part of the season when water use is at its peak. The rate of water use during this peak period provides the basis for determining the rate at which irrigation water should be delivered to the field. Examples of typical seasonal and peak daily crop water requirements are given in [Table 3](#).

Table 1 Range in available water-holding capacity of soils of different texture.

Soil texture	Water-holding capacity	
	Range (mm/range)	Average (mm/range)
Very coarse texture—very coarse sands	33–62	42
Coarse texture—coarse sands, fine sands, and loamy soils	62–104	83
Moderately coarse texture—sandy loams	104–145	25
Medium texture—very fine sandy loams, loams, and silt loams	125–192	167
Moderately fine texture—clay loams, silty clay loams, and sandy clay loams	145–208	183
Fine texture—sandy clays, silty clays, and clays	133–208	192
Peats and mucks	167–250	208

Source: Adapted from Soil Conservation Services (SCS), Irrigation Water Requirements.^[1]

SOIL MOISTURE MANAGEMENT

A general rule of thumb for many field crops in arid and semiarid regions is to maintain the soil moisture deficit (SMD) within the root zone above 50% of the total available water-holding capacity. This is a management-allowed deficit (MAD = 50%) because it is also desirable to bring the moisture level back to field capacity with each irrigation. (In humid regions, it is necessary to allow for rains during the irrigation period. However, the 50% limitation on soil moisture depletion should be followed as a general guide for field crops.)

Soil management, water management, and economic considerations determine the amount of water used in irrigating and the rate of water application. The standard approach has been to determine the amount of water needed to fill the entire root zone to field capacity and then to apply a sufficiently larger amount to account for evaporation, leaching, and efficiency of application. The traditional approach to the frequency of application has been to take the depth of water in the root zone reservoir that can be extracted assuming MAD = 50% and, using the daily consumptive use rate of the crop, to determine how long this supply will last. Such an approach is useful only as a guide to irrigation requirements, as many factors affect the volume and timing of applications for optimal design and operation of a system.

Table 4 is presented as a guide for selecting the appropriate MAD or for near optimum production of various crops. As indicated in Table 4 for crops with a high market value, it is often profitable to irrigate well before

Table 2 Effective crop root depths that would contain approximately 80% of the feeder roots in a deep, uniform, well-drained soil profile.^a

Crop	Root depth (M)
Alfalfa	1.2–1.8
Almonds	0.6–1.2
Apple	0.8–1.2
Apricot	0.6–1.4
Artichoke	0.6–0.9
Asparagus	1.2–1.8
Avocado	0.6–0.9
Banana	0.3–0.6
Barley	0.9–1.1
Bean (dry)	0.6–1.2
Bean (green)	0.5–0.9
Bean (lima)	0.6–1.2
Beet (sugar)	0.6–1.2
Beet (table)	0.4–0.6
Berries	0.6–1.2
Broccoli	0.6
Brussel sprouts	0.6
Cabbage	0.6
Cantaloupe	0.6–1.2
Carrot	0.4–0.6
Cauliflower	0.6
Celery	0.6
Chard	0.6–0.9
Cherry	0.8–1.2
Citrus	0.9–1.5
Coffee	0.9–1.5
Corn (grain and silage)	0.6–1.2
Corn (sweet)	0.4–0.6
Cotton	0.6–1.8
Cucumber	0.4–0.6
Eggplant	0.8
Fig	0.9
Flax	0.6–0.9
Grapes	0.5–1.2
Lettuce	0.2–0.5
Lucerne	1.2–1.8
Oats	0.6–1.1
Olives	0.9–1.5
Onion	0.3–0.6
Parsnip	0.6–0.9
Passion fruit	0.3–0.5
Pastures	0.3–0.8
Pea	0.4–0.8

(Continued)

Table 2 Effective crop root depths that would contain approximately 80% of the feeder roots in a deep, uniform, well-drained soil profile.^a (Continued)

Crop	Root depth (M)
Peach	0.6–1.2
Peanuts	0.4–0.8
Pear	0.6–1.2
Pepper	0.6–0.9
Plum	0.8–1.2
Potato (Irish)	0.6–0.9
Potato (sweet)	0.6–0.9
Pumpkin	0.9–1.2
Radish	0.3
Safflower	0.9–1.5
Sorghum (grain and sweet)	0.6–0.9
Sorghum (silage)	0.9–1.2
Soybean	0.6–0.9
Spinach	0.4–0.6
Squash	0.6–0.9
Strawberry	0.3–0.5
Sudan grass	0.9–1.2
Tobacco	0.6–1.2
Tomato	0.6–1.2
Turnip (white)	0.5–0.8
Walnuts	1.7–2.4
Watermelon	0.6–0.9
Wheat	0.8–1.1

^aApproximately 80% of the feeder roots are in the top 60% of the soil profile. Soil and plant environmental factors often offset normal root development; therefore, soil density, pore shapes and sizes, soil–water status, aeration, nutrition, texture and structure modification, soluble salts, and plant root damage by organisms should all be taken into account.

Source: Adapted from Keller & Bliesner.^[2]

half of the soil moisture in the root zone has been depleted, i.e., SMD = 50%.

Irrigation Depth

The maximum net depth of water to be applied per irrigation, d_x , is the same as the maximum allowable depletion of soil water between irrigations. Assuming no additional water is necessary for leaching purposes, it is computed by as follows:

$$d_x = \frac{MAD}{100} W_a Z \quad (1)$$

where d_x is maximum net depth of water to be applied and stored in the root zone per irrigation event [mm (in.)]; MAD is management-allowed deficit, which can be estimated

from Table 4 (%); W_a is available water-holding capacity of the soil, which can be estimated from Table 1 [mm/m (in./ft)]; and Z is effective root depth, which can be taken from Table 2 [mm (ft)].

Irrigation Interval

The appropriate irrigation interval, which is the time that should elapse between the beginning of two successive irrigation events, is determined by as follows:

$$f' = \frac{d_n}{U_d} \quad (2)$$

where f' is irrigation interval or frequency (days); d_n is net depth of water applied and stored in the soil per irrigation event, to meet consumptive use requirements [mm (in.)]; U_d is conventionally computed average daily crop water requirement, or use rate, during the peak-use month, which can be estimated from Table 3 [mm/day (in./day)].

The value selected for d_n will depend upon system design and environmental factors and should be equal to or less than d_x . When d_n is replaced by d_x in Eq. 2, f' becomes the maximum irrigation interval, f_x .

IRRIGATION SYSTEM DESIGN, MANAGEMENT, AND SCHEDULING

General Design Concepts

Design of all irrigation systems is process of synthesis, where properties, such as a soil's intake rate and crop water requirements; items, such as canals, pipes, and pumps; and processes, such as trenching, coaxing water down furrows, or moving pipe, must be integrated to form a good irrigation system. The irrigation designer's art is to know the kinds of hardware and management techniques appropriate for a given cropping system and site and to have a clear mental image of what the system can accomplish and how the completed system will appear. Careful site analysis is essential as it provides data that lead to an understanding of the physical, economic, and social resources that determine what can and ought to be accomplished by a proposed irrigation system.

Generally, irrigation systems are designed to meet average peak water use requirements. Sometimes, to reduce costs or to stretch limited water supplies, systems are designed to optimize production per unit of water applied. In such cases, systems can be designed to apply only about 80% of peak water requirements and still obtain up to 95% of optimum yields. For deep-rooted crops in fine-textured soils, an appreciable amount of water can be stored prior to the critical peak-use periods. By drawing on this stored water, peak system delivery

Table 3 Typical peak daily and seasonal crop water requirements in different climates.

Crop	Type of climate and water requirements (mm)									
	Daily	Seas	Daily	Seas	Daily	Seas	Daily	Seas	Daily	Seas
Alfalfa	5.1	635	6.4	762	7.6	914	8.9	1016	10.2	1219
Pasture	4.6	508	5.6	610	6.6	711	7.6	762	8.9	914
Grain	3.8	381	5.1	457	5.8	508	6.6	533	5.8 ^a	508 ^a
Beets	4.6	584	5.8	635	6.9	711	8.1	732	9.1	914
Beans	4.6	330	5.1	381	6.1	457	7.1	508	7.6	559
Corn	5.1	508	6.4	559	7.6	610	8.9	660	10.2	762
Cotton	—	—	6.4	559	7.6	660	—	—	10.2	813
Peas	4.6	305	4.8	330	5.1	356	5.6	356	5.1 ^b	356 ^b
Tomatoes	4.6	457	5.1	508	5.6	559	6.4	610	7.1	660
Potatoes	4.6	406	5.8	457	6.9	553	8.1	584	6.9 ^b	533 ^b
Truck vegetables	4.1	305	4.6	356	5.1	406	5.6	457	6.3 ^b	508 ^b
Melons	4.1	381	4.6	406	5.1	457	5.6	508	6.4 ^b	559 ^b
Strawberries	4.6	457	5.1	508	5.6	559	6.1	610	6.6	660
Citrus	4.1	508	4.6	559	5.1	660	—	—	5.6	711
Citrus (w/cover)	5.1	635	5.6	711	6.4	813	—	—	6.9	889
Dec orchard	3.8	483	4.8	533	5.8	584	6.6	635	7.6	762
Dec orchard (w/cover)	5.1	635	6.4	711	7.6	813	8.9	914	10.2	1016
Vineyards	3.6	356	4.1	406	4.8	457	5.6	508	6.4	610

^aWinter planting.^bFall or winter planting.**Source:** From Keller & Bliesner.^[2]

requirements can be reduced without reducing yield potential providing the quality of the irrigation water is high.

Salinity Control

All irrigation water contains some dissolved salts that are pushed downward by rainfall and application of irrigation water. By applying more water than the plants consume, most of the salts can be pushed or leached below the root zone. The first step in computing the additional water

required for leaching is to determine the leaching requirement by:

$$LR = \frac{EC_w}{5EC_e - EC_w} \quad (3)$$

where LR is leaching requirement ratio for sprinkle or surface irrigation; EC_w is electrical conductivity of the irrigation water [dS/m (mmhos/cm)]; EC_e is estimated electrical conductivity of the average saturation extract of the soil root zone profile for an appropriate yield reduction [dS/m (mmhos/cm)].

Unless more specific information on specific crop cultivars is available, the EC_e values presented in Table 5, which are taken by Ayers and Westcott,^[3] can be used in Eq. 3. These are values that will give an approximate 10% yield reduction, as presented by Ayers and Westcott.^[3] (For conversion purposes, 1.0 ppm = 640 × EC in dS/cm.)

Under full irrigation, where $LR < 0.1$, the annual deep percolation losses—even in most of the least watered areas—will normally be sufficient to provide the necessary leaching. However, under deficit irrigation or when $LR < 0.1$, water in addition to the consumptive use should be applied or available at some time during the year to satisfy leaching requirements. The ratio of the total depth

Table 4 Guide for selecting management-allowed deficit (MAD), values for various crops.

MAD (%)	Crop and root depth
25–40	Shallow-rooted, high-value fruit, and vegetable crops
40–50	Orchards, ^a vineyards, berries, and medium-rooted row crops
50	Forage crops, grain crops, and deep-rooted row crops

^aSome fresh fruit orchards require lower multiwavelength anomalous diffraction or values during fruit finishing for sizing.**Source:** From Keller & Bliesner.^[2]

Table 5 Values of EC_e that will give 10% yield reduction for various crops.

Crop	EC_e (dS/m)
Field crops	
Barley	10
Cotton	9.6
Sugar beets	8.7
Wheat	7.4
Soybean	5.5
Sorghum	5.1
Groundnut	3.5
Rice	3.8
Corn	2.5
Flax	2.5
Broadbeans	2.6
Cowpeas	2.2
Beans	1.5
Fruit and nut crops	
Date palm	6.8
Fig, olive	3.8
Pomegranate	3.8
Grapefruit	2.4
Orange	2.3
Lemon	2.3
Apple, pear	2.3
Walnut	2.3
Peach	2.2
Apricot	2
Grape	2.5
Almond	2
Plum	2.1
Blackberry	2
Boysenberry	2
Avocado	1.8
Raspberry	1.4
Strawberry	1.3
Vegetable crops	
Beets	5.1
Broccoli	3.9
Tomato	3.5
Cucumber	3.3
Cantaloupe	3.6
Spinach	3.3
Cabbage	2.8
Potato	2.5
Sweet corn	2.5

(Continued)

Table 5 Values of EC_e that will give 10% yield reduction for various crops. (Continued)

Crop	EC_e (dS/m)
Sweet potato	2.4
Pepper	2.2
Lettuce	2.1
Radish	2
Onion	1.8
Carrot	1.7
Beans	1.5
Forage crops	
Tall wheat grass	9.9
Bermuda grass	8.5
Barley (hay)	7.4
Rye grass	6.9
Crested wheat grass	6
Tall fescue	5.8
Sudan grass	5.1
Wild rye grass	4.4
Vetch	3.9
Alfalfa	3.4
Corn (forage)	3.2
Berseem clover	3.2
Orchard grass	3.1
Clover	2.3

Source: Adapted from Ayers & Westcott.^[3]

of irrigation water required with and without leaching is equal to $1/(1 - LR)$.

Drainage

Drainage of the excess water from the soil profile and conveying it to a sump for reuse or disposal is as important as irrigation. In fact natural or man-made subsurface drainage is essential for sustaining irrigated agricultural production. Without it leaching will not be possible, causing salts to accumulate until they become toxic to plant growth, and the water table will rise and literally drown the plants. Furthermore, the productive capability of the land itself may be severely damaged and require major reclamation to become productive again. Details regarding drainage needs are presented elsewhere in this encyclopedia.

Application Depth and Frequency

For periodic-move, and low-frequency, continuous-move systems, such as traveling sprinklers, it is desirable to irrigate as infrequently as practical to reduce labor costs and

Eq. 1 will apply. For drip solid-set and center-pivot sprinkle systems, labor costs are not a major consideration and the irrigation frequency can be selected to provide the optimal environment for plant growth, water conservation, and economic production within the physical limitations of the system.

Systems are usually designed so that their discharge, depths of application, and irrigation frequency meet crop water requirements during the peak consumptive use period. Therefore, systems must be managed to avoid wasting water, labor, and energy, and leaching nutrients from the soil during periods of the crops' growth cycle when water requirements are less, when the crops' roots may not have penetrated to their full depth, and during rainy periods. For optimum efficiency, irrigation applications should be scientifically scheduled from water

budgets based on crop evapotranspiration estimates or soil moisture observations.

REFERENCES

1. Soil Conservation Services (SCS), Irrigation Water Requirements. *Technical Release 21. Chapter 1, Section 15, Soil Conservation Service's (SCS) National Engineering Handbook*; U.S. Department of Agriculture: Washington, 1970.
2. Keller, J.; Bliesner, R.D. Soil-water-plant relations. In *Sprinkle and Trickle Irrigation*, 1st Ed.; Van Nostrand-Reinhold: New York, 1990.
3. Ayers, R.S.; Westcott, D.W. Salinity problems. In *Water Quality for Agriculture*; Irrigation and Drainage Paper 29; Rev. 1; Food and Agricultural Organization of the United Nations: Rome, 1985; 31–33.

Irrigation: Efficiency

Terry A. Howell

Conservation and Production Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Bushland, Texas, U.S.A.

Abstract

Irrigation efficiency is an important engineering term that involves understanding soil and agronomic sciences to achieve the greatest benefit from irrigation. The enhanced understanding of irrigation efficiency can improve the beneficial use of limited and declining water resources needed to enhance crop and food production from irrigated lands.

INTRODUCTION

Irrigation efficiency is a critical measure of irrigation performance in terms of the water required to irrigate a field, farm, basin, irrigation district, or an entire watershed. The value of irrigation efficiency and its definition are important to the societal views of irrigated agriculture and its benefit in supplying the high-quality, abundant food supply required to meet our growing world's population. "Irrigation efficiency" is a basic engineering term used in irrigation science to characterize irrigation performance, to evaluate irrigation water use, and to promote better or improved use of water resources, particularly those used in agriculture and turf/landscape management.^[1–4] Irrigation efficiency is defined in terms of: 1) the irrigation system performance; 2) the uniformity of the water application; and 3) the response of the crop to irrigation. Each of these irrigation efficiency measures is interrelated and will vary with scale and time. Fig. 1 illustrates several of the water transport components involved in defining various irrigation performance measures. The spatial scale can vary from a single irrigation application device (a siphon tube, a gated pipe gate, a sprinkler, and a microirrigation emitter) to an irrigation set (basin plot, a furrow set, a single sprinkler lateral, or a microirrigation lateral) to broader land scales (field, farm, an irrigation canal lateral, a whole irrigation district, a basin or watershed, a river system, or an aquifer). The timescale can vary from a single application (or irrigation set), a part of the crop season (preplanting, emergence to bloom or pollination, or reproduction to maturity), the irrigation season, to a crop season, or a year, partial year (premonsoon season, summer, etc.), or a water year (typically from the beginning of spring snow melt through the end of irrigation diversion, or a rainy or monsoon season), or a period of years (a drought or a "wet" cycle). Irrigation efficiency affects the economics of irrigation, the amount of water needed to irrigate a specific land area, the spatial uniformity of the crop and its yield, the amount of water that might percolate beneath the crop root

zone, the amount of water that can return to surface sources for downstream uses or to groundwater aquifers that might supply other water uses, and the amount of water lost to unrecoverable sources (salt sink, saline aquifer, ocean, or unsaturated vadose zone).

The volumes of the water for the various irrigation components are typically given in units of depth (volume per unit area) or simply the volume for the area being evaluated. Irrigation water application volume is difficult to measure, so it is usually computed as the product of water flow rate and time. This places emphasis on accurately measuring the flow rate. It remains difficult to accurately measure water percolation volumes, groundwater flow volumes, and water uptake from shallow ground water.

IRRIGATION SYSTEM PERFORMANCE EFFICIENCY

Irrigation water can be diverted from a storage reservoir and transported to the field or farm through a system of canals or pipelines; it can be pumped from a reservoir on the farm and transported through a system of farm canals or pipelines; or it might be pumped from a single well or a series of wells through farm canals or pipelines. Irrigation districts often include small to moderate size reservoirs to regulate flow and to provide short-term storage to manage the diverted water with the on-farm demand. Some on-farm systems include reservoirs for storage or regulation of flows from multiple wells.

Water Conveyance Efficiency

The conveyance efficiency is typically defined as the ratio between the water that reaches a farm or field and that diverted from the irrigation water source.^[1,3,4] It is defined as

$$E_c = 100 \frac{V_f}{V_t} \quad (1)$$

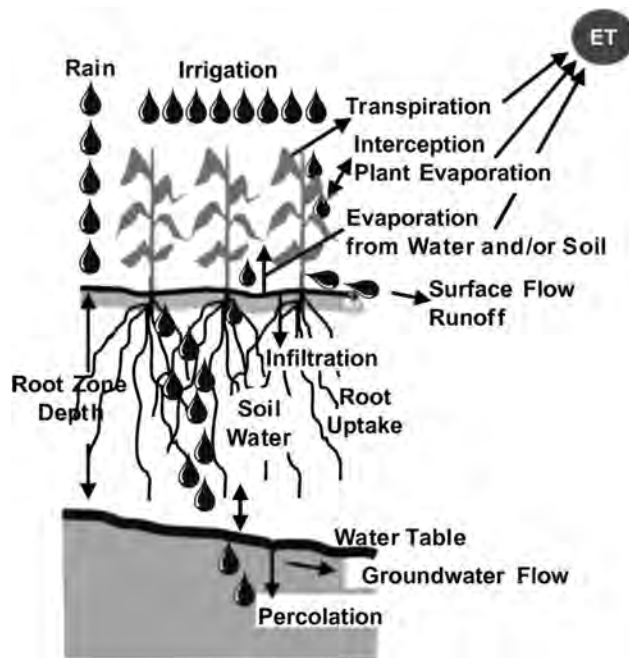


Fig. 1 Illustration of the various water transport components needed to characterize irrigation efficiency.

where E_c is the conveyance efficiency (%), V_f is the volume of water that reaches the farm or field (m^3), and V_t is the volume of water diverted (m^3) from the source. E_c also applies to segments of canals or pipelines, where the water losses include canal seepage or leaks in pipelines. The global E_c can be computed as the product of the individual component efficiencies, E_{ci} , where i represents the segment number. Conveyance losses include any canal spills (operational or accidental) and reservoir seepage and evaporation that might result from management as well as losses resulting from the physical configuration or condition of the irrigation system. Typically, conveyance losses are much lower for closed conduits or pipelines^[4] compared with unlined or lined canals. Even the conveyance efficiency of lined canals may decline over time due to material deterioration or poor maintenance.

Application Efficiency

Application efficiency relates to the actual storage of water in the root zone to meet the crop water needs in relation to the water applied to the field. It might be defined for individual irrigations or parts of irrigations (irrigation sets). Application efficiency includes any application losses to evaporation or seepage from surface water channels or furrows, any leaks from sprinkler or drip pipelines, percolation beneath the root zone, drift from sprinklers, evaporation of droplets in the air, or runoff from the field. Application efficiency is defined as

$$E_a = 100 \frac{V_s}{V_f} \quad (2)$$

where E_a is the application efficiency (%), V_s is the irrigation needed by the crop (m^3), and V_f is the water delivered to the field or farm (m^3). The root zone may not need to be fully refilled, particularly if some root zone water-holding capacity is needed to store possible or likely rainfall. Often, V_s is characterized as the volume of water stored in the root zone from the irrigation application. Some irrigations may be applied for reasons other than meeting the crop water requirement (germination, frost control, crop cooling, chemigation, fertigation, or weed germination). The crop need is often based on the “beneficial water needs.”^[5] In some surface irrigation systems, the runoff water that is necessary to achieve good uniformity across the field can be recovered in a “tailwater pit” and recirculated with the current irrigation or used for later irrigations, and V_f should be adjusted to account for the “net” recovered tailwater. Efficiency values are typically site specific. Table 1 provides a range of typical farm and field irrigation application efficiencies^[6–8] and potential or attainable efficiencies for different irrigation methods that assume irrigations are applied to meet the crop need.

Storage Efficiency

Since the crop root zone may not need to be refilled with each irrigation, the storage efficiency has been defined.^[4] The storage efficiency is given as

$$E_s = 100 \frac{V_s}{V_{rz}} \quad (3)$$

where E_s is the storage efficiency (%) and V_{rz} is the root zone storage capacity (m^3). The root zone depth and the water-holding capacity of the root zone determine V_{rz} . The storage efficiency has little utility for sprinkler or microirrigation because these irrigation methods seldom refill the root zone, while it is more often applied to surface irrigation methods.^[4]

Seasonal Irrigation Efficiency

The seasonal irrigation efficiency is defined as

$$E_i = 100 \frac{V_b}{V_f} \quad (4)$$

where E_i is the seasonal irrigation efficiency (%) and V_b is the water volume beneficially used by the crop (m^3). V_b is somewhat subjective,^[4,5] but it basically includes the required crop evapotranspiration (ET_c) plus any required leaching water (V_l) for salinity management of the crop root zone.

Leaching requirement (or the leaching fraction)

The leaching requirement,^[9] also called the leaching fraction, is defined as

Table 1 Example of farm and field irrigation application efficiency and attainable efficiencies.

Irrigation method	Field efficiency (%)		Farm efficiency (%)			
	Attainable	Range	Average	Attainable	Range	Average
Surface						
Graded furrow	75	50–80	65	70	40–70	65
w/tailwater reuse	85	60–90	75	85	—	—
Level furrow	85	65–95	80	85	—	—
Graded border	80	50–80	65	75	—	—
Level basins	90	80–95	85	80	—	—
Sprinkler						
Periodic move	80	60–85	75	80	60–90	80
Side roll	80	60–85	75	80	60–85	80
Moving big gun	75	55–75	65	80	60–80	70
Center pivot						
Impact heads w/end gun	85	75–90	80	85	75–90	80
Spray heads wo/end gun	95	75–95	90	85	75–95	90
LEPA wo/end gun	98	80–98	95	95	80–98	92
Lateral move						
Spray heads w/hose feed	95	75–95	90	85	80–98	90
Spray heads w/canal feed	90	70–95	85	90	75–95	85
Microirrigation						
Trickle	95	70–95	85	95	75–95	85
Subsurface drip	95	75–95	90	95	75–95	90
Microspray	95	70–95	85	95	70–95	85
Water table control						
Surface ditch	80	50–80	65	80	50–80	60
Subsurface drain lines	85	60–80	75	85	65–85	70

Note: LEPA, low energy precision application.

Source: From Howell,^[6] Merriam & Keller,^[7] and Hart.^[10]

$$L_t = \frac{V_d}{V_f} = \frac{EC_i}{EC_d} \quad (5)$$

where L_r is the leaching requirement, V_d is the volume of drainage water (m^3), V_f is the volume of irrigation (m^3) applied to the farm or field, EC_i is the electrical conductivity of the irrigation water ($dS\ m^{-1}$), and EC_d is the electrical conductivity of the drainage water ($dS\ m^{-1}$). The L_r is related to the irrigation application efficiency, particularly when drainage is the primary irrigation loss component. The L_r would be required “beneficial” irrigation use

$$V_l \equiv L_T V_i$$

so only V_d greater than the minimum required leaching should reduce irrigation efficiency. Then, the irrigation efficiency can be determined by combining Eqs. 4 and 5.

$$E_i = 100 \left(\frac{V_b}{V_f} + L_T \right) \quad (6)$$

Burt et al.^[5] defined the “beneficial” water use to include possible off-site needs to benefit society (riparian needs or wildlife or fishery needs). They also indicated that V_f should not include the change in the field or farm storage of water, principally soil water but it could include field (tailwater pits) or farm water storage (a reservoir) that was not used within the time frame that was used to define E_i .

IRRIGATION UNIFORMITY

The fraction of water used efficiently and beneficially is important for improved irrigation practice. The uniformity of the applied water significantly affects irrigation efficiency. This uniformity is a statistical property of the applied water’s distribution. This distribution depends on many factors that are related to the method of irrigation, soil topography, soil hydraulic or infiltration characteristics, and hydraulic characteristics (pressure, flow rate, etc.) of the irrigation system. Irrigation application distributions are usually based on depths of water (volume per unit area);

however, for microirrigation systems, they are usually based on emitter flow volumes because the entire land area is not typically wetted.

Christiansen's Uniformity Coefficient

Christiansen^[11] proposed a coefficient intended mainly for sprinkler systems based on the catch volumes given as

$$C_U = 100 \left[\frac{1 - (\sum |X - \bar{x}|)}{\sum X} \right] \quad (7)$$

where C_U is the Christiansen's uniformity coefficient (%), X is the depth (or volume) of water in each of the equally spaced catch containers (mm or ml), and $\bar{x}$ is the mean depth (volume) of the catch (mm or ml). For C_U values >70%, Hart^[10] and Keller and Bliesner^[8] presented

$$C_U = 100 \left[1 - \left(\frac{\sigma}{\bar{x}} \right) \left(\frac{2}{\pi} \right)^{0.5} \right] \quad (8)$$

where σ is the standard deviation of the catch depth (mm) or volume (ml). Eq. 8 approximates the normal distribution for the catch amounts.

The C_U should be weighted by the area represented by the container^[12] when the sprinkler catch containers intentionally represent unequal land areas, as is the case for catch containers beneath a center pivot. Heermann and Hein^[12] revised the C_U formula (Eq. 8) to reflect the weighted area, particularly intended for a center pivot sprinkler, as follows:

$$C_{U(H\&H)} = 100 \left\{ 1 - \left[\frac{\left(\sum S_i |V_i - \left(\frac{\sum V_i S_i}{\sum S_i} \right)| \right)}{\sum (V_i S_i)} \right] \right\} \quad (9)$$

where S_i is the distance (m) from the pivot to the i th equally spaced catch container and V_i is the volume of the catch in the i th container (mm or ml).

Low-Quarter Distribution Uniformity

The distribution uniformity represents the spatial evenness of the applied water across a field or a farm as well as within a field or farm. The general form of the distribution uniformity can be given as

$$D_{Up} = 100 \left(\frac{\bar{V}_p}{\bar{V}_f} \right) \quad (10)$$

where D_{Up} is the distribution uniformity (%) for the lowest p fraction of the field or farm (lowest one-half $p = 1/2$, lowest one-quarter $p = 1/4$), $\bar{V}_p$ is the mean application volume (m^3), and $\bar{V}_f$ is the mean application volume (m^3) for the whole field or farm. When $p = 1/2$ and $C_U > 70\%$, then the D_U and C_U are essentially equal.^[13] The

USDA-NRCS (formerly, the Soil Conservation Service) has widely used D_{U1q} ($p = 1/4$) for surface irrigation to access the uniformity applied to a field, i.e., by the irrigation volume (amount) received by the lowest one-quarter of the field from applications for the whole field. Typically, D_{Up} is based on the post-irrigation measurement^[5] of water volume that infiltrates the soil because it can more easily be measured and better represents the water available to the crop. However, the post-irrigation infiltrated water ignores any water intercepted by the crop and evaporated and any soil water evaporation that occurs before the measurement. Any water that percolates beneath the root zone or the sampling depth will also be ignored.

The D_U and C_U coefficients are mathematically interrelated through the statistical variation (coefficient of variation, $\sigma/\bar{x}$, C_v) and the type of distribution. Warrick^[13] presented relationships between D_U and C_U for normal, log-normal, uniform, specialized power, and beta and gamma distributions of applied irrigations.

Emission Uniformity

For microirrigation systems, both the C_U and D_U concepts are impractical because the entire soil surface is not wetted. Keller and Karmeli^[14] developed an equation for microirrigation design as follows

$$E_U = 100 \left[1 - 1.27(C_{vm})n^{-1/2} \right] \left(\frac{q_m}{\bar{q}} \right) \quad (11)$$

where E_U is the design emission uniformity (%), C_{vm} is the manufacturer's coefficient of variability in emission device flow rate (1 hr^{-1}), n is the number of emitters per plant, q_m is the minimum emission device flow rate (1 hr^{-1}) at the minimum system pressure, and $\bar{q}$ is the mean emission device flow rate (1 hr^{-1}). This equation is based on the D_{U1q} concept^[4] and includes the influence of multiple emitters per plant that each may have a flow rate from a population of random flow rates based on the emission device manufacturing variation. Nakayama et al.^[15] developed a design coefficient based more closely on the C_U concept for emission device flow rates from a normal distribution given as

$$C_{Ud} = 100(1 - 0.798(C_{vm})n^{-1/2}) \quad (12)$$

where C_{Ud} is the coefficient of design uniformity (%) and the numerical value, 0.798, is

$$\left(\frac{2}{\pi} \right)^{0.5}$$

from Eq. 8.

Many additional factors affect microirrigation uniformity including hydraulic factors, topographic factors, and emitter plugging or clogging.

WATER USE EFFICIENCY

The previous sections discussed the engineering aspects of irrigation efficiency. Irrigation efficiency is clearly influenced by the amount of water used in relation to the irrigation water applied to the crop and the uniformity of the applied water. These efficiency factors impact irrigation costs, irrigation design, and more important, in some cases, the crop productivity. The water use efficiency has been the most widely used parameter to describe irrigation effectiveness in terms of crop yield. Viets^[16] defined water use efficiency as

$$WUE = \frac{Y_g}{ET} \quad (13)$$

where WUE is water use efficiency (kg m^{-3}), Y_g is the economic yield (g m^{-2}), and ET is the crop water use (mm). WUE is usually expressed by the economic yield, but it has been historically expressed as well in terms of the crop dry matter yield (either total biomass or aboveground dry matter). These two WUE bases (economic yield or dry matter yield) have led to some inconsistencies in the use of the WUE concept. The transpiration ratio (transpiration per unit dry matter) is a more consistent value that depends primarily on crop species and the environmental evaporative demand,^[17] and it is simply the inverse of WUE expressed on a dry matter basis.

Irrigation Water Use Efficiency

The previous discussion of WUE does not explicitly explain the crop yield response to irrigation. WUE is influenced by the crop water use (ET). Bos^[3] defined a term for water use efficiency to characterize the influence of irrigation on WUE as

$$WUE = \frac{(Y_{gi} - Y_{gd})}{(ET_i - ET_d)} \quad (14)$$

where WUE is irrigation water use efficiency (kg m^{-3}), Y_{gi} is the economic yield (g m^{-2}) for irrigation level i , Y_{gd} is the dryland yield (g m^{-2} ; actually, the crop yield without irrigation), ET_i is the evapotranspiration (mm) for irrigation level i , and ET_d is the evapotranspiration of the dryland crops (or of the ET without irrigation). Although Eq. 14 seems easy to use, both Y_{gd} and ET_d are difficult to evaluate. If the purpose is to compare irrigation and dryland production systems, then dryland rather than non-irrigated conditions should be used. If the purpose is to compare irrigated regimes with an unirrigated regime, then appropriate values for Y_{gd} and ET_d should be used. Often, in most semiarid to arid locations, Y_{gd} may be zero. Bos^[3] defined irrigation water use efficiency as

$$IWUE = \frac{(Y_{gi} - Y_{gd})}{IRR_i} \quad (15)$$

where IWUE is the irrigation efficiency (kg m^{-3}) and IRR_i is the irrigation water applied (mm) for irrigation level i . In Eq. 15, Y_{gd} may be often zero in many arid situations.

CONCLUSION

Irrigation efficiency is an important engineering term that involves understanding soil and agronomic sciences to achieve the greatest benefit from irrigation. The enhanced understanding of irrigation efficiency can improve the beneficial use of limited and declining water resources needed to enhance crop and food production from irrigated lands.

REFERENCES

1. Israelsen, O.W.; Hansen, V.E. *Irrigation Principles and Practices*, 3rd Ed.; Wiley: New York, 1962; 447 pp.
2. ASCE. Describing irrigation efficiency and uniformity. *J. Irrig. Drain. Div. ASCE* **1978**, *104* (IR1), 35–41.
3. Bos, M.G. Standards for irrigation efficiencies of ICID. *J. Irrig. Drain. Div. ASCE* **1979**, *105* (IR1), 37–43.
4. Heermann, D.F.; Wallender, W.W.; Bos, M.G. Irrigation efficiency and uniformity. In *Management of Farm Irrigation Systems*; Hoffman, G.J., Howell, T.A., Solomon, K.H., Eds.; Am. Soc. Agric. Engrs: St. Joseph, 1990; 125–149.
5. Burt, C.M.; Clemmens, A.J.; Strelkoff, T.S.; Solomon, K.H.; Bliesner, R.D.; Hardy, L.A.; Howell, T.A.; Eisenhauer, D.E. Irrigation performance measures: Efficiency and uniformity. *J. Irrig. Drain. Eng.* **1997**, *123* (3), 423–442.
6. Howell, T.A. Irrigation efficiencies. In *Handbook of Engineering in Agriculture*; Brown, R.H., Ed.; CRC Press: Boca Raton, 1988; Vol. I, 173–184.
7. Merriam, J.L.; Keller, J. *Farm Irrigation System Evaluation: A Guide for Management*; Utah State University: Logan, 1978; 271 pp.
8. Keller, J.; Bliesner, R.D. *Sprinkle and Trickle Irrigation*; The Blackburn Press: Caldwell, 2000; 652 pp.
9. U.S. salinity laboratory staff. *Diagnosis and Improvement of Saline and Alkali Soils*; USDA Handbook 60; U.S. Govt. Printing Office: Washington, 1954; 160 pp.
10. Hart, W.E. Overhead irrigation by sprinkling. *Agric. Eng.* **1961**, *42* (7), 354–355.
11. Christiansen, J.E. *Irrigation by Sprinkling*; California Agric. Exp. Bull. No. 570; Univ. of Calif. Berkeley, 1942; 94 pp.
12. Heermann, D.F.; Hein, P.R. Performance characteristics of self-propelled center-pivot sprinkler machines. *Trans. ASAE* **1968**, *11* (1), 11–15.
13. Warrick, A.W. Interrelationships of irrigation uniformity terms. *J. Irrig. Drain. Eng. ASCE* **1983**, *109* (3), 317–332.
14. Keller, J.; Karmeli, D. *Trickle Irrigation Design*; Rainbird Sprinkler Manufacturing: Glendora, 1975; 133 pp.
15. Nakayama, F.S.; Bucks, D.A.; Clemmens, A.J. Assessing trickle emitter application uniformity. *Trans. ASAE* **1979**, *22* (4), 816–821.
16. Viets, F.G. Fertilizers and the efficient use of water. *Adv. Agron.* **1962**, *14*, 223–264.
17. Tanner, C.B.; Sinclair, T.R. Efficient use of water in crop production: Research or re-search? In *Limitations to Efficient Water Use in Crop Production*; Taylor, H.M., Jordan, W.R., Sinclair, T.R., Eds.; Am. Soc. Agron., Crop Sci. Soc. Am., Soil Sci. Soc. Am.: Madison, 1983; 27 pp.

Irrigation: Erosion

Robert E. Sojka

Northwest Irrigation and Soils Research Lab, U.S. Department of Agriculture (USDA), Kimberly, Idaho, U.S.A.

David L. Bjorneberg

Northwest Irrigation and Soil Research, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Kimberly, Idaho, U.S.A.

Abstract

This entry focuses on unique aspects of irrigation-induced soil erosion that are important when managing and simulating soil erosion on irrigated lands. Soil erosion mechanics can be divided into three components: detachment, transport, and deposition. Water droplets and flowing water detach soil particles; flowing water then transports these detached particles downstream; and deposition occurs when flowing water can no longer transport the soil particles because flow rate decreases as water infiltrates or as rill slope or roughness changes. Some particles are deposited within a few meters, although others are transported off the field with runoff water. These mechanisms are the same for surface irrigation, sprinkler irrigation, and rainfall; however, there are some systematic differences between irrigation and rainfall erosion, especially between surface irrigation and rainfall.

INTRODUCTION

Irrigation is important to global food production. About 15% of cropland^[1] and 5% of food production land, which includes rangeland and permanent cropland,^[2] are irrigated. However, irrigated land produces more than 30% of the world's food,^[3] which is 2.5 times as much per unit area compared with non-irrigated production.^[1] In the United States, approximately 15% of the harvested cropland is irrigated; however, almost 40% of the total crop value is produced on irrigated land.^[4]

Although sprinkler- and drip-irrigated areas are increasing, most of the world's irrigated land uses surface or flood irrigation. The countries with the largest irrigated areas are India—59,000,000 ha, China—52,580,000 ha, the United States—21,400,000 ha, and Pakistan—18,000,000 ha.^[2] These countries account for 55% of the world's irrigated land; all other countries have less than 10 million hectares each of irrigated land.^[2] About 50% of the irrigated land in the United States is surface irrigated^[5] although 95% to 99% of the irrigated land in India, China, and Pakistan is surface irrigated.^[6]

Soil erosion from irrigated fields has been discussed previously,^[7,8] and we focus on unique aspects of irrigation-induced soil erosion that are important when managing and simulating soil erosion on irrigated lands. Soil erosion mechanics can be divided into three components: detachment, transport, and deposition. Water droplets and flowing water detach soil particles; flowing water then transports these detached particles downstream;

and deposition occurs when flowing water can no longer transport the soil particles because flow rate decreases as water infiltrates or as rill slope or roughness changes. Some particles are deposited within a few meters, although others are transported off the field with runoff water. These mechanisms are the same for surface irrigation, sprinkler irrigation, and rainfall; however, there are some systematic differences between irrigation and rainfall erosion, especially between surface irrigation and rainfall.

SURFACE IRRIGATION

Soil erosion is often a serious problem on surface-irrigated land (Figs. 1 and 2). Erosion rates as high as 145 Mg/ha in 1 h^[9] and 40 Mg/ha in 30 minutes^[10] were reported in some early surface irrigation erosion studies. These extreme losses do not represent a sustained seasonal rate. Annual soil losses of 1 to 141 Mg/ha from surface-irrigated fields were reported in a 1980 southern Idaho study.^[11] Within-field erosion rates on the upper quarter of a furrow-irrigated field can be 10 to 30 times more than the field average erosion rate.^[12] Some soil eroded from the upper end of a field is deposited on the lower end, whereas some soil leaves the field with runoff. Losing topsoil from the upper end of the field can decrease crop yields 25% compared with the lower end of the field.^[13]

Sediment cannot be transported without runoff. Runoff is planned with many surface irrigation schemes in order to irrigate all areas of the field adequately. Under ideal



Fig. 1 White area in the field caused by erosion from more than 80 years of surface irrigation.



Fig. 2 Eroded irrigation furrows near the inflow end of a field.

conditions, properly designed and managed sprinkler irrigation systems will not have any runoff from the irrigated area. However, economic and water supply constraints, along with variable slope and soil conditions, often force compromises in sprinkler irrigation design.

SPRINKLER IRRIGATION

Runoff is rarely a problem with solid-set sprinkler irrigation systems because stationary sprinklers uniformly apply water at low rates (e.g., 2 mm/h). At the other end of the spectrum are systems with continuously moving laterals (center pivot and lateral move systems), which apply water to smaller areas (5–20 m wide) at higher rates than solid-set systems (e.g., 80 mm/h). Traveling lateral systems must irrigate large fields to reduce cost per unit area; this necessitates high instantaneous application rates to meet crop

water requirements over the entire field. Application rates for center pivot and lateral move irrigation systems often exceed the soil infiltration rate; therefore, runoff is almost always a potential problem. Sprinkler type, nozzle pressure, and nozzle size influence runoff and soil erosion by affecting application rate, wetted area, and droplet size. Low-pressure sprinklers, which reduce energy costs, have smaller pattern widths and therefore greater application rates. Lower pressure also produces larger drops with greater impact energy on the soil.

Sprinkler systems, particularly center pivots, operate on variable slopes and topography. Slope direction relative to the lateral affects how runoff accumulates. If the lateral is perpendicular to the slope direction, runoff will tend to move away from the lateral where water is being applied, allowing water to infiltrate before traveling very far. However, if the slope is parallel to the lateral, runoff can accumulate down-slope and begin flowing in erosive streams. Furthermore, if the lateral is traveling up-slope, runoff will flow onto a previously wetted area, whereas with down-slope travel runoff can flow onto dry soil. These factors are further complicated by wheel tracks from moving sprinkler systems that create compacted channels for water flow.

SURFACE IRRIGATION AND RAINFALL EROSION DIFFERENCES

The most obvious difference between soil erosion from rain or sprinkler irrigation and from surface irrigation is the lack of water droplets impacting the soil during surface irrigation. This fundamental difference is important because droplet kinetic energy affects both erosion and infiltration.^[14] When rain begins, droplets wet the soil surface and detach soil particles; as runoff begins, rills form in wet soil. Water flowing in rills is also exposed to falling raindrops, which affects detachment, transport, and deposition in the rills.

For furrow irrigation, rills are mechanically formed in dry soil before irrigation begins. Water is applied to only a small portion of the soil surface. As water advances down the field, it flows over dry, loose soil on the first irrigation and dry, consolidated soil on subsequent irrigations. Irrigation water instantaneously wets the soil, rapidly displacing air adsorbed on internal soil particle surfaces.^[15] The rapid replacement of air with water breaks apart soil aggregates,^[7] increasing the erodibility of the soil. Preliminary results from a southern Idaho field study showed that soil erosion from initially dry furrows was greater than soil erosion from furrows that were prewet by drip irrigation.

The hydraulics of rill flow from rain differ from furrow irrigation. Rill flow rate tends to increase downstream as additional rain water plus sheet and rill flow combine. During furrow irrigation, flow rate decreases with distance down the furrow as water infiltrates and increases with time as infiltration rate decreases, which changes sediment detachment and transport capacities with distance and time.

The duration of furrow irrigation runoff (typically 12 hours or more) is generally longer than most rain runoff events. Temporal changes in infiltration, soil and water temperature, rill size and shape, and soil erodibility become more important for longer runoff events. Sediment concentration tends to decrease with time during furrow irrigation. Flow rate, however, increases with time, which should increase sediment detachment and transport. This indicates that soil erodibility decreases during furrow irrigation by phenomena such as armoring, surface sealing, or other unrecognized processes.

Predicting small erosion events is important for irrigation. Seasonal irrigation-induced erosion occurs during numerous controlled and often small events rather than during one or two large erosion events. In southern Idaho, e.g., a cornfield may be sprinkler irrigated 15 to 20 times or furrow irrigated 6 to 8 times during the growing season. The magnitude of a single irrigation erosion event is usually much smaller and less dramatic than that of the erosion from a single 50-mm thunderstorm occurring on freshly tilled soil without an established crop. However, the cumulative soil loss from irrigation during the growing season may be substantial.

Chemical quality of rainfall varies less from location to location than surface water and groundwater quality. Irrigation water quality can also vary during the season as return flow is added to surface water sources or as groundwater and surface water sources are mixed. Water quality can significantly impact erosion from furrow- and sprinkler-irrigated fields. Increasing electrical conductivity (EC) tends to decrease erosion, whereas increasing sodium adsorption ratio (SAR) tends to increase erosion.^[16,17] Interactions among EC, SAR, clay flocculation, soil chemistry, rainfall application rate, etc. influence the effects of water quality on infiltration and erosion.

REFERENCES

1. Kendall, H.W.; Pimentel, D. Constraints on the expansion of the global food supply. *Ambio* **1994**, *23* (3), 198–205.
2. Food and Agriculture Organization. *FAOSTAT: Agriculture Data*. Food and Agriculture Organization Online Database. Available at <http://faostat.fao.org/> (accessed August 20, 2015).
3. Tribe, D. *Feeding and Greening the World, the Role of Agricultural Research*; CAB International: Wallingford, 1994.
4. National research council. *A New Era for Irrigation*; National Academy Press: Washington, 1996; 203 pp.
5. United States Department of Agriculture. *Farm and Ranch Irrigation Survey*. National Agricultural Statistics Service, 1998. Available at www.nass.usda.gov/census/ (accessed September 2000).
6. Food and Agriculture Organization. *AQUASTAT—Country Profiles*. Food and Agriculture Organization On-line Database. Available at www.fao.org/WAICENT/FAOINFO/AGRICULT/AGL/AGLW/aquastat (accessed September 2000).
7. Carter, D.L. Soil erosion on irrigated lands. In *Irrigation of Agricultural Crops*, Agronomy Monograph no. 30; Stewart, B.A., Nielson, D.R., Eds.; American Society of Agronomy: Madison, 1990; 1143–1171.
8. Koluvek, P.K.; Tanji, K.K.; Trout, T.J. Overview of soil erosion from irrigation. *J. Irrig. Drain. Eng.* **1993**, *119* (6), 929–946.
9. Israelson, O.W.; Clyde, G.D.; Lauritzen, C.W. *Soil Erosion in Small Irrigation Furrows*; Utah Agricultural Experiment Station: Logan, 1946.
10. Mech, S.J. Effect of slope and length of run on erosion under irrigation. *Agr. Eng.* **1949**, *30*, 379–383.
11. Berg, R.D.; Carter, D.L. Furrow erosion and sediment losses on irrigated cropland. *J. Soil Water Conserv.* **1980**, *35* (6), 367–370.
12. Trout, T.J. Furrow irrigation erosion and sedimentation: On-field distribution. *T. ASAE* **1996**, *39* (5), 1717–1723.
13. Carter, D.L.; Berg, R.D.; Sanders, B.J. The effect of furrow irrigation erosion on crop productivity. *Soil Sci. Soc. Am. J.* **1985**, *49* (1), 207–211.
14. Thompson, A.L.; James, L.G. Water droplet impact and its effect on infiltration. *T. ASAE* **1985**, *28* (5), 1506–1510.
15. Kemper, W.D.; Rosenau, R.; Nelson, S. Gas displacement and aggregate stability of soils. *Soil Sci. Soc. Am. J.* **1985**, *49* (1), 25–28.
16. Lentz, R.D.; Sojka, R.E.; Carter, D.L. Furrow irrigation water-quality effects on soil loss and infiltration. *Soil Sci. Soc. Am. J.* **1996**, *60* (1), 238–245.
17. Kim, K.-H.; Miller, W.P. Effect of rainfall electrolyte concentration and slope on infiltration and erosion. *Soil Technol.* **1996**, *9*, 173–185.

Irrigation: Historical Perspective

Robert E. Sojka

Northwest Irrigation and Soils Research Lab, U.S. Department of Agriculture (USDA), Kimberly, Idaho, U.S.A.

David L. Bjorneberg

Northwest Irrigation and Soil Research, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Kimberly, Idaho, U.S.A.

J.A. Entry

U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Kimberly, Idaho, U.S.A.

Abstract

Irrigation can be broadly defined as the practice of applying additional water (beyond what is available from rainfall) to soil to enable or enhance plant growth and yield and, in some cases, the quality of foliage or harvested plant parts. The water source could be groundwater pumped to the surface or surface water diverted from one position on the landscape to another. The development of irrigation water often entails the development of large-scale, geographically significant dams and water impoundments and/or diversions that can provide additional functions apart from crop growth enhancement, e.g., flood control, recreation, or generation of electricity. In many cases, sustainable irrigation development requires concomitant development of surface and/or subsurface drainage.

ANCIENT ORIGINS AND IMPORTANCE

Irrigation may be the single most strategically important, intentional, environmental modification humans have learned to perform. While irrigation's impact has not always been as critical to the global agricultural economy and food supply as it is today, it has always had major local impacts and profound historical and social consequences. In the Bible's book of Genesis, we are told that God's creation of humans was accompanied shortly thereafter by His assignment to Adam of the stewardship of the irrigated orchard that was Paradise. The four life-giving water heads of Judeo-Christian Paradise are also mentioned in the 47th sura of the Koran.^[1] Some anthropologists and historians point to the development of irrigation as the catalyst for the interaction of engineering, organizational, political, and related creative or entrepreneurial skills and activities that produced the outcome referred to as "civilization."^[2–5] In the ancient Persian language, the word "abadan," civilized, is derived from the root word "ab," water.^[1] Fundamental differences in social, cultural, religious, political, esthetic, economic, technological, and environmental outlook have been attributed to modern groupings of humankind related to their use of irrigation.^[5]

The earliest archeological evidence of irrigation in farming dates to about 6000 B.C. in the Middle East's Jordan Valley.^[1] It is widely believed that irrigation was being practiced in Egypt at about the same time,^[6] and the earliest pictorial representation of irrigation is from Egypt around 3100 B.C.^[1] In the following millennia, irrigation spread throughout Persia, the Middle East, and westward along the Mediterranean. In the same broad time frame, irrigation technology sprang up more or less independently across the Asian continent in India, Pakistan, China, and elsewhere. In the New World, the Inca, Maya, and Aztec made wide use of irrigation. The technology migrated as far north as the current Southwestern United States., where the Hohokam built some 700 miles of irrigation canals in what is today called as central Arizona to feed their emerging civilization, only to mysteriously abandon it in the 14th century A.D.^[3]

In the ancient world, the level of irrigation sophistication varied from one setting to the next. The differences, however, stemmed mostly from variations in understanding of both large- and small-scale hydraulic principles, as well as the capabilities to construct feats of hydraulic engineering. The Assyrians, for example, built an inverted siphon into the Nineveh Aqueduct 700 years before the birth of Christ, an engineering feat unrivaled until

the 1860 construction of the pressurized siphons of the New York Aqueduct.^[3]

Some ancient irrigation schemes have survived to the present day where geologic, soil, and climatic conditions were favorable and where then-known management principles were adequate for the prevailing conditions. However, some ancient schemes failed. In the Mesopotamian Valley, Syria, Egypt, and other areas throughout the Middle East, there were many cases where the principles of salt management and drainage were insufficiently understood, resulting in eventual permanent impairment of the land.^[1] Siltation of ancient dams and reservoirs is a testament to inadequate soil conservation measures that eventually reduced the productivity of the land as well as destroyed the capacity of reservoirs to provide an adequate supply of water.^[3] Erosion of irrigation channels, in geologically unstable areas like the Chilean deserts, and catastrophic failure of irrigation channels after earthquakes often defeated the best efforts of ancient engineers to maintain water supplies.^[3]

Modern irrigation technology probably began with the Mormon Settlement of the Utah Great Salt Lake Basin in 1847, and their eventual cultivation of nearly 2.5 million hectares irrigated across the intermountain Western United States by the turn of the century. Whereas the relationships between mass, energy, and turbulence of flow were mastered at remarkably high levels of proficiency in ancient cultures, understanding of chemistry and physicochemical interactions of soil and salt-bearing water was relatively meager even into the 19th century.

MODERNIZATION OF IRRIGATION

The mid-19th century marked a conjunction of several ascending areas of scientific learning, including chemistry, physical chemistry, physics, mineralogy, and biology. These were adapted, blended, and applied in important emerging new subdisciplines of soil chemistry, soil physics, plant physiology, and agronomy, whose fundamental principles were to prove essential for sustainable irrigation system design and operation.

In ancient irrigation developments, soils, climate, and water quality were used in more forgiving combinations at some locations than at others. Where seasonal rains provided leaching, where soils were permeable and well drained, and/or where irrigation water had favorable combinations of electrolyte concentrations and specific cations, irrigation has continued to the present day, even without sophisticated management. In other areas, salinization, increased soil sodicity, and elevated water tables have limited the life spans of irrigation schemes or impaired their productivity. As irrigation moved into more marginal settings, with less productive soils, poorer drainage, and greater salinity and sodicity

problems, the success or failure and ultimate longevity of the schemes became more dependent on knowledgeable application and adaptation of scientific principles. America's Mormon pioneers, choosing to settle in a remote salt-impaired desert habitat, were forced of necessity to use trial and error and the enlightened application of all available new knowledge to reclaim their lands from the desert and to practice a sustainable irrigated crop husbandry. They were so successful in their efforts that their approaches to irrigation and salt-threatened arid land reclamation and management provided the guiding principles for the development of irrigation throughout the Western United States from 1902 (with passage of the Reclamation Act) to the close of the 20th century.^[3] The science of irrigated agriculture and arid zone soil science, in general, relied mostly on the foundation and contributions stemming from these mid-19th century origins.^[7] The development of irrigation in the Western United States was further spurred by passage of the Desert Land Act of 1877 and the Carey Act of 1894, which provided land for settlement and governmental infrastructure for development. The first university-level irrigation course is believed to have been taught by Elwood Mead (Lake Mead's namesake) at the Agricultural College of Colorado in Fort Collins, Colorado.^[8] Mead later took positions with the U.S. Department of Agriculture and eventually was a commissioner for the Bureau of Reclamation. Worldwide, many of the practical modern principles of irrigation system design and irrigated soil management can be traced to the lessons learned in the settling of the American West from 1847 to the close of World War II, when the total irrigated area in the United States had grown to 7.5 million hectares.^[6]

Following World War II, irrigation development worldwide entered a heady period of rapid expansion. World populations were increasing, in part because of increased life expectancies resulting from new medicines and use of dichlorodiphenyltrichloroethane to control malaria and other disease-carrying insects. The advances in technology spurred by the First and Second World Wars were being applied to all avenues of life including agriculture. Electrical, steam, and internal combustion power sources became available to pump and pressurize water. New pump designs, the patenting of the center pivot, and other sprinkler delivery systems came together in a few short decades between and immediately following the wars to revolutionize the ability to deliver water.^[7]

EXISTING STATUS

In the United States, Soviet Union, Australia, and Africa, huge government-sponsored programs were initiated in the 1930s, 1940s, and 1950s to build dams for

hydropower, flood control, and irrigation and to encourage settlement and stabilization of sparsely populated frontiers. The worldwide total irrigated area was about 94 million hectares in 1950 and grew to 198 million hectares by 1970.^[9] In contrast, the world total irrigated area grew to only about 220 million hectares by 1990^[9] and to 263 million hectares by 1996.^[10] Not surprisingly, the easiest, least technically challenging, and least expensive irrigation developments occurred first, and more difficult, more technically challenging, and more expensive projects dominate the remaining potential for water development. In some instances, dams and large-scale water development projects have been hampered by poor economies and the instability of the countries in the potential development areas, rather than by the cost or technical challenges per se.

60% of the earth's grain production and half the value of all crops harvested result from irrigation.^[10] Perhaps, most remarkable is the agricultural production efficiency that irrigation provides worldwide. Some 50 million hectares of the earth's most productive irrigated cropland (4% of the earth's total cropland) produces a third of the entire planet's food crop.^[11] Hectare for hectare, irrigated land produces two to two and a half times the yield and three times the crop value per hectare compared with non-irrigated land.^[10,12,13] The irrigated portion amounts to only about one-sixth of the world's total cropped area^[14] and about 5% of the world's total production area, which includes cropland, range, and pasture.^[15] In the United States, most fresh fruits and vegetables in grocery stores come from irrigated agriculture. Beyond survival and economic impact, even our entertainment and esthetics rely heavily on irrigation. Nearly all-garden nursery stock in the United States is propagated and maintained under irrigation and parks, play fields, golf courses, and commercial landscaping are seldom established and maintained without irrigation.

To put the global production impact of irrigated agriculture in perspective, it would require over quarter billion hectares of new rainfed agricultural land (an area equal to the size of Argentina) to supply the average additional production that irrigation's high yield and efficiency provide. Actually, this estimate is conservative. If the land irrigated was no longer irrigated but left in production, its output would be well below the mean of existing rainfed land; this is because the lion's share of irrigation occurs in arid or semiarid environments. Furthermore, additional rainfed land brought into production to replace irrigated agriculture would be well below the existing rainfed average productivity; this is because the rainfed land with greatest yield potential has already been brought into production. A more realistic estimate might be double or triple the quarter billion hectare nominal replacement estimate.

In a world of six billion people, irrigation has become essential by providing another benefit that cannot be

immediately quantified, but which is as important as or more important than production efficiency or economic gain, or even the often uncredited benefits in many irrigation development schemes of hydropower, flood control, transportation, and rural development. The overriding benefit is security—security derived from food production stability. Substantial portions of the world food supply are subject to precipitous and often unpredictable yield reductions owing to drought. Irrigation was a key component of the “Green Revolution” of the 1960s and 1970s, which stabilized food production in the developing world, providing a new tier of nations the opportunity to turn some of their monetary and human resources to non-agricultural avenues of economic and social development. Much of the drop in the rate of increase of worldwide food production relates to the decrease in the rate of irrigation development since 1980.

ISSUES AFFECTING THE FUTURE

Although there are large projects underway or planned for the near future, notably in China, Pakistan, Brazil, Canada, Spain, and Portugal, the equal of the great dam-building era from 1930 to 1970 will likely never be seen again. Much of the development of irrigation has been achieved through exploitation of groundwater or by smaller-scale entrepreneurial surface water developments. In Australia, e.g., with the disastrous deflation of the world wool market in the 1990s, substantial numbers of individual sheep stations ceased raising animals and developed their surface water supplies to grow vast hectares of irrigated cotton and rice.

Worldwide, further expansions in irrigated area are unlikely to be large because of the limited remaining surface water sources to exploit and because of the growing environmental concerns, especially related to soil water-logging, salinization, and sodication problems. Future increases in irrigated area will likely result mainly from the development of the so-called “supplemental” irrigation in humid rainfed areas; from improvements in water use efficiencies associated with the utilization of existing irrigation resources; and from improvements in the reuse of municipal, industrial, and agricultural wastewaters. Howell^[10] noted that improved efficiencies have resulted in a reduction in the mean applied depth of water in the United States from about 650 mm annually in 1965 to 500 mm. These increased efficiencies have come in great part from the improved understanding of the energy physics of water, which led to modern evapotranspiration (ET) theory and ET-based crop irrigation scheduling.^[9,10] Many other water conservation practices were developed in the last half of the 20th century, including drip and micro-irrigation, which have spread from the hyperarid conditions of Israel in the early 1950s^[11] to nearly every climate and rainfall environment where there is a need, for one reason or another, to conserve water.

Loss of productive capacity caused by soil salinization, sodication, and waterlogging, as well as by runoff contamination, riparian habitat impairment, and species losses, is often cited by critics of irrigation as evidence of fundamental drawbacks of irrigated agriculture. Surveys have indicated that of the existing irrigated lands, some 40–50 million hectares show measurable degradation from waterlogging, salinization, and sodication.^[16,17] Erosion and sedimentation of reservoirs and channels caused the failures of ancient irrigation schemes and have limited the life expectancy of some modern dams to only a few decades as well.^[3,18] These problems should not be trivialized. They demonstrate the need for intensified research and conservation, as well as improved dissemination and the use of known prophylactic and remedial technologies. However, neither should they be overstated nor presented without due consideration of mitigating factors.

If rates of production loss from these problems are weighted by relative yield or economic value of irrigation compared to rainfed agriculture, and if other positive effects of irrigation are considered, the relative magnitude of negative impacts of irrigation is greatly diminished; e.g., runoff contamination from irrigated land would have to be three times the mean for rainfed land, on a crop value basis, or two to two and a half times the mean, on a yield basis, to be “comparable” to problems from non-irrigated agriculture because of the respective relative efficiencies of irrigated agriculture. Both the absolute and relative areas of impaired production, plus the degree of impairment, need to be compared on a global basis to rainfed losses, as well as the potential for remediation and production expansion under either circumstance. Positive impacts of irrigation water development include many social and economic benefits such as hydropower, flood control, transportation, recreation, and rural development. Positive environmental effects result from crops, field borders, canals, ditches, and reservoirs that provide significant expansions of habitat for a variety of wildlife compared to undeveloped arid land.

As with all agriculture methods, irrigated agriculture has greatly improved its ability to provide humanity's essential needs in closer harmony with environmental needs. This is a key priority in modern irrigated agricultural research along with continued improvement in production potential to meet the needs of a growing population. Population growth is occurring mostly in underdeveloped nations, where there is an added expectation of improved diet and standard of living. This expectation raises the need for improved production per capita above a simple linear extrapolation based on population. Only high yield intensive production from irrigated agriculture has shown the potential to meet these projected needs.

The knowledge and technology exist to design and operate irrigated agricultural systems sustainably and without environmental damage or irreversible soil

impairment.^[9,16,19] The problem lies in implementing known scientific principles and technologies in a timely fashion as part and parcel of irrigation project and system design and management. This is true both on a regional or a project basis and at the farm or field level. Politics and economics play pivotal roles in how well-known science and technology are applied. In this respect, irrigated agriculture is no different than the myriad manifestations of rainfed agriculture, or any other environmentally impacting activity. Because of modern political and economic considerations, there is usually great pressure, when designing and developing a large-scale irrigation project, to allocate resources for the development of as many irrigated hectares as possible at the outset. This often occurs without provision of an adequate technical or social support network to the farming community making the transition to irrigated agriculture. Many schemes fail to provide sufficient financial or technical resources to install drainage systems, to educate farmers, or to include them in policy formulation. The resources are needed to help guarantee prudent water application and salinity or drainage management compatible with the social, technical, and financial capabilities of the water users. These are not failures of irrigation. They are failures of human institutions. In this respect, human political, economic, and institutional considerations rather than technical advances or water availability may represent the real challenges for irrigation in the 21st century. These obstacles must be overcome if irrigated agriculture is to provide the production advantage required to satisfy future human needs and to meet improved dietary and living standard expectations.

REFERENCES

1. Hillel, D. *Rivers of Eden: the Struggle for Water and the Quest for Peace in the Middle East*; Oxford University Press: New York, 1994; 355 pp.
2. Mitchell, W.P. The hydraulic hypothesis: A reappraisal. *Curr. Anthropol.* **1973**, *14*, 532–534.
3. Reisner, M. *Cadillac Desert: The American West and its Disappearing Water*; Penguin Books: New York, 1986; 582 pp.
4. Wittfogel, K.A. *Oriental Despotism: A Comparative Study of Total Power*; Yale University Press: New Haven, 1956.
5. Worster, D. *Rivers of Empire: Water, Aridity & the Growth of the American West*; Pantheon Books: New York, 1985; 402 pp.
6. Hoffman, G.J.; Howell, T.A.; Solomon, K.H. Introduction. In *Management of Farm Irrigation Systems*; Hoffman, G.J., Howell, T.A., Solomon, K.H., Eds.; American Society of Agricultural Engineers: St. Joseph, 1990; 5–10.
7. Morgan, R.M. *Water and the Land—A History of American Irrigation*; The Irrigation Association: Fairfax, 1993; 208 pp.

8. Heerman, D.F. Where we have been, what we have learned and where we are going. In *National Irrigation Symposium, Proceedings of the 4th Decennial Symposium*; Phoenix, AZ, Nov 14–16, 2000; Evans, R.G., Benham, B.L., Trooien, T.P., Eds.; American Society of Agricultural Engineers: St. Joseph, 2000; 40–51.
9. Jensen, M.E.; Rangeley, W.R.; Dieleman, P.J. Irrigation trends in world agriculture. In *Irrigation of Agricultural Crops*; Stewart, B.A., Nielsen, D.R., Eds.; Agronomy Monograph 30; American Society of Agronomy: Madison, 1990; 31–67.
10. Howell, T.A. Irrigation's role in enhancing water use efficiency. In *National Irrigation Symposium, Proceedings of the 4th Decennial Symposium*; Evans, R.G., Benham, B.L., Trooien, T.P., Eds.; American Society of Agricultural Engineers: St. Joseph, 2000; 66–80.
11. Tribe, D. *Feeding and Greening the World, the Role of Agricultural Research*; CAB International: Wallingford, 1994; 274 pp.
12. Bucks, D.A.; Sammis, T.W.; Dickey, G.L. Irrigation for arid areas. In *Management of Farm Irrigation Systems, ASAE Monograph*; Hoffman, G.J., Howell, T.A., Solomon, K.H., Eds.; American Society of Agricultural Engineers: St. Joseph, 1990; 499–548.
13. Kendall, H.W.; Pimentel, D. Constraints on the expansion of the global food supply. *Ambio* **1994**, 23 (3), 198–205.
14. Gleick, P.H. *Water in Crisis: A Guide to the World? Fresh Water Resources*; Oxford University Press: New York, 1993; 473 pp.
15. Food and Agriculture Organization. *FAOSTAT—Agriculture Data, Food and Agriculture Organization On-Line Database*. Available at <http://apps.fao.org> (accessed September 2000).
16. Rhoades, J.D. Sustainability of irrigation: An overview of salinity problems and control strategies. In *CWRA 1997, Annual Conference on Footprints of Humanity: Reflections on Fifty Years of Water Resource Developments*, Lethbridge, Alta, Jun 3–6, 1997; 1–42.
17. Ghassemi, F.; Jakeman, A.J.; Nix, H.A. *Salinisation of Land and Water Resources, Human Causes, Extent, Management, and Case Studies*; CAB International: Wallingford, 1995.
18. Fukuda, H. *Irrigation in the World*; University of Tokyo Press: Tokyo, 1976.
19. Sojka, R.E. Understanding and managing irrigation-induced erosion. In *Advances in Soil and Water Conservation*; Pierce, F.J., Frye, W.W., Eds.; Sleeping Bear Press: Ann Arbor, 1998; 21–37.

Irrigation: Soil Salinity

James D. Rhoades

Agricultural Salinity Consulting, Riverside, California, U.S.A.

Abstract

Irrigation has resulted in considerable salination of associated land and water. Surviving the salinity threat requires that the seriousness of the problem be recognized more widely, the processes contributing to the salination of irrigated lands be understood, effective control measures that will sustain the viability of irrigated agriculture be developed and implemented, and that practical reclamation measures be implemented to rejuvenate the degraded lands.

INTRODUCTION

Irrigation is an ancient practice that predates recorded history. While irrigated farmland comprises only about 15% of the world's total farmland, it contributes about 36% of the total supply of food and fiber, and it stabilizes production against the vagaries of weather.^[1] In 30-year time, irrigated agriculture is expected to have to supply 50% of the world's food production requirements.^[1] However, irrigation growth has actually slowed to a rate that is inadequate to keep up with the projected expanding food requirements.^[1] Furthermore, irrigation has resulted in considerable salination of associated land and water. It has been estimated variably that the salinized area is as low as 20% and as high as 50% of the world's irrigated land.^[2–4] Worldwide, about 76.6 Mha of land have become degraded by human-induced salination.^[3] It has been estimated that the world is losing at least 3 ha of arable land every minute to soil salination (about 1.6 Mha per year) and then only to erosion as the leading worldwide cause of soil degradation.^[5–7] These data imply that the rate of salinization in developed irrigation projects exceeds the rate of irrigation expansion.^[8]

Surviving the salinity threat requires that the seriousness of the problem be recognized more widely, the processes contributing to the salination of irrigated lands be understood, effective control measures be developed and implemented that will sustain the viability of irrigated agriculture, and that practical reclamation measures be implemented to rejuvenate the degraded lands.^[9,10]

DELETERIOUS EFFECTS OF SALTS ON PLANTS, SOILS, AND WATERS

Salt-affected soils have reduced the value for agriculture because of their content and proportions of salts, consisting mainly of sodium, magnesium, calcium, chloride, and

sulfate and secondarily of potassium, bicarbonate, carbonate, nitrate, and boron. Saline soils contain excessive amounts of soluble salts for the practical and normal production of most agricultural crops. Sodic soils are those that contain excessive amounts of adsorbed sodium in proportion to calcium and magnesium, given the salinity level of the soil water. An example of a salt-affected irrigated soil is shown in Fig. 1.

Soluble salts exert both general and specific effects on plants, both of which reduce crop yield.^[11] Excess salinity in the seedbed hinders seedling establishment and in the crop root zone causes a general reduction in growth rate. In addition, certain salt constituents are specifically toxic to some plants. For example, boron is highly toxic to susceptible crops when present in the soil water at the concentrations of only a few parts per million. In some woody crops, sodium and chloride may accumulate in the tissue over time to toxic levels. These toxicity problems are, however, much less prevalent than is the general salinity problem.

Salts may also change soil properties that affect the suitability of the soil as a medium for plant growth.^[12] The suitability of soils for cropping depends appreciably on the readiness with which they conduct water and air (permeability) and on their aggregate properties (structure), which control the friability (ease with which crumbled) of the seedbed (tilth). In contrast to saline soils, which are well aggregated and whose tillage properties and permeability to water and air are equal to or higher than those of similar non-saline soils, sodic soils have reduced permeabilities and poor tilth. These problems are caused by the swelling and dispersion of clay minerals and by the breakdown of soil structure (slaking and crusting), which results in loss of permeability and tilth. Sodic soils are generally less extensive but more difficult to reclaim than saline soils.

Beneficial use of water in irrigation consists of transpiration and leaching for salinity control (the leaching requirement). Plant growth is directly proportional to water consumption through transpiration.^[13] From the point of



Fig. 1 Photograph of salt-affected irrigated field.

view of irrigated agriculture, the ultimate objective of irrigation is to increase the amount of water available to support transpiration. Salts reduce the fraction of water in a supply (or in the soil profile) that can be consumed beneficially in plant transpiration.^[14] In considering the use of a saline water for irrigation and in selecting appropriate policies and practices of irrigation and drainage management, it is important to recognize that the total volume of a saline water supply cannot be consumed beneficially in crop production (i.e., transpired by the plant). A plant will not grow properly when the salt concentration in the soil water exceeds some limit specific to it under the given conditions of climate and management.^[11] This is even true for halophytes.^[15] Thus, the practice of blending or diluting excessively saline waters with good quality water supplies should be undertaken only after consideration is given to how it affects the volumes of consumable (usable) water in the combined and separated supplies.^[14]

CAUSES OF SALINATION INDUCED BY IRRIGATION AND DRAINAGE

While salt-affected soils occur extensively under natural conditions, the salt problems of the greatest importance to agriculture arise when previously productive soils become salinized as a result of agricultural activities (the so-called secondary salination). The extent and salt balance of salt-affected areas has been modified considerably by the redistribution of water (hence salt) through irrigation and drainage. The development of large-scale irrigation and drainage projects, which involves diversion of rivers, construction of large reservoirs, and irrigation of large landscapes, causes large changes in the natural water and salt balances of entire geohydrologic systems. The impact of such developments can extend well beyond that of the immediate irrigated area. Excessive water diversions and applications are major causes of soil and water salination in irrigated lands. It is not unusual to find that less than

60% of the water diverted for irrigation is used in crop transpiration.^[9] This implies that about 40% of the irrigation water eventually ends up as deep percolation. This drainage water contains more salt than that added with the irrigation water because of salt dissolution and mineral weathering^[14] within the root zone. It often gains additional salt load as it dissolves salts of geologic origin from the underlying substrata through which it flows in its down-gradient path. This drainage water often flows laterally to lower lying areas, eventually resulting in shallow saline groundwaters of large areas of land (waterlogging). Salination occurs in soils underlain by saline shallow groundwater through the process of “capillary rise” as groundwater (hence, salt) is driven upward by the force of evaporation of water from the soil surface. Correspondingly, saline soils and waterlogging are closely associated problems.

Seepage from unlined or inadequately lined delivery canals occurs in many irrigation projects and is often substantial. Law et al.^[16] estimated that 20% of the total water diverted for irrigation in the United States is lost by seepage from conveyance and irrigation canals. Biswas^[17] estimated that 57% of the total water diverted for irrigation in the world is lost from conveyance and distribution canals. Analogous to on-farm deep percolation resulting from irrigation, these seepage waters typically percolate through the underlying strata (often dissolving additional salts in the process), flow to lower elevation lands or waters, and add to the problems of waterlogging and salt loading associated with on-farm irrigation there. A classic example of the rise in the water table following the development of irrigation has been documented in Pakistan and is described by Jensen et al.^[9] and Ghassemi et al.^[2] The depth to the water table in the irrigated landscape located between three major river tributaries rose from 20 to 30 m over a period of 80–100 years, i.e., from pre-irrigated time (about 1860) to the early 1960s, until it was nearly at the soil surface. In one region, the water table rose nearly linearly from 1929 to 1950, demonstrating that deep percolation and seepage resulting from irrigation were the primary causes. Ahmad^[18] concluded that about 50% of the water diverted into irrigation canals in Pakistan eventually goes to the groundwater by seepage and deep percolation.

The role of irrigated agriculture in salinizing soil systems has been well recognized for hundreds of years. It is of relatively more recognition that the salination of water resources from agricultural activities is a major and widespread phenomenon of likely equal concern to that of soil salination. The causes of water salination are essentially the same as those of soils, where the final reservoir of the discharged salt load is only a water supply in the former case.^[14] The volume of the water supply is reduced through irrigation diversions and irrigation; thus, its capacity to assimilate such received salts before reaching use-limiting levels is reduced proportionately. Only lately has it become apparent that trace toxic constituents, such as selenium, in

agricultural drainage waters can also cause serious pollution problems.^[19]

IRRIGATION AND DRAINAGE MANAGEMENT TO CONTROL SOIL SALINITY

The key to overall salinity control is strict control that maintains a net downward movement of soil water in the root zone of irrigated fields over time while minimizing excess irrigation diversions, applications, and deep percolation.^[20] The direct effect of salinity on plant growth is minimized by maintaining the soil–water content in the root zone within a narrow range at a relatively high level, while at the same time avoiding surface ponding and oxygen depletion and minimizing deep percolation. Combined methods of pressurized, high-frequency irrigation and irrigation scheduling have been developed that permit substantially the desired control to be achieved.^[21,22] These systems transfer the control of water distribution and infiltration from the soil to the irrigation equipment. This results in less excess water (and hence, less salt) being applied overall to the field to meet the needs of a part of the field area having the lowest intake rate, as done in the more traditional gravity irrigated systems. However, gravity irrigation systems can be designed to achieve good irrigation efficiency and salinity control even though surface ponding does occur. The so-called level-basin, multiset, cablegation, surge, and tailwater-return systems are among them.^[21,22] The need for irrigation and the amount required to meet evapotranspiration and leaching requirement is determined from plant stress measurements, the calculations of evapotranspiration amounts, measurements of soil–water depletion, measurements of soil (or soil–water) salinity, or a combination of them.^[21,22]

In addition to effective methods of irrigation scheduling and application, appropriate irrigation and salinity management also require an effective delivery system. Delivery systems have generally been designed to provide water on a regular schedule. Efficient irrigation systems require more flexible deliveries that can provide water on demand as each crop and particular field have need of it. Delivery systems can be improved by lining the canals, by containing the water within closed conduits, and by implementing techniques that increase the flexibility of delivery.

As briefly discussed earlier, irrigated agriculture is a major contributor to the salinity of many rivers and groundwaters, as well as soils. Reducing deep percolation generally lessens the salt load that is returned to rivers or groundwater and their pollution.^[14] Additionally, saline drainage waters should be intercepted before being allowed to mix with water of better quality. The intercepted saline drainage water should be desalted and reused, disposed of by pond evaporation or by injection into some suitably isolated deep aquifer, or better should be used for irrigation in a situation where brackish water is appropriate. Various

irrigation and drainage strategies have been developed for minimizing the pollution of waters from irrigation and for using brackish waters for irrigation.^[14,23] The desalination of agricultural drainage waters is not economically feasible, but improved techniques for doing this exist and some are being implemented. However, more needs to be done in this regard.

Traditionally, the concepts of leaching requirement and salt-balance index have been used to plan and judge the appropriateness of irrigation and drainage systems, operations and practices with respect to salinity control, water-use efficiency, and irrigation sustainability. However, these approaches are inadequate. The recommended method is to monitor directly the root-zone salinity levels and distributions across fields as a means to evaluate the effectiveness of salinity, irrigation, and drainage management practices, to detect problems, to help determine the underlying causes of problems, and to determine source areas of major water and salt-load contributions to the underlying groundwater. Theory, equipment, and practical technology have been developed for these purposes.^[24] More information about irrigation and drainage management to control soil and water salinity is found elsewhere.^[25–27]

REFERENCES

1. FAO. *World Agriculture toward 2000: An FAO Study*; Alexandratos, N., Ed.; Bellhaven Press: London, 1988; 338 pp.
2. Ghassemi, F.; Jakeman, A.J.; Nix, H.A. *Salination of Land and Water Resources. Human Causes, Extent, Management and Case Studies*; CAB International: Wallingford, 1995; 526 pp.
3. Oldeman, L.R.; van Engelen, V.N.P.; Pulles, J.H.M. The extent of human-induced soil degradation. In *World Map of the Status of Human-Induced Soil Degradation: An Explanatory Note*; Oldeman, L.R., Hakkeling, R.T.A., Sombroek, W.G., Eds.; International Soil Reference and Information Center (ISRIC): Wageningen, 1991; 27–33.
4. Adams, W.M.; Hughes, F.M.R. Irrigation development in desert environments. In *Techniques for Desert Reclamation*; Goudie, A.S., Ed.; Wiley: New York, 1990; 135–160.
5. Buringh, P. Food production potential of the world. In *The World Food Problem: Consensus and Conflict*; Sinha, R., Ed.; Pergamon Press: Oxford, 1977; 477–485.
6. Dregne, H.; Kassas, M.; Rozanov, B. A new assessment of the world status of desertification. *Desertification Control Bull.* **1991**, *20*, 6–18.
7. Umali, D.L. *Irrigation-induced Salinity: A Growing Problem for Development and Environment*; Technical Paper No. 215; World Bank: Washington, 1993.
8. Seckler, D. *The New Era of Water Resources Management: From "Dry" to "Wet" Water Savings*; Issues in Agriculture No. 8; Consultative Group on International Agricultural Research: Washington, 1996.
9. Jensen, M.E.; Rangeley, W.R.; Dieleman, P.J. Irrigation trends in world agriculture. In *Irrigation of Agricultural Crops*; Stewart, B.A., Nielsen, D.R., Eds.; American Society

- of Agronomy Monograph No. 30; ASA: Madison, 1990; 31–67.
10. UNEP. *Saving Our Planet: Challenges and Hopes*; Nairobi, United Nations Environment Program: Nairobi, 1992; 20 pp.
11. Maas, E.V. Crop salt tolerance. In *Agricultural Salinity Assessment and Management Manual*; Tanji, K.K., Ed.; ASCE Manuals and Reports on Engineering No. 71; ASCE: New York, 1990; 262–304.
12. Rhoades, J.D. Principal effects of salts on soils and plants. In *Water, Soil & Crop Management Relating to the Use of Saline Water*; Kandiah, A., Ed.; FAO (AGL) Misc. Series Publication 16/90; Food and Agriculture Organization of the United Nations: Rome, 1990; 1933 pp.
13. Sinclair, T.R. Limits to crop yield? In *Physiology and Determination of Crop Yield*; Boote, K.J., Ed.; American Society of Agronomy: Madison, 1994; 509–532.
14. Rhoades, J.D.; Kandiah, A.; Mashali, A.M. *The Use of Saline Waters for Crop Production*; FAO Irrigation and Drainage Paper 48; FAO: Rome, 1992; 133 pp.
15. Miyamoto, S.; Glenn, E.P.; Oslon, M.W. Growth, water use and salt uptake of four halophytes irrigated with highly saline water. *J. Arid Environ.* **1996**, *32*, 141–159.
16. Law, J.P.; Skogerboe, G.V.; Denit, J.D. The need for implementing irrigation return flow control. In *Managing Irrigated Agriculture to Improve Water Quality*, Proceedings of Math. Conf. Manag. Irrig. Agric. Improve Water Avail., Denver, CO; Graphics Management Corporation: Washington, 1972; 16–18.
17. Biswas, A.K. Conservation and management of water resources. In *Techniques for Desert Reclamation*; Goudie, A.S., Ed.; Wiley: New York, 1990; 251–265.
18. Ahmad, N. Planning for future water resources of Pakistan. In *Proceedings of Darves Borno Spec. Conference, National Committee of Pakistan*; ICID: New Delhi, 1986; 279–294.
19. Letey, J.; Roberts, C.; Penberth, M.; Vasek, C. *An Agricultural Dilemma: Drainage Water and Toxics Disposal in the San Joaquin Valley*; Special Publication 3319; University of California: Oakland, 1986; 56.
20. Rhoades, J.D. Soil salinity—Causes and controls. In *Techniques for Desert Reclamation*; Goudie, A.S., Ed.; Wiley: New York, 1990; 109–134.
21. Hoffman, G.J.; Rhoades, J.D.; Letey, J.; Sheng, F. Salinity management. In *Management of Farm Irrigation Systems*; Hoffman, G.J., Howell, T.A., Solomon, K.H., Eds.; ASCE: St. Joseph, 1990; 667–715.
22. Kruse, E.G.; Willardson, L.; Ayars, J. On-farm irrigation and drainage practices. In *Agricultural Salinity Assessment and Management Manual*; Tanji, K.K., Ed.; ASCE Manuals and Reports on Engineering No. 71; ASCE: New York, 1990; 349–371.
23. Rhoades, J.D. Use of saline drainage water for irrigation. In *Agricultural Drainage*; Skaggs, R.W., van Schilfgaarde, J., Eds.; ASA Drainage Monograph 38; American Society of Agronomy: Madison, 1999; 619–657.
24. Rhoades, J.D.; Chanduvi, F.; Lesch, S. *Soil Salinity Assessment: Methods and Interpretation of Electrical Conductivity*; FAO Irrigation and Drainage Paper 57; FAO, United Nations: Rome, 1999; 152 pp.
25. Rhoades, J.D. Use of saline and brackish waters for irrigation: Implications and role in increasing food production, conserving water, sustaining irrigation and controlling soil and water degradation. In *Proceedings of the International Workshop on “The Use of Saline and Brackish Waters for Irrigation: Implications for the Management of Irrigation, Drainage and Crops” at the 10th Afro-Asian Conference of the International Committee on Irrigation and Drainage*, Bali, Indonesia, Jul 23–24; Ragab, R., Pearce, G., Eds.; International Committee on Irrigation and Drainage: Bali, 1998; 261–304.
26. Rhoades, J.D.; Loveday, J. Salinity in irrigated agriculture. In *Irrigation of Agricultural Crops*; Stewart, B.A., Nielsen, D.R., Eds.; Agron. Monograph No. 30; American Society of Agronomy: Madison, 1990; 1089–1142.
27. Tanji, K.K. Nature and extent of agricultural salinity. In *Agricultural Salinity Assessment and Management*; Tanji, K.K., Ed.; ASCE Manuals and Reports on Engineering No. 71; ASCE: New York, 1990; 1–17.

Jhum: Cultivation

Arun Jyoti Nath

*Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.
and Ecology and Environmental Science, Assam University, Silchar, India*

Biplab Brahma

*Department of Ecology and Environmental Science, Assam University,
Silchar, India*

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Ashesh Kumar Das

*Department of Ecology and Environmental Science, Assam University,
Silchar, India*

Abstract

Shifting cultivation (locally jhum) is a traditional agricultural system and involves slashing of the native vegetation and burning the slash to generate nutrients *in situ* to support the crop cultivation for few years before the area is abandoned. The system is practiced in Eastern Himalayas and represents one of the predominant land uses in the mountainous region of North East India (NEI) covering an area of 1.5 million hectares (Mha). The shifting cultivation has been sustainable because of relatively longer fallow periods. However, the system is outdated and irrelevant because of the population-driven reduction in the duration of the fallow cycle (3–5 years), with the attendant degradation of soil and other natural resources. The annual loss ($\text{Mg ha}^{-1} \text{ yr}^{-1}$) of top soil, nitrogen (N), and potassium (K) under shifting cultivation area in NEI is 58.9, 7.1, and 4, respectively. Soil erosion, nutrient loss, and other ecosystem disservices exacerbated from cultivation with short fallows are jeopardizing the soil resilience and ultimately leading to poverty and food insecurity among the shifting cultivators. Even though a number of alternatives for shifting cultivation (terrace cultivation and agroforestry intervention) have been proposed and experimented upon, little progress has been made in finding viable alternatives. Therefore, this entry aims to develop a comprehensive mechanism that sustains the forest ecosystems and enhances livelihood security of the hill farmers in NEI. This entry proposes feasibility of introducing “grains for forest management program” through provision food grains to farmers as an alternative to shifting cultivation. The proposed program is also envisaged to achieve: 1) soil restoration and strengthening of ecosystem resilience; 2) food security through provisioning of grains under grains for forest management; and 3) carbon offset under climate negotiations.

INTRODUCTION

Shifting cultivation, variously termed as “slash-and-burn agriculture,” “swidden,” and “rotational bush fallow agriculture,” is called “jhum” in India. It practiced in the wet tropics and covers ~30% of the world’s uplands from Africa, Latin America, Oceania, and South and Southeast Asia.^[1] Being the oldest farming system of the world and representing the dominant land use in the mountainous regions of South and Southeast Asia, it involves clearing of mountainous land by cutting the vegetation (tree or bushes), leaving the biomass *in situ* for drying, and finally burning it for production of charred material for soil fertility enrichment. Subsequently, multiple crops (annual and perennial) are grown simultaneously on the same land until soil fertility is depleted, and farmers move to a new forest patch and repeat the cycle.^[2] In sub-Saharan Africa, this traditional

agriculture consists mainly of 1) bush fallow-food crop rotation; 2) permanent tree crop farming; 3) taungya, a form of agroforestry; and 4) permanent compound farming. The latter not only involves food crops but also plants that are known for their effectiveness in restoring soil fertility.^[3] This system embodies the agroforestry concept of combining crops and trees and ensures the dominance of effective tree species during the fallow period.^[4,5] Over the centuries, shifting agriculture has been sustainable with satisfactory yield on a long-term basis.^[2,6] The time difference between two subsequent cultivations on the same land (the jhum cycle), which was more than 20 years,^[2] is reduced to 3–5 years. Such population-driven abrupt decline in jhum cycle leads to accelerated soil erosion, nutrient loss, decline in productivity, and reduction in biodiversity and that ultimately exacerbates ecosystem disservices. Moreover, vicious cycle can lead

to irreversible degradation of soil and a disintegration of the ecosystem (under review Nath et al., 2015). Therefore, sustainability of this old-age subsistence agricultural system is questionable. Terrace cultivation and agroforestry systems were introduced in North East India (NEI) as an alternative to shifting cultivation with little success. Therefore, this entry aims to identify viable alternatives to shifting cultivation, especially those which are ecologically compatible, socially acceptable, economically feasible, and environmentally sustainable. In this entry, shifting cultivation in NEI, part of Eastern Himalaya, a biodiversity hotspot, is analyzed with respect to soil quality as an indicator of sustainability. In addition, a viable alternative for this cultivation system is proposed so that farmers can set aside the fragile ecosystem for restoration and nature conservancy.

SHIFTING CULTIVATION IN NEI

North East India (NEI) comprised of eight states (Arunachal Pradesh, Assam, Manipur, Meghalaya, Mizoram, Nagaland, Sikkim, and Tripura) and covered 26.3 million hectares (Mha) or ~8% of total geographical area of India (329 Mha). NEI is situated at the confluence of Indo-Malayan, Indo-Chinese, and Indian bio-geographical realms. Because of its geographical location, NEI

represents diverse ecosystem types dominated by mountains and is endowed with rich forest resources. There are over 100 different tribes in NEI, differing linguistically and culturally^[2] practicing shifting agriculture for millennia. This agricultural practice has evolved as a part of the culture of the hill people of the region, and the agricultural procedures are closely linked with the sociocultural practices and religious beliefs.^[2] However, increasing population pressure has drastically reduced the jhum cycle and has been the cause of poverty and food insecurity for the hill farmers and environmental degradation in the NEI landscape. Schematic presentation of short fallow cycle in practice and the consequences of this degradative process are presented in Fig. 1. Land area measuring 1.5 Mha is under shifting cultivation in NEI.^[7] An indiscriminate practice of shifting cultivation decreased forest area by ~625 km² over a short period between 2011 and 2013.^[8] During the same period, Nagaland experienced severe deforestation (274 km²) followed by that in Tripura (111 km²), Manipur (100 km²), and Arunachal Pradesh (89 km²).

Soil Erosion and Nutrient Loss

As a large proportion of the mineral nutrients in the tropical ecosystem are stored in the vegetation,^[14,15] slashing and

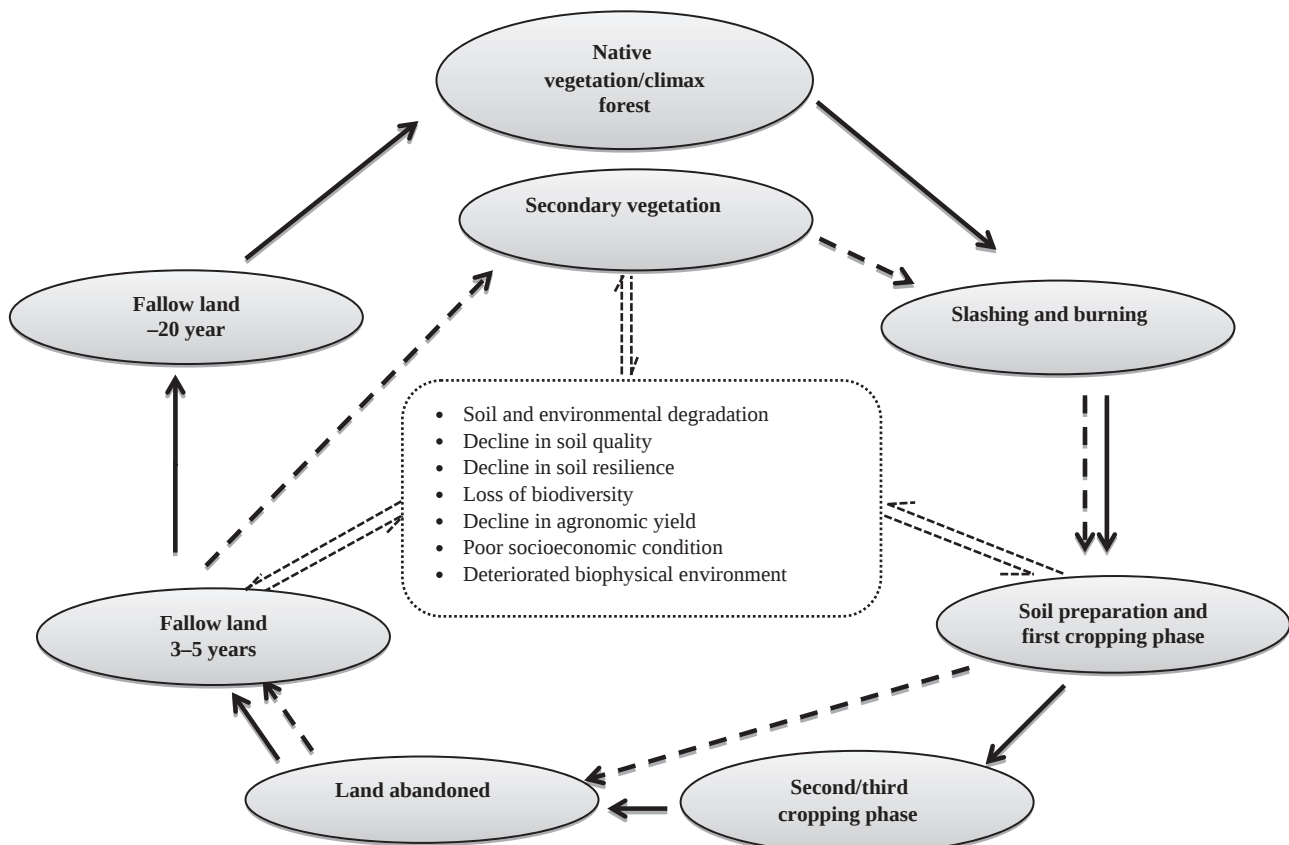


Fig. 1 Schematic presentation of shifting cultivation 1) with short fallow cycle and 2) with long fallow cycle.

burning appear to be the only way to prepare the land for cultivation to incorporate nutrients into the soil that have been accumulated in vegetation. The effects of slashing and

burning involve changes in soil physical and chemical properties and in nutrient status.^[16,17] The effects of fallow periods on soil physical and chemical properties in

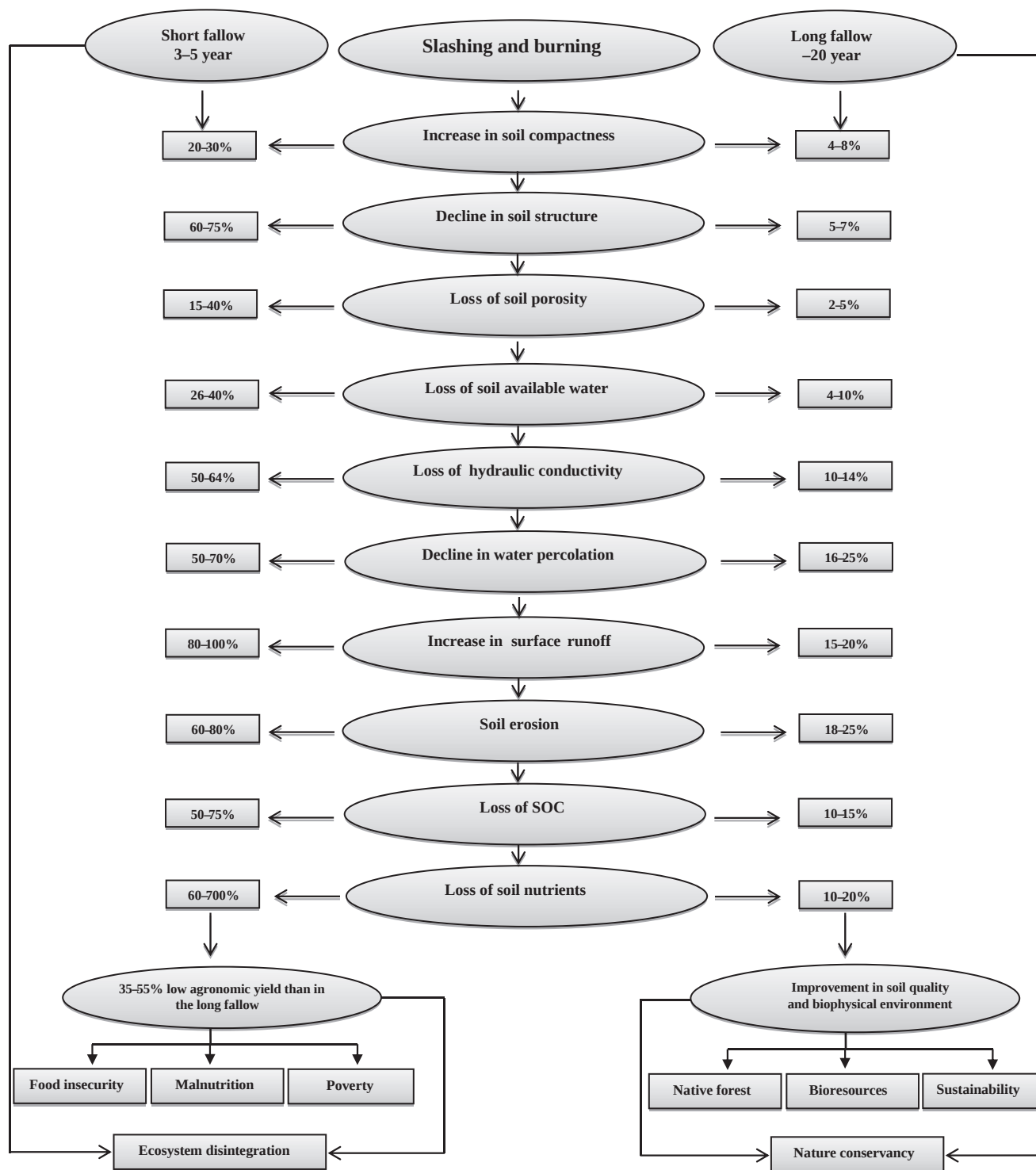


Fig. 2 Schematic representation of effect of fallow periods on soil physical and chemical properties in North East India. Values represent percentage increase or decrease of particular property than that under native climax forest.

Source: From Ramakrishnan,^[2] Ramakrishnan & Toky,^[9] Mishra & Ramakrishnan,^[10] Saha & Mishra,^[11] Grogan, Lalnunmawia, et al.,^[12] and Singh, Bordoloi, et al.^[13]

comparison with native forest in NEI are outlined in Fig. 2. Drastic changes occur in soil macro- (>250 μm) and micro-aggregates (<250 μm) and porosity because of densification under short fallow period. Soil compaction reduces water in soil (θ) and hydraulic conductivity (K_s ; Fig. 2; Table 1). Soil erosion (60–80% more) and surface runoff (80–100% more) are more under short fallow than under native forest. Therefore, vegetation slashing and burning with short fallows deteriorate soil physical properties by disintegrating soil stable aggregates and disrupting continuity of pores from surface to subsurface soil that reduces water infiltration and percolation and increases surface runoff. Soil under short fallow is prone to a high nutrient loss (60–700% more than native forest), resulting into low agronomic yield. Similar to NEI, a study from Zambia indicated decline in maize yield (*Zea mays*) under short fallows.^[32] Changes in soil properties under shifting cultivation in different parts of the tropics are presented in Table 1. These data in Table 1 indicate trends similar to those observed in NEI (Fig. 2). The magnitude of annual losses ($\text{Mg ha}^{-1} \text{ yr}^{-1}$) of top soil, N, and K under shifting cultivation area in NEI may be as high as 58.9, 7.1, and 4, respectively.^[33] Decline in soil fertility leads to decline in crop yields^[6,32,34,35] and forces cultivator to abandon the land and shift to another forest patch and begin the new cycle. Therefore, shifting cultivation with short fallow is the prime driver of poor socioeconomic status of the hill farmers and of the degraded biophysical environment. In contrast, long fallow period improves the soil quality by improving soil structure (Fig. 2). During the long fallow period, changes in soil structure are closely linked with increase in soil organic matter (SOM) and soil humus contents.^[6,15] In NEI, litterfall increased from 1.2 $\text{Mg ha}^{-1} \text{ yr}^{-1}$ during the first-year fallow to 9.7 $\text{Mg ha}^{-1} \text{ yr}^{-1}$ in the 20-year-old fallow,^[36] almost equivalent to the mature forests,^[37] implying a critical role of biomass accumulation in long fallows to soil restoration. In case of older fallow, organic inputs through above- and belowground litterfall and development of macro- and microfaunal population^[38] accelerate the decomposition process and subsequently improve soil quality, and the system slowly approaches to the level of climax forest.

Soil Organic Carbon (SOC) and Shifting Cultivation

The SOC derived from the partially combusted biomass is the basis for maintenance of productivity of soil under shifting cultivation. Nutrients derived from biomass burning are stored in organic form and held in the mineral form on the exchange sites of SOC.^[39,29] In the absence of vegetation cover in the first and second year of cropping phase, SOC generated remains prone to the accelerated erosion. A decrease of 14–20% of SOC in

Table 1 Changes in soil parameter under shifting cultivation.

Parameter	Site	Native forest	Cropping phase/ short fallow period	References
Physical properties				
BD (g cm^{-3})	Bangladesh	1.16	1.23	[18]
	Amazon Basin	1.05	1.19	[19]
	Honduras	0.82	1.18	[20]
	Nigeria	0.70	1.44	[21]
Pore space (%)	Honduras	58.0	42.0	[20]
Infiltration rate (cm hr^{-1})	Nigeria	152	21	[21]
Hydraulic conductivity (cm hr^{-1})	Nigeria	308	46	[21]
Runoff and erosion				
Water runoff (mm)	Nigeria	3.0	31.0	[22]
	Brazil	16.0	35.0	[23]
Soil erosion ($\text{Mg ha}^{-1} \text{ yr}^{-1}$)	Nigeria	0.2	5.9	[22]
	Laos	0.3	30.0	[24]
	Thailand	3.0	11.7	[25]
	Brazil	0.10	6.0	[23]
Chemical properties				
SOC (%)	Bangladesh	3.01	2.88	[18]
	Cameroon	2.60	2.30	[26]
SOC (g kg^{-1})	Madagascar	64.1	40.3	[27]
	China	34.8	23.9	[28]
	Amazon Basin	27.6	14.9	[19]
	Ecuador	48.0	27.0	[29]
CO ₂ evolution ($\text{g C m}^{-2} \text{ day}^{-1}$)	Costa Rica	2.52	4.51	[30]
Total N (g kg^{-1})	Bangladesh	4.18	2.91	[18]
	Madagascar	4.7	1.7	[27]
	Costa Rica	7.0	5.5	[31]
Average P (Mg kg^{-1})	Madagascar	4.9	4.6	[27]

top soil during the cropping phase of shifting cultivation has been reported,^[29] and it may take up 35 years to regain the content comparable to that under a primary forest.^[39] The oxidative loss of SOC during biomass burning releases CO₂ and, therefore, has positive feedback on global climate change. However, fallow systems apparently regain the SOC content (Fig. 2). A study from Madagascar showed that a longer fallow length supported more mature soil with high

SOC content.^[40] In a Rio Negro watershed, an increase in SOC by 18% was observed after 5 years of fallow period.^[41] Therefore, accretions of SOC with increase in fallow periods are the result of organic matter input from the above- and belowground litter deposited by fallow vegetation.

Proposed Alternative Program for Shifting Cultivation in NEI

Synthesis of data on soil erosion, loss of nutrients, and SOC reveals that a short fallow cannot fulfill the livelihood requirements of the increasing hill populace due to

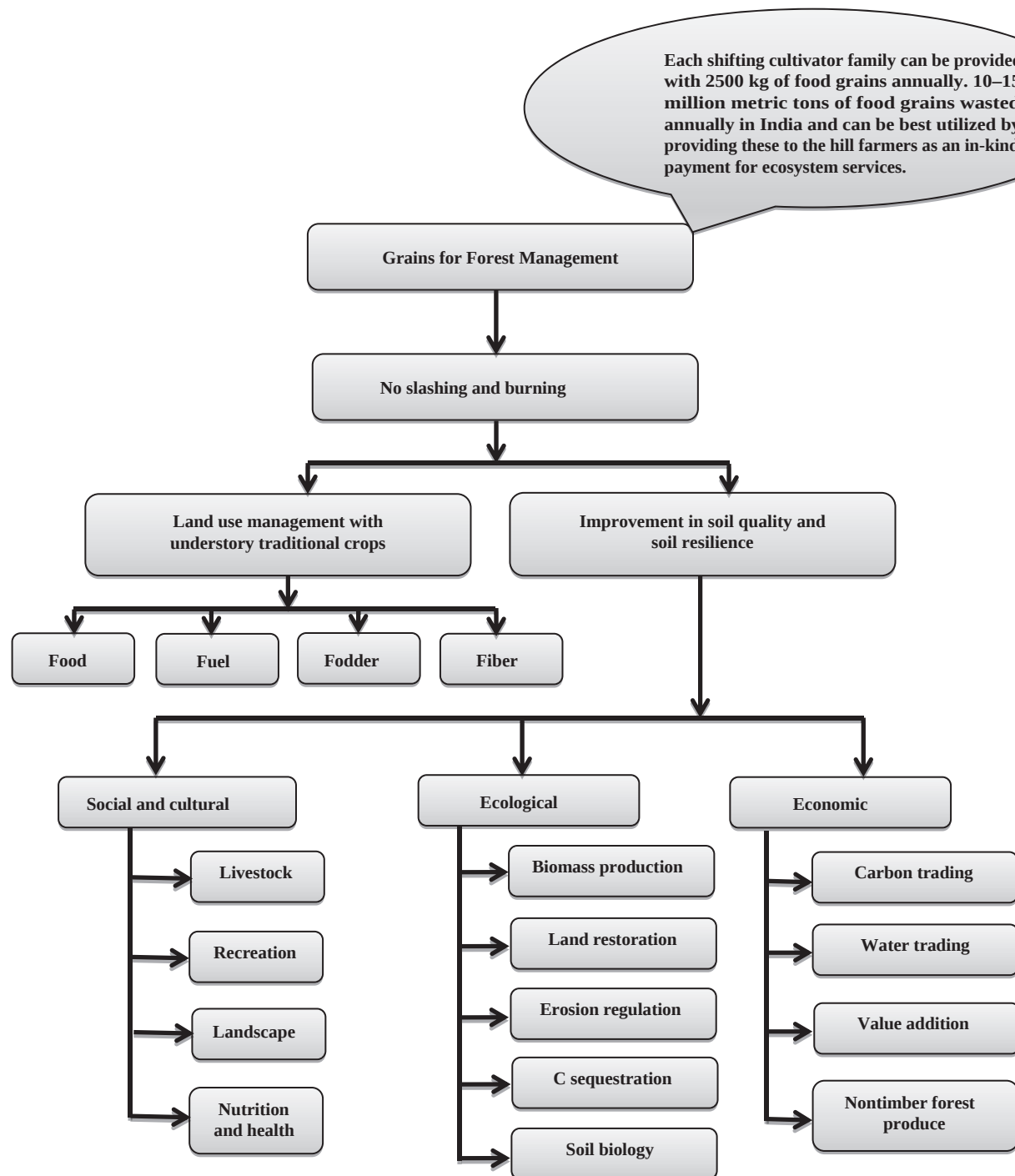


Fig. 3 Proposed viable alternative for shifting cultivation to set-aside the fragile ecosystem for restoration and nature conservancy in North East India.

severe decline in agronomic yield. Poverty and unavailability of alternate sources of gainful employment drive hill agricultural communities toward unsustainable utilization of even marginal lands. The effect is loss of the soil resilience, leading to the unsustainability of the shifting agricultural system. Sustainability of the fragile ecoregion can be managed through maintaining longer fallow periods, but this also seems to be unachievable due to rise in population and their subsequent livelihood requirement in the form of fuel, food, and fodder. Further, maximizing yield from short-term fallow is not a viable option considering soil degradation and its attendant socioenvironmental problems. Thus, there is a need to develop a comprehensive plan so that forest can be managed on long-term basis and, at the same time, the livelihood requirement of hill farmers is fulfilled. In this context, we propose the introduction of “grains for forest management program” as an alternative to shifting cultivation, where farmers can be provided with food grains that they otherwise grow by shifting cultivation (Fig. 3). Rather than a subsidy, food grains provided are an in-kind payment for ecosystem services (PES) for restoration of degraded lands. India has made commendable success in food grain production with an annual production rate of 264 million metric ton.^[42] However, 10–20% of the total production is wasted annually due to lack of adequate storage facility.^[43] Therefore, implementing “grains for forest management program” in NEI and provisioning food grains are appropriate use of surplus food grains, while restoring degraded soils, strengthening ecosystem services, and earning C credits through PES from soil and ecosystem C sequestration. The establishment of such program will also enable the hill farmers to pursue their traditional semiperennial and perennial crop cultivation understorey of the restoring vegetation. This program will fulfill their food, fodder, and other daily life requirements. Considering the soil C sequestration rate of 0.61–0.64 Mg ha⁻¹ yr⁻¹ with conversion of shifting cultivated area to a fallow land in India,^[9,44] total C sequestration in 1.5 Mha of jhum land in NEI should be around 1 Tg yr⁻¹. Lal^[45] reported the monetary equivalent of inherent cost or societal value of SOC is US\$ 0.13 kg⁻¹. Therefore, societal value of 1 Tg of C sequestered is around US\$ 130 million yr⁻¹. Assuming total number of shifting cultivators in NEI at 443,000,^[46] each family can earn US\$ 294 yr⁻¹. Therefore, the proposed “grains for forest management program” will help in improving the socioeconomic status of the hill farmers practicing the subsistence system of shifting cultivation.

CONCLUSION

Considering the socioeconomic condition of hill farmers, increasing population pressure and decreasing dense

forest cover in NEI proposed “grains for forest management program” may be a viable strategy to restore degraded fragile ecoregion and enhance livelihood security. Implementation of the program will also strengthen the ecosystem resilience of this biodiversity hotspot to offer critical ecosystem services. In addition to SOC, C sequestered in the biomass in successional fallows of restored shifting cultivation landscape can bring additional income to the land managers of NEI in the form of offsetting C credits under climate negotiations. Moreover, the implementation of this program will also lead to achieve United Nations proposed program of reducing the rate of land degradation to achieve land degradation neutrality or zero net land degradation.^[47]

ACKNOWLEDGMENT

Senior author greatly acknowledges the research fellowship granted by the Department of Biotechnology, Government of India in the form of Overseas Associateship.

REFERENCES

1. FAO. *Global Forest Resources Assessment 2010*, FAO Forestry Paper No. 163; FAO: Rome, 2010.
2. Ramakrishnan, P.S. *Shifting Agriculture and Sustainable Development: An Interdisciplinary Study from North-Eastern India*, Man and the Biosphere Series 10; Parthenon Publications: Park Ridge, 1992; 424 pp.
3. Okigbo, B.N.; Lal, R. Soil fertility maintenance and conservation for improved agroforestry systems in the lowland humid tropics. In *Soil Research in Agroforestry*; Mongi, H.O., Huxley, P.A., Eds.; International Council for Research in Agroforestry: Nairobi, 1979; 41–77.
4. MacDonald, L.H., Ed. *Agroforestry in the African Humid Tropics*; United Nations University Press: Tokyo, 1982.
5. Benites, V.M.; Moutta, R.O.; Coutinho, H.L.; Balieiro, F.C. Discriminant analysis of soils under different land uses in the Atlantic rain forest area using organic matter attributes. *Rev. Arvore* **2010**, *34*, 685–690.
6. Ziegler, A.D.; Bruun, T.B.; Guardiola-Claramonte, M.; Giambelluca, T.W.; Lawrence, D.; Lam, N.T. Environmental consequences of the demise in swidden cultivation in montane mainland Southeast Asia: Hydrology and geomorphology. *Human Ecol.* **2009**, *37*, 361–373.
7. Roy, P.S.; Kushwaha, S.P.S.; Mirthy, M.S.R.; Roy, A.; Kushwaha, D.; Reddy, C.S.; Behera, M.D.; Mathur, V.B.; Padalia, H.; Saran, S.; Singh, S.; Jha, C.S.; Porwal, M.C. *Biodiversity Characterization at Landscape Level: National Assessment*; Indian Institute of Remote Sensing: Dehradun, 2012; 140 pp.
8. Forest Survey of India (FSI). *India State of Forest Report*; FSI: Dehradun, 2013.
9. Ramakrishnan, P.S.; Toky, O.P. Soil nutrient status of hill agro-ecosystems and recovery pattern after slash and burn agriculture (jhum) in north-eastern India. *Plant Soil* **1981**, *60*, 41–64.

10. Mishra, B.K.; Ramakrishnan, P. Slash and burn agriculture at higher elevations in north-eastern India: II. Soil fertility changes. *Agr. Ecosyst. Environ.* **1983**, *9*, 83–96.
11. Saha, R.; Mishra, V.K. Long-term effect of various land use systems on physical properties of silty clay loam soil of N-E hills. *J. Indian Soc. Soil Sci.* **2007**, *55*, 112–118.
12. Grogan, P.; Lalnunmawia, F.; Tripathi, S.K. Shifting cultivation in steeply sloped regions: A review of management options and research priorities for Mizoram state, Northeast India. *Agrofor. Syst.* **2012**, *84*, 163–177.
13. Singh, A.K.; Bordoloi, L.J.; Kumar, M.; Hazarika, S.; Parmar, B. Land use impact on soil quality in eastern Himalayan region of India. *Environ. Monit. Assess.* **2014**, *186*, 2013–2024.
14. Richards, P.W. *The Tropical Rain Forest*; Cambridge University Press: Cambridge, 1952; 450.
15. Nye, P.H.; Greenland, D.J. *The Soil under Shifting Cultivation*; Commonwealth Bureau of Soils: Harpenden, 1960; 156 pp.
16. Christanty, L. Shifting cultivation and tropical soils: Patterns, problems, and possible improvements. In *Traditional Agriculture in Southeast Asia: A Human Perspective*; Marten, G.G., Ed.; Westview Press: Boulder, 1986; 226–240.
17. Khresat, S.; Al-Bakri, J.; Al-Tahhan, R. Impacts of land use/cover change on soil properties in the Mediterranean region of northwestern Jordan. *Land Degrad. Dev.* **2008**, *19*, 397–407.
18. Osman, K.S.; Jashimuddin, M.; Haque, S.M.S.; Miah, S. Effect of shifting cultivation on soil physical and chemical properties in Bandarban hill district, Bangladesh. *J. For. Res.* **2013**, *24*, 791–795.
19. McGrath, D.A.; Smith, C.K.; Gholz, H.L.; Assis Oliveira, A. Effects of land-use change on soil nutrient dynamics in Amazonia. *Ecosystems* **2001**, *4*, 625–645.
20. Paniagua, A.; Kammerbauer, J.; Avedillo, M.; Andrews, A.M. Relationship of soil characteristics to vegetation successions on a sequence of degraded and rehabilitated soils in Honduras. *Agr. Ecosyst. Environ.* **1999**, *72*, 215–225.
21. Lal, R. Deforestation and land-use effects on soil degradation and rehabilitation in western Nigeria: 1. Soil physical and hydrological properties. *Land Degrad. Dev.* **1996**, *7*, 19–45.
22. Lal, R. Deforestation and land-use effects on soil degradation and rehabilitation in western Nigeria: 3. Runoff, soil erosion and nutrient loss. *Land Degrad. Dev.* **1996**, *7*, 99–119.
23. Thomaz, E.L. Slash-and-burn agriculture: Establishing scenarios of runoff and soil loss for a five-year cycle. *Agr. Ecosyst. Environ.* **2013**, *168*, 1–6.
24. Roder, W.; Phengchanh, S.; Keoboulapha, B. Relationships between soil, fallow period, weeds and rice yield in slash-and-burn systems of Laos. *Plant Soil* **1995**, *176*, 27–36.
25. Valentin, C.; Agus, F.; Alamban, R.; Boosaner, A.; Bricquet, J.P.; Chaplot, V.; de Guzman, T.; de Rouw, A.; Janeau, J.L.; Orange, D.; Phachomphonh, K.; Phai, D.D.; Podwojewski, P.; Ribolzi, O.; Silvera, N.; Subagyono, K.; Thiebaut, J.-P.; Toan, T.D.; Vadari, T. Runoff and sediment losses from 27 upland catchments in Southeast Asia: Impact of rapid land use changes and conservation practices. *Agr. Ecosyst. Environ.* **2008**, *128*, 225–238.
26. Njomgang, R.; Yemefack, M.; Nounamo, L.; Moukam, A.; Kotto-Same, J. Dynamics of shifting agricultural systems and organic carbon sequestration in southern Cameroon. *Tropicultura* **2011**, *29*, 176–182.
27. Vagen, T.; Andrianorofanomezana, M.A.; Andrianorofanomezana, S. Deforestation and cultivation effects on characteristics of oxisols in the highlands of Madagascar. *Geoderma* **2006**, *131*, 190–200.
28. Yang, J.C.; Huang, J.H.; Pan, Q.M.; Tang, J.W.; Han, X.G. Long-term impacts of land-use change on dynamics of tropical soil carbon and nitrogen pools. *J. Environ. Sci.* **2004**, *16*, 256–261.
29. Bahr, E.; Zaragocin, D.C.; Makeschin, F. Soil nutrient stock dynamics and land-use management of annuals, perennials and pastures after slash-and-burn in the Southern Ecuadorian Andes. *Agr. Ecosyst. Environ.* **2014**, *188*, 275–288.
30. Ewel, J.; Berish, C.; Brown, B.; Price, N.; Raich, J. Slash and burn impacts on a Costarican wet forest site. *Ecology* **1981**, *62*, 816–829.
31. Guggenberger, G.; Zech, W. Soil organic matter composition under primary forest, pasture, and secondary forest succession. Region Huatar Norte, Costa Rica. *For. Ecol. Manag.* **1999**, *124*, 93–104.
32. Ando, K.; Shinjo, H.; Noro, Y.; Takenaka, S.; Miura, R.; Sokotela, S.B.; Funakawa, S. Short-term effects of fire intensity on soil organic matter and nutrient release after slash- and-burn in Eastern Province, Zambia. *Soil Sci. Plant Nutr.* **2014**, *60*, 173–182.
33. Sharma, U.C. Methods of selecting suitable land use system with reference to shifting cultivation in NEH region. *Indian J. Soil Conserv.* **1998**, *26*, 234–238.
34. Tinker, P.B.; Ingram, J.S.I.; Struwe, S. Effects of slash-and-burn agriculture and deforestation on climate change. *Agr. Ecosyst. Environ.* **1996**, *58*, 13–22.
35. Gafur, A.; Jenson, J.R.; Borggard, O.K.; Peterson, L. Runoff and losses of soil and nutrients from small watersheds under shifting cultivation (Jhum) in the Chittagong hill tracts of Bangladesh. *J. Hydrol.* **2003**, *279*, 293–309.
36. Toky, O.P.; Ramakrishnan, P.S. Secondary succession following slash and burn agriculture in north-eastern India: II. Nutrient cycling. *J. Ecol.* **1983**, *71*, 747–757.
37. Swamy, S.L.; Dutt, C.B.S.; Murthy, M.S.R.; Mishra, A.; Bargali, S.S. Floristics and dry matter dynamics of tropical wet evergreen forests of Western Ghats, India. *Curr. Sci.* **2010**, *99*, 353–364.
38. Reza, S.K.; Baruah, U.; Nath, D.J.; Sarkar, D.; Gogoi, D. Microbial biomass and enzyme activity in relation to shifting cultivation and horticultural practices in humid subtropical North-Eastern India. *Range Manag. Agrofor.* **2014**, *35*, 78–84.
39. Palm, A.A.; Swift, M.J.; Wooster, P.L. Soil biological dynamics in slash-and-burn agriculture. *Agr. Ecosyst. Environ.* **1996**, *58*, 61–74.
40. Raharimalala, O.; Buttler, A.; Ramohavelo, C.D.; Razanaka, S.; Sorg, J.; Gobat, J. Soil vegetation patterns in secondary slash and burn successions in Central Menabe, Madagascar. *Agr. Ecosyst. Environ.* **2010**, *139*, 150–158.
41. Delang, C.O.; Li, W.M., Eds. *Ecological Succession on Fallowed Shifting Cultivation Fields*; Springer: Dordrecht, Heidelberg, New York, London, 2013; 126.

42. Department of Agriculture and Cooperation, Ministry of Agriculture Annual Report 2013–14; Government of India: New Delhi, 2014.
43. Nellemann, C.; MacDevette, M.; Manders, T.; Eickhout, B.; Svihus, B.; Prins, A.G.; Kaltenborn, B.P., Eds. *The Environmental Food Crisis – The Environment's Role Inverting Future Food Crises, A UNEP Rapid Response Assessment*; United Nations Environment Programme, GRID-Arendal: Norway, 2009.
44. Lenka, N.K.; Choudhury, P.R.; Sudhishri, S.; Dass, A.; Patnaik, U.S. Soil aggregation, carbon build up and root zone soil moisture in degraded sloping lands under selected agroforestry based rehabilitation systems in eastern India. *Agr. Ecosyst. Environ.* **2012**, *150*, 54–62.
45. Lal, R. Societal value of soil carbon. *J. Soil Water Conserv.* **2014**, *69*, 186–192.
46. Kumar, S. *Report of the Inter-Ministerial National Task Force on Rehabilitation of Shifting Cultivation Areas*; Government of India: New Delhi, 2008.
47. United Nations Convention to Combat Desertification (UNCCD). *Zero Net Land Degradation: A Sustainable Development Goal for Rio+20*; UNCCD: Bonn, 2012.

Jhum: Shifting Agriculture

P.S. Ramakrishnan

School of Environmental Sciences, Jawaharlal Nehru University, New Delhi, India

Abstract

The forest farmer in the tropics has managed the traditional shifting agriculture (*slash-and-burn agriculture*, locally known in India as *jhum*), which is essentially an agroforestry system organized both in space and time for centuries. Shifting agriculture is largely confined to the humid tropics of Asia, Africa, and South America, with highly variable Oxisols, Ultisols, Inceptisols, and Entisols, where the major soil constraint is the presence of some toxic chemicals with low nutrient reserves. Formal ecological knowledge derived through the hypothetico-deductive method—validated traditional ecological knowledge derived largely through societal experiences and perceptions accumulated by traditional societies during their interaction with nature and natural resources—has a strong human element attached to it.

INTRODUCTION

The forest farmer in the tropics has managed the traditional *shifting agriculture* (*slash-and-burn agriculture*, locally known in India as *jhum*), which is essentially an agroforestry system organized both in space and time for centuries. The small-scale perturbations, in the past, ensured enhanced *biological diversity* in the forest, with enriched crop and associated biodiversity, capitalizing on the nutrient released through slash-and-burn. With increasing pressure on forest resources from outside and on population pressure from within, and the consequent declining soil fertility through land degradation, *agricultural cycle* has shortened. It is suggested^[1] that, in the late 1980s, about 500 million people were dependent on shifting agriculture in 90 countries, covering an area of approximately 400 million hectares of tropical forest land area (Table 1). A subsequent forest resource assessment^[2] found that more than 7% of the 1980 forest area underwent change during the period 1980–1990, with more than half of this change due to shifting agriculture, resulting in moderate to severe degradation.

The net consequence is drastic reduction in shifting agricultural cycle, leading to: 1) drastic yield reduction; 2) reduced system stability and resilience, leading to social disruption; 3) biodiversity decline because of weed takeover, biological invasion, and/or eventual site desertification; and 4) substantial carbon dioxide emitted into the atmosphere. All attempts made so far in finding an alternate to shifting cultivation, for want of a holistic approach in dealing with the complex issues, have had little or no impact on the farmer.^[3] It is in this context that an evaluation of ecological impacts and the finding of an acceptable but sustainable solution(s) to the problem become critical.

SOIL FERTILITY AND NUTRIENT BUDGET UNDER SHIFTING AGRICULTURE

Shifting agriculture is largely confined to the humid tropics of Asia, Africa, and South America, with highly variable Oxisols, Ultisols, Inceptisols, and Entisols, where the major soil constraint is the presence of some toxic chemicals with low nutrient reserves.^[4] The complex tropical rain forests are often extremely fragile. First, these forests, which have developed over centuries, often grow on highly infertile soil, with biomass as the chief storage compartment for nutrients. Second, in oligotrophic areas, stability is ensured by the presence of a thick surface root mat, which picks up nutrients released from decaying leaf litter on the mineral soil and pumps these nutrients back into the living biomass before they have the chance to enter the mineral soil. Frequent and large-scale perturbations, as are occurring now, upset this delicate balance in nutrient cycling.

In general, nitrogen (N) and potassium (K), being very labile, are often limiting under many situations. Phosphorus (P) deficiency is a widely reported soil constraint in tropical America, which is further compounded by high P fixation related to soil acidity. There are also other problems related to soil acidity—low availability of calcium (Ca) and magnesium (Mg), often times with aluminum (Al) and manganese (Mn) toxicities.

After clear-cutting and burning of the forest, the ecosystem loses its ability to hold nutrients. Losses occur through the volatilization of carbon (C) and N during the burn. Substantial nutrient losses through wind blowing of ash, runoff, and leaching through water may all occur before adequate vegetal cover builds up, during both the cropping and fallow phases. During the cropping phase, losses also occur through the uptake and removal of

Table 1 Forest fallows (thousands of hectares) under shifting agriculture in the late 1980s.

Region	Closed forest fallows	Open forest fallows
South America and the Pacific	108,600	61,650
Africa	61,700	104,350
Asia	69,250	4,000
Tropical world (total)	239,550	170,000

Source: From FAO/UNEP.^[1]

nutrients through biomass that gets harvested. A reduced cycle length (5 years or less) in many parts of the world has led to a drastic decline in soil fertility under short agricultural cycles imposed on the same site over a period of time (Table 1). Largely herbaceous vegetation that develops under very short cycles of 5 years or so does not help in adequate regeneration of the lost soil fertility.

The rapid regeneration of forest vegetation following clearing and burning reduces nutrient loss and allows a return to the steady state cycling characteristics of mature forests. Although farmers deal with the decline in soil fertility in different ways, a minimum of 10–15 years is required for fallow regrowth in order to recoup most of the soil fertility lost during the cropping phase (Fig. 1). The C sequestration in the system depends largely on the cycle length of these systems,^[1] as also observed in the Indonesian case study.^[5] The N budgeting under different cycle lengths of 15, 10, and 5 years is illustrative of the kind of issues involved (Table 2). During one cropping phase, the agroecosystem loses about 600 kg ha⁻¹ N (the difference between the soil N capital before and after one cropping). With the plot under a 5-year cycle having the same cycle length during the last 20 years, this system had lost 1.28×10^3 kg ha⁻¹ N from its initial capital of $7.68\text{--}6.40 \times 10^3$ kg ha⁻¹. While 10- and 15-year agricultural cycles are long enough to restore the original N status in the soil before the next cropping, it seems unlikely that the 600 kg ha⁻¹ N lost during one cropping could be restored under a 5-year cycle—an observation similar to other nutrients, too,

Table 2 Net change of N (10³ kg ha⁻¹ yr⁻¹) in the soil under jhum in northeast India.

	15-year fallow cycle	10-year fallow cycle	5-year fallow cycle	
			First year crop	Second year crop
Soil pool before burning	7.68	7.74	6.40	5.98
Soil pool at the end of cropping	7.04	7.15	5.98	5.60
Net difference	0.64	0.59	0.42	0.38

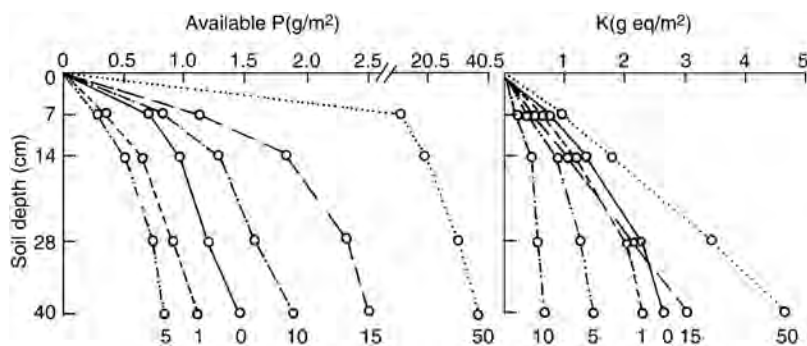
Source: From Ramakrishnan.^[3]

such as K. The increased frequency of fire and cropping, with too short a fallow phase, thus results in rapid site degradation.

The first step in site degradation is the replacement of forests by an arrested weed stage. Large tracts of forested lands in the Asian tropics, e.g., are taken over by the grass *Imperata cylindrical* (thatching grass, locally known in the Asian region as “alang-alang”). Exotic weed invasion is yet another major consequence of frequent perturbation under short agricultural cycles in many parts of the world, with consequences for ecosystem processes.^[6] In extreme cases, the end result is a bald, totally desertified landscape^[1] (e.g., northeastern India). The consequence of this is less land area of shifting agriculture and a further shortening of the cycle. Therefore, the success of shifting agriculture is, to a large extent, related to nutrient cycling patterns and processes of the forest fallow phase.

KNOWLEDGE SYSTEMS FOR LAND USE MANAGEMENT

“Formal” ecological knowledge derived through the hypothetico-deductive method—validated “traditional ecological knowledge” (TEK) derived largely through societal experiences and perceptions accumulated by traditional societies during their interaction with nature and natural resources—has a strong human element attached to it. This knowledge needs to be effectively integrated

**Fig. 1** Changes in cumulative quantity of available P and K within a soil column of 40 cm depth under 0-, 1-, 5-, 10-, 15-, and 50-year-old jhum fallows (e.g., g eq/m²).

to ensure the participatory land use development of traditional societies.^[7] Linking ecological processes with social processes is the key issue here.

Thus, e.g., the concept of ecological keystone, an end product of a social selection process, is illustrative of the linkages that exist between the traditional and the formal knowledge systems. Thus, in many areas in northeast India where the landscape is highly degraded, a legume crop of lesser known food value, *Flemingia vestita* (locally known in Meghalaya, India, as “Soh-phlong,” yielding up to 3000 kg of edible juicy tuber), which is socially valued, is used both in space and in time under a 2- to 5-year rotational fallow system. By fixing 250 kg ha⁻¹ yr⁻¹ N, this keystone species ensures the sustainability of these low-input agroecosystems, under conditions of extreme pressure on land under low soil fertility. Using organic residues and manipulating soil biodiversity through socially valued and ecologically important keystone earthworm species, sustainable fertility management has been made possible. It is a patented technology that offers opportunities^[8] for effective fallow management practices.

Nepalese alder (*Alnus nepalensis*), a socially valued species, is another N-fixing tree conserved by traditional societies of northeast India in their shifting agricultural plots. It happens to be a socially significant keystone species. This early successional tree species in the northeastern hill region is conserved by the shifting agricultural farmer, during both the cropping and fallow phases. With its roots nodulated by *Frankia*, this species can fix up to about 125 kg ha⁻¹ yr⁻¹ N and has the potential to recover all the 600 kg of N lost from the system over a 5-year cycle period. It would, otherwise, take a minimum of 10 years of natural fallow regrowth to recover all these N back into the system.

In other words, there is a close connection between the ecological and social dimensions of this validated TEK, and there are implications for fallow management through “incremental pathway” and/or “contour pathway” for agroecosystem redevelopment.^[3,9] Adapting this knowledge to modern scientific inputs is important for community participation in soil fertility management and acceleration of the developmental process itself.

SHIFTING AGRICULTURE AND SUSTAINABILITY

Realizing that sustainable soil fertility management is the key issue for finding a solution to the vexed problem of shifting agriculture-affected areas, two different approaches are possible. If the society under consideration is more “traditional,” an *incremental pathway* is more appropriate to be able to relate to their value system. For others who are less traditional and are ready to make a more drastic departure, the *contour pathway* could be the solution.

Incremental Pathway

Building on traditional technology in an incremental fashion is one of the options for shifting agriculture, at least in the short term. Management of forest fallows^[3] or grass fallows^[10] seems to be an attractive cost-effective solution to the problem, as has been suggested through many studies. Such an approach forms the basis for a major initiative, which aims at redeveloping shifting agriculture in Nagaland in northeast India,^[11] through a participatory process of fallow management in over 5500 replicated test plots in farmers’ fields spread across 1200 villages. The key to the management lies in appropriately constructed village-level institutions that are based on the local value system. In parts of South America (Venezuela), natural secondary forest succession has been suggested as a model to replace traditional shifting agriculture: bean (*Phaseolus vulgaris*), corn (*Zea mays*), sugarcane (*Saccharum officinarum*), and pineapple (*Ananas comosus*) in the first year; followed by woody yucca (*Manihot esculenta*), cashew (*Anacardium occidentale*), or papaya (*Carica papaya*); followed by larger trees such as Brazil nut (*Bertholletia excelsa*) and jackfruit (*Artocarpus* sp.). At another level, the home garden concept could be the model for developing a plantation economy (found in many Asian and Latin American countries).

Contour Pathway

As a possible medium strategy, this type of management acknowledges and works with the ecological forces that provide the base on which the system must be built, while acknowledging the social, economic, and cultural requirements of the farming communities. Working with nature instead of dominating it, this approach seeks active planning, keeping in mind the nature of the background ecosystem. Slope management has long been a major element of farming in the Western Pacific region and in the uplands of the Asian tropics. The Sloping Agricultural Land Technology (SALT) developed by the Mindanao Baptist Rural Life Center in the early 1980s in the southern part of the Philippines is based on planting field and perennial crops in 3- to 5-m bands between double rows of N-fixing trees and shrubs planted on contours for soil conservation.^[12] The crop species and the tree/shrub species could vary. Although SALT technology has been tested successfully in many countries in the Asian tropics, socioeconomic and cultural problems stand in the way of its large-scale acceptance.

CONCLUSION

Involving local communities in managing forestry and non-timber forest product-related activities for cash economy is important, as shifting agriculture cannot be viewed in

isolation from forestry and forest-related activities of local communities. In the ultimate analysis, as a long-term strategy, it may be desirable to have a mosaic of agroecosystem types using all of the pathways just mentioned, along with intensive modern agricultural systems, coexisting with natural ecosystem types, managed or unmanaged. The maintenance of the overall sustainability of the system requires a patchwork mosaic that would, albeit inadvertently, be the best plan for effectively managing natural resources of the landscape, in the shifting agriculture-affected areas.

REFERENCES

1. FAO/UNEP. *Tropical Forest Resources (by Jean-Paul Lanley)*; Forestry Paper 50; Food and Agriculture Organization (FAO): Rome, 1982.
2. FAO. *Forest Resources Assessment 1990: Global Synthesis*; Forestry Paper 124; Food and Agriculture Organization (FAO): Rome, 1995.
3. Ramakrishnan, P.S. *Shifting Agriculture and Sustainable Development: An Interdisciplinary Study from North-Eastern India*; UNESCO-MAB Series, Paris; Parthenon Publishers: Carnforth, 1992.
4. Sanchez, P.A. Soils. In *Tropical Rain Forest Ecosystems*; Leith, H., Werger, M.J.A., Eds.; Elsevier: Amsterdam, 1989; 73–88.
5. Tomich, T.P.; van Noordwijk, M.; Budidaresono, S.; Gillison, A.; Kusumanto, T.; Murdiyarso, D.; Stolle, F.; Fagi, A.M. *Alternatives to Slash-and-Burn in Indonesia: Summary Report and Synthesis of Phase II*; ASB Indonesia and ICRAF-S.E. Asia: Bogor, 1998.
6. Ramakrishnan, P.S.; Vitousak, P.M. Ecosystem-level processes and consequences of biological invasions. In *Biological Invasion: A Global Perspective*; Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Rejmanek, M., Williamson, M., Eds.; SCOPE 37; John Wiley: New York, 1989; 281–300.
7. Ramakrishnan, P.S. *Ecology and Sustainable Development*; National Book Trust: New Delhi, 2001.
8. Senapati, B.K.; Naik, S.; Lavelle, P.; Ramakrishnan, P.S. Earthworm-based technology application for status assessment and management of traditional agroforestry systems. In *Traditional Ecological Knowledge for Managing Biosphere Reserves in South and Central Asia*; Ramakrishnan, P.S., Rai, R.K., Katwal, R.P.S., Mehndiratta, S., Eds.; UNESCO and Oxford IBH: New Delhi, 2002; 139–160.
9. Swift, M.J.; Vandermeer, J.; Ramakrishnan, P.S.; Anderson, J.M.; Ong, C.K.; Hawkins, B. Biodiversity and agroecosystem function. In *Functional Roles of Biodiversity: A Global Perspective*; Mooney, H.A., Cushman, J.H., Medina, E., Sala, O.E., Schulze, E.D., Eds.; SCOPE Series; John Wiley: Chichester, 1996; 261–298.
10. Lal, R.; Wilson, G.F.; Okigbo, B.N. Changes in properties of an alfisol produced by various cover crops. *Soil Sci.* **1979**, *127*, 377–382.
11. NEPED; IRRR. *Building Upon Traditional Agriculture in Nagaland*; Nagaland Environmental Protection and Economic Development, Nagaland India and International Institute of Rural Reconstruction: Philippines, 1999.
12. Pratap, T.; Watson, H.R. *Sloping Agricultural Land Technology (SALT): A Regenerative Option for Sustainable Mountain Farming*; International Centre for Integrated Mountain Development: Khatmandu, 1994.

Land Capability Analysis

Michael J. Singer

*Department of Land, Air, and Water Resources, University of California—Davis,
Davis, California, U.S.A.*

Abstract

The amount of land available for the production of food, fiber, and other necessary products for human sustainability is limited. Land capability systems are designed to help determine appropriate uses and management of available land. Land rating systems for agriculture include those that evaluate the potential for bringing new lands into production and others that evaluate the production potential and conservation needs for those already in production. Additional land capability systems rate lands for non-agricultural uses.

INTRODUCTION

All soils are not the same, and they are not of the same capability for every use. Because the earth's human population exceeds 7 billion on its way to 9 or 10 billion by the mid-21st century, the proper use of land becomes a critical component in human sustainability. Land capability implies that the choice of land for a particular use contributes to the success or failure of that use. It further implies that the choice of land for a particular use will determine the potential impact of that use on surrounding resources such as air and water. To make the best use of land and to minimize the potential for negative impacts on surrounding lands, land capability analysis is needed. The assessment of land performance for specific purposes is land evaluation.^[1] A system that organizes soil and landscape properties into a form that helps differentiate among useful and less useful soils for a purpose is land capability classification (LCC). Land capability is a broader concept than soil quality, which has been defined as the degree of fitness of a soil for a specific use.^[2] Bouma^[3] points out that land capability or land potential needs to be evaluated via various scales and gives the example of precision agriculture, which requires land capability analysis in more detail than that required to determine if an investment should be made to initiate agriculture.

Land capability or suitability classification systems have been designed to rate land and soil characteristics for specific uses (Table 1). Huddleston^[7] has reviewed many of these systems. They may also rate land qualities. Land qualities have been defined by the U.N. Food and Agricultural Organization (FAO)^[1] as “attributes of land that act in a distinct manner in their influence on the function of land for a specific kind of use.” An example of a land quality is the plant-available water stored in soil, and an example of a soil characteristic is the clay content that contributes to the plant-available water-holding capacity.

It is generally recognized that a single soil characteristic is of limited use in evaluating differences among soils,^[15] and that use of more than one quantitative variable requires a system for combining the measurements into a useful index.^[13] Gersmehl and Brown^[16] advocate regionally targeted systems.

KINDS OF SYSTEMS

Land rating systems for agriculture include those that are used to evaluate the potential for agricultural development of new areas and others that evaluate the potential for agriculture in already developed areas, e.g., land capability for grazing in Australia^[17] and fertility capability classification for the tropics.^[18] Many other land capability assessments exist to help planners rate the suitability of agricultural lands for non-agricultural uses (e.g., see Steiner et al.^[19]). Some examples of agricultural land capability systems are described in this entry. All systems have, in common, a set of assumptions on which the analysis is based, and each system answers the question “capability for what use.”

Development Potential

Examples of systems designed to determine the potential for agricultural development include the FAO framework for land evaluation and the U.S. Bureau of Reclamation (USBR) irrigation suitability classification. The FAO framework combines soil and land properties with a climatic resources inventory to develop an agroecological land suitability assessment.

The USBR capability classification was frequently used to evaluate land's potential for irrigation in the Western United States during the period of rapid expansion of water delivery systems.^[4–6,20] It combines social and economic evaluations with soil and other ecological variables to

Table 1 Examples of land capability systems.

System	Purpose	Property	References
FAO framework	Development potential	Used for large-scale development of agriculture	[1,4]
USBR irrigation suitability	Potential for irrigation development	Used for determining potential to repay costs of developing irrigation	[5,6]
USDA LCC	Land capability for agriculture	Uses 13 soil, climate, and landscape properties to determine agricultural capability	[7,8]
SIR	Land capability for agriculture	Uses nine soil and management factors to determine agricultural capability	[9–11]
Soil potential	Soil suitability for specific uses	Uses a cost index to rate land for any potential use	[12]
Soil quality	Determining status of soil profile	Used for determining the status of selected soil properties	[12–14]

determine whether the land has the productive capacity, once irrigated, to repay the investment necessary to bring water to an area. It recognizes the unique importance of irrigation to agriculture and the special qualities of soils that make them irrigable.

Land Capability

The U.S. Department of Agriculture (USDA) LCC is narrower in scope than either the FAO or USBR capability rating systems. The purpose of the LCC is to place arable soils into groups based on their ability to sustain common cultivated crops that do not require specialized site conditioning or treatment.^[8] Non-arable soils, unsuitable for long-term, sustained cultivation, are grouped according to their ability to support permanent vegetation and according to the risk of soil damage if mismanaged.

Several studies have shown that lands of higher LCC have higher productivity with lower production costs than lands of lower LCC.^[21–23] In a study of 744 alfalfa-, corn-, cotton-, sugar beet- and wheat-growing fields in the San Joaquin Valley of California, those with LCC ratings between 1 and 3 had significantly lower input/output ratios than the fields with ratings between 3.01 and 6.^[23] The input/output ratio is a measure of the cost of producing a unit of output and is a better measure of land capability than output (yield) alone. This suggests that the LCC system provides an economically meaningful assessment of agricultural soil capability.

Quantitative systems result in a numerical index, typically with the highest number being assigned to the land or soil with the highest capability for the selected use. The final index may be additive, multiplicative, or more complex functions of many land or soil attributes. Quantitative systems have the following two important advantages over non-quantitative systems: 1) they are easier to use with geographic information system and other automated data retrieval and display systems and 2) they typically provide a continuous scale of assessment.^[12] No single national system is in use, but several state or regional systems exist.

One example of a quantitative system is the Storie Index Rating (SIR). Storie^[9] determined that land productivity is

dependent on 32 soil, climate, and vegetative properties. He combined only nine of these properties into the SIR to keep the system from becoming unwieldy. The nine factors are soil morphology (A), surface texture (B), slope (C), and management factors such as drainage class (X_1), sodicity (X_2), acidity (X_3), erosion (X_4), microrelief (X_5), and fertility (X_6). Each factor is rated from 1% to 100%. These are converted to their decimal value and multiplied together to yield a single rating for a soil map unit as follows^[9–11]:

$$SIR = \left(A \times B \times C \times \prod_{i=1}^6 X_i \right) \times 100 \quad (1)$$

An area-weighted SIR can be calculated by multiplying the SIR for each soil map unit within a parcel by the area of the soil unit within the parcel, followed by summing the weighted values and dividing by the total area as follows:

$$\text{Area-weighted SIR} = \frac{1}{\text{total area}} \times \sum_{i=1}^n \text{SIR soil}_i \times \text{area soil}_i \quad (2)$$

Values for each factor were derived from Storie's experience mapping and evaluating soils in California, and in soil productivity studies in cooperation with California Agricultural Experiment Station cost-efficiency projects relating to orchard crops, grapes, and cotton. Soils that were deep, which had no restricting subsoil horizons, and held water well had the greatest potential for the widest range of crops. Much of what was non-quantitative scoring has been modernized using computer-based decision support systems.^[24]

Reganold and Singer^[23] found that area-weighted average SIR values between 60 and 100 for 744 fields in the San Joaquin Valley had lower but statistically insignificant input/output ratios than fields with indices <60. The lack of statistical significance is scientifically meaningful but may not be interpreted the same by planners or farmers for whom small differences in profit can be significant.

Designers of these systems have selected thresholds or critical limits that separate one land capability class from another. Frequently, little justification has been given for

the critical limit selected. The issue of critical limits is a difficult one in soils because of the range of potential uses and the interactions among variables.^[14]

Soil Potential

Soil potential is an interpretive system that is used for both agricultural and non-agricultural capability assessments. It is an example of a land capability system that is both quantitative and highly specific. Soil potential is defined as the usefulness of a site for a specific purpose using available technology at an indexed cost. Experts in the location where the system will be used determine the soil properties that most influence the successful use of a soil for a particular purpose. The well-suited soils for the use are given a rating of 100. Soil limitations that can be removed or reduced are identified, and continuing limitations that occur even after limitations have been addressed are also identified. The cost of available practices and technologies used to remove limitations and the cost of continuing limitations are indexed, and these index values are subtracted from 100 to yield a soil potential rating for each soil:

$$\text{SPI} = \text{PI} - (\text{CI} + \text{CL})$$

The best soils in an area are those that have the lowest indexed cost for operating or maintaining use, while ensuring the lowest possible environment.

CONCLUSION

All soils are not the same and are not equally capable of sustained use. Various systems have been created that rate soil properties for these uses. Examples include FAO land suitability, USBR irrigation suitability, USDA LCC, SIR, and soil potential. Each has strengths and weaknesses that the user must recognize before using the system.

REFERENCES

1. A Framework for Land Evaluation. *Soils Bulletin*; Food and Agricultural Organization (FAO): Rome, 1976; 1–32.
2. Gregorich, E.G.; Carter, M.R.; Angers, D.A.; Monreal, C.M.; Ellert, B.H. Towards a minimum data set to assess soil organic matter quality in agricultural soils. *Can. J. Soil Sci.* **1994**, *74*, 367–386.
3. Bouma, J. Land evaluation for landscape units. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; E-393–E-412.
4. Food and Agricultural Organization. *Land Evaluation Criteria for Irrigation. Report of an Expert Consultation*. World Soil Resour. Rep. 50; FAO: Rome, 1979.
5. USBR. *Land Classification Handbook*; Bur. Recl. Pub. V; Part 2; USDI: Washington, 1953.
6. Maletic, J.T.; Hutchings, T.B. Selection and classification of irrigable lands. In *Irrigation of Agricultural Lands*; Hagen, R.M., Ed.; Soil Science Society of America: Madison, 1967; 125–173.
7. Huddleston, J.H. Development and use of soil productivity ratings in the United States. *Geoderma* **1984**, *32*, 297–317.
8. Klingebiel, A.A.; Montgomery, P.H. *Land-Capability Classification. Agriculture Handbook No. 210*; Soil Conservation Service; USDA: Washington, 1973.
9. Storie, R.E. *An Index for Rating the Agricultural Value of Soils*; Bulletin 556; California Agricultural Experiment Station: Berkeley, 1932.
10. Wier, W.W.; Storie, R.E. *A Rating of California Soils*; Bull. 599; California Agricultural Experiment Station: Berkeley, 1936.
11. Storie, R.E. *Handbook of Soil Evaluation*; Associated Students Store; University of California: Berkeley, 1964.
12. Ewing, S.A.; Singer, M.J. Soil quality. In *Handbook of Soil Science*; Huang, P.M., Li, Y., Sumner, M.E., Eds.; CRC Press: Boca Raton, 2012; 26-1–26-28.
13. Halvorson, J.J.; Smith, J.L.; Papendick, R.I. Integration of multiple soil parameters to evaluate soil quality: A field example. *Biol. Fert. Soils* **1996**, *21*, 207–214.
14. Arshad, M.A.; Coen, G.M. Characterization of soil quality: Physical and chemical criteria. *Am. J. Alternative. Agr.* **1992**, *7*, 25–31.
15. Reganold, J.P.; Palmer, A.S. Significance of gravimetric versus volumetric measurements of soil quality under biodynamic, conventional, and continuous grass management. *J. Soil Water Conserv.* **1995**, *50*, 298–305.
16. Gersmehl, P.J.; Brown, D.A. Geographic differences in the validity of a linear scale of innate soil productivity. *J. Soil Water Conserv.* **1990**, *45*, 379–382.
17. Squires, V.R.; Bennett, F. Land capability assessment and estimation of pastoral potential in semiarid rangeland in South Australia. *Arid Land Res. Manage.* **2004**, *18*, 25–37.
18. Sanchez, P.A.; Palm, C.A.; Buol, S.W. Fertility capability soil classification: A tool to help assess soil quality in the tropics. *Geoderma* **2003**, *114*, 157–185.
19. Steiner, F.; McSherry, L.; Cohen, J. Suitability analysis for the upper Gila River watershed. *Landscape Urban Plan.* **2000**, *50*, 199–214.
20. McRae, S.G.; Burnham, C.P. *Land Evaluation*; Clarendon Press: Oxford, 1981.
21. Patterson, G.T.; Mackintosh, E.E. Relationship between soil capability class and economic returns from grain corn production in southwestern Ontario. *Can. J. Soil Sci.* **1976**, *56*, 167–174.
22. Van Vliet, L.J.P.; Mackintosh, E.E.; Hoffman, D.W. Effects of land capability on apple production in southern Ontario. *Can. J. Soil Sci.* **1979**, *59*, 163–175.
23. Reganold, J.P.; Singer, M.J. Comparison of farm production input/output ratios of two land classification-systems. *J. Soil Water Conserv.* **1984**, *39*, 47–53.
24. O'Geen, A.T.; Southard, S.B.; Southard, R.J. *A Revised Storie Index for Use with Digital Soils Information*; University of California Division of ANR Publication: Oakland, 2008; Vol. 8355.

Land Capability Classification

Thomas E. Fenton

Soil Morphology and Genesis, Agronomy Department, Iowa State University, Ames, Iowa, U.S.A.

Abstract

Land classification may be defined as a classification of specific bodies of land according to their characteristics or to their capabilities for use. A natural land classification system is one in which natural land types are placed in categories according to their inherent characteristics. Some sort of land classification system was used in the transfer of public lands to private owners and users as the United States settled. However, a scientific approach was not possible until soil surveys, topographic maps, economic analyses of production, and other activities became available in the early 1900s. One of the most commonly used land classification systems for land management is the land capability classification. Hockensmith defined this classification as a systematic arrangement of different kinds of land according to those properties that determine the ability of the land to produce permanently. There are three major groupings: 1) capability classes; 2) capability subclasses; and 3) capability units.

INTRODUCTION

A common definition of land is the surface of the earth and all its natural resources. This is interpreted to include the atmosphere, the soil, and underlying geology, hydrology, and plants on the earth's surface.^[1] In the *1938 Yearbook of Agriculture*, land is defined as the total natural and cultural environment within which production must take place. Its attributes include climate, surface configuration, soil, water supply, and subsurface conditions together with its location with respect to centers of commerce and population. It should not be used as synonymous with soil or in the sense of the earth's surface only.^[2] Other definitions included the results of the past and present human activities as well as the animals within this area when they exert a significant influence on the present and future uses of land by man. Thus, the concept of land is much broader than soil. However, because soils are a major component of terrestrial ecosystems,^[3] they integrate and reflect internal and external environmental factors. They have been considered an important factor—if not the major factor—in many land classification systems.

CLASSIFICATIONS

Land classification may be defined as a classification of specific bodies of land according to their characteristics or to their capabilities for use. A natural land classification system is one in which natural land types are placed in categories according to their inherent characteristics. A land classification according to its capabilities for use may be defined as one in which the bodies of land are classified on the basis of physical characteristics with or without

economic considerations according to their capabilities for man's use.^[2]

Olson^[4] defined land classification as the assignment of classes, categories, or values to the areas of the earth's surface (generally excluding water surfaces) for immediate or future practical use. The project, product, or proposal resulting from this activity may be also generally referred to as land classification. Any land classification involves two parts or phases: resource inventory and analysis and categorization. The inventory consists of gathering data and delineating land characteristics on maps. The analysis and categorization put the basic data into a form that can be used generally for a specific use. Thus, there are potentially many land classification systems that could be developed depending on the objectives of the system.

HISTORICAL PERSPECTIVE

Some sort of land classification system was used in the transfer of public lands to private owners and users as the United States settled. However, a scientific approach was not possible until soil surveys, topographic maps, economic analyses of production, and other activities became available in the early 1900s. These sources provided essential data for a scientific approach to land classification. This approach gained added momentum in the mid-1930s due to the need for land classification for many new government programs.

A report submitted to the President of the United States by the National Resources Board dated December 31, 1934,^[5] included a section that dealt with the physical classification of the productivity of the land. The total land area of the United States was rated and divided into five grades

based on the factors thought to affect productivity—soil type, topography, rainfall, and temperature. Norton^[6,7] defined five classes of land in the regions of arable soils according to their use capability. Two other classes were defined for land that should not be cultivated. A National Conference on Land Classification (NCLC) was held in the campus of the University of Missouri in October 1940. The proceedings of this conference were published in the Bulletin 421 of the Missouri Agricultural Experiment Station.^[8] At this conference, it was recognized that land classification in the United States had begun many years prior to 1940. Simonson^[9] summarized the purpose of the NCLC meeting as follows: to encourage and provide opportunity for the discussion of land classification in all of its different aspects.

In the years following the conference, there were many entries published that examined questions related to land classification. General principles associated with technical grouping^[10] or solution of soil management problems^[11] using natural soil classification systems were discussed in detail. Hockensmith and Steele^[12] presented a land capability classification developed for conservation and development of land and for land use adjustments. The system used eight land capability classes with the first four suited for cultivation and the remaining four not suited for cultivation. Within a capability class, the subclasses were determined by the kind of limitation. Within each subclass, the land that required the same kind of management and the same kind of conservation treatment was called a land capability unit. The highest level of abstraction, the capability class, provided a quick, easy understanding of the general suitability of the land for cultivation. Hockensmith^[13] discussed the use of soil survey information in farm planning. A land capability map was an essential part of the farm plan. This map was an interpretative map of the soil map together with other pertinent facts. It was used to help the farmers understand the limitations of their soils and to aid in selection of those practices that would preserve their soils and keep them productive. Klingebiel^[14] defined capability classification as the grouping of individual kinds of soils, called soil mapping unit, into groups of similar soils, called capability units, within a framework of eight general capability classes that were divided into subclasses representing four kinds of conservation problems. This land classification system has been used as a basis for all the conservation farm plans developed by the Soil Conservation Service (now Natural Resources Conservation Service) in the United States since the 1950s and continues to be used. Because of its widespread use, this system is discussed in more detail in the following section.

LAND CAPABILITY CLASSIFICATION

One of the most commonly used land classification systems for land management is the land capability classification. Hockensmith^[15] defined this classification as a systematic

arrangement of different kinds of land according to those properties that determine the ability of the land to produce permanently. Classification was based on the detailed soil survey (scale of 1:15,840 or 1:20,000). The system was described in detail in the Agriculture Handbook No. 210^[16] and continues to be in use, with no significant changes except that Arabic numerals are used to identify the capability classes rather than Roman numerals. There are a number of land capability classifications used throughout the world, but all these methods are patterned after the U.S. system.^[17]

The capability grouping of soils is designed to

1. help landowners and others use and interpret the soil maps;
2. introduce users to the detail of the soil map itself; and
3. make possible broad generalizations based on soil potentialities, limitations in use, and management problems.

There are three major groupings: 1) capability classes; 2) capability subclasses; and 3) capability units. All map unit components, including miscellaneous areas, are assigned a capability class and subclass.^[18]

Capability Classes

The broadest unit is the capability class and there are eight classes. The probability of soil damage or limitations in use become progressively greater from Class I to VIII. There are no additional subdivisions of soils in Class I because by definition the soils in this category have no limitations in use. The proper grouping of the soils assumes that the recommended use will not deteriorate the soil over time.

- Class I soils have few limitations that restrict their use.
- Class II soils have some limitations that reduce the choice of plants or require moderate conservation practices.
- Class III soils have severe limitations that reduce the choice of plants or require special conservation practices or both.
- Class IV has very severe limitations that restrict the choice of plants, require very careful management, or both.
- Class V soils have little or no erosion hazard but has other limitations that are impractical to remove and limit their use largely to pasture, range, woodland, or wildlife food and cover.
- Class VI soils have severe limitations that make them generally unsuited to cultivation and limit their use largely to pasture or range, woodland, or wildlife food and cover.
- Class VII soils have very severe limitation that make them unsuited to cultivation and that restrict their use largely to grazing, woodland, or wildlife.

- Class VIII soils and landforms limitations that preclude their use for commercial plant production and restrict their use to recreation, wildlife, water supply, or to aesthetic purposes.

Capability Subclasses

Subclasses are groups of capability units within classes with the same kind of major limitations for agricultural use. The capability subclass is a grouping of capability units having similar kinds of limitations and hazards. Four general kinds of limitations or hazards are recognized:

1. Erosion hazard
2. Wetness
3. Rooting-zone limitations
4. Climate

The limitations recognized and the symbols used to identify them are risk of erosion designated by the symbol e; wetness, drainage, or overflow, w; rooting-zone limitations, s; and climatic limitations, c. There is a priority of use among the subclasses, if the limitations are approximately of the same degree. The order of use is e, w, s, and c. For example, if a group of soils has both erosion and excess water hazard, e takes precedence over w. However, for some uses, it may be desirable to show two kinds of limitations. This combination is rarely used but when used, the higher priority one is shown first, i.e., *ilew*.

Capability Units

Capability units provide more specific and detailed information than class or subclass. They consist of soils that are nearly alike in suitability for plant growth and responses to the same kinds of soil management. However, they may have characteristics that place them in different soil series or soil map units. Soil map units in any capability unit are adapted to the same kinds of common cultivated and pasture plants and require similar alternative systems of management for these crops. Another assumption is that the longtime estimated yields of adapted crops for individual soil map units within the unit under comparable management do not vary more than about 25%.

CONCLUSION

The grouping of soil map units as used in the land capability classification is a technical classification. The basis for this classification is the natural soil classification system in which the soils and soil map units are defined based on their characteristics and interpretations made based on those properties that affect use and management. For conservation planning, environmental quality, and generation of interpretive maps, it is important to know the kind of soil,

its location on the landscape, its extent, and its suitability for various uses.

REFERENCES

1. Brinkman, R.; Smyth, A.J. *Land Evaluation for Rural Purposes*. Publication 17; International Institute for Land Reclamation and Improvement: Wageningen, the Netherlands, 1973; 116.
2. United States Department of Agriculture. *Soils and Men: Yearbook of Agriculture*; U.S. Govt. Printing Office: Washington, DC, 1938; 1232.
3. Jenny, H. *The Soil Resource: Ecological Studies*; Springer: New York, 1980; Vol. 37, 377.
4. Olson, G.W. *Land Classification*; Cornell University Agricultural Experimental Station Agronomy 4: Ithaca, New York, 1970.
5. National Resources Board Report. Part II. *Report of the Land Planning Committee, Section II*; U.S. Govt. Printing Office: Washington, DC, 1934; 108–152.
6. Norton, E.A. Classes of land according to use capability. *Soil Sci. Soc. Am. Proc.* **1939**, *4*, 378–381.
7. Norton, E.A. *Soil Conservation Survey Handbook; Misc. Publication 352*; U.S. Department Agricultural Soil Conservation Service: Washington, DC, 1939; 40.
8. Miller, M.F. The Classification of Land. *Proceeding of First International Conference on Land Classification, Bulletin 421*, Missouri Agricultural Experimental Station, 1940; 334.
9. Simonson, R.W. The National Conference on Land Classification. *Soil Sci. Soc. Am. Proc.* **1940**, *5*, 324–326.
10. Orvedal, A.C.; Edwards, M.J. General principles of technical grouping of soils. *Soil Sci. Soc. Am. Proc.* **1941**, *6*, 386–391.
11. Simonson, R.W.; Englehorn, A.J. Interpretation and use of soil classification in the solution of soil management problems. *Soil Sci. Soc. Am. Proc.* **1942**, *7*, 419–426.
12. Hockensmith, R.D.; Steele, J.G. Recent trends in the use of the land-capability classification. *Soil Sci. Soc. Am. Proc.* **1949**, *14*, 383–387.
13. Hockensmith, R.D. Using soil survey information for farm planning. *Soil Sci. Soc. Am. Proc.* **1953**, *18*, 285–287.
14. Klingebiel, A.A. Soil survey interpretation-capability groupings. *Soil Sci. Soc. Am. Proc.* **1958**, *23*, 160–163.
15. Hockensmith, R.D. Classification of land according to its capability as a basis for a soil conservation program. Reprinted from *Proceedings of the Inter-American Conference on Conservation of Renewable Natural Resources*, Denver, CO, Sept 7–20, 1948.
16. Klingebiel, A.A.; Montgomery, P.H. *Land Capability Classification*. Agricultural Handbook No. 210; U.S. Department of Agricultural Soil Conservation Service: Washington, DC, 1961; 21.
17. Hudson, N. Land use and soil conservation. In *Soil Conservation*, 3rd Ed.; Iowa State University Press: Ames, IA, 1995; 391.
18. U.S. Department of Agriculture, Natural Resources Conservation Service. *National Soil Survey Handbook, Title 430-VI*. Available at <http://soils.usda.gov/technical/handbook/> (accessed October 2012).

Land Evaluation

J. Herbert Huddleston

Department of Soil Science, Oregon State University, Corvallis, Oregon, U.S.A.

Abstract

Soil resource evaluation often serves as a surrogate for land evaluation. In fact, the only real difference between comprehensive land evaluation and soil resource evaluation is that land evaluation may include assessments of social and economic factors, whereas soil resource evaluation includes such factors only in a very general way, if at all.

INTRODUCTION

Land refers to a specific area of the earth's surface that is characterized by its soil, climatic, geologic, topographic, hydrologic, floral, and faunal resources, as well as by the kinds of human activities that take place on it.^[1–4] *Land use* refers to those biological and technological activities that manage and improve land resources for economic and social purposes.^[2] *Land evaluation* refers to an assessment of the properties of the several resources that characterize land as they bear on the requirements of one or more specific land uses.^[3] It is undertaken both to assess the *performance* of land under specified uses^[3] and to *communicate* information about alternative uses of the same tract of land.^[5]

Soil, strictly speaking, is only one of the natural resources that characterize the land resource. But the same set of natural resources that define land also define the environment in which soil forms,^[6] and soil expresses both the integrated effects of its internal properties and the combined effects of its surroundings. Soil resource evaluation, therefore, often serves as a surrogate for land evaluation. In fact, the only real difference between comprehensive land evaluation and soil resource evaluation is that land evaluation may include assessments of social and economic factors,^[3,4,7] whereas soil resource evaluation includes such factors only in a very general way, if at all. Readers who are interested in the comprehensive land evaluation approach will find excellent resources in FAO^[3] and McRae and Burnham.^[4]

HISTORICAL OVERVIEW

Land evaluation in some form has been going on for as long as human societies have been able to observe their environment and adapt their activities to the resource qualities available to them. Over four millennia ago, the Chinese were making soil maps as a basis for taxation and

agricultural management, and the early Romans clearly understood that different regions had different soils, and these differences were manifest in different crop yields.

Land evaluation in the United States began in earnest as the Division of Soils initiated soil surveys to direct agricultural development onto the more productive soils^[8] and the Bureau of Reclamation began classifying land suitability for irrigation development. The Bureau of Reclamation later wrote a comprehensive land evaluation manual^[7] in which all related natural resources, as well as the economic factors affecting the development and use of the land, were assessed. The U.S. Department of Agriculture developed the Land Capability Classification,^[9,10] which was based on an interpretation of soil survey information and defined capability explicitly in terms of limitations to the use of soil and risk of damage to soil when cultivated.

After World War II, soil survey information was used increasingly to develop interpretations for land use planning^[11] and other nonagricultural uses of land.^[12] Specific procedures for making these interpretations are spelled out in the *National Soil Survey Handbook*.^[13] In the 1980s, two new interpretations, namely the soil potential rating (SPR) and the land evaluation and site assessment (LESA) program, were developed, both of which incorporate a degree of economic analysis in their evaluation criteria. SPRs^[13,14] are based on net returns from specific agricultural or non-agricultural uses on specific soils. They are determined by expressing the yield, output, or performance of a soil in economic terms and subtracting from the output value the costs of inputs or management practices needed to achieve the desired output. LESA combines an evaluation of resource quality with an evaluation of various social, cultural, and economic impacts on land use.^[15–17] The land evaluation part is really soil resource evaluation because it is based on SPRs, soil productivity ratings, land capability classes, or some combination of these. Site assessment considers impacts associated with parcel size and shape, access to supporting services and markets, historical or cultural values, and compatibility among adjacent and nearby land uses.

STEPS IN THE DEVELOPMENT OF LAND EVALUATION MODELS

The first and most important task is to *specify the objective* as thoroughly and completely as possible. Land evaluation is always undertaken for some specific land use, and because different land uses require different sets of soil and other resource attributes, the objective must be stated clearly to develop evaluation criteria properly. An important corollary of objective specification is that a land evaluation developed for one specific purpose should not be used as a surrogate to classify land for a different use. Land capability classes, e.g., are neither measures of soil productivity nor classes of agricultural suitability and should not be used in land use planning programs that require evaluations of productivity or suitability.

The second step is to *identify those resource properties that most affect the desired use*. This is best done by assembling a committee of local people—namely farmers, farm service advisers, planners, developers, and realtors—who know both the resources of the area and the issues associated with the intended use. Involving local people ensures that the widest possible range of ideas and issues is represented and gives credibility to the land evaluation system that emerges from the process. The committee should first make a long list of all factors that might influence the land use of interest and then, through discussion and debate, narrow the list to a few important factors that are the least correlated with each other as possible.

The third step is to *develop specific criteria for evaluating the impact of each factor selected*. Land evaluation models may be either qualitative or quantitative, deductively or inductively derived.^[18] Qualitative criteria assign ratings such as good, fair, or poor suitability to soil property characteristics. Quantitative criteria assign numerical ratings on a scale of 10, or 100, to the values associated with each property. Deductive models use actual data, such as crop yields, to create classes of land suitability. Inductive models use the values of various soil properties that are highly correlated with performance to develop empirical ratings of land suitability. In all cases, criteria should be specified as completely as possible to maximize consistency among all users. Soil and landscape complexity, however, may defy absolute prescription of criteria that are applicable to all possible situations, in which case the model should leave some flexibility for users to subjectively determine exactly how a criterion should be evaluated.

The fourth step is to *combine all of the individual factor ratings into an overall evaluation* of land quality for the specified objective. Additive models, multiplicative models, or combination models^[18] may all be appropriate, depending on the objective and the consensus of the committee undertaking the land evaluation. Additive models simply sum up all of the individual factor ratings or subtract them from an ideal score of, e.g., 100. They are easy to use,

but these models risk minimizing the full impact of one or more particularly limiting factors, especially if the model includes a large number of factors. Multiplicative models, which create a product of all individual factor ratings,^[19] do allow single factors to control the overall evaluation, but the final rating generated can be significantly lower than each of the individual factor ratings, even when all factor ratings are high. Combined models capture the advantages of both additive and multiplicative models by using additive techniques to combine property ratings within major factors and then by multiplying major factor ratings to generate the overall rating.

The final steps in the process are to *set thresholds for creating land suitability classes* and to *validate the results of the model*. Setting class limits is another committee effort, and it may be done using overall ratings only, ratings of component factors, or both.^[20] Model validation is necessary to ensure the usefulness and credibility of the final product. Several test cases are selected, all model factors are evaluated, and the committee determines subjectively whether the results are realistic and “make sense.” If not, the model is revised—by adding or deleting factors, changing factor ratings, or adjusting criteria for setting thresholds—and then tested again, until consensus is reached.

ROLE OF SOIL SURVEYS IN LAND EVALUATION AND LAND CLASSIFICATION

Soil surveys are the backbone of any land evaluation system for the following three reasons: 1) soils are essential land resources; 2) soil survey interpretations are often sufficient to meet the land evaluation objective; and 3) soil surveys provide a cartographic base from which maps of resultant land evaluation classes can be made. Two issues must be fully understood to use soil surveys properly for land evaluation: the composition of soil map units and the limitations of scale.

Most soil map units are defined by a single kind of soil. Each delineation of these map units represents a single element of the landscape within which soil properties and behavior are reasonably homogeneous. Because soils are naturally variable, however, each soil-landscape element contains small areas whose soils are different from the dominant soil for which the map unit is named. These areas are called *inclusions*, and their soils are identified in map unit descriptions. The problem is that, although inclusions may behave quite differently from the dominant soil for a given kind of land use, it is a common practice to treat each delineation as though it were composed entirely of the dominant, named soil, classify its land use suitability accordingly, and map it as such. Users must understand that some error is inherent in such interpretive maps and avoid the temptation to assume that, once mapped, all areas shown in a particular land use suitability class are 100% pure.

Most soil surveys used for land evaluation purposes are order 2 or order 3 surveys having scales ranging from 1:15,840 to 1:63,360. Because landscape scale soil variability occurs within meters, order 2 and order 3 surveys necessarily entail generalizations that result in inclusions, particularly in the vicinity of soil boundaries. Some detailed land use planning projects may invoke detailed soil surveys at scales of 1:1200 or 1:4800, which do permit higher degrees of accuracy, but it is more common to enlarge order 2 or order 3 maps to overlay the results of a land evaluation on larger-scale planning or plat maps. This is particularly easy to do when a digitized soil map is brought into a computerized geographic information system, and scales can be changed easily at the click of a mouse. Users must understand that simply changing the scale at which a soil or land evaluation map is displayed does not change its accuracy, and all the inclusions present in the original delineations are still present in their representation at a larger scale.

Soil surveys, when used appropriately with full understanding of their limitations, are excellent tools for conducting land evaluations for many kinds of uses. Many land evaluations can be based entirely on existing soil survey interpretations. Others may be developed by identifying relevant factors, evaluating each one according to precisely specified criteria, combining factor ratings into a land evaluation model, and setting thresholds for land suitability classes. Existing soil surveys may then serve as templates for mapping the results of land evaluation efforts.

REFERENCES

1. Brinkman, R.; Smyth, A.J. *Land Evaluation for Rural Purposes*, Publication 17; International Institute for Land Reclamation and Improvement: Wageningen, 1973.
2. Vink, A.P.A. *Land Use in Advancing Agriculture*, Advanced Series in Agricultural Sciences 1; Springer-Verlag: New York, 1975.
3. FAO. *A Framework for Land Evaluation*, Soils Bulletin; Food and Agriculture Organization of the United Nations: Rome; Vol. 32, 1976.
4. McRae, S.G.; Burnham, C.P. *Land Evaluation*; Oxford Science Publications Monographs on Soil Survey; Oxford University Press: New York, 1981.
5. Beek, K.J. From soil survey interpretation to land evaluation Part 1. From the past to the present. *Soil Surv. Land Eval.* **1981**, 2, 6–12.
6. Jenny, H. *Factors of Soil Formation. A System of Quantitative Pedology*; Dover Publications, Inc.: New York, 1941.
7. USBR. Bureau of Reclamation Manual, Part 2 Land classification; U.S. Dept. of Interior: Washington; Vol. V, 1953.
8. Whitney, M. *Announcement, Circ. 1*; U.S. Dept. Agr. Bureau of Soils: Washington, 1894.
9. Norton, E.A. Classes of land according to use capability. *Soil Sci. Soc. Am. Proc.* **1939**, 4, 378–381.
10. Klingebiel, A.A.; Montgomery, P.H. *Land Capability Classification, Agr. Handbook 210*; U.S. Dept. Agr. Soil Conservation Service: Washington, 1961.
11. Bartelli, L.J.; Klingebiel, A.A.; Baird, J.V.; Heddleson, M.R. *Soil Surveys and Land Use Planning*; Soil Science Society of America: Madison, 1966.
12. Simonson, R.W. *Non-Agricultural Applications of Soil Surveys*; Developments in Soil Sci. 4; Elsevier: Amsterdam, 1974.
13. Soil survey staff. U.S. natural resources conservation service. In *National Soil Survey Handbook, Title 430-IV*; U.S. Govt. Printing Office: Washington, 1999.
14. McCormack, D.E.; Stocking, M.A. Soil potential ratings: I. an alternative form of land evaluation. *Soil Surv. Land Eval.* **1986**, 6, 37–42.
15. Wright, L.E. Agricultural Land Evaluation and Site Assessment (LESA): A new agricultural land protection tool in the U.S.A. *Soil Surv. Land Eval.* **1984**, 4, 25–38.
16. Pease, J.R.; Coughlin, R.E. *Land Evaluation and Site Assessment. A Guidebook for Rating Agricultural Lands*; Soil and Water Conservation Society: Ankeny, 1996.
17. Huddleston, J.H.; Pease, J.R.; Forrest, W.G.; Hickerson, H.J.; Langridge, R.W. Use of agricultural land evaluation and site assessment in Linn County, Oregon, USA. *Environ. Manage.* **1987**, 11, 389–405.
18. Huddleston, J.H. Development and Use of soil productivity ratings in the United States. *Geoderma* **1984**, 32, 297–317.
19. Storie, R.E. *Storie Index Rating*; Spec. Pub. 3203; University of California Division Agriculture Science: Davis, 1976.
20. Huddleston, J.H. Importance of the LESA objective in selecting LE methods and setting thresholds for decision making. In *A Decade with LESA. The Evolution of Land Evaluation and Site Assessment*; Steiner, F.R., Pease, J.R., Coughlin, R.E., Eds.; Soil and Water Conservation Society: Ankeny, 1994; 78–93.

Land Forms

Carolyn G. Olson

National Leader Soil Survey Investigations, Lincoln, Nebraska, U.S.A.

Abstract

Landforms are features of the earth that together comprise the land surface. They may be of large features, such as plains, plateaus, or mountains, or smaller features, such as hills, eskers, or hillslopes. There are several ways to describe a landform. Surface form is the most common method of description and includes an analysis of shape factors. Early classification generally falls into the following five categories: genetic, process, size, topographic expression (morphology), and degree of erosion.

INTRODUCTION

Landforms are features of the earth that together comprise the land surface. They may be of large features, such as plains, plateaus, or mountains, or smaller features, such as hills, eskers, or hillslopes. The glossary of geology defines a landform as “any physical, recognizable form or feature of the earth’s surface having a characteristic shape and produced by natural causes.”^[1] A similar definition for landform can be found in almost any geology or geomorphology textbook. The word “landform” is derived from the Old English or Old German word *land* and the Latin *forma*.^[2]

DESCRIPTION AND MORPHOMETRY

There are several ways to describe a landform. Surface form is the most common method of description and includes an analysis of shape factors. Shape is characterized by breaks and changes of slope, directions and angles of maximum slope, and other similar information. A minimum of three parameters are used to describe slope shape: 1) slope length or vertical curvature; 2) slope width or horizontal curvature; and 3) gradient. Slope length is measured by a radial that crosses topographic contour lines at right angles. This dimension is sometimes referred to as a longitudinal profile. Slope width, measured perpendicular to the slope length, parallels topographic contour lines. The curvature or shape of a simple landform such as a hillslope may be defined by a series of forms (Fig. 1). Ruhe represented changes in hillslope curvature by a matrix of nine basic forms.^[4] Huggett added surface flow lines to basic slope shapes,^[5] and Pennock et al. combined curvature with landform elements to identify seven different hillslope positions.^[3]

Since there are hundreds of landforms, it might be useful to choose a commonly recognized landform such as a dune

to discuss some of the aspects of shape. Characteristically, many dunes have a steeper, shorter, leeward side and a longer, lower gradient, or windward side and in cross section have a distinctive asymmetry of slope length. Slope width is dependent on the amount of material composing the dune. The curvature of the windward side is often linear to linear convex and that of the leeward side almost any possible combination of forms. In the case of dunes, each distinctive slope also provides information on the wind direction during formation. The map view on topographic maps and aerial photographs of a barchan dune with its commonly recognized crescentic patterns illustrates its direction of movement. Lancaster provides an excellent series of drawings and photographs illustrating several types of dune landforms.^[6]

The third slope-shape factor is gradient, the angle of inclination of the slope with respect to the horizontal. On a topographic map, it is measured at right angles to the contour lines. Soil scientists measure gradient in percentage. Geologists and engineers generally measure gradient in degrees.

In addition to shape, certain landforms are best described by waveforms which include the dimensions of length, height, and periodicity. Examples of landforms for which waveform description is useful are meander curves and dune fields.

Internal form or structure is often included in a landform description. This is an important distinguishing criterion for the identification of eroded or degraded landforms in which characteristic surface morphology has been obscured. For example, many landforms such as dunes, eskers, or barrier islands are composed primarily of sand. However, the distribution of sands and the cross-bedding in a dune are internal structural elements unique to a landform developed by eolian processes.

The shape and size of a landform are not random. These parameters are determined by the processes that build landforms and by the properties of their materials.

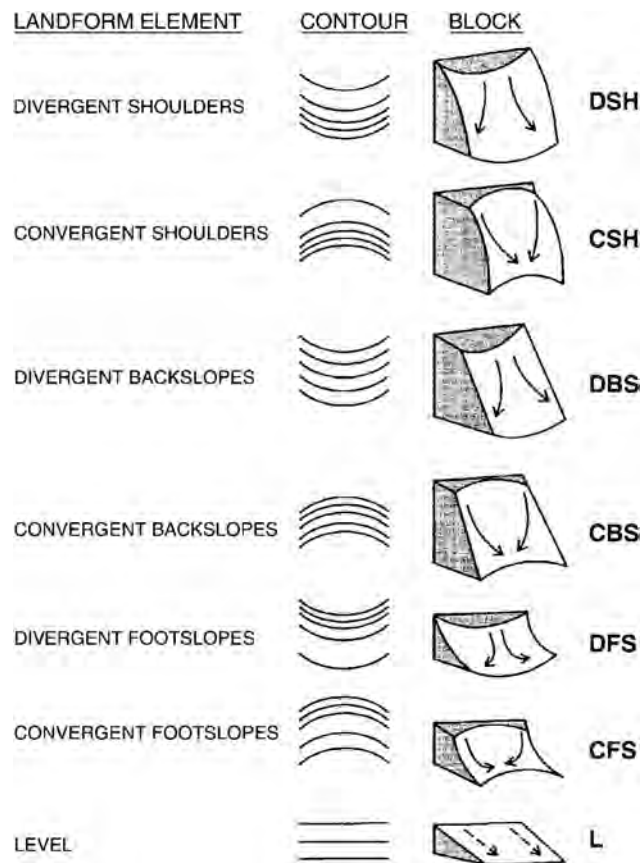


Fig. 1 A matrix of representative hillslope shapes.

Source: From Pennock, Zebarth, et al.^[3]

In the case of a depositional feature such as a dune, the size of the feature is also closely related to the amount of depositional material available during formation. A dune can form that is a few centimeters to several hundred meters in height. The former can occur in the dry bed of a small ditch or in intermittent streams, as the wind blows sandy bed material into dune forms. Dunes in the central deserts of China or the Sahara of Northern Africa are examples of the latter.

Landforms have a natural relationship to each other and to the surrounding landscape. Fractal geometry has been used to produce realistic computer-generated reproductions of landforms.^[7] The mathematical concept of fractal geometry also eliminates the problems associated with the scale of observation. A landform is observed and defined differently by an individual on the ground than is a feature observed from earth orbit.

CLASSIFICATION

There are numerous approaches to the classification of a landform.^[8] A summary of early classification schemes^[9] first presented in the study by Howard and Spock^[10] generally fall into the following five categories: genetic,

process, size, topographic expression (morphology), and degree of erosion.

Features have been classified by size.^[11] Landforms were grouped into numbered orders. First-order relief features included continents and ocean basins. Second-order relief features were tectonic mountain belts, plateaus, and plains, and third-order relief features were erosional and constructional features.

Groups of landforms have been organized into physiographic provinces.^[12,13] The provinces are subdivided into sections and grouped into divisions. Each grouping was based on landforms resulting from a set of similar processes or a succession of processes. Similar regional groupings have been produced for the coterminous United States,^[14–16] and for Alaska.^[17]

A genetic classification, or the mode of origin, has some overlap with descriptive systems because different manner of origin may produce different forms. Examples of genetic systems include the classification of volcanic rocks,^[18] the classification of glacial landforms,^[19,20] and landslide types and processes.^[21]

LANDFORM DEVELOPMENT

Landforms may be erosional or constructional in form, or a composite of both. Most landforms are the products of erosion, but many are formed by the deposition of sediments, by volcanic activity, or by movements from within the earth's crust. Examples of depositional features include dunes, moraines, and spits. Examples of erosional features are water gaps, deflation basins, or arêtes. Lahars, lava plains, and volcanoes result from volcanic activity. The formation of many alluvial fans and subsequent episodes of erosion and sedimentation in the basin and range of the Western United States are triggered by tectonic uplift.

LANDFORMS AND MAPPING

Landforms are part of a continuum of near surface features, soils, and surficial sediments. Bloom^[22] describes a landform in a continuum as a unit of systematic analysis. When mapping, distinctive landform populations need to be delineated. The patterns are identifiable in the field with aids such as topographic maps, aerial photography, geophysical surveys, land use maps, and other available information.

Delineating landforms can sometimes simplify soil mapping. For example, in traversing a hillslope from its summit to its toe, one might recognize several differing soil map units. The sequence may repeat itself on adjacent hillslopes resulting in a predictable pattern of soils that can be related back to a distinctive landform population.

In mapping large tracts of land or in remote areas, it is theoretically possible to systematically subdivide surficial deposits into textural, morphological, or genetic

components. In reality, it is often difficult. Instead, the composition and nature of map unit materials over wide areas are generally inferred from landforms.^[23,24] For example, in flying over a region, drumlins are observed, then drumlins and eskers, then disintegration features, followed by moraines at right angles blocking valleys. If the only advance information one has is that the area had been glaciated, the sequence of landforms becomes the key to interpreting that one is crossing the periphery of a former ice sheet, possibly its maximum extent. One can then map the active zone, the transition to the proglacial, ice-contact zone, and the outer edge of the glacial advance from the landform distribution.

REFERENCES

1. Goldthwait, R.P.; Matsch, C.L. *Glossary of Geology*, 4th Ed.; American Geological Institute: Alexandria, 1997; 769 pp.
2. Goldthwait, R.P.; Matsch, C.L. *The Random House Dictionary of the English Language, Unabridged*, 2nd Ed.; Random House, Inc.: New York, 1987.
3. Pennock, D.J.; Zebarth, B.J.; deJong, E. Landform classification and soil distribution in hummocky terrain saskatchewan, Canada. *Geoderma* **1987**, *40*, 297–315.
4. Ruhe, R.V. *Geomorphology, Geomorphic Processes and Surficial Geology*; Houghton Mifflin: Boston, 1975; 246 pp.
5. Huggett, R.J. Soil landscape systems: A model of soil genesis. *Geoderma* **1975**, *13*, 1–22.
6. Lancaster, N. *Geomorphology of Desert Dunes*; Routledge: New York, 1995; 290 pp.
7. Mandelbrot, B. *The Fractal Geometry of Nature*; W.H. Freeman: New York, 1983; 468 pp.
8. Small, R.J. *The Study of Landforms, a Textbook of Geomorphology*, 2nd Ed.; Cambridge University Press: New York, 1978; 502 pp.
9. Goldthwait, R.P.; Matsch, C.L. *The Encyclopedia of Geomorphology, Encyclopedia of Earth Sciences*; Dowden, Hutchinson and Ross: Stroudsburg, 1968; Vol. 3, 1295 pp.
10. Howard, A.D.; Spock, L.E. A classification of landforms. *J. Geomorphol.* **1940**, *3*, 332–345.
11. Salisbury, R.D. *Physiography*, 3rd Ed.; Henry Holt: New York, 1919; 676 pp.
12. Fenneman, N.M. *Physiography of Western United States*; McGraw-Hill: New York, 1931; 534 pp.
13. Fenneman, N.M. *Physiography of Eastern United States*; McGraw-Hill: New York, 1938; 714 pp.
14. Hammond, E.H. Classes of land-surface form in the forty-eight states, USA, 1963: Map Supp. No. 4. Assoc. Am. Geograph. Ann. **1964**, *54* (1), map supplement no. 4, 1: 5,000,000.
15. Thornbury, W.D. *Regional Geomorphology of the United States*; Wiley: New York, 1965; 609 pp.
16. Hunt, C.B. *Natural Regions of the United States and Canada*; Freeman, W.H., Ed.; San Francisco, 1974; 725 pp.
17. Wahrhaftig, C. *Physiographic Divisions of Alaska*; US Geol. Survey Prof. Paper 482; US Department of Interior Geological Survey: Washington, 1965; 52 pp.
18. Rittman, A. *Volcanoes and Their Activity*; Wiley–Interscience: New York, 1962; 305 pp.
19. Sugden, D.E.; John, B.S. *Glaciers and Landscape, A Geomorphological Approach*; Routledge/Chapman, & Hall: New York, 1976; 376 pp.
20. Goldthwait, R.P.; Matsch, C.L. *Genetic Classification of Glaciogenic Deposits; Final Report of the Commission on Genesis and Lithology of Glacial Quaternary Deposits of the International Union for Quaternary Research (INQUA)*; Balkema: Rotterdam, 1988; 294 pp.
21. Cruden, D.M.; Varnes, D.J. *Landslide Types and Processes*. Turner, A.K., Schuster, R.L., Eds.; National Research Council Transportation Research Board Special Report 247; National Academy Press: Washington, 1996; 673 pp.
22. Bloom, A.L. *Geomorphology, A Systematic Analysis of Late Cenozoic Landforms*, 2nd Ed.; Prentice-Hall: Engle-Wood Cliffs, 1991; 532 pp.
23. Fulton, R.J. Surficial geology mapping at the geological survey of Canada: its evolution to meet Canada's changing needs. *Can. J. Earth Sci.* **1992**, *30*, 232–242.
24. McMillan, A.A.; Powell, J.H. *British Geological Survey Rock Classification Scheme, 4: Classification of Artificial (Man-Made) Ground and Natural Superficial Deposits—applications to Geological Maps and Datasets in the UK*; British Geological Survey Research Report Resonance Raman 99-04; NERC: Nottingham, 1999; 65 pp.

Land Husbandry

Francis Shaxson

Consultant, Dorset, U.K.

Malcolm Douglas

Consultant, York, U.K.

Abstract

Agriculture has not only damaged much land already in use but has also expanded into marginal areas—as a result of population pressure and land scarcity—provoking further the declining usefulness of landscapes for a range of purposes. Almost all the land capable of easy sustainable agricultural use is already being farmed. Therefore the challenge is to enhance and sustain productivity of areas already in use. Improvements in husbandry of land are resulting in rapid expansion of ecologically sustainable types of agronomic, environmental, social, and economic benefits to people in different parts of the world.

INTRODUCTION

Inappropriate management of land in both small and large areas results in declining productivity and/or rising costs, and—especially in countries under the intense climatic conditions of the Inter-Tropical Convergence Zone—in avoidable loss of inputs and water in runoff, water stress in plants, soil erosion, sedimentation, flooding, and related problems. This is particularly serious for the millions of resource-poor rural people in this zone, where there is a strong correlation between environmental degradation, poverty, and insecurity of food and water provision.^[1] Agriculture has not only damaged much land already in use but has also expanded into marginal areas—as a result of population pressure and land scarcity—provoking further the declining usefulness of landscapes for a range of purposes. Almost all the land capable of easy sustainable agricultural use is already being farmed. Therefore the challenge is to enhance and sustain productivity of areas already in use. The holistic “better land husbandry” approach interweaves different specific aspects of management of soil, water, and crops in addressing this, within the constraints and opportunities of diverse natural resource-based livelihood systems.

DEFINING LAND HUSBANDRY

Scope

Much of the failure of past remedial efforts has been a result of their narrow sectoral focus, overreliance on simplistic technical solutions, and concentration on alleviating symptoms rather than addressing real causes.

Experiences in many countries show that improvements in soil properties—notably porosity—resulting from better

husbandry of land enable more efficient use of inputs and increased sustainability of production, with considerable benefits to both people and the environment.

Land husbandry is concerned with maintaining the ongoing usefulness of land for chosen purposes and, more specifically, with sustaining the capacity of its soils to support plant growth and—wherever conditions allow—simultaneously to yield water from its catchments. This, in turn, depends not only on adequate levels of organic matter and plant nutrients, but also on the optimum porosity of its soil architecture, whose restoration after degradation, buildup, and continuing stability depends on optimizing and sustaining conditions fit for the transformative actions of its macro-, meso-, and microorganisms.

Because the management of land is undertaken by rural people, good land husbandry has both agro-ecologic and socioeconomic principles.

Meanings

The word “husbandry” is widely understood when applied to animals and crops. As a concept signifying active care for their well-being, it is equally applicable to land. Animal husbandry, crop husbandry, and land husbandry have, in common, the need for the farmer/land user to:

1. understand the characteristics of different types of plants, animals, and land;
2. predict their likely responses to changes in management and to severe environmental events;
3. devise means of strengthening them against damage; and
4. maintain their productivity and usefulness for the chosen purposes (Fig. 1).^[2]

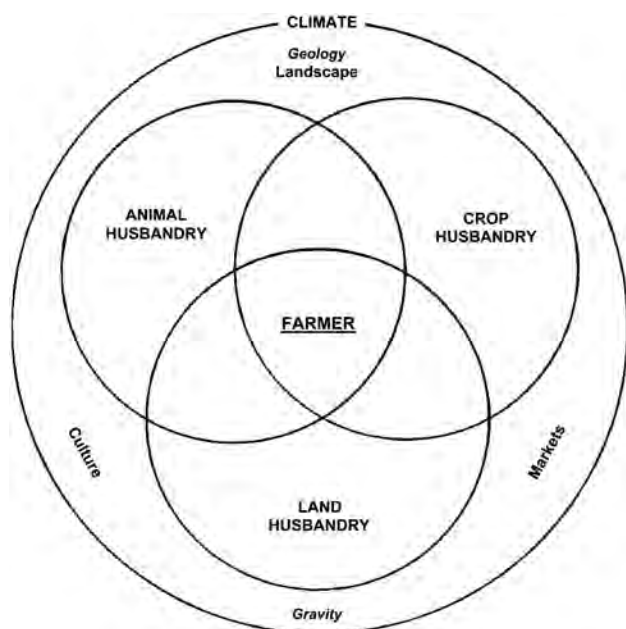


Fig. 1 Three interconnecting husbandries. The farmer is the ultimate decision-maker.

Source: From Shaxson.^[3]

Land husbandry is a broader concept than just “soil-and-water conservation” or land reclamation. Good land husbandry can be defined as: “Care and management of the land to provide outputs of value to the land user and to

society as a whole, while sustaining and enhancing the land’s potentials to continue providing them in the future.”

As a subject for advisers, researchers, and students, “land husbandry” requires integration of disciplines rather than compartmentalization, combining understandings from a range of agronomic, social, environmental, and economic perspectives in ways that acknowledge the land’s ecological nature, and indicate the practicalities of making safe use of it.

Agro-Ecologic Components

Land varies from place to place in a landscape. Different areas of land have different characteristics and different capabilities. Its nature in any place is determined by the particular combination of the features of which it is composed: topography, soil, fauna and flora, and the climate in which it is located.^[4]

Soil stability and productivity derives from multiple interactions between the living and the non-living components of the land, and more specifically from the interactions between its physical, hydrological, biological, and chemical components, under the influences of climate, gravity, and the actions of people. The results may be negative, neutral or—by design—positive in terms of their conservation effectiveness.

Altered hydrology, caused by loss of porosity, and the long periods when bare soil is exposed to the effects of sun, wind, and rain, together with the failure to replenish lost nutrients, are the basic causes of land degradation (Fig. 2).^[5]

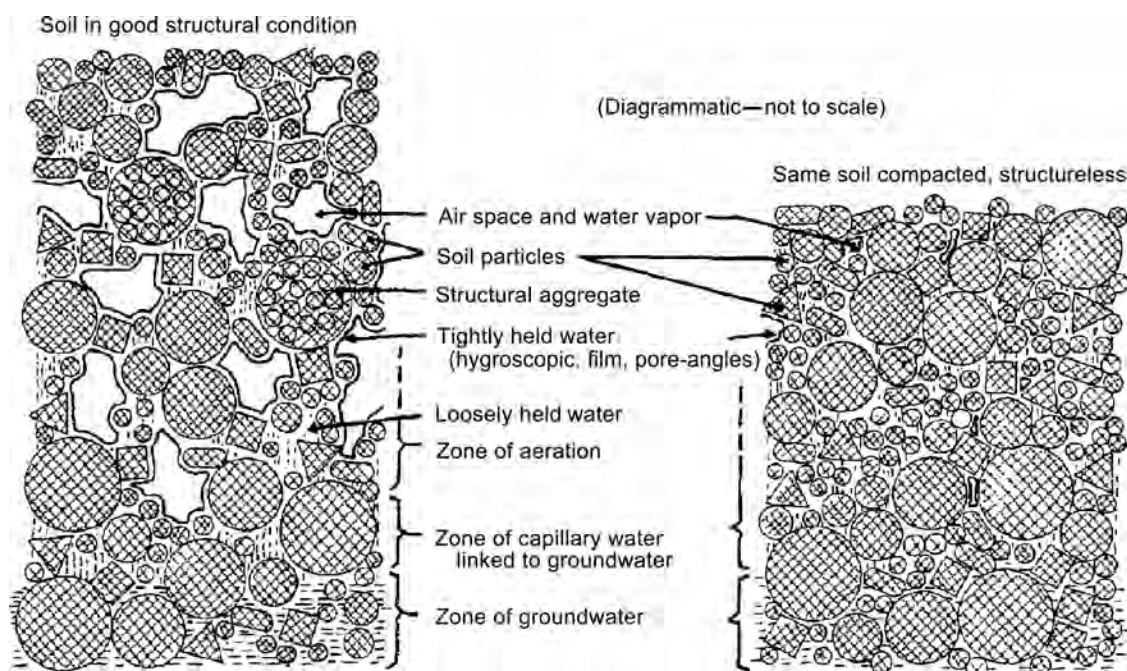


Fig. 2 Loss of pore spaces from the soil architecture diminishes the soils’ capacity to accept, absorb, and transmit rainwater, and alters its suitability for growth and functioning of roots.

Source: From Shaxson, Hudson, et al.^[2]

Good land husbandry aims to increase and sustain biomass production—as crops, trees, shrubs, and grasses—by ensuring that there are no avoidable hindrances to soil's acceptance of rain- or irrigation-water nor to its exploration by roots, and that roots are able to access sufficient oxygen, nutrients, and water within the soil. These features also favor both groundwater recharge by rainfall and the regularity of streamflow, as usable water supplies. Minimization of runoff and erosion, and improvements in efficiency of the use of water and other inputs, follow as consequences (Fig. 3).

The total volume, connectedness, and size distribution of pore spaces in the soil architecture (which affect the volume, distribution, and availability of soil water to plants) are greatly improved by the diversity and activity of the various soil organisms that are present—from earthworms, termites, beetles, and so on, down to the soil-inhabiting fungi and bacteria.

After the porosity of soil architecture has been damaged, mechanical or chemical actions cannot provide self-sustaining remedies. In almost all soils, these are only possible via biological actions, which therefore form the true basis of sustainability of stable productive land uses year by year (Fig. 4).

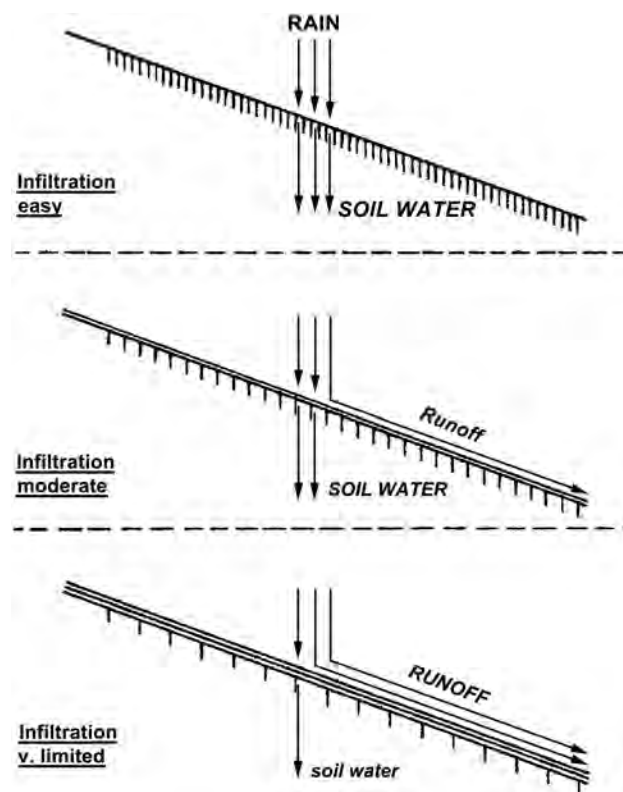


Fig. 3 The condition of the upper layers of the soil determines the partitioning of rainwater between infiltration and runoff, thus affecting the proportion of rainfall that becomes soil water, and affecting the streamflow regime.

Source: From Shaxson.^[3]

Other key concepts in the approach include:

1. plant growth and yields are reduced more directly and rapidly by a shortage or excess of soil water than they are by a loss of soil particles; therefore there should be first emphasis on rainwater management, and improved soil porosity, more than on conservation of soil per se;
2. minimizing runoff (by enabling infiltration) should precede the controlling of its overland flow; therefore agronomic features that manage rainwater's energy are potentially more effective than mechanical measures for limiting soil loss;
3. because of organic matter's multiple beneficial effects—most importantly for biotic activity, water relations, and nutrient relations in the soil—improvement in the amount and management of organic matter is the key to maintaining soil productivity;
4. only after farmers have made improvements in the biological, physical, and hydrological properties of their soils can they expect to obtain the full benefits to their crops from supplying the soil with any additional nutrients in mineral fertilizers.

The starting point for good land husbandry is to characterize the particular area of land with respect to its topography, the nature and condition of its soils, and the likely hazards of degradation attending its use. On this basis, the physical layout and management of the chosen land uses should conform with the land's biophysical characteristics, and the hazards should be counterbalanced by the adoption of conservation-effective husbandry practices.



Fig. 4 Clods from two adjacent plots of a farmer's trial. That from the plot under several years' zero tillage (left, held sideways) shows the residues of previous crops in the process of transformation into soil organic matter, evidence of worms' activities, and good porosity. That from the plot under conventional annual disk-tillage for land preparation (right) has never been protected by crop residues, and shows less darkening by organic matter, less evidence of organisms' activities, and poorer porosity.

Source: Photo by F. Shaxson.

In the tropics and subtropics in particular, for greatest sustainable improvement in productivity, the most important aspects of conservation-effective farming are:

1. maintaining an even surface cover—of at least 30%—of plant materials, particularly crop residues and cover crops;
2. choice of appropriate crop rotations; and
3. no mechanical soil disturbance except that necessary for direct seeding or planting of the crop.^[6]

Synergistic benefits are obtained, in both sustaining soil productivity and enhancing farm livelihoods, through combinations of locally appropriate practices that improve management of rainwater, organic materials, nutrition, and plants, and proactively avoid soil damage.

Socioeconomic Components

Farmers are primarily concerned with stable and economic production, not with the conservation of soil and water per se. They are only likely to adopt and adapt recommendations for better land husbandry if they judge them to be capable, singly or jointly, to:

1. stabilize or increase output;
2. be economic;
3. confer other benefits that are important to them; and
4. simultaneously release resources—time, energy, cash, and so on—for additional favored actions or investments.

Rural people make rational decisions within the “envelope” that bounds their circumstances, as defined by their skills and resources, and by other internal and external potentials and limitations. Improvements in land husbandry are more likely to occur if the “shape” of this envelope is changed and enlarged, than if attempts are made to change their rationality within an unaltered envelope.

These considerations indicate the need for “bottom-up” rather than “top-down” participation with farmers and interchange of information, and for development of the credibility of advisers in the minds of the land users.

Using our ability to foresee land’s reactions to changes, it is possible to assist land users—having the central role as stewards and managers of their soil, water, and vegetation resources—to choose and implement combinations of land use enterprises and active management matching the land’s characteristics, with conservation-positive results in terms of productivity and sustainability.^[7,8]

Central to the promotion, realization, and spread of better land husbandry is the adoption of:

1. people-centered learning approaches, involving field-based discovery exercises through “Farmer Field Schools”;

2. community-based approaches to land management that empower communities to assess the status of their natural resources and to plan, implement, and monitor results of land husbandry improvements; and
3. market-oriented approaches that encourage better land husbandry as the basis for raising productivity and profitability of specific land use enterprises.

BENEFICIAL EFFECTS

There is a growing body of evidence from many countries across the world that improvements in land husbandry resulting in microscale positive effects on land’s productivity also result in benefits at macroscale, both to the environment and to people. The range and size of the derived benefits depend on the nature, degree, and duration of the improvements to the soil conditions.

Table 1 was compiled from reports, from researchers, advisory workers, government agencies, project evaluators, and from farmers’ own observations, of the effects of conservation-effective farming. They come from high-rainfall (1600 mm) to low-rainfall (300 mm) areas, from many countries (including Australia, Brazil, Burkina Faso, Chile, Costa Rica, El Salvador, Honduras, India, Italy, Kenya, Morocco, Niger, Pakistan, Paraguay, and Zimbabwe).

On this basis, comparable effects of such improvements may reasonably be expected elsewhere—in different mixes and to different degrees, according to the particular agroenvironmental and socioeconomic circumstances—where farmers’ balancing of likely risks, costs, and benefits in their particular situations suggest it is worthwhile to adopt. Interchange of information between enthusiastic farmers has proved to be the most effective means of spread.

The most complete expression of the range of multiple effects of improvements in land husbandry is seen in Brazil, where mulch-based rotational zero tillage systems are being implanted on increasingly large areas—e.g., from 1000 ha in 1972 to 14 million ha by 2000. Other countries have experiences of comparable nature, although often of lesser range; but all validate the overall thesis and highlight the urgent necessity to encourage widespread practical application of the principles of better land husbandry in locally appropriate ways.^[6,9]

IMPLICATIONS

Introduction of small or less complex improvements in land husbandry may be relatively straightforward. However, achieving the fullest synergistic and mutually supporting effects of more comprehensive benefits of

Table 1 Direct and derived benefits following better land husbandry.

Better land husbandry	Physical planning		National policy and strategy
+Cover +organic matter, +nutrients ▼ Improved porosity ▼			
For plant	For farm family	For catchment/community	For society
Less destructive impact of rainfall (vertically) and runoff (laterally)	Increased use-efficiency of rain and other inputs	Less damage by floods and sedimentation	Less “greenhouse” gas
Reduced soil loss (e.g., –90%)	Reversal of slow yield decline	Greater recharge of groundwater and regularity of streams' base flow	More carbon fixed
Less direct evaporation from soil surface	Less severe effects of common storms	More water available for irrigation	Less pressure to clear forests for farms' expansion
More biological activity	Yield variations smoothed	More secure dry-weather water supplies	Hydropower supplies maintained
More cation exchange capacity	Reduced costs (e.g., –30%) and increased returns (e.g., +30%)	Less disruption of transport routes	Options retained for a range of land uses in future
More stable aggregates	Less need for irrigation	Reduced costs of infrastructure repairs	etc.
Better pore space	Greater financial resilience re markets, weather	Less pollution of water and soil	
Better aeration in root zone	Greater security of daily food, water	Reduced costs of water treatment	
More water into soil (e.g., +40%)	Less fuel and energy needed	More community cohesion and interest in managing surroundings	
More plant-available water	More flexible timing of operations	etc.	
Prolonged availability to plants of water and nutrients	More diversity possible	Leads to next column ►	
Better soil T^o regime	Improved nutrition and health		
Better root exploration and function	Safer use of sloping lands		
Better growth and yield of biomass (e.g., +30% or more)	etc.		
Soil continually forms top–downward etc.	Leads to next column ►		
Leads to next column ►			Positive support justified!
▲▲▲▲▲	◀◀◀◀◀	◀◀◀◀◀	◀◀◀◀◀
Feedback	Feedback	Feedback	Feedback

conservation-effective farming over significant areas requires development of local and national policies and strategies both for overcoming constraints and for facilitating adoption and adaptation to fit the various circumstances.^[10] Zero-tillage systems should not be seen as a “quick fix” solution.^[11]

Range managers, agronomists, hydrologists, farm economists, soil scientists, ecologists, geographers, and other

professionals need to link in “an ecology of disciplines” to provide sound advice and appropriate skills that marry the land users’ needs with the land’s characteristics.

By reassessing a common view of land degradation, it is possible to see that attention to increasing cover over the soil, increasing organic matter levels, balancing nutrient supply, and restoring porosity to the soil provides a way of engaging collaboratively with farmers

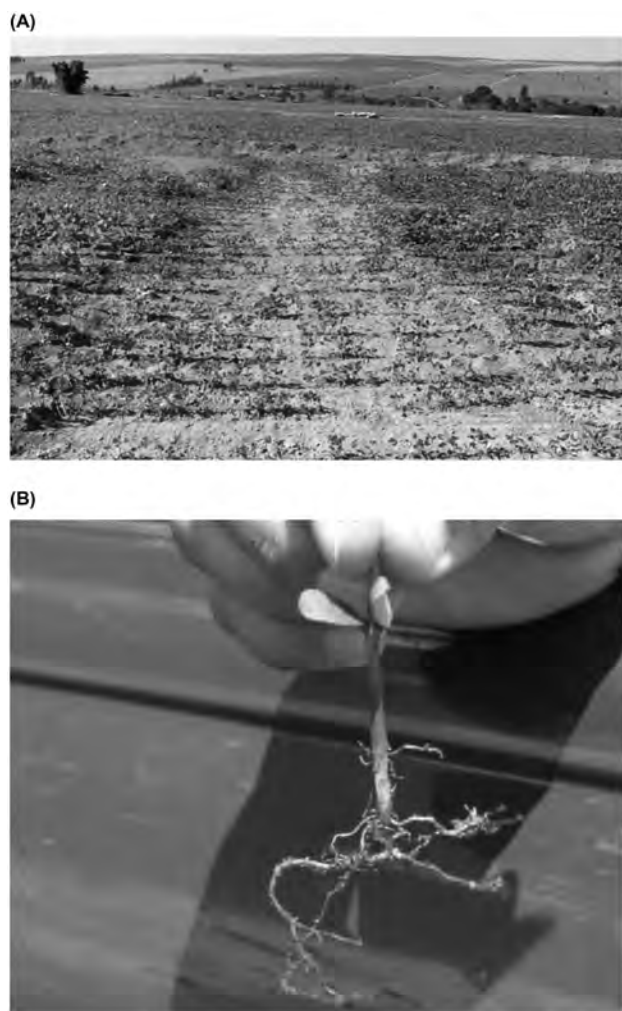


Fig. 5 (A) In pedological terms, this soil is about 2 m deep, but compaction induced by annual disk-tillage has reduced its effective depth to about 5 cm. This has resulted in severe runoff and soil loss. (B) The roots of this soya seedling show the effects of the compaction on their growth.

Source: Reproduced photo of “Roots of Soybeans” from FAO Bulletin no. 75, 1999 and both photos by F. Shaxson.

in countering the consequences of deforestation, excessive cultivation, and overgrazing, by emphasizing the positive effects of bringing soils alive again, rather than invoking the negative effects of punitive (and usually ineffective) legislation (Fig. 5).^[3]

CONCLUSION

Improvements in husbandry of land are resulting in rapid expansion of ecologically sustainable types of agronomic, environmental, social, and economic benefits to people in different parts of the world. As an indicator, Fig. 6A shows a field at a cereal experiment station near Foggia, Italy, planted with Durum wheat. The left side



Fig. 6 (A) The field planted to Durum wheat at the cereal experiment station near Foggia, Italy. (B) The yield of the ZT plot (left side) subsequently was equivalent to 2.63 t/ha at 15.5% protein; that of the conventionally tilled plot was 2.11 t/ha at 11.4% protein.

Source: Information provided by M. Pisante, upper photo by F. Shaxson; lower photo by D. McGarry, with permission of D. McGarry; both taken on May 31, 2002.

(slightly darker) had been under zero tillage, with crop residues left on the surface, for 3 years. The right-hand side had been under conventional tillage for at least that long. Fig. 6B shows the striking difference between representative ears of wheat—that on the left (zero tillage area) evidently was still drawing on available soil water sufficient for the plant to continue photosynthesizing and filling the grain; by comparison, that on the right indicates that exhaustion of a lesser volume of plant-available soil water had seriously limited the time during which grain-filling could occur.

It is increasingly being recognized and demonstrated that an integrated approach is needed for reversing the decline in the productive potentials of land—in both tropical and less severe climatic zones—thus increasing its capacity to produce plant materials and usable water on a sustainable and regular basis.^[12] The indications are that optimizing the husbandry of land provides an essential basis for addressing the problems of hunger and rural poverty.

The subject of land husbandry provides a conceptual matrix within which fit other topics described in this Encyclopedia.

REFERENCES

1. Hamilton, P. "Goodbye to hunger!". ENABLE—Newsletter of the Association for Better Land Husbandry **2003**, 16, 33–38. Available at <http://www.landhusbandry.cwc.net/> (accessed June 2003).
2. Shaxson, T.F.; Hudson, N.W.; Sanders, D.W.; Roose, E.; Moldenhauer, W.C. *Land Husbandry—A Framework for Soil and Water Conservation*; Soil and Water Conservation Society: Ankeny, 1989.
3. Shaxson, T.F. *New Concepts and Approaches to Land Management in the Tropics with Emphasis on Steeplands*; FAO Soils Bulletin; Food and Agriculture Organization of the United Nations: Rome, 1999; Vol. 75.
4. Hamblin, W.K. *The Earth's Dynamic Systems*, 5th Ed.; Macmillan Publishing Co.: New York, 1989.
5. Downes, R.G.; Shaxson, T.F.; Douglas, M.G. *An Ecological Background to Concepts of Land Husbandry, and Principles of Good Land Husbandry*; Association for Better Land Husbandry: Winterborne Kingston, 1997. Available at <http://www.landhusbandry.cwc.net/> (accessed June 2003).
6. Benites, J.R.; Ashburner, J.E. FAO's role in promoting conservation agriculture. In *Conservation Agriculture: A Worldwide Challenge*; Garcia-Torres, L., Benites, J., Martinez-Vilela, A., Eds.; Food and Agriculture Organization of the United Nations with the European Conservation Agriculture Federation: Córdoba, 2001; Vol. 1, 133–147.
7. Douglas, M.G. *Sustainable Use of Agricultural Soils—A Review of the Pre-requisites for Success or Failure*; Development and Environment Reports; Group for Environment and Development, Institute of Geography, University of Berne: Berne, 1994; Vol. 11.
8. Roose, E. *Introduction à la gestion conservatoire de l'eau, de la biomasse et de la fertilité des sols (GCES)*; Bulletin Pédologique de la FAO; Food and Agriculture Organization of the United Nations: Rome, 1994; Vol. 70.
9. Derpsch, R. Frontiers in conservation tillage and advances in conservation practice. Sustaining the Global Farm—Selected Readings from the 10th International Soil Conservation Organization Meeting, West Lafayette, USA, May 24–29, 1999; Stott, D.E., Mohtar, R.H., Steinhardt, G.C., Eds.; 2001; 248–254. ISCO in cooperation with the USDA and Purdue University: West Lafayette (USA), K008. CD-ROM available from the USDA-ARS National Soil Erosion Laboratory, West Lafayette, IN. Available at <http://topsoil.nserl.purdue.edu/nserlweb/isco99/pdf/isco99pdf.htm> (accessed May 2003).
10. Pieri, C.; Evers, G.; Landers, J.; O'Connell, P.; Terry, E. *No-Till Farming for Sustainable Rural Development*; Agriculture and Rural Development Working Paper; Rural Development Department, The International Bank for Reconstruction and Development: Washington, 2002; 65 pp.
11. Landers, J.N. *Zero Tillage Development in Tropical Brazil—The Story of a Successful NGO Activity*; FAO Agricultural Services Bulletin; Food and Agriculture Organization of the United Nations: Rome, 2001; Vol. 147.
12. Food and agriculture organization of the United Nations. *The Sub-Saharan Africa Soil Fertility Initiative—A Review of Progress and Lessons Learned*; Final Draft of Report no. 02/041 CP-SSA; FAO: Rome, 2002.

Land Restoration

Richard W. Bell

School of Environmental Science, Murdoch University, Perth, Western Australia, Australia

Abstract

Land is a finite resource. Every attempt must be made to prevent or minimize land degradation. In addition, restoration of already degraded land needs to be undertaken. Land restoration is a relatively new science that is increasingly informed by the emerging discipline of restoration ecology. Land restoration occurs at a range of different scales. Mine rehabilitation has developed many successful practices of land restoration at a site-specific scale. Most success at both regional and landscape scales has been concerned with restoring key ecosystem functions such as nutrient cycling and water balance.

INTRODUCTION

Soils are essentially a finite and nonrenewable resource^[1] but are subject to various forms of degradation. Monographs by Lal and coworkers^[1–3] and papers therein have dealt in depth with soil and land degradation processes and their measurement, impact, and management. Conacher and Conacher^[4] define land degradation broadly as “alteration to all aspects of the natural (or biophysical) environment by human actions, to the detriment of vegetation, soils, landforms and water (surface and subsurface, terrestrial and marine) and ecosystems.” The challenge for sustainable use of land resources embraces both the minimization of the degrading processes and the restoration of previously degraded land. In this entry, the author focuses on restoration of degraded land from a biophysical perspective.

The term restoration is used in this entry as a generic term after the usage of Hobbs and Norton^[5] who suggest “that restoration occurs along a continuum and that different activities are simply different forms of restoration.” Given that part of this entry also concerns restoration of land degraded by mining, the term rehabilitation as defined by Aronson et al.^[6] is also appropriate. Restoration will usually focus on restoring ecosystem functions such as nutrient cycling, hydrological balance, and ecosystem resilience (see the entry *Restoration Ecology*, p. 1932) although restoring the original flora may on occasions be a realistic and appropriate goal. Reclamation is the term that describes the replacement of ecological functions by planting a different vegetation to that which previously grew; it is commonly used in the United States.

LAND RESTORATION PRINCIPLES

Land restoration comprises the following components: determination of end land use, determining the main

limiting factors for restoration and means of alleviating them, and finally planning and implementation of the restoration program. This entry will deal mostly with the first two of these components.

End Land Use

A clear definition of the end land use is a prerequisite for effective restoration of highly disturbed land. This will determine the stakeholders, the scope for restoration, and the key constraints that have to be alleviated and help to define the goals for determining success. End land use has a major bearing on the degree of difficulty of the task of restoration and hence its cost. Restoration of a complete ecosystem, and its function, is a markedly more complex task than restoration of a stable surface or creating a pasture for low intensity grazing.^[5] The former may require decades of systematic research before restoration goals can be reliably accomplished and demonstrably sustainable.

Prior land use and existing use of the surrounding areas will generally determine the end land use except where the substrate or landscape is so radically altered by degrading processes as to make this impossible. Apart from restoring natural ecosystems, agriculture, forestry, housing, industry, amenity and recreation, wetlands, and waste disposal are all possible end land uses.^[7] Economic and sociopolitical factors have a major bearing on end land use, particularly on production lands, in protected landscapes, and in densely settled areas.

While setting end land use is necessary in order to develop tangible goals for restoration, there are also merits in an adaptive rather than a prescriptive approach. As practice improves, new benchmarks for completion can be set, and new possibilities for end land use emerge, as shown in

Table 1 Biophysical factors limiting land restoration, their consequences for site stability and plant growth, and common treatment methods.

Factor	Constraint	Consequence	Treatment
Climate	Drought	Failure to germinate, poor emergence or establishment, plant death	Irrigation, drought tolerant species, adjust time of sowing
	High temperature	Poor germination and emergence, plant death	Mulching
	Low temperature/frost	Delayed emergence, plant death, poor seed set	
	Extreme rainfall/wind events	Water or wind erosion episodes	Contour banks, soil cover by foliage, mulch or stubble, wind breaks
Landform	Slope	Land slippage, soil creep	Deep rooted plants, reshape to lower slope, contour banks, engineering design
	Runoff	Water erosion	Contour banks, reshape to lower slope
	Exposure	Drought, high winds, extreme temperatures	Tolerant plant species
	Aspect	High winds, extreme temperatures	Tolerant plant species
Hydrology	Runoff	Erosion, waterlogging, flooding	Contour banks, increased drainage intensity
	Limited profile water storage	Drought, runoff, increased groundwater recharge	Tolerant plant species, deep ripping
	Groundwater discharge	Water-filled voids, acid mine drainage	Containment of water, wetland treatment ponds
Substrate properties	Acidity	Poor plant growth especially roots, plant death	Lime, acid tolerant species, P fertilizer
	Alkalinity	Poor plant growth, plant death	Gypsum, leaching, alkaline-tolerant species, acidifying materials
	Salinity	Poor plant growth, plant death	Leaching, salt tolerant species, drainage
	Sodicity	Soil dispersion, crusting, poor seedling emergence, erosion	Gypsum, leaching, organic matter addition
	Nutrient deficiency	Poor plant growth	Fertilizer
	Metal toxicity	Poor plant growth, plant death	Lime, tolerant species, phytoremediation
	Low water availability	Poor plant growth, plant death	Irrigation, mulching, organic matter, deep ripping
	Waterlogging	Poor plant growth, plant death	Drainage, tolerant plant species
	Poorly structured soils	Crusting, poor water holding capacity	Mulching, organic matter, gypsum
	Mycorrhiza	Poor plant growth, nutrient deficiency	Topsoil management, inoculation of nursery plants
	Rhizobium	Nitrogen deficiency	Topsoil management, inoculation
	Soil microbes	Poor mineralization of soil organic matter	Topsoil management

the changes with time in bauxite mine rehabilitation in Southwest Australia.^[8]

Problem Diagnosis and Definition: Biophysical Factors Limiting Land Restoration

Land degradation has many different forms and hence different consequences for restoration.^[3] Diagnosing the main form(s) of degradation at a site or in a landscape is a prerequisite for appropriate land restoration. The main biophysical factors limiting land restoration are as follows: climate, landform, hydrology, substrate properties, and plant establishment. These are described briefly later (see also Table 1). Many examples of their

importance and treatment can be found elsewhere in this encyclopedia.

Climate sets the underlying growing conditions for plants in land restoration and in a broad sense determines the range of species able to survive and grow in a particular location. Landform is a prime consideration in restoration because of the need to achieve slope stability and effective erosion control. The important components of landform that limit restoration are slope angle, elevation, aspect, and drainage. Any clearing or significant disturbance of vegetation alters hydrology. Increased runoff is usually the first on-site response, followed by erosion, and then downstream consequences such as flooding. Other changes in hydrology include

altered profile water storage and increased groundwater recharge.

Substrate Properties

Physical properties: Land degradation often alters the soil physical conditions in ways that decrease its suitability for plant establishment and growth, particularly by changing water storage and availability. Water erosion may strip away topsoil; decrease soil depth; and expose subsoil material with different texture, lower organic matter levels, and poorer soil structure. Wind erosion selectively removes clay and humus from the soil surface, increasing the prevalence of coarse materials. Sand deposits from wind erosion may bury topsoil and vegetation.

Chemical properties: Nutrient levels, acidity, alkalinity, salinity, sodicity, and toxic metals and organic compounds are the chemical factors in soils and substrates that can limit plant growth and therefore hamper effective land restoration. Deficiencies of nutrient elements can be diagnosed or predicted by several means.^[9] Once identified, deficiencies can generally be corrected by fertilizer application. The selection of plants with high tolerances of the toxicities from acidity or heavy metals is often the most cost-effective strategy of revegetation even if it means planting a different species mix to that which existed before disturbance.^[10] However, the preferred modern practice in mining is to identify adverse materials and selectively place them in waste dumps where there is no contact with the root zone.

Biological properties: Mining activities and topsoil storage have quite profound negative effects on microbial biomass and activity.^[11] Restoration of microbial biomass may take 7–10 years on revegetated mine sites. Careful handling and reuse of topsoil can often minimize the problems associated with decreased biological activity of soil after land restoration. Maximum activity is retained if topsoil is reused immediately without a period of storage.

Plant Establishment

The prerequisite for effective plant establishment is to ameliorate substrate conditions most likely to limit plant growth. In mining, the aim is to plan mine operations, so that the most benign materials are placed on top of surfaces to be revegetated. This minimizes the need for expensive amelioration of adverse soil conditions.

Plant establishment relies on three strategies, either individually or in combination. Topsoil spreading will often be the most reliable strategy if it is available because it contains seed native to the area, adds soil organisms, and creates a seedbed for seed germination and establishment. However, in some plant communities, seed is stored in the plant canopy rather than in the soil. For the restoration of

these communities, cutting branches and spreading them on the soil surface have proved effective. Nursery-raised seedlings will generally be reliable for plant establishment but are relatively expensive. In addition, there are often many species that are difficult to propagate from seed in the nursery, or they generally regenerate by vegetative means. Direct seeding is usually cheaper than other options although there is significant cost associated with the collection of native seed. Establishment of seedlings especially from direct seeding is hampered by weed competition and grazing by herbivores.

Sustainability

After plant establishment, some measure of the success of restoration is needed. According to Hobbs and Norton,^[5] the following ecosystem characteristics should be considered when setting goals for land restoration: composition, structure, pattern, heterogeneity, function, species interactions, and dynamics and resilience. This raises the need for reliable low-cost indicators of success.^[11] Most indicators have relied on composition and structure of vegetation. Microbial biomass has been advocated as an indicator of nutrient cycling functions. Indicators for pattern, heterogeneity, dynamics, and resilience are less advanced in their development, in part because few land restoration activities have run for long enough to measure these characteristics let alone develop indicators for them.

CONCLUSION

Land is a finite resource. Every attempt must be made to prevent or minimize land degradation. In addition, the restoration of already degraded land needs to be undertaken. Land restoration is a relatively new science that is increasingly informed by the emerging discipline of restoration ecology. Land restoration occurs at a range of different scales. Mine rehabilitation has developed many successful practices of land restoration at a site-specific scale. The challenge ahead is to develop effective land restoration practices at a regional or landscape scale. Most success at both scales has been concerned with restoring key ecosystem functions such as nutrient cycling and water balance.

REFERENCES

1. Lal, R.; Blum, W.H.; Valentine, C.; Stewart, B.A., Eds. *Soil Restoration*; Advances in Soil Science; CRC Press: Boca Raton, 1992; Vol. 17, 456.
2. Lal, R.; Blum, W.H.; Valentine, C.; Stewart, B.A., Eds. *Soil Degradation*; CRC Press: Boca Raton, 1990; Vol. 11, 345 pp.
3. Lal, R.; Blum, W.H.; Valentine, C.; Stewart, B.A., Eds. *Methods for Assessment of Soil Degradation*; CRC Press: Boca Raton, 1998.

4. Conacher, A.J.; Conacher, J. *Rural Land Degradation in Australia*; Oxford University Press: Melbourne, 1995; 170 pp.
5. Hobbs, R.J.; Norton, D.A. Towards a conceptual framework for restoration ecology. *Restor. Ecol.* **1996**, *4*, 93–110.
6. Aronson, J.; LeFoc'h, E.; Floret, C.; Ovalle, C.; Pontanier, R. Restoration and rehabilitation of degraded ecosystems in arid and semiarid regions. II. Case studies in Chile, Tunisia and Cameroon. *Restor. Ecol.* **1993**, *1*, 168–187.
7. McRae, S.G. Land reclamation after open-pit mineral extraction in Britain. In *Remediation and Management of Degraded Lands*; Wong, M.H., Wong, J.W.C., Baker, A.J.M., Eds.; Lewis Publishers: Boca Raton, 1999; 47–62.
8. Koch, J.M.; Ward, S.C. The technology of bauxite mine rehabilitation in the jarrah forest of western Australia. In *Proceedings of Remade Lands 2000, The International Conference on Remediation and Management of Degraded Lands, Fremantle, Nov 30– Dec 2, 2000*; Brion, A., Bell, R.W., Eds.; Promaco Conventions: Canning Bridge, 2000; 38–39.
9. Bell, R.W. Chapter 16 diagnosis and prognosis of soil fertility constraints for land restoration. In *Remediation and Management of Degraded Lands*; Wong, M.H., Wong, J.W.C., Baker, A.J.M., Eds.; Lewis Publishers: Boca Raton, 1999; 163–173.
10. Ho, G.E.; Samaraweera, M.K.S.A.; Bell, R.W. Revegetation directly on salt-affected gold ore refining residues—A case study. In *Remediation and Management of Degraded Lands*; Wong, M.H., Wong, J.W.C., Baker, A.J.M., Eds.; Lewis Publishers: Boca Raton, 1999; 123–135.
11. Jasper, D.A.; Sawada, Y.; Gaunt, E.; Ward, S.C. Indicators of reclamation success-recovery patterns of soil biological activity compared to remote sensing of vegetation. In *Land Reclamation: Achieving Sustainable Benefits*; Fox, H.R., Moore, H.M., McIntosh, A.D., Eds.; A. A. Balkema: Rotterdam, 1998; 21–24.

Land Use: Historic

J. Douglas Helms

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Washington, District of Columbia, U.S.A.

Hari Eswaran

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS) (Retired), Washington, District of Columbia, U.S.A.

Paul Reich

Natural Resources Conservation Service, U.S. Department of Agriculture (USDA-NRCS), Washington, District of Columbia, U.S.A.

Abstract

Land, as a fundamental factor in production, has always been integrally related to the economic growth of a country. Until or about the time of the industrial revolution, land resources and their productivity constituted much of a society's wealth. Societies use land differently based on their culture, technology, and political, and economical aspirations. European colonization of the much of the world since Columbus' explorations radically altered global land use. In addition to mineral exploration and mining, the colonial powers introduced new agricultural production systems to serve their political and economic goals. The land use patterns in the developing countries were, to a large extent, determined by the resources and raw material needs of the colonial powers. The soil and climatic endowments and the cultural habits of the people, of course, control the specific land use within a region or country.

INTRODUCTION

There were many events and reasons that resulted in land use changes. It is not the intent here to examine these in detail. The purpose of this entry is to evaluate, on a global perspective, some major changes that took place in the last few centuries and evaluate how these have resulted in the global land use patterns. This entry will focus on agricultural land uses, although it is recognized that increasing urbanization is a serious issue for preserving valuable agricultural land.

ORIGINS OF AGRICULTURE

The term "land use" implies human choices and actions that change and alter the landscape. Initially, agriculture was the main use of the land, and land was valued for its agricultural productivity. Agriculture altered land use in multiple and reinforcing ways. First, humans began altering the land surface as more of it was devoted to growing domesticated plants and raising domesticated animals. Next, agriculture supported an increasing population, which in turn brought more land into production. While

it is true that some hunter-gatherers lived in communities, agriculture freed people to pursue other livelihoods and to congregate in towns and cities and to be supplied with food from the countryside.

Scholars have debated the origins and spread of agriculture. Jared Diamond's *Guns, Germs, and Steel* argues for five centers where food production arose independently: Southwest Asia (also referred to as the Near East or Fertile Crescent); China; Mesoamerica (central and southern Mexico and areas of Central America); the Andes of South America, and the Eastern United States. There are four other possibilities for the independent agricultural origins: the Sahel of Africa, tropical West Africa, Ethiopia, and New Guinea.^[1]

Wheat had its origins in Mesopotamia in the Tigris and Euphrates river valleys about 10,000 years ago. By about 1000–3000 B.C., Egyptians were already consuming yeast-leavened bread. Corn is considered to have its origin in the Mexican plateau. Native Americans knew corn, squash, and beans as the three sisters. Archeological studies in Hunan province, China suggests that cultivation of rice dates back 6000–7000 years and quickly spread across the East Asian crescent from Assam in Northeast India to Yun-nan, China. Soon it became a dominant cultivar, occupying

the major river valleys and making it the most important cereal grain in Asia. It is a staple food for more than three billion people, with most Asians depending on it for their caloric intake. These—wheat, corn, and rice—are the three basic grain crops that together dominate the agricultural scene of the world.

FLUCTUATIONS IN AGRICULTURAL USES

However, the expansion of cropland and population did not proceed unabated at all times and places. The Black Death of 1348–1349 was followed by repeated, smaller epidemics. During the 15th century, there were fewer people in the European countryside than at any time since about 1150. By the mid-15th century, probably 60% of the plowed cropland in the Parisian basin was no longer planted. Forests reclaimed some land, and pasture replaced some cropland. As a result of less intensive land use, peasants enjoyed more meat and higher quality breads in their diet because of the depopulation of the countryside.^[2] Native Americans succumbed to disease such as smallpox after European contact.^[3] The “Indian old fields” of the Eastern woodlands, the Indian mounds of urban Mississippian settlements, and the abandoned irrigation works of the Southwest attested to Native Americans’ land uses.^[4] Indians repeatedly used fire to renew grasses for grazing animals and left open woodland, where little underbrush hindered travel of the arriving Europeans.^[5]

MAKING LAND MORE PRODUCTIVE

As there is a strong relationship between natural chemical elements in the soil and the ability to feed people,^[6,7] early economists such as Malthus saw population as being limited by the amount of arable land. Through the centuries and especially in the 20th century, humans succeeded in producing more per acre, so that the correlation between the amount of arable land and the food production is no longer as strong as it once was. Various societies learned to rotate crops utilizing legumes. Irrigation and drainage, often on fertile soils, have increased production, as have plant selection and breeding and pest protection. As agricultural chemists began understanding more about plant growth in the 19th and 20th centuries, humans began mining guano and phosphorous for transportation around the world. The most far-reaching development affecting land use for agriculture has been the development of the Haber–Bosch process to synthesize ammonia. By one estimate, these nitrogen compounds provide between 60% and 80% of the nutrients available to the most commonly produced grain crops and are perhaps equal to half of the total nitrogen fixed by all bacteria in natural resource systems.^[8]

LEGACY OF COLONIALISM

The creation of new nations from the former European colonies triggered vast sociopolitical changes that have altered land use. In places where traditional agricultural systems had been sufficient to feed a small population, western medical advances expanded the population.^[9,10] Many new nations could not, or did not, build agricultural supply and production systems, including agricultural chemicals, to grow additional food. With more pressure on production, the result was too often indiscriminate exploitation of land in most developing countries. The considerations of ecosystem integrity and the need to maintain biodiversity are not considered in many developing countries. Large-scale deforestation as in Southeast Asia and South America and expansion of cultivation into wildlife reserves became rampant for several reasons, including increasing population, greater number of people searching for land, and economics. In the richer western countries, the application of soil and water conservation technology and financial support from governments initiated a land use system more dependent on science.^[7] In the centrally controlled economies of the Former Soviet Union and other socialist countries, land use was strictly determined by the state. Although land use was designed according to landscape conditions, pressures to enhance productivity resulted in less care for the land, leading to severe land degradation.

GLOBAL LAND USE PATTERNS

In a modeling study, Ramankutty and Foley demonstrate the extent of natural ecosystems that were converted for agricultural use during the last three centuries.^[11] Fig. 1 is based on the empirical data they generated and shows the changing areas for forest, grassland, and cultivated land in the world and for several regions. The pattern of land use change in Africa (Fig. 1B) mimics the global pattern (Fig. 1A) due to the availability of land for agricultural expansion. Fig. 1C shows the situation of South Asia, a region where limits of available land for agriculture have been met. The remaining forest and Savanna land in this region have many constraints for agricultural use. Fig. 1 shows the case of the former Soviet Union countries where major change in land use has been the conversion of Savanna to agricultural use. The forested land of this region has low temperature as a major constraint for grain production and so less-forested land is converted for agricultural purposes.

Land use is not only the growing of crops or grazing of animals but also the architecture of fields. The layout and distribution of fields came about initially for efficiency of farm practices (access to the fields by the farmers) and later by concepts of soil and water conservation. Field layout had its best design in modern irrigated systems, and these

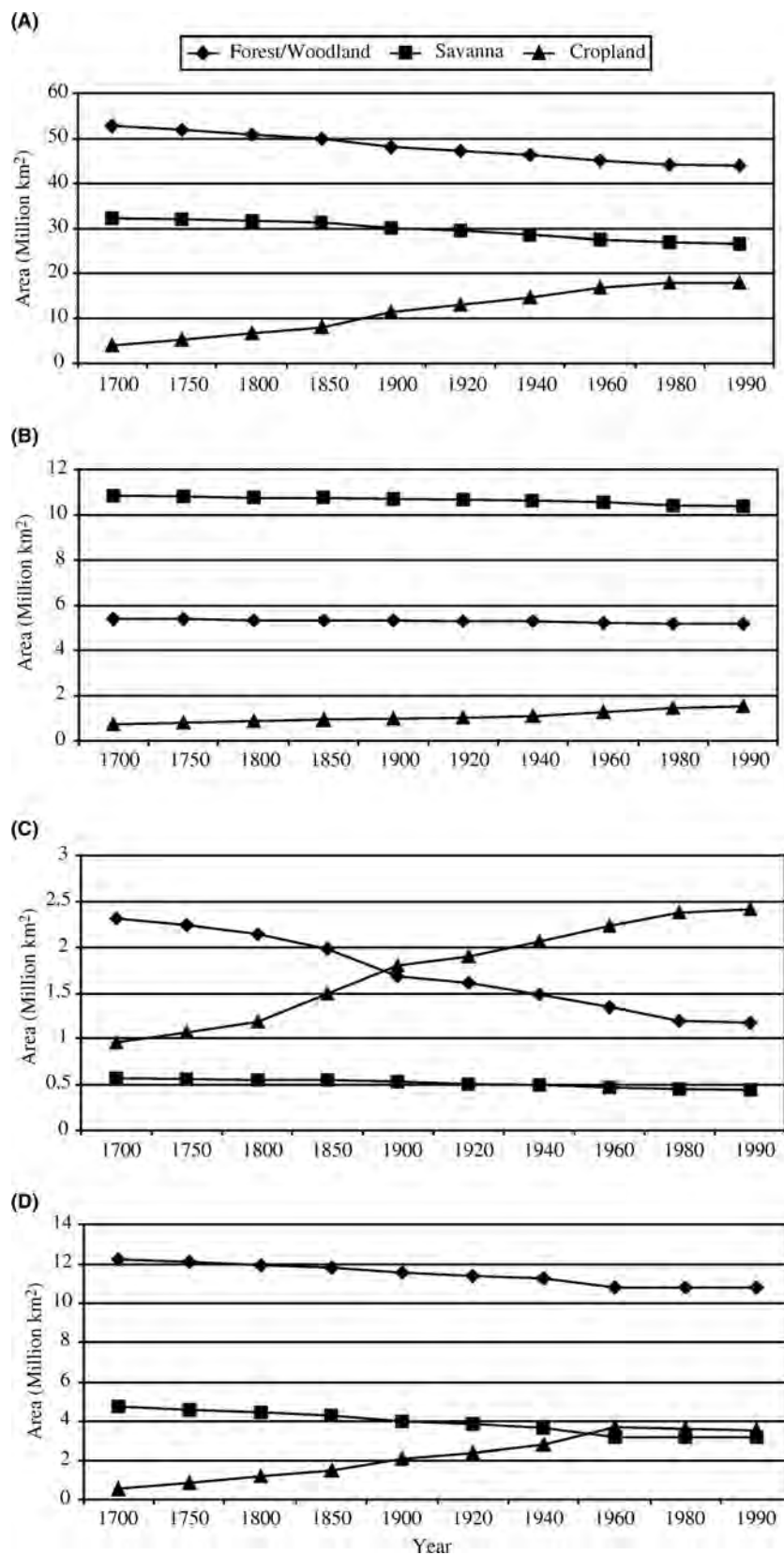


Fig. 1 Change in land cover area from 1700 to 1990. Graph (A), Global; Graph (B), Africa; Graph (C), South Asia; Graph (D), Former Soviet Union countries.

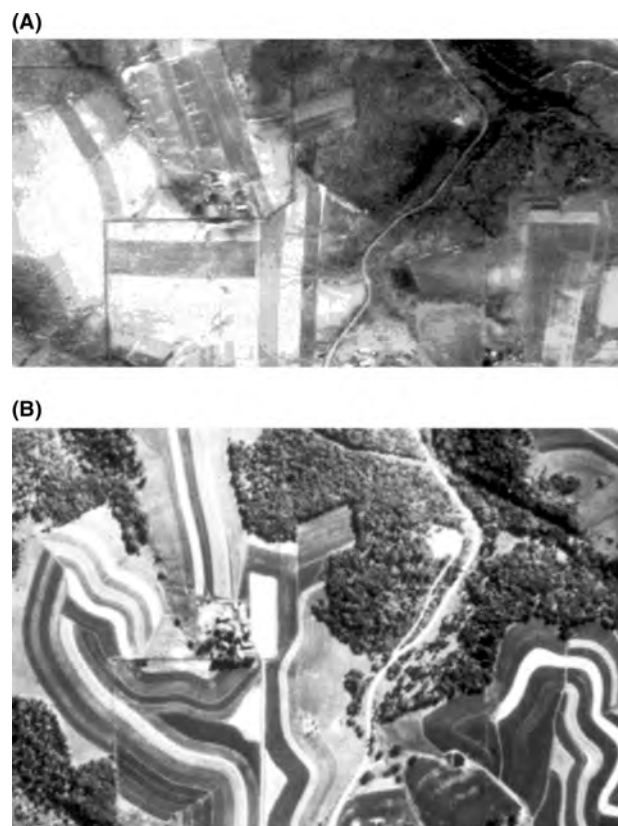


Fig. 2 (A) Soil Conservation Service demonstration project at Coon Creek, La Crosse County, Wisconsin, U.S.A. 1934. (B) Soil Conservation Service demonstration project at Coon Creek, La Crosse County, Wisconsin, U.S.A., 1967 (of approximately the same area of the 1934 photograph in A).

Source: (A) Photograph is Records of the Natural Resources Conservation Service, negative 114-2501-167, National Archives, 1934. (B) Photograph is negative BII-2HH-291, Farm Services Agency, Salt Lake City, Utah, U.S.A., 1967.

are evident even in developing countries. However, field size, shape, and patterns show a lack of organization in traditional systems. Many western countries went to great extent to reallocate land to enhance efficiency and economies of farming. In Belgium, e.g., this process took place in the 1960s and was called “ruilverkaveling” or land reallocation. In other countries, economic pressures induced such changes. Fig. 2A and B shows an example of change in field architecture over a 50-year period in the United States.

The beginning of the new millennium is characterized by a number of conditions that affect or even control agricultural sustainability. The world population of six billion people is projected to increase to nine billion in the next few decades, and much of the increase is to take place in the countries that can least afford to enhance the quality of life for population.^[7] Agricultural expansion has taken place at the expense of other kinds of land use (Fig. 1). To obtain an idea of the kinds of soils where conversion of land to agriculture was taking place, the authors combined a global land use/land cover map with the soil map of the world. Table 1 shows the distribution of land use as a function of soils.

About 21% of the global land is cultivated and the dominant soils exploited are the Entisols and Inceptisols (mostly for rice cultivation in Asia); Alfisols, Mollisols, and Ultisols (for upland grain crops); and Oxisols (for plantation crops in the tropics). Some of the semiarid grasslands and shrublands in the developing countries are used for grazing, usually nomadic. About 658,000 km² of land (0.5%) is used for urban purposes. The actual amounts vary between countries, and, as shown in the study of Nizeyimana et al. the better quality lands succumb to urbanization faster. This study has shown that the highly productive soils of the United States have a higher level of urbanization.^[12]

Table 1 Land use and land cover as a function of soils (data in 1000 km²).

Soil orders	Urban	Cultivated land	Grass land	Shrub land	Shrubland–Grassland	Savanna	Forest	Wetland	Barren	Tundra
Alfisols	142	5,708	652	454	24	2,669	3,296	26	97	11
Andisols	15	207	93	80	18	78	437	3	17	53
Aridisols	49	1,286	1,984	5,851	216	1,013	582	2	4,551	66
Entisols	94	3,365	1,695	4,125	133	2,516	2,156	62	7,108	65
Gelisols	2	104	271	448	176	76	4,204	787	246	4,208
Histosols	3	240	48	24	4	66	1,068	45	2	20
Inceptisols	89	3,748	504	509	97	1,865	2,217	20	99	46
Mollisols	102	4,811	2,195	611	40	323	1,003	3	19	47
Oxisols	13	2,007	88	48	5	1,923	5,905	6	12	4
Spodosols	33	450	40	102	5	32	2,487	19	3	77
Ultisols	86	3,505	246	265	4	1,834	5,585	17	12	13
Vertisols	21	1,112	365	493	2	949	189	14	15	0
Rocky land	7	736	2,006	1,019	539	579	3,448	275	952	2,901
Shifting sands	3	157	622	1,535		427	13	0	2,557	0
Total	658	27,436	10,809	15,564	1,263	14,350	32,592	1,279	15,690	7,514

CONCLUSION

There are a number of factors that determine land use patterns in a country. In a global perspective, historical land use reflects sociocultural evolution, the impact of pests, diseases, conflicts between nations, developments in technologies, and the pressures introduced by population increases. The conversion of land for agricultural uses is at an increased pace particularly in developing countries although the reverse trend of restoring the original habitat is taking place in some countries such as the U.S. Developments in geographic information technology with appropriate models are enabling the study of evolution of land use. These developments will provide a better analysis in the forthcoming years.

REFERENCES

1. Diamond, J. *Guns, Germs, and Steel: The Fates of Human Societies*; W.W. Norton: New York, 1997.
2. Dyer, C. Rural Europe. In *The New Cambridge Medieval History*; Allmand, C., Ed.; Cambridge University Press: Cambridge, 1998.
3. Crosby, A.W. *The Columbian Exchange: Biological and Cultural Consequences of 1492*; Greenwood Press: Westport, 1973.
4. Hurt, R.D. *Indian Agriculture in America: Prehistory to the Present*; University Press of Kansas: Lawrence, 1987.
5. Helms, D. Soil and southern history. *Agr. Hist.* **2000**, *74* (Fall), 723–758.
6. Buol, S.W.; Eswaran, H. Assessment and conquest of poor soils. In *Proceedings of Workshop on "Adaptation of Plants to Soil Stresses"*; INTSORMIL Pub. 94-2; University of Nebraska: Lincoln, 1993; 17–27.
7. Eswaran, H.; Reich, P. A preliminary assessment of the human impact on land systems of the world. *A World Soils Agenda—Discussing International Actions for the Sustainable Use of Soils*, 2002; 38–42.
8. Smil, V. *Enriching the Earth: Fritz Haber, Carl Bosch, and the Transformation of World Food Production*; The MIT Press: Cambridge, 2001.
9. Nye, P.H.; Greenland, D.J. Greenland. In *The Soil under Shifting Cultivation*; Technical Communication, 51; Commonwealth Bureau of Soils: Bucks, 1960.
10. Sanchez, P. *Properties and Management of Soils in the Tropics*; John Wiley and Sons, Inc.: New York, 1976.
11. Ramankutty, N.; Foley, J.A. Estimating historical changes in global land cover: Croplands from 1700 to 1992. *Global Biogeochem. Cy.* **1999**, *13*, 997–1027.
12. Nizeyimana, E.L.; Peterson, G.W.; Imhoff, M.L.; Sinclair, H.R.; Waltman, S.W.; Reed-Margaretan, D.S.; Levine, E.R.; Russo, J.M. Assessing the impact of land conversion to urban use on soils with difference productivity levels in the USA. *Soil Sci. Soc. Am. J.* **2001**, *65*, 391–402.

Land Use: Land Cover

Andrew K. Skidmore

Division of Agriculture, Conservation and Environment, International Institute for Aerospace Survey and Earth Sciences, Enschede, the Netherlands

Abstract

The mapping and monitoring of land cover and land use is mature, as production systems for making maps are now routine and available globally. But users continue to demand more information, of a higher accuracy and precision, and at a cheaper price. Innovative methods for mapping land cover at higher levels of accuracy continually develop. At the same time, techniques are becoming more efficient, thereby decreasing costs per mapped unit. This gain in accuracy and efficiency comes from higher resolution satellite sensors as well as improvements in computer algorithms.

INTRODUCTION

Land use and land cover are components of land; land is defined as:^[1]

- Any delineable area of the earth's terrestrial surface, involving all attributes of the biosphere immediately above or below this surface, including those of the near-surface climate, the soil and terrain forms, the surface hydrology, near-surface layers and associated ground water and geohydrological reserves, the plant and animal populations, the human settlement patterns, and human activity (terracing, water storage or drainage structures, roads, buildings, etc.).

Thus, the main characteristics of land are terrain (e.g., landform, soil, and geomorphology), biotic elements (e.g., fauna and flora), results of past land use, and climate. Numerous definitions of land use exist—in general they refer to management activities related to a tract of land or an ecosystem:^[2]

- A series of operations in land carried out by humans with the intention to obtain products and/or benefits through using land resources.

In contrast, land cover is defined as a 3-D element of land, which occurs on the surface of the land:^[1]

- The vegetation (natural or planted) or man-made constructions (buildings, etc.) that occur on the earth's surface. Water, ice, bare rock, sand, and similar surfaces also count as land cover.

Land cover is identified by direct observation, while land use implies that information is determined through an interview or using similar indirect methods. The input of labour,

or implements used in land use, may not be observable but may impact on the land cover. Thus remote sensing can be used for land-cover mapping but not for land use mapping. However, land use may be determined indirectly from land cover; in other words, land use may be mapped using land cover as the intermediate step.^[3] Examples of land cover include wheat, forest, and water. For wheat, the land use may be “grain production,” “grazing,” or “green fertiliser.” Many people confuse the definitions of land use and land cover.^[4]

MAPPING LAND COVER

Prior to 1945, land resources were surveyed by field sampling, often by examination or measurement along transect lines, with the intervening areas filled in by interpolation. An example of this approach, updated using modern Kriging techniques, is shown in [Fig. 1](#), where a series of soil pits were dug across a hillside, and the soil chemistry for each pit analyzed in the laboratory.^[5]

By using aerial photographs, land managers can map visible features in their entirety, allowing field work to concentrate on dubious points or on quantitative sampling within (already) recognized units or strata. The process of manually extracting information from aerial photographs is usually referred to as “interpretation.”

To interpret aerial photographs, one uses a range of clues under the principle referred to as “convergence of evidence.” For example, using this integrated approach, features such as the drainage pattern, or changes in vegetation, may suggest differences in geology, land form, or soil type. An interpreted photograph for a forested area in Australia is shown in [Fig. 2](#). The line work interpreted on an aerial photograph is transferred to a cartographic base map by the “photogrammetric” process.^[6–9] Further information on the application of aerial photographs to a

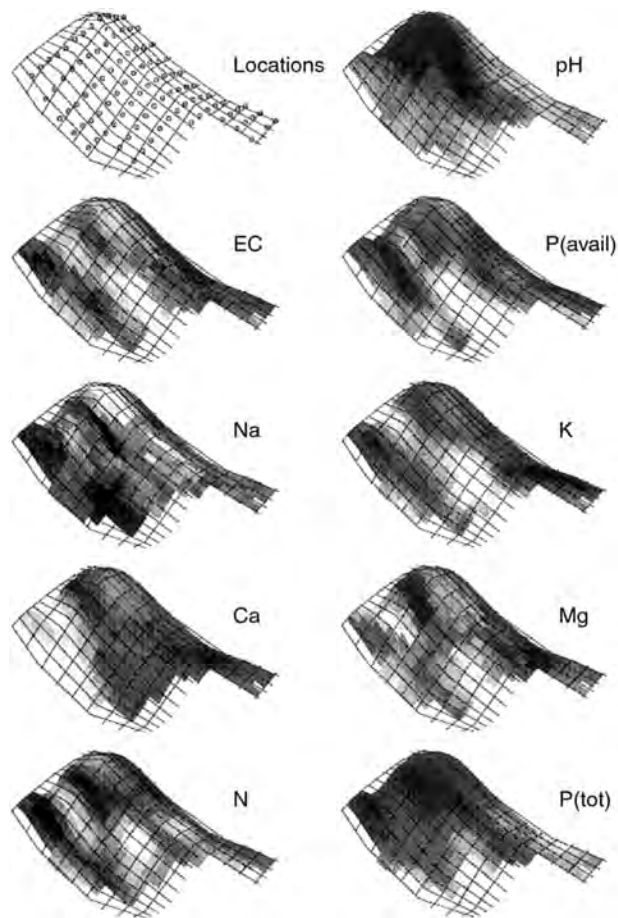


Fig. 1 Sample locations and interpolated soil properties for a forested hillside in southeast New South Wales, Australia.

variety of land cover mapping applications can be found in the study by Colwell^[10]

The main problem in using aerial photographs is a high labor requirement. In addition, the assignment of boundaries is subjective, leading to inconsistencies by individuals as well as between interpreters, especially over time. Interpretation may be automated using computers; the cover-type classes are labeled on the basis of the brightness values recorded in many bands at a pixel, and the process is fast and objective. Use of spatial recognition (e.g., of field boundaries) is, however, not made. There are many computer techniques available to derive land cover maps from imagery, and these can be split into two broad strategies, namely, unsupervised and supervised methods.

With unsupervised processing, pixels in an image are assigned to spectral classes without users having foreknowledge of the existence or names of the classes. The unsupervised classification uses clustering algorithms to merge pixels into common spectral groups, and then the analyst names the clusters. In other words, the analyst takes the classified map produced by the unsupervised classifier and then assigns a label to each class.^[11]

Supervised classification is where the analyst identifies cover-type classes in the field or from existing maps and then uses this information to identify the classes on the image. The supervised classification method involves three phases. First, gathering prototype (training) data so that areas can be identified on the image. Second, statistics are generated describing the shape and structure of the training area data in the image feature (multispectral) space. Third, the computer labels each unknown pixel into one of the classes based on the class statistics. One of the most commonly used supervised methods is the maximum likelihood classifier.^[11]

An innovation in the 1970s was to combine imagery with ancillary geographic data creating geographic information systems (GIS), in order to increase the accuracy of land cover maps.^[12–15] The GIS handle geographical data, both in the form of “digital maps” and as “attribute data.” Attributes are, for example, tabular data connected to geometric features such as points (grids), lines, and surfaces on a map.

However, soil maps in particular continued to disappoint land managers, especially where vegetation cover obscured the soil response.^[16] To improve the accuracy of soil maps, expert systems based on known relationships between soil characteristics and landscape features were developed.^[17] An example of an expert system classification is given in Fig. 3, where the relationship between soil and landscape (defined in rule bases by soil scientists) is used to map land cover from landscape data (such as elevation, slope, aspect, solar radiation, parent material, etc.) and remotely sensed imagery. The mapping of land cover with expert systems yields higher accuracy maps compared with more conventional methods, such as a supervised classifier.^[17]

LAND-COVER MONITORING

Changes in land cover may be rapid (e.g., clearing a forest for agriculture) or relatively slow (e.g., tree damage and death due to acid rain). Changes may also be predictable and cyclical (e.g., seasonal variation in tree leaf cover) or unpredictable (e.g., fire damage to vegetation). Using a single aerial photograph or satellite image, an interpreter may identify patterns (either manually or by computer enhancement and classification) indicating that land cover has altered. It is possible to monitor land cover change by mapping an area on two dates and then calculating the difference in land cover with a GIS, using a technique often referred to as “postclassification comparison.”

Various techniques have been developed to compare multirate remotely sensed imagery.^[18] The techniques have been grouped into categories of change detection method by various authors. Postclassification comparison has been described above and involves a comparative analysis of independently classified maps. Other

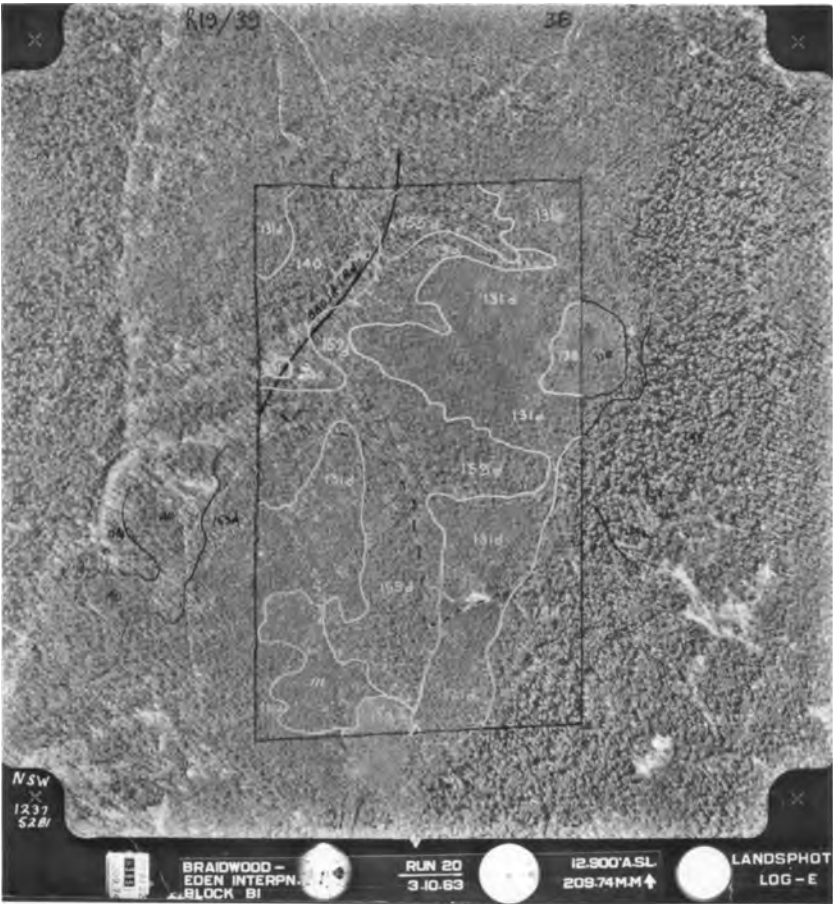


Fig. 2 An interpreted black and white aerial photograph showing forest-type boundaries on the south coast of New South Wales Australia.

methods include multivariate classification, which is a mathematical combination of two images, as well as simultaneous analysis of multitemporal data.^[19–21] Innovative methods for change detection using wavelets have been suggested.^[22]

All authors agree that accurate geometric registration of imagery is essential for good results. In addition, there is no single optimal method for change detection, as contradictory results have been generated under different (empirical) conditions. However, it is obvious that dramatic differences

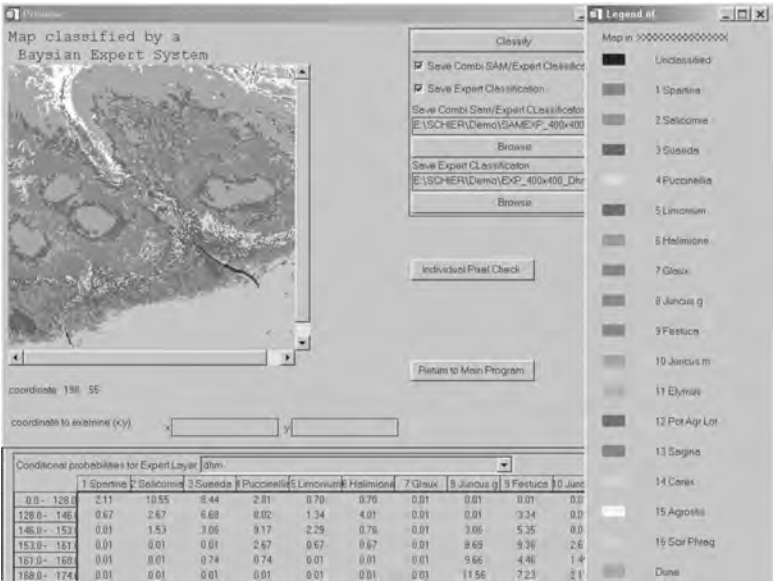


Fig. 3 Expert system interface written in IDL. The accuracy of this map, showing coastal wetland vegetation, is 62%.

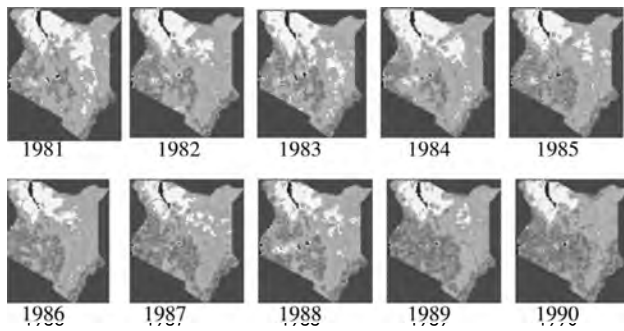


Fig. 4 Annual average NDVI calculated for Kenya from 1981 to 1990. A red to green color indicates high biomass and high vegetation vigor (e.g., 1990 was a wet year), while yellow indicates drought conditions (e.g., 1984).

occur between images (see Fig. 4), which show the average annual normalized difference vegetation index (NDVI) derived for Kenya from 1981 to 1990. In this series of figures, a red to green color indicates high biomass and high vegetation vigor (e.g., 1990 was a wet year), while yellow indicates drought conditions (e.g., 1984). Such data sets may be processed using the change detection techniques described above.

ACCURACY OF LAND-COVER MAPS

The main products from a GIS, as well as from remotely sensed images, are maps (or a summary of information from maps). Possibly the most important question to be asked by a user is “how accurate is the spatial information?” Quantifying error in maps is based on various statistics derived from the error matrix (also called a contingency table or confusion matrix), which was first expounded for remotely sensed data in the 1970s.^[12,23] The aim of the error matrix is to estimate the mapping accuracy (i.e., the number of correctly mapped points) within an image or map. An error matrix is constructed from points sampled from the map. The reference data collected in the field (also called ground-truth data) are compared with the classified (or image) data.

Two points are particularly relevant to the discussion of accuracy. First, should we map class boundaries (e.g., Fig. 2) or should continuous surfaces (e.g., Fig. 1) be used to represent a landscape? The standard method of calculating error matrices relies on class boundaries. However, as most landscapes are actually continuums, and as it is impossible to map the subtlety inherent in a continuous surface using classes, new methods for assessing the accuracy of a continuous surface are required. Second, when attempting to verify maps, a mapped area becomes a historical record. A land cover map classified using an old image is impossible to check (e.g., a farmer may plant different crops each year). A complete description of accuracy assessment procedures and issues is given by Skidmore.^[24]

LAND USE DATABASES

As noted above, land cover may be derived from remotely sensed data and aerial photographs. It was remarked earlier that land cover and land use are frequently misunderstood and confused. A land use system is defined as:^[3]

- A specific land use, practised during a known period on a known and contiguous area of land with reasonably uniform land characteristics.

The importance of the land use concept is that “use” is often not detectable on an image, e.g., consider the land use “applying fertilizer.” The impact of this use operation on land cover may not be observable (e.g., due to a short time period between application and observation, or the spatial resolution of the remote-sensor system being too coarse). Thus, information on land use requires an interview with the person who controls or carries out the land use. Note that land use may often be correlated with impacts on actual land cover, allowing land use to be mapped by using land cover as an intermediate step.

Land use data collected by numerous agricultural and regional development projects are generally not poor accessible. These data are hidden away in survey reports, and when available, may be difficult to use due to non-standard descriptors. A database has been developed^[3] to consistently record land use data (see <http://www.itc.nl/ha2/ace/>).

CONCLUSION

The mapping and monitoring of land cover and land use is mature, as production systems for making maps are now routine and available globally.^[25] But users continue to demand more information, of a higher accuracy and precision, and at a cheaper price. Innovative methods for mapping land cover at higher levels of accuracy continually develop. At the same time, techniques are becoming more efficient, thereby decreasing costs per mapped unit. This gain in accuracy and efficiency comes from higher resolution satellite sensors as well as improvements in computer algorithms. The trend is clear; more sophisticated land cover products at lower prices and available all over earth.

REFERENCES

1. FAO. *Integrated Approach to the Planning and Management of Land Resources*; Food and Agriculture Organisation of the United Nations (FAO): Rome, 1994.
2. De Bie, C.A.; van Leeuwen, J.A.; Zuidema, P.A. *The Land Use Database*; ITC: PO Box 6, 7500 AA Enschede: The Netherlands, 1996.
3. De Bie, C.A.J.M. *Comparative Performance Analysis of Agro-Ecosystems*; Ph.D. dissertation; Wageningen University and ITC Enschede, ITC publication no. 75, Wageningen, 2000; 232 pp.

4. Maidment, D.R. GIS and hydrologic modeling. In *Environmental Modeling with GIS*; Goodchild, M.F., Parks, B.O., Steyaert, L.T., Eds.; Oxford University Press: Oxford, 1993.
5. Varekamp, C.; Skidmore, A.K.; Burrough, P.A. Using public domain geostatistical and GIS software for spatial interpolation. *Photogramm. Eng. Rem. S.* **1996**, *62* (7), 845–854.
6. Wolf, P.R. *Elements of Photogrammetry*; McGraw-Hill: New York, 1974.
7. Avery, T.E.; Berlin, B.G.L. *Fundamentals of Remote Sensing and Airphoto Interpretation*, 5th Ed.; Macmillan: New York, 1992.
8. Baltsavias, E.; Stallmann, D. From satellite images to GIS with digital photogrammetry using SPOT data. In *Proceedings of EGIS'92*, Munich, Germany, EGIS Foundation: Paris, 1992; 945–946.
9. Kraus, K. *Photogrammetry*; Ummeler: Bonn, 1993.
10. Colwell, R.N. *Manual of Remote Sensing*; ASPRS: Falls Church, 1983.
11. Richards, J.A. *Remote Sensing Digital Image Analysis: An Introduction*, 2nd Ed.; Springer: Berlin, 1993.
12. Hoffer, R.M. *Natural Resource Mapping in Mountainous Terrain by Computer Analysis of ERTS-1 Satellite Data*; Laboratory of Remote Sensing, Purdue University: West Lafayette, 1975.
13. Strahler, A.H.; Logan, T.L.; Bryant, N.A. Improving forest cover classification accuracy from landsat by incorporating topographic information. In *Proceedings of the 12th International Symposium on Remote Sensing of Environment*; ERIM: Ann Arbor, 1978; Vol. 2, 927–942.
14. Hoffer, R.M.; Fleming, M.D.; Bartolucci, L.A.; Davis, S.M.; Nelson, R.F. *Digital Processing of Landsat MSS and DMA Topographic Data to Improve Capabilities for Computerized Mapping of Forest Cover Types*; Laboratory of Remote Sensing, Purdue University: West Lafayette, 1979.
15. Strahler, A.H.; Estes, J.E.; Maynard, P.F.; Mertz, F.C.; Stow, D.A. Incorporating collateral data in landsat classification and modelling procedures. In *Proceedings of the 14th International Symposium on Remote Sensing of Environment*, San Jose, Costa Rica; ERIM: Ann Arbor, 1980; Vol. 2, 1009–1026.
16. Tucker, C.J.; Miller, L.D. Soil spectra contributions to grass canopy spectral reflectance. *Photogramm. Eng. Rem. S.* **1977**, *43* (6), 721–726.
17. Skidmore, A.K.; Watford, F.W.; Luckananurug, P.; Ryan, P.J. An operational expert system for mapping forest soils. *Photogram. Eng. Rem. S.* **1996**, *62* (5), 501–511.
18. Addink, E. *Change Detection with Remote Sensing*; Wageningen University: Wageningen, 2001; 113 pp.
19. Singh, A. Digital change detection techniques using remotely-sensed data. *Int. J. Remote Sens.* **1989**, *10* (6), 989–1003.
20. Mas, J.F. Monitoring land-cover changes: a comparison of change detection techniques. *Int. J. Remote Sens.* **1999**, *20* (1), 139–152.
21. Coppin, P.R.; Bauer, M.E. Digital change detection in forest ecosystems with remote sensing imagery. *Remote Sens. Rev.* **1996**, *13*, 207–234.
22. Carvalho, L. *Mapping and Monitoring Forest Remnants*; Wageningen University: Wageningen, 2000.
23. Cohen, J. A coefficient of agreement for nominal scales. *Educ. Psychol. Meas.* **1960**, *20* (1), 37–46.
24. Skidmore, A.K. Accuracy assessment of spatial information. In *Spatial Statistics for Remote Sensing*; Stein, A., van der Meer, F., Gorte, B., Eds.; Kluwer Academic Publishers: Dordrecht, 1999; 197–209.
25. Skidmore, A.K. A taxonomy of environmental models. In *Environmental Modeling Using GIS and Remote Sensing*; Skidmore, A.K., Ed.; Taylor and Francis: London, 2002; 8–24.

Land Use: Planning and Geographic Information System (GIS)

Egide Nizeyimana

*Department of Agronomy and Environmental Resources Research Institute,
Pennsylvania State University, University Park, Pennsylvania, U.S.A.*

Jacob Opadeyi

*Department of Surveying and Land Information, University of West Indies,
Saint Augustine, Trinidad and Tobago*

Abstract

Geographic information system (GIS) and related technologies (e.g., remote sensing and global position system) have proven to be a valuable tool in land use planning activities. The GIS approach is important in this area because it provides functions to capture, store, organize, and analyze spatially referenced data. Moreover, GIS has been coupled to a variety of models (e.g., crop productivity, hydrology, and water quality simulations) and may be an important component of spatial decision support systems and land resource information systems. GIS enhances model flexibility and efficiency in these systems where it is often regarded as a centralized data analysis, management, and planning system. As a result, decision-makers and land use planners in private companies, in universities, and at various levels of the government are using GIS to develop spatial environmental databases, perform land evaluations, and analyze and manage resources. There is no doubt that the use of GIS and GIS-based systems in land use planning activities will continue to increase in the future, as more detailed digital environmental data sets become available and the capability of computers to handle large volumes of data increases.

INTRODUCTION

Agricultural scientists are required to provide information needed to address land degradation and land use conflicts confronting the world. As the population increases and land becomes a commodity in many parts of the world, careful planning of the use of land must be undertaken to accommodate conflicting people's needs and preserve and/or protect the environment. The decision about the use of land must be made based on analyses of each potential use in terms of its economic and biophysical suitability to the specific tract of land and possible impact to environmental degradation.

GEOGRAPHIC INFORMATION SYSTEM (GIS) USE IN LAND USE PLANNING ACTIVITIES

The goal of land use planning is to make decisions about the use of land and resources.^[1] Its implementation is often driven by future people's needs in terms of productivity and environmental sustainability. Land use planning is important in highly populated communities primarily due to conflicts between competing uses and interests of users.^[2] In this case, planning activities are tailored to make the optimal use of the limited land resources. In general, the land use planning involves sequentially an organization of

thoughts and establishment of long-term goals, a land evaluation that includes appraising alternatives and finally designing and implementing the plan. Land evaluation is the most important aspect of this process.

The information within a GIS consists of a spatial component represented by points (e.g., well locations), lines (e.g., streams and road networks), or polygons (e.g., soil delineations) and attribute data or information that describes the characteristics of spatial features. The spatial entity is referenced to a geographic coordinate system and is stored in either a vector or a raster format.^[3] GIS is primarily used in the development of spatial databases and land evaluation, a procedure in the land use planning process that is aimed at determining the suitability of land units to existing and alternative uses and the potential impact of each on the environment.

Development of Spatial Databases

GIS stores, retrieves, and allows the efficient manipulation of database information. It provides powerful analysis and relational database facilities to modify and/or integrate spatial data from different sources and resolutions as well as advanced visualization functions to display output data in the form of interpretative maps. In this case, land attributes and qualities are derived from geographic databases and used to determine land suitability, limitations, or ratings for

various land use types. The analysis results may be presented in tabular or graphical form and are intended to provide key information necessary for land users or planners to make meaningful decisions about land management and conservation and/or land use planning.

Site Suitability Analyses

A site suitability analysis typically involves the assessment of the level of affinity a specific land has for a particular use. Soil information available in soil databases is rarely enough for site evaluations. In addition to soils, the analysis often integrates local information on landforms, land uses, the relative location of the land, and associated social and political restrictions. The proposed use may have additional limitations that should be taken into account; e.g., an analysis for suitable sites for the land application of sewage sludge should consider the physical, chemical, and biological properties of the waste in soil and water.^[4,5]

GIS is efficient in identifying and ranking sites for various land use planning activities. First, site-specific analyses often require many and detailed data sources. Second, GIS overlay features, logical operations, and display functions are tailored to speed up data processing and therefore allow efficiently suitability class assignment and graphical display of results. A good example is the use of GIS for locating appropriate sites for the forestland application of sewage waste.^[6] The authors derived physical site suitability ratings for an area in Vermont based on U.S. Environmental Protection Agency guidelines^[7] and merged them with social and political restrictions of the state and counties to derive a land applicability classification. Similar GIS-based approaches have been used to locate sites for solid waste disposals.^[8]

Linking GIS and Models

The spatial databases and associated attribute data described above may be part of a GIS/model graphical user interface (GUI). These models may be those that determine land suitabilities or those that simulate water quality for environmental impact assessments. As an entity of GIS/model GUI, GIS allows easy access to database attributes by various algorithms, statistical software, and environmental models for a variety of land use analyses. In addition to model parameterization, a full integration of GIS and models also allows the user to interact with various modules, select data input and module utilities, and display the graphical and/or tabular representations of modeling results.^[9] The use of this type of interface reduces significantly the processing time and resources required to develop input data and run the model. GIS/model linkage has been accomplished for field- and watershed-scale hydrology/water quality models such as the Leaching Estimation and Chemistry Model^[10] and Agricultural Non-Point Source Pollution^[11] model. Results of these analyses are used to support management and land use planning activities in the farm or watershed.

The GIS/model GUI may also be part of a spatial decision support system (SDSS). In addition to modeling parameterization, SDSSs offer the users with functions to evaluate different land use scenarios necessary for making management recommendations and/or land use planning decisions. Results may be used by farmers to adjust the management practices of distinct fields in a given farm or may be used by field officers to help farmers set priorities while providing technical assistance for nutrient management planning. Fig. 1 shows a flow diagram indicating data

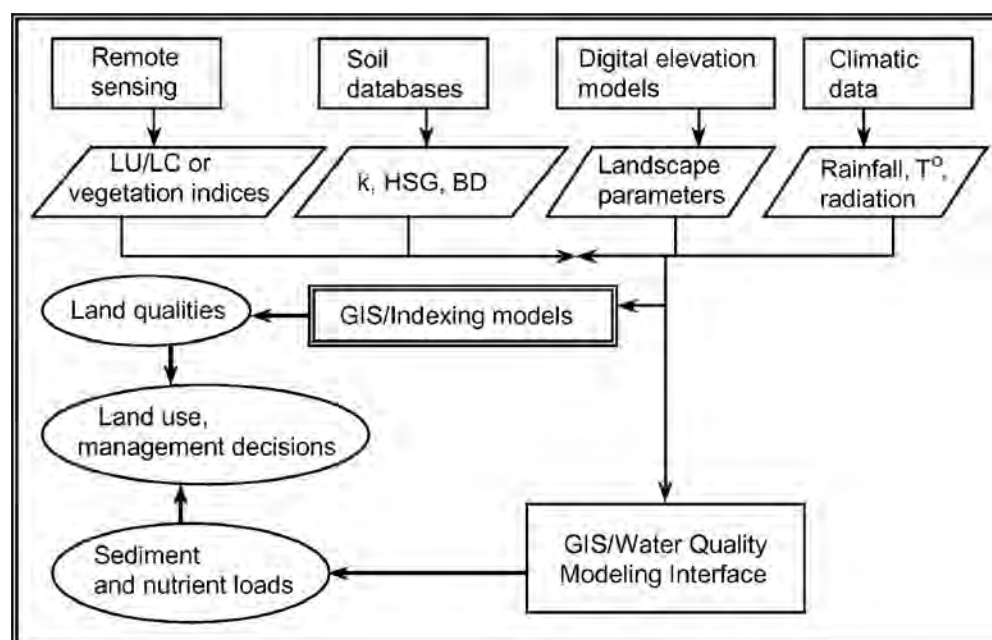


Fig. 1 The integration of GIS, spatial database, and models.

integration in a GIS with models for water quality assessment in watersheds.

GIS has been incorporated in many land evaluation systems, so that planners and public officials can take advantage of its spatial modeling and visualization capabilities. An example of such a system is the U.S. Department of Agriculture–Natural Resources Conservation Service's Land Evaluation and Site Assessment (LESA).^[12] The land evaluation portion of LESA determines soil productivity levels, farm size, and agricultural sales volume; the site assessment portion deals with factors such as location, amount of non-agricultural land, and zoning restrictions.^[13]

GIS, in GIS-based LESA systems, is used in both modules and at all levels of the analysis.

Development of Land Resource Information Systems (LRISs)

A number of land information delivery systems have been developed to make spatial data available to users for application in various aspects of land use planning. These systems, also called LRIS, are multipurpose systems that integrate geographic databases and GIS tools to analyze, record, report, and display spatial data relationships. Some of these systems have been embedded into the World Wide Web for quick and easy access and analysis via Internet browsers.

RELIABILITY OF GIS-BASED ANALYSIS RESULTS

GIS and GIS-based systems used in spatial data analyses for land use planning have grown lately. GIS is particularly attractive in these areas because it allows overlay of spatial data sets and the merging and analysis of attribute data from different sources. The resulting data are obtained using data from digital and paper maps of different scales or acquired at different resolutions such as in the case of digital elevation models and remote sensing imagery. The combination of such data layers may produce unrealistic data and consequently lead to erroneous predictions. The question is, how reliable are these results when used for developing land use and management plans? A discussion on the propagation of error and uncertainty in GIS-based systems is provided elsewhere.^[14]

The accuracy of GIS-based land evaluations is a function of the quality of attribute data and mapping delineations of databases, and the type of model or assessment scheme used in the analyses. Various algorithms for assessing the quality of GIS analysis results as affected by error and uncertainty in GIS layers and their propagation are part of popular GIS and image analysis software (e.g., Arc/Info and ERDAS Imagine). The results, which are often in the form of reliability diagrams, are not used by average GIS

practitioners. This is probably due to the fact that they are not easy to understand or interpret. Similarly, models vary depending on how each represents various processes of the system. Lumped parameter models and indexing/ranking schemes are mostly used in land evaluations because they are easy to parameterize. However, these models ignore spatial variations of parameters throughout the field, watershed, or region of study. Furthermore, models originally designed for fields and watersheds are often applied to regional analyses, thus adding some level of uncertainty in modeling predictions; e.g., most land use planning programs use conventional methods of land evaluation. Each land parameter is given a range of values with corresponding ratings showing its suitability to crop production. These indexes are added or multiplied to create a single index that is to rank land units. The method is simple but carries a high uncertainty because breaks between two ranges or ranks are subjective.

The effect of map scale and resolution on environmental assessment and modeling output data has been subject to many studies. Raster-based GIS systems require that a grid cell size be defined prior to the analysis. However, as pixel size increases above the resolution of the original data, the spatial variability decreases. This causes a decrease in the predictive power of generated input parameters particularly for small land areas.

CONCLUSION

GIS has been used primarily in land evaluation, a procedure in the land use planning process that deals with determining land suitability to existing and alternative uses and the potential impact of each use on the environment. GIS and GIS-based systems are often considered to as integrated spatial information systems for data analysis needed in land use management and planning activities. Despite the advantages of GIS outlined above, the land use planner should be aware of the error and uncertainty in GIS and GIS-based analyses, resulting from digitizing and scaling inaccuracies, data conversion between vector and raster formats, and others. Finally, GIS implementation in land use planning activities can be expensive. In addition to costs associated with hardware and software purchase and maintenance, a high level of technical expertise is required to perform complex modeling tasks in land evaluations for alternative uses and to sustain databases. Nonetheless, the demand for GIS and GIS-based analysis systems in land use planning is to increase in the future as more detailed spatial data sets become available.

REFERENCES

1. FAO. *Guidelines for Land Use Planning; FAO Development series 1*; Soil Resources, Management and Conservation Service, FAO: Rome, 1993; 96 pp.

2. Brinkman, R. Recent developments in land use planning, with special reference to FAO. In *The Future of the Land: Mobilizing and Integrating Knowledge for Land Use Options*; Fresco, L.O., Stroosnijder, L., Bouma, J., van Keulen, H., Eds.; Wiley: Chichester, 1994; 13–21.
3. Burrough, P.A. Matching databases and quantitative models in land resource assessment. *Soil Use Manage.* **1989**, *5*, 3–8.
4. Nizeyimana, E.; Petersen, G.W.; Looijen, J.C. Land use planning and environmental impact assessment using GIS. In *Environmental Modeling with GIS and Remote Sensing*; Skidmore, A., Prins, H., Eds.; Taylor and Francis Ltd.: London, 2002; 227–251.
5. Petersen, G.W.; Nizeyimana, E.; Miller, D.A.; Evans, B.M. The use of soil databases in resource assessments and land use planning. In *Handbook of Soil Science*; Summes, M.E., Ed.; CRC Press: Boca Raton, 2000; H69–H94.
6. Hendrix, W.G.; Buckley, J.A. Use of a geographic information system for selection of a sites for land application of sewage waste. *J. Soil Water Conserv.* **1992**, *92*, 271–275.
7. U.S. Environmental Protection Agency. *Process Design Manual for Land Treatment of Municipal Waste*; Environmental Protection Agency: Cincinnati, 1981.
8. Weber, R.S.; Jenkins, J.; Leszkiewicz, J.J. Application of geographic information system technology to landfill site selection. In *Proceedings of the Application of Geographic Information Systems, and Knowledge-Based Systems for Land Use Management*; VPI and State University: Blacksburg, 1990.
9. Petersen, G.W.; Nizeyimana, E.; Evans, B.M. Application of GIS to land degradation assessments. In *Methods for Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentine, C., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1997; 377–391.
10. Inskeep, P.P.; Wraith, J.M.; Wilson, J.P.; Snyder, R.D.; Macur, R.E.; Gaber, H.M. Input parameter and model resolution effects on predictions of solute transport. *J. Environ. Qual.* **1996**, *25*, 453–462.
11. Tim, U.S.; Jolly, R. Evaluating agricultural nonpoint source pollution using integrated geographic information systems and hydrology/water quality models. *J. Environ. Qual.* **1994**, *23*, 25–35.
12. Wright, L.E.; Zitzmann, W.; Young, K.; Googins, R. LESA—Agricultural land evaluation and site assessment. *J. Soil Water Conserv.* **1983**, *38*, 82–86.
13. Daniels, T. Using LESA in a purchase of development rights program. *J. Soil Water Conserv.* **1990**, *45*, 617–621.
14. Heuvelink, G.B.M. *Error Propagation in Environmental Modeling with GIS*; Taylor & Francis: London, 1998; 127 pp.

Landfill Sites: Revegetation

Gilbert Yuk-sing Chan

Department of Applied Biology and Chemical Technology, Hong Kong Polytechnic University, Hong Kong, China

Ming H. Wong

Institute for Natural Resources and Land Management, Hong Kong Baptist University, Hong Kong, China

Abstract

The disposal of waste to land has been the prime means of waste disposal since the evolution of humans. In the past, people disposed of refuse at dumping points within a reasonable distance, e.g., outside the city wall. As cities expanded, dumping points expanded and wastes were piled up until they reached an unacceptable level and were eventually covered by soil. These rudimentary open dumping methods are common in small and remote villages. Landfill was developed from open dumping and is a method of codisposal of municipal solid waste and soil, on land with proper engineering consideration. Alternative names are sanitary landfill or controlled tipping; however, it has become more common simply to call it “landfill.” It can be regarded as the only way for the ultimate disposal of municipal solid waste in a controlled manner that causes minimal nuisance to public health or safety. Hazardous or radioactive wastes are generally not accepted in ordinary landfill sites.

LANDFILL DESIGN AND OPERATION

The daily operation of a landfill site is to spread and compact the waste unloaded from vehicles by using waste-moving equipment. A better practice is to cover the waste with another layer of waste or with a temporary cover soil of about 0.2 m thick and compact with compactors to maximize the landfill capacity. The layers of waste and soil formed are called landfill cells. The final top cover is a layer of soil about 0.2 to 1 m in depth. To prevent differential settlement, a high degree of compaction over the whole area of the cell is maintained. An *in situ* density of 1.2 t m^{-3} can be achieved using a bulldozer.^[1] Where densities of only $0.6\text{--}0.7 \text{ t m}^{-3}$ are achieved, settlement of 20% or greater is expected; while on sites where a high density of $1.0\text{--}1.2 \text{ t m}^{-3}$ is achieved, the settlement may be 10% or less.^[2] A site is considered completed after its filling capacity is reached. Completed sites have to be redeveloped, and special techniques have to be applied for vegetation establishment.

There are two contrasting approaches to landfill design: 1) to consider landfill as a bioreactor and try to maximize waste degradation; and 2) to attempt to isolate the embedded waste from the outside environment as far as technology can achieve and costs allow. The former principle relies upon attenuation of the leachate both within the waste and in the adjacent geology by biological and physicochemical processes. However, the contamination of adjacent environments and groundwater has been experienced.^[3]

The latter is also called containment landfill, where the rate of release of leachate into the environment is extremely low and landfill gas is either flared on-site or collected for energy recovery.^[4] In modern landfill design, the latter concept is commonly adopted to protect nearby environments and minimize pollutant discharge.^[5] To achieve this, an impermeable bottom liner is laid at the base of the site to avoid seepage of leachate into groundwater or adjacent water bodies. The very top of the landfill cells is covered by a thicker layer of inert soil, usually about 1–1.5 m thick, which is compacted by machinery.^[1] In order to further restrict gas migration, a costly synthetic impermeable layer may be laid together with the soil, but the efficiency of the composite liner (also called landfill cap) depends on the material used.^[6] Between the bottom liner and the landfill cap is a network of pipes to collect leachate and gas. The top liner not only prevents gas migration into the topsoil, but also prevents the infiltration of rainwater into the landfill.

Other criteria to be considered in the design, planning, and development of a landfill include engineering aspects, mode of landscaping, environmental impact, operational issues, and legal questions.^[1,2,7]

ANAEROBIC DEGRADATION OF MUNICIPAL SOLID WASTE

Anaerobic biodegradation of waste embedded in landfill cells generates landfill gas. The typical composition of pure

landfill gas is about 60% carbon dioxide (CO_2) and 30% methane (CH_4 ; v v^{-1}); however, concentrations as high as 89.3% CO_2 and 77.1% CH_4 have been detected.^[1] In addition to the major gases present in landfill cover soil, other organics, such as alkanes, aromatic compounds, cycloalkanes, terpenes, alcohols, ketones, and phosphine, and over 140 volatile organic compounds that are generally at levels of $<0.1 \text{ mg m}^{-3}$ have been identified in landfill gas.^[8,9] The production of gases depends on climatic factors and the chemical nature of the waste.

The aqueous secondary waste percolating through embedded municipal solid waste and its degradation by-products is called landfill leachate. In most cases, rainwater contributes the greater portion of water that forms the leachate. Groundwater and other water sources from the adjacent environment may also supply moisture to the waste and produce leachate. Biodegradation of waste also produces water. The properties of leachate depend mainly on the nature of the embedded waste and the design of the site, which determines the infiltration rate of rainwater. High ammonium–nitrogen ($\text{NH}_4^+\text{-N}$) is a general characteristic of landfill leachate, as $\text{NH}_4^+\text{-N}$ oxidation generally requires aerobic conditions. It is not uncommon for leachate to contain in excess of $1000 \text{ mg NH}_4^+\text{-N l}^{-1}$.^[10] The ratio of biochemical oxygen (O_2) demand to chemical O_2 demand of leachate collected from a new site is generally higher than that of leachate from an old site.^[11]

LANDFILL FACTORS AFFECTING PLANT GROWTH

Although the design and operation of landfills in different countries are not the same, the basic problems of landfill gas and leachate contamination are common to all.^[12]

Landfill Gas

Flower, Gilman, and Leone^[13] presented one of the early reports on how landfill gas affects tree growth and how its injurious effects may be prevented. Gas problems were experienced in sites where the clay layer was not formed properly or was cracked by uneven settlement of topsoil. Similarly, for landfill sites without an impermeable top liner, or for areas in larger sites where only temporary cover was laid, high CH_4 and CO_2 and low O_2 were detected in the cover soil and tree growth was affected.^[14] The tolerance of 10 subtropical woody plants to landfill gas was compared,^[15] and the results indicated that legumes were generally more resistant to landfill gas. For sensitive plants, such as *Liquidambar formosana* and *Castanopsis fissa* (both nonlegumes), chlorosis and stunted growth were observed after they had been fumigated with simulated landfill gas. Adverse effects of landfill gas on plants were caused by high levels of CO_2 ($>20\%$) in the rhizosphere,

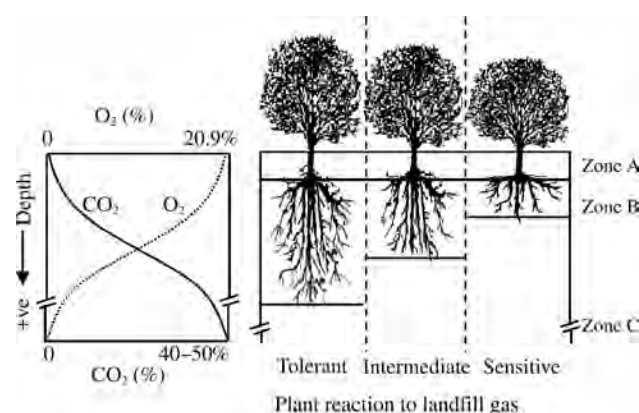


Fig. 1 Stratification of landfill gas in cover soil. Light penetration and occasional drying make zone A unsuitable for root growth. In zone B, root growth is stimulated by low levels of CO_2 (e.g., $<10\%$). Zone C is restricted for root growth by high levels of CO_2 (e.g., $>10\%$). Plants with roots sensitive to CO_2 (e.g., $>10\%$) have a shallow root system in zone B. Plants with moderate sensitivity to CO_2 have an intermediate depth of zone B for root growth. Plants tolerant to high CO_2 level have a deep root system.

rather than by low partial pressure of O_2 ($<10\%$). However, low levels of CO_2 ($<10\%$) fumigation in the rhizosphere stimulated root growth, as low levels of gaseous CO_2 or bicarbonate stimulate cellular respiration in root cells. Therefore, the stratification of CO_2 in landfill cover soil stimulates shallow root system development (Fig. 1). A landfill gas fumigation test indicated that differences in sensitivity of plants to gas level also caused differential stimulatory (*Lotus corniculatus* and *Trifolium pratense*) and inhibitory effects (*Vicia villosa* and *Trifolium repens*) on root growth subjected to landfill gas (CO_2).^[16] However, the mechanisms by which CO_2 , especially at high concentrations (e.g., $>10\%$), differentially affects tissue respiration are not fully understood.

When the O_2 partial pressure is low, leguminous nitrogen gas (N_2) fixation declines. Under a subambient partial pressure of O_2 , N_2 fixation by a nodule is proportional to the O_2 concentration.^[17] The influence of CO_2 after landfill gas contamination on the growth and activity of N fixing nodules is complicated. CO_2 at high levels as may exist in landfill cover soil was shown to be effective in inhibiting rhizobia-legume N_2 fixation, and the effect was reversible within 1 hour.^[18] However, it has been observed that rhizobia are effective in nodulating woody legumes (*Acacia confusa* and *Leucaena leucocephala*) under landfill gas fumigation conditions and are effective in N fixation.^[19]

CH_4 , a major ingredient of landfill gas, is physiologically inert to plants. However, in the presence of CH_4 , O_2 in the rhizosphere is depleted. CH_4 in cover soil can be converted to CO_2 by methanogenic bacteria.^[20] This process decreases the level of CH_4 and increases the level of CO_2 in cover soil.

Landfill Leachate

The problem of leachate seepage normally happens in confined areas and usually at the edge of a site. "Leachate breakout" is caused by high leachate levels within the waste and ingress into a site through weaknesses in the cap or on an uncapped site.^[21] The breakout can cause injury to trees due to leachate toxicity. In West Virginia, the United State, six tree species were irrigated with leachate for 4 years resulting in significant mortality.^[22] In Hong Kong, *A. confusa* irrigated with leachate exhibited growth depression to about 25% of the control after 50 days.^[23] The phytotoxicity of treated and untreated leachate samples collected from three landfills in Germany was evaluated using luminescent bacteria and bioassays conducted using the aquatic plant *Lemna minor* and terrestrial plants *Brassica rapa* and *Lepidium sativum*.^[24] In addition to high levels of $\text{NH}_4^+\text{-N}$, the accumulation of chloride and increases in soil salinity are major causes of phytotoxicity of leachate.^[25] On the contrary, when leachate concentration and irrigation rate were properly controlled, leachate irrigation could benefit plant growth.^[26,27]

Leachate recirculation on landfill sites can be regarded as an alternative approach to leachate treatment.^[28] It was found that leachate recycling increased the rate of biodegradation of organics and the landfill would become "stabilized."^[29] When leachate recycling was practiced, the duration of CH_4 production could be shortened from 20–30 years to a few years.^[30] Application of leachate to the surface of a landfill can result in a significant decrease in leachate volume due to evapotranspiration. However, leachate recycling is not commonly adopted in cities demanding groundwater for their main water supply and for landfill without proper bottom liners. For example, leachate recycling has been banned in New Jersey^[30] and is also not allowed by law in the United Kingdom.

Other Landfill Factors

In addition to gas and leachate, a number of unique characteristics found in landfill final cover soil might affect plant growth. For landfill sites with a high-quality composite cap, leachate contamination of cover soil is unlikely. The physical and chemical properties of the cover soil being laid on top of the impermeable cap are therefore crucial for the performance of the vegetation established there. Nutrient deficiency, especially N, is a common characteristic of low-quality cover soil, such as demolition waste or unweathered soil.^[31] Application of fertilizer is therefore necessary.

Waterlogging is common in landfill sites where the evaporation rates are low or the rainfall is intense.^[21] On the contrary, the problem of drought was reported in subtropical landfill sites.^[32] Drought problems may be severe in landfills located in places with a high evaporation rate. The impermeable top liner imposes a constraint

on plant growth because of the relatively shallow effective root zone for water storage.

Another common feature is the high bulk density in cover soil caused by the compaction of soil and waste. Excessive compaction of soil might hinder the penetration of common plant roots. The restriction of root growth depends not only on bulk density but also on soil particle size. Plant roots will rarely penetrate into light textured soils with a bulk density greater than $1.7\text{--}1.8\text{ g cm}^{-3}$ or heavy textured soils with a bulk density greater than $1.5\text{--}1.6\text{ g cm}^{-3}$. It was demonstrated that the growth of Austrian pine roots on a silt loam and sandy loam was restricted at bulk density of $1.4\text{--}1.8\text{ g cm}^{-3}$.^[33] A study conducted in Hamburg, Germany, indicated that the reduction of compaction during construction of landfill cover soil or the use of amelioration increased the water retention capacity in the topsoil layer.^[34]

Elevated temperature in cover soil, e.g., 45°C ,^[15] caused by the heat release process of the biodegradation of waste and solar radiation on bare lands, is another common feature of landfill sites. The adverse effects of elevated temperature in landfill cover soil on plant growth have been summarized.^[21]

Landfill cover soil may have elevated metal levels after contamination by leachate that has high metal levels. However, landfill cover soil does not normally accumulate metals to such high levels that it has harmful effects on vegetation. On old landfill sites, other landfill factors such as landfill gas are more significant than metals in their affect vegetation performance.^[35]

There is a lack of information related to the role of mycorrhizal fungi on tree growth on closed landfills. The need for mycorrhizal tree seedlings in site restoration programs has been questioned.^[36,37]

Suitable Tree Species for Landscaping Completed Landfill

There are two basic criteria that determine the suitability and success of a plant to be established on a landfill site: tolerance to landfill gas and/or leachate and drought tolerance. The superior performance of *A. confusa*, *Acacia mangium*, and *Acacia auriculiformis* is mainly due to their high drought tolerance^[38] and capability to fix N when growing on N-deficient soil.^[15,39] Due to their high drought tolerance, *Tristania conferta*, *Eucalyptus citriodora*, and *Eucalyptus torelliana* are some of the non-legumes that also show superior performance in landfill.^[31] Tree species suitable for woodland establishment in subtropical completed landfill are listed in the studies by, Chan, Wong, and Whitton,^[19] Chan and Wong,^[31] and Chan, Wong, and Whitton,^[40] in which about one half of the trees are legumes. Suitable and unsuitable trees for planting in the United Kingdom are listed in the study by Dobson and Moffat,^[21] of which five of the 30 "likely to be tolerant" trees are legumes.

ECOLOGICAL RESTORATION AND END LAND USE

The number and size of landfill developments are increasing in most cities. As a direct consequence, more completed landfill will be sites awaiting redevelopment in the near future. Due to uneven settlement of the top profile and the potential accumulation of explosive landfill gas in infrastructure, building estates or large civil constructions is not recommended on completed landfill sites. Therefore, closed landfill sites are commonly converted into parks, golf courses, or botanical gardens.

A sustainable landfill should be properly managed so that the environmental risk is acceptable.^[41] However, information about sustainable development of closed landfill is limited. To protect the environment, an assessment of the impact of landfill on ecosystems is required throughout the operation and postoperation stages of a site, both within the site boundaries and in the vicinity of the site.^[2]

The interaction between landfill factors and organisms inhabiting landfill sites is complex. In the rhizosphere, vegetated landfill caps may enhance CH₄ oxidation, but the effects seem not to be significant or consistent.^[42] Introduction of earthworms to cover soil on landfills in the United Kingdom was successful, but the physical soil conditions of the cover soil have not changed significantly after 3 years.^[43] *In situ* flaring of landfill gas may produce oxide deposits containing rather high concentrations of elements that form volatile species such as phosphorus, arsenic, antimony, and tin.^[44] However, adverse effects of such deposits on ecosystems established within landfill sites have not been reported.

On the contrary, all landfill sites are not necessarily unsuitable for ecosystem development. An ecological study was conducted on temporary cover soil in two landfill sites in Hong Kong, on which the cover soils were slightly contaminated by landfill gas and leachate. Higher plant coverage and diversity and higher densities of soil and litter animals were detected on these two sites, compared with normal sites. This was due to the seepage into landfill cover soil of leachate elevated in nutrient levels, especially N, which benefited plant growth. Indirectly, the presence of plant coverage and leaf litter benefited soil and litter fauna in the sites.^[45] The absence of plant cover at a completed landfill site limited the occurrence of some common soil animals such as Diplopoda, Hemiptera, Isopoda, and Isoptera.^[46]

A 4-year revegetation study was conducted on one of the world's largest landfills in Hong Kong, constructed with composite impermeable top liners, and no landfill gas and leachate problems were detected. The performance of 24 tree species was tested and the results indicated that *A. confusa*, *A. mangium*, *A. auriculiformis* (legumes), *E. citriodora*, and *E. torelliana* (nonlegumes) served as excellent pioneer species, while other species, such as *Castanopsis fissa*, *Cinnamomum camphora*, and *Machilus thunbergii* (native species to Hong Kong) had extremely

high mortalities on the test site. After 3 years, the pioneer species provided shelter for native species. Animals, including insects and small vertebrates, started to inhabit the newly established woodland, with a species richness comparable to nearby natural landscapes.^[31] Tree planting has not been recommended in closed landfill sites in the United Kingdom as trees were suspected to damage the landfill top liner. However, evidence indicates that tree roots will not penetrate the top liner if it has a high bulk density of about 1.8 g cm⁻³.^[14,47] The roles of trees in ecologically sound and sustainable reclamation of old landfill sites have been reviewed.^[35]

Wildlife colonization may also take place in woodlands established on landfill sites. The succession and maturation processes and the sustainability of the woodland rely on the physical and chemical properties of the man-made cover soil and also on climatic factors. Sustainable development of an ecosystem with vegetation comparable to ecosystems in the adjacent natural or less disturbed lands has proved to be feasible on closed landfills. However, special technologies, management skill, and understanding of landfill design and operation are generally required.

REFERENCES

1. Department of the Environment (U.K.). *Landfilling Wastes*; Waste Management Paper No. 26, 6th Impression; HMSO: London, 1994; 205 pp.
2. Street, A.; Dumble, J.P. Above-ground landfill: The engineered approach. *J. Inst. Water Env. Man.* **1990**, *4*, 430–435.
3. Kuajara, O.; Sanchez, J.C.D.; Ballestrin, R.A.; Teixeira, E.C. Environmental monitoring of the North Porto Alegre landfill Brazil. *Water Environ. Res.* **1997**, *69*, 1170–1177.
4. Wuebben, P.; George, S.; Watkins, L.; Bonny, A. South coast air-quality management district (SCAQMD): The permitting and commercialization of landfill gas-based methanol production facilities. *Appl. Biochem. Biotech.* **1996**, *57–58*, 729–740.
5. Gronow, J. The role of landfills in the United Kingdom. *Waste Manage. Res.* **1999**, *17*, 409–412.
6. Bachmann, J.; Von Felde, D. Comparison of different surface sealing for municipal waste landfills with respect to gas emission and recultivation. *Zietschrift Fuer Kulturtechnik und Landentwicklung* **1998**, *39*, 220–227.
7. Burton, S.A.Q.; Watson, C.I.A. Ammonia and nitrogen fluxes in landfill sites. Application to sustainable landfilling. *Waste Manage. Res.* **1998**, *16*, 41–53.
8. Allen, M.R.; Braithwaite, A.; Hills, C.C. Trace organic compounds in landfill gas at seven U.K. waste disposal sites. *Environ. Sci. Technol.* **1997**, *31*, 1054–1061.
9. Jenkins, R.O.; Morris, T.A.; Craig, P.J.; Ritchie, A.W.; Ostah, N. Phosphine generation by mixed- and monoseptic-culture of anaerobic bacteria. *Sci. Total Environ.* **2000**, *250*, 73–81.
10. Robinson, H.; Gronow, J. Ground water protection in the U.K.: Assessment of landfill leachate source-term. *J. Inst. Water Env. Man.* **1992**, *6*, 229–236.
11. Obbard, J.P.; Barr, M.J.; Robinson-Howard, D.; Carville, M.S. Landfill leachate: Characteristics and biological

- treatment in Hong Kong. *Resour. Env. Biotechnol.* **1999**, *2*, 235–248.
12. Lisk, D.J. Environmental effects of landfills. *Sci. Total Environ.* **1991**, *100*, 415–468.
 13. Flower, F.B.; Gilman, E.F.; Leone, I.A. Landfill gas, what it does to trees and how its injurious effects may be prevented? *J. Arboricult.* **1981**, *7*, 43–52.
 14. Dobson, M.C.; Moffat, A.J. A re-evaluation of objections to tree planting on containment landfills. *Waste Manage. Res.* **1995**, *13*, 579–600.
 15. Chan, Y.S.G.; Wong, M.H.; Whitton, B.A. Effects of landfill gas on subtropical woody plants. *Environ. Manage.* **1991**, *15*, 411–431.
 16. Marchiol, L.; Cesco, S.; Pinton, R.; Zerbi, G. Germination and initial root growth of four legumes as affected by landfill biogas atmosphere. *Restor. Ecol.* **2000**, *8*, 93–98.
 17. Heckmann, M.O.; Drevon, J.J. Inhibition of symbiotic nitrogen fixation by nitrate. In *Physiological Limitations and the Genetic Improvement of Symbiotic Nitrogen Fixation*; O'Gara, F., Manian, S., Drevon, J.J., Eds.; Kluwer Academic Publishers: The Netherlands, 1988; 97–106.
 18. Zhang, J.; Liang, J.; Wong, M.H. The effect of high CO₂ and low O₂ concentrations in simulated landfill gas on the growth and nodule activity of *Leucaena leucocephala*. *Plant Cell Physiol.* **1995**, *36*, 1431–1438.
 19. Chan, Y.S.G.; Wong, M.H.; Whitton, B.A. Effects of landfill gas on growth and nitrogen fixation of two leguminous trees (*Acacia confusa*, *Leucaena leucocephala*). *Water Air Soil Pollut.* **1998**, *107*, 409–421.
 20. Fielding, E.R.; Archer, D.B.; Macario, E.C.; MaCario, A.J.L. Isolation and characterization of methanogenic bacteria from landfills. *Appl. Environ. Microbiol.* **1988**, *54*, 835–836.
 21. Dobson, M.C.; Moffat, A.J. *The Potential for Woodland Establishment on Landfill Sites*; Report for the Department of the Environment; HMSO: London, 1993; 88 pp.
 22. Menser, H.A.; Winant, W.M.; Bennett, O.L. Spray irrigation with landfill leachate. *Biocycle* **1983**, *24*, 22–25.
 23. Wong, M.H.; Leung, C.K. Landfill leachate as irrigation water for tree and vegetable crops. *Waste Manage. Res.* **1989**, *7*, 311–324.
 24. Devare, M.; Bahadir, M. Biological monitoring of landfill leachate using plants and luminescent bacteria. *Chemosphere* **1994**, *28*, 261–271.
 25. Stephens, W.; Tyrrel, S.F.; Tiberghien, J.E. Irrigation short rotation coppice with landfill leachate: constraints to productivity due to chloride. *Bioresource Technol.* **2000**, *75*, 227–229.
 26. Ettala, M.O. Short rotation tree plantations at sanitary landfills. *Waste Manage. Res.* **1988**, *6*, 291–302.
 27. Cureton, P.M.; Groenvelt, P.H.; McBride, R.A. Landfill leachate recirculation: Effects on vegetation vigour and clay surface cover infiltration. *J. Environ. Qual.* **1991**, *20*, 17–24.
 28. Gordon, A.M.; McBride, R.A.; Fiskien, A.J. Effect of landfill leachate irrigation on red maple (*Acer Rubrum* L.) and sugar maple (*Acer Saccharum* Marsh.) seedling growth and on foliar nutrient concentrations. *Environ. Pollut.* **1989**, *56*, 327–336.
 29. O'-Keefe, D.M.; Chynoweth, D.P. Influence of phase separation, leachate recycle and aeration on treatment of municipal solid waste in simulated landfill cells. *Bioresource Technol.* **2000**, *72*, 55–66.
 30. Lee, G.F.; Jones, R.A.; Ray, C. Sanitary landfill leachate recycle. *Biocycle* **1986**, *37*, 36–38.
 31. Chan, G.Y.S.; Wong, M.H. *Third Year Plant Performance Monitoring Report, South East New Territories (SENT) Landfill*, Hong Kong, PRC, Submitted to Green Valley Landfill Ltd; 2001; 54 pp.
 32. Wong, M.H.; Yu, C.T. Monitoring of gin drinkers' bay landfill, Hong Kong: II. Gas contents, soil properties, and vegetation performance on the side slope. *Environ. Manage.* **1988**, *13*, 753–762.
 33. Zisa, R.P.; Halverson, H.G.; Stout, B.J. *Establishment and Conifers on Compact Soils in Urban Areas*; Research Paper NE; USDA Forest Service: Broomall, PA, 1980; 45 pp.
 34. Tresselt, K.; Groengroeft, A.; Leonhardt, T.; Miehllich, G. Soil physical properties of topsoil layers in landfill cover systems and their influence on the growth of vegetation: Case study Hamburg–Francop. *Zeitschrift Fuer Kulturtechnik und Landentwicklung* **1998**, *39*, 228–233.
 35. Dickinson, N.M. Strategies for sustainable woodland on contaminated soils. *Chemosphere* **2000**, *41*, 259–263.
 36. Tosh, J.E.; Senior, E.; Smith, J.E.; Watson-Craik, I.A. Ectomycorrhizal seedling response to selected components of landfill leachate. *Mycol. Res.* **1993**, *97*, 129–135.
 37. Tosh, J.E.; Senior, E.; Smith, J.E.; Watson-Craik, I.A. The role of ectomycorrhizal inoculations in landfill site restoration programmes. *Letter Appl. Microbiol.* **1993**, *16*, 187–191.
 38. Webb, R. *Tree Planting and Maintenance in Hong Kong*; Standing Interdepartmental Landscape Technical Group; Hong Kong Government Printer: Hong Kong, 1991; 53 pp.
 39. Chan, Y.S.G.; Wong, M.H.; Whitton, B.A. Effects of landfill factors on tree cover—A field survey at 13 landfill sites in Hong Kong. *Land Contam. Reclam* **1996**, *4*, 115–128.
 40. Chan, Y.S.G.; Wong, M.H.; Whitton, B.A. Effects of landfill leachate on growth and nitrogen fixation of two leguminous trees (*Acacia Confusa*, *Leucaena Leucocephala*). *Water Air Soil Pollut.* **1999**, *111*, 29–40.
 41. Westlake, K. Sustainable landfill—possibility of pipe-dream? *Waste Manage. Res.* **1997**, *15*, 453–461.
 42. Hilger-Helen, A.; Wollum-Arthur, G.; Barlaz-Morton, A. Landfill methane oxidation response to vegetation, fertilization, and liming. *J. Environ. Qual.* **2000**, *29*, 324–334.
 43. Butt, K.R.; Frederickson, J.; Lowe, C.N. Colonisation, survival and spread of earthworms on a partially restored landfill site. *Pedobiologia* **1999**, *43*, 684–690.
 44. Glindemann, D.; Morgenstern, P.; Wennrich, R.; Stottmeister, U.; Bergmann, A. Toxic oxide deposits from the combustion of landfill gas and biogas. *Environ. Sci. Pollut. R. Int.* **1996**, *3*, 75–77.
 45. Chan, Y.S.G.; Chu, L.M.; Wong, M.H. Influence of landfill factors on plants and soil fauna—An ecological perspective. *Environ. Pollut.* **1997**, *97*, 39–44.
 46. Wong, M.H.; Cheung, K.C.; Lan, C.Y. Factors related to the diversity and distribution of soil fauna on Gin drinkers' bay landfill, Hong Kong. *Waste Manage. Res.* **1992**, *10*, 423–434.
 47. Handel, S.N.; Robinson, G.R.; Parsons, W.F.J.; Mattei, J.H. Restoration of woody plants to capped landfills: Root dynamics in an engineered soil. *Restor. Ecol.* **1997**, *5*, 178–186.

Landscape

George Hall

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Landscape is the portion of land that can be seen from a single vantage point. The shape of the land has an effect on soil conditions. This entry reviews the categorization of the parts of the landscape and their impact on land use and management.

INTRODUCTION

The landscape is defined in several different ways. It has been defined as the portion of the land surface that the eye can comprehend in a single view.^[1] The glossary of soil science terms^[2] states that the landscape is a collection of related landforms; usually the land surface which the eye can comprehend in a single view. Landforms are any physical, recognizable form or feature on the earth's surface.^[2] Thus, the scope of the landscape is variable depending on the position of the observer, whether in a deep valley or on an airplane flying over the valley. The landforms are the features that make up that landscape.

LANDSCAPE TERMINOLOGY

The terminology used by soil scientists is based on the historic concepts of William Morris Davis and Walther Penck. Wood^[3] and King^[4] formulated models for the fully developed hillslope which included the elements of waxing slope, free face, debris slope, and pediment (Fig. 1). Ruhe^[5,6] developed a soil-geomorphic model in which the soil properties were integrated with the hillslope models. Ruhe^[6,7] modified the Wood and King models by proposing the hillslope elements of summit, shoulder, backslope, footslope, and toeslope (Fig. 1). Following the introduction of the hillslope positions, Ruhe and coworkers^[7,8] formalized terms for geomorphic slope components: headslope for the concave position, sideslope for the linear position, and noseslope for the convex position (Fig. 2).

A different approach was taken by Conacher and Dalrymple.^[9] They used the catena concept and defined the slopes as a 3-D unit extending from summit to the valley floor. The model consisted of nine units, all of which need not occur on a given landscape. Of particular importance in the model were the interactions among soil materials and mobilization, translocation, and redeposition of the material by water and gravity.

LANDSCAPE UNITS

The specific landscape position elements have differing processes and as a result have differing properties. Although these five positions are separately defined, as is the case with many natural phenomenon, the positions and thus the processes grade from one to the other. The following discussion is most applicable for humid climates.

Summit

Where the summit is greater than approximately 30 m in width, much of the water is retained on the surface. This position is considered the most stable, and the water movement is predominantly vertical except near the transition to the shoulder position or on small undulations on the summit.

Shoulder

Convexity is the rule for the shoulder positions. Surface runoff of water is maximized in this element resulting in a highly erosional and relatively unstable land surface. Lateral movement of surface material (soil creep) may become an important process on this part of the landscape. Lateral subsurface water movement or throughflow is often an important process in this landscape position.

Backslope

General linearity is characteristic of the backslope position. The dominant process on this position is the transport of material as well as water. Surface transport of material may be in the form of surface wash, creep, flow, or slump. Lateral subsurface water movement (throughflow) is important in this element. The position is considered to be relatively unstable depending on the degree of slope.

Landscape

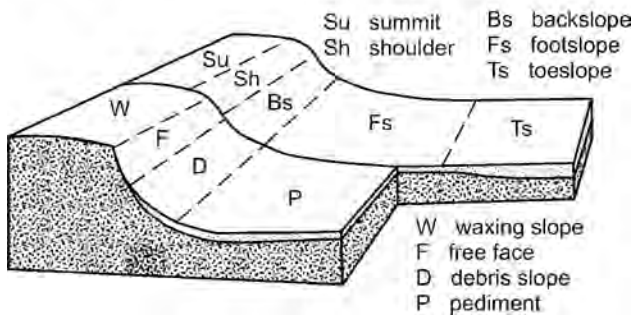


Fig. 1 Elements of a hillslope model.

Source: From Ruhe.^[6]

Footslope

Concavity is characteristic of this landscape position. This concavity results in deposition from upslope of particulate material as well as material carried in solution. Seepage zones may be present, and water retention is high. Soils in footslope positions may be heterogeneous due to mass movement, irregular seepage, and non-uniform deposition. Paleosols often can be identified in these positions.

Toeslope

The toeslope element is unstable as a result of its dominantly constructional nature. Materials in this position are

derived from up valley and to some extent from superjacent footslope and backslope positions.

CATENA

The catena concept was proposed by Milne^[10,11] as an interlocking of elements of the landscape. The term is derived from the latin word for chain. In United States, catena is used almost interchangeably with toposequence. Toposequence or local relief carries with it a morphological connotation that is related to a relative elevation and thus to changes in hydrology. Catena on the other hand carries with it a process-response connotation. The soils of a catena differ not only in morphology but also are considered to differ as a result of erosion, transport, and deposition of surficial material as well as leaching, translocation, and deposition of chemical and particulate constituents in the soil. The concept of catena has been used extensively to study the soil genesis.^[12-14]

The morphology and processes of each member of the catena are related to every other member of the catena. It is a dynamic process where in the members of the catena are continuously adjusting to the environmental changes of a given landscape.^[15]

LANDSCAPE MORPHOLOGY

Landforms are individually transformed by erosion and deposition during the process of landscape evolution. Water

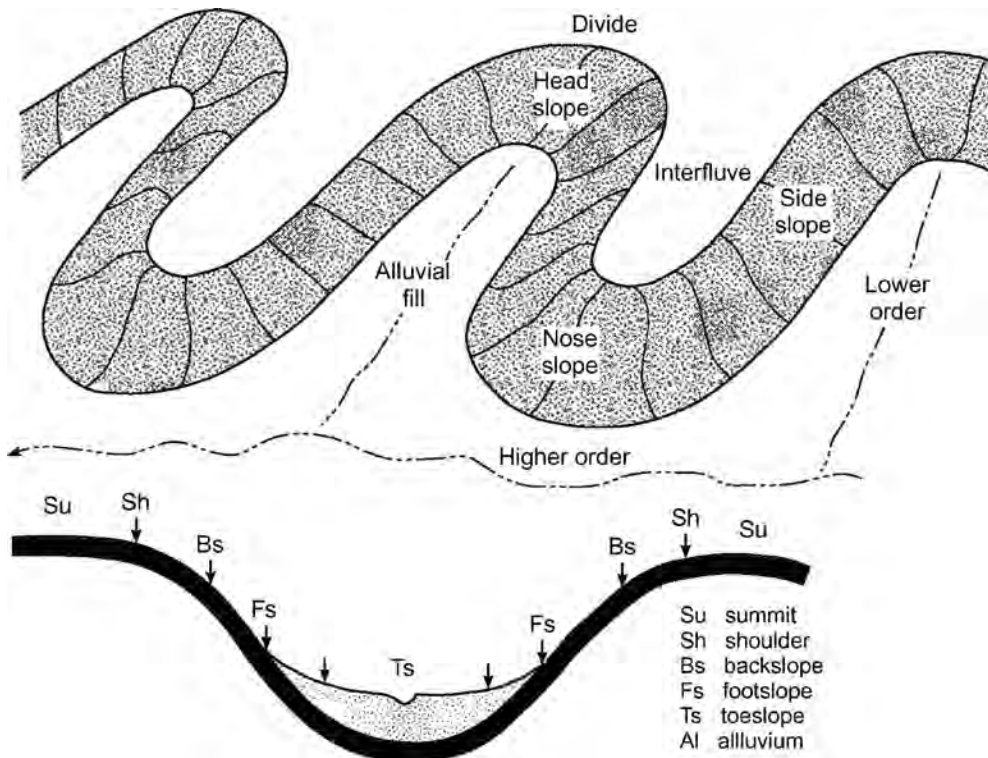


Fig. 2 Geomorphic slope components and hillslope profile from an incised valley.

Source: From Ruhe.^[8]

movement is a major driving force that shapes and modifies the morphology of the landscape. To properly evaluate the processes occurring on the landscape, the morphology of the landscape must be determined. This evaluation should include at a minimum: gradient, aspect, and vertical and horizontal curvature.

Water movement on the slopes is largely determined by the planar curvature of the slopes; both the longitudinal and latitudinal profile. Huggett^[13] added surface flow lines to the basic slope shapes (Fig. 3).

Pennock et al.^[16] studied the complexity of water movement on various hillslope elements and curvatures. They used the terms convergent and divergent to identify the slope positions in which water accumulates or disperses. One combination of convergent and divergent slope positions is shown in Fig. 4. In this example, the landscape element with the least moisture would be the divergent shoulder and the element with the most moisture would be the convergent footslope. Other elements would be expected to have intermediate moisture contents.

Landscape models have shown that landscapes are predictable; they have a large non-random variability component. The landscape must be defined in three dimensions, considering the surface configuration as well as the vertical changes in stratigraphic materials.

SLOPE ORIENTATION/ASPECT

Landscape modifying processes are influenced by the slope orientation or aspect. The orientation affects the amount of

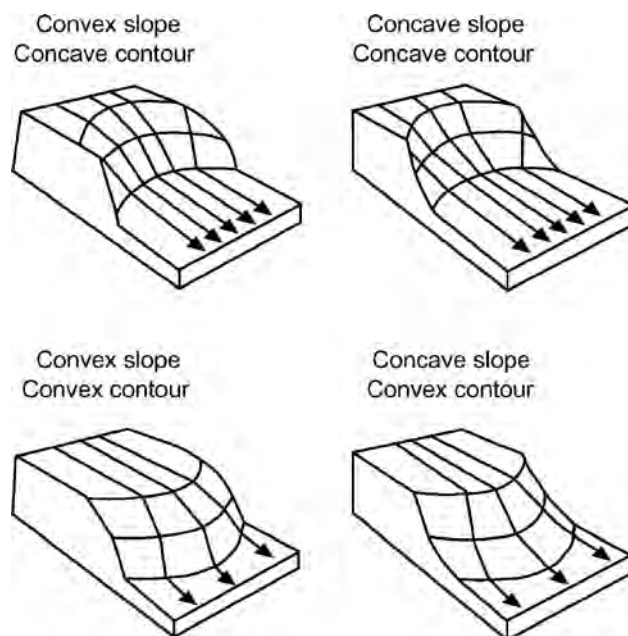


Fig. 3 Four basic slope shapes with flow lines.

Source: From Huggett.^[13]

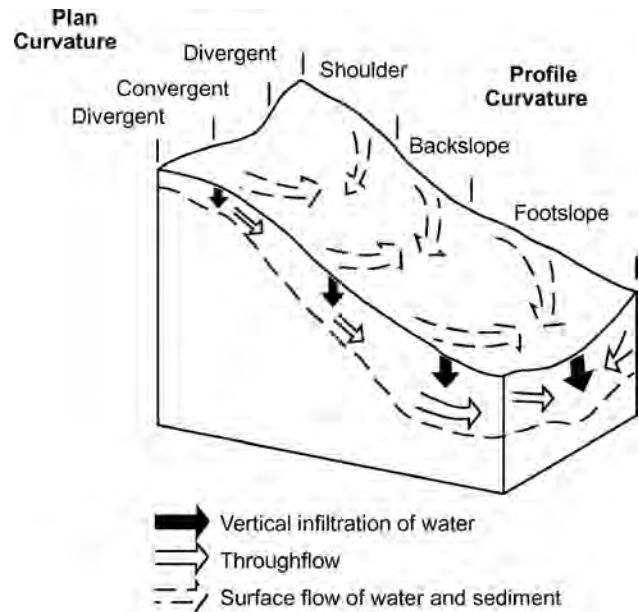


Fig. 4 Water movement in a hillslope system.

Source: From Pennock, Zebarth, et al.^[16]

radiation reaching the soils surface which in turn influences the microclimate, and thus, the moisture and vegetation on slopes. In the areas of low relief, aspect has little influence while in the areas of high relief the influence becomes more important.

Global latitude is also a determinant because the angle of solar radiation increases with increased latitude. Thus, shading increases with increased latitude. Aspect becomes most important in the high relief areas between 40° and 60° latitude.^[17] In the northern hemisphere, south-facing slopes receive more radiation and are thus warmer and dryer than north-facing slopes.^[14] The differences in moisture and temperature are reflected in the vegetative and soil process differences for different slope orientations.

IMPLICATIONS FOR LAND USE AND MANAGEMENT

Landscape and Land Use

Most land use decisions are influenced by the landscape characteristics. In agriculture, cropping patterns and cultivation practices are conditioned by the gradient and configuration of the landscape. Conservation practices must be invoked as slope gradients increase. These practices include contour strip cropping and conservation tillage to control erosion. Increasing slope gradients also limit the use of agricultural equipment and increases the cost of production. In steeper portions of the landscape, the land must be utilized for grazing or forestry. Footslope and toeslope positions are more subject to seepage and flooding and thus

may restrict the cropping options on these portions of the landscape.

In urban land use planning, consideration must also be given to landscape. As in the case of agriculture, footslope and toeslope positions are subject to water seepage and flooding, and urban development of these areas must take these limitations into consideration when land use decisions are made. Although housing developments on steep gradients are aesthetically pleasing, they are more expensive to build and maintain and may be subject to slippage and slope failure.

Risks of Soil Erosion

Water, wind, and gravity are each considered agents of soil erosion. Erosion resulting from water and gravity is most commonly related to landscape positions. Wind erosion is more closely related to vegetative soil cover but may be modified by landscape conditions. Soil loss increases as the length of the slope increases up to some maximum. Soil loss increases exponentially as the slope gradient increases. Erosion as the result of gravity (mass wasting) is most common on very steep gradient slopes but may also be found on lesser slopes where the surface and subsurface conditions such as a seep may cause slumps and slips to occur.

On-Farm and *In situ* Water Management

Water management is important in the control of erosion and to provide optimum moisture for plant growth. Diversions and waterways are often used to control water movement on the landscape. On steeper gradient slopes, contour and strip cropping may be used to slow water movement over the soil and thus encourage greater moisture infiltration and retention. Tile drainage may be used to intercept drainage from seeps in the footslope positions. In level or depressional areas where soil water accumulates at or near the surface, tile drainage must be installed to remove the excess water.

REFERENCES

1. Ruhe, R.V. *Quaternary Landscapes in Iowa*; Iowa State Univ. Press: Ames, 1969; 255 pp.
2. Roth, C.B. *Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 1997; 134 pp.
3. Wood, A. The development of hillside slopes. *Geol. Assoc. Proc.* **1942**, 53, 128–138.
4. King, L.C. Canons of landscape evolution. *Geol. Soc. Am. Bull.* **1953**, 64, 721–752.
5. Ruhe, R.V. Geomorphic surfaces and the nature of soils. *Soil Sci.* **1956**, 82, 441–455.
6. Ruhe, R.V. Elements of the soil landscape. *Trans. Int. Congr. Soil Sci.* 7th. **1960**, 4, 165–170.
7. Ruhe, R.V. *Geomorphology*; Houghton Mifflin: Boston, 1975; 246 pp.
8. Ruhe, R.V.; Walker, P.H. Hillslope model and soil formation: I. Open systems. *Trans. Int. Congr. Soil Sci.* 9th. **1968**, 4, 551–560.
9. Conacher, A.J.; Dalrymple, J.B. The nine unit landscape model: An approach to pedogeomorphic research. *Geoderma* **1977**, 18, 1–154.
10. Milne, G. *A Provisional Soil Map of East Africa*; Amani Memoirs No. 28 East African Agric. Res. Stn.: Tanganyika Territory, 1936; 34 pp.
11. Milne, G. Normal erosion as a factor in soil profile development. *Nature* **1936**, 138, 548.
12. Yaalon, D.H. Soil-forming processes in time and space. In *Paleopedology: Origin, Nature and Dating of Paleosols*; Yaalon, D.H., Ed.; Israel University Press: Jerusalem, 1971; 29–39.
13. Huggett, R.J. Soil landscape systems: A model of soil genesis. *Geoderma* **1975**, 13, 1–22.
14. Birkeland, P.W. *Soils and Geomorphology*; Oxford University Press: Oxford, 1999; 430 pp.
15. Dan, J.; Yaalon, D.H. The application of the catena concept in studies of pedogenesis in mediterranean and desert fringe regions. *Trans. 8th Int. Congr. Soil Sci.* **1964**, 5, 751–758.
16. Pennock, D.J.; Zebarth, F.J.; deJong, E. Landform classification and soil distribution in hummocky terrain, saskatchewan, canada. *Geoderma* **1987**, 40, 297–315.
17. Hunckler, R.V.; Schaetzl, R.J. Spodosol development as affected by geomorphic aspect, Baraga county, Michigan. *Soil Sci. Soc. Am. J.* **1997**, 61, 1105–1115.

Landscape: Classification

Philip Schoeneberger

*U.S. Department of Agriculture—Natural Resources Conservation Service
(USDA-NRCS), Lincoln, Nebraska, U.S.A.*

Abstract

The earth's surface is not uniform, but it can be understood and consistently described. It is a universal human activity to evaluate and categorize the earth's surface into groupings useful for management decisions. The most robust and versatile classification schemes include attributes that capture the surficial form, internal composition, spatial arrangement and distribution, with attention to how these parameters affect vegetation and land management options. Geomorphology provides a framework that integrates these parameters to describe the inherent architecture upon which natural systems and processes operate.

INTRODUCTION

Groups of spatially associated landforms that share common forms, patterns, composition, or site history are called landscapes. Landscapes are a general and convenient way to summarize the context or setting of a land area. Identifying various landscapes and placing them into an orderly array constitute landscape classification. Project purpose and scale (minimum polygon size) determine the most appropriate approach to landscape classification.

Landscapes

The word *landscape* is widely used in at least three different ways, all of which are legitimate and can be readily found in soil-related literature:

1. *Informal context:* Landscape—a portion of land that the eye can comprehend in a single view.^[1] This “everyday” context refers to whatever portion of the earth's surface that surrounds you or whatever you can see outside your window (Fig. 1). It is utterly dependent upon where you happen to be standing. There is a huge difference between “what the eye can comprehend” (see) from a mountain top vs. the view from a coastal plain marsh. This perspective depends exclusively upon your field of view. It encompasses everything, and yet only, what you happen to see (doesn't address things out-of-view).
2. *Explicit Geomorphic context—e.g., mountains (any mountains) vs. plains (any plains):* “Landscape—a broad or unique land area comprised of an assemblage or collection of natural landforms that define a general geomorphic form or setting (e.g., mountain range, lake plain, river valley, etc.). Landforms within a landscape share spatially associated formational processes but can vary in details and age.”^[2] This context recognizes landform groups that are spatially associated in a unique, recognizable pattern (clustered, oriented, and arranged) indicating a common link (Fig. 2). The link(s) may be similar topography (elevation and relief), composition (bedrock or sediment types), structure (tectonic, crustal deformations), surficial modification (erosional or depositional alterations), etc. and are tangibly different from adjoining landform groups. If landscape subtypes (e.g., different types of mountains) are recognized, the emphasis is on specific geologic or geomorphic attributes (e.g., fault-block mountains vs. volcanic mountains) rather than differences in climate, vegetation, or geography (e.g., the Organ Mountains vs. the Cascade Mountains). A list of common landscapes is shown in Table 1.
3. *Physiographic context:* Landscape—a collection or group of individual landforms that are spatially associated in a unique, recognizable pattern (clustered, oriented, and arranged) indicating common links. The links are not restricted to geomorphic and geologic parameters but may also include geography (where on a continent), native plant communities (desert vs. savannah), dominant cropping or management practices, or some combination of these. If subtypes of a physiographically based landscape are recognized (e.g., different types of mountains), the emphasis is on differences in geography, climate, vegetation or geography, dominant land use practices,^[4] or plant communities, not just geologic or geomorphic attributes. For example, interest has grown in ecosystem or “ecoregions” basis^[5,6] that emphasizes potential native vegetation and/or real-time plant communities or management systems.



Fig. 1 An idealized Landscape painted by Asher Durand, circa 1850.

The ecoregions approach poses both advantages and disadvantages for soils information. Advantages include more overt recognition of climatic nuances and biogeochemical impact of specific plant communities on land management and soil morphology. This approach partitions landscapes into “present-day” or “real-time” environments rather than historical norms. A disadvantage can be an overemphasis on dynamic, potentially ephemeral plant communities that can change dramatically and rapidly, both in geologic and human time frames. Plant communities can change quickly and dramatically with fire, insect or disease infestations, human-caused land use changes, climate shifts, etc. Such vegetation changes can



Fig. 2. 2 A scenic “landscape” (vernacular context) overlook at Waubonsie State Park, Iowa, U.S.A. The view also presents spatially associated groups of individual Landforms that together form several discrete Landscapes (soil geomorphic/technical context): 1) a river valley landscape (a Fluvial Geomorphic setting) composed of natural levees, backswamps, overflow stream channels, etc. during the 2011 floods of the Missouri River (background); 2) a loess hills landscape (an Eolian Geomorphic setting) composed of individual loess hills, loess bluffs, hillslopes, etc. (foreground).

Table 1 A partial list of landscapes commonly used in soil inventory.^[2,3]

Landscapes (broad or unique groups or clusters of natural, spatially associated features)	
Alluvial plain	Kegel karst
Alluvial plain remnant	Lagoon
Badlands	Lake plain
Bajada	Lava field
Barrier island	Lava plain
Basin	Lava plateau
Batholith	Lowland
Bay (coast)	Marine terrace
Bolson	Meander belt
Breached anticline	Mountain range
Breaklands	Mountains
Breaks	Mountain system
Caldera	Ocean
Canyonlands	Outwash plain
Coastal plain	Peninsula
Cockpit karst	Piedmont
Cone karst	Piedmont slope
Continental glacier	Plains
Delta plain	Plateau
Dissected breaklands	Rift valley
Dissected plateau	River valley
Drumlin field	Sandhills
Dune field	Sand plain
Estuary	Scabland
Everglades	Sea
Fan piedmont	Semibolson
Fault-block mountains	Shield volcano
Fluviokarst	Shore complex
Fluviomarine terrace	Sinkhole karst
Fold-thrust mountains	Sound
Foothills	Strait
Glaciokarst	Tableland
Gulf	Thermokarst
Hills	Till plain
Ice-margin complex	Tower karst
Intermontane basin	Upland
Island	Valley
Karst	Volcanic field

occur much faster than subsequent soil changes resulting in a lag-offset between prevailing ecosystem and a corresponding pedologic record of it. Examples include: 1) Climate changes: A rise in atmospheric temperature (e.g., hypsithermal interglacial, or other global warming)

can cause the tree line on mountain slopes to climb, such that formerly cold and barren areas become forested. Similarly, some continental proglacial environments (e.g., periglacial) appear to have changed rapidly following the most recent continental glacier “retreat.” 2) Human-caused land use changes, such as conversion of native woodlands to pasture or croplands and back again (e.g., the initial reduction in native woodlands in New England and the Piedmont following European settlement in the 18th and 19th centuries, with a substantial reversal of that trend in the late 20th century). Another example is the radical expansion of sage and mesquite plant communities associated with livestock grazing in the Southwestern United States since the late 19th century. 3) Introduced plant species: Subtle but no less pervasive ecosystem changes in plant community composition can result from human-introduced plant species (e.g., crabgrass, kudzu, bamboo, and lespedeza in the United States). Landscape classifications based on dynamic plant communities or management conditions are susceptible to misrepresentation over time.

Of these confounding contexts, it is the second, geologic/geomorphic context of landscapes that has consistently proven to have the greatest utility for understanding, describing, predicting, and successfully managing soils.^[3] This success stems from an emphasis on the basic architecture and constituents (the form, internal structure, and composition) of the regolith from which the soil is derived and their effect on water dynamics. Consequently, geologic/geomorphic-based landscape classifications are inherently more durable and reflect dominant conditions (longer term equilibrium conditions). Soils do not immediately re-equilibrate to reflect ecosystem shifts. Soils do adjust to longer term (10s to 100s of years) environmental conditions and eventually reflect new ecosystem equilibriums in their morphology and properties.

Landscape classification

The identification and subsequent placement of different types of landscapes into some order or array.

Grouping individual but related landforms into clusters (i.e., landscapes) provides a meaningful, utilitarian, and common sense way to subdivide the natural continuum of the earth’s surface. Historically, landscapes have been defined by transportation, management, or resource circumstances. Each identified landscape establishes a context, a suite of realities that establish land management opportunities and limitations. Areas that present similar restrictions to travel (e.g., mountains) or are subject to similar hazards (e.g., floodplain) give rise to practical groups. In geology and soil science, emphasis has been placed on morphometric and compositional attributes that have useful applications, such as limitations for road or housing construction, opportunities for food or fiber production, or building material extraction. To these ends,

Table 2 Geomorphic process environments and other useful groupings of landscapes.

Geomorphic environments and other groupings: Land-features [landscapes, landforms, and microfeatures grouped by geomorphic process (e.g., fluvial) or common settings (e.g., water bodies)].^[3]

Environments Other	
1	Coastal marine and estuarine (wave, tidal, or shallow marine related)
2	Lacustrine (related to non-fluvial inland water bodies)
3	Fluvial [related to concentrated channel flow (e.g., stream)]
4	Solution (dominated by dissolution and subsurface drainage)
5	Eolian (wind dominated)
6	Glacial (directly related to glaciers)
7	Periglacial (related to non-glacial, cold climate)
8	Mass wasting (mass movement; gravity-dominated)
9	Volcanic and hydrothermal
10	Tectonic and structural (bedrock structures, crustal movement)
11	Subaqueous features (submerged features generally capable of supporting rooted plants)
12	Wetlands (related to vegetated or shallow wet areas, wet soils)
13	Water bodies (permanent water features)
14	Erosional (dominated by non-channel, non-perennial water erosion)
15	Depressional (low area or declivity terms, excludes permanent water bodies)
16	Slope (generic slope forms, geometry, or arrangement rather than process)

traditional geomorphic process environments provide coherent and functional groupings of landscapes (Table 2).

From a soils perspective, geology strongly influences soils by determining: 1) the materials which are present and 2) their internal arrangement and composition. Concurrently, geomorphology strongly affects soils through the processes and dynamics that move and modify the materials present (e.g., removal, transportation, deposition, and/or geochemical alterations). Geomorphic processes deserve special attention in soil science as they modify residuum (in-place materials) and dictate the transportation and redistribution of unconsolidated materials (e.g., alluvium and colluvium). Active geomorphic processes determine present conditions and strongly impact management choices, and thereby provide the practical link between “what is here?” and “how can we successfully manage it?” for specific purposes. Most important to soil geography (soil patterns), geomorphology is an effective basis from which to explain and predict sediment body distribution (where do specific materials or soils start and stop, and where are you likely to find more of the same?).

Much has been written about various geomorphic processes (e.g., glacial and fluvial) from a strictly geologic point of view.^[7,8] Indeed, these processes strongly influence the macroscale structure of landscapes and the natural distribution of the raw materials in which soils form. There is a related, pedologic perspective of soil geomorphology. It encompasses the relatively thin veneer of regolith (unconsolidated materials) and the processes that generate, redistribute, and alter the regolith at or near the earth's surface.^[9–11] Nuances that are of little or no interest to a geologic point of view may be crucial in a pedologic context and have substantial consequences at the human-operative scale. For example, bedrock geology maps commonly omit relatively thin surficial layers (e.g., <1–3 m) of regolith and rarely recognize “thin” but pedologically important eolian inputs (dust and Aerosols; e.g., carbonates) and outputs (deflation removal of fine particles; e.g., the Dust Bowl). Soil geomorphology deals directly with these near-surface layers and includes the hydrologic behavior that profoundly influences ecosystem behavior and soil patterns (e.g., the red-edge effect^[10,12]) in the same zone.

There has been a shift in the United States away from an emphasis on bedrock inventory and toward a greater emphasis on surficial sediments (regolith). This shift has focused attention on finer-scale geologic/geomorphic processes and sediments and leads directly to soils and soil geomorphology. For example, the relatively thin (50–150 cm) volcanic tephra mantle covering much of Idaho and western Montana is rarely captured on geologic maps, yet has much greater influence on timber growth and ecosystem function than does the much thicker, underlying colluvium/residuum of the tectonically derived mountains. Landscape classifications that directly recognize pedo-geomorphic processes and materials need to be developed further and used.

Scale—the hidden landscape qualifier

Project or application scale predetermines the relative size of features or areas that can be recognized (e.g., minimum polygon size) and therefore what constitutes a landscape (i.e., what features and geomorphic processes are recognized). For a given area of land, a continental scale assessment will reflect a different and obviously more generic suite of landscapes than an assessment focused on smaller, more localized areas. Historically, small-scale-dominated (coarse resolution) landscape assessment.^[13–16] In soil geography, some programs also deal with very large areas (e.g., continental) as in Food and Agriculture Organization's soil maps of the world. Other soil inventory or geography programs are much more localized and detailed. In the United States, soil geography scales range from very coarse (continental) to very detailed (Order 1 site investigations; e.g., 1 in. = 330 m). Within that range, generally soil surveys in the United States (e.g., the

National Cooperative Soil Survey) are more locally scaled (e.g., Order 2 county-level surveys, to regionally scaled (e.g., Order 3 reconnaissance surveys^[17]).

The value of (and to) soils in understanding landscapes

Soil is a synthograph. Soil retains a record of both dramatic and subtle geomorphic processes and environmental site history in its properties and morphology.^[10,12,17,18] Soils can thereby be used to decipher the prevailing past conditions, the history of geomorphic processes (how did an area become what it is?), as well as the present environment and soil behavior. This historical imprint serves as a feedback loop whereby geomorphology can shed light on soil geography and properties, and conversely, soils can and are used to confirm or refine geomorphic interpretations of landscapes.

CONCLUSION

Landscapes are an efficient way to subdivide the earth's surface into areas that share common management or resource opportunities and limitations. Organizing landscapes into an array, such as geomorphic process environments, helps to communicate natural relationships useful for land use management and in simply understanding the world around us. Greater scientific effort is needed to denote landscapes by pedogeomorphic relationships that are not adequately addressed by traditional geologic or geomorphic landscapes.

REFERENCES

1. Bailey, R.G.; Avers, P.E.; King, T.; McNab, W.H. *Webster's Seventh New Collegiate Dictionary*; G.&C. Merriam Co.: New York, 1972.
2. Soil Survey Staff. Glossary of landforms and geologic materials. In *National Soil Survey Handbook—Part 629*; USDA, Natural Resources Conservation Service, National Soil Survey Center: Lincoln, 2001.
3. Schoeneberger, P.J.; Wysocki, D.A. *A Geomorphic Description System (Version 3.1)*; National Soil Survey Center, Natural Resources Conservation Service, USDA: Lincoln, 2002.
4. Soil Conservation Service. Land resource regions and major land resource areas of the United States. In *Agricultural Handbook # 296*; U.S. Dept. Agriculture: Washington, DC, 1981.
5. Bailey, R.G.; Avers, P.E.; King, T.; McNab, W.H. *Ecoregions and Subregions of the United States (map)*; US Geological Survey: Washington, DC, 1994. Scale: 1:7,500,000; colored. Accompanied by a supplementary table of map unit descriptions compiled and edited by McNab, W.H. and Bailey, R.G. Prepared for the U.S. Dept. Agriculture-Forest Service.

6. Omernick, J.M. Ecoregions of the United States. *Ann. Assoc. Am. Geogr.* **1987**, 77 (1), 118–125.
7. Bloom, A.L. *Geomorphology: A Systematic Analysis of Late Cenozoic Landforms*, 3rd Ed.; Prentice Hall: Englewood Cliffs, 1997.
8. Ritter, D.F.; Kochel, R.C.; Miller, J.R. *Process Geomorphology*, 3rd Ed.; Wm C. Brown Publ.: Dubuque, 1995.
9. Daniels, R.B.; Hammer, D. *Soil Geomorphology*; John Wiley & Sons: New York, 1992.
10. Wysocki, D.A.; Schoeneberger, P.J.; LaGarry, H.E. Geomorphology of soil landscapes. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press LLC: Boca Raton, 1999.
11. Fenneman, N.M. *Physical Divisions of the United States*; U.S. Geological Survey, U.S. Gov. Print. Office: Washington, DC, 1946. (reprinted 1957); (map) 1 sheet; 1:7,000,000.
12. Daniels, R.B.; Buol, S.W.; Kleiss, H.J.; Ditzler, C.A. *Soil Systems in North Carolina*. Technical Bulletin 314; Soil Science Department, North Carolina State University, Raleigh, 1999.
13. Thornbury, W.D. *Regional Geomorphology of the United States*; John Wiley & Sons, Inc.: New York, 1965.
14. Hunt, C.B. *Physiography of the United States*; W.H. Freeman & Co: London, 1967.
15. Wahrhaftig, C. *Physiographic Divisions of Alaska*. US Geological Survey Professional Paper 482; US Government Printing Office: Washington, DC, 1965.
16. Ruhe, R.V. *Geomorphology: Geomorphic Processes and Surficial Geology*; Houghton-Mifflin: Boston, 1975.
17. Soil Survey Staff. *Soil Survey Manual*. USDA, Soil Conservation Service, *Agricultural Handbook No. 18*. U.S. Government Printing Office: Washington, DC, 1993; 503 pp.
18. Schoeneberger, P.J.; Wysocki, D.A.; Benham, E.C. Soil Survey Staff. *Field Book for Describing and Sampling Soils, Version 3.0*. USDA-NRCS. US Government Printing Office: Washington, DC, 2012.

Landscape: Development and Erosion

Jon Harbor

Department of Earth and Atmospheric Sciences, Purdue University,
West Lafayette, Indiana, U.S.A.

Abstract

Landscapes develop as a result of the interaction between rates and patterns of uplift and denudation. Landscape development occurs over timescales that are typically extremely large from the perspective of soil scientists, and the landscape itself is a spatial scale far greater than those tackled in most soil studies. Scientists interested in modern soil erosion processes and controls usually consider the landscape and landscape features as fixed boundary conditions. Sediment yield data demonstrate the importance of linkages between temporal scales and spatial patterns in soil erosion.

INTRODUCTION

Landscapes develop as a result of the interaction between rates and patterns of uplift and denudation. Therefore, styles of landscape development depend in part on rates and patterns of soil formation and soil erosion.^[1] Although the similarity of some aspects of landscape is seen across spatial scales (Fig. 1) and suggests that fundamental balances of erosional and depositional processes are scale independent,^[2] scale issues are extremely important in understanding how soil erosion and empirical data on soil erosion relate to landscape development.

TIMESCALES

Landscape development occurs over timescales that are typically extremely large ($\approx 10^5$ years) from the perspective of soil scientists, and the landscape itself is a spatial scale far greater than those tackled in most soil studies. Thus, scientists interested in modern soil erosion processes and controls usually consider the landscape and landscape features as fixed boundary conditions (slope angles, lengths, and shapes do not typically change over months or years). Jenny's^[3] often-quoted list of factors of soil formation (including climate, organic matter, relief, and parent material) is in fact a list of independent variables controlling spatial patterns of soil erosion over short timescales (Table 1).

However, over thousands to millions of years, rates and patterns of soil formation and erosion are important in controlling the long-term development of landscape form. Over these long timescales if soil accumulation at a site (soil production plus deposition) exceeds soil loss by erosion, then the denudational balance^[4] results in increasing soil thickness, decreasing subsoil weathering,

and a situation where the erosion rate is limited by the effectiveness of the processes transporting soil on the slope. In such transport-limited situations, long-term slope evolution is toward convexo-concave forms that become progressively moderate over time.^[5] In contrast, if the soil accumulation rate is less than the potential soil erosion rate, then weathered material is removed as soon as it is produced and the rate and pattern of weathering limit the amount and pattern of soil erosion. Under such weathering-limited situations, slopes have prominent straight sections and develop over time by parallel retreat in which slope angles are maintained in dynamic equilibrium.^[5]

Thus, over long timescales, soil erosion rates and patterns are not only affected by landscape form but also by a determinant of landscape form.

More generally, as discussed in detail by Schumm and Lichty,^[5] as the timescale of interest changes, what we consider to be relevant, dependent, and independent variables in soil erosion also change (Table 1). Over short timescales, it is irrelevant what the landscape was like when it was first exposed by uplift, or how much time has elapsed since then, and morphometric variables such as relief and slope shape are independent variables in a study of soil erosion. However, on the timescale of landscape development, variables such as relief and valley morphology change over time as a function of the pattern of geomorphic processes, including soil erosion, and so must be treated as dependent variables. Thus, feedback relationships develop between changes in slope form and soil erosion patterns: Soil erosion patterns depend on slope form, but on long timescales soil erosion patterns change slope form, and thus slope form and soil erosion patterns evolve interdependently.

Timescales are also important in soil erosion measurement programs and in the ways in which we use



Fig. 1 Scale and soil erosion features. Topography is approximately fractal^[2] so, without scale indicators, this landscape photo could have been taken from an airplane of a landscape element on the order of 10^2 km^2 , or it could be a close up of a microscale feature on the order of a 10^0 m^2 .

Source: Photo courtesy of Christian Renschler, University at Buffalo. Soil-surface approximately 1 m^2 , Badlands National Park, U.S.A.

observations of soil erosion processes to make inferences about landscape development. Over short timescales, our understanding of spatial and temporal patterns of soil erosion is based on monitoring erosion plots and on determining sediment yields from watersheds across a range of spatial scales.^[7] Over geologic timescales, our understanding of spatial and temporal patterns of soil erosion is based on depositional records from lake and ocean sediments and also on cosmogenic nuclide techniques.^[8] The cosmogenic nuclide approach is based on the observation that reactions between cosmic rays and minerals within about 2–3 m of the soil or

rock surface produce a range of cosmogenic nuclides within the mineral structure, i.e., the radionuclides such as ^{26}Al , ^{10}Be , and ^{36}Cl . These nuclides accumulate at known rates, and the production rate decreases exponentially with depth below the soil or rock surface. The concentrations of radionuclides in eroded sediments can thus be used to back calculate the amount of time it took for a sample to go from approximately 3-m deep into the surface and thus is a measure of the surface-lowering (soil erosion) rate. The applications of this technique and comparison with other sources of data allow evaluation of the scale of variations in soil erosion rates at different timescales. For example, sediment yields for a hyperarid drainage basin in southern Israel based on a 33-year sediment budget exceed long-term sediment generation rates based on ^{10}Be and ^{26}Al by 53–86%.^[9] In comparison, over a timescale of several decades, different averaging periods in data from 189 gauging stations in the Eastern United States and Canada produced sediment yield variations of 10–50%.^[10] At the shortest timescales, variations in erosion rates increase dramatically; because soil erosion is an episodic process, measurements over periods of days, hours, or even shorter times fluctuate wildly depending on whether or not an erosional event occurs during the measurement period. Thus, soil erosion rate variability decreases as a function of increasing timescale.

SPATIAL SCALES

Sediment yield data also demonstrate the importance of linkages between temporal scales and spatial patterns in soil erosion. In the data from gauging stations in the Eastern United States and Canada,^[10] in addition to temporal variation in sediment yield as a function of the timescale used for averaging, the spatial pattern of sediment yield varied dramatically between averaging periods. If only one averaging period had been used to derive inferences about spatial patterns of soil loss, this could lead to misleading conclusions concerning longer term patterns of landscape development. Thus, understanding spatial patterns of soil erosion for application to landscape development requires the use of information on soil erosion patterns at the appropriate temporal scale.

Table 2 Impact of spatial scale on denudation rates.

Basin area (km^2)	Mean denudation rate ($\text{mm}/1000 \text{ yr}$)
3×10^{-1}	12,600
3×10^0	2,550
8×10^1	220–60
3.9×10^3	100–30
3.7×10^4 – 3.3×10^6	60–30

Source: Adapted from Chorley, Schumm, et al.^[11]

Table 1 Impact of timescale on the status of some variables in soil erosion.

Variable	Status during the designated timescale	
	Short timescale $\leq 10^2 \text{ years}$	Long timescale $\leq 10^4 \text{ years}$
Time of landscape formation	n.a.	I
Initial relief	n.a.	I
Geology	I	I
Paleoclimate	I	I
Paleohydrology	I	I
Relief	I	D
Valley dimensions	I	D
Climate	I	
Vegetation	I	

Note: I, independent; D, dependent.

Source: Adapted from Schumm & Lichty.^[6]

Spatial scale also has a significant impact on soil erosion process and rate data. Soil erosion rates, as indicated by sediment yields back calculated to derive denudation rates (Table 2), typically are greatest at small spatial scales. This important observation is largely because of differences in the relative magnitudes of processes operating at different spatial scales and on sampling techniques. At larger spatial scales, soil erosion, as measured by sediment yield, includes both erosion and deposition. Loss of sediment to deposition on hillslope sections with decreasing gradient and on floodplains significantly reduces net soil loss rates from a landscape,^[7] and this pattern of erosion and deposition is important for landscape form development. In contrast, soil erosion studies at plot scales rarely include areas of deposition and are dominated by studies of small-scale soil detachment and transport processes. Thus, from a landscape development perspective, the extrapolation of rates from small spatial scales to predict landscape change rates is likely to result in the significant overestimation of rates and styles of landscape change.

REFERENCES

1. Kneupfer, P.; McFadden, L. *Soils and Landscape Evolution*; Elsevier: Amsterdam, 1990.
2. Chase, C. Fluvial land sculpting and the fractal dimension of topography. *Geomorphology* **1992**, *5*, 39–57.
3. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
4. Jahn, A. Denudational balance of slopes. *Geogr. Polonica* **1968**, *13*, 9–29.
5. Carson, M.; Kirkby, M. *Hillslope Form and Process*; Cambridge University Press: Cambridge, 1972.
6. Schumm, S.A.; Lichty, R.W. Time, space and causality in geomorphology. *Am. J. Sci.* **1965**, *263*, 110–119.
7. Lane, L.; Hernandez, M.; Nichols, M. Processes controlling sediment yield from watersheds as a function of spatial scale. *Environ. Modell. Softw.* **1998**, *12*, 355–369.
8. Granger, D.; Kirchner, J.; Finkel, R. Spatially averaged long term erosion rates measured from *in situ* produced cosmogenic nuclides in alluvial sediment. *J. Geol.* **1996**, *104*, 249–257.
9. Clapp, E.; Bierman, P.; Schick, A.; Lekach, J.; Enzel, Y.; Caffee, M. Sediment yield exceeds sediment production in arid region drainage basins. *Geology* **2000**, *28*, 995–998.
10. Conrad, C.; Saunderson, H. Temporal and spatial variability in suspended sediment yields from Eastern North America. In *Uplift, Erosion and Stability: Perspectives on Longterm Landscape Development*; Smith, B.J., Whalley, W.B., Warke, P.A., Eds.; Special Publication No. 162; The Geological Society of London: London, 1999; 219–228.
11. Chorley, R.; Schumm, S.; Sudgen, D. *Geomorphology*; Methuen: London, 1984.

Landscape: Elements

Douglas Wysocki

Research Soil Scientist, National Soil Survey Center, Lincoln, Nebraska, U.S.A.

C. William Zanner

University of Nebraska, Lincoln, Nebraska, U.S.A.

Abstract

Soils are landscape entities that possess geomorphic, geometric (surface form), and geochemical character. This is not mere scientific coincidence but reflects the link between landscape evolution and soil formation. Landforms are the basic units that combine to form a landscape. Various often-episodic processes create landforms. Thus, landforms in a landscape may not be comparable to or repeated in other landscapes.

INTRODUCTION: SOIL LANDSCAPES—THE INTRICATE LINK

Landscapes and soils form an intricate, 3-D continuum at the earth's surface. The soil landscape continuum displays remarkable diversity and complexity. To more easily comprehend and effectively manage the soil landscape, we recognize and delineate individual soils. To recognize a soil, we must identify three boundaries—upper, lower, and lateral. The upper soil boundary, the atmosphere to earth's surface interface,^[1] is readily apparent. The lower soil boundary is less distinct, and more difficult to define, but can be an abrupt contact (e.g., bedrock layer) or a gradational change with depth. Soil processes (biologic activity, water and solute transmission, and rooting depth), however, respect no specific limit (e.g., 2 m) and often extend several meters.^[2] Lateral soil boundaries are of greatest concern, as they define soil arrangement and extent on landscapes. Soils occur in a landscape sequence, known as a catena^[3,4] or a soilscape.^[5] No physical interfaces or barriers necessarily separate soils in a catena, rather soil properties change gradually with distance. Soil differences arise from combined geomorphic, atmospheric, and biologic processes that provide energy and material inputs. Soil boundaries often coincide with natural landscape breaks. Soil and landscape unit boundaries occur at observable, predictable positions where energy and mass transfers are restricted or controlled by landscape form, internal structure, or both.^[6] Soil formation and landscape evolution are coeval. Soils originate and evolve by processes that also drive landscape evolution. For example, erosion and deposition concurrently consume and construct adjacent hillslope elements. Erosion partially or entirely removes soil in higher areas. Deposition blends eroded sediments with existing soil or furnishes fresh parent material (sediment) by burying lower areas. Soils are a landscape history and a portrait of geomorphic process.

LANDSCAPES, LANDFORMS, AND BASIC LANDSCAPE UNITS

Landscape unit boundaries include slope gradient inflections, slope shape or aspect differences, contacts between landforms of different origin and composition, and near-surface contacts between rock types or sediment bodies.^[6] Note that landscape unit boundaries are visible or inferable by surface form, are defined and identified independent of soils, and occur in all landscapes. The strong linkage between soils and landscape units allows consistent, reliable, and economical soil prediction.

Simply, a landscape is a geographically associated landform assemblage visible in a single view.^[7,8] Four primary agents (water, wind, ice, and gravity) independently or in combination create and alter landforms through episodic and spatially focused processes. Landforms are earth surface features that have a specific origin, form, and composition; they are the basic landscape unit. Landforms may originate through a single, time-constrained process (e.g., a glacial advance). Generally, however, landforms originate through various processes acting over geologic time. Most landscapes contain landforms of different ages and origins. Landform kind and location depend upon geomorphic processes and history. Soil landscape relationships, therefore, are specific to a geographic region. Specific soil-landform knowledge is only transferable to landscapes with similar geomorphic history.

HILLS AND HILLSLOPES

Hills possess three valuable characteristics of a fundamental landscape unit. Hills occur in most geomorphic settings, are easily identified, and contain segments with consistent linkage to soil occurrence. Hills may initially form through constructional processes (e.g., dunes—eolian

deposition, kames—glacial deposition, and fault scarps—tectonic uplift). Fluvial erosion and deposition, however, are the overarching processes that sculpt and create hills. Gravity and the hydrologic cycle are the universal forcing mechanisms in hillslope evolution and soil formation.

HILLSLOPE DESCRIPTION

A few geometric and geomorphic descriptors accurately define hillslope characteristics. Primary descriptors include slope gradient, aspect, shape, complexity, position, and geomorphic components. Slope gradient is the ground surface inclination from a horizontal plane. We express slope gradient as degrees (e.g., 45°) or as a rise/run percent (e.g., 100%). Aspect, the direction a slope faces, is a compass azimuth (e.g., 225°) or a cardinal direction (e.g., southwest). In the northern hemisphere, north and east facing slopes are cooler and moister than south or west facing slopes. Noted soil differences occur due to aspect.^[9–11]

Slope shape refers to surface form. The vertical profile (upslope and downslope) and elevation contour (across slope) are either straight (linear) or curved. If curved, the shape is convex (upward or outward curve) or concave (downward or inward curve). Combining the vertical profile and across slope shape yields nine possible 3-D forms (Fig. 1). Slope shape influences water movement both overland and throughflow. A slope linear in both profile and contour produces parallel, lateral flow. A convex slope in both profile and contour causes divergent flow, and a slope concave in both profile and contour creates convergent flow (Fig. 1). Slope complexity refers to periodicity and downslope flow path (Fig. 2). Simple slopes are smooth with few undulations, or flow obstructions, and few

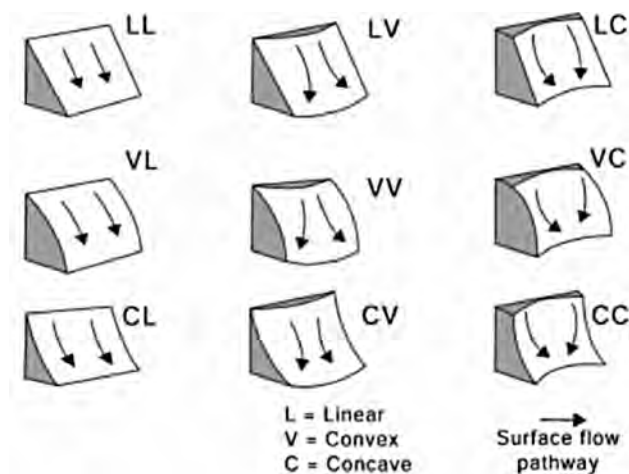


Fig. 1 Slope shapes with surface flow lines based on profile and contour.

Source: From Wysocki, Schoeneberger, et al.^[6] ©2000 CRC Press.

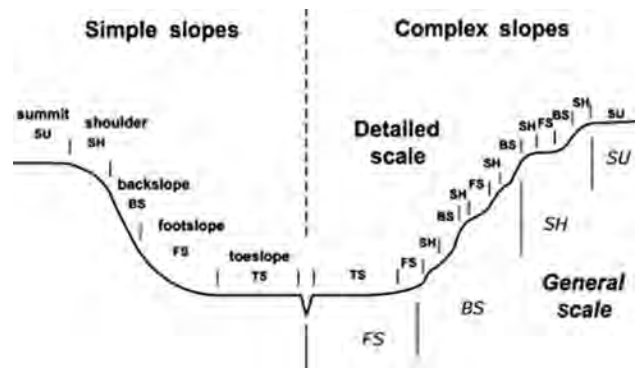


Fig. 2 Simple vs. complex slopes.

Source: From Wysocki, Schoeneberger, et al.^[6] ©2000 CRC Press.

deposition points. Complex slopes contain substantial irregularities or undulations that interrupt surface flow, impede runoff velocity, and create deposition points.

GEOMORPHIC DESCRIPTORS

Landscape elements along a hillslope profile are distinguishable by shape, position, erosion degree, and sediment occurrence and characteristics.^[12] Ruhe^[13] defined five slope elements: 1) the summit, 2) shoulder, 3) backslope, 4) footslope, and 5) toeslope (Fig. 3). The summit is the nearly level, uppermost element. Summit soils commonly exhibit the greatest profile development in a landscape. The shoulder is the convex element below the summit. The shoulder undergoes greater erosion than the summit. Soils tend to be similar to, but thinner than those on the summit, and may be vertically compressed or truncated. The shoulder descends to the steepest and linear slope element, the backslope, where surface runoff and sediment transport are at a maximum. Backslope soils display less development than summit or shoulder soils. The backslope descends to a concave element, the footslope. Surface water flow velocity and transport capacity decrease at the footslope, which results in deposition. Footslope soils tend to accrete upward, are wetter, and lesser developed than soils on higher positions. The footslope merges downslope with the toeslope, which is predominantly linear or slightly concave. The comparatively low slope gradient and

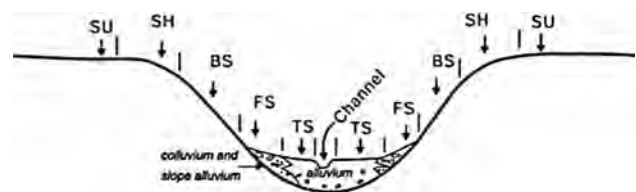


Fig. 3 Hillslope profile elements.

Source: From Ruhe.^[8] ©1975.

position promote alluvial processes as either surface run-on or stream inputs. Toeslope soils tend to be deep, comparatively moist, and composed of alluvial sediments. Together the five slope elements provide a succinct, but eloquent set of landscape units that effectively describe both hillslope positions and soil location.

The 3-D landscape relationships require additional geomorphic descriptors. Ruhe^[8,14] formalized four geomorphic components for open drainage systems the interfluvial, head slope, side slope, and nose slope (Fig. 4). Wysocki, Schoeneberger, and LaGarry's study^[6] has added a fifth, the base slope (Fig. 4). The interfluvial, the uppermost landscape area, is the oldest, most stable component. Regionally, it contains the most well-developed soils. Below the interfluvial, landscapes can undergo backwearing (erosion), which initiates at stream heads and is focused by overland flow. Head slopes, side slopes, and nose slopes direct overland flow: converging (head slope), parallel (side slope), or diverging (nose slope). Flow direction in turn determines erosion and sediment transport. Below the nose slope, head slope, or side slope is the base slope^[6] where sediment deposition or decreased transport capacity dominates over erosion, which results in a colluvial apron.^[15]

Other landscapes (mountains, terraces, and plains) have dynamics and complexity that warrant distinct descriptors. Mountains embody a unique geomorphic setting of size and slope complexity. Mountainsides are complex slopes up to thousands of meters long with steep slope gradients, highly diverse sediment mantles, and elaborate, near-surface hydrology. Mass movement processes and features are more prevalent than in hills. Deep soils, however, may exist over hard rock on steep slopes or mountainsides.^[16,17] Consequently, Wysocki, Schoeneberger, and LaGarry^[6] proposed distinct geomorphic components (e.g., mountaintop, mountainflank, and mountainbase) for mountains (Fig. 5).

Low slope gradient and limited relief characterize two landscape forms. These are terraces and plains. Terraces are

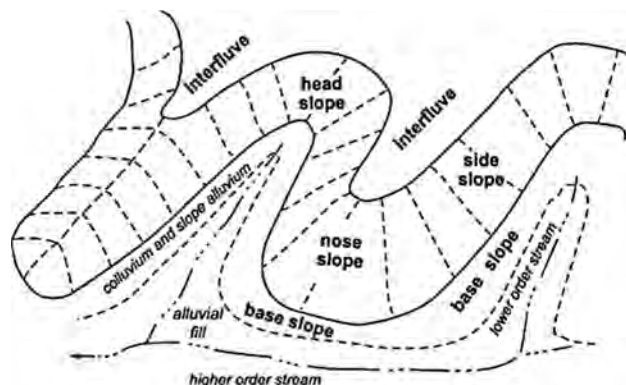


Fig. 4 Geomorphic components: hillslopes.

Source: From Wysocki, Schoeneberger, et al.^[6] ©2000 CRC Press.

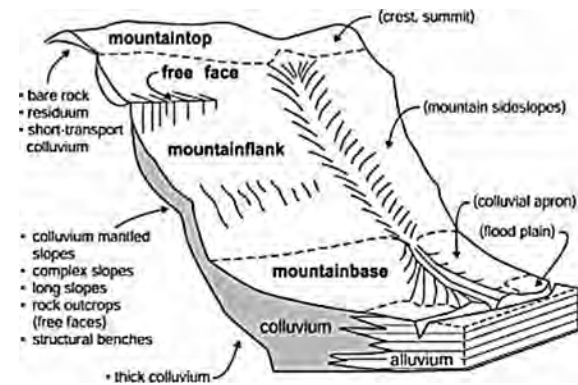


Fig. 5 Geomorphic components: mountains.

Source: From Wysocki, Schoeneberger, et al.^[6] ©2000 CRC Press.

level or gently inclined landforms that border a stream, lake, or sea. Stream terraces develop via alluvial deposition and/or erosion, rather than slope processes. Two geomorphic components exist for terraces (Fig. 6). The tread is a broad, generally level terrace surface. Treads can extend laterally for many kilometers.^[18,19] The riser component is an escarpment that separates terrace or flood-plain levels. The riser is a short, steep slope cut into sediments below the upslope tread. Risers can extend for considerable distances up and down a valley but may be only tens of meters across. Risers mark an abrupt change to a lower hydrologic base level, which suppresses the water table along the higher tread margin. Soils above and adjacent to a riser have better drainage than those farther away do. Daniels et al.^[20] called this the “red edge effect.” Risers can separate soils, sediments, and geomorphic surfaces of significant age difference.

A plain is any landform with limited relief and slope gradient regardless of origin. Lacustrine, coastal, and

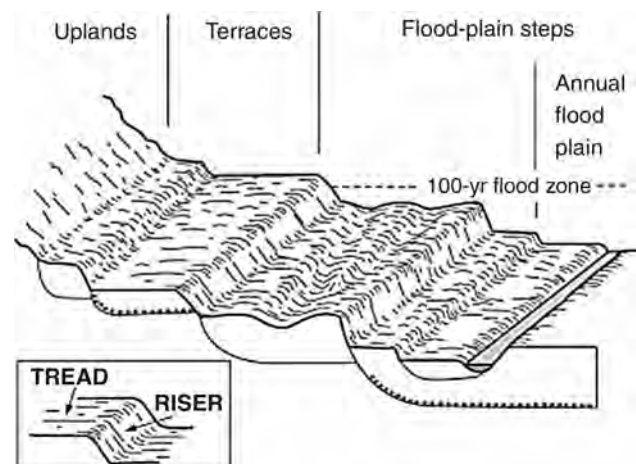


Fig. 6 Geomorphic components: terraces.

Source: From Wysocki, Schoeneberger, et al.^[6] ©2000 CRC Press.

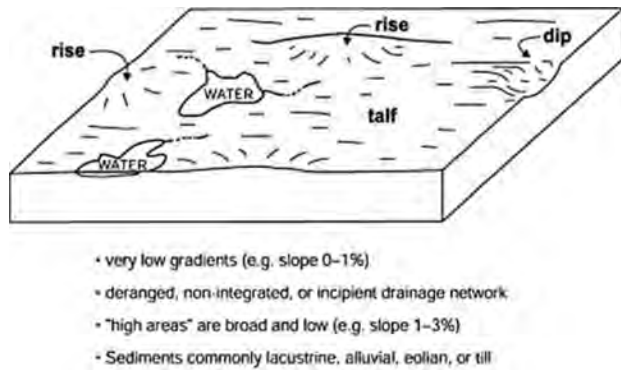


Fig. 7 Geomorphic components (provisional): plains.

Source: From Wysocki, Schoeneberger, et al.^[6] ©2000 CRC Press.

glacial processes can produce distinctly low-relief plains with limited outlets for surface runoff. Low potential energy to drive water flow is the common, critical geomorphic character. Political interest and recognized ecological value of wetlands and hydric soils^[21] highlight the need for unique geomorphic components to describe plains. Small elevation differences (e.g., 15 cm or less) on a plain can result in significant soil and hydrologic differences.^[22] Provisional geomorphic components have been proposed^[15] for plains. Broad, low slope gradient (1%), low-relief areas on plains are “talfs” (Fig. 7). A “rise” is a slightly elevated plain component that exhibits subtle but significant relief (i.e., microhigh) and/or slope gradient differences. A rise is marked by observable soil and ecological differences compared to the adjacent plain or talf. Closed depressions within a plain are termed “dips.” Low relief restricts potential energy in plains. Drainage systems tend to expend energy by transport rather than erosion or incision. Consequently, drainage networks tend to be weakly to moderately developed with limited channel incision, except in active uplift areas. Precipitation tends to pond locally due to low hydraulic gradient and extended paths to outlets. Runoff is comparatively slow both above and below ground. These conditions favor organic matter accumulation (e.g., organic soils and upland bogs) and hydric soil formation.

CONCLUSION

Soils are landscape entities that possess geomorphic, geometric (surface form), and geochemical character. This is not mere scientific coincidence but reflects the link between landscape evolution and soil formation. Landforms are the basic units that combine to form a landscape. Various often-episodic processes create landforms. Thus, landforms in a landscape may not be comparable to or repeated in other landscapes. The most widely known and used geomorphic-based landscape

elements (summit–shoulder–backslope–footslope–toeslope; interfluvial–side slope–head slope–nose slope–base slope) are based on hillslope process and form and incorporate both geomorphic and soil-forming processes within their definition. These term sets succinctly convey landscape position, surface form, and water flow direction and indicate soil occurrence. The terms are most applicable and work well in moderate relief landscapes. Mountainous terrain and low-relief plains warrant greater scientific study and need additional geomorphic descriptors.

REFERENCES

1. Soil survey staff. *Soil taxonomy. A Basic System of Soil Classifications for Making and Interpreting Soil Surveys*, 2nd Ed.; Ag. Handbook 436. U.S. Government Printing Office: Washington, DC, 1999; 1–869.
2. Stone, E.L.; Comerford, N.B. Plant and animal activity below the solum. In *Whole Regolith Pedology*; Cremeens, D.L., Brown, R.B., Huddleston, J.H., Eds.; Special Publication 34; Soil Science Society of America: Madison, 1994; 57–74.
3. Milne, G. Some suggested units of classification and mapping, particularly for East African soils. *Soil Res.* **1935**, *4*, 183–198.
4. Milne, G. *A Provisional Soil Map of East Africa*; East African Agriculture Research Station, Amani Memoirs: Tanganyika Territory, 1936.
5. Hole, F.D.; Lee, G.B. *Introduction to the Soils of Wisconsin*; University of Wisconsin Geological and Natural History Survey Bulletin: Madison, 1955; 79 pp.
6. Wysocki, D.A.; Schoeneberger, P.J.; LaGarry, H.E. Geomorphology of soil landscapes. In *Handbook of Soil Science*; Sumner, M., Ed.; CRC Press: Boca Raton, 2000; E1–E39.
7. Bates, R.L.; Jackson, J.A. *Glossary of Geology*; American Geological Institute: Alexandria, 1987.
8. Ruhe, R.V. *Geomorphology: Geomorphic Processes and Surficial Geology*; Houghton Mifflin: Boston, 1975.
9. Lotspeich, F.B.; Smith, H.W. The Palouse catena: Part 1 soils of the Palouse loess. *Soil Sci.* **1953**, *76*, 467–480.
10. Finney, H.R.; Holowaychuk, N.; Heddleston, M.R. The influence of microclimate on the morphology of certain soils of the Allegheny Plateau of Ohio. *Soil Sci. Soc. Am. Pro.* **1962**, *26*, 287–292.
11. Franzmeier, D.P.; Pedersen, E.J.; Longwell, T.J.; Byrne, J.G.; Losche, C.K. Properties of some soils in the Cumberland Plateau as related to slope aspect and position. *Soil Sci. Soc. Am. Pro.* **1969**, *33*, 755–761.
12. Wood, A. The development of hillside slopes. *P. Geologist. Assoc.* **1942**, *53*, 128–138.
13. Ruhe, R.V. Elements of the soil landscape. In *Transactions of the 7th International Congress of Soil Science*, Madison, 1960; Vol. 4, 165–170.
14. Ruhe, R.V. *Quaternary Landscapes in Iowa*; Iowa State University Press: Ames, IO, 1969.
15. Schoeneberger, P.J.; Wysocki, D.A. Geomorphic descriptors for landforms and geomorphic components: Effective

- models, weaknesses and gaps. Abstract, In *American Society of Agronomy, Annual Meetings*, Indianapolis, IN, 1996.
16. Knox, J.C. Quaternary history of the Kickapoo and Lower Wisconsin River valleys, Wisconsin. In *Quaternary History of the Driftless Area*; Knox, J.C., Ed.; Wisconsin Geological and Natural History Survey: Madison, 1982; 1–65.
 17. Graham, R.C. Geomorphology, mineral weathering, and pedology in an area of the blue ridge front, North Carolina. Ph.D. Dissertation. North Carolina State University: Raleigh, 1986.
 18. Saucier, R. *Geomorphology and Quaternary Geologic History of the Lower Mississippi Valley*; U.S. Army Corps of Engineers Waterways Experiment Station: Vicksburg, 1994.
 19. Gamble, E.E. *Geomorphic Study in the Upper Gasconade River Basin, Laclede and Texas Counties, Missouri*. Soil Survey Investigation Report No. 43; National Soil Survey Center: Lincoln, 1993.
 20. Daniels, R.B.; Kleiss, J.; Buol, S.W.; Byrd, K.J.; Phillips, J.A.. *Soil Systems in North Carolina*; North Carolina State University Agricultural Research Service: Raleigh, 1984; Vol. 467, 1–77.
 21. Mausbach, M.J.; Richardson, J.L. Biogeochemical processes in hydric soil formation. *Curr. Top. Wetland. Biogeochem.* **1994**, *1*, 68–127.
 22. Richardson, J.L.; Arndt, J.L.; Freeland, J. Wetland soils of the Prairie Potholes. *Adv. Agron.* **1994**, *52*, 121–171.

Landscape: Relationships

Timothy A. Quine

School of Geography and Archaeology, University of Exeter, Exeter, U.K.

Abstract

Tillage erosion is inextricably linked to landscape characteristics. The magnitude of tillage erosion rates is determined by the characteristics of the landscape and the spatial variation in soil redistribution by tillage contributes to the evolution of spatial variation in the landscape. The dynamic relationships between tillage erosion and landscape are, therefore, addressed under the headings *landscape sensitivity* and *landscape heterogeneity*. Sensitive landscapes are topographically complex or have a high density of field boundaries, or display both characteristics. Tillage erosion contributes to the evolution of landscape heterogeneity through creation of distinctive landforms, such as lynchets, terraces, and field boundary steps, and through progressive but relatively rapid redistribution of soil from spurs to hollows. The resultant variability in soil properties is an important influence on crop production and water erosion.

LANDSCAPE SENSITIVITY

Tillage erosion potential, described by the tillage transport coefficient, is controlled by and may be predicted from cultivation practice; however, in order to determine rates of tillage erosion, this erosion potential must be combined with information concerning landscape sensitivity to tillage erosion.

Tillage erosion occurs where the outflux of soil (usually downslope) due to tillage translocation exceeds the influx of soil (usually from upslope) due to tillage translocation—accumulation occurs under the opposite conditions. Net soil loss and gain due to tillage erosion are focused on areas in the landscape with discontinuous or rapidly changing flux. Therefore, high landscape sensitivity to tillage erosion is found in landscapes with complex within-field topography or where there are numerous field boundaries located across steep slopes. In general terms, for landscapes of the same average slope angle, complexity and sensitivity to tillage erosion are inversely related to the distance between slope crest and valley, and for landscapes with similar crest to valley distance, relative sensitivity to tillage erosion will be proportional to average slope angle (Table 1).

The location of such sensitive landscapes is determined by lithology and long-term erosion history and, consequently, strong spatial variation in landscape sensitivity to tillage erosion exists even where there is relatively little variation in tillage erosion potential. In a study of topographic controls on medium-term soil redistribution on five agricultural fields in the United Kingdom, slope curvature (and, by implication, tillage erosion) dominated soil redistribution on the four fields underlain by sands, siltstone, and chalk, whereas on the field underlain by clay with flints, no such relationship was identified.^[1] All the fields were

subject to similar cultivation practice and the differences observed represent variation in landscape sensitivity due to lower topographic complexity. This is most clearly reflected in the range of slope profile curvatures: On the clay soils, the range was only 25% of the smallest range observed on the other fields.

High sensitivity landscapes include strongly rolling glacial till lands, incised loess plains, and dry valley systems (Fig. 1).

LANDSCAPE HETEROGENEITY

The discussion of landscape sensitivity has addressed the control that large-scale landscape heterogeneity exercises on spatial variation in tillage erosion rates. Changing the scale of examination to the field/subfield scale sees a reversal in causation and the control of tillage erosion rates on the existence and degree of heterogeneity may be examined. Two aspects are considered: landform creation and landform modification.

Landform Creation

The creation of distinctive landforms provided some of the first evidence for the importance of tillage erosion as a pedo-geomorphic process on agricultural land^[2] and “monuments” of tillage erosion (Fig. 2) associated with boundaries of cultivation may be seen in a wide range of agricultural systems.

Field boundaries form lines of zero tillage flux. In landscapes where relatively steep slopes are divided by cross-slope boundaries, these lines of zero flux represent *significant* flux discontinuities with high influx and zero

Table 1 Indicative mean soil redistribution rates due to tillage erosion.

Soil redistribution rates (t ha ⁻¹ yr ⁻¹)					
Distance (m)	Maximum (slope 10°)	Maximum (slope 5°)	Maximum (slope 3°)	Slope change from 3° to 10° or from 10° to 3°	Slope change from 3° to 5° or from 5° to 3°
50	18	9	5	12	4
40	22	11	7	15	4
30	29	15	9	21	6
20	44	22	13	31	9
10	88	44	26	62	18
5	176	87	52	124	35

Notes: 1) An annual tillage transport coefficient of 500 kg m⁻¹ has been used—This is indicative of one typical mouldboard plough operation and two supplementary operations, e.g., disking and harrowing. 2) These are mean rates and mask significant smaller scale variability. Where the distance refers to the land surface either from the upslope boundary to the location of maximum slope or over which slope angle increases, the rates are mean rates of erosion over the specified distances. Where the distance refers to the land surface either from the location of maximum slope to the downslope boundary or over which slope angle decreases, the rates are mean rates of accumulation over the specified distance.

outflux on the upslope side and high outflux and zero influx on the downslope side. This flux discontinuity results in soil accumulation adjacent to the boundary in the upslope field and soil loss adjacent to the boundary in the downslope field with the consequent formation of a linear step along the boundary line (Fig. 3). Such steps are near universal features of agricultural landscapes:

1. Where fields are or have been narrow contour strips, bench terraces,^[3] or strip lynchets are created—the latter have provided landscape archaeologists with evidence of past land use.
2. When larger fields are located on steep slopes subject to mechanized agriculture, then the steps may develop into large changes in slope—for example, steps 3–4 m high have been identified in the Palouse region of the Pacific Northwest of the United States.^[2]
3. Even in landscapes where water erosion processes dominate geomorphological evolution and cultivation is undertaken by hand, discontinuity in cultivation leads to step formation^[4] and such steps can become



Fig. 1 A landscape of high sensitivity to tillage erosion—seasonally dry valleys and rolling hills near Cordoba in southern Spain. This landscape is also highly sensitive to water erosion.

unstable and act as foci for small-scale mass failure and pipe or gully initiation (Fig. 4).

The formation of these distinctive landforms is associated with local change in soil properties associated with accumulation and loss of plow soil. In landscapes in which rolling topography is divided into relatively large fields, this may have relatively little impact on field-scale soil variation and productivity, although



Fig. 2 Dr. Dino Torri (shown) and Dr. Paolo Bazzoffi coined the phrase “monuments of tillage erosion” to describe features such as this relict land surface surrounding a solitary pine on a strongly convex hill in Tuscany, Italy.



Fig. 3 Evidence of tillage erosion is widely seen in large banks at contour field boundaries on sloping agricultural land subject to mechanized agriculture. This example is from Aarslev in Denmark. **Source:** From Papendick & Miller.^[2] ©1977.

boundary zones of depleted crop production are often visible. However, where moderately steep slopes are divided into narrow strips and rapid lynchet or terrace development takes place (Table 1), the landscape will become characterized by large variation in soil properties over small distances.^[5–7] The significance of this variation will be dependent on the nature of the subplow horizons. If the subplow material is nutrient rich and easily weathered the impact on soil productivity may be small^[7] but if the subplow soil is significantly less productive than the plow soil, significant variation in crop production over distances of a few meters can be expected^[6,8] and a large proportion of the available agricultural land may become unproductive.

Landform Modification

The effect of differential flux on complex landforms is in marked contrast to the effect of zero flux on relatively



Fig. 4 Water erosion is the dominant erosion process on agricultural fields in the rolling hills region of the Loess Plateau but, even here, tillage steps develop at the lower boundaries of fields and these can be focused for mass failure and pipe or gully initiation. **Source:** From Quine, Govers, et al.^[4]



Fig. 5 Tillage erosion of soil from the ridges and transport into the hollows in this field in Minnesota, U.S.A, has led to the development of visible strong within-field spatial variation in soil properties. In this case, the land user is attempting to remediate the problem by manuring the convexities.

simple landforms. While the latter leads to landform creation, the former leads to gradual landform obliteration, although the rate is such that landform modification will usually be a more appropriate description. In essence, tillage erosion leads to a gradual smoothing of the topography and reduction in relief intensity. Convex slope elements are subject to greatest soil loss (Fig. 2) and concavities such as seasonal or relict channel lines are subject to soil accumulation. Tillage acts like a conveyor belt transporting soil from crest to hollow leading to gradual infilling of the hollow and reduction in crest altitude. This creates important synergy with water erosion processes.

This homogenization of landform is, nevertheless, associated with an increase in landscape heterogeneity because of the associated increase in soil variability. Differences in soil properties between crest and hollow become more marked as (topsoil rich) plow soil accumulates in the hollows leading to the formation of overdeepened Ap horizons, while the plow soil on convexities becomes diluted by subsoil inclusion (Fig. 5) and depleted in surface-derived material (e.g., organic matter and surface applied nutrients).^[9] Even where no net soil loss occurs on linear slopes below convexities, downslope tillage translocation of soil from the crest leads to soil depauperation (due to subsoil incorporation).^[10,11]

Systematic spatial variation in water movement and availability driven by topographic controls on hillslope hydrological processes will combine with this erosion-produced systematic soil variability to create a landscape of strong within-field spatial variation in plant growing conditions (Fig. 6)^[12,13] that may be economically very significant where water supply is a limiting factor for crop growth.

While landform modification is usually the most appropriate term, landform obliteration may take place



Fig. 6 Spatial variation in soil properties combines with hillslope hydrology to create significant spatial variation in conditions for crop growth seen in variation in time of flowering in this oilseed rape crop in Devon, U.K.

Source: From Quine & Zhang.^[13]

where slope curvature is very high, slopes are short and implements are large. Under these conditions, planning of landforms^[14] can result in much more rapid landform evolution and strong variation in soil properties over very short distances.

REFERENCES

1. Quine, T.A.; Walling, D.E. Use of Caesium-137 measurements to investigate relationships between erosion rates and topography. In *Landscape Sensitivity*; Thomas, D.S.G., Allison, R.J., Eds.; John Wiley: Chichester, 1993; 31–48.
2. Papendick, R.I.; Miller, D.E. Conservation tillage in the Pacific Northwest. *J. Soil Water Conserv.* **1977**, *32*, 49–56.
3. Dabney, S.M.; Liu, Z.; Lane, M.; Douglas, J.; Zhu, J.; Flanagan, D.C. Landscape benching from tillage erosion between grass hedges. *Soil Till. Res.* **1999**, *51*, 219–231.
4. Quine, T.A.; Govers, G.; Walling, D.E.; Zhang, X.; Desmet, P.J.J.; Zhang, Y. Erosion processes and landform evolution on agricultural land—New perspectives from Caesium-137 data and topographic-based erosion modelling. *Earth Surf. Proc. Land.* **1997**, *22*, 799–816.
5. Agus, F.; Cassel, D.K.; Garrity, D.P. Soil-water and soil physical properties under contour hedgerow systems on sloping oxisols. *Soil Till. Res.* **1997**, *40*, 185–199.
6. Lewis, L.A. Terracing and accelerated soil loss on Rwandan Steeplands: A preliminary investigation on the implications of human activities affecting soil loss. *Land Degrad. Rehabil.* **1992**, *3*, 241–246.
7. Quine, T.A.; Walling, D.E.; Zhang, X. Tillage erosion, water erosion and soil quality on cultivated terraces at xifeng in the loess plateau, China. *Land Degrad. Dev.* **1999**, *10*, 251–274.
8. Turkelboom, F.; Poesen, J.; Ohler, I.; Ongpraset, S. Reassessment of tillage erosion rates by manual tillage on steep slopes in Northern Thailand. *Soil Till. Res.* **1999**, *51*, 245–259.
9. Papendick, R.I.; Young, D.K.; McCool, D.K.; Krauss, H.A. Regional effects of soil erosion on crop productivity—Pacific Northwest. In *Soil Erosion and Crop Productivity*; Follet, R.F., Stewart, B.A., Eds.; American Society of Agronomy: Madison, 1985; 305–320.
10. Van Oost, K.; Govers, G.; Van Muysen, W.; Quine, T.A. Modeling translocation and dispersion of soil constituents by tillage on sloping land. *Soil Sci. Soc. Am. J.* **2000**, *64*, 1733–1739.
11. Quine, T.A.; Walling, D.E.; Chakela, Q.K.; Mandiringana, O.T.; Zhang, X. Rates and patterns of tillage and water erosion on terraces and contour strips: evidence from Caesium-137. *Catena* **1999**, *36*, 115–142.
12. Schumacher, T.E.; Lindstrom, M.J.; Schumacher, J.A.; Lemme, G.D. Modeling spatial variation in productivity due to tillage and water erosion. *Soil Till. Res.* **1999**, *51*, 331–339.
13. Quine, T.A.; Zhang, Y. An investigation of spatial variation in soil erosion, soil properties and crop production within an agricultural field in Devon, UK. *J. Soil Water Conserv.* **2002**, *57* (1), 55–65.
14. Lobb, D.A.; Kachanoski, R.G.; Miller, M.M. Tillage translocation and tillage erosion in the complex upland landscapes of southwestern Ontario, Canada. *Soil Till. Res.* **1999**, *51*, 189–209.

Landslide and Landcreep

Jau-Yau Lu

*Department of Civil Engineering, National Chung-Hsing University,
Taichung, Taiwan*

Hao-Jung Shieh

Taiwan Slope Land Disaster Prevention Association, Taichung, Taiwan

Abstract

Landslide and slope failure are soil mass movement that cause costly damage to property, structures, and even human life. In this entry, “slope failure” is used when the interface between the moving soil mass and the stationary underlayer is unclear and the moving speed is fast. The term “landslide” is used when the interface is clear or gradually varied and the moving speed is relatively slow. The former is usually for small-scale surface failure, while the latter is for large-scale movement of soil mass. The occurrences of both slope failure and landslide are related to the soil types and properties. Also, mass movement becomes flow, such as debris flow, if the soil mass is deformed during the movement. First, the mechanisms that trigger the slope failure, including surface saturation, piping, pressurized groundwater, surface erosion, flow, and earthquake are discussed. The major differences between landslide and slope failure are also identified. Second, the characteristics, classifications, investigation, and mitigation measures of landslide are illustrated. Finally, the distinction between debris flow and mudflow is explained. In addition, three important conditions required for the initiation of debris flow are clearly identified.

INTRODUCTION

Landslide and slope failure are soil mass movement causing costly damage to property, roads, utilities, structures, stream channels, and many times human life. In geomorphology, there are many different classifications for soil mass movement. In general, “slope failure” is used when the interface between the moving soil mass and the stationary underlayer is unclear and the moving speed is fast. The term “landslide” is used when the interface is clear or gradually varied and the moving speed is relatively slow. Also, mass movement becomes flow, such as rock flow, debris flow, and earth flow, if the soil mass is deformed during the movement.

The occurrences of both slope failure and landslide are related to the soil types and properties. Slope failure may occur when a rainstorm saturates the soil layer, and mechanisms such as piping, pressurized groundwater, and/or earthquake trigger it. Landslide is frequently initiated by the weakened cohesive soil or clay layer with a rising groundwater table due to rainfall.

In nature, the distinction between landslide and slope failure sometimes is not very clear. The scale of the failure and movement speed can be used to distinguish them. Slope failure is for rapid, small-scale surface failure and landslide for slow, large-scale movement of soil mass. Mudflow is an extreme case of the debris flow. It mainly carries sediment finer than coarse sand (2 mm).

SLOPE FAILURE

Slope failure refers to the abrupt failure of steep hill slope due to the effects of seepage water and the weight of soil mass. The displacement of the soil mass varies from more than 10 to several hundred meters. Slope failure occurs when a rainstorm saturates the soil layer and one or more of the following mechanisms usually trigger it.

Surface Saturation

When there is a dense and impermeable subsurface layer, a saturated surface may form after a prolonged rainstorm. Water saturation increases the weight of the surface layer and decreases the soil strength. On a steep slope, the weakened surface layer can easily initiate a slope failure. This is the most common type of slope failure, and it usually occurs near the peak of the storm. Although this mechanism is simple, the variability of the soil and geologic properties on the landscape as well as the variability of the rainfall pattern makes it very difficult to predict when and where the slope failure may occur.

Piping

This type of slope failure occurs in sandy soils where interflows or piping is easily formed. First, piping causes the seepage flow to increase suddenly, and a small slope failure

occurs at a certain point on the hill slope. Gradually, larger slope failure occurs above the initial point of piping-induced slope failure. This process may occur at many parts of a hill slope. This type of slope failure starts during the early stage of a rainstorm.

Pressurized Groundwater

This type of slope failure occurs in the places where surface soil layer is thick and the groundwater is easy to concentrate. The mass movement occurs in the valley-shaped soil layer with little disturbance. Due to the topographical and geological conditions, slope failure occurs near bedrock due to the decrease of effective stress from the pressurized groundwater.

Surface Erosion

Initially, a small slope failure occurs at a certain point near the upper portion of the slope surface. When the soil mass of the small slope failure flows down the slope, it may trigger additional down slope failure through surface erosion or failure of saturated surface soil. This type of slope failure usually occurs on the shallow surface layer for a steep, valley-shaped slope.

Flow

When the coarse and loose surface soil layer, containing fine grains and organic material, is saturated, it loses its strength and starts to flow. Some of the material may move as clods, but most of the material tends to move as an earth flow. This type of slope failure usually occurs on bare lands or grass slopes with gentle gradient.

Slope failures may also be induced by an earthquake. According to the analysis of aerial photographs by the Soil and Water Conservation Bureau in Taiwan, slope failures occurred at 2365 locations covering a total area of 14,347 ha in central Taiwan on September 21, 1999, due to a devastating earthquake (100-year return period), 7.3 on the Richter scale.

Several slope failures occurred along the Chin-Shui River as a result of the September 21 earthquake, resulting in an upstream impoundment of water, below the village of Tsao-Ling. The minimum crest of the slope failure at the impoundment was estimated to be approximately 540 m, with an estimated total storage volume of about 43 million cubic meters. This location has a long history of slope failures caused by either earthquakes or heavy rainfall, resulting in the impoundment of water and subsequent breach outflow causing downstream damage and loss of life. The massive slope failure covered an area of 620 ha (about 5 km along the river channel) and with an amount of 126 million cubic meters of deposited material.

Table 1 shows the major differences between landslide and slope failure indicated by Watari.^[1] However, the

Table 1 Differences between landslide and slope failure.

	Landslide	Slope failure
Geology	Occurs in places with particular geology or geological formation	Seldom related to geology
Soils	Moves with cohesive soil as slip surface	Frequently occurs even in sandy soils
Topography	Occurs on gentle slopes of 5–30°, especially if the upper portion is plateau topography	Frequently occurs on the slopes steeper than 30°
Occurrence	Occurs gradually and continuously	Occurs suddenly
Moving speed	Usually low at 0.01–10 mm/day	High speed, greater than 10 m/s
Surface deformation	Soil moves with original shape and little disturbance	Soil mass being disturbed
Provoking causes	Greatly affected by groundwater	Affected by rainfall, especially rainfall intensity
Scale	Large scale, between 1 and 100 ha	Small scale
Warning signs	Cracks, depressions, upheavals, groundwater fluctuations occur before the landslide	Seldom has symptoms and slips down suddenly

Source: Revised from Watari.^[1]

dividing slope between landslide and slope failure (30°) and the high moving speed of slope failure (>10 m/s) are modified.^[2]

LANDSLIDE

Landslide usually occurs continuously or intermittently on gentle slopes at a large scale (usually greater than 1 ha). Sometimes, landslide may shift to an abrupt slope failure or even becomes a debris flow.

The characteristics of landslide include the following:

1. It usually occurs in places with particular geology.
2. The slip surface usually occurs at layers with clay or clayey soils. The movement is mainly induced by the groundwater effect.
3. It usually occurs on gentle slopes. More than 50% of the landslide occurs on slopes with slope angle less than 30°.
4. The large-scale soil mass (1–100 ha) usually moves with its original shape with little surface disturbance. Cracks, depressions, upheavals, and groundwater fluctuations have been observed before the landslide, but in general, landslide is a long-term, continuous, and slow soil mass movement.

5. The moving speed of landslide is usually low at 0.01–10 mm/day. The movement can be either continuous or intermittent. However, the final slope failure may occur rapidly.

The distribution of landslide is highly influenced by the geological structure. In Japan, it usually occurs in 1) Neogene (new tertiary); 2) tectonics; 3) areas affected by the volcanic activity-induced upheavals, depressions, or areas influenced by strong geothermal gradients from volcanic gas or hot springs; 4) areas with the Eocene (old tertiary) shale and tuff belt containing cap rock, or areas along the fault or with coal pit. But the areas with a low degree of consolidation for the quaternary deposits seldom have this type of landslide. In England and North European countries such as Sweden and Norway, landslides occurring in areas with thick clay deposits are of major concern.

Fig. 1 shows the landslide that occurred at Keelung–Hsieh section of Highway 3 in Taiwan on April 25, 2010, due to a dip slope failure (the beds in the slope were inclined subparallel to the slope surface). The total amount of the sliding material was about 100,000 m³. It took the Highway Bureau about 2 months to restore the traffic. According to the field observations, the slip surface occurred at a clay layer, which was consistent with the second characteristics of landslide mentioned above. The rock formations above and below the slip surface are sandstone and shale layers, respectively.

Classifications of Landslide

Landslide can be classified according to the characteristics of geologic structure, formation features and history of landslide development, type of movement, and shape of



Fig. 1 Extreme landslide disaster on Highway 3 in Taiwan on April 25, 2010.

Source: Photo by National Airborne Service Corps, Ministry of the Interior, Taiwan.

longitudinal profile for slip surface.^[3] One of the most famous landslide classifications was given by Varnes.^[4] For example, based on the type of movement, landslide can be classified into the following: falls, topples, rotational and translational slides, lateral spreads, flows and complex movement, etc. Similar information also can be found in U.S. Geological Survey “fact sheet” online.^[5]

Causes of Landslide

Besides geological factors, landslide may be triggered by many causes but can be generally classified into two categories, i.e., natural phenomena and human activities. The natural triggers of landslide include rainfall, snowmelt, groundwater, earthquake, channel incision, and hill slope failure. The human-induced factors include cut and fill of roadways, mining, excavation of tunnels, and construction of reservoirs.

Investigation of Landslide

Investigations of landslide include the collection of information on topography, soil, and geology; monitoring the weather, groundwater, and movement of the soil mass; and the performance of slope stability assessment for slope failure hazard potential. More detailed procedure can be found in the nomograph published by Japan landslide society.^[2]

Landslide Mitigation Measures

Landslide mitigation measures are selected based on the provoking causes. In general, they can be classified into two categories, i.e., the precaution measures and the prevention measures. The precaution measures rely on a detailed geologic and hydrologic database to predict the potential for landslide occurrence and a real-time monitoring system to provide information such that warning can be issued when a slope failure event becomes imminent. The prevention measures include the control works and the restraint works. The control works are used to eliminate the provoking causes of landslides. The restraint works are used to increase the shear strength of the soil layers and to slow down the movement of the soil mass. Fig. 2 shows the detailed landslide mitigation measures.

DEBRIS FLOW AND MUDFLOW

Debris flow is a mixture of water, sand, and large solid particles. It has strong impacting and destructive forces as it moves downstream a channel. Fig. 3 shows a severe debris flow disaster that occurred on October 9, 1973, near the Zhiban bridge, Zhiban Creek in eastern Taiwan. Zhiban Creek basin has abundant hot spring

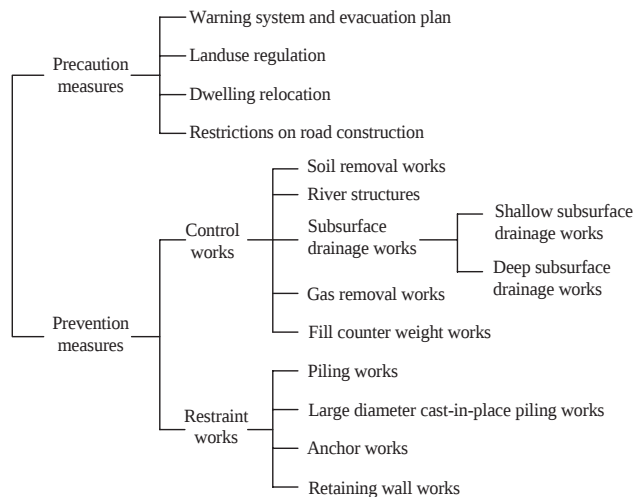


Fig. 2 Landslide mitigation measures.

resources. The gravel/sand, slate, and metasandstone layers were identified based on the drilling information. The debris flow was caused by the heavy rainfall from Typhoon Nora. The amount of deposited sediment was so large that the surface of the deposited material almost reached the lower surface of the bridge deck. As shown in the figure, the temple was almost completely buried.

Mudflow is an extreme case of the debris flow. It mainly carries sediment finer than coarse sand (2 mm) but sometimes also a small amount of rock fragments. There are two types of mudflow.^[6] The first type occurs in the active volcano areas, and it contains volcanic ash. The second type occurs in the tertiary or dormant volcano areas, and it contains the old volcanic ash and weathered tephra. The moving speed of the mudflow is usually at least twice faster than the ordinary debris flow, which contains more coarse particles.

In general, the initiation of debris flow is related to the following three conditions: 1) available loose solid material; 2) steep slope gradient; and 3) heavy rainfall. The loose



Fig. 3 A severe debris flow disaster near the Zhiban bridge in Taiwan due to the heavy rainfall from Typhoon Nora (October 9, 1973).

solid material may be due to the prior landslides, land creeps, typhoons, earthquakes, and volcanic eruptions. The gravitational force associated with steep slopes is an important driving force for the initiation of debris flow. An intensive rainstorm is usually the triggering factor for the formation of a debris flow.

REFERENCES

1. Watari, M. *Mechanism and Mitigation Measures of Slope Disasters*; Sankaido Co.: Tokyo, 1986; 170 pp.
2. Japan Landslide Society. *Landslides in Japan*; National Conference of Landslide Control: Japan, 2002; 64 pp.
3. Takai, A. *Soil Conservation*; Sankaido Co.: Tokyo, 1990.
4. Schuster, R.L.; Krizek, R.J., Eds. *Landslide, Analysis and Control*; National Academy of Sciences: Washington, DC, 1978.
5. USGS. *Landslide Types and Processes. Fact Sheet 2004-3072*, July 2004; 2 pp. Available at <http://pubs.usgs.gov/fs/2002/3072>.
6. Ikeya, H. *Debris Flow Disaster Investigation*; Sankaido Co.: Tokyo, 1980.

Law and Legal Issues

Ian Hannam

Centre for Natural Resources, Department of Infrastructure, Planning and Natural Resources, Sydney, New South Wales, Australia

Ben Boer

School of Law, University of Sydney, Sydney, New South Wales, Australia

Abstract

This entry provides a short background to the international and national soil law and outlines moves to promote reform of soil law including progress made on developing appropriate frameworks for successful management of soil as an ecological element.

INTRODUCTION

At a national level, soil law means a body of law to promote soil conservation enacted by a legislature, e.g., an act, decree, regulation, or other formal legal instrument that is legally enforceable. Soil law, or “soil legislation” as it may also be referred, includes those laws that have primary responsibility for soil conservation, soil and water conservation, and land rehabilitation. They are generally characterized by provisions to mitigate and manage soil erosion and soil degradation and methods to conserve soil resources. Internationally, the legal framework for the conservation of soil can include conventions, protocols, agreements, and covenants, which are expressed to be legally binding. Worldwide, soil law is managed by a variety of legal and institutional systems, which are the individual organizational and operational regimes that have the administrative authority over soil.

WHY LAW FOR SOIL?

Soil bodies are effectively large ecosystems and comprise fundamental components of the earth's biodiversity. Soil is thus seen as the basis for the conservation of terrestrial biological diversity and the sustenance of all terrestrial organisms, including people. The widespread soil degradation as a result of human use of soil provides the imperative for enactment of soil law. The ever-increasing demand for food by rapidly growing populations in many countries in the past few decades has exerted increasing environmental stress on the soil leading to widespread soil degradation.^[1] The following definitions provide the context for soil law.

Soil

Soil forms an integral part of the earth's ecosystems and is situated between the earth's surface and bedrock. It is

subdivided into successive horizontal layers with specific physical, chemical, and biological characteristics. From the standpoint of history of soil use, and from an ecological and environmental point of view, the concept of soil also embraces porous sedimentary rocks and other permeable materials together with the water that these contain and the reserves of underground water.^[2]

Soil Degradation

Soil degradation is a loss or reduction of soil functions or soil uses. It includes aspects of physical, chemical, and biological deterioration, including loss of organic matter, decline in soil fertility, decline in structural condition, erosion, adverse changes in salinity, acidity, or alkalinity, and the effects of toxic chemicals, pollutants, or excessive flooding.^[1]

The Sustainable Use of Soil

The sustainable use of soils preserves the balance between the processes of soil formation and soil degradation while maintaining the ecological functions and needs of soil. In this context, the use of soil means the role of soil in the conservation of biological diversity and the maintenance of human life.^[3]

INTERNATIONAL LAW AND SOIL

International environmental law is an essential component for setting and implementing global, regional, and national policy on environment and development. There is an increasing recognition of the role of international environmental law to overcome the global problems of soil degradation, including its ability to provide a juridical basis for action by nations and the international community.^[4] A number of international and regional instruments introduced in the past years contain elements that can

contribute to achieving sustainable use of soil. None are sufficient on their own. Some of the instruments could assist by promoting the management of some of the activities that can control soil degradation. However, this role is not readily apparent except for those that include provisions specifically directed to soil (e.g., see Article IV “Soil”—1968 *African Convention on the Conservation of Nature and Natural Resources*, final revision text adopted by the African Union Assembly on July 11, 2003).

Declarations

A number of non-binding declarations and charters draw attention to the fact that soil degradation and desertification are reaching alarming proportions and seriously endangering human survival. They call on states to cooperate and develop the tools to conserve soils. Key declarations relevant to soil include the 1972 *Stockholm Declaration on the Human Environment*, the 1981 FAO World Soil Charter, the 1982 *World Charter for Nature*, the 1982 *Nairobi Declaration*, the 1992 *Rio Declaration on Environment and Development*, and the 2002 *Johannesburg Declaration on Sustainable Development*. Also of relevance is the Programme for the Development and Periodic Review of Environmental Law for the First Decade of the Twenty-First Century, known as the Montevideo Programme; this program includes provisions to improve the conservation, rehabilitation, and sustainable use of soils.^[5]

International Conventions, Covenants, Treaties, and Agreements

Many multilateral agreements include provisions that could be used to promote sustainable use of soil, but the provisions are generally tangential to the needs of soil as such. Key global instruments relevant to soil include the 1992 *Convention on Biological Diversity*, the 1992 *United Nations Framework Convention on Climate Change*, the 1997 *Kyoto Protocol*, and the 1994 *United Nations Convention to Combat Desertification*. Relevant regional instruments include the 1968 *African Convention on the Conservation of Nature and Natural Resources* (Revised July 2003), the 1985 *ASEAN Agreement on the Conservation of Nature and Natural Resources*, the 1986 *Convention for the Protection of the Natural Resources and Environment of the South Pacific Region*, the 1986 *European Community Council Directive*, the 1995 *Convention Concerning the Protection of the European Alps*, and the 1998 *Protocol for the Implementation of the Alpine Convention of 1991 in the Area of Soil Protection*.^[6]

NATIONAL SOIL LAW

Legislation has been used for some years in many countries to control soil degradation problems and to manage

soil. A worldwide examination of national legal and institutional frameworks indicates that most countries approach the management of soil in a fragmented manner. The term “soil law” also covers those situations where comprehensive provisions for soil protection and management have been integrated in legislation that protects other aspects of the environment, such as forests, water, biodiversity, and desertification. In general, soil law thus provides for farm planning, implementation of soil erosion control measures, establishing community groups, planning catchment schemes, and compliance and enforcement. Some jurisdictions, such as the United Kingdom, have multiple soil legislation mechanisms that cover a broad range of functions including soil planning, access to sensitive land types, organic farming practices, nitrate sensitive areas, and soil restoration. On the other hand, federally organized countries often have a system where each state or province has its own soil legislation and supportive legal mechanisms. Hybrid situations also exist, such as in the People’s Republic of China, which has enacted the *Water and Soil Conservation Law 1991* and the *Desertification Law 2002* at a national level, but causes them to be implemented through a comprehensive provincial system of law and regulations.

There is a wide variety of types of legal mechanisms used to protect and manage soil, including acts, decrees, resolutions, ordinances, codes, regulations, circulars, decisions, orders, and bylaws. Whereas these are generally appropriate, many need to be applied in more inventive ways to effectively manage the soil in an ecosystem context.^[3]

EFFECTIVENESS OF SOIL LAW

The effectiveness of international and national soil law is generally dependent on two matters: first, the capacity of a legal and institutional framework to manage soil—which is measured by the ability of a legislative and institutional system to achieve sustainable use of soil—and second, by the number and type of essential legal and institutional elements present in a soil statute in a format that enables soil degradation issues to be identified. These need to be backed by the legal, administrative, and technical capability in the particular instrument as a basis of some form of effective action. Capacity is also represented in the form of legal rights, the type of legal mechanisms, and importantly, the number and comprehensiveness of the essential elements and their functional capabilities. Legal and institutional “elements” for soil are the basic, essential components of a legal and institutional system. An individual law can include a number of legal mechanisms in a well thought-out structure that gives an organization the power it needs, through its executive and administrative structure, to address soil degradation. It is also possible that the necessary elements may be distributed among a number of individual laws within a comprehensive national legal and institutional system.^[7]

Most key soil management issues are multifactorial (i.e., many include a sociological, a legal, and a scientific component), so it is obvious that generally more than one piece of environmental legislation (along with detailed regulations) and many types of legal and institutional elements will be needed to effectively manage soil degradation issues.^[7] Legal and institutional elements can be used to assist in the evaluation of an existing law or legal instrument to determine its capacity to meet certain prescribed standards of performance for the sustainable use of soil. They can also be used to guide the reform of an existing soil law or to develop new legislation for the sustainable use of soil. The manner and degree in which an “essential element” is applied will vary according to the particular type of legal mechanism concerned and its expected role in a particular jurisdiction. For example, an international legal instrument may include a provision for dispute resolution, but the actual implementation of this provision between states might not rely on, or be influenced by, the existence of similar provisions within a law of either of the disputing states.^[7]

IUCN COMMISSION ON ENVIRONMENTAL LAW

The Commission on Environmental Law of International Union for Conservation of Nature (The World Conservation Union) has carried out extensive investigation into the options for a new international instrument focusing on soil. The commission has also identified a variety of ways available for states to approach the task of a detailed legal and institutional analysis and the design of appropriate legal and institutional systems that provides for the effective management of soil. Arising from this work, in which the authors have been centrally involved, two principal strategies can be considered for the development of legal and institutional arrangements for soil. These are as follows:

A non-regulatory strategy that is characterized by elements for education, participatory approaches, soil management, and incentive schemes

A regulatory approach that is characterized by statutory soil use plans that prescribe legal limits and targets of soil and land use, issue of licenses or permits to control soil use, and the use of restraining orders and prosecution for failure to follow prescribed standards of sustainable soil use.

These strategies can be approached on a short-term time frame for implementation or a longer term time frame,

which involves substantial reform of existing laws, policies, and institutional and sectoral change.^[7]

CONCLUSION

Soil law has been neglected at the international level and, in many of the world's regions, at the domestic level. However, the growing recognition of soil degradation as a major international environmental issue in the context of the conservation of biological diversity is gradually being addressed, and this is starting to change attitudes toward the benefits of improved international and national legal and institutions for soil.^[8] Soil bodies represent complex terrestrial ecosystems. They require careful management of their ecological characteristics through the medium of soil law at a national and international level. This approach is essential for the long-term sustainable use of soil and to meet the food production requirements of the expanding human population of the world as well as to meet the needs of all flora and fauna that depend on the soil for sustenance.

REFERENCES

1. Bridges, E.M.; Hannam, I.D.; Oldeman, L.R.; Penning deVries, F.; Scherr, S.J.; Sombatpanit, S. *Response to Land Degradation*; Science Publishers Inc.: Enfield, 2001.
2. Council of Europe European Conservation Strategy. *Recommendations for the 6th European Ministerial Conference on the Environment*; Council of Europe: Strasbourg, 1990.
3. Hannam, I.D.; Boer, B.W. *Legal and Institutional Frameworks for Sustainable Soils*; The World Conservation Union: Gland, 2002.
4. Khan, R. International law of land degradation. *International Studies* **1993**, 30 (3), 255–275.
5. UNEP. *The Montevideo Programme III—The Programme for the Development and Periodic Review of Environmental Law for the First Decade of the Twenty-First Century*; 2001 Decision 21/23 of the Governing Council of UNEP, UNEP: Nairobi, 2001.
6. Sands, P. *Principles of International Environmental Law*; Cambridge University Press: Cambridge, 2003.
7. Boer, B.W.; Hannam, I.D. Legal aspects of sustainable soils: International and National. *Rev. Eur. Commun. Int. Environ. Law* **2003**, 12 (2), 149–163.
8. WSSD (World Summit on Environment and Development). *A Framework for Action on Agriculture*; WEHAB Working Group: New York, 2002.

Law and Legal Issues: Land as Property

Paul Martin

*Australian Centre for Agriculture and Law, University of New England, Armidale,
New South Wales, Australia*

John Page

Bond University, Gold Coast, Queensland, Australia

Abstract

Soil management exists at the intersection between natural systems and human systems. Property rights are an important element in the transactions between man and the land. Soil health relies on property rights regimes that encourage the owner of the property right (individual or communal) to achieve their private goals, in ways that are sustainable. In this entry, the linkages between property, society, and soil health and some myths about property rights are explored.

INTRODUCTION

This entry considers the nature of property rights as institutional arrangements for the exploitation of natural resources and the role that they can play in conservation. It looks at myths about property and dangers that these pose for policy. This entry begins by considering property as a key element underpinning the transactions between social and natural systems.

A transaction between man and soil involves a decision, triggering a flow of either information or material things. Thus, a soil scientist may interpret “signals” from the soil, while the farmer may decide to plant and irrigate, and then interpret the results of these decisions in deducing what to do next. Behind these transactions and the elements that are shaping them are institutions made up of rules and organizations. Examples of rules include zoning for land use, environmental controls, water allocation rules, rules applied by financiers or communities, and rules agreed between neighbors.^[1] Property is a fundamental institutional arrangement in both capitalist and non-capitalist economies (though the nature and management of property vary with jurisdiction). It comprises both the legal rules and the institutional arrangements of ownership. Owners (those with a property right) are motivated to pursue gain, using those resources in an economically productive way.

PROPERTY

Legal property consists of rights to things, which society will enforce on behalf of the citizen. “What society

will enforce” is not universal or fixed. It changes with culture and society over space and time. This is contrary to the common perception of property law as an unchanging structure. Rigidity may be true for short term and for most purposes, but it is not true across time and space. Carbon trading, biodiversity markets, and weather futures require property rights unknown to a prior generation.

From Roman law, there is a legal distinction made between things unowned (*res nullius*), owned by the state, and owned by individuals (*res privatus*) or by groups of individuals (*res communales*). Under the British “common law” heritage, property is a theoretical construct with the flexibility to create new rights to meet new social or economic requirements. Property rights can be envisaged as a “bundle of rights” capable of division and adjustment by agreement (the term bundle of rights was first cited in the 1888 *Treatise on the Law of Eminent Domain in the United States* by John Lewis, cited in Goldstein^[2]). This bundle includes the right to possess, the right to alienate, the right to exploit, the right to control (and thereby exclude), and the right to the “fruits” of the thing, among others. These hallmark rights are core attributes of property. As a minimum, they need to be stable, ascertainable, and capable of assumption by third parties, for the law to give the imprimatur of property.

Because of the link between wealth, political freedom, and increased economic productivity (for those with property or a lack of wealth and social injustice for those without), property is a frequent element in political conflicts. In some societies such as the United States, a history of conflict led to an entrenchment of

private property as a civil right.^[3] In others, struggles to overcome poverty and injustice led to an elevation of other social interests over ownership. Property rights are a tool of social policy.^[4]

The linkages between soil health and property rights are many. Ownership of private land carries the right to exploit that resource to achieve private outcomes that may be harmful. The owner may place a premium on exploitation to achieve short-term gain rather than the long-term public good of soil health. An example is depletion of soil through excessive chemical addition to secure short-term profits. Conversely, ownership also provides an incentive to maintain soil health for further exploitation and to protect its productive capacity.

MARKETS

The trend in developed economies has been to extend the private property rights to things that were previously not the subject of ownership. They include rights of extraction (water), rights to contaminate (sulfur dioxide emissions), rights to conserve (wildlife), and rights to use (grazing). The ownership interests span time-limited interests (licenses, permits, or rights to profits) through to permanent allodial-like ownership. The divisibility of property rights allows new or recast rights to be added to the overall bundle as “green wood” rights.^[2] These can facilitate environmentally positive outcomes but equally can enable further exploitation of the resource.

Along with property rights, markets are often created. This is done by legally excluding valued use by non-owners. This then means that non-owners have to buy interests from the owners of the newly created rights. Examples include water markets, carbon markets (where emitters in effect buy sequestered carbon), pollution offsets, and “bio-banking” (where those who wish to reduce biodiversity in one area are obliged to purchase protection or rehabilitation in another).^[5]

Increasing ecological concerns are reflected in the trend to restrict the exercise of traditional property rights by regulation. Where this constraint is very strong, it can be argued that the property right has been removed or fundamentally changed. This may be sufficient in some jurisdictions to trigger rights to compensation for “taking.”^[6] Regulatory control for soil conservation has long been identified by prescient commentators as necessary to reduce loss of soil organic matter, to properly regulate consumption, and to increase the level of soil organic matter over time.^[7]

As with the use of property rights for social and economic purposes, regulation can also result in undesirable effects. Economic models often suggest that tradable entitlements using property are more efficient than regulation to achieve environmental and social policy goals. In practice (given social and economic complexity

and varying institutional performance), the results are more ambiguous than the models. The empirical evidence is that both regulation and market approaches share the potential to fail or cause perverse effects, unless carefully tailored and well implemented by effective institutions.

MYTHS

Myths and misunderstandings exist about what is required to use property rights successfully. Property is cultural. A property instrument that works in one society can fail in another. The most useful example of ignoring cultural effects is the belief that common property systems fail as methods of resource management in developed economies. This is referred to as the “Tragedy of the Commons.”^[8] There is sufficient truth in the Tragedy of the Commons myth to pay attention to the risks of common property, but sufficient empirical evidence to suggest that tragedy is not preordained. Even in developed economies, common property can be pivotal to economic activity (as the conventional public company with ownership allocated by shares demonstrates). It is often overlooked that one of Hardin’s prescriptions to avoid the tragedy was the use of less than fee simple property rights in threatened resources to encourage their protection and sustainable use.^[9]

A property myth that can mislead policymakers is the concept of unattenuated property rights. Many commentators suggest that a private market cannot efficiently work unless the owners of property have very secure rights. While it is understandable that private owners will want secure rights, the empirical evidence is that, provided there is a sufficient perceived opportunity for gain, entrepreneurs will innovate and exploit even given property right uncertainties. The historical evidence of the flowering of private enterprise at times when the State had absolute freedom to take is sufficient to prove this point. The modern example of trading in options where exercise is uncertain serves to reinforce it. The danger in property right mythologies is that they may lead to naive policy formulation that will not achieve optimal results.

CONCLUSION

Property rights are a powerful tool, but like all tools they depend on being used well, in the right situation. Property rights can adjust interests and harness self-interest to social goals. However, their effect is conditioned by other institutional arrangements and the context into which they are introduced. Good design of a property regime is not a matter of adopting some category of right and then expecting that this alone will ensure desirable outcomes. Sophisticated design and efficient institutions are critical. Even then it is necessary to manage

implementation with an eye to potential failures. The most difficult concern is social justice for a private market works, by facilitating the movement of resources from the less economically competent to those most efficient at the use of resources. Ensuring that this occurs in a way that is not unjust is a difficult challenge that few societies have achieved.

REFERENCES

1. Robert, C.E. *Order Without Law: How Neighbors Settle Disputes*; Harvard University Press: Cambridge, 1991.
2. Goldstein, R.H.J. Green wood in the bundle of sticks: Fitting environmental ethics and ecology into real property law. *B. C. Envtl. Aff. L. Rev.* **1998**, *25*, 347.
3. Paul, E.F.; Dickman, H., Eds. Liberty, property and the future of constitutional development. In *SUNY Series in the Constitution of Economic Rights*; State University of New York Press: Albany, 1990.
4. Weimer, D., Ed. *The Political Economy of Property Rights: Political Economy of Institutions and Decisions*; Cambridge University Press: Cambridge, 1997.
5. Martin, P.; Verbeek, M. *Sustainability Strategy*; Federation Press: Sydney, 2006.
6. Sax, J.L. Property rights and the economy of nature: Understanding *Lucas v South Carolina Coastal Council*. *Stanford Law Rev.* **1992–1993**, *45*, 201.
7. Albrecht, W.A. Loss of soil organic matter and its restoration. In *Soils and Men, Yearbook of Agriculture 1938*; U.S. Department of Agriculture: Washington, 1938; 347–360.
8. Hardin, G. The tragedy of the commons. *Science* **1968**, *162*, 1243–1248.
9. Adler, J. Back to the future of conservation: Changing perceptions of property rights & environmental protection. *1 N. Y. Univ. J. Law Liberty* **2005**, *987*, 1021.

Leaching and Illuviation

Lynn E. Moody

Earth and Soil Sciences Department, California Polytechnic State University,
San Luis Obispo, California, U.S.A.

Abstract

The dynamic system of soils includes various types of translocations, involving movements of materials in aqueous solution or suspension, from one part of the solum (leaching or eluviation) into another part of the solum (illuviation). During translocation, organic and inorganic materials may be deposited into the upper part of the soil profile or produced or liberated in the upper part of the profile by physical, chemical, and biological weathering. The dominant translocation process in a soil profile depends partially on the form of the materials (inorganic or organic; solutes or particles). At a given stage of soil development, one process may be dominant or several may operate simultaneously or sequentially. In all cases, percolating water is the mechanism of translocation, and the rates and intensities of translocation processes are controlled by the factors that regulate water percolation: climate, topography, and tortuosity of soil pores. The last is usually quantified and expressed as hydraulic conductivity.

LEACHING

Leaching is the translocation of solutes. Some authorities specify that leaching is the removal of solutes entirely out of the solum, representing a loss of materials from the soil profile,^[1] but according to many experts, leaching includes the translocation of solutes within the solum.^[2] Solute arise mainly by chemical weathering and include ions, complex ions, and ion pairs.

Mineral weatherability depends on several factors of mineral and soil chemistry. It also depends on particle size (smaller → more soluble). If chemical bonds between ions in a mineral are ionic in character, then the ions readily detach from the mineral surface and enter solution,^[3] as in the congruent dissolution of halite into $\text{Na}^+(\text{aq})$ and $\text{Cl}^-(\text{aq})$. Chemical bonds with some covalent character, as well as ionic character, are less easily disrupted. Because they generally require a strong polarizer such as H^+ , dissolution proceeds less readily,^[3] as in the incongruent weathering of feldspar: $2\text{KAlSi}_3\text{O}_8 + 11\text{H}_2\text{O} + 2\text{CO}_2 \rightarrow 2\text{K}^+(\text{aq}) + \text{Al}_2\text{Si}_2\text{O}_5(\text{OH})_4 + 2\text{HCO}_3^-(\text{aq}) + 4\text{H}_4\text{SiO}_4(\text{aq})$. In this reaction, particulate kaolinite, as well as solutes, is produced.

The relative weatherability of the common silicates was tabulated by Goldich.^[4] Olivine, pyroxenes, and calcium (Ca) plagioclases are most weatherable, and quartz is most resistant (Fig. 1). Of the common non-silicates, sodium (Na) salts (including sodium carbonates) and chlorides of Ca, magnesium, and potassium are soluble, sulfates less so, and carbonates are slowly soluble. The solubility of iron (Fe) and manganese (Mn) oxides depends on the oxidation state of the metals; reduction of Fe and Mn in saturated soils causes their oxides to dissolve. As minerals weather and

elements are leached away, the soil unit decreases in mass and volume.^[5]

When chemical conditions in the soil are suitable, solutes precipitate, usually deeper than the origin of the solutes. One or more of several conditions, including the following, may initiate precipitation of solutes. Desiccation, the removal of water by evaporation or sorption by roots, may cause precipitation of soluble salts, gypsum, carbonates, or silica (Si). Reduction in carbon dioxide partial pressure below the zone of maximum biological activity is an additional cause of pedogenic carbonate precipitation. Solubility of many minerals is partially dependent on pH; therefore, a change in soil pH may also cause precipitation. Oxidation of Mn and Fe causes the oxides and oxyhydroxides of these metals to precipitate. As materials precipitate in the soil, mass is added and the volume increases.^[5]

PODZOLIZATION

Podzolization is a specific term for such leaching and precipitation of Fe and aluminum (Al) chelates within the solum. Such leaching may occur in humid climates, acidic soil conditions, and characteristically under coniferous or mixed coniferous–deciduous forests. The types of soils formed by podzolization are Podzols, generally equivalent to Spodosols in the U.S. taxonomy. The morphological expression of podzolization is a Bs, Bh, or Bhs (spodic) horizon, where the h symbolizes humus and s denotes Al and Fe sesquioxides. Often, the spodic horizon is overlain by a highly leached E (eluvial) horizon. A spodic horizon may be cemented by enrichment in humus, Al, Fe, and Si,

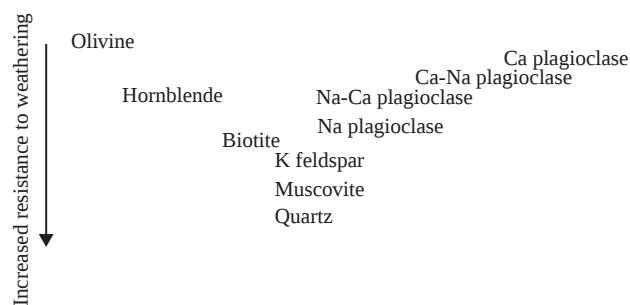


Fig. 1 Goldich's mineral stability series in weathering of the common rock-forming silicate minerals is displayed. Minerals least resistant to chemical weathering are at the top; resistance to weathering increases downward. Quartz is most resistant.

Source: From Goldich.^[4]

in which case it is known as *ortstein*. A placic horizon is a thin, wavy, cemented layer enriched in humus and Fe or Fe and Mn that is often associated with Podzols.

Two models have been developed to describe the podzolization process. The first model, favored by most authorities, involves the translocation of Al and Fe as organometallic complexes.^[6] Fulvic acids, components of humus, are formed during the decomposition of acidic leaf litter, especially litter from conifers and ericaceous (heath) plants. Humification is complex and probably a combination of processes involving the breakdown of lignin and the synthesis of phenols and aromatics from carbohydrates and amino acids.^[7] Fulvic acids are water-soluble anions under acidic soil conditions and, thus, mobile with percolating water. They are rich in phenols and carboxyls; the juxtaposition of these functional groups enables the organic matter to chelate trivalent cations Al and Fe. The organometallic complexes are carried downward in the percolating water. In an aerated, acidic environment, the metal cation/organic anion ratio mainly controls the solubility of the organometallic complex. If the cation/anion ratio is low, the complexes are soluble and, thus, mobile; they become less soluble by increases in the cation/anion ratio.

Precipitation of the organometallic complexes may be initiated at depth in the solum because of an increased ratio of the chelated trivalent cation to the organic anion. This shift in the cation/anion ratio may be brought about by microbial decomposition of the fulvic acids or by saturation of the complexes by greater concentrations of Al and Fe released by weathering. As all available chelation sites are filled, the molecules lose polarity, become hydrophobic, and precipitate. Precipitation also may be initiated by desiccation, sorption of the complexes onto mineral surfaces, or flocculation in the presence of divalent cations. As the organic matter eventually decomposes, Al and Fe are released from complexation and are free to form oxyhydroxides or sesquioxides. With the addition of Si, Al can form imogolite and allophane.

The second model, contrasting model of Podzol formation, holds that the dominant translocation process is inorganic and that organic acids are involved mainly in weathering rather than in transport.^[8,9] Weathering of primary minerals by carbonic and organic acids at the soil surface releases Al, Fe, and Si. At pH < 5, hydroxyaluminum cations react with Si to form stable, mobile, inorganic "proto-imogolite" complexes. The complexes precipitate in deeper horizons as imogolite and allophane, by desiccation or reaction to higher pH. Evidence supporting this model consists of Podzols containing Al predominantly as imogolite, allophane, and proto-imogolite and Fe as oxides, with Al-fulvic acid chelates sorbed and precipitated on allophane surfaces.^[10]

These two models may not be exclusive. Early podzolization in some Greenland soils involved translocation of Al and Fe by inorganic processes, whereas later stages involved their transport in complexes with organic acids.^[11] However, most studies of Podzols favor the organic model,^[12,13] even in the early stages of podzolization.^[14]

ILLUVIATION OF CLAYS

As the finest particles in soils, clays (<2 mm diameter) are most susceptible to eluviation (removal) and illuviation (deposition). Fine clays (<0.2 mm), in particular, are small enough to be mobile in soil profiles. Larger particles, even silts and very fine sands, may be mobile in some soils, depending on the size and geometry of pores and velocity of percolating water.^[15] Clays originate in soils by physical reduction of particles to clay size, by chemical transformation or precipitation, or by introduction as wind-transported dust. Once clays are present, their translocation in a soil profile involves mobilization, removal and transport, and deposition in a lower part of the profile.

Clay particles must be detached from the greater soil mass to enter into aqueous suspension. *Slaking* is physical detachment, by swelling during wetting, dislodgment by the shear of water flow, raindrop impact, or wind shear. *Dispersion* is chemical detachment that can result from the introduction of waters with low electrolyte concentration, dominance of Na compared to divalent cations on the exchange complex, or high pH. Clay minerals differ in their mobility: of the phyllosilicates, smectites are most easily dispersed, and kaolinite and illite, less so.^[16] Once detached, particles are then free to move in percolating water.

Pore space in most soils consists of micropores and macropores. Macropores may be animal burrows, channels left by decayed roots, large interstitial pores between coarse fragments, or spaces between peds. Under saturated conditions or during infiltration of free water from the surface, percolating water carrying detached clays

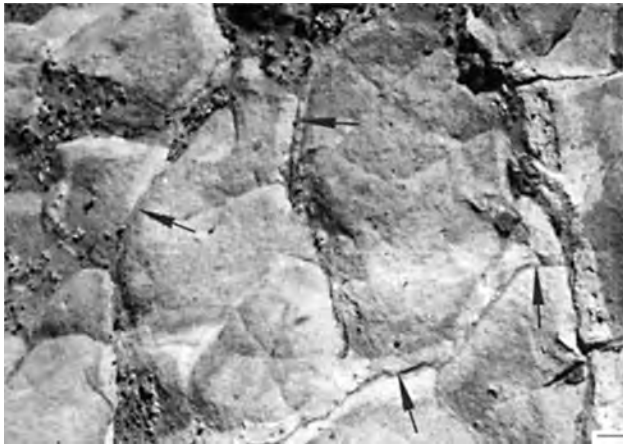


Fig. 2 Argillans (arrows) filling cracks between coarse prisms. This view is looking down on exposed tops of the prisms.

flows preferentially and, with greater velocity, through macropores than through micropores of the soil matrix. Water in the large pores is drawn into drier and microporous fabric by matric suction. The micropores serve a filtering function, and clay particles coat the walls of the large pores and the faces of peds as clay coatings (known as argillans, clay films, clay skins, or clay linings; Figs. 2 and 3). Phyllosilicates assume a face-to-face configuration, with the c-crystallographic axis approximately perpendicular to the ped face or pore wall. The orientation of phyllosilicate particles gives the argillan a strong optical orientation. To the unaided eye, and at low magnification, the argillan appears shiny or waxy. At high magnification of petrographic microscopy, the argillan displays undulatory extinction as the microscope stage is rotated under crossed polarizers.

Pore walls that do not allow appreciable water absorption, such as between rock fragments and sand grains, also

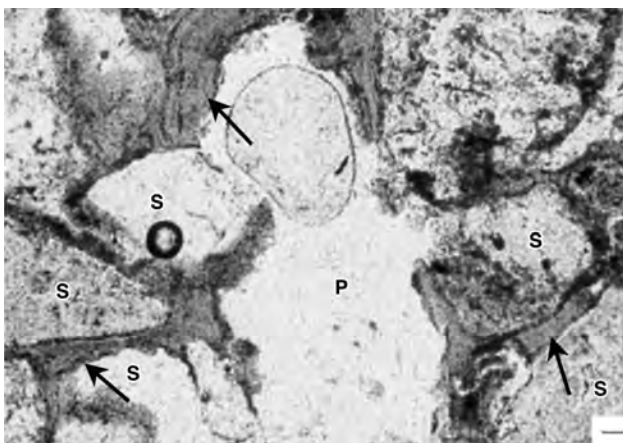


Fig. 3 Microscopic view under plane polarized light of loamy fine sand, with argillans (arrows) filling interstitial pores between sand grains (S) and lining a large pore (P).

develop clay coatings by illuviation. Drying of clay suspensions following downward flow of suspensions draws in the water interface progressively closer to the grain surfaces and to contact points between grains. Clay particle deposition follows the pattern of drying, argillans are thicker at the contact points than away from them, and phyllosilicate particles orient with faces parallel and crystallographic axis perpendicular to the surface.^[17] Thus, clay coatings on grains, and as bridges between grains, have a strong optical orientation.

The depth to which the suspension percolates determines the depth at which the deposition of clay particles occurs. The wetting front, driven by a potential gradient, stops where and when gravitational, pressure, and matric forces acting on the water reach equilibrium. The retarded water evaporates or is sorbed by roots, and suspended particles are deposited at the depth of water percolation.^[18,19]

Soil scientists accept the strongly oriented argillan as evidence of clay illuviation. However, phyllosilicate clays, which are deposited by flocculation (upon encountering high electrolyte concentrations or in the presence of di- and trivalent cations), tend to assume edge-to-edge or edge-to-face configuration and may not develop strong optical orientation. Thus, their illuvial origin may be difficult to identify. Finally, after deposition, argillans may be modified or destroyed by shrink-swell or mixing of soil material by animal burrowing.

CONCLUSION

Because the processes of leaching and precipitation, podzolization, and illuviation are highly dependent on water movement into and through the soil, any activity or land use that interrupts water infiltration and percolation will disturb those processes. Physical disturbances of the soil, such as tilling, may mix soil materials in such a way that soil horizons will be disturbed, mixed, or effaced.

REFERENCES

1. Buol, S.W.; Hole, F.D.; McCracken, R.J.; Southard, R.J. *Soil Genesis and Classification*, 4th Ed.; Iowa State University Press: Ames, 1997.
2. Tan, K.H. *Environmental Soil Science*; Marcel Dekker: New York, 1994.
3. Sposito, G. *The Chemistry of Soils*; Oxford University Press: New York, 1989.
4. Goldich, S.S. A study in rock weathering. *J. Geol.* **1938**, *46*, 17–58.
5. Chadwick, O.A.; Nettleton, W.D. Quantitative relationships between net volume change and fabric properties during soil evolution. In *Soil Micromorphology: Studies in Management and Genesis*; Ringrose-Voase, A.J., Humphreys, G.S., Eds.;

- Developments in Soil Science: Elsevier, 1994; Vol. 22, 353–359.
6. DeConinck, F. Major mechanisms in formation of spodic horizons. *Geoderma* **1980**, 24, 101–128.
 7. Oades, J.M. An introduction to organic matter in mineral soils. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; Soil Science Society of America: Madison, 1989; 89–159.
 8. Farmer, V.C.; Russell, J.D.; Berrow, M.L. Imogolite and proto-imogolite allophane in spodic horizons: Evidence for a mobile aluminum silicate complex in podzol formation. *J. Soil Sci.* **1980**, 31, 673–684.
 9. Farmer, V.C. Significance of the presence of allophane and imogolite in podzol Bs horizons for podzolization mechanisms: A review. *Soil Sci. Plant Nutr.* **1982**, 28, 571–578.
 10. Anderson, H.A.; Berrow, M.L.; Farmer, V.C.; Hepburn, A.; Russell, J.D.; Walker, A.D. A reassessment of podzol formation processes. *J. Soil Sci.* **1982**, 33, 125–136.
 11. Jakobsen, B.H. Multiple processes in the formation of subarctic podzols in Greenland. *Soil Sci.* **1991**, 152, 414–426.
 12. Barrett, L.R.; Schaetzel, R.J. An examination of podzolization near Lake Michigan using chronofunctions. *Can. J. Soil Sci.* **1992**, 72, 526–541.
 13. Gustafsson, J.P.; Bhattacharya, P.; Bain, D.C.; Fraser, A.R.; McHardy, W.J. Podzolisation mechanisms and the synthesis of imogolite in Northern Scandinavia. *Geoderma* **1995**, 66, 167–184.
 14. Certini, G.; Ugolini, F.C.; Corti, G.; Agnelli, A. Early stages of podzolization under corsican pine (*Pinus nigra* Arn. ssp *laricio*). *Geoderma* **1998**, 83, 103.
 15. Nettleton, W.D.; Brasher, B.R.; Baumer, O.W.; Darmody, R.G. Silt flow in soils. In *Soil Micromorphology: Studies in Management and Genesis*; Ringrose-Voase, A.J., Humphreys, G.S., Eds.; Developments in Soil Science: Elsevier, 1994; Vol. 22, 361–371.
 16. Stern, R.; Ben-Hur, M.; Shainberg, I. Clay mineralogy effect on rain infiltration, seal formation, and soil losses. *Soil Sci.* **1991**, 152, 455–462.
 17. Sullivan, L.A. Clay coating formation on impermeable materials: Deposition by suspension retention. In *Soil Micromorphology: Studies in Management and Genesis*; Ringrose-Voase, A.J., Humphreys, G.S., Eds.; Elsevier: Amsterdam, 1994; Vol. 22, 373–380.
 18. Dijkerman, J.M.; Cline, M.G.; Olson, G.W. Properties and genesis of textural subsoil lamellae. *Soil Sci.* **1967**, 104, 7–16.
 19. Moody, L.E.; Graham, R.C. Pedogenic processes in thick sand deposits on a marine terrace, Central California. In *Whole Regolith Pedology*; Cremeens, D.L., Brown, R.B., Huddleston, J. Herbert, Eds.; Special Pub.; Soil Science Society of America: Madison, 1994; Vol. 34, 41–55.

Liming and Lime Materials

Mark K. Conyers

New South Wales Department of Primary Industries, Wagga Wagga Agricultural Institute, Wagga Wagga, New South Wales, Australia

Abstract

“Liming” is the application of an alkaline compound of calcium (Ca) to soil. The main aim is to neutralize excessive acidity in the soil by reaction with the alkaline anion, and the secondary aim is to supply Ca. Liming is a major tool, along with tolerant plants, in the management of acidic soils.

ACIDITY

The quantity of acidity that is regarded as “excessive” varies with the soil and with the species and cultivar of plant. Direct hydrogen ion (H^+) toxicity is uncommon in mineral soils but can occur in peaty soils of low pH. A high concentration of H^+ also has an indirect effect on legume growth by inhibiting the symbiotic relationship between the host plant and rhizobia. In most mineral soils, low pH is associated with aluminum (Al) toxicity to plants. Plants vary in their tolerance to Al (e.g., medics are generally sensitive; tropical grasses are generally tolerant). In some soils, manganese (Mn) toxicity is also a limitation to plant growth. Plant tolerance to Al and to excessive Mn is independently inherited. Plant breeding for tolerance to Al and Mn toxicity is used as either an alternative to using limestone, where the cost of liming is uneconomic, or in conjunction with liming, where subsurface acidity or seasonal conditions limit the effectiveness of liming.^[1] The application of liming materials does not always overcome Mn toxicity, as it can occur at higher pH than Al toxicity and may be induced by hot, dry soil conditions or by waterlogging. On the other hand, too much alkali added to a soil can induce deficiencies of Mn and also of other trace elements such as iron, zinc, and boron. In view of the potential for imbalanced plant nutrition, what constitutes “excessive” soil acidity is best determined by trials on local soils with local species and cultivars.

Deficiencies of calcium (Ca) and magnesium (Mg) also occur in some acidic soils. Ca deficiency is more commonly found in tropical species such as the peanut. Ca is also of benefit to soil structure; it is mainly applied to soils with large proportions of sodium on the cation exchange capacity and also to non-cracking clay soils with a large proportion of Mg on exchange sites. Where soil structure is a concern, gypsum is normally the preferred source of Ca, sometimes in combination with fine limestone. Gypsum is not an alkaline salt and hence does not add alkalinity to a soil.

While most liming materials can overcome Al toxicity and Ca deficiency, the alleviation of Mg deficiency requires the application of a liming material with a significant component of Mg. This may be either dolomite or a Mg-rich limestone, commonly known as dolomitic limestone. There is in fact a continuum between pure limestone and pure dolomite. Magnesite may be used to provide Mg but is rarely used as a liming material per se.

LIMING MATERIALS

The application rates of liming materials are best expressed relative to a standard such as calcium carbonate ($CaCO_3$). To overcome an existing problem of excessive acidity as opposed to a prophylactic dressing, application rates range from as little as 1 tonne $CaCO_3$ per hectare on weakly buffered soils, growing Al- or Mn-tolerant plants, to 10 tonne $CaCO_3$ per hectare on strongly buffered soils supporting sensitive plants. Weakly buffered soils include sandy textured soils and tropical Oxisols, particularly with low concentrations of organic carbon (OC). Strongly buffered soils include acidified Mollisols and soils with high contents of clay or OC. In high lime requirement situations, the $CaCO_3$, or its chemical equivalent, need not be applied in a single dose. Higher application rates than outlined here may be required for the amelioration of acid sulfate soils; however, this requires a separate consideration from the present focus.

Some common lime materials are listed in [Table 1](#). Most agricultural lime is some form of $CaCO_3$. This is generally in the mineral form of calcite, which occurs in limestone, marble, marl, and aged coralline deposits. Crushed shells and formed coralline material may be in the aragonite form of $CaCO_3$. The aragonite form is more soluble than the calcite form. Dolomite has a harder crystalline form than either of the $CaCO_3$ forms and is generally less reactive in soils.^[2] It has to be crushed finer than limestone to react in a similar time frame, but if Mg is required the only alternative

Table 1 NV and Ca content of pure liming materials.

Substance	Formula	% NV ^a	% Ca ^a
Limestone	CaCO ₃	100	40
Dolomite	(Ca _{0.5} ·Mg _{0.5})CO ₃	109	22
Burnt lime	CaO	178	71
Hydrated or slaked lime	Ca(OH) ₂	135	54
Ca silicate	CaH ₂ SiO ₄	75	30
Magnesite	MgCO ₃	119	0

^aRounded to nearest whole percent.

to dolomite is to spread both crushed limestone and a separate source of Mg such as magnesite.

Industrial waste materials are also commonly used for liming. Precipitator dusts from clinker and cement manufacturing contain varying proportions of CaCO₃ and calcium oxide (CaO). The latter has an exothermic and caustic reaction with water and so should be handled with care.^[3] The risks increase as the proportion of oxide in the waste increases. Ca silicates from steel manufacturing can also be good liming materials.

Liming materials can be broadly grouped as follows:

Natural deposits—dolomite; calcite (limestone, coralline and earthy limestones, and marble); aragonite (fresh shell and coralline materials). Further refinement is available.^[4]

Manufactured forms—burnt lime; slaked or hydrated lime.

By-products—precipitator dusts (mixtures of carbonates, oxides, etc.); slags (usually Ca silicate wastes from steel production).

Incidental forms—where liming is not the principal function of the material: reactive rock phosphate, dicalcic phosphate, sewage ash, manures, and composts.

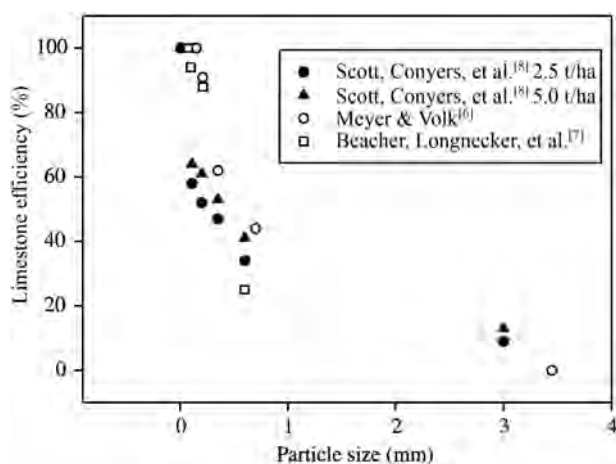
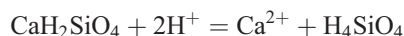
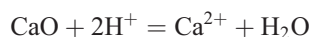


Fig. 1 The increase in limestone efficiency in raising soil pH with decrease in particle diameter.

Source: From Meyer & Volk,^[6] Beacher, Longnecker, et al.,^[7] and Scott, Conyers, et al.^[8]

The reactions, which neutralize acidity, other than for incidental liming materials, can be exemplified as follows:



In each case, 2 mol of acid are consumed per mole of liming compound. While the second reaction liberates the greenhouse gas carbon dioxide, as does the increase in OC mineralization that follows liming, in the longer term, the removal of the constraint of soil acidity increases plant root and top growth, and so net C fixation can be enhanced.

Purity and Fineness

The effectiveness of traditional liming materials such as limestone and dolomite depends on their purity and fineness. Clearly, the more pure the limestone or dolomite, the greater the quantity of acid that can be neutralized by each tonne of product. In more general terms, the ability of any type of liming material to neutralize acidity is termed neutralizing value (NV). It is a measure of the ability of a given mass of liming material to react with acidity, expressed relative to a standard such as CaO^[5] or CaCO₃. The latter is adopted here. For pure minerals and compounds, NV is easily calculated by the following calculation:

$$\text{NV} = \frac{\text{mole wt of CaCO}_3}{\text{mole wt of compound}} \quad (2)$$

For example, for CaO,

$$\text{NV} = 100/56 = 1.78 \quad (3)$$

Results for pure compounds are given in Table 1. For compounds of mixed or unknown composition, the NV is obtained by titration after the dissolution of liming material in excess acid. Details of the method can be obtained by reference to the latest edition of the Official Methods of Analysis of the Association of Official Analytical Chemists.

The fineness of a liming material is a more controversial issue, not because of differences in experimental results but because of differences in opinion about the cost of production of fine lime and the ease of spreading. Fig. 1 shows the results of three studies on the ability of calcitic limestone to raise soil pH as a function of particle size.^[6–8] The degree of agreement between these pot and field studies, conducted by different workers on different continents 50 years apart, is remarkable. In none of these studies was there an upper limit where fineness was of no further benefit in lifting pH. The concept of a “plateau” in particle-size effectiveness, referred to in some texts, is a subjective assessment of, firstly, the practical limitations of spreading fine limestone and, secondly, the cost of the liming material relative to its agricultural value.

Despite the clear demonstration of the value of fineness, there are two reasons why it is not necessarily helpful to define an “ideal” particle-size distribution for all calcitic liming materials. First, the agricultural context has to be considered. On the one hand, the need for finer limestone is most apparent for crops grown in regions of relatively low rainfall. On the other hand, for sandy soils receiving 2000 mm rainfall annually, it may be unnecessary or wasteful to have a high proportion of fines as there may be losses of the limestone from surface soil through mass flow. On the one hand, where cartage and spreading costs are a large proportion of the total cost of liming, it is more cost effective to cart and spread a product that is almost 100% effective. On the other hand, if the farm is near to a limestone quarry and if the farmer has his own lime spreader, then he may be able to regularly spread higher tonnages of relatively coarse limestone quite cheaply. On the one hand, a high ratio of lime cost to agricultural product value also means that limestone has to be used as cost effectively as possible. On the other hand, in an intensive horticulture, the cost of lime may be relatively minor and a higher application rate of coarser limestone may be as cost effective. Therefore, the agricultural context of the liming has to be considered.

Second, the lime crusher’s industrial perspective needs to be considered. If the limestone crushing plant services industries such as glass, plastics, and collieries, there is more chance that the plant will have a ball mill, enabling the manufacture of fine limestone. If the main focus of the plant is to produce agricultural limestone, it may not be reasonable to expect the milling of very fine products. The cost of producing a fine limestone simply for soil amelioration may diminish the cost competitiveness of the crushing plant. On the other hand, the milling of fine limestone may give a lime producer a strong advantage over the competition. Local circumstances need to be considered. The same principle holds for by-product liming materials; the introduction of a processing step such as milling may in some instances make the by-product uneconomic. However, without milling, a coarse slag may be nearly useless. Therefore, the ways in which local by-products are best used should be determined by field trials and economic analysis.

The purpose of quantitative models of efficiency for liming materials should not be to dictate terms to crushers and other producers but to enable consumers to assess the relative cost effectiveness of alternative products available to them. The concept of an “ideal” liming material is likely to vary with local circumstances.

CONCLUSION

Despite claims in some popular literature, limestone has no “cure all” properties; it is a part of the solution for the specific problem of excessive soil acidity. The best management practice (type of lime, fineness, rate of application, incorporation method, etc.) needs to be determined for each mix of agricultural and industrial circumstances.

REFERENCES

1. Cregan, P.D.; Hirth, J.R.; Conyers, M.K. Amelioration of soil acidity by liming and other amendments. In *Soil Acidity and Plant Growth*; Robson, A.D., Ed.; Academic Press: Marrickville, 1989; 205–264.
2. Barber, S.A. Liming materials and practices. In *Soil Acidity and Liming*; Adams, F., Ed.; 2nd Ed.; Monograph 12, Agronomy; ASA, CSSA, SSSA: Madison, 1984; 171–209.
3. Boynton, R.S. *Chemistry and Technology of Lime and Limestone*, 2nd Ed.; John Wiley and Sons: New York, 1980.
4. Oates, J.A.H. *Lime and Limestone*; Wiley-VCH: Weinheim, 1998; 14–15.
5. Cooke, G.W. Calcium fertilizers and liming materials. In *Fertilizing for Maximum Yield*, 3rd Ed.; Granada: London, 1982; 161–173.
6. Meyer, T.A.; Volk, G.W. Effect of particle size of limestone on soil reaction, exchangeable cations and plant growth. *Soil Sci.* **1952**, 73, 37–52.
7. Beacher, R.L.; Longnecker, D.; Merkle, F.G. Influence of form, fineness and amount of limestone on plant development and certain soil characteristics. *Soil Sci.* **1952**, 73, 75–82.
8. Scott, B.J.; Conyers, M.K.; Fisher, R.; Lill, W. Particle size determines the efficiency of calcitic limestone in amending acidic soil. *Aust. J. Agr. Res.* **1992**, 43, 1175–1185.

Loess

Fred E. Rhoton

National Sedimentation Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Oxford, Mississippi, U.S.A.

Helaine W. Markewich

U.S. Geological Survey (USGS), Atlanta, Georgia, U.S.A.

Abstract

Loess is a wind-blown silt that originates from outwash alluvium downstream of receding glaciers and from the surface of desert soils, covering approximately 10% of the earth's surface. Both the thickness and particle size of Loess decrease precipitously within a short distance from the source area. In addition to being highly fertile and productive, Loess and Loess-derived soils have an inherently low resistance to erosion by water and wind. Consequently, in the presence of adequate rainfall and/or windstorms, Loess becomes the most erosion-prone material on earth. Environmental problems resulting from Loess erosion include loss in productivity, air and water quality degradation, and coincident medical ailments resulting from particulate matter and chemicals/trace elements associated with the eroded material.

INTRODUCTION

For almost two centuries, the study of Loess and Loess-like deposits has provided controversy and consternation. In general, Loess is considered largely unstratified silt-sized dust/material that mantles the landscape in blanket-like deposits. Loess deposits are widely distributed, with the most extensive deposits occurring in the Central United States, eastern China, Northern Europe, and Argentina. Loess deposits are also recognized in Alaska, Siberia, and Canada.

Environmentally and culturally, Loess deposits are considered as both assets and detriments. Some of the most fertile, highly productive agricultural soils in the world are developed on Loess,^[1] which is inherently high in plant nutrients, available water-holding capacity, and porosity. Conversely, there are many problems associated with Loess due to its unfavorable physiochemical properties. Most unweathered Loess is devoid of cementing agents such as clay, iron (Fe) oxides, and organic matter. This lack of cementing agents results in unbound silt-sized particles that do not form erosion-resistant structural units, which in turn results in Loess and Loess-derived soils having very low resistance to water and/or wind erosion. Loess erosion results in significant local-to-regional degradation of soil, water, and air quality.

ORIGIN OF LOESS

Four major events have been suggested^[2] during the formation of a Loess deposit: formation of the silt-sized

material (20–60 μm), transportation, deposition, and post-depositional changes. One of the original arguments as to the origin of Loess deposits was whether they were alluvial or eolian.^[3] Since the early 1920s, Loess has been traditionally defined as wind-deposited silt, which in many cases is difficult-to-impossible to differentiate from Loess-like colluvium and/or alluvium. Making this distinction addresses the problems of how Loess material is made and how it is transported.

Much of the world's Loess material is “made” by glacial grinding, resulting in finely ground rock powder (flour) being deposited by glacial meltwater. Since the middle of the 20th century, a number of researchers showed that weathering processes in very cold alpine and desert environments can also produce significant amounts of Loess material.^[4–6] Small amounts of Loess material also form in warmer deserts, but the resulting Loess deposits generally are of local extent.^[7]

In a dry state, the silt-sized material associated with each process eventually becomes airborne and is transported by strong, generally prevailing, winds at distances determined by particle size.^[8] Wind action during the transport phase results in physical sorting and a deposit characterized by homogeneous particle sizes, primarily in the coarse-to-medium silt size range (20–50 μm , U.S. Department of Agriculture scale).

Origins of Loess deposits are explicitly linked to eolian processes that transport sediment from bare surfaces of glacial till, outwash alluvium, and desert soils. These represent the primary sources of fine sediment available for wind transport.^[9] Finely divided rock



Fig. 1 Typical topography of Loess deposits in the Lower Mississippi River Valley near Vicksburg, Mississippi.

powder (or flour) produced from glacial scour eventually is transported, sorted, and deposited by meltwater streams. The meltwater deposits are entrained by winds, sorted again, and redeposited along the edge of continental glaciers and adjacent to major meltwater river valleys. As a result of multiple sortings, Loess of glacial origins consists of finer particle sizes compared to Loess derived from desert origins, which may have undergone only one sorting event.^[10]

Silt-sized particles are most likely to be transported by winds blowing across source areas. The silt transport process is initiated by the saltation of sand grains moving across surfaces that dislodge and suspend silt grains to altitudes as high as 3 km and to downwind distances that exceed several hundred kilometers.^[9] A decrease in wind velocity initiates Loess deposition in amounts and particle sizes that are greatest to the source areas. As the downwind distance increases, the thickness of Loess deposits and particle size decrease proportionately. Once deposited, Loess appears as a

blanket draped over the landscape (Fig. 1), with the lower contact conforming to the Preloess topography.

LOESS DISTRIBUTION

Loess deposits are widely distributed on a global basis, covering approximately 10% of the total landmass (Fig. 2) primarily between latitudes 30° N and 50° N, the zone that generally corresponds to the southern extremities of glaciation in the northern hemisphere.^[10] The most extensive areas of Loess occur in Europe ($1.8 \times 10^6 \text{ km}^2$), the United States ($1.6 \times 10^6 \text{ km}^2$), Argentina ($7.2 \times 10^5 \text{ km}^2$), and China ($6.2 \times 10^5 \text{ km}^2$).^[9,11,12] The thickest Loess deposits are found in China, averaging about 150 m.^[13] By contrast, Loess thickness in the Central United States is 8–15 m, with maximum thickness of 20–35 m being recorded along eastern edges of major river valleys.^[9] Average Loess thickness adjacent to major rivers in Western and Eastern Europe is 5–10 m, with maxima

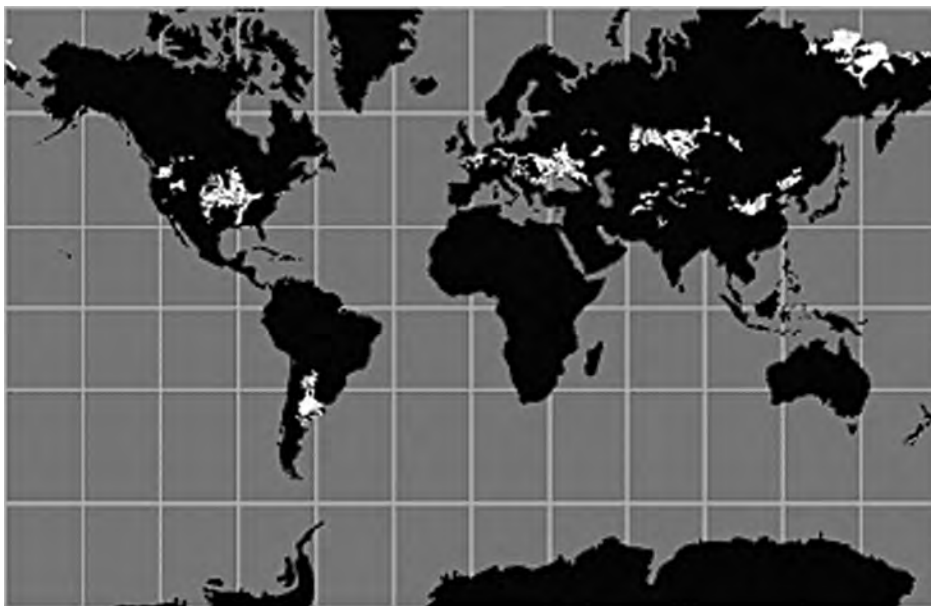


Fig. 2 Global distribution of major Loess deposits.

Source: <http://www.geog.ouc.bc.ca/physgeog/contents/11r.html>.

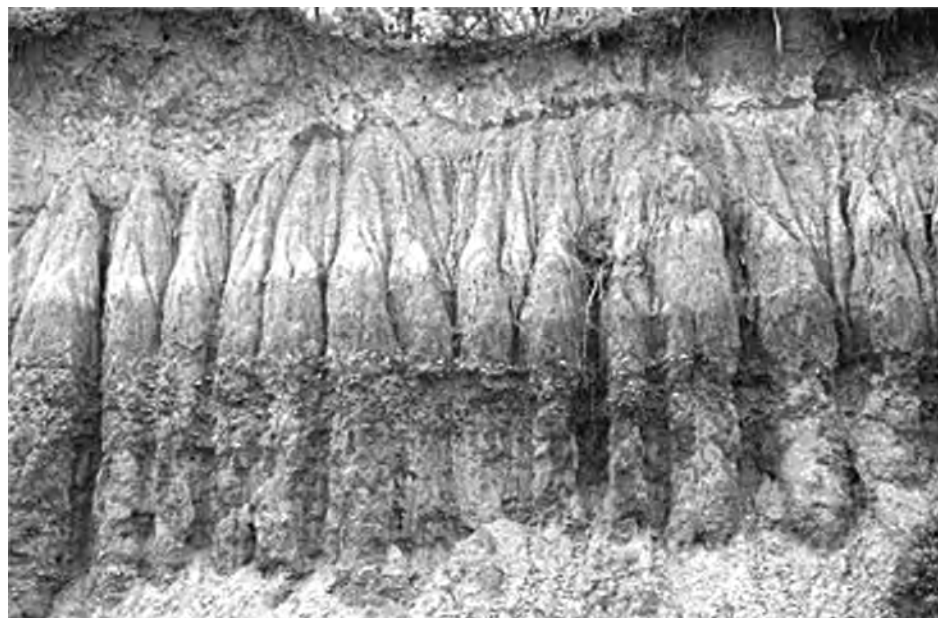


Fig. 3 Weathered Peoria Loess overlying a Paleosol in Carroll Co., Mississippi.

between 30 and 50 m.^[9] Argentinean Loess thickness ranges from 10 to 30 m over vast expanses and locally may exceed 100 m.^[11] Loess deposits in Alaska^[9] and Siberia^[14] range from < 10 to 60 m.

LOESS CHARACTERISTICS

Loess deposits appear to have a unique suite of characteristics regardless of location. These similarities can be attributed to the eolian processes responsible for Loess formation. Deposits are massive except for clay lenses and zones of carbonate and/or organic matter accumulations resulting from weathering and pedogenic activity. Where present, Paleosols (Fig. 3) mark periods of minimal or non-deposition. In a relatively unweathered state, Loess colors are predominately pale yellow to brown.^[15] As weathering reactions progress, relative Fe concentrations increase, and Loess colors will tend toward more brown or red hues.

Loess deposits, by virtue of their wind-blown origin, commonly possess an open structure with a pore space in the range of 50–60%. This high porosity value likely contributes to the instability of Loess in a wet state. Although remarkably stable in vertical cliffs (Fig. 4) when dry, upon wetting, Loess has a tendency to collapse. This is in large part due to the high percentage of pore space and the lack of cementing agents such as clay, Fe oxides, organic matter, or carbonates.

Characterization data for Loess deposits in the mid-continental United States, Central and Eastern Europe, China, and Argentina are very comparable relative to changes in particle size as a function of distance from source.^[16] The major factor then influencing the particle size composition of Loess deposits in different locales is proximity to the source. As the downwind distance of the source increases, both Loess thickness and particle size decrease dramatically in the first few kilometers and continue to decrease logarithmically for short



Fig. 4 Vertical slope in a Loess deposit near Vicksburg, Mississippi.

distances before reaching a minimum that remains constant regardless of distance from source.^[8]

Although Loess deposits may show uniformity of particle size at distance from source areas on a global basis, their chemical and mineralogical compositions are often dissimilar. In relatively unweathered Loess, this variability can be attributed to differences in the composition of parent materials in glaciated and non-glaciated source areas and the composition of strata incised by streams transporting the outwash; e.g., Loess deposits in the lower Mississippi River Valley were ultimately derived from granitic rocks in the Canadian Shield, whereas Argentinean Loess was derived from volcanic rocks (pyroclastic).^[12] The chemical composition of the two Loesses is similar to the extent that both are dominated by silica and aluminum, but the calcium and magnesium concentrations of the U.S. Loess are considerably greater and Fe concentrations are lower than those of the Loess from Argentina.^[17]

CULTURAL AND ENVIRONMENTAL ASPECTS OF LOESS

The importance of Loess to civilization for several millennia has been documented worldwide. Soils developed on the Loess Plateau, China, have been cultivated for 6000 years and are credited with allowing for the establishment of early Chinese culture.^[18] Loess soils had similar impacts in India and Mesopotamia. The early agricultural settlements in Europe were established in areas with Loess-derived soils.^[19] In the American Midwest, crop yields from loess-derived soils that dominate the region have been unrivaled for centuries.

In addition to the favorable aspects of Loess, there are several problems endemic to these regions. Soil/Loess loss to erosion by water can reach astronomical levels. Loess and Loess-derived soils have the distinction of being more susceptible to erosion than any other soil material. On a global basis, Loess regions sustain the greatest soil erosion losses. The northwest and midcontinent regions of the United States and the Pampas region of Argentina are excellent examples of Loess-blanketed areas with high erosion rates. The Loess Plateau of China undoubtedly has the most serious erosion problem in the world. The region is responsible for 90% of the estimated 1.6 billion tons of Loess received by the Yellow River on an annual basis.^[18,20] By comparison, the Mississippi River at Vicksburg, Mississippi, U.S.A., transports an annual average of 225 million tons of sediment, even though this region receives two to three times more rainfall than the Loess Plateau of China.

The effects of erosion are twofold: loss of productivity and degradation of the environment. In areas of extreme erosion, such as the Loess Plateau region, environmental damage related to sedimentation poses a greater problem.

The Yellow River carries between 10% and 40% of its weight as silt-derived sediment, or approximately 10 times the content of other rivers of the world.^[21] This load is equivalent to 10 million cubic meters of sediment annually. Much of this sediment is caught in reservoirs, decreasing their storage capacity by about 10% per year.^[22] The consequences are a reduction in hydroelectric output, water available for irrigation, flood control potential, and navigable waterways.

In summary, Loess is a wind-blown silt that originates from outwash alluvium downstream of receding glaciers and from the surface of desert soils, covering approximately 10% of the earth's surface. The most extensive deposits, in terms of area, exist in Europe, the United States, Argentina, and China, in this order. The thickest deposits are found on the leeward side of major stream valleys. Both the thickness and particle size of Loess decrease precipitously within a short distance from the source area. In addition to being highly fertile and productive, Loess and Loess-derived soils have an inherently low resistance to erosion by water and wind. Consequently, in the presence of adequate rainfall and/or windstorms, Loess becomes the most erosion-prone material on earth. Environmental problems resulting from Loess erosion include loss in productivity, air and water quality degradation, and coincident medical ailments resulting from particulate matter and chemicals/trace elements associated with the eroded material.

REFERENCES

1. Chesworth, W. Late Cenozoic geology and the second oldest profession. *Giosci. Can.* **1982**, *9*, 54–61.
2. Smalley, I.J.; Smalley, V. Loess material and loess deposits: Formation, distribution and consequences. In *Eolian Sediments and Processes*; Brookfield, M.E., Ahlbrandt, T.S., Eds.; Developments in Sedimentology; Elsevier: New York, 1983; Vol. 38, 51–68.
3. Rutledge, E.M.; Guccione, M.J.; Markewich, H.W.; Wysocki, D.A.; Ward, L.B. Loess stratigraphy of the lower Mississippi valley. *Eng. Geol.* **1996**, *45*, 167–183.
4. Zeuner, F.E. Frost soils of Mount Kenya, and the relation of frost soils to aeolian deposits. *J. Soil Sci.* **1949**, *1*, 20–30.
5. Brockie, W.J. Experimental frost shattering. In *Proceedings of the 7th New Zealand Geography Conference*; Conf. Ser. 7; New Zealand Geographical Society: Hamilton, 1972; 177–186.
6. Lautridou, J.P.; Ozouf, J.C. Experimental frost shattering: 15 years of research at the Centre de Geomorphologie du CNRS. *Prog. Phys. Geog.* **1982**, *6*, 215–232.
7. Yaalon, D.H.; Dan, J. Accumulation and distribution of loess-derived deposits in the semi-desert and desert fringe area of Israel. *Z. Geomorphol. Suppl.* **1980**, *20*, 91–105.
8. Waggoner, P.E.; Bingham, C. Depth of loess and distance from source. *Soil Sci.* **1961**, *96b*, 366–401.

9. Flint, R.F. *Glacial and Quaternary Geology*; John Wiley and Sons, Inc.: New York, 1971; 892.
10. <http://www.msu.edu/user/larsong/g/g412/lecture-12.pdf> (accessed February 2002).
11. Teruggi, M.E. The nature and origin of argentine loess. In *Loess: Lithology and Genesis*; Smalley, I.J., Ed.; Dowden, Hutchinson, and Ross, Inc.: Stroudsburg, 1975; 195–205.
12. Wang, Y.; Zhang, Z. *Loess in China*; Shaanxi People's Art Publishing House, 1980; 170 pp.
13. Yoong, H. Loess cave dwellings in Shaanxi Province, China. *Geojournal* **1990**, *21*, 95–102.
14. Chlachula, J.; Kemp, R.A. Late Pleistocene climatic variations in Siberia based on loess-palaeosol records. *Geolines* **2000**, *11*, 83–85.
15. Snowden, J.O., Jr.; Priddy, R.R. Loess investigations in Mississippi. *Miss. Geol. Surv. Bull.* **1968**, *111*, 1–203.
16. Krinitzsky, E.L.; Turnbull, W.J. *Loess Deposits of Mississippi*. GSA Special Paper 94; Geological Society of America Inc.: Boulder, 1967; 1–64.
17. Pye, K.; Johnson, R. Stratigraphy, geochemistry, and thermoluminescence ages of lower Mississippi valley loess. *Earth Surf. Proc. Land.* **1988**, *13*, 103–124.
18. Kleine, D. Who will feed China? *J. Soil Water Conserv.* **1997**, *52*, 398.
19. <http://www.geocities.com/Athens/Agora/7768/nature3.html> (accessed February 2002).
20. Li, J. Comment: Population effects of deforestation and soil erosion in China. *Popul. Dev. Rev.* **1990**, *16*, 254–258.
21. <http://www.geocities.com/Tokyo/Island/7602/history.htm> (accessed February 2002).
22. <http://www.library.utoronto.ca/pes/state/chinaco/land.htm> (accessed February 2002).

Loess Plateau

Fenli Zheng

Ximeng Xu

Institute of Soil and Water Conservation, Northwest A&F University, Yangling, China

Xiubin He

Institute of Mountain Hazards and Environment, Chinese Academy of Sciences, Chengdu, China

Abstract

The Loess Plateau area, the most widely distributed region of the Loess on the earth, is mostly located in the middle reaches of the Yellow River, covering 8 latitudes (33–41°N) and 14 longitudes (100–114°E) with an area of 624,000 km². Current landform of the Loess Plateau formed due to neotectonism, rising–falling movement, fault activities on a large scale, and Loess deposition process. The Loess Plateau area located at a transitional zone from monsoon climate of the eastern China to semiarid and arid climate in the northwestern China, the Loess Plateau experiences a prominent continental monsoon climate. The Loess Plateau can be bioclimatically divided into forest, forest steppe, and steppe zones from the southeast to the northwest, while soil is both determined by climate condition and human activities. As a consequence of special landform and human activities, the Loess Plateau is one of the most severely eroded areas in the world. However, a set of soil and water conservation projects has been implemented on the Loess Plateau and obtains great ecological and economic benefits.

INTRODUCTION

The Loess Plateau area is located in the upper and middle reaches of the Yellow River (the Huanghe River). It is bordered by the Yinsan Mountain in the north, extending southward to the Qinling Mountain, and by the Riyue Mountain in the west, stretching eastward to the Taihang Mountain (Fig. 1). The Loess Plateau area covers 8 latitudes (33–41°N) and 14 longitudes (100–114°E) with an area of 624,000 km². The Loess Plateau area and its dusty soil cover most areas of Shanxi, Shaanxi, and Gansu provinces and cover parts of Qinghai and Henan provinces, Ningxia, and Inner Mongolia Autonomous Regions. The Loess deposits with depth of 50 to 200 m thick. Loess is the name for the silty soil that has been deposited by wind storms on the plateau over the ages. Loess is a highly erosion prone soil that is susceptible to the erosional forces of wind and water.

The Loess Plateau area is the most widely distributed region of the Loess on the earth, where the thickest Loess–Paleosol sequence has been developed. The Loess Plateau area, with an altitude of 1000–1600 m characterized by various geomorphologic units, includes high plains with deep-cutting gullies (Fig. 2), hilly gully (Fig. 3), rock mountains, wide valley plain, and sand-depositing land (sandy land). Its surface is covered by averagely 100 m thickness of Loess–Paleosol layers. The Loess was geologically transported from the

northwestern Gobi desert by winds and accumulated on the Loess Plateau since the beginning of the quaternary (about 2.5 million years before).^[1] The alternations of Loess and Paleosols, in conjunction with their enclosed faunas and distinctive mineralogies, have been widely interpreted as reflecting quaternary glacial/interglacial episodes.^[2]

As one of the most severe soil erosion regions in the world, over 60% of the land in the Loess Plateau area has been subjected to soil erosion, with an average soil loss of 2000–2500 t km⁻² yr⁻¹.^[3] For example, the amount of soil erosion from some watersheds in the Huangfuchuan basin reached 59,700 t km⁻² yr⁻¹, and maximum sediment concentration in the river water was measured as high as 1640 kg m⁻³.^[4] Soil erosion has seriously depleted land resources and degraded the eco-environment in the Loess Plateau, which directly affects local socio-economic sustainable development. Furthermore, soil loss on the Loess Plateau area is the major source of sediment load in the lower reaches of the Yellow River. The sediment delivery in the Yellow River is increased after it flows through the Loess Plateau area. Mean annual suspended sediment concentration in the Yellow River measured at the Shanxian Observatory, is 37.6 kg m⁻³, and a sediment load of 1.64 billion tons per year (before 1980)^[5] was observed at the Sanmen Gorge, Henan Province (Table 1), which is 9 to 21 times greater than that of the most major rivers in the world. Therefore, soil erosion control is a critical issue in the Yellow River basin.

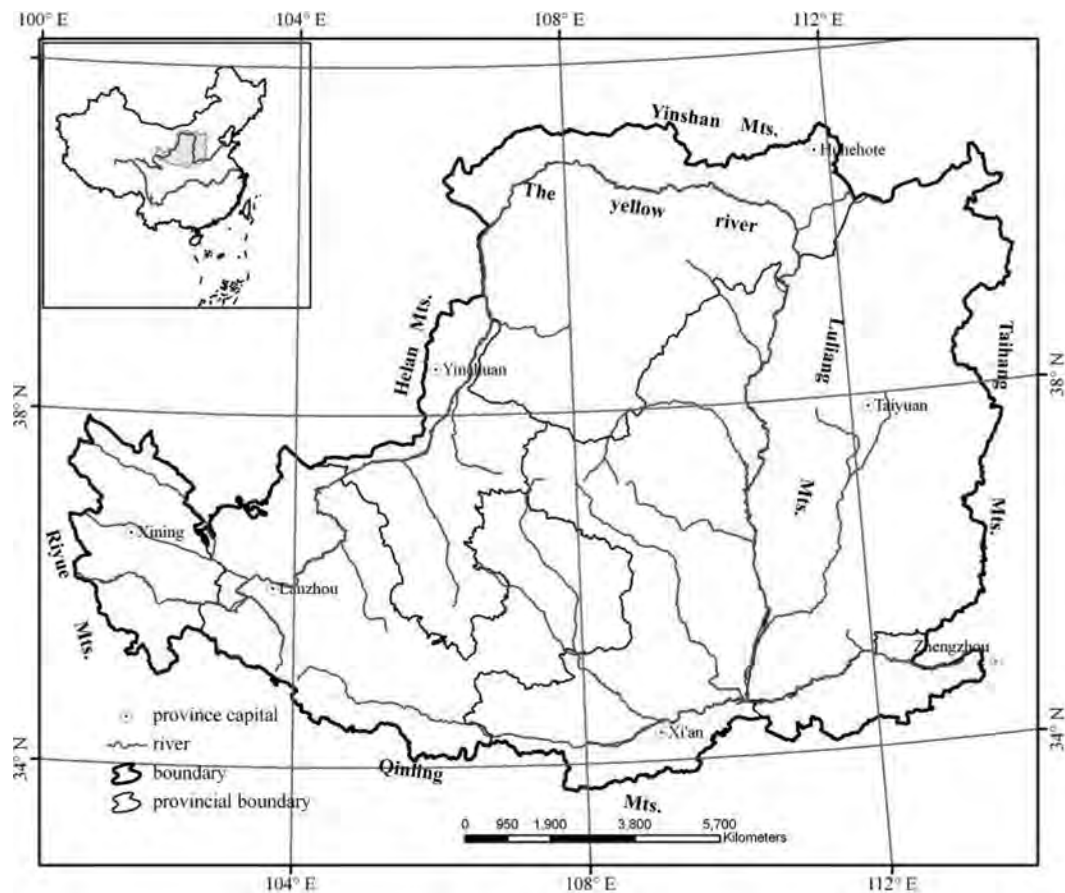


Fig. 1 Location of the Loess Plateau area.

Of the 1.6 billion tons of annual sediment delivery, 0.4 billion tons was deposited in the downstream river channel at a sedimentation rate of about 10 cm yr^{-1} before 1980. Thus, lower reaches of the Yellow River's riverbed

is 4–10 m higher than the adjacent land. Since the 1990s, sediment delivery has greatly decreased and thereafter, sediment delivery of the Yellow River is about 0.3 billion tons per year.



(A) High plain

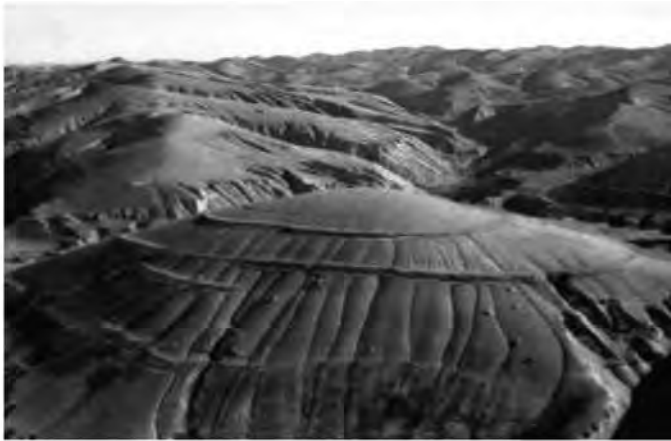


(B) Deep gully along high plain edge

Fig. 2 The high plain (A) with deep-cutting gullies (B) of the Loess Plateau.



(A) Hilly gully landscape



(B) Hillslope landscape

Fig. 3 The hilly gully landscapes of the Loess Plateau (A and B).

FORMATION OF THE LOESS PLATEAU AND LOESS DEPOSITION

The Loess Plateau at the end of the Cretaceous period was in peneplain stage. At the beginning of Eocene epoch, the peneplain started to fold and arch, and was completely broken up at the end of the Oligocene epoch. Thereafter, because of neotectonism, rising–falling movement, and fault activities on a large scale, the basic landforms such as mountains and basins were formed. The landforms before Loess deposition on the Loess Plateau may be roughly grouped into three types:

1. In the east, the Wutai, Lüliang, and Zhongtiao old upland in Shanxi Province, being repeatedly under the process of orogenic movement, denudation, deposition, and the effects of magma explosion in the north had formed the two parallel folded mountain series of the Taihang and Lüliang Mountains and a series of basins. Here, Loess covered piedmonts, basins, and valleys below the altitude of 1500 m. The thickness of Loess deposition is generally less than 30 m, except in the Taiyuan Basin and wide valleys where Loess deposit is thicker.

2. In the middle, the high plain and the rolling flat mound were formed through the successive cutting and denudation of the Ordos platforms. Although the platform has risen gradually and evenly, the gentle folds within the platform were formed under the influence of Yanshan Movement, and the grabens and faults were formed at the edge of the platform due to the influence of Himalayan Orogeny. The high plain formed after the vast areas were covered with thick Loess deposit in the typical Loess Plateau, with an area of about 280,000 km², an altitude of 1000–14,000 m, and Loess layer of about 100 m thickness. The grabens and valley basins are at an altitude of 400–500 m only.
3. In the west, Longxi basin between the Liupan Mountain and the Qilian Mountain has frequently been subjected to the influence of denudation, accumulation rising and falling since the Nanshan movement and has formed a series of long mounds, pointed mountains, low gentle hills, and piedmont plains. The Loess could deposit as high as 2000 m or more above the sea level.

Table 1 Erosion episodes of valley erosion.

Erosion/ deposition cycle	Oxygen isotope of deep sea	Age (×1000 years)	Erosion episodes
Erosion	Interval glacial epoch	10	Present
Deposition	Glacial epoch	29	
Erosion	Interval glacial epoch	61	The third stage
Deposition	Glacial epoch	75	
Erosion	Interval glacial epoch	128	The second stage
Deposition	Glacial epoch	185	
Erosion	Interval glacial epoch	242	The first stage

During deposition of Loess in different periods, the neotectonic rising movement was still going on. Hence, Loess deposition was often accompanied by erosion. Loess in this region has been mostly transported by wind for about 2.5 million years. Natural conditions of this region and the original region of the Loess have gone through many periodical changes over a very long time. Accordingly, Loess deposition also had periodical intermittence, and during the intermittent periods, Paleosol profiles reflecting the initial landscape and biometeorology formed.

In the beginning of quaternary period, the Loess Plateau had Loess deposition on a large scale; Wucheng Loess, Lishi Loess, Malan Loess, and Loess of the Holocene epoch had deposited continuously. In the meantime, erosion process

occurred. Loess–Paleosol profile shows that four erosion periods were experienced during the Loess deposition (Table 1).

CLIMATE CONDITION AND WATER RESOURCE

The Loess Plateau is located at a transitional zone from monsoon climate of the eastern China to semiarid and arid climate in the northwestern China. Continental monsoon climate is very prominent. Annual average temperature is 4.3°C in the northwest and 14.3°C in the southeast. Temperature in January (the coldest month) is from −14°C to −8°C in the northwest and −4°C to 0°C in the southeast, and temperature in July (the warmest month) is 20°C to 24°C in the northwest and 24°C to 28°C in the southeast. Annual precipitation ranges from 200 mm in the northwest to 700 mm in the southeast, and more than 50% of the precipitation occurs in June, July, August, and September. Dominant rainfall pattern is rainstorm, wherein the rainfall from single rainstorm accounts for 15–25% of the annual precipitation while rainfall from a large storm accounts for 40–50% of the annual precipitation.^[6]

Moreover, there is much sunshine in the Loess Plateau area. Annual sunshine time is 2100–3100 hr. The annual solar radiation is about 5–6.3 billion J per m². More than and equal to 0°C accumulated temperature is 3200–3900°C, and the period without frost is 150–200 days.^[6]

Annual average runoff volume from the Yellow River basin is about 56.3 billion m³, and the ground water volume is about 26.3 billion m³.

VEGETATION AND SOIL

The Loess Plateau can be bioclimatically divided into forest, forest steppe, and steppe zones from the southeast to the northwest. Paleo-pollen analysis shows that there had been luxurious forest cover during warm–wet periods in the quaternary. However, vegetation clearance started about 3000 years ago for cultivation, resulting in vegetation cover becoming very sparse. According to investigation on the residual vegetation in field edges, gully slopes, and rock mountain areas, there are more than 500 species of grasses, 250 species of trees, and 3000 species of herbs and bushes. The dominant species are *Pinus*, *Quercus*, *Ulmus*, *Artemisia*, *Bothriochloa ischaemum*, *Stipa bungeana*, *Lespideza dahurica*, and *Cleistogenes chinensis*.^[7]

According to the natural bioclimatic condition, there would be brown soil or cinnamon (Heilutu) within the forest zone in the south, chemozem or chestnuts in forest steppe and steppe zone in the middle, and desert soil or brown calcium soil in desert-steppe zone in the north. However, for several thousands of years, human activities have caused extensive destruction of the natural

vegetation cover and resulted in a profound modification of the original soils. Chinese soil scientists called it agricultural Loess soil and subdivided it into three groups: Lutu soil, Zhong Heilutu soil (heavy black loam), and Qin Heilutu soil (light black loam) in Chinese soil classification system.^[8] They are comparable to Ustalfs or Udalfs in the U.S. Department of Agriculture system or Luvisols in the Food and Agriculture Organization system.^[1]

The Loess is characterized by low soil organic matter content (0.2–1%), weak alkaline (pH value ranges 7.2–7.8) reaction with carbonate specks, and coarse to medium granular structure with biopores and vertical joints. Its profile is typically characterized by an alternation of silty or sandy Loess (yellow Loess layer) with more clay-rich Paleosols (brown–red layer). The pedogenesis process in the Loess–Paleosol sequence is rather unique. A Loess layer formed under the condition of dust accumulation rate outweighs the pedogenesis rate, and during the period of Paleosol formation, the dust accumulation was not stopped but was only slower than the pedogenesis.^[2] The Loess layers, which represent strong deposition process, have been affected by pedogenetic process in different degrees. Their soil types are mainly steppe or forest-steppe soils. In addition, there are aggregates, pores of earthworm and plants, and certain tree composition in pollen assemblage. It shows that the Loess layer had been subjected to pedogenetic process of steppe or forest-steppe undersemiarid climate.

GEOLOGY AND LANDFORM

The Loess Plateau is one of the most active areas of tectonic movement in China. Right from the quaternary period, there have been large areas of crustal uplift of about 150–200 m. With crustal uplift, valleys are formed and water concentrates into rivers in the valleys. The energy of water in the rivers causes increased soil erosion in the valleys. In addition, the Loess Plateau has become an area with frequent earthquakes due to tectonic movements. A harmful earthquake not only causes gravitational erosion but also destroys the structure of the Loess, leading to large areas of collapse and sliding. Moreover, the composition of Loess generated by rock and soil-forming processes results in substantial amounts of fine sandy loam and silt loam with strong tensile and stress joints so that collapse of the soil may happen only after 1–2 minutes in water. This Loess characteristic leads to high erodibility including channel and cave formation.

The main geomorphic landforms on the Loess Plateau are Yuan (a large flat surface) (Fig. 2A), ridge (“liang” in Chinese) (Fig. 3A), hill (“mao” in Chinese) (Fig. 3B) and various gullies. The degree of slope is also an important factor affecting soil erosion in Loess Plateau. Soil erosion becomes more intense as the slope increases.

SOIL EROSION ON THE LOESS PLATEAU

Vegetation removal and agricultural development started about 3000 years ago on the Loess Plateau area. Consequently, soil erosion became increasingly severe. The Loess Plateau is one of the most severely eroded areas in the world. Land area affected by soil erosion is 430,000 km², accounting for 81.1% of the total area of the Loess Plateau, of which severe soil erosion is 280,000 km², accounting for 52.8% of the total area. Every year, about 0.01–2 cm of topsoil is washed away.

Due to different climatic conditions, geographical features, and human activities, the regional distribution of soil erosion in the Loess Plateau is obvious. Generally speaking, there are three erosion areas, i.e., the wind erosion area, the wind–water erosion criss-cross area, and the water erosion area (Fig. 4). Among the three erosion regions, the water erosion area (Area III in Fig. 4) is the most important region because population is mainly concentrated in this region, and it is also a main sediment source of the Yellow River. Depending on the distinctly different geomorphology and erosion patterns, the water erosion area is divided into two subregions, i.e., the hilly gully subregion (Fig. 3) and the high Loess plain, with deep-cutting gullies, subregion (Fig. 2). Gully erosion occupies more than 60% of total soil erosion in a watershed in the hilly gully subregion and more than 80% in the high Loess plain with deep-cutting gullies. According to a gully survey, there are about 660,000 gullies, which are more than 500 m in length. In the Loess Plateau, the area with the highest gully density is near the Yellow River in the Shanxi and Shaanxi provinces, which became the main coarse sediment producing area in

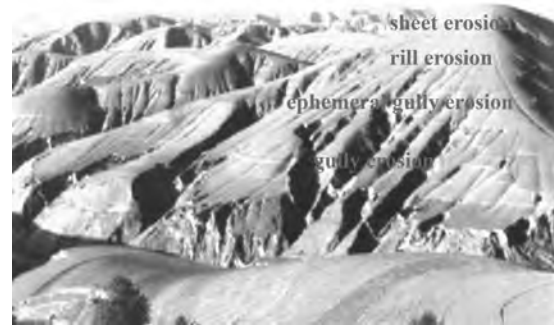


Fig. 5 Vertical distribution of water erosion patterns along hill-slope in the Loess Plateau.

the middle reaches of the Yellow River;^[9] for example, sediment from the North Shaanxi province occupies about 50% of total sediment delivery in the Yellow River.

Water erosion patterns along the slope shows a vertical distribution. From upslope to downslope, dominant erosion process follows the sequence of splash–sheet erosion–rill erosion–ephemeral gully erosion–gully erosion. Fig. 5 is a typical example of vertical distribution of water erosion patterns in the hilly gully region of the Loess Plateau. With the change of erosion patterns along the slope length, erosion intensity gradually increases. Generally speaking, sheet erosion, rill erosion, and ephemeral gully erosion frequently distribute in the cropland; all these inducing a decline of soil quality and reduction in crop productivity, and consequently affecting vegetation restoration.

Wind erosion is a secondly important environmental problem in the Loess Plateau, and is one of the main dust sources of sand storms in China. The wind erosion area

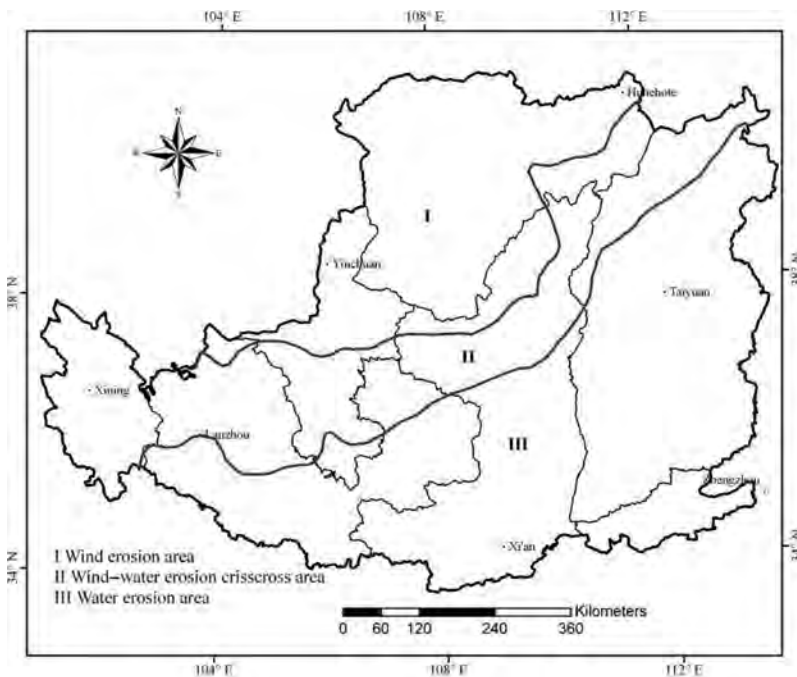


Fig. 4 Regional distribution of soil erosion of the Loess Plateau.

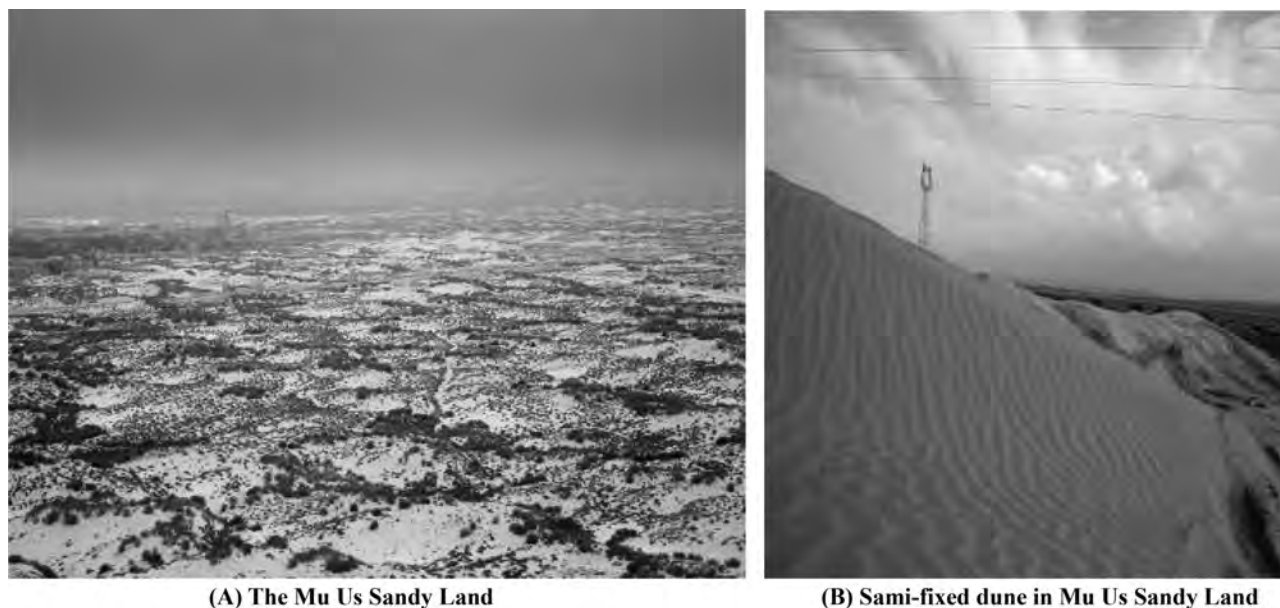


Fig. 6 Desert landscape of the Mu Us Sandy Land in the Loess Plateau (A and B).

mainly covers part of the Ordos Plateau (Area I in Fig. 4), which is bordered by the Great Wall in the south, extending northward to the Yinshan Mountain, and by the Helan Mountain in the west, stretching eastward to Linger–Dongsheng–Yulin line. It includes the Maowusu Sandy Land (Mu Us Sandy Land), Kubuqi Desert (Hobq Desert), Hedong Sandy Land, and Hetao Plain. The total area of wind erosion is about 180,000 km², accounting for 25.08% of the Loess Plateau.^[10,11] The domestic terrain is high plains, and the desert (sandy land) landscape is obvious (Fig. 6). Average annual precipitation is almost below 300 mm, annual evaporation is 2500–3000 mm, and the coefficient of humidity is 0.23–0.1 or less. The drought environmental and geographical attributes lead to the shrub species becoming dominant and make the vegetation transmitting from steppe to desert steppe.^[12,13] However, as the wind erosion area on the Loess Plateau mainly consisted of the desert, sandy land, and the Gobi, the erosion hazard is not obvious and has not gained much attention.

Another specific erosion area related to wind erosion which contributed to high soil erosion, is the wind–water erosion criss-cross area on the Loess Plateau. The wind–water erosion criss-cross area is a corridor between the water erosion area and the wind erosion area, extending mainly between 35°25′–40°38′ N and 103°00′–113°53′ E (Area II in Fig. 4). This area is notorious for dramatic climate changes, with frequent occurrences of natural disasters, such as flood, rain gush, drought, and sandstorm. Annual precipitation is 250–400 mm, and the uneven inter-annual precipitation distribution rate ranged between 2 to 7 times that of the normal year.^[11] Therefore, soil erosion occurs all year round, in which water erosion is dominant in

summer and autumn, and wind erosion mainly occurs in winter and spring. Due to the complex environmental and geographical attributes, about 216,000 km² out of the total 178,000 km² of the wind–water erosion criss-cross area is severe erosion region with the average annual erosion intensity more than 10,000 t km⁻²; and the extremely severe erosion center of the Loess Plateau, whose annual erosion intensity can reach 150,000 to 250,000 t km⁻², is located in the contiguous area of Shanxi, Shaanxi Provinces, and Inner Mongolia Autonomous Region.^[14,15]

EROSION CONTROL ON THE LOESS PLATEAU

Since 1950, a set of soil and water conservation projects have been implemented on the Loess Plateau, comprising construction of large reservoir at the main stream and branches of the Yellow River, backbone dam and check dams in gullies, and integrated management of small watershed. Implementing these projects has brought great benefits for both ecology and economy. Especially in 1999, Grain for Green Project (GGP) was issued as one of the government policies for Chinese ecological restoration, which means steep cropland was converted to permanent vegetation cover with the government giving some compensation to local farmers, including grain, cash, and seedling fee. After more than 10 years of GGP implementation, local landscape has obviously changed (Fig. 7) and local farmers' livelihoods have greatly improved. Owing to implementation of several soil and water conservation projects, obvious stepwise decrease in sediment load has occurred in the Yellow River, and thus as a result, ecological



Fig. 7 Local landscape change caused by GGP on the Loess Plateau.

security is greatly improved. Both on-site and off-site positive impacts on ecology and economy of erosion control are remarkable.

REFERENCES

1. He, X.B.; Tang, K.L.; Lei, X.Y. Heavy mineral record of the holocene environment on the Loess Plateau in China and its pedogenetic significance. *Catena* **1997**, *29*, 323–332.
2. Liu, T.S. *Loess and the Environment*; China Ocean Press: Beijing, 1985.
3. Yang, W.-Z.; Yu, C.-Z. *Regional Management and Evaluation of Loess Plateau*; Science Press: Beijing, 1992; 241–297.
4. Tang, K.L.; Xiong, G.S.; Liang, J.Y. *Varieties of Erosion and Runoff Sediment in Yellow River Basin*; Chinese Sciences and Technique Press: Beijing, 1993; 12–13.
5. Sediment Specility Committee of Chinese Water Resources Association. *The Handbook of Sediment*; Chinese Environmental Sciences Press: Beijing, 1989.
6. Zhu, X.M. The formation of Loess Plateau and its harnessing measures. *Bull. Soil Water Conserv.* **1991**, *11* (1), 1–9.
7. Yang, Q.Y.; Yuan, B.Y. *Environment on the Loess Plateau Regions and Its Evolution*; Science Press: Beijing, 1991.
8. Zhu, X.M. *Lutu Soil*; Agriculture Press: Beijing, 1962.
9. Qian, N.; Wang, K.Q.; Yan, L.D.; Fu, R.S. The source of coarse sediment in the middle reaches of the Yellow River and its effect on the siltation of the lower Yellow River. In *Chinese Society of Hydraulic Engineering. Proceeding of the International Symposium on River Sedimentation*, Chinese Society of Hydraulic Engineering, March 24–28, 1980; Guanghua Press: Suzhou, Vol. (1), 53–62.
10. Huang, B.-W. Experience and lesson of mapping soil erosion region in middle reaches of Yellow River. *Bull. Sci.* **1955**, *12*, 15–21.
11. Tang, K.L. Importance and urgency of harnessing the interlocked area with both water and wind erosion in the Loess Plateau. *Soil Water Conserv.* **2000**, *11* (244), 11–17.
12. Tang, K.L. *Soil and Water Conservation in China*; Science Press: Beijing, 2004; 845 pp.
13. Fu, B.J. Soil erosion and its control in the Loess Plateau of China. *Soil Use. Manage.* **1989**, *5* (2), 76–82.
14. Zhu, X.M. Proper development and careful protection of land resources in the Loess Plateau. *Sci. Geogr. Sinica* **1984**, *4*, 97–105.
15. Jing, K.; Wang, W.Z.; Zheng, F.L. *Soil Erosion and Environment in China*; Science Press: Beijing, 2005; 359 pp.

Low-Carbon Technology: Brazilian Semiarid Ecosystems

Antonio Pereira Filho

Federal University of San Francisco Valley, Juazeiro, Brazil

Vanderlise Giongo

Tony Jarbas Ferreira Cunha

Ministry of Agriculture, Livestock, and Food Supply, Brazilian Agricultural Research Corporation (Embrapa Semiarid), Brasília, Brazil

Jose Texeira Filho

Agricultural Engineering, University of Campinas, Campinas, Brazil

Tayane de Lima Santos

Federal University of Ceara, Fortaleza, Brazil

Luiz Fernando Carvalho Leite

Brazilian Agricultural Research Corporation (Embrapa Mid-North), Teresina, Brazil

Abstract

Agriculture is a vulnerable sector to the climate changes, and Brazil is one of the major contributor of greenhouse gases (GHGs). Thus, the implementation of effective strategies for both mitigation and adaptation to climate change for agriculture is very important for development and also to infer sustainability to the Brazilian agricultural sector. The “low-carbon agriculture” aims to develop processes and technologies that promote mitigation of GHG emissions in agriculture and enable the adaptation of the agricultural sector to climate change. Thus, in Brazilian semiarid region, developing technologies with no-tillage systems concomitant with green manure and agrosilvopastoral systems are adapted to climatic conditions to infer sustainability in the livestock and agricultural sectors.

INTRODUCTION

The greenhouse gas (GHG) emissions have their dynamic emission to the atmosphere related to agricultural production processes, changes in land use, the burning of fossil fuels, and industrial activities. In a global level, changes in land use added to the agricultural production processes contributed 22% of carbon dioxide (CO₂) emissions, 43% of methane (CH₄) emissions, and 85% of nitrous oxide (N₂O) emissions. In Brazil, historically, this sector contributed 75% of CO₂ emissions, 91% of CH₄ emissions, and 94% of N₂O emissions, considerably higher in relation to global averages.^[1] Thus, conservation technologies, based on low-carbon (C) agriculture, are fundamental to mitigate the impact of land use change and climate change on GHG emissions.

Studies directed to determining the impact of land use change and climate changes in C stock are sporadic in the Brazilian semiarid region, mainly because of large variability and mosaic distribution of the soils and vegetation.^[2] This question assumes greater importance in the face of

climate change scenarios for the Brazilian semiarid region in which extreme events would negatively impact agricultural production.^[3–4] Thus, this entry presents the main concerns related to the land use change and agriculture in Brazilian semiarid region, and the low-C agriculture technologies adapted to climate conditions that will imply sustainability to regional production system.

LAND USE CHANGE IN THE BRAZILIAN SEMIARID

The Brazilian semiarid region, with 969,589 km², has 28 million people, including urban areas, representing 11% of the country, and 1.6 million agricultural establishments, with 95% classified as family farms. Caatinga is the most representative Brazilian semiarid biome, with an approximate area of 844,453 km²,^[5] which is formed mainly by low trees and shrubs, many of which have thorns, microphyll and xerophytic characteristics, and herbaceous with great importance because of its foraging,

medicinal and apicultural value.^[6] However, the change of land use with the use of woody plants for energy production together with the use conversion aimed at agricultural production is responsible for the removal of 46.38% of caatinga vegetation.^[7] Livestock is the main land use followed by rainfed agriculture and irrigated agriculture, changing the soil C stock. In this sense, Sampaio and Costa^[8] estimated that the average soil C stocks in caatinga areas, native pasture, planted pasture, and crops are 100, 90, 80, and 70 Mg ha⁻¹, respectively.

TECHNOLOGIES TO MITIGATE THE LAND USE CHANGE AND CLIMATE CHANGE IMPACTS

No-till farming, green manures adoption, agroforestry systems, silvopastoral systems, and organic farming are technologies used in semiarid region to increase the soil organic C content, consequently reducing the GHG emissions.^[9–11] They are fundamental to consolidate a low-C agriculture adapted to the soil and climatic conditions of the Brazilian semiarid.

No-Till System and Green Manure

In the Brazilian agricultural soils, no-till contributes to both increased soil microbial biomass and C sequestration, with increments from 5.2 to 8.5 Mg C ha⁻¹, superior to soil under conventional tillage.^[12] However, there are difficulties to implement that practice in the Brazilian semiarid region due to climatic limitations and the traditional requirement of crop residues to livestock feed.^[13] Conversely, the priority of the soil conservation in Brazilian semiarid region with the use of cover crops and its residues allows the increase of water infiltration and retention, the increase of organic matter content, the decrease of soil temperature oscillations and evaporation, and, subsequently, the decrease of the salinization process. In this sense, many studies have focused on no-tillage system development for corn crops,^[14,15] melon,^[16] cowpea,^[17] and watermelon.^[18] All the proposed crop systems consist of rotation, succession, or intercropping, with emphasis on the green manure use. Among the technologies studied and implanted with no-till system in Brazilian semiarid region

stands out the use of plant cocktails. Thus, the studies of irrigated areas in Brazilian semiarid region have been developed using plant cocktail as green manure in mango crop (since 2005) and melon crop (*Cucumis melo* L.; since 2011), as technology for low-C agriculture.

Studies have already been developed with species cultivated as plant cocktail, covering different proportions of grasses, legumes, and oilseeds to evaluate the best composition in the culture under study.^[19] Grasses generally contributed to relatively large amounts of biomass and characterized by a high C/nitrogen (N) ratio that increases the persistence of soil cover over time.^[20] On the contrary, the leguminous species, which fix atmospheric N, have high levels of N in plant biomass, and their residues generally have low C/N ratio with relatively fast decomposition that promotes a small soil cover.^[21] The Brazilian semiarid region has a potential to add large amounts of C and nutrients in the soil of agricultural systems in a short period by means of the green manure crops (Table 1). The plant cocktail biomass increases the rate of adding C that may be stored in the soil system over time.

There is a lack of studies on no-till system impact and green manure on C dynamics in the Brazilian semiarid soil, both taking into account the topsoil along the profile. In Brazil, the studies carried out throughout the territory estimated that soil C stocks are 39, 52, 72, and 105 Pg in 0.0–0.3, 0.0–0.5, 0.0–1.0, and 0.0–2.0 m layers, respectively.^[22] The distribution of C on profile decreases significantly with the soil use, initially in surface and after in subsurface,^[23] demonstrating that there are significant differences between the surface C and C in depth.^[24]

The roots contribute twice as much to the soil organic C compared to plant aerial part due to a higher proportion of lignified tissue.^[25] The production of biomass and the root cycling, especially the thin ones, influence the amount and dynamics of C in soil. The root cycling is regulated by soil moisture and temperature and microbial activity due to their biochemical characteristic^[26] or associated with plant hormonal and physiological reactions. In arboreal stratum, due to the high renewal rate over the tree life cycle, there is a high rate of adding C to the root system.^[27] Consequently, in addition to the aerial biomass contribution of the green manure is also important to quantify the production of root

Table 1 Production of dry matter and nutrient accumulation in the aerial part of plant cocktail and spontaneous vegetation (Petrolina, Embrapa semiarid region, 2013).

Green manure	Dry phytomass						
	(Mg ha ⁻¹)	N (kg ha ⁻¹)	P (kg ha ⁻¹)	K (kg ha ⁻¹)	Ca (kg ha ⁻¹)	Mg (kg ha ⁻¹)	S (kg ha ⁻¹)
75% L + 25% NL	10.61	212.41	55.47	319.31	133.15	100.00	74.87
25% L + 75% NL	11.00	126.50	49.18	360.07	177.02	155.85	75.76
Spontaneous vegetation	3.99	78.68	22.66	105.52	43.43	38.81	29.68
CV (%)	21.68	23.12	32.15	26.75	24.91	27.84	23.06

Note: L, legumes; NL, no legumes.

biomass and the nutrient accumulation by the roots along the soil profile. It was verified that green manure cultivated in the Brazilian semiarid region added quantity of biomass ($5.0\text{--}5.6\text{ Mg ha}^{-1}$) and nutrient [N: $70.1\text{--}80.4$; phosphorus (P): $4.2\text{--}5.3$; potassium (K): $39.9\text{--}40.0$; calcium (Ca) $39.8\text{--}47.7$; magnesium (Mg): $5.8\text{--}7.3$; and sulfur (S): $6.3\text{--}6.8$] along the upper soil profile when compared with the spontaneous vegetation (biomass: 1.5 Mg ha^{-1} and N: 38.6 ; P: 2.0 ; C: 10.9 ; Ca: 17.7 ; Mg: 2.8 ; and S: 3.3). These observations do not include plant exudates and mucigels released by the root system during the development of the green manure. Thus, the use of plant cocktails as green manure in both mango crop and melon crop, even for a short period of time, increased the content of soil organic matter, initially occurring in the surface and subsurface layers. It was also observed that the highest C/N ratio influences the behavior of the humic fractions, indicating the formation of humus with less fulvatic character.^[28]

Understanding the dynamics of organic matter and accurately estimating C stocks in surface and subsurface are important to the development of technologies and sustainable agricultural systems and involve long-term studies. Lima^[29] evaluated the melon C footprint produced in Brazilian semiarid region, comparing the following production systems: conventional system used by most producers in the region and the conservationist system, in development and experimental scale with three-year cultivation, using no-tillage with green manure via plant cocktail. The conservationist system uses plant cocktail (composition: 75% legume species and 25% of no legumes species) as green manure without incorporation in soil. The C footprint results for a ton of melon produced and transported to the consumer market (State of São Paulo, about 2000 km) were $821\text{ kg CO}_2\text{eq}$ in conventional system and about $639\text{ kg CO}_2\text{eq}$ in conservationist system. The conventional system demands larger amounts of synthetic fertilizers, resulting in direct N_2O emissions and indirect NH_4 and NO_3^{2-} emissions that affect the melon C footprint, whereas conservationist system has lower contribution due to lower loss C, considering the amount of C returned to the soil by green manure biomass used.

Agroforestry System

The agroforestry system is a sustainable practice of natural resource management that combines forest species, agricultural crops, and/or livestock in the same exploitation area, simultaneous or in temporal sequence. This technology aims to ensure the production stability and diversity, increase productivity, improve soil fertility, and increase the supply of a good-quality forage.^[30] Besides, it reduces GHG emissions and increases the C stock in the soil and plant system.^[31]

The agrosilvopastoral system is the predominant agroforestry system in Brazilian semiarid region, combining crops, forest species, and animals in the same area. There is also the silvopastoral system with the introduction of

animals in areas with permanent and commercial tree species or with the introduction or maintenance of the tree component (native or exotic) in cultivated pastures and adapted to the region.^[32] Studies have shown that the preservation of natural trees during the cutting of caatinga trees or cultivation of native or introduced species may contribute to the preservation or restoration of soil fertility in agricultural or pasture areas in northeast semiarid region of Brazil. Sacramento et al.^[11] evaluated the changes in soil C stocks in the layer of $0\text{--}40\text{ cm}$, after 13 years of experiment installed in a typical Orthic Luvisols. They observed that silvopastoral agroforestry farming systems (97.59 Mg ha^{-1}) and agrosilvopastoral (71.64 Mg ha^{-1}) and traditional (with clearing of vegetation, burning, and cultivation during 2 to 3 years; 68.16 Mg ha^{-1}) systems had lower C stocks compared to the remaining caatinga vegetation (134.65 Mg ha^{-1}); however, the agroforestry systems outperformed the traditional system. But, some studies have shown that measurements of changes in some soil variables in agroforestry systems, such as total organic C, are imperceptible in less than 10 years.^[33]

Drumond^[34] highlighted that the main arboreal species with multiple potential use for agroforestry systems in the Brazilian semiarid region are as follows: *Leucaena* (*Leucaena leucocephala*), *Gliricidia* (*Gliricidia sepium*), Mesquite (*Prosopis juliflora*), Sabiá (*Mimosa caesalpiniiifolia*), and Eucalyptus (*Eucalyptus* sp.). These species have forestry development in areas dependent on rain with an average annual rainfall ranging from 500 to 700 mm.

Among the agrosilvopastoral models, the following stand out: 1) pastures cultivated with grasses: Buffel (*Cenchrus ciliaris*), Aridus grass (*Cynodon dactylum* var *aridus*), and Urochloa (*Urochloa moçambisensis*); 2) *Leucaena* protein banks (*Leucaena leucocephala*) grown in rows ($4.0 \times 1.0\text{ m}$) and intercropped with maize and/or beans; 3) *Gliricidia* protein banks (*Gliricidia sepium*) grown in rows ($4.0 \times 1.0\text{ m}$) and intercropped with maize; 4) cactus areas planted with the giant varieties (*Opuntia ficusindica*) and round varieties (*Opuntia stricta*), in a dense system, spacing $1.0 \times 0.25\text{ m}$ and $1.0 \times 0.5\text{ m}$, respectively, and in single-row system ($3.0 \times 0.25\text{ m}$) intercropped with *Gliricidia* (*Gliricidia sepium*), on lines and corn in between rows; 5) reforested areas with Sabiá (*Caesalpinia echinata*) established in spacing of $10.0 \times 3.0\text{ m}$; and 6) live fences with forage gliricidia.^[35] Among all systems, we highlight the importance of using leguminous species for biological N fixation.

The planting in alleys is also a technique used in Brazilian semiarid region to improve the quality of soil. It consists of plant trees in rows properly spaced between them, where among the lines are cultivated crops at the beginning of the rainy season. The main objective of this system is to increase the organic matter content, nutrient cycling, and addition of N in by green manure.^[36]

The main leguminous species used as alley farming are as follows: *Crotalaria* (*Crotalaria juncea*), Jack Bean

(*Canavalia ensiformis*), Black mucuna (*Stylobium aterrimum*), Guandu (*Cajanus cajan*), Leucaena (*Leucaena leucocephala*), Gliricidia (*Gliricidia sepium*), Sabia (*Mimosa caesalpinifolia* Benth), and Canafistula (*Peltophorum dubium*). The choice of these species will depend largely on weather conditions, soil types, and culture characteristics.^[37]

Gliricidia sepium, e.g., is a legume largely used in alleys farming due to good development conditions under water stress and high capacity of biomass production.^[38] Studies by Marin et al.^[38] evaluated the corn production and chemical composition and soil characteristics modifications with the use of *Gliricidia* in alley farming system in the semiarid region. They observed that the use of *G. sepium* did not alter the levels of total organic matter but significantly increased the light organic matter content, available P, and extractable K in soil.

Oliveira et al.^[39] evaluated the carbon management index (CMI), calculated from the C stock index and lability index,^[40] among the conventional agricultural systems and agroforestry systems in the semiarid region of Ceará, Brazil. They observed that the CMI showed the highest values for the agrosilvopastoral system (120–200%) compared to the silvopastoral (0–60%) and traditional (100–160%) systems. Therefore, according to the authors, agroforestry systems are effective for improving soil quality, increasing C stock, and promoting sustainability in these environments.

There are few studies involving the agroforestry systems impact on C stocks in soil and plant system, and also there are no systematic studies to assess the impact that integrates scenarios of global climate change in areas or long-term experiments in Brazilian semiarid region.

CONCLUSION

Agriculture and livestock are very important activities in the semiarid economy, characterized by caatinga biome. However, the extractive livestock, monoculture, excessive soil preparation, and lack of soil cover, promoted by land use changes, contribute to soil erosion, decomposition and mineralization, salinization, and/or sodification, decreasing the C stock in soil. Predicted impacts on climate change scenarios may potentialize the C stock reduction. Thus, the technologies and farming systems, such as no-tillage system associated with green manure and the agroforestry systems, are alternatives that can reduce GHG emissions, increase C sequestration, and mitigate the impact of climate change in the Brazilian semiarid region.

REFERENCES

1. Cerri, C.C.; Maia, S.M.F.; Galdos, M.V.; Cerri, C.E.P.; Feigl, B.J.; Bernoux, M. Brazilian greenhouse gas emissions: The importance of agriculture and livestock. *Sci. Agric.* **2009**, *66* (6), 831–843.
2. Salcedo, I.H.; Sampaio, E.V.S.B. Matéria orgânica do solo no bioma caatinga. In *Fundamentos da Matéria Orgânica do Solo: Ecossistemas Tropicais e Subtropicais*, 2nd Ed.; Metropole: Porto Alegre, 2008; 419–441.
3. Lima, R.D.C.C.; Cavalcante, A.D.M.B.; Marin, A.M.P. *Desertificação e Mudanças Climáticas no Semiárido Brasileiro*. Ministério; Instituto Nacional do Semiárido – INSA: Campina Grande, 2011.
4. IPCC. 2013. In Intergovernmental panel on climate change. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change; Cambridge University Press: Cambridge and New York.
5. IBGE. Instituto Brasileiro de Geografia e Estatística. Estatísticas sobre agropecuária e produção, 2010. Available at www.sidra.ibge.gov.br.
6. Araujo Filho, J.A. *Manipulação da Vegetação Lenhosa da Caatinga Para Fins Pastoris*; Embrapa-CNPQ: Sobral, 1990.
7. Campello, B. O uso da energia de Biomassa no Bioma Caatinga. *Proceedings of the V Semana do Meio Ambiente / Mudanças climáticas e o Nordeste Brasileiro*; Recife, Pernambuco, Brazil, Jun 3–5, 2008; Fundação Joaquim Nabuco (Fundaj/MEC), Recife, 2008. 1 CD ROM.
8. de Sá Sampaio, E.V.; da Costa, T.L. Estoques e fluxos de carbono no semi-árido Nordeste: Estimativas preliminares. *Rev. Bras. Geogr. Física* **2011**, *6*, 1275–1291.
9. Maia, S.M.F.; Xavier, F.A.D.S.; Oliveira, T.S.D.; Mendonça, E.D.S.; Araújo Filho, J.A.D. Impactos de sistemas agroflorestais e convencional sobre a qualidade do solo no semi-árido cearense. *Rev. Árvore* **2006**, *30*, 837–848.
10. Aguiar, M.I.; Maia, S.M.F.; Xavier, F.A.D.S.; de Sá Mendonça, E.; Filho, J.A.A.; de Oliveira, T.S. Sediment, nutrient and water losses by water erosion under agroforestry systems in the semi-arid region in northeastern Brazil. *Agrofor. Syst.* **2010**, *79*, 277–289.
11. Sacramento, J.A.A.S.; Araújo, A.C.M.; Escobar, M.E.O.; Xavier, F.A.S.; Cavalcante, A.C.R.; de Oliveira, T.S. Soil carbon and nitrogen stocks in traditional agricultural and agroforestry systems in the semiarid region of Brazil. *Rev. Bras. Ci. Solo* **2013**, *37* (1), 784–795.
12. Pereira, M.F.S.; de Sá, J.R.; Linhares, P.C.F.; Neto, F.B.N. Ciclagem do carbono do solo nos sistemas de plantio direto e convencional. *ACSA – Agropecuária Científica no Semi-Árido* **2013**, *8* (1), 21–32.
13. Brandão, S.; Giongo, V.; Santana, S.; Monteiro, A.; Mendes, S.; Petrere, C. Efeito da adubação verde nos teores de matéria orgânica e fósforo em Vertissolo cultivado com meloeiro irrigado no Semiárido, I Reunião Nordestina de Ciências do Solo, Areia, Paraíba, Brazil, Sept. 22–26, 2013; Centro de Ciências Agrárias/Universidade Federal da Paraíba; 2013. Available at <http://ainfo.cnptia.embrapa.br/digital/bitstream/item/91544/1/Vanderlise.pdf> (accessed April 2015).
14. Pereira, R.G.; Medeiros, V.Q.D.P.; Cavalcante, M.; Cruz, S.C.S.; da Barros, E.S. Avaliação de espécies forrageiras como plantas de cobertura sobre os componentes de produção do milho cultivado no sistema plantio direto. *Rev. Caatinga* **2009**, *22* (3), 1–04.
15. Silva, A.S.; Da Silva, I.D.F.; Da Silva Neto, L.D.F.; De Souza, C. Semeadura direta na produção do milho em

- agricultura de sequeiro na região Nordeste do Brasil. *Ciência Rural* **2011**, *41* (9), 1556–1562.
16. Teófilo, T.M.S.; Freitas, F.C.L.; Medeiros, J.F.; Fernandes, D.; Grangeiro, L.; Tomaz, H.V.Q.; Roberto, A.P.M.S. Eficiência no uso da água e interferência de plantas daninhas no meloeiro cultivado nos sistemas de plantio direto e convencional. *Planta Daninha* **2012**, *30* (3), 547–556.
 17. de Freitas, R.M.O.; Dombroski, J.L.D.; de Freitas, F.C.L.; Nogueira, N.W.; de Pinto, S.J.R. Crescimento de feijão-caupi sob efeito de veranico nos sistemas de plantio direto e convencional. *Biosci. J.* **2014**, *30* (2), 393–401.
 18. Silva, D.V.; Santos, J.B.; Ferreira, E.A.; Silva, A.A.; França, A.C.; Sedyama, T. Manejo de plantas daninhas na cultura da melancia nos sistemas de plantio direto e convencional. *Hortic. Bras.* **2013**, *31* (3), 901–910.
 19. Chaves, V.C.; Ferreira, G.B.; Mendonça, C.E.S.; Petrere, V.G.; Cunha, T.J.F.; da Silva, M.S.L. *Potencialidade de coqueiros para adição de matéria seca ao sistema solo na cultura da mangueira*; 2007. Available at <https://www.embrapa.br/busca-de-publicacoes/-/publicacao/160492/potencialidade-de-coqueiros-vegetais-para-a-adicao-de-materia-fresca-e-seca-ao-sistema-solo-na-cultura-da-mangueira> (accessed April 2015).
 20. Andreola, F.; Costa, L.M.; Olszewski, N.; Jucksch, I. A cobertura vegetal de inverno e a adubação orgânica e, ou, mineral influenciando a sucessão feijão/milho. *R. Bras. Ci. Solo* **2000**, *24*, 867–874.
 21. Perin, A.; Santos, R.H.S.S.; Urquiaga, S.S.; Cecon, P.R.; Guerra, J.G.M.; De Freitas, G.B. Sunnhemp and millet as green manure for tropical maize production. *Sci. Agric. (Piracicaba, Braz.)* **2006**, *63* (5), 453–459.
 22. Leite, L.F.C.; Petrese, V.G.; Sagrilo, E. Sequestro de carbono em solos da região semiárida brasileira estimado por modelo de simulação em diferentes sistemas produtivos. *ICID+18 Conferência Int. – Sustentabilidade e Desenvolvimento em Regiões Semiáridas*; 2010; 11 pp. Available at <http://ainfo.cnptia.embrapa.br/digital/bitstream/item/60095/1/Vanderlise-2010.pdf> (accessed April 2015).
 23. Meersmans, J.; Van Wesemael, B.; De Ridder, F.; Van Molle, M. Modelling the three-dimensional spatial distribution of soil organic carbon (SOC) at regional scale (Flanders, Belgium). *Geoderma* **2009**, *152*, 43–52.
 24. Santruckova, H.; Kaštovská, E.; Kozlov, D.; Kurbatova, J.; Livečková, M.; Shibistova, O.; Tatarinov, F.; Lloyd, J. Vertical and horizontal variation of carbon pools and fluxes in soil profile of wet southern taiga in European Russia. *Boreal Environ. Res.* **2010**, *15* (3), 357–273.
 25. Pergoraro, R.F.; da Silva, I.R.; de Novais, R.F.; de Barros, N.F.; Fonseca, S.; Dambroz, C.S. Estoques de carbono e nitrogênio nas frações da matéria orgânica em argissolo sob eucalipto e pastagem. *Rev. Ciência Florest.* **2011**, *21* (2), 261–273.
 26. Mello, S.L.M.; Gonçalves, J.L.M.; Gava, J.L. Pre and post-harvest fine root growth in *Eucalyptus grandis* stands installed in sandy and loamy soils. *For. Ecol. Manage.* **2007**, *246*, 186–195.
 27. Gonçalves, J.L.M.; Mello, S.L.M. O sistema radicular das árvores. In *Nutrição e Fertilização Florestal*; de Gonçalves, J.L.M., Benedetti, V., Eds.; IPEF: Piracicaba, 2000; 219–268.
 28. Medeiros, M.G. de. *Frações da matéria orgânica do solo devido ao cultivo de coqueiros vegetais, em condições irrigadas, em um argissolo amarelo do semiárido brasileiro*. 2012, 64 p. Dissertação (mestrado em Ciências do Solo) – Universidade Federal Rural do semiárido, Mossoró, Brazil, 2012. Available at <http://www2.ufersa.edu.br/portal/view/uploads/setores/81/Dissertacao%20Monalisa%20Gurgel.pdf> (accessed april 2015).
 29. Santos, T.L. *Avaliação de impactos ambientais da produção de melão em sistema convencional e conservacionista no submédio São Francisco*, 111 p. Dissertação (Mestrado em Engenharia Civil: Saneamento Ambiental); Centro de Tecnologia, Universidade Federal do Ceará, Fortaleza, Brazil, 2015. Available at http://www.repositorio.ufc.br/bitstream/riufc/13008/1/2015_dis_tlsantos.pdf (accessed August 2015).
 30. Altieri, M.A. *Agroecology: The Science of Sustainable Agriculture*, 2nd Ed.; Westview Press Inc.: Boulder, 1995.
 31. Lorenz, K.; Lal, R. Soil organic carbon sequestration in agroforestry systems: A review. *Agron. Sustain. Dev.* **2014**, *34* (2), 443–454.
 32. Balbino, L.C.; Barcellos, A.O.; Stone, L.F. *Marco Referencial em Integração Lavoura-Pecuária-Floresta (iLPF)* Embrapa Informação Tecnológica: Brasília.
 33. Barreto, A.C.; Fernandes, M.F. Cultivo de *Gliricidia sepium* e *Leucena leucocephala* em alamedas visando a melhoria dos solos dos tabuleiros costeiros. *Pesqui. Agropecu. Bras.* **2001**, *36*, 1287–1293.
 34. Drumond, M.A. Espécies arbóreas potenciais para sistemas integrados de produção (iLPF) no semiárido Brasileiro. In *Integração Lavoura-Pecuária-Floresta: Potencialidades e Técnicas de Produção*; Santos, L.D.T., Mendes, L.R., Duarte, E.R., Glória, J.R., Andrade, J.M., Carvalho, L.R., Sales, N.L.P., Eds.; Instituto de ciências Agrárias da UFMG: Montes Claros, 2012; 85–99.
 35. Sá, C.O.; Sá, J.L.; de Rangel, J.H.A.; Muniz, E.N. Sistema agrossilvipastoril para convivência com o semi-árido sergipano. *Rev. Bras. Agroecol.* **2009**, *4* (2), 2781–2785.
 36. Vasconcelos, M.C.C.A.; Silva, A.F.A.; Lima, R.S. Cultivo em aléias: Uma alternativa para pequenos agricultores. *Rev. ACSA.* **2012**, *8*, 18–21.
 37. Eiras, P.P.; Coelho, F.C. Utilização de leguminosas na adubação verde para a cultura de milho. *Rev. Cient. Int.* **2011**, *4*, 28.
 38. Marin, A.M.P.; Menezes, R.S.C.; Salcedo, I.H. Produtividade de milho solteiro ou em aléias de gliricídia adubado com duas fontes orgânicas. *Pesqui. Agropecu. Bras.* **2007**, *42* (1), 669–677.
 39. Oliveira, T.S.; Nogueira, R.D.S.; Teixeira, A.D.S.; Campanha, M.M. Distribuição espacial do índice de manejo do carbono em luvisolos sob sistemas agrícolas tradicionais e agroflorestais no município de sobral-CE. *Rev. Bras. Agroecol.* **2009**, *4* (2), 589–592.
 40. Blair, G.J.; Lefroy, R.D.B.; Lisle, L. Soil carbon fractions based on their degree of oxidation, and the development of a carbon management index for agricultural system. *J. Agric. Res.* **1995**, *46*, 1459–1460.

Lower Himalayas: Soil Management

Anup Das

Ramkrushna Gi

Badapmain Makdoh

Research Complex for North Eastern Hill Region, Indian Council of Agricultural Research (ICAR), Meghalaya, India

Dibyendu Sarkar

Research Complex for North Eastern Hill Region, Indian Council of Agricultural Research (ICAR), Manipur, India

Jayanta Layek

Sandip Mandal

Research Complex for North Eastern Hill Region, Indian Council of Agricultural Research (ICAR), Meghalaya, India

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

The Eastern Himalayan region of India, also called the North Eastern Region (NER), comprises of eight states, namely, Arunachal Pradesh, Assam, Manipur, Meghalaya, Mizoram, Nagaland, Sikkim, and Tripura. The NER of India has geographical area of 26.3 million hectares and population of 45 million. Almost 85% of soils in this region are moderate to strongly acidic. About 47.5% land is prone to degradation by soil erosion, acidity, etc. Nutrient mining by extractive agriculture is the major reason for depletion of soil fertility. Decline in soil organic matter because of residue burning, excessive tillage, soil erosion, etc. along with the depletion of soil biological quality is the major concern. With the efficient management of available natural resources and need-based application of fertilizer and manure, soil health can be improved and sustained. Agroforestry, microwatershed-based integrated farming system, integrated nutrient management, conservation tillage, and residue management are among the options for sustaining soil health and agricultural productivity.

INTRODUCTION

The Himalayas represents the youngest but complex mountain system on the earth, which is still evolving and is recognized as 1 of the 34 global biodiversity hot spots. India's Eastern Himalayan region constitutes about 52% of total Himalayas [total of 52.42 million hectare (Mha)] and consists of eight states (Arunachal Pradesh, Assam, Manipur, Meghalaya, Mizoram, Nagaland, Sikkim, and Tripura) and Darjeeling hills of West Bengal. Located at the tri-junction of Indo-Chinese, Indo-Malayan, and Palaearctic biogeographic realms, India's Eastern Himalayan region has diverse hilly terrain with wide-ranging altitude. It shares the borders with China, Myanmar, Nepal, Bangladesh, and Bhutan. The temporal and spatial variations caused by diversity in geological orogeny have

resulted into a marked difference in climate and physiography and consequently in distribution pattern of biotic elements, including the domesticated ones. The Eastern Himalayan region, also called the North Eastern Region (NER), lies between 22°05' and 29°30' N latitudes and 87°55' and 97°24' E longitudes. The region represents three provinces (East Himalayas, Brahmaputra Valley, and North East Hills) and covers about 7.7% of the total geographic area of India.

Despite rich endowment of agroclimatic and geographical diversity, the NER has lagged behind in economic development. Nonetheless, there is an ample potential for development with abundance of natural resources. Soils in the valleys are rich in soil organic matter (SOM) but are mostly degraded. The SOM content is low to medium in arable lands on steep slopes because

of accelerated soil erosion. Around 56% of the cultivated area of the NER is under low altitude (valley or lowland), 33% under mid-altitude (flat upland), and the rest under high altitude (upland terrace). Traditionally, farmers in both upland terrace and valleys practice monocropping under rainfed agriculture, where rice (*Oryza sativa*) is the major crop occupying more than 80% of the cultivated area followed by maize (*Zea mays*). The cropping intensity of the NER is 130%.

The “slash and burn” agriculture (shifting cultivation or *Jhum*) is practiced on about 0.88 Mha. Deforestation and biomass burning in *Jhum* aggravate soil erosion and ecosystem degradation. *Bun* cultivation (raised beds of 0.15–0.30 m height and 0.75–1 m width with almost equal width under sunken area made along the slopes) is a modification of shifting cultivation and is mostly followed in the Meghalaya plateau. In the *bun* cultivation, biomass is burnt under the soil, and the land is abandoned after two or 3 years. This practice aggravates soil erosion and degradation. Annual soil erosion on steep slopes (44–53%) under shifting cultivation can be as much as 40.9 Mg/ha along with attendant losses (in kg/ha) of 702.9 of soil organic carbon (SOC), 63.5 of phosphorus (P) and 5.9 of potassium (K).^[1] Soil erosion, during the 1st and 2nd years, on the abandoned land has been estimated at 147, 170, and 30 Mg/ha, respectively.^[1] Steep slopes, cultivated along the slope, with negligible nutrient replacement and high rainfall are among the major causes of land degradation in the NER.^[2]

Maintaining and enhancing soil quality are crucial to sustaining agricultural productivity and environment quality. Indiscriminate use of intensive agriculture has adversely impacted soil and environment over the past decades. Biological methods of soil and water conservation, especially grass or vegetation cover, are suitable for hilly ecosystems and are cost-effective.^[2] Soil health management in the fragile hill ecosystems of the NER should be based on recycling of available plant residues, agroforestry, conservation agriculture, integrated nutrient management (INM), microwatershed-based integrated farming system (IFS) and permanent pasture grasses on sloping lands, and intensive IFS in valleys. The objective of this entry is to describe the status of soil fertility, explain different types of soil degradation, and outline some management options considering the availability of natural resources to sustain soil health and productivity in the NER.

Land Use and Agroclimatic Conditions

About 54% of the geographical area of the region is under forest—reserve, protected, and unclassed forest under the forest department. The land is mostly owned by communities and allotted to villagers for *Jhum* and other uses by the village headman or the land council. The states like Meghalaya and Tripura have no intact

forest. Forests in Mizoram (71.7%) and Nagaland (53.3%) are relatively core forest. Sikkim, Manipur, and Tripura have considerable area of forests which is threatened by severe fragmentation.^[3]

The net and gross cultivated area in the NER is about 4 and 5.7 Mha, respectively. The state of Assam has the maximum cultivated area followed by that in Nagaland and Meghalaya. Cropping intensity is the highest in the state of Tripura (184%) and the lowest in Mizoram (106%). The share of total cultivated area under food grain is the highest in Manipur (78.7%) followed by that in Nagaland and Arunachal Pradesh.

The NER has eight distinct agroclimatic zones starting from subtropical to temperate to alpine. Each state can be divided into agro-ecozones for agricultural planning and development. Rice is the main crop, and tropical and temperate fruits are also commonly grown. In Assam, Manipur, and Tripura, double-cropping of rice is practiced in plains, and fish farming is common in valleys. Tea (*Camellia sinensis*) husbandry is common in hills of Assam and Tripura. In low altitudes of Assam, Tripura, Manipur, and Meghalaya, rice, oilseeds, and vegetables are major crops. Rice, maize, ginger (*Zingiber officinale*), turmeric (*Curcuma longa*), and citrus (*Citrus* spp.) are important crops in hills. Maize and cardamom (*Elettaria cardamomum*) are important crops of Sikkim, which is the only organic state in India. In Tripura, there are three agro-ecoregions where double-cropping of rice is prevalent in plains. Pineapple (*Ananas comosus*), areca nut (*Areca catechu*), and vegetables are other important crops.

Soil Fertility and SOC Status

Most soils of the NER are rich in SOC and nitrogen (N) content, except those under shifting cultivation but are low to medium in P and medium to high in K. Soils of the humid subtropical regions (e.g., Tripura, Assam, and other foothills) are deficient in SOC and K. More than 85% of soils are acidic and have mild to severe acidity attributable to the leaching of basic cations due to heavy rainfall. P fixation is a problem in the soils, and substantial part of the P from applied phosphatic fertilizer gets immediately fixed and unavailable. Soils of the region are broadly classified under five orders, namely Inceptisols, Entisols, Alfisols, Ultisols, and Mollisols.^[4]

In Arunachal Pradesh, SOC concentration ranges from 14 g/kg in Dirang [1475–1775 m above mean sea level (AMSL)] to as high as 59 g/kg in Tawang (2700–2775 m AMSL). The SOC concentrations in other high altitude areas (1725–2300 m AMSL) like Bomdila (52.8 g/kg), Jerigaon (38.9 g/kg), Bomdir (34.5 g/kg), and Jung (48.7 g/kg) are also very high. In Manipur, SOC concentration ranges from 1–34 g/kg in Bishnupur valley to 5.1–36.6 g/kg in Imphal valley. Conversely, in the hill districts of Manipur such as Chandel (14.4–49.1 g/kg), Senapati (18.3–27.7 g/kg), Ukhrul (14.4–51.0 g/kg), Churachandpur

(15.2–26.8 g/kg), and Tamenglong (18.4–31.2 g/kg), the SOC concentration is very high. In Meghalaya, the SOC concentration is very high and ranges from 5.2 to 43.0 g/kg in East Khasi Hill and from 5.2 to 31.3 g/kg in West Khasi Hill districts. About 13% of soils in the valleys and 5.5% in the high altitude (>1700 m MSL) are low in SOC in Meghalaya. The soils of Mizoram are low to medium in SOC (4.2–13.4 g/kg) concentration. In Sikkim, the SOC concentration is high in most of the soils ranging from 23.5 to 44.5 g/kg. In high altitude areas of Sikkim (>1200 m AMSL), the SOC concentration is relatively higher in places like Gangtok (44.5 g/kg), Nagi (40.5 g/kg), Namthang (38.2 g/kg), and Turung (36.0 g/kg). However, SOC concentration is low in low altitude areas (<600 m MSL) of Sikkim like Singtam (23.5 g/kg), Rangpo (24.6 g/kg), Miming (29.2 g/kg), Majitar (26.0 g/kg), and Central Pendam (31.5 g/kg).^[4,5]

Under shifting cultivation, land is cleared off the existing vegetation, allowing the felled biomass to dry and finally burning the biomass in the month of March/April. Burning of aboveground biomass increases pH and cations and decreases carbon (C) and N contents in surface soils. The SOC content decreases after burning because of oxidative loss of carbon dioxide (CO₂). Soil loss to the tune of 40.9 Mg/ha and corresponding nutrient loss of 703 kg SOC, 63.5 kg P/ha, and 6 kg K/ha have been reported from shifting cultivation on steep slopes of 44–53%. Annual loss of topsoil, N, P, and K to the extent of 88,346, 10,669, 372, and 6051 Gg from the region had been reported, respectively.^[4]

SOC under Various Land Use Systems

Agricultural soils have large potential for SOC stock and sequestration and can be managed to reduce atmospheric concentration of CO₂. The SOC stock of different land uses including long-term experimental fields of Lembucherra, Tripura (<100 m AMSL); Umiam, Meghalaya (950–1000 m AMSL); and Tadong, Sikkim (>1500 m AMSL) was estimated at the following two depths: 0–15 and 15–30 cm. The objective was to recommend best soil management practices for SOC sequestration of different altitudes.

Lembucherra, Tripura (Low Altitude)

Soil sampling was done to analyze bulk density and SOC concentrations to estimate SOC stock from existing long-term land uses (>10 years) except for jatropha (*Jatropha curcas*) block that was 5 years old. In the studied soils of different land use systems of Lembucherra, Tripura, the SOC concentrations ranged from 8.6 to 16.4 g/kg at 0–15 cm and from 7.4 to 15.2 g/kg at 15–30 cm layers (Fig. 1). Among all the land uses, SOC stock was the maximum under bamboo (*Bambusa* spp.) plantations followed by that in *Tephrosia purpurea* alley cropping (AC) and duck-based farming

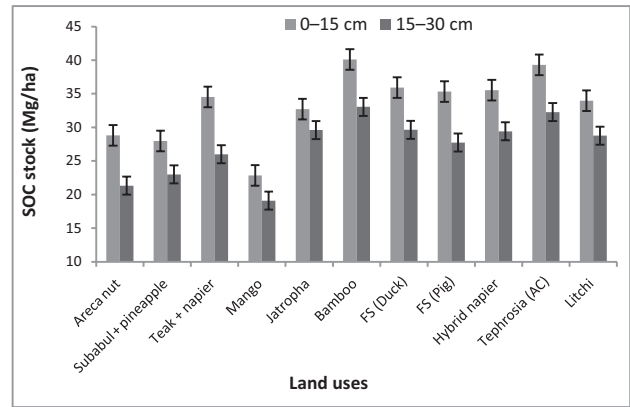


Fig. 1 SOC stock under different land uses in Lembucherra, Tripura.

system [FS (duck)]. Soils under mango (*Mangifera indica*) and arecanut had the lowest SOC stock.^[5]

Umiam, Meghalaya (Mid-Altitude)

At mid-altitude of Meghalaya, land uses comprising conventional tillage (CT), no-till (NT), recommended dose of fertilizer (RFD), organic farming (OF), *in situ* residue retention (IRR) in rice, maize, and pine (*Pinus kesiya*) forest (dominant forest), *Jhum*, and IFS [upper slope (US), middle slope (MS), and lower slope (LS) of hills] involving crop-livestock fodder on terrace risers were evaluated for SOC stocks. The SOC concentrations ranged from 16.4 to 34.7 g/kg in the 0–15 cm and 15.5 to 33.5 g/kg in the 15–30 cm layers (Fig. 2). Among all the land uses, SOC stock was recorded to the maximum under IRR in rice (10 years) in lowland followed by that in rice under OF. Pine forest had the lowest SOC stock among all the land uses. IFS involving crop-livestock (agropastoral) components had higher SOC stock in surface layer at US, MS, and LS than that under *Jhum* fields. Thus, IRR in lowland rice and the adaptation of IFS approaches are favorable for building soil fertility especially the SOC stocks.^[5]

Tadong, Sikkim (High Altitude)

In some soils of different land use systems of Tadong, Sikkim, the SOC concentrations vary from 17.6 to 29.2 g/kg in the 0–15 cm and 13.5 to 24.5 g/kg in the 15–30 cm layers (Fig. 3). Among all the land uses, SOC stock was the maximum in *Alnus nepalensis* system + cardamom (>20 years) and sal tree (*Shorea robusta*) + cardamom (>20 years) in the surface and subsurface layers, whereas the minimum SOC stock was recorded in soils under orange (*Citrus reticulata*) plantation (>15 years) in both the layers.

Impact of agroforestry, perennial forage crops, crop residue recycling and conservation agriculture^[6] on

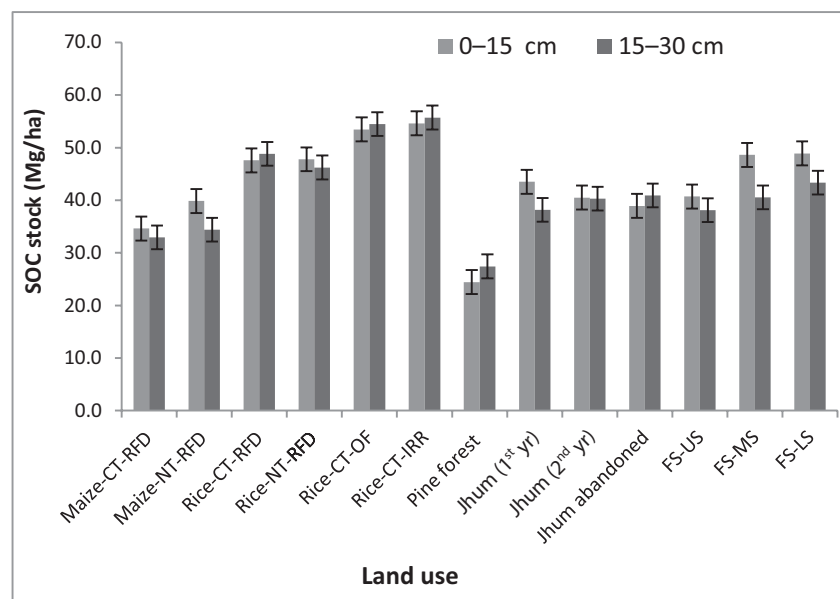


Fig. 2 SOC stock under different land uses in Umiam, Meghalaya.

improving soil health and C sequestration has been reported under the NER.

Some Approaches for Soil Health Management

Integrated farming system

The alternative farming systems (eight IFS models) to shifting cultivation were developed by Indian Council of Agricultural Research, Research Complex for North Eastern Hill Region, Umiam, in 1983. Soil physicochemical properties were assessed after 20 years. The maximum accumulation of exchangeable Ca^{2+} was observed under cropping system and exchangeable K under agri-horti-silvi-pastoral system.^[7] There was a substantial buildup of available N in soils under all eight systems. The rise in available P in agricultural livestock-based system could be due to heavy and continuous dressing of cow dung litter

for a long period. Available K increased in all systems except that in the livestock-based farming system. The overall fertility buildup followed the trend of agriculture > agri-horti-silvi-pastoral > livestock-based farming system. The improvement in overall soil fertility due to various farming systems over shifting cultivation-based land use has been reported.^[8]

Integrated nutrient management

The use of N, P, and K fertilizer in the NER is only 13.4%, 11.1%, and 11.1%, respectively, of the crop removal, indicating a perpetual depletion and low productivity. In most cases, only N fertilizer is applied and P and K are neglected. In case of Manipur, the ratio of N/P/K application is 76.9:7.5:1, respectively. Integrated use of chemical fertilizers of 60 kg N/ha along with 5 Mg/ha farmyard manure (FYM) with the seed inoculation of *Azotobacter* can enhance rice yield, nutrient uptake, and N buildup in soils.^[8] Similarly, the application of single super phosphate with *Rhizobium* inoculation in black gram produced 69.1% increase in nodulation over control in acidic soils of Tripura.

Agroforestry systems (AFSs)

In hill ecosystems, AFSs play an important role in sustainability, resource conservation, and food security. Multistoried AFSs/home gardens are the classic example of agroforestry. Agri-horticulture, agri-silviculture, agri-horti-silvi-pasture, agri-pisciculture, and multitier systems are some of the important AFSs in the NER. Tree species (e.g., *A. nepalensis* and *P. kesiya*) have favorable effects on soil physical properties, particularly on soil bulk density, moisture content, and mean weight diameter of soil

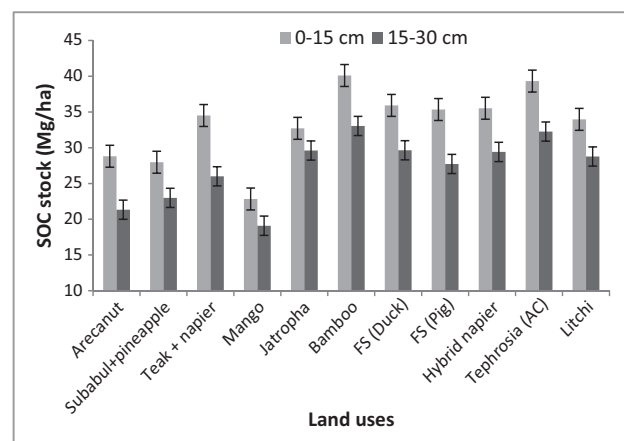


Fig. 3 SOC stock under different land uses in Lembucherra, Tripura.

aggregates. About 33%, 50%, and 76% increase in soil available N, P, and K content, respectively, has been observed under *A. nepalensis* as compared to control. *A. nepalensis* and *Michelia oblonga* considerably favored the accumulation of SOC and available nutrients compared to other tree species under mid hill condition of Meghalaya.^[9]

Conservation agriculture

The retention of crop residues on the soil surface in combination with NT initiates processes that lead to improved soil quality and overall enhancement of resource-use efficiency. Higher SOC and total N contents under NT and residue retention in rice–pea (*Pisum sativum*),^[6] rice–lentil (*Lens culinaris*),^[10] maize–rapeseed (*Brassica campestris* var. toria),^[5] and maize–French bean (*Phaseolus vulgaris*)^[11] have been reported from the diverse ecosystems of the NER. Increase in SOC and soil nutrient status due to residue management under groundnut (*Arachis hypogaea*)–rapeseed system is also reported.^[12]

Organic farming

The fertilizer consumption is less than 12 kg/ha (excluding Manipur and Tripura) in the NER. The total potential for nutrient supply through crop residues in the region has been estimated as 9.86, 2.12, and 35.5 Mg of N, P, and K, respectively, thus creating on an average 2.46 kg N, 0.53 kg P₂O₅, and 8.87 K₂O/ha. Thus, a good quantity of potash can be added through crop residues.^[4] Growing leguminous shrubs such as *Crotalaria juncea* and *Tephrosia purpurea* around the farm fences can produce 5–6 Mg/ha green leaf manure containing 2.4–2.7% N, 0.3–0.6% P, and 0.8–2.0% K. It is an added advantage that every family maintains some cattle, and thus, the NER has about 13 million cattle population. A 5-year study indicated that the application of recommended dose of N through FYM or FYM + vermicompost is equally effective as organic amendment and their continuous application improved soil health and crop productivity.^[13] Organic production of rice, pea, lentil, carrot (*Daucus carota*), French bean, tomato (*Solanum lycopersicum*), potato (*Solanum tuberosum*), and okra (*Abelmoschus esculentus*) can sustain soil health and productivity in the NER.^[11]

Liming

Acid soils occupy nearly 85% of geographical area in the NER. Nearly 65% of the soils of the region are extremely acidic (below 5.5). Crops are affected by aluminum and iron toxicity, deficiency, and low base saturation. The furrow application of lime 250–500 kg/ha of high quality uniform grades/sizes at furrows every year can optimize the yields of crops in acid soils of the NER. The use of acid tolerant varieties and the application of organic manure also improve the productivity of such soils. The optimum

yield can be obtained by applying liming materials at 0.25 LR (lime requirement) or 1–1.25 equivalent soil exchangeable aluminum.^[14]

Micronutrient management

Acid soils are also characterized by the deficiency of micronutrients, particularly of boron (B) and zinc (Zn). The deficiency of B is commonly observed in light-textured acidic soils receiving high precipitation, which cover about 35% of the NER, whereas Zn deficiency occurs in 34% of cultivated soils of Assam plains and in about 55% soils of the hilly areas of the NER. Direct and residual effects are observed in B and Zn application to crops grown in deficient (B and Zn) soils. Soil management such as liming and organic matter application increases the availability of B and Zn in acid soils. Residual effects of Zn, B, and lime or FYM amendments have also been observed. Micronutrient deficiency in acid soils may, therefore, be ameliorated through liming and application of organic manure.^[15]

Rainwater Management—An Integral Component of Sustainable Soil Management

Rainwater management in the NER is equally important for sustainable soil health and agriculture. The NER receives very high rainfall (>2000 mm annually). However, more than 75% of rain is received within a short span of 4–5 months (June to October). Later the rains are deficient, causing a scarcity of even drinking water in the highest rainfall receiving NER (Cherrapunjee–Mawsynram range of Meghalaya receives about 11,000 mm rainfall annually). Crop fortunes are linked to the behavior of rains, and quite often there are crop failures even for want of life-saving irrigation at the critical stage of crop growth as the irrigation development is almost nil. Under such situations, it is important to harvest the rainwater where it falls. Watershed approach, microrainwater harvesting structure (*Jalkund*), roof water harvesting, *in situ* rainwater management, etc. have been recommended for efficient soil and water management in sloping lands of the Himalayas.^[16,17]

CONCLUSION

The management approaches for the soils of the NER should be site-specific and as per the resources available to a farm community. Since the region is a mix of plain, valleys, and steep slopes, the problems and opportunities are also site-specific. Thus, integrated natural resource management comprising IFS, INM, and AFSs along with the adoption of principles of conservation agriculture, and rainwater management offers the most sustainable soil management approach for the NER.

REFERENCES

1. Singh, A.; Singh, M.D. *Soil Erosion Hazards in North Eastern Hill Region*, Research Bulletin No. 10; ICAR Research Complex for NEH Region: Meghalaya, 1981.
2. Ghosh, P.K.; Saha, R.; Gupta, J.J.; Ramesh, T.; Das, A.; Lama, T.D.; Munda, G.C.; Bordoloi, J.S.; Verma, M.R.; Ngachan, S.V. Long-term effect of pastures on soil quality in acid soil of North-East India. *Aust. J. Soil Res.* **2009**, *47*, 372–379.
3. FSI 2009. *Forest Survey of India*. India State of Forest Report, Ministry of Environment and Forests, Government of India, Dehradun India.
4. Sharma, U.C.; Datta, M.; Samra, J.S. *Soils and Their Management in North East India*; ICAR Research Complex for NEH Region: Umiam, 2006; 122–170.
5. Das, A.; Munda, G.C.; Ghosh, P.K.; Ramkrushna, G.I.; Patel, D.P.; Choudhury, B.U.; Mandal, S.; Ngachan, S.V.; Layek, J. Managing degraded land and soil organic carbon for climate resilient agriculture in North East India. In *Soil Carbon Sequestration for Climate Change Mitigation and Food Security*; Srinivasa Rao, Ch., Venkateswarlu, B., Srinivas, K., Sumanta, K., Singh, A.K., Eds.; Central Research Institute for Dry land Agriculture: Hyderabad, 2011; 241–265.
6. Ghosh, P.K.; Das, A.; Saha, R.; Enboklang, K.; Tripathi, A.K.; Munda, G.C.; Ngachan, S.V. Conservation agriculture towards achieving food security in North East India. *Curr. Sci.* **2010**, *99* (7), 915–921.
7. Singh, A.K.; Munda, G.C.; Ngachan, S.V.; Panwar, A.S.; Ghosh, P.K.; Das, A.; Patel, D.P.; Choudhury, B.U.; Tripathi, A.K.; Mohapatra, K.P. *Natural Resource Management for Sustainable Hill Agriculture – Need for a Paradigm Shift*; ICAR Research Complex for NEH Region: Umiam, 2012; 268 pp.
8. Majumdar, B.; Kumar, K.; Saha, R.; Satapathy, K.K. Patiram. Different forms of nitrogen as affected by bun system of cultivation on hill slopes of Meghalaya. *Indian J. Hill Farm.* **2004**, *17* (1–2), 9–14.
9. Ramesh, T.; Manjaiah, K.M.; Tomar, J.M.S.; Ngachan, S.V. Effect of multipurpose tree species on soil fertility and CO₂ efflux under hilly ecosystems of Northeast India. *Agroforest. Syst.* **2013**, *87*, 1377–1388.
10. Das, A.; Patel, D.P.; Ramkrushna, G.I.; Munda, G.C.; Ngachan, S.V.; Buragohain, J.; Kumar, M.; Naropongla. Crop diversification, crop and energy productivity under raised and sunken beds: Results from a seven-year study in a high rainfall organic production system. *Biol. Agr. Hortic.* **2014**, *30* (2), 73–87.
11. Das, A.; Ramkrushna, G.I.; Choudhury, B.U.; Ngachan, S.V.; Tripathi, A.K.; Singh, R.K.; Patel, D.P.; Tomar, J.M.S.; Mohapatra, K.P.; Layek, J.; Munda, G.C. Conservation agriculture in rice and maize based cropping systems for enhancing crop and water productivity – Participatory technology demonstration in north east India. *Indian J. Soil Conserv.* **2014**, *42* (2), 196–203.
12. Kuotsu, K.; Das, A.; Lal, R.; Munda, G.C.; Ghosh, P.K.; Ngachan, S.V. Land forming and tillage effects on soil properties and productivity of rainfed groundnut (*Arachis hypogaea* L.) – Rapeseed (*Brassica campestris* L.) cropping system in northeastern India. *Soil Till. Res.* **2014**, *142*, 15–24.
13. Patel, D.P.; Das, A.; Kumar, M.; Munda, G.C.; Ngachan, S.V.; Ramakrishna, G.I.; Layek, J.; Naropongla, B.J.; Somirreddy, U. Continuous application of organic amendments enhance soil health, produce quality and system productivity of vegetable based cropping systems at subtropical eastern Himalayas. *Exp. Agr.* **2014**, *51* (1), 85–106.
14. Patiram. Furrow application of lime is economical to rectify the acid soils of Sikkim. *Indian Farm.* **1994**, *5*, 21.
15. Sarkar, D.; Mandal, B.; Kundu, M.C. Increasing use efficiency of boron fertilizers by rescheduling the time and methods of application for crops in India. *Plant Soil* **2007**, *301*, 77–85.
16. Saha, R.; Ghosh, P.K.; Mishra, V.K.; Bujarbaruah, K.M. Low-cost micro-rainwater harvesting technology (Jalkund) for new livelihood of rural hill farmers. *Curr. Sci.* **2007**, *92*, 1258–1265.
17. Acharya, C.L.; Kapur, O.C.; Sharma, S. *Management of Rain Water and Other Natural Water Resources for Sustainable Hill Agriculture*, Proceedings of National Seminar on Water Management for Sustainable Agriculture – Problems and Perspectives for the 21st Century, New Delhi, April 15–17, 1998, Indian Agricultural Research Institute, Water Technology Centre; Indian Society of Water Management: New Delhi, 249–254.

Macroporosity

Daniel Gimenez

Department of Environmental Sciences, Rutgers University, New Brunswick, New Jersey, U.S.A.

Daniel Hirmas

Department of Geography and Atmospheric Science, University of Kansas, Lawrence, Kansas, U.S.A.

Abstract

Macropores constitute a small volumetric fraction of the total porosity of soils but profoundly influence both the movement of water when soils are at saturated or near saturated states and the flux of gases when soils are at moisture states less than saturation. Macropores are recognized as one of the causes of preferential flow, a phenomenon that continues to defy our ability to predict transport processes in field soils. This entry reviews the most salient properties of soil macropores, including their origin and morphological properties, field and laboratory characterization methods, and strategies used to model flow of water through macroporous soils. Applications of geophysical and multistripe laser triangulation techniques to map macropores in the field, together with the increasing availability of high-resolution 3-D computer-aided tomography, are promising developments that could help improve the theoretical foundation of transport processes in soils containing macropores.

INTRODUCTION

Macroporosity refers to the areal or volumetric proportion of macropores in soils. The term macropore conveys connotations of rapid pathways for water, chemicals, and gases in soils. The Glossary of Soil Science Terms^[1] defines macropores as pores with an equivalent cylindrical diameter larger than 75 μm ; however, the range of sizes found in the literature for defining macropores varies between 30 and 3,000 μm .^[2] The hydrological properties of macropores are not only dependent on size but also on their continuity, tortuosity, and shape.^[3,4]

Densities of macropores reported in the literature vary from 10 to 10,000 pores/ m^2 ,^[5–7] and the distribution of their sizes is highly skewed, with the number of pores decreasing exponentially as their size increases.^[5] Larger pores tend to have elongated (planar or tubular) shapes whereas smaller ones are more rounded.^[8] Macropore density and continuity decrease with depth and are usually interrupted by tillage.^[7]

ORIGIN OF MACROPORES

Macropores can originate from cultivating a soil, fauna activity, root decay, shrinking of a soil matrix upon drying, or dissolution of chemical substances. Aggregation may result in a bimodal distribution of pores, where the larger pores are located in between aggregates (interaggregate pores). Tillage operations result in the formation of

structurally unstable macropores that subside during the growing season, thus altering the distribution of pore sizes in a very short period of time.^[9,10] Among soil fauna, earthworms have received much attention because of their abundance and potential impact in creating preferential flow pathways in soil,^[6,11] but fossorial fauna can also create extensive systems of burrows affecting large areas.^[12,13] Another source of biopores is plant roots, which can form an intricate network of macropores influencing water movement and chemical transport.^[14,15] Taproot systems are particularly effective in providing stable and continuous pores that increase infiltration rates.^[16] Shrinking of a soil matrix upon drying forms planar macropores that are important for transport processes in initially dry soils.^[4]

QUANTIFICATION OF MACROPORE PROPERTIES

Direct quantification of macropore abundance (number and volumes), sizes, and spatial distribution in the field is achieved by tracing or photographing macropores on planes parallel or perpendicular to the soil surface.^[7] Dyes are sometimes used to enhance the identification of continuous macropores.^[17] An alternative to the use of dyes is to cast the macropores with a viscous fluid that hardens with time and to subsequently excavate and quantify the hardened cast.^[18] Materials used to cast pores include epoxy, plaster, wax, and molten metal.^[19,20] Multistripe laser triangulation (MLT) scanning has been applied to quantify



Fig. 1 MLT scanner (foreground) being used to digitize macropores in the bottom portion of an excavation wall. The parallel laser stripes projected from the scanner can be seen on the pit surface in the background. For scale, each laser stripe is approximately 40 cm in length.

interpedal macropores from vertical walls exposed by excavation.^[21] As the lasers sweep across the exposed surface (Fig. 1), interpedal macropores act as “traps” of the signal, creating data gaps in the digital surface mesh that can be projected in 2-D and used to map and quantify the abundance, orientation, distribution, size, and shape of macropores. Non-invasive geophysical methods have also been applied to map subsurface macropores, but detection is limited to macropores of several centimeters in diameter.^[22,23] Laboratory techniques are restricted in the soil volume that can be evaluated but are capable of producing high-resolution images. Among these techniques are the use of resin-impregnated soil^[24] and X-ray and gamma-ray computed tomography.^[25,26] The number of studies using the latter technology has increased steadily since the early 1990s, as the instruments have become more accessible to soil scientists.

Indirect quantification of macropores has been made through measurements of infiltration at various suctions and of air permeability.^[27] These methods require defining a lower limit of macropore radius, which is typically influenced by the methodology used.

IMPACT OF MACROPORES ON WATER MOVEMENT AND CHEMICAL AND GAS TRANSPORT

Typically, more than 70% of the total water moving through a soil is transmitted through pores that represent less than

1% of a soil volume.^[28] The effect of macropores on water movement through soil was recognized in the late 19th century but received increased attention during the 1980s because of the risk macropores pose to contamination of subsurface water.^[2,3] In the presence of macropores, matter can bypass the soil matrix. For instance, heavy rainfalls occurring shortly after a chemical application can leach below the root zone up to 5% of the applied mass in sandy and loamy soils,^[29] but deep transport of chemicals has also been observed in heavy clay soils.^[30] Flow regimes influenced by macropores have been studied in relation to the transport of chemicals, microorganisms, and colloidal particles through soils,^[31–33] using both undisturbed soil^[26,33] and packed soil containing artificial macropores.^[34] Maximum flow is obtained under ponded conditions and with macropores connected to the surface, but macropore flow can also occur under a non-ponded regime once the macropore walls are wet.^[3]

Macropores also impact the exchange with the atmosphere of gases that are produced or consumed in soils as result of nutrient cycling (e.g., nitrogen and carbon). The effect of macropores on gas transport is more pronounced in soils with water contents near saturation. Under those conditions, only macropores are air filled and provide open pathways for gas exchange in the form of diffusion or mass flow. For instance, the flush release of methane following postharvest drainage of rice paddies is facilitated by the presence of macropores that allow gases to bypass the water-saturated soil matrix. Experimental observations in packed and undisturbed soils indicate that diffusion coefficients in soil around macropores are several fold greater than the average diffusion coefficient of the soil matrix.^[35]

Modeling Transport Processes in Macroporous Soils

Prediction of transport processes through macroporous soils is complicated by the intricate geometry of macropores. Laminar flow of water through pores with cylindrical and planar geometries can be theoretically estimated using Poiseuille’s equation modified to account for the finite nature of planar voids^[36] and the turbulent or transitional nature of flow through macropores.^[37,38] Preferential flow has been modeled as films of uniform thickness flowing along the surface of macropores as laminar flow.^[39] Prediction of flow through natural macropores is, however, highly complicated by their irregular shape. The use in flow equations of measured diameter, length, and volume of macropores does not satisfactorily predict macropore hydraulic properties,^[37] probably because the smallest section of the irregular macropores has a disproportionate effect on the flow of water through macropores and, therefore, a potential to influence geometrical estimations from flow measurements. The prediction of flow through macropores may be aided by studies replicating the complex geometries of

these pores through the use of rapid fabrication techniques such as 3-D printing.^[40,41]

Conceptual models of flow through macroporous soils divide the soil matrix in two or more domains with or without exchange of mass among them.^[3,4] The experimental determination of the relatively large number of input parameters needed to simulate multiple flow domains is a significant challenge for the application of these models.^[42] A more pragmatic approach to modeling water movement and gas transport in macroporous soils is to work with the concept of an effective porosity (i.e., the proportion of a soil volume occupied by macropores) coupled with relative simple relationships to estimate transport coefficients. Effective porosity can be determined in a number of ways from areal macroporosity mapped in the field to laboratory measurements of air-filled porosity at an arbitrary pressure potential close to saturation. Power law relationships between effective porosity and saturated hydraulic conductivity or air permeability can be used to predict these properties or to compare the characteristics of macropore systems.^[43,44] Similarly, Eck et al.^[45] coupled the size of macropores determined from MLT scanning with coefficient of linear extensibility to predict saturated hydraulic conductivity. These approaches are promising and require minimal data but do not provide information on mechanistic relationships between macropore morphology and water flow.

CONCLUSION

Macropores are ubiquitous in soils, and despite their minuscule volumetric presence, their influence on soil hydrology and gas exchange is profound. Their intricate geometry and multiple origins complicate any attempt to generalize their effect on water movement and chemical transport through soil. Despite the complexities of transport processes in the presence of macropores, advances in measurements techniques and in conceptual models have improved our ability to simulate transport of matter through soils with macropores.

REFERENCES

1. Soil Science Society of America. *The Glossary of Soil Science Terms*; Soil Science Society of America: Madison, WI, 2008. Available at <https://www.soils.org/publications/soils-glossary#> (accessed June 30, 2015).
2. White, R.E. The influence of macropores on the transport of dissolved and suspended matter through soil. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer: New York, 1985; 95–120.
3. Jarvis, N.J. A review of non-equilibrium water flow and solute transport in soil macropores: Principles, controlling factors and consequences for water quality. *Eur. J. Soil Sci.* **2007**, *58* (3), 523–546.
4. Köhne, J.M.; Köhne, S.; Šimůnek, J. A review of model applications for structured soils: (a) Water flow and tracer transport. *J. Contam. Hydrol.* **2009**, *104* (1–4), 4–35.
5. Smettem, K.R.J.; Collis-George, N. Statistical characterization of soil biopores using a soil peel method. *Geoderma* **1985**, *36* (1), 27–36.
6. Edwards, W.M.; Norton, L.D.; Redmond, C.E. Characterizing macropores that affect infiltration into non-tilled soils. *Soil Sci. Soc. Am. J.* **1988**, *52* (2), 483–487.
7. Logsdon, S.D.; Allmaras, R.R.; Wu, L.; Swan, J.B.; Randall, G.W. Macroporosity and its relation to saturated hydraulic conductivity under different tillage practices. *Soil Sci. Soc. Am. J.* **1990**, *54* (4), 1096–1101.
8. Pachepsky, Y.; Yakovchenko, V.; Rabenhorst, M.C.; Pooley, C.; Sikora, L.J. Fractal parameters of pore surfaces derived from micromorphological data: Effect of long-term management practices. *Geoderma* **1996**, *74* (3–4), 305–319.
9. Ahuja, L.R.; Fiedler, F.; Dunn, G.H.; Benjamin, J.G.; Garrison, A. Changes in soil water retention curves due to tillage and natural reconsolidation. *Soil Sci. Soc. Am. J.* **1998**, *62* (5), 1228–1233.
10. Leij, F.J.; Ghezzehei, T.A.; Or, D. Modeling the dynamics of the soil pore-size distribution. *Soil Till. Res.* **2002**, *64* (1–2), 61–78.
11. Lee, K.E.; Foster, R.C. Soil fauna and soil structure. *Aust. J. Soil Res.* **1991**, *29* (6), 745–775.
12. Whitford, W.G.; Kay, F.R. Bioperturbation by mammals in deserts: A review. *J. Arid Environ.* **1999**, *41* (2), 203–230.
13. Gabet, E.J.; Reichman, O.J.; Seabloom, E.W. The effects of bioturbation on soil processes and sediment transport. *Annu. Rev. Earth Pl. Sc.* **2003**, *31*, 249–273.
14. Li, Y.; Ghodrati, M. Preferential transport of nitrate through soil columns containing root channels. *Soil Sci. Soc. Am. J.* **1994**, *58* (3), 653–659.
15. Ghestem, M.; Sidle, R.C.; Stokes, A. The influence of plant root systems on subsurface flow: Implications for slope stability. *Bioscience* **2011**, *61* (11), 869–879.
16. Mitchell, A.R.; Ellsworth, T.R.; Meek, B.D. Effect of root systems on preferential flow in swelling soil. *Commun. Soil Sci. Plan.* **1995**, *26* (15–16), 2655–2666.
17. Droogers, P.; Stein, A.; Bouma, J.; de Boer, G. Parameters for describing soil macroporosity derived from staining patterns. *Geoderma* **1998**, *83* (3–4), 293–308.
18. Abou Najm, M.R.; Jabro, J.D.; Iversen, W.M.; Mohtar, R.H.; Evans, R.G. New method for the characterization of three-dimensional preferential flow paths in the field. *Water Resour. Res.* **2010**, *46* (2), W02503.
19. Hasiotis, S.T.; Bourke, M.C. Continental trace fossils and museum exhibits: Displaying organism behavior frozen in time. *Geol. Curator* **2006**, *8* (5), 211–226.
20. Tschinkel, W.R. Methods for casting subterranean ant nests. *J. Insect Sci.* **2010**, *10*, 88.
21. Eck, D.V.; Hirmas, D.R.; Giménez, D. Quantifying soil structure from field excavation walls using multistripe laser triangulation scanning. *Soil Sci. Soc. Am. J.* **2013**, *77* (4), 1319–1328.
22. Gormally, K.H.; McIntosh, M.S.; Mucciardi, A.N.; McCarty, G.W. Ground-penetrating radar detection and three-dimensional mapping of lateral macropores: II. Riparian application. *Soil Sci. Soc. Am. J.* **2011**, *75* (4), 1236–1243.

23. Leslie, I.N.; Heinse, R. Characterizing soil-pipe networks with pseudo-three-dimensional resistivity tomography on forested hillslopes with restrictive horizons. *Vadose Zone J.* **2013**, *12* (4), DOI:10.2136/vzj2012.0200.
24. Ringrose-Voase, A.J. Measurement of soil macropore geometry by image analysis of sections through impregnated soil. *Plant Soil* **1996**, *183* (1), 27–47.
25. Warner, G.S.; Nieber, J.L.; Moore, I.D.; Geise, R.A. Characterizing macropores in soil by computed tomography. *Soil Sci. Soc. Am. J.* **1989**, *53* (3), 653–660.
26. Perret, J.; Prasher, S.O.; Kantzas, A.; Langford, C. Three-dimensional quantification of macropore networks in undisturbed soil cores. *Soil Sci. Soc. Am. J.* **1999**, *63* (6), 1530–1543.
27. Edwards, W.M.; Shipitalo, M.J.; Owens, L.B. Gas, water, and solute transport in soils containing macropores: A review of methodology. *Geoderma* **1993**, *57* (1–2), 31–49.
28. Watson, K.W.; Luxmoore, R.J. Estimating macroporosity in a forest watershed by use of a tension infiltrometer. *Soil Sci. Soc. Am. J.* **1986**, *50* (3), 578–582.
29. Flury, M. Experimental evidence of transport of pesticides through field soils—A review. *J. Environ. Qual.* **1996**, *25* (1), 25–45.
30. Kelly, B.P.; Pomes, M.L. Preferential flow and transport of nitrate and bromide in claypan soil. *Ground Water* **1998**, *36* (3), 484–494.
31. Czapar, G.F.; Horton, R.; Fawcett, R.S. Herbicide and tracer movement in soil columns containing an artificial macropore. *J. Environ. Qual.* **1992**, *21* (1), 110–115.
32. Murphy, S.L.; Tate, R.L. Bacterial movement through soil. In *Soil Biochemistry*; Stotzky, G., Bollag, J.-M., Eds.; Marcel Dekker: New York, 1996; 253–286.
33. Jacobsen, O.H.; Moldrup, P.; Larsen, C.; Konnerup, L.; Petersen, L.W. Particle transport in macropores of undisturbed soil columns. *J. Hydrol.* **1997**, *196* (1–4), 185–203.
34. Arora, B.; Mohanty, B.P.; McGuire, J.T. Inverse estimation of parameters for multidomain flow models in soil columns with different macropore densities. *Water Resour. Res.* **2011**, *47*, W04512.
35. Allaire, S.E.; Lafond, J.A.; Cabral, A.R.; Lange, S.F. Measurement of gas diffusion through soils: Comparison of laboratory methods. *J. Environ. Monitor.* **2008**, *10* (11), 1326–1336.
36. Towner, G.D. Formulae for calculating water flow in macropores in soil. *Int. Agrophys.* **1987**, *3*, 1–5.
37. Wang, D.; Norman, J.M.; Lowery, B.; McSweeney, K. Nondestructive determination of hydrogeometrical characteristics of soil macropores. *Soil Sci. Soc. Am. J.* **1994**, *58* (2), 294–303.
38. Logsdon, S.D. Flow mechanisms through continuous and buried macropores. *Soil Sci.* **1995**, *160* (4), 237–242.
39. Nimmo, J.R. Theory for source-responsive and free-surface film modeling of unsaturated flow. *Vadose Zone J.* **2010**, *9* (2), 295–306.
40. Otten, W.; Pajor, R.; Schmidt, S.; Baveye, P.C.; Hague, R.; Falconer, R.E. Combining X-ray CT and 3D printing technology to produce microcosms with replicable, complex pore geometries. *Soil Biol. Biochem.* **2012**, *51*, 53–55.
41. Bacher, M.; Schwen, A.; Koestel, J. Three-dimensional printing of macropore networks of an undisturbed soil sample. *Vadose Zone J.* **2015**, *14* (2), DOI:10.2136/vzj2014.08.0111.
42. Larsbo, M.; Jarvis, N. Simulating solute transport in a structured field soil: Uncertainty in parameter identification and predictions. *J. Environ. Qual.* **2005**, *34* (2), 621–634.
43. Rawls, W.J.; Brakensiek, D.L.; Logsdon, S.D. Predicting saturated hydraulic conductivity utilizing fractal principles. *Soil Sci. Soc. Am. J.* **1993**, *57* (5), 1193–1197.
44. Rosenberg, R.J.; McCoy, E.L. Tillage- and traffic-induced changes in macroporosity and macropore continuity: Air permeability assessment. *Soil Sci. Soc. Am. J.* **1992**, *56* (4), 1261–1267.
45. Eck, D.V.; Qin, M.; Hirmas, D.R.; Giménez, D.; Brunsell, N. Relating quantitative soil structure metrics to saturated hydraulic conductivity. *Vadose Zone J.* **2016**, *15* (1), DOI: 10.2136/vzj2015.05.0083.

Magnesium

N.S. Bolan

*Institute of Natural Resources, Soil and Earth Sciences, Massey University,
Palmerston North, New Zealand*

K. Arulmozhiselvan

*Department of Soil Science and Agricultural Chemistry, Tamil Nadu Agricultural
University, Coimbatore, India*

P. Paramasivam

Tamil Nadu Agricultural University, Coimbatore, India

Abstract

Among the elements that constitute the solid surface of the earth, magnesium (Mg) is the eighth most abundant and an essential plant nutrient. Mg, the central component of chlorophyll, plays a major role in plant nutrition. Although Mg is regarded as a macronutrient, it is not always considered in fertilizer management possibly because Mg is added to soil as an accessory element in many fertilizers. Increasing incidences of Mg deficiency are being observed mainly as a result of the use of Mg-free fertilizers. Furthermore, modern agricultural practices are inducing accelerated soil acidification. If uncorrected, this acidification will lead to a reduction in exchangeable basic nutrient cations, leading to the increasing incidence of Mg deficiency in soils. The role of Mg in plants and animals is discussed in relation to the supply of Mg in soils.

MAGNESIUM (Mg) IN SOILS

Mg is a normal component of both igneous and sedimentary rocks and the soils developed from such rocks.^[1] Soils developed from basic rocks (basalts and limestone) generally contain higher levels of Mg (0.27–2.86% Mg) than those developed from granite and sandstones (0.01–0.34% Mg).^[2] The bulk of Mg in soil is usually in forms that are not readily available to the plant and exists in primary minerals and secondary silicate clay minerals.^[3] Some of the important silicate mineral species carrying Mg are shown in [Table 1](#). The formulas presented represent the “ideal” composition of the minerals. In nature, however, the composition varies, depending on the extent to which elements substitute for each other in the crystalline structure of these mineral species.

Soil Mg is often subdivided into soluble, rapidly exchangeable, slowly exchangeable, and structural (mineral) forms.^[4] These arbitrary subdivisions account for differences in bioavailability.^[5] The availability of Mg in soils for plant uptake is as follows: solution > exchangeable > mineral. There is an equilibrium between the forms of Mg that allows the release of Mg from less-available forms to the more-available forms. Only a small fraction of the total Mg is present in soil solution,

and plants absorb Mg from soil solution, which is buffered by the readily exchangeable forms.

Most soils hold important reserves of Mg in primary and secondary minerals.^[5,6] These reserves contribute a large proportion of the Mg needs of annual and perennial crops.^[7] The rate of release of Mg for plant uptake depends on the rate of weathering of these minerals. Studies on the kinetics of Mg release from soils and soil fractions using cation exchange resins, dilute salts, and organic acids showed that as weathering proceeds, the slowly exchangeable Mg originates from progressively coarser particle-size fractions.^[8] The rate of release of slowly exchangeable Mg increases with a decrease in particle size. Further, the presence of surface coatings on a mineral surface acts as a semipermeable barrier, reducing the rate of mineral weathering and, consequently, the rate of Mg release to soil solution.^[9]

Mg IN PLANTS

Mg forms the central unit of chlorophyll in plant leaves and cannot be substituted by other metals. Mg is involved in the production of starch during photosynthesis. Because Mg is a component of chlorophyll, an insufficient supply reduces chlorophyll formation, which

Table 1 Mg minerals in soils.

Mg minerals	Chemical formula	Mg content (g Mg/kg)
Forsterite	Mg ₂ SiO ₄	320–350
Pyrope	3MgO · Al ₂ O ₃ · 3SiO ₂	60–130
Iolite	H ₂ (Mg, Fe) ₄ Al ₈ Si ₁₀ O ₃₇	50–80
Diopside	CaMg(SiO ₃) ₂	20–140
Augite	CaMg(SiO ₃) ₂	45–100
Enstatite	MgSiO ₃	180–220
Actinolite	Ca(Mg, Fe) ₃ Si ₄ O ₁₂	100–160
Hornblende	CaMg metasilicate	10–90
Serpentine	H ₄ Mg ₃ Si ₂ O ₉	19–26
Talc	H ₂ Mg ₃ Si ₄ O ₁₂	160–200
Phlogopite	H ₃ Mg ₃ Al(SiO ₄) ₃	130–180
Biotite	(H,K) ₂ (Mg, Fe) ₂ Al ₂ Si ₃ O ₁₂	10–160
Clinochlore	H ₈ (Mg, Fe) ₅ Al ₂ Si ₃ O ₁₈	100–120

Source: Adapted from Beeson.^[2]

is likely to affect the photosynthetic ability of the plants. Mg has a controlling action on the swelling of plasma and plays a major role in many enzyme functions in plants.^[10]

Mg deficiency symptoms generally show up rather clearly in plant leaves, and Mg is readily mobile in plants. It moves from the older leaves to the younger leaves when Mg is deficient, and the deficiency symptoms therefore show first on the older leaves as Mg is withdrawn from them. The deficiency is initially characterized by an interveinal chlorosis; although in acute stages, the leaf may be generally deficient in both green and yellow pigments (variegated coloration), and necrosis may occur in the areas of the leaf first affected by Mg deficiency. A deficiency of Mg also induces the formation of anthocyanins in some plant species, such as cotton (*Gossypium hirsutum*). Economically, one of the most important Mg deficiency diseases is identified in tobacco (*Solanum tobaccum*), commonly referred to as “sand drown.” This occurs when tobacco leaves contain less than 0.25% Mg on a dry-weight basis. Similarly, Mg deficiency in pasture leads to “grass tetany” in grazing animals. The deficiency of Mg in pasture occurs when the concentration is less than approximately 0.2%.^[11]

Mg deficiency in plants generally occurs in soils that are low in exchangeable Mg, light textured, highly leached, acidic, and low in cation exchange capacity (CEC).^[12] In addition, induced deficiency may occur on some soils as a result of nutrient imbalances. Soils that are heavily fertilized, particularly with materials lacking in Mg or high in calcium (Ca), potassium (K), and methane (NH₄), can also induce Mg deficiency.^[13] By growing crops that require high levels of Mg

throughout the growing season (e.g., tobacco), Mg deficiency in soils may be exacerbated.^[12,14]

Mg IN ANIMALS

Mg plays an important role in the enzymatic metabolism of carbohydrates, lipids, proteins, and nucleic acids. Generally, Mg is an activator for numerous enzymes, such as phosphatases. It is also involved in the nerve conduction and muscular contraction.

Deficiency of Mg in blood plasma causes a disorder known as hypomagnesemia (grass tetany or staggers), which usually occurs in dairy cows in the early part of lactation.^[15] Mg deficiency in animals can be overcome by the regular use of Mg salts as drench or water trough treatment, as a lick, or in pasture after foliar application.

The application of excessive amounts of K fertilizer and K-rich farm effluents has been shown to increase the incidence of Mg deficiency leading to grass stagger. In pasture and fodder crops, grass stagger index (GSI; Eq. 1) is used to predict the occurrence of Mg deficiency.

$$\text{GSI} = [\text{K}^+]/([\text{Ca}^{2+}] + [\text{Mg}^{2+}]) \quad (1)$$

where $[\text{K}^+, \text{Ca}^{2+}, \text{Mg}^{2+}]$ = milliequivalents/kg-DM.

It has been suggested that GSI values in the pasture in excess of 2.2 enhance the risk of grass stagger development. This condition is generally linked with animal serum Mg levels less than 1.0–1.5 mg/100 ml, compared with normal levels in cattle of 1.7–3.0 mg/100 ml. This condition arises in response to diets low in Mg- or K-induced inhibition of Mg absorption in the rumen.^[15]

Mg FERTILIZERS

Mg deficiency in soils can be overcome by adding Mg-containing compounds (Table 2).^[16] Epsom salt (MgSO₄) is soluble in water and used as a fast-release Mg source. Other fertilizers are insoluble in water and are used as a slow-release source. The manufacture and use of serpentine superphosphate as a fertilizer material are declining. Dolomite, which contains both Ca and Mg, is more effective in acidic soils because the Mg is brought into solution by the acidic soil. Dolomite is the most widely used source of Mg, both as an ingredient of mixed fertilizers and as a separate amendment for liming.

All the slightly soluble Mg fertilizers listed in Table 2 are acid-neutralizing materials and would thus reduce the acid-forming potential of the fertilizer to which they might be added. Mg silicates, including magnesites, are ineffective sources of Mg for plants. Selectively calcined dolomite, in which the Mg component is oxidized to MgO, is more reactive than dolomite. Calcined magnesite can be added to soils or dusted onto pasture as a source of Mg.

Table 2 Mg fertilizers.

Mg minerals	Chemical formula	Mg solubility (mmol/liter)	Solubility product (pK _{SP})	Mg content (g Mg/kg)
Dolomite	MgCO ₃ · CaCO ₃	0.038	17.09	100
Calcined dolomite	MgO · CaCO ₃			160
Hydrated dolomite	MgO · Ca(OH) ₂			170
Magnesite	MgCO ₃	0.076	8.24	260
Brucite	Mg(OH) ₂	0.091	11.41	360
Magnesia	MgO	0.150		560
Kieserite	MgSO ₄ · H ₂ O	4,943		160
Epsom salt	MgSO ₄ · 7H ₂ O	127.3		90
Kainite	MgSO ₄ · KCl · 3H ₂ O			70
Langbeinite	2MgSO ₄ · K ₂ SO ₄			110
Forsterite	Mg ₂ SiO ₄	0.067 × 10 ⁻³	28.11	320–350

Source: Adapted from Augustin.^[16]

FACTORS AFFECTING Mg AVAILABILITY TO PLANTS

Plants differ markedly in their response to Mg deficiency in soil. The input and losses of Mg, forms of Mg in soils, and the interaction of Mg with other cations are some of the important properties affecting the plant availability of Mg.^[17] Mg concentration in plants is depressed by high concentrations of other cations, such as Ca, K, NH₄, and aluminum.^[18] Field trials have indicated that continuous use of ammonium-based fertilizers such as diammonium phosphate along with muriate of potash (KCl) results in Mg deficiency in pasture.^[11] The addition of KCl decreases the concentration of exchangeable Mg in soils, which is attributed mainly to the displacement of Mg by K ions added through the fertilizer.

Lime-induced reductions in tissue Mg level and Mg uptake by plants have been observed by a number of researchers.^[19] This effect occurs following an increase in Mg adsorption resulting from an increase in pH-dependent adsorption sites in soils containing variable charge components. Liming has also been shown to increase the leaching potential of Mg mainly because the Ca added through lime exchanges with Mg on the soil surfaces leading to the leaching of Mg in soil solution.^[20]

A number of soil test methods are used to predict Mg availability to plants, including 1N NH₄OAc-exchangeable Mg, percentage CEC saturated with Mg, and indexes of relative Mg concentration in soil solution (pMg in soil solution, pMg–pCa in solution, and half pMg–pK in solution).^[21] The ability of these indexes to predict the

availability of Mg to plants varies depending on the relative concentration of Mg, Ca, and K in soil solution.^[22] Field calibrations for the Mg soil test are available that enable the soil testing service to predict the Mg status of the soils and to make necessary Mg fertilizer recommendations.

CONCLUSION

As the central unit of chlorophyll, Mg is one of the most important plant nutrients. Mg is usually added as an accessory element along with other fertilizer materials. Plants take up Mg from soil solution, which is replenished through the continuous release from the exchange sites. Epsom salt is one of the most important soluble Mg sources added as a fertilizer material. Dolomite is a sparingly soluble material and is used extensively both as a source of Mg and as a liming material.

Excessive amounts of K in fertilizers and farm effluents generally decrease the uptake of basic cations, especially Ca and Mg by pasture. This is mainly the result of depletion of these cations in soil solution that results from leaching because of the competition from K ions for the cation exchange sites of the soil. To reduce the deficiency of Mg in grazing animals, it is important that this nutrient is added to soils, which receive regular effluent irrigation or K fertilizer application.

The timing of the application of K-based fertilizers to pasture soils is important in regulating the supply of Mg for dairy cattle. Increased Mg requirements at the end of pregnancy and in early lactation periods need to be supplemented by drenching with Mg salts. The Mg requirement of dairy cattle can be regulated by adjusting the dietary cation–anion balance (DCAB). An acid DCAB in the animal ration is likely to overcome grass tetany disorders in dairy cattle caused by Mg deficiency.

REFERENCES

1. Aitken, R.L.; Scott, B.J. Magnesium. In *Soil Analysis—An Interpretation Manual*; Peverill, K.I., Sparrow, L.A., Reuter, D.J., Eds.; CSIRO Publishing: Collingwood, 1999; 255–262.
2. Beeson, K.C. Magnesium in soils—sources, availability and zonal distribution. In *Magnesium and Agriculture, Proceedings of the Symposium*; Anderson, G.C., Jencks, E.M., Horvath, D.J., Eds.; West Virginia University: Morgantown, 1959; 1–11.
3. Uzo Mokwunye, A.; Melsted, S.W. Magnesium forms in selected temperate and tropical soils. *Soil Sci. Soc. Am. Proc.* **1972**, *36*, 762–764.
4. Metson, A.J.; Brookes, J.M. Magnesium in New Zealand soils. II. Distribution of Exchangeable “reserve” magnesium in the main soil Groups. *NZ. J. Expt. Agr.* **1975**, *18*, 317–335.
5. Hailes, K.J.; Aitken, R.L.; Menzies, N.W. Magnesium in tropical and subtropical soils from North-Eastern Australia.

- I. Magnesium fractions and interrelationships with soil properties. *Aust. J. Soil Res.* **1997**, *35*, 615–627.
6. Mayland, H.F.; Wilkinson, S.R. Soil factors affecting magnesium availability in plant-animal systems: A Review. *J. Anim. Sci.* **1989**, *67*, 3437–3444.
7. Christenson, D.R.; Doll, E.C. Magnesium uptake from exchangeable and non exchangeable sources in soils as measured by intensive cropping. *Soil Sci.* **1978**, *126*, 166–168.
8. Lombin, G.; Fayemi, A. Release of exchangeable and non-exchangeable magnesium from Nigerian soils on cropping with maize or chemical extraction. *J. Sci. Food Agr.* **1975**, *27*, 101–108.
9. Courchesne, F.; Turmel, M.; Beauchemin, P. Magnesium and potassium release by weathering in spodosols: Grain surface coating effects. *Soil Sci. Soc. Am. J.* **1996**, *60*, 1188–1196.
10. Krauss, R.W.; Gauch, H.G. Roles of magnesium in plants. In *Magnesium and Agriculture*; Anderson, G.C., Jencks, E.M., Horvath, D.J., Eds.; West Virginia University: Morgantown, 1959; 39–61.
11. Bolan, N.S.; Horne, D.J.; Wilson, G.F.; Dawood, D.; Selvarajah, N. Composition and nutritive value of pasture irrigated with farm effluents. In *Soil Research: A Knowledge Industry for Land-Based Exporters*; Currie, L.D., Loganathan, P., Eds.; Massey University: Palmerston North, 2000; 157–164.
12. Pinkerton, A. The fate of magnesium applied to flue-cured tobacco, and its effect on leaf quality and magnesium content. *Aust. J. Expt. Agr. Anim. Husbandry* **1971**, *11*, 99–104.
13. Embleton, T.W. Magnesium. In *Diagnostic Criteria for Plants and Soils*; Chapman, H.D., Ed.; University of California Press: Berkeley, 1966; 225–263.
14. Sinclair, A.H. Availability of magnesium to rye grass from soils during intensive cropping in the glass house. *J. Agr. Sci.* **1980**, *96*, 635–642.
15. Grunes, D.L.; Rending, V.V. *Grass Tetany*; ASA Special Publication No 25, Am. Soc. Agron., Crop Sci; Soc. Am., Soil Science Society of America: Madison, 1979; 23 pp.
16. Augustin, S.; Mindrup, M.; Meiwes, K.J. Soil chemistry. In *Magnesium Deficiency in Forest Ecosystem*; Huttli, R.F., Schaaf, W., Eds.; Kluwer Academic Publishers: London, 1997; Vol. 1, 255–273.
17. Metson, A.J. Magnesium in New Zealand soils. 1. some factors governing the availability of soil magnesium: A review. *NZ. J. Expt. Agr.* **1974**, *2*, 277–319.
18. Ellis, R., Jr. Influence of soil, liming, magnesium, potassium and nitrogen on magnesium composition of plants. In *Grass Tetany*; Rending, V.V., Grunes, D.L., Eds.; Special Publication No. 35; American Society of Agronomy: Madison, 1979; Vol. 35, 70–92.
19. Grove, J.H.; Sumner, M.E. Lime induced magnesium stress in corn: Impact of magnesium and phosphorus availability. *Soil Sci. Soc. Am. J.* **1985**, *49*, 1192–1196.
20. Edmeades, D.C.; Judd, M.J. The Effects of Lime on the magnesium status and equilibria in some New Zealand top soils. *Soil Sci.* **1979**, *129*, 156–161.
21. Doll, E.C.; Lucas, R.E. Testing soil for potassium, calcium and magnesium. In *Soil Testing and Plant Analysis*; Walsh, L.M., Beaton, J.D., Eds.; Soil Science Society of America: Madison, 1973; 133–151.
22. Rahmatullah; Baker, D.E. Magnesium accumulation by corn (*Zea mays* L.) as a function of potassium-magnesium exchange in soils. *Soil Sci. Soc. Am. J.* **1981**, *45* (5), 899–903.

Manganese: Brazilian Soils

Diego Antonio França de Freitas

Nilton Curi

Federal University of Lavras, Lavras, Brazil

Nestor Kämpf

Federal University of Rio Grande do Sul, Porto Alegre, Brazil

Abstract

Manganese (Mn) content in Brazilian soils is extremely varied, ranging from very low values in soils derived from sandstone up to extremely high values in soils derived from ferruginous dolomite. This variability is also related to the environmental conditions, highlighting the soil drainage because although Mn is insoluble under oxidizing conditions, it is soluble under reducing conditions. The peculiar Mn behavior in Brazilian soils impact cropping systems and the environment, mainly due to the high sorption of Mn and consequently large accumulation capacity of trace elements.

INTRODUCTION

Manganese (Mn) oxides generally occur in small proportions in parent rocks and soils; however, they have significant influence on their chemical properties and show high sorption capacity of trace elements and therefore are able to accumulate large amounts of these trace element ions. Mn is also necessary for human and plant health in small quantities.

The majority of Brazilian soils have concentrations of Mn near the world average.^[1] However, due to the environmental variability of Brazilian soils, related to the parent material and environmental conditions, mainly soil drainage (Mn is sensitive to reduction–oxidation reactions), a wide variability of Mn values ranging from very low to extremely high, impacting cropping systems and the environment, is found.

Mn IN SOILS

Mn oxides commonly have authigenic origin in soils and are formed by chemical or biochemical precipitation from the solution and by crystallization of dispersed colloids.^[2] In soils presenting oxidizing pedoenvironments (well-drained soils), Mn content is generally related to the parent materials (normally associated to iron oxides); Mn tends to occur disseminated in the soil as finely dispersed particles, with higher concentrations in the A horizon (through biocycling) and in the C horizon (through accumulation). In soils having variable oxidizing–reducing environments (imperfectly to poorly

drained soils), higher Mn contents occur in the form of discontinuous (black and brown) coatings (mangans) on the surface of aggregates or filling the pores, forming concretions and nodules, generally situated in the zone where the water table fluctuates. However, the accumulation of Mn in soils does not necessarily indicate the redox environment, as they may be relicts of preterit wetter conditions.^[2] In the form of mangans, Mn^{4+} (oxidized form) can be easily reduced to Mn^{2+} (reduced form) and cause toxicity to plants, considering the latter is mobile in soils solution. It is worthy to mention this reduction (gain of electrons) is frequently accelerated by microorganisms.

In the acid soils (pH 4–5), the contents of soluble Mn generally reach toxic levels for crops, primarily in soybeans [*Glycine max* (L.) Merr], which may be neutralized by the addition of lime. In soils with high Mn contents and without apparent evidence of Mn toxicity, prolonged periods of water residence time in the rainy periods can favor the dissolution of oxides by reduction, temporarily releasing soluble Mn in toxic levels to sensitive plants.

In soils, the occurrence and formation of Mn oxides are influenced by the higher proportion of different ions, microorganisms, and organic compounds contributing to their small particle size, high surface area, and low crystallinity.^[2] Mineralogical characterization of the Mn oxides in soils is a challenge due to the large number of minerals, inaccurate knowledge of some of their structures, the diffuse nature of X-ray diffraction (XRD) patterns of some minerals, and the coincidence of reflections with other associated minerals.

Table 1 Minimum and maximum total Mn values in the fine earth fraction of soils (<2 mm) from Brazil.

Location	Soils	Parent rocks	Mn (mg kg ⁻¹)	References
Southern	Cambisols (Inceptisols)	Rhyolite-Dacite	123–354	[3]
Southern	Cambisols (Inceptisols)	Basalt	309–1,171	[3]
Southern	Latosols (Oxisols)	Sandstone	149–306	[3]
Southern	Latosols (Oxisols)	Basalt	490–12,644	[3]
Southern	Nitisols (Ultisols)	Basalt	902–28,504	[3]
Various	Latosols (Oxisols)	Basalt and mafic rocks	472–1,712	[4]
Various	Latosols (Oxisols)	Clayey sediments	47–376	[4]
Various	Latosols (Oxisols)	Anfibolites	170–338	[4]
North	Nitisols (Ultisols)	Mafic rocks	527–3,467	[5]
North	Argisols (Ultisols)	Felsic rocks	15–58	[5]
Southeastern	Sandy soils and medium texture	Sandstone	14–257	[6]
Southeastern	Neosols and Latosols (Oxisols)	Mafic rocks	710–2,395	[6]
Northeast	Various	Various	15–1,668	[7]
Central Plateau	Latosols (Oxisols)	Various	19–1,030	[8]
Central Plateau	Latosols (Oxisols)	Sedimentary deposits	85–347	[9]
Central Plateau	Latosols (Oxisols)	Basalt	1136–1,720	[9]
Central Plateau	Latosols (Oxisols)	Gneiss and schist	238–1,826	[9]
Iron Ore Province	Perferric soils	Itabirites	241–14,000	[10]
Iron Ore Province	Perferric soils	Ferruginous Dolomites	113–73,000	[10]
Iron Ore Province	Perferric soils	Serpentinites	366–9,500	[10]

Mn IN BRAZILIAN SOILS

Brazilian soils have a wide range of total content of Mn (Table 1). Thus, very low contents were recorded in Quartzarenic Neosols (Psamments) derived from sandy sediments of Southeastern Brazil, São Paulo State.^[6] This parent material generates mostly sandy soils, with low organic matter contents, which have a low capacity to retain Mn. Also, these soils are low in weatherable minerals and do not have a source for Mn. In soils of Rio Grande do Sul State, Southern Brazil, the highest contents of Mn were found in Latosols (Oxisols) originating from basalt.

In the Cerrado region of the Central Plateau of Brazil, soils are, in general, highly weathered, leached, and acidic, with small amounts of available nutrients for plant growth. In this region, where Latosols (Oxisols) cover 47% of the Cerrado biome, particularly in the highlands, Mn toxicity is more frequent than deficient due to the high soil acidity. In the sandy soils of this region, which are naturally low in Mn, there is commonly a deficiency of available Mn to crops. The highest values of Mn were found in soils developed from basalt in this region. According to Marques et al.^[9] due to the long weathering–leaching process that these soils had undergone, the average concentration of total Mn found, i.e., 216 mg kg⁻¹ in soils developed from sedimentary deposits, is equivalent to half of the world average, 558 mg kg⁻¹.

However, the highest Mn contents in the world were reported, especially in perferric soils derived from ferruginous dolomite in the Environmental Protection Area of Belo Horizonte city (APA Sul RMBH), a representative area of the Iron Ore Province, covering 1625.32 km², located between the coordinates 19°55′–20°15′S and 42°55′–44°15′ WGR (Fig. 1), Minas Gerais state: extreme values of 35,882 mg Mn kg⁻¹ of soil by sulfuric acid digestion,^[11] 73,000 mg Mn kg⁻¹ of soil by triacid digestion,^[12] 84,033 mg Mn kg⁻¹ of clay by sodium dithionite-citrate-bicarbonate,^[13] and 33,728 mg Mn kg⁻¹ of clay by acid ammonium oxalate.^[10] In these Mn-rich soils, the influence of the parent material is stronger than the soil landscape position and the degree of weathering–leaching of these soils. Due to their high content of Mn, these soils from the Iron Ore Province are a natural base for a variety of studies, ranging from basic research to environmental sustainability, as suggested for South Africa.^[14]

When compared to soils from temperate regions, Brazilian soils generally have higher Mn contents. The environmental conditions have created systems with acidic pH, high oxidation of organic matter, oxidation–reduction, and weathering, presenting a typical situation in which hardly reducible forms of Mn contribute significantly to the more available forms of Mn.^[5]

Identification data of Mn minerals in Brazilian soils are rare. In soils from the State of Rio Grande do Sul, Mn

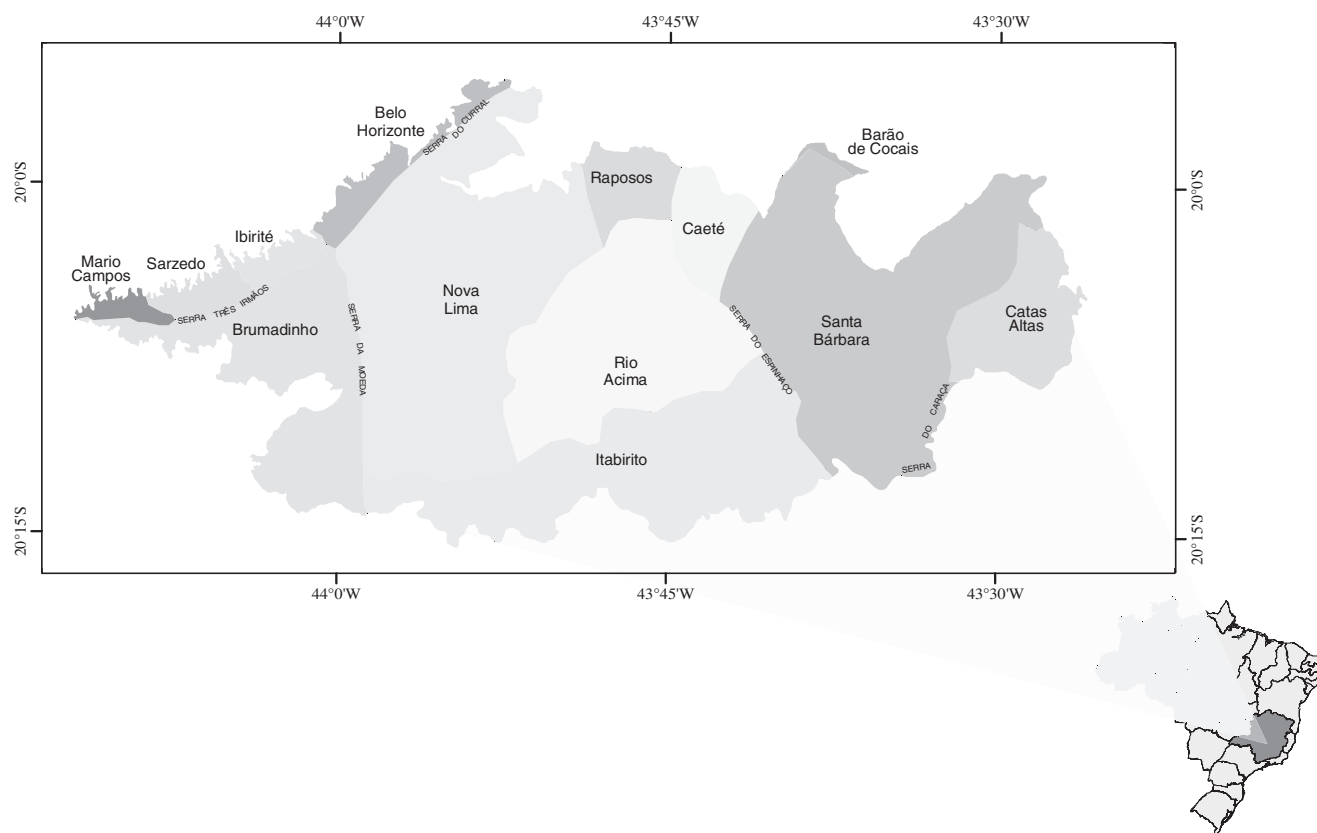


Fig. 1 Geographic location of APA Sul RMBH, Iron Ore Province, Minas Gerais State, Brazil.

Source: From Carvalho Filho, Curi, et al.^[10]

oxides were identified by XRD in 6 out of 12 samples of nodules and mangans, following concentration pretreatments, and birnessite was the most frequently found Mn oxide.^[15] Due to the peculiar Mn-rich nature of soils from the Iron Ore Province, Minas Gerais State, todorokite, lithiophorite, and pyrolusite were identified by XRD in the clay fraction of soils without any concentration pretreatments.^[10]

CONCLUSION

Brazilian soils have a wide range of Mn contents with some of the lowest to some of the highest values in the world. The variability is primarily related to the parent material from which the soils were formed. The climate and soil environment also provide more variability. Some soils in Brazil are affected by seasonal water tables, which impact the vertical distribution of Mn. Under oxidizing conditions, Mn is insoluble but it is soluble under reducing conditions. Additionally, phytocycling alters the vertical distribution of Mn. Commonly, some of the highest contents of Mn in a soil profile occur in the upper horizon. When soils are cultivated, Mn deficiencies may appear in well-drained sandy soils. Conversely, Mn in

some acidic soils may be toxic to sensitive crops. The wide range of Mn contents in Brazilian soils is related to the parent material and environmental conditions, mainly soil drainage.

REFERENCES

1. Kabata-Pendias, A. *Trace Elements in Soils and Plants*, 4th Ed.; CRC Press: Boca Raton, 2010.
2. Kämpf, N.; Curi, N.; Marques, J.J. Óxidos de alumínio, silício, manganês e titânio. In *Química e Mineralogia do Solo*, 1st Ed.; Melo, V. de F., Alleoni, L.R.F., Eds.; Sociedade Brasileira de Ciência do Solo: Viçosa, 2009; 573–610.
3. Kämpf, N. *Die Eisenoxidmineralogie einer Klimasequenz Von Boden ans Eruptiva in Rio Grande do sul, Brasilien*; PhD Thesis, Technische Universität Munchen, Freising, 1981, 271 pp.
4. Ker, J.C. *Mineralogia, Sorção e Dessorção de Fosfato, Magnetização e Elementos Traços de Latossolos do Brasil*; PhD Thesis, Universidade Federal de Viçosa, Viçosa, Brazil, 1995, 181 pp.
5. Singh, R. *Disponibilidade de Micronutrientes em Classes Dominantes de Solos do Trópico Umido Brasileiro: Manganês*; EMBRAPA: Belem, 1984.

6. Valadares, J.M.A.S.; Camargo, O.A. Manganês em solos do Estado de São Paulo. *R. Bras. Ci Solo* **1983**, *7* (2), 123–130.
7. De Oliveira, A.B.; Nascimento, C.W.A. Formas de manganês e ferro em solos de referência de Pernambuco. *R. Bras. Ci Solo* **2006**, *30* (1), 99–110.
8. Vendrame, P.R.S.; Brito, O.R.; Quantin, C.; Becquer, T. Disponibilidade de cobre, ferro, manganês e zinco em solos sob pastagens na Região do Cerrado. *Pesq. Agropec. Bra.* **2007**, *42* (6), 859–864.
9. Marques, J.J.; Schulze, D.G.; Curi, N.; Mertzman, S.A. Trace element geochemistry in Brazilian Cerrado soils. *Geoderma* **2004**, *121* (1), 31–43.
10. Carvalho Filho, A.; Curi, N.; Marques, J.J.; Freitas, D.A.F.; Jesus, E.A.; Massahud, R.T.L.R. Óxidos de manganês em solos do Quadrilátero Ferrífero (MG). *R. Bras. Ci Solo* **2011**, *35* (3), 783–804.
11. Empresa Brasileira de Pesquisa Agropecuária—EMBRAPA. *Manual de Métodos de Análises de Solo*, 2nd Ed.; Ministério da Agricultura e do Abastecimento: Rio de Janeiro, 1997.
12. Jackson, M.L. *Chemical Analysis: Advanced Course*; University of Wisconsin: Madison, 1974.
13. Mehra, O.P.; Jackson, M.L. Iron oxide removal from soils and clays by a dithionite-citrate system buffered with sodium bicarbonate. *Clay Clay Miner.* **1960**, *7* (1), 317–327.
14. Dowding, C.E.; Fey, M.V. Morphological, chemical and mineralogical properties of some manganese-rich Oxisols derived from dolomite in Mpumalanga province. *S. Afr. Geoderma* **2007**, *141* (1), 23–33.
15. Kämpf, N.; Azevedo, A.C. Óxidos de Mn em solos do Rio Grande do Sul. In *Congresso Brasileiro de Ciência do Solo*; Sociedade Brasileira de Ciência do Solo: Goiânia, 1993; 239–240.

Manure Management: Gaseous Emissions

Veerasamy Sejian

*Animal Physiology Division, National Institute of Animal Nutrition and Physiology,
Bangalore, India*

Lipismita Samal

*College of Agriculture, Odisha University of Agriculture and Technology,
Bhawanipatna, India*

M. Bagath

R.U. Suganthi

Raghavendra Bhatta

National Institute of Animal Nutrition and Physiology, Bangalore, India

Rattan Lal

*Carbon Management and Sequestration Center, Ohio State University, Columbus,
Ohio, U.S.A.*

Abstract

Livestock contributes both directly and indirectly to climate change through the emissions of greenhouse gases (GHGs) such as carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O). Animal manure and its use as fertilizer contribute to gaseous emissions. CH₄ and N₂O are the two major GHGs produced from livestock manure, which has a direct impact on global warming. CH₄ emissions from manure storage globally was estimated to be 470 Mt CO₂e/yr in 2010 with an expected 11% increase by 2020, and N₂O emissions from fertilizer use, manure application, and deposition by grazing livestock was estimated at 2482 Mt CO₂e/yr in 2010 with an expected 18% increase by 2020. Besides these, other emissions from livestock manure management systems are ammonia, nitric oxide, and non-CH₄ volatile organic compounds. The nature of the nitrogen cycle and its interaction with the carbon cycle demands a holistic approach for addressing gaseous emissions and mitigation research by developing suitable abatement strategies from manure management. Most representative and management technology combined with data for the specific area or region must be considered to obtain reliable results.

INTRODUCTION

The major source of gaseous emissions from livestock production sites is the waste or manure in solid, slurry, or liquid forms, with diverse physical properties. Once excreted, processes of biological decomposition and formation of gaseous compounds continue but diminish as manure cools and dries. Animal manure and its use as fertilizer contribute to gaseous emissions. Manure contains complex organic compounds (e.g., carbohydrates and proteins), which are broken down by bacteria resulting in the production of carbon dioxide (CO₂) under aerobic and methane (CH₄) under anaerobic conditions. With open grazing, manure is spread thinly, and aerobic decomposition predominates. In intensive livestock practices, animals are housed or kept in confined spaces for part of the year, and manure is concentrated and stored in tanks, lagoons, or pits where anaerobic conditions predominate and CH₄ is

produced. Bacteria, archaea, and other groups are also involved in microbial activities taking place during manure storage. Thus, manure is a source of greenhouse gas (GHG) emissions due to a multitude of microbial activities. These GHG emissions contribute to radiative forcing, which ultimately add to global warming. Besides these GHGs, other emissions from livestock manure management systems are ammonia (NH₃), nitric oxide (NO), and non-CH₄ volatile organic compounds (NMVOCs). This entry collates and synthesizes information pertaining to GHG emissions from livestock manure and its abatement strategies.

SIGNIFICANCE OF LIVESTOCK MANURE FOR GLOBAL WARMING

Increased emission of GHG has drawn worldwide attention due to risks of global warming and stratospheric ozone

depletion. Livestock contribute both directly and indirectly to climate change through the emissions of GHGs [e.g., CO₂, CH₄, and nitrous oxide (N₂O)]. Globally, the livestock industry contributes 18% (7.1 billion tons CO₂ equivalent) of global GHG emissions. Although it accounts for only 9% of global CO₂, it generates 65% of human-related N₂O and 35% of CH₄, which has 298 times and 25 times the global warming potential of CO₂, respectively. Much of the global GHG emissions from agriculture sector arise from enteric fermentation and manure from grazing animals. Animal wastes contain organic compounds such as carbohydrates and proteins. During the decomposition of livestock wastes under moist, oxygen (O₂)-free (anaerobic) environments, anaerobic bacteria transform the biomass carbon (C) into CH₄. Animal wastes also contain nitrogen (N) in the form of some complex compounds. The microbial processes of nitrification and denitrification of animal waste form N₂O, which is emitted to the atmosphere. CH₄ emissions from manure storage globally are estimated to be 470 Mt CO₂e/yr in 2010 with an expected 11% increase by 2020, and N₂O emissions from fertilizer use, manure application, and deposition by grazing livestock are estimated at 2482 Mt CO₂e/yr in 2010 with an expected 18% increase by 2020.^[1] Chianese et al.^[2] reported average CH₄ emissions from covered slurry, uncovered slurry, and stacked manure to be 6.5, 5.4, and 2.3 kg/m²/yr, respectively, although rates vary with temperature and time of storage. N₂O emissions from soil application of manure are a major contributor to total GHG emissions from agriculture with animal waste representing 30–50% of the global agricultural emissions. [Table 1](#) describes the amount of manure produced annually by different species of livestock.

FACTORS INFLUENCING MANURE GAS PRODUCTION

Principal factors that affect gaseous emissions from manure include temperature, O₂ level (aeration), moisture, and sources of nutrients. These factors are affected, in turn, by manure type (livestock type), diet, storage, and handling (pile, anaerobic lagoon, pit, etc.), and application of manure (injected, incorporated, etc.). These emissions arise from

the excreta of livestock deposited in and around buildings and from manure stores following land spreading of manures and collected as liquid slurry, solid manure, or litter-based farmyard manure (FYM). Relative importance of these emissions depends not only on manure composition and local management practices with respect to treatment, storage, and field application but also on ambient climatic conditions. The specific factors influencing each gas production are discussed below.

DIFFERENT GASES PRODUCED FROM LIVESTOCK MANURE

Important gases produced from livestock manure with direct and indirect impacts on global warming are discussed below.

Methane

The term “manure” is used here collectively to include both dung and urine (i.e., the solids and the liquids) produced by livestock. Besides enteric CH₄ production, manure storage is also a source of CH₄, especially in liquid form as slurry or compacted form, and the magnitude of CH₄ production depends on storage conditions and anaerobic decomposition of organic matter. Enteric fermentation from dairy cattle usually accounts for 80% of CH₄ with manure contributing about 20%. Methanogenesis occurs only under strict anaerobic conditions where it is coupled to other processes involved in the breakdown of manure organic matter. These conditions occur most readily when large numbers of animals are managed in a confined area (e.g., dairy farms, beef feedlots, and swine and poultry farms) and where manure is disposed of in liquid-based systems. Principal factors affecting CH₄ emissions from manure are the amount of manure produced and the portion of the manure that decomposes anaerobically. The former depends on the rate of waste production per animal, number of animals, feed intake, and digestibility, and the latter depends on how the manure is managed (e.g., whether it is stored as liquid or spread as solid). When manure is stored or treated as a liquid (e.g., in lagoons, ponds, tanks, or pits), it decomposes anaerobically and can produce a significant quantity of CH₄. The CH₄ conversion factor (MCF) for different sources of manure is described in [Table 2](#). The temperature and the retention time of the storage unit greatly affect the amount of CH₄ produced. The CH₄-producing potential of the manure varies by animal type ([Table 3](#)) and the quality of the feed consumed. For example, slurry from swine emits more GHG than that from cattle.^[3] Long-term storage of manure at high temperature promotes biological activity, thereby enhancing CH₄ production. In humid climates, air is excluded, soils become anaerobic, along with increase in the potential for CH₄ production.

Table 1 Amount of manure produced annually per kg live weight.

Animal	Solids (kg)	Urine (kg)	Total wet weight (kg)	Total dry matter (kg)
Horse	14.4	3.6	18	4
Chicken	8.4	0	8.4	4.3
Cow	19	8	27	3.8
Pig	18.2	12.2	30.4	4
Sheep	8.2	4.2	12.4	4

Table 2 MCF of different sources.

Source	MCF (%)
Pasture/dry lot	1
Solid storage	1
Liquid storage	39
Slurry channels <1 month	0
Slurry channels >1 month	39
Anaerobic (co)digestion	0–100

Little is known about the microbiological basis of methanogenesis in fresh manure, but the methanogenic potential in fresh manure is low, probably because of inhibitory concentrations of NH_3 derived from urine. There are, however, slow growing methanogens capable of adapting to as much as 7 g total ammoniacal N/l (i.e., *Methanosarcina* spp.), which develop in manure and anaerobic digesters.^[4] CH_4 is oxidized mainly by aerobic bacteria. Both methanogens and CH_4 -oxidizing bacteria are present in solid manure. Duan et al.^[5] reported the potential CH_4 oxidation in a surface crust to be 50–100 times more sensitive to nitrite (NO_2^-) than to ammonium (NH_4^+) or nitrate (NO_3^-), and <1 mM NO_2^- significantly inhibited CH_4 oxidation activity. Besides storage temperature, moisture, and duration of storage, there are several other factors influencing slurry-derived CH_4 emissions, such as stirring, aeration, and slurry tank coverage. Another important factor is the use of an inoculum to reduce the start-up time of CH_4 production by providing already established microbe populations. Additionally, the dilution factor of slurry may also affect CH_4 emissions. Hindrichsen et al.^[6] observed that feeding treatments causing a higher enteric CH_4 formation resulted in lower CH_4 emissions from the corresponding slurry, probably because of the lack of fermentable matter, which has already been spent in the animal's digestive tract.

Table 3 CH_4 produced from manure management.

Animal	Manure management (kg CH_4 /head/yr)
Dairy cows	13
Beef	6
Other cattle >1 year	6
Other cattle <1 year	6
Pigs	3
Sheep	0.2
Lambs <1 year	0.1
Goats	0.1
Horses	1.4
Poultry	0.1
Deer	0.3

Nitrous Oxide

N_2O is produced, directly and indirectly, during the storage and treatment of manure before it is applied to land. Direct emissions occur from different systems such as pasture, range, and paddock, and indirect emissions occur from the soil due to other forms of N loss from the systems. Even indirect N_2O losses (i.e., derived from NO_3^- leaching and NH_3 volatilization) can at times be larger than direct N_2O losses. The emission of N_2O from manure during storage and treatment depends on the N and C contents of manure and on the duration of the storage and type of treatment. Whereas CH_4 production and oxidation processes are associated with anoxic and oxic conditions, respectively, and emissions of N_2O are stimulated under O_2 -limited conditions. Direct N_2O emissions occur through combined nitrification and denitrification of N contained in the manure. Nitrification does not occur under anaerobic conditions. It is a two-step oxidation of $\text{NH}_3\text{-N}$ to $\text{NO}_3\text{-N}$, primarily carried out by NH_3 -oxidizing bacteria (AOB) and nitrite-oxidizing bacteria (NOB). NH_3 -oxidizing archaea have also been observed in composting manure and soil, but are not responsive to N inputs, and hence, their role is uncertain. N_2O is produced either by AOB, as a side-product in the conversion of NH_3 , or by a subsequent conversion of NO_2^- through nitrifier-denitrification process. Denitrification is the transformation of nitrites and nitrates to N_2O and dinitrogen (N_2). The ratio of $\text{N}_2\text{O}/\text{N}_2$ increases with increasing acidity, nitrate concentration, and reducing moisture content. Denitrification is carried out by a phylogenetically diverse group of heterotrophic bacteria, although the process has also been observed among fungi and archaea. Most denitrifiers are facultative anaerobes, i.e., they prefer O_2 as electron acceptor but can use NO_3^- or NO_2^- in the absence of O_2 . The expression of genes for denitrifying enzymes is stimulated at low O_2 levels, but N_2O reductase is unstable even with traces of O_2 . Therefore, N_2O can accumulate around oxic–anoxic interfaces in the manure. Chemical gradients can also develop, which may influence N_2O emissions. Around air–liquid interfaces, CO_2 and NH_3 will be lost to the gas phase, thereby removing the buffering capacity and alkalinity. As NH_3 oxidation is an acidifying process, there is thus a potential for significant reduction in pH because of nitrification. Fukumoto^[7] showed that the addition of NOB to a compost prevented NO_2^- accumulation and N_2O emission. In summary, the production and emission of N_2O from managed manures require the presence of either nitrites or nitrates in an anaerobic environment preceded by aerobic conditions necessary for the formation of these oxidized forms of N. In addition, conditions preventing reduction from N_2O to N_2 (e.g., a low pH or limited moisture) must also be present. If pH declines, rates of nitrification and denitrification may also decline, but the $\text{N}_2\text{O}/\text{N}_2$ ratio of denitrification will increase. Table 4 describes the composition of manure in different livestock species.

Table 4 Composition of manure in different livestock species.

Livestock	Moisture content (% wet basis)	Total manure (L/day/animal)	Total solids content (kg/day/animal)	Total volatile solids (kg/day/animal)
Dairy cattle	87	68.14	9.07	7.71
Beef cattle	92	29.60	2.31	1.89
Poultry broiler	74	0.10	0.03	0.02
Pig	90	4.73	0.45	0.37

Nitric Oxide

NO is emitted from soils through nitrification in the surface layers of stored manure or in manure aerated to reduce odor or to promote composting. Increased nitrification is likely to occur following the application of manures and deposition of excreta during grazing.

Ammonia

Agricultural activities and concentrated animal feeding operations account for up to 80% of the total NH_3 emissions on a global scale.^[8] NH_3 can volatilize from solids and liquids separation area, manure storage facilities (under floor pits and anaerobic lagoons), and also land application of manure. The relationships between NH_3 volatilization and factors such as air velocity and turbulence, manure temperature, and pH have been well documented.^[9] The source of NH_3 emission from manure management is the N excreted by the livestock. The excretion of N, and the subsequent emissions of NH_3 , varies between livestock species (e.g., cattle and pigs). Within a livestock species, there are large differences between animals raised for different purposes (e.g., dairy cattle vs. beef cattle). In general, only the $\text{NH}_3\text{-N}$ is susceptible to loss by volatilization from dairy wastewater into the atmosphere.^[10] About 75–80% of the total N entering a dairy facility is lost from an anaerobic lagoon through gaseous emissions and only 20–25% of total N retained in the lagoon liquid being available for fertilizing croplands or pastures.^[11] NH_3 volatilization is generally the largest pathway of the loss of manure N, with losses typically accounting for 30–70% of the NH_4^+ content of cattle manure. In ruminants, the conversion of feed N into milk and meat is an inefficient process, and approximately 75–80% of the feed N intake is excreted in urine and feces.^[12] NH_3 can only be volatilized when urinary N (urea) is hydrolyzed by the enzyme urease present in the feces and not in the urine. NH_3 emissions from livestock manures during and after field application depend on properties of the manure, including viscosity, total $\text{NH}_3\text{-N}$, C contents, and pH.

Non- CH_4 Volatile Organic Compounds

The NMVOCs are defined as all those artificial organic compounds other than CH_4 , which can produce

photochemical oxidants by reaction with N oxides in the presence of sunlight. NMVOCs from animal production originate from feed, especially silage, and from partly digested and undigested fat, carbohydrates, and protein that decompose in manure. Consequently, anything that affects the rate of manure management (e.g., manure management in the stable and storage, straw added to the manure, duration of storage, and manure application) affects NMVOC emission. Sites of emission include livestock buildings, yards, manure stores, and fields on which the manure is spread. These emissions depend on the actual temperature and the wind speed over the surface. Up to 50% of the NMVOC from animal handling is isopropanol and n-propanol followed by acetaldehyde and short-chain acids such as acetic acid, propionic acid, and butanoic acid.^[13] Dimethyl sulfide comprises approximately 1–3% of the NMVOC emission. Ethyl acetate and dimethyl disulfide occur in larger concentrations in cattle and poultry barns, respectively. These compounds contribute strongly to the odor associated with manure.

GENERAL MANURE MANAGEMENT STRATEGIES

The following are the general strategies by which manure can be effectively managed:

1. Using animal genetics, phase feeding (altering feed to match the age, growth, and production level of animals), and amino acid supplements as management tools to reduce GHG emissions.
2. Selecting livestock to genetically improve the efficiency of feed conversion by the animal.
3. Feeding less frequently and increasing the feed digestibility by mechanical (chopping, grinding, or pelleting feed), chemical, or biological processing.
4. Feeding livestock based on age, sex, and stage of production to match diet according to the nutritional requirements.
5. Using additives and improved nutrient utilization in animals' diets to simplify manure management at all stages of handling and disposal by reducing the amount of nutrients in waste.^[14] The goal of feed management is to match the nutrient needs of animals more closely with nutrients in feed.

6. Trapping air vented from production houses and treating it before discharge to the atmosphere to reduce odorous compounds, NH_3 , and other gases. Treatment processes include ozonation and biofilters,^[15] which do not affect the N content of manure.
7. Spreading manure evenly around the pasture and maintaining healthy pastures by implementing beneficial management grazing practices, such as rotational grazing, to improve the quality of forages resulting in increased animal productivity and lower gas emissions.
8. Applying swine manure treatment strategies based on physical, chemical, and biological processes that are able to reduce manure's pollution potential and convert them into valuable by-products such as biomethane (i.e., heat and electricity), organic fertilizers, and C credits (certified emissions reductions).^[16]
9. Avoiding application of excess amounts of manure because nutrients can be lost to the environment. Testing both the soil and the manure before application ensures the proper nutrient balance for plant needs and can help reduce the loss of nutrients as gases.
10. Integrating animal and crop production systems by reusing manure in plant production.

EVALUATION OF MITIGATION STRATEGIES

The nature of the N cycle and its interaction with the C cycle demand a holistic approach for addressing gaseous emissions and mitigation research.

- Handling manure as a solid or depositing it on pasture rather than storing it in a liquid-based system such as a lagoon would likely reduce CH_4 emissions but may increase N_2O emissions.
- Developing of compact models of biogas plants efficiently by using dung and urine for production of biogas, while reducing emission of CH_4 and N_2O .
- Using manure application techniques such as subsurface injection to reduce NH_3 and CH_4 emissions but resulting in increased N_2O emissions. Injection works well when combined with anaerobic (co)digestion and solids separation by improving infiltration.^[17]
- Using practices that result in the reduction of CH_4 production but increase in N_2O emissions (e.g., aeration of manure during storage to reduce CH_4 emissions but increase in N_2O emissions when aeration rate is sufficient to create an aerobic environment because of the nature of the antagonistic processes resulting in CH_4 and N_2O emissions).
- Recognizing the complex relationship between manure NH_3 volatilization and N_2O emission, because emissions of both may be reduced by diet manipulation or manure management. If a mitigation technology reduces NH_3 losses, the preserved NH_4^+ may later

increase soil N_2O emissions.^[18] On the other hand, gaseous losses of N may reduce the availability of N for nitrification and denitrification processes and, consequently, N_2O formation.^[19]

- Including low protein levels and creating a proper balance of amino acids in the diet to minimize the amount of N excreted, particularly in urine, namely feeding pigs low-protein diets with synthetic amino acids to reduce N being excreted in urine, reducing NH_3 volatilization, and decreasing NO_2 emissions.
- Using soil/plant testing to determine fertilizer needs, limiting N mineralization by minimizing fallow periods, optimizing split application schemes, and matching N supply with crop demand reduce emissions of NH_3 , N_2O , and NO gases.

CONCLUSIONS

There are numerous interactions between the animal, storage, and land application phases of the manure management process, and management at one stage affects manure composition and gaseous emissions at subsequent stages. Therefore, mitigation practices should not be evaluated individually in isolation but as the entire continuum of manure management and treatment. The latter warrants the use of whole-farm models. System-based modeling must play a key role in integrating the complexity of management and environmental controls on GHG emissions. As production of these gases is of microbial origin, the dry matter content and temperature of manure and soil are key factors in farm manure management decisions that influence the magnitude of their emission. There are a number of manure management practices that are feasible and can effectively reduce GHG emissions from manure storage and/or land application.

FUTURE PERSPECTIVES

The main barrier to widespread implementation of manure GHG mitigation strategies is the lack of knowledge about the technology or conservatism within the sector. This warrants education of the farming community about the positive benefits of anaerobic (co)digestion, supported by incentives through grants or long-term, low-interest loans to reduce the initial costs of capital equipment. Assistance in setting up farming cooperatives with shared digester facilities would also aid the development of community level anaerobic (co)digestion schemes. It is crucial to evaluate and develop cost-effective methods that producers can choose to mitigate GHG emissions from concentrated animal feeding operations in order to comply with contemporary and future legislations. To set standards for all these gaseous pollutants, technologies are needed to estimate their respective concentrations in the air. Additional

research is needed about the impact of soil type on the emission factors and on the C sequestration in relation with use of mineral fertilizer and FYM.

REFERENCES

1. U.S. Environmental Protection Agency (USEPA). *Global Anthropogenic Non-CO₂ Greenhouse Gas Emissions: 1990–2020*; USEPA: Washington, 2006.
2. Chianese, D.S.; Rotz, C.A.; Richard, T.L. Whole-farm gas emissions: A review with application to a Pennsylvania dairy farm. *Appl. Eng. Agric.* **2009**, *25*, 431–442.
3. Dinuccio, E.; Berg, W.; Balsari, P. Gaseous emissions from the storage of untreated slurries and the fractions obtained after mechanical separation. *Atmos. Environ.* **2008**, *42*, 2448–2459.
4. De Vrieze, J.; Hennebel, T.; Boon, N.; Verstraete, W. Methanogenesis: The rediscovered methanogen for heavy duty biomethanation. *Bioresour. Tech.* **2012**, *112*, 1–9.
5. Duan, Y.F.; Elsgaard, L.; Petersen, S.O. Inhibition of methane oxidation in slurry surface crust by inorganic nitrogen: An incubation study. *J. Environ. Qual.* **2013**, *42*, 507–515.
6. Hindrichsen, I.K.; Wettstein, H.R.; Machmüller, A.; Jörg, B.; Kreuzer, M. Effect of the carbohydrate composition of feed concentrates on methane emissions from dairy cows and their slurry. *Environ. Monit. Assess.* **2005**, *107*, 329–350.
7. Fukumoto, Y.; Suzuki, K.; Osada, T.; Kuroda, K.; Hanajima, D.; Yasuda, T.; Haga, K. Reduction of nitrous oxide emission from pig manure composting by addition of nitrite-oxidizing bacteria. *Environ. Sci. Technol.* **2006**, *40*, 6787–6791.
8. Liang, Z.S.; Westerman, P.W.; Arogo, J. Modelling ammonia emission from swine anaerobic lagoons. *Trans. ASAE* **2002**, *45* (3), 787–798.
9. Ndegwa, P.M.; Hristov, A.N.; Arogo, J.; Sheffield, R.E. A review of ammonia emission mitigation techniques for concentrated animal feeding operations. *Biosys. Eng.* **2008**, *100*, 453–469.
10. Sommer, S.G.; Zhang, G.Q.; Bannink, A.; Chadwick, D.; Misselbrook, T.; Harrison, T.; Hutchings, N.J.; Menzi, H.; Monteny, G.J.; Ni, J.Q.; Oenema, O.; Webb, J. Algorithms determining ammonia emission from buildings housing cattle and pigs and from manure stores. *Adv. Agron.* **2006**, *89*, 261–335.
11. ICL (Idaho Conservation League) and IDEAL (Independent Dairy Environmental Action League) Report. *Dairy Air Emissions Analysis: Focus: Ammonia Emissions for Typical Dairy Management Systems in Idaho Summary of Methodology and Findings: Technical Reference Document Final Draft Developed in a Collaborative Negotiation*, Feb 10–11, 2005; ICL and IDEAL: Boise, 2005.
12. de Boer, I.J.M.; Smits, M.C.J.; Mollenhorst, H.; van Duinkerken, G.; Monteny, G.J. Prediction of ammonia emissions from dairy barns using feeds characteristics Part I: Relation between feed characteristics and urinary urea concentration. *J. Dairy Sci.* **2002**, *85*, 3382–3388.
13. US EPA. 2012. Available at <http://www.epa.gov/oecaagct/airmonitoringstudy.html> (accessed January 2011).
14. Abt Associates. *Air Quality Impacts of Livestock Waste. Report Prepared for U.S. EPA Office of Policy*; Abt Associates: Bethesda, 2000.
15. Jacobson, L.D.; Moon, R.; Bicudo, J.; Janni, K.; Noll, S.; Shurson, G.; Zhu, J.; Schmidt, D.; McGinley, C.; Goodrich, P.; Nicolai, R.; Clanton, C.; Davis, K.; Brosseau, L.; Bruns, J.; Pijoan, C.; Blaha, T. *Generic Environmental Impact Statement on Animal Agriculture: A Summary of the Literature Related to air quality and Odour (H)*. Report Prepared for Minnesota Environmental Quality Board, Minnesota Department of Agriculture. University of Minnesota: St. Paul, 1999.
16. Vanotti, M.B.; Szogi, A.A.; Vives, C.A. Greenhouse gas emission reduction and environmental quality improvement from implementation of aerobic waste treatment systems in swine farms. *Waste Manage.* **2008**, *28*, 759–766.
17. Montes, F.; Meinen, R.; Dell, C.; Rotz, A.; Hristov, A.N.; Oh, J.; Waghorn, G.; Gerber, P.J.; Henderson, B.; Makkar, H.P.S.; Dijkstra, J. Special topics—Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options. *J. Anim. Sci.* **2013**, *91*, 5070–5094.
18. Petersen, S.O.; Sommer, S.G. Ammonia and nitrous oxide interactions: Roles of manure organic matter management. *Anim. Feed Sci. Tech.* **2011**, *166–167*, 503–513.
19. USEPA, U.S. Environmental Protection Agency. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2008*; USEPA: Washington, 2010. Available at www.epa.gov/climatechange/emissions/usinventoryreport.html (accessed April 2014).

Manure: Compost and Biosolids

Bahman Eghball

*U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
Lincoln, Nebraska, U.S.A.*

Kenneth A. Barbarick

Colorado State University, Fort Collins, Colorado, U.S.A.

Abstract

Manure, compost, and biosolids (municipal sludge) are organic residuals that contain nutrients and organic matter. They are excellent substitutes for chemical fertilizers. The organic matter in these renewable organic residuals can significantly improve the chemical and physical properties of soil and enhance biological activities. Because manure, compost, and biosolids contain nutrients and organic matter, they can be used to improve degraded, eroded, or less productive soils as soil amendments. If not used properly, manure, compost, and biosolids can be sources of environmental pollution.

MANURE

Manure (animal waste) is generated in beef cattle feedlots, swine operations, dairy barns, poultry houses, and other livestock operations. The number of animals and the number of large production facilities in United States have significantly increased in the past years.^[1] Manure, as well as composts and biosolids, is a renewable resource and an excellent source of macronutrients [nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), and sulphur (S)] and micronutrients [zinc (Zn), copper (Cu), iron (Fe), manganese (Mn), etc.] that are essential for growing plants. For centuries, manure was used throughout the world for improving soil fertility and enhancing crop productivity. However, with the advent of synthetic fertilizers after World War II, manure was considered more a liability than a nutrient resource for crop production.

When animals are grazing on pastures and rangelands, manure is dispersed across a large area and little management is needed because the material is not concentrated and decomposes rapidly. However, when animals are concentrated in small feeding areas, the quantity of manure requiring proper management increases greatly. Significant amounts of manure are generated each year in United States from the confined feedlots of major livestock species (Table 1). The amount of N, P, and K present in the manure from these species would replace 25%, 25%, and 45% of the purchased N, P, and K fertilizers, respectively, in United States, if utilized at agronomic application rates (Table 1). However, because of the high hauling cost, replacement of fertilizer is limited to specific areas in the country where the animal feeding operations are located.

Crop producers are also reluctant to use manure because of factors such as hauling and spreading costs, potential introduction of weed seeds, non-availability of manure where needed, uncertainty about availability of manure nutrients to plants, and problems of odor and application uniformity. The global numbers of major livestock species are given in Table 2.

Even though manure is an excellent source of multiple nutrients and organic matter, it can also contribute to water, air, and land pollution because of the potential for environmental loading with excess P, nitrate, salts, undesirable microorganisms, pathogens (disease-causing organisms), and greenhouse gases. Manure application in excess of crop needs can cause a significant build up of P, N, trace elements [arsenic (As), cadmium (Cd), lead (Pb), mercury (Hg), molybdenum (Mo), nickel (Ni), Cu, Fe, Mn, selenium (Se), and Zn], and salts in soils. Trace-element limits in soil are given in Table 3. The elevated P and N levels in soil are of environmental concern when these nutrients are carried by runoff to streams and lakes and cause “eutrophication,” which is the nutrient enrichment of water that can promote algal growth and depletion of dissolved oxygen in water. This oxygen is essential for aquatic animals. Pathogens (such as bacteria, viruses, and parasites) in runoff from fields treated with manure can be another source of water pollution. Pathogens and odorous materials can also be carried by wind from the feeding operations to neighboring areas. Excess manure application can contaminate the groundwater with nitrate–N. Nitrate is a water-soluble ion that moves with water into the soil and can reach the groundwater within few days after application. The U.S. Environmental Protection Agency (USEPA) has set a 10 mg NO₃–N/L standard for drinking water.

Table 1 Annual manure, N, P, and K generated by animals confined in beef cattle feedlots, dairy barns, poultry and swine operations, and fertilizer use in United States.

Animal species	Animals on feed ^a (million)	Manure (dry weight; million Mg)	N ^b (Mg × 1000)	P ^b (Mg × 1000)	K ^b (Mg × 1000)
Beef cattle	13.22	31.67	602	206	633
Dairy cows	13.14	22.01	782	140	522
Chickens (broilers and layers) ^c	8263.00	13.26	544	186	278
Turkeys ^c	284.00	3.09	142	65	65
Swine	59.41	15.15	709	451	709
Total		85.18	2,779	1,048	2,207
1999 Fertilizer use in United States ^d			12,436	4,345	5,016
1996 Global fertilizer use ^d			78,353	13,543	17,516
(Manure nutrient/U.S. fertilizer use) × 100 (%)			25	25	45

^aFrom U.S. Department of Agriculture.^[1]^bManure weight and N, P, and K contents taken from U.S. Department of Agriculture.^[2]^cYearly production numbers.^dFrom The Fertilizer Institute.^[3]

COMPOST

Composting is the aerobic decomposition of organic materials in the thermophilic temperature range of 40–65°C. The composted material should be an odorless, fine textured, low-moisture content material that can be bagged and sold for use in gardens, potting, and nurseries or used as a source of nutrients and organic matter on cropland with little fly-breeding potential. Other advantages of composting include improving the handling characteristics of any organic residue by reducing its volume and weight. Composting also has the potential to kill pathogens and weed seeds. Disadvantages of composting organic residues include loss of N and other nutrients during composting, the time taken for processing, cost of handling equipment, need for available land for composting, odors during composting, marketing, diversion of manure or residue from cropland, and slow release of available nutrients. Similar to manure or biosolids, composts can cause water, air, and land pollution if not used properly.

Temperature, water content, C/N ratio, pH level, aeration rate, and the physical structure of organic materials are important factors influencing the rate and efficiency of the composting process. Ideal values for these factors include a temperature of 54–60°C, C/N ratio of 25:1–30:1, 50–60% moisture content, oxygen concentration >5%, pH of 6.5–8.0, and particle size of 3–13 mm. The requirement set

by the USEPA regulations for composting municipal waste is that the temperature should be maintained at 55°C or above for at least three days so as to destroy the pathogens. A temperature of 63°C within the compost pile is needed to destroy the weed seeds.

Homogeneous manure solids can be composted alone without mixing with bulk materials. Bulking agents are required to provide structural support when manure solids, or other organic residues, are too wet to maintain air space within the composting pile, to reduce water content, and/or to change the C/N ratio. Dry and fibrous materials, such as saw dust, leaves, and finely chopped straw or peat moss, are good bulking agents for composting wet manure or organic residues. Depending on the ambient temperature, a complete composting process may take 2 to 6 months.

There are a number of methods for composting organic materials. These include active windrow (with turning), passive composting piles, passively aerated windrow (supplying air through perforated pipes embedded in the windrow), active aerated windrow (forced air), bins, rectangular agitated beds, silos, rotating drums, containers, anaerobic digestion, and vermicompost (using earthworms). Carcass composting can be done by using all types of animals. Mortality composting can be accomplished in backyard-type bins, indicator composter bins, and temporary open bins using layers of saw dust or chopped straw and dead animals. Water content is an important factor to be considered when composting dead animals and should be maintained at about 40–50%.

Table 2 Global numbers of major livestock species in 1997.

Animal species	Animals ^a (million)
Cattle	1333
Chickens	14156
Sheep and goats	1754
Swine	837

Source: ^aFrom Food and Agricultural Organization.^[4]

BIOSOLIDS

Treatment of municipal wastewater results in a mostly organic by-product known as “biosolids.” Land application of biosolids for beneficial use has been practiced since the early 20th century in United States. The USEPA announced

Table 3 USEPA (40CFR503.13, revised July 1, 1999), trace element limits, and concentrations in Littleton/Englewood, CO biosolids, July 27, 1999.

Trace element	Agronomic rate concentration limit (mg/kg)	Ceiling concentration limit (mg/kg)	Annual soil loading limit (kg/ha)	Cumulative soil loading limit (kg/ha)	Littleton/Englewood biosolids (mg/kg)
As	41	75	2.0	41	2.7
Cd	39	85	1.9	39	5.6
Cu	1500	4300	75	1500	256
Pb	300	840	15	300	46
Hg	17	57	0.85	17	1.2
Mo	—	75	—	—	8.0
Ni	420	420	21	420	15
Se	100	100	5.0	100	4.6
Zn	2800	7500	140	2800	198

Note: Concentration and quantities are on dry weight basis.

requirements regarding beneficial use of biosolids with promulgation of the 40 CFR503 regulations in February 1993. The USEPA and the state agencies that control land application of biosolids encourage the judicious recycling of biosolids on crop- or rangeland, as they contain essential plant nutrients and organic matter.

A key aspect of USEPA and Colorado Department of Health (CDH) regulations requires the application of biosolids at an agronomic rate. The CDH^[5] defines agronomic rate as “the rate at which biosolids are applied to land such that the amount of N required by the food crop, feed crop, fiber crop, cover crop, or vegetation grown on the land is supplied over a defined growth period, and such that the amount of N in the biosolids which passes below the root zone of the crop or vegetation grown to groundwater is minimized.” The USEPA trace-element limits for land application of biosolids are shown in Table 3. State agencies that control biosolids recycling on land are required to adopt these limits as minimum requirements to protect the environment and public health. Risk assessment of different biological pathways served as the foundation for establishing the trace-element restrictions. For example, the concentrations for Littleton/Englewood biosolids shown in Table 3 indicate that it meets the agronomic-rate limits and can, therefore, be applied at an agronomic rate with minimal restriction. New, aggressive pretreatment programs have significantly reduced trace-element concentration in biosolids since about 1970, and therefore, environmental and public health risks are even more minimal.

The USEPA requires municipal wastewater treatment facilities to reduce pathogens and to reduce the attraction of insects and animals before applying biosolids to land. Most municipal wastewater treatment plants use heat and attack by beneficent microorganisms through anaerobic (without air) or aerobic (with air) digestion to kill potential pathogens and reduce odors that may reside in wastewater. Municipalities accomplish further reduction of pathogens and stabilization by composting, drying, or other techniques.

The major reason that the USEPA promotes land application of biosolids is that the plant nutrients and organic matter can benefit the soil–plant agroecosystem. For example, Littleton/Englewood biosolids used at two research locations contained up to 5.0% organic–N, 1.3% ammonium–N, 140 mg/kg nitrate–N, 3.7% P, and 0.30% K. Biosolids can also provide plant micronutrients such as Fe and Zn. The organic carbon in biosolids can help to develop and stabilize soil structure with a concomitant increase in precipitation capture and decrease in soil erosion. Efficacious land application of biosolids changes the perspective from disposal of a waste (i.e., a nuisance) to recycling a valuable resource (i.e., a beneficial process).

CONCLUSION

Organic residuals (wastes) can serve as excellent sources of plant nutrients such as N, P, and micronutrients, namely Fe and Zn. Proper management is required, however, to match nutrient amounts supplied by the organic materials with crop needs so as to avoid potential pollution problems.

REFERENCES

1. United States Department of Agriculture. *Agricultural Statistics*; Government Printing Office: Washington, 2000.
2. United States Department of Agriculture. *Agricultural Uses of Municipal, Animal, and Industrial Byproducts*; Conserv. Research Report No. 44; Government Printing Office: Washington, 1998.
3. The Fertilizer Institute. *Fertilizer Statistics*. The Fertilizer Institute: Washington, 2001. Available at <http://www.tfi.org>.
4. Food and Agricultural Organization. *Statistical Database*; United Nations: New York, 2001. Available at <http://fao.stat.fao.org/> (accessed August 20, 2015).
5. Colorado Department of Health. *Biosolids Regulation 4.9.0*, 1993; 46 pp.

Mapping Soil Carbon

Budiman Minasny

Brendan Malone

Alex B. McBratney

Department of Environmental Sciences, University of Sydney, Sydney, New South Wales, Australia

Abstract

The soil system is recognized as a significant terrestrial sink of carbon. Mapping and predicting the spatial distribution of soil carbon have been of great interest, as demonstrated by the increasing number of publications in mapping soil carbon stock globally and nationally. This entry gives an overview of mapping soil carbon based on traditional soil maps to modern digital mapping technologies.

The soil system is recognized as a significant terrestrial sink of carbon. Mapping and predicting the spatial distribution of soil carbon have been of great interest, as demonstrated by the increasing number of publications in mapping soil carbon stock globally and nationally.^[1,2] However, it is also a challenge as evidenced by the large variation of soil carbon concentration observed at field, continental, and global scales (Fig. 1). Soil carbon mapping and the knowledge of the spatial distribution of soil carbon are useful to: 1) provide a baseline carbon level; 2) assist in soil management for agricultural production and ecosystem services; 3) identify critical areas for monitoring soil conditions; 4) identify potential areas for soil-based carbon sequestration; and 5) serve as an input into process-based simulation models.

There is a fundamental difference between mapping soil carbon and carbon auditing. For carbon auditing, we are only interested in the total amount of carbon in the soil within a given spatial domain. Therefore, it is not necessary to be mapped.^[3] The product of digital mapping, however, is maps of the spatial distribution of soil carbon, and while we can use mapping for temporal soil carbon auditing, it will generally be an expensive exercise.

Traditionally, soil carbon estimates are based on soil maps using soil–landscape and vegetation associations.^[4] At the continental scale, the resulting maps are usually at the cartographic scale of 1:1,000,000 or coarser, e.g., Africa^[5] and Brazil.^[6] These maps are useful where there is little soil information for the area of interest. There are several versions of global soil carbon maps, and these are mainly derived from the Food and Agriculture Organization soil map and the Harmonized World Soil Database, such as^[7] at a resolution of 30 arc seconds or the United Nations Environment Programme's World Conservation Monitoring Centre updated Global Carbon Map.^[8] Global estimations of soil carbon stocks from the global maps range between 1200 and 2470 for the upper 1-m profile^[4,7]

that were derived from different data and regression models. In any case, these estimates of soil carbon stocks have no quantified uncertainty associated with them.

Traditional soil mapping has been difficult and time-consuming due to the great soil spatial variation. In addition, much soil information is lost because they are mapped in polygons reflecting soil types or classes and there is no quantified uncertainty associated with them. The development of digital soil mapping technologies arose in the 1990s^[9] and formalized by McBratney, Mendonça Santos, and Minasny^[10] These technologies align with the progress in digital earth observation with massive spatial data generated by remote and on-the-ground sensors. Concurrently, mathematical and statistical techniques have been developed that allow for the prediction of soil properties in areas with little information as well as indicating the uncertainty of the predictions. The digital revolution has transformed many of the sciences, e.g., soil carbon concentration is mapped as a continuous function of depth, as soil carbon stock can be readily calculated at any depth.^[11] These maps are stored and manipulated in digital form within a geographic information system environment, creating the possibility of vast arrays of data for analysis and interpretation. A review of digital mapping of soil carbon is presented in the study by Minasny et al.^[2]

The basis of digital soil mapping is that the soil property (carbon content) is related to its environmental factors and can be expressed as a spatial soil prediction function. This is called the “*scorpan*” spatial prediction function:

$$C_{x,z} = f(s, c, o, r, p, a, n) + e$$

where soil carbon concentration C (either as a mass fraction kg C per kg soil or as a density kg C per m³ volume of soil) at spatial position x and depth z is a function of soil factors (s), climate (c), organisms that include land use, human effects, and management (o), relief (r),

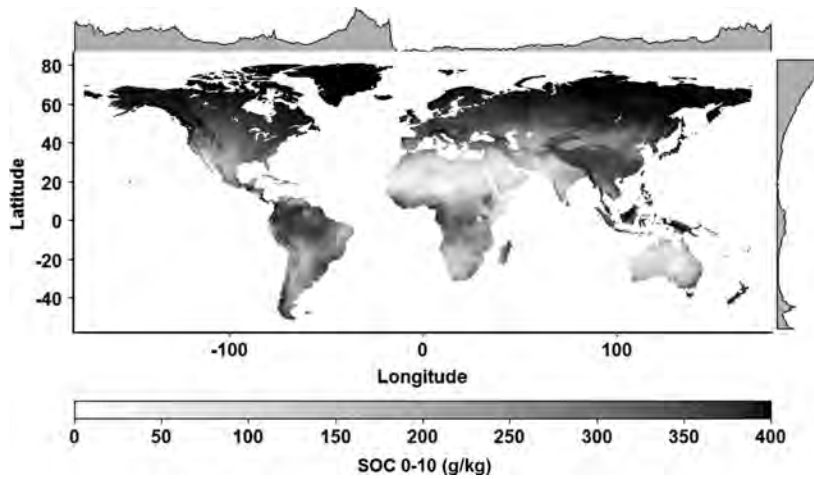


Fig. 1 A global map of soil organic carbon content at the depth of 0–10 cm. The map was generated using an empirical model relating the soil carbon data from the International Soil Reference and Information Centre–World Inventory of Soil Emission global profile database to climate (mean annual temperature, precipitation, radiation, and evapotranspiration) and a topographic variables (topographic wetness index). The graph on the right represents the variation of soil organic carbon with latitude.

parent materials (p), age or time (a), spatial position (n), and the spatially correlated errors (e). Soil variables such as an existing soil map “s” can also be used as a predictor. The form of *f* can be simple linear models^[12] to more sophisticated data mining tools.^[13] Soil observations are fitted to model the relationship between the soil carbon concentration and their *scorpan* factors. This is followed by using the fitted model to predict at all locations of the mapping extent. This approach is quantitative and relies on the assumption that the soil observations cover the whole-range variation in environments, so that the model can be extrapolated to the whole area.

The soil carbon maps, produced using digital soil mapping techniques, have been applied at various grid spacing and extents, from global,^[12] continental,^[14,15] national,^[13,16,17] regional, catchment,^[18–20] farm or field,^[21,22] and urban areas.^[23]

Soil organic carbon is most often mapped; however, inorganic carbon has also been mapped,^[24] as has different pools of carbon^[19] and soil carbon saturation index.^[25] Another important challenge is in mapping areas with peat soils, where the environmental (*scorpan*) relationships between soil organic carbon in peat and mineral soils can be quite different.^[26]

Global soil initiatives such as GlobalSoilMap have been developed by a consortium that aims to create a digital map of the world’s soil properties, including soil carbon.^[27] This global effort will provide access to the best available map of soil properties across the globe at a resolution of 100 m along with its 90% confidence of prediction, in a consistent format at the depth ranges of 0–5, 5–15, 15–30, 30–60, 60–100, and 100–200 cm. Realizing that it is a significant effort to apply the *scorpan* prediction function across the globe, the approach taken for global soil mapping is a pragmatic one. The methods that are used (or will be used) for mapping consider the nature, availability, and density of the existing soil data. A first approach of mapping soil carbon in the United States is based on a 1:250,000 soil map from the U.S. Department of

Agriculture–Natural Resources Conservation Service, where the soil polygons were converted to raster estimates of organic carbon content for the 6 depth intervals of the GlobalSoilMap specification.^[28] This effort is mirrored elsewhere around the globe.

In addition to mapping the baseline soil carbon, digital maps can be used to map or estimate soil carbon change with time. When soil carbon data over space and time are available, they can be interpolated using spatiotemporal kriging. However, this approach is not dynamic and cannot be extrapolated for scenario modelling. On the other hand, the digital map of soil carbon is usually used in a mechanistic model to predict the change of C under different scenarios. The challenge in mapping soil carbon in space and time is to use a more mechanistic and dynamic relationships analyzing the spatial and temporal carbon data.^[29]

REFERENCES

1. Grunwald, S. Multi-criteria characterization of recent digital soil mapping and modeling approaches. *Geoderma* **2009**, *152*, 195–207.
2. Minasny, B.; McBratney, A.B.; Malone, B.P.; Wheeler, I. Digital mapping of soil carbon. *Adv. Agron.* **2013**, *118*, 1–47.
3. McBratney, A.; Minasny, B.; De Grujter, J.; Mulvey, P.J. Method of quantifying soil carbon. U.S. Patent Application 13/701, Patent No. 2011261179, 2011; 786 pp.
4. Batjes, N.H. Total carbon and nitrogen in the soils of the world. *Eur. J. Soil Sci.* **1996**, *47*, 151–163.
5. Batjes, N.H. Mapping soil carbon stocks of Central Africa using SOTER. *Geoderma* **2008**, *146*, 58–65.
6. Bernoux, M.; Arrouays, D.; Cerri, C.C.; Bourennane, H. Modeling vertical distribution of carbon in oxisols of the Western Brazilian Amazon (Rondonia). *Soil Sci.* **1998**, *163*, 941–951.
7. Hiederer, R.; Köchy, M. *Global Soil Organic Carbon Estimates and the Harmonized World Soil Database*; Global Soil Organic Carbon: Ispra, 2011; 79 pp.

8. Scharlemann, J.P.W.; Tanner, E.V.J.; Hiederer, R.; Kapos, V. Global soil carbon: Understanding and managing the largest terrestrial carbon pool. *Carbon Manage.* **2014**, *5*, 81–91.
9. Moore, I.D.; Gessler, P.E.; Nielsen, G.A.; Peterson, G.A. Soil attribute prediction using terrain analysis. *Soil Sci. Soc. Am. J.* **1993**, *57*, 443–452.
10. McBratney, A.B.; Mendonça Santos, M.L.; Minasny, B. On digital soil mapping. *Geoderma* **2003**, *117*, 3–52.
11. Malone, B.P.; McBratney, A.B.; Minasny, B.; Laslett, G.M. Mapping continuous depth functions of soil carbon storage and available water capacity. *Geoderma* **2009**, *154*, 138–152.
12. Hengl, T.; de Jesus, J.M.; MacMillan, R.A.; Batjes, N.H.; Heuvelink, G.B.; Ribeiro, E.; Samuel-Rosa, A.; Kempen, B.; Leenaars, J.G.; Walsh, M.G. SoilGrids 1 km—global soil information based on automated mapping. *PLoS One* **2014**, *9*, e105992.
13. Poggio, L.; Gimona, A. National scale 3D modelling of soil organic carbon stocks with uncertainty propagation – An example from Scotland. *Geoderma* **2014**, *232–234*, 284–299.
14. Bui, E.; Henderson, B.; Viergever, K. Using knowledge discovery with data mining from the Australian soil resource information system database to inform soil carbon mapping in Australia. *Global Biogeochem. Cy.* **2009**, *23*. Article No. GB4033, DOI:10.1029/2009GB003506.
15. Viscarra Rossel, R.A.; Webster, R.; Bui, E.N.; Baldock, J.A. Baseline map of organic carbon in Australian soil to support national carbon accounting and monitoring under climate change. *Glob. Change Biol.* **2014**, *20*, 2953–2970.
16. Adhikari, K.; Hartemink, A.E.; Minasny, B.; Kheir, R.B.; Greve, M.B.; Greve, M.H. Digital mapping of soil organic carbon contents and stocks in Denmark. *PLoS One* **2014**, *9*, e105519.
17. Chaplot, V.; Bouahom, B.; Valentin, C. Soil organic carbon stocks in Laos: Spatial variations and controlling factors. *Glob. Change Biol.* **2010**, *16*, 1380–1393.
18. Dorji, T.; Odeh, I.O.A.; Field, D.J.; Baillie, I.C. Digital soil mapping of soil organic carbon stocks under different land use and land cover types in montane ecosystems, Eastern Himalayas. *Forest Ecol. Manag.* **2014**, *318*, 91–102.
19. Karunaratne, S.B.; Bishop, T.F.A.; Baldock, J.A.; Odeh, I.O.A. Catchment scale mapping of measureable soil organic carbon fractions. *Geoderma* **2014**, *219–220*, 14–23.
20. Veronesi, F.; Corstanje, R.; Mayr, T. Landscape scale estimation of soil carbon stock using 3D modelling. *Sci. Total Environ.* **2014**, *487*, 578–586.
21. Razakamanarivo, R.H.; Grinand, C.; Razafindrakoto, M.A.; Bernoux, M.; Albrecht, A. Mapping organic carbon stocks in eucalyptus plantations of the central highlands of Madagascar: A multiple regression approach. *Geoderma* **2011**, *162*, 335–346.
22. Miklos, M.; Short, M.G.; McBratney, A.B.; Minasny, B. Mapping and comparing the distribution of soil carbon under cropping and grazing management practices in Narrabri, north-west New South Wales. *Aust. J. Soil Res.* **2010**, *48*, 248–257.
23. Vasenev, V.; Stoorvogel, J.J.; Vasenev, I.I.; Valentini, R. How to map soil organic carbon stocks in highly urbanized regions? *Geoderma* **2014**, *226–227*, 103–115.
24. Rawlins, B.G.; Henrys, P.; Breward, N.; Robinson, D.A.; Keith, A.M.; Garcia-Bajo, M. The importance of inorganic carbon in soil carbon databases and stock estimates: A case study from England. *Soil Use Manage.* **2011**, *27*, 312–320.
25. Angers, D.A.; Arrouays, D.; Saby, N.P.A.; Walter, C. Estimating and mapping the carbon saturation deficit of French agricultural topsoils. *Soil Use Manage.* **2011**, *27*, 448–452.
26. Rawlins, B.G.; Marchant, B.P.; Smyth, D.; Scheib, C.; Lark, R.M.; Jordan, C. Airborne radiometric survey data and a DTM as covariates for regional scale mapping of soil organic carbon across Northern Ireland. *Eur. J. Soil Sci.* **2009**, *60*, 44–54.
27. Arrouays, D.; Grundy, M.G.; Hartemink, A.E.; Hempel, J.W.; Heuvelink, G.B.M.; Hong, S.Y.; Lagacherie, P.; Lelyk, G.; McBratney, A.B.; McKenzie, N.J.; Mendonca-Santos, M.D.; Minasny, B.; Montanarella, L.; Odeh, I.O.A.; Sanchez, P.A.; Thompson, J.A.; Zhang, G.L. GlobalSoilMap: Toward a fine-resolution global grid of soil properties. *Adv. Agron.* **2014**, *125*, 93–134.
28. Odgers, N.P.; Libohova, Z.; Thompson, J.A. Equal-area spline functions applied to a legacy soil database to create weighted-means maps of soil organic carbon at a continental scale. *Geoderma* **2012**, *189–190*, 153–163.
29. Viaud, V.; Angers, D.A.; Walter, C. Towards landscape-scale modeling of soil organic matter dynamics in agroecosystems. *Soil Sci. Soc. Am. J.* **2010**, *74*, 1–14.

Mapping Soil Carbon: Stocks in Turkey

Gönül Aydın

*Department of Soil Science and Plant Nutrition, Adnan Menderes University,
South Campus, Aydın, Turkey*

Mehmet Ali Çullu

Department of Soil Science and Plant Nutrition, Harran University, Şanlıurfa, Turkey

Sabit Erşahin

Department of Forestry Engineering, Çankırı Karatekin University, Çankırı, Turkey

Erhan Akça

Center of Remote Sensing, Adiyaman University, Adiyaman, Turkey

Emrah Erdoğan

*General Directorate of Agrarian Reform, Ministry of Food, Agriculture and
Livestock, Ankara, Turkey*

Levent Atatanır

Alper Yorulmaz

*Department of Soil Science and Plant Nutrition, Adnan Menderes University,
South Campus, Aydın, Turkey*

Ahmet Çilek

Merve Ersoy

Department of Landscape Architecture, University of Cukurova, Adana, Turkey

Somayyeh Rezzaghi Miavaghi

Department of Soil Science and Plant Nutrition, University of Cukurova, Adana, Turkey

Selim Kapur

*Departments of Soil Science, Landscape Architecture, and Agricultural
Engineering, University of Cukurova, Adana, Turkey*

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

The soil organic carbon (SOC) stocks of each geographical region based on the World Reference Base (WRB) soil classes of the country were determined seeking their contemporary and/or future use for national crop/soil use policies and for the credible assessment to mitigate atmospheric carbon dioxide and restore degraded soils. Once the SOC stocks are mapped, it is important to take the necessary measures for improving land management for increasing SOC stock and enhancing quality of agricultural soils. Total SOC stock of soil in Turkey is 9.23 Pg up to 0.7 m depth, and annual emission from the agricultural sector is 25.7 Tg C.

INTRODUCTION

The total carbon dioxide (CO₂) emissions of 25.7 Tg (Tg = teragram = 10¹² g = 1 million metric ton) from the agricultural sector (Turkish Statistical Institute, 2011) were only about 7% of the total national emissions (369.7 Tg) for

Turkey in 2009.^[1] The carbon-sink capacity of the two land-based reservoirs in Turkey, namely the soils and vegetation, is affected by anthropogenic activities such as soil sealing and land use change. Drastic changes in land use and the resultant degradation of soil quality have depleted the carbon (C) reserves in the country, i.e., a reduction in soil

organic matter (SOM) content of the managed soils and, in turn, of the forest ecosystems. Nevertheless, there exists a vast potential to sequester C through conservation and restoration of soil quality and improvement in the vegetation cover of the different Terroirs and/or Anthroscapes or human-resaped historical landscapes^[2,3] of Turkey by “sustainable land management.” Thus, the objective of this study was to prepare the soil organic carbon (SOC) stock map and assess the C-sequestration potential of the Turkish soils.

MATERIALS AND METHODS

The first step in mapping the SOC stocks of the soils of Turkey was accomplished by the partial renovation of the previous soil maps prepared by the General Directorate (GD) of Agricultural Reform (the previous GD of Rural Affairs and GD of Soil and Water) of the Ministry of Food, Agriculture, and Livestock of Turkey via site-specific field work and profile descriptions based upon expert knowledge. Additionally, recently accomplished 5368 SOM analyses, of the 0–20 cm layer, by the modified Walkley Black Method,^[4] for soils of different slightly-modified Köppen–Geiger climatic regions (Central, Western, Eastern, South Eastern, and Northern Anatolia regions and the region of the Mediterranean–Aegean–Northern Marmara)^[5,6] were compiled for C-stock calculations. Moreover, land use and plant cover characteristics were integrated in the SOM database. The SOM analyses were provided from the laboratories of the Universities of Adnan Menderes, Çukurova, Ege, Harran, Selçuk and 19 Mayıs and the Institutes of Central Research

of Soil, Fertilisers and Water and Soil, Water in Ankara and Agricultural Meteorology Research of Atatürk in Kırklareli.

The SOM values of the World Reference Base (WRB) soil classes were extended countrywide for each of the six geographical soil regions by means of geographic information system, i.e., the SOM values of each WRB soil groups of a specific region were extended/assigned (pedo-transferred via local expert knowledge) to similar soils of that geographical region. Ultimately, the Carbon Stocks Map of the Soils of Turkey was accomplished via the formula given according to the WRB (2014) soil map for each soil group at 0–20 cm depth.

$$\text{SOC stock (Mg ha}^{-1}\text{)} = \sum_i^n \text{soil volume (m}^3 \text{ ha}^{-1}\text{)} \\ \times \text{soil bulk density (Mg m}^{-3}\text{)} \\ \times \text{SOC concentration (Mg Mg}^{-1}\text{)} \\ \times \text{area (ha)}$$

where n is the number of the soil layers (0–20 cm soil depth only). The bulk density was calculated by the Saxton texture triangle.^[7] Data reported as SOM concentration were converted to SOC by multiplying with a conversion factor of 0.58.^[8]

RESULTS AND DISCUSSION

An enhanced diversity in the type of soils of the country was determined based on the highly variable and undulating topography, parent materials, vegetation, and

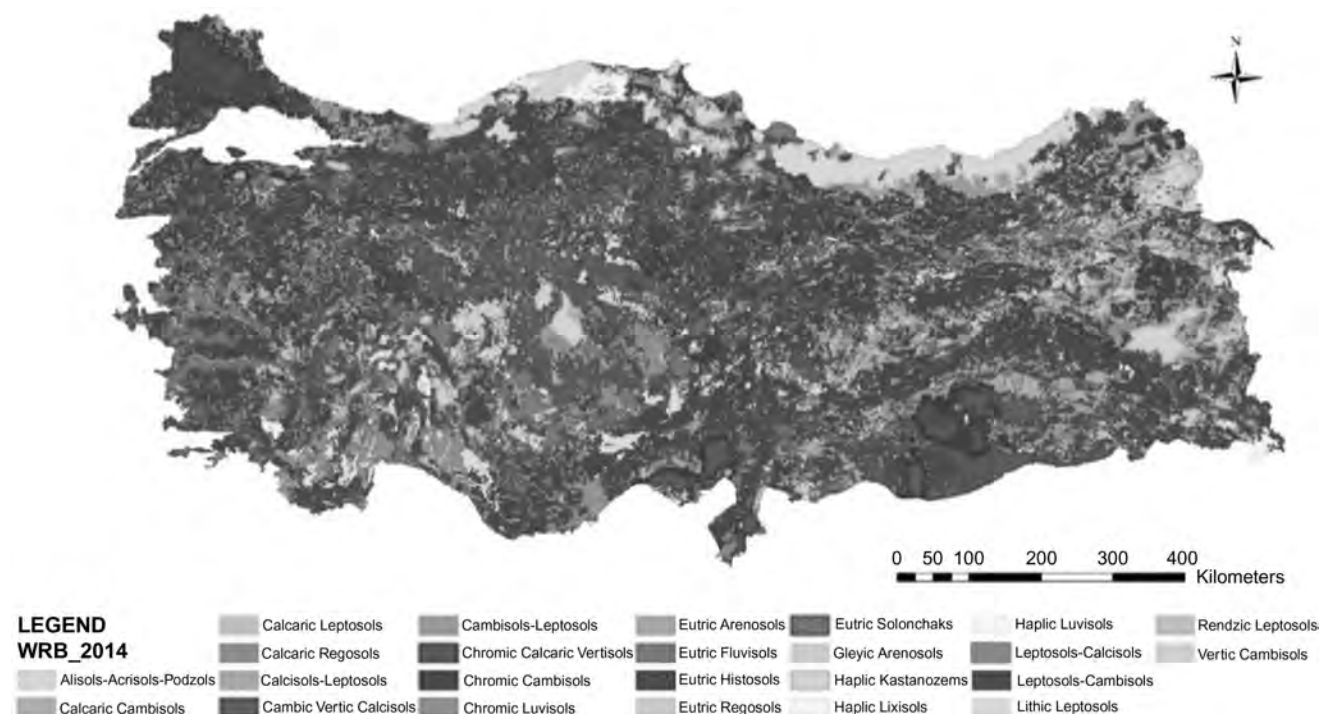


Fig. 1 The soil map of Turkey according to the IUSS Working Group on WRB (2014).

the effects of the present and/or past climates on soil development with evidence of prevalence at numerous locations (Fig. 1).

In general, higher SOC stocks occur in the Mediterranean, Aegean, and Northern Marmara and Northern Anatolia, Central Anatolia, and the lowest in Southeastern Anatolia (Fig. 2). The SOC stocks follow a *decreasing* order from West Anatolia to the East and Southeast. The higher SOC stock of the Mediterranean region (706.22 Tg) than that of the Black Sea region (501.85 Tg), despite higher precipitation (~2500 mm annual rainfall) and dense vegetation cover (coniferous forests and cultivated *Corylus avellana*, *Camellia sinensis*, and *Zea mays*; Table 1 and Fig. 2) of the latter, are due to its larger land area. On the other hand, the Tauride Mountain forests of the Mediterranean region, such as the maquis (the Mediterranean scrub), the traditional olives, and carobs of the Tauride slopes are the major land cover which increase the SOC stock of the Mediterranean and Aegean regions. Koçak and Kapur^[9] and Rezzaghi et al.^[10] have determined high amounts of SOM in the red pine–oak community (11%), maquis (10%), *Olea europaea* (≥7%), and *Ceratonia siliqua* (≥8%in the) root-zone soils of the eastern Mediterranean

Table 1 Areas and SOC stocks according to the regions.

Regions	Area (ha)	SOC stock (Tg)
Central Anatolia	9,467,349.77	249.33
Mediterranean, Aegean, and North of the Marmara Regions	23,806,283.69	706.22
Northern Anatolia	10,893,328.70	501.85
Western Aegean	13,499,603.54	332.26
Eastern Anatolia	14,551,530.26	278.23
Southeastern Anatolia	7,541,446.08	158.97
Total	79,759,542.05	2,226.85
		2.23 (Pg)

part of Turkey, i.e., the eastern Tauride slopes. The widely spread Calcisols and Cambisols of Central Anatolia (Fig. 1) and the Cambisols and Leptosols of eastern Anatolia have been used extensively as small grazing rangelands since the Roman and Ottoman periods and for the cultivation of rainfed wheat. These soils along with the inland Arenosols (Fig. 1) of the Konya-Karapınar area are the potentially high C sequestering soils of the country, if appropriately

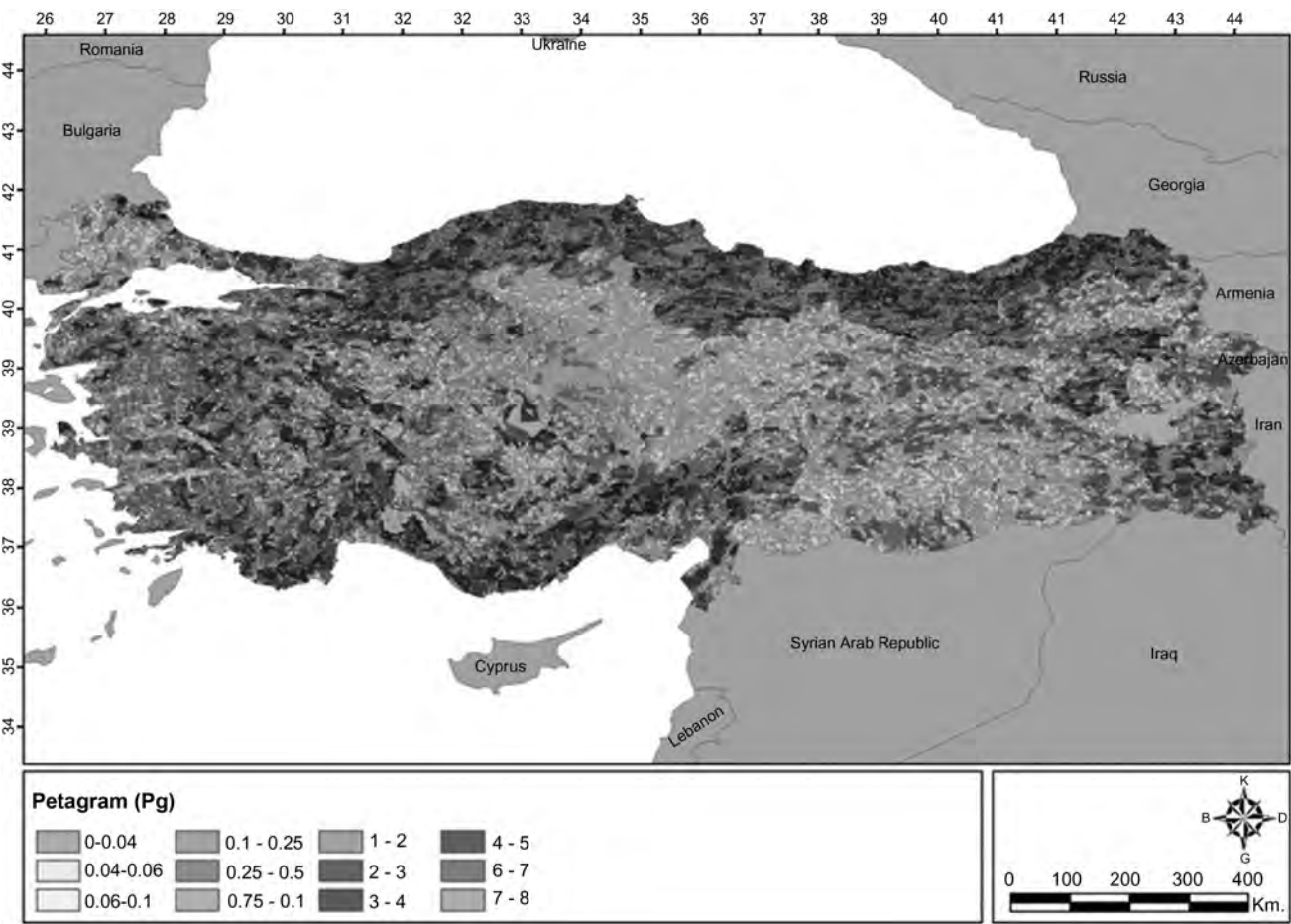


Fig. 2 SOC stocks of Turkey.

managed.^[11] This area is one of the bread baskets of Turkey together with eastern Anatolia area used for cultivation of wheat irrigated with ground water and surface sources collected in reservoirs. The coastal Arenosols and that include the wetlands are also the potentially high C sequestering areas of the country. Yaktı et al.^[12] stated that the SOC contents of leveled Arenosols were increased several folds in 20 years thanks to governmental forest management projects under cypress and eucalypts canopies. Despite the contradictory views repeatedly stated by nature conservation scientists, Polat and Kapur^[13] revealed an eightfold increase within 40–50 years in the country indicating the high potential of Arenosols in sequestering C in the soil, under appropriate forest management.

CONCLUSION

The SOC stocks of Turkey for the top 20 cm (2.23 Pg C) are comparable to that of France's 3.26 ± 0.872 Pg C,^[14] a country almost as large as Turkey.^[15]

It is widely recognized that with judicious cultivation and management, despite the large vegetative diversity and high agricultural potential, Turkey is faced with a dilemma of soil degradation by salinization and desertification. Thus, recarbonization of soils of Turkey is critical to the mitigation of greenhouse gas emissions and restoration of soil fertility. Further, the SOC stock must be credibly assessed to mitigate atmospheric CO₂ and restore degraded soils. Once the SOC stocks are mapped, it is important to take the necessary measures for improving land management for increasing SOC stock and enhancing quality of agricultural soils.

REFERENCES

1. Turkish Statistical Institute. *Greenhouse Gas Emissions Inventory, 1990–2011, No: 13482*. 2013. Available at <http://www.turkstat.gov.tr/PreHaberBultenleri.do?id=13482> (accessed October 16, 2015).
2. Kapur, S.; Akça, E.; Özden, D.M.; Sakarya, N.; Çimrin, K.M.; Alagöz, U.; Ulusoy, R.; Dancı, C.; Kaya, Z.; Düzenli, S.; Gülcan, H. Land degradation in Turkey. In *Land Degradation in Central and Eastern Europe*. European Soil Bureau Research Report No. 10, EUR 20688 EN, 324; Jones, R.J.A., Montanarella, L., Eds.; Office for Official Publications of the European Communities: Luxembourg, 2003; 303–319.
3. Eswaran, H.; Berberoğlu, S.; Cangir, C.; Poyraz, D.; Zucca, C.; Özevren, E.; Yazıcı, E.; Zdruli, P.; Dingil, M.; Dönmez, C.; Akça, E.; Çelik, İ.; Watanabe, T.; Koca, K.Y.; Montanarella, L.; Cherlet, M.; Kapur, S. The Anthroscape approach in sustainable land use. In *Sustainable Land Management*; Kapur, S., Eswaran, H., Blum, W.E.H., Eds.; Springer: London/Berlin, 2013; 1–50.
4. Walkley, A. A critical examination of a rapid method for determining organic carbon in soils: Effect of variations in digestion conditions and of inorganic soil constituents. *Soil Sci.* **1947**, *63*, 251–263.
5. Köppen, W.; Geiger, G. *Klima der Erde (Map)*. Justus Perthes, Darmstadt, Germany. American Distributor; A. J. Nystrom and Co: Chicago, 1954.
6. Türkeş, M. Contemporary climate of Turkey. In *Turkey's Climatic Characteristics*. Available at <http://www.webhatti.com/kultur/49511-turkiyenin-iklim-ozellikleri.html#ixzz208QP7Xcwhkaynak,2014> (accessed October 16, 2015).
7. Saxton, K.E.; Rawls, W.J.; Romberger, J.S.; Papendick, R.I. Estimating generalized soil water characteristics from texture. *Soil Sci. Soc. Am. J.* **1986**, *50*, 1031–1036.
8. Zinn, Y.L.; Lal, R.; Resck, D.V.S. Changes in soil organic carbon stocks under agriculture in Brasil. *Soil Till. Res.* **2005**, *84*, 28–40.
9. Koçak, Z.; Kapur, S. Determination of some quality parameters of the Olive and Carob root-zone soils in the eastern Mediterranean region of Turkey. *J. Sci. Eng.* **2011**, *25* (2), 128–138.
10. Rezzaghi, M.S.; Kapur, S.; Şenol, S. Mineralogical and micromorphological characteristics of red pine (*Pinus pinea*) and oak (*Quercus coccifera*) root zone soils in the Göksu catchment. *J. Inst. Sci.* **2014**, in press.
11. Akça, E. Determination of the Soil Development in Karapınar Erosion Control Station Following Rehabilitation. Ph.D. Thesis. University of Çukurova, Institute of Natural and Applied Sciences, Adana, Turkey.
12. Yaktı, S.; Akça, E.; Kapur, S. Management of coastal sand dunes by an agroecosystem approach. In *Workshop Proceedings of Determining an Income Product Generating Approach for Soil Conservation Management*; Zdruli, P., Liuzzi, G.T., Eds.; February 12–16, 2004, Marrakech, Morocco; CIHEAM; 227–234.
13. Polat, O.; Kapur, S. Soil quality parameters in Arenosols under Stone Pine (*Pinus Pinea* L.) canopy in the Turan Emeksiz sand dune Area, Tarsus, Mersin. *J. Sci. Eng.* **2010**, *3* (2), 167–174.
14. Martin, M.P.; Wattenbach, M.; Smith, P.; Meersmans, J.; Jolivet, C.; Boulonne, L.; Arrouays, D. Spatial distribution of soil organic carbon stocks in France. *Biogeosciences* **2011**, *8*, 1053–1065.
15. Martial, B.; Maria da Conceição, S.C.; Boris, V.; Carlos, C.C. Soil science society. Brazil's soil carbon stocks. *Soil Sci. Soc. Am. J.* **2002**, *66*, 888–896.

Materials and Soils

Selim Kapur

Departments of Soil Science, Landscape Architecture, and Agricultural Engineering, University of Cukurova, Adana, Turkey

Erhan Akça

Center of Remote Sensing, Adiyaman University, Adiyaman, Turkey

Takanori Nagano

Graduate School of Agricultural Science, Kyoto University, Kobe, Japan

Abstract

Soil has long been used for manufacturing diverse materials alongside its major function as the medium of plant growth offering all the knowledge attained over millennia via its consumers. Its major use as an ingredient of pottery and construction materials is a process that causes the irreversible overuse of this highly precious natural material. However, contemporary land use approaches such as the Anthroscape—a renovated land use approach based on the traditional knowledge attained by long-standing experience—promise a sustainable and appropriate use of soil.

INTRODUCTION

A Historical Perspective of Appropriate Use of Soil/Land

The soil, alongside its numerous uses, has been misused for the purpose of making pottery and constructing buildings since millennia by many civilizations, particularly, in the Mediterranean Basin, the cradle of the ancient societies. Fortunately, the initial and local efforts to limit such misuse were followed by the establishment of law and order during the Roman Empire. Terraces were built along the coastal slopes embellishing the bountiful nature that was and still is generous to all its inhabitants by conserving the soil and its water content. Such measures ultimately lead to the establishment of agroecosystems with dominant crop/tree systems such as the olive orchards of the Emperor Sebastian (Elaiussa Sebaste) as well as in Kızkalesi (Korykos) in Erdemli, Mersin, Turkey.^[1]

The terraces were the products of a unique long-developed concept through the integrated thinking that had advanced since the teachings of Aristotle of the School of Athens on the protection and management of nature. The Aristotelian holistic/organismic context developed for nature was based on the strong relationship and harmony between biotic and abiotic components (i.e., the living animals and plants and the non-living things such as rocks and soils). [Due to a demand for the implementation of integrated sustainable land and water management programs, which are in vain sought the Aristotelian holistic/organismic context developed for nature was based on the strong

relationship and harmony of the inanimate and animated components, i.e., the living (e.g., animals and plants) and the non-living things (e.g., the rocks and soils).]

The promising approach of the “Anthroscape” is based on reshaped long-standing human landscape that seeks a continuum in the Anthropocene. The concept was proposed by Crutzen,^[2] it delineates land degradation since the last 300 years^[3] and attempts to reconstruct a socioeconomic and physical methodology with regional indigenous specificities for the people. It is most likely the contemporary version of the Aristotelian concepts about nature and its management. This approach recommends the determination of sample sites of variable scales with particular and unique traditional knowledge throughout the developed and developing world and the development of sustainable land and water management programs. Finally, the concept also allocates land for industry, agriculture, and forestry, while it permits the extraction of raw materials for civil engineering structures.

Soil as a Building Material

Laterite

Laterite, formed by pedogenic processes, is a traditional building material. It has been used extensively as bricks in constructing local houses in India, throughout the Indian subcontinent, other parts of Asia, and South America. The sturdy buildings constructed from laterite have been proven to be resistant to the highly degrading effects of nature during the monsoon period as well as during the hot and

wet/dry summer months. Laterite houses are also preferred for their durability against earthquakes. These houses were also highly popular during the historical periods in Cambodia, which used laterite for the construction of the famous Angkor monuments^[4] and which endured the extreme environment for millennia. Laterite, a naturally occurring soil to civil engineers and an ultimate weathering product to pedologists, is a soil/material rich in goethite, hematite, kaolinite, and quartz. It is being extensively introduced for use as an additive in durable and cheap concrete in the tropics and subtropics. The material is also used as a “filler” for foundations, as a base material for highway construction, and for economic building “Blocks.”^[5]

Making cement

Soil materials are used in the manufacturing of cement and ultimately concrete, which are referred to as soil cement/roller-compacted concrete (RCC). The RCC has a zero-slump consistency. It is placed and compacted with earth-moving or paving equipment as soil cement. Soil cement is also known as the cement-modified soil and cement-treated aggregate base with materials denoted as pumice (scoria, expanded perlite, tufa, zeolite, etc.).

Soil cement is a dense, highly compacted mixture of soil or roadway material with Portland cement and water. Soil material can consist of any combination of sand, silt, clay, gravel, or crushed stone. Granular soils are preferred, however, because they pulverize more easily and require less cement to achieve the required strength and durability.^[6] Cement–soil mortars have been found to have a high tensile bond strength for use in soil–cement block couplets when compared to cement mortar and cement lime mortar.^[7]

As the soil cement is placed and compacted, the cement hydrates and the mix becomes a structural slab-like material. After construction and curing, soil cement is hardly affected by water or the freeze–thaw cycle. Therefore, it does not slump under construction traffic or rut during spring thaws, and it can also bridge over soft subgrade. This is the primary requirement for attaining high-quality and long-lived concrete, whereby the process of hydration has reached a state of maturity with the optimal formation and distribution of end-product minerals such as ettringite and portlandite^[8] (Fig. 1). The perfection of the hydration process is most likely due to the optimal ionic character of the soils and their pH. These properties influence the speed and uniformity of the ongoing reactions which make the unique Anatolian mortar—the Horasan mortar, and that was extensively used by the Romans, by Byzantines, and later by the Seljuks.

The contents of crushed/ground brick and lime, long slaked and softly burnt,^[9] together with the illite or smectite containing dominantly kaolinitic soil impart a unique character to the hydraulic/hydration properties of the Horasan mortar. In addition to its economic merits, Roman initial Horasan mortars with tufa fragments containing clay/soil

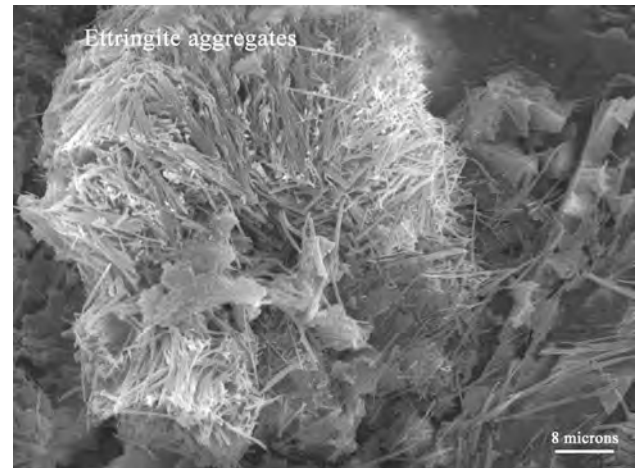


Fig. 1 Scanning electron microscopy (SEM) image of aggregates of clewing ettringites developing during hydration.

Source: Photo courtesy of Dr. Hanifi Binici, K.S. University, Kahramanmaraş, Turkey.

materials were also probably leached. Clay coatings onto the sedimentary tufa may have affected the binding strength as well as the consistency in the speed of the long slaking and slow burning of the Roman mortars recovered from several Spanish sites.^[10]

Soil in Earthfill Dams

Soils, other than the prime quality suitable for cultivation and agro-ecosystem management, are the most important components affecting the construction of earthfill embankments by their distribution, method of placement, water content, and compaction. Soils are mainly classified by their engineering properties into two main divisions using their textural characteristics, as soils of either coarse grains or fine grains. Coarse grains are those >200 sieve mesh size and include rocks, gravels, and sands. Fine grains are <200 sieve mesh size and include silts and clays. Coarse-grain material is used for the outer zones of an earthfill embankment, and fine-grain material is used for the impervious core or central portion of the dam.

The soil material must be placed in horizontal layers no more than 15 cm thick after being compacted and should be homogeneous and free from lenses, pockets, organic material, or other imperfections. Prior to placement, the material should have the optimum moisture content required for compaction. The optimum moisture content, or the water content that produces the maximum density, may be obtained by the Proctor test. Good compaction of a cohesive soil reduces permeability and increases the shear strength and the stability of the dam. Clay mineral contents of the fine core materials are primarily significant as the clay/soil core settles even after the construction of the dam and/or the completion of the field compaction process. Soil/clay cores extracted from soil types rich in smectite

(especially nontronite) tend to expand on wetting and create potentially disruptive forces in rockfill sections. These soils settle differentially and lead to erosion of the core, thus jeopardizing the overall safety of the dam.^[11–13]

SOIL IN CERAMICS

Ancient Pottery

Ancient India of the Indus Valley civilizations (Harappa, Mohenjo Daro, and Lothal) had marvellous craftsmen skilled in pottery. The necessary raw material sources/alluvial soils (for making high-quality products like terracotta figurines) were bountiful throughout the Indus Basin. These materials were replenished year-round by the sequential deposition of soil via flooding by the waters collected from the Hindu Kush ranges.^[14,15] Much earlier, the Neolithic Çatalhöyük from Anatolia (the modern Turkey of today) was the cradle of civilization. People from this region were sustainably using soils and sediments for making mud bricks and pottery. These observations were based on the clues from the utilization of plant remains as well as the existence of suitable soil/clay sources in the vicinity (Fig. 2).^[16] The Hittites, who lived in present-day Turkey, were the masters of firing technologies in constructing updraft furnaces and of using the most appropriate soil/clay sources to successfully attain high and somewhat consistent temperatures as is indicated by the high vitrification observed in high-resolution electron images (Fig. 3).^[17] The Urartus, the early settlers of eastern Turkey (800–600 B.C.), were the expert craftsman of ceramics and were highly proficient in the firing technologies and production of vitrified slipware by the use of appropriate volcanic soil materials.^[18] The late Seljuks and Ottomans of İznik (the Byzantine Nycheia withholding the primary

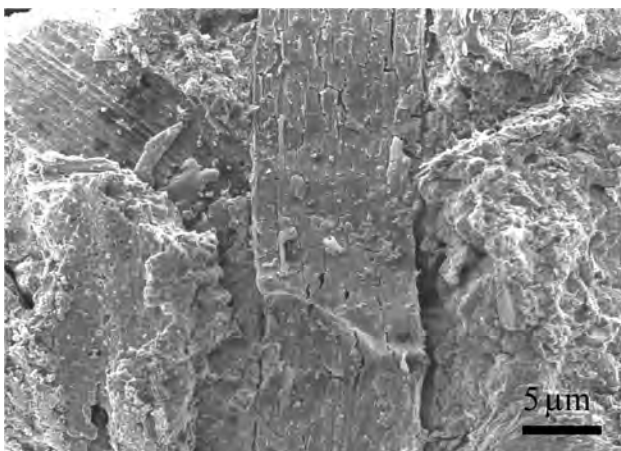


Fig. 2 Broken reed remnants in a Neolithic shard from Çatalhöyük, Central Anatolia (SEM image).

Source: Data from the archive of Department of Archeometry, University of Çukurova, Adana, Turkey.

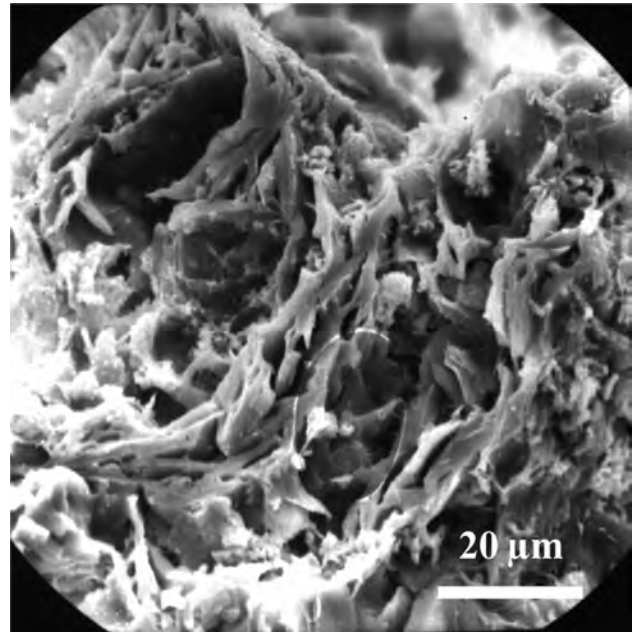


Fig. 3 Intergrown quasi-linear stress cutans and curved cutans developed around pores in matrix of a Hittite ceramic in Karatepe Aslantaş Museum.

Source: Data from the archive of Department of Archeometry, University of Çukurova, Adana, Turkey (SEM image).

clues of Anatolian pottery) discovered the superiority of buried furnaces with uniform and gradual low-rate increase of temperature along with the use of earth/soil materials rich in kaolinite and quartz as well as calcite, acting as a flux in heat control. These highly appropriate soil materials were located on the ancient terraces/shores of the Lake İznik and on the volcanic highlands of central and southern Turkey, which leads to the development of an ultimately superb microstructure (with high-temperature minerals) of high durability and artistic properties (Figs. 4 and 5).

The renowned Deruta ceramics from Umbria, Italy, belonging to the Renaissance period, were manufactured from soils referred to as the clay raw materials. This was also the case in the production of indigenous ceramics throughout the historical periods. The material used was rich in calcite and was the driving force in the kinetics of firing together with the low heating rate.^[19] The process was similar to the earlier but highly advanced technology of the İznik ceramics of Turkey.^[20]

Advanced Ceramics

Besides the “traditional ceramics,” there is a new class of ceramics, called “advanced ceramics.” It emerged in the 20th century as the material systems became more refined, and special compounds and processes were developed for structural and electronic applications. These advanced ceramics

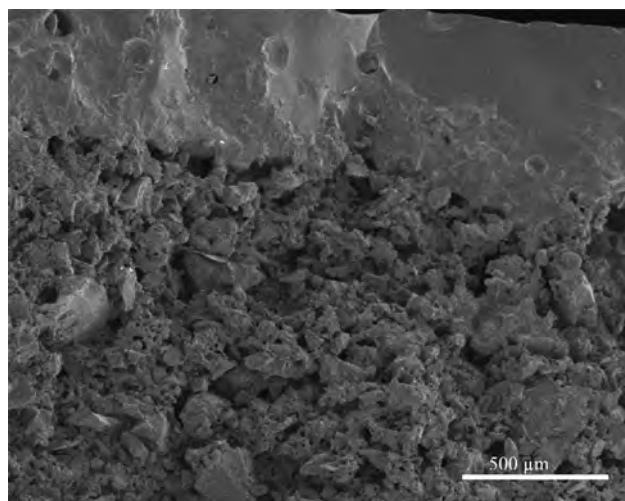


Fig. 4 Uniform glaze and coating in Iznik Ceramic (13th century).

Source: Data from the archive of the Department of Archeometry, University of Çukurova, Adana, Turkey.

are distinguished by their high chemical purity, careful processing, and high values of the useful properties.^[21]

Despite the development of the advanced high-technology ceramics, traditional ceramic production is used at a high rate due to the renovation of the furnace technologies and the increasing knowledge on the selection of the raw materials. Among the numerous contemporary examples for this are the studies undertaken by Kapur et al.^[17,22] and Kelling et al.^[23] These studies specifically

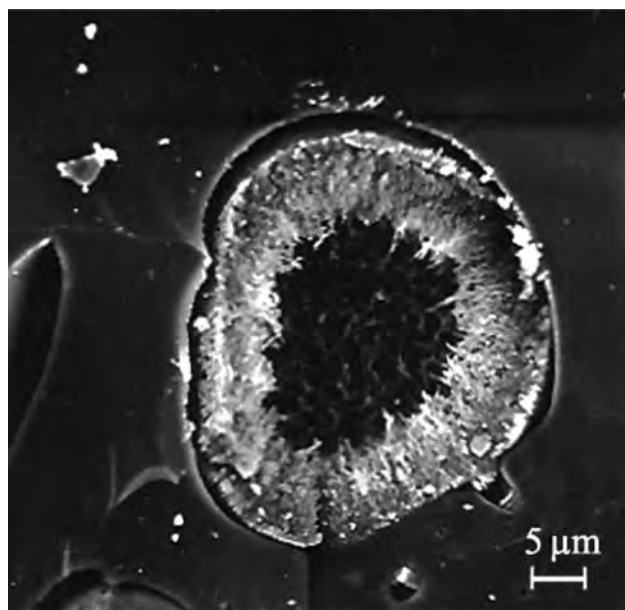


Fig. 5 Fibrous high-temperature minerals developing in glaze pore of a Seljuk ceramic (12th century).

Source: Data from the archive of the Department of Archeometry, University of Çukurova, Adana, Turkey.

involve the use of novel raw materials—rocks—such as basalts along with soils formed in/on basaltic rocks with conventional furnace technologies. These technologies have been renovated for economic and sustainable production, as well as for generating products that are more resistant to physical and chemical stresses.^[24]

ACKNOWLEDGMENTS

The authors are deeply indebted to Profs. E.A. FitzPatrick (University of Aberdeen, U.K.), Gilbert Kelling (University of Keele, U.K.), and Halet Çambel (Karatepe Open Air Museum, Turkey) for their invaluable contributions to the more than two decade-long studies conducted on the ancient ceramic materials in the Department of Archeometry, University of Çukurova. The principal author is also highly indebted to the valuable review undertaken by the editor.

REFERENCES

1. Akca, E.; Kelling, G.; Kapur, S.; Bal, Y.; Gültekin, E.; Everest, A.; Yetiş, C.; Şenol, S.; Darıcı, S. Preserving our heritage: A case study from southern Anatolia. In *Problems of Urban Growth: Preserving While Developing*, Conference Proceedings of International Urban Fellow Association, Pub of the Mersin Univ., Faculty of Architecture, 2002; 68–76. Proceedings of the 31st Annual Conference International Urban Fellows Program, Annual IUFA Conference, Mersin, Turkey, Jun. 9–15, 2001.
2. Crutzen, P.J. The Anthropocene: Geology of mankind. *Nature* **2002**, *415*, 23.
3. Eswaran, H.; Kapur, S.; Akça, E.; Reich, P.; Mahmoodi, S.; Veerasilp, T. Anthrosapes: A landscape unit for assessment of human impact on land systems. In *Application of the Emerging Soil Research to the Conservation of Agricultural Ecosystems*; Yang, J.E., Sa, T.M., Kim, J.J., Eds.; Korean Society of Soil Science and Fertilizers: Seoul, 2005; 175–192.
4. Uchida, E.; Maeda, N.; Nakawa, T. The laterites of Angkor monuments, Cambodia. The grouping of the monuments on the basis of the laterites. *J. Miner. Petrol. Econ. Geol.* **1999**, *94*, 162–175.
5. Udoeyo, F.; Iron, U.; Odum, O. Strength performance of laterized concrete. *Constr. Build. Mater.* **2006**, *20* (10), 1057–1062.
6. Choi, Y.K.; Hansen, K.D. RCC/soil–cement: What's the difference? *J. Mater. Civil Eng.* **2005**, *17* (4), 371–378.
7. Reddy, V.B.V.; Gupta, A. Characteristics of cement–soil mortars. *Mater. Struct.* **2005**, *38* (280), 639–650.
8. Binici, H.; Çağatay, İ.H.; Shah, T.; Kapur, S. Mineralogy of plain Portland and blended cement pastes Elsevier. *Build. Environm.* **2008**, *43* (7), 1318–1325.
9. Binici, H.; Arocena, J.; Kapur, S.; Aksoğan, O.; Kaplan, H. Investigation of the physico-chemical and microscopic properties of Ottoman mortars from Erzurum (Turkey). *Constr. Build. Mater.* **2010**, *24* (10), 1995–2002.

10. Pavia, S.; Caro, S. An investigation of Roman mortar technology through the petrographic analysis of archaeological material. *Constr. Build. Mater.* **2008**, *22* (8), 1807–1811.
11. Goodman, R.E. *Engineering Geology: Rock in Engineering Construction*; John Wiley & Sons: Singapore, 1993; 432 pp.
12. Çetin, H.; Laman, M.; Ertunç, A. Settlement and slaking problems in the world's fourth largest rock-fill dam, the Atatürk Dam in Turkey. *Eng. Geol. Int. J.* **2000**, *56*, 225–242.
13. Grishin, M.M.; Sliskii, S.M.; Rasskazov, L.N. Earth-rock dams. Principles of design and construction. *Power Technol. Eng.* **1978**, *12* (6), 637–641.
14. Greenwood, C. *Economics of the Indus Valley Civilization*, 2007. Available at <http://www.csuchico.edu/~cheinz/syllabi/asst001/fall97/2chd.htm> (accessed June 2007).
15. Kharakwal, J.S. Indus civilization: An overview. In *Indus Civilization, Text and Context*; Osada, T., Ed.; Indus Project of the Research Institute for Humanity and Nature: Kyoto, 2007; 15–59.
16. Düring, B.S. Cultural dynamics of the Central Anatolian Neolithic: The early ceramic Neolithic–late ceramic Neolithic transition. In *The Neolithic of Central Anatolia, Internal developments and external relations during the 9th–6th millennia CAL BC*, Istanbul, Nov. 23–24, 2001. Proceedings of the International CANeW Table Ronde; Gerard, F., Thissen, L., Eds.; Ege Publications: Istanbul, 2002; 219–236.
17. Kapur, S.; Sakarya, N.; Fitzpatrick, E.A. Mineralogy and micromorphology of chalcolithic and early bronze age İkiztepe ceramics. *Geoarchaeology* **1992**, *7* (4), 327–337.
18. Akça, E.; Arocena, J.; Kılıç, S.; Dingil, M.; Kapur, S. Preliminary chemical and micromorphological observations on Urartu (800–600 B.C.) ceramics, Eastern Turkey. *Geoarchaeology* **2010**, *25* (2), 233–244.
19. Moroni, B.; Conti, C. Technological features of Renaissance pottery from Deruta (Umbria, Italy): An experimental study. *Appl. Clay Sci.* **2006**, *33* (3–4), 230–246.
20. Kapur, S.; Sakarya, N.; Fitzpatrick, E.A.; Pagliai, M.; Kelling, G.; Akça, E.; Karaman, C.; Sakarya, B. Mineralogy and micromorphology of İznik ceramics. *Anatolian Stud. J. Brit. Inst. Archaeol. Ankara* **1998**, *48*, 181–189.
21. Taylor, D.A. Advanced ceramics. *Mater. Austr.* **2001**, *33* (1), 20–22.
22. Kapur, S.; Sakarya, N.; Karaman, C.; Fitzpatrick, E.A.; Pagliai, M. Micromorphology of basaltic ceramics. *Brit. Ceram. T.* **1995**, *94* (1), 33–37.
23. Kelling, G.; Kapur, S.; Sakarya, N.; Akça, E.; Karaman, C.; Sakarya, B.; Robinson, P. Basaltic tephra: Potential new resource for ceramic industry. *Brit. Ceram. T.* **2000**, *99* (3), 129–136.
24. Kapur, S.; Akça, E.; Sakarya, N.; Karaman, C.; Sakarya, B.; Kelling, G.; Robinson, P. A preliminary study on basaltic tephra-rich ceramic bodies. In *8th National Clay Symposium*, Ankara, Sep. 24–27, 1997, University of Dumlupınar: Kütahya, Turkey, 1997; 353–368 [in Turkish].

Mediterranean Agriculture: Rainfed, Nitrogen Use Efficiency

Àngela D. Bosch Serra

Department of Environmental and Soil Sciences, University of Lleida, Lleida, Spain

Abstract

Mediterranean regions are the ones with a typical rainfall pattern, characterized by rainfall in winter and dry, hot summers. Strong water deficits occur for several months. Thus, nitrogen use efficiency (NUE) and its increase are linked to improvements in water availability and water use efficiency. NUE is lower in Mediterranean regions than in temperate areas, and large opportunities exist to improve it. Rainfed cereal crops are major nitrogen (N) consumers in these areas, and increases in NUE may be achieved through the proper design of the agricultural system. Proper timing of fertilizer operations, appropriate choice of N fertilizer type, and implementation of analytical tools adapted to the specific conditions are key issues, in addition to others, like conservation tillage or crop breeding. Some attention is paid to organic fertilization and more intensive rotations, avoiding bare fallow as a way to increase soil quality and overall NUE. To face these challenges, as the management complexity of the agricultural system increases, there is a strong need for developing channels for communication with land managers.

INTRODUCTION

Nitrogen use efficiency (NUE), a generic term in agronomic literature,^[1] is used here to indicate an overall nitrogen (N) efficiency; when possible, more specific terms are used.

Rainfed Mediterranean agriculture mainly encompasses cereals, such as barley and wheat (soft and durum), and tree crops such as almonds, olive trees, and vineyards.

Cereals are the greatest N fertilizer consumers in Mediterranean regions, a pattern that some authors^[1,2] signal to be worldwide, since cereals account for about 60% of N fertilizer.

The following discussion will mostly deal with agricultural practices and how improvements in N and agricultural system management may increase NUE. The discussion is restricted to areas that can support regular cultivation.

MEDITERRANEAN REGIONS

Mediterranean regions, associated with Mediterranean climate,^[3] are present in all continents.

The largest area lies between the temperate region (42° N) and the subtropical desert region (32° N), from the Eastern Atlantic coast to the Near East. Elsewhere, such denomination is given to South-Western Africa, western parts of America (California, Chile), and Southern Australia.

Winter rainfall is much larger than summer rainfall, and total rainfall amount per year ranges from 200 mm to over 900 mm. Some recharge of the soil profile takes place during the moist period, and some leaching might occur.

Water deficit may reach 8 months but always in the hottest period, with long days, and its intensity goes from arid to humid Mediterranean. Agronomic yields are affected by this erratic and irregular pattern of rainfall.

In Mediterranean regions, carbonate redistribution and clay illuviation are two of the main soil-forming processes.^[4] Soils are calcareous in the topsoil, salinity is present in some soils, soil water-holding capacity is highly variable, and soils have a low organic matter content. The pattern of net N mineralization in the major Mediterranean soils is similar, although the amounts of N mineralized may largely differ.^[5]

Since water is the most limiting factor in rainfed Mediterranean regions, its availability determines the characteristics of the agricultural systems.

The region around the Mediterranean may be divided into four main areas: arid areas with annual rainfall <250 mm (barley yields around 1500 kg/ha); semiarid areas with annual rainfall between 250 and 450 mm (average barley yields around 3500–4000 kg/ha); subhumid areas with annual rainfall between 400 and 700 mm, where soft wheat rather than barley is the main crop and, sometimes, legumes such as lentils, oilseed crops such as sunflower, or summer fiber crops such as cotton are included in the rotation; finally humid areas with an annual rainfall >700 mm, where summer crops such as alfalfa are easily introduced.

In California's dryland regions,^[6] annual rainfall ranges from 120 to 450 mm, and a crop–fallow system may be practiced in areas with rainfall <300 mm. Annual cropping with small grains (wheat and barley) is practiced where moisture conditions are more favorable.

In Southern Australia, where a Mediterranean climate prevails, ley farming (self-regenerating legumes such as *Trifolium* and *Medicago* cropped with wheat) practiced after 1945^[7] was very successful, but changes in the relative prices of wool and wheat have introduced a more intensive cropping, which relies on commercial fertilizers.

NITROGEN USE EFFICIENCY

There are many definitions of N efficiency parameters,^[8] but this discussion uses three of them, namely: nitrogen agronomic efficiency (NAE, kg/kg), defined as the ratio of (yield at N_x to yield at N_0) to applied N at N_x , N_0 means no N is applied; apparent nitrogen recovery (NRF, kg/kg), defined as the ratio of (N uptake at N_x to N uptake at N_0) to applied N at N_x ; and nitrogen physiological efficiency (NPE, kg/kg) is the ratio of (yield at N_x to yield at N_0) to (N uptake at N_x to N uptake at N_0). Thus, NAE (yield efficiency) is the product of NRF (N recovered) and NPE (yield and N recovered) and therefore reflects the overall efficiency with which applied N is used. NPE is directly analogous to NAE, where N recovered is substituted for N rate.

This analysis can be used to assess management impacts on the components of N efficiency,^[8] N utilization, and N retention in the cropping system.

The efficiency of N fertilizers applied to cereals across different climates, including the Mediterranean one, has been reviewed.^[9] Factors such as soils, fertilizer rate, source and management, and crop husbandry seem to have contributed as much to this variability as climatic factors. Such variability is also observed (Table 1) in semiarid Mediterranean regions in Spain where NAE ranges from 0 to 16 kg/kg, and NRF ranges from 0.12 to 0.9 kg/kg. Similar data have been reported in Greece.^[10]

Globally,^[1] average NAE for wheat is 18 kg grain/kg fertilizer. Apparent fertilizer recovery through the grain

cereal is estimated globally at 33%^[2] and is 42% and 29% in the developed and developing nations and about 50% for the aboveground biomass.^[1] Studies with ¹⁵N including the soil compartment have indicated higher figures^[11] of 66% for the first year.

Most of the available data are from research experiments. Thus, on-farm NUE may be less than these figures. Some authors guess,^[1] with available information, that actual farmer NRF values for aboveground wheat are between 20% and 30%, and under improved N management, NRF practices may reach 40%.

The low NRF under on-farm conditions indicates numerous opportunities to improve N use efficiency through management practices. Furthermore, NRF declines with increased rate of fertilizer use, which is usually low in many cases in the Mediterranean regions.

Increasing NUE in Mediterranean regions is based on a strategy of maximizing water use efficiency (WUE, aboveground biomass or grain dry weight per millimeter of water use) and increasing water use (transpiration plus evaporation from the soil) by the crop. These in turn call for a wide array of measures as rotations or conservation tillage.

Losses of the soil–crop system, which decrease sharply the N recovery efficiency, have been studied at different scale levels and environments: ammonium (NH_4^+) plant release,^[12] nitrate leaching, denitrification, ammonia (NH_3) volatilization. Losses due to volatilization, denitrification, and nitrate leaching altogether account for 20–50% of fertilizer N^[2] in studies using ¹⁵N. Under Mediterranean conditions, nitrate leaching is highly variable^[13] and denitrification may be low because of relatively good soil aeration. However, NH_3 volatilization may be high,^[14] because most of the soils are calcareous, with high pH in the topsoil.

Excess of available N in cereals grown in Mediterranean regions often leads to a situation of excessive vegetative growth and delayed maturity, which in turn results in low grain yields, a widely recognized process.

Table 1 Maximum grain yield (MGY), NAE, NRF, and NPE in different climatic Mediterranean regions in Spain.

Crop	MGY (10^{-3} kg/ha)	NAE (kg/kg)	NRF (%)	NPE (kg/kg)	References	Remarks
W	4.8	4.0–11.4	12–34	12–33	[15]	
B	5.4	4.3–16.1	33–65	—	[16]	
B	4.9	0–12.6	32–50	—	[16]	
B	2.5	0–10.6	20–42	—	[16]	
W ^a	5.7		50–85	—	[13]	
B ^a	4.3		31–46	—	[13]	
W ^a	6.7		35–84	—	[13]	
W ^a	7.4		47–81	—	[13]	
W ^a	7.1		60–90	—	[13]	
W	7.4		30–80	—	[13]	
B	5.8		76–94	—	[13]	Irrigated
B–B–W–B rotation	4.0	2.6–9.4	—	—	[17]	Organic fertilization

^aTotal crop nitrogen includes N in the root system. It accounts for 10–23% of total N; B: winter barley; W: winter wheat.

STRATEGIES TO IMPROVE N EFFICIENCY

Monitoring N in Soils and Crops

Large quantities of fertilizer N accumulate as nitrate in the soil profile in Mediterranean regions, particularly when high rates of N (mostly manure) are applied.^[18] Monitoring mineral N (N_{min}) in soils has two principal advantages: first, it allows to adjust N at sowing, according to the amount of residual soil mineral N, at tillering, or at the beginning of stem elongation in cereals, and second, it helps to control vegetative growth according to stored water at the end of winter period. Also, it prevents soil salinity to build up when an excessive amount of organic fertilizers is applied.^[19]

Available quick analysis methods in order to quantify N_{min} , at the time of fertilizer application, facilitate controls. This methodology can be complemented with other tools for N plant status diagnostic, such as optical measurements.

Nevertheless, usefulness of tools for N fertilization recommendations has to be locally tested and adapted; experience shows that they work very poorly when strong water deficit exists and fine tuning is needed.

N Fertilization: Type of Fertilizers and Timing

In the arid and semiarid Mediterranean climate of Ebro river valley in Spain, barley grain can increase 9–9.5 kg/ha/mm water conserved,^[5] raising yields by >300 kg/ha. In order to take the advantage of stored water at critical stages of flowering and grain filling, the amount of N applied and timing, in order to control unnecessary vegetative growth, is a key factor. Thus, applying N close to the time of maximum N demand of plants will increase N use efficiency. Usually, the greatest response to N fertilization occurs when the final increment of N is applied just to stem elongation in cereals.^[20]

A substantial proportion of applied N can be lost not only by leaching but also to the atmosphere as NH_3 gas,^[21] and NH_3 does not volatilize from dry soils. Nevertheless, when urea or NH_4^+ salts, or even pig slurry rich in NH_4^+-N ,^[14] are applied on the surface of calcareous soils, and a shower occurs, NH_3 losses can be important. Therefore, urea should be incorporated into the soil. It is also recommended to choose mineral fertilizers with a significant NO_3-N content, rather than other forms as $(NH_4)_2SO_4$. Management of organic fertilizers has to deal with different problems.^[22] In such cases, equipment that allows direct injection into the soil at sowing, at competitive cost or prompt (before 24 hr after application) plug-ins are effective. It is also useful to introduce some strategies in later cereal applications, such as dilution, to facilitate slurry infiltration into the soil to avoid losses.

Nitrification inhibitors may also be used, but do not seem to have any significant effect on NH_3 emissions; further research is needed, particularly under dryland and high pH conditions.

Organic fertilizers [manures, high in organic carbon (C) content] are also important in rainfed agriculture because of their effect in increasing organic matter and, thereby, improving physical soil properties as structure or water-holding capacity^[23] as well as biological ones. In arid and semiarid areas, organic fertilizers with high C-to-N ratios contribute modestly to maintain the organic N levels in the soil. Short-term N recycling occurs and, in some cases, can be significant in plant N uptake, but most of the added organic N remains in the soil organic N fraction.^[24]

University of Lleida researchers (Boixadera, Bosch, Sió, and Teira) have summarized^[25] how to improve agronomic efficiency of organic fertilizers in Mediterranean countries from a wide broad perspective.

Crop Management

Under Mediterranean conditions where NUE is linked to WUE,^[26] the choice of a winter or a summer crop and the cultivars are of the utmost importance. In semiarid and arid areas, barley is advantageous over wheat because of its shorter vegetative period, avoiding water stress and high temperatures during grain filling. Nevertheless, there are also differences, in the responses to drought, between barley cultivars.^[27] Decreasing soil evaporation through a fast initial growth of the crop also seems to be relevant because biomass accumulation takes place under low vapor pressure deficit.^[28] Adequate plant density would help to keep water and control excessive vegetative growth not just in cereals but also in legumes.^[29]

Tillage intensity is important in Mediterranean regions because as its intensity increases, it reduces soil water availability; it increases the oxidation of organic matter and the amount of N mineralized^[16] and erosion. Advantages of shifting from conventional to conservation tillage have been described^[30–32] in terms of soil aggregation, soil organic matter content, C sequestration, N conservation, microbial biomass C, enzymatic activity, and available water capacity. Nevertheless, the influence of tillage systems and the associated changes on soil properties on yields is very diverse. In arid and semiarid areas, conservation tillage has proved to be the best option to increase crop water use and WUE.^[33] In humid Mediterranean regions or in wet years in other subareas, yields and N concentrations in plants are less influenced by tillage systems and usually, with full availability of N fertilizers, conventional tillage gives higher yields.^[34] There is also a positive yield answer to N fertilization rates. In this case, no-tillage systems lose their economic interest in front of conventional tillage (the most productive), but minimum tillage remains economically advantageous.

Rotations

NUE can be evaluated in one cropping season or through all rotation.

Because of water stress, a significant portion can remain in soil in organic or inorganic forms than can be effectively used by the next crop by mineralization or directly, if water availability increases.

Crop rotations can be combined with fallow periods in Mediterranean arid areas. In arid and semiarid areas, it has been demonstrated^[35] that cereal production would be greater with annual cropping than with cereal–fallow rotation. One of the reasons could be that in conventional tillage systems, under fallow, residue cover is low (<10%). When no-till fallow is introduced in the rotation in semiarid environments, where surface is rewetted during the fallow period, the amount of residues plays an important role in improving soil water conservation^[36] as well as soil quality. No-till fallow strategies would theoretically allow to take advantage of mineralized N in low-input agricultural systems. It is well-known that N mineralization increases with the percentage of soil pores filled with water up to 60%.^[37] Nevertheless, a risk of leaching and also denitrification exists, with punctual summer rains. Additionally, since yields are low, straw production is low and recycled organic matter is low. If water is limiting, straw decomposition will be slow, but crop residues will help to control water and wind erosion. Thus, it may be concluded that fallows are not a reliable store for available N.^[38] If conservation tillage can be introduced, long fallowing is no longer recommendable in semiarid areas.^[39]

Legumes, in semiarid environments, are an alternative to fallow if their productivity is low and they produce relatively small biomass,^[40] thus reducing the impact on the next cereal crop. This cereal–legume rotation is an alternative to weed control and reduces the need for fertilizers, but increases complexity, and its economic advantages are little.

Cultivated pasture, except for humid areas, is considered very risky.

In humid areas, catch crops can be effective in reducing nitrate leaching potential by absorbing residual soil mineral N from earlier crops and available water.^[41] As the catch crop is buried, some of the absorbed N is returned to the soil, and it is available to the following crop.

Balanced nutrition is important to achieve optimum NUEs. One key element is phosphorus, playing a double role;^[42] first, it enhances root growth and thus increases the soil volume explored by roots, which, in turn, increases water available to the crop; second, it accelerates crop maturity by 10–14 days, which might allow the crop to escape to hot, dry conditions in the last part of the growing season.

CONCLUSION

In rainfed Mediterranean agricultural systems, rainfall and rainfall distribution are the most limiting factors for crop production. Increasing available water during the crop-growing period, through different agronomic strategies, will allow to obtain a positive yield answer to N fertilization,

increasing, at the same time, WUE. Thus, agricultural management must be inspired in the principle of minimizing each production resource that is needed to allow the maximum utilization of all other resources.^[43]

Agroclimatic analysis, soil characterization, choice of crops and cultivars adapted to water availability period, crop rotations, and the introduction of soil conservation practices, which allow to store water for critical periods and to match N availability to potential demand, are the major challenges.

Technological tools, such as the quantification of water and N_{min} available in soils, testing organic manure for nutrient and dry matter content, and better assessment of management practices as future nutrient application during crop growth for a site-specific area, are the main goals in order to increase the efficiency of N use in the agricultural systems of Mediterranean regions and to reduce environmental impacts in other systems.

Also, a good standard education is needed, for land managers, to face these challenges on complexity management, in a more competitive and open agricultural market.

REFERENCES

1. Ladha, J.K.; Pathak, H.; Krupnik, T.J.; Six, J.; van Kessel, C. Efficiency fertilizer nitrogen in cereal production: Retrospects and prospects. *Adv. Agron.* **2005**, *87*, 85–155.
2. Raun, W.R.; Johnson, G.V. Improving nitrogen use efficiency for cereal production. *Agron. J.* **1999**, *91* (3), 357–363.
3. Elías, F.; Castellví, F.; Bosch Serra, A.D. Clasificaciones climáticas. In *Agrometeorología*; Elías, F., Castellví, F., Eds.; Mundi-Prensa: Madrid, 1996; 279–315.
4. Yaalon, D.H. Soils in the Mediterranean region: What makes them different? *Catena* **1997**, *28*, 157–169.
5. Matar, A.E.; Beck, D.P.; Pala, M.; Garabet, S. Nitrogen mineralization potentials of selected Mediterranean soils. *Commun. Soil Sci. Plant Anal.* **1991**, *22* (1–2), 23–36.
6. Hatfield, J.L. Cropping practices: Pacific Southwest. In *Dryland Agriculture*; Dregne, H.E., Willis, W.O., Eds.; American Society of Agronomy-Crop Science Society of America-Soil Science Society of America: Madison, 1983; Vol. 23, 427–433.
7. McWilliam, J.R. Striving for sustainability in dryland farming: The Australian experience. In *Challenges in Dryland Agriculture: A Global Perspective*, Proceedings of the International Conference on Dryland Farming; Unger, P.W., Jordan, W.R., Sneed, T.V., Eds.; Amarillo/Bushland: Texas, 1988; 45–49.
8. Huggins, D.R.; Pan, W.L. Nitrogen efficiency component analysis: An evaluation of cropping system differences in productivity. *Agron. J.* **1993**, *85*, 898–905.
9. Craswell, E.T.; Godwin, D.C. The efficiency of nitrogen fertilizers applied to cereals grown in different climates. In *Advances in Plant Nutrition*; Tinker, P.B., Läuchi, A., Eds.; Praeger: New York, 1984; 1–55.
10. Simonis, A.D. Studies on nitrogen efficiency in cereals. In *Nitrogen Efficiency in Agricultural Soils*; Jenkinson, D.S., Smith, K.A., Eds.; Elsevier Applied Science: London/New York, 1988; 110–124.

11. IAEA (International Atomic, Energy Agency). *Management of Crop Residues for Sustainable Crop Production*; IAEA TECHDOC-1354. IAEA: Vienna, 2003.
12. Francis, D.D.; Schepers, J.S.; Vigil, M.F. Post-anthesis nitrogen loss from corn. *Agron. J.* **1993**, *85*, 659–663.
13. Quemada, M., Ed. *Balance de nitrógeno en sistemas de cultivo de cereal de invierno y de maíz en varias regiones españolas*; Monografías INIA, Serie agrícola 22: Madrid, 2006.
14. Teira Esmatges, R.M. Emission of NH₃ and N₂O from Irrigated Semi-Arid Calcareous Soils. PhD thesis, University of Gent, Gent, 1998.
15. López-Bellido, R.J.; López-Bellido, L. Efficiency of nitrogen in wheat under Mediterranean conditions: Effect of tillage, crop rotation and N fertilization. *Field Crop. Res.* **2001**, *71*, 31–46.
16. Angás, P.; Lampurdanés, J.; Cantero-Martínez, C. Tillage and N fertilization. Effects on N dynamics and barley yield under semiarid Mediterranean conditions. *Soil Till. Res.* **2006**, *87*, 59–71.
17. Bosch Serra, A.D.; Iglesias Fernández, N.; Amat Bové, M.; Boixadera, L.J. Efficiency of nitrogen in slurry and mineral fertilization under rain-fed Mediterranean agriculture. In *Towards a Better Efficiency in N Use*, Proceedings of the 15th N Workshop, Lleida, Spain, 2007; Bosch Serra, A.D., Teira Esmatges, M.R., Villar Mir, J.M., Eds.; 2007; 164–166.
18. Ortiz, C.; Bosch, A.D.; Boixadera, J. Nitrogen pig slurry effects in a dry land cereal system. In *N Management in Agrosystems in Relation to the Water Framework Directive*, Proceedings of the 14th N Workshop; Schröder, J.J., Neeteson, J.J., Eds.; Plant Research International B.V.: Wageningen, 2006; 157–159.
19. Arbonés, A.; Sió, J.; Ortiz, C. Fertilización con purines de ganado porcino en almendro de secano. *Butll. Inf. Agrícoles* **2005**, *21* (7), 22–25.
20. Mossedaq, F.; Smith, D.H. Timing nitrogen application to enhance spring wheat yields in Mediterranean climate. *Agron. J.* **1994**, *86*, 221–226.
21. Hagin, J.; Tucker, B. *Fertilization of Dryland and Irrigated Soils*; Springer-Verlag: Berlin/Heidelberg/New York, 1982.
22. Boixadera, J.; Bosch, A.D. Gestión de la aplicación al suelo de residuos orgánicos. In *I Encuentro Internacional: Gestión de Residuos Orgánicos en el Ámbito Rural Mediterráneo*; Cátedra Zurich de Medio Ambiente de la Universidad de Navarra: Pamplona, 2001; 59–74.
23. Hudson, B.D. Soil organic matter and available water capacity. *J. Soil Water Conserv.* **1994**, *49* (2), 189–194.
24. Seligman, N.G.; Feigenbaum, S.; Feinerman, D.; Benjamin, R.W. Uptake of nitrogen from high C-to-N ratio, 15N-labeled organic residues by spring wheat grown under semi-arid conditions. *Soil Biol. Biochem.* **1986**, *3*, 303–307.
25. PAM/PNUE-CAR/PP. *Lignes directrices pour l'application des meilleures pratiques environnementales (MPE) en vue de l'utilisation rationnelle des engrais et de la réduction des pertes d'éléments nutritifs par l'agriculture dans la région méditerranéenne*; PAM/PNUE: Athens, 2004.
26. Oweis, T.; Zhang, H.; Pala, M. Water use efficiency of rainfed and irrigated bread wheat in a Mediterranean environment. *Agron. J.* **2000**, *92*, 231–238.
27. Cantero-Martínez, C.; Villar, J.M.; Romagosa, I.; Fereres, E. Growth and yield responses of two contrasting barley cultivars in a Mediterranean environment. *Eur. J. Agron.* **1995**, *4*, 317–326.
28. Tambussi, E.A.; Bort, J.; Araus, J.L. Water use efficiency in C₃ cereals under Mediterranean conditions: A review of physiological aspects. *Ann. Appl. Biol.* **2007**, *150*, 307–321.
29. Sau, F.; Mínguez, M.I. Adaptation of indeterminate Faba beans to weather and management under a Mediterranean climate. *Field Crop. Res.* **2000**, *66*, 81–99.
30. Mrabet, R.; Saber, N.; El-Brahli, A. Total, particulate organic matter and structural stability of a Calcixeroll soil under different wheat rotations and tillage systems in a semi-arid area of Morocco. *Soil Till. Res.* **2001**, *57*, 225–235.
31. Bescansa, P.; Imaz, M.J.; Virto, I. Soil water retention as affected by tillage and residue management in semiarid Spain. *Soil Till. Res.* **2006**, *87*, 19–27.
32. Madejón, E.; Moreno, F.; Murillo, J.M. Soil biochemical response to long-term conservation tillage under semi-arid Mediterranean conditions. *Soil Till. Res.* **2007**, *94*, 346–352.
33. Cantero-Martínez, C.; Angás, P.; Lampurdanés, J. Long-term yield and water use efficiency under various tillage systems in Mediterranean rainfed conditions. *Ann. Appl. Biol.* **2007**, *150*, 293–305.
34. Mazzoncini, M.; di Bene, C.; Coli, A.; Antichi, D.; Petri, M.; Bonari, E. Rainfed wheat and soybean productivity in a long-term tillage experiment in central Italy. *Agron. J.* **2008**, *100* (5), 1418–1429.
35. Austin, R.B.; Cantero-Martínez, C.; Arrúe, J.L.; Playán, E.; Cano-Marcellán, P. Yield-rainfall relationships in cereal cropping systems in the Ebro river valley of Spain. *Eur. J. Agron.* **1998**, *8*, 239–248.
36. Lampurdanés, J.; Cantero-Martínez, C. Hydraulic conductivity, residue cover and soil surface roughness under different tillage systems in semiarid conditions. *Soil Till. Res.* **2006**, *85*, 13–26.
37. Linn, D.M.; Doran, J.W. Aerobic and anaerobic microbial population in no-till and plowed soils. *Soil Sci. Soc. Am. J.* **1984**, *48*, 794–799.
38. Seligman, N.G.; Feigenbaum, S.; Benjamin, R.W. Efficiency of fallow as a store for fertilizer nitrogen in a semi-arid region. *J. Agric. Sci. Camb.* **1985**, *105*, 245–249.
39. Moret, D.; Arrúe, J.L.; López, M.V. Winter barley performance under different cropping and tillage systems in semiarid Aragon (NE Spain). *Eur. J. Agron.* **2007**, *26*, 54–63.
40. Díaz-Ambrona, C.H.; Minués, M.I. Cereal-legume rotations in a Mediterranean environment: Biomass and yield production. *Field Crop. Res.* **2001**, *70*, 139–151.
41. Domingo Olivé, F.; Roselló Martínez, A.; Serra Gironella, J.; Teixidor Albert, A. Els cultius captadors de nitrogen: Una bona pràctica agrícola per minimitzar el rentatge de nitrat del sòl en cultius extensius. Departament d'Agricultura, Ramaderia i Pesca, Generalitat de Catalunya. Bona Pràctica Agrària (I). Dossier Tècnic **2005**, *6*, 17–23.
42. Khattari, S.; Tell, A. Response of wheat varieties to P fertilizers under a wide range of rainfall. In *Challenges in Dryland Agriculture: A Global Perspective*, Proceedings of the International Conference on Dryland Farming, Amarillo/Bushland, Texas, 1988; Unger, P.W., Jordan, W.R., Sneed, T.V., Eds.; 1988; 429–431.
43. de Wit, C.T. Resource use efficiency in agriculture. *Agric. Sys.* **1992**, *40*, 125–151.

Metal—Clay Interactions

Dean Hesterberg

Department of Soil Science, North Carolina State University, Raleigh,
North Carolina, U.S.A.

Abstract

Soils are vital for regulating the biological effects and mobility of metals in nature. Some metals such as potassium and calcium are essential nutrients for plants and animals, while others such as lead and mercury are potentially toxic. Therefore, a detailed understanding of the chemistry of metals in soils is essential for managing their agricultural and ecological impacts. Two soil components that are particularly important for regulating metal impacts are clays and organic matter. Clays consist of inorganic minerals in the fine particle-size fraction of soils (<0.002 mm particles) and may include associated organic matter. Modern concepts of metal interactions within soils evolved from the discovery in the mid-1800s that soils retain (bind) cations. Since then, researchers have applied a wide variety of analytical techniques and chemical principles to understand metal interactions within soils and with soil components.

INTRODUCTION

Most chemical elements are metals, and many of these occur naturally in soils. Metals usually occur as cations in soils. However, metal cations of higher charge such as chromium (Cr^{6+}) and molybdenum (Mo^{6+}) combine with oxygen to form stable molecules called oxyanions (CrO_4^{2-} and MoO_4^{2-}). The most abundant metals in mineral soils, aluminum (Al^{3+}) and iron (Fe^{3+} or Fe^{2+}), are mostly combined with oxygen anions (O^{2-}) into soil minerals such as aluminosilicates, Al hydroxides, and Fe oxides. Potassium (K^+), sodium (Na^+), calcium (Ca^{2+}), and magnesium (Mg^{2+}) are abundant in all but highly acidic soils. Metals having a density greater than 5 g/cm^3 are called heavy metals. Except for manganese (Mn) and Fe, concentrations of heavy metals in non-contaminated soils are typically <100 mg/kg. In soils that are impacted by nuclear wastes, metals such as uranium, plutonium, cesium, and strontium are of concern.

Clays are especially important for binding metal ions because they have a high amount of surface area and the surfaces can be charged. Because oxygen constitutes, on an average, 60% of the atoms in soils, the surfaces of clay particles are typically dominated by oxygen atoms.^[1] Phyllosilicate (layer silicate) clay surfaces have a permanent negative charge because of minor substitutions of equivalently sized but lower-charged metal cations for more abundant cations within the crystal structure. The surfaces on the edges of aluminosilicate clay particles and surfaces of Fe-, Al-, and Mn-oxide minerals have a variable charge. This so-called pH-dependent surface charge from a loss or gain of hydrogen ions (H^+) on the surface oxygens as the concentration of H^+ (i.e., pH) in the surrounding water changes.

MECHANISMS OF METAL RETENTION BY CLAYS

Metal ions are retained by clays through: 1) adsorption; 2) surface cluster formation; 3) surface precipitation; 4) diffusion into an existing mineral structure; or 5) coprecipitation. The first three mechanisms occur at clay surfaces, the interface between clay particles and the surrounding water (aqueous solution). Diffusion results from a metal concentration gradient from the clay surface into the particle and occurs very slowly.^[2] Coprecipitation involves incorporation of metal ions into a clay structure as it forms.

Adsorption—Macroscopic Aspects

Adsorption is the net accumulation of a substance into a 2-D structure at the clay surface.^[3] Adsorption processes occur over time periods of seconds to days.^[4] The amount of a given type of metal ion adsorbed depends mainly on the total concentration of metal ions present, their affinity for the clay surfaces relative to the aqueous (water) solution, and factors such as pH and the presence of other (competing) ions. The concentration and affinity effects at constant pH and temperature are indicated by an adsorption isotherm, a plot of adsorbed vs. dissolved metal. Isotherms are classified according to their shape (Fig. 1). An H-curve reflects a high affinity of metal ions for the soil or clay, as is common for heavy metals at low concentrations. The H-curve can be fit mathematically with a Freundlich adsorption isotherm model.^[3] An L-curve suggests that the mineral particles have a high affinity for the metal ions at low concentration and a decreasing affinity, as the adsorbed metal concentration approaches the maximum adsorption capacity of the soil or clays.^[7]

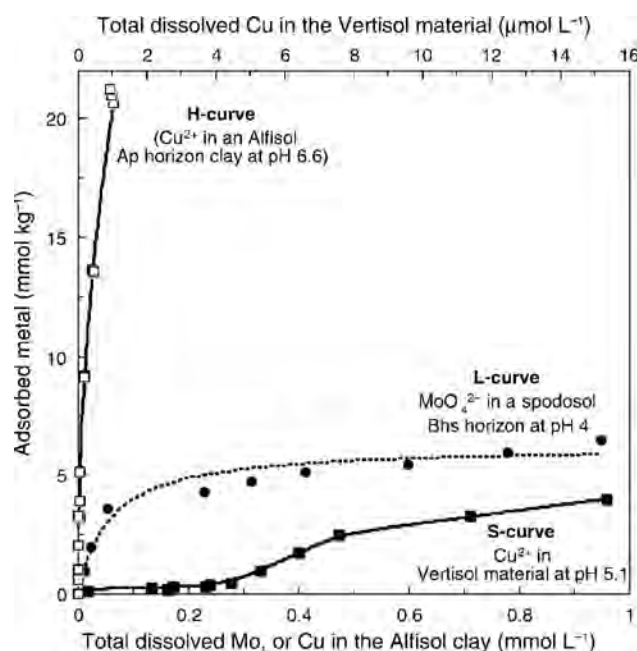


Fig. 1 Classes of adsorption isotherms for metal ions in soils or clays. H-curve data are for <0.001 mm soil clay,^[5] adsorbed Cu values are scaled by a factor of 0.1, and data are fit with a Freundlich isotherm model. L-curve data^[6] are fit with a Langmuir isotherm model. For the S-curve data,^[3] adsorbed Cu values are scaled by a factor of 0.1 and fit with a smooth curve.

However, an isotherm does not reveal the molecular aspects of metal–clay associations, and metal retention mechanisms other than adsorption can also produce such a curve.^[8] The L-curve is fit mathematically with a Langmuir or Freundlich isotherm model.^[3] One explanation of an S-curve is that the presence of a metal-binding agent in the aqueous solution such as dissolved organic matter competes with the surfaces for the metal ions. As the total metal concentration increases, the capacity of the dissolved agent to bind metals is exceeded and adsorption increases.^[3]

Typical pH dependence of metal-ion adsorption on oxide minerals is illustrated in Fig. 2. With increasing pH, metal-cation adsorption increases and metal oxyanion adsorption decreases. These trends are related to the pH-dependent surface charge (Fig. 2). However, shifts in the adsorption curves along the x-axis for different metal cations added at the same level suggest that metal–clay interactions other than electrostatic attractions between the surface and oppositely charged ions are important. Heavy metal binding on such pH-dependent surfaces is strong under favorable pH conditions. Therefore, adsorption is not affected much by the presence of weakly bound cations such as Na⁺ and K⁺. Adsorption of heavy metal cations on permanent charge sites of phyllosilicate minerals is usually weaker than that on pH-dependent surfaces,^[2] so other cations more effectively compete for adsorption sites through a process called cation exchange.

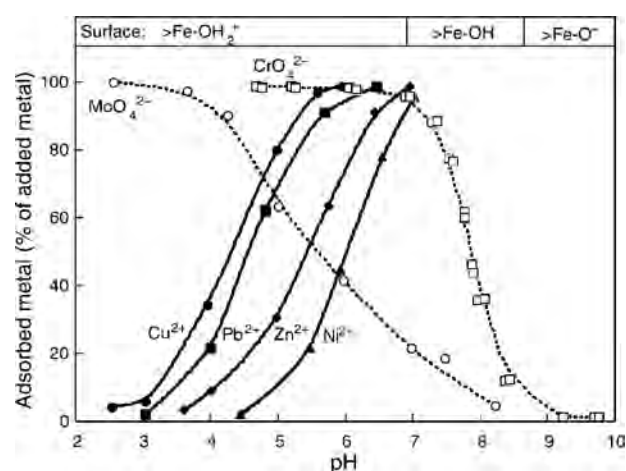


Fig. 2 Effects of pH on the adsorption of metal cations (Cu²⁺, Pb²⁺, Zn²⁺, and Ni²⁺) and oxyanions (MoO₄²⁻ and CrO₄²⁻) on the Fe-oxide mineral goethite (α-FeOOH). The generalized number of protons and charge on reactive surface oxygens (e.g., >Fe–OH₂⁺) at different pH ranges assumes that the goethite particles have no charge at pH 7.8.^[7] Cation data^[9] (added metal: 20 mmol/kg goethite), molybdate data^[10] (added metal: 300 mmol/kg goethite), and chromate data^[11] (added metal: 0.6 mmol/kg goethite).

Adsorption—Molecular Mechanisms

On a molecular scale, three mechanisms of metal-ion adsorption are distinguished: inner-sphere surface complexes, outer-sphere surface complexes, and diffuse layer ions (Fig. 3). Inner-sphere complexation, also called chemisorption, involves metal bonding directly to atoms at the clay surface as one or more water molecules that

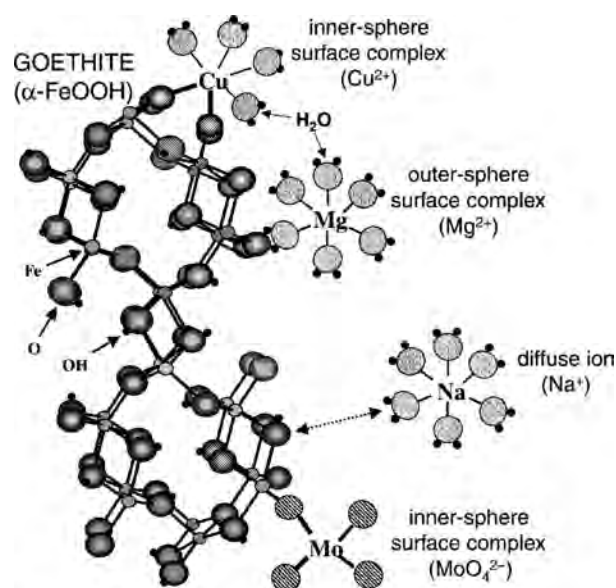


Fig. 3 Schematic illustration of molecular-scale mechanisms of metal cation (Cu²⁺, Mg²⁺, and Na⁺) oxyanion (MoO₄²⁻) adsorption on a goethite surface.

Source: Adapted from Alcacio, Hesterberg, et al.^[12] ©2001.

surround the dissolved ion are lost. A metal cation would bond directly to a surface oxygen that has lost its associated hydrogen ion (H^+). An oxyanion would bond through its own oxygen to the metal ion in the material that lies one atomic layer behind the surface oxygen that replaces (Fig. 3).^[2,3,7] As such, metal cations and oxyanions may compete for the same surface sites. Chemisorption involves a combination of covalent (electron sharing) and ionic (electrostatic) bonding. Outer-sphere surface complexes and diffuse ions are primarily bound to the surface through electrostatic attractions between the ions and oppositely charged surface sites. Outer-sphere surface complexes involve weaker binding of the metal ion directly on the clay surface but with water molecules positioned between the ions and the surface. Diffuse layer ions are not located directly on the surface but are moving about within a region called the diffuse double layer that extends nanometer-scale distances from the surface. Conceptually, the diffuse double layer exists because of the opposing forces of electrostatic attraction of ions to the charged surface and diffusion of the accumulated ions away from the surface.

Metal-cation adsorption on clays is modified by adsorbed organic matter. Adsorption involving interactions between clay surfaces, metal ions, and organic matter may consist of either the metal-ion bridging between the clay and the organic matter (Type A ternary complex) or the metal-ion adsorbing to the organic matter that is adsorbed on the clay surface (Type B ternary complex).^[12]

Surface Clusters and Surface Precipitates

With increasing concentration of adsorbed metal, metal ions may cluster together on the clay surface and form a 3-D structure comprising metal cations and hydroxyl anions (OH^-).^[8,13] As metal concentration on the surface increases, these surface clusters may grow into a surface precipitate that entirely covers the surface.

Coprecipitation

Metal ions can be incorporated into the bulk clay structure by a process called coprecipitation, which is the simultaneous precipitation (formation) of one compound in conjunction with another.^[14] When a mineral is formed, coprecipitation may involve the replacement of ions that are common to the mineral by metal ions of similar size and charge as the mineral forms. The substituted ions are dispersed as a minor component throughout the mineral. Another type of coprecipitation involves a metal accumulating as its own pure mineral phase encapsulated within a different pure mineral phase. A third type of coprecipitation involves an adsorption mechanism, whereby metal ions bind at surfaces of newly formed mineral particles, then become incorporated within the particles as they continue to grow.

CONCLUSION

Interactions between metals and clays in soils help regulate the plant availability of metals that are essential for agronomic crop growth, while affecting the mobility and environmental impacts of potentially toxic heavy metals. Because soils comprise a complex mixture of clays, organic matter, coarse-grained particles, water, air, and biological entities with properties that change both in space and time, accurately predicting chemical reactions and long-term fate of metals in a given soil is challenging.

REFERENCES

- Schulze, D.G. An introduction to soil mineralogy. In *Minerals in Soil Environments*, 2nd Ed.; Dixon, J.B., Weed, S.B., Eds.; SSSA Book Series 1; Soil Science Society of America: Madison, 1989; 1–34.
- McBride, M.B. Chemisorption and precipitation of inorganic ions. In *Environmental Chemistry of Soils*; Oxford University Press: New York, 1994; 121–168.
- Sposito, G. Inorganic and organic solute adsorption in soils. In *The Surface Chemistry of Soils*; Oxford University Press: New York, 1984; 113–153.
- Amacher, M.C. Methods of obtaining and analyzing kinetic data. In *Rates of Soil Chemical Processes*; Sparks, D.L., Suarez, D.L., Eds.; SSSA Special Publication Number 27; Soil Science Society of America: Madison, 1991; 19–59.
- Weitzel, S.C. Flocculation of Soil Clay as Affected by Adsorbed Metals, Organic Matter, PH. Master of Science Thesis; North Carolina State University, 1999.
- Bibak, A.; Borggaard, O.K. Molybdenum adsorption by aluminum and iron oxides and humic acid. *Soil Sci.* **1994**, *158* (5), 323–328.
- Sparks, D.L. Sorption phenomena on soils. In *Environmental Soil Chemistry*; Academic Press: San Diego, 1995; 99–139.
- Manceau, A.; Charlet, L.; Boisset, M.C.; Didier, B.; Spadini, L. Sorption and speciation of heavy metals on hydrous Fe and Mn oxides. From microscopic to macroscopic. *Appl. Clay Sci.* **1992**, *7*, 201–223.
- McKenzie, R.M. The adsorption of lead and other heavy metals on oxides of manganese and iron. *Aust. J. Soil Res.* **1980**, *18*, 61–73.
- Chu, Z.P.; Sparks, D.L. Kinetics and mechanisms of molybdate adsorption/desorption at the goethite/water interface using pressure-jump relaxation. *Soil Sci. Soc. Am. J.* **1989**, *53* (4), 1028–1034.
- Ainsworth, C.C.; Girvin, D.C.; Zachara, J.M.; Smith, S.C. Chromate adsorption on goethite: Effects of aluminum substitution. *Soil Sci. Soc. Am. J.* **1989**, *53* (2), 411–418.
- Alcacio, T.E.; Hesterberg, D.; Chou, J.W.; Martin, J.D.; Beauchemin, S.; Sayers, D.E. Molecular scale characteristics of Cu(II) bonding in goethite–humate complexes. *Geochim. Cosmochim. Acta.* **2001**, *65*, 1355–1366.
- McBride, M.B. Reactions controlling heavy metal solubility in soils. *Adv. Soil Sci.* **1989**, *10*, 1–56.
- Sposito, G. Mineral solubility. In *Chemical Equilibria and Kinetics in Soils*; Oxford University Press: Oxford, 1994; 93–137.

Microbial Biomass: Measurement

K.R. Islam

S.R. Wright

Ohio State University South Centers, Piketon, Ohio, U.S.A.

Abstract

Common and new methods were discussed, including their advantages and disadvantages. Over the years, the indirect methods have been more widely used compared to direct methods for simple, rapid, and precise measurement of soil microbial biomass (SMB). Among the indirect methods, chloroform fumigation incubation has been used as a baseline for calibrations and correlations for other methods. Extraction methods, in particular the microwave soil extraction, are gaining increasing acceptance for their simplicity, rapidity, and precision in determining SMB. No method is universally accepted because of various limitations. Consequently, there is a demand for further improvement in SMB measurement

INTRODUCTION

Soil microbial biomass (SMB) is an active component of the terrestrial ecosystem that regulates many critical functions and properties related to soil and environmental qualities. The functions and processes include source-sink in nutrient cycling, decomposition of organic residues, structural stability, and indicator of soil pollution and bioremediation.^[1–3] No standard method for measuring SMB is available, but several widely differing approaches have been developed. Methods used for measuring SMB are briefly discussed.

DIRECT METHODS TO MEASURE SMB

In direct methods, microorganisms are determined by colony forming units counted on soil dilution series using most probable number (MPN) and/or by direct microscopic counting methods. In MPN method, soil samples are dispersed in a series of dilution to estimate population density based on the presence or absence of microbial cells.^[4] Thus, if microbial growth is observed in the 10^{-4} but not in the 10^{-5} dilution, the number of cells is estimated in between 10^4 and 10^5 .

The direct microscopic method involves dispersing a known amount of fresh soil in a known volume of water, or dilute agar media is smeared over a known area on a glass slide to count for microbial cells using fluorochromes. The measurement of SMB from cell dry weight and volume characteristics^[5] is as follows:

$$\text{Bacterial biomass}(\mu\text{g/g oven-dry soil}) = \text{NVB}^b 10^6 \quad (1)$$

where nitrogen (N) is the number of bacteria/g of oven-dry soil, V is the average volume of bacterial cell (μm^3), and B^b is the biomass/volume conversion factor ($0.22\text{--}0.33 \times 10^{-12} \text{ g}/\mu\text{m}^3$).

$$\text{Fungal biomass}(\mu\text{g/g oven-dry soil}) = \text{L}\Pi r^2 B^f 10^6 \quad (2)$$

where L is the mycelia length (in $\mu\text{m/g}$ oven-dry soil), r is the average radius of mycelia (μm), B^f is the biomass/volume conversion factor ($0.2\text{--}0.33 \times 10^{-12} \text{ g}/\mu\text{m}^3$).

Soil MPN method based on dilution is widely applicable to bacterial cell counting, but for fungi, it is only valuable for spore or other propagule counting. The direct microscopic method is a tedious procedure, and often yields result 10–100 times greater than the soil dilution techniques.

INDIRECT METHODS TO MEASURE SMB

In indirect methods, the SMB is assessed by using biochemical, chemical, and physical principles for determination of a particular cell constituent such as carbon (C), N, phosphorous (P), sulfur (S), adenosine triphosphate (ATP), and phospholipids of microbes.^[6–14]

Chloroform (CHCl_3) Fumigation Incubation (CFI) and Extraction Methods

CHCl_3 fumigation has been a reference method to determine SMB since it was first developed.^[9] The rationale for the CFI method is that the fumigant lysed the soil microbes and the resulting increase in carbon dioxide (CO_2) evolution from fumigated soil, compared to unfumigated soil,

over a 10-day incubation period at 25°C, is directly proportional to the amount of C in the SMB.^[9] The amount of released CO₂ is determined by absorption in 0.5 M sodium hydroxide (NaOH), followed by acid–base titration to calculate the SMB.

$$\text{SMB} = F_c/K_c \quad (3)$$

where F_c is the net flush of CO₂ from fumigated and unfumigated soils, respectively, during incubation, and K_c is a coefficient of 0.45.^[9]

In the CHCl₃ fumigation extraction (CFE) method, the postfumigated soil is extracted with suitable extractants for flush of C, N, P, and S compared to unfumigated soils.^[6] Soil extracts were analyzed for C, ninhydrin-reactive N, P, and S to calculate SMB.^[6,13]

$$\text{SMB} = 2.68V - 44.1 \quad (4)$$

where V is the net flush of C from fumigated and unfumigated soils, respectively extracted by neutral 0.5 M potassium sulfate (K₂SO₄).

$$\text{SMB} = 20.0 \times \text{NRN if soil pH is} > 5.0 \quad (5)$$

$$\text{SMB} = 35.3 \times \text{NRN if soil pH is} \leq 5.0 \quad (6)$$

Both CFI and CEF methods yield good estimates of SMB, but CHCl₃ is a biohazard. Also, the CFI method is affected by high organic matter content, organic amendments, low pH, and soil waterlogged conditions and is time-consuming and involves several steps.^[1,9] The CFE method is fast and useful where CFI does not work.^[6] A portion of the SMB may be insensitive to CHCl₃ fumigation.

Microwave (MW) Irradiation Incubation and Extraction Methods

The underlying principle of the MW irradiation method is to use non-thermal MW energy at 800 J/g oven-dried equivalent of field-moist soil in plastic tubes with punctured caps to disrupt the microbial cells and then incubate or extract the MW and unmicrowaved soils to measure flushes of C.^[8] In MW irradiation incubation method, both MW and unmicrowaved soils are incubated for 10 days in the dark at 25°C, and the flush of CO₂ is absorbed in dilute solution of NaOH followed by an acid–base titration to calculate the SMB.

$$\text{SMB} = \text{CO}_2 - C_{\text{MW}}/K_{\text{MI}} \quad (7)$$

where $\text{CO}_2 - C_{\text{MW}}$ is the net flush of CO₂ from MW and unmicrowaved soils, respectively, and K_{MI} is a coefficient of 0.341.

In MW irradiation extraction method, the MW soil is extracted for flush of C by neutral 0.5 M K₂SO₄ compared to unmicrowaved soil.^[8] An automatic analyzer

with ultraviolet (UV) persulfate oxidation and infrared (IR) detection, or rapid colorimetric^[8,15] or titrimetric method, is used to determine extracted C for the SMB measurement.

$$\text{SMB} = C_{\text{EXTMW}}/K_{\text{ME}} \quad (8)$$

where C_{EXTMW} is the net flush of C from MW and unmicrowaved soils, respectively, and K_{ME} is the extraction coefficient (0.213) of the SMB.^[8]

The MW irradiation method is an alternate approach to CFI and CFE methods for rapid, precise, safe, and reliable measurement of SMB.^[8] This method is very economical. To avoid the release of nonbiomass C from soil, the MW oven has to be calibrated before use.

Rehydration Method

Rewetting of air-dry soils with dilute salt solution or water is the basis for the rehydration method for determining SMB.^[16,17] Upon rehydration, the desiccated microbial cells in air-dried soils are disrupted and they release intracellular C compounds. The extracted C is analyzed by a rapid potassium dichromate oxidation method to determine the SMB content.

$$\text{SMB} = E_c/0.23 \quad (9)$$

where E_c is the net flush of 0.25 M K₂SO₄ extracted C from air-dried and field-moist soils, respectively, and 0.23 is the extraction coefficient.^[17]

The rehydration method is a simple procedure for SMB determination that does not use hazardous chemicals, although prolonged air-drying of soil often releases non-biomass C. A portion of the SMB may be insensitive to air-drying.

Freeze-Dried Soil Extraction Method

The principle of the method is that extraction of freeze-dried soils with either 0.5 M K₂SO₄ or 0.5 M sodium bicarbonate (NaHCO₃) releases cytoplasmic C compounds from desiccated and disrupted microbial cells.^[7] The extracted C is analyzed by a rapid colorimetric method^[15] to calculate the SMB contents.

$$\text{SMB} = C - K_2\text{SO}_4/0.152 \quad (10)$$

$$\text{SMB} = C - \text{NaHCO}_3/0.257 \quad (11)$$

where $C - K_2\text{SO}_4$ and $C - \text{NaHCO}_3$ are the net differences in C extracted by 0.5 M K₂SO₄ and NaHCO₃ from freeze-dried and field-moist soils, respectively.

This is a precise, reliable, and safe method for measuring SMB but it requires trained personnel and sophisticated equipment.

Adenosine Triphosphate Extraction Method

The principle of this method is that, by using suitable reagents, soil microbial cells are rapidly disrupted. This is followed by stabilization of ATP with deactivating synthesis and degradative enzyme processes, and extraction of ATP from the soil matrix. Then, using luciferin–luciferase light reactive system, SMB contents can be calculated.^[18,19]

$$1 \mu\text{g ATP} = 250 \mu\text{g of SMB} \quad (12)$$

As ATP is rapidly degraded during extraction and adsorbed by soil constituents, SMB measurement is often uncertain because of storage conditions, season of collection, low and irregular recovery, and weak correlation to SMB measured.^[1]

Phospholipid Fatty Acids Extraction Method

This method is based on extraction of phospholipid fatty acids (PFLA) from soil microbial cell membranes via suitable extractants.^[11,14] The lyophilized soil is extracted with the single-phase CHCl_3 – CH_3OH –buffer system, followed by analysis of PFLA by a capillary gas chromatography (GC) with flame ionization detection to SMB.^[14]

$$1 \text{ nmol PFLA/g soil} = 2.4 \mu\text{g SMB} \quad (13)$$

Extracts are easily analyzed to identify the different PLFAs for the determination of SMB,^[5] but this is a time-consuming, complex, and expensive method.

Substrate-Induced Respiration (SIR) Method

The SIR method, which was first reported by Anderson and Domsch,^[10] utilizes the initial change in the soil respiration response as a result of adding glucose or sucrose and nutrients. The SMB is calculated from the maximum initial respiratory response (MIRR) measured at 22°C, as follows:

$$\text{SMB} = (40.04 \text{ MIRR}) + 0.37 \quad (14)$$

where MIRR is $\mu\text{L CO}_2/\text{g soil}$.

This is a fast method to measure SMB but may often overestimate by measuring the glucose responsive active portion of the SMB. The use of glucose may shut down the metabolism of other microbes. The SIR method requires a GC to measure evolved CO_2 .

UV SPECTROSCOPIC METHOD

This technique is a rapid and inexpensive method to estimate SMB based on the near-UV light absorption of certain molecules (nucleic acids/nucleotides) of microbial cells

extracted from CHCl_3 fumigated soils.^[3] UV absorbance at 280 nm is significantly correlated to SMB measured by CHCl_3 method as follows:

$$\text{SMB} = 21,747 \times E_{280 \text{ nm}} \quad (15)$$

Although a fast and simple method to determine SMB, this process uses CHCl_3 . The results are often compromised by soil colloidal interferences and electrolyte precipitation.^[3]

CONCLUSION

Common and new methods were discussed, including their advantages and disadvantages. Over the years, the indirect methods have been more widely used compared to direct methods for simple, rapid, and precise measurement of SMB. Among the indirect methods, CFI has been used as a baseline for calibrations and correlations for other methods. Extraction methods, in particular the MW soil extraction, are gaining increasing acceptance for their simplicity, rapidity, and precision in determining SMB. No method is universally accepted because of various limitations. Consequently, there is a demand for further improvement in SMB measurement.

REFERENCES

1. Martens, R. Current methods for measuring microbial biomass-C in soil: Potentials and limitations. *Biol. Fert. Soils* **1995**, *19*, 77–81.
2. Beck, T.; Joergensen, R.G.; Kandler, E.; Makeschin, F.; Nuss, E.; Oberholzer, H.R.; Scheu, S. An inter-laboratory comparison of ten different ways of measuring soil microbial biomass C. *Soil Biol. Biochem.* **1997**, *29*, 1023–1032.
3. Nunan, N.; Morgan, M.A.; Herlihy, M. Ultraviolet absorbance (280 nm) of compounds released from soil during chloroform fumigation as an estimate of the microbial biomass. *Soil Biol. Biochem.* **2002**, *30*, 1599–1603.
4. Alexander, M. Most probable number method for microbial populations. In *Methods of Soil Analysis, Part II. Chemical and Microbiological Properties*; Page, A.L., Miller, R.H., Keeney, D.R., Eds.; 2nd Ed.; American Society of Agronomy: Madison, 1982; Vol. 9, 815–820.
5. Bottomley, P.J. Light microscopic methods for studying soil microorganisms. In *Methods of Soil Analysis, Part II. Microbiological and Biochemical Properties*; Weaver, R.W., Angle, J.S., Bottomley, P.S., Eds.; Soil Science Society of America, Inc.: Madison, 1994; 81–106.
6. Vance, E.D.; Brooks, P.C.; Jenkinson, D.S. An extraction method for measuring soil microbial biomass-C. *Soil Biol. Biochem.* **1987**, *19*, 703–707.
7. Islam, K.R.; Weil, R.R.; Mulchi, C.L.; Glenn, S.D. Freeze-dried soil extraction method for the measurement of microbial biomass C. *Biol. Fert. Soils* **1997**, *24*, 205–210.

8. Islam, K.R.; Weil, R.R. Microwave irradiation of soil for the routine measurement of microbial biomass C. *Biol. Fert. Soils* **1998**, *27*, 408–416.
9. Jenkinson, D.S.; Powlson, D.S. The effects of biocidal treatments on metabolism in soil. V. A method for measuring soil biomass. *Soil Biol. Biochem.* **1976**, *8*, 209–213.
10. Anderson, J.P.E.; Domsch, K.H. A physiology method for quantitative measurement of microbial biomass in soils. *Soil Biol. Biochem.* **1978**, *10*, 519–525.
11. White, D.; Davis, W.M.; Nickels, J.S.; King, J.D. Determination of the sedimentary microbial biomass by extractable lipid phosphate. *Oecologia* **1979**, *40*, 51–62.
12. Webster, J.J.; Hampton, G.J.; Leach, F.R. ATP in soil: A new extractant and extraction procedure. *Soil Biol. Biochem.* **1984**, *16*, 335–342.
13. Joergensen, R.G. Quantification of the microbial biomass by determining ninhydrin-reactive N. *Soil Biol. Biochem.* **1996**, *28*, 301–306.
14. Baily, V.L.; Peacock, A.D.; Smith, J.L.; Bolton, H., Jr. Relationship between soil microbial biomass determined by chloroform fumigation–extraction, substrate-induced respiration, and phospholipid fatty acid analysis. *Soil Biol. Biochem.* **2002**, *34*, 1385–1389.
15. Islam, K.R.; Weil, R.R. A rapid microwave digestion procedure for spectrophotometric measurement of soil organic C. *Commun. Soil Sci. Plant Anal.* **1998**, *29*, 2269–2284.
16. Blagodatskaya, S.A.; Blagodatskaya, Y.V.; Yu, A.; Panikov, N.S. A rehydration method of determining the biomass of microorganisms in soil. *Pochvovedeniye* **1987**, *4*, 64–71.
17. Sikora, L.J.; Yakovchenko, V.; Kaufman, D.D. Comparison of the rehydration method for biomass determination to fumigation–incubation and substrate-induced respiration method. *Soil Biol. Biochem.* **1994**, *26*, 1443–1445.
18. Oades, J.M.; Jenkinson, D.S. Adenosine triphosphate content of the soil microbial biomass. *Soil Biol. Biochem.* **1979**, *11*, 201–204.
19. Parkinson, D.; Paul, E.A. Microbial biomass. In *Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties*; Page, A.L., Ed.; 2nd Ed.; American Society of Agronomy: Madison, 1982; Vol. 9, 821–830.

Microbial Communities

K.R. Islam

S.R. Wright

Ohio State University South Centers, Piketon, Ohio, U.S.A.

Abstract

The soil is a complex and dynamic component of the terrestrial ecosystem with a diverse community of microbes dominated by bacteria, fungi, actinomycetes, and algae. The microbial community is vital to the cycle of life on earth. They incorporate plant and residues into the soil and digest them, returning carbon dioxide to the atmosphere, where it can be recycled through higher plants. Simultaneously, they release nutrients during decomposition of organic residues and form soil organic matter, which is vital for soil quality. With the diversity in a soil ecosystem, many types of community interactions are possible, including parasitism, competition, and symbiosis.

INTRODUCTION

The soil, a complex and heterogeneous component of terrestrial ecosystems, sustains an immense number and diversity of microorganisms, which require much more study. Actually, less than 1% of the soil microorganisms have been cultured, identified, and characterized in the soil ecosystem. Soil microorganisms, prokaryotic and eukaryotic, spend all or part of their lives in the soil environment.^[1,2] These microorganisms live together by interacting with each other and with other organisms in the soil.

The soil microbial community presents a complex and variable association among the different levels of biological organization, which encompasses genetic variability and the richness and relative evenness in communities. There is a growing interest in the relationships among ecosystem diversity, structure, and function. A number of theories have been formulated on how microbial species composition and diversity relate to the functional capability of the terrestrial ecosystem.^[3] It has been suggested that enhanced microbial species diversity is beneficial to ecosystem function.^[4,5] Others suggest that the properties of an ecosystem depend more on the functional capabilities of a particular microbial species than on the total number of species.^[6–8]

COMPOSITION OF SOIL MICROBIAL COMMUNITIES

The soil contains billions of organisms representing nearly every phylum.^[9] Microbial communities are the most complex and diverse group of soil organisms, ranging in size from 0.5 to 5.0 μm , which consists predominantly of bacteria, fungus, actinomycetes, and lichens.^[2,10] A simplified,

general classification of soil microorganisms is presented in [Table 1](#).

Soil bacteria are single-celled, prokaryotic organisms in various shapes and sizes (0.5–5 μm). They are perhaps the most complex and diverse group of soil microorganisms (about 20,000 different species per gram of soil) and are adapted to most environments.^[2,10] In general, bacteria are not energy-efficient organisms in soil (only about 20% of the organic substances decomposed may become bacterial tissue).^[11]

Soil fungi are eukaryotic organisms that may form visible macroscopic structures in soil. Fungi comprise an extremely diverse group of microorganisms with tens of thousands of species present in the soil. Although they are usually in somewhat smaller numbers than bacteria, fungi dominate the biomass and metabolic activity in many soils because of their relatively large size and branching. Fungi are energy-efficient organisms in soil (30–50% of the organic substances decomposed may become bacterial tissue).^[11]

Like the fungi, actinomycetes are filamentous and often profusely branched cells, although their mycelia threads are much smaller than those of fungi. Actinomycetes were previously classified with the fungi; however, they are now classified as bacteria. They have no nuclear membrane and often dissociate into spores that closely resemble bacterial cells. We will consider them separately because of the unique roles they play in the soil compared to bacteria.^[2,10]

Although actinomycetes develop best in moist, warm, well-aerated soil, they are functionally important in arid-region, salt-affected soils.

Algae are eukaryotic cells ranging in size from 2 μm to 20 μm . Several hundred algal species have been identified in soil, but only a small number of species are prominent throughout the world. Most algae grow best

Table 1 Relative numbers and biomass of microbial species at 0–15-cm depth of soil.

Microorganisms	Number per gram of soil	Biomass (g/m ²)
Bacteria	10 ⁸ –10 ⁹	40–500
Actinomycetes	10 ⁷ –10 ⁸	40–500
Fungi	10 ⁵ –10 ⁶	100–1500
Algae	10 ⁴ –10 ⁵	1–50

Source: From Adu & Oades.^[11]

under moist to wet conditions, but some thrive in hot or cold environments.

DETERMINATION OF SOIL MICROBIAL COMMUNITIES

A number of techniques have been developed to culture, identify, and characterize microbial communities. Such methods characterize substrate utilization by *in situ* microbial communities or by part of the community that is culturable in Biolog GN microplates, which are used to produce community-level physiological profiles. As no single universal method is available to determine the whole microbial community, a combination of methods was developed over time. Methods to analyze microbial community structure and function have been generalized for dominant groups such as bacteria and fungi. Population may be determined through direct counting,^[12,13] specific cell component analysis,^[14–16] substrate-induced respiration,^[17] chloroform fumigation and microwave incubation, and extraction methods^[18–20] and DNA analysis.^[21] The use of molecular technology, including polymerase chain reaction amplification, cloning, and sequence analysis of the 16S subunit of ribosomal RNA, is becoming a much more common means of estimating the composition of the bacterial community.^[21]

FACTORS AFFECTING SOIL MICROBIAL COMMUNITIES

The soil ecosystem comprises a variety of microhabitats with different physicochemical gradients and discontinuous environmental conditions that are influenced by the physical and chemical properties of the soil, the type of vegetative cover, the climate,^[22] and the soil-crop management practices. The structure and function of soil microbial communities reflect between hosts of biotic and abiotic components of the terrestrial ecosystem.

Amount, Quality, and Placement of Organic Residues

Organic carbon (C) compounds, from plant residues and soil organic matter, are used as energy and food sources by

the heterotrophic microorganisms in soil. The quality of the plant litter reflects the biochemical composition of the substrates and the physical availability of the substrates to the microorganisms.^[23] Bacteria tend to respond most rapidly to additions of simple C compounds such as starch, sugars, and amino acids, while fungi and actinomycetes dominate if complex C compounds such as cellulose and more resistant lignin materials are available.^[24] In addition, if organic residues are deposited on the soil’s surface (under conifer forest and no-till systems), fungi dominate the microbial activity. Bacteria commonly play a larger role if the substrates are mixed into the soil, as by earthworms, root distribution, and tillage.^[25]

Soil Moisture, Aeration, and Temperature

While most microbes are aerobic and use oxygen as the electron acceptor in their metabolism, some microbes are anaerobic and use chemical compounds with combined oxygen other than gaseous oxygen. Facultative microbes use either aerobic or anaerobic forms of metabolism. Usually, these three metabolisms simultaneously occur in different habitats within a soil. Soil microbial activity is generally greatest when temperatures are within 20–40°C. The warmer end of this range tends to favor actinomycetes. Except for certain cryophilic species, most microorganisms cease metabolic activity below about 5°C, a temperature sometimes referred to as biological zero.^[2,10]

Soil Reaction

Soil reaction is a factor in determining which specific microorganisms dominate and function in a particular soil. Although some bacterial species may thrive in any chemical condition of the soil, near-neutral pH results in the largest and most diverse composition of bacterial populations. Acidity enhances soil fungi activity. The acidity effect explains why fungi tend to dominate in forested soils while bacterial biomass generally dominates over fungi in slightly acidic subhumid to semiarid prairie and rangeland soils.

Management Practices

Type, timing, and degree of tillage affect microbial processes by placing plant residues on soil or in soil contact, exposing native and protected C, and by altering soil aeration, moisture, and temperature conditions.^[26,27] Once in the soil, plant residues serve as energy and C sources for heterotrophic organisms.^[28] In general, greater additions of plant residue to soil under conservation management practices are almost always followed by a flush of biological activities, coinciding with an increase in microbial biomass and metabolic activity.^[27] The incorporation of easily decomposable plant residues with relatively low C:nitrogen (N) ratio is a unique feature of legume-based cropping

system, which enhances soil biological processes compared to conventional systems.

ROLE OF SOIL MICROBIAL COMMUNITIES

The processes and functions of the terrestrial ecosystem are regulated by the activities of soil microorganisms. Some of the important ecological functions related to the activities of the microbial community are litter decomposition, nutrient cycling, bioremediation, C sequestration, and improvement of the physical properties of the soil.

Litter Decomposition and Nutrient Availability

Litter decomposition is perhaps the most significant contribution of the soil microbes in terrestrial ecosystem. Through this process, the complex chemical compounds in dead leaves, roots, and other plant tissues are broken down into simpler chemical compounds, converting organically held nutrients into mineral forms available for renewed plant uptake. The release of N from decomposition of organic residues is a prime example. Soil microbes also assimilate wastes from animals and other organic materials added to soils. As a by-product of their metabolism, microbes synthesize new compounds, some of which help stabilize soil structure, while others contribute to formation of soil organic matter and improve soil quality over time.^[27]

Inorganic Nutrient Transformations

The transformation of inorganic nutrient elements and compounds is of great significance to the ecological functions of the soil systems, including plant growth. Nitrate, sulfate, and, to a lesser degree, phosphate ions are present in soil primarily because of microbial activities. Bacteria and fungi assimilate some of the N, phosphorous (P), and sulfur (S) in the organic materials they digest. Excess amounts of these nutrients may be excreted into the soil system in inorganic forms usable either by the bacteria and fungi themselves or by other organisms that feed on them. In this process, organically bound forms of N, P, and S are converted into inorganic forms that are available to higher plants. Likewise, the solubility and availability of the other essential elements, such as iron (Fe) and manganese (Mn), are largely determined by microbial activities in soil. In well-drained soils, these elements are oxidized by autotrophic microorganisms to their valence states, in which forms they become quite insoluble, which keeps Fe and Mn mostly in non-toxic forms even under fairly acidic conditions. Microbial oxidation also controls the potential for toxicity in soil contaminated with selenium, arsenic, and chromium compounds.

Biological N Fixation

N is one of the primary nutrients for plant growth. Atmospheric N₂ is an inert gas that cannot be directly used by higher plants. Biological N fixation is one of the most important microbial processes in soils. Actinomycetes of the genus *Frankia* fix major amounts of atmospheric N in forest ecosystems; cyanobacteria are important in flooded rice systems, wetlands, and deserts; and rhizobia are the most important group for the fixation of gaseous inert N in agricultural soils.^[10] By far, the greatest amount of N fixation by these microorganisms occurs in root nodules or in other close associations with plants. Worldwide, every year, microbes in soils fix enormous quantities of atmospheric N into forms usable by higher plants.^[2]

Soil Aggregate Stability and Carbon Sequestration

Aggregate structure is a key biologically mediated soil property that results from close associations of fungal hyphae with plant roots and from release of low molecular weight organic acids associated with clays, metals, and microaggregates.^[29] In addition to C retention through efficient cell assimilation, extensive fungal hyphal associations with plant roots can contribute to protect and accumulate C by enhancing macroaggregation through physical enmeshing of soil microaggregates or releasing extracellular polysaccharide as binding agents or stabilization through complex association with metals as microaggregates.^[30,31]

The microbially derived decomposition products cement primary particles, organic debris, and microaggregates together to form and stabilize macroaggregates.^[32] An enhanced physical protection of fragmented organic debris and microbial decomposition products within macroaggregates is an important mechanism enabling the soil C accumulation.^[32] These mechanisms, in turn, increase the proportion of C in soil aggregates that is physically protected from microbial decomposition and protects soil from accelerated soil erosion.^[27]

Breakdown of Toxic and Xenophobic Compounds

Many organic compounds toxic to plants or animals find their way into the soil. Some of these toxins are produced by soil microbes as metabolic by-products; some are applied as agrochemicals to kill pests and insects and to control weeds; and others are deposited in the soil because of environmental contamination. If these compounds continuously accumulate, they would do enormous ecological damage. Some toxins are xenophobic compounds foreign to biological systems, and these may resist attack by commonly occurring microbes. Soil bacteria and fungi are especially important in helping maintain a non-toxic soil environment by breaking down toxic compounds.

Plant Protection and Diseases

Certain soil microbes protect higher plants from invasion by soil parasites and pathogens; however, others attack plant roots. According to several research studies, many plants that are resistant to soilborne diseases have camouflaged rhizosphere,^[10,33] that is, although the rhizosphere of these resistant plants is enriched in microbial numbers, the types of microbes are more similar to those in the bulk soil than in the rhizosphere of disease-susceptible plants. Having a rhizosphere microbial community that is similar to the bulk soil community may make the rhizosphere of these plants less identifiable to pathogens.

Beneficial rhizobacteria have an intriguing mode of action called induced systematic resistance, which prevents infection of plants by diseases, insects, or pests, both above and below ground. In many cases studied so far on crops, the resistance-inducing organism was *Pseudomonas* or *Serratia* bacterium.^[10,33] This mechanism was shown to effectively reduce damages by numerous fungal, bacterial, and viral pathogens, and several leaf-eating insect pests.^[33]

Disease infestations occur in great variety and are induced by many different soil microorganisms. Among the microorganisms, bacteria, actinomycetes, and fungi are responsible for most of the common soilborne diseases of crops. Examples of such diseases are wilts, damping-off, root rots, clubroot of cabbage, and blight of potatoes.

INTERACTIONS OF SOIL MICROBIAL COMMUNITIES

With the diversity in a soil ecosystem, many types of interactions of the microbial community with plant roots and other soil organisms, and between several microorganisms, are reported. The most important associations are given below.

Antagonistic Interactions

Among the antagonistic interactions, parasitism, where one species resides within another, and predation, where one species actively pursues another, are deleterious associations of two microbial populations in the sense that one organism gains an advantage at the expense of another. Bacteria that prey on other bacteria are active in soil ecosystems.^[2,34,35] Protozoa and nematodes^[36] that feed on bacteria are also active in the soil ecosystem. This can have a significant effect on population.^[37] Parasitism of *Pythium coloratum*, the fungi responsible for cavity-spot disease in carrots, by actinomycetes has been reported.^[38]

Competition Interactions

Competition occurs when the growth of one species or community is suppressed by another species or community that more effectively obtains a limiting resource, or when one species or community is inhibited by the products of a second species or community. Limiting resources may be electron acceptors,^[37,38] mineral resources,^[39] host organisms,^[40] and carbon sources.^[41]

When fresh organic residues are added to the soil, the vigorous heterotrophic soil microbes compete with each other for this food resource. If simple compounds such as sugars, starches, and amino acids are available in the organic residues, the bacteria initially dominate because of their rapid growth and preference for simple compounds. As these simple compounds are metabolized, the fungi, and particularly the actinomycetes, become more competitive in soil by altering habitat conditions. Certain microbes alter the acidity level in their vicinity to the disadvantage of competitors. Others produce strong bonding compounds to protect the food source.

Production of antimicrobial compounds may also limit the development of other microbes. The potato soft-rot pathogen, *Erwinia carotovora* subsp. *atroseptica* (van Hall) Dye, is controlled by the production of the antibiotic 2,4-diacetylphloroglucinol by *Pseudomonas fluorescens* (Trevisan) Migula F113.^[42] However, the production of antibiotics does not eliminate other possible mechanisms of action. It has been reported that non-pathogenic strains of *Agrobacterium* prevent expression of disease rather than killing the pathogens.^[43]

Mutualistic Interactions

A mutually beneficial relationship between and among microorganisms is common in soil. The most familiar is the relationship between leguminous plants and rhizobia to fix atmospheric N. Symbiotic relationships are also found between populations of N-fixing communities and non-fixing communities,^[44] and *Rhizobium* and arbuscular fungi.^[45] One of the most ecologically and economically important activities of soil fungi is the mutually beneficial association between certain fungi and the roots of higher plants. This association is called mycorrhizae, a term meaning “fungus root.” In natural ecosystems, especially in degraded sites, many plants are quite dependent on mycorrhizal relationships and cannot survive without them; however, both parties benefit from such relationship.

The microbiotic crusts are intricate living systems in arid rangelands. They consist of mutualistic associations that usually include algae or cyanobacteria along with fungi, mosses, bacteria, and liverworts.^[10] The presence of microbiotic crusts is considered a sign of healthy or recovering ecosystems.

CONCLUSION

The soil is a complex and dynamic component of the terrestrial ecosystem with a diverse community of microbes dominated by bacteria, fungi, actinomycetes, and algae. The microbial community is vital to the cycle of life on earth. They incorporate plant and residues into the soil and digest them, returning carbon dioxide to the atmosphere, where it can be recycled through higher plants. Simultaneously, they release nutrients during decomposition of organic residues and form soil organic matter, which is vital for soil quality. With the diversity in a soil ecosystem, many types of community interactions are possible, including parasitism, competition, and symbiosis.

REFERENCES

- Whipps, J.M. Microbial interactions and biocontrol in the rhizosphere. *J. Exp. Bot.* **2001**, *52*, 487–511.
- Tate, R.L., III. *Soil Microbiology*; John Wiley & Sons, Inc.: New York, 1995; 147–170.
- Muller, A.K.; Westergaard, K.; Christensen, S.; Sorensen, S.J. The diversity and function of soil microbial communities exposed to different disturbances. *Microb. Ecol.* **2002**, *44*, 49–58.
- Naeem, S.; Thompson, L.J.; Lawler, S.P.; Lawton, J.H.; Woodfin, R.M. Empirical evidence that declining species diversity may alter the performance of terrestrial ecosystems. *Philos. Trans. R. Soc. Lond.* **1995**, *347*, 249–262.
- Tilman, D.; Wedin, D.; Knops, J. Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* **1996**, *379*, 718–720.
- Hopper, D.U.; Vitousek, P.M. The effects of plant composition on ecosystems processes. *Science* **1997**, *277*, 1302–1305.
- Tilman, D.; Knops, J.; Wedin, D.; Reich, P.; Ritchie, M.; Siemann, E. The influence of functional diversity and composition on ecosystem processes. *Science* **1997**, *277*, 1300–1302.
- Wardle, D.A.; Zackrisson, O.; Horberg, G.; Gallet, C. The influence of island area on ecosystem properties. *Science* **1997**, *277*, 1296–1299.
- Rosello-Mora, R.; Amann, R. The species concept of prokaryotes. *FEMS Microbiol. Rev.* **2001**, *25*, 39–67.
- Brady, N.C.; Weil, R.R. Organisms and ecology of the soil. In *Nature and Properties of Soils*, 13th Ed.; Prentice Hall: Upper Saddle River, 2002; 449–497.
- Adu, J.K.; Oades, J.M. Utilization of organic materials in soil aggregates by bacteria and fungi. *Soil Biol. Biochem.* **1978**, *10*, 117–122.
- Alexander, M. Most probable number method for microbial populations. In *Methods of Soil Analysis, Part II. Chemical and Microbiological Properties*; Page, A.L., Miller, R.H., Keeney, D.R., Eds.; 2nd Ed.; American Society of Agronomy: Madison, 1982; Vol. 9, 815–820.
- Bottomley, P.J. Light microscopic methods for studying soil microorganisms. In *Methods of Soil Analysis, Part II. Microbiological and Biochemical Properties*; Weaver, R.W., Angle, J.S., Bottomley, P.S., Eds.; Soil Science Society of America, Inc.: Madison, 1994; 81–106.
- Webster, J.J.; Hampton, G.J.; Leach, F.R. ATP in soil: A new extractant and extraction procedure. *Soil Biol. Biochem.* **1984**, *16*, 335–342.
- Joergensen, R.G. Quantification of the microbial biomass by determining ninhydrin-reactive N. *Soil Biol. Biochem.* **1996**, *28*, 301–306.
- White, D.; Davis, W.M.; Nickels, J.S.; King, J.D. Determination of the sedimentary microbial biomass by extractable lipid phosphate. *Oecologia* **1979**, *40*, 51–62.
- Anderson, J.P.E.; Domsch, K.H. A physiology method for quantitative measurement of microbial biomass in soils. *Soil Biol. Biochem.* **1978**, *10*, 519–525.
- Jenkinson, D.S.; Powlson, D.S. The effects of biocidal treatments on metabolism in soil. V. A method for measuring soil biomass. *Soil Biol. Biochem.* **1976**, *8*, 209–213.
- Vance, E.D.; Brooks, P.C.; Jenkinson, D.S. An extraction method for measuring soil microbial biomass-C. *Soil Biol. Biochem.* **1987**, *19*, 703–707.
- Islam, K.R.; Weil, R.R.; Mulchi, C.L.; Glenn, S.D. Freeze-dried soil extraction method for the measurement of microbial biomass C. *Biol. Fert. Soils* **1997**, *24*, 205–210.
- Marilley, L.; Vogt, G.; Blanc, M.; Aragno, M. Bacterial diversity in the bulk soil and rhizosphere fractions of *Lolium perenne* and *Trifolium repens* as revealed by PCR restriction analysis of 16S rDNA. *Plant Soil* **1998**, *198*, 219–224.
- Clark, F.E.; Paul, E.A. The microflora of grassland. *Adv. Agron.* **1970**, *22*, 373–435.
- Wardle, D.A.; Giller, K.E. The quest for a contemporary ecological dimension to soil biology. *Soil Biol. Biochem.* **1996**, *28*, 1549–1554.
- Moller, J.; Miller, M.; Kjoller, A. Fungal-bacterial interactions on beech leaves: Influence on decomposition and dissolved organic C quality. *Soil Biol. Biochem.* **1999**, *31*, 367–374.
- Beare, M.H.; Parmelee, R.W.; Hendrix, P.F.; Cheng, W.; Coleman, D.C.; Crossley, D.A. Microbial and faunal interactions and effects on litter N and decomposition in agroecosystems. *Ecol. Monogr.* **1992**, *6*, 569–591.
- Dick, W.A. Tillage system impacts on environmental quality and soil biological parameters. *Soil Till. Res.* **1997**, *41*, 165–167.
- Islam, K.R.; Weil, R.R. Soil quality indicator properties in mid-Atlantic soils as influenced by conservation management. *J. Soil Water Conserv.* **2000**, *55*, 69–78.
- Van de Geijn, S.C.; Van Veen, J.A. Implication of increased carbon dioxide levels for carbon input and turnover in soils. *Vegetation* **1993**, *104/105*, 283–292.
- Tisdall, J.M.; Oades, J.M. Organic matter and water-stable aggregates in soils. *J. Soil Sci.* **1982**, *33*, 141–163.
- Oades, J.M. Soil organic matter and structural stability: Mechanisms and implications for management. *Plant Soil* **1984**, *76*, 319–337.
- Chenu, C. Influence of a fungi polysaccharide, scleroglucan, on clay microstructures. *Soil Biol. Biochem.* **1989**, *21*, 299–305.
- Jastrow, J.D.; Miller, R. *Soil Aggregation in the Rhizosphere: Optimal Conditions for Multiple Mechanisms*. In The Ecological Society of America 85th Annual Meeting, Snowbird, UT, 2000; 21 pp.

33. Imran, A.S.; Shaukat, S.S. Mixtures of plant disease suppressive bacteria enhance biological control of multiple tomato pathogens. *Biol. Fert. Soils* **2002**, *36*, 260–268.
34. Casida, L.E. Competitive ability and survival in soil of *Pseudomonas* strain 679-2, a dominant, nonobligate bacterial predator of bacteria. *Appl. Environ. Microbiol.* **1992**, *58*, 32–37.
35. Germida, J.J.; Casida, L.E. *Ensifer adherens* predatory activity against other bacteria in soil, as monitored by indirect phage analysis. *Appl. Environ. Microbiol.* **1983**, *45*, 1380–1388.
36. Mattison, R.G.; Harayama, S. The predatory soil flagellate *Heteromita globosa* stimulates toluene biodegradation by a *Pseudomonas* sp. *FEMS Microbiol. Lett.* **2001**, *194*, 39–45.
37. Mikola, J.; Sulkava, P. Responses of microbial-feeding nematodes to organic matter distribution and predation in experimental soil habitat. *Soil Biol. Biochem.* **2001**, *33*, 811–817.
38. El-Tarabily, K.A.; Hardy, G.E.; Sivasithamparam, K.; Hussein, A.M.; Kurtböke, D.I. The potential for the biological control of cavity-spot disease of carrots, caused by *Pythium coloratum*, by streptomycete and non-streptomycete actinomycetes. *New Phytol.* **1997**, *137*, 495–507.
39. Kotsyurbenko, O.R.; Glagolev, M.V.; Nozhevnikova, A.N.; Conrad, R. Competition between homoacetogenic bacteria and methanogenic archaea for hydrogen at low temperature. *FEMS Microbiol. Ecol.* **2001**, *38*, 153–159.
40. van Bodegom, P.; Stams, F.; Mollema, L.; Boeke, S.; Lefelaar, P. CH₄ oxidation and the competition for O₂ in the rice rhizosphere. *Appl. Environ. Microbiol.* **2001**, *67*, 3586–3597.
41. O'Hara, G.W. Nutritional constraints on root nodule bacteria affecting symbiotic nitrogen fixation: A review. *Aust. J. Exp. Agric.* **2001**, *41*, 417–433.
42. Cronin, D.; Moenne-Loccoz, Y.; Fenton, A.; Dunne, C.; Dowling, D.N.; O'Gara, F. Ecological interaction of a biocontrol *Pseudomonas fluorescens* strain producing 2,4-diacetylphloroglucinol with the soft rot potato pathogen *Erwinia carotovora* subsp. *atrosetica*. *FEMS Microbiol. Ecol.* **1997**, *23*, 95–106.
43. Johnson, K.B.; DiLeone, J.A. Effect of antibiosis on antagonist dose–plant disease response relationships for the biological control of crown gall of tomato and cherry. *Phytopathology* **1999**, *89*, 974–980.
44. Bever, J.D.; Simms, E.L. Evolution of nitrogen fixation in spatially structured populations of *Rhizobium*. *Heredity* **2000**, *85*, 366–372.
45. Biro, B.; Koves-Pechy, K.; Voros, I.; Takacs, T.; Eggenberger, P.; Strasser, R.J. Interrelations between *Azospirillum* and *Rhizobium* nitrogen-fixers and arbuscular mycorrhizal fungi in the rhizosphere of alfalfa in sterile, AMF-free or normal soil conditions. *Appl. Soil Ecol.* **2000**, *15*, 159–168.

Micromorphology and Soil Quality

Ahmet R. Mermut

Department of Soil Science, University of Saskatchewan, Saskatoon,
Saskatchewan, Canada

Abstract

Soil micromorphology has numerous applications to soil quality assessment. These include the assessment of soil pores, aggregates, crusting, biology including ecosystems and faunal activity, soil chemistry, environmental chemistry, and mineralogy. Whereas numerous advances were made in the 1970s and 1980s, there is a vast scope for the use of these techniques.

INTRODUCTION

The book *Micropedology*^[1] and a large number of publications have induced many people to practice soil micromorphology since 1950s. Major areas where micromorphological analyses were used to determine soil quality in the 1970s and 1980s were in soil physics (soil pores, aggregates, and crusting), biology including ecosystems and faunal activity, soil chemistry, environmental chemistry, and mineralogy. These applications were made possible by the advent in computer science and development of image analyses technologies^[2,3] for better quantification of soil features.

So far, quantitative data were obtained from two-dimensional measurements, on the basis of aerial percentage. As is performed in stereology, 2-D measurements are extended to volume percent by the Delesse principle; i.e., aerial percentage is an unbiased estimate of volume percentage.^[4]

SOIL PORES, AGGREGATES, TILLAGE, AND SOIL CRUSTING

Of all the physical measurements, pore size distribution is the most pertinent parameter for plant growth.^[5] Soil porosity is dynamic^[6] and exhibits a high degree of physical anisotropy. Primary focus in porosity studies has been on the measurement of total porosity and shape and size classes of pores. It was discovered that the measurements of porosity itself are not sufficient unless it is coupled with other soil physical, biological, and chemical analyses.^[7] Despite the difficulties, attempts were made to measure the full spectrum of pore sizes, from pore diameters in the order of centimeters to those of nanometers.^[8]

Application of micromorphology in soil structure and aggregates gained considerable momentum in the 1960s, 1970s, and 1980s (Fig. 1).^[9–12] Impact of tillage implements, including zero tillage and crop rotation on soil

structure, received considerable attention.^[13] This was followed by studies dealing with surface crusting and sealing.^[14–16] These studies suggested that the susceptibility of soils to surface crusting and sealing depends on a combination of several physical and chemical properties of soils (Fig. 2).

Several investigators have studied the effect of different kinds of tillage operations on soil porosity and structure (Table 1).^[17–19] The data led to insight of different properties and functions of soil structure and porosity. Pagliai^[20] suggested that conservation or zero-tillage practices, when compared with conventional tillage, maintain soil structure and optimize soil porosity. Several workers have attempted to directly quantify porosity using thin sections or impregnated and polished soil blocks.^[21–23] Yet the influence of tillage operations on soil porosity is not completely appreciated.

Mermut et al.^[24] used image analyses to evaluate the impact of deep ripping and paraplowing on soil porosity in two lacustrine soils in Saskatchewan. Deep ripping increased soil porosity in the surface horizon, whereas the influence of paraplowing was evident up to 30–60 cm depth (Fig. 3). Two tillage methods used in this study increase large planar voids, but non-planar intra-aggregate voids were not influenced by tillage. It has been suggested that pores should be measured more frequently and over a long time, so that a clearer picture of the dynamics is better understood.

Mineralogy of clay sized particles and rainstorm characteristics are among the major factors that determine the nature of soil sealing.^[25] Southard et al.^[26] found that soil material suspended in water containing electrolytes resulted in less dense and more porous crust. Several crust types were identified.^[15,16,27]

SOIL FAUNA

The impact of organisms on the function and quality of soil is an important area of research.^[28,29] By using interactive image analysis system and a video camera,

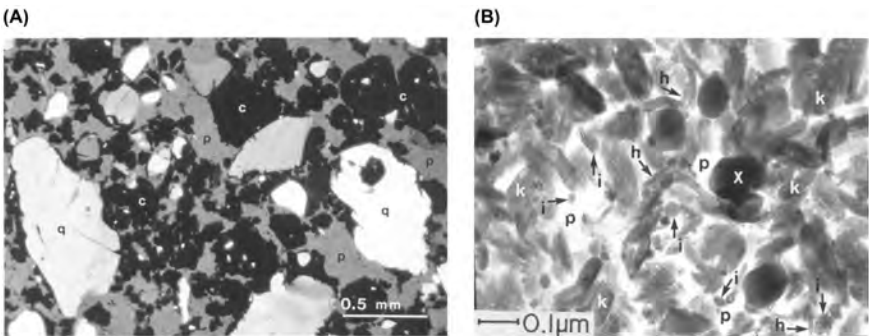


Fig. 1 (A) Thin section micrograph showing interconnected simple and compound and (B) packing void (interaggregate pores) under partially crossed polarized light (p = pores; q = quartz; c = soil aggregates). Right: high-resolution TEM micrographs of an ultracut of primary aggregate (p = porosity; k = kaolinite; h, l, and x are various soil minerals). **Source:** From Bui, Mermut, et al.^[8]

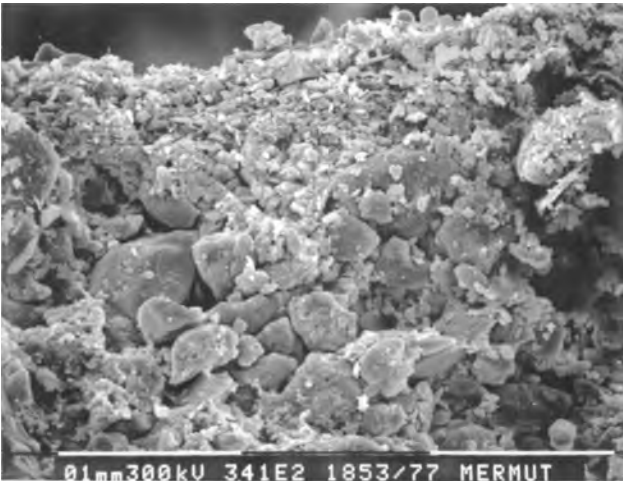


Fig. 2 Scanning electron micrograph of a crust. Note the fine particles covering the surface top central part (the white bar is 100 μm). **Source:** From Arshad & Mermut.^[15]

micromorphologists can measure the size and shape of fecal pellets of animals observed in soil thin sections. Such an analysis can determine the possible ecological significance of the fecal pellets of animals in soils, and the data generated are essential to the protection, use, and management of soils and forests (Fig. 4).

SOIL ORGANIC MATTER

Micromorphologists are also interested in soil organic matter and its dynamics.^[30] Several textbooks and articles were published by Kubiena^[31] between 1938 and 1970 on soil

organic matter, mainly to determine the function and turnover of organic matter in soils. Submicroscopic techniques were also used to study organic matter *in situ* by histochemical staining methods in terms of organic matter function in ecosystem.^[32] This is an important and high-priority area of research.

MICROCHEMISTRY AND SUBMICROSCOPIC STUDIES

Pioneering works in undisturbed *in situ* microchemical analyses of polluted soils have been carried out on heavy metals.^[33] When coupled with image analyses, these techniques provide not only the location, but also quantity and distribution of the pollutant in soils. With the help of the new instrumentation, such as synchrotron, revolutionary scientific research can be performed.

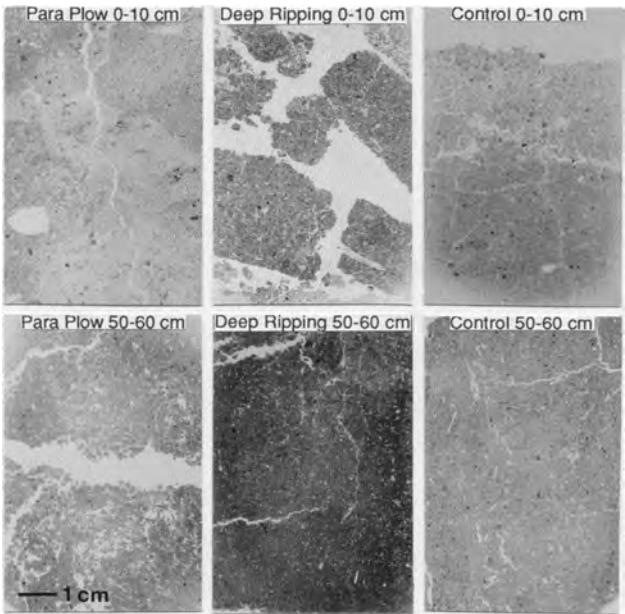


Fig. 3 Contact images of this section representing the control, deep ripping, and paraplow parcel from the Tisdale site, Saskatchewan, Canada. **Source:** From Mermut, Grevers, et al.^[23]

Table 1 Average porosity of the upper 1-cm thick layer of soil measured by image analyses after different irrigation system.

Seed bed		Sprinkler	Furrow irrigation track	Furrow irrigation ridge
29	Bare soil	16	5	16
	Cropped soil	14	16	17

Note: Values are expressed in % of porosity.

Source: From Tedeschi, Melle, et al.^[19]

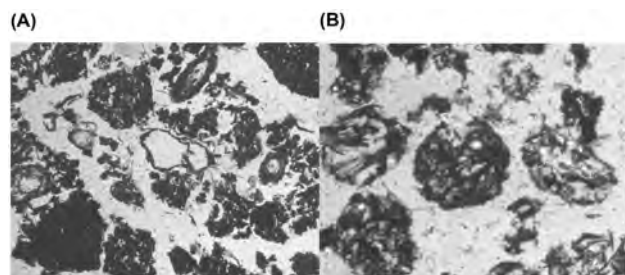


Fig. 4 (A) A soil thin section showing the fecal pellets of a soil-ingesting animal in the surface soil and (B) more rounded fecal pellets. The frame length is about 0.5 cm.

CONCLUSION

Soil micromorphology has numerous applications to soil quality assessment. These include the assessment of soil pores, aggregates, crusting, biology including ecosystems and faunal activity, soil chemistry, environmental chemistry, and mineralogy. Whereas numerous advances were made in the 1970s and 1980s, there is a vast scope for the use of these techniques.

REFERENCES

- Kubiena, W. *Micropedology*; Collegiate Press: Ames, 1938.
- Miedema, R.; Mermut, A.R. *Soil Micromorphology: An Annotated Bibliography 1968–1986*; CAB International: Wallingford, 1990.
- Protz, R.; Sweeney, S.J. An application of spectral image analyses to soil micromorphology. 1. Methods of analyses. *Geoderma* **1992**, *53*, 275–287.
- Weibel, E.R. *Stereological Methods*; Academic Press: London, 1979; Vol. 1.
- Thomasson, A.J. Towards an objective classification of soil structure. *J. Soil Sci.* **1978**, *29*, 38–46.
- Cassel, D.K. Spatial and temporal variability of soil physical properties following tillage of norfolk loamy sand. *Soil Sci. Soc. Am. J.* **1983**, *47*, 196–201.
- Bouma, J.; Jongreius, A.; Boersma, O.; Jager, A.; Schoonderbeck, D. The functions of different types of macro-pores during saturated flow through four swelling soil horizons. *Soil Sci. Soc. Am. Proc.* **1977**, *41*, 945–950.
- Bui, E.; Mermut, A.R.; Santos, M.C.D. Microscopic and ultramicroscopic porosity of an oxisol from pernambuco, Brazil. *Soil Sci. Am. J.* **1989**, *53*, 661–665.
- Bouma, J. *Microstructure and Stability of Two Sandy Loam Soils with Different Soil Management*; Agricultural Research Reports; PUDOC: Wageningen, 1969; Vol. 724.
- Tokaj, J. Micromorphological investigations on soil aggregates. Third international working meeting on soil micromorphology, wroclaw, poland. *Zesz. Probl. Postep. Nauk Rol.* **1972**, *123*, 733–746.
- de Meeter, T.; Jongerius, P.D. The relationship between soil erodability factor K (universal soil loss equation, aggregate stability and micromorphological properties of soils in the Hornos area, Southern Spain. *Earth Surf. Proc.* **1978**, *3*, 379–391.
- Santos, M.C.D.; Mermut, A.R.; Ribeiro, M.R. Submicroscopy of clay aggregates in an oxisol from pernambuco, Brazil. *Soil Sci. Soc. Am. J.* **1989**, *53*, 575–582.
- Jongerius, A. Some micromorphological aspects of regrouping phenomena in Dutch soil. *Geoderma* **1970**, *4*, 311–331.
- Pagliai, M.; Bisdom, E.B.A.; Ledin, S. Changes in surface structure (crusting) after application of sewage sludge and pig slurry to cultivated agricultural soil in Northern Italy. *Geoderma* **1983**, *30*, 35–53.
- Arshad, M.A.; Mermut, A.R. Micromorphological and physico-chemical characteristics of soil crust types in North-western Alberta, Canada. *Soil Sci. Soc. Am. J.* **1988**, *52*, 724–729.
- Bresson, L.M.; Valenti, C. *Comparative Micromorphological Studies of Soil Crusting in Temperate and Arid Environments*. In Transactions 14th International Congress of Soil Science, Kyoto, Japan, August, 12–18, 1990; 236–243.
- Pawluk, S. Micromorphological investigations of cultivated Gray Luvisols under different management practices. *Can. J. Soil Sci.* **1980**, *60*, 731–745.
- Collins, J.F.; Larney, F. Micromorphological observations of compacted horizons (cultivated pans) from various horizons in Irish Tillage Soils. In *Soil Micromorphology*; Fedoroff, N., Bresson, L., Courty, M.A., Eds.; Association Francaise, 1987; 451–457.
- Tedeschi, A.; Melle, G.; Terribile, F. Soil pore geometry changes in surface horizons, under different irrigation regime. In *17th World Congress of Soil Science*; Transactions International Union of Soil Science (IUSS), Bangkok, Thailand, 2002, Symposium 35, 1–12.
- Pagliai, M. Effects of different management practices on soil structure and surface crusting. In *Soil Micromorphology*; Fedoroff, N., Bresson, L., Courty, M.A., Eds.; Association Francaise pour l'Etude du Sol: Plaisir, 1987; 415–421.
- Shipitalo, M.J.; Protz, R. Comparison of morphology and porosity of a soil under conventional and zero tillage. *Can. J. Soil Sci.* **1987**, *67*, 445–456.
- Moran, C.J.; McBratney, A.B.; Koppi, A.J. A rapid method for analyses of soil micropore structure. I. Specimen preparation and digital binary image production. *Soil Sci. Soc. Am. J.* **1989**, *53*, 921–928.
- Grevers, M.C.J.; de Jong, E. Soil structure changes in sun-soiled solonchic and chernozemic soils measured by image analyses. In *Digitization, Processing and Quantitative Interpretation of Image Analysis*; Mermut, A.R., Norton, D.L., Eds.; Geoderma; Elsevier: Amsterdam, 1992; Vol. 53, 289–307.
- Mermut, A.R.; Grevers, M.C.J.; de Jong, E. Evaluation of pores under different management systems by image analysis of clay soils in saskatchewan, Canada. In *Digitization, Processing and Quantitative Interpretation of Image Analysis*; Mermut, A.R., Norton, D.L., Eds.; Geoderma; Elsevier: Amsterdam, 1992; Vol. 53, 357–372.

25. Mermut, A.R.; Luk, S.H.; Romkens, M.J.M.; Poesen, J.W. A. Micromorphological and mineralogical components of surface sealing in loess soils from different geographic regions. *Geoderma* **1995**, *66*, 71–84.
26. Southard, R.J.; Shainberg, I.; Singer, M.J. Effect of electrolyte concentration on the micromorphology of artificial depositional crust. *Soil Sci.* **1988**, *145*, 278–288.
27. Onofriok, O.; Singer, M.J. Scanning electron microscope studies of surface crusts formed by simulated rainfall. *Soil Sci. Soc. Am. J.* **1984**, *48*, 1137–1143.
28. Babel, U. Echytraeen-losungsgefuge in loess (Dropping fabrics from Anchytraeidae in Loess). *Gedoderma* **1969**, *2*, 57–63.
29. Kositra, M.J. The interpretation and classification of features produced by pelecypods (Mollusca) in marine intertidal deposits in the Netherlands. *Geoderma* **1981**, *26*, 83–94.
30. Bal, L. Micromorphological analyses of soils. In *Lower Level in the Organization of Soil Organic Materials*; Soil Survey Papers; Netherlands Soil Survey Institute: Wageningen, 1973; Vol. 6, 174 pp.
31. Kubiena, W. *Micromorphological Features of Soil Geography*; Rutgers University: New Brunswick, 1970.
32. Foster, R. *In situ* localization of organic matter in soils. *Quaest. Entomol.* **1985**, *21*, 609–633.
33. Bisdorf, E.B.A.; Henstra, S.; Werner, H.W.; Boudewijn, P.R.; Knippenberg, W.F.; deGrefte, H.A.M.; Gourgout, J.M.; Migeon, H.N. Quantitative analyses of trace and major elements in thin sections of soils with the secondary ion microscope (Cameca). *Geoderma* **1983**, *30*, 117–134.

Micronutrients and Human Health

Ram Phal Narwal

Ministry of Agriculture, Government of India, New Delhi, India

R.S. Malik

Department of Soil Science, CCS Haryana Agricultural University, Hisar, India

S.K. Malhotra

Ministry of Agriculture, Government of India, New Delhi, India

Bal Ram Singh

Department of Environmental Sciences, Norwegian University of Life Sciences, Aas, Norway

Abstract

Nutritional health and well-being of humans are entirely dependent on plant foods either directly or indirectly. There are silent epidemics of vitamin and mineral deficiencies affecting people in many developing countries. Micronutrient deficiency is affecting over 2 billion people worldwide. The malnutrition is the result of insufficient intakes of micronutrients through staple foods. Deficiency of zinc (Zn), iron (Fe), manganese (Mn), copper (Cu), boron (B), and molybdenum (Mo) has been observed in 49%, 12%, 5%, 4%, 33%, and 13% soils of India, respectively. Among micronutrients, Zn is highly deficient micronutrient and the deficiency of Zn in soils is further expected to increase from 49% to 63% by the year 2025. Low availability of micronutrients in soils such as Zn, Cu, and Mo showed a good relationship with micronutrient malnutrition but this relationship was not so evident for Fe. Diversification of cropping system and fertilization for meeting nutritional requirement is essential for reducing malnutrition. This review focuses on causes and amelioration of micronutrients deficiency in human diet by manipulating the agricultural practices that may result in supplying required intake of vitamins and minerals for better health and productivity of humans and animals.

INTRODUCTION

Nutrient sufficiency is the basis for good health, productive, and longevity of life. Primary aim of agricultural productivity is to ensure food security and safety. There is little concern on nutritional value or health promoting qualities of food produced. If agricultural systems fail to provide enough food diversity, quality, and quantity to satisfy all the nutrients essential to human life, health of people will deteriorate and national development efforts will stagnate. Although major emphasis of nutritionist has been on protein energy malnutrition, World Health Organization (WHO)^[1] data show that deficiency of micronutrients is far more severe. Approximately 1.7 million (2.8%) of deaths worldwide are attributed to micronutrients deficiency caused by lower consumption of fruits and vegetables that is regarded as top 10 selected risk factors for global mortality (WHO^[2]). These deficiencies affect the population of Europe and more severely in the transition countries of Eastern Europe, the former Soviet Union, and countries of Central Asia. World Bank has estimated that most affected malnutrition countries may lose up to 2–3% per year of their gross domestic product.

According to WHO, iron (Fe), zinc (Zn), vitamin A (beta-carotene), selenium (Se), and iodine (I) are the most prevalent trace element deficiencies in living being. Population affected by their deficiencies showed low productivity, decreased cognition, learning disability among children, lower worker productivity, increased morbidity and mortality rates, impact on immune system, and high health-care costs.^[3] Pregnant women lacking key nutrients not only increase the risks of death during childbirth but also the risk not to provide proper nutrients to her infant child.

Feeding the world's poor with biofortified seeds/edible parts can significantly improve the content of nutrients in the target population. Challenge ahead lies in enhancing substances (e.g., ascorbic acid, S-containing amino acids, etc.) that promote micronutrient bioavailability or decrease antinutrient substances (e.g., phytate, polyphenolics, etc.) that inhibit micronutrient bioavailability. Fighting Fe-deficiency-created anemia through the use of transgenic and meeting the Zn demands of human heart through the Zn-fortified food grains are the immediate challenges in micronutrient research in soils and plants.

Based upon Zn and Fe deficiency in soils, there is no standard method of relating it to their suboptimal levels in human.

While Zn deficiency can be alleviated by increasing Zn density in food crops by Zn fertilization, such possibilities for managing Fe deficiency are economically unfavorable. Hence, soil conditions and agronomic practices are conducive to be the incidence of micronutrient deficiencies in plants. The production of low micronutrient foodstuff has created concern about nutritional deficiency-related health hazards especially in poor masses of the society in developing countries.^[4] It is imperative to initiate studies on such global problems with due attention.

MAGNITUDE OF MICRONUTRIENT DEFICIENCIES IN SOILS

According to Food and Agriculture Organization estimates, 50% of world's soils growing cereal crops are Zn deficient. In India, analysis of 250,000 surface soil samples collected from arable soils revealed the predominance of Zn deficiency in divergent soils. Out of these samples, 49%, 12%, 5%, 4%, 33%, and 13% soils are tested to be deficient in Zn, Fe, Mn, Cu (DTPA – Diethylene triamine penta acetic acid extractable), B (hot water-soluble) and Mo (Ammonium oxalate extractable), respectively.^[5]

Shukla et al.^[6] prepared micronutrients-deficiency (Zn, Fe, Cu, Mn, and B) status maps based on the status of micronutrients availability in soils of the different districts of India. Map depicted Zn-deficiency status in soils of country (Fig. 1) revealed that out of 379 districts, 84 districts fall in the category where Zn deficiency is less than 10%, 53 districts fall in the range where Zn deficiency varied from 10% to 20%, 44 districts in 20–30%, 38 districts in 30–40%, and 40 districts in the category of 40–50% deficiency. Even after regular use of Zn fertilizer in many parts of the country, Zn deficiency in 120 districts more than 50%, indicating that about one-third of the country's vast area is acute deficient in Zn. The incidence of Zn deficiency was observed more in cereal belts of the country, particular rice and wheat growing areas, however, the distribution of Zn deficiency has changed and has extended to coarse cereals and pulse growing areas too. Increase in Zn-deficiency areas is due to intensification of agricultural systems inducing imbalance use of macronutrient, increasing Zn demand of new cultivars, altering Zn availability, and hastening depletion of readily-available soil Zn pools. Crops grown in these soils have low Zn content in shoot and seed.

A strong relationship between Fe availability in soil and occurrence of Fe anemia deficiency disorder is lacking in various parts of the country.^[6] By and large most of the soils do not respond to Cu and Mn fertilization. Deficiency of B is emerging fast in many soils due to extensive depletion of native soil B.

STATUS OF RESEARCH ON THE PROBLEM

In general, discussions on sustainable growth of agriculture revolve around food security. Conscientious deliberations

on the role of agriculture in human health remain a typical miss. Contrary to this prevailing apathy, agriculture is viewed as an instrument for public health. Since the use of pulses has been on the decline and intake of foods from animals is rather low, possible prevalence of micronutrient becomes common. In order to alleviate micronutrients and vitamin A malnutrition affecting over 3 billion people, a global challenging program on biofortification has been approved by the Consultative Group on International Agricultural Research. However, this single bullet solution does not seem wholesome to lessen micronutrient malnutrition when adversaries such as phytates, which reduces bioavailability continue to diminish value of ultra-nourishing crops. The high demand of marginalized sections of society and those who require more nutrients than others may not be entirely met by biofortified cereals alone. Hence, there is a need for innovating holistic-diet approaches combining high-yielding nutrient dense crops, inhibiting adverse effects of anti-nutrient factors, nutrient enhancing food enrichment techniques, and supplementation programmed to tackle hunger and malnutrition.^[5]

The physiological impacts of micronutrients are complex, relating to many bodily functions. Trails conducted in several countries indicate that duration and severity of major baby-killer such as diarrhea and pneumonia can be reduced by 30–50% by supplying adequate amount of vitamin A and Zn.^[7] In developing countries, Zn deficiency ranks 5th along with the Fe deficiency that is at 6th position globally, among the leading 10 risk factors.

In human nutrition, protein content of staple food grains has received maximum attention. Studies on other nutrients, such as beta-carotene, Fe, Zn, etc., have been initiated from 2002 to 2003 but that too in a non-inclusive fashion. Thus, whether it is unlocking the genetic diversity, improving bioavailability by promoting or demoting buildup, respectively of synergistic and antagonistic substances, enhancing bioavailability by modernizing indigenous cooking treatments or fortifying staple foods with nutrient-rich supplements and complements is in rudimentary and a highly fragmented condition. There is hardly any concerted effort to relate and quantify relationship within soil–crops–human health continuum. This sordid state of affairs prevails in India also when there is a strong database emanating from a 40-year program on micronutrient in soils and plants.

GLOBAL STATUS OF MINERAL NUTRITION DISORDERS AND THEIR EFFECT ON HUMAN HEALTH

Since the scientific and technical discoveries that food contains important vitamins and minerals and their deficiency may cause health problem, micronutrients became the focus of research for amelioration of their problems. The

Spatial Variation in Available Zinc Deficiency Status in Soils of Different Districts of India

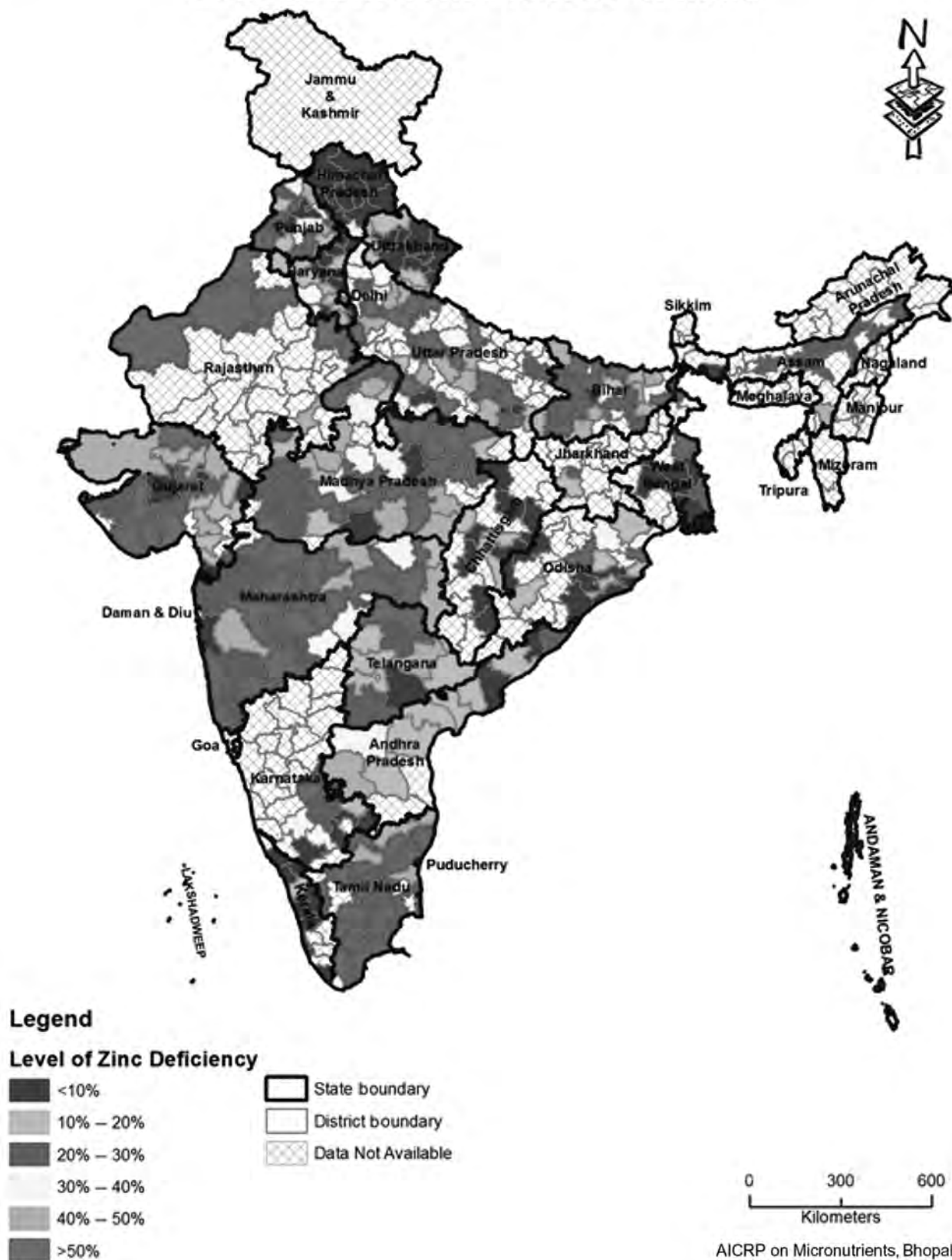


Fig. 1 Spatial variation in available Zn-deficiency status in soils of different districts of India.

details of the disorders on account of mineral nutrition deficiencies and their likely impact on human health are listed as below:

Zn

Zn deficiency in human diet was reported as early as 1961 and its deficiency in human has been observed in Africa and Asia. It is estimated that >90% coverage with Zn supplementation program to prevent Zn deficiency would reduce child mortality by ~5% globally (WHO^[1] and Graham et al.^[8]). The WHO studies indicated that approximately 30% of children worldwide had stunted growth and thus suggesting wide spread occurrence of Zn deficiency. People feeding on grains and vegetation showed lower Zn content in their blood plasmas serum compared to areas that had high-available Zn status and lesser deficiency of Zn in soils.^[9]

Zn deficiency in children has been connected with allergic illness, eczema, learning problems, and hyperactivity. Zn deficiency also cause anorexia (failure to eat), growth retardation, skin lesions, rough and dry skin, immunosuppression, loss of taste, infertility, premature birth, and low birth weight. Zn supplements in diet reduced diarrhea in infants, improved intestinal permeability and appetite in children hence reduced significantly the health risk from Mn deficiency. The suggested safe recommended intake of Zn is 15 $\mu\text{g day}^{-1}$ but WHO^[1] recommended 45 $\mu\text{g Zn day}^{-1}$ as the upper limit and concentration of 150 $\mu\text{g day}^{-1}$ are toxic and cause impairment of health.

Fe

Fe deficiency in human largely results from its low Fe content in diet or its low dietary intake especially by poor individual in developing countries depending largely on cereal diet having low Fe content as well as low bioavailability. Prevalence of Fe deficiency anemia is reported wide spread in woman and children as a major micronutrient disorder problem in several parts of India. Severe Fe anemia was found in 34% in adolescent girls of Rajasthan and Gujarat state.^[10] Fe deficiency anemia is often associated with malaria and hookworm infections. The clinical syndromes in Fe deficient individuals are usually nervous, easily fatigued, lower weight gain, reduced appetite, palpitation on mild exercise, sore tongue, and angular stomatitis. Fe supplements in the form of capsule, tablets, and syrup with folic acid and vitamins are easily available in the market and are prescribed to the targeted Fe-deficient individuals and are helping in reducing the health risk from its disorders.

Cu

Cu is component of many enzymes that are involved in reducing molecular oxygen. Being an essential micronutrient

for man, it involved in lipid metabolism, bones development, and maturation of connective tissue. Signs of its deficiency in human are anemia (hypocupremic), neutropenia and leucopenia (lack of white blood cells), skeletal defects, degradation of nervous system, defective melanin synthesis that manifests as depigmentation or hypopigmentation (lack of color) of hair and skin, keratinization of hair, steely hair, cardiovascular disorders, osteoporosis, arthritis, infertility, and diarrhea. Cu depletion in man leads to elevated levels of serum cholesterol and reversed upon its supplementation. An intake of 2 mg Cu day^{-1} is adequate for healthy adults and 80 $\mu\text{g day}^{-1}$ for infants.^[8]

I

Deficiency of I is the world's most prevalent cause of brain damage. Goiter is the most common form of I deficiency disorder. Endemic goiter has been reported from all over Asia. Its deficiency causes cretinism in children born of parents residing in goiter endemic areas.^[7] The recommended dietary intake level is 150 $\mu\text{g capita}^{-1} \text{day}^{-1}$ or 1 mg or less in the thyroid gland.

Se

Se is a component of enzyme glutathione peroxidase, which protects against oxidative stress. The biological action of Se is similar to that of vitamin E and both substances are antioxidants in their own way and at its specific sites that protect the cells of the body against oxidative damage.

The most important disease that results from Se deficiency is "nutrient muscular dystrophy" termed as "white muscle disease." It manifests as chalky white striations and necrosis in skeletal and heart muscles. It causes hepatic necrosis, edema of colon, lungs, dystrophy of skeletal, poor reproductive performance, impaired immune responses, reduced milk, and egg production, etc., generally it causes death of swine. A daily Se intake below 20 mg caused "Keshan" disease—development of a cardiac disease, identified in 1935 in human and first Se disorder discovered in 1849 in the mountains and hills of Central China (WHO^[11]).

Vitamin A

Vitamin A deficiency (VAD) is a common cause of preventable blindness in children and a risk factor for increased severity of infectious disease and mortality. Its deficiency can lead to poor night vision, eye lesions and in severe cases, permanent blindness, increased illness, and death from infections. In pregnant women, VAD causes night blindness and may increase the risk of maternal mortality.

STRATEGIES FOR CORRECTION OF MICRONUTRIENTS MALNUTRITION IN HUMAN

Some of the interventions have been and could be used to alleviate micronutrient malnutrition problem in human globally are discussed below:

1. Supplementation

Micronutrient deficiencies have been successfully corrected through their supplements in the targeted population. Supplementation approaches are expensive short-term strategies as these rely largely on government or donor support and individual compliance. It is suitable for certain populations whose micronutrient status must be improved over a short period. Some of programs initiated by WHO for supplementation are salt iodization, vitamin A supplementation for children (6–59 months), etc.

2. Fortification

Fortification of food with micronutrients could be a cost-effective sustainable strategy for correcting their deficiencies to reduce health risks as the major cost of fortification and distribution is born largely by the consumers. At national level, fortification with multiple micronutrients could be a cost-effective method for correcting their deficiency disorders in risk groups or regions. Fortification strategy has been successful in many countries especially in the western countries where almost all the processed food are fortified.

3. Dietary modification

It involves changes in food selection patterns and/or in the traditional household methods for preparing and processing indigenous food so that foods with a high content and bioavailability of micronutrient throughout the year. Soaking has been advocated as a practical, non-enzymic, household method to reduce the phytic acid content of certain cereals maize and of legumes soybean, green gram, etc. Also flours produced from germinated and non-germinated staple foods can be mixed together and then soaked and fermented using microbial starter culture for 16–24 hours after which the flours are mixed with water to slurries and cooked to form porridges. This combination of approaches have been shown to reduce the phytic acid content by 90% as well as enhance Zn absorption, protein quality, protein digestibility, microbiological safety, and keeping quality besides decreasing toxins such as hemagglutinins and cyanide.

Improving Micronutrient Content in Seed

Biofortification through efficient crops/crop variety selection

Agriculture is a vital tool of ameliorating micronutrient malnutrition as it is a primary source of all the micronutrients consumed by human and animals worldwide. Approaches

include agronomic practices like cultivation of high density seed or varietal selection, advanced fertilization and organic matter practices, cropping system or farming planning soil, tillage and water management practices, and increased diversity by including of share of traditional millets, legume, or vegetable crops and cropping systems.

Narwal et al.^[12] revealed that the magnitude of susceptibility to Zn, Fe, and Mn deficiencies in different wheat varieties primarily depends on their capacity to utilize soil Zn, Fe, and Mn. However, there is an inevitable need to Zn, Fe, and Mn application on severely deficient soils regardless of wheat variety. While studying ferti-fortification of maize cultivars with Zn at different growth stages, Dhaliwal et al.^[13] found the increase in Zn concentration in all the five maize cultivars.

Microenriched fertilization/ferti-fortification of crops

Application of micronutrients through soil or foliar increased the micronutrients in seed and enhanced the yield. For certain essential micronutrient elements, e.g., Zn, Ni, I, and Se increasing soil available supplies to food crops can result in significant increases in their concentrations in edible plant products.

Micronutrients content in seeds and their bioavailability

Increased micronutrient concentration in seed does not qualify that it will be bioavailable to human and animals. Plant food can contain substances, i.e., antinutrients that interfere with the absorption or utilization of these nutrients in human. Bioavailable micronutrients in the edible parts of staple crops at concentrations high enough to impact on human health can be obtained through breeding, provided that sufficient genetic variation for a given trait exist, or through transgenic approaches. Studies in Thailand also suggest that additional Zn derived from rice grown on soils treated with Zn fertilizer could get absorbed and be potentially at a level that reduces the prevalence of Zn deficiency and has potential to improve growth and immune function in north eastern Thai school children (WHO^[11]).

Agronomic management for better nutrition

Agronomic practices besides genetic improvement play a vital role in increasing yield, high micronutrient density, and reducing antinutritional constituents in seed and fodders. Soil, foliar application, or seed treatment of micronutrient fertilizers are found quite efficient to enhance the micronutrient yield in micronutrient-deficient conditions and thereby improving cattle's and human nutritional health.^[5] Therefore, in order to enhance biofortification of micronutrients in plant parts and their translocation and accumulation in edible part, better understanding of mechanisms

that influences changes in rhizosphere, root cell sap pH, and production of siderophores is very much needed.

Management of Antinutrients

Absorption or utilization of micronutrients depends upon the substances like antinutrients present in food. Breeding of genotypes that contain lower concentrations of antinutrients or altering plant genes in ways that reduce or even eliminate antinutrient from plant foods is very much desired for enhancing Zn or Fe bioavailability. However, doing this involves risk, so it should be done with caution because many antinutrients are major plant metabolites may play important role in plant metabolism and growth and resistance to crop pest or pathogens. It is therefore, recommended that plant breeders closely scrutinize the strategy for increasing bioavailability of micronutrients for human food.

Zn bioavailability is markedly reduced because of high intake of phytate (inositol hexaphosphate) that adversely affects Zn absorption in the body. Phytate: Zn ratio of <5:1, 5–15:1, and >15:1 is considered as an index of high, moderate, and low bioavailability of Zn. Bioavailability of Fe depends upon quantity of heme- and non-heme-Fe diet and also balance between Fe absorption inhibitor and absorption promoter constituents (British Nutrition Foundation Task Force^[14]).

REFERENCES

1. WHO. *Reducing Risk and Promoting Healthy Life. Micro and Secondary Nutrient Research in India*. World Health Organization Report—2002, WHO: Geneva, Switzerland, 2002.
2. WHO. *Global Strategy on Diet, Physical Activity and Health*. World Health Organization Report—2014; WHO: Geneva, Switzerland, 2014; 14–18.
3. Welch, R.M.; Graham, R.D. Breeding for micronutrients in staple food crops from a human nutrition perspective. *J. Exp. Bot.* **2004**, *55* (396), 353–364.
4. Graham, R.D.; Welch, R.M. Plant food micronutrient composition and human nutrition. *Commun. Soil Sci. Plan.* **2002**, *31*, 1627–1640.
5. Singh, M.V. Micronutrient nutritional problems in soils of India and improvement for human and animal health. *Indian J. Fert.* **2009**, *5* (4), 19–26. 56.
6. Shukla, A.K.; Tiwari, P.K.; Chandra, P. Micronutrients deficiencies vis-à-vis food and nutritional security of India. *Indian J. Fert.* **2014**, *10* (12), 94–112.
7. Aswathanarayana, U. Geoenvironment-related disease endemicity in Africa. In *Geomedicine*; Lag, J., Ed.; CRC: Boca Raton, 1995; 235–244.
8. Graham, R.D.; Welch, R.M.; Bouis, H.E. Addressing micronutrient malnutrition through enhancing the nutritional quality of staple foods: Principles perspectives and knowledge gaps. *Adv. Agron.* **2001**, *70*, 77–142.
9. Bhupal Raj, G.; Singh, M.V.; Patnaik, M.C.; Khadake, K.M. Four decades of research of micro and secondary nutrients and pollutants elements in Andhra Pradesh, India. *Research Bulletin. No. 9, 2009*; 1–102. AICRP Micronutrients, IISS, Nabibagh, Bhopal and AICRP micronutrient Unit, ANGRAU, Hyderabad, India.
10. Seshadri, S.; Sharma, K.; Raj, A.E.; Thakore, B.; Saiyed, F. Iron supplementation to control pregnancy anemia. *Proc. Nutr. Soc. India* **1994**, *41*, 131–140.
11. WHO. *Trace Element in Human Nutrition and Health. World Health Organization Report—1996*; WHO: Geneva, Switzerland, 1996.
12. Narwal, R.P.; Dahiya, R.R.; Malik, R.S.; Ramkala. Influence of genetic variability on zinc, iron and manganese responses in wheat. *J. Geochem. Explor.* **2012**, *121*, 45–48.
13. Dhaliwal, S.S.; Sadana, U.S.; Manchanda, J.S. Fertilization of maize cultivars with Zn in relation to food security and alleviation of Zn malnutrition. *Indian J. Fert.* **2013**, *9* (3), 24–30.
14. British Nutrition Foundation Task Force. *Iron Nutritional and Physiological Significance*; Chapman and Hall: London, 1995.

Mine Soils: Measuring Geogenic Carbon

David A.N. Ussiri

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Soil organic carbon (SOC) of the reclaimed minesoils (RMS) formed from coal mining often contains carbon (C) associated with coal particles from mining and the reclamation activities. This geogenic organic C (GOC, i.e. formed as a product of geological processes) derived from coal must be quantified in order to accurately quantify SOC pools, sequestration potential, and ecosystem C budgets of the RMS. The GOC contamination in RMS could lead to overestimation of recent organic C (ROC) pools derived from decomposition of plant biomass and sequestration rates and can also present major challenge due to large C background against which small changes in ROC must be measured. Standard procedures for quantifying SOC pools cannot distinguish GOC and ROC, and there is no standardized method for quantifying GOC in the environment. Methods for quantifying GOC in RMS have been grouped into microscopic, thermal, chemical, spectroscopic, isotopic, and combination of some of these methods. Radiocarbon analysis, one of the most reliable methods for quantifying GOC in RMS, needs specialized instrumentation, advanced computational skills, and high analytical costs, which preclude its adoption for routine soil analysis. Additional research and further evaluations of the existing techniques are needed to develop some reliable and cost-effective methods and ultimately propose standard GOC quantification methods that can be widely adopted for routine analysis.

INTRODUCTION

Minesoils are soils formed on landscapes altered by human activities of surface mining. Their original soil profile has been disturbed and sometimes partially or completely replaced by earth materials from the depths below 1 m, which are often less weathered than the original undisturbed soil. When these materials are exposed to surface or near surface environment, accelerated weathering may develop soils with properties that differ greatly from the original soil, depending on the parent material and the rate of weathering. Minesoils have developed as a result of disturbance and depositions through various anthropogenic activities such as coal mining, metal mining, sand and gravel mining, limestone quarrying, water bodies dredging, road building, construction, or other mining related activities. Sometimes referred to as spoils, anthropogenic soils or “Anthrosols,” these drastically disturbed soils exhibit profile characteristics, physical, chemical, and biological conditions that reflect anthropogenic perturbations rather than natural soil-forming processes. Most research dealing with development and properties of minesoils has focused on surface mining of coal, mainly due to extensive coal mining, which has resulted in disturbance of large expanse of land, and also the economic importance of

coal. Coal mining spoils include all non-coal materials disturbed by coal mining operations—rocks, rock fragments, soil, and other natural materials.

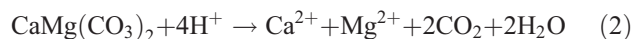
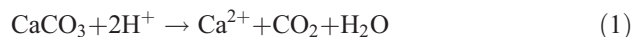
Surface mining operations reduce soil quality drastically by depleting soil organic carbon (SOC) as a result of enhanced mineralization, reduced litter input, soil erosion, and leaching. In addition to recent organic carbon (ROC) from plant litter, geogenic organic carbon [GOC, i.e., organic carbon (OC) that has been subjected to geological processes) may contribute to total OC of the minesoils formed from coal mining operations. The GOC collectively represents lithified plant debris at different degree of coalification formed by biological and geological processes over millions of years and contain a variable amount of aromatic structures depending on the degree of coalification. Decomposition of recent plant litter and other pedogenic processes lead to mixing of recently formed soil organic matter (SOM) with coal particles. Minesoils developed on calcareous parent materials such as siltstone, sandstone, and limestone may also contain significant quantities of soil inorganic C (SIC). Therefore, total carbon (TC) in minesoils developed from coal mining operations can be a mixture of: 1) SIC; 2) GOC; and 3) ROC. Due to the dark color of humus and its intimate mixing with fine coal particles, ROC and GOC cannot be distinguished by visual means. Characterization of ROC accrued as a result of

biochemical processes in reclaimed minesoils (RMS) is necessary for understanding soil development because the quantity and properties of recent organic matter (OM) from plant litter influence chemical and physical properties of soil. It is also important in C dynamics, ecosystem C budgets, and SOC sequestration rates in these soils. Standard methods commonly used for the determination of SOC such as dry combustion do not distinguish between ROC and GOC and may lead to overestimation of SOC pools and sequestration rates in RMS due to coal contamination. Available data suggest that fully restored minesoils have high potential for C sequestration compared to other terrestrial soils, especially during the first few decades following restoration. However, accurate quantification of SOC pools and C sequestration rates in these soils is a challenge, mainly due to coal contamination. Failure to correct for GOC contribution to SOC leads to overestimation of pools and rates of C sequestration in RMS. Available methods for the determination of GOC and correcting SOC pools for coal C and SIC are summarized in the next section.

ANALYTICAL METHODS FOR QUANTIFYING C OF DIFFERENT SOURCES IN RECLAIMED MINED LANDS

Inorganic C

In general, SIC occurs in neutral to alkaline RMS, predominantly as carbonate minerals—calcite (CaCO_3) and dolomite ($\text{CaMg}(\text{CO}_3)_2$). However, solid-phase carbonates in the form of nodules are also known to exist in some acid soil environments. Calcite is usually the dominant form in active pedogenic environments. As presence of SIC interferes with quantification of SOC, it must be removed prior to quantification of ROC and GOC. The SIC is usually removed by acid dissolution the following reactions (Eq. 1 and 2):



Addition of ferrous chloride or ferrous sulfate to the acid is recommended to minimize SOM decomposition during the acid treatment.^[1]

Geogenic and Recent SOC Separation and Quantification

Although some overlaps exist in the chemistry of recent SOM and coal, there are dissimilarities among the two OC sources, including chemical structures, functional groups composition, heat, and chemical reactivity resistance.^[2] For example, recent OM is dominantly composed of easily reactive and oxidizable organic compounds such as carbohydrates, protein, and lipids,^[3] while coal exhibits

highly aromatic and condensed structures as a result of partial decomposition and polymerization during the coalification process over a geological timescale. These differences have been exploited to develop different methods for distinguishing GOC from ROC and quantify the contribution of GOC to TC pool of RMS. Changes in structure during coalification are accompanied by a loss of easily mineralizable organic compounds while more recalcitrant moieties, mostly aromatic structures, are preserved. Although microbial decomposition of geogenic C in minesoils has been reported,^[4] coal is generally considered inert to chemical and microbial degradation^[3] due to the structural recalcitrance and inaccessibility of the small pores in the coal matrix to microorganisms. The quality and type of the ROC available for decomposition generally regulate microbial communities in RMS. Coal contamination contributes to irregular depth distribution of SOC in RMS (Table 1). Coal contamination also occurs in undisturbed soils adjacent to coal combustion plants, and soils ameliorated with coal combustion by-products such as coal ash.

The GOC quantification techniques in soil can be grouped into the following seven general categories: (1) optical and microscopic; (2) thermal; (3) chemical; (4) spectroscopic; (5) molecular markers; (6) isotopic; and (7) radiocarbon analysis. A summary of analytical methods for GOC quantification in RMS and sediments is presented in Table 2. These methods are based on the assumption that GOC and ROC can be distinguished on the basis of their optical, chemical, and/or spectroscopic properties. In practice, violation of these assumptions occurs to a varying degree depending on sample composition and method. Sometimes, to increase the confidence and accuracy of the results, combinations of methods are used. Some of these techniques are labor intensive and may require a level of instrumentation not readily available in many typical soil laboratories. The effectiveness of different methods in relation to coal rank, coal reactivity, and/or coal structure is presented in Fig. 1.

Radiocarbon Analysis Technique

Radiocarbon analysis is one of the most reliable methods for the quantitative measurement of GOC and closest to standard method. However, the specialized analytical equipment, which is not available in many laboratories, specialized skills, and high analytical costs preclude the adoption of this technique for routine analysis. The technique relies on distinguishing GOC and ROC based on the activity of radioactive C (^{14}C), which is unstable and decays over time based on the half-life of 5730 years. ^{14}C is present in environment in low quantities ($10^{-9}\%$), and coal, which formed millions of years ago, does not have ^{14}C activity (dead C)^[3] due to radioactive decay, while recent OM contains ^{14}C activity (active C) due to plant uptake of $^{14}\text{CO}_2$ from the atmosphere. After correction for the stable isotope and isotopic fractionation, plant biomass

Table 1 Distribution of geogenic C in minesoils.

Age since reclamation	Horizon/soil depth	Total OC (g/kg)	Geogenic C (g/kg)	Geogenic C (% of TOC)	References
36-year-old Red oak	Oh (forest floor)	224	56	25	[5]
	Ai (0–5 cm)	110	54	49	
	Cv (1 m)	37	36.9	96	
22-year-old pine	F	350–483	10–28	3–6	[6]
	H	116–346	6–49	5.2–14	
	0–4	15–72	10–66	67–92	
	4–10	22–83	20–78	91–94	
11 years pine	Of (1)	234	96	41	[7]
	Ai 0–2	102	86	84	
	Cv 100	36	36	100	
17 years pine	Of (2 cm)	320	96	30	
	Ai1 (0–1)	80	54	68	
	Ai2 (1–3)	46	37.7	82	
	Cv (3–100)	18		Nd	
32 years pine	Ai (0–5)	180	115	64	
	Cv (5–100)	36	35.6	99	
28 years grassland	0–50 cm	3.6–155.8	0–24.4	0–92	[8]

has the same ^{14}C activity as the atmospheric CO_2 , incorporated into their tissue during photosynthesis, and ^{14}C concentration in living plant tissue is in equilibrium with $^{14}\text{CO}_2$ in the atmosphere. Once the tissue dies and metabolic function ceases, radioactive decay of tissue ^{14}C occurs. By measuring the ^{14}C concentration of the organic material and comparing the measured activity with that of standard modern C sample, the number of decay events per gram of C can be calculated based on the half-life of ^{14}C radioisotope.

The SOC in RMS consists of ROC with variable ^{14}C activity depending on the age of OM and GOC without ^{14}C activity. Based on difference in ^{14}C activity between the two C sources, the relative proportion of GOC can be calculated based on Eq. 3:

$$F = \left[1 - \frac{A_{\text{SOC}}}{A_{\text{recent OC}}} \right] \times 100 \quad (3)$$

where F is the fraction of GOC (%), A_{SOC} is the measured ^{14}C concentration of RMS, and $A_{\text{recent OC}}$ is the ^{14}C concentration of ROC derived from plant material. However, the relative ^{14}C activity in recent OM fraction is not well constrained due to variation in atmospheric $^{14}\text{CO}_2$ concentration as a result of atomic weapons testing from 1950. The ^{14}C concentration in plant tissues has systematically changed over time to reflect the atmospheric changes.^[9] This is also reflected in plant tissues. Rumpel et al.^[10] developed a modeling approach for estimating ^{14}C activity of recent OM in RMS by using time series approach. The ^{14}C activity of the RMS sample is determined using accelerator mass spectroscopy (AMS) after

removing carbonates by HCl treatment.^[11] In radiocarbon dating, year 1950 is year 0 before present (BP) by convention.^[12] The proportion of GOC and ROC can then be calculated using Eqs. 4 and 5:

$$\text{ROC}(\%) = \frac{A_{\text{SOC}}}{A_{\text{recent OC}}} \times \text{SOC} \quad (4)$$

$$\text{GOC}(\%) = 1 - \frac{A_{\text{SOC}}}{A_{\text{recent OC}}} \times \text{SOC} \quad (5)$$

where SOC is the organic C concentration of RMS.

Optical and Microscopic Technique

Optical and microscopic methods rely on optical properties of black color of coal materials to distinguish coal particles from other particles in soil. Optical method uses less sophisticated expertise of observation and microscope to draw the appropriate inferences. The morphology (particle size and shape), surface characteristics, and distribution of particles can also be determined using optical methods. Comparison with standard reference materials allows the identification and quantification of coal particles. Coal particles can be quantified *in situ* on petrographic thin sheets or polished sections. Coal particles can be concentrated by sieving and using sequential washing through a series of sieves of different mesh sizes or chemical pretreatments to improve the accuracy of microscopic analysis (Table 2). Chemical pretreatments include deflocculation of soil aggregates [generally with potassium hydroxide (KOH)] and removal of carbonates and silicates (with HCl and

Table 2 Summarized methods for determination of coal C in soils and sediments.

Method	Material analyzed	Pretreatment	Oxidation/treatment	Geogenic C determination	Merits	Limitations
Optical/Microscopy	Sediment	Sieving, deflocculating, removal of carbonates by HCl	None	Light microscope, electron microscope (SEM or TEM)	Simple Less expensive	Can only detect larger particles Cannot detect coal degradation products qualitative
Thermal oxidation	Soil	Sieving (<2 mm) Removal of carbonates and silicates with 1 M HCl, and 10% HF, respectively	Thermal, heated at 375°C for 24 hr	C, N elemental analysis or mass balance	Less expensive Adaptive for routine analysis	Oxidation temperature overlap for recalcitrant OM and coal Not suitable for lower rank coal
Thermal oxidation and acid hydrolysis	Soils	As above	6 M TFA at 100°C for 18 hr, 6 M HCl 110°C 24 hr, thermal oxidation at 375°C for 24 hr	C, N elemental analysis of the residual		
Chemical oxidation method	Sediments	HCl, HF	Hydrogen peroxide	Counting	Less expensive Adaptive for routine analysis	Labor intensive Chemical oxidation overlap between recalcitrant OM and lower rank coal Overestimation of coal contamination due to oxidation overlap Not suitable for lower rank coal
	Sediments	HCl, HF	$\text{Cr}_2\text{O}_7^{2-}$, H_2SO_4 , heated at 50–60°C for 60 hr	Counting, mass loss		
	Coal	HCl, HF	H_2O_2 , KOH	Infra-red spectroscopy		
	Sediments	HCl, HF	$\text{Cr}_2\text{O}_7^{2-}$, H_2SO_4 , heated to 60°C, 72 hr + KOH, H_2O_2 , heated to 60°C 8 hr	Elemental C, N, H analyzer		
	Sediments	HCl, HF	Hot nitric acid (HNO_3)	GC, combustion, MS		
	Soils	none	$\text{Na}_2\text{S}_2\text{O}_8$ with NaHCO_3 buffer	Elemental analyzer		
	Soils	none	NaOCl + acetic acid + DI water	Elemental analysis, ^{13}C NMR spectroscopy		
Chemical extraction, chemical oxidation, thermal oxidation	Soils	HCl, HF	0.5 M NaOH extraction, 65% HNO_3 , 1% HCl, thermal treatment, 340°C in oxygen or air	Elemental analysis, mass balance	Improved coal recovery	Labor intensive
	Soils					

(Continued)

Table 2 Summarized methods for determination of coal C in soils and sediments. (*Continued*)

Method	Material analyzed	Pretreatment	Oxidation/treatment	Geogenic C determination	Comments	
					Merits	Limitations
Specific molecular marker	Soils	Sieving (<2 mm), carbonates removal	0.5 M NaOH extraction, 60% HNO ₃ oxidation, hydrolysis, 10% HF, thermal oxidation at 340°C for 3 hr	C, N analysis of the residual, mass balance	Adaptive for routine analysis	Labor intensive Requires specialized gas chromatograph for quantifying molecular biomarkers
Ultraviolet photo-oxidation	Soils	Sieving, carbonates and silica removal	65% HNO ₃ 170°C for 8 hr, pressure filtered, washed with DI water, freeze dried	Benzene-carboxylic acid as molecular biomarkers analyzed by GC with FID detector		
¹³ C NMR Spectroscopy	Soils	Digestion (4 M TFA) at 104°C	Digestion (65% HNO ₃ , 170°C 8 hr. BC converted to BPCA	BPCA quantified with gas chromatograph (flame ionization detector)		
	Soils	Physical fractions	Ultraviolet photo-oxidation	¹³ C CPMAS NMR spectroscopy		Overestimation of coal due to incomplete oxidation of ROC Requires NMR spectroscopy
¹³ C stable isotope measurement	Soils and sediments	Grinding, 10% HF treatment to remove paramagnetic materials	None	Relative abundance of functional groups associated with coal by NMR spectroscopy	Simple, adaptive for routine analysis	Requires NMR spectroscopy which may not be available in many soil analysis laboratories Semi-quantitative
	Soils and sediments	Pretreatment to remove inorganic carbon	None	Isotope ratio mass spectroscopy	Simple, adaptive for routine soil analysis	Requires stable isotope ratio mass spectrometer which may not be available in routine soil analytical laboratory
Radiocarbon activity measurement	Soils	Removal of carbonates, grinding and sieving through 250 µm	None	AMS, ¹⁴ C activity	Highly accurate and suitable for calibrating other methods can be used as reference method	Requires AMS for carbon dating, which is not available in many soil analysis laboratories for routine analysis Requires specialized computational skills, which may not be available in routine soil analysis laboratory Expensive

Note: TFA, trifluoroacetic acid; BPCA, benzene poly-carboxylic acid.**Source:** Adapted from Ussiri, Jacintho, et al.^[1,3]

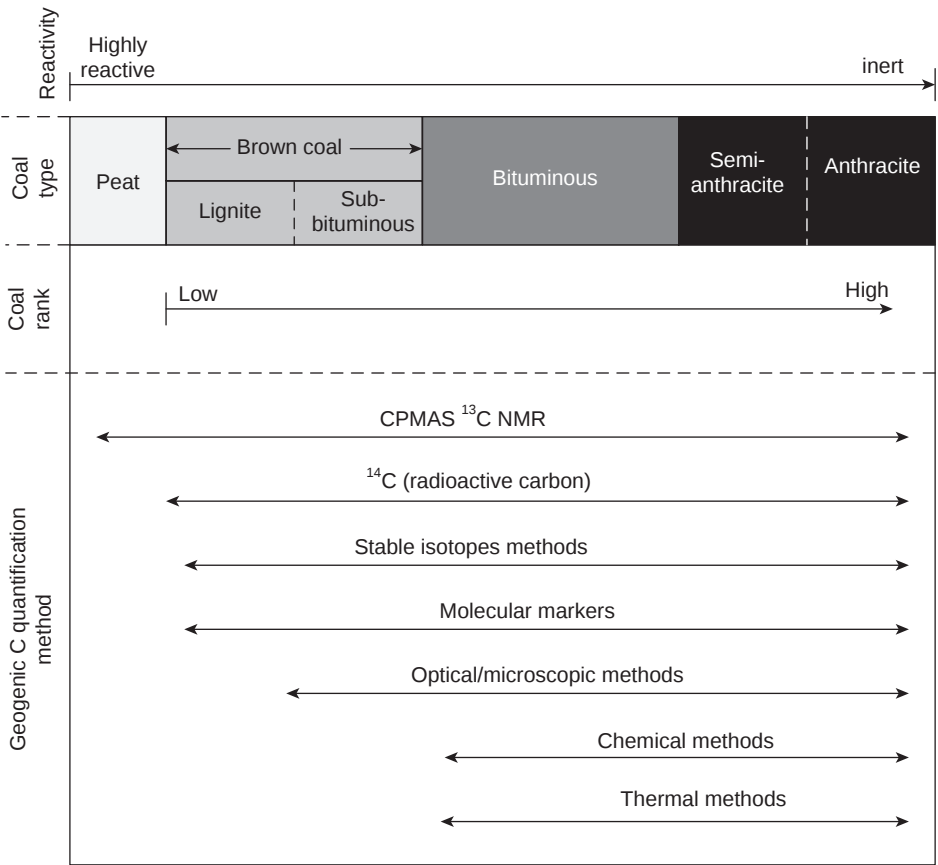


Fig. 1 Coal reactivity analysis continuum suitability of different methods of in the reclaimed minesoils. **Source:** Adapted from Ussiri, Jacinthe, et al.^[13]

HF, respectively) followed by microscopic observation under oil suspension on glass slides. Brightfield light microscope, scanning electron microscope (SEM),^[5] or transmission electron microscope (TEM) can be used depending on the particle size. Coal particles are distinguished from other mineral particles based on shape and color and quantified based on distribution area.^[5] Coupling SEM with energy dispersive X-ray spectroscopy allows the determination of the chemical composition of the particles and their O–C ratios at a nanometer resolution.^[14] If the SEM is coupled with an elemental analyzer, coal particles can be identified and the amount present can be quantified based on composition (C, H, and N) and element ratios.^[15] This procedure is time-consuming and its accuracy relies on correct differentiation between coal and other dark-colored (opaque) debris in the soils. Microscopic technique can only detect relatively larger particles and cannot detect coal degradation products.

Thermal Oxidation Techniques

Thermal oxidation methods utilize the difference in heat resistance between recent SOM and coal particles under specific heating conditions (temperature and oxygen). Heating conditions are selected such that ROC is oxidized and GOC remains unaffected after sample treatment. By adjusting the combustion temperature and duration, ROC

can be oxidized and the remaining GOC can subsequently be quantified using elemental C analysis or C mass balance (Table 2). The efficiency of the thermal oxidation method varies with coal rank. If the RMS contain low-rank brown coal particles, the thermal oxidation method is not suitable, because some of the coal particles will be oxidized along with recent OM fractions, leading to underestimation of geogenic C concentration.

An example of thermal analysis technique that allows for the determination of physical properties of OM being oxidized (mass, temperature, and enthalpy), as the sample is subjected to a controlled temperature program.^[16,17] It involves heating a sample incrementally in a temperature controlled thermogravimetric analyzer, where changes in mass, temperature, and enthalpy of the sample are recorded during the oxidation process.^[16] Weight losses at temperatures >200, 250, and 395, 400–520, and above 600°C are generally attributed to loss of various forms of moisture, SOM oxidation, coal oxidation, and decomposition of carbonate minerals, respectively.^[16] Other variations of thermal oxidation approach are summarized in Table 2.

Thermal oxidation methods are based on the assumption that temperature conditions can be selected to thermally oxidize ROC fractions while GOC remains unoxidized. However, due to close associations and incorporation of coal particles within the macromolecular structure of SOM,

selective oxidation of recent SOM without oxidizing coal, especially those of lower rank remains a methodological challenge. Both recent SOM and coal comprise a continuum of OM compounds that overlap in their thermal resistance and chemistry.^[18] Less refractory C compounds of coal are oxidized along with ROC while highly refractory ROC remains unoxidized at the selected temperature. This overlap proves to be problematic, if low rank coal particles such as lignite are the sources of coal particles present in RMS. There is no single, universally accepted temperature limit for separating OM from the coal of different ranks. As a result, operationally defined temperature cutoffs are set empirically and probably will vary with the rank of coal particles present in RMS (Fig. 1).

Chemical Methods

Similar to thermal oxidation, these techniques rely on resistance of coal to chemical oxidizing agents for quantifying GOC in soils and sediments. For effective oxidation, soil or sediment samples must be deflocculated to expose occluded OM and also pretreated to remove carbonates and silicates prior to chemical treatment. The presence of carbonates and silicates interferes with the chemical oxidants and reduces the efficiency of SOM removal. In calcareous soils, e.g. carbonate coatings favor occlusion and thus physical protection of OM against chemical reaction. Carbonates are removed by dilute acid reaction, while hydrofluoric acid (HF; usually $\leq 10\%$ by volume) is commonly used for demineralization prior to chemical treatment (Table 2). The ROC, comprising humified SOM, is removed by chemical oxidants such as sodium hypochlorite (NaOCl), hydrogen peroxide (H_2O_2), potassium dichromate ($K_2Cr_2O_4$), sulfuric acid (H_2SO_4), nitric acid (HNO_3), disodium peroxodisulfate ($Na_2S_2O_8$), permanganate ($KMnO_4$), or ultraviolet light. Mild alkaline solutions (i.e., <0.5 M NaOH) have also been used to extract recalcitrant ROC.^[8] The efficiency of recent SOM removal from soils by chemical reagents is determined by reaction conditions (temperature, time of contact, and addition of catalysts) and RMS properties (soil pH, mineralogy, SOC concentration, and quality). Some of the chemical reagents are often used in combination and at selected elevated temperatures to increase the efficiency of recent OM removal. The remaining resistant OC fraction can be characterized using different methods, including elemental, microscopic, and spectroscopic methods.

Accurate quantification of GOC concentration in soils requires complete oxidation of recent SOM without removing GOC. Reaction with coal can cause underestimation, and incomplete removal of SOM could lead to overestimation of GOC concentration in RMS. Due to incomplete oxidation of recent OM, spectroscopic techniques such as ^{13}C nuclear magnetic resonance (NMR) analysis have the potential of distinguishing GOC and recalcitrant ROC, based on differences in functional

groups, and are used for quantifying the recalcitrant ROC resistant to oxidizing agents.

Both ROC and GOC consist of a continuum of organic material ranging in chemical oxidation resistance. Resistance of coal to chemical treatment is rank dependent, such that coals of lower rank (e.g., brown coal) are less resistant to chemical oxidation and may be oxidized along with recent OM. Similarly, highly recalcitrant recent OM is also resistant to chemical oxidants. Therefore, in RMS contaminated with low-rank coal particles, chemical methods could underestimate GOC. In addition, recalcitrant recent OM may remain after the chemical treatment and that could lead to the overestimation of GOC.

Ussiri and Lal^[8] developed chemi-thermo method, which combines chemical extraction, chemical oxidation, and heat oxidation at a controlled temperature for removing ROC and quantifying GOC using elemental C analyzer. It involves subjecting a soil sample to a series of chemical treatments and heat oxidation, to remove various fractions of ROC; then the remaining GOC is quantified by elemental C analysis.^[8]

Molecular Biomarkers

In the molecular approach, functional groups and/or classes of organic compounds associated with coal are first identified and, based on the concentration of these chemical markers, the coal content of the samples is determined.^[14,19] Oxidative degradation of coal produces benzenepolycarboxylic acids (BPCA), resulting from conversion of polycyclic C or substituted aromatic C. The oxidative degradation of coal can be achieved through the use of concentrated acid digestion. GOC is estimated based on reference standards by quantifying BPCA using gas chromatography.^[19]

Spectroscopic Techniques

Spectroscopic methods use a spectroscopic analysis to quantify GOC based on relative abundance of specific functional groups characteristic to coal, as revealed by spectral analysis, generally after oxidative removal of non-coal SOM by a chemical pretreatment.^[2,20] Two methods commonly applicable to GOC determination are infrared (IR) and NMR spectroscopies. The IR spectroscopy is based on the specific response of organic groups to IR light in different regions of the IR spectrum, and NMR spectroscopy identifies spectra regions characteristic of functional groups in coal and estimates coal concentration in the RMS samples based on the relative strength of the identified bands. Fourier transformed IR spectroscopy and diffuse reflectance Fourier transform IR spectroscopy analyses are commonly used to examine complex organic substances in the solid state. Interpretation of spectra information and quantification of different OC sources is achieved through

Multivariate data analysis and predictive modeling using partial least squares.^[2]

Solid state ^{13}C CPMAS NMR technique determines structural information of SOM based on distribution of ^{13}C content of functional groups and infers the relative contribution of different C sources in the sample based on relative functional groups distribution. The peak intensity of NMR spectroscopy is approximately proportional to the fraction of C represented by the structural fingerprint assigned to the abscissa of the spectrum (chemical shift). The structural composition of coal differs considerably from that of recent SOM, and their functional group differences can be utilized to quantify the contribution of GOC in RMS. However, for mixtures of macromolecules such as those encountered in RMS, the NMR spectra becomes a complex assemblage of overlapping peaks where only the average structural entities can be discerned.^[3] The distinct dominance of aliphatic (alkyl C) and aromatic C in brown coal was used as a fingerprint for brown coal presence in RMS and the signal intensity ratio (A) as an indicator of the presence of lignite in soils, where A is computed as

$$A = \frac{(\text{alkyl C} + \text{aromatic C})}{(\text{O} - \text{alkyl C} + \text{carboxylic C})} \quad (6)$$

The value of A is typically between 0.8 and 1.2 for non-contaminated soils and between 1.6 and 2.2 for soils contaminated with brown coal.^[20] Thus, the value of A increases as lignite contribution increases.

^{13}C Stable Isotope Approach

The ^{13}C isotope technique is also a potential tool for estimating GOC and ROC contribution in RMS. The GOC is normally enriched in ^{13}C compared to recent OM probably due to selective preservation of OM during coalification and microbial discrimination against the heavier isotope. Microbial discrimination against ^{13}C leads to relative enrichment of ^{13}C in plant materials resistant to microbial mineralization.^[21] Plant remains underwent extensive transformation and decomposition prior to geological coalification process,^[22] resulting in relatively narrow range of $\delta^{13}\text{C}$ stable isotope composition of coal (-22‰ and -27‰) compared to that of C_3 recent OM (-22‰ to -35‰).^[23] The $\delta^{13}\text{C}$ stable isotope ranges are very stable and does not change significantly with rank or maturity of coal.^[22] A simple two-compartment model can be used to estimate the proportions of coal C and ROC in RMS. However, this approach has not been used for quantifying GOC in RMS where recent OM can be dominantly C_3 or C_4 plant sources.

CONCLUSION

Minesoils formed from coal mining can play a significant role in C sequestration climate change mitigation. However, accurate potential of these soils can only be estimated

if GOC contamination can be quantified and adjusting OC pools for coal contamination. To obtain accurate estimations of SOC pool and sequestration rates, cost effective, reliable, and standardized methods are needed for routine quantification of the relative contribution of coal C to SOC in RMS. Compared to recent OM, coal is highly aromatic and hydrophobic in nature, a characteristic that make coal-derived OC relatively inert to various chemical oxidants and highly resistant to thermal at lower temperatures, depending on coal rank. The differences in chemical and thermal oxidation properties and also functional group composition among coal and recent OM form the basis for the various techniques for estimating GOC and ROC in RMS. The major limitation of thermal and chemical oxidation methods is their inability to remove highly recalcitrant ROC and also oxidation of lower rank coals together with ROC. The ^{13}C NMR spectroscopy method generally differentiates between ROC and GOC based on relative abundance of specific functional groups characteristic of coal or recent SOM. While structural and compositional differences between GOC and ROC are fairly well understood, overlaps exist in the spectral signature of these C sources, which makes it a challenge to quantitatively estimate GOC using spectroscopic methods. Radiocarbon analysis is one of the most reliable methods for quantitative measurement of GOC in RMS and closest to reference method, but high analytical costs, requirement of specialized analytical instrument, as well as, specialized expertise preclude its adoption for routine soil analytical laboratory. Although various methods have been employed in quantifying GOC in RMS, there are no standard methods for that purpose. Thermal and chemical methods have several shortcomings including incomplete oxidation of ROC and variable efficiency of the methods depending on reaction conditions and rank of the coal particles in RMS. Nonetheless, with further research, they offer a great potential for methods that could be widely adopted for GOC determination in RMS. Systematic inter-laboratory comparison of analysis of well-defined standard materials could be a crucial step toward that goal.

ACKNOWLEDGMENT

This work was supported by the Biomass Research Initiative Program (2012-10008-20302) USDA National Institute of Food and Agriculture.

REFERENCES

1. Loeppert, R.H.; Suarez, D.L. Carbonate and gypsum. In *Methods of Soil Analysis: Part 3. Chemical Methods*; Sparks, D.L., Ed.; SSSA, ASA Inc.: Madison, 1996; 437–474.
2. Rumpel, C.; Janik, L.J.; Skjemstad, J.O.; Kogel-Knabner, I. Quantification of carbon derived from lignite in soils using

- mid-infrared spectroscopy and partial least squares. *Org. Geochem.* **2001**, *32* (6), 831–839.
3. Rumpel, C.; Skjemstad, J.O.; Knicker, H.; Kogel-Knabner, I.; Huttel, R.F. Techniques for the differentiation of carbon types present in lignite-rich mine soils. *Org. Geochem.* **2000**, *31* (6), 543–551.
 4. Rumpel, C.; Kogel-Knabner, I. Microbial use of lignite compared to recent plant litter as substrates in reclaimed coal mine soils. *Soil Biol. Biochem.* **2004**, *36* (1), 67–75. DOI: 10.1016/j.soilbio.2003.08.0205.
 5. Rumpel, C.; Knicker, H.; Kogel-Knabner, I.; Skjemstad, J.O.; Huttel, R.F. Types and chemical composition of organic matter in reforested lignite-rich mine soils. *Geoderma* **1998**, *86* (1–2), 123–142.
 6. Fettweis, U.; Bens, O.; Huttel, R.F. Accumulation and properties of soil organic carbon at reclaimed sites in the Lusatian lignite mining district afforested with *Pinus* sp. *Geoderma* **2005**, *129* (1–2), 81–91. DOI:10.1016/j.geoderma.2004.12.034.
 7. Rumpel, C.; Kogel-Knabner, I.; Huttel, R.F. Organic matter composition and degree of humification in lignite-rich mine soils under a chronosequence of pine. *Plant Soil* **1999**, *213*, 161–168.
 8. Ussiri, D.A.N.; Lal, R. Method for determining coal carbon in the reclaimed minesoils contaminated with coal. *Soil Sci. Soc. Am. J.* **2008**, *72* (1), 231–237. DOI:10.2136/sssaj2007.0047.
 9. Manning, M.R.; Lowe, D.C.; Melhuish, M.H.; Sparks, R.J.; Wallace, G.; Brenminkmijer, C.A.M.; McGill, R.C. The use of radiocarbon measurements in atmospheric studies. *Radiocarbon* **1990**, *32*, 37–245.
 10. Rumpel, C.; Balesdent, J.; Grootes, P.; Weber, E.; Kogel-Knabner, I. Quantification of lignite- and vegetation-derived soil carbon using C-14 activity measurements in a forested chronosequence. *Geoderma* **2003**, *112* (1–2), 155–166.
 11. Morgenroth, G.; Kretschmer, W.; Scharf, A.; Uhl, T.; Fettweis, U.; Bens, O.; Huttel, R.F. ^{14}C measurement of soil in post-mining landscapes. *Nucl. Instr. Meth. B* **2004**, *223*, 568–572. DOI:10.1016/j.nimb.2004.04.105.
 12. Taylor, R.A. *Radiocarbon Dating: Anarcheological Perspective*; Academic Press: Orlando, 1987.
 13. Ussiri, D.A.N.; Jacinthe, P.-A.; Lal, R. Methods for determination of coal carbon in reclaimed minesoils: A review. *Geoderma* **2014**, *214*, 155–167. DOI:10.1016/j.geoderma.2013.09.015.
 14. Brodowski, S.; Amelung, W.; Haumaier, L.; Abetz, C.; Zech, W. Morphological and chemical properties of black carbon in physical soil fractions as revealed by scanning electron microscopy and energy-dispersive X-ray spectroscopy. *Geoderma* **2005**, *128*, 116–129. DOI:10.1016/j.geoderma.2004.12.019.
 15. Fernandes, M.B.; Skjemstad, J.O.; Johnson, B.B.; Wells, J.D.; Brooks, P. Characterization of carbonaceous combustion residues. I. Morphological, elemental and spectroscopic features. *Chemosphere* **2003**, *51* (8), 785–795. DOI: 10.1016/s0045-6535(03)00098-5.
 16. Maharaj, S.; Barton, C.D.; Karathanasis, T.A.D.; Rowe, H.D.; Rimmer, S.M. Distinguishing “new” from “old” organic carbon in reclaimed coal mine sites using thermogravimetry: II. Field validation. *Soil Sci.* **2007**, *172* (4), 302–312. DOI:10.1097/SS.0b013e3180314702.
 17. Maharaj, S.; Barton, C.D.; Karathanasis, T.A.D.; Rowe, H.D.; Rimmer, S.M. Distinguishing “new” from “old” organic carbon on reclaimed coal mine sites using thermogravimetry: I. Method development. *Soil Sci.* **2007**, *172* (4), 288–301. DOI:10.1097/SS.0b13e31803146e8.
 18. Rumpel, C.; Kogel-Knabner, I. The role of lignite in the carbon cycle of lignite-containing mine soils: Evidence from carbon mineralisation and humic acid extractions. *Org. Geochem.* **2002**, *33* (3), 393–399. DOI:10.1016/s0146-6380(01)00169-3.
 19. Glaser, B.; Haumaier, L.; Guggenberger, G.; Zech, W. Black carbon in soils: The use of benzenecarboxylic acids as specific markers. *Org. Geochem.* **1998**, *29* (4), 811–819.
 20. Schmidt, M.W.I.; Knicker, H.; Hatcher, P.G.; Kogel-Knabner, I. Impact of brown coal dust on the organic matter in particle-size fractions of a Mollisol. *Org. Geochem.* **1996**, *25* (1–2), 29–39.
 21. Hobbie, E.A.; Macko, S.A.; Shugart, H.H. Insights into nitrogen and carbon dynamics of ectomycorrhizal and saprotrophic fungi from isotopic evidence. *Oecologia* **1999**, *118* (3), 353–360. DOI:10.1007/s004420050736.
 22. Whiticar, M.J. Stable isotope geochemistry of coals, humic kerogens and related natural gases. *Int. J. Coal Geol.* **1996**, *32*, 191–215.
 23. Redding, C.; Schoell, M.; Monin, J.C.; Durand, B. Hydrogen and carbon isotopes in coals and kerogen. In *Advances in Organic Geochemistry 1979*; Douglas, A.G., Maxwell, J.R., Eds.; Pergamon: Oxford, 1980; 711–723.

Mine Soils: Miscanthus Plantations

Jose G. Guzman

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

In the United States, surface mining has created over 1.82 Mha of degraded land, mostly in the Appalachian region. With demand for bioenergy derived from renewable biomass resources increasing and a finite availability of land and water resources, interest for bioenergy crop production with low input demands on marginal lands (i.e., miscanthus) is subsequently increasing. However, physical (available water content, high compaction), chemical (pH, lack of nutrients, plant toxicities), and biological (low organic carbon content) soil properties have to be appropriately amended for miscanthus to have a large impact as a biomass energy crop in minesoils in the Appalachian region.

INTRODUCTION

Termed as a second-generation biofuel crop, the use of miscanthus (*Miscanthus giganteus*) is a positive step toward a more sustainable bioenergy resource compared to first-generation biofuels that are derived from plant sugars, starches, and oils. This is in large part due to miscanthus having a perennial life cycle, low maintenance and nutrient requirements, and high biomass yield potential.^[1] Consequently, the establishment of miscanthus compared to annual crops such as maize (*Zea mays* L.) over the long term can typically improve soil nutrient recycling, soil carbon (C) sequestration, and controlling soil erosion, especially in degraded lands.^[2,3]

However, there are concerns about bioenergy crops competing for finite land and water resources with annual grain crop production in the United States.^[4] One approach to addressing these concerns is to reserve land that is rich in soil fertility with no environmental restrictions for food and feed production, while marginal lands are primarily used for establishment of bioenergy crops that have low input demands and are tolerable to environmental stresses. This entry provides an overview of the extent and types of minesoils and the constraints and limitations for miscanthus production on those marginal lands.

DEFINING MINESOILS

Extent of Degraded Land

Surface mining for coal and other minerals has created ~1.82 Mha of degraded land across the United States, of which only 65,000 ha (4%) is reclaimed and bond released (Fig. 1).^[5] Much of these degraded lands have gone

through reclamation practices that reestablish native vegetation or grass and legume mixtures, which promote soil formation and stabilization during the first 5 years in humid and subhumid regions in the United States. These soils are termed minesoils, because they are human-made and differ from other disturbed surface mined soils that were abandoned and little to no attempt of reclamation and revegetation was made.^[6] There have been at least 34 minesoils that have been identified as individual soil series, mostly in the Appalachian region in land mined for coal.^[6]

Reclamation of Mined Land

Surface mining for minerals and metals results in the destruction of the existing vegetation and soil profile, as well as severe disruption of soil microbe populations and nutrient cycling. In 1977, the U.S. government enacted the Surface Mining Control and Reclamation Act (SMCRA), which requires grading, of the top soil replacement, and reestablishment of vegetation of the disturbed land to conditions prior to mining.^[7] These regulations were relaxed to establishing landscape conditions and vegetation, which minimized soil erosion, acid mine drainage, and pollution of natural waters.

During surface mining, the vegetation and top soil are first scalped and stockpiled for later use. Then, explosives are used to remove the overburden material (subsoil and rock fragments) from the attended extraction material (typically coal in the Appalachian region), creating a mix of spoil. Once the attended minerals or metals have been extracted, the pit that has been formed is backfilled with the spoil. Grading of spoil is done to decrease the steepness of slopes and reduce soil surface runoff and slippage of spoil piles. However, the heavy equipment used during the

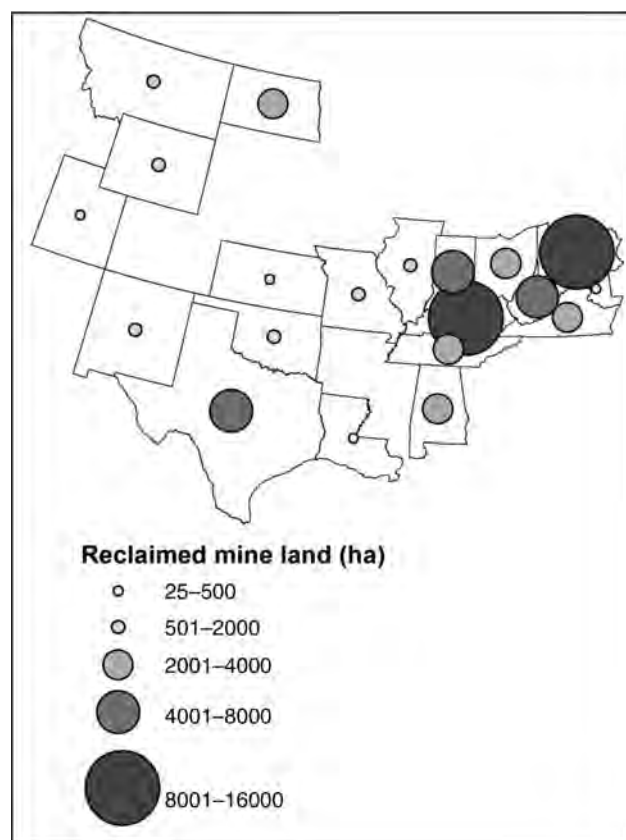


Fig. 1 Amount of reclaimed mine land in the United States.
Source: From Office of Surface Mining.^[5]

grading causes a high level of compaction. Finally, the original stockpiled soil is replaced back onto the surface where appropriate soil amendments and seeding mixtures are applied for revegetation.

Given that minesoiils are human-made, landform decisions can be made based on post-mining land uses that would be more valuable for agriculture or forestry. In the Appalachian region, hilly terrain is common, and although surface mine reclamation generally requires that lands be restored to the approximate original contour, it would be advantageous to construct land that is flat for row crop production.^[8] Ideal grading of minesoiils for row crop production should have a 2–3% gently sloping surface in humid regions to avoid depression areas that cause water logging, while still minimizing soil erosion.^[8] Minesoiils with slopes greater than 15% are typically not suitable for miscanthus production, as full canopy establishment takes over 2 years, leaving the soil surface vulnerable to erosion without some form of cover vegetation.

Top soil replacement with greater silt and clay contents and higher nutrient capacity, biological activity, and water retention than overburden spoil is a critical step in providing a suitable growth media for revegetation. The 1977 SMCRA required that at least 15–20 cm of the top soil

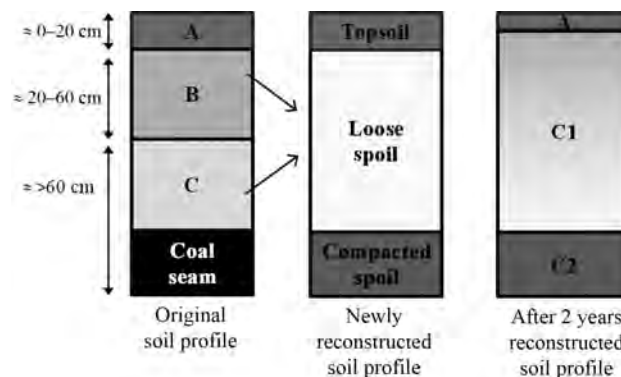


Fig. 2 Soil profiles starting from non-disturbed to development of new soil horizons in minesoil over time.

be removed prior to mining, stockpiled under appropriate conditions (aerobic conditions), and replaced back on the surface. However, in sloping regions such as the Appalachian, the top soil is often too shallow, rocky, infertile, and too acidic to use for revegetation. Under these circumstances, a substitute material such as overburden material mixed with sewage sludges, residue mulch, manure, sawdust, wood chips, and papermill sludges can be used in the place of original top soil.^[9]

General Properties of Minesoiils

Minesoiils are pedogenically young soils developing from replacement of top soil and mine spoil, typically expressing weak soil structure that contributes to high compaction and poor hydraulic conductivity (Fig. 2). Minesoiils generally have an A–C or A–AC–C horizon soil profile, the formation of a shallow A horizon often takes as little as a year, characterized by spoil loosening and aggregation formation.^[6] After 5 years, A horizons have been reported to range from 5 to 20 cm deep in the Appalachian region (Table 1), with weak granular or subangular blocky structure and enriched with organic matter.^[6]

Table 1 Range and mean of soil physical characteristics of minesoiils in the Appalachian region.

Variable	Range	Mean
Coarse fragments >2 mm (%)	4–71.5	26.5
Clay (%)	13–51.7	31.7
Bulk density (g cm ⁻³)	1.32–1.70	1.53
Cone penetrometer resistance (MPa)	0.9–3.7	1.9
Available water capacity (cm ³ cm ⁻³)	3–33	13.5
A horizon depth (cm)	5–20	12.6

Source: From Guzman & Lal.^[12]

CONSTRAINTS AND LIMITATIONS OF ESTABLISHING MISCANTHUS IN MINE SOILS

Miscanthus Production in Minesoils

There is only sparse, readily available published data on bioenergy crop production in marginal lands, especially in minesoils, which is the focus of this entry. During the first 2 years of miscanthus establishment in Ohio, biomass yields ranged from 0.1 to 4 Mg ha⁻¹,^[10] although harvesting is not recommended until at least the 3rd year. This would allow for sufficient time of the rhizomes to increase their nutrient reserves and increase tiller production, so as not to reduce population stands. Once fully established, miscanthus yields ranged from 2 to 18.7 Mg ha⁻¹ in Ohio and West Virginia, varying due to stand establishment, which is significantly affected by physical and chemical properties of minesoils and environmental stresses such as flooding and drought.^[10,11]

Physical Soil Properties

Minesoils in the Appalachian region typically have low water-holding capacity and high soil compaction due to their inherently high coarse texture and rock fragments content ranging from 4% to 72% (mean 26%; Table 1) and are the primary constraints for bioenergy crop production.^[12] Minesoils with greater than 40% coarse fragments (>2 mm), which in general are those with no or little top soil with silt and clay applied, have a water-holding capacity ranging from 3% to 15% resulting in limiting plant growth.^[13] Additionally, in regions that receive less than 70 cm of rainfall, yields could fall lower than maize and switchgrass (*Panicum virgatum*), due to miscanthus having a larger water uptake requirements.^[11,14]



Fig. 3 Soil compaction effect on miscanthus growth in a mine-soil near Zanesville, Ohio.

Severe soil compaction also reduces water-holding capacity and pore space, restricting root growth in drought conditions, especially in minesoils with less than 60 cm of effective rooting depth (Fig. 3).^[12] For many plants, a soil bulk density greater than 1.7 g cm⁻³ or cone penetrometer resistance greater than 2 MPa in minesoils (Table 1) is very restrictive for root growth. Additionally, in excessive wet settings, soil compaction can also cause perched water tables resulting in anaerobic conditions for root growth. Furthermore, overwintering survival during the first year of establishment largely depends on soil and rhizome moisture content.^[15] In Ohio, overwinter survival rates of miscanthus in minesoils ranged from 40% to 70% in non-compacted soil.^[10] However, in the same study, soils that were water logged due to compaction had overwinter survival rates only from 20% to 55%.

Chemical Soil Properties

Chemical soil properties of minesoils important to growth of miscanthus include pH, concentration of soluble salts, nutrient availability, organic (C), and metal toxicity. In the Appalachian region, pH in minesoils ranged from 6.5 to 7.8 in calcareous soils and 4 to 6.2 in non-calcareous spoils.^[12] The ideal soil pH range for most plants is 6.0 to 7.5, although miscanthus can tolerate soils with pH ranging from 4.9 to 7.6.^[12] Potential uptake of toxic levels of metals such as copper, lead, nickel, zinc, and aluminum is primarily only a concern in acidic soils. Other potential toxicities from additions of industrial wastes and soil amendments include mercury, arsenic, cadmium, and selenium.^[16]

Nutrient cycling, specifically nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), and soil organic carbon (SOC), is severely disrupted during mining, due to declines in soil microbial activity and recycling of nutrients between plants and microbes. In minesoils, soil fertility varies from low to ideal conditions in well-established minesoils, although young minesoils frequently have very low nutrient concentrations in the soil.^[12] The SOC concentrations in minesoils in the Appalachian region range from 5 to 44 g kg⁻¹ (mean 15 g kg⁻¹). Total soil N concentrations in minesoils in the Appalachian region range from 0.4 to 1.9 g kg⁻¹ (mean of 1.2 g kg⁻¹), within range of what is considered adequate for crop growth (1–1.5 g kg⁻¹). However, N uptake demand is much higher in miscanthus compared to annual row crops after the 3rd year of establishment, and N fertilization may be needed to maintain or increase productivity in minesoils. Total soil P concentrations in minesoils in the Appalachian region range from 27 to 129 mg kg⁻¹ (mean of 56 mg kg⁻¹), well below adequate soil levels for crop growth at 100–150 mg kg⁻¹. This is of particular concern as miscanthus nutrient requirements for P (2–5 kg Mg⁻¹ of dry matter), K (0.8–1.2 kg Mg⁻¹ of dry matter), and Ca (0.3–1.1 kg Mg⁻¹ of dry matter) are high.^[17] Although miscanthus has low demand for inputs due to its nutrient cycling between rhizome and

aboveground biomass, attention must be given to how much nutrients are removed when harvesting biomass and, subsequently, how much nutrients are needed to be added back to maintain long-term productivity.^[12] This is especially true in minesoils, where these soils typically do not have high nutrient reserves, due to limitations in nutrient capacity from high content of coarse fragments and low organic matter in some cases.^[12]

CONCLUSION

Minesoils that are under cool season forage grasses may be underutilized when future economic potential of bioenergy crop production such as *miscanthus* is considered. Once physical (available water content, high compaction), chemical (pH, lack of nutrients, plant toxicities), and biological (low SOC) soil properties have been appropriately amended, *miscanthus* has a large potential as biomass energy crop in minesoils in the Appalachian region. However, there have been few field studies that have examined fertilization requirements for bioenergy crop production in minesoils to determine optimum rates while minimizing negative environmental impacts. Further research is needed before *miscanthus* can be implemented at a commercial scale in minesoils.

ACKNOWLEDGMENTS

This work was supported by the Biomass Research and Development Initiative Program (2012-10008-20302) from USDA National Institute of Food and Agriculture.

REFERENCES

1. Heaton, E.A.; Dohleman, F.G.; Long, S.P. Meeting US bio-fuel goals with less land: The potential of *miscanthus*. *Glob. Change Biol.* **2008**, *14* (9), 2000–2014.
2. Shrestha, R.K.; Lal, R. Soil carbon and nitrogen in 28-year-old land uses in reclaimed coal mine soils of Ohio. *J. Environ. Qual.* **2007**, *36* (6), 1775–1783.
3. Keene, T.; Skousen, J. Switchgrass production potential on reclaimed surface mines in West Virginia. In *Proceedings of the 2010 National Meeting, American Society for Mining and Reclamation*, Jun 5–11, 2010.
4. Qin, Z.; Zhuang, Q.; Chen, M. Impacts of land use change due to biofuel crops on carbon balance, bioenergy production, and agricultural yield, in the conterminous United States. *Glob. Change Biol. Bioenerg.* **2012**, *4* (3), 277–288.
5. Office of Surface Mining. *Annual Report 2011*; Office of Surface Mining, Department of Interior: Washington, 2012.
6. Sencindiver, J.C.; Ammons, J.T.; Barnhisel, R.I.; Darmody, R.G.; Daniels, W.L. Minesoil genesis and classification. In *Reclamation of Drastically Disturbed Lands*, ASA Monograph No. 41; Barnhisel, R., Darmody, R., Daniels, W., Eds.; American Society of Agronomy; Madison, 2000; 595–614.
7. Public Law 95-87. Surface Mining Control and Reclamation Act, U.S. Code Vol. 30, Sec 1265, 1977.
8. Barnhisel, R.I.; Hower, J.M. Coal surface mine reclamation in the eastern United States: The revegetation of disturbed lands to hayland/pasture or cropland. *Adv. Agron.* **1997**, *61*, 233–275.
9. Haering, K.C.; Daniels, W.L.; Feagley, S.E.; Barnhisel, R.I.; Darmody, R.G. Reclaiming mined lands with biosolids, manures, and papermill sludges. In *Reclamation of Drastically Disturbed Lands*, ASA Monograph No. 41; Barnhisel, R., Darmody, R., Daniels, W., Eds.; American Society of Agronomy; Madison, 2000; 615–644.
10. Guzman, J.G.; Ussiri, D.; Lal, R. Impact of anaerobic digestate effluent on bioenergy crop production and soil quality in a minesoil from Ohio. **2014**.
11. Skousen, J.; Keene, T.; Marra, M.; Gutta, B. Reclamation of mined land with switchgrass, *Miscanthus*, and *Arundo* for biofuel production. *J. Am. Soc. Mining Recl.* **2013**, *2*, 177–191.
12. Guzman, J.G.; Lal, R. *Miscanthus* and switchgrass feedstock potential for bioenergy and carbon sequestration on minesoils. *Biofuels* **2014**, *5* (3), 313–329.
13. Coile, T.S. Moisture content of small stone in soil. *Soil Sci.* **1952**, *75*, 203–207.
14. Wu, Y.; Liu, S. Impacts of biofuels production alternatives on water quantity and quality in the Iowa River Basin. *Biomass Bioenerg.* **2012**, *36*, 182–191.
15. Clifton-Brown, J.C.; Lewandowski, I. Overwintering problems of newly established *miscanthus* plantations can be overcome by identifying genotypes with improved rhizome cold tolerance. *New Phytol.* **2000**, *148*, 287–294.
16. Gaskin, J.W.; Brobst, R.B.; Miller, W.P.; Tollner, E.W. Long-term biosolids application effects on metal concentrations in soil and bermudagrass forage. *J. Environ. Qual.* **2003**, *32* (1), 146–152.
17. Lewandowski, I.; Scurlock, J.M.O.; Lindvall, E.; Christou, M. The development and current status of perennial rhizomatous grasses as energy crops in the US and Europe. *Biomass Bioenerg.* **2003**, *25*, 335–361.

Mine Soils: Open Cut Mine Rehabilitation

Douglas J. Dollhopf

Department of Land Resources and Environmental Sciences, Montana State University, Bozeman, Montana, U.S.A.

Abstract

Land rehabilitation sciences at open pit mines has developed at an exponential rate. Grading the mined landscape to the approximate original contour, use of adequate amounts of—and quality of—coversoil and development of *in situ* soil treatments at sites where coversoil resources were absent have all provided means to establish a diverse plant community.

INTRODUCTION

Mining directly disturbs approximately 240,000 km² of the earth's surface.^[1] Surface mining methods may be classified as: 1) open pit mining; 2) strip mining; 3) dredging; and 4) hydraulic mining.^[2] Open pit mining includes quarries used to produce limestone, sandstone, marble, and granite; pits used to produce sand, gravel, and bentonite; and large excavations used to produce talc, copper, gold, iron, silver, and other metals. This mining method is distinguished by having one large pit or numerous small pits across the landscape. State or federal governments establish regulations that require the operator to restore the land to an approved land use(s) that is equal to or better than the premine land use. The approved postmine land use(s) is instrumental in determining the final graded topography and vegetation community.

LANDSCAPE REGRADING

Reclamation of an open pit mine must address the open pit itself, waste rock removed from the pit to gain access to the ore, tailing impoundments, and access roads (Fig. 1). For open pit mines, complete backfilling of the pit is not usually economically feasible. Because the pit remaining after mineral extraction may be hundreds of feet deep, the cost of moving waste rock back into the depression is prohibitive. Surface and groundwater may flow into the depression after mining is terminated and an impoundment develops. Pit slopes should be reduced to the limits required for safe access for humans, livestock, and wildlife. Pit slopes should not be permitted at gradients that jeopardize the success of postmine reclamation.

COVERSOIL RESOURCES

Successful plant community establishment on an open pit mine site is a direct function of the coversoil quality

applied. Coversoil suitability criteria vary in the United States from state to state but are similar to those presented in Table 1. The coversoil resource emanates from two procedures: 1) salvaging the natural soil resource from the area to be disturbed; and 2) mining unconsolidated geologic stratum when the soil resource is absent. Under U.S. federal and state mine regulatory programs, all portions of the soil resource shall be salvaged in a project area if it meets physicochemical suitability criteria. Soil materials that do not meet these suitability criteria are generally not used because they may impair plant establishment and growth. Once the soil resource is salvaged, it should be directly hauled to an area that has been backfilled and is ready to receive coversoil. Direct hauled soil contains a viable seed bank, mycorrhizal associations, organic matter, and nutrients that aid in plant establishment. Conversely, if the soil resource is stockpiled, these resources will deteriorate with time.

Prior to implementation of mine land reclamation regulations during the 1970s and 1980s, open pit U.S. mine operations often did not institute land reclamation. The soil resource was not salvaged, the disturbed landscape was not graded to the approximate original contour, and the site was not seeded. Consequently, the soil resource was lost. These lands are being coversoiled using unconsolidated geologic stratum. Valley fill areas adjacent to the disturbed landscape often contain alluvial materials beneath the soil resource that meet all soil suitability criteria (Table 1), except organic matter is absent. This alluvial stratigraphy may be 10–30 m thick or more. Earth moving equipment is used to stockpile the soil resource, separating true topsoil (A horizon) from subsoil (B and C horizons), to enable excavation of deeper alluvial material. Following placement of this alluvial coversoil on the disturbed landscape, the pit created is contoured and the stockpiled soil resource is replaced. Lands receiving coversoil emanating from a geologic stratum should have organic matter applied to



Fig. 1 Large open pit mine in the United States with water impounded in bottom.

expedite establishment of nutrient cycles in the plant root zone. Cattle manure and municipal compost are two sources of organic matter used in land reclamation projects.

COVERSOIL THICKNESS REQUIREMENTS

The thickness of coversoil required for maximum vegetation performance is dependent on several factors including the 1) vegetation species present; 2) local climate; 3) coversoil quality; and 4) physicochemical quality of the substrate beneath the coversoil. The maximum rooting depth of many rangeland ecosystem plant species is less than 45 cm and generally does not exceed 100 cm.^[3] Investigators found that a minimum of 40 cm to a maximum of 150 cm of coversoil is needed for optimum plant growth.^[4–8] This wide variation of findings is a function of site specific conditions identified before. The Barth^[4] investigation is representative of most research. This investigator found that generic spoil, defined as non-alkaline and non-saline loams with sodium adsorption ratio (SAR) below seven,

Table 1 Coversoil suitability criteria.

Soil parameter	Suitability criteria
pH	>6.5 and <8.0 standard units
Electrical conductivity	<4.0 dS/cm
Exchangeable sodium percentage	<12%
USDA textural classes (12 types)	All suitable except clay, loamy sand, sand
Rock content (particles >2 mm diameter)	
Slope gradients <25%	<35%, weight basis
Slope gradients >25%	>35% and <60%, weight basis
As, Cd, Cu, Pb, Zn, other metals	Near back ground levels, i.e., no enrichment
Organic matter	>0.5%, weight basis

showed an increase in plant production up to a coversoil depth of approximately 50 cm. Sodic spoil, or material with SAR of 26 or higher, required a minimum of 70 cm of coversoil to reach maximum cool season grass production. Coversoiled acid spoils, defined as spoils exhibiting pH values between 3.6 and 4.3, revealed increased grass production up to the maximum coversoil depth used in this study (152 cm).

IN SITU SOIL RECLAMATION

In situ soil reclamation at open cut mine sites means no coversoil resource is used. Chemical and/or organic amendments are applied to the graded spoil, tailings, or waste rock landscape to enable plant establishment and growth.

In situ Sodic Minesoil Remediation

Thousands of hectares of land in Wyoming and Montana were open pit mined for the mineral bentonite prior to passage of state regulations requiring land reclamation. The absence of a soil resource to reclaim these lands means the plant ecosystem must be established on graded spoils. Spoils are overburden materials cast aside to access the underlying mineral, bentonite. These overburden materials have a sodic condition [exchangeable sodium percentage (ESP) 20–40], clay texture (60% clay) dominated by swelling clay minerals such as smectite, and preclude plant establishment and growth. *In situ* soil remediation requires a twofold approach: 1) permanently reduce ESP to less than 10 within the 0–30 cm soil profile and 2) provide an organic amendment to immediately prevent soil crust development and increase water infiltration. Crust development in these clayey-sodic soil systems is precluded with applications of wood chips from saw mill waste or manure.^[9] Applications of gypsum, calcium chloride, magnesium chloride, and sulfuric acid to these sodic soils have been shown to be effective at reducing the ESP to acceptable levels in minesoils and provide a root zone suitable for plant growth.^[10]

In situ Acid Minesoil Remediation

Open pit mines associated with gold, silver, and metal extraction from sulfide ore bodies frequently produce tailings impoundments and waste rock areas that are acidic (pH 2.5–5.5) and fail to support plant growth. Soil acidity is formed when sulfide minerals, such as pyrite, are exposed to oxidizing conditions in the presence of water. In the absence of a coversoil resource, tailings and waste rock material are treated with alkaline amendments to permanently neutralize the soil acidity. Soil analysis of the acid base account and active acidity enables

determination of the calcium carbonate requirement.^[11] Amendments used include calcium carbonate, calcium hydroxide, and calcium oxide. *In situ* treatment with amendments should be at least to the 45 cm soil depth and special engineered plowing equipment is required to attain this depth of incorporation.^[12]

ACID MINE DRAINAGE

Upon exposure of open pit mine wastes to water and oxygen, sulfide minerals oxidize to form acidic, sulfate rich drainage.^[13] Iron-oxidizing bacteria, e.g., *Thiobacillus ferrooxidans*, expedite the acid producing reaction rate up to a million times. Metal composition (aluminum, copper, iron, manganese, lead, zinc, and others) and concentrations in acid mine drainage (AMD) depend on the type and quantity of sulfide minerals present. Approaches to prevent or treat AMD include: 1) removal of air and water from the sulfide body that is disturbed; 2) wetland construction to precipitate contaminants with oxidation ponds and anaerobic-microbial reactions in plant root zones; 3) underground anoxic- and aboveground open-limestone drains to raise water pH; and 4) treatment of drainage with neutralizing chemicals such as hydrated lime, quicklime, soda ash, caustic soda, and ammonia.^[14]

STEEP SLOPE RECLAMATION

Open pit mines are frequently located in mountainous terrain and reconstructed slopes may have gradients as steep as 50%. Sediment yields tend to increase linearly with slope gradient. There is an inverse relationship between soil loss on slopes and rock cover.^[15] High rock content in soils increases infiltration rate and surface roughness decreasing runoff and soil loss. As erosion occurs on rocky soil, rock cover increases as the coarse fragments below the surface are exposed. This armoring of the soil surface can help reduce soil erosion. It has been reported that plant growth may be impaired when soil rock content exceeds 35%.^[16] High rock contents of 35–60% applied on steep slopes (Table 1) may impair plant growth but are considered a best management practice to facilitate plant establishment and slope stability.

Construction of shallow pits across steep slopes is commonly practiced to increase water storage on the slope and minimize runoff. Pitting techniques using equipment referred to as Dammer-Diker, gouger, and dozer basin blade have been shown to be effective in controlling sediment loss.^[17]

CONCLUSION

Since the 1970s, land rehabilitation sciences at open pit mines developed at an exponential rate. Grading the mined

landscape to the approximate original contour, use of adequate amounts of—and quality of—coversoil and development of *in situ* soil treatments at sites where coversoil resources were absent have all provided means to establish a diverse plant community. Techniques have been developed to revegetate and stabilize slopes approaching a 50% gradient, which is an asset for mines located in mountainous terrain. Methods to control and treat AMD from open pit mines continue to improve, but this impact to surface water resources is unresolved. Further research is required to develop better AMD treatment methods.

REFERENCES

1. Solomons, W. Mining impacts worldwide. In *Proceedings of the International Hydrology Program*; UNESCO, Asian Institute of Technology: Bangkok, 1988; 7–9.
2. Paone, J.; Struthers, P.; Johnson, W. Extent of disturbed lands and major reclamation problems in the United States. In *Reclamation of Drastically Disturbed Lands*; Schaller, F.W., Sutton, P., Eds.; American Society of Agronomy: Madison, 1978; 11–22.
3. Wyatt, J.W.; Dollhopf, D.J.; Schafer, W.M. Root distribution in 1- to 48-year old strip-mine spoils in southeastern Montana. *J. Range Mgmt.* **1980**, *33*, 101–104.
4. Barth, R.C. *Soil-Depth Requirements to Reestablish Perennial Grasses on Surface-Mined Areas in the Northern Great Plains*; No. 1, Publication 0192-6179/84/2701-0001; Mineral & Energy Resources Research Institute, Colorado School of Mines: Golden, 1984; Vol. 27.
5. Doll, E.C.; Merrill, S.D.; Halvorson, G.A. *Soil Replacement for Reclamation of Strip Mined Lands in North Dakota*; Agricultural Experiment Station Bulletin 514; North Dakota State University: Fargo, 1984; 23 pp.
6. Halvorson, G.A.; Melsted, S.W.; Schroeder, S.A.; Smith, C.M.; Pole, M.W. Topsoil and subsoil thickness requirements for reclamation of nonsodic mined-land. *Soil Sci. Soc. Am. J.* **1986**, *50*, 419–422.
7. Power, J.F.; Sandoval, F.M.; Ries, R.E. Topsoil–subsoil requirements to restore North Dakota mined lands to original productivity. *Mining Engng.* **1979**, *31* (12), 1708–1712.
8. Schuman, G.E.; Taylor, E.M., Jr.; Rauzi, F.; Pinchak, B.A. Revegetation of mined land: Influence of topsoil depth and mulching method. *J. Soil Water Conserv.* **1985**, *40*, 249–252.
9. Schuman, G.E.; King, L.A.; Smith, J.A. Reclamation of bentonite mined lands. In *Reclamation of Drastically Disturbed Lands*; Barnhisel, R.I., Darmody, R.G., Daniels, W.L., Eds.; American Society of Agronomy: Madison, 2000; Vol. 41, 687–707.
10. Dollhopf, D.J.; Rennick, R.B.; Smith, S.C. *Long-Term Effects of Physicochemical Amendments on Plant Performance at a Bentonite Mine Site in the Northern Great Plains*; Reclamation Research Unit Publication 88-02, Montana State University: Bozeman, 1988; 126 pp.
11. Sobek, A.A.; Schuller, W.A.; Freeman, J.R.; Smith, R.M. *Field and Laboratory Methods Applicable to Overburdens and Minesoils*. Publication 600/2-78-054;

- U.S. Environmental Protection Agency, Office of Research and Development: Cincinnati, 1978; 47–67.
12. Dollhopf, D.J. *Deep Lime Incorporation Methods for Neutralization of Acidic Minesoils*; Reclamation Research Unit Publication 9201; Montana State University: Bozeman, 1992; 1–88.
 13. Stumm, W.; Morgan, J.J. *Aquatic Chemistry*; Wiley: New York, 1970; 583.
 14. Skousen, J.G.; Sexstone, A.; Ziemkiewicz, P.F. Acid mine drainage control and treatment. In *Reclamation of Drastically Disturbed Lands*; Barnhisel, R.I., Darmody, R.G., Daniels, W.L., Eds.; American Society of Agronomy: Madison, 2000; Vol. 41, 131–168.
 15. Kapolka, N.M.; Dollhopf, D.J. Effect of slope gradient and plant growth on soil loss on reconstructed steep slopes. *Int. J. Surf. Mining Reclam. Environ.* **2001**, *15* (2), 86–89.
 16. Munn, L.; Harrington, N.; McGirr, D.R. Rock fragments. In *Reclaiming Mine Soils and Overburden in the Western United States; Analytical Parameters and Procedures*; Williams, D.E., Schuman, G.E., Eds.; Soil Conservation Society of America: Ankeny, 1987; 259–282.
 17. Larson, J.E. *Revegetation Equipment Catalog*; Stock No. 001-00518-5 Forest Service; U.S. Department of Agriculture, U.S. Government Printing Office: Washington, DC, 1980; 198 pp.

Mine Soils: Reclamation and Soil Carbon Sequestration

Vasant A. Akala

Ohio State University, Columbus, Ohio, U.S.A.

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Soil organic carbon (SOC) sequestration can be an environmentally friendly use of reclaimed minelands, and long-term carbon storage in such systems can offset part of the carbon dioxide emissions. The formation of primary organomineral complexes is the first step in the process of SOC sequestration. Soil aggregation and humification in reclaimed minelands can be higher than the undisturbed sites, resulting in high SOC sequestration. Pasture with topsoil application can achieve the highest SOC sequestration in a short period of time.

INTRODUCTION

The revegetation of mined lands is one of the management options for mitigation of the negative impacts of mining. The area of minelands that have been reclaimed since 1970 is 1 Mha in the United States alone.^[1,2] Land restoration measures can reverse the degradation trends, leading to positive changes in the ecosystem. These include improved water, air, and soil qualities and the attendant socioeconomic benefits.^[3] An important ancillary benefit of mine-land reclamation is its potential to sequester soil organic carbon (SOC).^[4] The potential for sequestration of carbon (C) in the aboveground biomass and minesoils of reclaimed minelands may be high and merit serious consideration as a significant benefit of restoration of highly disturbed land.

MINESOIL DEVELOPMENT AND SOC SEQUESTRATION

The rate of formation of minesoil or minesoil development over time may be the single most important edaphic factor determining the success of the reclamation plan. The rate of minesoil development is a function of reclamation plan management. Freshly exposed mine spoil may be thought of as soil at time zero, in respect to formation, because it consists only of rock and pulverized rock material.^[5] Mine spoils reflect the properties of the parent material more closely than natural soils because of the initial stage of pedogenic development.^[6]

Although there are different theories regarding the rate of minesoil development, there is a general consensus that soil weathering is rapid in the early stages of reclamation, and the rate of change decreases over time.^[7–9] Very rapid chemical reactions occur in spoil as the exposed material

begins to come into equilibrium with its new environment.^[7] Also, minesoils are exposed to processes of physical weathering, such as cycles of wetting and drying, freezing and thawing, and mechanical disruption by roots.^[9] Rapid changes in particle size distribution take place as spoil material weathers.^[10]

The potential of SOC enhancement in minelands depends on biomass productivity, root development in subsoil, and changes in minesoil properties resulting from overburden weathering.^[11] Improvement in soil aggregation is an important factor influencing minesoil development.^[12,13] The changes in minesoil properties impact on biomass productivity and pedogenic processes that lead to SOC sequestration in minesoils. Soil processes that are important in the C dynamics of minesoils and those that lead to SOC sequestration are weathering, stable structure formation, erosion, compaction, aeration, aggregation, nutrient recycling, humification, and mineralization. [Table 1](#) shows how some of these processes relate to SOC sequestration in minesoils.

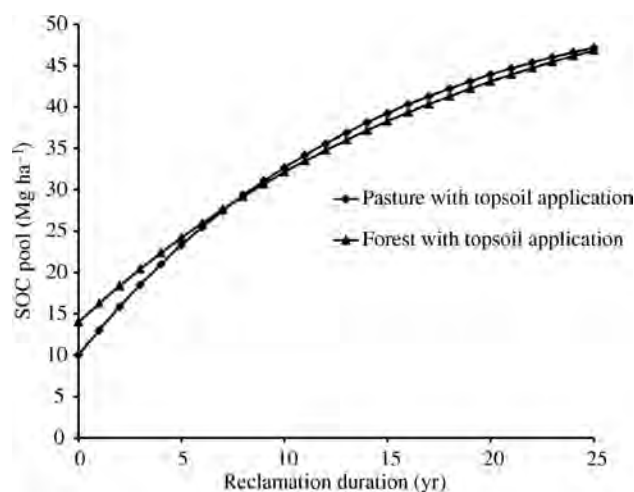
CASE STUDY

Akala and Lal^[14] conducted a study on a chronosequence of reclaimed minelands, where the effects of reclamation duration (time), land use (pasture and forest, with and without topsoil application), and soil processes (aggregation and humification) on SOC sequestration were observed. The study showed that soil disturbance by mining resulted in high loss of the antecedent SOC pools and reclamation of mined lands regained SOC. Temporal changes in the SOC pool of reclaimed minesoils showed that the potential to sequester SOC was high. The SOC pool of the soil surface (0–15 cm depth) during the initial phases of reclamation

Table 1 Minesoil processes and SOC sequestration.

Processes	Relation to SOC sequestration
Weathering	Formation of soils from the geologic material in the soil reduces the particle size of rock and coarse fragments, leading to the formation of stable peds and aggregates.
Erosion	Probably, the most important process to be addressed in the initial stages of reclamation because the topsoil is susceptible to erosion. Control of erosion in the initial phases assures establishment of vegetation and favorable impact on environmental quality.
Aggregation	Increasing clay and organic matter content favor the eventual formation of minesoil peds and minesoil structure. The presence and development of minesoil structure by aggregation lead to SOC sequestration. Indirectly, aggregation reduces erodibility and increases pore space, thereby increasing infiltration and permeability, and thus improving minesoil quality.
Compaction	Vehicular traffic involved in implementing the reclamation plan causes compaction. This result in increased density and soil strength and decreased aeration of the regraded spoil material. As a result of these changes, permeability is reduced and root proliferation and rooting depth are reduced. Water erosion may also be accelerated.

was 10–15 Mg ha⁻¹, which increased to 45–55 Mg ha⁻¹ in 25–30 years (Fig. 1). The low SOC pool during the initial phases of reclamation was attributed to the drastic disturbance caused by mining. Soil that was removed and stored was applied during reclamation lost significant amounts of SOC by decomposition and erosion during mining. Reclamation, minimum perturbation, and management of restored ecosystems, over a period of 20–25 years, had an ameliorative effect on minesoils because of decreases in

**Fig. 1** Change in SOC during reclamation over time.

soil compaction (reduction in soil bulk density), improvement in soil structure (high aggregation), and increases in SOC accumulation (enhanced biomass productivity). The SOC pool for 15–30 cm depth also increased over the period of reclamation. Possible mechanisms of C sequestration in reclaimed mineland were development of an A horizon, increased aggregation through the formation of organomineral complexes, and humification of soil organic matter (SOM).

Minesoil aggregation and C sequestration are closely related. Akala and Lal^[14] observed a high correlation between water stable aggregates and SOC content. Land disturbance caused by mining not only decreased SOC content of the soil aggregates (secondary organomineral complexes) but also caused a drastic decrease in SOC content associated with the primary soil particles (primary organomineral complexes). The loss in SOC content (in comparison to the antecedent levels in control sites) was 65% in the clay fraction, 75% in the silt fraction, and 40% in the sand fraction. The temporal increase in SOC content in all particle size fractions over the reclamation duration reflected the increase in total SOC pool. The increase (difference in SOC contents between the end and beginning of reclamation period) was 3 to 6 times the antecedent level for the clay and silt fractions and 1 to 2 times for the sand fraction. A large increase in the clay fraction showed the reactive nature of minesoils and its ability to respond to changes in SOC pool. The reactive nature of the primary particles coupled with the increase in SOM input was the most important factors for SOC sequestration in the initial stages of mineland reclamation.

DISCUSSION

The rates of SOC sequestration in reclaimed minelands can be 2–3 Mg ha⁻¹ yr⁻¹ for 0–15 cm depth and 1–2 Mg ha⁻¹ yr⁻¹ for 15–30 cm depth and are higher than those observed in other land uses. For example, SOC sequestration rates by adoption of conservation tillage are in the range of 0.1–0.5 Mg ha⁻¹ yr⁻¹ in temperate regions and 0.05–0.2 Mg ha⁻¹ yr⁻¹ in tropical regions.^[15] Similarly, SOC sequestration rates are in the range of 0.3–0.5 Mg ha⁻¹ yr⁻¹ by restoration of wetlands.^[16] Rates of SOC sequestration reported above can be sustained for a period of 10–15 years. The rate of change of SOC pools or SOC sequestration in reclaimed minelands depends on the magnitude of loss by perturbation (mining) and how far the reclaimed sites are from attaining the equilibrium SOC level. Soils that initially have low SOC content and those that are converted to restorative land management have a potential SOC sink capacity. Reclaimed minelands are an example of such aggrading ecosystems.

The SOC pool that can be potentially attained in reclaimed minelands in the United States is 25 million metric tonnes C.^[15] The net emission of carbon dioxide (CO₂)

increased from 1037 Tg (1 Tg = 1 Teragram = 1×10^{12} g = 1 million metric ton) C equivalent in 1990 to 1263 Tg C equivalent in 1996 in the United States.^[17] Assuming that the reclaimed minelands were at least 25 years old, they had a potential to offset 4% of CO₂ emissions for the United States. The importance of this potential may be realized when considered in combination with the restoration of all degraded soils and ecosystems and in the context of other soil management strategies.^[15]

CONCLUSION

1. The potential to sequester SOC in reclaimed minelands is high at 2–3 Mg ha⁻¹ yr⁻¹.
2. SOC sequestration can be an environmentally friendly use of reclaimed minelands, and long-term C storage in such systems can offset part of the CO₂ emissions (2–4%).
3. The formation of primary organomineral complexes is the first step in the process of SOC sequestration.
4. The total SOC content of the 25–30-year-old-reclaimed sites can be 30–35% higher than that of the undisturbed sites, but not all of the SOC are sequestered in the secondary organomineral complexes, thereby making it vulnerable to future land disturbance.
5. Soil aggregation and humification in reclaimed minelands can be higher than the undisturbed sites, resulting in high SOC sequestration.
6. Pasture with topsoil application can achieve the highest SOC sequestration in a short period of time.

REFERENCES

1. Office of Surface Mining (OSM). *1998 Annual Report*; Department of Interior: Washington, 1998.
2. National Mining Association (NMA). *Facts About Coal*; National Mining Association (NMA): Washington, 1998.
3. Hossner, L.R., Ed. *Reclamation of Surface Mined Lands, Vols. I and II*; CRC Press: Boca Raton, 1991.
4. United States Department of Energy (USDOE). *Energy Department Launches Thirteen New Research Projects to Capture and Store Greenhouse Gases*; DOE Fossil Energy Techline: Washington, 2000.
5. Kohnke, H. The reclamation of coal minesoils. *Adv. Agron.* **1950**, 2, 317–349.
6. Sobek, A.A.; Smith, R.M.; Schuller, W.A.; Freeman, J.R. *Overburden Properties that Influence Minesoils*; NCA/BCR: Pittsburgh, PA, 1976; 159 pp.
7. Struthers, P.H. Chemical weathering of strip minesoils. *Ohio J. Sci.* **1964**, 64, 125–131.
8. Caspall, F.C. *Soil Development on Surface Mine Spoils in Western Illinois*; NCA/BCR: Pittsburgh, PA, 1975; 228 pp.
9. Schafer, W.M.; Nielsen, G.A.; Dollhopf, D.J.; Temple, K. *Soil Genesis, Hydrological Properties, Root Characteristics and Microbial Activity of 1 to 50 Year Old Strip Mine Spoils*. EPA-600/7-79-100, USEPA: Cincinnati, OH, 1979.
10. Van Lear, D.H. *Effects of Spoil Texture on Growth of K-31 Tall Fescue*. Research Note NE-141, USDA Forest Service, 1971.
11. Haering, K.C.; Daniels, W.L.; Roberts, J.A. Changes in minesoil properties resulting from overburden weathering. *J. Environ. Qual.* **1993**, 22 (1), 194–200.
12. Boerner, R.E.J.; Scherzer, A.J.; Brinkman, J.A. Spatial patterns of inorganic N, P availability, and organic C in relation to soil disturbances: a chronosequence analysis. *Appl. Soil Ecol.* **1998**, 7 (2), 59–177.
13. Malik, A.; Scullion, J. Soil development on restored open-cast coal sites with particular reference to organic matter and aggregate stability. *Soil Use Manage.* **1998**, 14 (4), 234–239.
14. Akala, V.A.; Lal, R. Potential of mineland reclamation for soil organic carbon sequestration in Ohio. *Land Degrad. Dev.* **2000**, 11, 289–297.
15. Lal, R.; Kimble, J.M.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Ann Arbor Press: Chelsea, 1998; 128 pp.
16. Mitsch, W.J.; Wu, X. Wetlands and global change. In *Soil Management and Greenhouse Effect*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1995; 205–230.
17. Environmental Protection Agency (EPA). *Inventory of US Greenhouse Gas Emissions and Sinks: 1990–1996*; Environmental Protection Agency: Washington, 1998.

Minerals: Amorphous

April L. Ulery

Agronomy and Horticulture, New Mexico State University, Las Cruces, New Mexico, U.S.A.

Abstract

The term amorphous literally means “without form” and refers to a class of materials that are non-crystalline or, at best, poorly crystalline. Amorphous materials do not have a regular or repeating, ordered, internal atomic structure that is detectable by X-ray diffraction. Mineralogists use X-ray diffraction as the criterion for crystallinity and to identify most inorganic solids. Other terms used to define amorphous materials include short-range-order or X-ray amorphous minerals because the distances between repeating, internal patterns are too short to diffract X-rays.

INTRODUCTION

Crystalline minerals, in contrast to amorphous or non-crystalline materials, have long-range-order, usually in 3-D, and diffract X-rays from repeating atomic planes. Instead of a sharp boundary between amorphous and crystalline materials, a continuum exists between perfectly ordered, highly crystalline minerals and disordered, poorly crystalline to amorphous, non-crystalline solids (Fig. 1).

VARIETIES OF AMORPHOUS MINERALS

Amorphous soil minerals include oxides and hydroxides of aluminum, silicon, iron, titanium, and manganese and silicates of aluminum and iron. They may be hydrated weathering products of primary minerals or unweathered volcanic glass and biogenic silica (SiO_2) phytoliths. Amorphous weathering products occur as coatings or gel hulls on larger mineral grains and as aggregates or discrete particles in the soil clay fraction ($< 2 \mu\text{m}$ equivalent diameter). Glass particles and phytoliths are often larger than $2 \mu\text{m}$ in diameter, and, compared to the smaller amorphous soil components, are not as reactive.

Allophane, as a general term, has previously been used to describe any amorphous aluminosilicate in soils derived from volcanic material. Allophane ($1\text{-}2\text{SiO}_2\text{Al}_2\text{O}_3\cdot n\text{H}_2\text{O}^+$) is recognized as a soil mineral group with a range in alumina and SiO_2 composition and specific diagnostic properties. Imogolite $[(\text{OH})\text{SiO}_3\text{Al}_2(\text{OH}_3)]$ is a mineral with threadlike or tubular morphology and long-range order in only 1-D. Although it exhibits unique diffraction characteristics, it is often included in the discussion of amorphous minerals because it has similar chemical and physical properties with allophane and is often associated with allophane in soils.^[1] For more detailed information on allophane and imogolite, please see the section “Allophane” in this encyclopedia.

Poorly crystallized halloysite ($\text{Al}_2\text{Si}_2\text{O}_{10}\text{OH}_4\cdot n\text{H}_2\text{O}$) is another aluminosilicate that is included in the group of amorphous minerals because of its broad X-ray diffraction bands and tubular or curling morphology. Hisingerite ($\text{Fe}_2\text{O}_3 \cdot 2\text{SiO}_2\cdot n\text{H}_2\text{O}$) is a “very finely crystalline” to amorphous variant of iron phyllosilicates or an iron silicate analog of allophane. Other iron minerals observed in soils but amorphous to X-rays include ferrihydrite ($5\text{Fe}_2\text{O}_3 \cdot 9\text{H}_2\text{O}$) and feroxyhite (FeOOH). Ferrihydrite may exist anywhere along the continuum between crystalline to poorly crystalline. It can range from having short-range order in 3-D to less order in only 2-D.^[2] Poorly crystalline varieties of aluminum and titanium are also identified as pseudomorphs of their corresponding well-ordered minerals. They include pseudoboehmite (AlOOH) and pseudorutile ($\text{Fe}_2\text{O}_3\cdot n\text{TiO}_2\cdot m\text{H}_2\text{O}$). Amorphous varieties of manganese oxides have not been specifically named.

SiO_2 may be present in soils as one or more mineral varieties along a continuum that ranges from well-ordered through disordered to no atomic order (Fig. 1). Quartz and cristobalite (opal-C) represent the well-ordered, crystalline phases of SiO_2 , whereas biogenic opal (opal-A) and volcanic glass are non-crystalline hydrous SiO_2 compounds. Short-range-ordered to disordered cristobalite (opal-CT) is also found along the SiO_2 continuum. Biogenic opaline phytoliths and diatoms are valuable indicators of paleoenvironments and are identified by the morphology of the biological structure in which they originated. However, both SiO_2 phytoliths and glass from pyroclastic deposits are amorphous.^[3] SiO_2 is not as chemically reactive as other amorphous minerals, but it is important as a cementing agent in arid zone soils.

PROPERTIES OF SOILS CONTAINING AMORPHOUS MINERALS

Soils derived from volcanic ash are called Andisols and contain the highest amounts of amorphous components.

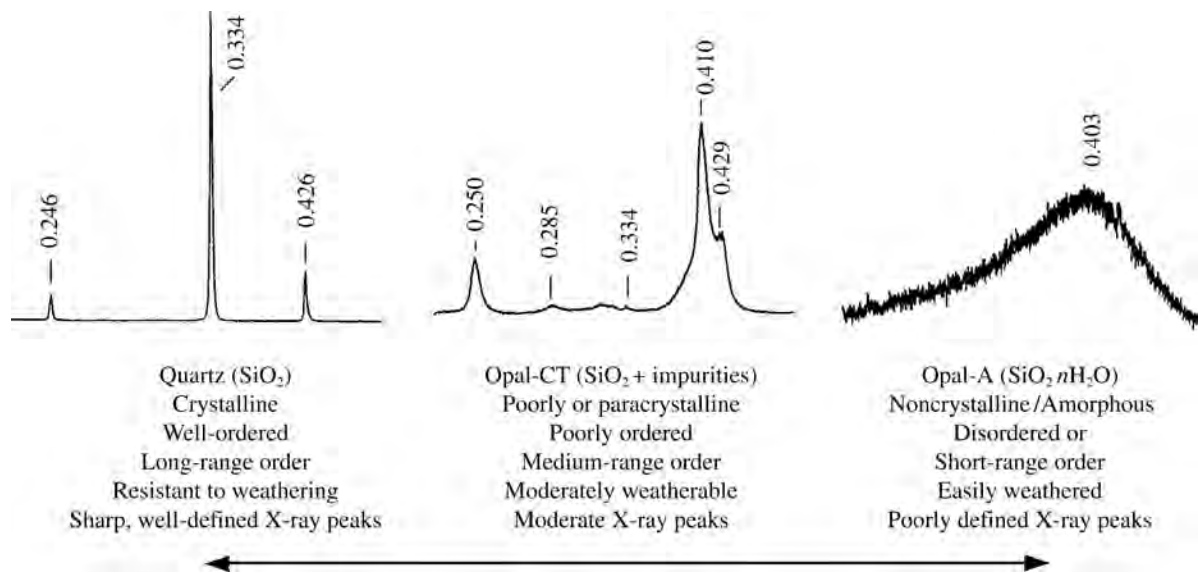


Fig. 1 A comparison of three SiO₂ varieties plotted along the crystallinity continuum. The X-ray peak positions are in nanometers, and the spectra were made using CuK α radiation. Note that the well-ordered quartz is typically purer than either the opal-CT or opal-A, which both may contain numerous impurities in the lattice structure.

Spodosols are soils with a diagnostic subsurface horizon containing up to 50% iron and aluminum hydrous oxides (a significant portion of which may be amorphous) and organic matter in the clay fraction. Hydrous iron and aluminum oxides, SiO₂, and kaolin minerals, some of them amorphous or poorly crystalline, dominate in oxic horizons, diagnostic for Oxisols. Even though in most soils amorphous materials comprise a relatively small part of the total mass, they contribute significantly to the physical and chemical properties of the soil.^[4]

Physical Properties

Amorphous materials are extremely important both physically and chemically because of their small particle size, large surface area, and reactive surface chemistry (Table 1). The surface area of amorphous minerals represents a large pool of active sites that can interact with cations, anions, and water to aggregate soil particles by forming bridges and

coatings. Large surface areas result from the tiny diameters of individual tubes or spheres of many amorphous materials and are several times higher than most crystalline soil minerals.

Amorphous materials strongly influence bulk density, aggregate stability, water holding capacity, and plasticity. Soils containing large amounts of amorphous materials have bulk densities ranging from 0.3 to 1.0 Mg m⁻³.^[5] In contrast, most mineral soils have bulk density values from 1.2 to 1.7 Mg m⁻³. Low bulk density and high void ratios are largely due to the formation of stable, porous aggregates by amorphous materials and the inability of lightweight glass shards and porous minerals to be compacted. Soils high in amorphous materials have void ratios ranging from 2 to 5, whereas sandy alluvial soils have values of 0.8 to 1.0 and clayey soils have values ranging from 1.5 to 2.5. These soils also exhibit large preconsolidation loads and anomalous compaction behavior.^[5]

Table 1 Ranges of selected physical and chemical properties of amorphous minerals.

Mineral	Particle			Surface area (m ² g ⁻¹)	CEC ^a (cmol _c kg ⁻¹)
	Density (Mg m ⁻³)	Diameter (nm)	Morphology ^b		
Allophane	2.7–2.8	3–5	Spherical	100–800	5–350
Ferrihydrite	2.4–3.8	3–7	Coatings/spherical	200–600	10–160
Halloysite	2.5–2.6	20–200	Tubular/spherical	10–45	10–40
Imogolite	2.6–2.8	10–30	Threadlike	700–1100	19–37
Opaline SiO ²	1.5–2.3	10–50	Spherical ^c	40–120	<10

^aCEC is strongly pH-dependent for most amorphous minerals and increases as the pH increases.
^bShape most commonly observed; others also may be reported.
^cAmorphous SiO₂ also exists as phytoliths, spicules, or shards visible to the naked eye (mm in length).

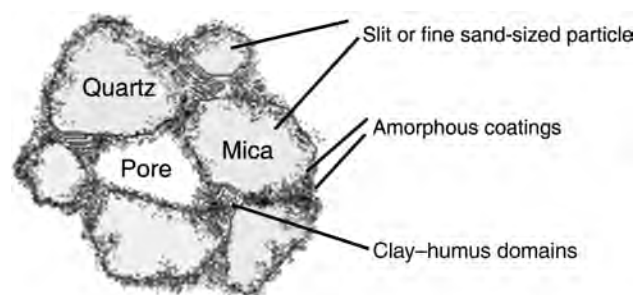


Fig. 2 Schematic representation of a soil aggregate stabilized by amorphous coatings surrounding larger primary minerals, biomass and connecting layers of clay-humus.

Stable aggregates are formed during the drying of soils rich in amorphous minerals. Amorphous or gel-like material-coating soil particles cements those particles into larger aggregates (Fig. 2). The amorphous coatings behave as viscous bodies when moist and elastic bodies when dry.^[6] The high content of organic matter commonly associated with amorphous materials also contributes to enhanced aggregate formation and stability and darkens the soil (in Japanese *ando* = black).

The moisture content of amorphous-rich or volcanic soils is much higher than that of other soils and ranges from 80% to 180% on an oven-dried basis.^[5] The high natural moisture content contributes to low bulk density values, which are based on dry soil weight per volume. The total porosity of many amorphous minerals and the aggregates they form provides numerous sites for water storage. Once dried, however, these minerals and soils tend to rehydrate incompletely.^[5] The irreversible wetting of these soils and minerals affects several physical and engineering properties including liquid and plasticity limits and water retention values.

Chemical Properties

Amorphous minerals have numerous molecular units, or surface functional groups, that when charged can affect organic matter retention, soil fertility, and the mobility of metals and organic compounds. Anion and cation exchange capacities (AEC and CEC) are variable for most amorphous minerals depending on solution pH and ionic composition. The pH-dependent or amphoteric charge tends to be positive at low pH and negative above the point of zero charge, which varies for each mineral but is usually between 3 and 7. Thus, AEC is typically higher at low pH, and CEC increases as the pH increases.

Because of the variable nature of surface charge on amorphous materials, the pH and method of analysis strongly influence the CEC and AEC values. Sample drying, solution concentration, temperature, pH, and mineral composition all affect CEC and AEC values.^[5] The CEC for individual amorphous minerals varies

widely (Table 1). Amorphous components, even in small amounts, contribute to the variable CEC and AEC in soils. For example, smectite-rich clayey soils containing 4–32% amorphous material had CEC values ranging from 24 to 300 cmol_c kg⁻¹.^[7]

Soil fertility and nutrient availability problems often arise in soils rich in amorphous materials. Strong covalent and ionic bonds form between oxyanions and amorphous mineral surfaces, especially under acidic conditions. The oxyanions, which include nutrients such as phosphate, sulfate, and molybdate or contaminants like selenite and arsenate, may be fixed or permanently adsorbed on the solid phase, which removes them from solution and makes them unavailable to plants. Anion exchange, precipitation reactions, and physical adsorption are other mechanisms that also remove oxyanions from soil solution.

IDENTIFICATION AND CHARACTERIZATION OF AMORPHOUS MINERALS

There is not a specific method universally accepted for the chemical characterization of soil amorphous minerals.^[8] The techniques used to identify and quantify amorphous components include selective dissolution, electron microscopy, differential thermal analysis and loss on ignition, differential X-ray diffraction, X-ray fluorescence, and infrared spectroscopy.

The most common method used to measure the chemically active portion of the amorphous fraction is extraction with acid ammonium oxalate in the dark.^[9] Following the extraction of non-crystalline aluminosilicates and hydrous oxides, boiling in 0.5 M sodium hydroxide or potassium hydroxide is used to remove amorphous SiO₂. It is useful to pretreat soil samples using one or both of these methods to clear the crystalline minerals of amorphous materials that coat, dilute, or otherwise interfere with the crystalline components of soil.^[9] Among several methods estimating the amorphous content of soils, it appears that the Rietveld method is one of the best to determine the total amorphous content of the clay fraction.^[8] For a variety of quantitative methods in soil mineral analysis, including amorphous components, the reader is referred to the work of Amonette and Zelazny.^[10]

CONCLUSION

Amorphous or non-crystalline minerals are atomic structures with short-range order to no order. Most amorphous minerals are extremely small and reactive both chemically and physically. They can exist as discrete particles and aggregates, but are most common as coatings on other soil particles. As coatings, they aid in the formation of aggregates, crusts, duripans, or other cemented features in soils. Soils high in amorphous minerals usually have high water

holding capacity, low bulk densities, and high void ratios. They also have slippery but non-sticky consistence, high Atterberg limits, greater values for liquid and plastic limits on undried than on dried samples, large preconsolidation loads, and anomalous compaction behavior.^[5] Amorphous minerals have large organic matter retention capability and pH-dependent ion exchange capacities. Phosphate, sulfate, and other ions or metals are commonly fixed by amorphous soil minerals.

REFERENCES

1. Wada, K. Allophane and imogolite. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; Soil Sci. Soc. Am. Book Series No. 1; 2nd Ed.; SSSA: Madison, 1989; 1051–1087.
2. Schwertmann, U.; Taylor, R.M. Iron oxides. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; Soil Sci. Soc. Am. Book Series No. 1; 2nd Ed.; SSSA: Madison, 1989; 379–438.
3. Drees, L.R.; Wilding, L.P.; Smeck, N.E.; Senkayi, A.L. Silica in soils: Quartz and disordered silica polymorphs. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; Soil Sci. Soc. Am. Book Series No. 1; 2nd Ed.; SSSA: Madison, 1989; 913–974.
4. Fey, M.V.; LeRoux, J. Properties and quantitative estimation of poorly crystalline components in sesquioxidic soil clays. *Clay. Clay Miner.* **1977**, *25*, 285–294.
5. Wada, K.; Harward, M.E. Amorphous clay constituents of soils. *Adv. Agron.* **1974**, *26*, 211–260.
6. Jones, R.C.; Uehara, G. Amorphous coatings on mineral surfaces. *Soil Sci. Soc. Am. Proc.* **1973**, *37*, 792–798.
7. Borchardt, G.A. Cation exchange capacity of noncrystalline clays in California landslides. In *Amorphous and Poorly Crystalline Clays Nature, Properties and Management: Extended Abstracts of; U.S.-Japan Seminar*; Harward, M.E., Wada, K., Eds.; Oregon State University: Corvallis, 1976; 57–58.
8. Jones, R.C.; Babcock, C.J.; Knowlton, W.B. Estimation of the total amorphous content of Hawaii soils by the Rietveld method. *Soil Sci. Soc. Am. Proc.* **2000**, *64* (3), 1100–1108.
9. Jackson, M.L.; Lim, C.H.; Zelazny, L.W. Oxides hydroxides, and aluminosilicates. In *Methods of Soil Analysis, Part 1. Physical and Mineralogical Methods*; Klute, A., Ed.; Agronomy Monogr. No. 9, 2nd Ed.; SSSA: Madison, 1986; 101–150.
10. Amonette, J.E.; Zelazny, L.W. *Quantitative Methods in Soil Mineralogy*; SSSA: Madison, 1994; 1–462.

Minerals: Primary

Nikolaos I. Barbayiannis

Aristotle University of Thessaloniki, Thessaloniki, Greece

Vissarion Z. Keramidas

Laboratory of Soil Science, School of Agriculture, Aristotle University of Thessaloniki, Thessaloniki, Greece

Abstract

Primary minerals identified in soils belong mainly to the classes of silicates, oxides of iron, zirconium, and titanium, and phosphates and are usually found in the sand and silt fractions. In soils, they are inherited from the parent material, and their presence is predicated by the nature of the mineral and the intensity of weathering, which results in the formation of secondary clay minerals in soils and the release of elements, essential for plant growth, in the soil solution. Therefore, primary minerals play a significant role in soil formation, and in the long run, they serve as sources of plant nutrients.

INTRODUCTION

According to the *Glossary of Soil Science Terms*,^[1] primary minerals are those that have not been altered chemically since their deposition or crystallization from molten lava (magma). Primary minerals identified in soils belong mainly to the classes of silicates, oxides of iron (Fe), zirconium (Zr), and titanium (Ti), and phosphates (apatite, $[\text{Ca}_{10}(\text{PO}_4)_6(\text{Cl}, \text{F}, \text{OH})_2]$). Their study is essential because: 1) they serve as sources of plant nutrients; 2) they are, in certain cases, the precursors of secondary clay minerals; and 3) they provide information about soil development.

SILICATES

Of the primary minerals found in soils, silicates are the most abundant, comprising nearly 40% of the common minerals. The building unit of the silicates is the silicon (Si) tetrahedron. Silicate structures may consist of single tetrahedron (nesosilicates), double tetrahedra (sorosilicates), rings (cyclosilicates), single or double chains (inosilicates), sheets (phyllosilicates), or framework patterns (tectosilicates). Typical silicate minerals most likely to be found in soils are presented in [Table 1](#).

Olivines

Olivines are olive green nesosilicates in which divalent cations mainly Mg^{2+} and Fe^{2+} join the Si tetrahedra by forming octahedra of Mg^{2+} or Fe^{2+} . The forsterite–fayalite series include the most abundant naturally occurring olivines, with Mg^{2+} and Fe^{2+} being the respective cations of the series end members.

Olivines are the first minerals to crystallize in the initial stages of magma solidification, and they are highly unstable minerals because the high ratio of the divalent cations to Si in its structure renders them vulnerable to chemical attack. Therefore, olivines are very easily weathered in soils, and for this reason, they are considered relatively rare soil constituents and can be found only in very young soils and in the coarser sand fractions.

Zircon

Zircon is another nesosilicate mineral. Its structure consists of alternating edge-sharing Si tetrahedra that are held together with Zr^{4+} ions that are located in the center of triangular dodecahedra.^[3]

Zircon is found as residual grains in the sand and silt fractions of soils, and because of its stability in pedogenic environments, it is frequently related to the degree of soil development and soil age.

Pyroxenes and Amphiboles

Pyroxenes and amphiboles are “ferromagnesian” minerals with single- and double-chain structures, respectively ([Table 1](#)). Pyroxene chains have Si tetrahedra sharing two oxygen (O) atoms. In amphiboles, the chains are formed by Si tetrahedra sharing alternatively two and three O atoms. In both classes of minerals, the chains are held together mainly by divalent cations (e.g., Mg^{2+} , Fe^{2+} , or Ca^{2+}) located at the center of an octahedron.^[4]

Augite and hornblende are the most important minerals of the pyroxenes and amphiboles, respectively. Pyroxenes and amphiboles are formed at lower temperatures and

Table 1 Common primary minerals of the silicate class found in soils.

Silicon tetrahe- dra arrangement	Name of mineral or group	Ideal formula
Nesosilicates (SiO ₄) ⁴⁻ (single)	Olivine	(Mg,Fe) ₂ SiO ₄
Zircon	ZrSiO ₄	
Sorosilicates (Si ₂ O ₇) ⁶⁻ (double)	Epidote	Ca ₂ (Al,Fe)Al ₂ O (SiO ₄) ₄ (Si ₂ O ₇)(OH)
Cyclosilicates (Si ₆ O ₁₈) ¹²⁻ (rings)	Tourmaline	(Na,Ca)(Li,Mg,Al) (Al,Fe,Mn) ₆ (BO ₃) ₃ (Si ₆ O ₁₈)(OH) ₄
Inosilicates		
Single chains (SiO ₃) ²⁻	Pyroxenes	
Augite	(Ca,Na)(Mg,Fe,Al) (Si,Al) ₂ O ₆	
Hypersthene	(Mg,Fe)SiO ₃	
Double chains (Si ₄ O ₁₁) ⁶⁻	Amphiboles	
Hornblende	(Ca,Na) ₂₋₃ (Mg,Fe,Al) ₅ Si ₆ (Si,Al) ₂ O ₂₂ (OH) ₂	
Phyllosilicates (Si ₄ O ₁₀) ²⁻ (sheets)	Micas	
Muscovite	KAl ₂ (AlSi ₃ O ₁₀)(OH) ₂	
Tectosilicates (SiO ₂) ⁰ (framework)	Feldspars	
Orthoclase	KAlSi ₃ O ₈	
Albite	NaAlSi ₃ O ₈	
Anorthite	CaAl ₂ Si ₂ O ₈	
Quartz	SiO ₂	

Source: From Shultz. [2] ©1989 SSSA.

pressures during magma solidification compared to olivines and are more stable in the weathering environment. In soils, however, they weather relatively faster and are largely confined in the sand and silt fractions. Some of them may occur in the clay fraction of soils that are young and have not been subjected to intensive weathering. [5]

Feldspars

Feldspars are tectosilicates, with 3-D framework of corner-linked Si and aluminum (Al) tetrahedra. The Al tetrahedral result from Al substitution for Si and the charge deficit created is compensated by sodium (Na), potassium (K), and calcium (Ca). Feldspars are virtually present in all sediments and soils in quantities that vary with the nature of the parent material and the stage of weathering. The majority of feldspars belong to the ternary system NaAlSi₃O₈ (albite)–KAlSi₃O₈ (K-feldspar)–CaAl₂Si₂O₈ (anorthite). In K-feldspars, one of every four Si is replaced by Al, with K balancing the charge. Most common K-feldspars are sanidine, orthoclase, microcline, and adularia. Na and Ca feldspars form a series called “plagioclase”

with compositions ranging from pure albite (Na-feldspar) to pure anorthite (Ca-feldspar).

The presence of feldspars in soils is related to the overall mineralogical composition and the prevailing environmental conditions, including climate, topography, degree of leaching, the presence of chelating agents, redox status, and soil solution composition. [6]

Feldspars usually follow the stability sequence: anorthite < albite < K-feldspars. [3]

Weathering of plagioclase minerals increases the supply of Ca in soils in a manner similar to that of weathering of K-feldspars in the clay, and silt fractions provide an important source of K in soils.

Micas

Micas are 2:1 phyllosilicate minerals formed by two Si tetrahedral sheets and an Al octahedral sheet in between. One of four Si in the tetrahedral sheet is replaced by Al, and the high negative layer charge developed is balanced by non-hydrated interlayer cations that hold the layers tightly together. [7] The most abundant and important micas in soils (muscovite and biotite) have K as the interlayer cation, which upon weathering is released in the soil solution and serves as a plant nutrient.

Micas are formed in soils from the parent material, and as they tend to weather to other minerals, they are expected to prevail in younger, less weathered soils. [8] In young soils, they tend to occur as discrete sand and silt particles, while in more weathered soils they are usually found in the clay fraction. Biotite is uncommon in most soils because it is transformed very easily to other secondary minerals, even under mild weathering conditions. Muscovite is commonly found in soils of advanced weathering.

Like the other primary minerals in soils, mica becomes unstable in the weathering conditions at the soil surface. This is demonstrated by the fact that biotite is almost absent from the surface horizons of most, except very young, soils. Micas are considered as the precursors of several secondary 2:1 minerals and clay minerals in soil like illites, vermiculites, and smectites. [8,9]

Quartz

Quartz is one of the seven polymorphs of silicon dioxide, found in soils and sediments: quartz, cristobalite, tridymite, coesite, stishovite, lechatelierite (silica glass), and opal. [10] Quartz and disordered cristobalite are the typical forms found in soils, while tridymite is present mostly in soils developed from siliceous volcanic rocks.

Quartz belongs to the tectosilicate class of the silicate minerals (Table 1), which have each O of Si tetrahedra linked to Si atoms of adjacent tetrahedra, forming a 3-D framework structure. Owing to its resistance to weathering, quartz is generally concentrated in the sand and silt fractions, and generally eluvial (surface) horizons are enriched owing to weathering and removal of less resistant minerals.

Quartz is considered as inert skeletal material and exhibits very small to nil anion and cation exchange properties. In certain horizons, however, poorly crystalline quartz and other silica minerals may serve as cementing agents. In normal soils, quartz particles are also considered as important components of the soil structure because they are linked together or to clay domains through organic matter bridges.^[11] Quartz is also a suitable mineral to assess parent material uniformity and degree of weathering because of its abundance, resistance to weathering, and immobility.

OXIDES

Fe

Of the several Fe oxides and hydroxides found in soils, magnetite (Fe_3O_4) is the only one of primary origin. The crystal structure of Fe_3O_4 , like all Fe oxides, consists of Fe ions surrounded by six O in a sixfold coordination.

Fe_3O_4 occurs as black grains in the heavy fraction of the sand and silt fraction of soils and is of lithogenic origin.^[12] As Fe_3O_4 is found in the coarser soil fractions and is easily transformed to maghemite in the finer fractions, its influence on soil chemistry is minor.

Ti

Rutile (TiO_2) and ilmenite are the main primary Ti minerals found in soils. TiO_2 is tetragonal and consists of TiO_6 polyhedra that share their opposite edges and form chains along the c-axis.

Ti oxides are common accessory minerals of igneous and metamorphic rocks and have been identified in the sand and silt fractions of soils originating from such rocks.

TiO_2 develops a negative charge through adsorption of OH^- on surface Ti ions. This charge is pH dependent. Charge development plays an important role in sandy acid soils and influences physical and chemical properties such as formation of stable aggregates and retention of nutrients such as phosphate.

Ti oxides, in the similar way as zircon, are most frequently related to soil genesis studies. Their occurrence in the non-clay fraction in the different horizons of soils and the parent material provides information about the profile uniformity and soil development.

PHOSPHATES

Phosphate minerals comprise only a small part of the inorganic fraction of soils, and their basic structural unit is the orthophosphate ion (PO_4^{3-}), which forms a tetrahedron where phosphorous (P) is surrounded by four O^{2-} . The high affinity of PO_4^{3-} for cations forms structures that are classified as framework, chain, and layer phosphates, similar to silicates.^[13] $[\text{Ca}_{10}(\text{PO}_4)_6(\text{Cl},\text{F},\text{OH})_2]$ is the most frequently

reported phosphate of igneous origin and has been identified with photographic microscopy as discrete grains in the sand and silt fraction of a number of soils. Along with their weathering products, apatites are considered as a natural source of P for plants.

CONCLUSION

Primary minerals identified in soils belong mainly to the classes of silicates, oxides of Fe, Zr, and Ti, and phosphates and are usually found in the sand and silt fractions. In soils, they are inherited from the parent material, and their presence is predicated by the nature of the mineral and the intensity of weathering, which results in the formation of secondary clay minerals in soils and the release of elements, essential for plant growth, in the soil solution. Therefore, primary minerals play a significant role in soil formation, and in the long run, they serve as sources of plant nutrients.

REFERENCES

1. Soil Science Society of America. *Glossary of Soil Science Terms*; Bigham, J.M., Ed.; SSSA: Madison, 1997; 134 pp.
2. Shultze, D.E. An introduction to soil mineralogy. In *Minerals in Soil Environments*, 2nd Ed.; Series 1; Dixon, J.B., Weed, S.B., Eds.; SSSA: Madison, 1989; 1–34.
3. Deer, W.A.; Howie, R.A.; Zussman, J. *An Introduction to the Rock Forming Minerals*; Longman Ltd.: Essex, 1992; 696 pp.
4. Huang, P.M. Feldspars, olivines and amphiboles. In *Minerals in Soil Environments*, 2nd Ed.; Series 1; Dixon, J.B., Weed, S.B., Eds.; SSSA: Madison, 1989; 975–1050.
5. Churchman, G.J. The alteration and formation of soil minerals by weathering. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; F3–F76.
6. Allen, B.L.; Hajek, B.F. Mineral occurrence in soil environments. In *Minerals in Soil Environments*, 2nd Ed.; Series 1; Dixon, J.B., Weed, S.B., Eds.; SSSA: Madison, 1989; 199–278.
7. Fanning, D.S.; Keramidas, V.Z.; El-Desoky, M.A. Micas. In *Minerals in Soil Environments*, 2nd Ed.; Series 1; Dixon, J.B., Weed, S.B., Eds.; SSSA: Madison, 1989; 551–634.
8. von Reichenbach, H.G.; Rich, C.I. Fine grained micas in soils. In *Soil Components. Part 2. Inorganic Components*; Gieseking, J.E., Ed.; Springer: Berlin, 1975; 59–95.
9. Borchardt, G. Smectites. In *Minerals in Soil Environments*, 2nd Ed.; Series 1; Dixon, J.B., Weed, S.B., Eds.; SSSA: Madison, 1989; 675–727.
10. Drees, L.R.; Wilding, L.P.; Smeck, N.E.; Senkeyi, A.L. Silica in soils: Quartz and disordered silica polymorphs. In *Minerals in Soil Environments*, 2nd Ed.; Series 1; Dixon, J.B., Weed, S.B., Eds.; SSSA: Madison, 1989; 913–974.
11. Emerson, W.W. The structure of soil crumbs. *J. Soil Sci.* **1959**, *10*, 235–244.
12. Schwertman, U.; Taylor, R.M. Iron oxides. In *Minerals in Soil Environments*, 2nd Ed.; Series 1; Dixon, J.B., Weed, S.B., Eds.; SSSA: Madison, 1989; 379–438.
13. Lindsay, W.L.; Vlek, L.G.; Chien, S.H. Phosphate minerals. In *Minerals in Soil Environments*, 2nd Ed.; Series 1; Dixon, J.B., Weed, S.B., Eds.; SSSA: Madison, 1989; 1089–1130.

Minerals: Secondary

Deborah A. Soukup

Geomatrix Consultants, Inc., Bakersfield, California, U.S.A.

Abstract

This entry presents a broad overview on the formation and characteristics of common secondary minerals in soils. The alteration pathways occurring in the soil environment are very complex and are influenced by a variety of abiotic and biotic factors. The secondary minerals discussed in this entry may also occur in soils as a result of inheritance from parent materials.

INTRODUCTION

There are three possible origins for minerals in soil environments including inheritance from parent materials, weathering transformations of existing minerals, and neoformation or crystallization from solution.^[1] Secondary minerals are defined as minerals formed later than the rock enclosing them, usually at the expense of an earlier-formed primary mineral, as a result of weathering, metamorphism, or solution.^[2] Alternatively, secondary minerals may be defined as recrystallized products of the chemical breakdown and/or alteration of primary minerals.^[3] Secondary minerals are generally characterized by smaller particle size, because the particle size of primary minerals is decreased during weathering and release of soluble materials. Therefore, secondary minerals are typically principal components of the silt and clay fraction of soils.^[4]

TYPES OF SECONDARY MINERALS IN SOIL

The major secondary minerals and the soil orders in which they commonly occur are listed in [Table 1](#). Primary minerals including feldspars, pyroxenes, amphiboles, micas, and primary chlorite may be altered to secondary minerals such as illite, vermiculite, clay chlorite, smectites, kaolinite, halloysite, and oxides of iron (Fe) and aluminum (Al) by the removal of silica and bases, and the addition of water. Additional details and the significance of these secondary minerals in soil environments are discussed in the following sections. Selected physical and chemical characteristics of the secondary minerals are summarized in [Table 2](#).

Iron and Aluminum Oxides

The most common oxides of Fe and Al, typically reported in soils, are goethite, hematite, and gibbsite.^[4,6,8] These minerals, sometimes referred to as *sesquioxides*, generally occur in soils subject to intense weathering, e.g., Oxisols

and Ultisols. The sesquioxide minerals are amphoteric in character and exhibit variable charges. These minerals are also characterized by high surface charge density, high specific surface area, and high cation and anion adsorption capacity.^[4–9] Additionally, sesquioxide minerals are reported to enhance soil aggregation. Finely divided iron oxides are believed to act as binding agents among other soil particles and may result in cementation of these particles into large units.^[6]

Even at low concentrations, the iron oxides goethite and hematite play an important role in influencing soil color because of their pigmenting power.^[6] Goethite minerals are naturally yellow, whereas hematite is bright red. In fact, the greater pigmenting power of hematite can mask the yellow color of higher concentrations of goethite. Gibbsite, in contrast, is colorless; however, its content is used in the oxidic ratio to indicate the relative degree of weathering

$$\text{Oxidic ratio} = \frac{(\% \text{ extractable Fe}_2\text{O}_3 + \% \text{ gibbsite})}{\% \text{ clay}} \quad (1)$$

If the oxidic ratio is 0.2, the soil is considered to be highly weathered.

Kaolinite and Halloysite

Kaolinite is one of the most common clay minerals in soils, particularly those of warm, moist climates. Kaolinite is a 1:1 aluminosilicate mineral composed of one octahedral sheet stacked above one tetrahedral sheet. The two crystal units comprising one kaolinite particle are held together by hydrogen bonds, and the space between the structural layers, therefore, has a fixed dimension.^[4,11] Halloysite is also a 1:1 aluminosilicate mineral with the same composition as kaolinite, except that halloysite may be hydrated and contain water between the structural layers. Both kaolinite and halloysite are products of acid weathering, but halloysite is formed more rapidly in soils of volcanic origin.

Kaolinite and halloysite typically have low surface areas and low cation exchange capacity (CEC) and anion

Table 1 Ideal chemical formula of secondary minerals commonly occurring in soils.

Mineral name	Chemical formula	Major occurrences
Oxides of Fe and Al		
Goethite	FeOOH	Oxisols, Ultisols
Hematite	Fe ₂ O ₃	Oxisols, Ultisols
Gibbsite	Al(OH) ₃	Oxisols, Ultisols
Aluminosilicate clays		
Kaolinite	Al ₂ Si ₂ O ₅ (OH) ₄	Ultisols, Oxisols, Alfisols
Halloysite	Al ₂ Si ₂ O ₅ (OH) ₄ ·2H ₂ O	Andisols
Illite	K(Si ₃ Al)(AlMg) ₂ O ₁₀ (OH) ₂	Alfisols, Mollisols
Chlorite ^a	[(M ²⁺ , M ³⁺) ₃ (SiAl) ₄ (Al ₂)O ₁₀ (OH) ₂] ^{x-} [(M ²⁺ , M ³⁺) ₃ (OH) ₆] ^{x+}	Common accessory mineral in many soils
Vermiculite	Mg(SiAl) ₄ (Al ₂)O ₁₀ (OH) ₂	Common in many soils
Smectite	Na(Si ₄)(AlMg) ₃ O ₁₀ (OH) ₂	Vertisols, Mollisols, Alfisols
Hydroxy-interlayer Vermiculite and Smectite	Chemical composition highly variable	Wide geographic distribution but most abundant in Ultisols and Alfisols
Carbonate and sulfate minerals		
Dolomite	CaMg(CO ₃) ₂	Aridisols, Mollisols, Alfisols, arid Entisols
Calcite	CaCO ₃	Aridisols, Mollisols, Alfisols, arid Entisols
Gypsum	CaSO ₄ ·2H ₂ O	Aridisols

^aM²⁺ and M³⁺ represent cations with charges of +2 and +3, respectively.

exchange capacity. Isomorphous substitution within the crystal is limited, contributing to the low permanent charge.

Table 2 Selected physical and chemical characteristics of secondary minerals commonly occurring in soils.

Mineral name	Surface area (m ² /g)	CEC (cmol _c /kg)
Oxides of Fe and Al		
Goethite ^a	50–200	3
Hematite ^a	50–200	3
Gibbsite ^b	58 > 600	1–3
Aluminosilicate clays		
Kaolinite	5–39	2–15
Halloysite	21–43	10–60
Illite	80–150	20–40
Chlorite	25–150	10–40
Vermiculite	600–800	100–200
Smectite	600–800	80–150
Hydroxy-interlayer Vermiculite and Smectite	25–150	CEC varies depending on degree of filling of interlayer space
Carbonate and sulfate minerals		
Dolomite ^c		
Calcite ^c		
Gypsum ^c		

^aSee McBride,^[5] Schwertmann & Taylor,^[6] and Bigham, Fitzpatrick, et al.,^[7] for additional information.

^bSee McBride,^[5] Hsu,^[8] and Huang, Wang, et al.,^[9] for additional information.

^cSee Doner & Lynn^[10] for additional information.

However, kaolinite and halloysite may develop variable or pH dependent negative charge because of the dissociation of protons for exposed OH groups.^[4,11] The typical range in CEC for kaolinite and halloysite is from 1 to 10 cmol(+)/kg.

Kaolinite-containing clays are used extensively in the production of brick, sewer pipes, and drain tiles. Kaolinite is also used in the ceramic industry, because of its low expansion and contraction capacity. Additionally, kaolinite is used for the production of false teeth, paper making, and in the pharmaceutical industry.^[4,11]

Illites

Illites are micaceous clays and have been referred to as *hydrous micas* and *hydrobiotite*.^[4] However, illite differs from micas, such as muscovite, in that it contains more silicon dioxide and less potassium (K). Illites are 2:1 aluminosilicate minerals consisting of one octahedral sheet sandwiched between two tetrahedral sheets. The 2:1 layers are bound together strongly by K ions, and thus, illites are non-expansive and do not exhibit shrink–swell behavior upon wetting and drying.^[4,12] Illite typically weathers to smectite under conditions of high precipitation, because of the loss of interlayer K.

Vermiculites

Vermiculite is also a 2:1 aluminosilicate mineral with magnesium (Mg) occupying the octahedral positions between the two tetrahedral sheets.^[4,13] Vermiculites are common weathering products of micas in well-

drained soils and are characterized by high CECs ranging from 144 to 182 cmol(+)/kg.^[13] These minerals exhibit *wedge zones*, attributed to marginal curling of layers on the mineral surface.^[4] These wedge zones provide partially enlarged interlayer spacings in which organic matter may be entrapped, or for fixation of K, NH_4^+ , and other cations. The high fixation capacity of many soils for K and NH_4^+ is primarily attributable to the presence of vermiculite. Vermiculite is also widely used as a potting medium in nurseries and in cat litter.

Smectites

Smectites are a group of expansive, 2:1 aluminosilicate minerals, commonly occurring in soils with impeded drainage and/or in soil environments characterized by high silicon (Si) and basic cation activities.^[14] Minerals within the smectite group are classified based upon the location of the layer charge, either within the tetrahedral or octahedral sheets. In montmorillonite, substitution of Mg for Al in the octahedral sheet produces the layer charge. The layer charges of beidellite and nontronite, two other important smectite minerals, are concentrated in the tetrahedral sheets. The predominant octahedral cations in beidellite and nontronite are Al and Fe, respectively.^[14]

Smectites are generally responsible for the high shrink-swell conditions in soil and result in more damage annually than any other natural disaster including earthquakes and floods. Annual estimates of damage to residential structures in the United States were approximately 798.1 million in 1970 and was projected to increase to 997 million by the year 2000.^[15] These figures increase by a factor of 2–3 if damage to industrial and commercial buildings and transportation infrastructure is included.

Hydroxy-Interlayered Vermiculite and Smectite

Hydroxy-interlayered vermiculite (HIV) and smectite (HIS) are believed to occur in soils as weathering products derived from chlorite or micas or from the deposition of hydroxy-Al or -Fe polymeric components within the interlayer spaces of vermiculite and smectite.^[16] HIV and HIS are widely distributed geographically and are found in several soil orders, although they are most commonly reported in Ultisols and Alfisols.^[16]

There is significant variability in the composition of HIV and HIS, because the composition is dependent on the basic 2:1 mineral structure and the type and amount of hydroxy-interlayer material within the interlayer spaces. The presence of hydroxy-Al and -Fe interlayer components may have a profound effect on some of the physical and chemical properties of HIV and HIS, including swelling, CEC, and adsorption of metals and anions. Many investigators have reported that formation of hydroxy-Al and -Fe interlayers resulted in reduced swelling and dispersion of

smectites. Hydroxy-Al polymers were more effective than hydroxy-Fe polymers in reducing swelling, presumably because the Al interlayer components are more uniformly distributed within the interlayer space, and their greater relative stability.^[16]

The CEC of HIV and HIS may also be reduced as the interlayer space is filled. The primary mechanisms proposed for the observed charge reduction in HIV and HIS compared to the non-interlayered end members are precipitation of Al on surfaces and/or into interlayer spaces, sterically blocking exchange sites, and adsorption of positively charged hydroxy-Al polymers that are non-exchangeable.^[16] Hydroxy-Al interlayers can also prevent fixation of K^+ , NH_4^+ , Cs^+ , and Rb^+ because the interlayer material inhibits collapse of the layers about these ions. Finally, the presence of hydroxy interlayers in 2:1 clay minerals significantly enhances the adsorption of anions, such as P^{3-} .^[16] The increase in anion adsorption in hydroxy-interlayered clays occurs because anions are only adsorbed at edge sites in minerals not containing hydroxyl interlayers.

Chlorites

Chlorites are hydrated Mg and Al silicates that are commonly green in color and resemble micas in appearance.^[4,16] Structurally, chlorite is a 2:2 aluminosilicate mineral consisting of one octahedral sheet and two tetrahedral sheets with the interlayer space occupied by $\text{Mg}(\text{OH})_2$ or brucite layers. The replacement of Mg by Al in the brucite layers creates enough positive charge to nearly neutralize the negative charges. Thus, chlorite has little or no charge and a small CEC. Chlorites are less stable than most of the other clays in acidic environments and are subject to rapid weathering.^[4,16]

Carbonate and Sulfate Minerals

Calcium (Ca) and magnesium carbonate minerals, including calcite and dolomite, may originate from several sources or combinations of sources, either directly in the form of carbonates, or by a solution-precipitation mechanism.^[10] Dissolution of more soluble Ca-bearing minerals, such as gypsum, may result in the subsequent precipitation of calcite. Carbonates may also be formed by the reaction of Ca ions in rainwater and surface water with CO_2 -charged H_2O in the soil. Coatings of calcium carbonate may eventually accumulate and result in cementation in some soils. Movement of clay in calcareous soils has also been reported to be restricted because of flocculation of the silicate clays by Ca^{2+} .^[10]

Gypsum may be precipitated in soil from surface or subsurface waters rich in Ca and SO_4 . It may also be formed through the oxidation of pyrite and subsequent reaction of acid sulfate with CaCO_3 .^[10] Gypsum has frequently been used to reclaim sodic soils because of

its high solubility. However, numerous cases of soil subsidence and deterioration of concrete due to the presence of gypsum have been reported.

CONCLUSION

In summary, it should be noted that the information presented in this entry on the formation and characteristics of common secondary minerals in soils is a broad overview that simplifies many relevant details. The actual alteration pathways occurring in the soil environment are very complex and are influenced by a variety of abiotic and biotic factors. The secondary minerals discussed in this entry may also occur in soils as a result of inheritance from parent materials. Additionally, it may be extremely difficult in some soil environments to determine the precise origin of a given mineral.

REFERENCES

1. Allen, B.L.; Hajek, B.F. Mineral occurrence in soil environments. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 199–278.
2. Bates, R.L.; Jackson, J.A. *Dictionary of Geological Terms*, 3rd Ed.; Doubleday: New York, 1984.
3. Brady, N.C.; Weil, R.R. *The Nature and Properties of Soils*, 12th Ed.; Prentice-Hall: Upper Saddle River, 1999.
4. Tan, K.H. *Environmental Soil Science*, 2nd Ed.; Marcel Dekker: New York, 2000; 28–79.
5. McBride, M.B. *Environmental Chemistry of Soils*; Oxford University Press: New York, 1994.
6. Schwertmann, U.; Taylor, R.M. Iron oxides. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 379–438.
7. Bigham, J.M.; Fitzpatrick, R.W.; Schulze, D.G. Iron oxides. In *Soil Mineralogy with Environmental Applications*; Dixon, J.B., Schulze, D.G., Eds.; Soil Science Society of America: Madison, 2002; 323–366.
8. Hsu, P.H. Aluminum hydroxides and oxyhydroxides. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 331–378.
9. Huang, P.M.; Wang, M.K.; Kämpf, N.; Schulze, D.G. Aluminum hydroxides. In *Soil Mineralogy with Environmental Applications*; Dixon, J.B., Schulze, D.G., Eds.; Soil Science Society of America: Madison, 2002; 261–289.
10. Doner, H.E.; Lynn, W.C. Carbonate, halide, sulfate, and sulfide minerals. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 279–330.
11. Dixon, J.B. Kaolin and serpentine group minerals. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 467–525.
12. Fanning, D.S.; Keramidas, V.Z.; El-Desoky, M.A. Micas. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 551–634.
13. Douglas, L.A. Vermiculites. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 635–674.
14. Borchardt, G. Smectites. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 675–727.
15. Reid-Soukup, D.A.; Ulery, A.L. Smectites. In *Soil Mineralogy with Environmental Applications*; Dixon, J.B., Schulze, D.G., Eds.; Soil Science Society of America: Madison, 2002; 467–499.
16. Barnhisel, R.I.; Bertsch, P.M. Chlorites and hydroxy-interlayered vermiculite and smectite. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America: Madison, 1989; 729–788.

Minerals: Solubility

Wayne P. Robarge

Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.

Abstract

Minerals are found in almost all soils, and the precipitation and dissolution reactions of minerals are an integral component of the biogeochemical cycles in soils. A mineral is considered a naturally occurring substance, inorganic in composition, with a definite chemical composition and an ordered atomic arrangement. Minerals in soils are divided into two broad categories: primary and secondary minerals.

MINERALS

Primary Minerals

A primary mineral is considered one that has not been altered chemically since its deposition and crystallization as a result of large-scale geological processes.^[1] Soils inherit primary minerals from the parent material from which the soil is derived.^[2] Chemical weathering of primary minerals (a dissolution reaction) represents the release of plant nutrients such as magnesium (Mg), calcium (Ca), potassium, phosphorus (P), iron (Fe), manganese (Mn), and boron for plant uptake and recycling in soil–plant ecosystems. Primary minerals are also a source of trace metals, which can either be essential or toxic to life. As most primary minerals are formed by tectonic activities, they are inherently unstable in soils and will dissolve.

Secondary Minerals

Secondary minerals form from the decomposition and restructuring of primary minerals or from precipitation reactions (formation of a new mineral) involving chemical constituents of primary minerals released during dissolution.^[1] Most secondary minerals in soils are phyllosilicates: silicate-based minerals having sheets of silicate anions linked to sheets of cations [usually aluminum (Al)] to form a layer.^[3] The cations in phyllosilicates also have reacted with oxygen such that the sheets of silicate anions and cations are linked together by sharing oxygen atoms. This yields a very stable structure that resists weathering. Phyllosilicates can be formed in soils or are inherited in the soil parent material, having been formed elsewhere as part of the weathering of primary minerals.^[4]

Additional secondary minerals found in soils include oxides, sulfates, sulfides, carbonates,^[5] and phosphates.^[6] In arid environments, the presence of sulfates and carbonates is common, but they are generally absent from

well-leached soils in humid environments. Most oxides are stable in well-aerated soils and are considered the end product of the decomposition of primary and secondary minerals in highly weathering environments. However, the presence of organic acids derived from the decomposition of plant material can dissolve most oxides in acid soils. Certain metal oxides based on Fe and Mn are readily soluble under reducing conditions induced in poorly drained soils or when soils are temporarily saturated with water. Sulfides are generally not stable in well-aerated soils, due to the conversion of the sulfide anion to sulfate in the presence of oxygen. Sulfides are common in anaerobic soils where there is a source of sulfur either from plant residues or sulfate secondary minerals. Phosphates encompass a wide range of minerals with differing solubility. In neutral and basic soils, the Ca-based phosphates, known as apatites, are relatively insoluble and can remain in soils for long periods of time. In acid soils, Al- and Fe-based phosphates are considered stable, especially in agricultural soils receiving P-containing fertilizers. The names and chemical formula of some common primary and secondary minerals are given in [Table 1](#).

DISSOLUTION REACTIONS OF MINERALS

Solubility of Minerals

The stability of primary and secondary minerals in soil, and therefore their solubility, is directly dependent on their chemical structure and the nature of the weathering environment.^[2] Differences in chemical bonding among minerals influence their shape, often exposing surfaces or defects in the minerals that react more readily with water or serve as points of attack for other constituents in the soil solution. The nature of the weathering environment is expressed through the composition of the soil solution [pH, ionic strength (presence of other cations and anions), presence of ligands, reductants, oxidants, or ions that can adsorb

Table 1 Names, chemical formula, and dissolution reaction for common minerals.

Name	Chemical formula	Dissolution reaction
Carbonates		
Calcite	CaCO_3	$\text{Ca}^{2+} + \text{CO}_3^{2-}$
Dolomite	$\text{MgCa}(\text{CO}_3)_2$	$\text{Mg}^{2+} + \text{Ca}^{2+} + 2\text{CO}_3^{2-}$
Siderite	FeCO_3	$\text{Fe}^{2+} + \text{CO}_3^{2-}$
Sulfates		
Gypsum	$\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$	$\text{Ca}^{2+} + 2\text{SO}_4^{2-} + 2\text{H}_2\text{O}$
Alunite	$\text{KAl}_3(\text{SO}_4)_2(\text{OH})_6$	$\text{K}^+ + 3\text{Al}^{3+} + 2\text{SO}_4^{2-} + 6\text{OH}^-$
Phosphates		
Variscite	$\text{AlPO}_4 \cdot 2\text{H}_2\text{O}$	$\text{Al}^{3+} + \text{PO}_4^{3-} + 2\text{H}_2\text{O}$
Strengite	$\text{FePO}_4 \cdot 2\text{H}_2\text{O}$	$\text{Fe}^{3+} + \text{PO}_4^{3-} + 2\text{H}_2\text{O}$
Hydroxyapatite	$\text{Ca}_{10}(\text{OH})_2(\text{PO}_4)_6$	$10\text{Ca}^{2+} + 2\text{OH}^- + 6\text{PO}_4^{3-}$
Fluorapatite	$\text{Ca}_{10}\text{F}_2(\text{PO}_4)_6$	$10\text{Ca}^{2+} + 2\text{F}^- + 6\text{PO}_4^{3-}$
Sulfides		
Alabanite	MnS	$\text{Mn}^{2+} + \text{S}^{2-}$
Pyrite	FeS_2	$\text{Fe}^{2+} + \text{S}_2^{2-}$
Black-metacinnibar	$\beta\text{-HgS}$	$\text{Hg}^{2+} + \text{S}^{2-}$
Oxides/hydrous oxides		
Quartz	$\alpha\text{-SiO}_2^a$	H_4SiO_4
Gibbsite	$\gamma\text{-Al}(\text{OH})_3$	$\text{Al}^{3+} + 3\text{OH}^-$
Hematite	$\alpha\text{-Fe}_2\text{O}_3^a$	$\text{Fe}^{3+} + 3\text{OH}^-$
Goethite	$\alpha\text{-FeOOH}^a$	$\text{Fe}^{3+} + 3\text{OH}^-$
Silicates		
Olivine	$\text{Mg}_{1.6}\text{Fe}_{0.4}\text{SiO}_4$	$1.6\text{Mg}^{2+} + \text{Fe}^{2+} + \text{H}_4\text{SiO}_4$
Diopside	$\text{CaMg}(\text{SiO}_3)_2$	$\text{Mg}^{2+} + \text{Ca}^{2+} + 2\text{H}_4\text{SiO}_4$
Pyroxene	$\text{CaAl}_2\text{SiO}_6$	$\text{Ca}^{2+} + 2\text{Al}^{2+} + \text{H}_4\text{SiO}_4 + 2\text{H}_2\text{O}$
Albite	$\text{NaAl}_2\text{Si}_3\text{O}_8$	$\text{Na}^+ + \text{Al}^{2+} + 3\text{H}_4\text{SiO}_4$
Microcline	KAlSi_3O_8	$\text{K}^+ + \text{Al}^{3+} + 3\text{H}_4\text{SiO}_4$
Anorthite	$\text{CaAl}_2\text{Si}_2\text{O}_8$	$\text{Ca}^{2+} + 2\text{Al}^{3+} + 2\text{H}_4\text{SiO}_4$
Phyllosilicates		
Muscovite	$\text{KAl}_2(\text{AlSi}_3\text{O}_{10})(\text{OH})_2$	$\text{K}^+ + 3\text{Al}^{3+} + 3\text{H}_4\text{SiO}_4$
Kaolinite	$\text{Al}_2\text{Si}_2\text{O}_5(\text{OH})_4$	$2\text{Al}^{3+} + 2\text{H}_4\text{SiO}_4 + \text{H}_2\text{O}$

^aDissolution of Si-containing minerals often requires H_2O to form the silicate anion: H_4SiO_4 .

onto the mineral surface] and soil drainage, which influences the rate at which constituents released from the mineral surfaces are transported from the local environment.^[7] Acid (H^+) or base (OH^- ; and even water molecules) destroys mineral structures by migrating into the interior of the mineral and breaking the chemical bonds between the mineral's anions and cations. Ligands are inorganic or

organic molecules that have the ability to bind tightly with the cations in minerals and help to remove them from a mineral surface. Reductants and oxidants cause the cations in minerals to change their charge (reductants reduce charge and oxidants increase charge). When a cation within a mineral structure changes charge, it weakens the original structure allowing the attack by acids, bases, or water molecules to be more successful. Soil drainage is important in mineral dissolution because the original ionic constituents of the mineral accumulate in the surrounding soil solution, mineral dissolution will eventually stop or slow significantly. Thus primary and secondary minerals are often found in weathering environments that would favor their relatively rapid dissolution because of poor drainage in the soil.

Chemical theories derived to predict the behavior of minerals in soils and that attempt to account both for differences in chemical structure and the nature of the soil weathering environment have evolved from consideration of either macroscale or microscale processes.^[8] Theories that describe macroscale processes are based on the science of thermodynamics and allow prediction of mineral solubility.^[9] The power of this approach is that a scientist can attempt to predict the presence of a mineral without actually seeing it in the soil. The disadvantage of this approach is that it cannot easily predict the rate at which minerals will dissolve. Theories that focus on microscale processes assume specific chemical reaction mechanisms occur at mineral surfaces and are more successful in predicting the rate of mineral dissolution due to changes in the composition of the surrounding soil solution.^[10] Unfortunately, microscale processes require information that can usually only be generated under laboratory conditions, which, at best, only approximate the actual weathering environment in the soil. A knowledge of both macroscale and microscale processes is important in understanding the solubility of minerals.^[8]

Macroscale Processes

Theories to describe mineral dissolution based on macroscale processes assume that the minerals, which may be present in a soil, are only sparingly soluble in the presence of water. In other words, only a tiny fraction of the mineral dissolves at any one time. The amount that dissolves is in proportion to the chemical formula for the mineral. Chemical analysis of the soil solution in contact with the soil minerals, therefore, should yield information about the type of minerals present. For example, the dissolution reaction for the secondary mineral gibbsite can be written as follows:



where the subscript (s) refers to the solid mineral, (aq) to the ion in solution, and (l) for water. The $\rightleftharpoons$ symbol reflects the

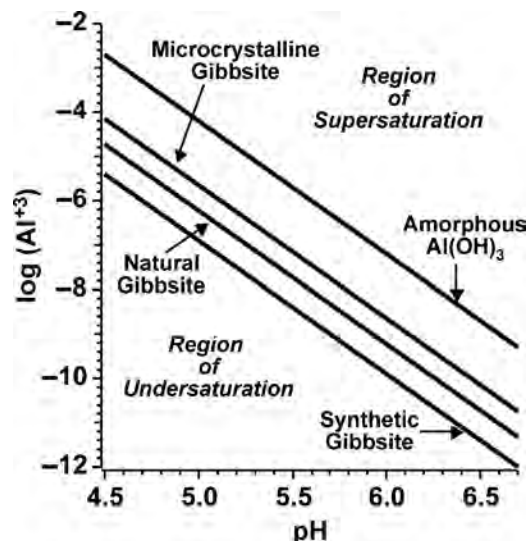


Fig. 1 Activity-ratio diagram for synthetic gibbsite, natural gibbsite, microcrystalline gibbsite, and amorphous aluminum hydroxide.

Source: K_{diss} values taken from Johnson, Driscoll, et al.^[11]

assumption that the solid gibbsite mineral is controlling the concentration of Al^{3+} ions in the soil solution. Thermodynamics predicts that if gibbsite is acting to control the composition of the soil solution, then the product of the concentrations of the dissolved ions in the soil solution is a constant (often referred to as the solubility product; K_{so}). The K_{so} of a mineral is in theory a unique quantity that is specific to the chemical composition and chemical structure of the mineral.^[9]

A visual representation of the ability to predict the presence of a mineral based on the concept of the K_{so} is shown in Fig. 1 for the mineral gibbsite, where the product of the concentrations of the dissolved ions in soil solution represented by Eq. 1 has been written in the form of a straight line as follows:

$$\log(\text{Al}^{3+}) = K' - 3\text{pH} \quad (2)$$

The term $\log(\text{Al}^{3+})$ represents the activity of Al^{3+} in solution (a measure of concentration), K' is a constant whose value is dependent on K_{so} , pH is the activity of H^+ in solution, and the number 3 represents the fact that it takes 3 H^+ ions to release one Al^{3+} ion. Experimental data points plotted on such graphs, which fall along the linear lines, are taken to indicate the presence of the mineral in the soil. If the data points fall in a region above the lines (region of supersaturation), conditions are favorable for the formation of the mineral (a precipitation reaction). If the data points fall in a region below the lines (region of undersaturation), conditions in the soil are considered too severe for the mineral to survive for long periods of time and unlikely that the mineral would be present in the soil.^[12] The four straight lines illustrate that solubility is not just dependent

on the chemical composition of a mineral, but also on its degree of crystallinity (amorphous is more soluble than synthetic) and size (microcrystalline is more soluble than more normal-sized particles).^[13]

The concept of the solubility product for minerals and the use of diagrams (often termed activity-ratio or activity-activity diagrams) as shown in Fig. 1 provide a useful means of predicting mineral behavior in soils, often by delineating boundaries for possible reactions between several minerals.^[13–15] The approach is dependent, however, on the assumption that the rate at which minerals dissolve is not limiting and thereby influencing the concentrations of ions in solution. Predicting rates of dissolution in specific soil environments is best viewed as a microscale process.

Microscale Processes

It is generally accepted that the rate of dissolution of minerals in soils is controlled by chemical reactions along mineral surfaces at distinct surface features that can include dislocations, etch pits, and micropores.^[16] The presence of surface features such as dislocations, etch pits, and micropores is taken as an indication of a population of surface reactive sites where dissolution reactions are favored above those possible along the total surface area of a mineral.^[8] The dissolution reaction at the surface reactive sites is considered as a two-step process. The first step involves the relatively rapid formation of a surface species as the result of the reaction with entities in soil solution such as H^+ , OH^- , H_2O molecules, and inorganic and organic ligands. The second step, and the one that is considered rate limiting, involves the transformation of the surface species into an activated complex with the eventual release of constituent ions to the bulk solution.^[7] Dissolution at the microscale, therefore, can be considered as a function of the reactive surface area of the mineral, the nature of the inherent chemical bonding structure that comprises the mineral, and the presence of various chemical species in the soil solution. Thus the dissolution rate of a mineral is not a fixed numerical quantity but dependent upon a number of variables which may change with time due to changes in the physical size of the mineral itself, as well as changes in the surrounding soil solution.^[8] It is possible to calculate approximate rates of dissolution for various classes of primary and secondary minerals, but local conditions within the soil should also be considered when predicting mineral solubility.

REFERENCES

1. S374 Glossary of Soil Science Terms Committee. *Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 1997; 100 pp.
2. Allen, B.L.; Hajek, B.F. Mineral occurrence in soil environments. In *Minerals in Soil Environments*; Dixon, J.B.,

- Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America Book Series Number 1; Soil Science Society of America: Madison, 1989; 199–278.
3. Bragg, L.; Claringbull, G.F.; Taylor, W.H. *Crystal Structures of Minerals: The Crystalline State*; Cornell University Press: Ithaca, 1965; Vol. IV, 409 pp.
 4. White, A.F. Chemical weathering rates of silicate minerals in soils. In *Chemical Weathering Rates of Silicate Minerals*; White, A.F., Brantley, S.L., Eds.; Reviews in Mineralogy; The Mineralogical Society of America: Washington, 1995; Vol. 31, 407–461.
 5. Doner, H.E.; Lynn, W.C. Carbonate, halide, sulfate, and sulfide minerals. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America Book Series Number 1; Soil Science Society of America: Madison, 1989; 279–330.
 6. Lindsay, W.L.; Vlek, P.L.G.; Chien, S.H. Phosphate minerals. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America Book Series; Soil Science Society of America: Madison, 1989; 1089–1130.
 7. Robarge, W.P. Precipitation/dissolution reactions in soils. In *Soil Physical Chemistry*; Sparks, D.L., Ed.; 2nd Ed.; CRC Press: Boca Raton, 1999; 193–238.
 8. Lasaga, A.C. Fundamental approaches in describing mineral dissolution and precipitation reactions. In *Chemical Weathering Rates of Silicate Minerals*; White, A.F., Brantley, S.L., Eds.; Reviews in Mineralogy; The Mineralogical Society of America: Washington, 1995; Vol. 31, 23–86.
 9. Sposito, G. *The Thermodynamics of Soil Solutions*; Oxford University Press: London, 1981; 223 pp.
 10. Lasaga, A.C. *Kinetic Theory in the Earth Sciences*; Princeton University Press: Princeton, 1998; 811.
 11. Johnson, N.M.; Driscoll, C.T.; Eaton, J.S.; Likens, G.E.; McDowell, W.H. 'Acid rain,' dissolved aluminum and chemical weathering at the Hubbard Brook experimental forest, New Hampshire. *Geochim. Cosmochim. Acta* **1981**, 45 (9), 1421–1437.
 12. Lindsay, W.L. Chemical equilibria. In *Soils*; John Wiley & Sons: New York, 1979; 449 pp.
 13. Dove, P.M. Kinetic and thermodynamic controls on silica reactivity in weathering environments. In *Chemical Weathering Rates of Silicate Minerals*; White, A.F., Brantley, S.L., Eds.; Reviews in Mineralogy; The Mineralogical Society of America: Washington, 1995; Vol. 31, 235–290.
 14. Garrels, R.M.; Christ, C.L. *Solutions, Minerals and Equilibria*; Harper and Row: New York, 1965; 450 pp.
 15. Stumm, W.; Morgan, J.J. *Aquatic Chemistry: Chemical Equilibria and Rates in Natural Waters*, 3rd Ed.; John Wiley & Sons, Inc.: New York, 1996; 1022 pp.
 16. White, A.F.; Brantley, S.L. Chemical weathering rates of silicate minerals: an overview. In *Chemical Weathering Rates of Silicate Minerals*; White, A.F., Brantley, S.L., Eds.; Reviews in Mineralogy; The Mineralogical Society of America: Washington, 1995; Vol. 31, 1–22.

Minirhizotron

Verónica Muñoz-Romero

Luis López-Bellido

Department of Agriculture and Forestry Science and Resources, University of Cordoba, Cordoba, Spain

F. Javier López-Bellido

University of Castilla–La Mancha, Ciudad Real, Spain

Rafael J. López-Bellido

Department of Agroforestry, University of Huelva, Huelva, Spain

Abstract

Minirhizotron is a useful technique to study root system dynamics by means of a transparent tube and a root image acquisition device. It is non-destructive and less time-consuming; has high sensibility to detect fine roots (down to 0.09 mm diameter); and tracks the appearance, growth, and disappearance of individual roots over time. However, its use is limited to some soil types, and the measurements of root density in the top layer soil obtained using this technique are often found to be underestimations.

INTRODUCTION

Traditionally, root extraction methods have been used at the different crop growth stages to characterize the root system. These methods are based on the collection of unaltered soil samples with a probe of a known volume, separation of the soil from the roots by washing, and digitalization of the roots to measure their length and surface area per unit volume using a scanner and root analysis software. The root surface area can also be measured using leaf area meters. A more basic option consists of counting the roots in a cross-sectional plane of the sample. The manual extraction of the roots from soil samples is, without a doubt, the most exact method. However, its disadvantages are as follows: 1) it is a destructive technique; 2) there is no effective distinction between live and dead roots; 3) fine roots can be lost depending on the opening of the mesh used; 4) the almost transparent nature of many roots prevents the analyst from recovering them; 5) normally, samples are taken from only one crop growth stage (The study of root dynamics would require samples to be obtained during all growth stages.); and 6) it is a laborious technique and requires a significant amount of time *in situ* as well as in the laboratory.

Continuous non-destructive assessment of root growth has been practiced under field conditions for decades. This involves observation of root growth against glass or rigid plastic barriers. The oldest systems consisted of barriers as simple as a vertical glass pane buried in the soil with a big hole or trench alongside, so that an observer can see the roots growing against the glass. Rhizotron is another type of permanent system designed for examining the root

growth of a plant; it contains enclosed columns of soil with transparent plastic windows that permit viewing, measuring, and photographing. Owing to its smaller size, rhizotron can be used in replicated experiments. Great advances have been made in another non-destructive system for the study of roots called minirhizotron.

WHAT IS A MINIRHIZOTRON?

Minirhizotron is a transparent tube that is inserted into the soil permanently into which an image capturing system is introduced (camera, video system, or a portable cylindrical scanner) which is connected to a computer via an image capturing software, thus allowing the roots intercepted by the wall of the tube to be observed (Fig. 1). Subsequently in the laboratory, the images obtained are analyzed using a computer equipped with specific software to obtain the root parameters.

TUBE CHARACTERISTICS AND INSTALLATION

Normally, minirhizotron is cylindrical and rigid and can be made of different materials (Plexiglas, polycarbonate, glass, etc.). Tube type can affect the root growth pattern within a species.^[1] Although the cost of the minirhizotron equipment is quite high, the placement of multiple tubes allows the cost to be reduced as compared to rhizotrons. Minirhizotron tubes are sealed or covered to exclude moisture. Typically, the above-ground portion of the minirhizotron is completely opaque and painted white. The upper part of the tube is

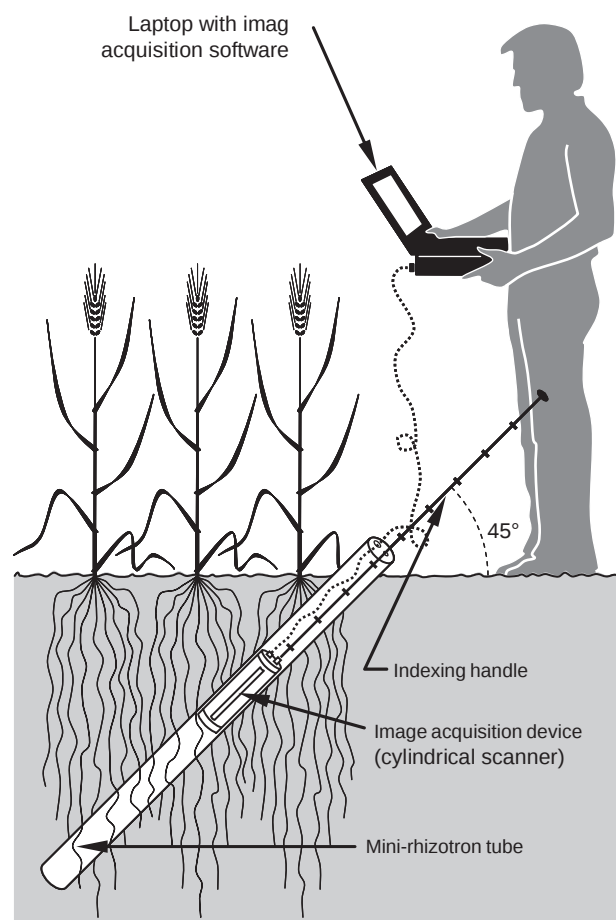


Fig. 1 Diagrammatic representation of minirhizotron with cylindrical scanner as image acquisition device.

covered with an opaque cap. These precautions are taken to reduce thermal fluctuations within the tube and exclude light, which can affect rooting along the tube.

Minirhizotrons have been installed in forests, orchards, grasslands, deserts, agricultural systems, controlled environment chambers, and greenhouse lysimeters.^[2] The tube installation procedure is crucial to guarantee a correct measurement. When installing the minirhizotron tube, disturbances around it, such as soil compaction or air gaps between the tube and the soil that affect the root growth, creating conditions around the tube that are not the same as those found in the surrounding soil, need to be minimized. A common approach is to use an auger or a corer to excavate a hole with the same diameter as that of the minirhizotron tube to the desired depth. The minirhizotron tube is then inserted into the hole. Another method, namely, two-step tube installation, can also be used.^[3] First, a spiral-slip auger, with a diameter somewhat less than the tube diameter, is used to remove the majority of the soil. Then the hole is cut to its final diameter using a reverse taper bit. As a final step before tube insertion, a fine circular wire brush is used to remove soil striations that may have been incurred during the boring procedure.

The soil type will affect the tube installation method used. The abovementioned methods work well for medium textured (sandy loam) soils. Heavy textured soils, soils with abundant rocks and cobbles, soils with argillic horizons, or loose structureless soils are more problematic. The time of the year can also affect tube installation. Dry clay soils offer a great deal of resistance to coring, while wet clay soils can smear the outside surface of the minirhizotron tubes reducing visibility. When clay soils dry, they may develop cracks, which, on the one hand, separate the soil from the tube and, on the other hand, allow light to penetrate the soil.

The fixed tube of the minirhizotron can be inserted into the soil at angles ranging from 0° (horizontal) to 90° (perpendicular). Image acquisition device is calibrated according to this angle. The orientation of the tube at 0° allows many images to be obtained at the same depth, while slanted tubes provide greater information about the distribution of the roots throughout the profile and, at the same time, are simpler to install. The vertical placement of the tube can instill in the roots a tendency to grow downward lengthwise along the tube. The most commonly used angles are 30° and 45° with the vertical.

Once the tube has been placed, a waiting period must be allowed before proceeding to the capture of images. This period is called the return to equilibrium. After this time, the images are taken *in situ* with a predetermined frequency depending on the species studied.

IMAGE ACQUISITION DEVICE

The still camera, video camera, or cylindrical scanner is connected to an indexing handle that allows the system to move rapidly and accurately from one field of view to the next (depths) and provides a precise method for rapidly collecting images at the same location over a period of time.^[4] The existing tendency is for these devices to be connected to a laptop computer, which, being more easily organized, verifies whether the images were correctly taken and allows overlaying them in the field to ensure that the same locations were photographed. The laptop computers are specifically suitable for field work, as the high light intensity does not allow conventional computers to be used. The use of these devices requires the use of batteries, which can be problematic when a large number of images are to be taken. When working with a miniature video camera inside the tube, it is necessary to take into account that it requires a lighting source, and cameras have been fitted with incandescent or ultraviolet (UV) lights. The use of incandescent or UV light, however, generates heat within the tubes, which would be transferred to the adjacent soil. The heat generated by minirhizotron lighting systems may alter root growth and development more at depth than at the surface, especially when soil temperatures are lower.^[5]

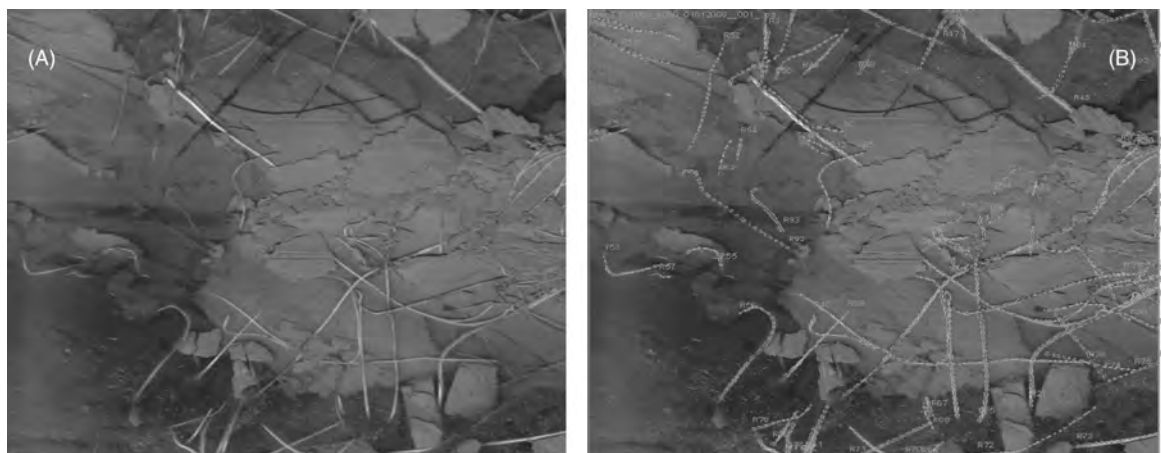


Fig. 2 Root image acquired by a minirhizotron. (A) Original image and (B) image after being analyzed with a root image analysis software. Root length, diameter, area, and volume are measured using a series of straight length lines called segments as illustrated.

IMAGE ANALYSIS

Normally, image analysis is carried out completely in the laboratory, manually using one of the existing image analysis software packages (WinRHIZO Tron & Tron MF, ARCOS, etc.). The operator performs all aspects of the analysis by hand, i.e., opening and aligning the images through tracing root length and diameter in order to assign root condition codes (Fig. 2). When more than one person is involved in the analysis of the images, errors can occur in the measurements as observation capacity varies from one person to another. Automated techniques for minirhizotron image processing have increased the speed of root data collection, as well as the number and size of the experiments that can reasonably be analyzed with a single minirhizotron.^[6] These techniques are in an initial stage of development and must be improved. For the study of roots, color images provide better results than black-and-white images. However, black-and-white images require less memory capacity. Background homogeneity and color are major determinants of quality as well as the behavior of a minirhizotron toward light and the uniformity of lighting. The best fit has been found in clay soil, owing to the greater homogeneity of its background.

The direct measurements that can be obtained from the processing of images from the minirhizotron are root numbers, length, area, volume, and diameter. From these measurements, the root length density is calculated, and studies are performed on the production and turnover (mortality) of fine roots over time, thus obtaining their life span. In addition, the minirhizotron allows the determination of root distribution throughout the soil profile. Root biomass also can be calculated with the minirhizotron, which is estimated using a previous calibration with a manual soil core extraction method in the same soil and for the same species.

The most important question raised about the use of minirhizotrons is whether the estimations are comparable

to those made with soil core extraction methods. The results obtained in different studies have shown how in some cases there is a very good match. In others cases, an underestimation is observed, and on occasion, there is no correlation whatsoever (Fig. 3). The main problem in obtaining a good correlation is experienced in the top layer soil (0–10 cm). It is for this reason that the top layer data are at times not included, so as to obtain a correlation with regard to the rest of the depths (Fig. 3). The problem arises because the top layer of the soil is most disturbed when the tube is inserted

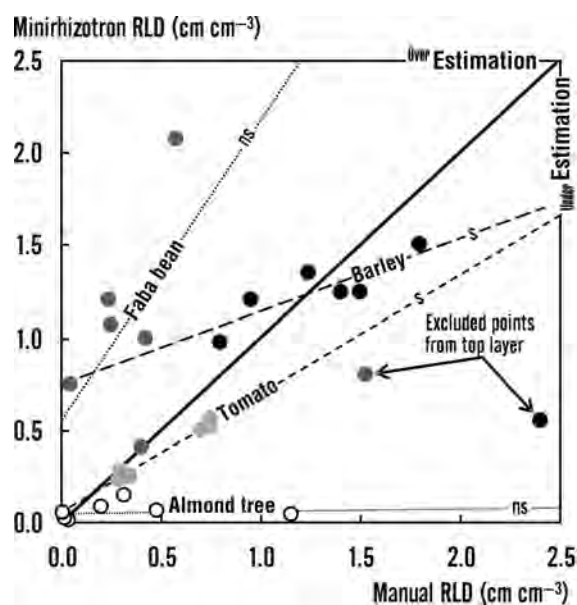


Fig. 3 Comparison of root length density (RLD) obtained by manual root extraction method and by minirhizotron for four crops (ns, not significant; s, significant).

Source: Adapted from Franco & Abrisqueta,^[3] Machado & Oliveira,^[7] and Heeraman & Juma.^[8]

and it is where the worst root contact exists with the tube, which is influenced by the type of soil and its water content.

CONCLUSION

Minirhizotron, in a right soil, offers an opportunity to take non-destructive root samples over different growth stages, with a low consumption of time and high sensibility to detect fine roots (down to 0.09 mm diameter). This device makes it possible to study the appearance, growth, and disappearance of individual roots over time. Its main disadvantage is that the results obtained are not in agreement with those from the manual extraction method if the top layer data are considered.^[7,9]

REFERENCES

1. Withington, J.M.; Elkin, A.D.; Buaj, B.; Olesinski, J.; Tracy, K.N.; Bouma, T.J.; Oleksyn, J.; Anderson, L.J.; Modrzynski, J.; Reich, P.B.; Eissenstat, D.M. The impact of material used for minirhizotron tubes for root research. *New Phytol.* **2003**, *160* (3), 533–544.
2. Johnson, M.G.; Tingey, D.T.; Phillips, D.L.; Storm, M.J. Advancing fine root research with minirhizotrons. *Environ. Exp. Bot.* **2001**, *45* (3), 263–289.
3. Franco, J.A.; Abrisqueta, J.M. A comparison between minirhizotron and soil coring methods of estimating root distribution in young almond trees under trickle irrigation. *J. Hortic. Sci. Biotechnol.* **1997**, *72* (5), 797–805.
4. Johnson, M.G.; Meyer, P.F. Mechanical advancing handle that simplifies minirhizotron camera registration and image collection. *J. Environ. Qual.* **1998**, *27* (3), 710–714.
5. Van Rees, K.C.J. Soil temperature effects from minirhizotron lighting systems. *Plant Soil* **1998**, *200* (1), 113–118.
6. Zeng, G.; Birchfield, S.T.; Wells, C.E. Detecting and measuring fine roots in minirhizotron images using matched filtering and local entropy thresholding. *Mach. Vision. Appl.* **2006**, *17* (4), 265–278.
7. Machado, R.M.A.; Oliveira, M.D.G.O. Comparison of tomato root distributions by minirhizotron and destructive sampling. *Plant Soil* **2003**, *255* (1), 375–385.
8. Heeraman, D.A.; Juma, N.G. A comparison of minirhizotron, core and monolith methods for quantifying barley (*Hordeum vulgare* L.) and faba bean (*Vicia faba* L.) root distribution. *Plant Soil* **1993**, *148* (1), 29–41.
9. Ephrath, J.E.; Silberbush, M.; Berliner, P.R. Calibration of minirhizotron readings against root length density data obtained from soil cores. *Plant Soil* **1999**, *209* (2), 201–208.

Mites

Hans Klompen

Museum of Biological Diversity, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Mites (Acari) form the most diverse lineage of Chelicerata. The basal division within Acari is between the orders, Parasitiformes and Acariformes. The former includes the Ixodida (ticks) and the Mesostigmata, a highly diverse lineage that is well-represented in soils. These mites tend to be relatively large, often with very discrete shields.

INTRODUCTION

Mites (Acari) form the most diverse lineage of Chelicerata. They are generally small (0.25–1 mm for most taxa) and are characterized morphologically by the division of their bodies. Functionally, there are two parts: one is the gnathosoma, containing the mouth, chelicera, and pedipalps and the other is the idiosoma, the body proper, containing the legs, eyes (if present), brain, and all remaining body systems. The nymphal and adult instars generally carry the four pairs of walking legs characteristic for Chelicerata, but the larval (and prelarval) instars carry only three pairs. An excellent review of mite morphology can be found in Evans.^[1]

The basal division within Acari is between the orders Parasitiformes (= Anactinotrichida) and Acariformes (= Actinotrichida). The former includes the Ixodida (ticks) and the Mesostigmata, a highly diverse lineage that is well-represented in soils. These mites tend to be relatively large, often with very discrete shields. Some Mesostigmata are fungal or pollen feeders but mostly retain the ancestral, predaceous lifestyle. In terms of their impact on soil ecosystems, a basic distinction between Uropodina (slow and often fungivorous) and Gamasina (active and mostly predaceous) is important.

The bulk of mite diversity is in the Acariformes, with at least four rather distinct taxa in soil environments: Astigmata, oribatid mites, “Endeostigmata,” and Prostigmata. Astigmata are often associated with ephemeral habitats. They have three nymphal instars, the second of which, the deutonymph, is highly modified for dispersal. It has no mouth, is well sclerotized, and generally carries an attachment organ (sucker plate) on the posterior venter. The other instars are often poorly sclerotized and can feed. The main group of interest for soil biologists are the oribatid mites. Like Astigmata, they are solid feeders, with generally well-developed chelate (biting) chelicera. However, oribatid mites are usually well sclerotized in the adults, can live for a long time (1–3 years of life cycles

are not unusual in temperate regions), have generally low reproductive rates, and disperse poorly.^[2] In contrast, Astigmata disperse very well, tend to have very high reproductive rates and live for much shorter periods of time.^[3]

The “Endeostigmata” form a paraphyletic assemblage of early derivative Acariformes. Poorly sclerotized and with diverse feeding behaviors, from fungivory to predation, these mites are particularly abundant in relatively poor quality habitats, including the deep soil. The final lineage of Acariformes is the Prostigmata sp. This group is enormously diverse in both morphology and life history. In studies of soil ecosystems, it tends to be ignored due to the difficulties in identifying these often very small and poorly sclerotized mites. Yet identification to at least major groups is desirable, as the main lineages, Anystina (including Parasitengona), Eupodina, Raphignathae, and Heterostigmae, are quite different in morphology, life history, and (most probably) ecosystem function. All of these mites are fluid feeders, with chelicera modified as stylets. Prostigmata include many fungivorous and predaceous taxa and a few true plant feeders.

Comprehensive keys to the species of soil inhabiting mites have been published for the former Soviet Union.^[4] Keys to the families (“Endeostigmata,” Prostigmata, oribatid mites) and genera (Mesostigmata, Astigmata) for North America can be found by Dindal.^[5] Comprehensive keys for soil mites in tropical areas are not yet available.

DISTRIBUTION AND ABUNDANCE

Mites are the most abundant group of arthropods within most soil and litter habitats, including deserts and arctic areas. Numbers of 10,000–500,000/m² are normal in temperate and tropical sites.^[6] Mesostigmata usually form about 20% of the total mite fauna (3000–4000/m²),^[6] with abundance and diversity generally being higher in forests

than in more open habitats. In temperate regions, the number of Uropodina is generally lower than that of Gamasina, but in the tropics Uropodina abundance can rival that of oribatid mites. Overall, Uropodina is strongly associated with soils of high organic matter content.

Astigmata have often been ignored in studies on soil fauna but can be common in disturbed sites, such as soil around isolated city streets, and cultivated fields.^[3,5] A similar association with disturbance is found for most Heterostigmae (Prostigmata). The contrast with oribatid mites is once again strong. Oribatid mites are generally very common in relatively stable habitats (10,000–300,000/m²)^[6] and usually form the bulk of mite biomass in forest ecosystems. However, many oribatid mites are quite sensitive to soil disturbance, and the fauna of conventionally tilled agricultural fields is generally depauperate. Unlike most other groups of mites, they cannot take advantage of the resource flushes that are characteristic of many agricultural practices (e.g., manure applications). Overall, the group is highly diverse, and most of the 150+ families described do occur in soil and litter habitats.

The “Endeostigmata” are not very diverse, although many species have a broad geographic range. Among the 126+ described families of Prostigmata, about 50 occur mostly in soil environments, but representatives of many other families may be encountered during sampling.^[7] The number of Prostigmatid and Endeostigmatid mites (these groups are often combined) varies dramatically by general habitat. Numbers vary from 7500/m² in moss turf banks in Antarctica to over 100,000/m² in pine forests in Northern Europe.^[7] Prostigmata plus Endeostigmata may form 40–50% of the total number of mites in deserts and grassland ecosystems, but this proportion drops sharply in temperate forests (less than 10%). The relative proportion of Prostigmata (+ Endeostigmata) to oribatid mites is occasionally reported, but this number may not be very informative. The different lineages in Prostigmata are sufficiently diverse; to lump them together into a single value may not be useful.

FUNCTION

The role of mites in soil ecosystems is only partially understood. It is clear that their role in overall energy flow is limited,^[6] and most research is concentrating more on indirect effects. These include fragmentation of litter (comminution) and soil formation, nutrient cycling, dispersal of microbial spores, and stimulation of the microflora (bacteria and fungi) by grazing. A second level of indirect regulation results from predation on both other microarthropods and nematodes. It is useful to distinguish the two: the first group of processes mostly impact the fungal decomposition pathway (few

mites feed directly on bacteria), while the second influences both the fungal and bacterial pathways.^[8] Most importantly from the standpoint of soil ecosystems, different groups of mites have distinctly different functions, and broad generalizations are nearly always obscuring, rather than illuminating.^[9]

Comminution, the breaking up of larger organic units such as dead leaves and wood into smaller pieces, is a process that is largely limited to oribatid mites and Astigmata. The other lineages are either predaceous, or feed by puncturing, not breaking up, fungal hyphae etc. Comminution greatly increases the surface areas on which bacteria can complete the actual process of decomposition. Through their feeding and production of fecal pellets, oribatid mites can alter the structure of the soil. As such, they are considered as the most important group of arachnids from the standpoint of direct or indirect effects on the formation and maintenance of soil structure.^[2,8] Many oribatid mites also sequester calcium and other minerals in their cuticle and thus their bodies may form important “sinks” for nutrients in some nutrient limited environments.

Oribatid soil mites disperse bacteria and fungi, both externally, on their body surface, and internally, by excreting undigested spores. Through this process they can enhance endomycorrhizal colonization.^[10] Such dispersal may be especially important in the early stages of saprophytic succession.^[11] Mites grazing on fungi influences both the composition and productivity of the microflora.^[11] What is poorly understood are the effects of these actions on decomposition rates.^[8,11] Compensatory growth of fungi in response to periodic grazing by mites and Collembola appears to be very important in the field and may significantly increase decomposition rates.^[11] In addition, comminution of litter and the disturbance of fungi by mite feeding may favor bacterial over fungal growth.

Mites are among the most important predators in the soil ecosystem, attacking most other components of the mesofauna. The two main taxa involved in this process are gamasine Mesostigmata and Prostigmata, but nematode feeding can be found in all taxa discussed.^[3] Strict specificity for microarthropods (most common among Prostigmata) or nematodes (some “Endeostigmata” and Mesostigmata) occurs, but the majority of mites are generalist predators, feeding on a variety of prey. So far, relatively little is known about the effect of arthropod predation in soil communities. In laboratory experiments (microcosms), predatory mites can suppress nematode numbers,^[12] but these results have not been translated to more realistic field conditions. In a broader sense, predation may alleviate grazing pressure by nematodes and oribatid mites on the primary decomposers, fungi and bacteria. Excessive grazing decreases, rather than increases, soil respiration and decomposition rates.

REFERENCES

1. Evans, G.O. *Principles of Acarology*; University Press: Cambridge, 1992; 563 pp.
2. Norton, R.A. Aspects of the biology and systematics of soil arachnids, particularly saprophagous and mycophagous mites. *Quaestiones Entomologicae* **1985**, *21*, 523–541.
3. Walter, D.E.; Proctor, H.C. *Mites: Ecology, Evolution and Behaviour*; CABI Publishing: New York, 1999; 322 pp.
4. Ghilyarov, M.S.; Bregatova, N.G., Eds. *A Key to the Soil-Inhabiting Mites Mesostigmata*; Zoologicheskogo Instituta Akademii Nauk SSSR: Leningrad, 1977; 1–718.
5. Dindal, D.L., Ed. *Soil Biology Guide*; John Wiley & Sons: New York, 1990; 1349 pp.
6. Petersen, H.; Luxton, M. A comparative analysis of soil faunal populations and their role in decomposition processes. *Oikos* **1982**, *39*, 288–388.
7. Kethley, J. Acarina: Prostigmata (actinedida). In *Soil Biology Guide*; Dindal, D.L., Ed.; John Wiley & Sons: New York, 1990; 667–756.
8. Moore, J.C.; Walter, D.E.; Hunt, H.W. Arthropod regulation of micro- and mesobiota in below-ground detrital foodwebs. *Annu. Rev. Entomol.* **1988**, *33*, 419–439.
9. Walter, D.E.; Ikonen, E.K. Species, guilds, and functional groups: Taxonomy and behavior in nematophagous arthropods. *J. Nematol.* **1989**, *21*, 315–327.
10. Klironomos, J.N.; Kendrick, W.B. Stimulative effects of arthropods on endomycorrhizas of sugar maple in the presence of decaying litter. *Funct. Ecol.* **1995**, *9*, 528–536.
11. Lussenhop, J. Mechanisms of microarthropod-microbial interactions in soil. *Adv. Ecol. Res.* **1992**, *23*, 1–33.
12. Laakso, J.; Setälä, H. Population- and ecosystem-level effects of predation on microbial-feeding nematodes. *Oecologia* **1999**, *120*, 279–286.

Modern Civilization and Soils

Kenneth R. Olson

*Department of Natural Resources and Environmental Sciences,
University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.*

Abstract

Contemporary use of soils is not greatly different from ancient use of soils. Many millions of hectares in the modern urban landscape have been altered or destroyed for construction of buildings, transportation, or mineral extractions. The archeological record shows that some ancient people abused their soils and that their civilizations were disrupted by the ecological and environmental consequences. One of history's greatest problems has been the cultivation and erosion of sloping lands. Repeated productivity losses on these soils would result in long-term costs to society, because this natural resource (capital) would eventually be depleted. Sustaining the long-term productivity and quality of all soil resources needs to be a national priority, which is essential to our nation's prosperity. It is in the national interest that management strategies and techniques continue to be developed to monitor and sustain soil productivity and quality.

INTRODUCTION

In many parts of the world, contemporary use of soils is not greatly different from ancient use of soils.^[1] The archeological record shows that some ancient people abused their soils and that their civilizations were disrupted by the ecological and environmental consequences. Some ancient populations, such as the Middle Mississippian culture near Cahokia, Illinois, were larger in population in 1200 A.D. than their modern descendants.^[2] The archeological record gives us reasons to ponder, whether contemporary societies are using their resources wisely.^[1] People should seek explanations for ancient population declines and shifts so they might avoid similar fates, because an important factor in the continuing prosperity of a nation is the nature of the husbandry of the soil and land resources through centuries of use and occupation.

The Middle Mississippian Indians settled in an area near Cahokia, in southwestern Illinois (adjacent to the current city of St. Louis, Missouri) in approximately 500 A.D.^[3–5] By 1350 A.D., the site was abandoned after peaking in the Lohmann phase (1050–1100 A.D.) and the Stirling phase (1100–1200 A.D.).^[2,5–7] During the later years of settlement, the population grew to 20,000 people, making it the largest city in the United States until the European settlement of Philadelphia and Pennsylvania exceeded 20,000 people in 1830. The Middle Mississippian people settled on the stream and lacustrine deposits in the American Bottoms of the Mississippi River valley. Previously, it would not have been possible to feed such a large population by hunting and fishing alone. The Middle Mississippian Indians

supplemented their diet by cultivating maize (corn) and garden crops on the silty soils (Fluvaquents) on floodplains.^[8] They could then store sufficient food for the winter season to feed the entire population. The people also cut down the adjacent forests for firewood and making structures and fences. This deforestation caused accelerated erosion of the steep, silty soils (Hapludalfs) on nearby hillsides. As a result, wood had to be transported to greater and greater distances over time. These Middle Mississippian Indians were also earthen mound builders. They created thousands of earthen burial and ceremonial (platform) mounds. The earthen mounds were built on clayey soils (Fluvaquents). They dug out the silty, clayey, and sandy stream deposits from the Mississippi River valley and carried the soil materials on their backs in sacks to the top of the mounds.^[2] Later, the original borrow areas were filled to build a large plaza by digging and transporting more soil materials in sacks from borrow areas farther away.

Some of these mounds (almost 800 years old) exist. The largest mound is called Monks Mound, which is 30 m high with a rectangular base of approximately 320 m × 294 m. War, fire, food shortages, and disease were the most likely causes for the collapse of the Middle Mississippian civilization, but the final causes for the collapse are the subject of considerable debate. It is difficult to understand how the wastes from 20,000 people could be disposed off without polluting the adjacent water and land. Waste disposal problems could have contributed to the contamination of the water supplies, especially the water in the borrow areas. Over time, the demand for maize without crop rotation and nutrient additions could have resulted in soil depletion and

reduced crop yields, which could have contributed to food shortages. Deforestation of forests, overgrazing of pastures and ranges, and overplowing of fields may have accelerated erosion in many ancient civilizations. Accelerated soil erosion may have diminished the ability of many farmers worldwide to produce satisfactory yields and maintain a reasonable standard of living.^[9] Tillage utilized in agriculture often accelerates soil loss from water and wind erosion.^[10] For survival and well-being, governments throughout the world need to respond to the challenges of the stabilizing erodible soils.

MODERN CIVILIZATIONS

World demand for food climbs higher each year as a result of both population growth and rising expectations. As a nation’s standard of living increases, diets often contain more of meat. This change may entail growing of feed crops, such as corn and soybeans, which lead to more erosion than of small grains and forage.^[11] In the Third World, farmers are being pushed onto steeply sloping, erodible lands that are rapidly losing their topsoil. Even in the American Midwest, many farmers have abandoned more biologically diverse and soil-conserving rotations that include forage and small grains (oats and wheat) in favor of continuous corn, or a corn and soybean rotation. In other parts of the world, farming has extended into semiarid regions where land is vulnerable to soil loss from wind erosion when plowed.^[12]

The loss of topsoil affects the ability of soil to grow food crops, by decreasing the availability of nutrients. This results in degrading the soil’s physical condition by increasing the need for fertilizers and irrigation, which, in turn, increases the cost of production. The loss of topsoil is a quiet crisis in the world economy, because unlike natural disasters such as earthquakes or volcanic

eruptions, it is largely a human-made crisis that unfolds gradually.^[11]

Soil degradation is more severe in sub-Saharan Africa than other tropics.^[12] This is where soil degradation has severely reduced the soil’s potential capacity to produce food, feed, and fiber. This is a result of physical, biological, and chemical deterioration due to erosion, desertification, laterization (formation of a stone-like layer high in iron), salt accumulation, excessive leaching and acidification, and nutrient imbalance.

Prior to the European settlement, many north-central U.S. soils were developed under prairie grass on nearly level plains and under forests on rolling hillslopes. European settlement and agricultural technology had significant impacts on north-central U.S. soils [Table 1](#).

During the Dust Bowl of the 1930s, severe droughts in the Great Plains of the United States resulted in crop failures and major dust storms. In 1935, with dust settling in the congressional offices in Washington, District of Columbia, the Congress passed a law, creating the Soil Conservation Service to address the erosion problem. At that time, soil erosion of row cropland became so severe, and farm debt became so high, that lands were abandoned in many states such as Illinois, Indiana, Kentucky, Tennessee, Oklahoma, and Missouri. Numerous research stations and demonstration farms were established in the north-central United States to test and disseminate information on how to restore soils and improve crop yields and farm income on hilly, highly eroded soils. Management techniques that helped achieve these goals included the removal, crushing, and application of limestone from nearby hills to highly eroded soils with acidic subsoils. Rock phosphate was added to the phosphorus-poor soils and forage crops were seeded as pasture for beef and sheep, reducing the need for a yearly tillage. Gullies were filled on the steep hillslopes as well.

In the 1950s, nitrogen fertilizer became available at low cost and the increasing acreage planted with soybeans

Table 1 Agricultural technology impacts of European settlement on north-central United States.

Timeline	Agricultural technology events related to European settlement
1814	The introduction of the cast iron plow made it possible to plow the forest land but not the wet, sticky prairie soils. Consequently, settlement in the early 1800s along the Ohio, Mississippi, and Missouri Rivers resulted in the clearing of forest and cultivation of sloping land which was easily eroded
1837	The John Deere Company started mass producing a steel plow which could be used to plow the prairie lands. Shortly thereafter, some of the agriculture from the highly eroded forest soils shifted or moved to the prairie soils
1850s	The steam engine was introduced which helped with the wheat harvest and increased wheat production. In 1879 the U.S. Drainage Act and Levee Act allowed the drainage (initially surface ditches and later tile systems) and farming of swampy prairie areas in states like Illinois
1890–1950	Gasoline tractors replaced horses resulting in a decline in the acreages of oats and hay
1920s	Agronomic research resulted in better oil seed varieties of soybean. Crop rotations which had previously included corn planted in rows approximately 1 m wide, drilled wheat and oats in very narrow rows approximately 7.5 cm wide, and perennial hay was replaced by an intensive rotation of corn and soybeans. Over time, the soybeans were grown in wide rows for grain instead of being closely grown for forage; consequently, accelerated soil erosion occurred

and corn increased erosion. In the 1960s, the moldboard plow was partially replaced with a chisel plow, which reduced soil loss by leaving plant residue on the surface (conservation or mulch tillage). In the 1970s, no-till planting of limited acreages of corn and soybean further reduced tillage and soil erosion, but often with increased use of herbicides. In the 1980s, ridge tillage, which left residues between the rows at planting, to reduce erosion, became more popular. Row cultivation was used to control weeds between the rows and to maintain ridges.

The U.S. Food Security Act of 1985 resulted in millions of hectares of highly erodible lands being put into forages or trees for 10 years or more, which further helped reduce soil movement from the row cropland. Many states developed conservation programs in the 1980s that required soil loss to be reduced over a specific time, to the tolerable soil loss level, ranging from 2 to 11 metric tons per hectare per year.

EFFECTS OF AGRICULTURE ON SOILS

Sheet and rill erosion rates on cropland have been declining in United States.^[13] Under the production of agricultural systems, however, 23% of U.S. cropland is eroding at rates higher than that of the tolerable soil loss.^[14,15] The effects of erosion on cropland soils depend, in part, on the characteristics of the soils, and the landscapes in which they occur. Even under favorable climatic conditions, topography, and parent materials, soil formation is a slow process, requiring hundreds of years to develop topsoils and subsoils. Adopted conservation management improvements, advances in conservation tillage technology, and land use changes have contributed to sheet and rill erosion-rate reductions; however, there can be a net soil loss and reduced long-term soil productivity even under no-till systems. When considered in the context of long-term productivity, erosion rates alone are not the only indicator of soil degradation. Degradation of soil structure and aggregation or loss of irreplaceable soil attributes are much more serious in certain soils than others when compared at the same erosion rates. Documented on-farm and off-site damages include loss of soil, sedimentation in streams, reservoirs, and fields, loss of nutrients, soil property changes, soil productivity losses, and air, soil, and water pollution.

Government programs either provide incentives or require soil loss to be held within tolerable levels. On average, the total soil loss resulting from wind and water activity has been declining, in part, as a consequence of government programs. However, erosion and sediment deposition are locally and regionally significant and, thus, perceived by the scientific and non-scientific communities as a serious threat to our economy and environment.

EFFECTS OF MODERN URBANIZATION AND INDUSTRIALIZATION ON SOILS

For the past 200 years, as European settlement increased in the United States, the disturbance and alteration of the soils have accelerated. Many millions of hectares in the urban landscape have been altered or destroyed for constructing buildings, transportation, or mineral extractions. Vast areas of soils near urban centers have been excavated or disturbed for the construction of buildings, roads, railroads, cemeteries, canals, utility lines, and landfills. In addition, there has been surface and subsurface mining for various minerals, including coal, limestone, clay, sand, and gravel apart from waste disposal. Topsoils and even subsoils are often removed and sold from constructions sites. Topsoil has become a commodity that is of value and is often transported and sold to the highest bidder. In the urban landscapes, soils (often both topsoil and subsoil) have been removed, exposing the dense parent materials, which are often built on and then landscaped using a thin layer of sod. This sod has difficulty in growing roots into the dense parent material. Often, soils are no longer present and cannot accept or hold the rainwater on the landscape. Runoff is much greater than it was prior to the alteration. The lack of soil water storage and covering of other soils in the landscape by roofs, sidewalks, driveways, and roads has resulted in more runoff and increased flooding. As humankind have tried to clean up the air and water, the industrial and urban pollutants, urban and industrial wastes, and sewage sludges have either been consolidated or buried under the soil materials, or transported and spread on soils as amendments. Fortunately, environmental regulations no longer allow urban and industrial wastes and sewage sludges to be transferred to the air, rivers, lakes, or oceans.

CONCLUSION

Contemporary use of soils is not greatly different from ancient use of soils. Many millions of hectares in the modern urban landscape have been altered or destroyed for construction of buildings, transportation, or mineral extractions. The archeological record shows that some ancient people abused their soils and that their civilizations were disrupted by the ecological and environmental consequences. One of history's greatest problems has been the cultivation in and erosion of sloping lands. Repeated productivity losses on these soils would result in long-term costs to society, because this natural resource (capital) would eventually be depleted. Sustaining the long-term productivity and quality of all soil resources needs to be a national priority, which is essential to our nation's prosperity. It is in the national interest that management strategies and techniques continue to be developed to monitor and sustain soil productivity and quality.

REFERENCES

1. Olson, G.W. *Soils and the Environment—A Guide to Soil Surveys and their Applications*; Chapman and Hall: New York, 1981.
2. Fowler, M.L. *The Cahokia Atlas: A Historical Atlas of Cahokia Archaeology*, Revised Ed.; University of Illinois at Urbana: Champaign, 1997.
3. Fowler, M.L. *The Cahokia Atlas: A Historical Atlas of Cahokia Archaeology*; Studies in Illinois Archaeology No. 6; Illinois Historical Preservation Agency: Springfield, 1989.
4. Milner, G.R. The later prehistoric cahokia culture system of the Mississippi river valley: Foundations, florescence, and fragmentation. *J. World Prehist.* **1990**, *4*, 1–43.
5. Pauketat, T.R. *The Ascent of Chiefs: Cahokia and Mississippian Politics in Native North America*; University of Alabama Press: Tuscaloosa, 1994.
6. Bareis, C.J.; Porter, J.W. *American Bottom Archaeology: A Summary of the FAI-270 Project Contributions to the Culture History of the Mississippi River Valley*; University of Illinois Press: Urbana, 1984.
7. Hall, R.L. Cahokia identity and interaction models of Cahokia Mississippian. In *Cahokia and the Hinterlands: Middle Mississippian Cultures of the Midwest*; Emerson, T.E., Lewis, R.B., Eds.; University of Illinois Press: Urbana, 1991.
8. Soil Survey Staff. *Soil Taxonomy, a Basic System of Soil Classification for Making and Interpreting Soil Surveys*; AH-436; U.S. Government Printing Office: Washington, 1975.
9. Lowdermilk, W.C. *Conquest of the Land through Seven Thousand Years*; Agricultural Information Bulletin No. 99; 2nd Ed.; U.S. Department of Agriculture: Washington, 1975.
10. Troeh, F.R.; Hobbs, J.A.; Donahue, R.L. *Soil and Water Conservation for Productivity and Environmental Protection*; Prentice-Hall: Englewood Cliffs, 1980.
11. Brown, L.R.; Wolf, E.C. *Soil Erosion: Quiet Crisis in the World Economy*; Worldwatch Institute: Washington, 1984.
12. Lal, R. Soil degradation and the future of agriculture in sub-Saharan Africa. *J. Soil Water Conserv.* **1988**, *43*, 444–451.
13. Lee, L.K. Land use and soil loss: 1982 update. *J. Soil Water Conserv.* **1984**, *39*, 226–228.
14. USDA Soil Conservation Service Staff. *Soil and Water Resources Conservation Act Program Report and Environmental Impact Statement*; U.S. Government Printing Office: Washington, 1981.
15. USDA Soil Conservation Service Staff. *Preliminary Data, 1982*; National Resources Inventory, Executive Summary: Washington, 1983.

Mulch Farming

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Mulch farming maintains or enhances soil quality by conserving soil and water resources, improving soil organic carbon content, increasing soil biodiversity, strengthening nutrient cycling mechanisms, and regulating soil temperature regime. Mulch farming increases the probability of achieving agricultural sustainability through: 1) increasing production; 2) decreasing costs of fertilizers and water inputs; and 3) preserving the resource base and its productive potential.

INTRODUCTION

Mulching, the practice of leaving crop residues and other biomass on the soil surface to protect it from climatic elements, is an ancient practice. Its application, however, is even more relevant than ever before because of the serious global issues relevant to human population growth, scarcity of prime agricultural land, ecological benefits of nutrient recycling and optimizing rates of fertilizer input, severe problems of soil degradation, adverse water quality, and the accelerated greenhouse effect. It is an ecological approach to addressing the problem of sustainability, soil degradation environment quality, and nutrient cycling. Conventional agricultural practices, based on clean cultivation and intensive use of inorganic fertilizers and chemical amendments, make soils susceptible to degradative processes, e.g., erosion, acidification, and emission of greenhouse gases to the atmosphere. Stabilizing the atmospheric concentration of greenhouse gases is a major global challenge.^[1] The global release of soil organic carbon (SOC) from agriculture is estimated at 800 Tg C/yr (T = tera = 10^{12} = 1 million metric ton).^[2] Soil restoration (for enhancement of SOC content) is an important and feasible option for mitigating the greenhouse effect^[3–5] and enhancing soil quality for increasing production and improving the environment.^[6,7] These desirable properties and processes are achievable through frequent applications of crop residue mulch and other mulch farming techniques.

MULCH TYPES

Mulch is any material, other than soil, specifically established at the soil–air interface to manage soil and water and create favorable environments for plant growth. Most mulches are organic materials of plant origin comprising crop residues, weed biomass, leaf litter, and by-products of agroindustries, e.g., saw dust, rice husk, and corn cobs.

However, there is a wide variety of mulch materials ranging from gravels and plastic to crop residues and planted fallows. Mulches can be broadly classified into two categories: organic and inorganic mulches (Fig. 1).

1. *Organic mulches.* Most mulch materials used in agriculture are of organic origin and comprise crop residue used both *ex situ* and *in situ*.
2. *Inorganic mulches.* Three most common inorganic mulches are plastic, gravels, and water.

MULCH PROCUREMENT FOR ARABLE LAND

Crop Residues

The most common practice of mulch procurement for use on agricultural land is the use of crop residues from the previous season's crops. Crop residue is a renewable resource, and annually a large quantity of residues are produced in the world.^[9,10] Residues produced in the world during 2001 were estimated at 3758 million Mg, of which 2802 million Mg (75%) were from cereals, 305 million Mg (8%) from legumes, and 651 million Mg (17%) from other crops (Table 1). The amount of residue produced by seven crops in the tropics is estimated at 1838 million Mg (Table 2). Properly used as mulch, these residues can cover the entire arable land of 1500 million hectares at an average rate of 2.53 Mg/ha. Therefore, crop residues are an important resource that need to be carefully and judiciously used for sustainable use of soil, water, and other natural resources.

Crop residues management

Management of crop residues is an important strategy of mulch procurement. Crop/plant residues, produced by annuals or perennials, *in situ* or *ex situ*, used as mulch can improve soil quality, increase productivity and sustainability,

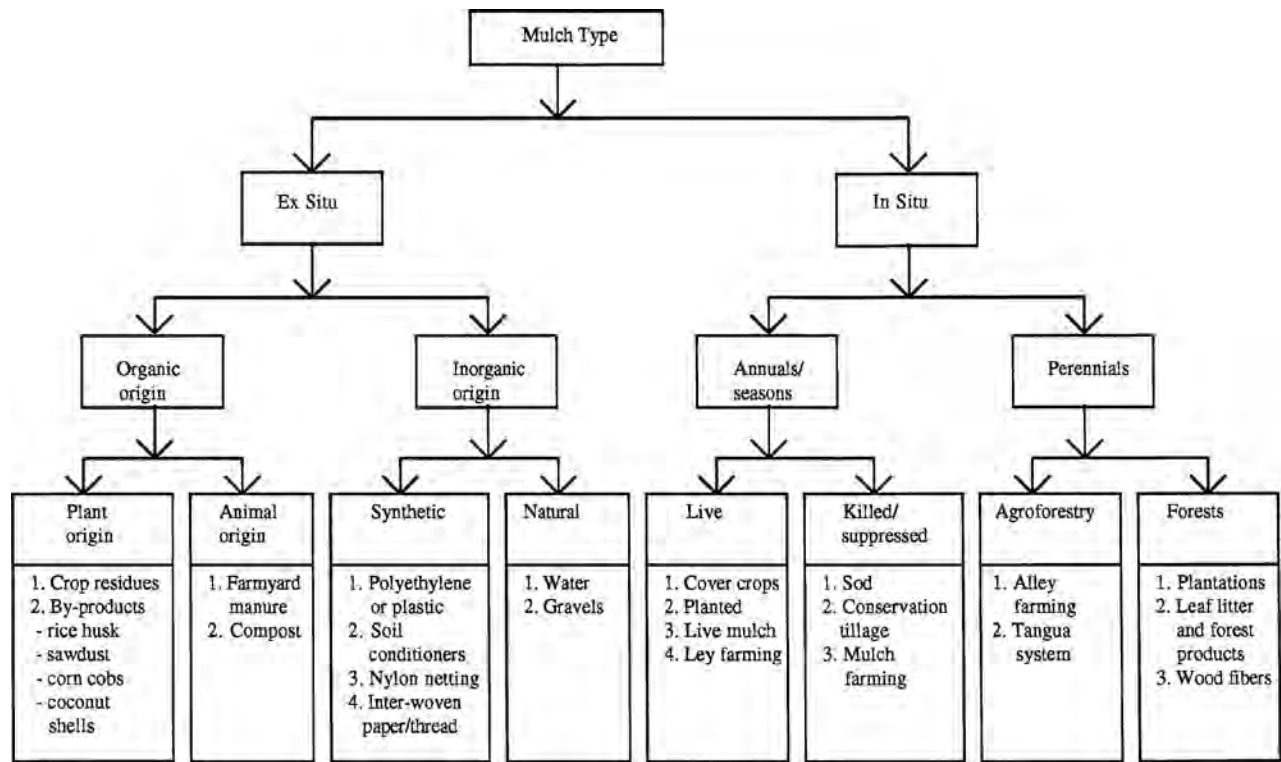


Fig. 1 Different types of mulch materials.
Source: From Larson.^[8]

and enhance environmental quality (Fig. 2). Principal management alternatives outlined in Fig. 3 indicate four options, namely: 1) mulching, 2) using as animal feed, 3) composting, and 4) burning. Using residues as mulch is the best option for controlling soil erosion, conserving soil water, and at the same time replenishing plant nutrient reserves.

Planted Fallows and Cover Crops

Cover crops, legumes, or grasses generally grown to produce biomass and provide ground cover are a good source of mulch. Suitable cover crops are those that: 1) establish rapid ground cover; 2) improve soil fertility through biological nitrogen (N) fixation and recycling nutrients;

Table 1 Estimates of crop residues produced by cereals, legumes, and other crops.

Crops	Residues produced (106 ton/yr)	
	1991	2001
Cereals	2563	2802
Legumes	238	305
Oil crops	162	108
Sugar crops	340	373
Tubers	145	170
Total	3448	3758

Source: From Lal.^[10]

3) suppress weeds and other pests; and 4) produce a large quantity of biomass that can be used as mulch. Selected legumes and grasses, appropriate for the soil and environment, can ameliorate soil properties and produce the mulch material in a short time interval.^[12] Deep rooted cover crops are also effective in improving subsoil properties.

BENEFITS OF MULCH FARMING

Mulch farming is an ecological approach to agriculture. Properly used, it can improve and sustain agricultural

Table 2 Estimates of crop residues produced in the tropics.

Crop	Asia	Africa	South America
	10 ³ Mg		
Rice straw	772	26	24
Rice husk	154	5	5
Wheat	380	27	26
Barley	34	7	2
Sugarcane	54	9	42
Cotton	6	0.3	0.07
Oats	2	0.3	2
Corn	166	39	55
Total	1568	114	156

Source: From Singh, Singh, et al.^[11]

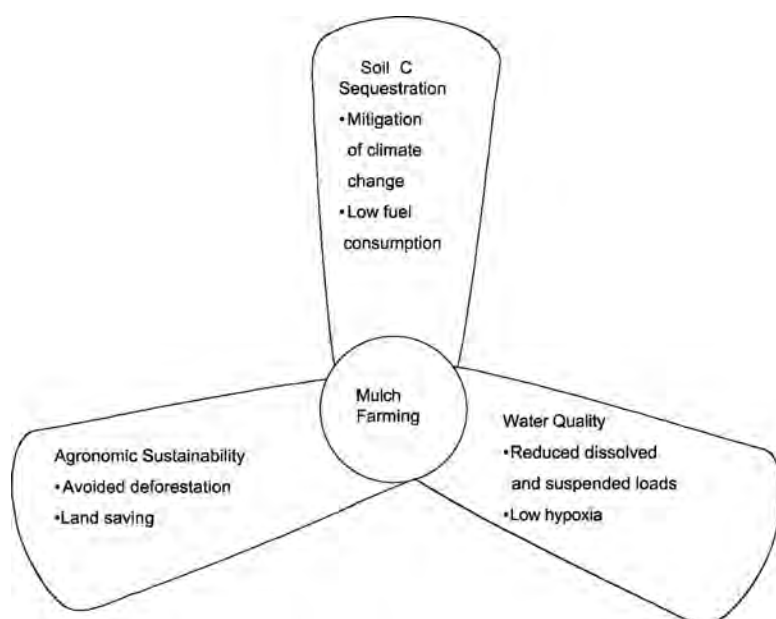


Fig. 2 Interactive effects of mulch farming on soil quality/agronomic productivity, water quality, and greenhouse effect.

production, enhance soil quality, and have minimal adverse effects on the environment (Fig. 4). Principal benefits of mulch farming are briefly described in this figure.

Soil-Surface Management and Erosion Control

Soil erosion by water and wind is the principal degradative process,^[13] and it adversely affects soil quality.^[14,15] Mulching, at the rate of 4–6 Mg/ha, is an effective erosion-control measure.^[16] Larson^[8] recommended the use of crop residue mulch for erosion control in the Mid-western United States. Moldenhauer and Wischmeier^[17] observed significant reduction in soil erosion by mulch farming practices. Both runoff and soil erosion decrease

exponentially with an increase in surface area covered by mulch (Fig. 5). Structural improvement by mulching is due to an increase in SOC or humus content that facilitates the formation of organomineral complexes and an increase in soil biodiversity, notably the activity of earthworms, termites, and other beneficial soil fauna.^[18]

Water Management

There are four types of water, namely: 1) surface water; 2) ground water; 3) soil water; and 4) rainwater. Mulching affects all four types of water, but has a major impact on soil water and surface water. Mulching decreases runoff by improving infiltration rate, replenishes soil water by

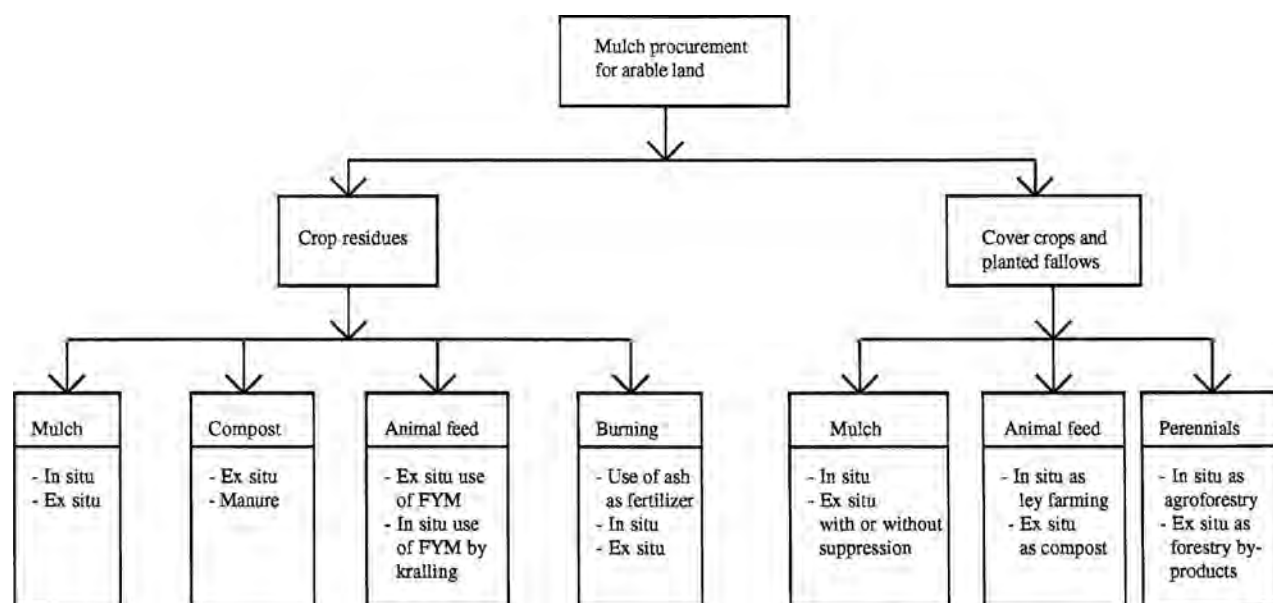


Fig. 3 Methods of procurement of mulch materials.

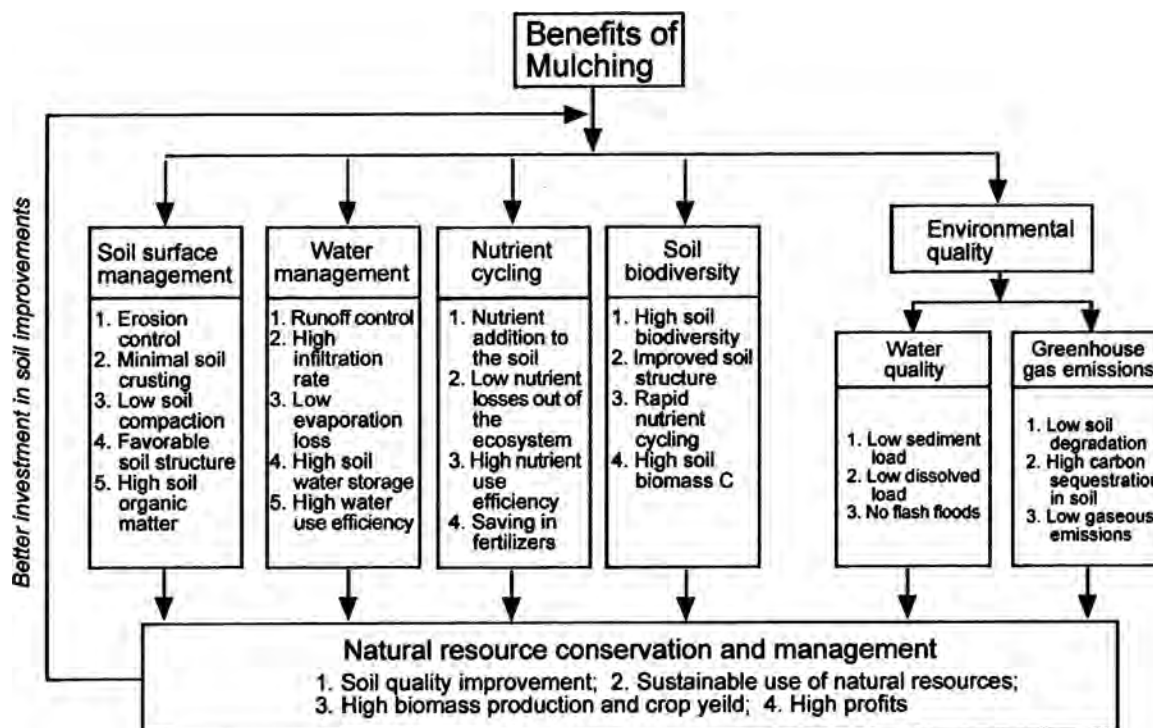


Fig. 4 Economic, ecologic, and natural resource improvements by mulch farming.

improving retention pores, and recharges groundwater by increasing percolation. Partitioning of rainwater into runoff, soil water, and groundwater components is influenced by mulching through its effect on the water balance. Mulching increases soil water and groundwater components by decreasing runoff and soil evaporation. Mulching improves soil water storage and minimizes the risks of drought. When the soil is wet at about field capacity level, as is the case soon after rainfall or irrigation, mulching decreases the evaporation rate and prolongs the duration during which soil remains moist (Fig. 6).

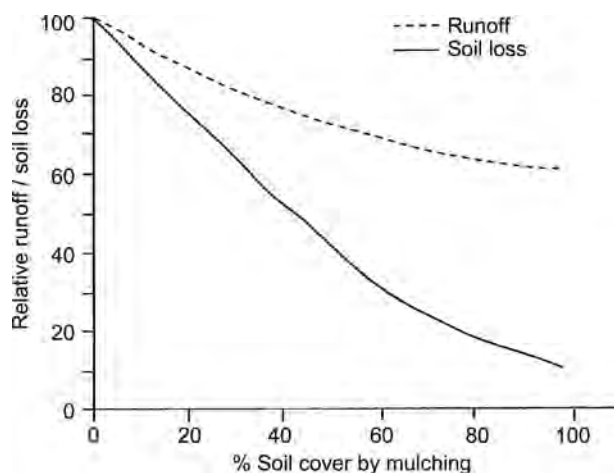


Fig. 5 A schematic depicting an exponential decline erosion with increasing soil cover by mulching.

Soil Temperature Management

Crop residue mulch has a buffering or moderating effect on soil temperature regime. It influences the extremes by decreasing the maximum soil temperature and increasing the minimum soil temperature. Therefore, soil temperature range is narrower in mulched than in unmulched soil. Mulching increases soil temperature in the morning and decreases it in the afternoon (Fig. 7). Plastic mulches are often used to increase soil temperature in horticultural and other high-value plants. Soil temperature effects of mulching are due to both direct and indirect effects. Direct effects are due to reduction in insolation or energy load on the soil surface. Mulching shades the soil and prevents the

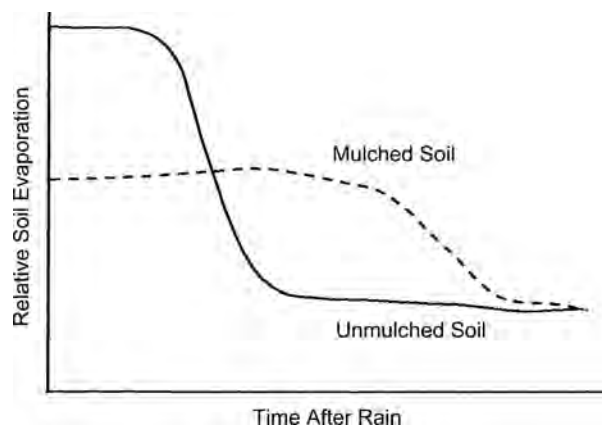


Fig. 6 A schematic depicting mulch effects on soil.

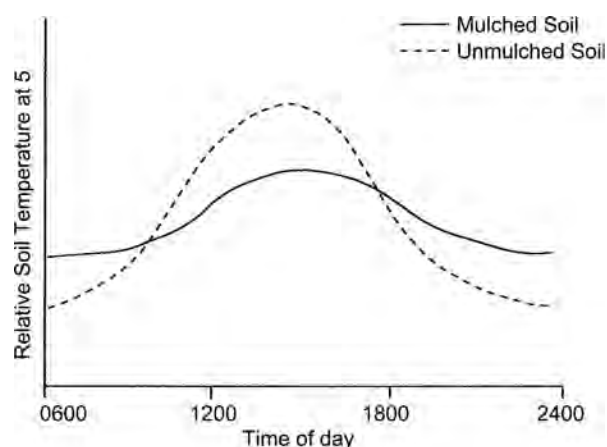


Fig. 7 A schematic diagram showing diurnal fluctuations in mulched and unmulched soil during summer.

radiation reaching the soil surface. Indirectly, mulching influences soil temperature through its effect on soil's heat capacity and thermal conductivity.

Nutrient Cycling and Soil Fertility Enhancement

Severe nutrient depletion in sub-Saharan Africa^[19,20] and elsewhere can be controlled through nutrient cycling. Depending on plant species and management systems, crop residues contain a considerable amount of plant nutrients. Total amount of plant nutrients in crop residues range from 40 to 100 kg/Mg of residues.^[9] On a global scale, crop residues contain 22.6 million Mg N, 3.6 million Mg phosphorus (P), 47.4 million Mg potassium (K; [Table 3](#)). Estimates of global fertilizer consumption show annual use of 113 million tons of NPK vs. 74 million Mg contained in crop residues. In comparison, annual fertilizer consumption is 16 million Mg of NPK in the United States vis-à-vis 9 million Mg contained in crop residues. Estimates of N, P, and K contained in crop residue in the world and in the developing countries are shown in [Tables 4, 5, and 6](#), respectively. Thus, cycling of nutrients contained in the

Table 3 Nutrients contained in crop residues of cereals and legumes produced in the world.

Nutrient	Plant nutrients in residues (million tons/yr)		Fertilizer use (million tons/yr)	
	The United States	World	The United States	World
N	2.98	22.62	10	77
P	0.47	3.58	2	16
K	5.70	47.39	4	20
Ca	1.87	12.11	—	—
Mg	0.85	6.16	—	—
N+P+K	9.15	73.59	16	113

Source: From Lal.^[9]

Table 4 Estimates of N contained in crop residue.

Crop	Asia	Africa and South America	World
	10 ³ Mg		
Rice	4,862	328	5,346
Wheat	1,899	308	5,651
Barley	222	60	1,521
Sugarcane	226	212	526
Cotton	64	4	69
Oats	15	12	325
Corn	781	788	5,393
Total	8,069	1,712	18,832

Source: From Singh, Singh, et al.^[11]

residue is an important factor affecting soil quality and agronomic yields. It is feasible, therefore, that a considerable quantity of fertilizer can be saved by returning crop residues to the soil. Indeed, mulching and residues management are effective nutrient recycling mechanisms.

Soil Biodiversity

Applications of crop residues mulch improve the activity of soil micro- and macrofauna and biomass carbon (C). The relative magnitude of biomass carbon is a good indicator of soil quality. Mulching improves population and activity of soil fauna especially those of earthworms and termites. These organisms improve soil structure by burrowing, mixing and soil turnover activities. They decompose and mineralize crop residues and make nutrients available to plants.

The Greenhouse Effect

The atmospheric carbon dioxide (CO₂) concentration has increased by approximately 30% from 280 ppm in 1850 to approximately 370 ppm in 2000.^[21,22] This increase is attributed to two principal human activities: 1) land use change and soil management (1.6 Pg C/yr; 1 Pg = petagram = 10¹⁵ g) and 2) fossil fuel consumption (6.8 Pg C/yr).

Table 5 Estimates of P contained in crop residue.

Crop	Asia	Africa and South America
	10 ³ Mg	
Rice	772	53
Wheat	266	37
Barley	31	8
Sugarcane	43	40
Cotton	10	1
Oats	4	3
Corn	216	149
Total	1342	291

Source: From Singh, Singh, et al.^[11]

Table 6 Estimates of K contained in crop residue.

Crop	Asia	Africa and South America	World
	10 ³ Mg		
Rice	4,600	446	5,322
Wheat	2,355	418	7,758
Barley	235	73	2,374
Sugarcane	361	338	839
Cotton	64	4	69
Oats	40	32	852
Corn	1,230	1,013	6,824
Total	8,885	2,324	24,038

Source: From Singh, Singh, et al.^[11]

The rate of increase of atmospheric CO₂ concentration is approximately 1.5 ppm or 3.5 Pg C/yr. Total crop residues production in the world estimated at 3460 million tons per annum^[9] contains 1.5 Pg of C. This is approximately 43% of the annual atmospheric increase of 3.5 Pg/C and cannot be ignored in the context of global C cycle. This C can have a major effect even if only 10% of the 1.5 Pg can be sequestered in soils as humus. In addition to the direct effect,

mulching also sequesters C through erosion control and increase in biomass production.^[23]

MULCHING AND AGRICULTURAL SUSTAINABILITY

Mulch farming affects agricultural sustainability through its impact on soil quality and productivity (Fig. 8). Mulch-induced changes in soil quality set in motion restorative processes leading to soil and water conservation, nutrient cycling and soil fertility improvement, improvement in soil structure and C sequestration, increase in soil biodiversity, and soil structure enhancement. Mulch farming improves energy use efficiency by 1) reducing fertilizer use through decreasing nutrient losses in runoff and erosion and recycling plant nutrients; 2) decreasing irrigation needs through increasing infiltration and reducing losses due to runoff and evaporation; 3) decreasing herbicide use through suppressing weed growth; and 4) decreasing fuel consumption through reduction of tillage operations and vehicular traffic.

Crop residues are also a renewable source of energy that can be used in the farm household. Energy content of straw, depending on crop species and other factors, ranges from 3000 to 4000 kcal/kg.^[24] Energy value of 1 Mg of residues may be as much as 9 million BTU, 18.6×10^9 J, or equivalent to two barrels of oil.^[25] Estimates of fuel energy in crop residues show that cereal straw produced annually in the world is equivalent to 47×10^{18} J of energy (Table 7).^[10] However, traditional use of biomass (crop residue, cattle dung, etc.) can have severe adverse impacts on the environment (Fig. 9).^[26]

LIMITATION OF MULCH FARMING

Biomass, the principal mulch material, has numerous alternative uses especially for resource-poor farmers of the tropics. Crop residues are used for feeding livestock for building fences and homesteads and as fuel for cooking and household energy use. It is a precious commodity, and little if any is left behind on the field as mulch on small farms in the developing countries. Under some circumstances, mulches may also have negative effects on crop growth due to cool soil temperature during spring in temperate regions, high incidence of pests and pathogens, and N immobilization.

Table 7 Fuel energy of cereal straw produced in the world.

Parameter	Global value
Total crop residue (million Mg/yr)	3758
Oil equivalent (106 barrels)	7560
Energy equivalent (Quads)	60

Source: From Lal.^[9]

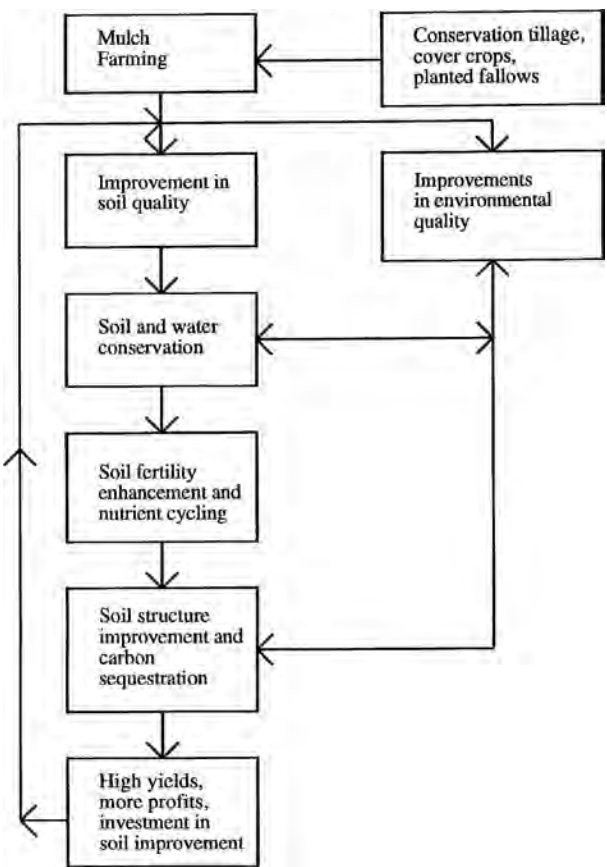


Fig. 8 Mulch farming and agricultural sustainability.

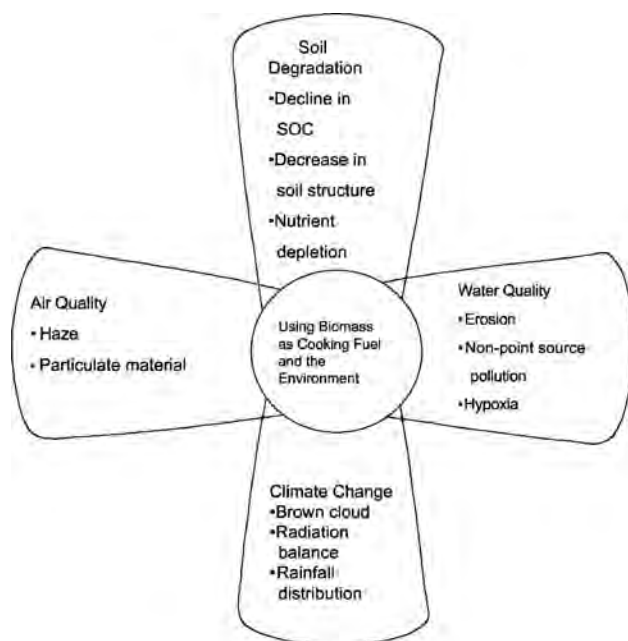


Fig. 9 Biomass burning as household fuel and the environment.

CONCLUSION

Mulch farming maintains or enhances soil quality by conserving soil and water resources, improving SOC content, increasing soil biodiversity, strengthening nutrient cycling mechanisms, and regulating soil temperature regime. Mulch farming increases probability of achieving agricultural sustainability through: 1) increasing production; 2) decreasing costs of fertilizers and water inputs; and 3) preserving the resource base and its productive potential.

Mulch farming has a potential to improve environmental quality by: 1) decreasing soil erosion and transport of chemicals in runoff, mulch farming decreases risks of eutrophication of surface waters and contamination of groundwater; and 2) reducing emission of radioactively active gases from soil to the atmosphere through increasing SOC content, stabilizing C within aggregates as organomineral complexes, decreasing losses due to soil erosion, and improving biomass production.

REFERENCES

- Engelman, R. Stabilizing the atmosphere: Population, consumption, and greenhouse gases. In *Population and Environment Program*; Population Action International: Washington, 1994; 48 pp.
- Schlesinger, W.H. Evidence from chronosequence studies for a low carbon-storage potential of soils. *Nature* **1990**, *348*, 232–234.
- Lal, R.; Kimble, J.M.; Levine, E.; Stewart, B.A., Eds. *Soils and Global Change*; CRC/Lewis Publishers: Boca Raton, 1995; 440 pp.
- Lal, R.; Kimble, J.M.; Levine, E.; Stewart, B.A. *Soil Management for Mitigating the Greenhouse Effect*; CRC/Lewis Publishers: Boca Raton, 1995; 385 pp.

- Lal, R. Soil management and restoration for C sequestration to mitigate the accelerated greenhouse effect. *Prog. Environ. Sci.* **1999**, *1*, 307–326.
- Doran, J.W.; Parkin, T.B. Defining and assessing soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., Eds.; SSSA Special Publication No. 35; SSSA: Madison, 1994; 3–21.
- Lal, R. *Soil Quality and Agricultural Sustainability*; Ann Arbor Press: Chelsea, 1998; 3–11.
- Larson, W.E. Protecting the soil resource base. *J. Soil Water Conserv.* **1981**, *36*, 13–16.
- Lal, R. The role of residues management in sustainable agricultural systems. *J. Sustain. Agr.* **1995**, *5*, 51–78.
- Lal, R. The world crop residue production and implications of its use as biofuel. *Environ. Int.* **2005**, *31*, 575–584.
- Singh, Y.; Singh, B.; Timsina, J. Crop residue management for nutrient cycling and improving soil productivity in rice-based cropping systems in the tropics. *Adv. Agron.* **2005**, *85*, 269–407.
- Smith, M.S.; Frye, W.B.; Varco, J.J. Legume winter cover crops. *Adv. Soil Sci.* **1987**, *7*, 95–140.
- Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 99–118.
- Lal, R. Applying soil quality concepts for combating soil erosion. In *Soil Quality and Soil Erosion*; Lal, R., Ed.; CRC Press: Boca Raton, 1999; 309–318.
- Lal, R.; Mokma, D.; Lowery, B. Relation between soil quality and erosion. In *Soil Quality and Soil Erosion*; Lal, R., Ed.; CRC Press: Boca Raton, 1999; 237–257.
- Lal, R. *Soil Erosion in the Tropics: Principles and Management*; McGraw-Hill: New York, 1990; 580 pp.
- Moldenhauer, W.C.; Wischmeier, W.H. Soil and water losses and infiltration rates on Ida silt loam as influenced by cropping systems, tillage practices, and rainfall characteristics. *Soil Sci. Soc. Am. Proc.* **1960**, *24*, 409–413.
- Lal, R. Soil structure and sustainability. *J. Sustain. Agr.* **1991**, *1*, 67–92.
- Stoorvogel, J.J.; Scaling, E.M. *Assessment of Soil Nutrient Depletion in Sub-Saharan Africa: 1983–2000*; Report No. 28; The World Starving Center: Wageningen, 1990.
- Stoorvogel, J.J.; Smaling, E.M.A.; Janssen, B.H. Calculating soil nutrient balances in Africa at different scales. I. Supranational scale. *Fert. Res.* **1993**, *35*, 227–235.
- Inter-Governmental Panel on Climate Change (IPCC). *Technical Summary*; WMO: Geneva, 1995; 44 pp.
- Kennel, C. *UC Revelle Program on Climate, Science and Policy*; Scripps Institute of Oceanography: La Jolla, 2000.
- Lal, R. Residue management, conservation tillage and soil restoration for mitigating greenhouse effect by CO₂ enrichment. *Soil Till. Res.* **1997**, *43*, 81–107.
- Stout, B.A. *Handbook of Energy for World Agriculture*; Elsevier: New York, 1990.
- Anderson, A.W.; Anderson, J.F. On finding a use of straw. In *Utilization and Recycles of Agricultural Wastes and Residues*; Shuler, M.L., Ed.; CRC Press: Boca Raton, 1980; 237–278.
- Ramanathan, V.; Crutzen, P.J.; Kiehl, J.T.; Rosenfeld, D. Aerosols, climate and the hydrological cycle. *Science* **2002**, *294*, 2118–2124.

Mycorrhiza of Forest Ecosystems

Ian A. Dickie

School of Forest Resources, University of Minnesota, St. Paul, Minnesota, U.S.A.

Abstract

Mycorrhiza are involved in every aspect of forest soil ecology—they are central to plant nutrient uptake, they can short-circuit the nitrogen cycle, they participate in primary weathering of soil nutrients, and they provide a major input of carbon into the soil ecosystem. In addition, a lack of mycorrhizal fungi may limit the ability of forest tree species to establish slowing forest succession on disturbed sites. At least a basic understanding of mycorrhiza is therefore critical to understanding both soil and plant ecology.

INTRODUCTION

Most land plants (>80% of angiosperms and almost all gymnosperms) form mycorrhiza: symbiotic associations between plant roots and fungi that increase plant nutrient uptake.^[1,2] Mycorrhizal fungi are an important component of the soil ecosystem, with ca. 200 m of mycorrhizal hyphae per gram of forest soil.^[1] Several types of mycorrhiza occur, including arbuscular mycorrhiza (dominants of grasslands, many tropical, and some temperate forests), ectomycorrhiza (dominants of boreal, most temperate, and some tropical forests), and ericoid mycorrhiza (common in temperate and boreal forests and heathlands). Mycorrhiza plays a critical role in plant nutrient uptake, particularly of phosphorus (P), NH_4^+ , and organic nutrients. In return for increased nutrient acquisition, plants direct 10–30% of total fixed carbon (C) to supporting mycorrhizas. This represents an important flow of C to the soil ecosystem, creating “mycorrhizosphere” communities in association with fungal hyphae.

MYCORRHIZAL TYPES

Arbuscular mycorrhizas are the ancestral state of all land plants, and approximately 67% of all angiosperms and nearly all non-Pinaceae gymnosperms retain this association.^[2] Arbuscular mycorrhizas are formed by most tropical and many temperate forest trees, as well as many forest understory plants. Arbuscular mycorrhizas are characterized by the internal colonization of root, with the fungus forming characteristic structures (arbuscules) within the root cells. Arbuscular mycorrhizas are formed by fungi in the order Glomales.

While less common than arbuscular mycorrhiza, ectomycorrhizas are particularly important in forest ecosystems (Fig. 1). Predominately ectomycorrhizal plant families include the Pinaceae, Fagaceae, Myrtaceae, Dipterocarpaceae, Salicaceae, Betulaceae, and the leguminous

subfamily Caesalpinioideae, among others.^[1] Although these species account for only around 3% of plant diversity, they include the dominants of most temperate and boreal forests and some tropical forests. In ectomycorrhiza, the fungus forms a mantle, which envelops root tips, and a Hartig net, which penetrates between the cells of the epidermis and cortex of the root but does not penetrate cells. The fungi forming ectomycorrhizas are primarily Basidiomycetes, as well as some Ascomycetes and some species of the Zygomycete genus *Endogone*. Many fungi forming ectomycorrhizas produce conspicuous sporocarps, some of which are economically important edibles (e.g., chanterelles, truffles, cepes, and matsutake).

Ericoid mycorrhizas are exclusively formed by plants in the Ericales (Ericaceae, Epacridaceae, and Empetraceae). These plants are an important component of high altitude, temperate, and boreal forests and heathland ecosystems. Ericoid mycorrhizas form a loose weft of undifferentiated hyphae on the root surface, with extensive intercellular structures within root cells. Fungi forming ericoid mycorrhizas are primarily Ascomycetes, and some are Basidiomycetes.^[3]

In general terms, arbuscular mycorrhizal forests tend to dominate low-latitude and low-elevation sites with P-limited soils. Ectomycorrhizal forests become dominant with increasing latitude and elevation and on more acidic and nitrogen (N)-limited soils. Ericoid mycorrhizas dominate at the highest elevations and latitudes and on highly acid or organic soils.^[1] Other forms of mycorrhiza also occur in forest ecosystems, including ectendomycorrhizas, arbutoid, orchid, and monotropoid mycorrhizas.^[1] None of these types dominate forest ecosystems.

NUTRIENT UPTAKE

Mycorrhizas can considerably increase plant nutrient uptake, and for many plants, they are the primary mechanism for nutrient uptake. All forms of mycorrhiza greatly

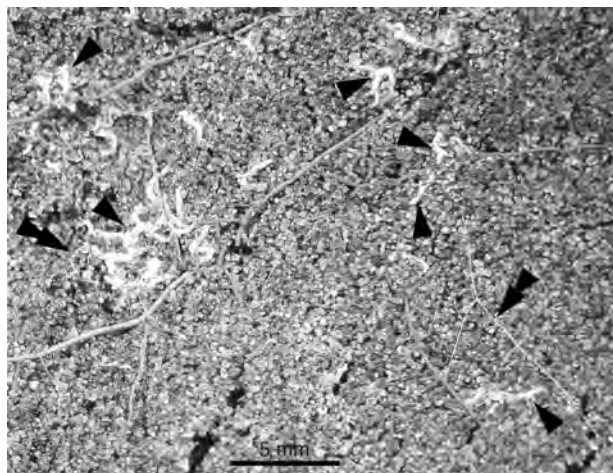


Fig. 1 Ectomycorrhiza formed by the fungus *Cortinarius* sp. on the roots of a potted oak seedling. Single triangles indicate ectomycorrhizal root tips, and double triangles indicate fungal rhizomorphs.

Source: Photo by the author; seedling courtesy of P.G. Avis.

increase the ability of plants to explore the soil. Estimates of the length of mycorrhizal hyphae in forest ecosystems range from 300 m to 8000 m of fungal hyphae per meter root length.^[4] By increasing effective root surface area, mycorrhizal fungi can substantially increase nutrient uptake, particularly of nutrients such as P and NH_4^+ that have low mobility in soils. The small size of fungal hyphae (down to $<2 \mu\text{m}$ diameter) also permits fungal hyphae to explore soil pore spaces that are inaccessible to roots.

While all mycorrhizas increase the root surface area, ectomycorrhiza and ericoid mycorrhiza are also capable of obtaining nutrients from sources which are otherwise unavailable to plants. Ectomycorrhiza and ericoid mycorrhiza are capable of obtaining N and P from organic sources, even where those organic nutrients are protected by structural components of litter.^[1] This has the potential to short-circuit the N cycle, with plants obtaining N directly from organic sources without that N first being mineralized. Ectomycorrhiza may also increase the uptake of P, calcium, potassium, and magnesium through direct weathering of mineral nutrient sources.^[5] In addition to nutritional benefits, ectomycorrhizal fungi can increase plant tolerance of drought and heavy metal toxicity and increase resistance to disease.^[1]

C COSTS

The fungi involved in mycorrhiza are dependent on plants for C. Available estimates indicate that forest trees may transfer 10–30% of total fixed C to mycorrhiza.^[1] These costs are typically offset by increased nutrient uptake (and hence increased total photosynthesis), but reports of

negative growth responses to mycorrhiza, particularly arbuscular mycorrhiza, are not uncommon.

Considerable interest has been generated by the suggestion that mycorrhiza may permit the transfer of C between plants. Individual fungi may infect roots of different plants, potentially linking these plants into a common mycelial network. It has been suggested that this may permit C fixed by one plant to flow through mycelial links into other plants.^[6] Some achlorophyllous plants apparently obtain all of their C from such mycorrhizal linkages.^[1] Evidence for flow of C between photosynthetic plants remains equivocal because of difficulties in demonstrating net C flow^[6,7] (as opposed to gross C flow, which has been clearly demonstrated). If such interplant C transfer does occur, the ecological importance of such transfer in increasing seedling growth remains to be quantified.

INFLUENCE ON SOIL ECOSYSTEMS

The interactions between mycorrhiza and soil microorganisms can be complex. Mycorrhiza can serve as an important channel for C flow from plants to soil, potentially increasing microbial biomass or altering microbial community composition. At the same time, mycorrhizal fungi may compete with some saprophytes for soil nutrients or may produce antibiotic compounds that inhibit the growth of bacteria and fungi. The outcome of mycorrhiza–microbial interactions appears to depend both on the mycorrhizal and microbial species involved and on soil factors.^[8] One group of bacteria, the so-called mycorrhization helper bacteria, appears to enhance mycorrhizal infection of plants.

Studies on the influence of mycorrhizal fungi on decomposition drew conflicting results. Mycorrhizal fungi may inhibit decomposition (the so-called Gadgil effect) through competition with or direct inhibition of soil saprophytes.^[1] Alternatively, mycorrhizal fungi may actively participate in the decomposition (increasing the overall rate) or may enhance the activities of some saprophytes through the exudation of C-containing compounds.^[8]

SEEDLING INFECTION

Mycorrhizas are a horizontal symbiosis: each sexually produced generation of plants must obtain their own symbiotic partners independent of the parent plant. This is an important consideration, as a lack of mycorrhizal fungi can greatly hinder the establishment of plants. Limitation due to a lack of mycorrhizal fungi has been clearly observed in the failure of ectomycorrhizal trees to establish in regions where ectomycorrhizal vegetation is nonnative.^[1] Low levels of ectomycorrhizal fungi can also be an important limitation on even relatively small-scale disturbed sites.^[9] On sites where mycorrhizal inoculum is otherwise limiting, seedlings may have increased infection near established

ectomycorrhizal plants.^[9] Arbuscular mycorrhizal fungi may be more ubiquitous in secondary succession; however, these fungi are often absent in early primary successional sites (e.g., glacial forefronts, sand dunes, and strip mines). Even where mycorrhiza are uniformly present, there may be patchiness in the effectiveness of mycorrhiza at increasing plant growth.^[9,10]

Efforts to manipulate mycorrhizal fungi to increase forest productivity have had mixed success. Clear benefits of ectomycorrhizal inoculation are observed when planting tree seedlings on particularly harsh sites or sites with no native ectomycorrhizal fungi. However, under more typical forest conditions, inoculated strains of fungi are often out-competed by native fungi and may have limited benefits for forest production.^[1]

CONCLUSION

Mycorrhizas are involved in every aspect of forest soil ecology: they are central to plant nutrient uptake, they can short-circuit the N cycle, they participate in primary weathering of soil nutrients, and they provide a major input of C into the soil ecosystem. In addition, a lack of mycorrhizal fungi may limit the ability of forest tree species to establish, slowing forest succession on disturbed sites. At least a basic understanding of mycorrhiza is therefore critical to understanding both soil and plant ecology. A number of reviews are available for further reading.^[1,11,12]

ACKNOWLEDGMENT

The author was supported by NSF/DEB grant 0080382.

REFERENCES

1. Smith, S.E.; Read, D.J. *Mycorrhizal Symbiosis*, 2nd Ed.; Academic Press: London, 1997.
2. Brundrett, M.C. Coevolution of roots and mycorrhizas of land plants. *New Phytol.* **2002**, *154* (2), 275–304.
3. Perotto, S.; Girlanda, M.; Martino, E. Ericoid mycorrhizal fungi: Some new perspectives on old acquaintances. *Plant Soil* **2002**, *244* (1–2), 41–53.
4. Rousseau, J.V.D.; Sylvia, D.M.; Fox, A.J. Contribution of ectomycorrhiza to the potential nutrient-absorbing surface of pine. *New Phytol.* **1994**, *128* (4), 639–644.
5. Landeweert, R.; Hoffland, E.; Finlay, R.D.; Kuyper, T.W.; van Breemen, N. Linking plants to rocks: Ectomycorrhizal fungi mobilize nutrients from minerals. *Trends Ecol. Evol.* **2001**, *16* (5), 248–254.
6. Simard, S.W.; Durall, D.; Jones, M. Carbon and nutrient fluxes within and between mycorrhizal plants. In *Mycorrhizal Ecology*; van der Heijden, M.G.A., Sanders, I.R., Eds.; Springer-Verlag: Berlin, 2002; 33–74.
7. Robinson, D.; Fitter, A.H. The magnitude and control of carbon transfer between plants linked by a common mycorrhizal network. *J. Exp. Bot.* **1999**, *50* (330), 9–13.
8. Cairney, J.W.G.; Meharg, A.A. Interactions between ectomycorrhizal fungi and soil saprotrophs: Implications for decomposition of organic matter in soils and degradation of organic pollutants in the rhizosphere. *Can. J. Bot.* **2002**, *80* (8), 803–809.
9. Dickie, I.A.; Koide, R.T.; Steiner, K.C. Influences of established trees on mycorrhizas, nutrition, and growth of *Quercus rubra* seedlings. *Ecol. Monogr.* **2002**, *72* (4), 505–521.
10. Lovelock, C.E.; Miller, R. Heterogeneity in inoculum potential and effectiveness of arbuscular mycorrhizal fungi. *Ecol. Monogr.* **2002**, *83* (3), 823–832.
11. van der Heijden, M.G.A.; Sanders, I.R. *Mycorrhizal Ecology*; Springer-Verlag: Heidelberg, 2002.
12. Smith, S.; Smith, A. *Diversity and Integration in Mycorrhizal Symbiosis*; Kluwer Academic: Dordrecht, 2002.

Mycorrhizae: Soil Quality

Ibrahim Ortaç

*Department of Soil Science and Plant Nutrition, University of Cukurova,
Adana, Turkey*

Abstract

Soil is a dynamic and complex natural system, which is interactively regulated by physical, chemical, and biological properties and processes to provide ecosystem services including crop production. The soil quality as the critical component of the ecosystems is associated with soil's usefulness or performance to environmental quality as well as food quality. Soil quality is related to the development of many soil properties such as providing soil health, reflecting the fitness of a soil body, supporting water quality, sustaining plant productivity, and promoting human health. Soil is a sink for soil organic carbon (SOC). SOC, infiltration, aggregation, pH, microbial biomass, nitrogen forms, topsoil depth, conductivity or salinity, and available nutrients are the important indicators for soil quality. Mycorrhizal fungi also have an effect on soil quality in terms of increase in SOC, aggregation, nutrient uptake, and also increase in soil capacity for better health. The strength of the mycorrhizal association and its relationship with soil structure depends on root morphology and mycorrhizal hyphae. Arbuscular mycorrhizal fungi inoculation often increases the success of the ecological restoration as well. Mycorrhizal fungi are also effect on the soil function and capacity to increase productivity, resilience, and health functions. Also mycorrhizae increase SOC content, rhizosphere fertility, aggregation, porosity, and bulk density and reduce the stress factors that are all increase the soil quality.

WHAT IS SOIL QUALITY?

Soil is a dynamic and complex natural system, which is interactively regulated by physical, chemical, and biological properties and processes to provide ecosystem services including crop production. Soil as part of natural systems is a basis for life and habitat for soil organisms, i.e., plants. Soil's ability and functions in ecosystem to sustain plant and animal productivity and to maintain or enhance water and soil quality are critically important for human health and habitation. The soil quality as the critical component of the ecosystems is associated with soil's usefulness or performance to environmental quality as well as food quality. Soil has a significant contribution on filter of water and nutrient cycles and buffer of the soil reactions. Karlen et al.^[1] indicated that organic matter, infiltration, aggregation, pH, microbial biomass, N forms, topsoil depth, conductivity or salinity, and available nutrients are the important indicators for soil quality. Moreover, biological activity, enzymes, total organic carbon (C) and nitrogen (N), cation exchange capacity, aggregation, bulk density, and water retention and release capacity are very sensitive and important indicators of soil quality.^[2] Although there is not a complete consensus that has yet reached on the definition of soil quality, soil quality is usually related to soil organic matter (SOM) content, number and diversity of organisms, microbial by-products, hydrologic characteristics, aggregate stability, etc.

Aggregation Is a Very Important Parameter of Soil Quality

The dynamics of soil aggregate characteristics are often associated with changes in management practices and environment. Generally, soils in harsh environment are characterized by poor and compacted soil structures, low water-holding capacity and SOM content, and widespread nutrient deficiency. For a successful crop production or vegetation, it is necessary to improve soil quality and the ability of the plant's adaptation.^[3]

Management of SOM is important to maintain soil quality including soil physical characteristics. Soils, in general, have varied C storage capacity and aggregate size distribution. Soil organic carbon (SOC), as the main component of SOM, is an important determinant of the soil physical quality influencing water-holding capacity, porosity, aggregation, and fertility.^[4] The SOC content with other essential nutrients directly affects the production of biomass by improving soil quality.^[5] Kavdir and Smucker^[6] reported that soil structure influences several important soil functions such as soil productivity, biological activity, root growth, soil stability, and nutrient cycling as well. Borie et al.^[7] reported that soil structure stability is strongly influenced by the nature and content of SOM. Moreover, land use and management practices influencing SOM content are determinant of soil aggregation. Soil aggregates especially both macro- and

microaggregates exhibited a significant positive relationship with SOC and particulate organic matter.^[8]

WHAT IS THE ROLE OF MYCORRHIZAE IN SOIL?

Arbuscular mycorrhizal fungi (AMF) are the largest symbiotic associations between plants and fungi, which make significant contribution on physical, chemical, and biological aspects of soil quality through AMF hyphae extending into the rhizosphere and thereby improving the absorption of nutrients especially phosphorus (P) and micronutrients.^[9–11] The establishment of mycorrhiza causes changes in the physiology of the host plants. Like other soil microorganisms, AMF act as ecosystem engineers on roots and root surface of the plants. The AMF as soil microorganisms play a contributory effect on soil aggregate formation because of the symbiosis that significantly changes the root functioning.^[12] Thus, mycorrhiza has a significant impact on soil resilience, which is also an important component of soil quality.

Several studies have reported that soil biology especially mycorrhizal fungi significantly influences soil fertility and soil quality. However, the study on applied mycorrhizae is very new avenue in the field of soil quality research. The AMF also influence other soil functions, such as C, N, and P cycling to support plant growth and nutrition in the agroecosystems. Mycorrhizal inoculated plants are effective competitors in P-limited soil conditions. Several studies reported that mycorrhizal associations improve soil structures and increase uptake of nutrients and water through the extension of AMF hyphae distribution within the rhizosphere.^[9–11] Medina and Azcon^[3] suggested that soil amended with mycorrhizal fungi inoculation can be used as a successful biostrategy to improve plant performance in P-deficient soils under Mediterranean conditions. As shown in Table 1, mycorrhizal inoculation under field conditions increased the N, P, and K contents of sweet corn.

Mycorrhizal inoculation has been found to decrease the concentrations of salts and electrical conductivity of the

soil.^[14] The AMF colonization can also improve plant tolerance to stress conditions caused by drought and salt. Cekic et al.^[15] showed that under different salt level applications, mycorrhizal inoculation increased relative water content, P, total chlorophyll, and carotenoid content of pepper plants.

Mycorrhizae and Roots Effect on Soil Quality

Soil quality is also related to the rhizosphere beneficial microorganisms, such as AMF and bacteria. The AMF are an active and major component of the soil microbial community and have an important role on soil quality. Barea et al.^[16] indicated that symbiotic mycorrhizal associations are fundamental in optimizing plant fitness and soil quality. The AMF symbiosis has been proposed as one of the most effective mechanisms of plant heavy metal tolerance and water stress avoidance due to their effect on rhizosphere physical, chemical, and biological activities.^[10] The AMF are expected to have a synergistic effect on crops by improving the soil quality. The potential role of AMF in soil quality is related to improved soil physical properties.^[17] The contribution of mycorrhizal inoculation to the SOC content and aggregate formation depends on roots and hyphae productivity, roots and hyphae turnover rates, and their exudation. They improve soil quality by binding microaggregates and primary soil particles together and consequently resulting in: 1) improved soil structure; 2) increased water infiltration and retention; and 3) reduced soil erosion by minimizing leaching and runoff, while making them available to plants. As expected, plant roots are different and have significant effect on soil aggregation. Plant roots and AMF have direct and indirect binding effects on soil aggregate formation and stability. Jastrow et al.^[18] have provided the evidence related to the direct effects of mycorrhizal hyphae on soil aggregation. Soil aggregates play an important role in sequestering and stabilizing SOC.^[19] Other soil properties are also affected by plant roots and rhizosphere by significantly changing the soil physical, chemical, and biological properties.^[20]

Nadian et al.^[21] demonstrated that colonized Berseem clover (*Trifolium alexandrinum* L.) root length was improved by 20%, as soil aggregate diameter increased. Six et al.^[22] indicated that plant roots influence soil structure in diverse ways such as through root exudation, which includes polysaccharides that help to bind microaggregates and clay particles, and also through root pressure and penetration, which help to increase the proportion of stable aggregates through root entanglement. Mycorrhizae inoculated root also has mycorrhizosphere, which expanded rhizosphere volume and improved soil aggregate formation. The adherence of soil particles and sand grains on the root via AMF mycelia was pictured by Ortaş in field grown sorghum plant (Fig. 1).

The strength of the mycorrhizal association and its relationship with soil structure depend on root morphology^[24]

Table 1 The effect of P application and mycorrhizal inoculation on N, P, and K concentration (%) in shoots of sweet corn at silking.

Treatments	N	P	K
–Mycorrhizae			
P0	2.31 ± 0.01	0.16 ± 0.01	0.80 ± 0.28
P1	2.44 ± 0.02	0.20 ± 0.00	0.90 ± 0.14
P2	2.44 ± 0.12	0.22 ± 0.01	0.60 ± 0.03
+Mycorrhizae			
P0	2.60 ± 0.00	0.22 ± 0.01	1.20 ± 0.28
P1	2.84 ± 0.12	0.23 ± 0.01	1.30 ± 0.14
P2	2.58 ± 0.11	0.24 ± 0.03	1.00 ± 0.28

Source: From Ortaş & Sari.^[13]

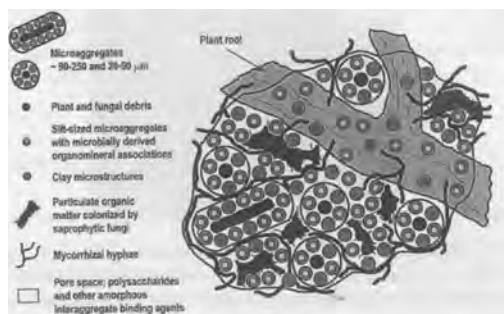


Fig. 1 Aggregate formation by the plant roots and AM mycelium.

Source: From Miller & Jastrow.^[23]

and mycorrhizal hyphae. AMF inoculation often increases the success of the ecological restoration.^[25] Rillig^[12] indicated that AMF are one of the important factors of soil quality through their effects on soil ecological interactions and their contributions to maintaining soil structure. However, AMF contributions on soil physical development depend on several factors and have a significant effect on ecosystem services. The ecosystem services provided by AMF are based on the modification of the root morphology and development of a complex ramified mycelia network in soil. These interactions improve plant–soil adherence and soil stability (binding action and improvement of soil structure), which in turn increase mineral nutrient and water uptake contributions by plants. It has been demonstrated that the mycorrhizal mycelial network can have a binding action on the soil and improve soil structure.^[26–28]

A major contribution of AMF in soils is their role on the maintenance of soil structure and plays a significant role in maintaining soil quality. The elongation of AMF from the mycorrhizal root to soil is a complex and ramifying network into the surrounding soil that can reach up to 30 m of fungal hyphae per gram of soil.^[29,30] Fig. 2 shows the adherence of soil particles and sand grains on the roots via AM mycelia in soil aggregation conceptual diagram.^[23]

Organic amendment and mycorrhizae are the well-known aggregate building agents.^[17] Extensive soil-based mycelium of fungus around roots can help soil aggregation processes (Fig. 3).

Mycorrhizal Hyphae, Glomalin, and Soil Quality

AMF enhance soil aggregate stability due to the production of extraradical hyphae and a protein known as glomalin. Glomalin, a brown to red–brown colored glycoprotein produced by AMF, is a major component of SOM that is defined operationally by the extraction method.^[31] The AMF, fed on C, found living on plant roots and appear to be the only producer of glomalin. A laboratory procedure reveals glomalin accumulation on soil aggregates as the green material (Fig. 4). Wright and



Fig. 2 Aggregate formation by the AM mycelia.

Source: From Ortas, unpublished data.

Upadhyaya^[32] have shown a strong non-linear relationship of the AMF hyphae product “glomalin” with water-stable aggregates across several soil types.

Glomalin was reported to be a non-water soluble, highly persistent glycoproteinaceous substance^[32] produced in mycorrhizal fungal cell walls, and it remains in soil after hyphae died.^[33] A strong correlation has been found between glomalin and water-stable aggregates in a wide variety of soils where organic material is the main binding agent.^[12,35] Wright and Upadhyaya^[36] indicated that increased aggregate stability, which leads to better soil structure, in turn, leads to control soil erosion for better plant production.

Bedini et al.^[37] showed that the secretion by AMF glomalin contributes to soil stability and water retention. Furthermore, it has been indicated that higher levels of

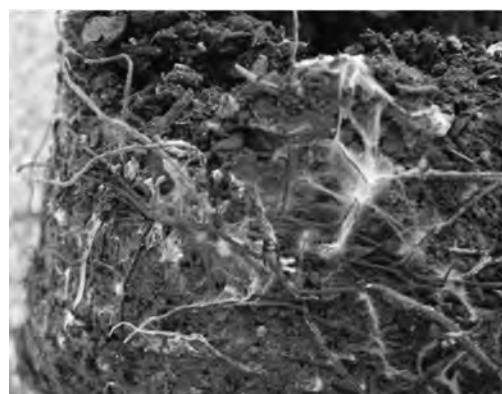


Fig. 3 Mycorrhizal hyphae through soil profile.

Source: Photo by Ortas, unpublished data.

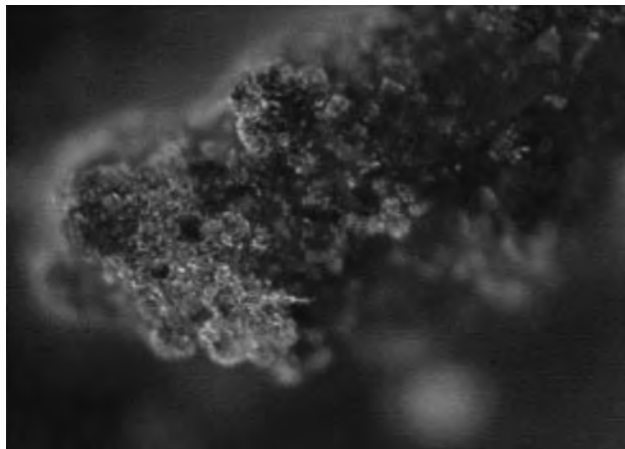


Fig. 4 Glomalin under microscope.
Source: From Nichols.^[34]

glomalin increase water infiltration, improve porosity and air permeability, better root development, improve microbial efficiency, and increase resistance to surface sealing and erosion. Driver et al.^[33] indicated that glomalin is thought to be deposited in soil by the degradation of extraradical hyphae of AMF. Bedini et al.^[37] indicated that (AM) fungi are key organisms of the soil–plant system, influencing and contributing to soil aggregation and soil structural stability by the combined action of extraradical hyphae and of a hydrophobic proteinaceous substance named glomalin. Rillig et al.^[38] shown that AMF hyphae and their glomalin products had significant contribution on water-stable aggregates of soil.

Treseder and Turner^[39] postulated that the role of glomalin in the ecosystem is not very clear and should be studied thoroughly under long-term and diverse management systems. Glomalin as a strong cementing agent is produced by a beneficial fungus that grows on plant roots. When hyphae stop transporting water, their protective glomalin sloughs off into the soil, where it helps soil build defenses against degradation and erosion and boosts its productivity.

As mycorrhizae produce enzymes (such as alkaline phosphatase), the activity of phosphatase can increase plant nutrient uptake as well.^[40] Phosphatase enzymes

are developed by plants for an effective P acquisition and possibly for N and less mobile micronutrients.

EFFECT OF MYCORRHIZA INOCULATIONS ON SOIL QUALITY UNDER LONG-TERM FIELD CONDITIONS

Long-term effects of mycorrhizal inoculation (after 14 years) on soil quality were presented in Table 2. Water-stable aggregation (WSA) had significantly affected by the organic (mycorrhiza + compost 10 ton ha⁻¹) and inorganic fertilizer treatments after 14 years' long-term experiment and ranged from 71.1% to 87.6%. The highest WSA was observed in compost + mycorrhizae treatment (87.6%), whereas the lowest WSA was found in control treatment (71.13%); mineral fertilizer has 74.6% of WSA. Mean weight diameter (MWD) of aggregates increased by 44% under the mycorrhizae + compost treatment as compared with the control.

Bedini et al.^[37] showed that MWD values of soil aggregates were positively correlated with the values of total hyphae length and hyphae density of the AMF. They also showed that the MWD of macroaggregates was significantly higher in mycorrhizal inoculated soils as compared to the non-mycorrhizal soils. Ortas et al.^[17] showed that the high values of MWD were associated with mycorrhizal inoculation rather than with application of organic fertilizer and that the MWD depended more on mycorrhizae application than on SOC content. Wilson et al.^[30] also reported that AMF abundance was a dominant factor affecting variability in soil WSA and MWD. The development of soil structure and the dynamics of WSA in many soils are closely related to the SOM cycling and the magnitude of root growth and mycorrhizal hyphae.^[41,42] Smith and Read^[10] concluded that the contribution of mycorrhizal root plus hyphae on aggregation is more than those of the hyphae alone, and that under pasture, the contribution of root and hyphae to WSA increases with increase in SOC content.

There were also significant differences in SOM content among the treatments (Table 2). The SOM content was 2.01% and 2.31% for chemical fertilizer and compost + mycorrhizae treatments, respectively, as compared with the SOM content in control (1.50%). Increasing SOM increases

Table 2 Effect of mineral fertilizer and mycorrhizae + compost application on soil properties after 14 years' long-term field experiment.

	SOM (%)	Bulk density (pb; g cm ⁻³)	Total porosity (St; %)	Soil-available water (cm ⁻³)	WSA (%)	MWD
Treatments						
Control	1.50	1.46	45	8.69	71.13	1.47
Mineral fertilizer	2.01	1.47	48	9.45	74.72	1.43
Mycorrhizal + compost	2.31	1.31	53	11.48	87.60	2.12

Source: From Ortas, unpublished data.

aggregate stability, infiltration, nutrient mineralization and availability, and cation exchange capacity. It suggests that mycorrhizal inoculation increased SOM content, which is directly related to soil quality. According to Harris et al.^[43] soil quality assessment scorecard, SOC has high score range. Soil bulk density, porosity, and soil water availability were significantly affected by mycorrhizal inoculation compared with the chemical fertilizer treatment.

There were also significant differences in total porosity among the treatments. As discussed earlier, Ortas et al.^[17] showed that soils treated with the compost + mycorrhizae organic fertilizers have the highest total porosity (St). The porosity was 45%, 48%, and 53% for control, chemical fertilizer, and compost + mycorrhizae treatments, respectively (Table 2).

Soil quality is related to the development of many soil properties such as providing soil health, reflecting the fitness of a soil body, supporting water quality, sustaining plant productivity, and promoting human health. Mycorrhizal fungi are also effect on the soil function and capacity to increase productivity, resilience, and health functions. Also mycorrhizae increase SOC content, rhizosphere fertility, aggregation, porosity, and bulk density and reduce the stress factors that are all increase the soil quality. Accordingly, it seems that mycorrhizal fungi have significant contribution on soil quality development. Usually, soil physical and chemical components were used for soil quality assessment; it may be possible to use mycorrhizae as a biological component as well. Usually, healthy soils have a rich diversity of soil useful microorganisms including mycorrhizal fungi. Mycorrhizal infection and spore germination can also be used as a soil quality indicator.

REFERENCES

- Karlen, D.L.; Mausbach, M.J.; Doran, J.W.; Cline, R.G.; Harris, R.F.; Schuman, G.E. Soil quality: A concept, definition, and framework for evaluation (A Guest Editorial). *Soil Sci. Soc. Am. J.* **1997**, *61*, 4–10.
- Askari, M.S.; Holden, N.M. Indices for quantitative evaluation of soil quality under grassland management. *Geoderma* **2014**, *230*, 131–142.
- Medina, A.; Azcon, R. Effectiveness of the application of arbuscular mycorrhiza fungi and organic amendments to improve soil quality and plant performance under stress conditions. *J. Soil Sci. Plant Nutr.* **2010**, *10*, 354–372.
- Tisdall, J.M.; Oades, J.M. Organic matter and water-stable aggregates in soils. *J. Soil Sci.* **1982**, *33*, 141–163.
- Lal, R. Soil quality impacts of residue removal for bio-ethanol production. *Soil Till. Res.* **2009**, *102*, 233–241.
- Kavdir, Y.; Smucker, A.J.M. Soil aggregate sequestration of cover crop root and shoot-derived nitrogen. *Plant Soil* **2005**, *272*, 263–276.
- Borie, F.; Rubio, R.; Morales, A. Arbuscular mycorrhizal fungi and soil aggregation. *Rev. Cienc. Suelo Nutr.* **2008**, *8*, 9–18.
- Mikha, M.M.; Benjamin, J.G.; Vigil, M.F.; Nielson, D.C. Cropping intensity impacts on soil aggregation and carbon sequestration in the Central Great Plains. *Soil Sci. Soc. Am. J.* **2010**, *74*, 1712–1719.
- Karandashov, V.; Bucher, M. Symbiotic phosphate transport in arbuscular mycorrhizas. *Trends Plant Sci.* **2005**, *10*, 22–29.
- Smith, S.E.; Read, D.J. *Mycorrhizal Symbiosis*; Academic Press: San Diego, 2008.
- Ortas, I. Effect of selected mycorrhizal inoculation on phosphorus sustainability in sterile and non-sterile soils in the Harran Plain in South Anatolia. *J. Plant Nutr.* **2003**, *26*, 1–17.
- Rillig, M.C. Arbuscular mycorrhizae, glomalin, and soil aggregation. *Can. J. Soil Sci.* **2004**, *84*, 355–363.
- Ortas, I.; Sari, N. Enhanced yield and nutrient content of sweet corn with mycorrhizal inoculation under field conditions. *Agricultura Mediterranea* **2003**, *133* (3–4), 188–195.
- Li, Y.; Chen, Y.L.; Li, M.; Lin, X.-G.; Liu, R.-J. Effects of arbuscular mycorrhizal fungal communities on soil quality and the growth of cucumber seedlings in a greenhouse soil of continuously planted to cucumber in North China. *Pedosphere* **2012**, *22*, 79–87.
- Cekic, F.O.; Unyayar, S.; Ortas, I. Effects of arbuscular mycorrhizal inoculation on biochemical parameters in *Cap-sicum annuum* grown under long term salt stress. *Turk. J. Bot.* **2012**, *36*, 63–72.
- Barea, J.M.; Palenzuela, J.; Cornejo, P.; Sánchez-Castro, I.; Navarro-Fernández, C.; Lopéz-García, A.; Estrada, B.; Azcón, R.; Ferrol, N.; Azcón-Aguilar, C. Ecological and functional roles of mycorrhizas in semi-arid ecosystems of Southeast Spain. *J. Arid. Environ.* **2011**, *75*, 1292–1301.
- Ortas, I.; Akpınar, C.; Lal, R. Long-term impacts of organic and inorganic fertilizers on carbon sequestration in aggregates of an entisol in Mediterranean Turkey. *Soil Sci.* **2013**, *178*, 12–23.
- Jastrow, J.D.; Miller, R.M.; Lussenhop, J. Contributions of interacting biological mechanisms to soil aggregate stabilization in restored prairie. *Soil Biol. Biochem.* **1998**, *30*, 905–916.
- Sundermeier, A.P.; Islam, K.R.; Raut, Y.; Reeder, R.C.; Dick, W.A. Continuous no-till impacts on soil biophysical carbon sequestration. *Soil Sci. Soc. Am. J.* **2011**, *75*, 1779–1788.
- Ortas, I. Determination of the extent of rhizosphere soil. *Commun. Soil Sci. Plan.* **1997**, *28*, 1767–1776.
- Nadian, H.; Hashemi, M.; Herbert, S.J. Soil aggregate size and mycorrhizal colonization effect on root growth and phosphorus accumulation by berseem clover. *Commun. Soil Sci. Plan.* **2009**, *40*, 2413–2425.
- Six, J.; Bossuyt, H.; Degryze, S.; Denef, K. A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. *Soil Till. Res.* **2004**, *79*, 7–31.
- Miller, R.M.; Jastrow, J.D. Mycorrhizal fungi influence soil structure. In *Arbuscular Mycorrhizas: Physiology and Function*; Kapulnik, Y., Douds, D.D., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 3–18.
- Miller, R.M.; Jastrow, J.D. Hierarchy of root and mycorrhizal fungal interactions with soil aggregation. *Soil Biol. Biochem.* **1990**, *22*, 579–584.

25. Jeffries, P.; Gianinazzi, S.; Perotto, S.; Turnau, K.; Barea, J.-M. The contribution of arbuscular mycorrhizal fungi in sustainable maintenance of plant health and soil fertility. *Biol. Fert. Soils* **2003**, *37*, 1–16.
26. Tisdall, J.M.; Smith, S.E.; Rengasamy, P. Aggregation of soil by fungal hyphae. *Aust. J. Soil Res.* **1997**, *35*, 55–60.
27. Rillig, M.C.; Wagner, M.; Salem, M.; Antunes, P.M.; George, C.; Ramke, H.-G.; Titirici, M.-M.; Antonietti, M. Material derived from hydrothermal carbonization: Effects on plant growth and arbuscular mycorrhiza. *Appl. Soil Ecol.* **2010**, *45*, 238–242.
28. Caravaca, F.; Alguacil, M.M.; Azcón, R.; Roldán, A. Formation of stable aggregates in rhizosphere soil of *Juniperus oxycedrus*: Effects of AM fungi and organic amendments. *Appl. Soil Ecol.* **2006**, *33*, 30–38.
29. Cavagnaro, T.R.; Smith, F.A.; Smith, S.E.; Jakobsen, I. Functional diversity in arbuscular mycorrhizas: Exploitation of soil patches with different phosphate enrichment differs among fungal species. *Plant Cell Environ.* **2005**, *28*, 642–650.
30. Wilson, G.W.T.; Rice, C.W.; Rillig, M.C.; Springer, A.; Hartnett, D.C. Soil aggregation and carbon sequestration are tightly correlated with the abundance of arbuscular mycorrhizal fungi: Results from long-term field experiments. *Ecol. Lett.* **2009**, *12*, 452–461.
31. Nichols, K.A.; Wright, S.F. Comparison of glomalin and humic acid in eight native U.S. soils. *Soil Sci.* **2005**, *170*, 985–997.
32. Wright, S.F.; Upadhyaya, A. A survey of soils for aggregate stability and glomalin, a glycoprotein produced by hyphae of arbuscular mycorrhizal fungi. *Plant Soil* **1998**, *198*, 97–107.
33. Driver, J.D.; Holben, W.E.; Rillig, M.C. Characterization of glomalin as a hyphal wall component of arbuscular mycorrhizal fungi. *Soil Biol. Biochem.* **2005**, *37*, 101–106.
34. Nichols, K.A. Glomalin production and accumulation in soilless pot cultures. *Can. J. Soil Sci.* **2010**, *90*, 567–570.
35. Rillig, M.C.; Ramsey, P.W.; Morris, S.; Paul, E.A. Glomalin, an arbuscular-mycorrhizal fungal soil protein, responds to land-use change. *Plant Soil* **2003**, *253*, 293–299.
36. Wright, S.F.; Upadhyaya, A. Extraction of an abundant and unusual protein from soil and comparison with hyphal protein of arbuscular mycorrhizal fungi. *Soil Sci.* **1996**, *161*, 575–586.
37. Bedini, S.; Pellegrino, E.; Avio, L.; Pellegrini, S.; Bazzoffi, P.; Argese, E.; Giovannetti, M. Changes in soil aggregation and glomalin-related soil protein content as affected by the arbuscular mycorrhizal fungal species *Glomus mosseae* and *Glomus intraradices*. *Soil Biol. Biochem.* **2009**, *41*, 1491–1496.
38. Rillig, M.C.; Wright, S.F.; Eviner, V.T. The role of arbuscular mycorrhizal fungi and glomalin in soil aggregation: Comparing effects of five plant species. *Plant Soil* **2002**, *238*, 325–333.
39. Treseder, K.K.; Turner, K.M. Glomalin in ecosystems. *Soil Sci. Soc. Am. J.* **2007**, *71*, 1257–1266.
40. Tarafdar, J.C.; Marschner, H. Phosphatase-activity in the rhizosphere and hyphosphere of VA mycorrhizal wheat supplied with inorganic and organic phosphorus. *Soil Biol. Biochem.* **1994**, *26*, 387–395.
41. Rillig, M.C.; Mardatin, N.F.; Leifheit, E.F.; Antunes, P.M. Mycelium of arbuscular mycorrhizal fungi increases soil water repellency and is sufficient to maintain water-stable soil aggregates. *Soil Biol. Biochem.* **2010**, *42*, 1189–1191.
42. Hallett, P.D.; Feeney, D.S.; Bengough, A.G.; Rillig, M.C.; Scrimgeour, C.M.; Young, I.M. Disentangling the impact of AM fungi versus roots on soil structure and water transport. *Plant Soil* **2009**, *314*, 183–196.
43. Harris, R.F.; Karlen, D.L.; Mulla, D.J. A conceptual framework for assessment and management of soil quality and health. In *Methods for Assessing Soil Quality*; Doran, J.W., Jones, A.J., Eds.; SSSA: Madison, 1996; 61–82.

Nanofertilizers

Ruiqiang Liu

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

The development and application of new types of fertilizers using nanotechnology are one of the potentially effective options for significant enhancement of the global food production. This entry summarizes reviews related to either engineered or natural nanomaterials, which improve overall plant growth and could be used as nanofertilizers for increasing agronomic yields. A list of the relevant nanomaterials is presented, and the entry is focused on chemical composition, particle size, particle concentration applied, plant species, and plant growth enhancements. In addition, this entry classifies nanofertilizers into the following four categories: macronutrient nanofertilizers [e.g., apatite nanoparticles (NPs), calcium carbonate NPs, and magnesium oxide NPs], micronutrient nanofertilizers (e.g., iron oxide NPs, manganese oxide NPs, zinc oxide NPs, and copper oxide NPs), nutrient-loaded nanofertilizers [zeolites, silica NPs, and carbon nanotubes (CNTs)], and plant growth stimulating nanomaterials (titanium oxide NPs and CNTs). Each category is discussed for its own specific importance, research direction, and requirements in new fertilizer development. This entry suggests that development of nitrogen and phosphorus macronutrient nanofertilizers are a high research priority for sustainable agriculture and environmental protection.

INTRODUCTION

The capacity of arable land to sustain human has increased from 1.9 to 4.3 person ha⁻¹ between 1908 and 2008. This increase is partly due to the invention and application of synthetic mineral fertilizers. Hence, given the limited additional arable lands and scarce water resources in the world, the development and application of new types of high-efficiency fertilizers are among the practical options for feeding the projected 9.6 billion population (a ~30% increase from 2012) by 2050 without seriously damaging ecosystems and the environment. Nanotechnology is a promising approach in new fertilizer development. Indeed, some new findings, as seen in this entry, in nanotechnology research have revealed encouraging perspectives in this regard. So, the major objective of this entry is to analyze and synthesize the knowledge on these nanoparticles (NPs) and nanomaterials, which enhance plant growth and yields while reducing the risks of environmental disruptions associated with the conventional fertilizer uses (such as eutrophication or nitrate contamination). These nanomaterials are potentially important as innovative fertilizers to meet challenges of food security and environmental protection.

Nanomaterials and Nanofertilizers

Nanomaterials are defined as materials that have a single unit, with size between 1 nanometer (nm) and 100 nm, in at least 1-D. Accordingly, nanofertilizers are either

nanomaterials that can supply one or more nutrients to the plants and enhance plant growth and yields or those that can improve the performance of conventional fertilizers but do not directly provide crops with nutrients. For convenience of discussion in this entry, the former is collectively called nanofertilizers and the latter as nanomaterial-enhanced fertilizers. Furthermore, nanofertilizers can be classified as macronutrient nanofertilizers and micronutrient nanofertilizers. Compared with the conventional nanofertilizers, these nanofertilizers are expected to significantly improve crop growth and yields, to enhance the efficiency of fertilizer use and reduce nutrients losses, and/or to minimize the adverse environmental impacts.

MACRONUTRIENT NANO FERTILIZERS

Macronutrient nanofertilizers are chemically composed of one or more macronutrient elements such as nitrogen (N), phosphorus (P), potassium (K), magnesium (Mg), and calcium (Ca), thus being able to supply one or more of these essential elements to plants. Total global consumption of conventional macronutrient fertilizers (N + phosphorus pentoxide + potassium oxide) was estimated at 176 million ton (Mt) in 2011 and is projected to increase to 263 Mt in 2050,^[1] indicating the critical role of these macronutrient fertilizers in agriculture and in food production. Moreover, due to the low efficiency (30–50%) and heavy application of these fertilizers, significant amounts of these nutrients

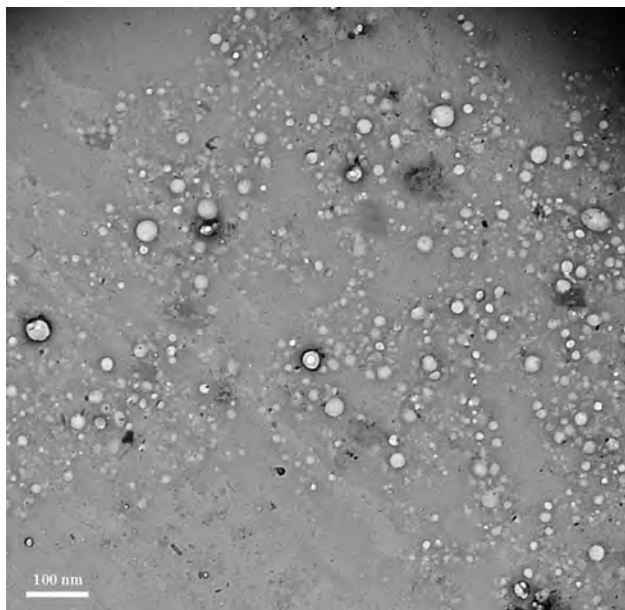


Fig. 1 A transmission electron microscopic image of hydroxyapatite NPs.

Source: Adapted from Liu & Lal.^[2]

(N and P) end up in surface and groundwater bodies, disrupting aquatic ecosystems and threatening human health. Therefore, an urgent and practical research direction is to develop highly efficient and environmentally friendly macronutrient (especially N and P) nanofertilizers to replace the conventional nanofertilizers for sustainable food production and environmental protection.

Liu and Lal^[2] synthesized a new type of hydroxyapatite ($\text{Ca}_5(\text{PO}_4)_3\text{OH}$) NPs of ~16 nm in size (Fig. 1) and assessed the effect of NPs as a P fertilizer on soybean (*Glycine max*) growth in a greenhouse experiment. Results showed that NPs' application increased the growth rate and seed yield by 32.6% and 20.4%, respectively, compared to those of a regular P fertilizer. Biomass productions were enhanced by 18.2% (aboveground) and 41.2% (belowground), respectively. Data revealed that soybean plants could absorb apatite NPs as an effective P nutrient source for normal metabolism. In addition, the NPs may also have supplied Ca nutrient. Using apatite NPs as a new class of P fertilizer can potentially enhance agronomical yield, increase efficiency of P application, and reduce risks of water eutrophication. Liu et al.^[3] and Delfani et al.^[4] reported using CaNPs (calcium carbonate, 20–80 nm, 160 mg L⁻¹ as Ca) and MgNPs (500 mg L⁻¹ as Mg, others unknown) that Ca and Mg fertilizers are used for enhancing peanut (*Arachis hypogaea*) and black-eyed pea (*Vigna unguiculata* subsp. *unguiculata*) growth.

MICRONUTRIENT NANOFERTILIZERS

Plant micronutrients include iron (Fe), manganese (Mn), zinc (Zn), copper (Cu), Mg, and molybdenum (Mo). Compared

with the plant macronutrients, only trace levels of micronutrients are required for healthy growth of crops as shown in the composition of Hoagland solution.^[5] Micronutrients are often added to the N, P, and K fertilizers at small amounts as soluble salts for crop uptake. Micronutrients in these composite fertilizers usually provide enough nutrients and cause no environmental damages. However, plant availability of the applied micronutrients may become low, and micronutrient deficiency may occur in some soils with alkaline pH, coarse texture, or low organic matter content.^[6] Micronutrient nanofertilizers are expected to enhance uptake by plants even in such worst-case scenarios. Table 1 includes a list of micronutrient NPs that have been tested and have exhibited potentials as micronutrient nanofertilizers.

NANOMATERIAL-ENHANCED FERTILIZERS

Nanomaterial-enhanced fertilizers are defined as these nanomaterials, when loaded with plant nutrient(s), aimed at increasing plant-uptake efficiency of the nutrient(s) and/or reducing the adverse impacts of fertilizer application, but the nanomaterials themselves do not contain or supply the targeted nutrient(s). An important example of this type is nutrient-loaded zeolites.

Nutrient-loaded Zeolite Nanofertilizers

Zeolites' particles are often not present at nanoscales. But the arrangement of aluminum and silicon in the 3-D framework of silicon tetroxide and AlO_4 tetrahedra of zeolites creates the channels and cages that are within nanoscale (0.3–10 nm in diameter). As a result, zeolites are termed as materials with nanostructures. Due to their unique nanoporous properties, zeolites often have extremely high specific surface area, high cation exchange capacity, and high selectivity toward plant macronutrients (potassium ion and ammonium ion) over others. Thus, these plant-essential elements may be absorbed onto zeolite exchange sites, where the nutrients can subsequently be released slowly for plant uptake, thereby reducing migration of these nutrients into groundwater or runoff and decreasing the potential for environmental pollution. Moreover, volatilization of gaseous N [e.g., as ammonia or dinitrogen (N_2)] can also be reduced, as the adsorbed N is unavailable for conversion to these gaseous phases by soil microbes. So far, zeolites have been an ideal candidate for improving N efficiency, reducing nutrient losses, and eliminating environmental risks.^[13] A few examples showing the benefits of nutrient-loaded zeolite nanofertilizers under field conditions are provided in several references.^[14–17]

Other Nanoparticulate Fertilizer Carriers

Laboratory observations indicate that some NPs (silica, Fe oxides, carbon-coated Fe, and polymers) can enter

Table 1 NPs reportedly enhanced plant growth by providing micronutrients and could be used as micronutrient fertilizers.

Nutrient provided	NP type, particle size, and concentration	Test plant, method, and medium	Growth enhancement		
			Over control 1 (no target nutrient)	Over control 2 (regular fertilizer with target nutrient)	Other factors
Micronutrient Fe	Superparamagnetic iron oxide NPs, ^[7] Fe ₃ O ₄ , 18.9–20.3 nm, 30, 45, and 60 mg L ⁻¹	Soybean, nutrient solution, perlite medium, 7-day greenhouse test	No reported	Fe-EDTA, chlorophyll content, (1.1) ^a at 30 mg L ⁻¹ and (1) at 45 mg L ⁻¹ but (0.8) at 60 mg L ⁻¹	N/A
Micronutrient Mn	Metallic Mn, ^[8] 20 nm, 0.05, 0.1, 0.5, and 1.0 mg L ⁻¹	Mung bean (<i>Vigna radiate</i>) plant, nutrient solution, perlite medium, 15-day in growth chamber	Root length (1.4–1.5), shoot length (1.2–1.4), dry weight (1–2), chlorophyll (1.1–1.8), and carotenoid (1.1–2) content, photosynthesis rate (1.1–1.7)	MnSO ₄ , root length (1–1.6), shoot length (1.1), dry weight (1–2), chlorophyll (1–1.6), and carotenoid (1.1–1.4) content, photosynthesis rate (1.1–1.3)	MnNPs did not show phytotoxic as MnSO ₄ did
Micronutrient Zn	ZnO, ^[9] 20 nm, 1–2000 mg L ⁻¹	Mung bean and chickpea (<i>Cicer arietinum</i>) seedlings, agar medium, 60 hr in an incubator	Shoot height (1.3) and biomass (1.6), root length (1.9) and biomass (1.4) for Mung bean at 20 mg L ⁻¹ ZnNPs; (1), (2.8), (1.5), (1.3) for chickpea at 1 mg L ⁻¹ ZnNPs	No reported	ZnNPs at levels higher than the optimum showed phytotoxicity
	ZnO, ^[10] 10 nm, 400 and 800 mg kg ⁻¹ soil	Cucumber (<i>Cucumis sativus</i>), soil mixture, a life cycle, 53-day greenhouse test	Root dry mass (1.1–1.6), fruit starch (1.1–1.6), glutelin (0.9–2), Zn (1.7–2.5)		No negative effects of ZnNPs found
Micronutrient Cu	70% CuO and 30% Cu ₂ O, ^[11] 30 nm, 0.025, 0.25, 0.5, 1, and 5 mg L ⁻¹ as Cu	<i>Elodea densa planch</i> , 3-day incubation, water	Photosynthesis rate (1–1.4) at <0.5 mg L ⁻¹	CuSO ₄ ; photosynthesis rate (1.4–3) at <0.5 mg L ⁻¹ ; leaf Cu (5) at <0.5 mg L ⁻¹	CuSO ₄ inhibitory at all concentrations
Micronutrient Mo	Mo, ^[12] 100–250 nm, 8 mg L ⁻¹ , others unknown	Chickpea, seed-soaked 1–2 hr, a life cycle, soil, rhizosphere examination	Nodule number/mass (11/6.2), activity of antioxidant enzymes (1.9–2.6), symbiotic bacteria (1.9–3.2)	Not reported	Microbial treatment further enhanced the effects

^aThe enhancement value in parentheses represents chlorophyll content under NPs treatment/chlorophyll content under control, and so on.

plant tissues/cells and transport DNA and chemicals inside the tissues.^[7,13–21] These findings led many researchers to speculate on using these NPs to deliver nutrients to the plants as a new fertilizing or nurturing technique. However, there are no specific reports on using these NPs as nutrient carriers to enhance plant growth and productivity.

NEW NANOPARTICULATE PLANT GROWTH ENHANCERS WITH UNCLEAR MECHANISMS

There are a few reports on other types of NPs that could enhance plant growth although these particles do not contain any essential plant nutrients. Typical examples of this

type are titanium dioxide (TiO₂) NPs (TiNPs) and carbon nanotubes (CNTs).

Titanium dioxide NPs

Traditionally, titanium (Ti) was not regarded as an essential nutrient for plants. Additionally, soils usually contain high levels of Ti ranging from 0.1% to 0.9%, and an average of ~0.03 mg L⁻¹ Ti is often reported in soil solution.^[22] Thus, the application of Ti to soil is not necessary for agronomic production. However, a few reports^[23,24] indicate that TiNP application (on exposure to sunlight) may increase plant photosynthesis efficacy, enhance plant enzymes activity, and improve availability of N through chemical fixation of N₂ in the air, thereby promoting plant growth. There was

Table 2 Research needs for nanofertilizers.

Nanofertilizer type	Importance in food security and sustainable agriculture	Research direction	Other important research items
Macronutrient	Very important	N and P fertilizers with high fertilizing capacity, high efficiency, low leaching rate, and low risks of environmental contamination	
Micronutrient	Not important	High efficiency at extreme soil pH, or in soils with low organic matter content, or coarse texture	To compare with regular fertilizers in plant growth/yield enhancements
NP carriers	Important	Increasing N and P use efficiency and decreasing nutrient leaching	To present nanotoxicity study of any new type of nanofertilizers
Plant growth simulating NPs	Important	More evidence and mechanism clarification	

a report published in the 1970s showing that leaf spray of chelate Ti (1 mg L^{-1}) enhanced sugar and chlorophyll contents in apples (*Malus domestica*), grapes (*Vitis vinifera*), and sugar beets (*Beta vulgaris*).^[25] But negative effects of TiNPs at 4 g L^{-1} on seed germination have also been reported.^[26]

Carbon Nanotubes

Some reports^[27–29] indicate that CNTs in low doses could be stimulators of seed germination and plant growth. Khodakovskaya et al.^[27] proposed that the surface charges of CNTs affected growth and expression of water channel proteins in tomato, thereby enhancing water uptake and utilization efficiency.

RESEARCH NEEDS AND FUTURE DIRECTIONS

Review of the nanotechnology research on enhancing plant growth and yield indicates potential applications of those nanofertilizers for crop production. The importance, research direction, and research priority of each type of nanofertilizers are summarized in Table 2.

CONCLUSION

Plants can utilize nutrient-containing NPs as essential mineral nutrient sources through roots or leaves, thereby maintaining or enhancing their metabolism, anti-pathogen capacity, growth, and yield. Some of these NPs lead to better performance than associated conventional fertilizers. Moreover, some inert NPs, when augmented with nutrients, can either enhance the fertilizer use efficiency or minimize environmental disruptions. Some other inert NPs, although supplying no nutrients, can improve crop yields by enhancing their physiological activities such as photosynthesis or water uptake/uses. Therefore, research and development of nanofertilizers is fairly important to global sustainable development. Among these, macronutrient nanofertilizer

and macronutrient-loaded NPs (especially for N and P) have a high research priority.

REFERENCES

- Alexandratos, N.; Bruinsma, J. *World Agriculture Towards 2030/2050: The 2012 Revision*, ESA Working Paper No. 12–03, Rome, FAO. Available at <http://www.fao.org/docrep/016/ap106e/ap106e.pdf> (accessed June 2014).
- Liu, R.; Lal, R. Synthetic apatite nanoparticles as a phosphorus fertilizer for soybean (*Glycine max*). *Sci. Rep.* **2014**, *4*, 5686–5691.
- Liu, X.M.; Zhang, F.D.; Zhang, S.Q.; He, X.S.; Wang, R.F.; Feng, Z.B.; Wang, Y.J. Responses of peanut to nano-calcium carbonate. *Plant Nutr. Fert. Sci.* **2005**, *11* (3), 385–389.
- Delfani, M.; Firouzabadi, M.B.; Farrokhi, N.; Makarian, H. Some physiological responses of black-eyed pea to iron and magnesium nanofertilizers. *Commun. Soil Sci. Plan.* **2014**, *45* (4), 530–540.
- Hoagland, D.R.; Arnon, D.I. *The Water-culture Method for Growing Plants without Soil*; University of California, College of Agriculture, Agricultural Experiment Station: Berkeley, 1950.
- Fageria, N.K. *The Use of Nutrients in Crop Plants*; CRC Press: Boca Raton, 2009.
- Ghafariyan, M.H.; Malakouti, M.J.; Dadpour, M.R.; Stroeve, P.; Mahmoudi, M. Effects of magnetite nanoparticles on soybean chlorophyll. *Environ. Sci. Technol.* **2013**, *47*, 10645–10652.
- Pradhan, S.; Patra, P.; Das, S.; Chandra, S.; Mitra, S.; Dey, K.K.; Akbar, S.; Palit, P.; Goswami, A. Photochemical modulation of biosafe manganese nanoparticles on *Vigna radiata*: A detailed molecular, biochemical, and biophysical study. *Environ. Sci. Technol.* **2013**, *47*, 13122–13131.
- Mahajan, P.; Dhoke, S.K.; Khanna, A.S. Effect of nano-ZnO particle suspension on growth of mung (*Vigna radiata*) and gram (*Cicer arietinum*) seedlings using plant agar method. *J. Nanotechnol.* **2011**, *2011*, 1–7.
- Zhao, L.; Peralta-Videa, J.R.; Rico, C.M.; Hernandez-Viezcas, J.A.; Sun, Y.; Niu, G.; Servin, A.; Nunez, J.E.; Duarte-Gardea, M.; Gardea-Torresdey, J.L. CeO_2 and ZnO nanoparticles change the nutritional qualities of cucumber (*Cucumis sativus*). *J. Agr. Food Chem.* **2014**, *62*, 2752–2759.

11. Nekrasova, G.F.; Ushakova, O.S.; Ermakov, A.E.; Uimin, M.A.; Byzov, I.V. Effects of copper (II) ions and copper oxide nanoparticles on *Elodea densa planch.* Russ. J. Ecol. **2011**, *42* (6), 458–463.
12. Taran, N.Y.; Gonchar, O.M.; Lopatko, K.G.; Batsmanova, L.M.; Patyka, M.V.; Volkogon, M.V. The effect of colloidal solution of molybdenum nanoparticles on the microbial composition in rhizosphere of *Cicer arietinum* L. Nanoscale Res. Lett. **2014**, *9*, 289–297.
13. Ming, D.W.; Allen, E.R. Use of natural zeolites in agronomy, horticulture, and environmental soil remediation. In *Natural Zeolites: Occurrence, Properties, and Applications: Mineralogical Society of America Reviews in Mineralogy and Geochemistry* (v. 45); Bish, D.L., Ming, D.W., Eds.; The Mineralogical Society of America: Chantilly, 2001; 619–654.
14. Perrin, T.S.; Drost, D.T.; Boettinger, J.L.; Norton, J.M. Ammonium-loaded clinoptilolite: A slow-release nitrogen fertilizer for sweet corn. J. Plant Nutr. **1998**, *21*, 515–530.
15. Malekian, R.; Abedi-Koupai, J.; Eslamian, S.S. Influences of clinoptilolite and surfactant-modified clinoptilolite zeolite on nitrate leaching and plant growth. J. Hazard. Mater. **2011**, *185*, 970–976.
16. Hershey, D.R.; Paul, J.L.; Carlson, R.M. Evaluation of potassium-enriched clinoptilolite as a potassium source for potting media. Hortic. Sci. **1980**, *15*, 87–89.
17. Zwingmann, N.; Mackinnon, I.D.R.; Gilkes, R.J. Use of a zeolite synthesized from alkali treated kaolin as a K fertilizer: Glasshouse experiments on leaching and uptake of K by wheat plants in sandy soil. Appl. Clay Sci. **2011**, *53*, 684–690.
18. Liu, Y.; Laks, P.; Heiden, P. Controlled release of biocides in solid wood. I. Efficacy against brown rot wood decay fungus (*Gloeophyllum trabeum*). J. Appl. Polym. Sci. **2002**, *86*, 596–607.
19. Torney, F.; Trewyn, B.G.; Lin, V.S.-Y.; Wang, K. Mesoporous silica nanoparticles deliver DNA and chemicals into plants. Nat. Nanotechnol. **2007**, *2*, 295–300.
20. Ambrogio, M.W.; Frascioni, M.; Yilmaz, M.D.; Chen, X. New methods for improved characterization of silica nanoparticle-based drug delivery systems. Langmuir **2013**, *29*, 15386–15393.
21. González-Melendi, P.; Fernández-Pacheco, R.; Coronado, M.J.; Corredor, E.; Testillano, P.S.; Risueño, M.C.; Marquina, C.; Ibarra, M.R.; Rubiales, D.; Pérez-de-Luque, A. Nanoparticles as smart treatment-delivery systems in plants: Assessment of different techniques of microscopy for their visualization in plant tissues. Ann. Bot. **2008**, *101*, 187–195.
22. Kabata-Pendias, A.; Pendias, H. *Trace Elements in Soils and Plants*; CRC Press: Boca Raton, 1984.
23. Yang, F.; Liu, C.; Gao, F.; Su, M.; Wu, X.; Zheng, L.; Hong, F.; Yang, P. The improvement of spinach growth by nano-anatase TiO₂ treatment is related to nitrogen photoreduction. Biol. Trace Elem. Res. **2007**, *119*, 77–88.
24. Song, G.; Gao, Y.; Wu, H.; Hou, W.; Zhang, C.; Ma, H. Physiological effect of anatase TiO₂ nanoparticles on *Lemna minor*. Environ. Toxicol. Chem. **2012**, *31* (9), 2147–2152.
25. Pais, I.; Fehér, M.; Farkas, E.; Szabó, Z.; Cornides, I. Titanium as a new trace element. Commun. Soil Sci. Plan. **1977**, *8* (5), 407–410.
26. Castiglione, M.R.; Giorgetti, L.; Geri, C.; Cremonin, R. The effects of nano-TiO₂ on seed germination, development and mitosis of root tip cells of *Vicia narbonensis* L. and *Zea mays* L. J. Nanopart. Res. **2011**, *13*, 2443–2449.
27. Khodakovskaya, M.V.; Kim, B.S.; Kim, J.N.; Alimohammadi, M.; Dervishi, E.; Mustafa, T.; Cernigla, C.E. Carbon nanotubes as plant growth regulators: Effects on tomato growth, reproductive system, and soil microbial community. Small **2013**, *9* (1), 115–123.
28. Cañas, J.E.; Long, M.; Nations, S.; Vadan, R.; Dai, L.; Luo, M.; Ambikapathi, R.; Lee, E.H.; Olszyk, D. Effects of functionalized and nonfunctionalized single-walled carbon nanotubes on root elongation of select crop species. Environ. Toxicol. Chem. **2008**, *27* (9), 1922–1931.
29. Lahiani, M.H.; Dervishi, E.; Chen, J.; Nima, Z.; Gaume, A.; Biris, A.S.; Khodakovskaya, M.V. Impact of carbon nanotube exposure to seeds of valuable crops. ACS Appl. Mater. Interfac. **2013**, *5*, 7965–7973.

Natural Resources: Interaction with Soil

Dennis J. Greenland

University of Reading, Reading, U.K.

Abstract

To preserve life as we know it, it is essential that we conserve our soils and increase their productivity. In the past, there have been several examples of the end of civilizations because of failure to control soil degradation. Only by careful conservation of our soils, and technical innovations to increase their productivity while avoiding environmental damage, can a similar fate for our modern civilization be prevented.

INTRODUCTION

It is impossible to overestimate the importance of soils. All life on earth, except for fish and other aquatic species, depends on the soil. The very first living things may have evolved in the oceans, but emergence onto the land followed and developed because of the unique properties of the soil. The relation of humans to the soil has long been recognized in religious beliefs. Hillel^[1] traces these relationships in Jewish beliefs, in the sayings of Buddha, in the Koran, and in early Greek civilization. The common theme is that earth is the mother of life. In some places, the earth had to be made fertile by water. In ancient Egypt, the waters of the Nile were worshipped, and in India, the Ganges is known as the Mother of India. But in China, the Huanghe (formerly known as the Yellow River) is called the Sorrow of China because of the huge quantities of silt eroded from the Loessial soils in its catchment and the fearsome floods that have devastated the downstream areas for thousands of years.

It is not only the interaction of soil with water that creates its unique and essential properties but also its interaction with all natural resources. The soil has been rightly described as the attic of the earth sciences and the basement of the plant sciences. Soil is formed from rocks, and its interaction with air and water enables plants to grow. Forests and grasslands, as well as the crops on which we live, are all dependent on soil. To grow, plants need air, water, and nutrients. Soil is the great storehouse of the nutrients essential for life. Not only does soil contain all essential nutrients, but it also exchanges them with soil solution in such a way that there is seldom an excess and normally a sufficient amount to allow the development of a varied population of invertebrates and microorganisms.

Nitrogen is an exception. It is seldom present in sufficient quantities within the rocks from which the soil was formed to support an active soil population. However, among the earliest organisms were those with the ability to process atmospheric nitrogen. The presence of air within

the soil provides the nitrogen for these organisms; as these organisms die, their remains are attacked by other organisms and their nitrogen partially assimilated and partially added to the nitrogen in soil organic matter.

THE UNIQUE PROPERTIES OF SOILS

To support life, soil must enable water to be stored, so that it does not drain away under the pull of gravity but can be available to organisms in and on the soil and be retained for plant use. The soil must also allow air to move into it, so that oxygen is available and carbon dioxide may escape. This requires the soil to have a structure consisting of large cracks and pores; large pores will drain under gravity (transmission pores), and smaller pores will hold water against gravity but not hold it so strongly that plants are unable to use it (storage pores).

Such a structure is possible because of the wide range of soil particle sizes, from fine [<2 micron equivalent spherical diameter (μ esd)] clay particles to silt particles (2 – 20μ esd) and larger sand, gravel, and stones ($>20 \mu$ esd). The physical forces associated with swelling and shrinkage—due to wetting and drying and freezing and thawing—create cracks and pores within the soil, between which are denser aggregates containing a wide range of smaller pores. Worms and other soil animals also create relatively large transmission pores, and fungi, bacteria, and plant roots release mucilaginous material to coat the sides of the pores and strengthen them against subsequent compression.

Thus, the special structure of soil essential to the development of the natural vegetation is built. This structure is able to provide water and nutrients, allow gases to move in and out, and provide channels through which plant roots can grow and seedlings can emerge. Once the vegetation is established, it contributes to the soil by adding organic matter as litter decays and nutrients are recycled. The organic matter feeds the soil population and contributes

to the exchange of nutrients between soils and organisms. When the organic matter decomposes, it releases nitrogen, sulfur, and phosphorus contained within to feed other organisms.

Among the materials that make up soil organic matter are many acidic compounds (e.g., humic and fulvic acids), which adsorb important nutrient cations and exchange them with others in the soil solution.

When clays are formed by minerals weathering, they too retain nutrient cations at their surfaces. These cations are held either by negative charges—which develop from isomorphous replacement of higher-charged cations with lower-charged cations from within the clay lattice—or by dissociation from acidic groups on the surfaces of hydrous oxides.^[2]

THE FORMATION AND DEGRADATION OF SOILS

Soils tend to be formed slowly over thousands of years, as rocks weather, organic matter accumulates, and as eluviation, sedimentation, and erosion occur under the natural effects of climate and topography, modified by the influence of vegetation.^[3,4] These processes which have given rise to the fertile soils of the world may be drastically modified by human intervention, which can both enhance and reduce the ability of soils to sustain life on earth. Soils are used in many ways to improve human life, but their misuse or exploitation can also destroy the essential features that have made them the essence of life on earth.

SOIL AND CIVILIZATION

Two hundred years ago, Malthus^[5] warned the world of the dangers that arise as the demands of a growing population place increasing stress on world resources. Evans^[6] has traced how the world population has grown and how it has produced the food to sustain it. Until the world population reached 3 billion in 1960, the main source of greater food production was the cultivation of more land. Since 1960, the world population has doubled. The extra food needed to sustain a population of six billion in 1999 had come from increased crop yields, associated with higher-yielding plant varieties and the use of fertilizers and pesticides—techniques of the Green Revolution. There is scope for further production increase where good quality water is available, more land can be irrigated, and the number of crops grown per year increased. The Food and Agriculture Organization (FAO) of the United Nations believes that there is still scope for more land to be cultivated for arable production,^[7] but Young^[8] and others believe that the FAO estimates are too high. Most good land was cultivated first; subsequently, agriculturists have been forced to develop poorer and more fragile land that is susceptible to degradation. Further, because the good land was close to cities and the cities have grown, the

alienation of land for buildings, roads, and factories has affected good land more than poor land.

Much of the essential resource for sustaining life has been lost by the alienation of good land and the degradation of poorer land. This is perhaps most vividly illustrated by erosion. The rate of natural erosion may be enormously accelerated by actions that expose the soil directly to rainfall, without the intervention of a forest or grass cover. Such actions mostly derive from deforestation to allow the land to be used for agriculture or animal production.^[8,9] Although there are several sources of soil degradation, erosion is the most widespread and that which most commonly leads to irreversible losses of productivity.^[10]

The continued intensification of land use that has led to increasing degradation of land and water supplies has brought the world closer to the Malthusian Precipice,^[11] where the rate of population growth exceeds the rate of increase in food production.

To preserve life as we know it, it is essential that we conserve our soils and increase their productivity. In the past, there have been several examples of the ends of civilizations because of a failure to control soil degradation.^[1,12] Only by careful conservation of our soils, and technical innovations to increase their productivity while avoiding environmental damage, can a similar fate for our modern civilization be prevented.

REFERENCES

1. Hillel, D. *Out of the Earth*; University of California Press: Berkeley, 1991.
2. Greenland, D.; Hayes, M. *Chemistry of Soil Constituents*; Wileys: Chichester, 1989.
3. Jenny, H. *Factors of Soil Formation*; McGraw Hill: New York, 1941.
4. Buol, S.W.; Hole, F.D.; McCracken, R.J. *Soil Genesis and Classification*; Iowa State University Press: Ames, 1973.
5. Malthus, T.R. *An Essay on the Principle of Population as It Affects the Future Improvement of Society*; Johnson, St.-Paul's Churchyard: London, 1798.
6. Evans, L.T. *Feeding the Ten Billion: Plants and Population Growth*; Cambridge University Press: Cambridge, 1998.
7. Alexandratos, N. *World Agriculture: Towards 2010. An FAO Study*; FAO, Rome, John Wiley and Sons: Chichester, 1995.
8. Young, A. *Land Resources: Now and for the Future*; Cambridge University Press: Cambridge, 1998.
9. Lal, R. *Soil Erosion in the Tropics*; McGraw Hill: New York, 1994.
10. Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 99–118.
11. Gregory, D.J.; Greenland, P.J.; Nye, P.H. *Land Resources: On the Edge of the Malthusian Precipice?* CAB International: Wallingford, 1998.
12. Hyams, E. *Soil and Civilisation*; Thames and Hudson: London, 1952.

Nematodes

Christien H. Ettema

Soil Biology Group, Wageningen Agricultural University, Wageningen,
the Netherlands

Abstract

Nematodes are generally microscopic animals, belonging to the phylum Nematoda. They are very numerous, representing 80% of all multicellular animals. Nematodes live in soil, freshwater, and marine environments. In addition, they occur as parasites of vertebrate (including humans) and invertebrate animals (such as insects). Soil nematodes live in water-filled pore spaces or water films surrounding soil particles. A handful of soil contains thousands of individuals, dozens of species, and several feeding groups. Plant disease caused by nematodes results in billions of dollars lost worldwide. However, most nematode species are beneficial—they enhance plant growth by promoting nutrient cycling and contribute to biological control of plant enemies, including insects, fungi, and plant-parasitic nematodes.

HISTORY

Free-living (non-parasitic) nematodes were first discovered in 1656 A.D., when Borellus observed *Turbatrix aceti* in natural vinegar, using one of the first microscopes.^[1] In 1743, Needham gave the first account of a plant-parasitic nematode, *Anguina tritici*, from wheat galls. In the mid-19th century, nematology developed as an independent branch of zoology. Because the study of soil nematodes has been driven by the economic impact of plant parasites, free-living nematodes have comparatively received little attention. However, the most intensively studied nematode is the free-living *Caenorhabditis elegans*, which is considered a model for genomic, developmental, and neurobiological systems with relevance to all of biology.^[2]

CLASSIFICATION

Taxonomic Classification

As nematology is a relatively young science, nematode taxonomy is in great flux, with no single classification agreed upon by all taxonomists. Taxonomic classification is generally based on morphological characteristics, but molecular phylogenetic research challenges the resulting higher level classification.^[3] Usually, Nematoda are divided into two classes, the Adenophorea (Aphasmsidia) and the Secernentea (Phasmsidia), primarily based on the absence or presence of posterior sense organs, the phasmids. Examples of soil-dwelling orders are given in Fig. 1. A comprehensive overview of taxonomic literature is provided by Bongers.^[4]

Trophic Classification

The soil nematode fauna typically includes several trophic (feeding) groups. Nematodes may be bacterial feeders (predominantly Rhabditida, Araeolaimida, and Monhysterida), fungal feeders (Aphelenchida, Dorylaimida, and Tylenchida), plant parasites and plant associates (Tylenchida, Aphelenchida, and Dorylaimida), carnivores (Mononchida and Enoplida), omnivores (Dorylaimida), algal feeders (Chromadorida, Desmodorida, and Enoplida), and insect parasites (Rhabditida). Trophic groups are generally distinguished by the mouth structure and esophageal shape. However, as some taxa lack the typical anterior morphology, it is recommended that nematodes be identified to genus and classified trophically according to Yeates, Bongers, et al.^[5] A second note of caution is that unequivocal classification into trophic groups is not always possible, because of opportunistic feeding habits, life-stage dependent diets, or lack of dependable information.^[5]

Life History Classification

For environmental monitoring, soil nematodes are classified into five life history groups, ranging from r-selected “colonizers” (commonly members of Rhabditida) to K-selected “persisters” (mostly Dorylaimida). Their relative abundances are weighted using the maturity index, indicating the level of soil disturbance.^[6] However, the challenge remains to avoid ad hoc descriptions and establish clear cause-and-effect relationships between nematode community changes and soil disturbance.^[7] A bibliography on environmental monitoring with nematodes is provided by Bongers.^[8]

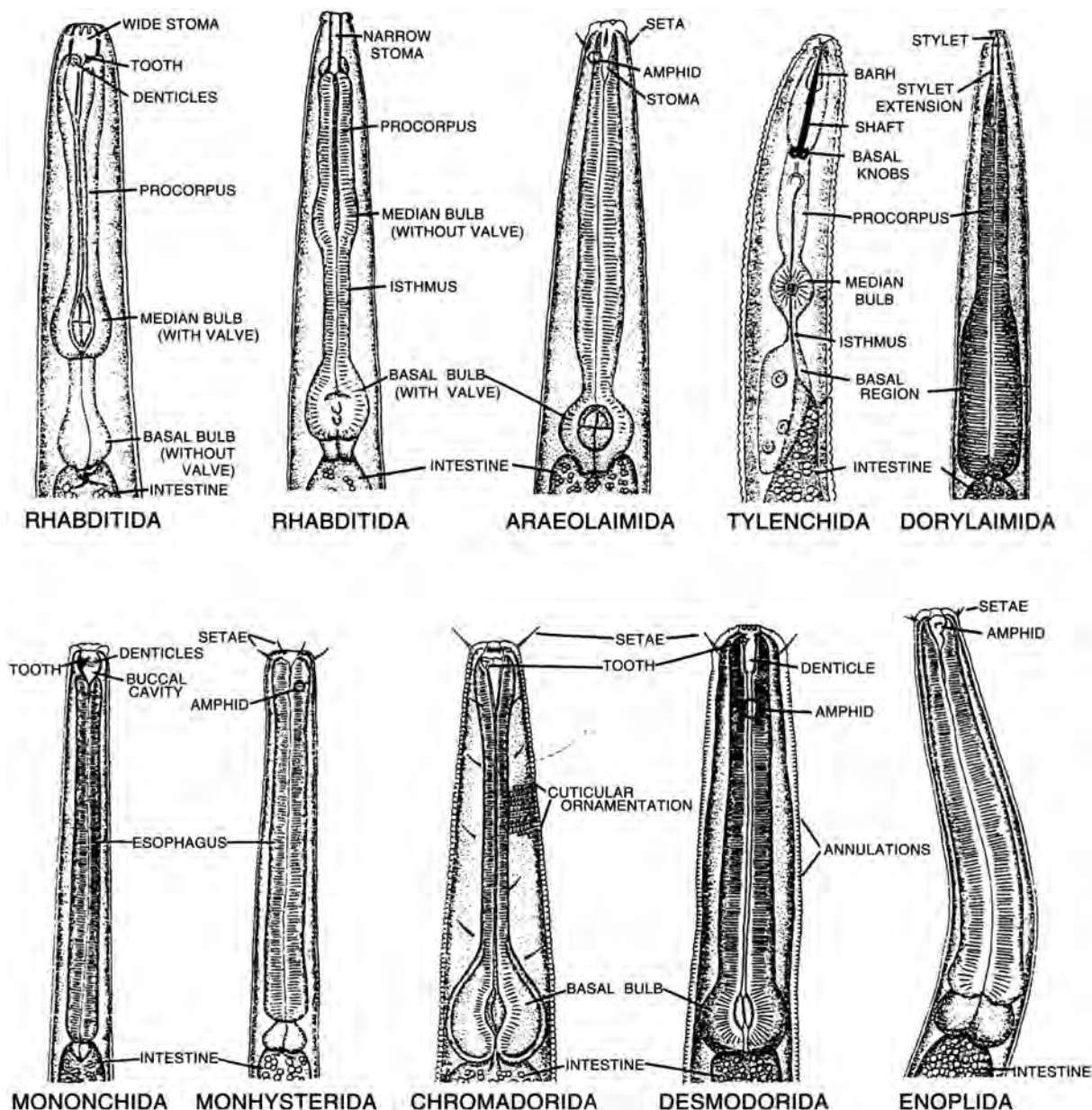


Fig. 1 The soil-dwelling nematode orders. Note that the order Aphelenchida is missing, as Ingham^[9] considered it a suborder (Aphelenchina) of the Tylenchida, following the classification used by Golden.^[10]

Source: From Ingham.^[9]

MORPHOLOGY

Nematodes are bilaterally symmetrical, non-segmented pseudocoelomates. Their body plan is simply a “tube within a tube.” The outer tube is the body wall, consisting of a flexible cuticle and longitudinal muscles, with which sinusoidal locomotion is achieved. The inner tube is the alimentary tract, which, starting at the anterior end (Fig. 1), consists of a stoma or stylet, esophagus, intestine, rectum, and anus. Nematodes are normally bisexual, but for some species reproduction is parthenogenetic

or hermaphroditic.^[11] Nematodes have excretory and nervous systems but lack discrete circulatory or respiratory systems. Internal structures are easily observed with substage lighting under a microscope. In the nematode life cycle, which depending on species and environment varies from 1 week to >1 year, the egg stage is followed by four juvenile stages before adulthood. In response to adverse environmental conditions (e.g., drought), nematodes may enter anabiosis, a state of metabolic dormancy that may last for years. Extensive information on nematode structure can be found in Bird and Bird.^[11]

BIOGEOGRAPHY

Distribution and Abundance

Soil nematodes are found in all biomes, with topsoil abundances ranging from 50,000/m² in Antarctic deserts^[12] up to tens of millions per square meter in temperate grasslands and deciduous forests.^[13] Generally, nematodes occur wherever there is moisture and food available. Because these resources have heterogeneous distributions, nematode populations are highly aggregated, which is a concern for quantitative soil sampling.^[14] Active migration is limited in range, but nematodes may be dispersed over long distances (notably in anabiotic form) by wind, water, insects, birds, and mammals.^[1]

Species Diversity

Less than 6000 terrestrial nematode species have been described, but it is estimated that there are at least 1,00,000 species.^[15] Species richness is generally greatest in temperate broadleaf forest (62 species per soil sample) followed by cultivated soil, grassland, tropical rainforest, temperate coniferous forest, and polar vegetation.^[16] The structural and functional implications of nematode biodiversity are being debated.^[17]

HARMFUL NEMATODES

Plant-parasitic nematodes can cause great damage to agricultural crops. Ectoparasitic nematodes stay outside the root and use their stylet to puncture root cells (e.g., the feeding nematode *Belonolaimus* and the stunt nematode *Tylenchorhynchus*). Endoparasitic nematodes enter the root and move around (the lesion nematode *Pratylenchus*) or stay in one feeding site (the root-knot nematode *Meloidogyne* and the cyst nematodes *Globodera* and *Heterodera*). The resulting root damage can cause wilting, stunting, nutrient deficiencies, and yield losses. Some species transmit viruses (*Longidorus*, *Trichodorus*, and *Xiphinema*). In other cases, nematodes may cause damage to above ground plant parts (the stem nematode *Ditylenchus dipsaci* and the wheat gall nematode *A. tritici*). Worldwide, nematode damage may amount to \$78 billion annually.^[18] Nematode problems of various crops and climates are reviewed by Nickle and Luc et al.^[19,20] An important observation is that low levels of root herbivory could promote plant growth by stimulating root branching and exudation, which in turn enhances rhizosphere microbial activity and nutrient availability.^[21]

Nematode control may include sanitary measures, crop rotation, use of resistant cultivars, application of nematicides and organic amendments, intercropping with nematode-antagonistic crops, and physical methods such as tillage, steaming, and solarization.^[22] Nematodes have



Fig. 2 A predacious nematode feeding on a gravid bacterivorous nematode. The predator itself is infected with *Pasteuria penetrans* (the “warts” attached to the predator’s cuticle are bacterial endospores).

numerous natural enemies including soil bacteria, fungi, predatory nematodes, and insects (Fig. 2),^[23,24] but soil communities are not easily manipulated for biological control. Promising results have been obtained with *Verticillium chlamydosporium*, an egg parasite of cyst and root-knot nematodes.^[25] The nematode-trapping fungus *Arthrobotrys* is commercially available and controls nematodes in cultivated mushroom and tomatoes.^[23]

BENEFICIAL NEMATODES

Microbial-Feeding Nematodes

Microbial-feeding nematodes (bacterivores and fungivores) may promote plant growth by enhancing nutrient mineralization, notably of nitrogen.^[26] As microbial-feeding nematodes have a higher carbon-to-nitrogen ratio than microbes, and low production efficiency, they mineralize the nitrogen immobilized in microbial biomass. In addition, they can stimulate microbial activity (and thus nitrogen mineralization) by reducing microbial competition, enhancing oxygen diffusion, and transporting microbial propagules to new substrates. Using knowledge of microbivorous nematode population dynamics, the application of organic amendments can be timed to optimize nitrogen availability to crops.^[27]

Predacious and Omnivorous Nematodes

By controlling microbivorous population, predacious (Fig. 2) and omnivorous nematodes prevent “overgrazing” of microbes and thereby indirectly control nutrient cycling processes.^[28] In terms of biological control, it is unlikely that predacious nematodes could significantly contribute to plant-parasitic nematode control.^[29] Their generation times are too long for a timely response to

plant-parasite population peaks, and there is an evidence that predacious nematodes prefer microbial feeding, not plant-feeding, nematodes as diet. Predacious and omnivorous nematodes may be useful indicators of soil ecosystem health, because of their long life cycles and sensitivity to disturbance.^[6]

Entomopathogenic Nematodes

Entomopathogenic (insect-parasitic) nematodes are successfully used in biological control of pest insects, including white grubs, leaf miners, and cockroaches.^[30] Entomopathogenic nematodes (order Rhabditida) carry a symbiotic bacterium. After penetrating the host insect, the nematode releases the bacterium, which kills the host within 72 hours, providing food to the nematode, which then multiplies inside the cadaver. Infective juveniles disperse to new hosts. (For detailed information on entomopathogenic nematodes, see Gaugler and Kaya.^[31])

METHODS

Methods for collection and extraction are reviewed by Nickle, Ingham, and McSorley.^[9,19,32] For ecological work with nematodes, useful methods are compiled by Wheeler et al.^[33] A bibliography of nematode identification keys is provided by Bongers,^[4] and an interactive diagnostic key to plant-parasitic, free-living, and predacious nematodes is found in <http://ianrwww.unl.edu/ianr/plntpath/nematode/key/nemakey.htm>.^[34]

REFERENCES

1. Poinar, G.O. Jr. *The Natural History of Nematodes*; Prentice Hall: Englewood Cliffs, 1983; 323 pp.
2. Caenorhabditis elegans, <http://helios.bto.ed.ac.uk/mbx/fgn/wow/celegans.html> (accessed August 2000).
3. Blaxter, M.L.; De Ley, P.; Garey, J.R.; Liu, L.X.; Scheldeman, P.; Vierstraete, A.; Vanfleteren, J.R.; Mackey, L.Y.; Dorris, M.; Frisse, L.M.; Vida, J.T.; Thomas, W.K. A molecular evolutionary framework for the phylum Nematoda. *Nature* **1998**, *392*, 71–75.
4. Bongers, T. *Nematode Identification Literature*. Available at http://www.spg.wau.nl/nema/ident_lit.html (accessed August 2000).
5. Yeates, G.W.; Bongers, T.; De Goede, R.G.M.; Freckman, D.W.; Georgieva, S.S. Feeding habits in soil nematode families and genera—An outline for soil ecologists. *J. Nematol.* **1993**, *25*, 315–331.
6. De Goede, R.G.M.; Bongers, T. The nematode maturity index. In *Techniques in Nematode Ecology*; Wheeler, T., Forge, T., Caswell-Chen, E., LaMondia, J.A., Eds.; Society of Nematologists, 2000. Available at <http://ianrwww.unl.edu/son/deGoede.htm> (accessed August 2000).
7. Niles, R.K.; Freckman, D.W. From the ground up: Nematode ecology in bioassessment and ecosystem health. In *Plant and Nematode Interactions*; Barker, K.R., Pederson, G.A., Windham, G.L., Eds.; Agronomy Monograph 36; American Society of Agronomy: Madison, 1998; 65–85.
8. Bongers, T. *Maturity Index Literature*. Available at http://www.spg.wau.nl/nema/MI_lit.htm (accessed August 2000).
9. Ingham, R.E. Nematodes. In *Methods of Soil Analysis. Part 2—Microbiological and Biochemical Properties*; Bigham, J.M., Ed.; Soil Science Society of America: Madison, 1994; 459–490.
10. Golden, A.M. Classification of the genera and higher categories of the order Tylenchida (Nematoda). In *Plant Parasitic Nematodes*; Zuckerman, B.M., Mai, W.F., Rohde, R.A., Eds.; 1971; Vol. 1, 191–232.
11. Bird, A.F.; Bird, J. *The Structure of Nematodes*; Academic Press: London, 1991.
12. Freckman, D.W.; Virginia, R.A. Low-diversity Antarctic soil nematode communities: Distribution and response to disturbance. *Ecology* **1997**, *78*, 363–369.
13. Sohlenius, B. Abundance, biomass and contribution to energy flow by soil nematodes in terrestrial ecosystems. *Oikos* **1980**, *34*, 186–194.
14. Webster, R.; Boag, B. Geostatistical analysis of cyst nematodes in soil. *J. Soil Sci.* **1992**, *43*, 583–595.
15. Andrassy, I. A short census of free-living nematodes. *Fund. Appl. Nematol.* **1992**, *15*, 187–188.
16. Boag, B.; Yeates, G.W. Soil nematode biodiversity in terrestrial ecosystems. *Biodiv. Conserv.* **1998**, *7*, 617–630.
17. Ettema, C.H. Soil nematode diversity: Species coexistence and ecosystem function. *J. Nematol.* **1998**, *30*, 159–169.
18. Wheeler, T.; Forge, T.; Caswell-Chen, E.; LaMondia, J.A., Eds. *Plant and Soil Nematodes: Societal Impact and Focus for the Future*; A Report by the Committee on National Needs and Priorities in Nematology; Society of Nematologists: Lawrence, 1994.
19. Nickle, W.R. *Manual of Agricultural Nematology*; Marcel Dekker: New York, 1991.
20. Luc, M.; Sikora, R.A.; Bridge, J. *Plant-Parasitic Nematodes in Subtropical and Tropical Agriculture*; CAB International: Wallingford, 1990.
21. Denton, C.S.; Bardgett, R.D.; Cook, R.; Hobbs, P.J. Low amounts of root herbivory positively influence the rhizosphere microbial community in a temperate grassland soil. *Soil Biol. Biochem.* **1999**, *31*, 155–165.
22. NEMABASE. *A Database of the Host Status of Plants to Plant-parasitic Nematodes*. Available at <http://www.ipm.ucdavis.edu/NEMABASE/index.html> (accessed August 2000).
23. Poinar, G.O. Jr.; Jansson, H.B. *Diseases of Nematodes*; CRC Press: Boca Raton, FL, 1988; Vol. I and II.
24. *Natural Enemies of Nematodes: Biological Control Images*. Available at <http://sacs.cpes.peachnet.edu/nemabc/> (accessed August 2000).
25. *Verticillium Chlamydosporium and Biological Control of Phytoparasitic Nematodes*. Available at <http://www.area.ba.cnr.it/~e085ac01/bkfair3444.html> (accessed August 2000).
26. Freckman, D.W. Bacterivorous nematodes and organic-matter decomposition. *Agr. Ecosyst. Environ.* **1988**, *24*, 195–217.

27. Ferris, H.; Venette, R.C.; Lau, S.S. Dynamics of nematode communities in tomatoes grown in conventional and organic farming systems, and their impact on soil fertility. *Appl. Soil Ecol.* **1996**, *3*, 161–175.
28. Hunt, H.W.; Coleman, D.C.; Ingham, E.R.; Ingham, R.E.; Elliott, E.T.; Moore, J.C.; Rose, S.L.; Reid, C.P.P.; Morley, C.R. The detrital food web in a short grass prairie. *Biol. Fert. Soils* **1987**, *3*, 57–68.
29. Yeates, G.W.; Wardle, D.A. Nematodes as predators and prey: Relationships to biological control and soil processes. *Pedobiologia* **1996**, *40*, 43–50.
30. Smart, G.C. Jr. Entomopathogenic nematodes for the biological control of insects. *J. Nematol.* **1995**, *4-S*, 529–534.
31. Gaugler, R.; Kaya, H. *Entomopathogenic Nematodes in Biological Control*; CRC Press: Boca Raton, 1990.
32. McSorley, R. Sampling and extraction techniques for nematodes. In *Techniques in Nematode Ecology*; Wheeler, T., Forge, T., Caswell-Chen, E., LaMondia, J.A., Eds.; Society of Nematologists, 2000. Available at <http://ianrwww.unl.edu/son/McSorley.htm> (accessed August 2000).
33. Wheeler, T.; Forge, T.; Caswell-Chen, E.; LaMondia, J.A. *Techniques in Nematode Ecology*; Society of Nematologists, 2000. Available at http://ianrwww.unl.edu/son/Ecology_Manual_TOC.htm (accessed August 2000).
34. *Interactive Diagnostic Key to Plant Parasitic, Freelifing and Predacious Nematodes*. Available at <http://ianrwww.unl.edu/ianr/plntpath/nematode/key/nemakey.htm> (accessed August 2000).

Nematodes and Soil

Wilfrida Decraemer

Department of Invertebrates, Royal Belgian Institute of Natural Sciences,
Brussels, Belgium

Abstract

Soil nematodes and their communities are suitable as bio-indicators to assess soil health and environmental changes. They are abundant, functionally diverse, and present at all levels of the food chain; show different reproductive and survival capacities; and respond in different degrees to changes in the ecosystem such as resource enrichment or disturbance. They recycle nutrients by feeding on microorganisms and plants and facilitate uptake of minerals by plants. The succession status of the nematode soil community reflects a history of disturbance or restoration. Several indices and ratios have been introduced, e.g., the structure index that describes whether a soil ecosystem is matured (high index) or disturbed (low index) and the fungivore/bacterivore ratio as indicator of the dominant decomposition pathway. Understanding ecosystem functioning and stability is indispensable for a sustainable agriculture and environmental monitoring. Nematodes form a suitable tool to analyze the effects of altered conditions due to climate change, or other forms of disturbance. More cost-effective tools are developed taking advantage of next-generation sequencing of faunal assemblages and search for key genera/species with a known sensitivity or response to specific types of disturbance.

INTRODUCTION

Nematodes or round worms are a very abundant and diverse group of small, unsegmented, and mostly transparent thread-like worms, occurring in almost every habitat either as free-living or as parasites of plants and/or animals. They appear simple and conservative in their morphology but show an enormous plasticity to adapt to a wide range of conditions and habitats. For their active life and locomotion, they depend on moisture. However, many nematodes can survive harsh conditions in an anhydrobiotic state. Survival of soil nematodes is directly affected by soil moisture, relative humidity, and other environmental factors such as soil texture. Soil pore size influences the ability with which nematodes move through the soil interstices. Soil pH and temperature may also affect nematode taxa. Nearly, a quarter of the total number of nematode species described are free-living (about 12,000 species), the majority being from terrestrial habitats; many species remain to be discovered.^[1] Nematodes are typically amphimictic, having separate males and females. However, many species lack males and reproduce either by parthenogenesis (common) or by hermaphroditism (rare). Nematodes typically have an egg stage, four juvenile stages, and the adult stage. Most nematodes molt four times before becoming an adult. In some groups, one juvenile stage (usually J3) is more resistant to environmental stress and is therefore specialized for dispersal and for surviving harsh conditions; it usually represents the infective stage of most animal-parasitic forms. In free-living bacterial-feeding rhabditid nematodes,

this modified stage is called dauer (German for endurance) and may represent an alternative pathway to the normal development process. Many free-living species with dauers are dispersed by insects or other arthropods, a relationship known as phoresy. The nematode life cycle varies from a few days for many bacterivorous colonizers such as rhabditids to more than a year for longidorid plant ectoparasites.^[2] Females are usually oviparous, but in some groups, the eggs can hatch inside the female body (ovoviviparity), usually resulting in her death. Ovoviviparity can be induced in normally reproducing species by pollutants (e.g., sulfur dioxide).^[3]

Nematodes play a variety of roles. In agriculture, they are one of the major pests, e.g., in potato in Europe where the increase of world trade provides the associated risk of spread of the potato cyst nematode *Globodera* spp. as well as an increased threat to the use of resistant cultivars as a major tool in reducing damage caused by these species.^[4] Worldwide, but especially in the developing and transition countries, plant-parasitic nematodes (PPNs) including root-knot nematodes (*Meloidogyne* spp.) cause important yield losses (overall average of 12.3%) to cash and other crops (Fig. 1).^[5] It has been estimated that a single acre of soil from arable land may contain as many as 3,000,000,000 nematodes, and a single wheat gall formed by *Anguina tritici* contains approximately 11,000–18,000 nematodes, although as many as 90,000 have been recorded.^[6] Several nematode species are, on the contrary, very promising as natural antagonists against pest insects. Entomopathogenic nematodes (EPNs) of the families



Fig. 1 Galling of tomato roots due to root-knot nematodes in Ethiopia.

Source: Awol Seid Ebrahim.

Steinernematidae and Heterorhabditidae are associated with symbiotic bacteria (*Xenorhabdus* and *Photorhabdus*, respectively), which are able to kill the insect host usually within 24–48 hr.^[7] EPNs are harmless to the environment and men and are increasingly used as environmentally safe biocontrol alternatives for chemical control products. Free-living nematodes are used as bio-indicators of pollution in terrestrial and aquatic environments. Nematodes play a key role in soil and aquatic food webs because of their ubiquitous presence, overwhelming densities (typically 100 g of soil yields around 3000 nematodes) and diversity (sometimes compared to insects), their presence at different trophic levels (as plant feeders, secondary detritivores, predators, and omnivores), low mobility, their difference in sensitivity, and quick response toward diverse types of disturbances (Fig. 2).^[8,9] To facilitate interpretation in ecological studies, terrestrial nematodes were allocated into functional groups (guilds) based on feeding type and life history strategy.^[10] Nematodes are an essential component in food security and interact in a synergistic or antagonistic way with other organisms.^[11] Different types of soil properties, soil management (forestry, agriculture, natural habitat), vegetation, and environmental disturbance such as pollution have an impact on soil nematode community structure, making the latter an interesting tool to assess changes in soil conditions.

This entry aims to highlight 1) the role of soil nematodes in soil ecosystems and 2) their importance as soil health bio-indicators, a function that will increase in importance due to the impact of climate change, the decrease in the use of chemical pesticides, and major biodiversity and environmental concerns nationally and internationally, inducing the need for further development of soil ecosystem strategies to improve soil quality,

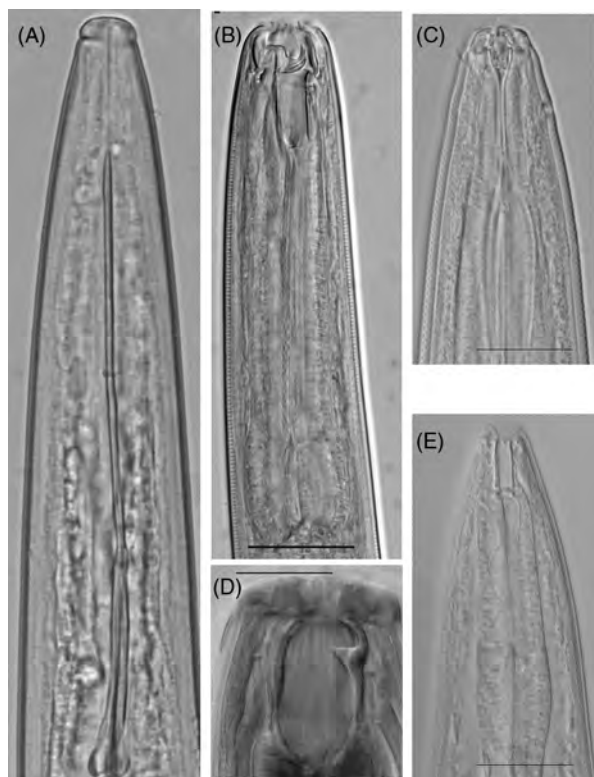


Fig. 2 Soil nematodes: a selection of different functional types: (A) a plant feeder (*Xiphinema*), (B) *Mononchoides composticola*, a mainly bacterivore also preying on other nematodes (omnivore), (C) *Plectus*, a bacterivore, and (D) a rhabditid bacterivore. Scale bar is 20 μ m.

Source: Courtesy of (B) H. Steels and (C–D) D. Slos.

including the reduction of PPN pests and improvement of plant health.

NEMATODES AND SOIL

Soil nematodes can be used to untangle biological mechanisms in soil and serve as environmental monitoring tools. Nematodes play a central role in the soil food web, but information on free-living soil nematodes is rather small compared with that of economically important PPNs. The active life of nematodes in soil is primarily determined by soil properties such as soil texture and soil structure, available soil moisture, pH, temperature, organic matter, available food, and impact of other soil biota. A range of nematode community indices has been proposed for environmental monitoring.

Soil Texture, Soil Structure, and Moisture Regime

Most ecological studies relate nematode behavior to soil texture. Nematodes occur in a wide range of soils, but their distribution, horizontally and vertically, is mostly limited by soil moisture regime and food availability. Soil

structure (= arrangement of soil particles or aggregation in crumb particles) has a major effect on pore space, movement of water, and consequently on locomotion of nematodes in the soil interstices of 25–100 μm diameter. Soil structure affects decomposition and mineralization process by its effect on the grazing intensity on bacteria. Loam and clay soils have a higher bacterial biomass, but bacteria are more correlated with pores of <2 μm diameter and thus not accessible for nematodes. Grazing on bacteria by bacterivorous nematodes in grassland appeared to be higher in sandy soils than in loams and clays, resulting in a higher percentage of soil organic nitrogen (N) per day in sandy soils.^[12] In natural soils, soil particles aggregate and provide a secondary structure. Distribution of nematodes is correlated with distribution of aggregate size. Nematode grazers are found inside these aggregates where bacterial density is greater than outside aggregates.^[13] A study on the distribution of the ecto-plant-parasite *Xiphinema diversicaudatum* in an undisturbed deciduous woodland over a period of 31 years showed that the historical infestation from a bordering hedge spread into the woodland was about 30 cm/yr. Vertical distribution of PPNs depends on the rhizosphere of the host plant; ectoparasites have been found up to 2 m depth.^[14] Nematodes occur in all types of soil. Coarse-textured soils favor hatching and subsequent invasion of roots by PPNs,^[15] but heavier soils on the contrary may have the advantage of reducing damage caused by PPNs because of lower population densities.^[2] Soil characteristics change with land use. Cultivation reduces abundance of some genera of free-living nematodes such as *Diptherophora*, *Prismatolaimus*, and *Tylenchorhynchus* consistently.^[16] Sensitivity of some taxa to mechanical disturbance is exploited, e.g., to control stubby-root nematodes by reducing population densities to beyond damage threshold by plowing.^[2] Management of soilborne pathogens often occurs by the application of synthetic chemical biocides, which results in an impoverished soil fauna. Soil fumigation, e.g., decreases the microbial population and strongly reduces nematodes, resulting in soils that are only inhabitable by primary colonizers.^[16] Also, change in soil moisture regime may have an impact on the incidence of damage by PPNs. Hockland (personal communication) observed that carrots grown for processing, which require less water, showed less damage by stubby-root nematodes. Some plant ectoparasites, e.g., *Xiphinema bergeri*, show a broad tolerance and occur in habitats that are subjected to periods of drought as well as waterlogged conditions as in rice fields.^[14]

Soil pH

In experiments where soil pH was manipulated, terrestrial nematode communities reflect the disturbances caused by induced changes of soil pH.^[17] In an arable

agroecosystem, increase of soil pH had a positive effect on the total number of bacterivorous nematodes but led to a reduction of hyphal feeders. A low soil pH in combination with higher copper levels increased the toxic effect of copper and reduced the total number of nematode species.^[18] The effects of soil pH on nematode behavior differ according to the species; e.g., *Trichodorus similis* (common European virus-vector nematode) prefers more acid soils, while *Trichodorus primitivus* occurs more in neutral to alkaline soils.^[19]

Temperature

Temperature affects nematode physiological processes (rates of assimilation and respiration) and behavior and determines individual development rate and population growth.^[20] Nematodes react to t° gradients resulting in oriented movement toward or away from the preferred t° ; e.g., newly hatched juveniles of root-knot nematodes migrate to a preferred t° range in close proximity to plant roots and attractant root diffusates.^[15] Predicted climate change in response to elevated carbon dioxide (CO_2) in the atmosphere may shorten generation time of nematodes, which may increase population density and dominance of species better adapted to warm habitats at the expense of species near the edge of their geographic range.^[21] Temperature rise in subarctic sites increases the abundance of bacterial- and fungal-feeding nematodes but reduces species richness.^[22]

Nutrient Cycling

Energy and nutrient flow in terrestrial ecosystems are primarily mediated by bacteria and fungi. Bacterivorous nematodes play an important role in regulating decomposition processes; they alter microbial activity, microbial community structure, and N mineralization.^[23] Decomposition rates are sensitive to physical disturbance. Bacteria and bacterivorous nematodes regulate decay rates in incorporated residues in conventional till soils, while fungi and fungivorous nematodes are more likely to regulate decomposition of surface residues in no-till soils.^[24] Nematodes affect N availability directly by the excretion of ammonium and indirectly by freeing N immobilized by microbes through metabolism, excretion, and dispersal; they contribute between 8% and 19% of N mineralization in conventional and integrated farming systems.^[25] Health of agricultural soils can be improved by organic matter amendments and forms a critical component of sustainable agricultural production.^[23] Most genera correlate positively with the addition of organic fertilizers, but a consistent effect across multiple seasons was only observed for a few genera of free-living nematodes (*Cruzema*, *Mesorhabditis*, *Mesodorylaimus*, and *Nygolaimus*).^[16] Continued detrimental loss of organic matter in some production systems as in

intensive cotton production may reduce the resilience to plant pathogens.^[26]

Impact of Soil Biota

Plant roots attract nematodes, which show consistent directions of movement toward CO₂.^[15] Monoculture of crops (maize) led to a drop in nematode diversity and an increase in the incidence of PPNs.^[27] Suppressive soils (=soils with antipathogenic potential) possess a biological buffering against pests such as nematodes, by microorganisms (fungi, *Verticillium chlamydosporium*, or bacteria, *Pasteuria* sp.), which are primarily responsible for maintaining nematode population densities below damage threshold.^[26] In contradiction to the hypothesis of soil food web models, carnivorous nematodes are less successful in controlling root feeding nematodes than soil microorganisms.^[28,29]

Nematode Assemblages and Response to Disturbance

Soil nematode communities vary among ecosystems not only because of differences in soil characteristics but also because of differences in plant diversity such as between agricultural and natural ecosystems and in the former between annual and perennial crops. Soil communities under perennial crops more closely resemble natural ecosystems.^[30] Plants show differences in rooting patterns, which favors some species. Mobile ecto- and semi-endoparasitic nematodes, e.g., are more abundant under grasses, while sedentary ecto- and endoparasites are more common under forbs where the soil appears to be more fungal dominated.^[31]

Understanding the relationship of biodiversity to ecosystem functioning and stability is indispensable for a sustainable agriculture, environmental monitoring, and the restoration of ecosystems based on nematodes. Nematode diversity can be assessed at species, genus, or trophic level. Diversity at trophic level reflects food web complexity and is in general greater in perennial and natural systems than in annual cropping systems. Change in community composition through time reflects seasonal dynamics and ecological succession.^[24] Within a food web (Fig. 3), three energy pathways can be recognized, namely: the plant root, bacterial, and fungal pathways (Fig. 3). Primary herbivores as plant-feeding nematodes decrease the primary productivity of the plant by altering water uptake and nutrients; nutrients will become incorporated in detritus that support the bacterial and fungal pathways, which include bacterivorous and fungivorous nematodes.^[32] All three pathways unite at higher levels in the food chain with omnivorous and predatory nematodes. Secondary and tertiary consumers as omnivores feed on species in any of the trophic groups. A healthy soil ecosystem is characterized by

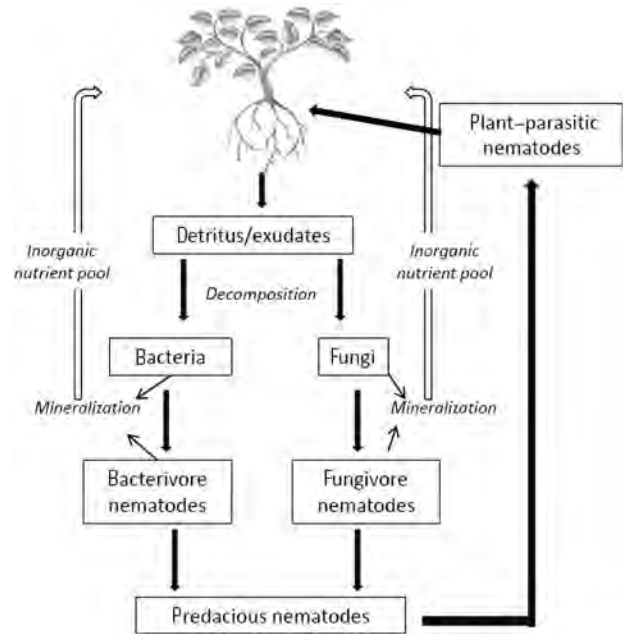


Fig. 3 Functional groups in a soil food web in relation to soil nutrient cycling.

Source: Modified after Ingham, Trofymow, et al.^[33]

high biodiversity and high community stability that can overcome short- or long-term disturbance and stress and is able to maintain the integrity of nutrient cycling and energy flow, suppress of pests and pathogens, and improve plant health.^[34] Insight into the functional role of nematodes in the ecosystem can be deduced from nematode feeding groups, established largely on stoma morphology. Depending on the author, five to eight main feeding types are distinguished: plant-feeding, hyphal- or fungal-feeding, bacterial-feeding, substrate ingesters, carnivores, unicellular eukaryote feeders, dispersal stages of animal parasites, and omnivores; many nematodes feed on more than one source and could be considered as omnivore.^[32] Insight into the effect of environmental disturbance (natural or anthropogenic) is revealed by changes in the nematofauna.^[26] A history of disturbance can be traced by looking at the successional status of a soil community. The early stage of succession favors the opportunistic species (colonizers) such as bacteria and their predators, small organisms with short generation time, rapid dispersal, and generalist feeding habits as in bacterivorous nematodes. This stage is generally transformed to a more diverse community with an increase in relative abundance of fungi and slower fungivorous nematodes. Secondary and tertiary consumers as predators establish later in a more stable condition. If disturbance is severe, few communities will persist. Chemical and physical disturbance are the main types of disturbance in agriculture.

To analyze the complex structure of nematode assemblages and ecological succession or disturbance, Bongers^[35]

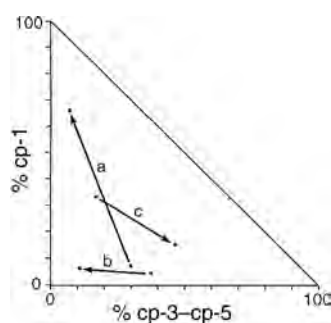


Fig. 4 Cp triangle showing (a) shifts during eutrophication, initial situation, and 2 weeks after adding powdered cow dung; (b) artificial acidification of coniferous forest soil; and (c) recovery, 33 and 44 weeks after organic manuring.

Source: Russian Journal of Nematology, 1995.

&classified nematode genera according to their feeding type and assigned them to a colonizer (r)–persister (K) life strategy scaling or a 1–5 cp scale; cp-1 are enrichment opportunists with a short life cycle and high fecundity; cp-2 are general opportunists with a moderate life cycle and lower fecundity and are stress tolerant and considered to be part of a basal fauna present in all soils; and cp-3–cp-5 are structure indicators with a long life cycle, large body, and low fecundity and are stress intolerant (Fig. 4). Bongers developed the nematode maturity index (MI) for free-living nematodes taking feeding type and life cycle into account. Later, several adaptations of MI were made, including a separate index for PPNs or a modified MI including all soil nematodes.^[36] The MI is the weighted mean of the frequency of the taxon with corresponding cp value:

$$MI = \sum \frac{v_i \times f_i}{n}$$

where v_i = cp value assigned to taxon (family), f_i = frequency of family i in sample, and n = total number of individuals in a sample. Large values for MI point to a late successional stage with persisters (e.g., Dorylaims), while a low MI value is typical for unstable habitats with many successful colonizers such as rhabditids. To evaluate a pollution-induced stress factor, Bongers and Bongers^[37] proposed MI_{2-5} same as MI but without cp-1 group.

Other measures of ecological succession include the fungivore/bacterivore ratio (f/b ratio) as indicator of the dominant decomposition pathway;^[38] the structure index (SI) that describes whether a soil ecosystem is matured (high SI) or disturbed (low SI); the enrichment index (EI) that is based on the expected responsiveness of the opportunistic guilds (bacterivores with cp-1 or Ba1) to the increase in food, reflecting the activity of primary detritus-feeding nematodes (high EI = nutrient enriched; low EI = depleted); and the channel index that gives the percentage of fungivores among the total fungivores and

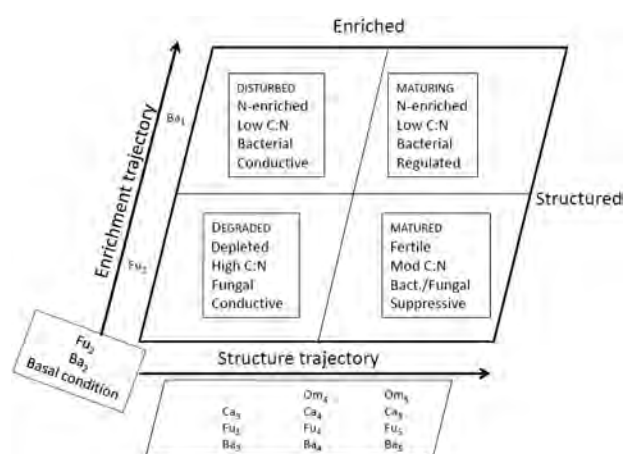


Fig. 5 A simplified food web structure showing the effect on EI and SI. Ba, bacterivore; Ca, carnivore; Fu, fungivore; Om, omnivore.

Source: Modified from Ferris, Bongers, et al.^[38] and Ferris & Bongers.^[39]

opportunistic bacterivores and describes the dominant decomposition channels in the food web.^[38] The calculation of SI and EI is based on guilds, which combines feeding type and cp value to cluster nematode taxa; they can be used to construct a faunal profile, with change in resources reflecting the succession of nematode communities from basal to enriched or to structured.^[39] Readily decomposed components are utilized fast by bacteria; more recalcitrant components are resources for fungi (Fig. 5). To obtain a well-functioning system, agricultural ecosystems may require input of organic material.

CONCLUSION

Nematodes are suitable as bio-indicators for overall soil condition because they occupy different trophic levels within soil food webs, reflect the succession stage of the system they inhabit, and show different sensitivities to toxins and stressors, such as chemicals and metals.^[40] More wide exploitation of nematode potentials as indicators for soil health is associated with technical innovations such as the use of advanced molecular tools. Time-consuming identification problems and small-scale fauna investigations will be overcome by application of next-generation sequencing on faunal assemblages or real-time polymerase chain reaction as diagnostic tool for quantification of nematodes. However, the interpretation of sequence data is not always straightforward, and the discrimination of related species for precise insight about abundance remains difficult.^[41,42] Nematode indices could become more cost-effective if they would be reduced to genera with known sensitivity or response to specific types of disturbance.^[16] Concerns on the impact of climate change are analyzed by modeling changes in

temperature, moisture, elevated CO₂, and ultraviolet B radiation. The effects of altered conditions were observed and either appeared taxon dependent^[43] for temperature and moisture or showed changes in functional groups to the latter factors.^[44]

REFERENCES

1. Coomans, A. Present status and future of nematode systematics. *Nematology* **2002**, *4* (5), 573–582.
2. Decraemer, W.; Geraert, E. Ectoparasites. In *Plant Nematology*; Perry, R., Moens, M., Eds.; CABI: Wallingford, 2006; 153–184.
3. Walker, J.T.; Tsui, R.K. Induction of ovoviviparity in *Rhabditis* by sulfur dioxide. *Nematologica* **1968**, *14*, 148–149.
4. Hockland, S.; Niere, B.; Grenier, E.; Blok, V.; Phillips, M.; den Nijs, L.; Anthoine, G.; Pickup, J.; Viaene, N. An evaluation of the implications or virulence in non-European populations of *Globodera pallida* and *G. rostochiensis* for potato cultivation in Europe. *Nematology* **2012**, *4* (1), 1–13.
5. Sasser, J.N.; Freckman, D.W. A world perspective on nematology: The role of the society. In *Vistas on Nematology: A Commemoration of the Twenty-Fifth Anniversary of the Society of Nematologists*; Veech, J.A., Dickson, D.W., Eds.; Society of Nematologists: Hyattsville, 1987; 7–14.
6. Decraemer, W.; Hunt, D. Structure and classification. In *Plant Nematology*; Perry, R., Moens, M., Eds.; 2nd Ed.; CABI: Wallingford, 2013; 1–51.
7. Boemare, N.E.; Boyer-Giglio, M.H.; Thaler, J.O.; Akhurst, R.J. The phages and bacteriocins of *Xenorhabdus* spp., symbiont of the nematodes *Steinernema* spp. and *Heterorhabditis* spp. In *Nematodes and the Biological Control of Insect Pests*; Bedding, R., Akhurst, R., Kaya, H., Eds.; CSIRO Publications: East Melbourne, 1993; 137–145.
8. Bongers, T.; Bongers, M. Functional diversity of nematodes. *Appl. Soil Ecol.* **1998**, *10*, 239–251.
9. Schouten, T.; Breure, A.M.; Mulder, C.H.; Rutgers, M. Nematode diversity in Dutch soils, from Rio to a biological indicator for soil quality. *Nematol. Monogr. Perspect* **2004**, *2*, 469–482.
10. Bongers, T.; Ferris, H. Nematode community structure as a bioindicator in environmental monitoring. *Trends Ecol. Evol.* **1999**, *14*, 224–228.
11. Sánchez-Moreno, S.; Nicola, N.; Ferris, H.; Zalom, F. Effects of agricultural management on nematode-mite assemblages: Soil food web indices as predictors of mite community composition. *Appl. Soil Ecol.* **2009**, *41*, 107–117.
12. Hassink, J.; Bouwman, L.A.; Zwart, K.B.; Bloem, J.; Brussaard, L. Relationships between soil texture, physical protection of organic matter, soil biota, and C and N mineralization in grassland soils. *Geoderma* **1993**, *57*, 105–128.
13. Savin, M.C.; Görres, J.H.; Neher, D.A.; Amador, J.A. Uncoupling of carbon and nitrogen mineralization: Role of microbivorous nematodes. *Soil Biol. Biochem.* **2001**, *33*, 1463–1487.
14. Taylor, C.E.; Brown, D.J.F. *Nematode Vectors of Plant Viruses*; CABI: Wallingford, 1997.
15. Perry, R.N. Hatching. In *Biology of Nematodes*; Lee, D.L., Ed.; Taylor & Francis: London, 2002; 147–170.
16. Zhao, J.; Neher, D.A. Soil nematode genera that predict specific types of disturbance. *Appl. Soil Ecol.* **2013**, *64*, 135–141.
17. de Goede, R.; Dekker, H.H. Effects of liming and fertilization on nematode communities in coniferous forest soils. *Pedobiologia* **1993**, *37*, 193–209.
18. Korthals, G.W.; Alexiev, A.D.; Lexmond, T.M.; Kamenga, J.E.; Bongers, T. Long-term effects of copper and pH on the nematode community in an agroecosystem. *Environ. Toxicol. Chem.* **1996**, *15*, 979–985.
19. De Waele, D.; Coomans, A. Occurrence and ecology of trichodorid nematodes in Belgium. *Rev. Nématol.* **1991**, *14*, 127–132.
20. Ferris, H.; Venette, R.C.; Lau, S.S. Dynamics of nematode communities in tomatoes grown in conventional and organic farming systems, and their impact on soil fertility. *Appl. Soil Ecol.* **1996**, *3*, 161–175.
21. McSorley, R.; Porazinska, D.L. Elements of sustainable agriculture experiment. *Nematropica* **2001**, *31*, 1–9.
22. Solenius, B.; Bostrom, S. Effects of global warming on nematode diversity in a Swedish tundra soil—A soil transplantation. *Nematology* **1999**, *1*, 695–709.
23. Djigal, D.; Brauman, A.; Diplo, T.A.; Chotte, J.L.; Villenave, C. Influence of bacterial-feeding nematodes (*Cephalobidae*) on soil microbial communities during maize growth. *Soil Biol. Biochem.* **2004**, *36*, 323–331.
24. Neher, D.A. Ecology of plant and free-living nematodes in natural and agricultural soils. *Annu. Rev. Phytopathol.* **2013**, *48*, 371–394.
25. Beare, M.H. Fungal and bacterial pathways of organic matter decomposition and nitrogen mineralization in arable soil. In *Soil Ecology in Sustainable Agricultural Systems*; Brussaard, L., Ferrera-Cerrato, R., Eds.; Lewis Publication: Boca Raton, 1997; 37–70.
26. Westphal, A. Detection and description of soils with specific nematode suppressiveness. *J. Nematol.* **2005**, *37*, 121–130.
27. Santiago, D.; de Oliveira Arieira, G.; de Almeida, E.; de Fatima Guimaraes, M. Responses of soil nematode communities to agroecological crop management systems. *Nematology* **2012**, *14*, 209–221.
28. Yeates, G.W.; Wardle, D.A. Nematodes as predators and prey: Relationships to biological control and soil processes. *Pedobiologia* **1996**, *40*, 43–50.
29. Piskiwics, A.M.; Duyts, H.; Berg, M.P.; Costa, S.R.; van der Putten, W. Soil microorganisms control plant ectoparasitic nematodes in natural coastal foredunes. *Oecologia* **2007**, *152*, 505–514.
30. Wasilewska, L. The structure and function of soil nematode communities in natural ecosystems and agroecosystems. *Pol. Ecol. Stud.* **1979**, *5*, 97–145.
31. Wardle, D.A.; Yeates, G.W.; Williamson, W.; Bonner, K.I. The response of a three trophic level soil food web to the identity and diversity of plant species and functional groups. *Oikos* **2003**, *102*, 45–56.
32. Yeates, G.W.; Bongers, T.; de Goede, R.G.M.; Freckman, D.W.; Georgieva, S.S. Feeding habits in soil nematode

- families and genera—An outline for soil ecologists. *J. Nematol.* **1993**, *25*, 315–331.
33. Ingham, R.E.; Trofymow, J.A.; Ingham, E.R.; Coleman, D.C. Interactions of bacteria, fungi, and their nematode grazers: Effects on nutrient cycling and plant growth. *Ecol. Monogr.* **1985**, *55*, 119–140.
 34. Wang, K.-H.; MacSorley, R. *Effects of Soil Ecosystem Management on Nematode Pests, Nutrient Cycling, and Plant Health*, 2005. 10.1094/APSnetFeatures/2005-0105. Available at <http://www.apsnet.org>.
 35. Bongers, T. The maturity index, an ecological measure of environmental disturbance based on nematode species composition. *Oecologia* **1990**, *83*, 14–19.
 36. Yeates, G.W. Modification and qualification of the nematode maturity index. *Pedobiologia* **1994**, *38*, 97–101.
 37. Bongers, T.; Bongers, M. Functional diversity of nematodes. *Appl. Soil Biol.* **1998**, *10*, 239–251.
 38. Ferris, H.; Bongers, T.; de Gouede, R.G.M. A framework for soil food web diagnostics of the nematode faunal analysis concept. *Appl. Soil Ecol.* **2001**, *18*, 13–29.
 39. Ferris, H.; Bongers, T. Nematode indicators of organic enrichment. *J. Nematol.* **2006**, *38* (1), 3–12.
 40. Ferris, H.; Griffiths, B.; Porazinska, D.; Powers, T.; Wang, K.; Tenuta, M. Reflections on plant and soil nematode ecology: Past, present and future. *J. Nematol.* **2012**, *44*, 115–126.
 41. Parazinska, D.L.; Sung, W.; Gibblins-Davis, R.M.; Thomas, W.K. Reproducibility of read numbers in high-throughput sequencing analysis of nematode community composition and structure. *Mol. Ecol. Resour.* **2010**, *10*, 666–676.
 42. Min, Y.Y.; Toyota, K.; Sato, E. A novel nematode diagnostic method using the direct quantification of major plant-parasitic nematodes in soil by real-time PCR. *Nematology* **2012**, *14*, 265–276.
 43. Bakonyi, G.; Nagy, P.; Kovács-Lang, E.; Kovács, E.; Barabas, S.; Répási, V.; Seres, A. Soil nematode community structure as affected by temperature and moisture in a temperate semiarid shrubland. *Appl. Soil Ecol.* **2007**, *37*, 31–40.
 44. Kuijper, L.D.; Berg, M.P.; Morriën, E.; Kooi, B.; Verhoef, H.A. Global change effects on a mechanistic decomposer food web model. *Glob. Change Biol.* **2005**, *11*, 249–265.

Nexus Approach: Resource Management for Soil Productivity

Kai Schwärzel

Reza Ardakanian

Tamara Avellán

Lulu Zhang

*Institute for Integrated Management of Material Fluxes and of Resource,
United Nations University, Dresden, Germany*

Abstract

Despite of all efforts, soil and land degradation is increasing across sub-Saharan Africa (SSA). One reason for this is that many homes in SSA have no access to modern energy supply. Most of the energy used for cooking and heating comes from firewood, charcoal, crop residues, or even dung. Since biomass collection for energy use is not counterbalanced by soil nutrient increasing measures these habits results in significant vegetation and soil degradation, and decreasing water supply in wells and rivers. As a consequence, the provision of water, food, and energy is not ensured in many regions of SSA. The entry shows that food, energy and water security are interlinked as a Gordian knot. At the heart of these interlinkages is the Nexus of water, soil, and waste. Solutions to food and water issues in SSA must involve the establishment of land use systems which produce more food and bioenergy based on a sustainably enhanced productivity of soils and land. It is discussed that multifunctional land use systems based on agroforestry concepts meet these needs. As an innovative strategy to improve soils and land in SSA, the integration of non-food bioenergy systems across the landscape on degraded soils and on soils with medium to low productivity is suggested. It is demonstrated how the productivity of these sites could be increased by integrating waste from households and farms into the biomass production cycle. Finally, the potential impacts of implemented multifunctional land use systems on environmental, social, and socioeconomic aspects are illustrated.

INTRODUCTION

Despite all efforts, the provision of water, food, and energy is not secured in many regions across sub-Saharan Africa (SSA). For instance, one in four people suffered from malnourishment in 2013.^[1] Besides increasing variability and change in climate and the growing demand for resources, such as water, soil, and biomass, poor management of these resources is also a major part of this problem. It results, among other things, in a detrimental decline of soil and land resources.

Many households in SSA have no access to modern energy supply. Instead, traditional biomass fuel—that is, mostly firewood (unprocessed woody biomass) and charcoal—supplies over 70% of the household energy in SSA and 93% of households rely on wood fuel (as firewood, charcoal, pellets, biogas, and cellulosic ethanol) for their daily cooking.^[2] Since the extraction of biomass for energy use is not compensated by soil fertility enhancing and nutrient recycling measures, in the midterm and long term, these practices result in significant vegetation and soil degradation, nutrient depletion of soils, and decreasing water supplies in wells and rivers. As a consequence, this in turn leads to a decrease in biomass production across SSA, thus further decreasing energy supply. Due to this

close interconnectedness between food and energy issues, sub-Saharan countries will not be able to solve their food problem without solving their energy problem, and without a satisfactory solution to both, their economic growth will be severely constrained.^[3] In the following, we will elaborate coping strategies that illustrate the interlinkages between food and energy security by advancing a nexus approach toward the integrated management of soil, water, and waste.

OVEREXPLOITATION OF BIOMASS FROM FOREST AND CROPLAND

In SSA, firewood is the main energy source in rural areas whereas charcoal is the major source of household energy for the majority of the urban dwellers. Traditional firewood collection is seen as sustainable because it involves collecting dead branches and small dead trees, whereas the collection rates are usually below regeneration rate. However, dead wood collection removes nutrient from soils, and it is unknown the extent to which natural forests are impacted, and whether or not impact translates into direct effects on vegetation growth and/or species composition.^[4] In contrast to traditional firewood collection,

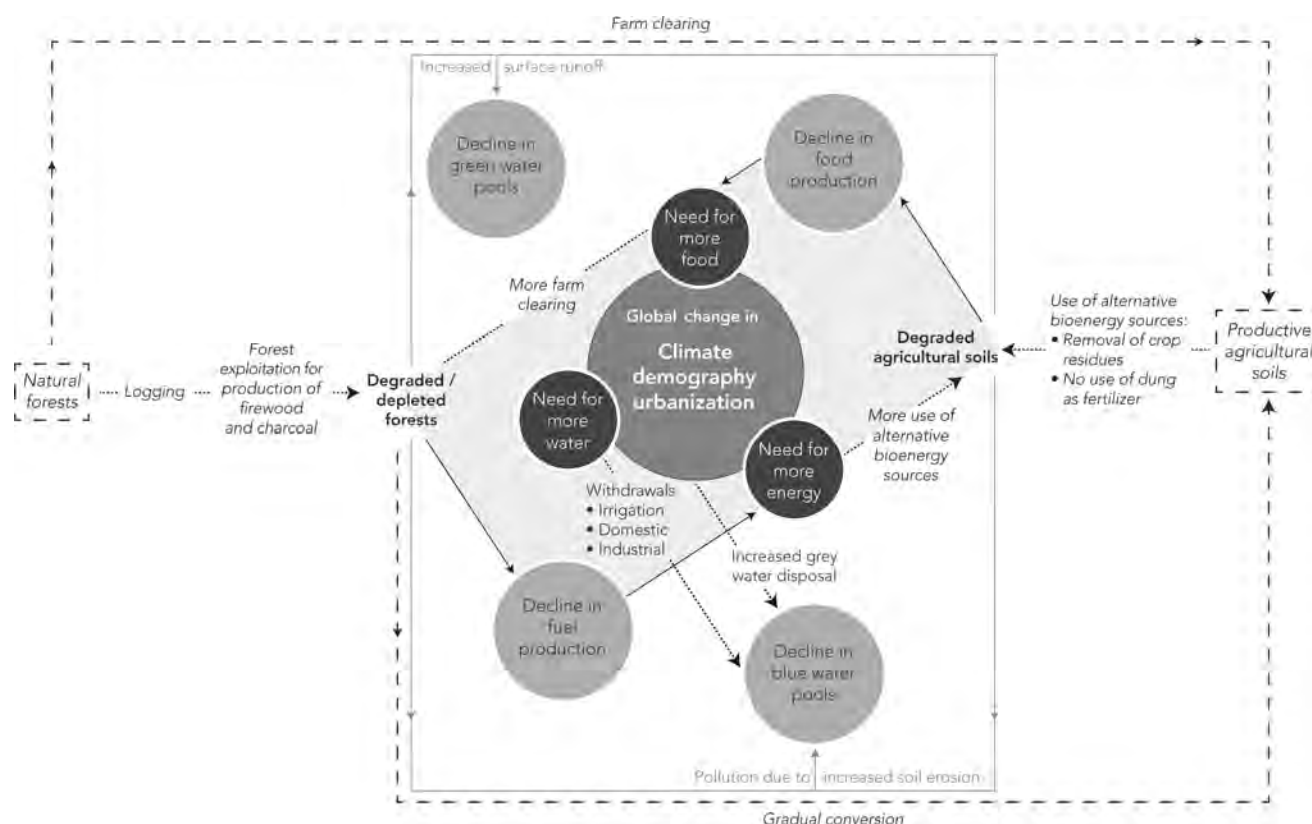


Fig. 1 Vicious cycle of overexploitation of biomass from forested and cropped land, concurrent degradation, downward spiral of food and fuelwood production, and decline in blue and green water pools around cities in SSA (developed based on the studies by Sitoe and Wertz-Kanounnikoff^[6] and Karlberg et al.^[11]). Blue water: fresh water in lakes, rivers, and aquifers; green water: plant available soil water; gray water: wastewater generated in households and offices.

charcoal production is based on logging and requires land clearing. Studies from Ethiopia,^[5] Mozambique,^[6] and additional countries in SSA show localized deforestation around large cities.^[7] But charcoal production is not the primary driver of deforestation around cities. It is assumed that peri-urban zones in which charcoal production for urban dwellers can be expected to be concentrated in the beginning of urban growth are likely to be areas that are also under pressure from clearance for agriculture. Meaning, the areas from which wood for charcoal production are often sourced are also the areas that are cleared anyway.

Research has shown that there are depletion hotspots in SSA where wood fuel harvest is unsustainable.^[8] The largest hotspot, rather a huge contiguous degradation area, occurs in East Africa, extending from Eritrea through western Ethiopia, Kenya, Uganda, Rwanda, and Burundi and affecting 26% of the region's population. Smaller hotspots can be found in western and southern Africa (e.g., in Lesotho or in southern Mozambique). In these hotspots, biomass scarcity is prevalent and many farmers are using crop residues or cattle dung instead of wood fuel as energy source for cooking and heating.^[9,10]

Fig. 1 illustrates how that periodic removal of organic matter and nutrients and the misuse of dung for cooking and heating instead of fertilization gradually lower soil fertility

and thus soil productivity and ultimately food security. These habits result in a self-propagating, vicious cycle of overexploitation of biomass from forested and cropped land, concurrent degradation, and downward spiral of food production.^[11] This leads to high risks with regard to achieving and sustaining food security by constituting major biophysical limitations for agricultural production in SSA.^[12] It is projected that SSA's population will double to 2.4 billion by 2050, as a result, the demand for food and fuelwood can also be expected to increase significantly.^[13] Moreover, urbanization will further exacerbate the pressure on rural and peri-urban zones around the cities; in 2014, SSA's urbanization level was 37%, and by 2050, 54% of SSA's population is projected to live in urban areas. In the absence of affordable alternatives, bioenergy use—in particular charcoal—is expected to increase significantly over the next decades.^[14] It is estimated that for each 1% in urbanization, there will be a 14% increase in charcoal use.^[15]

The problem of securing food and energy for an ever growing urban population is inextricably linked to forest and cropland degradation in the hinterland of SSA's cities. But this degradation is also associated with change in soil properties relevant for the partition of rainfall into surface and subsurface water. How much water infiltrates,

evaporates, or leaks depends not only on vegetation and climate but also on the geometry of the soil pore space and therefore on soil structure. The latter is strongly related to permeability and water storage capacity of soils; it is well known that changes in land cover and available nutrient pools can induce the loss of soil structure and hence modify soil properties relevant for transmission and storage of water. As a consequence, the decline in forests and cropland favors increased surface runoff transporting excess soil into surface water and inducing silting and eutrophication of these (i.e., a decline in blue water pools; Fig. 1).

The lack of vegetated soil surfaces also leads to excess water erosion, which in turn reduces the infiltration rate of rainwater into the soil and lowers soil water pools available for plant growth (i.e., decline in green water pools). The decline in blue and green water pools due to the combination of contaminated surface waters and reduced infiltration of rainfall into the soil further promotes increased withdrawals for agricultural, domestic, and industrial use. Withdrawals often end up exceeding recharge rates, thus resulting in a decline in available blue-water pools. In the long run, these practices are not sustainable.

INTEGRATING ORGANIC WASTES INTO THE BIOMASS PRODUCTION CYCLE

It has been proposed to substitute traditional wood fuel with renewable energy from wind and sunlight,^[11] but the quick implementation of such technology is challenging.^[16] Another alternative could be biofuel made from short rotation forests, but its production would compete with food production for limited soil and water resources.^[17,18] Growing biofuel crops on marginal soils has been suggested as a way out of this dilemma. But examples across SSA have shown that water and nutrient shortage limit the yield of biofuel crops on such soils.^[19,20]

A sustainable solution to produce economically viable biofuel yields from marginal soils would be to integrate nutrients and water from organic waste (sewage sludge, compost, wastewater, etc.) into the biomass production cycle. There are promising examples from all over the world that illustrate how the application of organic waste can increase growth and yield, for instance in forest plantations.^[21] Municipalities in Sweden and Estonia are using pretreated wastewater, sewage sludge, and methanogenic landfill leachate in short rotation energy forests.^[22,23] Application of organic waste in energy forests is not just a cost-effective method to purify municipal wastewater and to produce bioenergy but also generates additional environmental benefits, such as soil carbon accumulation.^[23] This may directly translate into higher soil productivity and improved soil structure.

Much like the agricultural sector, SSA's wastewater management sector is facing challenges due to changes in

demography. The problem arises of how to deal with the ever increasing volume of wastewater in light of ineffective wastewater management, which covers only a small portion of the total wastewater volume generated. The largest part of SSA's wastewater flows into bodies of water and the environment more or less untreated. As using wastewater as a resource in food and bioenergy production may overcome some of the challenges farmers face, wastewater could be used where available. The safe use of wastewater for biomass production would also decrease the pollution of bodies of water used for irrigation. A typical example of the water–soil–waste nexus, this informal irrigation sector along streams and rivers is usually unregulated and unmonitored.

MANAGEMENT OF THE INTERLINKAGES BETWEEN FOOD, WATER, AND ENERGY SECURITY

SSA's food problem cannot be solved without solving its energy problem. Thus, the real challenge is how to manage the interlinkages. At the heart of this challenge is the nexus of water, soil, and waste (Fig. 2).

Use and alterations of one of these resources have a direct impact on the other resources. An interconnected management of soil, water, and waste is therefore an approach to ensure a sustainable intensification of biomass production. Such a strategy includes soil as well as blue and green water conservation; use of treated gray water as a source for irrigation and fertilizing; use of crop and animal waste for enhancing soil quality, preventing soil erosion, and minimizing unproductive evaporation by mulching and crop residue management; and the reuse and recycling of by-products.

Moreover, solutions to food and energy issues in SSA must involve the establishment of land use systems that produce more food and bioenergy based on a sustainable increased productivity of soils and land. Multifunctional land use systems developed according to agroforestry concepts, meet these needs. Agroforestry can play an important role in addressing the crisis of food security and environmental resilience, for instance by integrating fertilizer trees into food crop agriculture or by soil conservation hedgerows. In this context, multifunctionality can be either addressed through a single land use type, such as multifunctional agriculture, or by integrating several land uses on one piece of land (e.g., agroforestry). Similarly, from a broader perspective, landscapes that consist of a set of different spatial units (land use types) can also have spatial multifunctionality, depending on the functions of each piece of land. Such multifunctional landscapes will not only provide multiple goods and services to meet local demands but also satisfy the social needs of other areas (e.g., downstream), while the environmental resources are maintained and improved.

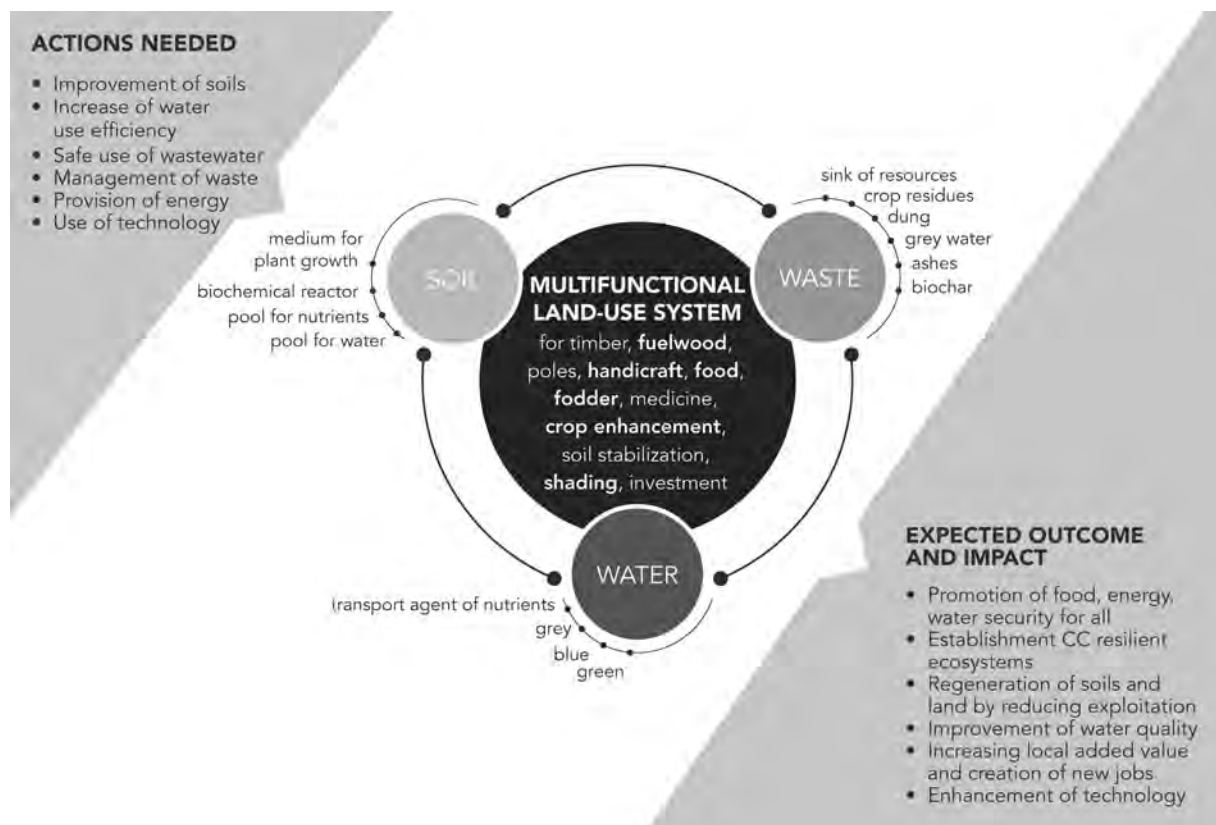


Fig. 2 Conceptual framework for the management of the interlinkages between food, water, and energy security.

An example for multifunctionality is to integrate bioenergy systems across the landscape on degraded soils and on soils with medium to low productivity. The productivity of these sites would then be increased by integrating organic waste (notably wastewater) from households and farms into the biomass production cycle. To this end, the concept of wastewater sanitation and reuse must be combined with multifunctional land use systems for the production of food and energy crops. In this approach, wastewater is treated in a target-oriented manner to be used for irrigation and nutrient supply of the respective land uses, considering hygienic as well as environmental aspects. Treating wastewater in this way can be cost-effective and reduces the demand for freshwater and fertilizers. Moreover, with the strategy to integrate bioenergy systems across landscapes direct and indirect effects on soils and land can be expected in the long run—direct enhancement of soil fertility at sites where bioenergy crops and improved farming systems will be established and indirect enhancement of soils and land at the landscape level because of regeneration of soils and land by reducing exploitation (firewood collection and tree logging will be significantly reduced). Other benefits of the integration of bioenergy systems across the landscape are the establishment of healthier ecosystems with better resilience to climate change and improved provision of water-related ecosystem services. Socioeconomic benefits include locally added

value, the creation of new jobs. Finally, the establishment of bioenergy crop production as part of multifunctional land use systems promotes the use of energy saving and advanced technology for electricity generation at the household and community level with social and economic benefits for households and local enterprise development.

CONCLUSION

The previous discussions revealed a major dilemma in biomass production and use in SSA: more and more food, feed, fiber, and wood for more and more people are needed, but less and less adequate soil and water resources are available. Therefore, a different approach to thinking about increased food and bioenergy production is necessary. This can be achieved, on one hand, by integrating organic waste from households and farms into the biomass production cycle and, on the other hand, by integrating bioenergy production in multifunctional land use systems. In these land use systems, the bioenergy production on degraded soils or of soils with low to medium productivity is combined with soil fertility enhancing measures, such as wastewater application. However, a successful implementation of such a concept requires the estimation of its economic feasibility and an assessment of its social acceptability. Finally, it needs to be investigated what

policy tools are necessary to reconcile competing resource uses (water, soil/land, and waste) and users. Then, the full consideration of social and environmental aspects can be considered when the concept of organic waste application is combined with multifunctional land use for the production of food and bioenergy crops or multi-purpose crops.

REFERENCES

1. FAO, IFAD and WFP. *The State of Food Insecurity in the World 2014. Strengthening the Enabling Environment for Food Security and Nutrition*; FAO: Rome, 2014.
2. Cerutti, P.O.; Sola, P.; Chenevoy, A.; Iiyama, M.; Yila, J.; Zhou, W.; Djoudi, H.; Eba'a Atyi, R.; Gautier, J.G.; Gumbo, D.; Kuehl, Y.; Levang, P.; Martius, C.; Matthews, R.; Nasi, R.; Neufeldt, H.; Njenga, M.; Petrokofsky, G.; Saunders, M.; Shepherd, G.; Sonwa, D.J.; Sundberg, C.; Noordwijk, M. The socioeconomic and environmental impacts of wood energy value chains in sub-Saharan Africa: A systematic map protocol. *Environ. Evid.* **2015**, *4*, 12.
3. Sachs, I.; Silk, D. *Food and Energy Nexus: Strategies for Sustainable Development*; United Nations University: Tokyo, 1990.
4. Morton, J. Fuelwood consumption and woody biomass accumulation in Mali, West Africa. *Ethnobot. Res. Appl.* **2007**, *5*, 37–44.
5. Alem, S.; Duraisamy, J.; Legesse, E.; Seboka, Y.; Mitiku, E. Wood charcoal supply to Addis Ababa city and its effect on the environment. *Energy Environ.* **2010**, *21*, 601–609.
6. Sitoe, A.S.A.; Wertz-Kanounnikoff, S. *The Context of REDD+ in Mozambique: Drivers, Agents and Institutions*; Center for International Forestry Research (CIFOR): Bogor, 2012.
7. Arnold, J.E.M.; Köhlin, G.; Persson, R. Woodfuels, livelihoods, and policy interventions: Changing perspectives. *World Dev.* **2006**, *34*, 596–611.
8. Bailis, R.; Drigo, R.; Ghilardi, A.; Masera, O. The carbon footprint of traditional woodfuels. *Nat. Clim. Change.* **2015**, *5*, 266–272.
9. Mahiri, I.O. Rural household responses to fuelwood scarcity in Nyando District, Kenya. *Land Degrad. Dev.* **2003**, *14*, 163–171.
10. Duguma, L.A.; Minang, P.A.; Freeman, O.E.; Hager, H. System wide impacts of fuel usage patterns in the Ethiopian highlands: Potentials for breaking the negative reinforcing feedback cycles. *Energy Sustain. Dev.* **2014**, *20*, 77–85.
11. Karlberg, L.; Hoff, H.; Flores-López, F.; Goetz, A.; Matuschke, I. Tackling biomass scarcity—from vicious to virtuous cycles in sub-Saharan Africa. *Curr. Opin. Environ. Sustain.* **2015**, *15*, 1–8.
12. Reynolds, T.W.; Waddington, S.R.; Anderson, C.L.; Chew, A.; True, Z.; Cullen, A. Environmental impacts and constraints associated with the production of major food crops in sub-Saharan Africa and South Asia. *Food Secur.* **2015**, *7*, 795–822.
13. United Nations, Department of Economic and Social Affairs, Population Division. 2015. *World Urbanization Prospects: The 2014 Revision*, (ST/ESA/SER.A/366).
14. Liyama, M.; Neufeldt, H.; Dobie, P.; Njenga, M.; Ndegwa, G.; Jamnadass, R. The potential of agroforestry in the provision of sustainable woodfuel in sub-Saharan Africa. *Curr. Opin. Environ. Sustain.* **2014**, *6*, 138–147.
15. Hosier, R.H. Charcoal production and environmental degradation. Environmental history, selective harvesting, and post-harvest management. *Energy Policy* **1993**, *21*, 491–509.
16. Mohammed, Y.S.; Bashir, N.; Mustafa, M.W. Overuse of wood-based bioenergy in selected sub-Saharan Africa countries: Review of unconstructive challenges and suggestions. *J. Clean. Prod.* **2015**, *96*, 501–519.
17. Gerbens-Leenes, P.W.; Hoekstra, A.Y.; van der Meer, T. The water footprint of energy from biomass: A quantitative assessment and consequences of an increasing share of bio-energy in energy supply. *Ecol. Econ.* **2009**, *68*, 1052–1060.
18. Giovannetti, G.; Ticci, E. Determinants of biofuel-oriented land acquisitions in sub-Saharan Africa. *Renew. Sust. Energ. Rev.* **2016**, *54*, 678–687.
19. Nhantumbo, I.; Salomaõ, A. *Biofuels, Land Access and Rural Livelihoods in Mozambique*; International Institute for Environment and Development: London, 2010.
20. von Maltitz, G.; Gasparatos, A.; Fabricius, C. The rise, fall and potential resilience benefits of *Jatropha* in Southern Africa. *Sustainability* **2014**, *6*, 3615–3643.
21. Marron, N. Agronomic and environmental effects of land application of residues in short-rotation tree plantations: A literature review. *Biomass Bioenerg.* **2015**, *81*, 378–400.
22. Hasselgren, K. Use of municipal waste products in energy forestry: Highlights from 15 years of experience. *Biomass Bioenerg.* **1998**, *15*, 71–74.
23. Holm, B.; Heinsoo, K. Municipal wastewater application to short rotation coppice of willows—Treatment efficiency and clone response in Estonian case study. *Biomass Bioenerg.* **2013**, *57*, 126–135.

Nitrate Leaching Index

Harold M. van Es

Department of Crop and Soil Sciences, Cornell University, Ithaca, New York, U.S.A.

Jorge A. Delgado

Soil Plant Nutrient Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.

Abstract

The Nitrate Leaching Index is a rapid assessment tool that evaluates nitrate (NO_3) leaching potential based on basic soil and climate information. It is the basis for many nutrient management planning efforts, but it has considerable limitations because of: 1) an oversimplification of the processes affecting NO_3 leaching; and 2) a lack of management considerations.

INTRODUCTION

Nitrogen (N) is an essential plant nutrient and the key to the sustainability and economical viability of agricultural systems.^[1] On average, N use efficiencies are being reported to be about 50%, and the economic worldwide average N losses are equivalent to millions of U.S. dollars. The Environmental Protection Agency considers water with over 10 mg nitrate–nitrogen ($\text{NO}_3\text{--N}$) L^{-1} concentration unsafe for drinking purposes,^[2] and several studies have shown that NO_3 leaching from agricultural systems can readily cause this level to be exceeded.^[3,4] The Nitrate Leaching Index (NLI) is an indicator of the potential for NO_3 to reach shallow groundwater and incorporates information on soils and precipitation.^[5] It can be used as a tool to identify land areas where good N management is critical. This entry discusses the processes affecting NO_3 leaching in soils, explains the NLI and its strengths and weaknesses, and provides suggestions for improvement.

PROCESSES

The pathways of N losses need to be viewed within the context of the N cycle and the need to budget all N sources.^[6] N can readily be transformed from organic forms and ammonia fertilizer forms into $\text{NO}_3\text{--N}$, which is mobile in soil and therefore subject to leaching. However, plants require most N to be in the NO_3 form; therefore, the main objective of good agricultural N management is to have sufficient $\text{NO}_3\text{--N}$ available in the root zone for adequate plant nutrition but to avoid any excess N that would lead to high leaching losses. This is a challenge in many crop production systems because: 1) soil by its very nature is “leaky” and water percolation and leaching of chemical

substances is often unavoidable; 2) N additions cannot always be precisely managed, especially when dealing with organic sources; and 3) soil N is subject to other loss pathways such as denitrification and subsequent gaseous emission, ammonia volatilization, surface runoff, and erosion. These processes are often difficult to predict because they are affected by site-specific factors (e.g., fine-textured soils experience more denitrification than coarse-textured ones^[7]) and are strongly dependent on spurious climate factors such as temperature and precipitation. Improved management of N requires an understanding of these processes within the context of the N cycle^[6] and a recognition that inadequate predictability of the various loss pathways often causes farmers to use excess (insurance) N rates to avoid plant N deficiencies under worst-case scenarios. Nevertheless, agricultural N can be effectively managed to reduce $\text{NO}_3\text{--N}$ leaching,^[8] and the NLI is a tool developed for this purpose. We will discuss the available tool and its applications, as well as possible future improvements.

THE NLI

The extent of soil water percolation and thereby $\text{NO}_3\text{--N}$ leaching *potential* mostly depends on permeability, pore-size distribution, soil depth to a restrictive layer, artificial drainage, and precipitation amount and distribution over the year. For a given precipitation pattern, well-drained soils generally have greater N leaching potential than poorly drained soils. The NLI estimates this percolation potential from the Natural Resources Conservation Service soil hydrologic unit designations, which range from Group A soils with high percolation rates (generally coarse-textured and well-drained soils) to Group D soils with low percolation rates (generally fine-textured and poorly

drained soils). The NLI is an indicator of the potential for NO_3 to reach shallow groundwater and incorporates information on soils and precipitation as follows:^[5]

$$\text{NLI} = \text{percolation index} \times \text{seasonal index} \quad (1)$$

The percolation index (PI) is a function of the annual average precipitation (P_A , in inches) and soil hydrologic group and is estimated by the following equations for each of the hydrologic groups:^[9]

$$\text{Hydrologic Group A: } \text{PI} = (P_A - 10.28)^2 / (P_A + 15.43)$$

$$\text{Hydrologic Group B: } \text{PI} = (P_A - 15.05)^2 / (P_A + 22.57)$$

$$\text{Hydrologic Group C: } \text{PI} = (P_A - 19.53)^2 / (P_A + 29.29)$$

$$\text{Hydrologic Group D: } \text{PI} = (P_A - 22.67)^2 / (P_A + 34.00) \quad (2)$$

Soils with mixed hydrologic grouping, e.g., B/D, as a result of artificial drainage, generally use the higher percolation potential in the NLI assessment. The seasonal index (SI) is determined by the annual precipitation (P_A) and the sum of the non-growing season precipitation (typically the winter period in the United States, P_N in inches)

$$\text{SI} = (2 \times P_N / P_A)^{1/3} \quad (3)$$

The SI reflects the fact that most water percolation occurs from precipitation after crop senescence and removal, as a result of the consequent drop in transpiration rate. NLI values are generally interpreted as follows

$\text{NLI} < 2$: Low nitrate leaching risk

$2 < \text{NLI} < 10$: Medium nitrate leaching risk

$\text{NLI} > 10$: High nitrate leaching risk

Despite its simplicity, the NLI is generally effective in identifying high-leaching scenarios from low-leaching scenarios^[10] and has been widely adopted in the United States for nutrient management planning on farms.

FRAMEWORK FOR AN IMPROVED NLI

Although the NLI allows for the assessment of NO_3 -N leaching potential, it also has significant limitations because of its oversimplification of complex processes.^[11] The NLI does not actually estimate the leaching of NO_3 -N and therefore cannot be directly tied to an enforceable water-quality standard (e.g., $10 \text{ mg L}^{-1} \text{ NO}_3$ -N). The index also ignores important processes that affect N leaching, such as denitrification, which is especially significant in fine-textured soils.^[4] van Es et al.^[10] concluded that despite this shortcoming, the NLI still effectively identifies soil and climate regions of high and low leaching potential.

Another major limitation of the NLI is the lack of a connection with management practices, such as N application rates and timing, crop type, and crop rotations, which greatly influence NO_3 -N leaching. This concern has, in some cases, been addressed by providing specific management guidelines for fields with high NLI values, such as conservative N applications, optimum timing of applications, cover cropping, etc.^[10]

Shaffer and Delgado^[11] proposed a framework for an improved NLI through a *technologically based index* that requires the integration of databases describing soil hydraulic properties, climate, off-site factors and management, the use of simulation models, and the Internet. This NLI is proposed to use a dynamic ranking system similar to that for the PI,^[12] which can facilitate a preliminary assessment by classifying combinations of management scenarios, soil conditions, climate, crop types and off-site factors into leaching potential ratings. This NLI should be simple enough that it can be used by consultants, agronomists, conservationists, farmers, and technical personnel. It is proposed to have a tiered approach, where initial efforts involve simple screening to separate the higher NO_3 -N leaching potential scenarios from the lower ones.^[11] Higher tiers of this NLI should be capable of assessing N leaching potential within the context of more complex N dynamics, transformations, and mobility in the root zone. Also, this NLI must be developed so it can be simultaneously used with the PI to perform integrated analyses of nutrient loss potential, which allows for the evaluation of environmental trade-offs associated with management practices. For example, early fall manure application in the cool temperate regions of the United States is often optimal for avoiding phosphorus runoff losses but poses high N leaching risk compared to other seasonal application periods.^[10]

At the first tier of application, the NLI should be simple enough to be applied across large regions. At the higher, more complex levels, the NLI should be capable of evaluating the leaching potential related to site-specific management scenarios by being coupled to geographic information system and capable of identifying variable NO_3 -N leaching areas across single fields of various soil characteristics and management scenarios. A final attribute of a new NLI should be its national scope and consistency but not necessarily identical formulation. This allows for uniform standards and effective communication among technical service providers and scientists.^[11]

A new NLI needs to be developed with mechanistic dynamic simulation models (e.g., GPFARM, EPIC, LEACHM, NLEAP, GLEAMS, and RZWQM) that can account for the complex N pathways, transformations, and interactions with other nutrients.^[13] This is especially important for the evaluation of cases where organic N sources are being applied to the fields, which are generally of the greatest concern with nutrient losses.

CONCLUSION

The NLI is a rapid assessment tool that evaluates N leaching potential based on basic soil and climate information. It is the basis for many nutrient management planning efforts but has considerable limitations because of: 1) an oversimplification of the processes affecting N leaching and 2) a lack of management considerations. Improved N management in the landscape requires a new NLI that considers the complex interactions of climate conditions, soil characteristics, crop type, off-site factors, and management scenarios. A tiered approach is proposed to achieve these multiple objectives.^[11]

REFERENCES

- Hallberg, G.R. Nitrate in ground water in the United States. In *Nitrogen Management and Ground Water Protection*; Follett, R.F., Ed.; Elsevier: New York, 1989; 35–74.
- U.S. EPA. *Basic Information about Nitrate in Drinking Water*. Available at <http://water.epa.gov/drink/contaminants/basicinformation/nitrate.cfm> (accessed August 20, 2015).
- Randall, G.W.; Huggins, D.R.; Russelle, M.P.; Fuchs, D.J.; Nelson, W.W.; Anderson, J.L. Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa and row crop systems. *J. Environ. Qual.* **1997**, *26*, 1240–1247.
- Sogbedji, J.M.; van Es, H.M.; Yang, C.L.; Geohring, L.D.; Magdoff, F.R. Nitrate leaching and N budget as affected by maize N fertilizer rate and soil type. *J. Environ. Qual.* **2000**, *29*, 1813–1820.
- Williams, J.R.; Kissel, D.E. Water percolation: An indicator of nitrogen-leaching potential. In *Managing Nitrogen for Groundwater Quality and Farm Profitability*; Follett, R.F., Ed.; Soil Science Society of America Inc.: Madison, 1991; 59–83.
- Delgado, J.A. Quantifying the loss mechanisms of nitrogen. *J. Soil Water Conserv.* **2002**, *57*, 389–398.
- Moisier, A.R.; Doran, J.W.; Freney, J.R. Managing soil denitrification. *J. Soil Water Conserv.* **2002**, *57*, 505–513.
- Meisinger, J.J.; Delgado, J.A. Principles for managing nitrogen leaching. *J. Soil Water Conserv.* **2002**, *57*, 485–498.
- Pierce, F.J.; Shaffer, M.J.; Halvorson, A.D. Screening procedure for estimating potentially leachable nitrate–nitrogen below the root zone. In *Managing Nitrogen for Groundwater Quality and Farm Profitability*; Follett, R.F., Ed.; Soil Science Society of America, Inc.: Madison, 1991; 259–283.
- van Es, H.M.; Czymmek, K.J.; Ketterings, Q.M. Management effects on N leaching and guidelines for an N leaching index in New York. *J. Soil Water Conserv.* **2002**, *57*, 499–504.
- Shaffer, M.J.; Delgado, J.A. Essentials of a national nitrate leaching index assessment tool. *J. Soil Water Conserv.* **2002**, *57*, 327–335.
- Sharpley, A.N.; Daniel, T.; Sims, T.; Lemunyon, J.; Stevens, R.; Parry, R. *Agricultural Phosphorus and Eutrophication*; USDA-ARS: Washington, 1999.
- Delgado, J.A. Use of simulations for evaluation of best management practices on irrigated cropping systems. In *Modeling Carbon and Nitrogen Dynamics for Soil Management*; Shaffer, M.J., Ma, L., Hansen, S., Eds.; Lewis Publishers: Boca Raton, 2001; 355–381.

Nitrate Leaching Management

John J. Meisinger

U.S. Department of Agriculture (USDA), Beltsville, Maryland, U.S.A.

Jorge A. Delgado

Soil Plant Nutrient Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.

Ashok Alva

Vegetable and Forage Crop/Production Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Prosser, Washington, U.S.A.

Abstract

Nitrate (NO_3) leaching is a significant nitrogen (N) loss process for agriculture that must be managed to minimize NO_3 enrichment of groundwater and surface waters. Managing NO_3 leaching should involve the application of basic principles of understanding the site's hydrologic cycle, avoiding excess rates of N, and applying N in phase with crop demand. Other specific techniques to reduce NO_3 leaching are presented in this entry.

INTRODUCTION

Nitrate leaching occurs when the soil nitrate–nitrogen ($\text{NO}_3\text{--N}$) concentrations are high and water moves beyond the root zone. Leaching losses in modern agriculture commonly account for 10–30% of the N additions.^[1–3] Leaching can contribute to NO_3 enrichment of groundwater, which has a health advisory limit of 10 mg $\text{NO}_3\text{--N/L}$, and to the eutrophication of surface waters that can lead to the development of hypoxic zones in receiving waters. Managing leaching requires development of site-specific practices that should be based on an understanding of the soil–crop–hydrologic cycle, avoiding excess N by applying a N rate to meet expected yields, and applying N in phase with crop demand. Specific NO_3 management approaches include adjusting irrigation inputs according to site water needs, employing cropping systems that fully utilize soil–water resources, and utilizing within-season and real-time N monitoring tools. The goal of this entry is to discuss practical techniques to reduce NO_3 leaching from modern agriculture.

APPROACHES FOR DECREASING NO_3 LEACHING

Primary Techniques for Managing NO_3 Leaching

The major NO_3 leaching management techniques include understanding the soil–crop–hydrologic cycle, applying the proper rate of N, and applying N in phase with crop demand. Because water movement is the driving force for

NO_3 leaching, it is essential to understand the hydrologic cycle before developing site-specific leaching management strategies.^[1] Understanding the hydrologic cycle of the site should identify the most likely times when leaching can occur, i.e., the times when the soil is at near field capacity and water inputs exceed water use by evapotranspiration (ET). In humid regions, leaching is usually minimal in summer when ET exceeds precipitation. But during the winter-to-spring season, precipitation exceeds ET and leaching is common, as shown by the percolate data of Fig. 1, adapted from large monolith lysimeter data from Ohio, U.S.A.^[3] In irrigated agriculture, leaching is most likely to occur during the growing season from excess water inputs on coarse-textured soils. Controlling leaching in humid regions should focus on management practices to keep the soil $\text{NO}_3\text{--N}$ levels low during the fall season and on providing a crop N sink during the non-growing season, such as a grass cover crop. In irrigated agriculture, avoiding excess irrigation inputs through irrigation scheduling is an effective method to manage leaching.

Avoiding excess N inputs, from fertilizer and/or manure, is the most fundamental approach to control leaching because crop N use efficiency is usually high for plants responding to N. By contrast, N rates on the non-responsive part of the yield curve are associated with lower efficiencies and high levels of unused N that is vulnerable to leaching. The data of wheat grain N removals (assuming 20 kg N/t grain) from the Rothamsted Broadbalk Study are summarized in Fig. 2; the associated estimate of N leaching is adapted from the study by Gouldings.^[4] These data show only small leaching losses for N rates in the responsive part

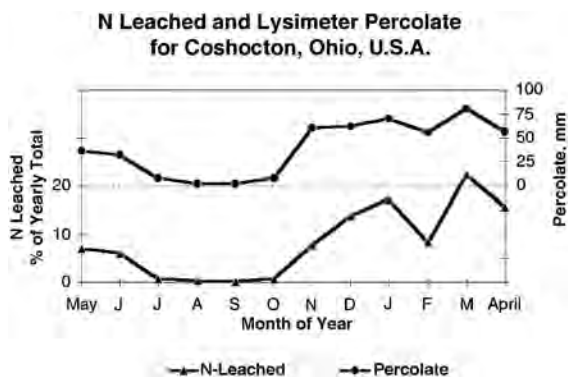


Fig. 1 Monthly lysimeter percolate (upper panel) and N leached (lower panel, as percent of total yearly N leached) for large monolith lysimeters in Coshocton, Ohio, U.S.A.

Source: From Chichester & Smith.^[3]

of the curve, with a marked increase in leaching in the non-responsive portion of the curve (Fig. 2). Applying the proper N rate usually involves estimating the crop N need from the expected yield and then subtracting N credits from residual $\text{NO}_3\text{-N}$, irrigation N inputs, prior legume crop credits, and adjusting for available manure N.^[1]

Applying N in phase with crop demand is also a fundamental approach for managing leaching that involves applying N after crop establishment, or after each forage harvest in grass-forage systems. Timing N applications to closely match crop N uptake minimizes the time that N is exposed to uncontrolled rainfall inputs and thus leads to lower leaching losses.

Irrigation and Cropping System Techniques for Managing NO_3 Leaching

Other NO_3 management techniques include irrigation scheduling, modifying the cropping system, and developing

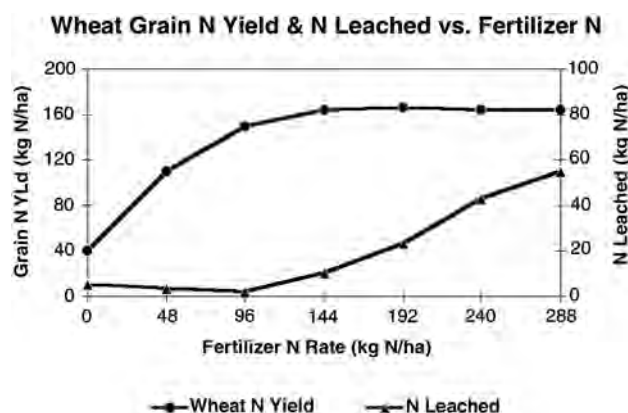


Fig. 2 Wheat grain N yield and N leaching for various fertilizer rates; Broadbalk Study, Rothamsted, U.K.

Source: From Gouldings.^[4]

riparian zones and conservation acres. Irrigation water management techniques are based on irrigation scheduling, which adds irrigation in accordance with ET and soil-water status. Irrigation scheduling can reduce leaching by avoiding excess water applications, by adjusting water inputs to local weather, and by adding water based on local soil properties and local soil-water content. For example, irrigating at 85% of ET, compared to 100% of ET, reduced leaching losses from about 110 to 60 kg N/ha on a sandy soil in Nebraska, U.S.A.^[5] Similarly, Diez et al.^[6] reported that scheduled irrigations to account for crop water demands can reduce the leaching of water and of $\text{NO}_3\text{-N}$ by 4 times, compared to excess irrigation. Meisinger and Delgado^[1] summarized cropping system effects on leaching and concluded that adding a grain legume, such as soybeans, can partially reduce leaching compared to continuous corn (typically 5–10%), but adding a forage crop, such as alfalfa or a forage grass, can substantially reduce leaching (typically 70–90%). However, changing the cropping system requires simultaneous changes in marketing and/or livestock enterprises. Delgado^[7] reported that cropping systems that have shallow-rooted crops and are heavily fertilized are most susceptible to $\text{NO}_3\text{-N}$ leaching. However, N recovery can be significantly improved by adding a deep-rooted crop such as malting barley. The barley served as a scavenger crop and even mined $\text{NO}_3\text{-N}$ from the system.^[7]

Adding a grass cover crop is another cropping system approach to reduce leaching by converting mobile $\text{NO}_3\text{-N}$ to immobile plant protein. A review of the effects of cover crops on water quality^[8] concluded that grass covers can reduce NO_3 leaching by an average of 70% compared to no-cover, while legume covers can reduce leaching by about 20%. Cover crops are especially useful in humid regions, where the winter leaching potential is high (Fig. 1). Developing riparian zones and conservation reserve areas can reduce the N loss from production fields into adjacent streams, especially in tile-drained watersheds.^[1]

Other Approaches for Managing NO_3 Leaching

Meisinger and Delgado^[1] have also discussed other leaching management techniques. Nitrification inhibitors can delay leaching losses but are most effective in conjunction with other techniques, such as reduced N rates. Changing tillage practices usually have only secondary effects on leaching. Drainage-ditch water control measures can reduce NO_3 transport to streams by impounding water and promoting denitrification but have limited geographic applicability. Improved N application equipment is a direct approach to improve N application accuracy and avoiding excess N rates; new application equipment can apply N at the meter and submeter spatial scale.

Within-Season Monitoring Techniques for Managing NO₃ Leaching

In-season testing and real-time sensors are tools developed for managing leaching. These monitoring approaches seek to identify N sufficient or deficient areas within a field and adjust subsequent N applications accordingly. The leaf chlorophyll meter (LCM) compares leaf greenness to the LCM reading in a well-fertilized reference strip. The relative LCM value has been shown to successfully predict the need for extra fertilizer N, especially in irrigated systems where N can be applied with irrigation water.^[9] The pre-sidedress soil nitrate test (PSNT) measures NO₃-N in the surface 30 cm of soil when corn is about 30 cm tall and compares it to a sufficiency concentration of 20–25 mg NO₃-N/kg. The PSNT is the most useful for diagnosing N sufficient sites. A study^[10] carried out in Connecticut, U. S.A., compared conventional N management with the PSNT and reported N leaching losses of 50 and 20 kg N/ha, respectively. Plant stem or petiole NO₃-N tests measure plant NO₃-N sufficiency at specific growth stages and are commonly used in vegetable crops.^[1]

Real-time sensors that measure crop biomass and greenness based on remotely sensed or tractor-mounted units sense red and near infrared reflectance^[11] and are new tools for managing N leaching. Data from real-time sensors have a meter to submeter resolution and can be combined with geographically mapped data of soil properties and previous crop yields to produce a real-time assessment of N status, potential yield, and the suggested N application based on crop simulation models such as nitrate leaching and economic analysis package.^[7] These above seasonal and real-time monitors have been shown to reduce NO₃ leaching by identifying N sufficient areas and avoiding the applications of excess N; they can also increase profitability by identifying N deficient areas with excellent small-scale resolution. An example of the benefits from Precision Agriculture has been reported by Bausch and Delgado^[11] who reported that using geographic information system (GIS) and remote sensing tools for in-season N management resulted in N applications that required 52% less N than that used under conventional practices (214 kg N/ha/yr). On average, the in-season N management saved 102 kg N/ha/yr which was worth about \$55/ha/yr, without yield reductions during two consecutive growing seasons. These results show that precision management can significantly improve N efficiency of corn systems without reducing grain yields, thus minimizing the potential for NO₃-N leaching.

CONCLUSION

NO₃ leaching is a significant N loss process for agriculture that must be managed to minimize NO₃ enrichment of groundwater and surface waters. Managing NO₃ leaching

should involve the application of basic principles of understanding the site's hydrologic cycle, avoiding excess rates of N, and applying N in phase with crop demand. Other specific techniques to reduce leaching include use of irrigation scheduling, grass cover crops, within-season monitoring with soil NO₃-N tests or LCMs, real-time sensors using red and near infrared reflectance, and use of remote sensing with GISs and simulation models to identify the best combination of practices to control leaching. The application of a specific combination of the above practices should increase crop N recovery with concomitant reductions in NO₃-N leaching.

REFERENCES

1. Meisinger, J.J.; Delgado, J.A. Principles for managing nitrogen leaching. *J. Soil Water Conserv.* **2002**, *57* (6), 485–498.
2. Legg, J.O.; Meisinger, J.J. Soil nitrogen budgets. In *Nitrogen in Agricultural Soils*. Monograph No. 22; Stevenson, F.J., Bremner, J.M., Hauck, R.D., Keeney, D.R., Eds.; American Society of Agronomy: Madison, 1982; 503–566.
3. Chichester, F.W.; Smith, S.J. Disposition of ¹⁵N-labeled fertilizer nitrate applied during corn culture in field lysimeters. *J. Environ. Qual.* **1978**, *7* (2), 227–233.
4. Gouldings, K. Nitrate leaching from arable and horticultural land. *Soil Use Manage.* **2000**, *16* (1), 145–151.
5. Hergert, G.W. Nitrate leaching through sandy soil as affected by sprinkler irrigation management. *J. Environ. Qual.* **1986**, *15* (3), 272–278.
6. Diez, J.A.; Roman, R.; Caballero, R.; Caballero, A. Nitrate leaching from soils under a maize-wheat-maize sequence, two irrigation schedules and three types of fertilizers. *Agr. Ecosyst. Environ.* **1997**, *65* (1), 189–199.
7. Delgado, J.A. Use of simulations for evaluation of best management practices on irrigated cropping systems. In *Modeling Carbon and Nitrogen Dynamics for Soil Management*; Shaffer, M.J., Ma, L., Hansen, S., Eds.; Lewis Publishers: Boca Raton, 2001; 355–381.
8. Meisinger, J.J.; Hargrove, W.L.; Mikkelsen, R.B.; Williams, J.R.; Benson, V.W. Effect of cover crops on groundwater quality. In *Cover Crops for Clean Water*; Hargrove, W.L., Ed.; Soil and Water Conservation Society of America: Ankeny, 1991; 57–68.
9. Blackmer, T.M.; Schepers, J.S. Use of a chlorophyll meter to monitor nitrogen status and schedule fertigation for corn. *J. Prod. Agric.* **1995**, *8* (1), 56–60.
10. Guillard, K.; Morris, T.F.; Kopp, K.L. The pre-sidedress soil nitrate test and nitrate leaching from corn. *J. Environ. Qual.* **1999**, *28* (6), 1845–1852.
11. Bausch, W.C.; Delgado, J.A. Ground base sensing of plant nitrogen status in irrigated corn to improve nitrogen management. In *Digital Imaging and Spectral Techniques: Applications to Precision Agriculture and Crop Physiology*; Van Toai, T., Major, D., McDonald, M., Schepers, J., Tarpley, L., Eds.; Am. Soc. Agronomy Spec. Pub. 66; American Society of Agronomy: Madison, 2003; 151–163.

Nitrogen and its Transformations

Oswald Van Cleemput

Pascal Boeckx

Faculty of Agricultural and Applied Biological Sciences, University of Ghent,
Ghent, Belgium

Abstract

Nitrogen (N) is essential to all life. It is the nutrient that most often limits biological activity. In agricultural and natural ecosystems, N occurs in many forms covering a range of valence states from -3 to $+5$. The change from one valence state to another depends primarily on environmental conditions. The transformations and flow from one form to another constitute the basics of the soil N cycle. The use of N fertilizers has become essential to increase the productivity of agriculture and has resulted in an almost doubling of the global food production. However, this also implies that the natural N cycle has substantially been disturbed. In the following sections, an overview of the different N transformation processes in the soil is given.

THE NITROGEN (N) CYCLE: GENERAL

Atmospheric nitrogen gas (N_2 ; valence 0) can be converted by lightening to various oxides and finally to nitrate (NO_3^- ; valence $+5$), which can be deposited and taken up by growing plants (Fig. 1). Also N_2 can be converted to ammonia (NH_3 ; valence -3) by biological N_2 fixation, with the NH_3 participating in a number of biochemical reactions in the plant. When plant residues decompose, the N compounds undergo a series of microbial conversions (mineralization), leading first to the formation of ammonium (NH_4^+ ; valence -3) and possibly ending up in NO_3^- (nitrification). Under anaerobic conditions, NO_3^- can be converted to various N oxides and finally to N_2 (denitrification). When mineral or organic N fertilizers are used, they also undergo the same transformation processes and influence the rate of other N transformations. In considering the soil compartment, there can be N gains (such as biological N_2 fixation) as well as N losses (such as leaching and denitrification). Furthermore, N can be exported from the soil via harvest products or immobilized in soil organic matter.

N TRANSFORMATIONS IN THE SOIL

The principal forms of N in the soil are NH_4^+ , NO_3^- , or organic N substances. At any moment, inorganic N in the soil is only a small fraction of the total soil N. Most of the N in a surface soil is present as organic N. It consists of proteins (20–40%), amino sugars, such as the hexosamines (5–10%), purine and pyrimidine derivatives (1% or less), and complex unidentified compounds formed by the reaction of NH_4^+ with lignin, polymerization of quinones with

N compounds, and condensation of sugars and amines. In the subsoil, an important fraction of the present N can be trapped in clay lattices (especially *illitic* clays) as non-exchangeable NH_4^+ and is consequently largely unavailable. Organic substances slowly mineralize by microorganisms to NH_4^+ , which could be converted by other microorganisms to NO_3^- (see further).

The NH_4^+ can be adsorbed to negatively charged sites of clay minerals and organic compounds. This reduces its mobility in the soil compared to the more mobile NO_3^- ion.

Microorganisms can use both NH_4^+ and NO_3^- to satisfy their need for N. This type of N transformation is called microbial immobilization.

The ratio between carbon (C) and N (C/N ratio) in organic matter determines whether immobilization or mineralization is likely to occur. When utilizing organic matter with a low N content, the microorganisms need additional N, decreasing the mineral N pool of the soil. Thus, the incorporation of organic matter with a high C/N ratio (e.g., cereal straw) results in immobilization. The incorporation of organic matter with a low C/N ratio (e.g., vegetable or legume residues) results in N mineralization. A value of the C/N ratio of 25–30 is often taken as the critical point toward either immobilization or mineralization.

Nitrification is a two-step process. In the first step, NH_4^+ is converted to nitrite (NO_2^- ; valence $+3$) by a group of obligate autotrophic bacteria known as *Nitrosomonas* species. The second step is carried out by another group of obligate autotrophic bacteria known as *Nitrobacter* species. Also a few heterotrophs can carry out nitrification, usually at much lower rates.

Soil water and aeration are crucial factors for nitrification. At a water potential of 0 kPa (saturation), there is little air in the soil and nitrification stops, due to oxygen

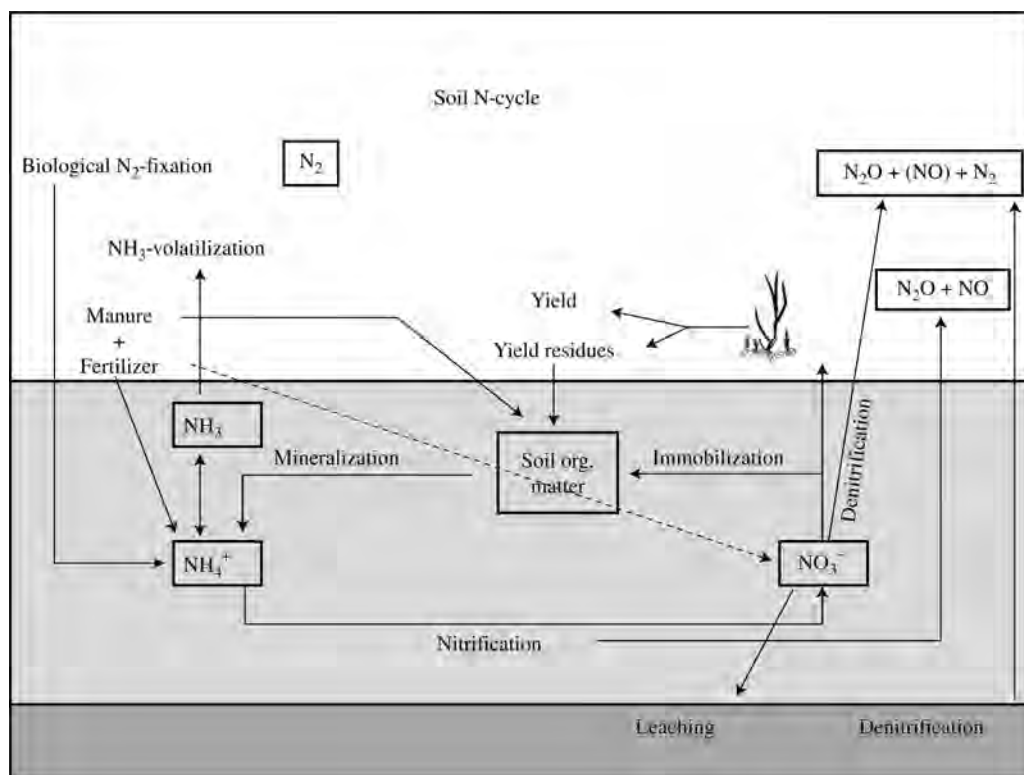


Fig. 1 The soil N cycle. The white compartment represents the atmosphere, the light gray compartment represents the biosphere, and the dark gray compartment represents the subsoil.

limitation; nitrification is greatest near field capacity (−33 kPa in medium- to heavy-textured soils to 0 to −10 kPa in light sandy soils). Also in dry soils, NH_4^+ and sometimes NO_2^- accumulate presumably because *Nitrobacter* species are more sensitive to water stress than the other microorganisms.

Nitrification is slow in acid conditions with an increasing rate at increasing pH. Mainly under alkaline conditions, NO_2^- is also accumulating, because *Nitrobacter* is known to be inhibited by NH_3 , which is formed under alkaline conditions. Nitrification is a process that acidifies the soil as protons (H^+) are liberated as follows:



During nitrification, minor amounts of nitrous oxide (N_2O ; valence +1) and nitric oxide (NO ; valence +2) are formed. Both compounds have environmental consequences, as discussed below.

The effect of temperature on nitrification is climate dependent. There is a climatic selection of species of nitrifiers, with those from cooler regions having lower temperature optima and less heat tolerance than species from warmer regions. All abovementioned factors influencing nitrification also influence the nitrifying population. The population and activity of nitrifiers can be reduced by the use of nitrification inhibitors, such as dicyanodiamide,

nitrapyrin, and neem (*Azadirachta indica*) seed cake. They are used mostly to retard the nitrification of manure; otherwise, their practicality is controversial, and they are not extensively used. More details about nitrification and nitrification inhibitors can be found in the studies by McCarty^[1] and Prosser.^[2]

N INPUT PROCESSES

Atmospheric N Deposition

The total atmospheric N (NH_4^+ and NO_3^-) deposition is in the order of 10–40 kg N ha^{−1} yr^{−1} in much of north-western and Central Europe and some regions in North America. It ranges from 3 to 5 kg N ha^{−1} yr^{−1} in pristine areas.^[3] It is originating from previously emitted NH_3 and NO_x from agricultural and industrial activities or traffic.

Biological N Fixation

Rhizobium species living in symbiotic relationship in root nodules of legumes—e.g., clover (*Trifolium*), lucerne (*Medicago*), peas (*Pisum*), and beans (*Faba*)—can convert atmospheric N_2 to NH_3 , which is further converted to amino acids and proteins. Parallel to this process, the *rhizobium* species receive from the legume the energy they

need to grow and to fix N_2 . Photosynthetic cyanobacteria are also N-fixing organisms and are especially important in paddy rice (*Oryza*). The amount of N fixed varies greatly from crop to crop, ranging from a few $kg\ N\ ha^{-1}\ yr^{-1}$ to a few hundred $kg\ N\ ha^{-1}\ yr^{-1}$. The process is depressed by ample N supply from other sources, and it is sensitive to the lack of phosphorus. The amount of globally fixed N is almost double the amount of applied fertilizer N. Next to symbiotic N-fixing bacteria, also non-symbiotic species (e.g., *Azotobacter*) occur in soils. In general, free-living diazotrophs make a small but significant contribution to the soil N status. Some non-leguminous trees and plants [e.g., alder (*Alnus*) and sugarcane (*Saccharum*)] host N-fixing bacteria as well. Much uncertainty exists about the association of N-fixing bacteria with nonlegumes (the so-called associative N-fixing bacteria).

Mineral and Organic N Fertilization

Theoretically, plants should prefer NH_4^+ to NO_3^- because NH_4^+ does not need to be reduced before incorporation into the plant. In most well-drained soils, the oxidation of NH_4^+ is fairly rapid, and therefore, most plants have developed to grow better with NO_3^- . However, a number of studies have shown that plants better develop when both sources are available. Rice, growing under submerged conditions, must grow in the presence of NH_4^+ , as NO_3^- is not stable under flooded conditions. When urea is applied, it rapidly hydrolyzes under well-drained conditions, unless a urease inhibitor is being added; under submerged conditions, rice plants may also absorb N directly as molecular urea. Organic manure can be of plant or animal origin or a mixture of both. However, most comes from dung and urine from farm animals. It exists as farmyard or stable manure, urine, slurry, or compost. Because its composition is not constant and because plant material (catch or cover crops and legumes) is often added freshly (green manure) to the soil, less than 30% of its nutrients becomes available for the next crop.

N UPTAKE BY PLANTS

Growing plants get their N from fertilizer N as well as from organic soil N upon mineralization. Plants take up N compounds both as NO_3^- and as NH_4^+ . In general, NO_3^- is the major source of plant N. There is some evidence that small amounts of organic N (urea or amino acids) can be taken up by plants from the soils solution. Plant uptake of N can be studied through the use of mineral fertilizers or organic matter labeled with the stable N isotope ^{15}N . The proportion of applied N taken up by the crop is affected by many factors, including crop species, climate, and soil conditions. Aboveground

parts of the crop can recover 40–60% of the fertilizer N applied.

N LOSS PROCESSES

NH_3 Volatilization

Losses of N from the soil by NH_3 volatilization amount globally to 54 Mt (or $10^{12}\ g\ NH_3-N\ yr^{-1}$), and 75% is of anthropogenic origin.^[4] According to the European Centre for Ecotoxicology and Toxicology of Chemicals,^[5] the dominant source is animal manure, and about 30% of N in urine and dung is lost as NH_3 . The other major source is surface application of urea or NH_4^+ bicarbonate and to a lesser degree other NH_4^+ -containing fertilizers. As urea is the most important N fertilizer in the world, it may lead to important NH_3 loss upon hydrolysis and subsequent pH rise in the vicinity of the urea till. The transformation of NH_4^+ to the volatile form NH_3 increases with increasing pH, temperature, soil porosity, and wind speed at the soil surface. It decreases with increasing water content and rainfall events following the application. NH_3 losses from soils can be effectively reduced by fertilizer incorporation or injection instead of surface application.

Emission of N Oxides (N_2O and NO) and Molecular N (Nitrification and Denitrification)

Microbial nitrification and denitrification are responsible for the emission of NO and N_2O .^[6] They are by-products in nitrification and intermediates during denitrification. Probably about 0.5% of fertilizer N applied is emitted as NO ^[7] and 1.25% as N_2O .^[8] However, wide ranges have been reported. The intensification of arable agriculture and of animal husbandry has made more N available in the soil N cycle increasing the emission of N oxides. The relative percentage of NO and N_2O formation very much depends on the moisture content of the soil. At a water-filled pore space (WFPS; or the fraction of total soil pore space filled with water) below 40%, NO is produced mainly from nitrification. Between a WFPS of 40% and 60%, the formation of NO and N_2O from nitrification occurs. Between a WFPS of 60% and 80%, N_2O is predominantly produced from denitrification, and the formation of NO is decreasing sharply. At a WFPS above 80%, the formation of N_2 by denitrification is dominant. In practice, these WFPS ranges will overlap and depend on the soil type.^[9] Next to water content, also temperature, land use, availability of N, and decomposable organic matter are important determining factors for N_2O formation. N_2O is a greenhouse gas contributing 5–6% to the enhanced greenhouse effect. Increased concentrations are also detrimental for the stratospheric ozone layer.^[10] In the presence of sunlight, NO_x (NO and NO_2) react with volatile organic compounds from evaporated petrol and solvents and from vegetation and

forms tropospheric ozone, which is, even at low concentration, harmful to plants and human beings. The major gaseous end product of denitrification is N_2 . The ratio of N_2O to N_2 produced by denitrification depends on many environmental conditions. Generally, the more anaerobic the environment, the greater the N_2 production. Denitrification is controlled by three primary factors (oxygen, NO_3^- , and C), which in turn are controlled by several physical and biological factors. Denitrification N loss can reach 10% of the fertilizer N input—more on grassland and when manure is also applied.^[11] Chemical denitrification is normally insignificant and is mainly related to the stability of NO_2^- and acid conditions.^[12] It is more difficult to reduce N_2O and NO from soils than NH_3 losses. A general principle is to minimize N surpluses in the soil profile via careful fertilizer adjustment, corresponding to the actual crop demands.

Leaching

Applied NO_3^- or NO_3^- , formed through nitrification from mineralized NH_4^+ or from NH_4^+ from animal manure, can leach out of the rooting zone. It is well possible that this leached NO_3^- can be denitrified at other places and returned into the atmosphere. The amount and intensity of rainfall, quantity and frequency of irrigation, evaporation rate, temperature, soil texture and structure, type of land use, cropping and tillage practices, and the amount and form of fertilizer N are all parameters influencing the amount of NO_3^- leaching to the underground water. NO_3^- leaching should be kept under control, as it may influence the NO_3^- content in drinking water influencing human health and in surface water causing eutrophication. NO_3^- losses can be minimized by reducing the mineral N content in the soil profile during the winter period by careful fertilizer adjustment, growing of cover crops, or riparian buffer areas.

REFERENCES

1. McCarty, G.W. Modes of action of nitrification inhibitors. *Biol. Fert. Soils* **1999**, 29, 1–9.
2. Prosser, J.I. *Nitrification, Special Publications of the Society of General Microbiology*; IRL Press: Oxford, 1986; 20 pp.
3. Lagreid, M.; Bockman, O.C.; Kaarstad, O. *Agriculture, Fertilizers and the Environment*; CIBA Publishing: Oxon, 1999; 294 pp.
4. Sutton, M.A.; Lee, D.S.; Dollard, G.J.; Fowler, D. International conference on atmospheric ammonia: Emission, deposition and environmental impacts. *Atmos. Environ.* **1998**, 32, 1–593.
5. ECETOC. *Ammonia Emissions to Air in Western Europe (No. 62)*; European Centre for Ecotoxicology and Toxicology of Chemicals: Brussels, 1994; 196 pp.
6. Bremner, J.M. Sources of nitrous oxide in soils. *Nutr. Cycl. Agroecosys.* **1997**, 49, 7–16.
7. Veldkamp, E.; Keller, M. Fertilizer-induced nitric oxide emissions from agricultural soils. *Nutr. Cycl. Agroecosys.* **1997**, 48, 69–77.
8. Mosier, A.; Kroeze, C.; Nevison, C.; Oenema, O.; Seitsinger, S.; Van Cleemput, O. Closing the global N_2O budget: Nitrous oxide emissions through the agricultural N cycle. *Nutr. Cycl. Agroecosys.* **1998**, 52, 225–248.
9. Davidson, E.A. Fluxes of nitrous oxide and nitric oxide from terrestrial ecosystems. In *Microbial Production and Consumption of Greenhouse Gases: Methane, Nitrogen Oxides, and Halomethanes*; Rogers, J.E., Whitman, W.B., Eds.; American Society for Microbiology: Washington, 1991; 219–235.
10. Crutzen, P.J. The influence of nitrogen oxides on the atmospheric ozone content. *Q. J. Roy. Meteor. Soc.* **1976**, 96, 320–325.
11. von Rheinbaben, W. Nitrogen losses from agricultural soils through denitrification—A critical evaluation. *Z. Pflanz. Bodenkunde* **1990**, 153, 157–166.
12. Van Cleemput, O. Subsoils: Chemo- and biological denitrification, N_2O and N_2 emissions. *Nutr. Cycl. Agroecosys.* **1998**, 52, 187–194.

Nitrogen Index

Jorge A. Delgado

Soil Plant Nutrient Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.

Abstract

Nitrogen (N) is the key for humanity in the 21st century to help us achieve food security. Adding N inputs to agricultural systems contributes to higher crop yields; however, when inputs are higher than crop needs, the excess N can be lost to the environment via different pathways and contribute to negative impacts on air and water quality. There is a need to continue improving N management practices to increase N use efficiencies and reduce losses of N to the environment. The N index is a quick tool and approach that can provide needed information to nutrient managers to quickly assess the effects of management practices on the potential risk of N losses via different pathways. By quickly identifying these risky management landscape scenarios, better management practices for the site could be implemented. The N index can provide important agronomical and environmental information to nutrient managers to help them make improved management decisions.

NITROGEN (N) IS KEY TO SOLVE THE CHALLENGE OF ACHIEVING GLOBAL FOOD SECURITY

The global human population is projected to increase by 2 billion people by 2050 to a total population of approximately 9.5 billion people.^[1] At the same time, improved economies around the globe are contributing to healthier diets with higher demand for protein from livestock and dairy products.^[2] These factors are projected to contribute to greater global demand for calories over the next few decades. We must increase crop production to meet this increased demand for calories and protein and feed the 9.5 billion people who will be living on the earth by 2050.^[2] Humans will have to produce 70% more food during the next few decades.^[3]

Both crops and animals respond to N inputs, and N will be key to feeding humanity during the next few decades since it will contribute to increases in food production per area as well as to improved, higher protein diets. N inputs will be key for increasing grain, livestock, and dairy product production. Conservation of natural resources will be key for sustaining higher agricultural production. In particular, managing the N cycle to reduce losses of reactive N to the environment will be important. In other words, good management practices to manage the N cycle will be essential.

Among other great challenges are irrigation systems that are threatened by overpumping of groundwater resources and reduction of snowpack. Sustainability of irrigation systems is key because, on average, these systems are more productive per average unit of land than nonirrigated systems. To increase the production of these

irrigated systems to 200% or 300% higher than that of nonirrigated systems, N inputs are essential. However, these systems need to be managed with improved irrigation practices as well as best N management practices to reduce the potential for reactive N losses since these systems on average have a higher nitrate (NO₃) leaching potential, especially cropping systems with shallow root systems or when crops are grown over irrigated, coarse-textured soils. It has been reported that other pathways for NO₃ leaching include tile systems that accelerate the movement of NO₃ out of the root zone and typically into drainage ditches, contributing to the contamination of surface water with problems such as hypoxia in the Gulf of Mexico. On average, NO₃ leaching has been reported as the main or one of the main mechanisms for N losses.

Another challenge is the potential impacts of climate change on agricultural systems. It has been projected that climate change and extreme events will increase erosion rates. Similarly, droughts could also contribute to increased erosion rates. Soil erosion, besides removing important nutrients like N off-site, reduces productivity by an average of 4% for every 10 cm loss of soil, with higher loss in yields, as the soil continues to be eroded.^[4] As climate change contributes to higher erosion rates, it is expected that these higher erosion rates will contribute to increased off-site N impacts since the transport of N off-site will increase. Fortunately, conservation practices will help reduce these higher erosion rates expected to result from climate change and extreme events and will contribute to climate change adaptation.

There are other general challenges such as desertification, deforestation, and conversion of productive lands to urban centers. If we maximize agricultural production,

we need to increase productivity ahead of all of these challenges that humanity is facing, and N inputs will be key for increasing grain and forage production to meet the higher demand for meat and dairy products. N will be essential for meeting the demand for agricultural and animal products for feed, biofuels, meat, and dairy products. Worldwide food security will be connected to good stewardship and good N management practices that increase productivity while minimizing environmental impacts of N.

ENVIRONMENTAL CHALLENGES AND N MANAGEMENT

N has contributed to food security across a large number of regions; however, in addition to the significant positive impacts of N use that have been reported, it has also been reported that inputs of organic and inorganic N fertilizers have contributed to increased losses of N. These increased flows of N out of agricultural systems are contributing to an increase in losses of reactive N via surface runoff, NO_3 leaching, nitrous oxide (N_2O) emissions, ammonia (NH_3) volatilization, and other pathways. These losses of N contribute to a phenomenon known as the N cascade, where there are multiple effects as the reactive N is cycled through the environment.^[5]

It has been reported that surface N losses, as well as NO_3 leaching losses via tile systems, are pathways for N losses that contribute to negative impacts on surface water (e.g., hypoxia, higher cost at drinking water plants to remove NO_3 from water). NO_3 leaching losses have contributed to negative impacts on groundwater quality by increasing background NO_3 -N levels. The U.S. Environmental Protection Agency has reported that drinking water with NO_3 -N concentrations above 10 ppm is unsafe for human consumption, requiring that the NO_3 is removed from the water before it can be used by urban centers and/or rural communities connected to potable water from drinking water plants.^[6] The cost of removing the NO_3 from water in the United States has been estimated at \$4.8 billion per year, and of this cost, an estimated \$1.7 billion per year is from the NO_3 -N coming from agricultural sources.^[7]

Atmospheric losses of reactive N can contribute to negative impacts on air quality. Emissions of NH_3 have been reported to increase the concentrations of particulate matter 2.5, which can potentially impact air quality. It can also contribute to the transport of N to natural areas and impact natural agroecosystems by contributing to shifts in plant ecosystems.^[8] Atmospheric emissions of N_2O are also impacting air quality. The Intergovernmental Panel on Climate Change has reported that atmospheric emissions of N_2O can contribute to global warming and climate change, and emissions of N_2O from agricultural systems are an important source of greenhouse gas emissions. In the United States, the major contributor to emissions of

greenhouse gases from agricultural systems is N_2O emission from soils.^[9] In the United States, N_2O is contributing to about 80% of the greenhouse gas emissions from agriculture in carbon dioxide equivalents. About 20% of these carbon dioxide equivalent emissions are due to the indirect losses of reactive N to the environment (e.g., NH_3 volatilization and NO_3 leaching).

Management can potentially be used to reduce N losses to the environment. We could use, for example, nitrification inhibitors and controlled-release fertilizers to slow down the conversion of NH_4 to NO_3 -N to reduce the potential for NO_3 leaching.^[10,11] These practices also reduce the emission of N_2O .^[10,11] We could use precision agriculture to improve the spatial and temporal management of N by applying the right source of N and the right rate of N, at the right place and at the right time, to increase its use efficiency and reduce the losses to the environment. Precision conservation goes a step further and contributes to reduced losses of N to the environment at the field level and to reduced transport of N through the landscape, by implementing practices at the field and farm level to reduce off-site transport. There is potential to increase N use efficiency and reduce transport of N by applying the right fertilizer, with the right method of fertilizer application, with the right source of fertilizer, applied with the right conservation practice, with the conservation practice applied at the right scale and at the right place, and with the fertilizer and conservation practice applied at the right time.

PREVIOUS N INDEX WORK

There is potential to use quick assessments of management practices to assess the potential risk of N losses to the environment.^[12,13] There is also potential to use a tiered approach to manage N.^[12] The N index is a tier-one tool that is simple to use but robust enough to quickly assess the effects of a given set of management practices on the potential risk of N loss (ranked as low, medium, or high risk) using information on soil properties and soil hydrology, crop N uptake, inorganic and organic N inputs, and weather. Tier-two and tier-three tools are more complex tools and approaches that require more time for calibration to implement N management practices.

Several authors have reported on several different indices and on the advantages and disadvantages of these indices.^[12,14] The leaching index was the initial work that accounted for soil and climate and how it potentially influenced NO_3 -N leaching.^[15] However, the initial leaching index was more like a percolation water index since it was not tied to N management or N inputs.^[15,16] Among other indices reported in the literature^[12,17] are the movement risk index, the NO_3 -N available for leaching index, the residual NO_3 -N index, NO_3 leaching hazard index, the N use efficiency index, annual leaching risk potential index, and the aquifer risk index.

Each one of these indices has some advantages and disadvantages. For example, some of the indices do not estimate the actual $\text{NO}_3\text{-N}$ leaching. Additionally, some of the indices are not tied to N management practices and do not account for estimated crop N uptake or estimated N dynamics and N sources. Some of the indices do not estimate the impact of crop rotations or rooting depths. Some of the indices do not account for hydrological properties or irrigation events or water budgets. Some of the indices do not account for potential NO_3 leaching during the off-season or do not consider off-site factors. Shaffer and Delgado proposed an N index that integrates these factors and that can be used as a tier-one tool to quickly conduct a robust estimate of the potential risk of N losses to the environment.^[12]

AN N INDEX FOR RISK ASSESSMENT AND IMPROVED N MANAGEMENT

The N index is a quick assessment tool that can be used to assess the potential risk of NO_3 leaching and of surface water and atmospheric impacts (Fig. 1). The tool assesses the effects of management practices and source of N fertilizer (e.g., use of organic manure inputs) by integrating information about management, weather, and site-specific soil properties to assess the potential risk for N losses via leaching, surface, and atmospheric pathways. The tool conducts an N mass balance to assess the losses via NH_3 volatilization, NO_3 leaching, denitrification, surface runoff, and N_2O emissions. The tool also estimates crop N uptake and residual soil NO_3 . Several peer-reviewed manuscripts have shown that the N index tool can be used to assess the effects of management practices on N losses for different cropping systems in the United States, Mexico, South America (Bolivia and Ecuador), Africa (Malawi), and a Mediterranean region of Spain.^[18]

Shaffer and Delgado reported on a tiered approach for the assessment of N management practices, describing tiers-two and three as having a data-intensive approach, requiring more time to set up the needed information, and being used in coordination with field studies.^[12] In contrast, a tier-one tool is easy to use, is faster, and requires minimal information. The tier-zero tool is even easier to use than a tier-one tool, with just a few screens and minimal information required, and which provides a robust analysis of N management practices and a qualitative and quantitative assessment of the potential risk of N losses (Fig. 1).

The N index is easy to use, and it takes into consideration the effects of management of fertilizer, manure, crops, irrigation, soils, climate, and off-site factors. The N index also integrates the use of best management practices being applied at the site when assessing the potential risk of N losses to the environment. The N index accounts for N management practices and estimates the crop N uptake, N dynamics, and N sources. It also accounts for crop rotations

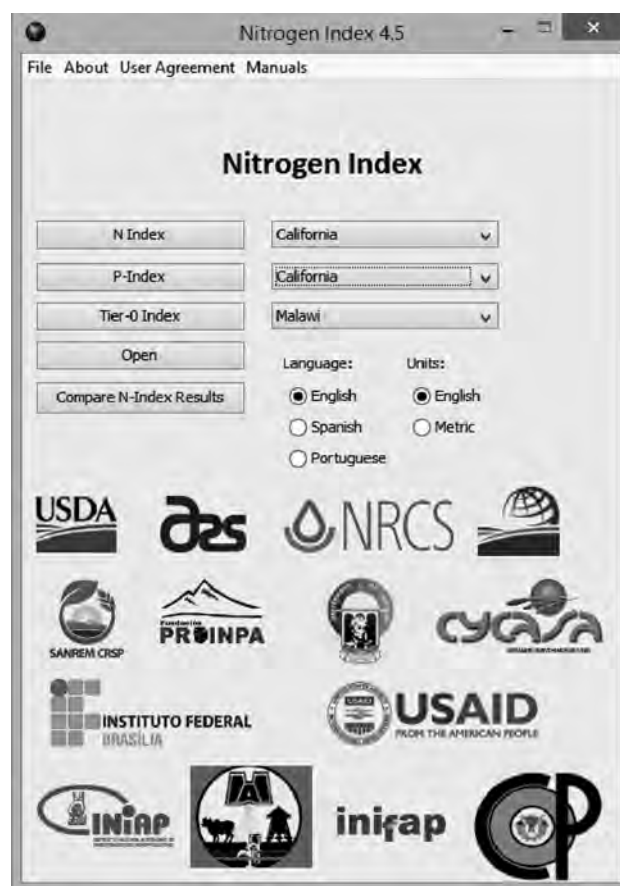


Fig. 1 Version 4.5 of the Nitrogen Index, a national and international tool.

and rooting depth. It integrates soil hydrology properties and water inputs as well as estimated water leaching. It accounts for precipitation during the growing season and nongrowing season and estimates $\text{NO}_3\text{-N}$ leaching losses, direct and indirect N_2O emissions, surface N losses, denitrification, NH_3 volatilization, crop N uptake, and residual soil $\text{NO}_3\text{-N}$. The management can be entered for a given crop at a site-specific location, and the tool will estimate the potential risk for N losses as well as the cropping system N use efficiency. The tool integrates soil management principles, soil chemical and physical principles, hydrological principles, and ecological engineering principles to assess the potential risk of N losses and help the user make management decisions (Fig. 2).

The N index can be joined to other nutrient indices and be used to help users assess practices with multiple indexes in one screen. The Kentucky N and phosphorous (P) index is one such example (Fig. 3).^[19] The N index can be used as an important tool for conservation management by integrating this approach into nutrient management plans.^[20] The tool is being used in California and is the official tool in the Natural Resources Conservation Service (NRCS) 590 nutrient management standards for Kentucky and Wisconsin. It has been transferred to NRCS—South

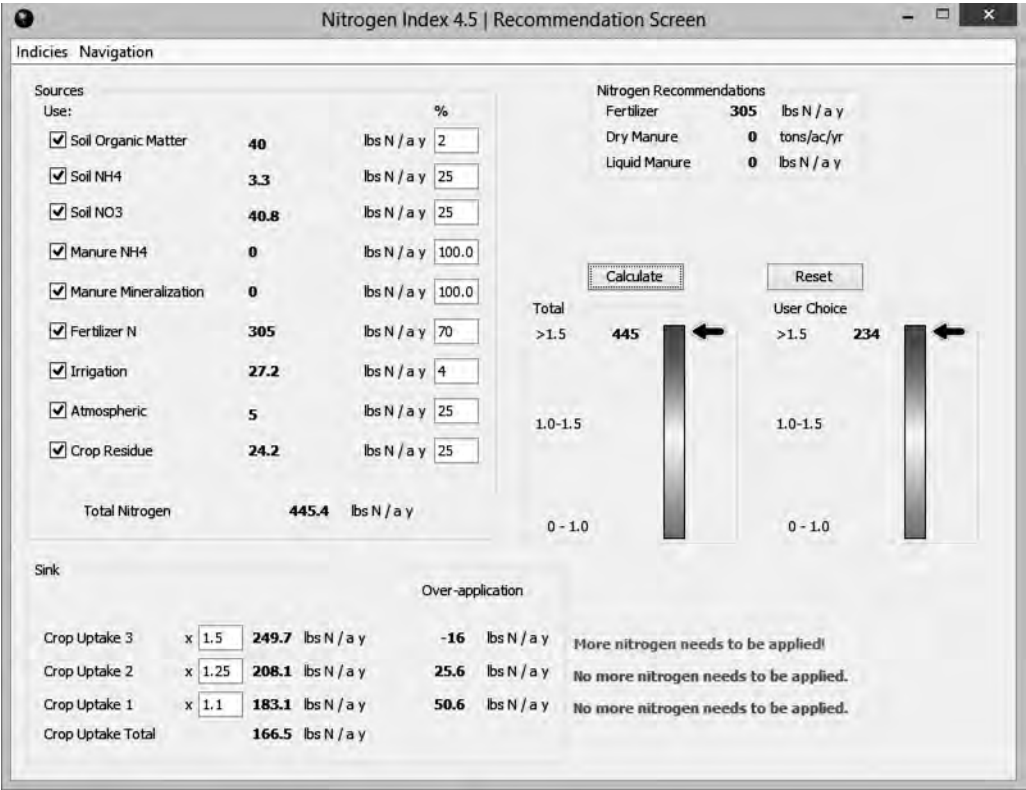


Fig. 2 Recommendation Screen of the Nitrogen index (version 4.5), a national and international tool.

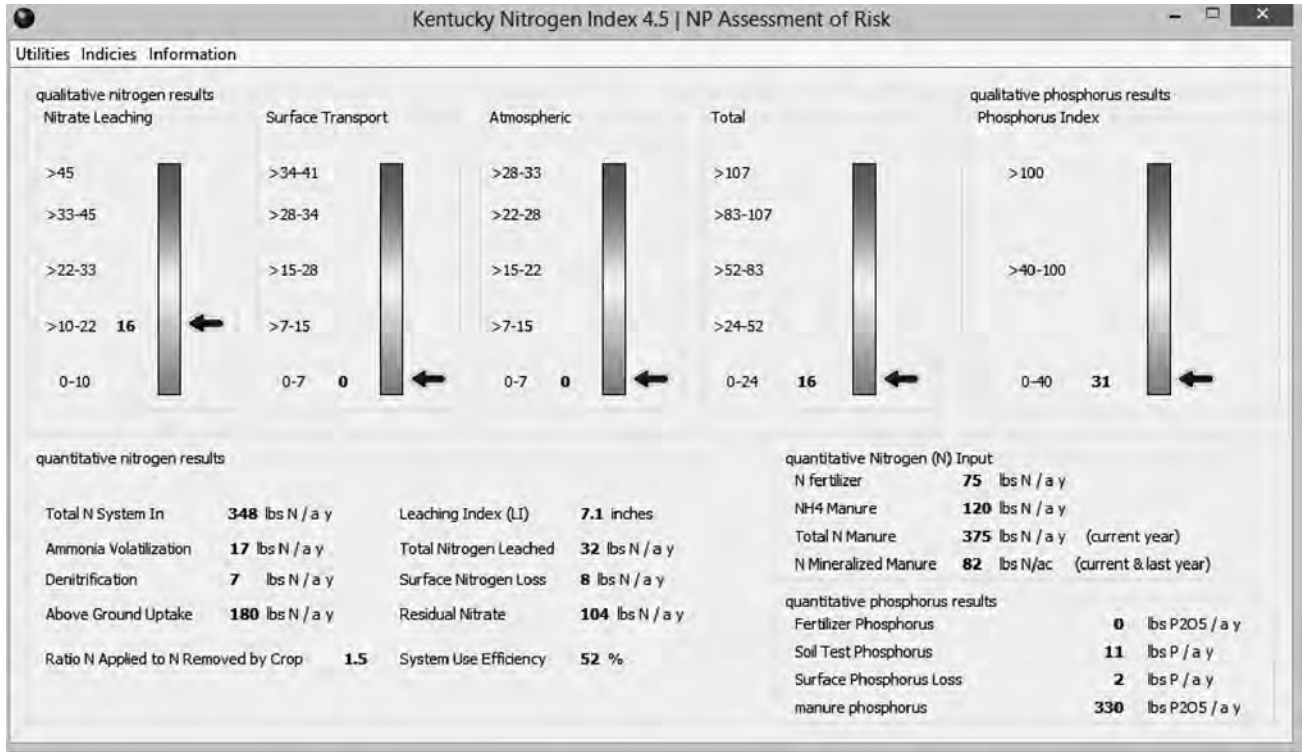


Fig. 3 Assessment of Risk Screen of the Kentucky Nitrogen and Phosphorus Index.

Dakota and to several international institutions such as Instituto Nacional de Investigaciones Forestales, Agrícolas y Pecuarias (INIFAP; in Mexico), Colegio de Estudios de Posgrado (COLPOS; in Mexico), Fundación para la Promoción e Investigación de Productos Andinos (PROINPA; in Bolivia), Instituto Nacional de Investigaciones Agropecuarias (INIAP; in Ecuador), University of Brasilia (in Brazil), and the Ministry of Agriculture and Food Security (in Malawi). The tool can be used in English, Spanish, and Portuguese. The tool can be used in metric or English units.

AN N INDEX FOR FERTILIZER RECOMMENDATIONS

Users of the N index can quickly enter the information on soil, weather, management, hydrology, weather, and expected yields. The user could select the source of inorganic N that is being applied (e.g., urea) and method (e.g., incorporated) from drop-down menus and enter the amount applied. Similarly, the user can enter the type of manure added to the field (e.g., liquid or dry), the method (e.g., incorporated), and the amount. Once the user has entered all the information into the tool, which just takes a few minutes, it is possible to proceed to the output screen. The user could select the recommendations screen to try to assess whether, when accounting for all the N sources, there is potential to apply less N while providing enough N for the crop uptake demand. The user could then go back to the N fertilizer and manure screens and change the N inputs. The user could then go back to the outputs screen to determine whether the changes in management provide enough N to the crop while reducing the N losses to the environment.

A QUICK N INDEX TOOL FOR 590 NUTRIENT MANAGEMENT PLANS

The N index has been adopted by NRCS as a quick assessment tool. The tool has been transferred to NRCS—California, NRCS—South Dakota, NRCS—Kentucky, and NRCS—Wisconsin. The tool has been adopted as the official tool for the 590 nutrient management plans in Kentucky^[19] and Wisconsin. The Kentucky N index also includes the assessment of potential risk of P losses to the environment. The tool can be used to help users assess the risk of reactive N losses to the environment and to make management decisions that can reduce the losses of reactive N. The N index tool has the potential to be used for other 590 nutrient management plans and for improving N management.

CONCLUSION

N is key for the humanity in the 21st century. N is part of our bodies, and protein inputs are important in the human diet.

However, N is also needed to maximize calorie production in the field because crops respond significantly to N inputs. To fulfill the daily caloric needs of the increasing human population during the 21st century, N will be key, especially in these challenging times when we confront big challenges such as a changing climate, depletion of water resources across different key agroecological systems, conversion of productive land to urban centers, and other great challenges. So adding N inputs to agricultural systems is the right approach for food security. However, we also need to conserve water, soil quality, and air quality, and N is also connected to environmental challenges.

There is a need to improve the management of N inputs to agricultural systems because they increase the potential for losses of reactive N to the environment, resulting in negative impacts on water and air resources. There is a need to reduce NO₃ leaching, emissions of N₂O from agricultural systems, N losses to surface water, and other losses of reactive N. There is a need to increase N use efficiencies across agroecosystems. The N index is a quick approach that can provide needed information to nutrient managers to quickly assess the effects of management practices on the potential risk of N losses via different pathways so nutrient managers could identify situations where risky combinations of management practices and landscape scenarios increase the potential for reactive losses of N (e.g., NO₃ leaching and N₂O emissions). By quickly identifying these risky management landscape scenarios, better management practices for that site could be implemented. The N index can provide important agronomical and environmental information to nutrient managers to help them make improved decisions. It can use an N balance to develop management recommendations. The U.S. Department of Agriculture—NCRS has started to apply the N index in state 590 nutrient management plans. The tool can be used in different languages and is being used in different countries and continents. The N index can be run in English or metric units. Similar to the P index that can assess the potential risk of P losses to the environment, the N index can be used to quickly assess the risk of N losses to the environment to improve N management, reduce losses of reactive N, increase N use efficiencies, and improve economic returns for farmers.

REFERENCES

1. United Nations Department of Economic and Social Affairs. *World Population Projected to Reach 9.6 Billion by 2050* 2013. United Nations Department of Economic and Social Affairs. Available at <https://www.un.org/en/development/desa/news/population/un-report-world-population-projected-to-reach-9-6-billion-by-2050.html> (accessed April 19, 2016).
2. Linehan, V.; Thorpe, S.; Andrews, N.; Kim, Y.; Beaini, F. *Food Demand to 2050: Opportunities for Australian Agriculture*; Conference paper 12.4; 42nd Australian Bureau of

- Agricultural and Resource Economics and Sciences (ABARES) Outlook Conference, Canberra, Mar 6–7. CC BY 3.0; Commonwealth of Australia, 2012.
3. Food and Agriculture Organization of the United Nations. How to feed the world in 2050. In *Paper prepared for How to Feed the World in 2050: High-Level Expert forum*, Rome, Oct 12–13, 2009; Food and Agriculture Organization of the United Nations, 2009. Available at www.fao.org/fileadmin/templates/wsfs/docs/expert_paper/How_to_Feed_the_World_in_2050.pdf (accessed April 16, 2016).
 4. Bakker, M.M.; Govers, G.; Rounsevell, M.D.A. The crop productivity-erosion relationship: An analysis based on experimental work. *Catena*. **2004**, *57*, 55–76.
 5. Galloway, J.N.; Aber, J.D.; Erismann, J.W.; Seitzinger, S.P.; Howarth, R.W.; Cowling, E.B.; Cosby, B.J. The nitrogen cascade. *BioScience*. **2003**, *53* (4), 341–356.
 6. USEPA (U.S. Environmental Protection Agency). *Federal Register*. 54 FR 22062, 22 May; USEPA: Washington, DC, 1989.
 7. Ribaud, M.; Delgado, J.; Hansen, L.; Livingston, M.; Mosheim, R.; Williamson, J. *Nitrogen in Agricultural Systems: Implications for Conservation Policy*; ERS Economic Research Report: Washington, DC, 2010.
 8. Davis, J.G.; Borch, T.; Marcillac, N.M. Agriculture's contribution to nitrogen deposition in Rocky Mountain National Park. In *Proceedings of the Great Plains Soil Fertility Conference, Volume 12*; Schlegel, A.J., Ed.; Denver, CO, Mar 4–5, 2008; International Plant Nutrition Institute: Peachtree Corners, GA; 2008; 45–53.
 9. USEPA. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2008*; USEPA: Washington, DC, 2010. Available at https://www3.epa.gov/climatechange/Downloads/ghgemissions/508_Complete_GHG_1990_2008.pdf (accessed May 25, 2016).
 10. Bronson, K.F.; Mosier, A.R. Nitrous oxide emissions and methane consumption in wheat and corn-cropped systems in northeastern Colorado. In *Agricultural Ecosystems Effects on Trace Gases and Global Climate Change*; Harper, L.A., Mosier, A.R., Duxbury, J.M., Rolston, D.E., Eds.; ASA Special Publication. 55; ASA: Madison, 1993; 133–144.
 11. Delgado, J.A.; Mosier, A.R. Mitigation alternatives to decrease nitrous oxides emissions and urea-nitrogen loss and their effect on methane flux. *J. Environ. Qual.* **1996**, *25*, 1105–1111.
 12. Shaffer, M.J.; Delgado, J.A. Essentials of a national nitrate leaching index assessment tool. *J. Soil Water Conserv.* **2002**, *57*, 327–335.
 13. Heathwaite, L.; Sharpley, A.; Gburek, W. A conceptual approach for integrating phosphorous and nitrogen management at watershed scales. *J. Environ. Qual.* **2000**, *29*, 158–166.
 14. Buczek, U.; Kuchenbuch, R.O.; Lennartz, B. Assessment of the predictive quality of simple indicator approaches for nitrate leaching from agricultural fields. *J. Environ. Manage.* **2010**, *91*, 1305–1315.
 15. Williams, J.R.; Kissel, D.E. Water percolation: An indicator of nitrogen-leaching potential. In *Managing Nitrogen for Groundwater Quality and Farm Profitability*; Follett, R.F., Keeney, D.R., Cruse, R.M., Eds.; Soil Science Society of America: Madison, 1991; 59–83.
 16. Van Es, H.M.; Czymmek, K.J.; Ketterings, Q.M. Management effects on nitrogen leaching and guidelines for a nitrogen leaching index in New York. *J. Soil Water Conserv.* **2002**, *57*, 499–504.
 17. Shaffer, M.J.; Halvorson, A.D.; Pierce, F.J. Nitrate leaching and economic analysis package (NLEAP): Model description and application. In *Managing Nitrogen for Groundwater Quality and Farm Profitability*; Keeney, D.R., Follett, R.F., Eds.; SSSA: Madison, 1991; 285–322.
 18. Delgado, J.A.; Hu, M.; Lavado, C.; Cueto-Wong, R.; Joosse, J.; Sotomayor, P.; Colon, D.; Follett, W.; Del Grosso, R.; Li, S.; Rimski-Korsakov, X.; H. An index approach to assess nitrogen losses to the environment. *Ecol. Eng.* **2008**, *32*, 108–120.
 19. Bolster, C.H.; Horvath, T.; Lee, B.D.; Mehlhope, S.; Higgins, S.; Delgado, J.A. Development and testing of a new phosphorus index for Kentucky. *J. Soil Water Conserv.* **2014**, *69*, 183–196.
 20. Gross, C.M.; Delgado, J.A.; Shaffer, M.J.; Gasseling, D.; Bunch, T.; Fry, R.A. tiered approach to nitrogen management: A USDA perspective. In *Advances in Nitrogen Management for Water Quality*; Delgado, J.A., Follett, R.F., Eds.; Soil and Water Conservation Society: Ankeny, 2010; 410–424.

Nitrogen-15: Utilization

Jorge Benitez-Vega

Luis López-Bellido

Department of Agricultural and Forestry Science and Resources, University of Cordoba, Cordoba, Spain

Ramón Redondo

Autonomous University of Madrid, Madrid, Spain

Rafael J. López-Bellido

Department of Agroforestry, University of Huelva, Huelva, Spain

Abstract

Nitrogen-15 (^{15}N) stable isotope has been and is a very useful tool to know the complexity of N cycling in soil. An ^{15}N isotope can be used to study N fertilizer efficiency, mineralization, nitrification, immobilization, leaching, etc. It helps to reduce the negative environmental impact of N fertilizer. The main limitation of ^{15}N is the high cost of the equipment for analysis and enriched fertilizer.

INTRODUCTION

Research in the past has supplied abundant information regarding many factors and processes that influence nitrogen (N) availability in soils. Many of these advances have resulted from the studies performed with the enriched nitrogen-15 (^{15}N) stable isotope. This isotope has been indispensable in knowing the rate of N in soil-plant systems. Two types of studies are performed with ^{15}N , namely studies of natural abundance levels and studies of enriched levels (^{15}N -labeled). In both these types of studies, the isotope content is measured with a mass spectrometer. The main uses of ^{15}N have been to determine the efficiency in the use of N fertilizer, the biological fixation of atmospheric nitrogen gas (N_2) by legumes, N balance, N transformation in soils, N availability from organic materials, the estimation of N rhizodeposition and root biomass, and studies on animal nutrition. The main limitation for the use of ^{15}N is the high cost of the equipment for analysis as well as the enriched fertilizers.

THE ^{15}N ISOTOPE

The most common N isotope has an atomic weight of 14. There is a heavier isotope, ^{15}N , which has one neutron more than the most abundant form (^{14}N). The ^{15}N concentration in air is very stable, reaching 0.3663% ($\pm 0.0004\%$) of the N in the atmosphere, with the rest as (99.6%) ^{14}N . This concentration is known as natural abundance. All fertilizers

with ^{15}N concentrations found outside the range of natural abundance are referred to as enriched.

The ^{15}N enrichment of any sample is expressed as the percentage of ^{15}N atoms in excess and is determined according to the following equation:

$$\% \text{ atom } ^{15}\text{N excess} = \% \text{ atom } ^{15}\text{N}_{\text{sample}} - \% \text{ atom } ^{15}\text{N}_{\text{air}} \quad (1)$$

where the % of ^{15}N atoms in the air is the natural abundance.

The most common method used to apply ^{15}N is through the use of enriched N fertilizer, presented in uric, nitric, or ammonial form or a combination of these, with enrichments that range between 2.5% and 99% ^{15}N atoms, being able to distinguish between fertilizers with only a fraction of enriched N or where all the N atoms present are ^{15}N .

N FERTILIZER RATE

The ^{15}N technology has allowed the N processes in soil to be examined, making it possible to identify the labeled forms of N and how these forms enter the system, transform, and even leave the system studied with adequate accuracy and precision.^[1] Nevertheless, there are limitations due to the internal biological recycling, which occurs in the soil, making the interpretation of ^{15}N data a meticulous work.^[2]

This technique is normally applied in mass balance studies in which the following premises are assumed:^[3]

1. The labeled and unlabeled materials have the same chemical behavior.
2. The pool that constitutes the N source is considered uniformly labeled by the isotope.
3. The addition of the isotope does not stimulate the transformation rate.

Once these premises are satisfied, by labeling the soil with $^{15}\text{NH}_4^+$ or $^{15}\text{NO}_3^-$, it is possible to determine the following:^[2,3]

1. mineralization rates, through the decrease in ^{15}N enrichment of $^{15}\text{NH}_4^+$;
2. nitrification, by studying the decrease in ^{15}N enrichment of NO_3^- ;
3. consumption/removal or immobilization of NH_4^+ or NO_3^- , through the disappearance of labeled ^{15}N ;
4. leaching and volatilization; and
5. the processes that exploit the ^{15}N and ^{14}N in all of the available pools, proportionally to its relative amounts.

To estimate the efficiency of N fertilizer using ^{15}N , a labeled fertilizer is added to the soil, and the amount of N fertilizer absorbed by the plant is determined. Different practices can be studied in this way, such as the type of fertilizer, time of application, location, and rate. In studies with ^{15}N -enriched fertilizers, the percentage that the plant nitrogen derived from the fertilizer (Ndff) is calculated on the following basis:

$$\text{Ndff}(\%) = \frac{\% \text{ } ^{15}\text{N excess}_{\text{sample}}}{\% \text{ } ^{15}\text{N excess}_{\text{fertilizer}}} \times 100 \quad (2)$$

The fertilizer nitrogen utilization efficiency (FNUE) is calculated according to the following equation:

$$\text{FNUE}(\%) = \frac{\text{N uptake} \times \text{Ndff}}{\text{Rate of N applied}} \times 100 \quad (3)$$

DETERMINATION OF BIOLOGICAL N_2 FIXATION

The methods that use ^{15}N have been highly used to understand the biochemistry of N_2 fixation by microorganisms. The study of the factors that influence the symbiotic fixation of N_2 also includes the evaluation of the fixation capacity of each microorganism, in addition to the quantification of the translocation of fixed N through the system (soil or plant).

The ^{15}N abundance in the environment is determined through the analysis of a reference plant (non-N-fixing plant) that is completely dependent on the N present in the soil for growth. The N_2 fixation can be determined by measuring the changes produced in the enrichment of ^{15}N in the fixing plant as a consequence of the assimilation of atmospheric N_2 . Various methods have been developed to determine N_2 fixation using ^{15}N . The most highly used methods are discussed in the following subsections.

^{15}N Dilution Method

The ^{15}N dilution method allows each of the different N sources of the plant to be evaluated. Before sowing, the application of a small amount of ^{15}N -enriched fertilizer is required, ranging from 1 kg/ha to 10 kg/ha with 1–10% N atoms in excess, with the objective of reducing any possible interference with the fixing capacity of the legume. The proportion of N of the plant from symbiotic fixation and from the soil can be calculated if the isotopic abundance between these two sources is different. This difference is obtained by comparing the ^{15}N enrichment of the nodulated legume with that of the plant used as a reference (non-fixing). The following equation is used to determine the plant nitrogen derived from the air (Ndfa):

$$\text{Ndfa}(\%) = \frac{1 - \% \text{ atom } ^{15}\text{N excess}_{\text{legume}}}{\% \text{ atom } ^{15}\text{N excess}_{\text{reference}}} \times 100 \quad (4)$$

The ^{15}N enrichment of the N derived from the soil present in the legume is determined from the ^{15}N enrichment of the reference plant, which should have grown in the same soil and at the same time as the legume. The use of this method implies the assumption of the following premises:

1. The $^{14}\text{N}/^{15}\text{N}$ ratio in the reference plant is the same as N ratio of the soil.
2. Both the legume and the reference plant explore the same volume of soil with an identical $^{14}\text{N}/^{15}\text{N}$ ratio.
3. Both plants should absorb the same relative amount of ^{15}N .
4. The dynamic of N absorption is same for both plants, as this is the most frequent cause of error when the fixed N_2 is determined using this method.

^{15}N Natural Abundance Method

This method is a variation of the isotopic dilution technique, based on the small differences in ^{15}N natural abundance between atmospheric N_2 and soil N. These differences are expressed as delta (δ) ^{15}N , which indicates the amount of ^{15}N with respect to the ^{14}N that exists in a sample, expressed in parts per thousand (‰). Elevated δ values indicate large amounts of ^{15}N . By definition, the standard considered (0.3663) has $\delta = 0\text{‰}$. Positive δ values indicate that the sample contains more ^{15}N than the standard, whereas the opposite is true for negative δ values. The calculation used is given as follows:

$$\delta^{15}\text{N}(\text{‰}) = \frac{\% \text{ atom } ^{15}\text{N}_{\text{sample}} - \% \text{ atom } ^{15}\text{N}_{\text{standard}}}{\% \text{ atom } ^{15}\text{N}_{\text{standard}}} \times 1000 \quad (5)$$

Soils are more enriched in ^{15}N (with a $\delta^{15}\text{N}$ of up to 16‰) than the atmosphere ($\delta^{15}\text{N} = 0\text{‰}$). This enrichment is due to the processes of isotopic discrimination and

denitrification, as well as other N transformations in the soil. The percentage of atmospheric N₂ fixed by the legume or Ndfa is calculated using the following equation:^[4]

$$\text{Ndfa}(\%) = \frac{\delta^{15}\text{N}_{\text{reference}} - \delta^{15}\text{N}_{\text{legume}}}{\delta^{15}\text{N}_{\text{reference}} - \text{B}} \times 100 \quad (6)$$

with B factor as the $\delta^{15}\text{N}$ of the legume that grows with atmospheric N₂ as it is the only source of N. Theoretically, the B value is 0‰ if it is assumed that no isotopic fractionation occurs. Considering that the soil N is generally more abundant in ^{15}N than in the atmospheric N₂, the reference plant is expected to have higher δ values than the legume, which obtains N from both the soil and the air. The $\delta^{15}\text{N}$ values in the plants used as reference present a high variability, where the $\delta^{15}\text{N}$ values can fluctuate by several units in plants separated by several centimeters.^[5,6]

The most significant advantages of this methodology with respect to the dilution technique can be summarized as follows:

1. It does not require the addition of ^{15}N .
2. The soil $\delta^{15}\text{N}$ values are relatively constant with depth and time.
3. The reference plant selection is not as important, taking precautions with the proximity of the fixing plants.

The following limitations of this methodology are worth mentioning:^[7]

1. It requires great meticulousness and precision in the analytic processes.
2. The B factor must be determined for each legume.
3. The technique is inappropriate if the $\delta^{15}\text{N}$ values of the soil are close to those of the air. For this reason, it is recommended that the soil $\delta^{15}\text{N}$ be greater than 5‰.
4. Small variations in the $\delta^{15}\text{N}$ values correspond to substantial variations in the estimation of N₂ fixation.

ESTIMATION OF N RHIZODEPOSITION

N rhizodeposition (N derived from the root exudates as well as the decomposition of fine roots) has been estimated with different methodologies. To determine the proportion of N present in the soil derived from rhizodeposition (NdfR), the plants must be labeled with ^{15}N . Subsequently, the ^{15}N flow is studied from the aerial part of the plant to the roots and then to the soil.^[8]

Because the N rhizodepositions are susceptible to being extracted again by plants and microorganisms, absorbed in the soil particles or lost from the system, the data always offer the resulting net rhizodeposition and related processes.^[9] The high ^{15}N enrichment of the fertilizer used in this technique (greater than 98%) allows for better

detection of the isotope even in small soil fractions. Five methods have been tested for effectiveness and practicability of plant ^{15}N labeling to study the N rhizodeposition as follows:^[10]

1. Root feeding:^[11] Roots of hydroponic plants are immersed in nutrient solution containing ^{15}N for 48 hours.
2. Vacuum infiltration:^[10] Roots of hydroponic plants are enclosed in a wet tissue paper/polythene sheet to prevent desiccation and ^{15}N contamination of the roots. Plants are turned upside down, and the leaves are immersed in labeling solution. After transfer to a desiccator, the plants are evacuated to 170 mbar for 5 minutes. Thereafter, the plants are left under the atmospheric pressure in the climate chamber for a further 48 hours.
3. Surface abrasion:^[10] The surface of the representative leaves of intact plants is abraded with sandpaper. Tissue paper is soaked with a labeling solution and is attached and fixed to the treated leaves for 48 hours.
4. Stem infiltration:^[12] A cotton wick is inserted into a hole in the stem of each plant. The wick is covered with siphon plastic tubing and connected to a reservoir containing the labeling solution. The height of wick insertion in relation to the reservoir is selected to avoid the dripping of the labeling solution into the rooting substrate (nutrient solution or soil). The reservoir is refilled with the labeling solution when necessary over the following 48 hours.
5. Leaf tip feeding:^[8] Leaf tips are cutoff, and the de-tipped grass blades are bent into glass reaction vials where they are immersed in labeling solution from 48 hours to 7 days. The leaves are fixed into this position with adhesive tape.

The NdfR is determined using the following equation:^[8]

$$\text{NdfR}(\%) = \frac{\% \text{ atom } ^{15}\text{N}_{\text{excess soil}}}{\% \text{ atom } ^{15}\text{N}_{\text{excess roots}}} \times 100 \quad (7)$$

The amount of rhizodeposited N (kg/ha) is estimated as follows:

$$\text{NdfR}(\text{kg/ha}) = N_s \times \frac{\text{NdfR}(\%)}{100} \quad (8)$$

where N_s = N content in the soil or soil fraction (kg/ha). The accuracy of estimating N rhizodeposits increases with increased enrichment of the plant roots and with a decreasing soil N content.^[11] However, it is crucial to include unlabeled control treatments, where the isotopic composition of plant and soil pools is measured and used as a background for calculating the atom % ^{15}N in excess values, instead of using the natural ^{15}N abundance of atmospheric N₂ (0.3663 at % ^{15}N).

CONCLUSION

^{15}N is a crucial tool to research N dynamic in soil–plant system. The results of studies with ^{15}N have reduced environmental impact of N fertilization and have improved the knowledge about the important role of legume crops in the sustainable agriculture.

REFERENCES

1. Di, H.J.; Cameron, K.C.; McLaren, R.G. Isotopic dilution methods to determine the gross transformation rates of nitrogen, phosphorus, and sulphur in soil: A review of the theory, methodologies and limitations. *Aust. J. Agr. Res.* **2000**, *38* (1), 213–230.
2. Jarvis, S.C.; Stockdale, E.A.; Shepherd, M.A.; Powlson, D.S. Nitrogen mineralization in temperate agricultural soils: Processes and measurement. *Adv. Agron.* **1996**, *57*, 187–235.
3. Dawson, T.E.; Mambelli, S.; Plamboeck, A.H.; Templer, P.H.; Tu, K.P. Stable isotopes in plant ecology. *Annu. Rev. Ecol. Syst.* **2002**, *33*, 507–559.
4. Unkovich, M.J.; Pate, J.S.; Sanford, P.; Armstrong, E.L. Potential precision of the $\delta^{15}\text{N}$ natural abundance method in-field estimates of nitrogen fixation by crop and pasture legumes in South-West Australia. *Aust. J. Agr. Res.* **1994**, *45* (1), 119–132.
5. Bedard-Haughn, A.; van Groenigen, J.W.; van Kessel, C. Tracing ^{15}N through landscapes: Potential uses and precautions. *J. Hydrol.* **2003**, *272* (1–4), 175–190.
6. Chalk, P.M. Dynamics of biologically fixed N in legume–cereal rotations: A review. *Aust. J. Agr. Res.* **1998**, *49* (3), 303–316.
7. Peoples, M.B.; Boddey, R.M.; Herridge, D.F. Quantification of nitrogen fixation. In *Nitrogen Fixation at the Millennium*; Leigh, G.J., Ed.; Elsevier Science: Amsterdam, 2002; 357–389.
8. Høgh-Jensen, H.; Schjoerring, J.K. Rhizodeposition of nitrogen by red clover, white clover and ryegrass leys. *Soil Biol. Biochem.* **2001**, *33* (4–5), 439–448.
9. Schmidtke, K. A model to predict the accuracy of measurements of legume N rhizodeposition using a split-root technique. *Soil Biol. Biochem.* **2005**, *37* (5), 829–836.
10. Hertenberger, G.; Wanek, W. Evaluation of methods to measure differential ^{15}N labeling of soil and root N pools for studies of root exudation. *Rapid Commun. Mass Sp.* **2004**, *18* (20), 2415–2425.
11. Schmidtke, K. How to calculate nitrogen rhizodeposition: A case study in estimating N rhizodeposition in the pea (*Pisum sativum* L.) and grass pea (*Lathyrus sativus* L.) using a continuous ^{15}N labelling split-root technique. *Soil Biol. Biochem.* **2005**, *37* (10), 1893–1897.
12. Russell, C.A.; Fillery, I.R.P. *In situ* ^{15}N labelling of lupin below-ground biomass. *Aust. J. Agr. Res.* **1996**, *47*, 1035–1046.

Nitrous Oxide Emissions: Agricultural Soils

John R. Freney

Plant Industry, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia

Abstract

Nitrous oxide emissions from agricultural soils are generally greater and more variable than those from uncultivated land. All soils are deficient in nitrogen (N) for the growth of plants, but the deficiency can be overcome by adding fertilizer N. The low efficiency of fertilizer N in agricultural systems is primarily caused by the large losses of mineral N from those systems by gaseous loss: nitrous oxide emission is directly linked to the loss processes.

INTRODUCTION

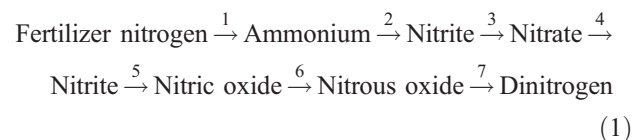
Nitrous oxide is a gas that is produced naturally by many different microorganisms in soils and waters, and as a result of human activity associated with agriculture, biomass burning, stationary combustion, automobiles, and the production of nitric and adipic acids for industrial purposes. According to the Intergovernmental Panel on Climate Change (IPCC),^[1] ~23.1 million metric tons (Mt) of nitrous oxide is emitted each year, 14.1 Mt as a result of natural processes (~4.7 Mt from the oceans, ~6.3 Mt from tropical soils, and ~3.1 Mt from temperate soils) and ~9Mt as a result of human activities (5.5 Mt from agricultural soils, 0.6 Mt from cattle and feedlots, 0.8Mt from biomass burning, and 2.1 Mt from mobile sources and industry). While there is considerable uncertainty associated with each of these estimates, it is apparent that most nitrous oxide is derived from soils.

Because of the intimate connection between the earth and the atmosphere, much of the nitrous oxide produced enters the atmosphere and affects its chemical and physical properties. Nitrous oxide contributes to the destruction of the stratospheric ozone layer that protects the earth from harmful ultraviolet radiation and is one of the more potent greenhouse gases that trap part of the thermal radiation from the earth's surface. The atmospheric concentration of nitrous oxide is ~313 parts per billion. It is increasing at the rate of ~0.7 parts per billion each year, and its lifetime is ~166 years.^[2] It seems that the increased atmospheric concentration results from the increased use of synthetic fertilizer nitrogen (N), biologically fixed N, animal manure, crop residues, and human sewage sludge in agriculture to produce food and fiber for the rapidly increasing world population.^[3]

NITROUS OXIDE EMISSION FROM AGRICULTURE

All soils are deficient in N for the growth of plants, but the deficiency can be overcome by adding fertilizer N. When

the fertilizer (e.g., urea or ammonia-based compounds) is applied to soil, it is transformed by microorganisms as follows



When the soil is aerobic (i.e., when oxygen is present), ammonium is oxidized to nitrite and nitrate (Steps 2 and 3). This process is called nitrification. After the addition of irrigation water or rain, the soil may become anaerobic (devoid of oxygen). The nitrate is then reduced by soil organisms to nitrite and the gases, namely nitric oxide, nitrous oxide, and dinitrogen (Steps 4–7) in a process termed denitrification.^[4]

When atmospheric scientists first expressed concern that nitrous oxide emission into the atmosphere, as a result of fertilizer use, would lead to destruction of the ozone layer, it was thought that nitrous oxide was produced mainly from the microbiological reduction of nitrate in poorly aerated soils. However, research in the latter part of the 1970s showed that significant nitrous oxide was emitted from aerobic soils during the nitrification of ammonium, and subsequent work has shown that nitrification is a major source of nitrous oxide.^[4]

Nitrous Oxide from Denitrification

Certain microorganisms in the absence of oxygen have the capacity to reduce nitrate (or other N oxides). Most denitrifying bacteria are heterotrophs—i.e., they require a source of organic matter for energy—but denitrifying organisms that obtain their energy from light or inorganic compounds also occur in soils. The capacity to denitrify has been reported in more than 20 genera of bacteria, and almost all are aerobic organisms that can

only grow anaerobically in the presence of N oxides. The dominant denitrifying organisms in soil are *Pseudomonas* and *Alcaligenes*. In addition to the free-living denitrifiers, rhizobia living symbiotically in root nodules of legumes are able to denitrify nitrate and produce nitrous oxide.^[4]

The general requirements for biological denitrification include the presence of microorganisms with denitrifying capacity, nitrate (or other N oxides) and available organic matter, the absence of oxygen, and a suitable pH and temperature environment. In aerobic soils, denitrification can occur in anaerobic microsites in soil aggregates or in areas of high carbon content, where active microbial activity rapidly consumes all of the available oxygen.^[4]

Nitrous Oxide from Nitrification

The process of nitrification is normally defined as the biological oxidation of ammonium to nitrate with nitrite as an intermediate.^[4] The first step in the reaction, the oxidation of ammonium to nitrite, is carried out mainly by the microorganism *Nitrosomonas*. The second step, the oxidation of nitrite to nitrate, is carried out by *Nitrobacter*. It has been shown in a number of publications that *Nitrosomonas europaea* produces nitrous oxide during the oxidation of ammonium.^[4]

The possibility that significant nitrous oxide can be produced in soils by nitrifying organisms was indicated by studies that showed that soils incubated under aerobic conditions with ammonium produced more nitrous oxide than soils amended with nitrate.^[4] In addition, treatment of aerobic soils with nitrapyrin, which inhibits the nitrification of ammonium but has little effect on denitrification, markedly reduced the emission of nitrous oxide.^[4] The production of nitrous oxide by nitrification in soils is increased by increasing temperature, pH, water content, available carbon, and the addition of ammonium-based fertilizers, plant residues, and animal manure.

Flooded Soils

In the past few years, increased attention has been given to nitrous oxide emission from paddy soils. The concern is that the introduction of management practices to reduce methane emissions from flooded soils may result in increased emissions of nitrous oxide.

Flooded soils are characterized by an oxygenated water column overlying an oxidized layer at the soil–water interface, an aerobic zone around each root, and anaerobic conditions in the remainder of the soil. This differentiation of the flooded soil into oxidized and reduced zones has a marked effect on the transformation of N.^[5] The resulting reactions are as follows:

1. Ammonium in the reduced zone diffuses to the oxidized zone;
2. Ammonium is oxidized to nitrate by nitrifying organisms;
3. The nitrate formed diffuses to the anaerobic zone;
4. Denitrification occurs with the production of nitrous oxide and dinitrogen; and
5. The gaseous products diffuse through the soil and water layers to the atmosphere.^[6]

It is apparent that the rate of diffusion of ammonium to an oxidized layer and the rate of nitrification in the oxidized layer are factors controlling the production of nitrous oxide in flooded soils. The rate of diffusion of nitrous oxide through the soil and water layers will control its rate of emission to the atmosphere, or its further reduction to dinitrogen.^[5]

A number of mechanisms have been identified for the transfer of nitrous oxide from the soil to the atmosphere.^[3] Nitrous oxide may diffuse from the zone of production through the saturated soil and water layer to the atmosphere. It may also enter the roots of the rice plant and move by diffusion through the plant to the atmosphere in the same way as methane. Bhadrachalam et al.^[6] studied the importance of the two pathways in intermittently flooded rice in the field in India and found that N gas fluxes were ~30% greater when transfer through the plants was included.

In the tropics, rice is usually transplanted and fertilized sometime after flooding. Because of the anaerobic conditions that develop before fertilization and the slow rate of diffusion of nitrous oxide in flooded soils, most of the nitrous oxide is reduced to dinitrogen and very little escapes to the atmosphere. Nitrous oxide emission from intermittently flooded rice was relatively large compared with that from permanently flooded rice, reflecting the different oxidation states of intermittently and continuously flooded soils.^[6] Studies of nitrous oxide emission from rice fields from the time the soils were drained for harvest, through flooding the soil in preparation for planting the next crop, showed that nitrous oxide was emitted continuously while the soil was not flooded. Overall, the rate of emission of nitrous oxide from flooded soils was less than that from upland soils after the application of N fertilizer.^[3]

Biomass Burning

During combustion, the N in the fuel can be converted into gaseous forms such as ammonia, nitric oxide, nitrous oxide, dinitrogen, and hydrogen cyanide. It is estimated that biomass burning contributes between 0.3 and 1.6 Mt nitrous oxide per year globally to the atmosphere.^[3] Most of the biomass burning (~90%) takes place in the tropics as a result of forest clearing, savanna

and sugarcane fires, and burning of agricultural wastes and firewood.^[7]

Biomass burning is not only an instantaneous source of nitrous oxide, but it also results in a longer term enhancement of the production of this gas. Measurements of nitrous oxide emissions from soils, before and after burning, showed that significantly more nitrous oxide was exhaled after the burn through alteration of the chemical, biological, and physical processes in soil.^[7]

Fertilizer Consumption and Nitrous Oxide Production

Nitrous oxide emissions from agricultural soils are generally greater and more variable than those from uncultivated land. The application of fertilizer N, animal manure, and sewage sludge usually results in enhanced emissions of nitrous oxide.^[7] Generally, there is a large emission of nitrous oxide immediately after the application of fertilizer. After about 6 weeks, the emission rate falls and fluctuates around a low value. Mosier^[8] concluded that interactions between the physical, chemical, and biological variables are complex, that nitrous oxide fluxes are variable in time and space, and that soil management, cropping systems, and variable rainfall appear to have a greater effect on nitrous oxide emission than the type of N fertilizer. Consequently, Mosier et al.^[9] recommend the use of one factor only for calculating the emission of nitrous oxide from different fertilizer types

$$\text{N}_2\text{O emitted} = 1.25\% \text{ of N applied (kg N/ha)} \quad (2)$$

This equation is based on data from long-term experiments with a variety of mineral and organic fertilizers and encompasses 90% of the direct contributions of N fertilizers to nitrous oxide emissions.

Mosier et al.^[3] developed a methodology to estimate agricultural emissions of nitrous oxide, taking into account all of the N inputs into crop production. They included direct emissions from agricultural soils as a result of synthetic fertilizer addition, animal wastes, increased biological N fixation, cultivation of mineral and organic soils through enhanced organic matter mineralization, and mineralization of crop residues returned to the field. Indirect nitrous oxide emissions resulting from the deposition of ammonia and oxides of N, leaching of nitrate and the introduction of N into sewage systems were also included. They concluded that in 1989, 9.9 Mt of nitrous oxide was emitted into the atmosphere directly or indirectly, as a result of agriculture (Table 1).

MANAGEMENT PRACTICES TO DECREASE NITROUS OXIDE EMISSION

The low efficiency of fertilizer N in agricultural systems is primarily caused by the large losses of mineral N

Table 1 Calculated emission of nitrous oxide from agricultural activities.

Mt nitrous oxide per year	
Direct soil emissions	
Synthetic fertilizer	1.4 (0.28–2.5)
Animal waste	0.9 (0.19–1.7)
Biological nitrogen fixation	0.16 (0.03–0.3)
Crop residue	0.6 (0.11–1.1)
Cultivation of Histosols	0.16 (0.03–0.3)
Total	3.3 (0.6–5.9)
Animal production	
Waste management systems	3.3 (0.9–4.9)
Indirect emissions	
Atmospheric deposition	0.47 (0.09–0.9)
Nitrogen leaching and runoff	2.5 (0.2–12.1)
Human sewage	0.3 (0.06–4.1)
Total	3.3 (0.35–17.1)
Total	9.9 (1.9–27.9)

Source: Modified from Mosier, Kroeze, et al.^[3]

from those systems by gaseous loss: nitrous oxide emission is directly linked to the loss processes. It is axiomatic that any strategy that increases the efficiency of N fertilizer use will reduce the emissions of nitrous oxide, and this has been directly demonstrated for a number of strategies.^[3]

The IPCC^[1] reported that some combination of the following management practices, if adopted worldwide, would improve the efficiency of the use of synthetic fertilizer and manure N, and significantly reduce nitrous oxide emission into the atmosphere to

1. Match N supply with crop demand.
2. Tighten N flow cycles by returning animal wastes to the field and conserving residues instead of burning them.
3. Use controlled-release fertilizers, incorporate fertilizer to reduce volatilization, use urease and nitrification inhibitors, and match fertilizer type to precipitation.
4. Optimize tillage, irrigation, and drainage.

The potential decrease in nitrous oxide emissions from synthetic fertilizer, as a result of the mitigation techniques, could amount to 20%.^[1]

REFERENCES

1. Intergovernmental Panel on Climate Change (IPCC). *Climate Change 1995. Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses*; Watson, R.T., Zinyowera, M.C., Moss, R.H., Eds.; Cambridge University Press: Cambridge, 1996; 1–878.

2. Hengeveld, H.; Edwards, P. An assessment of new research developments relevant to the science of climate change. *Climate Change Newslett.* **2000**, *12*, 1–52.
3. Mosier, A.; Kroeze, C.; Nevison, C.; Oenema, O.; Seitzinger, S.; van Cleemput, O. Closing the global N₂O budget: Nitrous oxide emissions through the agricultural nitrogen cycle. *Nutr. Cycl. Agroecosys.* **1998**, *52*, 225–248.
4. Bremner, J.M. Sources of nitrous oxide in soils. *Nutr. Cycl. Agroecosys.* **1997**, *49*, 7–16.
5. Patrick, W.H., Jr. Nitrogen transformation in submerged soils. In *Nitrogen in Agricultural Soils*; Stevenson, F.J., Ed.; American Society of Agronomy: Madison, 1982; 449–465.
6. Bhadrachalam, A.; Chakravorti, S.P.; Banerjee, N.K.; Mohanty, S.K.; Mosier, A.R. Denitrification in intermittently flooded rice fields and N-gas transport through rice plants. *Ecol. Bull.* **1992**, *42*, 183–187.
7. Granli, T.; Bøckman, O.C. Nitrous oxide from agriculture. *Nor. J. Agr. Sci.* **1994**, *12*, 128–134.
8. Mosier, A.R. Chamber and isotope techniques. In *Exchange of Trace Gas between Terrestrial Ecosystems and the Atmosphere*; Andreae, M.O., Schimel, D.S., Eds.; John Wiley & Sons: Chichester, 1989; 175–187.
9. Mosier, A.R.; Duxbury, J.M.; Freney, J.R.; Heinemeyer, O.; Minami, K. Nitrous oxide emissions from agricultural fields: Assessment, measurement and mitigation. *Plant Soil* **1995**, *181*, 95–108.

Nitrous Oxide Emissions: Paddy Soils

Jianfu Xue

Xin Zhao

Hailin Zhang

College of Agronomy and Biotechnology, China Agricultural University, Beijing, China

Abstract

Global warming is an increasing concern for scientists, policy makers, and public all over the world. Nitrous oxide (N_2O) is one of the most important anthropogenic greenhouse gases. Paddy fields are the primary sources of N_2O emissions due to mid-season and drying–wetting alternation during the growing season. Agricultural management practices (e.g., water management and fertilizers application) have great impacts on N_2O emissions because of the microenvironmental changes of paddy fields. A further understanding of the mechanisms of N_2O emissions and optimizing farming practices on paddy fields are of fundamental significance for the mitigation of climate change.

INTRODUCTION

Globally, climate change has become a significant issue and challenge to human well-being. Next to carbon dioxide and methane (CH_4), nitrous oxide (N_2O) is one of the principal anthropogenic greenhouse gases, which can result in an increased photolysis rate of stratospheric ozone due to the increase of N_2O concentration at an average rate of ~ 0.75 ppb yr^{-1} .^[1] The N_2O emissions from agriculture are the primary anthropogenic source, especially due to widespread use of synthetic and manure fertilizers. It has been reported that the amount of N_2O emissions from fertilizers, including the following subdomains with synthetic fertilizers, manure management, and manure applied to soils, are 2016.4, 680.8, 508.7, 116.4, and 40.4 Gg in Asia, the United States, Europe, Africa, and Oceania, respectively.^[2]

Paddy soils are not only the source of CH_4 emissions but also a significant source of N_2O emissions. For example, it is estimated that N_2O emission from paddy fields accounted for 22% of the total emission from croplands in China.^[3] It is critical to understand the mechanism of N_2O generation and the influencing factors of N_2O emissions from paddy soils in order to effectively mitigate climate change.

PATHWAYS OF N_2O EMISSION FROM PADDY SOILS

Soil N_2O as an intermediate product is primarily generated during the process of nitrification and denitrification. Generally, the main sources of soil N_2O are from nitrification under aerobic conditions and from denitrification under anoxic conditions.^[4] The oxidation process in nitrification

includes oxidizing ammonium (NH_4^+) to nitrite (NO_2^-) and oxidizing NO_2^- to nitrate (NO_3^-). In denitrification, oxidized inorganic forms of nitrogen (N) are sequentially reduced as follows: $\text{NO}_3^- \rightarrow \text{NO}_2^- \rightarrow \text{NO} \rightarrow \text{N}_2\text{O} \rightarrow \text{N}_2$ under anoxic/suboxic conditions. Paddy soils have a unique profile including oxidizing and reducing layers in the plow layer due to long-term flooding. Generally, N_2O is mainly generated through both nitrification in the oxidizing layer and denitrification in the reducing layer of paddy soils during rice (*Oryza sativa* L.) growing and non-growing seasons. However, less N_2O emission is released into the atmosphere through denitrification due to lower water content during rice non-growing season. Previous studies showed that 80% of N_2O emissions from paddy fields were released into the atmosphere through the rice plant under continuous flooding conditions.^[5] However, N_2O emissions from paddy soils mainly occurred during the drainage-drying periods. Compared with the flooding condition, 82.5% of N_2O emissions were released into the atmosphere through the soil surface under the drainage-drying period.^[6] In addition, additional research has been conducted concerning the influence of floodwater on the pathway of N_2O emission from rice plants.^[6] Cutting the rice stems below water surface can immediately decrease N_2O emission compared with cutting under flooded condition, while no variation is observed among uncut plants.^[6] However, there is almost an equal amount of N_2O emission after drainage flooding regardless of cutting the rice plants.

Water Regime

Water regime has a significant influence on N_2O emissions from paddy soils. Generally, less N_2O emission from paddy fields is observed under continuous flooding conditions

compared with non-flooding conditions during the rice-growing season. However, mid-season aeration and drying-wetting alternation are widely applied in rice production in order to save agricultural water and to increase rice yield, which can stimulate N_2O generation and emissions from paddy soils. The peak value of N_2O fluxes is observed during the mid-season aeration and drainage-drying period before the rice harvest. The N_2O emission during the drainage-drying phase accounts for 87–98% of the total N_2O emission in the whole rice-growing season.^[7] Li et al. also found that N_2O emissions during the mid-season aeration period and 5–8 days after reflooding can account for 70–94% of the total N_2O emission in the whole rice-growing season.^[8] But the N_2O emissions from paddy fields decrease gradually with the postponement of the mid-season aeration. Soil moisture status [e.g., water-holding capacity (WHC) and water-filled pore space (WFPS)] affects N_2O emission from paddy fields through changing soil aeration and redox potential. The largest N_2O emission in a paddy ecosystem is observed under 90–100% of WHC or 77–86% of WFPS.^[9] These peaks of N_2O emission in paddy soils are observed at different WFPS levels due to differences in soil types and cropping systems.^[9–11]

Fertilizer Application

Several studies have reported the effects of fertilizer application on N_2O emissions in paddy fields. The N_2O emission increases with increase in the application rate of N fertilizer regardless of the urea or NH_4^+ content of fertilizer during the rice-growing season,^[12] due to the supplement of NO_3^- and NH_4^+ substrate for the process of nitrification and denitrification. Differences in N_2O emissions could be observed with the application of different synthetic fertilizer types in paddy fields.^[13] The use of ammonium sulfate can lead to larger N_2O emissions than that of urea during the rice-growing season in paddy fields.^[13] The effects of diverse forms of N fertilizers on N_2O emissions from paddy fields indicate the following order, i.e., from the highest N_2O emissions to the lowest: anhydrous ammonia > NH_4^+ > calcium nitrate > ammonium nitrate > urea.^[14] In some cases, the adoption of controlled availability of fertilizer reduced N_2O emissions by 30% compared with urea during the whole rice-growing season in paddy fields,^[15] but the difference in N_2O emissions between the fertilizers was not always be observed.^[16] Such contradictory response may be due to the slow release and sustained release properties of controlled availability of fertilizer. The controlled release can hold higher NH_4^+ concentrations and promote nitrification during mid-season aeration, thereby accumulating NO_3^- , which benefits denitrification during reflooding and ultimately leads to the enhancement of N_2O emission. Fertilizer application methods also have a significant effect on N_2O emissions in paddy fields. Compared with surface broadcasting in paddy field, incorporating fertilizer into soil can reduce N_2O emissions. This trend

can be attributed to easier oxidation from NH_4^+ to NO_3^- in the surface solution and NO_3^- diffusion to deeper aerobic layer, which accelerates microbial denitrification for surface broadcasting practice.

However, there are more complex mechanisms by which organic fertilizers influence N_2O emissions, compared with synthetic fertilizers. In addition to acting as an N source, organic fertilizers provide useful nutrition supplements to the soil [e.g., organic carbon (C)]. Due to nitrification and denitrification processes, the higher C/N ratio of organic fertilizers used in paddy fields can assimilate and consume soil N, resulting in an increase in N_2O emissions. Application of organic fertilizer could change soil structure and microenvironment, thus affecting N_2O emissions as well. Similar to synthetic fertilizer, the types, amounts, and application methods of organic fertilizer have an important effect on N_2O emissions in paddy field. Combined application of organic and synthetic fertilizers is a common trend in paddy fields, but few studies have examined the interactive effect of these practices on N_2O emissions.

Others Factors

Besides water management and fertilizer application, other farming practices also affect N_2O emissions from paddy fields. Tillage and rice residue retention also have great influences on N_2O emissions in paddy fields due to changes in the soil properties (e.g., soil moisture, soil porosity, and redox potential). Generally, rice residues retention can enhance N_2O emissions. When rice residues are retained, no-till (NT) practices reduce N_2O emissions compared with plow tillage and rotary tillage during the double paddy growing season, which may be due to the increase of saturated hydraulic conductivity causing increased NH_4^+ and NO_3^- leaching under NT.^[17] There are significant differences in water and N fertilizer management under different cropping systems in the paddy ecosystem, which can lead to large variations among them. The N_2O emissions from the rice-flooding fallow cropping system are often lower than those from the rice-wheat and the double rice-wheat cropping systems due to a lower quantity of N fertilizer application and a longer flooding period. Soil texture also has a significant effect on average N_2O fluxes in paddy fields. In general, N_2O emissions from sandy soil are significantly higher than those from loam and clay soils.^[5] Such texture-induced variations in N_2O flux may be due to faster diffusion of N_2O emissions, weaker buffer action of redox potential, and lower storage capacity of organic matter in sandy soil.

MITIGATION AND CHALLENGE ON N_2O EMISSION FROM PADDY SOILS

Water management and N fertilizer application are the main factors influencing N_2O emissions from paddy soils.

Enhancement of the flooding duration can reduce N₂O emissions from paddy soils. There is a trade-off between CH₄ and N₂O emissions from paddy fields. Increases in flooding time can lead to higher CH₄ emissions, even though N₂O emissions are reduced in the paddy ecosystem. However, CH₄ emissions have a more predominant contribution to integrated and cumulative global warming potential compared with N₂O emission. Suitable late aeration may reduce N₂O emission from paddy soils, but information on its mechanism of action is scarce. Thus, it is necessary to explore appropriate water regimes to mitigate both N₂O and CH₄ emissions from paddy soils. Meanwhile, improving N fertilizer management is an easier way to mitigate N₂O emissions from paddy soils. Some cropland management options are suggested for reducing N₂O emissions from paddy fields, which include 1) changing the N fertilizer application rate; 2) choosing appropriate fertilizers types; 3) appropriate timing and precision application of fertilization; and 4) using urease and nitrification inhibitors. Adoption of other agricultural practices can reduce N₂O emissions from paddy fields through optimizing tillage practice, residues management, and cultivating rice varieties with higher N use efficiency.

REFERENCES

1. IPCC. Observations: Atmosphere and surface. In *Climate Change 2013: The Physical Science Basis, Working Group I Contribution to the IPCC Fifth Assessment Report (AR5)*; Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M., Eds.; Cambridge University Press: Cambridge and New York, 2013; 168 pp.
2. FAO. *FAOSTAT Emissions Database*, 2013. Available at <http://faostat.fao.org/> (accessed September 2014).
3. Xing, G.X. N₂O emission from cropland in China. *Nutr. Cycl. Agroecosys.* **1998**, *52*, 249–254.
4. Ussiri, D.; Lal, R. Formation and release of nitrous oxide from terrestrial and aquatic ecosystems. In *Soil Emission of Nitrous Oxide and its Mitigation*; Ussiri, D., Lal, R., Eds.; Springer: Dordrecht, 2013; 63–96.
5. Yu, K.; Wang, Z.; Chen, G. Nitrous oxide and methane transport through rice plants. *Biol. Fert. Soils.* **1997**, *24*, 341–343.
6. Yan, X.; Shi, S.; Du, L.; Xing, G. Pathways of N₂O emission from rice paddy soil. *Soil Biol. Biochem.* **2000**, *32*, 437–440.
7. Xu, H.; Xing, G.X.; Cai, Z.C.; Tsuruta, H. Effect of soil water regime and soil texture on N₂O emission from rice paddy field. *Acta Pedologica Sin.* **2000**, *37*, 499–505.
8. Li, X.L.; Xu, H.; Cao, J.L.; Cai, Z.C.; Yagi, K. Effect of water management on N₂O emission in rice-growing season. *Soils* **2006**, *38*, 703–707.
9. Zheng, X.; Wang, M.; Wang, Y.; Shen, R.; Gong, Y.; Luo, D.; Zhang, W.; Jin, J.; Li, L. Impact of soil humidity on N₂O production and emission from a rice-wheat rotation ecosystem. *Chin. J. Appl. Ecol.* **1996**, *7*, 273–279.
10. Peng, S.Z.; Hou, H.J.; Xu, J.Z.; Mao, Z.; Abudu, S.; Luo, Y.F. Nitrous oxide emissions from paddy fields under different water managements in southeast China. *Paddy Water Environ.* **2011**, *9*, 403–411.
11. Hou, H.; Peng, S.; Xu, J.; Yang, S.; Mao, Z. Seasonal variations of CH₄ and N₂O emissions in response to water management of paddy fields located in Southeast China. *Chemosphere* **2012**, *89*, 884–892.
12. Hua, X.; Guangxi, X.; Cai, Z.C.; Tsuruta, H. Nitrous oxide emissions from three rice paddy fields in China. *Nutr. Cycl. Agroecosys.* **1997**, *49*, 23–28.
13. Cai, Z.C.; Xing, G.X.; Yan, X.Y.; Xu, H.; Tsuruta, H.; Yagi, K.; Minami, K. Methane and nitrous oxide emissions from rice paddy fields as affected by nitrogen fertilisers and water management. *Plant Soil* **1997**, *196*, 7–14.
14. Granli, T.; Bøckman, O.C. Nitrous Oxide from Agriculture. *Norw. J. Agr. Sci.* **1994**, *12*, 7–128.
15. Li, F.; Fan, X.; Liu, F.; Wang, Q. Effects of controlled release fertilizers on N₂O emission from paddy field. *Chin. J. Appl. Ecol.* **2004**, *15*, 2170–2174.
16. Yan, X.; Du, L.; Shi, S.; Xing, G. Nitrous oxide emission from wetland rice soil as affected by the application of controlled-availability fertilizers and mid-season aeration. *Biol. Fert. Soils* **2000**, *32*, 60–66.
17. Cui, S.Y.; Xue, J.F.; Chen, F.; Tang, W.G.; Zhang, H.L.; Lal, R. Tillage effects on nitrogen leaching and nitrous oxide emission from double-cropped paddy fields. *Agron. J.* **2014**, *106*, 15–23.

Nitrous Oxide Emissions: Sources, Sinks, and Strategies

Katsu Minami

*Environmental Resources Division, National Institute of Agro-Environmental Sciences,
Tsukuba, Japan*

Abstract

Although it has been known for more than 50 years that nitrous oxide (N_2O) is a regular constituent of the atmosphere, it was not considered to be of any importance as an air constituent until the early 1970s. Atmospheric scientists hypothesized that N_2O released to the atmosphere through denitrification of nitrates in soils and natural waters may trigger reactions in the stratosphere leading to partial destruction of ozone layer protecting the earth from biologically harmful ultraviolet radiation from the sun. N_2O is also one of the natural components of earth's atmosphere and contributes to the natural greenhouse effect; therefore, the increasing of N_2O in the atmosphere may be contributing to global warming. The atmospheric concentration of N_2O has been increasing at an accelerated rate for several decades. The global warming potential of each molecule of N_2O is about 210 times (20-year horizon) greater than each molecule of carbon dioxide.

INTRODUCTION

Although it has been known for more than 50 years that nitrous oxide (N_2O) is a regular constituent of the atmosphere, it was not considered to be of any importance as an air constituent until the early 1970s. Atmospheric scientists hypothesized that N_2O released to the atmosphere through denitrification of nitrates in soils and natural waters may trigger reactions in the stratosphere leading to partial destruction of ozone layer protecting the earth from biologically harmful ultraviolet radiation from the sun.^[1] N_2O is also one of the natural components of earth's atmosphere and contributes to the natural greenhouse effect; therefore, the increasing of N_2O in the atmosphere may be contributing to global warming.^[2]

The atmospheric concentration of N_2O has been increasing at an accelerated rate for several decades at a rate of 0.8 ppbv/yr, and the lifetime of N_2O is 120 years. It has been estimated that doubling the concentration of N_2O in the atmosphere would result in a 10% decrease in the ozone layer that would increase the ultraviolet radiation reaching the earth by 20%,^[3] eventually leading to an increase in the occurrence of skin cancer and other health problems. The global warming potential (GWP) of each molecule of N_2O is about 210 times (20-year horizon) greater than each molecule of carbon dioxide. N_2O accounts for 6% of total GWP.^[4]

THE BUDGET OF ATMOSPHERIC N_2O

Atmospheric Distribution of N_2O

As a result of biotic and anthropogenic activities, the concentration of N_2O in the atmosphere is increasing at the

rate of about 0.25%/yr. The concentration of N_2O is about 0.75 ppbv higher in the northern hemisphere than in the southern hemisphere, suggesting the presence of greater source of N_2O in the northern hemisphere than in the southern hemisphere. Ice core measurements show that the preindustrial value of N_2O was relatively stable at about 285 ppbv for most of the past 2,000 years and started to increase around the year 1700 associated with anthropogenic activity.^[4,5]

Sinks for N_2O

The major atmospheric loss process for N_2O is photochemical decomposition by sunlight (wavelengths 180–230 nm) in the stratosphere and is calculated to be 12.3 (range 9–16) Tg N/yr. Tropospheric sinks such as surface loss in aquatic and soil systems are considered to be small as compared to atmospheric sink.^[5] However, the paucity of data does not enable us to estimate the importance of this sink on a global basis.

Sources of N_2O

There are many sources of N_2O , both natural and anthropogenic, which cannot be easily quantified. Undoubtedly, the earth and oceans are significant N_2O sources. N_2O fluxes from upwelling regions of the Indian and Pacific Oceans clearly suggest that oceans may be a larger source. The ocean flux estimate is 3 (range 1–5) Tg N/yr. Tropical forest soils are probably the single most important source of N_2O emission to the atmosphere. The total N_2O emission from tropical soils (forest, savannah) is estimated at 4 (range 2.7–5.7) Tg N/yr. The magnitude of N_2O emissions from intensively fertilized tropical agricultural soils

Table 1 The amount of N₂O emission from agriculture (Mt/yr).

Mineral fertilizer	1.5 (0.5–2.5)
N-fixation	0.5 (0.25–0.75)
Soils after burning	0.1 (0.05–0.2)
Animal wastes	1.5 (0.5–2.5)
Biomass burning	0.2 (0.1–0.3)
Forest conversion	0.4 (0.1–1)
Total	4.2 (1.5–7.25)

has not been quantified. Also, no attempt has been made to separate the tropical soil sources into natural and anthropogenic components.^[5]

The main anthropogenic sources are derived from agriculture and a number of industrial processes such as adipic acid and nitric acid production. New research suggests that N₂O emissions from cropped, nitrogen-fertilized agricultural systems are significant on a global scale as shown in Table 1.

Agricultural Fields

About 40% of N₂O sources are anthropogenic and, among them, fertilized soils account for about 60%.^[5] This figure could be an underestimate because tropical agricultural soil sources resulting from human activities have not been separated from natural tropical soil sources. Agricultural N₂O emissions are considered to arise from fertilization of soils with mineral nitrogen (N) and animal manures, N derived from biological N fixation, and from enhanced soil N mineralization. N₂O is directly evolved during biomass burning, and produced in soil after burning, and enhanced emissions arise during the clearing of tropical forests for agricultural activities.^[4]

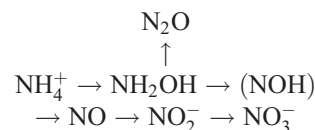
Bouwman^[6] estimated the total emissions of N₂O from a regression equation: total annual direct field N₂O–N loss = 1 + 0.0125 N application (kg N/ha). The value of 1 kg N₂O–N/ha represents the background N₂O–N evolved and the 0.0125 factor expresses for the contribution from fertilization. This estimate includes N sources from a variety of mineral and organic N fertilizers and was based on long-term data sets. About 40% of the estimated N₂O production is derived from North and Central America, Europe, and the former Soviet Union, where about 20% of the world human population resides. Asian countries that hold about 55% of the global human population contribute about 40% of the estimated annual N₂O production.^[4]

MECHANISM OF N₂O PRODUCTION

Nitrification

Nitrification is the reaction whereby ammonium is oxidized to nitrate. In soils, autotrophic and heterotrophic bacteria mediate this process. The most common ammonium

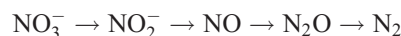
oxidizers are *Nitrosomonas* spp., which are involved in the formation of nitrite, while nitrite oxidation to nitrate is usually achieved by *Nitrobacter* spp. The overall nitrification sequence is as follows:



Two nitrogenous gases may evolve through nitrification, NO and N₂O. The nitrifiers are active over a wide range of temperatures (2–40°C). The overall nitrification process is controlled primarily by ammonium and oxygen concentrations. Because oxygen supply is moderated by soil moisture, the effect of soil water content on N transformation probably reflects its impact on oxygen diffusion in the normal soil moisture range. N₂O emissions from N-fertilizer-applied agricultural fields have been detected by Bremner and Blackmer.^[7]

Denitrification

Biological denitrification refers to the dissimilatory reduction of nitrate and nitrite to produce NO, N₂O, and N₂ by a taxonomically diverse group of bacteria which synthesize a series of reductases that enable them to utilize successively more reduced N oxides as electron acceptors in the absence of oxygen. The general reductive sequence is as follows:



The most abundant denitrifiers are heterotrophs that require sources of electron-reducing equivalents contained in available organic matter. Soil factors that most strongly influence denitrification are oxygen, nitrate concentration, pH, temperature, and organic carbon. N₂O reductase appears to be more sensitive to oxygen than either nitrate or nitrite reductase. Therefore, N₂ production predominates in more anoxic sites and N₂O production may be higher under more aerobic conditions.

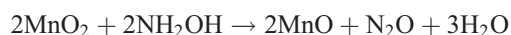
Chemical Decomposition of Nitrite (Chemodenitrification)

There is evidence that nitrite produced by nitrifying or denitrifying microorganisms can also react chemically to form N₂O via “chemodenitrification.”^[8] High nitrite concentrations have been attributed to the inhibition of nitrite oxidation, which is presumed to result from ammonia toxicity to *Nitrobacter*. Several investigators have noted that gaseous loss of N (via NO, N₂O, or N₂) may accompany temporary nitrite accumulation. High concentration of nitrite is sometimes found in anaerobic soils where ammonium and ammonium type fertilizers are applied at high

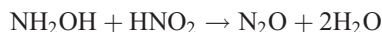
doses. Nitrite ions react chemically with organic molecules forming nitroso groups ($-N=O$) that are unstable.

Other Mechanisms

Some of the N_2O evolved from soils may be formed by chemical decomposition of hydroxylamine (NH_2OH) produced by nitrifying or nitrate-reducing microorganisms, because NH_2OH has been identified as an intermediate in oxidation of ammonium to nitrate by *Nitrosomonas europaea* and has been postulated as an intermediate in microbial reduction of nitrate to ammonium, and several investigations have shown that NH_2OH is decomposed rapidly in soils with formation of N_2O and N_2 by processes that appear to be largely chemical. Other investigations indicated that manganese compounds are involved in the reactions leading to the formation of N_2O and N_2 by chemical decomposition of NH_2OH in soils, and that calcium carbonate and iron compounds are involved in the reactions leading to formation of N_2 in calcareous soils. The reaction is as follows:



Several workers^[1] have postulated that N_2O is produced in soils through interaction of hydroxylamine and nitrite produced by soil microorganisms as follows:



MEASUREMENT OF N_2O EMISSION

N_2O emissions from N-fertilized agricultural fields have been found to vary between 0.001% and 6.8% of the N applied to the field.^[9–11] A proportion of this variability in N_2O estimates, relative to the amount and type of fertilizer applied and to the type of cropping system such as grassland, upland crop, rice paddy, and others, has been attributed to spatial and temporal change in the processes, which produce N_2O in soil.

MITIGATION STRATEGY

Many of the strategies were proposed by Mosier et al.^[12] 1) match N supply with crop demand; 2) close N flow cycles; 3) use advanced fertilizer technologies; and 4) optimize tillage, irrigation, and drainage, in which gaseous emissions that deal primarily with cropping systems could be minimized. Although most of the practices listed are assumed to decrease N_2O emissions, there have been relatively few systematic studies in which various farming practices were compared as to their ability to conserve N and limit N_2O emissions. A number of field studies have been conducted with nitrification inhibitors that could decrease N_2O emissions when used. There are a few studies

in which the potential of using controlled-release fertilizer for decreasing N_2O emission was evaluated as follows.

Nitrification Inhibitors

Because ammonia- and ammonium-producing compounds are the main sources of fertilizer N, maintenance of the applied N in the ammonium form should result in a lower emission of N_2O from cultivated soils. One mechanism of maintaining added ammonium N is to use nitrification inhibitors with the fertilizer. Mosier et al.^[12] summarized the effects of nitrification inhibitors on N_2O emission from fertilized soils both in laboratory and field studies. A number of field studies indicate that nitrification inhibitors do limit N_2O emissions from ammonium-based fertilizers. Several field tests also show that the utilization of a variety of nitrification inhibitors does significantly limit N_2O emissions from the application of ammonium-based fertilizers. To illustrate this point, a study to quantify the effect of nitrification inhibitors N-(2, 5-dichlorophenyl) succinamic acid (DCS) and the application of ammonium sulfate on N_2O emissions was conducted in field lysimeters using carrot (*Daucus carota* L.) as a test crop.^[11] The addition of DCS reduced about 30% of N_2O emission and leaching of nitrate.

Controlled-Release Fertilizer

The use of controlled-release fertilizer has the potential to improve N-use efficiency by matching nutrient release with crop demand, reducing NO_3^- leaching, and denitrification losses. Polyolefin-coated fertilizers are a type of controlled-release fertilizer where fertilizer granules are covered with a thermoplastic resin. The release of the N fertilizer is temperature dependent and is not controlled by hydraulic reactions or by microbial attack of the coating. Greenhouse studies have revealed that controlled-release fertilizer can increase yields with more N-fertilizer use efficiency and reduce NO_3^- leaching.

For example, Minami^[11] observed that a controlled-release fertilizer reduced N_2O emissions in lysimeter studies of carrot at Tsukuba, Japan, in which the fertilizer-induced emissions of N_2O-N during an 83-day period of cultivation were 0.14% and 0.02% of the 250 kg N applied following ammonium sulfate and a controlled-release fertilizer, respectively.

REFERENCES

1. Bremner, J.M. Sources of nitrous oxide in soils. *Nutr. Cycl. Agroecosys.* **1997**, *49*, 7–16.
2. IPCC. In *Climate Change: The IPCC Scientific Assessment*; Houghton, J.T., Jenkins, G.J., Ephraume, J.J., Eds.; Cambridge University Press: Cambridge, 1990.

3. Crutzen, P.J.; Ehhalt, D.H. Effect of nitrogen fertilizers and combustion on the stratospheric ozone layer. *Ambio* **1977**, *6*, 112–117.
4. IPCC. In *Climate Change 1995, Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses*; Watson, R.T., Zinyowera, Mos, R.H., Eds.; Cambridge University Press: Cambridge, 1996.
5. IPCC. In *Climate Change 1994: Radiative Forcing of Climate Change and an Evaluation of the IPCC IS92 Emission Scenarios*; Houghton, J.T., Meira Filho, L.G., Bruce, J., Lee, H., Callander, B.A., Haites, E., Harris, N., Maskell, K., Eds.; Cambridge University Press: Cambridge, 1995.
6. Bouwman, A.F. *Direct Emission of Nitrous Oxide from Agricultural Soils*. RIVM Report No. 773004004; RIVM: Bilthoven, The Netherlands, 1994; 1–28.
7. Bremner, J.M.; Blackmer, A.M. Effects of acetylene and soil water content on emissions of nitrous oxide from soils. *Nature* **1979**, *280*, 380–381.
8. Chalk, P.K.; Smith, C.J. Chemodenitrification. In *Gaseous Loss of Nitrogen from Plant-Soil Systems*; Freney, J.R., Simpson, J.R., Eds.; Martinus Nijhoff/Dr. W. Junk Publishers: The Hague, 1983; 65–90.
9. Eichner, M.J. Nitrous oxide emissions from fertilized soils: Summary of available data. *J. Environ. Qual.* **1990**, *19*, 272–280.
10. Bouwman, A.F. Exchange of greenhouse gases between terrestrial ecosystems and the atmosphere. In *Soils and the Greenhouse Effect*; Bouwman, A.F., Ed.; John Wiley & Sons: New York, 1990; 61–127.
11. Minami, K. Nitrous oxide emissions from agricultural fields. In *Trace Gas Emissions and Plants*; Singh, S.N., Ed.; Kluwer Academic Publishers: Dordrecht, 2000; 215–230.
12. Mosier, A.R.; Duxbury, J.M.; Freney, J.R.; Heinemeyer, O.; Minami, K. Nitrous oxide emissions from agricultural fields: assessment, measurement and mitigation. *Plant Soil* **1996**, *131*, 95–108.

No Till

Paul W. Unger

Conservation and Production Research Laboratory, U.S. Department of Agriculture (USDA), Bushland, Texas, U.S.A.

Abstract

Conservation tillage and no-till farming systems are based on retaining sufficient crop residues on the soil surface, mainly to control erosion. Other benefits include greater water conservation; improved environmental protection; equipment, energy, and labor savings; and often greater net returns to the producer. Some disadvantages that occur under some conditions and the systems, especially no-till, may not be suitable for all conditions. Most disadvantages, however, can be overcome or minimized by careful management.

INTRODUCTION

No-till, also known as no-tillage or zero-tillage, is a type of conservation tillage, which is a planting system that results in at least 30% cover of crop residues on the soil surface after planting the next crop.^[1] Use of conservation tillage provides major soil erosion control benefits and also helps conserve water. In contrast, conventional tillage refers to tillage operations normally used for crop production that bury most residues and result in <30% cover of residues on the soil surface after planting. Tillage that incorporates all residues into soil is clean tillage. Conventional and clean tillage are less effective for controlling erosion and conserving water than conservation tillage under many conditions.

Tillage methods such as sweep, chisel, paraplow, subsoiling, slit, and strip rotary usually qualify as conservation tillage. Even disk tillage may qualify, provided adequate residues are retained on the surface. The ultimate conservation tillage method is no-till for which the next crop is planted without any soil disturbance since harvesting the previous crop. A special planter is usually needed to prepare a narrow, shallow seedbed for the seed, which is planted by the no-till method.^[1] Sometimes, no-till is used in combination with a subsoiling operation that facilitates crop seeding and plant root growth but that leaves the surface residues virtually undisturbed, except for the slot caused by the subsoiling implement.^[1]

Adequate residues are not always produced to provide 30% cover [e.g., dryland (non-irrigated) crops]. Also, a crop such as cotton (*Gossypium hirsutum* L.) may not produce enough residues under some conditions to satisfy the required ground cover for conservation tillage. Under such conditions, some conventional or even clean tillage methods can provide for soil and water conservation. Any tillage method that results in a rough or ridged surface helps reduce soil erosion by wind. Listing (ridge-forming tillage)

is commonly used to help control wind erosion in the cotton-producing area of west Texas, U.S.A., where residue amounts are usually low (personal observation). Even plowing, which brings erosion-resistant clods to the surface, helps control wind erosion on some sandy soils.^[2] Any tillage method that impedes or prevents water flow across the surface helps reduce soil erosion by water and usually helps conserve water. Listing on the contour retains water on the surface, thus reducing erosion and conserving water. Furrow diking in conjunction with listing improves water retention where contour tillage is not used.^[3] Graded-furrow tillage allows excess water to flow slowly from land, thus reducing the potential for erosion; it also provides water conservation benefits.^[4]

ADVANTAGES AND DISADVANTAGES OF USING CONSERVATION TILLAGE AND NO-TILL

Compared with clean tillage, advantages of different conservation tillage types, including no-till, include improved erosion control, a cleaner environment, greater water conservation, equal or greater crop yields, less equipment and lower maintenance costs, lower energy and labor requirements, and greater net returns. Erosion control benefits from conservation tillage are due to more residues being retained on the soil surface. For controlling erosion by wind, residues shield the surface and reduce wind speed at the surface to below the threshold required for erosion to occur. Erosion by water is reduced because residues reduce the rate and amount of water flow across the surface. Residues also result in less soil particle detachment and transport due to raindrop splash and flowing water. The value of surface cover provided by crop residues for controlling erosion by wind and water is shown in Fig. 1.^[5]

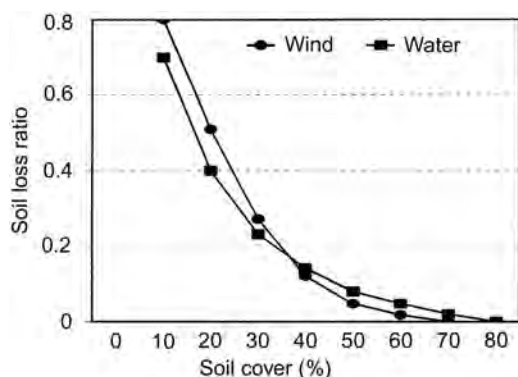


Fig. 1 Relationship of the soil loss ratio (soil loss with cover divided by soil loss from bare soil).

Source: From Papendick, Parr, et al.^[5]

Water conservation with conservation tillage results from residues retarding the rate of water flow across the surface, thus providing more time for infiltration. Residues also shield the surface against raindrop impact, thus dissipating the energy of raindrops, reducing surface sealing, and maintaining favorable infiltration rates. Residues reduce soil water evaporation by shading the soil and slowing the wind at the soil surface. Of course, the soil must have adequate storage capacity for the water to be retained for later use by crops.

Use of conservation tillage reduces erosion, thus resulting in a cleaner environment. Erosion by wind damages crops, causes health and visibility problems, clogs roads and waterways, damages machinery and homes, and pollutes the air. Erosion by water damages crops, roads, machinery, and homes. It also pollutes water with soil particles, chemicals adhering to the particles, and chemicals dissolved in water.

Crop yields are affected by numerous factors. Yields with conservation tillage systems are often greater than with clean tillage, provided no major problems are encountered. Increase in yield, especially with no-till, is usually attributable to greater soil water conservation, especially in subhumid and semiarid regions without irrigation. More favorable soil temperatures may also be involved. In warm or hot regions, high soil temperatures may injure plants, and surface residues with no-tillage result in temperature decreases of up to 10°C,^[6,7] which result in better crop performance. In cool regions, low temperatures with no-till are usually detrimental to crop yields because planting is delayed beyond the optimal date.^[8]

Advantages of lower equipment inventories, equipment maintenance, and energy and labor requirements with conservation tillage are interrelated. With most conservation tillage methods, and especially no-till, tillage frequency and intensity are lower than with clean tillage. As a result, less equipments may be needed, smaller tractors may be satisfactory (for no-till), and the tractors and equipments are used less frequently. This results in less equipment

maintenance and in lower fuel and labor requirements. Some fuel energy savings, however, may be partially offset by the energy required to produce herbicides and fertilizer, especially where no-till is used. The no-till system is based on using herbicides for weed control, and more nitrogen fertilizer is used under some conditions, especially when first converting to the system.

As for yields, many factors affect net returns for a crop production system. However, if production costs are not greater and yields are equal to or exceed those with clean tillage, then net returns should be equal or greater with conservation tillage, especially with no-till, because equipment inventories and maintenance and labor and energy requirements are lower.^[9,10]

Disadvantages

Problems with conservation tillage, especially no-till, occur under some conditions.^[8,11–13] A greater use of herbicides results in concern regarding the potential for polluting soil and water resources. Lower soil temperatures in cool regions delay crop planting, thereby potentially reducing crop yields. On poorly drained soils, additional water retained by using no-till aggravates the excess soil water problem, thus generally reducing crop yields. Some weeds are difficult to control with herbicides, which, along with the high cost of some herbicides, may increase production costs. The possible need for new equipment may also increase production costs, especially when first converting to a no-till system. Because crop residues are retained on the surface when a no-till system is used, there is the potential for increased pest problems (insects, diseases, and rodents). Problems are greater with some insects and less with others, indicating that insect populations must be closely evaluated regardless of tillage system used. Organisms of some plant diseases are carried over to the next crop when residues are retained on the soil surface. Surface residues also provide shelter for rodents, which may be detrimental for the production of some crops. Other possible disadvantages include limited residue availability, greater soil compaction, and a need for greater managerial ability. Certainly, conservation tillage and no-till are not suitable for all conditions. However, with good management, most problems (real or potential) can be minimized or avoided.

RESULTS ACHIEVED BY USING CONSERVATION TILLAGE OR NO-TILL

The value of conservation tillage and no-till farming methods for controlling erosion, conserving water, and increasing crop yields has been shown in numerous studies, but only a few examples will be given here. Probably, the most dramatic example regarding the value of no-till for controlling erosion occurred during a rainstorm on watersheds

planted with corn (*Zea mays* L.) in Ohio, U.S.A.^[14] Treatments were clean tillage with sloping rows (land slope 6.6%), clean tillage with contour rows (land slope 5.8%), and no-till with contour rows (land slope 20.7%). On the respective treatment areas, rainfall was 140, 140, and 129 mm; runoff was 112, 58, and 64 mm; and sediment loss was 50.7, 7.2, and 0.07 Mg/ha. Even though the slope was much greater, soil loss was negligible from the no-till area. Runoff was also low, which provided an opportunity to store more soil water, but soil water information was not given.

Soil losses determined by using a rainfall simulator averaged 2.1 and 5.7 Mg/ha where no-till and conventional tillage, respectively, were used for wheat (*Triticum aestivum* L.) and soybean (*Glycine max* L.) production in Paraná, Brazil. Losses from bare soil averaged 100.2 Mg/ha during the same period.^[15] No-till is widely used to control soil losses in Brazil^[16] and Paraguay (personal observations). Crop residues retained on the soil surface by using no-till was highly effective for reducing runoff and soil losses on Oxisols in Brazil.^[17]

Rainfall totaled 346 mm in July and August 1992 in Australia. Runoff from an Alfisol averaged 136 mm from bare soil where no-till, shallow tillage, and deep tillage treatments were imposed, but only 4 mm with the same treatments when the soil was covered by straw.^[18] As a result, water storage was much greater in the straw-covered soil. The infiltration of water (simulated rainfall at 108 mm/hr for 20 minutes) was greater (less runoff) in Queensland, Australia, when a clay soil was treated with gypsum or when a surface cover of wheat straw was present. With gypsum applied at 0 (check), 2, and 5 Mg/ha, cumulative infiltration was 12%, 37%, and 50%, respectively, with no surface cover, and 45%, 68%, and 76%, respectively, with an 80% surface cover.^[19] The residue cover was more effective than gypsum for enhancing infiltration, thereby improving soil water conservation for crop production. Retaining crop residues on the soil surface by using no-till was found to be effective for increasing water storage in soil in other studies in Australia.^[20,21]

In India, soil water conservation and crop yield increases occurred when plant residues were placed on the soil surface as a mulch.^[22,23] Applying the mulch soon after plant emergence was the most effective treatment.^[23] Similar results should occur by using no-till, provided adequate residues retained on the soil and weeds are effectively controlled. In some studies in India, however, weed control with no-till was not as good as with plowing^[24] or hand weeding,^[25] which resulted in lower crop yields with no-till. Effective weed control is a prerequisite for successful crop production with no-till.

After harvesting irrigated winter wheat, moldboard, rotary, disk, sweep, and no-till treatments were imposed to manage the residues during the fallow period until planting dryland grain sorghum [*Sorghum bicolor* L. (Moench)] 10–11 months later at Bushland, Texas, U.S.A. Weed control was similar with all treatments. Plant-available soil water

contents averaged 149, 143, 158, 179, and 207 mm at sorghum planting and sorghum grain yields averaged 2.56, 2.19, 2.37, 2.77, and 3.34 Mg/ha with the respective treatments. Greater water contents and yields with conservation tillage (sweep and especially no-till) resulted from the retention of more residues on the surface than with other treatments. The presence of residues resulted in greater infiltration and lower evaporation, but the effect of the different processes could not be determined.^[26]

A field study at Akron, Colorado, U.S.A., clearly showed the value of surface residues with conservation tillage (minimum and no-till) for reducing evaporation. Soil water contents one day after 13.5 mm rain were similar to a depth of 15 cm where conventional, minimum, and no-tillage treatments were imposed after harvesting winter wheat. The treatments resulted in 1.2, 2.2, and 2.7 Mg/ha of surface residues, respectively. After 34 rainless days, the soil had dried to $<0.1 \text{ m}^3/\text{m}^3$ water content at depths of 12, 9, and 5 cm, respectively.^[27] The value of surface residues for reducing evaporation was also shown under laboratory conditions.^[28,29]

CONCLUSION

Conservation tillage and no-till farming systems are based on retaining sufficient crop residues on the soil surface, mainly to control erosion. Other benefits include greater water conservation; improved environmental protection; equipment, energy, and labor savings; and often greater net returns to the producer. Some disadvantages that occur under some conditions and the systems, especially no-till, may not be suitable for all conditions. Most disadvantages, however, can be overcome or minimized by careful management.

REFERENCES

1. Soil Science Society of America (SSSA). *Glossary of Soil Science Terms*, 1996; Soil Science Society of America: Madison, 1997.
2. Fryrear, D.W. Wind erosion: Mechanics, prediction, and control. In *Dryland Agriculture, Strategies for Sustainability Advances in Soil Science*; Singh, R.P., Parr, J.F., Stewart, B.A., Eds.; Springer-Verlag: New York, 1990; Vol. 13, 187–199.
3. Clark, R.N.; Jones, O.R. Furrow dams for conserving rainwater in a semiarid climate. In *Proceeding of Conference on Crop Production with Conservation in the 80's*, Chicago, IL, December 1980; American Society of Agricultural Engineers: St. Joseph, 1981; 198–206.
4. Richardson, C.W. Runoff, erosion, and tillage efficiency on graded-furrow and terraced watersheds. *J. Soil Water Conserv.* **1973**, 28 (4), 162–164.
5. Papendick, R.I.; Parr, J.F.; Meyer, R.E. Managing crop residues to optimize crop/livestock production systems for dryland agriculture. In *Dryland Agriculture, Strategies for*

- Sustainability Advances in Soil Science*; Singh, R.P., Parr, J.F., Stewart, B.A., Eds.; Springer-Verlag: New York, 1990; Vol. 13, 253–272.
6. Allen, R.R.; Musick, J.T.; Wiese, A.F. *No-Till Management of Furrow Irrigated Continuous Grain Sorghum*; Prog. Rpt. PR-3332 C, Texas Agricultural Experiment Station: College Station, 1975.
 7. Rockwood, W.G.; Lal, R. Mulch tillage: A technique for soil and water conservation in the tropics. *Span* **1974**, *17*, 77–79.
 8. Radke, J.K. Managing early season soil temperatures in the northern corn belt using configured soil surfaces and mulches. *Soil Sci. Soc. Am. J.* **1982**, *46* (5), 1067–1071.
 9. Crosson, P.; Hanthorn, M.; Duffy, M. The economics of conservation tillage. In *No-Tillage and Surface Tillage Agriculture*; Sprague, M.A., Triplett, G.B., Eds.; John Wiley and Sons: New York, 1986; 409–436.
 10. Harman, W.L.; Martin, J.R. Economics of conservation tillage in Texas. In *Conservation Tillage: Today and Tomorrow*; Gerik, T.J., Harris, B.L., Eds.; Misc. Publ. Mp-1634; Texas Agric. Exp. Stn: College Station, 1987; 24–37.
 11. Phillips, R.E.; Phillips, S.H. *No-Tillage Agriculture, Principles and Practices*; Van Norstrand-Reinhold: New York, 1984.
 12. Sprague, M.A.; Triplett, G.B. *No-Tillage and Surface-Tillage Agriculture*; John Wiley and Sons: New York, 1986.
 13. Unger, P.W. Reduced tillage. In *Semiarid Lands and Deserts, Soil Resource and Reclamation*; Skujinš, J., Ed.; Marcel Dekker: New York, 1991; 387–422.
 14. Harrold, L.L.; Edwards, W.M. A severe rainstorm test of no-till corn. *J. Soil Water Conserv.* **1972**, *27* (1), 30.
 15. Sidiras, N.; Derpsch, R.; Mondardo, A. Effect of tillage systems on water capacity, available moisture, erosion, and soybean yield in Parana, Brazil. In *No-Tillage Crop Production in the Tropics*, Proceedings of a Symposium, Monrovia, Liberia; Akobunda, I.O., Deutsch, A.E., Eds.; International Plant Protection Center, Oregon State University: Corvallis, 1983.
 16. Muzilli, O. Soil conservation policies in the state of Parana, Brazil: The role of agricultural research to attain sustainability. In *Abstracts: 10th International Soil Conservation Organization Conference*, Purdue University, West Lafayette, May 1999; 96–97.
 17. Derpsch, R.; Sidiras, N.; Roth, C.H. Results of studies made from 1977 to 1984 to control erosion by cover crops and no-tillage techniques in Paraná, Brazil. *Soil Till. Res.* **1986**, *8*, 253–263.
 18. Cogle, A.L.; Rao, K.P.C.; Reddy, M.V.; Srinivasan, S.T.; Megarry, D.; Smith, G.D.; Yule, D.F. The impact of the soil biota and cover on runoff and infiltration in a hard setting Alfisol in the SAT. In *Soil and Water Conservation, Challenges and Opportunities*, Proceedings of 8th International Soil Conservation Conference, New Delhi, Dec1994; Bhushan, L.S., Abrol, I.P., Rama Mohan Rao, M.S., Eds.; A.A. Balkema: Rotterdam, Brookfield, 1998; Vol. 2, 1546–1553.
 19. Freebairn, D.M.; Woodruff, D.R.; Rowland, P.; Wockner, G.; Hamilton, A.; Radford, B.J. Options for improving water conservation and utilization in semiarid southern Queensland, Australia. In *Challenges in Dryland Agriculture—A Global Perspective*, Proceedings of the International Conference on Dryland Farming, Amarillo/Bushland, TX, August 1988; Unger, P.W., Sneed, T.V., Jordan, W.R., Jensen, R., Eds.; Texas Agricultural Experiment Station: College Station, 1988; 523–526.
 20. Carter, M.R.; Mele, P.M.; Steed, G.R. The effects of direct drilling and stubble retention on water and bromide movement and earthworm species in a duplex soil. *Soil Sci.* **1994**, *157*, 224–231.
 21. Carter, M.R.; Steed, G.R. The effects of direct-drilling and stubble retention on hydraulic properties at the surface of duplex soils in north-eastern Victoria. *Aust. J. Soil Res.* **1992**, *30*, 505–516.
 22. Friesen, G.H.; Korwar, G.R. Conservation tillage systems for sorghum production under semi-arid conditions in India. *Trop. Pest Manage.* **1987**, *33* (4), 364–366.
 23. Nandan, M.D.; Prasad, S.M. Effect of tillage on weed control and yield of maize. *Indian J. Agron.* **1980**, *25* (1), 146–148.
 24. Agarwal, S.K.; Rajat, D.E. Effect of application of nitrogen, mulching and antitranspirants on the growth and yield of barley under dryland conditions. *Indian J. Agric. Sci.* **1977**, *47* (4), 191–194.
 25. Mane, V.S.; Shingte, A.K. Use of mulch for conserving moisture and increasing the yield of sorghum in dryland. *Indian J. Agric. Sci.* **1982**, *52* (7), 458–462.
 26. Unger, P.W. Tillage and residue effects on wheat, sorghum, and sunflower grown in rotation. *Soil Sci. Soc. Am. J.* **1984**, *48* (4), 885–891.
 27. Smika, D.E. Seed zone soil water conditions with reduced tillage in the semi-arid central Great Plains. In *Proceedings 7th Conference of the International Soil Tillage Research Organization*, Sweden, 1976.
 28. Unger, P.W.; Parker, J.J. Evaporation reduction from soil with wheat, sorghum, and cotton residues. *Soil Sci. Soc. Am. J.* **1976**, *40* (6), 938–942.
 29. Ji, S.; Unger, P.W. Soil water accumulation under different precipitation, potential evaporation, and straw mulch conditions. *Soil Sci. Soc. Am. J.* **2001**, *65* (2), 442–448.

Nutrient Management

Jorge A. Delgado

Soil Plant Nutrient Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.

Abstract

Nutrient management has been defined as “the science and art directed to link soil, crop, weather, and hydrologic factors with cultural, irrigation, and soil and water conservation practices to achieve the goals of optimizing nutrient use efficiency, yields, crop quality, and economic returns, while reducing off-site transport of nutrients that may impact the environment.” Nutrient management, together with soil and water conservation practices, will be key to adapt to the challenges of the 21st century. We need to apply the right amount of fertilizer at the right place at the right time with the right fertilizer sources (4 Rs) and also incorporate the right conservation practices to increase conservation effectiveness, improve nutrient use efficiency, and reduce the transport of nutrients from the farm and across the landscape; this is the 7 Rs of nutrient management and conservation.

INTRODUCTION

“Nutrient management is the science and art directed to link soil, crop, weather, and hydrologic factors with cultural, irrigation, and soil and water conservation practices to achieve the goals of optimizing nutrient use efficiency, yields, crop quality, and economic returns, while reducing off-site transport of nutrients that may impact the environment.”^[1] Although this sounds like a simple task, it is very difficult to work with natural agroecosystems and spatial variability of soil, including variabilities in soil chemical, physical, and biological properties. These variabilities together with temporal and spatial variabilities of weather (e.g., temperature, precipitation, etc.) influence plant physiological responses as well as responses from weeds, pests, and diseases that could also impact crop yield and quality. Even the management of nutrients at a site-specific location and how the field is managed in a given year, will impact the cycling of nutrients and/or residual nutrients in a given year, impacting nutrient availability and yield responses in the following year. The crop rotation, the use of a cover crop, applications of manures, and how the soil is tilled are examples of management practices that will impact the soil biological, physical, and chemical properties and nutrient cycling and management in the following year.

Nutrient management is not a simple task; even management practices that may reduce losses of a nutrient via one pathway (e.g., volatilization and surface runoff) may increase losses of the same nutrient or another nutrient via another pathway (e.g., leaching). Nutrient management is a skillful task of matching the nutrient demands of a crop with the nutrient sources during the growing season considering site-specific field soil properties, the site-specific climate, and site-specific crop management practices. With

that said, nutrient management is key for agricultural production and for food security and global stability, since supplying and/or managing essential crop nutrients is an important way to increase yields and crop quality and contribute to economically viable systems.

NUTRIENT MANAGEMENT IS KEY FOR ADDRESSING GLOBAL CHALLENGES

There are several global challenges that are creating a need for significant increases in worldwide agricultural production per unit area, and nutrient management will be essential to supply adequate amounts of essential nutrients that are needed to increase and sustain these desired higher yield responses to meet global food demands. One of the main challenges driving this need for higher yields is the continuously increasing world population. It is anticipated that within the next few decades, we will have to increase agricultural production by 70% to feed the additional two billion people that it has been projected the world population will have by 2050.^[2] Another challenge related to this projected increase in population by two billion people is that it is anticipated that the demand for higher protein diets will increase in the next few decades due to higher consumption of protein from livestock and dairy products.^[3]

A changing climate with extreme events is another challenge that can impact agricultural production and how we manage nutrients. For example, reports suggest that a changing climate could potentially have a negative impact on agricultural production by 2050 or 2100 due to extreme events such as droughts and/or flooding,^[4] higher temperatures, and potential negative impacts on soil erosion^[5] that could contribute to reduced yields. Fortunately, we could

use nutrient management and conservation practices to contribute to mitigation of and adaptation to the negative effects of a changing climate and extreme events.^[6] However, under conditions of extreme drought, there may be limitations to adaptation capabilities.^[6,7] Water is the most important nutrient and a limiting factor for maximum yields, and under extreme events (droughts), there may be lower potential for adaptation.

Another global challenge is the depletion of water resources, which are so important for maintaining agricultural production across key worldwide agroecosystems.^[4] Some of the most productive worldwide agroecosystems depend on available irrigation water resources. For these regions, irrigated agriculture on average can have twice the yields of nonirrigated agriculture.^[8] Nutrient management in irrigated agriculture is an important component of the challenges being faced, and nutrient managers need to consider the hydrologic cycle when managing nutrients to ensure that management contributes to meeting the higher nutrient demands during the growing season as well as to minimizing the higher potential for nutrient losses via leaching and/or off-site surface transport from potential irrigation erosion and/or surface water runoff. Nutrient management will be important and may change with the different irrigation systems used for each respective region (e.g., surface irrigation, sprinkler irrigation, and drip irrigation). Different irrigation systems could be applied, or in some cases, farmers may even incorporate limited irrigation management practices depending on the year and availability of water for the region.

Nutrient management will be a key part of efforts to adapt to changes in water resources.^[6] Other examples of global challenges that may impact agricultural production systems are desertification, deforestation, salinization, and energy resources. Responses to these challenges and others will have to include nutrient management plans at the site-specific region. Global challenges will likely impact management practices at a given site, and the management responses for crops, forages, and agroforestry systems will need to include nutrient management plans to maximize agricultural production and economic returns at a given site.

NUTRIENT MANAGEMENT AND ENVIRONMENTAL CHALLENGES

Nutrient management is key for increasing yields; however, when nutrients are applied at higher rates than necessary or when they are not managed using the best management practices recommended for a given management-landscape combination, the potential for losses of nutrients to the environment increases. These losses could significantly contribute to environmental challenges. It has been reported that the increase in use of nitrogen inputs has

contributed to increased losses of reactive nitrogen. Adding nitrogen to agricultural systems increases the probability of higher rates of nitrogen losses from the nitrogen cycle as ammonia, ammonium, nitrogen oxides, nitrous oxide, and nitrate (NO_3); these nitrogen compounds are all reactive forms of nitrogen.^[9]

The excess application of these nutrients could contribute to higher losses that could significantly impact the function of terrestrial, freshwater, and marine ecosystems, including impacts to quality of groundwater resources.^[9–11] Several reports in the literature about the effects of nutrient losses have listed different impacts to air, soil, and water quality. These environmental impacts are reported to contribute to lower pH of forest soils, resulting in acidification. Other negative impacts can include poor water quality conditions that can have negative societal impacts due to eutrophication of aquatic ecosystems, hypoxia in freshwater and coastal areas, harmful algae blooms, and contamination of drinking water (surface and groundwater) resources with NO_3 concentrations above levels considered safe for consumption. Impacts to air quality from nutrient losses can have negative effects on society such as odors and contributions to trace gas emissions (e.g., nitrous oxide) that are reported to contribute to climate change.^[4] The scientific literature shows that nutrients are critical for food security and societal stability; however, it also shows that nutrient management must be used to reduce the environmental challenges created due to their intensive use, in order to minimize nutrient losses and their potential negative impacts to air, soil, and water quality.

The loss of nitrogen from fields contributes to an effect known as the nitrogen cascade, where the environmental impacts are multiplied as the reactive nitrogen is cycled through the environment and contributes to additional losses of reactive nitrogen.^[9] Accumulations of these nutrients (e.g., phosphorous) could also contribute to legacy effects where negative impacts are increased several years after continued increases in the soil content, to the point that the losses are significantly increased.^[11] Fortunately, nutrient management, together with conservation practices, can help minimize the losses of nutrients and minimize these environmental challenges.^[12] Delgado^[12] reported that there is a need to increase the communication between nutrient managers and conservation practitioners to increase nutrient use efficiencies at the farm level while reducing the off-site transport of nutrients.

KEEPING UP WITH ADVANCES IN NUTRIENT MANAGEMENT

New technological advances in software, electronics, and engineering that are applied to the field of nutrient

management are continually being developed, and nutrient managers need to keep up with these advances that can potentially contribute to higher yields and nutrient use efficiencies and to reduced nutrient losses to the environment. Specifically, advances in remote sensing, field sensors, unmanned aerial vehicles, drones, computer models, nutrient indexes, geographic information systems, mobile phones, and new nutrient management apps are examples of areas where new technologies are achieving significant advances that are being applied for nutrient management and/or have nutrient management implications. A large number of these advances in nutrient management are being driven by advances in linking and/or managing databases on soils, crops, weather, and hydrological properties for quick integration and analysis of information to develop improved management decisions.

More and more, these advances are also integrating spatial and temporal properties in their analysis. The accessibility of the Internet and a faster flow of information have been contributing to the integration of these large databases to provide quick analysis and recommendations at the field level and on the go as machinery is moving across the field. Advances of mobile technologies, such as smartphones that can connect to the cloud quickly, in combination with new, advanced phone apps, are facilitating the flow of information and helping users make decisions from wherever they are. By integrating the information from “big data” quickly, nutrient managers can make decisions on the go and respond to nutrient

management needs. The capabilities of global positioning systems are also allowing the development of highly accurate maps (within a few centimeters), and the controls and sensors of new nutrient management/agricultural machinery are helping nutrient managers improve management decisions to increase nutrient use efficiencies on the go. There are several private sector companies that are expanding these capabilities to allow farmers and nutrient managers to integrate yield maps and remote sensing maps with nutrient analysis maps to develop improved management recommendations. An example is the John Deere Mobile Farm Manager software applications.^[13] These apps can help save information on where soil samples were collected in the field, access field information, and share the information with trusted advisors or export information to models. This is just an example and there are several other private industry apps and new software that are being used by consultants, service providers, nutrient managers, extension personnel, and farmers.

Spatial variability of nutrients across the field can be monitored with these new technologies, and we can use precision farming to improve management of nutrients and reduce nutrient losses to the environment.^[14,15] We could use these new technologies to apply variable rates to better match the rate of nutrient application with the rate of nutrient uptake across the field during the growing season, thus improving nutrient use efficiencies and reducing nutrient losses.^[14,15] Or, we could divide the field into management zones and manage this spatial

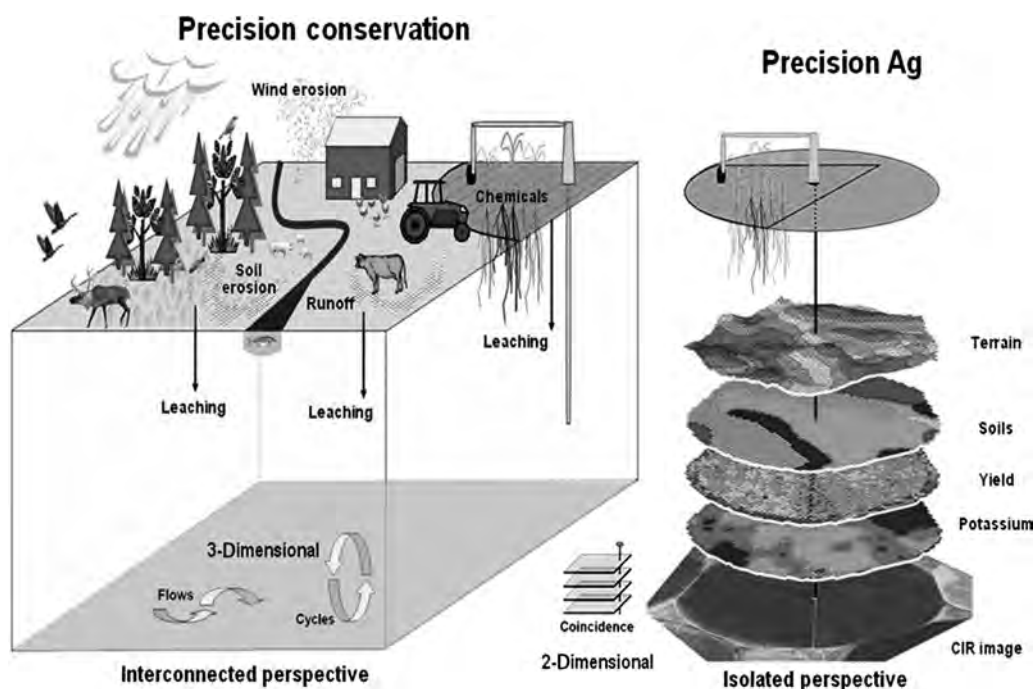


Fig. 1 The site-specific approach can be expanded to a 3-D scale approach that assesses inflows and outflows from fields to watershed and region scales.

Source: From Berry, Delgado, et al.^[16] ©2003 Journal of Soil and Water Conservation.

variability by areas of the field to better improve the nutrient sources for a given subarea of the field, matching the average nutrient uptake in the zone with the average nutrient source per zone during the growing season.^[17] Sensors are tools that can help provide *in situ* rapid evaluations of the nutrient status of a field to manage this spatial variability and integration of information across the field to improve nutrient management decisions.^[14,15]

NEED FOR A JOINT NUTRIENT MANAGEMENT AND CONSERVATION APPROACH TO INCREASE EFFICIENCY AND REDUCE ENVIRONMENTAL LOSSES OF NUTRIENTS

Even with the advances as far as integrating large databases of information to improve nutrient management and apply the right amount of fertilizer at the right place at the right time with the right fertilizer sources (4 Rs),^[18] there is a need to also incorporate the right conservation practices using precision information to increase conservation effectiveness, improve nutrient use efficiency, and reduce the transport of nutrients out of a farm and across the landscape.^[12] Delgado reported that the 4 Rs for nutrient management alone are not enough to reduce nutrient losses. The 4 Rs alone do not integrate variable information about how to maintain and/or improve soil health and soil quality, or how to increase soil carbon sequestration, or how to reduce erosion to maintain soil sustainability and soil health at the site-specific location.

Practices that maintain and/or improve soil health and quality, increase soil carbon sequestration, or reduce erosion will impact nutrient availability and cycling at a given site and should be part of the decision process of site-specific nutrient management but they are not part of the 4 Rs decision process.^[12] Additionally, even if the 4 Rs are applied at a site-specific field, due to the spatial and temporal variability of weather, soils, and crops, the potential for NO_3 losses could be significantly high unless precision conservation practices are implemented across the field, farm, and landscape to minimize the transport of nutrients. For example, the 2012 drought reduced yields significantly across large regions of the Midwest, significantly reducing the potential for nitrogen uptake and increasing the potential for NO_3 leaching losses.^[7] This is an example of the need for and potential of incorporating precision conservation practices at a field level and across farms and the landscape to reduce the potential for nutrient losses.^[12,16,19] There is a need to bring crop advisors and nutrient managers together with soil and water conservation practitioners to increase application of precision conservation so we can increase nutrient use efficiencies at a farm level while reducing the transport of nutrients across the farming landscape and natural areas (Figs. 1–3).

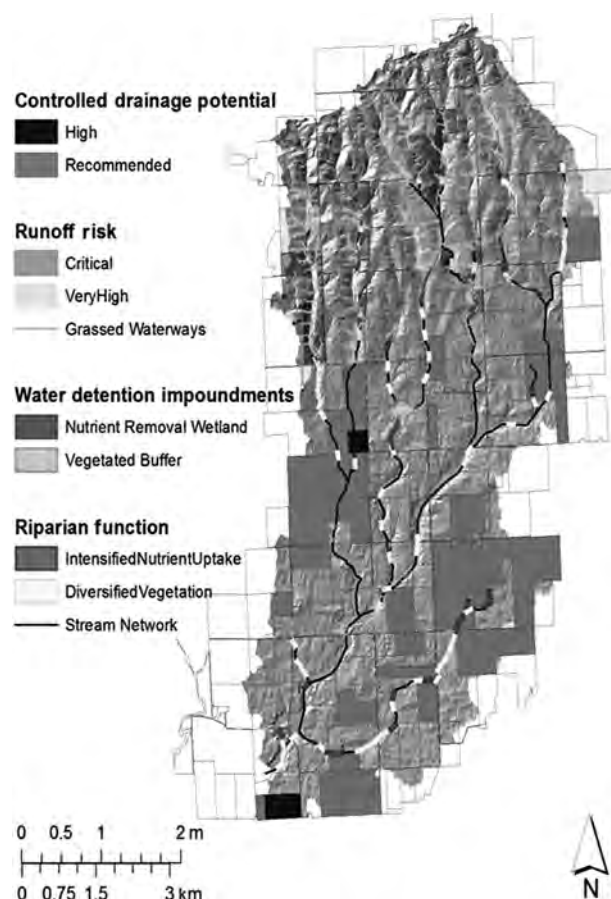


Fig. 2 One possible conservation planning scenario for Lime Creek.

Source: From Tomer, Porter, et al.^[19] ©2013 Journal of Soil and Water Conservation.

Precision conservation was defined as a set of spatial technologies that accounted not only for the spatial variability across the agricultural systems but also through the natural areas outside of the field. Precision conservation analysis could help connect the flows of nutrients from site-specific fields to surrounding landscape areas where buffers, riparian zones, wetlands, and forests could be applied to reduce the transport of these flows.^[16,19] It has been reported that a precision conservation approach could contribute to the management of nutrients while they are in the farm to help reduce movement of nitrogen and other chemicals across the landscape of a farm and natural areas.^[12] It has also been reported that “if we are to increase conservation effectiveness and nutrient use efficiency and minimize the losses of sediment and nutrients to the environment, we need to apply the right product (fertilizer), at the right fertilizer rate, with the right method of fertilizer application, with the right conservation practice, and with conservation practice at the right place, and at the right scale of conservation practice, with both the fertilizer and the conservation practice applied at the right time (right product, right rate, right method, right practice, right place,

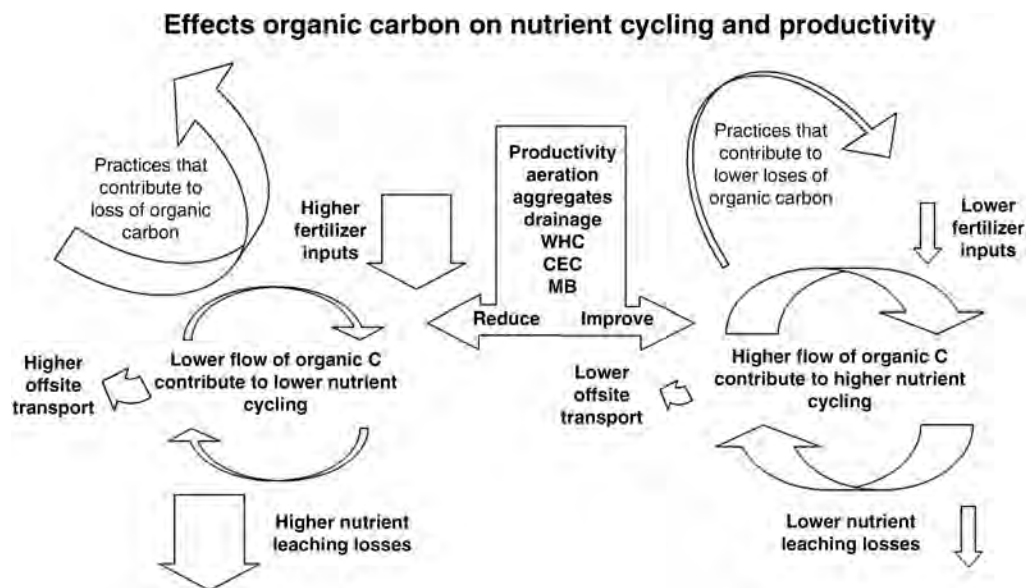


Fig. 3 Effects of organic carbon on nutrient cycling and productivity.

Source: From Delgado & Follett.^[20] ©2002 Journal of Soil and Water Conservation.

right scale, and right time: the 7 Rs of nutrient management and conservation).”^[12]

CONCLUSION

Nutrient management has been defined as “the science and art directed to link soil, crop, weather, and hydrologic factors with cultural, irrigation, and soil and water conservation practices to achieve the goals of optimizing nutrient use efficiency, yields, crop quality, and economic returns, while reducing off-site transport of nutrients that may impact the environment.”^[1] Nutrient management will be one of the most important tools in efforts to achieve food security during the 21st century and feed the increasing population with growing demand for higher protein diets. At the same time, improved nutrient management will also contribute significantly to addressing other environmental challenges of the globe by reducing losses of nutrients to the environment.

Nutrient management technologies are being developed in the world to allow for rapid analysis of large databases for precision farming, but the best pathway ahead to increase nutrient use efficiencies and minimize the losses of nutrients involves incorporation of precision conservation to increase conservation effectiveness across the landscape for air, soil, and water quality. In areas where these new technologies are not being used, other basic principles of nutrient management can be applied to increase nutrient use efficiencies while increasing air, soil, and water conservation.

There is a need to increase communication and cooperation between nutrient managers and conservation practitioners to implement a 7 Rs approach to apply the “right

product, right rate, right method, right practice, right place, right scale, and right time: the 7 Rs of nutrient management and conservation” to maximize nutrient use efficiencies, decrease losses of nutrients to the environment, and reduce their transport across the landscape.^[12]

DISCLAIMER

Manufacturers’ names are necessary to report factually on available data; however, the USDA neither guarantees nor warrants the standard of the product, and the use of a given name by the USDA does not imply approval of that product to the exclusion of others that may be suitable.

REFERENCES

1. Delgado, J.A.; Lemunyon, J. Nutrient management. In *Encyclopedia of Soil Science*, 2nd Ed.; Lal, R., Ed.; Markel and Decker: New York, 2006; 1119–1121.
2. Food and Agriculture Organization of the United Nations. How to Feed the World in 2050; Food and Agriculture Organization of the United Nations. In *Paper Prepared for How to Feed the World in 2050: High-level Expert Forum*, Rome, Oct 12–13, 2009. Available at www.fao.org/fileadmin/templates/wsfs/docs/expert_paper/How_to_Feed_the_World_in_2050.pdf (accessed April 16, 2016).
3. Linehan, V.; Thorpe, S.; Andrews, N.; Kim, Y.; Beaini, F. *Food Demand to 2050: Opportunities for Australian Agriculture*; Conference paper 12.4; 42nd Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) Outlook Conference, Canberra, Mar 6–7, CC BY 3.0; Commonwealth of Australia, 2012.

4. IPCC (Intergovernmental Panel on Climate Change). Summary for policymakers. In *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*; Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L., Eds.; Cambridge University Press: Cambridge and New York, 2007. Available at <https://www.ipcc-wg1.unibe.ch/publications/wg1-ar4/ar4-wg1-spm.pdf> (accessed April 21, 2016).
5. Nearing, M.A.; Pruski, F.F.; O'Neal, M.R. Expected climate change impacts on soil erosion rates: a review. *J. Soil Water Conserv.* **2004**, *59*, 43–50.
6. Delgado, J.A.; Groffman, P.M.; Nearing, M.A.; Goddard, T.; Reicosky, D.; Lal, R.; Kitchen, N.R.; Rice, C.W.; Towery, D.; Salon, P. Conservation practices to mitigate and adapt to climate change. *J. Soil Water Conserv.* **2011**, *66*, 118A–129A.
7. Lal, R.; Delgado, J.A.; Gulliford, J.; Nielsen, D.; Rice, C.W.; Van Pelt, R.S. Adapting agriculture to drought and extreme events. *J. Soil Water Conserv.* **2012**, *67*, 162A–166A.
8. Rangely, W.R. Irrigation and drainage in the world. In *Proceedings of Water and Water Policy in World Food Supplies*, College Station, Texas, May 25–30, 1985; Jordan, W.R., Ed.; Agriculture and Mining University Press: College Station, 1987; 29–35.
9. Galloway, J.N.; Aber, J.D.; Erisman, J.W.; Seitzinger, S.P.; Howarth, R.W.; Cowling, E.B.; Cosby, B.J. The nitrogen cascade. *BioScience* **2003**, *53* (4), 341–356.
10. Vitousek, P.M.; Aber, J.D.; Howarth, R.W.; Likens, G.E.; Matson, P.A.; Schindler, D.W.; Schlesinger, W.H.; Tilman, D.G. Human alteration of the global nitrogen cycle: Sources and consequences. *Ecol. Appl.* **1997**, *73* (3), 737–750.
11. Smith, D.R.; King, K.W.; Williams, M.R. What is causing the harmful algal blooms in Lake Erie? *J. Soil Water Conserv.* **2015**, *70*, 27A–29A.
12. Delgado, J.A. 4 Rs are not enough: We need 7 Rs for nutrient management and conservation to increase nutrient use efficiency and reduce off-site transport of nutrients. In *Soil-Specific Farming: Precision Agriculture*; Lal, R., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2015; 89–126.
13. Deere & Company. John Deere Mobile Farm Manager. Deere & Company. Available at https://www.deere.com/en_US/products/equipment/ag_management_solutions/information_management/john_deere_mobile_farm_manager/john_deere_mobile_farm_manager.page (accessed April 21, 2016).
14. Raun, W.R.; Solie, J.B.; Johson, G.V.; Stone, M.L.; Mullen, R.W.; Freeman, K.W.; Thompson, W.E.; Lukina, E.V. Improving nitrogen use efficiency in cereal grain production with optical sensing and variable rate application. *Agron. J.* **2002**, *94*, 815–820.
15. Scharf, P.C.; Shannon, D.K.; Palm, H.L.; Sudduth, K.A.; Drummond, S.T.; Kitchen, N.R.; Mueller, L.J.; Hubbard, V.C.; Oliveira, L.F. Sensor-based nitrogen applications out-performed producer-chosen rates for corn in on-farm demonstrations. *Agron. J.* **2011**, *103*, 1683–1691.
16. Berry, J.K.; Delgado, J.A.; Khosla, R.; Pierce, F.J. Precision conservation for environmental sustainability. *J. Soil Water Conserv.* **2003**, *58*, 332–339.
17. Khosla, R.; Fleming, K.; Delgado, J.; Shaver, T.; Westfall, D. Use of site-specific management zones to improve nitrogen management for precision agriculture. *J. Soil Water Conserv.* **2002**, *57*, 513–518.
18. Roberts, T.L. Right product, right rate, right time and right place ... the foundation of best management practices for fertilizer. In *Fertilizer Best Management Practices: General Principles, Strategies for Their Adoption and Voluntary Initiatives vs Regulations*. Paper presented at the IFA International Workshop on Fertilizer Best Management Practices, Brussels, Belgium, Mar 7–9, 2007, International Fertilizer Industry Association: Paris, 1997; 29–32.
19. Tomer, M.D.; Porter, S.A.; James, D.E.; Boomer, K.M.B.; Kostel, J.A.; McLellan, E. Combining precision conservation technologies into a flexible framework to facilitate agricultural watershed planning. *J. Soil Water Conserv.* **2013**, *68*, 113A–120A.
20. Delgado, J.A.; Follett, R.F. Carbon and nutrient cycles. *J. Soil Water Conserv.* **2002**, *57*, 455–464.

Nutrient Management: Fertility and

John L. Havlin

Department of Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.

Abstract

Future population growth will drive increased food and fiber production per unit of land. Increasing crop productivity increases soil nutrient removal and the importance of replenishing soil fertility through effective and efficient nutrient management. Native soil nutrient supply depends on the soils' ability to buffer nutrient loss through crop removal. Mineralization of soil organic fractions provides limited supplies of nitrogen, sulfur, and micronutrients, while mineral dissolution and surface exchange reactions resupply phosphorus, potassium, calcium, magnesium, and micronutrients. Nutrient mobility in soil influences ion transport to plant roots, the evaluation of nutrient availability to plants, and ultimately nutrient management decisions. Effective nutrient management requires quantifying crop nutrient requirement and the nutrient supplying capacity of the soil through soil testing. Once these are determined, the nutrient management plan includes the determination of optimum nutrient rate, appropriate nutrient source(s), most effective nutrient placement method(s), and nutrient application timing. Management decisions will vary depending on the specific nutrient. Effective management of soil nutrients will ensure crop productivity meet consumption demand, while minimizing impacts of nutrient use on the environment.

INTRODUCTION

Nutrients are essential to support life and are acquired through diverse food sources, ultimately supplied by the soil. While plants are a major direct human food source, animals used as human food also obtain nutrients from a variety of plants (forage and grains) in their diets. As plants are consumed or removed from a field, the nutrients are also removed. With continued crop removal over time, soil nutrients are depleted and plant productivity declines. As food demand increases with increasing population growth, a greater agricultural output demand requires substantial replacement of soil nutrients through the addition of inorganic fertilizers and organic wastes. Therefore, to optimize nutrient availability to crops and to minimize environmental risk of nutrient use, it is essential to understand nutrient reactions and processes in soils (*soil fertility*) and to efficiently manage nutrient inputs (*nutrient management*) to ensure adequate soil nutrient supply to support life. Without effective management of soil nutrients, 30–40% more land area will be needed to produce the same crop yields depending on the crop, soil, and climatic region. Therefore, crop producers must take advantage of technologies that increase productivity as well as those that minimize productivity loss.

SOIL FERTILITY

Soil fertility represents the diverse interactions between physical, chemical, and biological soil processes that control plant nutrient supply. Understanding these processes

and interactions is critical to optimizing nutrient availability to crops. Maximum crop production depends on the growing season environment and the producers' skill to minimize yield-limiting factors using appropriate management technologies. Crop productivity is maximized when the *most* limiting factor to yield potential is corrected first, the second most limiting factor next, and so on (*Law of the Minimum*). For example, the correct nutrient supply may not result in the highest yield potential if plant-available water (H₂O) is the *most* limiting factor. Thus, minimizing H₂O as a yield-limiting factor must be accomplished before the crop can fully respond to management of any other factor.

Essential Plant Nutrients

Seventeen elements are considered essential plant nutrients since they are involved in metabolic functions such that the plant cannot complete its life cycle without these elements (Table 1). The most abundant non-mineral nutrients (carbon, hydrogen, and oxygen) are obtained from carbon dioxide (CO₂) and H₂O and converted into carbohydrates that form amino acids, proteins, nucleic acids, and other compounds through photosynthesis. The supply of CO₂ has increased from ~310 to 390 ppm CO₂ since 1960. The supply of H₂O rarely limits photosynthesis directly but reduces plant growth when plant-available H₂O is limited. The remaining 14 macronutrients and micronutrients are classified on their relative abundance in plants. The nitrogen (N) concentration in most plants is greater than the

Table 1 Essential plant nutrients and their relative and average concentration.

Classification	Nutrient		Concentration in plants ^a	
	Name	Symbol	Relative	Average
Macronutrients	Hydrogen	H	60,000,000	6%
	Carbon	C	40,000,000	45%
	Oxygen	O	30,000,000	45%
	Nitrogen	N	1,000,000	1.5%
	Potassium	K	250,000	1.0%
	Calcium	Ca	125,000	0.5%
	Magnesium	Mg	80,000	0.2%
	Phosphorus	P	60,000	0.2%
	Sulfur	S	30,000	0.2%
	Chloride	Cl	3,000	100 ppm
Micronutrients	Iron	Fe	2,000	100 ppm
	Boron	B	2,000	20 ppm
	Manganese	Mn	1,000	50 ppm
	Zinc	Zn	300	20 ppm
	Copper	Cu	100	6 ppm
	Nickel	Ni	2	<1 ppm
	Molybdenum	Mo	1	<1 ppm

^aConcentration expressed on a dry matter weight basis.

other 13 nutrients combined. Although nutrient content varies greatly between plant species, most plants also require substantial amounts of calcium (Ca) and potassium (K; Table 1). Plants absorb many non-essential elements present in the soil solution. For example, aluminum ions (Al^{3+}) in plants can be high when soils contain relatively large amounts of soluble Al (low pH soil). In some severely acid soils, Al toxicity can reduce plant growth and yield. In high-pH soils, sodium ion (Na^+) can be elevated in soil solution and/or on the exchange complex, increasing Na^+ in the plant, which can also be detrimental to plant growth.

Many soil, climate, and management factors influence plant nutrient availability. When nutrient concentration is *deficient* enough to severely reduce plant growth and yield, distinct visual deficiency symptoms appear (Fig. 1). Extreme deficiencies can result in plant death. Visual symptoms may not appear when a nutrient is marginally deficient (*hidden hunger*), but plant yields may still be reduced. Plant nutrient concentration above which plant growth or yield is not increased is the *critical level* or *range*. Critical nutrient ranges vary among plants but always occur in the transition between nutrient deficiency and sufficiency (Fig. 1). *Sufficiency* or *luxury consumption* is the concentration range, where added nutrient does not increase yield but increases nutrient concentration. Nutrient concentration is considered *excessive* or *toxic*, when plant growth and yield are reduced.

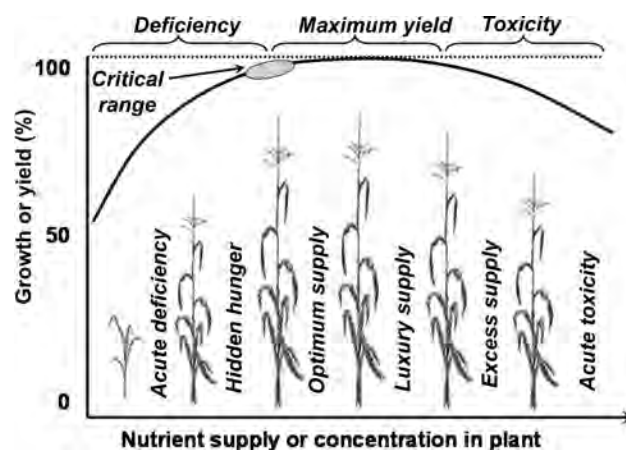


Fig. 1 Optimum nutrient supply is essential for maximizing plant growth and nutrient use. Nutrient addition beyond critical nutrient range does not increase plant growth.

Soil Processes Supplying Plant-Available Nutrients in Soil

After plants absorb nutrients from soil, nutrients are exported in harvested plant materials or returned to soil in plant residues left in the field. These nutrients are subject to the same biological and chemical reactions as nutrients added as fertilizers and organic wastes. Although this “cycle” varies among nutrients, understanding nutrient dynamics in the soil–plant–atmosphere system is essential to successful nutrient management.

As plants absorb nutrients from soil solution, reactions occur to resupply the soil solution (Fig. 2). These reactions

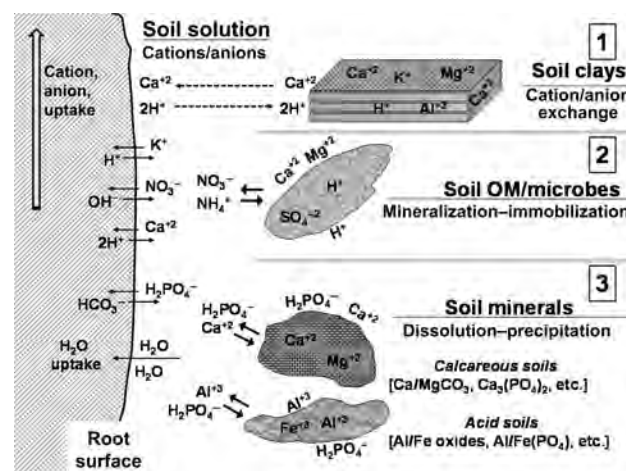


Fig. 2 Primary processes of buffering nutrients in soil solution absorbed by plant roots. Clay minerals (cation/anion exchange), OM (mineralization), and mineral compounds (dissolution) resupply nutrients to soil solution, as they are removed from solution by plant uptake.

Source: From Barker & Pilbeam,^[1] Grant & Milkha,^[2] and Havlin, Tisdale, et al.^[3]

are important to nutrient availability; however, the dominant reaction varies between nutrients. For example, biological processes are more important for N and sulfur (S) availability, and surface exchange reactions are important for Ca, magnesium (Mg), and K supply to roots.

Cation and anion exchanges

Exchange of cations and anions on surfaces of clay minerals, inorganic compounds, organic matter (OM), and roots is one of the most important soil chemical properties and greatly influences nutrient availability (Fig. 2). Ions adsorbed on mineral and organic surfaces are reversibly exchanged with other ions in the solution. *Cation exchange capacity* (CEC) represents the quantity of negative (–) charge available to attract cations in solution, and the *anion exchange capacity* (AEC) represents positive (+) charge attracting anions in a solution. In most soils, CEC is greater than AEC.

The source of (+) or (–) surface charges found in clay minerals is comprised of permanent charge unaffected by solution pH, and on (+) and (–) charges located on clay mineral edges and soil OM (Fig. 3).

At low pH, more (+) charge exists due to higher hydrogen ions (H^+) on mineral edges and OM; as pH increases, H^+ concentration decreases, which increases (–) edge or surface charge. At $pH > 7$, all H^+ on edges are neutralized, which maximizes (–) pH-dependent charge (Fig. 3). Soils with greater clay and OM contents have a higher CEC than sandy, low OM soils (Table 2).

CEC influences nutrient availability, as most adsorbed cations are essential plant nutrients, except for Al^{3+} in acid soils and Na^+ in alkaline pH soils. Cations are adsorbed into CEC with different strengths, which influence the relative ease of desorption. The *lyotropic series*, or relative strength of adsorption, is Al^{3+} (H^+) > calcium ions (Ca^{2+}) > magnesium ions (Mg^{2+}) > potassium ion (K^+) = ammonium ion

Table 2 Typical range in CEC for moderately weathered (Mollisol) and highly weathered (Ultisol) soils.

Soil textural class	Mollisol	Ultisol
	(cmol kg ^{–1})	
Sands (light colored)	3–5	~1
Sands (dark colored)	10–20	1–3
Loams	10–15	1.5–5
Silt loams	15–25	2–6
Clay and clay loams	20–50	3–5
Organic soils	50–100	20–40

(NH_4^+) > Na^+ . Cations with greater charge are more strongly adsorbed. For cations of similar charge, adsorption strength is determined by the size of the hydrated cation; greater hydrated cation radii (Mg^{2+}) reduce adsorption because they cannot get as close to the exchange surface as smaller cations (Ca^{2+}).

Base saturation (BS) represents the percentage of CEC occupied by bases [Ca^{2+} , Mg^{2+} , K^+ , and (Na^+)], which increases with increasing soil pH (Fig. 4). The availability of Ca^{2+} , Mg^{2+} , and K^+ increases with increasing %BS. For example, 80% BS provides essential nutrients (cations) to plants easier than 40% BS. At pH 5 and 6, most soils have ~50% and 80% BS, respectively.

Soil solution anions are adsorbed into (+) charged sites on the surface of clay minerals and soil OM. AEC increases with decreasing soil pH (Fig. 3). The adsorption strength is dihydrogen phosphate ion ($H_2PO_4^-$) > sulfate ions (SO_4^{2-}) >> nitrate ion (NO_3^-) = chloride ion (Cl^-). In most soils, $H_2PO_4^-$ is the primary anion adsorbed although in severely acid soils SO_4^{2-} adsorption represents a major source of plant-available S.

Soil acidity and alkalinity. Acid soils usually occur in regions where annual precipitation is >600–800 mm. Natural soil acidification is enhanced with increasing rainfall since rain $pH \leq 5.7$, depending on pollutants such as sulfur

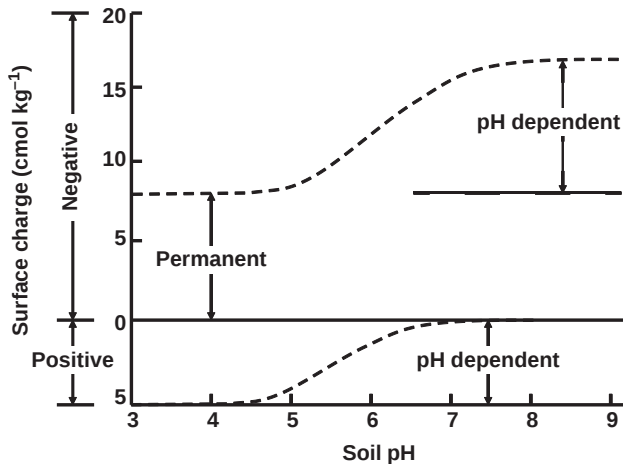


Fig. 3 Permanent and pH-dependent charge associated with clay minerals. As soil pH increases (+), charge decreases and (–) charge increases, increasing CEC.

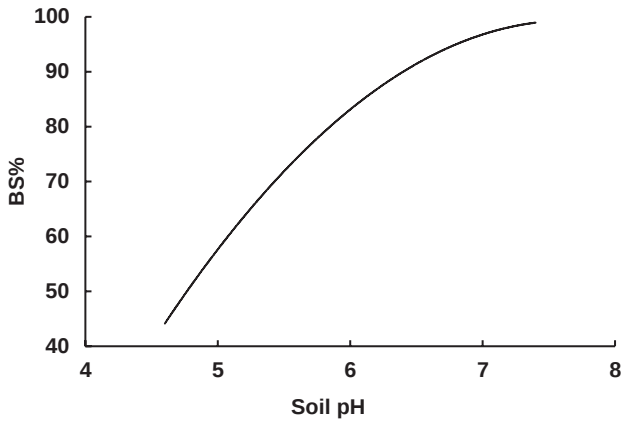


Fig. 4 Influence of soil pH on BS% in mineral soils.

dioxide, nitrogen dioxide, and others which lower rainfall pH. Management factors increasing soil acidity include the following: 1) use of acid forming fertilizers; 2) crop removal and/or leaching of cations decreasing %BS; and 3) the decomposition of organic residues. Optimum soil pH depends on the crop and ranges between 4.5 and 7.5. Neutralizing soil acidity by liming to $\text{pH} \geq 5.5$ will reduce exchangeable Al^{3+} to ~20% of the CEC and will usually prevent Al toxicity-related yield loss.

Calcareous soils contain the solid mineral calcium carbonate (CaCO_3), exhibit soil $\text{pH} \geq 7.2$, and occur in regions of <500 mm annual precipitation. In areas where annual rainfall is >800 mm, the acidifying effect of rain increases depth to CaCO_3 to below the root zone. If CaCO_3 exists, all of it must be dissolved and neutralized before soil pH could decrease. In most situations, reducing soil pH by neutralizing CaCO_3 is too costly. High soil pH in calcareous soils reduces the availability of several micronutrients [e.g., ferric ions and zinc ions (Zn^{2+})].

Excessive salts in alkaline soils can reduce plant growth and yield depending on crop sensitivity. In arid and semiarid regions as H_2O evaporates, salts are deposited near the soil surface to form saline and/or sodic soils and are characterized by their electrical conductivity (EC_{se}), soil pH, and exchangeable Na% (ESP; Table 3). In saline soils, soluble salts interfere with plant growth, although salt tolerance varies with plant species. In sodic soils ($\text{ESP} > 15\%$), soil aggregates disperse reducing infiltration and soil H_2O permeability, while excess Na can also create nutritional disorders. Saline-sodic soils exhibit both high salt and Na content.

Soil OM

Soil OM represents organic materials in various stages of decay (Fig. 5). The most important fraction to soil fertility is the stable OM that is relatively resistant to further decomposition (Fig. 5). Soil humus, the largest component of soil OM, is relatively resistant to microbial degradation but is essential for maintaining optimum soil physical conditions (aggregation, infiltration, aeration, etc.) important for plant growth. Fresh plant and animal residues undergo rapid decomposition. The microbial biomass represents soil microorganisms responsible for mineralization and

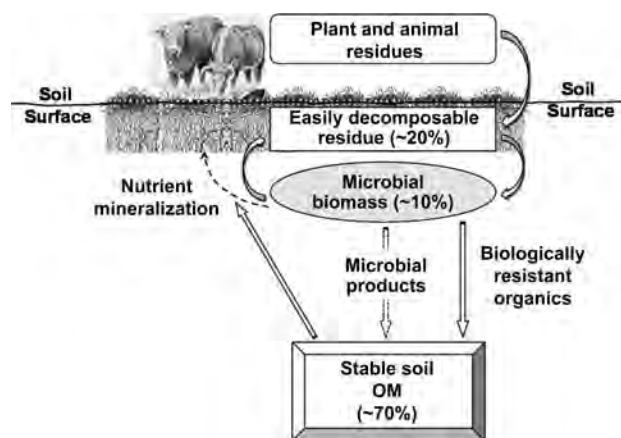


Fig. 5 Pathway of the degradation of plant and animal residues to stable soil OM. The mineralization of organic compounds produces plant-available nutrients.

immobilization processes that influence the availability of N, phosphorus (P), S, and other nutrients. Residue decomposition and stable OM formation depend on climate, soil type, and soil and crop management practices. Soil OM greatly influences various soil processes and properties that influence nutrient availability (Table 4). For example, increasing soil OM increases soil aggregation and decreases bulk density, which improves H_2O infiltration, air exchange, root proliferation, and crop productivity. Soil OM levels depend on soil and crop management. If management is changed, a new OM level is eventually attained that may be lower or higher than the previous level. Management systems should be adopted that sustain or increase soil OM and productivity. Proper nutrient management will produce high yields, which will increase the quantity of residue and soil OM.

Mineral solubility in soils

Mineral solubility refers to the concentration of ions in solution supported or maintained by solid phase mineral(s). There are many soil minerals influencing nutrient concentrations in soil solution. For example, in acid soils, $\text{FePO}_4 \cdot 2\text{H}_2\text{O}$ influences P availability by the following reaction:



The K_{sp} for this reaction is as follows:

$$K_{\text{sp}} = (\text{H}_2\text{PO}_4^-)(\text{H}^+)$$

As H_2PO_4^- decreases with P uptake, strengite dissolves to resupply or maintain solution H_2PO_4^- concentration. This reaction also shows that as H^+ increases (decreasing pH), H_2PO_4^- decreases. Therefore, specific P minerals present in soil and the concentration of solution P supported by these

Table 3 Classification and properties of salt-affected soils.

Classification	EC_{se} (dS m^{-1}) ^a	Soil pH	ESP% ^b	Physical condition
Saline	>4	<8.5	<15	Normal
Sodic	<4	>8.5	>15	Poor
Saline-sodic	>4	<8.5	>15	Normal

^a EC_{se} represents electrical conductivity of soil H_2O removed from a sample of saturated soil. Increasing salt concentration increases EC_{se} .

^bESP% (exchangeable Na percent) represents the percentage of CEC occupied with Na^+ .

Table 4 Characteristics of soil OM and associated effects on soil and plants.

Property	Effect on soil	Effect on plant
Soil color	Imparts dark color	Darker color causes soils to warm faster; >surface soil temperature advances germination and seedling growth (depending on residue cover)
H ₂ O holding capacity	Holds ~20 times weight in H ₂ O	>H ₂ O holding capacity, >plant-available H ₂ O especially in sandy soils
OM–clay interaction	Cements soil particles into aggregates	Improve soil structure and porosity and enhances gas exchange, infiltration, root proliferation in soil
Ion exchange/buffering	Increases CEC and AEC 20–70%	>Nutrient retention and availability
Mineralization/immobilization	Nutrient cycling	Increases nutrient availability and retains/conserves nutrients
Chelation	OM forms stable metal (M ⁿ⁺) complexes	Enhance micronutrient availability
Solubility	Insoluble humus–clay complexes, many soluble low MW organic compounds	OM-bound nutrients less likely to leach

MW: molecular weight.

minerals are dependent on solution pH. Solubility relationships are particularly important for plant availability of P and many of the micronutrients.

Buffer capacity

As nutrient concentration in the soil solution decreases by plant uptake, nutrients in soil solution are replenished from exchange surfaces, soil OM, or the dissolution of soil minerals (Fig. 2). Nutrient supply to plants depends on nutrient concentration in solution and on *buffering capacity* (BC) or the ability of the soil to resupply nutrients to solution (Fig. 6). For example, as plant roots absorb K⁺, adsorbed K⁺ on the CEC is desorbed to resupply solution K⁺. Similarly, H₂PO₄[−] adsorbed into the AEC will desorb, while P-bearing minerals may also dissolve to supply solution H₂PO₄[−] (Fig. 2). Both processes, desorption from exchange sites and the dissolution of minerals, comprise soil BC.

Nutrient Transport from Soil to Roots

Nutrient absorption by plant roots requires contact between the nutrient ion and the root surface (Fig. 2). Nutrients

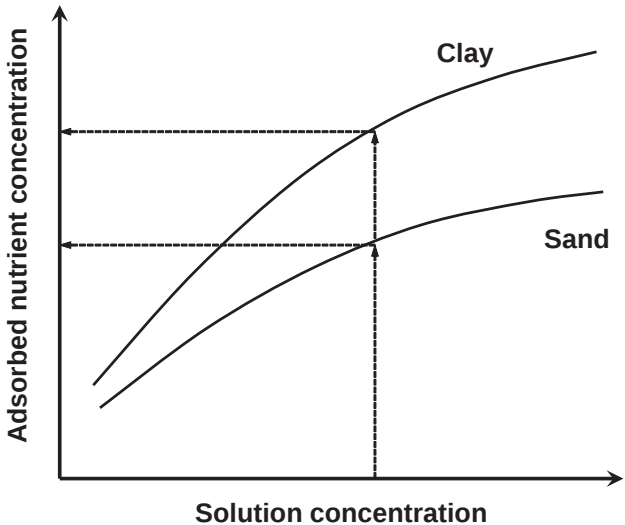


Fig. 6 Influence of soil texture on the quantity of adsorbed and solution nutrient concentrations. Buffer capacity is the ratio of adsorbed to solution nutrient concentration. To maintain the same solution concentration in both soils, the clay soil should have a greater adsorbed nutrient concentration than the sand soil. As solution concentration decreases with the nutrient uptake, the clay soil will have a greater supply of adsorbed nutrients to resupply the soil solution.

reach the root surface by *root interception*, *mass flow*, and *diffusion*. Although each is important, mass flow and diffusion dominate nutrient transport (Table 5).

Root interception represents ion exchange through physical contact between the root and exchange surfaces. Ions

Table 5 Significance of root interception, mass flow, and diffusion in nutrient transport to roots. For some nutrients, the quantity transported can exceed that required by the crop.

Nutrient	Nutrients required (kg ha ^{−1}) for 10 ton ha ^{−1} of corn	Percentage supplied by		
		Root interception	Mass flow	Diffusion
N	200	1	99	0
P	40	2	4	94
K	180	2	20	78
Ca	45	120	440	0
Mg	50	27	280	0
S	22	4	94	2
Cu	0.11	8	400	0
Zn	0.36	25	30	45
B	0.22	8	350	0
Fe	2.2	8	40	52
Mn	9.3	25	130	0
Mo	0.01	8	200	0

Notes: The contribution of diffusion was estimated by the difference between total nutrient needs and amounts supplied by interception and mass flow. If interception + mass flow is >100%, then diffusion = 0.

such as H^+ adsorbed into root surfaces may exchange with ions adsorbed into mineral and OM surfaces through direct contact of both surfaces. As roots and associated mycorrhiza develop, more ions adsorbed into soil surfaces are contacted by increasing root mass. The quantity of nutrients in direct contact with plant roots is the amount in a soil volume equal to the root volume (~2% of soil volume).

Mass flow occurs when ions in soil solution are transported to the root by the transpiration of H_2O by the plant, H_2O evaporation at the soil surface, and percolation of H_2O in the soil profile. The quantity of nutrients transported by mass flow is determined by the rate of H_2O flow and the nutrient concentration in the soil H_2O . Decreasing soil moisture decreases H_2O transport to the root surface.

Diffusion occurs when an ion moves from an area of high to low concentration. As roots absorb nutrients from the soil solution, concentration at the root surface is lower than that in solution not influenced by the root. Therefore, increasing nutrient concentration gradient increases diffusion rate toward the root. A high plant requirement for a nutrient results in a large concentration gradient, favoring a high diffusion rate. Most of the P and K move to the root by diffusion (Table 5).

The importance of diffusion and mass flow in supplying ions to the root surface depends on the ability of the solid phase of the soil to replenish or buffer the soil solution. Ion concentrations are influenced by the types of clay minerals and OM in the soil and the distribution of cations and anions on CEC or AEC.

Mass flow and diffusion processes are also important in nutrient management. Soils that exhibit low diffusion rates because of high BC, low soil moisture, or high clay content may require the application of immobile nutrients near the roots to maximize nutrient availability and plant uptake.

Nutrient Mobility in Soil

Nutrient mobility in soil influences ion transport to plant roots, the evaluation of nutrient availability to plants, and ultimately nutrient management decisions. Nutrient mobility varies between ions, where NO_3^- , SO_4^{2-} , Cl^- , and $H_3BO_3^0$ are not strongly attracted to exchange sites and are soluble in soils, so they can readily move through the root zone with H_2O . As a result, mobile nutrients within the whole soil volume occupied by the plant root system are available for transport to the root in percolating and transpirational H_2O (Fig. 7). The relative mobility of each nutrient will depend on soil pH, temperature, moisture, soil texture, type of clay, and OM content.

Immobile nutrients interact with mineral and OM surfaces, are less soluble, and do not readily move throughout the root zone (Fig. 7). While classified as immobile nutrients in soil, some are more mobile than others. Generally, NH_4^+ , K^+ , Ca^{2+} , and Mg^{+2} are more soluble and mobile than the micronutrient cations and much more mobile than $H_2PO_4^-/HPO_4^{2-}$ and MoO_4^{2-} . As these nutrients are relatively immobile in soil, plant roots access them from a small soil volume surrounding individual roots. Plants create a small zone around the root that has the very low concentration of these immobile nutrients due to plant uptake, causing a concentration gradient encouraging nutrient diffusion. If the soil has a high buffer capacity for an immobile nutrient, then the solution can be replenished and diffusion continues. With a low buffer capacity, solution concentration (and diffusion) ultimately decreases, causing a nutrient deficiency.

Understanding nutrient mobility in soils is essential to managing nutrient applications to maximize plant growth and recovery of applied nutrients by the plant. For example,

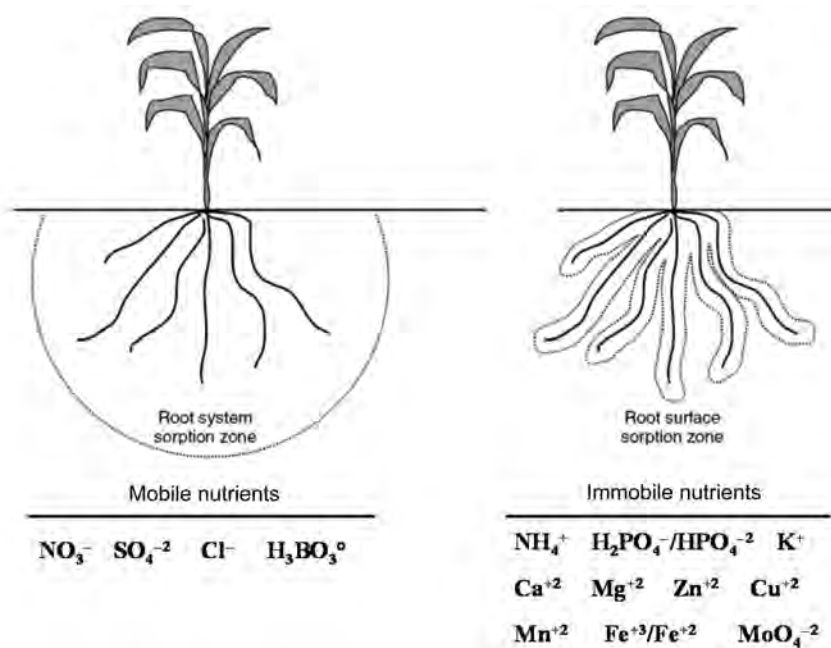


Fig. 7 Relationship between nutrient mobility and nutrient extraction zones. Plants obtain mobile nutrients from the whole soil volume occupied by plants roots. In contrast, plants obtain immobile nutrients from the small soil volume immediately surrounding the plant root.

mobile nutrients can be broadcast or band applied with fairly similar results because of their mobility in soil. However, P is generally placed in concentrated bands because it is generally immobile in soil.

NUTRIENT MANAGEMENT

Efficient nutrient management programs supply plant nutrients in adequate quantities to sustain maximum plant growth and yield while minimizing environmental impacts of nutrient use. Substantial economic and environmental consequences occur when nutrients limit plant productivity or when nutrients are applied in excess of plant requirement.

The essential requirements of an effective nutrient management plan are as follows: 1) assessment of crop nutrient requirement; 2) evaluation of nutrient supplying capacity of soil; 3) quantify optimum nutrient rate; 4) identify appropriate and available nutrient source(s); 5) determine most efficient nutrient placement method; and (6) schedule nutrient applications to meet plant nutrient demand. Management decisions will vary depending on the specific nutrient and crop.

Assessing Plant Nutrient Requirement

The quantity of nutrients required by plants depends on the nutrient (Table 1), plant characteristics (specific crop, yield level, variety, or hybrid), growing season environmental factors (moisture, temperature, etc.), soil characteristics (soil properties, soil fertility, and landscape position), and soil and crop management. Management practices that enhance crop productivity will increase nutrient requirements.

Evaluation of Nutrient Supplying Capacity of the Soil

The *soil testing–nutrient recommendation system* is comprised of the following four consecutive steps: 1) collect a representative soil sample from the field; 2) determine the quantity of plant-available nutrient in the soil sample (soil test); 3) interpret the soil test results (Fig. 8); and 4) estimate the quantity of nutrient required by the crop. The greatest potential for error is in collecting a soil sample that accurately represents the field sampling area. Great care is required in identifying the sampling area from which a composited sample is sent to a laboratory for analysis. Selected parameters used in separating sampling areas within a field include differences in topography, soil type, crop productivity, and past management.

Soil testing is essential for determining the relative availability of plant nutrients in soil. Soil tests extract part of the total nutrient content in soil that is related to (but not

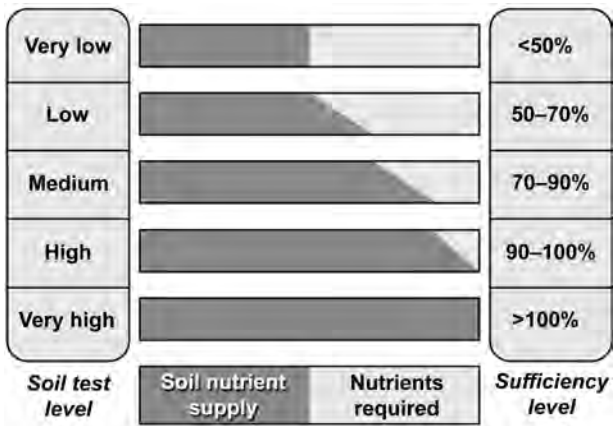


Fig. 8 As soil test levels increase, the capacity of the soil to provide sufficient plant nutrients increases, reducing the quantity of nutrients added to meet crop requirement.

equal to) the quantity of nutrients removed by plants. Thus, the soil test level represents an *index* of nutrient availability representing the relative nutrient sufficiency of a soil (Fig. 8).

Quantify Optimum Nutrient Rate

Soil test interpretation for the purposes of making nutrient recommendations is influenced by the mobility of the nutrient (Fig. 7). With mobile nutrients, crop yield is proportional to total quantity of nutrient present in the root zone because of minimal interaction with soil constituents (Fig. 2). For NO_3^- , SO_4^{2-} , and Cl^- , a 0.5–1.0 m profile sample is important for accurately assessing mobile nutrient availability (Fig. 7). For N, recommendations are usually based on yield goal, where N required to produce each unit of yield is known (e.g., $0.3 \text{ kg N} \cdot \text{kg}^{-1} \text{ grain}$). This concept is also evident when additional in-season N is recommended because better-than-average growing conditions increased the yield potential above initial estimates provided before planting. Additional factors affecting optimum N rate include the estimates of N mineralized during the growing season from previous legume crops and organic N amendments (manures, biosolids, etc.). With immobile nutrients (e.g., H_2PO_4^- , K^+ , and Zn^{2+}), crop yield potential is limited by the quantity of nutrient available at the soil–root interface. Generally, the solution concentrations of immobile nutrients are low, and replenishment occurs through exchange, mineralization, and mineral solubility reactions (Fig. 2). If nutrient uptake demand exceeds the soil’s capacity to replenish solution nutrients, then plant growth and yield will be limited. Immobile nutrient recommendations are based on sufficiency levels determined through soil testing (Fig. 8). Soil tests for immobile nutrients provide an index of nutrient availability that is generally independent of environment.

Identify Nutrient Source(s)

The primary criteria for selecting a nutrient source are availability and cost. For example, if organic nutrient sources (manures, biosolids, etc.) are available, they should be utilized if the cost per unit of plant-available nutrients is comparable to inorganic fertilizers. There are important benefits of using organic sources beyond their nutrient value, which should also be considered (Table 4). Nutrient content of organic materials must be quantified to determine the application rate needed to meet crop nutrient requirement in the year of application.

In most cases, there are few differences in crop response between inorganic fertilizer sources. Although most plants prefer a mixture of NH_4^+ and NO_3^- , most NH_4^+ or NH_4^+ -forming N sources (urea-based products) readily dissolve and nitrify to form NO_3^- . Under the conditions of high denitrification, volatilization, and/or nitrification potential, numerous additives or coatings (solid fertilizers) are available to reduce losses of applied NH_4^+ and NO_3^- . When applying nutrients with the seed, nutrient sources with a low salt index are important in reducing salt damage to germinating seeds and seedlings.

Nutrient Placement Methods

Nutrient placement decisions involve knowledge of crop and soil characteristics, whose interactions determine nutrient availability. Although numerous placement methods have been developed, the primary goal is to ensure optimum nutrient availability from plant emergence to maturity. Vigorous seedling growth (i.e., no early growth stress) is essential to optimize plant growth and yield. Merely applying nutrients does not ensure that they will be absorbed by the plant.

Fertilizer placement options generally involve surface or subsurface applications before, at, or after planting. Placement practices depend on the crop and crop rotation, degree of deficiency or soil test level, nutrient mobility in the soil,

degree of acceptable soil disturbance, and equipment availability.

Preplant application

Preplant applications are important to provide adequate nutrient supply during germination and early plant growth stages. Broadcast nutrients are applied uniformly on the soil surface over the entire area before planting and can be incorporated by tillage (Fig. 9). In no-till, there is limited opportunity for incorporation; thus, broadcast N applications may reduce N recovery by the crop due to enhanced immobilization, denitrification, and volatilization of applied N. Crop recovery of nutrients can be increased by placement below the soil surface where soil moisture might be more favorable for nutrient uptake (Fig. 9). Surface band applied fertilizers can be effective before planting (Fig. 9); however, if not incorporated, dry surface soil conditions can reduce nutrient uptake, especially with immobile nutrients. Surface band N applications can improve N availability compared with broadcast application in some soils and cropping systems due to reduced interaction with surface plant residues.

Animal waste and other organic nutrient sources are commonly preplant applied, usually as a broadcast or subsurface band placement. Broadcast applications, especially unincorporated, are subject to greater denitrification and volatilization N losses and surface runoff losses of all nutrients.

At planting application

Fertilizer placement can occur at numerous locations near the seed (2–8 cm below and/or to side of seed), depending on equipment and crop (Fig. 9). Fertilizer application with the seed is a subsurface band used to enhance early seedling vigor, especially in cold, wet soils. Usually, low nutrient rates are applied to avoid germination or seedling damage.

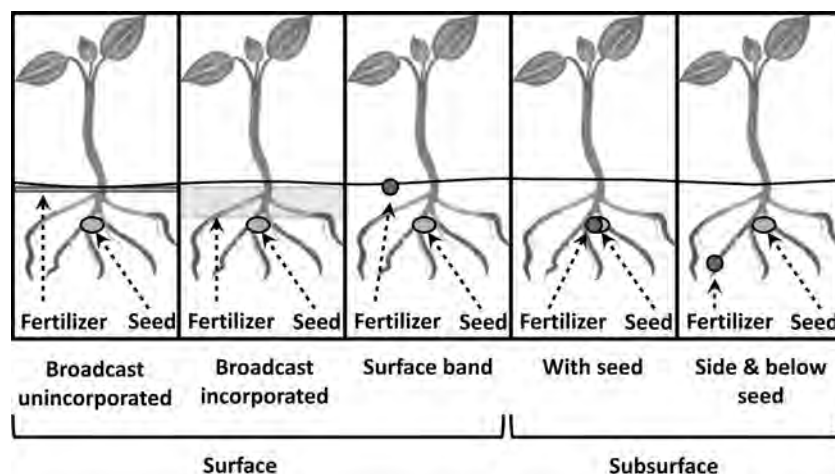


Fig. 9 Options for the application of nutrients at or below the soil surface.

Fertilizers can be surface applied at planting in bands directly over or to the side of the row (Fig. 9).

After planting application

Broadcast application after planting is commonly referred to as topdressing. Topdress N applications are common on permanent or close seeded crops (turf, small grains, pastures, etc.); however, N immobilization with high surface residue reduces the recovery of topdress N. Topdressed P and K are not as effective as preplant applications. Side-dress N application is common in row crops (corn, grain sorghum, etc.) as either a surface or subsurface band. Side-dress applications increase flexibility; since applications can be made almost anytime, equipment is operated without damage to the crop. Side-dress placement is particularly suited for spatially variable N application using remote sensors mounted on the applicator to guide application rate. Subsurface side-dress applications too close to the plant can cause damage by either root pruning or nutrient toxicity. The side-dress application of immobile nutrients (e.g., P and K) is not recommended because most crops need P and K early in the season.

Timing of Nutrient Applications

Nutrients are needed in the greatest quantities during the periods of maximum plant growth. Thus, nutrient management plans are designed to ensure adequate nutrient supply before the exponential growth period. In this case, N should be applied preplant or split applied before the stem extension phase. All of the immobile nutrients, including P and K, should be applied before planting. Knowledge of crop-uptake patterns facilitates improved management for maximum productivity and recovery of applied nutrients.

Variable Nutrient Management

Variable or *site-specific* nutrient management can improve nutrient use efficiency by distributing nutrients based on spatial variation in yield potential, soil test levels, and other spatially variable factors that influence soil nutrient

availability and crop nutrient demand. Preseason and in-season assessments of spatial information are two variable nutrient management approaches used to guide nutrient application decisions.

CONCLUSION

Meeting food needs for a growing population requires increased food production on the same or less agricultural land area. Land managers must adopt economically viable technologies that maintain, enhance, or protect the productive capacity of our soil resources to ensure future food and fiber supplies. While organic nutrient sources are important to meeting the nutritional needs of higher-yielding cropping systems, inorganic fertilizer nutrients will remain the predominant nutrient source. The challenge to the agricultural community is to ensure maximum recovery of applied nutrients, regardless of source, through use of diverse soil, crop, H₂O, nutrient, and other input management technologies to maximize plant productivity. Accomplishing this will significantly reduce nutrient losses to the environment.

REFERENCES

1. Barker, A.V.; Pilbeam, D.J. *Handbook of Plant Nutrition*; CRC Press: Boca Raton, 2007.
2. Grant, C.A.; Milkha, M.S. *Integrated Nutrient Management for Sustainable Crop Production*; The Hanworth Press: NY, 2008.
3. Havlin, J.L.; Tisdale, S.L.; Nelson, W.L.; Beaton, J.D. *Soil Fertility and Fertilizers: An Introduction to Nutrient Management*, 8th Ed.; Pearson Inc.: Upper Saddle River, 2014; 1–516.

BIBLIOGRAPHY

1. Rengel, Z., Ed. *Handbook of Soil Acidity*; Marcel Dekker: New York, 2003.
2. Sparks, D.L. *Environmental Soil Chemistry*; Academic Press: San Diego, 2003.

Nutrients: Deficiency and Toxicity Symptoms

Noel J. Grundon

Land and Water, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Atherton, Queensland, Australia

Abstract

When plants suffer from too much (a toxicity) or too little (a deficiency) of the mineral elements that are their food, they develop visual symptoms on their leaves, stems, fruits, and roots that are characteristics for the deficiency or toxicity. If we can understand these visual symptoms, we are better able to diagnose the causes of the disorders and to commence correcting the problem. In order to understand this visual language of sick plants, it is advantageous to understand the processes that underlie symptom development.

SYMPTOM DEVELOPMENT

Because mineral nutrients are associated with specific metabolic process, the development of symptoms and their location on the plant can be considered in relation to the function and mobility of the mineral elements within the plant.

In Relation to the Functions of Elements

The functions of mineral elements have been covered widely in the literature (see Marschner)^[1] and include the following:

1. Being an integral constituent of an essential metabolite, complex, or macromolecular assembly, e.g., amino acids; proteins (nitrogen and sulfur); enzymes and coenzymes (nitrogen, sulfur, iron, copper, zinc, manganese, nickel, and molybdenum); nucleic acids (DNA and RNA), adenosine diphosphate, and adenosine triphosphate (nitrogen and phosphorus); cell walls (calcium); and chlorophyll (nitrogen and magnesium).
2. Being an activator of an enzyme or regulator of an enzyme-mediated process, e.g., potassium, calcium, magnesium, zinc, manganese, and nickel.
3. Playing a nonstructural role in a physiological process, e.g., protein metabolism; pollen tube growth (boron); auxin metabolism (boron and zinc); membrane integrity and function (phosphorus, calcium, boron, and zinc); photosynthesis (potassium, magnesium, chlorine, and manganese); carbohydrate metabolism and partitioning (potassium, sulfur, magnesium, copper, and zinc); stomatal movement (potassium and chlorine); and messenger in environmental signals (calcium).

Mineral elements may have nonspecific roles in addition to specific roles. For example, the ions of many elements play a role in cation–anion and pH balance, osmoregulation, and maintaining electrical neutrality within cells. Another

example is the quality of harvested products: Calcium deficiency can lead to decreased quality of fruit causing bitter pit in apple, blossom-end rot in tomato, and jelly-fruit in mango.

If an element is involved in the uptake or metabolism of another element, the symptoms may not be clearly defined, and it can be difficult to identify the causal element. For example, nitrogen and sulfur are biochemically related in the formation of proteins. In forest trees, there is no inorganic nitrogen in the foliage, and the close biochemical linkage between organic nitrogen and organic (and total) sulfur causes the rate of nitrogen uptake to be limited by the rate of sulfur accumulation.^[2] Therefore, in many forest and horticultural tree species, the symptoms of mild nitrogen and sulfur deficiency are very similar.^[3]

In Relation to the Mobility of the Elements

Mineral elements differ markedly in their mobility and extent and retranslocation. All mineral elements are highly mobile within the xylem, but their mobility within the phloem can be divided into the following three classes: 1) readily mobile; 2) variably mobile; and 3) poorly mobile. Mineral elements that are readily mobile in the phloem (e.g., nitrogen, potassium, phosphorus, and magnesium) are easily retranslocated from older to younger tissues when nutrient supply becomes limiting. Deficiency symptoms of these elements develop first in the older tissues and advance toward the younger tissues as the deficiency becomes more severe. Nutrients that have low phloem mobility (e.g., calcium, boron, and iron) develop symptoms first in immature tissues. However, elements that are readily mobile under one set of conditions may appear to be only partly mobile or even immobile under different conditions. For example, magnesium is normally classed as a mobile element, and deficiency symptoms appear first on older leaves when a constant but inadequate supply occurs. However, when magnesium supply to the roots of subterranean

clover was interrupted suddenly, symptoms developed first on the younger leaves.^[4]

Some elements show variable mobility depending on the degree of deficiency and either the nitrogen or phosphorus status of the plant. For example, copper, zinc, and molybdenum are relatively immobile when nitrogen supply is adequate, and there is little or no movement of these elements from old leaves of plants deficient in them. Here, symptoms occur mainly in young tissues. However, should nitrogen become limited, copper, zinc, and molybdenum can be remobilized from older to younger leaves along with the retranslocation of nitrogen,^[5] and the symptoms of nitrogen deficiency are likely to dominate.

If a toxic nutrient element interacts with the metabolism of another element, symptoms of the toxicity occur on older tissues, while symptoms of deficiency of the other element occur on those tissues characteristic for that nutrient. For example, on acidic soils, excessive levels of soluble manganese can induce iron deficiency in some plants, thereby causing the development of manganese toxicity symptoms on older leaves and iron deficiency symptoms on younger leaves.^[6]

DIAGNOSTIC SYMPTOMS

Descriptions and color photographs of mineral nutrient deficiencies and toxicities in a range of species have been published (see Grundon et al.^[7] for a comprehensive list). Symptom expression can vary greatly from one species to another and also among cultivars within species. It is most important to check the specific symptoms described for a particular species. However, the first requirement in the use of visual symptoms is knowledge of the appearance of a healthy plant of the species concerned. Knowledge of the symptoms caused by insect pests and diseases also aids in the final diagnosis.

Deficiency

Because one element may have many functions, the characteristic symptoms of a deficiency are those associated with the function that is most sensitive or dominates. For example, zinc is associated with auxin metabolism, nucleotide synthesis, and membrane integrity, but it is an impairment of auxin metabolism in zinc-deficient plants that leads to the characteristic symptoms of leaf distortion and failure of the internodes to elongate. General symptoms of deficiency of some nutrient elements are listed in Table 1.

Toxicity

Symptoms of nutrient toxicity occur more commonly in those tissues that have had a longer period of time to accumulate ions, e.g., the margins of older leaves in dicotyledons or the leaf tips in monocotyledons and gymnosperms.

Table 1 General symptoms of mineral element deficiency (note that symptoms for specific crops and cultivars may differ).

Element	General symptoms
Nitrogen	Pale yellow chlorosis of the older leaves spreading to younger leaves, often with reddening in cold weather
Phosphorus	Dark green foliage; reddening or purpling of older leaves spreading to younger leaves
Potassium	Near-marginal to marginal chlorosis and necrosis of older leaves
Calcium	Death of the terminal meristems; distorted growth or death of leaf tips; disorders of fruit (e.g., blossom-end rot of tomato; bitter pit in apple; jelly-fruit in mango)
Magnesium	Marginal or interveinal chlorosis and necrosis of older leaves
Sulfur	Pale yellow chlorosis of immature leaves
Iron	Interveinal chlorosis or bleaching of immature leaves
Copper	Distorted growth of immature leaves or death of leaf tips; failure of fertilization and fruit set
Zinc	Little leaf, rosetting, and shortening of the internodes
Manganese	Interveinal chlorosis, necrotic spots, or streaks on older or middle leaves
Boron	Death of growing points; failure of fertilization and fruit set
Nickel	Interveinal chlorosis (monocotyledons) or leaf tip necrosis (dicotyledons)
Molybdenum	General chlorosis (legumes); mottled pale appearance (nonlegumes)

Source: From Dell, Malajczuk, et al.,^[3] Grundon, Edwards, et al.,^[6] Blamey, Edwards, et al.,^[8] and Grundon.^[9]

Thus, marginal chlorosis and necrosis in older leaves are common symptoms of many nutrient toxicities.^[6,8,9] General symptoms of toxicity of some mineral elements are listed in Table 2.

Advantages, Disadvantages, and Limitations

The use of visual symptoms to diagnose nutrient disorders has distinct advantages, including: 1) the technique can be applied in the field; and 2) it is not dependent on costly laboratory support services. However, even in the hands of experienced practitioners, there are a number of major disadvantages in relying on visual symptoms as the sole diagnostic tool, including the following:

1. When the disorder is mild or transient in nature, no foliar symptoms may be produced, and clearly defined symptoms may occur only when the disorder is severe enough to depress yield.

Table 2 General symptoms of mineral element toxicity (note that symptoms for specific crops and cultivars may differ).

Nutrient	Visual symptoms
Nitrate-nitrogen	Marginal scorch of leaves followed by interveinal collapse
Ammonium-nitrogen	Necrosis and blackening of margins of leaves or leaf tips
Phosphorus	Marginal scorch and interveinal necrosis and death of older leaves
Sodium	Marginal scorch and necrosis of older leaves
Sulfate	Marginal scorch and necrosis of older leaves
Chloride	Bronzed chlorosis and marginal scorch of older leaves
Manganese	Marginal chlorosis and brown necrotic peppering on older leaves
Aluminum	Roots stunted, brown, and multibranched; symptoms on shoots may resemble phosphorus deficiency
Zinc	Interveinal necrosis on older leaves
Boron	Interveinal chlorosis becoming marginal necrosis on older leaves
Nickel	Transverse light and dark bands on immature leaves

Source: From Dell, Malajczuk, et al.,^[3] Grundon, Edwards, et al.,^[6] Blamey, Edwards, et al.,^[8] and Grundon.^[9]

- For some micronutrient disorders, environmental factors (e.g., light and zinc deficiency symptoms) have a profound effect on the appearance or severity of the symptoms.
- Under certain conditions, different nutrient disorders can produce similar symptoms.
- By the time definitive symptoms become evident, it is often too late to correct the problem in that growing season.

The Final Diagnosis

Despite the limitations to its application, the use of visual symptoms are a valuable diagnostic tool. It is important to recognize that the use of visual symptoms provides a preliminary diagnosis only. Confirmation by other methods such as plant and soil analyses, pot culture assays, field experiments, or test strips is an essential second step.

REFERENCES

- Marschner, H., Ed. Functions of mineral nutrients: Macronutrients. In *Mineral Nutrition of Higher Plants*, 2nd Ed.; Academic Press: London, 1995; 229–404.
- Turner, J.; Lambert, M.J. Nutrition and nutritional relationships in *Pinus radiata*. *Annu. Rev. Ecol. Syst.* **1986**, *17*, 325–350.
- Dell, B.; Malajczuk, N.; Grove, T.S. Atlas of deficiencies. In *Nutrient Disorders in Plantation Eucalypts*; ACIAR Monograph 31; Australian Centre for International Agricultural Research: Canberra, 1995; 9–92.
- Scott, B.J.; Robson, A.D. Distribution of magnesium in subterranean clover (*Trifolium subterraneum* L.) in relation to supply. *Aust. J. Agr. Res.* **1990**, *41*, 499–510.
- Hill, J.; Robson, A.D.; Loneragan, J.F. The effects of copper and nitrogen supply on the retranslocation of copper in four cultivars of wheat. *Aust. J. Agr. Res.* **1978**, *29*, 925–939.
- Grundon, N.J.; Edwards, D.G.; Takkar, P.N.; Asher, C.J.; Clark, R.B. *Nutritional Disorders of Grain Sorghum*; ACIAR Monograph 2; Australian Centre for International Agricultural Research: Canberra, 1987; 99 pp.
- Grundon, N.J.; Robson, A.D.; Lambert, M.J.; Snowball, K. Nutrient deficiency and toxicity symptoms. In *Plant Analysis: An Interpretation Manual*, 2nd Ed.; Reuter, D.J., Robinson, J.B., Eds.; CSIRO Publishing: Collingswood, 1997; 37–51.
- Blamey, F.P.C.; Edwards, D.G.; Asher, C.J. *Nutritional Disorders of Sunflower*; Department of Agriculture, The University of Queensland: St. Lucia, 1987; 72 pp.
- Grundon, N.J. *Hungry Crops: A Guide to Nutrient Deficiencies in Field Crops*; Queensland Department of Primary Industries: Brisbane, 1987; 246 pp.

Nutrients: Diffusion, Bioavailability, and Plant Uptake

Niels Erik Nielsen

Plant Nutrition and Soil Fertility Lab, Department of Agricultural Sciences, Royal Veterinary and Agricultural University, Frederiksberg, Denmark

Abstract

Entry of nutrient elements into plants and, therefore, into the food web of human beings depends on the capability of the soil to release and maintain a concentration higher than the minimum concentration ($c_{\min}$) of elements in the soil solution at the root surface, and on the uptake capacity of the plant roots. Root-induced rhizosphere processes influence nutrient availability, maintenance of the nutrient concentration in the soil solution and plant uptake. The entry into plants and the associated flux of a nutrient element toward root surface may be controlled by a number of rate-limiting and/or rate-determining processes. One of the rate-limiting processes may be the flux by diffusion from the solid constituents in the soil via the soil solution to the nutrient-absorbing cell membrane in root tissue near the root surface. The goal of good agronomy is always to obtain a useful and sustainable modification of rate-limiting and/or rate-determining processes in the soil—plant atmosphere system, aiming at high yields and qualities of the crops. This goal requires identification of the rate limiting and/or rate-determining steps, an understanding of their dynamics, and knowledge of how to obtain appropriate modifications of these steps.

SOIL PLANT SYSTEM

The movement of any nutrient element, M , from solid soil constituents to the root surface, and its entry into plants, can be divided into a sequence of processes (steps), as shown in Fig. 1, which also indicates major agronomical actions used to improve nutrition and growth of crop plants. The $\leftrightarrow$ denotes solid-phase processes slowly approaching equilibrium, or microbial-mediated net mineralization of nitrogen, sulfur, and phosphorus, for example. Also denoted are the source/sink processes by which diffusible nutrients are being produced or removed by chemical, physical, and biological transformation processes. In Fig. 1, L denotes ligands that are any dissolved solute reacting with M to form ML , which are organic complexes and ion pairs dissolved in the soil solution. The occurrence of L and, therefore, of ML , increases the total concentration ($M + ML$) and mobility of the nutrient element. The $\rightleftharpoons$ denotes reversible processes, which are spontaneously approaching equilibrium. Depending on ion species, ion concentration at the root surface, and plant age, the symbol $\rightleftharpoons$ denotes processes that may be irreversible. The irreversible processes are always rate limiting and/or rate determining, whereas reversible processes may be rate limiting, only. Processes 2 and 3 are in the vicinity of the soil particle, whereas process 7 is in the vicinity of the root. Processes 4, 5, and 6 are transport processes by mass flow and diffusion due to water uptake and nutrient uptake (process 8) by cell membranes of root cells near the root surface or root hairs. Process 9 is the nutrient translocation in the plant. Process 10 is the plant growth that also integrates the absorbed nutrient into the plant tissue. Processes 8, 9,

and/or 10 create the concentration (electro-chemical) gradients for irreversible net flux of nutrients from the soil–soil solution system into the plants. Hence, at any time, the rate-determining processes (Fig. 1) are then either the release of the nutrient into the pool of plant available nutrient (source/sink processes in the soil) or the nutrient uptake into the roots, its translocation/circulation in the plant, and/or the rate by which the nutrient is built into new tissue. Mass flow and diffusion may be rate limiting only, and not rate determining. Usually only a small fraction of the plant-available nutrients is dissolved in the soil solution. This implies that the bioavailability of nutrients to plant roots is governed by several soil properties including, e.g., the characteristics of process 2 in Fig. 1 and the possibilities for movement via soil solution to the root surface by mass flow and diffusion. The concept of a *bioavailable nutrient* can then be defined as a nutrient element that is present in a pool of diffusible (available) nutrients that are close enough to arrive at water- and/or nutrient-absorbing root surfaces during a period of 10 days, for example. This seems to fit with the observation that most of the depletion zones of slowly moving nutrients, such as phosphorus, are created during the first 10 days after root growth into a new soil volume unit.^[1] The bioavailable quantity of a nutrient in the soil is affected by at least five different groups of processes as indicated in Table 1.

DIFFUSION

Diffusion is the net movement of a solute or a gas from a region that has a higher concentration, to an adjacent region

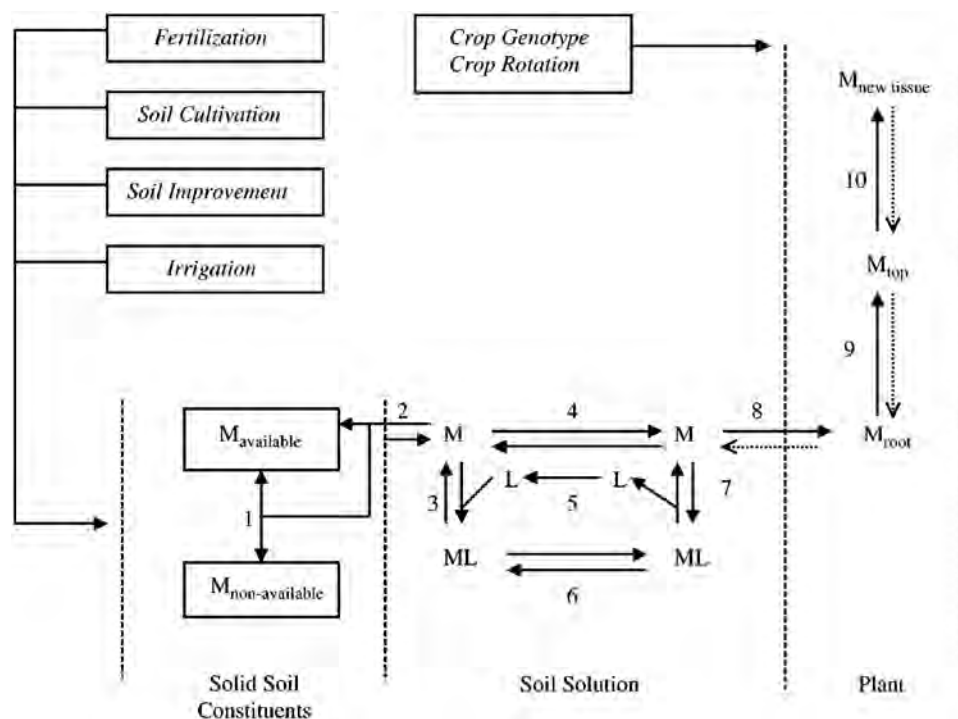


Fig. 1 Flux of a nutrient element in the soil plant atmosphere system and agronomy actions; nutrient element (M), ligands (L), reversible $\rightleftharpoons$ and irreversible processes.

Source: From Nielsen.^[2]

that has a lower concentration. Diffusion is a result of the random thermal motions of molecules in the considered solids, solution, or air. The net movement caused by diffusion is a statistical phenomenon because the probability of

the molecules' movement from the concentrated to the diluted region is greater than vice versa. Fick (1855) was one of the first to examine diffusion on a quantitative basis. The basic equation Eq. 1 to express diffusion, e.g., to a root is known as Fick's first law of diffusion:

$$F = -D \frac{\partial C}{\partial r} \quad (1)$$

in which F is the diffusive flux ($\text{mol cm}^{-2}\text{s}^{-1}$) of a nutrient in the r -direction normal to the root cylinder. The driving force (the gradient in the electro-chemical potential) is, in most cases, approximated by the concentration gradient

$$\frac{\partial C}{\partial r} \text{ (mol cm}^{-4}\text{)}$$

and D is the diffusion coefficient (cm^2s^{-1}). They way to describe diffusion processes mathematically under various conditions has been presented by Jost^[3] and Crank.^[4] Great contributions to our understanding of solute movement in the soil root system by mass flow and diffusion have been given or reviewed by Nye and Tinker^[5,6] and Barber.^[7] Willigen and colleagues^[8] reviewed some aspects of the modeling of nutrient and water uptake by plant roots.

Nutrient Movement by Mass Flow and Diffusion from Soil to Plant Roots

Nutrients bound to the solid-soil phase are virtually immobile in the sense of its movements to roots. The nutrient has to be released into the soil solution as shown in Fig. 1. Furthermore, contact between the root

Table 1 Processes and factors involved in nutrient transfer from soil to plant roots.

Process	Factors
Release, mineralization, and dissolvment of the nutrient in the soil solution	Chemical and physical properties of the solid phases, temperature, soil water content, and activity of the microbial biomass
Root development	Root length, distribution of roots in the root zone, root morphology, root hairs, rate of root growth, and root surface contact area with soil solution
Solute movement by mass flow and diffusion to roots	Transpiration rate (w_o), concentration of the nutrient in the soil solution (c_b), effective diffusion coefficient (D_e), nutrient buffer power of the soil (b)
Rhizosphere processes, increasing the rate of nutrient release in the soil	Depletion of the soil solution for nutrients by the roots; root exudates as protons, reducing agents, chelates, organic anions, enzymes; modification of microbial activity; mycorrhizal and Rhizobium symbioses
Nutrient uptake	Concentration of the nutrient at the root surface (c_o); transport kinetic parameters of nutrient uptake by the roots ($\bar{I}_{\max}$, K_m , and $c_{\min}$)

Source: From Nielsen.^[2]

and nutrient-absorbing membranes in the root tissue near the root surface and the soil solution is a prerequisite for nutrient uptake. Contact to nutrient pools can be brought about by two means as follows: 1) growth of roots to the sites where nutrient pool are located (root interception) and 2) movement of the nutrient from the bulk (the pool) of the soil to the root surface. Even so, nutrients may at any time move over a certain distance in the soil solution and cell wall before they reach the outside cell membrane of a root hair or a root cortical cell for uptake. The mechanisms for these transports are mass flow and diffusion.^[5,9] The driving force for the net movement of nutrients (Fig. 1) is the water and the selective nutrient uptake by the plant root, creating a concentration gradient (dc/dr). The general equation of continuity (mass balance) used to describe movements in a direction normal to a root cylinder at radial distance r and time t may partly be developed from Eq. 1, extended and expressed as

$$\left[b \frac{\partial c}{\partial t} \right]_r = - \left[\frac{1}{r} \frac{\partial r F_T}{\partial r} \right]_t + U_{r,t} \quad (2)$$

in which

1. U is the production/consumption term (mole $\text{cm}^{-3} \text{s}^{-1}$) at r (radial distance from the center of the root) and t (time).
2. b is the buffer power (dC/dc) in which C is the total concentration of diffusible solute in the soil and c is the concentration of solute in the soil solution.
3. F_T is total net flux of solute by mass flow and diffusion (mole $\text{cm}^{-2} \text{s}^{-1}$).

$$F_T = F_m + F_d$$

where $F_m = wc$ in which w is the flux of soil solution in the direction of the root ($\text{cm}^3 \text{cm}^{-2} \text{s}^{-1}$) and c is the nutrient concentration of soil solution (mole cm^{-3}). The expected rates of water flux at the root surface are $0.2\text{--}1 \cdot 10^{-6} \text{cm s}^{-1}$.^[7] The flux, F_d , by diffusion can be expressed by Fick's first law

$$F_d = -D_e b \frac{dc}{dr}$$

The $b = dC/dc$ is the soil buffer power defined previously. The C is the sum of the amount of nutrient in the soil solution and the amount of adsorbed nutrient that is able to replenish the nutrient in the soil solution spontaneously. Hence, b is the parameter mediating the effects of the soil chemical conditions on nutrient uptake by plants. D_e denotes the effective diffusion coefficient in the soil. D_e differs between media, but it can be related to the diffusion coefficient D_o for the nutrient in free soil solution. The influences of soil on diffusion, and thereby the relation between D_e and D_o , can be expressed by Eq. 3:^[10]

$$D_e = D_o \theta f / b \quad (3)$$

where θ is the volumetric water content expressed as a fraction, and f is the impedance factor that essentially allows for the increase in the actual diffusion distance because of the tortuous pathway of water filled soil pores and water films. The volumetric water content that allows a reasonable root activity is between 0.1 and 0.4. The value of f increases with increase in water content,^[11] whereas the buffer power remains constant with changes in soil moisture at the same bulk density.^[12] It has been observed^[13] that the relation between f and θ can be expressed empirically by $f = 1.58\theta - 0.17$ for $\theta > \text{about } 0.11$. From this it may be estimated that D_e decreases about 18 times if θ decreases from 0.40 to 0.15. Hence D_e is the parameter mediating the effects of soil moisture, soil chemical, and soil physical conditions on diffusion in soil.

Almost all studies on solute movement in the soil plant system neglect U in Eq. 2 because our understanding of the biology caused by root-induced processes and its effects on production or consumption of available nutrients is incomplete as yet.

To solve Eq. 2 for a given soil plant system is a complicated process, and in most cases, difficult or even impossible because of the lack of information on the soil root interactions and root behavior. The method for obtaining analytical and numerical solutions of Eq. 2 under a number of often simplified soil plant conditions has been summarized.^[6-8] However, illustration of the importance of diffusion for the bioavailability of nutrients in soils may be based on Eq. 4:

$$\Delta r = \sqrt{2D_e t} \quad (4)$$

in which Δr is the average distance of diffusion, e.g., in a direction normal to a root. The mathematics behind Eq. 4 has been presented by Jost.^[3] Based on Eq. 4, the equivalent soil volume (V in cm^3) of soil depleted for diffusible (available) nutrients—the quantity of bioavailable nutrients—can then be estimated as follows (Eq. 5) for roots without root hairs:

$$V = \pi(\Delta r + r_o)^2 L_v \quad (5)$$

in which Δr is estimated from Eq. 4, r_o is the root radius and L_v is the root density in cm cm^{-3} of soil. The data in Table 2 show the expected effective diffusion coefficient of a number of plant nutrients in soil and corresponding influences on the nutrient bioavailability; in addition, the data show how a decrease of the soil moisture from $\theta = 0.40$ to 0.15 affects the bioavailability at a root density of 5 cm^{-2} of roots without root hairs. It may be calculated from $f = 1.58\theta - 0.17$ and Eq. 3 that the diffusive flux decreases by a factor of $18 = D_e^{\theta=0.40} / D_e^{\theta=0.15}$. At field capacity of water content, the expected, effective diffusion coefficient of nitrate in soil is $10^{-6} \text{cm}^2 \text{s}^{-1}$. This is almost 10 times slower than in pure water. Hence, a pored media, such as soil, physically decrease the possibility for solute movement with a factor of nearly 10. As the soil dries out, this

Table 2 Expected effective diffusion coefficients of some nutrients in soil at field capacity of water content (e.g., $\theta = 0.40$), and estimated bioavailability as a fraction of diffusible (available) nutrient (Eqs. 4 and 5) at a root density ($L_v = 5 \text{ cm}^{-2}$) of roots without root hairs (mean root radius, $r = 0.01 \text{ cm}$; time, $t = 10$ days).

Element	$D_e^a \text{ (cm}^2 \text{ s}^{-1}\text{)}$	Bioavailable nutrient as a fraction of available nutrient ($V, \text{ cm}^3$)	
		$\theta = 0.40$	$\theta = 0.15$
Nitrate	1×10^{-6}	27.558	1.574
Potassium	1×10^{-7}	2.847	0.180
Boron	1×10^{-7}	2.847	0.180
Magnesium	1×10^{-8}	0.314	0.026
Calcium	1×10^{-8}	0.314	0.026
Phosphorus	1×10^{-9}	0.042	0.006
Manganese	1×10^{-9}	0.042	0.006
Molybdenum	1×10^{-9}	0.042	0.006
Zinc	1×10^{-9}	0.042	0.006
Iron	1×10^{-10}	0.008	0.003

Source: From ^aValues obtained from Nielsen^[2] and Barber.^[7]

factor increases as shown in Fig. 2. Apart from nitrate and chlorine, nutrient elements are adsorbed more or less to the solid soil constituents. This is the main cause of the decrease of the diffusion coefficients below $10^{-6} \text{ cm}^2 \text{ s}^{-1}$. The diffusion coefficient of phosphorus is as low as $10^{-9} \text{ cm}^2 \text{ s}^{-1}$ mainly because approximately 0.1% of the diffusible (available) phosphorus is dissolved in the soil solution only. This has a large effect on the bioavailability of the plant-available quantities of the various nutrient elements in soil as illustrated in columns 3 and 4 of Table 2. The V values ≥ 1 indicate that the root at a density of 5 cm^{-2} is able to deplete all the available nutrient as seen for nitrate, even under dry conditions, whereas only 4% of the available phosphorus is bioavailable inside

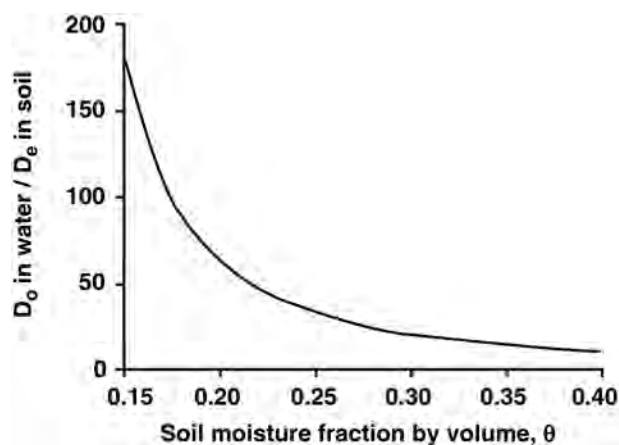


Fig. 2 Effects of soil and soil moisture on nutrient element mobility.

a period of 10 days. If the soil dries out to $\theta = 0.15$, the bioavailability decreases to only 0.6%. This illustrates that the decrease of soil moisture may create nutrient deficiency even in soil with high phosphorus fertility. However, phosphorus uptake is increased by the activity of root hairs (discussed in the following).

IMPORTANCE OF ROOT HAIRS

Root hairs are outgrowths from specialized root epidermal cells (trichoblasts). Root hair length, diameter, and number per unit length of root vary among plant species and genotypes within the same species.^[6,7,14] Frequency and size of root hairs are affected by many environmental factors, as well. In nature, the length of root hairs varies from 0.01 to 0.15 cm, the radius varies from 0.0005 to 0.002 cm, and the number per unit of length varies from 100 to 1000 per cm root. The importance of root hairs for phosphorus uptake has been demonstrated directly in the laboratory^[15,16] and under field conditions.^[14] It is reasonable to assume that the clusters of root hairs' outer tips form a fairly well-defined cylinder to which phosphorus diffuses, on average, a distance Δr in 10 days, and that root hair density and its period of function are long enough to withdraw the entire available nutrient in the soil penetrated by root hairs. The bioavailability of phosphorus, e.g., as affected by root hair length, can then be estimated from the following extension of Eq. 10:

$$V = \pi(\Delta r + \sigma + r_o)^2 L_v \quad (6)$$

in which σ is the root hair length in cm.

Fig. 3 shows that the bioavailability of phosphorus increases exponentially with the root hair length. Hence, root hairs play a very important role for the bioavailability of nutrients having a low effective diffusion coefficient (D_e) in soil.

BOUNDARY CONDITIONS AND NUTRIENT ENTRY

If depletion zones around the roots do not overlap, the solute concentration converges to the solute concentration c_b in the bulk solution, at which $F_T = F_m + F_d = 0$. The boundary conditions at the surface of the root are

$$F_T = F_m + F_d = \alpha c_o \quad (7)$$

in which $\alpha \text{ (cm s}^{-1}\text{)}$ is the root-absorbing power defined by Tinker and Nye^[6] and c_o is the concentration of solute at the root surface. It can be learned from Eq. 7 that the actual concentration (c_o) at the root surface and, therefore, the rate of nutrient flow per unit length of root, is determined by the ratio F_T/α . The kinetics of net uptake of nutrients^[7,17-19] may be expressed by:

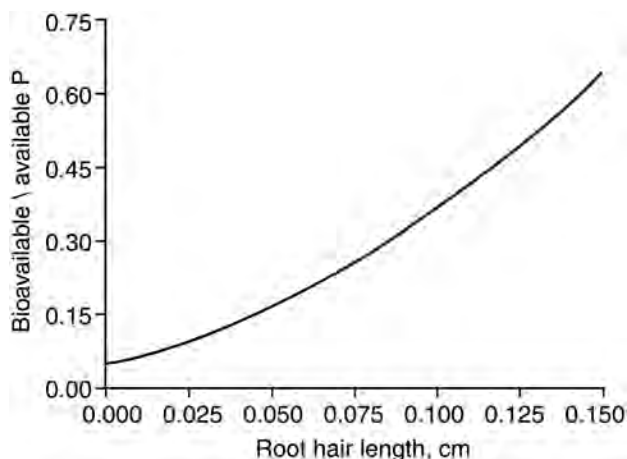


Fig. 3 The effect of root hair length on phosphorus bioavailability. Soil moisture ($\theta = 0.4$); root density 5 cm^{-2} ; root radius 0.01 cm .

$$\overline{\ln L^*} = \frac{L^* \bar{I}_{\max} (c_o - c_{\min})}{K_m + c_o - c_{\min}} \quad (8)$$

in which L^* is the root length per unit of plant biomass, $\bar{I}_{\max}$ ($\text{mole cm}^{-1} \text{ s}^{-1}$) is the mean maximal net influx, K_m (mole cm^{-3}) is the Michaelis–Menten factor, c_o is the concentration of the nutrient at the root surface, and $c_{\min}$ is the nutrient concentration at which $\bar{\ln} = 0$. The values of the parameters $\bar{I}_{\max}$, K_m , and $c_{\min}$ vary according to the plant nutrient, temperature, and plant species/genotype and plant age. Furthermore, kinetics of nutrient uptake by roots may be influenced by ion interactions. Determined values of L^* , $\bar{I}_{\max}$, K_m , and $c_{\min}$ for uptake of several nutrients by several plant species or genotypes, obtained under conditions in which the rate-determining step of nutrient uptake was located in the roots, have been noted.^[20] The data show that the values of L^* , $\bar{I}_{\max}$, K_m , and $c_{\min}$ vary considerably among nutrients and among plant species and genotypes. This illustrates the efficiency by which these plants utilize soil as a source of nutrients. It is possible from $F_T = F_m + F_d = \alpha c_o$ and Eq. 8 to develop how α varies at varying solute concentration at the root surface by:

$$\alpha = \frac{\bar{I}_{\max} (c_o - c_{\min})}{2\pi r_o c_o (K_m + c_o - c_{\min})} \quad (9)$$

Figs. 4A and 4B illustrate the variation of α for phosphorus uptake at low concentration (c_o) at the root surface of some plant species and barley genotypes. Hence, the root-absorbing power (α) varies also at low concentration (c_o) of solute at the root surface. This implies that phosphorus uptake at low P concentration is more under the control of the plant parameters determining the size of α than under the control of P diffusion in the soil, whereas at the range of c_o at which α has achieved its maximum, uptake is controlled by diffusion.

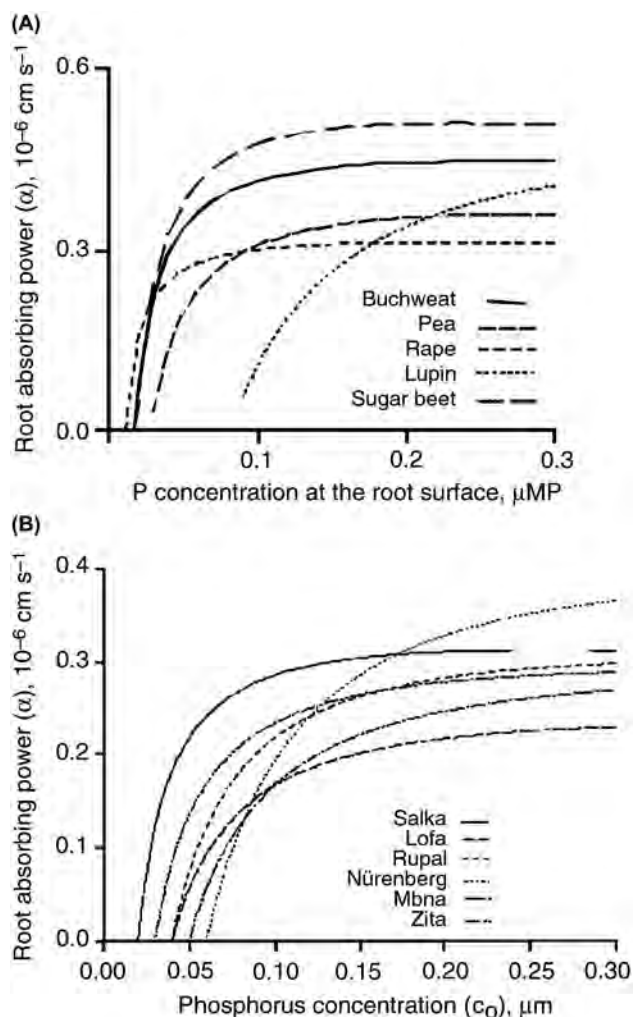


Fig. 4 Absorption power (α) at varying phosphorus concentration at the root surface of some plant species (A) and barley genotypes (B).

Source: From Nielsen.^[2,20]

CONCLUSION

Even though the movement—and the main factors affecting the movement—of nutrient elements to root by mass flow and diffusion is well known, the effect of soil conditions on crop growth is not properly understood. It is obvious that the big variation (Table 2) of the effective diffusion coefficient (D_e), caused mainly by the variation of the soil chemistry of the various nutrient elements, has a large impact on the bioavailability of nutrient elements. The mobility of phosphorus and micronutrients is so low in most soils that the soil exploited by root hairs is the main source of these elements. The root-induced modifications to the soil in the rhizosphere would then have a considerable impact on the efficiency by which plants use the rhizosphere soil as a source of nutrients. The understanding of how root-induced processes accelerate solute

movement and the transformation of non-available nutrients to bioavailable nutrients is increasing.^[6,21,22] Root hair length and root-induced processes appear to vary between genotypes of our crop plants.^[14] Hence, improvement of the efficiency by which plants use soil as a source of nutrients seems to be a possibility by targeted plant breeding.

REFERENCES

- Gahoonia, T.S.; Raza, S.; Nielsen, N.E. Phosphorus depletion in the rhizosphere as influenced by soil moisture. *Plant Soil* **1994**, *159*, 213–218.
- Nielsen, N.E. Bioavailability of nutrients in soil. In *Roots and Nitrogen in Cropping Systems of the Semi-Arid Tropics*; Ito, O., Johansen, C., Adu-Gyamfi, J.J., Katayama, K., Kumar Rao, J.V.D.K., Rego, T.J., Eds.; International Crop Research Institute for the Semi-Arid Tropics: India, 1996; 411–427.
- Jost, W. *Diffusion in Solids, Liquids and Gases*; Academic Press: New York, 1960; 49.
- Crank, J. *The Mathematics of Diffusion*, 2nd Ed.; Clarendon Press: Oxford, 1975.
- Nye, P.H.; Tinker, P.B. *Solute Movement in the Soil-Root System*; Blackwell: Oxford, 1977.
- Tinker, P.B.; Nye, P.H. *Solute Movement in the Rhizosphere*; Oxford University Press: Oxford, 2000.
- Barber, S.A. *Soil Nutrient Bioavailability*, 2nd Ed.; Wiley: New York, 1995.
- Willigen, P.; Nielsen, N.E.; Claassen, N.; Castrignano, A.M. Modelling water and nutrient uptake. In *Root Methods a Handbook*; Smit, A.L., Bengough, A.G., Engels, C., Noordwijk, M., Pellerin, S., Geijn, S.C., Eds.; Springer: Berlin, 2000; 511–543.
- Barber, S.A. A diffusion and mass-flow concept of soil nutrient availability. *Soil Sci.* **1962**, *93*, 39–49.
- Nye, P.H. The measurement and mechanism of ion diffusion in soil. I. The relation between self-diffusion and bulk diffusion. *J. Soil Sci.* **1966**, *17*, 16–23.
- Rowell, D.L.; Martin, M.W.; Nye, P.H. The measurement and mechanisms of ion diffusion in soils. III. The effect of moisture content and soil solution concentration on the self-diffusion of ions in soils. *J. Soil Sci.* **1967**, *18*, 204–222.
- Bhadoria, P.B.S.; Classen, J.; Jungk, A. Phosphate diffusion coefficient in soil as affected by bulk density and water content. *Z. Pflanzenernaehr. Bodenk.* **1991**, *154*, 53–57.
- Barracough, P.B.; Tinker, P.B. The determination of ionic diffusion coefficients in field soils. 1. Diffusion coefficient in sieved soils in relation to water content and bulk density. *J. Soil Sci.* **1981**, *32*, 225–236.
- Gahoonia, T.S.; Nielsen, N.E.; Lyshede, O.B. Phosphorus acquisition of cereal cultivars in the field at three levels of P fertilization. *Plant Soil* **1999**, *211*, 269–281.
- Barley, K.P.; Rovira, A.D. The influence of root hairs on the uptake of phosphorus. *Commun. Soil Sci. Plant Anal.* **1970**, *1*, 287–292.
- Gahoonia, T.S.; Nielsen, N.E. Direct evidence on the participation of phosphorus (P^{32}) uptake from soil. *Plant Soil* **1998**, *198*, 147–152.
- Classen, N.; Barber, S.A. A method for characterizing the relation between nutrient concentration and flux into roots of intact plants. *Plant Physiol.* **1974**, *54*, 564–568.
- Nielsen, N.E. A transport kinetic concept for ion uptake by plants. III. Test of the concept by results from water culture and pot experiments. *Plant Soil* **1976**, *45*, 659–677.
- Nielsen, N.E.; Barber, S.A. Differences among genotypes of corn in the kinetics of P uptake. *Agron. J.* **1978**, *70* (5), 695–698.
- Nielsen, N.E. Bioavailability, cycling and balances of nutrient in the soil plant system. In *Integrated Plant Nutrition Systems*; Dudal, R., Roy, R.N., Eds.; Fertilizer and Plant Nutrition Bulletin 12; FAO: Rome, 1995; 333–348.
- Jungk, A.; Classen, N. Ion diffusion in the soil-root system. *Adv. Agron.* **1997**, *61*, 53–110.
- Hensinger, P. How do plant roots acquire mineral nutrients? Chemical processes involved in the rhizosphere. *Adv. Agron.* **1998**, *64*, 225–265.

Nutrients: Interactions in Soil—Plant System

Fusuo S. Zhang

College of Resources and Environmental Sciences, China Agricultural University, Beijing, China

J. Shen

Department of Plant Nutrition, China Agricultural University, Beijing, China

Y.-G. Zhu

Research Centre for Eco-Environmental Sciences, Chinese Academy of Sciences, Beijing, China

Abstract

The soil–plant system is one of the most important components in agricultural and natural ecosystems. Nutrient dynamics in soil–plant systems not only reflect the pattern of nutrient flow but also influence food production and quality and the contaminant pathways in agricultural and natural ecosystems. In soil–plant systems, interactions between nutrients occur when the supply of one nutrient affects the movement, absorption, or utilization of another nutrient within the soil, soil–root interface (rhizosphere), or plant. There are several mechanisms of nutrient interaction in soil–plant systems. In this review, some specific examples are chosen to illustrate the mechanisms and effects of nutrient interactions, which include in the following: 1) nutrient interactions in soil; 2) nutrient interactions in the rhizosphere; and 3) nutrient interactions in plants.

NUTRIENT INTERACTIONS IN SOIL

Interactions between soil nutrients can occur through several soil physicochemical processes, particularly adsorption/desorption, dissolution/precipitation, and mineralization/immobilization. In these processes, nutrient interactions affect plant growth by affecting nutrient bioavailability. Adsorption and precipitation are the two key processes affecting nutrient interactions in soil.

Soil pH can affect the adsorption of nutrient ions and then affect nutrient uptake. The application of calcium carbonate (CaCO_3) can markedly increase the uptake of *molybdate* (Mo) by plants because the adsorption of MoO_4^{2-} by soils and by aluminum (Al) and iron (Fe) oxides decreases as pH increases due to CaCO_3 application. Additionally, interactions can also occur through the effects of one ion on the adsorption of another. For example, CaSiO_3 may displace dihydrogen phosphate (H_2PO_4^-) adsorbed onto Al and Fe oxides in soil. The application of H_2PO_4^- can increase Mo concentration in soil solutions by displacing MoO_4^{2-} from the surface of soil colloids. For cations, the *manganese* (Mn) oxides can readily adsorb cobalt (Co) from soil solution. The phosphates adsorbed by hydrous oxides of Fe and Al can further adsorb zinc (Zn). Interaction between H_2PO_4^- and Zn could occur through the formation of a complex between H_2PO_4^- and Zn on the surface of the oxides.^[1]

For precipitation effects, the application of CaCO_3 can decrease the toxicity of Al to plants by increasing soil pH

and thus forming precipitation of $\text{Al}(\text{OH})_3$. Factors that decrease soil pH can result in Al toxicity. Aluminum toxicity can also be modified by phosphorus (P) supply through forming a precipitate of AlPO_4 . Similarly, the concentration of Mn^{2+} in soil solution can be affected by the soil pH and oxidation–reduction potential. The availability of P for plants may be controlled by the dissolution and precipitation of Ca phosphates on calcareous soils and sparingly soluble Fe or Al phosphates on acid soils, which can also be modified by soil pH.^[2] Chloride or sulfate (SO_4^{2-}) can form stable complexes with cadmium ion (Cd^{2+}), thus increasing the concentration of *chloride* ion (Cl^-) or SO_4^{2-} in the growth medium can significantly increase plant uptake of Cd^{2+} .^[3]

NUTRIENT INTERACTIONS AND DYNAMICS IN THE RHIZOSPHERE

The rhizosphere is the narrow zone of soil adjacent to roots where root, soil, and microorganisms play joint roles. Therefore, the rhizosphere is regarded as a special habitat with intense nutrient interactions and biological activities. All mechanisms of nutrient interactions in bulk soil are also applicable to the rhizosphere, but some aspects of nutrient interactions in the rhizosphere are distinct from bulk soil due to specific rhizosphere processes. Only the effects of major rhizosphere processes on nutrient interactions are considered here.^[2]

Rhizosphere pH can affect adsorption–desorption and dissolution–precipitation of nutrient ions, thus affecting nutrient bioavailability to plants. Rhizosphere pH depends primarily on relative differences in root uptake of cations and anions. The supply of NO_3^- -N normally increases soil pH in the rhizosphere, whereas the supply of NH_4^+ -N decreases rhizosphere pH. Symbiotic nitrogen fixation by legume plants can decrease rhizosphere pH. Rhizosphere pH can also be decreased due to P deficiency for non-graminaceous species. This acidification can enhance the dissolution of sparingly soluble minerals containing P, K, Fe, Zn, and Mn.^[2]

The oxidation–reduction status in the rhizosphere plays an important role in nutrient interactions. The heavy precipitation of Fe^{3+} oxide hydrates can form a Fe film at the root surface due to the release of O_2 into the rhizosphere of rice, which severely hampers the uptake of other nutrients. An important factor for the decreased Mn^{2+} concentration in rhizosphere soil is adsorption of Mn^{2+} to the freshly precipitated Fe^{3+} oxide hydrates. In calcareous soils with low availability of both Fe and Mn, the response to Fe deficiency may thus overcome Mn deficiency to some extent, whereas in calcareous soils with low Fe and high Mn availability, this effect may cause excessive mobilization of Mn in the rhizosphere and Mn toxicity in plants.^[2]

Roots can release considerable amounts of organic substances into the rhizosphere. Phosphorus deficiency can increase the exudation of citrate in white lupin growing on P-deficient calcareous soils. Citrate has considerable effects on the mobilization of micronutrients and P in the rhizosphere. Citrate may contribute to both desorption and chelation of Al and Fe from Fe and/or Al phosphates and thus enhance P mobilization. Under Fe deficiency, phyto-siderophores are released from roots in grasses, which can form stable chelates not only with Fe^{3+} but also with Cu^{2+} , Zn^{2+} , and less distinctly with Mn^{2+} .^[2,4]

NUTRIENT INTERACTIONS IN PLANTS

Movement of nutrient ions from soil solution into the cell walls of individual cells or roots is the first step for nutrient uptake. The interaction between *silicon* (Si) and Al may reduce the toxicity of Al by producing low-solubility aluminosilicate on the root cell wall.^[5,6] Higher concentrations of Ca^{2+} in solution may depress the absorption of Zn^{2+} and Mn^{2+} and enhance the absorption of anions such as H_2PO_4^- and SO_4^{2-} , possibly through the modification of the surface electrical potential of cell walls. Increased concentrations of Mg and Ca in solution may protect plants

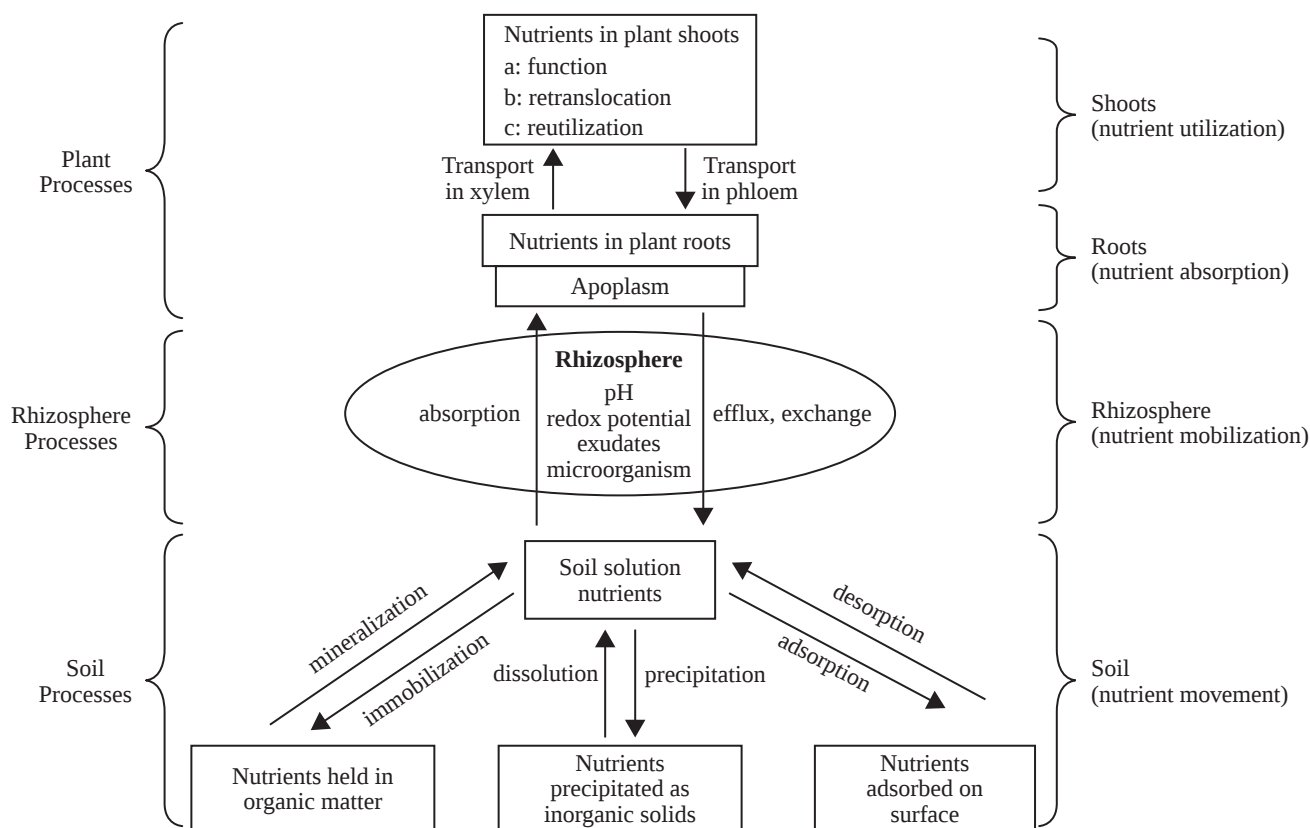


Fig. 1 Mechanisms and processes of nutrient interactions from soils to plants.

Source: Adapted from Robson & Pitman.^[1]

from Al toxicity by improving Mg or Ca nutrition and alleviating the toxic effect of Al on root development. Competition at absorption sites of root cell plasmamembranes is another mechanism for nutrient interactions. The absorption of MoO_4^{2-} by plant roots is strongly inhibited by SO_4^{2-} but is associated with an increase in H_2PO_4^- concentrations in solution. Nutrient interactions can also be operated through the regulation of efflux. For example, plants growing with higher external K^+ concentrations may have a higher efflux of Cs^+ than those growing with lower external K^+ concentrations, which contributes to the overall effect of K^+ – Cs^+ interactions.^[7] Altered expression of gene(s) related to the ion transport system of root cell plasma membranes may also be involved in nutrient interactions, e.g., P/Zn.

Another source of interaction among ions occurs due to the regulation of the cation/anion balance in cells. Nitrate ($\text{NO}_3\text{-N}$) and ammonium ($\text{NH}_4\text{-N}$) account for more than 70% of the total cations and anions taken up by plants. Iron nutrition of plants can be significantly improved by the supply of $\text{NH}_4^+\text{-N}$.^[8] Moreover, $\text{NH}_4\text{-N}$ supply could contribute to Fe translocation from primary leaves and roots to young leaves.^[2]

In nutrient utilization, the transport of one nutrient in xylem or phloem may be impaired by another nutrient. Furthermore, the function of the nutrient may be affected by this site.^[1] For example, Na^+ loading was larger than Mg^{2+} loading and was competed by K^+ . Chloride loading competed with SO_4^{2-} and phosphate. Additionally, high concentration of Ca or Mg may form precipitates in phloem, particularly in the presence of phosphates. The supply of Si can alleviate Mn toxicity in cereals by altering the distribution of Mn within plant leaves or regulating the uptake rate of Mn. However, the mechanisms for interactions between Si and Mn are not yet understood.

Nutrient interactions may occur during functioning of nutrients. For example, Mo is a constituent of nitrate reductase. When MoO_4^{2-} is replaced by WO_4^{2-} , the activity of NO_3^- reductase in higher plants can be inhibited. Some species can use Na in place of K for some, but not all, physiological or metabolic functions. Additionally, Mo, Ca, and Cu may be the components of the system of symbiotic N fixation.

Nutrient interactions can involve not only one mechanism but also several processes from soil to plant. For example, the mechanisms of interactions between Zn and P may include as follows: P depresses Zn bioavailability in soils and Zn absorption by roots and translocation of Zn from roots to shoots; P-induced growth response causes Zn dilution in plant tops; P–Zn imbalance leads to metabolic disorder; and P-activated interference affects Zn functioning. Additionally, P can also decrease the development of arbuscular mycorrhizas (AM), resulting in a decrease in plant Zn uptake mediated by AM.^[9] Overall, there is possibly a triangle interaction between P, Zn, and AM; however, the mechanisms involved are not clear.

CONCLUSION

Interactions between nutrients can occur through several different mechanisms involving nutrient movement from soil to root surface; nutrient dynamics in the rhizosphere; nutrient absorption and utilization by plants; nutrient mobilization and retranslocation within plants; and the expression of genes involved in nutrient uptake, translocation, and function (Fig. 1). It is important to identify the limiting step(s) of the overall effects of the interactions and what effects on plant growth are produced through nutrient interactions. Although some independent research on nutrient interactions in soils or plants has been done, systematic studies on nutrient interactions in the soil–plant system from molecular to ecosystem level should be emphasized in future research. Another dimension of nutrient interactions that needs attention is the control of plant uptake of trace elements (nutrients or contaminants) for food quality and environmental remediation.

ACKNOWLEDGMENT

This project was supported by MSBRDPC (No. G1999011709), NSFC (No. 39790100 and 30000102) and the “Hundred-Talent Program” of the Chinese Academy of Sciences.

REFERENCES

1. Robson, A.D.; Pitman, M.G. Interactions between nutrients in higher plants. In *Inorganic Plant Nutrition*; Läuchli, A., Bielecki, R.L., Eds.; Springer-Verlag: New York, 1983; 147–180.
2. Marschner, H. *Mineral Nutrition of Higher Plants*, 2nd Ed.; Academic Press: London, 1995.
3. McLaughlin, M.J.; Lambrechts, R.M.; Smolders, E.; Smart, M.K. Effects of sulfate on cadmium uptake by Swiss chard: II. Effects due to sulfate addition to soil. *Plant Soil* **1998**, *202*, 217–222.
4. Zhang, F.S.; Römhild, V.; Marschner, H. Role of the root apoplasm for iron acquisition by wheat plants. *Plant Physiol.* **1991**, *97*, 1302–1305.
5. Hodson, M.J.; Evans, D.E. Aluminum/silicon interaction in higher plants. *J. Exp. Bot.* **1995**, *46*, 161–171.
6. Cocker, K.M.; Evans, D.E.; Hodson, M.J. The amelioration of aluminum toxicity by silicon in higher plants: Solution chemistry or an in plant mechanism. *Physiol. Plant* **1998**, *104*, 608–614.
7. Zhu, Y.-G.; Shaw, G.; Nisbet, A.F.; Wilkins, B.T. Effects of external potassium on compartmentation and flux characteristics of radiocaesium in intact spring wheat roots. *Ann. Bot.* **1999**, *84*, 639–644.
8. Mengel, K.; Planker, R.; Hoffman, B. Relationship between leaf apoplast pH and iron chlorosis of sunflower (*Helianthus annuus* L.). *J. Plant Nutr.* **1994**, *17*, 1053–1065.
9. Smith, S.E.; Read, D.J. *Mycorrhizal Symbioses*, 2nd Ed.; Academic Press: London, 1997.

Nutrients: Water Interactions

Ardell D. Halvorson

U.S. Department of Agriculture (USDA), Fort Collins, Colorado, U.S.A.

Abstract

Water plays a critical role in the availability of nutrients to plants. Adequate levels of both water and nutrients are needed to optimize plant growth and productivity. Fertilizer and water management practices can influence the efficient use of water and nutrients by plants and their subsequent impact on environmental quality.

INTRODUCTION

Water is a major factor in nutrient availability to plants.^[1–4] It is the vehicle through which nutrients move through soil to access plant roots for uptake. Nutrients move via mass flow and diffusion in soil water to the root surface. Root interception is a third way in which plants obtain soil nutrients as root hairs develop and contact the soil particles and/or solution.

WATER AND NUTRIENT AVAILABILITY

In mass flow, nutrient ions are transported with water flow to the root as the plant absorbs water for transpiration. Many mobile nutrients, such as calcium (Ca), magnesium, nitrate–nitrogen (NO₃-N), and sulfate, are transported to the root by mass flow. Diffusion of nutrients to the plant root occurs as ions move from high-concentration areas to low-concentration areas in the soil solution. Phosphorus (P) and potassium are two nutrients that move by diffusion.

If soil water becomes limiting, as it frequently does under dryland or rainfed conditions, nutrient availability to plants can be affected.^[5] Water is held as a film around soil particles. As the water content of the soil decreases, the thickness of the film decreases. Most plant nutrients are readily available when the soil is near field capacity, which is about the water content of the wet soil after two days of rain has saturated it and free drainage has ceased. Nutrient availability is at a minimum as the soil water content approaches the permanent wilting point, which is the water content at which plant roots cannot extract water from the soil. As soil water content diminishes, some less-soluble nutrients may precipitate out of the soil solution and become unavailable to plants. However, these minerals will dissolve and become available once again as the soil is rewetted. Thus, soil water content influences nutrient availability and plant growth.

Micronutrients are generally supplied to plant roots by diffusion in soil. Therefore, low soil moisture conditions will reduce micronutrient uptake. Plants require smaller quantities of micronutrients to optimize productivity than those of macronutrients such as P; thus, drought stress effects on micronutrient deficiency are not as serious as for P. However, iron and zinc deficiencies are frequently associated with high soil moisture conditions.^[2]

Soil water content is an important factor in microbial activity in soils. Soil microbial activity is important in the breakdown of organic plant and animal residues, which release nutrients such as N and P for plant uptake. Microbial activity tends to be the greatest when soil water is near field capacity with soil temperatures ranging from 25°C to 35°C. As soils dry, microbial activity decreases and lowers the rate of nutrient release from soil organic matter.^[6,7]

NUTRIENT AND WATER USE EFFICIENCY (WUE)

Adequate levels of plant nutrients are needed to optimize rooting depth and water extraction from the soil.^[2,3,5] Healthy plants tend to root deeper into the soil profile, using more of the soil water in the root zone. Thus, plants not only need adequate water to optimize yield potential, but also require an adequate level of nutrients to allow the crop to take advantage of the available water supplies. Under dryland conditions, the crop will often use all of the available water (precipitation plus soil water in the root zone) during the growing season. The application of N and P fertilizers will frequently increase crop yields, thus increasing crop WUE. WUE is the amount of crop produced per unit of available water from precipitation, soil, and irrigation. The influence of N fertilization on WUE of winter wheat, corn, and sorghum in a dryland wheat–corn or sorghum–fallow rotation is shown in [Fig. 1](#).

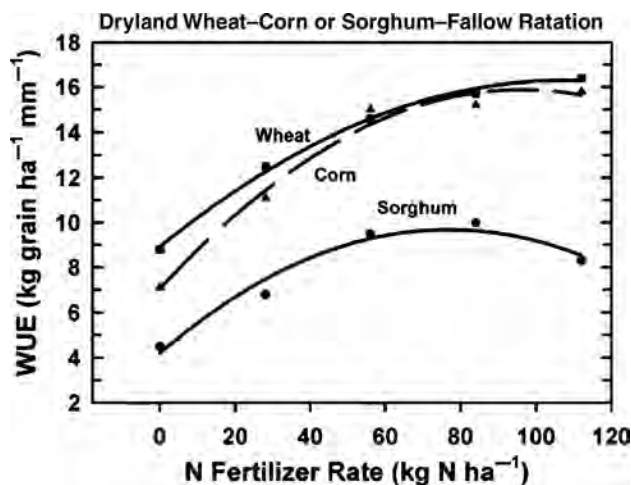


Fig. 1 WUE of wheat, corn, and sorghum as a function of N fertilizer rate in a dryland wheat–corn or sorghum–fallow rotation near Akron, Colorado, U.S.A.

When plant-available water is limited, the overapplication of N can also result in reduced grain yields owing to increased vegetative growth and water use in the early growth stage, with insufficient water remaining to maximize grain development and yield. The application of N will not increase yields without adequate plant-available water, and increasing plant-available water will not increase crop yield without adequate N supply. The percentage increase in response of crops such as wheat to P fertilization tends to be greater in dry years than in wet years on P-deficient soils, while both N and P are needed to optimize yields in wetter years.

Water is important for the activation and movement of fertilizer nutrients applied to soils.^[1–3,7,8] Dry fertilizer granules must dissolve in the soil water before they become available to plants. When applied to dry soil, liquid fertilizers may become unavailable to plants until precipitation or irrigation water rewets the soil and they become part of the soil solution again. Rainfall affects the volatilization loss of N from ammonia (NH₃)-based fertilizers such as urea and urea ammonium nitrate (UAN). Rainfall received within 36 hours after the surface applications of urea or UAN fertilizers will greatly reduce N volatilization losses and improve the N fertilizer use efficiency by crops. Rainfall moves the surface-applied N fertilizer into the soil where it can react and reduce NH₃ losses to the atmosphere. Excessive soil water, however, can result in anaerobic conditions and the loss of NO₃-N by denitrification. NO₃-N is converted to various N gases, which are lost to the atmosphere under anaerobic conditions.

Water is essential for optimizing crop yields. Under irrigation, water is generally not a yield-limiting factor. Under dryland or rainfed conditions, crop yields are dependent on available soil water supplies and growing season precipitation. Adequate levels of essential plant nutrients are

needed to optimize crop yields and WUE (i.e., kg grain produced/mm crop water use). Under rainfed conditions, crop water supplies during the growing season can vary weekly and annually. During the periods of drought (i.e., low supply of plant-available water), less plant nutrients are needed to optimize crop yields than during the years of average or above-average precipitation. In wetter years, both the crop yield potential and the nutrients needed to optimize crop yield increase.

Soil management practices, such as reduced- and no-till systems, that increase soil organic matter and improve soil physical quality also improve soil aggregation and porosity. This, in turn, improves water infiltration into the soil and water availability for increased crop productivity and improved nutrient use efficiency.

IRRIGATION WATER QUALITY AND FERTILIZER APPLICATION

Irrigation water quality can affect the application of fertilizer nutrients through irrigation systems.^[3,8] For example, the addition of anhydrous NH₃ or liquid ammonium polyphosphate fertilizers to irrigation waters high in Ca can result in the formation of lime and Ca phosphate precipitates. The precipitates can plug sprinkler and drip irrigation systems. In some instances, the precipitation of Ca can result in a higher sodium hazard of the irrigation water, which may subsequently reduce the water intake capacity of the soil.

Applying fertilizers with both flood and furrow irrigation systems requires that a uniform distribution of water be achieved throughout the field to obtain a uniform distribution of fertilizer nutrients to the crop. With flood and furrow irrigation systems, fertilizer should not be applied with the initial flush of irrigation water because of the generally non-uniform distribution of water during the initial wetting of the soil surface by the irrigation water. The reactions of fertilizers with the irrigation water and the fertilizer distribution to the crop are affected by (irrigation) water quality. If fertigation (i.e., the application of fertilizer nutrients through an irrigation system) is to be used, the compatibility of fertilizers to be applied with the quality of irrigation water available must be examined to avoid poor distribution of fertilizer nutrients.

ENVIRONMENTAL QUALITY

N is generally transported from soils into surface and groundwater by runoff, erosion, and leaching.^[7,9] Runoff water from watersheds with high levels of soluble N and P sources on the soil surface can contribute to the eutrophication of streams, lakes, ponds, bays, and estuaries. Placing or positioning applied N and P sources below the soil surface and using soil management practices to minimize

runoff will help reduce agriculture's impact on the eutrophication of water bodies. Water erosion of soil not only carries soluble plant nutrients from a watershed, but also carries soil particles with sorbed nutrients, such as P, into water bodies that can then contribute to the degradation of water quality.

Soil management practices such as no-till and other conservation tillage practices can reduce soil erosion by water. Water moving through soil in excess of field-capacity water content can move soluble nutrients, such as $\text{NO}_3\text{-N}$, below the root zone of crops and into groundwater. In summary, using cropping systems and an adequate fertility program to optimize crop WUE will help reduce loss of plant-available water and nutrients below the crop root zone.

CONCLUSION

Water plays a critical role in the availability of nutrients to plants. Adequate levels of both water and nutrients are needed to optimize plant growth and productivity. Fertilizer and water management practices can influence the efficient use of water and nutrients by plants and their subsequent impact on environmental quality.

REFERENCES

1. Engelstad, O.P. *Fertilizer Technology and Use*, 3rd Ed.; SSSA: Madison, 1985.
2. Havlin, J.L.; Beaton, J.D.; Tisdale, S.L.; Nelson, W.L. *Soil Fertility and Fertilizers: An Introduction to Nutrient Management*, 6th Ed.; Prentice Hall: Upper Saddle River, 1999.
3. Mortvedt, J.J.; Murphy, L.S.; Follett, R.H. *Fertilizer Technology and Application*; Meister Publishing Co.: Willoughby, 1999.
4. Troeh, F.R.; Thompson, L.M. *Soils and Soil Fertility*, 5th Ed.; Oxford University Press: New York, 1993.
5. Taylor, H.M.; Jordan, W.R.; Sinclair, T.R., Eds. *Limitations to Efficient Water Use in Crop Production*; ASA-CSSA-SSSA: Madison, 1983.
6. Follett, R.F.; Stewart, J.W.B.; Cole, C.V., Eds. *Soil Fertility and Organic Matter as Critical Components of Production Systems*; SSSA: Madison, 1987.
7. Pierzynski, G.M.; Sims, J.T.; Vance, G.F. *Soils and Environmental Quality*, 2nd Ed.; CRC Press: Boca Raton, 2000.
8. Ludwick, A.E.; Bonczkowski, L.C.; Bruice, C.A.; Campbell, K.B.; Millaway, R.M.; Petrie, S.E.; Phillips, I.L.; Smith, J.J., Eds. *Western Fertilizer Handbook*, 8th Ed.; California Fertilizer Association, Interstate Publishers: Danville, 1995.
9. Follett, R.F., Ed. *Nitrogen Management and Ground Water Protection*; Elsevier: New York, 1989.

Organic Agriculture

Kathleen Delate

Departments of Agronomy and Horticulture, Iowa State University, Ames, Iowa, U.S.A.

Abstract

Organic agriculture is based on minimal use of off-farm inputs and on management practices that restore, maintain, or enhance ecological harmony. The term “organic” is defined by law, but the labels “natural,” “eco-friendly,” and similar statements do not guarantee complete adherence to organic practices as defined by law.

INTRODUCTION

The USDA National Organic Standards Board defines organic agriculture as “an ecological production management system that promotes and enhances biodiversity, biological cycles, and soil biological activity.”^[1] It is based on minimal use of off-farm inputs and on management practices that restore, maintain, or enhance ecological harmony. The primary goal of organic agriculture is to optimize the health and productivity of interdependent communities of soil life, plants, animals and people.”^[1] The term “organic” is defined by law. The labels “natural,” “eco-friendly,” and similar statements do not guarantee complete adherence to organic practices as defined by law.

HISTORY

In 1990, the U.S. Congress passed the Organic Food Production Act (OFPA). This law was heralded as the first U.S. law established to regulate a system of farming. The OFPA requires that anyone selling products as “organic” must follow a set of prescribed practices that include avoidance of synthetic chemicals in crop and livestock production, and in the manufacturing of processed products. “Certified organic” crops must be raised on land to which no synthetic chemical (any fertilizers, herbicides, insecticides, or fungicides) inputs were applied for 3 years prior to the crops’ sale. Organic certification agencies became established in the United States to deal with a required “third-party certification.” There are at least 20 private certification agencies and 15 state agencies certifying organic production and processing in the United States. Proposed rules implementing the federal OFPA law were promulgated in 1997, after 7 years of revisions. Unfortunately, these rules did not meet private certification agencies’ standards, and a record number of complaints (275,000) were issued in the public comment period. The federal rules are established (released in 2001), all certifiers must utilize the federal standards as the minimum standard for the “certified organic” label in

the United States. European regulation is under the auspices of the International Federation of Organic Agriculture Movements with national certification agencies in each country.^[2] Japan certifies under the Ministry of Agriculture and Forestry. Certification for the European Union and Japan is extended to several U.S. certifiers that meet international standards.

Organic agriculture is the oldest form of agriculture on earth. Farming without the use of petroleum-based chemicals (fertilizers and pesticides) was the norm for farmers in the developed world until post-World War II (WWII). The war era led to technologies that were adapted for agricultural production. Ammonium nitrate used for munitions during WWII evolved into ammonium nitrate fertilizer; organophosphate nerve gas production led to the development of powerful insecticides. These technical advances since WWII have resulted in significant economic benefits, as well as unwanted environmental and social effects. Organic farmers seek to utilize those advances that yield benefits (e.g., new varieties of crops, more efficient machinery) while discarding those methods that have led to negative impacts on society and the environment, such as pesticide pollution and insect pest resistance.^[3] Instead of using synthetic fertilizers and pesticides, organic farmers utilize crop rotations, cover crops, and naturally based products to maintain or enhance soil fertility.^[4,5] These farmers also rely on biological, cultural, and physical methods to limit pest expansion and increase populations of beneficial insects on their farms. By managing their ecological capital through efficient use of on-farm natural resources, organic farmers produce for diverse and specialized markets that provide premium prices.

Because genetically modified organisms (GMOs) constitute synthetic inputs and pose unknown risks, GMOs, such as herbicide-resistant seeds, plants, and product ingredients, are disallowed in organic agriculture. Organic livestock, like organic crops, must be fed 100% organic food or feed in their production. Synthetic hormones and antibiotics are disallowed in organic livestock production. Traditional farmers throughout the world have relied on natural

production methods for centuries, maintaining consistent yields within their local environment. While “green revolution” technologies have led to increased yields in many less-developed countries, many farmers have seen an increase in pest problems with new varieties and high input-based systems.

Motivations for organic production include economic, food safety, and environmental concerns. All organic farmers avoid the use of synthetic chemicals in their farming systems, but philosophies differ among organic farmers regarding methods to achieve the ideal system. Organic farmers span the spectrum from those who completely eschew external inputs, create on-farm sources of compost for fertilization, and encourage the activity of beneficial insects through conservation of food and natural habitats, to those farmers who import their fertility and pest management inputs. A truly sustainable method of organic farming would seek to eliminate, as much as possible, reliance on external inputs.

WORLDWIDE STATISTICS

USDA does not publish systematic reports on organic production in the United States. The census in 1994 identified 1.5 million acres of organic production in the United States with 4050 farmers reporting organic acreage.^[6] This figure underrepresents production because many organic farmers opt to sell their products as organic without undergoing certification. The U.S. organic industry continues to grow at a rate of 20% annually. The industry was listed as a \$4.5 billion industry in 1998, with predicted future growth to \$10 billion by 2003. The organic industry is a consumer-driven market. According to industry surveys, the largest purchasers of organic products are young people and college-educated consumers. Worldwide consumption of organic products has experienced tremendous growth, often surpassing U.S. figures of 20% annual gain. Much of the increase in consumption worldwide has been fueled by consumers’ demand for GMO-free products. Because GMOs are disallowed in organic production and processing, organic products are automatically segregated as GMO-free at the marketplace. European consumers have led the demand for organic products, particularly in countries such as the Netherlands and Scandinavia. Two percent of all German farmland, 4% of Italian farmland, and 10% of Austrian farmland are managed organically.^[2] Prince Charles of England has developed a model organic farm and established a system of government support for transitioning organic farmers. Major supermarket chains and restaurants in Europe offer a wide variety of organic products in their aisles and on their menus. Industry experts predict that the establishment of federal rules will advance organic sales in the United States. Although the organic industry began as a niche market, steady growth has led to its place in a “segment” market since 1997. The organic dairy

industry, e.g., expanded by 73% from 1996 to 1997, and continues to grow. Organic markets can be divided into indirect and direct markets. Indirect or wholesale markets include cooperatives, wholesale produce operations, brokers, and local milling operations. Many supermarket chains buy directly from farmers or from wholesalers of organic products. Because meat can be labeled as “organic,” as of 1999, the marketing of organic beef, pork, chicken, and lamb has been significantly simplified. Roadside stands, farmers’ markets, and community supported agriculture farms constitute the direct marketing end of the organic industry. Most consumers relate their willingness to pay premium prices for food that has been raised without synthetic chemicals because of their concern for food safety and the environment. Supporting local family farmers also enters into their purchasing decisions.

CROP AND PEST PERFORMANCE IN ORGANIC SYSTEMS

The basis for all organic farming systems is the health of the soil.^[4,7] In addition to maintaining adequate fertility, organic farmers strive for biologically active soil, containing microbial populations required for nutrient cycling.^[8,9] Crop rotations (required for all organic operations) provide nutrients such as nitrogen in the case of legume crops (alfalfa and clover) and carbonaceous biomass upon which beneficial soil microorganisms depend on survival.^[10,11] A crop rotation plan is required as protection against pest problems and soil deterioration.^[12–15] Ideally, no more than four out of 6 years should be in agronomic crops, and the same row crop cannot be grown in consecutive years on the same land. Legumes (alfalfa, clovers, and vetches) alone, or in combination with small grains (wheat, oats, and barley), must be rotated with row crops (corn, soybeans, amaranth, vegetables, and herbs) to ensure a healthy system. A typical six-year rotation in the Midwestern United States would be corn (with a cover of winter rye)–soybeans–oats (with an underseeding of alfalfa)–alfalfa–corn–soybeans.^[16,17] Horticultural crops must be rotated with a leguminous cover crop at least once every 5 years.

Pest management in organic farming systems is based on a healthy plant able to withstand some pest injury and on the inherent equilibrium in nature, as most insect pests have natural enemies that regulate their populations in unperturbed environments.^[18,19] Because only naturally occurring materials are allowed in organic production, insect predators, parasites, and pathogens exist without intervention from highly toxic insecticides.^[20] Most organic farmers rely on naturally occurring beneficial insects on their farms, but some farmers purchase and release lacewings and other natural enemies every season, for example. There are also commercial preparations of natural insect pathogens, such as *Bacillus thuringiensis* (Bt), which are used to manage pestiferous larvae, such as corn borers. Botanical

insecticides, such as neem and ryania, are also allowable in organic production, but as with all insecticides, sprays should be used only as a last resort. Although these materials are naturally based, some materials may affect natural enemies. Prevention is a cornerstone of organic farming.^[21] Pest-free seeds and transplants, along with physical and cultural methods, are used to prevent pest infestations. Physical methods include the use of row covers for protection against insects such as cabbage butterflies and aphids. Cultural methods include sanitation and resistant varieties. Plant varieties are used that have been bred traditionally (i.e., no manipulated gene insertion or engineering involved) for insect, disease, and nematode resistance or tolerance.

Most organic farmers rely on multiple tactics for their weed management.^[22,23] Allelopathic crops, cultivation, mulching, and flame burning are all methods available for organic farmers. Allelopathic crops, such as rye and oats, produce an exudate that mitigates against small weed seed germination. Depending on the crop, cultivation offers the least labor-intensive method of organic weed management. Timely cultivation is key; without specific schedules, weeds proliferate. Propane flame burning is generally used in conjunction with cultivation, particularly during times of high field moisture. Mulching with straw or wood chips is commonly used in many organic horticultural operations.

Yields comparable to conventional crops have been shown for organic crops in three university long-term experiments in the United States (South Dakota State University,^[24] Iowa State University,^[16] the University of California-Davis,^[25] and in many European studies^[26]). Factoring in an organic premium (ranging from 50% to 400%, depending upon crop and season), organic systems consistently out-performed conventional systems in terms of economics.^[17,27,28] Pest problems were not a critical factor in these organic systems. Other studies have shown the benefits of organic practices, such as composting, in mitigating root-borne diseases.^[8]

KEY ISSUES REQUIRING ADDITIONAL RESEARCH

Continued verification of the long-term benefits of organic vs. conventional farming in terms of soil quality,^[29] pest management, and nutritional benefits^[30] is needed. Key issues include the development of management practices to increase nutrient cycling for maintenance of crop yields and optimize biological control of plant pests and diseases.^[31] Economic analysis, including risks of the three-year transition required for organic certification, will provide useful information for growers interested in alternative systems.^[26,32] Appropriate tillage systems, which protect soil quality and provide adequate soil preparation, are important issues for organic producers. The improvement of natural parasiticide formulations, such

as diatomaceous earth, is required for optimum organic livestock production. Marketing and support needs include the availability of reliable statistics for organic operations and prices. Although many European countries support their farmers in their organic production practices through environmental subsidies,^[33] the United States has made small gains in this area. Some state agencies (Minnesota Department of Agriculture) and the USDA Natural Resources Conservation Services through the Environmental Quality Indicators Program offer financial incentives to organic farmers during their transitioning years. More of these support services are needed to encourage farmers interested in the conversion to alternative production.^[34]

REFERENCES

1. National Organic Program. USDA Agricultural Marketing Service: Washington. Available at <http://www.ams.usda.gov/nop/> (accessed June 2000).
2. Lampkin, N.H. *The Policy and Regulatory Environment for Organic Farming in Europe*; Institut für Landwirtschaftliche Betriebslehre, Universität Hohenheim: Stuttgart, 1999; 1–379.
3. Altieri, M. *Agroecology*, 2nd Ed.; Westview Press: Boulder, 1995; 1–433.
4. Lampkin, N.H.; Measures, M. 1999 *Organic Farm Management Handbook*, 3rd Ed.; Welsh Institute of Rural Studies, University of Wales: Hamstead Marshall, 1999; 1–163.
5. Lockeretz, W.; Shearer, G.; Kohl, D. Organic farming in the corn belt. *Science* **1981**, *211*, 540–547.
6. Lipson, M. *Searching for the “O-Word”: Analyzing the USDA Current Research Information System for Pertinence to Organic Farming*; Organic Farming Research Foundation: Santa Cruz, 1997; 1–85.
7. Macey, A.; Kramer, D. *Organic Field Crop Handbook*; Canadian Organic Growers, Inc.: Ottawa, 1992; 1–256.
8. Drinkwater, L.E.; Letourneau, D.K.; Shennan, C. Fundamental differences between conventional and organic tomato agroecosystems in California. *Ecol. Appl.* **1995**, *5* (4), 1098–1112.
9. Wander, M.M.; Traina, S.J.; Stinner, B.R.; Peters, S.E. Organic and conventional management effects on biologically active soil organic matter pools. *Soil Sci. Soc. Am. J.* **1994**, *58*, 1130–1139.
10. Drinkwater, L.E.; Wagoner, P.; Sarrantonio, M. Legume-based cropping systems have reduced carbon and nitrogen losses. *Nature* **1998**, *396*, 262–265.
11. Yeates, G.W.; Bardgett, R.D.; Cook, R.; Hobbs, P.J.; Bowling, P.J.; Potter, J.F. Faunal and microbial diversity in three welsh grassland soils under conventional and organic management regimes. *J. Appl. Ecol.* **1997**, *34* (3), 453–470.
12. Adee, E.A.; Oplinger, E.S.; Grau, C.R. Tillage, rotation sequence, and cultivar influences on brown stem rot and soybean yield. *J. Prod. Agr.* **1994**, *7* (3), 341–347.
13. Karlen, D.L.; Varvel, G.E.; Bullock, D.G.; Cruse, R.M. Crop rotations for the 21st century. *Adv. Agron.* **1994**, *53*, 1–45.
14. Lipps, P.E.; Deep, I.W. Influence of tillage and crop rotation on yield, stalk rot, and recovery of *Fusarium* and *Trichoderma* spp. from corn. *Plant Dis.* **1991**, *75* (8), 828–833.

15. Stinner, D.H.; Stinner, B.R.; Zaborski, E.R.; Favretto, M.R.; McCartney, D.A. *Ecological Analyses of Ohio Farms under Long-Term Sustainable Management*; Ecological Society of America: Knoxville, 1994; 7–11.
16. Delate, K. *Comparison of Organic and Conventional Rotations at the Neely-Kinyon Long-Term Agroecological Research (LTAR) Site*; Leopold Center for Sustainable Agriculture Annual Report; Iowa State University: Ames, 1999; 1–12.
17. Welsh, R. *The Economics of Organic Grain and Soybean Production on the Midwestern United States*; Henry, A., Ed.; Wallace Institute for Alternative Agriculture: Greenbelt, Policy Studies Program Reports, 1999.
18. Neher, D.A. Nematode communities in organically and conventionally managed agricultural soils. *J. Nematol.* **1999**, *31*, 142–154.
19. Pfiffner, L.; Niggli, U. Effects of bio-dynamic, organic and conventional farming on ground beetles and other epigeic arthropods in winter wheat. *Biol. Agr. Hort.* **1996**, *12*, 353–364.
20. Kromp, B.; Meindel, P.; Harris, P.J.C. *Entomological Research in Organic Agriculture*; Academic Publishers: Bicester, 1999; 1–386.
21. Lockeretz, W. *Environmentally Sound Agriculture: Selected Papers from the Fourth International Conference of the International Federation of Organic Agriculture Movements (IFOAM)*; Praeger: New York, 1983; 1–426.
22. Lanini, W.T.; Zalom, F.; Marois, J.; Ferris, H. Researchers find short-term insect problems, long-term weed problems. *Ca. Agric.* **1994**, *48*, 27–33.
23. Liebman, M.; Ohno, T. Crop rotation and legume residue effects on weed emergence and growth: Applications for weed management. In *Integrated Weed and Soil Management*; Hatfield, J.L., Buhler, D.D., Stewart, B.A., Eds.; Ann Arbor Press: Chelsea, 1997; 181–221.
24. Dobbs, T. *Report on Organic and Conventional Grain Trials at South Dakota State University. USDA–ERS Conference on the Economics of Organic Farming Systems*; USDA: Washington, DC, 1999; 1–4.
25. Klonsky, K.; Livingston, P. Alternative systems aim to reduce inputs, maintain profits. *Ca. Agric.* **1994**, *48* (5), 34–42.
26. Lampkin, N.H.; Padel, S. *The Economics of Organic Farming: An International Perspective*; CAB International: Wallingford, 1994; 1–468.
27. Hanson, J.C.; Lichtenberg, E.; Peters, S.E. Organic versus conventional grain production in the Mid–Atlantic: An economic and farming system overview. *Am. J. Alt. Agr.* **1997**, *12*, 2–9.
28. Stanhill, G. The comparative productivity of organic agriculture. *Agr. Ecosys. Environ.* **1990**, *30*, 1–26.
29. Lockeretz, W.; Shearer, G.; Klepper, R.; Sweeney, S. Field crop production on organic farms in the midwest. *J. Soil Water Conserv.* **1978**, *33*, 130–134.
30. Woese, K.; Lange, D.; Boess, C.; Bögl, K.W. A comparison of organically and conventionally grown foods—results of a review of the relevant literature. *J. Sci. Food Agr.* **1997**, *74*, 281–293.
31. Walz, E. *Final Results of the Third Biennial National Organic Farmers' Survey*; Organic Farming Research Foundation: Santa Cruz, 1999; 1–75.
32. Chase, C.; Duffy, M. An economic comparison of conventional and reduced-chemical farming systems in Iowa. *Am. J. Alt. Agr.* **1991**, *6* (4), 168–173.
33. Zygmunt, J. International issues pertaining to organic agriculture. In *Proceedings of the Workshop: Organic Farming and Marketing Research—New Partnerships and Priorities*; Lipson, M., Hammer, T., Eds.; Organic Farming Research Foundation: Santa Cruz, 1999; 317–379.
34. D'Souza, G.; Cyphers, D.; Phipps, T. Factors affecting the adoption of sustainable agricultural practices. *Agr. Resour. Econ. Rev.* **1993**, *22* (2), 159–205.

Organic Carbon: Assessment Methods

K.R. Islam

Ohio State University South Centers, Piketon, Ohio, U.S.A.

Abstract

Carbon, the principal component of soil organic matter (SOM), exerts profound effects on soil quality. A simple and rapid measurement of soil labile carbon by mild permanganate oxidation provides opportunities for early detection of changes in SOM quality. All available methods to determine soil organic carbon have inherent problems associated with them, and the user should select the method most applicable to the type of soils analyzed, expected recovery, and the required precision of the results.

INTRODUCTION

Soil organic matter (SOM), a complex, dynamic component of terrestrial ecosystem, is a core indicator of soil quality. Carbon (C) is the prime element present in SOM, comprising 48–58% of the total weight.^[1] Rapid and precise measurement of soil organic C (SOC) is essential to monitor temporal changes in SOM content. Since soil C is the sum of both organic and inorganic C pools, several methods have been developed or modified over years to determine SOC. The methods are: 1) dry and wet combustion analyses for total C and inorganic C separately and subtraction of the amount of inorganic C from the total C present; 2) dry and wet combustions for direct analysis of SOC after removal of inorganic C by acid pretreatment; 3) rapid dichromate oxidation of SOC from subsequent determination of unreduced Cr^{+6} or reduced Cr^{+3} by redox titration or colorimetric methods; 4) near-infrared reflectance spectroscopic analysis of SOC; and 5) rapid mild permanganate oxidation to measure soil active C by colorimetric method. A brief description of the measurement, principles, merits, and limitations of the commonly used methods for measurement of SOC is discussed (Table 1).

COMBUSTION METHODS TO MEASURE SOC AS CALCULATED FROM DIFFERENCE BETWEEN TOTAL C AND INORGANIC C DETERMINATIONS

Many of the total C methods are basic for the procedures modified to use for determination of SOC by dry and wet combustion methods. In dry combustion method, total C is measured in resistance or induction furnace using manual procedures or automated equipments.^[1,2] The use of resistance furnace to measure total C is based on oxidation of organic C from carbonaceous materials and thermal decomposition of carbonates with catalysts heating at about 1000°C in a stream of O_2 . In the use of induction furnace,

the soil sample is mixed with catalysts and rapidly heated to about 1650°C in a stream of O_2 to convert all C to carbon dioxide (CO_2). The CO_2 evolved is commonly trapped in Ascarite or suitable reagents and determined by titrimetric, gravimetric, volumetric, conductimetric, or colorimetric procedures. Using sequential dry combustion at different temperatures, both soil organic and inorganic C pools can be determined.^[6]

Most of the problems associated with the manual dry combustion techniques have been overcome by the development of automated combustion instruments. The widely used commercial instruments to measure total C are: LECO, Carlo-Erba, vario MAX CN, and Perkin–Elmer CHN analyzers. The citation of the above-mentioned instruments does not imply that they are superior or inferior to others in the market or being marketed.^[1,2] In automated dry combustion method, the soil sample is mixed with catalysts and heated in resistance or induction furnace in a stream of O_2 to convert all C to CO_2 . The CO_2 thus evolved is determined by gas chromatographic methods using thermal conductivity, infrared absorption, or flame ionization detectors.^[1,2]

The wet combustion of soil has long been a standard procedure for determination of total C. By using combustion train or Van Slyke–Neil apparatus, the soil sample is heated at about 210°C with a mixture containing $\text{K}_2\text{Cr}_2\text{O}_7$ – H_2SO_4 , and H_3PO_4 in a stream of O_2 to ensure complete oxidation of all C to CO_2 .^[7] The evolved CO_2 is determined gravimetrically, titrimetrically, or manometrically.

Determination of Soil Inorganic C (Carbonates)

Several methods are available for determination of soil inorganic C. These include: 1) neutralization of the carbonates with hydrochloric acid and back titration of the excess acid; 2) determination of calcium and magnesium in an acid leachate; and 3) dissolution of carbonates in

Table 1 Methodologies used for determination of organic and active C in soils.

Measurement	Basic methods	Principle	Merits	Limitations
From difference between soil total C and inorganic C	Both dry and wet combustions	Total C and inorganic C are determined on separate soil samples: $\text{SOC} = \text{total C} - \text{inorganic C}$	These methods yield reliable results for acid soils. Effective to oxidize resistant Cs. Automated methods are fast and precise	Two separate analyses are required. These methods are not suitable for high pH or recently limed soils. Incomplete thermal oxidation of carbonates
From determination of soil total C after removal of inorganic C	Both dry and wet combustions	Total soil C is determined after removal of inorganic C with acid pretreatment: $\text{SOC} = \text{total C}$	Total C measured after removing inorganic C is equal to SOC contents. Effective for acid soils	These methods are not very effective for complete removal of inorganic C
From rapid dichromate redox reactions	Wet oxidation	Dichromate (Cr^{+6}) oxidizes C in strong acid medium and the amount of Cr^{+6} reduced to Cr^{+3} is related to SOC	These methods are simple, rapid, and easy to use for a wide range of soils to measure SOC	Recoveries of SOC by the simple heat of dilution are incomplete and highly variable (60–86%)
a) Redox titration	Heat of dilution	Amount of Cr^{+6} reduced to Cr^{+3} is titrated with reducing agent and redox indicator	This method is rapid and simple to estimate C in different soils	Required a correction factor (1.32). Errors from variable C content in soils or horizons
With external heating	This is same as the Cr^{+6} oxidation procedure except that SOC is oxidized by external heat	No correction factor is required. The method, has reportedly permitted a total recovery of SOC	Errors due to thermal decomposition of Cr^{+6} . Interferences from Cl, Fe, and Mn oxides	
Colorimetric	Heat of dilution	Amount of oxidized Cr^{+6} or reduced Cr^{+3} species determined in solution is proportional to SOC	This is a simple and rapid method for SOC. No interferences from Cl, Fe, and Mn oxides	Low and variable recoveries of SOC from incomplete oxidation by simple heat of dilution
With controlled ionizing heating (e.g., hotplate)	This is same as the Cr^{+6} oxidation procedure except that SOC is assumed oxidized by external heating	This is a precise method with total SOC recovery. Minimum interferences from Fe, Cl, Mn, CO_3 , and charcoal	Thermal decomposition of Cr^{+6} due to ionizing heating; generates a substantial volume of wastes	
With nonionizing external heat (e.g., microwave oven)	This is same as the Cr^{+6} oxidation procedure except that SOC is assumed oxidized by nonionizing heating	This is a rapid, precise, and low-waste method. No interferences from Fe, Cl, Mn, CO_3 , and charcoal	A correction factor (1.09) is required. Need specialized equipments (e.g., microwave oven)	
From near-infrared reflectance spectrometry	Spectroscopic	SOC is determined from distinct spectral features of H–C bonds in SOM	This is a rapid and nondestructive method to measure SOC	Expensive and complex. Spectral responses affected by soil water, inorganic C, and total N
From permanganate oxidation of labile C	Wet mild oxidation	Amount of permanganate consumed is related to SOC from bleaching of pink color of the solution	Compared to SOC, the active C pool is more sensitive to management effects, and related to soil quality properties	The chemicals used are photosensitive and not stable over time. Yields best with dry soils

Source: From Nelson & Sommers,^[1] Tabatabai,^[2] Islam & Weil,^[3] Chang & Laird,^[4] and Weil, Islam, et al.^[5]

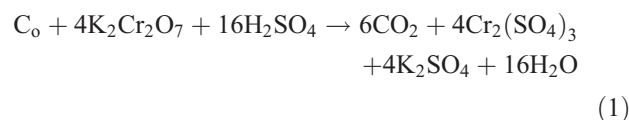
acid and determination of the inorganic C by measuring the volume or pressure of the CO₂ or by titrimetry, sample weight loss, infrared spectrometry, gas chromatography, and thermogravimetry.^[1,2]

COMBUSTION METHODS FOR DIRECT MEASUREMENT OF SOC AFTER REMOVAL OF INORGANIC C

Prior to direct measurement of SOC by dry or wet combustion method, inorganic C must be removed from soil, especially calcareous or limed soils. Soil inorganic C is removed by pretreating the sample with a mixture of dilute H₂SO₄ and FeSO₄ to minimize oxidation and decarboxylation of SOM or with H₂SO₃ followed by heating to remove H₂SO₃.^[7] It has been reported that inorganic C could be effectively removed by a metaphosphoric acid treatment at 130°C.^[8]

RAPID DICHROMATE OXIDATION METHODS FOR INDIRECT MEASUREMENT OF SOC

Previous works^[9,10] suggested that oxidation of SOC can be accomplished by a K₂Cr₂O₇–H₂SO₄ mixture as follows:



Over the years, a number of modifications^[11–13] have been proposed to measure SOC by rapid dichromate (Cr⁺⁶) oxidation methods (Table 1). The present variants of Cr⁺⁶ oxidation fall into two groups: one group is employed with simple heat of dilution and the another group is employed with external heat.

Rapid Titrimetric Determination of SOC After Dichromate Oxidation with Simple Heat of Dilution

Rapid titration methods to measure SOC are based on simple heat of dilution from K₂Cr₂O₇–H₂SO₄ mixture. The amount of Cr⁺⁶ remaining after oxidation is back titrated with ferrous ammonium sulfate in the presence of redox indicators.

Rapid Titrimetric Determination of SOC After Dichromate Oxidation with External Heating

A controlled use of external heating to oxidize C by K₂Cr₂O₇–H₂SO₄ mixture under reflux with two reagent blanks was suggested for complete recovery of SOC.^[11,13] One blank is used for standardization of the ferrous ammonium sulfate due to variable stability, and the another, a boiled blank to account for thermal decomposition of the

Cr⁺⁶. This method has been widely used with modifications to perform digestion in Folin–Wu tubes placed in an Al-heating block at 150°C^[14] or digestion of soil with K₂Cr₂O₇–H₂SO₄ mixture for 30 minutes in glass tubes in a heating block at 170°C.^[15] Others suggested that digestion of soil sample with a mixture of K₂Cr₂O₇–H₂SO₄ at 135°C is optimum for complete oxidation of SOC.^[16] The remaining Cr⁺⁶ is back titrated with ferrous ammonium sulfate in the presence of redox indicators.

Colorimetric Determination of SOC After Rapid Dichromate Oxidation with Simple Heat of Dilution or External Heating

The Cr⁺⁶ oxidation methods with or without external heating for measurement of SOC have been modified colorimetrically after removal of soil particles from the suspension by filtration or centrifugation.^[3,17–19] As the oxidation of SOC is accompanied by the reduction of Cr⁺⁶ to Cr⁺³, a pronounced change in color, orange to green, provides a rapid colorimetric measurement of SOC that is preferred over the tedious titration procedures.^[3,17–19] The Cr⁺³ exists as stable hexaquo ion [Cr(H₂O)₆]⁺³ in an acid medium and has two absorption peaks at 450 and 590 nm, respectively.^[18] In a comparison of methods that used to measure Cr⁺³ and Cr⁺⁶ absorption, it has been suggested that colorimetric detection of Cr⁺³ green color intensity in K₂Cr₂O₇–H₂SO₄–soil mixture at 590–600 nm wavelengths, without any masking interference from Cr⁺⁶, is proportional to the amount of SOC.^[18,19] An external heating at 135°C for 30 minutes is enough to oxidize C by Cr⁺⁶ for colorimetric determination of SOC from Cr⁺³ concentration measured at 600 nm against sucrose C standards.^[19]

Since the heat of dilution raised the temperature of K₂Cr₂O₇–H₂SO₄–soil mixture sufficiently to induce a substantial oxidation within a minute or so, a rapid colorimetric method based on nonionizing microwave energy has been developed for improved oxidation to determine SOC.^[3] The rapid microwave energy supplemented K₂Cr₂O₇–H₂SO₄ soil digestion followed by colorimetric measurement of Cr⁺³ absorption at 590 nm provides a reliable measurement of SOC. The results obtained by the rapid microwave energy supplemented colorimetric method are significantly accounted for 99.2% variations in SOC content (Fig. 1) measured by the LECO dry combustion method.^[3]

NEAR-INFRARED REFLECTANCE SPECTROSCOPIC (NIRS) ANALYSIS OF SOC

The NIRS measurement of SOC based on distinct spectral features of H–C, C–N, and H–O bonds in SOM within the visible and near-infrared regions of the spectrum.^[4,20] It has been reported that NIRS can be used

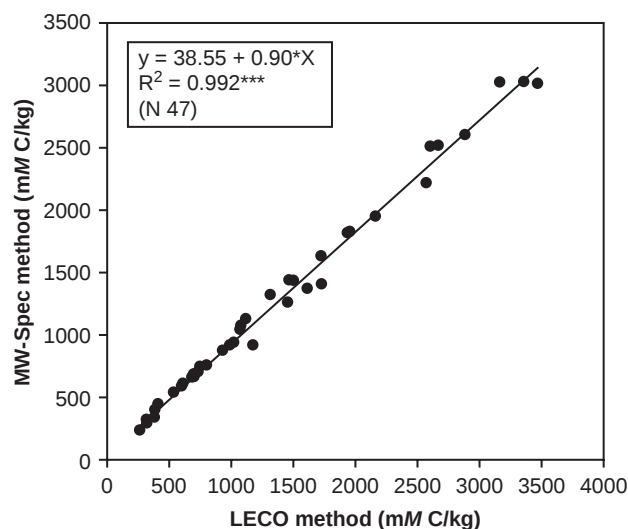


Fig. 1 Relationship between SOC measured by a rapid microwave digestion method and the LECO dry combustion method. **Source:** From Islam & Weil.^[3]

to measure independently and simultaneously SOC, inorganic C, and total N for soils with diverse C and N compositions.^[4]

COLORIMETRIC DETERMINATION OF SOIL LABILE OR ACTIVE C AFTER PERMANGANATE OXIDATION OF SOIL

Changes in small but relatively labile fractions of SOC may provide an early indication of soil quality in response to management practices.^[5,21–23] Most research on labile C has used the 0.333 M KMnO_4 to oxidize a fraction of soil C considered biologically active.^[5,21–23] A highly simplified colorimetric method has developed to determine active C after 2 minute shaking of air-dried soil with neutral dilute solutions (0.02–0.025 M) of KMnO_4 provides reliable and consistent results in response to soil management practices.^[5] Results from the 0.02 M reactive labile C significantly accounted for 81% variations in SOC content (Fig. 2) determined by LECO dry combustion method.^[5]

CONCLUSION

C, the principal component of SOM, exerts profound effects on soil quality. Among the methods used to determine SOC, automated dry combustion method provides an accurate measure of SOC over other methods. The developments in rapid dichromate oxidimetric, colorimetric methods that involve refluxing under external heating provide SOC values comparable to those obtained by automated dry combustion methods. A simple and rapid measurement of soil labile C by mild permanganate oxidation provides opportunities for early detection of changes in

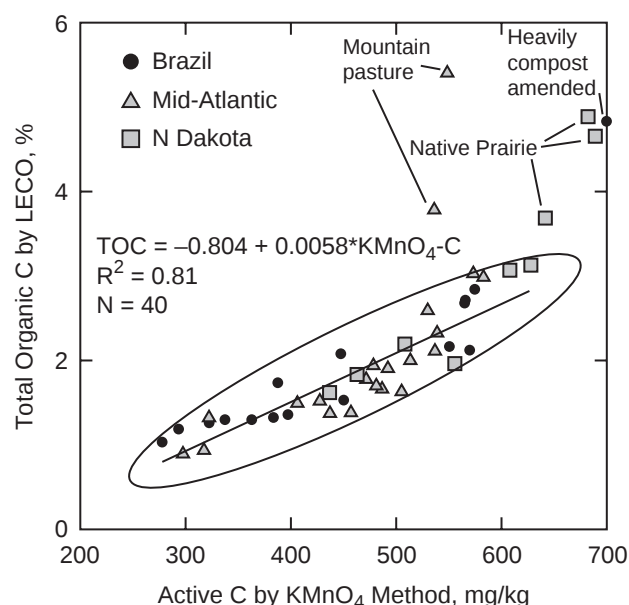


Fig. 2 Relationship between soil active C measured by a rapid permanganate oxidation method and the SOC measured by the LECO dry combustion method. **Source:** From Weil, Islam, et al.^[5]

SOM quality. All available methods to determine SOC have inherent problems associated with them, and the user should select the method most applicable to the type of soils analyzed, expected recovery, and the required precision of the results.

REFERENCES

1. Nelson, D.W.; Sommers, L.E. Total carbon, organic carbon, and organic matter. In *Methods of Soil Analysis, Part 3—Chemical Methods*; Bigham, J.M., Ed.; SSSA Book Series: 5; American Society of Agronomy: Madison, 1996; 1001–1006.
2. Tabatabai, M.A. Soil organic matter testing: An overview. In *Soil Organic Matter: Analysis and Interpretation*; Magdoff, F.R., Tabatabai, M.A., Hanlon, E.A., Jr., Eds.; SSSA Spl. Publ. No. 46; Soil Science Society of America: Madison, 1996; 1–9.
3. Islam, K.R.; Weil, R.R. A rapid microwave digestion method for colorimetric measurement of soil organic carbon. *Commun. Soil Sci. Plant Anal.* **1998**, *29* (15 and 6), 2269–2284.
4. Chang, C.W.; Laird, D.A. Near-infrared reflectance spectroscopic analysis of soil C and N. *Soil Sci.* **2002**, *167*, 110–116.
5. Weil, R.R.; Islam, K.R.; Stine, M.A.; Gruver, S.B.; Liebig, S.E.S. Estimating active carbon for soil quality assessment: A simplified method for laboratory and field use. *Am. J. Altern. Agric.* **2003**, *18*, 3–17.
6. Rabenhorst, M.C. Determination of organic and carbonate carbon in calcareous soils using dry combustion. *Soil Sci. Soc. Am. J.* **1988**, *52*, 965–969.

7. Allison, L.E. Wet-combustion apparatus and procedure for organic and inorganic carbon in soil. *Soil Sci. Soc. Am. Proc.* **1960**, *24*, 36–40.
8. Nommik, H. A modified procedure for determination of organic carbon in soils by wet combustion. *Soil Sci.* **1971**, *111*, 330–336.
9. Warrington, R.; Peake, W.A. On the determination of carbon in soils. *J. Chem. Soc.* **1880**, *37*, 617–625.
10. Ames, W.J.; Gaither, E.W. Determination of carbon in soils and soil extracts. *J. Ind. Eng. Chem.* **1914**, *6*, 561.
11. Schollenberger, C.J. A rapid approximate method for determining soil organic matter. *Soil Sci.* **1927**, *24*, 65–68.
12. Walkley, A.; Black, T.A. An examination of the Degtjareff method for determining soil organic matter and a proposed modification of the chromic acid titration method. *Soil Sci.* **1934**, *37*, 29–38.
13. Mebius, L.J. A rapid method for the determination of organic carbon in soil. *Anal. Chim. Acta* **1960**, *22*, 120–121.
14. Nelson, D.W.; Sommers, L.E. A rapid and accurate procedure for estimation of organic carbon in soils. *Proc. Indiana Acad. Sci.* **1975**, *84*, 456–462.
15. Yeomans, J.C.; Bremner, J.M. A rapid and precise method for routine determination of organic carbon in soil. *Commun. Soil Sci. Plant Anal.* **1988**, *19*, 1467–1476.
16. Soon, Y.K.; Abboud, S. A comparison of some methods for soil organic carbon determination. *Commun. Soil Sci. Plant Anal.* **1991**, *22*, 947–954.
17. Graham, E.R. Determination of soil organic matter by means of a photoelectric colorimeter. *Soil Sci.* **1948**, *65*, 181–183.
18. Sims, J.R.; Haby, V.A. Simplified colorimetric determination of soil organic matter. *Soil Sci.* **1971**, *112*, 137–141.
19. Heanes, D.L. Determination of total organic-C in soils by an improved chromic acid digestion and spectrophotometric procedure. *Commun. Soil Sci. Plant Anal.* **1984**, *15*, 1191–1213.
20. Henderson, T.L.; Baumgardner, M.F.; Franzmeier, D.P.; Stott, D.E.; Coster, D.C. High dimensional reflectance analysis of soil organic matter. *Soil Sci. Soc. Am. J.* **1992**, *63*, 865–872.
21. Loginow, W.; Wisniewski, W.; Gonet, S.S.; Ciescinska, B. Fractionation of organic carbon based on susceptibility to oxidation. *Pol. J. Soil Sci.* **1987**, *20*, 47–52.
22. Blair, G.J.; Lefroy, R.D.B.; Lise, L. Soil carbon fractions based on their degree of oxidation, and the development of a carbon management index for agricultural systems. *Aust. J. Agr. Res.* **1995**, *46*, 459–1466.
23. Bell, M.J.; Moody, P.W.; Connolly, R.D.; Bridge, P.J. The role of active fractions of soil organic matter in physical and chemical fertility of ferrosols. *Aust. J. Soil Res.* **1998**, *36*, 809–819.

Organic Carbon: Sequestration Concepts

Kenneth R. Olson

*Department of Natural Resources and Environmental Sciences,
University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.*

Abstract

One source of the conflicting soil organic carbon (SOC) sequestration findings relates to the general nature of the definition of SOC sequestration. The sequestered SOC process should increase the net SOC stock at the end of a study to above the pretreatment SOC stock baseline and result in a net reduction in the carbon dioxide (CO₂) levels in atmosphere. In agricultural land areas, no-tillage (NT) systems have been proposed to replace moldboard plow and chisel systems as a way to sequester SOC. Numerous estimates have been published on SOC sequestration rates as a result of a switch to NT systems. For SOC sequestration to occur as a result of a treatment applied to a land unit, all of the SOC sequestered must have come from atmosphere and be transferred into the soil humus through the unit plants, plant residues, and other organic solids. To unequivocally demonstrate that SOC sequestration at a specific site has occurred, a temporal increase must be documented relative to pretreatment SOC level and linked to a net depletion of atmospheric CO₂.

INTRODUCTION

The increase in atmospheric levels of carbon dioxide (CO₂) has been largely due to the burning of fossil fuels, deforestation, cultivation of the grasslands, drainage of the land, and land use changes. This has led to an increase in greenhouse gases (GHGs) and created concerns about the potential for long-term climate change and interest in developing methods to sequester some of this atmospheric carbon (C). In agricultural land areas, no-tillage (NT) systems have been proposed to replace moldboard plow (MP) and chisel plow (CP) systems as a way to sequester soil organic carbon (SOC). Numerous estimates have been published on SOC sequestration rates as a result of a switch to NT systems. Other researchers have proposed the use of cover crops, synthetic fertilizers, organic fertilizer, manure, liming, agricultural systems and management, agroforestry, forages, compost, crop rotations, and reduced row crop use as ways to sequester SOC. The amount of SOC present in the soil humus at the end of the study has to be greater than the pretreatment SOC levels in the same land unit, and as a result, there needs to be a net depletion of atmospheric CO₂. One source of the conflicting SOC sequestration findings relates to the general nature of the definition of SOC sequestration. The sequestered SOC process should increase the net SOC stock at the end of a study to above the pretreatment SOC stock baseline and result in a net reduction in the CO₂ levels in atmosphere. In agricultural land areas, NT systems have been proposed to replace MP and CP systems as a way to sequester SOC. Numerous estimates have been published on SOC sequestration rates as a result of a switch to NT systems. For SOC

sequestration to occur as a result of a treatment applied to a land unit, all of the SOC sequestered must have come from atmosphere and be transferred into the soil humus through the unit plants, plant residues, and other organic solids. When claiming that SOC sequestration at a site has happened, a temporal increase must be measured relative to pretreatment SOC stock and linked to a net reduction in the atmospheric CO₂.

Application of the SOC Sequestration Concept to a Specific Site

One source of the conflicting SOC sequestration findings relates to the general nature of the definition. SOC sequestration was defined by Olson^[1] as a “Process of transferring CO₂ from the atmosphere into the soil of a land unit through unit plants, plant residues and other organic solids, which are stored or retained in the unit as part of the soil organic matter (humus).” The retention time of sequestered C in the soil (terrestrial pool) can range from short-term (not immediately released back to atmosphere) to long-term (millennia) storage. The sequestered SOC process should increase the net SOC storage during and at the end of a study to above the previous pretreatment baseline levels and result in a net reduction in the CO₂ levels in atmosphere. The land unit concept with borders was added to the definition previously proposed by Olson^[2] to add clarity and to prevent the loading or adding SOC stocks to the land unit soil naturally or artificially from external sources without accounting for the C additions. C not directly from the atmosphere and from outside the land unit should not be counted as sequestered SOC. These external inputs could

include organic fertilizers, manure, plant residues, and topsoil or natural input processes such as erosion of a sloping soil and sediment-rich C deposition on a soil located on a lower landscape position or in a waterway. The land unit could be a plot, plot area, parcel, tract, field, farm, landscape position, landscape, wetland, forest, or prairie with defined and identified boundaries. This entry only discusses SOC sequestration as defined in the proposed definition and not soil inorganic carbon (IC), OC, or C sequestration.

Atmospheric C is cycled to the plant by photosynthesis, the plant cycles the organic C to the soil as residue, and it becomes humus or soil organic matter (SOM). It is impossible for most researchers, with the possible exception of modelers, to quantify changes in both the terrestrial and atmospheric pools. Therefore, the soil sequestration definition needs meaningful boundaries to be used by the researchers who want to measure actual changes in a specific part of a terrestrial (soil) pool.

Determination of Long-Term SOC Levels and Trends in Agricultural Lands

U.S. researchers have reported that the SOC content of timberland and prairie soils declines by 20–30% with cultivation.^[2,3] Rapid losses frequently occur in the first few years after clearing and cultivation; land drainage and land use change can result in additional SOC losses for many more years with the most significant being within the first 5–40 years.

Depth Distribution of SOC (Surface Layer vs. Root Zone)

When determining SOC sequestration, storage, and retention, it is important to include the entire root zone commonly to a depth of 1–2 m unless there is a root-restricting layer present, such as a fragipan or bedrock. Some tillage systems can enhance the SOC storage or retention in the surface and subsurface layers while decreasing it in the subsoil layer. The SOC needs to be totaled for the entire

root zone. Most soil sampling techniques use distance from the soil surface as a primary metric. Deep SOC profiles can differ between tillage and other treatments.

Identify Appropriate Experimental Designs for Use in Determining SOC Sequestration

Much of the literature^[4–8] suggests that paired comparison method, among various tillage treatments with one treatment, such as MP, used as baseline or control can be used to determine SOC sequestration rates. These comparison plots are often sampled only once during or at the end of a long-term study to determine the rate of SOC sequestration (Table 1). These researchers apparently assumed that the treatment used as baseline (often MP) at the end of a study had the same SOC stock it had prior to the treatment application or at the start of a long-term study. The MP baseline level of SOC was then maintained at a steady state during the study. If true, then any increase in SOC of the comparison treatment (NT) above the baseline treatment (MP) at the end of study would represent the amount of SOC sequestered (Table 1). However, these studies often lack or do not report the pretreatment SOC stock of the baseline treatment.^[2,9] Without such pretreatment SOC data for the baseline treatment (MP), the NT SOC sequestration rate findings cannot be verified.^[2,10]

If SOC in comparison treatment (NT) at the end of experiment was higher than that in the baseline treatment (MP) before the tillage treatments were applied, then SOC sequestration occurred. Alternatively, when the comparison treatment (NT) did not have more SOC stock at the end of the study than the baseline treatment at the start and the end of the study, then no SOC sequestration occurred. If the SOC sequestration rate is based on an NT comparison with SOC stock of the baseline treatment (MP) in the final study year^[11] when baseline treatment is not at a steady state, findings might not be correct (Table 1). It is possible that all treatments including baseline treatment (MP) lost SOC over time, and if so the differences between treatments at the end of the study only mean that one treatment (NT) lost

Table 1 Twenty-year effects of tillage treatments on the net SOC stock ($\text{Mg C ha}^{-1} \text{ layer}^{-1}$) amount of difference and SOC retention rate differences on the Grantsburg soil using the paired comparisons method.

Tillage treatment (six replications)	Depth (cm)	June 2009 SOC stock after 20 years of treatment ($\text{Mg C ha}^{-1} \text{ layer}^{-1}$)	June 2009 comparison method with SOC retention difference between NT and MP ($\text{Mg C ha}^{-1} \text{ layer}^{-1}$)	Paired comparison method with MP as baseline, NT vs. MP 20-year total SOC retention rate above 2009 MP baseline ($\text{Mg C ha}^{-1} \text{ layer}^{-1} \text{ yr}^{-1}$)
NT	0–15	25.2a	+7.9	0.40
	15–75	20.1a	+1.2	0.06
	0–75 (all)	45.3a	+9.1	0.46
MP	0–15	17.3b		
	15–75	18.9a		
	0–75 (all)	36.2b		

Note: Mean of six replications with the same letter in the same year and depth with a different tillage treatment is not significantly different at $P = 0.05$.

less SOC or retained more SOC in storage or is losing SOC at a lower rate (Table 1). When the pretreatment SOC for baseline treatment (MP) at the beginning of the long-term experiment is higher than at the end, no SOC sequestration can be claimed for any other comparison treatment (NT) without subtracting this baseline treatment SOC stock decrease from NT.^[1,2]

Develop a Procedure with Pretreatment Measurements of SOC Levels

The use of the comparison method with MP as baseline without establishing the pretreatment SOC content of the baseline treatment before the establishment of experiment could, in some cases, overestimate the amount of SOC sequestration, i.e., the SOC sequestration rate, and underestimate the amount of GHG released to the atmosphere from SOC during the study. SOC losses can occur from water erosion, MP tillage mixing, disturbance during planting in NT, disturbance when nitrogen (N) injection occurs in corn (*Zea mays* L.) years, aeration, and mineralization. This can and often does offset the SOC gains from plant and root growth and residue being returned to the humus or SOM.

Previously eroded soils with low SOC content were identified as having significant potential to sequester SOC.^[8,12] For 20 years, Olson^[2] studied previously eroded soils on 6% slopes with low SOC stocks in an attempt to quantify the amount and rates of SOC storage and retention as a result of a conversion to NT system. Pretreatment SOC baseline of the plot area was used to determine SOC loss from NT and MP plots.^[2] No SOC sequestration actually occurred in the NT and MP plots since the SOC level of the plot area was higher at the start of the experiment than at the end of the study.^[2] The NT plots did retain or store more SOC after 20-year tillage study than MP plots (Table 1). However, there was no increase in sequestered SOC in NT system.^[1,2]

Findings suggest that a pretreatment SOC baseline is essential in all tillage studies to determine the amount and rate of SOC sequestration, steady state, and loss. A pretreatment SOC baseline was needed in these tillage studies when determining the amount and rate of SOC sequestration, storage, retention, and loss, especially on sloping and eroding soils with more intensive cropping rotations (more row crops and fewer years of forages) during the study than in years preceding the study. Clearly, MP treatment was not at steady state during the 20-year tillage experiment.^[1,2] In fact, MP plots lost 15.2 Mg C ha⁻¹ from the root zone as a result of mixing, intensity of crop rotation, aeration, and eroded SOC-rich sediments being transported off the plot area.^[2] NT treatment only reduced the magnitude and rate of SOC loss over time. The pair comparison method used by many researchers with MP as baseline suggested that +9.1 Mg C ha⁻¹ of SOC sequestration occurred (Table 1) during the 20-year experiment. However, the 1988 baseline method did not validate the SOC sequestration rate.^[2] At

this site, the sloping and eroding NT plot actually lost 6.8 Mg C ha⁻¹ during the 20-year study, so no actual SOC sequestration occurred.^[2] The assumption that the MP was at a steady state, made by researchers using the comparison method with 1 year of SOC sampling near the end, was incorrect and resulted in a SOC sequestration finding that was invalid.^[2]

It is true that if a researcher had decided to use MP instead of NT, the amount of SOC retained in the soil after 20 years of MP treatment^[2] would be significantly less (−9.1 Mg C ha⁻¹) than after 20 years of NT treatment, and therefore, the GHG emissions from the SOC in storage or retained in the NT plots would be lower than those from the SOC in storage or retained in the MP plots. During the 12-year experiment, the SOC stocks in the subsoil and for the entire root zone increased by 2012 under the NT, CP, and MP systems with cover crops (Table 2). The SOC of the 0–15 layer increased under NT but not under CP and MP. By 2012, cover crop treatments raised SOC stocks significantly for the rooting zone of all tillage treatments except CP. Using the pretreatment baseline 2000 SOC stocks, NT, CP, and MP with cover crops sequestered SOC in root zone.^[13] The results shown in Table 2 illustrate how the pretreatment method can be used to validate the SOC findings.

To determine whether SOC sequestration has occurred, it is critical that researchers establish the baseline SOC stock prior to any treatment applications including cover crops. NT, CP, and MP treatments with cover crops did sequester and maintain more SOC stocks above the 2000 pretreatment baseline levels. The SOC gains from cover crops were equal to or more than any loss from any disturbance or mixing during planting, N injection in corn years, and erosion during corn and soybean production. SOC sequestration occurred in root zone (0–75 cm) for all of the tillage treatments with cover crops which were compared with 2000 pretreatment SOC stock levels (baseline) which validated the findings.

Rate of SOC Stock Change

It is not possible to determine a rate of change if all the data are collected at one time and near the end of the study. A rate of change study requires the sampling at two or more different times during the length of the study. The way some researchers get the rate of change is by assuming that the SOC stock for the baseline, usually MP, is at a steady state. That allows the researcher to assume that the SOC of the MP plots was the same before and after the tillage treatment application. Then, the NT gain above the baseline (MP) in measured year is divided by the number of years since the start of experiment to get the SOC sequestration rate.

Baseline Steady-State Assumption

All of these land use and agricultural system changes over time have had an impact on the SOC levels and trends (gains,

Table 2 Use of pretreatment 2000 SOC stock values for each tillage plot as baseline to determine the total SOC change in 12-year cover crops for the Grantsburg soils.

Cover crops by tillage treatment	Depth (cm)	September 2000 SOC stock (Mg C ⁻¹ ha ⁻¹ layer ⁻¹)	June 2012 SOC stock (Mg C ⁻¹ ha ⁻¹ layer ⁻¹)	12-year tillage effect on SOC total change by treatment using baseline		12-year tillage effect on SOC rate of change
				Mg C ⁻¹ ha ⁻¹ layer ⁻¹	% change	Mg ⁻¹ ha ⁻¹ layer ⁻¹ yr ⁻¹
NT						
Yes	0–15	26.8a	31.7b	5.0	19	0.42
	15–75	20.2a	29.5b	9.3	46	0.78
	0–75 (all)	47.0a	61.1b	14.3	30	1.21
CP						
Yes	0–15	25.0a	23.7a	−1.3	−5	−0.11
	15–75	18.7a	24.2b	5.5	29	0.46
	0–75 (all)	43.7a	47.9a	4.2	10	0.35
MP						
Yes	0–15	19.9a	18.9a	−1.0	−5	−0.08
	15–75	17.8a	25.4b	7.6	36	0.63
	0–75 (all)	37.7a	44.3b	6.6	14	0.55

Note: Mean of three replications with the same letter, in different years, same tillage treatment, and depth with cover crops is significantly different at $P = 0.05$.

steady state, and loss) in the United States. Each change affected the SOC stocks and trends. Some management practices raised the SOC levels and some lowered them, but the SOC of the soils seldom reached a true “steady state” since many of these land use and practice changes were adopted at various times and had an impact over different time periods. It usually takes years to reach a steady state, especially if the soils are sloping and eroding, and SOC-rich sediment is being transported from the land unit.

Microbial Role in SOC Sequestration, Net Storage, or Retention Rate Change over Time

The primary reason that SOC sequestration amounts and rates are often overestimated in tillage and management systems is a result of enhanced microbial activity over time. During the decomposition process, the C in the residue and some of the SOC in storage are released to the atmosphere as CO₂. This release rate of CO₂ can be accelerated over time, as more residue and amendments are added to the soil. During a long-term experiment of 20–50 years, much of the SOC formed in early years of the study will be released before the end of the experiment. If net SOC storage is increased in the 1st year at a rate of 0.5 Mg C ha⁻¹ yr⁻¹, the rate will drop the next year as a result of microbial activity. The 2nd year's net gain (0.5 Mg C ha⁻¹ yr⁻¹ rate) would be the same as in the 1st year, but the 1st year's rate will be reduced by the microbial release of some of the 1st year's SOC storage and effectively lower the annual rate for the 1st year. This will happen each year with some of the previous year's net SOC storage being released in the subsequent year as a result of water erosion, mixing by tillage,

disturbance during planting, microbial activity, when crop rotation is intensified, when synthetic N is injected, aeration, and mineralization, which would effectively lower the net annual SOC sequestration rate in a long-term study.

Account for the Loading of C-Rich Amendments

The loading of C-rich amendments by researcher or the natural deposition of SOC-rich sediment^[2,9] from outside the land unit needs to be accounted for when claiming SOC sequestration. The proposed definition of SOC sequestration requires that the land unit used has specified boundaries and only C gains from the atmosphere to the soil humus to be based on C inputs from the land unit plants and roots and their residue only. Any loading of external C-rich amendments (not directly from the atmosphere through unit plants to the soil humus) such as manure and organic fertilizers must be accounted for in the analysis and should be deducted from the amount claimed as SOC sequestration since it was not created in the land unit and did not lead to a reduction in the atmospheric CO₂ levels. The treatment applied could still result in SOC sequestration. The C loading on a land unit would most likely increase the amount of CO₂ given off, from the land unit during decomposition to the atmosphere even if some additional humus formation were to occur in the land unit as a result of the C loading.

The loading of C-rich amendments from an external source onto a land unit creates a number of issues. Some researchers will want to claim part of the C-rich amendment as sequestered SOC. This could be in the form of OC amendments (manure, saw dust, or organic fertilizers), IC

amendments (lime or dolomite), and SOC amendments (new topsoil or sediment deposition). These amendment cases illustrate the necessity of specifying the land unit and origin of the C sources in order to accurately measure the change in SOC derived from the atmosphere. This specification prevents the loading of C-rich amendments from outside the land unit boundaries and claiming the C additions as SOC sequestration when the SOC was not sequestered in the land unit and the atmosphere was not depleted of CO₂ as a result. The addition of land unit boundaries in the definition of SOC sequestration prevents the overestimation or underestimation of SOC sequestered by C loading. The C-rich amendments from external sources could in some cases eventually raise the land unit humus or net SOC storage levels, but that C addition to the humus was not sequestered SOC and may increase GHG emissions.

CONCLUSION

It is important to determine SOC sequestration rates for various agricultural land treatments and to establish a protocol to validate the amount and rate of SOC sequestration. Previous soil science research has built an important beginning foundation to assess the capacity of soil to store and sequester C. However, there are inconsistencies and errors in the application of SOC sequestration concept, plot area experiment designs, and depth of measurements. Most critical is the inability to accurately verify C drawn from the atmosphere and sequestered in the soil. Land unit boundaries need to be incorporated into the definition of SOC sequestration when applied to a specific site that pretreatment baseline measurements are essential to all studies, and that field experiments be designed to more carefully measure, monitor, and assess internal and external inputs. That amount of SOC loss from the soil storage during the time of experiment needs to be subtracted from SOC sequestration amount to determine the change in net SOC storage. The proposed protocols that have been discussed earlier are necessary to move the science forward and to attempt to address climate trends. The amount of SOC sequestered as a result of alternative agricultural systems such as NT and its effects on the net SOC storage changes in terrestrial pool and SOC released where the water and atmospheric pools need to be measured or calculated. If the losses from the terrestrial pool are greater than the gains in SOM during the time of the experiment, then no net SOC

sequestration would have occurred, and the release of CO₂ gas would have increased rather than being depleted in the atmosphere.

REFERENCES

1. Olson, K.R. Soil organic carbon sequestration in U.S. cropland: Protocol development. *Geoderma* **2013**, *195*–196, 201–206.
2. Olson, K.R. Impacts of tillage, slope and erosion on soil organic carbon retention. *Soil Sci.* **2010**, *175*, 562–567.
3. Johnson, M.G.; Kern, J.S. *Sequestering C in Soils: A Workshop to Explore the Potential for Mitigating Global Climate Change*; USEPA Rep. 600/3-91-031; USEPA: Corvallis, 1991; 85 pp.
4. Franzluebbers, A.J. Achieving soil organic carbon sequestration with conservation agricultural systems in the south-eastern United States. *Soil Sci. Soc. Am. J.* **2010**, *74*, 347–357.
5. Johnson, J.M.F.; Reicosky, D.C.; Allmaras, R.R.; Sauer, T.J.; Venterea, R.T.; Dell, C.J. Greenhouse gas contributions and mitigation potential of agriculture in the central USA. *Soil Till. Res.* **2005**, *83*, 73–94.
6. Liebig, M.A.; Morgan, J.A.; Reeder, J.D.; Ellert, B.H.; Gollany, H.T.; Schuman, G.E. Greenhouse gas contributions and mitigation potential of agricultural practices in northwestern USA and western Canada. *Soil Till. Res.* **2005**, *83*, 25–52.
7. Franzluebbers, A.J.; Follett, R.F. Greenhouse gas contributions and mitigation potential is agricultural region of North America. *Soil Till. Res.* **2005**, *83*, 1–8.
8. Lal, R. Soil management and restoration for C sequestration to mitigate the accelerated greenhouse effect. *Prog. Environ. Sci.* **1999**, *1*, 307–326.
9. Khan, S.A.; Mulvaney, R.L.; Ellsworth, T.R.; Boast, C.W. The myth of nitrogen fertilization for soil carbon sequestration. *J. Environ. Qual.* **2007**, *36*, 1821–1832.
10. Sanderman, J.; Baldock, J. Accounting for soil carbon sequestration in national inventories: A soil scientist's perspective. *Environ. Res. Lett.* **2010**, *5*, 1–6.
11. West, T.O.; Post, W.M. Soil organic carbon sequestration rates by tillage and crop rotation: A global data analysis. *Soil Sci. Soc. Am. J.* **2002**, *66*, 1930–1946.
12. Lal, R.; Kimble, J.M.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Sleeping Bear Press, Inc., Ann Arbor Press: Chelsea, 1998; 128 pp.
13. Olson, K.R.; Ebelhar, S.A.; Lang, J.M. Long-term effects of cover crops on crop yields, soil organic carbon stocks and sequestration. *Open J. Soil Sci.* **2014**, *4* (7), 284–292.

Organic Carbon: Subsoil Pools

Klaus Lorenz

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Routine soil surveys are restricted to about 1 m depth but more soil organic carbon (SOC) is stored globally below this depth (1325 Pg C to 1 m depth vs. 3000 Pg C to soil profile depth). Deposition of carbon (C) from roots and associated biota is the main subsoil C input and inevitable where deep soils coincide with deep rooting. The SOC at depth is characterized by slow turnover times, as environmental conditions in subsoil are less favorable for decomposition, and a range of physical and biological mechanisms stabilize SOC. Thus, subsoil SOC sequestration can contribute to climate change mitigation. However, SOC in subsoil may also respond dynamically on decadal timescales to environmental change, and more research is needed on this poorly understood pool of the contemporary C cycle.

INTRODUCTION

Soils comprise the largest pool of organic carbon (OC) in the terrestrial biosphere. The vertical distribution of soil organic carbon (SOC) within soil profiles is, however, unclear. In the subsoil below 30 cm depth, soil organic matter (SOM) stocks and fluxes are crucial for the soil functions. Aside from storing soil inorganic carbon and SOC, the subsoil contributes to the cycling of elements with consequences, e.g., for groundwater chemistry, in plant nutrition. Despite lower concentrations, the subsoil is probably also an important factor for the long-term storage of SOC, as the radiocarbon age and turnover time of organic matter (OM) increase with increase in soil depth. Associations of OM with pedogenic minerals (i.e., mineral–organic associations) result in the long-term stabilization of OC especially in the subsoil. In the deep soil profile, environmental conditions are detrimental to decomposition. Increases in the relative amount of mineral-associated carbon (C) compared to C relatively free of mineral particles at depth may contribute to SOC stabilization. The OM in the subsoil has therefore the potential to contribute to the mitigation of atmospheric increases in carbon dioxide (CO₂) by SOC sequestration.

SOURCES OF SUBSOIL OC

Among sources of subsoil OC are surface residues incorporated into the mineral soil by physical mixing and solubilization, transport, and subsequent adsorption.^[1]

Soil burial by volcanic, aeolian, alluvial, colluvial, glacial, and anthropogenic depositional processes also contributes to deep SOC storage.^[2] However, plant roots (i.e., litter and rhizodeposition) are the primary vector for most C entering the subsoil.^[3] The relative importance of surface residue incorporation vs. root litter and rhizodeposition for profile SOC distribution and dynamics depends on climate, soil, and vegetation types.^[4] Residues of plant litter decomposition, SOM, and compounds released by microorganisms are the major sources of water-soluble organic C. The dissolved organic carbon (DOC) is transported from upper to deeper soil layers with the percolating water. The organic C contained in the soil aqueous phase represents a pool of highly mobile, predominately solvated molecular fragments of relatively small molecule size.^[5] With increasing depth, DOC may be strongly adsorbed to poorly crystalline iron and aluminum (hydr)oxides with a high specific surface area resulting in the stabilization of OC.^[6] In contrast to studies in temperate forest soils, there are only a few studies on DOC in temperate arable and grassland soils and in tropical agroecosystems.^[7] For example, the process of DOC movement and retention was responsible for 9% of the total SOC pool to 1 m depth in a temperate grassland soil.^[8] The soil fauna (i.e., earthworms, ants, and termites) translocate OM to deeper soil layers by burrowing activities.^[9] In particular, earthworm activity may be of great significance to the long-term stabilization of OM in the soil.^[10] However, the major determinant of the relative distribution of SOC with depth is probably the allocation of net primary productivity

Table 1 Global rooting depths and vertical distribution of SOC.

Parameter	Grasses/grasslands	Shrubs/shrublands	Trees/forests	References
Maximum rooting depth	15.0 m	5.2 m	7.3 m	[12]
Additional SOC in 1–2 m relative to 0–1 m	0.30	0.38	0.29	[11]
Additional SOC in 2–3 m relative to 0–1 m	0.13	0.39	0.27	[11]

belowground as root litter and rhizodeposition.^[3] Thus, the vertical root distribution leaves distinct imprints on the depth distribution of SOC (Table 1).^[11] Roots (and their associated biota) contribute more C to SOM than aboveground residues. Because of their maximum rooting depths, grasses and trees transfer C to considerably deeper soil layers than shrubs, with maximum rooting depths of 68 m.^[12]

COMPOSITION OF SUBSOIL OC

The composition of the belowground plant and microbial C input to SOC may be substantially different from that of the aboveground input.^[13] However, beyond the decadal time frame, selective preservation of relatively unaltered plant-derived compounds due to biochemical recalcitrance (i.e., aliphatic compounds such as lipids and waxes) may not be as an important long-term stabilization mechanism as previously thought.^[14,15] Similarly, microbial-derived OM may not be recalcitrant or composed of complex compounds, as mean residence times of microbial biomarkers in soils do not exceed several hundred years.^[16] Any input of OM to soils can increase the subsoil SOC pool when it is stabilized and not completely mineralized to CO₂.^[17] Charring of OM creates biochemically recalcitrant black carbon which may be stabilized preferentially at deeper soil depths.^[18] At lower soil depths, decomposed plant material may directly interact with metal oxides resulting in OM stabilization, whereas in deep mineral soil, OM could mainly interact with metal oxides after microbial turnover.^[19] In subsoil horizons, easily available polysaccharides and nitrogen-rich microbial material

may be stabilized by interactions with the mineral phase but not lignin.

QUANTITY OF SUBSOIL OC

Routine soil surveys are restricted to about 1 m depth with exploring greater depths of profile development, and measurements of SOC stocks below 1 m depth are rare.^[20] Estimates for the vertical distribution of SOC are highly uncertain (Table 2). The reliability of estimates of SOC storage is affected by unreliable *in situ* measurements of SOC, soil depth, and bulk density.^[23] Variability in the estimates of global SOC distribution is due to variation in the definitions of soil units, differences in soil property databases, scarcity of information about SOC at depths >1 m in peatlands, and variation in the definitions of “peatland.”^[22] Given the limitations, 39–70% of the total SOC in the upper 1 m are stored in the first 30 cm.^[21] The global SOC stock increases by 30–60% when the second meter of soil is included and by additional 18% when the third meter is included.^[11,21] About 56% of the total profile SOC stock without depth limit is found below 1 m depth.^[22]

In particular, the root OM may be translocated below 1 m depth as is indicated by the maximum rooting depths for grasses, shrubs, and trees (Table 1) and by the global average maximum rooting depth of 4.6 m.^[12] The root densities, however, are highest in the upper 20 cm of the soil profiles.^[24] It is estimated that soils in permafrost regions store globally more SOC below 1 m than up to 1 m depth (i.e., 1176 Pg C vs. 496 Pg C).^[25] Further, peatlands may contain about 6% of the tropical SOC mass within the first meter (22.0 Pg C) and approximately 21% of the total tropical SOC mass (88.6 Pg C; without depth limit).^[26] Estimated totals of SOC stock for the second and third meters indicate that globally tropical evergreen forests, grasslands, and savannas store most SOC between 1 and 3 m depth.^[11]

The SOC stocks in the upper meter of the most soil profiles drop off quickly, but stocks in underlying horizons are not negligible (Table 1). Especially where deep soils (profiles greater than 5 m deep) coincide with deep rooting, the biological deposition of C from roots and their associated biota is inevitable at great soil depths; e.g., soils from sites in the deeply weathered regolith in Australia contained 59% of the total SOC store to bedrock (mean depth 21 m) in the surface 5 m.^[27]

Table 2 Global estimates of the vertical distribution of SOC.

Depth (m)	Total SOC storage (10 ¹⁵ g C) estimated by		
	Batjes ^[21]	Köchy et al. ^[22]	Jobbágy and Jackson ^[11]
0–0.3	684–724		
0–1	1462–1548	1325	1342–1662
0–2	2376–2456		1758–2228
0–3			2104–2584
Without depth limit		3000	

SUBSOIL OC TURNOVER TIMES

Environmental conditions are important for regulating SOC turnover times.^[28] The slow turnover times and old radiocarbon ages of SOC in deep soils may be the result of 1) depressed microbial biomass and activity due to low rates of oxygen diffusivity, low soil moisture, and nutrient availability; 2) the microbial recycling of aged C compounds; and 3) aging during the slow, downward migration of C that escapes mineralization from upper soil horizons.^[2] Incorporation of OC into soil aggregates and associations with mineral surfaces, cations, or metals also contribute to slow turnover times of SOC at depth. However, deep SOC may be more dynamic as previously thought and may play a role in the contemporary C cycle. Specifically, the ¹⁴C age of SOM fractions is not necessarily related to molecular structure or thermodynamic stability.^[29] For example, modern ¹⁴C is incorporated into subsurface soils, and ancient SOC can respond to disturbances on annual to decadal timescales.^[30,31]

CONCLUSION

Studies on the vertical distribution of SOC below 1 m depth are rare. The subsoil may, however, receive appreciable amounts of OM as DOC with the percolating water, because of bioturbation by soil animals, by soil microorganisms, and by soil burial but probably most notably by root litter and rhizodeposition. As the turnover time of SOM increase with depth, the subsoil has the potential of C sequestration. Future work must focus on understanding the dynamics of deep SOC depending on land use and climate changes and how the subsoil OC stock can be enhanced. Soil ecosystem services will also benefit from higher amounts of subsoil SOC as, e.g., the subsoil contributes to plant nutrition.

REFERENCES

1. Lorenz, K.; Lal, R. The depth distribution of soil organic carbon in relation to land use and management and the potential of carbon sequestration in subsoil horizons. *Adv. Agron.* **2005**, *88*, 35–66.
2. Chaopricha, N.T.; Marín-Spiotta, E. Soil burial contributes to deep soil organic carbon storage. *Soil Biol. Biochem.* **2014**, *69*, 251–264.
3. Rasse, D.P.; Rumpel, C.; Dignac, M.F. Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. *Plant Soil* **2005**, *269*, 341–356.
4. Rumpel, C.; Kögel-Knabner, I. Deep soil organic matter—a key but poorly understood component of terrestrial C cycle. *Plant Soil* **2011**, *338*, 143–158.
5. Kleber, M.; Eusterhues, K.; Keiluweit, M.; Mikutta, C.; Mikutta, R.; Nico, P.S. Mineral–organic associations: formation, properties, and relevance in soil environments. *Adv. Agron.* **2015**, *130*, 1–140.
6. Bolan, N.S.; Adriano, D.C.; Kunhikrishnan, A.; James, T.; McDowell, R.; Senesi, N. Dissolved organic matter: Biogeochemistry, dynamics, and environmental significance in soils. *Adv. Agron.* **2011**, *110*, 1–75.
7. Chantigny, M.H. Dissolved and water-extractable organic matter in soils: A review on the influence of land use and management practices. *Geoderma* **2003**, *113*, 357–380.
8. Sanderman, J.; Amundson, R. A comparative study of dissolved organic carbon transport and stabilization in California forest and grassland soils. *Biogeochemistry* **2008**, *89*, 309–327.
9. Wolters, V. Invertebrate control of soil organic matter stability. *Biol. Fert. Soils* **2000**, *31*, 1–19.
10. Bossuyt, H.; Six, J.; Hendrix, P.F. Rapid incorporation of carbon from fresh residues into newly formed stable microaggregates within earthworm casts. *Eur. J. Soil Sci.* **2004**, *55*, 393–399.
11. Jobbágy, E.G.; Jackson, R.B. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecol. Appl.* **2000**, *10* (2), 423–436.
12. Canadell, J.; Jackson, R.B.; Ehleringer, J.R.; Mooney, H.A.; Sala, O.E.; Schulze, E.-D. Maximum rooting depth of vegetation types at the global scale. *Oecologia* **1996**, *108*, 583–595.
13. Kögel-Knabner, I. The macromolecular organic composition of plant and microbial residues as inputs to soil organic matter. *Soil Biol. Biochem.* **2002**, *34*, 139–162.
14. Krull, E.S.; Baldock, J.A.; Skjemstad, J.O. Importance of mechanisms and processes of the stabilization of soil organic matter for modelling carbon turnover. *Funct. Plant Biol.* **2003**, *30*, 207–222.
15. Lorenz, K.; Lal, R.; Preston, C.M.; Nierop, K.G.J. Strengthening the soil organic carbon pool by increasing contributions from recalcitrant aliphatic bio(macro)molecules. *Geoderma* **2007**, *142*, 1–10.
16. Amelung, W.; Brodowski, S.; Sandhage-Hofmann, A.; Bol, R. Combining biomarker with stable isotope analyses for assessing the transformation and turnover of soil organic matter. *Adv. Agron.* **2008**, *100*, 155–250.
17. Kleber, M.; Johnson, M.G. Advances in understanding the molecular structure of soil organic matter: Implications for interactions in the environment. *Adv. Agron.* **2010**, *106*, 77–142.
18. Lorenz, K.; Lal, R. Biochar application to soil for climate change mitigation by soil organic carbon sequestration. *J. Plant Nutr. Soil Sci.* **2014**, *177*, 651–670.
19. Rumpel, C.; Baumann, K.; Remusat, L.; Dignac, M.F.; Barré, P.; Deldicque, D.; Glasser, G.; Lieberwirth, I.; Chabbi, A. Nanoscale evidence of contrasted processes for root-derived organic matter stabilization by mineral interactions depending on soil depth. *Soil Biol. Biochem.* **2015**, *85*, 82–88.
20. Scharlemann, J.P.W.; Tanner, E.V.J.; Hiederer, R.; Kapos, V. Global soil carbon: Understanding and managing the largest terrestrial carbon pool. *Carbon Manage.* **2014**, *5*, 81–91.
21. Batjes, N.H. Total carbon and nitrogen in the soils of the world. *Eur. J. Soil Sci.* **1996**, *47*, 151–163.

22. Köchy, M.; Hiederer, R.; Freibauer, A. Global distribution of soil organic carbon – Part 1: Masses and frequency distributions of SOC stocks for the tropics, permafrost regions, wetlands, and the world. *SOIL* **2015**, *1*, 351–365.
23. Jandl, R.; Rodeghiero, M.; Martinez, C.; Cotrufo, M.F.; Bampa, F.; van Wesemael, B.; Harrison, R.B.; Guerrini, I.A.; deB Richter, D., Jr.; Rustad, L.; Lorenz, K.; Chabbi, A.; Miglietta, F. Current status, uncertainty and future needs in soil organic carbon monitoring. *Sci. Total Environ.* **2014**, *468–469*, 376–383.
24. Schenk, H.J.; Jackson, R.B. The global biogeography of roots. *Ecol. Monogr.* **2002**, *72* (3), 311–328.
25. Tarnocai, C.; Canadell, J.G.; Schuur, E.A.G.; Kuhry, P.; Mazhitova, G.; Zimov, S. Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biogeochem. Cy.* **2009**, *23*, GB2023.
26. Page, S.E.; Rieley, J.O.; Banks, C.J. Global and regional importance of the tropical peatland carbon pool. *Glob. Change Biol.* **2009**, *17*, 798–818.
27. Harper, R.J.; Tibbett, M. The hidden organic carbon in deep mineral soils. *Plant Soil* **2013**, *368*, 641–648.
28. Schmidt, M.W.I.; Torn, M.S.; Abiven, S.; Dittmar, T.; Guggenberger, G.; Janssens, I.A.; Kleber, M.; Kögel-Knabner, I.; Lehmann, J.; Manning, D.A.C.; Nannipieri, P.; Rasse, D.P.; Weiner, S.; Trumbore, S.E. Persistence of soil organic matter as an ecosystem property. *Nature* **2011**, *478*, 49–56.
29. Kleber, M.; Nico, P.S.; Plante, A.; Filley, T.; Kramer, M.; Swanston, C.; Sollins, P. Old and stable soil organic matter is not necessarily chemically recalcitrant: implications for modeling concepts and temperature sensitivity. *Glob. Change Biol.* **2011**, *17*, 1097–1107.
30. Koarashi, J.; Hockaday, W.C.; Masiello, C.A.; Trumbore, S.E. Dynamics of decadal cycling carbon in subsurface soils. *J. Geophys. Res.* **2012**, *117*, G03033.
31. Paul, E.A.; Morris, S.J.; Conant, R.T.; Plante, A.F. Does the acid hydrolysis–incubation method measure meaningful soil organic carbon pools? *Soil Sci. Soc. Am. J.* **2006**, *70*, 1023–1035.

Organic Farming: China

Yuxin Miao

Fusuo Zhang

Fanghao Wang

College of Resources and Environmental Sciences, China Agricultural University, Beijing, China

Abstract

Although Chinese farmers have practiced traditional organic farming for over 4000 years, China's organic farming history started only in 1990 and has witnessed fast developments since 2000. This entry introduces the history, status and challenges, and future prospects of organic farming in China, as well as other eco-labeled foods (pollution-free and green foods).

INTRODUCTION

China has been internationally acclaimed during the past few decades for achieving food security for the largest population in the world and that too with a limited cropland area. However, this success was achieved at the cost of high external inputs, low resource use efficiencies, serious ecological and environmental problems, and increasing food safety concerns. To address the problems associated with conventional agriculture, China has explored different alternative practices, especially those requiring less external chemical inputs. As a result, the development of ecological agriculture, organic farming, and eco-labeled foods (pollution-free, green, and organic foods) have received increasing attention. This new development of organic farming complies with international organic farming standards and has only a short history in China. This entry introduces the definition, history, status, challenges, and future prospects of organic farming in China.

DEFINITION

Although there are many definitions, organic farming in this entry refers to agricultural management systems that are based on ecological and sustainable development principles and agronomic, biological, manual, or mechanical methods without using synthetic chemical fertilizers, pesticides, growth regulators, livestock feed additives, or genetically modified organisms during the production process and are internationally certifiable (with controls and traceability).^[1,2]

HISTORY OF ORGANIC FARMING IN CHINA

China has a long history of traditional organic farming. For over 4000 years, Chinese farmers used strategies of crop

rotation and diversification, intercropping, thorough tillage, terracing, various sources of organic manure (e.g., animal waste, human excreta, cooking ash, compost, and dredged sediments), nitrogen (N)-fixing green manures, and manual methods to produce food, maintain soil fertility, and control pests.^[3,4] Increase in population and urgency to achieve food security led to a replacement of this natural, subsistence-oriented traditional organic farming by conventional (or intensive) agriculture. This transition started during the middle of the 20th century and was based on the Green Revolution technologies to maximize agricultural output. The rich experience of millennia in traditional farming, however, formed a good foundation for the development of China's organic farming systems.

To address the problems of conventional agriculture, China started ecological agriculture in the beginning of the 1980s, which has specific limitations with regard to inputs of agrochemicals. By the mid-1990s, China had established about 1200 demonstration eco-villages and farms,^[2] which was the first step toward the development of organic farming in China. China started the Green Food program in 1990.

The Green Food is produced and processed with controlled and reduced use of synthetic inputs and domestically certified and labeled to be safe from chemical contamination.^[1,5] It is administered by China Green Food Development Center under the Ministry of Agriculture (MOA), which was established in 1992. Green Food production has seen steady increase since its start, due to reduced chemical input costs without productivity reduction and 10–50% price premium.^[5] The Green Food program has paved the way for China's organic farming development since 2000.

Also in 1990, China's Organic Green Tea from Lin'an County of Zhejiang Province was certified by the Dutch certifier SKAL and exported to the Netherlands, setting the milestone of organic farming in China. Nanjing Scientific

Research Institute's Rural Ecology Research Lab, under National Bureau of Environmental Protection (NBEP), joined International Federation of Organic Agriculture Movement (IFOAM) as China's first IFOAM member in 1989. This developed into Organic Food Development Center (OFDC) of NBEP in 1994 and later was renamed as OFDC of State Environmental Protection Administration (SEPA). The OFDC was finally given full accreditation by IFOAM in 2003, and Chinese organic products certified by OFDC can be sold as organic food around the world.^[2] During the same year, the administration of China's organic products certification was taken over from SEPA by Certification and Accreditation Administration of China (CNCA).

The year 2005 was an important year for China's organic farming development. The General Administration of Quality Supervision, Inspection and Quarantine issued the "Regulatory Measures on Organic Product Certification Management" and National Standard for Organic Products (GB/T 1963–2005). This standard includes the following four components: production, processing, logo, and marketing and management system.^[2] The CNCA published "Implementation Rules for Organic Product Certification," and a standard national logo for organic products was introduced.

China has seen the fastest organic farming development in the world, moving from 45th in 2000 in terms of hectares under organic management to 16th in 2005, and then to the 2nd position in 2006, accounting for 11% of world's total organic area.^[6] This rapid development was facilitated by the Green Food program, which acted as a stepping-stone for the farmers to move from conventional chemical farming to modern organic farming. Green Food started as a local standard and later was differentiated into Grade A and Grade AA. Grade AA is very close to organic food, prohibiting the use of any chemicals in the production process. It will be phased out gradually and merged with the international organic standard.^[1,6,7] A key difference between Green Food and organic certifications is that Green Food certifies both production and outcome, while organic certification certifies only the production process.

EXISTING STATUS

According to CNCA, the area under organic farming management was only 34.26×10^4 ha by the end of 2003 and increased to 97.8×10^4 ha (not including aquaculture, wild production, and poultry and livestock production) in 2005.^[2] In 2006, certified organic farm land area and production were 2.1 million ha (Mha) and 2.80 million tons, respectively, making China as the 3rd largest country in terms of area after Australia (11.8 Mha) and Argentina (3.9 Mha), according to the China Organic Food Certification Center (COFCC), as cited by Scoones.^[8] There was more than 2 Mha of wild collection (with 0.05 million ton

production) and additional 1.1 Mha in conversion (with 1 million ton production).^[8] If these were all added up, China would rank as the 2nd country in the world.

The main certified farm lands are located in Northeast China (Liaoning, Jilin, and Heilongjiang provinces) for grain and soybean production, East China (Shandong, Jiangsu, and Zhejiang provinces) for vegetable production, and South China (e.g., Zhejiang, Jiangxi, and Yunnan provinces) mainly for organic tea production. The processing companies are mainly distributed in developed areas of the east (i.e., Shanghai, Beijing, Zhejiang, Shandong, and Jiangsu provinces).^[9]

China's organic products are mainly export driven and oriented, especially during the early development stage. Before 1999, over 95% of organic products were exported to Japan (mainly vegetables and some field crops like rice and soybeans), the European Union (organic tea and field crops), and North America.^[8] According to COFCC, the value of exported organic products increased from US\$0.3 million in 1995 to US\$0.35 billion at the end of 2004, which accounted for 1.7% of total Chinese agricultural export and 1.2% of global trade in organic products. Domestic organic market started from 2000 and is concentrated mainly in some large cities, with a retail value of US\$750 million in 2006 (COFCC, as cited by Scoones).^[8] The price of conversion to organic vegetables was reported to be 3 to 5 times that of the conventional vegetables in Beijing and Shanghai, yet only 1.5 to 3.0 times in other cities such as Nanjing.^[2]

In addition to organic and green food, pollution-free food is another eco-labeled food, which was launched in 2002 and certified by the Center for Agri-Food Quality and Safety under MOA. This is a Chinese standard to address basic food safety issues and thus has lower standards compared with green and organic foods. Pollution-free food label is only in Chinese, while labels of the other two types of certified foods are in both Chinese and English, including conversion to organic food (Fig. 1). In 2007, there was a total of 34.18 Mha eco-labeled foods in China, with 21.07, 10, and 3.11 Mha for pollution-free, green, and organic food, respectively.^[7]

Organic food production in China is organized in the following three main models: 1) company owns organic production base; 2) company rents land from farmers for long term, and farmers are employees of the company; and



Fig. 1 Labels denote the types of food producers: (A) pollution-free food, (B) green food, (C) organic food, and (D) conversion to organic food.

3) “Company plus Farmers” model, which is the most widely adopted one based on contracts signed between a company and farmers. The farmers grow organic products in their own fields under the company’s guidelines, and the company purchases their products.^[9] One effective management method to ensure that the farmers will follow the guidelines is to group local contract farmers together for collective responsibility, and if one farmer has violations, none of the products from the group is purchased by the company.^[8]

CHALLENGES AND FUTURE PROSPECTS

Organic farming is at its early stages in China and lacks sufficient research, technology, and policy support. Organic farming is knowledge intensive and has higher requirements from the producers. However, many organic farming technologies are not mature in China, and most Chinese farmers have much less education and training than those in developed countries. Most young farmers with good education generally seek off-farm jobs. Thus, developing a good research, extension, and training system for organic farming technologies is crucially important and urgently needed. Crop yields are generally low during the initial conversion period, and it requires more financial and labor investment. Thus, farmers take some financial risks and need policy support. Ensuring organic farming production standards and marketing can also be challenging with scattered and small-scale production.

Achieving food security is the top priority in China. To meet the food demand of the increasing population, China needs to continue to increase crop yields. Crop yield in organic farming is generally lower than that in conventional intensive farming, and it requires rotation with leguminous crops or N-fixing green manures once every 2 or 3 years (or 1/3–1/2 of crop land each year) to maintain soil fertility. Therefore, organic farming is not likely to become the mainstream farming in China in the near future. Pollution-free and green food production systems, on the one hand, may become more widely accepted in China, as they can meet the food safety demand without sacrificing

crop yields. Organic farming, on the other hand, may gain more development in production systems with more emphasis on product quality rather than high agronomic yields.

REFERENCES

1. Giovannucci, D. *Organic Agriculture and Poverty Reduction in Asia: China and India Focus*; Report No. 1664; International Fund for Agricultural Development (IFAD): Rome, 2005.
2. Chinese Rural Technology Development Center of Ministry of Science and Technology. *Organic Agriculture in China*; Chinese Agricultural Science and Technology Press: Beijing, 2006.
3. Wittwer, S.; Youtai, Y.; Sun, H.; Wang, L. *Feeding a Billion: Frontiers of Chinese Agriculture*; Michigan State University Press: East Lansing, MI, 1987.
4. King, F.H. *Farmers of Forty Centuries—Organic Farming in China, Korea and Japan*; Dover Publications: New York, 2004.
5. Paull, J. Green food in China. *Elementals—J. Bio-Dynamics Tasmania*. **2008**, *91*, 48–53. Available at <http://orgprints.org/14720/> (accessed May 2000).
6. Paull, J. China’s organic revolution. *J. Org. Syst.* **2007**, *2* (1), 1–11.
7. Paull, J. The greening of China’s food—Green food, organic food and eco-labelling, Sustainable Consumption and Alternative Agri-Food Systems Conference (SUSCONS), Liege University, Arlon, Belgium, May 27–30, 2008. Available at <http://www.orgprints.org/13563> (accessed May 2000).
8. Scoones, S. *Organic Agriculture in China: Current Situation and Challenges*; An EU-China Trade Project Report; 2008. Available at http://www.euchinawto.org/index.php?option=com_content&task=view&id=233&Itemid=71 (accessed May 2000).
9. Zhang, F.; Qiao, Y.; Wang, F.; Zhang, W. A Perspective on organic agriculture in China—Opportunities and challenges, Paper presented at Zwischen Tradition und Globalisierung—9. Wissenschaftstagung Ökologischer Landbau, Universität Hohenheim, Stuttgart, Deutschland, Germany, Mar 20–23, 2007. Available at <http://orgprints.org/10388/> (accessed May 2000).

Organic Fertilization: Heavy Metal Uptake

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Alexandra Izosimova

Saint Petersburg Agricultural Physical Research Institute, Saint Petersburg, Russia

Renata Gaj

Institute of Soil Science, Agricultural University, Poznan, Poland

Abstract

Organic fertilizers affect the heavy metal balance of agroecosystems by supply, mobilization, and immobilization. Supply and mobilization are useful options in terms of feeding plants with essential heavy metals but are a risk for loading sensitive environments. Immobilization in and removal from soil of heavy metals by phytoremediation are deceptive options: The former leaves the risk in place, while the latter is too inefficient in terms of masses removed and just shifts the problem of contamination to other ecological compartments. The only feasible option for protecting soils, food, and environment is applying the precautionary principle by balancing all sources of input and output of heavy metals to an ecosystem.

INTRODUCTION

Heavy metals are elements with atomic weights ranging between 63.54 [copper (Cu)] and 200.59 [mercury (Hg)] and specific gravity greater than 4.0 g/cm³. Some elements may stimulate growth of an organism, although the evidence for this process is lacking. Micronutrients important to plant growth are Cu, cobalt, manganese (Mn), iron, zinc (Zn), and molybdenum (Mo). Other trace elements do not play any physiological function and are toxic in extremely small quantities.^[1,2] There is a strong link between micronutrient nutrition, the uptake, and the impact of contaminants of plants, animals, and humans.

Contamination with heavy metals has become one of the most serious problems for the functionality of ecosystems. Because of the industrial development and past disposal activities, heavy metals are considered to be among the most important environmental contaminants that affect terrestrial, aquatic, and atmospheric systems. Concerns about these contaminants are based on the fact that certain trace elements [e.g., arsenic, cadmium (Cd), lead (Pb), or uranium (U)] are accumulating in the food chain threatening the health of humans and animals.^[3] The risks of human, crop, and/or environmental toxicity posed by these elements are a function of their mobility and availability in soil.^[4] The concentration of heavy metals in the soil solution, as the main parameter for the bioavailability of heavy metals to plants, is affected by numerous factors. Among others, the uptake of heavy metals by plants from soils is also affected by management factors like organic fertilization. Despite the fact that organic fertilizers themselves can

be a source for heavy metals, nevertheless organic matter (OM) added with manures, compost, and sludges greatly increases soil physical features like colloidal stability and cation exchange capacity and chemical features like plant available nutrients and chelating compounds which have a beneficial effect on soil biology.

Organic fertilization may affect the transfer of heavy metals to plants by several mechanisms: enhancing metal adsorption as a result of increased surface charge and increasing the formation of organic complexes including changes of the redox potential in soils toward negative values. It may additionally not only increase the solubility of some elements like Mn but also decrease the solubility of other elements like U.^[5] The oxidation of OM releases protons and lowers the pH of the soil which increases Mn and Zn solution concentration or lowers the mobility of Mo.^[6] A higher cation exchange capacity and increased concentrations of chelating agents, because of the organic fertilization, improve soil's storage capacity for heavy metals. This function acts as a buffer system against heavy metal transfers in ecosystems, either by preventing them from leaching or by counteracting their accumulation in food plants.^[7] Metal retention processes are generally much more important than metal leaching processes.

SOURCES OF ORGANIC FERTILIZERS

Differences between organic fertilizers must also be considered. Some typical metal concentrations in organic fertilizers are given in Table 1. Narwal and Singh^[8] observed

Table 1 Typical heavy metal concentrations in selected organic fertilizers.

Organic fertilizers	Zn (mg/kg dry weight, range)	Cd (mg/kg dry weight, range)	Cu (mg/kg dry weight, range)	Ni (mg/kg dry weight, range)	Pb (mg/kg dry weight, range)	References
Manure						
Pig	206–716	0.19–0.53	160–780	3–24.3	1.01–4.65	[9,10]
Cattle	41–238	0.15–0.40	26.2–55.8	1.7–9.1	1.5–8.4	[10]
Poultry	350–632	0.44–2.04	49.4–74.8	4.5–11.4	3.36–14.8	[10]
Slurry						
Pig	639–2115	0.32–1.08	197–773	8.6–11.6	2.7–7.0	[11,12]
Cattle	68–235	0.11–0.53	17.5–48.7	1.9–20.4	4.1–8.4	[10,13]
Sewage sludge	700–50,000	2–1,500	50–3,300	16–5,300	50–3,000	[9]
Green compost	72–181	0.11–1.7	17–432	5.7–25	15–131	[14]
Biowaste compost	125–371	0.11–0.7	34–215	7.2–37.7	19–435	[15]

that increasing rates of cow and pig manure decreased the amounts of extractable Cd and nickel (Ni), whereas the addition of peat, at the same rate of as OM, had an increasing effect. At this point, the effects are linked to farming systems, which affect, either directly or indirectly, the quantities of compounds of plant or animal origin introduced into the soil.^[16]

All factors involved in the effects of OM with heavy metal mobility show distinct interaction, like pH and the formation of metal-organic complexes. Most metals (in free ionic form) have a higher mobility in acidic, coarse-textured soils,^[17] but mobility, however, can be also significant at about neutral or higher pH owing to the fact that dissolved OM becomes itself more soluble at those pH levels and because of the formation of soluble organometallic complexes. That is why the release of Cd into the soil solution as metal-organic complex becomes significant when the soil pH exceeds 5.5,^[4,18] and Pb leaching increases with a higher concentration of dissolved OM and pH.^[19] Such interactions are of great importance when the remediation of heavy metal polluted soils is discussed. In terms of preventing heavy metals from transferring to ecosystems, the aim of remediation is not decontamination but immobilization through increasing the binding capacity of the soil.^[20]

The largest sources for OM in agricultural systems are plant residues. Gupta et al.^[21] observed a considerably higher content of Cd in wheat grown after the addition of pulses than when the same crop was grown after cereals. This may indicate an effect of nitrogen applied with the organic fertilizers by a synergistic effect of the former on heavy metal uptake, which has been described by Schnug.^[22]

In the point of view of negative anthropogenic activities on soil quality, the loss of OM is a particularly serious problem, which also diminishes the buffer capacity of soils for heavy metals. The utilization of wastes, as soil amendments to counteract losses of OM,^[23,24] is only an apparent and short-sighted solution because these sources are usually highly polluted with heavy metals (Table 1).

PLANT UPTAKE OF HEAVY METALS

Typical heavy metal concentrations in not accumulating higher plants are (mg kg^{-1}) as follows: Cd, 0.1–1; Cu, 3–15; Hg, 0.1–0.5; Ni, 0.1–5; Pb, 1–5; and Zn, 15–150.^[23] There are numerous evidences of an increase in heavy metal phytoavailability and crop uptake related to the land application of sewage sludge, fly ash, poultry litter, pig slurry, and other biosolids.^[25–27] Supplementing swine feed with Cu has been a common practice for many years, and just environmental concerns have been expressed over elevated Cu concentration in soils receiving the long-term application of swine waste. On such sites, Cu is preferentially adsorbed by OM associated with the coarse fraction in soil.^[28] Sludge-derived OM contributes significantly to the metal adsorption capacity, and the slow mineralization of this OM can release metals into more soluble forms.^[29] Because the decomposition of sludge OM is often associated with the acidification of soil, further increased bioavailability of the sludge-born heavy metals is to be considered^[30] resulting in enhanced availability of heavy metals, decades after sludge application.^[31] Wastes with unsuitable agronomic features can be processed to suitable fertilizers by a variety of methods such as drying, composting, alkaline stabilizing, or incineration to reduce sludge mass, volume, concentration, and mobility of metals.^[4]

Any use of organic amendments must consider possible effects on heavy metal availability, especially when the initial contamination of a soil is already high. Under these circumstances, even the use of composts with low metal concentrations can be questionable because of the mobilization action of soluble organic compounds.^[32]

CONCLUSION

Organic fertilizers affect the heavy metal balance of agroecosystems by supply, mobilization, and immobilization. Supply and mobilization are useful options in terms of

feeding plants with essential heavy metals (e.g., Zn and Cu) but are a risk for loading sensitive environments. Immobilization in and removal from soil of heavy metals by phytoremediation [increased uptake by accumulating plant species (e.g., *Polygonum spec.*)] are deceptive options: The former leaves the risk in place, while the latter is too inefficient in terms of masses removed and just shifts the problem of contamination to other ecological compartments. The only feasible option for protecting soils, food, and environment is applying the precautionary principle by balancing all sources of input and output of heavy metals to an ecosystem.

REFERENCES

- Kabata-Pendias, A. *Trace Elements in the Biological Environment*; Geological Edition: Warsaw, 1979.
- Tiller, K.G. Heavy metals in soils and their environmental significance. *Adv. Soil Sci.* **1989**, *9*, 113–142.
- Iskandar, I.K.; Kirkham, M.B. *Trace Elements in Soil: Bioavailability, Flux and Transfer*; Lewis Publishers: Boca Raton, 2001.
- Richards, B.K.; Steenhus, T.S.; Peverly, J.H.; McBride, M.B. Effect of sludge-processing mode, soil texture and soil pH on metal mobility in undisturbed soil columns under accelerated loading. *Environ. Pollut.* **1999**, *109*, 327–346.
- Bolan, N.S.; Duraisamy, V.P. Role of inorganic and organic soil amendments on immobilisation and phytoavailability of heavy metals: A review involving specific case studies. *Aust. J. Soil Res.* **2003**, *41*, 533–555.
- Khoshgoftarmanesh, A.H.; Kalbasi, M. Effect of municipal waste leachate on soil properties and growth and yield of rice. *Commun. Soil Sci. Plant Anal.* **2002**, *33*, 2011–2020.
- Hansen, L.G.; Schaeffer, D.J. Livestock and domestic animals. Soil amendments. In *Impact on Biotic Systems*; Rechcigl, J.E., Ed.; Lewis Publishers: Boca Raton, 1995; 41–79.
- Narwal, R.P.; Singh, B.R. Effect of organic materials on partitioning, extractability and plant uptake of metals in alum shale soil. *Water Air Soil Pollut.* **1998**, *103*, 405–421.
- Kabata-Pendias, A.; Pendias, H. *Biogeochemistry of Trace Elements*; PWN: Warsaw, 1999.
- Nicholson, F.A.; Chambers, B.J.; Williams, J.R.; Unwin, R.J. Heavy metal contents of livestock feeds and animal manures in England and Wales. *Bioresour. Technol.* **1999**, *70*, 23–31.
- Marb, C.; Riedel, H. Changes in hazardous substances in sewage sludges during cold drying. *Korrespondenz Abwasser.* **1997**, *44* (8), 1386–1393.
- Kühnen, V.; Bien, B.; Goldbach, H.E. Heavy metal balances (Cd, Cr, Cu, Ni, Pb, Zn) of pig farming. In *VDLUFA-Kongress vom 2001 in Berlin*, September 17–21, 2001; VDLUFA-Verlag, Kongressband 57/II, VDLUFA-Schriftenreihe, Darmstadt.
- Menzi, H.; Kessler, J. Heavy metal contents of manures in Switzerland. In *Proceedings of the Eighth International Conference on the FAO Escorena Network on Recycling of Agricultural, Municipal and Industrial Residues in Agriculture*, Rennes, France, May 26–29, 1998; Martinez, J., Maudet, M.-N., Eds.; 1998; 26–5.
- Schmoll, H.; Held, T. Use of green compost in forestry. *Forst. Holz.* **1997**, *52* (9), 245–249.
- Schaaf, H.; Janßen, E. *Heavy Metal Concentrations in Manures and Secondary Raw Material Fertilizers and Heavy Metal Loads in Application of Codes of Good Agricultural Practice*; Kongressband 2000, VDLUFA-Verlag, VDLUFA-Schriftenreihe: Darmstadt, 144–150.
- Oliver, D.P.; Schultz, J.E.; Tiller, K.G.; Merry, R.H. The effect of crop rotations and tillage practices on cadmium concentration in wheat grain. *Aust. J. Agr. Res.* **1993**, *44*, 1221–1234.
- McBride, M.B. *Environmental Chemistry of Soils*; Oxford University Press: New York, 1994.
- Naidu, R.; Harter, N.D. Effect of different organic ligands on cadmium sorption and extractability from soils. *Soil Sci. Soc. Am. J.* **1998**, *62*, 644–650.
- Klitzke, S.; Lang, F.; Kaupenjohann, M. Influence of pH and DOM on the mobilization of colloid-bound lead. *DIAS Report. Plant Prod.* **2002**, *80*, 261–265.
- Senesi, N. Metal-humic substance complexes in the environment. Molecular and mechanistic aspects by multiple spectroscopic approach. In *Biogeochemistry of Trace Metals*; Adriano, D.C., Ed.; Lewis Publishers: Chelsea, 1992; 429–496.
- Gupta, S.K.; Herren, T.; Wenger, K.; Krebs, R.; Hari, T. *In situ* gentle remediation measures for heavy metal-polluted soils. In *Phytoremediation of Contaminated Soil and Water*; Norman, T., Gary, B., Eds.; Lewis Publishers: Chelsea, 2000; 303–322.
- Schnug, E. Differentiation between primary and secondary effects of N-fertilization on the micronutrient supply of cereals. *Landwirtsch. Forsch.* **1984**, *40*, 231–240.
- McGrath, S.; Chaundri, A.M.; Giller, K.E. Long-term effects of metal in sewage sludge on soils microorganisms and plants. *J. Ind. Microbiol.* **1995**, *14*, 94–104.
- Joergensen, R.G.; Meyer, B.; Roden, A.; Wittke, B. Microbial activity and biomass in mixture treatments of soil and biogenic municipal refuse compost. *Biol. Fert. Soils* **1996**, *2*, 43–49.
- Shuman, M.L. Organic waste amendments effect on Zn fractions of 2 soils. *J. Environ. Qual.* **1999**, *28*, 1442–1447.
- Jackson, B.P.; Miller, W.P.; Shuman, A.W.; Sumner, M.E. Trace elements solubility from land application of fly ash organic waste mixtures. *J. Environ. Qual.* **1999**, *28*, 639–647.
- Basta, N.T.; Sloan, J.J. Bioavailability of HM in strongly acidic soils treated with exceptional quality biosolids. *J. Environ. Qual.* **1999**, *28*, 633–638.
- Wu, J.; Laird, D.A.; Thompson, M.L. Sorption and desorption of Cu on soil clay component. *J. Environ. Qual.* **1999**, *28*, 334–338.
- McBride, M.B. Toxic metal accumulation from agricultural use of sludge: Are USEPA regulations protective? *J. Environ. Qual.* **1995**, *2*, 5–18.
- McGrath, S.P.; Zhao, F.J.; Dunham, S.J.; Grosland, A.R.; Coleman, K. Long-term changes in the extractability and bioavailability of Zn and Cd after sludge application. *J. Environ. Qual.* **2000**, *29*, 875–883.
- McBride, M.B.; Richards, B.K.; Steenhuis, T.; Russo, J.; Sauve, S. Mobility and solubility of toxic metals and nutrients in soil fifteen years after sludge application. *Soil Sci.* **1997**, *162*, 487–500.
- Madrid, L.; Diaz-Barrientos, E.; Cardo, I. Sequential extraction of metals from artificially contaminated soils in the presence of various composts. In *Trace Elements in Soil: Bioavailability, Flux and Transfer*; Iskandar, I.K., Kirkham, M.B., Eds.; Lewis Publishers: Boca Raton, 2001; 43–62.

Organisms and Soil Food Webs

David C. Coleman

Institute of Ecology, University of Georgia, Athens, Georgia, U.S.A.

Abstract

Soils may be viewed as the organizing centers for terrestrial ecosystems. This is largely the result of organismal activities in the soil. Major functions such as ecosystem production, respiration, and nutrient recycling are controlled by the rates at which nutrients are released by decomposition in the soil and litter horizons. The array of biota, including microbes, microbe-feeding fauna, vegetation, and consumers are all influenced by soil processes, and the organisms in turn have an impact on the soil system.

INTRODUCTION

Soils provide a wide range and variety of microhabitats, thus accommodating a very diverse biota. The enormous surface area (hundreds of m²/g of soil) of soil particles ranges in size classes from clays (0.1–2 µm in diameter) to silts (2–50 µm in diameter) and sands (0.05–2 mm in diameter). Numerous microbes and micro- and mesofauna (protozoa and nematodes) exist in water films on these particles and in or on the surfaces of microaggregates formed from the primary particles.^[1] In turn, the more mobile fauna, from collembola and mites (larger mesofauna) to the macrofauna (earthworms, millipedes, ants, termites, and fossorial or earth-dwelling vertebrates) move through macro- and micropores in the soil. The macrofauna play a role in moving parts of the soil profile around and form many sorts of burrows and pores, and they are often termed “ecological engineers.”

SOIL FOOD WEBS

The initial breaking up or “comminution” of plant litter (above- and below-ground) results from the chewing and macerating action of both large and small animals. This comminution benefits the fauna, which derive nutritional benefit from the litter and/or microbes initially colonizing the plant material. The increased surface area and further inoculation of the smaller pieces enhance the microbial access to, and breakdown of, these tissues. The decomposition process drives complex food webs^[2,3] in the soil, with numerous interactions between the initial agents of decomposition, the bacteria and fungi, and the fauna that in turn feed upon them, which facilitates nutrient return in the soil matrix (Fig. 1). These feeding relationships have several trophic levels, with bacteria and fungi being fed upon by microbe feeders, such as protozoa, nematodes, and microarthropods, which are in turn preyed upon by predatory nematodes or mites, and these in turn fed upon

by higher predators (Fig. 1). In forests or no tillage agroecosystems, where fungi dominate in the surface litter, the dominant flows of energy and nutrients will go via fungal pathways. In contrast, in conventional tilled fields where the organic matter is incorporated in the plow layer (usually 6–8 in. = 15–20 cm), the dominant flows of energy and nutrients may be more bacterially dominated, usually decomposing faster than in the no tillage system.^[1]

ZONES OF INFLUENCE

The heterogeneous distribution of food resources in the soil matrix makes it difficult to sample adequately for abundances and activities of the biota in a repeatable fashion. A useful approach is to consider soils as being comprised of zones of influence (ZOI) that can be targeted for further study. These ZOI, also termed “hot spots,” are located in the root rhizosphere, in regions of organic detritus accumulation, or detritusphere, and also in earthworm-influenced regions, such as burrows, which are termed a drilosphere^[4] (Fig. 2). These ZOI may represent less than 10% of the volume of the surface A horizon but account for up to 90% of the total biological activity in soils worldwide.

ROLES OF BIOTA IN SOIL FOOD WEBS

The functional roles of soil organisms can be compared most usefully in terms of body width. The microbes and microfauna inhabit soil water films and are restricted to this aquatic milieu. In contrast, the meso- and macrofauna, from acari (mites) to earthworms, inhabit gas-filled pores and move around in the soil matrix for considerable distances (Fig. 3).^[5]

Bacteria

Bacteria are unicellular prokaryotes (organisms lacking a unit membrane-bounded nucleus and other organelles),

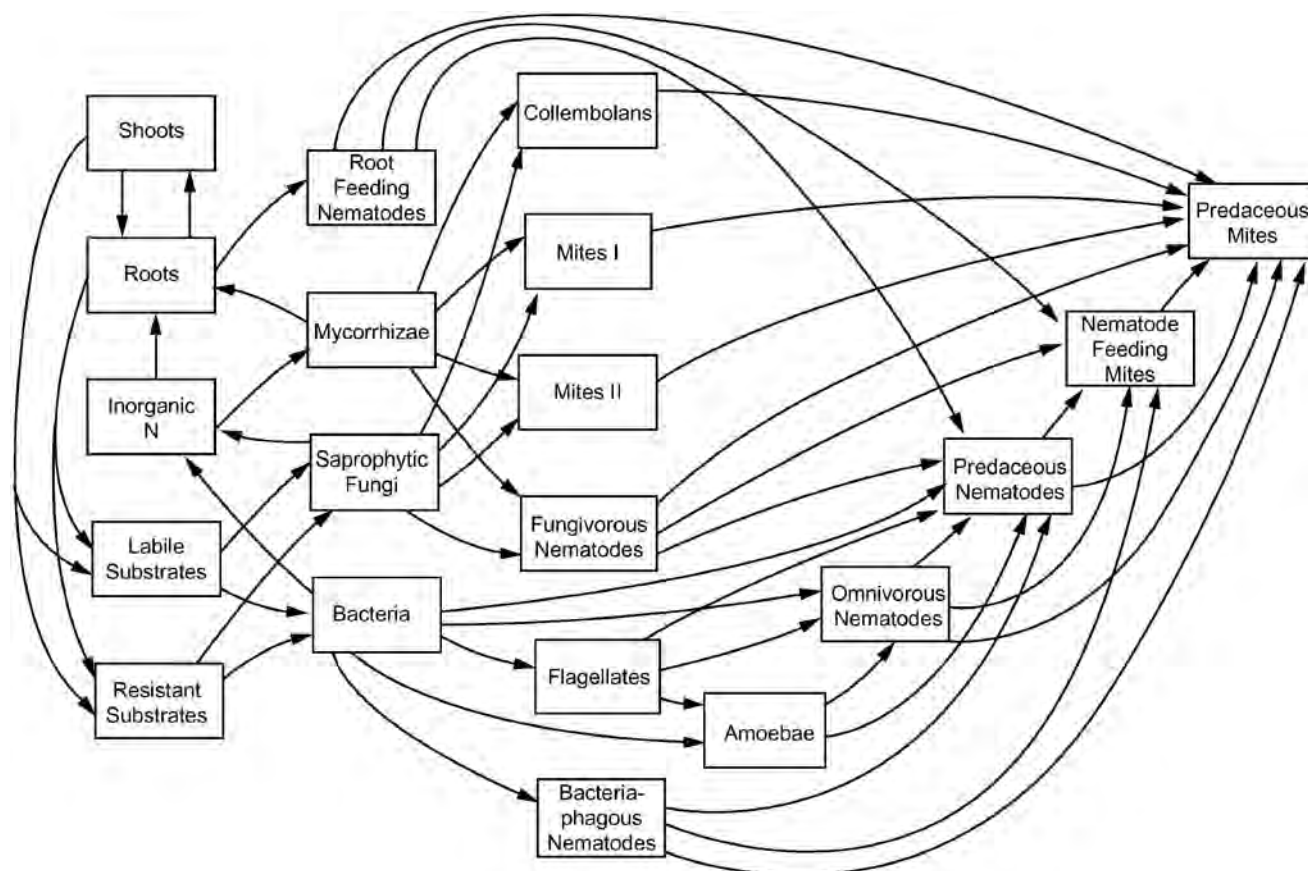


Fig. 1 Representation of detrital food web in shortgrass prairie. Fungal-feeding mites are separated into two groups (I and II) to distinguish the slow-growing oribatids from faster-growing taxa. Flows omitted from the figure for the sake of clarity include transfers from every organism to the substrate pools (death) and transfers from every animal to the substrate pools (defecation) and to inorganic N (ammonification).

Source: From Hunt, Coleman, et al.^[2]

which are found in all habitats on earth. They are exceedingly numerous (more than 10^{30} or one million trillion trillion^[6]) and diverse, comprising over 35 kingdoms in two domains, the Archaea and Eubacteria. They are active in all aspects of elemental cycling, and needed for nitrogen (N) cycling, both in N fixation (splitting N_2 and incorporating N into organic compounds) and subsequent transformational pathways as well. They are also primary agents of decomposition in many habitats and are particularly active in rhizospheres.^[4]

Fungi

Fungi are multicellular eukaryotes that are found in many habitats worldwide. They have long, ramifying strands (hyphae) which can grow into and explore many microhabitats and are used for obtaining water and nutrients. The hyphae secrete a considerable array of enzymes, such as cellulases, and even lignases in some specialized forms (useful in breaking down wood), decomposing substrates *in situ*, taking up the decomposed subunits, and

translocating them back through the hyphal network. Fungi are very abundant, particularly in undisturbed forest floors, in which literally thousands of kilometers of hyphal filaments will occur per gram of leaf litter.

The roles of mycorrhizas (literally “fungus–root,” or symbiotic fungi associated with many plants) in soil systems are being increasingly viewed as central to much of terrestrial ecosystem function. Mycorrhizas are essential to the growth and reproduction of numerous families of plants.^[1]

Microfauna

The unicellular eukaryotes, or Protoctista, are more often called protozoans. They include the flagellates, naked amoebae, testacea, and ciliates. These organisms range in size from a few cubic micrometers in volume (micro flagellates) to larger ciliates, which may be up to 500 μm in length and 20–30 μm in width. Protozoa are abundant, reaching densities of from 100 to 200 thousand per gram of soil. Bacteria, their principal prey, often exist in numbers up to one billion per gram of soil. All of these organisms are

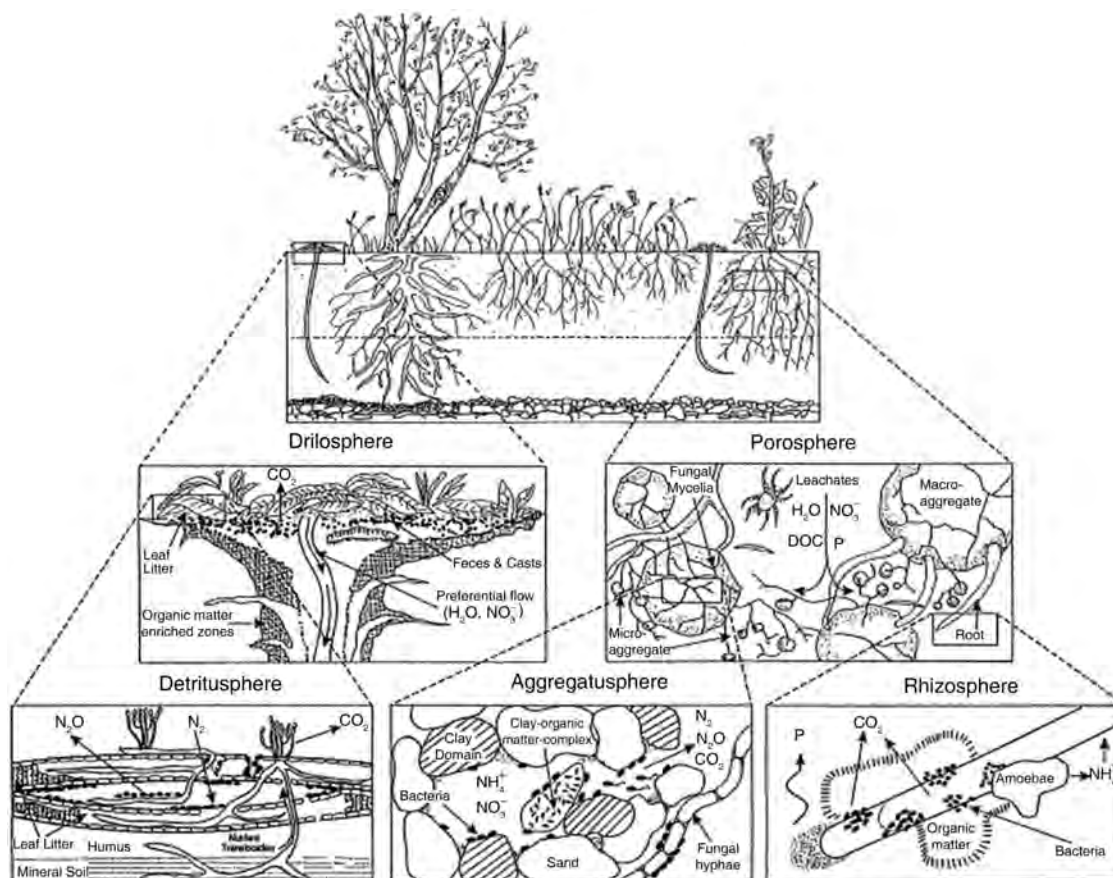


Fig. 2 Areas of activity in soil systems. These “ZOI” may be <10% of the total soil volume but represent >90% of the total biological activity in most soils worldwide.

Source: From Beare, Coleman, et al.^[4]

true water-film dwellers and become dormant or inactive during episodes of drying in the soil. They can exist in inactive or resting stages literally for decades at a time in very xeric environments.

Mesofauna

Nematodes

Nematodes have a wide range of feeding preferences. A general trophic grouping is bacterial feeders, fungal feeders, plant feeders, and predators and omnivores. Anterior (stomal or mouth) structures can be used to differentiate general feeding or trophic groups. Because nematodes reflect the developmental stages of the systems in which they occur (e.g., annual vs. perennial crops, or old fields and pastures and more mature forests), they have been used as indicators of overall ecosystem condition.^[7,8]

Collembola

Collembolans, or “springtails,” are primitive apterygote (wingless) insects. They are called “springtails” because

many of them have a spring-like lever, or furcula, which enables them to move many body lengths away from predators in a springing fashion. Collembolans are ubiquitous members of the soil fauna, often reaching abundances on 100,000 or more per m². They occur throughout the soil profile, where their major diet is decaying vegetation and associated microbes (usually fungi). However, like many members of the soil fauna, collembolans defy placement in exact trophic groups. Many collembolan species will eat nematodes when those are abundant. Some feed on live plants or their roots. One family (Onychiuridae) may feed in the rhizosphere and consume mycorrhizas or even plant pathogenic fungi.^[9]

Mites (Acari)

The soil mites, Acari, are chelicerate arthropods related to the spiders. They are often the most abundant microarthropods in soils. A 100 g sample may contain as many as 500 mites representing nearly 100 genera. This diverse array includes participants in three or more trophic levels, with varied strategies for feeding, reproduction, and dispersal.

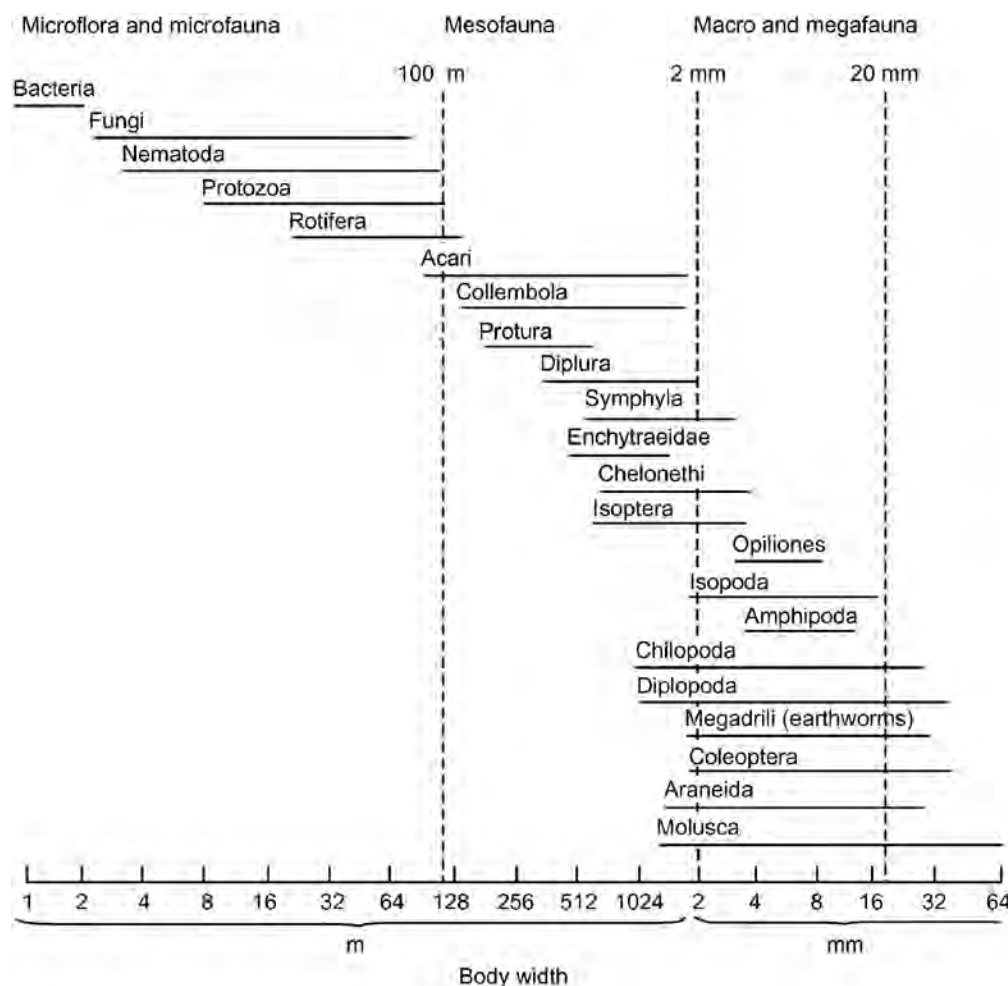


Fig. 3 Size classification of organisms in decomposer food webs by body width.

Source: From Swift, Heal, et al.^[5]

The oribatid mites (Oribatei) are the characteristic mites of the soil and are usually fungivorous or detritivorous. Mesostigmatid mites are nearly all predators on other small fauna, although a few species are fungivores and may become numerous at times. Astigmatid mites are associated with rich, decomposing N sources and are rare except in agricultural soils. The Prostigmata contains a broad diversity of mites with several feeding habits.^[1]

Macrofauna

Termites (Isoptera) are one of the major ecosystem “engineers,” particularly in tropical regions. Termites are social insects with a well-developed caste system. By their ability to digest wood, they have become economic pests of major importance in some regions of the world. The termites in a primitive family, the Kalotermitidae, possess a gut flora of protozoans, which enables them to digest cellulose. Their normal food is wood that has come into contact with soil. Many species of termites construct runways of soil, or along root channels, and some are builders of large, spectacular mounds. Members of the phylogenetically advanced family Termitidae possess a formidable

array of microbial symbionts (bacteria and fungi but not protozoa), and elevated pH in their hindguts,^[10] which enable them to process and digest the humified organic matter in tropical soils and to thrive on it.

Termites parallel earthworms in ingestive and soil turnover functions. The principal difference is that earthworms egest much of what they ingest in altered form (that enriches microbial action), whereas termites transfer large amounts of soil (organic material into building nests and mounds (carbon sinks).

Earthworms

Much of the evidence for earthworm effects on soil processes comes from agroecosystems and involves a small group of European lumbricids (Lumbricidae family in the Oligochaeta order). Impacts of exotic earthworms on native species are not well understood, although there is evidence that when native habitat is destroyed and native earthworm species extirpated, exotic earthworms colonize the newly empty habitat. As more extensive studies are carried out, it is becoming clear that earthworms are present in a wide variety of tropical as well as temperate ecosystems.

Earthworms have important roles in the fragmentation, breakdown, and incorporation of soil organic matter (SOM). This affects the distribution of SOM and also its chemical and physical characteristics. Changes in any of these soil parameters may have significant effects on other soil biota, by changing their resource base (e.g., distribution and quality of SOM, microbes, or microarthropods) or by changing the physical structure of the soil.^[11] Earthworm activities impact the communities of other soil biota through their effects on the chemical and physical characteristics of SOM, causing changes in the microbial and microarthropod communities, and also having impacts elsewhere in the soil food web.^[12]

CONCLUSION

Soil biota are very interconnected with each other by a variety of trophic and nutrient-flow pathways. The possibility of enhancing biotic activity in agricultural systems via conservation or no tillage regimes is very great, and much research is focusing on this area of interest. This approach permits a melding of interests between those who want to reduce fossil fuel inputs to agroecosystems and those who are concerned with enhancing carbon sequestration in soils as well.

REFERENCES

1. Coleman, D.C.; Crossley, D.A., Jr. *Fundamentals of Soil Ecology*; Academic Press: San Diego, 1996; 205 pp.
2. Hunt, H.W.; Coleman, D.C.; Ingham, E.R.; Ingham, R.E.; Elliott, E.T.; Moore, J.C.; Rose, S.L.; Reid, C.P.P.; Morley, C.R. The detrital food web in a shortgrass prairie. *Biol. Fert. Soils* **1987**, *3*, 57–68.
3. Moore, J.C.; de Ruiter, P.C. Invertebrates in detrital food webs along gradients of productivity. In *Invertebrates as Webmasters in Ecosystems*; Coleman, D.C., Hendrix, P.F., Eds. CABI: Wallingford, 2000; 161–184.
4. Beare, M.H.; Coleman, D.C.; Crossley, D.A., Jr. Hendrix, P.F.; Odum, E.P. A hierarchical approach to evaluating the significance of soil biodiversity to biogeochemical cycling. *Plant Soil* **1995**, *170*, 5–22.
5. Swift, M.J.; Heal, O.W.; Anderson, J.M. *Decomposition in Terrestrial Ecosystems*; University of California Press: Berkeley, 1979; 379 pp.
6. Whitman, W.B.; Coleman, D.C.; Wiebe, W.J. Prokaryotes: The unseen majority. *Proc. Natl. Acad. Sci.* **1998**, *95*, 6578–6583.
7. Bongers, T. The maturity index: An ecological measure of environmental disturbance based on nematode species composition. *Oecologia* **1990**, *83*, 14–19.
8. Yeates, G.W.; Bongers, T.; de Goede, R.G.M.; Freckman, D.W.; Georgieva, S.S. Feeding habits in soil nematode families and genera—an outline for soil ecologists. *J. Nematol.* **1993**, *25*, 315–331.
9. Lartey, R.T.; Curl, E.A.; Peterson, C.M. Interactions of mycophagous collembola and biological control fungi in the suppression of *Rhizoctonia Solani*. *Soil Biol. Biochem.* **1994**, *26*, 81–88.
10. Bignell, D.E.; Eggleton, P. On the elevated intestinal pH of higher termites (isoptera termitidae). *Insect. Soc.* **1995**, *42*, 57–69.
11. Hendrix, P.F., Ed. *Earthworm Ecology and Biogeography in North America*; Lewis Publishers: Boca Raton, 1995; 335 pp.
12. Fu, S.; Cabrera, M.L.; Coleman, D.C.; Kisselle, K.W.; Garrett, C.J.; Hendrix, P.F.; Crossley, D.A., Jr. Soil carbon dynamics of conventional tillage and no-till agroecosystems at Georgia Piedmont—HSB-C models. *Ecol. Model.* **2000**, *131*, 229–248.

Organo-Mineral Relationships

Claire Chenu

National Institute of Agronomic Research (INRA), Versailles, France

Alain F. Plante

Science of the Sun Unit, National Institute of Agronomic Research (INRA), Versailles, France

Pascale Puget

School of Engineering and Agriculture (ESITPA), Rouen, France

Abstract

Organo-mineral relationships are a fundamental feature of soils and are often used to define and differentiate soils from geological parent materials. Soil is primarily a mineral matrix (except in organic soils such as bog peat) but receives inputs of organic materials from various natural sources such as litterfall, root exudates, or from various anthropogenic sources such as manure additions. Organo-mineral relationships range in degree of association from the spatial distribution of particulate organic matter and mineral particles with minimal interaction to the inseparable organic matter intercalated between clay layers. Here, the term association is used to describe an arrangement of organic matter and minerals that has an undefined degree of cohesion. The term complex is restricted to cases where adsorption is the dominant mechanism.

INTRODUCTION

Organo-mineral associations cover a wide range of spatial scales, from nanometric (e.g., the complex of an organic acid with a clay sheet) to decimetric (e.g., a soil ped); however, they all originate at the molecular scale where physico-chemical interactions lead to the formation of bonds. The formation of organo-mineral associations may require only minutes, but their lifetime is related to that of the organic component, which can be short or extend to thousands of years. The impact of organo-mineral complexes in soils is significant; 40–80% of soil carbon is present in clay-sized separates and cannot be separated easily from clay minerals.^[1] Their importance is also qualitative and functional because the formation of organo-mineral complexes in soils strongly impacts the properties of the mineral phase and influences the biodegradation of organic matter.

CHARACTERIZATION OF ORGANO-MINERAL RELATIONSHIPS

Historically, organic matter has been studied by methods that involved their solubilization and separation from soil minerals. On the other hand, mineralogists analyzed soil minerals after comprehensive destruction of organic matter with hydrogen peroxide. Therefore, most of the knowledge concerning organo-mineral associations was gained from

studies performed on synthetic organo-mineral complexes, prepared from well-defined constituents such as clay minerals from geological deposits and pure organic compounds (e.g., polysaccharides and proteins).^[2] Physical methods for soil fractionation such as size or density separation enable the separation of intact organo-mineral associations from soils and their direct study with a wide range of non-destructive methods such as nuclear magnetic resonance spectroscopy.^[3]

Primary organo-mineral associations are associations of organic matter with individual soil mineral particles.^[4]

Their separation requires complete dispersion of soil using mechanical means and size or density separation techniques. Secondary organo-mineral associations, namely aggregates, are made by grouping together of several primary organo-mineral associations. Secondary organo-mineral associations are normally isolated from soil by sieving techniques after mild soil dispersion (e.g., a simple immersion of soil sample in water and agitation).^[5]

In primary organo-mineral associations, more organic matter is more closely associated with minerals as particle size decreases: sand-sized organic matter (i.e., decomposing plant debris) is separated easily from sand-sized minerals by densimetric techniques,^[6,7] whereas little free mineral or organic matter seems to occur in the clay-sized fraction.^[8] A variety of compounds are involved in primary organo-mineral associations. The clay-sized associations consist of highly processed and humified compounds, as

well as microorganisms and their metabolites.^[1,4,9] Clay-organic matter associations also show diverse microstructures, from organic interlayer complexes^[10,11] and organic coatings on clay mineral surfaces to more complex structures such as microaggregates of bacteria or plant debris with clay particles^[12,13] (Fig. 1).

In various soil types and management contexts, secondary organo-mineral complexes are hierarchical: particles associate into microaggregates, which in turn associate into larger aggregates.^[15–18] The smaller the aggregate hierarchical level, the higher is its physical stability, because the binding agents involved change with the spatial scale. Clay-sized associations are bound by sesquioxides, humic materials, and polysaccharides. These associations are bound into <250 μm microaggregates by bacterial and fungal debris, which may further be bound into macroaggregates >250 μm by transient agents (e.g., polysaccharides) and temporary ones (e.g., fine roots and fungal hyphae; Fig. 2).^[19,20] Aggregates appear

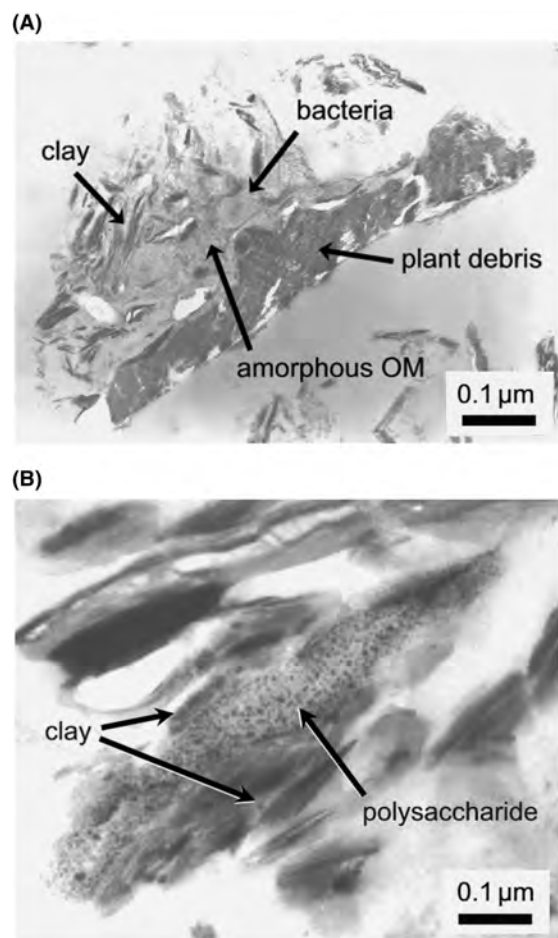


Fig. 1 Microstructure of primary organo-mineral associations observed with transmission electron microscopy in clay-sized fractions from soils: (A) complex microaggregate, (B) microaggregate in which OM occurs as layers between stacks of clay.

Source: From Chenu & Stotzky.^[14]

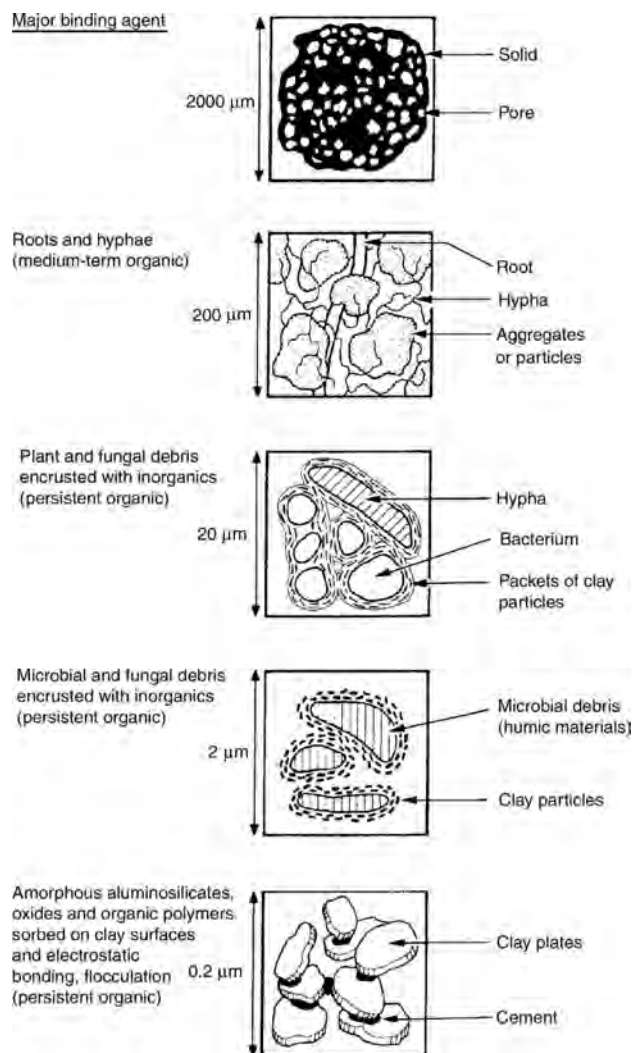


Fig. 2 Nature and scale of various organo-mineral associations. **Source:** From Tisdall & Oades.^[19]

to be formed and stabilized around decomposing plant debris that act as hot spots for microbial activity^[17,21–24] (Fig. 3).

Binding Forces within Organo-Mineral Associations

A variety of bonds may be formed when organic molecules in solution come into contact with the surface of soil minerals (see Fig. 4).^[26] For small organic molecules, these interactions depend on their ability to establish electrostatic interactions or dipole interactions. Polarity and polarizability are thus important adsorption-related properties of such organic molecules. For macromolecules, weak interactions such as van der Waals interactions and hydrogen bonding become very important. The conformation of macromolecules largely influences their adsorption, which increases with molecular weight.^[2] High-molecular-weight polymers are generally adsorbed in an irreversible fashion and

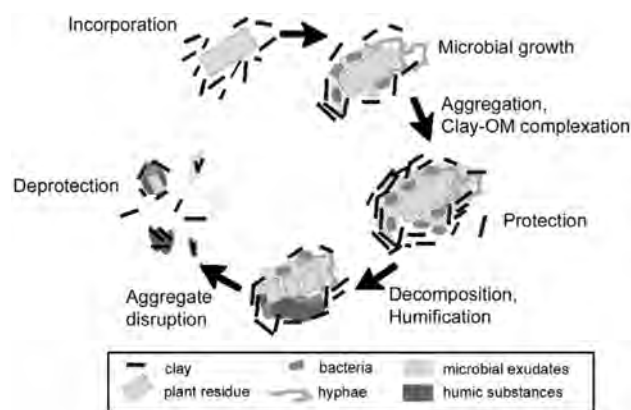


Fig. 3 Schematic presentation of the interaction between soil organic matter decomposition and aggregate formation and destruction.

Source: Adapted from Golchin, Oades, et al.,^[22] Chenu, Puget, et al.,^[23] Balesdent, Chenu, et al.^[25]

therefore the adsorption of macromolecules into soil minerals often leads to the formation of stable organo-mineral complexes. Organic polymers such as polysaccharides, proteins, fulvic, and humic acids are active in the formation of organo-mineral complexes because of their high molecular weight and charge. Similarly, clay minerals are the most active mineral constituents in the formation of organo-mineral complexes because of their charge and their high specific surface area.^[2] Solid particles such as bacteria, fungi, or plant debris also establish physico-chemical interactions with mineral surfaces through the process of adhesion.

Physico-chemical interactions at the molecular scale (i.e., adsorption and adhesion) provide tensile forces that increase both the tensile strength and the compressive strength of primary as well as secondary organo-mineral complexes and associations.^[27–29] At a larger scale, the binding of aggregates also involves physical mechanisms.

Fungal hyphae and fine roots stabilize aggregates by entangling the soil particles^[30–32] and thereby increasing aggregate strength through compressive forces. Physical processes such as those related to shrink–swell or freeze–thaw also alter organo-mineral associations by 1) creating failure zones that separate aggregates^[15,33] or 2) increasing the contacts between the organic and mineral surfaces and thus aiding the creation of new bonds.^[34,35]

ORGANO-MINERAL RELATIONSHIPS ALTER THE PROPERTIES OF SOIL MINERALS

Soil properties observed at a macroscopic scale are largely due to changes in the properties and associations of soil primary particles at a microscopic scale, in particular that of clay minerals. For example, the basic physical and physico-chemical mechanisms by which organic matter stabilizes soil aggregates against the disruptive action of water are an increased cohesion of the aggregate or a decreased wettability. Increased cohesion helps the aggregate withstand mechanical disruption by raindrops and resist forces exerted by compressed air upon wetting. Adsorption of large and flexible organic polymers increases the tensile strength of minerals (see above). Decreased wettability of aggregate surfaces slows the rate of wetting of the aggregates and thus the extent of slaking. Synthetic complexes of clays and humic substances, as well as natural complexes, are less wettable than pure clays.^[36–38] Associated organic matter also changes the swelling of clays and organo-mineral associations.^[39–41]

Organo-mineral associations alter the reactivity of soil minerals, particularly of soil clays, thereby altering the retention of cations, trace metals, or organic pollutants in soils. Associated organic matter increases the cation-exchange capacity of soils and soil clays,^[42–44] which contributes to the preferential retention of heavy metals by clay-sized fractions of soils. Increased hydrophobicity in

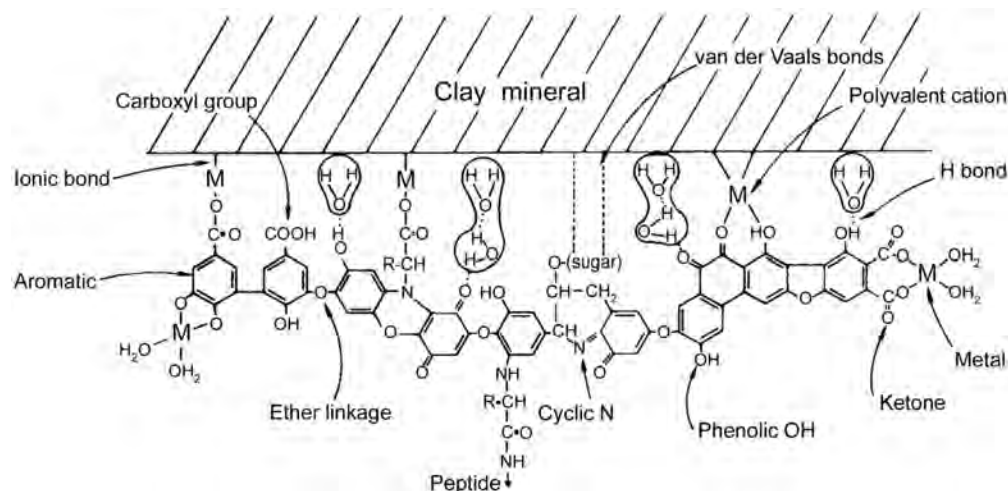


Fig. 4 Mechanisms of interaction between clay minerals and organic molecules.

Source: From Paul & Clark.^[45]

the clay fraction, caused by associated organic matter, facilitates the sorption of non-polar organic pollutants.^[46]

ORGANO-MINERAL RELATIONSHIPS REDUCE THE BIODEGRADATION RATE OF ORGANIC MATTER

Many easily decomposed compounds are retained in soil for longer periods than expected from their biochemistry. Association of organic matter with soil mineral particles and aggregates usually results in a decrease in the rate of biodegradation of the organic matter. Bartlett and Doner^[47] measured the decomposition of two radiolabeled-amino acids in four treatments: in a non-aggregated and non-adsorbed state; in a non-aggregated but adsorbed state; in an aggregated but non-adsorbed state; and in an aggregated and adsorbed state. The results showed that decomposition was more rapid in the non-aggregated vs. aggregated state, and that adsorption further decreased decomposition. The experiment clearly demonstrates two major mechanisms involved in organic-matter protection: “chemical protection” and “physical protection.”^[48]

Chemical protection refers to the reduced availability to microorganisms (and therefore decomposition) of organic matter in soil due to chemical interactions with soil constituents, such as complexation or adsorption on reactive mineral surfaces. Many natural and contaminant organic compounds are not available to microorganisms when adsorbed.^[49–52] Models that best describe the decomposition of compounds in soil couple biodegradation with sorption and assume that adsorbed compounds are not degraded.^[53] While some organisms can access some adsorbed compounds, the rate of decomposition is largely controlled by the desorption of the compound.^[53,54] The effect of minerals is also indirect because extracellular enzymes adsorb to clay minerals and become inactivated.^[55]

The physical protection of organic matter is the reduction in organic-matter decomposition rates caused by the architecture of the organo-mineral soil matrix. Evidence of physical protection comes primarily from experiments or agronomic situations in which soil structure is disrupted, leading to a flush of mineralization.^[25,56–58] A large proportion of labile soil organic matter appears to be physically protected in microaggregates.^[56] Some degree of contact between a microorganism or extracellular enzyme and the organic substrate is needed to allow biodegradation. Soil microorganisms generally occupy less than 2% of soil surface area and less than 1% of soil porosity.^[14] Therefore, large distances can exist between organism and substrate. Soil structure controls the continuity of pores filled with water and their accessibility to microorganisms. Habitable pore space consists of 15–25% of the total porosity, while the remainder is inaccessible to microorganisms because pore necks are smaller than 0.2 μm or pores are air-filled

rather than water-filled.^[59] Furthermore, soil structure may create sites in which the local physico-chemical conditions are unfavorable to mineralization due to oxygen depletion in anoxic center of aggregates.^[60]

While the chemical protection provided by adsorption is often permanent, physical protection is highly dynamic. Soil structure is continuously changing due to the action of soil fauna, root growth, wet–dry cycles, and tillage (see Fig. 3). Therefore, physical protection depends on the life expectancy of the arrangement of the protection sites and is therefore subject to changes due to soil aggregate turnover.^[61] While the relative contribution of physical protection to the stabilization of organic matter may be low, it often affords the time for more permanent chemical-protection mechanisms to occur.

CONCLUSION

The existence of “primary particles” and “clean” mineral surfaces in natural soils occurs only rarely (except in the case of sands). Instead, soil is an intimate association of organic matter and mineral surfaces that interact to various degrees. While the binding of organic matter with mineral surfaces occurs at the nano- to micrometer scale, its impact is felt upward to the centimeter and meter scales. Organo-mineral relationships are dynamic processes that alter the properties of both the organic matter and mineral particles involved.

REFERENCES

1. Christensen, B.T. Carbon in primary and secondary organo-mineral complexes. In *Structure and Organic Matter Storage in Agricultural Soils*; Carter, M.R., Stewart, B.A., Eds.; CRC Press Inc.: Boca Raton, 1995; 97–165.
2. Theng, B.K.W. *Formation and Properties of Clay-Polymer Complexes*; Elsevier Scientific Publishing Company: Amsterdam, 1979.
3. Kögel-Knabner, I. Analytical approaches for characterizing soil organic matter. *Org. Geochem.* **2000**, *31* (7/8), 609–625.
4. Christensen, B.T. Physical fractionation of soil and organic matter in primary particle size and density separates. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer: New York, 1992; Vol. 20, 1–90.
5. Elliott, E.T. Aggregate structure and carbon, nitrogen and phosphorus in native and cultivated soils. *Soil Sci. Soc. Am. J.* **1986**, *50* (3), 627–633.
6. Feller, C. Une méthode de fractionnement granulométrique de la matière organique des sols. Application aux sols tropicaux à textures grossières, très pauvres en humus. *Cah. ORSTOM, Sér. Pédologie* **1979**, *XVII*, 339–345.
7. Balesdent, J.; Pétraud, J.P.; Feller, C. Effet des ultrasons sur la distribution granulométrique des matières organiques des sols. *Sci. Sol.* **1991**, *29* (2), 95–106.
8. Turchenek, J.M.; Oades, J.M. Fractionation of organomineral complexes by sedimentation and density techniques. *Geoderma* **1979**, *21* (4), 311–343.

9. Oades, J.M. An introduction to organic matter in mineral soils. In *Minerals in Soil Environments*; Dixon, D.E., Ed.; Soil Science Society of America: Madison, 1989; 89–158.
10. Theng, B.K.G.; Chuchman, G.J.; Newman, R.H. The occurrence of interlayer clay–organic complexes in two New Zealand soils. *Soil Sci.* **1986**, *142* (5), 262–266.
11. Righi, D.; Dinel, H.; Schulten, H.R.; Schnitzer, M. Characterization of clay–organic matter complexes resistant to oxidation by hydrogen peroxide. *Eur. J. Soil Sci.* **1995**, *46* (3), 423–429.
12. Feller, C.; François, C.; Villemin, G.; Portal, J.M.; Toutain, F.; Morel, J.L. Nature des matières organiques associées aux fractions argileuses d'un sol ferrallitique. *C.R. Acad. Sci. Paris.* **1991**, *312* (12), 1491–1497.
13. Chenu, C.; Arias, M.; Besnard, E. The influence of cultivation on the composition and properties of clay–organic matter associations in soils. In *Sustainable Management of Organic Matter*; Rees, R.M., Ball, B.C., Campbell, C.D., Watson, C.A., Eds.; CABI International: Wallingford, 2001; 207–213.
14. Chenu, C.; Stotzky, G. Interactions between microorganisms and soil particles: An overview. In *Interactions between Microorganisms and Solid Particles in the Soil Environment*; Huang, P.M., Ed.; John Wiley & Sons, Ltd.: Manchester, 2002; 1–40.
15. Dexter, A.R. Advances in characterization of soil structure. *Soil Till. Res.* **1988**, *11* (3/4), 199–238.
16. Oades, J.M.; Waters, A.G. Aggregate hierarchy in soils. *Aust. J. Soil Res.* **1991**, *29* (6), 815–828.
17. Puget, P.; Chenu, C.; Balesdent, J. Dynamics of soil organic matter associated with primary particle size fractions of water-stable aggregates. *Eur. J. Soil Sci.* **2000**, *51* (4), 595–605.
18. Six, J.; Paustian, K.; Elliott, E.T.; Combrink, C. Soil structure and organic matter: I. Distribution of aggregate-size classes and aggregate-associated carbon. *Soil Sci. Soc. Am. J.* **2000**, *64* (2), 681–689.
19. Tisdall, J.M.; Oades, J.M. Organic matter and water-stable aggregates. *J. Soil Sci.* **1982**, *33* (2), 141–163.
20. Jastrow, J.D.; Miller, R.M.; Lussenhop, J. Contributions of interacting biological mechanisms to soil aggregate stabilization in restored prairie. *Soil Biol. Biochem.* **1998**, *30* (7), 905–916.
21. Beare, M.H.; Hendrix, P.F.; Coleman, D.C. Water-stable aggregates and organic matter fractions in conventional-tillage and no-tillage soils. *Soil Sci. Soc. Am. J.* **1994**, *58* (3), 777–786.
22. Golchin, A.; Oades, J.M.; Skjemstad, J.O.; Clarke, P. Soil structure and carbon cycling. *Aust. J. Soil Res.* **1994**, *32* (5), 1043–1068.
23. Chenu, C.; Puget, P.; Balesdent, J. Clay–organic matter associations in soils: Microstructure and contribution to soil physical stability. In 16th World Congress of Soil Science, Montpellier, France, August 20–26, 1998; CD-ROM; Cirad: Montpellier.
24. Golchin, A.; Baldock, J.A.; Oades, J.M. A model linking organic matter decomposition, chemistry, and aggregate dynamics. In *Soil Processes and the Carbon Cycle*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; CRC Press Inc.: Boca Raton, 1998; 245–266.
25. Balesdent, J.; Chenu, C.; Balabane, M. Relationship of soil organic matter dynamics to physical protection and tillage. *Soil Till. Res.* **2000**, *53* (3/4), 215–230.
26. Mortland, M.M. Clay–organic matter complexes and interactions. In *Advances in Agronomy*; Bray, N.C., Ed.; Academic Press: New York, 1970; Vol. 22, 75–117.
27. Dowdy, R.H. The effect of organic polymers and hydrous oxides on the tensile strength of clay. In *Soil Conditioners*; Stewart, B.A., Ed.; SSSA: Madison, 1975; 25–33.
28. Chenu, C.; Guérif, J. Mechanical strength of clay minerals as influenced by an adsorbed polysaccharide. *Soil Sci. Soc. Am. J.* **1991**, *55* (4), 1076–1080.
29. Czarnes, S.; Hallett, P.D.; Bengough, A.G.; Young, I.M. Root- and microbial-derived mucilages affect soil structure and water transport. *Eur. J. Soil Sci.* **2000**, *51* (3), 435–443.
30. Tisdall, J.M. Fungal hyphae and structural stability of soil. *Aust. J. Soil Res.* **1991**, *29* (6), 729–743.
31. Dorioz, J.M.; Robert, M.; Chenu, C. The role of roots, fungi and bacteria on clay particle organization: An experimental approach. *Geoderma* **1993**, *56* (1–4), 179–194.
32. Degens, B.P. Macro-aggregation of soils by biological bonding and binding mechanisms and the factors affecting these: A review. *Aust. J. Soil Res.* **1997**, *35* (3), 431–459.
33. Preston, S.; Griffiths, B.S.; Young, I.M. Links between substrate additions, native microbes, and the structural complexity and stability of soils. *Soil Biol. Biochem.* **1999**, *31* (11), 1541–1547.
34. Haynes, R.J.; Swift, R.S. Stability of soil aggregates in relation to organic constituents and soil water content. *J. Soil Sci.* **1990**, *41* (1), 73–83.
35. Caron, J.; Kay, B.D.; Stone, J.A. Improvement of aggregate stability of a clay loam with drying. *Soil Sci. Soc. Am. J.* **1992**, *56* (5), 1583–1590.
36. Jouany, C. Surface free energy components of clay–synthetic humic acid complexes from contact-angle measurements. *Clay Clay Miner.* **1991**, *39* (1), 43–49.
37. Jańczuk, B.; Hajnos, M.; Białopiotrowicz, T.; Kliszcz, A.; Biliński, B. Hydrophobization of the soils by dodecylammonium hydrochloride and changes of the components of its surface energy. *Soil Sci.* **1990**, *150* (5), 792–798.
38. Chenu, C.; Bissonnais, Y.L.; Arrouays, D. Organic matter influence on clay wettability and soil aggregate stability. *Soil Sci. Soc. Am. J.* **2000**, *64* (4), 1479–1486.
39. Chenu, C. Influence of a fungal polysaccharide, scleroglucan, on clay microstructures. *Soil Biol. Biochem.* **1989**, *21* (2), 299–305.
40. Chenu, C. Clay- or sand-polysaccharides associations as models for the interface between microorganisms and soil: Water-related properties and microstructure. *Geoderma* **1993**, *56* (1–4), 143–156.
41. Emerson, W.W. Water retention, organic C and soil texture. *Aust. J. Soil Res.* **1995**, *33* (2), 241–351.
42. Thompson, M.L.; Zhang, H.; Kazemi, M.; Sandor, J. Contribution of organic matter to cation exchange capacity and specific surface area of fractionated soil materials. *Soil Sci.* **1989**, *148* (4), 250–257.
43. Curtin, D.; Rostad, H.P.W. Cation exchange capacity and buffer potential of Saskatchewan soils estimated from texture, organic matter and pH. *Can. J. Soil Sci.* **1997**, *77* (4), 621–626.

44. Bigorre, F. Contribution des argiles et des matières organiques à la rétention d'eau dans les Sols. Signification et rôle fondamental de la capacité d'Échange en cations. C.R. Acad. Sci. Paris **1999**, 330 (3), 245–250.
45. Paul, E.A.; Clark, F.E. *Soil Microbiology and Biochemistry*; Academic Press: San Diego, CA, 1996.
46. Wershaw, R.L. A new model for humic materials and their interactions with hydrophobic organic chemicals in soil water or sediment water systems. J. Contam. Hydrol. **1986**, 1 (1/2), 29–45.
47. Bartlett, J.R.; Doner, H.E. Decomposition of lysine and leucine in soil aggregates: Adsorption and compartmentalization. Soil Biol. Biochem. **1988**, 20 (5), 755–759.
48. Baldock, J.A.; Skjemstad, J.O. Role of the soil matrix and minerals in protecting natural organic materials against biological attack. Org. Geochem. **2000**, 31 (7/8), 697–710.
49. Aardema, B.W.; Lorenz, M.G.; Krumbein, W.E. Protection of sediment adsorbed transforming DNA against enzymatic inactivation. Appl. Environ. Microbiol. **1983**, 46 (2), 417–420.
50. Ogram, A.; Jessup, R.E.; Ou, L.T.; Rao, P.S.C. Effects of sorption on biological degradation rates of (2,4-dichlorophenoxy) acetic acid in soils. Appl. Environ. Microbiol. **1985**, 49 (3), 582–587.
51. Dashman, T.; Stotzky, G. Microbial utilization of amino acids and a peptide bound on homoionic montmorillonite and kaolinite. Soil Biol. Biochem. **1986**, 18 (1), 5–14.
52. Jones, D.L.; Edwards, A.C. Influence of sorption on the biological utilization of two simple carbon substrates. Soil Biol. Biochem. **1998**, 30 (14), 1895–1902.
53. Scow, K.M.; Johnson, C.R. Effect of sorption on biodegradation of soil pollutants. Adv. Agron. **1997**, 58 (1), 1–56.
54. Rao, P.S.C.; Bellin, C.A.; Lee, L.S. Sorption and biodegradation of organic contaminants in soils: Conceptual representations of process coupling. In *Environmental Impact of Soil Component Interactions: Volume 1: Natural and Anthropogenic Organics*. Proceedings of a Workshop Entitled 'Impact of Interactions of Inorganic, Organic and Microbiological Soil Components on Environmental Quality,' Edmonton, AB, Aug 11–15, 1992; Lewis Publishers: Boca Raton, 1995; 263–274.
55. Quiquampoix, H. Mechanisms of protein adsorption on surfaces and consequences for soil extracellular enzyme activity. In *Soil Biochemistry*; Bollag, J.M., Stotzky, G., Eds.; Marcel Dekker: New York, 2000; Vol. 10, 171–206.
56. Gregorich, E.G.; Kachanovskii, R.G.; Voroney, R.P. Carbon mineralization in soil size fractions after various amounts of aggregate disruption. J. Soil Sci. **1989**, 40 (3), 649–659.
57. Balesdent, J.; Boissgonier, D.; Mariotti, A. Effect of tillage on soil organic carbon mineralization estimated from ¹³C abundance in maize fields. J. Soil Sci. **1990**, 41 (4), 587–596.
58. Beare, M.H.; Cabrera, M.L.; Hendrix, P.F.; Coleman, D.C. Aggregate-protected and unprotected organic matter pools in conventional-tillage and no-tillage soils. Soil Sci. Soc. Am. J. **1994**, 58 (3), 787–795.
59. Hassink, J.; Bouwman, J.; Brussaard, L.B. Relationships between habitable pore space, soil biota and mineralization rates in grassland soils. Soil Biol. Biochem. **1993**, 25 (1), 47–55.
60. Sextone, A.J.; Revsbech, N.P.; Parkin, T.B.; Tiedje, J.M. Direct measurement of oxygen profiles and denitrification rates in soil aggregates. Soil Sci. Soc. Am. J. **1985**, 49 (3), 645–651.
61. Plante, A.F. *Soil Aggregate Turnover and the Physical Protection of Soil Organic Matter as Measured using Dy-labelled Tracer Spheres*; University of Alberta: Edmonton, 2001.

Oxisols

Stanley W. Buol

*Department of Soil Science, North Carolina State University, Raleigh,
North Carolina, U.S.A.*

Abstract

Oxisols are physically deep but nutrient poor and sparsely inhabited soils in the world. Most Oxisols are located in warm tropical areas with ustic or udic soil moisture regimes that permit the growth of one or two grain crops each year, without irrigation. Without lime and fertilizer, even subsistence slash and burn agriculture is not possible on the poorest Oxisols. With stable economic infrastructure and markets, chemical fertilizer and lime needed to initially correct natural chemical infertility and replace nutrients harvested, intensive mechanized agriculture and grazing of improved pastures are realities.

INTRODUCTION

The name Oxisol was introduced in 1960 to avoid connotations associated with laterite and associated terms.^[1] Oxisols are sandy loam or finer textured soils. Their subsoils have an apparent cation exchange capacity at pH 7 of 16 cmol/kg clay or less, have an apparent effective cation exchange capacity of 12 cmol/kg clay or less, and contain less than 10% calcium (Ca), magnesium (Mg), or potassium (K) bearing weatherable minerals in the 50–200 μm sand fraction. If the surface 18 cm contains less than 40% clay, the amount of clay in the subsoil must be less than 20% greater than the clay content of the surface 18 cm. If the surface 18 cm contains 40% or more clay, increased clay content with depth (i.e., a kandic horizon) is permitted provided that it contains less than 10% weatherable minerals.^[2]

Reaction and base saturation are not criteria for defining Oxisols. Most Oxisols are acidic in reaction and have low base saturation percentages, but some have high base saturation and neutral pH values. Most Oxisols have low silt content; low available water-holding capacity; low bulk density; strong, fine, and very fine granular structures; and a more rapid hydraulic conductivity than would be predicted by particle size class.

Almost all Oxisols are located within tropical latitudes and have average annual soil temperatures of 15°C or more. The mean soil temperature of June, July, and August differs by less than 6°C from that of December, January, and February. Oxisols are divided into five suborders according to their soil moisture regime (SMR):

1. Aquox (3% of all Oxisols) is poorly drained with aquatic conditions and most often located in depressions and seepage areas in association with other Oxisols.

2. Torrox (0.3% of all Oxisols) has an aridic SMR with less than 90 consecutive days of plant-available moisture in normal years.
3. Ustox (53% of all Oxisols) has an ustic SMR with from 90 to 270 consecutive days of plant-available moisture in normal years. One or two crop-growing seasons are possible each year.
4. Udox (32% of all Oxisols) has less than 90 consecutive days that are too dry for reliable crop growth in normal years. Two and sometimes three consecutive crops can be grown each year.
5. Perox (12% of all Oxisols) has rainfall that exceeds potential evapotranspiration every month of normal years. With no dry season, it is difficult to harvest grain crops, and slash and burn management is difficult because of non-availability of reliable time to burn.

GEOLOGIC SETTINGS

The most extensive contiguous areas of Oxisols are related to mid- to late-tertiary geomorphic surfaces composed of material that has been weathered, eroded, transported, and redeposited many times.^[3,4] The materials within which Oxisols form on such surfaces are those that are an accumulation of resistant minerals that have the properties of Oxisol subsoil at the time of their deposition. Such deposits, known as oxic soil material sometimes have stone lines of quartz and/or oxide-cemented gravel. The inert nature of oxic soil material precludes many pedogenic processes, and Oxisols are present on alluvial deposits of such material.^[5]

Oxisols also form on stable surfaces where easily weatherable materials, such as basic and ultrabasic rocks and geologically old volcanic deposits, are exposed to warm humid conditions.^[6,7] These occurrences are generally of limited spatial extent.

PROCESSES OF FORMATION

Processes that concentrate aluminum (Al) and deplete silicon (Si) favor the formation of oxic soil material. Low Si/Al ratios may have their origin in the magma of the earth's crust and persist when exposed to the soil environment. Si loss is to be expected in the surface of almost all soils, as rainwater infiltrates and moves through the surface layer. The amount of Si removed depends on the type of silicate mineral and residence time of water around the silicate mineral. Net Si loss results in destruction of 2:1 lattice clays and favors kaolinite, halloysite, and gibbsite enrichment.

Iron released by the dissolution of iron bearing silicates is concentrated as iron oxides unless reducing conditions are present. Almost all of the iron in Oxisols are present as iron oxides. Iron oxides impart many red, red-yellow, and yellow hues to soil. Red hues are indicative of hematite, while the yellow colors indicate goethite. Mixtures of these minerals have intermediate colors.^[8] Hematite is more easily reduced, and thus more readily dissolved, than goethite.^[9,10] Even short periods of reduction in microsites around decaying roots will, over time, cause the removal of hematite leaving goethite as the dominate iron oxide in the more yellow-colored Oxisols. If soil material is subjected to reducing conditions, all iron oxides are chemically reduced to soluble ferrous ions and removed, thereby leaving only the gray color of the clays and sand.^[11]

Oxisols contain more organic carbon than most other mineral soils.^[12] The organic carbon in the subsoil is relatively unavailable for microbial oxidation and masked by red- and yellow-colored iron oxides.^[13]

LOCATIONS

It is estimated that about 981 million hectares of the earth's surface is dominated by Oxisols. Of this 76% is in South America, 22% in Africa, and 2% in Asia. In South America, the central plateau between the Amazon and Paraná rivers and lower Amazon basin is perhaps the most extensive area of Oxisols. Multiple erosion-deposition cycles have created deep sedimentary deposits of material rich in quartz sand, iron, and Al oxides and 1:1 layer clays. Quartzipsamments are present in deposits too sandy to classify as Oxisols. The eastern part of the Amazon basin is filled with oxic materials eroded from adjacent plateaus and Oxisols predominate. In the western part of the Amazon basin, materials eroded from the Andes enrich the parent materials with weatherable minerals, and almost no Oxisols are present.^[14] In Central Africa, most Oxisols are in the central Congo River basin but seldom present in areas where relatively volcanic materials contribute to the present soil.

The presence of Oxisols within all SMRs and on formed geomorphic surfaces argues against generalizations about climatic and time factors necessary for Oxisol formation. The close association of Oxisols with parent materials

almost devoid of weatherable silicate minerals and the near absence of Oxisols in adjacent geologic materials clearly identify parent material composition as the controlling factor in Oxisol formation.

HUMAN USE

Oxisols are chemically infertile when compared to most other soils. Except when formed in basic and ultrabasic material, Oxisols are extremely poor in basic cations, and exchangeable Al dominates the cation exchange sites. Size and density of trees in undisturbed "cerrado" vegetation growing on Oxisols in Central Brazil are closely related to chemical fertility.^[15,16] Even the more highly base saturated "Eutro" great groups of Oxisols contain small total quantities of exchangeable Ca, Mg, and K.^[11] Phosphorous contents are relatively low, and phosphorus reacts with Al and iron oxides to such a degree that it is very slowly available to plant growth. Fast-growing food-crop plants seldom sustain satisfactory growth on unfertilized Oxisols.

Many of the relatively base-rich Oxisols support natural forest growth. Forest stands slowly accumulate plant-essential nutrients in their biomass. When a sufficient quantity of forest vegetation is cut, allowed to dry, and then burned, enough nutrients are rapidly released from the ashes to allow subsistence farmers to harvest one or two food crops. The site is then abandoned and a natural succession of woody vegetation that is able to grow with a slower rate of nutrient uptake than crop plants reestablishes. After several years, the biomass of the natural vegetation again acquires enough essential nutrients that subsistence farmers can repeat the slash and burn process.

On the least fertile Oxisols, only poor quality grass and shrub-type trees grow. An extensive area of this condition is the "cerrado" savanna vegetation in Central Brazil. The "cerrado" vegetation is so nutrient poor that the bones of grazing cattle deteriorate from Ca and phosphorous deficiency, unless supplemental mineral concentrates are used. Farmers attempting slash and burn techniques are discouraged by negligible yields. Inability to sustain indigenous human populations is a characteristic of the least fertile Oxisols.

With the advent of a clear understanding of chemical limitations and in those locations where economic stability enabled infrastructure necessary for commercial agriculture, an entirely different relationship of human interaction with Oxisols is possible. In 1992, with only 10 million hectares of the estimated 204 million hectares of Oxisol dominate "cerrado" cultivated, 28% of the grain production in Brazil was from the "cerrado" area.^[17] Beef production has rapidly increased in fertilized pastures. With market access, it is economically feasible to make initial applications of lime and phosphate fertilizer to overcome the natural acidity and phosphate fixation capacity of the most infertile Oxisols. Small amounts of

zinc, boron, molybdenum, and copper are needed on some sites. The initial application of lime and phosphate is mixed as deeply as possible to maximize a favorable rooting depth. Facilitated by the low cation capacity of the Oxisol, subsoil Ca moves downward, replaces exchangeable Al, increases the exchangeable Ca/Al ratio, and facilitates deeper root growth.^[18] The deeper rooting zone enhances the available moisture supply during rainless periods in the growing season. Total amounts of exchangeable Al in the subsoil are low, and relatively small amounts of Ca are required. Gypsum is especially favorable for Ca movement and supplies sulfur that is often needed in Oxisols. The initial investment in lime and phosphate fertilizer is large but not prohibitive. In subsequent years, nitrogen, phosphorous, and K fertilizers and lime are needed to replace nutrients exported in harvested crops and maintain desirable pH values, but annual amounts are no greater than on other soils growing similar crops.

Many Oxisol-dominated landscapes are nearly level, and trafficability for large equipment and road construction is facilitated by physically stable low activity silicate clay and iron oxide. Low indigenous populations facilitate the acquisition of large management units for efficient mechanized agriculture. Oxisols with ustic and udic SMRs and isothermic and isohyperthermic soil temperature regimes have reliable rains during at least one growing season each year, followed by a period of dryness as grain crops mature. The reliable onset of a dry season decreases the risk of grain spoilage and the cost of drying harvested grain and permits maximum efficiency of harvest equipment and marketing infrastructure.

CONCLUSION

Oxisols are physically deep but nutrient poor and sparsely inhabited soils in the world. Most Oxisols are located in warm tropical areas with ustic or udic SMRs that permit the growth of one or two grain crops each year, without irrigation. Without lime and fertilizer, even subsistence slash and burn agriculture is not possible on the poorest Oxisols. With stable economic infrastructure and markets, chemical fertilizer and lime needed to initially correct natural chemical infertility and replace nutrients harvested, intensive mechanized agriculture and grazing of improved pastures are realities.

REFERENCES

1. Buol, S.W.; Eswaran, H. Oxisols. *Adv. Agron.* **2000**, *68*, 151–195.
2. Soil Survey Staff. *Soil Taxonomy*; 2nd Agricultural Handbook No. 436; US Department of Agriculture, Natural Resources Conservation Service: Washington, 1999; 869 pp.
3. Lepsch, I.F.; Buol, S.W.; Daniels, R.B. Soils-landscape relationships in the occidental plateau of Sao Paulo State, Brazil. I. Geomorphic surfaces and soil mapping units. *Soil Sci. Soc. Am. J.* **1977**, *41*, 104–109.
4. Lepsch, I.F.; Buol, S.W.; Daniels, R.B. Soils-landscape relationships in the occidental plateau of Sao Paulo State, Brazil. II. Soil morphology, genesis and classification. *Soil Sci. Soc. Am. J.* **1977**, *41*, 109–115.
5. Odell, R.T.; Dijkerman, J.C.; Van Vuure, W.; Melsted, S.W.; Beavers, A.H.; Sutton, P.M.; Kurtz, L.T. *Characteristics, Classification and Adoption of Soils in Selected Areas of Sierra Leone, West Africa*; Bull. 748; Agric. Exp. Stn., University of Illinois Urbana-Champaign: Urbana, 1974; 194 pp.
6. Buurman, P.; Soepraptohardjo, M. Oxisols and associated soils on ultramafic and felsic volcanic rocks in Indonesia. In *Red Soils in Indonesia*; Buurman, P., Ed.; Bull. No. 4; Soil Research Inst: Bogor, Indonesia and Centre for Agric. Pub. and Documentation: Wageningen, 1980; 71–87.
7. Beinroth, F.H. Some highly weathered soils of Puerto Rico. 1. Morphology, formation and classification. *Geoderma* **1982**, *27*, 1–73.
8. Bigham, J.M.; Golden, D.C.; Buol, S.W.; Weed, S.B.; Bowen, L.H. Iron oxide mineralogy of well-drained ultisols and oxisols. II. Influence on color, surface area and phosphate retention. *Soil Sci. Soc. Am. J.* **1978**, *42*, 825–830.
9. Macedo, J.; Bryant, R.B. Morphology, mineralogy, and genesis of a hydrosequence of oxisols in Brazil. *Soil Sci. Soc. Am. J.* **1987**, *51*, 690–698.
10. Macedo, J.; Bryant, R.B. Preferential microbial reduction of hematite over goethite in Brazilian oxisol. *Soil Sci. Soc. Am. J.* **1989**, *53*, 1114–1118.
11. Buol, S.W.; Camargo, M.N. Wet oxisols. In *Characterization, Classification, and Utilization of Wet Soils*. In Proceedings of Eighth International Soil Correlation Meeting (VIII ISCOM), Kimble, J.M., Ed.; USDA, Soil Conservation Service, National Soil Survey Center: Lincoln, 1992; 41–49.
12. Eswaran, H.; Van Den Berg, E.; Reich, P. Organic carbon in soils of the world. *Soil Sci. Soc. Am. J.* **1993**, *57*, 194.
13. Couto, W.; Sanzonowicz, C.; De, O.; Barcellos, A. Factors affecting oxidation–reduction processes in an oxisol with a seasonal water table. *Soil Sci. Soc. Am. J.* **1985**, *49*, 1245–1248.
14. EMBRAPA. *Mapa de Solos do Brasil (Escala 1:5,000,000)*. Serviço Nacional de Levantamento e Conservação de Solos; EMBRAPA: Geocarta, 1981.
15. Lopes, A.S.; Cox, F.R. A survey of the fertility status of surface soils under Cerrado vegetation in Brazil. *Soil Sci. Soc. Am. J.* **1977**, *41*, 742–747.
16. Lopes, A.S.; Cox, F.R. Cerrado vegetation in Brazil: An edaphic gradient. *Agron. J.* **1977**, *69*, 828–831.
17. Lopes, A.S. Soils under Cerrado: A success story in soil management. *Better Crops Int.* **1996**, *10* (2), 9–15.
18. Ritchey, K.D.; Sousa, D.M.G.; Lobato, E.; Sousa, O.C. Calcium leaching to increase rooting depth in a Brazilian Savannah oxisol. *Agron. J.* **1980**, *32*, 40–44.

Pampas, The

Martín Díaz-Zorita

Experimental Agricultural Station, National Institute of Technical Agriculture
(EEA INTA), General Villegas, Argentina

Daniel E. Buschiazzo

La Pampa University and National Institute of Agricultural Technology,
Buenos Aires, Argentina

Abstract

The soils of the Pampas are mostly Mollisols under udic and thermic moisture and temperature regimes (Chernozems, Kastanozems, and Phaeozems under the Food and Agriculture Organization soil classification system) used for agricultural practices for continuous annual crops, integrated crops, and livestock or extensive livestock production systems. Wind and water soil erosion and soil organic matter and nutrient depletion are the major constraints for crop production. For the management of these limitations, the use of no-till production systems, crop rotations, and fertilization and inoculation for legumes practices are recommended for the establishment of sustainable production systems. However, more research is required for the development of integrated nutrient and crop management practices for the different crop sequences within the subregions of the Pampas.

INTRODUCTION

The Pampas region is a vast plain of approximately 52 Mha located along the central part of Argentina. It is an extension of a bigger plain wedged between the Andes in the West and the Guyana–Brazilian shield in the east including the Venezuelan llanos, the Amazonian lowlands, and the Chaco–Pampas region. According to rainfall and soil quality patterns,^[1] the Pampas can be divided into five subregions: Rolling Pampas, Inland or Central Pampas, Southern Pampas, Flooded Pampas, and Mesopotamian Pampas (Fig. 1).

The climate is warm temperate with adequate to less than adequate rainfall for normal crop production. The mean temperature is 18°C and 14°C in the north and the south of the Pampas, respectively.^[2] In part of the region, the temperature and the frost-free period are adequate for growing double crops (i.e., soybean or maize planted after wheat). Rainfall amounts show high inter-annual variability. Most rainfall occurs between October and April (spring to fall seasons), and the long-term annual average is 500 and 1000 mm in the southwest and in the northeast of the region, respectively.^[2] Although there has been an increase in precipitation, at the rate of 100 mm per year,^[3] the water balance during the summer is negative with long periods of droughts of 2–6 weeks.

Most of the region was originally covered by grasslands, interrupted only by gallery forests along the margins of the Paraná, Uruguay, and La Plata rivers. Most of the region is under agricultural land use and, excluding the flooded Pampas, covers approximately 34 Mha. Almost

60% of the agricultural area is covered by perennial pastures comprising alfalfa (*Medicago sativa* L.), fescue (*Festuca arundinacea* L.), and other grasses under direct grazing livestock systems. The rest of the area is involved in the annual crop production, comprising 10 Mha of soybean [*Glycine max* (L.) Merrill], 2.4 Mha of maize (*Zea mays* L.), 6.3 Mha of wheat (*Triticum aestivum* L.), 1.8 Mha of sunflower (*Helianthus annuus* L.), and other crops.^[4] In the rolling Pampas and in the eastern parts of the inland Pampas, almost 100% of the area falls under the annual crop sequences while the proportion of area covered by pastures increases toward the boundaries of the Pampas. The flooded Pampas is mostly under native grasslands for cattle production systems.

MAIN SOILS OF THE PAMPAS

The parent materials are homogeneous comprising Pleistocene and Holocene Loessial sediments, with variation in texture according to the distance from the Andes, being sandier to the southwest of the Pampas. The mineralogy of the coarse fractions is dominated by the almost unweathered volcanic material (volcanic glasses) and plagioclases and lithic fragments and has a minor proportion of quartz and orthoclases. The clay fraction of the topsoil contains mostly illite along with kaolinite and illite–smectites intergrades (rarely well-crystallized montmorillonite).^[2]

The most frequent cropped soils of the Pampas are Mollisols, with prevailing udic and thermic moisture and

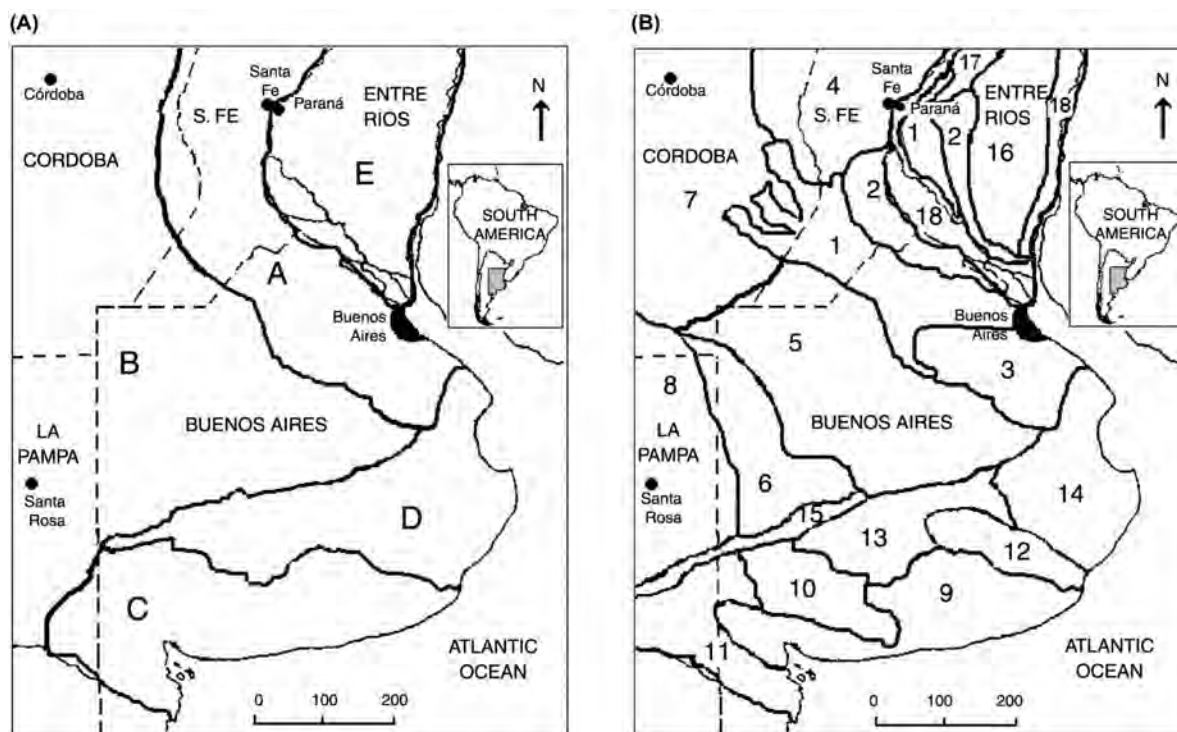


Fig. 1 Location, (A) subregions and (B) main soil associations of the Pampas region, Argentina, showing boundaries for the subregions (solid line). Provinces lying partly within the area of interest are named and their boundaries shown (dashed line). Inset shows location of area within South America. The references for subregions and soil types are presented in [Table 1](#).

temperature regimes [mostly Chernozems, Kastanozems, and Phaeozems after the Food and Agriculture Organization (FAO) classification system]. A summary of prevailing soil associations within each subregion of the Pampas is presented in [Table 1](#). In general, surface soil is moderately acidic with a high base saturation dominated by Ca^{2+} and K^+ .^[2] Soil organic matter (SOM) is a relevant soil property related to crop productivity because of its contribution to the soil structure, water-holding capacity, and nutrient supply.^[5] The SOM of the top layer decreases from almost 40 g/kg to less than 15 g/kg from northeast to southwest. A change in texture of the soil, i.e., from a fine to coarse texture, is noticed along the northeast to southwest direction.^[6] Nitrogen (N) and to a lesser extent phosphorus (P) supply are low for normal crop production in most of the areas along the Pampas region. In localized areas, sulfur and several micronutrients (i.e., boron, chloride, and zinc) are deficient for high yielding crops. From the point of view of crop production, water storage capacity is an important differentiating factor among soil types in the Pampas. Potential available water in the top 100 cm is 83 and 172 mm on Entic Haplustolls and Vertic Argiudolls, respectively.^[2] Shallow soils have a serious limitation to crop production in the Southern Pampas, where the topsoil depth is less than 50 cm in approximately 50% of the areas because of the presence of a petrocalcic horizon ([Figs. 2–4](#)).

SOIL RELATED CONSTRAINTS TO CROP PRODUCTION

Soils of the Pampas are susceptible to erosion by water and wind. Erosion by water predominates in the Rolling and the Mesopotamian Pampas where intensive rainfalls, silty topsoil textures, and long slopes predominate.^[2] Wind erosion, caused by a combination of tillage operations and strong winds during the dry season (winter), is a severe hazard in the west part of the Inland and the Southern Pampas.^[7] In shallow depressions, in the Inland Pampas, temporary ponds of water and saline soils are frequently observed. In saline patches, salts are leached down the profile in years with normal rainfall. The area under severe saline conditions is reduced and located close to permanent water formations.

SOM declines are mostly associated with annual row crop production systems using tillage.^[7] The use of soil pastures in rotation with annual crops contributes to recover SOM levels, and it is a widely adopted production system toward the marginal areas of the Pampas.^[2,7] The use of no-tillage practices, mostly for annual crops, is increasing every year and it contributes, among other factors, to the conservation of SOM levels.^[8,9]

The risk of topsoil compaction by machinery traffic increases in areas having low SOM levels and high silt content. The risk of compaction is more toward the Rolling

Table 1 Main soil associations within subregions of the Pampas.

Subregion	USDA soil classification system	FAO soil classification system
A Rolling Pampas	Typic Argiudolls	Luvic Phaeozems
	Vertic Argiudolls	Vertic Luvisols
	Typic Argiudolls-Aquic Argiudolls-Argiaquic Argialbolls	Luvic Phaeozems, luvic Phaeozems, mollic Planosols
	Typic Argiudolls-Typic Argiabolls-Typic Argiaquolls	
B Central or Inland Pampas	Entic Hapludolls-Typic Hapludolls-Thaptoargic Hapludolls	Haplic Phaeozems, luvic Phaeozems (on buried Argic horizon)
	Entic Hapludolls-Typic Hapludolls	Haplic and calcic Kastanozems, Arenosols
	Typic Haplustolls-Entic Haplustolls-Udic Haplustolls	
	Entic Haplustolls-Petrocalcic Paleustolls-Typic Ustipsamments	
C Southern Pampas	Petrocalcic Paleustolls-Typic Argiudolls	Kastanozems (on calcrete), luvic Phaeozem
	Udic Petrocalcic Paleustolls-Typic Argiudolls	Kastanozems (on calcrete), luvic Phaeozem
	Ustic Torripsaments-Petrocalcic Calciustolls-Aridic Haplustolls	Arenosols, calcic and haplic Kastanozems
	Typic Argiudolls-Lithic Argiudolls-Udic Petrocalcic Paleustolls	Luvic Phaeozem (on bedrock), Phaeozems, Planosols
D Flooding Pampas	Petrocalcic Natraquolls-Abruptic Argiacuolls-Udic Petrocalcic Paleustolls	Mollic-gleyic Solonetz (on calcrete), gleyic Greyzems, Planosols (on calcrete)
	Typic Natraquolls-Typic Argialbolls-Abruptic Argiaquolls Aquic Natrustalfs-Entic Haplustolls	Mollic-gleyic Solonetz, luvic Chernozems, gleyic Gryzems (with sharp limits)
E Mesopotamian Pampas	Typic Pelluderts	Pellic Vertisol
	Typic Ochraqualfs-Typic Natraqualfs	Chromic-orthic Luvisol, gleyc Solonetz
	Typic Udifluvents and alluvial complexes	Fluvisols and alluvial complexes

Note: For geographic distribution, see Fig. 1.

Source: From Viglizzo, Lértora, et al.,^[1] Hall, Rebella, et al.,^[2] Moscatelli, Salazar Lea Plaza, et al.,^[10] and Pazos & Moscatelli.^[11]



Fig. 2 Typic Hapludoll.

and the Mesopotamian Pampas and when summer crops are harvested soon after the rains.^[12]

Grain yields of crops and livestock production per unit of area are increasing every year.^[4] However, replacement of nutrients harvested in the grains and other agricultural products is lower than the amount applied by fertilizers. On average, only an equivalent of 40% of the P in the grains harvested in major crops of the Pampas is applied annually with the fertilizers. Thus, the area with



Fig. 3 Typic Ustisamment.



Fig. 4 Typical Argiudoll.

extractable P deficiency has increased from less than 50% to almost 80% of the Pampas between the late 1970s and the end of the 1990s (Fig. 5).^[13,14]

MANAGEMENT OPTIONS FOR SUSTAINABLE LAND USE

No-till farming and crop sequences integrating cereals (i.e., maize and wheat) with oil crops (i.e., soybean and sunflower) are the recommended management practices for sustainable land use in most of the Pampas.



Fig. 5 Wind erosion in central Pampas.

But, toward the marginal boundaries of the region (i.e., west part of the Inland Pampas and the southwest of Southern Pampas), the rotation of annual crops and perennial pastures under livestock production is recommended. Numerous research and extension reports show that the adoption of continuous no-till cropping practices, also in rotation with pastures, reduces soil erosion, conserves SOM, consolidates soil structure, reduces the risk of soil compaction, and improves productivity.^[8] Soil tillage increases the disturbance of soil, promotes the loss of SOM, and reduces its productivity in most of the region. Use of crop residue mulch is also recommended for conserving soil water. In the Mesopotamic Pampas, terracing and no-till cropping systems are the recommended soil management practices for decreasing the risk of soil erosion by water.

The efficient management of SOM and related properties depend on the lack of soil disturbance and carbon (C) input to the soil based on above- and below-ground productivity of the crops. In most of the Pampas, the use of N and P fertilizers are recommended for agronomically optimum crop production. In the case of soybean and other legumes (i.e., peanuts and alfalfa), an efficient and yearly use of inoculants providing selected strains of rizobia is also recommended (Fig. 6).

RESEARCH AND DEVELOPMENT PRIORITIES

During the last years, the agricultural practices broadly expanded from the grain belt (mostly in the Rolling Pampas) through the rest of the region. Because of market and ecological conditions, soybean is the most common crop and, despite the recommendation of growing it in rotation with cereals, is mostly planted as monoculture. Thus, focused research is needed for developing sustainable production systems capable of sequestering C and related beneficial effects on soil properties, along with high profit for



Fig. 6 Central or inland Pampas maize crop.

the farmers, so as to meet the competition arising out of the production of monoculture soybean.

Developing no-till production systems involving integration of annual crops and livestock production on marginal areas is a high priority. These lands have been under cultivation since the beginning of the 21st century. There is also a need to conduct research for strengthening the nutrient-cycling mechanisms within different production systems for improving the development of integrated nutrient management practices for different crop sequences and subregions of the Pampas.

CONCLUSION

The soils of the Pampas are mostly Mollisols under udic and thermic moisture and temperature regimes (Chernozems, Kastanozems, and Phaeozems under the FAO soil classification system) used for agricultural practices for continuous annual crops, integrated crops and livestock or extensive livestock production systems. Wind and water soil erosion and SOM and nutrient depletion are the major constraints for crop production. For the management of these limitations, the use of no-till production systems, crop rotations, and fertilization and inoculation for legumes practices are recommended for the establishment of sustainable production systems. However, more research is required for the development of integrated nutrient and crop management practices for the different crop sequences within the subregions of the Pampas.

REFERENCES

- Viglizzo, E.F.; Lértora, F.; Pordomingo, A.J.; Bernardos, J.N.; Roberto, Z.E.; Del Valle, H. Ecological lessons and applications from one century of low external-input farming in the pampas of Argentina. *Agr. Ecosyst. Environ.* **2001**, *83*, 66–81.
- Hall, A.J.; Rebella, C.M.; Ghersa, C.M.; Culot, J.P. Field-crop systems of the Pampas. In *Field Crop Ecosystems*; Pearson, C.J., Ed.; Elsevier: Amsterdam, 1992; 413–450.
- Sierra, E.M.; Hurtado, R.H.; Spescha, L.; Barnatan, I.; Messina, C. Corrimiento de las isoyetas anuales medias decenales (1941–1990) en la región pampeana (Displacement in the annual mean rainfall isolines (1941–1990) in the Pampas region). *Rev. Fac. Agron. UBA* **1995**, *15*, 137–143.
- Argentine Secretary of Agriculture. *Livestock, Fisheries and Food*, 2003. Available at <http://www.sagpya.gov.ar>.
- Díaz-Zorita, M.; Buschiazso, D.E.; Peinemann, N. Soil organic matter and wheat productivity in the semiarid Argentine Pampas. *Agron. J.* **1999**, *91*, 276–279.
- Alvarez, R.; Lavado, R.S. Climate, organic matter and clay content relationships in the Pampa and Chaco soils, Argentina. *Geoderma* **1998**, *83*, 127–141.
- Buschiazso, D.E.; Panigatti, J.L.; Unger, P.W. Tillage effects on soil properties and crop production in the subhumid and semiarid Argentinean Pampas. *Soil Till. Res.* **1998**, *49*, 105–116.
- Díaz-Zorita, M.; Duarte, G.A.; Grove, J.H. A review of no-till systems and soil management for sustainable crop production in the subhumid and semiarid Pampas of Argentina. *Soil Till. Res.* **2002**, *65*, 1–18.
- García, F.O.; Ambroggio, M.; Trucco, V. No-tillage in the pampas of Argentina: A success story. *Better Crops Int.* **2000**, *14*, 24–27.
- Moscatelli, G.; Salazar Lea Plaza, J.C.; Godagnone, R.; Grinberg, H.; Sanchez, J.; Ferrao, R.; Cuenca, M. Mapa de suelos de la provincia de Buenos Aires (Buenos Aires province soil map). In *Proceedings IX Argentine Soil Science Meeting*, Buenos Aires, Argentina, 1980. *Asociación Argentina de la Ciencia del Suelo*, Eds.; 1079–1089.
- Pazos, M.S.; Moscatelli, G. The WRB applied to Pampean soils—Argentina. In *Proceedings XVI World Congress of Soil Science*; Montpellier, France, 1998. *International Soil Science Society*, Eds.; in CD.
- Díaz-Zorita, M.; Grosso, G.A. Effect of soil texture, organic carbon and water retention on the compactability of soils from the Argentinean Pampas. *Soil Till. Res.* **2000**, *54*, 121–126.
- Montoya, J.; Bono, A.A.; Suárez, A.; Darwich, N.; Babinec, F. Cambios en el contenido de fósforo asimilable en suelos del este de la Provincia de La Pampa, Argentina (Changes in available P levels in soils from the Eastern part of La Pampa province, Argentina). *Ciencia del Suelo* **1999**, *17*, 45–48.
- García, F.O. Phosphorus balance in the Argentine Pampas. *Better Crops Int.* **2001**, *15*, 22–24.

Pantanal, The

Henrique de Oliveira

Brazilian Corporation of Agricultural Research (EMBRAPA), Corumba, Brazil

Amaury de Carvalho Filho

Soils and Crops Research Centre, Brazilian Corporation of Agricultural Research (EMBRAPA), Rio de Janeiro, Brazil

Carlos Ernesto Reynauld G. Schaefer

Soil Department, Federal University of Vicosa, Vicosa, Brazil

Evaldo Luis Cardoso

Brazilian Corporation of Agricultural Research (EMBRAPA), Corumba, Brazil

Abstract

In the Pantanal, four pedological zones can be distinguished: 1) a Vertisols–Planosols–Solonetz area of Miranda, Aquidauana, Abobral, Nabileque, and Porto Murtinho subregions; 2) a Hydromorphic Podzol and Arenosols area in the central part, encompassing the alluvial fan of Taquari River (Paiguás and Nhecolândia subregions); 3) Plinthosol–Planosol area to the north of São Lourenço River (Barão de Melgaço, Poconé, and Cáceres subregions); and finally 4) the Gleisols/Alluvials of the Paraguai floodplain, where Hydromorphic Planosols and associated Solonetz are dominant (Paraguai subregion).

INTRODUCTION

The Pantanal is an extensive wetland located in the center of South America (16–21°S; 55–58°W), possessing an extraordinary variety of ecosystems, which constrains a general characterization of the area. Basically, it is a vast floodplain that is seasonally inundated, with approximately 140,000 km² in the Brazilian territory (Fig. 1), extending over Bolivia and Paraguay, and with low-lying relief, ranging from 80 to 150 m.

GEOLOGICAL AND CLIMATIC ASPECTS

The main feature of the area is an intracratonic tectonic depression infilled by Quaternary sediments and surrounded by crystalline rock terrains, which are the main water and sediment source area. Its formation is linked with later events of Andean orogeny, in the late Tertiary/early Quaternary, with intense block faulting, from the Miocene onward.^[1] The active subsidence of tectonic blocks in Central South America was accompanied by an upward movement of the “Bodoquena Platô Horst,” creating a rock barrier, which dammed the outlet of Paraguai River, and damming “lag” sediments within the depression of the Pantanal.^[2] The most abundant Quaternary deposits are interpreted as products of intense erosion of the bordering marginal plateaux, infilling the low-lying

Paraguai depression. In the early Quaternary, this basin was virtually endorheic. Then, the Taquari River, coming from highland areas of the Paraná/Araguaia/Amazonas watershed, broke its outlet onto the friable, soft sediments, building a vast sandy, alluvial fan under semiarid conditions.

This semiaridity prevailed during the early mid-Quaternary deposition of the “Pantanal Formation,” with a trend of increasing aridity. Eventually, increasing rainfall of late Quaternary time broke the dammed Paraguai, creating its outlet into the Paraná basin, with increasing water volume.

The climate is tropical subhumid (Aw, Koeppen) with mean annual temperature of 26°C, and rainfall in the range of 800–1200 mm.^[3]

SUBDIVISION OF THE PANTANAL AND HYDROLOGICAL ASPECTS

The geographic position of the Pantanal represents a crossroad between Amazonia, Chaco, Cerrado, and Atlantic Forest biomes, favoring the exchange of species.^[4] The interplay between complex biotic and abiotic factors resulted in high heterogeneity of landscapes within the plains, characterizing distinct subregions. On hydrological and geomorphological basis, the Pantanal is formed by 10 subregions.^[5] Reviewing the cartographic database,



Fig. 1 Localization of the Pantanal in South America.

11 subregions were identified;^[6] these are called the Pantanaís of Cáceres, Poconé, Barão de Melgaço, Paiaguás, Nhecolândia, Aquidauana, Miranda, Abobral, Paraguai, Nabileque, and Porto Murtinho (Fig. 2).^[7]

Local and regional climatic forces influence the hydrological behavior of the Pantanal. Rains are concentrated between October and March and usually lead to widespread flooding because of insufficient drainage outlet. In the north sector of the Pantanal, the high water level occurs during January–March, whereas the southern part is affected from April until June. The timing of the highest level is reached in February in the north, and in June in the south Pantanal, as a result of slow movement of Paraguai River waters.^[5]

The widespread floodings are time/space-constrained ecological phenomena, being variable with respect to intensity, duration, and depth. Extensive flooding may be due to either pluvial water or runoff from adjacent high plateau,

being exacerbated by high water-table level and sluggish movement of rivers in the Pantanal. Between half and a third of the Pantanal area is subjected to fluvial inundations, whereas the majority is due to local rainfall.^[8]

In this periodic inundation regime, hydromorphism is a dominant process, which is due to ill-drainage, which affects all soils of the Pantanal, especially those developed from alluvial sediments. High-nutrient inputs are prevalent only in fluvial inundation, whereas rains can cause extensive waterlogging without enrichment.

SOILS OF THE PANTANAL

The main soils of the Pantanal are Planosols, Hydromorphic Podzol, Plinthosols, Gleisols, solodized Solonetz, Vertisols, and Arenosols (hydromorphic or well-drained).

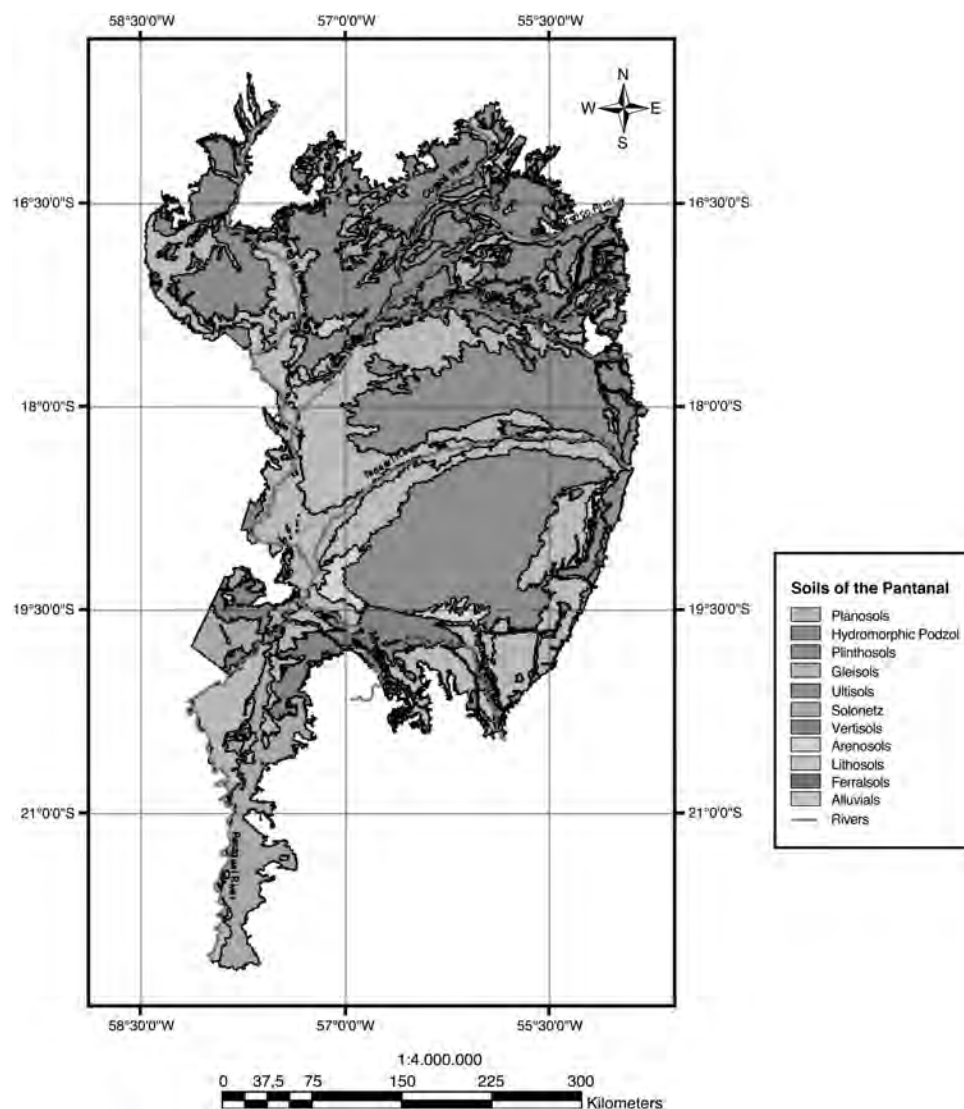


Fig. 3 Soils of the Pantanal.

Planosols are typically found in flat, low-lying terraces across the whole Pantanal, associated with dry savanna woodland. These soils are closely associated with sedimentary deposits of the Pantanal Formation (Quaternary), possessing shallow profiles under hydromorphic regimes. They show an argillic horizon with an abrupt contact with an overlying E, which is normally fractured along the horizontal contact during the dry period. The B horizon (planic) is dense and impervious, showing features related to seasonal ill-drainage, such as mottling and gleying. These soils commonly have a plinthic or petroplinthic character and can be either eutric or dystric. Most are solodic, due to the high sodium (Na) percentage of the cation exchange capacity (CEC), particularly in the south and northwest Pantanal. The agronomic importance is limited due to chemical and physical constraints, and natural pastures are the common land use.^[9–11]

Solodized Solonetz are soils with great similarities with the Planosols, but their B horizons present an even higher

Na saturation, which makes them sodic (or natric), which is associated with poor drainage conditions. Vegetation is called “Bosque Chaquenho,” characterized by xerophytic vegetations rich in cactaceae and deciduous trees mixed with savanna species and Carandá (*Copernicia* sp.), sometimes forming dense palm forests called “Carandazal.” They are often found in the south Pantanal in the Nabileque/Apa interfluves.^[9–11]

Plinthosols can be either hydromorphic or not but are always related to hard, dense subsoils with limited internal drainage. They have a plinthic horizon within the first 40 cm, and abundant mottling with greater depths. They are usually associated with open savanna grassland, being either dystric or eutric, depending on the parent material, but the former is dominant. They occur north of the Piquiri River and in the Paraguai depression.^[9]

Gleisols (humic or nonhumic) are hydromorphic mineral soil with a glei horizon not accompanied by a plinthic or an argillic one. They cannot have vertic, natric, or salic properties and are less important in the Pantanal lowlands

compared with the former hydromorphic soil classes. Vegetations on Gleisols are typically hydrophilous fields, and nutrient status varies considerably, but eutric types are dominant.^[9–11]

In the top of the well-drained uplands of the Pantanal, Arenosols (quartzous sands) are distinguished by sandy texture, little evolution, and A and C horizon sequences. They are very frequent in large extension of the Taquari alluvial fan, associated with ill-drained Hydromorphic Podzols in the lower landscape positions. The vegetation is closed savanna woodland, which progressively becomes grassy with high water-table level. The transition zone between the quartzous sands and Podzols is hydromorphic quartzous sands, always dystic and with very low CEC.^[9–11]

Vertisols are hydromorphic clayey soil, which occurs in the Pantanal wherever high activity clays formed from alluvium or wherever calcareous rocks are dominant, showing little horizonation (A–C). Their shrinking–swelling character gives them the typical vertic morphology, with slickensides and columnar structure, with or without calcium carbonate (CaCO₃). They are mostly associated with CaCO₃-rich sediments of the Xaraés Formation, being common between the Aquidauana and Paraguai Rivers, in the south Pantanal, and in the Cuiabá River, in the north Pantanal. The vegetation is dry forest with palms and cacti, and pastures are the dominant land use.^[9,10]

CONCLUSION

In the Pantanal, four pedological zones can be distinguished: 1) a Vertisols–Planosols–Solonetz area of Miranda, Aquidauana, Abobral, Nabileque, and Porto Murtinho subregions; 2) a Hydromorphic Podzol and Arenosols area in the central part, encompassing the alluvial fan of Taquari River (Paiaguás and Nhecolândia subregions); 3) Plinthosol–Planosol area to the north of São Lourenço River (Barão de Melgaço, Poconé, and Cáceres subregions); and finally 4) the Gleisols/Alluvials of the Paraguai floodplain, where Hydromorphic Planosols and associated Solonetz are dominant (Paraguai subregion).

REFERENCES

1. Godoi Filho, J.D. Aspectos Geológicos do Pantanal Mato-Grossense e de Sua Área de Influência. In *Simpósio Sobre Recursos Naturais e Sócio-Econômicos do Pantanal*, 1, 1984; EMBRAPA-DDT: Corumbá, 1986; 63–76.
2. Silva, T.C. Contribuição da Geomorfologia Para o Conhecimento e Valorização do Pantanal. In *Simpósio Sobre Recursos Naturais e Sócio-Econômicos do Pantanal*, 1, 1984; EMBRAPA-DDT: Corumbá, 1986; 63–76.
3. Júnior, J.H.C.; Sandanielo, A.; Canappele, C.; Priante Filho, N.; Musis, C.R.; Soriano, B.M.A. Climatologia. In *BRASIL. Ministério do Meio Ambiente, dos Recursos Hídricos e da Amazônia Legal. Plano de Conservação da Bacia do Alto Paraguai (Pantanal)-PCBAP; Diagnóstico dos Meios Físico e Biótico: Meio Físico*, 1997; Vol. 2, t.1, 295–334.
4. Adámoli, J. *Diagnóstico do Pantanal: Características Ecológicas e Problemas Ambientais*; Programa Nacional do Meio Ambiente: Brasília, 1995; 50 pp.
5. Hamilton, S.K.; Sippel, S.J.; Melack, J.M. Inundation patterns in the Pantanal wetland of South America determined from passive microwave remote sensing. *Arch. Hydrobiol.* **1996**, 137 (1), 1–23.
6. dos Silva, J.S.V.; dos Abdon, M.M. Delimitação do Pantanal brasileiro e suas sub-regiões. *Pesqui. Agropecu. Bras.* **1998**, 33, 1703–1711.
7. Adámoli, J. Vegetação do Pantanal. In *Recursos Forrageiros Nativos do Pantanal Mato-Grossense*; Allem, A.C., Valls, J.F.M., Eds.; EMBRAPA-CENARGEN: Brasília, 1987; 339.
8. dos Silva, J.S.V. Elementos fisiográficos para delimitação do ecossistema Pantanal: Discussão e proposta. *Oecol. Bras.* **1995**, 1, 439–458.
9. Santos, R.D.; Carvalho Filho, A.; Naime, U.J.; de Oliveira, H.; Motta, P.E.F.; Baruqui, A.M.; Barreto, W.O.; Melo, M.E. C.C.M.; Paula, J.L.; Santos, E.M.R.; Duarte, M.N. Pedologia. In *BRASIL. Ministério do Meio Ambiente, dos Recursos Hídricos e da Amazônia Legal. Plano de Conservação da Bacia do Alto Paraguai (Pantanal)-PCBAP; Diagnóstico dos Meios Físicos e Bióticos: Meio Físico*, 1997; Vol. 2, t.1, 121–293.
10. Oliveira, J.B.; Jacomine, P.K.T.; Camargo, M.N. *Classes Gerais de Solos do Brasil: Guia Auxiliar Para Seu Reconhecimento*; FUNEP: Jaboticabal, 1992; 201 pp.
11. Camargo, M.N.; Klant, E.; Kauffman, J. Classificação de solos usada em levantamento pedológico no Brasil. *Bol. Inf. Soc. Bras. Ciên. Solo, Campinas* **1987**, 12 (1), 11–33.

Paramos Soils

Pascal Podwojewski

Institute for Research and Development (IRD), Hanoi, Vietnam

Jérôme Poulenard

Sciences of the Sun Laboratory, Interdisciplinary Scientific Center of Montagne,
University of Savoie, Savoy, France

Abstract

The Spanish word *páramo* refers to the deserted, windy, humid, and cold North Andean areas. This high-altitude grassland ecosystem, which has large plant diversity, covers 35,000 km² and is distributed as discontinuous islands between the high altitudinal forest and the permanent snow line. In the *Páramos*, soils evolve in relation with low temperature, high soil moisture, and aluminum availability of the parent rock. Carbon content and water retention are very high. *Páramos* soils are considered as a “sponge” which enhance water retention and buffer the water flow downstream. However, new agricultural practices modify soil properties, altering infiltration and runoff regime and disturbing the primary hydrological function of the *Páramos*.

PÁRAMOS ENVIRONMENT

Páramo Location

The Páramo is located in intertropical Central and South America, between 11° latitude north and 8° latitude south (Fig. 1), mainly situated in Colombia (14,500 km²), in Ecuador (12,600 km²) but also in Venezuela with small extensions in Costa Rica, Panama, and Northern Peru (Table 1).^[1–3] It occurs at an altitude generally ranging between 3200 and 4800 m.

Páramos Climate

With an average decrease of 0.6°C for 100 m change in elevation, temperatures are cold with an annual average of 9°C at 3500 m. Day–night amplitude is over 15°C with temperatures dropping randomly below 0°C over 4200 m. Annual variation is very low.

The windward external sides of the cordilleras receive more rainfall than the leeward internal sides (interandean valleys side), where rainfall increases with altitude. Annual Precipitation ranges between 1000 and 1500 mm with extremes between 500 and 3000 mm. In the northern tropical areas, the rainfall has a unimodal distribution with one dry season, in the southern area closer to the equator, rainfall occurs all year long with a bimodal distribution of two drier seasons. Intensities are very low, rarely exceeding 40 mm/day. Frequent fogs, drizzles, and hailstorms contribute to maintain a humid climate, and air moisture ranges between 70% and saturation.^[4]

Páramos Vegetation

The Páramo is a grassland biotic formation of the high Andes and contains between 3000 and 4000 species of vascular plants with 60% estimated endemism.^[5]

The vegetation of the Páramos can be divided into three belts:^[3]

1. The *super-Páramo*, at high altitudes ranging between 4200 and 4800 m, represents a periglacial environment with sparse vegetation cover and species adapted to nocturnal freezing and high solar radiation by leaf xeromorphic adaptation (reduced surfaces, folded, densely woolly, whitish leaves).
2. The *proper-Páramo* (3500–4200 m) has an open-type vegetation dominated by bunchgrasses (tussocks). Poaceae (*Calamagrostis* sp. and *Festuca* sp.) and dwarfed bamboos form the *pajonal* and cover 70% of the soil surface. The giant rosettes are the most remarkable plants [*Espeletia* sp. (Fig. 2) and *Puya* sp.]. Acaulescent rosettes, cushion-like vegetation (especially in swampy areas), and some shrubs complete vegetation. Patches of forest (*Polylepis* sp.) grow up to an altitude of 4200 m.
3. The *sub-Páramo* (3200–3500 m) is a transitional zone between the Páramo and the Andean forest; the coverage of small trees and scrubs is greater than in the proper-Páramo. The sub-Paramo surface extends toward lower altitude with human land clearing to provide pasture and farmland.



Fig. 1 World distribution of the *Páramo* Ecosystem.

Source: From Robert Hofstede, Proyecto Páramo (<http://www.paramo.org>).

Table 1 Extension of the *Páramo* in different countries (% of the total).

Country	Surface (km ²)	% of total
Colombia	14,430	1.3
Costa Rica	80	0.2
Ecuador	12,600	5.1
Peru	4200	0.3
Venezuela	3990	0.4

Source: From Robert Hofstede, Proyecto Páramo (<http://www.paramo.org>).

Geology–Geomorphology

The Páramos follow the summital portion of the main northern Andes Mountain chain. The most important part of this belt from the center of Colombia to the center of Ecuador has a strong volcanic activity. Ash falls are mainly rhyodacitic to andesitic in composition and cover the basement of the Andean belt made of sedimentary or metamorphic rocks.

The geomorphology of the Páramo was modeled by glacier extension during the major glacier phases at altitudes of over 4000 m, with large U-formed valleys, many lakes and smoothened relieves.^[4] Two main forms can be distinguished: 1) highly erodible forms affected by volcanic ash deposits, with uniformed small hills of convex



Fig. 2 Profile of a Melanudand in the *Páramo* of the ecological reserve of El Angel, Carchi province, north of Ecuador.

concave forms, gentle slopes, and isolated volcanic cones (>5000 m in altitude) emerging from these chains and 2) non-volcanic rocks or hard lava flows with notched summits and steep slopes.^[4]

SOILS

Páramos soils have been described by Jenny since 1948.^[6] Soils evolve in function of the convergent effects of low temperature, high soil moisture, and aluminum (Al) availability.^[7]

In the supra-*Páramo*, organic matter production decreases with altitude, and soils are very shallow, with umbric epipedon^[8] and periglacial features. In non-volcanic Venezuelan areas, Cryepts are dominant. The presence of giant rosette plants gives the soil pattern discontinuous properties with higher carbon (C) content only at soil surface and around the rosette.^[9] In volcanic areas, during glacier periods, ice caps protected soils from ash deposits. The deposits are weakly weathered and form Vitrandic Cryents.

At lower altitudes, in humid conditions, the availability of Al is the second important factor in soil formation. When Al availability is very low, Humods and Fragiaquods are observed on gneiss and micaschists rich in quartz (Podocarpus *Paramo* in the south of Ecuador). With higher Al availability, Inceptisols (Dystrudept) form with an

umbric epipedon. With high Al availability, Al forms very stable complexes with organic matter.^[10] The main pedological process is an acidic complexolitic andosolization with Al + 1/2 Fe oxalate extract >2% and Al% pyrophosphate/Al% oxalate >0.5 characteristic of non-allophanic Andisols,^[11] which are the more extended soil types in the *Páramos*. On volcanic ashes, important Al availability leads to a very fast soil formation, with Vitrands forming in less than 1000 years. The farther the deposits are from their emission source, the finer are the ashes, and the faster is the rate of weathering. The older, most evolved soil profiles (± 3000 years) have the lowest base reserve and can be classified as Udands. The non-allophanic andosolization process could also be active on nonvolcanic Al-rich substrates (paleo-Oxisols or metamorphic rocks). All of these soils are acidic and have anionic retention capacities (especially for phosphorous and sulfur).

***Páramo* Soils and C Content**

All *Páramo* soils are very rich in organic C. In the *Paramo* of Northern Peru, soils contain over 10 kg m⁻² C, whatever the nature of the parent rock (limestones, volcanic rocks, or sandstones).^[12] Low temperatures and stable organo-metallic complexes^[7] reduce the biological activity and therefore decrease the rate of organic matter mineralization. High C accumulation results from different pedologic processes on successive ash layers but also on colluvial buried soils on steep slopes. C sequestration can exceed 85 kg m⁻² in the polygenic non-allophanic matured Andisols of Northern Ecuador (Table 2, Fig. 2).^[13] Altitude increases the soil organic C content with a maximum density (reaching 10 g cm⁻³) at around 3900 m. These Andisols have black thick melanic epipedon attributed to the presence of humic acids produced by the degradation of Poaceae family plants^[11] and can be classified as Melanudands (Fig. 2).^[8]

***Páramos* and Water Cycle Regulation**

The water content in non-allophanic Andisols at 1500 kPa is over 1000 g kg⁻¹ (hydic properties)^[8] and can be 2000 g kg⁻¹ at field capacity. The formation of an organo-metallic complex network leads to important microporosity. The more mature the soils, the richer they are in humic substances, and the higher is their microporosity. For matured Andisols, micropores with a radius <0.1 μ m can form over 50% of the soil volume.^[14] Water content is

Table 2 C Pool (kg m⁻²) of some *Páramos* soils

Place	Pichincha (1)	Carchi (2)	Cajas (3)	Fierro Urcu (4)	Cerro Toledo (5)	Sabanilla (6)
A	35.6	46.3	36.4	34.1	15.8	11.2
B	56.7	86.4				

Note: 1, Thaptic Hapludand; 2, Hydric Pachic Melanudand; 3, Hydric Melanudand; 4, Humic Andic Hapludox; 5, Humic Dystrudept; 6, Typic Fragiaquod. A, first meter of the profile; B, whole profile.

stable all year long. The measured hydraulic conductivity of this mature Andisol reaches 60 mm hr^{-1} and runoff is unlikely.

After drying, a new and extremely rigid structure appears: drying is therefore partially irreversible. As porosity increases, so does sensitivity to modification of the pores architecture and the loss of water-storage properties after drying.^[14] Water-repellent surfaces develop during drying.

Degradation of the Páramo Soils

More than half of the *Páramos* surface was used as extended land with an increasing rural population for grazing and cultivation. First, burning formed temporary bare surfaces,^[2,15] which are receptive to structural decay by the kinetic energy of raindrops or hailstorms. Second, black C-rich bare surfaces absorb solar radiation, which favors irreversible drying and water repellency at soil surface. The upper 20 cm of Andisols are deeply affected with a convergent decrease of more than a third of their original value in C, water, and Al contents. A highly water-repellent soil microstructure replaces the stable macrostructure.^[16]

After tillage, juvenile vitric Andisols crust and runoff coefficient increases by a factor of 3. On mature Andisols, soil packing and the development of water repellence lead to severe decrease in hydraulic conductivity. At the same time, soil loss is very high (over 1500 g m^{-2}) and occurs via the entrainment of hydrophobic aggregates in the surface runoff.

CONCLUSION

Though covering relatively small surfaces, paramos soils present specific properties in water retention and C sequestration. Any degradation or inappropriate land use alters these soils durably with a negative effect on the water availability for a large population living downstream.

REFERENCES

1. <http://www.paramo.org>.
2. Balslev, H.; Luteyn, J.L. *Páramo. An Andean Ecosystem Under Human Influence*; Academic Press: London, 1992.
3. Troll, C. Geo-ecology of the mountainous regions of the tropical Americas. *Collect. Geogr.* **1968**, *9*, 223.
4. Winckell, A.; Zebrowski, C.; Sourdat, M. Los paisajes andinos de la sierra del Ecuador. In *Los Paisajes Naturales del Ecuador*; Winckell, A., Ed.; Geografía Básica del Ecuador. Tomo IV, volumen 2, Geografía Física, CEDIG Quito, 1997; 3–207.
5. Luteyn, J.L. Páramos. A checklist of plant diversity, geographical distribution and botanical literature. *Mem. New York Botan. G.* **1999**, *84*, 278.
6. Jenny, H. Great soil groups in the equatorial regions of Colombia, South America. *Soil Sci.* **1948**, *66*, 5–28.
7. Aran, D.; Gury, M.; Jeanroy, E. Organo-metallic complexes in an andosol. Comparative study with a cambisol and podzol. *Geoderma* **2001**, *99*, 65–79.
8. Soil Survey Staff. *Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys*; USDA-NRCS: Washington, 1999.
9. Perez, F.L. Plant-induced spatial patterns of surface soil properties near Caulescent Andean Rosettes. *Geoderma* **1995**, *68*, 101–121.
10. Boudot, J.P.; Hadj, A.B.; Chone, T. Carbon mineralization in andosols and aluminum-rich highland soils. *Soil Biol. Biochem.* **1986**, *4*, 457–461.
11. Shoji, S.; Nanzyo, M.; Dahlgren, R. *Volcanic Ash Soils. Genesis; Development in Soil Science*; Elsevier: Amsterdam, 1993; Vol. 17.
12. Escobedo-Urquiza, J. Les sols des páramos. Etude pédogénétique dans les hautes andes du Pérou septentrional [doctoral dissertation]. In *Thèse de sciences agronomiques*; Faculté des sciences de Gembloux, 1980.
13. Poulenard, J.; Podwojewski, P.; Herbillon, A.J. Characteristics of non-allophanic andisols with hydric properties in Ecuadorian Páramo. *Geoderma* **2003**, *117*, 267–281.
14. Poulenard, J.; Bartoli, F.; Burtin, G. Shrinkage and drainage in volcanic soil aggregates: a structural approach using air under vacuum drying kinetics and mercury porosimetry. *Eur. J. Soil Sci.* **2002**, *53* (4), 563–574.
15. Hofstede, R.G.M. *Effects of Burning and Grazing on a Colombian Páramo Ecosystem*. Ph.D. dissertation, University of Amsterdam: Amsterdam, 1995.
16. Podwojewski, P.; Poulenard, J.; Zambrana, T.; Hofstede, R. Overgrazing effects on vegetation cover and volcanic ash soil properties in the páramo of Llangahua and La Esperanza (Tungurahua, Ecuador). *Soil Use Manag.* **2002**, *18* (1), 45–55.

Particle Density

Keith R.J. Smettem

Soil Science and Plant Nutrition, University of Western Australia, Nedlands,
Western Australia, Australia

Abstract

This entry gives an overview of particle density and standard methods for measurement.

INTRODUCTION

Density is a standard physical term defined as weight (mass) per unit volume. In soils, the density of the solid particles in a particular sample is referred to as the particle density. SI units are kilograms per cubic meter, but the fractional units of grams per cubic meter are also used ($1 \text{ g cm}^{-3} = 1 \text{ Mg m}^{-3}$). In most soils, the average density of particles, ρ_s , is in the range $2.6\text{--}2.7 \text{ Mg m}^{-3}$. This narrow range reflects the predominance of quartz and clay minerals in the soil matrix. An average particle density of 2.65 Mg m^{-3} (the density of quartz) is often applied to soils comprising principally silicate materials. Exceptions can occur if the soil is high in organic matter, which reduces the soil particle density. Humus, e.g., has a density that is usually less than 1.5 Mg m^{-3} . Conversely, soils that are rich in iron can have high particle densities. Ferromagnesian minerals, e.g., have densities ranging from 2.9 to 3.5 Mg m^{-3} . Density of iron oxides and other so-called heavy minerals can exceed 4 Mg m^{-3} .

Ovendry soil is essentially a two-phase porous medium comprising air and solids. The porosity can be determined if the bulk density (ρ_b ; the ratio of the mass of dry solids to the bulk volume of the soil) and the particle density are known. The porosity, ϵ , is then given by the following equation:

$$\epsilon = 1 - \rho_b / \rho_s \quad (1)$$

The particle density is also necessary for the calculation of sedimentation parameter required for the estimation of particle size by sedimentation methods.^[1,2] It is also used to determine the particle settling velocity in studies of sediment deposition from flowing water.^[3]

MEASUREMENT OF PARTICLE DENSITY

Widely accepted standard methods are available for measuring the weighted mean particle density of a sample^[4] and are based on measuring the mass and volume. The

mass is determined by weighing, and the volume is then calculated from the mass and density of a fluid displaced by the sample. The two most common methods of measuring particle density are the pycnometer method^[5] and the submersion method.^[6]

A pycnometer is a specific gravity flask that is sometimes fitted with a thermometer. For large samples, a volumetric flask may also be used with sufficient measurement accuracy for most applications. In brief, the method involves careful weighing to obtain the particle density from the following equation:

$$\rho_s = \rho_w (W_s - W_a) / [W_{sw} - (W_s - W_a)] \quad (2)$$

where ρ_w is the density of water at the observed temperature, W_s is the pycnometer weight containing a soil sample at oven-dry water content, W_a is the empty (air-filled) pycnometer weight, W_{sw} is the pycnometer weight when filled with soil and water, and W_w is the pycnometer weight when filled with water at the observed temperature. Note that the denominator represents the volume of water displaced by the soil particles.

The soil in the pycnometer is covered with a layer of water and evacuated to ensure complete saturation of the soil before topping up the pycnometer. Note that de-aired distilled water is used throughout the procedure, and organic matter is generally removed prior to use.

An air pycnometer can be used as an alternative method, particularly if it is important to retain organic matter in the sample. The method uses gas rather than water as the displacing fluid and the ideal gas law to calculate the volume of solids. Care must be taken to ensure that the procedure is performed at a constant temperature.

The submersion method involves measuring the volumetric displacement of water or a non-polar liquid by a soil sample of known air-dry weight. The samples of air-dried soil, usually about 25 g in weight, are formed into spaghetti-like threads by moistening to a plastic consistency and forcing through a 2-mm sieve. After oven-drying at 105°C and cooling in a desiccator, the prepared sample is placed in a weighing dish attached to a weighing beam by a thin

wire, and both the dish and the sample are immersed in a container of liquid. The particle density then follows from the equation:

$$\rho_s = \rho_l(W_{sd} - W_d) / [(W_{sd} - W_d) - (W_{sdl} - W_{dl})] \quad (3)$$

where ρ_l is the density of water or organic liquid used, W_{sd} is the oven-dried weight of both the soil and the weighing dish, W_d is the weight of the weighing dish, W_{sdl} is the weight of sample and weighing dish submerged in liquid, and W_{dl} is the weight of the submerged dish without the soil sample.

The submersion method sacrifices some precision compared to the pycnometer method but is easy to perform and has the advantage of being less laborious. The method cannot, however, be used with sandy soils because coherence is usually too low to allow the spaghetti-like sample threads to be prepared.

The particle density of finely divided active soil measured in water is usually greater than that measured in a non-polar liquid.^[4] As the proportion of minerals with expanding lattices increases in a given sample, so too does the specific gravity.^[4] Water density is also known to be affected by the surfaces of finely divided particles, so a more accurate measure of particle density would be obtained using a non-polar inorganic liquid in a pycnometer.^[4] However, if the particle density information is required in order to determine the volume of solids in a soil in contact with water, then it is logical to use water as the measurement liquid.

If the soil is high in organic matter, then it is easier to achieve wetting using a non-polar organic liquid. Organic particles also have a tendency to be buoyant in water, whereas they sediment more readily in non-polar liquids. This is obviously an advantage when attempting to measure the weight of a submerged sample.

The density of a suspension can also be measured very precisely using a vibrating tube,^[7] which is filled with a

solution or suspension. The resonant frequency of the vibrating tube is directly related to the density of the suspension.

REFERENCES

1. Gee, G.W.; Bauder, J.W. Particle size analysis by hydrometer: A simplified method for routine textural analysis and a sensitivity test of measurement parameters. *Soil Sci. Soc. Am. J.* **1979**, *43*, 1004–1007.
2. Gee, G.W.; Bauder, J.W. Particle-size analysis. In *Methods of Soil Analysis 1: Physical and Mineralogical Methods*; Klute, A., Ed.; 2nd Ed.; American Society of Agronomy: Madison, 1986; 383–411.
3. Julien, P.Y.; Simons, D.B. Sediment transport capacity of overland flow. *Trans. Am. Soc. Agr. Eng.* **1985**, *28*, 755–761.
4. Blake, G.R.; Hartge, K.H. Particle density. In *Methods of Soil Analysis 1: Physical and Mineralogical Methods*; Klute, A., Ed.; 2nd Ed.; American Society of Agronomy: Madison, 1986; 377–382.
5. American Society for Testing and Materials. *Procedures Testing Soils*; American Society for Testing and Materials: Philadelphia, 1958.
6. Capek, M.; 1933; Cited by DiGleria, J.A.; Klimes-Szmik, A.; Dvoracek, M.; *Bodenphysik und Bodenkolloidik*, 1962; German Edition Jointly by Akademiai Kiado, Budapest, and VEB Gustav Fischer Verlag, Jena.
7. Elder, J.G. Density measurement by the mechanical oscillator. In *Methods in Enzymology Series*; Academic Press: New York, 1979; 12–25.

BIBLIOGRAPHY

1. Gradwell, M.W. The determination of specific gravities of soils as influenced by clay-mineral composition. *NZJ. Sci. Technol.* **1955**, *37B*, 283–289.

Particle Packing

Satish C. Gupta

Department of Soil, Water, and Climate, University of Minnesota, St. Paul, Minnesota, U.S.A.

D.P. Thoma

University of Minnesota, St. Paul, Minnesota, U.S.A.

Abstract

Particle packing refers to arrangement of solids in a given volume. This is important in many fields. This entry discusses the packing characteristics of various mono- and multiple-size particles in a mixture. It also outlines the corrections for particle sphericity, roundness, and roughness.

INTRODUCTION

Packing refers to the arrangement of solids in a given volume. Packing is important in many different fields, including aeronautics, agriculture, ceramics, chemical engineering, chemistry, foundation engineering, electrical engineering, foods, geology, mechanical engineering, medicine, pharmaceuticals, and polymer science.^[1] A few specific applications include packing of solid fuel for rockets and missiles, packing and storage of seeds, strength of powder compacts, packing of fluidized beds, packing of soil and roadway foundations, packing of pharmaceutical pills, and strengths of polymer structure. The underlying premise of packing is to identify the most efficient way of packing solids (whether it be particles, aggregates, powder, solid fuel, fruits, etc.) such that it provides maximum strength, uses minimum space, maximizes the efficiency of chemical reaction, and/or minimizes the shipping costs. In soils, geology, and hydrology, packing is important in understanding and quantifying the pathways of water and chemical transport and the sites of their retention.

INDICES OF PACKING

There are several indices that can be used to characterize particle packing. These include bulk density (ρ_b), void ratio (e), and porosity (f) as follows:

$$\rho_b = \frac{\text{mass of particles}}{\text{bulk volume}} \quad (1)$$

$$e = \frac{\text{volume of voids}}{\text{volume of particles}} \quad (2)$$

$$f = \frac{\text{volume of voids}}{\text{bulk volume}} \quad (3)$$

The above three indices are related to one another through their relationship with particle density (ρ_p).

$$f = \left(1 - \frac{\rho_b}{\rho_p}\right) = \frac{e}{1 + e} \quad (4)$$

$$\rho_p = \frac{\text{mass of particles}}{\text{volume of particles}} \quad (5)$$

In some fields, packing density has also been expressed as fractional density—defined as a fraction of the theoretical density of the material. Another index that characterizes a packing arrangement is the coordination number, the number of neighboring particles touching a given particle.

TYPES OF PACKING

Particle packing can be geometrically systematic (regular) or random (disordered). There are five different ways of systematically packing spherical particles of one size. These are cubic, cubical-tetrahedral, tetragonal-sphenoidal, pyramidal, and tetrahedral.^[2] Over the years, different terminology has been used to describe these packing arrangements. Graton and Fraser^[3] defined these packing arrangements as cubical, orthorhombic, rhombohedral, tetragonal-sphenoidal, and rhombohedral, respectively. White and Walton^[4] defined them as cubical, single stagger, double stagger, pyramidal, and tetrahedral, respectively (Fig. 1).

In cubical packing, spheres are arranged at 90° angles (Fig. 2). Each sphere touches four spheres in the same layer and one sphere in the layer above and one sphere in the layer below. Therefore, the coordination number for the cubical packing is 6 (Table 1). For other packing

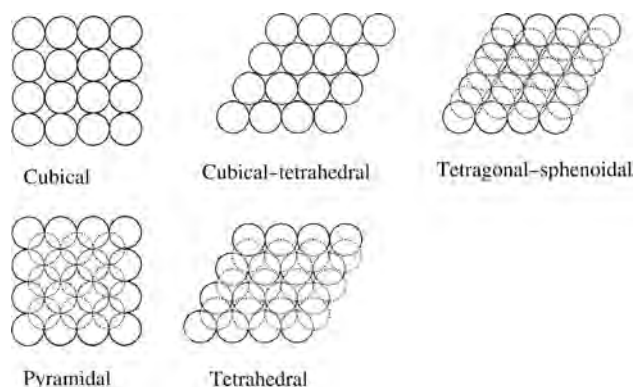


Fig. 1 The full circles represent spheres in one plane while the dotted circles represent the spheres in the layer above.

arrangements, the coordination number varies from 8 to 12 (Table 1).

PACKING DENSITY

The packing density or porosity of a given packing arrangement can be calculated by knowing the geometry of packing. For example, in cubic packing, each sphere (with diameter of $2r$) is surrounded by an imaginary cube (unit prism) with a unit volume equal to $8r^3$. As the volume of the sphere is $(4/3)\pi r^3$, the porosity and void ratio of the cubical packing are as follows:

$$f = \frac{8r^3 - (4/3)\pi r^3}{8r^3} = \left(1 - \frac{\pi}{6}\right) = 0.4764 \quad (6)$$

$$e = \frac{8r^3 - (4/3)\pi r^3}{4/3(\pi)r^3} = \left(\frac{\pi}{7} - 1\right) = 0.9099 \quad (7)$$

Similar calculations of porosity for other types of packing arrangements have been made by White and Walton^[4] and are listed in Table 1.

Porosities listed in Table 1 are independent of sphere radius. Cubical is the loosest and pyramidal and tetrahedral and the densest packing of any spherical particles of a given size. Additional reduction in the earlier

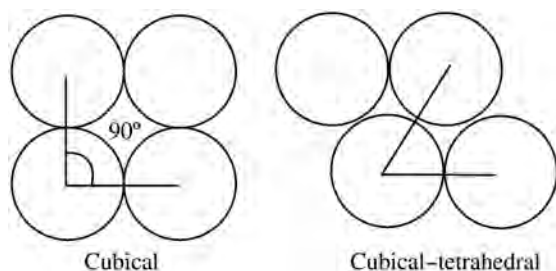


Fig. 2 Angle at which particles are aligned in cubical and cubical-tetrahedral packing.

discussed porosities can occur by filling the irregular void spaces between the primary particles with smaller secondary, tertiary, quaternary, and quinary spheres. White and Walton^[4] showed that the porosity of tetrahedral packing can be reduced to as low as 3.9% (Table 2) through the introduction of smaller particles. Similar reductions in porosity are also possible for other packing arrangements.

MODELS OF PARTICLE PACKING

Over the years, attempts have been made to predict the packing density of a mixture of particles varying in diameter. Westman and Hugill^[5] and later Bodman and Constantin^[6] and Staples^[7] showed that one could approximate the packing density of a mixture of two or more particle fractions using an analytical expression based on the packing density of individual fractions. The assumption underlying this method is that if larger particles are dominant, then the contribution of the finer particles to bulk volume is minimal. However, if the finer particles are dominant, then the bulk volume of the mixture is equal to the bulk volume of finer particles plus the particle volume of larger particles. Extrapolating these two assumptions to various proportions of coarse and finer particles then leads to the identification of an optimum mixture of coarse and fine particles that can result in maximum density.

The analytical model has been described in Fig. 3.^[5] The abscissa refers to the volume fraction of coarse (X_1) and fine (X_2) particles, whereas the ordinate refers to particle volume, void volume, and bulk volume of coarse and fine fractions. Points C and F refer to the bulk volume of coarse and fine fractions when packed individually. If these fractions are stacked one above the other without mixing, then the bulk volume of the mixture is given by the line CF. Lines CE and HE refer to the particle and bulk volumes of the coarse fraction, respectively. Lines FD and GD refer to particle and bulk volumes of the fine fraction, respectively. If the coarse fraction dominates the mixture then the fine particles will be enclosed in the voids of the coarse fraction. The bulk volume describing this relationship is line CE. If on the other hand, the fine fraction is dominant, then the addition of any coarse particles adds to the bulk volume the equivalent of the particle volume of the coarse fraction. The line FH then describes the bulk volume of the mixture. The intersection of lines CE and FH defines the volume fraction of coarse and fine particles with the minimum bulk volume or the maximum packing density.

In Fig. 3, we assumed that the particle density of both fractions was the same. If there is a difference in particle density of various fractions then the ordinate can be expressed in normalized volume by dividing the bulk volume by the particle volume. Equations of the two

Table 1 Various types of systematic packing and the corresponding coordination number, radius of the largest cavity, volume of unit prism, porosity, and void ratio.

Type of packing	Coordination number	Radius of largest cavity	Volume of unit prism	Porosity	Void ratio
Cubical	6	$0.732r^a$	$8r^3$	0.4764	0.9099
Cubical–tetrahedral	8	$0.528r$	$4r^3 \times 3$	0.3955	0.6540
Tetragonal–sphenoidal	10	$0.285r$	$6r^3$	0.3020	0.4324
Pyramidal	12	$0.414r$	$4r^3 \times 2$	0.2595	0.3504
Tetrahedral	12	$0.414r$	$4r^3 \times 2$	0.2595	0.3504

^a r is the radius of the packing particles.

lines (CE and FH), along with the assumption that the sum of the volume fraction of the two components is 1.0, help define the minimum volume or the maximum density.^[5]

Gupta and Larson^[8] presented a computer model that simulated maximum, random, and minimum bulk densities of a soil based on the particle size distribution and the packing density of individual fractions. The packing model is based on the concept that certain particles (donor) will be enclosed in the voids of the packing assemblages of other particles (acceptor) of larger radii. The decision to designate which particle fraction acts as the acceptor and which acts as the donor depends upon whether the maximum density or random density is to be predicted. The model uses the cavity radius of stable packing arrangements (tetragonal or pyramidal–tetrahedral) as a guide in deciding when donor fractions can be fitted into the voids of the acceptor fractions.

Maximum density refers to the conditions when all the voids of the coarse fractions are systematically filled with finer fractions. Random density refers to conditions when particle fractions are randomly selected as acceptor or donor particles. Minimum density refers to conditions when all the fractions are acceptor particles and there is no mixing between the fractions.

Gupta and Larson^[8] tested the predictions of random packing from the packing model against the measured values for 22 mixtures of various size glass beads^[7] and 43 soils and dredged sediment mixtures. In many cases, the predicted bulk densities were close to the

measured values. Spivey et al.^[9] showed that artificially compacted (at 2 MPa) bulk densities of coastal plain soils in the southeastern United States compared well with random bulk densities predicted from the packing model. Reinsh and Grossman^[10] showed that mean bulk densities estimated from the packing model were an acceptable approximation of oven-dry bulk densities after inundation.

Gupta and Larson^[11] also extended the packing model to predict the bulk density of artificially prepared aggregated soils consisting of aggregates that varied in diameter from 0.053 to 50.8 mm. For aggregate size distributions varying in geometric mean diameter from 0.52 to 9.06 mm, the random packing density of aggregated soils was within 0.17 Mg/m^3 of the measured values.

With the availability of high-speed computing, there have been several advances in modeling particle packing. These advances have been summarized in a review article by Davis.^[12]

PARTICLE SHAPE EFFECTS

Concepts of particle packing have generally been developed assuming the particles to be spherical. Next to the particle size distribution, the shape of the particles has a major influence on the packing density of granular materials. Particle shape can be described by parameters such as sphericity and roundness.^[13,14] Sphericity refers to the degree to which particles approach spherical shape,

Table 2 Radius of secondary (B), tertiary (C), quaternary (D), and quinary (E) particles that can fit in the irregular spaces between the primary (A) particles in tetrahedral packing and the corresponding porosities.

	A	B	C	D	E	Filler
Radius of sphere	r	$0.414r$	$0.225r$	$0.177r$	$0.116r$	Very small
Relative number of spheres	1	1	2	8	8	
Volume of a sphere	$4.189r^3$	$0.298r^3$	$0.0476r^3$	$0.0225r^3$	$0.0066r^3$	
Total solid volume of spheres	$4.189r^3$	$4.487r^3$	$4.582r^3$	$4.762r^3$	$4.815r^3$	$5.437r^3$
Porosity, %	25.95	20.7	19.0	15.8	14.9	3.9

Source: From White & Walton.^[4]

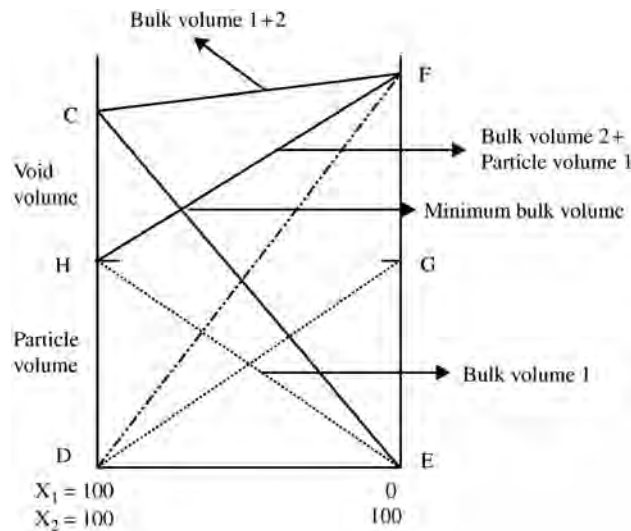


Fig. 3 Relationship between particle volume, void volume, and bulk volume of coarse and fine fractions in the packing model of Westman and Hugill. The intersection of lines CE and HF represents the minimal volume or maximum bulk density of a mixture. **Source:** Adapted from Westman & Hugill.^[5]

whereas roundness describes the angularity of particle corners. Wadell^[15] defined the sphericity index as the ratio of the surface area of a sphere (with the same volume as the test particle) to the surface area of the test particle. Riley^[16] defined the sphericity as the ratio of the diameter of the smallest circumscribing circle to the largest inscribed circle. The closer the particles are to a perfect sphere, the larger the sphericity index. German^[1] calculated the sphericity index of various shapes using Wadell's procedure.^[15,17] The sphericity index

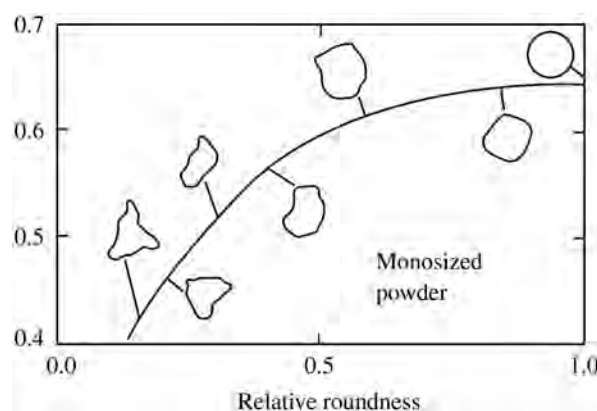


Fig. 4 Effect of particle shape and smoothness on random fractional packing density. A common method of assessing particle roundness is to compare images of individual particles to standard charts showing particles with varying roundness. German showed that packing density increases with an increase in roundness of randomly packed monosized particles (Fig. 5). **Source:** From German,^[1] Krumbein,^[13] and Griffiths.^[14]

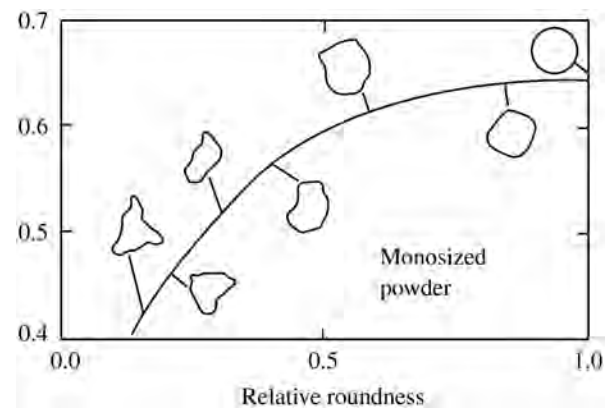


Fig. 5 Variation in fractional density vs. roundness of randomly packed monosized particles.

Source: Adapted from German.^[1]

varied from 0.87 for a cylinder of 1:1 diameter to length, 0.83 for a cylinder of 1:2 diameter to length, 0.47 for a disk of 5:1 diameter to thickness, 0.81 for a cube, and 0.76 for a rectangle of 2:2:1 length, width, and height ratio. German^[1] showed that for monosized particles, the particle packing increases with an increase in the sphericity of the particles (Fig. 4).

Another particle shape parameter affecting packing density is the surface roughness. This parameter refers to inter-particle friction owing to irregularities on the particle surface.^[1] German^[1] showed that the greater the surface roughness, the lower the particle density of monosized particles (Fig. 4).

CONCLUSION

Particle packing refers to arrangement of solids in a given volume. This is important in many fields. This entry discusses the packing characteristics of various mono- and multiple-size particles in a mixture. It also outlines the corrections for particle sphericity, roundness, and roughness.

REFERENCES

1. German, R.M. *Particle Packing Characteristics*; Metal Powder Industries Federation: Princeton, 1989; 443 pp.
2. Gray, W.A. *The Packing of Solid Particles*; Chapman and Hall: London, 1968; 134.
3. Graton, L.C.; Fraser, H.J. Systematic packing of spheres with particular relation to porosity and permeability. *J. Geol.* **1935**, *43*, 785–909.
4. White, H.E.; Walton, S.F. Particle packing and particle shape. *J. Am. Ceram. Soc.* **1937**, *20*, 155–166.
5. Westman, A.E.R.; Hugill, H.R. The packing of particles. *J. Am. Ceram. Soc.* **1930**, *13*, 767–779.

6. Bodman, G.B.; Constantin, G.K. Influence of particle size distribution on soil compaction. *Hilgardia* **1965**, *36*, 567–591.
7. Staples, W.J. The influence of size distribution on the bulk density of uniformly packed glass particles. *Soil Sci. Soc. Am. Proc.* **1975**, *39*, 404–408.
8. Gupta, S.C.; Larson, W.E. A model for predicting packing density of soils using particle size distribution. *Soil Sci. Soc. Am. J.* **1979**, *43*, 758–764.
9. Spivey, L.D.; Busscher, W.J.; Campbell, R.B. The effect of texture on strength of southeastern coastal plain soils. *Soil Till. Res.* **1986**, *6*, 351–363.
10. Reinsh, T.G.; Grossman, R.B. A method to predict bulk density of tilled Ap horizons. *Soil Till. Res.* **1995**, *34*, 95–104.
11. Gupta, S.C.; Larson, W.E. Modeling soil mechanical behavior during tillage. In *Predicting Tillage Effects on Soil Physical Properties and Processes*; Unger, P.W., Van Doren, D.M., Eds.; Special Pub. 44; ASA, SSSA: Madison, 1982; 198.
12. Davis, I.L. Particle pack influence on highly filled material properties. *Curr. Opin. Solid Stat. Mater. Sci.* **1999**, *4*, 505–513.
13. Krumbein, W.C. Measurement and geological significance of shape and roundness of sedimentary particles. *J. Sediment. Petrol.* **1941**, *11*, 64–72.
14. Griffiths, J.C. *Scientific Methods in Analysis of Sediments*; McGraw-Hill: New York, 1967; 508.
15. Wadell, H. Volume, shape and roundness of rock particles. *J. Geol.* **1932**, *40*, 443–451.
16. Riley, N.A. Projection sphericity. *J. Sediment. Petrol.* **1941**, *11*, 94–97.
17. Wadell, H. Sphericity and roundness of rock particles. *J. Geol.* **1933**, *41*, 310–311.

Particle Shape

Brian M. Schafer

Agriculture and Horticulture, University of Queensland, Blue Mountain Heights, Queensland, Australia

Abstract

Particle shape classes and secondary clay mineral content have been used in combination to determine soil texture grades. Texture grades and their arrangement in soil structure largely determine pore volume in soils, which can be related to water and air-filled porosity as well as soil strength which has implications for soil–plant root relationships. Most of the use of particle shape analysis has been confined to soil genesis studies. These studies together with secondary clay mineral formation include the interpretations of initial parent materials, inferred factors, and processes used to describe the nature and genesis of soil particulate mineralogical properties and hence the distribution of contiguous soils in a landscape.

INTRODUCTION

Soil particle shape refers to the morphology of grains of essentially individual or composite mineral grains comprising the fine earth fraction of soil. The term “grain” has been used in geological literature to describe minerals ranging in size from 2 mm to submicroscopic dimensions, while mineral grains larger than 2 mm are classed as phenocrysts.^[1] In soils, particulate material is generally described as particles smaller than 2 mm average diameter being measured as the various ratios of a particle’s long, intermediate, and short axes.^[2] They are categorized into size classes that in turn are used to determine texture grades in soils. Generally, the particles are in tangential contact with each other rather than in continuous contact with their neighbors as found in igneous rocks and collectively determine soil fabric classes by virtue of their packing arrangement.

The use of size and shape analysis in soil science has been extended from sedimentary petrology origins to express the frequency distribution of size and shape ranges of particulate matter.^[3] They form an integral part of both structure and mineral analysis used to describe quantitatively and to interpret the origin and processes of the formation of soil material. Cementing materials such as carbonates and iron oxides are excluded except where they are formed separately from the soil matrix as secondary formations. Thus, most work in soil particle shape has been confined to primary minerals that are residual or transported from other sources. Secondary clay minerals also play a significant role in structure and mineral analysis. However, clay particle shape has not been subjected to the same analytical treatment as the coarser fractions.

Particle shape is related to size in that the size is reflected as volume expressed as true nominal diameter whereby the particles form an open 3-D network. This implies that the

particle volume has a porosity component determined by a packing arrangement of different diameter particles, but invariably leaving a residual pore space which can be filled with cementing agents, fluids, or air.

METHODS OF MEASUREMENT

Shape is based on the concept of sphericity and roundness.^[4–6] Sphericity is a description of the overall form of the particle irrespective of angularity of edges and corners, whereas roundness refers only to sharpness.^[3] Both of these properties have been defined mathematically.^[7,8]

Sphericity and roundness assessments are commonly made visually on individual grains and are tedious and imprecise because of the partly altered nature of minerals that makes it difficult to positively identify their origin in source material. They require the skills of optical mineralogy techniques and an appreciation of sedimentary petrology.

Particle shape analysis is usually performed on samples obtained from particle size analysis. This method results in separation of particles into grade scale classes based on a geometric scale. The broadness of the individual classes in the international grade scale tends to hide the continuum of sizes meaning that the nature of the size distribution cannot be determined from the grade scale classes. Consequently,

Table 1 Shape classes.

Class I	Disc	$b/a > 2/3$	$c/b < 2/3$
Class II	Spheroid	$b/a > 2/3$	$c/b > 2/3$
Class III	Blade	$b/a < 2/3$	$c/b < 2/3$
Class IV	Rod	$b/a < 2/3$	$c/b > 2/3$

Note: a, b, and c are the crystallographic axes.

Source: Adapted from Zingg.^[7]

Table 2 Roundness grades.

Grade term	Class limits			Rho scale ^a
	Pettijohn ^[9]	Powers ^[10]	Folk ^[11]	
Very angular		0.12–0.17	0.00–1.00	
Angular	0–0.15	0.17–0.25	1.00–2.00	
Subangular	0.15–0.25	0.25–0.35	2.00–3.00	
Subrounded	0.25–0.40	0.35–0.49	3.00–4.00	
Rounded	0.40–0.60	0.49–0.70	4.00–5.00	
Well-rounded	0.60–1.00	0.70–1.00	5.00–6.00	

^aBased on the class limits from the study by Powers.^[10]

Source: Adapted from Brewer.^[3]

shape analysis conducted on each size grade scale class also cannot be expressed as a function of the whole, and therefore, measurement is semiquantitative.

A number of grade scales have been proposed for shape analysis. Krumbein^[8] used a sphericity classification proposed by Zingg^[7] (Table 1), and Brewer^[3] suggested the addition of planar, acicular, and acicular-planar shape classes to this classification. Roundness grade scales have been proposed^[9–11] and are listed in Table 2. The scales are arbitrary divisions of a geometric scale of sizes or shapes (sphericity), so that each unit or grade serves as a class interval for analytical purposes. The units are mathematical derivations of particle diameter (millimeters) and phi scale^[3] and alternatively a rho scale.^[11] Combining sphericity and roundness scales has not been attempted, but the estimation of two parameters is used in defining particle shape.

PEDOGENESIS

Sphericity and roundness are the properties of particulate matter that have been used to interpret the abrasion effect of transport processes on specific detritus resulting from predominant physical comminution of source rocks and sedimentary bodies. The more spherical or rounder the particles, the greater the effect of abrasion and hence weathering and transport.

These processes are common in diagenesis and weathering processes associated with sedimentary rocks and sedimentary bodies (alluvium and colluvium) and are integral in the rock cycle of igneous, sedimentary, and metamorphic rocks.

In soils, shape analysis is used to determine uniformity of parent material and origin and mode of soil formation and used in mineral and structure analysis.

Soil-forming processes at or near the earth’s surface can then be inferred to link parent material origin and genesis of soil materials and soil profiles.

CONCLUSION

Particle shape classes and secondary clay mineral content have been used in combination to determine soil texture grades. Texture grades and their arrangement in soil structure largely determine pore volume in soils, which can be related to water and air-filled porosity as well as soil strength which has implications for soil–plant root relationships.

Most of the use of particle shape analysis has been confined to soil genesis studies. These studies together with secondary clay mineral formation include the interpretations of initial parent materials, inferred factors, and processes used to describe the nature and genesis of soil particulate mineralogical properties and hence the distribution of contiguous soils in a landscape.

REFERENCES

1. American Geological Institute. *Dictionary of Geological Terms*, 2nd Ed.; Dolphin Reference Books; Doubleday: New York, 1962; 260.

2. Pettijohn, F.J.; Potter, P.E.; Siever, R. *Sand and Sandstone*; Springer: New York, 1987.

3. Brewer, R. *Fabric and Mineral Analysis of Soils*; Wiley: New York, 1964.

4. Wadell, H. Volume, shape, and roundness of rock particles. *J. Geol.* **1932**, *40*, 443–451.

5. Wadell, H. Sphericity and roundness of rock particles. *J. Geol.* **1933**, *41*, 310–331.

6. Wadell, H. Volume, shape and roundness of quartz particles. *J. Geol.* **1935**, *43*, 250–279.

7. Zingg, Th. Beitrag zur Schotteranalyse. *Schweiz. Mineral. Petrog. Mitt.* **1935**, *15*, 39–140.

8. Krumbein, W.C. Measurement and geological significance of shape and roundness of sedimentary particles. *J. Sediment. Petrol.* **1941**, *11*, 64–72.

9. Pettijohn, F.H. *Sedimentary Rocks*, 2nd Ed.; Harper Brothers: New York, 1957; 718.

10. Powers, M. A new roundness scale for sedimentary particles. *J. Sediment. Petrol.* **1953**, *23*, 117–119.

11. Folk, R.L. Student operated error in determination of roundness, sphericity and grain size. *J. Sediment. Petrol.* **1955**, *25*, 297–301.

Particle Size

Hwat Bing So

School of Agriculture and Food Sciences, University of Queensland, St. Lucia, Queensland, Australia

Harold R. Geering

Department of Agricultural Chemistry, University of Sydney, Sydney, New South Wales, Australia

Abstract

Qualitatively, soil texture is the consistency of a moist soil felt between the fingers and is associated with its quantitative aspect, i.e., the particle size distribution, which is determined in the laboratory. It is affected by the content and type of clay, silt and organic matter contents of the soil, and the ion composition on the clay fraction. The particle size distribution is generally plotted in a soil texture triangle, which delineates textural classes, and the names of these classes are correlated with the field texture descriptions. Texture is a fundamental property that determines the specific surface area of the soil, which in turn affects many of the physical and chemical properties and behavior of the soil, such as the soil water- and nutrient-holding capacities, soil hydraulic conductivity, soil consistency, resistance to cultivation, and hardsetting properties. Soil texture (sand or clay content) is generally one of the first properties used in developing pedotransfer functions for soil water characteristic properties.

INTRODUCTION

Besides color, one of the principal descriptors of a mineral soil (i.e., one with less than 10% organic matter content) is its texture, which is well correlated with the particle size distribution of the fine earth fraction (<2 mm fraction). The distribution of particle size is conveniently partitioned by particle size analysis (PSA) into subfractions of sand, silt, and clay, the proportions of which will have a dominant influence on many of the practical soil properties important to agriculture, the environment, and engineering purposes, such as ease of cultivation, nutrient- and water-holding capacities, water transmission characteristics, suitability for earth dam construction, and susceptibility to erosion.

PARTICLE SIZE DISTRIBUTION

With respect to particle size ranges, the subdivision into sand, silt, and clay fractions varies with country and professional institution.^[1] Examples of four of these particle size classifications are shown in Fig. 1.^[2] In defining soil texture, the first three systems are the most widely used in textural class designation. For describing a soil and/or soil horizon portion of a soil profile, the soil is allocated to a textural class. The textural class descriptors provide an indication of which particle size fraction is dominant, with the exception of a loam where no single size range exerts a

dominant influence. In the field, the textural class is estimated by the handfeel of a moist sample of fine earth fraction, when molded between the thumb and forefingers at just below the “sticky point.” In this moisture condition, the soil just begins to stick to the fingers, and its moisture content is approximately equivalent to field capacity or the amount of water retained at a water potential of -33 kPa.^[3] Because the process of hand molding destroys the structure of the soil, the estimate of the textural class will also be related to the particle size distribution. Initially, the soil is remolded several times to form a uniform moist bolus of about the size of a golf ball. Then, the bolus is deformed into threads^[4] (see Fig. 2) or ribbons of approximately 3-mm thick^[5] (see Fig. 3). The force necessary to deform and shear the bolus and the length of these threads or ribbons increase with clay content. At the same time, if the sample contains a particular size fraction that is readily identifiable by feel to the fingers, this is taken into account in deciding on the soil texture class, e.g., the grittiness of coarse sand or the slippery silky feel of silt. Alternatively, medium and coarse sand can also be seen with a hand lens, and fine sand can be heard by its crunchy grinding sound when remolding and shearing are done close to the ears. Finally, organic matter when present as humus at >10% usually confers cohesion to sandy textures and greasiness to clayey textures,^[6,7] which tend to make sandy and clayey soils feel more loamy in texture.

Taking all of the above description and criteria into account, soil texturing by hand is essentially a consistency

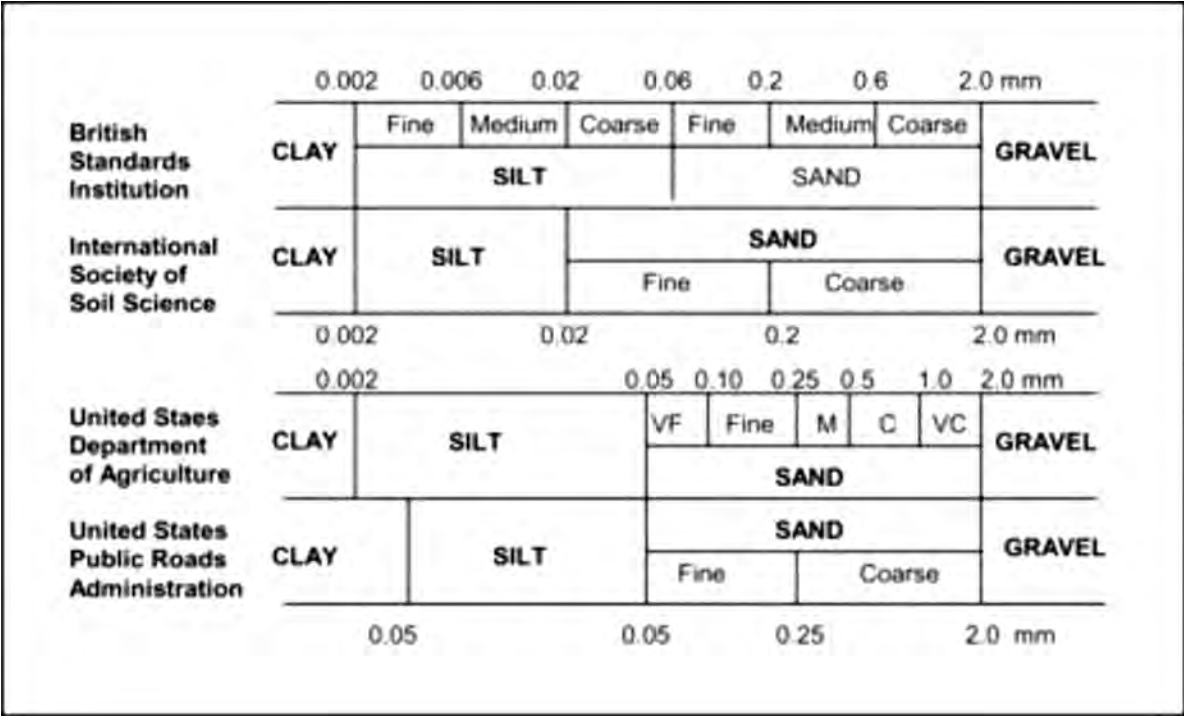


Fig. 1 The designation of particle size ranges from sand, silt, and clay by four professional institutions. The British Standards Institution and Massachusetts Institution of Technology used the same classification.
Source: From Brady.^[2]

test (see the entry *Consistence*, p. 471) at the sticky point.^[8] It then follows that since the amount of work and cohesion increase in the order of the three principal texture classes of sand, loam, and clay, they are often well correlated with the texture classes derived from PSA.^[1,9] Because additional criteria are used in describing the texture of a soil by hand, other than just particle size distribution per se, there are usually more classes derived from hand texturing than can be represented solely in terms of particle size distribution in a soil texture diagram.

PARTICLE SIZE ANALYSIS (PSA)

When accurate data on the soil’s particle size distribution are desired, a PSA can be conducted in the laboratory on a sample (25–50 g of oven-dry soil) of the fine earth fraction. The sample is treated to allow the individual primary soil particles to be freely suspended in water prior to the determination of the proportion of sand, silt, and clay. As primary soil particles are generally found as conglomerates or bonded aggregates, they are treated with one or more of the following sequence of treatments: boiling to soften the bonds, 30% hydrogen peroxide to oxidize organic matter bonds, and 1 M hydrochloric acid to dissolve silica and iron–aluminum bonds. This is then followed by the addition of sodium (Na) cations (as sodium hydroxide–Na hexa–metaphosphate mixture)

and mechanical shaking using a milk shaker or an end-over-end shaker (for 16 hours) to ensure that all clay particles are fully dispersed and remain dispersed. Alternatively, the ultrasonic probe is also routinely used to achieve effective separation of individual particles. Separation of the various size classes of particles is achieved using a combination of sieving, sedimentation, and decantation. Each size fraction is expressed as an oven-dry mass fraction or percentage of the original sample. Details of PSA methodologies can be found in a number of standard soils methodology texts.^[4,5,10] The percentage of sand, silt, and clay fractions are then plotted on a texture triangle that is appropriate for the size fractions selected. Some of the most frequently used texture triangles are shown in Fig. 4. The selection of texture diagram is dependent on the silt size fraction used.

SOIL PROPERTIES AFFECTING FIELD TEXTURE

Soil properties affecting the determination of field texture by hand^[1] are as follows:

- 1. Clay content (particles <0.002 mm in diameter)
- 2. The type of clay mineral (kaolinite, illite, or montmorillonite)
- 3. Silt content (particles between 0.002 and 0.02 mm in diameter)

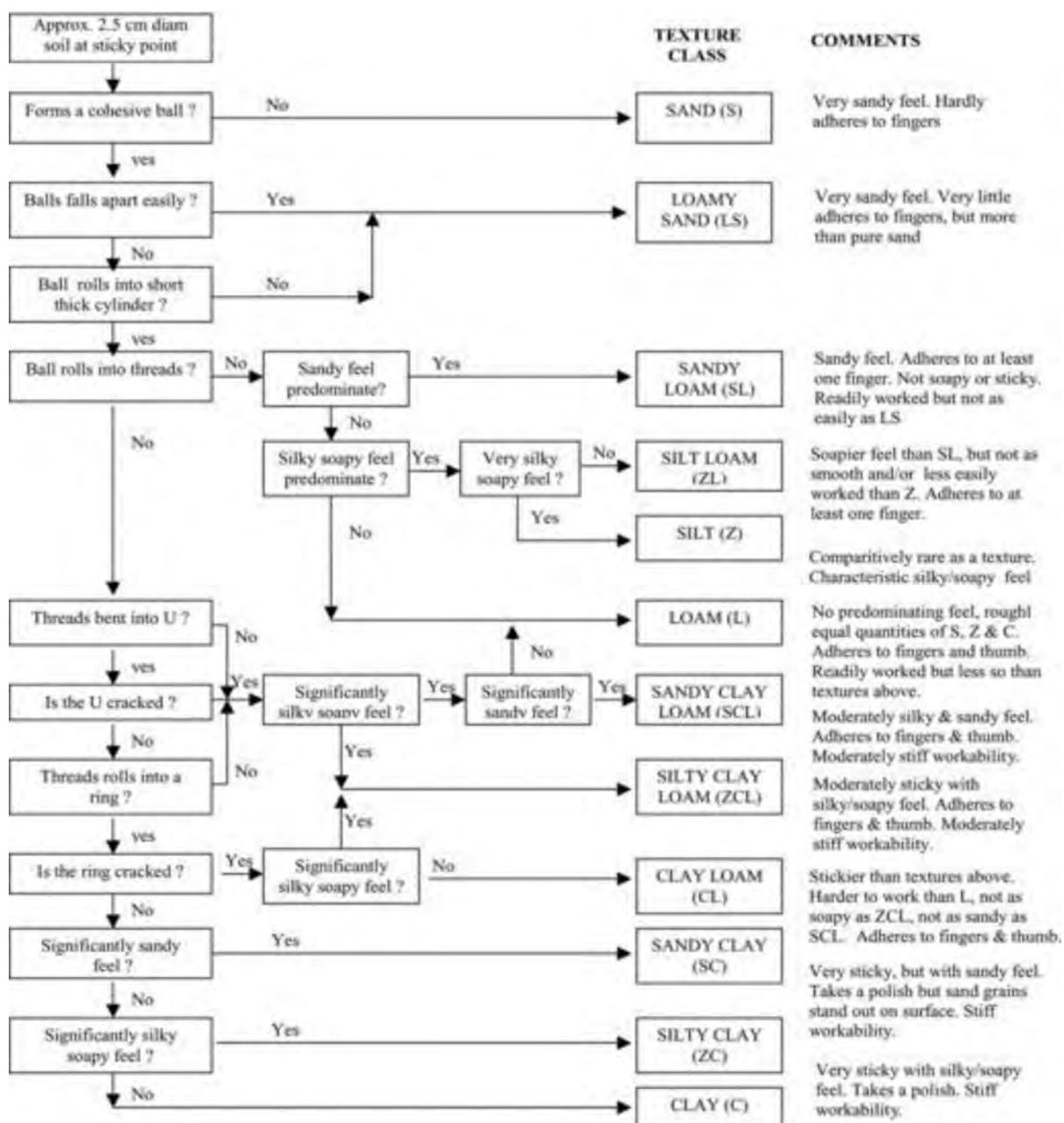


Fig. 2 A guide to field assessment of texture for mineral soils in the United Kingdom by S. Northcliff, University of Reading, and J. R. Landon, Booker Agricultural International.

Source: From Loveland & Whalley.^[4]

4. Organic matter content
5. Oxides
6. Calcium and magnesium carbonate content
7. Cation composition
8. Strong, fine-textured aggregation

The details of their effects are given in the study by Hodgson,^[1] and the dominant effects of clay and silt particles are the reasons for why the final texture designation is based on PSA.

RELEVANCE OF SOIL TEXTURE

The most fundamental property associated with particle size distribution is the specific surface area (SSA) of the soil expressed as surface area per unit mass or $\text{m}^2 \text{g}^{-1}$ (see the entry *Surface Area: Specific*, p. 2255). Smaller particles possess larger SSA; for example, SSA for sand is approximately $0.01\text{--}0.1 \text{ m}^2 \text{g}^{-1}$, for silt, it is approximately $1 \text{ m}^2 \text{g}^{-1}$, and for clay, it may range from 10 to $800 \text{ m}^2 \text{g}^{-1}$. As most soils contain a considerable amount

Field Texture Groups	Ribbon length (mm)	Coherence of the bolus at sticky point	Feel	Other features	Texture Grade (Synonymous with Texture class)	Approx Clay (%)
1 <i>The Sands</i>	Nil	Nil	Sandy	Single sand grains adhere to fingers	1. Sand (S)	Commonly <5
	5	Slight	Sandy	Discolors fingers with an organic stain	2. Loamy Sand (LS)	5–10
	5–15	Slight	Sticky	Sand grains sticks to fingers and discolors with a clay stain	3. Clayey Sand (CS)	5–10
2 <i>The Sandy Loams</i>	15–25	Just Coherent	Very sandy	Medium Sand readily visible	4. Sandy Loam (SL)	10–20
	15–25	Just Coherent	Very sandy	Fine sand may be heard	5. Fine Sandy Loam (FSL)	10–20
	20–25	Strong	Sandy	Medium Sand easily visible	6. Light Sandy Clay Loam (SCL)	15–20
3 <i>The Loams</i>	About 25	Coherent	Spongy & greasy	No obvious sandiness	7. Loam (L)	25
	About 25	Coherent	Slightly Spongy	Fine sand	8. Loam Fine Sandy (Lfsy)	25
	About 25	Coherent	Smooth	Silky; very smooth when manipulated	9. Silt Loam (SiL)	25 (>25% silt)
	25–40	Strong	sandy	Medium Sand in fine matrix	10. Sandy Clay Loam (SCL)	20–30
4 <i>The Clay Loams</i>	40–50	Strong	Smooth	No obvious sand grains	11. Clay Loam (CL)	30–35
	40–50	Coherent	Smooth	Silky feeling	12. Silty Clay Loam (SiCL)	30–35 (>25% silt)
	40–50	Coherent	Smooth & sandy	Fine sand can be felt and heard	13. Fine Sandy Clay Loam (FSCL)	30–35
5 <i>The Light Clays</i>	50–75	Coherent	Plastic	Fine to medium sand	14. Sandy Clay (SC)	35–40
	50–75	Coherent	Plastic	Smooth and silky	15. Silty Clay (SiC)	35–40 (>25% silt)
	50–75	Coherent	Plastic	Smooth with slight resistance to shearing	16. Light Clay (LC)	35–40
	>75	Coherent	Plastic	Smooth with a little resistance to shearing	17. Light Medium Clay (LMC)	40–45
6 <i>The Medium & Heavy Clays</i>	>75	Coherent	Plastic	Fair resistance to shearing	18. Medium Clay (MC)	45–55
	>75	Coherent	Plastic	Firm resistance to shearing	19. Heavy Clay (HC)	>50

Fig. 3 A summary guide for assessment of soil texture for mineral soils in Australia; silt fraction is 0.02–0.002 mm.

Source: From McDonald, Isbell, et al.^[5]

of clay, it is characterized by large SSA. All physical and chemical processes in the soil occur at the interface of the soil solids; hence, the larger the SSA, the greater the cumulative effect will be of these processes, and therefore the clay fraction plays the major role in this regard. Clays are referred to as the active fraction of the soil, whereas sand and silt are referred to as the skeleton of the soil.

Soil Consistency, Ease of Cultivation, and Soil Workability

Soil consistency is the deformation as a response of the soil to an externally applied force (see the entry *Consistence*,

p. 471) and is strongly affected by texture and water content. Moving from a high to low water content on a particular soil, the soil changes from a liquid to a plastic, semisolid, and solid phase. Each phase responds differently to an applied force, and the transitions are characterized by the Atterberg limits; these are the liquid, plastic, and shrinkage limits, which are all strongly dependent on the soil texture, specifically the clay content. The liquid and plastic limits are important engineering parameters of clayey, silty, and loamy soils that have to carry loads such as building foundation. From the agricultural perspective, the plastic phase of the soil should be avoided during cultivation, as the soil in this phase is highly compactible. The semisolid

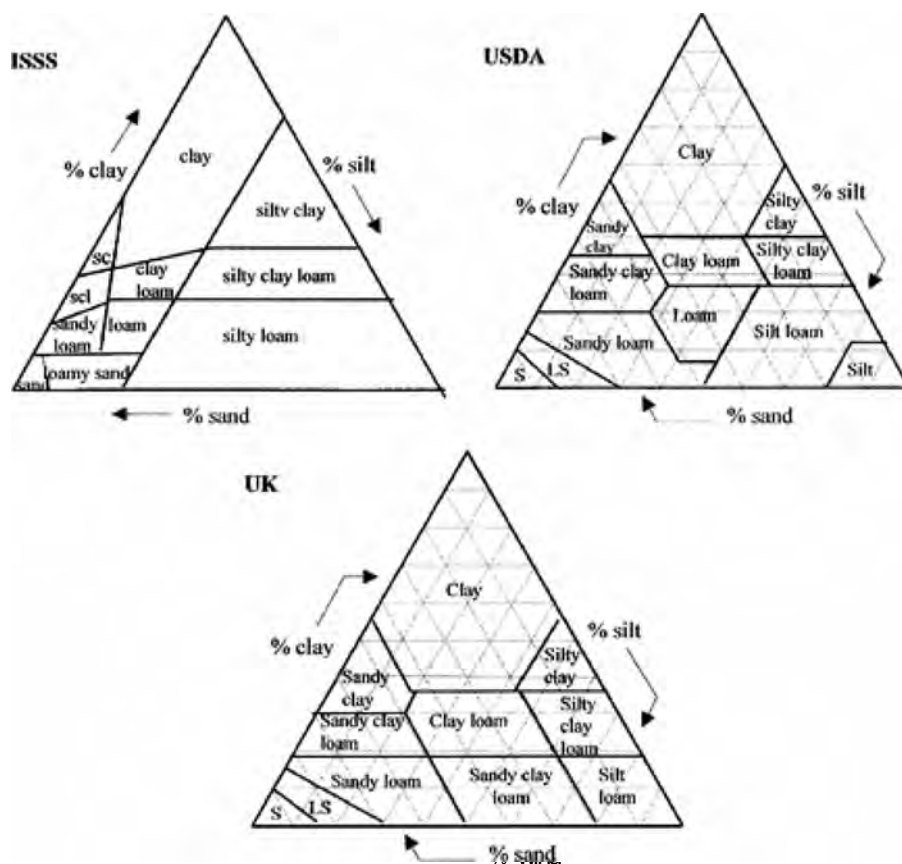


Fig. 4 Three of the most frequently used texture triangles. The International Soil Science Society is based on particle size of sand (2–0.02 mm), silt (0.02–0.002 mm), and clay (<0.002 mm). The U.S. Department of Agriculture (USDA) triangle is based on silt fractions of 0.05–0.002 mm and the UK triangle on silt fractions of 0.06–0.002 mm.

Source: From Klute,^[10] Gee & Bauder,^[11] Rowell,^[12] and Hillel.^[13]

phase is the most suitable for cultivation and within this phase is an optimal water content for cultivation or the soil's workability. The range of water content that defines the semisolid phase and the soil's workability is dependent on the soil's texture and modified by the soil structure.

The ease of cultivation for soils of different textures gives rise to the expressions of "light-textured soils" and "heavy-textured soils" being used for sandy and clayey soils, respectively. Coarse-grained, sandy soils tend to be loose, well aerated, and easy or light to cultivate. Fine-textured soils tend to absorb much water and may become plastic and sticky when wet. Hence, they require much more energy and are heavy to cultivate. Upon drying, they tend to become dense and hard.^[14]

Nutrient- and Water-Holding Capacities and Transmission Properties

The SSA and the magnitude of the electrical charges per unit area of clay and silt determine the cation exchange capacity (CEC) of the soil, which is the amount of cations that a unit mass of soil can hold (see the entry CEC, p. XXXX). The larger the SSA, the larger is the soil's nutrient-holding capacity, e.g., CEC for kaolinite, illite, and montmorillonite clays range from 3 to 15, 10 to 40, and 60 to 120 mmol/kg, respectively. However, fine particle size distribution is generally associated with

lower infiltration and drainage rates. Therefore, sandy soils tend to have higher infiltration rates and generally experienced greater leaching, resulting in acid pH and low base saturation. On the other hand, clayey soils tend to have lower infiltration rates and less leaching, resulting in neutral to alkaline pH and high base saturation.

Particle size distribution and SSA are important determinants of the soil's water-holding capacity. Water is adsorbed on soil surfaces, and water will accumulate in the pore space between the particles as capillary water, which is held in the soil by the force exerted by the water menisci across the pores between particles. This force is inversely proportional to the size of the pore; hence, water is readily removed or drained from the larger pores. At field capacity, which is the water content of an undisturbed soil at a water potential of –10 kPa, sandy soils tend to have lower water contents than clayey soils. Similarly, at the wilting point, which is the water content where water is adsorbed onto soil surfaces by a force of –1.5 MPa, sandy soils have lower water contents than clayey soils. However, the plant-available water capacity (PAWC), or the amount of water held between field capacity and wilting point, is also lower in sandy soils. The approximate relationship between PAWC and texture is shown in Fig. 5.^[15] As water at wilting point is adsorbed on the surface of soil particles, it is predominantly affected by the particle size distribution or texture

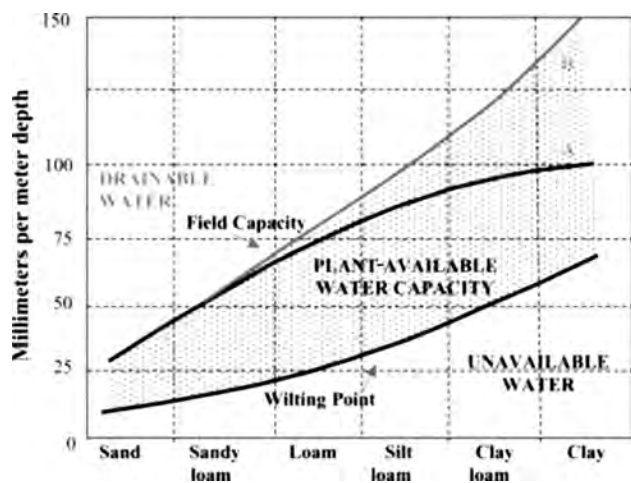


Fig. 5 Approximate water-holding capacities of different textured soils.

Source: From Foth.^[15]

class. However, water at field capacity is accumulated in pore spaces and is capillary water. The latter is strongly affected by soil structure; this effect increases in surface area and clay content. Where the soil is poorly structured or finely structured, field capacity tends to be highly associated with the predominance of fine pores such as the black earths (Curve B in Fig. 5), whereas strongly structured soils have a predominance of large pores, field capacity tends to be lower for the same clay content such as the red Latosols (Curve A in Fig. 5). Particle size distribution will affect the pore size distribution of a soil, and, consequently, it will influence the soil's transmission characteristic, which is termed the soil's hydraulic conductivity K (see the entry *Hydraulic Conductivity*, p. 1133). As measurements of K are resource intensive and expensive, pedotransfer functions are generally used to estimate K and soil texture generally featured as the first factor considered in the development of suitable pedotransfer functions.^[16]

Packing of Particles—Hardsetting Characteristic of Soils

Particle size distribution also affects the potential packing density of the soil or soil aggregates.^[17–19] The presence of sufficient quantities of small particles that will fill the space between the larger particles will result in a massive structureless soil. This generally occurs on soils with 30–35% sand content. One specific soil characteristic associated with the presence of silt and fine sand particles (silty or fine sandy-textured soils) is hardsetting, which is a soil characteristic where the aggregates readily break down upon wetting. The structure will not be reformed, and the soil dries out rapidly into a hard consistency. It softens considerably upon rewetting.^[20] Hardsetting soils are characterized by low water-holding capacities, and they alternate rapidly

between soft and hard consistencies, often resulting in poor soil physical conditions for most plants. Such soils are sometimes referred to locally as “Sunday soils,” which are too wet to cultivate on Saturday and too hard on Monday with Sunday the only timeslot for cultivation.

CONCLUSION

Qualitatively, soil texture is the consistency of a moist soil felt between the fingers and is associated with its quantitative aspect, i.e., the particle size distribution. It is affected by the content and type of clay, silt and organic matter contents of the soil, and the ion composition on the clay fraction. Texture is a fundamental property that determines the SSA of the soil, which in turn affects many of the physical and chemical properties and behavior of the soil, such as the soil water- and nutrient-holding capacities, soil hydraulic conductivity, soil consistency, resistance to cultivation, and hardsetting properties.

REFERENCES

1. Hodgson, J.M. *Soil Sampling and Soil Description*; Oxford University Press: London, 1978.
2. Brady, N.C. *The Nature and Properties of Soils*, 8th Ed.; Macmillan Publishing Co. Inc.: New York, 1974.
3. Cassel, D.K.; Nielson, D.R. Field capacity and available water capacity. In *Methods of Soil Analysis: Part 1—Physical and Mineralogical Methods*; Klute, A., Ed.; American Society of Agronomy: Madison, 1986; 901–926.
4. Loveland, P.J.; Whalley, R.W. Particle size analysis. In *Soil and Environmental Analysis: Physical Methods*; Smith, K.A., Mullins, C.E., Eds.; Marcel Dekker, Inc.: New York, 1991; 281–315.
5. McDonald, R.C.; Isbell, R.F.; Speight, J.G.; Walker, J.; Hopkins, M.S. *Australian Soil and Land Survey Field Handbook*, 2nd Ed.; Inkata Press: Melbourne, 1990; 115–123.
6. Northcote, K.H. *A Factual Key for the Recognition of Australian Soils*; Rellim Technical Publications: Adelaide, 1979.
7. Clark, G.R. *The Study of Soil in the Field*; Clarendon Press: Oxford, 1971; 46 pp.
8. Collis-George, N. Chapter 3: The mechanical and structural properties of soil. In *The Fundamentals of Modern Agriculture*; Blake, C.D., Ed.; Sydney University Press: Sydney, 1967; 61–88.
9. Hodgson, J.M. *Soil Survey Field Handbook*; Soil Survey of England and Wales Technical Monograph No. 5; Harpenden, 1974.
10. Klute, A., Ed. *Methods of Soil Analysis: Part 1—Physical and Mineralogical Methods*; American Society of Agronomy: Madison, 1986.
11. Gee, G.W.; Bauder, J.W. Particle-size analysis. In *Methods of Soil Analysis: Part 1—Physical and Mineralogical Methods*; Klute, A., Ed.; American Society of Agronomy: Madison, 1986; 383–412.

12. Rowell, D.L. *Soil Science: Methods and Applications*; Longman Scientific & Technical: New York, 1994.
13. Soil Survey Staff. *Soil Taxonomy, a Basic System of Soil Classification for Making and Interpreting Soil Surveys*; USDA Soil Conservation Service, R.E. Krieger Pub. Co.: Malabar, 1988.
14. Hillel, D. *Introduction to Soil Physics*; Academic Press, Inc.: New York, 1982.
15. Foth, H.D. *Fundamentals of Soil Science*, 8th Ed.; John Wiley & Sons: New York, 1990.
16. Ballard, V.; Pallacco, J.A.P.; Arp, P.A. Modelling soil hydraulic properties for a wide range of conditions. *Ecol. Model.* **2008**, *219*, 300–316.
17. Bodman, G.B.; Constantin, G.K. Influence of particle size distribution in soil compaction. *Hilgardia* **1965**, *36*, 567–591.
18. Gupta, S.C.; Larson, W.E. A model for predicting packing density of soils using particle-size distribution. *Soil Sci. Soc. Am. J.* **1979**, *43*, 758–764.
19. Smith, G.D.; Coughlan, K.J.; Fox, W.E. The role of texture in soil structure. In *Modifications of Soil Structure*; Emmer-son, W.W., Bond, R.D., Dexter, A.R., Eds.; John Wiley & Sons: New York, 1978; 79–86.
20. So, H.B.; Smith, G.D.; Raine, S.R.; Schafer, B.M.; Loch, R.J. *Sealing, Crusting and Hardsetting Soils: Productivity and Conservation*; Australian Society of Soil Science, Inc.: Brisbane, 1995; 527 pp.

Peatlands

Dale H. Vitt

Department of Plant Biology and Center for Ecology, Southern Illinois University, Carbondale, Illinois, U.S.A.

Paul Short

Canadian Sphagnum Peat Moss Association, St. Albert, Alberta, Canada

Abstract

Peatlands occupy only 3% of the global terrestrial surface, but hold 10% of the world's drinking water, and contain stored in peat about one-third of the earth's soil carbon (about equal to the total carbon dioxide in the atmosphere) and 9–16% of the world's soil nitrogen. Peatlands act as watershed filters and serve as habitat for a number of unique suites of animal species. Fungi are common and abundant, and insects are diverse, but little studied. Plant diversity is low but includes a number of rare and uncommon species. Peat has long been used as a soil amendment, and the harvesting of peat moss is a small, but important, industry where peatlands are abundant. The leading producer of commercial peat moss is Canada, and the industry maintains a view of responsible environmental management that strives to restore peatlands to their before-harvest condition. Additionally, peatlands are managed for enhanced tree growth through drainage and fertilization in some countries, and peat is harvested for electrical generation in a few northern countries. Peatlands contain a large, unstable stockpile of carbon that is sensitive to both climatic and anthropogenic disturbances. These disturbances have the potential to change the long-term peatland carbon sink into a source of carbon to the atmosphere; thus, *Wise Use*, especially conservation, of our peatland resources should be a priority.

INTRODUCTION

Peatlands contain abundant natural resources and are ecosystems that over the long term have net primary production of organic matter exceeding losses from decomposition and export and at millennial timescales have large stores of carbon.^[1] As a result, peatlands develop a deep layer of organic soil, or peat, that is composed of at least 30%, but usually more than 90% organic matter. Deposits of peat accumulate in place, contain a historical record of community change over time, as well as serve as proxies for past environmental and climatic changes^[2] and in some places hold a record of past human activities on the site. These records indicate that nearly all peatlands have initiated and developed over the past 10,000–12,000 years and since the last glaciation. Globally, peatlands represent a major carbon stockpile, storing approximately one-third of the world's soil carbon (450 Pg; petagram = 1×10^{15} g) and about 10% of the global drinkable water, yet cover only 3% of the world's terrestrial area.^[3] As compared to ocean and lake sediments that also contain large reservoirs of stored carbon and are relatively unaffected by atmospheric processes, peatland ecosystems, and the carbon stored in the layers of peat, are extremely vulnerable to climate change and land use changes. Additionally, peatlands are a sink for nitrogen (N), and it has been estimated that boreal peatlands contain about 9–16% of the global pool of N.^[4]

While all peatlands are wetlands, not all wetlands are peatlands, and the need to distinguish the difference is important to the recognition and management of peatland resources. In general, non-peat-forming wetlands are either treed or shrubby (swamps) or without woody vegetation (marshes). Both swamps and marshes have fluctuating water levels and relatively high amounts of nutrient inputs; hence, organic matter produced in these systems decomposes rapidly, and peat formation is limited. Peatlands (sometimes called “mires” in Europe) are usually classified as either fens or bogs (Fig. 1), depending on the source of the water. Fens receive waters from precipitation as well as from the surrounding uplands (i.e., geogenous water source), whereas bogs are somewhat raised above the surrounding area and receive water only from precipitation (i.e., ombrogenous). The ionic quality of these incoming waters greatly influences the structure and functioning of the ecosystem and dictates the amount and type of peat produced. Bogs and poor fens are dominated by the moss genus *Sphagnum* (Fig. 2), and it has been estimated that there is more carbon stored in *Sphagnum* than in any other plant genus, whereas rich fens have ground layers dominated by true mosses.^[5]

Peatlands are estimated to have originally occupied 4,258,000 km², of which about 74% are in non-tropical parts of the world. North America has 46%, Asia 35%, Europe 13%, South America 4%, Africa 1%, and Australia



Fig. 1 Right side: A poor fen in the foreground and treed bog in the background. The fen dominated by sedges and *Sphagnum*. Left side: Continental bog of Western Canada characterized by a dense tree layer of *P. mariana*.

Source: Photos by Kimberli Scott.

and Antarctica less than 1%. Peatlands are concentrated in the boreal and arctic regions of the world where precipitation exceeds potential evapotranspiration. Countries with more than 50,000 km² of peatland are Russia (1,410,000 km²), Canada (1,235,000 km²), the United States (625,000 km²), Indonesia (270,000 km²), Finland (96,000 km²),



Fig. 2 Ground cover of *Sphagnum lenense* with young leaves of *R. chamaemorus* (cloudberry) from the Yukon Territory, Canada.

Source: Photo by Dale Vitt.

Sweden (70,000 km²), and Peru (50,000 km²). Loss of natural peatlands to anthropogenic disturbance is variable and ranges from 52% of the natural peatlands being lost in Europe and 50% in non-tropical Africa, 20% in non-tropical South America, 8% in non-tropical Asia, and 5% in North America. About 16% of non-tropical peatlands have been lost to disturbance, with agriculture accounting for 50%, forestry 30%, peat extraction 10%, and the remainder from urbanization, inundation, and erosion. Restoration efforts, especially in peat harvested bogs of North America, have been developed to return some of these areas to functioning peatlands.^[6]

SENSITIVITY TO DISTURBANCE

The large reservoirs of carbon and N found in peatlands have developed relatively over the past 10,000–12,000 years. Over this period of time, it is estimated that peatlands accumulated carbon at a rate of about 17–20 g m⁻² yr⁻¹ and that globally peat accumulation is about 40–70 Tg C yr⁻¹ (teragram = 1 × 10¹² g). As a natural resource, peatlands are subject to both anthropogenic and natural disturbance impacts. In North America, particularly in the Western Canadian provinces, wildfires account annually for the largest emissions of carbon, estimated at 6300 Gg C yr⁻¹ (gigagram = 1 × 10⁹ g), while anthropogenic impacts that include flooding from hydroelectric construction (–100 Gg C yr⁻¹), peat harvesting (–140 Gg C yr⁻¹), and mining (–50 Gg C yr⁻¹) account for –290 Gg C yr⁻¹. Considering that these Western Canadian peatlands sequester ±8940 Gg C yr⁻¹ and the results of permafrost melting contribute +160 Gg C yr⁻¹, the net estimate of annual carbon sequestration is +1320 Gg C yr⁻¹.^[7]

Globally, the conversion of peatlands to agriculture and forestry is estimated to contribute 100–200 Tg of C yr⁻¹ to the atmosphere, and this combined with global peat extraction of approximately 15 Tg of C yr⁻¹ would account for peatlands at the global scale to be a source rather than a sink for carbon.^[8] However, these estimates are rapidly changing, with changes in forestry practices along with regional differences in anthropogenic activities and the application of appropriate responsible peatland management practices.

NATURAL RESOURCES OF PEATLANDS

Although peatlands have often been considered as wasteland, even cancers on the landscape (illustrated in *The Lord of the Rings*, “Meres of Dead Faces” NW of Mordor), they actually contain a wealth of natural resources and perhaps more importantly carry out valuable ecosystem functions on the landscape. This view of peatlands as wastelands is exemplified by their use as prisons (e.g., the Gulags of the former Soviet Union and the German concentration camps such as Dachau), their use as landfills, and their use as

military training grounds, with the latter allowing some peatlands to be preserved in areas of concentrated human occupation (such as Western Europe).

Given the poor regard for these natural peatland resources, responsible peatland management has been a challenge. The *Wise Use of Mires and Peatlands*, published in 2002,^[7] provided an excellent framework for decision-making when considering the use of peatlands. The *Strategy for Responsible Peatland Management*, published by the International Peat Society in 2011,^[9] has provided a comprehensive management construct for peatlands worldwide. It provides those involved in or responsible for peatland management with strategic objectives and actions for implementation that for the first time defines objectives and actions for the conservation, management, and rehabilitation of peatlands globally, based on the principles of *Wise Use of Mires and Peatlands*. According to the *Strategy for Responsible Peatland Management*, peatland management includes organizing, controlling, regulating, and utilizing peatland and peat for specific purposes, which should be appropriate to the peatland type and use while respecting cultural, environmental, and socioeconomic conditions. The *Strategy for Responsible Peatland Management* is applicable to all types of peatlands under every use, including non-use, and it is directed to everyone responsible for or involved in the management of peatlands, or in the peat supply chain.

Water Resources

In areas where large portions of watersheds are covered by peatlands, these ecosystems provide important source areas for drinking water. For example, in Yorkshire, Britain, 45% of the public supply of water is from watersheds draining peatlands. Haworth Moor of *Wuthering Heights* fame was owned by a company providing drinking water. In the Northern Andes, peatlands are source areas for drinking water for millions of people living in the large cities of Colombia and Peru. In Western Canada's boreal forest, lake chemistry and functions are influenced by watersheds having large cover of peatlands. These peatland-dominated watersheds provide filtered waters to the downstream lakes and in some cases acidifying the lakes.^[10] The nutrient chemistry of these lakes is significantly different from lakes with only upland inputs.^[11] Thus, peatlands can provide a modifying influence on downstream ecosystems. The large dissolved organic carbon (DOC) component of peatlands, especially bogs, colors downstream lakes and streams. With warming and/or drying climatic conditions, coupled with the increased atmospheric deposition and pasturing of animals, bogs can degrade releasing large amounts of DOC to downstream drainages. This is a loss of carbon from the long-term store found in these peatlands. This loss from agricultural uses has been reported as significant in the British Isles.

Soil Resources

Peat has been extracted from peatlands for centuries for a number of uses, foremost of which is an organic substrate in agriculture and gardening. Its uses include biodegradable pots for the cultivation of seedlings, soil amendments for gardens, and growing media for both professional and amateur flower and vegetable growers. It also has been used as a substrate for mushroom growers and for tree nurseries. Among its unique properties for use as a soil amendment and as a growing medium are: 1) excellent water-holding capacity—holding up to 20 times its weight in water and retaining water for exceptionally long periods of time; 2) a low pH that functions to reduce the microbial activity, thus lessening the number of pathogens; and 3) lowering the rate of decomposition, and as a result, the amendment remains in the soil medium longer. Additionally, peat is easy to harvest (Fig. 3), handle, and ship to market. Peat harvesting is a small, but important, industry in Canada ($10.3 \times 10^6 \text{ m}^3$ harvested in 1999) and Germany ($9.5 \times 10^6 \text{ m}^3$ in 1999).

In Canada, the horticultural peat industry is well-established, although it is a relatively recent commercial industrial use of peatlands. Peat for energy was the initial focus of peat use in Canada, but there is no industrial use of peat for energy production. Since the postwar era, horticultural peat has become a major resource industry in rural areas of Canada, particularly in Québec and the Maritime Province of New Brunswick. Western Canada's Prairie Provinces (Alberta, Saskatchewan, and Manitoba) are areas of expansion for the industry and account for approximately 40% of the national production. The Canadian production principally services the horticultural industry of both Canada and the United States.

The Canadian industry, through policy and practice, is committed to the return of postharvest sites to functioning peatland ecosystems. Recognized throughout the world as leaders in the restoration of peatlands is the Industrial Chair for Peatland Management and the Peatland Ecology



Fig. 3 Vacuum harvesting of a drained peat field in Québec, Canada.

Source: Photo by Dale Vitt.

Research Group at the Laval University in Laval, Québec. Beyond the applied level of peatland restoration, the industry continues to actively promote responsible management of peatland resources throughout the provincial and federal agencies accountable for natural resource management. Peatlands (bogs and fens) need to be acknowledged as natural biological resources and managed to ensure that their environmental, social, and economic values are sustained. The key is responsible management that sets aside areas for protection and conservation and puts in place management practices that ensure the retention of ecosystem goods and services following development. The peat-harvesting industry has a commitment to support the continued leading-edge peatland research, application of science-based restoration techniques, and improved responsible peatland management in order to provide sustainable peatland management that recognizes the environmental, social, and economic values of peatland resources.

Historically, peat has also been used in a number of fascinating ways. During the 1800s and early 1900s, peat was used as litter in horse stables, especially for cavalry, railways, and transportation companies that stabled a large numbers of horses. In fact, this use led to the rapid development of peat extraction in Western Europe. It has been estimated that one French army with a 13,500-horse cavalry needed 22,000 tons of peat in one year.^[8] *Sphagnum* peat was used extensively during the Napoleonic and the Franco-Prussian Wars, by the Japanese in the 1904–1905 war with Russia, and in World War I by both sides as a substitute for cotton in surgical dressings. In Britain, “War Work for Women” included collecting and processing *Sphagnum* in the extensive British open bogs. Aboriginal people in North America used *Sphagnum* moss for diapers, and in the 1980s, *Sphagnum* was used in the commercial production of feminine hygiene products. Peat also forms a component of filtering systems for water purification. The cation exchange properties of *Sphagnum* allow the exchange of hydrogen ions for all base cations, including cationic forms of heavy metals. The absorbent properties of peat allow the absorption of large quantities of water, but when dried to less than 8% moisture, peat becomes hydrophobic and absorbs large quantities of petroleum products; as such, it has been used in oil spill cleanup and as floor mats in industries with petroleum spillage. The use of peat for building and insulation was widespread in Europe in the past. In Ireland, canal banks were lined with peat; in Finland, it was used as a liner for roadways; and in Russia and Belarus, it was pressed into sheets and used as insulation in buildings and industry.

Peat is also utilized for energy production. The use of peat for electrical generation is an important national strategic use of peat in Ireland, Finland, and Sweden. Within the Nordic countries of the European Union, the energy generation of peat is one of the most important uses of peat and will almost certainly remain important for countries with large peat reserves and in producing energy for

isolated locations. The use of peat briquettes for burning in household stoves, furnaces, and fireplaces is also common in rural Ireland, but peat is mixed with *Miscanthus* biomass to form more energy-efficient renewable products. Bord na Móna reports that overall about 4 megatons (=4 Tg) of peat are burned for energy generation each year, of which 1 megaton (=1 Tg) is used for local burning in households.

Biotic Resources

Since the early 1800s, and originating in Germany, peat has been used in balneology (baths). It is reported that owing to the biologically active substances in peat, such as humic acids, peat may influence the immune system and is effective against microorganisms. Peat is an important component in the distillation of Scotch whiskey. The special peaty flavor of Scotch whiskey is imparted by slowly drying the “green malt” over a smoldering peat fire.

Fens have long been cut for hay in both Canada and Eurasia. This “slough hay” is used for both fodder and straw for domestic animals. Also, drier peatlands have been long used as pasture. Wild plants are actively collected and eaten by local and indigenous people across the boreal zone. Cranberries (*Vaccinium oxycoccos* and *Vaccinium macrocarpon*) are commercially grown and processed for a number of juice products in Eastern United States and Canada. In Fennoscandia and Québec, Canada, the berries of cloudberry (*Rubus chamaemorus*; Fig. 2) and lingonberry (*Vaccinium vitis-idaea*; Fig. 4) are distilled for liquors, as well as used in preserves and jellies. A large number of medicinal preparations are produced from sundews (*Drosera* spp.). First Nations peoples of Canada used plants from peatlands extensively for both food and



Fig. 4 *V. vitis-idaea* (lingonberry) with red berries, growing on a mat of the reindeer lichen, *C. mitis*.

Source: Photo by Kimberli Scott.



Fig. 5 Flowering branch and characteristic leaves of *L. groenlandicum* (Labrador tea).

Source: Photo by Kimberli Scott.

medicinal preparations; e.g., leaves of *Ledum groenlandicum* (Labrador tea; Fig. 5) were boiled and drank for nausea.^[5]

Drainage of peatlands to improve forest growth has long been practiced in Fennoscandia, Great Britain, and Russia. The estimates are that 15×10^5 ha of northern peatlands have been drained for forestry use. In oceanic areas where peatlands are treeless, often non-indigenous tree species have been planted with subsequent fertilization, whereas in more continental areas that have stunted trees growing directly on the peat, the soils have been enhanced with drainage and fertilization. These practices, common in Europe and Asia, have not been implemented in any large scale in North America; however, black spruce (*Picea mariana*) is harvested from natural peatlands in Eastern Canada.

Biodiversity

Peatlands are among the last large undisturbed ecosystems in the world and serve as habitat for many animals and birds. Although there are no vertebrate species that occur exclusively in peatlands, peatlands do serve as primary habitat for Caribou in North America, especially bogs with an abundance of reindeer lichens (mainly *Cladonia mitis*; see Fig. 3). Both moose and black bear utilize peatlands for food and shelter. In northern areas, peatland specialists include the Arctic shrew, northern bog lemming, and southern bog lemming. Birds also occupy peatlands, but few are exclusive to these areas. Sandhill cranes nest in fens in Western Canada; American bittern, Wilson's snipe, and upland sandpipers also use peatlands as major habitats in Eastern Canada. Beavers, both in Europe and North America, may disturb peatlands, but the lack of suitable substrate and food resources limits their activities to the edges. Surprisingly, little is known about the

apparent rich invertebrate fauna of peatlands, where many undescribed species may occur. In the few studies completed, the invertebrate diversity is extremely high, yet little studied. In comparison, such environmental indicators as testate amoebae have been well studied and have been extensively utilized in studies that reconstruct past environmental conditions of the local area.^[12]

Carbon Resources

Over the long term, peatlands extract large amounts of carbon dioxide (CO₂) from the atmosphere and store it in deposits of peat. Peatlands have stored about the same amount of carbon as there is in the atmosphere. Some believe that carbon extracted from the atmosphere by peatlands during the interglacials reduced atmospheric CO₂ concentrations and resulted in the development of the glacial periods. There is some evidence that global CO₂ concentrations are coupled to peatland initiation and global climate.^[13] Although pristine peatlands have served as a sink for atmospheric CO₂, peatlands are a source for methane [a potent greenhouse gas (GHG)] and also for nitrous oxides. Wetlands, rice paddies, and animal livestock dominate global methane production, and emissions from peatlands are variable, but of less importance. While bogs produce little or no methane, fens may provide a rich source. Methane is a product of anaerobic decomposition and is often the highest in wet habitats with relatively high rates of organic matter turnover (such as rich fens and eutrophic marshes). Nitrous oxide emissions from pristine peatlands are low.

Net GHGs from the horticultural peat harvesting process reveal that the entire life cycle of peat extraction emitted 0.54 Gg of CO₂ in 1999 rising to 0.89 Gg CO₂ in 2000. Seventy-one percent of the emissions were associated with peat decomposition, 15% from land use change, 10% from transportation to market, and 4% from processing.^[14] Canadian peat horticultural emissions from all sources (0.89 Gg) represent 0.03% of all degraded peatlands (3 Pg) worldwide. Emissions are 0.006% of all total global net anthropogenic emissions (15.7 Pg). Within the Canadian context (given that the national total GHG in Canada at 771 Pg CO₂ in 2006), the peat industry represented 0.1% of total GHGs. Environmental Life Cycle Analysis, such as this, has led both the Canadian Sphagnum Peat Moss Association and European Peat and Growing Medium Association to recognize that peat harvesting is a significant source of carbon emissions, and peat harvesting companies are examining their emissions' footprint base and identifying opportunities to reduce their impacts.

PEATLAND RESTORATION

Following peat harvest, peat fields are bare and without vegetation. Decomposing peat produces GHGs, and these

areas provide little in the form of suitable habitat or ecosystem services. Beginning in the late 1990s, however, techniques were developed (mainly in Eastern Canada) to restore a viable living vegetative cover. Using the moss transfer technique developed through the research program at Laval University's Peatland Ecology Group, a *Sphagnum*-dominated plant cover has been reestablished within 3–5 years following restoration procedures, with biodiversity and hydrology approaching preharvest conditions. It is predicted that carbon sequestration should become a net sink within 15–20 years.^[6]

Forestry practices result in a steady decrease in the carbon store due to increased aerobic conditions in the upper peat profile, leading to increased decomposition. Even though greater initial tree growth of drained peatlands may increase carbon storage and exceed the losses from peat decomposition, longer-term cumulative losses from the peat result in an increase in carbon emissions to the atmosphere.^[15]

CONCLUSION

Peatlands occupy about 3% of the terrestrial surface of the world. These ecosystems are characterized by organic soils that contain about one-third of the world's soil carbon—about equal to the CO₂ in the atmosphere. This carbon store has accumulated gradually over the past 10,000–12,000 years and is extremely sensitive to land use changes and climatic change. Peatlands function as habitats for animals and provide valuable ecosystem services such as water storage and filtration. A variety of plant species that serve as food and medicine for local as well as aboriginal people are abundant in peatlands. *Sphagnum* moss has been used in a variety of commercial applications. Peat has been harvested for horticultural purposes and is a valuable soil amendment, and peatlands have been drained for agricultural and forestry practices, releasing CO₂ to the atmosphere, but the restoration of harvested peatlands provides a successful avenue for future sustainable management of this valuable resource.

REFERENCES

1. Yu, Z.; Loisel, J.; Brosseau, D.P.; Beilman, D.W.; Hunt, S.J. Global peatland dynamics since the Last Glacial Maximum. *Geophys. Res. Lett.* **2010**, *37*, 1–5.
2. Kuhry, P.; Nicholson, B.J.; Gignac, L.D.; Vitt, D.H.; Bayley, S.E. Development of *Sphagnum*-dominated peatlands in boreal continental Canada. *Can. J. Bot.* **1993**, *71*, 10–22.
3. Holden, J. Peatland hydrology and carbon release: Why small-scale process matters. *Philos. T.: Math. Phys. Eng. Sci.* **2005**, *363*, 2891–2913.
4. Limpens, J.; Heijmans, M.M.; Berendse, F. The peatland nitrogen cycle. In *Boreal Peatland Ecosystems*; Wieder,

- R.K., Vitt, D.H., Eds.; Springer Verlag: Berlin, 2006; 195–230.
5. Vitt, D.H. Peatlands: Ecosystems dominated by bryophytes. In *Bryophyte Biology*; Shaw, A.J., Goffinet, B., Eds.; Cambridge University Press: Cambridge, 2000; 312–343.
6. Rochefort, L.; Lode, E. Restoration of degraded boreal peatlands. In *Boreal Peatland Ecosystems*; Wieder, R.K., Vitt, D.H., Eds.; Springer Verlag: Berlin, 2006; 381–423.
7. Turetsky, M.R.; Wieder, R.K.; Halsey, L.; Vitt, D. Current disturbance and the diminishing peatland carbon sink, 10.1029/2001GLO14000. *Geophys. Res. Lett.* **2002**, *29* (11), 21–1–21–4.
8. Joosten, H.D.; Clarke, D. *Wise-use of Mires and Peatlands*; International Mire Conservation Group and International Peat Society: Helsinki, 2002.
9. Clarke, D.; Rieley, J., Eds. *Strategy for Responsible Peatland Management*; International Peat Society: Jyväskylä, 2011.
10. Halsey, L.; Vitt, D.H.; Trew, D. Influence of peatlands on the acidity of lakes in northeastern Alberta, Canada. *Water Air Soil Poll.* **1997**, *96*, 17–38.
11. Prepas, E.E.; Planas, D.; Gibson, J.J.; Vitt, D.H.; Prowse, T.D.; Dinsmore, W.P.; Halsey, L.A.; McEachern, P.M.; Paquet, S.; Scrimgeour, G.J.; Tonn, W.M.; Paszkowski, C.A.; Wolfstein, K. Landscape variables influencing nutrients and phytoplankton communities in Boreal Plain lakes of northern Alberta: A comparison of wetland- and upland-dominated catchments. *Can. J. Fish. Aquat. Sci.* **2001**, *58*, 1286–1299.
12. Charman, D. *Peatlands and Environmental Change*; John Wiley & Sons Ltd.: Chichester, 2002.
13. Yu, Z.; Campbell, I.D.; Campbell, C.; Vitt, D.H.; Bond, G.C.; Apps, M.C. Carbon sequestration in western Canadian peat highly sensitive to Holocene wet-dry climate cycles at millennial timescales. *Holocene* **2003**, *13*, 801–808.
14. Cleary, J.; Roulet, N.T.; Moore, T.R. Greenhouse gas emissions from Canadian peat extraction, 1990–2000: A life-cycle analysis. *Ambio* **2005**, *34*, 456–461.
15. Laine, J.; Laiho, R.; Minkinen, K.; Vasander, H. Forestry and boreal peatlands. In *Boreal Peatland Ecosystems*; Wieder, R.K., Vitt, D.H., Eds.; Springer Verlag: Berlin, 2006; 331–357.

BIBLIOGRAPHY

1. Bauerochse, A.; Haßmann, H., Eds. *Peatlands Archaeological Sites – Archives of Nature – Nature Conservation – Wise Use*; Verlag Marie Leidorf: Rahden, 2003.
2. Crum, H. A focus on peatlands and peat mosses. In *Mires—Swamp, Bog, Fen and Moor. General Studies; 4B – Regional Studies, 1983*; Gore, A.J.P., Ed.; University of Michigan Press: Ann Arbor & Elsevier Publishing Co.: Amsterdam, 1992.
3. Rydin, H.; Jeglum, J. The Biology of Peatlands. In *Boreal Peatland Ecosystems 2006*; Wieder, R.K., Vitt, D.H., Eds.; Oxford University Press: Oxford & Springer Verlag: Berlin, 2006.

Pedological Modeling

Ronald G. Amundson

College of Natural Resources, University of California—Berkeley, Berkeley, California, U.S.A.

Abstract

While models are powerful means of testing hypotheses and synthesizing contemporary knowledge in a concise way, models of all types are simplified, incomplete, mathematical descriptions of the “real” world. Increasingly, it is emphasized that models can never be fully verified. Experience shows that multiple models (or versions of the same model) may faithfully mimic empirical observations of interest, a dilemma illustrating a practical verification problem. In addition, mathematical models may be internally correct, but they may poorly represent the phenomena they intend to describe because of incomplete knowledge of the system and, as a result, incorrect assumptions. The truly unique role for pedologists, in addition to applying mathematics on their own, is to provide the unique conceptual foundation peculiar to soils—one informed by extensive field observations guided by ideas generated during previous modeling attempts—that will make pedological models relevant to scientists and society.

INTRODUCTION

“The fascinating impressiveness of rigorous mathematical analysis, with its atmosphere of precision and elegance, should not blind us to the defects of the premise that condition the whole process” by T. C. Chamberlin,^[1] commenting on Lord Kelvin’s (ultimately incorrect) calculation of the age of the Earth.

In pedology, and in other sciences, “models” are increasingly used as tools for understanding natural phenomena. But what is a model, how are models used in pedology, and how are models developed and modified?

OVERVIEW

To begin, the term “model” has been used interchangeably in pedology with other concepts, sometimes leading to confusion or miscommunication. In simple terms, a “model” has been described by some as “a form of highly complex scientific hypothesis,”^[2] i.e., “a simplified and idealized description or conception of a particular system, situation, or process (often in mathematical terms) that is put forward as a basis for calculations, predictions, or further investigation.”^[3] As briefly outlined below, the mathematical approaches to describe a phenomenon can be varied, but all must rest on a solid understanding of the soil, and the factors and processes, which affect it. This empirical knowledge in turn constrains the “assumptions,” which underlie any mathematical model development. A model based on an incorrect, or poorly developed, understanding of a soil and the

processes that affect it will likely be unable to describe the processes of interest, but that inability may in turn inspire the modeler to better understand the soil. Therefore, modeling can help refocus attention to fieldwork and to the type of data to be collected. A specific example of how assumptions affect the development of a model is given later in this entry.

The first step in modeling soils—or anything—is to define the object of interest. In applying models to pedology, it should be recognized that soil is, in reality, a continuum of objects distributed across the earth’s surface—both in space and in time. The exact lateral boundary between one “soil” and another, or the vertical boundary between soil and non-soil, is arguably impossible to determine. Jenny^[4] first applied principles derived from the physical sciences to the conceptualization and modeling of soils. Jenny’s approach was to divide the continuum of soils on the earth’s surface into “systems,” which are arbitrarily defined, discrete, 3-D segments of the landscape that are amenable to mass or energy budgeting. The volume of these systems (both the chosen area and depth) is arbitrary, but it sets the stage for the mathematical formulations that are chosen to represent or describe it. A second important aspect of soils is the vast amount of time, and the array of unknown processes, that may have affected any soil system. This complex history in turn forces pedologists to develop tools, concepts, and modes of enquiry not always confronted by their experimental colleagues. Finally, soil formation is the result of an incompletely understood array of processes, and pedological models invariably are attempts to mathematically capture one, or at the most, a very restricted subset of the full suite of biogeochemical processes that affect a soil system.

In the pedological literature, more attention has possibly been devoted to classifying and discussing models^[5] than actually developing them. Unfortunately, much discussion has focused on the “pros and cons” of the *Factors of Soil Formation: A System of Quantitative Pedology*^[4] in the realm of pedological models. Briefly, the factorial “model” discussed in a book-length treatise by Jenny^[4] can be symbolically represented as follows:

$$s = f(\text{cl}, \text{o}, \text{r}, \text{p}, \text{t}, \dots) \quad (1)$$

where s is the soil properties; cl , climate; o , biota; r , topography; p , parent material; and t , time.^[4,6] The general truthfulness of this statement is almost beyond dispute (virtually every pedologist would ultimately have to agree that soil forms in response to variations in these factors, and that soil properties can also be numerically correlated with variations in these factors). Indeed, soil is “defined” in terms of this statement:

Soil is the “collection of natural bodies occupying portions of the earth’s surface that support plants and that have properties due to the integrated effect of climate and living matter, acting upon parent material, as conditioned by relief, over periods of time.”^[7]

Based on this definition, it would be more correct to define the state factor “model,” or “theory” as it has been alternately called, as a pedological “law” given its universal truthfulness and common definitions of scientific laws.^[8] At the very least, it is a fundamental underlying theory of pedology in the sense of Kuhn.^[9] This definition would also allow the discussion of models in pedology to move beyond this fundamental truism (and a useful quantitative and mathematical tool in its own right) to the development of mathematical models, as they are commonly considered and used in geochemistry, geophysics, and related fields. The remainder of this entry deals with these more practical modeling issues.

MODELING PROCESSES

Typical approaches to mathematically modeling soil processes involve the development of a mass or energy balance model. The mathematics used will ultimately hinge upon one’s understanding of the soil properties, the processes that presumably control them, and how these processes may vary over time. Some of the simplest models may be analytical models with time-independent variables. Alternative modeling approaches may involve the abandonment of time invariant parameters and, ultimately, the incorporation of relatively random changes in the rate of the process and factors that affect it.

As an example, it began with possibly the first true mathematical model of a pedogenic process for this entry—a time-dependent, analytical, mass-balance model of O

horizon formation in forest soils developed by Jenny, Gessel, and Bingham.^[10] Jenny, Gessel, and Bingham^[10] defined the system of study, discussed the processes that affect it, and, for the simplest cases they considered (tropical forests with nearly constant litter inputs with time), described the change in O horizon mass (F , in mass per area) with time as follows:

$$dF = A dt - k(F + A) dt \quad (2)$$

where A is litter inputs (mass/area/time), and k , decomposition constant (1/time; Jenny, Gessel, and Bingham^[10] give a fuller discussion of calculation and definition of k). Upon integration, the solution to Eq. 2 provided by Jenny was as follows:

$$F = \frac{A(1 - k)}{k} (1 - e^{-kt}) \quad (3)$$

This model, as well as the permutations of it, has served as the foundation for decades of research and modeling of the soil organic carbon budget. The model has been extended for use in stable isotopic studies of soil organic matter^[11] and in modeling turnover times of soil organic carbon.^[12] However, the analytical model, as developed by Jenny, assumes constant inputs with time (a restriction that he noted does not occur in all situations) and constant decomposition rates with time. It also assumes that all organic matter is homogeneous (and by implication, has the same decomposition rates).

Work since the early 1990s in particular has revealed that soil organic matter (and even litter layers) can best be viewed as multiple pools of soil organic matter, each with their own characteristic input and decomposition rates. In simplest terms, multiple pool models of soil organic matter can be expressed as follows:

$$\frac{dC}{dt} = \sum_{i=1}^n (I_i - k_i C_i) \quad (4)$$

where C is the total mass of soil carbon (mass/volume); I_i , inputs of pool i (mass/volume/time); k_i , decomposition constant of pool i (1/time); and C_i , mass of soil carbon in pool i (mass/volume).^[11] Even these multiple pool models do not capture other aspects of soil formation, the variation in soil carbon with depth, and the likelihood that the process may have varied unpredictably over time. To address the first issue, models may include a downward transport term and depth-dependent inputs. For a single pool of organic matter, a basic depth-dependent model is as follows:

$$dC/dt = I - v \frac{\partial C}{\partial Z} - kC \quad (5)$$

where v is the advection coefficient (distance/time).^[11] Even these models can be made more complex, e.g., to include the process of diffusion.^[13]

Analytical solutions to all the aforementioned models require time invariant parameters. The inclusion of time-dependent parameters may be accomplished through numerical means. Yet, soils form over thousands to millions of years, with numerous unknown perturbations to the system that elude the fundamental framework of these simple models.^[14] It is these real-world complications imposed by nature that weaken the utility of relatively simple, time invariant, deterministic models. Phillips^[14] began the discussion of applying non-linear dynamical system theory to soil modeling, with the goal of explicitly incorporating the role that random differences in initial conditions and historical contingencies have on the processes of soil formation. Undoubtedly, work of this nature is one of the future challenges in the field of pedological modeling.

CONCLUSION

While models are powerful means of testing hypotheses and synthesizing contemporary knowledge in a concise way, the previous discussion serves to illustrate that models of all types are simplified, incomplete, mathematical descriptions of the “real” world. Increasingly, it is emphasized that models can never be fully verified (confirmed as the establishment of truth).^[2] Experience shows that multiple models (or versions of the same model) may faithfully mimic empirical observations of interest, a dilemma illustrating a practical verification problem. In addition, mathematical models may be internally correct, but they may poorly represent the phenomena they intend to describe because of incomplete knowledge of the system and, as a result, incorrect assumptions. Knowledge of the soil is essential in modeling, for as Baker^[15] has noted, “mathematics (is) the science that draws necessary conclusions without regard to facts.”

Given the ultimate simplicity and abstractness of models, and the ultimate complexity of nature in general, and soils in particular, the question “why model in the first place?” may be asked.^[2] Pedological processes are first-order controls on global atmospheric^[16] and aquatic chemistry, and even relatively simple analytical models of soil processes have thus far proven useful to link these global reservoirs. The truly unique role for pedologists, in addition to applying mathematics on their own, is to provide the unique conceptual foundation peculiar to soils—one informed by extensive field observations guided by ideas

generated during previous modeling attempts—that will make pedological models relevant to scientists and society.

REFERENCES

1. Chamberlin, T.C. *Lord Kelvin's Address on the Age of the Earth as an Abode Fitted for Life*; Smithsonian Institution Annual Report; American Association for the Advancement of Science: Washington, 1899; 223–246.
2. Oreskes, N.; Shrader-Frechette, K.; Belitz, K. Verification, validation, and confirmation of numerical models in the earth sciences. *Science* **1994**, *263*, 641–646.
3. Simpson, J.A.; Weiner, E.S.C. *The Oxford English Dictionary*, 2nd Ed.; Clarendon Press: Oxford, 1989.
4. Jenny, H. *Factors of Soil Formation: A System of Quantitative Pedology*; McGraw Hill Book Co.: New York, 1941; 281 pp.
5. Hoosbeek, M.R.; Amundson, R.; Bryant, R.B. Pedological modeling. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 1999; E77–E116.
6. Amundson, R.; Jenny, H. On a state factor model of ecosystems. *Bioscience* **1997**, *47*, 536–543.
7. Soil Survey Staff. *Soil Survey Manual*; U.S. Dep. Agric. Handbook No. 18; U.S. Government Printing Office: Washington, 1951; 503 pp.
8. Morris, C. *Academic Press Dictionary of Science and Technology*; Academic Press: New York, 1992.
9. Kuhn, T.S. *The Structure of Scientific Revolutions*, 2nd Ed.; University of Chicago Press: Chicago, 1970; 210 pp.
10. Jenny, H.; Gessel, S.P.; Bingham, F.T. Comparative decomposition rates of organic matter in temperate and tropical regions. *Soil Sci.* **1949**, *68*, 419–432.
11. Amundson, R.; Baisden, W.T. Stable isotope tracers and models in soil organic matter studies. In *Methods in Ecosystem Science*; Sala, O., Mooney, H., Howarth, B., Jackson, R.B., Eds.; Springer Verlag: New York, 2000; 117–137.
12. Trumbore, S.E. Comparison of soil carbon dynamics in tropical and temperate soil using radiocarbon measurements. *Global Biogeochem. Cy.* **1993**, *9*, 515–528.
13. Elzein, A.; Balesdent, J. Mechanistic simulation of vertical distribution of carbon concentrations and residence times in soils. *Soil Sci. Soc. Am. J.* **1995**, *59*, 1328–1335.
14. Phillips, J.D. On the relations between complex systems and the factorial model of soil formation (with discussion). *Geoderma* **1998**, *86*, 1–42.
15. Baker, V.R. Geosemiosis. *Geol. Soc. Am. Bull.* **1999**, *111*, 633–645.
16. Amundson, R.; Stern, L.; Baisden, T.; Wang, Y. The isotopic composition of soil and soil-respired CO₂. *Geoderma* **1998**, *82*, 83–114.

Pedometrics

Alex B. McBratney

Budiman Minasny

Department of Environmental Sciences, University of Sydney, Sydney,
New South Wales, Australia

Abstract

Pedometrics is defined as the use of quantitative methods for the study of soil distribution, genesis, and sustainability. Pedometrics addresses soil science problems from the perspective of emerging approaches such as wavelet analysis, fuzzy set theory, etc. It also deals with utilizing modern analytical tools to quantify soil properties rapidly over space and time. Key areas of research include numerical soil classification, fuzzy sets, geostatistics, sampling, soil inference systems, and modeling pedogenesis.

INTRODUCTION

The term *pedometrics* was coined by McBratney in the late 1980s and is defined as the use of quantitative methods for the study of soil distribution, genesis, and sustainability. Pedometrics is a neologism derived from the Greek roots *pedos*, soil, and *metron*, measurement. Pedometrics is used analogously with other words such as *biometrics*, *psychometrics*, *econometrics*, *chemometrics*, and the oldest of all, *geometrics*.^[1] Pedometrics uses mathematical and statistical methods to deal with the variation in soil properties and processes over space and time. Pedometrics is also sometimes referred to as “soil science under uncertainty,” which addresses soil-related problems when there is uncertainty due to deterministic or stochastic variation, vagueness, and lack of knowledge of soil properties and processes.

A BRIEF HISTORY

Although the term *pedometrics* was coined in the 1980s and formally recognized as a different branch of soil science, pedometrics is not a new subject. Mathematical and statistical methods have been applied to soil studies since the late 19th century and early 20th century with applications in agronomy, soil survey, and soil physics.

For several decades of the 20th century, pedometrics (although not formalized and defined) was a tool for designing experiments and surveys and in advisory work. In the 1960s, pedometrics research was concerned with the problem of soil classification and applied the methods of numerical taxonomy. In the late 1970s, pedometricians began to treat soil properties as spatially correlated random processes and to utilize geostatistics for analysis and prediction. In the 1990s, pedometricians started to experiment with contemporary mathematical and statistical techniques

such as fuzzy sets, fractals, wavelets, and data-mining techniques for the description and prediction of soil properties. The application is largely focused on digital soil mapping to efficiently produce maps of soil properties. Pedometrics has also begun to attempt to elucidate pedogenesis by quantifying relations between individual soil properties and controlling factors.

Webster^[1] reviewed many early pedometric studies. The textbook by Webster and Oliver^[2] provided a concise, yet comprehensive introduction to geostatistics and its application in soil science. Grunwald^[3] presented and discussed pedometric techniques applied to soil-landscape modeling.

A bibliometric study of the composition of papers of the soil science journal *Geoderma*, from its inception in 1967 until 2001 (Fig. 1),^[4] showed that papers on pedometrics

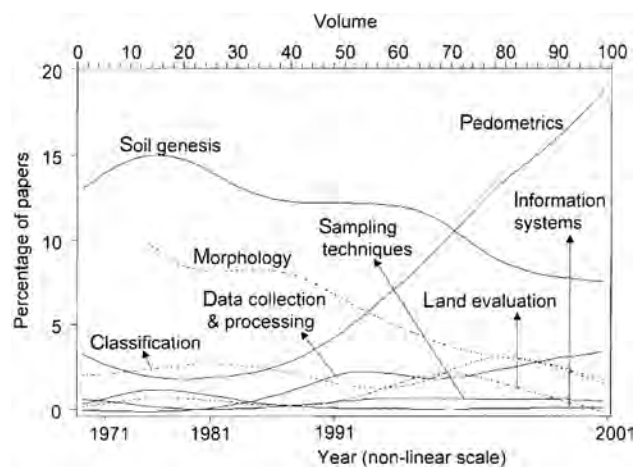


Fig. 1 Trends in pedological subdisciplines and subjects reported in *Geoderma* between 1967 and 2001. The ordinate is the percentage of papers.

Source: From Hartemink, McBratney, et al.^[4]

have risen from less than 3% in 1967 to around 18% of all papers in 2000. It is the fastest growing research area in pedology.

KEY AREAS OF RESEARCH

Pedometrics addresses soil-related questions from a quantitative point of view. The need for the quantitative approach arises from a general demand for quantitative soil information in order to improve economic production and environmental management. There are several key areas of research in pedometrics, which attempt to understand the pattern of soil distribution in character space, soil spatial and temporal variation, the utility and quality of soil, and, ultimately, the genesis of soil.

Numerical and Continuous Classification

Pedometrics attempts to resolve some of the polemics of soil classification by a search for insight into the structure of soil character space. Pedometrics approaches soil classification problems by creating soil classes based on measurable soil information with objective statistical criteria. This began in the late 1950s, with the invention of the digital computer, which allowed the calculation of taxonomic distances between soil profiles. Similarities were calculated between soil profiles, represented as similarity and distance matrices, which allowed a representation of the multivariate data in a few dimensions. This is called ordination, by which a view of the soil property universe was first revealed.^[5,6] Numerical classifications can be created both hierarchically and non-hierarchically using various clustering algorithms.^[7]

Conventional methods of soil classification are based on the logical model of hierarchically arranged, mutually exclusive classes. Conventional methods imply that soil classes are discrete and discontinuous with sharp boundaries and can be represented by a central concept idea known as a typical profile. The intergrading nature of the soil population was recognized in a formal way using the concept of fuzzy sets. This allowed the creation of non-hierarchical continuous classes based on the fuzzy k-means algorithm.^[7] Each class is defined by a centroid or a list of average properties, and the membership of a soil individual to the classes is defined by the relative taxonomic distance between the soil individual and the centroid for each class. In addition to intergrades, extragrades were also recognized. This allowed for more continuous soil mapping, allowing a more natural approach when using geostatistical methods.

Fuzzy Sets

Soil phenomena are sometimes imprecisely defined, while soil scientists like to allocate soil attributes to discrete

classes. The fuzzy set theory is a means of dealing with inexact concepts. In a conventional set theory, objects are allowed to belong only to one set, whereas in fuzzy set theory, they may belong totally, partly, or not at all to a set. The fuzzy set theory is a generalization of the abstract set theory and stems from inexactness and uncertainty, as a result of insufficient data or partly defined classes. The main feature of fuzzy sets is the grouping of individuals into classes where boundaries are not, or cannot be, sharply defined. Fuzzy logic models soil data as memberships to classes, which can have gradual transition from one class to the others, as opposed to sharp boundaries in classical set theory.^[8] A review of the development and application of the fuzzy theory in soil science is given by McBratney and Odeh.^[9]

Geostatistics

Soil attributes vary spatially and temporally in a sometimes continuous, sometimes discrete, sometimes haphazard, fashion. Using the existing and conventional technologies, we can only measure most attributes of the soil at a finite number of places and times on relatively small volumes, and therefore, statements concerning the soil at other places or times involve estimation, prediction and, an inevitable uncertainty. One of the tasks of pedometrics is to quantify this variation so that it can be known and managed accordingly. Pedometrics seeks insight into such spatiotemporal patterns using geostatistical and other spatial and temporal description and prediction tools.

The basics of geostatistics is to be able to measure how soil varies in a field and predict its values at places where it is not measured. The underlying principle is treating soil attributes as random variables, assuming stationarity (some statistics of the random process are same everywhere in a field). The assumption is that the soil properties in a random field are spatially auto-correlated and can be modeled using a variogram. Using information on the variogram, soil properties can be predicted over the field using a technique called kriging (Fig. 2). There are variants of kriging to deal with non-normal distributions and changes in the mean and variance (non-stationarity) such as indicator kriging, universal kriging, and regression kriging. The detailed explanations of the theory and applications in soil science are given by Webster and Oliver^[2] and Goovaerts.^[10]

The more rigorous statistically based approaches for spatial prediction are the residual maximum likelihood approach and best linear unbiased prediction.^[11] The geostatistical model of spatial variation is a special case of the linear mixed model where the data are modeled as the additive combination of fixed effects (e.g., the unknown mean and coefficients of a trend model), random effects (the spatially dependent random variation), and independent random error (nugget variation in geostatistics).

The Bayesian maximum entropy approach was developed for spatial and temporal prediction as an alternative to

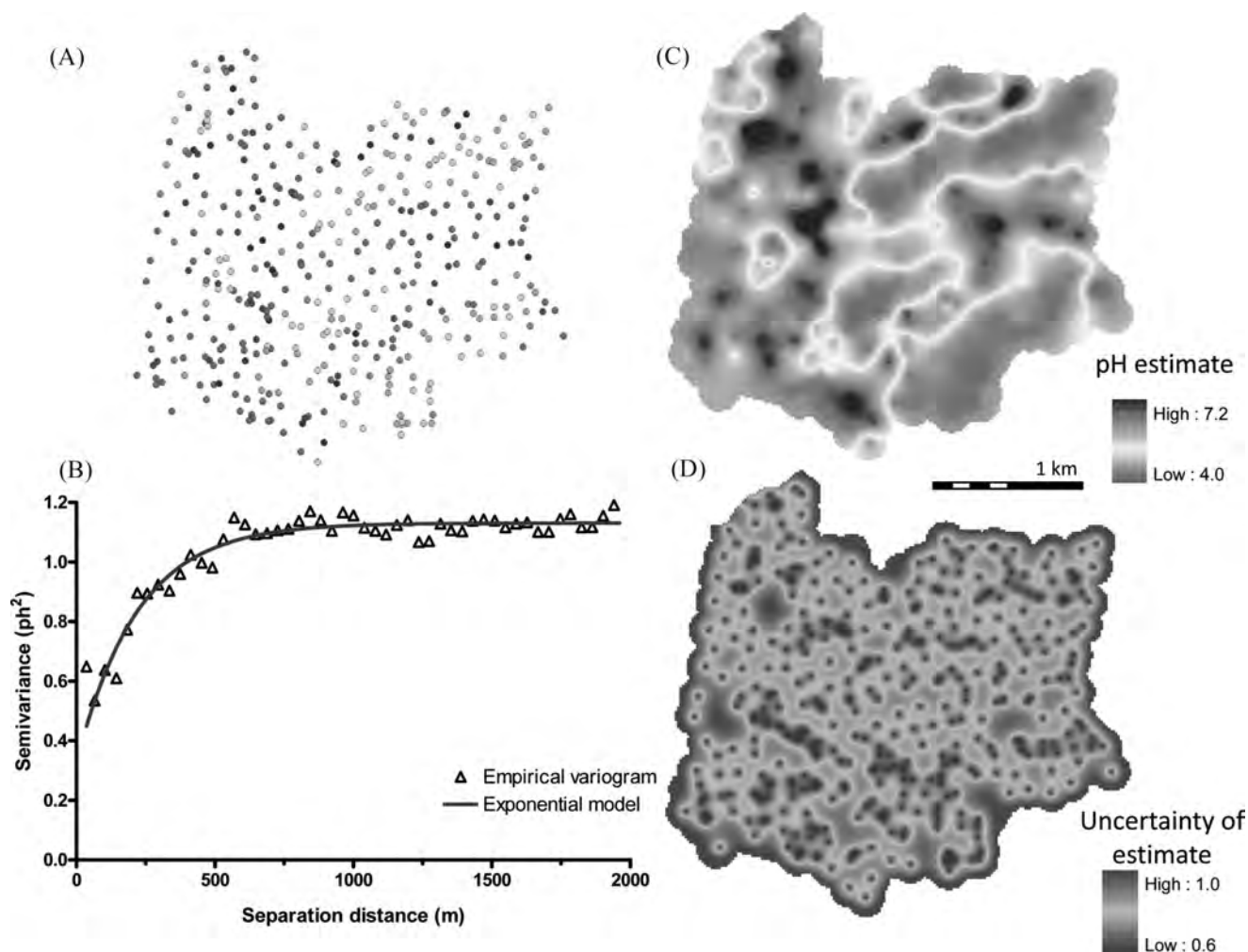


Fig. 2 Example of geostatistical prediction of subsoil pH. From field observations (A), spatial variability is characterized by a variogram (B). We can see that the pH values have a strong spatial correlation up to a separation distance of 600 m. The information on the soil variogram is used in a spatial prediction technique called kriging to produce a map of soil pH (C). Kriging provides the estimation of pH value at unknown location as a weighted average of the data in its neighborhood. The weights associated with neighboring observations depend on the way pH varies in space as described by the variogram. In addition to the estimate, kriging also calculates the uncertainty of the prediction (D).

kriging.^[12] It can take into account simultaneously data of various types and qualities. The data can be exact measurements, intervals of values, or probability density functions.

Soil Sampling

As soil varies across space and time, and under existing technology, it is impossible to capture the soil properties completely. The purpose of sampling is to obtain data that enable the estimation of some statistical parameters or spatial predictions of some properties over an area. Sampling is constrained by financial and available resources; thus, an efficient sampling strategy is sought for applications in soil survey for mapping and for establishing sites for monitoring networks. As an example, a good statistically based soil sampling scheme for estimating the mean nutrient concentration of fields can be used as a tool

for environmental regulation of the application rates of manure.^[13] Several sampling strategies have been developed in soil science, namely, for estimating the mean concentration of nutrient or contaminant in a field, for estimating a variogram, and for estimating variables within an area using spatial interpolation. This includes methods that minimize the kriging variance, provide an optimal spatial coverage, and establish a prediction model from secondary or ancillary variables. This topic is covered extensively by de Gruijter et al.^[13]

Soil Inference Systems

Soil analysis is expensive, and some functional properties such as available water capacity needed in many crop and environmental models are limited and not available in most areas. They need to be predicted from more other simple and

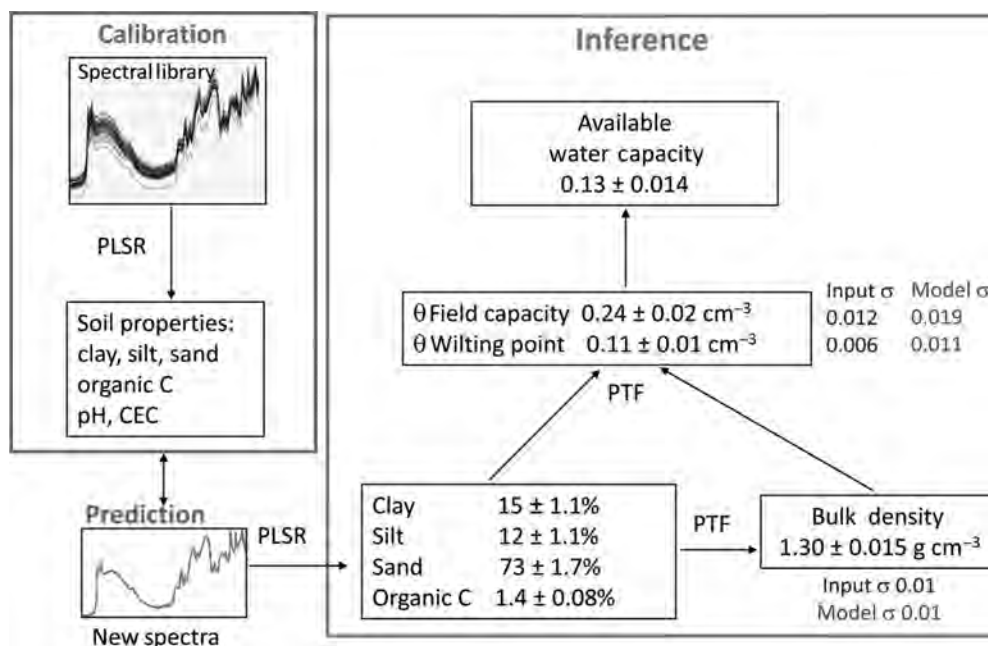


Fig. 3 Spectral soil inference system. Spectral calibration (left box) and an example of a soil inference system (right box). Spectral calibration involves the generation of prediction model called partial least squares regression (PLSR) to predict soil properties. Given a new spectrum, PLSR will predict the basic soil properties (clay, silt, sand, and organic C content) along with their uncertainties. These basic properties are then used to calculate other properties (bulk density), and the resulting input and model uncertainties are quantified. Further, basic properties and the predicted bulk density are used to calculate water content at field capacity and wilting point with subsequent available water capacity.

Source: From McBratney, Minasny, et al.^[16]

easy-to-measure properties. Soil inference systems take measurements with a given level of certainty and infer data that are not known with minimal uncertainties by means of logically linked predictive functions.^[14] These predictive functions are referred to as pedotransfer functions.^[15] The basic assumption underlying soil inference systems is that if we know or are able to predict the basic fundamental properties of the soil, we should be able to infer all other physical and chemical properties using pedotransfer functions.

Spectral soil inference systems, which combine prediction of soil properties from infrared spectroscopy and inference of soil properties using pedotransfer functions, were proposed by McBratney et al.^[16] as a means of predicting soil functional properties from a combination of various inputs, such as diffuse reflectance spectroscopy and basic soil properties (Fig. 3).

Modeling Soil Formation

Ultimately, pedometrics would attempt to provide a theory of soil variation through mechanistic models of soil evolution. This could be achieved by encapsulating knowledge of pedological processes in mathematical forms as an alternative to purely statistical approaches. The success of such a modeling approach will depend on pedological knowledge and the non-linearity of processes. The advantages of successful modeling of soil formation are substantial and

manifold. Soil spatial and temporal variation and soil genetic classes would be predictable from such a model. The first mechanistic soil profile model as a component of hillslope models was proposed by Kirkby.^[17] Initial work on modeling pedogenesis in the landscape has begun.^[18]

CONCLUSION

Pedometrics generally addresses the same issues as pedology but focuses on problems that can be formulated quantitatively and can be solved with mathematical and statistical techniques. Pedometrics is also closely related to other soil science subdisciplines: 1) soil physics; 2) chemistry; and 3) biology (Table 1). The following are the key areas of application:

1. Precision agriculture: This studies the spatial variation in crop yield and the soil and environmental factors that contribute to it. Pedometrics research has improved our understanding of the impact of spatial variability on crop yield and management and suggested differential treatments over a field.
2. Digital soil mapping: See the entry *Digital Soil Mapping* (p. 675).
3. Land evaluation: This is the process of predicting the potential use of land on the basis of its attributes.

Table 1 Pedometrics research in soil science subdisciplines.

Subdisciplines	Pedometrics research
Pedology	Modeling soil formation, numerical soil classification, spatial prediction of soil properties and soil classes, and chronofunctions
Soil survey	Soil sampling and digital soil mapping
Soil physics	Scaling, pedotransfer functions, uncertainty analysis, and time series analysis
Soil chemistry	Chemometrics and infrared spectroscopy for rapid analysis of soil properties
Soil biology	Quantifying soil biodiversity and spatial prediction of soil microbial diversity

A variety of analytical models can be used in these predictions, ranging from qualitative to quantitative, empirical to mechanistic, and specific to general.^[19]

- Contaminated sites: Pedometrics has been using geostatistical techniques to describe spatial distribution of heavy metals, pesticides, and salinity. These techniques provide estimates of concentration for an unvisited site and also the probability of the concentration at that location exceeding a critical threshold.^[10]

REFERENCES

- Webster, R. The development of pedometrics. *Geoderma* **1994**, *62* (1–3), 1–15.
- Webster, R.; Oliver, M.A. *Geostatistics for Environmental Scientists*, 2nd Ed.; John Wiley & Sons: Chichester, 2007.
- Grunwald, S., Ed. *Environmental Soil-Landscape Modeling. Geographic Information Technologies and Pedometrics*; CRC Press: Boca Raton, 2006.
- Hartemink, A.E.; McBratney, A.B.; Cattle, J.A. Developments and trends in soil science: 100 volumes of *Geoderma* (1967–2001). *Geoderma* **2001**, *100* (3–4), 217–268.
- Webster, R.; Burrough, P.A. Computer-based soil mapping of small areas from sample data. I. Multivariate classification and ordination. *J. Soil Sci.* **1972**, *23* (2), 210–221.
- Hole, F.D.; Hironaka, M. An experiment in ordination of some profiles. *Soil Sci. Soc. Am. Proc.* **1960**, *24* (4), 309–312.
- McBratney, A.B.; de Gruijter, J.J. A continuum approach to soil classification by modified fuzzy k-means with extra-grades. *J. Soil Sci.* **1992**, *43* (1), 159–175.
- Burrough, P.A. Fuzzy mathematical methods for soil survey and land evaluation. *J. Soil Sci.* **1989**, *40* (3), 477–492.
- McBratney, A.B.; Odeh, I.O.A. Application of fuzzy sets in soil science: Fuzzy logic, fuzzy measurements and fuzzy decisions. *Geoderma* **1997**, *77* (2–4), 85–113.
- Goovaerts, P. *Geostatistics for Natural Resources Evaluation*; Oxford University Press: New York, 1997.
- Lark, R.M.; Cullis, B.R.; Welham, S.J. On spatial prediction of soil properties in the presence of a spatial trend: The empirical best linear unbiased predictor (E-BLUP) with REML. *Eur. J. Soil. Sci.* **2006**, *57* (6), 787–799.
- Christakos, G. *Modern Spatiotemporal Geostatistics*; Oxford University Press: New York, 2000.
- de Gruijter, J.J.; Brus, D.J.; Bierkens, M.F.P.; Knotters, M. *Sampling for Natural Resource Monitoring*; Springer: Berlin, 2006.
- McBratney, A.B.; Minasny, B.; Cattle, S.R.; Vervoort, R.W. From pedotransfer functions to soil inference systems. *Geoderma* **2002**, *109* (1–2), 41–73.
- Bouma, J. Using soil survey data for quantitative land evaluation. *Adv. Soil Sci.* **1989**, *9*, 177–213.
- McBratney, A.B.; Minasny, B.; Viscarra Rossel, R. Spectral soil analysis and inference systems: A powerful combination for solving the soil data crisis. *Geoderma* **2006**, *136* (1–2), 272–278.
- Kirkby, M.J. Soil development models as a component of slope models. *Earth Surf. Proc.* **1977**, *2* (2–3), 203–230.
- Minasny, B.; McBratney, A.B. A rudimentary mechanistic model for soil production and landscape development II: A two-dimensional model incorporating chemical weathering. *Geoderma* **2001**, *103* (1–2), 161–180.
- Rossiter, D.G. A theoretical framework for land evaluation. *Geoderma* **1996**, *72* (3–4), 165–202.

Pedotransfer Functions

J.H.M. Wösten

*Alterra Research Institute, Wageningen University and Research Center,
Wageningen, the Netherlands*

Abstract

Pedotransfer functions (PTFs) are a powerful tool in estimating physical, chemical, and fertility properties of soils. Because PTFs predict properties that are difficult to obtain from already available basic soil properties, they have the clear advantage that they are relatively inexpensive and easy to derive and to use. For application on a specific location, use of PTFs might not be appropriate. Most successful PTFs are developed from large and reliable databases. Accuracy and reliability of PTFs will be appropriate for many applications on a regional and national scale. The search for data mining tools to develop better, more flexible PTFs, and the search for additional soil properties, as inputs in PTFs are important directions for improving PTF accuracy and reliability. Uncertainty in PTFs can be quantified. Its effects on calculated functional aspects of soil behavior will help to assess if an input parameter needs further detailing, and decide which one to assess to arrive at more accurate results.

INTRODUCTION

Simulation models, which are indispensable tools in modeling water and solute movement into and through soil, require as key input parameters easily accessible and representative hydraulic characteristics. Techniques to measure these characteristics are relatively time-consuming and therefore costly. At the same time, good predictions of the characteristics instead of direct measurements may be accurate enough for many applications. Considering the desired accuracy and the available financial resources, it is rewarding to analyze existing databases containing measured hydraulic characteristics and to establish relationships that predict the characteristics from measured basic soil data. These predictive relationships are called “pedotransfer functions” (PTFs)^[1] and they essentially translate data “we have” into data “we need.” Basically, PTFs relate soil characteristics being assembled during soil survey to more complex characteristics needed for simulation. Predicting soil hydraulic characteristics dominates the research field, though soil chemical and soil biological characteristics are also being predicted. Several reviews on PTF development and use have been published.^[2,3]

Large databases on measured hydraulic characteristics, such as UNSODA,^[4] HYPRES,^[5] WISE,^[6] and United States Department of Agriculture Natural Resource Conservation Service pedon database,^[7] form the essential, basic sources of information for the derivation of PTFs. In using PTFs, insight is needed to determine the input variables that are to be included in a PTF, what technique

is to be used to establish a PTF, and how accuracy and reliability of PTFs are to be quantified.

FUNCTIONS USED TO DESCRIBE THE WATER RETENTION AND HYDRAULIC CONDUCTIVITY CHARACTERISTICS

Describing hydraulic characteristics as functions rather than as tables has the clear advantage that they can be easily incorporated in simulation models. There exists a wide range of different equations for the description of the characteristics. The following equations to describe volumetric soil water content, θ , and hydraulic conductivity, K , as functions of pressure head, h , are widely used:^[8]

$$\theta(h) = \theta_r + \frac{\theta_s - \theta_r}{(1 + |\alpha h|^n)^{1-1/n}} \quad (1)$$

$$K(h) = K_s \frac{[(1 + |\alpha h|^n)^{1-1/n} - |\alpha h|^{n-1}]^2}{(1 + |\alpha h|^n)^{(1-1/n)(l+2)}} \quad (2)$$

In these equations, the subscripts r and s refer to residual and saturated values, and θ , n , and l are parameters that determine the shape of the curve. The residual water content θ_r refers to the water content, where the gradient $d\theta/dh$ becomes zero ($h \rightarrow -\infty$). The parameter α (1/cm) approximately equals the inverse of the pressure head at the inflection point. The dimensionless parameter n reflects the steepness of the curve. The dimensionless parameter l

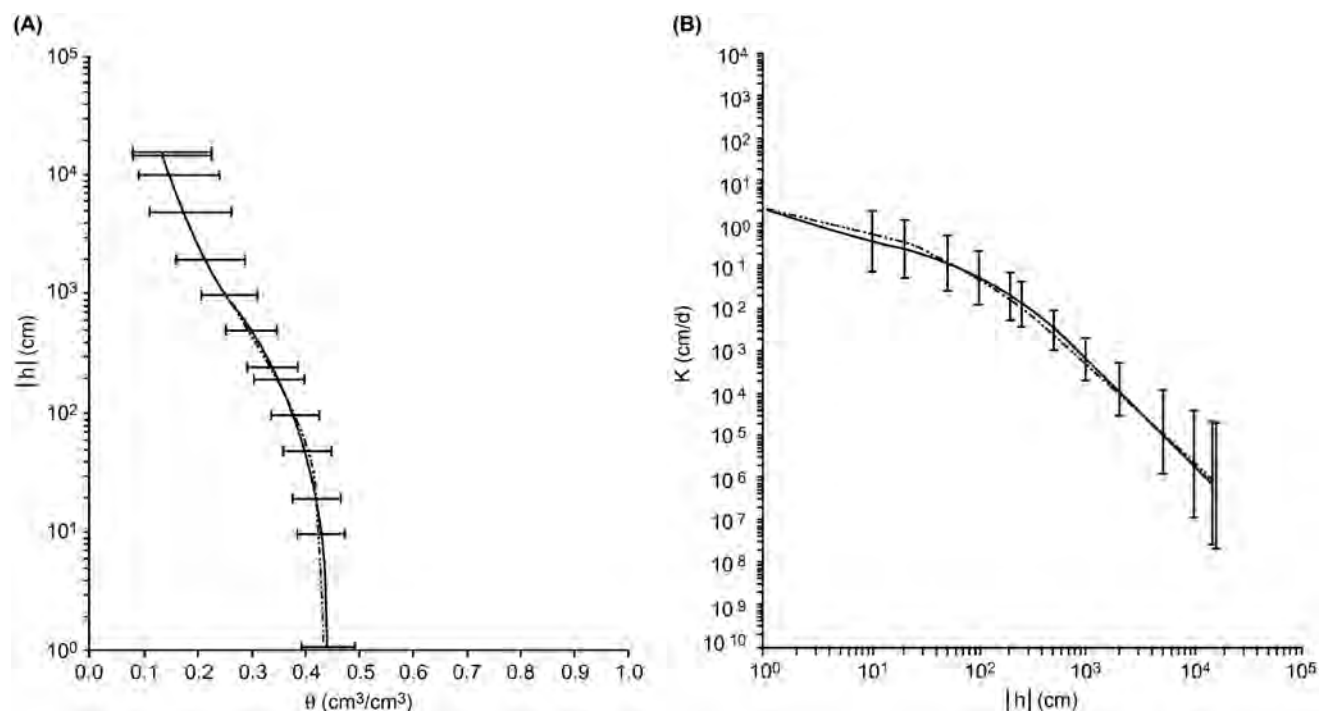


Fig. 1 Geometric mean water retention (A) and hydraulic conductivity (B) characteristic solid lines, standard deviations bars, and van Genuchten fits dotted lines for the texture class “medium fine topsoil.”

determines the slope of the hydraulic conductivity curve in the range of more negative values of h .

PTFs to predict the model parameters θ_r , θ_s , K_s , α , l , and n from basic soil data were built by many authors.^[9] Fig. 1 shows the mean water retention and hydraulic conductivity characteristics, also called class PTFs, for the texture class “medium fine topsoil.”^[5]

SELECTION OF PTF PREDICTOR VARIABLES

Soil properties affecting water retention and hydraulic conductivity are manifold.^[10] Table 1 lists the properties used most often as predictors because of their availability and because they proved to be the most promising ones.

Table 1 Soil properties often used in PTFs.

Particle size properties	Hydraulic characteristics	Morphological properties	Chemical/mineralogical properties	Mechanical properties
Sand, silt, clay	Water content at	Bulk density	Organic carbon	Penetration resistance
Fine sand	−33 kPa	Porosity	Organic matter	
	−1500 kPa	Horizon	CEC	
Very coarse sand, coarse fragments	Reference moisture retention curve	Structure	Clay type	
Median or geometric mean particle size		Grade	CaCO ₃	
		Size	Iron	
		Shape		
Water-stable aggregates		Color		
		Consistence		
		Pedality		
		Landscape position		

“Particle size distribution” is used in almost all PTFs. Particle size classes differ in different national and international classification systems, and so the number and the size of classes used in PTFs may also differ. Using sand, silt, and clay contents is a common approach.

“Limited, measured water retention data” at, for instance, two pressure heads may dramatically improve predictions of the complete water retention characteristic.

“Porosity or bulk density” is an important variable in many PTFs.

“Soil structure and morphology descriptors” such as the parameter topsoil and subsoil are successfully included in PTFs.

“Landscape position” is used as a topographic variable in PTFs.

“Organic matter content” is often used as predictor, but because bulk density and organic matter content are correlated, bulk density may effectively substitute organic matter content.

“Mechanical properties and shrink–swell parameters,” as characterized by the coefficient of linear extensibility, are used to estimate both water retention and K_s .

METHODS TO DEVELOP PTFs

When the set of PTF input parameters is defined and the PTF output is decided upon, a method is selected to relate input and output. The most prominent methods to create this relationship are discussed below.

Regression Analysis

The oldest and most widely used method to build PTFs was, exclusively, linear regressions, but this method is gradually being replaced by non-linear regression. An advantage of using regression techniques is that the essential predictor variables can be found automatically using stepwise regression. A drawback of regression is that it requires a model of the dependency of soil properties, which is almost impossible now that large digitized databases containing numerous soil properties are widely available. One remedial approach is to use principal component analysis. It allows identification of a small number of new variables that are linear combinations of the original predictors and that can explain a large percentage of variability within samples. An example of this type of PTF predicting the saturated water content θ_s is presented in Table 2.

Artificial Neural Networks

Artificial neural networks (ANNs) are becoming a common tool for modeling complex “input–output” dependencies. The advantage of ANNs is that no *a priori* model that relates input to output is required. After establishing the

Table 2 Continuous PTFs developed from the HYPRES database.

$$\begin{aligned}\theta_s = & 0.7919 + 0.001691C - 0.29619D - 0.000001491S^{2a} \\ & + 0.0000821OM^2 + 0.02427C^{-1} + 0.01113/S \\ & + 0.01472\ln(S) - 0.0000733OMC - 0.000619DC \\ & - 0.001183DOM - 0.0001664 \times \text{topsoil} \times S \\ & (R^2 = 76\%) \end{aligned}$$

^a θ_s is the saturated water content in the van Genuchten equations; C, percentage clay (i.e., percentage $<2 \mu\text{m}$); S, percentage silt (i.e., percentage between 2 and $50 \mu\text{m}$); OM, percentage organic matter; D, bulk density; topsoil and subsoil are qualitative variables having the value of 1 or 0; and $\ln$, natural logarithm.

network structure and finding coefficients to express the degree of influence of the network components on each other, an ANN generates a complex formula relating input with output values. This formula can be used like a regression formula. ANNs consisting of many interconnected simple computational elements, called nodes or neurons, are comparable to the best non-linear statistical regression approaches.

Group Method of Data Handling

The group method of data handling (GMDH) has, unlike ANNs, a built-in algorithm to retain only essential input variables in a flexible net of regression equations, which relates inputs to outputs. GMDH is a technique of finding an approximate relationship between a set of input variables $x_1, x_2, \dots, x_N$ and an output variable y . When the number of input variables is very large, or the relationship between inputs and output is very complex, GMDH successfully competes with statistical regression.

Regression Tree Algorithm

Regression tree algorithm is a recursive data partitioning algorithm that initially splits the data set into two subsets based on a single best predictor variable (the variable that minimizes the variance in the response). It then does the same on each of the subsets and so on recursively. The output is a tree with branches and terminal nodes. The predicted value at each terminal node is the average at that node.

Fig. 2 shows the accuracy of predicting water content at -33 kPa pressure head with the last three methods.

ACCURACY AND RELIABILITY OF PTFs

As empirical equations, PTFs are routinely evaluated in terms of the correspondence between measured and estimated values. When measured values are those used to develop the equation, the accuracy of the equation is

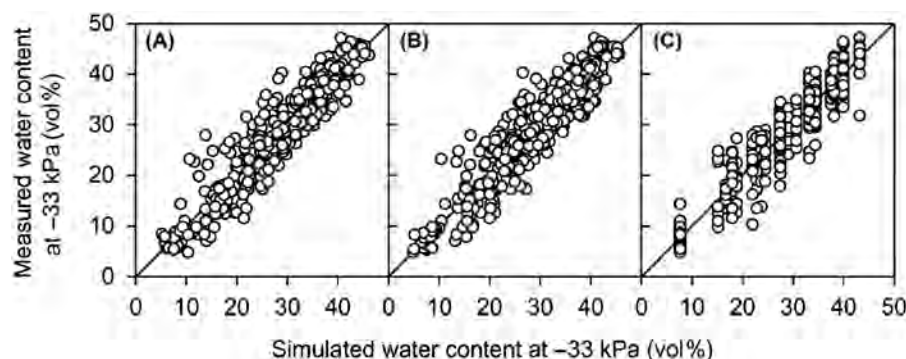


Fig. 2 One-to-one diagrams showing accuracy of predicting water content at -33 kPa suction with three methods: (A) back propagation ANN; (B) GMDH; and (C) regression tree algorithm. As can be seen, the accuracy of the three methods is quite comparable. In all three cases, the RMSE is around 3.4 vol%, the R^2 value of the regression is around 0.9 , the slope of the regression is close to one, and the intercept is about 0.001 vol%.

evaluated. When the measured values are different from the ones used to develop the equation, the reliability is evaluated. A multitude of statistics is used in PTF development including: multiple determination coefficient (R^2), root mean square error (RMSE), mean error (ME), and t-statistics to test the null hypothesis.

Accuracy of PTFs

Using the same statistics, the accuracy of existing PTFs varies appreciably. Accuracy estimates themselves may serve as benchmarks. However, they should be compared to variability of other measured input data. The model should not be more accurate than data used in the model development. Therefore, the PTF can be thought to be accurate if the variability of the PTF errors does not differ significantly from the variability in data, and if the average error does not significantly differ from zero.

Reliability of PTFs

The reliability of PTFs can be evaluated by cross-validation, i.e., splitting the available data set in a development and validation subset, or by using an independent data set.

Many studies assessed the reliability of PTFs by applying them to independent, regional data sets. From this, it appears that PTFs developed from regional databases give good results in regions with a similar soil and landscape history.

CONCLUSION

A number of conclusions can be formulated.

1. PTFs are a powerful tool in estimating physical, chemical, and fertility properties of soils. Because PTFs predict properties that are difficult to obtain from already available basic soil properties, they have the clear advantage that they are relatively inexpensive and easy to derive and to use.
2. For application on a specific location, use of PTFs might not be appropriate. In this case, direct measurement is the only option. PTFs should not be used to make predictions for soils that are outside the range of soils used to derive the PTFs. In other words, use of PTFs for interpolation purposes is safe but dangerous to use for extrapolation.
3. Most successful PTFs are developed from large and reliable databases. This implies that well structured and easily accessible national and international databases of measured soil hydraulic characteristics need to be created. PTFs should be periodically updated as more measured data become available from these databases.
4. Accuracy and reliability of PTFs will be appropriate for many applications on a regional and national scale. On these scales, temporal and spatial variability of other than hydraulic characteristics most likely will also have an important impact on the modeling results.
5. The search for data mining tools to develop better, more flexible PTFs, and the search for additional soil properties, as inputs in PTFs are important directions for improving PTF accuracy and reliability.
6. Uncertainty in PTFs can be quantified. Its effects on calculated functional aspects of soil behavior will help to assess, if an input parameter needs further detailing, and decide which one to assess, to arrive at more accurate results.

REFERENCES

1. Bouma, J.; van Lanen, J.A.J. Transfer functions and threshold values: from soil characteristics to land qualities. In *Quantified Land Evaluation, Int. Inst. Aerospace Surv. Earth Sci. Publ. No. 6*, Proceedings of Workshop ISSS and SSSA, Washington, DC, April 27–May 2, 1986; Beek, K.J., Ed.; ITC Publ.: Enschede, The Netherlands, 1987; 106–110.
2. Van Genuchten, M.Th.; Leij, F.J. On estimating the hydraulic properties of unsaturated soils. In *Indirect Methods for Estimating the Hydraulic Properties of Unsaturated Soils*; van Genuchten, M.Th., Leij, F.J.,

- Lund, L.J., Eds.; University of California: Riverside, 1992; 1–14.
3. Pachepsky, Y.A.; Rawls, W.J.; Timlin, D.J. The current status of pedotransfer functions: their accuracy, reliability, and utility in field- and regional-scale modelling. In *Assessment of Non-Point Source Pollution in the Vadose Zone*; Corwin, D.L., Loague, K., Ellsworth, T.R., Eds.; Geophysical Monograph 108; American Geophysical Union: Washington, 1999; 223–234.
 4. Leij, F.; Alves, W.J.; van Genuchten, M.Th.; Williams, J.R. *The UNSODA Unsaturated Soil Hydraulic Database*. User's Manual Version 1.0.; EPA/600/R96/095; National Risk Management Laboratory, Office of Research and Development: Cincinnati, 1996.
 5. Wösten, J.H.M.; Lilly, A.; Nemes, A.; Le Bas, C. Development and use of a database of hydraulic properties of European soils. *Geoderma* **1999**, *90*, 169–185.
 6. Batjes, N.H. Development of a world data set of soil water retention properties using pedotransfer rules. *Geoderma* **1996**, *71*, 31–52.
 7. U.S. Department of Agriculture. *Soil Survey Manual*. U.S. Department of Agriculture Handbook No. 18; U.S. Department of Agriculture: Washington, 1951.
 8. Van Genuchten, M.T. A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. *Soil Sci. Soc. Am. J.* **1980**, *44*, 892–898.
 9. Wösten, J.H.M.; van Genuchten, M.T. Using texture and other soil properties to predict the unsaturated soil hydraulic functions. *Soil Sci. Soc. Am. J.* **1988**, *52*, 1762–1770.
 10. Rawls, W.J.; Gish, T.J.; Brakensiek, D.L. Estimating soil water retention from soil physical properties and characteristics. *Adv. Soil Sci.* **1991**, *16*, 213–234.

Permafrost: Soil Temperature and Special Problems

Douglas Kane

Julia Boike

Water and Environmental Research Center, University of Alaska, Fairbanks, Alaska, U.S.A.

Abstract

Permafrost and seasonally frozen ground are very extensive in the high latitudes of earth. Also, resource development and population in these areas are increasing. Because of climatic change, the permafrost is not in thermal equilibrium. Instead, it is warming in many areas, and this needs to be considered in the design criteria for all engineering structures built on permafrost. Warming, with subsequent thawing of permafrost, will affect the surface energy budget, water and gas fluxes (potential for release of greenhouse gases), and vegetation; hence, there is a direct feedback with the climate system. Just as it took a long time for permafrost to develop, it will take a long time for it to completely melt, but the consequences of thawing permafrost can impact our daily lives in the first or second season of melting.

INTRODUCTION

On an extended south-to-north transect at northern latitudes, a transition from ground that never experiences seasonal freezing to those that occasionally freeze during the winter, to those that freeze every year, and to those that may remain frozen for an extended time could be encountered. The topic of discussion in this entry is those surface soils and deeper geologic layers that remain at or below freezing for a duration of 2 years or more and how they impact people living in this environment. Such frozen ground, both unconsolidated and bedrock, is commonly referred to as permafrost.^[1] Although at or below the freezing point of bulk water (0°C), the term “permafrost” implies that neither water is present nor water, if present, is frozen. In fact, it is possible for significant amounts of water to remain unfrozen in permafrost; this is also true for water in seasonal frost.

PERMAFROST CHARACTERISTICS

Spatially extensive permafrost can be found in Russia and Canada as far south as 45° N and even farther south on the elevated Tibetan Plateau and Himalayan Mountains.^[2] Approximately 25 million km² of permafrost exists in the northern hemisphere. In the higher latitudes, permafrost is continuous under the land surfaces. At intermediate latitudes, permafrost is discontinuous or sporadic. Leggett^[3] reported that 20% of the land surface of the world is underlain by permafrost. More than

50% of Russia and Canada is underlain by permafrost. Alaska has continuous permafrost in the northern one-third of the state and discontinuous permafrost in the rest of the state, excluding the coastal areas from the Aleutian Islands to southeastern Alaska (Fig. 1). In the southern hemisphere, permafrost distribution is confined to Antarctica and high alpine or mountainous regions. Isolated permafrost is common at higher elevations, and evidence of past permafrost is common in areas that no longer have permafrost. The thickness of permafrost can vary from a thin lens of less than 1 m to greater than 1000 m (Fig. 1). Permafrost can also be found in coastal areas at the bottom of seas.

Permafrost ground is interesting to the engineer and scientist because the medium is usually composed of two solids (porous medium and ice) and two fluids (air and liquid water). The more components that are present in a mixture, the more difficult it is to predict the medium's response to an input of energy or mass. The amount of unfrozen water in saturated ground is strictly a function of the grain size (more specifically, surface area) and the freezing temperature. A fine-grained soil such as clay has a very high surface area relative to coarse-grained soils such as sand; in frozen ground, this translates into much higher unfrozen water contents at the same temperature. The unfrozen water found in permafrost exists as a film of water around each soil particle. It is via these unfrozen films that water moves in permafrost. Frozen clay can have as much as 5–7% unfrozen water by volume at –15°C. As the temperature decreases, the amount of unfrozen water also decreases. Thermal and hydraulic properties of permafrost are quite

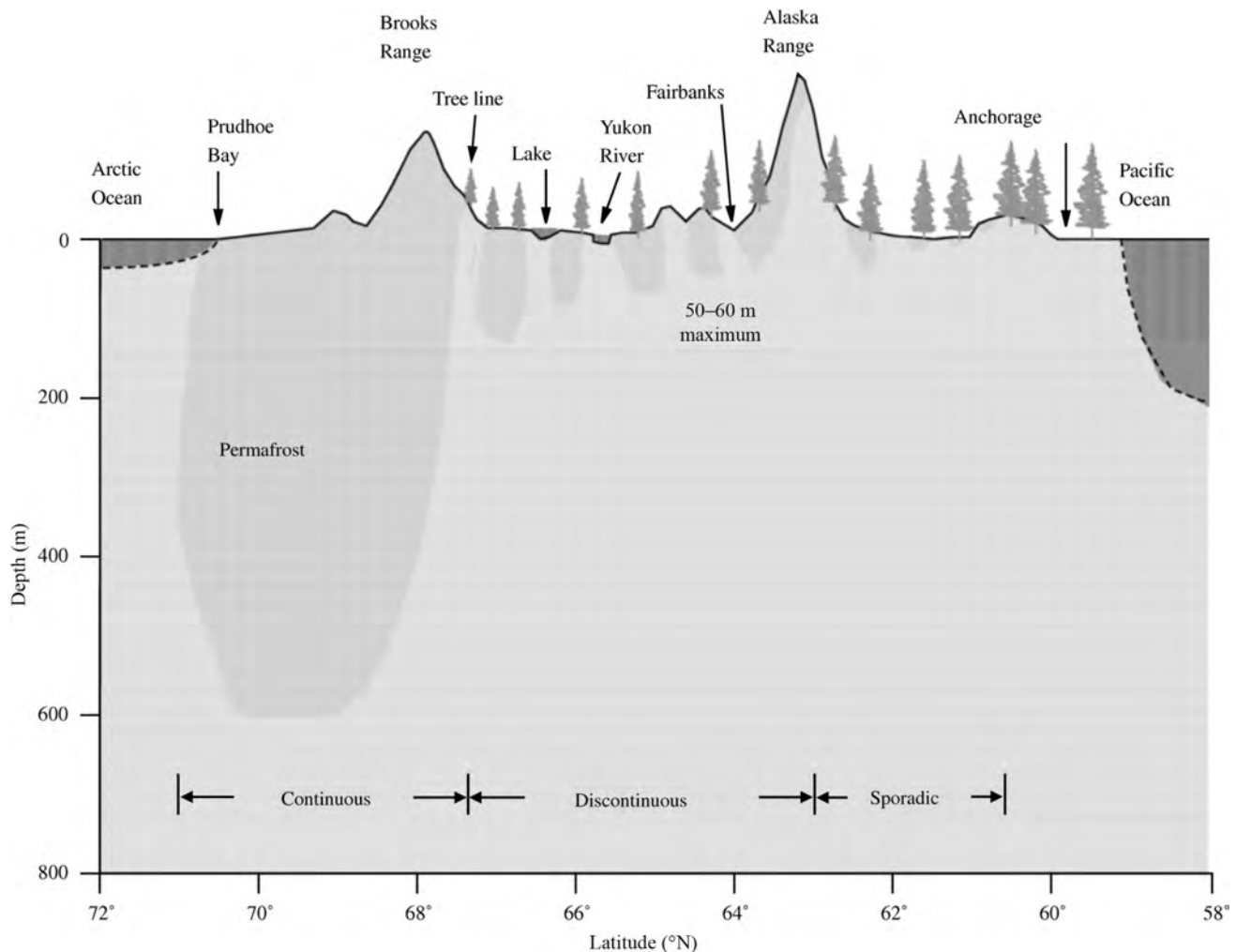


Fig. 1 The schematic representation of permafrost distribution in Alaska on a south-to-north transect.

variable and depend on the percentages of the various ground components. Most heat transfer in permafrost is by conduction and can be modeled by Fourier's law although simpler methods have been developed.

A typical temperature profile of permafrost is shown in Fig. 2. Since there is a geothermal flux outward from the center of the earth, this heat has to be successfully transferred to the ground surface or the permafrost will warm and melt. In order to maintain the thermal integrity of the permafrost, the soils above the top of the permafrost table must completely freeze during the winter, and so there is a continuously decreasing thermal gradient along which the geothermal heat can be transferred to the surface by conduction (Fig. 2).

The seasonally thawed soil layer at the ground surface that goes through freezing/thawing annually and mantles the permafrost is called the active layer (Fig. 2). This layer acts as a buffer to heat and mass transfer to the permafrost. A typical active layer in the continuous permafrost zone would typically thaw to a maximum depth of 60 cm. The top layers within 15–25 cm of these soils are generally

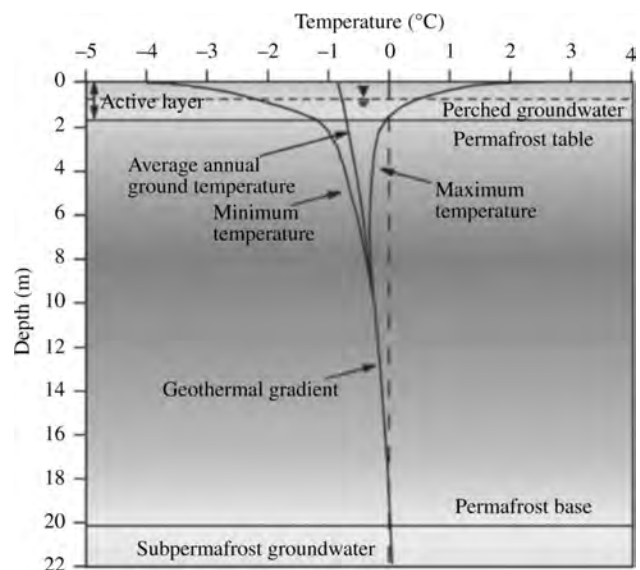


Fig. 2 Typical temperature profile for permafrost, with annual variation indicated near the ground surface.

composed of organic material, with the deeper soils being mineral. Organic soils are good thermal insulators, and when coupled with snow cover during the winter months, they minimize heat loss from the ground. The thermal balance of the permanently frozen ground is maintained by heat loss to the atmosphere that occurs at high latitudes over the extended winter months.

Permafrost is usually considered to be impermeable to water movement. This is usually a good assumption for short durations of days or even a few months. For longer periods of time, the redistribution of water within permafrost can be appreciable. During snowmelt and major rainfall events, considerable water enters the active layer. Most of this water resides in the organic soils as the mineral soils are usually already near saturation. Since permafrost has relatively low hydraulic conductivity, there is no hydraulic connection between the perched water above the permafrost (suprapermfrost groundwater) and the subpermafrost groundwater below the permafrost (Fig. 2). In continuous permafrost, the subsurface hydrology is confined to the active layer. For areas of discontinuous permafrost, the subsurface hydrology is a combination of shallow flow over the frozen ground and deeper flow around and under it.

SURFACE ENERGY BALANCE

Any time the ground surface is disturbed, the surface energy balance that sustains permafrost is upset. This generally results in warming of the permafrost and thickening of the active layer. Much sporadic and discontinuous permafrost is maintained at temperatures just below freezing; climatic warming of just a few degrees would result in the melting of this frozen ground and warming of colder permafrost. Also where permafrost exists, surface disturbances such as removing vegetation or surface soils and ponding of water are sufficient to alter the surface energy balance and cause permafrost degradation. Permafrost generally appears where the mean annual surface temperature is at least a few degree Celsius below freezing; for instance, Hay River, Northwest Territories, Canada, has sporadic, shallow permafrost with a mean annual temperature of -3.4°C .

In areas of discontinuous permafrost, the south-facing slopes are generally permafrost free (Fig. 1), while north-facing slopes have permafrost; the east- and west-facing slopes and valley bottoms may or may not have permafrost, depending on subtle site-specific conditions that impact the energy balance (vegetation, slope, moisture content, etc.). Because it took thousands of years for permafrost to develop, deep permafrost is particularly a good recorder of past climates, and changes in the temperature profile will reflect these impacts. Many general atmospheric circulation models predict variable global warming in the high latitudes, and in some areas, there is

already field evidence of atmospheric warming^[4] and permafrost warming.^[5] To sustain permafrost, the geothermal heat radiating out from the core of the earth must be removed; this only occurs if the active layer is completely frozen during winter.

LIVING WITH PERMAFROST

Permafrost influences many facets of everyday living that are often taken for granted in warmer climates. Permafrost negatively impacts both the quality and the quantity of the available groundwater and limits the potential for the infiltration of wastewater. As a result, housing units with individual water and wastewater systems must be designed with innovative options to replace the traditional groundwater well and septic system for wastewater treatment. Utilities that are typically buried are often placed in aboveground utilidors. Also as roadways age, they become very bumpy as differential movement of the road surface occurs.

The significant property of permafrost from an engineering viewpoint is the ice content. The amount of ice can range from essentially none in well-drained ground to values that exceed the porosity of poorly drained ground. Within the permafrost, segregated ice with minimal entrained soil exists as ice lenses and wedges. As long as the permafrost stays frozen, the ground is relatively stable. However, thawing of ice-rich permafrost causes ground to settle or subside. This thaw settlement compromises the structural integrity of buildings, roadways, pipelines, and other structures built on it. For areas where construction must take place on frozen ground, special efforts—e.g., the use of insulation and thermosyphons—are necessary to maintain the structural integrity of the permafrost. Thermosyphons are vertical devices for removing heat from the ground, and they are used extensively under buildings and pipelines to maintain below freezing conditions in the ground.

Soil freezing causes additional engineering problems near the ground surface when strong thermal gradients induce water movement from warm ground to cold ground through unfrozen films of water on the surfaces of soil particles, regardless of whether it is frozen or not. This process causes “frost heave,” as it results in an accumulation of ice in the form of lenses that can cause the ground to expand and deform. Surface heaving becomes significant over time periods of several years, and it is important for those engineering structures with design lives of tens of years to compensate for this additional stress.

CONCLUSIONS

Permafrost and seasonally frozen ground are very extensive in the high latitudes of earth. Also, resource

development and population in these areas are increasing. Initial efforts directed at building an infrastructure (roads, airports, water and sewer distribution, etc.) on permafrost assumed that the frozen ground was in thermal equilibrium. It is obvious that because of climatic change, the permafrost is not in thermal equilibrium. Instead, it is warming in many areas, and this needs to be considered in the design criteria for all engineering structures built on permafrost. Warming, with subsequent thawing of permafrost, will affect the surface energy budget, water and gas fluxes (potential for release of greenhouse gases), and vegetation; hence, there is a direct feedback with the climate system. Just as it took a long time for permafrost to develop, it will take a long time for it to completely melt, but the consequences of thawing permafrost can impact our daily lives in the first or second season of melting.

REFERENCES

1. Muller, S.W. *Permafrost or Permanently Frozen Ground and Related Engineering Problems*. U.S. Engineers Office, Strategic Engineering Study, Special Report No. 62, Edwards: Ann Arbor, 1943; 136 pp.
2. Brown, J.; Ferrians, O.J., Jr.; Heginbottom, J.A.; Melnikov, E.S. *Circum-Arctic Map of Permafrost and Ground-Ice Conditions*; U.S. Dept. of Interior, Geological Survey, Map CP-45; National Snow and Ice Data Center: Boulder, CO, 1998.
3. Leggett, R.F. Permafrost research. *Arctic* **1954**, 7 (3–4), 153–158.
4. Serreze, M.C.; Walsh, J.E.; Chapin III, F.S.; Osterkamp, T.; Dyurgerov, M.; Romanovsky, V.; Oechel, W.C.; Morison, J.; Zhang, T.; Berry, R.G. Observational evidence of recent change in the northern high-latitude environment. *Clim. Change* **2000**, 46, 159–207.
5. Lachenbruch, A.H.; Marshall, B.V. Changing climate: Geothermal evidence from permafrost in the Alaskan Arctic. *Science* **1986**, 234, 689–696.

Petrocalcic Horizons

Alain Ruellan

National Superior School of Agronomy of Rennes, Montpellier, France

Abstract

Petrocalcic horizons are strongly developed horizons that have accumulated calcium or other carbonates. Petrocalcic horizons distribution in the soil covers permits to prove the existence of pedological systems, i.e., lateral relationships between morphological features and dynamics of the soil mantle.

INTRODUCTION

In Mediterranean and tropical arid regions, B or C horizons of soils frequently contain, at shallow depths (some decimeters), more than 50% calcium carbonate. Generally, these horizons are layered (platy), hard, and 10–100 cm thick. These Bca and Cca horizons are called “lime crusts,” “calcretes,” or “petrocalcic horizons.” Soils with petrocalcic horizons are extensive on plateaus, slopes, and river terraces.

Petrocalcic horizons are strongly developed horizons that have accumulated calcium or other carbonates.^[1] They are indurated or cemented calcic horizons; vertically, they are related to the A and B horizons above and to C horizon below, which may or may not be calcareous.^[2]

THE CALCIC HORIZONS

Calcic horizons^[3,4] are horizons of calcium carbonate accumulation. In the calcic horizons, the calcium carbonate:

1. may transfer from the top of the soil profile, from the upper part of the pedological toposequence, from the air (wind), and from the underground water.
2. precipitates giving rise to very rich calcium carbonate forms that are either discontinuous (pseudomycelia, cutans, nodules, or veins) or continuous (petrocalcic horizons).

As development proceeds, the carbonate accumulates, first filling the pores and then replacing pre-existing minerals (epigeny).^[5–8]

THE PETROCALCIC HORIZONS

When the calcium carbonate concentration in B or C horizons increases sufficiently to mask all or most of the pedological or lithological structure within which it develops, it becomes a petrocalcic horizon (or calcrete

or lime crust).^[2–4] When consolidated, the carbonate content of the horizon is 50–90%.

There are several types of petrocalcic horizons:

1. Unlayered (non-platy) petrocalcic horizons: the structure can be massive, angular, finely lamellar, or nodular. The color is nearly white.
2. Layered (platy) petrocalcic horizons:

Layered, non-compact, petrocalcic horizons are made up of superposed, clearly individualized layers of crusty material, hard, but not petrified (Fig. 1). The thickness of these layers varies from a few millimeter to several centimeter; they are not continuous vertically but are separated from each other by more or less horizontal fissures arranged in a braided manner. The internal basic structure can be massive, nodular, or finely lamellar; the color is generally white to creamy white, with frequent black spots. As the crusty material becomes harder, it tends to be pink in color.

Layered compact petrocalcic horizons (slabs) (Fig. 2) are made up of one or several calcium carbonate layers, extremely hard, gray- or, more often, pink in color. Each layer can be 10–20 cm thick. These layers are petrified, usually very continuous. The internal basic organization is very massive without fine lamellae.

Finely layered compact petrocalcic horizons are very hard formations with thickness of a few millimeter to a few centimeter. The horizons are clearly stratified, consisting of one or more series of very fine superimposed lamellae; the color being white or pink.

In addition to calcium carbonate accumulation, structures, and hardness, other important data useful for characterizing and distinguishing different types of petrocalcic horizons are water pH and natural drainage:



Fig. 1 Below an A horizon (1) is a layered non-compact petrocalcic horizon (crust) (2), over an unlayered (non-platy) petrocalcic horizon (3), on a calcic horizon (4).

1. Basic types have a water pH between 8.0 and 8.7; ultrabasic types have a water pH > 8.7. A very high pH indicates the presence of magnesium carbonate.^[9]
2. Soils with petrocalcic horizons in environments with restricted drainage tend to form montmorillonite, attapulgite, and sepiolite clay minerals.^[6-8,10]

VERTICAL SUCCESSION OF THE CALCIC AND PETROCALCIC HORIZONS

Below A or B horizons, the accumulation of calcium carbonate can be of three types:

1. Slightly differentiated: the distribution of carbonates is diffuse, sometimes with pseudomycelia with very diffuse upper and lower limits.
2. Moderately differentiated: the carbonates in the calcic horizon occur partly as diffuse distributions in the soil matrix and partly as concentrations, forming cutans, soft and hard nodules, or veins, with diffuse upper and lower limits.
3. Highly differentiated (petrocalcic): the carbonates form laterally continuous concentrations resulting in one or more superposed petrocalcic horizons. The vertical complete succession from the top is: finely layered, over thicker compact layers (slabs), over



Fig. 2 A layered compact petrocalcic horizon (slab) (2) below an A horizon (containing pebbles of slab) (1), over a layered non-compact petrocalcic horizon (crust) (3), on a calcic horizon (4).

thick non-compact layers, over a massive horizon. There are four main types of vertical successions of petrocalcic horizons:

- single massive horizons;
- layered non-compact horizons, over unlayered horizons;
- finely layered horizons, over layered non-compact horizons, over unlayered horizons; and
- finely layered horizons, over layered compact horizons, over layered non-compact horizons, over unlayered horizons.

Petrocalcic horizons always have sharp upper boundaries and carbonate contents that decrease with depth. A gradual transition and discontinuous concentrations of carbonate mark the lower boundary of the petrocalcic horizon. The solum above the petrocalcic horizon is generally between 10 and 50 cm thick.

LATERAL DISTRIBUTION OF THE CALCIC AND PETROCALCIC HORIZONS

As with vertical transitions, progressive lateral transitions frequently occur between different forms of calcic and petrocalcic horizons. From this, it can be concluded that the vertical and horizontal structures of calcium carbonate accumulation result from the same mechanisms.

In a landscape, the lateral distribution of the calcic and petrocalcic horizons is mainly a function of topography and age.

Along a slope or pediment, calcic horizon development increases downslope. In a complete toposequence (catena), differentiation progresses from a soil with minimal calcic horizon development upslope, to a soil with moderate calcic horizon development immediately downslope, and finally to soils with more and more strongly developed petrocalcic horizons, the different types and superposition of petrocalcic horizons appearing successively (Fig. 3).

In time, calcic horizons evolve (Fig. 3) from diffuse carbonate distributions to discontinuous accumulations (pseudomycelia, cutans, nodules, or veins) to unlayered petrocalcic horizons to layered non-compact petrocalcic horizons to slab. The finely layered (ribboned) horizons can exist as soon as non-compact layered horizons appear.

On the other hand, decarbonation of the A and B horizons above the calcic and petrocalcic horizons does not increase with age. It is only on the younger Quaternary surfaces that a small decarbonation can be observed as these surfaces age. However, this decarbonation does not increase on the older surfaces and, when petrocalcic horizons appear, the calcium carbonate content of the upper horizons may increase due to erosion and subsequent formation of the upper horizons from the calcrete.

These facts confirm the following interpretations:

1. There is a logical order of appearance of the calcic and petrocalcic accumulations and horizons; this logical order is the same in space, vertically and laterally, and in time. These accumulations and horizons are thus genetically linked, by toposequences and chronosequences.

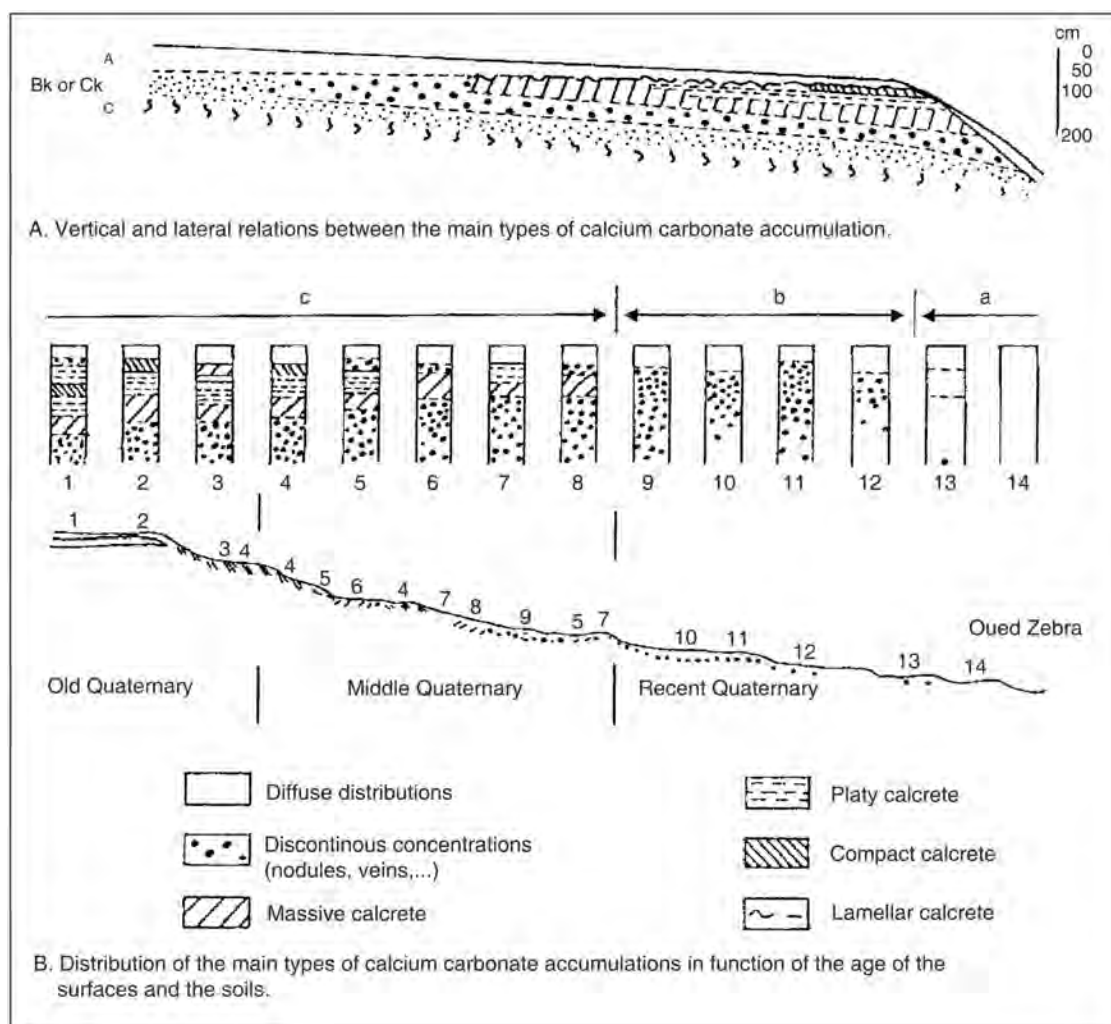


Fig. 3 Relationships between calcic and petrocalcic horizons in space and time (North Morocco). The length of the sequences may vary between some tens and several hundreds of meters; the difference in altitude, between the old and recent Quaternary surfaces, is some tens of meters.

Source: From Soil Survey Staff.^[1]

2. The vertical leaching of the calcium carbonate, which impoverishes the upper horizons in favor of the calcic and petrocalcic horizons, is a limited phenomenon; the major part of the calcium carbonate that accumulates in the soils comes from lateral redistributions.

In arid and semiarid regions, which are the privileged domains of the petrocalcic horizons, calcic and petrocalcic horizons can occur in soils formed from non-calcareous or non-calcic rocks: this happens when landscapes upstream furnish calcium by lateral lixiviation. However, in very arid regions near the sea, very strong petrocalcic horizons occur in soils formed from non-calcareous, non-calcic rocks without possible upstream sources of calcium. So calcium carbonate can also arrive by air as calcareous dust and calcium from sea spray.

SOIL USE

Petrocalcic horizons in soils at depths <50 cm limit the soil's agriculture uses, because of the physical (hardness and low porosity) and chemical (calcium carbonate, high pH) obstacles the horizon constitutes for the roots. It is possible to fracture the non-layered and the layered non-compact petrocalcic horizons to facilitate the penetration of water and roots for cotton, alfalfa, or fruit trees, but it is impossible to fracture when slabs are present, and it is costly too. Chemically, the non-layered, hard but non-compact petrocalcic horizons are more crop limiting because they contain too much active calcium and magnesium carbonate. Layered petrocalcic horizons in some areas are used for the construction of houses and building of roads.

CONCLUSION

Petrocalcic horizons distribution in the soil covers permits to prove the existence of pedological systems that is lateral relationships between morphological features and dynamics of the soil mantle.

REFERENCES

1. Soil Survey Staff. *U.S. Department of Agriculture, Natural Resources Conservation Service. Keys to Soil Taxonomy*, 2nd Ed.; USDA-NRCS: Washington, 1999.
2. Ruellan, A. *Contribution à la Connaissance des Sols des Régions Méditerranéennes: les Sols à Profil Calcaire Différencié de la Basse Moulouya (Maroc Oriental)*; Mémoire ORSTOM 54, ORSTOM: Paris, 1971.
3. Ruellan, A. Calcisols. In *World Reference Base for Soil Resources, Draft*; Spargaren, O.C., Ed.; ISSS, ISRIC, FAO: Wageningen, Rome, Italy, 1994; 106–111.
4. Ruellan, A. Calcisols. In *World Reference Base for Soil Resources, Introduction*; Deckers, J.A., Nachtergaele, F.O., Spaargaren, O.C., Eds.; Acco: Leuven, 1998; 53–56.
5. Boulet, R. *Toposéquences de Sols Tropicaux en Haute-Volta: Equilibre et Déséquilibre Pêdo-Bioclimatique*. Mémoire ORSTOM 62, ORSTOM: Paris, 1974.
6. Nahon, D.; Paquet, H.; Ruellan, A.; Millot, G. Encroûtements calcaires dans les altérations des marnes éocènes de la falaise de thiés (sénégal). organisation morphologique et minéralogie. *Sci. Géol. Bull. (Strasbourg)* **1975**, *28*, 29–45.
7. Millot, G.; Nahon, D.; Paquet, H.; Ruellan, A.; Tardy, Y. L'epigénie calcaire des roches silicatées dans les encroûtements carbonatés en pays subaride, anti-atlas, maroc. *Sci. Géol. Bull. (Strasbourg)* **1977**, *30* (3), 129–158.
8. Paquet, H.; Ruellan, A. Calcareous epigenetic replacement (epigénie) in soils and calcrete formation. In *Soils and Sediments, Mineralogy and Geochemistry*; Paquet, H., Clauer, N., Eds.; Springer: Berlin, 1997; 21–48.
9. Ruellan, A. Les sols salés et alcalisés en profondeur de la plaine du zebra (basse moulouya, maroc); premiers résultats d'une expérimentation destinée à étudier leur amélioration et leur evolution sous irrigation. In *Proceedings of the 8th International Congress of Soil Science, Bucharest, Romania, Aug 31–Sept 9, 1964*; Vol. II, 937–948.
10. Millot, G.; Paquet, H.; Ruellan, A. Néof ormation de l'attapulgitite dans les sols à carapaces calcaires de la basse-moulouya (maroc oriental). *C.R. Ac. Sci. Paris* **1969**, *268*, 2771–2774.

pH

Grant W. Thomas

University of Kentucky, Lexington, Kentucky, U.S.A.

Abstract

pH and buffering (reserves) of soils are controlled by a number of soil components such as clay minerals, organic matter, oxides of aluminum and iron, and compounds of calcium and sodium. In a few cases, the oxidation of sulfide overrides these more common controllers of pH, causing exceedingly low pH values. Including the latter, a range of pH values from 2 to 10 can be encountered. In most soils, however, a pH range of 4 to 9 is more common. The chemical reactions that control the soil pH are usually what interest us. In other words, pH reveals the reactions that dominate the soils. As such, pH is very useful as a clue about what must be done to the soil to make it ideal for the purpose we desire. No other single soil measurement gives us so much information so easily.

WHAT SOIL pH VALUES MEAN

Although pH values are not adequate to determine lime requirement, they do indicate rather precisely what is going on in the soil in terms of the major chemical reactions. In addition, pH infers secondary reactions such as the availability of both primary nutrients and micronutrients. The major pH categories are described below, together with their significance.

THE PRESENCE OF FREE ACIDS

Soil pH values around 2 to 3 indicate the presence of a free mineral acid. In almost all cases, the acid is sulfuric acid (H_2SO_4) which generally arises from the oxidation of iron (Fe) sulfides. The usual reason for their oxidation is exposure to the atmosphere during surface mining, excavation during construction, or drainage of submerged soils. Obviously, not all materials contain sulfides, but if they are present, exposure begins their oxidation, and the oxidation results in free H_2SO_4 . This acid then begins to dissolve soil components, including the clay minerals. If pH in the range of 2 to 3 persists with time, this indicates that there is sufficient free acid present to dominate all soil reactions. The long-term effect is the virtual dissolution of clay minerals.

In addition to the long-term destruction of the soil, plant growth is almost totally inhibited and the cost of neutralizing the acidity will be very high. On a practical level, neutralization of free acid is too expensive to attain by depending of the income received from growing crops. Therefore, it usually is achieved only by high expenditures for agricultural limestone, commonly mandated by government regulation; e.g., the use of 100 Mg of calcium carbonate (CaCO_3) per hectare is

commonly required to neutralize the free acid. That amount of limestone commonly costs more than the value of the land treated.^[1]

THE PRESENCE OF Al^{3+} IONS

At pH values of 4 to 5, trivalent exchangeable aluminum (Al) is always present in soils unless they are quite high in organic matter. The pH of the soil is dominated by the hydrolysis reaction:



which has a minimum pH of 3.84 in montmorillonite and a maximum pH of 4.89 in hectorite.^[2]

This pH range indicates problems with the growth of most crop plants because root growth is strongly inhibited by the presence of Al^{3+} . Nevertheless, the case is not nearly as serious as with free mineral acid for two reasons: First, plants are not killed outright, and second, correction of the problem usually is not an economic impossibility. Instead of requiring 100 Mg of CaCO_3 per hectare, a range of 5 to 10 Mg per hectare is generally sufficient to correct the problem, and this treatment will last for several years and will produce crop yield responses that will pay for the treatment.

THE PRESENCE OF HYDROXY-AL

At pH 5.5 and above, there is no longer any significant trivalent, exchangeable Al.^[3] The chemistry is dominated by complex, polymerized hydroxy-Al of the general composition:



These polymers are not exchangeable from either clay minerals or organic matter^[4] so that they are reactive only with the H⁺ ions formed by nitrogen fertilizer (or acid rain) or OH⁻ ions that come from the application of lime. The adsorption of H⁺ tends to reduce the OH⁻ ratio, and the OH⁻ tends to raise it—to a maximum of three, where the charge becomes zero.

Because of these characteristics, hydroxy-Al and, in some soils, hydroxy-Fe form the so-called pH-dependent charge in soils. In the case of clay minerals, the hydroxy-Al polymers lose positive charge by gaining hydroxyls. In the case of organic matter, the hydroxy-Al polymers block the strongest carboxylic groups effectively reducing the ionization of the remaining carboxylic groups at a given pH.^[5,6] Thus, in both mineral and organic fractions, the effect of the hydroxy-Al polymers is to make the soil an apparently weaker acid that greatly increases the buffer capacity.

Although hydroxy-Al is not directly toxic to plants as trivalent-Al is, it does have a negative effect of short-term phosphorus (P) availability because it represents a sink for soluble P. Whether this has a practical effect on long-term P availability is not at all clear because Al phosphate is surely one of the dependable sources of P in acid soils.^[7]

NEUTRAL SOIL: pH 6 TO 7

A soil with pH 6 to 7 is free of trivalent exchangeable Al, has variable levels of non-exchangeable hydroxy-Al, and has calcium (Ca) as the dominant exchangeable cation. In addition, there are no elements such as manganese present in toxic quantities. On the other hand, manganese, zinc, boron, and Fe are usually available in amounts adequate for good crop growth. The pH range of 6 to 7 increases molybdenum availability, which, in turn, favors crops that depend on nitrogen fixation, such as soybeans and alfalfa. This pH also favors mineralization of organic nitrogen with the eventual formation of nitrate nitrogen.

Hence, the pH range of 6 to 7 generally indicates optimum soil conditions for crop growth, with the fewest negative consequences.

THE PRESENCE OF CaCO₃

Passing to alkaline soils, the presence of excess CaCO₃ in the soil is diagnostic in the pH range of 7.6 to 8.3, depending on the partial pressure of carbon dioxide (CO₂). In this pH range, a fizz test with hydrogen chloride will reveal the presence of CaCO₃, which shows there is no need to worry about the use of lime because soil acidity will not develop. In addition, soil structure usually is superior because Ca so dominates the system. On the other hand, the availability of certain elements

such as zinc, manganese, and Fe can be reduced enough to cause deficiency problems. On the whole, a calcareous soil is favorable, but some problems can arise. Knowing whether a soil is calcareous or not can be very important in a landscape that has both calcareous and non-calcareous soils, a common occurrence. Treatments for soils and crop variety response will vary with the presence or absence of CaCO₃ in the soil.

THE PRESENCE OF SODIUM CARBONATE (Na₂CO₃)

As pH values rise above pH 8.3, the cause is invariably the presence of Na₂CO₃. As the problem worsens, the pH can rise to 9 and even 10. In this pH range, most exchangeable Ca is precipitated as CaCO₃, leaving the soils dominated by exchangeable sodium. Soil structure gradually deteriorates because both clay minerals and organic matter are dispersed. As a rule, water infiltration is greatly reduced as is internal drainage in the soil. The result is that dry soils stay dry and wet soils stay wet so that crop growth is strongly inhibited and soil productivity declines radically. Rooting is poor, and as an additional restrictive factor, Fe deficiency is common.

Because this transformation to sodium-dominated soils does not occur uniformly, the usual situation is a normal landscape interspersed with sodium-affected circles. Crop growth is so much inferior within these circular zones as compared to growth in the normal soil areas that it is quite easy to determine the proportion of the land that is strongly affected by sodium. It is very difficult to reverse the soil degradation and make these circular areas productive again. This often results in abandonment of the fields because of poor crop production. Probably, the exception to this is when there are very high value crops such as citrus where treatment with H₂SO₄ and/or gypsum can recover production with time and expense.

A summary of the foregoing discussion on the significance of soil pH is given in Table 1. These values are extremely important in field diagnosis; a table such as this carried in the clipboard or in the head is of constant value in making recommendations.

Table 1 Diagnostic pH values and the dominant reaction at each one.

pH	Dominant reaction
2–3	Free mineral acid, usually H ₂ SO ₄
4–5	Presence of trivalent, exchangeable Al
≥5.5	Presence of hydroxy-Al
≥6.0–6.7	An ideal pH for many crops
7.6–8.3	Presence of CaCO ₃
9–10	Presence of Na ₂ CO ₃

STRENGTH OF SOIL ACIDITY

As was shown clearly in 1931,^[8] there is a wide difference in the strength of soil acidity from one soil to the other. Mehlich^[9] showed similar results and attempted to relate the strength of soil acidity to the clay mineralogy present. Briefly, he showed that montmorillonite was strongly acid, whereas kaolinite was much weaker.

Rich^[10] showed that interlayers of hydroxy-Al greatly reduced apparent acid strength when added to clay minerals. Coleman and Thomas^[11] showed the effect of adding both hydroxy-Fe and hydroxyl-Al to montmorillonite. Whereas pure montmorillonite had strongly acid titration curves, the addition of hydroxyl-Fe or hydroxyl-Al compounds to the clay gave titration curves of very weak acids (Fig. 1).

Both hydroxy Fe and Al compounds act as hydrogen ion sinks, which give exactly the same effect as if the acid were very slightly disassociated. They also act as OH⁻ sinks, which makes the pH very difficult to raise. The positive side of this high buffering capacity is that pH remains quite stable even with the use of high nitrogen fertilizer rates or with a degree of overliming.

ORGANIC MATTER

In the case of organic matter, the differences already mentioned in clay minerals—hydroxyl-Fe or hydroxyl-Al—are

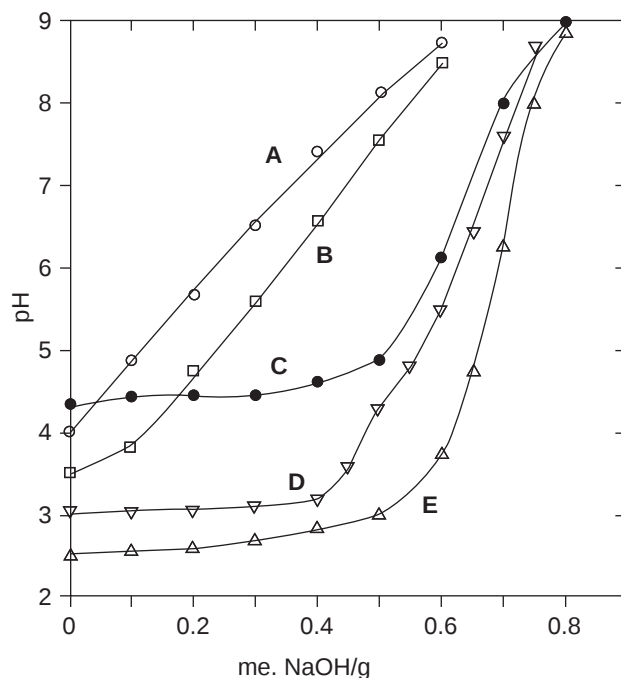


Fig. 1 Buffer curves (in 1 N KCl) for hydroxy-Al and hydroxy-Fe complexes with montmorillonite and for Al-, Fe-, and H-montmorillonites: A. hydroxy-Al complex; B. hydroxy-Fe³⁺ complex; C. Al-clays; D. Fe³⁺-clay; E. H-clay.

Source: From Coleman & Thomas.^[11]

also observed. The strength of carboxylic acids in organic matter should be in the order of pKa~4, where, in fact, most soil organic matter has an apparent pKa of around 6. Martin and Reeve^[5] showed why this was so. Organic matter contained Fe and Al compounds, and as they were removed, the pKa of organic matter approached the expected pKa of 4. Schnitzer and Skinner^[12] demonstrated how common the interaction between hydroxy Fe and Al and organic matter really was.^[6] Fig. 2 shows how strongly the adsorption of Al by organic matter affects apparent acid strength. The exchangeable Al in organic matter is almost insignificant. Instead, Al becomes hydroxylated and virtually non-exchangeable. At pH values of ≥4, the average composition of Al adsorbed is Al(OH)₂⁺.

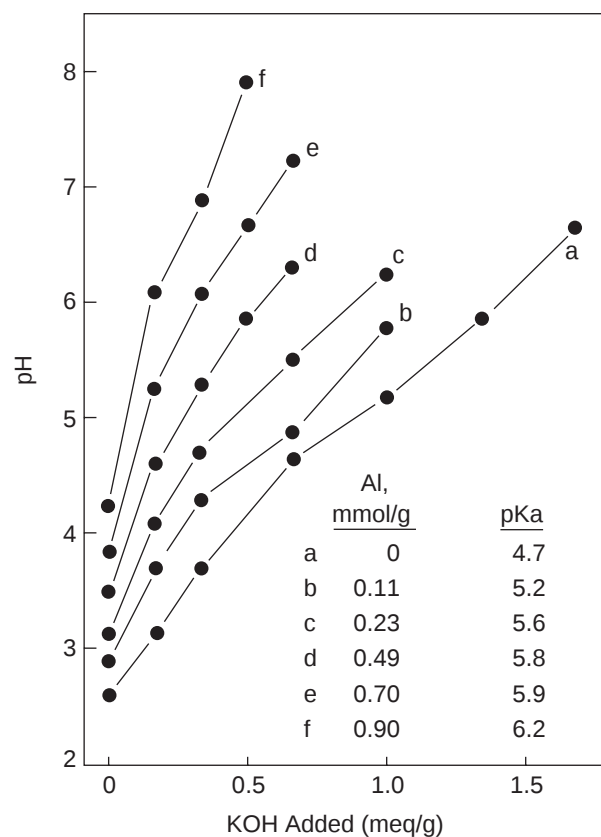


Fig. 2 The effect of adsorbed Al on Michigan Peat titration curves and on pKa values.

Source: From Hargrove & Thomas.^[6]

Table 2 The effect of organic matter on exchangeable Al in montmorillonite + sand at given pH values.

pH	Exchangeable Al cmol/kg (+)	
	No organic matter	10% organic matter
4.0	1.70	0.25
4.5	1.20	0.05
5.0	0.80	0.02

The fact that organic matter essentially immobilizes exchangeable Al helps explain why it can be so beneficial in acid soils. In two artificial soils made of sand and montmorillonite and with 10% organic matter in one, the amount of exchangeable Al was greatly affected by organic matter, as shown in Table 2.^[6]

OTHER FACTORS THAT AFFECT SOIL pH

Salt Content

The salt content of the soil–water slurry greatly affects soil pH. Salt content tends to be higher in soils that have been fertilized heavily, soils that have been irrigated with fairly high salt water, and samples that have been taken in the fall vs. those taken in the spring.^[13] Increased salt content tends to give lower pH values in temperate zone soils. This pH variation can be from one- or two-tenths, up to perhaps a half unit. In tropical soils, this same salt effect can be exactly the opposite, causing a rise in pH of about the same amount.^[14]

This salt problem has led many researchers to add salts ranging from 0.001 to 1 M to soil–water slurries in an attempt to “standardize” the pH values. The real effect is that the pH values all change, at the highest salt concentration up to 2 units^[15] with the result that there is another set of relationships to record mentally for use in interpretations. In practical terms, adding salt is not generally worth the trouble. It certainly is true that some extreme conditions will give pH values that cause some misinterpretations. But, the vast majority are good enough without adding salt to make reasonable deductions and recommendations.

Dilution

Another factor that can affect pH is the soil/water ratio in the slurry. Diluting soils by 10 times^[16] will raise pH by about 0.4 unit. Since most slurries range from 1:1 to 1:2.5, the change in pH is only about 0.1 unit, which is not a big problem.

Carbon Dioxide

CO₂ content of calcareous soils can affect the pH greatly, pH 8.3 at the atmospheric partial pressure (0.0003 atm) to 6.2 at 1.0 atm. Because soil levels of CO₂ are almost always higher than those in the aboveground atmosphere, there is a tendency for the following reaction to move toward the right.



On dried and ground soil, the effect should be small, but on pH measurements done in the field, this effect can be quite important.

Suspension Effect

In most soils, there will be a reduction in pH when the electrodes are placed in the suspension as compared to the supernatant solution. (In tropical soils, the effect may be exactly the reverse.) This suspension effect has been interpreted in two ways. First, according to Marshall,^[17] the higher activity of H⁺ ions near the clay surface causes the lowest pH. Second, according to Coleman et al.^[18] the suspension effect is mostly from the electrical charge on the soil, which differentially affects the mobilities of K⁺ and Cl[−] in the calomel electrode, not the glass electrode. In a strongly negatively charged soil, the mobility of K⁺ would be high and that of Cl[−] low, while in a positively charged tropical soil the opposite would occur.

It is generally accepted that the suspension effect is largely spurious.^[19] Nevertheless, there is evidence that H⁺ activity is somewhat higher at the clay surface.^[20]

CONCLUSION

Soil pH is one of the most useful, economic, and rapid measurements that can be made on soils. Remembering the reasons for unusually low and high pH values, it also interprets well the intermediate values rather precisely. Because of this, a rapid soil pH value can be one of the most useful measurements taken on the soil, whether the news is favorable or bad.

REFERENCES

1. Miller, W.L.; Godfrey, C.L.; McCully, W.G.; Thomas, G.W. Formation of soil acidity in carbonaceous soil materials exposed by highway excavations in East Texas. *Soil Sci.* **1976**, *121*, 162–169.
2. Frink, C.R.; Peech, M. Hydrolysis and exchange reactions of the aluminum ion in hectorite and montmorillonite suspensions. *Soil Sci. Soc. Am. Proc.* **1963**, *27*, 527–530.
3. Thomas, G.W.; Hargrove, W.L. The chemistry of soil acidity. In *Soil Acidity and Liming*, 2nd Ed.; Adams, F., Ed.; Agron. Monogr. 12; ASA and SSSA: Madison, 1984; 3–56.
4. Thomas, G.W. Beyond exchangeable aluminum: Another ride on the merry-go-round. *Commun. Soil Sci. Plan.* **1988**, *19* (7–12), 833–856.
5. Martin, A.E.; Reeve, R. Chemical studies of podzolic illuvial horizons. III. Titration curves of organic matter suspensions. *J. Soil Sci.* **1958**, *9*, 89–100.
6. Hargrove, W.L.; Thomas, G.W. Effect of organic matter on exchangeable aluminum and plant growth in acid soils. In *Chemistry in the Soil Environment*; Dowdy, R.H., Ryan, J.A., Volk, V.V., Baker, D.E., Eds.; ASA and SSSA: Madison, 1981; 151–166.

7. Chu, C.R.; Moschler, W.W.; Thomas, G.W. Rock phosphate transformation in acid soils. *Soil Sci. Soc. Am. Proc.* **1962**, *26*, 476–478.
8. Pierre, W.H.; Scarseth, G.D. Determination of the percentage base saturation of soils and its value in different soils at definite pH values. *Soil Sci.* **1931**, *31*, 99–114.
9. Mehlich, A. The significance of percentage base saturation and pH in relation to soil colloids. *Soil Sci. Soc. Am. Proc.* **1943**, *7*, 167–174.
10. Rich, C.I. Aluminum in interlayers of vermiculite. *Soil Sci. Soc. Am. Proc.* **1960**, *24*, 26–32.
11. Coleman, N.T.; Thomas, G.W. Buffer curves of acid clays as affected by the presence of ferric iron and aluminum. *Soil Sci. Soc. Am. Proc.* **1964**, *28*, 187–190.
12. Schnitzer, M.; Skinner, S.I.M. Organo-Metallic interactions in soils. 3. Properties of iron- and aluminum-organic matter complexes, prepared in the laboratory and extracted from a soil. *Soil Sci.* **1964**, *98*, 197–203.
13. Baver, L.D. Factors affecting the H^+ -ion concentration of soils. *Soil Sci.* **1927**, *23*, 399–414.
14. Van Raij, B.; Peech, M. Electrochemical properties of some oxisols and alfisols of the tropics. *Soil Sci. Soc. Am. Proc.* **1972**, *36*, 587–593.
15. Okusami, T.A.; Rust, R.H.; Juo, A.S.R. Reactive characteristics of certain soils from South Nigeria. *Soil Sci. Soc. Am. J.* **1987**, *51*, 1256–1262.
16. Davis, L.E. Measurements of pH with the glass electrode as affected by soil moisture. *Soil Sci.* **1943**, *56*, 405–422.
17. Marshall, C.E. *The Physical Chemistry and Mineralogy of Soils*; John Wiley & Sons, Inc.: New York, 1964; Vol. I.
18. Coleman, N.T.; Williams, D.E.; Nielsen, T.R.; Jenny, H. On the validity of interpretations of potentiometrically measured soil pH. *Soil Sci. Soc. Am. Proc.* **1951**, *15*, 106–114.
19. Olsen, R.A.; Robbins, J.E. The cause of the suspension effect in resin-water systems. *Soil Sci. Soc. Am. Proc.* **1971**, *35*, 260–265.
20. Swoboda, A.R.; Kunze, G.W. Reactivity of montmorillonite surface with weak organic bases. *Soil Sci. Soc. Am. Proc.* **1968**, *32*, 806–811.

Phosphorus

John Ryan

International Center for Agricultural Research in the Dry Areas (ICARDA), Aleppo, Syria

Abdul Rashid

Pakistan Atomic Energy Commission, Islamabad, Pakistan

Abstract

Though we have entered the 21st century, the problems associated with soil phosphorus (P) chemistry still abound. In developing, resource-poor countries, soil P deficiency is as severe a constraint as ever, especially in acid sandy soils and in large areas of Africa, where P fertilizer use is minimal. In such conditions, fertilizer use can potentially have a dramatic effect on crops. However, the main constraint is fertilizer availability on the local market and, when available, being at an affordable price. Factors such as remoteness from ports, inadequate transport networks, poorly developed distribution systems, absence of credit, weak or non-existent extension organizations, poor product prices, and vagaries of weather are some of the main factors militating against fertilizer use in many countries, especially Africa. The inevitable consequence is poor crop yields and poverty and hunger.

INTRODUCTION

Of all the nutrient elements that support life on this earth, phosphorus (P) is the most vital. As one of the three major elements in plant nutrition along with nitrogen (N) and potassium (K), P occurs in the least amount in soils and is the most growth-limiting factor. The growing plant derives this vital nutrient from the soil itself, which is dependent on P concentration in the rocks from which the soil was formed,^[1] or mainly from fertilizers in the case of most commercially grown crops.

Soil P deficiency, on a global basis, has negative implications for crop yields and the entire food chain. Under natural conditions, weathering and dissolution of rocks and relatively insoluble P-containing minerals is a slow process and is only capable of supporting slow-growing vegetation and crops adapted to low soil P availability. The widespread use of fertilizers in the developed world since mid-20th century or so has largely eliminated serious cases of P deficiency in crops, animals, and humans and has been a major factor contributing to increase in food production to feed the world's expanding population.

Few areas of soil chemistry–plant nutrition are as tantalizing as P research. Broad reviews documenting the state of P research, highlighting knowledge gained and challenges ahead, serve as building blocks for what we know.^[1,2] The phenomenon of soluble P reacting with soil constituents, i.e., adsorption/precipitation, results in only a small proportion of applied P (5–20%) being utilized by the first year's crop, and decreasing amounts thereafter, thus

giving rise to the notion of P “fixation.” The review of Larsen^[3] focused on unfertilized soils, considering in detail the equilibria between solution P and various solid phases. A major milestone was “The Role of Phosphorus in Agriculture,”^[1] having authoritative chapters on various aspects of the soil–plant P continuum (Fig. 1).^[4–6]

At the global level, fertilizer use mirrors the country's overall state of economic development. In developed, intensive-agriculture countries, many soils have high P levels due to continuous P fertilizer use. Consequently, there is frequently little or no response to P, and indeed soluble P may be lost from agricultural land, causing eutrophication in lakes and rivers.^[7] Despite this shift in emphasis, the intractable nature of P is an issue in developing and developed countries alike as efforts are made to get the most efficient use of this essential nutrient, with a balance of production and environmental concerns.

This brief overview broadly looks at the equilibria of P fractions or phases in soils; how fertilizer P reacts in the soil from the short- and long-term perspective; the significance of P and its absorption by plants; and how chemical reactions ultimately impact on crop growth and influence the environment—in short, the dynamic nature of P in soils and plants.

SOIL P EQUILIBRIA

Although the total amount of P in soils is 0.1–0.3%, it is chemically variable; the largest fraction is inorganic,



Fig. 1 Crop response to P fertilizer in the field.

with a smaller organic fraction, the relative distribution between the two fractions being dependent on the particular environment and soil type.^[3] In acid soils, various amorphous and crystalline forms of iron (Fe) and aluminum (Al) oxides dominate, while in calcareous soils P is found mainly in the form of calcium (Ca) compounds of varying solubility. However, as there is little or no relationship between total P and the relatively tiny fraction that is plant available (less than 1%), the main focus as far as crop production is concerned is the equilibria or dynamic changes in P solubility that influence crop nutrition.

The immediate source of P for growing plants is the soil solution, which is very low in P, i.e., a few mg kg^{-1} or even less, i.e., intensity factor. Labile P refers to a more sizeable fraction (capacity) and the rate that is capable of replenishing the tiny pool of solution P during plant uptake, i.e., buffering capacity. Within any one crop season, this exchange takes place many times as P is removed from the solution phase, and a new equilibrium is continually

re-established, albeit at a lower level. When root absorption is rapid during active growth, solution P levels may drop to such an extent as to limit growth. Similarly, slow release rates from the labile phase may limit growth. The labile phase represents a suite of soil P compounds of varying degrees of solubility. The non-labile P is that fraction of total P, often the largest, that is inert and has no short-term influence on labile or solution P.

The equilibria among the three P phases of varying solubility under natural conditions are disturbed mainly by plant P uptake.^[3] Fertilization raises solution P levels initially, but this is subsequently followed by a slow, inexorable shift in the direction of less-soluble P compounds. Thus, Johnston^[4] illustrated an agronomically embracing P cycle, including an expanded concept of “P pools” representing a continuum of availability, crop P uptake and removal, and the potential for P loss through leaching and erosion (Fig. 2). This raises the issue of the reactions that occur when soluble P fertilizers are applied to soils.

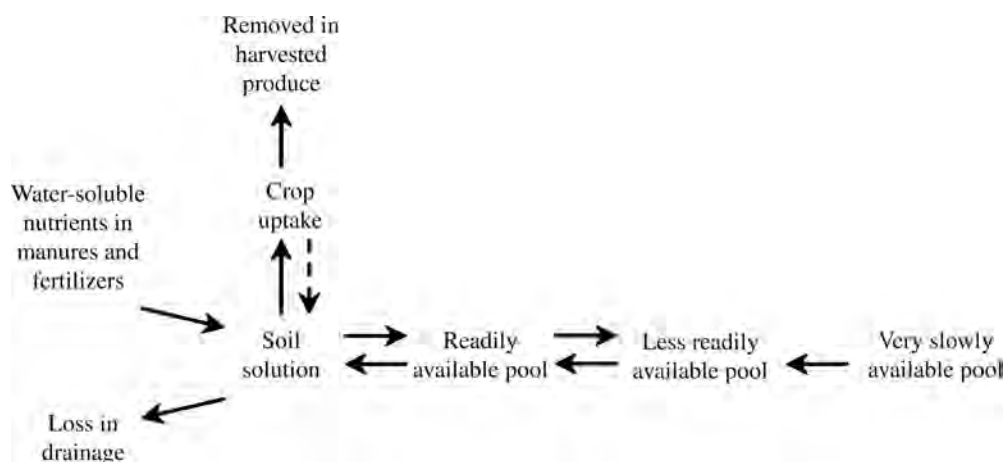


Fig. 2 Schematic representation of the soil-P cycle under fertilized conditions.
Source: From Johnston.^[4]

SOIL-FERTILIZER P REACTIONS

The reactions of soluble P fertilizer with soil constituents causing reduced P solubility and availability can be categorized as short-term and long-term ones;^[8] a reversal of this process, desorption, can also occur.

Initial Rapid Reactions

Early research on removal of soluble P from soil solution identified two mechanisms: precipitation, based on solubility products of soil-P minerals, readily occurs with cations such as Ca, Al, and Fe in the soil solution and adsorption occurs on solid-phase surfaces. The understanding envisages more refined mechanisms for P retention, including physical adsorption, chemisorption, anion exchange, surface precipitation, and precipitation of solid phases. Soil P reactions are usually a matter of minutes to several hours depending on soil constituents.^[9] Sorbed P can vary by several orders of magnitude depending on soil type. These reactions can be described by short-duration soil: soluble P equilibration studies using adsorption isotherms, e.g., Langmuir, Freundlich, etc. In practice, it is difficult to partition the different soil P reactions; the process is a continuous one—rapid at first, but continuing slowly to a more stable hypothetical equilibrium state whereby metastable compounds revert to more stable ones. In short, more and more of the plant-available P gets converted to less available or insoluble forms, if not taken up by the plant.

Slow Reaction Phase

Although initial reactions may indicate the magnitude of long-term P availability, they may not always dictate the ultimate direction. Studies of longer duration range from incubation experiments of several months^[10] to studies of a few years or more.^[11,12] Barrow^[13] gave an overview of the transformation process of initial metastable P to less available forms. The changes that occur within months to years can be monitored by crop growth or uptake responses or by changes in availability indices or by relative changes in soil chemical fractions;^[12] when considered together, these approaches clearly reflect the slow transition to insoluble or inert P compounds. The actual mechanisms whereby P reacts under prolonged P:soil contact are unclear; some arguments include migration of adsorbed P from original sites, occlusion of adsorbed P, and slow crystal formation of precipitated P compounds. The reversion process is influenced by soil moisture and aeration, fertilizer mixing or incorporation in the soil, fertilizer form (granular or powder) as well as mycorrhizae, and other modifying influences of the growing plant.

Under field conditions, the general reversion or transformation of P to less soluble forms can be influenced by leaching, especially in coarse-textured soils,^[7] and assimilation of soluble/available P in the soil microbial biomass.

Conversely, mineralization of organic P sources can contribute to the available P pool. Tillage or the extent of soil disturbance can also influence residual P. Given the many and varied influences on the fate of residual P, loss of P-fertilizing effectiveness tends to decrease with time. Barrow^[13] suggested that geometric progression may describe the rate of change of effectiveness over a period of a few years but may not do so within a larger timescale.

Desorption

Any discussion on P dynamism in soils has to consider the possible reversal of the normal process, i.e., the extent to which non-labile or sorbed P becomes labile or available P. While sorption quantity–intensity (Q/I) relationships are used to describe P availability in soils, desorption curves reflect P release from soil to plants.^[14] Evidence from exhaustive cropping and successive chemical desorption studies suggest that this process can occur, albeit very slowly. Nevertheless, the relevance of such desorption studies to actual field conditions is tenuous at best given the disparity between the two sets of conditions. The well-documented adsorption–desorption phenomenon observed in normal aerated soils does not pertain to flooded rice soils.

Soil P Availability

Notwithstanding the complexity of soil P reactions and the resulting equilibria, the critical issue for the growing plant is that the soluble P pool is adequately maintained by replenishment from the adsorbed or less readily available pool, i.e., “available” P fraction. In order to assess a soil’s adequacy for crop growth and determine fertilization needs, various chemical indices have been developed to reflect the “available P pool;” these include reagents that range from water to weak acids and bases as well as more methods such as ion exchange resins and Fe-impregnated paper strips that purportedly do not “dissolve” insoluble P fractions. Some of the most common test include Mehlich 1, Bray 1, and Oslen. The emphasis has been placed on multi-nutrient extractants. However, regardless of the method or availability index used, only the growing plant can accurately identify this fraction. Soil P availability in relation to the roots, i.e., “positional availability,” and limited P movement (diffusion) in soils are important considerations for plant growth.

The ideal test or availability index should identify a critical level of soil P beyond which no fertilizer P is needed and should also identify a P-response range. The soils that test very low would require the largest amount of fertilizer P, and decreased P rate as the test level increases to the “satisfactory” level, and no P beyond that point. Whether a fertilizer strategy of buildup or maintenance is feasible depends on economic considerations. A crucial concern for soil P testing in countries where fertilizer P is extensively used or over-used, e.g., United States and Western Europe,

is to identify the threshold value for soil P that is likely to indicate release or loss of soluble P from the soil, with potential to cause contamination of water sources.

P Nutrition of Plants

P, essential for many physiological processes, especially energy functions, is also a major component of numerous plant structural compounds. Some generalizations on soil-plant P relations follow.^[15]

Absorption and Translocation

P is absorbed by plant roots mainly as dihydrogen phosphate ion ($\text{H}_2\text{PO}_4^{2-}$), using metabolic energy. Crop genotypes vary in their ability to absorb soil P and respond to fertilizer P. Plant factors related to these differences include root development (length and geometry), mycorrhizal infection, and the plant's internal P requirement. Thus, differences in P uptake are associated primarily with root characteristics. Conversely, root proliferation is influenced by the P level in various soil profile layers. Growing plants store very little of the P taken up by the roots, so there must be an ample supply in the soil.^[4]

Mycorrhizae increase the efficiency with which crops extract P from soils and thus effectively decrease the external P requirement of many crop species. Legumes are more mycorrhizal than cereals, and hence are relatively tolerant to P deficiency. Notwithstanding the potential benefits of mycorrhizae in relation to crop P nutrition, the intractable problem of introducing mycorrhizal fungi into non-sterile field conditions remains. In short, it is difficult to manipulate these fungal-plant associations, which occur naturally, for practical economic benefit. However, these associations of soil organisms do help to explain differences in P uptake between plant species.

Most P ions entering the root are taken up by the root hairs or outermost cell layers; accumulation in root cell vacuoles competes with the processes leading to accumulation in the xylem and subsequent transport to the plant top. Phosphate passing radially across the root to the endodermis may move in the transpiration stream through the "outer free space" in the epidermal and cortical cell walls and the intercellular moisture films. Phosphate taken into the symplasm moves by diffusion and protoplasmic streaming.

Once absorbed, P is rapidly translocated to all plant parts, upward in the xylem and downward in the phloem. With soil P deficiency, the roots retain a large part of the absorbed P, and needed P for inflorescence development comes mainly from the stem and leaves by retranslocation. With an adequate P supply, the roots retain a smaller proportion of the total, but again most of the P in the stems and leaves is retranslocated to the developing grain heads. In early vegetative growth, the leaves accumulate the major part of the absorbed P, reaching relatively high concentrations. At inflorescence, leaf P accumulation

decreases, and P translocation from the roots may exceed the absorption rate.

Deficiency Symptoms

Plants unable to get adequate P exhibit stunted growth. As P is mobile in the plant, deficiency symptoms first appear on older leaves. In cereals, i.e., wheat and barley, dark green leaves are typical symptoms. In corn, the leaves and lower parts of the stem become purple or reddish. Deficiency affects not only plant growth but eventually decreases the development of seeds and fruits and invariably delays crop maturity.

P Requirement of Plants

Crop species differ in their external (soil solution P concentration), as well as internal, P requirement (plant tissue P concentration) for optimum growth. Lettuce (non-mycorrhizal) requires $0.3 \text{ mg P kg}^{-21}$ in the soil solution, soybean and tomato $0.2 \text{ mg P kg}^{-21}$, corn and sorghum $0.05\text{--}0.06 \text{ mg P kg}^{-21}$, and cassava (highly mycorrhizal) can grow optimally at $0.005 \text{ mg P kg}^{-21}$. Although the magnitude of these differences may seem small, it dramatically affects the amount of P required to attain the soil P concentrations to optimum levels.

P concentration in plant tissues may vary from 0.1% to 1.0%. Critical concentration of nine tropical grasses, nine tropical pasture legumes, and alfalfa fall within the range of 0.16–0.25% P. Values for individual species vary. The generalized critical level for young cereal plants (wheat, barley, and oats) at head emergence is 0.20% P.^[16] P concentration in leaves is greater than whole shoots; critical P concentration for youngest mature rice leaves is 0.33% at mid-tillering and 0.26% at maximum tillering and panicle differentiation. Critical concentration range in ear leaves of corn at tasseling is 0.23–0.25%. Such indices serve as guidelines for assessing P nutrition and serving as a basis for fertilizer P use in crops.

CONCLUSION

As with most chemically complex elements, P exists in a variety of mineralogical species and solubility phases in soils and is an essential part of all plant life. Like N, the dynamic nature of P constitutes a "P cycle", with P being removed in crop offtake and returned to the soil as residues, manures, and chemical fertilizers, primarily derived from phosphate rock. At the chemical level in the soil, the dynamic processes include dissolution, precipitation, adsorption, and desorption, while biological processes are mainly immobilization within the plant roots and bacterial biomass and subsequent mineralization or decomposition.

While there are gaps in our knowledge of soil, fertilizer and plant P at the basic science level—and they may always

persist—much is known from the applied standpoint. We now have a balanced view of the issues at stake for agriculture, resource conservation, and the environment. Indeed, as Johnston^[4] pointed out, “improving the efficiency of P in agriculture and minimizing its loss from soil to water has much to offer not only to farmers, but to society as a whole.” A number of key questions—now largely answered—provide us with the basis of a strategy of integrated plant nutrient management. Knowing the levels of available soil P for different crops and crop-management systems and having at one’s disposal soil and plant tests, which help in such diagnosis, are prerequisites to provide a P balance in nature. Such an approach will ensure high-P use efficiency, both from the soil and fertilizer (organic or inorganic) standpoint, and at the same time avoid damage to our aquatic environment and its attendant health consequences.

REFERENCES

1. Khasawneh, F.E.; Sample, E.C.; Kamprath, E.J., Eds. *The Role of Phosphorus in Agriculture*; ASA, CSSA, SSSA: Madison, 1980.
2. Wild, A. The retention of phosphate by soils. *J. Soil Sci.* **1949**, *1*, 221–238.
3. Larsen, S. Soil phosphorus. *Adv. Agron.* **1967**, *19*, 151–210.
4. Johnston, A.E. *Soil and Plant Phosphate*; International Fertilizer Industry Association: Paris, 2000.
5. Matar, A.; Torrent, J.; Ryan, J. Soil and fertilizer Phosphorus and crop responses in the dryland Mediterranean zone. *Adv. Soil Sci.* **1992**, *18*, 82–146.
6. Buresh, R.J.; Sanchez, P.A.; Calhaun, F., Eds. *Replenishing Soil Fertility in Africa*; SSSA: Madison, 1997.
7. Tunney, H.; Carton, O.T.; Brookes, P.C.; Johnson, A.E. *Phosphorus Losses from Soil to Water*; CAB International: Wallingford, 1997.
8. Sample, E.C.; Soper, R.J.; Racz, G.J. Reactions of phosphate fertilizers in soils. In *The Role of Phosphorus in Agriculture*; Khasawneh, F.E., Sample, E.C., Kamprath, E.J., Eds.; ASA, CSSA, SSSA: Madison, 1980; 263–310.
9. Ryan, J.; Curtin, D.; Cheema, M.A. Significance of Iron oxides and calcium carbonate particle size in phosphorus sorption and desorption in calcareous soils. *Soil Sci. Soc. Am. J.* **1985**, *49*, 74–76.
10. Ryan, J.; Hasan, H.; Baasiri, M.; Tabbara, H.S. Availability and transformation of applied phosphorus with time in calcareous lebanese soils. *Soil Sci. Soc. Am. J.* **1985**, *49*, 1215–1220.
11. Devine, J.R.; Gunary, D.; Larsen, S. Availability of phosphate as affected by duration of fertilizer contact with soil. *J. Agr. Sci.* **1968**, *71*, 359–364.
12. Hooker, M.L.; Peterson, G.A.; Sander, D.H.; Daigger, L.A. Phosphate fractions in calcareous soils as altered by time and amounts of added phosphate. *Soil Sci. Soc. Am. J.* **1980**, *44*, 269–277.
13. Barrow, N.J. Evaluation and utilization of residual phosphorus in soils. In *The Role of Phosphorus in Agriculture*; Khasawneh, F.E., Sample, E.C., Kamprath, E.J., Eds.; ASA, CSSA, SSSA: Madison, 1980; 333–359.
14. Raven, K.P.; Hossner, L.R. Phosphorus desorption quantity–intensity relationships in soils. *Soil Sci. Soc. Am. J.* **1993**, *57*, 1501–1508.
15. Ozanne, P.G. Phosphate nutrition of plants—A general treatise. In *The Role of Phosphorus in Agriculture*; Khasawneh, F.E., Sample, E.C., Kamprath, E.J., Eds.; Soil Science Society of America: Madison, 1980, 559–589.
16. Walsh, L.M.; Beaton, J.D. *Soil Testing and Plant Analysis*; Soil Science Society of America: Madison, 1973.

Phosphorus: Manure and Diet Modification

Rory O. Maguire

Virginia Polytechnic Institute and State University, Blacksburg, Virginia, U.S.A.

John T. Brake

Department of Soil Science, North Carolina State University, Raleigh, North Carolina, U.S.A.

Peter W. Plumstead

North Carolina State University, Raleigh, North Carolina, U.S.A.

Abstract

Since the earliest development of agriculture, animal manures have been applied to cropland to supply essential nutrients and increase crop yield. Modernization of agriculture has been typified by greater yields of both crop and animal products from smaller land areas and a greater emphasis on external inputs such as chemical fertilizers and concentrated animal feeds. Intensification and confinement of animal production have led to a situation in some areas where there are more nutrients available in animal manure than what the local crops require. These excesses of manure have raised concerns about the fate of the nutrients they contain, as losses of nutrients such as nitrogen and phosphorus (P) from agricultural land can cause problems such as eutrophication in rivers and lakes. To address these regional excesses of P, research efforts have been focused on reducing the P concentrations in diets fed to animals, as well as on improving the availability thereof, both of which result in a reduction in the P concentration in manure generated. These dietary alterations affect not only the concentrations of P in manures produced but also the forms of P, their fertilizer value, and environmental impact.

PROGRESS IN NUTRIENT MANAGEMENT PRACTICES

With increasing human population and awareness of environmental problems associated with pollution of rivers and streams with P, the end of the 20th century saw a refocusing of nutrient management strategies from increasing production toward environmental protection, in developed countries.^[1] Initially, nutrient management strategies were aimed at avoiding overapplication of nutrients, and voluntary best management practices were developed. However, manure tended to be applied at a rate to supply crops with sufficient nitrogen (N), which typically leads to overapplication of phosphorous (P), as manures have N:P ratios of around 2:1 to 4:1 (depending on species) while crop requirements have an N:P ratio of 8:1. When P is overapplied to soils, it builds up the soil test P (STP) concentration and over a period of many years, this leads to STP rising to levels in excess of crop requirement.^[2] STP in excess of crop requirement is undesirable, as studies have shown that when STP rises above agronomically optimum values P losses to rivers and streams increase to the detriment of water quality,^[2] and it can also lead to a negative effect on some micro-nutrients, e.g., zinc.

Nutritional Strategies to Reduce the Manure Phosphorus Concentration

In the past, dietary P levels frequently included a safety margin that allows for any potential variation in both the requirement of the animals and the degree of dietary P utilization, with little attention paid to the resultant manure P concentration. However, research in broilers and broiler breeders demonstrates that levels of inorganic P, supplemented to both broiler and broiler breeder diets, may be considerably reduced relative to minimum requirements published by the National Research Council without an adverse effect on performance.^[3,4]

When considering strategies to increase P digestion and retention, species can be split into two categories according to their ability to digest phytate P: ruminants (cattle and sheep) can digest phytate P, while monogastric species (poultry and swine) can digest little. As approximately 66% of the P in corn and soybean is in the form of phytic acid, and these ingredients frequently constitute about 60–90% of the diet, up to 80% of the total fecal P output may originate directly from corn and soybean meal in poultry diets.^[5] The primary strategy to address the low digestibility of phytate P in monogastric species has been the application of exogenous phytase enzymes to these

diets and is done on the basis of replacing 0.1% of available P from supplemental calcium phosphate with a more or less standard amount of enzyme. Alternate strategies other than the use of phytase include the development of low phytate variants of corn and soybean cultivars, because these possess similar, if not greater, potential in reducing manure P excretion. However, the commercial application of these new cultivars has been limited as a result of the practical complications arising from the identity preservation of these ingredients from harvest to point of feeding.

IMPACT OF DIET MODIFICATION ON NUTRIENT MANAGEMENT

The total P and soluble P concentrations in manures have been shown to be affected by dietary P concentration, P contributing ingredients, and dietary phytase supplementation. Total P is important because applications of manure total P contribute to long-term changes in STP. Once optimum STP for crop production is attained, it is beneficial to balance manure total P applications with crop removal of P. This will maintain crop production without raising STP to levels of environmental concern; although in some soils, P can revert to less available forms so P availability needs to be monitored with soil testing. Applications of soluble P in manure have been correlated to soluble P losses in runoff from manure-amended soils, so minimizing the soluble P concentration in manure is important when P losses to rivers and streams are of concern. Reducing the P concentration in animal feeds (high vs. low non-phytate P diets) and using phytase (phytase vs. no phytase diets) can reduce the total and soluble P concentrations in poultry wastes (Fig. 1A and B).^[6] It is estimated that using available technology, dietary amendment can potentially decrease the total P concentration in poultry by 40%, swine by 50%, and dairy by 30%, while advances could raise these figures to 60% for both poultry and swine.^[7] Research on the impact of dietary amendments on soluble P in manure and P losses in runoff from manure-amended soils is at an early stage; however, trends are starting to appear. Supplying P at an adequate level to satisfy animal requirements is generally achieved through a reduction in the added inorganic dietary calcium phosphate component, and results in a reduction in both total and soluble P in the manures generated. While most studies show that feed additives such as phytase reduce manure total P while having little effect on soluble P, this effect is very dependent on the level of dietary P relative to the requirements of the animals.^[6,7] Phytase supplementation to diets containing P in excess of animal requirements, or without an adequate reduction in the P from supplemental calcium phosphate, will result in a potential increase in the water soluble P when expressed on a percentage basis.^[5]

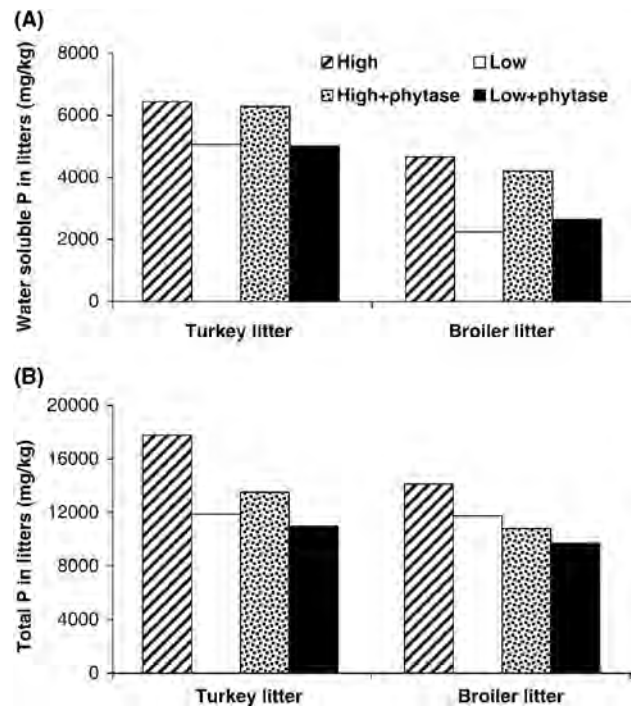


Fig. 1 Effects of diets high and low in dietary non-phytate P, and the impact of dietary phytase on (A) water soluble P in litters and (B) total P in litters produced.

Source: From Maguire, Sims, et al.^[6]

Traditionally, manure application rates to cropland have been decided by individual farmers. However, increasing concerns over the impact of the land application of nutrients in excess of crop requirement on water quality in rivers and streams have led to increasing regulation of land application of manures in many developed countries. These regulations are primarily based on N and P application rates that are often linked to problems identified in localized areas and are based on the levels of soil N and P, and the annual rate of removal of these nutrients through their incorporation into harvested crops. As forage crops typically have an N/P ratio of 8:1 and manure contains N/P ratios of approximately 2:1 to 4:1, nutrient management regulations based on P application may require up to four times more cropland than previous N-based application guidelines. The limitation of available land in areas proximate to intensive livestock production can cause problems associated with what to do with the manure if it cannot be land applied, as other uses of manure are fairly limited and transport costs limit the potential transportation of the wastes over any appreciable distance. If nutrient management regulations are based on P, then reducing the P concentration in manures could greatly alleviate these problems. For example, if one could reduce the total P concentration in a manure by 40%, then application of 40% more manure by weight to meet the same total P application limit would be possible. This may eliminate problems with

excess manure P in some areas of intensive animal production but may not be sufficient for other areas that already have high STP.

CONCLUSION

Dietary amendment to reduce the P concentration in feeds and hence manures generated will help address concerns over the fate of manure nutrients in areas of intensive animal production. If manures have the same N concentrations but reduced P concentrations, then these manures will come closer to meeting crop N needs without oversupply of P. However, dietary amendment alone will not solve the problem of excessive manure P in all areas of animal production. Dietary amendment will, however, certainly help alleviate the situation where applying animal manures to meet crop N requirements leads to overapplication of P, buildup of STP, and eventually increased losses of P to rivers and streams.

REFERENCES

1. Beegle, D.B.; Carton, O.T.; Bailey, J.S. Nutrient management planning: Justification, theory, practice. *J. Environ. Qual.* **2000**, *29*, 72–79.
2. Sims, J.T.; Edwards, A.C.; Schoumans, O.F.; Simard, R.R. Integrating soil phosphorus testing into environmentally based agricultural practices. *J. Environ. Qual.* **2000**, *29*, 60–71.
3. National Research Council. In *Nutrient Requirements of Poultry*; 9th revised; National Academy Press: Washington, 1994.
4. Brake, J.; Williams, C.V.; Lenfestey, B.A. Optimization of dietary phosphorus for broiler breeders and their progeny. In *Nutritional Biotechnology in the Feed and Food Industries*, Proceedings of Alltech 19th International Symposium, Lyons, T.P., Jacques, K.A., Eds.; Nottingham University Press: Nottingham, 2003; 77–84.
5. Brake, J.; Plumstead, P.; Gernat, A.; Maguire, R.; Kwan-yuen, P.; Burton, J. Reducing phytate versus adding phytase to reduce fecal phosphorus. In *Session PCP 3: Phytate in Soybeans and Related Environmental Concerns*, Proceedings of the 95th AOCS Meeting, Cincinnati, OH, 2004; 118 pp.
6. Maguire, R.O.; Sims, J.T.; Saylor, W.W.; Turner, B.L.; Angel, R.; Applegate, T.J. Influence of phytase addition to poultry diets on phosphorus forms and solubility in litters and amended soils. *J. Environ. Qual.* **2004**, *33*, 2306–2316.
7. CAST (Council for Agricultural Science and Technology). In *Animal Diet Modification to Decrease the Potential for Nitrogen and Phosphorus Pollution. Issue Paper No. 21*; CAST: Washington, 2002.

Phosphorus: Plant Nutrition and Microorganism Contribution

Bettina Eichler-Löbermann

Christel Baum

Institute of Land Use, University of Rostock, Rostock, Germany

Silke Ruppel

Institute of Vegetable and Ornamental Crops, Grossbeeren, Germany

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

The economical use of phosphorus (P) is a key issue for sustainable agriculture since natural P resources are limited. In this entry, the influence of microorganisms on the P nutrition of plants is described. Special attention is given to organic matter management and its effects on the microbial turnover of P in soil.

INTRODUCTION

The lack of mineral phosphorus (P) fertilizers in many regions worldwide and the increasing prices lead to an exploration for alternative strategies to improve the P supply of plants. These can be the use of organic matter and raw materials, the cultivation of P-efficient plants, and the application of microorganisms (MOs) able to mobilize P. Organic matter in the soil is one of the most important factors of increasing the productivity in plant production. It can be provided as manure, compost, organic household wastes, green fertilizer crops, or perennial fodder crops. Besides serving as a source of nutrients itself, organic fertilization can improve the availability of nutrients in soil.^[1,2] The mineralization of the organic P compounds in soil is mainly influenced by MOs. On the other hand, the organic matter in soil affects the activity of MOs.^[3,4]

Like plants, MOs are also capable of improving the availability of inorganic P compounds.^[5] This active mobilization of inorganic P sources by plants and MOs is an elemental factor to counteract the loss of P plant availability by fixation and adsorption and thus a source of energy for high P utilization. Plant growth-promoting effects of MOs were found for arable crops as well as for trees (Table 1).^[6]

In the following sections, the influence of MOs on plant P availability is described with special attention on organic matter management.

MICROBIAL P AND ORGANIC MATTER MANAGEMENT

Biological activity and diversity of MOs in soil are affected positively by the supply of organic matter such

as compost^[7] and manure.^[8] The organic matter supply also plays an important role in the successful inoculation of soil with MOs.^[9] Bünemann et al.^[10] emphasized the important role of organic substances in soil for the conversion of P through soil MOs and found that the microbial biomass is limited by carbon (C) and nitrogen (N) rather than P availability. In a laboratory experiment, the microbial P increased substantially when soil was supplied with C and N. The linkage between soil organic C and microbial P is a major reason why cropped soils usually have lower microbial P contents than forest soils or grassland.^[11]

Microbial P uptake and its subsequent release strongly affect the availability of P to plants. P uptake by MOs may be less than, equal to, or greater than P mineralization from organic material. Therefore, P in soil solution can be replenished or depleted during P mineralization, which has consequences for plant P nutrition.^[12]

The microbial turnover of P in soil was defined by Oberson and Joner^[13] as a sum of all microbial mediated transformations and related fluxes of P in soil. It can be described quantitatively by the P flux (ψ) through the soil microbial biomass as follows:

$$\psi = P_{\text{mic}}/T \quad (1)$$

where P_{mic} is microbial P pool and T the turnover time.

Rapid growth of soil microbial biomass resulted in an increase in P_{mic} and a decrease in water-extractable P.^[4] However, immobilization in MOs may retain P in a potentially plant-available form.^[10] The time between P immobilization and re-mineralization depends on substrate quality and soil properties. It was found that the P in bacterial cells, presumably present in organic compounds,

Table 1 Effects of microbial inoculum on plant growth and P uptake of *Salix caprea* × *Salix viminalis* in a pot experiment after a 6-month vegetation time.

Microbial inoculum	Shoot biomass (g DM)	P uptake into the shoot (mg)
Control (sterile inoculum)	65.0	179.6
<i>Laccaria laccata</i>	78.4 ^a	176.9
<i>Hebeloma crustuliniforme</i>	72.7 ^a	187.6 ^a
<i>Laccaria laccata</i> with the associated bacterium	70.0 ^a	178.5
<i>Orchrobactrum anthropi</i>		
<i>Hebeloma crustuliniforme</i> with the associated bacterium	107.3 ^a	220.5 ^a
<i>Sphingomonas paucimobilis</i>		

^aSignificant effect of inoculation, $p < 0.05$.

Source: From Baum, Stetter, et al.^[6]

is mineralized within a few days after addition into soil.^[14] Leitfeld, Siebert, and Kögel-Knabner^[7] found compost-derived microbial biomass as an easily degradable C source.

ROLE OF PHOSPHATASES

About 50% of the total P in agricultural soils are present in organic forms, which include a range of inositol, phosphate ester, phospholipids, and nucleic acids among others.^[15,16]

Decomposition by MOs is the dominating process for P mineralization from organic matter and is closely linked to C mineralization. Additionally, P is also released from organic matter through enzymatic hydrolysis outside the cells by phosphatase excretion, a process called biochemical P mineralization. George et al.^[17] found significantly decreased organic P concentrations in the rhizosphere when phosphohydrolase activity increased. The organic NaHCO_3^- and sodium hydroxide-extractable P fractions are supposed to be sources for plant P nutrition since Heilmann et al.^[18] found a significant positive correlation between these parameters and acid phosphatase activity.

Acid or alkaline phosphatases prevail in soils depending on the soil pH.^[19] Acid phosphatases originate from many sources, including plant root exudates and soil MOs. Alkaline phosphatases are produced by soil MOs and soil fauna, whereas higher plants are devoid of alkaline phosphatase. Organisms excrete non-specific phosphatases as well as specific phosphatases such as phytase. Investigations for the excretion of phosphatases were carried out for different MOs, such as *Aspergillus* spp.,^[16] *Penicillium* spp.^[20] (Table 2), and *Pseudomonas* spp.^[21] Unno et al.^[22] found different *Burkholderia* strains to utilize less-soluble phytates [aluminum (Al)-inositol hexaphosphate and iron (Fe)-inositol

Table 2 Activity of acid and alkaline phosphatase in soil ($\mu\text{g p-nitrophenyl g}^{-1} \text{ soil h}^{-1}$) in dependence of the cultivated crop and addition of the fungus *Penicillium bilaii* in a pot experiment (6 kg P-poor sandy loam pot^{-1}) after 7 weeks of vegetation time under greenhouse conditions.

Cultivation	Acid phosphatase		Alkaline phosphatase	
	Without <i>P. bilaii</i>	With <i>P. bilaii</i>	Without <i>P. bilaii</i>	With <i>P. bilaii</i>
Oilseed rape	227.8 ^c	236.2 ^c	33.8 ^{bc}	33.7 ^{cd}
Maize	276.7 ^d	279.0 ^e	27.0 ^b	43.6 ^{ef*}
Phacelia	122.6 ^b	70.9 ^{a*}	50.6 ^e	23.1 ^{b*}
Buckwheat	262.1 ^d	204.8 ^{b*}	33.8 ^{bc}	51.2 ^{g*}
Common ryegrass	73.0 ^a	77.8 ^a	6.8 ^a	15.2 ^{a*}
Lupin	217.1 ^c	246.6 ^{cd}	48.4 ^{de}	38.8 ^{de}
Pea	231.8 ^c	266.0 ^{de*}	41.6 ^{cd}	34.9 ^{cd*}
Serradella	302.6 ^e	256.5 ^{cde}	38.2 ^c	49.5 ^{fg*}
Amaranth	301.5 ^e	262.1 ^{cde*}	56.0 ^e	29.2 ^{bc*}
Control (without)	225.0 ^c	191.2 ^b	55.1 ^e	24.8 ^{b*}

Note: Letters (a–e) indicate significant differences between the crops. Asterick indicates a significant difference from the crop variant without fungal treatment (Duncan, $\alpha = 0.05$).

Source: From Eichler, Caus, et al.^[20]

hexaphosphate] in the rhizosphere of lupines. Hayes, Simpson, and Richardson^[23] reported that a range of plants are unable to utilize P supplied as soluble myo-inositol hexakisphosphate. Their growth was improved when either a purified phytase or a phytase-producing MO (*Aspergillus niger*) was added. The growth of maize was not improved when only calcium (Ca) phytate was added to the soil. However, plant growth increased when the phytate in combination with a phytase was added.^[24]

ACTIVE MOBILIZATION OF MINERAL P BY CROP PLANTS AND MOs

Like plants, MOs possess a range of mechanisms to mobilize or desorb poorly available inorganic P. Usually, production of organic acids and excretion of protons improve the P availability in the soil.^[25] Deubel,^[9] Ruppel,^[5] and Bünemann et al.^[10] found *Enterobacter* and *Pseudomonas* strains to dissolve poorly available mineral Fe–P, Al–P, and Ca–P compounds. These organisms secreted organic acids in dependency of the C source given.

MYCORRHIZAL FUNGI

Mycorrhizal fungi, especially arbuscular mycorrhizal fungi (AMF), are considered to be the most important

microbial symbionts for the majority of crops and have a special importance regarding the mobilization and transport of P in soils.^[26] Hyphae of AMF greatly increase the absorbing surface area of the roots, bridge the depletion zone, and grow into areas with an adequate supply of P. The activity of AMF is enhanced by organic matter,^[27] while the increase in the soil P content affects them negatively.^[28] Microbial inoculants, such as rhizobia, *Trichoderma* strains, fluorescent *Pseudomonas*, and *Enterobacter*, added to promote plant growth affect the AMF. Usually, positive synergistic effects were observed.^[29,30]

CONCLUSION AND FUTURE PROSPECTS

In this entry, the potential of different MOs for plant growth promotion and for P uptake was demonstrated. However, practical use in the temperate climate conditions is limited, and a prediction of practical success of applied MO inoculants is difficult. This has different reasons as follows: 1) The successful use of microbial inoculants depends on many environmental factors. Thus, more information is required of microbial P as a function of land use and the environmental factors; 2) caused by the complexity and the strong relation between soil MOs, inoculated MOs, and crop plants, it is very complicated to distinguish between the contribution of each complement to plant P uptake; and 3) the mechanisms for the plant growth promotion by inoculated MOs are not known in detail and are very indifferent. Positive effects on the P uptake of the plants may be based on an increased availability of nutrients in the rhizosphere, positive effects on the root growth and morphology, and the promotion of beneficial plant–microbe symbioses.

REFERENCES

1. Dorado, J.; Zancada, M.; Almendros, G.; López-Fando, C. Changes in soil properties and humic substances after long-term amendments with manure and crop residues in dryland farming systems. *J. Plant Nutr. Soil Sci.* **2003**, *166*, 31–38.
2. Hartl, W.; Erhart, E. Crop nitrogen recovery and soil nitrogen dynamics in a 10 year field experiment with biowaste compost. *J. Plant Nutr. Soil Sci.* **2005**, *168*, 781–788.
3. Frossard, E.; Skrabal, P.; Sinaj, S.; Bangerter, F.; Traoré, O. Form and exchangeability of inorganic phosphate in composted solid organic wastes. *Nutr. Cycl. Agroecosys.* **2002**, *62*, 103–113.
4. Oehl, F.; Frossard, E.; Fliessbach, A.; Dubois, D.; Oberson, A. Basal organic phosphorus mineralization in soils under different farming systems. *Soil Biol. Biochem.* **2004**, *36*, 667–675.
5. Ruppel, S. Bedeutung der rhizosphären-und endphytischen Bakterien für die Pflanzenernährung. *Arch. Acker-Pfl. Boden.* **2000**, *45*, 329–341.
6. Baum, C.; Stetter, U.; Makeschin, F. Growth response of *Populus trichocarpa* to inoculation by the ectomycorrhizal fungus *Laccaria laccata* in a pot and a field experiment. *Forest Ecol. Manage.* **2002**, *163*, 1–8.
7. Leitfeld, J.; Siebert, S.; Kögel-Knabner, I. Biological activity and organic matter mineralization of soils amended with biowaste composts. *J. Plant Nutr. Soil Sci.* **2002**, *165*, 151–159.
8. Oberson, A.; Fardeau, J.C.; Besson, J.M.; Sticher, H. Soil phosphorus dynamics in cropping systems managed according to conventional and biological agricultural methods. *Biol. Fert. Soil* **1993**, *16*, 111–117.
9. Deubel, A. Einfluss wurzelbürtiger organischer Kohlenstoffverbindungen auf Wachstum und Phosphatmobilisierungsleistung verschiedener Rhizosphärenbakterien. *Diss. Univ. Halle.* **1996**.
10. Bünemann, E.; Smithson, P.C.; Jama, B.; Frossard, E.; Oberson, A. Maize productivity and nutrient dynamics in maize-fallow rotations in western Kenya. *Plant Soil* **2004**, *264*, 195–208.
11. Selles, F.; Campbell, C.A.; Zentner, R.P. Effect of cropping and fertilization on plant and soil phosphorus. *Soil Sci. Soc. Am. J.* **1995**, *59*, 140–144.
12. Ross, D.J.; Tate, K.R.; Scott, N.A.; Feltham, C.W. Land-use change: Effect on soil carbon, nitrogen and phosphorus pools and fluxes in three adjacent ecosystems. *Soil Biol. Biochem.* **1999**, *31*, 803–813.
13. Oberson, A.; Jöner, E.J. Microbial turnover of phosphorus in soil. In *Organic Phosphorus in the Environment*; Turner, B.L., Frossard, E., Baldwin, D.S., Eds.; CABI Publishing: Wallingford, 2005; 133–164.
14. van Veen, J.A.; Ladd, J.N.; Martin, J.K.; Amato, M. Turnover of carbon, nitrogen and phosphorus through the microbial biomass in soils incubated with ¹⁴C-, ¹⁵N- and ³²P-labelled bacterial cells. *Soil Biol. Biochem.* **1987**, *19*, 559–565.
15. Tarafdar, J.C.; Yadav, R.S.; Meena, S.C. Comparative efficiency of acid phosphatase originated from plant and fungal sources. *J. Plant Nutr. Soil Sci.* **2001**, *164*, 279–282.
16. Turner, B.L.; Frossard, E.; Baldwin, D.S. *Organic Phosphorus in the Environment*; Cabi Publishing: Wallingford, 2005.
17. George, T.S.; Gregory, P.J.; Wood, M.; Read, D.; Buresh, R.J. Phosphatase activity and organic acids in the rhizosphere of potential agroforestry species and maize. *Soil Biol. Biochem.* **2002**, *34*, 1487–1494.
18. Heilmann, E.; Leinweber, P.; Ollesch, G.; Meißner, R. Spatial variability of sequentially extracted P fractions in a silty loam. *J. Plant Nutr. Soil Sci.* **2005**, *168*, 307–315.
19. Eivazi, F.; Tabatabai, M.A. Phosphatases in soils. *Soil Biol. Biochem.* **1977**, *9*, 167–172.
20. Eichler, B.; Caus, M.; Schnug, E.; Köppen, D. Soil acid and alkaline phosphatase in relation to crop species and fungal treatment. *FAL Agric. Res.* **2004**, *54* (1), 1–6.
21. Richardson, A.E.; Hadobas, P.A.; Hayes, J.E.; O'Hara, C.P.; Simpson, R.J. Utilization of phosphorus by pasture plants supplied with myo-inositol hexaphosphate is enhanced by

- the presence of soil micro-organisms. *Plant Soil* **2001**, *229*, 47–56.
22. Unno, Y.; Okubo, K.; Wasaki, J.; Shinao, T.; Osaki, M. Plant growth promotion abilities and microscale bacterial dynamics in the rhizosphere of Lupin analysed by phytate utilization ability. *Environ. Microbiol.* **2005**, *7*, 396–404.
 23. Hayes, J.E.; Simpson, R.J.; Richardson, A.E. The growth and utilization of plants in sterile media when supplied with inositol hexaphosphate, glucose 1-phosphate or inorganic phosphate. *Plant Soil* **2000**, *220*, 165–174.
 24. Findenegg, G.R.; Nelemans, J.A. The effect of phytase and the availability of P from myo-inositol hexaphosphate (phytate) for maize roots. *Plant Soil* **1993**, *154*, 189–196.
 25. Illmer, P.; Schinner, F. Solubilizing of inorganic calcium phosphates—Solubilising mechanisms. *Soil Biol. Biochem.* **1995**, *24*, 389–395.
 26. Read, D.J.; Perez-Moreno, J. Mycorrhizas and nutrient cycling in ecosystems—A journey towards relevance? *New Phytol.* **2003**, *157*, 475–492.
 27. Gryndler, H.; Hrševola, H.; Sudová, R.; Gryndlerová, H.; Řezáčová, V.; Merhautová, V. Hyphal growth and mycorrhiza formation by the arbuscular mycorrhizal fungus *Glomus claroideum* BEG 23 is stimulated by humic substances. *Mycorrhiza* **2005**, *15*, 483–488.
 28. Mohammad, M.J.; Hamad, S.R.; Malkawi, H.I. Population of arbuscular mycorrhizal fungi in semi-arid environment of Jordan as influenced by biotic and abiotic factors. *J. Arid. Environ.* **2003**, *53*, 409–417.
 29. Gamalero, E.; Trotta, A.; Massa, N.; Copetta, M.; Martinotti, M.G.; Berta, G. Impact of two fluorescent pseudomonads and an arbuscular mycorrhizal fungus on tomato plant growth, root architecture and P acquisition. *Mycorrhiza* **2004**, *14*, 185–192.
 30. Martinez, A.; Obertello, M.; Pardo, A.; Ocampo, J.A.; Godeas, A. Interactions between *Trichoderma pseudokoningii* strains and the arbuscular mycorrhizal fungi *Glomus mosseae* and *Gigaspora rosea*. *Mycorrhiza* **2004**, *14*, 79–84.

Phosphorus: Spatial Speciation in Agricultural Soils

Anja Gassner

*Institute of Science and Technology, Malaysia Sabah University,
Kota Kinabalu, Malaysia*

Ewald Schnug

*Institute for Crop and Soil Science, Julius Kuhn Institute (JKI),
Braunschweig, Germany*

Abstract

Phosphorus (P) is present in soils not as a definite and easily separated species, but as a continuum of compounds of different compositions and plant availability, in equilibrium with each other. This heterogeneous equilibrium is constantly disturbed by the uptake of the growing plant and by physical, chemical, or biochemical changes in the soil as well as fertilizer input. Within fields and across short distances, these factors can vary significantly in well-defined patterns, resulting in the spatial speciation of P. Consequently, soil P tests, evaluating the P status of a field, have to be adjusted to the site-specific soil characteristics. Furthermore, the interpretation of these results for subsequent management practices has to consider site-specific factors such as topography and past management history. Finally, the concept of spatial speciation should be applied to other nutrients.

INTRODUCTION

Phosphorus (P) mirabilis, the light-bearing nutrient, was probably discovered by an Arabian alchemist named Alchid Bechil, although its discovery is usually attributed somewhat later to Henning Brandt.^[1] Since the early work of Justus Liebig (1803–1873), agricultural researchers have tried to tackle the mystery of P availability to plants. P research intuitively reminds one of Heisenberg's theory: "The more you see, the less you know." Despite the substantial amount of information available about the transformation products of P fertilizer within temperate^[2] as well as tropical soils^[3] and numerous attempts to conduct elaborate soil test methods to predict P availability to crops and algae,^[2] the management of phosphate, a finite nonrenewable resource, is far from being in accordance with the principles of sustainability.^[4]

This is partly because of the prevailing misconception that soil is a homogenous, static entity. P is regarded traditionally as immobile, but its chemical reactivity with environmental factors, acting at different spatial scales and over different time periods, results in the formation of P species, which not only differ in their plant availability but also in their spatial distribution throughout the field. This spatial speciation^[5] is the key for a new approach to assess soil analysis methods.

CHEMICAL SPECIATION OF P

With the exception of small molecular organic P fractions, plants can only utilize P in its soluble form as

orthophosphate.^[6] Unlike carbon and nitrogen, which can be added to the soil system from the atmosphere, the P status of natural systems is essentially controlled by the occurrence of primary apatite minerals.^[7,8] Consequently, the P enrichment of soils depends directly on P inputs by mineral fertilizers and manure. Although the fate of P when applied to soil remains something of an enigma, it is widely accepted that more than 80% is immobilized by the soil because of precipitation and sorption processes,^[3] whereby the limiting step to furnish crop requirements is the dissolution of initial reaction products during the cropping season.^[9]

For a more practical approach, the soil P continuum is thought as three functional pools: a readily available pool, a reversibly available pool, and a sparingly available P pool,^[10] whereby the readily available P pool is related to the so-called "intensity factor," which is a measure of the gradient in the electrochemical potential of the phosphate ions across the adsorbing surfaces of plant roots and, in its simplest form, can be regarded as the P concentration in the soil solution.^[11] This pool represents P that is readily accessed by plant roots. The reversibly available P denotes the soil P reserve that can be converted into soluble (readily available) P, by either living organisms or by weathering during the growth season. This pool relates to the so-called "quantity factor" or "richness factor."^[12] Whereas the sparingly available P is not available on a short time scale such as one or more crop cycles, a small fraction of this pool may become available during long-term soil transformation.

MOBILITY OF P IN SOILS

As a tetrahedral oxyanion, phosphate has a very low solubility in soils and, in general, does not move with solvent fluxes, apart from small distance diffusion.^[6] In this sense, P is regarded as an “immobile” nutrient. Thus the physical movement of P is restricted to the movement of P associated with soil particles and large molecular-weight organic matter (particulated P) by either bioturbation, soil tillage activities, or soil erosion during flow events.^[13,14] This behavior of soil P was recognized long ago by European geographers and since then has been used in archaeology to trace back ancient settlements.^[15] Conway^[16] introduced the use of total P distribution patterns for the analysis of small-scale occupation deposits. For example, one building showed evidence of having been demolished and partially reincorporated into the courtyard of a subsequent structure. The floor area of the remnant original structure was protected by a layer of small stones and contained high levels

of P. That portion of the floor, which was subsequently converted into a courtyard, unprotected by stones, had less total P, having lost it by exposure and erosion. In an agricultural context, this translates to the following conclusions. First, as far as its total amounts are concerned, P applied with fertilizers may not move from the place it is applied to, and, second, keeping in mind that most soils are naturally poor in P, nearly all the spatial distribution of total P in agricultural soils should be more or less random, reflecting only the spatial sum of distribution faults of past anthropogenic activities (e.g., fertilization and animal husbandry). Spatial relationships may only have developed under the influence of erosion processes.^[16]

SPATIAL SPECIATION OF P

During the last years, the study of spatial variation of soil fertility parameters has expanded considerably, but studies

Table 1 Parameters of autocorrelation for soil P extracted by different P methods in selected investigations.

References	Soil texture	Sampling design (m)	Extraction method	Variogram model	Range of autocorrelation (m)
Trangmar (cited in Gassner ^[17])	—	1.5 × 1.5 m	Truog	Spherical	5.6
Boyer et al. (cited in Gassner ^[17])	μl	2 m transects	Bray I	Spherical	37
Doberman et al. (cited in Gassner ^[17])	T	5-m triangular grid	Olsen	Nested	48
Simard et al. (cited in Gassner ^[17])	μT—T	12 × 15 triangular	Mehlich III	Exponential	139
Karlen et al. (cited in Gassner ^[17])	—	15 × 15 grid	Bray I	Spherical	70
Webster and McBratney (cited in Gassner ^[17])	—	16 × 16 random	Morgan	Spherical	241
Gupta et al. (cited in Gassner ^[17])	sT	20 × 20 grid	Mehlich I	Exponential	29
Romanokov (cited in Gassner ^[17])	μl	20 × 20 grid	0.2 N HCl	Spherical	50–60
Nolin et al. (cited in Gassner ^[17])	μT—T	30 × 30 grid	Mehlich III	Exponential	39
Haneklaus et al. (cited in Gassner ^[17])	IS—sL	30 × 30 grid	CAL	Spherical	153
Gassner et al. ^[18]	μL	30 × 30 grid	AAC—EDTA	Spherical	110
Haneklaus et al. (cited in Gassner ^[17])	sL—μl	50 × 50 grid	DL	Spherical	115
Haneklaus et al. (cited in Gassner ^[17])	IS—sL	50 × 50 grid	DL	Spherical	131
Gassner et al. ^[5]	IS—sL	50 × 50 grid	CAL	Spherical	253
Chien et al. (cited in Gassner ^[17])	sL—μl	250 × 250 grid	Mehlich III	Spherical	580
Yost et al. (cited in Gassner ^[17])	—	1–2 km transect	Olsen	Exponential	1000

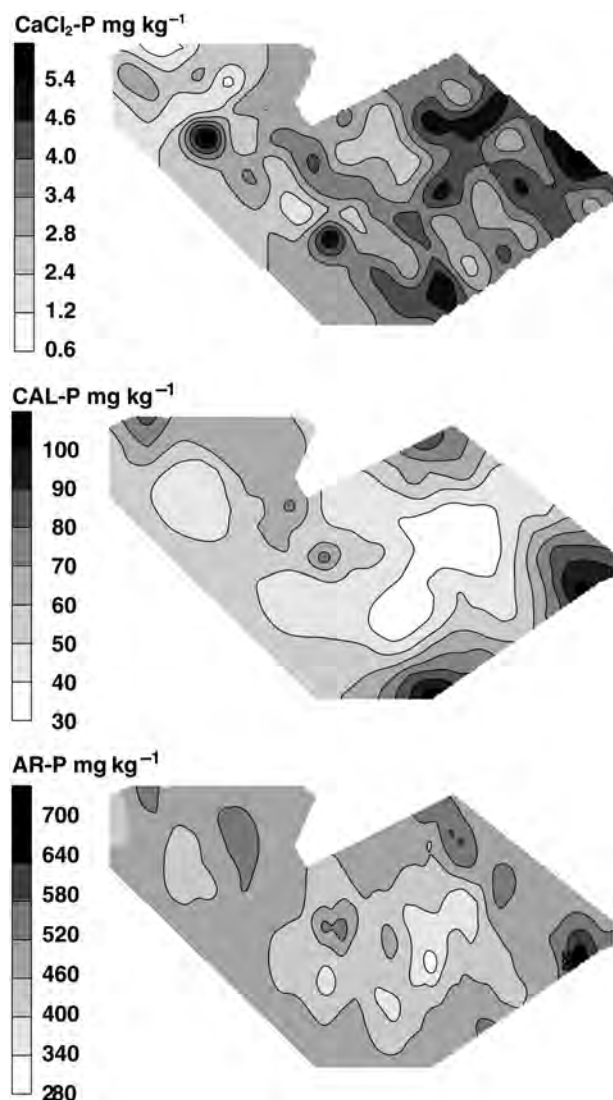


Fig. 1 Spatial distribution of three different P fractions: CaCl_2 extractable P (upper map), Ca-lactate extractable P (middle map), and Aqua Regia extractable P (bottom map), at Kassow (E12°06' and N53°10'), Northern Germany.

that investigated the spatial distribution of soil P generally focused solely on the distribution of so-called plant-available P. The results of these studies showed that plant-available P does not fluctuate randomly but shows distribution patterns with well-defined lag ranges, where the ranges differ, depending on the sampling procedure and scale of investigation (Table 1). For an introduction into geostatistical terminology, see the entry *Geostatistics* (p. 1012).

Additionally, it is generally accepted that the distribution of available P does not necessarily resemble the distribution of total P.^[19] It appears that the speciation of soil P is dependent on site-specific factors and, as such, is a spatial process.^[20–22] In this sense, “spatial speciation” is defined as the chemical reactivity of a nutrient with site-specific

environmental factors, and the subsequent formation of geochemical species that display different spatial dependencies (Fig. 1).^[5]

Insights into the environmental processes that result in the spatial speciation of soil P and, as such, govern the behavior of applied fertilizer are necessary to predict the interconversion and equilibrium distribution of different soil P pools under specific conditions such as geomorphology, field management, and soil types.

The ratio between the readily available P pool (intensity factor) and the reversible available P pool (quantity factor), reflecting the ease of P withdrawal by the plant, is expressed as the “capacity factor.”^[12] The capacity factor, the P adsorption capacity of the soil, is predominantly dependent on the negative surface charges as well as the specific surface area of soil particles, and is mainly a function of the amount and nature of available adsorption sites in the soil.^[23]

For a homogeneous soil, most of the potential adsorption sites will have a similar bonding energy for P. Thus the speciation of P will mainly be a function of the soil pH and the aging of initial reaction products of freshly applied fertilizer P. In this case, the spatial speciation is assumed to be low (Fig. 2). For a heterogeneous soil, the nature of the chemical reactions between P and particle surface is more diverse and the chemical speciation will result in a differentiated distribution of P among soil components. In this case, the spatial speciation is assumed to be more pronounced, as the distribution of different soil components with a particular bonding energy for P will result in a spatial differentiation of P species.

Apart from the site-specific adsorption capacity of a field, site-specific anthropogenic and environmental factors such as management, biological, and physical factors can result in a spatial speciation of P. Whereas the total P—and, therefore, the sparingly available P pool—is relatively inert to short-term environmental impacts, such as fertilizer placement or changes in tillage practices,^[5,24] the reversibly available P pool is most affected by management.^[25,26] Although the readily available P pool is directly influenced by crop selection and management, McDowell and Sharpley^[27] showed that under similar management conditions, the specific adsorption capacity of the soil (capacity factor) controlled the rate of P release into solution.

In a study investigating the spatial speciation of P at three different study sites, which differed in climate, parent material, topography, P fertilizer regime, and land management, the main environmental factors that controlled the spatial distribution of individual P pools were as follows: soil texture for the readily available P; degree of aging of fresh soil P fractions, precipitated from application of soluble fertilizer P, for the reversibly available P; and primary P minerals and geomorphology for the sparingly available P.^[17]

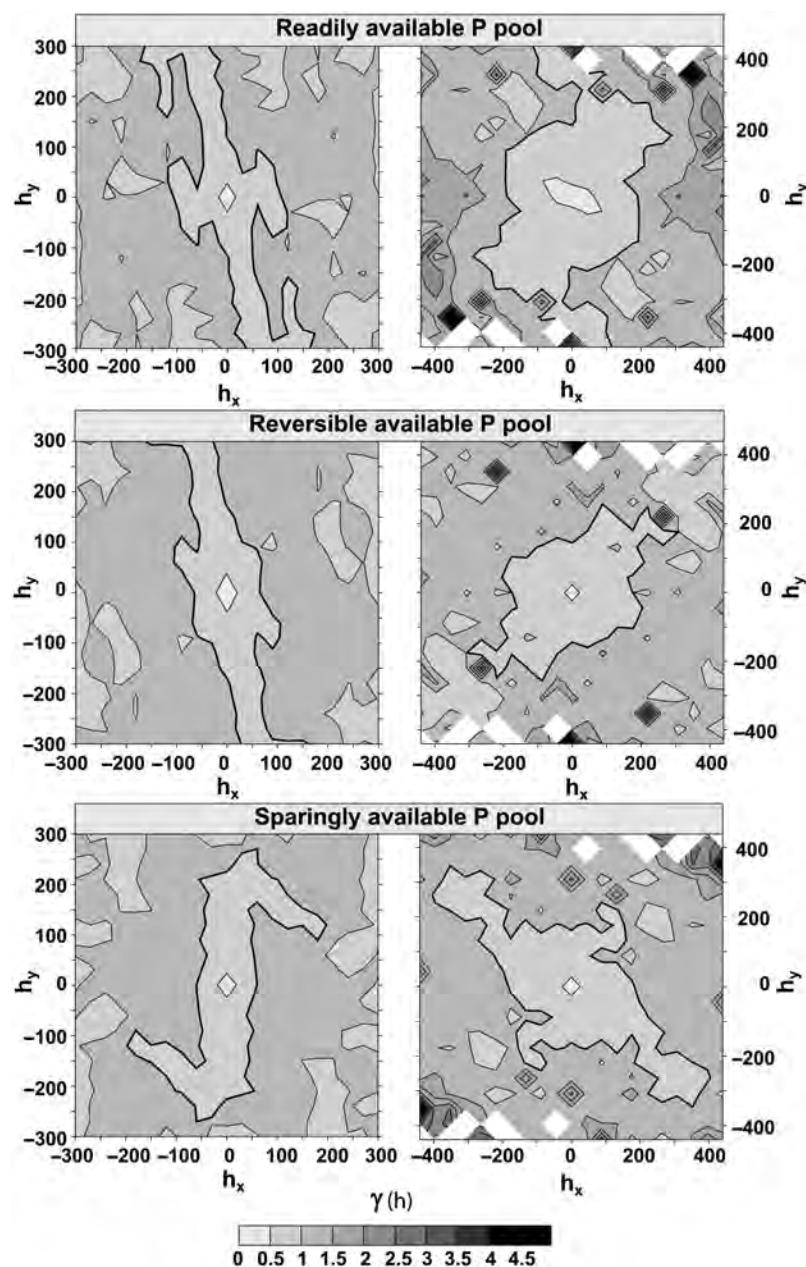


Fig. 2 Variogram maps showing the spatial distribution of the semivariance of different P pools in a loamy sand (right) and a loamy clay (left). The semivariance is a measure of the average degree of dissimilarity between two data points.

Source: From Gassner & Schnug.^[17]

CONCLUSION

P is present in soils not as definite and easily separated species, but as a continuum of compounds of different compositions and plant availability, in equilibrium with each other. This heterogeneous equilibrium is constantly disturbed by the uptake of the growing plant and by physical, chemical, or biochemical changes in the soil as well as fertilizer input. Within fields and across short distances, these factors can vary significantly in well-defined patterns, resulting in the spatial speciation of P. Consequently, soil P tests, evaluating the P status of a field, have to be adjusted to the site-specific soil characteristics. Furthermore, the interpretation of these results for subsequent management

practices has to consider site-specific factors such as topography and past management history. Finally, the concept of spatial speciation should be applied to other nutrients.

REFERENCES

1. Shapiro, J. Introductory lecture at the international symposium 'phosphorus in freshwater ecosystems,' Uppsala, Sweden, October 1985. *Hydrobiologia* **1985**, *170*, 9–17.
2. Frossard, E.; Condron, L.M.; Oberson, A.; Sinaj, S.; Fardeau, J.C. Processes governing phosphorus availability in temperate soils. *J. Environ. Qual.* **2000**, *29*, 15–23.
3. Buehler, S.; Oberson, A.; Rao, I.M.; Friesen, D.K.; Frossard, E. Sequential phosphorus extraction of a P-33-labeled oxisol

- under contrasting agricultural systems. *Soil Sci. Soc. Am. J.* **2002**, *66* (3), 868–877.
4. Steen, I. Phosphorus availability in the 21st century—Management of a non-renewable resource. *Phosphorus Potassium* **1998**, *217*, 25–31.
 5. Gassner, A.; Fleckenstein, J.; Haneklaus, S.; Schnug, E. Spatial speciation—A new approach to assess soil analysis methods. *Commun. Soil Sci. Plant Anal.* **2002**, *33* (15–18), 3347–3357.
 6. Barber, S.A. Soil–plant interactions in the phosphorus nutrition of plants. In *The Role of Phosphorus in Agriculture*; Khasawneh, F.E., Sample, E.C., Kamprath, E.J., Eds.; ASA, CSSA, and SSSA: Madison, 1980; 591–613.
 7. Walker, T.W.; Syers, J.K. The fate of phosphorus during pedogenesis. *Geoderma* **1976**, *15*, 1–19.
 8. Bowman, R.A.; Rodriguez, J.B.; Self, J.R. Comparison of methods to estimate occluded and resistant soil phosphorus. *Soil Sci. Soc. Am. J.* **1998**, *62*, 338–342.
 9. Schnug, E.; Rogasik, J.; Haneklaus, S. Die ausnutzung von phosphor aus düngemitteln unter besonderer berücksichtigung des ökologischen landbaus—The utilisation of fertiliser with special view of organic farming. *Landbau-forsch. Voelkenrode (FAL Agric. Res.)* **2003**, *53* (1), 1–11.
 10. Guo, F.; Yost, R.S. Partitioning soil phosphorus into three discrete pools of differing availability. *Soil Sci.* **1998**, *163* (10), 822–833.
 11. Olsen, S.R.; Khasawneh, F.E. Use and limitation of physical–chemical criteria for assessing the status of phosphorus in soils. In *The Role of Phosphorus in Agriculture*; Khasawneh, F.E., Sample, E.C., Kamprath, E.J., Eds.; ASA, CSSA, and SSSA: Madison, 1980; 361–410.
 12. Williams, D.E.G. *The Intensity and Quality Aspects of Soil Phosphate Status and Laboratory Extraction Values*; Anales de Edafología y Agrobiología, 1966.
 13. Catt, J.A.; Johnston, A.E.; Quinton, J.N. Phosphate losses in the woburn erosion reference experiment. In *Phosphorus Loss from Soil to Water*; Tunney, H., Carton, O.T., Brookes, P.C., Johnson, A.E., Eds.; CAB International: Wallingford, 1997; 374–377.
 14. Sharpley, A.; Foy, B.; Withers, P. Practical and innovative measures for the control of agricultural phosphorus losses to water: An overview. *J. Environ. Qual.* **2000**, *29*, 1–9.
 15. Arrhenius, O. Die phosphatfrage. *Z. Pflanzenernähr. Düng. Bodenkd.* **1929**, *14* (A), 185–194.
 16. Conway, J.S. An investigation of soil phosphorus distribution within occupation deposits from a Romano-British hut group. *J. Archaeol. Sci.* **1983**, *10*, 117–128.
 17. Gassner, A. Factors controlling the spatial speciation of phosphorus in agricultural soils. *Landbauforsch. Völkenrode (Braunschweig)* **2003**, *244*.
 18. Gassner, A.; Habib, L.; Haneklaus, S.; Schnug, E. Significance of the spatial speciation of phosphorous in agricultural soils for the interpretation of variability. *Landbauforsch. Völkenrode (FAL Agric. Res.)* **2003**, *53* (1), 19–25.
 19. Kamprath, E.J.; Watson, M.E. Conventional soil and tissue tests for assessing the phosphorus status of soils. In *The Role of Phosphorus in Agriculture*; Khasawneh, F.E., Sample, E.C., Kamprath, E.J., Eds.; ASA CSSA and SSSA: Madison, 1980; 433–469.
 20. Nowack, K.-H. *Phosphorversorgung Biologisch Bewirtschafteter Äcker und Möglichkeiten der Bioindikation*; University of Göttingen: Göttingen, 1990.
 21. Strohbach, B. Gesetzmäßigkeiten der arealen verteilung des gesamtphosphorgehaltes auf landwirtschaftlich genutzten flächen. *Arch. Acker-Pflanzenbau Bodenkd. (Berlin)* **1986**, *30* (9), 573–580.
 22. Jentsch, U. *Kapazitäts-, Quantitäts-, Intensitäts- und Kinetikparameter des Phosphats in verschiedenen Böden der DDR*; Humboldt-University of Berlin: Berlin, 1986.
 23. Schwertmann, U.; Taylor, R.M. Iron oxides. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; SSSA: Madison, 1989; 379–427.
 24. Beckett, P.H.T.; Webster, R. Soil variability: A review. *Soils Fert.* **1971**, *34*, 1–15.
 25. Schepers, J.S.; Schlemmer, M.R.; Ferguson, R.B. Site-specific considerations for managing phosphorus. *J. Environ. Qual.* **2000**, *29*, 125–130.
 26. Mallarino, A.P. Spatial variability patterns of phosphorus and potassium in no-tilled soils for two sampling scales. *Soil Sci. Soc. Am. J.* **1996**, *60*, 1473–1481.
 27. McDowell, R.; Sharpley, A. Availability of residual phosphorus in high phosphorus soils. *Commun. Soil Sci. Plant Anal.* **2002**, *33* (7&8), 1235–1246.

Phytoremediation

Khan Towhid Osman

Md. Abul Kashem

Department of Soil Science, University of Chittagong, Chittagong, Bangladesh

Abstract

Inorganic soil pollutants include nitrates and phosphates; micronutrients; heavy metals such as copper, iron, manganese, molybdenum, nickel, zinc, arsenic, cadmium, chromium, cobalt, fluorine, mercury, selenium, lead, and vanadium; and radionuclides such as uranium-238, caesium-137, and strontium-90. Major organic pollutants include trichloroethylene, atrazine, trinitrotoluene, gasoline, benzene, toluene, polycyclic aromatic hydrocarbons, methyl tertiary butyl-ether, polychlorinated biphenyls, etc. They are added to soils through industrial processing, solvent and fuel spills, military operations, disposal of industrial, municipal, agricultural and domestic wastes, use of agrochemicals, traffic, etc. These pollutants are hazardous to the health of most life forms including human. Therefore, remediation of soil has become an important environmental consideration. However, the conventional physical and chemical methods, although can effectively remove contaminants, may not be suited for the treatment of widespread yet low levels of contamination of soils. They are expensive and can have high risk of spreading contaminants to non-contaminated areas. Phytoremediation is effective on soils of low level of contamination, has little risk of secondary pollution, and can be applied to most regions and land types as an inexpensive, aesthetically pleasing, and socially acceptable technology for soil remediation. The mechanisms of phytoremediation include phytovolatilization, phytostabilization, phytoextraction, phytodegradation, and rhizofiltration. Many plant species for effective phytoremediation have been identified, and some more have been genetically engineered. For enhancing phytoremediation, some agronomic practices and chemical amendments may be necessary. Details of the mechanisms, factors, and enhancement of remediation are discussed.

INTRODUCTION

Soil pollution has become a serious problem, not alone in the industrially developed countries where huge amounts of organic xenobiotics are synthesized and metal products are manufactured and used, but also in other regions. Organic and inorganic contaminants find their way into the soils through industrial processing; solvent and fuel spills; military operations; disposal of industrial, municipal, agricultural, and domestic wastes; use of agrochemicals; traffic, etc. Major organic pollutants are solvents such as trichloroethylene (TCE); herbicides such as atrazine; explosives such as trinitrotoluene; hydrocarbons such as oil, gasoline, benzene, toluene, and polycyclic aromatic hydrocarbons (PAHs); fuel additives such as methyl tertiary butyl-ether; and the widely discussed polychlorinated biphenyls (PCBs). Inorganic pollutants can occur as natural elements in the earth's crust and are added to soils through anthropogenic pathways. They include plant macronutrients such as nitrates and phosphates; micronutrients such as copper (Cu), iron (Fe), manganese (Mn), molybdenum (Mo), nickel (Ni) and zinc (Zn); non-essential elements such as arsenic (As), cadmium (Cd), chromium (Cr), cobalt (Co), fluorine (F), mercury (Hg), selenium (Se), lead (Pb), and

vanadium (V); and radionuclides such as uranium-238 (^{238}U), caesium-137, and strontium-90. According to NRC,^[1] most frequently priority pollutants at hazardous waste sites in North America and Europe are TCE, Pb, benzene, toluene, Cr, dichloromethane, Zn, 1,1,1-TCE, As, chloroform, dichloroethene, vinyl chloride, barium, ethyl benzene, Ni, di(ethylhexyl)phthalate, xylenes, and phenol. Persistent organic pollutants (POPs) and heavy metals are particularly hazardous to the health of most life forms, including microorganisms, plants, animals, and humans. Many of these contaminants reach concentrations in soil to toxic and lethal extents, and many of them are carcinogenic to human. Moreover, soil pollutants may contaminate other environmental components. Concern has also arisen because these persistent and toxic xenobiotics are biomagnified along trophic webs and transported over long distances. So, remediation or the improvement of contaminated sites through prevention, minimization, or mitigation of damage to various life forms has become a dire necessity for the existence of ecosystems. Several *in situ* or *ex situ* physical, chemical, and biological cleanup methods involving: 1) removal of harmful chemicals from the environment; 2) treatment of harmful chemicals to change into less harmful forms; and 3) containment of

harmful chemicals left in the ground to prevent them from spreading within the environment have evolved. The conventional physical, chemical, and engineering methods can remove contaminants or reduce their hazards often rapidly and effectively from highly polluted sites in confined areas but may not be suited for the treatment of widespread yet low levels of contamination found in soils of many parts of the world. Moreover, there is high risk of spreading contaminants to non-contaminated places and most of these methods are prohibitively expensive.^[2] Meanwhile, phytoremediation, which is used by plants to extract, sequester, and/or detoxify pollutants, has emerged with a set of low technology and low cost but effective and viable techniques that can be carried out on soils of low level of contamination over small or larger areas with little risk of secondary pollution.^[3] Phytoremediation consists of a set of innovative technologies based on the unique extractive and metabolic capabilities of plants to clean up heavy metals, pesticides and xenobiotics, organic compounds, toxic aromatic pollutants, acid mine drainage, etc. Phytoremediation should be applicable to most regions, climates, and land types as an effective, non-intrusive, inexpensive, aesthetically pleasing, and socially acceptable technology for remediation of soil and water. Phytoremediation is cheaper than conventional remediation methods, which does not require expensive equipment or highly specialized personnel, can be applied *in situ*, and does not disturb soil and spread contaminants.

MECHANISMS OF PHYTOREMEDIATION

The following five general mechanisms of phytoremediation are generally recognized: phytovolatilization, phytostabilization, phytoextraction, phytodegradation, and rhizofiltration.^[4] Table 1 gives examples of some contaminants

in soil and water that can be removed through these mechanisms.

Phytovolatilization

Phytovolatilization is the removal of contaminants from water and soils by plants through absorption by roots and the subsequent release of an unchanged volatile contaminant, a volatile degradation product of a contaminant, or a volatile form of an initially non-volatile contaminant from the stomata at the leaves, where gas exchange occurs.^[10] This strategy of remediation differs from others, which has no control over the migration of contaminants to other areas after releasing them into the air. The contaminants may be degraded by hydroxyl radicals or stay in the air as a pollutant depending on the chemical properties of the contaminant. Some plants absorb the metals Hg and Se by the roots from polluted soils and volatilize them by the foliage into the air. Sites for phytovolatilization may not require much management after the original planting and it involves minimal site disturbance, less erosion, and no need to dispose of contaminated plant materials. However, phytovolatilization efforts should be carried out at sites away from population centers because release of toxic substances may affect human health. It may not be successful at places with unique meteorological conditions that promote the rapid redeposition of volatile compounds. On the other hand, some studies on volatilization of Se and Hg revealed that the pollutants were dispersed and diluted to such an extent that volatilization did not pose a threat.^[11] Volatilization of Se involves assimilation of inorganic Se into the organic selenoamino acids selenocysteine and selenomethionine (SeMet). SeMet is methylated to form a volatile, less toxic compound, dimethylselenide (DMSe). Se enzymes of the S assimilation pathway mediate Se volatilization. Overexpression of one of these enzymes, cystathionine-γ-synthase, promotes Se volatilization. Volatile Se compounds, such as DMSe, are less toxic than inorganic forms of Se found in the soil.^[12] Se contamination is a serious problem in many parts of the world where there are areas of Se-rich soil. Indian mustard and canola (*Brassica napus*) may be effective for phytovolatilization of Se. Some natural plants, which can volatilize Hg very slowly from soil, cannot be considered effective in Hg remediation processes. It inspired to engineer plants with a microbial gene, *merA*, which encodes an enzyme that catalyzes the reduction of tissue Hg(II) to volatile and less toxic Hg(0).^[13] Phytovolatilization of organic contaminants, such as TCE, can also occur in conjunction with other phytoremediation processes. In a study of poplar cuttings in hydroponic solution, about 20% of the benzene and TCE in the initial solution was volatilized from the leaves, with little remaining within the plant. Once released into the atmosphere, compounds with double bonds such as TCE and perchloroethylene could be rapidly oxidized in the atmosphere by hydroxyl radicals. About 10% of toluene, ethylbenzene, and xylene

Table 1 Some contaminants in soil and water and mechanisms of their phytoremediation.

Phytoremediation mechanism	Contaminants
Phytovolatilization	Chlorinated solvents, Hg, and Se ^[5]
Phytostabilization	Heavy metals, phenols, and chlorinated solvents ^[6]
Phytoextraction	Cd, Cr, Pb, Ni, Zn, radionuclides, BTEX, pentachlorophenol, and short chained aromatic compounds ^[7]
Phytodegradation	Nitrobenzene, nitroethane, nitrotoluene, atrazine, and chlorinated solvents like dichlorodiphenyltrichloroethane, chloroform, PAH, BTEX, PCB, and tetrachloroethane ^[8]
Rhizofiltration	Heavy metals, radionuclides, and organics ^[9]

was volatilized. There was little volatilization of nitrobenzene and no volatilization of 1,2,4-trichlorobenzene, aniline, phenol, pentachlorophenol, or atrazine.^[14] Phytovolatilization is most applicable to contaminants with a Henry's constant $K_H > 10 \text{ atm m}^3 \text{ water} \cdot \text{m}^{-3} \text{ air}$, such as benzene, toluene, ethylbenzene, and xylenes (BTEX), TCE, vinyl chloride, and carbon tetrachloride.^[15] Chemicals with $K_H < 10 \text{ atm m}^3 \text{ water} \cdot \text{m}^{-3} \text{ air}$ such as phenol and pentachlorophenol are not suitable for the air-stripping mechanism because of their relatively low volatility.^[16]

Phytostabilization

Phytostabilization refers to the reduction in the mobility and bioavailability of environmental pollutants through absorption into and adsorption onto roots, or precipitation within the root zone of plants, and prevention of contaminant migration via wind and water erosion, leaching, and soil dispersion. The principal advantage of phytostabilization is that the disposal of hazardous residues is not required and it is very effective when rapid immobilization is needed to preserve groundwater and surface water.^[17]

Metals are chemically precipitated or sequestered by complexation and sorption within soils, sediments, and mine tailings. Plants stabilize metals in different ways:^[18]

1) Plants harvest water in the root zone and transpire several hundred thousand liters of water per hectare during the growing season. Removal of water from soil has a significant impact on the volume of water that would move downward to reach the groundwater table. It reduces contamination of groundwater with metals and As; 2) Plants stabilize the landscape from erosion, greatly reducing surface water runoff and sediment available to receiving streams. Plants also reduce erosion caused by wind; 3) Plants absorb metals and As and sequester them in their tissues for a considerable period of time. Surface phenomenon of root exudates is also an important potential mechanism of phytostabilization. For example, binding of Cu to root exudates of two cultivated plants, wheat (*Triticum aestivum*) and rape (*B. napus*), and two weeds associated with wheat, dog daisy (*Matricaria inodora*) and cornflower (*Centaurea cyanus*), was reported by Sylvie et al.^[19] Phytochemical complexation in the root zone, transport protein inhibition, and vacuolar storage in the root cells are the three mechanisms of phytostabilization.^[20] Grasses, sedges, forage plants, and reeds with high transpiration rates are widely used in phytostabilization. The combinations of hardy, perennial, dense rooted plants or deep rooting trees (e.g., poplar and cottonwoods) have a particularly positive effect on the remediation of soil contamination.

Phytoextraction

Some plants can absorb considerable amounts of contaminants from polluted sites and store them in their biomass that can be harvested, treated, and disposed. The process of

using this capacity of plants for successful remediation of pollutants from the environment is called phytoextraction.^[21] Some metals can be locked for a long time in the wood of trees, and then the mechanism is called phytosequestration. Phytoextraction, known also as phytoaccumulation, can isolate and remove contaminants from soil without destroying its structure and fertility. It is suitable for the remediation of diffusely polluted areas, where pollutants occur only superficially at relatively low concentration. Factors such as availability of suitable native plant species, growth rate, element selectivity, resistance to diseases, method of harvesting, and disposal are important aspects regarding success of phytoextraction.^[22] A number of plant species of different families have the unique ability to grow on metal contaminated or metalliferous soils and to accumulate extraordinarily high amounts of heavy metals in the aerial organs, far in excess of normal levels, without suffering from phytotoxic effects.^[23,24] For example, species of the genus *Alyssum* may accumulate tissue concentrations exceeding 2% of plant dry weight of Ni and species of the genus *Thlaspi* may accumulate more than 3% Cd and 0.8% Pb.^[25] Although Jaffre et al.^[26] reported for the first time high accumulation of Ni in the New Caledonian tree species *Sebertia acuminata*, the credit of coining the term hyperaccumulator goes to Brooks et al.^[27] for plants that actively take up exceedingly large amounts of one or more heavy metals from the soil, translocate to the shoot where they accumulate in aboveground organs, especially leaves, at concentrations 100–1000 fold higher than those found in non-hyperaccumulating species. The term hyperaccumulator referred initially to plants that could accumulate more than 1000 mg kg^{-1} Ni (dry weight) in the shoot which was an exceptionally high heavy metal concentration because in most plants Ni toxicity starts from 10 to 15 mg kg^{-1} in vegetative organs.

Threshold values were successively provided for hyperaccumulation of other heavy metals. Hyperaccumulators are plants that, when growing on native soils, concentrate $>10,000 \text{ mg kg}^{-1}$ (1%) Mn or Zn; $>1,000 \text{ mg kg}^{-1}$ (0.1%) As, Co, Cr, Cu, Ni, Pb, Sb, Se, or Ti; and $>100 \text{ mg kg}^{-1}$ (0.01%) Cd in the aerial organs, without suffering from phytotoxic damage.^[28] So far, about 450 angiosperm species have been identified as heavy metal (As, Cd, Co, Cu, Mn, Ni, Pb, Sb, Se, Ti, and Zn) hyperaccumulators. However, new reports of this kind of plants continue to accrue adding new plants to the list of hyperaccumulators.^[29] However, most of these plants are of low biomass, slow growing, or are adapted to extreme environments and can accumulate only one or few specific metals.^[30] Besides a high concentration of metals in aboveground parts, hyperaccumulator plants meet the following requirements: 1) the metal concentrations in shoots must be greater than that in roots; 2) aboveground metal concentrations must be 100–500 times higher than those of the same plant species from non-polluted environments; and 3) a hyperaccumulator is regarded as a plant in which the concentrations of

heavy metal in the shoots are greater than that in soils. More than 75% hyperaccumulator plants accumulate Ni, and only five species to date has been identified for Cd hyperaccumulation.^[24] Ni has been found to reach the highest concentration in a tree endemic to the serpentine soil from New Caledonia *S. acuminata*, which accumulates up to 26% Ni (dry mass) in its latex.^[31] *Thlaspi calaminare* and *Phyllanthus serpentinus* were reported to accumulate 39,600 mg kg⁻¹ Zn and 38,100 mg kg⁻¹ Ni, respectively, in their leaves.^[32] About 25% of discovered hyperaccumulators belong to the family of Brassicaceae and, in particular, to genera *Thlaspi* and *Alyssum*. The majority of metal hyperaccumulating plant species discovered so far are restricted to tropical areas. Habit of these plants ranges from annual herbs to perennial shrubs and trees, and some species, such as *Sedum alfredii*, show the capacity of accumulating two or more elements.^[33] A list of hyperaccumulator plants obtained from different investigators^[6,30,34,35,36] is given below.

As: *Pteris vittata*, *Agrostis capillaris*, *Agrostiscastellana*, *Agrostis tenerrima*, and *Sarcosphaera coronaria*.

Cr: *Alyxia rubricaulis*, *Maytenus bureaviana*, *Maytenus pantheriana*, *Maytenus sebertiana*, *Garcinia amplexicaulis*, *Austromyrtus bidwillii*, *Eugenia clusioides*, *Eugenia* sp., *Beaupreopsis paniculata*, *Macadamia angustifolia*, *Macadamia neurophylla*, *Astragalus stanleya*, *Haplopappus*, and *Machaeranthera*.

Co and Cu: *Pandiaka metallorum*, *Anisopappus davyi*, *Cyanotis longifolia*, *Ascolepis metallorum*, *Bulbostylis pseudoperennis*, *Phyllanthus williamiioides*, *Crotalaria cobalticola*, *Vigna dolomitica*, *Aeollanthus subacaulis* var. *linearis*, *Haumaniastrum robertii*, *Eragrostis racemosa*, *Actiniopteris* sp., *Buchnera henriquesii*, *Sopubia neptunii*, *Triumfetta dekindtiana*, *Triumfetta welwitschii* var. *descampii*, and *Xerophyta retinervis* var. *equisetoides*.

Cu: *Clerodendrum infortunatum*, *Croton bonplandianus*, *Geniosporum tenuiflorum*, *Tephrosia villosa*, and *Waltheria indica*.

Cd, Pb and Zn: *Arabidopsis halleri*, *Thlaspi caerulescens*, *Thlaspi brachypetalum*, *Thlaspi ochroleucum*, *Thlaspi cepaeifolium*, *Thlaspi praecox*, *Thlaspi stenopterum*, *Thlaspi tatrensiniuartia verna*, *Polycarpha synandra*, *Dichapetalum gelonioides*, *Armeria maritime*, *Agrostis tenuis*, *Arrhenatherum elatius*, *Festuca ovina*, *Rumex acetosa*, and *Viola calaminaria*.

Mn: *Vaccinium myrtillus*, *A. bidwillii*.

Ni: *Berkheya coddii*, *Pentacalia* (10 species), *Senecio* (9 species), *Alyssum* (52 taxa), *Bornmuellera* (6 taxa), *Cassia kleinii*, *Cochlearia aucheri*, *Crotalaria biflora*, *Evolvulus alsinoides*, *Hybanthus enneaspermus*, *Peltaria emarginata*, *Streptanthus polygaloides*, and *Thlaspi* (23 taxa).

Phytodegradation

Phytodegradation involves the chemical modification of the pollutants, usually by decomposing them into less

harmful products through the action of enzymes, acids, and other metabolic by-products. Degradation of organic pollutants in the rhizosphere can be enhanced directly by root-derived enzymes and indirectly by aeration due to root activity, microbial activity, and modified microbial composition due to C inputs and metabolic precursors (e.g., phenolics) exuded by roots. These substances can attack the pollutant through enzymatic and metabolic activities and can produce unspecific effects of changes in pH, osmotic potential, redox potential, partial pressures of oxygen/carbon dioxide (O₂/CO₂) in soil, etc. Thus, phytodegradation mainly remediates organic pollutants in soils and sediments and can occur in three different manners:

1. Organic pollutants are decomposed and converted into less harmful substances in soil by metabolites in root exudates. For example, TCE can be degraded in soil by the roots of hybrid poplars (*Populus trichocarpa* × *Populus deltoides*) and Eastern cottonwoods (*P. deltoides*). These phreatophytic plants exude dehalogenase enzyme that transforms or mineralizes TCE. Important rhizodegrader plants include *Elymus canadensis*, *Festula arundinacea*, *Festula rubra*, *Lolium prene*, *Lotus corniculatus*, *Melilotus officinalis*, *Panicum virgatum*, *Trifolium pretense*, *Trifolium repens*, *Helianthus annuus*, *Pteris vittata*, *Betula pendula*, *Illex* spp., *Morus rubra*, *Populus* spp., *Salix* spp., etc.^[6]
2. Plants absorb contaminants through roots and convert them into harmless substances within the plant tissue. For example, the phreatophytic plants, hybrid poplars and Eastern cottonwood, can accumulate TCE in their tissues and metabolize them converting it all the way to normal end points such as CO₂ and salts. Some plants contain enzymes that can breakdown and convert ammunition wastes, chlorinated solvents such as TCE, and other herbicides. The enzymes are usually dehalogenases, oxygenases, and reductases. Pilon-Smits^[37] reported the mechanism of degradation of xenobiotics, upon entering the symplast, as the modification through oxidation, reduction, and/or hydrolysis reactions followed by conjugation with glutathione (GSH), sugars, or organic acids. The latter step renders them more soluble and probably facilitates their subsequent binding to enzymes, transporters, and other relevant proteins. Conjugated xenobiotics are then sequestered as part of insoluble cell wall polymers or in cellular compartments such as vacuoles, where they can be metabolized further ideally to CO₂ (mineralization). Uptake and translocation of a POP molecule in plant tissues depend on the pollutant's concentration in the soil solution, its ability to enter the root system, and the rate of transpiration in the plant.^[38] Since most POPs are irreversibly held on soil particle surfaces and usually very little desorption occurs, only low concentrations of POPs are present in soil solution. So, the

efficacy of phytoremediation of POPs is limited by the bioavailability of the pollutant that depends on the physical and chemical properties of their own as well as those of the soil. As uptake of POPs and their translocation within the plant tissue occurs usually in passive processes, removal of POPs from soil in significant quantity is possible only by increasing their solubility. Cyclodextrins amendments have been used to increase the elution of organic compounds from soils.^[39]

3. Biochemical breakdown of organic contaminants is caused by microorganisms stimulated by rhizosphere activity (often called rhizodegradation which is a slow process).^[40] Plant roots release, in their vicinity in soil, a wide variety of organic compounds that can have a very important influence, positive and negative, on growth of other plants and microorganisms and on their activities. Changes in microbial populations following exudate inputs may selectively increase the proportion of pollutant degraders in the soil. But it is more commonly observed that the proportion of pollutant degraders remains unchanged in polluted soil receiving root exudates, even when degradation is enhanced.^[41] The compounds in root exudates include sugars dominated by glucose, proteinogenic amino acids, organic acids, flavonols derivatives of lignin, coumarins, aurones, glucosinolates, anthocyanins, indole compounds, fatty acids, sterols, allomones, proteins and enzymes, etc. Many of these compounds stimulate the growth of microorganisms in the rhizosphere and accelerate the degradation of xenobiotics. Prolific microbial growth due to various root exudates in the rhizosphere provides improved degradation.^[16] Many of the soil contaminants are petroleum hydrocarbons (PHCs) derived from crude oil or refined petroleum products. Trees and herbaceous plants are widely and effectively used in the remediation of PHC-contaminated soils.

Rhizofiltration

The absorption of contaminants, usually metals, from aquatic environments by plants is known as rhizofiltration. Rhizofiltration uses plant roots to absorb heavy metals from polluted effluents, extracted groundwater, surface water, and wastewater with low contaminant concentrations of metals like Pb, Cd, Cu, Ni, Zn, and Cr.^[42] Rhizofiltration can partially treat industrial discharge, agricultural runoff, or acid mine drainage. Rhizofiltration can be employed to treat surface waters used for irrigation. It occurs *in situ* or *ex situ* and species other than hyperaccumulators can also be used. Indian mustard has proven to be effective in removing a wide concentration range of Pb (4–500 mg L⁻¹).^[43] The technology has been tested in the field with U-contaminated water at concentrations of 21–874 µg L⁻¹,

and the treated U concentration reported by Dushenkov et al.^[44] was <20 µg L⁻¹ before discharge into the environment. Two natural divisions of plants are involved in rhizofiltration: 1) purely aquatic plants; and 2) plants with submerged root systems. Among aquatic plants, water hyacinth is the most popular in phytoremediation of polluted waters. It can grow fast and can accumulate nutrients as well as heavy metals. Water hyacinth can remove high amounts of Cd, Co, Ni, and silver and appreciable amounts of Pb and Hg.^[45] But water hyacinth has a number of problems that limit its commercial use. It has become a noxious weed in many countries. For example in Sudan, it has completely covered a large area of the Nile River with a dense mat known as *Sudd*. Water hyacinth can grow only in tropical or warm temperate climate, which are frost free in winter. Other aquatic plant species that have been demonstrated to absorb high amounts of heavy metals from water include *Azolla* spp., *Elodea* spp., *Eichhornia crassipes*, *Lemna* spp., *Myriophyllum* spp., *Typha* spp., and *Vallisneria* spp. Removal of Pb by *Azolla pinnata* was demonstrated by Thayaparan et al.^[46] *Pistia stratiotes* has been found to accumulate very high amounts of heavy metals including Cr, Pb, and Ni. It is capable of growing fast and reproducing within the shortest possible time. It is an excellent agent of rhizofiltration of heavy metal from polluted waters.^[47] Several terrestrial plants including sunflower (*H. annuus*), Indian mustard (*Brassica juncea*), tobacco (*Nicotiana tabacum*), rye (*Secale cereale*), spinach (*Spinacia oleracea*), and corn (*Zea mays*) have been studied for their ability of rhizofiltration.^[17] As pollution in surface and groundwater is a great problem in many tropical, particularly Asian and temperate countries, and rhizofiltration is an important mechanism of remediating As from groundwater.

FACTORS AFFECTING PHYTOREMEDIATION

Phytoremediation is influenced by climatic, topographical, hydrological, and edaphological conditions including moisture, temperature, altitude, and soil type and the accessibility of agricultural equipments of the site to be cleaned. The efficiency of phytoremediation largely depends on the interaction of contaminants with plant roots, their transport to other plant parts, and their metabolism in the plant tissue. Depth and extent of the root systems of the plants used for phytoremediation are also important aspects in relation to their success of removal of pollutants. Soil contaminants are organic and inorganic chemical elements, compounds, and complexes, and rhizosphere action on them is affected primarily by their mobility. Their physicochemical characteristics such as solubility in water, molecular size, charge, vapor pressure, etc. and interactions with surrounding molecules determine their mobility and ease of remediation. Soil properties such as pH, texture, structure, and organic

matter contents are relevant in this context.^[48] Some of the metals are immobile in soils and their availability and phytoextraction rate are limited by solubility and diffusion to the root surface.^[49] One way to summarize many of the limitations of phytoremediation is that contaminants must be available to a plant and its root systems for their successful inactivation and eradication.^[37]

ENHANCEMENT OF PHYTOREMEDIATION

Often phytoremediation needs to be enhanced for which some agronomic practices and chemical amendments become necessary. Artificially guiding plant roots toward polluted zones, supplemental irrigation, fertilization, or root oxygenation can also be beneficial. Moreover, suitable plants can be selected or engineered for appropriate root architecture and metabolic capability.^[50] A key to effective phytoremediation, especially phytoextraction, is to enhance metal phytoavailability and to sustain adequate metal concentrations in the soil solution for plant uptake. Several chemical amendments, including some chelating agents and acids, can increase mobility of contaminants. Chelating agents form metal chelate complexes and prevent precipitation and sorption of the metals in the soil. Chelates can also bring metals into solution through desorption of sorbed species and dissolution of precipitated compounds. Additionally, the application of certain chelates to the soil increases the translocation of heavy metals into the shoots. Luo et al.^[51] reported that the application of chelating agents increased the root-to-shoot ratios of the metals Cu, Pb, Zn, and Cd in *Z. mays* and *Phaseolus vulgaris*. Common chelating agents for phytoextraction of metals include *trans*-1,2-diaminocyclohexane-*N,N,N',N'*-tetraacetic acid (CDTA), *N*-(2-hydroxyethyl)iminodiacetic acid (HEIDA), *N*-hydroxyethylenediaminetriacetic acid (HEDTA), diethylenetriaminepentaacetic acid (DTPA), ethyleneglycol-bis(β -aminoethyl ether) (EGTA), *N,N,N,N'*-tetraacetic acid (EGTA), or ethylenediaminedio-hydroxyphenylacetic acid (EDDHA). Some organic acids, including the malic, citric, and nitriloacetic acids,^[52] have been suggested useful for enhancing phytoextraction of metals. Many reports indicate that ethylenediaminetetraacetic acid (EDTA) and SS-ethylenediaminedissuccinic acid (EDDS) successfully improve heavy metal uptake by plants.^[53] Marques et al.^[53] reported that EDTA addition to contaminated soils increased the accumulation of Zn by *Solanum nigrum* by 231% in leaves, 93% in stems, and 81% in roots. Similarly, EDDS application enhanced the accumulation in leaves, stems, and roots up to 140%, 124%, and 104%, respectively. The decrease in soil pH following biosolid application is the major reason for the greater mobility of heavy metals in biosolid-treated soil. Moreover, dissolved organic matter can facilitate metal transport in soil by acting as a carrier through formation of soluble metal-organic complexes.

GENETIC ENGINEERING FOR PHYTOREMEDIATION

In genetic engineering of plants, a foreign piece of DNA, from any organism ranging from bacteria to mammals, or other plants, is stably inserted into the genome of a cell, which is regenerated into a mature transgenic plant. When the transformed plant is propagated, the foreign gene is inherited by its offspring. Likewise, desirable genes involved in metal uptake, translocation, and sequestration can be transferred into suitable plants for improved phytoremediation traits. The ideal plant species to engineer for phytoremediation should be capable of fast growth and high biomass production, sufficiently hardy and competitive in the climate where it is to be used, has a good phytoremediation capacity to start with, has a widespread, highly branched root system and ability to accumulate metals in the aboveground parts, and is amicable for genetic transformation.^[54] Research on hyperaccumulator species from the Brassicaceae, including *Allysum bertolonii*, *Allysum murale*, *A. halleri*, and *T. caerulescens*,^[55] and the multielement accumulator *B. juncea* has greatly benefited from the availability of functional genomics tools already utilized in *Arabidopsis thaliana* research.^[56] *A. thaliana* L. and tobacco (*N. tabacum* L.) have been genetically modified with bacterial organomecurial lyase (*MerB*) and mercuric reductase (*MerA*) genes.^[13] These plants absorb elemental Hg(II) and methyl Hg from the soil and release volatile Hg(0) from the leaves into the atmosphere. Thus, species of Brassicaceae have become model organisms for the molecular genetic analysis of metal hyperaccumulation and tolerance. Possible genetic manipulation for phytoremediation could be achieved in the following areas:^[54] 1) metallothioneins—the transfer of human metallothionein gene in tobacco resulted in plants with enhanced Cd tolerance and pea metallothionein gene transfer to *A. thaliana* resulted in increased Cu accumulation; 2) phytochelators—transgenic *B. juncea* overexpressing different enzymes involved in phytochelatin synthesis could extract more Cd, Cr, Cu, Pb, and Zn than wild plants; 3) organic acids—the overexpression of citrate synthase has shown to promote enhanced Al tolerance; 4) phytosiderophores—the overexpression of nicotianamine aminotransferase in rice resulted in the overproduction of Fe-chelator deoxy mugineic acid, a phytosiderophore, and consequently promoted a more efficient growth in Fe-deficient soils; 5) ferritin—the overexpression of Fe-binding protein ferritin has shown to increase up to 1.3-fold higher the Fe level in tobacco leaves;^[57] 6) metal transporters—transfer of Zn transporter (ZAT) gene from *Thlaspi goesingense* to *A. thaliana* resulted in twofold higher Zn accumulation in its roots; 7) alteration of metabolic pathways—transfer of *Escherichia coli* ars C and γ -ECS genes to *Arabidopsis* plants resulted in individuals that could transport oxyanion arsenate to aboveground tissues, reduce to arsenite, and sequester it to thiol peptide complexes;

8) alteration of oxidative stress mechanisms—overexpression of GSH-S-transferase and peroxidase in *Arabidopsis* plants resulted in enhanced Al tolerance; and 9) alteration in biomass—increasing phytohormones synthesis can increase biomass of transgenic plants.

CONCLUSIONS

Environmental pollution is a serious global problem. Organic and inorganic contaminants, particularly POPs and heavy metals, which are released into the environment through anthropogenic activities can reach toxic and lethal concentrations in soil, water, and air. Their remediation from polluted sites is, therefore, a dire necessity. Of the variety of techniques available at present, phytoremediation is environmentally, socially, aesthetically, and economically effective, safe, and universally acceptable for low-level pollution of soil and water. Many indigenous plants can be used and explored, and some plants can be genetically modified for improving their remediation capability. Phytoremediation is effective when the contaminants become available to plant roots; hence some sort of enhancement may be needed. Several chemical amendments, including some chelating agents and acids, are used for enhancement. Small-scale farm holders may be familiarized with these techniques of phytoremediation.

REFERENCES

1. National Research Council (NRC). *Alternatives for Groundwater Cleanup*; National Academy Press: Washington, DC, 1994; 314 pp.
2. Ensley, B.D. Rationale for the use of phytoremediation. In *Phytoremediation of Toxic Metals: Using Plants to Clean Up the Environment*; Raskin, I., Ensley, B.D., Eds.; John Wiley and Sons: New York, 2000; 3–11.
3. Shukla, K.P.; Singh, N.K.; Sharma, S. Bioremediation: Developments, current practices and perspectives. *Genet. Eng. Biotechnol. J.* **2010**, *3*, 1–20.
4. Khan, M.S.; Zaidi, A.A.; Wani, A.P.A.; Mohammad, A. Role of plant growth promoting rhizobacteria in the remediation of metal contaminated soils. *Environ. Chem. Lett.* **2009**, *7* (1), 1–19.
5. Terry, N.; Zayed, A.; Pilon-Smits, E.; Hansen, D. Can plants solve the selenium problem? In *Will Plants have a Role in Bioremediation?* Proceedings 14th Annual Symposium on Current Topics in Plant Biochemistry Physiology Molecular Biology, Columbia, April 19–22, 1995, University of Missouri: Columbia, 1995; 63–64.
6. McCutcheon, S.C.; Schnoor, J.L. *Phytoremediation: Transformation and Control of Contaminants*; Wiley-Interscience, Inc.: Hoboken, 2003.
7. Blaylock, M.J.; Huang, J.W. Phytoextraction of metals. In *Phytoremediation of Toxic Metals: Using Plants to Clean up the Environment*; Raskin, I., Ensley, B.D., Eds.; John Wiley & Sons: New York, 2000; 53–70.
8. Hutchinson, S.L.; Schwab, A.P.; Banks, M.K. Biodegradation of petroleum hydrocarbons in the rhizosphere. In *Phytoremediation: Transformation and Control of Contaminants*; McCutcheon, S.C., Schnoor, J.L., Eds.; Wiley-Interscience, Inc.: Hoboken, NJ, 2003; 355–386.
9. Horne, A.J. Phytoremediation by constructed wetlands. In *Phytoremediation of Contaminated Soils and Waters*; Terry, N., Bañuelos, G., Eds.; CRC Press LLC: Boca Raton, 2000; 13–40.
10. Tangahu, B.V.; Abdullah, S.R.S.; Basri, H.; Idris, M.; Anuar, N.; Mukhlisin, M. A review on heavy metals (As, Pb, and Hg) uptake by plants through phytoremediation. *Int. J. Chem. Eng.* **2011**, *21*, 1–31.
11. Lin, Z.Q.; Scemenauer, R.S.; Cervinka, V.; Zayed, A.; Lee, A.; Terry, N. Selenium volatilization from the soil-plant system for the remediation of contaminated water and soil in the San Joaquin Valley. *J. Environ. Qual.* **2000**, *29*, 1048–1056.
12. De Souza, M.P.; Lytle, C.M.; Mulholland, M.M.; Otte, M.L.; Terry, N. Selenium assimilation and volatilization from dimethylselenoniopropionate by Indian mustard. *Plant Physiol.* **2000**, *122* (4), 1281–1288.
13. Rugh, C.L.; Senecoff, J.F.; Meagher, R.B.; Merkle, S.A. Development of transgenic yellow poplar for mercury phytoremediation. *Nat Biotechnol.* **1998**, *16* (10), 925–928.
14. Burken, J.G.; Schnoor, J.L. Uptake and metabolism of atrazine by poplar trees. *Environ. Sci. Technol.* **1997**, *31* (5), 1399–1406.
15. Alvarez, P.J.J.; Illman, W.A. *Bioremediation and Natural Attenuation: Process Fundamentals and Mathematical Models*; Wiley-Interscience: Hoboken, NJ, 2006.
16. Kamath, R.; Schnoor, J.L.; Alvarez, P.J.J. Effects of plant-derived substrates on expression of catabolic genes using a nah-lux reporter. *Environ. Sci. Technol.* **2004**, *38* (6), 1740–1745.
17. Chhotu, D.; Jadia, D.; Fulekar, M.H. Phytoremediation of heavy metals: Recent techniques. *Afr. J. Biotechnol.* **2009**, *8* (6), 921–928.
18. Neuman, D.; Ford, K.L. *Phytostabilization as a Remediation Alternative at Mining Sites*. Technical Note 420. BLM/ST/ST-06/003+3720. Bureau of Land Management: Denver, CO, 2006; 48.
19. Sylvie, D.; Morel, J.L.; Jacobson, A.; Bitton, G. Copper binding capacity of root exudates of cultivated plants and associated weeds. *Biol. Fert. Soils* **2001**, *34*, 230–234.
20. Interstate Technology & Regulatory Council (ITRC). *Phytotechnology Technical and Regulatory Guidance and Decision Trees Revised*. PHYTO-3; ITRC: Washington, DC, 2009.
21. Marques, A.P.G.C.; Rangel, A.O.S.S.; Castro, P.M.L. Remediation of heavy metal contaminated soils: Phytoremediation as a potentially promising clean-up technology. *Crit. Rev. Env. Sci. Tec.* **2009**, *39*, 622–654.
22. Seth, C.S.; Remans, T.; Keunen, E.; Jozefczak, M.; Gielen, H.; Opdenakker, K.; Weyens, N.; Vangronsveld, J.; Cuypers, A. Phytoextraction of toxic metals: A central role for glutathione. *Plant Cell Environ.* **2012**, *35*, 334–346. DOI: 10.1111/j.1365-3040.2011.02338.x.
23. Reeves, R.D. Hyperaccumulation of trace elements by plants. In *Phytoremediation of Metal-contaminated Soils*; Morel, J., Echevarria, G., Goncharova, N., Eds.; Springer: New York, 2006.

24. Rascio, N.; Navari-Izzo, F. Heavy metal hyperaccumulating plants: How and why do they do it? And what makes them so interesting? *Plant Sci.* **2011**, *180* (2), 169–181.
25. Baker, A.J.M.; Brooks, R.R. Terrestrial higher plants which hyperaccumulate metallic elements—A review of their distribution, ecology and phytochemistry. *Biorecovery* **1989**, *1*, 81–126.
26. Jaffre, T.; Brooks, R.R.; Lee, J.; Reeves, R.D. *Sebertia acuminata*: A hyperaccumulator of nickel from New Caledonia. *Science* **1976**, *193* (4253), 579–580.
27. Brooks, R.R.; Lee, J.; Reeves, R.D.; Jaffré, T. Detection of nickeliferous rocks by analysis of herbarium specimens of indicator plants. *J. Geochem. Explor.* **1977**, *7*, 49–57.
28. Verbruggen, N.; Hermans, C.; Schat, H. Molecular mechanisms of metal hyperaccumulation in plants. *New Phytol.* **2009**, *181* (4), 759–776.
29. Karimi, N.; Ghaderian, S.M.; Raab, A.; Feldmann, J.; Meharg, A.A. An arsenic-cumulating, hypertolerant brassica, *Isatis cappadocica*. *New Phytol.* **2009**, *184* (1), 41–47.
30. Reeves, R.D.; Baker, A.J.M. Metal-accumulating plants. In *Phytoremediation of Toxic Metals: Using Plants to Clean-up the Environment*; Raskin, I., Ensley, B., Eds.; Wiley and Sons: New York, 2000; 193–229.
31. Sagner, S.; Kneer, R.; Wanner, G.; Cosson, J.-P.; Deus-Neumann, B.; Zenk, M.H. Hyperaccumulation, complexation and distribution of nickel in *Sebertia acuminata*. *Phytochemistry* **1998**, *47* (3), 339–347.
32. Siegel, F.R. *Environmental Geochemistry of Potentially Toxic Metals*; Springer-Verlag: Heidelberg, 2002.
33. Yang, X.E.; Long, X.X.; Ye, H.B.; He, Z.L.; Stoffella, P.J.; Calvert, D.V. Cadmium tolerance and hyperaccumulation in a new Zn-hyperaccumulating plant species (*Sedum alfredii* Hance). *Plant Soil* **2004**, *259* (1), 181–189.
34. Wither, E.D.; Brooks, R.R. Hyperaccumulation of nickel by some plants of Southeast Asia. *J. Geochem. Explor.* **1977**, *8*, 579–583.
35. Rajakaruna, N.; Bohm, B.A. Serpentine and its vegetation: A preliminary study from Sri Lanka. *J. Appl. Bot.* **2002**, *76*, 20–28.
36. Ma, L.Q.; Komar, K.M.; Tu, C.; Zhang, W.; Cai, Y.; Kennelley, E.D. A fern that hyperaccumulates arsenic. *Nature* **2001**, *409* (6820), 579.
37. Pilon-Smits, E. Phytoremediation. *Annu. Rev. Plant Biol.* **2005**, *56*, 15–39.
38. Komives, T.; Gullner, G. Phytoremediation. In *Plant-environment Interactions*; Wilkinson, R.E., Ed.; Marcel Dekker, Inc.: New York, 2000; 437–452.
39. Komives, T.; Gullner, G.; Bittsanszky, A.; Pascal, S.; Laurent, F. Phytoremediation of persistent organic pollutants. In *Presented in 8th Alps Adria Scientific Workshop*, Neum, Bosnia And Herzegovina, Apr 27 to May 2, 2009.
40. Shukla, K.P.; Sharma, S.; Singh, N.K.; Singh, V.; Tiwari, K.; Singh, S. Nature and role of root exudates: Efficacy in bioremediation. *Afr. J. Biotechnol.* **2011**, *10* (48), 9717–9724.
41. Joner, E.J.; Corgié, S.; Amellal, N.; Leyval, C. Nutritional constraints to PAH degradation in a rhizosphere model. *Soil Biol. Biochem.* **2002**, *34*, 859–864.
42. United States Environmental Protection Agency (USEPA). *Introduction to Phytoremediation*. EPA 600/R-99/107, U.S. Environmental Protection Agency, Office of Research and Development: Washington, DC, 2000; 94.
43. Raskin, I.; Ensley, B.D. *Phytoremediation of Toxic Metals: Using Plants to Clean Up the Environment*; John Wiley & Sons, Inc.: New York, 2000.
44. Dushenkov, S.; Vasudev, D.; Kapulnik, Y.; Gleba, D.D.; Ting, F.K.C.; Ensley, B. Removal of uranium from water using terrestrial plants. *Environ. Sci. Technol.* **1997**, *31* (12), 3468–3474.
45. Outridge, P.M.; Noller, B.N. Accumulation of oxic trace elements by the freshwater vascular plants. *Rev. Environ. Contam. T.* **1991**, *121*, 1–63.
46. Thayaparan, M.; Iqbal, S.S.; Chathuranga, P.K.; Iqbal, M.C. Rhizofiltration of Pb by *Azolla pinnata*. *Int. J. Env. Sci.* **2013**, *3* (6), 1811.
47. Abubakar, M.M.; Ahmad, M.M.; Getso, B.U. Rhizofiltration of heavy metals from eutrophic water using pistia stratiotes in a controlled environment. *IOSR J. Environ. Sci. Toxicol. Food Technol.* **2014**, *8* (6, Ver III), 1–3.
48. Dzantor, E.K. Phytoremediation: The state of rhizosphere engineering for accelerated rhizodegradation of xenobiotic contaminants. *J. Chem. Technol. Biot.* **2007**, *82* (3), 228–232.
49. Turan, M.; Esring, A. Phytoremediation based on canola (*Brassica napus* L.) and Indian mustard (*Brassica juncea* L.) planted on spiked soil by aliquot amount of Cd, Cu, Pb, and Zn. *Plant. Soil. Environ.* **2007**, *53* (1), 7–15.
50. Wang, H.; Inukai, Y.; Yamauchi, A. Root development and nutrient uptake. *Crit. Rev. Plant Sci.* **2006**, *25*, 279–301.
51. Luo, C.; Shen, Z.; Li, X. Enhanced phytoextraction of Cu, Pb, Zn and Cd with EDTA and EDDS. *Chemosphere* **2005**, *59* (1), 1–11.
52. Quartacci, M.F.; Argilla, A.; Baker, A.J.M.; Navari-Izzo, F. Phytoextraction of metals from a multiply contaminated soil by Indian mustard. *Chemosphere* **2006**, *63*, 918–925.
53. Marques, A.P.G.C.; Oliveira, R.S.; Samardjieva, K.A.; Rangel, A.O.S.S.; Pissarra, J.; Castro, P.M.L. EDDS and EDTA-enhanced zinc accumulation by *Solanum nigrum* inoculated with arbuscular mycorrhizal fungi grown in contaminated soil. *Chemosphere* **2007**, *70* (6), 1002–1014.
54. Eapen, S.; D'Souza, S.F. Prospects of genetic engineering of plants for phytoremediation of toxic metals. *Biotechnol. Adv.* **2005**, *23* (2), 97–114.
55. Chaney, R.L.; Angle, J.S.; McIntosh, M.S.; Reeves, R.D.; Li, Y.M.; Brewer, E.P.; Chen, K.Y.; Roseberg, R.J.; Perner, H.; Synkowski, E.C.; Broadhurst, C.L.; Wang, S.; Baker, A.J.M. Using hyperaccumulator plants to phytoextract soil Ni and Cd. *Z. Naturforsch. C.* **2005**, *60* (3–4), 190–198.
56. Cobbett, C. Heavy metals and plants—Model systems and hyperaccumulators. *New Phytol.* **2003**, *159*, 289–293.
57. Goto, F.; Yoshihara, T.; Saiki, H. Iron accumulation in tobacco plants expressing soybean ferritin gene. *Transgenic Res.* **1998**, *7* (3), 173–180.

Plant Growth: Oxygen Diffusion Rate

Witold Stepniewski

Technical University of Lublin, Lublin, Poland

Abstract

Oxygen diffusion rate (ODR) is of importance because availability of oxygen to plant roots is a basic factor of soil productivity. A knowledge of the method of measurement of ODR helps one to evaluate oxygen requirements of particular plant species at different stages of their development. Moreover, this can be used as a diagnostic tool to assess the oxygen availability in a particular soil under definite conditions.

INTRODUCTION

What Is Oxygen Diffusion Rate (ODR)?

ODR is an electrochemical method of assessment of soil oxygen availability to plant roots. It is based on the analogy of oxygen uptake by plant roots and by platinum wire electrode placed in the soil.

Concept and Principle of the Method

As early as 1926, Hutchins^[1] expressed a conviction that it is not so much the concentration of oxygen in the soil as the possibility of its uptake by plant roots that determines the plant response to the oxygen conditions in the soil. This idea was realized in 1952 by Lemon and Erickson,^[2] who designed an electrode simulating oxygen uptake by plant roots. An important role in this concept is played by the presence of water or exudate films on the root surface. These films, due to the low diffusion coefficient of gases in water, form a significant obstacle in the path of oxygen from the soil air to the root.^[2,3]

The principle of this method consists in the measurement of the amount of oxygen diffusing onto the surface of a platinum wire electrode (usually 0.5–1.0 mm in diameter and several millimeter in length), where it is reduced electrochemically to hydroxide ions or water.^[2,3] In practice, the value of ODR is determined on the basis of measurement of the diffusion current intensity on a platinum electrode, which is negatively polarized in order to provide specific conditions for the reduction of oxygen molecules only. The platinum wire is therefore a model of oxygen absorbing root, and the intensity of oxygen flux to the electrode indicates the maximum amount of oxygen that would be available for a plant root placed in the same spot as the electrode.

SOIL PARAMETERS AND ODR

The ODR index reflects comprehensively the soil oxygen availability, as it comprises the effect of all the factors such as respiration, moisture, and the physical particle arrangement that influence the concentration of oxygen in soil air, the effective thickness of the water films around the roots, and their diffusion characteristics.

The ODR values in soils vary, most frequently, within a range from 0 to 200 $\mu\text{g m}^{-2} \text{sec}^{-1}$ (see Figs. 1–3). They decrease with soil-moisture content and with soil compaction, and they increase with moisture tension, with air-filled porosity of the soil, and with oxygen content in soil air.^[4] Due to this, a decrease in ODR is expected usually with depth, especially just above the groundwater table. An example of the relationship of ODR on soil-moisture tension and bulk density is presented in Fig. 1.

Beyond the soil moisture content and bulk density, the value of ODR is related to many other soil parameters. The dependence on the gas diffusion coefficient^[5] is shown in Fig. 2. The relationship with air permeability of the soil^[5] is illustrated in Fig. 3. The value of ODR is also related to redox potential^[4] and to dehydrogenase activity.^[6]

ODR AND PLANT RESPONSE

Seedling Emergence

In a typical relationship between plant emergence and ODR^[4] starting with the highest values of ODR, initially a plateau range is observed with insignificant differentiation in emergence. Then, after reaching a certain limiting value, a significant decrease in the final emergence percentage in comparison with the germination capacity is observed. Thereafter, there is a rapid linear decline in

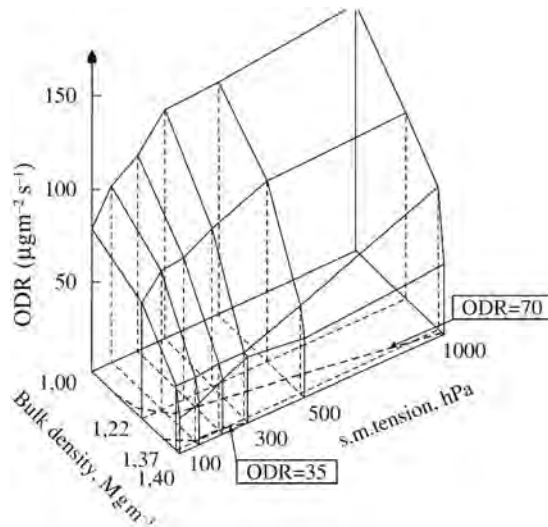


Fig. 1 ODR vs. bulk density and soil moisture tension in a loamy textured Chernozem Rendzina soil.

Source: Adapted from Stepniewski.^[7]

the number of emerging plants as the ODR decreases, this number falling to zero at the critical ODR. It was found that the limiting values for the plant emergence range from 25 to 100 $\mu\text{g m}^{-2} \text{sec}^{-1}$ and the critical values from 7 to 40 $\mu\text{g m}^{-2} \text{sec}^{-1}$.^[4]

Root Response

Root tissue is that part of a plant which is subjected first to oxygen deficiency or anoxia, and the consequence is a decrease of the root biomass at low ODR.^[4,8] It was confirmed that wheat-root population in soil^[9] increased

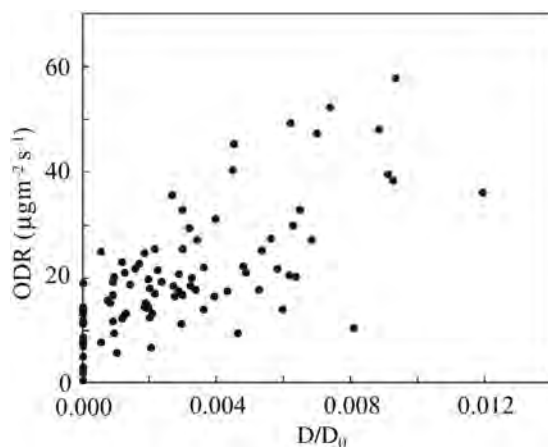


Fig. 2 ODR vs. relative gas diffusion coefficient D/D_0 in soil, where gas diffusion coefficient D of a gas relates to the soil and D_0 to the atmospheric air at the same temperature and pressure conditions. Collective data for 24 soil horizons of six different soil profiles from Hungary.

Source: Adapted from Stepniewski.^[5]

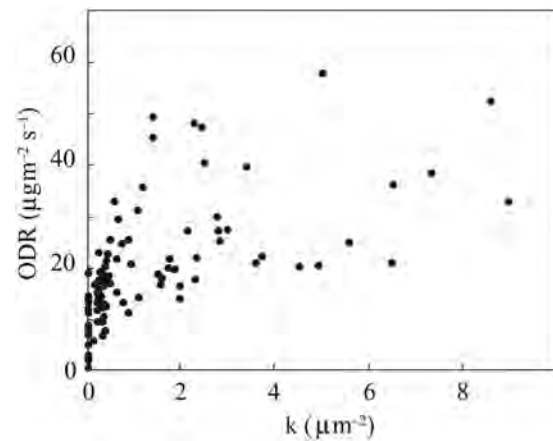


Fig. 3 ODR vs. air permeability (k) of the soil. Collective data for 24 soil horizons of six different soil profiles from Hungary.

Source: Adapted from Stepniewski.^[6]

linearly with ODR in the interval from 40 to 60 $\mu\text{g m}^{-2} \text{sec}^{-1}$ more than fivefold. Root-elongation rate of three desert shrubs also increased with ODR in the interval 30–90 $\mu\text{g m}^{-2} \text{sec}^{-1}$.^[10] It was found that for numerous grass species critical ODR value is below 10 $\mu\text{g m}^{-2} \text{sec}^{-1}$.^[4] In case of lowland rice, root porosity decreased from 23% at ODR < 20 $\mu\text{g m}^{-2} \text{sec}^{-1}$ to about 5% at ODR as high as >100 $\mu\text{g m}^{-2} \text{sec}^{-1}$.^[4]

Shoot Response

Shoot response to root hypoxia expressed by the ODR values has been studied intensively. As it was found, the uptake of water by orange seedling was 2 to 3 times higher at ODR > 60 $\mu\text{g m}^{-2} \text{sec}^{-1}$ as compared to that at ODR < 40 $\mu\text{g m}^{-2} \text{sec}^{-1}$.^[11] Another effect is the closure of stomata and an increase of stomatal diffusive resistance at ODR below 30 $\mu\text{g m}^{-2} \text{sec}^{-1}$.^[8,12] An obvious decrease of the shoot biomass observed by numerous authors for plants such as sorghum, cabbage, soybean, maize, and potato^[4] is the result of physiological disorders. These disorders comprise appearance of ethanol observed in xylem exudate of tomato plants at ODR below 20 $\mu\text{g m}^{-2} \text{sec}^{-1}$,^[13] reduced chlorophyll content,^[8,14] reduced net photosynthetic rate,^[15] reduced carotenoid and protein content,^[15] and reduced activity of superoxide dismutase related to the plant defense system.^[8]

The observed changes in the mineral composition of the plants result from combined effect of redox transformations of nutrients in the soil as well as from the modification of metabolic processes in the plant tissues. Thus, the concentrations of zinc, copper, and boron (B) in the soybean seeds and shoots tended to decrease at ODR values below 20 $\mu\text{g m}^{-2} \text{sec}^{-1}$, while manganese (Mn) and iron concentrations, in turn, tended to increase under such conditions^[16] due to increased availability of the two

latter elements. These opposite tendencies lead to a more distinct differentiation of their ratios, demonstrated for Mn/B ratio.^[17]

These and previously reviewed papers^[4] indicate that plants are differentiated with their response to low ODR values, and the limitation of the growth occurs usually in the range 35–70 $\mu\text{g m}^{-2} \text{sec}^{-1}$.

CONCLUSION

It has been documented that an ODR value depends on numerous soil parameters such as water content, bulk density, air-filled porosity, gas diffusion coefficient, air permeability, and oxygen concentration in soil air, and that it affects redox potential and dehydrogenase activity. It was also shown that plant parameters such as emergence percentage, root population, root elongation, biomass production, as well as numerous physiological parameters like chlorophyll, carotenoids, proteins, superoxide dismutase, ethanol, microelements, stomatal diffusive resistance, and net photosynthesis are related to ODR values.

REFERENCES

- Hutchins, L.M. Studies on the oxygen-supplying power of the soil together with quantitative observations on the oxygen-supplying power required for seed germination. *Plant Physiol.* **1926**, *1*, 95–150.
- Lemon, E.R.; Erickson, A.E. The measurement of oxygen diffusion in the soil with a platinum microelectrode. *Soil Sci. Soc. Am. Proc.* **1952**, *16*, 160–163.
- Lemon, E.E.; Erickson, A.E. Principle of the platinum microelectrode as a method of characterizing soil aeration. *Soil Sci.* **1955**, *79*, 383–392.
- Gliński, J.; Stepniowski, W. *Soil Aeration and Its Role for Plants*; CRC Press: Boca Raton, 1985; 229 pp.
- Stepniowski, W.; Stepniowska, Z.; Przywara, G.; Brzezińska, M.; Włodarczyk, T.; Varallyay, G. Relations between aeration status and physical parameters of some selected Hungarian soils. *Int. Agrophys.* **2000**, *14*, 439–447.
- Brzezińska, M.; Stepniowska, Z.; Stepniowski, W. Soil oxygen status and dehydrogenase activity. *Soil Biol. Biochem.* **1998**, *30* (13), 1783–1790.
- Stepniowski, W. Oxygen diffusion and strength as related to soil compaction. I. ODR. *Pol. J. Soil Sci.* **1980**, *13*, 3–13.
- Bennicelli, R.P.; Stepniowski, W.; Zakrhevsky, D.A.; Balakhnina, T.I.; Stepniowska, Z.; Lipiec, J. The effect of soil aeration on superoxide dismutase activity, malondialdehyde level, pigment content and stomatal diffusive resistance in maize seedlings. *Environ. Exp. Bot.* **1998**, *39*, 203–211.
- Silberbush, M.; Gornat, B.; Goldberg, D. Effect of irrigation from a point source (trickling) on oxygen flux and on root extension in the soil. *Plant Soil* **1979**, *52*, 507–514.
- Lunt, O.R.; Letey, J.; Clark, S.B. Oxygen requirements for root growth in three species of desert shrubs. *Ecology* **1973**, *54*, 1356–1362.
- Stolzy, L.H.; Van Gundy, S.D.; Labanauskas, C.K.; Szuszkiewicz, T.E. Response of *Tylenchulus semipenetrans* infected citrus seedlings to soil aeration and temperature. *Soil Sci.* **1963**, *96*, 292–298.
- Sojka, R.E.; Stolzy, R.H. Soil-oxygen effects on stomatal response. *Soil Sci.* **1980**, *130* (6), 350–358.
- Fulton, J.M.; Erickson, A.E. Relation between soil aeration and ethyl alcohol accumulation in xylem exudate of tomatoes. *Soil Sci. Soc. Am. Proc.* **1964**, *28*, 610–614.
- Huang, B.G.; Liu, X.Z.; Fry, J.D. Shoot physiological responses of two bentgrass cultivars to high temperatures and poor soil aeration. *Crop Sci.* **1998**, *38*, 1219–1224.
- Bennicelli, R.P.; Stepniowska, Z.; Balakhnina, T.A.; Stepniowski, W.; Zuchowski, J. Effect of soil re-oxidation on wheat (*Triticum aestivum* L.) defence system. *Int. Agrophys.* **1999**, *13*, 309–314.
- Stepniowski, W.; Przywara, G. The influence of oxygen availability on the content and uptake of B, Cu, Fe, Mn and Zn by soybean. *Folia Soc. Sci. Lublinensis* **1990**, *30*, 79–89.
- Gliński, J.; Stepniowski, W.; Stepniowska, Z.; Ostrowski, J.; Włodarczyk, T.; Brzezińska, M. Agroecological aspects of oxygen conditions in arable soils. *Acta Agrophys.* **2000**, *32*, 1–87.

Plant Nutrients

Volker Roemheld

Institute for Plant Nutrition, University of Hohenheim, Stuttgart, Germany

Abstract

Great progress in the knowledge on requirements and the function of plant nutrients for plant growth has initiated the recognition of the use of mineral fertilizers in the late 19th and the early 20th centuries. Considering a still growing world population, further improvements and achievements in an adequate supply of needed plant nutrients as organic or mineral fertilizers, together with innovative application techniques with maximum use efficiency, will be required in the future.

INTRODUCTION

In the late 19th century, the importance of mineral elements for the growth and development of plants was established, particularly, by Carl Sprengel (1787–1859) and Justus von Liebig (1803–1873). In the following decades, mineral elements were classified as essential (“plant mineral nutrients”), non-essential, and beneficial elements by the strict criteria that were laid down by Arnon and Stout (1939).

As a consequence of the great progress in understanding the function of these mineral nutrients in plants, the use of increasing amounts of mineral fertilizers in agriculture was initiated. Further progress in use of mineral nutrients for improvement in crop yield and crop quality is required to match the need of a worldwide growing population.

PLANT NUTRIENTS

Historical Retrospect

Although the beneficial effect of adding mineral elements (e.g., plant ash and lime) as soil fertilizers for growth promotion had been known for a long time, the first who compiled and summarized the importance of distinct mineral elements were Sprengel (1787–1859) and von Liebig (1803–1873).^[1]

In particular, von Liebig’s well-known textbook *Die Chemie in ihrer Anwendung auf Agricultur und Physiologie*^[2] rose awareness of farmers and scientists of plant mineral nutrients. Thus, as a consequence of his achievements, the use of large amounts of potash, superphosphate, and later inorganic nitrogen in agriculture and horticulture was initiated to improve plant growth. But also investigations into the essentiality of the various mineral elements were undertaken by different scientists at the end of the 19th century to support von Liebig’s “mineral element theory.”

From those investigations into the mineral composition of different plant species grown on various substrates, it was obvious that the presence or the concentration of a distinct mineral element in plant tissue is not a criterion for essentiality. Beside the essential, due to a limited selectivity in nutrient uptake, non-essential and even toxic elements can also be taken up by plant roots. On the other hand, improved analytical methods and better purification of chemicals in the 20th century resulted in discovery of further microelements essential for plant growth. This is reflected in the timescale of discoveries of the essentiality of the various micronutrients (Table 1). Nickel, with one of the lowest required plant tissue concentration, has been identified as an essential element by Brown, Welch, and Cary as late as 1987.^[3] As a consequence of continuous improvements in analytical techniques, it can be expected that further mineral elements at very low concentrations will be discovered as micronutrients for plant growth.

Definition of Plant Nutrients

The inorganic elements that are exclusively required by higher plants to maintain life and growth are called *essential mineral elements* or *mineral nutrients* (plant nutrients) by Arnon and Stout.^[4] These authors laid down the following three criteria for an element to be considered essential:

1. In the absence of the distinct mineral element, the life cycle could not be completed.
2. The function of one mineral element cannot be replaced by another mineral element.
3. The mineral element must be directly involved in the plant metabolism, either as component of a plant constituent such as plasma membrane or enzymes or required for a distinct metabolic step in enzymatic processes.

Table 1 Mineral nutrients separated for macro- and micronutrients, average concentrations in plant shoot dry matter for adequate growth and discovery of the essentiality of micronutrients for higher plants.

Element	Shoot tissue concentration ($\mu\text{mol g}^{-1}$ dry wt)	Year of discovery
Macronutrients		
Nitrogen	1000	
Potassium	250	
Calcium	125	
Magnesium	80	
Phosphorus	60	
Sulfur	30	
Micronutrients		
Chlorine	3.0	1954
Boron	2.0	1923
Iron	2.0	1860
Mn	1.0	1922
Zinc	0.3	1926
Copper	0.1	1931
Nickel	0.001	1987
Molybdenum	0.001	1938

Source: Adapted from Marschner.^[5]

According to these rather strict criteria, 14 mineral elements have been classified as essential or plant mineral nutrients (Table 1). Other mineral elements that can replace mineral nutrients in some of their less specific functions, such as sodium in the maintenance of osmotic pressure, or that can compensate for toxic effects of other elements [e.g., manganese (Mn)] such as silicon, are not essential, but can be described as beneficial elements. This definition is particularly true for the mineral element selenium. There are increasing indications that this mineral element is involved in metabolic processes of stress resistance. It has been shown that selenium at low dosage can enhance antioxidative power and diminish lipid peroxidation due to enhanced activities of glutathione peroxidase and superoxid dismutase.^[6] This would mean that selenium is required as beneficial element under adverse growth conditions such as drought, high temperature, or ultraviolet irradiation stress.

Classification and General Function of Plant Nutrients

The 14 plant mineral nutrients can be divided into two groups: six macronutrients and eight micronutrients. In general, plant tissue concentrations of macronutrients are distinctively higher than those of micronutrients (Table 1). However, such division of plant nutrients into these two groups is not always that clear. In various cases, the

differences between macro- and micronutrients are decisively less pronounced as indicated in Table 1. For example, tissue concentrations of iron and Mn can be sometimes nearly as high as concentrations of macronutrients sulfur or magnesium.

Hence, from a different viewpoint, a physiological or biochemical classification of plant mineral nutrients into four groups has been proposed by Mengel and Kirkby.^[7] The first group includes the major constituents of the organic plant material: nitrogen and sulfur. Both elements are involved in enzymatic processes and assimilated by oxidation–reduction reactions. Phosphorus, boron, and silicon constitute the second group of essential elements and show similar biochemical behavior such as the formation of esters with native alcohol groups with importance in stabilizing cell walls and/or biomembranes.

The third group of plant nutrients is made up of potassium, calcium, magnesium, Mn, and chloride. Besides more specific reactions such as conformation of enzyme proteins, bridging of reaction partners, and controlling membrane permeability, they perform some non-specific functions like establishing osmotic potential and balancing anions.

The plant nutrients of the fourth group, namely iron, copper, zinc, and molybdenum, are all predominately present as chelates in plants, often incorporated in prosthetic groups, and enable electron transport by valency change.

Besides these abovementioned essential mineral elements, the beneficial mineral elements sodium and silicon and also chlorine and selenium have to be considered as an additional group. Cobalt is also well documented as an essential element for certain higher plants. This is closely related to biological nitrogen fixation, as this mineral nutrient is a constituent of the vitamin B₁₂.^[5] For the same plant species supplied with adequate inorganic (NO_3^- and NH_4^+) or organic nitrogen forms (amino acids), cobalt is not essential or beneficial.

It will always be difficult to generalize which mineral elements are essential for plant growth. In this way, silicon and sodium are established as essential only for some higher plant species. According to Epstein,^[8] silicon is essential for rice and other wetland grasses, sugar cane, and species of the *Equisetaceae* family. This does not exclude the beneficial effect of silicon for most plant species, e.g., in the suppression of plant diseases and pests.^[5] On the other hand, sodium at relatively low concentrations is essential for some C₄ plant species excluding *Zea mays*.^[9] The function of sodium as an essential mineral element in C₄ plants such as *Panicum miliaceum* (NAD⁺ malic enzyme type) has been shown from experiments with isolated chloroplasts. In *P. miliaceum*, sodium enhances pyruvate uptake, which indicates a sodium/pyruvate cotransport through the membrane into the chloroplast, driven by a light-stimulated Na⁺ efflux pump.^[10]

Other Aspects

Besides essentiality or beneficial functions for growth and development of plants, there is an increasing interest in mineral nutrients from the viewpoint of quality for human food. Particularly, iodine, iron, zinc, and selenium are widespread limiting mineral nutrients for humans in specific regions;^[11] e.g., in an area with a diet based mainly on rice (Southeast Asia) or on millet (Africa), iron deficiency is a serious problem for children. Supreme effort is required to enhance iron tissue concentration and bioavailability in these crop plants by breeding and modern molecular approaches to overcome this nutritional problem.

CONCLUSION

Great progress in the knowledge on requirements and function of plant nutrients for plant growth has initiated the recognition of the use of mineral fertilizers in the late 19th and early 20th centuries. von Liebig's text book *Die Chemie in ihrer Anwendung auf Agricultur und Physiologie* was a milestone for the importance of mineral nutrients for plants at this time.

Considering a still growing world population, further improvements and achievements in an adequate supply of needed plant nutrients as organic or mineral fertilizers together with innovative application techniques with maximum use efficiency will be required in the future. Particularly, mineral nutrients like iodine, iron, and zinc have to be seriously considered for high-quality human food in regions with respective deficiency problems.

REFERENCES

1. Trenel, M. Zur frühgeschichte der agrikulturchemie. Berl. Forsch. Lehre Landwirtschaftswissenschaften **1956**, 86–102.
2. von Liebig, J. *Die Chemie in ihrer Anwendung auf Agricultur und Physiologie*; Druck und Verlag von Friedrich Vieweg und Sohn: Braunschweig, 1876.
3. Brown, P.H.; Welch, R.M.; Cary, E.E. Nickel: A micronutrient essential for higher plants. *Plant Physiol.* **1987**, *85*, 801–803.
4. Arnon, D.I.; Stout, P.R. The essentiality of certain elements in minute quantity for plants with special reference to copper. *Plant Physiol.* **1939**, *14*, 371–375.
5. Marschner, H. *Mineral Nutrition of Higher Plants*, 2nd Ed.; Academic Press Ltd.: London, 1995.
6. Xue, T.; Hartikainen, H.; Piironen, V. Antioxidative and growth-promoting effect of selenium on senescing lettuce. *Plant Soil* **2001**, *237*, 55–61.
7. Mengel, K.; Kirkby, E.A. *Principles of Plant Nutrition*, 5th Ed.; Kluwer Academic Publishers: Dordrecht, 2001.
8. Epstein, E. Silicon. *Annu. Rev. Plant Physiol.* **1999**, *50*, 641–644.
9. Brownell, P.F.; Crossland, C.J. The requirement for sodium as a micronutrient by species having the C₄ dicarboxylic photosynthetic pathway. *Plant Physiol.* **1972**, *49*, 794–797.
10. Ohnishi, J.; Flügge, U.-I.; Heldt, H.W.; Kanai, R. Involvement of Na⁺ in active uptake of pyruvate in mesophyll chloroplasts of some C₄ plants. *Plant Physiol.* **1990**, *94*, 950–959.
11. Cakmak, I. Plant nutrition research: Priorities to meet human needs for food in sustainable ways. In *Plant Nutrition—Food Security and Sustainability of Agro-ecosystems*; Horst, W., Ed.; Kluwer Academic Publishers: Dordrecht, 2001; 4–7.

Plant Nutrients: Intensity, Quantity, and Buffer Power

Ram C. Dalal

Department of Science, Information Technology, and Innovation,
Queensland Government, Dutton Park, Queensland, Australia

Abstract

This entry is concerned with the relative contribution of the intensity, quantity, and buffer power and their interactions in nutrient availability to plants. For routine laboratory assessment of available plant nutrients in soil, we need to measure at least two of the three parameters of availability, i.e., quantity and buffer power, or quantity and intensity, so as to improve nutrient management of ecosystems. The laboratory data can then be utilized in nutrient uptake models, which also include kinetics and diffusion of nutrients as well as plant–soil interactions. It is imperative that we understand and appropriately measure the extent of availability of plant nutrients from soil and added sources such as fertilizers and manures, for efficient and economic use of nutrients and for minimizing environmental pollution.

INTRODUCTION

Plant nutrient availability in soil is dominated by soil solution phase processes such as hydrolysis, hydration, carboxylation, and oxidation–reduction reactions. In addition, microbes and plant roots, through their synthetic and metabolic activities, mediated by both intracellular and extracellular enzymes, affect these solution phase processes. These processes determine the activity of plant nutrient ions in the soil solution and, in dilute soil solution, nutrient ion activity may be taken as concentration of nutrient or nutrient intensity factor (C_i). Sorption, desorption, precipitation, dissolution, complexation, and ion exchange with the nutrient on the solid phase provide quasi equilibrium with nutrient concentration in the soil solution. The concentration of nutrient in the entire soil mass is the quantity factor (C_s). The change in intensity factor with a change in quantity factor—i.e., dC_s/dC_i —is the nutrient buffer power. Because growing plants continually remove nutrient from the soil solution (intensity), the measure of capacity of the solid (and solution) phase nutrient (quantity) to maintain nutrient concentration during its withdrawal by plants (buffer power) is important for continual nutrient supply for plant growth. This is presented in Fig. 1.

This entry is concerned with the relative contribution of the intensity, quantity, and buffer power and their interactions in nutrient availability to plants.

INTENSITY FACTOR

The initial concentration of a nutrient in soil solution may be obtained by soil solution displacement^[1] and by equilibrating in dilute salt solutions such as 0.01 M calcium chloride (CaCl_2) or 0.02 M potassium chloride.

The relationship between low concentration range of a nutrient and plant uptake is described by the following equation:^[2]

$$U = 2\pi r \alpha C_r \quad (1)$$

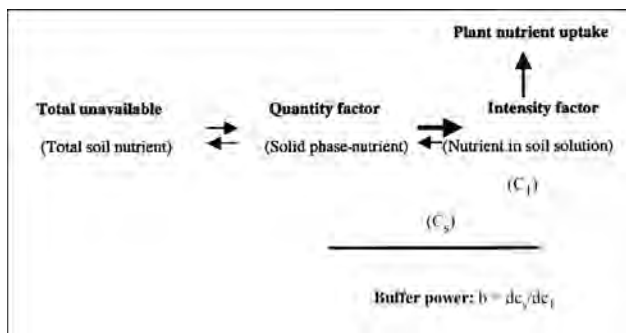
where U is the nutrient uptake by a unit m root segment, r is the radius of the root, C_r is the nutrient concentration at the root surface, and α is the root absorbing power. Diffusion and/or mass flow and root interception control the nutrient concentration at the root surface.

The nutrient uptake by mass flow is closely related to plant water uptake. In most soils, plant uptake of NO_3^- , Ca^{2+} , Mg^{2+} , and Cl^- is primarily governed by mass flow.^[1] Thus, increasing water influx increases proportionately nutrient supply by mass flow. For example, by increasing plant water transpiration from 106 L/kg of plant shoot to 444 L/kg of shoot increased Ca^{2+} uptake from 72.8 mg to 177.6 mg/pot.^[1] However, nutrient ions such as H_2PO_4^- , K^+ , and Zn^{2+} , which are strongly sorbed on soil surface, and usually low in soil solution, are primarily supplied to root surface by diffusion flow.

Nutrient concentration at the root surface cannot be measured directly. However, it can be estimated by using Fick's first law

$$F = -DA \frac{dC}{dx} \quad (2)$$

where F is the nutrient flux, dC/dx is concentration gradient, A is the area for diffusion, and D is the diffusion coefficient. The negative D means net nutrient movement from high concentration to low concentration. However, in dilute solutions D may be taken as constant over a range of concentrations in the soil solution.^[3]



For highly mobile nutrients such as NO_3 and Cl , $b \leq 1$; for these ions, b is insignificant in plant nutrient availability. For relatively immobile nutrients such as P and Zn , $b \geq 1$; significance of b in plant nutrient availability increases as its value increases.

Fig. 1 A diagrammatic representation of intensity factor, quantity factor, and buffer power.

Source: From R.C. Dalal, unpublished.

For strongly sorbed ions such as P and Zn^{2+} , initial nutrient concentration in the soil solution rarely meets plant nutrient demand. Thus, intensity factor alone does not sufficiently explain the relationship between nutrient concentration and plant nutrient uptake. For example, H_2PO_4^- activity or concentration in the soil solution is usually not related to grain yield or nutrient uptake^[4] except in the early stages of plant growth. Quantity factor becomes an important consideration for plant availability of these nutrients.

QUANTITY FACTOR

The plant available quantity of most cations is that measured by displacing cations from the exchange complex with a strong concentration of salt solution such as ammonium acetate, sodium acetate, or silver thiourea.^[5] For micronutrient cations (Fe , Mn , Zn , and Cu), complexation agents, such as ethylenediaminetetraacetic acid or diethylenetriamine pentaacetate (DTPA), have been used to reduce the precipitation and resorption of extracted cations from the soil solution.^[5]

Numerous methods are available for measuring the plant available quantity of phosphorus. These methods include extraction of P from soil by dilute and strong acids, dilute and strong alkali solutions, neutral salt solutions, and their combinations and iron oxide impregnated paper,^[6] isotopic exchangeable phosphate and ion-exchange resin,^[6] and electroultrafiltration.^[2]

The nutrient concentration of the quantity factor is 1–4 orders of magnitude greater than that of the intensity factor. For example, in 20 Australian soils, mean intensity of P was 0.03 mg/L (0.3 mg/kg), whereas mean quantity was 15 mg/kg (L value and Olsen P) or 30 mg/kg soil (Colwell P).^[6] Also, quantity of nutrients, such as P ,^[7]

K^+ , and Zn^{2+} , in soil often exceeds that taken up by plants in field conditions, although intensive cropping could remove as much quantities of P , K^+ , and Zn^{2+} as measured by the quantity factor.

The quantity factor for P , K^+ , and Zn^{2+} correlates better than the intensity factor with plant nutrient uptake. In 20 Australian soils, the intensity factor (0.01 M CaCl_2 P) accounted for only 19% each for the variation in P uptake at 35 and 150 days of wheat growth, while the quantity factor (Colwell P) accounted for 2 to 3 times more, 38 and 56% for the variation in P uptake at 35 and 150 days of wheat growth, respectively.^[4] Similarly, in eight Australian soils, the intensity and quantity factors accounted for 29 and 81%, respectively, for wheat grain yield in the field.^[4] This also applies to Zn^{2+} availability. For example, in 13 Vertisols, relative yield response to Zn application was better correlated with DTPA-extractable Zn (C_s), $r^2 = 0.69$, than soil solution Zn (C_i), $r^2 = 0.56$.^[8]

BUFFER POWER

The availability of nutrients to growing plants is not only influenced by nutrient intensity and the quantity but also by the ability of soil to replace the nutrients as plants take it up from soil solution. The effective diffusion of the nutrient to the root surface depends on the soil's buffer power for that nutrient, volumetric soil moisture, and the impedance factor. This is described by an adaptation of Eq. 2 to the soil system by the following equation:^[9]

$$D_e = D_1 \Theta f_1 \frac{dC_i}{dC_s} + R \quad (3)$$

where D_e is the effective diffusion coefficient of nutrient ion in soil, D_1 is the diffusion coefficient of ion in water, Θ is the volumetric soil water content, f_1 is an impedance factor or tortuosity factor that is measured using non-adsorbed ions such as Cl^- ions, C_i is the concentration of nutrient ion in the soil solution (intensity factor), C_s is the quantity of nutrient ion in the entire mass of soil (quantity factor), and R is the residual term accounting for diffusion through the adsorbed phase, which is probably negligible in neutral soils for most ions.

The nutrient buffer power, b , is the reciprocal of dC_i/dC_s , i.e.,

$$b = \frac{dC_s}{dC_i} \quad (4)$$

The nutrient buffer power, b , is measured from either an adsorption isotherm or desorption isotherm. In adsorption isotherm, simulating fertilizer application, known amount of nutrient is added to soil and the soil incubated for periods from minutes (exchangeable cations) to several months to attain quasi equilibrium between the

sorbed or exchangeable ions and those in the soil solution. At the end of the incubation period, the concentration of nutrient in the soil solution is measured and C_1 and C_s are calculated. Desorption isotherms are constructed from desorption of nutrient by an electrolyte, electro-ultrafiltration, or ion-exchange resin, simulating the nutrient uptake by roots.

For a narrow range in nutrient concentration, there is a linear relationship between C_s and C_1 , i.e.,

$$C_s = b_1 C_1 + a_1 \quad (5)$$

where b_1 is the distribution coefficient and a_1 is a constant. The buffer power, b , is calculated from the following equation:^[10]

$$b = \rho b_1 + \theta \quad (6)$$

where ρ is the bulk density and θ is the volumetric moisture content.

However, for a wide range in quantity and intensity, the relationship between C_s and C_1 is frequently described by the Freundlich isotherm:

$$C_s = k C_1^n \quad (7)$$

where k and n are constants, k is related to nutrient sorption capacity of the soil, and n is related to nutrient bonding energy to soil or curvilinearity between the corresponding C_s and C_1 values in a soil.

Langmuir isotherm has also been used to describe the relationship between C_s and C_1 values in a soil,^[4,11]

$$C_s = \frac{MBC_1}{1 + bC_1} \quad (8)$$

On differentiating with respect to C_1 , Eq. 8 becomes

$$\frac{dC_s}{dC_1} = \frac{MB}{(1 + bC_1)^2} \quad (9)$$

As $C_1 \rightarrow 0$, Eq. 9 becomes

$$\frac{dC_s}{dC_1} = MB \quad (10)$$

where M is Langmuir adsorption maximum, B is bonding energy term, and the product, MB , is the maximum buffering capacity.

For 20 Australian soils, the P buffer power varied from 32 to 364, except in one soil, an oxisol, which had the value of 3476.^[6] For 33 United States and Canadian soils, the values varied from 33 to 299.^[12]

The P buffer values are much lower, from 19 to 75, for Central European soils.^[3] For 14 Australian Vertisols, Zn buffer power varied from 217 to 790.^[8] The K buffer power varied from 142 to 592 for southern Indian soils.^[3] The relative importance of nutrient buffer power, intensity, and quantity factors is illustrated for a number of experiments in Table 1.

PERSPECTIVES

Other factors that affect nutrient availability from soil such as climate (temperature and moisture), pH, nutrient placement, soil biota, and plant type should also be considered along with the parameters of plant nutrient availability. For strongly sorbed ions, nutrient concentration in the soil solution is usually low. It is not determined routinely in soil testing laboratories. While quantity of the nutrient is routinely measured in soil testing services, accounting for the variation in plant nutrient uptake almost always improves by considering the soil's nutrient buffer power. Nutrient uptake models also include buffer power, so it should be widely measured. Optimal assessment of plant available nutrients from soil as well as by fertilizer applications is essential to ensure the efficient use of nutrients for economic food and fiber production while minimizing nutrient loss/pollution to the environment. While techniques for measuring buffer power, quantity, and intensity factors should be further refined, consideration of buffer power along with quantity/intensity factors^[13] will improve estimates and hence the efficient use of plant available nutrients from soil.

CONCLUSION

For routine laboratory assessment of available plant nutrients in soil, we need to measure at least two of the three parameters of availability, i.e., quantity and buffer power, or quantity and intensity, so as to improve

Table 1 Percentage of variation in nutrient uptake accounted for by intensity factor, quantity factor, and a combination of intensity or quantity and buffer power.

Nutrient	Intensity (C_1)	Quantity (C_s)	Buffer power (b)	C_1 and b	C_s and b	References
P	17	75	27	49	86	[4]
P	15	51	—	79	89	[4]
K	—	6	—	—	19	[3]
Zn	13	—	—	38	—	[3]
Zn	27	27	0.08	27	29	[8]

nutrient management of ecosystems. The laboratory data can then be utilized in nutrient uptake models, which also include kinetics and diffusion of nutrients as well as plant–soil interactions. It is imperative that we understand and appropriately measure the extent of availability of plant nutrients from soil and added sources such as fertilizers and manures, for efficient and economic use of nutrients and for minimizing environmental pollution.

REFERENCES

1. Barber, S.A. *Soil Nutrient Bioavailability: A Mechanistic Approach*, 2nd Ed.; John Wiley: New York, 1996; 1–414.
2. Mengel, K. Dynamics and availability of major nutrients in soils. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer-Verlag: New York, 1988; Vol. 2, 65–131.
3. Nair, K.P.P. The buffering power of plant nutrients and effects on availability. In *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press: Toronto, 1996; Vol. 57, 237–287.
4. Dalal, R.C.; Hallsworth, E.G. Evaluation of the parameters of soil phosphorus availability factors in predicting yield response and phosphorus uptake. *Soil Sci. Soc. Am. J.* **1976**, *40*, 541–546.
5. Rayment, G.E.; Higginson, F.R. *Australian Laboratory Handbook of Soil and Water Chemical Methods*; Inkata Press: Melbourne, 1992; 1–330.
6. Dalal, R.C.; Hallsworth, E.G. Measurement of isotopic exchangeable soil phosphorus and interrelationship among parameters of quantity, intensity, and capacity factors. *Soil Sci. Soc. Am. J.* **1977**, *41*, 81–86.
7. Moody, P.W.; Bolland, M.D.A. Phosphorus. In *Soil Analysis: An Interpretation Manual*; Peverill, K.I., Sparrow, L.A., Reuter, D.J., Eds.; CSIRO Publishing: Melbourne, 1999; 187–220.
8. Dang, Y.P.; Dalal, R.C.; Edwards, D.G.; Tiller, K.G. Understanding zinc reactions in vertisols to predict zinc responses by wheat. *Plant Soil* **1993**, *155/156*, 247–250.
9. Nye, P.H.; Tinker, P.B. *Solute Movement in the Soil-Root System*; Blackwell Scientific Publications: Oxford, 1977; 1–342.
10. van Rees, K.C.J.; Comerford, N.B.; Rao, P.S.C. Defining soil buffer power: Implications for ion diffusion and nutrient uptake modelling. *Soil Sci. Soc. Am. J.* **1990**, *54*, 1505–1507.
11. Holford, I.C.R. Soil phosphorus: Its measurement, and its uptake by plants. *Aust. J. Soil Res.* **1997**, *35*, 227–239.
12. Kovar, J.L.; Barber, S.A. Phosphorus supply characteristics of 33 soils as influenced by seven rates of P addition. *Soil Sci. Soc. Am. J.* **1988**, *52*, 160–165.
13. Probert, M.E.; Moody, P.W. Relating phosphorus quantity, intensity, and buffer capacity to phosphorus uptake. *Aust. J. Soil Res.* **1998**, *36*, 389–393.

Plant Nutrients: Sufficiency and Requirements

Kanwar L. Sahrawat

*International Crops Research Institute for the Semi-Arid Tropics (ICRISAT),
Panthacheru, India*

Abstract

The research on determining the nutrient sufficiency and nutrient requirements of plants is a prerequisite for the efficient use of purchased inputs of nutrients and for making the overall agricultural production more efficient through optimal use of plant nutrients. However, there have been problems in consistency of the results, which are greatly influenced by growing conditions and complex interactions among environmental, cultural, and genetic conditions. Despite these limitations, the knowledge of nutrient sufficiency and requirement is helpful in improving the prediction of plant growth and yield. A combination of soil and plant tests may be more effective in determining the nutrient sufficiency and requirements of crops. While soil tests are effective in determining the long-term nutrient requirement of crops, plant tests are necessary to diagnose and correct the nutrient deficiency of the crops.

INTRODUCTION

To meet the ever-increasing need for food, feed, and fiber, the production has to keep pace with the population growth. This can be achieved by enhancing the soil fertility and making crop production efficient through optimal use of plant nutrients. Assessment of the nutrient sufficiency and requirements is an integral component of research on efficient and rational use of plant nutrient inputs. The objective of this entry is to provide a concise summary of the major developments in the research on plant nutrient sufficiency and requirements. The emphasis is placed on the salient research results from field studies. Readers are referred to an excellent text by Black^[1] for detailed discussion and summary of the past research.

BACKGROUND INFORMATION

The need for food, feed, and fiber production is ever increasing because of the continuous, rapid population growth. The need is most pressing in the developing world.^[2] Apart from water shortages, soil infertility is the major constraint on the plant growth and yield in most of tropical regions. The increased crop production can be achieved through enhanced soil fertility.^[1] Soil fertility can only be sustained if the nutrients removed from the soil are replenished through addition. Also, crop production needs to be made efficient through optimal use of plant nutrients. Assessment of the nutrient sufficiency and requirements is an important component of research for efficient and rational use of nutrients from external sources.

Nutrient Sufficiency

Nutrient sufficiency or critical nutrient concentration, as more commonly termed, is a relative term because an absolute sufficiency cannot be determined. Nutrient sufficiency is a measure of nutrient concentration in the plant and is determined by plant analysis. It is preferably expressed as a range of concentrations rather than a single concentration. Sufficiency of a given nutrient lies between critical deficiency value and an excess or toxic concentration. Alternatively, nutrient sufficiency is the range of concentrations for which the yield reduction and nutrient stress symptoms are not taken into account.^[3] A critical deficiency concentration of a nutrient is the one that corresponds to a yield that is 10% below the maximum on the low side, and a critical toxicity concentration of a nutrient is the one that corresponds to a yield that is 10% below the maximum on the high side.^[4] It is perhaps more useful to consider critical concentration as the concentration that separates plants that would respond to addition of more nutrients from those that would not respond.^[5] For practical purposes, economic criterion would appear more appropriate and the critical nutrient concentration may be termed as critical economic concentration.^[6]

Nutrient Requirement

Nutrient requirement is the amount of nutrient required to achieve a yield target that is normally 90% of the maximum yield obtained under optimal nutrient regime. As for nutrient sufficiency, economic considerations are equally relevant for determining the nutrient requirements of various crop plants.^[1] Nutrient requirements include both external

and internal nutrient requirements. External nutrient requirement is the concentration in the growing medium, usually soil, while the internal requirement is the concentration in the plant tissue at a yield target, which is normally 90% of the maximum yield.

DETERMINING NUTRIENT SUFFICIENCY AND REQUIREMENTS

Soil and plant analyses are used for determining the nutrient requirements of crops, while plant analysis is used for establishing the nutrient sufficiency under well-defined conditions of experimentation and collection of soil and plant samples. Data on nutrient sufficiency range in plant tissue of selected grain crops^[7] for macronutrients are summarized in Table 1.

Since the early finding that the chemical composition of plants changes with the nutrient supplies, attempts have been made to use the elemental concentrations for evaluating nutrient requirements of crops. For historical developments covering the period up to 1990, the readers are referred to the excellent text entitled *Soil Fertility Evaluation and Control* by C.A. Black.^[1] The literature is briefly covered in this entry. Evidently, much research has been done in determining the concentrations of nutrient elements as indexes of the sufficiency of nutrient supplies for crop growth and for determining nutrient requirements.^[2,7,8]

A book entitled *Plant Analysis Handbook* by H.A. Mills and J.B. Jones, Jr.^[7] provides tables of interpretive plant analysis data for over 1000 individual horticultural, agronomic, and plantation crops. The publication provides data on sufficiency or critical concentration range for macro- and micronutrient elements for various crop plants where the deficiency, sufficiency, and toxicity concentrations have been established over a wide range of conditions in the

solution culture to field. In addition, survey range and survey average values for various nutrient elements are also presented for growing conditions under which the upper and lower limits of nutrient concentrations have not been as clearly defined as in the case of sufficiency range. Survey average value refers to the nutrient level found in healthy, normal plants without the presence of visual symptoms on the foliage showing deficiency or toxicity. Another comprehensive treatise on *Plant Analysis: An Interpretation Manual* edited by Reuter and Robinson^[8] is also an excellent source of information on nutrient levels for deficiency, toxicity, and sufficiency for temperate and tropical crops, pasture species, fruits, vines, nuts, vegetable crops, ornamentals, and forest plantations. The treatise discusses in detail the concepts and principles for interpreting plant analysis and describes nutrient deficiency and toxicity symptoms in the plant, apart from providing concentration ranges of nutrients covering the range from deficiency to toxicity.

The determination of nutrient sufficiency and requirements is based on the relationships with plant growth and yield. The nutrient elemental composition of the plant at optimal yield should approximate the nutrient sufficiency levels, expressed either as individual nutrient concentrations or ratios of the various nutrient elements.^[1,8] Before soil and plant tests can be used for determining nutrient sufficiency and requirements, appropriate standards should be developed through field experimentation.^[9] Field experiments are considered essential for calibration because standards developed under controlled conditions, e.g., greenhouse pots may not be representative of the real world situation and may not hold under field conditions. Using the primary standards developed under field conditions, the relative sufficiency and requirements of nutrients can be determined under non-experimental conditions. Although it is generally accepted that the determination of sufficiency requirements for nutrients leads to improved

Table 1 The range of sufficiency levels of N, P, and K in plant tissue for selected grain crops.

		Sufficiency range concentration (%)		
Crop growth stage	Plant part	N	P	K
Corn or maize (<i>Zea mays</i>)				
Prior to tasseling	Leaf below the whorl	3.00–3.50	0.25–0.45	2.00–2.50
Rice (<i>Oryza sativa</i> L.)				
Maximum tillering	Newest, fully developed leaf	2.80–3.60	0.10–0.18	1.20–2.40
Barley (<i>Hordeum vulgare</i> and <i>Hordeum distichon</i>)				
Emergence of head from boot	Whole tops	1.75–3.00	0.20–0.50	1.50–3.00
Sorghum (<i>Sorghum bicolor</i>)				
37–56 days after planting	Newest, fully developed leaf	3.20–4.20	0.13–0.25	2.00–3.00
Winter wheat (<i>Triticum aestivum</i>)				
Just before heading	Top two leaves	1.75–3.00	0.20–0.50	1.5–3.00
Spring wheat (<i>Triticum aestivum</i>)				
Emergence of head from boot	Whole tops	2.00–3.00	0.20–0.50	1.50–3.00

prediction of yields, it is emphasized that these values depend on the growing conditions.^[1]

Several equations, which are based on the response of crop yield or relative yield or growth to nutrient concentrations in soil or plant, have been proposed for determining the nutrient requirements of crops. Black^[1] provides an excellent historical perspective to the developments in research on the use of soil and plant tests for determining the nutrient requirements of crops. It is evident from the discussion that while qualitative aspects of nutrient status are useful and at times, especially in the past, was considered acceptable, no generally satisfactory system of testing plants for nutrient deficiencies has emerged. This is a major obstruction in the use of nutrient concentrations for determining requirements in a quantitative and consistent manner. The lack of consistency, however, is not entirely unexpected in view of the fact that the concentrations of various nutrient elements are determined by the total mass of the plant, which in turn is influenced by the growing conditions relative to environment, nutrient supply and interactions among various nutrient elements, and the genetics of the crop.

Soil characteristics greatly influence the use of soil tests for determining the nutrient sufficiency or critical concentration levels for various nutrient elements. For example, one study with grain sorghum under rainfed cropping on nearby Alfisol and Vertisol sites at the International Crops Research Institute for the Semi-Arid Tropics, Patancheru (India) showed that on the Vertisol site, 90% relative grain yield was obtained at 2.8 mg/kg Olsen extractable phosphorous (P) while on the Alfisol site, 90% relative grain yield was achieved at 5.0 mg/kg Olsen P.^[10] These results demonstrate that a single critical limit of available P does not hold true for sorghum on the two soil types under similar agroclimatic conditions and that the critical limit is lower for the clayey Vertisol with higher phosphate buffering capacity than the sandy Alfisol with lower buffering capacity.

Field study with upland rice on an Ultisol in the humid forest zone of Ivory Coast showed that the critical limit of Bray 1 P in the soil at 90% relative rice grain yield varied from 12.5 and 15.0 mg P/kg soil for the four cultivars tested. The critical or sufficiency value tended to be lower for the P-efficient rice cultivars.^[11] Using plant tissue P concentration as the criterion, the critical concentration of P in the whole rice plant tops at the tillering stage, in the 90% relative grain yield, was found to be 2 g P/kg for the four rice cultivars. It was indicated that while the rice cultivars differed in the external P requirement (concentration in the growing medium), they did not differ in their internal P requirement (concentration in the plant tissue).^[11,12] In these field experiments, a significant linear relationship was observed between total P uptake in the biomass and rice grain or rice straw yield.^[12]

Sahrawat et al.^[13] provided results on the calibration of plant P test for predicting sorghum grain yield under rainfed

cropping of a Vertisol site (Typic Haplustert) in the semi-arid zone of India. Despite variability in the rainfall received during the 3 years of field study, the leaf (newest, fully developed leaf at 50% flowering stage of the crop) P concentration was found to be linearly related to grain yield (r^2 varied from 0.724 to 0.993). The relationships between leaf P and sorghum yield during the 3 years of study were described by the following equations:

In 1987

$$\text{Grain yield (t/ha)} = 4.057 + 28.300 \text{ leaf P (\%)} r^2 = 0.993 \quad (1)$$

In 1988

$$\text{Grain yield (t/ha)} = -7.693 + 50.0085 \text{ leaf P (\%)} r^2 = 0.724 \quad (2)$$

In 1989

$$\text{Grain yield (t/ha)} = -3.414 + 21.787 \text{ leaf P (\%)} r^2 = 0.979 \quad (3)$$

The critical or the sufficiency leaf P concentration at 90% of the maximum yield was found to be about 2.5 g P/kg (Fig. 1). Phosphorus content in the grain was not significantly correlated to yield. Two important points emerged from this study: 1) linear relationship between leaf P and yield over a range of applied P and 2) the Cate and Nelson split method^[14] of graphic presentation of relationship between leaf P and relative sorghum grain yield was found to be useful and had the practical

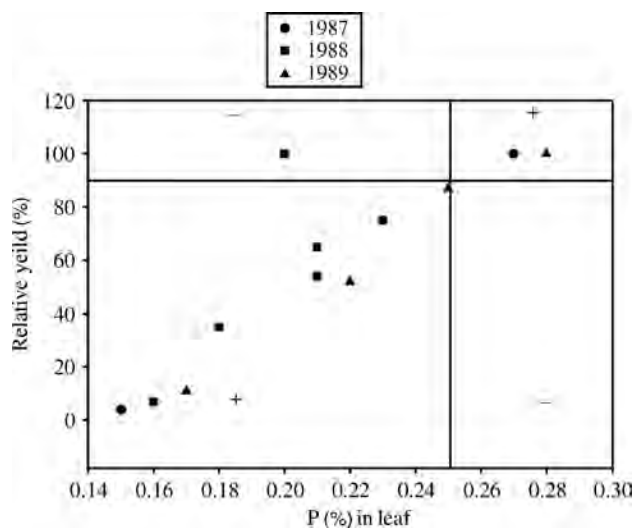


Fig. 1 Relationship between relative grain yield and P content in the index leaf (new, fully developed leaf) at flowering in sorghum grown on a Vertisol for 3 years (1987–1989). Each point represents an average value of four replications.

Source: From Sahrawat, Rahman, et al.^[13]

advantage in dividing the data into responsive and non-responsive populations.^[13]

SIMPLE MODELS FOR DETERMINING NUTRIENT REQUIREMENTS

Sahrawat^[15] evaluated two methods for determining the fertilizer phosphorus requirement (FPR) of sorghum. The first method used a simple method based on P uptake^[16,17] using the following equation:

$$\text{FPR} = (U_p - U_0)/\text{PRF} \quad (4)$$

where U_p is P uptake at a given yield, U_0 is P uptake from unfertilized soil, and PRF is the P recovery fraction of applied P. The parameters U_0 , U_p , and PRF were determined in a three-year field experiment with sorghum grown on a calcareous Vertisol site under rainfed conditions. The second method, proposed by the author, was based on the P applied, P uptake, and grain yield relationships. First, P uptake at a given sorghum yield was determined from the relationship between total P uptake and grain yield. The amount of fertilizer P applied for the given P uptake and grain yield was then determined from the relationship between P applied and P uptake. Based on 3 years results, it was found that the fertilizer P applied and total P uptake by sorghum were significantly and linearly related, which is described by regression Eqs. 5 and 6

$$\begin{aligned} \text{P uptake (kg/ha)} &= 1.67 + 0.200 \text{ P applied (kg/ha)} \\ (r^2 &= 0.97; n = 12) \end{aligned} \quad (5)$$

Total P uptake by the crop, in turn, was closely linearly related to sorghum grain yield and described by the following regression equation:

$$\begin{aligned} \text{P uptake (kg/ha)} &= 0.51 + 2.62 \text{ grain yield (t/ha)} \\ (r^2 &= 0.95) \end{aligned} \quad (6)$$

There was a close agreement between the observed values of FPR and the predicted values determined by the two methods. These results suggested that the simple models based on P uptake could be utilized for determining the nutrient requirements of crops.

Evaluation of the two methods described, for determining the fertilizer P requirements of upland rice cultivars grown for two seasons (1992–1993) under a range of applied P on an Ultisol site in the humid forest zone of Ivory Coast, showed that there was a good agreement between the observed values of FPR and the predicted values of FPR that were determined by the two methods. This indicates the usefulness of these simple methods based on P uptake for determining the P requirements of crops.^[18] As observed for sorghum, fertilizer P applied and P uptake and grain yield and P uptake by four upland rice cultivars were significantly and linearly related. Overall, the results

obtained with upland rice on an Ultisol site in the humid forest zone and sorghum on a Vertisol site in the semiarid tropical zone indicate a wide applicability of these simple models for determining P requirements of crops and merit further evaluation.

It was further indicated that using the Cate and Nelson split method,^[14,19] of graphic presentation of relationships between P applied and relative rice grain yield or between P uptake and relative yield provided a simple procedure for determining the P requirement of rice.^[18]

The author's unpublished results showed that a significant and linear relationship exists between fertilizer nitrogen (N) (as urea) and N uptake, and between grain yield and N uptake for lowland rice cultivars grown under irrigated conditions on Alfisols in central Ivory Coast. It was further indicated that these relationships could be used for determining the N requirements of lowland rice cultivars.

In a study, Pathak et al.^[20] used the Quantitative Evaluation of Fertility of Tropical Soils (QUEFTS) model^[21] for estimation of N, P, and potassium (K) requirements and fertilizer recommendations for targeted yields of wheat in India. The model considers the interactions of N, P, and K, and climate adjusted potential yield region wise in the country. Published data from field experiments conducted between 1970 and 1998 across the wheat-growing environments of India, covering a wide range of soils and climatic conditions, were used to capture the environmental variability effects on wheat yield. The relationships between soil's inherent N supply and soil organic carbon, P supply and Olsen extractable P in soil, and K supply and ammonium acetate extractable soil K were established. The results showed that required N, P, and K accumulation in the plant biomass for 1 t grain yield was 23.1, 3.5, and 28.5 kg, respectively. The observed yields of wheat with different amounts of N, P, and K were in agreement with the values predicted by the model, indicating that the QUEFTS model can be used for determining nutrient requirements and fertilizer recommendations for wheat production.

CONCLUSION

The research on determining the nutrient sufficiency and nutrient requirements of plants is a prerequisite for the efficient use of purchased inputs of nutrients and for making the overall agricultural production more efficient through optimal use of plant nutrients. However, there have been problems in consistency of the results, which are greatly influenced by growing conditions and complex interactions among environmental, cultural, and genetic conditions.^[1] Despite these limitations, the knowledge of nutrient sufficiency and requirement is helpful in improving the prediction of plant growth and yield. A combination of soil and plant tests may be more effective in determining the nutrient sufficiency and requirements of crops. While soil tests are effective in determining the long-term nutrient

requirement of crops, plant tests are necessary to diagnose and correct the nutrient deficiency of the crops.^[13,22,23]

There is need for future research to strengthen the use of models for determining nutrient requirements and fertilizer recommendations for crop production, as field experimentation is expensive and at times not practical.

REFERENCES

1. Black, C.A. *Soil Fertility Evaluation and Control*; Lewis Publishers: Boca Raton, 1993; 155–269.
2. IFPRI. *Feeding the World, Preventing Poverty and Protecting the Earth: A 2020 Vision*; International Food Policy Research Institute (IFPRI): Washington, 1996.
3. Jones, J.B. Jr.; Eck, H.V.; Voss, R. Plant analysis as an aid in fertilizing corn and grain sorghum. In *Soil Testing and Plant Analysis, Soil Science Society of America Series 3*; Westerman, R.L., Ed.; 3rd Ed.; Soil Science Society of America: Madison, 1990; 521–547.
4. Ohki, K. Zinc nutrition related to critical deficiency and toxicity levels for sorghum. *Agron. J.* **1984**, *76*, 253–256.
5. Escano, C.R.; Jones, C.A.; Uehara, G. Nutrient diagnosis in corn grown on hydric dystrandepts. I. Optimum nutrient concentration. *Soil Sci. Soc. Am. J.* **1981**, *45*, 1135–1139.
6. Malavolta, E.; Da Cruz, V.F. A meaning for foliar diagnosis. In *Recent Advances in Plant Nutrition*; Samish, R.M., Ed.; Lavoisier Publishing: New York, 1971; Vol. 1, 1–3.
7. Mills, H.A.; Jones, J.B. *Plant Analysis Handbook: A Practical Sampling, Preparation, and Interpretation Guide*; MicroMacro Publishing, Inc.: Athens, 1996.
8. Reuter, D.J.; Robinson, J.B. *Plant Analysis: An Interpretation Manual*, 2nd Ed.; CSIRO: Australia, 1997.
9. Westerman, R.L. *Soil Testing and Plant Analysis, Soil Science Society of America Series 3*, 3rd Ed.; Soil Science Society of America: Madison, 1990.
10. Sahrawat, K.L.; Pardhasaradhi, G.; Rego, T.J.; Rahman, M.H. Relationship between extracted phosphorus and sorghum yield in a vertisol and an alfisol under rainfed cropping. *Fert. Res.* **1996**, *44*, 23–26.
11. Sahrawat, K.L.; Jones, M.P.; Diatta, S. Extractable phosphorus and rice yield in an ultisol of the humid forest zone in West Africa. *Commun. Soil Sci. Plant Anal.* **1997**, *28* (9,10), 711–716.
12. Sahrawat, K.L.; Jones, M.P.; Diatta, S. Plant phosphorus and rice yield in an ultisol of the humid forest zone in West Africa. *Commun. Soil Sci. Plant Anal.* **1998**, *29* (7,8), 997–1005.
13. Sahrawat, K.L.; Rahman, M.H.; Rao, J.K. Leaf phosphorus and sorghum yield under rainfed cropping of a vertisol. *Nutr. Cycl. Agroecosyst.* **1999**, *54*, 93–97.
14. Cate, R.B.; Nelson, L.A. *A Rapid Method for Correlation of Soil Test Analysis with Plant Response Data*; International Soil Testing Series Technical Bulletin 1; North Carolina Agricultural Experiment Station: Raleigh, 1965.
15. Sahrawat, K.L. Assessing the fertilizer phosphorus requirement of grain sorghum. *Commun. Soil Sci. Plant Anal.* **1999**, *30* (11,12), 1593–1601.
16. Driessen, P.M. Nutrient demand and fertilizer requirements. In *Modelling of Agricultural Production: Weather, Soils and Crops*; van Keulen, H., Wolf, J., Eds.; PUDOC: Wageningen, 1986; 182–202.
17. Cornforth, I.S.; Metherell, A.K.; Sorn-Srivichai, P. Assessing fertilizer requirements. In *Phosphorus Requirements for Sustainable Agriculture in Asia and Oceania*; International Rice Research Institute: Manila, Mar 6–10, 1990; 157–166.
18. Sahrawat, K.L. Determining fertilizer phosphorus requirement of upland rice. *Commun. Soil Sci. Plant Anal.* **2000**, *31* (9,10), 1195–1208.
19. Dahnke, W.C.; Olson, R.A. Soil test calibration, and the recommendation. In *Soil Testing and Plant Analysis*; Westerman, R.L., Ed.; 3rd Ed.; Soil Science Society of America: Madison, 1990; 45–71.
20. Pathak, H.; Aggarwal, P.K.; Roetter, R.; Kalra, N.; Bandyopadhyaya, S.K.; Prasad, S.; Van Keulen, H. Modelling the quantitative evaluation of soil nutrient supply, nutrient use efficiency, and fertilizer requirements of wheat in India. *Nutr. Cycl. Agroecosyst.* **2003**, *65* (2), 105–113.
21. Smalling, E.M.A.; Janssen, B.H. Calibration of QUEFTS, a model predicting nutrient uptake and yields from chemical fertility indices. *Geoderma* **1993**, *59* (1–4), 21–44.
22. Sahrawat, K.L.; Sika, M. Direct and residual phosphorus effects on soil test values and their relationships with grain yield and phosphorus uptake of upland rice on an ultisol. *Commun. Soil Sci. Plant Anal.* **2002**, *33* (3,4), 321–332.
23. Rashid, A.; Awan, Z.I.; Ryan, J. Diagnosing phosphorus deficiency in spring wheat by plant analysis: Proposed critical concentration ranges. *Commun. Soil Sci. Plant Anal.* **2005**, *36* (4–6), 609–622.

Plastic Properties

Gunnar Kirchhof

New South Wales Department of Agriculture, Tamworth, New South Wales, Australia

Abstract

The application of a force to soil can result in an elastic, non-permanent, deformation that only lasts while the force is applied or a plastic, permanent deformation that persists after the load is removed. The latter may, or may not, result in a decrease in the soil volume. How easily and how much the volume changes is determined by the soil's plasticity. It is an important factor influencing ease and outcome of soil tillage operations. Soil plasticity is caused by the lubricating film of water surrounding soil particles that allow the soil to change shape without rupturing upon the application of forces. As an inherent soil property, it is governed primarily by the surface area of the soil particles. Plasticity increases with increasing clay content, activity of clay minerals, position of the adsorbed cations in the lyotropic series, and organic matter content.

DEFINITION

The adverb plastic is derived from the Latin noun *plasticus* [from Greek: plastikós (plássein)] meaning form or mold. Plasticity is the property of a solid body being molded, receiving form, or being brought into form. If the force leading to plastic deformation exceeds a yield value, permanent change in shape or size occurs. The definition of plasticity for soils includes the conditions that the deformation must not result in rupture and that the shape remains unchanged after water is removed during drying.^[1]

Although soil plasticity is closely related to soil consistence or consistency, there are two main differences: 1) Consistence and consistency usually refer to the resistance to deformation at a moisture content of the soil in the field^[2] and are a manifestation of the forces of cohesion and adhesion within the soil at various water contents.^[3] Plasticity is usually described with the soil nearly saturated with water^[4] or at water contents where the soil can easily be deformed. 2) Soil consistency or consistence is usually evaluated on undisturbed soil aggregates, fragments, or peds. Plasticity is assessed on disturbed, often wetted, and molded soil material. The latter can therefore be regarded as an inherent soil property that does not change with the soil's macro structure.

The Atterberg limits^[5] are most commonly used as parameters to describe soil plasticity, which is measured by the plasticity index (PI). PI is equal to the water content difference at the liquid limit (LL) and the plastic limit (PL). Although rarely used by the agriculturally oriented discipline of soil science, soil engineers frequently use the Casagrande Plasticity Chart^[6] to describe soil plasticity. It is a plot of the PI against the LL. Many fine-grained soils lie scattered along a line on the Casagrande Plasticity Chart:

$$PI = 0.73(LL - 20) \quad (1)$$

The units for the PI and LL are g water/100 g dry soil.

The line defined by this equation is called the Casagrande A-line. Engineering classification schemes use the position of soils on the chart in terms of their relation to the A-line for classification purposes. Soils with similar LLs tend to have greater soil toughness and dry strength when the PI increases.^[7]

The PL is generally regarded as the most suitable soil water content for seedbed preparation. Compared with wetter or drier soil water contents, it tends to give the best tilth and least cloddiness. However, the soil's compactibility tends to be the greatest at soil water contents slightly wetter than the PL.

THEORY OF SOIL PLASTICITY

Water Films

Water in soil is held inside pore spaces and is adsorbed along the surfaces of the soil particles as a thin water film. The latter determines the soil's plastic behavior because the water film acts as a lubricant between particles. The lubricating effect is most pronounced if particles are small, are platy, and are in face-to-face contact.

Cohesion

The same forces that determine the soil's behavior during the application of stress govern the dynamics of the soil's plastic behavior: cohesion, adhesion, and angle of internal friction. Soil properties that determine the response to these forces, and therefore degree of plasticity, are texture, the presence of cementation agents, cation exchange capacity and type of cations, salt concentration, soil water content, pH, and organic matter content.^[8–10]

Cohesion can be separated into true and apparent cohesion. Apparent cohesion holds particles together through capillary forces and interlocking of rough particle surfaces.^[7,11] It allows sands to be deformed without rupture, but this deformation is not plastic deformation since molded sands fall apart when dry. True cohesion holds particles together through electrostatic and electromagnetic forces of attraction and cementing agents. True cohesion is the force field for plasticity.

FACTORS AFFECTING PLASTICITY

Clay Content and Clay Type

The specific surface area of soils and thus their plasticity are largely determined by clay content. However, despite a highly significant empirical relationship between the Atterberg limits and clay content, the reliable prediction of one from the other is often not possible, unless similar soil types are considered.^[12–16]

The plasticity of different types of clay minerals is related to the size of their surface area, shape of the clay lattice, and the type of exchangeable cations. Platy or sheet-like structures are more plastic than tetrahedra-type structures. An assessment of plasticity, or any other soil property, in relation to clay mineralogy must bear in mind that natural soils usually do not contain appreciable amounts of pure primary clay minerals. Therefore, only a broad relationship can be indicated when relating clay mineralogy to plasticity.

Soils can be differentiated into the following three groups by the way in which their physical properties relate to plasticity.^[17]

1. Soils where the void volume changes little with changes in soil water content. These soils are generally governed by low activity or 1:1 clay minerals such as kaolinite, halloysite, serpentine, and berthierine. Owing to their relatively low specific surface area ($10\text{--}150\text{ m}^2\text{ g}^{-1}$) and low cation exchange capacity ($<15\text{ cmol} + \text{kg}^{-1}$), they have low values of plasticity.
2. Soils where the void space changes reversibly with changes in soil water content. These soils are typically governed by high activity or 2:1 clay minerals. The most common types of clay minerals in soils are illites, vermiculites, smectites, and chlorites. Their specific surface area is typically $50\text{--}800\text{ m}^2\text{ g}^{-1}$ and cation exchange capacity $20\text{--}200\text{ cmol} + \text{kg}^{-1}$, thus having a much greater plasticity than the low activity clay minerals.
3. The last group are soils where the void volume changes with soil water content but is largely irreversible. These soils contain appreciable amounts of allophane. Upon drying, the plasticity of these soils decreases irreversibly possibly due to the permanent collapse of a large

number of dehydrated micropores.^[18] The degree with which allophane soils change their plastic behavior upon drying can be used for classification purposes.

The impact of other clay size particles in soils [sesquioxides, manganese and silicon oxides, and calcium (Ca) and magnesium (Mg) carbonates] on soil plasticity is largely associated through their ability to cement particles together and thus tends to decrease plasticity or causes subplastic behavior.

Cations

The impact of different types of cations on the plasticity of soils is related to the size of the hydrated cation as well as to its charge, i.e., their position in the lyotropic series. Divalent cations (Ca and Mg) tend to increase both Atterberg limits and the plasticity of soils. Due to its larger effective ionic radius, Mg has a stronger effect on the Atterberg limits than Ca.^[19] Monovalent cations [potassium (K) and sodium (Na)] lower both Atterberg limits, but the K cation tends to decrease and the Na cation tends to increase plasticity.^[20]

pH

The effect of pH on plasticity is indirect and associated with the influence of pH on cation exchange capacity, amount of exchangeable bases, and soil organic matter.

Organic Carbon

Soil organic matter has a high absorptive capacity for water. Lubricating water films tend not to form unless the hydration of organic matter is fairly complete. Therefore, organic matter increases the Atterberg limits but tends to have little effect on the PI.^[1,13–16] Besides the total amount of organic matter, type or quality of organic matter influences plasticity. On soils with similar texture and organic matter content, the presence of long-chained lipids tends to shift the Atterberg limit to higher soil water contents.^[21]

TYPES OF PLASTICITY

Three different types of plasticity, based on the apparent change in clayiness when a standard field texture test is carried out, can be distinguished as follows:^[2]

1. Soil with normal plasticity only shows minimal apparent change in texture during the field texture test.
2. Subplastic soils are those where the texture appears to become finer or heavier as the soil bolus is worked for the field texture test.
3. Superplastic soils are those where clayiness diminishes as the soil is worked.

Super plasticity may be of little practical importance. Soils that could fall into this category are very sticky clays found in buried soils formed through weathering of glacial drift. Most soils have normal plasticity. Although subplastic clay soils are mainly reported in Australia, they probably occur worldwide. Soils with subplastic properties appear to be associated with very strongly bonded silt size aggregates consisting largely of fine clay.^[22] Subplasticity of Oxisols was attributed to cementation by iron oxides, while on Alfisols it was attributed to the presence of packets or domains of strongly oriented clay but not Ca carbonates.^[23] Removal of the cementation agents in subplastic clays revealed the presence of highly active (2:1) clay minerals.^[24] The degree of dispersion of subplastic clays has a strong impact not only on the outcome of texture analysis but also on the outcome of many soil chemical analyses.

REFERENCES

1. Baver, L.D. *Soil Physics*, 3rd Ed.; John Wiley & Sons, Inc.: New York; Chapman & Hall Limited: London, 1956.
2. Butler, B.E. A system for the description of soil structure and consistence in the field. *J. Aust. I. Agr. Sci.* **1955**, *21*, 239–249.
3. SSSA. *Internet Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 1997. Available at <https://www.soils.org/publications/soils-glossary> (accessed August 21, 2015).
4. Troeh, F.R.; Thompson, L.M. *Soils and Soil Fertility*, 5th Ed.; Oxford University Press: New York, 1993; 462 pp.
5. Atterberg, A. Die plastizität der tone. *Bodenkundliche Mitteilungen* **1911**, *1*, 10–43.
6. Casagrande, A. Classification and identification of soils. *Proc. Am. Soc. Civil Eng.* **1947**, *73*, 783–810.
7. Lambe, T.W.; Whitman, R.V. *Soil Mechanics, SI Version*; John Wiley & Sons: New York, Chichester, Brisbane, Toronto, Singapore, 1976.
8. Means, R.E.; Parcher, J.V. *Physical Properties of Soils*; Constable and Company Ltd.: London, 1964.
9. Hartge, K.H. *Einführung in die Bodenphysik*; Ferdinand Enke Verlag: Stuttgart, 1978.
10. Zhang, H.Q.; Hargte, K.H. Die wechselwirkung zwischen winkel der inneren reibung, organischer substanzen und wasserspannung. *Mitteilungen der Deutschen Bodenkundlichen Gesellschaft* **1989**, *59*, 279–282.
11. Mitchell, J.K. Fundamentals of Soil Behaviour. In *Series in Soil Engineering*; Lambe, T.W., Whitman, R.V., Eds.; John Wiley & Sons Inc.: New York, London, Sydney, Toronto, 1976; 422 pp.
12. Warkentin, B.P. Physical properties related to clay minerals in soils of the Caribbean. *Trop. Agr.* **1974**, *51*, 279–287.
13. Lal, R. Physical properties and moisture retention characteristics of some Nigerian soils. *Geoderma* **1979**, *21*, 209–223.
14. Rosa de la, D. Relation of seven pedological characteristics to engineering qualities of soil. *J. Soil Sci.* **1979**, *30*, 793–799.
15. Larney, F.J.; Fortune, R.A.; Collins, J.F. Intrinsic soil physical parameters influencing intensity of cultivation procedures for sugar beet seedbed preparation. *Soil Till. Res.* **1988**, *12*, 253–267.
16. Jong de, E.; Acton, D.F.; Stonehouse, H.B. Estimating the Atterberg limits of southern Saskatchewan soils from texture and carbon contents. *Can. J. Soil Sci.* **1990**, *70*, 543–554.
17. Maeda, T.; Takenaka, H.; Warkentin, B.P. Physical properties of allophane soils. *Adv. Agron.* **1977**, *29*, 229–264.
18. Ross, G.J. Mineralogical, physical and chemical characteristics of amorphous constituents in some pozzolanic soils from British Columbia. *Can. J. Soil Sci.* **1980**, *60*, 31–43.
19. Misopolinos, N.D.; Silleos, N.G.; Prodromou, K.P. The influence of exchangeable Mg on certain physical soil properties in a number of Mg-affected Soils. *Catena* **1988**, *15*, 127–136.
20. Baver, L.D. The Atterberg consistency constants: Factors affecting their values and a new concept of their significance. *J. Am. Soc. Agron.* **1930**, *22*, 935–948.
21. Leinweber, P.; Kahle, P.; Schulten, H.R. Einfluss der qualität der organischen substanz auf die konsistenz landwirtschaftlicher böden. *Zeitschrift für Pflanzenernährung und Bodenkunde* **1991**, *154*, 169–170.
22. Walker, P.H.; Hutka, J. Sub-plasticity in Australian soils. III. Particle-size properties of soil materials of varying plasticity. *Aust. J. Soil Res.* **1976**, *14*, 248–260.
23. Brewer, R.; Blackmore, A.V. Sub-plasticity in Australian soils. II Relationship between sub-plasticity rating, optically oriented clay, cementation and aggregate stability. *Aust. J. Soil Res.* **1976**, *14*, 237–248.
24. Blackmore, A.V. Sub-plasticity in Australian soils. IV Plasticity and structure related to cementation. *Aust. J. Soil Res.* **1976**, *14*, 261–272.

Plinthite and Petroplinthite

Eswaran Padmanabhan

Geoscience, University Teknologi Petronas, Bandar Seri Iskandar, Malaysia

Hari Eswaran

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS) (Retired), Washington, District of Columbia, U.S.A.

Abstract

Plinthite is a material that is made up of iron oxides and forms deep in soils under certain specific conditions. Other minerals such as quartz, kaolinite, gibbsite, and birnessite may also occur in this material. Petroplinthite forms from the hardening of plinthite. Recementation of plinthites or petroplinthites forms the petroferic contact. Such features can be found in several soil types. Plinthite is distributed over large areas in India, and petroplinthite gravel is present in the peneplains. West Africa has the largest contiguous extent of soils with plinthite. The largest extent of soils with petroferic contacts also occurs in West Africa (Ivory Coast, eastern Ghana, Burkina Faso, and Central African Republic) and sporadically in the eastern part of southern Africa. Irreversible hardening upon exposure marks the advent of major water percolation problems and root penetration difficulties in soils having these features. Such soils complicate site yield predictions in plantations and also cause a drop in overall average yield for the land. Soils with plinthites can be easily mismanaged, especially in zero to low input agricultural systems. There is a lot to discover regarding the resilience of such soils.

INTRODUCTION

Plinthite is a term used to describe a soil feature that culminated from a change of physical characteristics through exposure to the atmosphere. This feature commonly occurs as reddish redox concentrations in a horizon with various kinds of patterns. This entry describes the genesis and morphology of these features as well as impact of their presence on use and management of the affected soils. Geologic deposits of ironstones and bauxite, which have other modes of formation, are not considered in this entry.

HISTORICAL PERSPECTIVE

Laterite is a highly weathered material, rich in secondary oxides of iron, aluminum (Al), or both.^[1] The material has very low amounts of basic cations and primary silicates but, conversely, may be enriched in quartz and kaolinite. Laterites are either hard or capable of hardening on exposure to wetting and drying cycles. The term laterite originated in 1807,^[2] but the first mention of it in formal scientific literature was in 1821.^[3] Geologists, engineers, and pedologists consider laterite as some kind of sesquioxide accumulation that ranges from weathered rock to hard and cemented ironstone. The term Latosol^[4] was introduced to distinguish it from laterite but was later suggested to be abandoned.^[5,6]

These older terms were replaced with modern terms such as plinthite,^[7] petroplinthite,^[8] and petroferic contact.^[7]

DEFINITIONS

The term plinthite (Fig. 1) is derived from the Greek word “plinthos” meaning brick. Other definitions for this term have been introduced.^[6,7,9] Absolute amount of iron in plinthite is site dependent. Initially, diffuse mottles separated by bleached interconnected zones form in the soil. A continuous phase of plinthite is formed eventually, through the accretion of iron with time. Plinthite usually forms deep in the soil and, therefore, has low amounts of organic matter. Erosion and truncation of the soil could expose the plinthite. Subsequent to these processes, secondary enrichment with organic matter could occur.

Plinthite normally has a sharp upper boundary to surficial layers of the soil but a diffuse lower boundary to reduced subsurface layers. The basic form of plinthite consists of quartz (silt and sand size) in a matrix of kaolinitic clay (with possibly some mica). In extreme cases, gibbsite, goethite, and hematite can also occur in addition to quartz and kaolinitic clay.^[10] Manganese (Mn) minerals such as birnessite and lithiophorite can also be present and, consequently, color the system black.

Petroplinthite (formerly known as ironstone or lateritic gravel) forms either directly in a soil or from the hardening

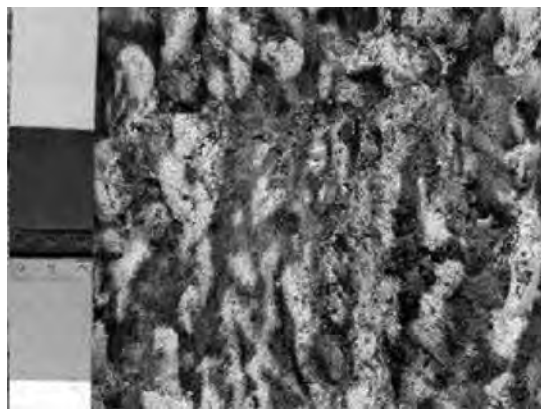


Fig. 1 Plinthudult from Peninsular Malaysia (plinthite is the combination of red and white mottles. The red blotches harden into hard petroplinthite when exposed and dried).

of plinthite. Petroplinthite usually occurs as loose or slightly cemented gravel.^[10,11] Recementation of plinthites or petroplinthites by rapid and cyclic influx of iron forms the petroferic contact. Alternatively, the contact can also develop on exposed ironstone gravel brought to the surface through uplift and erosion.

MODE OF FORMATION AND MORPHOLOGY

The pedological environment in the tropics is conducive for the formation of plinthite and related sesquioxide accumulations.^[6] The chemical processes resulting in the formation of these sesquioxide forms are quite complicated.^[6] A large supply of iron is needed to form plinthite. The origin of iron is normally difficult to ascertain.^[12] Downslope migration and subsequent accumulation of iron at lower levels in the landscape constitute one mode of formation.^[5] It has been suggested that iron can diffuse upwards (with fluctuating groundwater) toward oxidizing zones.^[13]

Cementation in soils may be due to two different chemical processes—1) ionic chemical bonding within the compounds (in petrogypsic and petrocalcic horizons); and 2) elements with intermediate electronegativity, such as silicon, Al, iron (Fe), and Mn,^[14] that form covalent bonds within the cementing agents and with other soil constituents.^[15] Even in crystalline forms, these compounds have outer surfaces composed of OH₂—OH groups that can be used to form covalent bonds with the same elements.^[6] This results in the high density of the cement.

Cementation and hardening of petroplinthite are attributed to the close packing of crystals.^[16] Submicroscopy studies indicate that the crust of Fe nodules is made up of densely packed Fe oxides.^[10,11,17] Iron accumulates on this rigid surface, subsequently increasing the thickness of the crust. Interior regions of the nodules usually have higher porosities as a consequence of the random orientation of minerals.^[10,11,17] It has been suggested that formation of

kaolinite is possible with Fe accretionary formations.^[18–20] Variations in ultramicroporosity and redox potentials appear to determine the type and geometry of minerals.^[17]

Alternating wet and dry conditions also favor hardening.^[6] Other factors that favor the hardening process include an ustic soil moisture regime^[7] or geomorphic changes in the landscape such as uplift or lowering of the groundwater table. Postuplift erosion eventually exposes the hardened zone (most evident at the edges of uplifted zones). The resistance to further erosion causes the formation of escarpments and inversion of the landscape.^[21] Consequently, the altered local hydrology usually favors the development of petroferic contacts in the soils.

OCCURRENCE AND DISTRIBUTION

The various forms of sesquioxide accumulations discussed earlier can be found in several soil types. The sharp upper and lower boundaries to the petroplinthite represent discontinuities in the soil. Active plinthite formation is usually found in relatively young landscapes that have fluctuating water tables in the soil. Plinthite is distributed over large areas in India, and petroplinthite gravel is present in the peneplains. West Africa has the largest contiguous extent of soils with plinthite. Such soils are present as a band from the West Coast of Gambia in the north to Cameroon in south and extend to southern Sudan and eastern Zaire.^[6] South of this band, soils with petroplinthite occur. The largest extent of soils with petroferic contacts also occurs in West Africa (Ivory Coast, eastern Ghana, Burkina Faso, and Central African Republic) and sporadically in the eastern part of southern Africa.^[22]

MANAGEMENT IMPLICATIONS

Pure Laterites and Latosols cover the greater part of the humid tropics and are either agriculturally poor or virtually useless.^[23] Essentially, the six major agro-management-related problems for such soils are irreversible hardening upon exposure, restriction to water flow, reduction in soil volume for root development, soil degradation, high remediation costs, and irreversible phosphorous fixation. The cause–effect relationships involving all these stresses, and the subsequent impact on land productivity remain unclear. Corrective measures to overcome these problems are usually not undertaken, because such projects are considered to be economically not viable. Economic and social factors have an impact on the use and management of such soils. However, these factors will not be discussed in this entry. Another effect of the presence of such soils would be the gradual reduction in carbon-sequestration capacity.

Irreversible hardening of plinthite occurs when the vegetative cover is removed.^[24] The presence of plinthite, petroplinthite, or petroferic contacts indicates that some restriction to water^[25,26] and root penetration as well as

natural soil degradation processes exists. Water flows faster above the plinthite but movement within this material is slow and restricted to the bleached zones.^[25,26] Perched water above the plinthite layer restricts the potential use of such soils to crops that require saturated conditions, such as rice. Landscape evolution and climatic change can modify the hydrology.^[27]

Quality of such soils declines (degradation) for agriculture and engineering purposes. Storage capacities for moisture and nutrients reduce with decrease in effective soil volume. Plinthite by itself poses few constraints to agricultural use of the soil.^[6] Global climatic changes (increased aridity) enhance the development of petroplinthite or petroferic contacts. The Fe nodules tend to have elevated phosphorus (P) contents and higher P sorption capacities compared to the soil fines.^[28]

In the humid tropics (e.g., Malaysia), such soils are being used for rubber and oil palm cultivation. Apart from lowering the average yield, the presence of such soils complicates site yield predictions, fertilizer recommendations, and also leaf and soil sampling for nutrient analyses. Increasing demands for food have forced these soils to be cultivated. Mismanagement of such soils has resulted in erosion rates of up to 100 ton/ha/yr. Eventual exposure of the petroferic material causes such soils to be abandoned. Soil remediation is very expensive. In addition to this, revegetation techniques are extremely time-consuming. Reforestation may be the only viable option for such soils.^[6] There have been studies that try to understand plinthite better in order to increase agricultural productivity.^[29–33]

Laterite has always been a cheap raw material for the construction of roads and walls in the tropics. However, the raw material is not considered to be very durable. Methods to enhance the performance of the material include treatment with lime, bitumen, or even cement.^[34] Stabilized soils have been used in road construction since the Roman times.^[35] Firing improves the durability of lateritic soil blocks.^[36] However, many of these methods are expensive and may not be viable options in developing nations.

CONCLUSION

The problem soils discussed in this entry are present in many developing countries in the tropics. Such soils can be easily mismanaged. Several agro-management and engineering-related problems can be identified. The average farmer cannot manage these soils at high input levels. Zero- and low-input agricultural systems on such soils enhance soil degradation. The exact areal extent of such soils in many tropical countries is unknown. There is also a lack of knowledge of the resilience of these soils to development. These issues merit rigorous and immediate attention.

REFERENCES

1. Alexander, L.T.; Cady, J.G. *Genesis and Hardening of Laterite in Soils*; Tech. Bull. 1282; Soil Conservation Service, U.S. Department of Agriculture: Washington, DC, 1962; 90 pp.
2. Buchanan, F. *A Journey from Madras Through the Countries of Mysore, Canara and Malabar*; East India Co.: London, 1807; Vol. 2, 440–441.
3. Babington, B. Remarks on the geology of the country between Tellicherry and Madras. Trans. Geol. Soc. London **1821**, 5, 328–329.
4. Kellogg, C.E. Preliminary suggestions for the classification and nomenclature of great soil groups in tropical and equatorial regions. Commonwealth Bur. Soil Sci. Tech. Commun. **1949**, 46, 76–85.
5. Maignien, R. *Review of Research on Laterites*. Natural Resources Research IV; UNESCO: Paris, 1966; 148 pp.
6. Eswaran, H.; De Coninck, F.; Varghese, T. Role of plinthite and related forms in soil degradation. In *Advances in Soil Science*; Lal, R., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1990; Vol. 11, 109–127.
7. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*. Hand book 436, 2nd Ed.; Natural Resources Conservation Service, U.S. Department of Agriculture, U.S. Govt. Print. Office: Washington, DC, 1999; 869 pp.
8. Sys, C. Suggestions for the classification of tropical soils with lateritic materials in the American classification. *Pedologie* **1968**, 18, 189–198.
9. Daniels, R.B.; Perkins, H.F.; Hajek, B.F.; Gamble, E.E. Morphology of discontinuous phase plinthite and criteria for its field identification in the southeastern United States. *Soil Sci. Soc. Am. J.* **1978**, 42, 944–949.
10. Eswaran, H.; Comerma, J.; Sooryanarayanan, V. Scanning electron microscopic observations on the nature and distribution of iron minerals in plinthite and petroplinthite, Lateritisation Processes. In *Proceedings of the International Seminar on Lateritisation Processes, Trivandrum, India*; Oxford and IBH Publishing Co.: New Delhi, 1980; 335–341.
11. Eswaran, H.; Raghumohan, N.G. The microfabric of petroplinthite. *Soil Sci. Soc. Am. Proc.* **1973**, 37, 79–81.
12. Van Wambeke, A.; Eswaran, H.; Herbillon, A.J.; Comerma, J. Oxisols. In *Pedogenesis and Soil Taxonomy: II. The Soil Orders. Developments in Soil Science 11B*; Wilding, L.P., Smeck, N.E., Hall, G.F., Eds.; Elsevier: Amsterdam, 1983; 325–354.
13. Lelong, F. Régime des nappes phréatiques continues dans les formations d'altération tropicale. Conséquence pour la pédogénèse. *Sci. Terre.* **1966**, 11, 203–244.
14. Sanderson, R.T. *Chemical Periodicity*; Reinhold: New York, 1964.
15. Chadwick, F.; Nettleton, D. Micromorphologic evidence of adhesive and cohesive forces in soil cementation. In *Soil Micromorphology: A Basic and Applied Science. Developments in Soil Science 19*; Douglas, L.A., Ed.; Elsevier: Amsterdam, 1990; 207–212.
16. Schwertmann, U. Some properties of soil and synthetic iron oxides. In *Iron in Soils and Clay Minerals*; Stucki,

- JW., Goodman, B.A., Schwertmann, U.D., Eds.; NATO ASI Series; Reidel Publishing Co.: Dordrecht, 1985; 203–204.
17. Padmanabhan, E.; Mermut, A.R. Submicroscopic structure of Fe-coatings on quartz grains in tropical environments. *Clays Clay Miner.* **1996**, *44* (6), 801–810.
 18. Tardy, Y.; Nahon, D. Geochemistry of laterites stability of Al-goethite, Al-hematite and Fe³⁺-kaolinite in bauxites and ferricretes: An approach to the mechanism of concretion formation. *Am. J. Sci.* **1985**, *285*, 865–903.
 19. Trolard, F.; Tardy, Y. A model of Fe³⁺-kaolinite, Al³⁺-goethite, Al³⁺-hematite equilibria in laterites. *Clay Miner.* **1989**, *24*, 1–21.
 20. Nahon, D.B. Self-organization in chemical lateritic weathering. *Geoderma* **1991**, *51*, 5–13.
 21. Sivarajasingham, S.; Alexander, L.T.; Cady, J.G.; Cline, M.G. Laterite. In *Advances in Agronomy*; Norman, A.G., Ed.; Academic Press: New York, 1962; Vol. 14, 1–60.
 22. Ghosh, S.; Guchhait, S.K. Lithofacies palaeogeography and sedimentology. Characterization and evolution of primary and secondary laterites in northwestern Bengal Basin, West Bengal, India. *J. Palaeogeogr.* **2015**, *4* (2), 96–123.
 23. Kamarck, A.M. *Climate and Economic Development*. EDI Seminar Paper No. 2; Economic Development Institute, World Bank: Washington, DC, 1972.
 24. Sombroek, W.G. Soils of the Amazon region. In *The Amazon Limnology and Landscape Ecology of a Mighty Tropical River and Its Basin*; Sioli, H., Ed.; Kluwer Press: Dordrecht, 1984; 521–535.
 25. Carlan, W.L.; Perkins, H.F.; Leonard, R.A. Movement of water in a plinthic paleudult using a bromide tracer. *Soil Sci.* **1985**, *139*, 62–66.
 26. Blume, L.J.; Perkins, H.F.; Hubbard, R.K. Subsurface water movement in an upland coastal plain soil as influenced by plinthite. *Soil Sci. Soc. Am. J.* **1987**, *51*, 774–779.
 27. Macedo, J.; Bryant, R.B. Morphology, mineralogy and genesis of a hydrosquence of oxisols in Brazil. *Soil Sci. Soc. Am. J.* **1987**, *51*, 690–698.
 28. Tiessen, H.; Frossard, E.; Mermut, A.R.; Nyamekye, A.L. Phosphorus sorption and properties of ferruginous nodules from semiarid soils from Ghana and Brazil. *Geoderma* **1991**, *48*, 373–389.
 29. Enya, O.O.; Omuetti, J.A.I.; Akinbola, G.E. Particle size and free iron oxides distribution along two toposequences in south western Nigeria. *Cont. J. Agron.* **2011**, *5* (2), 22–31.
 30. Costantini, E.A.C.; Priori, S. Pedogenesis of plinthite during early pliocene in the Mediterranean environment: Case study of a buried paleosol at Podere Renieri, central Italy. *Catena* **2007**, *71*, 425–443.
 31. Eghbal, M.K.; Givi, J.; Torabi, H.; Miransari, M. Formation of soils with fragipan and plinthite in old beach deposits in the south of the Caspian Sea, Gilan Province, Iran. *Appl. Clay Sci.* **2012**, *64*, 44–52.
 32. Lawal, B.A.; Adeboye, M.K.A.; Tsado, P.A.; Elebiyo, M.G.; Nwajoku, C.R. Properties, classification and agricultural potentials of lateritic soils of Minna in sub-humid agroecological zone, Nigeria. *Int. J. Dev. Sustain.* **2012**, *1* (3), 903–911.
 33. Eze, P.N.; Udeigwe, T.K.; Meadows, M.E. Plinthite and its associated evolutionary forms in soils and landscapes: A review. *Pedosphere* **2014**, *24* (2), 153–166.
 34. Okagbue, C.O.; Yakubu, J.A. Limestone ash waste as a substitute for lime in soil improvement for engineering construction. *B. Eng. Geol. Environ.* **2000**, *58*, 107–113.
 35. Krebs, R.D.; Walker, R.B. *Highway Materials*; McGraw Hills: New York, 1971; 428 pp.
 36. Anifowose, A.Y.B. Stabilization of lateritic soils as a raw material for building blocks. *B. Eng. Geol. Environ.* **2000**, *58*, 151–157.

Podzols

Peter Buurman

Wageningen University, Wageningen, the Netherlands

Abstract

Podzols are formed in boreal climates, on poor parent materials. Iron and aluminum are transported from upper horizons to the subsoil, predominantly as organic matter complexes. Organic matter accumulation takes place because of the root input and dissolved carbon and is regulated by its mean residence time. Agricultural use of Podzols is limited by climate, fertility, water availability, and rootability.

BACKGROUND

Podzols are mineral soils with transport and accumulation of organic matter, extractable iron (Fe) and aluminum (Al) oxyhydrates and, sometimes, amorphous aluminum silicates. The soils are common in boreal climates on all kinds of parent materials and in other climates on siliceous parent materials. Podzolization in siliceous parent materials is enhanced by hydromorphic circumstances and lateral water flow.

In the following, the typical morphology and mineralogy of Podzols are described and explained by complexation and humus dynamics.

MORPHOLOGY

Macromorphology

Podzols have a characteristic horizon sequence, consisting of O–Ah–E–Bh–Bs horizons.^[1] The O-horizon is well developed in boreal Podzols while the Ah horizon (requiring biological mixing) is more pronounced in Podzols on siliceous materials (henceforth, non-boreal Podzols). The B-horizon of boreal Podzols is generally colored more by iron compounds than by organic matter (Fig. 1). In non-boreal Podzols, organic matter accumulation in the B-horizon is more pronounced, resulting in dark brown and black colors (Fig. 2). With time, the B-horizon moves toward greater depth and the E-horizon becomes thicker. Non-boreal Podzols may show clearly distinguishable Bh1–Bh2–Bs horizons, in which case the Bh1-horizon is depleted of iron compounds and shows the maximum concentration of extractable aluminum.

Poorly drained sandy Podzols with lateral water flow frequently have pronounced parallel humus bands in the B-horizon (hardpans, Fig. 3), while the E–B transitions may fade as a result of groundwater movements. Poorly drained Podzols may be completely devoid of extractable

iron compounds. Fig. 4 depicts morphological transitions in relation to the groundwater level.

Micromorphology

Podzol profiles show various humus forms, the relative abundance of which may change with depth.^[2] In the litter layer (O-horizon), relatively undecomposed plant remains dominate and humified material appears mainly as coprolithic pellets (Fig. 5). In the E-horizon, organic matter contents are very low and remnants generally occur as very thin coatings, which are remains of former B-horizon accumulations. In well-drained B-horizons, pellet-like organic matter originating from decay of roots by mesofauna dominates. Such pellets eventually merge into aggregates and even into coating-like structures on peds or sand grains. In poorly drained Podzols, the non-rooted parts are dominated by organic matter coatings (Fig. 6). Such coatings are also found in the deeper parts of well-drained Podzols, frequently in horizontal, irregular humus bands.

MINERALOGY

Through low pH and complexation of metals by organic matter, the podzolization process causes severe weathering of minerals in Ah and E horizons. Feldspars, muscovites, biotites, apatites, pyroxenes, and amphiboles disappear eventually, while their weathering products are either transported downward (Al and Fe compounds) or degraded further (clay minerals). Muscovites and chlorites lose their interlayer material and open up, eventually exhibiting properties of vermiculites or even (Al)-smectites. The accumulation of aluminum in the B-horizon may cause local chloritization of vermiculite and smectite (soil chlorite). B-horizons may further have an accumulation of amorphous aluminum silicates (proto-imogolite) or gibbsite. The iron minerals accumulated in the B-horizon are usually goethite and ferrihydrite.



Fig. 1 Boreal Podzol on schist, Swiss Alps.

FORMATION OF PODZOLS

The major processes in Podzols are removal of Al and Fe from Ah- and E-horizons and their accumulation in the B-horizon, together with varying amounts of organic matter.

The removal of Fe and Al compounds from the upper horizons and their accumulation together with organic matter in the B-horizon have been explained in various ways. It is generally accepted that Fe and Al can be complexed by water-soluble organic matter and be transported away from Ah- and E-horizons by percolating



Fig. 2 Non-boreal Podzol on quartz sand, Belgium.

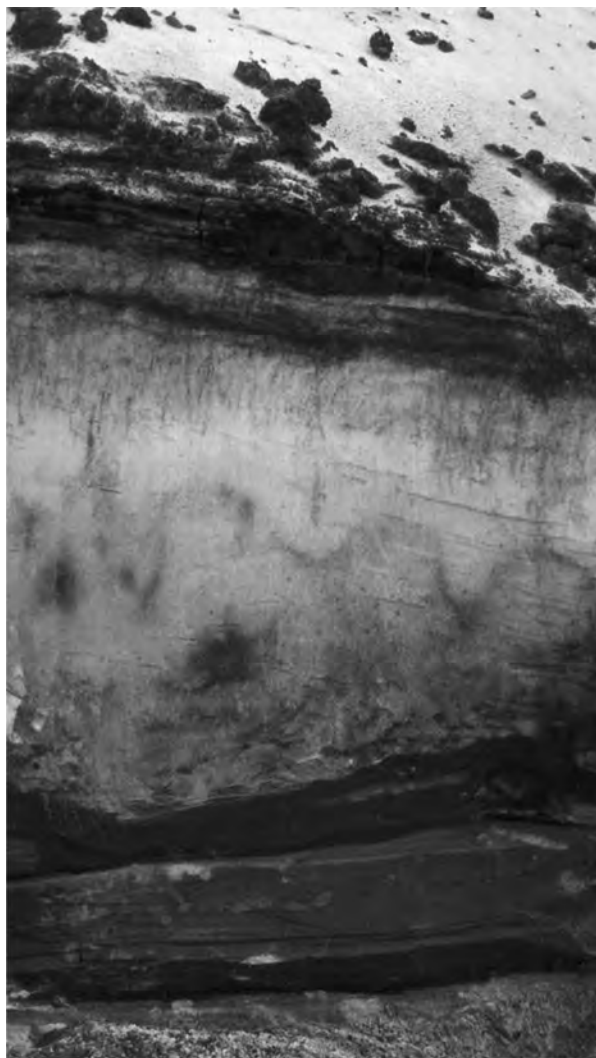


Fig. 3 Hydromorphic Podzol on quartz sands with thick Ortstein bands in the subsoil, the Netherlands.

water. There is more discordance about the process of accumulation. The classical “fulvate” theory^[3] postulates that organic molecules precipitate when all complexing groups are bound to metal ions. The fact that the carbon to sesquioxides molecular ratio of extracted organic matter and Al + Fe in the top of the B-horizon has similar values in many profiles and that this value is dependent on pH seems to point in this direction. This process, however, would mainly produce organic matter coatings, which is at variance with the fact that root-derived, pellet-like organic matter dominates in most well-drained Podzol-B-horizons. Because of their relative abundance of nutrients and better water-holding capacity, well-drained Podzol-B-horizons usually show abundant roots, and the fact that such horizons are dominated by root-derived material is not surprising. The relative abundance of coatings and pellets is related to the relative contribution of dissolved organic carbon

(DOC) and root organic matter and to organic matter dynamics.^[4]

Also, the relative dominance of sesquioxides in B-horizons of boreal Podzols and that of organic matter in B-horizons of Podzols on poor parent materials can be explained by soil organic matter dynamics.^[4] At a relatively high soil fertility, organic matter is rapidly broken down, while under nutrient-poor circumstances, breakdown is much slower. This is illustrated by high mean residence times (MRTs) of organic matter on poor parent materials, independent of climate (MRTs of 1000–4000 years) and much lower MRTs (350–750 years) in the much nutrient-richer boreal Podzols. The higher the MRT, the stronger is the accumulation of organic matter in the B-horizon. In boreal Podzols, accumulated DOC is much less than root-derived organic matter and therefore organic coatings are scarce. In non-boreal Podzols, DOC is better preserved and it dominates especially under hydromorphic circumstances when root contribution is low.

Formation and deepening of the E-horizon require removal of organic matter from the top of the B-horizon. There is no evidence of remobilization, and metal-complexed organic matter is relatively stable against microbial attack. The removal of metal ions from the top of the B-horizon by percolating unsaturated organic matter allows microbial decay of the metal-depleted humus.^[5] This results in breakdown of the upper fringe of the B-horizon and a gradual deepening of the E-horizon. This is in agreement with the predominance of relatively recalcitrant organic fractions in the E-horizon. The preservation of root-derived organic matter is similar to that of dissolved organic matter bound Al and Fe ions protect against microbial decay, and decay sets in when such metal ions are removed by percolating organic molecules.

Alternative Theories

There are some alternative theories about Podzol formation. Because Podzol-B-horizons on parent materials rich in weatherable materials may have proto-imogolite in the B-horizon, some authors postulate that Fe and Al are transported as positively charged silicate sols that, upon increase in pH, precipitate in the B-horizon as amorphous silicates.^[6] Percolating organic matter would then be adsorbed on these sols. Because organic molecules would strongly compete for metal ions with the silicate moiety, such an explanation would require a separation in time of Al/Si and organic matter accumulation. It seems more likely that percolating Si (as silicic acid) would recombine with Al to form proto-imogolite upon breakdown of the organic carrier of the Al ion. This is compatible with rapid humus dynamics (short MRT). Podzols with proto-imogolite are typically boreal Podzols with little organic matter accumulation.

Some authors emphasize the importance of low-molecular weight organic acids as carriers of complexed

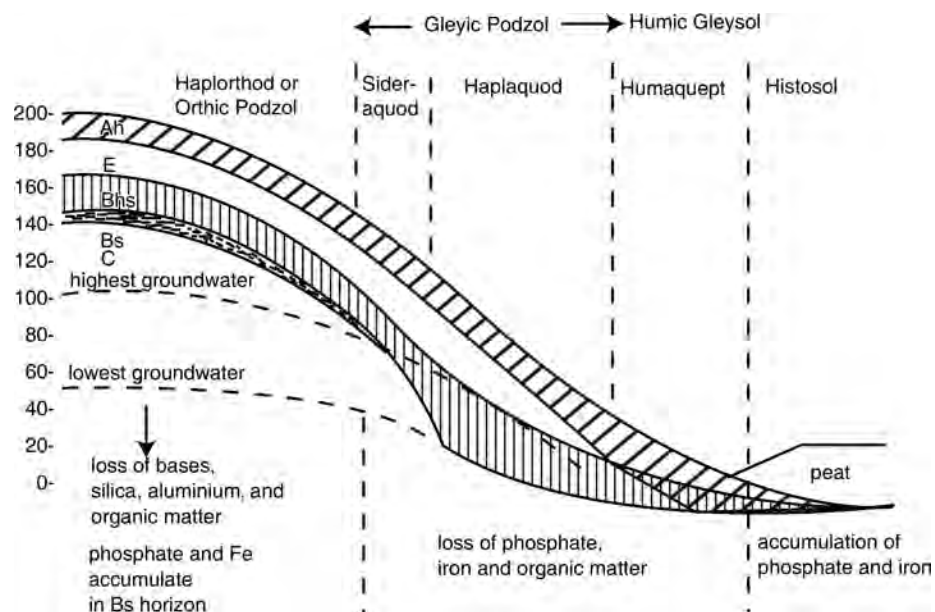


Fig. 4 Podzol hydrosequence on quartz sands.

Source: From Buurman.^[1]

metals. The metals would precipitate upon microbial breakdown of the carrier. This is probably a major mechanism in boreal Podzols and an important one in others. In non-boreal Podzols, it seems that upon breakdown of the carrier, complexed metals are transferred to organic moieties of larger molecular weight and higher recalcitrance.^[7]

ORGANIC MATTER CHEMISTRY

Pyrolysis-gas chromatography-mass spectrometry indicates that Podzol A horizons closely reflect the litter input and have abundant lignin-derived compounds. E-horizons are dominated by recalcitrant aliphatic moieties, and the composition of B-horizons depends on

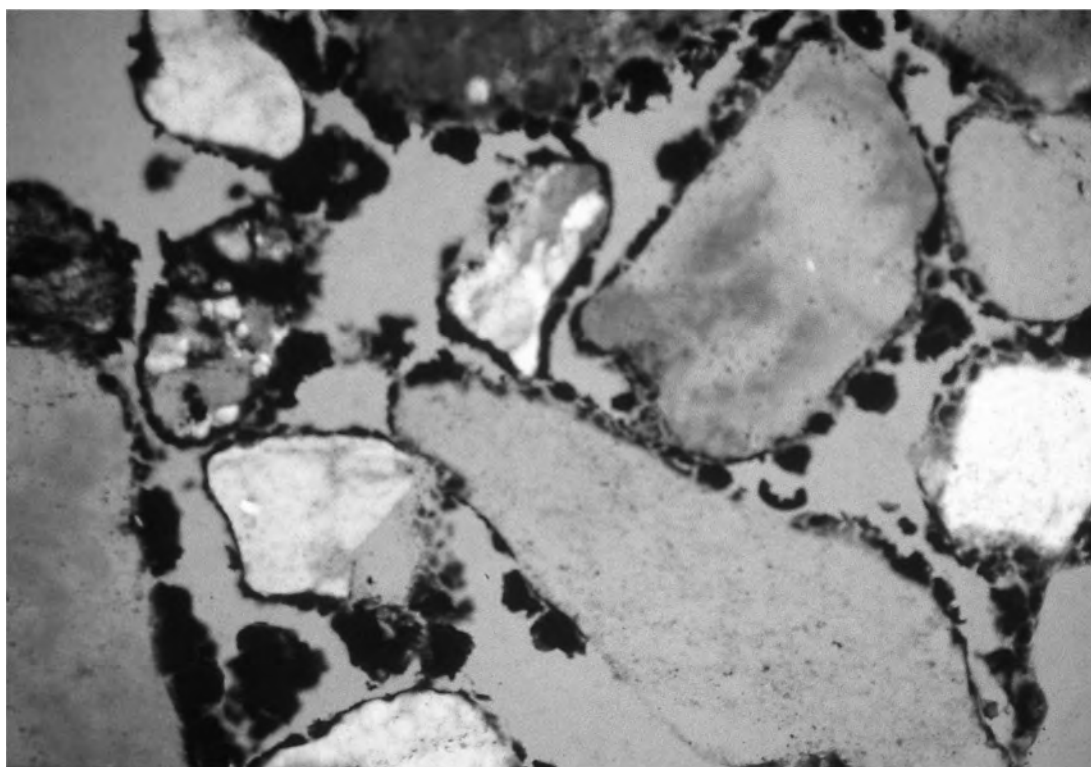


Fig. 5 Pellet-like organic matter in a well-drained Podzol-B-horizon, the Netherlands.

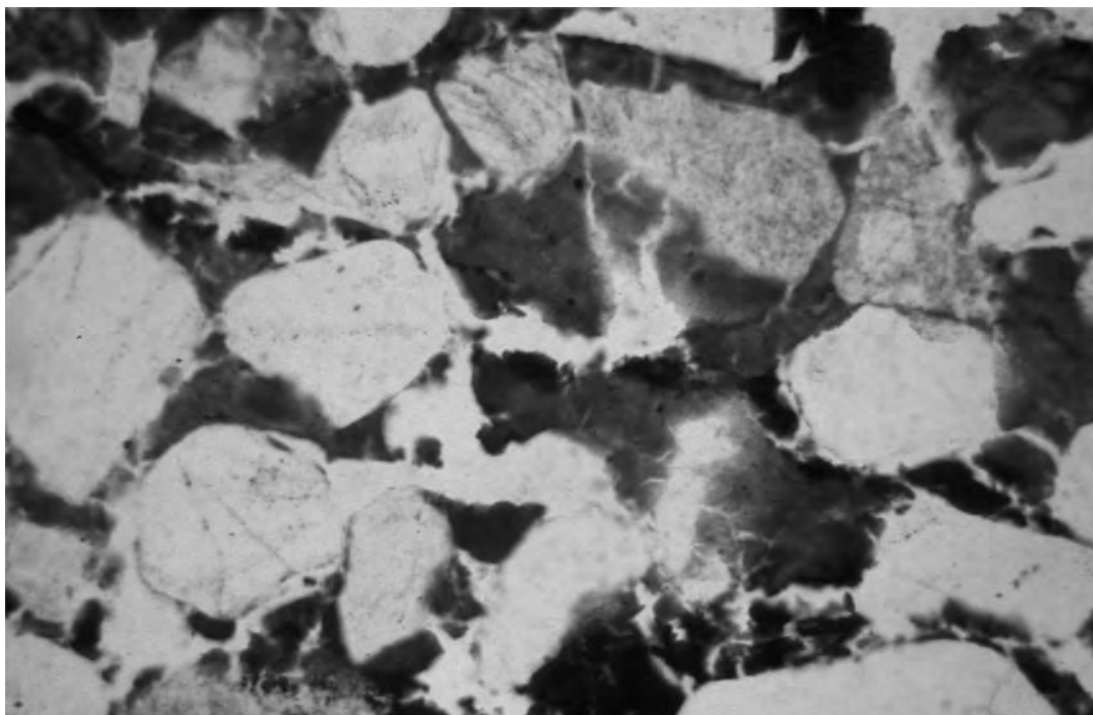


Fig. 6 Thick, fragmented organic coatings in a Podzol on quartz sands from New Zealand.

the relative abundance of root-derived and DOC-derived organics. Root-dominated B-horizons have high polysaccharide contents, while phenols and other small aromatic molecules dominate DOC-derived organic matter.^[5]

OCCURRENCE OF PODZOLS

[Fig. 7^{\[8\]}](#) depicts the worldwide distribution of Podzols. Podzols are the predominant soil type in boreal climates

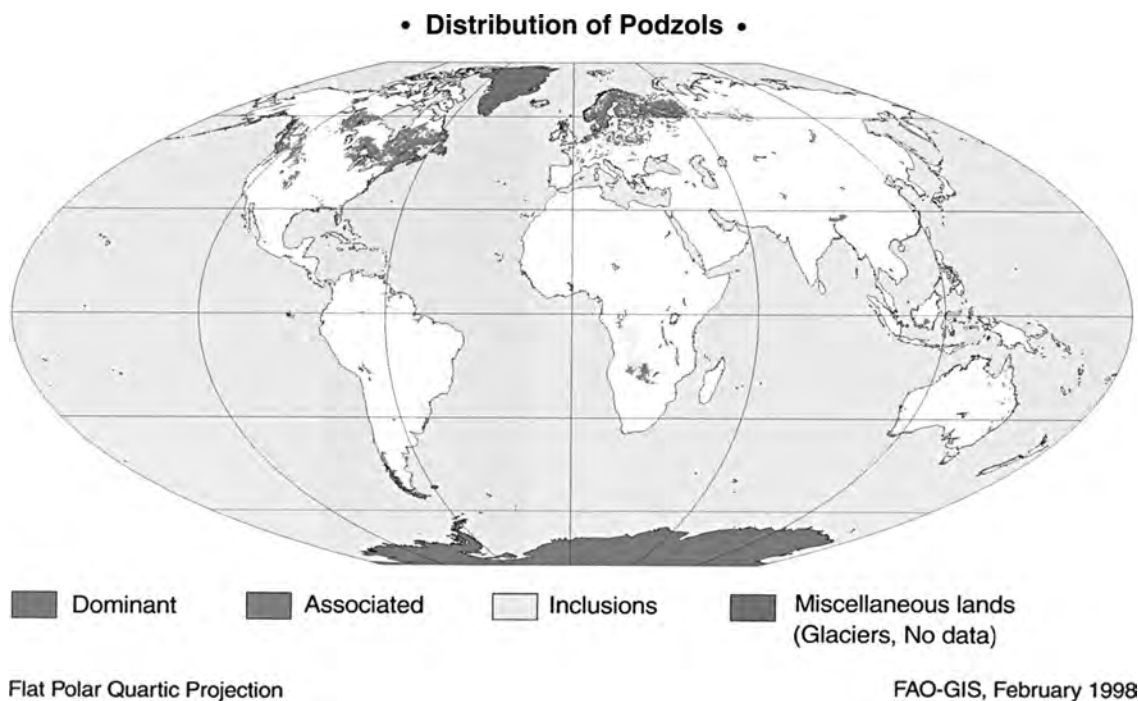


Fig. 7 Worldwide distribution of Podzols.
Source: From Driessen, Deckers, et al.^[9]

Table 1 Extent of Podzols by continent (1000 km²).

Europe	2136
North and Central Asia	218
South and South East Asia	60
Australia	85
North America	2208
South and Central America	55
Africa	113
Total	4875

Source: From Driessen, Deckers, et al.^[9]

and in pine and birch forests. This is caused by the slow decay of litter and consequent continuous production of complexing and water-soluble organic acids that weather topsoil minerals and transport Al and Fe in a complexed form. Toward milder climates, Podzols give way to other soils because the breakdown of complexing organic matter is too rapid and because biological mixing is more intense (obliterating any differentiation that might be caused by eluviation). In drier cold climates, there is insufficient production of litter and organic acids, along with lower percolation. Boreal climates are found both at relatively high latitudes and at high altitudes, and mountain Podzols have essentially the same properties as the Podzols of high latitudes.

Podzolization on nutrient-poor parent materials such as quartz sands is found in virtually all climate zones with sufficient rainfall. They are well documented from places like Central Europe, the Northern United States (Lake Michigan dunes), the Pigmy forest of California, the Amazon basin, uplifted beaches in Malaysia and Borneo, dune sands of Central Africa, and sandstone debris in eastern Australia. Such Podzols frequently are formed under the influence of both a poor parent material and an impeded drainage combined with a lateral groundwater flow. Sometimes, Podzols develop on Oxisols as a result of impeded drainage and clay destruction by ferrollysis, or on siliceous volcanic ash (ignimbrites). The extent of Podzols by continent is given in Table 1.

AGRICULTURAL USE

Podzols impose various restrictions to agricultural use. In boreal Podzols, climate is the main restriction. In non-boreal Podzols, both the low fertility and the stratified structure impose restrictions on the plant growth. Well-drained Podzols can be deep plowed to erase the layered structure and increase the water availability of the plowed zone. Poorly drained hardpan Podzols need both improved drainage and deep plowing to increase rootability. All Podzols tend to be poor in nutrients, especially nitrogen and phosphate. Poorly drained Podzols frequently lack trace elements, as they are removed from the profile through lateral water flow. In general, Podzols will produce adequately only under good management.

REFERENCES

1. Buurman, P., Ed. *Van Nostrand Reinhold Soil Science Series*; Podzols Van Nostrand Reinhold Company: New York, 1984; 1–450.
2. DeConinck, F. Major mechanisms in formation of spodic horizons. *Geoderma* **1980**, *24*, 101–128.
3. Mckeague, J.A.; Ross, G.J.; Gamble, D.S. Properties, criteria of classification and genesis of podzolic soils in Canada. In *Quaternary Soils*; Mahaney, W.C., Ed.; Geo Abstracts: Norwich, 1978; 27–60.
4. Buurman, P.; Jongmans, A.G. Podzolisation and soil organic matter dynamics. *Geoderma* **2005**, *125*, 71–83.
5. Buurman, P.; van Bergen, P.F.; Jongmans, A.G.; Meijer, E.L.; Duran, B.; van Lagen, B. Spatial and temporal variation in podzol organic matter studied by pyrolysis-GC/MS and micromorphology. *Eur. J. Soil Sci.* **2004**, *56*, 253–270.
6. Anderson, H.A.; Berrow, M.L.; Farmer, V.C.; Hepburn, A.; Russell, J.D.; Walker, A.D. A reassessment of podzol formation processes. *J. Soil Sci.* **1982**, *33*, 125–136.
7. Van Breemen, N.; Buurman, P. *Soil Formation*; Kluwer Academic Publishers: Dordrecht, 2002; 1–404.
8. FAO. *World Soil Resources*. World Soil Resources Report 66; FAO: Rome, 1991.
9. Driessen, P.; Deckers, J.; Spaargaren, O.; Nachtergaele, F. *Lecture Notes on the Major Soils of the World*. World Soil Resources Reports 91; FAO: Rome, 2001.

Polar Regions, The

Charles Tarnocai

Agriculture and Agri-Food Canada, Ottawa, Ontario, Canada

Iain Campbell

Land and Soil Consultancy Services, Nelson, New Zealand

Abstract

Soils of the polar regions cover approximately $20 \times 10^6 \text{ km}^2$. They occur in the coldest parts of the globe; therefore, their development is dominated by cryogenic processes, resulting in unique soil properties. These soils sequester organic carbon and store this carbon in the subsoil for thousands of years. Most of this carbon is in a frozen state and could be released into the atmosphere as greenhouse gases due to climate warming. These released greenhouse gases could then further accelerate the global warming.

INTRODUCTION

The polar regions encompass all land and water areas surrounding the north and south poles and include all of the global permafrost regions. The northern treeless part of this region is commonly known as the Arctic in the northern hemisphere and as Antarctica in the southern hemisphere. Since these regions have the earth's coldest climates, most of their soil materials are perennially frozen, and their development is dependent on the amount of moisture in the soil and the stability of the landscape.

Knowledge of the polar soils has advanced greatly since the 2000s, especially for the Northern Circumpolar Soils (NCPS). The development of seamless soils and soil carbon databases are able to demonstrate the huge amounts of soil carbon stored in these soils. Using these databases, several types of soil maps were produced (Fig. 1). Because of the global significance of this soil carbon, accelerated research was carried out by various disciplines on the characteristics, decomposability, and fate of this soil carbon due to climate change. Most of these activities were associated with significant amounts of field work in spite of the remoteness and inaccessibility of this region. To make the polar soils more available for a wide range of people, the Soil Atlas of the northern circumpolar region (NCPR) was published and made available for the government departments, libraries, and the general public.

ENVIRONMENTAL SETTING

The NCPS covers approximately $19 \times 10^6 \text{ km}^2$, and about 54% of this soil area is associated with perennially frozen soils. Most of the perennially frozen soils in the northern hemisphere occur in Canada and Russia. Glaciers cover approximately $1.9 \times 10^6 \text{ km}^2$ of this area, with most glaciers (92% or $1.7 \times 10^6 \text{ km}^2$) occurring in Greenland.

Although Antarctica is large (approximately $14 \times 10^6 \text{ km}^2$), 99.7% of it is covered by ice caps. The ice-free portion ($0.05 \times 10^6 \text{ km}^2$) that includes soils is found in many widely scattered small areas, the largest being the McMurdo Dry Valley region.

Short, cold summers and long, extremely cold winters characterize the climates of both polar regions. Mean daily temperatures above 0°C occur only during the warmest part of the summer. Total annual precipitation is generally low and occurs mostly as snow. In Antarctica, the annual precipitation averages 50 mm, ranging from less than 10 mm in some parts of the Transantarctic Mountains to greater than 100 mm in northern coastal regions. Effective precipitation is low, however, since most snowfalls either blow away or sublime. Vapor exchange is important in soil moisture transfers. However, soil moistening from snow thaw persists for only a few days.

The common vegetation associated with the NCPS is a conifer forest in the southern part of this region, and north of the tree line is a shrub-tundra grading into a sparse vegetation cover moving northward. Most of the Antarctic soils are unvegetated. Permafrost is continuous in both the Arctic and Antarctic, and the terrain is associated with patterned ground. Based on the Northern Circumpolar Soil Carbon Database,^[1] Turbels^[2] are the dominant soils covering approximately $6.5 \times 10^6 \text{ km}^2$. The second most common soils are the Histels^[2] ($2.7 \times 10^6 \text{ km}^2$) and followed by the Orthels^[2] ($1.1 \times 10^6 \text{ km}^2$). Seasonally frozen soils occur commonly in the southern part of the NCPR, and these are the Inceptisols, Spodosols, Histosols, Entisols, and Molisols.^[2]

SOIL-FORMING PROCESSES

Soil formation in the NCPR is dominated by cryogenic processes, which are driven by the presence and mobility

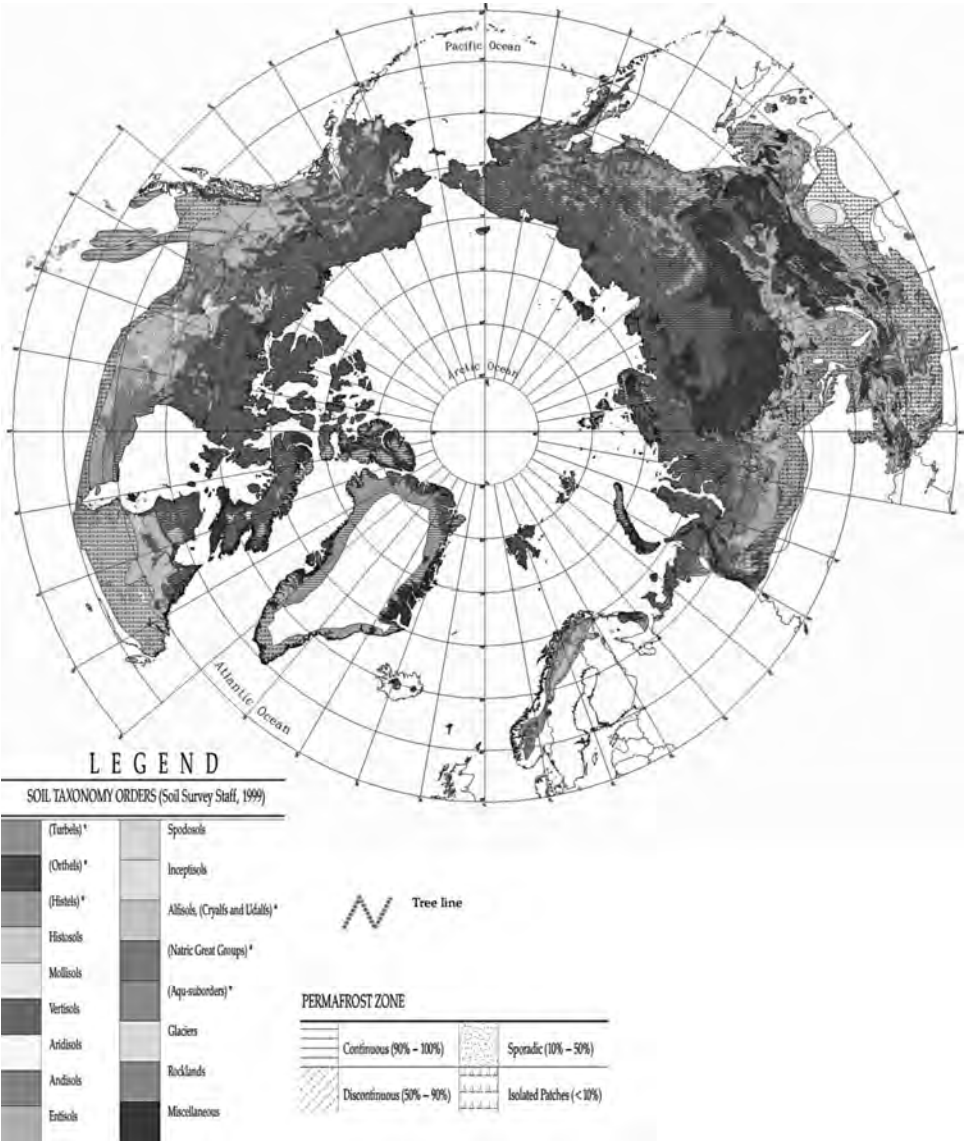


Fig. 1 Map showing the distribution of permafrost-affected soils (Turbels, Orthels, and Histels) and seasonally frozen soils in the northern circumpolar area. The various permafrost zones and the location of the arctic tree line are also shown in this map. **Source:** From Tarnocai, Canadell, et al.^[1]

of unfrozen soil water as it migrates toward the frozen front, along the thermal gradient in the frozen system. These processes, which produce a dynamic landscape, include freeze-thaw, cryoturbation or frost churning (Fig. 2), frost heave, cryogenic sorting, thermal cracking, patterned ground, and ice segregation. Other soil-forming processes that can leave an imprint on these soils include the gleying process, brunification, eluviation, and salinization.

In the Antarctic or southern circumpolar region, however, soil formation is dominated by weathering processes because of the extraordinary landscape stability resulting from the exceedingly cold and arid climate. Patterned ground develops widely in these dry Antarctic soils, especially on younger land surfaces, but the presence of volcanic ash in the cracks indicates that cryoturbic mixing is limited. Since almost all precipitation occurs as snow, which blows away or sublimates, little moisture is transferred

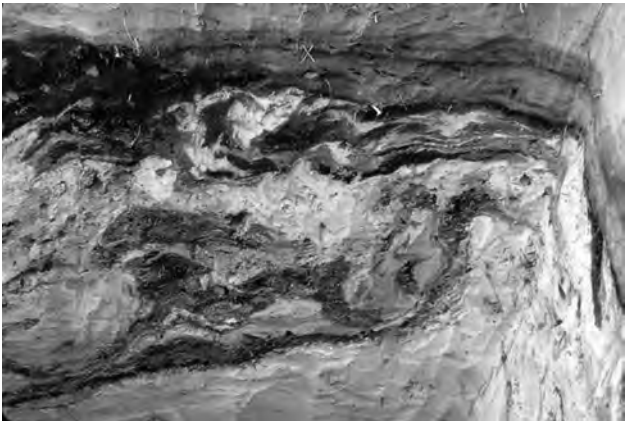


Fig. 2 Strongly cryoturbated permafrost-affected soil from Arctic Canada. Cryoturbation is displayed by the strongly contorted mineral and organic horizons.



Fig. 3 A Miocene-age soil weathering profile on till from near the head of Hatherton Glacier, Antarctica. There is a well-developed surface pavement and a salt horizon at 15 cm. The soil is dry-frozen with the active layer depth at approximately 10 cm. Soil colors are reddish brown (5YR 4/4) at the surface passing to light yellowish brown (10YR 6/4) at depth.

to the soil; therefore, cryoturbation and ice buildup are minimal or non-existent. As a result, cryogenic processes have little or no effect, and weathering processes, which operate over long periods on this stable landscape, dominate, even though they act extremely slowly. The soils thus produced are brunified (Fig. 3) and often contain salt accumulations.

Weathering processes operate extremely slowly in the Arctic as well as in the Antarctic. Physical weathering takes place mostly at the surface. In the Antarctic, clast reduction occurs through abrasion, exfoliation, granular disintegration (salt weathering), and thermal fracture. Frost weathering (frost shattering) is minimal in the Antarctic because of the absence of water, but it is more active in the Arctic, where the soil moisture content is higher. The dominant chemical weathering processes in Antarctic soils are oxidation and salinization.^[3,4] Because evaporation exceeds precipitation, salts accumulate in the Antarctic soils in a variety of forms, commonly in distinct horizons. Although salt composition of the soil partially reflects that of the soil parent material, it typically mirrors atmospheric and climatic gradients, indicating that aerosolic transport is the main source. Clay mineral formation is limited in both Arctic and Antarctic soils.

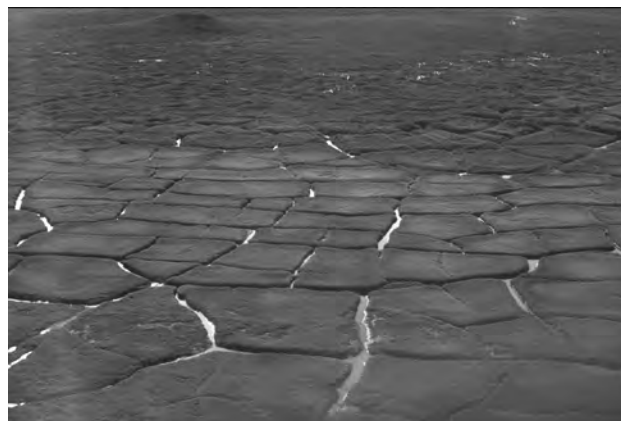


Fig. 4 Lowland ice-wedge polygon landscape, Mackenzie River Delta area, Canada. The ice wedges are located in the polygonal trenches.

SOIL CHARACTERISTICS

NCPS

Cryoturbation, which is very common in this region, results in irregular or broken soil horizons, involutions, organic intrusions (Fig. 1), organic matter buildup in the subsoil (often with concentration of this material along the top of the permafrost table), oriented rock fragments, silt-enriched layers, and silt caps.^[5] Permafrost-affected soils contain various amounts of ice in the form of ice crystals, vein ice, ice wedges, and massive ground ice several meters thick (Figs. 4 and 5).

The NCPS generally has high moisture content, especially near the permafrost table, so gleying associated grayish colors and redoximorphic features develops on most landforms, including those on better drained positions. These soils are very often saturated because the active layer (the upper layer of the soil that freezes and thaws



Fig. 5 Cross section of an ice-wedge polygon area associated with Cryosols. The ice wedge is approximately 3 m wide; Mackenzie River Delta area, Canada.

annually) is shallow. Thin, eluvial, or leached horizons are common in soils in the southern part of the discontinuous permafrost zone but very seldom occur in the northern regions. The brunification associated with coarse-textured soils is usually associated with this eluvial horizon.

The active layer not only supports biological activity but also protects the underlying permafrost. The thickness of this layer is controlled by soil texture and moisture, thickness of the surface organic layer, vegetation cover, aspect, and latitude. In perennially frozen soils, the active layer normally extends from 25 to 120 cm below the soil surface.^[6,7]

Particle size distribution in perennially frozen soil, ranges from silty clay to coarse, gravelly sand. The composition of the fine earth fraction is commonly dependent on the mode of deposition of the parent materials. The pH in these soils varies greatly and also depends on the chemistry of the parent materials. The similarity of pH to that of the parent material is caused, in part, by cryoturbation, which not only mixes parent material with soil materials but also mixes soil material among the horizons.

The nitrogen, potassium, and phosphorus contents of perennially frozen soils are generally low since most of these nutrients are locked into compounds contained in the surface organic layer.^[8] Salt crusts, which form by translocation of salts from the subsurface horizons, develop during dry periods in the summer because of increased evaporation from the soil surface. Such salt crusts are common on the surfaces of high Arctic soils. Calcareous crusts on the underside of rocks are also a common phenomenon, especially in coarse-textured soils. Soils developed on fine-textured marine sediments usually have both high conductivity and high salt content.

Soils in the NCPR contain the single largest component of the terrestrial carbon pool. An estimate made by Tarnocai et al.^[1] shows that organic soils (Histosols and Histels) and perennially frozen cryoturbated mineral soils (Turbels) have the highest organic carbon contents (32.2–69.6 kg m⁻³). Organic carbon pools estimated by the same authors also show that the perennially frozen soils in the NCPR contain approximately 191.3 Pg for 0–30 cm depth, 495.80 Pg for 0–100 cm depth, and 1024.00 Pg for 0–300 cm depth. Carbon pools in depths greater than 300 cm were estimated to be 407 Pg for yedoma deposits and 241 Pg for deltaic deposits. Therefore, based on these estimates, the soils in the NCPR contain 1672 Pg of organic carbon with approximately 88% occurring in permafrost-affected soils and deposits. This 1672 Pg of NCPR carbon accounts for approximately 50% of the total belowground global organic soil carbon. This total is greater than the amount of carbon stored in the atmosphere, forests, or biosphere. Only the oceans store a greater amount of carbon than do these soils, and the oceanic carbon is much more difficult to release into the atmosphere.

The diversity of microorganisms in NCPS is much greater than that in the SCPS soils because of the higher concentrations of available water and organic matter in NCPS. These microorganisms play an important role in the

decomposition of the organic matter and in nutrient recycling.^[9]

Southern Circumpolar Soils

Although conditions for soil formation in the Antarctic are minimal, considerable spatial soil variation results from differences in soil-forming factors. The main physical features of Antarctic soils are the following:

1. A distinctive surface pebble or boulder pavement;
2. Differing characteristics associated with differing rock types;
3. A soil form dominated by coarse, but highly variable, sandy to bouldery textures;
4. Only a small increase in fine particle size, associated with increasing age;
5. Almost a complete lack of cohesion or structural development;
6. Increased reddening and increased oxidation depth with increasing soil age;
7. Increased accumulation of soluble salts with increasing soil age;
8. An active layer from < 5 cm to > 70 cm, depending on climate; and
9. Permafrost that is commonly hard and ice-cemented in coastal situations and on younger land surfaces, but dry-frozen in areas of high aridity.

In the Transantarctic Mountains, the soils may thaw for only 2 months, while moisture contents of < 1–5% are common.^[10] Distinctions based on soil temperature and moisture differences are evident. In the Antarctic Peninsula and some northern coastal regions, the soils are moister and may have organic accumulations associated with sporadic moss or grass occurrences in damp sites.

Biologically, Antarctic soils typically lack organic accumulations. Life forms are restricted to more favorable moist environments where small patches of mosses, lichens, and algae may be present. Bacteria and algae occur within the soils, supporting grazers such as nematodes, tardigrades, and rotifers.^[11]

CONCLUSION

Soils of the circumpolar regions are the coldest soils on earth. These soils have been studied since the mid-1900s, but these studies greatly accelerated since the 2000s. There is now considerable knowledge of the processes that led to the formation of these soils, and of their role in relation to the unique ecosystems of these polar regions. However, their global significance was recognized by the role they play in global climate and due to the huge amounts of organic carbon they sequester annually, and they store this carbon for thousands of years due to their unique processes and properties.

REFERENCES

1. Tarnocai, C.; Canadell, J.G.; Schuur, E.A.G.; Kuhry, P.; Mazhitova, G.; Zimov, S. Soil organic carbon pools in the northern circumpolar permafrost Region. *Global Biogeochem. Cy.* **2009**, *23*, GB2023 DOI:10.1029/2008 GB03327.
2. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; Agriculture Handbook Number 436; United States Department of Agriculture; Natural Resources Conservation Service: Washington, DC, 1999; 869 pp.
3. Campbell, I.B.; Claridge, G.G.C. *Antarctica: Soils, Weathering Processes and Environment*; Elsevier Science Publishers: Amsterdam, 1987; 368 pp.
4. Bockheim, J.G. Relative age and origin of soils in eastern wright valley, Antarctica. *Soil Sci.* **1979**, *128*, 142–152.
5. Bockheim, J.G.; Tarnocai, C. Recognition of cryoturbation for classifying permafrost-affected soils. *Geoderma* **1998**, *81*, 281–293.
6. Smith, C.A.S.; Swanson, D.K.; Moore, J.P.; Ahrens, R.J.; Bockheim, J.G.; Kimble, J.M.; Mazhitova, G.G.; Ping, C.L.; Tarnocai, C. A Description and classification of soils and landscapes of the lower Kolyma river, northeastern Russia. *Polar Geogr. Geol.* **1996**, *19* (2), 107–126.
7. Tarnocai, C. Arctic permafrost soils. In *Permafrost Soils, Soil Biology Series*; Margesin, R., Ed.; Springer-Verlag: Heidelberg, 2008; Vol. 16, 3–16.
8. Broll, G.; Tarnocai, C.; Müller, G. Interactions between vegetation, nutrients and moisture in soils in the Pangnirtung Pass Area, Baffin Island, Canada. *Cryosols and Cryogenic Environments, Special Issue of Permafrost and Periglacial Processes*; Tarnocai, C., King, R., Smith, S., Eds.; **1999**, *10*, 265–277.
9. Robinson, C.H.; Wookey, P.A. Microbial ecology, decomposition and nutrient cycling. In *Ecology and Arctic Environment*; Woodin, S.J., Marquiss, M., Eds.; Special Publication 13, British Ecological Society; Blackwell Science: Oxford, 1997; 41–68.
10. Campbell, I.B.; Claridge, G.G.C.; Campbell, D.I.; Balks, M.R. The soil environment. In *Ecosystem Dynamics in a Polar Desert: The McMurdo Dry Valleys, Antarctica*; Prisco, J.C., Ed.; Antarctic Research Series; American Geophysical Union: Washington, DC, 1998; Vol. 72, 297–322.
11. Freckmann, D.W.; Virginia, R.A. Soil biodiversity and community structure in the Mcmurdo Dry Valleys, Antarctica. In *Ecosystem Dynamics in a Polar Desert: the McMurdo Dry Valleys, Antarctica*; Prisco, J.C., Ed.; Antarctic Research Series; American Geophysical Union: Washington, DC, 1998; Vol. 72, 323–335.

Pollution

Zulkuf Kaya

Department of Soil Science, University of Cukurova, Adana, Turkey

Abstract

Excess growth of population and urbanization on global scale along with industrialization induce pollution of soils which seriously threatens human health and ecosystem by causing irreversible hazardous effects. Thus, reaction of pollutants (decomposition, accumulation, and mobility properties) should be monitored for mitigation or if possible preventing perilous effects on living organisms. Since pollutants and heavy metals such as persistent chlorinated organic compounds (xenobiotica) and mercury could cause mutagenous and teratogenous side effects, great care and effort should be paid to prevent the transportation of the said substances to soil for sustaining biodiversity. Hence, via the sustainable ecosystem/agroecosystem concept, the status of the pollutants in soil should be monitored permanently.

INTRODUCTION

Soil—the medium of life and foundation of human existence, the contact of lithosphere, atmosphere, and biosphere—has been dominated by numerous physical (temperature and pressure changes, wind and water movements, etc.), chemical (oxidation, plant root, microorganism, excretions, solution, and neosynthesis), and biological (humus formation) processes in the past and the present, ranging from hundreds to millions of years. Ensurance of sustainable life, i.e., bound to the preservation of the natural courses of the processes as mentioned, is ultimately secured by the “quality of soil” in physical, chemical, and biological terms.

SOIL POLLUTION: THE HUMAN IMPACT

The impact of urbanization and industrialization has been a major factor against the need of preserving the quality of soil, by reducing it via chemical contaminants, use of polluted waters for irrigation, and deposition of harmful particulates to the atmosphere. Ecosystems are threatened by such contaminants and their interactions in the environment. Harmful chemicals can be studied via concepts of ecosystem/agroecosystem planning developed, keeping in mind the sources of chemical contamination, which are responsible for the degradation of environment. These sources are as follows: 1) the wastewaters; 2) the agricultural wastes; 3) the airborne pollutants; 4) the pesticides; 5) the urban wastes: a) sewage sludge, b) composts, and c) fly ash; and 6) the industrial wastes: a) pesticides and fungicides, b) fertilizers, c) detergents, and d) chlorinated solvents.^[1]

These sources may cause fatal effects and/or irreversible destructions for human health and the environment. The levels of contamination together with their ability and

mobility in decomposition and accumulation of the chemicals in the soil have been scientifically proven to be harmful to animals, plants, and microorganisms via destroying the natural structure of water, soil, and air, which is balanced by nature.^[2]

Inactivated enzymes are the measure of heavy metal (divalent) toxicity readily reacting with proteins, amines, and sulfohydryl (–SH) groups as well as mercury (Hg) and cadmium (Cd) replacing zinc (Zn) in metallic enzymes. Metal toxicity is known to decrease cell membrane permeabilities and change genetic characteristics of cells increasing risks of cancer. Hg accumulates in fatty tissues as methyl-Hg, whereas Cd replaces calcium in bones and kidneys destroying their excretory functions.

According to Förstner,^[2] the behavior of synthetic organic compounds in soils are not “sorptive” processes but are related to dissolution mechanisms on lipid-like particle surfaces ousted from soil solutions. Functional groups varying in molecular size, form, and valency such as the cationic, basic, and acidic solutions together with non-polar compounds are also responsible for the “sorption” of organic chemicals by solids.

Ecological studies have been carried out on the decomposition or hydrolysis of chemical bonds from water-soluble pesticides such as methylcarbamates or organophosphoric insecticides revealing shorter duration periods in soils than organochlorine compounds. However, weak bonding compounds of the chlorophenol group are easily mobilized by pH changes and other unknown environmental impacts and expose higher risks for human and environmental health.

The highly persistent and biologically unsynthesizable “xenobiotica,” i.e., the organic compounds with the highest poisoning risk such as 2,3,7,8-tetrachlorodibenzodioxine with a lethal dose of 1 µg/kg body weight are 1000-fold more poisonous than sodium cyanide.^[2]

Table 1 Contents of source materials causing pollution in cultivated soils (mg/kg).

Elements	Fertilizers ^a	Sewage sludge	Domestic compost	Fly ash	Precipitation ^b
Cd	P(50)	12	10	10	0.25
Cr	P(200)	250	120	280	1.4
Hg	Trace	4.4	—	—	0.05
Ni	Trace	80	120	270	7.3
Pb	P(100)	700	1200	330	11.0
Zn	P(150)	3000	2000	360	29

Note: P, phosphate fertilizers, the figures in parentheses are normal levels. Critical levels are in italics.

^aFertilizer containing highest amounts of heavy metals.

^bThe estimated accumulation in surface soils (top 20 cm) in 100 years by atmospheric precipitation.

Source: Adapted from Berrow.^[3]

Metals such as lead, arsenic, Hg, chromium, Cd, and copper are found at acceptable limits in nature. However, these limits might increase with contamination from agricultural, industrial, and infrastructural wastes (Table 1). For instance, concentrations of lead might increase by the gaseous emissions of vehicles and use of garbage compost as a fertilizer together with pesticides. Arsenic is naturally found at a level of 10 ppm in soils and might also increase to 500 ppm with industrial wastes and agricultural applications. Lead arsenate applications in orchards are determined to increase lead to 121 ppm.^[1] Hg, which is determined to be between 90 and 250 g/ha in normal soils, might increase at a rate of 5 g/ha/yr with the use of chemicals for cereal seeds. Additionally, about the same amount of Hg is added to the soil by rain and irrigation waters as well as garbage compost (Table 1).^[1] The increase of Cd contents in the soils of Japan has been the main reason in establishing a law for the control of soil contaminants. The behavior as well as the qualitative and quantitative distributions and conversion mechanisms of

the foreign organic materials in soils have always been obscure (Fig. 1). Fig. 1 illustrates that the independent enzymes are responsible for the biotic conversions of organic chemicals occurring together with the non-biotic. Soil quality on the long run is extensively affected by such conversions, forming non-extractable-bonded residues,^[3] even by suitable agricultural practices.^[1]

A specific extraction method for each non-extractable residue should be determined for residues absorbed within clay mineral stacks and bonded by non-covalent bonding in macromolecular spaces of humic substances, hydrogen bridges, van der Waals forces, and attractions occurring via charge transfers as well as covalent bonding to monomeric humic substances and macromolecular structure of phenols and aromatic amines.

Chemical residues bonded by humic substances in soils should be carefully controlled owing to their limited availability to plants, and lack of mineralization preventing their inclusion to the C-cycle,^[1] thus algae, fungi and lichens, protozoa, and simple metazoa are used to isolate the

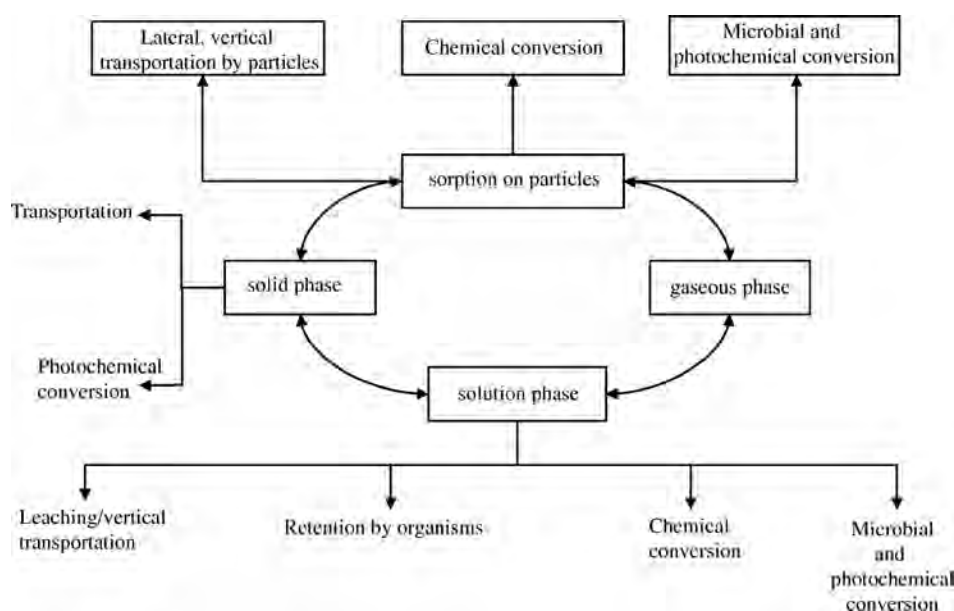


Fig. 1 Behavior of pesticides in soils.

Source: Adapted from Korte.^[1]

anthropogenic chemicals from soils and soil solutions. However, information obtained on the conversion of environmental chemicals in laboratory conditions is not usually valid for field conditions, which are highly complex with different densities of populations, nutrient statuses, and conversion activities.^[4]

Microorganisms, insects, and larvae have yielded estimable results^[5] at numerous experiments conducted in field or laboratory conditions on the conversion and transportation of chemicals in soils. In spite of being tedious to handle, lysimeters with soil columns have also been documented to yield realistic results.

Cd, Pb, AND Cr CONTAMINATION IN SOILS

The natural Cd content of soils at the upper 20 cm depth is <0.5 mg/kg. Cd contents of soils in the European Union have significantly increased to 0.16–0.32 mg/kg Cd in the last decades by excess input.^[1] The sources of increasing Cd are the polluted (~1.5–35 g/ha/yr) sewage sludges (~1 g/ha/yr), phosphate fertilizers (~5–6 g/ha/yr), and others (~1 g/ha/yr).^[1]

The major source of Cd contamination in agricultural soils is the excess use of fertilizers with varying amounts of Cd contents (0.1–90 mg/kg) depending on the phosphate rock materials utilized in production. Such contaminations in soils will cause a 0.1 mg/kg increase of Cd in 20–30 years. Other spots of high Cd contamination are adjacent to Zn mines. Another significant Cd contamination source is the sewage sludge used in cultivated soils. The Cd content in the dry matter of the sludge is 7–17 mg/kg.^[1] Cd adsorption by organic or inorganic soil compounds (minerals) depends on the soil texture, pH, and Ca together with other elements. The retention of low amounts of Cd in soil solutions in surface soils might take tens or hundreds of years depending on texture.^[6] The carryover of Cd to plants does not only depend on the type and growth period but also on the availability of Cd in soils. Exchange, adsorption, fixation, and diffusion are the dominant mechanisms in soils controlling Cd uptake/carryover by plants.

Lead is less toxic than Cd in soils with contents of about 15 mg/kg in the earth's crust. This amount increases to 700 mg/kg in soils close to highway traffic. Lead is very immobile in soils compared to Cd and Zn, being slightly soluble in pH <5, whereas it is more soluble and available to plants at pH = 4–4.5.^[7]

Soil Pb contents generally vary from 1 to 20 mg/kg,^[2,6] with Pb solubility decreasing at pH > 6 and increasing in pH < 6 with increasing availability to plants. The decrease of solubility/availability has been shown in many cases and especially in Pb-free grains of corn grown in soils with high Pb (320 kg/day).^[8] Tolerable limits of Pb in soils are given by Hock and Elstner^[8] as 100 mg/kg and 50–300 mg/kg for EU standards by Förstner.^[2]

Chromium, the essential element for human and animal diets, is not considered essential for plant nutrition with a tolerable level of 100 mg/kg in soils.^[1,8] Cr³⁺ is irrevocable in its role in the induction of the effect of insulin and proteins as well as stabilizing nucleic acid structures and activating some enzymes in glucose metabolism.^[9]

Varying parent materials seem to be responsible for the wide range of chromium contents of soils (5–3000 mg/kg). High pH conditions favor the formation of new insoluble compounds—Cr(OH)₃ and Cr₂O₃ · xH₂O—with iron oxides. In strong acidic conditions, Cr is readily soluble, with a low uptake (carryover) by plants, whereas it is readily available even at very low amounts if retained by roots.^[7]

NICKEL CONTAMINATIONS IN SOILS

Soils contain as low as 5–50 mg/kg Ni,^[7] which may increase to 5000 mg/kg in soils developed from olivine and serpentine rocks. Ni reaches the soil from air, via burning coal (10–50 mg/kg Ni) and derivatives of petroleum (49–145 mg/kg Ni) transformed to Nickel carbonyl, which is cancerogenous.^[9] Sewage sludge containing high amounts of Ni is also responsible for the pollution of the soil. Clay minerals, magnesium, iron, and aluminum oxides adsorb and/or bind immobilize soil Ni depending on pH.^[10] At pH < 5.5, the solubility and plant availability of Ni increases. In soils with pH > 6—the threshold reaction for Ni—the extractable Ni content by ethylenediaminetetraacetate is 25 mg/kg. At pH < 6, lower values of Ni concentrations are needed to avoid toxicity.

Hg CONTAMINATION

Hg compounds (especially CH₃–Hg⁺) are highly poisonous for humans and animals. Excess Hg affects the central nervous system, develops blindness, and causes mutagenous and teratogenous effects via CH₃ · HgCl. The Hg content of the earth's crust is approximately 0.02 mg/kg, whereas slightly contaminated soils and sediments together with soils developed from Hg-rich parent materials (located near volcanic areas) may contain 0.5 mg/kg Hg (dominantly <0.1 mg/kg) and 40 mg/kg Hg, respectively.^[8]

Soils of highly populated areas and ore production sites contain 0.1–0.4 and 1.8 mg/kg Hg, respectively. Similarly soils, irrigated by wastewaters, contain up to 1.6 mg/kg of Hg.^[7]

The following reaction, together with microorganisms taking part in the process, shows the conversion of Hg in soils as



The low solubility of Hg in soils and its immobilization caused by the effective groups (–S–H and –S–S) in soil organic matter, by inducing accumulation in the root zone, consequently reduce the carryover to the plants.^[11]

Vegetables like lettuce and beans also seem to be suitable for growing in Hg-contaminated soils (7 mg/kg Hg) owing to their low uptake rates (0.08–0.12 mg/kg Hg), in contrast to grass containing surprisingly high amounts of Hg at golf courses with 130 mg/kg Hg.

CONCLUSION

Excess growth of population and urbanization on global scale along with industrialization induce pollution of soils which seriously threatens human health and ecosystem by causing irreversible hazardous effects. Thus, reaction of pollutants (decomposition, accumulation, and mobility properties) should be monitored for mitigation or if possible preventing perilous effects on living organisms. Since pollutants and heavy metals such as persistent chlorinated organic compounds (xenobiotica) and Hg could cause mutagenous and teratogenous side effects, great care and effort should be paid to prevent the transportation of the said substances to soil for sustaining biodiversity. Hence, via the sustainable ecosystem/agroecosystem concept, the status of the pollutants in soil should be monitored permanently.

REFERENCES

1. Korte, F. *Lehrbuch der Ökologischen Chemie*. 3. Auflage; Georg Thieme: Stuttgart, New York, 1992; Vol. 3, 373 pp.
2. Förstner, U. *Umweltschutztechnik*. 2. Auflage; Georg Thieme: New York, 1992; 507 pp.
3. Berrow, M.L. An overview of soil contamination problems. In *Proceedings of International Conference on Chemicals in the Environment*, Lester, J.N., Perry, R., Sterritt, R.M., Eds.; Jelper Ltd: London, 1986; 543–552.
4. Klein, W.; Scheunert, I. Bound pesticide residues in soil, plants and food with particular emphasis on the application of nuclear techniques in agrochemicals. In *Fate in Food and the Environment*; International Atomic Energy Agency: Wien, 1982.
5. Führ, F.; Steffens, W. Transport und Bilanzierung von umweltchemikalien in agroökosystemausschnitten. In *AGE Bericht Wege und Wirkungen von Umweltchemikalien Bonn*; AGE: Bad Godesberg, 1985; 36–38.
6. Poelstra, P.; Friesel, M.J.; Bassam, N.E.L. Transport and accumulation of Cd ions in soils and plants. *Z. Pflanzenern Bodenk* **1979**, *142*, 848–864.
7. Schachtschabel, P.; Blume, P.; Brummer, G.; Hartge, K.H.; Schwertmann, U. *Lehrbuch der Bodenkunde* 14. Auflage; Ferdinand Enke: Mannheim, 1998; 494 pp.
8. Hock, B.; Elstner, E.F. *Schadwirkungen auf Pflanzen*. 2. Auflage; Wissenschafts: Mannheim, 1988; 348 pp.
9. Merian, E. *Metalle in der Umwelt*; Verlag Chemie: Weinheim, 1984; 722 pp.
10. Tiller, K.G.; Geith, J.; Brummner, G. The relative affinities of Cd, Ni, and Zn for different soil clay fractions and goethite. *Geoderma* **1984**, *34*, 17–35.
11. Fushitey, S.G.; Frank, R. Distribution of mercury residues from the use of mercurial fungicides on golf course greens. *Can. J. Soil Sci.* **1981**, *61*, 525–527.

Pollution: Industrial Waste

Winfried E.H. Blum

Walter W. Wenzel

Institute of Soil Research, University of Natural Resources and Applied Life Sciences, Vienna, Austria

Wolfgang Friesl-Hanl

Health and Environment Department, Environmental Resources and Technology, Austrian Institute of Technology GmbH (AIT), Seibersdorf, Austria

Martin H. Gerzabek

Institute of Soil Research, University of Natural Resources and Applied Life Sciences, Vienna, Austria

Abstract

Pollution by industrial waste is widespread throughout the world, adversely affecting soil and environment qualities. Management of these wastes is necessary to achieve a more sustainable system. After exhausting the two approaches of preventing waste generation and recycling suitable materials, the use of wastes, by-products, or mixtures of different materials for amelioration, remediation, or recultivation is one option to regulate material fluxes.

INTRODUCTION

Anthropogenic activities such as mining and smelting, combustion of waste and fossil fuels, as well as the use of various organic and inorganic chemicals and radionuclides in agriculture and industry entail risks for the overall environment and human beings. The awareness of these risks due to industrial wastes emerged through numerous cases of severe environmental impact many years after their disposal. Notorious examples include hazardous waste disposal in the Love Canal, New York, U.S.A.,^[1] Lekkerkerk near Rotterdam, the Netherlands,^[2] or the collapsed tailing dam containing acidic pyrite sludge in Donana, Spain.^[3]

Besides the negative effects of industrial waste, their reuse potential for soil amelioration, remediation, and recultivation should also be considered. Future perspectives and visions for handling industrial waste must be discussed in terms of the potential environmental risks and how to minimize them.

POTENTIAL POLLUTION BY INDUSTRIAL WASTE

Generally, industrial activities generate the following three categories of waste: 1) gaseous emissions; 2) wastewater; and 3) solid waste. Gaseous pollutants or particulates are generated by combusting fossil fuels and wastes; they are found both close to the emittent and far away, especially in forest ecosystems due to their filter capacity.^[4] Dispersion

and long-term transport of heavy metals, radionuclides, and persistent organic pollutants occur.^[5] Acidification and eutrophication are due to the transboundary distribution of nutrients involved in combustion processes (e.g., sulfur and nitrogen compounds). Wastewater is derived from domestic and industrial establishments; in both the cases, they are known as municipal wastewater. Surface and groundwater are the most threatened sinks of pollutants, besides irrigated areas. Upon removal of water, wastewater turns to sludge and further to solid waste. Generally, solid waste generation starts with mining and petroleum production and with the subsequent processing of these raw materials to goods for consumption. Emerging waste includes mineral ores, tailings, slags, oil-contaminated soils, and residues that stem from the processing of these materials.

Industrial activities generate mostly mixed wastes, yielding transient waste types. Pollutants can be released in gaseous, particulate, aqueous, or solid form, depending on the industrial processes involved. They emanate from point or diffuse sources [see the entries *Pollution: Point Source* (p. 1776) and *Pollution: Non-Point Source* (p. 1765) for more information]. Generally, pollution tends to be highest close to emittents and declines with increasing distance. However, the elevated concentrations of persistent organic chemicals have been found in environments far from pollution sources, such as in the Arctic, in mountain regions, and in forest soils.^[6]

Soil quality is prone to degradation through 1) physical; 2) chemical; and 3) biological processes, resulting in soil

erosion, reduced productivity, and soil and groundwater contamination. Detrimental effects of industrial waste on soil comprise 1) siltation and compaction through the application of fine-grained and salt-rich materials; 2) any type of pollution, acidification, or landfilling; and 3) loss of macro- and microfaunal diversity by intoxication. Heavy metals (lead, mercury, and cadmium), metalloids (arsenic and antimony), and organic compounds (e.g., dioxins and other persistent organic pollutants) are the most relevant soil pollutants. Waste generation has dramatically increased from chemical manufacturing, wastewater treatment, agriculture, mining and smelting, food processing, energy production, pulp and paper production, and other industries. The subsequent waste–environmental interactions have potential consequences on soil and water qualities. Estimates show that up to 75% of the materials used in industrial processes do not end up in final products.^[7]

The prevention of industrial waste production, enhanced by the optimization of processes reducing the amount of generated waste and the concentration of pollutants in waste streams, has a higher priority than end-of-pipe technologies such as soil remediation.

POTENTIAL USE OF INDUSTRIAL WASTE

Soil Amelioration

In the case of metals, humankind has imbalanced biogeochemical cycles through the use of metals. Considerable amounts are lost in technical processes into the abovementioned waste categories. Electronic waste is one of the most rapidly growing problems. Waste volume estimation for 1998 is given in Table 1. Instead of exporting this waste from “developed” to “developing” countries and thereby contaminating soil and water due to the lack of legal regulations, the strategy should be to promote recycling and reuse of metals. Besides heavy metal-contaminated arable land, nutrient deficient areas are widespread over the world. Here, the reuse of copper, manganese (Mn), molybdenum,

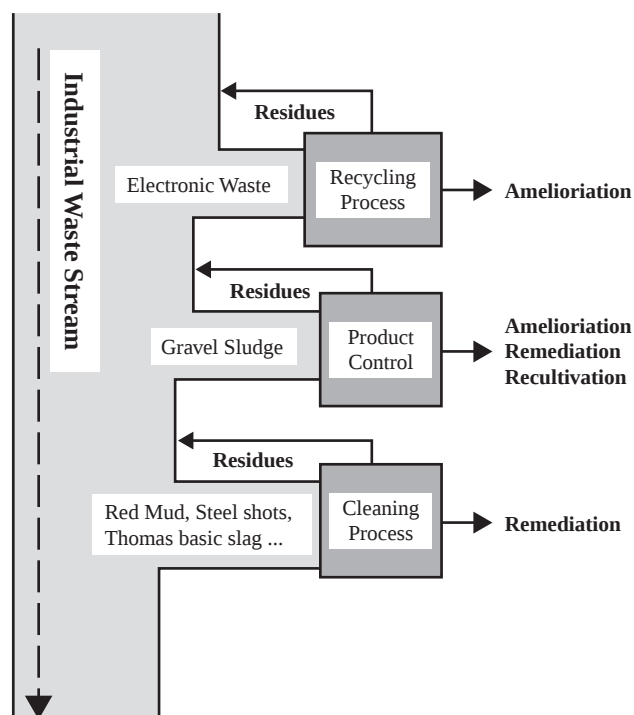


Fig. 1 A schematic overview of waste reuse possibilities.

iron (Fe), and zinc from electronic waste as nutrients could contribute toward more sustainable systems.

Gravel sludge, a fine-grained by-product of the gravel industry—applied on sandy soils poor in nutrients—can improve soil physical properties and crop production.

On the other hand, the use of industrial waste materials as soil amendments has been estimated to contaminate thousands of hectares of productive agricultural land in countries throughout the world. This is a particular problem in the Asian Pacific region, where land-based disposal of untreated wastes is being practiced due to lack of regulatory guidelines.^[12] If crops are cultivated on contaminated soils, they may become less competitive on the world market, provided efficient quality control is implemented. Fig. 1 shows a schematic overview of waste reuse possibilities.

Table 1 Industrial waste quantities.

Industry	Industrial waste	Produced amount (Mg ^a)
Electronic industry	Electronic waste	5–7 Mio worldwide ^[8]
Alumina industry	Red mud	60 Mio worldwide ^[9]
Pulp and paper industry	Paper mill sludge	9 Mio in the United States ^[10]
Iron and steel industry	Slags	300 Mio ^a worldwide ^[11]
Energy production	Gypsum	7 Mio in Europe ^[11]
Energy production	Fly ash	37 Mio in Europe ^[11]

^aCalculated from the produced amount of iron and steel.

Soil Remediation

In situ fixation of labile metals in soils by the addition of amendments is one option to remediate heavy metal-contaminated soils. The aim is to reduce the risk of contamination by balancing ecological and economical needs. Several industrial wastes or by-products have been screened for their ability to immobilize heavy metals. Red mud is produced by the alumina industry around the world in countries like Jamaica, Australia, the United States, and Germany (Table 1). Results of greenhouse and outdoor experiments indicate immobilization of many heavy metals in soil by red mud.^[13] Similar results were obtained for

steel shots, a waste product of the metal shaping industry, Thomas phosphate basic slag from the steel industry, and "Iron Rich," generated by the titanium dioxide production. Fly ash, a coal combustion residue, and beringite, a modified aluminosilicate that originates from the fluidized bed burning of coal refuse from a former coal mine in Beringen (Belgium), also have potential for metal immobilization.^[14] The composition of these amendments varies considerably, and the immobilizing effect is mostly due to aluminum-, Fe-, Mn-oxides, phosphates, silicates, and alkaline materials.

The application rates of these amendments up to 5% (w/w) are practicable and result in establishing a vegetation cover on bare land, the reduction of visual symptoms of metal phytotoxicity, and a significant reduction of the metal accumulation in aboveground biomass.^[15]

Besides the advantages of these materials, disadvantages or failures may also occur due to indigenous contamination of the materials.^[13] The recommendation is to adjust the amendment and application rate on a case-by-case basis. In certain cases, removal of contaminants may be required.

CONCLUSION

Pollution by industrial waste is widespread throughout the world, adversely affecting soil and environment qualities. Management of these wastes is necessary to achieve a more sustainable system. After exhausting the two approaches of preventing waste generation and recycling suitable materials, the use of wastes, by-products, or mixtures of different materials for amelioration, remediation, or recultivation is one option to regulate material fluxes. Besides considering the advantages of using waste materials, the disadvantages have to be taken into account and must be assessed on a case-by-case basis.

REFERENCES

1. Manhan, S.E. *Environmental Chemistry*, 6th Ed.; Lewis Publisher: New York, 1994.
2. Alloway, B.J.; Ayres, D.C. *Chemical Principles of Environmental Pollution*; Chapman & Hall: London, 1993.
3. Meharg, A.A.; Osborn, D.; Pain, D.J.; Sanchez, A.; Naveso, M.A. Contamination of donana food chain after the
4. Schulte, A.; Blum, W.E.H. Schwermetalle im waldökosystem. In *Geochemie und Umwelt*; Matschullat, J., Tobschall, Voigt, H.J., Eds.; Springer: Berlin, 1997; 53–74.
5. Blum, W.E.H. Soil degradation caused by industrialization and urbanization. In *Towards Sustainable Land Use*; Blume, H.P., Eger, H., Fleischhauer, E., Hebel, A., Reij, C., Steiner, K.G., Eds.; Catena: Reiskirchen, 1998; Vol. I, 755–766.
6. Wania, F. On the origin of elevated levels of persistent chemicals in the environment. *Environ. Sci. Pollut. Res.* **1999**, *6* (1), 11–19.
7. Onida, M. Innovation in prevention of quantitative and qualitative waste production. In *Innovation in Waste Management*, Proceedings of the IV European Waste Forum, Milano, Italy, Nov 30–Dec 1, 2000; Centro de Ingegneria per la Protezione dell'Ambiente (C.I.P.A.), Ed.; European Waste Forum: Milano, 2000; 91–105.
8. Puckett, J.; Byster, L.; Weservelt, S.; Gutierrez, R.; Davis, S.; Hussain, A.; Dutta, M. *Exporting Harm: The High-Tech Trashing of Asia*; The Basel Action Network: Seattle, 2002.
9. Wong, J.W.C.; Ho, G.E. Cation exchange behavior of bauxite refining residues from western Australia. *J. Environ. Qual.* **1995**, *24*, 461–466.
10. Bonomo, L.; Higginson, A.E. *International Overview on Solid Waste Management*; Academic Press: London, 1988.
11. Hackl, A.E.; Sammer, G.; Winter, B. *UN-ECE Task Force Management on By-Products/Residues Containing Heavy Metals and/or Persistent Organic Pollutants*; Status Report CP-031; Federal Environment Agency Ltd: Vienna, 2001.
12. Ullah, S.M.; Nuruzzaman, M.; Gerzabek, M.H. Heavy metal and microbiological pollution of water and sediments by industrial wastes, effluents and slums around Dhaka city. In *Tropical Limnology III*; Timotius, K.H., Göltenboth, F., Eds.; Satya Wacann University Press: Salatiga, 1995; 179–186.
13. Friesl, W. Gentle soil remediation: Immobilization of heavy metals by soil amendments. Thesis, University of Agricultural Sciences, Institute of Soil Science, Vienna, 2002.
14. Knox, A.S.; Seaman, J.C.; Mench, M.J.; Vangronsveld, J. Remediation of metal- and radionuclides-contaminated soils by *in situ* stabilization techniques. In *Environmental Restoration of Metal-Contaminated Soils*; Iskandar, I.K., Ed.; Lewis Publishers: Boca Raton, 2000; 21–60.
15. Vangronsveld, J.; Van Assche, F.; Clijsters, H. Reclamation of a bare industrial area contaminated by non-ferrous metals: *In situ* metal immobilization and revegetation. *Environ. Pollut.* **1995**, *87*, 51–59.

Pollution: Non-Point Source

Ravendra Naidu

Mallavarapu Megharaj

Peter Dillon

Rai Kookana

Ray Correll

*Commonwealth Scientific and Industrial Research Organisation (CSIRO),
Adelaide, South Australia, Australia*

Walter W. Wenzel

*Institute of Soil Research, University of Natural Resources and Applied Life
Sciences, Vienna, Austria*

Abstract

Environmental contaminants can have a deleterious effect on non-target organisms and their beneficial activities. The sampling requirements of non-point source pollution (NPSP) are quite different from those of the point source contamination. This entry reviews some strategies for evaluating and combatting the effects of NPSP.

INTRODUCTION

Non-point source pollution (NPSP) has no obvious single point source discharge and is of diffuse nature (Table 1). An example of NPSP includes aerial transport and deposition of contaminants such as sulfur dioxide from industrial emissions leading to acidification of soil and water bodies. Rain water in urban areas could also be a source of NPSP as it may concentrate organic and inorganic contaminants. Examples of such contaminants include polycyclic aromatic hydrocarbons, pesticides, and polychlorinated biphenyls that could be present in urban air due to road traffic, domestic heating, industrial emissions, and agricultural treatments.^[1–3] Other examples of NPSP include fertilizer [especially cadmium (Cd), nitrogen (N), and phosphorus (P)] and pesticide applications to improve crop yield. Use of industrial waste materials as soil amendments have been estimated to contaminate thousands of hectares of productive agricultural land in countries throughout the world.

CONTAMINANT INTERACTIONS

Non-point pollution is generally associated with low-level contamination spread at broad acre level. Under these circumstances, the major reaction controlling contaminant interactions is sorption–desorption processes, plant uptake, surface runoff, and leaching. However, certain contaminants, in particular organic compounds are also subjected

to volatilization, chemical, and biological degradation. Sorption–desorption and degradation (both biotic and abiotic) are the two most important processes controlling organic contaminant behavior in soils. These processes are influenced by both soil and solution properties of the environment. Such interactions also determine the bioavailability and/or transport of contaminants in soils. Where the contaminants are bioavailable, risk to surface- and ground-water and soil, crop, and human health is enhanced.

IMPLICATIONS TO SOIL AND ENVIRONMENTAL QUALITY

Environmental contaminants can have a deleterious effect on non-target organisms and their beneficial activities. These effects could include a decline in primary production, decreased rate of organic matter break down, and nutrient cycling as well as mineralization of harmful substances that in turn cause a loss of productivity of the ecosystems. Certain pollutants, even though present in very small concentrations in the soil and surrounding water, have potential to be taken up by various microorganisms, plants, animals, and ultimately human beings. These pollutants may accumulate and concentrate in the food chain by several thousand times through a process referred to as biomagnification.

Urban sewage, because of its nutrient values and source of organic carbon in soils, is increasingly being disposed to

Table 1 Industries, land uses, and associated chemicals contributing to NPSP.

Industry	Type of chemical	Associated chemicals
Agricultural activities	Metals/metalloid	Cd, mercury, arsenic, selenium
	Non-metals	Nitrate, phosphate, borate
	Salinity/sodicity	Sodium, chloride, sulfate, magnesium, alkalinity
	Pesticides	Range of organic and inorganic pesticides including arsenic, copper, zinc, lead, sulfonylureas, organochlorine, organophosphates, etc., salt, geogenic contaminants (e.g., arsenic and selenium)
Automobile and industrial emissions	Irrigation	Sodium, chloride, arsenic, selenium
	Dust	Lead, arsenic, copper, Cd, zinc, etc.
	Gas	Sulfur oxides, carbon oxides
	Metals	Lead and lead organic compounds
Rain water	Organics	Polyaromatic hydrocarbons, polychlorbiphenyls, etc.
	Inorganic	Sulfur oxides, carbon oxides acidity, metals, and metalloids

Source: From Barzi, Naidu, et al.^[4]

land. The contaminants present in sewage sludge (nutrients, heavy metals, organic compounds, and pathogens), if not managed properly, could potentially affect the environment adversely. Dumping of radioactive waste (e.g., radium, uranium, and plutonium) onto soil is more complicated because these materials remain active for thousands of years in the soil and thus pose a continued threat to the health of the ecosystem.

Industrial wastes, improper agricultural techniques, municipal wastes, and use of saline water for irrigation under high evaporative conditions result in the presence of excess soluble salts (predominantly sodium and chlorine ions) and metalloids such as selenium and arsenic in soils. Salinity and sodicity affect the vegetation by inhibiting seed germination, decreasing permeability of roots to water, and disrupting their functions such as photosynthesis, respiration, and synthesis of proteins and enzymes.

Some of the impacts of soil pollution migrate a long way from the source and can persist for some time. For example, suspended solids can increase water turbidity in streams, affecting benthic and pelagic aquatic ecosystems, filling reservoirs with unwanted silt, and requiring water treatment systems for potable water supplies. P attached to soil particles, which are washed from a paddock into a stream, can

dominate nutrient loads in streams and downstream water bodies. Consequences include increases in algal biomass, reduced oxygen concentrations, impaired habitat for aquatic species, and even possible production of cyanobacterial toxins, with serious impacts for humans and livestock consuming the water. Where waters discharge into estuaries, N can be the limiting factor for eutrophication; estuaries of some catchments where fertilizer use is extensive have suffered from excessive sea grass and algal growth.

More insidious is the leaching of nutrients, agricultural chemicals, and hydrocarbons to groundwater. Incremental increases in concentrations in groundwater may be observed over long periods of time resulting in initially potable water becoming undrinkable and then some of the highest valued uses of the resource may be lost for decades. This problem is most severe on tropical islands with shallow relief and some deltaic arsenopyrite deposits, where wells cannot be deepened to avoid polluted groundwater because underlying groundwater is either saline or contains too much As.

SAMPLING FOR NPSP

The sampling requirements of NPSP are quite different from those of the point source contamination. Typically, the sampling is required to give a good estimate of the mean level of pollution rather than to delineate areas of pollution. In such a situation, sampling is typically carried out on a regular square or a triangular grid. Furthermore, gains may be possible by using composite sampling.^[5] However, if the pollution is patchy, other strategies may be used. One such strategy is to divide the area into remediation units and to sample each of these. The possibility of movement of the pollutant from the soil to some receptor (or asset) is assessed, and the potential harm is quantified. This process requires an analysis of the bioavailability of the pollutant, pathway analysis, and the toxicological risk. The risk analysis is then assessed and decisions are then made as to how the risk should be managed.

MANAGEMENT AND OR REMEDIATION OF NPSP

The treatment strategies used for managing NPSP are generally those that modify the soil properties to decrease the bioavailable contaminant fraction. This is particularly so in the rural agricultural environment where soil–plant transfer of contaminants is of greatest concern. Soil amendments commonly used include those that change the ion-exchange characteristics of the colloid particles and those that enhance the ability of soils to sorb contaminants. An example of NPSP management includes the application of lime to immobilize metals because the solubility of most heavy metals decreases with increasing soil pH. However, this approach is not applicable to all metals, especially those that form oxyanions—the bioavailability of such species

increases with increasing pH. Therefore, one of the prerequisites for remediating contaminated sites is a detailed assessment of the nature of contaminants present in the soil. The application of a modified aluminosilicate to a highly contaminated soil around a zinc smelter in Belgium was shown to reduce the bioavailability of metals thereby reducing the Zn phytotoxicity.^[6] The simple addition of rock phosphates to form lead (Pb) phosphate has also been demonstrated to reduce the bioavailability of Pb in aqueous solutions and contaminated soils due to immobilization in the metal.^[7] Nevertheless, there is concern over the long-term stability of the processes. The immobilization process appears attractive given that there are very few cheap and effective *in situ* remediation techniques for metal-contaminated soils. A novel, innovative approach is using higher plants to stabilize, extract, degrade, or volatilize inorganic and organic contaminants for *in situ* treatment (cleanup or containment) of polluted topsoils.^[8]

PREVENTING WATER POLLUTION

The key to preventing water pollution from the soil zone is to manage the source of pollution. For example, nitrate pollution of groundwater will always occur if there is excess nitrate in the soil at a time when there is excess water leaching through the soil. This suggests that we should aim to reduce the N in the soil during wet seasons and the drainage through the soil. Local research may be needed to demonstrate the success of best management techniques in reducing nutrient, sediment, metal, and chemical exports via surface runoff and infiltration to groundwater. Production figures from the same experiments may also convince local farmers of the benefits of maintaining nutrients and chemicals where needed by a crop rather than losing them off site and facilitate uptake of best management practices.

GLOBAL CHALLENGES AND RESPONSIBILITY

The biosphere is a life supporting system to the living organisms. Each species in this system has a role to play and thus every species is important and biological diversity is vital for ecosystem health and functioning. The detection of hazardous compounds in Antarctica, where these compounds were never used or no man has ever lived before, indicates how serious is the problem of long-range

atmospheric transport and deposition of these pollutants. Clearly, pollution knows no boundaries. This ubiquitous pollution has had a global effect on our soils, which in turn has been affecting their biological health and productivity. Coupled with this, over 100,000 chemicals are being used in countries throughout the world. Further focus has been on the endocrine disruptor chemicals that mimic natural hormones and do great harm to animal and human reproductive cycles.

These pollutants are only a few examples of contaminants that are found in the terrestrial environment.

REFERENCES

1. Chan, C.H.; Bruce, G.; Harrison, B. Wet deposition of organochlorine pesticides and polychlorinated biphenyls to the Great Lakes. *J. Great Lakes Res.* **1994**, *20*, 546–560.
2. Lodovici, M.; Dolara, P.; Taiti, S.; Del Carmine, P.; Bernardi, L.; Agati, L.; Ciappellano, S. Polycyclic aromatic hydrocarbons in the leaves of the evergreen tree *Laurus Nobilis*. *Sci. Total Environ.* **1994**, *153*, 61–68.
3. Sweet, C.W.; Murphy, T.J.; Bannasch, J.H.; Kelsey, C.A.; Hong, J. Atmospheric deposition of PCBs into green bay. *J. Great Lakes Res.* **1993**, *19*, 109–128.
4. Barzi, F.; Naidu, R.; McLaughlin, M.J. Contaminants and the Australian soil environment. In *Contaminants and the Soil Environment in the Australasia-Pacific Region*; Naidu, R., Kookana, R.S., Oliver, D., Rogers, S., McLaughlin, M.J., Eds.; Kluwer Academic Publishers: Dordrecht, 1996; 451–484.
5. Patil, G.P.; Gore, S.D.; Johnson, G.D. *Manual on Statistical Design and Analysis with Composite Samples*, Technical Report No. 96-0501; EPA Observational Economy Series, Center for Statistical Ecology and Environmental Statistics: Pennsylvania State University: University Park, 1996; Vol. 3.
6. Vangronsveld, J.; Van Assche, F.; Clijsters, H. Reclamation of a bare industrial area, contaminated by non-ferrous metals: *In situ* metal immobilisation and revegetation. *Environ. Pollut.* **1995**, *87*, 51–59.
7. Ma, Q.Y.; Logan, T.J.; Traina, S.J. Lead immobilisation from aqueous solutions and contaminated soils using phosphate rocks. *Environ. Sci. Technol.* **1995**, *29*, 1118–1126.
8. Wenzel, W.W.; Adriano, D.C.; Salt, D.; Smith, R. Phytoremediation: A plant-microbe based remediation system. In *Bioremediation of Contaminated Soils*; Adriano, D.C., Bollag, J.M., Frankenberger, W.T., Jr., Sims, W.R., Eds.; Soil Science Society of America Special Monograph No. 37; Soil Science Society of America: Madison, 1999; 772 pp.

Pollution: Organic and Inorganic

A. Paul Schwab

Department of Agronomy, Purdue University, West Lafayette, Indiana, U.S.A.

Abstract

This entry focuses on pollutants that have anthropogenic origins and addresses organic and inorganic chemicals. Organic contaminants can include pesticides, solvents, preservatives, and petroleum products. Inorganic pollutants include heavy metals, non-metals, metalloids, radionuclides, and simple soluble salts. Whether the source of the contaminants is natural or anthropogenic, an understanding of the chemistry, toxicity, and bioavailability is crucial in responding to soil pollution.

INTRODUCTION

A soil pollutant can be broadly defined as any chemical or other substance that either is not normally found in soil or is present at high enough concentrations to be harmful to any living organisms.^[1] This very general definition could be applied to man-made (anthropogenic) as well as naturally occurring constituents. This entry will focus only on those pollutants that have anthropogenic origins and will address carbon-based (organic) and non-carbonaceous (inorganic) chemicals. Organic contaminants can include pesticides, solvents, preservatives, and petroleum products. Inorganic pollutants include heavy metals [e.g., lead (Pb) and mercury (Hg)], non-metals (selenium), metalloids [arsenic (As) and antimony (Sb)], radionuclides, and simple soluble salts (sodium chloride). Whether the source of the contaminants is natural or anthropogenic, an understanding of the chemistry, toxicity, and bioavailability is crucial in responding to soil pollution.

ORGANIC POLLUTANTS

Thousands of organic chemicals exist in nature, and many are acutely toxic as well as carcinogenic. When discussing toxic organic chemicals in soils, we normally focus on those pollutants resulting from human activities because their release has the potential to be controlled. Typical synthetic organic pollutants include fuels, lubricants, herbicides, fungicides, insecticides, solvents, and propellants.

Soil Contamination by Organic Chemicals

Organic chemicals can find their way into the soil accidentally through spills, leaking storage tanks, and unintentional discharges. However, not all soil pollution is accidental. More than one-half million metric tons of pesticides are used annually, the majority of which are used on agricultural fields.^[1] Approximately, 20,000 metric tons of

pesticides are used in non-agricultural applications including railroad right-of-way weed control, turf, and horticulture. Although many pesticides do not persist in soils, others have been studied extensively because of their potential negative impacts. Chlordane (termite control), dichlorodiphenyltrichloroethane (DDT; mosquitoes), and atrazine (weeds) are excellent examples of organic pesticides that were thought to have human health effects; chlordane and DDT have been banned in the United States, and the banning of atrazine has been debated.

Thousands of organic chemicals are in use, and listing all the specific compounds that contaminate soils is impossible. However, it is possible to discuss the categories of chemicals that are used frequently and are commonly found in soils (Table 1). Pesticides may be the most frequently encountered, and some pesticides are quite toxic. Many of the other classes of the compounds listed are of great concern when found in soil, again due to their potential toxicity to a wide range of organisms.

Potential Impacts of Organic Contaminants

After an organism is exposed to an organic pollutant, a number of antagonistic effects are possible if concentrations are high enough. The most dramatic impact is acute toxicity in which symptoms are quickly apparent and readily identified. At lower concentrations, the impacts become less obvious and take longer to be expressed. For many carcinogens, e.g., decades are required to develop the cancerous tumors. Pathway of exposure, concentrations, and duration of the exposure all dictate the resulting health effects and are essential components of risk analysis.

Bioremediation of Organic Contaminants

Although contamination of soil by organic compounds is an important environmental problem, these pollutants can be removed from the soil through bioremediation. Soil

Table 1 Some classes of organic chemicals found in soils.

Contaminant class	Example(s)	Sources and impacts
Insecticides	DDT, chlordane, diazinon	Used for insect control; DDT and chlordane have serious health effects; diazinon found in water supplies
Herbicides	Atrazine, 2,4-D, 2,4,5-T	Atrazine used on corn; 2,4-D widely used in lawns and agriculture; defoliant 2,4,5-T implicated in health effects in “agent orange”
Fungicides	Benomyl, propiconazole, chlorothalonil	Agricultural fungicides applied to soil directly
Nematicides	Methyl bromide	Used in fumigation of soil to remove nematodes; neurotoxin
Solvents	Trichloroethylene, trichloroethane	Widely used solvents and degreasers; chronic health effects not clear
Fuels	Diesel, kerosene	From spills, leaking tanks. Can lead to groundwater contamination
Polyaromatic hydrocarbons	Chrysene, benzo[a]pyrene	Components of petroleum, particularly after combustion; many of these are carcinogenic
Explosives, propellants	TNT, RDX	Residuals from manufacture the largest source of contamination; acute and chronic toxicities documented

Source: From Schwarzenbach, Gschwend, et al.^[2] and Evangelou.^[3]

microorganisms have a remarkable capacity to degrade organic contaminants. Degradation can be direct, using the organics as a source of energy and carbon, or indirect in which the compounds are cometabolized by organisms seeking similar compounds. End products can be CO₂(g) after total mineralization, alteration, and humification, or incorporation into the microbial biomass. (See the entry *Pollution: Persistent Organic*, p. 1771.)

INORGANIC POLLUTANTS

Inorganic pollutants are those contaminant compounds that do not contain carbon and include the acutely toxic heavy metals, As, radioactive elements, and the far less toxic soluble salts. Although all elements are naturally occurring, thousands of inorganic compounds are formed only through human activities. Some are highly toxic in minute amounts, but all can be harmful when concentrations are high enough.

Classes of Inorganic Pollutants and Concentration Ranges

As with organic pollutants, there are thousands of inorganic contaminants, and they are found over a wide range of concentrations. For discussion, it is convenient to group them in classes (Table 2) consistent with the periodic table. Metals include Pb, zinc (Zn), nickel (Ni), and cadmium (Cd), and their typical soil concentrations vary over a wide range, as the concentrations considered anomalous. Most metals in soils occur as the free cations in solution and in the solid phase. Nearly all metals can have toxic effects on humans although some metals (such as Zn) require high concentrations for toxicity. Metalloids, including As and Sb, generally occur in soil solution and solid phases as oxyanions (such as AsO₄³⁻), resulting in a significantly different chemical behavior than the metals. Metalloids can be highly toxic. Soluble salts are typical inorganic species that, in normal concentrations, do not constitute a health or environmental threat. Typical salts are sulfates, chlorides, and

Table 2 Inorganic contaminants in soil, their typical ranges found in uncontaminated soils, and those concentrations considered to be unusual.

Contaminant	Sources and impacts	Typical range in soil (mg/kg)	Extreme concentrations (mg/kg)
Metals			
Pb, Zn, Cd, Cu, Ni, Ba, Sr, Mn, Cr	Mining, smelting, electroplating	0.1–1,000	25 to >10,000 (metal dependent)
Metalloids			
As, B, Ge, Sb	Industrial activities, smelting	<0.1–100	>50
Non-metals			
Se, I, P, S	Manufacturing, processing	<0.1–1,000	Up to 2,000 (element dependent)
Soluble salts			
Halides, alkalis, alkaline earths, sulfates, nitrates	Mining, manufacturing, petroleum extraction, natural sources	<500	>1,000

Source: From Evangelou^[3] and McBride.^[4]

nitrate of calcium, magnesium, potassium, and sodium. These components are always present in soil and do not constitute an environmental problem until concentrations become excessive (Table 2).

Pathways of Contamination

Once in the soil organic and inorganic pollutants can follow many pathways to the target organisms.^[5] For all contaminants, the direct consumption of contaminated soil can be an important mechanism of exposure for soilborne organisms, grazing mammals that also consume the soil, and humans. For some pollutants, the direct soil consumption is the only means of exposure if they have very low mobility and low bioavailability, e.g., Pb and 5- or 6-ring polyaromatic hydrocarbons.

There are approximately eight pathways by which contaminants in soil can reach humans as the target organism.^[5] Half of the mechanisms involve plant uptake and subsequent consumption of the plants. Most anionic inorganic species are readily assimilated by plants, as are many cationic species. However, some inorganic species form highly insoluble solid phases or are otherwise non-bioavailable, and they are not translocated with the plant. For non-ionic organic species, plant uptake is a function of water solubility and hydrophobicity/lipophilicity. Compounds with intermediate lipophilicity are most suitable for plant uptake. Once the pollutants are in the plant, organisms that consume the vegetation may be exposed to the pollutant, and some of the contaminants are passed along the food chain.

Other pathways of exposure include movement of the contaminant from the soil to the water, air, or becoming airborne on dust/direct volatilization. All of these mechanisms have been shown to contribute to negative health effects. Drinking contaminated water is one of the better known pathways. If the drinking water originates from groundwater, then the pollutants must be soluble in water and readily percolate through the soil and into the groundwater. When surface water is used for drinking, the transport mechanism can be movement of dissolved contaminant in the water, or contaminants may be adsorbed onto the suspended sediment in the soil runoff. Virtually, every soil pollutant can be transported by this mechanism, and only careful control of runoff water and sediments can avoid this problem.

Direct volatilization from the soil into the air is important for a limited number of contaminants, but when the volatilization of a potentially compound or element occurs, exposure is directly through the lungs. Hg is a well-known, volatile metal, and the direct inhalation of high concentrations of gaseous Hg or dimethyl Hg can be poisonous. The inhalation of airborne, contaminated particles is a more frequent occurrence but less insidious. After inhalation, contaminants must be released from the dust before they can be assimilated. The inhalation of contaminated particulates can be reduced possibly through standard soil erosion measures, such as establishing wind breaks and adequate plant cover on contaminated soils.

Soil Remediation Options for Inorganic and Organic Pollutants

The best approach to keeping soil free from contamination is prevention. When this approach fails, removing the contaminants may be necessary, and many options are available.^[6] One of the most commonly used approaches is to excavate the soil and bury it in a hazardous waste landfill. Soils may be incinerated to burn off organic pollutants and to volatilize metals. Soil washing/leaching procedures are available in which the soil is exposed to a solvent that can dissolve that contaminants. The solutions of chelating agents can be effective in the soil washing of metals, and organic solvents can be used to extract organic pollutants. Other approaches include *in situ* vitrification (melting the soil into a block of glass using heat) and chemical immobilization. In all these cases, considerable waste is generated, and the soil is usually lost as a viable resource.

Less destructive approaches are often preferred because the soil is not seriously altered or destroyed, and the techniques are often less expensive. Electrokinesis or electromigration involves applying an electrical potential to the soil and forcing metal ions to migrate to the electrodes. An evolving technology is the use of higher plants to remove contaminants from soil, and this technique is called phytoremediation.^[7,8] For inorganic contaminants, the pollutants are physically removed from the soil. Organic contaminants can be taken up by the plants, degraded in the soil, or volatilized. Electrokinesis and phytoremediation leave the soil as a productive resource, but they take longer than some of the more aggressive alternatives.

REFERENCES

1. Pierzynski, G.M.; Sims, J.T.; Vance, G.F. *Soils and Environmental Quality*; Lewis Publishers: Boca Raton, 1993; 47–50.
2. Schwarzenbach, R.P.; Gschwend, P.M.; Imboden, D.M. *Environmental Organic Chemistry*; Wiley-Interscience: New York, 1992; 8–41.
3. Evangelou, V.P. *Environmental Soil and Water Chemistry*; Wiley-Interscience: New York, 1998; 476–499.
4. McBride, M.B. *Environmental Chemistry of Soils*; Oxford Press: New York, 1994; 308–350.
5. Chaney, R.L.; Ryan, J.A.; Brown, S. Environmentally acceptable endpoints for soil metals. In *Environmentally Acceptable Availability of Chlorinated Organics, Explosives, and Metals in Soils*; Anderson, W.C., Loehr, R.C., Reible, D., Eds.; American Academy of Environmental Engineering: Annapolis, 1999; 111–155.
6. LaGrega, M.D.; Buckingham, P.L.; Evans, J.C. *Hazardous Waste Management*; McGraw-Hill: New York, 1994; 437–831.
7. Terry, N.; Banuelos, G. *Phytoremediation of Contaminated Soil and Water*; Lewis Publishers: New York, 1999; 388 pp.
8. Banks, M.K.; Govindaraju, R.S.; Schwab, A.P.; Kulakow, P.; Finn, J. *Phytoremediation of Hydrocarbon Contaminated Soils*; Lewis Publishers: Boca Raton, 1999; 175 pp.

Pollution: Persistent Organic

Mark Radosevich

University of Delaware, Newark, Delaware, U.S.A.

E. Danielle Rhine

Department of Plant and Soil Sciences, University of Delaware, Newark, Delaware, U.S.A.

Abstract

Biodegradation is the primary dissipation pathway for most organic pollutants in soils. Three prevalent factors related to chemical structure that commonly inhibit the biodegradation of organic pollutants are as follows: 1) inherent susceptibility to enzymatic attack; 2) bioavailability; and 3) nutrient availability. Inherently recalcitrant xenobiotic chemicals (synthetic chemicals with no natural analogs) can persist in environments that lack sufficient numbers of microorganisms with the necessary catabolic pathways. However, the capacity of soil microbial communities to degrade toxic chemicals can increase with repeated chemical exposure. This suggests that the genetic potential to metabolize the chemical is initially present and the capacity to degrade the pollutant could be realized through community adaptation. Understanding the biological fate of these hazardous substances is of critical importance for the protection of human and ecosystem health.

MOLECULAR RECALCITRANCE

Many toxic organic chemicals are intrinsically stable (i.e., recalcitrant) in natural environments due to their fundamental structural properties. The structure of a compound can influence its stability by: 1) inhibiting its susceptibility to enzymatic attack or 2) causing favorable interactions (i.e., sorption) with the soil matrix that result in rate-limited mass transfer between the soil and aqueous phases thus reducing the amount of contaminant that is available for microbial metabolism.

Electrophillic Attack by Oxygenase Enzymes

Several generalizations can be made regarding the chemical structure and susceptibility to enzymatic attack involving two key mechanisms: steric hindrance and electron distribution effects. Under aerobic conditions (oxygen present), many xenobiotic compounds, especially aromatic and aliphatic hydrocarbons can be transformed by the electrophilic addition of oxygen by a class of enzymes known as oxygenases. These enzymes produce an electrophilic form of oxygen at the enzyme active site that subsequently attacks electron-dense sites on the substrate molecule.^[1] In the case of aromatic ring structures, electron-withdrawing substituents, such as halogens, lower the electron density in the ring, rendering it less susceptible to attack by oxygenase enzymes. Conversely, electron-donating substituents such as alkyl groups increase both the electron density in the ring and the oxygenase-mediated reactions. A classic example

of the electronic effects on enzymatic reaction rates is the relative rate of transformation of the herbicide 2,4-D compared to its structural analog 2,4,5-T (Fig. 1). These two molecules are identical with the exception of an additional chlorine atom in position five on the aromatic ring of 2,4,5-T. Addition of this chlorine atom inhibits the electrophilic attack of the acetic acid side chain and dramatically increases the stability of this compound relative to 2,4-D, in natural environments. Halogen substitution of aromatic and aliphatic compounds in key positions can not only block metabolism, producing dead-end metabolites, but also can yield products (e.g., acyl halides) that are lethal to the microorganism.^[1] Another classic example is that of polychlorinated biphenyls and pentachlorophenol (PCP). Biological degradation of these compounds decreases dramatically as the degree of halogenation increases (Fig. 1).

Increasing the degree of halogenation generally stabilizes the organic compounds in aerobic environments. Under anaerobic conditions (oxygen absent), however, these chemicals can be transformed via reductive processes. Reductive reactions usually result in detoxification of the compound and in some cases are involved in the energy-yielding metabolism of some organisms. In such cases, the organism directly transfers electrons (reducing equivalents) from an electron donor such as lactate, acetate, or H₂(g) to the halogenated compound. This terminal electron accepting process is a form of respiration and is analogous to oxidation of carbon substrates using oxygen as an electron acceptor.

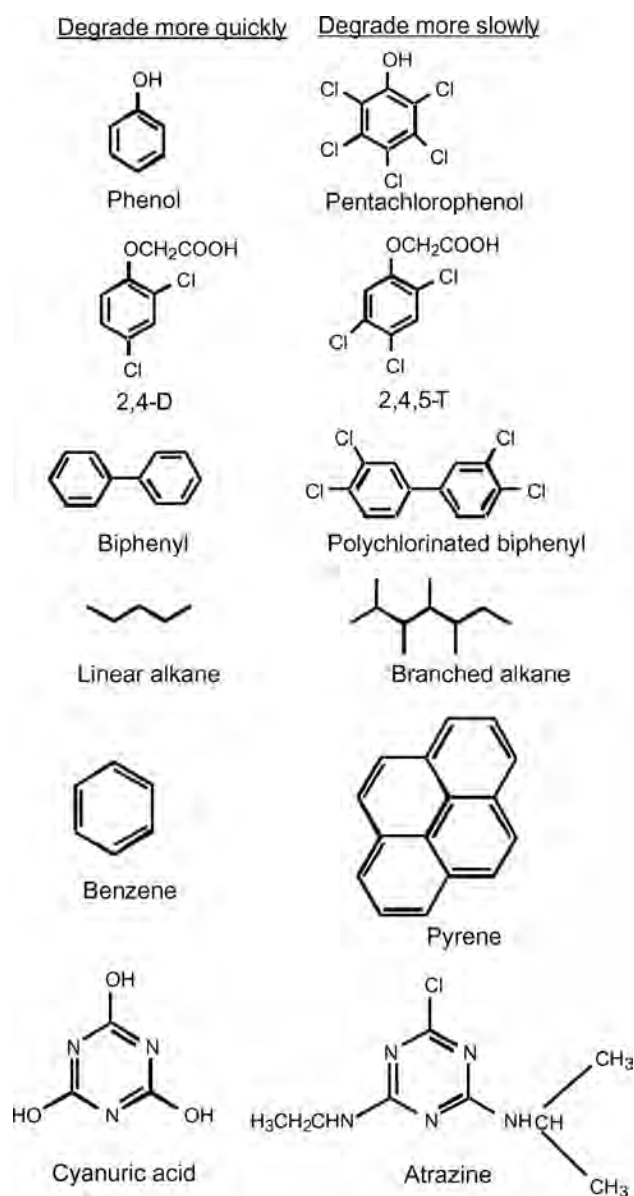


Fig. 1 Examples of persistent organic chemicals and their more degradable structural analogs illustrating the effect of various functional groups on biodegradability.

Hydrolytic Reactions

Microorganisms can degrade organic pollutants using hydrolytic reactions.^[1] Enzymes mediating these reactions are generally referred to as hydrolases and act by replacing substituents on the substrate with water. In these reactions, water acts as a nucleophile attacking centers of positive charge on the substrate molecule. Because oxygen does not participate in the reaction, hydrolytic transformations can occur in aerobic or anaerobic environments. Functional groups commonly occurring in many pesticides that are susceptible to hydrolytic attack include amides, esters, carbamates, C-halogen bonds, and sulfonyl ureas to name a few.

Steric Hindrance

Steric hindrance is another mechanism related to chemical structure that can influence enzymatic reactions. The addition of functional groups alters the structure in such a way that it is no longer recognized by the enzyme. Common examples of steric hindrance include highly branched aliphatic compounds or substituents, polycyclic aromatic hydrocarbons (PAHs), and various polymers. As the molecular weight of the substrate increases by branching, polymerization, or addition of fused aromatic rings, susceptibility to enzymatic attack decreases (Fig. 1).

BIOAVAILABILITY

Sorption and Sequestration

Removal of contaminants from the bulk soil aqueous phase can limit their bioavailability to microorganisms,^[2] thus reducing the degradation rate. The reduction of aqueous phase contaminant concentrations can result from a variety of processes that are primarily a function of a chemical's properties and soil characteristics such as clay and organic matter content. These processes include slow dissolution of the contaminant solid or liquid phase,^[3] slow diffusion and sequestration in soil nanopores,^[4] soil sorption (either adsorption or partitioning),^[5] and partitioning of the contaminant in a non-aqueous phase liquid (NAPL; Fig. 2).^[6] Ionizable functional groups on various organic acids and bases (e.g., 2,4-D, atrazine, and quinoline) result in pH-dependent charge of these molecules that can enhance or reduce interaction with sorption sites on soil surfaces. Hydrophobic interactions of non-polar organic compounds, primarily with soil organic matter, are greater for molecules

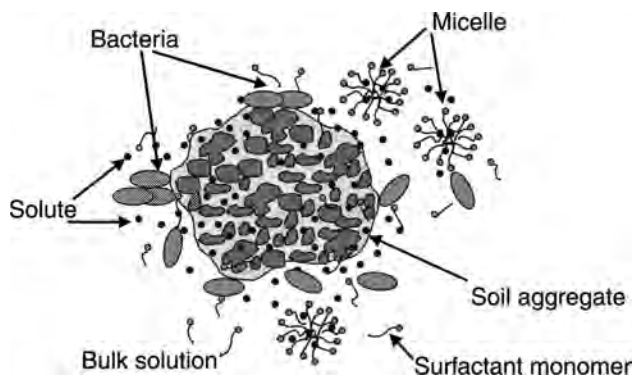


Fig. 2 Diagram illustrating interactions of solute molecules with the soil matrix resulting in a reduction in bioavailability to microorganisms. Rate limited processes depicted in the figure include sorption-desorption via electrostatic or hydrophobic interactions, entrapment and slow diffusion through nano/micropores, and release of solute from the soil matrix by interactions with surfactant micelles.

with low water solubility (aromatic compounds with alkyl or halide substituents).

Data regarding sorption-limited biodegradation in soils and sediments suggest the existence of available and unavailable compartments. However, the concentration of contaminant in these two pools does not strictly correspond to the soil aqueous and solid phases, respectively. Substrate localized in the “unavailable” compartment(s) may be an exploitable resource for appropriately adapted microorganisms. For example, differences have been observed in the bioavailability of sorbed PAHs naphthalene- and phenanthrene-degrading bacteria.^[7] Bacterial attachment at the interface between non-NAPL; a class of liquid organic contaminants such as solvents and fuels) and water is known to influence the uptake of NAPL and NAPL-partitioned organic compounds.^[8] Little is known about the role of attachment in the biodegradation of soil-sorbed pollutants. However, the ability of microorganisms to utilize surface accumulated substrates in dilute media has been recognized since the early 1940s.^[9] Biodegradation of soil-sorbed phenanthrene varied substantially between two PAH-degrading bacteria with differing cell surface properties.^[10] These studies indicate that bioavailability may be species-dependent and suggest that cellular attachment and as yet unidentified adaptations may allow some species greater access to sorbed pollutants.

Enhancing Bioavailability

Increasing the degradation of sorbed, trapped, or insoluble pollutants focused on enhancing the contaminant release from the soil matrix through the addition of cosolvents and surfactants. Enhancement and inhibition of biodegradation in the presence of surfactants have been reported. In general, enhanced degradation is attributed to enhanced solubility,^[3] an increase in the interfacial surface area between immiscible fluids,^[11] or enhanced desorption (Fig. 2).^[12] Inhibition of biodegradation can result from surfactant toxicity, sequestration of the contaminant within surfactant micelles,^[13] or preferential utilization of the surfactant as a substrate.^[11] Understanding how microbes enhance desorption or gain access to sorbed substrates is paramount to reducing the persistence of sorbed organic pollutants.

NUTRIENT AVAILABILITY

Effect of Added Nutrients on Hydrocarbon Degradation

Most soil and subsurface environments are limited in available carbon for microbial growth. However, in hydrocarbon contaminated soils, the ratio of reduced C to other nutrients is very large and as a result one or more nutrients other than C may become rate-limiting factors for contaminant biodegradation. Addition of inorganic nutrients usually

enhances hydrocarbon biodegradation. This approach is frequently considered to stimulate bioremediation of petroleum contaminated environments. Optimal ranges of C/N/P/S ratios vary enormously but a C/N/P ratio of 100:10:1 is generally accepted as optimal for hydrocarbon biodegradation in most environments. Inorganic salts such as ammonium phosphate are commonly used as nutrient fertilizers in hydrocarbon contaminated environments. Inorganic salts are inexpensive and provide nutrients in bioavailable forms, but in some situations these fertilizers can become diluted or washed out of the contaminated zone and therefore do not provide a long-term supply of the limiting nutrient(s). This is particularly a concern in near shore and saturated environments. In other cases, use of these fertilizers can lead to rather dramatic changes in pH or result in osmotic stress that can adversely affect biodegradation rates. These problems can sometimes be overcome by using organic nutrient sources with low C/N and C/P ratios such as urea, oleophilic forms of N and P, or a variety of other slow-release fertilizers.^[14]

Accelerated biodegradation in response to fertilization is frequently reported in hydrocarbon-contaminated soils and aquifer sediments.^[15] Some studies, however, have observed no increase or an inhibition in degradation rates.^[16] These contradictory results suggest that the response of the microbial community to manipulation of various environmental factors to optimize biodegradation is site specific and reflects the complex nature of heterogeneous systems such as soils.^[17]

Organic Chemicals That Are Macronutrient Sources

Ammonia is almost invariably the preferred N source for most bacteria. However, in natural environments ammonia is not always available, and bacteria have evolved the ability to use a wide array of organic and inorganic compounds as N sources. From a pragmatic viewpoint, coordinated regulation of degradative pathways and N-assimilatory pathways makes sense if the organic chemical in question is a poor carbon and energy source but potentially a good N source.

Biodegradation of a N-Containing Herbicide

Unlike homocyclic aromatic compounds such as 2,4-D, atrazine (Fig. 1) can serve as a C and N source for some microorganisms. Hence the level of induction and expression of the atrazine-degrading genotype in soil microorganisms is likely to be affected by the C and N status in soils. Atrazine has a low C/N ratio (8:5) and its complete mineralization results in release of excess N. Therefore, atrazine is potentially a good N source and its degradation is likely to be favored in N-limited soils. Addition of inorganic fertilizers to soils has been shown to inhibit atrazine mineralization.^[18,19] The general suppression of atrazine

mineralization in the presence of exogenous nitrogen suggests that regulation and expression of atrazine degradation genes in soil microbial communities are linked to the N status of soil. Atrazine and its N-heterocyclic derivatives are apparently substrates for a complex pathway that may be catabolic as well as linked to assimilatory pathways of C and N. A study with ^{15}N -ring-atrazine has definitively shown that atrazine-N is incorporated into the biomass of several bacterial cultures known to mineralize atrazine.^[19] Evidence for a regulatory linkage between atrazine catabolism and N assimilation has also been observed in an atrazine-mineralizing soil bacteria.^[19,20] Given the apparent negative correlation between atrazine degradation rates and soil N levels, perhaps the majority of soil bacteria capable of utilizing triazine-N can repress expression of the degradative pathway enzymes in favor of more readily available sources of N.

MICROBIAL ECOLOGY AND BIODEGRADATION

Soil is perhaps the most biologically diverse and abundant ecosystem in the biosphere. A few grams of topsoil may contain 5000–15,000 different microbial species. Only 0.1–1% of this biodiversity has been cultured and studied in the laboratory. Understanding of pollutant biodegradation is largely based on the study of microorganisms obtained via classical enrichment techniques. However, since these organisms were isolated under very selective conditions and represent less than 0.1% of the biodiversity of soil, they may not be representative of the full genetic biodegradation potential of soil communities and may play only a minor role in the detoxification of contaminated soil. To more fully understand environmental processes mediated by microorganisms, various culture-independent methodologies, based on the analysis of soil genomic DNA, are becoming widely used. Many of these approaches are based on the detection of 16S rRNA genes (16S rDNA) from community members that do not grow well in the laboratory.

One such technique amenable to ecological studies is denaturing gradient gel electrophoresis (DGGE). This DNA fingerprinting technique allows separation of DNA fragments of similar size but different sequence composition.^[21] When applied to microbial community analysis, conserved 16S rDNA sequences are used as primers to amplify community DNA. The resulting mixture of PCR products is then separated by DGGE. The result is a pattern of bands that roughly corresponds to the number of predominant members in a microbial community. Statistical analysis of banding patterns can then be used to monitor changes in the structure of microbial communities or to compare communities in different environments. The selection of the PCR primers used to amplify community DNA determines the relative breadth or narrowness of community members targeted for analysis. In some instances,

DNA fragments isolated from DGGE bands can be sequenced to identify phylogenetically the organisms represented by that particular band.

This approach has been used to characterize microbial communities in soils with and without prior atrazine exposure histories (H and NH soil, respectively) and different C/N ratios. The DNA fingerprints were then related to atrazine mineralization capacity in the acclimated soils (soils exposed to three atrazine amendments in a lab acclimation study; Fig. 3). The banding patterns revealed highly complex microbial communities. Although several shared bands were observed, the DNA fingerprints of the H and NH soils showed two distinctly different communities (lanes E and M). The bands (representing distinct phylotypes) that intensified in response to atrazine amendment in the H soil were different from those that intensified in the NH soil. For example, band 1 in the H soil was not present in the NH soil and bands 2 and 3 were present in the NH soil amended with atrazine but absent in the H soil. In soils amended with atrazine + NH_4NO_3 (low C/N), several bands in the H soil became more intense (bands 4–6, lane K). These bands were absent in the corresponding NH soil (lane C). Interestingly, band 4 was even more

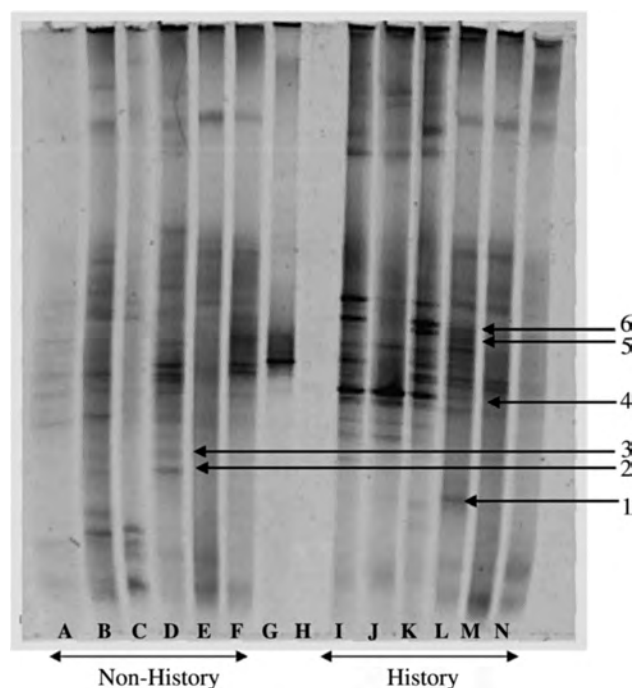


Fig. 3 PCR amplified 16S rDNA fingerprint of microbial communities obtained using DGGE analysis of DNA extracted from amended NH (A–F) and H soils (I–N). Soil amendments include: non-amended field soil (F and N), water control (E and M), atrazine (D and L), atrazine + NH_4NO_3 (C and K), atrazine + acetate (B and J), and atrazine + acetate + NH_4NO_3 (A and I). A pure bacterial strain, isolate P5-2 was used as a positive control (G) and a water control (H) was also included.

intense in the atrazine + acetate (high C/N) treated soil while bands 5 and 6 were absent. In general, the communities in the H and NH soils responded quite differently to the various amendments (i.e., variations in C/N ratio). Presumably, the organisms represented by these bands in the DGGE gel play a very significant role in atrazine degradation in this soil.

REFERENCES

- Hickey, W.J. Biochemistry and metabolism of xenobiotic chemicals. In *Principles and Applications of Soil Microbiology*; Sylvia, D.M., Fuhrmann, J.J., Hartel, P.G., Zuberer, D.A., Eds.; Prentice Hall: Upper Saddle River, 1998; 447–468.
- Ogram, A.V.; Jessup, R.E.; Ou, L.T.; Rao, P.S.C. Effects of sorption on biological degradation rates of (2,4-dichlorophenoxy) acetic acid in soils. *Appl. Environ. Microbiol.* **1985**, *49*, 582–587.
- Tiehm, A. Degradation of polycyclic aromatic hydrocarbons in the presence of synthetic surfactants. *Appl. Environ. Microbiol.* **1994**, *60*, 258–263.
- Nam, K.; Alexander, M. Role of nanoporosity and hydrophobicity in sequestration and bioavailability: Tests with model solids. *Environ. Sci. Technol.* **1997**, *32*, 71–74.
- Smith, S.C.; Ainsworth, C.C.; Traina, S.J.; Hicks, R.J. The effects of sorption on the transformation and degradation of quinoline. *Soil Sci. Soc. Am. J.* **1992**, *56*, 737–746.
- Efroymsen, R.A.; Alexander, M. Reduced mineralization of low concentrations of phenanthrene because of sequestering in nonaqueous-phase liquids. *Environ. Sci. Technol.* **1995**, *29*, 515–521.
- Guerin, W.F.; Boyd, S.A. Differential bioavailability of soil-sorbed naphthalene to two bacterial species. *Appl. Environ. Microbiol.* **1992**, *58*, 1142–1152.
- Stelmack, P.L.; Gray, M.R.; Pickard, M.A. Bacterial adhesion to soil contaminants in the presence of surfactants. *Appl. Environ. Microbiol.* **1999**, *65*, 163–168.
- Zobell, C.E. The effect of solid surfaces upon bacterial activity. *J. Bact.* **1943**, *46*, 39–56.
- Dean, S.M.; Jin, Y.; Cha, D.K.; Wilson, S.V.; Radosevich, M. Phenanthrene degradation in soils co-inoculated with phenanthrene-degrading and biosurfactant-producing bacteria. *J. Environ. Qual.* **2001**, *30* (4), 1126–1133.
- Rouse, J.D.; Sabatini, D.A.; Suflita, J.M.; Harwell, J.H. Influence of surfactants on microbial degradation of organic compounds. *Crit. Rev. Environ. Sci. Technol.* **1994**, *24*, 325–370.
- Aronstein, B.N.; Alexander, M. Surfactants at low concentrations stimulate biodegradation of sorbed hydrocarbons in samples of aquifer sands and soil slurries. *Environ. Toxicol. Chem.* **1992**, *11*, 1227–1233.
- Guha, S.; Jaffe, P.R. Biodegradation kinetics of phenanthrene partitioned into the micellar phase of nonionic surfactants. *Environ. Sci. Technol.* **1996**, *30*, 605–611.
- Churchill, S.A.; Griffin, R.A.; Jones, L.P.; Churchill, P.F. Biodegradation rate enhancement of hydrocarbons by an oleophilic fertilizer and a rhamnolipid biosurfactant. *J. Environ. Qual.* **1995**, *24*, 19–28.
- Jackson, A.; Pardue, J.H. Seasonal variability of crude oil respiration potential in salt and fresh marshes. *J. Environ. Qual.* **1997**, *26*, 1140–1146.
- Swindoll, C.M.; Aelion, C.M.; Pfaender, F.K. Influence on inorganic and organic nutrients on aerobic biodegradation and on adaptation response of subsurface microbial communities. *Appl. Environ. Microbiol.* **1988**, *54*, 212–217.
- Leahy, J.G.; Colwell, R.R. Microbial degradation of hydrocarbons in the environment. *Microbiol. Rev.* **1990**, *54*, 305–315.
- Alvey, S.; Crowley, D.E. Influence of organic amendments on biodegradation of atrazine as a nitrogen source. *J. Environ. Qual.* **1995**, *24*, 1156–1162.
- Bichat, F.; Sims, G.K.; Mulvaney, R.L. Microbial utilization of heterocyclic nitrogen from atrazine. *Soil Sci. Soc. Am. J.* **1999**, *63* (1), 100–110.
- Gebendinger, N.; Radosevich, M. Inhibition of atrazine degradation by cyanazine and exogenous nitrogen in bacterial isolate M91-3. *Appl. Microbiol. Biotechnol.* **1999**, *51*, 375–381.
- Muyzer, G.; Smalla, K. Application of denaturing gradient gel electrophoresis (DGGE) and temperature gradient gel electrophoresis (TGGE) in microbial ecology. *Antonie van Leeuwenhoek* **1998**, *73*, 127–141.

Pollution: Point Source

Ravendra Naidu

Mallavarapu Megharaj

Peter Dillon

Rai Kookana

Ray Correll

*Commonwealth Scientific and Industrial Research Organisation (CSIRO),
Adelaide, South Australia, Australia*

Walter W. Wenzel

*Institute of Soil Research, University of Natural Resources and Applied Life
Sciences, Vienna, Austria*

Abstract

Environmental pollution is one of the foremost ecological challenges. Pollution is an offshoot of technological advancement and overexploitation of natural resources. From the standpoint of pollution, the term environment primarily includes air, land, and water components including landscapes, rivers, parks, and oceans. Pollution can be generally defined as an undesirable change in the natural quality of the environment that may adversely affect the well-being of humans, other living organisms, or entire ecosystems either directly or indirectly. Although pollution is often the result of human activities (anthropogenic), it could also be due to natural sources such as volcanic eruptions emitting noxious gases, pedogenic processes, or natural change in the climate. Where pollution is localized, it is described as point source (PS). Thus, PS pollution is a source of pollution with a clearly identifiable point of discharge that can be traced back to the specific source such as leakage of underground petroleum storage tanks or an industrial site.

INTRODUCTION

Some naturally occurring pollutants are termed geogenic contaminants, and these include fluorine (F), selenium, arsenic (As), lead, chromium, fluoride, and radionuclides in the soil and water environment. Significant adverse impacts of geogenic contaminants (e.g., As) on environmental and human health have been recorded in Bangladesh, West Bengal, India, Vietnam, and China. The presence of geogenic cadmium and the implications to crop quality in Norwegian soils are also studied.^[1]

The terms contamination and pollution are often used interchangeably but erroneously. Contamination denotes the presence of a particular substance at a higher concentration than would occur naturally, and this may or may not have harmful effects on human or the environment. Pollution not only refers to the presence of a substance at higher level than would normally occur but also is associated with some kind of adverse effect.

NATURE AND SOURCES OF CONTAMINANTS

The main activities contributing to point source (PS) pollution include industrial, mining, agricultural, and

commercial activities as well as transport and services (Table 1). Uncontrolled mining, manufacturing, and disposal of wastes inevitably cause environmental pollution. Military land and land used for recreational shooting are also important sites of PS contamination. The contaminants associated with such activities are listed in Table 1. Contamination at many of these sites appears to have resulted because of lax regulatory measures prior to the establishment of legislation protecting the environment.

CONTAMINANT INTERACTIONS IN SOIL AND WATER

Inorganic Chemicals

Inorganic contaminant interactions with colloid particulates include the following: adsorption–desorption at surface sites, precipitation, ion exchange with clay minerals, binding by organically coated particulate matter or organic colloidal material, or adsorption of contaminant ligand complexes. Depending on the nature of contaminants, these interactions are controlled by solution pH and ionic strength of soil solution, nature of the species, dominant

Table 1 Industries, land uses, and associated chemicals contributing to PS and non-PS pollution.

Industry	Type of chemical	Associated chemicals
Airports	Hydrocarbons	Aviation fuels
	Metals	Particularly aluminum, magnesium, chromium
Asbestos production and disposal	Asbestos	
Battery manufacture and recycling	Metals	Lead, manganese, zinc, cadmium, nickel, cobalt, mercury, silver, antimony
	Acids	Sulfuric acid
Breweries/distilleries	Alcohol	Ethanol, methanol, esters
Chemicals manufacture and use	Acid/alkali	Mercury (chloralkali), sulfuric, hydrochloric and nitric acids, sodium and calcium hydroxides
	Adhesives/resins	Polyvinyl acetate, phenols, formaldehyde, acrylates, phthalates
	Dyes	Chromium, titanium, cobalt, sulfur and nitrogen organic compounds, sulfates, solvents
	Explosives	Acetone, nitric acid, ammonium nitrate, pentachlorophenol, ammonia, sulfuric acid, nitroglycerine, calcium cyanamide, lead, ethylene glycol, methanol, copper, aluminum, bis(2-ethylhexyl) adipate, dibutyl phthalate, sodium hydroxide, mercury, silver
	Fertilizer	Calcium phosphate, calcium sulfate, nitrates, ammonium sulfate, carbonates, potassium, copper, magnesium, molybdenum, boron, cadmium
	Flocculants	Aluminum
	Foam production	Urethane, formaldehyde, styrene
	Fungicides	Carbamates, copper sulfate, copper chloride, sulfur, chromium
	Herbicides	Ammonium thiocyanate, carbonates, organochlorines, organophosphates, As, mercury
	Paints	
	Heavy metals	As, barium, cadmium, chromium, cobalt, lead, manganese, mercury, selenium, zinc

(Continued)

Table 1 Industries, land uses, and associated chemicals contributing to PS and non-PS pollution. (Continued)

Industry	Type of chemical	Associated chemicals
	General	Titanium dioxide
	Solvent	Toluene, oils natural (e.g., pine oil) or synthetic
	Pesticides	As, lead, organochlorines, organophosphates
	Active ingredients	Sodium, tetraborate, carbamates, sulfur, synthetic pyrethroids
	Solvents	Xylene, kerosene, methyl isobutyl ketone, amyl acetate, chlorinated solvents
	Pharmacy	Dextrose, starch
	General/solvents	Acetone, cyclohexane, methylene chloride, ethyl acetate, butyl acetate, methanol, ethanol, isopropanol, butanol, pyridine methyl ethyl ketone, methyl isobutyl ketone, tetrahydrofuran
	Photography	Hydroquinone, pheidom, sodium carbonate, sodium sulfite, potassium bromide, monomethyl para-aminophenol sulfates, ferricyanide, chromium, silver, thiocyanate, ammonium compounds, sulfur compounds, phosphate, phenylene diamine, ethyl alcohol, thiosulfates, formaldehyde
	Plastics	Sulfates, carbonates, cadmium, solvents, acrylates, phthalates, styrene
	Rubber	Carbon black
	Soap/detergent	
	General	Potassium compounds, phosphates, ammonia, alcohols, esters, sodium hydroxide, surfactants (sodium lauryl sulfate), silicate compounds
	Acids	Sulfuric acid and stearic acid
	Oils	Palm, coconut, pine, tea tree
	Solvents	
	General	Ammonia
	Hydrocarbons	e.g., BTEX (benzene, toluene, ethylbenzene, xylene)
	Chlorinated organics	e.g., trichloroethane, carbon tetrachloride, methylene chloride

(Continued)

Table 1 Industries, land uses, and associated chemicals contributing to PS and non-PS pollution. *(Continued)*

Industry	Type of chemical	Associated chemicals
Metal treatments	Electroplating metals	Nickel, chromium, zinc, aluminum, copper, lead, cadmium, tin
	Acids	Sulfuric, hydrochloric, nitric, phosphoric
	General	Sodium hydroxide, 1,1,1-trichloroethane, tetrachloroethylene, toluene, ethylene glycol, cyanide compounds
	Liquid carburizing baths	Sodium, cyanide, barium, chloride, potassium chloride, sodium chloride, sodium carbonate, sodium cyanate
Mining and extractive industries		As, mercury and cyanides and also refer to Explosives under chemicals, manufacture and use
Power stations		Asbestos, PCBs, fly ash, metals
Printing shops		Acids, alkalis, solvents, chromium (see Photography under chemicals, manufacture and use)
Scrap yards		Hydrocarbons, metals, solvents
Service stations and fuel storage facilities		Aliphatic hydrocarbons
BTEX (i.e., benzene, toluene, ethylbenzene, xylene)		
PAHs (e.g., benzo(a) pyrene)		
Phenols		
Lead		
Sheep and cattle dips		As, organochlorines and organophosphates, carbamates, and synthetic pyrethroids
Smelting and refining		Metals and the fluorides, chlorides and oxides of copper, tin, silver, gold, selenium, lead, aluminum
Tanning and associated trades	Metals	Chromium, manganese, aluminum
	General	Ammonium sulfate, ammonia, ammonium nitrate, phenolics (creosote), formaldehyde, tannic acid

*(Continued)***Table 1** Industries, land uses, and associated chemicals contributing to PS and non-PS pollution. *(Continued)*

Industry	Type of chemical	Associated chemicals
Defense works		See explosives under chemicals manufacture and use, foundries, engine works, service stations
Drum reconditioning		See chemicals manufacture and use
Dry cleaning		Trichlorethylene and ethane Carbon tetrachloride Perchloroethylene
Electrical		PCBs (transformers and capacitors), solvents, tin, lead, copper
Engine works	Hydrocarbons	
	Metals	
	Solvents	
	Acids/alkalis	
	Refrigerants	
	Antifreeze	Ethylene glycol, nitrates, phosphates, silicates
Foundries	Metals	Particularly aluminum, manganese, iron, copper, nickel, chromium, zinc, cadmium and lead and oxides, chlorides, fluorides and sulfates of these metals
		Phenolics and amines
		Coke/graphite dust
Gas works	Inorganics	Ammonia, cyanide, nitrate, sulfide, thiocyanate
	Metals	Aluminum, antimony, As, barium, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, silver, vanadium, zinc
	Semivolatiles	Benzene, ethylbenzene, toluene, total xylenes, coal tar, phenolics and PAHs
Iron and steel works		Metals and oxides of iron, nickel, copper, chromium, magnesium and manganese, and graphite
Landfill sites		Methane, hydrogen sulfides, heavy metals, complex acids
Marinas		Engine works, electroplating under metal treatment
	Antifouling paints	Copper, tributyltin (TBT)

(Continued)

Table 1 Industries, land uses, and associated chemicals contributing to PS and non-PS pollution. (*Continued*)

Industry	Type of chemical	Associated chemicals
Wood preservation	Metals	Chromium, copper, As
	General	Naphthalene, ammonia, pentachlorophenol, dibenzofuran, anthracene, biphenyl, ammonium sulfate, quinoline, boron, creosote, organochlorine pesticides

Source: Adapted from Barzi, Naidu, et al.^[2]

cation, and inorganic and organic ligands present in the soil solution.^[3]

Organic Chemicals

The fate and behavior of organic compounds depend on a variety of processes including sorption–desorption, volatilization, chemical and biological degradation, plant uptake, surface runoff, and leaching. Sorption–desorption and degradation (both biotic and abiotic) are perhaps the two most important processes as the bulk of the chemicals is either sorbed by organic and inorganic soil constituents, or chemically or microbially transformed/degraded. The degradation is not always a detoxification process. This is because in some cases the transformation or degradation process leads to intermediate products that are more mobile, more persistent, or more toxic to non-target organisms. The relative importance of these processes is determined by the chemical nature of the compound.

IMPLICATIONS TO SOIL AND ENVIRONMENTAL QUALITY

Considerable amount of literature is available on the effects of contaminants on soil microorganisms and their functions in soil. The negative impacts of contaminants on microbial processes are important from the ecosystem point of view, and any such effects could potentially result in a major ecological perturbation. Hence, it is most relevant to examine the effects of contaminants on microbial processes in combination with communities. The most commonly used indicators of metal effects on microflora in soil are as follows: 1) soil respiration; 2) soil nitrification; 3) soil microbial biomass; and 4) soil enzymes.

Contaminants can reach the food chain by way of water, soil, plants, and animals. In addition to the food chain transfer, pollutants may also enter via direct consumption or dust inhalation of soil by children or animals. The

accumulation of these pollutants can take place in certain target tissues of the organism depending on the solubility and nature of the compound. For example, dichlorodiphenyltrichloroethane and polychlorinated biphenyls (PCBs) accumulate in human adipose tissue. Consequently, several of these pollutants have the potential to cause serious abnormalities including cancer and reproductive impairments in animal and human systems.

SAMPLING FOR PS POLLUTION

The aims of the sampling system must be clearly defined before it can be optimized.^[4] The type of decision may be to determine land use, how much of an area is to be remediated, or what type of remediation process is required. Because sampling and the associated chemical and statistical analyses are expensive, careful planning of the sampling scheme is therefore a good investment. One of the best ways to achieve this is to use the ancillary data that are available. These data could be in the form of emission history from a stack, old photographs that give details of previous land uses, or agricultural records. Such data can at least give qualitative information.

As discussed before, PS pollution will typically be airborne from a stack or waterborne from some effluent such as tannery waste, cattle dips, or mine waste. In many cases, the industry will have modified its emissions (e.g., cleaner production) or point of release (increased stack height); hence, the existing pattern of emission may not be closely related to the historic pattern of pollution. For example, liquid effluent may previously have been discharged into a bay, but that effluent may now be treated and perhaps discharged at some other point. Typically, the aim of a sampling scheme in these situations is to assess the maximum concentrations, the extent of the pollution, and the rate of decline in concentration from the PS. Often, the sampling scheme will be used to produce maps of concentration isopleths of the pollutant.

The location of the sampling points would normally be concentrated toward the source of the pollution. A good scheme is to have sufficient samples to accurately assess the maximum pollution and then space additional samples at increasing intervals. In most cases, the distribution of the pollutant will be asymmetric, with the maximum spread down the slope or down the prevailing wind. In such cases, more samples should be placed in the direction of the expected gradient. This is a clear case of when ancillary data can be used effectively. A graph of concentration of the pollutant against the reciprocal of distance from the source is often informative.^[5] Sampling depths will depend on both the nature of the pollution and the reason for the investigation. If the pollution is from dust and it is unlikely to be leached, only surface sampling will be required. An example of this is pollution from silver smelting in

Wales.^[6] In contrast, contamination from organic or mobile inorganic pollutants such as F compounds may migrate well down to the profile, and deep sampling may be required.^[7,8]

ASSESSMENT

In order to assess the impacts of pollution, reliable and effective monitoring techniques are important. Pollution can be assessed and monitored by chemical analyses, toxicity tests, and field surveys. Comparison of contaminant data with an uncontaminated reference site and available databases for baseline concentrations can be useful in establishing the extent of contamination. However, this may not be always possible in the field. Chemical analyses must be used in conjunction with biological assays to reveal site contamination and associated adverse effects. Toxicological assays can also reveal information about synergistic interactions of two or more contaminants present as mixtures in soil, which cannot be measured by chemical assays alone.

Microorganisms serve as rapid detectors of environmental pollution and are thus of importance as pollution indicators. The presence of pollutants can induce the alteration of microbial communities, reduction of species diversity and inhibition of certain microbial processes (organic matter breakdown, mineralization of carbon and nitrogen, enzymatic activities, etc.). A measure of the functional diversity of the bacterial flora can be assessed using ecoplates (see http://www.biolog.com/section_4.html). It has been shown that algae are especially sensitive to various organic and inorganic pollutants and thus may serve as a good indicator of pollution.^[9] A variety of toxicity tests involving microorganisms, invertebrates, vertebrates, and plants may be used with soil or water samples.^[10]

MANAGEMENT AND/OR REMEDIATION OF PS POLLUTION

The major objectives of any remediation process are to 1) reduce the actual or potential environmental threat and 2) reduce unacceptable risks to man, animals, and the environment to acceptable levels.^[11] Therefore, strategies to either manage or remediate contaminated sites have developed largely from the application of stringent regulatory measures set up to safeguard ecosystem function as well as to minimize the potential adverse effects of toxic substances on animal and human health.

The available remediation technologies may be grouped into two categories: 1) *ex situ* techniques that require removal of the contaminated soil or groundwater for treatment either on-site or off-site and 2) *in situ* techniques that attempt to remediate without the

excavation of contaminated soils. Generally, *in situ* techniques are favored over *ex situ* techniques because of 1) reduced costs due to the elimination or minimization of excavation, transportation to disposal sites, and sometimes treatment itself; 2) reduced health impacts on the public or the workers; and 3) the potential for the remediation of inaccessible sites, e.g., those located at greater depths or under buildings. Although *in situ* techniques have been successful with organic contaminated sites, the success of *in situ* strategies with metal contaminants has been limited. Given that organic and inorganic contaminants often occur as a mixture, a combination of more than one strategy is often required to either successfully remediate or manage metal contaminated soils.

GLOBAL CHALLENGES AND RESPONSIBILITY

The last 100 years have seen massive industrialization. Indeed, such developments were coupled with the rapid increase in world population and the desire to enhance economy and food productivity. While industrialization has led to increased economic activity and much benefit to human race, the lack of regulatory measures and appropriate waste management strategies until early 1980s (including the use of agrochemicals) has resulted in the contamination of our biosphere. Continued pollution of the environment through industrial emissions is of global concern. There is, therefore, a need for politicians, regulatory organizations, and scientists to work together to minimize environmental contamination and to remediate contaminated sites. The responsibility to check this pollution lies with every individual and country although the majority of this pollution is due to the industrialized nations. There is a clear need of better coordination of efforts in dealing with numerous forms of PS pollution problems that are being faced globally.

REFERENCES

1. Mehlum, H.K.; Arnesen, A.K.M.; Singh, B.R. Extractability and plant uptake of heavy metals in alum shale soils. *Commun. Soil Sci. Plan.* **1998**, *29*, 183–198.
2. Barzi, F.; Naidu, R.; McLaughlin, M.J. Contaminants and the Australian Soil Environment. In *Contaminants and the Soil Environment in the Australasia-Pacific Region*; Naidu, R., Kookana, R.S., Oliver, D., Rogers, S., McLaughlin, M.J., Eds.; Kluwer Academic Publishers: Dordrecht, 1996; 451–484.
3. McBride, M.B. Reactions controlling heavy metal solubility in soils. *Adv. Soil Sci.* **1989**, *10*, 1–56.
4. Patil, G.P.; Gore, S.D.; Johnson, G.D. *EPA Observational Economy Series Volume 3: Manual on Statistical Design and Analysis with Composite Samples*. In Technical Report No. 96-0501; Center for Statistical

- Ecology and Environmental Statistics, Pennsylvania State University: Pennsylvania, 1996.
5. Ward, T.J.; Correll, R.L. Estimating background concentrations of heavy metals in the marine environment. In *Proceedings of a Bioaccumulation Workshop: Assessment of the Distribution, Impacts and Bioaccumulation of Contaminants in Aquatic Environments*, Sydney, 1990; Miskiewicz, A.G., Ed.; Water Board and Australian Marine Science Association: Sydney, 1992; 133–139.
 6. Jones, K.C.; Davies, B.E.; Peterson, P.J. Silver in welsh soils: Physical and chemical distribution studies. *Geoderma* **1986**, *37*, 157–174.
 7. Barber, C.; Bates, L.; Barron, R.; Allison, H. Assessment of the relative vulnerability of groundwater to pollution: A review and background paper for the conference workshop on vulnerability assessment. *J. Aust. Geol. Geophys.* **1993**, *14* (2–3), 147–154.
 8. Wenzel, W.W.; Blum, W.E.H. Effects of fluorine deposition on the chemistry of acid Luvisols. *Int. J. Environ. Anal. Chem.* **1992**, *46*, 223–231.
 9. Megharaj, M.; Singleton, I.; McClure, N.C. Effect of pentachlorophenol pollution towards microalgae and microbial activities in soil from a former timber processing facility. *B. Environ. Contam. Tox.* **1998**, *61*, 108–115.
 10. Juhasz, A.L.; Megharaj, M.; Naidu, R. Bioavailability: The major challenge (constraint) to bioremediation of organically contaminated soils. In *Remediation Engineering of Contaminated Soils*; Wise, D., Trantolo, D.J., Cichon, E.J., Inyang, H.I., Stottmeister, U., Eds.; Marcel Dekker: New York, 2000; 217–241.
 11. Wood, P.A. Remediation methods for contaminated sites. In *Contaminated Land and Its Reclamation*; Hester, R.E., Harrison, R.M., Eds.; Royal Society of Chemistry, Thomas Graham House: Cambridge, 1997; 47–73.

Encyclopedia of Soil Science

Third Edition

Volume III

Porosity through World
Pages 1782–2653

Porosity –
Rapid

Rare Earth –
Seepage

Selenium –
Soil Enzymes

Soil Organic Matter –
Structure

Sub-Irrigation –
Testing

Texture –
Value

Variability –
Water-Repellent

Watershed –
World

Porosity: Pore Size Distribution

Keith C. Cameron

Soil and Physical Science Group, Lincoln University, Canterbury, New Zealand

Graeme D. Buchan

Centre for Soil and Environmental Quality, Lincoln University, Canterbury, New Zealand

Abstract

The relative arrangement of the soil units (primary particles and aggregates) defines the pore space of the soil. Pore space is essential as it provides the capacity to store water and air as well as enables drainage and root growth. For some soil processes, the total porosity is less important than the pore size distribution; e.g., a sandy soil has a lower total porosity than a clayey soil, but usually drains faster. The ability of a pore to transmit water decreases dramatically with its size. The sandy soil has a higher proportion of large soil pores than the clay and thus drains much faster. Small pores are, however, essential as they provide the ability to store water.

THE SOIL PORE SYSTEM

Structured or aggregated soils are effective “dual porosity” systems. Their porosity can be separated approximately into two components.

1. “Intra-aggregate porosity” is the microscopic pore space created by the geometrical packing of individual soil particles.
2. “Interaggregate porosity” represents the pore space caused by the arrangement of soil aggregates. It is created by biological activity, such as earthworm burrowing and root growth, shrinkage during soil drying, and cultivation of plants.

Definitions

Pore size

Soil particles are normally specified by a single size, the diameter of an equivalent idealized sphere. By contrast, we assume that the soil pore space can be partitioned into a system of idealized cylindrical tubes, each with an equivalent cylinder radius, r . This enables us to link the size of a pore with its ability to attract water by capillary action, as measured by the suction head h (cm) required to empty that pore:

$$h = (2\gamma/r)\cos \alpha \quad (1)$$

where γ is the surface tension, r the pore radius (cm), and α the contact angle.

Fig. 1 shows how soil pore sizes can be classified using either soil micromorphology or the hydraulic properties of soil.

Total porosity, ϵ

This is the ratio of the volume of pores to the total volume of soil

$$\epsilon = \frac{\text{volume of pores}}{\text{total volume of soil}} = \frac{V_p}{V_t} \quad (2)$$

Most soils have their porosity between 0.3 and 0.6, usually expressed as a percentage (i.e., 30% and 60%). Organic soils (e.g., peats) may have porosity up to 90%.

Air-filled porosity, ϵ_a

This is the ratio of the volume of air to the total volume of soil:

$$\epsilon_a = \frac{\text{volume of air}}{\text{total volume of soil}} = \frac{V_a}{V_t} \quad (3)$$

For field soils, an approximate minimum value of ϵ_a required to maintain adequate aeration (especially oxygen supply to respiring roots and microbes) is about 10%, although a range of values from 10% to 25% may occur.^[1]

If ϵ_a is less than 10% at “field capacity,” then soil loosening or drainage is recommended. A desirable ϵ_a for horticultural potting mixes is in the range of 20–25%, which provides rapid drainage and low mechanical impedance for root growth, while the organic matter provides adequate water-holding capacity.^[2]

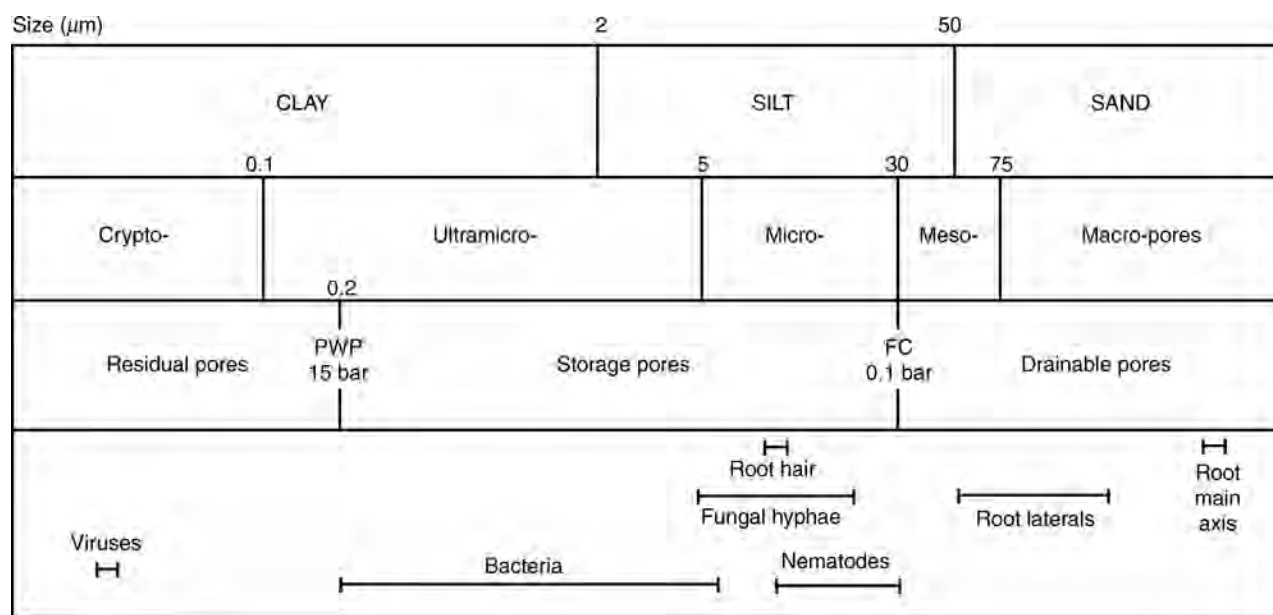


Fig. 1 Sizes of particles, pores, and biota in soil. (Top to bottom) particle sizes (μm) in the U.S. Department of Agriculture classification; pore sizes defined morphologically by Brewer; pore sizes defined hydraulically, with field capacity (FC, 0.1 bar) defining the lower limit of “drainable porosity,” and permanent wilting point (PWP, 15 bar) defining the limit of “storage pores;” typical sizes (diameters) of roots and microbiota in soil. Root sizes are for cereal crops.

Source: From Brewer^[3] and Gregory.^[4]

Macroporosity

Unfortunately there is no single, clear, or agreed criterion for the definition of “macropores” in soil. The minimum equivalent diameter, d , for macropores reported in the literature ranges between 30 and 3000 μm . Here, we adopt the lower limit and define macroporosity as the fraction of soil volume occupied by pores with $d > 30 \mu\text{m}$. This definition of macroporosity represents the air-filled porosity at field capacity, so that macropores are the “drainable pores” (Fig. 1).

Methods of Measurement

“Total porosity” can be measured by various techniques. The simplest method involves the collection and weighing of undisturbed soil cores of known volume (V_t). The ratio of the mass of dry soil to V_t gives the soil “bulk density,” ρ_b . ϵ is then calculated as follows:

$$\epsilon = 1 - \rho_b / \rho_p \quad (4)$$

where ρ_p is the particle density. For most common soil minerals, $\rho_p = 2.65 \text{ g/cm}^3$, but ρ_p decreases as organic matter increases; e.g., a soil with $\rho_b = 1.3 \text{ g/cm}^3$ and $\rho_p = 2.65 \text{ g/cm}^3$ would have a porosity of 51%.

Porosity can also be measured using a gas pycnometer.^[5] The original method, developed by Torstensson and Eriksson,^[6] is based on Boyle’s law of volume–pressure relationships. An oven-dried soil core is sealed into a gas-tight chamber of the pycnometer, and the air in the chamber

is compressed by a known amount. The porosity is calculated from the difference between the volume of compressed air in the chamber when it contains a soil sample and when it is empty.

Macroporosity depends on the definition chosen for the lower limit of macropore diameter. It can be measured from the soil moisture characteristic (SMC) that specifies where macroporosity may also be measured optically, using image analysis of thin sections of soil. Bullock and Thomasson^[7] in their pioneering work developed techniques for scanning thin sections of soil with a television camera and setting an image analyzer to detect differences in gray-scale indicating solids or pores. The volume and geometrical arrangement of the soil pores can be determined.

PORE SIZE DISTRIBUTION

Pores can be classified according to size or function, as shown in Fig. 1.

The macropores ($d > 30 \mu\text{m}$) are usually air-filled and contain water only when the soil is saturated or partially drained. The smaller pores (micropores, $d < 30 \mu\text{m}$) include the storage pores retaining plant-available water and are generally found within, rather than between the soil aggregates.

Methods of Measurement

Lawrence^[8] reviews techniques for measuring pore sizes in soil. The commonest method is to determine the SMC.

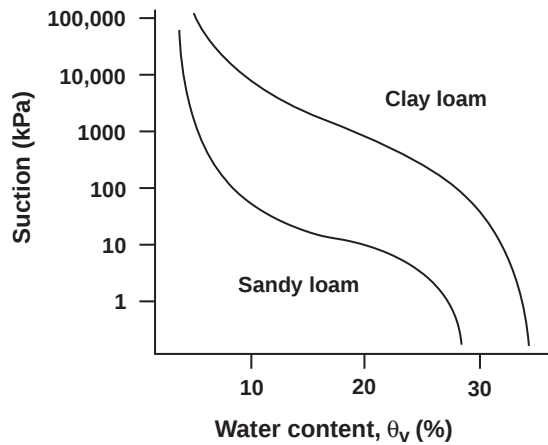


Fig. 2 Comparison of a SMC of a sandy vs. a clay soil.
Source: Adapted from McLaren & Cameron.^[9]

The SMC is the relationship between soil water content θ and the soil matric potential ψ_m (Fig. 2). As ψ_m is negative in unsaturated soil, we drop the negative sign and refer instead to the suction h (refers to $-\psi_m$).

The amount of water retained by the soil over a range of matric potentials can be measured by applying a suction to the soil to overcome the forces of capillarity and adhesion holding the water, using the apparatus shown in Fig. 3. An undisturbed core taken from the field is saturated with water and placed on the porous plate of the “tension table.” When the column of water is level with the soil, all the pores are water-filled. As the column is lowered, increasing suction removes water, first from the larger soil pores, followed by progressively smaller pores. The loss of water at each suction is measured by reweighing the soil core once it has ceased to drain at each particular suction.

The suction (or tension table) technique is limited to suctions less than approximately 10 kPa. Water held at lower matric potentials ($\psi_m < -10$ kPa) must be measured using a “pressure plate” technique.^[5]

The clay soil in Fig. 2 has a higher water content at each matric potential than the sandy soil, because the clay has a finer texture with smaller pores. The sandy soil pores are larger and can be emptied at relatively low suctions. The shape of the curve for the sandy soil reflects the restricted size range of pores, most of which empty around the same suction (e.g., 10 kPa in Fig. 2). The clay soil has a more gradual curve reflecting the more uniform distribution of pore sizes.

Soil structure affects the shape of the moisture release curve at low suctions (e.g., 0 to ca. 10 kPa), because it is the shape and packing of the structural units that mainly define the volume of macropores. Differences between soils at low suctions provide information about the relative drainage and aeration capacities of the soils.

Pore size distribution can also be measured using mercury intrusion porosimetry.^[8] Pressure is used to force mercury into the soil pores. The relationship between applied pressure and the volume of intruded mercury provides the measure of

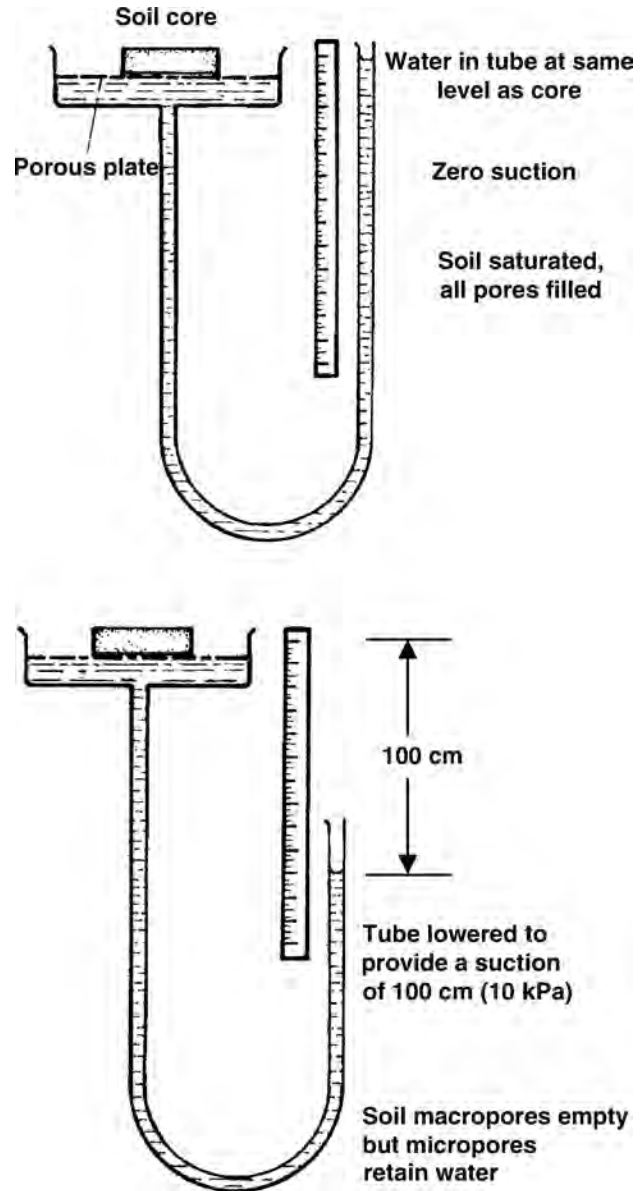


Fig. 3 Measurement of soil moisture release characteristic using tension table equipment.

Source: Adapted from McLaren & Cameron.^[9]

pore size distribution. Although the method is fast, errors may occur because of the differences in the contact angle of the mercury on different mineral surfaces.^[5]

KEY FUNCTIONS OF THE PORE SYSTEM

Porosity and pore size distribution control a wide range of soil phenomena, including the following:

1. *Water storage in soil.* This is one of the critical soil properties that determine the ability of a soil to sustain plant growth. The amount of water that is stored in a

soil depends not just on the total porosity but also on the pore size distribution.

2. *Transmission characteristics of the soil.* The hydraulic conductivity of the soil controls its drainage rate and its ability to support plant growth. Different size classes of pores perform different functions. Macropore flow controls drainage at or near saturation. Micropore flow regulates the rate of water supply to roots and hence determines the onset of plant water stress. The soil pore system also affects the transport of solutes and particles with the soil solution. For saturated or near-saturated flow, macropores can provide a “preferential” or a “bypass” pathway for water, solutes, or particles. Soil micropores provide a “storage volume” that can protect solutes against leaching. Diffusion of solutes out of soil micropores helps to supply nutrients to plant roots. Soil porosity and pore size distribution control the rate of mass flow and diffusion of soil gases. Plant root respiration requires a constant supply of oxygen and produces carbon dioxide. The soil pore system influences the rate of supply of oxygen at the root surface of the plant and rate of diffusion of carbon dioxide away from the root.
3. *Soil mechanical properties, including penetration resistance, soil consistency, compactibility, and “bearing capacity.”* Plant roots grow through the soil pore system in search of water and nutrients. Although roots are able to expand the diameter of a soil pore, the rate of root growth can be severely limited in a compact soil with low porosity. The greater the porosity of a soil, the more easily it can be compacted, and the more likely it is to yield or deform under stress or load.
4. *Location of biological activity, including migration of worms, nematodes, or pathogens.* Although soil organisms can create their own pores by burrowing through soil, they also utilize the existing pore system.

CONCLUSION

Soil pore space is essential because it provides the capacity to store water and air in the soil as well as enables drainage and root growth through the soil. Soil pore space is defined by the relative arrangements of the soil particles and soil aggregates.

REFERENCES

1. da Silva, A.P.; Kay, B.D.; Perfect, E. Characterization of the least limiting water range of soils. *Soil Sci. Soc. Am. J.* **1998**, *58*, 1775–1781.
2. Handreck, K. *Gardening Down-under: A Guide to Healthier Soils and Plants*; Landlinks Press: Melbourne, 2001.
3. Brewer, R. *Fabric and Mineral Analysis of Soils*; Wiley: New York, 1964; 470 pp.
4. Gregory, P.J. Growth and functioning of plant roots. In *Soil Conditions and Plant Growth*; Wild, A., Ed.; Longman: New York, 1988; 113–167.
5. Danielson, R.F.; Sutherland, P.L. Porosity. In *Methods of Soil Analysis. Part 1. Physical and Mineralogical Methods*, 2nd Ed.; Klute, A., Ed.; Agronomy Monograph No. 9; American Society of Agronomy: Madison, 1986; 443–461.
6. Torstensson, G.; Eriksson, S. A new method for determining the porosity of the soil. *Soil Sci.* **1936**, *42*, 405–417.
7. Bullock, P.; Thomasson, A.J. Rothamsted studies of soil structure. II. Measurement and characterisation of macroporosity by image analysis and comparison with data from water retention measurements. *J. Soil Sci.* **1979**, *30*, 391–413.
8. Lawrence, G.P. Measurement of pore sizes in soils: A review of existing techniques. *J. Soil Sci.* **1977**, *28*, 527–540.
9. McLaren, R.G.; Cameron, K.C. *Soil Science: Sustainable Production and Environmental Protection*; Oxford University Press: Auckland, 1996; 304 pp.

Potassium

Philippe Hinsinger

National Institute of Agronomic Research (INRA), Montpellier, France

Abstract

Potassium (atomic weight = 39.098) is an alkaline metal, as revealed by the etymology of its symbol (K) that derives from the Latin word “kalium” and from the Arabic word “qali” (alkali). K is thus strongly electropositive and always occurs as a monovalent cation. Consequently, its physico-chemistry and speciation are rather simple. K is an abundant alkaline metal cation, reaching a concentration of 26 g kg^{-1} in the earth's crust. It is a major nutrient for all living organisms.

POTASSIUM IN PLANTS

Potassium (K) is the major cation in most plants, occurring at concentrations ranging from 5 to 50 g kg^{-1} , twice as much as calcium (Ca) and slightly less than nitrogen (N).^[1–3] The etymology of its name accounts for the abundance of K in plant-derived ash material (potash). K is involved in a large number of physiological processes: osmoregulation and cation–anion balance, protein synthesis, and activation of enzymes.^[2,3] K is often referred to as “a cation for anions” as it balances the abundant negative charges of inorganic (nitrate) and organic (carboxylates) anions in plant cells. It therefore occurs at large concentrations, 100–200 mM in the cytosol, 5 to 10 times less in the vacuole. Being a major inorganic solute, it plays a key role in the water balance of plants: maintenance of the osmotic potential and turgor pressure involved in cell extension. K-controlled changes in turgor pressure in guard cells are a key process of stomatal opening and closure and hence the regulation of plant transpiration. Many of these physiological roles are related to the high mobility of K at all levels in the plant. This unique, considerable mobility of K in the plant is essentially due to the large permeability of cell membranes to K ions, which arises from the occurrence of a range of highly K selective, low and high affinity ion channels and transporters. These are being increasingly characterized at a molecular level.^[4] Large rates of K uptake can thereby be achieved in plant roots. In addition, K ions can easily be leached out of living plant tissues, as documented for tree foliage that contributes a large flux of K back to the soil via throughfall.^[3] K also rapidly leaves dead roots and other plant debris compared with N and P, which requires hydrolysis of organic molecules. At an agronomic level, the demand for K largely varies with plant species and productivity. The uptake of K essentially occurs during the vegetative stage and can reach values of $10 \text{ kg ha}^{-1} \text{ day}^{-1}$ and above. Depending also on the agricultural practices (removal of straw, for instance), the

amount of K removed with the harvested material will range from $5\text{--}50 \text{ kg ha}^{-1}$ for cereal grains to $50\text{--}500 \text{ kg ha}^{-1}$ for forage, root, and tuber or plantation crops.

POTASSIUM IN SOILS

Among major nutrients, K is usually the most abundant in soils as total K content ranges from 0.1 to 40, with an average of 14 g kg^{-1} .^[1,5] A major proportion of soil K occurs as structural K in feldspars and interlayer K in mica-ceous minerals (Fig. 1).^[1,6,7] Some minor proportion of soil K (usually much less than 1%) is adsorbed on negatively charged soil constituents, namely clay minerals and organic matter. A marginal part is present as free K ions in the soil solution. Bulk soil solution concentrations usually amount to $100\text{--}1000 \mu\text{M}$ (less than 0.01–0.1% of total K). The reason for this rather low concentration of K in the soil solution and hence restricted mobility of K in soils, compared to other metal cations such as sodium (Na) or Ca is related to its selective adsorption onto some clay minerals. Because of its ionic radius and small hydration energy, K ions indeed perfectly fit into the interlayer sites of mica-ceous minerals (micas, illites, and mica-derived clays).^[8] These sites and, to a lesser degree, the sites on the frayed edges of these minerals have thus a considerably larger affinity for K than for other cations, including divalent cations such as Ca or magnesium (Mg; Fig. 2). Clay minerals also bear sites with larger affinity for divalent cations than for monovalent cations such as K. These sites are located on the planar faces of clay minerals and are thus dominant in clays such as kaolins and smectites. They also occur in organic compounds. K is thus much less strongly held in soils dominated by kaolins (tropical soils), sand, or organic matter than in soils dominated by illite–vermiculite clay minerals. Traces of mica-derived clay minerals can dramatically influence the dynamics of soil K as evidenced in tropical soils that are apparently dominated by kaolins.^[9]

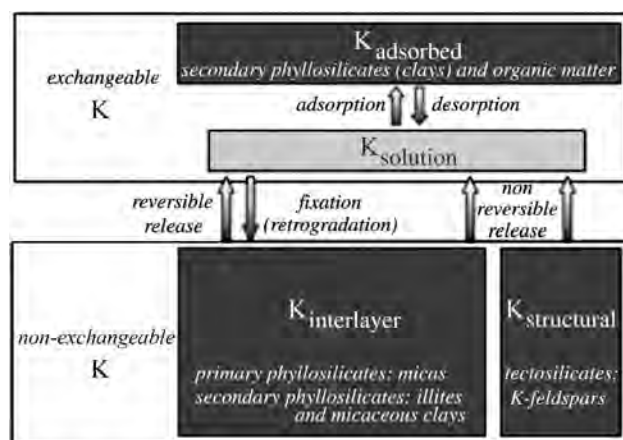


Fig. 1 The various forms of soil K and the chemical processes involved in soil K dynamics.

More generally, K dynamic is largely dependent on soil mineralogy that determines both ion exchange and release-fixation processes,^[6,7] i.e., the dynamics of “non-exchangeable K.” The latter is defined as that (major) portion of soil K that cannot be exchanged by ammonium (NH_4) ions. NH_4 ions have the same charge and radius than K ions and can successfully desorb K only from low K-affinity sites. Exchangeable K thus comprise soil solution and easily desorbable K ions. Non-exchangeable K mostly comprise interlayer K (high K-affinity sites) of micaceous minerals and structural K of feldspars (Fig. 1), i.e., 90–99% of total K in many soils. The release of K from feldspars requires a complete and irreversible dissolution of the mineral and is enhanced under acidic conditions.^[6,7] The release of interlayer K from micaceous minerals can

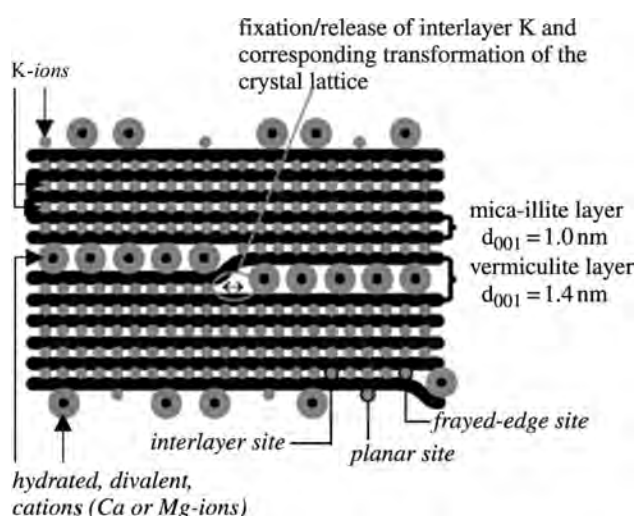


Fig. 2 The various sites of exchangeable K ions in micaceous clay minerals and the transition between mica and vermiculite layer that occurs when interlayer K is exchanged by hydrated, divalent cations.

proceed similarly or involve an ion exchange process (Fig. 1) leading to an expansion of the phyllosilicate (Fig. 2). This reversible release is essentially governed by the concentrations of K and competing cations in the outer solution.^[8,10,11] Cations that can be responsible for this release, such as Ca and Mg ions, have a large hydration energy, contrary to K or NH_4 ions. Therefore, they remain hydrated when exchanging interlayer K ions and expand the interlayer space (Fig. 2), making it possible for the release to proceed further, whereas NH_4 ions would block the reaction.^[12] However, because of the considerable affinity of these interlayer sites for K relative to Ca or Mg, the release can occur only for extremely low concentrations of K in the soil solution (Fig. 3), in the micromolar range.^[11,12] Conversely, elevated concentrations of K are prone to the reverse reaction of fixation, i.e., the collapse of expanded layers and concomitant increase in non-exchangeable K at the expense of readily available K. As long as exchangeable K was assumed to be the only plant-available K fraction, i.e., the poor recovery of applied K in the exchangeable fraction was seen as a poor efficiency of K fertilization. This apparent loss of applied K has therefore received considerably more interest than the reverse release process.^[7] In addition, the release of non-exchangeable K was considered unlikely to occur to any great extent, especially because K concentration in the bulk soil solution of fertilized soils is usually far above the critical concentrations that are prone to K release. However, numerous long-term fertilizer trials have shown that the release of non-exchangeable K contributes a major proportion of soil K supply in unfertilized and possibly fertilized plots too, although an overall net fixation is often found in the latter (Fig. 4).^[5,13] Non-exchangeable K can contribute up to $100 \text{ kg ha}^{-1} \text{ yr}^{-1}$, 80–100% of soil K supply. Such annual fluxes of release are fairly large compared with K dissolution rates commonly estimated by geochemists (in the order

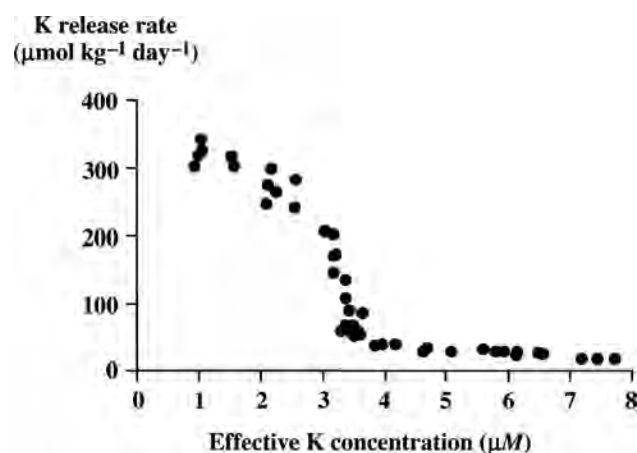


Fig. 3 Effect of solution K concentration on the rate of release of non-exchangeable K from a soil.

Source: Adapted from Springob & Richter.^[11]

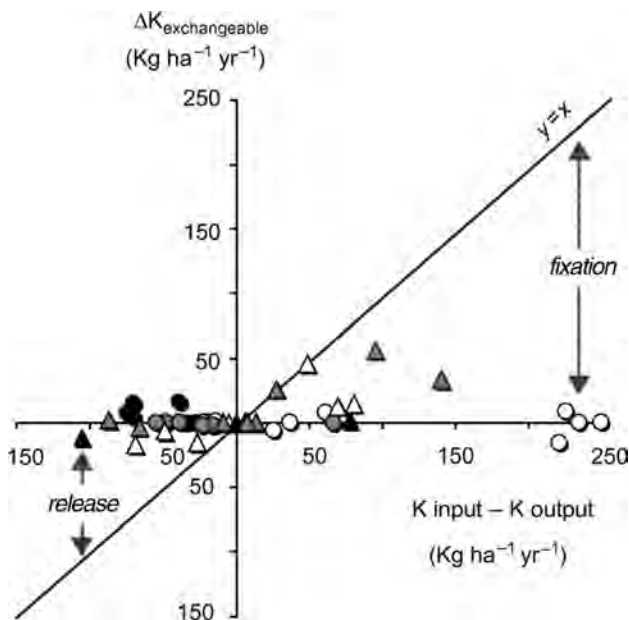


Fig. 4 Annual change in exchangeable K ($\Delta K_{\text{exchangeable}}$) as a function of the annual K budget in various K treatment plots of long-term fertilizer trials. The K input comprehends organic and inorganic, applied K fertilizers. The K output corresponds to the offtake of K in the harvested product.

Source: Adapted from Blake & Mercik^[5] and Gachon.^[13]

of 5–15 kg ha⁻¹ yr⁻¹). However, many geochemical models do not take into account the amount of K taken off in the vegetation, leading to large underestimates of the actual dissolution rates.^[14] Conversely, many K budgets provided by agronomists do not take into account atmospheric inputs and leaching of soil K, assuming that these terms are fairly negligible in most cases. This certainly holds true for atmospheric inputs that rarely exceed values in the order of a few kilogram per hectare per year. Leaching can, however, vary over a much wider range of values from several kilogram per hectare per year in most cases up to several tens (and up to 1–3 hundreds) of kilogram per hectare per year in those situations that are the most prone to leaching: bare soil or poor soil coverage by the vegetation, excessive fertilizer rates, and coarse-textured soils. Omitting the leaching term would, however, have led to underestimating the actual release rate.^[5] Hence, considerable amounts of non-exchangeable K can be released in agricultural soils and contribute a significant proportion of plant uptake, in contradiction to the widespread viewpoint shared by numerous agronomists and soil scientists. Reasons for this can be found when considering root–soil interactions occurring in the rhizosphere.

POTASSIUM IN PLANT–SOIL INTERACTIONS

K occurs at rather low concentrations in the soil solution, compared to other nutrient cations and to the large

requirements of plants for K. The transfer of soil K via mass-flow toward plant roots (i.e., the convective flow of solute accompanying transpiration-driven water flow) contributes about 1–20% of plant demand.^[3,15–17] A direct consequence is the rapid depletion of K ions from the soil solution in the vicinity of plant roots, i.e., the rhizosphere. The resulting concentration gradient generates a diffusion of K ions in the rhizosphere, which plays a key role in the transport of K toward plant root (i.e., 80–99% of plant demand). Such depletion results in a shift of the cation exchange equilibria, which rule the dynamics of both exchangeable and interlayer K. This ultimately results in a desorption of exchangeable K and eventually of interlayer K,^[7,16,17] as shown by their depletion in the rhizosphere (Fig. 5). The extent of the depletion of exchangeable K will depend on chemical parameters such as the initial level of exchangeable K and the K buffering capacity of the soil and on physical parameters that directly determine the diffusive transport of K ions: soil texture and structure and soil water content.^[17] The K depletion zone will extend over several millimeters in clayey, dry soils up to several centimeters in wet, sandy soils.^[7,16,17] The intensity of the depletion will also depend on how far the K concentration of solution is decreased, which may vary among plant species according to the K uptake ability of their root. Plants with a lower external K efficiency, i.e., with a higher affinity transport system, will have the capability to take up K at lower K concentrations and may thus deplete soil K further.^[17] In the vicinity of roots, solution K concentration can indeed decrease by 2–3 orders of magnitude, down to as little as 2–3 μM .^[16]

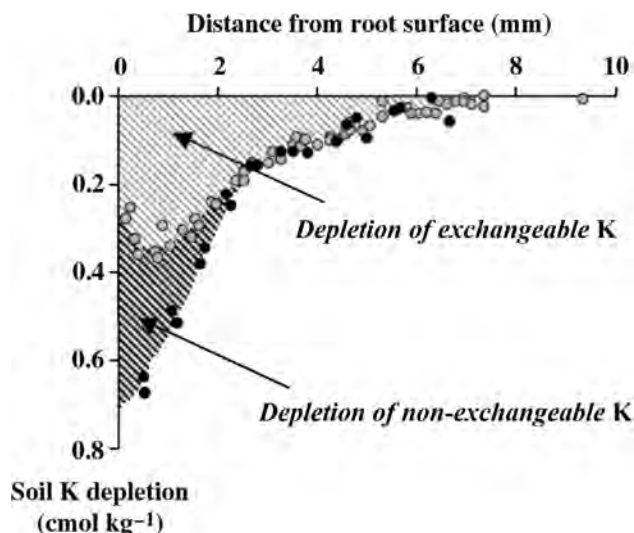


Fig. 5 Depletion of both exchangeable K (gray dots) and HCl-extractable K (black dots) as a function of the distance from rape roots.

Source: Adapted from Jungk & Claassen.^[17] ©1997.

At such low K concentrations, the release of non-exchangeable K can occur at large rates, whereas it would be dramatically restricted at bulk soil K concentrations of several hundreds of micrometer (Fig. 3).^[11] Plants thus play a major role in the dynamic of inter-layer K via the root-induced depletion of solution K.^[16] Measurements in pot experiments have indeed revealed that within several days of growth, the release of non-exchangeable K can amount up to 90% of K supplied to the plant.^[7,16] Soil-root chemical interactions in the rhizosphere thus largely explain the unexpectedly large contribution of the release of non-exchangeable K to plant uptake that is found in many agricultural soils, including fertilized soils.

ASSESSING AND MANAGING POTASSIUM FERTILITY

Soil K fertility is most often evaluated by measuring exchangeable K^[1,7] most frequently with molar NH₄ acetate in batch conditions. However, the adequacy of exchangeable K to predict plant response, i.e., the actual bioavailability of soil K, is rather poor in many soils. This arises from the major contribution of the release of non-exchangeable K in some soils, especially when exchangeable K is low and/or when large reserves of non-exchangeable K are readily available as a consequence of: 1) soil mineralogical composition or 2) fertilization history (build up of fixed K due to excessive K-fertilizer rates). In these situations, quantitative evaluation of the potential release of non-exchangeable K would be highly recommended for a better prediction of plant response and fertilizer needs.^[7] There are several methods for assessing non-exchangeable K but none of them is routinely used on a broad scale because of their cost. These are either based on the use of 1) concentrated, strong acids that dissolve K-bearing minerals or 2) cationic resins or chemicals such as Na tetraphenylboron that can promote the release of interlayer K by removing K ions from soil solution and by shifting the exchange equilibria.^[6,7] Alternatively, correction factors can be used when interpreting exchangeable K values, which account either for the cationic exchange capacity (or clay content) or for the soil type and K release potential.^[7] Exchangeable K is nonetheless often used alone for fertilizer recommendations, resulting in frequently overestimated fertilizer needs to compensate for the expected large fixation and negligible release. Many long-term fertilizer trials have shown that adequate yields of crops can be obtained at fairly low rates of K fertilizer application, or even, for the least demanding crops such as cereals, without any K fertilizer for several years or decades.^[7,13] Other more demanding crops, however, require the application of K fertilizer to achieve high yield and quality in the harvested

products.^[18] The need for K fertilizers will thus depend on the release potential of the soil and on the demand of the plant, the latter being increasingly accounted for in fertilizer recommendations. Fertilizer trials have also shown that commonly used soluble K fertilizers and organic sources such as manure or crop residues have fairly comparable efficiencies. This is not surprising as K is highly mobile in organic compounds where it occurs as soluble or exchangeable K ions. These sources are thus equally important as K fertilizers and absolutely need to be accounted for in K budgets.

CONCLUSION

K is the major nutrient cation for plants and thus taken up at large rates by plant roots. These are achieved by both high and low affinity transport systems, which explain the considerable mobility of K within the plant. In comparison, K is much less mobile in soils because of the strong affinity of some exchange sites of clays. The large K uptake rates achieved by roots result in a steep depletion of solution K in the rhizosphere, and hence in a shift of the equilibria of cation exchange. Exchangeable K and even non-exchangeable K can thereby be significantly depleted and contribute a substantial proportion of plant uptake. This is confirmed by K balance in both short-term pot experiments and long-term field trials. In addition to the desorption-adsorption of exchangeable K, release and fixation processes thus need to be accounted for when evaluating soil K fertility.

REFERENCES

1. Munson, R.D. *Potassium in Agriculture*; American Society of Agronomy, Crop Science Society of America, Soil Science Society of America: Madison, 1985; 1223.
2. Mengel, K.; Kirkby, E.A.; Kosegarten, H.; Appel, T. *Principles of Plant Nutrition*, 5th Ed.; Kluwer Academic Publishers: Dordrecht, 2001; 673.
3. Marschner, H. *Mineral Nutrition of Higher Plants*, 2nd Ed.; Academic Press: London, 1995; 889.
4. Schachtman, D.P. Molecular insights into the structure and function of plant K⁺ transport mechanisms. *Biochim. Biophys. Acta* **2000**, *1465*, 127–139.
5. Blake, L.; Mercik, S.; Koerschens, M.; Goulding, K.W.T.; Stempen, S.; Weigel, A.; Poulton, P.R.; Powlson, D.S. Potassium content in soil, uptake in plants and the potassium balance in three European long-term field experiments. *Plant Soil* **1999**, *216*, 1–14.
6. Sparks, D.L. Potassium dynamics in soils. *Adv. Soil Sci.* **1987**, *6*, 1–63.
7. International Potash Institute. Methodology in soil-K research. In Proceedings of the 20th colloquium of the International Potash Institute, Baden bei Wien, Austria; International Potash Institute: Bern, 1987; 428.

8. Dixon, J.D.; Weed, S.B. *Minerals in Soil Environment*; Soil Science Society of America: Madison, 1989; 1244.
9. Fontaine, S.; Delvaux, B.; Dufey, J.E.; Herbillon, A.J. Potassium exchange behaviour in Carribean Volcanic ash soils under banana cultivation. *Plant Soil* **1989**, *120*, 283–290.
10. Schneider, A. Influence of soil solution Ca concentration on short-term release and fixation of a loamy soil. *Eur. J. Soil Sci.* **1997**, *48*, 513–522.
11. Springob, G.; Richter, J. Measuring interlayer potassium release rates from soil materials. II. A percolation procedure to study the influence of the variable solute K in the $< 1 \dots 10 \mu\text{M}$ range. *Z. Pflanzen. Bodenk.* **1998**, *161*, 323–329.
12. Springob, G. Blocking the release of potassium from clay interlayers by small concentrations of NH_4^+ and Cs^+ . *Eur. J. Soil Sci.* **1999**, *50*, 665–674.
13. Gachon, L. *Phosphore et Potassium dans les Relations Sol–Plante: Conséquences sur la Fertilisation*; Institut National de la Recherche Agronomique: Paris, 1988; 566.
14. Taylor, A.B.; Velbel, M.A. Geochemical mass balances and weathering rates in forested watersheds of the southern blue ridge II. Effects of botanical uptake terms. *Geoderma* **1991**, *51*, 29–50.
15. Barber, S.A. *Soil Nutrient Bioavailability. A Mechanistic Approach*, 2nd Ed.; Wiley: New York, 1995; 414.
16. Hinsinger, P. How do plant roots acquire mineral nutrients? Chemical processes involved in the rhizosphere. *Adv. Agron.* **1998**, *64*, 225–265.
17. Jungk, A.; Claassen, N. Ion diffusion in the soil–root system. *Adv. Agron.* **1997**, *61*, 53–110.
18. <http://www.ipipotash.org/en/publications.php> (accessed August 21, 2015).

Potassium: Dynamics and Mineralogy

A.V. Shanwal

Department of Soil Science, Chaudhary Charan Singh Haryana Agricultural University, Hisar, India

S.S. Dahiya

Chaudhary Charan Singh Haryana Agricultural University, Hisar, India

Abstract

Potassium (K) exists in different forms, which are interrelated and maintain equilibrium in the soil system. The existence of these various forms of soil K and its incessant transformation from one form into another as well as the gain and losses generate a dynamic system in soil. The most important component of this dynamics is soil mineralogy, including primary and secondary minerals. The status of different forms of K in soil, their release characteristics, fixation, etc. are the other important components of K dynamics, which in turn are regulated by the soil mineralogical makeup.

INTRODUCTION

Soil contains some reserves of nutrients required by plants, which maintain the natural vegetation and food crops, the growth of which is regulated by the amount available. The natural reserve of potassium (K), a major essential plant nutrient, is fairly rich in soil. It comprises an average of 2.6% of the earth's crust, making it the seventh most abundant element and the fourth most abundant mineral nutrient in the lithosphere.^[1] K is the largest cation in non-hydrated state among the plant nutrients with 1.33-Å radius and 8–12 coordination number. K is generally preferred in ion-exchange reaction due to its higher polarizability (0.88 Å^3) and causes negligible swelling because of low hydration energy ($142.5 \text{ kJ g}^{-1} \text{ ion}^{-1}$) compared to other elements.^[2]

K occurs most frequently in the mineral, which is crystallized latest in the lithosphere and thereby concentrated in the earth's outer crust. The origin of K in natural soil resides in these K-bearing minerals, which release this element upon weathering during the soil-forming processes. Varying contents of K (0.4–3.0%) are thus found in the mineral soils depending upon the kind of K-bearing minerals, their degree of weathering, and the intensity of soil-forming processes. Most K in tropical and subtropical regions has been leached out of the soil profile because of strong weathering condition.

K MINERALOGY

K is an important constituent of layer and tecto aluminosilicates of primary and secondary minerals. The primary layer aluminosilicates include dioctahedral micas (muscovite

and glauconite) and trioctahedral micas (biotite and phlopopite). The tecto aluminosilicates are mainly K-feldspar, sanidine, orthoclase, and microcline feldspars. The secondary minerals include illite, transitional or interstratified clay minerals, and allophanes. Primary minerals contain 10% K. Among secondary minerals, illite contains a sufficient amount of K (3–8%), whereas transitional clay minerals and allophanes contain less than 1%.

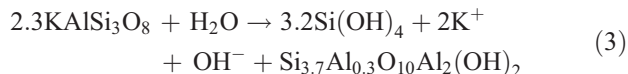
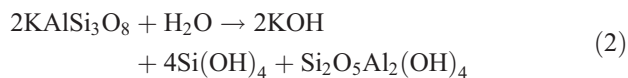
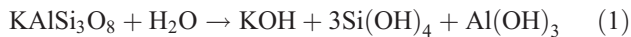
Feldspars, Their Structure and Weathering

Feldspars are abundant in the igneous, sedimentary, and metamorphic rocks and constitute about 15% of the total lithosphere. Therefore, it forms a most important group of minerals and has made a special place in the classification from geochemical point of view.

The structure of K-feldspars is a 3-D framework of [silicon (Si) and aluminum (Al)]-O₄ tetrahedral groups by sharing of corner oxygen as in the case of SiO₂ polymorphs. The framework of four-membered rings of tetrahedral groups encloses large interstices occupied by K, sodium (Na), calcium, barium, and other cations. In sanidine, orthoclase, and microcline, these sites are occupied by K. The K–O distances ($\sim 3 \text{ Å}$) indicate an irregular nine-fold coordination for the K⁺ ion. The polymorphism of K-feldspars (sanidine, orthoclase, microcline, and adularia) has an ideal chemical composition of KAlSi₃O₈, where one out of every four Si atoms in the framework is replaced by Al. In nature, it hardly occurs but contains foreign cations either by replacing Si or Al in four-fold coordination or by K⁺ ions in 8–12-fold coordination.^[3]

The exact mechanism of feldspar weathering is not very clear. Proteolysis is probably the most important reaction

resulting in the release of K from feldspar. Depending upon the amount of H_3O^+ ion, the hydrolysis of feldspar takes place as shown in Eqs. 1–3.



In the presence of excess hydronium ion, at low pH, total hydrolysis releases K^+ and solubilizes the silica and reprecipitates Al as $\text{Al}(\text{OH})_3$. Medium hydronium ions transform the feldspar into kaolinite through partial hydrolysis at low pH. However, the partial hydrolysis also produces smectites (montmorillonite and beidellite) but at alkaline pH and poor drainage condition.^[4]

Unlike mica, the weathering of feldspar proceeds through surface reactions in which K^+ is replaced by H_3O^+ . The K^+ ion diffusion slows down with time due to the development of thin (30-nm) residual layer of amorphous material around the feldspar grains. Apart from the weathering environment, the structural properties of the feldspars affect the weathering rate including its composition, Na content, the order of Si and Al in the crystal structure, crystal and particle size, purity, and nature of the twinning. Based on the degree of disorder of the Al and Si distribution in K-bearing feldspar, the weathering sequence follows as: sanidine > orthoclase > intermediate microcline and maximum microcline. In perthitic structures, albite is more weatherable than the K-feldspars due to the presence of Na^+ ion. Among feldspars, maximum microcline is most resistant to weathering due to Si/Al order, low Na content, lack of perthitic structure, and crystal size.

Micaceous Minerals: Their Structure and Weathering

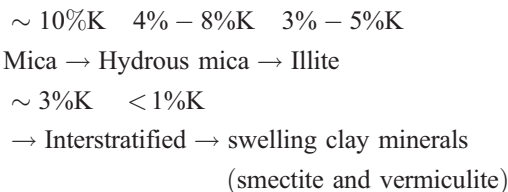
Micaceous minerals in the soil and rocks include primary mica (muscovite, biotite, and phlogopite), hydrous mica or degraded mica, and illite. Micas are third most extensive group of minerals after feldspars and quartz in igneous and metamorphic rocks.

Structurally all the micaceous minerals are similar and belong to group 2:1 phyllosilicates of the silicate mineral class but differs in K content. Primary mica contains 10% of K while hydrous micas and illites contain only 3–5%. The structural unit of mica consists of one layer of Al^{3+} octahedra sandwiched between two layers of Si^{4+} tetrahedra. Similar to feldspars, every fourth Si^{4+} tetrahedron is replaced by Al^{3+} creating a unit negative charge per four tetrahedra. Finally, when each such layer is joined face to face with another, the residual charge is neutralized

by K^+ fitted in the 12 coordinated hexagonal holes. But, K^+ is 6 rather than 12 coordinated because in each hexagonal holes three alternate oxygen move inwards and remaining three move outwards and are at a distance of 2.81–3.40 Å from K ions.^[5]

In the trioctahedral micas, the central cation is divalent (Mg^{2+} or Fe^{2+}) and all the three octahedral positions are filled, whereas in dioctahedral micas, the central cation is trivalent (Al^{3+} or Fe^{3+}) and only two out of three octahedral cation positions are filled. The stylized mica structure corresponds to a unit cell with approximate dimensions: $a = 5$ Å, $b = 9$ Å, $c = 10$ Å, which is monoclinic with $\beta = 95^\circ$. This unit cell chemical formula is $\text{K}_2\text{Al}_4(\text{Si}_6\text{Al}_2)\text{O}_{20}(\text{OH})_4$ for dioctahedral and $\text{K}_2\text{Mg}_6(\text{Si}_6\text{Al}_2)\text{O}_{20}(\text{OH})_4$ for trioctahedral.

Approximately 95% of the clay minerals of soils are produced by weathering of micas. The transformation of micaceous clay begins at a particle size of <0.2 μm. Although the weathering of mica involves proteolytic reaction, unlike feldspars, the H_3O^+ ions attack the lateral edges of the particle and proceed zonally inward toward the center of the mica lattice. During this process, the edges expand to ~18 Å by releasing K^+ ion and reducing the layer charge. The expansion of interlattice space of mica and release of K^+ ion and reduction in layer charge eventually result in a series of mineral transformations as follows:



This series of mica transformation ends with swelling clay minerals such as smectite and vermiculite (Fig. 1) in neutral or alkaline pH soils, but in acidic soils the release of K^+ ion from micas proceeds with the dissolution of the mineral followed by the formation of weathering products including kaolin and hydroxides of iron and aluminum.^[6]

However, the weathering of mica is mainly governed by the amount of H_3O^+ ion in the soil environment although structural characteristics of the mineral such as tetrahedral rotation and tilting, hydroxyl ion orientation, octahedral cations, particle size, and K depletion also play an important role. The weathering of the trioctahedral mica (biotite and phlogopite) is much more rapid than the dioctahedral mica (muscovite) due to its structural imperfection.

Transitional Clay Minerals and Allophanes

Although K is not part of the structure of these minerals and its content is also much less (~1%), they play an important role in plant nutrition through cation-exchange reaction. Transitional or interstratified clay minerals are the intermediate stage of mica transformation in which some or more

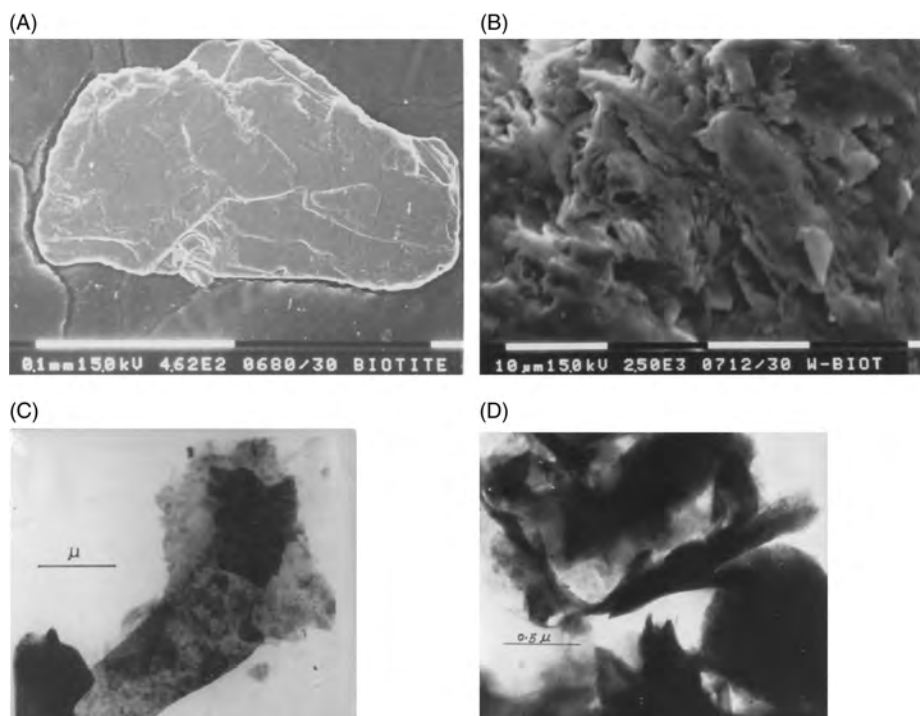


Fig. 1 Optical and electron micrographs of various stages of mica transformation in southern Haryana soils: (A) fresh biotite mineral without exfoliation and wrinkles; (B) weathered biotite showing fissures and crevices; (C) biotite particles transformed into interstratified minerals; and (D) biotite completely transformed into smectites.

layers of mica gets transformed into swelling minerals (smectite or vermiculite) either in regular, irregular, or random fashion.

Allophanes are series of hydrous aluminosilicate minerals of widely varying composition. Allophane has a definite affinity for K because of its structural configuration and has high CEC.

SOIL K DYNAMICS

Plant absorbs K from soil solution but its amount is so low that plant cannot complete its life cycle merely on this basis and is replenished from the reserve pool. As a result, K maintains equilibrium between its various phases referred to as K dynamics (Fig. 2).

Sharp distinction between various phases of K is not possible and often there is an overlap between these forms due to their dynamic equilibrium. The available K (solution K + exchangeable K) usually forms a small fraction (~2%) of the total K but is often large enough to satisfy plant requirements of short duration crop, although too small to be able to meet the needs of one large duration crop or several short duration crops. Therefore, for sustainable crop production, the available K must be continually replenished through non-exchangeable and mineral K reserves.

The dynamic equilibrium between solution, exchangeable, non-exchangeable, and mineral forms of K strongly influences K chemistry. The rate and direction of these reactions determine the release, fixation, plant uptake, and leaching of applied and native K. The kinetics of reactions

between these phases is prerequisite for making proper recommendations for K fertilizer use.

Kinetics of K Release

The release of reserve K is actually not the result of dissolution of K-bearing minerals but is an exchange reaction, which is so slow that it cannot be measured with normal methods of determining exchangeable K. During the course of plant growing season, K in soil solution is continually being replenished through release from exchangeable form, which in turn is replenished from non-exchangeable and mineral K.^[7]

Various methods and extractants such as salts, ion-exchange resins, sodium tetraphenyl boron (NaTPB), and organic and mineral acids have been used to study the kinetics of K release from the soil and its different fractions.^[8]

Normally nitric acid (HNO_3) extracts the highest amount of non-exchangeable K from soils followed by H-resin, NaTPB, and oxalic and citric acids. A number of

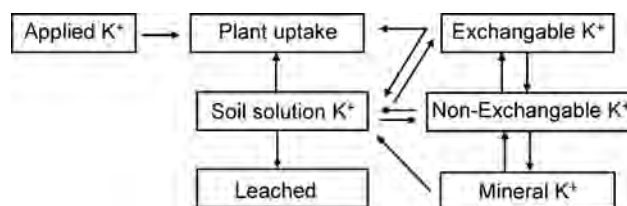


Fig. 2 The dynamics between various forms of K.

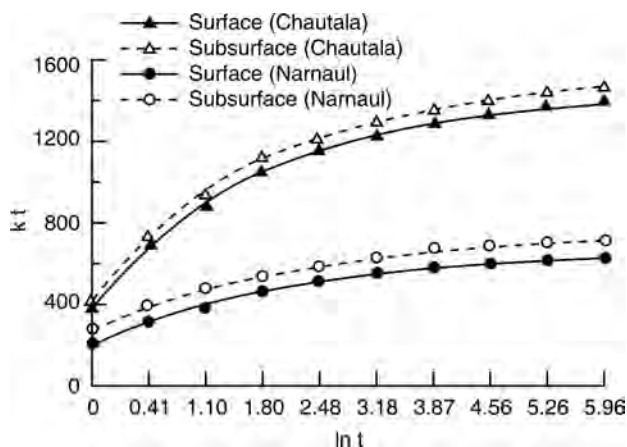


Fig. 3 Kinetics of non-exchangeable K release from the soils of North India to hydrogen region as described by elovich equation.

equations have been used to describe K^+ release kinetics in soils. These include the zero-order, first-order, elovich, and parabolic diffusion equations.^[9] The first-order equation is most successful to describe K kinetic release and K uptake by plants by most methods except H^+ resin, which obeys elovich equation with curvilinear pattern (Fig. 3).

The release of K is directly governed by the amount and mineralogical composition of soils. The illitic soils show the highest rate of K release followed by smectitic and kaolinitic soils regardless of the content of exchangeable K in the soils. The high rate of K release at the initial stage is mainly due to external domain (surface and edges) and the slower rate due to the restricted domain (interlayer sites) through diffusion.

K Fixation

Fixation of K is conversion of soil solution or exchangeable K into non-exchangeable forms. Normally the fixation and release of K occur simultaneously. In soils containing predominantly micaceous and vermiculite minerals, a large fraction of added K can be fixed. The fixation may convert the swelling clay minerals into micaceous minerals. The swelling clay minerals (14 Å) when dried in the presence of K ions lose their shell of water molecule and allow K ions to enter the pseudohexagonal space between tetrahedral layers and become electrostatically bound.^[10]

SOIL TEST CROP RESPONSE OF K

Various extractants and methods have been used for assessing the portion of soil K that is available to plants.^[11] Probably most widespread is the use of neutral 1 M NH_4OAc solution. Other extractants frequently used are Bray solution, various salts, buffered or unbuffered NH_4 -oxalate, double acid ($HCl-H_2SO_4$), Morgan's extract, dilute H_2SO_4 , organic acids, NaTPB, HNO_3 , etc.^[10]

Backett's quantity and intensity relationship (Q/I) and electro ultra filtration (EUF) at various potentials are also methods for estimating the available K but are not in common use due to the cost-effective and cumbersome nature.

Soil tests for K are used as bases for fertilizer recommendations for crops. Therefore, any method that samples the available fraction of plant nutrient can be used as a satisfactory soil test, and hence as a basis for making fertilizer recommendations provided a calibration curve relating soil test vs. yields through the normal response range is available.

Normally a very high correlation coefficient is observed between K extracted by neutral 1 M NH_4OAc and K uptake by plants and plant yields. However, this method fails in light textured soils of arid and semiarid regions where exchangeable K content is almost negligible. Under such conditions, 0.5 M HNO_3 , NaTPB, and EUF are more reliable methods in estimating the K available to plants as they also extract non-exchangeable K. The long duration crops also extract non-exchangeable K particularly in light textured soils under stress condition.^[9,12]

CONCLUSION

K is an important constituent of feldspars and micas, which weather and release K through proteolytic reaction but by different mechanisms. The weathering of feldspars proceeds through surface reactions and is a function of soil environment (pH, Eh, etc.) and its structural imperfection. Microcline feldspar is most resistant to weathering due to Si/Al order, lack of perthitic structure, and crystal size. Unlike feldspars, micas weather through interlattice expansion. Trioctahedral micas weather faster than dioctahedral micas due to shorter bond length and greater bond strength of K-O in the latter. Transitional clay minerals contain a negligible amount (~1%) of K but play an important role in plant nutrition.

K exists in different forms, which are in dynamic equilibrium. The rate and direction of these equilibria determine the release, fixation, plant uptake, and leaching of applied and native K. The kinetics of K release by most methods is explained by first order equation. Illitic soils release the highest K followed by smectitic and kaolinitic soils. The swelling clay minerals (smectite and vermiculite) fix appreciable amounts of added K.

Soil tests for K are used as basis for fertilizer recommendations for crops. Neutral 1 M NH_4OAc method is widely used for assessing available K. However, inclusion of non-exchangeable K is also relevant.

REFERENCES

1. Hurbut, C.S., Jr.; Klein, C. *Manual of Mineralogy*, 19th Ed.; John Wiley and Sons: New York, 1977.

2. Sparks, D.L.; Huang, P.M. Physical chemistry of soil potassium. In *Potassium in Agriculture*; Munson, R.D., Ed.; American Society of Agronomy: Madison, 1985; 201–276.
3. Rodoslovich, E.W. The cell dimension and symmetry of layer silicate: N. interatomic forces. *Am. Miner.* **1975**, *48*, 176–179.
4. Pedro, G. Current aspects of the mineralogy of clays and soils. In *Methodology in Soil-K Research*; Steineck, O., Ed.; International Potash Institute: Switzerland, 1987; 11–43.
5. Rich, C.I. Mineralogy of soil potassium. In *Role of Potassium in Agriculture*; Kilmer, V.I., Ed.; American Society of Agronomy: Madison, 1968; 76–96.
6. Shanwal, A.V.; Ghosh, S.K. Smectite formation in catena of soils derived from micaceous alluvium. *Int. J. Trop. Agric.* **1985**, 273–286.
7. Sparks, D.L. Potassium dynamics in soils. *Adv. Soil Sci.* **1987**, *6*, 1–63.
8. Novozamsky, I.; Houba, V.J.G. Critical evaluation of soil testing methods for K. In *Methodology in Soil-K Research*; Steineck, O., Ed.; International Potash Institute: Switzerland, 1987; 177–197.
9. Shanwal, A.V.; Singh, S.P. Mineralogy and dynamics of potassium in soils of arid and semi-arid regions of India. In *Potassium in Indian Agriculture*; Pasricha, N.S., Bansal, S.K., Eds.; Potash Research Institute of India: Gurgaon, 2001; 45–74.
10. Goulding, K.W.T. Potassium fixation and release. In *Methodology in Soil-K Research*; Steineck, O., Ed.; International Potash Institute: Switzerland, 1987; 137–154.
11. Bertsch, P.M.; Thomas, G.W. Potassium status of temperate regions' soils. In *Potassium in Agriculture*; Munson, R.D., Ed.; American Society of Agronomy: Madison, 1985; 131–162.
12. Sudjadi, M.; Sri Adiningsih, J.; Gill, D.W. Potassium availability in soils of Indonesia. In *Potassium in the Agricultural System of the Humid Tropics*; Cooke, G.W., Ed.; International Potassium Research Institute: Switzerland, 1985; 185–196.

Potassium: Nutrition in Semi-Arid Tropics

D.L. Oswalt

(Retired), Plymouth, Michigan, U.S.A.

Abstract

Short periods of intense rainfall causing waterlogging in warm Alfisol or Vertisol topsoils apparently reduce the uptake of potassium (K) ions from the active root zone. Highest grain yields, comparing 13 years of data in the semiarid tropics, unexpectedly occurred during the years with adequate but lower rainfall. K deficiency symptoms may appear prior to nitrogen or drought symptoms as rainfall diminishes. Midseason soil tests indicated reduced available K, as sample depth was increased. Sorghum embryo number may be reduced owing to inadequate nutrients being transported to the fertilized cells, resulting in lower grain yield when the topsoil becomes waterlogged at flowering time. Muriate of potash or potassium chloride applied to soils in the semiarid tropics, where salts are not normally leached through the soil profile, may not show a positive response in plant growth.

INTRODUCTION

Personal observations in northeastern Nigeria found small spots of well-developed grain sorghum [*Sorghum bicolor* (L.) Moench] in farmer's fields where grass or brush heaps had been burned, where wood ashes were present in the soil around compounds, on old termite mounds, and in drip areas of certain trees with leaves high in nitrogen (N) and potassium (K). Farmers often grow vegetables on raised beds or ridges in semiarid regions. These observations indicate that added K improves plant growth. Dry season soil tests of Alfisols and Vertisols often indicated adequate available K for normal plant growth; however, grain yields are low.

DEMONSTRATIONS

Vegetative growth of rain-fed grain sorghum and other crops in the semiarid regions with basal applications of fertilizer often appears normal prior to flowering (personal observations); however, grain yields are seldom 50% of the same variety grown in temperate regions. Personal observations found general yellowing of the older leaves occurring soon after flowering where topsoils had been waterlogged during high rainfall events. Plant tissue tests^[1] conducted on such plants indicated that N and P were present and K was very low in the leaf or stem sap. Soon after rainfall ceased, the lower leaves yellowed from N deficiency and drought resulted in brown lower leaves.

Long-term muriate of potash trials in Nigeria and potash fertilizer trials in India all indicated no benefit, and in some incidents, there was no plant growth at the higher applications of muriate of potash. Semiarid tropical soils often form salt deposits on the surface due to poor drainage or lack of normal leaching of salts through the profile.^[2] This

poses the question of potential toxicity to the chlorine ion^[3] from muriate of potash. Personal demonstrations using potassium chloride, potassium nitrate, and ammonium nitrate applied to sorghum at flowering produced a noticeable yield increase only from the potassium nitrate.

Two hybrid grain sorghum varieties were used as checks during 13 years of replicated yield trials by the International Crops Research Institute for the Semi-Arid Tropics (ICRISAT) trainees. Of the highest yielding experiments, 8 or 10 were selected from each year's trials. These average grain yields were compared with rainfall data. Unexpectedly, the lowest grain yields were produced in years when rainfall during June, July, and August exceeded 600 mm (Fig. 1). These sorghum hybrids indicated little variation in seed weight from year to year. Therefore, grain yield differences were due to variations in seed number. These results were not predicted for the semiarid tropics where rainfall is considered as yield-limiting. It has been reported that depleted oxygen supply and repeated wetting and drying constrain normal water and nutrient uptake.^[3–6] This would indicate that saturation of the soil in midseason limits grain yield. (See also the entry *Aeration: Tillage Effects*, p. 47.)

Personal tests of the sorghum leaf tissue from a yield trial on a Vertisol (when it was very difficult to walk through the wet plots) indicated high levels of N and P and a low level of K at the time of initial flowering. In replicated plots, Hoagland's solution was applied in the row between plants through tubes to a depth of 15–20 cm. In adjacent plots, Hoagland's solution minus K was applied to maintain the same application of water and other nutrients. Eight sorghum plants at 50% flowering and with similar plant heights (similar physiological development) were tagged on the same date in all plots. A few days later, the plant sap indicated a lower level of K, but higher levels of

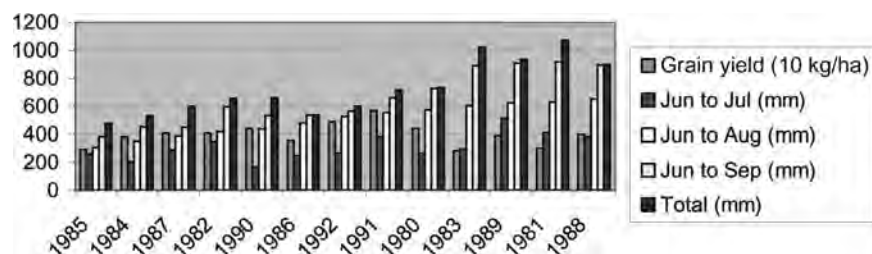


Fig. 1 Hybrid grain sorghum grain yield compared to rainfall (10 kg/ha or mm).

N and P in the plots receiving Hoagland's solution minus K. Plant sap from the plots receiving the complete Hoagland's solution indicated high levels of N, P, and K. Subsequent grain yields indicate a 30% increased grain weight where complete Hoagland's solution had been applied (Table 1). The similar seed weight indicates an increased number of embryos survived and matured where the complete Hoagland's solution was applied. Therefore, the roots remained active;^[3] however, they were not taking K from the soil after the topsoil had been wet from intense rainfall events.^[3,7]

Moist Vertisol soil samples (corrected for moisture content) collected from a fallow crop rotation plot during 1992 and their dried samples were tested for ammonium acetate-extractable K (mg kg^{-1}). Apparent soluble K was lower in all samples than that in dry soil samples taken during the previous dry season.^[3] The extractable K was lower at greater depths and in the moist samples before drying (Table 2), indicating that the available K had not leached downward. The 1992 annual rainfall was among the lower years (Fig. 1).

CONCLUSIONS

Demonstrations and observations indicate a midseason reduction of K uptake from soils (normally testing high in available K in the semiarid tropics where soil organic matter is usually low) that unexpectedly reduces sorghum grain yield during years with more than 600 mm rainfall during June, July, and August. Factors reducing the uptake of K from temporarily waterlogged root zones, caused by intense rainfall, merit the identification of the systems and time involved. The identification of effective topsoil drain-

Table 1 Grain harvested from tagged heads (average per plant).

	Grain/head (g)	1000 seed weight (g)
Hoagland solution	47.7	19.3
Hoagland solution (-K)	36.7	19.5

Table 2 Soluble K (mg/kg) from corrected moist samples when compared with their dried samples (rainfall—219 mm; August 1–18, 1992; 2 mm: August 19–24, 1992).

			Aug 12	Aug 19	Aug 24
Spot 1	0–15 cm	Moist	58	89	89
	15–30 cm	Moist	101	29	30
Spot 1	0–15 cm	Air dry		125	139
	15–30 cm	Air dry		89	85
Spot 2	0–15 cm	Moist		88	116
	15–30 cm	Moist		36	68
Spot 2	0–15 cm	Air dry		130	156
	15–30 cm	Air dry		93	91

age systems and methods of K fertilization merit further investigations into semiarid tropical soils.

ACKNOWLEDGMENT

The author acknowledges the use of the ICRISAT Training Program data from replicated yield trials, ICRSAT rainfall data, and ICRISAT facilities for laboratory analysis of soil samples.

REFERENCES

1. *Plant Check*, 4200 Woodville Pike, Urbana, Ohio.
2. Sparks, D.L. *Environmental Soil Chemistry*, 2nd Ed.; Academic Press: Amsterdam, Boston, 2003.
3. Munson, R.D., Ed. *Potassium in Agriculture*; ASA, CSSA, SSSA: Madison, 1985.
4. Quemener, J. *The Measurement of Soil Potassium*; International Potash Institute: Berlin, 1979.
5. Fitter, A.H.; Hay, R.K.M. *Environmental Physiology of Plants*, 3rd Ed.; Academic Press: London, 2002.
6. Mengel, K.; Kirby, E.A. *Principles of Plant Nutrition*; Kluwer Academic Press: Dordrecht, 2001.
7. Stucki, J.W.; Hou, D. Continued studies on the behavior of potassium in soils 1997. In *Illinois Fertilizer Conference Proceedings*, Jan 27–29, 1997.

Power Plants: Carbon Dioxide Capture

Sean I. Plasynski

Thomas J. Feeley

National Energy Technology Laboratory, U.S. Department of Energy, Pittsburgh, Pennsylvania, U.S.A.

Timothy Fout

National Energy Technology Laboratory, U.S. Department of Energy, Morgantown, West Virginia, U.S.A.

Howard G. McIlvried

Rameshwar D. Srivastava

National Energy Technology Laboratory, Science Applications International Corporation, Pittsburgh, Pennsylvania, U.S.A.

Abstract

An option that has been proposed to combat the buildup of greenhouse gases in the atmosphere is the capture of carbon dioxide (CO₂) at a power plant and injection of CO₂ into a geologic formation for permanent storage, a process generally referred to as carbon capture and storage. However, because CO₂ in power plant flue gas is at a low partial pressure, the technology for CO₂ capture is expensive and could raise the cost of power by 50–100% for new pulverized coal plants, with considerably higher increases possible for plants retrofitted with CO₂ capture systems, when the cost of replacement power is included. This entry describes the processes for CO₂ capture and some new technologies that hold promise of increasing efficiency and reducing costs.

INTRODUCTION

There is a growing concern that the buildup of carbon dioxide (CO₂) in the atmosphere is contributing to global climate change. The combustion of fossil fuels, particularly coal, in power plants for the production of electricity is a major source of CO₂ emissions. A promising approach to reducing greenhouse gas emissions from fossil-fuel-fired power plants and other large point sources is carbon capture and storage (CCS), which involves the capture of CO₂ from a large point source, transport to a storage site, and injection into a geologic formation for essentially permanent storage. The first, and most expensive, step in this chain is CO₂ capture.

CO₂ CAPTURE

There are three basic schemes for the capture of CO₂ from fossil-fuel-fired power plants ([Fig. 1](#)): postcombustion capture, precombustion capture, and oxygen-combustion (oxy-combustion). In postcombustion capture, CO₂ is recovered from the flue gas produced when the fuel is burned. In precombustion capture, the fuel is decarbonized before combustion, and in oxy-combustion, the fuel is burned in an oxygen-enriched environment using pure oxygen

diluted with recycled flue gas or steam to produce a flue gas consisting predominantly of CO₂ and water vapor (H₂O), from which CO₂ is recovered by condensing the water. The advantages and disadvantages of these three options are summarized in [Table 1](#).

POSTCOMBUSTION CAPTURE

Most fossil-fuel-fired power plants use air as the oxidant. The result is that the flue gas produced contains a large amount of nitrogen (typically about 70% for a coal-fired plant) with CO₂ concentration being in the range of 13–15%. Since flue gas is typically at about atmospheric pressure, the driving force (CO₂ partial pressure) for CO₂ capture is very low.

Because CO₂ removal from streams, such as natural gas and hydrogen (H₂), is important in the oil and gas industries, a number of processes have been commercialized for recovering CO₂ from gas streams. These processes break down into two categories: those based on physical solvents, in which the CO₂ merely dissolves in the solvent like CO₂ in seltzer water, and those based on chemical solvents, in which a weak chemical reaction occurs between the CO₂ and the solvent. The capacity of a physical solvent is a

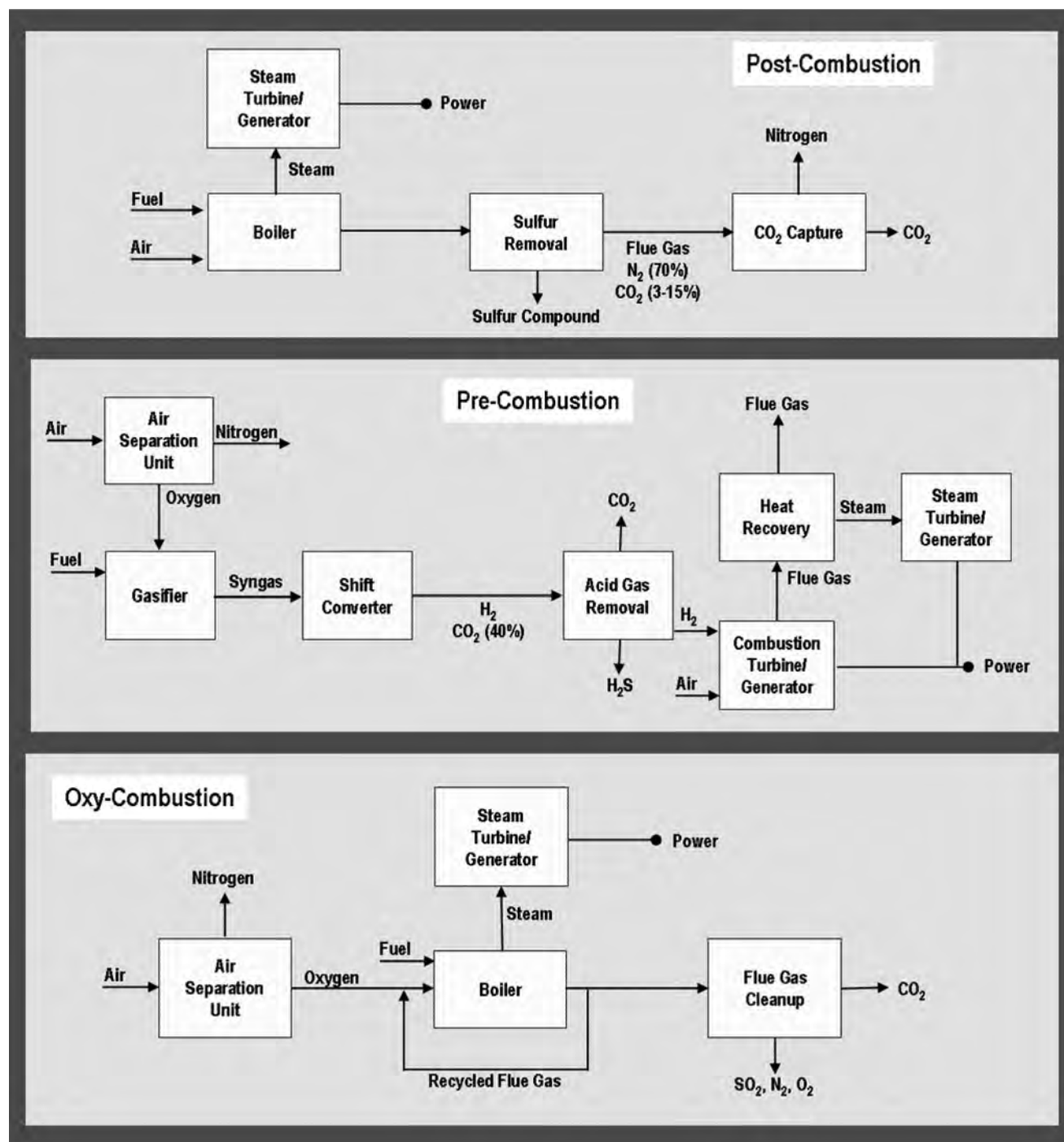


Fig. 1 Schematic diagram showing carbon capture options.

Source: From Figueroa, Fout, et al.^[1]

function of the partial pressure of CO₂. Physical solvents are regenerated by reducing pressure, increasing temperature, or both. Because of the low CO₂ partial pressure in flue gas, physical solvent-based processes are not feasible for postcombustion CO₂ capture.

Since chemical solvents act through a different mechanism (forming a weak chemical compound with the CO₂), they can absorb CO₂ even when the CO₂ has a very low

partial pressure. The solvent is regenerated by heating, which breaks the chemical bond and releases the absorbed CO₂. Most of these processes are based on the use of an amine as solvent. Popular solvents are monoethanolamine and diethanolamine (DEA), although other amines are sometimes used. With DEA, the reaction can be represented by:

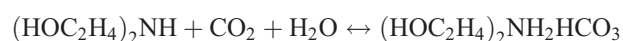


Table 1 Advantages and disadvantages of different CO₂ capture approaches.

	Advantages	Disadvantages
Postcombustion	Compatible with the state-of-the-art pulverized coal-fired boilers Could be retrofit to the majority of coal-fired power plants	Flue gas is diluted in CO₂ and at ambient pressure Low CO₂ partial pressure requires use of a chemical solvent for a high capture level CO₂ recovery at low pressure requires a large compressor load The above factors result in a significant decrease in plant efficiency and output
Precombustion	Synthesis gas is at high pressure and has a high CO₂ concentration High CO₂ partial pressure results in a large driving force for CO₂ capture Able to use physical solvents for acid gas capture Recovery of CO₂ at higher pressure results in reduced compression costs	Applicable mainly to new plants, as few gasifier-based plants are currently in operation Gasifier-based power plants are not cost competitive, which limits the applicability of precombustion CO₂ capture
Oxy-combustion	Very high CO₂ concentration in flue gas Possibility exists to retrofit and repower plants	Large cryogenic O₂ production requirement Cooled CO₂ recycle required to maintain temperatures within limits of materials in furnace Need for oxygen plant and CO₂ recycle increases auxiliary power load and decreases plant efficiency

Processes have also been developed that use a potassium carbonate (K₂CO₃) solution, such as UOP's Benfield™ process. Carbonate-based systems work by the conversion of carbonate to bicarbonate in the presence of CO₂, followed by the conversion back to carbonate by heating with release of the absorbed CO₂, as indicated by the following reaction:

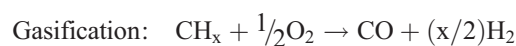


A major advantage of carbonate systems over amine-based systems is the significantly lower energy required for regeneration. Additives are sometimes included to improve absorption rate or provide other benefits.

In contrast to physical solvents, the capacity of a chemical solvent does not depend strongly on CO₂ partial pressure. Rather, it depends mainly on the concentration of the sorbent and the completeness of regeneration. Amine solvents are typically limited to concentrations of about 30% because of corrosion problems at higher concentrations. Another problem with chemical solvents is that they can form stable compounds with impurities in the flue gas and, thus, gradually lose their effectiveness. There is also a certain amount of degradation that occurs from oxygen in the flue gas. Furthermore, amine-based units have yet to be built at the scale necessary to scrub CO₂ from the flue gas of a large power plant. Some cost reductions are expected from continuing work to increase capacity and stability and reduce energy requirements.

PRECOMBUSTION CAPTURE

In precombustion capture, the fuel is decarbonized before it is burned. This typically involves gasifying the fuel, usually coal, by contacting it with a mixture of oxygen and steam to form synthesis gas (referred to as syngas), a mixture of H₂ and carbon monoxide (CO), along with smaller amounts of other gases, such as H₂O, CO₂, hydrogen sulfide (H₂S), ammonia (NH₃), and methane (CH₄). Gasification is the first step in a power-producing process known as integrated gasification combined cycle (IGCC), as shown schematically in the second row of Fig. 1. The syngas is subjected to the water–gas shift (WGS) reaction to convert CO into CO₂ plus H₂. Typical reactions occurring are as follows:



Other reactions also occur, but the above mentioned are the most important. After CO₂ removal, the H₂ is burned as a fuel in a combustion turbine and produces only H₂O. Since H₂ has different properties than CH₄ or CO, the combustion turbine must be redesigned to burn a high H₂ content fuel.

There are several advantages to this approach. First, gasifiers can operate at moderate pressures (500–1000 pound/in.²). This permits the use of physical solvents for CO₂ capture, rather than the more expensive chemical solvents. The most popular acid gas removal processes based on physical solvents are Rectisol[®], which uses methanol as a solvent, and Selexol[™], which uses a mixture of dimethyl ethers of polyethylene glycol. Second, because the gasifier runs at reducing conditions (as opposed to oxidizing conditions in a furnace), sulfur in the fuel is converted to H₂S rather than SO_x, and the nitrogen in the fuel is converted to NH₃ rather than NO_x. H₂S can be removed by the same physical solvent used for CO₂ removal, using a two-stage process, and converted to elemental sulfur, a saleable product. NH₃ can be scrubbed out with a water wash. Thus, with these systems, it is easier to remove sulfur and nitrogen in the form of H₂S and NH₃ than with systems requiring the removal of SO_x and NO_x. The biggest disadvantage of IGCC is that an oxygen plant is required, which significantly increases costs.

OXY-COMBUSTION

Oxy-combustion involves burning the fuel in an oxygen-enriched environment, where pure oxygen is diluted with recycled flue gas or steam so that the flue gas is almost entirely CO₂ and H₂O. When the H₂O is condensed, a nearly pure CO₂ stream is left. Depending on specifications and regulations, it may be necessary to remove trace contaminants, such as SO_x and NO_x, from the CO₂ before it can be injected into a pipeline for transport to a site for enhanced oil recovery or storage in a geologic formation. If CO₂ cleanup is required, costs for the oxy-combustion option will be somewhat increased.

Because combustion in pure oxygen would result in much higher furnace temperatures than that can be tolerated by materials of construction, flue gas recycle is necessary, which increases costs. Another disadvantage of oxy-combustion is the requirement for an oxygen plant. In the precombustion option, only part of the fuel is burned in pure oxygen, but in oxy-combustion, all the fuel is burned in pure oxygen, so a larger oxygen plant is needed. Oxy-combustion has the potential for higher efficiency, when compared to air-fired combustion, if boiler materials that can withstand higher temperatures are developed.

EMERGING CAPTURE TECHNOLOGIES

If a new pulverized coal power plant were built incorporating the state-of-the-art CO₂ capture technology, the cost per tonne of CO₂ avoided would be about \$75, and the cost of electricity would increase by over 80%, with considerably higher increases possible for plants retrofitted with CO₂ capture systems, when the cost of replacement power is included. CO₂ capture is by far the most expensive step in CCS, accounting for about 75% of the cost. Therefore, for large-scale implementation of CCS to be practical, more cost-effective CO₂ capture technology is needed. Fig. 2 depicts some of the emerging technologies and the approximate time frame when they may be available. Some of these technologies are discussed below.

Aqueous NH₃

NH₃-based systems behave similarly to amine-based systems, but have several advantages, including significantly lower heat of regeneration, higher CO₂ capacity, lack of

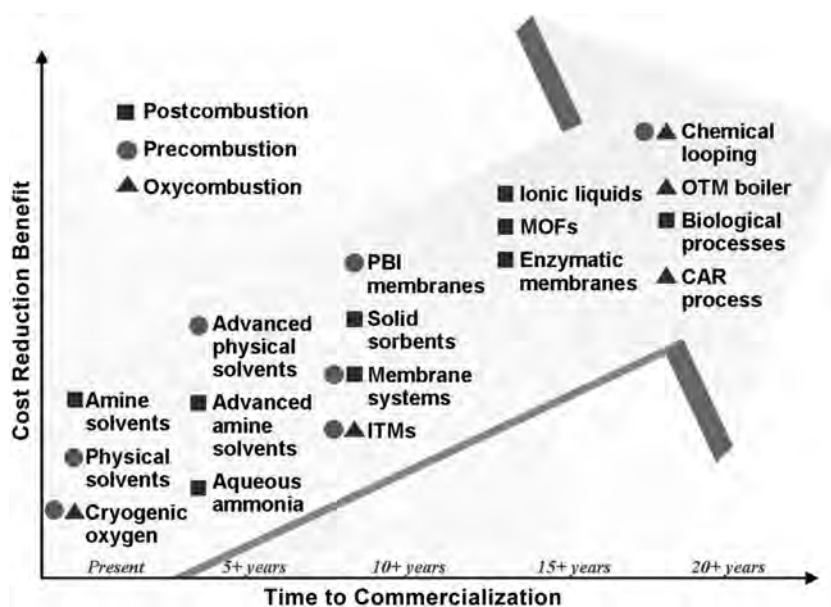
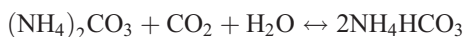


Fig. 2 Schematic diagram showing cost reduction benefits of innovative CO₂ capture technologies versus time to commercialization. ITM, ion transport membrane; CAR, ceramic autothermal recovery; OTM, oxygen transport membrane.

Source: From Figueroa, Fout, et al.^[1]

degradation during absorption/regeneration, and low cost. The chemical reaction is as follows:



There is also a potential for the simultaneous absorption of SO_x and NO_x to form ammonium sulfate and ammonium nitrate, which can be sold as fertilizer.

A major concern with NH_3 is its high volatility. This may require cooling the flue gas to the temperature range of 60–80°F to reduce NH_3 losses and then reheating it after CO_2 removal, which would increase costs.

Membranes

Another option under investigation is the use of various kinds of membranes to recover CO_2 . As a way of separating CO_2 from flue gas, membranes have the same problem as physical solvents—a low driving force (the difference in CO_2 partial pressure on the two sides of the membrane). Since the partial pressure of CO_2 in a typical flue gas is considerably below 1 atm, a vacuum is required on the permeate side. Compressing CO_2 from subatmospheric pressure to 2000 pound/in.² (a typical pressure for geologic storage) is expensive.

Because of the higher partial pressure of CO_2 , the use of membranes with precombustion capture may be more promising. Fig. 3 depicts the results from a polybenzimidazole (PBI) membrane under development at U.S. Department of Energy's Los Alamos National Laboratory. This membrane has shown hydrothermal stability up to 400°C and sulfur tolerance when operating on simulated coal-derived syngas, and its performance exceeds the Robeson upper bound for H_2/CO_2 selectivity versus permeability.

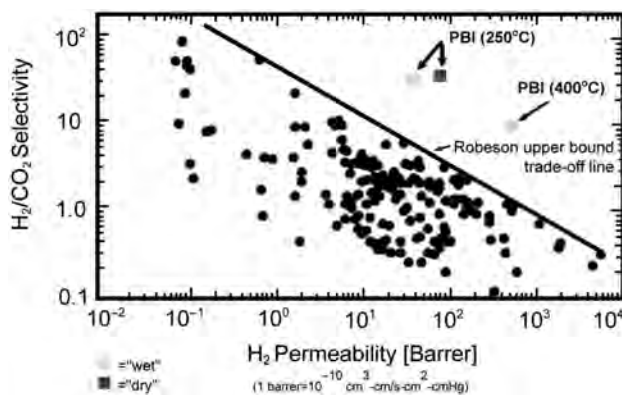


Fig. 3 Trade-off plot between H_2 permeability and H_2/CO_2 selectivity for polymers. The PBI membrane shows excellent performance.

Source: From Berchtold.^[2]

Solid Sorbents

Solids can act as physical sorbents (e.g., molecular sieves) or form a chemical compound, which is regenerated by heat. Many CO_2 solid sorbent processes under development tend to mirror liquid sorbent processes by being either amine based or carbonate based. Amine-based systems use an amine that is stabilized by being absorbed on a porous substrate. After a CO_2 absorption cycle, the sorbent is heated to release absorbed CO_2 . A carbonate system being investigated is sodium carbonate, which reacts to form sodium bicarbonate. The bicarbonate is regenerated to carbonate by heating.

Solid-based systems are inherently more complex than liquid-based systems because solids are more difficult to handle. Two approaches are possible: 1) moving (or fluidized) beds for the solids and 2) fixed beds with periodic gas flow switching. Because of the large volume of flue gas from a typical coal-fired power plant, equipment sizes will be large in either case.

A novel solid sorbent being investigated is metal organic frameworks (MOFs). These MOFs are a new class of hybrid material built from metal ions with well-defined coordination geometry and organic bridging ligands. They are extended structures with carefully sized cavities that can adsorb CO_2 . High CO_2 capacity should be possible, and the energy required for regeneration is low.

Novel Liquid Sorbents

Novel liquids being investigated as potential sorbents for CO_2 include ionic liquids (ILs). ILs are a broad category of salts, typically containing an organic cation and either an inorganic or an organic anion. They can dissolve CO_2 and are stable at temperatures up to several hundred degrees centigrade, thus offering the possibility of being able to recover CO_2 from flue gas without having to cool it first. Since ILs are physical solvents, little heat is required for regeneration.

Chemical Looping

Chemical looping is a variant of oxy-combustion in which the oxygen is provided by an oxygen carrier, such as a metal oxide, rather than by an oxygen plant. The fuel is contacted by the metal oxide in a combustor, where the fuel is oxidized to produce a flue gas, which is predominantly CO_2 and H_2O , while the metal oxide is reduced. After reduction, the reduced metal is contacted with air to regenerate the oxide. This reaction is exothermic, and the hot nitrogen leaving the oxidation stage is used to raise steam for power production. CO_2 is recovered by condensing the water in the flue gas. A possible embodiment of the concept is to use two fluidized beds (similar to fluid catalytic cracking units used in petroleum refining), oxidation occurring in one vessel and reduction in the other. After heat recovery

to raise steam, the oxygen-depleted air is exhausted to the atmosphere. This approach is a way to practice oxy-combustion without requiring an oxygen plant.

Improved Auxiliary Processes

If power plants are required to reduce CO₂ emissions and CCS is adopted as a CO₂ mitigation strategy, it seems likely, for the reasons discussed earlier, that new plants will be based on IGCC or oxy-combustion technology. However, deployment of these technologies must overcome high capital costs that affect commercial competitiveness and the ability to raise capital. One way to improve efficiency and economics for IGCC and oxy-combustion is to develop improved auxiliaries, such as air separation and WGS technology. One possibility is the use of tubular membranes; e.g., tubes made of an oxygen transport membrane could be inserted into a furnace. When air is blown through the tubes, oxygen diffuses into the furnace and combusts the fuel, thus achieving oxy-combustion without the need for a cryogenic air separation unit.

Catalyst-filled membrane tubes could be used in a WGS reactor to remove one of the products (H₂ or CO₂) from the reaction, thus allowing complete reaction in one reactor, instead of requiring multiple reactors operating at successively lower temperatures.

Another area where improvements are possible is the compression of CO₂. Studies are underway to determine the most efficient compressor design for compressing CO₂ to about 2000 pound/in.^[2], including looking at Ramjet technology. The option of compression to an intermediate pressure, followed by liquefaction and pumping to 2000 pound/in.², is also being evaluated.

CONCLUSION

CCS is a promising technology for reducing CO₂ emissions from fossil-fuel-fired power plants. However, the state-of-the-art technology is expensive and significantly reduces plant efficiency. Furthermore, this technology has not been implemented at the scale necessary to treat the flue gas from a large power plant. Thus, if CCS is to have an impact on mitigating global climate change effects, improved capture and auxiliary processes will need to be developed. Further information on CO₂ capture is provided in the references.^[1,3–5]

REFERENCES

1. Figueroa, J.D.; Fout, T.; Plasynski, S.; McIlvried, H.; Srivastava, R.D. Advances in CO₂ capture technology—The U.S. Department of Energy's carbon sequestration program. *Int. J. Greenh. Gas Con.* **2008**, *2*, 9–20.
2. Berchtold, K. Novel polymeric-metallic composite membranes for CO₂ separation at elevated temperatures. In *Presented at 5th Annual Conference on Carbon Capture and Sequestration*, Alexandria, VA, May 8–11, 2006, Paper #176.
3. White, C.M.; Strazisar, B.R.; Granite, E.J.; Hoffman, J.S.; Pennline, H.W. Air & Waste Management Association. Separation and capture of CO₂ from large stationary sources and sequestration in geologic formations-coalbeds and deep saline aquifers. *J. Air Waste Manage.* **2003**, *53*, 645–715.
4. Wilson, E.; Gerard, D., Eds. *Carbon Capture and Sequestration*; Blackwell Publishing: Oxford, 2007.
5. International Panel on Climate Change. *Carbon Dioxide Capture and Storage*; University of Cambridge Press: Cambridge, 2006.

Precision Agriculture: Engineering Aspects

Joel T. Walker

Department of Food, Agricultural and Biological Engineering, Ohio State University, Columbus, Ohio, U.S.A.

Reza Ehsani

Ohio State University, Columbus, Ohio, U.S.A.

Matthew O. Sullivan

Department of Food, Agricultural and Biological Engineering, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

It seems possible that all agricultural operations could be monitored and recorded, linked by digital transmissions to databases containing weather, remote sensing, and historical data, and controlled through a general model of the crop's predicted response to specific inputs. Such a system may not only control specific agricultural operations but also be involved in scheduling, ordering seed, fertilizer, and supplies, providing records to regulating agencies, and feeding valuable information back into the research system for further refinement of the control model.

INTRODUCTION

Information technology is playing an increasingly important role in agricultural production systems of all sizes, commodities, and management philosophies. Precision agriculture^[1–3] or site-specific management is an information-based management technique that has the potential to improve profitability^[4] and reduce the environmental impact^[5] of crop production. It also has the potential to improve the quality and nutrient content of the product. Precision agriculture, rather than the “one-size-fits-all” management strategy, provides for differential treatment of selected areas of a production field, called management zones, based upon the expectation of increased yield, profit, or some other agronomic goal.^[6–9] Management zones may be selected for differential treatment based upon various documented differences such as soil type, soil fertility or pH, yield history, presence of weeds, insects, or diseases, or other measures for which a differential treatment helps the producer achieve a selected goal. The ability to provide differential treatment to management zones, also called site-specific management, depends upon the availability of both proper equipment and effective treatment algorithms.

WHAT MAKES IT POSSIBLE?

Precision agriculture techniques have been made possible by the advent of global positioning system (GPS) and high-speed computer processing. GPS provides real-time

location information to a computer that, from stored information, determines the management zone, selects appropriate treatment for that management zone, and controls mechanisms to provide the treatment. Fig. 1 is a graphic representation of the precision agriculture paradigm. GPS provides position information for a variety of data gathering processes or for the control of site-specific treatments. The information of various types (shown as layers) may be used in analysis of yield results or to develop an application map to control site-specific treatments. The whole system taken together is often called precision agriculture or site-specific agriculture. Note that a feedback loop is implied where results of the previous growing season (yield) become part of the information that influences the treatment practices. Given the proper treatment algorithms, the treatment practices may optimize the goal parameter. Maximum yield is not necessarily the best goal because the cost of treatments required to achieve that yield may be greater than the increased crop value.^[8,9]

GPS consists of a minimum of 24 satellites circulating around the earth sending signals to a local receiver for which location is desired.^[10] Each satellite broadcasts encoded information with particular timing. By measuring the time a signal travels (at the speed of light) to reach the receiver, the distance from a satellite to the receiver may be calculated. Determining distances from four or more satellites of known location may establish the receiver's location (latitude, longitude, and elevation). Even simple, inexpensive receivers are capable of accuracy better than 15 m, close enough to return to a favorite fishing spot. With

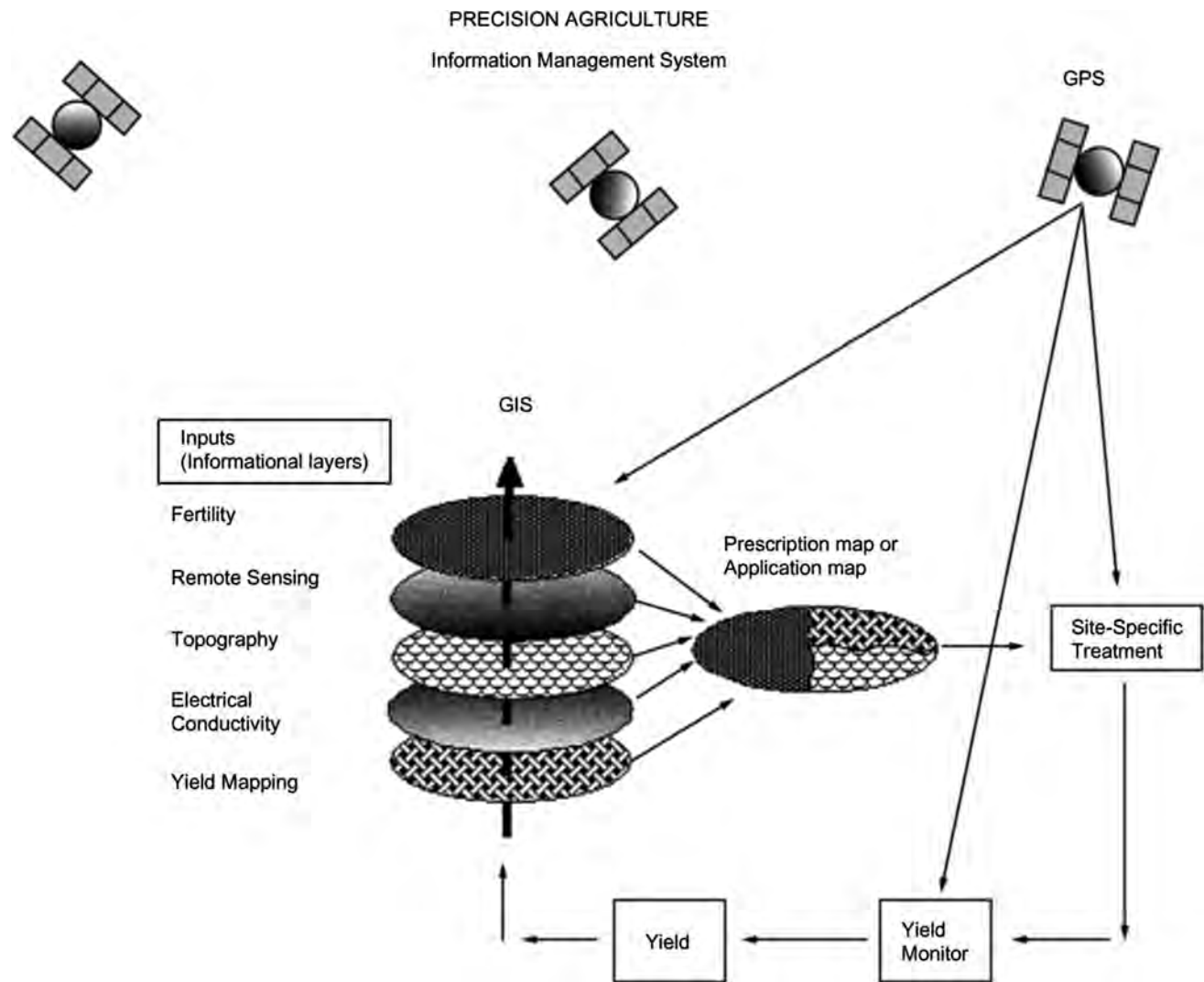


Fig. 1 Paradigm for precision agriculture.

specialized transmissions containing information to correct for known errors (called differential signals), accuracy better than 1 m may be achieved. Specialized local transmitters make possible real-time kinematic (RTK-GPS) systems with an accuracy of better than 1 cm. RTK systems are used in surveying and guidance, where the fine precision may justify a relatively high equipment cost.

High-speed computer processing systems have also played an important role in the advent of precision agriculture. Precision agriculture requires collection and storage of data, decision-making computation, and controlling of equipment by computers operating at billions of operations per second. GPS locations are recorded in real time by the computer. Digital maps of field conditions and parameters are carried in memory or storage media. Digital data such as digital still or video images of weeds, insects, or disease damage, soil properties, crop spectral reflectance, or climatic conditions may also be collected in real time by the computer. Using such information, the computer may work

through a response model to decide what actions should be taken with existing equipment. For example, a precision agriculture capable planter may be able to adjust planting rate and depth or change the seed variety on the go. Thus, the computer may decide that areas with a selected soil type and relative elevation will get a reduced population (investment of seeds) because less yield is expected. The computer may dictate that another soil type with a high yield potential and high existing moisture content will get a different variety planted to a shallower depth. Across an entire field, many combinations of the controlled variables will be chosen to optimize the desired result (yield, profit, or other goal). A large field would require constant decisions and adjustment of the equipment that would not be possible without computerized systems. Finally, volumes of data may be produced by precision agriculture techniques, and computer methods are being developed to extract useful information; models form these data^[11–16] and present them to the public in a readily accessible form.^[17]

SENSING FOR PRECISION AGRICULTURE

Real-time sensors will be important to many precision agriculture systems.^[18] A real-time sensor provides data in a nearly constant stream as the machine traverses the field. For example, a camera may provide digital images from which the presence of weeds may be determined. A sprayer may then be directed to spray only where weeds are present. Organic matter and moisture levels in soil affect the performance of some herbicides. Sensors that measure organic matter and/or moisture on the go allow for optimum rates of chemical application—adequate to control the weeds but no more than necessary to preserve environmental quality.

One of the most popular real-time sensors measures grain yield on combines.^[11] Yield sensors are also available or are being developed for a variety of crops such as cotton, potatoes, tree fruits, and strawberries. Yield sensors provide a measure of yield over a whole field. Areas of the field with unusually high or low yield may be identified, and corrective action (subject to some predictive model created by the producer or by computer) can be taken for the following year. This one aspect of precision agriculture has created a lot of activity in adjusting soil drainage and fertility and crop management decisions such as the relative value of fertilizer or chemical inputs.

For many agronomic parameters, site-specific soil sampling is more practical, either because a real-time sensor is not yet available or because the spatial variation of the parameter is more gradual and can be estimated with a few site samples. Soil samples to determine fertility are common. The values of pH (acidity/alkalinity), nitrogen, potassium, and phosphorus are particularly important in the prediction of yield levels. Soil type is another parameter that is often determined by a site visit.

Remote sensing is becoming increasingly important to precision agriculture. Early data that came from random satellite observations had low resolution and limited value. Now specially equipped satellites and aircraft may be hired to collect specific crop or soil information. Crop growth or health may be deduced from these data. Certain wavelengths of the electromagnetic spectrum are particularly helpful. The visible light frequencies provide some information. Unhealthy crops tend to reflect more yellow and red light. Various frequencies in the infrared range also have been correlated with plant health and soil moisture conditions. Multispectral systems provide data on the visible spectrum (often three primary colors) and a limited range of the infrared. More sophisticated equipment, called hyper-spectral, can provide data from a much broader range of the spectrum and from narrower sample bands. This type of data offers greater opportunity to correlate specific crop or soil conditions to measured spectral data.

Other types of remote sensing are becoming available. Light detection and ranging (LIDAR), e.g., is a laser

ranging technique that may be used to measure topography and plant height simultaneously.^[19] By applying to the forest industry and relatively expensive, this method holds promise for rapid feedback of information on crop growth problems to the producer, perhaps allowing solution of the problem before yield is permanently affected. For example, nematodes in soybeans are a common cause of reduced vigor and yield if left untreated. Conventional crop scouting or remote sensing may not detect small areas of infestation or provide feedback in time to take corrective action. With regular LIDAR imaging, a producer could detect and treat such trouble spots in a timely and efficient manner.

GEOGRAPHIC INFORMATION SYSTEMS (GISs)

GIS techniques are an integral part of precision agriculture. Basically, GIS is a storage system for geographically referenced digital data. Many of the data types discussed above can be digitized (if not already in digital form) and referenced to specific locations in a field. Each parameter or variable then may be represented as a layer of information. Geographic position of the information matches the position on a map of the field. The value of a parameter may be represented on a map as a shade of gray or a color. So a GIS information layer representing soil type can be drawn in the physical shape of the field with patches of color—each color representing a different soil type. Many such layers may exist in a GIS data set for a particular field. Depending upon the need, these layers may be viewed as overlays—simultaneous presentation of several variables in one field. As many layers of information are added to the GIS system, it becomes apparent that human vision is inadequate to detect the important patterns. Computers, however, are infinitely more able to “see” data patterns with a potential to produce an economic advantage.

FUTURE

The future of precision agriculture depends on economic results. Can the cost of additional equipment and operations be more than offset by increased economic return and value of environmental protection? It seems possible that all agricultural operations could be monitored and recorded, linked by digital transmissions to databases containing weather, remote sensing, and historical data, and controlled through a general model of the crop’s predicted response to specific inputs. Such a system may not only control specific agricultural operations but also be involved in scheduling, ordering seed, fertilizer, and supplies, providing records to regulating agencies, and feeding valuable information back into the research system for further refinement of the control model. Already, manufacturers are producing prototype agricultural machines that perform without an operator. These machines are guided by GPS, are controlled by

computer, and have onboard sensors to detect obstructions or people in the way. The advancement of agriculture will come with technology—technology to feed the world.

REFERENCES

1. Morgan, M.; Ess, D. *The Precision-Farming Guide for Agriculturalists*; John Deere Publishing: Moline, 1997.
2. Pierce, F.J.; Sadler, E.J. *The State of Site-Specific Management for Agriculture*; ASA/CSSA/SSSA: Madison, 1997.
3. Sonka, S.T. *Precision Agriculture for the 21st Century: Geospatial and Information Technologies in Crop Management*; National Academy Press: Washington, 1998.
4. Erickson, K. *Precision Farming Profitability*. Agricultural Research Programs; Purdue University: West Lafayette, 2000.
5. Wang, X.; Tim, U.S. *Problem-Solving Environment for Evaluating Environmental and Agronomic Implications of Precision Agriculture*. Available at <http://www.esri.com/library/userconf/proc00/professional/papers/PAP102/p102.htm> (accessed April 2001).
6. Adams, M.L.; Cook, S.; Corner, R. Managing uncertainty in site-specific management: What is the best model? *Precis. Agr.* **2000**, *2*, 39–54.
7. Boote, K.J.; Jones, J.W.; Pickering, N.B. Potential uses and limitations of crop models. *Agron. J.* **1996**, *88*, 704–716.
8. Bullock, D.S.; Bullock, D.G. From agronomic research to farm management guidelines: A primer on the economics of information and precision technology. *Precis. Agric.* **2000**, *2*, 71–101.
9. Swinton, S.M.; Lowenberg-Deboer, J. *Site-Specific Management Guidelines—Profitability of Site-Specific Farming*; Publication SSMG-3; Phosphorus and Potash Institute, 1998.
10. Kennedy, M. *The Global Positioning System and GIS: An Introduction*; Ann Arbor Press: Chelsea, 1996.
11. Data Mining for Site-Specific Agriculture. Available at <http://www.gis.uiuc.edu/cfardatamining> (accessed April 2001).
12. Data Mining Puts Geographic Information to Work for Public, Private Sectors. Available at <http://csd.unl.edu/csd/resource/Vol.13/datamine.htm> (accessed April 2001).
13. Kerschberg, L.; Lee, S.W.; Tischer, L. *A Methodology and Life Cycle Model for Data Mining and Knowledge Discovery for Precision Agriculture*; Final Report; Center for Information Systems Integration and Evolution, George Mason University: Fairfax, 1997.
14. Lazarevic, A.; Fiez, T.; Obradovic, Z.A. *Software System for Spatial Data Analysis and Modeling*; Report to INEEL University Research Consortium Project No. C94-175936, 2000.
15. Precision Farming Support. Available at <http://www.umac.org/farming/precision.html> (accessed April 2001).
16. Regression and Emerging Data Mining Tools Modeling. Available at <http://www.gis.uiuc.edu/cfardatamining/APR/Report.htm> (accessed April 2001).
17. Ohio State Precision Agriculture. Available at <http://www.ag.ohio-state.edu/~precisfm/> (accessed April 2001).
18. Gaultney, L.D. *Prescription Farming Based on Soil Property Sensors*. ASAE Paper No. 89-1036, ASAE Annual International Meeting; ASAE: St. Joseph, 1989.
19. Ackermann, F. Airborne laser scanning—Present status and future expectations. *ISPRS J. Photogram. Remote Sens.* **1999**, *54* (2–3), 64–67.

Precision Agriculture: Fertilization

Silvia H. Haneklaus

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

Soils are neither static nor homogenous in space and time, and this directly influences the concentration of plant-available nutrients and, consequently, the small-scale spatiotemporal variation of the fertilizer demand. Precision agriculture comprises the technology to determine the spatial variation of soil and plant features and to transfer this information into site-specific field operations. Variable rate fertilization within precision agriculture is applied to match spatially distributed nutrient pools and fluxes with fertilizer rates. In comparison, a uniform treatment of fields causes inevitably nutrient surpluses and undersupplies and thus is neither ecologically sound nor economically viable.

INTRODUCTION

The fertilizer rate is usually calculated from the balance between the natural supply by soil and environment, the nutrient demand of the crop, and the inevitable losses to the environment. The agricultural production system fulfills the criteria of sustainability if the losses in this balance can be minimized while maximizing crop productivity and economic profits. The major obstacles to achieving this ideal case of adjusting nutrient balances are, first, the accuracy of the methods employed to quantify the contribution from the soil, to determine the physiological requirement of the plant, and to calculate the losses to the environment; second, to address the spatial variability of soil fertility parameters and to match it with variable fertilizer rates; and, third, to warrant the profitability of implementing the technology.

GEOREFERENCED SOIL AND CROP INFORMATION

Access to validated georeferenced data is a major obstacle to the further development of precision agriculture. The applicability of different data sources mainly depends on the scale of farming. Big farms usually operate on a distinctly higher level of technology than small farms. Thus, big farms rely on access to digital data covering large areas. In comparison, on small farms, local knowledge is an important source of information, providing data in the form of amateurish maps, field files, and any other materialized form of store.^[1]

For variable rate fertilization, only maps with a scale of >1:5000, preferably in digital form, provide suitable information.^[2] Particularly in undulated areas, terrain

information is an important data source, and digital terrain models may be processed from elevation data or can be assessed directly on the farm.^[1] Experiences from grid soil sampling reveal that it is superior to random sampling because of the spatial correlation of values.^[3] Relevant in this context is that optimum sampling distances vary in different landscapes, depending on the scale of variation of field characteristics as a result of parent material and pedogenesis.^[4] The disadvantage of grid sampling is that it is time-consuming, labor-consuming, and costly. Strategies that reduce sampling efforts are, for instance, directed sampling and self-surveying.^[2]

In principle, remotely sensed data offer a fast and economic way to optimize soil and plant sampling strategies and to design variable field management. For variable rate fertilization, it is vital that the difference between nutrient demand and actual nutrient supply be determined in time, which is a major handicap because of the problem of cloud coverage at phenological relevant growth stages, the restricted revisit time of satellites, and the physical availability of airplanes. This would require the real-time surveillance of crops. Low-Altitude Stationary Surveillance Instrumental Equipment is an innovative concept for the continuous recording of real-time images of crop and soil surfaces, which are automatically rectified and georeferenced in a geographic information system.^[5]

On-the-go sensors for assessing the spatial variation of soil water, nitrate, organic matter, pH, electrical conductivity, and soil texture exist.^[6] One of the most promising developments in generating data for variable rate fertilization is the yield sensor because nutrient removal from a field is an important information for making decisions for variable rate fertilization. Another major field of application is the evaluation of the plant nitrogen status; here,

fundamental objections exist because of principal incorrect assumptions.^[1]

DECISION-MAKING STRATEGIES

Recommendation Schemes

Common rules for fertilizer application have been proposed for decision making and were directly transferred into fertilizer rates. However, there are two major criticisms of such approach. First, the basic fertilizer response curves were mainly established in the 1940s–1960s, and databases need to be updated and validated.^[7] Second, established critical thresholds refer to small homogenous plots and are counterproductive with view to the target, which is to identify variability, to match it with a variable input, and to improve recommendations.^[7] Additionally, soils and plants were sampled in the past without reference to spatial autocorrelation, so that the whole system of interpretation used for recommendation is based on averages, too.^[8]

On-Site Experimentation

Using precision agriculture technologies for positioning, variable rate application, and on-line yield measurements, it is possible to investigate multivariate experimental designs on each farm including the whole bandwidth of ecological variability.

One of the first and most crucial steps in decision making is the ranking of the nutritional minimum factors of a site. A suitable approach to identify yield-limiting factors in the mineral nutrition of a particular site is the directed sampling in high-yielding and low-yielding areas.

Ideally, fertilizer rates are calculated on the basis of response curves. The boundary line development system is a suitable tool to establish site-specific response curves from georeferenced yield and soil/plant data.^[1]

ACCURACY OF TECHNICAL EQUIPMENT AND DATA

In general, the risk that a mismatch of information and field operations causes adverse effects increases the smaller in the distance between two significantly different spatial events. A great problem to define quality criteria for variable rate fertilization is variability, which is searched for, so that erroneous variability can easily be hidden.

Data summarized in [Table 1](#) reveal that the accuracy of georeferenced information should no longer be a problem. The guaranteed quality standards for purchasable information are also satisfactory. The greatest source for errors will still be failures by the operator. Satellite-aided navigation of fertilizer spreaders meets the same accuracy than positioning (see [Table 1](#)).

Table 1 Positional accuracy of different data sources and technical equipment for variable rate fertilization.

Source	Positional accuracy (m)
Satellite positioning	
Standard	2–3
Differential	<1
With S/A	25
Ordnance survey maps	
Maps (scale 1:5000)	3
Digital elevation data	0.5
Soil survey	
Digitizing of map features (sampling points)	3
Boundaries of choropleth maps	15–20
Remote sensing images	10–20
Yield maps	10–15

Source: From Capelle.^[2]

The precise match of position and variable rate with the data of the fertilizer map is more problematic. The error during variable rate application of fertilizers has several main components such as the error of the navigation system, the hysteretic effect between the time where the system is advised to apply a certain amount of fertilizer,^[9] unevenness in the lateral distribution pattern,^[10] and inhomogeneities in the fertilizer material itself.^[11] Depending on the fertilizer used and the application system employed, more or fewer errors can occur. In general, the relative size of this error increases in the following order: liquids < solids; manufactured fertilizers < farm sources; slurries < manures; pneumatic spreaders < spin disk spreaders; and irrigation < spreading.^[10,11] The great number of error sources and non-spatial variability supports the hypothesis that successful variable rate fertilization requires the development and implementation of quality standards and control measures to enabling the secure implementation of the concept for farmers.

Saved fertilizer inputs or prevented environmental costs will not pay for the implementation of precision agriculture technologies.^[1] On average, fertilizer savings ranged from 10% to 20%, but yield increases varied only between 3% and 5%.^[1] The intrinsic hypothesis for the environmental friendliness of variable rate fertilization is evident because less fertilizer is applied to areas with increased vulnerability to nutrient losses. Only a few experimental proofs with direct measurements are available. Whitley et al.,^[12] for instance, found that the variable rate nitrogen fertilization of potato increased nitrate fluxes in the surface soil by 33%, whereas those in the subsurface soil were not affected. At the same time, the nitrate content in the soil was lower when rates were applied variably, resulting in an increased nitrogen utilization efficiency of the crop.

CONCLUSION

Agriculture was associated positively with food production and food security less than a century ago. It is commonly linked to problems such as the eutrophication of water bodies and enrichment of pesticides in soils and plants. Precision agriculture technologies could be an important step to regain the confidence of consumers by limiting the input of fertilizers and agrochemicals to the minimum requirements. This minimum equals fertilizer rates that are sufficient to maintain the site-specific crop productivity, whereas nutrient losses come close to those in low-input systems such as organic farming. Last, but not least variable, rate fertilization needs to be profitable, but reduced fertilizer expenditures and yield increases will not justify the costs for implementing the technology. Therefore, other profitable applications such as weed control measures and improved logistics and traffic on the field must pay for the change in the production system.^[1]

REFERENCES

1. Haneklaus, S.; Schnug, E. Site-specific nutrient management: Objectives, current status and future needs. In *Handbook of Precision Agriculture: Principles and Applications*; Srinivasan, A., Ed.; CRC Press: Boca Raton, 2006; 91–152.
2. Capelle, A. Die Eignung von Bodenkarten unterschiedlicher Massstaebe fuer die Erfassung der kleinraeumigen Heterogenitaet des Bodens. KTBL-Arbeitspapier. Erfass. Kleinraeumigen Heterogenitaet **1999**, 264, 42–46.
3. Sabbe, W.E.; Marx, W. Soil sampling: spatial and temporal variability. In *Soil Testing: Sampling, Correlation, Calibration and Interpretation*; SSSA Special Publication 21; SSSA: Madison, 1987; 1–14.
4. Burrough, P.A. Soil variability: A late 20th century view. *Soils Fert.* **1993**, 56, 529–562.
5. Schnug, E.; Haneklaus, S.; Lilienthal, H.; Panten, K. LASSIE—An innovative approach for the continuous remote sensing of crops. *Asp. Appl. Biol.* **2000**, 60, 147–154.
6. Viscarra Rossel, R.A. *Development of a Proximal Soil Sensing System for the Continuous Management of Acid Soil*. Ph.D. thesis, Australian Centre for Precision Agriculture, Department of Agricultural Chemistry and Soil Science, The University of Sydney: Sydney, 2001; 1–458.
7. Hergert, G.W.; Pan, W.L.; Huggins, D.R.; Grove, J.H.; Peck, T.R. The adequacy of current fertilizer recommendations for site-specific management in the USA. In *Proceedings of the 1st European Conference on Precision Agriculture*, Warwick, 1997; Vol. 1, 371–378.
8. Nowak, P. Agriculture and change: The promises and pitfalls of precision. *Commun. Soil Sci. Plant Anal.* **1998**, 29 (11–14), 1537–1541.
9. Griepentrog, H.-W.; Persson, K.A. Model to determine the positional lag for fertiliser spreaders. In *Proceedings of the 3rd European Conference on Precision Agriculture*, Montpellier, France, 2001; Vol. 2, 671–682.
10. Persson, K.; Bangsgaard, J. Accuracy of fertiliser distribution for site-specific fertilisation. In *Proceedings of the 2nd European Conference on Precision Agriculture*, Odense, Denmark, 1999; Vol. 2, 815–824.
11. Thirion, F.; Zwaenepol, P.; Chabot, F. Integrated weighing platform for manure spreader regulation. In *Proceedings of the 4th International Conference on Precision Agriculture*; ASA–CSSA–SSSA: Madison, 1999; 1219–1226.
12. Whitley, K.M.; Davenport, J.R.; Manley, S.R. Difference in nitrate leaching under variable and conventional nitrogen fertilizer management in irrigated potato systems. In *Proceedings of the 5th International Conference on Precision Agriculture*; ASA–CSSA–SSSA: Madison, 2001; CD-ROM.

Precision Agriculture: Nutrient Cycling

Robert J. Lascano

*U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
Lubbock, Texas, U.S.A.*

Abstract

Precision farming, also known as site-specific management, refers to the practice of applying agronomic inputs, mainly fertilizers and other chemicals, across a farm at variable rates based on soil nutrients or chemical tests, soil textural changes, weed pressures, and yield maps for each field in the farm. The technology for crop yield mapping is more advanced than methodologies for determining and understanding causes of yield variability. As the cost of agronomic inputs increase, producers will likely adopt variable rate strategies and manage their crops to increase crop yield and to maximize profitability.

INTRODUCTION

“Precision farming,” also known as site-specific management, refers to the practice of applying agronomic inputs, mainly fertilizers and other chemicals, across a farm at variable rates based on soil nutrients or chemical tests, soil textural changes, weed pressures, and/or yield maps for each field in the farm. In most large fields (e.g., >40 ha), crop yield is notoriously variable. The sources of this variation are related to the physical and chemical properties of the soil, pests, microclimate, genetic and phenological responses of the crop, and their interactions. The technology for crop yield mapping is more advanced than methodologies for determining and understanding causes of yield variability. Prevailing and traditional management practices treat fields uniformly as one unit. However, reports^[1–3] show that to understand underlying soil processes that explain crop yield variability, research must be done at the landscape level and using appropriate statistical tools for large-scale studies.^[1,3,4]

BACKGROUND

There is a linear relation between crop yield and water use when the only limiting factor is water;^[5] however, root uptake of water is synergistically related to nutrient uptake and the two processes cannot be separated. Precision farming has the potential to improve water and nutrient use efficiency on large fields provided that there is a quantitative understanding of what factors affect crop water and nutrient use and where in the field. It is known that crop water and nutrient use are a function of many biotic and abiotic variables—including managed inputs—and

that harvestable yield is a manifestation of how these variables and inputs interact and are integrated during the growing season. However, it is difficult to determine a hierarchy of the contribution of each input and variable to the measured yield using classical statistics.^[2,3] Often, variables that affect water and nutrient supply to the plant contribute to yield at a high level assuming an adequate plant stand and weed control. The cause and effect relation between a single state variable and crop yield is site specific and is difficult to establish without considerable sampling of the soil and/or crop. The establishment of response functions, i.e., crop water and nutrient use as a function of variable x_i , only gives a partial answer to explain crop water and nutrient use and yield based on inputs. The general idea of precision farming is to optimize input application to the measured crop yield at each sampling location. This is a simple premise; however, the decisions for variable rate application of any agronomic input must consider temporal and spatial variability of the soil's properties affecting crop growth, water and nutrient use, and yield. Soil factors that affect stored water, such as depth to root restricting layer and soil textural differences, must be considered in any precision farming operation that attempts to improve crop water use and yield related to agronomic inputs. Similarly, to improve the use of any soil macro- and micronutrient by the crop, the overall cycle of the nutrient must be considered, including its availability in the soil and demand by the crop.

Precision farming must incorporate the inherent spatial and temporal variability of soil physical, chemical, and biological factors within a field for input management. Accurate representation of spatial and temporal variability in a field requires taking and analyzing many samples. Sampling is normally done on a grid with a scale that can vary from one to several hundred meters.^[6] Once properties

are measured, geostatistical tools (e.g., semivariogram, kriging, and cokriging) and other spatial statistical tools (e.g., autocorrelation, crosscorrelation, and state-space analysis) can be used to establish statistical relations in space and to minimize the number of soil samples to characterize and map fields.^[2,3,7] The number of samples required *a priori* to determine spatial and temporal variability is perhaps the single largest deterrent in the application of precision farming practices to manage and improve crop water and nutrient use.

There is very little information published on crop water and nutrient use across large fields at the landscape level and in the context of precision farming.^[1,8,9] An exception is a study,^[1] where cotton water and total nitrogen use were measured along a 700 m transect with the objective to: 1) illustrate the landscape pattern of cotton water and total nitrogen use and 2) determine the underlying soil processes governing cotton lint yield variability. In this study, state-space analysis^[1,3] is used to formulate management decisions that may improve crop water and nitrogen use and, thus, the yield, using precision farming practices.

LANDSCAPE CROP WATER AND NITROGEN USE

The concept of crop water and nitrogen use in a large field is illustrated by the study of Li, Lascano, et al.^[1] In 1999, a field experiment was conducted near Lamesa, Texas, on a research farm of Texas A&M University on the southern edge of the High Plains of Texas. The soil was classified as an Amarillo sandy loam. The field was 60 ha with slopes ranging between 0.3% and 6.3%.^[1] To assess the effect of soil water, nitrate-nitrogen ($\text{NO}_3\text{-N}$), and topography on cotton lint yield across the landscape, two irrigation levels were used. The irrigation treatments consisted of water applications at the 50% and 75% potential evapotranspiration (ET) with a center pivot low energy precision application (LEPA) irrigation system.^[10] At each irrigation level, one transect was established following the circular pattern of the center pivot. The two transects were instrumented with 50 neutron access tubes that were 15 m apart from each other, and volumetric water content (θ_v) was measured periodically throughout the growing season. At each point θ_v was measured in 0.3 m depth increments to a depth of 2.0 m using a neutron probe calibrated for this soil. In addition, at each transect point soil texture, soil and plant $\text{NO}_3\text{-N}$, leaf area index, lint yield, slope, plant density, and other parameters were measured.^[1]

It has been shown that the use of classical statistics, such as regression analysis and analysis of variance, fails to completely explain the cause and effect between, e.g., crop yield and measured soil variables in precision farming experiments.^[1-4,11] Instead, there are other more appropriate statistical tools for relating the variability of

soil and plant parameters measured in space and time. For example, the structure of the spatial variance between measurements may be derived from the sample “semivariogram,” which is the average variance between neighboring measurements spatially separated by the same distance. Spatial structure between variables is often determined using “autocorrelation” and “crosscorrelation” functions. Autocorrelation measures the linear correlation of a variable in space along a transect. Crosscorrelation is the comparison of two variables measured along a transect and is used to describe the spatial correlation between two landscape variables, i.e., where one variable, the tail variable, lags behind the head variable by some distance. The spatial association between several variables can be described using “state-space analysis,” which is a multivariate autoregressive technique.^[1-4,7,11]

To illustrate the variability of crop water use or ET, values measured along the 50% irrigation transect were selected.^[1] In Fig. 1, the relation between the scaled ET and elevation, where both are given as a function of distance along the transect, is shown. The ET data are scaled to the maximum of 426 mm of water measured 210 m from the south end of the transect. These results show that higher ET was measured at lower elevations and that ET decreased at higher elevations. Spatial crosscorrelation between lint yield and soil water, lint yield and site elevation, and soil water and site elevation is shown in Fig. 2. For a 95% confidence interval, the cotton lint yield was positively crosscorrelated with soil

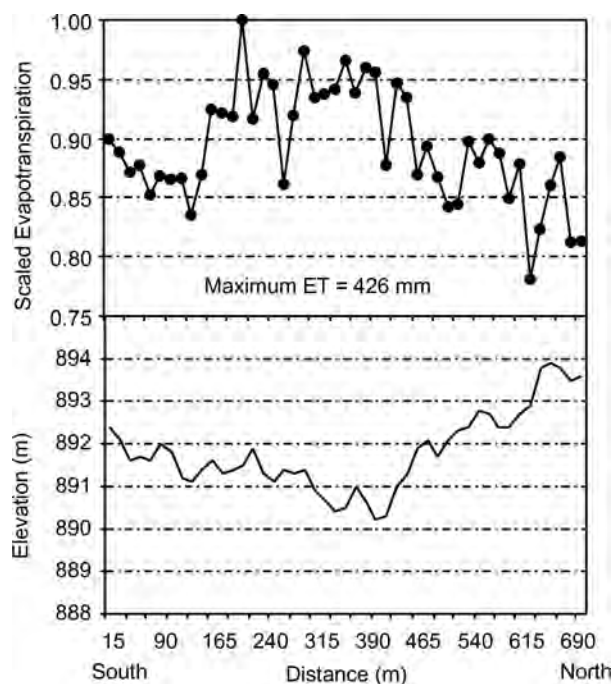


Fig. 1 Scaled evapotranspiration and elevation as a function of distance along a 700 m transect.

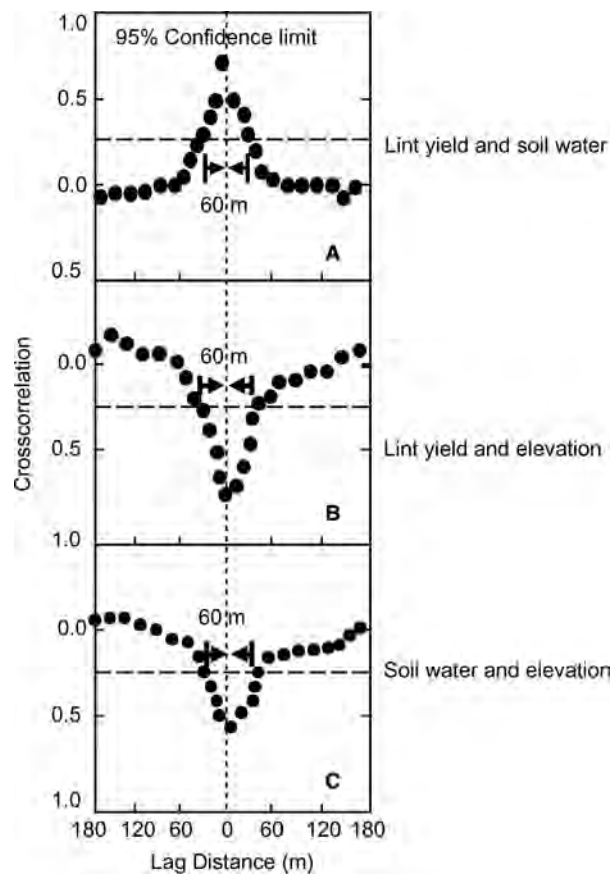


Fig. 2 Crosscorrelation as a function of lag distance: (A) Lint yield and soil water; (B) lint yield and elevation; and (C) soil water and elevation. Shown is the 95% confidence for the cross-correlation distance.

Source: From Li, Lascano, et al.^[1]

θ_v across a lag distance of ± 30 m. Lint yield and θ_v were negatively crosscorrelated with elevation at a lag distance of ± 30 m. These results show the effect of topography on the θ_v and crop water use measured along the transect. Similar results are given in other reports.^[1,8,9,11] In this example the crosscorrelation between θ_v and elevation shows the spatial structure of measured variables and further that more water was stored in lower elevations resulting in higher ET.

Linear regression analysis between θ_v and lint yield and relative site elevation is shown in Fig. 3, and the state-space analysis for the relation between lint yield and three measured parameters is shown in Fig. 4. Results in Fig. 3 show the shortcomings of using an inappropriate statistical tool to understand underlying processes explained with the state-space analysis. This analysis (Fig. 4) quantified how cotton lint yields varied as a function of distance and showed that by using θ_v , soil $\text{NO}_3\text{-N}$, and elevation the variation in lint yield can be explained with a high level of confidence.

Benefits of precision farming to improve crop water and nutrient use may be obtained by an economic analysis of

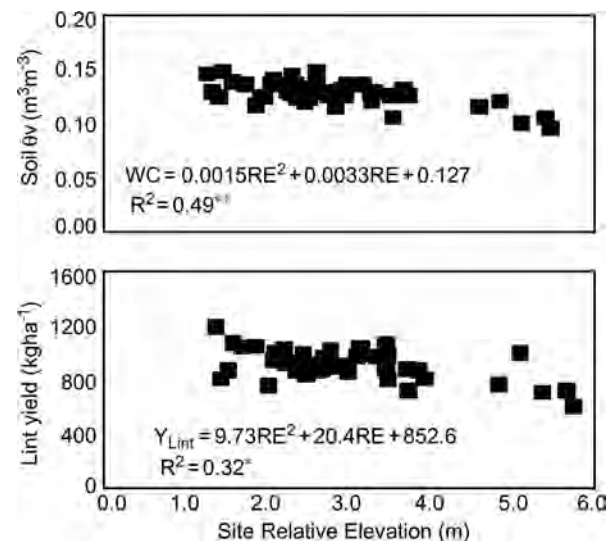


Fig. 3 Soil water content (θ_v) and cotton lint yield as a function of site relative elevation.

maximizing crop yield as a function of application of N fertilizer and irrigation water as given by the state-space equation. In the example given, the decision can be made to apply more N fertilizer to the lower areas of the field that also hold more water and increase crop water use and yield. With the introduction of variable rate planters, it will be possible in the near future to discriminate site locations and plant more “drought” tolerant varieties or change the seeding rate in areas that are prone to have less soil water. This implies the delineation of management zones within a field that are defined based on potential crop water and nutrient use and their interaction with other input variables to maximize economic yield across the field. This type of precision farming is not practiced but remains within the realm of possibilities that this type of farming has to offer.

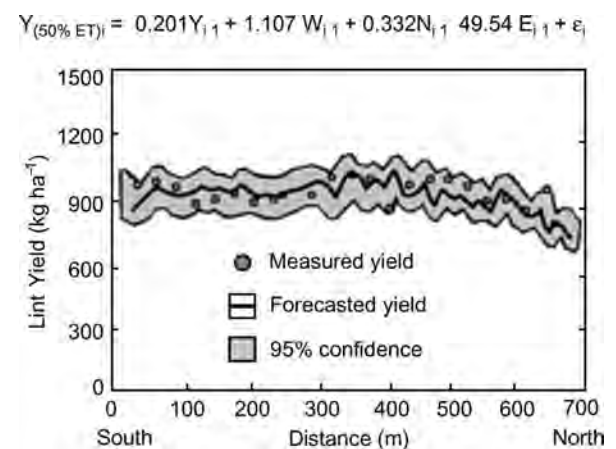


Fig. 4 State-space equation relating cotton lint yield (Y) to water content (W), nitrogen (N), and elevation (E) as a function of distance and location (i) along a 700 m transect.

Source: From Li, Lascano, et al.^[1]

A final consideration is the cost/benefit of precision agriculture practices and its impact on agriculture. Hardware for variable rate application of agronomic inputs is relatively expensive and in many cases unavailable; however, with increased adaptation and use of these practices the cost will be reduced. Further, environmental concerns for a given area will probably place limits on the amount of certain nutrients, e.g., N fertilizer, used for crop production. This will force producers to apply N and other nutrients across the field according to crop needs and position along the landscape. These practices will be beneficial from both an environmental and an economical point of view.

CONCLUSION

As the cost of agronomic inputs increase, producer's will likely adopt variable rate strategies and manage their crops to increase crop yield and to maximize profitability.

REFERENCES

1. Li, H.; Lascano, R.J.; Booker, J.; Wilson, L.T.; Bronson, K.F. Cotton lint yield variability in a heterogeneous soil at a landscape scale. *Soil Till. Res.* **2001**, *58*, 245–258.
2. Nielsen, D.R.; Wendroth, O.; Pierce, F.J. Emerging concepts for solving the enigma of precision farming research. In *Precision Agriculture, Proceedings of the Fourth International Conference*, Minneapolis, MN, Jul 19–22, 1998; Robert, P.C., Rust, R.H., Larson, W.E., Eds.; ASA, CSSA, SSSA: Madison, 1999; 303–318.
3. Wendroth, O.; Al-Oman, A.M.; Kirda, C.; Reichardt, K.; Nielsen, D.R. State-space approach to spatial variability of crop yield. *Soil Sci. Soc. Am. J.* **1992**, *56*, 801–807.
4. Cassel, D.K.; Wendroth, O.; Nielsen, D.R. Assessing spatial variability in an agricultural experiment station field: Opportunities arising from spatial dependence. *Agron. J.* **2000**, *92* (4), 706–714.
5. Kramer, P.J.; Boyer, J.S. *Water Relations of Plants and Soils*; Academic Press: San Diego, 1995.
6. Sadler, E.J.; Busscher, W.J.; Bayer, P.J.; Karlen, D.L. Spatial requirements for precision farming: A case study in the Southern U.S.A. *Agron. J.* **1998**, *90*, 191–197.
7. Shumway, R.H.; Stoffer, D.S. *Time Series Analysis and Its Application*; Springer Verlag: New York, 2000.
8. Halvorson, G.A.; Doll, E.C. Topographic effects on spring wheat yield and water use. *Soil Sci. Soc. Am. J.* **1991**, *55*, 1680–1685.
9. Hanna, A.Y.; Harlan, P.W.; Lewis, D.T. Soil available water as influenced by landscape position and aspect. *Agron. J.* **1982**, *74*, 999–1004.
10. Lyle, W.M.; Bordovsky, J.P. Low energy precision application (LEPA) irrigation system. *Trans. ASAE* **1981**, *24*, 1241–1245.
11. Timlin, D.J.; Pachepsky, Y.; Snyder, V.A.; Bryant, R.B. Spatial and temporal variability of corn grain yield on a hill-slope. *Soil Sci. Soc. Am. J.* **1988**, *62*, 764–773.

Precision Agriculture: Remote Sensing

Kerstin Panten

Holger Lilienthal

Institute of Plant Nutrition and Soil Science, Federal Agricultural Research Centre (FAL), Berlin, Germany

Erik Zillmann

Institute of Plant Nutrition and Soil Science, Federal Agricultural Research Centre (FAL), Braunschweig, Germany

Silvia H. Haneklaus

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

Remote sensing has been used in various investigations to determine the variability of soil parameters. Soil reflectance in the field is affected adversely by factors such as soil water dynamics, soil roughness, and surface effects because of cultivation practices. The interaction between all these factors reveals that ground truth campaigns are a prerequisite for calibrating reflectance data. Furthermore, complex algorithms need to be elaborated to correct interfering factors. Only then can reliable recommendations for farm management decisions be expected. Noteworthy is that with the exception of the radar technology, the penetration depth of the recorded reflectance is extremely small. Usually agricultural investigations comprise the whole top soil layer. Other problems such as the availability of sensor and platform, weather conditions, spatial, spectral and temporal resolution of data, trained operators, and, last but not least, cost-effective provision of data sets have to be overcome before remote sensing can be used extensively for data acquisition. Despite these basic restrictions, remote sensing has a high potential to identify in-field variation of soil features as large areas can be surveyed, which is not feasible by any other technique.

INTRODUCTION

Fertile soils are one of the most important resources on earth. Plant production practices such as fertilization, ploughing, and sowing are often applied in rates following certain soil parameters. Soils are heterogeneous in space and time so that the common way of uniform fertilizer application rates results in, side by side, oversupply and undersupply. Precision agriculture is an innovative farm concept, which uses modern techniques of geostatistic, geographic information systems, and global positioning systems for an ecologically and economically optimized use of agricultural inputs. Adoption of precision agriculture is unlikely to be uniform across farm types and sizes. Numerous factors such as soil variability, field sizes, and economical and technological possibilities influence the decision for or against precision agriculture.^[1] Mainly big farms with large fields and certain soil variability are interested or have already installed precision agriculture management systems. The efficient

acquisition of geocoded soil information is a major obstacle to the implementation of precision agriculture technologies. Soil information data sampled at densities that reflect the scale of spatial variability are required for the development of soil maps and decision support strategies to develop variable rate algorithms (e.g., for fertilizer applications).

Remote sensing is an efficient tool for retrieving information on large regions including data on spatial soil variability. Most agricultural fields have periods without vegetation so that the bare soil is visible, but the date depends on crop species and crop rotation. Bare soils can be found usually in the late autumn and early spring in Central Europe. However, with regard to remote sensing, these periods are unfavorable because of low sun elevation, which causes additional illumination and shading effects in the imagery.

In laboratory studies under standardized conditions, relationships were found between reflectance data and soil parameters such as soil organic matter, soil moisture,

particle size, mineral composition, iron oxides, soluble salts, and parent material. The three major factors causing soil heterogeneity on agricultural fields are soil organic matter, soil texture, and soil water regime. Therefore, this entry focuses on these parameters.

DATA SOURCES

The availability of a multitude of sensors and platforms results in a wide variety of remote sensing systems for data acquisition (Fig. 1). Not all sensor–platform combinations are operational, but theoretically, each sensor could be installed on each platform.

Sensors usually operate with wavelengths between 0.3 μm and 1 m. In general, four wavelength sections can be distinguished, namely: visible (0.4–0.7 μm), reflective infrared (0.7–3 μm), thermal infrared (8–14 μm), and microwave (1 mm–1 m). The geometrical resolution and distortion depend on the sensor system. Analog imagery can be transformed into digital data using scanning systems. The available remote sensing techniques are described in more detail by Richards and Jia^[2] and Lillesand and Kiefer.^[3] Soil reflectance measurements in the field are burdened by problems such as variations in illumination, viewing angle, and soil roughness. In particular, in two narrow ranges at about 1.4–1.9 μm , atmospheric water effects are problematic, besides the superimposing absorption features of gases.^[4]

SOIL ORGANIC MATTER

Soil organic matter is one of the most important soil fertility parameters of agricultural soils. In-field

variability of soil organic matter causes heterogeneous soil productivity. Coefficients of variation of 25–30% were determined.^[6] Thus, the determination of the variability of soil organic matter is essential for developing variable rate strategies, for instance, for fertilization and pesticide input.

The strong influence of soil organic matter on soil reflectance is described by Baumgardner et al.^[7] and Irons, Weismiller, and Petersen.^[8] If the soil organic matter content is higher than 2%, wavelengths of 0.4–2.5 μm proved to be suitable to establish quantitative relationships under laboratory conditions.^[7] Remotely sensed images are not used to map the variability of soil organic matter for precision agriculture^[9] because other factors such as soil moisture and surface roughness strongly influence the spectral reflectance and its suitability under practical conditions (Fig. 2). However, the variation of reflectance within fields can be used for directed sampling campaigns followed by laboratory analysis, resulting in information on spatial variation of soil parameters such as soil organic matter.^[10] A case study of McCann et al.^[11] described the use of reflectance characteristics of bare soil images to define management zones that need to be verified by ground truth campaigns.

SOIL TEXTURE

In rainfed agriculture, soil texture is the major factor influencing the effective field capacity. Particle size distribution, soil structure, and surface roughness affect the spectral reflectance. In single-factorial experiments, a decreasing particle size resulted in an increase of reflectivity.^[8] However, under field conditions, clayey soils often appear darker than sandy soils although the particle

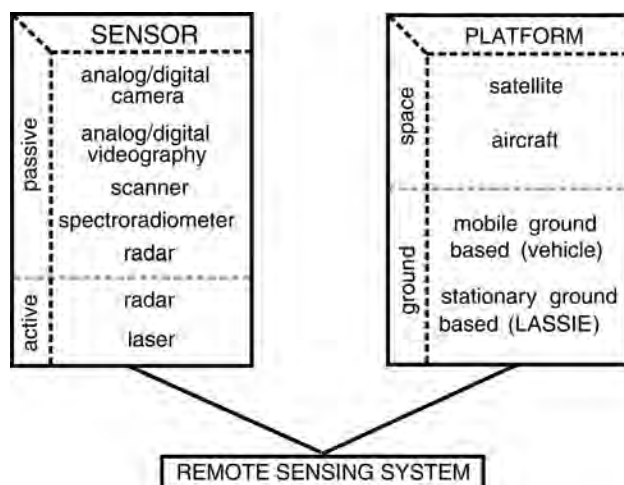


Fig. 1 Remote sensing systems for acquisition of reflectance data.

Source: From Lilienthal, Schnug, et al.^[5]



Fig. 2 Aerial image of a bare soil field with reflectance superimposition of soil organic matter content, soil texture, and soil moisture.

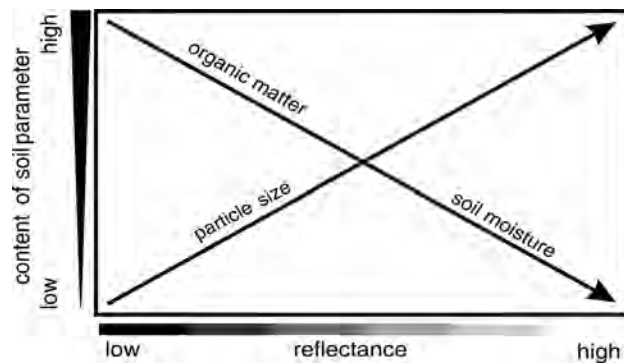


Fig. 3 Influence of major soil fertility factors on the reflectance of remotely sensed images.

size of clay is $<0.2\text{--}2\text{ }\mu\text{m}$, distinctly smaller than that of sand ($63\text{--}2000\text{ }\mu\text{m}$).^[12] The reasons are differences in mineralogy, the fact that clay forms agglomerates that are larger than sand grains, and the superimposition of soil moisture and soil organic matter.^[8] Fig. 3 illustrates the problem of superimposition of the major soil fertility parameters on the reflectance in a simplified scheme.

Extensive ground truthing is required to verify reflectance data. As soil texture is a constant soil parameter, ground data acquisition is required only once so that remote sensing could be an efficient tool for precision agriculture. However, at the moment, remotely sensed images are not used to map soil variability on an operational basis.^[9]

SOIL MOISTURE

As mentioned before, soil water regime is an important characteristic for the assessment of variability of soil fertility. Major problems in recording soil moisture are its dynamics and, again, the superimposition of the reflectance characteristics by other soil features (Figs. 2 and 3). Active radar sensing and passive radar sensing were used successfully to measure differences in soil moisture.^[13] (See also the entry *Water Content: Passive and Active Microwave Remote Sensing*, p. 2480.) Remote sensing in the microwave sector has substantial advantages compared with optical remote sensing techniques. The wavelengths penetrate clouds, haze, and light rain so that image acquisition is more or less independent of weather conditions. The microwave signal is not affected by solar illumination angles, resulting in bidirectional reflection effects. However, other problems such as geometrical distortion (e.g., layover and foreshortening) have to be considered. With respect to its use in precision agriculture, passive radar data are not available at an adequate spatial resolution. In comparison, it is possible to record the in-field variability of soil moisture by active radar.^[14] Here, the sensitivity to superimposition effects by soil roughness and vegetation

density limits the practical application for precision agriculture.

CONCLUSION

Remote sensing was used in various investigations to determine the variability of soil parameters.^[9] In laboratory studies, unifactorial experiments provided usually good results, whereas measurements under field conditions revealed clearly problems because of the superimposition of soil parameters. Soil reflectance in the field is affected adversely by factors such as soil water dynamics, soil roughness, and surface effects because of cultivation practices. The interaction between all factors reveals that ground truth campaigns are a prerequisite for calibrating reflectance data. Furthermore, complex algorithms need to be elaborated to correct interfering factors. Only then can reliable recommendations for farm management decisions be expected. Noteworthy is that with the exception of the radar technology, the penetration depth of the recorded reflectance is extremely small. Usually, agricultural investigations comprise the whole top soil layer. Other problems such as the availability of sensor and platform, weather conditions, spatial, spectral and temporal resolution of data, trained operators, and, last but not least, cost-effective provision of data sets have to be overcome before remote sensing can be used extensively for data acquisition. Despite these basic restrictions, remote sensing has a high potential to identify in-field variation of soil features as large areas can be surveyed, which is not feasible by any other technique.

REFERENCES

1. National Research Council, Board on Agriculture, Committee on Assessing Crop Yield. Site-specific farming, information systems, and research opportunities. In *Precision Agriculture in the 21st Century: Geospatial and Information Technologies in Crop Management*; National Academy Press: Washington, 1997.
2. Richards, J.A.; Jia, X. Sources and characteristics of remote sensing image data. In *Remote Sensing Digital Image Analysis—An Introduction*, 3rd Ed.; Revised and Enlarged Ed.; Springer: Berlin, 1999; 1–38.
3. Lillesand, Th.M.; Kiefer, R.W. *Remote Sensing and Image Interpretation*, 3rd Ed.; John Wiley and Sons, Inc.: New York, 1994.
4. Ben-Dor, E.; Irons, J.R.; Epema, G.F. Soil reflectance. In *Remote Sensing for the Earth Science, Manual of Remote Sensing*, 3rd Ed.; Rencz, A.N., Ed.; John Wiley and Sons, Inc.: New York, 1999; Vol. 3, 111–188.
5. Lilienthal, H.; Schnug, E.; Panten, K.; Haneklaus, S. Ground based remote sensing—A new tool for agricultural monitoring. In *Precision Agriculture*, Proceedings of the

- 6th International Conference on Precision Agriculture, Minneapolis, MN, Jul 14–17, 2002.
6. Beckett, P.H.T.; Webster, R. Soil variability—a review. *Soils Fert.* **1971**, *34*, 1–15.
7. Baumgardner, M.F.; Silva, L.F.; Biehl, L.L.; Stoner, E.R. Reflectance properties of soils. *Adv. Agron.* **1985**, *38*, 1–44.
8. Irons, J.R.; Weismiller, R.A.; Petersen, G.W. Soil reflectance. In *Theory and Application of Optical Remote Sensing*; Asrar, G., Ed.; John Wiley and Sons, Inc.: New York, 1989; 66–106.
9. Moran, M.S.; Inoue, Y.; Barnes, E.M. Opportunities and limitations for image-based remote sensing in precision crop management. *Remote Sens. Environ.* **1997**, *61*, 319–346.
10. Pocknee, S.; Boydell, B.C.; Green, H.M.; Waters, D.J.; Kvien, C.K. Directed Soil sampling. In *Precision Agriculture*, Proceedings of the 3rd International Conference on Precision Agriculture, Minneapolis, MN, Jun 23–26, 1996; Robert, P.C., Rust, R.H., Larson, W.E., Eds.; ASA–CSSA–SSSA: Madison, 1996.
11. McCann, B.L.; Pennock, D.J.; van Kessel, C.; Walley, F.L. The Development of management units for site-specific farming. In *Precision Agriculture*, Proceedings of the 3rd International Conference on Precision Agriculture, Minneapolis, MN, Jun 23–26, 1996; Robert, P.C., Rust, R.H., Larson, W.E., Eds.; ASA–CSSA–SSSA: Madison, 1996.
12. Schroeder, D. *Bodenkunde in Stichworten*, 5th Ed.; Ferdinand Hirt: Berlin, 1992.
13. Dobson, M.C.; Ulaby, F.T. Mapping soil moisture distribution with imaging radar. In *Principles and Application of Imaging Radar, Manual of Remote Sensing*, 3rd Ed.; Henderson, F.M., Lewis, A.J., Eds.; John Wiley and Sons, Inc.: New York, 1998; Vol. 2, 407–433.
14. Moran, M.S.; Hymer, D.C.; Qi, J.; Kerr, Y. Radar imagery for precision crop and soil management. In *Precision Agriculture*, Proceedings of the 4th International Conference on Precision Agriculture, Minneapolis, MN, Jul 19–22, 1998; Robert, P.C., Rust, R.H., Larson, W.E., Eds.; ASA–CSSA–SSSA: Madison, 1999.

Productivity: Natural and Anthropogenic Disturbances

Douglas G. Maynard

*Natural Resources Canada, Pacific Forestry Centre, Victoria,
British Columbia, Canada*

Charlotte E. Norris

*Department of Renewable Resources, University of Alberta, Edmonton,
Alberta, Canada*

Abstract

Forest ecosystems are dynamic systems that are continually undergoing changes and disturbances, whether they are natural or anthropogenic in origin. These disturbances influence soil pedogenesis and therefore soil productivity. Soil productivity is the capacity of a soil to support plant growth. The degree, type, and frequency of disturbance will variously affect plant growth through the changes they cause to the soil. A thorough knowledge of soil processes is therefore required to understand the potential effects of disturbances on productivity but with the understanding that short-term results may be different to long-term results in forest ecosystems.

INTRODUCTION

Disturbances are relatively discrete events, either natural or anthropogenic (i.e., human induced), that change the existing condition of an ecological system. Soil productivity refers to the capacity of a soil to support plant growth. Disturbances are an important process of virtually all forest ecosystems. It is not if a disturbance will happen but when, where, and how. Natural disturbances, such as fire, windthrow, insect and disease outbreaks, mass wasting (e.g., landslides), surface erosion, and catastrophic events (e.g., hurricanes, earthquakes, and floods), control forest succession and affect soil productivity. The influence of human activity has increased anthropogenic disturbances such as harvesting, mining, human-caused fires, air pollution (e.g., acid deposition), and climate change, which may affect forest succession and soil productivity. Human intervention (e.g., fire suppression) subsequently also has altered the frequency and degree of natural disturbances affecting soil productivity. How these disturbances alter soil processes and ultimately affect soil productivity depends on numerous factors, including the frequency, type, degree of disturbances, soil properties, and forest ecosystem.

The most obvious effect associated with disturbance natural or anthropogenic is the removal or destruction of aboveground biomass. However, changes as a result of disturbance may affect soil processes that result in either positive or negative changes to soil productivity. The changes in soil productivity from disturbance may be direct or indirect. They include changes to the chemical, physical,

and biological properties of the soil, such as loss of organic matter, soil compaction, soil displacement or mixing, exposure of subsurface material and bedrock, changes in nutrient supply, and soil organisms.

NATURAL DISTURBANCES

Fire

The most widely occurring and extensively studied natural disturbance is fire. Forest fires are a major large-scale disturbance in most forests of the world. These include temperate and boreal forests, most Australian ecosystems, Mediterranean-type ecosystems, and all but the wettest of tropical rainforests,^[1] including lowland rain forests of northwestern Amazonia.^[2] The intensity, type, and frequency of fire are important factors that affect soil productivity. High-intensity burns with high temperatures will result in volatilization of various nutrients. Carbon (C), nitrogen (N), and sulfur (S) will volatilize at relatively low temperatures (as low as 302°F to <932°F). Phosphorus (P) and potassium (K) are less volatile but will volatilize at temperatures of more than 1112°F. Losses of other non-volatile nutrients (>1832°F) such as calcium (Ca), magnesium (Mg), and manganese (Mn) may occur through blowing of fine ash. Similarly, in low-temperature burns, P also can be lost through the removal of fine ash. The accumulation of cations (e.g., Ca and Mg) has been observed in surface soils following high-intensity fires and results in increased soil pH.

Soil biological processes also may be influenced by fire, either directly by affecting soil organisms or indirectly as a result of chemical changes. It is difficult, however, to assess the relative influence of fire on soil biological processes. Some soil organisms are killed in ground fires, thus the species composition of the soil fauna and microbial population may change. This could indirectly influence soil productivity by affecting the microbial population of the soil (i.e., mycorrhizal fungi), thus altering the rate of nutrient release from the soil.

In the short term, fires can increase nutrient availability by releasing those tied up in soil organic matter (humus). This is probably beneficial; however, under certain conditions (e.g., relatively infertile soils), organic matter and nutrients may be lost that could decrease the long-term productivity of the soil.^[3] Fire is rarely a uniform event, and, generally, large burns have a variety of intensities across the landscape. Thus, generalizations are not entirely appropriate, although fire disturbance is considered essential to the rejuvenation of most forest ecosystems through the release of nutrients found in humus and renewal of forest stands.

Erosion

Surface erosion would be expected to be low in natural forests. Significant surface erosion may result from natural disturbances such as landslides, intense fires, and anthropogenic disturbances, which remove vegetative cover and expose mineral soil. Mass wasting (e.g., landslides and snow avalanches) is a natural process in mountainous areas and steep terrain. Generally, mass-wasting events tend to have longer lasting effects on soil productivity, because surface soil is often removed or severely disturbed. The soils on upper portions of slides generally are shallower and coarser with more bedrock exposure than the original soils. The lower portion of slides consists mainly of materials deposited or mixed with original soil. Additionally, for several years after failure, slides are susceptible to surface erosion. Revegetation of slide areas generally occurs on modified rather than primary soil materials. Productivity of these areas is reduced, although in many cases colonizing plant species are similar to adjacent areas.

Windthrow

Gaps are the most obvious result of windthrow, but soils can also be affected^[4] by altering structure and mixing subsurface soil with surface material. In some forests, small-scale gaps are the dominant canopy disturbances between large-scale events such as fire.

Mixing of soil may affect nutrient availability and soil-forming processes and expose a variety of substrates that may negatively affect surface soil chemical and physical properties. In boreal forests, long-term effects of windthrow may persist for 100–200 years.

Insects and Disease

Insects and disease indirectly accelerate or change soil nutrient dynamics. These processes include 1) altering the rate and amount of nutrients leached or deposited as litter; 2) changing light intensities; 3) reducing competition among plants; 4) altering plant species composition; and 5) stimulating translocation of nutrients from boles and branches to high turnover components such as leaves, buds, and flowers.^[5] Insect grazing occurs more slowly than obvious disturbances such as fire or forest harvesting. Similarly, pathogens (i.e., disease organisms) also differ from large-scale disturbances by selectively eliminating less vigorous individuals.^[6] The influence of insects and diseases, however, is similar to large-scale disturbances in that they recycle nutrients and renew forest stands within the landscape. Pathogens and, to some extent, insects can also increase tree susceptibility to windthrow and fire, which will affect soil productivity as outlined previously.

ANTHROPOGENIC (HUMAN-INDUCED) DISTURBANCES

Harvesting

The most common human-related activity is harvesting and its associated practices. The influence on soil productivity will vary greatly among species (forest type), time of harvest, and inherent soil properties.

Biomass removal is the main pathway for nutrient loss from harvesting. Losses will depend on the type of harvest (e.g., whole tree vs. boles only), harvest system used (e.g., clear cut vs. selective logging), forest type, and time of year. Generally, the influence of forestry practices on nutrient loss is long term, and harvesting rotation times that are less than the time needed to replace lost nutrients will eventually reduce soil productivity. The primary N sources in forest soils are biological fixation or deposition from the atmosphere. In most temperate and boreal forests, loss of N and soil processes related to N availability is the most critical. Availability of most other nutrients other, however, is largely dependent on weathering of parent material. In some temperate and boreal forests, particularly those affected by acid deposition, the loss of cations, particularly calcium (Ca), also may reduce soil productivity.^[7]

Some tropical forests, such as tropical montane forests, appear to function similar to temperate and boreal forests with respect to N.^[8] However, in highly weathered clay soils of the tropics (e.g., Oxisols and Ultisols), there is little replacement of nutrients lost.^[9] In these forests (i.e., lowland rain forests), changes in the phosphorus P cycle (and possibly Ca) following harvesting may be the most critical process affecting soil productivity.

Increased leaching (less nutrient uptake and lower evapotranspiration) is another pathway for nutrient loss following harvest. In most studies of temperate and boreal forests, nutrient loss from increased leaching in harvesting does not appear to have a major influence on soil productivity in the short term.^[10]

Deforestation through anthropogenic fire (i.e., slash-and-burn agriculture) and land clearing is a special type of harvesting disturbance of particular concern in tropical forests. This phenomenon has contributed to reduced productivity across a number of tropical soil types and forest ecosystems including the Amazon.^[11] Following land clearing involving fire, loss of organic matter and nutrients is generally more rapid and complete in tropical forests than in temperate or boreal forests.^[12] This is believed to be caused by a combination of the climate, easily decomposable organic matter, and reduced organic inputs. In nutrient-poor ecosystems, disruption of the nutrient conservation processes will quickly result in reduced productivity following land clearing. Nutrient-rich sites do not lose their potential productivity in the short term, but the long-term implications are still a concern.^[13]

Other important factors affecting soil productivity in harvesting are soil compaction, displacement, and organic matter loss. Altered soil physical properties from forestry practices represent deviations from the natural range of soil conditions, usually with negative effects on soil productivity.^[14] Physical disturbances associated with harvesting, including roads, skid trails, and landings, may result in soil compaction, soil displacement and mixing, exposure of unfavorable subsurface material, and organic matter removal.

Soil compaction is a concern in virtually all forest ecosystems.^[15] Tracked areas generally have lower soil productivity because of soil compaction and possibly soil displacement (including removal of organic matter). Generally, the loss of productivity is related to the amount of compaction. Compaction affects soil structure, aeration, water infiltration, runoff, and surface erosion that will reduce soil productivity. Recovery of compacted soils is dependent on several factors and has been found to vary from several years to several decades in boreal, temperate, and tropical forests.^[15] In contrast, tree growth has been found to be higher on displaced material such as berms and sidecasts compared with the harvested undisturbed soils. This has been related to improved edaphic conditions (e.g., aeration and porosity), increased soil organic matter, and lack of vegetative competition.

Site Preparation

Site preparation is used to produce more appropriate seedbeds for seedlings. In the short term, soil productivity may be improved. However, potential detrimental factors as a result of organic matter and nutrient loss could have long-term effects. The severity of disturbance is related to the

degree of organic matter and nutrient losses as a result of mechanical site preparation.^[3]

Mining

Mining produces large amounts of waste material and disrupts the biogeochemical cycles (e.g., water, carbon, and nutrient cycles) within the local landscape and therefore causes losses in soil productivity. Reclamation practices are determined by a thorough knowledge of forest conditions predisturbance and an understanding of how to reintegrate the disturbed area into the surrounding landscape. Fundamental to reclamation is the type of soil used and the soil's physical, chemical, and biological properties.^[16] Establishment of soil productivity and a natural climax of forest succession following land reclamation is unique to each site where (as noted in Harvesting above) tropical, temperate, and boreal soils have undergone different soil-forming processes and each area therefore requires different considerations. For example, coarse-textured acidic soils were best for subsequent tree growth in the temperate Appalachian Mountains.^[17] While in the western boreal forest, landscape level variability was shown to be an important consideration in reclamation.^[18] Restoring soil productivity with resilience and landscape integrity are the objectives of ecosystem reclamation; however, there is a growing understanding that novel and not natural ecosystems may develop.^[19]

Acid Deposition

Acid deposition is a large-scale (i.e., regional or greater) disturbance that may influence soil productivity. Acid deposition is decreasing in many parts of the world; however, it remains a concern in most of Europe and eastern North America and is of increasing concern in many countries such as China. In high-deposition areas, soil acidification, reduced base cations in surface soils, increased aluminum, and nutrient imbalances (particularly N saturation) have been observed. The loss in soil productivity will depend on the amount of soil acidification and the inherent physical and chemical properties. For example, the influence of acid deposition on the soil productivity of a sandy soil with low-cation exchange capacity will probably be greater than on a fine-textured soil with high-cation exchange capacity.

Climate Change

The spatial scale of climate change is similar to acid deposition; however, the potential influence of climate change on soil productivity is largely unknown. Changing climatic conditions may directly influence biological and chemical processes and thus influence soil productivity. Indirect effects include changing the frequency and degree of natural disturbances.

CONCLUSION

Disturbances are important processes that affect forest ecosystems. Increasingly, human activities have resulted in changes to the frequency, type, and degree of disturbance on forest ecosystems. Disturbances influence the physical, chemical, and biological properties of soil, and changes to these properties subsequently affect soil productivity. The difficulties in assessing changes associated with disturbances are that forest ecosystems are evolving and not static, and available information is usually based on short-term results that may not be indicative of long-term effects. Thus, a fundamental understanding of the soil physical, chemical, and biological processes is needed to assess the consequences of natural or anthropogenic disturbances on soil productivity.

REFERENCES

1. Attiwill, P.M. The disturbance of forest ecosystems: The ecological basis for conservation management. *Forest Ecol. Manag.* **1994**, *63*, 247–300.
2. Sanford, R.L., Jr.; Saldarriaga, J.; Clark, K.E.; Uhl, C.; Herrera, R. Amazon rain-forest fires. *Science* **1985**, *227* (4682), 53–55.
3. Prescott, C.E.; Maynard, D.G.; Laiho, R. Humus in northern forests: Friend or foe? *Forest Ecol. Manag.* **2000**, *133* (1–2), 23–36.
4. Ulanova, N.G. The effects of windthrow on forests at different spatial scales: A review. *Forest Ecol. Manag.* **2000**, *135* (1–3), 155–167.
5. Mattson, W.J.; Addy, N.D. Phytophagous insects as regulators of forest primary production. *Science* **1975**, *190* (4214), 515–522.
6. Castello, J.D.; Leopold, D.J.; Smallidge, P.J. Pathogens, patterns and processes in forest ecosystems. *BioScience* **1995**, *45* (1), 16–24.
7. Maynard, D.G.; Paré, D.; Thiffault, E.; Lafleur, B.; Hogg, K.E.; Kischuk, B. How do natural disturbances and human activities affect soils and tree nutrition and growth in the Canadian boreal forest? *Environ. Rev.* **2014**, *22* (2), 161–178.
8. Tanner, E.V.J.; Vitousek, P.M.; Cuevas, E. Experimental investigation of nutrient limitations of forest growth on wet tropical mountains. *Ecology* **1998**, *79* (1), 10–22.
9. Vitousek, P.M.; Sanford, R.L., Jr. Nutrient cycling in moist tropical forest. *Annu. Rev. Ecol. Syst.* **1986**, *17*, 137–167.
10. Grigal, D.F. Effects of extensive forest management on soil productivity. *Forest Ecol. Manag.* **2000**, *138* (1–3), 167–185.
11. Laurance, W.F.; Cochrane, M.A.; Bergen, S.; Fearnside, P.M.; Delamônica, P.; Barber, C.; D'Angelo, S.; Fernandes, T. The future of the Brazilian Amazon. *Science* **2001**, *291* (5503), 438–439.
12. Scholes, R.J.; van Breemen, N. The effects of global change on tropical ecosystems. *Geoderma* **1997**, *79* (1–4), 9–24.
13. Jordan, C.F.; Herrera, R. Tropical rain forests: Are nutrients really critical? *Am. Nat.* **1981**, *117* (2), 167–180.
14. Norris, C.E.; Maynard, D.G.; Hogg, K.E.; Benton, R.; Titus, B.D.; Curran, M.P. Ten-year results of seedling growth on calcareous soils in the interior of British Columbia, Canada. *Forest Ecol. Manag.* **2015**, *346*, 65–80.
15. Kozłowski, T.T. Soil compaction and growth of woody plants. *Scand. J. Forest Res.* **1999**, *14* (6), 596–619.
16. Cooke, J.A.; Johnson, M.S. Ecological restoration of land with particular reference to the mining of metals and industrial materials: A review of theory and practice. *Environ. Rev.* **2002**, *10* (1), 41–71.
17. Zipper, C.E.; Burger, J.A.; Barton, C.D.; Skousen, J.G. Rebuilding soils on mined land for native forests in Appalachia. *Soil Sci. Soc. Am. J.* **2013**, *77*, 337–349.
18. Quideau, S.A.; Swallow, M.J.B.; Prescott, C.E.; Grayston, S.J.; Oh, S.-W. Comparing soil biogeochemical processes in novel and natural boreal forest ecosystems. *Biogeosciences* **2013**, *10*, 5651–5661.
19. Hobbs, R.J.; Higgs, E.S.; Hall, C.M. *Novel Ecosystems: Intervening in the New Ecological World Order*; John Wiley & Sons, Ltd.: Chichester, 2013.

Protection: Policy of the European Union

Irene L. Heuser

Department of International Affairs, State Chancellery of the State of Brandenburg, Berlin, Germany

Luca Montanarella

European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy

Abstract

Although different European Union (EU) policies (for instance, on water, waste, chemicals, industrial pollution prevention, nature protection, and pesticides) are already contributing to soil protection, they are not sufficient to ensure an adequate level of protection for all soils in Europe. This is due to the fact that legislation has not been orientated toward the different soil endangerment paths and the protection of the various soil functions. Therefore, the European Commission has developed a soil protection strategy in order to find answers to the most urgent soil threats, i.e., the priority threats of organic matter decline, soil erosion, and soil contamination as well as the additional aspects of sealing, compaction, decrease of biodiversity, salinization, and floods and landslides. The proposal for a Soil Protection Framework Directive sets out common principles for protecting soils across the EU and will help the EU Member States to better protect the soil on their territory and to use it in a sustainable way.

INTRODUCTION

The environment was a latecomer to the policy agenda of European integration. When the European (Economic) Community was founded in 1957, it concentrated on the quantitative dimensions of building the common market, with relatively little attention paid to its qualitative aspects. Meanwhile, air and water resources are protected by European Union (EU) law, whereas soil degradation is a serious problem in Europe. Soil performs a variety of crucially important functions for human life: 1) storage and buffering of pollutants and chemical compounds; 2) basis for food and fiber production; 3) storage of cultural heritage; and 4) source of raw materials. Soil degradation and its consequences are perceived only with considerable temporal delay because of the large resilience of soil systems to changes. Building up fully functioning soil layers of 30 cm requires a period of 1000–10,000 years.^[1] Therefore, soil is considered mainly as a non-renewable resource.^[2] In the EU, more than 150 million hectares of soil are affected by erosion, in the most extreme cases leading to desertification. Western Europe is highly urbanized and competition for the land available results in soil sealing. Moreover, soil contamination is a problem and 45% of the European soils have a low content of organic matter.^[3] Europe is characterized by great differences with respect to the soils' physical, chemical, and biological properties and more than 320 major soil types are reported on the soil map of Europe (Fig. 1). The common soil map^[4]

has helped to describe the high spatial variability. Furthermore, the majority of our soils are subject to private property rights. Owing to the threats to the soils of Europe, the European Commission has developed a soil protection strategy in order to find answers to these urgent questions. The following section investigates to which extent soil protection meanwhile is realized in the EU.

FEATURES OF EU SOIL PROTECTION POLICY

EU environmental policy and law have a huge impact on each Member State, with approximately 85% of national environmental legislation being derived from the Community. Although the Single European Act of 1987 established the possibility of enacting regulations about soil protection, this field has remained almost neglected. There is no comprehensive legal instrument dealing with soil protection. Theoretically, chemical, biological, and physical threats of the soils could be tackled by the objectives of Articles 174 ff. EC. [The extent and description of these environmental competencies will be preserved with almost the same wording according to the new Articles 191 ff. of the Treaty on the Functioning of the European Union which entered into force (as a part of the Lisbon Treaty) on December 1, 2009.^[5]] Any proposed legislation also has to take into account the complexity of environmental situations (e.g., different soil types) in the various regions of the Community.



Fig. 1 The distribution of major soil groups in Europe according to the World Reference base.

Source: From European Soil Data Centre, European Commission, 2008.

Community legislation on environmental aspects directly or indirectly affects the soils. The introduction of the Sewage Sludge Directive^[6] of 1986 forms the first legal instrument partly contributing to the protection of soils by regulating the use of sewage sludge in agriculture in such a way that it takes the plants' nutrient requirements and the quality of the soil into account. The 1996 Directive on Integrated Pollution and Prevention Control^[7] contains transmedial requirements on the operation of certain industrial plants, in particular permission provisions for larger

industrial plants on the basis of the application of the best available technique. (In December 2007, the Commission proposed a new directive on industrial emissions which will integrate the IPPC approach in a consolidated version.^[8]) Waste management is a key element in preventing soil contamination. According to the EU waste hierarchy, waste should primarily be prevented and only secondarily reused, recycled, and recovered as much as feasible with landfills as the third option. (The new Waste Framework Directive will incorporate the content of the Hazardous Waste

Directive and the Waste Oils Directive.^[9]) The legal regime includes horizontal legislation on the waste framework, hazardous waste and the shipment of waste, directives dealing with waste installations, and those with specific waste streams.^[10]

Furthermore, EC chemicals law and water protection law prevent the entry of dangerous substances into the soil. With the new “REACH” system of registration, evaluation and authorization of chemicals, a comprehensive life cycle analysis, are to be carried out for highly soil-damaging harmful substances. The Nitrate Directive^[11] has caused limitations to the maximum amounts of nitrogen and contributed especially in endangered areas to a concentration reduction. Only plant protection products whose active substances are listed in the Pesticide Directive^[12] and do not pose a risk to the soils are authorized in the EU. With the Biocide Directive,^[13] a useful supplement is ensured for non-agricultural sectors. Because of the approach of the Water Framework Directive^[14] with its environmental quality standard of achieving a good ecological surface and groundwater status and its environmental action goal of implementing this condition within 15 years, the indirect aim of preventing pollutant concentrations in the soil has to be realized. Flood risk management policy also aims to reduce the impact of floods.^[15] Because of the inclusion of soil protection into the protection of special areas of conservation, the “Habitats Directive”^[16] forms the basis for a comprehensive consideration of all natural components of a habitat including the soil and contributes to the maintenance of soil biodiversity, especially because these community areas are such with often very sensitive soils.

EU air pollution control law^[17] aims to prevent the entry of various substances into the soil by an installation-related approach with pollutant-referred supplements. Member States can introduce stricter limit values at any time. The control of major accident hazards involving dangerous substances is ensured by the “Seveso II Directive,”^[18] which applies to some thousands industrial establishments where dangerous substances are present in high quantities. Also, general environmental provisions that have a strong procedural orientation and provide informative elements make a further contribution to a more conscious handling of the problems of soil protection. For example, the Environmental Impact Assessment Directive and the Strategic Environmental Assessment Directive^[19] are supporting instruments for the protection of the soils from harmful developments. The Environmental Liability Directive^[20] ensures that companies have to pay the costs for damages to the soils, only if they create a significant risk for human health. Moreover, research is needed to fill the gaps in soil knowledge and prepare for political and legislative action.

To sum up, the EU has already set up some standards in important fields of environmental law with direct or indirect effects on the soils, but a special soil protection policy

is in its infancy. Because of the threats to the soils in all EU Member States, an effective soil protection policy will have to be introduced.

INTERNATIONAL CONTEXT

International environmental policy is also far from achieving comprehensive soil protection. These problems in their entirety are not conceptually laid down by the law of international organizations in an obligatory form. In most cases, there is only “*pré-droit*” or “soft law,” e.g., in the World Soil Charter. Binding instruments in other fields of environmental policy might influence the use of the soils, especially the UN Convention to Combat Desertification (UNCCD), the Convention on Biological Diversity (CBD), and the UN Framework Convention on Climate Change (UNFCCC).^[21] Geographically relevant for the EU context, the achievements of the Council of Europe, which initiated international attention to the problems of soil protection by the adoption of the European Soil Charter already in 1972, go far beyond other regulations of international law, in particular by the decision on the “Revised European Charter for the protection and sustainable management of soil” on May 28, 2003. The idea of a international soil protection convention or at least a protocol to the CBD or the UNCCD, which would contribute to a more intensified dealing with the problems in all its facets, points in the right direction. A draft “Protocol for the Protection and Sustainable Use of Soil” as prepared by the Specialist Group of the IUCN Commission on Environmental Law is being discussed.

TOWARD A NEW EU SOIL PROTECTION FRAMEWORK DIRECTIVE

The Sixth Environmental Action Programme adopted on July 22, 2002 marks the beginning of an important phase of development with the intention to work on a specific “Thematic Strategy for Soil Protection” in the Community as a part of its plan to protect and preserve the natural resources. This strategy^[22] comprises a communication from the Commission and a legislative proposal that requires Member States to tackle the three priority threats of organic matter, i.e., decline, soil erosion, and soil contamination as well as the additional aspects of sealing, compaction, decrease of biodiversity, salinization, and floods and landslides. The EU institutions are discussing the proposed “Soil Protection Framework Directive,” which is not adopted. Several elements are being explored and assessed to decide whether to include them in the legislative instrument. The communication also identified the decline in soil biodiversity as a threat, but owing to the knowledge gap, it is not addressed in the directive proposal; closing this gap through research is regarded as an important aspect as well.

Among the elements being assessed and explored are the obligation to identify risk areas for erosion, organic matter decline, compaction, salinization, and landslides and the possibility of requiring Member States to establish programs of measures to combat the risks in those areas. The draft directive fills the gap between the classification and provisions required for the treatment of contaminated land that are completely missing in EU law yet and proposes setting a common definition of contaminated sites, a soil status report, and a common list of potentially soil-polluting activities that could be used by Member States to identify the contaminated sites on their territory and to establish a national remediation strategy. Moreover, preventive requirements are also being considered in the draft directive, e.g., to limit soil sealing.

There are many hurdles to face before the Soil Protection Framework Directive will be adopted. At least, it is clear that the legal base for soil protection provisions on EU level is Article 192 (1) of the Treaty on the Functioning of the European Union, which requires majority voting in the Council. In December 2007, EU Environment Ministers failed to find a compromise owing to the fear of administrative burdens for the Member States and the principle of subsidiarity under Article 5 (2) EU Treaty as a (false) counterargument. With respect to the strong local dimension of soils, action should mainly be taken on local level. But nevertheless there are reasons for EU-wide measures for soil protection, in particular the multifunctionality of soils, the transboundary consequences of, e.g., soil erosion and contamination, and the close correlation between soil degradation and other environmental problems (like climate change). Failure in protecting the soils will not only undermine sustainability, but also the long-term competitiveness of the EU.^[23]

In general, soil protection should be understood not only as minimization of damages, but also as comprehensive care and development of precautions and prevention of irreversible pollution and of harmful impacts on its functions.^[24] A mix of regulative and non-regulative instruments like in other fields of environmental policy would be appropriate for soil protection too. In the long term, the discussions within the EU institutions and in the Member States will hopefully pave the way for a policy that will ensure the protective and sustainable use of these natural resources.

CONCLUSION

The soil is a vital and largely non-renewable resource, and it has been subjected to several sector-related policy measures in the EU. The legislation covers some aspects of chemical and biological soil protection, but it is fragmented and achieves the aspect of safeguarding the soils

only as a side effect. Furthermore, EU law hardly takes any account of the requirements for the prevention of soil compaction, erosion, sealing, and other physical threats to the soil. This situation seems mainly due to the fact that EC legislation has not been orientated toward the different soil endangerment paths and the protection of the various soil functions. The proposed Soil Framework Directive tackles many of these critical aspects and develops instruments to solve the most severe problems. Agreeing on a international soil protection convention or at least a protocol to the existing UN conventions would also mean a great step forward. The need for the action in this field is particularly obvious and there are fundamental possibilities for improvement. In the course of these activities, we will hopefully have a genuine soil protection policy on EU level soon.

ACKNOWLEDGMENT

The views expressed in this entry are personal and do not reflect those of her employer.

REFERENCES

1. European Environment Agency/United Nations Environment Programme. *Down to Earth*; EEA: Copenhagen, 2000; 7 pp.
2. Commission of the European Communities (COM). 179 *Final of 16.4.2002*; CEC: Brussels, 2002; 8 pp.
3. Commission of the European Communities (COM). 231 *Final of 22.09.2006*; CEC: Brussels, 2006; 1 pp.
4. Montanarella, L. The thematic strategy for soil protection of the European Commission. In *Soils on the Global Agenda*; Hurni, H., Giger, M., Meyer, K., Eds.; Geographica Bernensia: Bern, 2006; 30–31.
5. European Union. Treaty of Lisbon amending the Treaty on European Union and the Treaty establishing the European Community. Off. J. Eur. Union **2007**, C 306, 1.
6. European Union. Directive 86/278/EEC. Off. J. Eur. Union **1986**, L 181, 6; Off. J. Eur. Union **1991**, L 377, 48. (See Marmo. In *Lowel/Hudson*; Proceedings of the CIWEM/6th European Biosolids and Organic Residuals Conference, 2001, p. 1 ff).
7. Integrated pollution prevention and control (IPPC) Directive 96/61/EC. Off. J. Eur. Union **1996**, L 257, 26.
8. Commission of the European Communities (COM). 844 *Final of 21.12.2007*; CEC: Brussels, 2007; 1 pp.
9. European Union. Directive 2006/12/EC of 5 April 2006 on waste. Off. J. Eur. Union **2006**, L 114, 9.
10. European Union. Especially the Landfills Directive 1999/31/EC. Off. J. Eur. Union **1999**, L 182, 1.
11. European Union. Directive 91/676/EEC. Off. J. Eur. Union **1991**, L 375, 1.
12. European Union. Directive 91/414/EEC. Off. J. Eur. Union **1991**, L 230, 1. (See proposal for a new Pesticide Framework

- Directive, COM (2006) 373 final of 12.07.2006, p. 1 ff.; COM (2006) 372 final of 12.07.2006, p. 1).
13. European Union. Directive 98/8/EC. Off. J. Eur. Union **1998**, L 123, 1.
 14. European Union. Directive 2000/60/EC. Off. J. Eur. Union **2000**, L 327, 1.
 15. European Union. Directive 2007/60/EC. Off. J. Eur. Union **2007**, L 288, 27.
 16. European Union. Directive 92/43/EEC. Off. J. Eur. Union **1992**, L 206, 7.
 17. European Union. Air Quality Framework Directive 96/62/EC. Off. J. Eur. Union **1996**, L 296, 55; Off. J. Eur. Union **2001**, L 309, 22.
 18. European Union. Directive 96/82/EC. Off. J. Eur. Union **1997**, L 10, 1.
 19. European Union. Directive 85/337/EEC. Off. J. Eur. Union **1985**, L 175, 40; Directive 2001/42/EC. Off. J. Eur. Union **2001**, L 197, 30.
 20. European Union. Directive 2004/35/EC. Off. J. Eur. Union **2004**, L 143, 56.
 21. Hannam, I.; Boer, B. *Legal and Institutional Frameworks for Sustainable Soils*; The World Conservation Union: Gland, Switzerland and Cambridge; Heuser. *Europäisches Bodenschutzrecht*, 350.
 22. COM. Thematic Strategy for Soil Protection, **2006**, 1; 231 Final of 22.09.2006, 1.
 23. Heuser. *Europäisches Bodenschutzrecht*; Erich Schmidt Verlag: Berlin, Vol. 80, 573.
 24. McCormick, J. *Environmental Policy in the European Union*; Belhaven: London, 2001; 75.

Protozoa

Bryan S. Griffiths

Soil Plant Dynamics Unit, Scottish Crop Research Institute, Dundee, Scotland

Abstract

The Kingdom Protista contains mainly unicellular eukaryotic organisms that cannot be classified into the three other kingdoms of plants, animals, and fungi. Protozoa are single-celled eukaryotic organisms commonly subdivided into four groups: flagellates, naked amoebae, shelled or testate amoebae (Testacea), and ciliates. Protozoa play a substantial part in energy and nutrient flows of terrestrial ecosystems. The impact of environmental and land management factors on protozoa results primarily from their effects on substrate supply and secondarily on soil structure, moisture, and climatic factors.

INTRODUCTION

The Kingdom Protista contains mainly unicellular eukaryotic organisms that cannot be classified into the three other kingdoms of plants, animals, and fungi. Protozoa are single-celled eukaryotic organisms^[1] commonly subdivided into four groups: flagellates, naked amoebae, shelled or testate amoebae (Testacea), and ciliates (Fig. 1). Naked amoebae are generally the most important group of protozoa in soil.^[2] Protozoa require a water film for locomotion and feeding, so their activity is limited to the water-filled pore space in soil. Most protozoa have adapted to fluctuating moisture by the ability to form resistant cysts, but ciliates require much higher soil moisture than flagellates or amoebae to be active.^[3,4] Reproduction is usually asexual by means of binary fission, but sexual reproduction also occurs.

IMPORTANCE IN SOIL, PLANT, AND ENVIRONMENTAL PROCESSES

Protozoa play a substantial part in energy and nutrient flows of terrestrial ecosystems, consuming 22% of the carbon (C) input to a beech-forest soil and 63% of the microbial biomass.^[5] Annual production is 39–98% of that of the total soil fauna,^[6] and their contribution to total net nitrogen mineralization in soil is 12–30%.^[7] Protozoa in soil systems regulate and modify microbial community activity and composition and inoculate new substrates with microorganisms.^[8,9] Their effects result mainly from their feeding activities because they are too small to directly influence soil structure. Soil protozoa are heterotrophic, feeding on bacteria, although mycophagous,^[10] predatory, and saprophytic protozoa occur.^[1] Approximately 40% of ingested nutrients are used for the production of protozoan biomass, while 60% are excreted.^[11] The effects of this nutrient turnover are such that the plants are significantly smaller and contain less nitrogen if protozoa are not present.^[2,8,9,11] Protozoa are a key

link in the terrestrial food chain as a major predator of bacteria and prey for larger organisms. The indirect effects of soil protozoa in stimulating plant growth are increasingly recognized and are probably more important than the direct nutritional effects.^[12]

FACTORS AFFECTING PROTOZOAN ACTIVITY AND COMPOSITION

The impact of environmental and land management factors on protozoa results primarily from their effects on substrate supply and secondarily on soil structure, moisture, and climatic factors. Protozoa decrease with soil depth, from approximately 10⁶/g at 10 cm to 10/g below 1 m, resulting from decreasing levels of organic matter down the soil profile.^[13] The application of organic manure to soil is an obvious source of added substrate and increases protozoan populations.^[14] Inorganic fertilizers also increase protozoan populations because of the increased plant growth, leading to a greater substrate input to the soil.^[14] The application of elemental sulfur, however, increased acidification of a Canadian pasture and reduced numbers of soil microorganisms and protozoa.^[15] Protozoa in soil are characterized by very rapid changes in density—amoebal numbers increase within two days of a rain due to a stimulation of bacterial prey.^[2] Seasonal maxima can be related to food supply, such as the spring and autumn peaks of *Dictyostelium mucoroides*^[16] and the peak of Testacea following autumn leaf fall.^[17] Population minima are typically related to unfavorable environmental conditions, such as dry summers and low temperature.^[13] Protozoa can rapidly encyst/excyst,^[18] such that 100% of the protozoa were active six days after watering a desert soil, compared to 0% before.^[19] Protozoa have a clumped horizontal distribution, again linked to substrate availability in the rhizosphere and in decomposing organic matter.^[20] Rhizosphere populations are, on average, sixfold higher than in

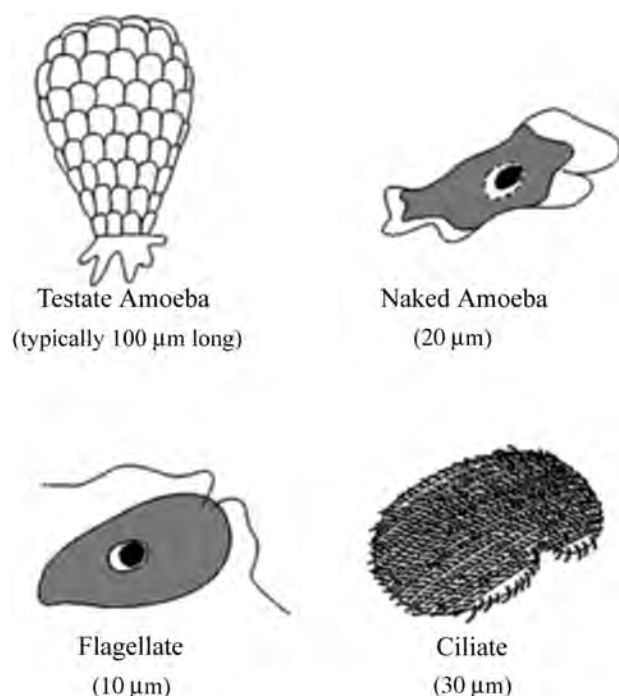


Fig. 1 Diagram of the four main groups of soil protozoa.

the surrounding soil.^[13] A 90-fold increase on decomposing grass residues in soil has been demonstrated.^[21] Protozoan species can be used as biological indicators of the environment.^[1] There were, e.g., marked differences in the species of testate amoebae among mor, acid-mull, and calcareous-mull litter types,^[22] and the ratio of *Colpodea* to *Polyhymenophora* is an indicator of soil disturbance.^[23] Soil protozoa are also used in laboratory bioassays to monitor pesticides and heavy metals.^[24]

METHODS OF STUDYING PROTOZOA

The study of protozoa in soil is hampered by their relatively small size, low numbers, and variable morphology. Direct enumeration has been most successfully used for testate amoebae^[25] and ciliates.^[26] Density-gradient centrifugation has enabled the direct observation of metabolically active protozoa.^[27] Culturing techniques rely on protozoan growth on laboratory media. A soil suspension is generally diluted using the most probable number technique,^[28] with modifications to allow the enumeration of mycophagous and active protozoa.^[10,29] Technological advances are being adapted to the study of protozoan populations in soil. Thus, a phospholipid fatty acid marker for protozoa has been identified,^[30] a β -N-acetyl-glucosaminidase-like enzyme has been identified as a putative marker for protozoan bacterivory,^[31] while molecular probes^[32] and analysis of 18S rRNA genes^[33] allow examination of protozoan communities without the need for culturing.

CONCLUSION

There is much we need to learn about this unknown and mostly unseen phylum and the many key roles its members have in the soil. Whether we like it or not, the protista are down there waiting, always in their thousands per gram soil—and for short but functionally important periods there will be millions of them.^[34]

REFERENCES

1. Foissner, W. Soil protozoa: Fundamental problems, ecological significance, adaptations in ciliates and testaceans, bioindicators, and guide to the literature. *Prog. Protistol.* **1987**, *2*, 69–212.
2. Clarholm, M. Effects of plant–bacterial–amoebal interactions on plant uptake of nitrogen under field conditions. *Biol. Fert. Soils* **1989**, *8*, 373–378.
3. Zwart, K.B.; Brussaard, L. Soil fauna and cereal crops. In *The Ecology of Temperate Cereal Fields*; Firbank, L.G., Carter, N., Darbyshire, J.F., Potts, J.R., Eds.; Blackwell Scientific Publications: Oxford, 1991; 139–168.
4. Darbyshire, J.F. Effect of water suctions on the growth in soil of the ciliate *Colpoda Steinii* and the bacterium *Azotobacter Chroococcum*. *J. Soil Sci.* **1976**, *27*, 369–376.
5. Meisterfeld, R. The importance of protozoa in a beech forest ecosystem. *Symp. Biol. Hung.* **1986**, *33*, 291–299.
6. Schönborn, W. The role of protozoan communities in freshwater and soil ecosystems. *Acta Protozool.* **1992**, *31*, 11–18.
7. Elliott, E.T.; Hunt, H.W.; Walter, D.E. Detrital food–web interactions in North American grassland ecosystems. *Agr. Ecosyst. Environ.* **1988**, *24*, 41–56.
8. Darbyshire, J.F. *Soil Protozoa*; CAB International: Wallingford, 1994; 1–209.
9. Bardgett, R.D.; Griffiths, B.S. Ecology and biology of soil protozoa, nematodes, and microarthropods. In *Modern Soil Microbiology*; Van Elsas, J.D., Trevors, J.T., Wellington, E.M.H., Eds.; Marcel Dekker: New York, 1997; 129–163.
10. Ekelund, F. Enumeration and abundance of mycophagous protozoa in soil, with special emphasis on heterotrophic flagellates. *Soil Biol. Biochem.* **1998**, *30*, 1343–1348.
11. Griffiths, B.S. Microbial-feeding nematodes and protozoa in soil: Their effects on microbial activity and nitrogen mineralization in decomposition hotspots and the rhizosphere. *Plant Soil* **1994**, *164*, 25–33.
12. Bonkowski, M. Protozoa and plant growth: The microbial loop in soil revisited. *New. Phytol.* **2004**, *162*, 617–631.
13. Schnürer, J.; Clarholm, M.; Rosswall, T. Fungi, bacteria and protozoa in soil from four arable cropping systems. *Biol. Fert. Soils* **1986**, *2*, 119–126.
14. Singh, B.N. The effect of artificial fertilizers and dung on the numbers of amoebae in Rothamsted soils. *J. Gen. Microbiol.* **1949**, *3*, 204–210.
15. Gupta, V.V.S.R.; Germida, J.J. Populations of predatory protozoa in field soils after 5 years of elemental S fertilizer application. *Soil Biol. Biochem.* **1988**, *20*, 787–792.

16. Kuserk, F.T. The relationship between cellular slime moulds and bacteria in forest soils. *Ecology* **1980**, *61*, 1474–1485.
17. Lousier, J.D.; Parkinson, D. Annual population dynamics and production ecology of Testacea (Protozoa, Rhizopoda) in an aspen woodland soil. *Soil Biol. Biochem.* **1984**, *16*, 103–114.
18. Bryant, R.J.; Woods, L.E.; Coleman, D.C.; Fairbanks, B.C.; Mclellan, J.F.; Cole, C.V. Interactions of bacterial and amoebal populations in soil microcosms with fluctuating moisture content. *Appl. Environ. Microbiol.* **1982**, *43*, 747–752.
19. Parker, L.W.; Freckman, D.W.; Steinberger, V.; Driggers, L.; Whitford, W.G. Effects of simulated rainfall and litter quantities on desert soil biota: Soil respiration, microflora, and protozoa. *Pedobiologia* **1984**, *27*, 185–195.
20. Feest, A. The quantitative ecology of soil mycetoza. *Prog. Protistol.* **1987**, *2*, 331–361.
21. Griffiths, B.S.; Caul, S. Migration of bacterial-feeding nematodes, but not protozoa, to decomposing grass residues. *Biol. Fert. Soils* **1993**, *15*, 201–207.
22. Stout, J.D. Some observations on the protozoa of some beechwood soils on the Chiltern Hills. *J. Anim. Ecol.* **1963**, *32*, 281–287.
23. Lüftenegger, G.; Foissner, W.; Adam, H. r- and K-Selection in soil ciliates: A field and experimental approach. *Oecologia* **1985**, *66*, 574–579.
24. Forge, T.A.; Berrow, M.L.; Darbyshire, J.F.; Warren, A. Protozoan bioassays of soil amended with sewage sludge and heavy metals, using the common soil ciliate *Colpoda steinii*. *Biol. Fert. Soils* **1993**, *16*, 282–286.
25. Lousier, J.D.; Parkinson, D. Evaluation of a membrane-filter technique to count soil and litter Testacea. *Soil Biol. Biochem.* **1981**, *13*, 209–213.
26. Lüftenegger, G.; Petz, W.; Foissner, W.; Adams, H. The efficiency of direct counting the numbers of microscopic soil animals. *Pedobiologia* **1988**, *31*, 95–102.
27. Griffiths, B.S.; Ritz, K. A technique to extract, enumerate and measure protozoa from mineral soils. *Soil Biol. Biochem.* **1988**, *20*, 163–173.
28. Rønn, R.; Ekelund, F.; Christensen, S. Optimizing soil extract and broth media for MPN-enumeration of naked amoebae and heterotrophic flagellates in soil. *Pedobiologia* **1994**, *39*, 10–19.
29. Rutherford, P.M.; Juma, N.G. Influence of soil texture on protozoa-induced mineralization of bacterial carbon and nitrogen. *Can. J. Soil Sci.* **1992**, *72*, 183–200.
30. Frostegård, Å.; Petersen, O.; Bååth, E.; Nielsen, T. Dynamics of a microbial community associated with manure hot-spots as revealed by phospholipid fatty acid analysis. *Appl. Environ. Microbiol.* **1997**, *63*, 2224–2231.
31. Vrba, J.; Šimek, K.; Nedoma, J.; Hartman, P. 4-Methylumbelliferyl-β-N-acetylglucosaminide hydrolysis by a high-affinity enzyme, a putative marker of protozoan bacterivory. *Appl. Environ. Microbiol.* **1993**, *59*, 3091–3101.
32. Zarda, B.; Mattison, G.; Hess, A.; Hahn, D.; Höhener, P.; Zeyer, J. Analysis of bacterial and protozoan communities in an aquifer contaminated with monoaromatic hydrocarbons. *FEMS Microbiol. Ecol.* **1998**, *27*, 141–152.
33. Van Hannen, E.J.; Mooij, W.; Van Agterveld, M.P.; Gons, H.J.; Laanbroek, H.J. Detritus-dependant development of the microbial community in an experimental system: Qualitative analysis by denaturing gradient gel electrophoresis. *Appl. Environ. Microbiol.* **1999**, *65*, 2478–2484.
34. Clarholm, M. Soil protozoa: An under-researched microbial group gaining momentum. *Soil Biol. Biochem.* **2005**, *37*, 811–817.

Pyrite

Michael J. Borda

Department of Plant and Soil Sciences, University of Delaware, Newark, Delaware, U.S.A.

Abstract

Pyrite (iron disulfide, FeS_2) is both a geologically and an environmentally relevant mineral, and is ubiquitous in nature, with varied implications. Its occurrence ranges from hydrothermal systems, to metamorphic systems, to anoxic sedimentary systems. FeS_2 oxidation is known to be a cause of acid sulfate soils, acid mine and, rock drainage with significant environmental harm, in addition to being implicated as a key mineral in the origin of life and as a technologically relevant material. It is, by any account, an important earth material with a rich future in scientific research.

INTRODUCTION

Pyrite (iron disulfide, FeS_2) is a ubiquitous mineral, both temporally and spatially, in the geologic record. It is the most common disulfide mineral in nature and is associated with nearly every ore body type. FeS_2 occurs in metamorphic, igneous, and sedimentary rocks and is also found in extant-reducing environments, such as anoxic soils.

The name “pyrite” is derived from the ancient Greek word for fire, “pyros,” because of the generation of sparks when the mineral is struck. FeS_2 has historically been used for the production of chemicals and also as a material for jewelry manufacturing. Early in the American history, it was polished by Native Americans and used as mirrors. FeS_2 was of interest both as a means for producing sulfuric acid and as a source of iron. Sulfuric acid is generated through the controlled oxidation of FeS_2 by combustion in the presence of air. In the past, FeS_2 was regarded as a source of iron ore; however, it is only used in countries where oxide ores are unavailable. As new sources of sulfuric acid have been discovered and developed, the use of FeS_2 has dramatically decreased. It is mainly considered a nuisance material generated as a fraction of the waste rock during ore and coal mining operations, although there is some interest in FeS_2 for technological purposes because of its semiconducting properties.

MINERALOGY AND CRYSTAL CHEMISTRY

FeS_2 is isometric ($2/\text{m}^3$) and typically occurs as crystals of several different forms. Commonly, it is found in a cubic crystalline habit where pentagonal dodecahedral (pyritohedral) and octahedral forms have also been observed. FeS_2 is known to form framboids, or raspberry-shaped clusters, typically in sedimentary environments (Fig. 1). In the cubic form, faces are often striated because of oscillatory growth between cubic and pyritohedral

forms. FeS_2 is one of the two dimorphs with FeS_2 composition, the second being marcasite. Marcasite is typically formed at lower temperatures and, more characteristically, under lower pH conditions ($\text{pH} < 4.5$). A stability diagram for the S–Fe– H_2O system is shown in Fig. 2; however, marcasite is not shown because FeS_2 is a thermodynamically favorable species under the conditions represented.

FeS_2 is brittle and is unusually hard for a sulfide mineral ($H = 6\text{--}6.5$). It occurs as a pale brass yellow to yellow-white color, whereas specimens exposed to oxic conditions tend to look gray because of surface oxidation (tarnish). FeS_2 has a modified sodium chloride structure with iron at the sodium cubic corner and face-center sites and disulfide in the chlorine sites. Each iron atom in FeS_2 is octahedrally coordinated to six sulfur atoms, and each sulfur atom is tetrahedrally linked to one sulfur atom and three iron atoms (Fig. 3).

FeS_2 is a semiconductor, as defined by its relatively small band gap of 0.95 eV. The band gap is defined as the energy difference between the highest completely filled band (valence band) and the lowest completely unoccupied band (conduction band). The valence band of FeS_2 consists of iron 3d valence electrons and sulfur 3p valence electrons. Because of crystal field splitting, the iron d orbitals are separated into the lower-energy t_{2g} levels and the higher-energy e_g levels. The iron in FeS_2 is present as Fe^{2+} in a low-spin state where all 6d electrons are paired in the t_{2g} energy level. The sulfur in FeS_2 is present as S^{2-} , where two sulfurs make up the S_2^{2-} moiety.

OCCURRENCE

FeS_2 is the most common sulfide mineral and is found worldwide. Important localities occur in the United States, Spain, and Peru. FeS_2 forms over a range of temperatures, with the high-temperature forms being dominant in volume. One of the largest sources is massive sulfide

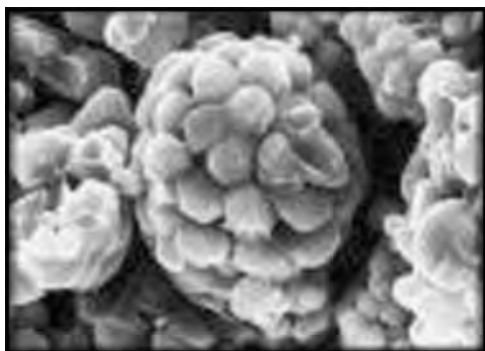


Fig. 1 Scanning electron microscopic image of framboidal pyrite.

Source: Photo courtesy of the U.S. Geological Survey.

deposits associated with hydrothermal activity. In some high-temperature systems, such as epithermal veins (a hydrothermal mineral deposit formed within ~ 1 km of the earth's surface and in the temperature range of 50 – 200°C), the tendency is to form euhedral (i.e., having well-formed faces) crystals ranging in size up to 10 cm. The largest FeS_2 crystals are typically associated with metamorphism and can exceed 20 – 30 cm.

FORMATION

The formation of FeS_2 has been recognized to proceed via two possible mechanisms. One mechanism is the direct nucleation of FeS_2 from solution, whereas the second mechanism is through a replacement reaction generating

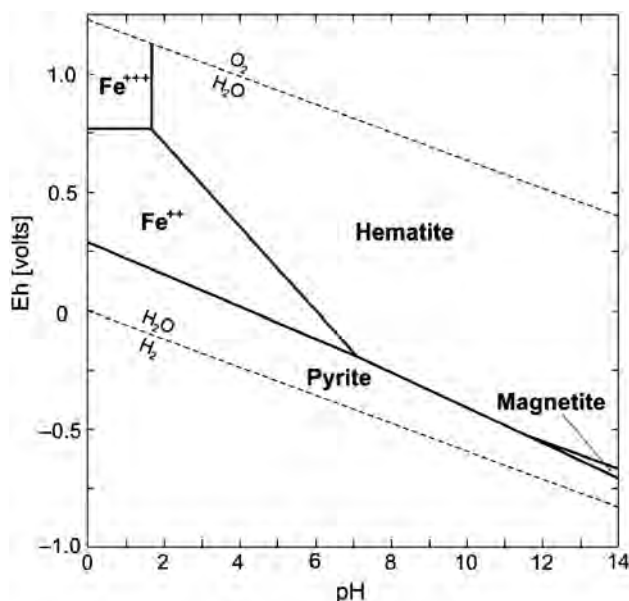


Fig. 2 A stability field diagram for the system $\text{Fe}^{2+}/\text{SO}_4^{2-}/\text{H}_2\text{O}$ where the activities of Fe^{2+} and SO_4^{2-} are set to 1×10^{-5} M. FeS_2 , hematite, and magnetite are the possible phases in the system, and the stability field of H_2O is shown for clarity.

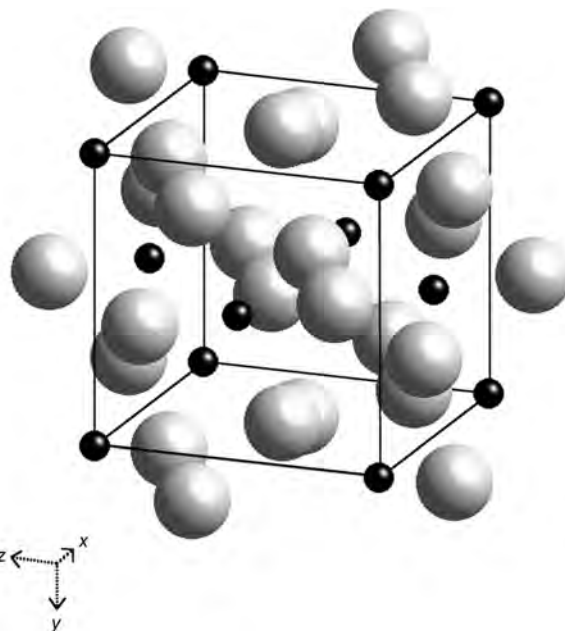


Fig. 3 Schematic image of the FeS_2 structure where black circles correspond to iron atoms and gray circles correspond to sulfur atoms.

Source: From Wyckoff Crystal Structures, 1969 (I), 346 (IV, g1).

FeS_2 from an iron monosulfide (FeS) precursor phase. The existence of euhedral crystals in hydrothermal systems has been one of the arguments for a direct precipitation mechanism. Experimental work has shown that the formation of FeS_2 through direct nucleation is insignificant compared with the formation via a FeS precursor.^[1] The source for sulfur species can be either hydrogen sulfide ($\text{H}_2\text{S}/\text{HS}^-$), thiosulfate, or polythionate species. This adds an important constraint on systems where FeS_2 is forming. FeS species are far more soluble and allow higher concentrations of iron and sulfur to be concentrated in systems before monosulfide formation occurs. Once the precursor FeS is formed, FeS_2 formation is facile and can produce macroscopic crystals.

An important texture of FeS_2 typically associated with soil environments and ore deposits is the framboidal form. Framboids are spherical structures composed of a number of microcrystals. The formation mechanism for framboidal FeS_2 is thought to proceed through a precursor phase, called greigite. However, some experimental works have shown that this phase is not a prerequisite, and framboidal FeS_2 can form through the reaction of amorphous FeS and H_2S .^[2]

REACTIVITY

The reactivity of FeS_2 , namely the oxidation reaction to form sulfuric acid and iron oxides, is a significant environmental concern. This process generates acid sulfate soils,

acid mine drainage, and non-anthropogenic acid rock drainage and may have harmful consequences for the entire watersheds that are affected. The oxidation of FeS_2 commonly proceeds because of the drainage of anoxic soils and sediments or through the excavation of pyritic materials during mining and subsequent exposure to aerobic surface conditions. Two main oxidants are important: molecular oxygen and ferric iron (Fe^{3+}); however, often overlooked is the significance of water. The reaction is catalyzed by bacteria (*Acidithiobacillus ferrooxidans*) through the oxidation of Fe^{2+} (product) back to Fe^{3+} (reactant). The products of oxidation reactions cause the acidification of streams draining the area affected and can result in the mobilization and transport of associated heavy metals. This ultimately has resulted in a \$1,000,000 per day environmental problem in the United States alone.^[3]

IMPLICATIONS

From an environmental standpoint, FeS_2 is a material that must be understood to develop abatement strategies to halt the acidification process. FeS_2 is a promising photocatalyst for environmental contaminant degradation and has also played a critical role in other fields. In the late 1980s, a hypothesis that the formation of FeS_2 may be the driving force for the chemoautotrophic (bacteria that fix carbon dioxide to make their own food source using fuel from chemical reactions) origin of life was proposed.^[4] FeS_2 has also been implicated as being a source of hydrogen peroxide on an early, anoxic earth, possibly playing a role in the evolution of oxygenic photosynthetic organisms.^[5] Along with its importance to the earth history, FeS_2 is also a technologically important material. Because of its semiconducting properties, FeS_2 is important to photoelectrochemistry and photocatalysis.

CONCLUSION

FeS_2 is both a geologically and an environmentally relevant mineral, and is ubiquitous in nature, with varied

implications. Its occurrence ranges from hydrothermal systems, to metamorphic systems, to anoxic sedimentary systems. FeS_2 oxidation is known to be a cause of acid sulfate soils acid mine/rock drainage with significant environmental harm, in addition to being implicated as a key mineral in the origin of life and as a technologically relevant material. It is, by any account, an important earth material with a rich future in scientific research.

ACKNOWLEDGMENTS

I would like to thank Dr. Martin A. Schoonen at Stony Brook University for giving me the opportunity to work on such a wonderful mineral and for the support and encouragement. I thank Alexander Smirnov for constructing the figures for this entry. I also thank Jerry Bigham for an excellent and constructive review and Rattan Lal for the opportunity to write this entry.

REFERENCES

1. Schoonen, M.A.A.; Barnes, H.L. Mechanisms of pyrite and marcasite formation from solution: I. nucleation of FeS_2 below 100°C. *Geochim. Cosmochim. Ac.* **1991**, *55*, 3491–3504.
2. Butler, I.B.; Rickard, D. Framboidal pyrite formation via the oxidation of iron (II) monosulfide by hydrogen sulphide. *Geochim. Cosmochim. Ac.* **2000**, *64* (15), 2665–2672.
3. Evangelou, V.P. Pyrite microencapsulation technologies: principles and potential field application. *Ecol. Eng.* **2001**, *17* (2–3), 165–178.
4. Wächtershäuser, G. Pyrite formation, the first energy source of life: A hypothesis. *Syst. Appl. Microbiol.* **1988**, *10*, 207–210.
5. Borda, M.J.; Elsetinow, A.R.; Schoonen, M.A.; Strongin, D.R. Pyrite-induced hydrogen peroxide formation as a driving force in the evolution of photosynthetic organisms on an early earth. *Astrobiology* **2001**, *1* (3), 283–288.

Pyrogenic Carbon

Barry G. Rawlins

British Geological Survey, Nottingham, U.K.

Abstract

This entry describes the range of substances in the soil that are encompassed by the term “pyrogenic carbon” (PyC). The two main processes for its generation are wildfire and purposeful charcoal/biochar production. It describes how it is quantified and characterized, with reference to further methodological advances. The impact of PyC on soil properties and the occurrence of PyC in soils in global ecosystems are described. The potential importance of PyC for climate change are presented, identifying a series of key knowledge gaps relating to PyC. It concludes by commenting on the paucity of research undertaken on PyC in the soil and identify the focus of such research in the forthcoming years.

INTRODUCTION

Substances referred to as pyrogenic carbon (PyC) are generally considered to form a continuum from charred plant materials, through charcoal, with an end point of graphite. Its common characteristic is an abundance of condensed aromatic ring structures and—when generated under higher temperatures—larger graphene sheets (Fig. 1). PyC is created naturally by wildfire and the combustion of fossil fuels and has been created by man for centuries as what is commonly referred to as charcoal. Interest in PyC has increased because of the need to better understand the global carbon (C) cycle and its potential role in enhancing C sequestration in the biosphere due to its greater chemical recalcitrance when compared with more common, labile soil C compounds, with implications for climate change.^[1] The term “pyrogenic carbon” that has been used in the soil science literature since around the year 2000 includes materials commonly referred to as black carbon (BC), elemental C, and biochar. The latter term has been used since around 2007 to refer to the purposeful charring of biomass with the intention that it is applied to soil to improve fertility, C storage, and filtration of percolating water.

GENERATION OF PyC

Two types of PyC are typically generated—soot condensates and solid residues. Soot comprises small molecules released by pyrolysis and recombination by free radical reactions. Solid residues of PyC form largely during the smoldering phases of combustion where fuel gases produced by pyrolytic (oxygen-depleted) reactions are insufficient to maintain a flame. Wildfire temperatures leading to the formation of PyC typically vary between 200°C and 500°C while ancient societal practices of charcoal production have been adapted using modern technology to

generate PyC (charcoal/biochar) residues on both local and industrial scales.^[2]

Measurement and Characterization

The chemical compositions of PyC and the terms used to define its components are shown in the van Krevelen diagram (Fig. 2). Methodologies have been developed to discriminate the BC component of PyC from two other forms of C in the soil: inorganic C and unaltered organic C from the decomposition products of plant materials. The first step in many methods for quantifying the BC in soil is the removal of inorganic C by pretreatment with hydrochloric acid. Many approaches for quantifying BC in soil use a variety of means to oxidize the unaltered soil C including heating, acids, or photooxidation followed by analysis of the residue for C. An alternative approach is the use of molecular markers such as benzene polycarboxylic acids. Further details of these and other approaches were discussed in greater detail.^[3] Methods that can provide more detail of PyC material along its continuum are being developed^[4] based on measuring the degree of aromatic condensation. Broad chemical characterization of PyC is also often undertaken by analysis with ¹³C nuclear magnetic resonance spectroscopy.

Influence on the Soil

PyC has several important characteristics that influence the properties of soils when it occurs in reasonable quantities. First, it is more recalcitrant than thermally unaltered soil C and so persists in soil for longer periods; because of its chemical composition, it is resistant to microbial decay and degrades slowly by abiotic processes. Numerous studies have attempted to estimate the longevity of PyC in soil; these range from a few decades to several thousand years depending on soil conditions.^[5] Second, it has a very large

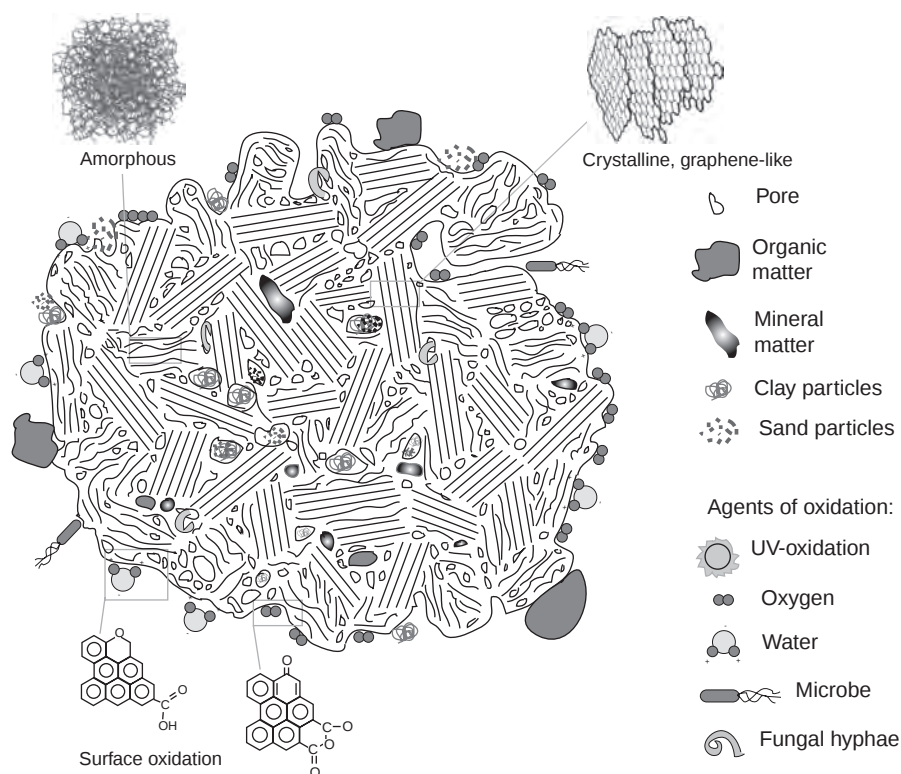


Fig. 1 Model of a PyC particle in soil comprising both crystalline graphene sheets surrounded by amorphous aromatic structures, with a range of pore sizes.

Source: Reproduced with permission of Earthscan Ltd., www.earthscan.co.uk.

surface area (ranges from 50 to 400 m² g⁻¹) due to its highly porous structure. This can markedly increase the water-holding capacity of the soil. After incorporation into the soil, the charged surfaces of PyC from the formation of functional groups contribute to progressively increasing soil cation-exchange capacities, improving fertility in agricultural soil. Some studies have suggested that greenhouse gas emissions

of nitrous oxide and methane decline when PyC has been applied to the soil.^[6] The charged surfaces of PyC enhance sorption with clay mineral surfaces (Fig. 3), leading to increased aggregation and improved soil structural characteristics. PyC is known to have a significant impact on mycorrhizal fungi in soil where they form intimate associations. Mechanisms through which PyC influence mycorrhiza in soil include the alteration of soil physical and chemical properties (fine pores providing a refuge), indirect effects through impacts on other soil microbes, and changing

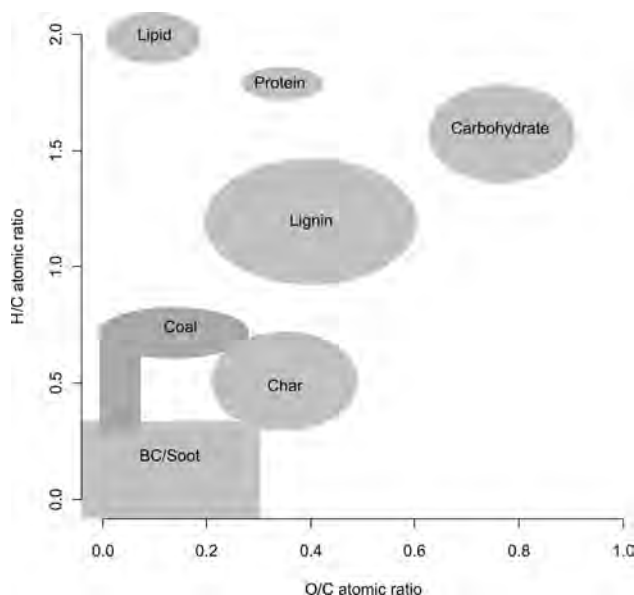


Fig. 2 van Krevelen plot of molar ratios for biomass and pyrogenic materials.

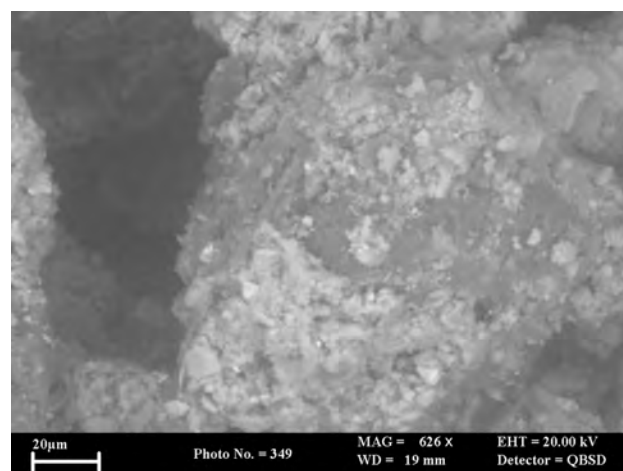


Fig. 3 Scanning electron micrographic image of a PyC particle in the soil with a coating of clay minerals determined by energy dispersive X-ray analysis.

plant–fungus signaling;^[7] however, there is little understanding of the relative importance of these mechanisms.

Occurrence of PyC in Soil

The quantity of PyC in soils generated by wildfire is related to the ecosystem type; the proportions of PyC in relation to total soil organic carbon (SOC) for forested and non-forested ecosystems have been summarized.^[8] For example, PyC comprises up to 33% of SOC in Japanese grasslands subject to frequent burning and up to 82% in the topsoils in parts of Australia.^[9] Where PyC has been generated by man and applied to the land forming the terra preta soils of the South America, PyC comprises up to 20% of SOC. Studies have suggested that physical breakdown of PyC residues results in particles with sizes finer than 50 μm .

Impacts on Climate Change

Greater understanding on the quantities and forms of PyC in soil has the potential to improve our understanding of feedbacks between projections of climate change and terrestrial ecosystems, including emission of carbon dioxide (CO_2) from the soil. A further study demonstrated that by incorporating accurate stocks of BC in climate prediction models, CO_2 emissions were reduced by 18.3% and 24.4% in two Australian savannah regions in response to a warming of 3°C over 100 years.^[9] In the case of biochar production of PyC and its purposeful incorporation into the soil, estimates have been made of the scale this contribution could make to terrestrial C sequestration. By adopting slash-and-char policies at a global scale, rather than slash-and-burn that releases all C to the atmosphere, 0.21 peta grams of carbon (PgC) could be sequestered in the soil each year. Using all agricultural, forestry, and urban wastes to create biochar and applying it to soil could sequester 0.16 PgC yr^{-1} , while if a proportion of renewable energy demand was met through pyrolysis, the figure increases substantially to 5.4 PgC yr^{-1} .^[10]

Knowledge Gaps

In areas prone to wildfire and managed burning, few data are available on the pools and fluxes of PyC, specifically that proportion converted into solid PyC and soot PyC. In terms of the C cycle, the fate of PyC residues is unknown—is it subsequently consumed by fire or oxidized, and if so at what rate? There is a lack of quantitative data on PyC storage in boreal peatlands. Losses of PyC to the atmosphere need to be tracked and its fate determined.

As a means of increasing soil C sequestration, some advocate vastly increasing the generation of biochar (PyC) and its application to soil yielding agronomic benefits. Despite the potential positive features of biochar, others are more cautious suggesting that little is known of its long-term impacts on soil properties and functions. In particular,

there are many outstanding questions concerning the impact of PyC on soil microbial communities, its long-term stability and dynamics in soil, its impact on the links between the C and nitrogen cycles (where models need to be developed), and its erosion, transport, and fate.

CONCLUSION

Despite the potential importance of PyC in the global C cycle and its implications for understanding the magnitude of climate change due largely to rising atmospheric CO_2 concentrations, remarkably little research has been undertaken on the biogeochemical cycle of soil PyC. Much of the research on characterization, impacts on soil properties, and fate will likely stem from the further increase in biochar-related research. Methods for characterizing PyC are likely to improve measurement along its continuum and aid understanding of how pyrolysis temperatures, duration, and feedstock influence its behavior and properties in the soil. This should enable a more thorough system for the classification of PyC substances in the soil and aid modeling of its turnover.

REFERENCES

1. Lehmann, J. A handful of carbon. *Nature* **2007**, *447*, 143–144.
2. Brown, R. Biochar production technology. In *Biochar for Environmental Management*; Lehmann, J., Joseph, S., Eds.; Earthscan: London, 2009; 127–146.
3. Manning, D.; Lopez-Capel, E. Test procedures for determining the quantity of biochar within soils. In *Biochar for Environmental Management*; Lehmann, J., Joseph, S., Eds.; Earthscan: London, 2009; 310–316.
4. McBeath, A.; Smernik, R. Variation in the degree of aromatic condensation of chars. *Org. Geochem.* **2009**, *40*, 1161–1168.
5. Lehmann, J.; Czimczik, C.; Laird, D.; Sohi, S. Stability of biochar in soil. In *Biochar for Environmental Management*; Lehmann, J., Joseph, S., Eds.; Earthscan: London, 2009; 183–206.
6. Yanai, Y.; Toyota, K.; Okazaki, M. Effects of charcoal addition on N_2O emissions from soil resulting from rewetting air-dried soil in short-term laboratory experiments. *Sci. Plant Nutr.* **2007**, *53*, 181–188.
7. Warnock, D.D.; Lehmann, J.; Kuyper, T.W.; Rillig, M.C. Mycorrhizal responses to biochar in soil—concepts and mechanisms. *Plant Soil* **2007**, *300*, 9–20.
8. Preston, C.M.; Schmidt, M.W.I. Black (pyrogenic) carbon: A synthesis of current knowledge and uncertainties with special consideration of boreal regions. *Biogeosciences* **2006**, *3*, 397–420.
9. Lehmann, J.; Skemstad, J.; Sohi, S.; Carter, J.; Barson, M.; Falloon, P.; Coleman, K.; Woodbury, P.; Krull, E. Australian climate–carbon cycle feedback reduced by soil black carbon. *Nat. Geosci.* **2008**, *1*, 832–835.
10. Lehmann, J.; Gaunt, J.; Rondon, M. Bio-char sequestration in terrestrial ecosystems—A review. *Mitigat. Adapt. Strateg. Glob. Chang.* **2006**, *11*, 403–427.

Quality

Ed G. Gregorich

*Eastern Cereal and Oilseed Research Centre, Agriculture and Agri-Food Canada,
Ottawa, Ontario, Canada*

Abstract

Depending on the use, soil quality may mean different things to different people. For example, in the agricultural context, within which much of the discussion of soil quality has taken place, the farmer may think of soil quality in terms of productive land that maintains or increases farm profitability, as well as the conservation of farm resources so that future farming generations can make a living. To the consumer, agricultural soil quality may mean a dependable supply of wholesome, affordable food and the protection of drinking water quality. To the environmentalist, it may be defined in terms of global nutrient cycling, water regulation, climate change, and biodiversity. And for high-level policy and decision makers, soil quality may be seen in terms of a nation's ability to stay competitive in the international marketplace and to honor its international environmental commitments.

INTRODUCTION

Concisely defined, soil quality is the degree of fitness of a soil for a specific use—its ability or capacity to function for a specific purpose.^[1,2]

BASIC SOIL FUNCTIONS

Soil performs six main functions: sustaining plant and animal life, regulating water, regulating gases, regulating energy, buffering against or filtering out potential environmental contaminants, and providing physical support for human structures such as buildings and roads. How well a soil carries out these functions determines its quality.

INHERENT AND DYNAMIC SOIL QUALITY

Soil quality has both an inherent or natural element determined by geological materials and soil formation processes (such as chemical and physical weathering and a dynamic element determined by management practices). Natural processes, such as erosion and the subsequent loss of organic matter, compaction, and salinization can degrade the inherent quality of soil. Human activity, such as agriculture, can accelerate these processes through various land use and management practices, hastening the symptoms and effects of soil degradation. On the other hand, some agricultural land uses and practices (such as various tillage methods, cropping systems, and nutrient management plans) help to stabilize or improve soil quality.

SUSTAINABLE DEVELOPMENT

The interest in soil quality is linked to sustainable development. A soil management system is considered sustainable only if it maintains or improves soil quality and does not compromise environmental quality beyond a degree acceptable to society. Only by understanding soil quality and promoting it can we achieve environmental protection and preserve soils in a usable state for future generations. Regular monitoring of soil quality allows land managers to assess the sustainability of various land uses and management practices.

DEVELOPING A FRAMEWORK TO ASSESS SOIL QUALITY

Soils are the interface between terrestrial ecosystems and both aquatic ecosystems and the atmosphere. Based on the linkages between ecosystems, soil quality is normally defined using environmental and economic performance criteria, including those of plant and animal health and productivity. Soil quality and soil health are often considered synonymous, but some researchers make a distinction between these terms by considering soil health as a subset of soil quality and by stressing that the focus of soil health is on the organic and more dynamic soil properties. This latter concept expands to a larger view of soils as a key component for sustainably supporting all terrestrial ecosystems.

A useful framework for assessing soil quality begins with a description of the functions on which such quality is to be based (i.e., those most likely to influence the performance criteria). For example, at the ecosystem or

global scale, soil functions include regulating both biotic processes (e.g., supplying plants with mineral nutrients and water) and the flux of elements [e.g., the turnover and storage of carbon (C), nitrogen, phosphorus, and sulphur]; mitigating the accumulation of greenhouse gases [e.g., carbon dioxide (CO₂), NO_x, and methane (CH₄)]; altering the chemical composition of precipitation and distributing water throughout the environment; contributing to the gas, water, and heat balance of the atmosphere; and serving as a biological habitat and genetic reserve.^[3,4]

Once soil functions are known, soil characteristics or properties that support these functions, along with important interactions of these characteristics, must be identified.^[5] Much of the work to characterize soil quality has been carried out for agricultural soils. Table 1 lists some of the physical, chemical, and biological characteristics and processes of agricultural soils that contribute to the soil quality functions that support agricultural productivity and sustainability.

Table 1 Some soil quality functions that are important for crop production.

Soil quality functions	Function characteristics and processes involved
Support plant growth	Provide suitable medium for seed germination and root growth
Show absence of adverse chemical conditions (e.g., acidity, salinity, sodicity)	
Supply balance of nutrients	
Provide suitable medium for microorganisms (for decomposition, nutrient cycling)	
Promote root growth and development	
Regulate water	Receive, store, and release water for plant use
Retain water adequately to buffer and reduce effects of drought	
Provide adequate infiltration and storage capacity to reduce runoff	
Regulate gases	Accept, hold, and release gases
Have conditions for adequate air exchange with atmosphere	
Regulate energy	Store energy-rich organic matter
Decompose, mineralize, and recycle nutrients and energy	
Buffer or filter	Accept, hold, and release nutrients
Sequester biotoxic elements	

Source: Adapted from Carter & Gregorich.^[5]

EVALUATING SOIL QUALITY

It is not feasible or even practical to measure all soil properties in order to assess the soil quality. Soil quality indicators are needed that encompass the whole range of physical, chemical, and physical properties of the soil; reflect soil function; are easy to measure for a variety of users and under various field conditions; and respond to changes in climate and management. The concepts underlying the criteria to select key indicators of soil quality, along with the interpretative framework needed to link the response of the measured indicator to management action and the issue of concern, are described in another section of the encyclopedia. Assessment of the indicators is critical in determining whether a soil process is unrestricted or a soil is fit for use. The approaches used to set quantitative critical limits or ranges of soil indicators, which help determine whether the value of the indicator adequately describes soil performance, are discussed in another section of the encyclopedia.

Land management practices used in agricultural production usually result in a degradation of the structures, or a perturbation of the processes, that help to maintain natural ecosystems in equilibrium. The inherent and dynamic aspects of soil quality that help accurately assess sustainable crop and animal production, and the link between soil quality and development of sustainable management systems (i.e., those that foster reduction in the inputs of non-renewable resources, maintain acceptable levels of productivity, and minimize impact on the environment) is described in other sections of the encyclopedia.

The framework used to interpret and assess soil quality must also be useful in evaluating effects in adjacent areas or systems. A description of soil properties important to the process of erosion and the consequences of erosion on soil functions, the relationship between soil quality and the quality of receiving waters, and the effects of both soil management and soil quality on the soil processes that produce and consume carbon and nitrogen gases are discussed other sections of the encyclopedia.

MONITORING SOIL QUALITY

In natural systems, soil quality is observed as a baseline value or set of values against that future changes in the system can be analyzed and compared. In agricultural systems, soil quality is monitored with the view toward managing the system to enhance production while not degrading soils and the environment.

The selection and quantification of properties used to assess soil quality are often based on their spatial and temporal variability, on whether they are desirable for crop production, and on whether they can be controlled or managed in a time frame relevant for annual crop production (i.e., relatively short-term) or for long-term sustainability.^[6] A set of basic soil properties and characteristics has been

Table 2 Some soil physical, chemical, and biological properties used as indicators of soil quality.

Soil property	Method	Information
Physical properties		
Soil texture	Hydrometer method	Retention and transport of water, nutrients, and chemicals; susceptibility to erosion; stabilization of organic matter and soil structure
Topsoil and rooting depth	Soil coring, excavation	Water and nutrient availability, rooting volume for crop production
Soil bulk density	Soil core, neutron probe	Volume of pore space, compaction
Infiltration, hydraulic conductivity	Pressure, tension infiltrometer; permeameter; laboratory measurement on cores	Runoff and leaching potential, drainage
Water content	Gravimetric measurement by oven drying; volumetric measurement by time domain reflectometry or core	Available water
Water release curve, water-holding capacity	Water content at 33 and 1500 kPa tension	Availability of air and water, retention and transport of water and chemicals, drainage
Chemical properties		
pH	pH meter	Acidity or alkalinity of soil, nutrient availability
Electrical conductivity	Conductivity meter	Presence and quantity of soluble salts
Available N, P, and K	Laboratory extraction and analysis	Plant available nutrients
Organic C and N	Wet or dry combustion	Organic matter reserves, nutrient cycling, soil structure
Biological properties		
Potentially mineralizable N	Aerobic or anaerobic incubation	Potential to supply plant-available N
Microbial biomass	Chloroform fumigation followed by incubation or extraction	Size of microbiological population, pool of rapidly cycling organic matter and nutrients
Soil respiration	<i>In situ</i> and laboratory measurements by CO ₂ gas analyzer or alkali traps	Availability of soil organic matter reserves, microbiological activity

Source: Adapted from Doran & Parkin.^[7]

proposed as indicators of soil quality.^[7] Table 2 lists a set of indicators that forms the core of most data sets used to characterize soil quality. From this data set, it is possible to derive other relevant indicators such as available water, microbial quotient, and respiratory quotient. It has been proposed that a suitable level of precision for soil quality indicators is the ability to detect a 10% change at the 90% confidence level.^[8]

Some indicators are less useful than others because of high temporal and/or spatial variability, because of strong correlation to other indicators, or because the response of the indicators to change is not readily interpreted.^[8] Selecting an appropriate suite of indicators or adopting surrogate indicators for a particular soil quality assessment is an important first step in monitoring soil quality, and there is a need to balance the cost of measuring particular indicators against the value of the information obtained.

Statistical quality control methods have been proposed to determine the range of variation and to set control limits for properties^[9] in order to provide the farmer with information to manage the soil in a sustainable manner. The theory behind these methods is that if upper and lower control limits, based on known or desired tolerances, can be established to represent minimum levels for sustainable

Table 3 Soil properties that can be estimated from input variables using pedotransfer functions.

Soil property	Basic input variable ^a
Cation exchange capacity	Clay type and quantity + organic carbon
Water retention characteristic; hydraulic conductivity; volumetric water content	% Sand, silt, clay + organic carbon + bulk density
Aerobic and anaerobic microbial activity	Water-filled pore space (calculated from bulk density, particle density, and water content)
Soil strength	Bulk density and water content
Rooting depth	Bulk density, water-holding capacity, pH, electrical conductivity
Leaching potential	% Sand, silt, and clay; pH; organic carbon; hydraulic conductivity, cation exchange capacity; depth

^aVariable used in pedotransfer function.

Source: Adapted from Doran & Parkin.^[11]

production, then changes in critical parameters can be identified and corrective action taken to adjust the management system.

When data are unavailable, pedotransfer functions can be used to estimate values of soil characteristics^[10,11] (Table 3). Pedotransfer functions are functional relationships (i.e., regression models) that transfer known soil properties (e.g., texture, bulk density, organic C content) into unknown soil properties (e.g., soil hydraulic conductivity, volumetric water content). The main advantage of pedotransfer functions is that they are relatively inexpensive and easy to derive and use. The disadvantage of these functions is that some may not be appropriate for application at a specific site and should not be used to make predictions for soils that are outside the range of soils that were used to derive the pedotransfer functions originally.

REFERENCES

1. Larson, W.E.; Pierce, F.J. The dynamics of soil quality as a measure of sustainable management. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezedick, D.F., Stewart, B.A., Eds.; American Society Agronomy: Madison, 1994; 37B51.
2. Gregorich, E.G.; Carter, M.R.; Angers, D.A.; Monreal, C.M.; Ellert, B.H. Towards a minimum data set to assess soil organic matter quality in agricultural soils. *Can. J. Soil Sci.* **1994**, *74*, 367–385.
3. Ellert, B.H.; Clapperton, M.J.; Anderson, D.W. An ecosystem perspective of soil quality. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 115–141.
4. Blum, W.E.H.; Santelises, A.A. A concept of sustainability and resilience based on soil functions. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 535–542.
5. Carter, M.R.; Gregorich, E.G.; Anderson, D.W.; Doran, J.W.; Janzen, H.H.; Pierce, F.J. Concepts of soil quality and their significance. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 1–9.
6. Singer, M.J.; Ewing, S. Soil quality. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; G271–G298.
7. Doran, J.W.; Parkin, T.B. Defining and assessing soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezedick, D.F., Stewart, B.A., Eds.; SSSA Special Publication No. 35; Soil Science Society of America: Madison, WI, 1994; 3–21.
8. Schipper, L.A.; Sparling, G.P. Performance of soil condition indicators across taxonomic groups and land uses. *Soil Sci. Soc. Am. J.* **2000**, *64*, 300–311.
9. Pierce, F.J.; Gilliland, D.C. Soil quality control. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 203–219.
10. Wösten, J.H.M. Pedotransfer functions to evaluate soil quality. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 221–245.
11. Doran, J.W.; Parkin, T.B. Quantitative indicators of soil quality: A minimum data set. In *Methods for Assessing Soil Quality*; Doran, J.W., Jones, A.J., Eds.; SSSA Special Publication No. 49; Soil Science Society of America: Madison, WI, 1996; 25–37.

Quality Index: Soil

Toru Nakajima

Department of Agriculture, Meiji University, Kawasaki, Japan

Abstract

Soil quality index (SQI) assessment has become an internationally accepted science-based tool for advancing the assessment, education, and understanding of soil resources. Soil quality is most often defined as the capacity of soil to function within ecosystem and land use boundaries to sustain productivity, maintain environmental quality, and promote plant growth as well as animal health. SQI assessment requires an evaluation of physical, chemical, and biological attributes to know how well the resource is functioning. Therefore, SQI seeks to characterize the overall agro-ecological functions of soil by selecting soil physical, chemical, and biological properties as indicators, measuring them, and calculating a score or index for both the individual properties and overall soil quality. There are three basic approaches for evaluating the sustainability of agricultural management: 1) comparing different management systems; 2) comparing the same site over time and finding trends; 3) comparing with problem and non-problem areas. Numerous investigators have proposed conceptual frameworks and models to evaluate soil quality including the scoring function analysis. SQI systems approach is needed to integrate the knowledge of soil science into solutions for natural resource problems.

INTRODUCTION

Soil quality indexes (SQIs) have been recognized as assessing tools for land use and soil management practices worldwide.^[1] The soil quality is important not only for the production of food and fiber but also in the maintenance of local, regional, and worldwide environmental quality.^[2] Soil quality is most often defined as the capacity of soil to function within ecosystem and land use boundaries to sustain productivity, maintain environmental quality, and promote plant growth as well as animal health.^[2] The SQI has to reflect the soil physical, chemical, and biological properties and processes and interactions within each soil resource and their interactions within each soil.^[3] Furthermore, SQI can be used to monitor trends in soil properties and functions over time to determine if soil quality under different land use and management is aggrading, sustaining, or degrading soil attributes.^[4] However, there is no universal method or tool to assess soil quality, although investigators have proposed conceptual frameworks and models to evaluate soil quality.^[5–10]

CONCEPT OF SOIL QUALITY ASSESSMENT

Assessing soil quality can help in understanding the emphasis on sustainable land use with a holistic focus emphasizing on sustainable soil managements (e.g., soil erosion tolerance and resiliency of crop products), and the concept has two aspects for education and assessment.^[4] Soil quality assessment and education may provide a better

understanding and awareness for essential ecosystem services.

There are three basic approaches for evaluating the sustainability of agricultural management: 1) comparing different management systems for differences in soil quality; 2) comparing the same site over time and finding trends; and 3) comparing within the areas with problem and non-problem.^[9] The second approach includes assessing changes in the soil properties over time to identify the trends.^[11] To find the trends, establishing baseline values for the various soil properties and monitoring changes of those properties are required. When SQI has positive trend, it indicates that soil is improving or aggrading in soil quality.^[9] Contrary, when SQI has negative trends, then soil is degrading. In addition, a flat trend could indicate sustainable. The processes include establishing criteria and condition to guide the evaluation processes, such as establishing ranges for indicator values that are appropriate for the specific soils, and determining the relative importance or weight that should be given to each indicators.^[9] This indexing soil quality assessment is an important step. It involves selecting appropriate soil quality indicators to efficiently and effectively monitor critical soil functions as determined by specific management goals.

SQI METHOD

Numerous investigators have proposed conceptual frameworks and models to evaluate soil quality.^[5–10] For instance, when calculating a soil quality index (SQI), linear

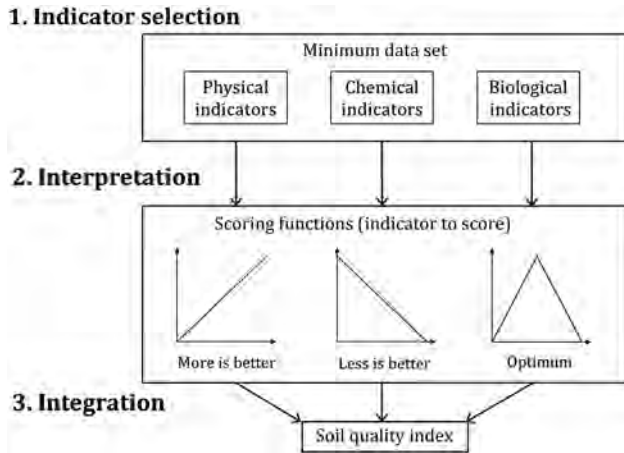


Fig. 1 Conceptual framework for scoring function analysis.

Source: Adapted from Andrews, Karlen, et al.^[5] ©2004 Soil Science Society of America.

and multiple regression analysis,^[12] pedotransfer functions analysis,^[13] principal component analysis (PCA),^[6,10] and scoring function analysis^[14] have been used. This entry describes one of the common methods termed “scoring function analysis” for SQI assessment.

The scoring function analysis for SQI assessment has three main steps, as followed by Andrews and Karlen: 1) identification of the minimum data set of indicators; 2) indicator interpretation; and 3) integration of the all indicator scores into one overall SQI value^[5] (Fig. 1). For the first step, SQI indicators are selected for soil physical, chemical, and biological properties according to the appropriate indicator for the management goal in the study. Doran and Parkin^[2] stated the importance of selecting appropriate indicators that could be measured which influence the capacity of a soil to perform and also be sensitive to the final outcome. In the second step, each indicator from the minimum data set is transformed into unit less combinable score ranging from 0 to 1 (where 1 represents the high level for indicators and contrary 0 represents the low level for indicator). There is a criterion for transforming indicators to scores for accounting their contribution to soil functions. In general, there are three general shapes of standard scoring function for SQI.^[8,15] Increasing the level of the indicators when the quality of the soil is increasing, “the more is better” curve is used. Conversely, “the less is better” curve is suitable for decreasing the level of the indicators by increasing the quality of soil. In addition, “the optimum” curve scores those indicators that have an increasingly positive association with soil quality up to an optimal level beyond which quality of soil decreases^[6] (Fig. 1). After considering the appropriate curve type for indicators, a scoring of functions chart for interpretation of SQI is created.

The third step, the score values (e.g., ranging from 0 to 1) of each soil indicators are given specific weights based on their contribution to management goal. The individual

scores are multiplied by weighting factors and combined by addition into an overall SQI (Fig. 1). The weightings were given subjectively; nevertheless, the soil physical, chemical, and biological properties are given almost equal weighting to emphasize the equal importance of these three categories of soil properties in their contribution to soil functions.

SOIL PHYSICAL, CHEMICAL, AND BIOLOGICAL INDEX FOR SQI

Research on soil quality assessment has focused on indicator selection and evaluation and minimum data set for SQI.^[4] Numerous researches are being conducted to describe accuracy, sensitivity, and contribution of various soil properties and processes. The indicators for SQI assessment must be evaluated based on multiple physical, chemical, and biological properties and their interactions. One aspect of indicator research to select the minimum data set is science-based statistical skill (e.g., PCA and pedotransfer function analysis) to select the suitable indicators. Larson and Pierce^[16] reported that pedotransfer analysis and potential soil indicators include: 1) bulk density; 2) water retention; 3) saturated hydraulic conductivity; 4) porosity; 5) random roughness; 6) soil productivity; 7) root depth; 8) cation exchange capacity; 9) soil organic carbon; and 10) phosphate sorption capacity.^[16] It is needed to be understood how soil properties and processes interact within ecosystems.^[7]

SUMMARY

The SQI assessment has become an internationally accepted science-based tool for advancing the assessment, education, and understating of soil resources.^[4] The SQI has to reflect the soil physical, chemical, and biological properties and processes and interactions within each soil resource. There are three basic approaches for evaluating the sustainability of agricultural management: 1) comparing different management systems; 2) comparing the same site over time and finding trends; 3) comparing with problem and non-problem areas. Numerous investigators have proposed conceptual frameworks and models to evaluate soil quality including the scoring function analysis. SQI systems approach is needed to integrate the knowledge of soil science into solutions for natural resource problems.

REFERENCES

1. Karlen, D.L.; Mausbach, M.J.; Doran, J.W.; Cline, R.G.; Harris, R.F.; Schuman, G.E. Soil quality: A concept, definition, and framework for evaluation (a guest editorial). *Soil Sci. Soc. Am. J.* **1997**, *61*, 4–10.
2. Doran, J.W.; Parkin, T.B. Defining and assessing soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A.,

- Eds.; Soil Science Society of America and American Society of Agronomy: Madison, WI, 1994; 1–21.
3. Karlen, D.L.; Andrews, S.S.; Doran, J.W. Soil quality: Current concepts and applications. *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press: London, 2001; Vol. 7, 1–40.
 4. Karlen, D.L.; Ditzler, C.A.; Andrews, S.S. Soil quality: Why and how? *Geoderma* **2003**, *114* (3–4), 145–156. DOI: 10.1016/S0016-7061(03)00039-9.
 5. Andrews, S.S.; Karlen, D.L.; Cambardella, C.A. The soil management assessment framework: A quantitative soil quality evaluation method. *Soil Sci. Soc. Am. J.* **2004**, *68* (6), 1945–1962. DOI:10.2136/sssaj2004.1945.
 6. Armenise, E.; Redmile-Gordon, M.A.; Stellacci, A.M.; Ciccicarese, A.; Rubino, P. Developing a soil quality index to compare soil fitness for agricultural use under different managements in the Mediterranean environment. *Soil Till. Res.* **2013**, *130*, 91–98. DOI:10.1016/j.still.2013.02.013.
 7. Hussain, I.; Olson, K.R.; Wander, M.M.; Karlen, D.L. Adaptation of soil quality indices and application to three tillage systems in southern Illinois. *Soil Till. Res.* **1999**, *50* (3–4), 237–249. DOI:10.1016/S0167-1987(99)00012-4.
 8. Karlen, D.L.; Stott, D.E. A framework for evaluating physical and chemical indicators of soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., Eds.; Soil Science Society of America: Madison, 1994; 53–72.
 9. Lee, C.H.; Wu, M.Y.; Asio, V.B.; Chen, Z.S. Using a soil quality index to assess the effects of applying swine manure compost on soil quality under a crop rotation system in Taiwan. *Soil Sci.* **2006**, *171* (3), 210–222. DOI:10.1097/01.ss.0000199700.78956.8c.
 10. Shukla, M.K.; Lal, R.; Ebinger, M. Determining soil quality indicators by factor analysis. *Soil Till. Res.* **2006**, *87* (2), 194–204. DOI:10.1016/j.still.2005.03.011.
 11. Wienhold, B.J.; Andrews, S.S.; Karlen, D.L. Soil quality: A review of the science and experiences in the USA. *Environ. Geochem. Hlth.* **2004**, *26* (2), 89–95.
 12. Mendham, D.S.; Smethurst, P.J.; Holz, G.K.; Menary, R.C.; Grove, T.S.; Weston, C.; Baker, T. Soil analyses as indicators of phosphorus response in young eucalypt plantations. *Soil Sci. Soc. Am. J.* **2002**, *66* (3), 959–968. DOI: 10.2136/sssaj2002.9590.
 13. Salchow, E.; Lal, R.; Fausey, N.R.; Ward, A. Pedotransfer functions for variable alluvial soils in southern Ohio. *Geoderma* **1996**, *73* (3–4), 165–181. DOI:10.1016/0016-7061(96)00044-4.
 14. Karlen, D.L.; Tomer, M.D.; Neppel, J.; Cambardella, C.A. A preliminary watershed scale soil quality assessment in north central Iowa, USA. *Soil Till. Res.* **2008**, *99* (2), 291–299. DOI:10.1016/j.still.2008.03.002.
 15. Wymore, A.W. *Model-Based Systems Engineering*, Vol. 3; CRC Press: Boca Raton, 1993.
 16. Larson, W.E.; Pierce, F.J. The dynamics of soil quality as a measure of sustainable management. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., Eds.; Soil Science Society of America and American Society of Agronomy: Madison, 1994; 37–51.

Quality: Assessment by Interpolation Techniques

Vincent Obade

School of Environment and Natural Resources, Ohio State University,
Columbus, Ohio, U.S.A.

Abstract

Environmental hazards such as pollution, flooding, and other repercussions linked to abrupt climate change cause social and economic hardships and are exacerbated by soil degradation. Thus, detailed and accurate soil quality status maps in digital and analog format are required to identify soils susceptible to degradation. This entry explains the scientific basis of soil quality interpolation and mapping and techniques of assessment. Indeed, the accurate determination of soil quality is constrained with problems that range from data availability to the requirement of complex models to capture the spatial heterogeneity of soil properties. Thus, the integration of different approaches and tools to handle data scarcity and minimize errors to generate simplified soil quality maps is discussed.

INTRODUCTION

The 21st century is faced with challenges caused by the spiraling population increase and demand for food, energy, and water; indiscriminate land use conversions; and the catastrophic impacts of abrupt climate change partially attributed to soil degradation.^[1] Addressing these challenges necessitate concrete availability of scientifically credible information backed by proactive and strategically targeted management. The judicious management of soil resources holds the key to sustainability of earth's resources and thus socioeconomic viability.^[2] Thus, models that synthesize soil property information into simplified formats can play a critical role in the conceptual understanding of soil systems and the identification of key sustainable practices. Soil quality is defined as the capacity of the soil to support ecosystem functions such as nutrient cycling and water purification and encompasses the fitness of the soil for specific uses (e.g., road construction and agricultural production).^[3]

Soil quality can be determined qualitatively (e.g., visually where darker soils are considered to have a high organic matter, thus of high quality) or quantitatively by measuring the soil physical, chemical, and biological properties (Table 1). Soil biological properties are indirectly inferred from other soil properties [e.g., soil organic carbon (SOC)], because of 1) inaccuracies in earthworm counts (i.e., by hand); 2) difficulty in accounting microbial species diversity; and 3) difficulty in interpreting the soil respiration tests. The SOC, a proxy of soil quality, influences the soil ecosystem functions and productivity.^[4] High SOC reduces risks of eutrophication from agricultural runoff laden with nitrate (NO₃) or pesticide leaching. Stocks of SOC are computed by multiplying the SOC content (%) with the dry bulk density (ρ_b), for each soil layer, minus the fraction of fragments >2 mm or by the use of pedotransfer functions (PTFs).

Characterizing soil quality is challenging because of the landscape spatial heterogeneity and the fuzzy definition of soil quality. Despite numerous challenges, the laboratory determination of soil quality has its own problems. For example, the determination of SOC, by chromate oxidation or “wet combustion” method, produces toxic wastes that must be properly disposed and is inaccurate because of the incomplete oxidation of soil organic matter, whereas the dry combustion is expensive and slow. Another technique, loss on ignition, though relatively cost-effective, is inaccurate because certain mineral fractions are also decomposed when heating to the high temperatures required by this method.^[5] Laboratory methods can be prohibitively expensive, when the number of samples to be analyzed is large.

Advances in automation, digital computation, or machine learning have ignited new prospects for characterizing and mapping complex features.^[6] Maps describe earth surface features, either on hard copy or digitally. Digital soil mapping entails the “computer-assisted” production of digital representations of soil type or soil properties, over a spatial domain.^[6] Machine learning methods interpolate and make predictions based on spatially exhaustive environmental variables and are therefore useful for enumerating the inter-relationship between soil properties and other covariates; a practical example being the *scorpan* paradigm^[7] that comprises the following: 1) s: soil, other or previously measured attributes of the soil at a point; 2) c: climate, climatic properties of the environment at a point; 3) o: organisms, including land cover and natural vegetation; 4) r: topography, including terrain attributes and classes (e.g., slope, aspect, area, and direction); 5) p: parent material, including lithology; 6) a: age, the time factor; and 7) n: space, spatial or geographic position.

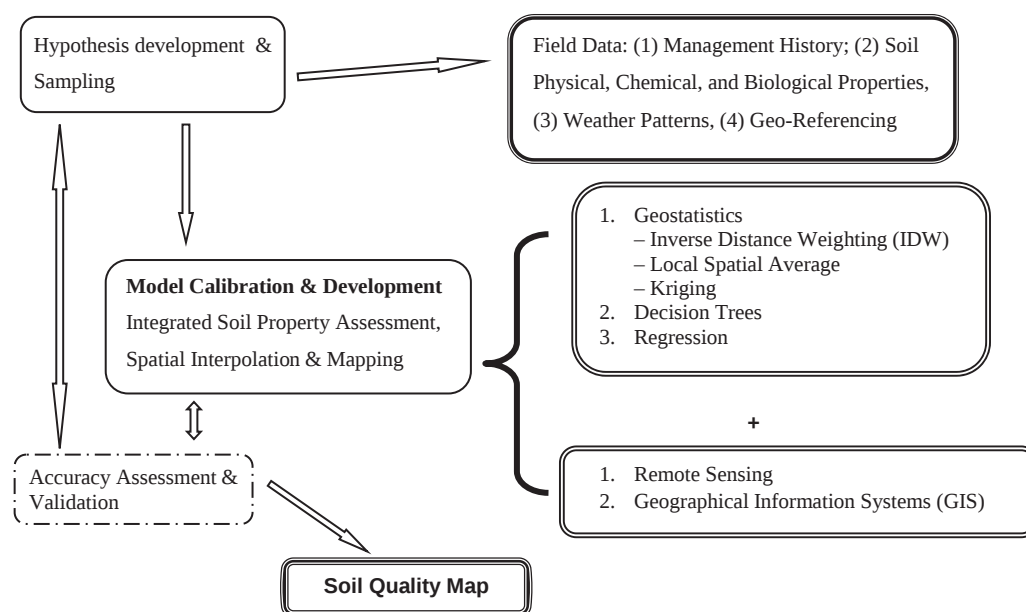
Generally, indicators need to be prioritized, ranked, or weighed based on importance or effectiveness in

Table 1 Soil quality assessment methods.

Qualitative (e.g., by visual observation)	Quantitative
Soil color: (i.e., darker soils perceived to be of relatively higher quality)	Soil physical, chemical, and biological properties (e.g., SOC, texture, ρ_b , water-holding capacity, pH, aggregate stability, electrical conductivity, earthworm count, and soil depth; http://soilquality.org/indicators.html)
Soil tilth: (good-quality soils have clods easily broken on tillage)	Soil erosion, pollution
Compaction (good-quality soils have no hard pan and have little resistance to plough on tillage; http://pubs.cas.psu.edu/FreePubs/pdfs/uc170.pdf)	Soil tensile strength/stability for construction or civil works
Water infiltration and drainage (e.g., good-quality soils drain well, compared with poor-quality soils that exhibit slaking and surface sealing, reduced water infiltration, and increased runoff and erosion)	Agricultural productivity (i.e., yields)
Hybrid methods: USDA Soil Quality Test Kit and Interpretive Guide; Cornell Soil Health Testing (http://soilhealth.cals.cornell.edu/).	Mathematical and statistical models (e.g., GIS, PTFs, and other regression models)

accomplishing or measuring selected tasks. The simpler or less detailed the indicator, the easier to identify, model, understand, and address the specific concerns that may include evaluating management efficacy or policy intervention. Geographical information system (GIS) generates maps from overlays of different data sets that can be used for prioritizing during decision-making and for evaluating alternative strategies. However, GIS is no panacea and has its own challenges. For example, the data used in overlays are derived from multiple sources, in different formats, time spans, and resolutions. Therefore, it is challenging to merge or standardize these data sets. Furthermore, mapping accuracy varies with the level of detail and generalization. Thus, it should not be construed that measurements made on the paper or digital map products reflect the exact location of all these features on the ground.

Thus, science-based techniques are required for establishing the least amount of data from an entire data set that mimic changes in soil quality vis-à-vis land management or geographic location. Due to the uncertainty with regard to soil quality benchmark selection, soil quality status may preferentially be determined on a relative basis. To date, there is no universal equation or soil quality prediction model that fits all ecoregions. Furthermore, different measurement and modeling techniques have different assumptions, which may create errors, exacerbating problems of interpretation and hampering efforts toward the judicious management of agroecosystems. However, it should be envisaged that measurements integrated with theories, hypotheses, or models can lead to discovery or insights on earth processes. Therefore, this entry expounds on the conceptual framework (Fig. 1) for interpolation and mapping of soil quality, limitations, and future research prospects.

**Fig. 1** Compartmentalized framework of soil quality spatial interpolation and mapping.

FRAMEWORK FOR SPATIAL INTERPOLATION

Sampling

Sampling entails the selection of a subset of data or observations to estimate the characteristics of the whole data set or to predict values at unsampled locations.^[8] Selecting a suitable sampling design can reduce analysis costs and time, enhance precision, and support repeatability in experiments. Examples of sampling designs include the simple random sampling (SRS), stratified random sampling, and systematic random sampling. The SRS considered as the reference method, selects calibration sites at random with all sites having the same probability of being sampled, and has no restrictions on their spatial locations. Although SRS is a simple method, some domains of the distribution and modalities of parameters may be missing or large gaps may occur in the sample. Alternately, stratified methods generate a set of samples that more precisely reflect the shape of a multidimensional distribution for a collection of chosen ancillary variables. To fulfill the scientific requirement of replicability in metrics, an unbiased estimate with the lowest errors is desirable.

Decision Trees

Decision trees are machine learning, data mining, and rule induction algorithms that categorize data by inferring the interconnectivity between a dependent variable and a set of predictors. Decision trees 1) handle non-parametric data where the predictors are not characterized as having a specific distribution; 2) are insensitive to missing data, to inclusion of irrelevant predictors or to the presence of outliers; 3) effectively operate using numerical, ordinal, binary, and categorical classes; and 4) identify complex hierarchical relationships between predictors and response variables.^[9] Decision trees consist of nodes and leaves with each node representing an *if-then* statement.^[10] The classification tree provides a categorical outcome, whereas the regression tree provides a continuous numerical outcome.

Random Forest (RF) is an ensemble of the Classification and Regression Tree algorithm and operates by aggregating multiple trees to enhance the model accuracy.^[11] RF has the advantage of incorporating “randomness” into its predictions through iterative bootstrap sampling and is less susceptible to overfitting.^[9] Comparatively, “bagging” aggregates the results of many trees, and boosting considers errors from previous classifier steps when sampling data for the next iteration.

Geostatistical Analyses

Geostatistical methods are based on the Tobler’s law and predict unsampled values at point locations from observations at neighboring positions, under the assumption that the values at different locations are spatially

autocorrelated.^[8] Tobler’s law states that observations or measurements close to each other are more likely to be similar than those farther apart.^[12] Examples of geostatistical methods include local spatial averaging, inverse distance weighing, and kriging. The local spatial average computes the value of unsampled locations from the mean of neighboring values; the problem being to define this local neighborhood. Comparatively, the inverse distance weighting computes the values for unsampled locations as the weighted mean of neighboring values, with the weights decreasing linearly from the prediction location, with the problem here being how to deal with the distances close to zero? In kriging, the linear model is fitted by ordinary least squares, and then, a variogram is estimated for the residuals. Cokriging is the multivariate modification of kriging that combines a sparsely measured primary variable (or target variable) with a denser set of ancillary data considered as secondary variable (e.g., remote sensing data) to enhance accuracy.^[13] The limitation of geostatistics is the requirement of dense point data sets.

Remote Sensing

Remote sensing is the science of gathering information by recording sensed or emitted energy from the earth’s surface without physical contact. Remote sensing provides spatially continuous spectral digital data useful for modeling earth system processes and downstream science applications. However, remotely sensed data can be expensive to collect, analyze, archive, and maintain. Furthermore, remote sensors measure only surrogate variables such as reflectance, brightness temperature, and backscatter, from which earth information can be gleaned.

The diffuse reflectance spectroscopy (DRS) visible (Vis) (400–780 nm) and near-infrared reflectance (NIR) (780–2500 nm) are non-destructive, fast, precise, inexpensive, proximal remote sensing tools useful for mapping soil properties.^[14] Practically, models are generated that relate soil spectra, from the Vis and NIR wavelengths with laboratory-based measurements of soil properties. The DRS has been utilized to estimate cation exchange capacity, base saturation, pH, exchangeable bases, and extractable phosphorus, clay content, extractable iron (Fe), total elements such as calcium, magnesium, Fe, manganese, potassium, and copper.^[15] The challenge remains to spatially interpolate spectra of these soil properties across a spatial domain, a feat that may be attainable by classifying soil spectra.

Soil classification for mapping can be done digitally by 1) grouping data/observations that present homogeneous attributes without prior knowledge (unsupervised) or 2) training a model to provide maps based on known soil type observations (i.e., supervised). In the same vein, feature selection and separability algorithms can

be used to separate relevant features from those that are irrelevant. Spectral mixture analyses decompose spectra within pixels based on proportional cover of each pure class, or end-member, enhancing clarity of map products. Unfortunately, the full range of spectral, spatial, and temporal properties for detailed soil classification is difficult to ascertain.

Time series analyses or temporal mapping with satellite imagery can be used for monitoring. Inherent monitoring challenges, such as data gaps from atmospheric scattering, cloud-covered, or shadowed pixels, can be eliminated by normalization. Normalization algorithms identify and merge the pseudo-invariant (i.e., temporally unchanged) features on both the ground and imagery. Mapping soil characteristics remotely may require sensors that generate signals that penetrate obstacles (e.g., vegetative cover, roads, and soil depths) or algorithms that consider these variables.

ASSESSING INTERPOLATION EFFICACY

A key challenge when interpreting maps is quantifying the accuracy and explanatory power of the information. Soil mapping errors can be attributed to the spatial heterogeneity in soil properties and the equipment errors. Accuracy can be assessed by comparing predicted with observed values. For example, a sample data set can be randomly split into 70% for model calibration and development and 30% for validation. Metrics such as the coefficient of determination (R^2), mean error (ME), and root mean square error (RMSE) can be used, with a high R^2 or smallest RMSE or ME indicating higher accuracy. For classified remotely sensed products, the error matrix or contingency table is computed, with the overall accuracy being the ratio of the correctly classified pixels (the sum of diagonal number of pixels in the *matrix*) to the total number of classified pixels, whereas the kappa index being the probability of a pixel classified by chance.

CONCLUSION

This entry provides a synopsis of issues to consider during soil quality interpolation especially under diverse land management scenarios. For comparability and monitoring purposes, fusing data from diverse sources to produce a synthesized indicator of soil quality would suffice. However, because one field can have several soil types, a limitation is the inherent scalar uncertainty when continuously mapping or predicting soil quality. Because interpolation approaches are data driven, interpretations should be done cautiously with local expert knowledge to avoid making

false assumptions, but the major challenge remains how to define the spatial dimensions (e.g., depth) for detailed analyses and interpretation of soil quality.

REFERENCES

1. Lal, R. Soil carbon management and climate change. In *Soil Carbon. Progress in Soil Science*; Hartemink, A.E., McSweeney, K., Eds.; Springer International Publishing: Switzerland, 2014; 339–361.
2. McBratney, A.; Field, D.J.; Koch, A. The dimensions of soil security. *Geoderma* **2014**, *213*, 203–213.
3. Doran, J.W.; Parkin, T.B. Defining and assessing soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicsek, D.F., Stewart, B.A., Eds.; Soil Science Society of America and American Society of Agronomy: Madison, 1994; 1–21.
4. Batjes, N.H. Soil organic carbon stocks under native vegetation – Revised estimates for use with the simple assessment option of the carbon benefits project system. *Agr. Ecosyst. Environ.* **2011**, *142* (3), 365–373.
5. Nelson, D.W.; Sommers, L.E. Total carbon, organic carbon, and organic matter. In *Methods of Soil Analysis Part 3—Chemical Methods*; Sparks, D.L., Page, A.L., Helmke, P.A., Loeppert, R.H., Eds.; SSSA book series 5.3; Soil Science Society of America and American Society of Agronomy: Madison, 1996; 961–1010.
6. Grunwald, S. Multi-criteria characterization of recent digital soil mapping and modeling approaches. *Geoderma* **2009**, *152* (3–4), 195–207.
7. McBratney, A.B.; Mendonca Santon, M.L.; Minasny, B. On digital soil mapping. *Geoderma* **2003**, *117* (1–2), 3–52.
8. Goovaerts, P. Geostatistics in soil science: State-of-the-art and perspectives. *Geoderma* **1999**, *89* (1–2), 1–45.
9. Heung, B.; Bulmer, C.E.; Schmidt, M.G. Predictive soil parent material mapping at a regional-scale: A random forest approach. *Geoderma* **2014**, *214–215*, 141–154.
10. Breiman, L.; Friedman, J.H.; Olshen, R.A.; Stone, C.J. *Classification and Regression Trees*; Chapman and Hall: Boca Raton, 1984.
11. Breiman, L. Random forests. *Mach. Learn.* **2001**, *45* (1), 5–32.
12. Tobler, W.R. A computer movie simulating urban growth in the Detroit region. *Econ. Geogr.* **1970**, *46* (2), 234–240.
13. Odeh, I.O.A.; McBratney, A.B.; Chittleborough, D.J. Further results on prediction of soil properties from terrain attributes: Heterotopic cokriging and regression-kriging. *Geoderma* **1995**, *67* (3–4), 215–226.
14. Gogé, F.; Gomez, C.; Jolivet, C.; Joffe, R. Which strategy is best to predict soil properties of a local site from a national Vis-NIR database? *Geoderma* **2014**, *213*, 1–9.
15. Sarkhot, D.V.; Grunwald, S.; Ge, Y.; Morgan, C.L.S. Comparison and detection of total and available soil carbon fractions using visible/near infrared diffuse reflectance spectroscopy. *Geoderma* **2011**, *164* (1–2), 22–32.

Quality: Carbon and Nitrogen Gases

Philippe Rochette

Soils and Crops Research Centre, Agriculture and Agri-Food Canada, Saint-Foy, Quebec, Canada

Reynald Lemke

Agriculture and Agri-Food Canada, Swift Current, Saskatchewan, Canada

Sean McGinn

Agriculture and Agri-Food Canada, Lethbridge, Alberta, Canada

Abstract

Several gases are either produced or consumed by biological and chemical transformations of carbon (C) and nitrogen (N) in soils. Some of these gases, including carbon dioxide, methane, and nitrous oxide, play a role in the earth's radiation balance (greenhouse effect) while others, such as carbon monoxide, ammonia, and nitrogen oxides (NO_x, i.e., NO and NO₂), have negative impacts on atmospheric chemistry. In this entry, we briefly describe how C and N gases are produced or consumed in soils and how soil management and soil quality can affect these processes.

CARBON (C) GASES

C is the major constituent of biomass and soil organic matter. However, more than 99% of global C is locked into sediments and fossil forms and is not available for biological processes. The small remaining active fraction of global C transits between atmospheric carbon dioxide (CO₂), biomass and soil organic matter, and detritus in the so-called C cycle (Fig. 1). The C cycle is driven by photosynthetic fixation of atmospheric CO₂ by plants. In global terrestrial ecosystems, it is estimated that plant photosynthesis fixes more than 200 Gt of CO₂ every year.^[1] Eventually, similar amounts are returned to the atmosphere by the respiration of animals and by the aerobic heterotrophic decomposition of soil organic matter and plant litter. In the absence of oxygen and other electron acceptors, methane (CH₄) is the final product of soil organic matter decomposition. Human activities, including changes in land use and soil management, are contributing to unprecedented rapid increases in atmospheric CO₂ and CH₄ concentrations, which may result in important modifications of the earth's climate. In dry soils, auto-oxidation of organic compounds can produce carbon monoxide (CO). CO can also be biologically oxidized to CO₂ in moist but well-aerated soils.

Carbon Dioxide

Soil-surface-emitted CO₂, or soil respiration, is the sum of the CO₂ produced by root respiration and heterotrophic

decomposition of root exudates, soil organic matter, and plant litter. Decomposition processes, while dominated by soil microbes, are the result of complex interactions between soil fauna, fungi, actinomycetes, and bacteria. During decomposition, complex molecules such as cellulose, hemicellulose, proteins, and lignin are broken down into low molecular weight substances and oxidized to CO₂ to produce energy and C for the growth of organisms.^[2,3] The rate of decomposition is regulated by the quality and quantity of organic substrates and by physical/environmental properties of soil, such as temperature, moisture, and aeration.^[4]

When ecosystems are in equilibrium, soil CO₂ emissions are the result of the natural recycling of nutrients and equal the amount of atmospheric CO₂ fixed by plant photosynthesis. However, in the past years, human activity has broken this equilibrium by burning fossil C reserves and by decreasing soil organic matter through land use changes. During this period, the additional CO₂ that entered the C cycle could not be completely absorbed by increases in autotrophic activity and atmospheric CO₂ concentrations increased from 280 to 365 ppm.

Several measures have been identified to reduce net CO₂ emissions from soils. Among them, it is believed that modifications in agricultural and forest management practices could result in increased C storage in soils as organic matter. Converting cultivated land into natural ecosystems usually increases soil C content because of increased return of plant litter and decreased

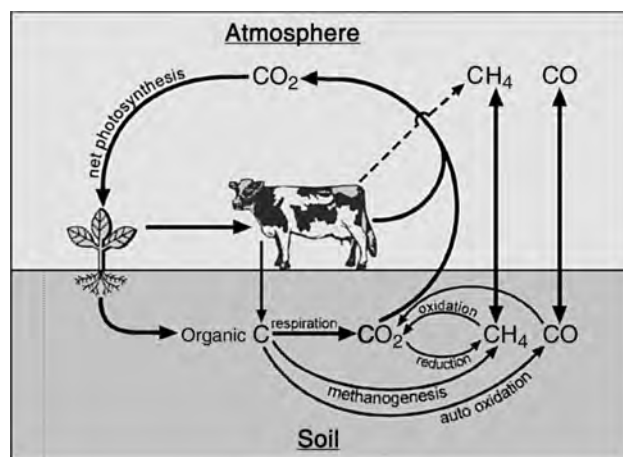


Fig. 1 The soil C cycle. In well-aerated soils, decomposition of soil organic matter produces CO_2 . Significant amounts of CH_4 are produced only under highly anaerobic (waterlogged) soils.

decomposition. Also, tillage breaks down aggregates and increases soil C losses by exposing organic matter to microbes that decompose it. Therefore, reducing tillage intensity, fallow frequency, and the amount of cultivated land (by increasing productivity) and increasing perennial crops in rotation and permanent grasslands are seen as potential management practices for mitigating global warming.^[5]

Methane

CH_4 is produced in soils by the decomposition of organic matter and by the reduction of CO_2 under highly anaerobic environments. Such conditions are found in wetlands and in rice paddies that, together with landfills, contribute about half of the total emissions of anthropogenic CH_4 . CH_4 is stable in waterlogged soils and can be emitted to the atmosphere via diffusion, ebullition, and transport through plants. In the presence of oxygen, certain bacteria can oxidize CH_4 to CO_2 (Fig. 1).^[6]

Human intervention can greatly influence CH_4 production or consumption in soils. Flooding of soils in natural or agricultural ecosystems usually results in increased emissions,^[7] while drainage of wetlands can turn a source of CH_4 into a sink.^[8] Flooded rice fields are a major source of anthropogenic CH_4 . Management practices have been proposed to reduce CH_4 emissions from rice paddies, including draining the fields during the growing season, replacing urea by other types of N fertilizers, and reducing the input of crop residues by using new cultivars and alternative cultural practices.^[5] Reduced rates of CH_4 oxidation in well-aerated soils have been observed following the cultivation and addition of N fertilizers.^[9]

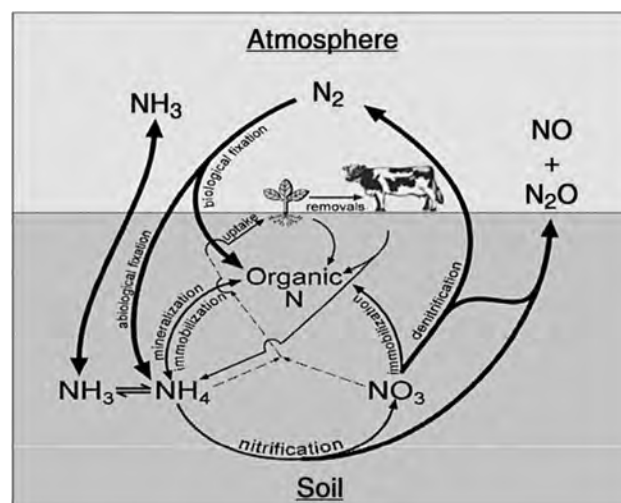


Fig. 2 Schematic representation of the N cycle depicted as a “loop-within-a-loop.” The outer loop traces the passage of N from the atmosphere through ecosystems and back to the atmosphere, while the inner loop traces the interconversion of N between organic and inorganic forms within the soil–plant system.

N GASES

The passage of N through ecosystems can be represented as a “loop-within-a-loop” (Fig. 2). N enters ecosystems primarily through biotic or abiotic processes that “fix” [convert molecular nitrogen (N_2) to biologically available forms] N_2 from the atmosphere. Within the soil–plant system, N undergoes a complex series of transformations, resulting in a continual transfer of N between inorganic and organic forms—the inner loop (Fig. 2). Nitrous oxide (N_2O) and nitrogen oxides (NO_x) are both produced and consumed during these transformations. With time, most of the N entering ecosystems returns to the atmosphere as N_2 , but an important fraction is emitted as gaseous ammonia (NH_3), NO_x , and N_2O . More than 70% of the estimated 18 Tg N_2O –N entering the atmosphere each year is emitted by soils.^[1,10] Above plant canopy emissions of NO_x from soils are probably in the range of 3.3–21 Tg N yr^{−1}.^[11,12] This exchange of N between the atmosphere and the soil–plant system—the outer loop—strongly mediates atmospheric concentrations of NH_3 , NO_x , and N_2O .

Nitric Oxide (NO) and NO_x

The majority of soil-emitted NO_x is NO ,^[13] therefore, the rest of this discussion will focus on N gases other than nitrogen dioxide (NO_2). Nitric oxide is derived primarily via autotrophic nitrification,^[14,15] whereas N_2O emissions can be a product of autotrophic nitrification,^[15] denitrification,^[16] or a combination of both.^[14] N_2O , and particularly NO, may also be emitted from various chemical reactions collectively known as chemodenitrification.^[17]

The magnitude of N_2O and NO emissions is strongly influenced by site-specific variables—most particularly soil attributes—and human intervention. Given that both gases are produced primarily during microbial transformations of inorganic N, the potential of a soil to produce and emit N_2O or NO increases with increasing N availability.^[18] Anthropogenic activities that increase the flow of N into the system also increase the emissions of N_2O and NO . Such activities include, e.g., the cultivation of legume grain or forage crops^[19] and intensive use of N fertilizers.^[20]

Total N_2O emissions tend to increase as soil organic carbon (SOC) content increases. This is understandable, since N turnover is closely tied to SOC turnover, and the amount of C available to drive microbial processes is directly related to the quantity and quality of SOC. The relative availability of C and N is of particular importance when livestock manure or sewage sludge is added to soils. The addition of both C and N tends to increase N_2O and NO emissions more than N alone, particularly in C-limited systems.^[21,22]

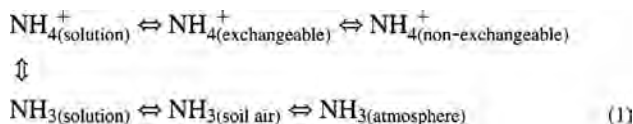
Agricultural management practices that minimize soil disturbance tend to increase soil water status and soil bulk density, favoring the development of anaerobic conditions and higher N_2O ^[4] but possibly lower NO emissions.^[23] Conversely, equal or lower emissions of N_2O have also been reported in no-tillage systems compared to conventional tillage systems.^[24] In this instance, reduced soil disturbance appeared to alter N cycling, resulting in lower nitrate (NO_3^-) availability—hence lower N_2O emissions—during the spring thaw period.

In general, soil conditions or management that leads to accumulations of ammonium (NH_4^+), NO_3^- , and particularly NO_2^- favor gaseous N losses. This may happen if N is released during soil C mineralization at a time when plant uptake is absent or minimal, an example being bare soil fallow periods commonly employed in the semiarid regions of North America, or if fertilizer or organic N is added at a time when crop growth is not vigorous. Low soil pH levels (< 5) inhibit NO_2^- oxidizers more than NH_4^+ oxidizers, resulting in accumulations of NO_2^- and abiotic production of NO .^[25] The latter may occur even in near-neutral soils as a result of very high concentrations of NH_4^+ localized in urine patches, or concentrated in bands or nests of NH_4^+ -based fertilizers.^[26]

Ammonia

The production of NH_3 in soils is closely linked to the accumulation of NH_4^+ resulting from the mineralization of organic matter (ammonification) by soil microbes. In agricultural soils, the NH_4^+ content is enhanced where inorganic N and livestock manure fertilizers are used. In soil solution, NH_4^+ dissociates and forms an equilibrium with NH_3 , which can volatilize to the soil air and

eventually reach the atmosphere (Eq. 1). NH_4^+ is also bound to negative exchange sites on soil organic matter and clay minerals (exchangeable NH_4^+) and can be “fixed” within clays. The latter is released slower than exchangeable NH_4^+ .^[27] NH_4^+ is converted through biological oxidation (nitrification) by soil bacteria (nitrifiers) to NO_3^- , which is taken up by plants. Atmospheric NH_3 and NH_4^+ can be recycled back to the soil through dry and wet deposition (Fig. 2).



The volatilized NH_3 from soil is not only a loss of valuable N for crop growth^[28] but also a significant source of atmospheric NH_3 ,^[29,30] which can damage plants^[31,32] and contribute to odor nuisances that are associated with the application of manure to soils. In the atmosphere, NH_3 neutralizes atmospheric acids, and the resulting aerosols contribute to atmospheric haze. Atmospheric NH_3 can also cause acidification of ecosystems by augmenting the capture of sulfur dioxide in clouds^[33] and through nitrification of deposited NH_4^+ and NH_3 .^[34]

Most management techniques that relate to minimizing NH_3 emissions pertain to managing inorganic and livestock manure fertilizers, which are the most significant sources of this gas.^[30,35] The key to these techniques is to reduce exposure of the amendment to the atmosphere either by limiting the amount applied or by the injection/incorporation of the amendment. Manipulating the chemistry of the soil can also be used to control NH_3 emissions especially in amended soils.^[36,37] For example, the addition of acids to fertilized soils can potentially reduce NH_3 emissions. Adding heavy metals or organic compounds with urea fertilizer can inhibit the conversion of urea to NH_4^+ by retarding urease enzyme activity. There is also the potential of adding Ca or Mg salts with urea fertilizers to reduce NH_3 emissions.

Meteorological and soil conditions also modify the emission of NH_3 . Application of manure when soil moisture is initially high but subject to rapid drying can result in higher emission than the normal emissions of NH_3 .^[36,38] High soil moisture content would put more NH_4^+ into solution near the soil surface and increase volatilization. However, irrigation or rainfall following application can leach near-surface NH_4^+ and decrease volatilization.

CONCLUSION

Soil quality and C and N gas dynamics are closely linked. In general, high-quality soils have high nutrient availability and aeration, good infiltration and retention of water,

and a stable structure, all of which are factors that control the production and consumption of gases in soils. Improving soil quality through higher SOC content or better soil structure often results in a reduction of net emissions of greenhouse gases into the atmosphere. Also, soil degradation increases the emissions of greenhouse gases as a result of greater use of N fertilizers, erosion, denitrification, deforestation, and grassland cultivation.^[39] Therefore, adoption of most management practices aimed at improving soil quality also help to improve atmospheric quality.

REFERENCES

- Duxbury, J.M.; Harper, L.A.; Mosier, A.R. Contributions of agroecosystems to global climate change. In *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*; Harper, L.A., Mosier, A.R., Duxbury, J.M., Rolston, D.E., Eds.; ASA Special Publication No. 55; American Society of Agronomy: Madison, 1993; 1–18.
- Kilham, K. The ecology of soil nutrient cycling. In *Soil Ecology*; Cambridge University Press: Cambridge, 1994; 89–108.
- Coleman, D.C.; Crossley, D.A. Decomposition and nutrient cycling. In *Fundamentals of Soil Ecology*; Associated Press: New York, 1996; 109–140.
- Linn, D.M.; Doran, J.W. Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. *Soil Sci. Soc. Am. J.* **1984**, *48*, 1267–1272.
- Intergovernmental Panel on Climate Change. Climate Change. In *Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses*; Watson, R.T., Zinyowera, M.C., Moss, R.H., Eds.; Cambridge University Press: New York, 1995.
- Topp, E.; Pattey, E. Soils as sources and sinks for atmospheric methane. *Can. J. Soil Sci.* **1997**, *77*, 167–178.
- Duchemin, E.; Lucotte, M.; Canuel, R.; Chamberland, A. Production of greenhouse gases CH₄ and CO₂ by hydroelectric reservoirs of the boreal region. *Global Biogeochem. Cy.* **1995**, *9*, 529–540.
- Roulet, N.T.; Ash, R.; Quinton, W.; Moore, T.R. Methane flux from drained northern peatlands: Effect of a persistent water table lowering on flux. *Global Biogeochem. Cy.* **1993**, *7*, 749–769.
- Mosier, A.R.; Scimel, D.; Valentine, D.; Bronson, K.; Parton, W. Methane and nitrous oxide fluxes in native fertilized and cultivated grasslands. *Nature* **1991**, *350*, 330–332.
- Kroeze, C.; Mosier, A.; Bouwman, L. Closing the global N₂O budget: A retrospective analysis 1500–1994. *Global Biogeochem. Cy.* **1999**, *13* (1), 1–8.
- Delmas, R.; Serca, D.; Jambert, C. Global inventory of NO_x sources. *Nutr. Cycl. Agroecosys.* **1997**, *48*, 51–60.
- Davidson, E.A.; Kinglerlee, W. A global inventory of nitric oxide emissions from soils. *Nutr. Cycl. Agroecosys.* **1997**, *48*, 37–50.
- Hutchinson, G.L.; Davidson, E.A. Processes for production and consumption of gaseous nitrogen oxides in soil. In *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*; American Society of Agronomy: Madison, 1993; 79–93.
- Skiba, U.; Smith, K.A.; Fowler, D. Nitrification and denitrification as sources of nitric oxide and nitrous oxide in a sandy loam soil. *Soil Biol. Biochem.* **1993**, *25* (11), 1527–1536.
- Hutchinson, G.L.; Guenzi, W.D.; Livingston, G.P. Soil water controls on aerobic soil emissions of gaseous nitrogen oxides. *Soil Biol. Biochem.* **1993**, *25* (1), 1–9.
- Williams, P.H.; Jarvis, S.C.; Dixon, E. Emission of nitric oxide and nitrous oxide from soil under field and laboratory conditions. *Soil Biol. Biochem.* **1998**, *30* (14), 1885–1893.
- Haynes, R.J.; Sherlock, R.R. Gaseous losses of nitrogen. In *Mineral Nitrogen in the Plant-Soil System*; Haynes, R.J., Ed.; Academic Press: New York, 1986; 242–302.
- Matson, P.A.; Vitousek, P.M. Cross-system comparisons of soil nitrogen transformations and nitrous oxide flux in tropical forest ecosystems. *Global Biogeochem. Cy.* **1987**, *1*, 163–170.
- Galbally, I.E. Biosphere-atmosphere exchange of trace gases over Australia. In *Australia's Renewable Resources: Sustainability and Global Change*; Gifford, R.M., Barson, M.M., Eds.; Bureau of Rural Resources: Canberra, 1992; 117–149.
- Smith, K.A.; McTaggart, I.P.; Tsuruta, H. Emissions of N₂O and NO associated with nitrogen fertilization in intensive agriculture, and the potential for mitigation. *Soil Use Manag.* **1997**, *13*, 296–304.
- Rochette, P.; van Bochove, E.; Prevost, D.; Angers, D.A.; Cote, D.; Bertrand, N. Soil C and N dynamics following application of pig slurry for the 19th consecutive year: II—mineral N and N₂O fluxes. *Soil Sci. Soc. Am. J.* **2000**, *64*, 1396–1403.
- Thornton, F.C.; Shurpali, H.J.; Bock, B.R.; Reddy, K.C. N₂O and NO emissions from poultry litter and urea applications to Bermuda grass. *Atmos. Environ.* **1998**, *32* (9), 1623–1630.
- Civerolo, K.L.; Dickerson, R.R. Nitric oxide soil emissions from tilled and untilled cornfields. *Agr. Forest Meteorol.* **1998**, *90* (4), 307–311.
- Lemke, R.L.; Izaurralde, R.C.; Nyborg, M.; Solberg, E.D. Tillage and N-source influence soil-emitted nitrous oxide in the Alberta parkland region. *Can. J. Soil Sci.* **1999**, *79*, 15–24.
- Yamulki, S.; Harrison, R.M.; Goulding, K.W.T.; Webster, C.P. N₂O, NO and NO₂ fluxes from a grassland: Effect of soil pH. *Soil Biol. Biochem.* **1997**, *29* (8), 1199–1208.
- Burns, L.C.; Stevens, R.J.; Smith, R.V.; Cooper, J.E. The occurrence and possible sources of nitrite in a grazed, fertilized, grassland soil. *Soil Biol. Biochem.* **1995**, *27* (1), 47–59.
- Mengel, K. Dynamics and availability of major nutrients in soils. *Adv. Soil Sci.* **1985**, *2*, 65–131.
- McGinn, S.M.; Janzen, H.H. Ammonia sources in agriculture and their measurement. *Can. J. Soil Sci.* **1998**, *78*, 139–148.

29. Bouwman, A.F.; Lee, D.S.; Asman, W.A.H.; Dentener, F.J.; van Der Hoek, K.W.; Olivier, J.G.J. A global high-resolution emission inventory for ammonia. *Global Biogeochem. Cy.* **1997**, *11* (4), 561–587.
30. Duxbury, J. The significance of agricultural sources of greenhouse gases. *Fert. Res.* **1994**, *38*, 151–163.
31. Van der Eerden, L.J.M. Toxicity of ammonia to plants. *Agr. Environ.* **1982**, *7*, 223–235.
32. Van der Eerden, L.J.M.; de Visser, P.H.B.; van Dijk, C.J. Risk of damage to crops in the direct neighbourhood of ammonia sources. *Environ. Pollut.* **1998**, *102*, 49–53.
33. ApSimon, H.M.; Kruse, M.; Bell, J.N.B. Ammonia emission and their role in acid deposition. *Atmos. Environ.* **1987**, *21* (9), 1939–1946.
34. Fahey, T.J.; Williams, C.J.; Rooney-Varga, J.N.; Cleveland, C.C.; Postek, K.M.; Smith, S.D.; Bouldin, D.R. Nitrogen deposition in and around an intensive agricultural district in central New York. *J. Environ. Qual.* **1999**, *28*, 1585–1600.
35. Buijsman, E.; Maas, H.F.M.; Asman, W.A.H. Anthropogenic NH₃ emissions in Europe. *Atmos. Environ.* **1987**, *21* (5), 1009–1022.
36. Hargrove, W.L. Soil, environmental, and management factors influencing ammonia volatilization under field conditions. In *Ammonia Volatilization from Urea Fertilizers*; Brock, B.R., Kissel, D.E., Eds.; National Fertilizer Development Center, Tennessee Valley Authority: Muscle Shoals, 1988; 17–36.
37. Fenn, L.B.; Hossner, L.R. Ammonia volatilization from ammonium or ammonium-forming nitrogen fertilizers. *Adv. Soil Sci.* **1985**, *1*, 124–169.
38. Brunke, R.; Alvo, P.; Schuepp, P.; Gordon, R. Effect of meteorological parameters on ammonia loss from manure in the field. *J. Environ. Qual.* **1988**, *17* (3), 431–436.
39. Lal, R.; Kimble, J.; Stewart, B.A. World soils as a source or sink of radiatively-active gases. In *Soil Management and Greenhouse Effect*; Lal, R., Kimble, J., Stewart, B.A., Eds.; Advances in Soil Science; CRC Press: Boca Raton, 1995; 1–7.

Quality: Critical Limits and Standardization

Martin R. Carter

Agriculture and Agri-Food Canada, Charlottetown, Prince Edward Island, Canada

Abstract

Critical (“threshold,” “trigger,” “baseline,” “reference”) limits in soil quality assessment refer to the specific value or range of a soil property or indicator that is required to ensure that a soil process or function is not restricted or adversely influenced. Based on the primary concept of “fitness for use” for soil quality, critical limits denote the boundary values or margins of tolerance required for soil indicators that are associated with optimum soil functioning and indicate if a soil is “fit” for a specific use. Standardization refers to the technical protocol for the sampling, storage, analysis, and interpretation of soil properties, attributes, and indicators as a means to provide reliable and comparable methodology for soil quality assessment.

INTRODUCTION

Concerns with soil degradation and sustainable soil management in agroecosystems have emphasized the need to define and evaluate soil and land quality.^[1–3] The basic idea of “fitness for use” in regard to agricultural and/or industrial use of soil, reflected in early attempts to classify “soil suitability” or “land capability,”^[4,5] is the basic premise of soil quality. Ecological concepts, such as function, processes, attributes, and indicators, provide a useful and logical framework to describe and evaluate soil quality.^[4] The framework is based on the following sequence: purpose, function, processes, properties/attributes (including critical values), indicators, and methods (including standardization; [Table 1](#)). The selection of suitable attributes and indicators, the establishment of critical limits for these attributes and standardization related to the protocol and methodology for attribute characterization are key aspects of soil quality evaluation.

CRITICAL LIMITS FOR SOIL QUALITY

A large range of attributes, such as chemical, physical, and biological properties, can be used to describe soil quality. A group of key soil properties/attributes or indicators is called a minimum data set (MDS).^[1,4,5]

Concept of Critical Limits

The concept of “critical values” (also called “threshold,” “trigger,” “baseline,” or “reference” values) is commonly used in soil pollution studies^[6] and is analogous to the “degrees of limitation” for rating land qualities in land use requirements.^[3] Critical limits are needed for each soil quality property/attribute or indicator, within a MDS, so that the measured value of the property can be classed as “optimum”

or “limited” for the specific land use and to ensure that dependent soil processes and function are not restricted or adversely affected.^[1] If the value obtained for the selected indicator falls outside the desired range this would raise questions about the fitness of the soil for the specific use and/or indicate the need for improved management.^[1]

Selecting Critical Soil Properties and Parameters

Past studies have noted the general relation between soil–land use and soil properties resulting in the development of land capability classes and soil productivity ratings.^[1,4,5] For example, the Canadian Land Suitability Rating System categorizes land into classes based on the limitations (e.g., climate, soil, and landscape restrictions) to productivity of a specific crop. The USDA Land Capability Classes based on the rating (eight classes) of six limitations/hazards (e.g., soil erosion, wetness, depth, salinity, sodicity, and climate) provide a meaningful assessment of soil productivity applicable to a wide range of soil types and agricultural crops.^[5] In other examples, a MDS of selected soil attributes (e.g., soil nutrient availability, organic matter, particle size, plant available water, structure, strength, rooting depth, pH, and salinity) is considered adequate to quantify and monitor soil quality.^[1]

Setting Critical Limits

The selection of critical soil properties is followed by the setting of critical limits, or ranges, that help characterize the value of the property as “optimum” or “limited.” If information on critical values is limited, qualitative reference values can be selected based on soil resource inventories or general consensus.^[1,5] An index can then be derived using scores (e.g., 0–1) for the value of each property, along with weighting values for the considered importance or

Table 1 Sequential framework to evaluate soil quality for specific purpose or fitness of use.

Sequence steps	Sequential framework	Questions implied by the framework
1	Purpose	What will the soil be used for?
2	Functions	What specific role is being asked of the soil?
3	Processes	What key soil processes support each function?
4	Properties/attributes	What are the critical soil properties for each process? What are their critical values or range?
5	Indicators/surrogates/pedotransfer functiona	When the soil property/attribute is difficult to measure or not available, what indirect or related property or properties can be used in its place?
6	Methodology/standardization	What methods are available to measure the attribute? Technical rules and protocols for soil sampling, handling, storage, analysis, and interpretation of data.

^aA pedotransfer function is a mathematical function that relates a soil property to another soil property.

Source: Adapted from Larson & Pierce^[1] and Carter & Gregorich.^[4]

priority of each property for soil quality.^[1] The combination of the scores for a group of selected properties provides a quantitative index as an overall measure of soil quality.^[5]

General Classes of Critical Limits

For many soil properties, a quantitative range of values (e.g., low, optimum, high) can be derived based on empirically derived relations between the property and soil processes or some aspect of soil function (e.g., plant growth). These values have general application to a wide range of situations. For example, soil hydraulic conductivity,^[7] macroporosity and aeration,^[8] bulk density,^[9] strength or penetration resistance,^[10] and the least limiting water range^[11] have been characterized on the basis of their

Table 3 Critical level or range of soil physical attributes or indicators for optimum root growth of irrigated fruit trees in duplex soils in southeastern Australia.

Soil quality attribute	Indicator	Critical level
Soil volume	Soil depth	500 mm
Soil strength	Penetrometer resistance	<0.5 MPa
Soil water parameters	Macropores (>30 μm diameters)	>15%
Storage pores (30–0.2 μm diameters)		>20%
Unsaturated hydraulic conductivity (–100 kPa)		>10 ^{–4} m day ^{–1}
Soil aeration	Air-filled porosity (after 24-hour drainage)	>15%
Soil temperature		18–25°C

Source: Adapted from Cockroft & Olsson.^[14]

critical range for plant root development and growth (Table 2). Critical levels and ranges are also available for soil oxygen content,^[10] salinity, and sodicity.^[7] Examples of desirable values are available for plant nutrients, soil pH, and soil rooting depth.^[1]

For soil properties that are multifunctional in nature (e.g., organic matter), the setting of critical limits can be problematic. Some studies have linked critical values of total soil organic matter for specific processes involved with soil fertility and erodibility,^[12] while others have attempted to link or elucidate cause–effect mechanisms between soil functions and soil organic matter and between soil microbiological indicators and macrofauna.^[13]

Site Specific Critical Limits

Although general classes or optimum ranges can be established for many soil attributes in soil quality assessment, it is recognized that the critical ranges must be determined and/or adapted and related to specific land use situations. Generally, critical values for soil attributes and indicators would be site or soil specific. For example, a high clay content may be favoured in a semiarid region, where soil

Table 2 Critical ranges for specific soil properties in relation to root and plant growth applicable over a range of soil types and conditions.

Range	Hydraulic conductivity (m s^{-1})	Macropores (>60 μm diameter; % soil volume)	Relative bulk density (%)	Penetration resistance (MPa)	Least limiting water range ($\text{m}^3 \text{m}^{-3}$)
Limiting (low)	<10–7	<5	<80	<0.5	<0.10
Optimum	10–7–10–4	10–20	82–87	0.5–2.0	0.10–0.20
Limiting (high)	>10–4	>25	>90	>2.5	—

Notes: The low range for hydraulic conductivity and macropore volume and high range for relative bulk density and penetration resistance are generally associated with compact soil conditions that inhibit root growth, while the contrasting limiting range is related to poor root-to-soil contact and poor retention of water. A limiting high range does not apply for least limiting water range.

Source: Adapted from Marshall & Holmes,^[7] Thomasson,^[8] Håkansson & Lipiec,^[9] Glinski & Lipiec,^[10] and Kay & Angers.^[11]

Table 4 Critical level for soil and associated non-soil indicators, based on improvements relative to community average, used to evaluate resource conservation and sustainability for rice culture on sloping land at Gubu, near Cebu City, Philippines.

Indicator	Critical level (relative to community average)
Crop yield	More than 20% of average yield
Profit	Better than 20% of community average
Frequency of crop failure	Average frequency or 20%, whichever is lower
Soil depth	Average of similar soil types or 50 cm
Soil organic carbon	Average of community or 10 g kg ⁻¹ , whichever is higher
Permanent ground cover	Average of community or 15%, whichever is higher

Source: Adapted from Gomez, Swete Kelly, et al.^[15]

moisture retention is an advantage but may be undesirable in humid conditions in which poor internal drainage may limit yields. In similar fashion, a certain soil bulk density can be optimal under a semiarid moisture regime but deleterious under a humid moisture regime due to changes in relative saturation and subsequent poor soil aeration.^[4] Table 3 provides an example of critical limits for some soil physical attributes selected for irrigated tree crops in Alfisols with a texture contrast in the A and B horizon in southeastern Australia.^[14] Other farm-based studies have developed critical limits for indicators to evaluate farmer satisfaction and resource conservation. Table 4 shows critical limits set for rice (*Oryza sativa* L.) culture in the Philippines based on consensus values at the farm level.^[15]

STANDARDIZATION

Once an attribute has been identified for a specific soil type or situation, information is needed in regard to soil quality standards for a given set of conditions. Soil quality standards are required to ensure that soil sampling, description, and analysis can set the limits for a quality soil and detect adverse changes in soil quality. Usually the critical soil attribute is related to a specific methodology.^[1,4] Standardization deals with the development and applications of technical rules, specification and protocols in regard to a measuring method.^[6] The International Organization for Standardization has developed various standards for soil quality measurements that address the different phases (e.g., soil sampling, handling, storage, analysis) involved in soil characterization.^[6,16]

REFERENCES

1. Larson, W.E.; Pierce, F.J. Conservation and enhancement of soil quality. In *Evaluation for Sustainable Land Management in the Developing World*; IBSRAM Proceeding 12(2); International Board Soil Research Management: Bangkok, 1991; Vol. 2, 175–203.
2. Doran, J.W.; Parkin, T.B. Defining and assessing soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezedick, D.F., Stewart, B.A., Eds.; Special Pub. No. 35; American Society of Agronomy: Madison, WI, 1994; 3–21.
3. FAO. *Guidelines: Land Evaluation for Rainfed Agriculture*; Soils Bull. 52; Food and Agriculture Organization, United Nations: Rome, 1983; 237 pp.
4. Carter, M.R.; Gregorich, E.G.; Anderson, D.W.; Doran, J.W.; Janzen, H.H.; Pierce, F.J. Concepts of soil quality and their significance. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 1–9.
5. Singer, M.J.; Ewing, S. *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, FL, 1999; G271–G298.
6. Nortcliff, S. Standardisation for soil quality attributes. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 187–201.
7. Marshall, T.J.; Holmes, J.W. *Soil Physics*; Cambridge University Press: Cambridge, 1978; 102, 249–250, 257–258.
8. Thomasson, A.J. Towards an objective classification of soil structure. *J. Soil Sci.* **1978**, *29*, 38–46.
9. Håkansson, I.; Lipiec, J. A review of the usefulness of relative bulk density values in studies of soil structure and compaction. *Soil Till. Res.* **2000**, *53*, 71–85.
10. Glinski, J.; Lipiec, J. *Soil Physical Conditions and Plant Roots*; CRC Press: Boca Raton, 1990; 250 pp.
11. Kay, B.D.; Angers, D.A. Soil structure. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 1999; A229–A276.
12. Feller, C.; Beare, M.H. Physical control of soil organic matter dynamics in the tropics. *Geoderma* **1997**, *79*, 69–116.
13. Carter, M.R. Organic matter and sustainability. In *Sustainable Management of Soil Organic Matter*; Rees, R.M., Ball, B.C., Campbell, C.D., Watson, C.A., Eds.; CABI: Oxford, 2001; 9–22.
14. Cockroft, B.; Olsson, K.A. Case study of soil quality in South-Eastern Australia: Management of structure for roots in duplex soils. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 339–350.
15. Gomez, A.A.; Swete Kelly, D.E.; Syers, J.K.; Coughlan, K.J. Measuring sustainability of agricultural systems at the farm level. In *Methods for Assessing Soil Quality*; Doran, J.W., Jones, A.J., Eds.; SSSA: Madison, 1996; 401–410.
16. Hortensius, D.; Welling, R. International standardization of soil quality measurements. *Commun. Soil Sci. Plant Anal.* **1996**, *27*, 387–402.

Quality: Erosion

Craig Ditzler

National Leader for Soil Classification and Standards, Lincoln, Nebraska, U.S.A.

Abstract

Soil erosion involves the detachment and removal of soil material from one site and its transport to another location. Soil erosion usually degrades soil quality and a soil of poorer quality is less able to withstand further erosion, thus creating a downward spiral of soil degradation. When the surface soil is removed through erosion, organic matter and clay particles may be lost, with consequent reductions in fertility, biological activity, aggregation, and rooting depth. Other potential effects of erosion on soil quality include reduced porosity and infiltration, formation of crusts on the soil surface, changes in soil texture, and compaction. These changes in turn reduce the capacity of the soil to supply and cycle nutrients, filter and degrade toxic materials, store and supply moisture, and sustain plant and biological productivity. They may also result in increased runoff, less biomass production and plant cover, and greater susceptibility to further erosion. Erosion increases the variability in soil quality across a field and, on a broad scale, is associated with widespread loss of agricultural productivity and declining water quality.

INTRODUCTION

Soil erosion is a natural process by which soil particles are detached and moved by water, wind, gravity, or ice. All soils have an inherent erodibility, or natural susceptibility to erosion, based on soil features, topography, and climate. However, human activities, such as logging, livestock grazing, tillage, removal of vegetation, and urban development, can greatly accelerate natural rates of erosion. Cleared and managed as they are for crop and livestock production, agricultural soils are particularly susceptible to wind and water erosion (Figs. 1 and 2), and also to a recognized process known as tillage erosion—the loosening of soil by tillage equipment and its downslope movement under gravity.^[1]

Eroded soil may move only a few meters in a field and come to rest in lower positions, resulting in increased variability of surface soil properties across the field as subsoil becomes exposed in some places and surface layers are buried and over-thickened in others. It may also move great distances, being deposited in neighboring fields, roadside ditches, and water bodies. In some cases of wind erosion, fine soil particles may travel many kilometers before being deposited.

EFFECTS OF SOIL EROSION ON SOIL QUALITY

The loss of surface soil in a landscape may impair soil function, and thus soil quality, through adverse effects on many physical, chemical, and biological properties of the

soil.^[2] [Table 1](#) summarizes some of the effects of erosion on the soil quality functions listed by Gregorich.

Physical Effects

Erosion selectively removes the finer, lighter particles from the soil surface, leaving coarser particles behind. Depending on the severity of erosion, an eroded soil may become very coarse in texture, sometimes with a gravelly surface. The deposition of the eroded material in lower topographic areas may result in a thickening of the topsoil and an increase in rooting volume.

Tillage of eroded soils may result in a mixing of subsoil with the surface soil, altering its composition. For example, cultivation of eroded soils having clay-enriched subsoils may increase the clay content of the surface soil. As erosion progresses, plant roots must enter progressively deeper into the subsoil layer to obtain nutrients and water. Where subsoil layers restrict root growth because of their physical and/or chemical properties, the depth of rooting is reduced, along with the capacity of the soil to supply water and nutrients to plants.

Individual soil particles are held together in aggregates, which form the structural fabric of the soil. Soils with good structural arrangement of aggregates and the pore spaces between them provide good aeration for soil roots and microbes; allow ready movement and storage of water and plant nutrients in the pore spaces; and retain their structure when exposed to stresses such as cultivation and the impact of raindrops.^[3,4] Abrasion by wind, rain, and tillage can disintegrate aggregates at the soil surface, and the resulting



Fig. 1 Deposition of wind-blown soil material.
Source: From USDA, Natural Resources Conservation Service—Soil Quality Institute.

fine particles can plug larger soil pores and form a hard, thin crust on the surface. This crust seals the surface and limits the infiltration of air and water, as well as impedes the emergence of seedlings. Finer soil particles are also more easily compacted as the pore spaces between them are reduced under the pressure of farm machinery. Deteriorating soil structure further increases the risk of soil erosion—fine soil particles created by the breakdown of aggregates at the soil surface are especially vulnerable to wind and water erosion. Furthermore, compacted, crusted soils resist the infiltration of water, increasing the volume of surface runoff and compounding the effects of erosion. Thus, erosion reduces soil quality, making the soil prone to further erosion and further degradation.

Chemical Effects

Organic matter and clay are important sites of cation exchange in the soil. Cation-exchange capacity is a measure



Fig. 2 Severe erosion by water has removed all of the surface layer and much of the subsoil.
Source: From USDA, Natural Resources Conservation Service—Soil Quality Institute.

Table 1 Some general effects of soil erosion on soil-quality functions.

Erosion effects	Soil-quality functions affected
Reduced cation-exchange capacity	Sustaining plant growth and animal life Buffering or filtering Regulating energy
Formation of surface crusts	Regulating water, gases, and energy Sustaining plant growth and animal life
Changes in rooting volume	Sustaining plant growth and animal life
Changes in surface layer texture	Sustaining plant growth and animal life Regulating water and gases
Loss of organic matter	Sustaining plant growth and animal life Regulating energy
Compaction	Regulating water and gases Sustaining plant growth and animal life
Deterioration of soil structure	Regulating water and gases Sustaining plant growth and animal life
Exposure of subsoil material	Sustaining plant growth and animal life Regulating water, gases, and energy

of negatively charged sites on the soil particles that are capable of holding positively charged ions, including many plant nutrients. As organic matter and clay particles are lost from the soil surface through erosion, attached nutrients are relocated in the landscape, often to adjacent water bodies where they contribute to declining water quality. Loss of these fine soil particles also impairs the ability to store nutrients, reducing soil fertility. This effect is more pronounced in sandy soils containing small amounts of clay, although it may be offset to some degree if erosion causes the surface layer to become more clayey. Where subsoils are more enriched in clay than the surface soil, the clay particles may form chemical bonds with phosphorus, fixing it into forms not easily available to plants and thus reducing fertility.

For some acid soils with subsoil pH <5.0, concentrations of available aluminum may be at levels toxic to plant roots and removal of surface soil by erosion can effectively reduce the rooting depth of the soil.

Subsoil layers having pH >8.5 often contain high levels of sodium. Surface soil removal and subsequent exposure of these subsoils to rainwater or irrigation waters with low ionic concentrations can lead to dispersion of clay particles, loss of soil structure, surface sealing, and greatly reduced water infiltration.

Biological Effects

Removal of organic matter and nutrients from the soil surface by erosion reduces the food and energy supply needed to sustain healthy populations of soil organisms and support plant growth. Soil organisms are the agents of organic matter decomposition and nutrient cycling in soil.

In addition, they play an important role in the stabilization of soil aggregates through the production of binding agents such as roots, fungal hyphae, polysaccharides, gums, and complex molecules consisting of humic substances combined with iron, aluminum, or aluminosilicates.^[5,6] So, as erosion proceeds, the food source for organisms is reduced, leading to declines in populations. Soil structure and stability then deteriorate, the soil becomes more susceptible to further erosion, and the cycle continues.

CONCLUSION

The interaction of accelerated soil erosion and soil quality is complex. Soil erosion usually reduces soil quality, and a soil of poorer quality is less able to withstand erosion, thus creating a downward spiral of soil degradation. However, many soil-conservation practices have been useful in mitigating the effects of erosion. Reduced tillage systems limit soil disturbance and build soil structure. Crop residue management, underseeding, cover cropping, and permanent cover (e.g., pasture) protect the soil from the action of wind and water. Contour cultivation, grassed waterways, and terracing alter the flow of surface water, curbing water erosion. Herbaceous wind barriers and woody windbreaks and shelterbelts help to control wind erosion. Nevertheless, large tracts of agricultural land throughout the world are subject to the unsustainable loss of soil as a result of erosion

and continued adoption of preventive methods are needed to protect and restore soil quality.

REFERENCES

1. Govers, G.; Lobb, D.A.; Quine, T.A. Tillage erosion and translocation: Emergence of a new paradigm in soil erosion research. *Soil Till. Res.* **1999**, *51*, 167–174.
2. Lal, R., Ed. *Soil Quality and Soil Erosion*; CRC Press: Boca Raton, 1998.
3. Karlen Douglas, L.; Stott Diane, E. A framework for evaluating physical and chemical indicators of soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., Eds.; SSSA Special Publication No. 35; Soil Science Society of America: Madison, 1994; 53–72.
4. Topp, G.C.; Wires, K.C.; Angers, D.A.; Carter, M.R.; Culey, J.L.B.; Holmstrom, D.A.; Kay, B.D.; Langille, D.R.; McBride, R.A.; Patterson, G.T.; Perfect, E.; Rasiah, V.; Rodd, A.V.; Webb, K.T. Changes in soil structure. In *The Health of Our Soil: Toward Sustainable Agriculture in Canada*; Acton, D.F., Gregorich, L.F., Eds.; Agriculture Canada: Ottawa, 1995; 51–60.
5. Tisdall, J.M.; Oades, J.M. Organic matter and water-stable aggregation. *J. Soil Sci.* **1982**, *33* (2), 141–163.
6. Wright, S.F.; Starr, J.L.; Paltineanu, I.C. Changes in aggregate stability and concentration of glomalin during tillage management transition. *Soil Sci. Soc. Am. J.* **1999**, *63* (6), 1825–1829.

Quality: Indicators

Graham P. Sparling

Landcare Research, Hamilton, New Zealand

Abstract

This entry presents criteria to select soil properties for soil quality indicators. A minimum data set to measure soil quality will include key dynamic properties of the soil quality aspect being monitored, with regard to the spatial scale, and the time span. An interpretative framework is needed to link indicator response to management action. Interpretation will differ according to the soil and land use, and monitoring trends through time is useful where critical limits have not been defined. Multiple indicators can be combined to form a simple index, but in so doing, some soil quality information may be masked.

CRITERIA FOR SELECTING SOIL QUALITY INDICATORS

Many soil properties and characteristics have been proposed as indicators of soil quality.^[1,2] In order to be considered as a valuable indicator, a soil property must meet certain criteria: 1) it needs to get our attention; and 2) it needs to tell us something useful about the condition of the soil. Desirable attributes include being quantitative and responsive within the specified time scale; interpretable; cost effective; scientifically justifiable; socially acceptable; internationally recognized; and preferably a part of historical monitoring procedures.^[1–4]

There is some overlap between the categories of soil quality and land quality, of which soil quality is a subset.^[2,5] Indicators for land quality generally describe the inherent and intrinsic characteristics of the soil, usually on a landscape scale. Inherent characteristics, such as soil mineralogy and texture, soil depth, and stoniness, are used to assess land capability or suitability for use, and generally are not greatly affected by land management. Other inherent characteristics, such as slope and aspect, overlap with soil quality indicators because they influence the more rapidly changing and dynamic soil properties that are responsive to land management. The variety of proposed indicators reflects the many ways in which soil quality has been defined^[5] and the many components of a soil's chemical, physical, and biological attributes that contribute to its overall character. Of its many definitions, quality can be most briefly described as “fitness for use.”^[1,5] Within a soil context, a quality rating depends on how well the soil characteristics match the suitability for a particular use. (See the entry *Quality: Critical Limits and Standardization*, p. XXX.) Soil characteristics are not fixed—what is good soil quality for one land use may be deemed poor soil quality for another use. Thus, quality ratings are not absolute but are instead values based on

interpretation depending on the intended use of the soil.^[6] The soil itself does not change, only our perception of it in relation to human needs.

A Minimum Data Set

Because it is not feasible to measure all soil quality characteristics, indicators need to be selected to address the issues of greatest concern. There are three major categories of soil quality concerns: soil erosion and redistribution; chemical and biological contamination; and soil degradation and depletion. The examples of these categories given in [Table 1](#) are of concern because they adversely influence the role of the soil in ecosystem functioning and services.^[5] Their degree of importance varies in different parts of the world.

Time and Spatial Aspects

Selected indicators must be appropriate to the scale being considered, within both spatial and time dimensions. We tend to classify these items based on human perceptions, and soil processes that geologists consider rapid may be considered slow or relatively static to soil biologists. Examples of time and spatial scales and some relevant soil quality characteristics are shown in [Table 2](#). Human-induced changes to soil, including soil redistribution through erosion and deposition, organic matter depletion, and chemical and radioactive contamination, can take hundreds of years to remediate.

Interpretation

An interpretative framework is needed to make sense of the numeric values obtained for the various soil characteristics. We need to know what constitutes a high or low value, and what target value is desirable for each

Table 1 Major aspects of soil quality requiring indicators.

Soil quality aspect	Examples
Erosion and deposition	Soil redistribution by wind and water, mass movement, slips and slumps, and tillage displacement
Contamination/pollution	Presence of potentially toxic chemicals Pathogenic organisms
Degradation/depletion	Depletion of soil organic matter Loss of fertility Soil acidification Salinization Structural degradation

particular soil and land use. An interpretive framework should also indicate whether there might be off-site consequences for the wider ecosystem. For example, soluble fertilizer contents in soil may benefit crop production

(good quality), but not matching the fertilizer to plant needs may result in an excess, increasing the risk of contaminating receiving waters (poor quality). In general, interpretative frameworks are better defined for contaminants than for other soil quality measures, because the presence of a contaminant can be related to an increased risk to plant and animal health. A critical value can be defined based on toxicity. Such dose–response curves are not available for most indicators used for soil loss, deposition, or soil depletion, and in many cases the response curve is an ill-defined continuum rather than a critical point.^[2] Much work needs to be done to define justifiable targets for soil properties, and even the broad categories of “more is better,” “less is better,” or “optimum range” have been the subject of detailed discussion.^[7] The units of expression can also influence interpretation—expression on a volume or area basis is generally preferred, particularly for comparisons between soils and land uses with differing bulk densities.^[8]

Table 2 Spatial and time scales of soil properties important for soil quality.

Spatial scale	Time scale	Soil properties showing change	Soil quality aspects
Rapidly changing, highly dynamic characteristics			
1–100 mm	Minutes and hours	Moisture content and temperature	Moisture and warmth for plant growth and biological activity
100 mm–1 m	Hours and days	Infiltration and drainage	Compaction due to treading or wheeled traffic, risk of run-off during rainfall, waterlogging in poorly draining soils, poor root environment
		Biological activity	Decomposition of fresh organic matter
Intermediate characteristics			
1 m–100 m	Days and months	Available nutrients and mineralization	Nutrient depletion and supply, nutrient imbalances for plant growth
		Soil structure	Soil damage from compaction; influence on root environment, aeration, moisture retention, and drainage
100 m–1 km	1–5 years	Organic matter content	Turnover of moderately decomposable organic matter, nutrient supply
		Salinity	Saline water in root zone, toxic stress
		Acidification	Soil chemical process leading to acidification in root zone, nutrient imbalances, toxicity
		Erosion and redistribution	Loss of topsoil, soil deposition
Slowly changing characteristics			
1–10 km	5–25 years	Soil loss and formation	Loss of topsoil and decrease in soil depth, soil deposition
		Humus formation or loss	Changes in soil structure and nutrient storage due to loss of organic matter
		Salinity	Salinity in the root zone, toxic stress, rising water tables
		Acidification	Increasing acidity due to leaching and cation/anion imbalance, toxic stress
		`Weathering	Nutrient release and soil formation from rapidly weathering minerals
10–100 km	25–100 years	Stable organic matter	Humus provides exchange sites to retain ions in soil and contributes to soil aggregate formation
>100 km	>100 years	Soil formation	Nutrient release and soil formation from slowly weathering minerals
		Mineralogy	Proportions of sand, silt, and clay, exchange characteristics of the clays

Trends

In the absence of defined critical values for soil quality indicators, the importance of the trend over time has been emphasized.^[9] Trends away from a target range are interpreted as a decline in soil quality; trends toward the target are interpreted as improved soil quality. However, it can also be argued that short-term trends away from a target value could be regarded as sustainable, provided the trend can be reversed within an acceptable time frame. Within the present context, recovery or restoration within a human generation of 20–25 years could be regarded as preserving intergenerational equity.^[10] Such exploitation–recovery cycles were much used during the shifting agriculture phase of human development. Soils differ in their resistance to change, and the rates at which they recover^[11,12] affects their soil quality rating. Generally, soils that are resistant to change, or do change but show a rapid rate of recovery, are regarded as having better quality.^[11]

Combining Indicators

Because multiple data sets are difficult to comprehend, it has been proposed that indicator sets be combined into a single index.^[1,2] These single indices are usually additive or multiplicative combinations of the individual indicators, and a weighting factor is used to adjust for the relative importance of each factor.^[13] While a single number is easier to remember and communicate, its use has the disadvantage of possibly masking information. When too many indicators are combined, poor-quality soil aspects may be masked by those of modest or good quality. Reporting on any soil environmental quality should reflect the limiting quality rather than the average value. For example, excellent chemical fertility is of little use if the physical structure of the soil is badly degraded. This poor physical structure would therefore dominate the soil characteristics.

CONCLUSION

Indicators must attract attention and inform. They need to be sufficiently sensitive for their intended task and to respond in a timely manner. Critical values can be set so that a particular indicator value will trigger a management response. The critical values will differ, depending on the soil and land use, and management options. The ability of soils to recover, and their susceptibility to change under use, will define which indicators are useful and where their critical limits may be set. Multiple indicators can be combined into a single index, using weighting factors to adjust for the relative importance of each indicator.

When combining indicators, care is needed not to mask poor indicator scores, and a safer option is to identify which soil characteristics are limiting factors

REFERENCES

1. Doran, J.W.; Parkin, T.B. Defining and assessing soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., Eds.; SSSA Special Publication No. 35; Soil Science Society of America: Madison, 1994; 3–21.
2. Karlen, D.L.; Mausbach, M.J.; Doran, J.W.; Cline, R.G.; Harris, R.F.; Schuman, G.E. Soil quality: A concept, definition, and framework for evaluation (Guest Editorial). *Soil Sci. Soc. Am. J.* **1997**, *61*, 4–10.
3. Doran, J.W.; Coleman, D.C.; Bezdicek, D.F.; Stewart, B.A. *Defining Soil Quality for a Sustainable Environment*; SSSA Special Publication, Vol. 35; Soil Science Society of America: Madison, 1994; 244 pp.
4. Cornforth, I.S. Selecting indicators for assessing sustainable land management. *J. Environ. Manage.* **1999**, *56*, 173–179.
5. Carter, M.R.; Gregorich, E.G.; Anderson, D.W.; Doran, J.W.; Janzen, H.H.; Pierce, F.J. Concepts of soil quality and their significance. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 1–19.
6. Singer, M.J.; Ewing, S. Soil quality. Interdisciplinary aspects of soil science. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: New York, 1999; 272–298.
7. Sojka, R.E.; Upchurch, D.R. Reservations regarding the soil quality concept. *Soil Sci. Soc. Am. J.* **1999**, *63* (5), 1039–1054.
8. Reganold, J.P.; Palmer, A.S. Significance of gravimetric versus volumetric measurements of soil quality under biodynamic, conventional, and continuous grass management. *J. Soil Water Conserv.* **1995**, *50* (3), 298–305.
9. Larson, W.E.; Pierce, F.J. The dynamics of soil quality as a measure of sustainable management. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., Eds.; SSSA Special Publication No. 35; Soil Science Society of America: Madison, 1994; 37–51.
10. Lunney, D.; Pressey, B.; Archer, M.; Hand, S.; Godhelp, H.; Curtin, A. Integrating ecology and economics: Illustrating the need to resolve the conflicts of space and time. *Ecol. Econ.* **1997**, *23*, 135–143.
11. Greenland, D.J.; Szabolcs, I. *Soil Resilience and Sustainable Land Use*; CAB International: Wallingford, 1994.
12. Seybold, C.A.; Herrick, J.E.; Brejda, J.J. Soil resilience: A fundamental concept of soil quality. *Soil Sci.* **1999**, *164*, 224–234.
13. Andrews, S.S.; Karlen, D.L.; Mitchell, J.P. A comparison of soil quality indexing methods for vegetable production systems in Northern California. *Agr. Ecosyst. Environ.* **2002**, *90*, 25–45.

Quality: Productivity

Mike H. Beare

Canterbury Agricultural and Science Centre, New Zealand Institute for Crop and Food Research, Christchurch, New Zealand

Abstract

The capacity to nurture and sustain plant and animal productivity is a key function of high quality soils. Indicators of soil quality reflect the key properties and processes that support this function and can be used to assess the fitness of soils for production. In addition to extrinsic factors (e.g., climate), productivity is influenced by both the intrinsic characteristics of a soil (i.e., inherent soil quality) and those processes or properties that are affected by its use and management (i.e., dynamic soil quality). Sustainable production depends on selecting land uses that are well suited to the capability of the soil (and wider environment) and maintaining soil conditions that minimize the risk of productivity declines. The soil quality conditions (e.g., indicator optimum ranges) required to sustain agricultural productivity may be different from, and perhaps less stringent than, those needed to ensure environmental, economic, or social sustainability.

PRODUCTIVITY INDICATORS

Plant productivity depends directly or indirectly on a soil's ability to carry out a variety of functions.^[1,2] The fitness of soil to carry out these functions within acceptable boundaries is the basis of soil quality assessment. Soil quality indicators that can characterize this fitness are useful in assessing plant productivity. For example, a soil's ability to accommodate unrestricted plant emergence and root growth and to permit the effective infiltration and supply of air, water, and nutrients may be described by physical indicators of soil quality.^[3] Chemical indicators of soil quality for plant productivity include properties associated with plant nutrition (e.g., nutrient availability and soil acidity) and properties that reflect an inhibitory or toxic effect on plant growth (e.g., presence of pesticide residues). Soil also supports a diverse community of organisms that both affect and are affected by its chemical and physical properties. Some biological indicators of soil quality reflect a risk to plant productivity (e.g., fungal pathogens), whereas others indicate a potential to enhance plant growth (e.g., plant growth-promoting bacteria). There has been much interest in using biological indicators to describe a soil's capacity to regulate or maintain important soil physical or chemical properties.^[4] For example, the soil microbial biomass may be used as a biological indicator because it reflects a pool of potentially available nutrients (i.e., chemical function) or the ability of soil to maintain soil organic matter levels (i.e., chemical function) or soil structural stability (i.e., physical function).

Animal productivity depends to a significant extent on plant productivity. However, many other factors influence animal productivity, some of which are determined directly

or indirectly by soil quality. These factors include forage nutrient content (e.g., selenium) and quality (e.g., protein content), chemical contaminants (e.g., dichlorodiphenyltrichloroethane), and soil-borne diseases (e.g., parasites). This discussion focuses on soil quality for plant productivity though many of the concepts also apply to animal productivity.

INHERENT SOIL QUALITY

Characteristics that define a soil's inherent capacity for plant production are usually static, changing little over relatively short time frames (years to decades).^[1] They are the product of soil formation driven by climate, topography, parent material, vegetation, and time. Soil mineralogy and particle size distribution are commonly included as properties of inherent soil quality for productivity.^[1] Other attributes, such as total soil carbon, cation exchange capacity, and sodicity, may be defined as inherent properties, where broad soil type or regional comparisons are of interest at one point in time even though they may be altered by management over longer time frames (e.g., decades to centuries).

Inherent soil properties are the basis of many land use capability or suitability assessments^[5,6] that are key components of land use planning and policy development in many regions of the world. Most were undertaken with the primary aim of evaluating potential soil productivity.^[7,8] These assessments are usually developed around a broad set of "land quality" criteria that integrate extrinsic factors (e.g., climate, topography, and hydrology) affecting the composition and productivity of plant communities in an

area with the inherent attributes of the soil. The interplay between these extrinsic factors and inherent soil quality determines the suitability of a land area for a particular agricultural use (e.g., arable cropping, extensive pastoral farming, and forestry).

Thus, the inherent quality of a soil should be viewed in light of its intended agricultural use. For example, in warm humid climatic zones, free-draining, stony soils may be judged as low quality for arable cropping but of high quality for viticultural production. Similar, but subtler, differences in inherent soil quality have been applied to the selection of specific crops, cultivars, or rotations. The Canadian Land Suitability Rating System,^[9] e.g., categorizes land into suitability classes based on the degree to which its environment limits the productivity of specific crops. Suitability classes are indexed from ratings that reflect limitations to productivity posed by climate (e.g., moisture and temperature), soil (e.g., topsoil depth and drainage), and landscape (e.g., slope and stoniness) factors.

DYNAMIC SOIL QUALITY

Soils of high inherent quality for a specific use may not necessarily function within their optimum range (i.e., exhibit maximum potential plant productivity) because of past use and management. This aspect of soil quality is determined by the dynamic properties of a soil.

Properties of dynamic soil quality are those that change in response to human use and management,^[1] normally over relatively short time frames (years to decades). Agricultural soils of high dynamic quality maintain high nutrient availability, permit adequate infiltration of air and water, have a relatively stable structure, and maintain a functionally diverse community of soil organisms that support relatively high levels of plant productivity. These processes are reflected in the specific physical, chemical, and biological properties of soils. The soil properties of greatest importance to productivity in a particular agricultural system are often grouped into a minimum data set^[3,5] defined by identifying the primary issues of soil management (e.g., nutrient deficiencies, risk to compaction, and water storage) that influence crop production on the soils and in the production systems in which they are to be used.^[10]

The terms “dynamic soil quality” and “soil health” are often used interchangeably.^[1] Two soils may be equally “healthy” but achieve different levels of plant productivity because of differences in their inherent quality. In terms of productivity, the optimum range or critical limit of a soil quality indicator is usually defined by applying established criteria to empirically derived relationships between the soil property and some measure of productivity (e.g., annual crop yield and root biomass at crop maturity; Fig. 1).^[10,11] Critical limits for productivity have been established for a number of soil physical and chemical properties. In many cases, as in the example given (Fig. 1), the criteria used in

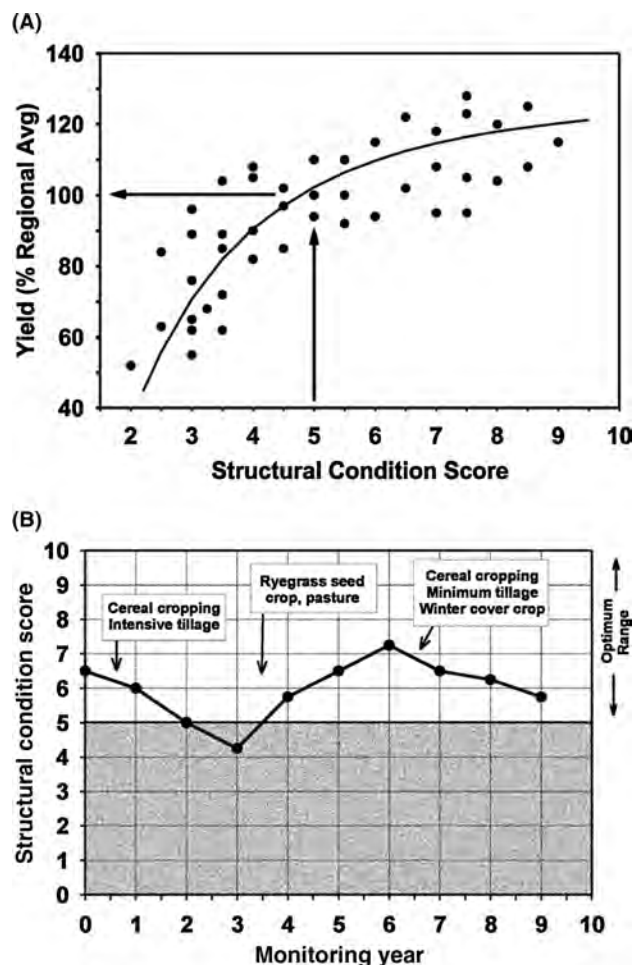


Fig. 1 (A) A relationship between soil structural condition and relative crop yield used to define the indicator's critical limit. (B) An example of how the critical limit for soil structural condition has been used to improve soil management decisions on the Canterbury Plains, New Zealand.

Source: From Beare, Williams, et al.^[10]

defining critical limits are based on the objectives of the user (e.g., yield targets and profit margins). Where relationships between individual properties and productivity are less clear, some studies have succeeded in indexing data from several different soil properties (minimum data set) into a single value that explains much more of the variation in productivity.^[12] While this approach has some advantages where complex interactions between factors may affect productivity, a single value can also mask information that is important to identify and manage the limitations to productivity.

Many factors influence the level of crop production in any 1 year (e.g., climatic conditions, cultivars, disease incidence, fertilizer rates, and timing). For this reason, the relationship between dynamic properties (or indicators) of soil quality and crop productivity is often difficult to define at a field scale. Where the relationship is known, the loss of dynamic soil quality is probably best described as a risk

to crop productivity. In this respect, the management of dynamic soil quality is often aimed at lowering the risk of productivity loss.

SOIL QUALITY AND SUSTAINABLE PRODUCTION

The use of soil quality information to evaluate the sustainability of soil management systems has involved two common approaches.^[3] With the *comparative assessment* approach, the performance of a system is determined in relation to alternative systems using one-off measures of soil properties. This approach is often used to evaluate potential impacts of land use change. With the *dynamic assessment* approach, the performance of a management system is assessed by measuring the changes in soil properties over time. For example, Fig. 2 shows the nutritive [nitrogen (N) and phosphorus (P)] benefits of different organic amendments (e.g., cattle manure, composted manure, barley straw, and pea hay) for restoring the productivity (wheat yield) of a desurfaced (artificially eroded) cropping soil in Southern Alberta, Canada.^[14] In this example, 87% of the restored yield could be attributed to N and P in the organic amendments, while 13% was attributed to soil structural improvements.

In agricultural systems, the applications of dynamic assessment approach usually involve both monitoring (i.e., repeated measurement) and control (i.e., regulation or management), and the focus shifts from describing soil conditions to maintaining soil conditions that sustain high productivity through improved management (Fig. 1). To this end, soil quality monitoring may be coupled with best management practice recommendations to develop soil

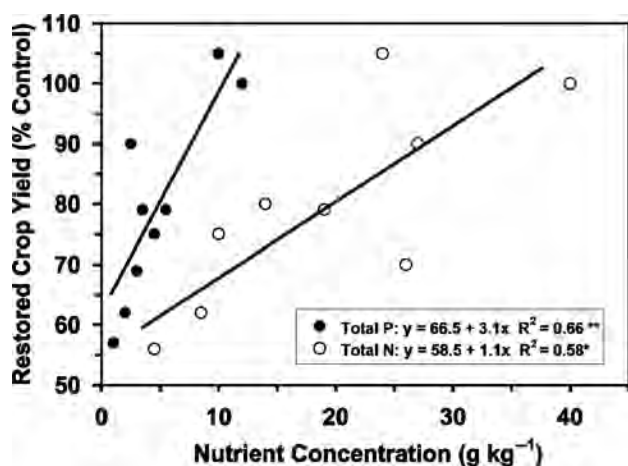


Fig. 2 Effects of total N and P concentrations in organic amendments on the restoration of wheat yields of a desurfaced (artificially eroded) cropping soil in Southern Alberta, Canada. Restoration of crop yields is expressed as a percentage of the non-desurfaced (uneroded topsoil) control.

Source: Adapted from Haynes, Swift, et al.^[13]

management decision support systems that improve the sustainability of production systems.

Using the dynamic assessment approach, sustainable soil management practices are often defined as those that maintain or improve dynamic soil quality.^[3] However, from the standpoint of productivity alone, this represents a somewhat narrow definition of sustainable management. In the short term, agricultural production often results in a degradation of the ordered complex structures that characterize natural ecosystems in equilibrium.^[15] Sustainable management of soil for crop production may involve practices that approach a new steady state or balance degradative processes by restorative processes over relatively short time frames (Fig. 3). For example, in New Zealand and other regions of the world, farmers practice mixed cropping rotations in which the degradative effects of short-term arable cropping are balanced by the restorative effects of pastoral management under grazing.^[13] Where best management practices are employed, the degradation of soil properties is not usually sufficient to significantly impair crop production (Fig. 3). However, the critical limit (or optimum range) for soil properties (or indicators) that sustain high levels of crop production may be less restrictive than those needed to achieve environmental, economic, or social sustainability. Furthermore, a decline in soil quality can be compensated for by an increase in production inputs (e.g., fertilizer, tillage, and pesticides) that may ultimately lower profitability or result in adverse environmental impacts (e.g., nitrate leaching).

The adoption of specific soil management practices is often driven by practical or economic considerations, rather

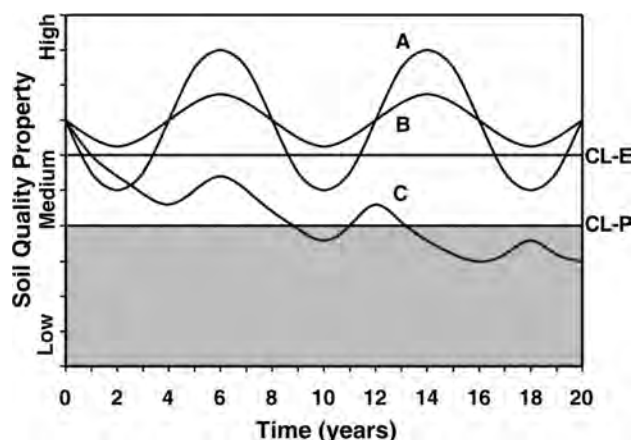


Fig. 3 A conceptual illustration of changes in soil quality under different types of management in relation to critical limits for productivity (CL-P) and environmental impacts (CL-E). Three types of mixed-cropping (degradative–restorative) management are shown: (A) intensive (e.g., cereal cropping with conventional tillage followed by grazed pasture); (B) extensive (e.g., cereal cropping with minimum tillage followed by grass seed crops and pasture); and (C) unbalanced (i.e., restorative practices insufficient to offset degradative management).

than the potential benefits of improved soil quality for crop production and the environment. Agricultural profitability is difficult to achieve and sustain without a relatively high productivity, but high productivity does not necessarily ensure environmental, economic, or social sustainability. Nevertheless, there are many documented cases in which soil management practices designed to lower production costs and to reduce environmental impacts have also resulted in improved crop performance and sustained economic performance.

REFERENCES

1. Carter, M.R.; Gregorich, E.G.; Anderson, D.W.; Doran, J.W.; Janzen, H.H.; Pierce, F.J. Concepts of soil quality and their significance. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 1–9.
2. Larson, W.E.; Pierce, F.J. The dynamics of soil quality as a measure of sustainable management. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezedick, D.F., Stewart, B.A., Eds.; Special Pub. No. 35; American Society of Agronomy: Madison, 1994; 37–51.
3. Doran, J.W.; Parkin, T.B. Defining and assessing soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezedick, D.F., Stewart, B.A., Eds.; Special Pub. No. 35; American Society of Agronomy: Madison, 1994; 3–21.
4. Gregorich, E.G.; Carter, M.R.; Angers, D.A.; Monreal, C.M.; Ellert, B.H. Towards a minimum data set to assess soil organic matter quality in agricultural soils. *Can. J. Soil Sci.* **1994**, *74*, 367–385.
5. Wösten, J.H.M.; Bouma, J.; Stoffelsen, G.H. Use of soil survey data for regional soil water simulation models. *Soil Sci. Soci. Am. J.* **1985**, *49*, 1238–1244.
6. Webb, T.H.; Claydon, J.J.; Harris, S.R. Quantifying variability of soil physical properties within soil series to address modern land-use issues on the Canterbury Plains, New Zealand. *Austr. J. Soil Res.* **2000**, *38*, 1115–1129.
7. Huddleston, J.H. Development and use of soil productivity ratings in the United States. *Geoderma* **1984**, *32*, 297–317.
8. Petersen, G.W.; Bell, J.C.; McSweeney, K.; Nielsen, G.A.; Robert, A.C. Geographic information systems in agronomy. *Adv. in Agron.* **1995**, *55*, 67–111.
9. Pettapiece, W.W. *Land Suitability Rating System for Agricultural Crops. 1. Spring-seeded Small Grains*. Technical Bulletin 1995-6E. Centre for Land and Biological Resources Research, Research Branch, Agriculture and Agri-Food Canada: Ottawa, 1995.
10. Beare, M.H.; Williams, P.H.; Cameron, K.C. On-farm monitoring of soil quality of sustainable crop production. In *Best Soil Management Practices for Production*, Proceedings of the Fertiliser and Lime Research Centre Conference, Curries, L.D., Hedley, M.J., Horne, D.J., Loganathan, P., Eds.; Massey University: Palmerston North, New Zealand, 1999; 81–90.
11. Pierce, F.J.; Gilliland, D.C. Soil quality control. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 203–219.
12. Hussain, I.; Olson, K.R.; Wander, M.M.; Karlen, D.L. Adaptation of soil quality indices and applications to three tillage systems in southern Illinois. *Soil Till. Res.* **1999**, *50*, 237–249.
13. Haynes, R.J.; Swift, R.S.; Stephen, R.C. Influence of mixed cropping rotations (pasture-arable) on organic matter content, water stable aggregation and clod porosity in a group of soils. *Soil Till. Res.* **1991**, *19*, 77–87.
14. Addiscott, T.M. Entropy and sustainability. *Eur. J. Soil Sci.* **1995**, *46*, 161–168.
15. Larney, F.J.; Janzen, H.H. Restoration of productivity to a desurfaced soil with livestock manure, crop residue, and fertilizer amendments. *Agron. J.* **1996**, *88*, 921–927.

Quality: Soil and Water

Todd Nissen

University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.

Michelle Wander

*Department of Natural Resources and Environmental Sciences,
University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.*

Abstract

Sediments and chemicals originating on farms impair more U.S. waters than any other source. Managing soil quality is an essential part of a strategy to reduce these forms of pollution and the costs they incur. Water quality concerns focus attention on soil functions related to regulating water flow and absorbing, buffering, and transforming chemical flows. Organic matter plays a significant role in a soil's ability to perform these functions and serves as one of several key indicators for measuring soil quality. Indicators, however, are scale dependent, and care must be taken in extending soil quality data—typically collected at the field level—to assess water quality at later temporal and larger physical scales. The use of interdisciplinary approaches to implement soil quality mirrors a trend already seen in the development of the water quality concept: after the technical details of indicator selection, sampling, and interpretation are addressed, disciplinary information is integrated into problem-solving frameworks for use by individuals, communities, municipalities, and nations.

SOILS AND WATER QUALITY

According to a U.S. Environmental Protective Agency report of 1995, the leading source of water pollution in the United States is agriculture (Table 1). Declines in water quality worldwide have also been blamed chiefly on the increases in agricultural inputs and crop productivity that have occurred since the 1950's.^[1,2] As efforts in industrialized countries have succeeded in mitigating municipal/industrial point source pollution, the chemicals and sediments that originate on farms impair more aquatic habitats and drinking water supplies than any other source.

In response, many land managers are becoming increasingly judicious in their use of chemical inputs. In the U.S. Corn Belt between 1982 and 1992, e.g., commercial nitrogen (N) and phosphorus (P) consumption dropped by 3.5% and 21%, respectively.^[3,4] But crop productivity is not simply a function of the amount of chemicals applied—soils differ in their ability to release nutrients, maintain adequate moisture and oxygen levels, and allow deep rooting and other functions related to productivity that determine the need for, and efficiency of, chemical inputs. The same is true of water quality—many soil factors affect water quality in addition to, and in feedback with, chemical applications (Fig. 1). For example, increased soil compaction can result from excessive or poorly timed tillage operations, lowering the rate of water infiltration into the soil and leading to increased overland

flow, soil erosion, and stream channel erosion. These processes carry adsorbed nutrients, pesticides, and salts into surface waters; reduce clarity and aquatic plant photosynthesis; and fill channels and reservoirs, limiting boat passage, water treatment capacity, and hydroelectric capacity. In the United States, off-farm erosion costs were estimated to be \$17 billion annually.^[5] That management practices cause differences in the ability of a soil to maintain or enhance water quality is one of the bases for evaluating soils in terms of their quality. Although soil quality is not limited to agricultural applications, it is most often considered in this context because of agriculture's large impact on water quality.

SOIL FUNCTIONS RELATED TO WATER QUALITY

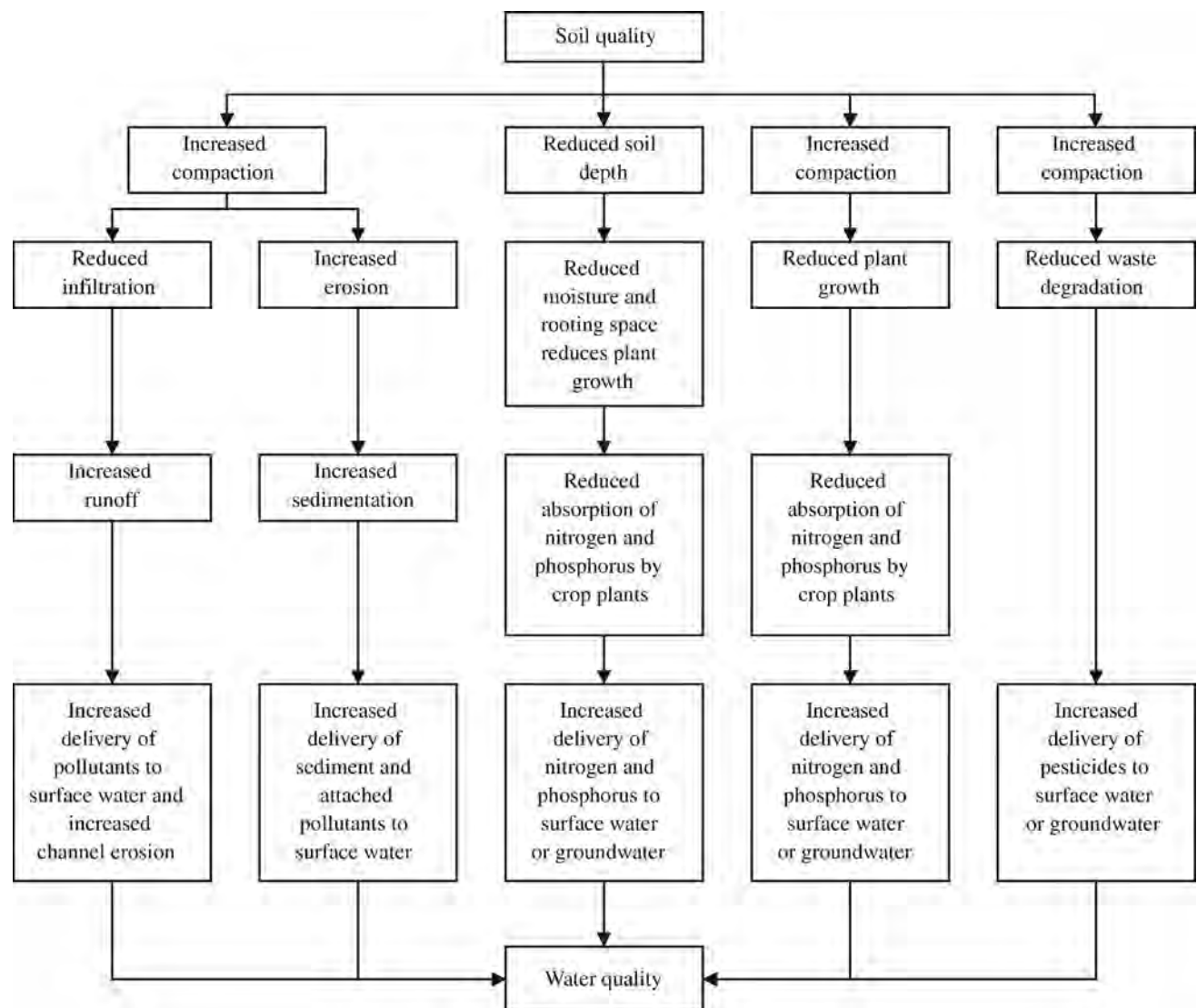
One approach to soil quality evaluates a soil's ability to perform its essential *functions*. Soil functions specifically related to water quality include the ability to partition and regulate water flow and absorb, buffer, and transform chemical flows.

Partition and Regulate Water Flow

Soils are an important switching station in the global water cycle, partitioning precipitation into overland or infiltrated flow. A soil's water-holding capacity and the rate at which water infiltrates the soil help determine the proportion of

Table 1 Leading sources of pollution of assessed waters in the United States: 1992 and 1993.

Sources	Rivers and streams (10 ³ miles) ^a	Rank	Lakes and reservoirs (10 ³ acres) ^b	Rank	Estuaries (sq mi) ^c	Rank
Agriculture	135	(1)	3350	(1)	3321	(3)
Hydro/habitat change	37	(4)	— ^d	— ^d		
Municipal/industrial point sources	53	(2)	2025	(2)	6436	(1)
Natural	42	(3)	965	(5)	2949	(4)
Unspecified non-point sources	— ^d	989	(4)	991	(5)	
Urban runoff/storm sewers	27	(5)	1200	(3)	4508	(2)

^aTotal river and stream mileage is 3.5 million miles.^bTotal lake, reservoir, and pond area is 40.8 million acres.^cTotal estuarine water area is 34.4 thousand square miles.^dNot among the top 5 sources.**Source:** From Water Quality and Agriculture: Status, Conditions, and Trends.^[3]**Fig. 1** Changes in soil quality affect water quality.**Source:** From National Academy of Sciences.^[6]

rainfall that flows through the soil matrix, which in turn controls the volume of stream flow and the rate of ground-water recharge.

Absorb, Buffer, and Transform Chemical Flows

In natural settings, soils receive and store inputs of N, P, and organic wastes, transforming them into forms available for plant uptake at rates that rarely exceed plant demand. Losses to surface and groundwaters are minimal. In agriculture, water pollution occurs when these inputs exceed the capacity of a soil/plant system to process them or when the soil itself is transported through erosion. Excess N and P levels in lakes and estuaries are the main causes of eutrophication, depleting oxygen levels for aquatic life.^[1,2] Elevated concentrations of nitrates, pesticides, and other organic compounds in waters pose direct risks to human health, although at such levels, these risks are less than to aquatic ecosystems.^[4]

Fig. 1 highlights the important feedback among productivity, soil quality, and water quality. As soil quality declines, so does productivity, reducing the uptake capacity of the plants and leaving more chemicals exposed to leaching and runoff. Profitability likewise declines as input-use efficiency is diminished. In general, farming profitability and water quality are complementary goals, as both are enhanced by improved input-use efficiency resulting from improved soil quality.^[6–8]

INDICATORS OF SOIL QUALITY RELATED TO WATER QUALITY

As these functions may be maintained or enhanced, research in soil quality has sought to identify the relative importance of management-affected soil indicators on multiple outcomes, the optimal levels of these indicators, and the extent to which these indicators are changed by management.

Selecting Indicators

The relative importance of an indicator depends on the outcome(s) of interest. As an example, Karlen and Stott^[9] selected indicators for erosion based on a sensitivity analysis of a simulation model (Table 2). Weightings were assigned to indicators based on their relative influence on sediment loss, with infiltration rate and aggregate stability emerging as the leading indicators. If runoff and leaching concentrations of N and P had been the outcomes of interest, macronutrient concentrations would have been weighted more heavily than they were for sediment loss.

For soil quality monitoring, selected indicators are evaluated either against themselves in relation to a temporal baseline or against “benchmark” soils. Because

water quality is generally high in natural systems such as forests and prairies, undisturbed soils are often used as benchmarks for managed soils with the same inherent properties.^[10–12]

Influence of Management on Indicators

As Table 2 indicates, maintaining soil cover and good soil physical conditions positively affects water quality. Both are related to organic matter management. On the surface, residues and cover crops protect against erosion by absorbing raindrop and flow energy. Together with roots, these additions become the young soil organic matter (SOM) that plays an important role in stable aggregation.^[13–15] Tillage and cropping regimes that favor organic matter accumulations generally maintain or improve water quality. For example, in data from Wisconsin where surface cover was not significantly different between conventional tillage and no-till, lower erosion in the no-till was associated with the enhancement of SOM-influenced characteristics (Table 3). Because of their contributions to SOM status, animal manures, municipal sludges, and some industrial by-products, if applied properly, can also have a beneficial impact on water quality, especially in the context of alternative disposal methods such as lagoons and landfills.^[16,17] Management strategies to improve SOM status and reduce erosion may nevertheless have other water quality tradeoffs. For example, a complete evaluation of no-till would include its reliance on pesticides compared to alternative weed control strategies such as cover crops.^[17]

SCALES OF SOIL AND WATER QUALITY

The influence of management on soil quality is typically measured at the field scale, yet its influence on water quality is expressed at the landscape or watershed level. The time scales over which practices affect soil and water properties also differ. Efforts to extrapolate soil quality information to larger scales are plagued by the fact that processes are not scale independent.^[18,19]

At larger physical and temporal scales, the connection between soil and water quality has been made indirectly by relating management practices to outcomes.^[20] The Universal Soil Loss Equation (USLE), which was developed and implemented before the soil quality concept was popularized, is the best example of this kind of soil and water management strategy.^[21] The USLE and related models have been used to estimate payments to farmers by U.S. government conservation programs, based in part on the amount of crop residue measured on individual farms. Those models have also provided the basis for efforts to scale up field-based information for use at the watershed level.^[22]

Table 2 Soil quality functions and indicators related to water erosion, based on a sensitivity analysis of an erosion simulation model.

Function	Weight	Indicator			
		Level I	Weight I	Level II	Weight II
Accommodate water entry	0.50	Infiltration rate	1.0	Surface crust	0.20
				Surface roughness	0.20
				Crop residue cover	0.50
				Macropores	0.10
Facilitate water transfer and absorption	0.10	Hydraulic conductivity	0.60	Soil texture	0.50
				Capillary water content	0.30
				Bulk density	0.20
		Porosity	0.15		
		Macropores	0.25	Plant roots	0.40
				Earthworms	0.60
Resist degradation	0.35	Aggregate stability	0.80	Mineralogy	0.20
				Soil carbohydrates	0.35
				Microbial biomass	0.30
				Available cations	0.15
Sustain plant growth	0.05	Shear strength	0.10		
		Soil texture	0.05		
		Heat transfer capacity	0.05		
		Rooting depth	0.25	Restrictive layer depth	0.70
				Texture	0.30
		Water relations	0.35	Available water capacity	0.70
				Drainage	0.20
				Organic C	0.10
		Nutrient relations	0.30	pH	0.15
				Organic C	0.25
				Macronutrients	0.40
				Micronutrients	0.20
		Chemical relations	0.10	Salinity	0.50
				Heavy metals	0.15
				Organics	0.25
				Radioactivity	0.10

Note: Weightings reflect relative influence of function or indicator on the next highest level.

Source: From Karlen & Stott.^[9]

The successful experience with the USLE points to opportunities for using spatially explicit models to extrapolate field scale indicators to larger scales. Model sensitivity analysis,^[9] metamodeling techniques,^[23,24] and spatial analyses of watershed land use patterns^[25] will be useful in relating soil quality indicators to desirable outcomes for water quality. Spatial analyses are also likely to increase the number of options for managing water quality. For example, watersheds may contain sinks for excess nutrients such as riparian zones to help mitigate plot-level problems.^[26] Some producers will find the construction of a wetland or riparian buffer strips a more economical way to reduce nutrient loadings in streams than reducing application rates of fertilizers.

DECISION MAKING AND POLICY

Establishing the scientific links between soil quality and water quality will not be enough to affect soil management decisions. In order for soil quality to become a valued decision-making tool, methodologies need to be developed that can estimate the economic or environmental value of marginal improvements in soil quality.^[27] Simulation models will be useful in assessing the dynamic biophysical tradeoffs for a wide range of what-if scenarios and management options relating soil and water quality.^[28] The tradeoffs that result from emphasizing one outcome over another—e.g., productivity over nitrate concentration in an adjacent water body—are difficult to evaluate using

Table 3 Soil quality data from surface soils (0–7.5 cm) in Wisconsin after 12 years of tillage treatments.

Parameter	No-till	Chisel	Moldboard	LSD
Bulk density (g cm ⁻³) ^a	1.48	1.4	1.42	0.06
Turbidity (log% transmittance) ^b	1.36	1.04	0.97	0.21
Total C in aggregates (g kg ⁻¹)	24	16	11	4
Biomass C (Mg C kg ⁻¹ soil)	696	394	260	276
Respiration (Mg C kg ⁻¹ soil)	352	139	74	114
Earthworms (0.25 m ⁻²)	78	52	53	18
Average surface cover (%)	68		58	NS
Soil quality index ^c	0.68	0.49	0.48	
Corn yield (t ha ⁻¹)	8.6	8.7	8.8	NS
Runoff amount (mm)	35 ^{a,d}		42 ^a	
Sediment concentration (g L ⁻¹)	1.4 ^b		5.0 ^a	
Estimated soil loss (Mg ha ⁻¹)	0.5 ^b		2.1 ^a	

Notes: LSD, least significant difference; NS, not significant. Measurements of runoff and sediment loss obtained from sprinkler-infiltration study.

^aBulk density calculated from porosity where $BD = (1 - \text{proportion porosity}) \times \text{particle density}$.

^bA measure of aggregate stability.

^cIndex value calculated by aggregating scores of individual parameters; scores assigned based on estimated optimal values.

^dValues followed by the same letter within the same row are not significantly different at the $p < 0.05$ level.

Source: Adapted from Karlen, Wollenhaupt, et al.^[18] and Halvorson, Smith, et al.^[19]

traditional cost–benefit analyses. Interdisciplinary efforts to identify human preferences and assign economic value to nature, health, and other elusive human welfare outcomes are being used to bridge this gap.^[29–32]

CONCLUSION

Several decades ago, the concept of water quality itself was unfamiliar and controversial.^[33] Evolving through this same interdisciplinary process, water quality has become the centerpiece, along with air quality, of resource management strategy and policy. The very dependence of water quality on soils ensures that farmers and developers, and individuals and governments, will also need to consider soil quality in making resource-use decisions.

REFERENCES

- Mannon, A.M. The environmental impact of agriculture in middle and high latitudes. In *Agriculture and Environmental Change: Temporal and Spatial Dimensions*; John Wiley and Sons: Chichester, 1995; 227–263.
- Tilman, D. Global environmental impacts of agricultural expansion: The need for sustainable and efficient practices. *Proc. Natl. Acad. Sci. U.S.A.* **1999**, *96* (11), 5995–6000.
- Lander, C. *Minimizing Agricultural Nonpoint Source Impacts with Management of Nutrients for Crop and Forage Production*; NRCS: Washington, 1994.
- Water Quality and Agriculture: Status, Conditions, and Trends*; Working Paper #16; National Resources Conservation Service, 1997; 1–125. Available at http://www.nrcs.usda.gov/wps/portal/nrcs/detail/ri/technical/dma/?cid=nrcs143_014066 (accessed August 21, 2015).
- Pimentel, D.; Harvey, C.; Resosudarmo, P.; Sinclair, K.; Kurz, D.; McNair, M.; Crist, S.; Shpritz, L.; Fitton, L.; Saffouri, R.; Blair, R. Environmental and economic costs of soil erosion and conservation benefits. *Science* **1995**, *267* (5201), 1117–1123.
- National Academy of Sciences. *Soil and Water Quality: An Agenda for Agriculture*; National Academy Press: Washington, 1993.
- Wander, M.M.; Walter, G.; Nissen, T.M.; Bollero, G.A.; Andrews, S.S.; Cavanaugh-Grant, D. Soil quality: Science and process. *Agron. J.* **2002**, *94*, 23–32.
- Cassman, K.G. Ecological intensification of cereal production systems: Yield potential, soil quality, and precision agriculture. *Proc. Natl. Acad. Sci. U.S.A.* **1999**, *96* (11), 5952–5959.
- Karlen, D.L.; Stott, D.E. A framework for evaluating physical and chemical indicators of soil quality. In *Defining Soil Quality for a Sustainable Environment*; SSSA Special Publication No. 35; Soil Science Society of America: Madison, 1994; 53–72.
- Seybold, C.A.; Mausbach, M.J.; Karlen, D.J.; Rogers, H.H. Quantification of soil quality. In *Carbon Sequestration in Soil. Advances in Agronomy*; Stewart, B.A., Lal, R., Eds.; Academic Press: San Diego, 1997; 387–404.
- Brye, K.R.; Norman, J.M.; Bundy, L.G.; Gower, S.T. Water-budget evaluation of prairie and maize ecosystems. *Soil Sci. Soc. Am. J.* **2000**, *64* (2), 715–724.
- Wander, M.M.; Bollero, G.A. Soil quality assessment of tillage impacts in Illinois. *Soil Sci. Soc. Am. J.* **1999**, *63* (4), 961–971.
- Chenu, C.; Le Bissonnais, Y.; Arrouays, D. Organic matter influence on clay wettability and soil aggregate stability. *Soil Sci. Soc. Am. J.* **2000**, *64* (4), 1479–1486.
- Six, J.; Elliott, E.T.; Paustian, K.; Doran, J.W. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Sci. Soc. Am. J.* **1998**, *62* (5), 1367–1377.
- Beare, M.H.; Hendrix, P.F.; Coleman, D.C. Water-stable aggregates and organic matter fractions in conventional-tillage and no-tillage soils. *Soil Sci. Soc. Am. J.* **1994**, *58* (3), 777–786.
- Karlen, D.L.; Wright, R.J.; Kemper, W.D. *Agricultural Utilization of Urban and Industrial By-Products*; ASA Special Publication 58; ASA, CSSA, and SSSA: Madison, 1995.
- Kelly, T.C.; Lu, Y.C.; Teasdale, J. Economic-environmental tradeoffs among alternative crop rotations. *Agric. Ecosyst. Environ.* **1996**, *60* (1), 17–28.
- Karlen, D.L.; Wollenhaupt, N.C.; Erbach, D.C.; Berry, E.C.; Swan, J.B.; Eash, N.S.; Jordahl, J.L. Long-term

- tillage effects on soil quality. *Soil Till Res.* **1994**, 32 (4), 313–327.
19. Halvorson, J.J.; Smith, J.L.; Papendick, R.I. Issues of scale for evaluating soil quality. *J. Soil Water Conserv.* **1997**, 52 (1), 26–30.
 20. Forster, D.L.; Richards, R.P.; Baker, D.B.; Blue, E.N. EPIC modeling of the effects of farming practice changes on water quality in two lake erie watersheds. *J. Soil Water Conserv.* **2000**, 55 (1), 85–90.
 21. Meyer, L.D. Evolution of the universal soil loss equation. *J. Soil Water Conserv.* **1984**, 39, 99–104.
 22. Karlen, D.L. Opportunities and challenges associated with watershed-scale assessments of soil and water quality. *J. Soil Water Conserv.* **1999**, 54 (4), 626–627.
 23. Bouzaher, A.; Lakshminarayan, P.G.; Cabe, R.; Carriquiry, A.; Gassman, P.W.; Shogren, J.F. Metamodels and nonpoint pollution policy in agriculture. *Water Resour. Res.* **1993**, 29 (6), 1579–1587.
 24. Wu, J.J.; Babcock, B.A. Metamodeling potential nitrate water pollution in the central United States. *J. Environ. Qual.* **1999**, 28 (6), 1916–1928.
 25. Lovejoy, S.B.; Lee, J.G.; Randhir, T.O.; Engel, B.A. Research needs for water quality management in the 21st century: A spatial decision support system. *J. Soil Water Conserv.* **1997**, 52 (1), 18–22.
 26. Groffman, P.M. Nitrogen in the environment. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; C-190–C-200.
 27. Jaenicke, E.C.; Lengnick, L.L. A soil-quality index and its relationship to efficiency and productivity growth measures: Two decompositions. *Am. J. Agr. Econ.* **1999**, 81 (4), 881–893.
 28. Wagenet, R.J.; Hutson, J.L. Soil quality and its dependence on dynamic physical processes. *J. Environ. Qual.* **1997**, 26 (1), 41–48.
 29. Smith, V.; Desvousges, W. *Measuring Water Quality Benefits*; Kluwer-Nijhoff: Boston, 1986.
 30. Fox, G.; Umali, G.; Dickinson, T. An economic analysis of targeting soil conservation measures with respect to off-site water quality. *Can. J. Agr. Econ.* **1995**, 43 (1), 105–108.
 31. Bockstael, N.E.; Freeman, A.M.; Kopp, R.J.; Portney, P.R.; Smith, V.K. On measuring economic values for nature. *Environ. Sci. Technol.* **2000**, 34 (8), 1384–1389.
 32. Andrews, S.S.; Lohr, L.; Cabrera, M.L. A bioeconomic decision model comparing composted and fresh litter for winter squash. *Agr. Syst.* **1999**, 61 (3), 165–178.
 33. Somlyódy, L. Water-quality management—can we improve integration to face future problems? *Water Sci. Technol.* **1995**, 31 (8), 249–259.

Radiometric Survey

Barry G. Rawlins

British Geological Survey, Nottingham, U.K.

Raphael A. Viscarra Rossel

Bruce E. Butler Laboratory, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australian Capital Territory, Australia

Abstract

Measurement of naturally emitted gamma radiation from soil at the land surface can be very effective to aid mapping of a range of primary soil properties, the creation of soil maps, and radon gas hazard assessment. It is unique among remote, soil sensing systems in that it can provide an estimate of bulk soil properties. Measurements are typically made a short distance above the surface of the earth (proximal sensing) or from an airplane flying at between 60 and 120 m altitude. Radiometric survey data are one of the secondary covariates increasingly used in digital soil mapping.

INTRODUCTION

A central objective of digital soil mapping (DSM) is the development and application of cost-effective methods for the creation of continuous soil property maps at suitable scales to aid soil management and policy-related decision making. The approach to DSM described by McBratney et al.^[1] identifies the need for high-resolution auxiliary, predictor variables for the estimation of soil properties. One of the most promising auxiliary variables is airborne (landscape scale) or proximal (field scale) measurements of natural gamma radiation emitted from the soil from radiometric survey. Research has shown that measurements of gamma radiation can be very effective to aid mapping of a range of primary soil properties, the creation of soil maps, and radon hazard assessment. It is unique among soil sensing systems in that it provides a measure of bulk soil properties—not just its surficial properties—with a depth of penetration between 0 and 50 cm of the solum. It has been used in precision agriculture and for understanding landscape-scale processes such as erosion.

THEORY

The application of radiometric survey requires consideration of the measuring system (the detector) and a range of environmental factors, such as soil moisture, rainfall, vegetation, and other sources of radiation. Gamma rays arise from radioactive decay and can penetrate around 30 cm of earth or rock and hundreds of meters of air. Each photon has an energy that is characteristic of its source isotope; the energy range from geological sources occurs in the range 0.2–3 MeV, which concerns this study here.

Three naturally occurring elements produce gamma radiation of sufficient energy for detection (listed here with their average natural abundance in the solum): potassium (K; 2%), thorium (Th; 8.5 mg kg⁻¹), and uranium (U; 2.7 mg kg⁻¹). The proportion of the radioactive isotope of K (⁴⁰K) is fixed, and so the gamma-ray flux can be used to estimate the total concentration of K present in the soil. In the case of U and Th, their concentrations are estimated from the gamma-ray emissions from their daughter products, which are bismuth-214 and thallium-208.^[2]

BACKGROUND RADIATION AND INTERACTIONS WITH MATTER

In addition to radiation from the ground, there are four other sources of “background” radiation flux measured in airborne surveys within this energy range as follows: atmospheric radon, cosmic background, aircraft background, and fallout from man-made atomic explosions and nuclear accidents (¹³⁷Cs). These background signals must be accounted for in the estimation of soil equivalent concentrations of K, Th, and U. Gamma rays also interact with matter, which decreases the intensity of some or all of sample signal before it is detected. The most significant of these processes is Compton scattering in which a photon loses part of its energy to an electron. This can occur in the source of gamma radiation, between the source and detector, and in the detector itself.

Radiation from the ground is attenuated by material between the source and the detector as well as within the source itself. The thicknesses of three different materials that reduce K gamma-ray energies by half as they pass through it are as follows: air, 102 m; water, 11.8 cm; and

concrete, 5.3 cm. Hence, both survey altitude (typically between 60 and 100 m for airborne surveys and less than 2 m for proximal sensing) and variations in soil moisture content can have a significant impact on gamma signals. This attenuation effect and the removal of the coarser (>2 mm) soil fraction in laboratory-based analyses of K, Th, and U may account for the larger measured concentrations of these elements when compared to estimates from radiometric survey.

AIRBORNE SURVEY

Sodium iodide scintillation crystals, which detect gamma rays, are laid in a fixed wing aircraft (Fig. 1) or helicopter to present a cross-sectional area to photons approaching from below. Flying height and aircraft speed determine the support or footprint over which measurements are made from the ground; the quality of gamma-ray spectrometry improves dramatically as elevation decreases because the reduction of gamma-ray intensity with height is approximately exponential. At 56 m altitude, 75% of the measured radiation is from a width of about 150 m, extending to around 220 m along the flight line.^[3] Typical radiometric surveys measure 256 channels monitoring the following four broad spectral windows: K, Th, U, and total counts (Fig. 2). The window count rates need to be corrected for counts resulting from the other elements and for the height of the detector above the ground, and the final data are expressed as equivalent concentrations.

PROXIMAL SURVEY

Proximal gamma-ray spectrometers can be used for surveying or down a borehole to acquire information on lithology

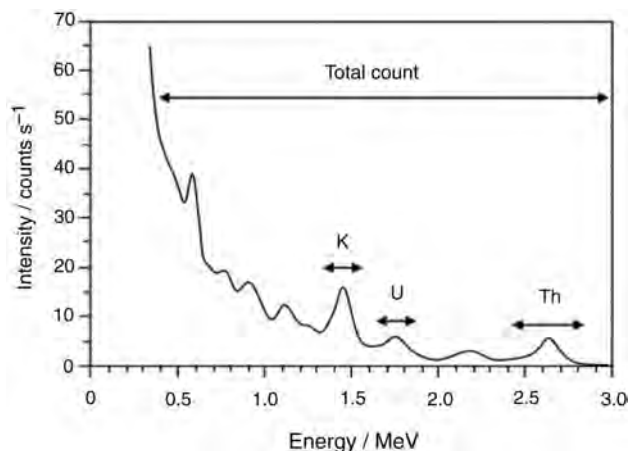


Fig. 2 Typical gamma-ray spectrum.

and geology. The systems can be mounted on vehicles (see Fig. 3) to attain a much greater coverage. Thallium-activated sodium iodide [NaI(Tl)] crystals are mainly used as detectors in these systems; however, thallium-activated caesium iodide [CsI(Tl)] is also available. NaI(Tl) detectors are hygroscopic; they age and are somewhat fragile, and the response of the photomultiplier tube depends on temperature. CsI(Tl) is not hygroscopic and is more robust, but it is more expensive. Proximal gamma-ray spectrometers use smaller crystal packs than airborne sensors, typically between 3 L and 8 L, and typically measure either 256 or 512 channels that comprise an energy spectrum ranging from 0 to 3 MeV (Fig. 2). As with airborne system, GPS navigation is used. The conventional approach to the acquisition and processing of proximal gamma-ray data is to monitor four broad spectral windows or regions of interest (ROIs) corresponding to K (ROI_K), U (ROI_U), Th (ROI_{Th}), and the total count (ROI_{TC}). ROI_K monitors the 1.460 MeV

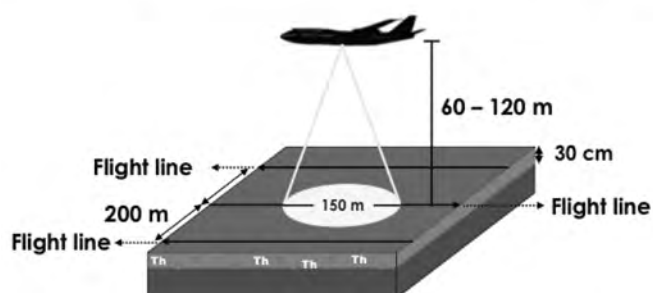


Fig. 1 Fixed wing aircraft used for airborne survey.



Fig. 3 Vehicle-mounted proximal sensor system.

gamma rays emitted by ^{40}K , while ROI_{U} and ROI_{Th} monitor gamma-ray emissions of the decay products (in parentheses) from the U (^{214}Bi) and Th (^{208}Tl) decay series.^[2] Only these three naturally occurring elements produce gamma rays of sufficient energy and intensity to be measured by ground and airborne surveys. Their isotopes have sufficiently long half-lives for them to remain relatively abundant in the earth, including its surficial deposits—including soil—from which gamma rays are detected. In addition to the gamma radiation from K, Th, and U and their decay products, isotopes emitting gamma radiation are created by cosmic ray interactions with the earth and its atmosphere; this is referred to as background radiation, which has to be removed from survey observations. For ROI_{U} , the energy of the photopeak is centered on 1.765 MeV, and for ROI_{Th} , it is centered on 2.614 MeV. The ROI_{TC} gives a measure of total radioactivity, and it is the integrated count over the 0.4–2.81 MeV range (Fig. 2). Viscarra Rossel et al.^[4] showed that the entire gamma-ray spectrum (256 energy bands) can be used for the prediction of various soil properties using multivariate calibration techniques.

APPLICATIONS

One of the first soil-related demonstrations of airborne radiometric survey was the airborne estimation and ground-based validation of soil moisture along flight lines based on the greater attenuation of the gamma signal in wetter soil.^[5] The negative correlation between wetter soils with the attenuation of gamma-ray K and larger quantities of soil organic carbon (SOC) was used to explain the utility of airborne gamma-ray K as an effective secondary covariate for mapping SOC across Northern Ireland.^[6]

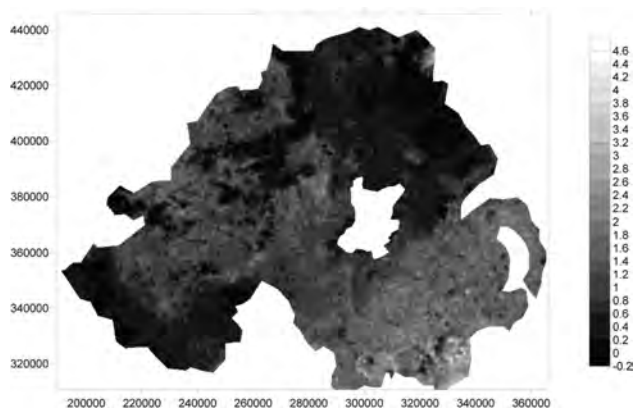


Fig. 4 The distribution of radiometric K across Northern Ireland. Coordinates are meters of the Irish National Grid.

The value of airborne radiometric survey for the identification of soil parent materials was highlighted by Cook et al.^[7] for highly weathered soil landscapes in Southwest Australia. Rawlins et al.^[8] demonstrated that radiometric K and Th are particularly effective for delineating soil parent material types in the young glaciated terrains of eastern England. Fig. 4 shows the spatial distribution of K from an airborne radiometric survey of Northern Ireland; the parent material map accounts for 68% of the variation in K showing that the bedrock and superficial deposits exert a strong influence on the distribution.

The identification of erosional and depositional features has been aided by the use of radiometric survey in Southeast Australia^[9] and aeolian dusts.^[10] Airborne surveys have also been shown to be effective for assessing terrestrial gamma dose rates in California^[11] and for radon potential mapping in Nova Scotia,^[12] which can be used to assess human health impacts.

Proximal gamma radiometrics has been used to establish relationships between ROI_{K} counts and apparent topsoil dust accumulations, which were identified as containing appreciable feldspar and illite.^[10] Rampant and Abuzar^[13] used ROI counts to identify yield management zones for precision agriculture. Many studies also report considerably better relationships between the ROI data and soil properties than do airborne surveys. Wong and Harper^[14] identified excellent linear relationships between ROI_{K} and available K. Pracilio et al.^[15] showed that there was a significant relationship between $\log \text{ROI}_{\text{Th}}$ and gravel content ($R^2 = 0.63$) at a site in Western Australia. They also identified significant relationships between the $\log \text{ROI}_{\text{TC}}$ and clay content ($R^2 = 0.63$). Taylor et al.^[16] identified similar linear relationships between ROI_{TC} and clay content ($R^2 = 0.71$). These authors also found relationships between ROI_{Th} and ironstone gravel ($R^2 = 0.23$) and between ROI_{K} and total feldspar content ($R^2 = 0.62$). Pracilio et al.^[17] attempted to improve the linear prediction models created by Pracilio et al.^[18] by using regression trees, which combined the $\text{ROI}_{\text{K,U,Th}}$ counts with

topographic data to predict soil properties. The technique improved R^2 values for the predictions of clay content but not for available K. Viscarra Rossel et al.^[4] surveyed two geographically and physiographically different fields in New South Wales, Australia, and collected hyperspectral gamma-ray data consisting of 256 energy bands at more than 20,000 sites in each field. They used bootstrap aggregation with bagging-partial least squares regression (bPLSR; see Viscarra Rossel^[19]) to calibrate the gamma-ray spectra of each field for the predictions of clay, coarse sand, and Fe contents in the 0–15 cm soil layer and pH and coarse sand contents in the 15–50 cm soil layer. The authors showed that proximally sensed gamma-ray spectrometry combined with bPLSR is a useful tool for predicting soil properties in different soil landscapes.

REFERENCES

- McBratney, A.B.; Mendonca Santos, M.L.; Minasny, B. On digital soil mapping. *Geoderma* **2003**, *117*, 3–52.
- Minty, B.R.S. Fundamentals of airborne gamma-ray spectrometry. *AGSO J. Aust. Geol. Geophys.* **1997**, *17*, 39–50.
- Pitkin, J.A.; Duval, J.S. Design parameters for aerial gamma-ray surveys. *Geophysics* **1980**, *45*, 1427–1439.
- Viscarra Rossel, R.A.; Taylor, H.J.; McBratney, A.B. Multivariate calibration of hyperspectral γ -ray energy spectra for proximal soil sensing. *Eur. J. Soil Sci.* **2007**, *58*, 343–353.
- Carroll, T.R. Airborne soil moisture measurements using natural terrestrial gamma radiation. *Soil Sci.* **1981**, *132*, 358–366.
- Rawlins, B.G.; Marchant, B.P.; Smyth, D.; Scheib, C.; Lark, R.M.; Jordan, C. Airborne radiometric survey data and a DTM as covariates for regional scale mapping of soil organic carbon across Northern Ireland. *Eur. J. Soil Sci.* **2009**, *60*, 44–54.
- Cook, S.E.; Corner, R.J.; Groves, P.R.; Grealish, G.J. Use of airborne gamma radiometric data for soil mapping. *Aust. J. Soil Res.* **1996**, *34*, 183–194.
- Rawlins, B.G.; Lark, R.M.; Webster, R. Understanding airborne radiometric survey signals across part of eastern England. *Earth Surf. Proc. Land.* **2007**, *32*, 1503–1515.
- Pickup, G.; Marks, A. Identifying large-scale erosion and deposition processes from airborne gamma radiometrics and digital elevation models in a weathered landscape. *Earth Surf. Proc. Land.* **2000**, *25*, 535–557.
- Cattle, S.R.; Meakin, S.N.; Ruszkowski, P.; Cameron, R.G. Using radiometric data to identify aeolian dust additions to topsoil of the Hillston district, western NSW. *Aust. J. Soil Res.* **2003**, *41*, 1439–1456.
- Wollenberg, H.A.; Revzan, K.L.; Smith, A.R. Application of airborne gamma spectrometric survey data to estimating terrestrial gamma-ray dose-rates—An example in California. *Health Phys.* **1994**, *66*, 10–16.
- Jackson, S.A. Estimating radon potential from an aerial radiometric survey. *Health Phys.* **1992**, *62*, 450–452.
- Rampant, P.; Abuzar, M. Geophysical tools and digital elevation models: Tools for understanding crop yield and soil variability. In *SuperSoil*, Third Australian/New Zealand Soils Conference; Singh, B., Ed.; University of Sydney: Sydney, 2004.
- Wong, M.T.F.; Harper, R.J. Use of on-ground gamma-ray spectrometry to measure plant-available potassium and other topsoil attributes. *Aust. J. Soil Res.* **1999**, *37*, 267–277.
- Pracilio, G.; Smettem, K.R.J.; Harper, R. New soil survey technologies to map landscape properties relevant to perennial plant performance. In *Salinity Solutions, Working with Science and Society*, Proceedings of the Salinity Solutions Conference, Bendigo, Victoria, Australia, Aug 2–5; 2004; Ridley, A., Feikama, P., Bennet, S., Rogers, M.-J., Wilkinson, R., Hirth, J., Eds.; [CD-ROM].
- Taylor, M.J.; Smettem, K.; Pracilio, G.; Verboom, W. Relationships between soil properties and high-resolution radiometrics, central eastern Wheatbelt, Western Australia. *Explor. Geophys.* **2002**, *33*, 95–102.
- Pracilio, G.; Adams, M.L.; Smettem, K.R.J. Gamma ray spectrometry used to spatially predict soil properties important to plant growth. In Proceedings of the Seventh International Conference on Precision Agriculture and Other Resources Management, July 25–28, 2004; Hyatt Regency: Minneapolis.
- Pracilio, G.; Adams, M.L.; Smettem, K.R.J. Use of airborne gamma radiometric data for soil property and crop biomass assessment, northern dryland agricultural region, Western Australia. In *Precision Agriculture '03*, Proceedings of the 4th European Conference; Stafford, J.V., Werner, A., Eds.; Wageningen Academic Publishers: Netherlands, 2003; 551–557.
- Viscarra Rossel, R.A. Robust modelling of soil diffuse reflectance spectra by bagging-partial least squares regression. *J. Near Infrared Spec.* **2007**, *15*, 39–47.

Radionuclides

Philip M. Jardine

Oak Ridge National Laboratory, Oak Ridge, Tennessee, U.S.A.

Abstract

The fate and transport of radionuclides through soil is controlled by coupled hydrologic, geochemical, and microbial processes. A multitude of complex soil processes are tightly linked that can both accelerate and impede the subsurface mobility of radioactive contaminants. Often the extent and magnitude of subsurface biogeochemical reactions is controlled by the spatial and temporal variability in soil hydrologic processes.

INTRODUCTION

Soil, the thin veneer of matter covering the earth's surface and supporting a web of living diversity, is often abused through anthropogenic inputs of toxic waste. The disposal of radioactive waste generated at U.S. Department of Energy (DOE) facilities within the Weapons Complex has historically involved shallow land burial in unsaturated soils and sediments. Disposal methods from the 1940s to the 1980s ranged from unconfined pits and trenches to single- and double-shell buried steel tanks. Most of the below-ground burial strategies were deemed to be temporary (i.e., an average life span of several decades) until suitable technologies were developed to deal with the legacy waste issues. Technologies for retrieving and treating the below-ground radionuclide waste inventories have been slow to evolve and are often cost prohibitive or marginally effective. The scope of DOE's disposal problem is massive, with landfills estimated to contain more than 3 million cubic meters of radioactive and hazardous buried waste; a significant proportion of which migrated into surrounding soils and groundwater. It is estimated that the migration of these waste plumes contaminated over 600 billion gallons of water and 50 million cubic meters of soil.

FATE AND TRANSPORT PROCESSES

Hydrologic Processes

Soil is a complex continuum of pore regions ranging from large macropores at the millimeter scale to small micropores at the sub-micrometer scale. It is the physical properties of the media (e.g., structured or layered), coupled with the duration and intensity of precipitation events that dictates the avenues of water and radionuclide movement through the subsurface. In humid environments where structured media is commonplace, transient storm events invariably result in the preferential migration of water.^[1–8]

Highly conductive voids within the media (e.g., fractures and macropores) carry water around low-permeability, high-porosity matrix blocks or aggregates resulting in water bypass of the latter. In these humid regimes, recharge rates are very high with more than 50% of the infiltrating precipitation resulting in groundwater and surface water recharge. This condition promotes the formation of massive contaminant plumes in the soil since storm flow and groundwater interception with waste trenches is frequent and long-lasting. Even in semiarid environments, where recharge is typically small, subsurface preferential flow is a key mechanism controlling water and solute mobility.^[9,10] Lithologic discontinuities and sediment layering promote perched water tables and unstable wetting fronts that drive both lateral and vertical subsurface preferential flow.

In both humid and semiarid regimes, water that is preferentially flowing through the soil media often remains in intimate contact with the porous matrix, and physical and hydrologic gradients drive the exchange of mass from one pore regime to another. Mass exchange is time-dependent and is often controlled by diffusion to and from the matrix. Thus, a significant inventory of radionuclide waste can reside within the soil matrix. This waste source is hydrologically linked to preferred flow paths which significantly enhances the extent and longevity of subsurface contaminant plumes. This scenario is commonplace at the Oak Ridge National Laboratory, located in eastern Tennessee, U.S.A., where thousands of underground disposal trenches and ponds have contributed to the spread of radionuclides such as ¹³⁷Cs, ⁶⁰Co, ⁹⁰Sr, and ^{235/238}U across tens of kilometers of landscape. Highly concentrated contaminant plumes move through soil and groundwater at time scales of meters per day (Fig. 1), since the soils are highly structured and conducive to rapid preferential flow. However, the soil matrix, which has a high porosity and low permeability, serves as a source/sink for contaminants.^[5,6,11] The preferential movement of water and radionuclides through the subsurface also significantly impacts geochemical and

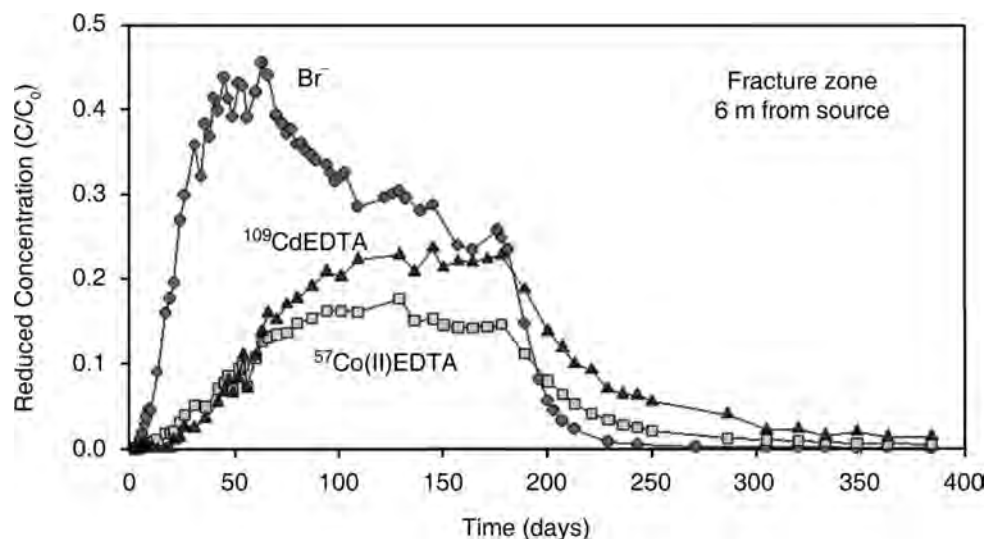


Fig. 1 Field-scale fate and transport of non-reactive Br^- and reactive $^{57}\text{Co(II)EDTA}^{2-}$ and $^{109}\text{CdEDTA}^{2-}$ in fractured subsurface media at the Oak Ridge National Laboratory. Although transport rates are rapid, geochemical reactions significantly impeded the mobility of the chelated radionuclides as are indicated by their delayed breakthrough.

Source: From Jardine, Mehlhorn, et al.^[12]

microbial processes by controlling the extent and rate of various reactions with the solid phase. It imposes kinetic constraints on biogeochemical reactions and limits the surface area of interaction by partially excluding water and mass from the matrix porosity.

Geochemical Processes

Radionuclide fate and transport in soil and sediments is also controlled by interfacial reactions with the soil solid phase. Most soils are a complex mixture of variably charged phyllosilicates, redox reactive iron (Fe)- and manganese (Mn)-oxides, organic matter, and mineral carbonates. Radionuclides interact with these solid phases through coulombic exchange, chemisorption, redox alterations, transformation processes such as polymerization, precipitation/dissolution, and complexation reactions. Both the extent and rate of these processes can be significantly influenced by variations in water content and the degree of pore regime connectivity. To make matters worse, radionuclide waste generated at the United States DOE facilities was often co-disposed with various chelating agents and organic acids. These synthetic organic constituents form highly stable, water-soluble complexes with a wide variety of radionuclides.^[13,14] The presence of the complexing agent significantly alters the geochemical behavior of the disposed contaminants in soils and sediments through increased solubility, accelerated redox reactions, and ionic charge reversal.

The geochemical mechanism controlling the fate and transport of chelated radionuclides has been well characterized in numerous soils and subsurface materials.^[15–23] Typically, Fe(III) and Mn(IV) oxyhydroxides are the dominant subsurface mineral assemblages that catalyze cocontaminant oxidation/reduction and dissociation reactions (Fig. 2). The mineral oxides have repeatedly been shown to catalyze the oxidation of $^{60}\text{Co(II)EDTA}^{2-}$ to $^{60}\text{Co(III)EDTA}^-$,

thereby adversely enhancing the transport and persistence of ^{60}Co in a variety of subsurface environments ranging from aquifer sands to fractured weathered shale saprolites.^[16,18,19,21,23] Further, Fe(III)-oxides have also been shown to effectively dissociate a large number of chelated metal and radionuclide complexes (e.g., $^{60}\text{Co}-$, $^{90}\text{Sr-EDTA}$) through ligand competition.^[16,21–23]

Certain radionuclides such as ^{137}Cs do not form strong bonds with many of the chelating agents and organic acids that were used during decontamination. Nevertheless, these radionuclides still interact aggressively with the soil solid phase. In the case of ^{137}Cs , 2:1 phyllosilicates and micas serve as excellent sorbents since the interlayer spaces of these mineral assemblages strongly attenuate the radionuclide. The migration tendency of ^{137}Cs in soils is often related to colloid mobility of contaminated sediments^[24] or cation competition for surface sites in harsh environments such as those found beneath the Hanford tank farms in western Washington State, U.S.A.^[25]

Microbial Processes

Radionuclides such as ^{60}Co and $^{235/238}\text{U}$ can exist in more than one oxidation state, and their behavior in the environment depends on their oxidation state. For example, U(VI) is soluble and mobile in the environment, whereas U(IV) is much less soluble and relatively immobile. Likewise, the oxidized $^{60}\text{Co(III)EDTA}$ complexes are much more stable and exhibit greater mobility in subsurface environments than the reduced $^{60}\text{Co(II)EDTA}$.^[16,18,19] Subsurface Fe- and Al-oxides can effectively dissociate the Co(II)EDTA complex to Fe(III)EDTA^[21] and Al(III)EDTA,^[17] respectively, and aqueous Co^{2+} is free to participate in sorption or precipitation reactions. Co(III)EDTA, on the other hand, is unaffected by Fe(III)- and Al-oxides. Therefore, the oxidized forms of these radionuclides and metals promote their undesirable enhanced migration through subsurface environments.

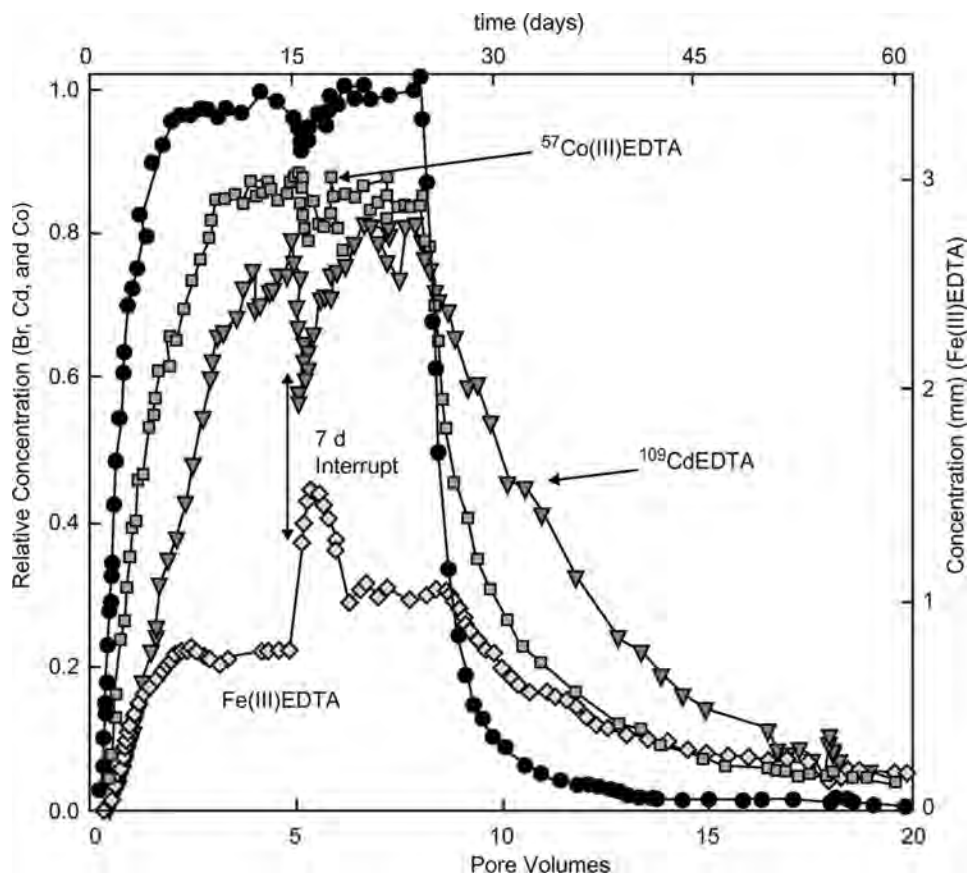


Fig. 2 Fate and transport of nonreactive Br^- and reactive $^{57}\text{Co(II)EDTA}^{2-}$ and $^{109}\text{CdEDTA}^{2-}$ in undisturbed soil columns of fractured weathered shale. Geochemical reactions impede the mobility of the chelated radionuclides. Co(II)EDTA^{2-} is oxidized to Co(III)EDTA^- where Mn-oxides serve as the oxidant. Fe-oxides effectively dissociate CdEDTA^{2-} complexes resulting in the formation of free Cd and Fe(III)EDTA^- . A flow interruption technique was employed to quantify the presence of physical and geochemical nonequilibrium processes.

Source: From Mayes, Jardine, et al.^[23]

Numerous metal-reducing bacteria have been isolated that enzymatically reduce toxic metals and radionuclides to stable end products. Microbial reduction of U(VI) to form the sparingly soluble U(IV) has been shown using chemostat experiments for a number of metal-reducing bacteria.^[26,27] Gorby et al.^[28] have also shown that certain metal-reducing bacteria can link the enzymatic reduction of $^{60}\text{Co(III)EDTA}^-$ to support cell growth. Important advances have been made toward implementing field scale microbially mediated metal reduction strategies in oxygen-deficient environments. Several studies have investigated contaminant reduction in the presence of solid-phase material.^[28–29] Gorby et al.^[28] have shown that the metal-reducing bacterium *Shewanella alga* preferentially reduced Co(III)EDTA^- to Co(II)EDTA^{2-} in the presence of Mn-oxides. Likewise, Wielinga et al.^[30] documented the bioreduction of U(VI) by *S. alga* in the presence of various Fe-oxide mineral phases. These authors noted that the rate of U(VI) bioreduction was unaffected in the presence of goethite and only slightly diminished in the presence of poorly crystalline Fe(III)-oxides, where the latter Fe solid phase effectively competed as a terminal electron acceptor. Studies by Brooks et al.^[29] showed the sustained microbial reduction of Co(III)EDTA^- under dynamic flow conditions. The net reduction of the Co(III)EDTA^- dominated the fate and transport of the contaminant even in the presence of strong mineral oxidants such as Mn- and Fe-oxides

that are known to effectively reoxidize Co(II)EDTA^{2-} back to Co(III)EDTA^- .^[16,19,21] The research findings of Brooks et al.^[29] provide new and important information on how to successfully implement a bioreduction strategy at the field scale. Their use of a dynamic flow system with sustained bacterial growth conditions in geochemically reactive media is consistent with contaminant migration scenarios *in situ*.

The studies of Brooks et al.,^[29] however, used uniformly packed media that contained little structure. Undisturbed subsurface soils and geologic material consist of a complex continuum of pore regions ranging from large macropores and fractures at the millimeter scale to small micropores at the sub-micrometer scale. Structured media, common to most subsurface environments throughout the world, accentuates this physical condition that often controls the geochemical and microbial processes affecting solute transport. Redox sensitive radionuclides such as U(VI) , Co(III)EDTA , and Tc(VI) reside within nearly all of the pore structure of the subsurface media, with the greatest concentration of contaminants held within micropores.^[2,3] Bacteria that are capable of reducing these contaminants are too big to reach a large fraction of the micropore regime and are largely restricted to macro- and mesopore domains.^[31,32] Fortunately, the pore structure of the media is hydrologically interconnected, and contaminants move from one pore class to another via hydraulic and concentration

gradients.^[4,6,7] This process is slow, however, and is often the rate-limiting factor governing the success of contaminant bioremediation. Thus, faster-flowing fracture-dominated regimes will most likely be physically more appealing for sustained bioreduction as long as a suitable electron donor can be supplied. In contrast, bioreduction processes in slower-flowing matrix regimes will most likely be limited by rate-dependent mass transfer of contaminants from smaller pores into larger pores.

Certain bacteria are also capable of degrading chelates and thus potentially immobilizing radionuclides *in situ*. The biodegradation of the commonly used aminopolycarboxylate chelates nitrilotriacetic acid (NTA), ethylenediaminetetraacetic acid (EDTA), and diethylenetriaminepentaacetic acid have been studied in soil and sediment systems for many years.^[33,34,35] Research has shown that NTA has the greatest potential for biodegradation in subsurface systems compared with the other aminopolycarboxylates.^[35,36]

Bolton et al.^[37] and Bolton and Girvin^[38] have shown that the bacterial strain *Chelatobacter heintzii* (ATCC 29600) is capable of degrading NTA in the presence of many different toxic metals and radionuclides. Likewise, Payne et al.^[39] and Liu et al.^[40] have deciphered the mechanisms by which certain bacteria degrade radionuclide-EDTA complexes. These studies lend promise to the potential for using bacteria to biodegrade chelates and enhance the geochemical immobilization of radionuclides *in situ*.

CONCLUSION

Radionuclide fate and transport in soils is controlled by coupled time-dependent hydrologic, geochemical, and microbial processes. Hydrologic processes such as preferential flow and matrix diffusion can serve to both accelerate and impede radionuclide migration, respectively. Preferential flow results in hydraulic, physical, and geochemical non-equilibrium conditions since differences in fluid velocities and solute concentrations in different-sized pores create hydraulic and concentration gradients that drive time-dependent inter-region advective and diffusive mass transfer. Thus, in soil systems with a large matrix porosity or a significant quantity of disconnected immobile water, radionuclide migration rates can be greatly retarded due to the slow transfer of mass to actively flowing preferential flow paths. Nevertheless, the prevalence of preferential flow can greatly accelerate the transport of mass in soil systems. Geochemical processes such as sorption, redox alterations, and dissociation reactions can also serve to both accelerate and impede radionuclide migration. Sorption and radionuclide-chelate dissociation reactions almost always result in retarded radionuclide migration rates, whereas oxidation reactions often result in more soluble, and thus more mobile, radionuclide species. Microbial processes can also potentially influence

the fate and transport of radionuclides in soil. Metal-reducing bacteria and chelate degraders can alter the geochemical behavior of redox-sensitive radionuclides which facilitates their immobilization via solid phase sorption and precipitation reactions.

Enhanced knowledge of the coupled hydrologic, geochemical, and microbial processes controlling radionuclide migration in soils will improve our conceptual understanding and predictive capability of the risks associated with spread of radioactive material in the subsurface environment. Too often risk assessment models treat soil and bedrock as inert media or assume that the media is in equilibrium with migrating contaminants. Failure to consider the time-dependent coupled processes that control radionuclide migration will greatly overpredict the off-site contribution of contaminants from the primary waste source and thus provide an inaccurate assessment of pending risk. By recognizing the importance of soil processes on radionuclide migration, we can improve our decision-making strategies regarding the selection of effective remedial actions and improve our interpretation of monitoring results after remediation is complete.

REFERENCES

1. Shuford, J.W.; Fitton, D.D.; Baker, D.E. Nitrate-nitrogen and chloride movement through undisturbed field soil. *J. Environ. Qual.* **1977**, *6*, 255–259.
2. Jardine, P.M.; Wilson, G.V.; Luxmoore, R.J. Unsaturated solute transport through a forest soil during rain storm events. *Geoderma* **1990**, *46*, 103–118.
3. Jardine, P.M.; Wilson, G.V.; McCarthy, J.F.; Luxmoore, R.J.; Taylor, D.L.; Zelazny, L.W. Hydrogeochemical processes controlling the transport of dissolved organic carbon through a forested hillslope. *J. Contam. Hydrol.* **1990**, *6*, 3–19.
4. Jardine, P.M.; O'Brien, R.; Wilson, G.V.; Gwo, J.P. Experimental techniques for confirming and quantifying physical nonequilibrium processes in soils. In *Physical Nonequilibrium in Soils: Modeling and Application*; Selim, H.M., Ma, L., Eds.; Ann Arbor Press: Chelsea, 1998; 243–271.
5. Jardine, P.M.; Wilson, G.V.; Luxmoore, R.J.; Gwo, J.P. Conceptual model of vadose-zone transport in fractured weathered shales. In *Conceptual Models of Flow and Transport in the Fractured Vadose Zone*; U.S. National Committee for Rock Mechanics, National Research Council; National Academy Press: Washington, 2001; 87–114.
6. Wilson, G.V.; Jardine, P.M.; O'Dell, J.D.; Collineau, M. Field-scale transport from a buried line source in variable saturated soil. *J. Hydrol.* **1993**, *145*, 83–109.
7. Wilson, G.V.; Gwo, J.P.; Jardine, P.M.; Luxmoore, R.J. Hydraulic and physical nonequilibrium effects on multi-region flow and transport. In *Physical Nonequilibrium in Soils: Modeling and Application*; Selim, H.M., Ma, L., Eds.; Ann Arbor Press: Chelsea, 1998; 37–61.
8. Hornberger, G.M.; Germann, P.F.; Beven, K.J. Through flow and solute transport in an isolated sloping soil block in a forested catchment. *J. Hydrol.* **1991**, *124*, 81–99.

9. Porro, I.; Wierenga, P.J.; Hills, R.G. Solute transport through large uniform and layered soil columns. *Water Resour. Res.* **1993**, *29*, 1321–1330.
10. Ritsema, C.J.; Dekker, L.W.; Nieber, J.L.; Steenhuis, T.S. Modeling and field evidence of finger formation and finger recurrence in a water repellent sandy soil. *Water Resour. Res.* **1998**, *34*, 555–567.
11. Jardine, P.M.; Sanford, W.E.; Gwo, J.P.; Reedy, O.C.; Hicks, D.S.; Riggs, R.J.; Bailey, W.B. Quantifying diffusive mass transfer in fractured shale bedrock. *Water Resour. Res.* **1999**, *35*, 2015–2030.
12. Jardine, P.M.; Mehlhorn, T.L.; Larsen, I.L.; Bailey, W.B.; Brooks, S.C.; Roh, Y.; Gwo, J.P. Influence of hydrological and geochemical processes on the transport of chelated-metals and chromate in fractured shale bedrock. *J. Contamin. Hydrol.* **2002**, *55*, 137–159.
13. Riley, R.G.; Zachara, J.M. *Chemical Contaminants on DOE Lands and Selection of Contaminant Mixtures for Subsurface Science Research*; DOE/ER-0547T; U.S. Government Printing Office: Washington, 1992.
14. Toste, A.P.; Osborn, B.C.; Polach, K.J.; Lechner-Fish, T.J. Organic analyses of an actual and simulated mixed waste: Hanford's organic complexant site revisited. *J. Radioanal. Nucl. Chem.* **1995**, *194*, 25–34.
15. Swanson, J.L. *Effect of Organic Complexants on the Mobility of Low-level Waste Radionuclides in Soils*; Status Report PNL-3927, UC-70, 1981.
16. Jardine, P.M.; Jacobs, G.K.; O'Dell, J.D. Unsaturated transport processes in undisturbed heterogeneous porous media: II. Co-contaminants. *Soil Sci. Soc. Am. J.* **1993**, *57*, 954–962.
17. Girvin, D.C.; Gassman, P.L.; Bolton, H. Adsorption of aqueous cobalt ethylenediaminetetraacetate by δ -Al₂O₃. *Soil Sci. Soc. Am. J.* **1993**, *57*, 47–57.
18. Zachara, J.M.; Gassman, P.L.; Smith, S.C.; Taylor, D. Oxidation and adsorption of Co(II)EDTA²⁻ complexes in subsurface materials with iron and manganese oxide grain coatings. *Geochim. Cosmochim. Acta.* **1995**, *59*, 4449–4463.
19. Brooks, S.C.; Taylor, D.L.; Jardine, P.M. Reactive transport of EDTA-complexed cobalt in the presence of ferrihydrite. *Geochim. Cosmochim. Acta.* **1996**, *60*, 1899–1908.
20. Read, D.; Ross, D.; Sims, R.J. The migration of uranium through clastic sandstone: The role of low molecular weight organics in enhancing radionuclide transport. *J. Contam. Hydrol.* **1998**, *35*, 235–248.
21. Szecsody, J.E.; Zachara, J.M.; Chilakapati, A.; Jardine, P.M.; Ferency, A.S. Importance of flow and particle-scale heterogeneity on Co(II/III)EDTA reactive transport. *J. Hydrol.* **1998**, *209*, 112–136.
22. Davis, J.A.; Kent, D.B.; Coston, J.A.; Hess, K.M.; Joye, J.L. Multispecies reactive tracer test in an aquifer with spatially variable chemical conditions. *Water Resour. Res.* **2000**, *36* (1), 119–134.
23. Mayes, M.A.; Jardine, P.M.; Larsen, I.L.; Brooks, S.C.; Fendorf, S.E. Multispecies transport of metal-EDTA complexes and chromate through undisturbed columns of weathered, fractured saprolite. *J. Contam. Hydrol.* **2000**, *45*, 243–265.
24. Solomon, D.K.; Marsh, J.D.; Larsen, I.L.; Wickliff, D.S.; Clapp, R.B. *Transport of Contaminants During Storms in the White Oak Creek and Melton Branch Watersheds*; ORNL/TM-11360; Oak Ridge National Laboratory: Oak Ridge, 1991; 127.
25. Serne, R.J.; Burke, D.S. *Chemical Information on Tank Supernatants, Cs Adsorption from Tank Liquids onto Hanford sediments, and Field Observations of Cs Migration from Past Tank Leaks*; Report No. PNNL-11495; Pacific Northwest National Laboratory: Richland, 1997.
26. Gorby, Y.A.; Lovley, D.R. Enzymatic uranium precipitation. *Environ. Sci. Technol.* **1992**, *26*, 205–207.
27. Francis, A.J.; Dodge, C.J.; Lu, F.; Halada, G.P.; Clayton, C.R. XPS and XANES studies of uranium reduction by *Clostridium* Sp. *Environ. Sci. Technol.* **1994**, *28*, 636–639.
28. Gorby, Y.A.; Caccavo, F.; Bolton, H. Microbial reduction of Co(III)EDTA⁻ in the presence and absence of manganese (IV) dioxide. *Environ. Sci. Technol.* **1998**, *32*, 244–250.
29. Brooks, S.C.; Carroll, S.L.; Jardine, P.M. Sustained bacterial reduction of Co(III)EDTA⁻ in the presence of competing geochemical oxidation during dynamic flow. *Environ. Sci. Technol.* **1999**, *33*, 3002–3011.
30. Wielinga, B.; Bostick, B.; Hansel, C.; Rosenzweig, R.F.; Fendorf, S. Inhibition of bacterially promoted uranium reduction: Ferric (hydr)oxides as competitive electron acceptors. *Environ. Sci. Technol.* **2000**, *34*, 2190–2195.
31. Smith, M.S.; Thomas, G.W.; White, R.E.; Ritonga, D. Transport of *Escherichia Coli* through intact and disturbed soil columns. *J. Environ. Qual.* **1985**, *14*, 87–91.
32. McKay, L.D.; Cherry, J.A.; Bales, R.C.; Yahya, M.T.; Gerba, C.P. A field example of bacteriophage as tracers of fracture flow. *Environ. Sci. Technol.* **1993**, *27*, 1075–1079.
33. Tiedje, J.M. Microbial degradation of ethylenediaminetetraacetic acid in soils and sediments. *Appl. Environ. Microbiol.* **1975**, *30*, 327–329.
34. Tiedje, J.M.; Mason, B.B. Biodegradation of nitrilotriacetic acid (NTA) in soils. *Soil Sci. Soc. Am. Proc.* **1974**, *38*, 278–283.
35. Bolton, H., Jr.; Li, S.W., Jr.; Workman, D.J.; Girvin, D.C. Biodegradation of synthetic chelates in subsurface sediments from the southeast coastal plain. *J. Environ. Qual.* **1993**, *22*, 125–132.
36. Means, J.L.; Kucak, T.; Crerar, D.A. Relative degradation rates of NTA, EDTA, and DTPA and environmental implications. *Environ. Pollut. Ser. B.* **1980**, *1*, 45–60.
37. Bolton, H.; Girvin, D.C.; Plymale, A.E.; Harvey, S.D.; Workman, D.J. Degradation of metal-nitrilotriacetate complexes by *Chelatobacter heintzii*. *Environ. Sci. Technol.* **1996**, *30*, 931–938.
38. Bolton, H., Jr.; Girvin, D.C., Jr. Effect of adsorption on the biodegradation of nitrilotriacetate by *Chelatobacter heintzii*. *Environ. Sci. Technol.* **1996**, *30*, 2057–2065.
39. Payne, J.W.; Bolton, H.; Campbell, J.A.; Xun, Y.L. Purification and characterization of EDTA monooxygenase from the EDTA-degrading bacterium BNC1. *J. Bacteriology* **1998**, *180*, 3823–3827.
40. Liu, Y.; Louie, T.M.; Payne, J.; Bohuslavsek, J.; Bolton, H.; Xun, L.Y. Identification, purification, and characterization of iminodiacetate oxidase from the EDTA-degrading bacterium BNC1. *Applied Environ. Microbiol.* **2001**, *67*, 696–701.

Raindrop Impact and Splash Erosion

Hossein Ghadiri

*School of Australian Environmental Studies, Griffith University, Nathan Campus,
Nathan, Queensland, Australia*

Abstract

The highest amount of splash loss occurs when the soil surface is at or just above saturation. Any degree of suction to soil water or depth of water on the surface reduces the splash loss. Surface slope only affects the distribution of splash particles with a greater proportion moving downslope as the slope angle increases. Release velocity and trajectory of splash droplets are dependent on the angle and velocity of splash corona. Crater size and shape are dependent upon both drop and surface characteristics. Craters formed in water or saturated soil pastes are hemispherical but shallow with a raised center in compacted soil and sand. Crater volume, which is a more accurate measure of total soil loss by the impacting drop, can be predicted using impact impulse and surface shear strength.

INTRODUCTION

The first comprehensive study of splash erosion and the mechanical action of falling raindrops on soils was carried out in 1940s.^[1–3] The recognition of the importance of raindrop splash in water erosion was a breakthrough which opened a new chapter in soil erosion research and resulted in the development of erosion prediction models such as Universal Soil Loss Equation (USLE),^[4] European Soil Erosion Model (EUROSEM),^[5] and Revised USLE (RUSLE),^[6] in which rain erosivity is expressed in terms of rainfall energy. However, after this big leap forward, research into the mechanics of raindrop impact and soil splash stagnated for nearly 40 years. Ellison's^[2] assumption of rainfall energy being the driving force of water erosion remained unchallenged and adopted by almost all the researchers and modelers in the field. The main reason for the lack of further progress in the study of raindrop impact during this period was the multitude and complexity of the processes that take place in a very short period of time following raindrop impact and the elaborate facilities such studies required.^[7] It was only after the application in erosion studies of high-speed photography technique, which was capable of slowing down the fast processes of impact and splash by several thousand times, that some of these complex processes began to be observed and explained.^[8]

Photographic studies of raindrop impact erosion of the 1970s and 1980s identified three main processes that take place concurrently or in quick succession following raindrop impact.^[8–10] They have been named as “impact,” “splash,” and “cratering” processes. Impact and splash processes have since been studied by many researchers,^[11–13] but “cratering” has not received much attention in soil erosion studies. On the other hand, cratering is the only aspect

of water drop erosion which researchers in the aerospace and turbine industries are concerned with.^[8,14,15] These three processes are discussed individually.

RAINDROP IMPACT

The studies of water drop impact on solid and liquid surfaces started as early as 1876.^[16] Simulating the conditions inside turbines revealed that water drops at very high speeds can cut through even the hardest metals, and the damage increases rapidly by some power of impact velocity.^[14–16] As the impact damage can be caused both by direct impact pressure and by the lateral outflow of drop water, the impact process is subdivided into “impact pressure” and “lateral flow stress.”

Impact Pressure

Early studies of the impact of liquid drops on solid surfaces have shown that high pressure develops under the impacting drop as a result of a process known as “water hammer effect.”^[15,17,18] The water hammer pressure “P” develops when the front end of a moving water drop is stopped instantaneously and lateral flow is prevented; thus, compressive pressure builds up. The general form of the water hammer equation is as follows.

$$P = \rho VC \quad (1)$$

where ρ is the liquid density, C is the velocity of sound in water, and V is the impact velocity.

The duration of the compressible phase of impact is extremely short, but its existence has been reported by several researchers.^[8,10,11,19] The measurement of impact pressure and its distribution over the contact area for free

falling water drops has been attempted using miniature pressure transducer and nylon mesh penetration techniques.^[8,10,11,19] A concentration of transient compressive stress around the periphery of the impact area with a magnitude several times higher than the average impact pressure and a duration of about 50 μ s has been reported.^[8] The measured pressure inside the high-pressure ring is comparable in value with the calculated water hammer pressure under similar conditions.^[8,19] The duration and the effective area of this pressure are so small that they cannot have any lasting effect on soil surface. However, the propagation of shock waves into soil aggregates may cause lines of weakness and promote their early collapse under subsequent impacts.

Stress Associated with Lateral Flow

Lateral flow of drop water follows the short compressive period of impact. This flow is several times faster than that of impact velocity.^[19,20] Lateral flow slows down to about impact velocity in less than 1 m and turns into a corona that expands both vertically and horizontally. The shear stress associated with the initial high-speed lateral flow is very high and operates on a circle just outside the impact area.^[8] Such a high shear stress could cause damage to the surface and remove some soil material no matter how small the drops or stable the aggregates are. There are, therefore, two possible causes for the existence of a high-pressure ring around the periphery of the impact area, a short-lived compressive impact, and the shear stress associated with lateral flow.^[8,19]

SPLASH

Formation and Characteristics of Splash Corona

Initial sideways flow and the early stages of splash corona appear to be made entirely from drop water. Surface water and material join in later when the third process, cratering, begins digging into the surface.

Sideways flow of the impacting drop soon turns into a splash corona, which continues to expand laterally and grow vertically. The characteristics of splash corona are affected mainly by the presence or absence of a water film on the surface and surface roughness.^[20] Splash on an unsaturated compacted soil produces a corona with a small angle with the surface and a short life. When the surface material is loose enough to be set in motion by the impacting drop (i.e., saturated soil paste), the angle of the splash corona increases, and as the surface material joins in, the corona gets larger and taller before breaking up into droplets (Fig. 1). With a water film on the surface, corona walls become almost vertical and splash duration increases significantly (Fig. 2). Splash corona on a

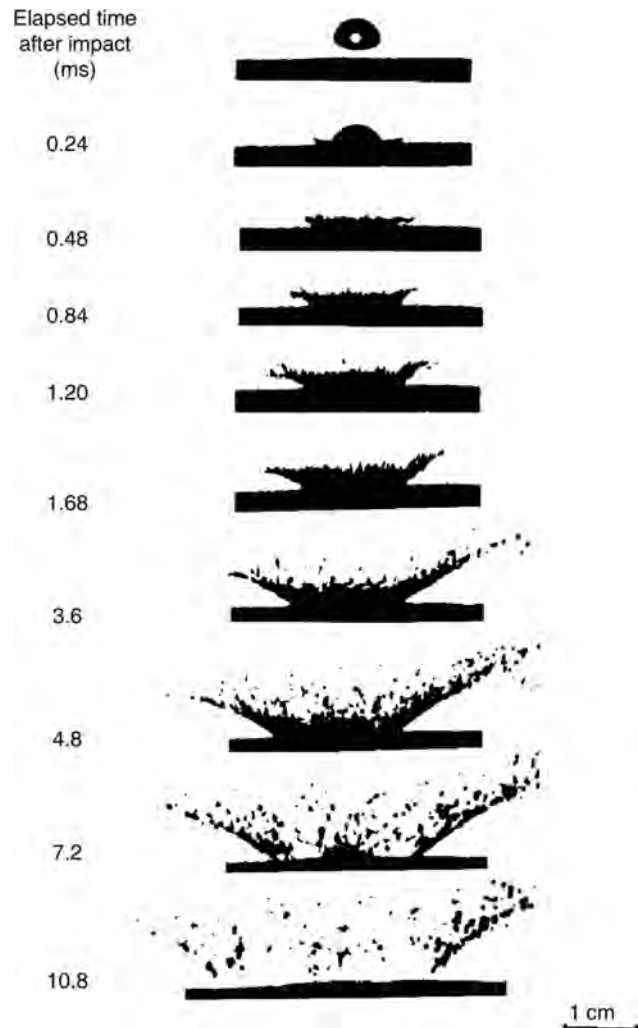


Fig. 1 Splash caused by a 5-mm water drop falling at its terminal velocity on a saturated soil paste.

bed of saturated stable aggregates or sand particles is discontinuous, consisting of a number of tall discrete jets formed when flow is intercepted by large particles.^[20]

Splash Droplets

Splash droplets start forming almost immediately after the lateral flow of the water drop begins, but the early droplets are very small and, because of their trajectory and size, cannot travel far. Splash droplets grow in size with the elapse time after impact. They get larger as the surface water and material join in to make the corona wall thicker and flow within the wall slower (Fig. 1).^[10,20]

The mechanism of the breakup of the top of the corona into jets and jets into droplets has received some attention but has not been fully studied.^[21] It has been established that a cylinder of water becomes unstable when its length exceeds its perimeter, and oscillation and surface tension cause segmentation of the jet with the length of each

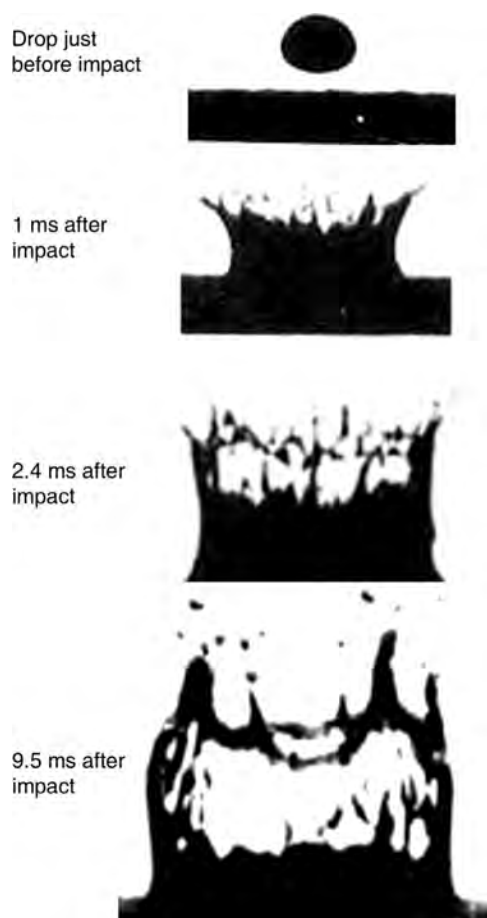


Fig. 2 Splash caused by a 5-mm water drop falling at its terminal velocity on a soil surface covered with a 5-mm water film.

segment depending on the velocity and the thickness of the jet. The breakup of the jets into droplets and the variation with time of the size and velocity of droplets are not affected by the presence or absence of soil material in the jets or in droplets. If soil particles are big and stable, they will be lifted toward the end of the splash when droplets are big enough to carry large particles. In other words, unless the duration of splash is long enough for the formation of droplets as big or bigger than soil particles, the lift up of these particles will not take place.^[8] The prolongation of splash duration is mainly associated with the presence of surface water and, to a lesser extent, with raindrop size. The smaller the target particles, intrinsically or following breakdown under impact, the earlier is their removal by splash and the faster and further they travel (Fig. 1). When soil surface is not covered with a thick water film, almost all the splashed droplets carry some soil material regardless of how stable the aggregates are.^[8,20,22]

The number and distribution of splash droplets are affected by the condition of soil surface, in particular the presence and the thickness of a water film on the soil surface and the size and velocity of the impacting drop. A 5-mm drop impacting at terminal velocity on a saturated

soil produces about 5000 splash droplets. The number decreases to about 1200 for a 2-mm drop.^[8,20] Others have reported lower numbers of splash droplets,^[23] possibly because of the inability of their techniques to capture and record very small droplets formed during the early stages of splash.^[22,24]

Loss of Soil Material by Splash

Between 15% and 25% of impact energy is used up in splashing water and soil material. The relationship between splash loss and kinetic energy is close but non-linear. The same relationship exists between soil shear strength and splash loss.^[25–28] Impact impulse appears to have a better and more linear relationship with splash loss, which suggests that an erosivity index based on impact force or stress may be more valid than the more popular energy-based index.^[8]

Splash loss appears to be dependent on surface water status and soil particle size. It has been said that a water film of depth of up to one-drop diameter increases the effectiveness of impact and increases the loss of underlying material. However, a detailed study carried out by Ghadri^[8] showed that the presence of a water film of any thickness cushions the impact and reduces its effectiveness in dislodging and splashing of surface material. It also reduces the traveling distance of splash droplets by making the corona walls vertical and the angle of splash close to 90° (Fig 2). Thus, the only effect that a water film, of any thickness, can have on splash is to reduce the splash loss of underlying material. Soil water tension also reduces the splash loss by making the splash duration shorter. Splash loss of surface material is, therefore, at its maximum when the soil is at or just above saturation (Fig. 3).

Total splash, water plus solid particles, increases with the depth of surface water up to about one-drop thickness. If, therefore, the first impact brings target particles into suspension to be splashed out by a subsequent impact, this may result in an increase in material loss with the presence of a thin film of water on the surface.

Slope steepness appears to have little effect on the total amount of material lost through splash, thus making splash

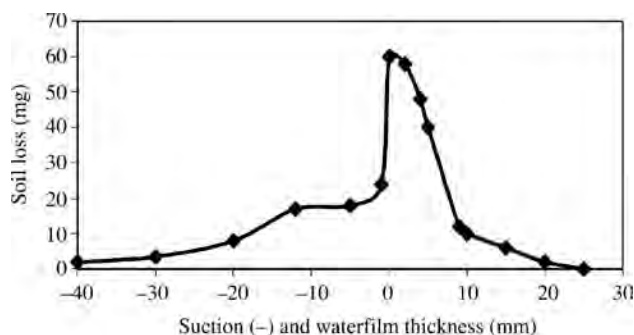


Fig. 3 Effect of surface water condition on the loss of soil material by rain splash.

loss a factor of the actual impact velocity rather than its normal component. The main effect of slope is on the distribution of splash material up and down the slope and their traveling distances.^[8,20] The following relationship has been suggested by Ghadiri^[8] between percent slope (S) and the downslope moving fraction of splash material (D)

$$D = 56 + S \quad (2)$$

CRATERING

During the first few microseconds of impact, all surfaces, including water, behave like a solid and have no involvement in the lateral flow of drop water.^[8,10,24,29,30] Once the surface water or material is set in motion, a crater begins to form and a splash corona starts to rise tangent to the crater sides. A crater formed in deep water soon obtains a hemispherical shape, with its depth and diameter steadily increasing to a maximum, to be followed by its collapse and the formation of a tall Raleigh jet.^[21]

Impact on different surfaces produces craters of different shapes and sizes, ranging from circular depressions on compacted sand and soil under suction to almost completely hemispherical shapes on the saturated soil pastes. One of the features of the craters formed on most soils is the formation of a relatively large rim around its edges due to the deposition of slow moving splash material toward the end of the cratering process. This rim makes up the bulk of the large difference between crater volume and the volume of splashed material. Rim material can become a part of the erosion loss if rainfall is followed by runoff or if the soil is on a steep slope where rim materials are prone to mobilization. Crater volume, therefore, may be a better indicator of raindrop's capacity to dislodge soil material than the amount of soil splashed out.

Between 10% and 20% of impact energy is spent in cratering, and both low and high-speed impact studies have reported direct, but curvilinear relationships between impact energy and crater volume.^[30,31] The following empirical equation has been developed between crater volume (V), impact energy (E_k), and target shear strength (σ) for high-speed impacts:^[31]

$$V = E_k / 4\sigma \quad (3)$$

where E_k/V was later shown to be dependent on drop size and impact velocity. E_k was later replaced with impact impulse (I) as I/V proved to be independent of drop size and velocity.^[8] The following equation was then developed between crater volume (V), impact impulse (I), and soil strength, the latter represented by the depth of cone penetration (P) of Hansbo's^[32] fall cone penetrometer:^[8]

$$V = kI P^2 \quad (4)$$

where k is a proportionality constant.

CONCLUSION

Impact impulse and surface shear strength appear to be the main drop and soil factors involved in all three processes of impact, splash, and cratering. As impulse is a product of impact force and its duration, it is possible that the entire prerunoff process of rain erosion is driven by impact force and its duration. This would account for the disproportionately higher erosivity of larger raindrops. The existence of a zone of transient high pressure around the peripheries of the impacting drops, caused by the compressive deformation of the drop (water hammer) or by the high-speed lateral flow, is established, but its significance in the breakdown of soil aggregates or their weakening by the shock waves generated under impact is not fully understood.

The highest amount of splash loss occurs when the soil surface is at or just above saturation. Any degree of suction to soil water or depth of water on the surface reduces the splash loss. Surface slope only affects the distribution of splash particles with a greater proportion moving downslope as the slope angle increases. Release velocity and trajectory of splash droplets are dependent on the angle and velocity of splash corona. Crater size and shape are dependent on both drop and surface characteristics. Craters formed in water or saturated soil pastes are hemispherical but shallow with a raised center in compacted soil and sand. Crater volume, which is a more accurate measure of total soil loss by the impacting drop, can be predicted using impact impulse and surface shear strength.

REFERENCES

1. Laws, J.O. Recent studies in raindrops and erosion. *Agr. Eng.* **1940**, *21*, 431–433.
2. Ellison, W.D. Studies of raindrop erosion. *Agr. Eng. Mich.* **1944**, *25*, 131–136, 181–182.
3. Ellison, W.D. Some effects of raindrops and surface flow on soil erosion and infiltration. *Trans. Am. Geophys. Union* **1945**, *26*, 415–429.
4. Smith, D.D.; Wischmeier, W.H. Rainfall erosion. *Adv. Agron.* **1962**, *14*, 109–149.
5. Morgan, R.P.C.; Quinton, J.N.; Smith, R.E.; Govers, G.; Poesen, J.W.A.; Auerswald, K.; Chisci, G.; Torri, D.; Styezen, M.E. The European Soil Erosion Model (EUROSEM): A dynamic approach for predicting sediment transport from field and small catchments. *Earth Surf. Proc. Land.* **1998**, *23* (6), 527–544.
6. Renards, K.G.; Foster, G.G.; Yoder, D.C.; McCool, D.K. RUSLE revisited: Status, questions, answers and future. *J. Soil Water Conserv.* **1994**, *49*, 213–220.
7. Alder, W.F. Rain impact: Retrospective and vision for the future. *Wear* **1999**, *233–235*, 25–38.
8. Ghadiri, H. *Raindrop Impact, Soil Splash and Cratering*. Ph.D. thesis, Reading University, Reading, UK, 1978.

9. Ghadiri, H.; Payne, D. The risk of leaving soil surface unprotected against falling rain. *Soil Till. Res.* **1986**, *8*, 119–130.
10. Ghadiri, J.; Payne, D. Raindrop impact stress. *J. Soil Sci.* **1981**, *32*, 41–49.
11. Nearing, M.A.; Bradford, J.M.; Holtz, R.D. Measurement of force vs. time relations for waterdrop impact. *Soil Sci. Soc. Am. J.* **1986**, *50*, 1532–1536.
12. Nearing, M.A.; Bradford, J.M. Relationship between waterdrop properties and forces of impact. *Soil Sci. Soc. Am. J.* **1987**, *51*, 425–430.
13. Nearing, M.A.; Bradford, J.M.; Holtz, R.D. Measurement of waterdrop impact pressures on soil surfaces. *Soil Sci. Soc. Am. J.* **1987**, *51*, 1302–1306.
14. Gardner, F.W. The erosion of steam turbine blades. *Engineer* **1932**, *153*, 202–205.
15. Engel, O.G. Waterdrop collisions with solid surfaces. *J. Res. Nat. Bur. Stand.* **1955**, *54*, 281–298.
16. Jenkins, D.C.; Booker, J.D. A photographic study of the impact between water drop and a surface moving at high speed. *R.A.E. Techn. Note No. Mech. Eng.* **1958**, 275.
17. Rich, G.R. Water-hammer. In *Hydraulic Transients*; McGraw-Hill Book Company Inc.: New York, 1951; 7–45.
18. Alder, W.F. The mechanics of liquid impact. In *Treatise on Material Science and Technology*; Preece, C.M., Ed.; Academic Press: New York, 1979; Vol. 16, 127–183.
19. Ghadiri, H.; Payne, D. Raindrop impact and soil splash. *J. Soil Sci.* **1978**, *38* (2), 38–57.
20. Ghadiri, H.; Payne, D. The formation and characteristics of splash following raindrop impact on soil. *J. Soil Sci.* **1988**, *39*, 563–575.
21. Rayleigh, L. On the capillary phenomena of jets. *Proc. R. Soc. London* **1879**, *29*, 71–97.
22. Mihara, Y. Raindrop and soil erosion. *Bull. Natl. Inst. Agr. Sci. Japan Ser. A* **1952**, 1–51.
23. Atkins, W.W. Hypervelocity penetration studies. *Proceedings of 3rd Hypervelocity Impact Symposium*, U.S. Naval Research Lab, 1959; Vol. 1, 199–211.
24. Mutchler, C.K. *Size, Travel and Composition of Droplets Formed by Waterdrop Splash of Thin Water Layers*. Ph.D. Thesis, Minnesota University: Minneapolis, 1970.
25. Cruse, R.M.; Larson, W.E. Effect of soil shear strength on soil detachment due to raindrop impact. *Soil Sci. Soc. Am. J.* **1977**, *41*, 777–781.
26. Al-Durrah, M.M.; Bradford, J.M. New method of studying soil detachment due to waterdrop impact. *Soil Sci. Soc. Am. J.* **1981**, *45*, 949–953.
27. Al-Durrah, M.M.; Bradford, J.M. Parameters for describing detachment due to single waterdrop impact. *Soil Sci. Soc. Am. J.* **1982**, *46*, 836–840.
28. Nearing, M.A.; Bradford, J.M. Single waterdrop detachment and mechanical properties of soils. *Soil Sci. Soc. Am. J.* **1985**, *49*, 547–552.
29. Worthington, A.M. *A Study of Splashes*; Longmans Green & Co.: New York, 1908.
30. Engel, O.G. Crater depth in fluid impacts. *J. Appl. Phys.* **1966**, *37*, 1798–1808.
31. Cook, M.A. Mechanism of cratering in ultra-high velocity impact. *J. Appl. Phys.* **1959**, *30*, 725–735.
32. Hansbo, S. A new approach to the determination of the shear strength of clay by the fall-conc test. *Roy. Swed. Geotech. Inst. Proc.* **1957**, *14*, 5–47.

Rapid Soil Analysis: Diffuse Reflectance Spectroscopy

Keith D. Shepherd

World Agroforestry Center, International Centre for Research in Agroforestry (ICRAF), Nairobi, Kenya

Markus G. Walsh

World Agroforestry Centre (ICRAF), Kisumu, Kenya

Abstract

Diffuse reflectance spectroscopy (DRS) is a technology for non-destructive characterization of the composition of materials based on the interaction of visible–infrared light (electromagnetic energy) with matter. There is potential for use of DRS to increase efficiencies and reduce costs in both large-area applications (e.g., soil survey, watershed management, pedotransfer functions, and soil quality indicators) and site-specific management problems (e.g., precision agriculture, farm advisory services, and process studies). In particular, the ability to rapidly characterize large numbers of samples with DRS opens up new possibilities for risk-based approaches to soil evaluations that explicitly consider uncertainty in predictions and interpretations of soil properties.

BACKGROUND

Near-infrared spectroscopy is routinely used for rapid analysis of a wide range of materials in many laboratory and process control applications in agriculture, food, geology, and biomedicine.^[1] Both the visible–near-infrared (0.35–2.5 μm) and mid-infrared (2.5–25 μm) wavelength regions have been investigated for non-destructive analysis of soils and simultaneous prediction of a number of soil properties.

Although diffuse reflectance spectroscopy (DRS) has been investigated in soils since the 1960s, it is not routinely used in soil analysis. The visible (0.4–0.7 μm) region has been used for color determinations in soil and geological applications as well as in the identification of iron oxides and hydroxides.^[2] There has been only limited success with sensing of soil properties in the field. However, research has demonstrated the ability of DRS to provide non-destructive rapid prediction of soil physical, chemical, and biological properties in the laboratory.^[3,4]

HOW DOES IT WORK?

Physics

Samples are illuminated with an artificial light source, and reflected light diffusing from the sample is collected and channeled through fiber optic cables to arrays of light detectors (Fig. 1). The detectors measure the

amount of light received in narrow wavelength bands (typically 3–10 nm in the visible–near-infrared range) as an electrical signal. The electrical signal in each wave band is expressed relative to the electrical signals obtained from a reference standard. The relative reflectance in each wave band comprises the reflectance spectrum for a sample (Fig. 2), which is displayed and stored on a computer.

The shape of the spectrum is dependent on all the structural groups of atoms that absorb visible and infrared light, which in turn are correlated to the major chemical and physical components of a substance. When light interacts with a sample, light is absorbed to different degrees in each wave band due to electronic transitions of atoms and vibrational stretching and bending of structural groups of atoms that form molecules and crystals. Fundamental features in reflectance spectra occur at energy levels that allow molecules to rise to higher vibrational states. For example, the fundamental features related to various components of soil organic matter (e.g., symmetric C–H stretching) generally occur in the mid-infrared to thermal infrared range (2.5–25 μm), but their overtones (at one-half, one-third, one-fourth, etc., of the wavelength of the fundamental feature) occur in the near-infrared region (0.7–2.5 μm). Soil minerals such as different clay types have very distinct spectral signatures in the near-infrared region because of strong absorption of the overtones of SO_4^{2-} , CO_3^{2-} , and OH^- and combinations of fundamental features of, e.g., H_2O and CO_3^{2-} .^[5] Absorption due to charge transfer and crystal field effects in Fe^{2+} and Fe^{3+} is particularly evident at 0.35–1.0 μm .^[5] However, because soil spectra are a product of many



Fig. 1 Portable visible–near-infrared spectrometer being used to scan a soil sample contained in a petri dish. With this setup, 500 samples per day can be routinely scanned.

overlapping absorption features of organic and mineral materials, they generally have few distinct absorption features. This makes qualitative interpretation of individual features of limited value, but even subtle differences in shape can yield quantitative information on soil properties.

From a theoretical standpoint, mid-infrared spectroscopy (Fourier transform spectroscopy) may be superior to visible–near-infrared spectroscopy because it detects fundamental features as opposed to their overtones and is sensitive to quartz. However, visible–near-infrared instruments are: 1) simpler and more portable; 2) able to scan larger soil samples more rapidly; and 3) better commercially supported than mid-infrared instruments.

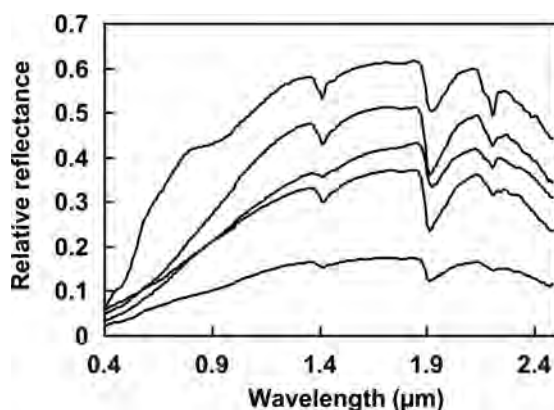


Fig. 2 Visible–near-infrared spectra of a range of soil samples from Kenya. The absorption features at 1.4 and 2.2 μm are due to both lattice and free water OH, whereas the adsorption feature at 1.9 μm is due to free water OH only.

Chemometrics

Extracting information about properties of interest from reflectance spectra requires specialized multivariate calibration and classification techniques—the field of chemometrics.^[6] The general aim is to find relationships between measurements made in the laboratory or field that are expensive or labor-intensive and the reflectance spectra, which are easy and cheap to acquire. To obtain robust calibrations, one must minimize information in the spectra that is not relevant to the prediction of the target variable. Various data transformations may be performed to minimize irrelevant information produced by the effects of light scattering, variation due to sample presentation (e.g., thickness, packing, and particle size) and optical setup, and statistical problems such as multicollinearity (correlation among wavelength bands) and non-linearity.^[6] Optimal transformations depend on the individual data set, but first derivative transformation has been commonly used for visible–near-infrared soil spectra.

Multivariate calibration methods are used to relate the measured soil property to reflectance values in a number of different wavelength bands. Methods that include compression of the spectral data are commonly used because of the problem of multicollinearity. The most common methods are principal components regression and partial least squares regression.^[6] However, non-linear parametric regression methods (e.g., multivariate adaptive regression splines), non-parametric regression methods (e.g., regression trees), and classification methods (screening tests using classification trees) have also been used.^[4] Careful procedures for sample selection and validation of calibration models are required to obtain robust calibrations with stable predictive ability.^[6] Standardization of calibration models among different spectrometers can be done by scanning standards on the master and host instruments and simple subtraction of master and host spectra before the unknown sample is predicted.^[7]

Soil Reference Methods

Calibration success will depend on how well the spectroscopic measurement relates to the reference measurement. Ideally, calibration performance should also be assessed relative to the error in reference measurements, e.g., due to variation among duplicates, batches, and laboratories (Fig. 3).

Primary properties of substances that significantly affect the shape of a soil spectrum generally calibrate well to soil reflectance. These include mineral composition, organic matter, water (e.g., hydration, hygroscopic, and free pore water), iron form and amount, carbonates, salinity, and particle size distribution.^[2] Interestingly, these properties also largely determine the capacity of soils to perform production, environmental, and engineering functions. Indirect information about secondary properties of soils (e.g., low

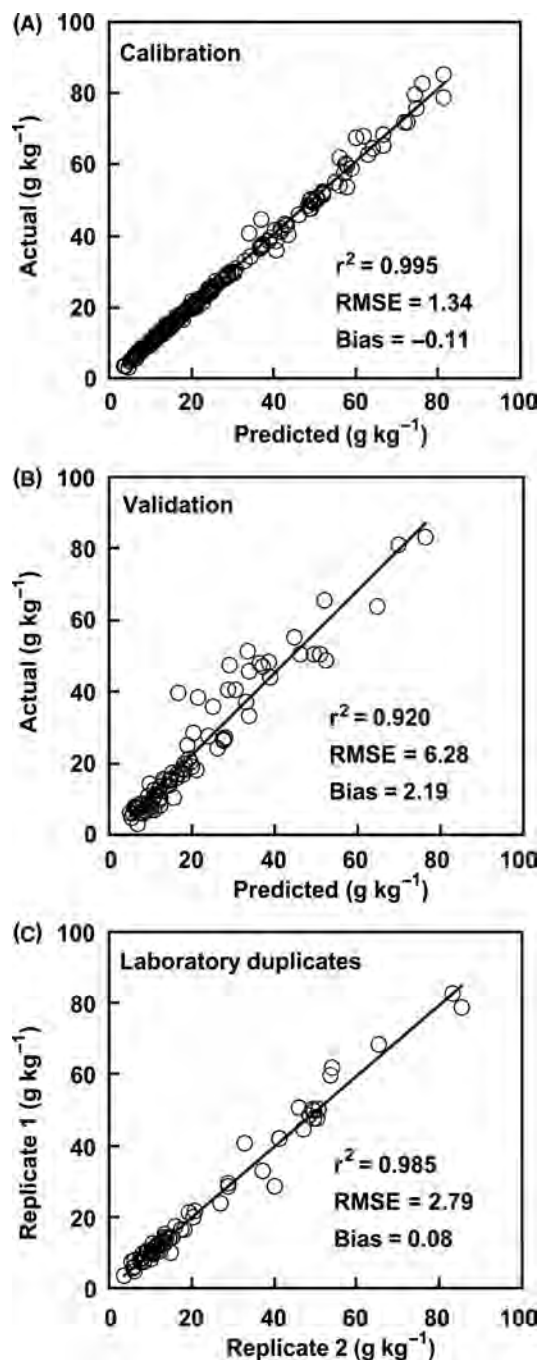


Fig. 3 Soil organic carbon measured in the laboratory against predicted values after calibration to visible-near-infrared reflectance for (A) calibration samples, (B) validation samples (25% of total data withheld), and (C) values for laboratory duplicates. RMSE is root mean square error. Note how the increased error in the calibration and validation data (A, B) at organic carbon values of $>20 \text{ g kg}^{-1}$ is closely related to the error in the laboratory data (C). At organic carbon values of $<20 \text{ g kg}^{-1}$, RMSE was 0.4 for calibration, 1.9 for validation, and 1.3 for laboratory duplicates. Organic carbon was determined by dry combustion on acidified samples for a wide range of Kenyan soils (pH, 4.5–9; clay, 200–800 g kg^{-1} , effective cation exchange capacity per unit mass of clay (ECECclay) = 11–127 $\text{cmol}_c \text{ kg}^{-1}$).

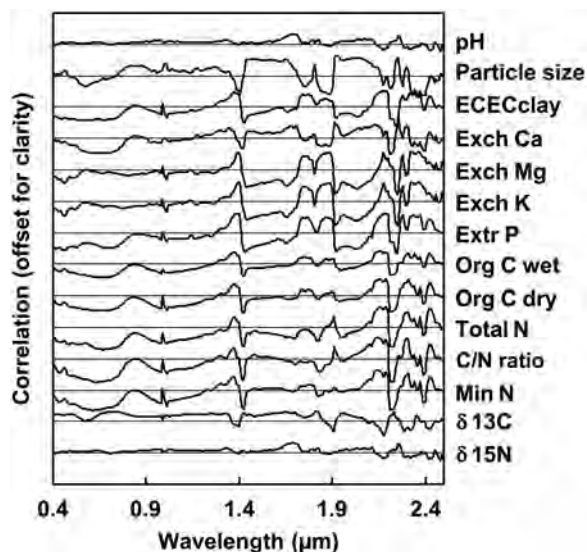


Fig. 4 Degree of correlation (–1 to +1) between soil properties and visible-near-infrared reflectance at different wavelengths, for pH, geometric mean particle size (particle size), effective cation exchange capacity per unit mass of clay (ECECclay), exchangeable calcium (Exch Ca), exchangeable magnesium (Exch Mg), exchangeable potassium (Exch K), modified Olsen extractable phosphorus (Extr P), organic carbon by wet oxidation (Org C wet), organic carbon by dry combustion (Org C dry), total nitrogen (Total N), C/N ratio (C/N ratio), anaerobic incubation mineralizable N (Min N), $\delta^{13}\text{C}$ isotopic ratio ($\delta^{13}\text{C}$), and $\delta^{15}\text{N}$ isotopic ratio ($\delta^{15}\text{N}$). The plots are based on large (>300) samples of diverse Kenyan soils.

concentrations of nutrients in soil extracts, potentially mineralizable carbon (C) and nitrogen, stable isotopes) can also be often obtained because of their interactions with primary soil properties (Fig. 4). For example, DRS has been successfully used to predict cadmium and zinc contamination in floodplain soils because of the associations of these heavy metals with soil organic matter and clay.^[8] Micronutrient concentrations have been predicted from mid-infrared reflectance because of their association with clay mineralogy.^[9] DRS has been used to predict several basic soil properties for a wide range of African topsoils (Table 1).

Merits and Limitations

The main merits of DRS are its repeatability and speed compared with conventional soil analyses. DRS can give better precision and accuracy than the actual reference measurement,^[6] particularly when the error in the reference measurement is large. Thus, repeatability among laboratories is expected to be greater with DRS than with conventional methods of analysis. The speed of analysis and the ability to estimate many soil properties from a single robust measurement are major advantages of DRS, especially for

Table 1 Calibration of soil properties to soil reflectance for a diverse set of African topsoils.

Soil property	No. of soils	Min.	Max.	Calibration data		Validation data	
				r ²	RMSE	r ²	RMSE
pH	1135	4.2	10.0	0.98	0.14	0.73	0.39
Exchangeable Ca, cmol _c kg ⁻¹	1109	0.16	47.0	0.99	1.1	0.86	3.2
Exchangeable Mg, cmol _c kg ⁻¹	1109	0.01	17.9	0.98	0.42	0.84	1.0
ECEC, cmol _c kg ⁻¹	1109	0.40	55.0	0.99	1.1	0.86	4.0
Organic C, g kg ⁻¹	1011	2.3	55.8	0.98	1.1	0.82	2.9
Sand, g kg ⁻¹	682	80	900	0.99	22	0.78	100
Clay, g kg ⁻¹	682	50	790	0.99	19	0.79	73
ECEC per unit clay, cmol _c kg ⁻¹	681	3.1	190	0.90	8.4	0.81	10

Note: Calibrations were done using a random sample of 67% of the soils and validation on the remaining 33% of soils.

RMSE: root mean square error; ECEC: effective (unbuffered) cation exchange capacity; C = carbon; Mg = magnesium; Ca = calcium.

Source: From Shepherd & Walsh.^[4] ©2002, reanalyzed using TreeNet[®] regression.

soil properties that are time-consuming to measure by conventional methods. The main limitations of DRS are the need to build calibration libraries for a given population of soils for all the soil properties of interest and the complexity of the data analysis that it involves. Development of global calibration libraries in centralized laboratory facilities and software development for automated data analysis could help to reduce these limitations.

APPLICATION

Soil spectral libraries can be used to generalize results of soil assessments that are conducted at a limited number of sites and thereby increase the efficiency of expensive time-consuming soil-related studies.^[4] The variability of soils in a study area is thoroughly sampled and spectrally characterized. Soil properties or attributes of soil functional capacity are measured on only a selection of soils, designed to sample the variation in the spectral library, and then calibrated to soil reflectance. The soil functional attributes can then be predicted for the entire library and for new samples from the study area. New samples that are classified as spectral outliers to the library are characterized and added to the calibration library, thereby increasing the predictive value of the library.

The spectral library approach has potential for direct and simultaneous prediction of soil attributes for agricultural, environmental, and engineering applications. Because soil reflectance provides an integrated measure of the number of fundamental soil properties, such calibrations could perform better and would certainly be more rapid than pedotransfer functions based on conventional measurements of soil properties.

The rapid nature of the measurement allows soil variability to be more adequately sampled than with conventional approaches and thereby facilitates risk-based

approaches and landscape-level soil assessments, including provision of more sites for calibration to digital terrain and remote sensing information for soil landscape modeling. For example, the authors are applying the approach using spectral libraries composed of thousands of samples to study the effects of historic land use changes on soil quality and to establish baselines for C offset projects. The approach also facilitates soil monitoring because large numbers of samples from repeated samplings can be characterized. For site-specific management, the method allows large numbers of samples to be taken from a field, which may give a better overall estimate of a given soil property than more accurate measurements of the property at a lower sampling density.

CONCLUSION

Developments can be expected toward cheaper and more portable spectrometers, coupled with more flexible software, and easier calibration methods. The technology should be increasingly used in a wide range of soil studies and surveys, and spectrometers are likely to become standard equipment in soil laboratories. Soil spectral libraries will likely form the basis for a new generation of expert systems and probabilistic advisory systems for predicting soil properties and responses to soil management. DRS can also be used for the characterization of organic resource quality for soil management.^[10]

REFERENCES

1. Davies, A.M.C.; Cho, R.K., Eds. *Near Infrared Spectroscopy: Proceedings of the 10th International Conference*; NIR Publications: Chichester, 2002.
2. Ben-Dor, E.; Irons, J.R.; Epema, G.F. Soil reflectance. In *Remote Sensing for the Earth Sciences: Manual of Remote*

Sensing; Rencz, N., Ed.; John Wiley & Sons: New York, 1999; Vol. 3, 111–188.

3. Janik, L.J.; Merry, R.H.; Skjemstad, J.O. Can mid infrared diffuse reflectance analysis replace soil extractions? *Aust. J. Exp. Agr.* **1998**, *38*, 681–696.
4. Shepherd, K.D.; Walsh, M.G. Development of reflectance spectral libraries for characterization of soil properties. *Soil Sci. Soc. Am. J.* **2002**, *66*, 988–998.
5. Clark, R.N. Spectroscopy of rocks and minerals, and principles of spectroscopy. In *Remote Sensing for the Earth Sciences: Manual of Remote Sensing*; Rencz, N., Ed.; John Wiley & Sons: New York, 1999; Vol. 3, 3–52.
6. Naes, T.; Isaksson, T.; Feam, T.; Davies, T. *A User-Friendly Guide to Multivariate Calibration and Classification*; NIR Publications: Chichester, 2002.
7. Shenk, J.S.; Workman, J.J., Jr; Westerhaus, M.O. Application of NIR spectroscopy to agricultural products. In *Handbook of Near-Infrared Analysis*, 2nd Ed.; Bums, D.A., Ciurczak, E.W., Eds.; Marcel Dekker, Inc.: New York, 2001; 419–474.
8. Kooistra, L.; Wehrens, R.; Leuven, R.S.E.W.; Buydens, L.M.C. Possibilities of visible–near-infrared spectroscopy for the assessment of soil contamination in river flood plains. *Anal. Chim. Acta* **2001**, *446*, 97–105.
9. Bertrand, I.; Janik, L.J.; Holloway, R.E.; Armstrong, R.D.; McLaughlin, M.J. The rapid assessment of concentrations and solid phase associations of macro- and micronutrients in alkaline soils by mid-infrared diffuse reflectance spectroscopy. *Aust. J. Soil Res.* **2002**, *40*, 1339–1356.
10. Shepherd, K.D.; Palm, C.A.; Gachengo, C.N.; Vanlauwe, B. Rapid characterization of organic resource quality for soil and livestock management in tropical agroecosystems using near infrared spectroscopy. *Agron. J.* **2003**, *95*, 1314–1322.

Rare Earth Elements

Zhengyi Hu

Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China

Gerd Sparovek

Luiz de Queiroz College of Agriculture (ESALQ) Piracicaba, University of Sao Paulo, Sao Paulo, Brazil

Silvia H. Haneklaus

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

The rare earth elements (REEs) comprise the elements scandium ($Z = 21$) and yttrium ($Z = 39$), and 15 lanthanides with successive atomic numbers (Z) from 57 to 71. REEs are applied to soils as fertilizer materials or as contaminations of industrial sludges so that an assessment of their behavior in soils is required for evaluating agro-environmental effects. This entry summarizes the knowledge of soil chemical properties of REEs.

ORIGINS OF SOIL RARE EARTH ELEMENTS (REEs)

Soil REEs mainly originate from parent materials.^[1–6] Application of phosphate fertilizer^[3] and phosphogypsum^[4] can supply REEs to the soil (Table 1). Some of the sludges, particularly those from the chemical industry, have been contaminated with REEs (Table 2). Continuous application of sewage sludges caused an accumulation of scandium and Samarium (Sm) in some soils in Japan.^[6] The use of REEs in agriculture is widely practiced in China. By 2001, 6.5 million hectares of land in China was treated with REE fertilizers.^[2] In total, 11,000 tons of REEs were applied in agricultural production in China.^[7] Besides the parent material, the application of REEs on agricultural farmland is going to be a major source of REEs if the practice of applying them regularly proceeds.

CHEMICAL SPECIATION OF REEs IN SOILS

Total Content

Representative background values of REEs in soils are available so far only for China and Japan.^[1,2,8] The REE content strongly depends on the parent material.^[1] The results of 853 soil analyses showed that the total REE content in soils varied between 18 and 583 mg kg⁻¹, with a mean value of 184 mg kg⁻¹.^[2] The light REEs lanthanum (La), cerium (Ce), praseodymium (Pr), neodymium (Nd), Sm and Europium (Eu) account for 90%

of the total REE content in soils (Table 3). On average, the La, Ce, Nd, Sm, and Eu content was 41, 73, 7.3, 27.5, and 5.6 mg kg⁻¹, respectively, in different soils in China ($n = 467$; Table 3).

Species of REEs in Soils

Binding forms of REEs in soils may be classified according to their availability for plants.^[1] Approximately 9 mg kg⁻¹ is exchangeable, 2, 5, 32, and 95 mg kg⁻¹ are bonded to carbonates, manganese (Mn) oxides, organic substances, and amorphous iron (Fe) oxides, while 59 mg kg⁻¹ is abundant in the form of crystal Fe oxides and 105 mg kg⁻¹ in residual forms (Table 4). Plants utilize exchangeable REEs most easily, while the uptake of other forms is limited.^[1,9] In contrast, residual forms are not plant available.^[1,9]

For determining the content of plant-available REEs, the following extractants have been proposed: 1 M HAc–NaAc (pH 4.8),^[1,9] 1.0 M NH₄NO₃ (pH 7.0),^[10] 0.1 M HCl,^[11] and 0.1 M malic–citric acid,^[12] with 1 M HAc–NaAc being most extensively used.^[1,2] Plant-available REE contents are highly variable and range from <1 to >200 mg kg⁻¹, with a mean value of about 12 mg kg⁻¹ ($n = 1790$).^[1] Physicochemical soil properties such as pH, E_h , cation exchange capacity (CEC), clay, H₂PO₄⁻, and carbonate content have a strong impact on the amount of exchangeable and plant-available REEs.^[1,9] Acid soils contain significantly higher amounts of plant-available REEs than more alkaline, calcareous soils.^[2] The availability of soil applied REEs is usually significantly higher than that in the original soil matrix.^[1,9]

Table 1 Symbol and atomic numbers of REEs.

Elements	Symbol	Atomic number	Descriptive classification
Scandium	Sc	21	Light earths
Yttrium	Y	39	
Lanthanum	La	57	
Cerium	Ce	58	
Praseodymium	Pr	59	
Neodymium	Nd	60	
Promethium	Pm	61	
Samarium	Sm	62	
Europium	Eu	63	
Gadolinium	Gd	64	
Terbium	Tb	65	
Dysprosium	Dy	66	
Holmium	Ho	67	
Erbium	Er	68	
Thulium	Tm	69	Heavy earths
Ytterbium	Yb	70	
Lutetium	Lu	71	

Source: From Liu.^[1]

Adsorption of REEs in Soils

In general, 95% of the added REEs are adsorbed.^[13] REEs added to soils are rapidly transformed, e.g., into exchangeable, organic matter–bonded, and Fe/Mn oxide–bonded species.^[1,9] The distribution coefficients for REEs added to red

Table 3 Mean content of REEs in soils extracted by Na₂O₂/NaOH (n = 467).

Light REEs	Content		Heavy REEs	Content		Total contents (mg kg ⁻¹) and ratios
	(mg kg ⁻¹)			(mg kg ⁻¹)		
La	41.2		Gd	4.8		Total REE (T) 172.8
Ce	73.4		Tb	0.7		Light REE (L) 156.0
Pr	7.3		Dy	4.4		Heavy REE (H) 16.8
Nd	27.5		Ho	0.9		L/H 9.3
Sm	5.6		Er	2.7		L/T 0.9
Eu	1.1		Tm	0.4		
			Yb	2.5		
			Lu	0.4		

Source: From Liu.^[1]

and yellow brown soils declined in the following order: residual >exchangeable >organic matter–bonded >Fe/Mn oxide–bonded REEs.^[9] The formation of bridged hydroxo complexes is probably the dominant sorption mechanism to clay minerals.^[14] Clay type, pH, CEC, organic matter, and amorphous Fe content regulate the adsorption kinetics of REEs.^[1,2,9] Langmuir and Freundlich equations were found to describe precisely the absorption of REEs in soils.^[1,15]

Translocation of REEs in Soils

The question open is whether the use of REEs in industry and agriculture may result in a pollution of soils, plants, and groundwater. In leaching experiments under controlled

Table 2 Concentrations and coefficient of variation of REEs in different sludges.

Elements	Night soil sludge ^a (n = 10)		Sewage sludge (n = 14)		Food industry sludge (n = 10)		Chemical industry sludge (n = 10)	
	Mean (mg kg ⁻¹)	CV (%)	Mean (mg kg ⁻¹)	CV (%)	Mean (mg kg ⁻¹)	CV (%)	Mean (mg kg ⁻¹)	CV (%)
La	3.39	37	6.70	47	0.89	72	2.46	98
Ce	6.98	44	14.10	58	1.83	77	2.69	105
Pr	0.82	38	1.48	46	0.22	82	0.48	95
Nd	3.18	34	6.00	47	0.91	82	2.04	98
Sm	0.53	36	1.02	40	0.17	81	0.36	95
Gd	0.53	34	1.18	45	0.17	79	0.48	101
Tb	0.07	45	0.16	36	0.03	81	0.06	103
Dy	0.39	53	0.93	33	0.14	76	0.39	110
Ho	0.07	54	0.19	32	0.03	79	0.09	119
Er	0.21	55	0.57	31	0.08	73	0.26	118
Tm	0.03	52	0.08	26	0.01	81	0.03	112
Yb	0.20	58	0.54	31	0.09	83	0.19	109
Lu	0.03	56	0.08	31	0.01	84	0.03	105

^aFeces and urine of humans.

Source: From Kawasaki, Kimura, et al.^[5]

Table 4 Mean content of different species of REEs in soils (mg kg⁻¹).

Soils	n	Exchangeable	Carbonate-bonded	Mn oxide-bonded	Organically bonded	Amorphous Fe oxide-bonded	Crystal ion oxides	Residual	Total
Latosol ^a	3	1.0	—	1.4	24.3	23.8	8.2	17.5	76.2
Red soil	5	14.8	—	8.0	30.1	108.0	33.9	199.8	394.6
Yellow brown soil	2	12.1	—	7.4	26.1	79.3	88.3	55.5	268.7
Brown soil	2	11.5	—	4.7	33.7	78.8	64.1	34.5	227.3
Black soil	2	1.4	5.3	1.2	46.8	88.8	74.8	64.0	282.3
Chernozem	2	1.8	11.4	2.8	41.2	105.3	94.1	79.0	335.5
Mean		9.1	1.9	5.3	32.2	95.4	58.8	105.3	307.9

^aLatosol (rhodic Ferralsol, FAO); red soil (ferralic Cambisol); yellow brown soil (haplic Luvisol); brown soil (haplic Alisol); black soil (luvic Phaeozems); Chernozem (Haplic Chernozems).

Source: From Liu.^[1]

conditions with ¹⁴¹Ce and ¹⁴⁷Nd, these elements were abundant only in the top soil layer because of their strong adsorption.^[9] For field conditions, a translocation depth of <1 cm has been estimated,^[9] but this value still needs to be validated for different soils and on a long-term basis.

Table 5 Total concentration of REEs in plants (mg kg⁻¹)

Species	n	Min (mg kg ⁻¹)	Max (mg kg ⁻¹)
Rice	319	<LLD	1.17
Wheat	440	<LLD	2.58
Corn	139	<LLD	0.92
Cucumber	41	<LLD	0.70
Leek	33	0.04	0.21
Spinach	41	<LLD	0.12
Cauliflower	61	<LLD	0.60
Lotus root	31	<LLD	0.76
Tomato	64	<LLD	0.18
Chinese cabbage	67	0.05	1.01
Pepper	31	<LLD	0.40
Potato	34	0.05	0.35
Cabbage	38	<LLD	1.20
Mushroom	33	<LLD	0.45
Orange	41	0.13	0.70
Litchi	30	<LLD	
Grape	61	<LLD	0.85
Longan	30	<LLD	0.71
Banana	33	<LLD	
Apple	62	0.07	0.80
Pear	34	<LLD	0.24
Watermelon	37	<LLD	0.26
Sugarcane	27	0.05	1.25
Peach	4	<LLD	

Note: LLD, Lower limit of detection.

Source: From Xiong, Zheng, et al.^[9]

Table 6 Ce content in grains and seeds of different crops in different countries and regions of China (mg CeO₂ kg⁻¹).

Crops	Region/country	Sample no.	Mean	Min	Max
Rice	Hubei/China	8	0.06	0.02	0.15
Rice	Jiangxi/China	21	0.06	<0.01	0.18
Rice	Beijing/China	17	0.04	<0.01	0.11
Rice	Nanjing/China	2	—		0.12
Wheat	Heilongjiang/China	21	0.04	<0.01	0.09
Wheat	Hubei/China	7	0.04	0.01	0.07
Wheat	Henan/China	6	0.10	<0.01	0.19
Wheat	Shandong/China	4	0.12	<0.07	0.17
Wheat	Beijing/China	10	0.03	<0.01	0.05
Wheat	Tianjiang/China	1	0.04	<0.01	0.08
Maize	Hubei/China	8	0.01	<0.01	0.02
Maize	Heilongjiang/China	10	0.04	<0.01	0.10
Maize	Tianjiang/China	15	0.04	<0.01	0.00
Barley	Beijing/China	14	0.04	<0.01	0.25
Wheat	Canada	5		0.04	0.16
Wheat	The United States	8		<0.01	0.20
Wheat	Australia	4		0.06	0.16
Wheat	Argentina	4		0.07	0.09
Soybean	The United States	3		<0.01	0.11
Maize	The United States	3		0.01	0.09
Mung bean	Thailand	1			0.09

Source: From Xiong, Zheng, et al.^[9]

UPTAKE OF REEs BY PLANTS

The uptake of REEs has been investigated for a wide range of plants.^[2,9] All rare elements except Pr were found in plants. Uptake of REEs was related to many factors such as element, plant type, and growth conditions.^[2,9] The light REEs such as Ce, La, and Nd were highest in plants.^[2,9] The total concentration of REEs in plant tissue ranges from <0.05 to 2.58 mg kg⁻¹ (Table 5). The concentration of Ce in food products is usually lower than 0.2 mg CeO₂ kg⁻¹ (Table 6). More attention should be paid to research on the uptake, translocation, and distribution of REEs in plants in order to follow up the biological effects of REEs and to assess agro-environmental effects.

CROP RESPONSE TO REEs

Research with a view to the use of REEs in agriculture was predominantly carried out in China and before in Russia. A small number of studies have been carried out in Australia, too.^[2,9] The increases in crop yield reported by workers from all parts of China range between 5% and 103% (Table 7), with an average response of 8–15%.^[2,9] Crop response to REEs is reported to be most probable when soils contain less than 10 mg kg⁻¹ of available REEs (in 1 M HAc–NaAc, pH 4.8), while a response on soils with more than 20 mg kg⁻¹ of available REEs is unlikely

Table 7 Yield increase (relative to control) after application of REEs to different crops.

Crop	Country	Yield increase (%)
Sugar beet	Bulgaria	17–24
Sugar beet	China	7
Wheat	China	6–17
Rape	China	4–48
Potato	China	5–6
Soybean	China	8–9
Cotton	China	5–12
Rice	China	7
Corn	China	9–103
Barley	Australia	18–19
Peanut	China	8–12
Tobacco	China	8–10
Rubber	China	8–10
Sugarcane	China	10–15
Cabbage	China	10–20
Litschi	China	14–17
Grape	China	8–12

Source: From Hu, Richter, et al.^[16]

Table 8 Critical values of plant-available REEs in soils.

REE content (mg kg ⁻¹)	Index	Crop response to REE
<5.0	Very low	Most likely
5–10	Low	Probable
11–15	Medium	Not expected
16–20	High	Not expected
>20	Very high	Unlikely

Source: From Liu^[1] and Xiong, Zheng, et al.^[9]

(Table 8). So far, there is no evidence that REEs are essential for plant growth.

CONCLUSION

Basic information about the chemistry of REEs in soils is available, but these data refer to specific regions and soil types. Besides this, the effect of REEs, e.g., on soil fertility and soil biological diversity is yet unclear. Crop response to REE applications depends on various factors, including soil properties. Further studies are also required in order to elaborate the most efficient application techniques.

REFERENCES

1. Liu, Z., Ed. Rare earth elements in soils. In *Microelements in Soils of China*; Jiangsu Science and Technology Publishing House: Nanjing, 1996; 293–329.
2. Xiong, B.K.; Chen, P.; Guo, B.S.; Zheng, W. *Rare Earth Element Research and Application in Chinese Agriculture and Forest*; Metallurgical Industry Press: Beijing, 2000.
3. Todorovsky, D.S.; Minkova, N.L.; Bakalova, D.P. Effect of the application of superphosphate on rare earth content in the soils. *Sci. Total Environ.* **1997**, *203* (1), 13–16.
4. Arocena, J.M.; Rutherford, P.M.; Dudas, M.J. Heterogeneous distribution of trace elements and fluorine in phosphogypsum. *Sci. Total Environ.* **1995**, *162* (2–3), 149–160.
5. Kawasaki, A.; Kimura, R.; Arai, S. Rare earth elements and other trace elements in wastewater treatment sludges. *Soil Sci. Plant Nutr.* **1998**, *44* (3), 433–441.
6. Zhang, F.S.; Yamasaki, S.; Kimura, K. Rare earth element content in various waste ashes and the potential risk to Japanese soils. *Environ. Int.* **2001**, *27* (5), 393–398.
7. Wang, S.M.; Zheng, W. Developing technology of rare earth application on agriculture and biological field in China. In *Proceedings of the 4th International Conference on Rare Earth Development and Application*, Beijing, China, Jun 16–18, 2001; Yu, Z.S., Yan, C.H., Xu, G.Y., Niu, J.K., Chen, Z.H., Eds.; Metallurgical Industry Press: Beijing, 2001; 237–240.

8. Yoshida, S.; Muramatsu, Y.; Tagami, K.; Uchida, S. Concentrations of lanthanide elements, Th, and U in 77 Japanese surface soils. *Environ. Int.* **1998**, *24* (3), 275–286.
9. Xiong, P.K.; Zheng, W.; Cheng, P.; Wang, F. *Rare Earth Elements in Agricultural Environment*; China Forestry Press: Beijing, 1999.
10. Zhai, H.; Yang, Y.G.; Zheng, S.J.; Hu, A.T.; Zhang, S.; Wang, L.J. Selection of the extractants for available rare earths in soils. *China Environ. Sci.* **1999**, *19* (1), 67–71.
11. Li, F.L.; Shan, X.Q.; Zhang, S.Z. Evaluation of single extractants for assessing plant availability of rare earth elements in soils. *Commun. Soil Sci. Plant Anal.* **2001**, *32* (15 & 16), 2577–2587.
12. Zhang, S.Z.; Shan, X.Q.; Li, F.L. Low-molecular-weight-organic-acid as extractant to predict plant bioavailability of rare earth elements. *Int. J. Environ. Anal. Chem.* **2000**, *76* (4), 283–294.
13. Zhu, J.G.; Xing, G.X.; Yamasaki, S.; Tsumura, A. Adsorption and desorption of exogenous rare earth elements in soils: I. Rate of forms of rare earth elements sorbed. *Pedosphere* **1993**, *3* (4), 299–308.
14. Dong, W.M.; Wang, X.K.; Bian, X.Y.; Wang, A.X.; Du, J.Z.; Tao, Z.Y. Comparative study on sorption/desorption of radioeuropium on alumina, bentonite and red earth: Effects of pH, ionic strength, fulvic acid, and iron oxides in red earth. *Appl. Radiat. Isotopes* **2001**, *54* (4), 603–610.
15. Gao, X.J.; Zhang, S.; Wang, L.J. The adsorption of La^{3+} and Yb^{3+} on soil and mineral and its environmental significance. *China Environ. Sci.* **1999**, *19* (2), 149–152.
16. Hu, Z.Y.; Richter, H.; Sparovek, G.; Schnug, E. Physiological and biochemical effects of rare earth elements on plants and their agricultural significance: A review. *J. Plant Nutr.* **2003**, *in press*.

Redox Phenomena

Bruce R. James

University of Maryland, College Park, Maryland, U.S.A.

Abstract

Oxidation–reduction reactions in soils are governed by the master variables, pH and Eh, and they are controlled kinetically by microbial metabolism and abiotic processes. The heterogeneous, metastable nature of soils makes the study of electron transfers in such environments challenging, but rewarding. Applications of fundamental studies of these processes are readily made to human concerns pertinent to agriculture, environmental quality, and human health.

INTRODUCTION

Electron transfers from electron-rich reductants to electron-poor oxidants in soils change the oxidation state of the electron donor and acceptor, and thereby commonly affect their form, solubility, and mobility. These oxidation and reduction processes involving losses and gains of electrons, respectively, are known as redox reactions, and studying them provides a means to observe and quantify key properties governing the speciation of many elements important in plant nutrition, soil contamination, and soil genesis.^[1–4] Studying how redox conditions change in soils in response to natural and anthropogenic perturbations also provides a way to observe the sensitivity and resilience of soil chemical properties related to electron transfers. Soils are metastable, open systems in a remarkably stable state of non-equilibrium in which electron transfer reactions are controlled by abiotic and biotic processes involving gaseous, aqueous, and colloidal phases.^[5] Interfaces between these phases are centrally important to the energetics and rates of reactions involving redox of soils and natural waters.^[6–8] Many redox reactions in soils involve a phase change for the element transformed, thereby making quantification of the redox status of soils difficult and subject to innovation and new interpretation.^[5] Relatively new understandings of the kinetics of redox reactions in natural waters, particularly as influenced by light, will provide guidance for innovation in the field of redox theory and measurement in soils.^[7,8]

THE NATURE OF THE ELECTRON AND THE PROTON AND THEIR ACTIVITIES IN SOILS

To understand and characterize “electron activity” and oxidation–reduction reactions in heterogeneous, multiphase soils and in soil solutions, an appreciation of the characteristics of electrons and closely allied protons is needed.^[4]

An examination of the complementary nature of electrons and protons affirms the importance of hydrogen (H) ion and electron activities as master variables in soils. The H atom, composed of one proton and one electron, may be visualized and modeled as a spherical puff of cotton candy with a radius of approximately 10 cm and a proton nucleus with a radius of 5 μm —essentially an invisible fleck of unspun sugar in the center! The remaining volume of the atom is occupied by the electron. The density of the spun sugar represents the probability of finding the electron in any one location, and it becomes increasingly thinner (less likely) with distance away from the positively charged proton.^[9] The radius of the H atom (0.3 Å), therefore, is approximately 20,000 times that of the proton (approximately 1.5×10^{-5} Å). The proton also may be visualized as the size of a 0.1 μm colloidal clay particle, compared to a 2000 μm sand grain in soil. In contrast to the large proportion of the volume of the H atom occupied by the negatively charged, wave-like electron, the electron has only negligible mass equal to approximately 550 $\mu\text{g/mol}$, 1/1836 of the mass of the H atom (10^6 $\mu\text{g/mol}$).

In soil chemical calculations and theory, one considers the electron as a “species,” designated “e[−]” with negligible mass and thermodynamically as a ligand, reactant, and product. The electron is not ionic, but it is “negatively charged” as the carrier of negative electricity, as described by J.J. Thompson,^[10] who discovered the electron in 1897. Its “activity” is conceptually analogous to that of H⁺, but its concentration in “mol/L” is undefined. All these caveats about the electron require that one understands that electron activity in soils and natural waters should be regarded as related strictly to energy functions. Such functions can be described simply as “the ability to do work,” “electrochemical potential,” and more colloquially, “electron pressure,” or quantified as a voltage.^[4]

Viewing the sibling concepts of “proton activity” (pH) and “electron activity” (pe) in soils, they must not be considered as twins. Recognition must be given to similarities

and differences in the formulation of conceptual and operational definitions for these key variables, and such comparisons are based on the differences in the nature of the proton and electron, as described above. In both cases, however, thermodynamic activity is defined as the ratio of e^- or H^+ in a given system relative to its activity under standard state conditions. In the case of the e^- , the standard state is at $(H^+)=1$ M, a partial pressure of $H_2 = 1$ atm, and a temperature of 298 K, characteristics of the standard hydrogen electrode (SHE) with 0.00 V under these conditions.^[6,11] For H^+ activity, the reference state is 1 M or pH 0. The measurement of pH and pe as analogous master variables in soils has been classically done with glass and platinum electrodes, respectively. The potential difference between the sensing electrode and a calomel or silver–silver chloride reference electrode is corrected for the potential of the reference electrode relative to the SHE. In this way, an oxidation–reduction potential is converted into a value known as Eh, or the platinum electrode potential relative to the SHE.

The familiar concept of buffer capacity for pH is defined as the change in acid or base added to affect a one unit change in pH.^[6] Similarly, the capacity factor in redox is referred to as poise and is defined as the change in added equivalents of reductant or oxidant to bring about a one unit change in pe (or Eh change of 59.2 mV).^[11] Poise in soils and natural waters has been less intensively studied than pH buffering, but it is a central concept governed by reductant and oxidant activities and by microbial processes. The above discussion of electron and proton activities in soils may be reviewed in more detail in publications.^[4,6]

The U.S. Environmental Protection Agency has developed Mineral Thermal Equilibria (MINTeq), a DOS-based computer program that calculates the distribution of myriad species based on minimizing the Gibbs free energy of a system of multiple chemical equilibria. It incorporates algorithms for redox reactions and allows the estimation of pe, pH, and reductant and oxidant activities in conjunction with dissolution and exchange equilibria. The use of MINTeq also permits an easy way to conduct a sensitivity analysis for estimated pe, pH, or ion activities when just one value for a given parameter is varied.^[12] Other models have been developed and verified for redox-controlled, multispecies systems to predict transport of particular ions.^[13,14]

APPLICATIONS OF REDOX PRINCIPLES IN SOIL ENVIRONMENTS

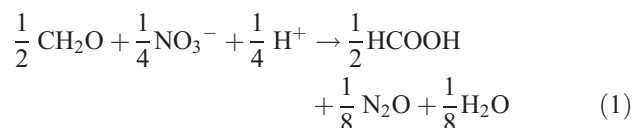
The thermodynamic and theoretical principles surrounding redox reactions have been used to predict and explain soil chemical phenomena in myriad environments and fields, including paddy rice production, wetland delineation, tidal marsh soil development, rhizosphere function, nutrient cycling, heavy metal remediation, synthetic organic compound cleanup, and other areas pertinent to environmental

protection, agricultural production, and ecosystem function. The coupling of Eh–pH diagrams with empirical measurements of pH, Eh, and activities of oxidants and reductants has proved to be a powerful tool for assessing redox processes in heterogeneous, colloidal environments.

Given the importance of paddy rice (*Oryza sativa* L.) in worldwide production of food, early redox work in soil chemistry focused on the reduction reactions coupled with organic matter oxidation in flooded soils of Asia.^[15] This provided the foundation for more practical applications of microbial and chemical redox reactions to understand wetland function and delineation and to study tidal marsh soil development. These two environments have attracted worldwide concerns over human encroachment into them, with controversies over allowable uses of such “wet soils.”

An environmental concern linking soil redox processes and atmospheric chemistry is that emissions of nitrous oxide (N_2O) from waterlogged soils may contribute to global warming as N_2O absorbs infrared radiation, as do carbon dioxide (CO_2), methane, and water in the troposphere. N_2O is an intermediate product in denitrification, the anaerobic microbial respiration process that converts nitrate (NO_3^-) to dinitrogen gas (N_2) as microbes oxidize reduced carbon compounds to organic acids or CO_2 . The energetics of denitrification and its role in releasing N_2O are summarized in the following equations:^[4]

Reduction of NO_3^- to N_2O by the oxidation of carbohydrate to an organic acid:



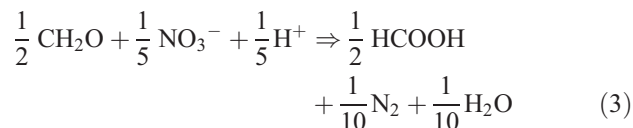
Gibbs free energy (kcal per mole of electrons transferred) equals –21 at pH 7.

Reduction of N_2O to N_2 by the oxidation of carbohydrate to an organic acid:



Gibbs free energy (kcal per mole of electrons transferred) equals –36 at pH 7.

Reduction of NO_3^- to N_2 by the oxidation of carbohydrate to an organic acid:



Gibbs free energy (kcal per mole of electrons transferred) equals –24 at pH 7.

The smaller Gibbs free energy of the reaction that produces N_2O from NO_3^- than for the conversion of N_2O to N_2 indicates that the first step in denitrification is less energetically favorable than is the second. This first step requires

higher electron pressures (more reduced soil conditions or lower Eh values) to effect the reduction, and it may be less kinetically favorable also, even though both reactions are enzymatically catalyzed. Soil conditions, such as pH, Eh, temperature, and carbon availability will affect how much N_2O is emitted from soils to the atmosphere rather than being reduced further to harmless N_2 . The overall reduction of NO_3^- to N_2 is energetically favorable, as indicated in Eq. 3, and is intermediate between the first and second steps in terms of energy released. The combination of thermodynamic calculations and kinetic experimentation can provide essential information to predict the nature of the linkage between soil redox processes and global warming.

Redox reactions in the rhizosphere are also important with respect to nutrient availability in this soil environment immediately adjacent to plant root surfaces (within 1 mm or so). On a broader scale, societal concerns about surface and groundwater chemical contamination and eutrophication of surface waters have focused attention on redox reactions with soil profiles as environments where nutrients, especially nitrogen, may undergo oxidation and reduction reactions leading to release of environmentally sensitive forms of nitrogen. For example, nitrification involves the oxidation of ammonium to NO_3^- , a form of nitrogen that is mobile in most soils and is a common pollutant of ground and surface waters.

Soils contaminated with chromium (Cr)(III, VI) or arsenic (As)(III, V) may undergo transformations between the oxidation states of these elements and thereby change the mobility, toxicity, and potential bioavailability of each element. Hexavalent Cr is a soluble and toxic anion, in contrast to Cr(III), a sparingly soluble, non-toxic cation in most soils. Because of this difference in solubility and toxicity of the common oxidation states of Cr in soil environments, reduction of Cr(VI) is being used to remediate soils without changing the total concentration of Cr.^[16] As(III) oxidation to As(V) results in a decrease in the solubility of As, and thereby may prevent contamination of groundwater, as has occurred in years in Bangladesh.^[17]

Synthetic organic chemicals added to soils by humans in wastes or as pesticides may undergo oxidation reactions that usually result in degradation and a decrease in toxicity. Examples are aromatic compounds used as organic solvents in industrial applications and pesticides, such as herbicides and insecticides, that may be retained in soils or be leached to groundwater, depending on redox reactions.^[18]

CONCLUSION

Oxidation–reduction reactions in soils are governed by the master variables, pH and Eh, and they are controlled kinetically by microbial metabolism and abiotic processes. The heterogeneous, metastable nature of soils makes the study of electron transfers in such environments challenging, but rewarding. Applications of fundamental studies of these

processes are readily made to human concerns pertinent to agriculture, environmental quality, and human health.

REFERENCES

1. Russell, E.W. The chemistry of waterlogged soils. In *Soil Conditions and Plant Growth*, 10th Ed.; Longman: London, 1973; 670–695.
2. Rowell, D.L. Oxidation and reduction. In *The Chemistry of Soil Processes*; Greenland, D.J., Hayes, M.H.B., Eds.; John Wiley & Sons: London, 1981; 401–461.
3. Bartlett, R.J.; James, B.R. Redox chemistry of soils. In *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press: San Diego, 1993; Vol. 50, 151–208.
4. James, B.R.; Bartlett, R.J. Redox phenomena. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; B169–B194.
5. Bartlett, R.J. Characterizing soil redox behavior. In *Soil Physical Chemistry*; Sparks, D.L., Ed.; 2nd Ed.; CRC Press: Boca Raton, 1999; 371–397.
6. Stumm, W.; Morgan, J.J. Oxidation and reduction; equilibria and microbial mediation. In *Aquatic Chemistry: Chemical Equilibria and Rates in Natural Waters*, 3rd Ed.; Wiley-Interface: New York, 1996; 425–515.
7. Stumm, W.; Morgan, J.J. Kinetics of redox processes. In *Aquatic Chemistry: Chemical Equilibria and Rates in Natural Waters*, 3rd Ed.; Wiley-Interface: New York, 1996; 672–725.
8. Stumm, W.; Morgan, J.J. Photochemical processes. In *Aquatic Chemistry: Chemical Equilibria and Rates in Natural Waters*, 3rd Ed.; Wiley-Interface: New York, 1996; 726–759.
9. Castellan, G.W. *Physical Chemistry*, 3rd Ed.; Addison-Wesley Publishing Co: Reading, 1983.
10. Thompson, J.J. *The Electron in Chemistry*; Franklin Institute Press: Philadelphia, 1923.
11. Lindsay, W.L. *Chemical Equilibria in Soils*; Wiley-Interscience: New York, 1979.
12. Allison, J.D.; Brown, D.S. MINTEQA2/PRODEFA2—A geochemical speciation model and interactive processor. In *Chemical Equilibria and Reaction Models*; Loeppart, R.H., Richard, H., Loeppart, A., Schwab, P., Goldberg, S., Eds.; Soil Sci. Soc. Am. Spec. Publ: Madison, 1995; 241–252.
13. LiuChen, W.; Narasimhan, T.N. Redox-controlled multiple-species reactive chemical transport 1. Model development. *Water Resour. Res.* **1989**, 25 (5), 869–882.
14. Liu Chen, W.; Narasimhan, T.N. Redox-controlled multiple-species reactive chemical transport 2. Verification and application. *Water Resour. Res.* **1989**, 25 (5), 883–910.
15. Ponnampertuma, F.N. The chemistry of submerged soils. In *Advances in Agronomy*; Brady, N.C., Ed.; Academic Press: New York, 1972; 29 pp.
16. James, B.R. The challenge of remediating chromium-contaminated soils. *Environ. Sci. Technol.* **1996**, 30, 248A–251A.
17. Bagla, P. India's spreading health crisis draws global arsenic experts. *Science* **1996**, 274, 174–175.
18. Schwarzenbach, R.P.; Gschwend, P.M.; Imboden, D.M. *Environmental Organic Chemistry*; John Wiley and Sons: New York, 1993.

Rehabilitation: Indicators and Monitoring

David Jasper

Centre for Land Rehabilitation, University of Western Australia, Nedlands, Western Australia, Australia

Abstract

The key role of soil microorganisms in processes of nutrient mineralization and acquisition by plants in sustainable ecosystems means that they are a relevant parameter to measure as an indicator of successful rehabilitation. However, it may be concluded from this integration of measurements from bauxite rehabilitation that plant productivity should be included as perhaps the most important indicator of revegetation success. This is not only because of its primary importance to recovery of soil processes, but also because it integrates all aspects of site fertility. Under the protocols for bauxite mine rehabilitation, it can be argued that if diverse and productive vegetation is successfully established, then appropriate populations of soil microorganisms, and the processes to which they contribute, will subsequently develop.

INTRODUCTION

Measuring soil parameters to use as indicators of ecosystem sustainability or to define soil quality is an increasingly common theme in land management. While substantially more research has occurred in relation to agricultural soils,^[1] the principles developed are equally applicable to soils that are being restored after mining. In the agricultural context, Karlen et al.^[2] listed four functions of soil which were integral to soil quality: 1) accommodating water entry; 2) retaining and supplying water to plants; 3) resisting degradation; and 4) supporting plant growth.

These four functions are equally applicable for rehabilitation of soils disturbed during mining, where reestablishment of a sustainable ecosystem requires vegetative diversity and productivity that are appropriate for the desired end use. Plant productivity, which is a key driver of many processes in a sustainable ecosystem, directly reflects the soil conditions that have been created. Therefore, constructing an appropriate soil environment is the crucial first step in rehabilitation.

The information that needs to be acquired in monitoring rehabilitated soils is a combination of measures that define the starting condition of the soil, together with those that may be more sensitive and can be used to demonstrate change over time (Table 1); e.g., total soil carbon (C) can be considered one of the most important indicators of soil quality and productivity,^[3] yet it is slow to respond to changes in management. As a result, microbial biomass and respiration have been used as a more sensitive indicator of long-term trends in organic matter dynamics in soils.^[4]

In general, indicators of successful ecosystem rehabilitation should have some key characteristics as follows:

1. Accommodate all land uses;
2. Describe generic factors and those that add resilience to the ecosystem;
3. Able to be repeatedly measured to indicate direction of change; and
4. Absolute value needs to be comparable with:
 - a. Premining environment (without specific expectations of replicating it), and
 - b. Postmining/prerehabilitation conditions (with the need to demonstrate appropriate development).

DEVELOPING SOIL BIOLOGICAL INDICATORS FOR ASSESSING REHABILITATION AFTER MINING

Key soil biological indicators may include soil microbial biomass and respiration, incidence of symbiotic microorganisms such as mycorrhizal fungi or rhizobia, and soil invertebrates. Soil microbial biomass is the living microbial fraction (bacteria, fungi, protozoa, microfauna, etc.) comprising 1–4% of soil organic matter.^[4] It is responsible for decomposition and mineralization of organic matter, contributes to soil structure development, and is a source and sink of plant nutrients. The soil microbial biomass has potential as an indicator because it is always present at some level and is very responsive to management changes. The relative proportion of soil microbial C to total soil C has been shown to be a sensitive index of ecosystem recovery in land that was disturbed by coal mining and subsequently reforested.^[5] In general, low or decreasing soil microbial biomass implies low or declining plant productivity.^[4]

Table 1 Selected soil properties that are useful indicators of the capacity of a soil to sustain a restored ecosystem.

Physical fertility	Chemical fertility	Biological fertility
Bulk density	pH	Organic C
Soil strength	Electrical conductivity	Microbial biomass and respiration
Aggregate stability	Plant-available and potentially available nutrients	Soil invertebrates
Water infiltration	Heavy metal availability	
Plant-available water		

Source: Adapted from Karlen, Mausbach, et al.,^[1] Karlen, Wollenhaupt, et al.,^[2] Sparling,^[4] Haigh,^[6] and Doran & Zeiss.^[7]

Mycorrhizal fungi assist in nutrient uptake for most plants and therefore will contribute to the sustainability of revegetation. These fungi can be severely reduced by soil disturbance and stockpiling during mining.^[8] The occurrence of mycorrhizal fungi depends on which plant species are present, and this restricts their potential as a generic indicator of rehabilitation success.

Soil invertebrates are responsible for the breakdown of plant residues, which enables subsequent mineralization, and through their activities, they enhance soil structure. Given their role and their widespread occurrence, soil animals do have potential as generic indicators.^[2,9] In contrast, nitrogen gas (N₂)-fixing microsymbionts have limited potential as indicators because they may have a specific host range and survive well without host plants in topsoils disturbed during mining.

All soil biological components and processes are driven ultimately by C inputs derived from plant production. Zak et al.^[10] demonstrated, on a continental scale, that microbial biomass and organic matter pools were positively related to the annual net productivity of vegetation. This relationship has also been demonstrated at the local scale in bauxite rehabilitation.^[11] In the following case study, Sawada has explored the concept that the rate of recovery of biological processes in rehabilitated mine soils is likely to reflect the productivity of the vegetation that has been established.

CASE STUDY OF THE DEVELOPMENT AND APPLICATION OF INDICATORS

In the bauxite mining industry in southwestern Australia, more than 700 ha of native eucalyptus forests are removed, and a similar area is replanted in previously mined areas every year. It is well recognized that the forest topsoil contains important reserves of viable seed, nutrients, and microorganisms; therefore, this soil is always salvaged

prior to mining.^[12] Where possible, the salvaged topsoil is directly returned to a previously mined area or it is stored in stockpiles until a suitable area becomes available. The relatively uniform nature of the soils and landscapes in the bauxite mining area and the consistent rehabilitation strategies that are applied make it feasible to use chronosequences of rehabilitation to estimate the changes in various soil biological components over time.

In addition to a research in nutrient cycling^[13] and soil development,^[12] there has been considerable research on soil organisms in the rehabilitated bauxite mines of southwestern Australia. These organisms are involved in processes that are essential in nutrient supply and uptake including:

1. mineralization of organic matter (soil invertebrates and soil microbial biomass);
2. enhanced uptake of immobile nutrients (mycorrhizal fungi); and
3. supply of nitrogen through N₂-fixation (rhizobia, *Frankia* spp.).

Total C content in surface soils is substantially lower in topsoils spread again after bauxite mining and increases very slowly, even under highly productive revegetation^[11] (Fig. 1). In contrast, soil microbial biomass increases rapidly and substantially from around 2 to 8 years^[14] (Fig. 1). Its rate and magnitude of response

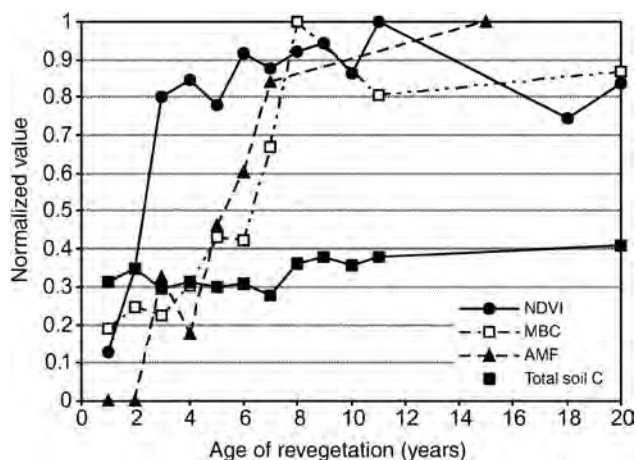


Fig. 1 The pattern of recovery of total soil C, infectivity of arbuscular mycorrhizal fungi (AMF) (Jasper, unpublished data), soil microbial C (MBC), and the NDVI with increasing age of revegetation following bauxite mining. [Total soil C is expressed as a proportion of the total C in premining forest soil (5.6%), while other parameters are expressed as a proportion of the maximum value reached. Values for total soil C and soil microbial C are derived from the same plots, but the other parameters are each from separate studies using different plots, within the same mining operation.]

Source: From Sawada^[11] and Jasper, Sawada, et al.^[14]

parallel those of the vegetation, and these attributes enhance its utility as an indicator.

Vesicular–arbuscular mycorrhizal fungi follow a similar recovery pattern to that of microbial biomass (Fig. 1). However, they are more difficult to measure and, given their constraints described before, are less suitable as an indicator. Although equivalent time sequences are not available for total biomass of soil animals, Greenslade and Majer^[9] found the total number of “decomposer” species of Collembola and the number of individuals, related very strongly to increasing plant and litter cover. Majer and Nichols^[15] reported similar relationships of ant species richness with plant species diversity and vegetative cover, over an equivalent time course to that in Fig. 1, although ant biomass was not measured.

Plant productivity is not easily measured on a large scale in bauxite rehabilitation. Relevant available data are restricted largely to estimates of cover.^[15] As an alternative, Jasper et al.^[14] used remote sensing technology to estimate photosynthetic leaf area in bauxite rehabilitation, using the normalized difference vegetation index (NDVI) (Fig. 1). The pattern of recovery of NDVI was closely related to that derived from direct measures of plant cover.^[14,15] NDVI values increased rapidly with an increasing age of vegetation in rehabilitated bauxite mines, reaching maximum values after 4–6 years. This rate of increase was more rapid than that of soil microbial biomass or mycorrhizal fungi, which by comparison was delayed by 3–4 years.

CONCLUSION

The key role of soil microorganisms in processes of nutrient mineralization and acquisition by plants in sustainable ecosystems means that they are a relevant parameter to measure as an indicator of successful rehabilitation. However, it may be concluded from this integration of measurements from bauxite rehabilitation that plant productivity should be included as perhaps the most important indicator of revegetation success. This is not only because of its primary importance to recovery of soil processes, but also because it integrates all aspects of site fertility. Under the protocols for bauxite mine rehabilitation, it can be argued that if diverse and productive vegetation is successfully established, then appropriate populations of soil microorganisms, and the processes to which they contribute, will subsequently develop.

The concept of using plant productivity as an integrated measure of rehabilitation performance is mirrored in a research in the Australian mining industry, which is investigating the application of “landscape function analysis” (LFA)^[16] as a tool for assessing mine rehabilitation. In the LFA approach, the capacity of the landscape to retain essential resources of water and nutrients is assessed. This retention capacity reflects both the underlying

characteristics of the constructed landform and the development of plant biomass and soil cover. Thus, it is an integrative measure of plant performance and the factors that affect the performance.

The importance of plant productivity, and its dependence on site fertility, reaffirms the importance of constructing a soil profile with physical and chemical characteristics that are appropriate for the particular vegetation to be established. If the right components of physical and chemical fertility are put in place, together with microbial and plant propagules, the key biological components, then the likelihood of successful rehabilitation is enhanced. In large-scale reestablishment of native ecosystems, such as in rehabilitation after bauxite mining, direct-returned fresh topsoil is the best source of the propagules.

ACKNOWLEDGMENT

The data presented in this entry were derived from studies that were generously supported by Alcoa World Alumina—Australia.

REFERENCES

1. Karlen, D.L.; Mausbach, M.J.; Doran, J.W.; Cline, R.G.; Harris, R.F.; Schuman, G.E. Soil quality: A concept, definition, and framework for evaluation. *Soil Sci. Soc. Am. J.* **1997**, *61*, 4–10.
2. Karlen, D.L.; Wollenhaupt, N.C.; Erbach, D.C.; Berry, E.C.; Swan, J.B.; Eash, N.S.; Jordahl, J.L. Crop residue effects on soil quality following 10-years of no-till Corn. *Soil Till. Res.* **1994**, *31*, 149–167.
3. Reeves, D.W. The role of soil organic matter in maintaining soil quality in continuous cropping systems. *Soil Till. Res.* **1997**, *43*, 131–167.
4. Sparling, G.P. Soil microbial biomass, activity and nutrient cycling as indicators of soil health. In *Biological Indicators of Soil Health*; Pankhurst, C.E., Doube, B.M., Gupta, V.V.S. R., Eds.; CAB International: Wallingford, 1997; 97–119.
5. Insam, H.; Domsch, K.H. Relationship between soil organic carbon and microbial biomass on chronosequences of reclamation sites. *Microb. Ecol.* **1988**, *15*, 177–188.
6. Haigh, M.J. Soil quality standards for reclaimed coal-mine disturbed lands: A discussion paper. *Int. J. Surf. Min. Reclam. Environ.* **1995**, *9*, 187–202.
7. Doran, J.W.; Zeiss, M.R. Soil health and sustainability: Managing the biotic component of soil quality. *Appl. Soil Ecol.* **2000**, *15*, 3–11.
8. Jasper, D.A.; Abbott, L.K.; Robson, A.D. The loss of VA mycorrhizal infectivity during bauxite mining may limit the growth of *Acacia Pulchella* R.Br. *Aust. J. Bot.* **1989**, *37*, 33–42.
9. Greenslade, P.; Majer, J.D. Recolonization by Collembola of rehabilitated bauxite mines in Western Australia. *Aust. J. Ecol.* **1993**, *18*, 385–394.

10. Zak, D.R.; Tilman, D.; Parmenter, R.R.; Rice, C.W.; Fisher, F.M.; Vose, J.; Milchunas, D.; Martin, C.W. Plant production and soil microorganisms in late-successional ecosystems: A continental-scale study. *Ecology* **1994**, *75* (8), 2333–2347.
11. Sawada, Y. *Indices of Microbial Biomass and Activity to Assess Minesite Rehabilitation*. In Proceedings Minerals Council of Australia 21st Annual Environmental Workshop, Newcastle, Australia, Oct 14–18, 1996; Minerals Council of Australia: Canberra, 1996; 223–236.
12. Ward, S.C. Soil development on rehabilitated bauxite mines in south-west Australia. *Aust. J. Soil Res.* **2000**, *38*, 453–464.
13. Todd, M.C.L.; Adams, M.A.; Grierson, P.F. Mineralisation of nitrogen in a chronosequence of rehabilitated bauxite mines. *Aust. J. Soil Res.* **2000**, *38*, 435–451.
14. Jasper, D.A.; Sawada, Y.; Gaunt, E.; Ward, S.C. Indicators of reclamation success—Recovery patterns of soil biological activity compared to remote sensing of vegetation. In *Land Reclamation: Achieving Sustainable Benefits*; Fox, H.R., Moore, H.M., McIntosh, A.D., Eds.; A.A. Balkema: Rotterdam, 1998; 21–24.
15. Majer, J.D.; Nichols, O.G. Long-term recolonization patterns of ants in western Australian rehabilitated bauxite mines with reference to their use as indicators of restoration success. *J. Appl. Ecol.* **1998**, *35*, 161–182.
16. Ludwig, J.; Tongway, D.; Freudenberger, D.; Noble, J.; Hodgkinson, K. *Landscape Ecology, Function and Management: Principles from Australia's Rangelands*; CSIRO: Melbourne, 1997.

Religious Aspects of Soil—Human Relationships

Verena Winiwarter

Centre for Environmental History, Institute of Social Ecology, Interdisciplinary Research and Continuing Education (IFF), Vienna, Austria

Winfried E.H. Blum

Institute of Soil Research, University of Natural Resources and Applied Life Sciences, Vienna, Austria

Abstract

The relationship between soil and humans has been an important part of religious rites and beliefs. The earth, as the basis for human life, is present in cosmogonies as mother earth and as an “element,” a basic constituent of natural bodies and humans alike. Ancient Greek and Roman, Chinese, Buddhist, Hindu, Inca, and the Abrahamic religious systems emphasize the spiritual aspects of human interventions (such as plowing) into soils. Worshipping the soil and a ritual framing of the central agricultural operation, plowing, are common in primal religions. While the Chinese are the only ones who developed a soil altar, the earth as the basis for all life is acknowledged in almost all primal belief systems. Studying the religious aspects of human relations to soils does allow us to appreciate the multifaceted ways of how societies dealt with the ambiguity of being human, being at once a part of and apart from nature.

INTRODUCTION

Since the advent of agriculture, dealing with the soil was the central pursuit of humans. Agriculture was as much a way of living as the mainstay of economy in the agricultural civilizations. The relationship between soil and humans was central to both individual subsistence and social life and was also an important part of religious rites and beliefs.^[1]

There are many, scholarly disputed definitions of “religion.” All religions contain three basic elements: systems of belief, ways of life, and important shapers of identity.^[2] According to Gottlieb, religions are “organized and overlapping systems of belief, ritual, institutional life, spiritual aspiration, and ethical orientation which are premised on an understanding of human beings as other or more than simply their purely social or physical identities. Religions also provide guidance that seeks to root everyday moral teachings in the ultimate nature or significance of a spiritual truth about who we really are. Religions therefore direct us toward particular ways of living with other people and with the world. Finally, religions provide rituals—acts of prayer, meditation, collective contrition, or celebration—whose goal is to awaken and reinforce an immediate and personal sense of our connection to the Sacred.”^[3]

When dealing with religious aspects of human relations to soils, two major kinds of religions need to be distinguished. Natural religions are those which embrace

all members of the society in question; taking part is an obligation. All small societies have such a religion; its main features are a myth of origin, care for the ancestors, rites of transition, and calendar rituals. This kind of religion is called “primal religion.” The expression “primal” means that historically it has a certain priority and that the basic aspects of religion are found within it. “Salvation religion,” “critical religion,” or “guidance religion” come under the second major type of religion. These religions are founded by religious leaders who offer a distinctive message against the usual assumptions of the society in which they live. Most major religions of the world belong to this type. During the complex history of humankind, Buddhism, Christianity, and Islam established themselves as major religions. As a result, they adapted themselves to various social and political pressures and adopted some of the functions of primal religions.

Over the long run of human history, when primal religions were dominant, soils were conceptualized as part of a world that was controlled by higher, unseen powers. Thus, soils were considered to bear immaterial qualities; they were—to some extent—objects of religious reverence and corresponding ritual practices to ensure their sustained fertility existed. Before giving an overview of religious concepts worldwide, a word of caution is needed. Comparative approaches to religions are fraught with difficulty. Translations are particularly problematic with transcendental concepts.

To discuss religions completely, we would need to consider their conceptual, behavioral, social, and subjective aspects. This entry focuses on conceptual aspects, discussing the earth's role in cosmogonies and the ritual aspects of dealing with the soil.

EARTH AS PART OF BELIEF SYSTEMS

The earth, as all-encompassing basis for human life, is presented in cosmogonies in two different forms: 1) as mother earth, deified as a numinous deity, with her body and organs as the different parts of nature and 2) as an "element," which is considered as a basic constituent of natural bodies and humans alike. The cult of the "mother earth goddess" is not directly a cult of the soil, and creation myths do not refer to the soil. Tillers of the soil come late in many cosmogonies, as a second inhabitant living on the earth.

Nevertheless, the earth plays a central role in most religions. For example, the Greek philosophy of nature knew four elements: earth, water, air, and fire. Its Chinese counterpart has five constituents: metal, wood, and again, earth, water, and fire. The Buddhist mahabhuds are water, soil, fire, air, and space. The Sanskrit word "*dharat*" is like "*bhumi*" a word for "soil," which is, at least in the Atharvaveda (XII.I), the deified subject of a hymn praising it as ruler and mother. The hymn mentions different types of soil. Another word for earth, "*prithvi*" and "*vasundhra*" are the most common names of the Hindu earth goddess and also one name of the element of earth in the fivefold cosmogony of Hinduism, comprising of earth, water, fire, air/wind, and ether.

In Greek cosmogony, the first beings are the giants. Gaia (Gea), the earth as goddess, according to Greek mythology the only being that emerged from primeval Chaos, Poseidon, and the Greek god of the sea, are the parents of the Giant Antaios. Antaios challenged all travelers passing his realm to a wrestling match, which he always won. He was slain by Herakles because the hero lifted the giant off the ground. Whenever he had fallen to the ground in battle before, touching the earth had regenerated his powers. The earth as embodied in the giant's mother Gaia plays a powerful role in this myth.

The concept of earth's healing powers has been preserved in European folklore over the millennia. In German folk belief, newborn babies were ritually placed on the earth to gain strength, as well as sick people, for the same reason. Magicians and sorcerers, much like Antaios, were believed to be able to escape when they touched the earth, and after capturing them, they had to be suspended above the ground. Swallowing a piece of earth was believed to heal hiccups, and after a snake bite, the affected limb was to be put into fresh earth.

Earth deities can be found in many religions. A Germanic Goddess called Nerthus was considered similar to

Tellus, or Terra mater, the Roman Gaia-analogue, by the Roman author Tacitus. These goddesses are most often chthonian (belonging to the underworld), and not fully, if at all, personalized. They are seen as primeval in many mythologies and embody the universal truth that all life comes from earth and reverts to earth after its death.

Place-based deities, owners of the space of a people, are also common in many religions. They are responsible for fertility, but their connection to soil is indirect. They are associated with the fruit of the land rather than with the soil as such. The Roman *Penates* and *Lares* can be subsumed under the category of place-based deities. They are both part of ancestor worship and a place-based (House of God) idea of bringing fertility and luck. There are shrines to ancestors and the spirits of a given home in many belief systems, but their relation to soil is an indirect one.

THE SPIRITUAL ASPECT OF HUMAN INTERVENTIONS INTO SOILS

One of the oldest European works with reference to agricultural practices, Hesiod's "Works and Days," is an example for the role of soils, or even more precise, of tilling the soil, in a primal religion's worldview, after the change from the chthonian deities Gaia and Chronos to the Olympic pantheon. Hesiod's poetic program is based on an immanent god. The divine will is not discovered by reading sacred or philosophical texts but rather in the day-to-day world in which he lives. Zeus' will is manifested in the life of the people. What succeeds in the world is what Zeus has prescribed for humans. Farming, which is the necessary way in which humanity procures food, is, therefore, necessarily also the will of Zeus. Farming knowledge in such a belief system is knowledge necessary to perform the will of the divine beings and has, therefore, a sacred dimension. Hesiod does not describe soil altars or rituals to it. Human life conduct is the main vehicle of worship. Tilling the soil well is a prerequisite for spiritual fulfillment. The various deities governing the conduct of agriculture in Greek and Roman mythology are part of the system Hesiod describes, a system in which the conduct of agriculture as the main worship to the Gods was also perceived as being at their mercy. The Greek/Roman Ceres/Demeter is one example for a powerful divine representation. The pantheon around agricultural operations was much differentiated in ancient Rome. Agricultural deities, while connected to the soil, are different from earth goddesses as they do not symbolize the entire universe or the earth, and they are also different in that they are most closely connected to the harvest and not to the fertility of the soil as such.

Many primal religions harbor similar concepts, in some cases, especially with Nomadic people, they can be of

opposite character: Touching the soil with an implement can be forbidden, and a spiritually fulfilling life requires not to touch the soil for planting.

As in other agrarian civilizations, a system of “elements,” of primary matter, was developed in Hellenic philosophy. Anaxagoras (ca. 500 B.C.E.) is said to have been the first to develop the scheme of the four qualities, namely hot, cold, wet, and dry, from which the qualities of the four elements are derived which lie at the very heart of Greek nature concepts. In Fragment 4, these four opposites were specifically mentioned, along with the bright and the dark, “much earth” and “innumerable seeds” as part of the primeval mixture, when all things were together. Earth does have a special place in this cosmogony. Empedocles, 4th century B.C.E., gives the earth none of the four qualities but rather identifies it with solidity, not unlike the earlier Anaximenes, who relates hot and cold to differences in density (rare/dense). Both Anaximenes and Empedocles describe qualities of the earth that are relevant to agriculture (density and solidity), rather than trying to abstract from it completely. Only their later readers and commentators rendered earth as one of the four abstract elements, which are the basis of later philosophy. This allows us a glimpse of how an abstract worldview such as Greek natural philosophy was developed on the basis of an earlier, perhaps more practical relation with the earth.

In ancient China, during the Sui and T’ang dynasties, starting from the late 6th century, a ritual honoring the earth was performed by the emperor on the day of the summer solstice. Other rituals were performed to honor the holy soil of China (Shên Tsou) or the Great Earth Deity (Ta-sheh).^[4] Later on, local soil altars were integrated into the greater belief system of Daoism. The best known part of this worship, and a major tourist attraction in Beijing, is the Altar of the Earth and Heavens in Sun-Yat-Sen Park, dating back to the 14th century. In it, hard-packed earth in five colors, yellow in the middle, green to the east, white to the west, red to the south, and black to the north is at the center of worship.

Plowing is the most significant agricultural operation, the one that engages most deeply with the soil. We know of ritual plowings in many religions and cultures. For ancient Greece, three locations of ritual plowing are known, one in Skiron, second one close to Eleusis, and the third one under the Akropolis in Athens. These have to be considered as a part of rites for Demeter, the goddess of fertility.^[5] During the Sukhothai Kingdom (13–15th century C.E.) in Thailand, a royal plowing ceremony was first recorded. After a lapse of 40 years, the Thai kings reinstituted it in 1960, and it is held every year in May. It is similar to many other plowing rites: A pair of sacred oxen is hitched to a wooden plow, opening a furrow on ceremonial ground.

The corresponding Inca rites are described in a report by a Spanish-Inca writer, Garcilaso de la Vega. According to him, a terrace in Cusco was tilled exclusively by those of royal blood (the Inca). Of special celebrations, songs

were based on the word *hailli*, which means triumph over the soil, which they ploughed and disemboweled so that it should give fruit (Garcilaso de la Vega 1989 (1609): 244, quoted after Bauer).^[6] It is important to point out that in the Inca tradition, agriculture and warfare were closely related ritually, and the furrowing was described in martial terms.

While primal religions almost universally ritualize the act of plowing or more generally, tilling, the Abrahamic religions, although they make frequent allusions to soils, do so only metaphorically. The Bible does not invoke any worship of the soil, as it—like all other Abrahamic religions—is directed against a numinous world view. This is not to say that there is no ecological awareness in these works. Much is yet to be learned about the European and also about the Jewish tradition. Of particular interest as an ecological concept is the Sabbat, the 7th day on which nothing in nature must be changed, a day of rest for nature, and the 7th year, in which fields shall rest. The etymological connection between the Hebrew words adam (man, in particular Adam, the first human in the Old Testament) and adamah (soil) has been discussed widely and controversially. Jewish Environmentalists drew on the connection to argue for a Jewish eco-theology, and a wealth of books on such issues has been published in the last years. The Judeo-Christian tradition is abundant with references to soils. The Bible contains multiple references to soil and its fertility, such as the Old Testament’s Psalm 65, 10, or Job 31, 38–40, in which the barrenness of the earth is invoked at the very end of Job’s plea, as the last and ultimate imaginable catastrophe. The interaction of the soil with seed is used as a metaphor to describe human souls and their ability to receive the word of the Lord. As an example from the New Testament, Mark 4, 3–9 shall be noted, where different soils are described on which the seed falls, with different results.

CONCLUSION

Worshipping the soil and a ritual framing of the central agricultural operation, plowing, are common in primal religions, and while the Chinese are the only ones who developed a soil altar, the earth as the basis for all life is acknowledged in almost all primal belief systems. In the cases of earth goddesses, the principle of fecundity was the focus of worship. They stood for all nature, for the rejuvenation of plant and animal life in the cycle of the seasons. Earth goddesses were no soil goddesses but embodied nature goddesses. Place-based deities are no soil gods either; again, soil fertility was conceptualized as part of the qualities of a particular place. The worship of the soil did not keep people from destroying it, unintentionally, through their efforts to procure a living from the soil, and there is no direct connection between the belief system and human actions. Studying the religious

aspects of human relations to soils does allow us, however, to appreciate the multifaceted ways how societies dealt with the ambiguity of being human, being at once a part of and apart from nature, a question that has not seized to be of importance.

ACKNOWLEDGMENT

This entry is based mainly on a longer discussion of the subject by the authors.^[1]

REFERENCES

1. Winiwarter, V.; Blum, W.E.H. Souls and soils: A survey of worldviews. In *Footprints in the Soil: People and Ideas in Soil History*; Warkentin, B.P., Ed.; Elsevier: Amsterdam, 2006; 107–122.
2. Gottlieb, R. *This Sacred Earth: Religion, Nature, Environment*; Routledge: London, 1996.
3. Gunn, T.J. The complexity of religion and the definition of “religion” in international law. *Harv. Hum. Rights J.* **2003**, *16*, 189–215.
4. Eichhorn, W. *Die Alte Chinesische Religion und das Staatskultwesen*; Handbuch der Orientalistik: Abt. 4, China: Bd. 4, Religionen und Brauchtum; Abschn. 1, 1976.
5. Kledt, A. Die Entführung Kores. Studien zur athenisch-eleusinischen demeterreligion. In *Palingenesia, Schriftenreihe Für Klassische Altertumswissenschaft*; Begründet von Stark, R., Nach Lendle, O., Steinmetz, P., Koster, S., Eds.; Steiner: Stuttgart, 2004; 84 pp.
6. Bauer, B.S. Legitimization of the state in Inca myth and ritual source. *Am. Anthropol.* **1996**, *98* (2), 327–337.

Rendzina

Thomas E. Fenton

Soil Morphology and Genesis, Agronomy Department, Iowa State University, Ames, Iowa, U.S.A.

Abstract

The term Rendzina has a long history of use to describe shallow soils that are calcareous and are underlain by parent materials high in carbonates. Based on the classification systems, it appears that the term Rendzina will be used less frequently in the future. However, the use of the “rend” root of the word will continue at least in Soil Taxonomy and the Food and Agriculture Organization–United Nations Educational, Scientific and Cultural Organization soil classification systems.

INTRODUCTION

Rendzina is a term that has a long history of use on a worldwide basis. In soil classification, it has always included soils high in carbonates. Most countries, except the United States, have used the name to describe shallow to very shallow soils, gray to black in color that developed in limestone or chalk. Early usage in the United States differed in that. Rendzinas included deeper, dark soils with profiles ranging from 38 to 152 cm in thickness, with many, but not all containing carbonates.^[1]

The origin of the word “Rendzina” has a complex history. Simonson^[2] described the original use of Rendzina as a folk term used to convey the clinking sound of a plow being drawn through certain stony soils. In the late 19th century, Polish scientists began using the name for soils formed in highly calcareous materials. Part of Poland was then in the Russian Empire, and the name was used in some of the early Russian soil classifications.

HISTORY OF USE

De'Sigmond^[3] described the conditions needed for the development of Rendzinas as parent rock rich in calcium carbonate or gypsum that has the fineness of structure and solubility necessary to neutralize the podzolizing effect of the humid climate under which they developed the forest vegetation. He considered these soils to be azonal, i.e., they did not have completely developed profiles.

Robinson^[4] described the Rendzinas of Central Europe as humus-carbonate soils. They were typically dark colored soils with 3–12% organic matter and varying amounts of free calcium carbonate. In the Russian classification system proposed by Glinka,^[5] Rendzinas were considered to be typical endodynamomorphic soils, which had not yet reached mature development, and in which the profile character was mainly determined by the nature

of the parent material. Glinka^[5] observed that these humus-carbonate soils usually contained considerable undecomposed rock, the amount of which decreased in soil development. All the Rendzina soils described by Glinka^[5] are shallow soils similar to those described by De'Sigmond.^[3] As described by Glinka,^[5] the A horizon was 15–30 cm in thickness with varying amounts of limestone fragments and graded through a thin transition of lighter-colored material into fragmental limestone at about 46 cm or less. Spanish red soils regarded as terra rossa were thought to be formed by the degradation of Rendzinas, consequent on deforestation under semiarid conditions. The placement of Rendzinas as endodynamomorphic soils was later criticized by Marbut^[6] because he thought Glinka's^[5] groupings were poorly balanced, in that they represented such a small part of the total soils of the world as compared to the more fully developed soils. Rode^[7] used the term “turfy-carbonate” soils as a synonym for Rendzina for Russian soils within Podzol zones. The presence of the high carbonate content in the parent rock resulted in a high content of calcium carbonate humus and was thought to have prevented these soils from becoming podzolized.

Marbut^[8] in his final classification system included Rendzinas in Category III which he defined as those soils in which extreme youth, excessive erosion, wetness, and excessive contents of carbonates, salt, or alkali prevented the development of normal soil. Rendzinas were the soils with excessive contents of carbonates.

Baldwin et al. retained the Rendzinas as intrazonal soils in their highest category order, calomorph in the suborder level, and named Rendzina as a great soil group. A general description of the soils included as Rendzinas in this classification system was an intrazonal group of soils, usually with brown or black friable surface horizons, underlain by light-gray or pale-yellow soft calcareous material, developed under grass vegetation or mixed grass and forest vegetation, in humid and semiarid regions from relatively soft, highly calcareous parent material.^[9]

After 12 years of the publication of the abovementioned soil classification system, Oakes and Thorp^[1] suggested that some of the soils included as Rendzinas, such as Houston Black, were markedly different from the Rendzinas of Europe. The included soils they objected to, were high in clay, generally dark in color, free of stones, and were deep soils. Many but not all were calcareous. The common property was shrink–swell that resulted in churning of the soil. They proposed that these soils be reclassified within a great soil group named Grumusols and developed a tentative definition. This proposal gradually gained acceptance both in the United States and other countries. The acceptance of the grumusol concept substantially reduced Rendzina soils to a minor group in the United States. Oakes and Thorp^[1] observed that the Lithosols of the prairies of the Southern United States conformed more closely to the Rendzina of other countries rather than the medium depth and deep soils that had been included with rendzinas in the United States.

Papadakis^[10] characterized the Rendzina group as those soils that contained active carbonates and gave effervescence with hydrogen chloride in all their horizons. He attributed the deep, dark accumulation of humus to the carbonate environment. As climate became drier, the accumulations of humus were less and desert Rendzinas (serozems) lack humus.

Bridges^[11] discussed Rendzinas in the category of intrazonal, calcimorphic soils. He described rendzinas in the following way: a shallow soil rich in organic matter and biological activity. It has a stable crumb structure and is dark in color. It is a relatively simple soil with an A horizon directly overlying a C horizon, which is the limestone parent material. The humus is well incorporated in the mull form and micromorphological evidence showed that these soils are largely composed of the fecal pellets of soil arthropods and the casts of earthworms.

PRESENT USE

A part of the word Rendzina is used in the U.S. soil classification system—Soil Taxonomy.^[12] The “rend” part of the word was used to coin a new term for a suborder in the Mollisol Order—Rendolls. The term was used to convey the concept of a high level of carbonates in these soils. Many of the soils previously called Rendzinas are included within the Mollisol Order and the Rendoll Suborder in Soil Taxonomy. Rendolls have mollic epipedons less than 50 cm in thickness that are underlain by calcareous parent material or a cambic horizon with a high carbonate content. The parent material is limestone, chalk, drift composed mainly of limestone, or shell bars. They occur in humid climates (udic moisture regimes) or cold climates with temperatures lower than 8°C (cryic soil temperature regime) or both and have a high calcium carbonate content (calcium equivalent of 40% or more) in the mollic epipedon or in the underlying mineral material. The native vegetation is thought to be forest or grass and shrubs. They are

not extensive in the United States but on a worldwide basis occupy approximately 0.2% of the land area.

Rendzinas were recognized in the Food and Agriculture Organization–United Nations Educational, Scientific and Cultural Organization (FAO–UNESCO) system for soil shallow to limestone prior to 1988.^[13] However, this term was dropped in the 1988 revision.^[14] These soils were then included with the Leptosols or thin shallow soils. The term rendzic is an adjective in the FAO–UNESCO legend having been added in 1988 and is used for naming map units at the second level.

CONCLUSION

The term Rendzina has a long history of use to describe shallow soils that are calcareous and are underlain by parent materials high in carbonates. Based on the classification systems, it appears that the term Rendzina will be used less frequently in the future. However, the use of the “Rend” root of the word will continue at least in Soil Taxonomy and the FAO–UNESCO soil classification systems.

REFERENCES

- Oakes, H.; Thorp, J. Dark-clay soils of warm regions variously called rendzina, black cotton soils, regur and tirs. *Soil Sci. Soc. Am. Proc.* **1950**, *15*, 347–354.
- Simonson, R.W. Rendzina—origin, history, and descendants. *Soil Surv. Horiz.* **1997**, *38* (3), 71–92.
- De'Sigmond, A.A.J. *The Principles of Soil Science*; Translation by Yolland, A.B. Thomas Murby: London, 1936; 236–244.
- Robinson, G.W. *Soils—Their Origin, Constitution, and Classification*; Wiley: New York, 1949; 373 pp.
- Glinka, K.D. *The Great Soil Groups of the World and Their Development*. Translated from the German by C.F. Marbut; USDA, 1927; 146–149.
- Marbut, C.F. Soils: Their genesis and classification. *Soil Sci. Soc. Am.* **1951**, *134*, Lectures Given in the Graduate School of USDA, 1928.
- Rode, A.A. *Soil Science*; Translated from Russian by Israel Program for Scientific Translations, Jerusalem; National Science Foundation: Washington, DC, 1962; 517 pp.
- Marbut, C.F. Soils of the United States. In *Atlas of American Agriculture*; USDA: Washington, 1935.
- Baldwin, M.; Kellogg, C.E.; Thorp, J. *Soils and Men: USDA Yearbook of Agriculture*; U.S. Government Printing Office: Washington, DC, 1938; 1232 pp.
- Papadakis, J. *Soils of the World*; Av. Cordoba: Buenos Aires, 1964; 141 pp.
- Bridges, E.M. *World Soils*; Cambridge University Press: Cambridge, 1970; 89 pp.
- Soil Survey Staff. *Soil Taxonomy*; 2nd Ed.; NRCS USDA, U.S. Govt. Printing Office: Washington, 1999; 869 pp.
- Food and Agriculture Organization. *FAO–UNESCO Soil Map of the World*; UNESCO: Paris, 1971–1981; Vol. I–X.
- Food and Agriculture Organization. *FAO–UNESCO Soil Map of the World: Revised Legend*; FAO: Rome, 1988.

Resilience

Moses M. Tenywa

J.G.M. Majaliwa

Department of Soil Science, University of Makerere, Kampala, Uganda

A. Lufafa

Makerere University, Kampala, Uganda

Abstract

The soil resilience concept is relatively new and has emerged alongside the efforts to define and measure sustainability of agricultural systems. Various efforts to define soil resilience can broadly be grouped into three categories: as a process, nature, and soil state. Soil resilience as a process focuses on the mechanical role of soil in countering degradative processes. The nature concept is defined as the ability of the soil system to remain productive through efficient processing over an extended period of time. The state concept views soil resilience as a lumped, dynamic macroscopic parameter that is a consequence of fine-scale interactions between properties, processes, microclimate, and management impacts within the soil body.

INTRODUCTION

The soil resilience concept is relatively new^[1,2] and has emerged alongside the efforts to define and measure sustainability of agricultural systems.^[3] It describes the level of resistance to forces driving soil degradation, and provides insight into mechanisms that endow soils with favorable properties and beneficial processes from physical, chemical, and biological perspectives. Like sustainability, which has remained a difficult concept to elucidate, the soil resilience concept is inexactly expressed^[4,5] and quantified, yet it is recognized as the most relevant parameter for understanding and management of soil ecosystems.^[6] Various efforts to define soil resilience can broadly be grouped into three categories: as a process, nature, and soil state.

A PROCESS

Soil resilience as a process focuses on the mechanical role of soil in countering degradative processes (e.g., rainsplash and detachment). It refers to the ability of the soil to resist or recover when subjected to degradative forces. This definition is primarily inspired by the Hooke's spring elasticity model. Using this model, a soil resilience coefficient was proposed as follows:^[3]

$$S_r = -2A/x^2 \quad (1)$$

where S_r is the resilience coefficient whose inverse indicates vulnerability to degradation; A is the amount of work required to move soil from one state to another (e.g., 1–2), and x is the variable reflecting changes in soil state.

A resilient soil, therefore, exhibits a higher S_r and minimal change in soil state (x) as stress accumulates. Although the analogy is satisfactory when used to elaborate resilience of the physical soil system, its utility is limited on five major premises:

1. Lack of means to translate the chemical, biological, and other forms of work into “unified mechanical form.”
2. The absence of a lumped parameter adequate in capturing “wholly” the changes in soil state.
3. Assumes classical resilience to be characteristically independent of stability. Yet for soil systems, the two may not necessarily be mutually exclusive, in that alterations in resilience from equilibrium may result in deterioration of soil structural attributes.
4. Eq. 1 holds if the alterations are slight and temporary. Otherwise, if the soil system undergoes irreversible changes and the degree or duration of perturbations is drastic and intense, the relationship ceases to be valid.^[7]
5. The model describes the macroscopic behavior of the soil matrix and therefore does not give a complete picture of what is happening at finer levels of soil organization.

A NATURE

This school of thought focuses on the functional integrity of the soil judged, based on its properties (e.g., soil formation stage, degree, and level of use/management), and

is the most common approach found in the work done by Lal.^[7] Soil resilience from this perspective is defined as the ability of the soil system to remain productive through efficient processing over an extended period of time.^[4,8] Accordingly, it refers to the longevity of the soil resource and its health measured in terms of overall performance and quality.^[9] The mass flux balance approach^[4] is typically used to express the functional integrity of the soil in the form of a soil resilience factor:

$$S_r = S_a + (S_n - S_d + I_m)dt \quad (2)$$

where S_r is soil resilience, S_a is the antecedent soil condition, S_n is the rate of new soil formation, S_d is the rate of soil degradation/depletion, and I_m connotes the management inputs from outside the ecosystem.

The major limitation of this approach lies in its failure to account for the impact of restorative processes (buffering capacities of soil) on the prevailing soil properties, that is, focuses on the “what” of the antecedent state has changed as a result of the interaction of soil formation, degradation, and management, but overlooks the mechanisms of change—the “how.”

A STATE

The state concept views soil resilience as a lumped, dynamic macroscopic parameter that is a consequence of fine-scale interactions between properties, processes, microclimate, and management impacts within the soil body.^[10] Difficulties exist in the development of a single indicator of soil resilience because the soil consists of at least three “states,” its physical, chemical, and biological properties (each of which consists of numerous subproperties), which interact in a complex manner but are quantified through extremely different procedures. Seen as a combined effect of both soil-buffering capacities (physical, chemical, and biological) and soil mass flux balance because of pedological and anthropological actions, a resilience factor as described by the following equation was suggested:^[3]

$$S_r = BC_{ph} + BC_{ch} + BC_b + (r_p + r_a)dt \quad (3)$$

where S_r is soil resilience, BC_{ph} is physical buffering, BC_{ch} is chemical buffering, BC_b is biological buffering, r_p is rate of pedological soil fluxes, r_a is rate of anthropological fluxes, and dt is change over time.

Although this approach is process based, its major limitations emanate from the assumption that the named different soil “states” are independent, ignoring their known interactions.^[8] Nonetheless, this “state” approach to resilience combines the three fundamentally different components of the soil, and as such is worthy of attention by those who seek to quantify soil impacts in a holistic manner.

With respect to agronomy, soil productivity is a crucial indicator of soil resilience. Although soil productivity also remains difficult to quantify,^[11,4] crop yield over time is a practical measure of productivity.^[11,12,13] Long-term crop performance under specific environmental and management conditions is mainly dependent upon relevant soil depth (RSD) characteristics.^[14,15] Assuming an infinitesimal pedon, a productivity index D_p of specific productivity level P ranging from P_{min} to P_{max} yield can be defined in a manner similar to the degree of polarization^[16,17] by the use of statistical mechanics on an RSD matrix, as:

$$D_p = 2(P - P_c)/g \quad (4)$$

where $g = (P_{max} - P_{min})$ is the maximum agroecological productivity gap, and $(P - P_c)$ is the gap between specific productivity level P and the critical productivity level $P_c = (P_{max} + P_{min})/2$ of the agroecological zone. D_p ranges from -1 for highly degraded (HD) land to $+1$ for highly productive (HP) land, insinuating the existence of three resilience phases: namely, high resilience (associated with HP), marginal resilience (associated with HD), and diminishing resilience (transitional phase).^[5]

The major advantage of using D_p as an index of soil resilience is that it is a macroscopic state parameter derived from microscopic properties. Considering that the soil can exhibit more than one accessible state (configuration of pedon of specific parameters of state; e.g., temperature, energy, chemical potential, volume, and pressure) at different equilibria, D_p provides an excellent opportunity for capturing all of them.^[5]

Finally, it is noteworthy that a similar index ($D_p = P/P_{max}$) for crop productivity is used in land evaluation and suitability assessment,^[18] and Eq. 1 can be applied near the limit of the HP macrostate. However, a great challenge remains to establish a state parameter for non-agricultural soils.

FACTORS AFFECTING SOIL RESILIENCE

The factors that affect soil resilience fall in two broad groups, namely, endogenous and exogenous.^[4] Endogenous factors are related to the inherent soil properties (e.g., soil depth, relief, drainage status, structure, texture, soil organic matter, humus quality, faunal and floral activities, and nutrient level) and climate especially precipitation.^[19] Exogenous factors are largely a result of land use and management. Good management aims at providing a buffer against sudden fluctuations in the system, minimizing degradative effects of atmospheric agents (e.g., heat, wind, rain, and runoff), enhancing restorative mechanisms, and providing adequate protection to soil against degradation. Several methods of management exist that impact resilience factors: agricultural operations

(e.g., irrigation, tillage, drainage, terracing, ridging, mulching, crop, and water harvesting) influence weathering, soil formation, and synthesis and hence the direction of the degradative process.

The net balance of interaction between the two categories of factors (exogenous and endogenous) determines the predominant mechanism, rate, duration, and direction of beneficial processes in soil.^[20] Under unsuitable land use and/or poor management, the restoration capacity of the beneficial processes becomes undermined and the net force balance is tilted negatively toward soil resilience by endogenous factors, hence the prevalence of the degradation mechanisms include:

1. Progressive removal of the top soil layer by soil erosion, tillage erosion, soil mining (e.g., brick making and sand quarrying).
2. Destruction of soil structure/matrix by bad tillage.
3. Nutrient depletion through crop uptake, harvest, and burning.
4. Degradation of subsoil by leaching of aluminum (Al) and silicon and formation of allophane and imogolite from residual silica and alumina.^[21]

On the other hand, when land is well managed, the positive effects of exogenous factors on soil resilience accrue through the following restorative mechanisms:

1. Invasion of subsoil by micro-organisms.
2. Subsequent differentiation of subsoil and plant succession (decomposition and humification).
3. Weathering enhanced by soil and water conservation measures of stabilization.^[21]
4. Silica absorbed by plants to form plant and Al-humus complex, which makes up the thick A horizon peculiar to the black soil.^[22]

CONCLUSION

Notwithstanding the great progress of years in the development of soil resilience theory, there is a pressing need for development of a more adequate, single measure of soil resilience that can be used by soil conservation/land husbandry engineers who are commonly confronted with various combinations of land degradation forms.

REFERENCES

1. Beinroth, F.H.; Eswaran, H.; Reich, P.F.; Van der Berg, P.F. Land related stresses in agroecosystems. In *Stressed Ecosystems and Sustainable Agriculture*; Virmani, S.M., Katyal, J.C., Abrol, I.P., Eds.; Oxford & IBH Publishing Co. Pvt. Ltd.: New Delhi, Bombay, Calcutta, 1994; 131–148.
2. Eswaran, H. Role of soil information in meeting the challenges of sustainable land management (18th Dr. R.V. Tamhane Memorial Lecture). *J Indian Soc. Sci.* **1992**, *40*, 6–24.
3. Szabolcs, I. The concept of soil resilience. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International Publishers: Wallingford, 1994; 33–39.
4. Lal, R. Sustainable lands use systems and soil resilience. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International Publishers: Wallingford, 1994a; 41–67.
5. Tenywa, M.M.; Lal, R.; Majaliwa, M.J.G. Characterization of the stages of soil resilience to degradative stresses: Soil erosion. In *Sustaining the Global Farm*. Selected papers from the 10th International Soil Conservation Organization Meeting, West Lafayette, May 24–29, 1999; Stott, D.E., Mohtar, R.H., Steinhard, G.C., Eds.; Purdue University and the USDA-ARS National Soil Erosion Research Laboratory: West Lafayette, 1990; 606–610.
6. Conway, G.R. Sustainability in agricultural development: tradeoff with productivity, stability and equitability. *J. Farm. Syst. Res.* **1994**, *4* (2), 1–14.
7. Lal, R. Soil degradation and agricultural sustainability. In *Soil and Water Conservation; Challenges and Opportunities*; Proc. of 8th ISCO Conference, New Delhi, India, Dec 4–8, 1994; Bhushan, L.S., Abrol, I.P., Rao, M.S.R.M., Eds.; Indian Association of Soil and Water Conservationists: Dehra Dun, India, 1998; 191–207.
8. Lal, R. Effects of macrofauna on soil properties in tropical ecosystems. *Agr. Ecosystem Environ.* **1988**, *24*, 101–115.
9. Hannam, I. Ecologically sustainable soil: the role of environmental policy and legislation. In *Sustaining the Global farm*; Selected Papers from the 10th International Soil Conservation Meeting, West Lafayette, May 24–29, 1999; Stott, D.E., Mohtar, R.H., Steinhard, G.C., Eds.; International Soil Conservation Organization in Cooperation with USDA and Purdue University: West Lafayette, 2001; 95–100.
10. Kay, B.D.; Rasiah, V.; Perfect, E. The structural aspect of soil resilience. In *Soil Resilience and Sustainable Landuse*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1993; 449–468.
11. Douglas, M. *Sustainable Use of Agricultural Soils. A Review of the Prerequisites for Success or Failure*; Development and Environment Reports No. 11; University of Berne: Switzerland, 1994.
12. Arnold, R.W.; Szabolcs, I.; Targulian, V.O. *Global Soil Change*; International Institute for Applied Systems Analysis: Laxenburg, 1990.
13. Uehara, G.; Tsuji, G.Y.; Beinroth, F.H., Eds. Extrapolating results of long-term experiments. In *Soil Management Experimental Basis for Sustainability and Environmental Quality*; Lal, R., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1995; 105–109.
14. Papendick, R.I.; Parr, J.F.; Van Schilgaarde, J. Soil quality: New perspective for a sustainable agriculture. In *Soil and Water Conservation Challenges and Opportunities*, Proceedings of 8th ISCO Conference, New Delhi, India, Dec

- 4–8, 1994; Bhushan, L.S., Abrol, I.P., Rao, M.S.R.M., Eds.; Indian Association of Soil and Water Conservationists: Dehra Dun, 1998; 227–237.
15. Passioura, J.B. Soil structure and plant growth. *Aust. J. Soil Res.* **1991**, *29*, 717–728.
16. Reif, F. *Fundamentals of Statistical and Thermal Physics*; McGraw-Hill International Editions: Singapore, 1987; 557 pp.
17. Rozanov, B.G. Human impacts on evolution of soils under various ecological conditions of the world. 14th International Congress of Soil Science, August 12–18, 1990, Kyoto, Japan, Plenary Papers, 1990; 53–62.
18. Sys, C.; Van Ranst, E.; Debaneye, J. *Land Evaluation. Part II. Methods in Land Evaluation*; Agri. Publ. 1991, (7), 247.
19. Vlek, P.L.G.; Vielhauer, K. Nutrient management strategies in stressed environments. In *Stressed Ecosystems and Sustainable Agriculture*; Virmani, S.M., Katyal, J.C., Eswaran, H., Abrol, I.P., Eds.; Oxford & IBH Publishing Co. PVT. Ltd.: New Delhi, Bombay, Calcutta, 1994; 203–229.
20. Lal, R. Sustainable land use system and soil resilience, soil quality and sustainability. *Soil Till. Res.* **1993**, *27*, 1–8.
21. Zolcinski, J. A new genetic physio-chemical theory of the formation of humus, peat and coal. The role and significance of biological factors in these processes. *Proc. First Intern. Congr. Soil Sci. Commission III and IV*, **1930**, 335–338.
22. Wada, K. Active aluminium in koroboko soils and non- and para-crystalline clay minerals. *Clay Sci.* **1977**, *17*, 143–151.

Resilience: Land Use and Soil Management

Soojin J. Park

Department of Geography, Seoul National University, Seoul, Korea

Abstract

Soil resilience is a relatively new concept in soil science and land management studies, and the understanding of soil resilience and interaction with soil and land management are rather qualitative and descriptive. Soil degradation trends give an alarming signal to the security of food production. Historically, the rapid increase of agricultural production has been achieved by the intensification of land use and the use of agrochemicals, but conservative soil management practices attract more attention than high-input agriculture in order to reduce soil degradation and agriculture-related pollution. The basic principles for conservative soil management are how to reduce the destructive influence of soil management practices within the resilience potentials of soils and what is the most appropriate soil management to maintain or even improve soil resilience characteristics in given natural and socioeconomic conditions. Our improved understanding of the processes in soil, water, and plant ecosystems in relation to highly variable local environmental conditions is essential for these tasks.

INTRODUCTION

Most human activities associated with land use cause direct and indirect changes in soil-forming processes and soil properties. Urban and industrial development might be the most destructive form of soil disturbance, causing total removal of existing soil layers, introduction of hazardous chemicals into soils, or drastic changes in soil-forming processes within a short time period. Even though the rate of change is slow and adverse effects are not clear in the short run, agricultural activities are also greatly contributing to soil disturbance and consequent loss of soil resilience. Land use practices and their influence on soil resilience are so diverse that it is virtually impossible to describe them in detail. Discussions here are mostly limited to agricultural land use in which soil management practices are essential human interventions to maintain agricultural production. A more theoretical discussion on the influence of other land use systems on soils may be found elsewhere.^[1,2] For agricultural land use, one should also note that the influence of certain soil management practices on soil resilience varies widely depending on soil types formed as a result of highly interacting environmental factors. The ignorance of the intrinsic and spatial variability of soils and environmental conditions may result in significant reduction in crop production and accelerated degradation of soil resources.

BASIC CONCEPT

Soils are the product of the highly complex interactions of many interdependent variables that have been categorized

as the five soil-forming factors: climate, parent material, organisms, topography, and time. Under natural conditions, soil evolves along two developmental pathways: progressive and regressive.^[3] The progressive pathway of soil evolution includes conditions, processes, and factors that promote horizonation, developmental upbuilding, and/or soil deepening. The regressive pathway promotes haploidization (destruction of soil horizons), retardant upbuilding, and/or subsurface removal. Either of these two pathways may be dominant for a soil in a given location, resulting in a steady state or equilibrium of a soil's physical, chemical, and biological properties in a given environment. Any kind of soil management practices leads to changes of such a steady state and accelerates either progressive or regressive pathway.

It does not necessarily mean that soils in progressive pathways are more beneficial for agricultural and engineering use than soils in regressive pathways, but soils in progressive pathways are more likely restored after temporal disturbances induced by soil management by favorable soil-forming conditions. Continuous or sometimes catastrophic dominance of regressive pathways caused by both natural processes and poor soil management may lead to the loss of resilience and even irreversible degradation of soils. However, it might also be possible to reverse the regressive pathway of soils and to restore highly degraded soils with good management and appropriate technological input. The main question in maintaining soil resilience for sustainable agriculture or other human activities is how to minimize the regressive influence of human-induced soil disturbance within the range of progressive pathways of soil formation, which can be modeled:

$$S_r = S_a + \int (S_n - S_d \pm I_m) dt$$

where S_r is the soil's resilience, S_a is the antecedent soil condition, S_n is the rate of new soil formation (progressive condition), S_d is the rate of soil degradation/depletion (regressive condition), and I_m is the management inputs from outside the ecosystem.^[4]

AGRICULTURAL SOIL MANAGEMENT EFFECTS ON SOIL RESILIENCE

Most land use practices have largely been accelerating regressive pathways of soil formation, which is evidenced by rapid soil degradation worldwide.^[5] Agricultural land use is one of the main land uses contributing to such soil degradation processes along with deforestation and overgrazing. Table 1 illustrates progressive and regressive changes induced by soil cultivation. When the original vegetation is removed for cultivation, the soil surface exposed to air

becomes vulnerable to air and wind erosion. It also modifies the microclimate of the near-ground atmosphere. With continuous cultivation, organic matter levels in soil gradually decrease. The decrease of organic matter causes the deterioration of soil structure, reduction of microbial activities and cation exchange capacity, increase of nutrient leaching, and sensitivity to erosion. The use of heavy machinery for tillage and farm traffics destroys soil structures and increases soil compaction, which greatly accelerates erosion processes.

Agricultural soil management can be grouped into five major activities: 1) tillage; 2) maintaining organic material content and nutrients; 3) application of agricultural chemicals; 4) drainage and irrigation; and 5) mechanical control of erosion. In addition to the mechanical control of erosion, soil erosion control is the most important consideration in selecting the different tillage types and maintaining the organic matter content. Tillage is frequently divided into two groups: conventional and conservation tillage. Conventional tillage turns over surface soils using a plow in order to remove weeds and to provide sufficient moisture and nutrient for planted seeds. Conservation tillage includes various tillage

Table 1 Possible progressive and regressive alteration of soil-forming factors by agricultural land use and soil management.

Soil-forming factor	Progressive influence	Regressive influence
Parent material	Adding mineral fertilizers	Removing through harvest more plants and animal nutrients than are replaced
	Accumulating shells and bones	Adding material in amounts toxic to plants or animals
	Accumulating ash locally	Altering soil constituents in a way to depress plant growth
	Removing excess amounts of substances such as salts	
Topography	Checking erosion through surface roughening	Causing subsidence by drainage of wetlands
	Land forming and structure building	Accelerating erosion
	Land leveling	Excavating
Climate	Adding water by irrigation	Subjecting soil to excessive insolation, to extended frost action, to expose to wind, to compaction
	Release of CO ₂ to atmosphere with possible warming trend in climate	Altering aspect by land forming
	Changing color of soil surface to change albedo	Clearing and burning off organic cover
	Removing water by drainage	
	Diverting wind	
Organisms	Introducing and controlling populations of plants and animals	Removing plants and animals
	Adding organic matter to soil directly or indirectly through organisms	Reducing organic content of soil through burning; plowing, overgrazing, harvesting, accelerating oxidation, and leaching
	Loosening soil by ploughing to admit more oxygen	Adding or fostering pathogenic organisms
	Removing pathogenic organisms, e.g., by controlled burning	
Time	Rejuvenating the soil by adding fresh parent material or through exposure of local parent material by soil erosion	Degrading the soil by accelerated removal of nutrients from soil and vegetation cover
	Reclaiming land from under water	Burying soil under solid fill or water

Source: Adapted from Bidwell & Hole,^[6] in which progressive and regressive influences were presented as beneficial and detrimental effects, respectively.

systems, such as no tillage, strip tillage, ridge tillage, and mulch tillage, which aims to reduce soil and water losses.^[7] Crop residue management designed to retain all or a portion of the previous crop's stalks on the soil surface is an important component of most conservation tillage. Maintaining an adequate level of organic materials by means of residue management, crop rotation, and manure is essential in soil management practices not only for providing major nutrients for crop growth but also for improving soil structure and reducing soil erosion.^[8] Without regular addition of adequate amounts of organic materials, the potential of erosion and leaching and gradual deterioration of soil physical properties increase. The strong positive correlation of organic matter content in soils with soil biodiversity has been well documented. Fertilizers and pesticides have led to a rapid increase of crop production, but wide use of agricultural chemicals also causes water pollution and soil degradation. Reduction of soil organic matter caused by replacing traditional organic fertilization through inorganic fertilizer is another negative influence on soil resilience. [Table 2](#) presents a more detailed description on soil management practices and their possible influence on soil's progressive and regressive pathways.

PRINCIPLES AND PRACTICES OF IMPROVING SOIL RESILIENCE

The basic principles of soil management to improve soil resilience are already well known. For agricultural soil management, they can be categorized as: 1) increase in soil organic matter content; 2) improvement in soil structure; 3) increase in soil biodiversity; 4) reduction of erosion rates; and 5) increase in nutrient capital and recycling mechanisms.^[9] Given the complexities of both soil-forming processes and socioeconomic causes of soil degradation, however, there is no single strategy or management policy for maintaining soil resilience. The rapid increase of agricultural production has been achieved by the intensification of land use, the use of motorized farm equipment, and the application of agrochemicals. These technological inputs frequently induce the loss of soil resilience and environmental pollution in many agricultural areas ([Table 2](#)).

New ideas have emerged for more sustainable soil management in both the developed and developing countries, replacing high-input agriculture.^[10] First, our improved understanding of the processes in the soil, water, and plant

Table 2 Major soil management practices and their possible influence on soil-forming processes.

Soil management	Management practice	Progressive influence on soil formation	Regressive influence on soil formation
Tillage	Conventional tillage	Increase aeration within plowed layers	Oxidative loss of organic matter
		Enhancing circulation of oxygen and water	Deterioration of soil structure
		Increase biological activity	Formation of subsurface plow pans
		Breaking surface and impermeable subsurface layers	Exposure of soil surface to raindrop impact, and water and wind erosion
		Increasing weathering rates	Reduction of plant diversity
	Conservation tillage (combined with residue management)	Increase of organic matter content in soil	Soil compaction (possibly in short term)
		Reducing erosion by increasing surface residue	Possible herbicide use
		Increase of biological activities	
		Reduced soil structure disruption (than conventional tillage)	
Maintaining organic matter and nutrients	Residue management (Crop residue mulching)	Protecting soil surface from raindrop impact	Reduction of soil temperature
		Reducing surface runoff	
		Increase organic matter content by the decomposition of residue	
		Increase biological activity	
		Less variation of soil temperature	

(Continued)

Table 2 Major soil management practices and their possible influence on soil-forming processes. (Continued)

Soil management	Management practice	Progressive influence on soil formation	Regressive influence on soil formation
Application of agricultural chemicals	Crop rotation (including cover and green manure crop)	Protecting soil surface by plant cover	Structural damage during harvesting of certain crops (e.g., potatoes, sugar beet)
		Increase organic matter input	
		Increase biodiversity in soils	
		Enhancing soil structure and stable aggregates	
		Enhancing nutrient balance	
	Organic fertilization (manure and compost)	Increase organic matter content in a short time period	Introduction of toxic chemicals and heavy metals
		Increase water retention capacity	Introduction of animal pathogens
		Promoting stable soil aggregates	Gaseous damage (e.g., ammonia) for certain plant species and microorganisms
	Shifting cultivation	Addition of nutrients into soils	Exposing soil surface to erosion processes
		Initial improvement of soil structure in heavy soils	Decrease of soil fauna population by heat
		Initial soil pH increases	Deterioration of soil structure through cropping
			Nutrient loss and soil acidification with cropping
Application of agricultural chemicals	Soil acidity adjustment (lime and gypsum application)	Reducing soil acidity	Decrease of infiltration rate with cropping
			Ionic imbalance of soil solution
	Inorganic fertilizer	Enhance stable soil aggregate formation	Selective leaching for certain ions (e.g., K^+)
		Increase of plant growth and crop yield	Reduction of soil organic matter replacing traditional soil organic management
		Enhance balance of solutes in soil	Increase of soil acidity by certain fertilizers (e.g., superphosphate)
			Nutrient imbalance and toxicity
Irrigation and drainage	Pesticide application		Possible loss of vital soil organisms
			Possible loss of vital soil organisms (e.g., earthworms, termites)
	Irrigation	Supplying water in soils	Accumulation of salts or alkali compounds in soil (in arid and semi-arid climate)
		Removing excessive salts	Deterioration of soil structure and consequent reduction of infiltration rate, when sodium is present
		Encouraging microbial activities	Increase of toxic elements (e.g., boron, selenium)
			Poor aeration caused by water logging
Mechanical erosion control	Drainage	Enhancing soil aeration	Rapid oxidation of organic matter
		Lowering the regional water table	Soil shrinkage and consequent surface lowering
	Contouring, terraces and waterways	Reducing erosion potential	Possible slope failure
			Removal of topsoil and expose of subsurface soil during construction

Note: This table is not intended to be exhaustive, it is also highly controversial for any generalization for all soil types.

ecosystem should be used for adapting management methods to the infinite variety of local conditions. Second, greater emphasis should be placed on efficiency of using soils, water, fertilizers, and plant resources instead of opening new fragile lands or relying on higher inputs. Finally, the joint efforts of resource scientists and the knowledge and skills of the local people should be used to address issues of soil management.

The first and second principles require more scientific and quantitative understanding of relationships between different soil management systems and long-term soil resilience. Soils are extremely complex media with many intrinsic feedback mechanisms, and the magnitude and degree of a given human intervention on soil resilience vary widely. It is also frequently recognized that a progressive influence of a human activity in one soil might be a regressive influence in other soils or the same type of soil in a different environment. Much information is available on how different soil management practices directly and indirectly regulate crop yields, but less is known about quantitative relationships between soil management and long-term soil processes that influence soil resilience characteristics. One important aspect of soil resilience studies is to develop appropriate restorative measures to improve degrading or already-degraded soils. This is especially important for many developing countries where cultivation of new land and advanced technological inputs are limited due to high population pressure and the lack of socioeconomic incentives.

Basic principles of enhancing soil resilience are known from many regions, but often those technical interventions are not available to farmers because of population pressures, limited natural resources, lack of secure land tenure systems, unfavorable macro-economic conditions, and poor national and international policies, etc.^[11] The processes of soil degradation are too locally specific to be effectively combated with the best land management developed elsewhere. The introduction of western farming techniques in many developing countries has been proven largely inadequate and sometimes disastrous for soil quality.^[10] Preserving and improving traditional land management techniques are increasingly considered the more sustainable land management system.^[12] Efforts to develop optimum sustainable soil management must include land users in both planning and implementation.

CONCLUSION

The rate of soil formation is slow, ranging from 0.001 to 0.4 cm a year,^[4] and the quality of such slowly forming soils is rapidly degrading due to inappropriate soil management in many parts of the world. Soil degradation trends give an alarming signal to the security of food production. Historically, the rapid increase of agricultural production has been achieved by the intensification of land use and the use of agrochemicals, but conservative soil management practices attract more attention than high-input agriculture in order to

reduce soil degradation and agriculture-related pollution. The basic principles for conservative soil management are how to reduce the destructive influence of soil management practices within the resilience potentials of soils and what is the most appropriate soil management to maintain or even improve soil resilience characteristics in given natural and socioeconomic conditions. Our improved understanding of the processes in soil, water, and plant ecosystems in relation to highly variable local environmental conditions is essential for these tasks. In addition, the joint efforts of resource scientists and the knowledge of the local people should be used to address issues of optimal local soil management. Soil resilience is a relatively new concept in soil science and land management studies, and the understanding of soil resilience and interaction with soil and land management are rather qualitative and descriptive. A high research priority should be given to develop objective and quantitative criteria of soil resilience, especially in relation to different soil-forming processes and also to assess the cause-effect relationship between different soil management practices and resilience.

REFERENCES

1. Amundson, R.; Jenny, H. The place of humans in the state factor theory of ecosystems and their soils. *Soil Sci.* **1991**, *151*, 99–109.
2. Seybold, C.A.; Herrick, J.E.; Brejda, J.J. Soil resilience: A fundamental component of soil quality. *Soil Sci.* **1999**, *164*, 224–234.
3. Johnson, D.L.; Watson-Stegner, D. Evolution model of pedogenesis. *Soil Sci.* **1987**, *143*, 349–366.
4. Lal, R. Sustainable land use systems and soil resilience. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 41–67.
5. Oldeman, L.R. Global extend of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 99–118.
6. Bidwell, O.W.; Hole, F.D. Man as a factor of soil formation. *Soil Sci.* **1965**, *99*, 65–72.
7. Bradford, J.M.; Peterson, G.A. Conservation tillage. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 1999; G-247–G-270.
8. Swift, M.J.; Sanchez, P.A. Biological management of tropical soil fertility for sustained productivity. *Nat. Resour.* **1984**, *20*, 2–10.
9. Lal, R. Soil quality and sustainability. In *Methods for Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentine, C., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1998; 17–30.
10. Young, A. *Land Resources, Now and for the Future*; Cambridge University Press: Cambridge, 1998; 319 pp.
11. Barrow, C.J. *Land Degradation*; Cambridge University Press: Cambridge, 1994; 316 pp.
12. Katyal, J.C.; Vlek, P.L.G. *Desertification—Concept, Causes and Amelioration*; ZEF Discussion Papers on Development Policy, No. 33; ZEF: Bonn, 2000; 73 pp.

Resilience: Quality and

Rattan Lal

*Carbon Management and Sequestration Center, Ohio State University,
Columbus, Ohio, U.S.A.*

Abstract

Soil resilience is an ecological concept being used to understand soil processes and their dynamics under stress. It refers to soil's capacity to resist change and restore itself when stress is removed and improved management systems are adopted. Soil resilience depends on inherent soil properties or soil quality, management and institutional factors, and climate. It can be measured for key soil properties and then aggregated to determine soil resilience class. Soil resilience has an important application to soil management for minimizing the risks of soil degradation and for restoring degraded soils. In view of the severity of soil degradation worldwide, the economic importance of soil resilience in maintaining soil quality and the multifunctions of soil cannot be overemphasized.

INTRODUCTION

Soil resilience, soil's ability to restore its productivity and environmental moderation/buffering capacity,^[1] is important because of the ever increasing risks of soil and environmental degradation due to anthropogenic activities. It is important to understand the factors and processes governing soil resilience and the techniques of measuring and modeling changes in it. The topic is more important and relevant because of at least five major global issues of the 21st century. These are as follows: 1) soil degradation and desertification; 2) accelerated greenhouse effect; 3) water scarcity; 4) land disposal of solid waste; and 5) agricultural sustainability. While all of these issues are influenced by soil resilience and quality, the global problem of soil degradation^[2] is an important issue that needs to be addressed.^[3] Soil degradation is also closely linked with the emission of greenhouse gases,^[4–6] water disposal,^[6] and water quality. Thus, an in-depth understanding of the mechanisms involved in imparting resilience to soils is a necessary prerequisite to identifying appropriate techniques for averting degradation and enhancing or maintaining soil resilience.

Soil quality, productivity, and environmental moderation capacity^[7] are the net balance between soil degradative processes and soil resilience (Fig. 1). In practice, soil quality is the capacity of the soil to perform a range of functions of value to humans. Important among these functions are as follows: 1) producing biomass; 2) filtering pollutants out of water; 3) buffering emissions of greenhouse gases; 4) moderating cycles of carbon (C), nitrogen, phosphorus, sulfur, water, and other elements; and 5) biodegrading contaminants. Soil resilience maintains or enhances these functions while keeping degradative processes under check and restoring soil's life-supporting and environmental cleansing

processes. Soil resilience can also decrease the risk of an accelerated greenhouse effect by C sequestration in stable microaggregates or deep incorporation in the soil profile; reduce eutrophication of surface water and contamination of groundwater by moderating its filtration and bioremediation processes; and facilitate achievement of agricultural sustainability by improving its life-supporting processes.

BASIC CONCEPTS AND DEFINITIONS

Applications of ecological concepts to soil science bring into focus several interrelated terms;^[8–12] some of these include the following:

Resilience: the capacity to restore.

Soil stability: the absence of change or constancy in soil quality.

Persistence: the length of survival.

Elasticity: the rate of restoration after perturbation.

Amplitude: the magnitude of change in soil quality from which restoration is possible.

These terms are subjective and to be useful in soil management and restoration, and it is important to operationalize them and develop methods to quantify their attributes.

There are two principal schools in relation to the concept of soil resilience.^[13] Per the first school, soil resilience refers to soil's capacity to restore itself.^[14,15] If not excessively degraded beyond the point of no return, soil, like any natural ecosystem, has the capacity to self-regulation and self-maintenance.^[16] Soil resilience, as per this school, is related to its: 1) filtration, detoxification, and transformation including bioremediation abilities; and

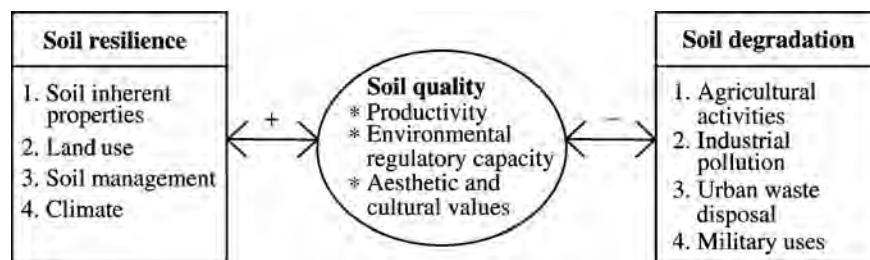


Fig. 1 Soil quality is the net effect of dynamic equilibrium between soil degradative and soil resilience processes.

2) its renewability, structural rejuvenation, and maintenance of a gene pool in soils.^[12,17] Per the second school, soil resilience refers to its ability to resist change^[2,18] or stability. According to this concept, therefore, soil resilience or stability is related to its buffering capacity. Although there may be some soils that can resist perturbation to a limit, most soils undergo change but may recover once the perturbation is removed.

Therefore, soil resilience may be defined as “soil’s ability to resist change and restore its quality following stress alleviation and adoption of improved management systems.” Land use and soil management are important to soil resilience.^[14] Similar to ecosystems, soils can also be grouped according to their resilience characteristics^[15,16] as follows:

High resilience: Soils that are not susceptible to degradation or are only slightly susceptible under extreme stresses. Some highly fertile and young soils come under this category, e.g., alluvial soils of the flood plains, Loess soils and Andisols on flat terrain, and Mollisols. These are dynamically stable soils.

Moderate resilience: Soils are moderately susceptible to degradative processes with continuous and intensive cultivation.

Slight resilience: These soils are easily susceptible to degradation and have low capacity to recover and restore.

Non-resilient: Such soils are easily degraded beyond the point of no return and cannot restore themselves. Some shallow soils on steep slopes are non-resilient against accelerated erosion. Similarly, a plinthic soil with a hardened layer at a shallow depth can be non-resilient. Non-resilient soils may be also called as fragile soils. These soils are easily destroyed in relation to their principal functions, e.g., productivity, environmental quality, social and aesthetic values, gene pool reserves, and support for engineering structures. Soil grouping according to this classification is shown in Table 1.

Susceptibility to degradation is just the reverse of soil resilience. However, the concept of “stability” cannot be applied to soils for agricultural use; e.g., plinthic (rocky) and gravelly soils may be highly stable but are not highly productive under intensive agricultural use. Such soils are

Table 1 Types of soil resilience in relation to agricultural land use.

Class	Type	Description	Soil order
1	Resilient	Highly responsive to science-based inputs; prime agricultural land, with a wide range of soil characteristics. These soils have a high buffering capacity.	Mollisols, Histosols, Inceptisols, Andisols, and other soils which are not easily degraded
2	Moderately resilient	These soils have fair to good agricultural potential and undergo changes in soil properties which can be restored with good soil management and science-based inputs. These soils respond to management. Changes in soil properties are modest, and soil properties can be restored by science-based management.	Oxisols, Ultisols, Alfisols, Entisol, and Aridisols
3	Slightly resilient	These soils have marginal or low agricultural potential, and the range of soil properties for productive use is narrow. These soils have only modest response to input. Critical limits for irreversible degradation are readily achieved.	Vertisols, Spodosols, and Psamments soils on steep terrain
4	Sensitive	These soils undergo drastic and adverse changes in soil properties, e.g., bulk density, infiltration, porosity, aggregation, and nutrient depletion, and are not suitable for agriculture. Examples include acid sulfate soils, shallow soils, soils on steep terrain, and ecologically sensitive regions. Intensive cultivation of poorly structured soils, draining of acid sulfate soils, and cultivation of marginal lands can lead to irreversible degradation by mismanagement or any intensive land use.	Sulfaquents, Psamments, and Entisols

Source: From Lal.^[19]

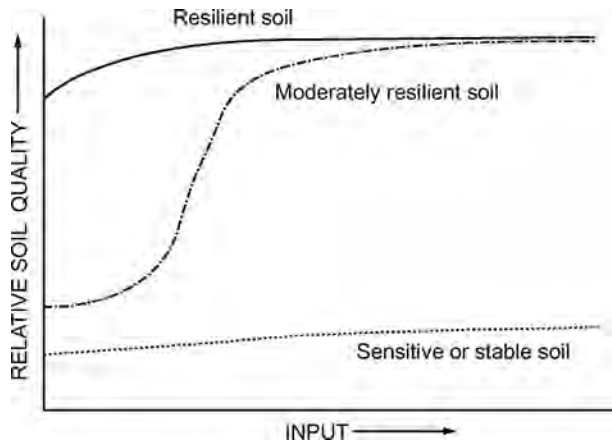


Fig. 2 Response to input of resilient and stable soils.

more suited for engineering (construction) than agricultural use. Soil is in dynamic equilibrium with its environment and management. A productive soil of high quality is not stable; it responds to management intensity. The term “management” is a broad-based concept, and “input” is a subset. The management may involve varying intensities of the same facets or different facets depending on the constraint to be alleviated. It is always undergoing a change and attains equilibrium at a level determined by its inherent properties and management (Figs. 2 and 3). Another important aspect of resilience and management is the economics of production. If the management intensity or the input is not economic, the soil loses its resilience for the specific use.

FACTORS AFFECTING SOIL RESILIENCE

Soil resilience is affected by a range of intrinsic soil properties (or endogenous factors) and external conditions (or exogenous factors) (Fig. 4). Soil quality constitutes an

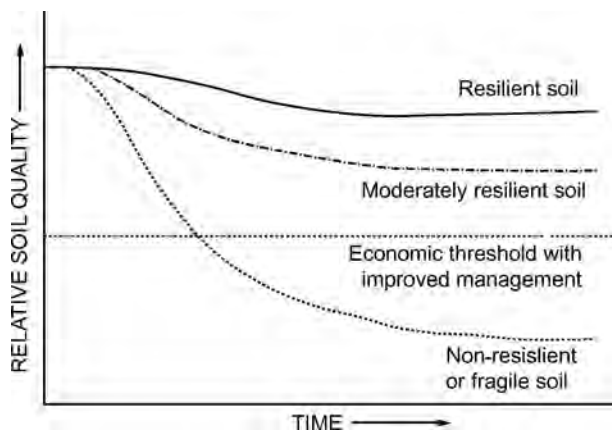


Fig. 3 Temporal changes in soil quality as affected by soil resilience and management.

important endogenous factor and includes physical, chemical, and biological processes and entities of soil quality. Soil resilience decreases with increasing soil capability class.^[19] Important among exogenous factors are land use and management, socioeconomic and institutional factors, and climate. Soils of arid regions are slightly to moderately resilient, all other factors remaining the same. Factors that affect productivity, profitability, and the farmer’s ability to reinvest in soil improvement affect soil resilience. Productivity, sustainability, and environments are linked through soil resilience (Eq. 1, Fig. 5).

$$A_s = f(S_r, P, E)_t \quad (1)$$

where A_s is the agricultural sustainability, S_r is the soil resilience, P is the productivity, E is the environment, and t is the time. Resilient soils are productive, are easy to manage, and have a wide range of management and farming systems options for sustainable use.

ASSESSMENT OF SOIL RESILIENCE

Soil resilience is an ecological concept applied to soil. Operationalization of the concept depends on the development of methods to assess and predict soil resilience. Lal^[1,14] proposed that soil resilience may be assessed by measuring components outlined in Eq. 2.

$$S_r = S_a + \int_0^t (S_n - S_d + I_m) dt \quad (2)$$

where S_a is the antecedent condition, S_n is the renewability, S_d is the degradation, and I_m is the input or management. S_a is in relative terms and refers to conditions prior to subjection to a stress. The magnitude and sign of the term $(S_n - S_d + I_m)$ is important in defining soil resilience class. The application of Eq. 2 involves a separate assessment of the individual soil properties [soil organic C (SOC) content, top soil depth, available water capacity (AWC), nutrient reserves etc.] and then combines these as per a rating system.^[20] Developing an aggregate or composite S_r index may also involve statistical techniques including principal component analysis and cluster analysis. Soil resilience can also be computed for individual soil properties, such as structural resilience in relation to erodibility and compactability.^[21] Quantitative assessment of structural resilience is important in developing and adopting soil and crop management systems that minimize the risks of physical degradation, e.g., erosion, compaction, and crusting. Soil resilience may also be assessed for properties that affect other degradative processes, e.g., acidification and salinization.

Because it is a new concept in soil science, not much progress has been made in predicting soil resilience. Roza-nov^[13] proposed a physical model to predict soil resilience (Eqs. 3 and 4).

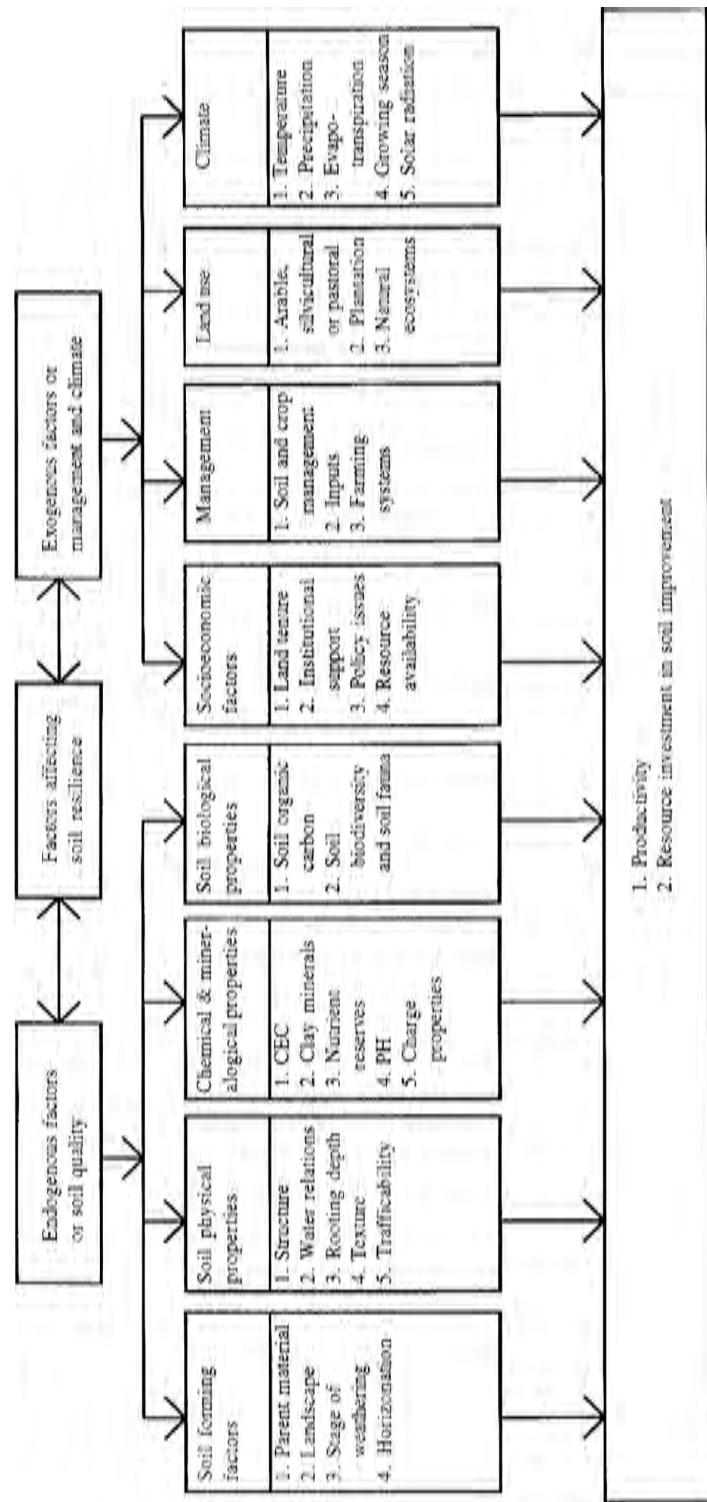


Fig. 4 Exogenous and endogenous factors and their interaction affecting soil resilience.

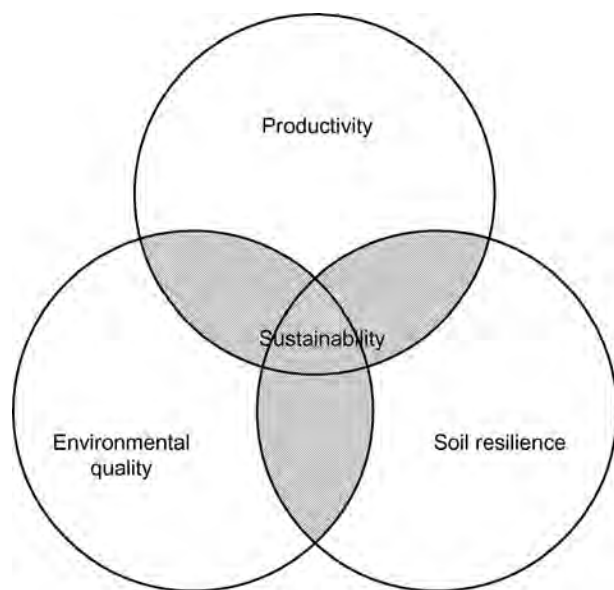


Fig. 5 Sustainability depends on productivity, environmental quality, and soil resilience.

$$S_r = \frac{dI/dx}{x} \quad (3)$$

$$S_r = -2I/x^2 \quad (4)$$

where S_r is the coefficient of soil resilience, I is the work or input required to change the soil property, x . Soil resilience of two soils or two land use systems is equal if a similar level of input is required to bring about the desired change in soil property, x .

SOIL RESILIENCE AND SOIL RESTORATION

Restoration of degraded soils and ecosystems is a high priority because global soil resources are finite and the world population is rapidly increasing at the rate of about 73 million per year. The soil quality depends on an optimal level of soil properties that a given soil needs to have in order to perform its functions satisfactorily. Therefore, the identification of appropriate techniques of soil restoration requires knowledge of the following: 1) soil resilience and 2) critical limits of key soil properties beyond which soil loses its resilience and becomes fragile and non-restorable for a given function. Key soil properties and their limits vary among soils and land use functions. However, important properties are rooting depth, SOC content, AWC, nutrient reserves, and soil structure as measured by aggregation and stability.

Knowledge of critical limits of soil properties is important because it may enable the land user to take soil out of a given land use before it becomes non-resilient or fragile. Similarly, an appropriate land use and management system may be chosen that will maintain or enhance soil quality, soil resilience, and soil stability. There are numerous

examples of soil quality and soil resilience enhancement by the choice of an appropriate land use system. Soils of the Machakos region of Kenya were severely degraded by the early to mid-1930s, and these badlands have been converted to productive farm land.^[22] Lessons learned from this example of soil resilience and land restoration can be applied to restore degraded soils elsewhere, e.g., the Himalayan–Tibetan ecosystem, Andean region, Central America, and the Caribbeans. Some soils of volcanic origin are highly stable and resist degradative processes.

Methods and degree of soil restoration or reclamation also depend on the land use or functions. With a change in soil resilience class, land use may also change; e.g., a soil capable of growing maize (*Zea mays*) can now grow only sorghum (*Sorghum bicolor*), or a soil capable of growing rice (*Oryza sativa*) can now only grow an upland crop. Therefore, the techniques and intensity of needed inputs depend on the land use or farming system intended.

There is a need to establish a link between the strategy or process of soil restoration and soil quality. This link is through key soil properties and the knowledge of their critical limits. Whether a degraded soil has been restored, through change in land use or management, can be determined through the analyses of key soil properties and their dynamics.

CONCLUSION

Soil resilience is an ecological concept being used to understand soil processes and their dynamics under stress. It refers to soil's capacity to resist change and restore itself when stress is removed and improved management systems are adopted. Soil resilience depends on inherent soil properties or soil quality, management and institutional factors, and climate. It can be measured for key soil properties and then aggregated to determine soil resilience class. Soil resilience has an important application to soil management for minimizing the risks of soil degradation and for restoring degraded soils. In view of the severity of soil degradation worldwide, the economic importance of soil resilience in maintaining soil quality and the multi-functions of soil cannot be overemphasized. However, additional research is needed to improve the concept and to enhance its application to sustainable management of soil and water resources.

REFERENCES

1. Lal, R. Degradation and resilience of soils. *Philos. T. Roy. Soc. B.* **1997**, 352, 997–1010.
2. Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 99–117.

3. Barrow, C.J. *Land Degradation: Development and Break-down of Industrial Environments*; Cambridge University Press: Cambridge, 1991; 295 pp.
4. Sombroek, W.G. *The Greenhouse Effect, Plant Growth and Soils*; Special report on 25th Anniversary of ISRIC, ISRIC: Wageningen, 1991.
5. Lal, R.; Kimble, J.M.; Levine, E.; Stewart, B.A. *Soils and Global change*; CRC/Lewis Publishers: Boca Raton, 1995; 544 pp.
6. Adger, W.N.; Brown, K. *Land Use and The Causes of Global Warming*; Wiley: Chichester, 1994; 271 pp.
7. Lal, R. Tillage effects on soil degradation, soil resilience, soil quality and sustainability. *Soil Till. Res.* **1993**, *27*, 1–8.
8. Oriens, G. Diversity, stability and maturity in natural ecosystems. In *Unifying Concepts in Ecology*; Van Dobben, W.H., Rowe-McConnell, R.H., Eds.; Junk Publishers: The Hague, 1975; 138–139.
9. Whitaker, R.H. *Communities and Ecosystems*, 2nd Ed.; MacMillan: New York, 1975.
10. Westman, W.E. Measuring the inertia and resilience of ecosystems. *Bioscience* **1978**, *28*, 705–710.
11. Peters, R.H. *A Critique for Ecology*; Cambridge University Press: Cambridge, 1991.
12. Blum, W.E.H. Soil resilience: General approaches and definitions. In *Proceedings of the XVth International Congress Soil Science Acapulco*, Mexico, Jul 10–16, 1994; Vol. 29, 233–237.
13. Rozanov, B. Stressed soil systems and soil resilience in drylands. In *Proceedings of the XVth International Congress Soil Science Acapulco*, Mexico, Jul 10–16, 1994; Vol. 29, 238–245.
14. Lal, R. Sustainable land use systems and soil resilience. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 41–67.
15. Kay, B.D.; Rasiyah, V.; Perfect, E. The structural aspects of soil resilience. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Wallingford, 1994; 535–542.
16. OECD. *The State of the Environment*; OECD: Paris, 1991; 295 pp.
17. Szabolcs, I. Introduction to the symposium on “stressed ecosystems and soil resilience”. In *Proceedings of the XVth International Congress Soil Science, Acapulco*, Mexico, Jul 10–16, 1994; Vol. 29, 231–237.
18. Johnson, D.L.; Lewis, L.A. *Land Degradation: Creation and Destruction*; Blackwell: Cambridge, 1995; 335 pp.
19. Lal, R. Land use and soil resilience. In *Proceedings of the XVth International Congress Soil Science, Acapulco*, Mexico, Jul 10–16, 1994; Vol. 29, 246–261.
20. Lal, R. *Methods and Guidelines for Assessing Sustainable Use of Soil and Water Resources in the Tropics*; SMSS Soil's Bulletin 21: Washington, 1994; 78 pp.
21. Kay, B.D. Rates of change of soil structure under different cropping systems. *Adv. Soil Sci.* **1990**, *12*, 1–30.
22. Tiffen, M.; Mortimore, F.; Gichuki, F. *More People, Less Erosion: Environmental Recovery in Kenya*; Wiley: Chichester, 1994; 311 pp.

Resilience: Restoration

F.W.T. Penning de Vries

International Board of Soil Research and Management, Pretoria, South Africa

Eric T. Craswell

Fenner School of Environment and Society, College of Medicine, Biology, and Environment, Australian National University, Canberra, Australian Capital Territory, Australia

Abstract

Processes of change in soils are relatively slow but common. In some cases, soils are improved so that their capacity for services increases. In most cases and particularly in developing countries, however, their services decrease. Some of the relevant processes are described briefly. Resilience is the property of the soil to resist changes in the level of services provided despite anthropogenic or environmental pressures. Because of resilience, soils can be degraded to some degree before their level of services declines significantly. Before reaching this level, degradation can be reversed and soils can be restored. Resilience also masks the degradation processes, particularly when the soils are deep and fertile, so that significant degradation may proceed unobserved.

INTRODUCTION

Soil is a substrate and a medium that provides many agricultural and non-agricultural services. Degradation is the broad term that describes the loss of soil value for a particular function. Soils are dynamic because the amount of soil at a particular location can increase or decrease, and the absolute and relative amounts of their components can change significantly. Some changes in soils are endogenous (e.g., weathering), while others are exogenous and are caused by animals (e.g., termite hills) and humans (e.g., reinforced drainage, deposition, and nutrient mining with crops). Cultivation on soil may cause it to degrade or to improve, depending on the management. From a global perspective, improvements are less common than degradation. Environmental conditions may also cause changes in soils.

Resilience, or resistance to change, varies greatly between soil types and is affected by climate and management practice. The degree of resilience and, hence, the period over which degradation can occur unnoticed is also quite variable, and so are the time and effort required to restore degraded soils. Typically, virgin soils converted to annual cultivation can be used for 3–30 years before the first sign of reduced function becomes apparent, after which degradation can take place for another 5–50 years until the soil is no longer economically productive.^[1–3] While restoration in general is technically feasible, the economic and sociological costs are often prohibitive.^[4,5] Preferably, interventions should be made early to mitigate degradative forces, taking advantage of inherent soil resilience to offset the restoration costs.

DEFINITIONS AND EXAMPLES

Resilience is the property of the soil to resist changes in the level of services it provides despite anthropogenic or environmental pressures. Degradation is the process by which soils change and its level of services declines. Anthropogenic degradation of soils has much accelerated in the past.^[4,6] Restoration is the process by which the soil is brought back to a prior condition, usually by improving its water and nutrient holding capacities.

Soils Provide Many Services

The services provided by soils are that they serve as a medium for agricultural crops, grasslands, and forests through which water and nutrients are absorbed, a medium to store and/or degrade pollutants, a place for recreation (golf!) and tourism, a place for temporary storage of excess water, generating water for streams and rivers by runoff from upland areas, and providing a surface for infrastructure and buildings. People enjoy several services simultaneously, but all of them are not important everywhere and at the same time. Soil degradation does not affect these services to the same extent. Hence, resilience with respect to these services can be very different. For instance, soils that are already mined for profitable cropping may still be of high value for infrastructure, recreation, or forestry. On the other hand, the service “generating water” need not be in conflict with “being a medium for crop production.”

Resilience and Lack of Degradation Are Not Identical

A soil may change considerably while its traditional service is not affected clearly, particularly in economic terms (e.g., erosion of a fertile and deep soil).^[7] In such cases, soil degradation may not appear to be a concern, and the need for soil and water conservation may not be recognized. If the changes continue, eventually leading to visible degradation, it will be more difficult to retain the same level of service, and the cost of restoration will escalate. In other cases, subtle changes in the soil may already have major consequences to its services (for instance, the addition of *Rhizobium* inoculum and small amounts of micronutrients can increase crop productivity). Fig. 1 shows examples of the resilience of different soil types to erosion. Other examples of resilience are discussed elsewhere.^[7-9]

Like any other object with mass, soils are resilient to change, meaning that their properties and components resist change in spite of external pressures that are constant or are sudden perturbations. An example is the alluvial soil of the Nile Valley, the agricultural function of which has remained unchanged for a thousand years or more. This example also shows that, in practice, resilience is a combined characteristic of the soil and its environment. Another example of resilience is the China Loess Plateau, where soils are very deep, and erosion of topsoil for hundreds of years has not diminished the productivity very much. Resilience is caused by: 1) the natural generation of soil from deeper layers, which occurs at a slow rate (<mm/yr) except for humid, tropical, volcanic soils, and alluvial and aeolian sedimentation (<cm/yr); 2) the weathering of rock minerals that provide all the nutrients for crops [phosphorus, potassium (K), micronutrients; a few kg/ha/yr] and biological nitrogen fixation (1–50 kg N/ha/yr); and 3) the buffering capacity of the soil mass. Degradation often goes hand in hand with transport of

soil, organic matter, and nutrients to another location where it may be beneficial (e.g., the Nile Valley clay) or detrimental (e.g., reservoir siltation). When soils change beyond their threshold of resilience, land users may be interested in restoration of the prior status of the soil. Restoration mainly refers to returning the capacity of the soil to support productive processes to the antecedent level, but not necessarily to the same physical, chemical, and biological state. While restoration is often technically feasible, economic and sociological conditions may not allow the process, and the land becomes barren. This is particularly pertinent to areas with low crop cover, such as in semiarid environments where crops are grown for short periods, and on infertile land where water harvesting and runoff redistribute soils and nutrients.

Sustainable soil and land management refers to situations where all biophysical, agronomic, economic, and sociological conditions for long-term farming are met simultaneously at the farm level.^[10,11] Soil resilience helps to overcome short periods of unsustainable land management. Such short-term benefits of resilience offer the advantage that, if appropriate management actions are taken, soil changes can be limited and the threshold of soil resilience is not exceeded.

EXTENT AND RATE OF DEGRADATION

Soil degradation is a widespread problem, to the extent that its productive uses have been diminished significantly. All continents suffer from loss in the capacity of soils to support agricultural crops, although some countries suffer more than others. An excellent description of the status of soil degradation was made by van Lynden and Oldeman.^[12,13] Yet it needs urgent updating, as the degraded land area is rapidly increasing where the resilience thresholds are surpassed ecologically and economically.^[3,4,14] For agricultural production, degradation implies that the yield ceiling is reduced, efficiency of inputs may be less (and hence the cost of production is increased), and the risk of crop failure escalates (Fig. 2).

Some degradation and erosion processes that put soil resilience to the test are elaborated below. Many technologies for soil and water conservation are relatively well known, but their adoption and adaptation depend heavily on whether the farmers' socioeconomic circumstances allow them to adopt the technology concerned. For instance, while terracing and alley cropping are known to be effective for controlling erosion, smallholder farmers often find such methods to be too expensive and labor demanding to match their resources. Willingness to follow soil conservation and restoration practices is particularly low when farmers do not own the land on which they farm and are not certain whether their investments will bring returns to their families.^[11] Therefore, lack of knowledge and land tenure and distance from markets, compounded

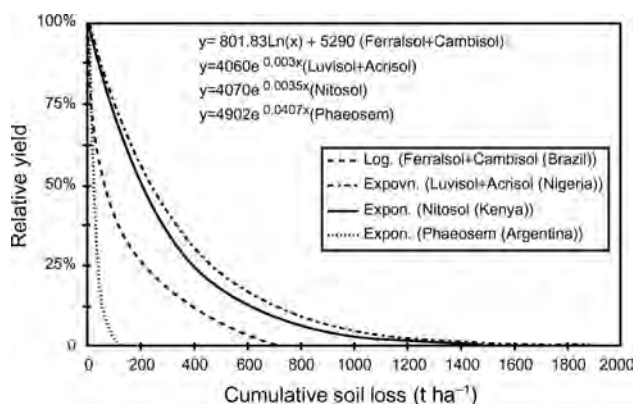


Fig. 1 Example of resilience of the service “relative crop yield” to erosion of different soil types.

Source: From Tengberg & Stocking.^[7]

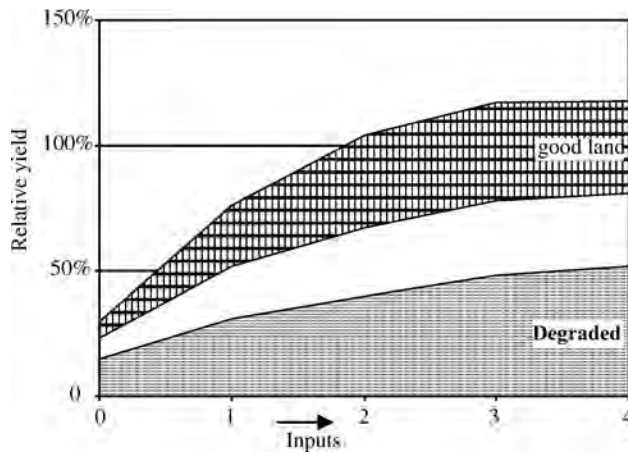


Fig. 2 Consequences of soil degradation for the response of the relative crop yield to management.

Source: From Penning de Vries.^[14]

by opportunities for extensification and shifting cultivation, may lead smallholders in the tropics to pursue exploitative and degrading land management practices and to continue these beyond the limits of the resilience of their environment.

Resilience, Degradation, and Restoration

The degree of resilience of soils to the numerous processes of degradation (erosion, nutrient mining, salinization, compaction, surface sealing, etc.) varies widely, depending on the nature of the process and the characteristics of the soil and environment. Some of the important dimensions are outlined below, and the bibliography provides a comprehensive list of recommended literature on this subject.

Globally, erosion is one of the most important degradation processes.^[1,2,4,15] It involves transport of soil particles, organic matter, and soil nutrients, via flowing water or wind. Soil depth is a key factor determining resilience to these surface-erosive processes. In the humid tropics, water erosion can cause all the topsoil to be removed in one human generation. Wind erosion occurs in semiarid environments and contributes to desertification. Eroded soil can be valuable downstream, but it can also silt up waterways and reservoirs. The degree of resilience strongly depends on soil type and land management (Fig. 1). Restoration is generally not feasible, but halting the process is possible in the right socioeconomic environment with small trenches, and/or dikes^[7] provide good examples. The more the soil surface is covered by vegetation, particularly when early season (erosive) rains arrive, the higher the erosion is prevented. This generally calls for the use of perennial crops and mulches.

Another key degradation process is soil nutrient mining. Mining typically occurs when cropping removes more

nutrients than that added by the farmer as the soil generates.^[16] Resilience to this process depends on both capacity and intensity factors.^[9,17] If only one crop is harvested in 5–20 years and the other years are fallow (the slash-and-burn management system), weathering may provide as much nutrients as removed by the single crop, and the system is sustainable. But when there is insufficient space or time for fallow, which is common these days, then mining reduces soil organic matter (SOM) and its nutrients. In many developing countries, SOM levels have declined by 50% over the past, with concomitant reduction in pH and increased aluminum toxicity, and particularly with loss of nutrient holding capacity.^[18] Conversely, soil nutrient levels have risen in some areas of developed countries among others because of soil fertility transfer. Even when the SOM level remains constant, K stock can be reduced (e.g., intensive rice cultivation) and productivity decreased. Resilience of some soils in terms of nutrient and organic reserves is generally significant. For example, Vertisols may be cultivated for 20 years with good yields even without fertilizer application.^[19] Soil mining is often reversible, provided that the process has not gone too far, by adding legumes to the crop cycle and greater use of organic and inorganic fertilizers.^[17,20] Soil fertility can be viewed as a valuable natural resource that may be depleted to a certain degree and then replenished when conditions allow.

Erosion and loss of soil fertility are processes that tend to reinforce each other: less soil nutrients > less crops and less SOM > less soil cover > more erosion > less nutrients, and so on.

Salinization is a common soil degradation process in irrigated areas.^[2,3] Resilience to salinization is not a soil characteristic per se but relates to the climate and the management of water, trees, and crops. In some areas, dryland salinity occurs when deep soil contains salts that come to the surface due to insufficient rain infiltration or pumping of groundwater. Large parts of Australia and areas such as Northeast Thailand suffer from this phenomenon. Restoration is possible with appropriate management of water and vegetation in the landscape and when the problem has not progressed too far.

Compaction is the increase in density of soils, particularly because of heavy mechanical traffic. When soils are too compact, roots cannot penetrate; the soils absorb less water and nutrients, and production is reduced. Resilience to compaction depends much on the particle size distribution and cropping pattern. Compaction generally increases water runoff. Some soils are much less resilient and more sensitive than others. Restoration under conventional tillage may require heavy equipment for deep plowing.

Sealing of the surface has a similar effect; it may be caused by physical, chemical, and biological agents and can be eliminated with light equipment. However, maintenance of a continuous plant cover is needed to prevent it from reoccurring.

CONCLUSIONS

Resilience is an important intrinsic property of soils, but the extent to which it can prevent change in the level of services that a soil provides depends on the anthropogenic and environmental pressures exerted on it. Soils provide many services, and the resilience in providing these may be different. Moreover, the level of resilience is affected by degradation processes, such as erosion and soil nutrient mining. Resilience to change allows moderate levels of degradation after which restoration is still feasible. For the same reason, however, resilience masks consequences of degradation processes and leads to underestimation of the actual change.

REFERENCES

1. Agassi, M. *Soil Erosion, Conservation and Rehabilitation*; Marcel Dekker Inc.: New York, 1996; 402 pp.
2. Penning de Vries, F.W.T.; Acquai, H.; Molden, D.; Scherr, S.J.; Valentin, C.; Cofie, O. *Integrated Land and Water Management for Food and Environmental Security*, Comprehensive Assessment Research Report 1; International Water Management Institute, Global Environmental Facility: Sri Lanka, Washington, 2002; 70.
3. Ponting, C. *Green History of the Earth*; Cambridge University Press: Cambridge, 1991; 350 pp.
4. Bridges, M.; Hannam, I.; Oldeman, R.; Penning de Vries, F.W.T.; Scherr, S.J.; Sombatpanit, S. *Response to Land Degradation*; Science Publishers: Plymouth, New Delhi, 2001; 510.
5. Scherr, S.J. *Soil Degradation. A Threat to Developing Country Food Security by 2020?* Food, Agriculture and Environment Discussion Paper 27; IFPRI: Washington, 1999; 60 pp.
6. Wood, S.; Sebastian, K.; Scherr, S.J. *Pilot Analysis of Global Ecosystems: Agroecosystems*; International Food Policy Research Institute and World Resources Institute: Washington, 2000; 110 pp.
7. Tengberg, A.; Stocking, M. Erosion-induced loss in soil productivity and its impacts on agricultural production and food security. In *FAO/Agritex Expert Consultation on Integrated Soil Management for Sustainable Agriculture and Food Security in Southern and Eastern Africa*, Harare; Land and Water Development Division, UN Food and Agriculture Organization: Rome, 1997; 42.
8. Conway, G. Agroecology and farming systems research. In *ACIAR Proceedings 11*; Remenyi, J.V., Ed.; Canberra, 1987; 43–59.
9. Greenland, D.J.; Szabolcs, I. *Soil Resilience and Sustainable Land Use*; CABI: Wallingford, 1994; 560 pp.
10. Drechsel, P.; Giordano, M.; Gyiele, L.A. *Valuing Nutrients in Soil and Water: Concepts and Techniques with Examples from IWMI Studies in the Developing World*, IWMI Research Report 82; International Water Management Institute: Sri Lanka, 2000.
11. International Board for Soil Research and Management (IBSRAM). *Farmers' Adoption of Soil-Conservation Technologies*, IBSRAM Proceedings No. 17; IBSRAM: Bangkok, Thailand, 1997; 144.
12. van Lynden, G.W.J.; Oldeman, L.R. *The Assessment of the Status of Human-Induced Soil Degradation in South and Southeast Asia*; UNEP/FAO/ISRIC: Nairobi/Rome, Wageningen, 1997.
13. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *World Map of the Status of Human-Induced Soil Degradation*, 2nd Ed.; ISRIC–UNEP: Wageningen, 1991.
14. Penning de Vries, F.W.T. Food security? We are losing ground fast. In *Crop Science: Progress and Prospects*; Noerberger, J., Geiger, H.H., Struik, P.C., Eds.; 1–14.
15. Lal, R. Erosion–crop productivity relationships for the soils of Africa. *Soil Sci. Soc. Am. J.* **1995**, *59*, 661–667.
16. Craswell, E.T.; Grote, U.; Henao, J.; Vlek, P.L.G. *Nutrient Flows in Agricultural Production and International Trade: Ecological and Policy Issues 2004*, ZEF Discussion Papers Development Policy, 78; University Bonn: Germany, 74 pp.
17. Buresh, R.J.; Sanchez, P.A.; Calhoun, F. *Replenishing Soil Fertility in Africa*, Agronomy Society of America, 51; Soil Science Society of America and ICRAF: Madison, 1997; 252 pp.
18. Noble, A.D.; Gillman, G.P.; Nath, S.; Srivastava, R.J. Changes in the surface charge characteristics of degraded soils in the wet tropics through the addition of beneficiated bentonite. *Aust. J. Soil Sci.* **2001**, *39*, 991–1001.
19. Syers, J.K.; Penning de Vries, F.W.T. *The Sustainable Management of Vertisols*; CABI and IBSRAM: Wallingford, 2001; 400.
20. Rey, C.; Scoones, I.; Toulmin, C. *Sustaining the Soil, Indigenous Soil and Water Conservation in Africa*; Earthscan: London, 1996; 260.

BIBLIOGRAPHY

1. Eswaran, H.; Lal, R.; Reich, P.F. Land degradation: An overview. In *Response to Land Degradation*; Bridges, E.M., Ed.; Oxford: New Delhi, 2001; 20–35.
2. Greenland, D.J.; Gregory, P.J.; Nye, P.H. Summary and conclusions. In *Land Resources: On the Edge of the Malthusian Precipice?* Greenland, D.J., Ed.; CABI, 1998; 1–7.
3. Penning de Vries, F.W.T. Potential and attainable food production and food security in different regions. *Phil. Trans. R. Soc. London B.* **1997**, *352*, 917–928.
4. WOCAT. *FAO Land and Water Digital Media Series 9*. CD-ROM, 2000. Available at <http://www.wocat.net>.

Respiration

Pierre A. Jacinthe

Indiana University—Purdue University Indianapolis, Indianapolis, Indiana, U.S.A.

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Soil respiration refers to the oxidation of organic materials by soil microorganisms, with generation of energy for microbial growth and maintenance, and production of carbon dioxide. The electrons needed for the oxidation of organic compounds originate, in most cases, from oxygen, which, therefore, is consumed by the respiration process. Because respiration is an attribute of all the life forms encountered in soils, it provides an integrative measure of soil biological activity and, appropriately, has extensively been used in soil biology research.

IMPORTANCE OF SOIL RESPIRATION

Soil respiration is a key driver of carbon (C) cycling in terrestrial ecosystems. While microbial decomposition of organic debris is essential to soil organic matter formation, the process also represents a major mechanism of C loss from soil systems. Global soil respiration is estimated at 68–100 Pg C year⁻¹,^[1] representing the second largest carbon dioxide (CO₂) flux in terrestrial ecosystems surpassed only by gross productivity.^[1,2]

As concerns over climate warming grow, it is becoming increasingly more important to document soil respiration in various ecosystems and to understand the biotic, environmental, and management factors affecting this process. Consequently, a tremendous amount of research has been dedicated to the study of soil respiration. It is not the goal of this contribution to provide an in-depth review of the extensive literature that this intensive research has generated. Several comprehensive reviews focusing on various aspects of soil respiration have been published.^[1,3–6] Instead, this entry is a cursory review of issues and knowledge of soil respiration addressed to the wider audience of soil scientists.

COMPONENTS OF SOIL RESPIRATION

The primary avenues of C entry into soil ecosystems are animal and plant debris (aboveground biomass, dead root tissues, and root exudates). Through the activity of soil microbes, these organic materials undergo decomposition with the more labile organic compounds readily converted into CO₂. The decomposition process also leads to the evolution of biochemically resistant secondary metabolites commonly referred to as humus or “stable” pools of soil

organic C (SOC). Depending on residue quality, soil, and climatic conditions, 10–20% of the residue-C returned to soil ends up in the humus pool.^[7] That pool also undergoes further decomposition but at much slower rates (20 to 50 times) than fresh residues. It is also important to note that not all the labile C fractions in soils are decomposed; however, through interaction with clays and inclusion within soil aggregates, some labile organic compounds become physically protected against microbial attack.

During photosynthesis, a fraction of the photosynthate C is translocated belowground. Some of the allocated C is consumed through respiration of root tissues (this is termed “autotrophic respiration”), while another fraction is released into the soil as root exudates, which could be rapidly decomposed by rhizosphere microbes. The decomposition of exudates and dead root tissue results in the production of CO₂, which, combined with autotrophic respiration, defines what is termed as “rhizosphere respiration.” “Total soil respiration,” which is reported in most studies, is defined as the sum of root respiration, litter decomposition, and stable SOC mineralization.

The contribution of rhizosphere respiration to total CO₂ production is variable, ranging from 10% to 90% of total soil respiration,^[4] with a generally higher contribution of root respiration to CO₂ flux in forests and grasslands than in agroecosystems mostly because of the difference in plant species and the duration of vegetative growth.

MEASUREMENTS OF SOIL RESPIRATION

CO₂ production is the most commonly used index of soil respiration intensity. Many procedures have been devised to measure CO₂ emission from soils, but the soil cover method is the most widely used approach.

An automated^[8] or a manually operated chamber of known dimensions (area, volume) is placed over the soil surface allowing CO₂ to accumulate inside. Soil respiration is expressed as the rate of CO₂ accumulation that is determined through: 1) measuring the amount of CO₂ absorbed by an alkaline solution (NaOH and KOH) placed inside the chamber; 2) regular sampling of the chamber headspace to determine change in CO₂ concentration using gas chromatography; and 3) interfacing the chamber with a flow-through system and an infrared gas analyzer for CO₂ determination. These procedures are sometimes combined with C labeling (administration of ¹³C- or ¹⁴C-labeled CO₂ to plants and monitoring the flux of labeled CO₂), root-exclusion techniques (comparing flux in areas with and without living roots) when assessing rhizosphere respiration.^[4] Bowen ratio/energy balance and eddy-covariance techniques have also been used and generally provide a more integrated estimate of flux over large areas.^[8] Review and comparison of these methods^[9,10] have shown that, within the bounds (e.g., duration of chamber closure, air flow speed) of a particular procedure, fluxes obtained with most methods are reasonably comparable. However, the alkali absorption method tends to overestimate fluxes probably because of an artificial lowering of CO₂ concentration at the soil surface by the alkali trap, resulting in an enhancement of respiratory activity, given reports^[10,11] of an enhancement of soil respiration at low CO₂ level.

CONTROLLING FACTORS OF SOIL RESPIRATION

Soil respiration is related to factors controlling the supply of organic carbon to soil microbes as well as factors controlling the abundance, distribution, and activity of the soil biota. Plant residue, dead roots, and root exudates constitute the primary sources of organic material entering soils. Studies conducted at both local and global scales have established the link between organic supply and soil respiration. Soil respiration was reported to increase nearly proportionally with the amount of residue added to soils.^[12] A summary report^[13] of CO₂ flux from 18 forest ecosystems within the Euroflux network concluded that ecosystem productivity, particularly the supply of young organic matter to fuel heterotrophic activity, was the main controller of soil respiration at these sites. A close relationship was also found between the net primary productivity (NPP is photosynthesis minus autotrophic respiration) of various natural and managed ecosystems and their mean annual soil respiration rates.^[1]

Temperature has been identified as the most important driver of soil respiration and several empirical models have been proposed to describe the relationship.^[1,5,6,14] The Q₁₀ coefficient, the relative change in respiration rate for a 10°C variation in temperature, is a commonly used index of the temperature sensitivity of soil respiration. It is derived from the equation $R_2 = R_1 Q_{10}^{(T_2 - T_1)/10}$, where R₁ and R₂ are rates

of respiration at temperatures T₁ and T₂, respectively. Typical Q₁₀ values reported in the literature ranged between 1.3 and 8.^[5] The temperature sensitivity of soil respiration varies inversely with temperature; i.e., at low temperatures, respiratory processes are highly sensitive to temperature variations but the sensitivity diminishes at higher temperatures.^[5,6,15,16]

Moisture is another controlling factor of soil respiration. A control of soil moisture on respiration is most often reported in semiarid regions where moisture deficit during the growing season is commonplace.^[17–19] Studies conducted in these regions have shown that soil CO₂ flux can increase by a factor of 2–5 between a wet and a dry year. A regional study^[18] of C dynamics across the U.S. Great Plains also concluded that, more than temperature and soil texture, soil moisture explained most of the regional variation in SOC decomposition. In a soil displacement experiment in Arizona^[15] in which respiration was monitored in mesocosms that were placed either upslope (more humid and cooler) or downslope to create a range of climatic conditions, it was observed that respiration increased by 40% under the mesic upslope conditions and decreased by 20% in mesocosms relocated in drier environments downslope. These findings further underscore the control of soil moisture on soil respiration in arid environments.

Several empirical models linking respiration to both soil temperature and moisture have been published.^[16,20] A synthesis of available data suggests that, under moist conditions, temperature is the main factor of soil respiration, while under dry conditions (a $\theta = 20\%$ is the proposed cutoff^[21]), temperature and moisture appear to have a multiplicative effect. This interpretation is consistent with reports of strong temperature control on respiration in studies conducted in humid regions,^[21–23] whereas greater effects of soil moisture are recorded in arid regions.^[15,17–19]

RESPIRATION AND CARBON SEQUESTRATION IN SOILS

An increase (1.4–5.8°C) in mean global temperature is predicted in the 21st century.^[24] The response of soil to climate warming has been the subject of considerable debate in the scientific literature. Reports of limited effect,^[3] acceleration,^[6] and adaptation^[25] of soil respiration in response to warming have been published. Given the linkage between temperature and respiratory C loss, one can infer that a warming of global climate would translate into increased transfer of C as CO₂ from soils into the atmosphere, thereby providing a positive feedback on climate warming.^[6] Assuming a Q₁₀ of 1.8, a 2°C increase in soil temperature would result in a 12% increase in global soil respiration (~9 Pg C year⁻¹), which is nearly three times the rate (3.2–3.4 Pg CO₂-C) of CO₂ accumulation in the atmosphere.^[24] On the other hand, it has been argued that increased temperature and atmospheric CO₂ concentration could enhance biomass production. Thus the rate of atmospheric CO₂

transfer into soils as plant biomass would increase.^[26] An analysis of these two conflicting scenarios^[5] concluded that stimulation of decomposition processes with climate warming will likely offset increase in NPP, thereby resulting in a net reduction in SOC storage. That analysis also suggested that changes in SOC storage will be slow and regionally variable with SOC gain in tropical region soils and net SOC loss in cooler regions.

EFFECTS OF MANAGEMENT AND SEASON ON SOIL RESPIRATION

The emission of CO₂ from soils is controlled by production and transport processes, which themselves are regulated by climatic and soil biophysical factors. Human intervention (management practices) can attenuate or enhance the effects of these factors on soil respiration through crop residue distribution, alteration of soil moisture and temperature regime, and disturbance of the soil biophysical environment.

Studies of soil respiration under optimum conditions in the laboratory have consistently found a larger labile C pool in soils amended with organic residue and soils under conservation tillage.^[12,23,27] They have also shown that both total and labile C pools are highly stratified in no-till (NT) soils compared with plowed soils.

We conducted laboratory incubation to determine the effects on soil respiration of tillage and soil disturbance as represented by soil sieving and erosion. In this evaluation, we compared C mineralization in runoff, sieved soil (< 2 mm) and in intact soil cores collected from research plots under NT and plow-till (PT) conditions for 38 years. Our data (Table 1) showed that after 200 days of incubation,

C mineralized in the sieved soil exceeded that in the core samples by 1153 and 146 mg C kg⁻¹ soil in the NT and PT soils, respectively. These figures also provide a clear indication of how disturbances (sieving and erosion) contribute to the release of protected C in NT soils. In conjunction with the high mineralization rate of runoff C, these data also indicate that a large fraction of the C that accumulates in soils through conservation tillage is labile and could be easily lost with soil disturbance.

Unlike the consistently positive effect of tillage on soil respiration observed with incubation studies, field-scale investigations of CO₂ emission in relation to tillage practices have yielded mixed results. Published data include comparable,^[19,27] higher^[19] and lower^[22,28] CO₂ emission from NT soils compared with plowed soils. Other studies^[23,29] have also reported a 10–20% lower CO₂ emission from NT soils compared with plowed soils. These variable results suggest that at the field-scale level, the effect of tillage on CO₂ flux is governed less by labile C availability but rather by soil moisture and temperature. Higher CO₂ emission from plowed soils has been linked to higher soil temperature.^[22]

As the driving variables (temperature, moisture, and organic substrates) vary with time, it follows that soil CO₂ emission exhibits strong seasonal variability with CO₂ flux maxima often recorded during late spring and summer and lowest fluxes generally occurring in the autumn.^[12] Overwinter soil respiration has generally been overlooked. However, several studies^[12,28] have shown that between 4% and 20% of the annual CO₂ emission occurs during the winter to spring-thaw period.

CONCLUSION

The oxidation of organic compounds to CO₂ by soil microbes is a major pathway of C loss in terrestrial ecosystems. Microbial processing of animal and plant residues is essential for the incorporation of biomass-C into SOC pools. These parallel processes (mineralization and biomass conversion) suggest that land use and management practices that optimize residue input and decomposition and that minimize respiratory C loss, likely lead to net increment of soil C stocks in the long run. In this synthesis of the literature, various drivers of soil respiration have been identified. Among them, soil temperature (e.g., with residue on soil surface) and soil disturbance (e.g., conservation tillage) are factors that can be managed to achieve C sequestration in soils. However, when soil respiration is placed in the context of climate warming or other anthropogenic disturbances such as N deposition, it appears that changes in soil respiration will be controlled by complex interactions between several direct and indirect factors. A synthesis of the literature on that issue reveals several schools of thought. These divergent views not only demonstrate the biogeochemical complexity of C flow in

Table 1 Organic carbon mineralization as affected by tillage practice and disturbance.

Tillage	Material incubated ^a	Duration of incubation (days)	CO ₂ evolved (g C kg ⁻¹ soil)
NT, 38 years	Core (5-cm diam, 5-cm L)	200	0.48 ± 0.09
Sieved soil (< 2 mm)	200	1.63 ± 0.02	
Runoff	100	12.4 ± 2.8	
PT, 38 years	Core (5-cm diam, 5-cm L)	200	0.34 ± 0.08
Sieved soil (< 2 mm)	200	0.22 ± 0.06	
Runoff	70	6.7 ± 2.8	

Note: Values are the means ± standard deviation of six observations.

^aAll materials were collected from a Crosby (fine, mixed, mesic Aeric Ochraqualf) soil at 0–10-cm depth. Triplicate cores and sieved soil (20 g) samples were incubated in a 450-ml jar and in 160-ml serum bottles, respectively. Details regarding runoff generation and incubation are available elsewhere.

Source: From Jacinthe, Lal, et al.^[30]

terrestrial systems but also highlight the incompleteness of our understanding of the alteration (rate and direction) in this flow in response to climate warming.

REFERENCES

1. Raich, J.W.; Schlesinger, W.H. The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. *Tellus B* **1992**, *44* (2), 81–99.
2. Mynemi, R.B.; Los, S.O.; Asrar, G. Potential gross primary productivity of terrestrial vegetation from 1982–1990. *Geophys. Res. Lett.* **1995**, *22* (19), 2617–2620.
3. Giardina, C.P.; Ryan, M.G. Evidence that decomposition rates of organic carbon in mineral soil do not vary with temperature. *Nature* **2000**, *404* (6780), 858–861.
4. Hanson, P.J.; Edwards, N.T.; Garten, C.T.; Andrews, J.A. Separating root and soil microbial contributions to soil respiration: A review of methods and observations. *Biogeochemistry* **2000**, *48* (1), 115–146.
5. Kirschbaum, M.U.F. Will changes in soil organic carbon act as a positive or negative feedback on global warming? *Biogeochemistry* **2000**, *48* (1), 21–51.
6. Lloyd, J.; Taylor, J.A. On the temperature dependence of soil respiration. *Funct. Ecol.* **1994**, *8* (3), 315–323.
7. Rasmussen, P.E.; Albrecht, S.L. Crop management effects on organic carbon in semi-arid Pacific northwest soils. In *Management of Carbon Sequestration in Soil*; Lal, R., Kimble, J.M., Follett, R.F.F., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1998; 209–219.
8. McGinn, S.M.; Akinremi, O.O.; McLean, H.D.J.; Ellert, B. An automated chamber system for measuring soil respiration. *Can. J. Soil Sci.* **1998**, *78* (4), 573–579.
9. Janssens, I.A.; Kowalski, A.S.; Longdoz, B.; Ceulemans, R. Assessing forest soil CO₂ efflux: An *in situ* comparison of four techniques. *Tree Physiol.* **2000**, *20* (1), 23–32.
10. Nakadai, T.; Koizumi, H.; Usami, Y.; Satoh, M.; Oikawa, T. Examination of the method for measuring soil respiration in cultivated land—Effect of carbon dioxide concentration on soil respiration. *Ecol. Res.* **1993**, *8* (1), 65–71.
11. Burton, A.J.; Zogg, G.P.; Pregitzer, K.S.; Zak, D.R. Effect of measurement CO₂ concentration on sugar maple root respiration. *Tree Physiol.* **1997**, *17* (7), 421–427.
12. Jacinthe, P.A.; Lal, R.; Kimble, J.M. Carbon budget and seasonal carbon dioxide emission from a central Ohio Luvisol as influenced by wheat residue amendment. *Soil Tillage Res.* **2002**, *67* (2), 147–157.
13. Janssens, I.A.; Lankreijer, H.; Matteucci, G.; Kowalski, A.S.; Buchmann, N.; Epron, D.; Pilegaard, K.; Kutsch, W.; Longdoz, B.; Grunwald, T.; Montagnani, L.; Dore, S.; Rebmann, C.; Moors, E.J.; Grelle, A.; Rannik, U.; Morgenstern, K.; Oltchev, S.; Clement, R.; Gudmundsson, J.; Minerbi, S.; Berbigier, P.; Ibrom, A.; Moncrieff, J.; Aubinet, M.; Bernhofer, C.; Jensen, N.O.; Vesala, T.; Granier, A.; Schulze, E.D.; Lindroth, A.; Dolman, A.J.; Jarvis, P.G.; Ceulemans, R.; Valentini, R. Productivity overshadows temperature in determining soil and ecosystem respiration across European forests. *Glob. Chang. Biol.* **2001**, *7* (3), 269–278.
14. Mielnick, P.C.; Dugas, W.A. Soil CO₂ flux in a tallgrass prairie. *Soil Biol. Biochem.* **2000**, *32* (2), 221–228.
15. Conant, R.T.; Klopatek, J.M.; Klopatek, C.C. Environmental factors controlling soil respiration in three semiarid ecosystems. *Soil Sci. Soc. Am. J.* **2000**, *64* (1), 383–390.
16. Qi, Y.; Xu, M.; Wu, J. Temperature sensitivity of soil respiration and its effects on ecosystem carbon budget: Nonlinearity begets surprises. *Ecol. Model.* **2002**, *153* (1–2), 131–142.
17. Akinremi, O.O.; McGinn, S.M.; McLean, H.D.J. Effects of soil temperature and moisture on soil respiration in barley and fallow plots. *Can. J. Soil Sci.* **1999**, *79* (1), 5–13.
18. Epstein, H.E.; Burke, I.C.; Lauenroth, W.K. Regional patterns of decomposition and primary production rates in the US Great Plains. *Ecology* **2002**, *83* (2), 320–327.
19. Franluebbbers, A.J.; Hons, F.M.; Zuberer, D.A. Tillage and crop effects on seasonal dynamics of soil CO₂ evolution, water content, temperature, and bulk density. *Appl. Soil Ecol.* **1995**, *2* (2), 95–109.
20. Epron, D.; LeDantec, V.; Dufrene, E.; Granier, A. Seasonal dynamics of soil carbon dioxide efflux and simulated rhizosphere respiration in a beech forest. *Tree Physiol.* **2001**, *21* (2–3), 145–152.
21. Rey, A.; Pegoraro, E.; Tedeschi, V.; De Parri, I.; Jarvis, P.G.; Valentini, R. Annual variation in soil respiration and its components in a coppice oak forest in Central Italy. *Glob. Biogeochem. Cycles* **2002**, *8* (9), 851–866.
22. Wagai, R.; Brye, K.R.; Gower, S.T.; Norman, J.M.; Bundy, L.G. Land use and environmental factors influencing soil surface CO₂ flux and microbial biomass in natural and managed ecosystems in southern Wisconsin. *Soil Biol. Biochem.* **1998**, *30* (12), 1501–1509.
23. Paul, E.A.; Harris, D.; Collins, H.P.; Schulthess, U.; Robertson, G.P. Evolution of CO₂ and soil carbon dynamics in biologically managed, row-crop agroecosystems. *Appl. Soil Ecol.* **1999**, *11* (1), 53–65.
24. IPCC. Climate change 2001: The scientific basis. In *Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*; Houghton, J.T., Ding, Y., Griggs, D.J., Noguer, M., van der Linden, P.J., Dai, X., Maskell, K., Johnson, C.A., Eds.; Cambridge University Press: Cambridge, 2001.
25. Luo, Y.Q.; Wan, S.Q.; Hui, D.F.; Wallace, L.L. Acclimatization of soil respiration to warming in a tall grass prairie. *Nature* **2001**, *413* (6856), 622–625.
26. Schimel, D.S.; Braswell, B.H.; McKeown, R.; Ojima, D.S.; Parton, W.J.; Pulliam, W. Climate and nitrogen controls on the geography and timescales of terrestrial biogeochemical cycling. *Glob. Biogeochem. Cycles* **1996**, *10* (4), 677–692.
27. Alvarez, R.; Santanoglia, O.; Garcia, R. Plant and microbial contribution to soil respiration under zero and disk tillage. *Eur. J. Soil Biol.* **1996**, *32* (4), 173–177.
28. Kessavalou, A.; Mosier, A.R.; Doran, J.W.; Drijber, R.A.; Lyon, D.J.; Heinemeyer, O. Fluxes of carbon dioxide, nitrous oxide, and methane in grass sod and winter wheat-fallow tillage management. *J. Environ. Qual.* **1998**, *27* (5), 1094–1104.
29. Fortin, M.C.; Rochette, P.; Pattey, E. Soil carbon dioxide fluxes from conventional and no-tillage small-grain cropping systems. *Soil Sci. Soc. Am. J.* **1996**, *60* (5), 1541–1547.
30. Jacinthe, P.A.; Lal, R.; Kimble, J.M. Carbon dioxide evolution in runoff from simulated rainfall on long-term no-till and plowed soils in southwestern Ohio. *Soil Tillage Res.* **2002**, *66* (1), 23–33.

Restoration Ecology

Richard Hobbs

Murdoch University, Murdoch, Western Australia, Australia

Abstract

The term “ecological restoration” covers a wide range of activities involved with the repair of damaged or degraded ecosystems. An array of terms has been used to describe these activities including restoration, rehabilitation, reclamation, reconstruction, and reallocation. Generally, restoration is used to describe the complete reassembly of a degraded system to its undegraded state, while rehabilitation describes efforts to develop some sort of functional protective or productive system on a degraded site.

INTRODUCTION

Restoration ecology is the science behind ecological restoration, which covers a wide range of activities involved with the repair of damaged or degraded ecosystems, and is usually carried out for one of the following reasons:

1. To restore highly disturbed, but localized sites, such as mine sites
2. To improve productive capability in degraded production lands
3. To enhance nature conservation values in protected landscapes
4. To restore ecological processes over broad landscape-scale or regional areas.

Ecological restoration occurs along a continuum from the rebuilding of totally devastated sites to the limited management of relatively unmodified sites. Restoration aims to return the degraded system to some form of cover that is protective, productive, esthetically pleasing, or valuable in a conservation sense.^[1] A further tacit aim is to develop a system that is sustainable in the long term.

The conceptual basis of restoration ecology has developed rapidly.^[2–6] The term “ecological restoration” covers a wide range of activities involved with the repair of damaged or degraded ecosystems. An array of terms has been used to describe these activities including restoration, rehabilitation, reclamation, reconstruction, and reallocation. Generally, restoration is used to describe the complete reassembly of a degraded system to its undegraded state, while rehabilitation describes efforts to develop some sort of functional protective or productive system on a degraded site. In addition, some authors also use the term “reallocation” to describe the transfer of a site from one land use to a more productive or otherwise beneficial use. Here I will follow^[1] and use the term restoration to refer broadly to activities that aim to repair damaged systems, although the other terms are used as above in particular examples.

Restoration ecology involves a number of interconnected activities, as summarized in Fig. 1.

DYNAMIC SYSTEMS AND RESTORATION GOALS

Ecosystem characteristics, which may be used when considering restoration goals, include as follows:^[1]

1. Composition: species present and their relative abundances
2. Structure: vertical arrangement of vegetation and soil components (living and dead)
3. Pattern: horizontal arrangement of system components
4. Heterogeneity: a complex variable made up of components 1–3
5. Function: performance of basic ecological processes (energy, water, nutrient transfers)
6. Species interactions: includes pollination, seed dispersal, etc.
7. Dynamics and resilience: succession and state-transition processes, recovery from disturbance.

Ecosystems are naturally dynamic entities, and hence the setting of restoration goals in terms of static compositional or structural attributes is problematic. Often, past system composition or structure is unknown or partially known, and past data provide only static snapshots of system parameters. Existing undegraded reference systems can therefore act as potential reference systems against which the success of restoration efforts in degraded systems can be measured. An alternative approach is to recognize explicitly the dynamic nature of ecosystems and to accept that there are a range of potential short- and long-term outcomes of restoration projects. The aim should be to have a transparent and defensible method of setting goals for restoration, which focuses on the desired characteristics for the system *in the future*, rather than in relation to what these were in the past. Where it is impossible or extremely expensive to restore composition and structure, alternative goals

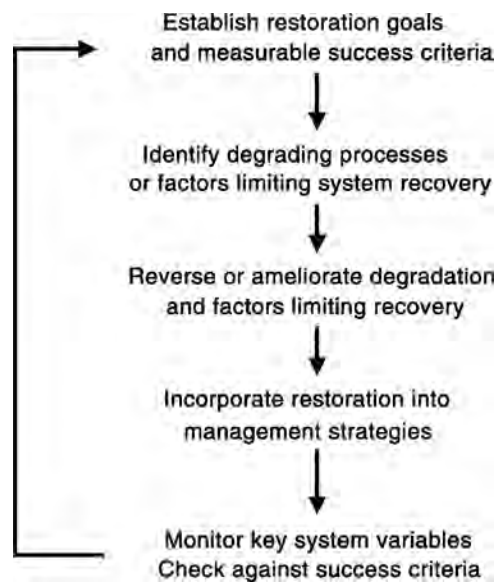


Fig. 1 Processes involved in ecological restoration.
Source: From Hobbs.^[7]

are appropriate. These may aim to repair damage to ecological function or ecosystem services, or to create a novel system using species not native to the region or those suited to changed environmental conditions.

Goal setting thus becomes an extremely important component of the restoration process. Goals for a particular site, or more broadly for a landscape, will need to be determined iteratively by considering the ecological potential for restoration and matching this against societal desires. This argues for an adaptive approach to restoration, which garners ecological knowledge from as many sources as possible (including on-ground practitioners), and uses this knowledge to develop ecological response models that can indicate the likely outcomes of restoration activities. Which of the restoration options is taken up is decided on the basis of the stakeholder expectations and goals, and the extent to which it is implemented depends on the degree of financial and resource input from various sectors, including individual investment and public subsidy or incentives.

RESTORATION OPTIONS

Arriving at clear restoration goals requires a clear picture of the restoration options available for a particular site, landscape or region. Restoration activities need to be prefaced by a rigorous assessment of the current state of the particular system or landscape, and the underlying factors leading to that state. Once this has been achieved, a clearer picture of the necessary restoration activities is possible, and a range of restoration options can be determined (Figs. 2 and 3).

Ecological restoration is required only where the system's resilience has been diminished in some way, or where



Fig. 2 Mine site in *Eucalyptus marginata* forest in Western Australia, following soil removal for extraction of alumina.

the normal recovery processes are too slow to achieve management goals within a desirable time frame. Restoration requires that the stressors acting on the system are removed and may also involve replacing components that have been lost during the degradation of the system. Degrading processes can result in a variety of ecosystem responses, depending on the intensity, duration, and scale of the impact, and on which system components and processes are affected. Stressors, which impair the processes of resource capture by plants (e.g., soil erosion, alteration of hydrology), are likely to have much greater impacts than stressors that remove or damage plants or consumers (e.g., pathogens or unsuitable fire regime).

A general feature of many systems seems to be the potential for the system to exist in a number of possible states, and the likelihood that restoration thresholds exist, which prevent the system from returning to a less-degraded state without the input of management effort.^[1] It has been suggested that two main types of such thresholds are likely, one that is caused by biotic interactions and the other



Fig. 3 Mine site in *Eucalyptus marginata* forest in Western Australia, following restoration involving soil ripping, topsoil replacement, and seeding with native species.

caused by abiotic limitations.^[6] If the system has degraded mainly due to biotic changes (such as grazing-induced changes in vegetation composition), restoration efforts need to focus on biotic manipulations that remove the degrading factor (e.g., the grazing animal) and adjust the biotic composition (e.g., replant desired species). If, on the other hand, the system has degraded due to changes in abiotic features (such as through soil erosion or contamination), restoration efforts need to focus first on removing the degrading factor and repairing the physical and/or chemical environment. In the latter case, there is little point in focusing on biotic manipulation without first tackling the abiotic problems.

The above argument is akin to ensuring that system functioning is corrected or maintained before questions of biotic composition and structure are considered. Considering system function provides a useful framework for the initial assessment of the state of the system and the subsequent selection of repair measures.^[8,9] Where function is not impaired, restoration can legitimately focus on composition and structure as parameters to be considered when setting goals.

MEASUREMENTS OF SUCCESS

There have been numerous attempts to provide categories of assessment that will contribute to a picture of the "healthy ecosystem," which have varying degrees of ease of measurement. Biological potential inventory is probably the earliest form of ecosystem assessment, typified by the species list. Although this can be extremely useful for assessing conservation status and is greatly improved by measurements over time, it often does not get to the basics of what is causing the degradation, rather simply reflecting the magnitude and direction of its effect.

More complex measurements of biological integrity can assess food-web complexity and the development of symbiotic relationships. However, difficulty of assessment increases greatly. Measurements relating to ecosystem function can include estimates of production, standing crop, mass balance, and mineral cycling pools and rates. The problem with all these measures lies in determining what the target should be, in relation to the problems discussed above concerning reference systems. More generally, indicators of soil quality improvement with restoration can include factors such as pH, organic matter content,

physical properties such as texture, erodibility, compaction, water-holding capacity and drainage, and chemical properties such as toxicities associated with heavy metals and other contaminants, and the availability of nitrogen, potassium, and phosphorus.^[10]

Measures of success have to be linked back to the clear definitions of goals for restoration. Assessment processes can be complicated and expensive, and if they are too complicated or expensive they will not be carried out. There is no point in assessing something unless it relates to specific goals. These goals can be set in relation to data on the preexisting ecosystem, or in relation to adjacent systems, or can be settled on by discussion with stakeholders about what may be possible and desirable on the site.

REFERENCES

1. Hobbs, R.J.; Norton, D.A. Towards a conceptual framework for restoration ecology. *Restor. Ecol.* **1996**, *4*, 93–110.
2. Allen, E.B.; Covington, W.W.; Falk, D.A. Developing the conceptual basis for restoration ecology. *Restor. Ecol.* **1997**, *5*, 275–276.
3. Aronson, J.; Floret, C.; Le Flo'h, E.; Ovalle, C.; Pontanier, R. Restoration and rehabilitation of degraded ecosystems in arid and semiarid regions. I. A view from the south. *Restor. Ecol.* **1993**, *1*, 8–17.
4. Aronson, J.; Le Flo'h, E.; Floret, C.; Ovalle, C.; Pontanier, R. Restoration and rehabilitation of degraded ecosystems in arid and semiarid regions. II. Case studies in Chile, Tunisia and Cameroon. *Restor. Ecol.* **1993**, *1*, 168–187.
5. Urbanska, K.M.; Webb, N.R.; Edwards, P.J. *Restoration Ecology and Sustainable Development*; Cambridge University Press: Cambridge, 1997.
6. Whisenant, S.G. *Repairing Damaged Wildlands: A Process-Oriented, Landscape-Scale Approach*; Cambridge University Press: Cambridge, 1999.
7. Hobbs, R.J. Restoration of disturbed ecosystems. In *Ecosystems of the World 16. Disturbed Ecosystems*; Walker, L., Ed.; Elsevier: Amsterdam, 1999; 673–687.
8. Tongway, D.J.; Ludwig, J.A. Rehabilitation of semi-arid landscapes in Australia. I. Restoring productive soil patches. *Restor. Ecol.* **1996**, *4*, 388–397.
9. Ludwig, J.; Tongway, D.; Freudenberg, D.; Noble, J.; Hodgkinson, K., Eds. *Landscape Ecology, Function and Management: Principles from Australia's Rangelands*; CSIRO Publishing: Melbourne, 1997.
10. Harris, J.; Birch, P.; Palmer, J. *Land Restoration and Reclamation: Principles and Practice*; Longman: Harlow, 1996.

Restoration: Success and Completion Criteria

Sam C. Ward

Bicton, Western Australia, Australia

Abstract

Restoration of a disturbed area entails restoring the land, so that the predisturbance conditions are replicated as closely as possible with all the area's environmental values intact. The term generally applies to the restoration of native ecosystems. Any project that aims to repair degraded land must have clear goals. This entry refers to success and completion criteria for restoration. It includes a case study of the restoration of a eucalyptus forest after mining to demonstrate the level of success and the completion criteria that have been used in a major project that is restoring around 500 ha of forest each year.

INTRODUCTION

Restoration of a disturbed area entails restoring the land, so that the predisturbance conditions are replicated as closely as possible with all the area's environmental values intact.^[1] The term generally applies to the restoration of native ecosystems. Approaches to success and completion criteria vary in different countries. In the United States, the system relies on vigorously enforced regulation, whereas in other countries, including Australia, a more flexible system of guidelines for rehabilitation or restoration is used.

Any project that aims to repair degraded land must have clear goals. Without goals, there is no way of determining whether a project is successful. This entry refers specifically to success and completion criteria for restoration. It includes a case study of the restoration of a eucalyptus forest after mining to demonstrate the level of success and the completion criteria that have been used in a major project that is restoring around 500 ha of forest each year.

Restoration infers that some aspects of the following characteristics of the area are restored (Adapted from Hobbs and Norton^[2]):

1. Composition: species' presence and their relative abundances;
2. Structure: vertical arrangement of vegetation and soil components;
3. Pattern: horizontal arrangement of system components;
4. Heterogeneity: a complex variable made up from the three variables above plus heterogeneity of soil characteristics, litter distribution, etc.;
5. Function: performance of basic ecological processes (energy, water, and nutrient transfers); and
6. Dynamics and resilience: successional processes, recovery from disturbance.

Debate surrounds the use of reference ecosystems against which restoration projects should be assessed. Pickett and Parker^[3] consider that choosing a reference state or system is a "trap" and a "pitfall" to be avoided. They make the point that natural systems are dynamic, rather than static and predictable, and therefore contend that it is futile to try to recreate an ecosystem, as it was at a particular time. However, Aronson et al.^[4] argue that for any restoration project, there must be a reference against which the results of the project can be assessed. Aronson et al.^[5] define an "ecosystem of reference" as "some standard of comparison and evaluation, even if the choice is somewhat arbitrary." Restoration after coal mining in the United States uses land that is representative of the premining abiotic and biotic conditions and large enough to withstand natural and man-induced perturbations as reference areas.^[6] Whether or not an "ecosystem of reference" is adopted, some of the desirable characteristics of the target ecosystem must be defined and measured in the developing restored ecosystem before restoration can be considered successful.

ENVIRONMENTAL INDICATORS

It is obvious that not all of the characteristics of a restored ecosystem can be measured. What needs to be measured are a number of indicators that summarize the characteristics of systems and make it possible to gauge the general status of a system. Environmental indicators are physical, chemical, biological, or socioeconomic measures that best represent the key elements of a complex ecosystem or environmental issues.^[7] The "Holy Grail" of restoration ecology is one measure that integrates all aspects of ecosystem function and structure into one or a small number of "one-off" measurements. Unfortunately, a single indicator or index (a mathematical combination of a number of indicators) is unlikely to be able to represent anything as complex as the successful restoration of an ecosystem.

For a general rehabilitation project, where the object is to return a stable, vegetated landform that does not replicate any existing predisturbance ecosystem, relatively simple methods to judge rehabilitation success may be appropriate. For example, a system called Ecosystem Function Analysis (EFA) has been developed in a project funded by the Australian mining industry.^[8] EFA consists of the following three modules: landscape function analysis, vegetation development, and habitat complexity. The system conceives rehabilitation as being land degradation in reverse with successful rehabilitation becoming more able to regulate the movement, and minimize the losses, of vital ecosystem resources such as water, topsoil, nutrients, and organic matter. This method offers a simple methodology for assessing the sustainability of a rehabilitated ecosystem and is especially useful for small or “one-off” rehabilitation efforts where financial and other resources may be limited.

For large restoration projects, especially those where rehabilitation is progressive over a long period, a more intensive assessment effort is required that can assess progression toward a specific end. The assessment should also be able to give feedback, so that restoration procedures on areas that are still undergoing rehabilitation can be continually improved.

General criteria for assessing all restoration projects have not been established. A large number of potential indicators of restoration success could be adopted. The challenge for the managers of any restoration project is to select the most relevant indicators for them to make decisions on the management of a restored area. The indicators ideally should be simple, repeatable, reliable, affordable, sensitive, and able to be aggregated with other measures. Indicators that can be measured repeatedly are valuable, as a time series of measurements allows the trajectory of the ecosystem to be visualized. “One-off” measurements often do not provide an accurate representation of a dynamic ecosystem.

Aronson et al.^[5] proposed a number of “vital ecosystem attributes” that may serve as indicators of ecosystem structure and function. Examples of attributes of ecosystem structure are perennial species richness, annual species richness, total plant cover, aboveground phytomass, beta diversity, life-form spectrum, keystone species, microbial biomass, and soil biota diversity. Examples of attributes of ecosystem function are biomass productivity, soil organic matter, maximum available soil water reserves, coefficient of rainfall efficiency, rain-use efficiency, length of water availability period, nitrogen-use efficiency, micro-symbiont effectiveness, and cycling indices.

Hobbs and Norton^[2] suggested that these attributes were a good starting point for developing a list of parameters (indicators) to be measured to assess rehabilitation development and success. Smyth and Dearden^[6] proposed a two-stage approach to measuring rehabilitation success. The first stage is successful landscaping and soil reconstruction. Soils hold some of the most important non-renewable

resources of an ecosystem,^[9] and the depth of soil available for root growth is a fundamental determinant of which plant species will establish and survive. The development of a stable soil with suitable physical and chemical characteristics is a basic requirement for successful restoration. The second stage is the evaluation of functional capacity and long-term successional trends. Ecosystems may not develop in an ordered and gradual manner but may undergo rapid transitions between different metastable states.^[10,11] The dynamic nature and possible multiple metastable states of natural ecosystems mean that the target values for “vital ecosystem attributes” will need to be defined with a mean and a variance. The values of the combined set of parameters in rehabilitated areas can then be compared against the target values or the natural variability found in “reference ecosystems” to provide a “scorecard” for the rehabilitation project.^[2]

COMPLETION CRITERIA

Completion criteria have been defined in a number of ways^[12,13] but can be considered as qualitative or quantitative indicators against which a restoration or rehabilitation project can be assessed, so that responsibility for the area can be relinquished. They may include physical, biological, water quality, and safety indicators. It is generally accepted that completion criteria need to be developed on a site-specific basis.^[13,14] This can be difficult and time-consuming and requires a detailed knowledge of the ecosystem being restored. For example, Smyth and Dearden^[6] present a flowchart of reclamation planning, implementation, and monitoring for surface coal mine rehabilitation in the United States. Identifying the major impediments to restoration can help in selecting the appropriate indicators.

Completion criteria for land disturbed by mining, construction, or other human-induced degradation serve two main purposes. First, they give both the landowner or land manager (e.g., private citizens or companies, or local, state, or federal government departments) and the parties responsible for the disturbance (e.g., mining companies) a clear direction for restoration. Second, they describe the state of the restored area and surrounding lands that were influenced by the disturbance at which any financial obligation or legal responsibilities can be relinquished by those responsible for the disturbance.

CASE STUDY

An example of established completion criteria comes from the restoration of the jarrah (*Eucalyptus marginata* Donn ex Smith) after bauxite mining in Southwest Australia, where Alcoa World Alumina, Australia, mines and restores around 500 ha of jarrah forest each year. Alcoa’s rehabilitation objective is “to establish a self-sustaining jarrah forest ecosystem planned to enhance or maintain water,

Table 1 Criteria used by Alcoa World Alumina to assess the establishment, development, and growth of the flora on rehabilitated bauxite mines in Southwest Australia.

Criteria and intent	Timing	Guidelines for acceptance	Standard	Corrective action
Has the area been deep ripped?	During rehabilitation	All ripping must prevent water runoff and soil erosion Rielines must not discharge water into forest Ripping must be according to criteria established	No uncontrolled water runoff or soil erosion	Areas of unacceptable ^a erosion to be reworked and erosion control methods applied
Have areas of caprock been fractured by blasting or ripping?	During rehabilitation	All caprock should have been broken by ripping or blasting. Small areas less than 0.1 ha are acceptable	No area greater than 0.1 ha has unbroken caprock	Areas of caprock not broken over 0.1 of a hectare must be broken and revegetated
Is there an adequate cover of topsoil?	During rehabilitation	Topsoil should be spread across the whole rehabilitated area. Areas (less than 0.5 ha) not receiving topsoil are acceptable provided these areas do not exceed 10% of the rehabilitated area	Topsoil is spread over a minimum of 90% of the rehabilitated area	Topsoil or additional seed/fertilizer may be spread over the bare areas
Are there adequate numbers of both jarrah and marri?	9 Months after rehabilitation	Rehabilitated areas must have a stocking rate which will meet proposed land use	An average of 1300 stems ha ⁻¹ to be present at 9 months over 65% of the pit, of which at least 200 stems are marri to ensure that dieback does not devastate the site	Rehabilitated areas not meeting the standard will be re-planted as required
Is there an adequate legume content?	9 Months after rehabilitation	Areas to have at least one legume/m ² . Areas up to 0.5 ha not meeting the standard are acceptable provided they are not greater than 10% of the pit	1 legume m ⁻² based on 9-month establishment monitoring	Areas will be scarified and seeded with legumes
Are there any bare areas other than sumps greater than 0.5 ha?	9 Months after rehabilitation	As above	There are no areas greater than 0.5 ha with less than 1 legume m ⁻²	Areas of 0.5 ha or greater not stocked at the rate of 1 legume m ⁻² to be reseeded the following autumn
Is there an appropriate species richness?	15 Months after rehabilitation	Areas to have a representative number of forest species present	Minimum of 50% of species in forest controls based on 15-month monitoring	Areas may need to receive additional seed. Sites to be scarified and seeded
Are there adequate stocking rates of eucalypts and understory species capable of regenerating after a wildfire?	Approximately 10 years after rehabilitation	Guidelines for acceptance still being established based on fire recovery patterns	The site is capable of recovering from a wildfire	Treatments may be needed such as reseeded or thinning of areas
Are there adequate numbers of both jarrah and marri?	Approximately 15–20 years after rehabilitation	Numbers need to be adequate to meet the designated land use. Tree crowns need to be healthy, of suitable density and in proportion to their tree height	Minimum of 300 stems ha ⁻¹ with the potential to produce trees with sawn timber potential	A thinning may be required

^aUnacceptable erosion is that which poses an accident hazard to a walking human or could lead to uncontrolled water discharge from a pit.Source: From Elliott, Gardner, et al.^[17]

timber, recreation, conservation and/or other forest values.” The restoration process involves landscaping, soil return, ripping, seeding, and fertilizing. Restoration is described more fully by Nichols et al.^[15] and Ward et al.^[16]

According to Elliott et al.^[17] the restored areas must:

1. Meet land use objectives.
2. Be integrated into the landscape.
3. Exhibit sustainable growth and development.
4. Be able to be integrated with forest management.
5. Ensure that vegetation is resilient to disturbance.

Jarrah forests may take several centuries before attaining maturity, much longer than it is practical for Alcoa to maintain responsibility for the area. The company developed completion criteria, with input from all the stakeholders, that can be measured during restoration or relatively early in the development of the restored ecosystem. Alcoa's completion criteria consist of both assessments of the quality of the restoration procedures and indicators of restoration development and success, similar to the two-stage strategy proposed by Smyth and Dearden.^[6] Any impediments to successful rehabilitation need to be identified as early as possible. Early identification of problems may allow them to be remedied, or for rehabilitation procedures to be changed in subsequent years.

Restoration of the jarrah forest after mining demands that tree roots have access to water stored deep in the soil profile, propagules of a wide range of species are available, adequate nutrients are available, and soil conditions are suitable for the establishment and growth of the flora. It is impossible to measure all the indicators of ecosystem function or structure that may be of interest on every hectare of restored land. Different indicators are measured at different times after the restoration process. Indicators are measured at different intensities for a number of reasons, not the least of which is cost. Examples of the criteria used to assess the establishment, growth, and development of the flora are given in Table 1. The establishment of eucalypts and legumes is assessed each year on transects that cover all of that year's rehabilitation. Plant species richness is measured on randomly located plots within each 2–4 ha of 15-month-old rehabilitation.

When all these criteria are met, monitoring and research that have been carried out in the past give a degree of confidence that the ecosystem is on a trajectory leading to a sustainable, diverse jarrah forest that does not require extra management costs, above those required for the native forest. Many examples of measures of ecosystem structure or function on Alcoa's bauxite mines of various ages, which are similar to, or trending toward, values found in the unmined jarrah forest, have been published. Examples include soil nitrogen,^[18,19] litter production,^[18,19] litter nutrient content,^[18,19] total ecosystem nitrogen,^[18,19,20] microbial

biomass and microbial quotient,^[21,22] botanical diversity (species richness),^[23,24] the abundance of VA mycorrhiza,^[25–27] depth to the water table,^[28] and the resilience to fire of the major plant species.^[29,30]

REFERENCES

1. EPA. *Best Practice Environmental Management in Mining—Rehabilitation and Revegetation*; Commonwealth of Australia Environment Protection Agency: Canberra, 1995.
2. Hobbs, R.J.; Norton, D.A. Towards a conceptual framework for restoration ecology. *Restor. Ecol.* **1996**, *4*, 93–110.
3. Pickett, S.T.A.; Parker, V.T. Avoiding the old pitfalls: Opportunities in a new discipline. *Restor. Ecol.* **1994**, *2*, 75–79.
4. Aronson, J.; Dhillon, S.; Le Floch, E. On the need to select an ecosystem of reference, however imperfect: A reply to pickett and parker. *Restor. Ecol.* **1995**, *3*, 1–3.
5. Aronson, J.; Floret, C.; Le Floch, E.; Ovalle, C.; Pontainier, R. Restoration and rehabilitation of degraded ecosystems in arid and semi-arid lands. I. A view from the south. *Restor. Ecol.* **1993**, *1*, 8–17.
6. Smyth, C.R.; Dearden, P. Performance standards and monitoring requirements of surface coal mine reclamation success in mountainous jurisdictions of western North America: A review. *J. Environ. Manage.* **1998**, *53*, 209–229.
7. Saunders, D.; Margules, C.; Hill, B. *Environmental Indicators for National State of the Environment Reporting—Biodiversity, Australia*. State of the Environment (Environmental Indicator Reports), Department of the Environment, Canberra, 1998.
8. Tongway, D.; Hindley, N.; Ludwig, J.; Kearns, A.; Barnett, G. Early indicators of ecosystem rehabilitation on selected minesites. In *Proceedings of 22nd Annual Environmental Workshop*, Adelaide, South Australia, Oct, 1997; Minerals Council of Australia: Canberra, 1997.
9. Bradshaw, A.D. The importance of soil ecology in restoration science. In *Restoration Ecology and Sustainable Development*; Urbanski, K.M., Webb, N.R., Edwards, P.J., Eds.; Cambridge University Press: Cambridge, 1997; 33–65.
10. Westoby, M.; Walker, B.; Noy-Meir, I. Opportunistic management for rangelands not at equilibrium. *J. Range Manage.* **1989**, *42*, 266–274.
11. Hobbs, R.J. Dynamics of vegetation mosaics: Can we predict responses to global change. *Ecoscience* **1994**, *1*, 346–356.
12. Farrell, T.P. Some considerations in planning for mine decommissioning. In *Proceedings of 18th Annual Environmental Workshop*, Burnie, Tasmania, Oct 24–29, 1993; Australian Mining Industry Council: Canberra, 1993; 235–247.
13. Tacey, W.; Treloar, J. What do we want completion criteria to achieve? In *Proceedings of 19th Annual Environmental Workshop*, Karratha, Western Australia, Oct 9–14, 1994; Australian Mining Industry Council: Canberra, 1994; 246–256.
14. Hollands, K. Lease relinquishment in New South Wales—completion criteria. In *Proceedings of 18th Annual Environmental Workshop*, Burnie, Tasmania, Oct 24–29, 1993; Australian Mining Industry Council: Canberra, 1993; 223–234.

15. Nichols, O.G.; Carbon, B.A.; Colquhoun, I.J.; Croton, J.T.; Murray, N.J. Rehabilitation after bauxite mining in south-western Australia. *Landscape Plan.* **1985**, *12*, 75–92.
16. Ward, S.C.; Slessar, G.C.; Glenister, D.J. Environmental resource practices of Alcoa of Australia Ltd. In *Australasian Mining and Metallurgy*; Woodcock, J.T., Hamilton, J.K., Eds.; 2nd Ed.; Australian Institute of Mining and Metallurgy: Melbourne, 1993; Vol. 1, 104–108.
17. Elliott, P.; Gardner, J.H.; Allen, D.; Butcher, G. Completion criteria for Alcoa of Australia limited's bauxite mine rehabilitation. In *Proceedings of 21st Annual Environmental Workshop*, Newcastle, New South Wales, Oct. 14–18, 1996; Minerals Council of Australia: Canberra, 1996; 79–89.
18. Ward, S.C.; Koch, J.M. Biomass and nutrient distribution in a 15-year old forest growing on a rehabilitated bauxite mine. *Aust. J. Ecol.* **1996**, *21*, 309–315.
19. Ward, S.C.; Pickersgill, G.E. Nutrient distribution in two eucalypt plantations growing on rehabilitated bauxite mines. *Aust. J. Ecol.* **1985**, *10*, 111–124.
20. Koch, J.M. Nitrogen accumulation in a rehabilitated bauxite mined area in the darling range, western Australia. *Aust. For. Res.* **1987**, *17*, 59–72.
21. Sawada, Y.; Sparling, G.P.; Jasper, D.A. Use of microbial biomass and activity indices to asses mine-site rehabilitation. In *Proceedings, Soil 1994, Australian Society Soil Science Incorporated*, Conference Bunbury, July, 1994; Australian Society Soil Science: Perth.
22. Sawada, Y. Indices of microbial biomass and activity to asses minesite rehabilitation. In *Papers, Environmental Workshop 1996*, Newcastle, New South Wales, Oct 7–11, 1996; Minerals Council of Australia: Canberra, 1996.
23. Koch, J.M.; Ward, S.C. Establishment of understorey vegetation for rehabilitation of bauxite-mined areas in the jarrah forest of western Australia. *J. Environ. Manage.* **1994**, *41*, 1–15.
24. Ward, S.C.; Koch, J.M.; Ainsworth, G.L. The effect of timing of rehabilitation procedures on the establishment of a jarrah forest after bauxite mining. *Restor. Ecol.* **1996**, *4*, 19–24.
25. Jasper, D.A.; Abbott, L.K.; Robson, A.D. The loss of VA mycorrhizal infectivity during bauxite mining may limit the growth of *Acacia Pulchella* R. Br. *Aust. J. Bot.* **1989**, *37*, 33–42.
26. Hutton, B.J.; Dixon, K.W.; Sivasithamparam, K.; Pate, J.S. Effect of habitat disturbance on inoculum potential of ericoid endophytes of western Australian heaths (Epacridaceae). *New Phytol.* **1997**, *135*, 739–744.
27. Gardner, J.H.; Malajczuk, N. Recolonisation of rehabilitated bauxite mine sites in western Australia by mycorrhizal fungi. *For. Ecol. Manage.* **1988**, *24*, 27–42.
28. Ruprecht, J.K.; Ainsworth, G.L.; Lareau, N.G.; Schofield, N.J. *Groundwater and Vegetation Response to Mining and Subsequent Rehabilitation within Del Park Catchment, South-West Western Australia*. Report No., WS 67, Water Authority of Western Australia, Perth, 1990.
29. Grant, C.D.; Koch, J.M.; Bell, D.T.; Loneragan, W.A. Tree species response to prescribed burns in rehabilitated bauxite mines in western Australia. *Austral. For.* **1997**, *60*, 84–89.
30. Grant, C.D.; Koch, J.M.; Bell, D.T.; Loneragan, W.A. The effect of burning, soil scarification and seeding on the understorey composition of 12 year old rehabilitated bauxite mines in western Australia. *Austral. For.* **1997**, *60*, 16–23.

Rice Paddies: Methane Emissions

Ronald L. Sass

Department of Ecology and Evolutionary Biology, Rice University, Houston, Texas, U.S.A.

Abstract

Methane (CH₄) emission from rice fields is the result of soil bacterial processes, i.e., production in flooded anaerobic microsites and consumption (oxidation) in aerobic microsites. The flooding of rice fields promotes the anaerobic fermentation of carbon sources supplied by the rice plants and other incorporated organics, resulting in the formation of CH₄. CH₄ reaches the atmosphere primarily through a gas-conducting system in the rice plants and by the ebullition of gas bubbles and diffusion. The process is governed by a complex set of parameters linking the physical and biological characteristics of flooded soil environments with specific agricultural management practices.

INTRODUCTION

Methane (CH₄) is an important greenhouse gas with a global warming potential approximately 21 times that of carbon dioxide. Of the agricultural sources that may account for as much as 40% of the total annual emission of atmospheric CH₄, a significant portion is contributed by rice cultivation. As one of the major crops of the world, rice accounts for approximately 23% of its per capita calorie intake. Modern estimates suggest that 50 ± 20 teragrams (Tg)/yr (1 Tg = 1 million metric tons) of CH₄ is emitted from global rice paddies. CH₄ is produced under anaerobic conditions generated in flooded rice fields. Irrigated rice fields account for 51%, rainfed fields for 27%, and deep-water fields for 8% of the world's total harvested rice lands. Dry upland rice lands (14%) are not flooded for any significant duration of time and do not produce CH₄.^[1]

FACTORS AFFECTING CH₄ EMISSIONS

Diel and Seasonal Patterns of CH₄ Emission

A positive dependence on temperature is observed in the diel pattern of CH₄ emission in rice fields and in incubation studies.^[1] Seasonal variations in CH₄ emission from rice paddies are more complex and differ among studies. Growth-seasonal CH₄ fluxes observed in temperate rice fields show a general correlation with temperature but respond primarily to seasonal trends in plant development and carbon (C) inputs. From negligible values at the beginning of the season, CH₄ emission gradually increases during the vegetative phase correlating with increasing plant biomass and peaks near panicle differentiation, a period of rapid root development. Emission remains relatively constant during the reproductive stage but may decrease during

late grain filling because of root degradation. Prior to the end of the season, a second emission peak may be observed. This late-season increase in emission can be attributed to an increase in soil C substrate due to accelerating leaf and root senescence.^[1] The addition of readily degradable C, such as rice straw, before planting results in an increase in the early-season CH₄ emission. The decomposition of the straw may also result in an enhanced or additional early-season emission maximum.^[1]

Photosynthetic Activity

In irrigated tropical rice paddies with double cropping, both CH₄ emission and rice grain yields are consistently higher in the dry season than in the wet season.^[2] One interpretation of this finding is that higher photosynthetic rates during the sunnier days of the dry season lead to larger amounts of C available to methanogenic bacteria and, consequently, greater production and emission rates of CH₄. Seasonal rates of CH₄ emission and rice grain yield have been positively correlated with accumulated solar radiation.^[3] A 1% increase in accumulative solar radiation is accompanied by a 1.1% increase in CH₄ emission and a 1% increase in rice grain yield. These findings are consistent with the hypothesis that solar radiation and hence photosynthetic activity of the rice plant correlate with CH₄ production and grain yield through the partitioning of non-structural carbohydrates to the root system and grain panicle.

Organic Amendments

Organic amendments of flooded rice paddies increase both CH₄ production and emission to the atmosphere. The amount of CH₄ emitted as a result of organic soil

amendments depends greatly on the amount and condition of readily available decomposable C contained in the treatment. CH₄ emission with incorporated rice straw has been found to be higher than that from either compost or mineral-fertilized plots.^[4] The effect of incorporating green manure is even more dramatic.^[5] Schütz et al.^[6] observed a 2.4-fold increase over control values with added rice straw. Cicerone et al.^[7] observed increases from a control value of 1.4 up to 58.18 g CH₄ m⁻² yr⁻¹ with added straw, a factor of over 40 times. Wassmann et al.,^[8] investigating the effect of fertilizers on CH₄ emission rates in Chinese rice fields (Hunan Province), found that the rate of increase in CH₄ emission depended on the total amount of organic manure applied. A single dose of organic manure increased the emission rates by the factors of 2.7 to 4.1, as compared with fields without organic manure. In field studies in the Philippines, Denier van der Gon and Neue^[9] found that fields treated with green manure applied at a rate of 22 t ha⁻¹ emitted over twice as much CH₄ as fields in which the application rate was 11 t ha⁻¹. Studies indicate that CH₄ emission enhancement due to added straw depends on soil type. In a study conducted in Bali,^[10] total amounts of CH₄ emitted during the rice growth period were 2.6–3.3 g m⁻² with applied chemical fertilizers compared with 3.9–6.8 g m⁻² with rice straw application in Alfisol plots, and 4.2–5.8 g m⁻² with applied chemical fertilizers compared with 6.9–10.7 g m⁻² with rice straw application in Inceptisol plots. These findings suggest that although CH₄ emissions from soils of volcanic origin are low, they show proportionate increases with added straw (C) comparable to those from other higher-emitting soils.

Inorganic Fertilization

Reports on the effects of mineral nitrogen fertilizer application (i.e., source, method of application, and amount) on CH₄ production and emission are difficult to interpret and inconsistent. Some form of fertilizer is necessary to ensure adequate plant development, which subsequently increases CH₄ emission through increased plant biomass and root activity. On the other hand, the lack of adequate fertilization could result in poor plant growth and yield as well as premature death, which would increase CH₄ emission through C made available by plant decay. The use of ammonium sulfate fertilizer may reduce CH₄ emission through the competition by sulfate-reducing bacteria for hydrogen or the presence of hydrogen sulfide toxicity.^[1]

Water Management

CH₄ emission rates are highly sensitive to soil aeration controlled through water management. Periodic drainage of irrigated rice paddies, which significantly decreases CH₄ emissions, may be the most effective method of mitigating them. In the Philippines, midseason drainage

of 2 weeks' duration at either mid-tillering or panicle initiation was very successful in suppressing CH₄ flux rates by up to 60%. However, the flux of nitrous oxide, a potent greenhouse gas, increased sharply during the drainage period.^[11] A single midseason drain reduced seasonal CH₄ emission rates by 50% in fields in the United States. In addition, multiple short periods of drainage (2–3 days) approximately every 3 weeks during the growing season reduced CH₄ emissions to an insignificant amount.^[12]

Soils

Rice soils prone to CH₄ production belong mainly to the orders of Entisols, Inceptisols, Alfisols, Vertisols, and Molisols. Oxisols, most of the Ultisols, and some of the Aridisols, Entisols, and Inceptisols are less favorable to CH₄ production when flooded. After the flooding of calcareous or alkaline soils (high pH), CH₄ production may occur almost immediately, whereas emission may take weeks to develop with acid soils and may not occur at all in highly acid soils. In general, sandy soils high in organic C produce more CH₄ than clay soils with similar C content.^[1] A strong positive correlation between seasonal CH₄ emission and % sand has been observed in Vertisols in Texas.^[13] However, CH₄ production may be limited in all soils if water percolation and the resultant oxygen levels are high.

Rice Cultivars

The effect of cultivar differences^[14] contributes to variation in CH₄ emissions from rice fields. A study of five rice cultivars in irrigated fields near Beijing, China, indicated that CH₄ emission during the tillering–flowering stage varied by a factor of two. CH₄ emission from eight different cultivars grown under comparable conditions near New Delhi, India, differed by as much as one order of magnitude. CH₄ emissions from six different cultivars grown in Louisiana fields showed that semidwarf varieties evolved significantly less CH₄ than tall varieties. Similarly, in an Indonesian paddy field, CH₄ emissions from eight popular modern varieties of rice varied, depending on the cultivar, from 32.6 to 41.7 g m⁻² and from 51.3 to 64.6 g m⁻².^[15] Fields planted with the Italian rice cultivar Lido showed CH₄ emissions (24–31%) lower than fields planted with the cultivar Roma.^[16] The difference in CH₄ emissions was attributed to a significantly lower gas transport capacity in the Lido cultivar. A study in Texas of 10 rice cultivars^[14] showed a cultivar-dependent variation in seasonal CH₄ emissions, ranging from 17.95 to 41.0 g m⁻². Cultivars exhibiting higher seasonal CH₄ emissions also exhibited higher soil acetate concentrations,^[17] suggesting that intervarietal differences in CH₄ emissions may be the result of different soil organic substrate levels and, hence, different rates of CH₄ production.

CONCLUSION

Significant progress has been made in understanding the processes of CH₄ flux, their temporal and spatial variations, and their relationship to country level and global atmospheric trace-gas inventories. This progress has come through international and interdisciplinary research collaborations and cooperation. The known factors governing the levels of CH₄ emissions from flooded rice fields, including climate and photosynthetic activity, added organic fertilizers, type of inorganic fertilizers, water management, soil type, and choice of rice cultivar, are largely understood. These factors have been incorporated into effective process-level models capable of calculating site-specific emission strengths throughout the world. Further integration of mechanistic models with geographic information systems of factors controlling emissions can provide improved estimates of the CH₄ source strength of various rice ecologies and farming practices and lead to environmentally and socioeconomically sound management and mitigation policies.

REFERENCES

1. Neue, H.U.; Sass, R.L. Rice cultivation and trace gas exchange. In *Global Atmospheric-Biospheric Chemistry*; Prinn, R.G., Ed.; Plenum Press: New York, 1994; 119–147.
2. Neue, H.U.; Lantin, R.S.; Wassmann, R.; Aduna, J.B.; Alberto, M.C.R.; Andales, M.J.F. Methane emission from rice soils of the Philippines. In *CH₄ and N₂O Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources*; Minami, K., Mosier, A., Sass, R.L., Eds.; NIAES Series 2; Yokendo Publishers: Tokyo, 1994; 55–63.
3. Sass, R.L.; Fisher, F.M.; Turner, F.T.; Jund, M.F. Methane emission from rice fields as influenced by solar radiation, temperature, and straw incorporation. *Global Biogeochem. Cy.* **1991**, *5*, 335–350.
4. Yagi, K.; Minami, K. Effect of organic matter applications on methane emission from Japanese paddy fields. *Soil Sci. Plant Nutr.* **1990**, *36*, 599–610.
5. Denier van der Gon, H.A.C.; Neue, H.U.; Lantin, R.S.; Wassmann, R.; Alberto, M.C.R.; Aduna, J.B.; Tan, M.J.P. Controlling factors of methane emission from rice fields. In *World Inventory of Soil Emission Potentials*; Batjes, N.H., Bridges, E.M., Eds.; WISE Report 2; ISRIC: Wageningen, Netherlands, 1993; 81–92.
6. Schütz, H.; Holzapfel-Pschorn, A.; Conrad, R.; Rennenberg, H.; Seiler, W. A 3-year continuous record on the influence of daytime, season and fertilizer treatment on methane emission rates from an Italian rice paddy. *J. Geophys. Res.* **1989**, *94*, 16405–16416.
7. Cicerone, R.J.; Delwiche, C.C.; Tyler, S.C.; Zimmermann, P.R. Methane emissions from California rice paddies with varied treatments. *Global Biogeochem. Cy.* **1992**, *6*, 233–248.
8. Wassmann, R.; Tölg, M.; Cheng, D.X.; Wang, M.X.; Papen, H.; Rennenberg, H.; Seiler, W. Spatial and seasonal distribution of organic amendments affecting methane emission from Chinese rice fields. *Biol. Fert. Soils* **1996**, *22*, 191–195.
9. Denier van der Gon, H.A.C.; Neue, H.U. Influence of organic matter incorporation on the methane emission from a wetland rice field. *Global Biogeochem. Cy.* **1995**, *9*, 11–23.
10. Subadiyasa, N.; Arya, N.; Kimura, M. Methane emissions from paddy fields in Bali island, Indonesia. *Soil Sci. Plant Nutr.* **1997**, *43*, 387–394.
11. Bronson, K.F.; Neue, H.U.; Singh, U.; Abao, E.B. Automated chamber measurements of methane and nitrous oxide flux in a flooded rice soil: I. Residue, Nitrogen, and water management. *Soil Sci. Soc. Am. J.* **1997**, *61*, 981–987.
12. Sass, R.L.; Fisher, F.M.; Wang, Y.B.; Turner, F.T.; Jund, M.F. Methane emission from rice fields: The effect of flood water management. *Global Biogeochem. Cy.* **1992**, *6*, 249–262.
13. Sass, R.L.; Fisher, F.M.; Lewis, S.T.; Turner, F.T.; Jund, M.F. Methane emission from rice fields: Effect of soil properties. *Global Biogeochem. Cy.* **1994**, *8*, 135–140.
14. Sass, R.L.; Fisher, F.M., Jr. Methane from irrigated rice cultivation. *Curr. Top. Wetland Biogeochem.* **1996**, *2*, 24–39.
15. Nugroho, S.G.; Sunyoto, Jumbanjaja, J.; Suprpto, H.; Ardjasa, W.S.; Kimura, M. Effect of rice variety on Methane emission from an Indonesian paddy field. *Soil Sci. Plant Nutr.* **1997**, *43*, 799–809.
16. Butterbachbahl, K.; Papen, H.; Rennenberg, H. Impact of gas transport through rice cultivars on methane emission from rice paddy fields. *Plant Cell Environ.* **1997**, *20*, 1175–1183.
17. Sigren, L.K.; Byrd, G.T.; Fisher, F.M.; Sass, R.L. Comparison of soil acetate concentrations and methane production, transport, and emission in two rice cultivars. *Global Biogeochem. Cy.* **1997**, *11*, 1–14.

Rice Paddies: Methane Emissions, Mitigating

Kazuyuki Yagi

National Institute for Agro-Environmental Sciences, Tsukuba, Japan

Abstract

The atmospheric concentration of methane (CH_4) is increasing rapidly. Because it is a radiative trace gas and takes part in atmospheric chemistry, the rapid increase could be of significant environmental consequence. Of the wide variety of sources, rice fields are considered as an important source of atmospheric CH_4 , because the harvest area of rice has increased by about 70% during the 20th century and it is likely that CH_4 emission has increased proportionally. Due to the large amount of the global emission from rice cultivation, reduction of CH_4 emission from this source is very important in order to stabilize atmospheric concentration. In addition, because of the possibility of controlling the emission by agronomic practices, rice cultivation must be one of the most hopeful sources for mitigating CH_4 emission.

PROCESSES CONTROLLING CH_4 EMISSIONS FROM RICE FIELDS

Table 1 provides a summary of measured CH_4 emissions at a number of specific research sites around the world.^[1,2] It should be noted that CH_4 fluxes from rice fields show pronounced diel and seasonal variations and vary substantially with different climate, soil properties, agronomic practices, and rice cultivars.

Processes involved in CH_4 emission from rice fields are illustrated in Fig. 1. Like other biogenic sources, CH_4 is produced by the activity of CH_4 producing bacteria, or methanogens, as one of the terminal products in the anaerobic food web in paddy soils. Methanogens are known as strict anaerobes that require highly reducing conditions. After soil is flooded, the redox potential of soil decreases rapidly by sequential biochemical reactions.

Flooded paddy soils have a high potential to produce CH_4 , but part of CH_4 produced is consumed by CH_4 oxidizing bacteria, or methanotrophs. In rice fields, it is possible that a proportion of CH_4 produced in the anaerobic soil layer is oxidized in the aerobic layers, such as the surface soil–water interface and the rhizosphere of rice plants.

The emission pathways of CH_4 that is accumulated in flooded paddy soils are as follows: diffusion into the flood water, loss through ebullition, and transport through the aerenchyma system of rice plants. In the temperate rice fields, more than 90% of CH_4 is emitted through plants,^[5] while significant amounts of CH_4 may evolve by ebullition, in particular during the early part of the season in the tropical rice fields.^[6] Therefore, it is concluded that possible strategies for mitigating CH_4 emission from rice cultivation can be made by controlling either production, oxidation, or transport processes.

OPTIONS FOR MITIGATING CH_4 EMISSION

Water Management

Mid-season drainage (aeration) in flooded rice fields supplies oxygen into soil, resulting in a reduction of CH_4 production and a possible enhancement of CH_4 oxidation in soil.^[7,8] A study using an automated sampling and analyzing system clearly showed that short-term drainage had a strong effect on CH_4 emission, as shown in Fig. 2. Total emission rates of CH_4 during the cultivation period were reduced by 42–45% by short-term drainage practices compared with continuously flooded treatment.^[9] These results indicate that improvement in water management can be one of the most promising mitigation strategies for CH_4 emission from rice fields. Increasing the rate of water percolation in rice fields by installing underground pipe drainage may also have an influence on CH_4 production and emission.

Soil Amendments and Mineral Fertilizers

The progress of soil reduction can be retarded by adding one of several electron acceptors in the sequential soil redox reactions. Sulfate is one of the most promising candidates for this strategy because it is commonly used as a component of mineral fertilizer and soil amendment. Field measurements have shown that CH_4 emission rate decreased at most 55–70% by application of ammonium sulfate or gypsum.^[10,11]

Additions of other oxidants, such as nitrate and iron-containing materials, may influence CH_4 emission from rice fields, as well as adding oxidants, dressing paddy fields with other soils that contain a large amount of free iron and manganese may decrease CH_4 emission. Other chemical candidates are nitrification inhibitors and acetylene releasing materials.

Table 1 CH₄ emission from rice fields in various world locations.^a

Country	Daily average (g/m ² day)	Flooding period (days)	Season total	
			Average (g/m ²)	Range (g/m ²)
China	0.19–1.39	75–150	13	10–22
India	0.04–0.46	60	10	5–15
Italy	0.10–0.68	130	36	17–54
Japan	0.01–0.39	110–130	11	3–19
Spain	0.10	120	12	
Thailand	0.04–0.77	80–110	16	4–40
U.S.A.	0.05–0.48	80–100	25	15–35

^aThe data are for the fields without organic fertilizer.

Organic Matter Management

In rice cultivation, fresh organic matter and animal wastes are often applied as fertilizers. In the fields, a proportion of the biomass of previous crops and weeds remains in soils at the start of rice cultivation. Such organic matter is decomposed in soils and acts as a substrate for fermentation reactions. Many researchers have demonstrated that incorporation of rice straw and green manure into rice paddy soils dramatically increases CH₄ emission.^[7,10,12] The impact of organic amendments on CH₄ emissions can be described by a dose–response curve which adopts

correction factors for composted and fermented organic matter.^[6] Mitigation of CH₄ emission requires the quantities of organic amendments to be minimized.

Field experiments also indicated that composted or fermented organic matter increased CH₄ emission much less than fresh organic matter due to a lower content of easily decomposable carbon.^[6,7] Therefore, stimulation of composting organic amendments appears to be a promising mitigation option. Plowing the fields during the fallow period and promoting aerobic degradation of organic matter are also likely to reduce CH₄ emission.

Others

Different tillage and cropping practices change the physical, chemical, and microbiological properties of the plow layer soil and may reduce CH₄ emission. These include deep tillage, no tillage, and flooded rice–upland crop rotation.

Selecting and breeding rice cultivars that emit lower CH₄ is a desirable approach because it is easy to adopt. There are four points to consider for selecting cultivars: 1) they should exude low levels carbon from their roots; 2) they should have a low level of CH₄ transport; and a high level of CH₄ oxidation in the rhizosphere; 3) they should have a higher harvest index in order to reduce organic matter input into soil after harvest; and 4) they should be suitable and have a high productivity when other mitigation options are performed.

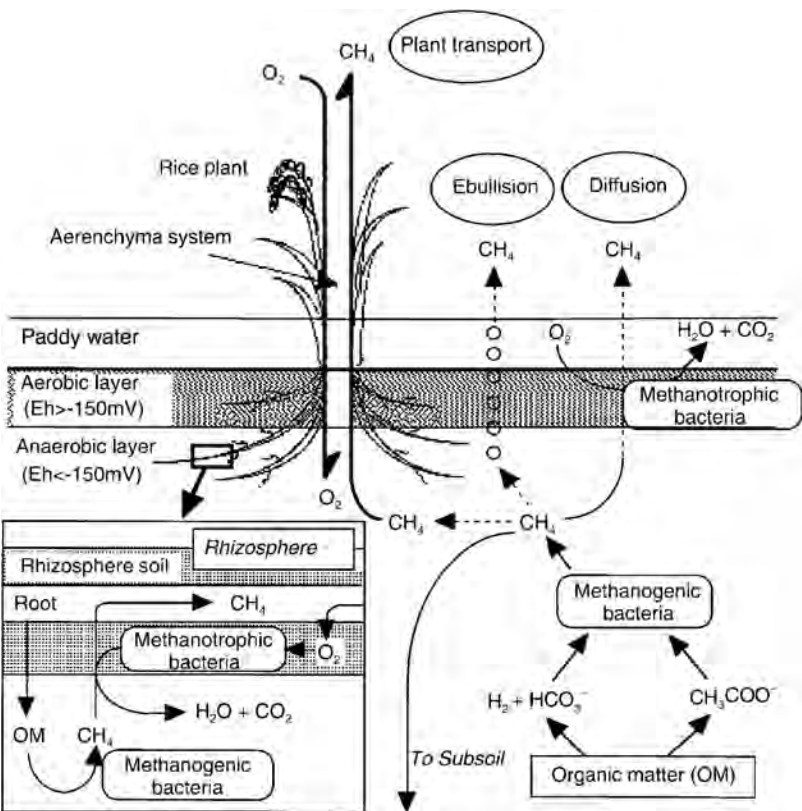


Fig. 1 Production, oxidation, and emission of CH₄ in rice paddy fields.

Source: Adapted from Conrad^[3] and Knowles.^[4]

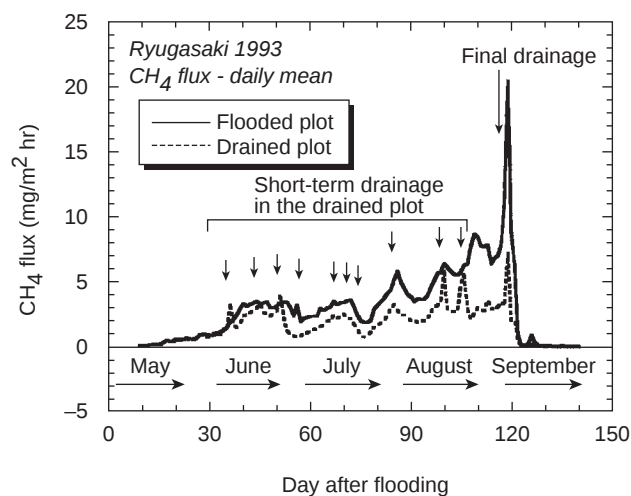


Fig. 2 Effect of water management on CH_4 emission from a rice paddy field. The arrows indicate the period of midseason drainage in the intermittent irrigation plot and the timing of final drainage in both of the plots.

PROBLEMS AND FEASIBILITY OF THE OPTIONS

If the above mitigation options could be applied to world's rice cultivation, global CH_4 emission from rice fields

could decrease significantly. However, there are several formidable obstacles to adopting the mitigation options into local rice farming. Table 2 summarizes the problems and feasibility of the individual mitigation options along with the efficiency of the options.

Application of some options is limited to specific types of rice fields. In particular, altering water management practices may be limited to rice paddy fields where the irrigation system is well equipped. Long midseason drainage and short flooding may cause possible negative effects on grain yield and soil fertility. Improving percolation by underground pipe drainage requires laborious engineering work. The increased water requirement is another problem in the water management options because water is a scarce commodity in many regions.

Cost and labor are serious obstacles for applying each option to local farmers. Most of the mitigation options will decrease profitability and the farmer net returns in the short run. To overcome these obstacles, an effort to maximize net returns by joining CH_4 mitigation and increased rice production will be needed as well as by political support.

It is recognized that the mitigation options should not have any significant trade-off effects, such as decreased rice yield, a decline in soil fertility, or increased environmental impact by nitrogen compounds. The development of anaerobic conditions in soil by flooding decreases the

Table 2 Evaluation of the mitigation options for CH_4 emission from fields.

Problem for application									
Applicability		Economy		Effects on		Fertility	Time span		Other trade-off effects
CH ₄ mitigation efficiency	Irrigated	Rainfed	Cost	Labor	Yield				
Water management									
Midseason drainage	□	○	•	~	↑	+	~	○	May promote N ₂ O emission
Short flooding	□	○	•	~	~	—	—	○	May promote N ₂ O emission
High percolation	□	○	•	↑	↑	+	~	○	May promote nitrate leaching
Soil amendments									
Sulfate fertilizer	□	○	○	↑	~	Δ	—	○	May cause H ₂ S injury
Oxidants	□	○	○	↑	↑	Δ	—	○	
Soil dressing	○	○	○	↑	↑	—	—	○	
Organic matter									
Composting	□	○	○	↑	↑	+	+	○	Causing atmospheric pollution
Aerobic decomposition	□	○	○	~	↑	~	~	○	
Burning	○	○	○	~	↑	~	~	○	
Others									
Deep tillage	○	○	○	↑	↑	—	—	○	
No tillage	?	○	○	~	↓	—	~	○	
Rotation	○	○	Δ	~	↑	—	—	○	
Cultivar	○	○	○	~	~	~	~	•	

Note: □, very effective; ○, effective/applicable; Δ, case by case; •, not applicable/require long time; ?, no information; ↑, increase; ↓, decrease; ~, about equal to previous situation; +, positive; -, negative.

Source: From Ranganathan, Neue, et al.^[13] ©1995 Springer-Verlag, Neue, Wassmann et al.^[14] ©1995 Springer-Verlag, and Yagi, Tsuruta, et al.^[15] ©1997.

decomposition rates of soil organic matter compared with aerobic soils, resulting in soil fertility being sustained for a long time. Flooded rice cultivation shows very little growth retardation by continuous cropping. Some mitigation options may reduce these advantages of rice fields. Application of sulfate-containing fertilizer may cause a reduction in rice yield due to the toxicity of hydrogen sulfide. Mid-season aeration and soil amendments may induce nitrogen transformation resulting in enhanced nitrous oxide emissions.^[16,17]

REFERENCES

1. Prather, M.; Derwent, R.; Ehhalt, D.; Fraser, P.; Sanhueza, E.; Zhou, X. Other trace gases and atmospheric chemistry. In *Climate Change 1994, Radiative Forcing of Climate Change and an Evaluation of the IPCC IS92 Emission Scenarios*; Houghton, J.T., Meira Filho, L.G., Bruce, J., Lee, H., Callander, B.A., Haites, E., Harris, N., Maskell, K., Eds.; Cambridge University Press: Cambridge, 1995; 73–126.
2. International Panel on Climate Change. *Greenhouse Gas Inventory Reference Manual; IPCC Guidelines for National Greenhouse Gas Inventories*; OECD: Paris, 1997; Vol. 3, 46–60.
3. Conrad, R. Control of methane production in terrestrial ecosystems. In *Exchange of Trace Gases Between Terrestrial Ecosystems and the Atmosphere*; Andreae, M.O., Schimel, D.S., Eds.; John Wiley & Sons Ltd.: New York, 1989; 39–58.
4. Knowles, R. Processes of production and consumption. In *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*; Harper, L.A., Mosier, A.R., Duxbury, J.M., Rolston, D.E., Eds.; American Society of Agronomy: Madison, 1993; 145–156.
5. Cicerone, R.J.; Shetter, J.D. Sources of atmospheric methane: Measurements in rice paddies and a discussion. *J. Geophys. Res.* **1981**, *86*, 7203–7209.
6. Denier van der Gon, H.A.C.; Neue, H.-U. Influence of organic matter incorporation on the methane emission from a wetland rice field. *Global Biogeochem. Cycles.* **1995**, *9*, 11–22.
7. Yagi, K.; Minami, K. Effect of organic matter application on methane emission from some Japanese paddy fields. *Soil Sci. Plant Nutr.* **1990**, *36*, 599–610.
8. Sass, R.L.; Fisher, F.M.; Wang, Y.B.; Turner, F.T.; Jund, M.F. Methane emission from rice fields: The effect of flood-water management. *Global Biogeochem. Cycles.* **1992**, *6*, 249–262.
9. Yagi, K.; Tsuruta, H.; Kanda, K.; Minami, K. Effect of water management on methane emission from a Japanese rice paddy field: Automated methane monitoring. *Global Biogeochem. Cycles.* **1996**, *10*, 255–267.
10. Schütz, H.; Holzapfel-Pschorn, A.; Conrad, R.; Rennenberg, H.; Seiler, W. A 3 yr continuous record on the influence of daytime, season, and fertilizer treatment on methane emission rates from an Italian rice paddy. *J. Geophys. Res.* **1989**, *94*, 16405–16416.
11. Denier van der Gon, H.A.C.; Neue, H.-U. Impact of gypsum application on methane emission from a Wetland rice field. *Global Biogeochem. Cycles.* **1994**, *8*, 127–134.
12. Sass, R.L.; Fisher, F.M.; Harcombe, P.A.; Turner, F.T. Mitigation of methane emission from rice fields: Possible adverse effects of incorporated rice straw. *Global Biogeochem. Cycles.* **1991**, *5*, 275–287.
13. Ranganathan, R.; Neue, H.-U.; Pingali, P.L. Global climate change: Role of rice in methane emission and prospects for mitigation. In *Climate Change and Rice*; Peng, S., Ingram, K.T., Neue, H.-U., Ziska, L.H., Eds.; Springer-Verlag: Berlin, Germany, 1995; 122–135.
14. Neue, H.-U.; Wassmann, R.; Lantin, R.S. Mitigation options for methane emissions from rice fields. In *Climate Change and Rice*; Peng, S., Ingram, K.T., Neue, H.-U., Ziska, L.H., Eds.; Springer-Verlag: Berlin, Germany, 1995; 137–144.
15. Yagi, K.; Tsuruta, H.; Minami, K. Possible options for mitigating methane emission from rice cultivation. *Nutr. Cycling Agro-Ecosys.* **1997**, *49*, 213–220.
16. Cai, Z.; Xing, G.; Yan, X.; Xu, H.; Tsuruta, H.; Yagi, K.; Minami, K. Methane and nitrous oxide emissions from rice paddy fields as affected by nitrogen fertilisers and water management. *Plant Soil* **1997**, *196*, 7–14.
17. Bronson, K.F.; Neue, H.-U.; Singh, U.; Abao, E.B., Jr. Automated chamber measurements of methane and nitrous oxide flux in a flooded rice soil: I. Residue, nitrogen, and water management. *Soil Sci. Soc. Am. J.* **1997**, *61*, 981–987.

Rice Paddies: Methanogenesis

Ronald L. Sass

Department of Ecology and Evolutionary Biology, Rice University, Houston, Texas, U.S.A.

Abstract

Irrigated rice cultivation is among the larger global sources of atmospheric methane (CH_4) and will become an even larger source due to increasing food demands. Because it is one of the few sources where the management of CH_4 emission is possible, it is also likely to be a critical focus of mitigation efforts. Several mitigation options include water management, soil amendments, organic matter management, cultivar selection, and field preparation, with water management being the most promising. However, because rice is also the world's most important wetland crop and the primary calorie source of a large fraction of the world's population, mitigation efforts must be based on sound agricultural practices as well as good scientific judgment.

INTRODUCTION

Methane (CH_4) is a radiatively active trace gas with approximately 21 times more infrared sorbing capability per molecule than carbon dioxide as a source of potential global warming.^[1] Worldwide, agricultural sources of CH_4 may account for as much as 40% of the total annual emission of atmospheric CH_4 with a significant portion contributed by rice cultivation. To meet the rice supply of growing populations, rice cultivation will continue to increase at or beyond its existing rate. Based on the projected population growth rates in countries where rice is the main food crop, it is estimated that the world's annual rough rice production must increase from a 1990 value of 473 million tons to 564 million tons by 2000 and 781 million tons by 2020—a 65% increase (1.7% per year). In South Asia, e.g., rice production is projected to double by the year 2020.^[2] Because arable land is limited in major rice-growing areas, increased production has to be achieved mainly by intensifying cropping (i.e., two or three crops per year) and developing new technologies rather than expanding the area of rice cultivation. Irrigated rice will continue to dominate production. Irrigated rice land comprises about half the total harvested area but contributes more than two-thirds of the total grain production. With available agronomic practices, this trend will lead to significantly increased CH_4 emissions.

CH_4 PRODUCTION AND EMISSION FROM RICE FIELDS

The processes involved in CH_4 emission from flooded rice to the atmosphere include CH_4 production in the soil by methanogens, CH_4 oxidation within oxic zones of floodwater and the soil by methanotrophs, and vertical transport from soil to the atmosphere. CH_4 is produced in the terminal step of several anaerobic degradation chains. The

amount of CH_4 produced in flooded rice soils is primarily determined by the availability of methanogenic substrates and the influence of environmental factors. The sources of organic carbon for methanogenic substrates are derived from rice plants via root processes and biomass litter or added organic matter. The environmental factors affecting CH_4 production include soil texture, climate, and agricultural practices, such as water regime and management. Plant-mediated transport is the primary mechanism for the emission of CH_4 from rice paddies, with approximately 90% of CH_4 transported to the atmosphere through the aerenchymal system of the rice plants. The rice aerenchymal system not only transports CH_4 from the rhizosphere to the atmosphere but also promotes the movement of atmospheric oxygen into the rhizosphere supporting CH_4 oxidation. More than 50% of the generated CH_4 is oxidized during the rice-growing season.^[3]

DISTRIBUTION OF CH_4 EMISSIONS FROM RICE AGRICULTURE

Rice is grown under a variety of climatic, soil, and hydrological conditions in over 90 countries of the world and on all continents except Antarctica. The ecosystems within which rice is grown vary widely with respect to elevation, rainfall pattern, depth of flooding and drainage, soil type, and the variety of rice planted. Fields of rice can be found from 50° N in the regions of China and Japan to 40° S in the southern regions of Australia. Rice is grown from sea level to altitudes of more than 2500 m. It is a unique crop since it adapts to a broad range of soil water content. It grows well in flood-prone areas of South and Southeast Asia (as much as 5 m of floodwater) and in drought-prone upland areas of Asia, South America, and Africa.^[4]

A rich variation in local cultural patterns as well as in natural conditions can be found in the various regions of the

world where rice agriculture is practiced. These variations are reflected in different regional and country values of CH_4 emission rates.^[5] Rice agriculture can be conveniently divided into the following four categories based on water availability: upland or dry rice, rainfed lowland rice, flood-prone rice, and irrigated rice. The relative CH_4 flux strength for different rice ecosystems follows the order: irrigated rice $\geq$ favorable rainfed rice $>$ flood-prone rainfed rice $\geq$ deepwater rice $>$ drought-prone rainfed rice $>$ tidal wetland rice. Upland rice is not a source of CH_4 because it is grown in well-aerated soils. Differences in crop residue treatment, types and amounts of organic amendments, scheduled or opportunistic aeration periods, soil classification and texture, fertilization practices, and choice of rice cultivars are major causes for variations in CH_4 fluxes in irrigated rice systems. Table 1 presents the scaling factors for CH_4 emissions with different ecosystem water regimes and organic amendments relative to continuously flooded (irrigated) fields without organic amendments. An estimate for each

of these different systems is obtained by multiplying the expected CH_4 emission from a continuously flooded field without organic amendments by the two relevant scaling factors. The highest CH_4 fluxes are observed in continuously flooded fields receiving organic amendments such as green or animal manure. The lowest CH_4 fluxes are recorded in fields with low residual carbon content, multiple aeration periods, poor soils, low fertilization, poor rice growth, and low grain yields.

The CH_4 source strength of rainfed rice is highly uncertain because of the high variability in all factors controlling CH_4 emission. CH_4 fluxes are generally lower from rainfed rice because of drought periods and low rice yields. The availability and use of recycled soil carbon residues and available organic amendments are also generally less in areas where rainfed rice is grown compared to the areas of irrigated rice. CH_4 fluxes may approach those from irrigated rice in highly flood-prone rainfed areas. Some areas of deepwater rice may exhibit an even higher CH_4 source strength than the irrigated rice because of long flooded growing periods and high rice biomass production, even though grain yields are low. The source strength of tidal wetland rice is small because generally it is affected by salt water high in sulfate.^[5,8]

Table 1 CH_4 emission scaling factors for rice ecosystems relative to continuously flooded fields without organic amendments.

	Scaling factor	Range
Rice ecosystem		
Upland	0	
Irrigated		
Continuously flooded	1.0	
Intermittently flooded, single aeration	0.5	0.2–0.7
Intermittently flooded, multiple aeration	0.2	0.1–0.3
Rain fed		
Flood prone	0.8	0.5–1.0
Drought prone	0.4	0–0.5
Deep water		
Water depth 50–100 cm	0.8	0.6–1.0
Water depth >100 cm	0.6	0.5–0.8
Organic amendments		
Straw, animal manure, green manure, agricultural wastes, etc.:		
0 t ha ⁻¹ (inorganic fertilizer only)	1.0	
1–2 t ha ⁻¹	1.5	1.0–2.0
2–4 t ha ⁻¹	1.8	1.5–2.5
4–8 t ha ⁻¹	2.5	11.5–3.5
8–15 t ha ⁻¹	3.5	2.0–4.5
15+ t ha ⁻¹	4.0	3.0–5.0
Fermented amendments, compost, biogas pit residue, etc.:		
1–6 t ha ⁻¹	1.0	1.0–1.5
6–12 t ha ⁻¹	1.5	1.0–2.0

Source: From Houghton^[6] and Denier van der Gon & Neue.^[7]

RANGE OF GLOBALLY OBSERVED SEASONAL EMISSIONS

Observed CH_4 emissions for a single rice-cropping season reported from several countries around the world show large ranges, reflecting local as well as regional differences in agricultural, biological, and climatic factors. An average of all such reported median country CH_4 emission values from irrigated rice is 27.23 g CH_4 m⁻², with a range from a minimum value of less than 1 g CH_4 m⁻² to a maximum value of 155 g CH_4 m⁻².^[5] A data set from five countries (China, India, Indonesia, Thailand, and the Philippines) obtained under a network coordinated by the International Rice Research Institute showed averages of less than 30 g CH_4 m⁻² under mineral fertilization.^[9] Emissions under organic amendments were generally higher than 20 g CH_4 m⁻² except for a site in northern India where average emissions remained as low as 2 g CH_4 m⁻².^[9] In a series of studies in Texas, conducted between 1991 and 1995, 32 seasonal CH_4 emission rates were measured under the conditions of continuous flooding and no organic amendments, in three different soil types with 11 different rice cultivars. These studies show a measured mean CH_4 emission of 21.99 $\pm$ 8.53 g CH_4 m⁻² and a median value of 19.84 g CH_4 m⁻².^[10] The situation is similar for rainfed and deepwater rice. For rainfed rice, the median of average observed value is 13.33 g CH_4 m⁻², and reported ranges are from 1 to 68 g CH_4 m⁻².^[5] Emission values^[11] for rainfed rice range from 8 g CH_4 m⁻² (drought-prone rice) to 16 g CH_4 m⁻² (flood-prone rice). The average CH_4 emission value for

deepwater rice ranges from 12 g CH₄ m⁻² (50–100 cm water depth) to 16 g CH₄ m⁻² (>100 cm water depth).^[5]

Estimates of the global release of CH₄ from rice paddies made range from 12 to 200 Tg yr⁻¹.^[10] Many early estimates are based on a limited number of field measurements and range widely. The Intergovernmental Panel on Climate Change (IPCC) suggested a most probable value of CH₄ emitted from global rice paddies to be 60 ± 40 Tg yr⁻¹.^[12] Three estimates based on extensive field data and model calculations report ranges of 25–54,^[13] 21–41,^[14] and 33–49 Tg yr⁻¹.^[5] Considering these results, a best estimate ranging between 26 and 50 Tg yr⁻¹ for the annual world CH₄ emission from rice fields is posited.^[10] Asia accounts for approximately 90% of the world rice production^[15] and over 90% of the annual CH₄ emissions from rice fields. The following seven Asian countries lead the world in rice production and CH₄ emission: China, India, Indonesia, Thailand, the Philippines, Japan, and the Republic of Korea.^[10]

MITIGATION OF CH₄ EMISSIONS FROM RICE FIELDS

The IPCC^[16] calculated that stabilizing the concentration of atmospheric CH₄ at the 1990 level would require a 15–20% reduction in total CH₄ emission. Irrigated rice cultivation is among the larger global sources of atmospheric CH₄ and will become an even larger source due to increasing food demands. Because it is one of the few sources where the management of CH₄ emission is possible, it is also likely to be a critical focus of mitigation efforts. Several mitigation options to reduce atmospheric CH₄ concentrations have been developed through research efforts.^[2,8,17,18] These include water management, soil amendments, organic matter management, cultivar selection, and field preparation, with water management being the most promising. However, because rice is also the world's most important wetland crop and the primary calorie source of a large fraction of the world's population, mitigation efforts must be based on sound agricultural practices as well as good scientific judgment.

REFERENCES

1. Rodhe, H. A comparison of the contribution of various gases to the greenhouse effect. *Science* **1990**, *248*, 1217–1219.
2. Sass, R.L.; Fisher, F.M.; Wang, Y.B.; Turner, F.T.; Jund, M.F. Methane emission from rice paddies: The effect of floodwater management. *Global Biogeochem. Cy.* **1992**, *6*, 249–262.

3. Huang, Y.; Sass, R.L.; Fisher, F.M., Jr. A semi-empirical model of methane emission from flooded rice paddy soils. *Glob. Change Biol.* **1998**, *4*, 247–268.
4. Neue, H.U.; Sass, R.L. Trace gas emissions from rice fields. In *Global Atmospheric-Biospheric Chemistry*, Environmental Science Res. 48; Prinn, R.G., Ed.; Plenum Press: New York, 1994; 119–148.
5. Neue, H.-U.; Sass, R.L. The budget of methane from rice fields. *IGActivities NewsLetter* **1998**, *12*, 3–11.
6. Houghton, J.T. *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories*; IPCC/OECD/IEA; Intergovernmental Panel on Climate Change: Paris, 1997; 3, 4–12.
7. Denier van der Gon, H.A.C.; Neue, H.U. Influence of organic matter incorporation on the methane emission from a wetland rice field. *Global Biogeochem. Cy.* **1995**, *9*, 11–22.
8. Neue, H.U. Fluxes of methane from rice fields and potential for mitigation. *Soil Use Manage.* **1997**, *13*, 258–267.
9. Wassmann, R.; Neue, H.-U.; Lantin, R.S.; Buendia, L.V.; Rennenberg, H. Characterization of methane emissions from rice fields in Asia. 1. Comparison among field sites in five countries. *Nutr. Cycl. Agroecosys.* **2000**, *58*, 1–12.
10. Sass, R.L.; Fisher, F.M.; Ding, A.; Huang, Y. Exchange of methane from rice fields: A model system for wetland emission modeling. *J. Geophys. Res.* **1999**, *104*, 26943–26951.
11. Wassmann, R.; Neue, H.-U.; Lantin, R.S.; Makarim, K.; Chareonsilp, N.; Buendia, L.V.; Rennenberg, H. Characterization of methane emissions from rice fields in Asia. 2. Differences among irrigated, rainfed and deepwater rice. *Nutr. Cycl. Agroecosys.* **2000**, *58*, 13–22.
12. Climate Change 1992. Supporting Material, Working Group III, Response Strategies, Subgroup AFOS. The IPCC Supplementary Report to the IPCC Scientific Assessment; WMO/UNEP: Geneva, 1992; 14–22.
13. Sass, R.L.; Fisher, F.M.; Lewis, S.T.; Jund, M.F.; Turner, F.T. Methane emission from rice fields: Effect of soil properties. *Global Biogeochem. Cy.* **1994**, *8*, 135–140.
14. Yagi, K.; Minami, K. Methane emission from paddy fields as affected by soil properties and its implications to the global estimation. In *Paper Presented at IRRI-UNDP Final Workshop*, Beijing, 1998; 34–38.
15. International Rice Research Institute. *World Rice Statistics 1993–94*; International Rice Research Institute: Manila, 1995; 2–5.
16. Houghton, J.T.; Callender, B.A.; Varney, S.K. *Climate Change*; The Supplementary Report to the IPCC Scientific Assessment; IPCC-Intergovernmental Panel on Climate Change; Cambridge University Press: Cambridge, 1992.
17. Sass, R.L.; Fisher, F.M. Methane emissions from Texas rice fields: A five-year study. In *Climate Change and Rice*; Peng, S., Ingram, K.T., Neue, H.U., Ziska, L.H., Eds.; Springer: New York, 1995; 46–59.
18. Yagi, K.; Tsuruta, H.; Minami, K. Possible options for mitigating methane emission from rice cultivation. *Nutr. Cycl. Agroecosys.* **1997**, *49*, 213–220.

Rice Paddies: Soil Management

Xin Zhao

Jianfu Xue

Hailin Zhang

College of Agronomy and Biotechnology, China Agricultural University,
Beijing, China

Abstract

As a global staple food, rice (*Oryza sativa* L.) plays an important role in food security. Due to the long-term flooded condition, paddy soils have a series of special physical, chemical, and biological characteristics. Therefore, managing paddy soils is critical to rice production and climate change mitigation. Management strategies of paddy soils should be based on the goals of increasing rice production, enhancing nutrient use efficiency, improving soil quality, and enhancing the economic and environmental benefits.

INTRODUCTION

Rice (*Oryza sativa* L.) is an important food staple of the world. About 166.1 million hectares (Mha) of paddy rice was harvested in the world in 2013.^[1] During the period of 1981–2013, the harvested area of paddy rice increased by 21.1 Mha with average yield increased from 2730.7 to 4547.8 kg ha⁻¹ globally.^[1] Asia is the most important producer and consumer, providing 91.0% of the world's paddy rice grown in 89.4% of world's paddy fields, and the yield increased from 2859.3 to 4486.7 kg ha⁻¹ during 1981–2013.^[1] China is one of the top contributors of paddy rice to the world, and the harvested areas and yield reached 30.2 Mha and 6725.7 kg ha⁻¹ in 2013, respectively.^[1]

Rice is mostly cultivated and managed as paddy soil, which is typically waterlogged and formed through seasonal irrigation. A marked alteration of hydromorphic soil properties induces specific processes of paddy soil formation through redox potential oscillations under long-term paddy rice cultivation.^[2] Such specific properties result in hydric Anthrosols because of the characteristic management of paddy soil.^[3] Therefore, suitable paddy soil management strategies play an important role in rice production and global food security. Moreover, paddy soil is a major source of global methane (CH₄) emissions on 11% of the total agricultural emissions. It is also an important source of nitrous oxide (N₂O).^[4] The formation of paddy soil is related to rice cultivation and is affected by many factors, e.g., terrain hydrological conditions, parent material, climate, and cropping system strategies. Thus, management strategies of paddy fields are vital to maintain rice production and reduce emissions of greenhouse gases (GHGs).

CHARACTERISTICS OF PADDY SOILS

Paddy fields are sited primarily on alluvial soils such as in deltas and flood plains of large rivers, coastal plains, fans, and lower terraces. Thus, paddy soils have many special characteristics that depend on frequently anthropogenic management practices under rice-based cropping systems. Alternate dry–wet cycles and soil tillage play an essential role in profile formation and a series of physical, chemical, and biological properties of paddy soils. Therefore, paddy soils share some special characteristics despite wide differences in parent materials. In addition, soil properties change strongly with increase in soil age.^[2]

Compared to upland soils, the water-holding capacity, soil organic matter (SOM), and microbial biomass carbon are higher under paddy soils. For example, Chen et al.^[5] reported that water-holding capacity, soil organic carbon (SOC), and microbial biomass carbon under paddy soil were 19.0%, 33.7%, and 12.1% higher than that under upland soil at 0–20 cm depth, respectively.^[5] Moreover, the seasonal variations of soil temperatures in paddy soils are more stable with the enhancement of the soil heat capacity through increased water and decreased air in the cultivated layer during the flooding period. Stabilization of soil temperatures benefits both rice growth and yield. Furthermore, changes in soil temperatures within a paddy field are related to the depth of the soil layers, soil saturated moisture regime, infiltration rate, and temperature of the irrigation water.^[6]

Paddy soils are generally comprised of oxidized and reduced zoned in the plow layer.^[7] Under flooding conditions, the oxidized layer is formed at the water–soil interface due to diffusion of aquatic oxygen into the surface soil.

The periodic change of redox reactions is the most obvious biochemical change in paddy soils, the intensity of which is indicated by the redox potential (E_h). The E_h of paddy soils ranges from -200 to -300 mv to $+500$ to $+700$ mv, which also reflects the proportional relationship between dissolved oxide and reduzate (a high E_h indicates more oxides).^[7]

Generally, waterlogging associated with rice cropping enhances accumulation of SOC. The accumulation of SOC in paddy soils may be due to the high input of rice residues combined with slow decomposition under anaerobic conditions. However, mechanisms of SOC accumulation in paddy soils are not well understood. Proposed mechanisms include the relation between SOC accretion and CH_4 emissions and the interactions between SOC and E_h or other environmental factors that will help to understand how the aeration condition and E_h affect the accumulation and stabilization of SOC.^[8–10]

The transformation of substances in paddy soils is a special biochemical process that occurs under long-term flooding conditions. The form of inorganic nitrogen exists as ammonium more than nitrate (NO_3^-) in paddy soils due to the reduction of NO_3^- by denitrification and leaching in the subsurface layer.^[7] Moreover, some noxious substances are formed due to incomplete decomposition of SOM under conduction of low oxidation reduction potential.^[2,7,8]

MANAGEMENT APPROACHES

Recommended management practices of paddy soils are based on the goals of increasing crop yield, enhancing nutrient use efficiency, ameliorating soil quality, and increasing economic and environmental benefits. On the contrary, the characteristics of paddy soils are also vital factors that affect the management strategies. Prior studies and farming experiences indicate that farming practices (e.g., rational tillage, water management, residue management, and fertilization) are conducive to maintaining paddy rice productivity, enhancing food security, mitigating global climate change, and sustaining the paddy system.

Tillage Practices

Tillage practices have been one of the most important factors that change the soil environment directly. For most paddy fields, plow tillage as a conventional practice has a long history on a global scale. In conventional tillage, a plow pan can develop at the 20–30 cm depth following an extended period of plowing, which can slow the transmission of water and nutrients across this layer. The plow pan can also reduce the leaching of nutrients in the rice paddy, even with long-term flooding.^[11] Moreover, the disruption of substance transmission through the plow pan can

decrease redox potential, reduce toxic materials, regulate organic matter accumulation by affecting the distribution of elements (i.e., iron), and maintain the nutrient supply.^[8,9] On the contrary, tillage practices also affect the formation and function of paddy soils, especially changing properties (i.e., soil bulk density, enzymatic activity, and microbial activity) in the plow layers.^[12] No-till (NT) farming (or direct seeding) in paddy soils has been gradually adopted in China, Pakistan, Bangladesh, India, and elsewhere, due to rising labor prices and short term around period for sowing the next crop (e.g., wheat). However, with the long duration of NT, the plow pan may gradually disappear and increase the water infiltration rate, thus increasing the risk of nutrient leaching. Thus, rational tillage systems in paddy soils need to be further studied for improving the nutrient use efficiency in rice-based systems.

Irrigation

In general, paddy cultivation consumes large quantities of water to meet the evaporative demand during the entire rice-growing season. Thus, the adoption of water-saving irrigation practices has a critical role in sustainable rice production due to the decreasing water availability and competing user of water. Meanwhile, water management is recognized as one of the most important practices associated with CH_4 and N_2O emissions from paddy fields. Traditional continuous flooding in paddy fields increases CH_4 emissions, even though it reduces N_2O emissions. Further, mid-season aeration and alternate dry–wet cycles have been widely adopted to save agricultural water and increase rice yield. However, there is a trade-off between CH_4 and N_2O emissions from paddy fields with mid-season aeration and dry–wet cycles.^[13] The adoption of controlled irrigation can effectively mitigate the integrative greenhouse effect produced by CH_4 and N_2O emissions from paddy fields while also ensuring high rice yield.^[13] Additional research is needed to quantify the integrative greenhouse effect under other major water-saving irrigation practices (e.g., intermittent irrigation, flooding–mid-season drainage–frequent waterlogging with intermittent irrigation, and flooding–mid-season drainage–reflooding–moist intermittent irrigation).

Residue Management

Large amounts of rice residues are produced in the rice-growing countries. Traditionally, large proportion of rice residues has been burned after rice harvest, causing loss of nutrients and SOM, as well as air pollution. The adoption of appropriate residue management practices can have a strong effect on soil physical, chemical, and biological properties, which can improve soil fertility, enhance SOC sequestration, and reduce GHG emissions.^[14]

Innovative technologies have greatly enhanced paddy rice production. Conservation agriculture, including

rational tillage systems, residue mulching, and crop rotation, has gained popularity on a global scale and has contributed to sustainable productivity of paddy soils through improved crop yield, enhanced SOC sequestration, and mitigation of GHG emissions.^[11,13] Other management strategies such as fertilization (new fertilizer type and methods), pesticides, and irrigation are conducive to improve the use efficiency of materials and inputs to enhance paddy rice production and reduce the GHG emissions.^[11,15] Integrated management systems are an efficient strategy to achieve the goals of sustaining food supply, mitigate climate change, and improve agroecosystem sustainability in paddy soils. High yield amelioration of paddy soils requires the incorporation of rational tillage, fertilization (with farmyard manure), and water management (aeration and reflooding).

CONCLUSION

Paddy rice production is critical to global food security. Thus, managing paddy soils is vital to both food security and mitigating global climate change. Therefore, a deep understanding of the formation process and characteristics of paddy soils is helpful to select the suitable managements for rice production. Adopting a holistic approach in the management of soil and water resources (including rational tillage systems and integrated nutrient and pest management practices) in paddy soils is important for decreasing water and soil losses, improving efficiency, enhancing rice production, reducing GHG emissions, and sequestering SOC.

REFERENCES

1. FAOSTAT. *FAOSTAT Database, Food and Agriculture Organization of the United Nations*, 1981–2013. Available at <http://faostat3.fao.org/faostat-gateway/go/to/home/E> (accessed September 2014).
2. Kölbl, A.; Schad, P.; Jahn, R.; Amelung, W.; Bannert, A.; Cao, Z.H.; Fiedler, S.; Kalbitz, K.; Lehndorff, E.; Müller-Niggemann, C.; Schloter, M.; Schwark, L.; Vogelsang, V.; Wissing, L.; Kögel-Knabner, I. Accelerated soil formation due to paddy management on marshlands (Zhejiang Province, China). *Geoderma* **2014**, *228–229*, 67–89.
3. IUSS Working Group WRB. *World Reference Base for Soil Resources 2006, 1st Update 2007: World Soil Resources Reports, 103*; FAO: Rome, 2007.
4. FAOSTAT. *FAOSTAT Database, Food and Agriculture Organization of the United Nations* 2012. Available at <http://faostat3.fao.org/faostat-gateway/go/to/home/E> (accessed September 2014).
5. Chen, X.; Wang, A.; Li, Y.; Hu, L.; Zheng, H.; He, X.; Ge, T.; Wu, J.; Kuzyakov, Y.; Su, Y. Fate of ¹⁴C-labeled dissolved organic matter in paddy and upland soils in responding to moisture. *Sci. Total Environ.* **2014**, *488–489*, 268–274.
6. Playan, E.; Perez-Coveta, O.; Martinez-Cob, A.; Herrero, J.; Garcia-Navarro, P.; Latorre, B.; Brufau, P.; Garces, J. Overland water and salt flows in a set of rice paddies. *Agr. Water Manage.* **2008**, *95*, 645–658.
7. Li, Q.Q. *Paddy Soils of China*; Press of Science: Beijing, 1992.
8. Kögel-Knabner, I.; Amelung, W.; Cao, Z.; Fiedler, S.; Frenzel, P.; Jahn, R.; Kalbitz, K.; Kölbl, A.; Schloter, M. Biogeochemistry of paddy soils. *Geoderma* **2010**, *157*, 1–14.
9. Wissing, L.; Kolbl, A.; Hausler, W.; Schad, P.; Cao, Z.H.; Kögel-Knabner, I. Management-induced organic carbon accumulation in paddy soils: The role of organo-mineral associations. *Soil Till. Res.* **2013**, *126*, 60–71.
10. Zhang, Y.; Su, S.; Zhang, F.; Shi, R.; Gao, W. Characterizing spatiotemporal dynamics of methane emissions from rice paddies in northeast China from 1990 to 2010. *PLoS One* **2012**, *7*, e29156.
11. Cui, S.Y.; Xue, J.F.; Chen, F.; Tang, W.G.; Zhang, H.L.; Lal, R. Tillage effects on nitrogen leaching and nitrous oxide emission from double-cropped paddy fields. *Agron. J.* **2014**, *106*, 15–23.
12. Pandey, D.; Agrawal, M.; Bohra, J.S. Effects of conventional tillage and no tillage permutations on extracellular soil enzyme activities and microbial biomass under rice cultivation. *Soil Till. Res.* **2014**, *136*, 51–60.
13. Cai, Z.C.; Xing, G.X.; Yan, X.Y.; Xu, H.; Tsuruta, H.; Yagi, K.; Minami, K. Methane and nitrous oxide emissions from rice paddy fields as affected by nitrogen fertilisers and water management. *Plant Soil* **1997**, *196*, 7–14.
14. Liu, C.; Lu, M.; Cui, J.; Li, B.; Fang, C. Effects of straw carbon input on carbon dynamics in agricultural soils: A meta-analysis. *Global Change Biol* **2014**, *20*, 1366–1381.
15. Zhang, H.L.; Lal, R.; Zhao, X.; Xue, J.F.; Chen, F. Opportunities and challenges of soil carbon sequestration by conservation agriculture in China. *Adv. Agron.* **2014**, *124*, 1–36.

Rocks

Vsevolod V. Dobrovolsky

Moscow State Pedagogical University, Moscow, Russia

Abstract

Soil develops, as a rule, on a mineral substrate that plays the role of soil skeleton and accumulates plant residues and products of their biochemical transformation. The significance of soil-forming mineral substrate (parent rock) is very high, since it constitutes about 95% of the total mass of most soils. Soil inherits all peculiarities of mineral substrate connected with the particle size distribution, mineralogical and bulk chemical composition, and many physico-chemical properties important for the agronomic technology. Fully developed soil with the regular sequence of certain horizons determined by climatic factors can be formed only on those mineral substrates that are loose and porous enough to provide the free migration of water, air, mezofauna, and easy penetration by plant roots. These are the properties of late-tertiary and quaternary loose substrates covering nearly all the terrestrial surface. The loose consistency is also characteristic of the volcanic eruptions, on which specific soils—Andosols—are formed.

INTRODUCTION

Consolidated rocks play a less significant role in pedogenesis because their high density is unfavorable for an active interaction between biota, water, gases, and mineral matter. Soils formed on these rocks consist, as a rule, of a single horizon of plant residues decomposed into different degrees with an admixture of fine eolian material. These soils can also include a shallow transitional horizon consisting of rock debris, loose allochthonous material, and illuvial humus. The composition of hard rocks can strongly affect the possibility of full-profile soils' development and the trend of pedogenesis. On dense quartzites in case of moderate climate, the regolith cannot be formed even for $n = 10^4$ yr. However, on clayey-calcareous rocks (marls) the soils with the complete sequence of soil horizons can be formed in a much shorter time. The high content of calcite in hard rocks and consequent neutralization of acid products of organic matter decomposition results in the occurrence of soils with a mollic horizon, even in the case of humid climate and forest vegetation. At the same time, extremely high content of some chemical elements in parent material [e.g., magnesium (Mg) in serpentinites] may lead to low bioproductivity of ecosystems and, consequently, to weak development of pedogenic features.^[1]

With respect to loose substrates, although mantle deposits differ by the mechanisms of transport and accumulation (glacial, fluvial, eolian, etc.), all of them consist of weakly consolidated products of rock weathering. The earth's crust is composed mostly of compact crystalline rocks consisting of minerals of the silicate class. Being exposed on the surface, these rocks undergo mechanical disintegration, as the crystalline and chemical structures

of minerals formed at a great depth become unstable. For this reason, a very slow but continuous processes of total decomposition (hydrolysis) and transformation into new mineral structures occur.^[2] Therein lies the essence of rock weathering (hypergenesis).

COARSE AND FINE MINERAL MATERIAL OF MANTLE DEPOSITS INHERITED BY SOILS

The mineral components of mantle deposits can be subdivided into three groups.^[3] The first one includes the coarse particles of prevailing bedrock minerals, i.e., quartz and silicates. The stability of these minerals is different and decreases along the following range: quartz (the most stable), feldspar, mica, amphibole, pyroxene, plagioclase, olivine.^[4] For this reason, redeposited products of weathering are characterized by the prevalence of quartz, lower contents of feldspars and micas, rare amphiboles, single plagioclases and pyroxenes, and very infrequent olivine.

Some of the minerals contained in widespread rocks in very small quantities are very stable. These are ilmenite, magnetite, garnet, epidote, zircon, rutile, disthene, staurolite, etc. They are insignificantly changed during the redeposition of weathering products and can be easily separated because of a considerable specific weight (2.9) as a single fraction of coarse particles (the so-called "heavy fraction"). Despite the mineralogical heterogeneity, this fraction cannot cause a significant variation in the bulk chemical composition of mantle deposits, since it usually constitutes less than 1% of their total mass. However, the composition of heavy fraction points to the original location of weathering products. Since microelements

specific for certain territories are concentrated in this fraction, its study allows us to define the mineralogical–geochemical provinces of mantle deposits, the peculiarities of which are inherited by soils.

The second group of mineral components of mantle loams comprises minerals formed as a result of hypergene transformation of structures of primary silicates. Secondary (hypergene) silicates have a specific (layered) structure and are represented by very fine particles (<0.002 mm) that can be observed only with the use of electron microscope and with magnifications of several thousands.^[5]

The structure of hypergene silicates and their large specific surface account for the ability of these minerals to adsorb cations (which can be equivalently substituted by other cations) from solution. Hypergene silicates are represented by the group of clay minerals: kaolinite, halloysite, montmorillonite, hydrochlorides, hydromicas, etc. They differ by the cation exchange capacity, which is the lowest for kaolinite and the highest (about 10 times higher) for montmorillonite.

Soils formed on mantle deposits inherit the content and composition of clay minerals. The cation exchange capacity of many soils is related to the mineralogical composition of mantle deposits being prevalent parent rocks.

The composition of clay minerals may predetermine not only soil characteristics but also microrelief. For example, in the case of tropical and subtropical climates, with alternations of dry and humid seasons on heavy substrates with montmorillonite predominance, Vertisols with well-pronounced microrelief gilgai are developed.^[6]

Another important property of mantle deposits is also connected with fine-earth minerals. This is the color of deposits inherited by soils. In loamy and clayey substrates, clay particles serve as chromophores (coloring agents), because F(III) hydroxides are firmly adsorbed (non-exchangeable) on their surface. These iron (Fe) hydroxides are neither destroyed during erosion and redeposition nor are dissolved in acids and alkalis and can be removed only in reduction conditions, as trivalent iron is reduced to an easily soluble bivalent iron. The latter is used for determining Fe(III) hydroxides in soils.^[7]

There are several modifications of iron oxides formed in different reduction–oxidation, acid–alkali, and temperature conditions. In mantle deposits beyond tropical areas, Fe(III) hydroxides (amorphous α -FeO*OH) on the surface of clay particles have yellow–brown colors: from 7.5 to 10YR according to the Munsell's Color Charts.^[8] In the tropics and partly in the subtropics, such iron substances are represented mainly by α -Fe₂O₃, and several modifications of FeO*OH and have bright red and orange colors (from 10R to 2.5YR), which could be changed only in upper horizons (Fig. 1).

The color of parent rock is well preserved in conditions of a free gas exchange between soil and atmosphere.



Fig. 1 Podzolized (bleached) soils on red-colored substrate.

Therefore, many tropical soils have red or orange colors, and soils of cold and temperate regions have yellow–brown colors. In conditions of hindered gas exchange (caused by overmoistening of loose deposits) and oxygen deficiency, parent rocks and soils acquire bleached bluish-gray color (from 5GY to 5BG) due to iron reduction and dissolution (Fig. 2).

In case of pedogenesis on light-colored limestones, the color of deep soil horizons is inherited from the rock, whereas the upper horizons turned black because of humus accumulation (Fig. 3).

CHEMICAL COMPOSITION AND PARTICLE SIZE DISTRIBUTION IN MANTLE DEPOSITS

The bulk chemical composition of mantle deposits and their soils depend on the proportion of fine and coarse mineral fractions. A high content of silicon (Si) results from the predominance of quartz; the contents of aluminum (Al), Fe, potassium (K), and Mg depend on the amount of clay minerals (fraction <0.002 mm). The average chemical composition of mantle deposits and soils differs from that of the earth's crust by an



Fig. 2 Gleyed soils with bluish-gray color due to iron reduction.

increased share of Si, decreased shares of Al, calcium (Ca), Mg, K, and sodium (Na), and by the prevalence of Fe(III) over Fe(II).

The particle size distribution is the content of different particle size fractions in parent rocks and soils expressed in percent. This is a very important characteristic of parent rocks, because it results in mechanical and hydrophysical properties (stickiness, swelling capacity, rupture resistance, porosity, permeability, water capacity, water availability for plants, etc.) of soils formed on these deposits.

The particle size distribution in mantle deposits varies considerably. It is connected with the chemical composition of mantle deposits and their soils: The higher the share of clay fraction, the lower is the Si content, and the higher are the contents of Al, Fe, Mg, and K.

THE TRANSFORMATION OF MANTLE DEPOSITS

The third group of mineral components of mantle loams includes epigenetic mineral neoformations appearing after the accumulation of loose deposits as a result of soil-hypergenic processes. These are concretions and



Fig. 3 Mollisol on limestone—inheritance of white color in deep soil horizons and black color of upper humus horizon.

cements of Fe(III) and Mn(IV) hydroxides; calcareous concretions, pore encrustation, and films on fissure walls; Ca, Mg, and Na sulfates and Na chlorides in the form of efflorescence, individual crystals, and druses. Some epigenetic minerals are formed not by free crystallization in voids and fissures, but as a result of the selective metasomatic substitution of certain elements, which mainly affects very fine clay particles, while coarse mineral grains are only weakly corroded.^[9] The phenomenon of hypergenic metasomatism is most typical for the oxides of Fe(III) and Mn(IV) and, to some extent, for calcium carbonate. Besides the neoformation of individual minerals, epigenetic processes play an important role in forming the micromorphological fabric, porosity, and bulk density of mantle deposits. Deposits with most common particle size distribution (silty loams) have a bulk density of 1.5 g/cm³ and a porosity varying from 40% to 50% of the total volume.

From the aforesaid it follows that mantle deposits, the most widespread soil-forming substrates, are one of the most important factors of pedogenesis responsible for the particle size distribution, chemical composition, physico-mechanical, hydrophysical, and physico-chemical properties of many soils.

REFERENCES

1. Graham, R.C. Soils and mineral weathering on phyllite colluvium and serpentinite in Northwestern California. *Soil Sci. Soc. Am. J.* **1990**, *54*, 1682–1690.
2. Birkeland, P.W. *Pedology, Weathering, and Geomorphological Research*; Oxford University Press: New York, 1992; 286 pp.
3. Dobrovolsky, V.V. *Biogeochemistry of the Worlds Land*; CRC Press: Boca Raton, 1994; 362 pp.
4. Goldish, S.S. A study of rock weathering. *J. Geol.* **1938**, *46* (1), 17–58.
5. Grim, R.E. *Clay Mineralogy*; McGraw-Hill: New York, 1953.
6. Wilding, L.; Puentes, R., Eds. *Vertisols: Their Distribution, Properties, Classification and Management*; Texas A&M University Printing Center: College Station, 1988; 193 pp.
7. Mehra, O.P.; Jackson, M.L. Iron oxide removal from soils and clays system buffered with sodium bicarbonate by a ditionite-citrate. *Clay Miner.* **1960**, *5*, 317–327.
8. Pendleton, R.L.; Nickelson, D. Soil colors and special munsell soil color charts. *Soil Sci.* **1951**, *71*, 35.
9. Dobrovolsky, V.V. Micromorphological effects of metasomatic and colloidal phenomena during hypergenesis. In *Soil Micromorphology*; Jongerius, A., Ed.; Elsevier: Amsterdam, 1964; 131–137.

Root Growth: Pressure at the Soil—Root Interface

Rabi K. Misra

Faculty of Science, University of the Sunshine Coast, Sunshine Coast, Queensland, Australia

Abstract

In order to avoid reduction in root and plant growth in soils of high strength, soil and crop management options need careful consideration. Permanent, long-term reduction of strength in soils of unfavorable structure is difficult even with tillage. Sustainable soil and crop management practices incorporating minimum traffic and high retention of organic matter through residue management hold the greatest promise for favorable root growth and crop production by maintaining the soil strength below critical levels.

INTRODUCTION

Plant roots apply force to make cylindrical cavities in soil, which allow them to grow and elongate. However, in certain situations, roots need to apply only a small amount of force, for instance, when a root tip reaches an existing channel (made by earthworms, roots of a previous crop, or cracks from shrinkage of soil). Studies in homogeneous soils of uniform strength usually show development of localized zones of compressed soil around roots.^[1]

A root growing in soil exerts force within the zone of elongation behind the root cap (Fig. 1) until the elongation zone reaches maturity and the force shifts to a new elongation zone developed through cell division. Roots apply force in axial and radial directions.^[2] As roots have a cylindrical shape (some departure from cylindrical shape is expected for roots in very strong soils), they apply force over a cylindrical area. Therefore, root growth pressure is an estimated quantity from separate measurements of force and area. For axial pressure, area over which the force is exerted is circular, which can be estimated from the measurement of diameter in the elongation zone of the root. For radial pressure, force is exerted over a cylindrical surface, so both diameter and length of the root are important. At the soil–root interface, axial and radial root growth forces are expected to vary with time, and so does the area over which these forces are exerted.

THEORETICAL BASIS

The estimation of root growth pressure from direct measurements of force and diameter of root is time-consuming, and these measurements are possible for single roots of seedlings in a laboratory. To overcome this, many studies rely on assessing soil resistance to root growth (equivalent to indirect estimation of root growth pressure) from rapid measurements of soil strength using penetrometers. Penetrometers measure pressure required to push root-shaped

(conical) metal probes into soil. These pressures are referred to as penetrometer resistances or cone indices in the literature. There is often a good correspondence between root growth pressure and penetrometer pressure^[3]

when roots are not growing in cracks or holes. However, penetrometers tend to overestimate root growth pressure by 1.8 to 7.5 times.^[3,4] Critical values of penetrometer resistance, which halves root elongation rates in soil, are shown in Table 1. Data in this table show substantial differences in root growth response between crops, which gives the opportunity to select crops tolerant to soils of high strength.

The nature of soil deformation by root tips and cones and the mechanical stress at the soil–root and soil–cone interfaces can be explained well using the theory of soil mechanics for homogeneous soils and aggregate.^[1,3,11,12]

These and similar theories and associated experimental evidence suggest that friction at the soil–root interface could be substantially lower than the friction at the soil–cone interface. Studies suggest that cells produced from sloughing of root cap might reduce soil–root friction, which would allow roots to penetrate strong soils without the need to exert as high a pressure as a penetrometer cone.^[13]

Despite the differences between penetrometer resistance and root growth pressure, cone penetrometers continue to be used as a surrogate measure of root growth pressure at the soil–root interface and in the diagnosis of soil compaction to determine the extent of soil limitation to root growth in the field.

ROOT GROWTH PRESSURE IN COMPACTED SOILS

In compacted soils of high bulk density and high soil strength, root growth rate is reduced because the axial pressure applied by roots is not sufficient to overcome soil strength, and root growth may stop when root is unable to deform soil. In this situation, axial root growth pressure is expected to reach a maximum. In less compacted soil,

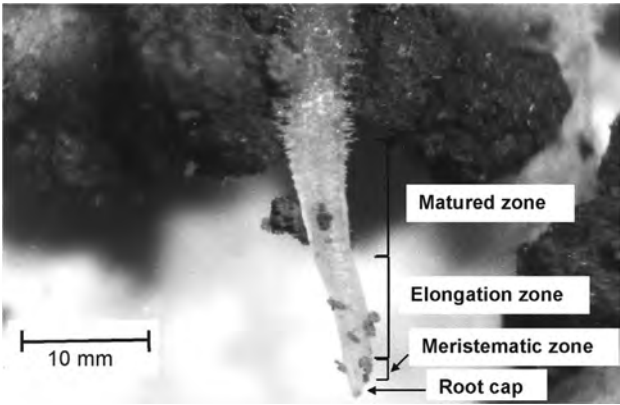


Fig. 1 Developmental zones in a seedling root of cotton.

axial force applied by roots increases with time until the pressure is sufficient to overcome soil strength and then it declines. Values of maximum axial growth pressure are well documented, and they are in the range of 0.2–1.5 MPa (Table 2). However, direct estimates of radial root growth force and pressure are scarce in the literature. The importance of radial root growth pressure for root penetration in compacted soils is also not so well understood as the axial root growth pressure.

MEASUREMENTS OF AXIAL ROOT GROWTH PRESSURE

In the late 19th century, Pfeffer reported maximum axial and radial growth pressures for primary roots of a limited number of seedlings.^[14] Although Pfeffer's measurement technique is now refined, the principle of measurement has changed little.

Most reports on root growth force are limited to primary roots of young seedlings (≤ 1 -week old) in laboratory experiments. Measurements on roots of older seedlings (e.g., nursery-raised tree seedlings) and comparison of forces exerted by lateral roots with primary roots suggest that the ability of roots to apply force may decline with age

Table 1 Critical values of soil penetrometer resistance P (MPa) at which elongation rate of the primary root is halved.

Plant species	P (MPa)	References
Maize (<i>Zea mays</i>)	1.30	[5]
Pea (<i>Pisum sativum</i>)	2.03	[6]
Tomato (<i>Lycopersicon esculentum</i>)	1.48	[6]
Ryegrass (<i>Lolium multiflorum</i>)	1.39	[6]
Peanut (<i>Arachis hypogaea</i>)	1.91	[7]
Cotton (<i>Gossypium hirsutum</i>)	0.72	[7]
Eucalypt (<i>Eucalyptus nitens</i>)	2.54	[8]
Pine (<i>Pinus radiata</i>)	1.30	[9]

Source: Data taken from Gooderham^[10] and Dexter.^[6]

Table 2 Maximum axial growth pressure applied by the seedling primary roots of various crop plants.

Plant species	Mean pressure (MPa)	References
Maize (<i>Z. mays</i>)	1.451	[14]
Pea (<i>P. sativum</i>)	0.497	[2]
Chickpea (<i>Cicer arietinum</i>)	0.450	[15]
Peanut (<i>A. hypogaea</i>)	1.150	[16]
Broad bean (<i>Faba vulgaris</i>)	1.082	[14]
Common vetch (<i>Vicia sativa</i>)	1.080	[14]
Sunflower (<i>Helianthus annuus</i>)	0.238	[2]
Cotton (<i>G. hirsutum</i>)	0.289	[2]
Lupin (<i>Lupinus albus</i>)	0.645	[17]
Eucalypt (<i>E. nitens</i>)	0.160 ^a	[18]

^aMeasured on lateral root of nursery seedlings.

and possibly with increased branching order.^[18] This might be one of the reasons for the development of herringbone rooting pattern at the highest branching order observed for many species of plants.

Axial root growth force is commonly measured on intact seedlings with a 20–40 mm long primary root. The root axis is anchored to the bottom of a pot, allowing 3–5 mm of the root tip to protrude through a hole at the bottom of the pot. This pot is filled with soil or vermiculite (containing adequate water and nutrients) with care to avoid any damage to the root. The protruding root tip is then pushed into a hole in the soil in a second pot to an extent that a narrow air gap of ~ 0.5 mm between the two pots is maintained (Fig. 2). A force-measuring device supports the second pot. Axial force is commonly measured using a data logger over short time intervals (5–10 minutes) until it reaches maximum and then declines (Fig. 3). Measurements are usually stopped when the maximum axial force is known. Root diameter is measured in the air gap following the termination of the experiment, and these data are used to calculate the circular area over which maximum axial force would have been exerted. Maximum axial growth pressure is calculated as maximum force divided by the area. In many experiments, a wet, porous ceramic block with a cylindrical hole

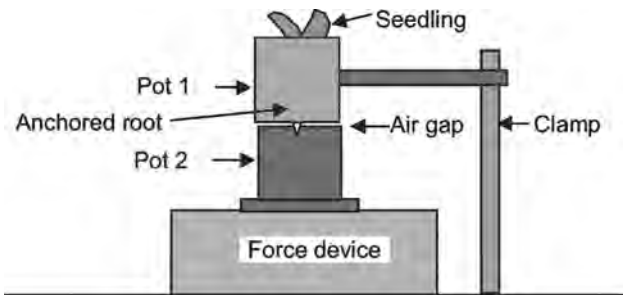


Fig. 2 Experimental arrangement for the measurement of axial root growth force exerted by a typical seedling at the soil–root interface.

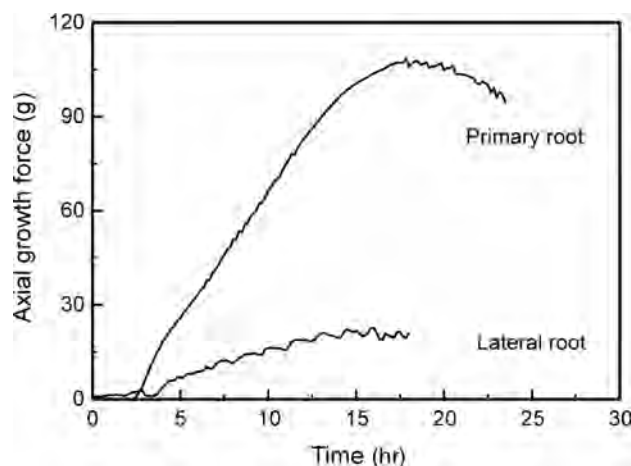


Fig. 3 Typical variation of axial root growth force with time for seedling roots of pea.

Source: From Misra.^[18]

(to position the root tip) replaces the second pot containing the soil. These experiments aim to measure maximum axial root growth pressure to establish the ability of seedling root to exert axial force.

CONCLUSION

Data on maximum root growth force are generally consistent between various experimental methods and laboratories. However, there are some uncertainties with root diameter measurements. Most measurements of root diameter are made in the air gap after removal of the seedling from the apparatus. This might overestimate the root diameter, as roots can expand slightly when the mechanical stress is removed.^[18] Root growth pressure in soil may also be influenced by variables other than soil strength such as root temperature, species of plants, and cultivar.

Roots must perform work to grow through soil. Energy requirement for root penetration is expected to be high in strong soils. As energy for root growth is derived through assimilates fixed by leaves, growth and partitioning of resources might be affected in plants growing on strong soils.

In order to avoid reduction in root and plant growth in soils of high strength, soil and crop management options need careful consideration. In situations where the zone of high soil strength is close to the surface, loosening of surface soil with shallow tillage (using discs, tines, and ploughs) will reduce soil strength. Tillage with deep ripping is recommended where high strength occurs in the subsoil. However, care is needed to avoid excessive tillage that is known to deteriorate soil structure, decrease organic matter, and accelerate erosion. Permanent, long-term reduction of strength in soils of unfavorable structure is difficult even with tillage. Sustainable soil and crop management practices incorporating minimum traffic and high retention of organic matter through residue management hold the

greatest premise for favorable root growth and crop production by maintaining soil strength below critical levels.

REFERENCES

1. Greacen, E.L.; Farrell, D.A.; Cockcroft, B. Soil resistance to metal probes and plant roots. In *Transactions of the 9th Congress of International Soil Science Society*, Adelaide, South Australia, 1968; 769–779.
2. Misra, R.K.; Dexter, A.R.; Alston, A.M. Maximum axial and radial growth pressures of plant roots. *Plant Soil* **1986**, *95*, 315–326.
3. Misra, R.K.; Dexter, A.R.; Alston, A.M. Penetration of soil aggregates of finite size. II. Plant roots. *Plant Soil* **1986**, *94*, 59–85.
4. Bengough, A.G.; Mullins, C.E. Penetrometer resistance, root penetration resistance and root elongation rate in two sandy loam soils. *Plant Soil* **1991**, *131*, 59–66.
5. Mirreh, H.F.; Ketcheson, J.W. Influence of soil water matric potential and resistance to penetration on corn root elongation. *Can. J. Soil Sci.* **1973**, *53*, 383–388.
6. Dexter, A.R. Mechanics of root growth. *Plant Soil* **1987**, *98*, 303–312.
7. Taylor, H.M.; Ratliff, L.F. Root elongation rates of cotton and peanuts as a function of soil strength and soil water content. *Soil Sci.* **1969**, *108*, 113–119.
8. Misra, R.K.; Gibbons, A.K. Growth and morphology of eucalypt seedling-roots, in relation to soil strength arising from compaction. *Plant Soil* **1996**, *182*, 1–11.
9. Zou, C.; Penfold, C.; Sands, R.; Misra, R.K.; Hudson, I. Effects of soil air-filled porosity, soil matric potential and soil strength on primary root growth of radiata pine seedlings. *Plant Soil* **2001**, *236*, 105–115.
10. Gooderham, P.T. *Soil Physical Conditions and Plant Growth*. Ph.D. thesis, University of Reading: Reading, 1973.
11. Farrell, D.A.; Greacen, E.L. Resistance to penetration of fine probes in compressible soil. *Aust. J. Soil Res.* **1966**, *4*, 1–17.
12. Misra, R.K.; Dexter, A.R.; Alston, A.M. Penetration of soil aggregates of finite size. I. Blunt penetrometer probes. *Plant Soil* **1986**, *94*, 43–58.
13. Bengough, A.G.; McKenzie, B.M. Sloughing of root cap cells decreases the frictional resistance to maize (*Zea mays* L.) root growth. *J. Exp. Bot.* **1997**, *48*, 885–893.
14. Pfeffer, W. Druck und Arbeitsleistung Durch Wachsende Pflanzen. *Abh. Königlich Sächsischen Ges. Wiss.* **1893**, *33*, 235–474.
15. Aggarwal, G.C.; Prihar, S.S. A simple technique to determine axial root growth force. *Plant Soil* **1975**, *42*, 485–489.
16. Taylor, H.M.; Ratliff, L.F. Root growth pressures of cotton, peas and peanuts. *Agron. J.* **1969**, *61*, 398–402.
17. Whalley, W.R.; Dexter, A.R. The maximum axial growth pressure of roots of spring and autumn cultivars of lupin. *Plant Soil* **1993**, *157*, 313–318.
18. Misra, R.K. Maximum axial growth pressures of the lateral roots of pea and eucalypt. *Plant Soil* **1997**, *188*, 161–170.
19. Clark, L.J.; Bengough, A.G.; Whalley, W.R.; Dexter, A.R.; Barraclough, P.B. Maximum axial root growth pressure in pea seedlings: Effects of measurement techniques and cultivars. *Plant Soil* **1999**, *209*, 101–109.

Root Monitoring System

Zhikuan Jia

Pute Wu

Qingfang Han

Junfeng Nie

Ruixia Ding

Yufeng Zou

Institute of Water Saving Agriculture in Arid Areas of China, Northwest A&F University, Yangling, China

Abstract

Plant root monitoring systems directly observe the growth status of plant roots in Plexiglas soil columns without damaging the plant, providing real-time and dynamic data of plant root formation, transpiration, and root growth throughout the day. The small retractable windows on the Plexiglas soil columns are used to monitor and sample the rhizosphere soil for moisture, nutrients, temperature, and microbiological indicators, allowing for the analysis of various physiological and biochemical indicators. The plant root monitoring system can also be used for root pruning experiments, which can reduce errors caused by single-field experiments and thus providing a way to conduct intensive study of plant roots functions. The system is easily maintained, widely used, and operates effectively.

INTRODUCTION

Plant roots are one of the most important components of plants, and its growth status is closely related to the growth of the whole plant. Roots absorb, transfer, and store nutrients and water, as well as function of fixing and supporting plant body,^[1] playing an important role in the biogeochemical cycle of ecosystems.

While the root systems have been focused in biological research, due to the complexity and difficulty of observing and sampling, its progress has been relatively slow. As it is relatively difficult to directly observe the growth status of root systems, physiological and biochemical research of root systems is not intensive enough. The primary method for analyzing root systems rely on methods that are destructive and result in incomplete sampling, such as digging,^[2] soil-coring,^[3] and profiling methods.^[4] These methods are time consuming and labor intensive, terminate the growth of plants, and additionally, the collected samples are incomplete or have different degrees of damage, making the results difficult to interpret. The commonly used method for *in situ* root system observation is by installing an observation device in the soil, such as the rhizobox method^[5] and the minirhizotron technique.^[6,7] The minirhizotron method is one of the most commonly used methods when monitoring the growth of root systems in the fields. The ray computed tomography (CT) method is the most commonly used method for monitoring root system morphology, and it

utilizes the mature ray imaging principle to generate images of the root system in soil samples, such as the X-ray CT^[8,9] or magnetic resonance imaging,^[10] however, this method requires relatively expensive equipment and it can only observe limited space, thus the application is not universal. The selection of research method for plant roots influences the results with respect to aboveground part of plant. Therefore, it is necessary to develop methods that allow for *in situ* observation of root growth and spatial distribution of root morphology. In order to better study plant root, it is of great significance to establish specialized research devices that allow automatic monitoring, analysis, and condition control of plant root system.

ROOT EXCAVATION METHOD

Analyzing the root system architecture and physiology and biochemistry of the roots could be realized through excavation of the root system close to the ground (Fig. 1). The method employed a sampling device to excavate the plant root system and its surrounding soil within certain scopes,^[2] in approximately 10 cm intervals. After washing through soil sieves with aperture size over 0.25 mm, and using scanned images of the root system within the sampling range, this method allows for quantitative observation of branched roots length at different levels and number of branches. It also allows measuring of relevant



Fig. 1 Excavation method.

physiological and biochemical indicators of the root system. The excavation method is less labor intensive, more cost effective, and less time consuming, but its shortcoming is that the sampling range is limited.

ROOT-CORING METHOD

Root auger with inner diameter over 6.0 cm is used to conduct stratified sampling of the plant root system (Fig. 2). Under field conditions, multipoint sampling is generally carried out along rows and vertical rows, and the sampling is usually conducted in 10 or 20 cm



Fig. 2 Root-coring method.

intervals. The soil sieve with bore diameter over 0.25 mm is used to wash the composite samples of roots and soil, and the morphological analysis of the root system within the sampling range and the measurement of physiological and biochemical indicators could be conducted through image scanning. The longitudinal sampling range of this method is not limited, but it requires high labor intensity.

MINIRHIZOTRON METHOD

The minirhizotron method installs minirhizotrons (transparent Plexiglas tubes with length of 1.8 m and external diameter of 5.2 cm) in field experiments.^[11,12] Before installation of the minirhizotrons, the bottom of the tube is sealed and then launched with a 45° angle to the ground, then exposed part above ground (about 30 cm) is covered with black tape after fixing the minirhizotrons. A rubber cap is installed on the tube nozzle in case water or dust falls into the tubes. The distance between the installed minirhizotrons is generally 50 cm and each group of four minirhizotrons is used to collect data from the same treatment. The minirhizotrons are adopted to conduct dynamic monitoring, and the system collects images from $1.05 \times 1.45 \text{ cm}^2$ range by connecting a microcamera with slide control to the computer using image scanning software for observation. After the images of the root systems are collected, the root analysis software WinRhizoTron is used for analysis to acquire root parameters such as root length, root diameter, number of root tips, and surface area. However, images acquired via minirhizotron method can have complicated soil background and poor root continuity (Fig. 3).



Fig. 3 Minirhizotrons technique.



Fig. 4 MRI of plant roots.

NUCLEAR MAGNETIC RESONANCE METHOD

Multislice spiral CT can observe the microscopic structure in human bodies, thus the multislice spiral CT machines can also be used to conduct scanning and 3-D imaging of plant root system. It allows for the collection of root system volume using a CT3D workstation, and then the image postprocessing software MIP can be used to generate images. The quality of images is influenced by the scanning technical solutions (parameters), imaging technical solutions (images processing and algorithm reconstruction), and density difference between plant root system and growth medium. The method is subject to the limitation of root growth environment; therefore, it is very difficult to obtain complete data of *in situ* root morphology, especially at the middle to late stage of crop growth (Fig. 4).^[13]

ROOT-OBSERVING SYSTEM

The observing system built with steel and concrete based on the plant root system, with an observing chamber (standard holes) in which 20 Plexiglas columns with height of 2.8 m can be placed (Fig. 5).

The aboveground part is designed according to the requirements of plant root system and adopts the arched architecture made of rustproof steel. It is composed of a roof made of sheet material that shields light and blocks rain, the surrounding parts made of waterproof material that enables ventilation, and the mobile greenhouse supplies power and water, with a steel lifting device and aboveground antiradiation plate (Fig. 6).

The roof of the underground part of the system is constructed by pouring steel and concrete according to the system requirements. In addition, the standard concrete holes for placing Plexiglas soil columns are reserved. It consists of Plexiglas soil columns for observing and



Fig. 5 Plexiglas soil columns for root observing.

monitoring (which is developed separately and contains 14 minirhizotron windows and lifting infrastructure), automatic monitoring system of evaporation mass balance, automatic monitoring system of soil column moisture, temperature monitoring, soil-shading equipment and lifting support, illumination system required for observation, and power supply system for electric equipment such as experiment instruments (Fig. 7).



Fig. 6 Aboveground part of root observing system.



Fig. 7 Underground part of root observing system.

Auxiliary facilities of the system include devices that allow for automatic monitoring, analyzing, and researching system of plant root system. These include SM200-DT80 soil moisture dynamic monitoring system (soil moisture transducer and digital collector), root analysis system WinRHIZO, precise electronic balance (placed in the bottom of soil columns and its measurement range is within 600 kg and the sensitivity is ± 50 g), monitoring and analyzing instruments such as plant root pruning trepans. In addition, according to the system requirements, there is a tank to wash the roots in the sedimentation tank.

Root growth status can be observed directly without doing damage to original plants using the root system automatic monitoring system (Fig. 8). The plant root monitoring subsystem allows for real-time and dynamic collection of data, along with images of the root system, and measurement of plant transpiration and growth parameters throughout the day using the auxiliary facilities. The plant root and leaf analyzing subsystem can be used to analyze images of plant root system and leaves and explore the relationship between leaf growth and root growth. Soil moisture, nutrients, temperature, and microbiological indicators can be monitored using the small retractable windows. The retractable windows can also be used to prune the roots to collect samples or carry out analyses on physiological and biochemical indicators, improving the reliability of root physiological and biochemical experiments and provide favorable conditions for intensive study of plant root system. This system can be used for a long period and is fast, intuitive, time saving, easy operation, fast measurement, and high working efficiency when compared to the traditional root sampling methods. By employing the

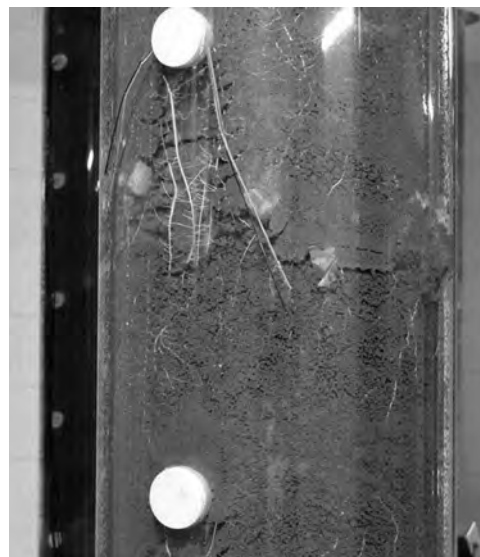


Fig. 8 Observation of *in situ* root system.

plant root research system, it is convenient to conduct a variety of analyses including the root physiology and biochemistry in the cultivation of crops, forage grass and flowers, as well as spatial distribution of root system morphology and ecological research of plant root system. Studies of crop growth and water-saving irrigation in agricultural ecosystem, physiological and biological responses of plant root system to adversity stress, such as salt, drought, and heavy metals, and senescence of plant root pruning are some of the examples that this method could be employed effectively.

REFERENCES

1. Steudle, E. The cohesion-tension mechanism and the acquisition of water by plant roots. *Annu. Rev. Plant Physiol. Plant Mol. Biol.* **2001**, 52, 847–875.
2. Wu, J.; Guo, Y. An integrated method for quantifying root architecture of field-grown maize. *Ann Bot-London* **2014**, 114 (4), 841–851. DOI:10.1093/aob/mcu009.
3. Bolinder, M.A.; Angers, D.A.; Dubuc, J.P. Estimating shoot to root ratios and annual carbon inputs in soils for cereal crops. *Agr. Ecosyst. Environ.* **1997**, 63, 61–66.
4. Li, Z.Y.; Qi, X.B.; Fan, X.Y.; Wu, H.; Du, Z.; Li, P.; Lü, M. Influences of biochars on growth, yield, water use efficiency and root morphology of winter wheat. *Trans. CSAE* **2015**, 31 (12), 119–124. (in Chinese with English abstract).
5. Hawkins, H.J.; George, E. Effect of plant nitrogen status on the contribution of arbuscular mycorrhizal hyphae to plant nitrogen uptake. *Physiol. Plantarum* **1999**, 105 (4), 694–700. DOI:10.1034/j.1399-3054.1999.105414.x.
6. Majdi, H. Root sampling methods—Applications and limitations of the minirhizotron technique. *Plant Soil* **1996**, 185 (2), 255–258. DOI:10.1007/BF02257530.
7. Qiu, X.Q.; Gao, Y.; Huang, L.; Li, X.Q.; Sun, J.S.; Duan, A.W. Temporal and spatial distribution of root morphology

- of winter wheat. *Sci. Agr. Sinica* **2013**, *46* (11), 2211–2219. (in Chinese with English abstract).
8. Gregory, P.J.; Hutchison, D.J.; Read, D.B.; Jenneson, P.M.; Gilboy, W.B.; Morton, E.J. Non-invasive imaging of roots with high resolution X-ray micro-tomography. *Plant Soil* **2003**, *255* (1), 351–359. DOI:10.1023/A:1026179919689.
 9. Luo, X.W.; Zhou, X.C.; Yan, X.L.; Luo, L.P.; Xiang, Z.Y. Visualization of plant root morphology *in situ* based on X-ray CT imaging technology. *Trans. CSAM* **2004**, *35* (2), 104–106. 133. (in Chinese with English abstract).
 10. Schulz, H.; Postma, J.A.; van Dusschoten, D.; Scharr, H.; Behnke, S. Plant root system analysis from MRI images. In *Computer Vision, Imaging and Computer Graphics. Theory and Application*, 7th International Joint Conference, Rome, Italy, Feb 24–26, 2012; Csurka, G., Kraus, M., Laramee, R.S., Richard, P., Braz, J., Eds.; Springer: Berlin Heidelberg, 2012; 359, 411–425.
 11. Mooney, S.J.; Pridmore, T.P.; Helliwell, J.; Bennett, M.J. Developing X-ray computed tomography to non-invasively image 3-D root systems architecture in soil. *Plant Soil* **2012**, *352* (1–2), 1–22. DOI:10.1007/s11104-011-1039-9.
 12. Johnson, M.G.; Tingey, D.T.; Phillips, D.L.; Storm, M.J. Advancing fine root research with minirhizotrons. *Environ. Exp. Bot.* **2001**, *45* (3), 263–289. DOI:10.1016/S0098-8472(01)00077-6.
 13. Shanghai Tainiu Electronic Technology Co. Ltd. 2014. *Introduction of Product Application: Nuclear Magnetic Resonance Imaging of Plant Roots*. Available at <http://www.niumag.com.cn>.

Salt-Affected Soils

Anna Eynard

Ohio State University, Columbus, Ohio, U.S.A.

Rattan Lal

*Carbon Management and Sequestration Center, Ohio State University,
Columbus, Ohio, U.S.A.*

Keith D. Wiebe

*Resource Economics Division, U.S. Department of Agriculture (USDA),
Washington, District of Columbia, U.S.A.*

Abstract

Salt-affected soils decrease crop productivity at local and global scales. The extent of soil salt problems is likely to increase if no measures are taken to limit soil salinization and sodication. Management and reclamation of salt-affected soils require a combination of agronomic practices, including hydraulic, mechanical, amending, and cropping practices. The selection of proper management practices is relative to the soil, water, crop, and human available resources.

INTRODUCTION

Salt-affected soils can be defined as soils on which the growth of most crop plants is limited by an excess of easily soluble salts. Salts are considered easily soluble when they are more soluble in water than gypsum. Salts may include chlorides, sulfates, carbonates, and bicarbonates of sodium, potassium, magnesium, and calcium. The salt concentration in the soil solution is usually measured by the electrical conductivity (EC) of the soil saturation extract. A soil is saline if the EC of the extract is higher than 4 dS m^{-1} .^[1] However, plant growth may be severely affected or inhibited at much lower levels of EC. Limiting EC values change with the ionic composition of the soil solution and with the plant species and variety grown. Sodic soils are salt-affected at $\text{EC} < 4 \text{ dS m}^{-1}$ (Table 1). In sodic soils, the high sodium content relative to other cations is the main factor affecting plant growth. The amount of sodium is usually expressed as exchangeable sodium percentage (ESP) or sodium adsorption ratio (SAR). The critical level of ESP for growth reduction depends on many interacting factors and may vary from <6% to >15%. There is a continuum of plant responses to the continuous increase of salt stress with increasing salt or sodium concentrations in the soil. Saline and sodic conditions, although very different, are often implicated in adversely affecting plant performance, so that terms such as salinity, saline, and salinization are generically referred to soil conditions and processes leading to salt and sodium stresses on plants.^[2]

EXTENT AND DISTRIBUTION

Salt-affected soils are most common in arid climates with evapotranspiration greater than precipitation for at least part of the year (aridic, ustic, or xeric soil moisture regimes), but they are present at any latitude and altitude. Salinization by improper irrigation management was a major cause of degradation of the Nile Delta, the Mesopotamian Plain, and the valley of the Yangtze and the Hwang Ho.^[3] Estimates of the extent of soil salinity and sodicity problems vary from 5% to 10% of the world land area. Estimates from the Food and Agriculture Organization, Land and Water Development (FAO-AGL)^[4] indicate that >800 million hectares are salt-affected, i.e., >6% of the world land area (Fig. 1). Salinization affects about 70 million hectares of irrigated lands, about one-third of the world irrigated area. The *World Map of the Status of Human-Induced Soil Degradation* indicates that about 77 million hectares are affected by human-induced salinization,^[5] comprising 45 million hectares of irrigated and 32 million hectares of rain-fed agricultural land.^[4] Salinity problems also affect greenhouse crops, mine spoils, and disposal areas. Part of the yield loss occurs on irrigated lands, which are otherwise highly productive. In the irrigation district of Ciudad Juarez Valley in Mexico, the productivity of 70% of the lands is hindered by salinity. On average, 25% of irrigated Mexican lands are salt-affected. The impact of yield losses is magnified at a local scale because salt-affected soils are not uniformly distributed, threatening the economic welfare of some regions and countries. In Bangladesh, 24% of the total land area is salt-affected with a rapid expansion

Table 1 General classification of salt-affected soils.

Soil class	EC (dS m ⁻¹)	SAR	ESP (%)
Non-saline non-sodic	<4	<13	<15
Saline	>4	<13	<15
Saline-sodic	>4	>13	>15
Sodic	<4	>13	>15

during the last-quarter of the 20th century from <1 million hectares to >3 million hectares. In Australia, almost 20% of the land is salt-affected.^[4]

Salt-affected soils not used for crop production (e.g., salt flats) are a potential source of salts for the salinization of surrounding fields. According to Oldeman et al.,^[5] the annual rate of loss of agricultural land by salinization, alkalization, and waterlogging is about 1.5 million hectares. Previous estimates reported 10 million hectares of agricultural land loss per year.^[6] The rate of increase is likely to vary widely across regions and from year to year, but all data suggest an increasing extent of salt-affected lands, especially where the natural resources are scarce and the growing human population increases pressure on soils. Changes in the global environment are likely to increase yield losses consequent to soil salinity. The predicted increase of the sea level from thermal expansion of sea water ranges from 15 cm to >50 cm by the year 2100.^[7] The rise of sea water is likely to worsen salinity problems from the tidal inundation of coastal lands. On the other hand, there is no evidence to suggest that elevated atmospheric carbon dioxide (CO₂) would increase the level of salinity suitable for plant growth.^[8] Increased CO₂ is expected to increase plant growth only on non-saline soils, magnifying yield losses due to soil salinity.

CROP TOLERANCE

Soil salinity and sodicity limit the potential area of growth of sensitive crops. All plants are sensitive to salts at some concentration, but the limiting concentration varies with plant species and variety. Crop tolerance is defined in relation to the level of salinity causing yield losses. High salt concentration may lead to plant death and no yield. Decreased growth occurs above a soil critical salt or sodium concentration (tolerance threshold). Crop yield reductions in salt-affected soils result from the alteration of various metabolic processes in plants under salt stress. Negative effects of excess of salts in the soil solution include increased osmotic pressure limiting water uptake (physiological drought), abnormal pH and ionic competition limiting nutrient uptake, and ionic toxicity. Total salinity effects always depend on the specific soil ionic composition. The negative response of plants to low water potential may prevail in saline soils, while single ion toxicity or nutritional imbalance may be particularly severe in sodic soils. Soil structural impedance of plant growth is common in the absence of sufficient Ca²⁺. Structural problems derive from crusting, formation of compacted layers, poor infiltration, and poor permeability to water and air. In waterlogged soils, anaerobic conditions may enhance salinity stresses.

In all plants growing on saline media, a part of the metabolic energy is diverted to ion transport, synthesis of organic osmolytes (contributing to osmotic adjustment to low water potential), and ion compartmentation preventing ion toxicity.^[2] Osmotic adjustments in salt-tolerant plants need to follow seasonal changes in soil salinity and water availability. In general, plants that are more salt-tolerant tend to grow more slowly at low

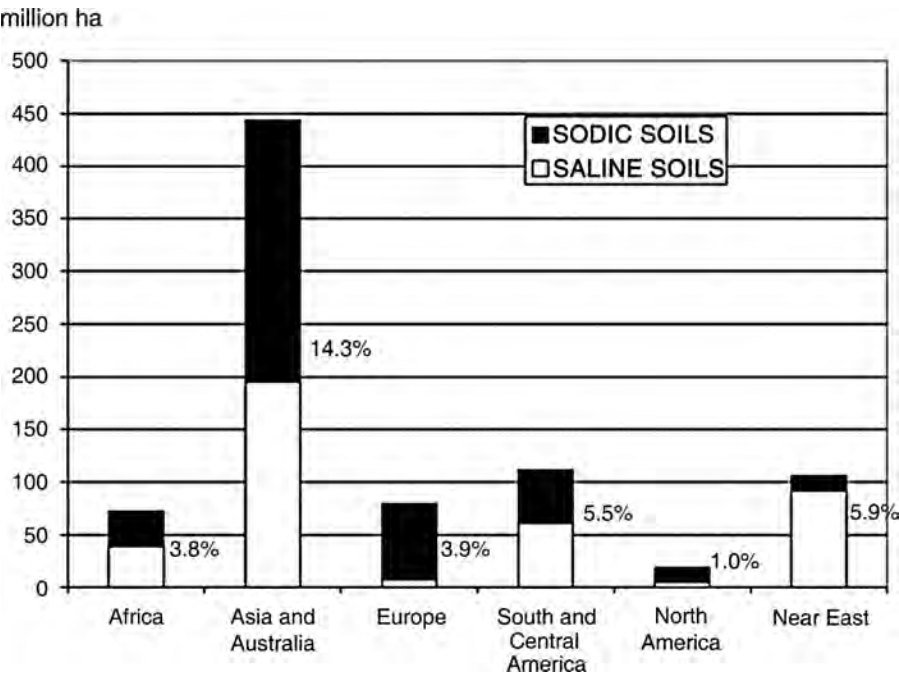


Fig. 1 The distribution of salt-affected soils. The percentage of salt-affected soils on the total land surface is reported for each region of the world.
Source: From FAO-AGL.^[4]

salinity levels than less-tolerant plants. Broadly adaptable plants can produce good yields where strong temporal changes of soil salinity occur in the soil. But such plants, tolerant to a wide range of salinities, perform less well at any salinity than less adaptable plants at their optimal salinity.^[8] Reduced yield in tolerant plants can be related to: 1) greater allocation of organic carbon (C) in the roots of tolerant plants at the expense of leaf area; 2) decreased use of the solar radiation; and 3) low transpiration rate. Soil salinity interacts with climate and other edaphic factors in determining yield. Reduced availability of water in the soil is concomitant to higher evaporative demand in the leaves. The adaptation to soil salinity is a reduction in the capacity for photosynthetic C assimilation consequent to the necessity of minimizing evapotranspiration. In general, crops are less sensitive to salinity in glasshouse conditions than outdoors. At low soil fertility levels, plants may appear more tolerant to salinity than at high fertility levels if salinity is not the limiting factor of yield. On the other hand, nutrient deficiencies induced by salinity and sodicity can further reduce low yields due to low fertility.

In natural conditions, soil salinity varies with depth and time. Plants extract water from the least saline parts of the soil surrounding the roots. Plants may be able to tolerate high salinity levels in the root zone if occurring only for short periods of time during the growing season. Soil salinity may peak at different stages of the crop, when plants may be more or less sensitive. Consequently, the critical point for yield reductions varies with the duration of salt stress and stage of development. Growth can be inhibited at any stage of the biological cycle. Severe reductions of yields can be due to low germination and limited early plant establishment. Yield reductions can be caused by reduced vegetative growth and/or by perturbation of the reproductive phases. Developmental shifts of relative salt-tolerance are common during plant life and vary with cultivar and environment. Often, qualitative yield reductions enhance total yield losses.

A common classification of crop salinity tolerance distinguishes five categories (Table 2). Sodium is not considered a limiting factor in the above classification.

Table 2 The classification of crop salt-tolerance based on the values of EC above which yield loss begins.

Relative tolerance	Critical EC (dS m ⁻¹)	Examples of crops
Sensitive	<1.3	Beans
Moderately sensitive	1.3–3.0	Rice, corn
Moderately tolerant	3.0–6.0	Wheat
Tolerant	6.0–10.0	Barley, cotton
Unsuitable	>10.0	Most crops

Source: From Ayers & Wescot.^[9]

Table 3 The classification of crop exchangeable sodium tolerance.

Relative tolerance	ESP (%)	Examples of crops
Sensitive	<15	Beans, corn
Semitolerant	15–40	Rice, wheat
Tolerant	>40	Barley, cotton

Note: Growth is usually already affected at lower values of ESP, and crop production is usually excluded at ESP >30.

Source: From Ayers & Wescot.^[9]

A classification of sodium tolerance of crops consists of three groups (Table 3). Sodium-tolerance tables are usually based on nutritional responses in absence of soil structural degradation, which generally excludes crop production at ESP >30.^[9] These classifications are based on the agronomic criterion of comparing the relative yield of each crop on saline media to its yield on normal non-saline media. Published tolerance tables provide a reference of relative salt tolerance of crops under irrigation without local climatic specifications. Such data refer to the duration of salinity from late seedling stage to maturation with irrigation, fertilization, and pest control maintained optimal for each crop species and variety at the time of the experiment. Tolerance tables may apply to individual cultivars in the absence of interactions between soil salinity and other environmental factors.

CONCLUSION

Salt-affected soils decrease crop productivity at local and global scales. The extent of soil salt problems is likely to increase if no measures are taken to limit soil salinization and sodication. Management and reclamation of salt-affected soils require a combination of agronomic practices, including hydraulic, mechanical, amending, and cropping practices.^[4] The selection of proper management practices is relative to the soil, water, crop, and human available resources.

Improved crop salt tolerance can contribute to the remediation of soil salt problems because it:

1. Extends the choice of crops that can grow at each soil salinity level.
2. Allows irrigation using more saline water.
3. Increases soil organic matter and improves soil structure.

Consequently, tolerant crops may assist establishing a proper water balance and reduce the need and cost of reclamation measures such as leaching.

Information on crop response to salinity and sodicity needs to be updated in relation to changes of plant materials, agronomic practices, and climatic conditions. A range of values in the continuum of salt and sodium stresses would

be better suited than a single critical value to represent the tolerance or sensitivity of a plant since the intensity of salt stress depends on many other factors determining yields.

REFERENCES

1. SSSA. *Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 1997; 134 pp.
2. Läuchli, A.; Epstein, E. Plant responses to saline and sodic conditions. In *Agricultural Salinity Assessment and Management*; Tanji, K.K., Ed.; ASCE Manuals and Reports No. 71; ASCE: New York, 1990; 113–137.
3. Szabolcs, I. Salt buildup as a factor of soil degradation. In *Methods for Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentine, C., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1998; 253–264.
4. FAO-AGL. *Global Network on Integrated Soil Management for Sustainable Use of Salt-Affected Soils*. FAO-AGL, Land and Plant Nutrition Management Service (2000). Available at <http://www.fao.org/ag/AGL/ag11/spush>.
5. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *Second Revised Edition. World Map of the Status of Human-Induced Soil Degradation. An Explanatory Note*; International Soil Reference and Information Center: Wageningen, 1991; 35 pp.
6. WCED. *Our Common Future*; Oxford University Press: Oxford, 1987; 383 pp.
7. Warrick, R.A.; Le Prevost, C.; Meier, M.F.; Oerlemans, J.; Woodworth, P.L. Changes in sea level. In *Climate Change 1995*; Houghton, J.T., Meira Filho, L.G., Callander, B.A., Harris, N., Kattenberg, A., Maskell, K., Eds.; Cambridge University Press: Cambridge, 1996; 363–405.
8. Ball, M.C.; Sobrado, M.A. Ecophysiology of mangroves: Challenges in linking physiological processes with patterns in forest structure. In *Physiological Plant Ecology*, The 39th Symposium of the British Ecological Society, Held at the University of York, Sep 7–9, 1998; Scholes, J.D., Barker, M.G., Press, M.C., Eds.; Blackwell Science: Oxford, 1999; 331–346.
9. Ayers, R.S.; Wescot, D.W. *Water Quality for Agriculture*; FAO Irrigation and Drainage Paper 29, Rev. 1; FAO: Rome, 1989; 174 pp.

Salt-Affected Soils: Nitrous Oxide Emissions

G.S. Dheri

Babu S. Brar

Debankur Sanyal

Department of Soil Science, Punjab Agricultural University, Ludhiana, India

Abstract

Nitrous oxide (N_2O) not only is an important greenhouse gas but can also contribute to the depletion of the stratospheric ozone layer. Agricultural sectors are emitting a large quantity of N_2O as nitrogen fertilizers are widely applied. However, a few studies have been conducted on N_2O emissions from salt-affected soils compared to not salt-affected soils. Possible reasons of variation in N_2O emissions are mainly due to differences in soil microbial activities caused by the presence of salt or manuring with certain organic amendments. More studies are needed to elucidate mechanisms of N_2O emissions from salt-affected soils.

INTRODUCTION

Nitrous oxide (N_2O) is the third most potent greenhouse gas (GHG), contributing 6% share (0.17 W m^{-2}) of total radiative forcing caused by well-mixed GHGs.^[1] It is also the most significant anthropogenic ozone-depleting compound.^[2] The mean global mole fractions of N_2O in 2013 reached 325.1 ± 0.1 ppb with an annual increase of 0.80 ppb for the decade ending in 2013.

Agricultural soils are the single largest contributor of the global N_2O emissions.^[3] In agricultural soils, N_2O emissions mainly originate from mineral nitrogen (N) applied with fertilizers and manures, mineralization of soil organic N including crop residues, and biologically fixed N.^[4] There is strong need to identify sources and sinks of N_2O .

Salt-affected soils contain excessive concentrations of soluble salts, exchangeable sodium, or both.^[5] Nearly 6–10% of earth's land area and 77 million hectares of irrigated land area are salt-affected.^[6] Saline and sodic water are used by farmers due to lack of better quality irrigation water in the arid and semiarid regions of the world. Continuous irrigation of soil with saline and saline-sodic water deteriorates soil physicochemical properties and adversely affects yields of different crops.^[7]

Emissions of N_2O from salt-affected soils are a well-reported phenomenon but not clearly well explained or documented. In this entry, we discuss the mechanisms of N_2O production in salt-affected soils and how it differs from the mechanisms in not salt-affected soils (i.e., normal soils).

SOIL EMISSIONS OF N_2O

Microbial processes of nitrification and denitrification are responsible for the production of N_2O in soils.

The underlying mechanism behind the production of N_2O is the depletion of oxygen (O_2), subsequent anaerobic conditions, and reduction of oxidized forms of N to their reduced forms as nitrate (NO_3^-) in denitrification process. NO_3^- is utilized as terminal electron acceptor by facultative and obligate anaerobic bacteria.^[8]

The principal regulatory mechanism of denitrification is fluctuation of partial pressure of molecular oxygen.^[9] The concentration of NO_3^- (or NO_2^-) in the soil is another important regulator, and the third important factor is the amount of available organic carbon (C).^[10] Soil water content is the main factor that controls O_2 diffusion in soil and, thus, nitrification and denitrification. Small changes in soil water content can have a significant effect on N_2O emission.^[11] Soils having high clay content are more prone to denitrification as those easily get waterlogged, causing anaerobic conditions.^[12]

HOW MECHANISMS OF N_2O PRODUCTION IN SALT-AFFECTED SOILS DIFFER FROM THOSE IN NOT SALT-AFFECTED SOILS

Nitrification and denitrification processes coexist within the same soil aggregate due to microsite variability.^[13] N_2O produced in soil is a by-product of the oxidation of NH_4^+ to NO_2^- and NO_2^- to NO_3^- during nitrification under aerobic conditions. Production of N_2O is also possible in denitrification during the reduction of NO_3^- to N_2O under anaerobic conditions or the oxidation of NH_4^+ to NO_2^- and NO_2^- to NO_3^- with a subsequent reduction to N_2O (nitrifier denitrification).^[14] During nitrification, N_2O is produced by two pathways: nitrifier denitrification, i.e., $\text{NH}_3 \rightarrow \text{NH}_2\text{OH} \rightarrow \text{NO}_2^- \rightarrow \text{N}_2\text{O} \rightarrow \text{N}_2$, and hydroxylamine oxidation, i.e., $\text{NH}_3 \rightarrow \text{NH}_2\text{OH} \rightarrow \text{NO} \rightarrow \text{N}_2\text{O}$.^[14] Thus,

under optimum moisture conditions because of high-rate NO_3^- transformation, production of N_2O remains high until all NO_3^- and NO_2^- are eliminated.

Soil salinity slows down N mineralization and accumulation of NO_3^- , thereby influencing the enzymatic activities responsible for denitrification and N_2O emissions.^[15] The high $\text{N}_2:\text{N}_2\text{O}$ ratios in moderately saline conditions having electrical conductivity (EC) of 12.5 dS m^{-1} and low $\text{N}_2:\text{N}_2\text{O}$ ratios in strongly saline conditions (EC of 56 dS m^{-1}) suggest that under highly saline soils a decline in N_2O reductase enzyme activity enhances N_2O emissions.^[16] The potential of denitrification rate in the saline wetland systems is primarily regulated by organic C, NO_3^- and activity of denitrifying bacteria, exchangeable Na^+ percentage (ESP), and the denitrification rate decreases with a rise in the ESP. Microorganisms in artificially created salt-affected soils, however, have limited time of adjustment to the adverse conditions, and their response to salts might be different from those of microorganisms living for an extended time in soils with high amounts of salts.^[16] The activity of NO_3^- reductase produced by ammonia-oxidizing bacteria (AOB) under anaerobic or limited O_2 conditions during nitrifier denitrification can denitrify NO_2^- to nitric oxide and subsequently to N_2O or N_2 . For example, the AOB *Nitrosomonas europaea* reduces NO_2^- produced during nitrification to N_2O using hydrogen or organic compounds as electron donors; thus, incomplete nitrification results in the release of N_2O .^[17]

The exhaustion of NO_3^- can be in excess to its reduction to NO_2^- -N, N_2O -N plus N_2 under anaerobic conditions. The inhibition of N_2O reductase and/or de novo synthesis may be the possible reason for higher N_2O -N emissions and N_2O -to- N_2 ratio in Texcoco-B (EC = 56 dS m^{-1}) as compared to that in Texcoco-A (EC = 12.6 dS m^{-1}).^[16] Increase in EC from 0.5 to 2.0 dS m^{-1} increased production of N_2O from $3.37 \text{ mg N}_2\text{O-N m}^{-2}$ to $34.30 \text{ mg N}_2\text{O-N m}^{-2}$ that may be due to variation in adaptation of microbial communities to high salt concentrations and ion toxicities.^[18] Another study^[19] discovered that N_2O emissions by *Bacillus subtilis* sp. 2 proceed until the content of sodium chloride (NaCl) in analytically pure culture is lower than or equal to 7%, whereas N_2O consumption occurs only when the content of NaCl is lower than 3%. Such differences in N_2O emission and consumption indicate the higher sensitivity of N_2O reductase to salts than NO_3^- reductase. The nature of salts has profound effects on the production of N_2O in soils incubated under anaerobic conditions. The cumulative production of N_2O in soil treated with different salts-enriched water was in the order of normal water < calcium chloride water < NaCl water (Fig. 1) that indicates that nature of salts influences the sensitivity of reductase and hence production of N_2O .^[20] The N_2O reductase is inhibited by acidic pH and low O_2 concentration, indicating that large salt concentrations will act in the same way.^[16]

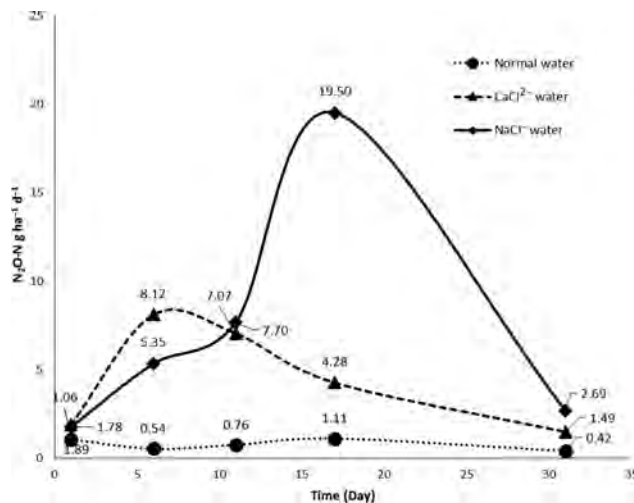


Fig. 1 Production of N_2O ($\text{g ha}^{-1} \text{ d}^{-1}$) in soil treated with CaCl_2 - and NaCl -enriched water during a 31-day incubation period under anaerobic conditions.

Source: From Dheri, Lal, et al.^[20] ©2016.

Ding^[21] observed effects of organic amendments such as manure, composts, and plant residues on the rates of N loss through denitrification from soils compared to unamended or mineral N-treated soils. The deleterious effect of high salts concentration on crops can be minimized by incorporating organic material^[22] and by applying gypsum, farmyard manure (FYM), or both under sodic and only FYM under saline-sodic water irrigation.^[23,24] The most probable cause of enhanced N_2O production in manured soil is the increased organic C availability. However, application of organic amendments and soil salinization can increase N_2O emissions.^[25]

CONCLUSION

Emission of N_2O is enhanced from some salt-affected soils, but there are no consistent reports to this effect. Thus, well-designed studies are needed to understand the underlying mechanism(s) of N_2O emission from salt-affected soils. Applications of some commonly used organic or inorganic amendments can enhance. Thus, choice of appropriate soil amendments is also essential to reducing N_2O emissions.

REFERENCES

1. Elkins, J.W.; Dutton, G.S. Nitrous oxide and sulfur hexafluoride. In *State of the Climate in 2009*; Arndt, D.S., Baringer, M.O., Johnson, M.R., Eds.; American Meteorological Society: Boston, MA, 2010; 44–45.
2. Ravishankara, A.R.; Daniel, J.S.; Portmann, R.W. Nitrous oxide (N_2O): The dominant ozone-depleting substance

- emitted in the 21st century. *Science* **2009**, *326* (5949), 123–125.
3. Smith, P.; Bustamante, M.; Ahammad, H.; Clark, H.; Dong, H.; Elsiddig, E.A.; Haberl, H.; Harper, R.; House, J.; Jafari, M.; Masera, M.; Mbow, C.; Ravindranath, N.H.; Rice, C.W.; Robledo Abad, C.; Romanovskaya, A.; Sperling, F.; Tubiello, F. Agriculture, forestry and other land use (AFOLU). In *Climate Change Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*; Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner, S., Seyboth, K., Adler, A., Baum, I., Brunner, S., Eickemeier, P., Kriemann, K., Savolainen, J., Schlömer, S., von Stechow, C., Zwickel, T., Minx, J.C., Eds.; Cambridge University Press: Cambridge and New York, 2014; 811–922.
 4. Ussiri, D.; Lal, R. *Soil Emission of Nitrous Oxide and Its Mitigation*; Springer: Dordrecht and New York, 2013.
 5. Richards, L.A. Diagnosis and improvement of saline and alkali soils. *Soil Sci.* **1954**, *78* (2), 154.
 6. Eynard, A.; Lal, R.; Wiebe, K. Crop response in salt-affected soils. *J. Sustain. Agr.* **2005**, *27* (1), 5–50.
 7. Bajwa, M.S.; Josan, A.S.; Choudhary, O.P. Effect of frequency of sodic and saline-sodic irrigations and gypsum on the buildup of sodium in soil and crop yields. *Irrigation Sci.* **1993**, *1*, 14.
 8. Rogers, C.W.; Brye, K.R.; Roberts, T.L.; Norman, R.J.; Fulford, A.M. Assessing redox potentials as related to greenhouse gases in flooded paddy soils. In *B.R. Wells Rice Research Studies: 2010*; AAES Research Series 591; Norman, R.J., Moldenhauer, K.A.K., Eds.; University of Arkansas Division of Agriculture, Fayetteville, 2010; 235–243.
 9. Yu, H.; Song, Y.; Xi, B.; Dub, E.; He, X.; Tu, X. Denitrification potential and its correlation to physico-chemical and biological characteristics of saline wetland soils in semi-arid regions. *Chemosphere* **2012**, *89*, 1339–1346.
 10. Tiedje, J.M. Ecology of denitrification and dissimilatory nitrate reduction to ammonium. In *Biology of Anaerobic Microorganisms*; Zehnder, A.J.B., Ed.; Wiley: New York, 1988; 179–244.
 11. Grant, R.F.; Pattey, E. Modeling variability in N₂O emissions from fertilized agricultural fields. *Soil Biol. Biochem.* **2003**, *35*, 225–243.
 12. Rochester, I.J. Estimating nitrous oxide emissions from flood-irrigated alkaline grey clays. *Aust. J. Soil Res.* **2003**, *41*, 197–206.
 13. Kuenen, J.G.; Robertson, L.A. Combination nitrification–denitrification processes. *FEMS Microbiol. Rev.* **1994**, *15*, 109–117.
 14. Wrage, N.; Velthof, G.L.; van Beusichem, M.L.; Oenema, O. Role of nitrifier denitrification in the production of nitrous oxide. *Soil Biol. Biochem.* **2001**, *33*, 1723–1732.
 15. Delwiche, C.C. Production and utilisation of nitrous oxide by *Pseudomonas denitrificans*. *J. Bacteriol.* **1959**, *24*, 55–59.
 16. Ruiz-Romero, E.; Alcántara-Hernández, R.; Cruz-Mondragon, C.; Marsch, R.; Luna-Guido, M.L.; Dendooven, L. Denitrification in extreme alkaline saline soils of the former lake Texcoco. *Plant Soil.* **2009**, *319*, 247–257.
 17. Thangarajan, R.; Bolan, S.N.; Tian, G.; Naidu, R.; Kunhikrishnan, A. Role of organic amendment application on greenhouse gas emission from soil. *Sci. Total Environ.* **2013**, *465*, 72–96.
 18. Adviento-Borbe, M.A.A.; Doran, J.W.; Drijber, R.A.; Dobermann, A. Soil electrical conductivity and water content affect nitrous oxide and carbon dioxide emissions in intensively managed soils. *J. Environ. Qual.* **2006**, *35*, 1999–2010.
 19. Menyailo, O.V.; Stepanov, A.L.; Umarov, M.M. The transformation of nitrous oxide by denitrifying bacteria in Solonchaks. *Eur. Soil Sci.* **1997**, *30* (2), 178–180.
 20. Dheri, G.S.; Lal, R.; Jha, P.; Nakajima, T. Effect of CaCl₂ and NaCl on N₂O production in soils amended with compost and gypsum. 2016 (Unpublished).
 21. Ding, W. Effects of long-term amendment of organic manure and nitrogen fertilizer on nitrous oxide emission in a sandy loam soil. *J. Environ. Sci.* **2007**, *19*, 185–193.
 22. Liang, Y.; Nikolic, M.; Peng, Y.; Chen, W.; Jiang, Y. Organic manure stimulates biological activity and barley growth in soil subject to secondary salinization. *Soil Biol. Biochem.* **2005**, *37*, 1185–1195.
 23. Choudhary, O.P.; Josan, A.S.; Bajwa, M.S.; Kapur, M.L. Effect of sustained sodic and saline-sodic irrigation and application of gypsum and farmyard manure on yield and quality of sugarcane under semi-arid conditions. *Field Crop. Res.* **2004**, *87* (2), 103–116.
 24. Brar, B.S. *Sodium Hazard of Bicarbonate Irrigation Waters as Affected by Leaching Fraction and Amendment Application*; PhD Dissertation, Punjab Agricultural University: India, 1986.
 25. Reddy, N.; Crohn, D.M. Effects of soil salinity and carbon availability from organic amendments on nitrous oxide emissions. *Geoderma* **2014**, *235*, 363–371.

Satellite Mapping

Bruce E. Frazier

Department of Crop and Soil Sciences, Washington State University, Pullman, Washington, U.S.A.

Abstract

Land-observing satellites have provided data about soil that were beneficial to the making of soil maps. In some cases, these data gave information about soil properties, like color, which was used to infer erosion or drainage status. In other cases, the evidence was about landform boundaries, vegetative indicators, or less well-understood spectral patterns expressed in terms of principal components that can be related to soil patterns. These data proved useful to map levels of soil organic carbon, iron oxide, and moisture relationships in plants. Satellites provide flexibility in dates of imagery, wavelength, resolution, and synoptic details, allowing much focused views of land areas to be surveyed.

INTRODUCTION

Soil is that “ecstatic skin of the earth,”^[1] containing much life, which is formed as a 3-D skin over the face of earth. The whole soil is seen only by probing, digging, or otherwise cutting into it to expose its 3rd dimension. For over a century, we have organized the soil’s properties according to classification schemes and mapped its geographic distribution. Only for the last years has there been an opportunity to view the surface of the soil from a satellite. From such a perspective, we must infer the properties that soil has in 3-D by convergence of several forms of evidence. Direct evidence of soil properties is derived by their spectral nature and patterns of distribution. Indirect evidence derives from geomorphology, differences in types of vegetation growing on the soils, differences in vegetation caused by soil properties acting as a source of stress to the plants, and land cover created by man in accordance with soil properties. The indirect evidence is related to three of the state factors of soil formation,^[2] namely: organisms, topography, and parent material.

For collecting direct evidence of soil properties with an optical space-borne sensor, the soil should be bare. The exception may be microwave sensing (not optical), which has shown some success in mapping soil moisture. Low-frequency microwaves partially penetrate vegetation and the soil surface, in addition to clouds, and can be used without light.^[3] Timing is critical for optical sensing except in those deserts where the soil is always bare. Some agricultural soils are exposed to a view from above at planting time, but with the adoption of cropping systems built around no-till and minimum till practices, many agricultural soils are no longer bare at any time. As a result, indirect evidence of soil properties is crucial for mapping by satellite or any other optical airborne sensor. This entry

first describe the available sensors, then research into direct evidence of soil properties, and finally the indirect evidence and applications as described in the literature and then the author’s experience.

AVAILABLE SENSORS

The number of sensors, wavelengths, resolution, and satellites are wide as we enter the new millennium. Private satellites are available that can provide a range of resolutions in the optical wavelengths from high (1 m) to low (1 km) and a range of wavelengths from visible (400–700 nm) and reflective infrared (700–2500 nm) to thermal infrared (8000–14,000 nm). Panchromatic (black and white) imagery that is similar to aerial photography is available from IKONOS,^[4] or QuickBird,^[5] if necessary. Acquisition of images can be timed to capture the ground scene when soils are bare or when soil response patterns are detectable in vegetation. Of course, no one has control over cloud cover; so for persistently cloudy areas, any clear image may dictate the choice of date, and all other factors become secondary.

A wealth of choices of satellite data inevitably leads to the realization that if one is planning to use satellite data to map soils, needs of the survey must be well defined. Are high-resolution data necessary? One-meter panchromatic data may be useful where highly detailed surveys are conducted, but a synoptic view of a wide region will not be available. Would normal color or color infrared imagery show patterns of vegetation that relate to soil properties? These data are available in resolutions from 5 m to 1 km, providing some choices in detail and synoptic views. Is information about water absorption patterns of the landscape important? These data are available from the 30-m

Landsat thematic mapper (TM) sensor that has two bands in the middle infrared region (1550–1750 and 2080–2350 nm). The future will witness more satellites launched by both government and private entities that will provide more choices of wavelength and resolution as dictated by economic feasibility.

DIRECT EVIDENCE

Direct evidence of soil properties in satellite data has been investigated and reported by several authors. Soil organic carbon (SOC), iron oxide, and water are constituents of soil that absorb energy. As long as the soil is bare and dry, information about constituents that appear at the surface can be collected by the satellite. This technique is especially adaptable to areas of the world that employ periods of bare fallow in multiyear cropping systems. Bowers and Hanks,^[6] Condit,^[7] Stoner et al.,^[8] and Baumgardner et al.^[9] provide examples of basic research on energy and soil property interactions that were necessary to understand the wavelengths that are important for satellite sensors and then to understand how to apply the satellite-acquired data. Taking into consideration these basic energy absorption features, Weismiller et al.^[10,11] were able to incorporate detailed soil patterns derived from Landsat multispectral scanner (MSS) data into the survey of Jasper County, Indiana. Spectral map overlays were used in the field with aerial photographs to help locate the hand-drawn boundaries. Spectral patterns were correlated with soil drainage classes. A few years later, Frazier and Cheng^[12] investigated Landsat TM bands and found that ratios of several of the bands could be used to reliably map soil patterns in eastern Washington State. They used bands centered at 483 (TM1), 60 (TM3), 825 (TM4), and 1650 nm (TM5). The ratios TM 1:4, 3:4, 5:4, and 5:3 scaled to 8-bit range formed images that were related to SOC levels and to eroded soils. TM 5:4 pixel values were well correlated ($R^2 = 0.98$) to SOC (g/kg) in samples gathered from the test fields. Likewise, TM 5:3 pixel values were well correlated ($R^2 = 0.96$) with the ratio of amorphous iron to SOC from the field samples. This meant that eroded soils could easily be mapped because elevated amorphous iron/SOC occurred where B-horizons were exposed. Both of these sensors, MSS at 80 m resolution and TM at 30 m resolution, provided data of sufficient quality to benefit soil surveys published at 1:15,840 scale.

INDIRECT EVIDENCE

Indirect evidence of soil properties derived from processing satellite images has been added to actual soil surveys in numerous instances since the Landsat series of satellites was first launched in 1972. Usually, Landsat provides a background image to be used in a reconnaissance survey in an area where little ancillary data are available. Most examples described in the literature were spectral maps

made from individual MSS bands or combinations of bands that covered the visible and near infrared spectra. Soil landscapes observed on images made from Landsat digital data were found to exhibit a characteristic surface geometry, type and density of vegetation, and hydrologic pattern.^[13] Frequent satellite overpasses were seen as an advantage over conventional aerial photography because comparison of multirate images would show soil-specific vegetation changes. Multirate capability also allowed surveyors to acquire low sun elevation images showing detailed hydrologic patterns.^[13]

Roudabush et al.,^[14] in surveys on Arizona rangeland, processed two dates of Landsat imagery. The resulting image consisted of clusters representing soil, landform, or geomorphic features. The clusters were related to soil map unit data within broad landform/parent material units. Upon completion of the survey it was concluded that Landsat spectral data added to the mapping of soils on about 35% of the mapping units.

In Washington State, a general soil map was produced with the intent of illustrating the wide diversity of soils in the state. In Washington, 10 of the 12 soil orders that occur worldwide have been identified.^[15] Normally, a general soil map is made by condensing the detailed map units to achieve an appropriate level of abstraction. In some cases, as in the state of Washington, large blocks of land devoted to wilderness and parks had never been surveyed, and thus, had no prior map units to generalize from. For these areas, a schematic soil map was made using corroborating evidence from geology, climate, landforms, and vegetation. Patterns observed in Landsat MSS data served to define prominent geomorphic boundaries and vegetative indicators that could be incorporated into logical map units.^[16] The contribution of satellite data to the mapping process is shown in Fig. 1 of southeastern Washington, showing glacial outburst flood channels cutting through the Loess-covered Columbia Plateau.^[17] Soils within the channels are formed in less than a meter of Loess over flood gravels and bedrock. Soils outside the channels are formed in deep Loess. Flood channel boundaries are seen because of plant cover differences between the two landforms. Deep Loess soils are cropped to small grains, while shallow soils of the channels primarily support range grasses. The image underlying the general soil map polygons is a mid-February Landsat TM band 5 scene that captures the bright return of middle infrared energy from the dry range grasses in contrast with the dark return from green winter grain crops and bare, moist soils. A few apparent mismatches between the satellite image patterns and soil boundary line locations are evident upon close examination. These occur where irrigation has made cropping within the channels possible, where light toned residue from the fall grain crop is still on the field, and where mapping of channel width was exaggerated to preserve their continuity at a publication scale of 1/750,000.

Wilson^[18] developed and implemented premapping techniques using rectified Landsat 7 multispectral imagery

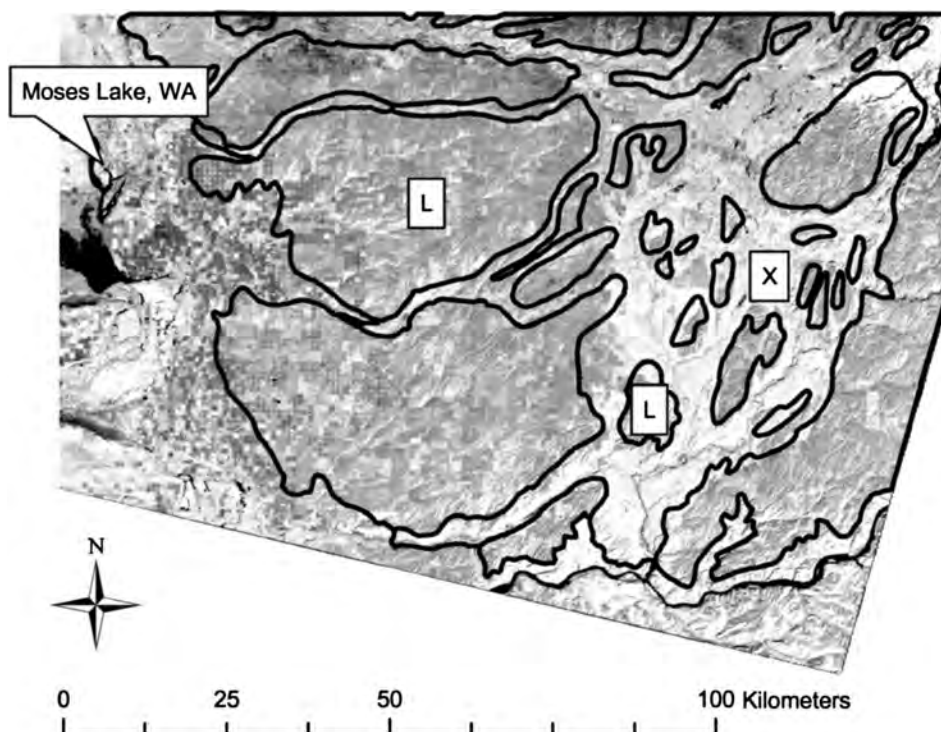


Fig. 1 Selected polygons from the Washington State general soil map superimposed on Landsat 7 TM band 5 (1550–1750 nm) scene, February 16, 2000. Light-toned X units are shallow soils of the channeled scablands. Dark-toned L units are deep soils of the Loess uplands.

at 30 m resolution for an area of previously unmapped desert in Arizona. He found color band combinations of (5, 4, 2) (5,4, 1) and band ratios of (5/7) (5/4) and (3/1) provided patterns that could be related to the geomorphic units.

What is common to all these examples of satellite contributions to soil mapping is that the data provided information to delineate soil boundaries by traditional standard map production procedures, i.e., lines drawn on photographic base, USGS quadrangle sheets, or mylar overlays. There are also examples of soil modeling,^[19] wherein satellite data are one of the several mapped databases that are used by a model to predict the occurrence of certain soils in wilderness areas. The entire map, a result of a GIS modeling process, is a computer-drawn product based on elevation data, geology maps, vegetation (from Landsat TM data), and climate data or models. Soil map units were classified at the subgroup level of soil taxonomy^[20] and designed to be consistent with requirements of a fourth-order survey.^[21] Landsat TM data were used to map vegetation classes, some of which include indicator species for certain soil properties, and to separate thin soils from bare rock and snow that occur in high elevation alpine areas. Verification was done by transecting the wilderness via existing trails and sampling soils within predicted soil groupings and landscape positions.

CONCLUSION

During the past years, land-observing satellites have provided data about soil that were beneficial to the making of

soil maps. In some cases, these data gave information about soil properties, like color, that was used to infer erosion or drainage status. In other cases, the evidence was about landform boundaries, vegetative indicators, or less well-understood spectral patterns expressed in terms of principal components that can be related to soil patterns. The launch of Landsat IV with the TM instrument brought the first wide-area access to the middle infrared bands. These data proved useful to map the levels of SOC, iron oxide, and moisture relationships in plants. Satellites provide flexibility in dates of imagery, wavelength, resolution, and synoptic details, allowing very focused views of land areas to be surveyed.

REFERENCES

1. Logan, W.B. Ecstatic skin of the earth. *Conserv. Voices* **1999**, 2 (2), 14–17.
2. Jenny, H. The soil resource: Origin and behavior. In *Ecological Studies*; Springer: New York, 1980; Vol. 37, 366–367.
3. Van de Griend, A.A.; Owe, M. The influence of polarization on canopy transmission properties at 6.6 GHz and implications for large scale soil moisture monitoring in semi-arid environments. *IEEE Trans. Geosci. Remote Sens.* **1994**, 32 (2), 409–415.
4. Space Imaging, <http://www.spaceimaging.com/level1/index23.htm>.
5. Digital Globe, <http://www.digitalglobe.com>.
6. Bowers, S.S.; Hanks, R.J. Reflection of radiant energy from soils. *Soil Sci.* **1965**, 100, 130–138.

7. Condit, H.R. The spectral reflectance of American soils. *Photogram. Eng.* **1970**, *36*, 955–966.
8. Stoner, E.R.; Baumgardner, M.F.; Biehl, L.L.; Robinson, B.F. *Atlas of Soil Reflectance Properties*; Research Bulletin 962; Agric. Exp. Sta, Purdue University: West Lafayette, 1980; 1–75.
9. Baumgardner, M.F.; Silva, L.F.; Biehl, L.L.; Stoner, E.R. Reflectance properties of soils. *Adv. Agron.* **1985**, *38*, 1–43.
10. Weismiller, R.A.; Van Scoyoc, G.E.; Pazar, S.E.; Latz, K.; Baumgardner, M.F. Use of soil spectral properties for monitoring soil erosion. In *Soil Erosion and Conservation*; El-Swaify, S.A., Moldenauer, W.C., Lo, A., Eds.; Soil Cons. Soc. Am.: Ankeny, 1985; 119–127.
11. Weismiller, R.A.; Kirschner, F.R.; Kaminsky, S.A.; Hinz, E.J. *Spectral Classification of Soil Characteristics to Aid the Soil Survey of Jasper County, Indiana*; LARS Tech. Rep. 040179; Laboratory for Application of Remote Sensing, Purdue University: West Lafayette, 1979; 342 pp.
12. Frazier, B.E.; Cheng, Y. Remote sensing of soils in the eastern Palouse region with Landsat thematic mapper. *Remote Sens. Environ.* **1989**, *28*, 317–325.
13. Westin, F.C.; Frazee, C.J. Landsat data, its use in a soil survey program. *Soil Sci. Soc. Am. J.* **1976**, *40*, 81–89.
14. Roudabush, R.D.; Herriman, R.C.; Barmore, R.L.; Schellentrager, G.W. Use of Landsat multispectral scanning data for soil surveys on Arizona rangeland. *J. Soil Water Conserv.* **1985**, *40* (2), 242–245.
15. Boling, M.S.; Frazier, B.E.; Busacca, A.J. *General Soil Map of Washington*; Dept. Crop and Soil Sciences, Washington State University: Pullman, 1998.
16. Boling, M.S. *General Soil Map of Washington State*; Unpublished MS Thesis, Department of Agronomy and Soils, Washington State University: Pullman, WA.
17. Baker, V.R.; Nummedal, D., Eds. *The Channeled Scabland: A Guide to the Geomorphology of the Columbia Basin, Washington*; NASA: Washington, 1978; 1–186.
18. Wilson, R. Soil survey operations using satellite imagery. *NCSS Newsletter* **2005**, (31), 3–6.
19. Rodgers, T.M. *Modeling Soils of the Sawtooth and Pasayten Wilderness Areas with a Geographic Information System (GIS)*. Unpublished MS Thesis, Department of Crop and Soil Sciences, Washington State University: Pullman, WA.
20. Soil Survey Staff. *Keys to Soil Taxonomy*, 8th Ed.; USDA, NRCS, USGPO: Washington, 1998.
21. Soil Survey Staff. *Soil Survey Manual*, USDA Handbook No. 18; U.S. Government Printing Office: Washington, 1993.

Scaling Effects

Yakov A. Pachepsky

Walter J. Rawls

Hydrology Lab, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Beltsville, Maryland, U.S.A.

John W. Crawford

University of Abertay, Dundee, Scotland

Abstract

Soil volume within a given measurement scale can be thought of as composed from soil volumes corresponding to smaller measurement scales. Averaging volumetric soil moisture content across the smaller volumes results in a correct value for volumetric moisture content for the larger scale. Similar averaging soil hydraulic properties, such as moisture retention and hydraulic conductivity, across small scales may not result in the correct values for the larger scale. The reason is that the larger scale represents levels of heterogeneity in soil structure that is absent in small-scale volumes. For the same reason, the spatial variability of soil moisture and soil hydraulic properties appears to be dependent on scale. Scaling as a noun means a relationship between soil moisture data at different scales, whereas scaling as a gerund means relating data at different scales. Upscaling (downscaling) refers to estimating soil moisture or soil hydraulic properties at a scale that is coarser (finer) than the one at which data are available.

INTRODUCTION

Soil moisture and soil hydraulic properties are measured over temporal and spatial measurement scales that are often different from the scales at which knowledge is required. For example, the measurement scale of the moisture release characteristic is equal to the volume of the core, whereas soil surface moisture measured from a satellite has the measurement scale equal to the pixel size of the imagery. Soil water balance calculations are based on subdividing soil into cells, the size of which defines the model scale. Both measurement and model scales define supports, i.e., minimum lengths, areas, volumes, or time intervals below which variation in properties is ignored.

PHYSICS-BASED SCALING

Models of soil structure used for scaling assume the preservation of some structure-related properties across a range of measurement scales or among samples of the same measurement scale.

Similitude-Based Scaling

The similitude model postulates that pore-size distributions in several soil samples of the same size can be converted into each other by simple multiplication of radii by a

sample-specific constant or by adding a sample-specific constant to the logarithms of pore radii as shown in Fig. 1. The Washburn's equation is often used to replace values of pore radii R with values of the maximum matric potential Ψ at which a pore is filled during the drainage, h is the absolute value of matric potential Ψ , and a is the capillary constant

$$h = |\Psi| = \frac{a}{R} \quad (1)$$

Using Eq. 1, one can transform similarity in pore-size distributions into similarity in dependencies of h on saturation degree S as shown in Fig. 1. Therefore, water retention curves of samples exhibiting the similitude should coincide in a “lg h vs. S ” coordinate system if appropriately moved along the axis lg h . Let the expression $h = f_i(S)$ be water retention curves in n samples, $i = 1, 2, 3, \dots, n$. The coincidence of these curves following the shifts will be achieved if, for the same values of h ,

$$\begin{aligned} \lg \lambda_1 + \lg f_1(S) &= \lg \lambda_2 + \lg f_2(S) = \dots \\ &= \lg \lambda_n + \lg f_n(S) = \lg f^*(S) \end{aligned} \quad (2)$$

or

$$\lambda_1 f_1(S) = \lambda_2 f_2(S) = \dots = \lambda_n f_n(S) = f^*(S) \quad (3)$$

where λ_i is the sample-specific scaling constant for the sample “ i ,” $f^*(S)$ is the reference water retention curve, lg

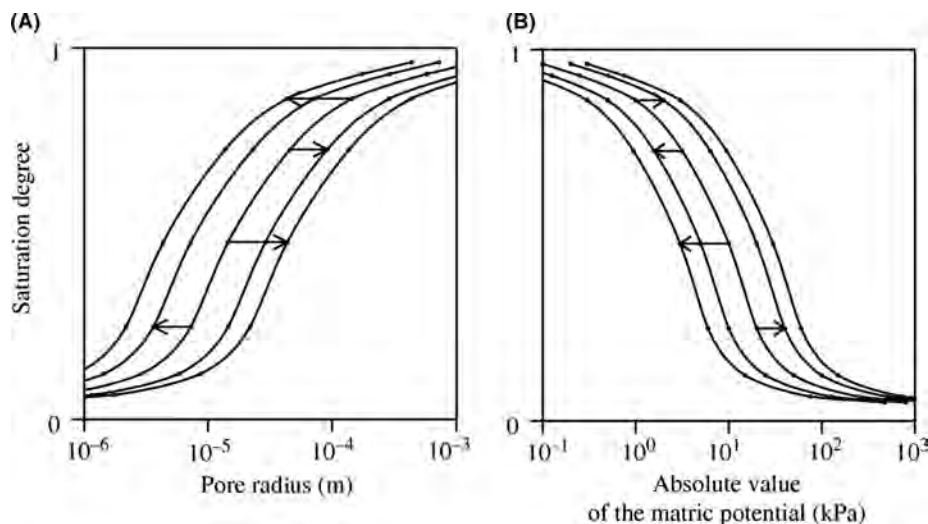


Fig. 1 Similarity of pore-size distributions under the similitude hypothesis; (A) is the distribution of pore volume expressed as saturated degree, and (B) is pore radii transformed into absolute values of matric potentials. Arrows show shifts of the middle reference curve to transform it into other curves.

λ_i is the shift along the axis $\lg h$ needed for the coalescence of the i^{th} curve with the reference curve. The scaling allows the compression of variations in functions $f_i(S)$ into variations in single number λ_i .^[1] The application of this scaling is shown in Fig. 2. Although no ideal similitude is

observed, a significant decrease in variability is achieved by applying scaling, i.e., shifting original water retention curves as shown in Fig. 2. No large different groups of water retention curves are present, and the distribution of values of λ_i is close to the lognormal distribution (Fig. 2).

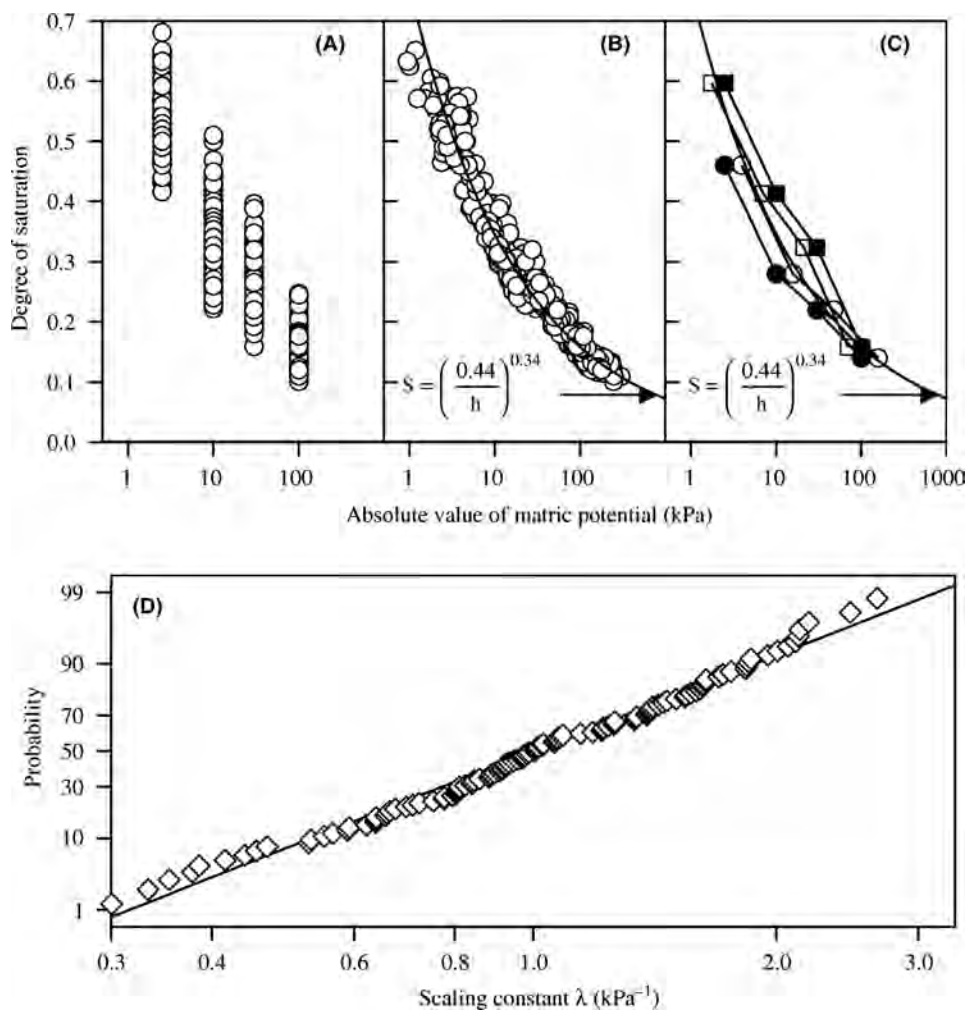


Fig. 2 Testing the similitude hypothesis with 120 of sandy siliceous, mesic Psammentic Hapludult across $170 \times 210 \text{ m}^2$ area; (A) is the variability of original water retention data; (B) is water retention curves shifted to be close to the reference water retention curves shown as a solid line with an equation; (C) is two examples of shifts of water retention curves; original and shifted curves are shown with filled and hollow symbols, respectively; and (D) is the probability distribution of scaling constant; symbols—experimental data, and line is the theoretical lognormal distribution.

The similitude assumption can be combined with the hypothesis that the Kozeny–Carman law for the hydraulic conductivity K is applicable in soil. Then a coalescence of dependencies $K(h)$ can be achieved.

Scale Invariance of Water Transport Equations

Soil water transport equations can be transformed into dimensionless forms that relate the dimensionless combination of the original dependent and independent variables. Dimensional or inspectional analysis is used for that purpose.^[2] Scaling invariance of the Richards' water transport equation requires, in general, water retention and hydraulic conductivity to be log-linear functions of volumetric water content as follows

$$\lg h = \lg h_0 - b_h \lg \theta \quad (4)$$

$$\lg K = \lg K_s + b_k \lg \theta \quad (5)$$

where the saturated hydraulic conductivity K_s and the air-entry pressure h_0 can be spatially variable, whereas slopes b_k and b_h are the same across the spatial extent of the sampling. Where this scaling invariance applies, the volumetric water content at the depth, z , at the time, t , can be obtained from the volumetric water content, θ' , at another depth z' and at time t' as follows

$$\frac{\theta}{\theta'} = \left(\frac{z}{z'}\right)^{-1/b_h}; \quad \frac{t}{t'} = \left(\frac{z}{z'}\right)^{(b_k+b_h-1)/b_h} \quad (6)$$

Simplified models of water transport, such as the Green–Ampt infiltration model, the unit gradient drainage model, or the model with a negligible gravity, also yield scaling relationships that compress spatiotemporal variations in

water content profiles into spatial variability of one or two dimensionless parameters. To use the same hydraulic properties at different scales, it was suggested to incorporate the scaling directly into water transport equations by using fractional derivatives.

Fractal Scaling Using Self-Similarity, Self-Affinity, and Multiscaling Hypotheses

Self-similarity models are based on the assumption of fractal geometry. Natural objects often have similar features when viewed at different scales. Measures of these features, e.g., the total number of features, the total lengths, the average roughness, and the total surface area, are dependent on the measurement scale. Fractal geometry assumes that this dependence is the same over a range of scales, i.e., scale invariant within this range. Many soil properties were shown to obey fractal scaling.^[3]

To apply fractal geometry, one must have in mind a model of formation of the fractal features in soil. The Sierpinski carpet and the Menger sponge (Fig. 3) are examples of such models that are often used to simulate soil pore space structure. These and other fractal models are obtained by an iterative extraction of similar pieces of progressively small sizes from an initially solid geometrical figure, square in case of the Sierpinski carpet, and cube in case of the Menger sponge. Fractal objects can be also generated from aggregation processes.

Fractal modeling of pore volume based on the Sierpinski carpet and Menger sponge led to the widely used Brooks–Corey equation of soil water retention. The Sierpinski carpet model has been successfully used to develop equations for saturated hydraulic conductivity of macropores in soil.^[4] A fractal model of pore outlines

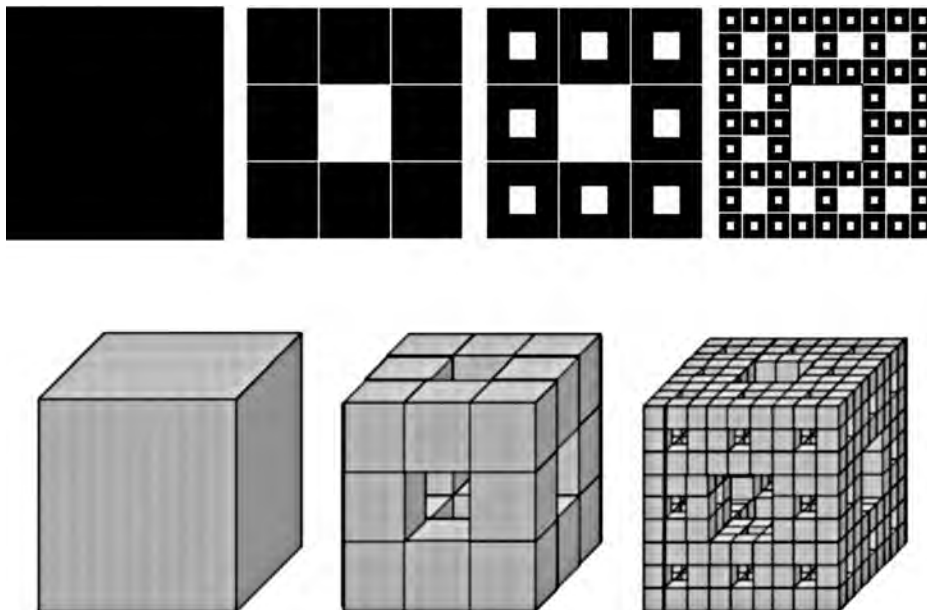


Fig. 3 The iterative construction of fractal objects; top is the first three stages of constructing fractal surface using the Sierpinski carpet model, and bottom is the first two stages of constructing fractal volume using the Menger sponge model.

has been useful in relating the unsaturated hydraulic conductivity k to soil water retention and the tortuosity caused by fractal surface of pores. Soils with water retention curves from fractal models satisfy the requirement 4 for scaling invariance of Richards' equation and also exhibit the similitude-based scaling 2 provided that the spatial variability of the fractal dimension is not substantial.^[5]

The self-affinity model appears to be appropriate to simulate the spatial variability of soil moisture for large resolutions and extents.^[6] A self-affine surface remains statistically similar to itself when the resolution in horizontal plane increases μ times and the resolution in vertical direction increases μ^H times. The variances of self-affine elevations, z_1 and z_2 , measured at two resolutions, d_1 and d_2 , obey the following scaling relationship: $\text{Variance}(z_1) = (d_1/d_2)^{2H} \text{Variance}(z_2)$. Assuming spatial dependencies of soil moisture contents to be self-affine, one can apply the self-affinity model as shown in Fig. 4.

It has been proposed that multiscaling, or multifractal, models expand the range of resolutions in which fractal scaling is applicable to soil moisture fields. Unlike the generation of Sierpinski carpet or Menger

sponge fractals in which the similarity of structure persists across iterative steps, multifractals are generated by cascade processes that create more heterogeneity as the resolution decreases.

EMPIRICAL SCALING

Physics-based scaling is usually applicable within a range of scales covering one or two orders of magnitude. At larger scales, soil heterogeneity manifests itself by the appearance of new types of heterogeneity, and self-similar scaling ceases to be applicable. A further increase in the scale may reveal new scaling. Empirical observations are usually substituted to proceed from one level of physics-based scaling to another.

Transition from Laboratory to Plot Scale

Transition from the laboratory soil sample scale of 5–10 cm to the plot scale of 1–5 m reveals a change in soil water retention. A regression equation has been developed to convert volumetric water content from the sample resolution to the plot resolution at the same soil matric potential $\theta_{\text{Plot}} = 1.287\theta_{\text{Lab}} - 0.977 \theta_{\text{Lab}}^2$. Saturated hydraulic conductivity measured in soil cores increases according to the power law $K_{\text{sat}} \propto R^m$, where R is the core radius, $m = 0.3 - 0.4$ ($0.6 - 0.7$) for coarse-textured (fine-textured) soils without macropores.

Using Pedotransfer Functions and Soil-Landscape Relationships

The use of pedotransfer functions, i.e., relationships between soil properties, is a common way to estimate effective soil hydraulic properties for large areas with data available from soil survey maps. Because the soil textural and other classes used in soil mapping are quite broad, nonlinear hydrologic models may give distorted answers if average class values are entered in pedotransfer functions. Soil-landscape relationships can be used to describe the variability within soil map units. In particular, the spatial variations of soil hydraulic properties within extents of 10–1000 m correlate with terrain attributes.^[7]

Inverse Modeling

Inverse modeling has been used to estimate effective soil hydraulic properties for large-area supports by matching measured and predicted soil surface moisture, runoff, or infiltration. Simulation studies show a good correspondence between weighted-average hydraulic parameters for individual soil units within the support and parameters found from the inverse modeling.

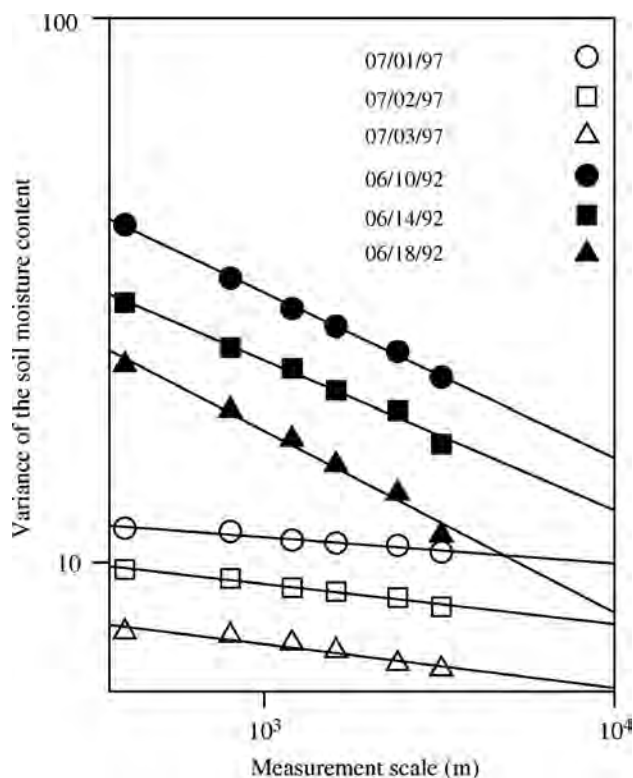


Fig. 4 Fractal scaling of the variance of soil surface moisture measured with passive microwave radiometry across Little Washita Watershed in Oklahoma in 1992 and 1997 during the drying periods. Observation dates are shown with different symbols, and measurement scale is the pixel size.

CONCLUSION

Scaling relationships both enhance the understanding of soil moisture dynamics and underlying processes and serve as a unique contrivance to overcome the mismatch of scales between measurements available in soil moisture studies, hydrological models, and the applications of soil moisture parameters and values in environmental decision making. One should expect scaling of soil moisture to play a major role in the fast-growing field of data assimilation, i.e., combining data obtained at different scales for environmental monitoring and prediction.

REFERENCES

1. Warrick, A.W.; Nielsen, D.R. Spatial variability of soil physical properties in the field. In *Applications of Soil Physics*; Hillel, D., Ed.; Academic Press: New York, 1980; 319–344.
2. Sposito, G. *Scale Dependence and Scale Invariance in Hydrology*; Cambridge University Press: Cambridge, 1998; 423 pp.
3. Pachepsky, Y.A.; Crawford, J.W.; Rawls, W.J. *Fractals in Soil Science*; Elsevier: Amsterdam, 2000; 295 pp.
4. Giménez, D.; Perfect, E.; Rawls, W.J.; Pachepsky, Ya.A. Fractal models for predicting soil hydraulic properties: A Review. *Eng. Geol.* **1997**, *48*, 161–183.
5. Hillel, D.; Elrick, E. *Scaling in Soil Physics, Principles and Applications*; SSSA Spec. Pub. 25; Soil Science Society of America: Madison, 1990; 122 pp.
6. Stewart, J.B.; Engman, E.T.; Feddes, R.A.; Kerr, Y. *Scaling up in Hydrology Using Remote Sensing*; John Wiley & Sons: New York, 1996; 255 pp.
7. Kabat, P.; Hutjes, R.W.A.; Feddes, R.A. The scaling characteristics of soil parameters: From plot scale heterogeneity to subgrid parameterization. *J. Hydrol.* **1997**, *190*, 363–396.

Sediment: Influence on Aquatic Life

Suzanne M. Gray

*School of Environment and Natural Resources, Ohio State University,
Columbus, Ohio, U.S.A.*

Abstract

Soil sediments enter aquatic systems through natural processes such as erosion, run-off, and resuspension and provide necessary components for aquatic ecosystem functioning. However, sedimentary inputs to aquatic systems have increased drastically in the 20th century due to human activities in and around water. This increased sediment load is expected to influence aquatic biota in multifaceted ways while in suspension and through sedimentation. Here, a broad overview of how aquatic life is influenced by increased sediment loads was given, highlighting the general trends of organismal and population-level responses to sediment levels that exceed natural or historical levels. Overall, increased sediment loads caused by human activities are associated with negative responses, such as reduced primary productivity due to the shading effects of suspended particles and a decrease in the depth of the photic zone, and population declines of macroinvertebrates and fish associated with a suite of physicochemical changes to aquatic systems and concomitant changes in the ecological structure of those systems. While extensive research has investigated the influence of increased sediment loads on aquatic biota, linking organismal-level responses to population- and community-level responses, and further to global scale impacts remains a challenge to the field.

INTRODUCTION

The movement of sediments into aquatic ecosystems is a natural process with many beneficial and essential results that contribute to the global sediment flux.^[1,2] However, global rates of sediment inputs to aquatic systems have increased drastically in the 20th century due to human activities:^[1,3,4] Syvitski et al.^[3] estimated an increased sediment load of approximately 2.3 billion metric tons per year to the earth's rivers, with much of that excess load being contained in freshwater systems due to impoundments that impede the natural flux of sediments to the oceans. What are the impacts of increased sediment load on aquatic biota and ecosystems? A number of excellent critical reviews have addressed this question, with foci ranging from terrestrial to ocean sediment flux,^[4,5] to responses of specific aquatic taxa to suspended sediments and sedimentation,^[2,6–9] and to community-level impacts.^[10,11] It is important to recognize that many organisms are adapted to relatively high sediment loads that vary through space and time; however, substantial increases over naturally experienced levels may push even those well-adapted species beyond their ability to cope with this stressor. We know from extensive research that all levels of biological organization are affected by increased sediment inputs to aquatic systems (i.e., individuals, populations, communities, and entire ecosystems) in all kinds of water bodies (e.g., streams, rivers,

lakes, reservoirs, estuaries, and coastal zones). Here, a broad overview of how aquatic organisms are affected by increased sediment loads is provided, as shown in Fig. 1.

CAUSES AND CONSEQUENCES OF INCREASED SEDIMENTARY INPUTS

The arrival, deposition, and transport of sediments in aquatic systems are natural and essential processes for ecosystem functioning.^[12,13] However, human activities have greatly increased sedimentary inputs throughout aquatic systems. Drivers of this increase include clearing of land for agriculture and poor agricultural land management, deforestation, mining, urban development, road and reservoir construction, and dredging.^[1–3,8,13,14] Such anthropogenic alterations of terrestrial and aquatic landscapes result in higher rates of soil erosion and mobilization of sediments into river catchments and ultimately to coastal marine waters. Overlaid on these human activities is global climate change,^[1] which contributes to more severe and unpredictable weather events that can further increase sediment movement to waterways. For example, in tropical areas, intense agriculture leads to higher erosion rates especially during post-drought severe rain events, changing the geomorphology of river and riparian landscapes, leading to even greater sediment loads being suddenly dumped into

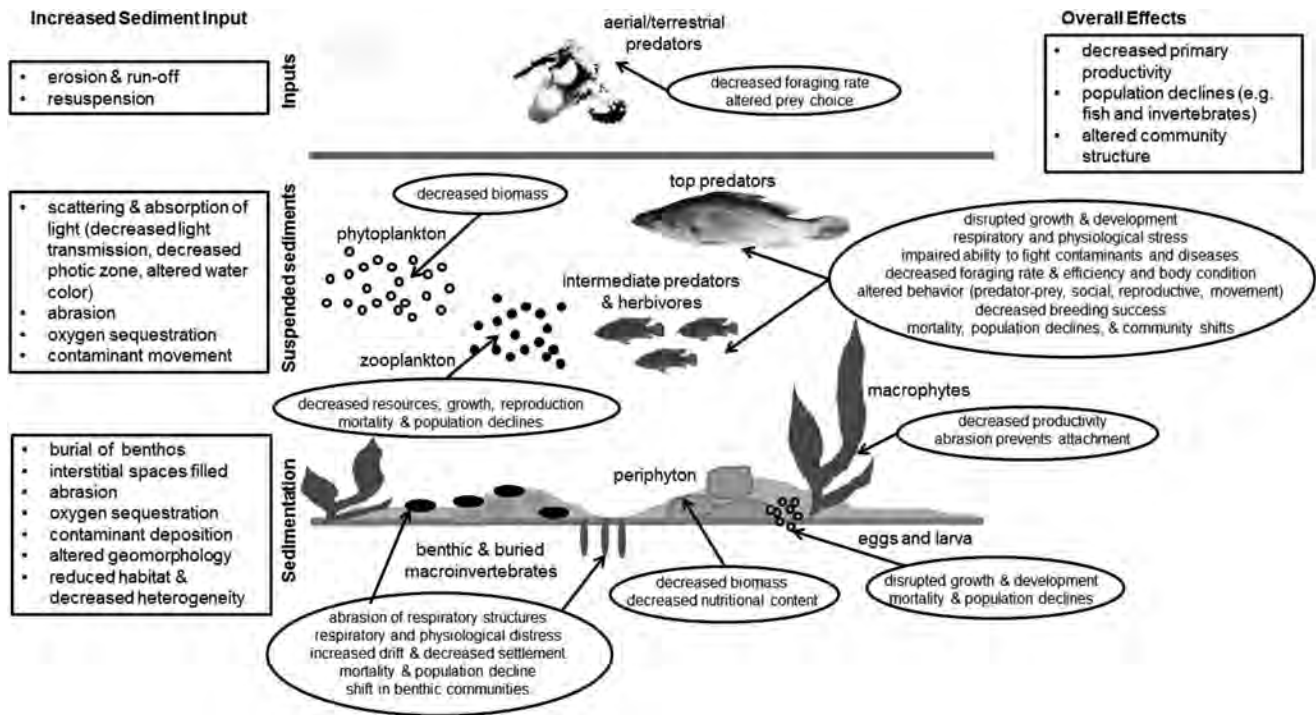


Fig. 1 Graphical representation of major influences of increased sediment inputs on aquatic biota. Text in boxes describes potential physico-chemical changes caused by suspended sediments and sedimentation, and text in ovals describes potential specific effects on aquatic organisms caused by these two stressors.

Source: Adapted from Bruton^[6] ©1985 Springer and Kemp, Sear, et al.^[2] ©2011 John Wiley & Sons.

freshwaters that contain some of the highest biodiversity on the planet.^[13] This exemplifies the fact that excessive sediment inputs can alter the physical and chemical properties of water, including altered flow regimes and hydrological processes in lotic systems, decreased photic zone in lakes and coastal zones, decreased availability of dissolved oxygen, and increased water temperature.^[6,15] In addition, eroded soils carry contaminants (e.g., heavy metals and pesticides) and organic materials with them, posing another source of stress for aquatic life.

Sediment composition and particle size will, in part, determine how aquatic organisms will be affected by this stressor. The impacts of fine sediments [particles less than 2000 μm and thus including sand (<2000 to >62 μm), silt (<62 to >4 μm), and clay (<4 μm)^[14] on aquatic biota have received the greatest amount of attention, as they affect aquatic life in the form of both suspended sediments and through sedimentation. Coarse sediments also contribute to sedimentation as they tend to be deposited faster and can have profound impacts on geomorphology, especially of lotic systems, e.g., through changes in channel morphology and in the heterogeneity of available habitats.^[16] Aquatic biota in systems experiencing human-induced increases in sediment inputs must face this challenge on two major fronts: in the form of suspended sediments and through sedimentation. In both forms, the severity of impacts on aquatic biota is attributed to several key factors: concentration,

duration of exposure, composition of sediment inputs, and particle size.^[7,12,17] The fate of sediment inputs will vary across hydrological regimes (e.g. rivers vs. lakes vs. estuaries) and spatiotemporally; however, the way that individual organisms experience increased sediments in water has some generalities. The focus here is on these general trends.

SUSPENDED SEDIMENTS

Suspended sediments have a multitude of indirect and direct effects on aquatic life, such as changing the amount and color of light underwater, reducing oxygen availability, increasing water temperature, and abrasion of sensitive respiratory structures, as discussed later (Fig. 1). In general, increased suspended sediments have been reported to induce population declines at all trophic levels within aquatic systems, including phytoplankton and periphyton, submerged macrophytes, zooplankton, macroinvertebrates, and fishes and other vertebrates.^[2,6,14]

The scattering of light caused by suspended particles, which creates the muddy or turbid appearance of water, increases the attenuation of light with depth, meaning an overall reduction in the amount of light available underwater.^[15] This effectively decreases the depth of the photic zone, or that area in the water column where photosynthesis can take place. The resultant decrease in primary

productivity is considered one of the most important alterations to aquatic systems experiencing increased turbidity because it can have cascading impacts on the entire aquatic community.^[11] the reduction in the size and depth of the photic zone leads to a reduction in phytoplankton biomass, which can affect the growth and survival of invertebrates that require primary production for food, in turn impacting those animals that feed on invertebrate herbivores and so on up the food chain.^[11] This also reduces the areas where visually orienting animals can see, potentially altering behavioral interactions.^[18,19] In areas that were historically clear, reduction in light availability at depth can also have profound effects by shading submerged macrophytes, leading to their decline or even removal from the ecosystem. All of the animals dependent on macrophytes for food and shelter are again expected to be affected by this change in resources, resulting in community-level shifts from complex, diverse food webs, to simplified webs dominated by turbidity tolerant species.^[16,20]

Increased turbidity, or the scattering of light from suspended particulates, is expected to disrupt visually mediated behaviors, such as foraging, social interactions, reproduction, and predator avoidance, in aquatic organisms and in terrestrial animals that forage in aquatic habitats.^[6,18,19] Suspended sediments change not only the amount of light available underwater through scattering and absorption of light, as noted above, but can also change the spectral composition, or color, of light underwater through the wavelength-dependent scattering of light.^[21] This is why some rivers, for example, appear grayish, while others with more mineral-rich geologies might appear a cloudy red when turbid. Thus behaviors, such as finding a mate based on color pattern for species recognition,^[21] can be compromised, leading to hybridization and the loss of biodiversity. Limited work has also suggested that terrestrial-based aquatic predators (e.g., kingfishers and cormorants) will be affected by changes to underwater light, compromising their ability to efficiently find prey.^[6,18]

Another direct effect of suspended sediments on aquatic animals is the abrasive nature of some particulates. Respiration in aquatic organisms is achieved in a number of ways, including through gills or the integument. Gills are particularly sensitive respiratory structures as they are in direct contact with the external environment, meaning that any particulates in the water column will be filtered across sensitive structures, potentially leading to abrasion or clogging. High concentrations of suspended particulates can lead to increased mucus surrounding gills and damaged gill lamellae that could directly impair oxygen uptake and potentially impact metabolically challenging activities such as swimming.^[2,6,9] Physical damage of respiratory structures caused by suspended particles has a serious physiological cost for the animal, as the ability to uptake oxygen is compromised. Evidence suggests that such physiological stress can lead to lowered body condition, reduced growth,

hyperplasia, respiratory impairment, reduced tolerance to disease and toxicants, and death.^[6,10]

High suspended sediment loads and sediment resuspension can, in addition to compromising respiration via abrasion of gills, reduce oxygen availability due to high oxygen sequestration capacity of sediment particles^[6] and increased temperature through the absorption of light energy.^[15] The creation of hypoxic (low dissolved oxygen) zones has serious negative impacts on aquatic communities, with mass fish kills the most obvious.^[20] Oxygen availability in water is much lower than in air (~1% in water vs. ~21% in air) and so many aquatic organisms are adapted to acquire oxygen under these circumstances (e.g., through gills or their integument). However, even small changes in oxygen availability can create physiological stress for many aquatic organisms, and a concomitant increase in temperature can exacerbate this problem in ectothermic animals. As an example, ectothermic fishes that experience increased ambient temperature will have increased metabolic demands, thus requiring more oxygen. If the warmer water also has less dissolved oxygen, the fish will not be able to meet their metabolic demands, possibly leading to death.^[22]

Many fishes maintain a top trophic position in aquatic food webs and are of high economic importance; therefore, they have been extensively studied in the context of responses to increased suspended sediments.^[2,7] Freshwater fish biodiversity, in particular, has decreased drastically, in part attributed to increased sedimentary inputs.^[20] These declines are likely due to both direct effects of sediments on fish (e.g., gill clogging) and indirect effects (e.g., reduced growth due to declines in food availability). Extensive experimental work examining the mechanisms by which suspended sediments might lead to fish population declines show a suite of traits that can be negatively impacted. For example, work on economically important salmonids (e.g., trout and salmon) has shown reductions in foraging rate and efficiency, growth rate, respiratory and physiological stress, altered behavior (e.g., reproductive, homing, and predator avoidance), and egg and larval mortality.^[2,6,7] Additional responses of fish generally include reduced toxicant tolerance, elevated stress hormones, disrupted growth and development, and population declines.^[2,6] While fish responses were highlighted here, other obligate aquatic vertebrates (e.g., amphibians) likely respond in similar ways to increased suspended sediment loads.

As more experimental work is performed on fishes, some ambiguity has been reported in species-specific responses to increased suspended sediments. For example, the “level” of turbidity or concentration of suspended particles varies widely across studies and responses, with some fishes responding lethally to very minor increases in turbidity, and others not responding in any negative way to very high suspended sediment loads.^[2,12] In some fishes, moderate increases in turbidity may impart a short-term benefit because scattered light creates a background against

which planktonic prey contrast strongly over short distances,^[9] making them easier to find. The same principle may also make it more difficult for top predators to find their fish prey at longer distances. Although only a few studies have directly tested this idea, it is only expected at low-to-moderate levels of turbidity.^[23] The long-term effects of moderate increases in turbidity on respiration and community dynamics, as examples, are not known.

In general, few studies have tested the full suite of factors imposed by suspended sediments on fish and other aquatic animals, or the synergistic effects of concentration, duration, composition, and particle size. It is clear from the literature that fishes and macroinvertebrates^[17] show species-specific responses to increased suspended sediments; however, proximate mechanisms underlying the link between turbidity and fish population declines are potentially complex and not well understood.^[7]

SEDIMENTATION

Many of the sediments deposited into waterways will sink to the substrate through the process of sedimentation. The effect of sediment deposition on aquatic life can be as dramatic as when sediments are suspended and has been well studied for benthic communities.^[9,12] Sedimentation is considered a major factor in the degradation of riverine habitats resulting in the loss of benthic biota.^[9,14] The major habitat changes associated with sedimentation, such as burial, filling of interstitial spaces, and reduction in oxygen availability, are discussed later.

The rate of sedimentation depends on a number of physical characters of the sediments and the hydrology and geomorphology of the water body. For example, silt deposition in fast flowing streams will be slower than in low-flow areas, such as behind impoundments (e.g., dams). Stream geomorphology is often altered as a result of sedimentation and results in what was a heterogeneous river (with riffles, pools, submerged structure, etc.) becoming a channelized and homogenized habitat lacking essential resources for many rheophilic and structure-loving animals.^[16] The removal of heterogeneous patches will greatly diminish the complexity of benthic communities of plants and animals and may provide the opportunity for non-native invasive species to infiltrate the altered habitat.

As benthic substrate is inundated with sediments, periphyton growth is reduced (i.e., due to covering of substrate and/or lack of light for photosynthesis), thereby decreasing the food supply for many benthic organisms. In addition, settlement and attachment of periphyton or other sessile organisms (e.g., corals, sponges, and mollusks) are prevented by the covering of suitable hard substrate by sediments. At the same time, interstitial spaces become filled. These spaces, found under and between benthic substrate such as cobble, are required for the persistence of many benthic invertebrates and the eggs and

larva of fish and amphibians, as they provide refuge from predators and nursery grounds.^[6,9,10,17] Changes in the invertebrate and fish communities reflect changes in habitat heterogeneity (e.g., as patches required by some species are lost) and also reduced oxygen availability (depending on the organic content of sediments).

The smothering action of sedimentation on the physical structure of the benthic habitat also functions to smother aquatic organisms via clogging of the gills and integument, thus reducing oxygen uptake. For example, demersal adhesive fish eggs have a sticky coating that will bind sediments falling out of suspension, effectively creating a hypoxic situation in which obtaining oxygen for routine metabolic function is diminished.^[7] The physiological costs associated with decreased capacity to obtain oxygen can lead directly to death in organisms not able to compensate, e.g., by moving away from areas of high sedimentation. The eggs and larva of many fish species are susceptible to this smothering action, resulting in high levels of mortality.^[2,6,7] This loss of early life history stages in many species likely scales up to population declines, although direct tests of this link are scarce.

CONCLUSION

The effects of increased sedimentary input into earth's waters highlight the critical idea that what we do on land will influence water quality and aquatic ecosystem functioning. All levels of biological organization in aquatic habitats and those in surrounding terrestrial habitats can be impacted by human-induced sediment movement to aquatic systems. Mitigation and management of land- and water-based activities will be important as we move forward. For example, where dredging activities take place on a regular basis, accounting for fish spawning seasons and the effects of sediment influx through resuspension and subsequent falling out of suspension on eggs and larvae might be particularly important.^[8] An interesting example of such considerations is the removal of dams no longer in commission, which has instigated new research to understand how the potentially sudden release of impounded sediments will influence both the physical and biological properties of rivers and downstream ecosystems.^[24] The spatial (e.g., downstream) and temporal (e.g., long-term) impacts of dam removal on aquatic communities are only beginning to be understood.^[24]

It is clear that increasing the amount and type of suspended sediments and sedimentation in aquatic systems will have profound influences on aquatic organisms. Considerable evidence suggests that changes in sedimentary inputs from the terrestrial to aquatic environments can drastically alter the community dynamics within aquatic ecosystems and may have cascading effects on riparian zones.^[13] Linking the mechanistic effects, such as abraded gills and physiological stress, with population declines and

community-level effects is required to fully understand the processes by which altering the terrestrial landscape will affect entire aquatic ecosystems at local and global scales.

REFERENCES

1. Walling, D.E.; Fang, D. Recent trends in the suspended sediment loads of the world's rivers. *Global Planet. Change* **2003**, *39*, 111–126.
2. Kemp, P.; Sear, D.; Collins, A.; Naden, P.; Jones, I. The impacts of fine sediment on riverine fish. *Hydrol. Process.* **2011**, *25*, 1800–1821.
3. Syvitski, J.P.M.; Vorosmarty, C.J.; Kettner, A.J.; Green, P. Impact of humans on the flux of terrestrial sediment to the global coastal ocean. *Science* **2005**, *308*, 376–380.
4. Walling, D.E. Human impact on land-ocean sediment transfer by the world's rivers. *Geomorphology* **2006**, *79*, 192–216.
5. Dearing, J.A.; Jones, R.T. Coupling temporal and spatial dimensions of global sediment flux through lake and marine sediment records. *Global Planet. Change* **2003**, *39*, 147–168.
6. Bruton, M.N. The effects of suspensoids on fish. *Hydrobiologia* **1985**, *125*, 221–241.
7. Newcombe, C.P.; MacDonald, D.D. Effects of suspended sediments on aquatic ecosystems. *N. Am. J. Fish. Manage.* **1991**, *11*, 72–82.
8. Wilbur, D.H.; Clarke, D.G. Biological effects of suspended sediments: A review of suspended impacts on fish and shellfish with relation to dredging activities in estuaries. *N. Am. J. Fish. Manage.* **2001**, *21*, 855–875.
9. Jones, J.I.; Murphy, J.F.; Collins, A.L.; Sear, D.A.; Naden, P.S.; Armitage, P.D. The impact of fine sediment on macro-invertebrates. *River Res. Appl.* **2012**, *28*, 1055–1071.
10. Waters, T.F. *Sediment in Streams: Sources, Biological Effects, and Control*; Monograph; American Fisheries Society: Bethesda, 1995.
11. Henley, W.F.; Patterson, M.A.; Neves, R.J.; Lemly, A.D. Effects of sedimentation and turbidity on lotic food webs: A concise review for natural managers. *Rev. Fish. Sci.* **2000**, *8* (2), 125–139.
12. Collins, A.L.; Naden, P.S.; Sear, D.A.; Jones, J.I.; Foster, I.D.L.; Morrow, K. Sediment targets for informing river catchment management: International experience and prospects. *Hydrol. Process.* **2011**, *25*, 2112–2129.
13. Wantzen, K.M.; Mol, J.H. Soil erosion from agriculture and mining: A threat to tropical stream ecosystems. *Agriculture* **2013**, *3*, 660–683.
14. Wood, P.J.; Armitage, P.D. Biological effects of fine sediment in the lotic environment. *Environ. Manage.* **1997**, *21* (2), 203–217.
15. Kirk, J.T.O. Effects of suspensoids (turbidity) on penetration of solar radiation in aquatic ecosystems. *Hydrobiologia* **1985**, *125*, 195–208.
16. Sullivan, S.M.P.; Watzin, M.C.; Hession, W.C. Understanding stream geomorphic state in relation to ecological integrity: Evidence using habitat assessments and macro-invertebrates. *Environ. Manage.* **2004**, *34* (5), 669–683.
17. Donohue, I.; Irvine, K. Effects of sediment particle size composition on survivorship of benthic invertebrates from Lake Tanganyika, Africa. *Arch. Hydrobiol.* **2003**, *157* (1), 131–144.
18. Abrahams, M.V.; Kattenfeld, M.G. The role of turbidity as a constraint on predator-prey interactions in aquatic environments. *Behav. Ecol. Sociobiol.* **1997**, *40*, 169–174.
19. Utne-Palm, A.C. Visual feeding of fish in a turbid environment: Physical and behavioural aspects. *Mar. Freshw. Behav. Phys.* **2002**, *35*, 111–128.
20. Donohue, I.; Garcia-Molinis, J. Impacts of increased sediment loads on the ecology of lakes. *Biol. Rev.* **2009**, *84*, 517–531.
21. van der Sluijs, I.; Gray, S.M.; Amorim, M.C.P.; Barber, I.; Candolin, U.; Hendry, A.P.; Krahe, R.; Maan, M.E.; Utne-Palm, A.C.; Wagner, H.J.; Wong, B.B.M. Communication in troubled waters: The evolutionary implications of changing environments on fish communication systems. *Evol. Ecol.* **2011**, *25*, 623–640.
22. Pörtner, H.O. Oxygen- and capacity-limitation of thermal tolerance: A matrix for integrating climate-related stressor effects in marine ecosystems. *J. Exp. Biol.* **2010**, *213*, 881–893.
23. Pangle, K.L.; Malinich, T.D.; Bunnell, D.B.; DeVries, D.R.; Ludsins, S.A. Context-dependent planktivory: Interacting effects of turbidity and predation risk on adaptive foraging. *Ecosphere* **2012**, *3* (12), 114.
24. Stanley, E.H.; Doyle, M.W. Trading off: The ecological effects of dam removal. *Front. Ecol. Environ.* **2003**, *1* (1), 15–22.

Sedimentation Control and Erosion: Engineering Techniques

Jose Miguel Reichert

Federal University of Santa Maria, Santa Maria, Brazil

Elemar Antonino Cassol

*Department of Soils, Federal University of Rio Grande do Sul (UFRGS),
Porto Alegre, Brazil*

Abstract

Engineering techniques for soil erosion/sedimentation control involve manipulation of surface topography and usually are complex and costly to install and maintain, thus requiring proper planning and implementation. However, they are essential for controlling and orienting the flow of excess water generated in agricultural, forest, pasture, or urban land and should be used in conjunction with agronomic and management practices.

INTRODUCTION

Water erosion is caused basically by raindrop impact and runoff of excess water (Fig. 1), thus erosion and sedimentation control strategies must be based on covering the soil against raindrop impact, increasing water infiltration to reduce runoff generation, and increasing surface roughness to reduce overland flow velocity. To pursue these objectives, several strategies (Fig. 2) can be adopted, including agronomic measures, soil management, and mechanical or engineering methods^[1,2] as discussed and illustrated in this entry.

Erosion is a natural process, which cannot be completely controlled, but engineering methods can help in reducing erosion rates to maintain a maximum sustained level of production while keeping soil loss below a threshold (soil loss tolerance). Erosion control is also needed to reduce nutrient and pesticide losses, to prevent pollution of surface water, and to avoid sedimentation or siltation of water bodies.

It is largely accepted that soil management and agronomic measures are more suitable for erosion control because of lower cost and greater effect on reducing soil detachment due to surface protection, increased water infiltration, and runoff reductions and are easily incorporated into existing and improved farming systems,^[3] with characteristics of sustainability.

ENGINEERING TECHNIQUES

Practices of engineering erosion control include contouring, terraces, contour barriers and stone terraces, waterways, conservation structures, gully stabilization structures, and geotextiles.^[1–3]

Contouring

Contouring consists of performing all field practices such as plowing, disking, planting, and cultivation on the contour

(Fig. 3). It is effective for low slopes and short slope length and for rainstorms of low intensity. The effectiveness of this practice is about 50%^[2] depending upon slope grade and length, and this effectiveness can be increased by establishing a series of depressions or pits (Fig. 4), which fill with water and sediment during rain.

Terraces

Terraces are mechanical structures constructed across a slope and consist of a channel and an earth or stone bank to intercept surface runoff, allow it to infiltrate and evaporate or divert the excess to a stable outlet at a non-erosive velocity. Depending upon their function or type of construction, terraces are classified as retention or absorption, diversion or graded, and bench terraces. Retention or absorption terraces are used when water is to be stored on the hillside of slopes less than 4.5°, having level channels able to accumulate runoff volume generated by a 10-yr return period rainfall (Fig. 5). Diversion or graded terraces aim at intercepting runoff and conduct it to a proper outlet through a channel of a slight grade, usually 1:250 (Fig. 6) on slopes less than 7°. Bench terraces are used on steep slopes, up to 30°, alternating a series of shelves (cuts) and raises (fills; Fig. 7). Research data from several authors^[1] indicate an effectiveness in soil loss control of bench terraces up to 93%. Terraces must be properly designed to avoid even greater soil erosion damage,^[2] and should be planned considering the whole farm, a group of neighboring farms, or even a small hydrographic catchment,^[3] to ensure optimization of landuse.

Contour Ridges and Stone Terraces

Contour ridges are earth banks, similar to narrow-based terraces, constructed across the slope using hand tools,



Fig. 1 Rill and interrill erosion and sedimentation because of improper soil conservation in southern Brazil in the early 1970s.



Fig. 2 Agricultural field with agronomic and engineering soil conservation methods in southern Brazil.



Fig. 3 Contouring in an agricultural field in southern Brazil.

animal traction implements, and sometimes motorized farm machinery. These structures are frequently used on small-scale farms and hilly regions. Their effectiveness in reducing soil loss varies but research data from several authors indicate values up to 80% and even 100% control.^[1] Contour stone bunds (Fig. 8) are an alternative technique on stony lands, where stone walls are set up in shallow trenches. These allow slow retention of sediment and are permeable to water. With time, upslope sedimentation will generate a gradual development into bench terraces. The construction of stone barriers requires considerable labor and time but is more permanent and easier to maintain than earth barriers.^[3] Spacing and construction details are presented by FAO.^[3]



Fig. 4 Sedimentation pits in a horticultural field in Costa Rica.



Fig. 5 Retention or absorption terraces in southern Brazil, accumulating runoff in a level channel.

Waterways

A system of graded terraces requires waterways to receive the excess drained runoff and safely conduct at a non-erosive velocity to lower parts of the landscape. Waterways must be carefully established (considering location, construction, and stabilization) and maintained (managing grass species and controlling incipient erosion) to avoid serious erosion problems, even causing gully formation. Grass waterways



Fig. 6 Diversion or graded terraces in southern Brazil.



Fig. 7 Bench terraces on high-grade hillslope in Guatemala.



Fig. 8 Contour stone barriers in an agricultural field in Cape Verde.

(Fig. 9), tile drains, or on steep slopes, concrete structures (Fig. 9) are located in natural depressions or, if needed, reshaped channels. The grass waterways reduce water flow velocity because of the retardance effects of vegetation and increased soil resistance to erosion due to soil coverage and aggressive rooting system holding soil particles together. To work properly, waterway design follows defined principles from channel hydraulics,^[1,4] and the flow ways must be maintained and cared for.^[3]



Fig. 9 Grass waterway draining to a concrete drop inlet structure in southern Brazil.



Fig. 10 Conservation structures showing drop spillway and flumes to convey water downslope, along with stone barriers, in Spain.

Conservation Structures

In certain situations, the only feasible way to convey safely large amounts of runoff water from a higher elevation to a lower one is through the use of conservation structures that are made of some type of non-erodible material, such as concrete or large rocks. Examples of these structures include chute drop inlets (Fig. 9), flumes and drop spillways (Fig. 10), and pipe spillways, which must follow standard functional requirements to maintain channel stabilization.^[4] Selection and design of these structures are based upon existing or anticipated erosion problem, slope steepness, estimated maximum water flow discharge, and cost of alternative systems. Conservation structures are usually used in conjunction with other erosion control techniques such as grass waterways (Fig. 9).

Gully Stabilization Structures

Gullies represent an advanced stage of rill erosion (Fig. 11) and are most important as a source of sediment



Fig. 11 Gully development on sandy soil in southern Brazil.

in streams. If the gully is small, recovery might be suitable by filling it with soil, but in many cases, recovery is technically and economically not viable. In such cases, gully stabilization is recommended by constructing small dams using diverse materials depending on availability. These materials may include stones, wood planks or branches, earth, etc., to trap sediment, thus reducing gully depth and slope. Along with these stabilization structures, plants must be established to reclaim the affected areas (Fig. 12). Sometimes, to control the overfall of water on the headwall on large gullies, more permanent structures are required.^[1,2]

Geotextiles

Erosion on road baks or urban areas is best controlled by vegetation but establishment on these highly erodible and usually infertile soil surfaces is difficult. Thus, other materials including straw and hay to help establishing vegetation are used but these do not hold tight to the soil and are washed away easily.^[2] Alternative materials include geotextiles that are permeable textiles for erosion control interacting with soil and vegetation. The geotechnical material is in the form of a mat, sheet, grid, or web of natural or synthetic fiber.^[1] The first type is biodegradable while the second type gives a more permanent protection to a slope, interacting with roots to improve soil cohesion and slope stability.



Fig. 12 Gully stabilization with used tires, stones, and vegetation in Costa Rica.

CONCLUSION

Engineering techniques vary in their purpose and suitability. For a given field condition, a variety of methods should be considered and adopted in an integrated manner, depending on whether water or wind erosion is the most prominent problem. Technology, money, and labor availability are also constraints in adopting a given method; thus, for different locations around the world, myriad ways of adaptation are developed and used by professionals and farmers.

REFERENCES

1. Morgan, R.P.C. *Soil Erosion and Conservation*, 2nd Ed.; Wiley: New York, 1995; 198 pp.
2. Hudson, N. *Soil Conservation*, 2nd Ed.; Cornell University Press: Ithaca, 1995; 391 pp.
3. FAO. *Manual on Integrated Soil Management and Conservation Practices*; FAO Land and Water Bulletin 8; IITA and FAO: Rome, 2000; 214 pp.
4. Schwab, G.O.; Fangmeier, D.D.; Elliot, W.J. *Soil and Water Conservation Engineering*, 4th Ed.; New York: Wiley, 1992; 528 pp.

Seepage

Patricia M. Gallagher

Drexel University, Philadelphia, Pennsylvania, U.S.A.

Abstract

Seepage is a complex process that can be difficult to quantify and analyze. In certain situations, such as flow through dams, simple analysis methods may be employed. More complex analysis is required for applications such as developing and managing groundwater resources. The difficulties associated with measuring the hydraulic conductivity accurately and accounting for the inherent variability of natural soil deposits is the most challenging aspects of seepage analysis.

INTRODUCTION

The term *seepage* is used to describe the flow of water through the subsurface. Soil formations consist of soil particles with interconnected voids; seepage occurs through the voids. Seepage is important in engineering applications because it affects the stability of dams and levees, as well as dewatering of excavations during construction. Additionally, since groundwater is often used for water supply and irrigation, it has to be protected from overuse and contamination. This entry provides a brief overview of seepage principles and applications. For more detailed information, see the studies by Cedergren,^[1] Coduto,^[2] Driscoll,^[3] Duncan,^[4] Fetter,^[5,6] Freeze,^[7] and Harr.^[8]

Water reaches the soil when precipitation percolates through the unsaturated zone under gravity flow until it reaches the *water table* or *phreatic surface*, which is the top of the zone of saturation and may vary seasonally. Once the water reaches the phreatic surface, seepage is typically horizontal through the soil. Eventually, the groundwater recharges streams and rivers or may be pumped from the ground as a water supply.

The quantity and speed of flow through a soil formation depend on the size of the voids through which the water flows. There are essentially two types of soil deposits as follows: sand and clay. Sand deposits have bulky particles and larger voids than clay deposits, so that more water tends to flow faster through sand than through clays. Deposits that readily transmit water are called aquifers, while aquiclude permit very little flow. Clays are often used as barriers to flow, e.g., in dams or as compacted liners while sand deposits often function as water-supply aquifers.

SEEPAGE PRINCIPLES

Water, both static and flowing, affects stresses and stability in soil. The static water pressure below the groundwater

table increases linearly with depth according to the following relationship

$$u = \gamma_w z \quad (1)$$

where u is the water pressure (lb/ft² or kN/m²), γ_w is the unit weight of water (lb/ft³ or kN/m³), and z is the depth beneath the groundwater table (ft or m). When water pressure is described as $z = u/\gamma_w$, it is called pressure head.

For water flowing through the subsurface, the total energy potential is a combination of pressure head, elevation head, and velocity head. Total head can be calculated using Bernoulli's equation as follows:

$$h = \frac{u}{\gamma_w} + z + \frac{v^2}{2g} = \text{constant} \quad (2)$$

where h is the total head, or the sum of the energy potentials, u is the water pressure (lb/ft² or kN/m²), z is the elevation head (ft or m), v is the water velocity (ft/s or m/s), and g is the gravity (ft/s² or m/s²). The velocity of groundwater is generally so small that velocity head can be considered to be zero for most flow problems.

While the total energy is constant at any point in the region of flow, the viscous resistance within the pores of the formation causes a loss of energy across the region of flow, described as *head loss*, Δh , as shown in Fig. 1. In Bernoulli's equation, the energy between points A and B can be written as follows:

$$\frac{u_A}{\gamma_w} + z_A = \frac{u_B}{\gamma_w} + z_B + \Delta h \quad (3)$$

where Δh is the total head loss over the flow distance ΔL .

The ratio of head loss to the flow distance is called the *hydraulic gradient*,

$$i = \frac{\Delta h}{\Delta L} \quad (4)$$

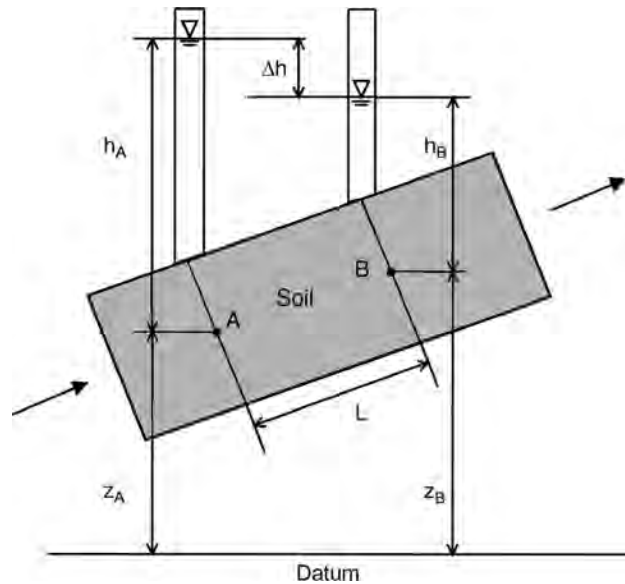


Fig. 1 The illustration of head loss between two points in a soil mass.

While the velocity head can usually be considered zero in Bernoulli's equation, the flow velocity is very important for determining the quantity of flow moving through the soil formation. Darcy's law is used to relate the flow of water through soil to the hydraulic gradient as follows:

$$v = ki \quad (5)$$

where v is the Darcy velocity or specific discharge (ft/s or m/s) and k is the hydraulic conductivity (ft/s or m/s).

The *Darcy velocity* is actually a quantity rather than a velocity, so it is more correctly called the *specific discharge* or *flux*. It accounts for the quantity of flow traveling through the entire cross-sectional area of flow, including both soil particles and voids. The *seepage velocity*, v_s , is the velocity at which water actually moves through the area occupied by the pore spaces. Therefore, it is always larger than the Darcy velocity. The area occupied by the pore spaces is typically quantified in terms of *porosity*, n , the ratio of the volume of voids to the total volume. Porosity ranges from about 25% to 50% for most soil deposits. The Darcy velocity and the seepage velocity are related by the porosity as follows:

$$v = v_s n \quad (6)$$

Darcy's law, which is valid for most types of soil formations, is the basic equation used to describe groundwater flow.

HYDRAULIC CONDUCTIVITY

Hydraulic conductivity, a measure of water's ability to move through the soil, is the proportionality constant that

relates the Darcy velocity to the hydraulic gradient. It describes the properties of both the formation and the fluid moving through it. A second parameter, *intrinsic permeability*, K , describes the properties of the media, such as the grain size and the tortuosity of the flow path. The hydraulic conductivity and the intrinsic permeability are related by the following equation:

$$k = K \frac{\rho g}{\mu} \quad (7)$$

where ρ is the density of the permeant (kg/m^3) and μ is the viscosity of the permeant (Pa s or cP).

Hydraulic conductivity is highly variable spatially due to the heterogeneous nature of soil deposits and/or the presence of discontinuous layers or faults. Additionally, hydraulic conductivity varies with the direction of flow and can vary by orders of magnitude over the distance of a few inches or feet, even in soil deposits that appear to be fairly homogeneous. The hydraulic conductivity is a function of the void ratio and the grain size distribution of the formation. In general, sands have much higher hydraulic conductivity values than clays and can transmit large quantities of water. Typically, hydraulic conductivity values of coarse sands and gravels are in excess of $>10^{-3}$ m/s; fine sands range from 10^{-3} m/s to 10^{-5} m/s; silty sands range from 10^{-5} m/s to 10^{-7} m/s; silts range from 10^{-7} m/s to 10^{-9} m/s; and clays tend to be $<10^{-9}$ m/s.

Hydraulic conductivity can be measured in either the laboratory or the field. Laboratory tests are done on small samples in a controlled environment, so while they may give accurate values, they cannot capture the variability of a deposit. Field methods may provide a better overall idea of the hydraulic conductivity of a formation, but they are expensive and can be difficult to interpret.

Two types of laboratory tests are commonly used to determine the hydraulic conductivity. The constant head test is used on highly permeable soils such as sand or gravel. The falling head test is more suitable for measuring the hydraulic conductivity of less permeable soils such as silt and clay. In both tests, the flow of water through a specimen of known area is measured, and Darcy's law is used to calculate the hydraulic conductivity.

Field tests typically require several wells, including a central pumping well and surrounding observation wells. Water is pumped out of the pumping well, and the water levels in the observation wells are measured when steady conditions are reached. The interpretation of pump tests is based on certain simplifying assumptions; if these assumptions are not valid for the formation in question, the results may be questionable. Single wells may also be used to measure the hydraulic conductivity. The tests are done by causing an instantaneous change in the water level through either

addition (a slug test) or removal (a bail test) of a known volume of water. The time taken by the water level to recover is measured and used to calculate the hydraulic conductivity. There are also indirect methods of assessing the hydraulic conductivity, including correlations based on the grain size distribution of the formation.

APPLICATIONS OF SEEPAGE THEORY

Dams and Levees

Dams are used for a variety of reasons, including flood control, water supply, and storage of liquid wastes. Levees are used primarily for flood control. Seepage through and beneath dams and levees is an important consideration in design. The primary concerns in seepage analysis are as follows: 1) the amount of seepage that will go through or beneath the structure; 2) the potential for erosion and piping at the downstream end; and 3) the uplift pressure acting underneath the structure. If a dam is used for water supply, care must be taken to limit the amount of seepage through or beneath the dam since water flowing beneath the dam cannot be used as water supply.

The simplest method for analyzing the flow through or beneath a dam is a *flow net*. An example of a flow net is shown in Fig. 2. A flow net is a graphical solution to the Laplace continuity equation, which describes steady flow in the x and y directions as follows.

$$\frac{\partial^2 h}{\partial x^2} + \frac{\partial^2 h}{\partial y^2} = 0 \quad (8)$$

As shown in Fig. 2, the solution consists of the following two families of mutually perpendicular lines: 1) *flow lines*, which show the path water travels along and 2) *equipotential* lines, which show the contours of total head. Rules for drawing flow nets can be found in most textbooks on introductory soil mechanics. Computer programs may also be used to solve the Laplace equation, but the flow net is simple to use and, considering the uncertainty associated with the determination of the hydraulic conductivity, often accurate enough for most engineering analyses.

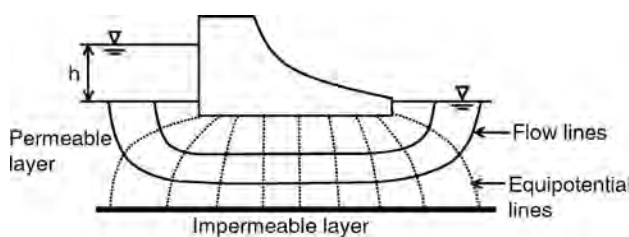


Fig. 2 Example flow net.

The flow net can be used to calculate the total head and the water pressure at any point beneath the structure, the seepage quantity, and the risk of damage to the dam from erosion and piping. Piping is the loss of soil from beneath the dam. If enough soil is removed through erosion, channels can form in the foundation, allowing uncontrolled flow of water beneath the dam and often failure. Erosion is most likely to occur where the hydraulic gradient is the highest, typically where the water exits from beneath the structure. If erosion is a problem, solutions include lowering the upstream water level, placing a filter at the downstream end of the dam, and increasing the length of the flow path by placing an impermeable layer at the upstream edge of the dam.

Water Supply and Groundwater Contamination

Groundwater must be properly managed as a resource. Aquifers are often the source of drinking and irrigation water. If overpumped, resources could be depleted, causing loss of water supply, loss of base flow to streams, or salt-water intrusion in areas near coastlines. Additionally, since groundwater contamination can affect water quality, it is necessary to protect against it.

Aquifers are either confined or unconfined. An unconfined aquifer is one in which the water level is below the top of the soil formation, so the water is under atmospheric pressure. If more water flows into the aquifer, the water level will rise. In contrast, a confined aquifer is one in which the top boundary is aquiclude, confining the water in the aquifer and causing it to be under pressure. If a well is drilled into a confined aquifer, the water will rise above the top of the aquifer. If the water flows to the ground surface, the well is called artesian.

Although groundwater modeling is a complex 3-D problem, there are some simple analyses for modeling the flow of water to a single well. For a confined aquifer (Fig. 3), the flow rate (Q) from the well may be calculated using the following equation:

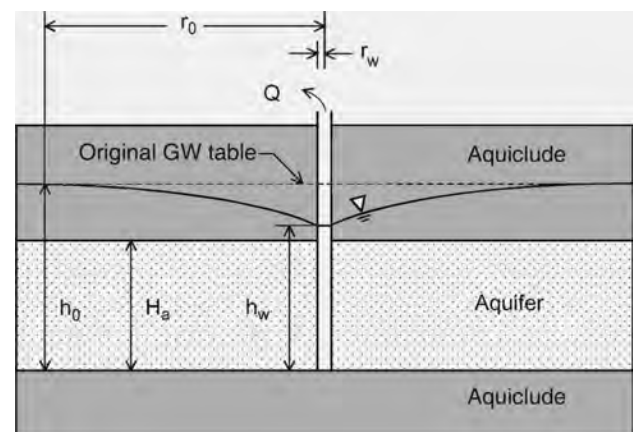


Fig. 3 Well in a confined aquifer.

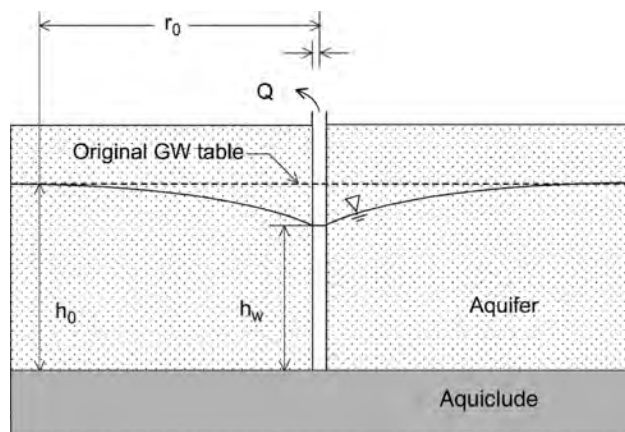


Fig. 4 Well in an unconfined aquifer.

$$Q = \frac{2\pi k H_a (h_0 - h_w)}{\ln\left(\frac{r_0}{r_w}\right)} \quad (9)$$

where H_a is the aquifer thickness, r_0 is the drawdown radius, h_0 is the water height at r_0 , r_w is the well radius, and h_w is the water height at well.

For an unconfined aquifer (Fig. 4), the equation for water quantity is as follows:

$$Q = \frac{\pi k (h_0^2 - h_w^2)}{\ln\left(\frac{r_0}{r_w}\right)} \quad (10)$$

While these simple equations are useful to demonstrate the factors that influence flow to a well, most groundwater modeling requires the use of numerical codes. The complexity of subsurface conditions, the heterogeneity and anisotropy of the parameters involved in modeling, and the uncertainty of the measurements make groundwater modeling extremely difficult.

As water resources dwindle and contamination affects the quality of drinking water supplies, proper management of groundwater resources becomes more important. Groundwater contamination can be caused by leakage from underground tanks, by solid waste landfills or lagoons, and by deep-well injection of liquid wastes. In addition, animal wastes, fertilizers, and pesticides may leach into the soil and contribute to groundwater contamination. The movement of contaminants through the subsurface depends on the type present.

Contaminants that dissolve in the groundwater are carried along with the flowing groundwater by the processes of advection, the bulk flow of contaminants along with the flowing groundwater, and dispersion, the spreading out of the contaminant in front of the advancing contaminant front. Dispersion is caused by the variation in pore size in the formation.

If contaminants are lighter than water and insoluble, they will float on top of the water table. These types of

contaminants are called light non-aqueous phase liquids. Contaminants that are heavier than water and insoluble (dense non-aqueous phase liquids) tend to sink to the bottom of the aquifer and get into larger pores and fractures. These types of contaminants can be very difficult to locate and treat.

Dewatering

Dewatering is another important application of seepage principles. When excavations are required below the groundwater table, dewatering is used to remove water from the excavation, so that construction can occur in the dry. The basic types of construction dewatering methods include open pumping, predrainage, and cutoffs. Selecting an appropriate method of dewatering depends on numerous factors, including the hydraulic conductivity and soil conditions, the depth of the excavation below the groundwater table, and the size of the excavation. It can be a complex problem and is typically left to specialty dewatering contractors.

Open pumping consists of controlling the water from the excavation using trenches and sumps. The water is directed to the sumps through the trenches and removed from the excavation by pumping from the sumps. It is a simple method of dewatering and is most suited for use in excavations where the amount of flow is small. In cases where the hydraulic conductivity is high and large amounts of water could flow into an excavation, predrainage may be used. Predrainage is used to intercept the groundwater before it reaches the excavation by pumping from wells or well points placed around the perimeter. It can involve pumping large quantities of water and has the potential to alter the regional groundwater level. In areas where the flow of water through the soil is significant, it can be difficult to keep an excavation dewatered. In cases like this, cutoffs may be used to prevent the water from entering the excavation. A cutoff, or seepage barrier, is designed to minimize the flow of water into an excavation. Typical examples include sheet pile walls and slurry walls. A slurry wall is a trench filled with a soil-bentonite backfill. It is excavated using a bentonite slurry to support the trench walls and then filled with a backfill that has a low hydraulic conductivity. Seepage barriers can be very effective in reducing the amount of water that can flow into the excavation.

CONCLUSION

Seepage is a complex process that can be difficult to quantify and analyze. In certain situations, such as flow through dams, simple analysis methods may be employed. More complex analysis is required for applications such as developing and managing groundwater resources. The

difficulties associated with measuring the hydraulic conductivity accurately and accounting for the inherent variability of natural soil deposits remain the most challenging aspects of seepage analysis.

REFERENCES

1. Cedergren, H.R. *Seepage, Drainage, and Flow Nets*, 3rd Ed.; Wiley Classics in Ecology and Environmental Science; Wiley: New York, 1997; 496 pp.
2. Coduto, D.P. *Geotechnical Engineering: Principles and Practices*, 5th Ed.; Prentice Hall: Upper Saddle River, 1999; 759 pp.
3. Driscoll, F.G. *Groundwater and Wells*, 2nd Ed.; Johnson Division: St Paul, 1986; 1089 pp.
4. Duncan, J.M. *Seepage Course Notes*; Center for Geotechnical Practice and Research; The Charles, E. Via, Jr., Ed.; Department of Civil and Environmental Engineering, Virginia Polytechnic Institute and State University: Blacksburg, VA, 1997.
5. Fetter, C.W. *Contaminant Hydrogeology*; Prentice Hall: Upper Saddle River, 1993; 458 pp.
6. Fetter, C.W. *Applied Hydrogeology*, 4th Ed., Prentice Hall: Upper Saddle River, 2000; 691 pp.
7. Freeze, R.A.; Cherry, J.A. *Groundwater*; Prentice Hall: Englewood Cliffs, 1979; 604 pp.
8. Harr, M.E. *Groundwater and Seepage*; Dover Publications: New York, 1992; 315 pp.

Selenium

Gunnar Gissel-Nielsen

Head of Program Plant Ecosystems, Riso National Laboratory, Roskilde, Denmark

Abstract

Selenium (Se) has been recognized as an essential mineral in human and livestock nutrition for more than 30 years but never proved to be essential for plants. Lack of Se in food has often been viewed as an example of a mineral imbalance related to intensive crop production as the main source of Se for animal and humans in the soil–plant system. This entry explains why field treatment with Se is an inexpensive, safe, and easy way of ensuring a desirable Se intake in humans and animals.

INTRODUCTION

The traditional definition of a plant nutrient is an element essential for the growth and development of plants. However, with the increasing concern on the quality of agricultural crops as animal and human food, elements or compounds that improve the quality should be considered as plant nutrients, too. A clear example is selenium (Se).

GEOGRAPHICAL DISTRIBUTION OF Se

The lower and the upper limits for adequate Se supply are approximately 0.05 mg/kg dry matter and 2 mg/kg, respectively.^[1] Outside Europe, crops containing toxic Se concentrations are found in the mid-western regions of the U.S. and Canadian prairies, in Venezuela, India, and China. Se deficiencies are more common and have been reported in both western and eastern coastal areas of North America, Venezuela, Australia, New Zealand, Japan, and China. However, no information is available about the Se status of most countries of the world. In Europe, Scandinavia is a natural low-Se area, while Central Europe ranges between deficiency and sufficiency. Se toxicity has only been observed in a few spots of Wales, Eire, and Russia.^[2]

Blood Se content is correlated with Se concentrations in the food for both humans and animals. Cereal and vegetables are dominant Se sources in the human diet and, along with pasture plants, in livestock fodder; the Se intake can be estimated to a great extent by evaluating the Se uptake by the fodder crops grown in the area in question. Consequently, an increase in the Se concentration of crops is an obvious way to remedy Se deficiency in humans or livestock (Fig. 1).

Se UPTAKE BY PLANTS

Low-to-moderate soil Se is not correlated with plant Se because of the large number of other factors influencing

the availability of soil Se to plants. In soils with high pH, inorganic Se occurs mainly as selenate, which is not fixed in the soil. Consequently, if precipitation and leaching are low, crops will be rich in Se. This is known in some places in India and in South Dakota, U.S.A.^[3] On the other hand, a low pH favors the selenite form, which is fixed strongly to the soil clay particles and iron hydroxides. Consequently, soils with similar Se content will produce crops with much lower Se content in low pH soils than in high pH soils. Volatile Se is lost to the atmosphere through microbial activity, but Se also returns to the soil from the atmosphere through precipitation. This leaves only a minor part of Se in the cycle to pass through the plant–animal system.

The Se concentration in plants also depends on the time of sampling. The release of Se from clay and organic matter is a slow process. Se released during the winter is available for crops during spring when the yield of grass is low, giving a relatively high Se concentration. When the growth of grass increases during summer, less Se is available and it is distributed over a far greater yield. Therefore, the Se concentration drops as a result of dilution.

FIELD TREATMENT WITH Se

Different ways of raising Se concentrations of crops have been investigated over the years. Of all the different ways, the application of selenate to the soil and foliar application of selenate or selenite are those of practical importance. Pre-sowing treatment of barley (*Hordeum vulgare* L.) and soybean (*Soya hispida* L.) seeds with selenite and selenate has also been tested.

Soil Application

The initial studies on soil application of Se involved spraying of selenite or selenate solutions onto the soil

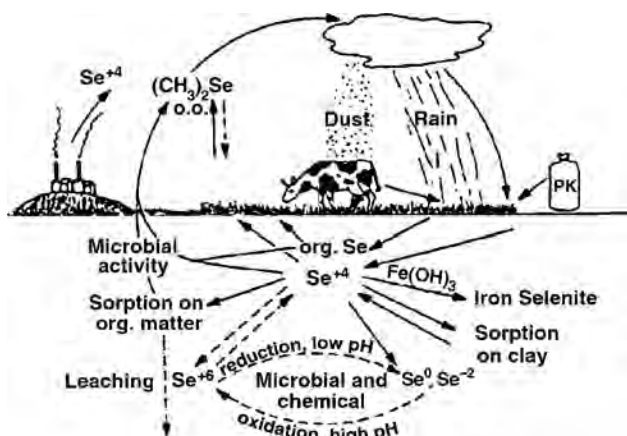


Fig. 1 Possible cycles of Se under field conditions with morainic soils.

Source: From Gissel-Nielsen, Gupta, et al.^[1]

surface in 1967.^[4,5] These relatively simple experiments showed the promising effects of Se addition, and they have been followed by comprehensive studies worldwide.^[2] Different Se salts have been tested in pot and field experiments. Selenates in general are 20 to 50 times more available than selenites, at least in the first year.^[1]

A large-scale field experiment involving an annual addition of 60 and 120 g Se/ha for 5 years as Na_2SeO_3 incorporated in a NPK compound fertilizer was carried out on 21 Danish farms in 1972–1973.^[3] The soils were all glacial deposit mineral soils with a pH of 5–7. The 120 g Se treatment raised the native Se concentration of 0.02–0.04 mg/kg of a number of fodder crops to 0.08–0.13 mg Se/kg, considered a sufficient and safe level for animal nutrition. Other experiments showed a similar effect when about 10 g Se/ha as selenate is added.

A strip of a paddock (5% of the area) was supplied with 340 g Se/ha, which is equivalent to 17 g Se/ha for the total area. The grazing sheep had the same blood Se content as sheep grazing a paddock with the even distribution of Se. The free movement of the grazing animals compensates for an uneven distribution of Se.^[2] For perennial pastures, a single application of 100 g Se/ha can give a desirable residual effect for 4–5 years.

Flue gas desulphurization waste, fly ash, and sewage sludge are examples of alternative sources of Se for field crops. They all resulted in varying increases in the Se concentration of crops. In most cases, the amount of waste needed for remedying a Se deficiency situation was so high that other problems occurred, such as too high cadmium concentrations or salinity problems. These waste products, therefore, could be acknowledged to have a positive effect on the Se concentration of crops when deposited in the field, but they should not be considered Se fertilizers as such.^[2]

Foliar Application

The complication of soil effect on the availability of added Se can be avoided by using foliar application. In a test of foliar applications of up to 50 g Se/ha as selenite in field experiments with spring barley, a linear correlation was found between added Se and Se concentration in the plants. Five to ten grams of Se per hectare resulted in the desired minimum level of about 0.1 mg/kg Se in the grain. The increase in Se concentration was greater in the mature grain than in the straw, indicating an effective translocation of the absorbed Se to the edible part of the plant. The addition of a detergent to the Se solution improved the efficiency of foliar application, giving the desirable minimum concentration of about 0.1 mg/kg in the grain using 2–5 g Se/ha. The foliar application was performed at two growth stages (end of May and end of June). The late application also resulted in a twofold Se increase. The likely explanation is a better plant cover of the soil and thereby a greater part of the solution wetting the leaves.^[2]

In another field experiment, foliar application in the spring of 10 g Se/ha as sodium selenite to an established pasture resulted in concentrations within the desirable range in all cuts during the growing season. The recovery of the added Se in crops was up to 35%.

Comprehensive Finnish field experiments included foliar applications of selenite and selenate. The results were in accordance with the earlier Danish studies: selenate is somewhat more effective than selenite, an amount of 5–10 g Se/ha is sufficient, and the effect depends on the stage of development of the crop at the time of spraying.^[6]

Seed Treatment with Se

Seed treatment with Se has been tested in barley and soybean.^[2,7] The amount of selenite needed to obtain a desirable Se concentration in harvested barley grain was the same as when selenite was added through the fertilizer. The pre-sowing treatment of barley seed with selenite resulted in a concentration of several 100 mg Se/kg seed making it highly toxic to animals and humans.

Experience on a Larger Scale

In Finland, field treatment with Se is performed to an extent that enables the study of overall long-term effects. Since 1985, all Finnish commercial fertilizers for food and feed crops are enriched with sodium selenate: 16 mg Se/kg fertilizer for cereals and horticulture crops and 6 mg Se/kg fertilizers for fodder beets and grass production. The Se concentrations in crops are monitored carefully, and Se content in all crops is elevated, typically from about 0.01 to 0.1–0.3 mg/kg dry matter.^[8]

It was decided to remove fertilizers containing 16 mg Se/kg from the market, leaving only those having 6 mg/kg for all crops. This change stabilized the average Se concentrations of Finnish crops at a somewhat lower level: about 0.1–0.2 mg/kg in cereal grain and about 0.13 mg/kg and 0.17 mg/kg in hay and grass silage, respectively. In 1998, the Finnish authorities decided to increase the Se content of fertilizers to 10 mg Se/kg. In any case, Finland has become a country producing crops with adequate Se levels for livestock and human nutrition instead of being a Se-deficient area.

Risk Assessments

A number of experiments were carried out to study possible harmful effects on the environment from using Se-enriched fertilizers. Fishes in lakes and streams, animals living in the soil or in the fields, or birds did not concentrate the added Se to any harmful level.^[2]

Main Guidelines

All these experiments lead to the following conclusion: Foliar application of about 5 g Se/ha as selenite or selenate and soil fertilization using about 10 g Se/ha as selenate, or about 120 g Se/ha as selenite, are effective annual treatments for raising the Se content of annual crops to a desirable level for human and animal nutrition. The effect of Se is enhanced when it is used with a detergent for foliar application, and for all treatments, the effect is the greatest when carried out on a well-established crop. The residual effect of these treatments is very small. A somewhat higher amount is needed for pasture crops, but this gives a residual effect lasting 2–3 years.

BIOAVAILABILITY OF Se

Whether Se is offered in premixed fodder, concentrates, mineral supplements, water, lick stones, or ruminant pellets, it occurs as inorganic selenite, whereas the Se in crops occurs predominantly as free seleno-amino acids or in proteins.^[2] Organic Se is often found to be more effective than inorganic Se to prevent Se deficiency in livestock. However, the significance of organic versus inorganic Se in animal nutrition is subject to much discussion, so one cannot recommend a single Se compound as the best in all situations.

It is well established that the bioavailability of different Se compounds varies greatly, and this has been shown in several experiments. Such variation stresses the importance of the form of Se in animal and human foodstuffs, and research has been carried out to evaluate the factors that influence the chemical form of Se in plants.

Earlier short-term experiments carried out with tomato roots (*Lycopersicum esculentum* Mill.) and other experiments with maize (*Zea mays* L.) showed that Se is translocated in the xylem sap as selenate when added to the nutrient solution as selenate, whereas selenite is metabolized immediately to seleno-amino acids and translocated as such.^[2]

To evaluate the significance of these findings, rye grass (*Lolium perenne* L.) and barley supplied Se as selenite or selenate through the roots or by foliar application in a pot experiment.^[2] A series of fractionations of Se compounds in grass and barley grain separated the compounds into total and water-soluble Se proteins, seleno-amino acids, selenite, and selenate. Neither the oxidation state of the added Se nor the method of application had any significant effect on the distribution of Se in the various compounds. In all four cases, only about 10% of Se was present in an inorganic form. This implies that any method of supplementing crops with inorganic Se leads to the same Se compounds in the mature plants. Therefore, the effect of different methods can be evaluated purely on the basis of total uptake as a percentage of the total added.

CONCLUSION

Se toxicity in humans and livestock is rare, and only China has reported severe cases of deficiency leading to the death of many humans. Contrary to this, moderate Se deficiencies leading to problems in animals are seen in many countries, and these deficiencies might be more dangerous to human health than often considered. As described earlier, in view of the high rates of cardiovascular disease and certain forms of cancer in Finland at the time, the Finnish authorities legislated the introduction of selenate in all compound fertilizers sold in Finland since 1985. In New Zealand, selenized compound fertilizers are available, and in some countries (e.g., Sweden and the United Kingdom), microelement solutions including Se are on the market for foliar application. In many countries, premixed feed enriched with inorganic Se is available for livestock production. This reflects the growing concern for Se deficiency.

Se is available in tablet form for human consumption in most countries, and in general a much more varied diet including relatively Se-rich foodstuffs such as fish and some vegetables are consumed by humans, but far more attention is paid to the Se nutrition of animals than to the human population. It is likely that increased attention will be paid to the importance of sufficient Se in human diets as well as in livestock feed. The fact that a reasonable Se supplementation of crops is an inexpensive, safe, and easy way of ensuring a desirable Se intake in humans and animals has been shown in this entry.

REFERENCES

1. Gissel-Nielsen, G.; Gupta, U.C.; Lamand, M.; Westermarck, M. Selenium in soils and plants and its importance in live-stock and human nutrition. *Adv. Agron.* **1984**, *37*, 397–460.
2. Gissel-Nielsen, G. Effects of selenium supplementation of field crops. In *Environmental Chemistry of Selenium*; Frankenberger, W.T., Jr., Engberg, R.A., Eds.; Marcel Dekker, Inc.: New York, 1998; 99–112.
3. Gissel-Nielsen, G. *Control of Selenium in Plants*; Risø Report No. 370: Roskilde, 1977; 1–42.
4. Cary, E.E.; Wiecek, G.A.; Allaway, W.H. Reactions of selenite-selenium added to soils that produce low-selenium forage. *Soil Sci. Soc. Am. Proc.* **1967**, *31*, 21–26.
5. Watkinson, J.H.; Davies, E.B. Uptake of native and applied selenium by pasture species. III. Uptake of selenium from various carriers. *N.Z.J. Agric. Res.* **1967**, *10*, 116–121.
6. Yläntä, T. Effect of selenite and selenate fertilization and foliar spraying on selenium content of timothy grass. *Ann. Agric. Fenn.* **1984**, *23*, 96–108.
7. Gupta, U.C.; MacLeod, J.A. Relationship between soybean seed selenium and harvested grain selenium. *Can. Soc. Soil Sci.* **1999**, *79*, 221–223.
8. Jukola, E.; Hakkarainen, J.; Saloniemi, H.; Sankari, S. Effect of selenium fertilization on selenium in feedstuffs and selenium, vitamin E, and beta-carotene concentrations in blood of cattle. *J. Dairy Sci.* **1996**, *79*, 831–837.

Serpentinitic Soils

Rebecca Burt

National Soil Survey Laboratory, U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Lincoln, Nebraska, U.S.A.

Michael A. Wilson

U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS), Lincoln, Nebraska, U.S.A.

Abstract

Serpentinitic soils develop from metamorphosed (serpentinized) ultramafic rocks. Primary mineral constituents are antigorite, chrysotile, and lizardite. While serpentinitic soils compose only small, widespread parts of the earth's surface, they have unique properties that dramatically alter productivity and land use compared to other soils. Elemental deficiencies, imbalances of major elements, and enrichments of nickel, chromium, and cobalt influence plant survival and productivity. Landscapes with these soils typically have sparse, low productive stands of shrubs and other diverse flora, and are not practical for agricultural production. These unique properties have warranted inclusion of specific taxonomic categories on soil classification systems and continue to generate interest of research scientists worldwide.

INTRODUCTION

Serpentinite is a rock consisting primarily of serpentine minerals, e.g., chrysotile, lizardite, and antigorite, with the generalized chemical formula $[\text{Mg}_3\text{Si}_2\text{O}_5(\text{OH})_4]$, derived from the alteration (serpentinization) of such minerals as olivines, pyroxenes, and amphiboles in dunites, peridotites, pyroxenites, and other ultramafic rocks.^[1] Lizardite and antigorite have platy morphology, whereas chrysotile has tubular or fibrous morphology. Fibrous chrysotile is a mineral of the asbestos group^[1] and is known to cause serious human-health problems.^[2,3]

U.S. *Soil Taxonomy*^[4] identifies serpentinitic soils at the family level as having magnesian mineralogy “any particle-size class, except for fragmental, with >40% by weight magnesium (Mg)-silicate minerals, such as serpentine minerals (antigorite, chrysotile, lizardite) plus talc, olivines, Mg-rich pyroxenes, and Mg-rich amphiboles, in the fine-earth fraction (<2 mm).” With little substitution of iron (Fe) for Mg in serpentine, Fe in peridotite minerals is incorporated into common accessory minerals such as magnetite and hematite.^[5]

IDENTIFICATION OF SERPENTINE MINERALS

Serpentine minerals present quantification problems by X-ray diffraction (XRD) or optical analysis without labor-intensive pretreatments to remove Fe oxides from mineral surfaces. Oxides attenuate peaks during XRD analysis and

mask mineral grains during optical analysis of sands or silts, preventing proper identification. Additionally, crystalline units of individual serpentine minerals are generally below the observational limit of the optical microscope, and two or more mineral species are often intermixed in sand-sized grains.^[6,7] This characteristic thus requires the use of an electron microscope for exact mineral identification and limits the utility of the optical microscope in quantification of individual mineral species. Thermogravimetric analysis (dehydroxylation at 600–650°C) and total analysis are other methods used in the identification and quantification of serpentine minerals,^[8] typically requiring no pretreatments and allowing determination on the entire <2 mm fraction.

SOIL PROPERTIES

Ultramafic rocks occupy <1% of the land surface of the earth but are found in many parts of the world, e.g., North America, Europe, continental Asia, Japan, Australia, New Zealand, Africa, Brazil, Puerto Rico, and Cuba.^[9] “Serpentine barrens” is a common term for soils developed from ultramafic (serpentinized) rock.^[10–12] Serpentinic soils are recognized for their unusual, diverse, and sparse flora.^[9] Vegetation on these soils has been linked to nutrient deficiencies and imbalances [low nitrogen (N), phosphorus (P), and potassium (K); adverse calcium (Ca)/Mg ratios], non-anthropogenic metal enrichments [Nickel (Ni), chromium (Cr), and cobalt (Co)], low-water holding capacity,

and susceptibility to erosion, with resulting slope instability and/or land sliding.^[13–18] Vegetation on serpentinic soils in the Siskiyou-Klamath Mountains of southwest Oregon is reduced in kind and number, relative to surrounding landscapes, with broad-leaved trees typically absent and shrubs and pines of greater importance.^[12]

Fertility studies of serpentinic soils documented Ca/Mg ratios <0.7 ^[9,13,19,20] as unfavorable for the growth of most plants.^[21–23] Plant deficiencies of Ca can be the result of low Ca and/or excess Mg, antagonistic to the uptake of Ca by plants.^[24] Low Ca^{2+} can also result in lowered ability to counteract the adverse effects of such ions as Na^+ , Mg^{2+} , and H^+ .^[25,26] In a study of 22 soils on serpentinized peridotite in California,^[14] productivity (timber-yield index) was related more to Ca/Mg ratio in surface horizons than to any combination of climatic and physical soil variables or Fe, manganese (Mn), Cr, Ni, or Co content. Research on associated mafic and serpentinic soils in Maryland found extractable Ca/Mg ratios <0.10 and has a 98% probability of having formed from serpentine rather than from non-serpentine mafic parent materials.^[27] A Ca/Mg ratio <0.1 is used in the Magne-sic Great Group in the Australian Soil Classification System,^[28] but this ratio has to be incorporated into U.S. *Soil Taxonomy*.^[4,12]

High levels of Ni, Cr, and Co in some serpentinic soils have been linked to potential toxicity and/or antagonistic effects on other ions.^[9,23,29,30] Many studies have focused on Ni because of characteristic symptoms exhibited in plants growing on these soils.^[31] Ni substitutes for Mg in the serpentine crystal structure^[5] and is generally considered a more likely candidate for chemical toxicity because of its greater availability to plants than Cr and because of its greater abundance than Cr and Co, respectively.^[9] A study of serpentinic subalpine soils in Switzerland^[32] showed exchangeable $\text{Ni} > 0.1$ mmol/kg and exchangeable $\text{Cr} < 0.01$ mmol/kg. Most Cr was located in the structure of primary minerals (e.g., garnet, pyroxene, and spinels) and secondary minerals (Fe oxides). However, research on serpentinic soils in northwest Spain^[33] showed the foliage of some agricultural crops (e.g., sugarbeet, cabbage, and pasture) accumulates considerable Ni and Cr, despite low to moderate ethylenediaminetetraacetic acid-extractable amounts of both elements in the soils.

Field observations of ultramafic rocks often indicate widely differing weathering rates within the same general area.^[19,34] Elemental distribution (e.g., Mg, Ni, and Cr) and mineral alteration have been used as indicators of pedogenesis of ultramafic soils. As Mg leaches from the soil profile, elements such as Fe, Al, and silicon (Si) may concurrently be enriched.^[35] Ni, Co, and Mn generally increase with depth, except in the driest environments, whereas Cr is less mobile, tending to accumulate in upper parts of the soil profile, associated

with high levels of Fe oxides.^[32,36] Mineral transformations during pedogenesis of ultramafic soils are determined by factors such as topography and the climate. Smectite formation is promoted on wetter landscape positions and concentrated in soils on footslope and toe slopes as opposed to higher landscape positions.^[37,38] Similarly, the formation of Mg-rich smectite in poorly drained landscape positions has been attributed to neo-synthesis in an environment with high pH and accumulations of Mg and Si by subsurface flow of soil water.^[39] A pedogenic study of ultramafic soils along a climatic gradient in southwestern British Columbia shows nearly equal proportions of hydroxyl-interlayered vermiculite/smectite, chlorite, serpentine, and talc in the wet-cool zones, whereas serpentine, talc, chlorite, and only small amounts of smectite are found in the dry-cool zones.^[40] In a soil catena on serpentine in northwestern Italy, serpentine minerals appear to weather to low charge vermiculite in upper and drier horizons or to smectite in poorly drained conditions.^[41]

MANAGEMENT STRATEGIES

Timber on some serpentine soils^[42] (Curry County Oregon Soil Survey; U.S. Department of Agriculture, Natural Resources Conservation Service, unpublished) is considered impractical to manage because of low productivity and sparse stands, owing to inherent infertility and nutrient imbalance, complicated by susceptibility to landsliding and erosion. However, other studies^[14] suggest that on some old peridotite terrains, which are at least partially serpentinized, the timber may be as productive as timber on some soils on more silicic (less mafic) rocks in the area. Little information exists on agricultural soils, and the effect of their management on heavy metal availability,^[31,33] owing partly to the limited agronomic use of these soils.^[9,33] Some studies^[33] have shown, however, that despite moderate amounts of soil-extractable heavy metals, crops such as cabbage (*Brassica oleracea*), barley (*Secale cereale*), and pasture grown on managed serpentinic soils can accumulate significant amounts of Cr and Ni, with sugarbeet (*Beta vulgaris*) accumulating dangerous levels of Ni. Some management strategies that help reduce plant availability of Ni and Cr include liming and additions of organic matter, other plant nutrients (N, K, and Ca), and phosphate fertilizers.^[43–46]

CONCLUSION

Serpentinic soils develop from metamorphosed (serpentinized) ultramafic rocks. Primary mineral constituents are antigorite, chrysotile, and lizardite. While serpentinic soils compose only small, widespread parts of the earth's surface, they have unique properties that dramatically alter productivity and land use compared to

other soils. Elemental deficiencies (e.g., N, P, and K), imbalances of major elements (e.g., Ca/Mg), and enrichments of Ni, Cr, and Co influence plant survival and productivity. Landscapes with these soils typically have sparse, low productive stands of shrubs and other diverse flora, and are not practical for agricultural production. These unique properties have warranted inclusion of specific taxonomic categories on soil classification systems and continue to generate interest of research scientists worldwide.

REFERENCES

1. Bates, R.L.; Jackson, J.A., Eds. *Glossary of Geology*, 3rd Ed.; American Geological Institute: Alexandria, 1987; 788 pp.
2. Skinner, H.C.W.; Ross, M.; Frondel, C. *Asbestos and Other Fibrous Minerals*; Oxford University Press: New York, 1988; 204 pp.
3. Schreier, H.E.; Omueti, J.A.; Lavkulich, L.M. Weathering processes of asbestos-rich serpentinitic sediments. *Soil Sci. Am. J.* **1987**, *51*, 993–999.
4. Soil Survey Staff. *Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys*; United States Department of Agriculture, Natural Resources Conservation Service Government Printing Office: Washington, DC, 1999; 869 pp.
5. Deer, W.A.; Howie, R.A.; Zussman, J. Serpentine. In *An Introduction to the Rock-Forming Minerals*, 2nd Ed.; Longman Scientific and Technology: Essex, 1992; 344–352.
6. Kerr, P.F. *Optical Mineralogy*, 4th Ed.; McGraw-Hill Publ. Co.: New York, 1977.
7. Dixon, J.B. Kaolin and serpentine group minerals. In *Minerals in Soil Environment*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; SSSA: Madison, 1989; 467–525.
8. Karathanasis, A.D.; Harris, W.G. Quantitative thermal analysis of soil materials. In *Quantitative Methods in Soil Mineralogy*; Amonette, J.E., Zelazny, L.W., Eds.; SSSA Misc. Publ.: Madison, 1994; 360–411.
9. Brooks, R.R. *Serpentine and Its Vegetation. A Multidisciplinary Approach (Ecology, Phytogeography, and Physiology Series)*; Dioscorides Press: Portland, 1987; Vol. 1, 454.
10. Kruckeberg, A.R. Plant life on serpentinite and other ferromagnesian rocks in northwestern North America. *Syesis* **1969**, *2*, 14–114.
11. Whittaker, R.H. The ecology of serpentine soils: IV. The vegetation response to serpentine soils. *Ecology* **1954**, *35*, 275–288.
12. Whittaker, R.H. Vegetation of Siskiyou mountains, Oregon and California. *Ecol. Monogr.* **1960**, *30*, 279–338.
13. Alexander, E.B.; Wildman, W.E.; Lynn, W.C. Ultramafic (serpentinitic) mineralogy class. In *Mineral Classification of Soils*; Kittrick, J.A., Ed.; SSSA Spec. Publ. No. 16, ASA and SSSA: Madison, 1985; 135–146.
14. Alexander, E.B.; Adamson, C.; Graham, R.C.; Zinke, P.J. Soils and conifer forest productivity on serpentinitized peridotite of the trinity ophiolite, California. *Soil Sci.* **1989**, *148*, 412–423.
15. Jenny, H. *The Soil Resource: Origin and Behavior*; *Ecol Series No. 37*; Springer-Verlag: New York, 1980.
16. Kruckeberg, A.R. *California Serpentine: Flora, Vegetation, Geology, Soils, and Management Problems*; Univ Calif. Publ. in Botany, University of California Press: Berkeley, 1984; Vol. 78, 1–180.
17. Proctor, J. The plant ecology of serpentine: II. Plant response to serpentine soils. *J. Ecol.* **1971**, *59*, 397–410.
18. Alexander, E.B. Morphology, fertility, and classification of productive soils on serpentinitized peridotite in California (U.S.A.). *Geoderma* **1988**, *41*, 337–351.
19. Johnston, W.R.; Proctor, J. Ecological studies on the lime hill serpentine, Scotland. *Trans. Proc. Bot. Soc. Edinb.* **1979**, *43*, 145–150.
20. Lee, B.D.; Graham, R.C.; Laurent, T.E.; Amrhein, C.; Creasy, R.M. Spatial distributions of soil chemical conditions in a serpentinitic wetland and surrounding landscape. *Soil Sci. Soc. Am. J.* **2001**, *6*, 1183–1196.
21. Proctor, J.; Woodell, S.R.J. The ecology of serpentine soils. *Adv. Ecol. Res.* **1975**, *9*, 255–366.
22. Woodell, S.R.J.; Mooney, H.A.; Lewis, H. The adaptation to serpentine soils in California of the annual species *Linanthus Androsaceus* (Polemoniaceae). *Bull. Torrey Bot. Club.* **1975**, *102*, 232–238.
23. Woolhouse, H.W. Toxicity and tolerance in the responses of plant to metals. In *Encyclopedia of Plant Physiology*; Lange, O.L., Ed.; Springer: New York, 1983; Vol. 12C, 246–300.
24. Proctor, J. The plant ecology of serpentine, III. The influence of a high calcium/magnesium ratio and high nickel and chromium levels in some British and Swedish serpentine soils. *J. Ecol.* **1971**, *59*, 827–842.
25. Wallace, A.; Frolich, E.; Lunt, O.R. Calcium requirements of higher plants. *Nature* **1966**, *209*, 634.
26. Wyn Jones, R.G.; Lunt, O.R. The function of calcium in plants. *Bot. Rev.* **1967**, *33*, 407–426.
27. Rabenhorst, M.C.; Foss, J.E. Soil and geologic mapping over mafic and ultramafic parent materials in Maryland. *Soil Sci. Soc. Am.* **1981**, *45*, 1156–1160.
28. Isbell, R.F. *The Australian Soil Classification*; CSIRO Publ.: Collingwood, 1996; 143.
29. Johnston, W.R.; Proctor, J. Growth of serpentine and non-serpentine races of *Festuca Rubra* in solutions simulating the chemical conditions in a toxic serpentine soil. *J. Ecol.* **1981**, *69*, 855–869.
30. Wild, H. Indigenous plants and chromium in Rhodesia. *Kirkia* **1974**, *9*, 233–241.
31. Proctor, J.; Baker, A.J. The importance of nickel for plant growth in ultramafic (serpentine) soils. In *Toxic Metals in Soil-Plant Systems*; Ross, S.M., Ed.; John Wiley & Sons: New York, 1994; 417–432.
32. Gasser, U.G.; Juchler, S.J.; Hobson, W.A.; Sticher, H. The fate of chromium and nickel in subalpine soils derived from serpentinite. *Can. J. Soil Sci.* **1995**, *75*, 187–195.
33. Fernandez, S.; Seoane, S.; Merino, A. Plant heavy metal concentrations and soil biological properties agricultural serpentine soils. *Commun. Soil Sci. Plant Anal.* **1999**, *30* (13 and 14), 1867–1884.
34. Proctor, J.; Woodell, S.R.J. The plant ecology of serpentine soil. *Adv. Ecol. Res.* **1971**, *9*, 255–366.

35. Wildman, W.E.; Jackson, M.L.; Whittig, L.D. Iron-rich montmorillonite formation in soils derived from serpentinite. *Soil Soc. Am. J.* **1968**, *32*, 787–794.
36. Bulmer, C.E.; Lavkulich, L.M. Pedogenic and geochemical processes of ultramafic soils along a climatic gradient in southwestern British Columbia. *Can. J. Soil Sci.* **1994**, *74*, 165–177.
37. Istok, J.D.; Edward, M.E. Influence of soil moisture on smectite formation in soils derived from serpentinite. *Soil Sci. Soc. Am. J.* **1982**, *46*, 1106–1108.
38. Lee, B.D.; Sears, S.K.; Graham, R.C.; Amrhein, C.; Vali, H. Secondary mineral genesis from chlorite and serpentine in an ultramafic soil toposequence. *Soil Sci. Soc. Am. J.* **2003**, *67*, 1309–1317.
39. Senkayi, A.L. *Clay Mineralogy of Poorly Drained Soils Developing from Serpentinic Rocks*. Ph.D. Thesis; University of California: Davis, 1977.
40. Bulmer, C.E.; Lavkulich, L.M.; Shreier, H.E. Morphology, chemistry, and mineralogy of soils derived from serpentinite and tephra in southwestern British Columbia. *Soil Sci.* **1992**, *154*, 72–82.
41. Bonifacio, E.; Zanini, E.; Boero, V.; Franchini-Angela, M. Pedogenesis in a soil catena on serpentinite in northwestern Italy. *Geoderma* **1997**, *75*, 33–51.
42. Mason, R.S. *Geology, Mineral Resources and Rock Material of Curry County, Oregon, Bulletin 93*; Department of Geology and Mineral Industries: Oregon, 1977; 1–10.
43. Halstead, R.L.; Finn, B.J.; MacLean, A.J. Extractability of nickel added to soils and its concentration in plants. *Can. J. Soil Sci.* **1969**, *49*, 335–342.
44. Hunter, G.R.; Vergnano, O. Nickel toxicity in plants. *Ann. Appl. Biol.* **1952**, *39*, 279–284.
45. Crooke, W.M. Effect of soil reaction on uptake of nickel from a serpentine soil. *Soil Sci.* **1956**, *81*, 269–276.
46. Kukier, U.; Chaney, R.L. Amelioration of nickel phytotoxicity in muck and mineral soils. *J. Environ. Qual.* **2001**, *30*, 1949–1960.

Sesquioxides

Joey N. Shaw

Agronomy and Soils, Auburn University, Auburn, Alabama, U.S.A.

Larry T. West

National Soil Survey Center, U.S. Department of Agriculture—Natural Resources Conservation Service (USDA-NRCS) (Retired), Fayetteville, Arkansas, U.S.A.

Abstract

Oxides, oxyhydroxides, or hydroxides (referred to collectively as oxides) of iron (Fe), aluminum (Al), and manganese (Mn) are termed sesquioxides. Oxide minerals largely affect soil chemical and physical properties, morphology, and classification. These minerals occur in soils as cations in fourfold (tetrahedral) and eightfold (octahedral) coordination with oxygen and hydroxyl groups. Substantial ionic substitution occurs in these minerals mostly due to the ability of many cations to reside in octahedral coordination. Most oxide minerals in soils form pedogenically after the release of these elements from primary mineral weathering, but a few Fe (e.g., magnetite) and Al (e.g., corundum) oxide minerals are predominantly of primary origin (inherited from the parent material). Some oxide minerals require an oxidizing (well-drained) soil environment for stability (e.g., most Fe and Mn oxides), while others do not (Al oxides). Virtually all soils possess sesquioxide minerals, with the possible exception of extremely young soils, extremely calcareous soils, or soils that possess seasonal high water tables for extended periods. In seasonally wet soils, the transformation and translocation of the redox sensitive Fe and Mn result in soil wetness (redoximorphic) features that are used to evaluate depths to seasonally high water tables. Fe oxides in particular, and Mn oxides in some instances, impart much color to soils. Substantial coulombic and specific adsorption of organic molecules, cations, and anions occur on sesquioxides.

INTRODUCTION

Research suggests most sesquioxides possess no permanent charge but develop surface charge dependent on the pH and ionic strength of the soil environment. Readers are encouraged to consult these following sources for additional information: Hsu,^[1] Schwertmann and Taylor,^[2] and McKenzie.^[3]

IRON (Fe) OXIDES

Fe Oxide Minerals

Most Fe oxides in soils form by pedogenic processes. In an oxidizing environment, Fe released from the weathering of primary minerals such as hornblende and biotite coordinates octahedrally with oxygens and/or OHs to form Fe oxide minerals. Goethite (α -FeOOH) and hematite (α -Fe₂O₃) are common hexagonal close-packed Fe oxides. Because Al³⁺ is similar in size to Fe³⁺, substantial aluminum (Al) substitution occurs in both of these minerals.^[2] The cubic close-packed lepidocrocite (γ -FeOOH), a polymorph of goethite found in periodically saturated soils, tolerates little Al substitution within its structure. Ferrihydrite, a para-crystalline Fe oxide, along with amorphous Fe

forms, is generically called “soil Fe.” No consensus on the formula and structure of ferrihydrite and other “soil Fe” phases exist, but it is believed these forms control Fe activities in soil solutions. Ferrihydrite is also considered a necessary precursor to hematite formation. Magnetite (Fe₃O₄) is a primary mineral, and most evidence suggests that it oxidizes to maghemite (Fe₂O₃) in soils. This oxidation may be facilitated in near-surface environments by fire,^[4] although most Fe oxides that possess some degree of magnetism, magnetite, and maghemite exhibit the strongest magnetic properties.

Properties of Fe Oxides

The amount and type of Fe oxide dictate subsurface soil colors because their fine-grained nature and charge properties allow them to coat other soil minerals. Red soil colors are due to the presence of hematite, yellow to brown colors indicate the presence of goethite, and orange colors arise from the presence of lepidocrocite (Table 1; Fig. 1A). Fe oxide color is also dependent on crystal size, degree of crystallinity, and the amount of ionic substitution (mostly Al) within the mineral structure.^[2] Soil colors are often the result of mixtures of Fe oxide minerals, and certain Fe oxides (e.g., hematite)

Table 1 General properties of oxide minerals commonly found in soils.

Element	Common oxide form in soil	Color	Soil environments	Major effects on soil properties
Fe	Goethite, α -FeOOH	Yellow to brown	Oxidizing, many environments	Color, aggregation, sorption of cations, anions, and organics
Fe	Hematite, α -Fe ₂ O ₃	Red to purple	Advanced weathering, oxidizing, warm and humid	Color, aggregation, sorption of cations, anions, and organics
Fe	Ferrihydrite, Fe ₅ HO ₈ · 4H ₂ O	Reddish-brown	Wet, rapid Fe release and oxidation, young soils	Color, aggregation, sorption of cations, anions, and organics
Al	Gibbsite, Al(OH) ₃	Colorless	Advanced weathering or rapidly leached aluminous environments	Aggregation, sorption of anions and cations
Mn	Many minerals	Black to dark brown	Oxidizing	Color, sorption of anions and cations

Source: From Schwertmann.^[5]

possess greater pigmenting ability than others.^[2] Many techniques have been developed that estimate soil Fe oxide content from either Munsell color notation or spectral reflectance values acquired via remote sensing.

Soil color is also used as a differentiating characteristic at many levels of soil classification schemes, including Soil Taxonomy. For example, Rhodic subgroups of Ultisols and Alfisols possess dark red hematite-enriched

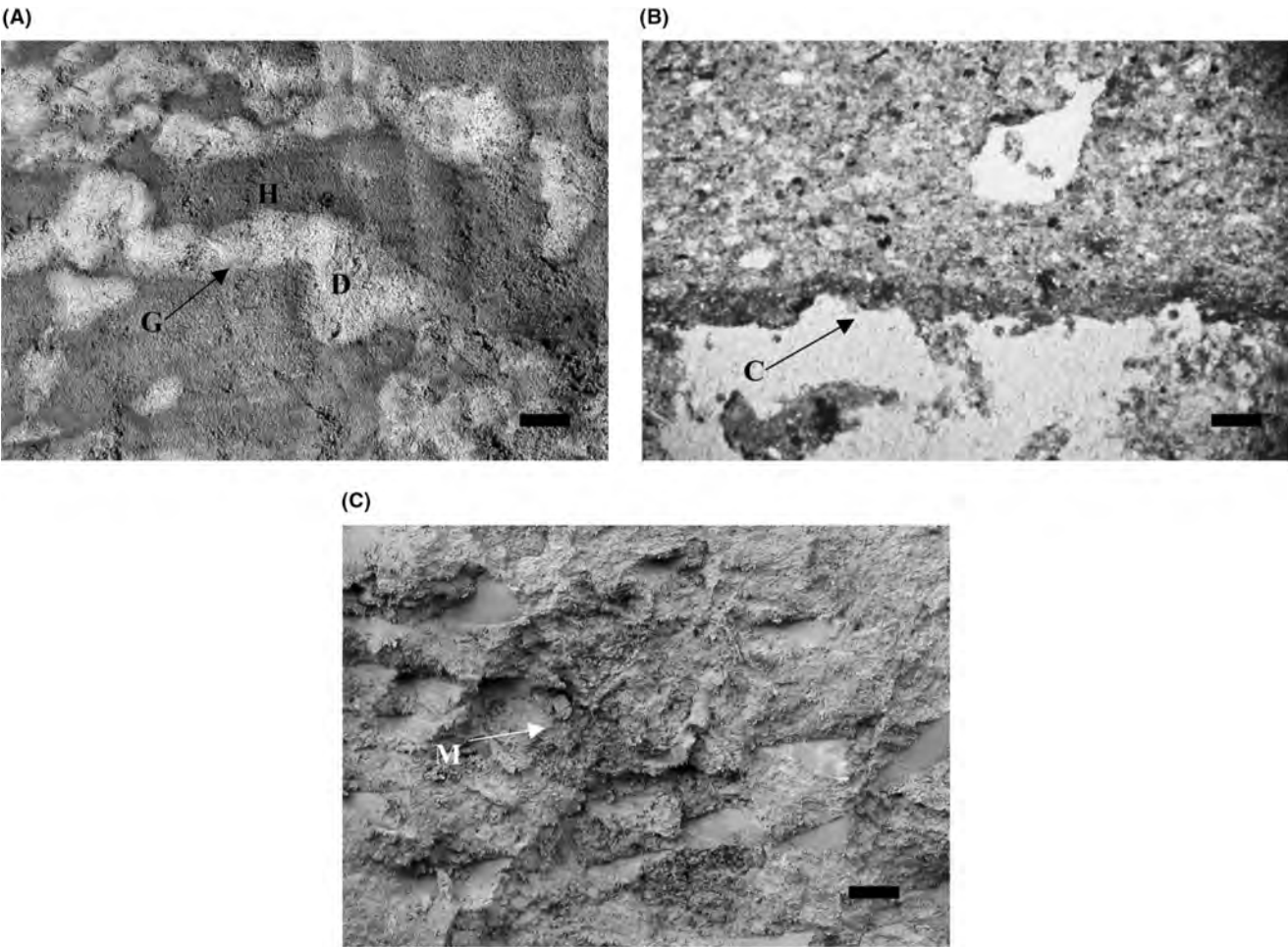


Fig. 1 (A) Hematite (H), goethite (G), and redox depletions (D) in a plinthic Paleudult; bar = 20 mm. (B) Photomicrograph (plane polarized light) of an Fe oxide concentration (C) on a pore wall of a Bw horizon in a Fluventic Dystrudepts; bar = 0.2 mm. (C) Mn oxide (M) concentrations in the Bt horizon of an Oxyaquic Hapludalf; bar = 20 mm.

argillic horizons, while wet soil taxa possess features related to Fe transformations in seasonally saturated soils.

Wet Soil Features

A discussion of Fe [and manganese (Mn)] oxides is not complete without a reference to biochemical reduction effects on their solubility. Fe and Mn possess oxidation states that are dependent on the redox potential of the soil environment, which is largely controlled by soil saturation and microbial activity.^[6] When soils become saturated for significant time periods, aerobic microbes deplete the available oxygen gas (O_2) and cause oxidized soil components such as Fe^{3+} and Mn^{4+} in oxide minerals to undergo chemical reduction induced by anaerobic microbial respiration and relatively high labile organic carbon content. As Fe and Mn are reduced, the oxide minerals bearing these elements become soluble. These elements can re-oxidize as water tables recede, precipitate as reduced forms in sulfide minerals, be adsorbed onto exchange sites, or move from one part of the soil to another by mass flow and diffusion. Reductive dissolution and loss of Fe and Mn oxide grain coatings exposes the silicate minerals that are commonly gray to white, resulting in the formation of redox depletions (see redox depletion in Fig. 1A). The oxidation of Fe and Mn in another part of the soil results in Fe and Mn oxide concentrations and a general reddening to blackening of the area. These depletions and concentrations, or redoximorphic features, are commonly used to interpret the depth to a seasonally high water table in soils (Fig. 1A and B).

Chemical and Physical Properties of Fe Oxides

Fe oxides greatly influence soil physical and chemical properties due to their charge characteristics, fine-particle size, and high surface area. Generally, Fe oxides possess surfaces that protonate (gain H^+), deprotonate (lose H^+), and develop charge based on soil solution pH and ionic strength. The pH at which the negative charge equals the positive charge has been termed the point of zero charge (PZC). Soil pH levels below this point result in (+) charged oxide surfaces, while the pH levels above the PZC result in (−) charged surfaces (Fig. 2A).^[7] Fe and Al oxides tend to have higher PZCs than Mn oxides. Because the PZC of most soil Fe oxides occurs above pH 7, many soils with $pH < 7$ have Fe oxides with (+) charged surfaces. Therefore, many highly weathered soils with high Fe (and Al) oxide content have substantial ability to sorb anions on their surfaces and possess appreciable anion exchange capacity (AEC). Phosphorus (P) and other anions are also sorbed through specific mechanisms involving coordination of the anion to the Fe within the oxide mineral (Fig. 2B). Because of this, P deficiency in crop and forest

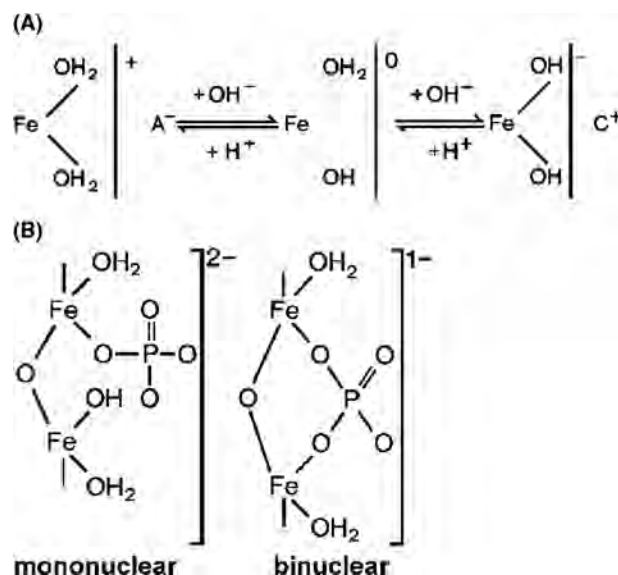


Fig. 2 Proposed mechanisms for the development of pH-dependent charge (A) and specific adsorption of oxyanions (PO_4); (B) by Fe oxides.

Source: From Schwertmann & Taylor.^[2]

production is common on soils possessing high Fe (and Al) oxide quantities.

Fe oxides also play a large role in soil aggregation. Because many Fe oxide surfaces possess a (+) charge, they tend to facilitate aggregation by forming bridges between (−) charged phyllosilicate minerals. Poorly crystalline Fe oxides may promote increased aggregation compared to more crystalline forms.^[2]

Al OXIDES

Al Oxide Minerals

Gibbsite [$Al(OH)_3$] is the most common Al oxide found in soils^[1] (Table 1). Corundum (Al_2O_3), boehmite ($AlOOH$), and diaspor ($AlOOH$) are primary minerals that are not typically found in soils but are common in bauxite ore deposits.^[1] Unlike Fe oxides, poorly crystalline phases of Al oxide have not been readily identified in soils but most likely exist. Al oxides are essentially colorless, and their visual identification in soils is difficult.

Gibbsite Formation

Gibbsite is a common constituent of tropical and subtropical soils, and its presence generally suggests an advanced stage of soil development. Gibbsite formation is mostly due to the desilication [weathering and leaching of silicon (Si)] of kaolinite or other aluminosilicate that occurs over long-term weathering under humid or extreme leaching

environments. However, in highly leaching aluminous environments, gibbsite can form rapidly.^[8] Because it can form rapidly as well as exist in transported parent materials, its presence does not always indicate an advanced degree of soil development.

Al Oxide Properties

Similar to Fe oxides, gibbsite develops surface charge based on soil solution pH and ionic strength. Most evidence suggests that Al oxides have a PZC ~pH 8.^[1] Thus, gibbsite is (+) charged in most soils and readily sorbs anions, cations, and organic molecules by both specific and coulombic attraction. The specific adsorption of ions (similar to Fe oxides) also occurs with gibbsite. Similar to Fe oxides, gibbsite contributes to soil aggregation by bridging (–) charged phyllosilicate surfaces.

Mn OXIDES

Mn Oxide Minerals

Mn ions are similar in size to Fe; thus, Mn is found in several primary minerals. However, Mn oxides are not as abundant as Fe oxides because lower quantities of Mn typically exist in parent materials. Because Mn is essential to plant growth, but high concentrations found in wet or acid soils are toxic, an understanding of its mineral forms is critical for crop management.^[3] McDaniel and Buol^[9] determined that approximately half of the Mn in soils developed from a gneiss parent material existed in oxide mineral form, while the other half was either organically bound or resided on exchange sites. Similar to Fe, most pedogenic Mn minerals exist as oxides with Mn in sixfold coordination with oxygens or OHs. Because Mn is redox sensitive, morphological features resulting from its transformations are also used as a soil wetness indicator. Mn oxides mostly exist as black, hardened concentrations, called concretions, that violently effervesce upon treatment with H₂O₂. They also commonly exist as black coatings on ped faces (Table 1; Fig. 1C).

Mn occurs in several oxidation states (Mn²⁺, Mn³⁺, and Mn⁴⁺) within oxide minerals of varying crystallinity.^[9] Mn oxides are classified as either layer or tunnel type, the difference depending on the presence or absence of tunnels within the mineral structure.^[3] Birnessite [(Na_{0.7}Ca_{0.3})Mn₇O₁₄ · 16H₂O], vernadite (MnO₂), and lithiophorite [(Al,Li)MnO₂(OH)₂] are layer-type Mn oxides that have been identified in soils, while hollandite (Ba₂Mn₈O₁₆) is a tunnel-type soil Mn oxide.^[3] Other primary Mn oxides sometimes found in soils include the tunnel-type pyrolusite (MnO₂) and todorokite [(Na,Ca,K,Ba,Mn²⁺)₂Mn₄O₁₂ · 3H₂O].^[3]

Mn Oxide Properties

Most evidence suggests that Mn oxides possess PZCs < pH 4.5.^[3] This renders most Mn oxide surfaces (–) charged except under extremely acid conditions. Cations, particularly heavy metals, are adsorbed onto Mn oxide surfaces by both specific mechanisms and coulombic attractive forces.^[3] For example, in a study of metal-contaminated soils in Colorado, zinc, cadmium, and lead were preferentially adsorbed onto Mn and Fe oxide surfaces, while copper was predominantly associated with the organic matter fraction.^[10] Mn in oxide minerals is reduced to the soluble Mn²⁺ in saturated environments by similar mechanisms as described in the Fe oxide section earlier. Soluble Mn²⁺ is either leached, sorbed onto organic or clay surfaces, or reoxidized to black concentrations.

GENERAL RELATIONSHIPS BETWEEN OXIDE MINERALS

The presence of high quantities of sesquioxides suggests an advanced degree of soil weathering. Although many soil orders possess these minerals, sesquioxides are concentrated in Oxisols and Ultisols. Because of the properties Fe and Al oxides impart to soils, these soils are often red, possess significant AEC, and are strongly aggregated. P deficiency is also common.

Table 2 Oxide dominated soil mineralogical families in soil taxonomy and their occurrence.

Mineralogical family	Criteria	Soil orders where commonly found
Ferritic	>40% Fe ₂ O ₃ in the < 2 mm fraction	Oxisols, Ultisols, Inceptisols
Gibbsitic	>40% gibbsite in the < 2 mm fraction	Oxisols, Ultisols
Sesquic	18–40% Fe ₂ O ₃ and 18–40% gibbsite in the < 2 mm fraction	Oxisols
Ferruginous	18–40% Fe ₂ O ₃ in the < 2 mm fraction	Oxisols, Ultisols
Allitic	18–40% gibbsite in the < 2 mm fraction	Oxisols, Ultisols (none currently correlated)
Ferrihydritic	(8 × Oxalate extractable Si + 2 × oxalate extractable Fe) > 5	Andisols, Spodosols
Parasesquic	>10% Fe ₂ O ₃ + gibbsite in the < 2 mm fraction	Ultisols, Alfisols, Mollisols, Aridisols, Spodosols, Inceptisols

Source: From Soil Survey Staff.^[11]

Oxide minerals can also act as cementing agents in soils. Plinthite, a common constituent of southeastern U.S. coastal plain soils, is a hardened mass of mostly Fe oxides that also typically contain quartz, phyllosilicates, and gibbsite. In large quantities, plinthite can have a major effect on soil properties and cause significant reductions in rooting depth and hydraulic conductivity.

Many soil mineralogy classes defined by oxide mineral quantities exist in Soil Taxonomy due to the influence these minerals have on soil properties.^[11] Most of the 12 soil orders have at least one soil series identified that has an oxide-dominated mineralogical class (Table 2).

REFERENCES

1. Hsu, P.H. Aluminum oxides and oxyhydroxides. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America Book Series No. 1; Soil Science Society of America: Madison, 1989; Vol. 1, 331–378.
2. Schwertmann, U.; Taylor, R.M. Iron oxides. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America Book Series No. 1; Soil Science Society of America: Madison, 1989; 379–438.
3. McKenzie, R.M. Manganese oxides and hydroxides. *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed., Soil Science Society of America Book Series No. 1; Soil Science Society of America: Madison, 1989; 439–466.
4. Schwertmann, U. Occurrence and formation of iron oxides in various pedoenvironments. In *Iron in Soils and Clay Minerals*; Stucki, J.W., Goodman, B.A., Schwertmann, U.D., Eds.; Reidel Publishers: Dordrecht, 1988; 267–308.
5. Schwertmann, U. Relations between iron oxides, soil color, and soil formation. In *Soil Color*; Bigham, J.M., Ciolkosz, E.J., Eds.; Soil Science Society of America Special Publication No. 31; Soil Science Society of America: Madison, 1993; 51–70.
6. Bohn, H.L.; McNeal, B.L.; O'Connor, G.A. *Soil Chemistry*, 2nd Ed.; Wiley: New York, 1985.
7. McBride, M.B. Surface chemistry of soil minerals. In *Minerals in Soil Environments*; Dixon, J.B., Weed, S.B., Eds.; 2nd Ed.; Soil Science Society of America Book Series No. 1; Soil Science Society of America: Madison, 1989; 35–88.
8. Norfleet, M.L.; Smith, B.R. Weathering and mineralogical classification of selected soils in the blue ridge mountains of south carolina. *Soil Sci. Soc. Am. J.* **1989**, *53*, 1771–1778.
9. McDaniel, P.A.; Buol, S.W. Manganese distributions in acid soils of the north carolina piedmont. *Soil Sci. Soc. Am. J.* **1991**, *55*, 152–158.
10. Levy, D.B.; Barbarick, K.A.; Siemer, E.G.; Sommers, L.E. Distribution and partitioning of trace metals in contaminated soils near leadville, colorado. *J. Environ. Qual.* **1992**, *21*, 185–195.
11. Soil Survey Staff, Soil Taxonomy. *A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; Handbook No. 436; USDA–NRCS: Washington, 1999.

Shrinkage

Des McGarry

Natural Resource Sciences, Queensland Department of Natural Resources and Mines, Indooroopilly, Queensland, Australia

Don F. Yule

Department of Natural Resources, Rural Water Use Efficiency Initiative, Indooroopilly, Queensland, Australia

Abstract

Soil shrinkage is one of the intrinsic characteristics of any soil, providing a unique insight into a soil's nature, specifically the response to change in soil water content. The nature and degree of soil shrinkage is determined by several inter-related soil properties with clay percentage, clay type, and the ratios of saturating cations predominating. Several tests are available to quantify and express this characteristic with each providing different aspects of a soil's shrinkage nature and potential. Soil classification systems are utilizing the characterization of soil shrinkage to aid in the definition of soil classes, recognizing the unique insight this provides into a soil's inherent nature.

INTRODUCTION

Soil shrinkage is the process of soil material contracting to a lesser volume, with the loss of water on drying. In agriculture, soil shrinkage is focused on two topics: the characterization and classification of soils containing clay, as the degree of shrinkage (and swelling) largely determines the potential for soil structure development,^[1] and the redevelopment of soil structure (soil resilience) compacted by wheels and implements.^[2] In engineering, soil shrinkage is a key determinant of material stability for both intrinsic and stress states associated with structures, such as foundations, pipelines, and roads, and soil conservation structures.^[3]

Negative aspects of soil shrinkage include overall volume change and uneven shrinkage of soil supporting loads in engineering and in agriculture root breakage caused by soil cracking and the super dehydration of soils from air thermals in cracks.^[4] Positive, agricultural aspects of soil shrinkage include soil cracking, providing aeration and facilitating water entry, and the formation of naturally, aggregated, fine seedbeds with wetting and drying cycles.^[5]

FACTORS AFFECTING SOIL SHRINKAGE

Many factors, singly and in combination, affect the degree of shrinkage in a soil, including soil fabric, mineralogy, saturating cation, electrolyte concentration and speciation, clay content, surface area, antecedent water content, frequency of wet-dry cycles, confining pressures, and soil

thickness.^[6] Shrinkage is positively correlated with greater total and fine clay contents, and particularly greater proportions of clay composed of expanding 2:1 clay minerals, such as the smectites.^[6] Antecedent water content is positively correlated with clay percent, i.e., the proportion of expansible clay minerals and surface area. Hence, it is also positively correlated with total shrinkage. Particle arrangement also affects shrinkage characteristics where random (edge to face) arrangements can trap larger volumes of water (or air) than stratified (face to face) arrangements, and hence have greater capacity for shrinkage.^[4] Shrinkage is reduced by factors such as increased proportions of sand relative to clay, as the interlocking of sand and silt grains increases the frictional forces,^[6] increased organic matter, as the organic matter dilutes the shrinkage potential of the mineral components,^[7] and, as cited by some studies, low cation exchange capacity and high exchangeable sodium levels.

MECHANISMS OF SOIL SHRINKAGE

Smectites, because of the charge deficiency, have strong ability to attract and adsorb water molecules onto clay domains.^[6] The mechanisms of soil shrinkage explain the loss of this water, there being two common representations (Figs. 1 and 2). The first involves the measurement of the volume of a soil sample as it slowly dries from a fully wet condition and provides a relationship between soil volume and water content (Fig. 1). This relationship is termed as a soil shrinkage characteristic curve^[8] and consists of four zones, though all four zones are not always

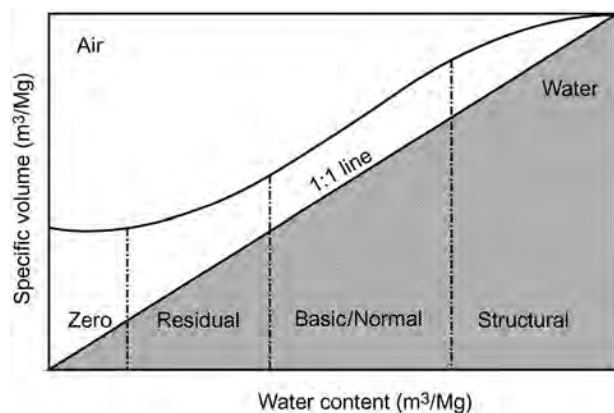


Fig. 1 A plot of a soil shrinkage characteristic curve showing the four shrinkage zones and the theoretical (1:1) shrinkage line.

present: structural, basic or normal, residual, and zero shrinkage.^[9] In structural shrinkage, volume change is less than the volume of water lost, arising from the drainage of large pores that do not completely close with water loss. Basic shrinkage, traditionally termed normal shrinkage, is commonly depicted as equal volume and water loss. This zone accounts for most shrinkage in agricultural soils, generally between soil water potentials of -33 to <-1500 kPa.^[10] Water is lost from space between clay domains (lumps of individual clay particles), and the random orientation of these domains cause shrinkage to be equidimensional, thereby resulting in crack formation. Residual shrinkage, like structural shrinkage, has volume change less than the volume of water lost. Water is lost from stable pores between clay particles that are unable to completely close as the particles retain the clay matrix structure. Zero shrinkage, recognized by some authors, has no volume change with water loss.^[11]

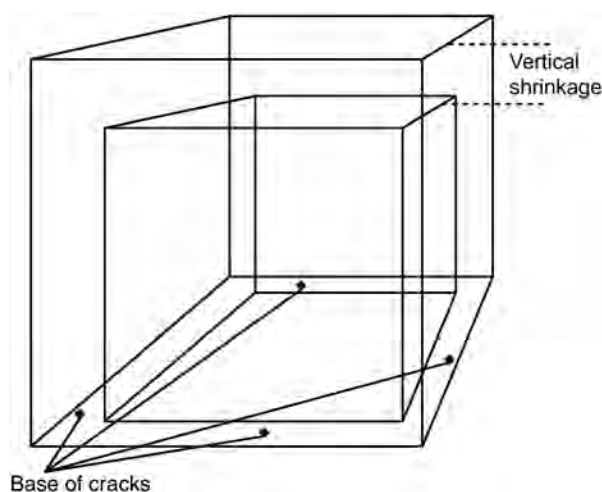


Fig. 2 Shrinkage of a soil cube showing equidimensional shrinkage, resulting in soil cracking.

On the soil shrinkage characteristic curve, the location of experimental data is referred to a 1:1 line, termed as the saturation line as it represents the shrinkage of a structureless (air-less) clay paste (Fig. 1). With reference to this theoretical line, the relative proportions of the specific volumes of the solid, liquid, and air for an experimental plotted value may be obtained.

The second representation of soil shrinkage aims to represent “whole soil” shrinkage behavior and is generally applicable to the shrinkage of large, intact soil units—either soil cores in the laboratory or field soil (Fig. 2). A measurable field result is a reduction in the vertical dimension, i.e., the soil surface moves down, and a visible field result is the formation of large cracks that form laterally at irregular intervals (Fig. 2). Importantly, the soil block between the cracks contains all the structural and residual porosity, and the cracks (including the “crack” above the soil surface) contain only the basic (or normal) shrinkage porosity.

THE MEASUREMENT OF SOIL SHRINKAGE

Soils may be characterized by measurement of total or partial shrinkage, of either linear dimensions or volume, as an initially wet sample loses water. There is no unique measure of soil shrinkage; different techniques provide different aspects of a soil’s shrinkage nature and potential.

Measurements may be made on soil in different states: 1) remolded samples of finely ground material; 2) natural soil aggregates; 3) intact blocks or cores; and 4) volume change in the field.^[12] A vital consideration is that for each of the tests, alteration of the start and end points in terms of water content will provide different results, hence negating the potential for soil comparisons.

Remolded Samples

The most common measure used in engineering applications is linear shrinkage: LS_{std} (based on the study by Standards Association of Australia^[13]). Determination is made of the percent decrease in length of a subsample of ground soil ($<425 \mu m$) that has been wet to its liquid limit, mixed to a homogeneous paste, placed in a mold, air-dried ($20^\circ C$) for 24 hours, and then oven-dried at $105^\circ C$. The LS_{std} of the specimen is

$$LS_{std} = \frac{L_s}{L} \quad (1)$$

where L is the length of the mold in millimeters and L_s is the longitudinal length of the oven-dry specimen in millimeter.

A modification to LS_{std} , toward providing a more agriculturally pertinent measure of soil shrinkage, is the modified linear shrinkage test: LS_{mod} ,^[14] where linear

shrinkage is determined on sieved (<2 mm), not remolded soil.

Natural Soil Aggregates

Several techniques are available, measuring soil volume by liquid displacement. Waterproofing coatings for the soil sample include paraffin wax, rubber balloons, and industrial resins. The most common test is the coefficient of linear extensibility (COLE),^[15] where determinations of volume change of intact soil clods are made between a moist and an oven-dry state. The test is integral in defining the Vertisol soil group.^[16]

Intact soil clods of size 0.05–0.08 m diameter are wetted on pressure plates to a specified matric potential (commonly, –33 kPa), before being coated in SARAN[®] resin. Two paired weighings are conducted—the first on the coated clod at the –33 kPa matric potential and then a second on the clod after oven-drying (105°C). The bulk density of the clod is calculated at each level of water content, and COLE is calculated as

$$COLE_{std} = 100[(BD_{OD}/BD_{33})^{1/3} - 1] \quad (2)$$

where BD_{OD} is the bulk density at oven-dry and BD_{33} is the bulk density at –33 kPa.

Several indices of soil shrinkage can be derived from the shrinkage of coated soil clods by collecting data from the whole range of the soil shrinkage characteristic curve, not one part alone as with COLE. McGarry^[17] specified 17 indices based on fitting straight lines to the shrinkage curve, and Braudeau et al.^[11] specified 13 parameters including consideration of the points of transition between the different shrinkage zones. Such indices have been used to quantify and rationalize differences in soils or treatments (e.g., see Pillai and McGarry^[12]).

Vertical Volume Change on Cores and in the Field

These measures of volume change are most commonly gained by measuring the physical dimensions of soil samples. Three types are recognized: volumetric and destructive (based on core bulk density measurements at various moisture states), non-destructive measures with dual gamma attenuation, and “1-D” measures based on monitoring surface elevation.^[18] Disadvantages with each are small cores may give non-representative sampling in cracking soils, access tubes can initiate preferential cracking of soil around the tube giving biased measures, and there are problems with relating a local surface elevation measurement with an average value of water contents collected around the gauge. New techniques employ electronic displacement transducers with block kriging interpolation to improve the precision of the water content estimates.^[18]

Table 1 Examples of classification schemes for the assessment of volume change potential based on linear shrinkage LS_{std} and COLE.

		Volume change rating
(a) LS_{std}		
Arid to semiarid climate	Humid climate	
0–0.05	0–0.12	Low
0.05–0.12	0.12–0.18	Moderate
>0.12	>0.18	High
(b) COLE		
<0.03		Low
0.03–0.06		Medium

Source: From Holland & Richards^[19] and Dasog, Acton, et al.^[20]

THE DEFINITION OF SOIL CLASSES FROM CUT-OFF VALUES OF SHRINKAGE TEST DATA

An important aim of determining shrinkage values is to define soil classes in terms of volume change activity for either engineering or soil (pedological) classifications. Engineering classes define broad categories of soil in terms of their predicted soil behavior for conservation earthworks or building foundations. Examples of the use of LS_{std} and COLE classes are given in Table 1. Holland and Richards^[19] included climatic considerations in their classification, recognizing LS_{std} becomes limiting at lower values in drier climates, as the potential for desiccation (hence shrinkage) is greater. In pedological studies (land survey and assessment), variable volume soils may be classed in terms of their potential for volume change. The aim is to provide a numeric, independent assessment of volume-change potential to replace or supplement visual assessments of crack intensity toward classifying Vertisols. For example, in South Africa, $LS_{std} > 0.12$ is required for vertic horizons and $LS_{std} < 0.12$ but > 0.08 for melanic horizons. There is also potential and a growing demand to use such classifications to express soil resiliency, i.e., a soil’s potential to self-repair soil compaction through wet-dry cycles.

CONCLUSION

“Soil shrinkage is one of the intrinsic characteristics of any soil, providing a unique insight into a soil’s nature, specifically the response to change in soil water content. The nature and degree of soil shrinkage are determined by several inter-related soil properties with clay percentage, clay type, and the ratios of saturating cations predominating. Several tests are available to quantify and express this characteristic with each providing different aspects of a soil’s shrinkage nature and potential. Soil classification systems

are utilizing the characterization of soil shrinkage to aid in the definition of soil classes, recognizing the unique insight that provides into a soil's inherent nature."

REFERENCES

1. Loveday, J. Field aspects of swelling and shrinking. In *Physical Aspects of Swelling Clay Soils*; McGarity, J.W., Hoult, E.H., So, H.B., Eds.; University of New England: Armidale, 1972; 45–52.
2. Pillai, U.; McGarry, D. Structure repair of a compacted vertisol with wet/dry cycles and rotation crops. *Soil Sci. Soc. Am. J.* **1999**, *63*, 201–210.
3. Aitchison, G.D. The quantitative definitions of the physical behaviour of expansive soils—An engineering viewpoint. In *Physical Aspects of Swelling Clay Soils*; McGarity, J.W., Hoult, E.H., So, H.B., Eds.; University of New England: Armidale, 1972; 63–74.
4. Yong, R.N.; Warkentin, B.P. Soil properties and behaviour. *Developments in Geotechnical Engineering Series*; Elsevier: Amsterdam, 1975; Vol. 5, 499 pp.
5. Pillai-McGarry, U.P.P.; Collis-George, N. Laboratory simulation of the surface morphology of self-mulching vertisols. I. Materials, methods and preliminary results. *Aust. J. Soil Res.* **1990**, *28*, 129–139.
6. Wilding, L.P.; Tessier, D. Genesis of vertisols: Shrink–swell phenomena. In *Vertisols: Their Distribution, Properties, Classification and Management*; Wilding, L.P., Puentes, R., Eds.; Technical Monograph No. 18, Soil Management Support Services: Texas A & M, 1988; 55–81.
7. McGarry, D. The structure and grain size distribution of vertisols. In *Vertisols and Technologies for Their Management, Developments in Soil Science*; Ahmad, N., Mermut, A., Eds.; Elsevier: Amsterdam, 1996; Vol. 24, 231–259.
8. Mitchell, A.R. Shrinkage terminology: Escape from “normalcy.” *Soil Sci. Soc. Am. Proc.* **1992**, *56*, 993–994.
9. Tariq, A.U.R.; Durnford, D.S. Analytical volume change model for swelling clay soils. *Soil Sci. Soc. Am. Proc.* **1993**, *57*, 1183–1187.
10. Yule, D.F.; Ritchie, J.T. Soil shrinkage relationships of Texas vertisols: I. Small cores. *Soil Sci. Soc. Am. Proc.* **1980**, *44*, 1285–1291.
11. Braudeau, E.; Costantini, J.M.; Bellier, G.; Colleuille, H. New device and method for soil shrinkage curve measurement and characterisation. *Soil Sci. Soc. Am. Proc.* **1999**, *63*, 525–535.
12. Warkentin, B.P. Soil shrinkage. In *Soil Sampling and Methods of Analysis*; Carter, M.R., Ed.; Lewis Publishers: London, 1993; 513–518.
13. Standards Association of Australia. AS 1289. C4.1—1977. Determination of the Linear Shrinkage of a Soil (Standard Method), 1977.
14. McKenzie, N.J.; Jacquier, D.J.; Ringrose-Voase, A.J. A rapid method for estimating soil shrinkage. *Aust. J. Soil Res.* **1994**, *32*, 931–938.
15. Grossman, R.B.; Brasher, B.R.; Franzmeier, D.P.; Walker, J.L. Linear extensibility as calculated from natural-clod bulk density measurements. *Soil Sci. Soc. Am. Proc.* **1968**, *32*, 570–573.
16. Soil Survey Staff. *Soil Survey Laboratory Methods and Procedures for Collecting Soil Samples*; Soil Survey Investigation Reports 1; Soil Conservation Service, USDA: Washington, 1972; 126 pp.
17. McGarry, D. Quantification of the effects of zero and mechanical tillage on a vertisol by using shrinkage curve indices. *Aust. J. Soil Res.* **1988**, *26*, 537–542.
18. Coquet, Y. *In situ* measurement of the vertical linear shrinkage curve of soils. *Soil Till. Res.* **1998**, *46*, 289–299.
19. Holland, J.E.; Richards, J. Road pavements as expansive clays. *Aust. Road Res.* **1982**, *12*, 173–179.
20. Dasog, G.S.; Acton, D.F.; Mermut, A.R.; De Jong, E. Shrink–swell potential and cracking in clay soils of Saskatchewan. *Can. J. Soil Sci.* **1988**, *68*, 251–260.

Silica: Pedogenic Accumulation

Katherine J. Kendrick

Western Earthquake Hazards, U.S. Geological Survey (USGS), Pasadena, California, U.S.A.

Abstract

Pedogenic silica is formed in arid, semiarid, and Mediterranean climates where there is adequate precipitation to mobilize the silica but not so much as to leach the silica out of the profile. In its most advanced form, it cements the soil fabric to form extremely hard horizons known as duripans or silcrete.

INTRODUCTION

Pedogenic silica is defined as silica that has precipitated in the soil environment, taking the form of opal-A, opal-CT, or microcrystalline quartz. Opaline silica, which is amorphous to poorly crystalline, is the most common form of pedogenic silica.^[1,2] Pedogenic silica is formed in arid, semiarid, and Mediterranean climates where there is adequate precipitation to mobilize the silica but not so much as to leach the silica out of the profile. Pedogenic silica accumulation has been reported throughout the Western United States, Australia, Italy, South Africa, and New Zealand.^[3] In its most advanced form, it cements the soil fabric to form extremely hard horizons known as duripans or silcrete.

SOURCES OF PEDOGENIC SILICA

Pedogenic silica is derived from two main sources. The weathering and hydrolysis of primary silicate minerals provide for the slow release of silica into solution as silicic acid. Easily weathered silicates, such as olivine and calcic plagioclase, are particularly likely to contribute silica into solution. The second source is from the weathering of primary amorphous silica, including volcanic glass and biogenic silica (e.g., phytoliths and diatoms). Such amorphous silica is the most soluble form of silica found in the soil environment. Weathering of glassy volcanic products, including ash and tephra, to release silicic acid into solution occurs readily in soils.

DISSOLUTION OF SILICA

Factors controlling the dissolution of silica include soil solution pH, the presence of organic matter, the particle size of the material, and the presence of coatings on grains. The solubility of silica is moderate throughout the pH range of most soils but is particularly high at pH values ≥ 9 , as found in sodium carbonate soil systems. Organic matter

enhances the dissolution of silica, including quartz, and impedes precipitation.^[3] The dissolution of quartz has been shown to be highest in the root zone due to organic complexation of monosilicic acid.^[4] On the other hand, organic matter may actually form a coating on opal, impeding its dissolution.^[5] Because of surface area effects, smaller particles are more susceptible to dissolution. This phenomenon accounts for the fact that quartz does not persist in the clay fraction of most soils.^[6] Aluminum and iron (Fe) oxyhydroxides can form insoluble coatings on grains, thereby decreasing rates of dissolution from those grains.^[3]

MOVEMENT AND PRECIPITATION OF SILICA

Silica is moved in solution through the soil as silicic acid. Incomplete leaching allows for the precipitation of this silicic acid in the pedogenic zone. During drying induced by evapotranspiration, silicon dioxide is adsorbed onto surfaces, forming amorphous opaline silica. This process led to the silica cementation of south-facing terrace edges on the Central California coast.^[7] Alternatively, *in situ* alteration of eolian dust to opal-A, rather than precipitation from silica in soil solution, has been proposed for duripans in Idaho.^[8] The eolian dust comprised volcanic glass and feldspars.

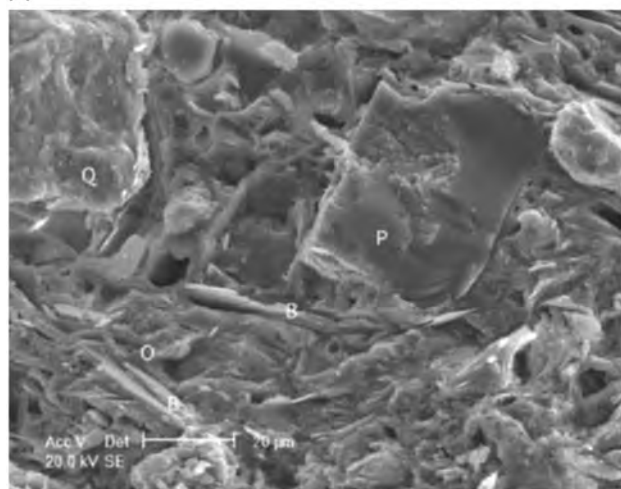
Conditions favoring the precipitation of silica from solution include pH values less than 7, a high available surface area within the soil fabric, and high ionic strength of the soil solution. Although a high available surface area favors precipitation, silica cementation is most common in medium-textured parent materials with abundant skeletal grains. The grain-to-grain cementation is an important component in the formation of a duripan.^[6]

Aluminum and Fe oxyhydroxides specifically adsorb soluble silica. Since these compounds are nearly ubiquitous in soils, often as coatings on other grains, they form a significant sink for the precipitation of silica on surfaces.^[3]

SILICA MORPHOLOGY

Stages of silica cementation have been defined for coarse grained deposits in arid climatic regimes.^[9] The first stage is precipitation of silica on the undersides of clasts. These are termed *pendants* and are equivalent to *opal beards* in Australian soil descriptions.^[10] Dramatic examples of opal pendants have been described in some Central Californian soils.^[11] The second stage is precipitation within the matrix, where silica forms bridging contacts between grains. The third stage is silica precipitation on all sides of the clasts, and the final stage is a laminar cap above a plugged matrix. Apart from silica precipitation on gravels, silica can cement the finer matrix materials into nodules termed *durinodes*.

(A)



(B)

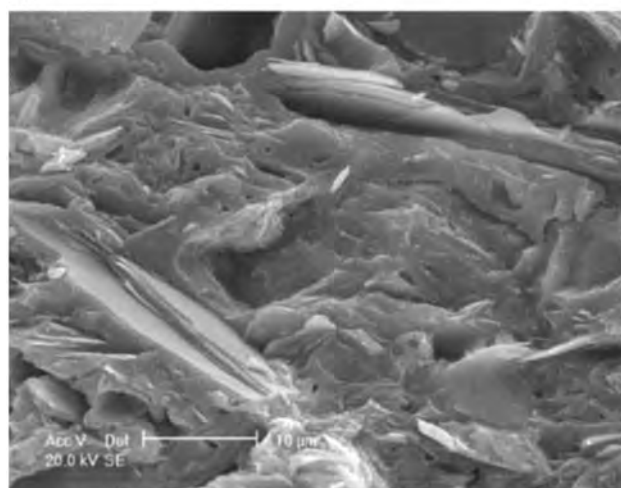


Fig. 1 Scanning electron micrograph of soil fabric in a duripan from southern California. (A) Primary mineral grains (P: plagioclase; B: biotite; and Q: quartz) cemented by opaline silica (O: opaline silica). (B) Close-up view of the lower left portion of Fig. 1A, showing biotite grains embedded in opaline silica.

Source: Photo by Dr. Krassimir Bozhilov, Central Facility for Advanced Microscopy and Microanalysis, UC Riverside.

Duripans are defined as soil horizons that are irreversibly cemented by various forms of silica such that they do not slake after soaking in water or hydrochloric acid.^[12] Duripans take two general forms. In Mediterranean climates, duripans often have prismatic structure and pedogenic calcite is minimal or absent.^[1] Small opal flocs within the matrix of the prisms are the precursors to durinodes. Further development of the duripan yields extensive grain-to-grain cementation in the soil matrix (Fig. 1) and silica coatings on the tops and upper sides of the prisms. Ultimately, silica laminae with platy structure cap the prismatic horizon.^[1] This is in contrast to duripans that form in arid environments, which typically have a large component of pedogenic calcite and are characterized by a platy structure throughout, with plates 1–15 cm thick.^[13] As little as 10% silicon as opaline silica is adequate to cement horizons effectively and form a fully developed duripan.^[1]

Thorough cementation of a soil horizon to form a duripan requires geomorphic stability of long duration, particularly in the absence of readily soluble volcanic ash. The ancient landscapes of Australia fit this criterion. Silica-cemented horizons in Australia are common and are referred to as silcrete, duricrust, and gray billy.^[10]

Silica cementation has also been reported in *hardsetting* soils. Hardsetting soils have one or more horizons with hard to very hard consistence when dry. This phenomenon is recognized in soils with alternating seasons of wetting and drying, primarily those in Australia. Hardsetting soils eventually slake on wetting and are thus not irreversibly cemented.^[14] Amorphous particles and coatings of silica have been documented in these soils,^[14] suggesting that silica is an important constituent in the process of seasonal cementation.^[15] The easily mobilized silica might be ephemeral and may or may not be related to incipient formation of a duripan.

REFERENCES

1. Flach, K.W.; Nettleton, W.D.; Gile, L.H.; Cady, J.G. Pedocementation: Induration by silica, carbonates, and sesquioxides in the quaternary. *Soil Sci.* **1969**, *107*, 442–453.
2. Chadwick, O.A.; Hendricks, D.M.; Nettleton, W.D. Silica in duric soils: 1. A depositional model. *Soil Sci. Soc. Am. J.* **1987**, *51*, 975–982.
3. Dress, L.R.; Wilding, L.P.; Smeck, N.E.; Senkayi, A.L. Silica to soils: Quartz and disordered silica polymorphs. In *Minerals in Soil Environments*, 2nd Ed.; Dixon, J.B., Weed, S.B., Eds.; Soil Science Society of America: Madison, 1989; 913–974.
4. Cleary, W.J.; Conolly, J.R. Embayed quartz grains in soils and their significance. *J. Sediment. Petrol.* **1972**, *42*, 899–904.
5. Wilding, L.P.; Drees, L.R. Contributions of forest opal and associated crystalline phases to fine silt and clay fractions of soils. *Clay. Clay Miner.* **1974**, *22*, 295–306.
6. Moody, L.E.; Graham, R.C. Silica-cemented terrace edges, central California coast. *Soil Sci. Soc. Am. J.* **1997**, *61*, 1723–1729.

7. Blank, R.R.; Fosberg, M.A. Duripans in Idaho, U.S.A. *in situ* alteration of eolian dust (loess) to an Opal-A/X-ray amorphous phase. *Geoderma* **1991**, *48*, 131–149.
8. Norton, L.D. Micromorphology of silica cementation in soils. In *Soil Micromorphology: Studies in Management and Genesis*, Proceedings of the IX International Working Meeting on Soil Micromorphology, Townsville, Australia, July 1992; Ringrose-Voase, A.J., Humphreys, G.S., Eds.; Elsevier: Amsterdam, 1994; 22, 811–824.
9. Harden, J.W.; Taylor, E.M.; Reheis, M.C.; McFadden, L.D. Calcic, gypsic and siliceous soil chronosequences in arid and semiarid environments. In *Occurrence, Characteristics, and Genesis of Carbonate, Gypsum, and Silica Accumulations in Soils*; Nettleton, W.D., Ed.; Soil Science Society of America: Madison, 1991; Vol. 26, 1–16.
10. Milnes, A.R.; Wright, M.J.; Thiry, M. Silica accumulations in saprolites and soils in South Australia. In *Occurrence, Characteristics, and Genesis of Carbonate, Gypsum, and Silica Accumulations in Soils*; Nettleton, W.D., Ed.; Soil Science Society of America: Madison, 1991; Vol. 26, 121–149.
11. Munk, L.P.; Southard, R.J. Pedogenic implications of opaline pendants in some California late-pleistocene paleosols. *Soil Sci. Soc. Am. J.* **1993**, *57*, 149–154.
12. Soil Survey Staff. *Soil Taxonomy*; U.S. Government Printing Office: Washington, DC, 1975.
13. Chadwick, O.A.; Graham, R.C. Pedogenic processes. In *Handbook Soil Science*; Summer, M.E., Ed.; CRC Press: Boca Raton, 2000; E-41–E-75.
14. Chartres, C.J.; Norton, L.D. Micromorphological and chemical properties of Australian soils with hardsetting and duric horizons. *Dev. Soil Sci.* **1994**, *22*, 825–834.
15. Chartres, C.J.; Kirby, J.M.; Raupach, M. Poorly ordered silica and aluminasilicates as temporary cementing agents in hard-setting soils. *Soil Sci. Soc. Am. J.* **1990**, *54*, 1060–1067.

Silicon and Sodium

Jian Feng Ma

Research Institute for Bioresources, Okayama University, Kurashiki, Japan

Abstract

Silicon (Si) and sodium (Na) are both beneficial elements for higher plants. Si is the second most abundant element in the earth's crust. All soil-grown plants contain Si. Na is a common element in the environment. It is well known that excess Na inhibits plant growth, but small quantities of Na are essential and beneficial for some plant species. This entry reviews the beneficial effects, use in fertilizers, and plant uptake of these two elements.

INTRODUCTION

Silicon (Si) and sodium (Na) are both beneficial elements for higher plants. Beneficial elements are defined as those for which positive effects are expressed in only certain plant species or under specific growth conditions.^[1]

SILICON

Si is the second most abundant element in the earth's crust. The concentration of Si in soil solution ranges between 3.5 and 40 mg/l. Si in soil solution is mainly present as the uncharged monomeric molecule, silicic acid $[\text{Si}(\text{OH})_4]$, when the pH solution is below 9.0. At a higher pH (>9.0), $\text{Si}(\text{OH})_4$ dissociates into silicate ions $[(\text{OH})_3\text{SiO}^{-1}]$. The solubility of $\text{Si}(\text{OH})_4$ in water is 2.0 mM at 25°C, and the polymerization of $\text{Si}(\text{OH})_4$ into silica gel ($\text{SiO}_2 \cdot \text{H}_2\text{O}$) occurs when the concentration of $\text{Si}(\text{OH})_4$ exceeds 2 mM.

Uptake, Translocation, and Accumulation of Si

All soil-grown plants contain Si. However, the Si concentration of plant shoots varies greatly between plant species, ranging from 0.1% to 10% Si on a dry weight basis. This variation is largely due to different capacities of Si uptake by plant roots. Three uptake modes have been suggested, namely, active, passive, and rejective uptake.^[2]

Active uptake

Plant species which employ active uptake of Si, such as rice, take up Si faster than water.^[3] The Si concentration in rice leaf blades can reach more than 10% Si on a dry weight basis. The uptake of Si by rice roots is inhibited by metabolic inhibitors such as 2,4-dinitrophenylhydrazine, iodoacetate, 2,4-dichlorophenoxyacetic acid, and low temperature, suggesting that Si uptake is energy dependent.^[4] This uptake system has been characterized in the

literature.^[5] A kinetic study indicated that Si uptake by rice roots is mediated by a proteinaceous transporter. The K_m value for uptake was estimated to be 0.32 mM, suggesting that the transporter has a low affinity for $\text{Si}(\text{OH})_4$. Si uptake is very rapid and not inducible. Mercuric chloride and phloretin significantly inhibited Si uptake, but 4,4'-diisothiocyanato-stilbene-2,2'-disulfonic acid and boric acid did not, suggesting that the transporter for Si contains cysteine residues but not lysine (Lys) residues, or that Lys residues, if present, are not located at sites which are important for transport activity. However, neither the gene encoding the transporter nor the transporter protein has been isolated.

Passive uptake

With the passive mode of uptake, the uptake rate of Si by plant roots is similar to that of water.^[3] The Si concentration in the shoot largely depends on Si concentration in the medium but rarely exceeds 2.5% Si. Cucumber shows this type of uptake.

Rejective uptake

Some plants, such as tomato, take up Si more slowly than water.^[3] The Si concentration in the shoots is typically lower than 0.5% Si. A rejective mechanism seems to be involved in this uptake mode although it is not well understood.

A comparative study has showed that at least two transporters (SIT1 and SIT2) are involved in Si transport from the external solution to the xylem in rice roots (Fig. 1). SIT1 is responsible for radial transport of Si from the external solution to root cortical cells, while SIT2 regulates the release of Si from cortical cells into the xylem (xylem loading).^[6] SIT1 also exists in non-Si accumulating species such as cucumber and tomato, but the density of this transporter differs among plant species, following the order:

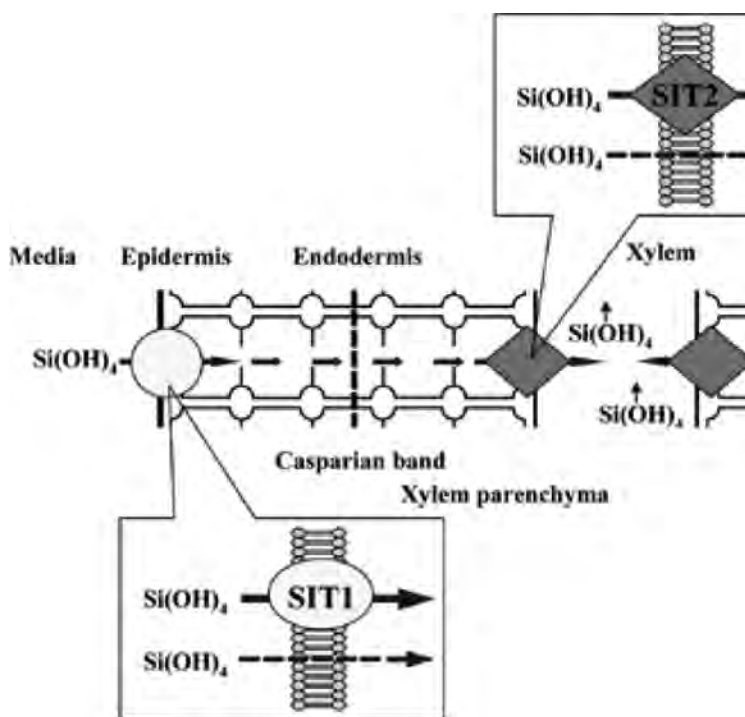


Fig. 1 The schematic representation of Si uptake system in different plant species. Radial transport of Si includes transporter-mediated transport and passive diffusion in rice, cucumber, and tomato. Xylem loading of Si is mediated by a transporter in rice and by diffusion in cucumber and tomato. SIT1: Si transporter from external solution to cortical cells; and SIT2: Si transporter for xylem loading.

Source: From Mitani & Ma.^[7]

rice > cucumber > tomato.^[7] However, in contrast to rice, xylem loading of Si was mediated by diffusion in cucumber and tomato. It seems that xylem loading is the most important determinant of a high accumulation of Si in the shoots. The much lower accumulation of Si in cucumber and tomato could be explained by a lower density of SIT1 and defective or an absence of SIT2.

Following uptake by the roots, Si(OH)_4 is translocated to the shoot via xylem. The form of Si in the xylem has been identified as monomeric Si(OH)_4 in wheat^[8] and rice.^[9] In the shoot, Si(OH)_4 is concentrated through the loss of water and is polymerized. The process of Si polymerization converts Si(OH)_4 to colloidal Si(OH)_4 and finally to silica gel with increasing Si(OH)_4 concentration. In rice plants, more than 90% of total Si in the shoot is present in the form of silica gel, while the concentration of colloidal plus monomeric Si is kept below 140–230 mg Si/l.^[3] A similar pattern of accumulation is observed in cucumber leaves, although the total Si concentration of cucumber is much lower than that of rice.

Si is deposited as a 2.5- μm thick layer in the space immediately beneath the thin (0.1 μm) cuticle layer, forming a cuticle–Si double layer in leaf blades of rice.^[10] There are two types of silicified cells in rice leaf blades, namely silica cells and silica bodies or silica motor cells (Fig. 2).^[3] Silica cells are located on vascular bundles and are dumbbell-like in shape, while silica bodies are in bulliform cells of rice leaves. The silicification of cells proceeds from silica cells to silica bodies. In addition to leaf blades, silicified cells are also observed in the epidermis and vascular tissues of the stem, leaf sheath, and hull of rice.

As there is no evidence that Si is involved in metabolic activities of plants, silica gel deposited on tissue surfaces is proposed to play an important role in alleviating biotic and abiotic stresses. The accumulation of Si in the shoots is a prerequisite for the beneficial functions of Si.

Beneficial Effects of Si

Positive effects of Si have been observed in a number of plant species, including rice, wheat, and barley.

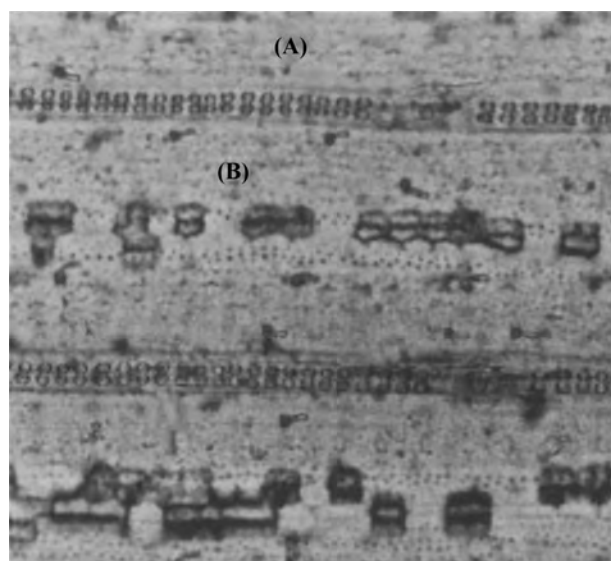


Fig. 2 Silica cell (A) and silica body (B) in rice leaf blades.

Source: From Ma, Miyake, et al.^[3]

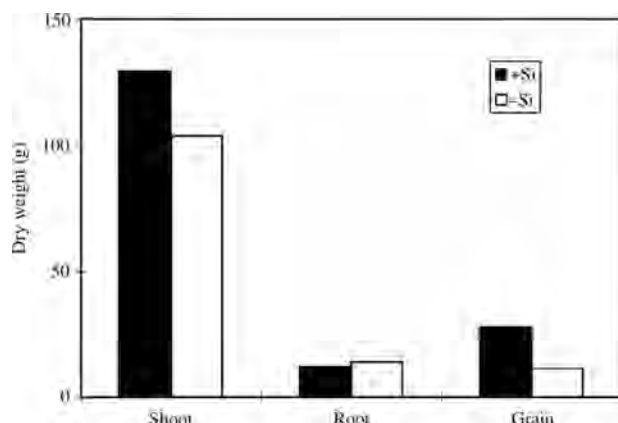


Fig. 3 Effect of Si application on the growth of rice. Rice was cultured in a nutrient solution with or without 1.67 mM Si as Si(OH)_4 .

The beneficial effects of Si differ between plant species. Beneficial effects are usually obvious in plants which actively accumulate Si in their shoots (Fig. 3). The more the accumulation of Si in the shoots, the larger is the effect that is gained. This is because most effects of Si are expressed through the formation of silica gel, which is deposited on the surface of leaves, stems, and other organs of plants. On the other hand, the beneficial effects of Si vary with growth conditions. The effects

are usually expressed more clearly when plants are under abiotic or biotic stresses. In addition, Si is the only element which does not damage plants when accumulated in excess. This is because $\text{Si}(\text{OH})_4$ remains undissociated at physiological pH and is also due to silica polymerization. Neither binding to cellular substances nor the creation of high osmotic pressure occurs in plants, even when accumulation is up to 10% Si on a dry weight basis. The beneficial effects of Si are outlined in Fig. 4.

Si-stimulated photosynthesis

Si deposited on the leaf blade keeps the leaf erect, thereby improving light interception by the leaf blade and indirectly stimulating photosynthesis. This is particularly important in dense plant stands when nitrogen (N) fertilizers are heavily applied to minimize mutual shading.

Si-decreased susceptibility to disease
and insect damage

It is well known that Si application reduces the severity of fungal diseases such as blast and sheath blight of rice, powdery mildew disease of barley and wheat, and vermin damage of rice by the plant hopper in the field. In addition

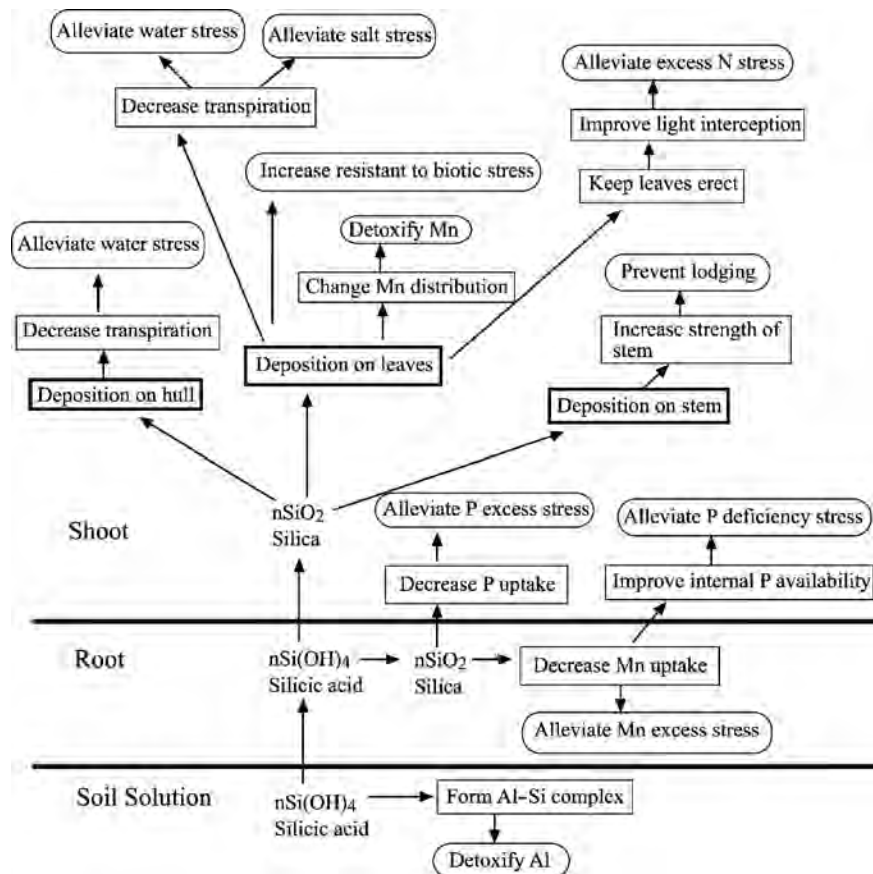


Fig. 4 The uptake, distribution, and accumulation of Si and its beneficial effects on crop growth in relation to biotic and abiotic stresses.

Source: From Ma, Miyake, et al.^[3]

to species with an active Si uptake mode, it was also reported that Si prevents powdery mildew disease of solution-cultured cucumber and muskmelon.^[11]

It has been reported that the foliar application of Si is effective in inhibiting powdery mildew development on cucumber, muskmelon, and grape leaves.^[12,13] Foliar applied Si may deposit on the surface of leaves and play a role similar to Si transported from the roots. This approach may be particularly useful for crops with passive or rejective Si uptake modes.

Si deposited on the tissue surface is proposed to be responsible for the protective effects of Si against biotic stress. The deposited Si prevents physical penetration by insects and/or makes the plant cell less susceptible to enzymatic degradation by fungal pathogens. It has been reported that Si application resulted in a stimulation of chitinase activity in cucumber and the rapid activation of peroxidases and polyphenol oxidases after infection with *Pythium* spp.^[14] Glycosidically bound phenolics extracted from Si-treated plants when subjected to acid- or beta-glucosidase hydrolysis displayed a strong fungistatic activity.

Si-increased resistance to climatic stress

Si application reduces injury to rice caused by climatic stresses such as typhoons, low or high temperature, and low light. Typhoon damage usually results from lodging and sterility. The deposition of Si in rice increases the thickness of the culm wall and the size of the vascular bundle and can help to prevent lodging. Si deposited at the hull decreases the transpiration of panicles by about 30% at either milky or maturity stage (Table 1)^[3] and prevents excess water loss. This is the means by which Si application can significantly increase the percentage of ripened grain.^[15]

Low temperature during summer can cause serious damage to rice crops. Low temperature decreases Si uptake by rice, and insufficient sunshine lowers the Si/N ratio in rice, which can lead to an outbreak of blast disease. The application of Si under such conditions remarkably reduces the incidence of blast in rice.

Table 1 Effect of Si application on transpiration of the panicle at different stages.

Treatment	Transpiration (mg H ₂ O/g fresh wt. 1 hr)		Si concentration in hull (%)
	Milky stage	Maturity stage	
+Si	204	39.2	6.36
–Si	279	50.4	0.18
+Si/–Si	1.37	1.29	

Note: The excised panicles, with or without Si supply, were placed in an incubator at 30°C, relative humidity of 30% for 1 hour.

Source: From Ma, Miyake, et al.^[3]

The positive effects of Si under low-light conditions are particularly evident. Si effects on rice growth under shading treatment are greater than for a non-shaded treatment.^[3] However, the mechanism responsible for this phenomenon is unknown. It has been hypothesized that Si deposited on the leaf epidermal system might act as a “window” to facilitate the transmission of light through photosynthetic mesophyll tissue. However, evidence supporting this hypothesis is not available.^[16]

Si-alleviated water stress

Si can alleviate water stress by decreasing transpiration. The transpiration of leaves is mediated through stomata and the cuticle. Rice plants have a thin cuticle, and the formation of a cuticle–Si double layer significantly decreases cuticular transpiration. The transpiration of rice decreases with increasing Si concentration in the shoot.^[3] Water stress causes stomatal closure and consequently decreases photosynthetic rate. Therefore, Si stimulates the growth of rice more under water-stressed conditions (low humidity) compared to non-stressed conditions (high humidity).

Si-alleviated mineral stress

Mineral stress can be either a deficiency of an essential element or an excess of an essential or other element. Many reports have shown the beneficial effects of Si under various mineral stresses including phosphorus (P) deficiency, excess of P, and toxicity of aluminum (Al), Na, and manganese (Mn).^[3]

Beneficial effects of Si under P deficiency stress have been observed in many plants, including rice and barley. Si does not have any effect on the availability of P in soil.^[17] The beneficial effects of Si on growth under P deficiency stress appear to result from decreased Mn and Fe uptake and thus increased P availability within P-deficient plants.^[18,19]

Si decreases P uptake when its supply is high.^[13] This phenomenon has been observed in rice and some non-Si accumulators such as tomato, strawberry, cucumber, and soybean.^[3] In rice, the organic P component is not affected by Si, but cellular inorganic P is significantly decreased by Si when P is supplied at high concentrations (0.7 mM).^[18] Therefore, stress caused by excess P can be alleviated by Si-induced decrease in P uptake.

Beneficial effects of Si under salt stress have also been observed in rice.^[20] The Na concentration in the shoot is decreased by about 50% with Si amendment. The translocation of Na to the shoot is partly regulated by transpiration. Since Si decreases transpiration, it is suggested that beneficial effects of Si during salt stress result from decreased transpiration.

Alleviative effects of Si on Mn toxicity have been reported in water-cultured rice, barley, bean, and pumpkin.^[3] Three different mechanisms seem to be involved,

depending on the plant species. In rice,^[21] Si reduces Mn uptake by promoting the Mn-oxidizing power of the roots. In bean, it was found that Si does not reduce Mn uptake but minimizes damage by effecting the homogenous distribution of Mn in the leaf blade.^[22] In a cultivar of pumpkin, Si causes localized accumulation of Mn around the base of trichomes.^[23]

Alleviative effects of Si on Al-induced inhibition of root elongation have been reported in many crop species, including maize, cotton, rice, teosinte, sorghum, and wheat.^[24] The formation of non-toxic Al-Si complexes in the culture solution and/or interaction between Al and Si within the plant have been suggested to be the mechanisms for this alleviative effect.^[25]

Silicate Fertilizers

In 1955, Si was first recognized as a fertilizer in Japan. Since then, 1.5–2.0 tons/ha of silicate fertilizers have been applied to paddy soils in Japan leading to a significant increase in rice production.^[2] Si fertilizers are also widely applied in Korea, China, and the United States. Various Si fertilizers have been developed including slags, which are by-products of the iron (Fe) manufacturing industry (mainly containing calcium silicate), and Na and potassium (K) silicates. The solubility of these fertilizers and their availability to plants vary greatly. They are used as a basal dressing or for topdressing. K and Na silicates are also used as foliar sprays for plants which are not able to take up Si via the roots.

SODIUM

Na is a common element in the environment. The content of this element in the earth's crust is about 2.8%, and Na concentrations in soil solutions range from 0.4 to 150 mM Na, depending on soil type. It is well known that excess Na inhibits plant growth, but small quantities of Na are essential and/or beneficial for some plant species.

Essentiality of Na

Certain plant species using the C₄ photosynthetic pathway (C₄ and CAM plants) require Na as an essential element. In these plants, Na deficiency causes chlorosis, necrosis, and decreased growth. Na is supposed to play a significant role in the conversion of pyruvate to phosphoenolpyruvate (PEP) in the mesophyll chloroplasts.^[26] PEP is the first product of carbon fixation in C₄ plants. Failure to regenerate the PEP from pyruvate will result in decreased photosynthesis in C₄ plants. A cotransporter of pyruvate/Na was found in mesophyll chloroplasts of *Panicum miliaceum*,^[27] suggesting that Na is involved in the transport of pyruvate from bundle sheath cells to mesophyll chloroplasts. In some plant species such as *Amaranthus tricolor* L., Na

Table 2 Effect of Na addition on plant growth and Na accumulation in two strains of tomato cultivated under low K stress.

Strain	Plant dry weight (g)		Na accumulation (mg/plant)
	–Na	+Na	
576	1.06	1.67	9.9
546	0.80	0.85	2.9

Source: From Figdore, Gerloff, et al.^[28]

enhances growth by stimulating nitrate uptake and reduction.^[29] Since only trace amounts of Na are required for its function in plants, Na deficiency rarely occurs in nature.

Beneficial Effects of Na

Beneficial effects of Na on plant growth are observed under both K deficiency and with optimal K supply. Under K deficiency, Na is able to act as a substitute for some functions of K such as charge compensation and osmoregulation. The K content required for optimal growth decreases from 4.3% to 1.0% in lettuce.^[30] However, the extent of beneficial effects of Na under low K stress differs between plant species. Na addition under low K stress is effective in rice, barley, and cotton but not in corn, potato, or soybean.^[1] The substitution capacity of Na also differs between cultivars within a species. The high Na substitution capacity of some strains of tomato is associated with a high accumulation of Na in the top under low K stress (Table 2).^[28] However, the beneficial effects of Na in these plant species are not apparent when K supply is sufficient.

On the other hand, in particular plant species, a beneficial effect of Na under optimal K supply is observed. This effect is not attributed to a partial substitution of Na for K, but for the stimulation of cell elongation and maintenance of water balance, although the exact mechanism is unknown. A typical example is sugar beet. Usually, Na concentration in plants is less than 10% of K levels, but in sugar beet, the Na concentration is similar to or greater than K, and increased Na enhances sugar beet growth.^[1]

Na Fertilizers

The application of Na may be effective for some plant species (e.g., natrophilic species) under K deficiency conditions. Na chloride or sulfate has been applied to sugar beet stands.^[1] Na fertilizers are also used to increase the Na content of forage and pasture plants for animal nutrition.

REFERENCES

1. Marschner, H. Beneficial mineral elements. In *Mineral Nutrition of Higher Plants*; Academic Press: San Diego, 1995; 405–426.

2. Takahashi, E.; Ma, J.F.; Miyake, Y. The possibility of silicon as an essential element for higher plants. *Comment. Agr. Food. Chem.* **1990**, *2*, 99–122.
3. Ma, J.F.; Miyake, Y.; Takahashi, E. Silicon as a beneficial element for crop plants. In *Silicon in Agriculture*; Datonoff, L., Ed.; Elsevier: Amsterdam, 2001; 17–39.
4. Okuda, A.; Takahashi, E. Effect of metabolic inhibitors on Si uptake by rice roots. *Jpn. J. Soil Sci. Plant Nutr.* **1962**, *33*, 453–455.
5. Tamai, K.; Ma, J.F. Characterization of silicon uptake by rice roots. *New. Phytol.* **2003**, *158*, 431–436.
6. Ma, J.F.; Mitani, M.; Nagao, S.; Konishi, S.; Tamai, K.; Iwashita, T.; Yano, M. Characterization of Si uptake system and molecular mapping of Si transporter gene in rice. *Plant Physiol.* **2004**, *136*, 3284–3289.
7. Mitani, M.; Ma, J.F. Uptake system of silicon in different plant species. *J. Exp. Bot.* **2005**, *56*, 1255–1261.
8. Casey, W.H.; Kinrade, S.D.; Knight, C.T.G.; Rains, D.W.; Epstein, E. Aqueous silicate complexes in wheat, *Triticum aestivum* L. *Plant Cell Environ.* **2003**, *27*, 51–54.
9. Mitani, M.; Ma, J.F.; Iwashita, T. Identification of silicon form in the xylem of rice (*Oryza sativa* L.). *Plant Cell Physiol.* **2005**, *46*, 279–283.
10. Yoshida, S.; Ohnishi, Y.; Kitagishi, K. Histochemistry of Si in rice tissues. III. The presence of cuticle–silica double layer in the epidermal tissue. *Soil Sci. Plant Nutr.* **1962**, *8*, 1–5.
11. Miyake, Y.; Takahashi, E. Effect of silicon on the resistance of cucumber plant to the microbial disease. *Jpn. J. Soil Sci. Plant Nutr.* **1982**, *53*, 106–110.
12. Menzies, J.; Bowen, P.; Ehret, D. Foliar application of potassium silicate reduces severity of powdery mildew on cucumber, muskmelon, and zucchini squash. *J. Am. Soc. Hort. Sci.* **1992**, *117*, 902–905.
13. Bowen, P.; Menzies, J.; Ehret, D. Soluble silicon sprays inhibit powdery mildew development on grape leaves. *J. Am. Soc. Hort. Sci.* **1992**, *117*, 906–912.
14. Cherif, M.; Asselin, A.; Belanger, R.R. Defense responses induced by soluble silicon in cucumber roots infected by *Pythium* Spp. *Phytopathology* **1994**, *84*, 236–242.
15. Ma, J.F.; Nishimura, K.; Takahashi, E. Effect of silicon on the growth of rice plant at different growth stages. *Soil Sci. Plant Nutr.* **1989**, *35*, 347–356.
16. Agarie, S.; Agata, W.; Uchida, H.; Kubota, F.; Kaufman, P.B. Function of silica bodies in the epidermal system of rice (*Oryza Sativa* L.): Testing the window hypothesis. *J. Exp. Bot.* **1996**, *47*, 655–660.
17. Ma, J.F.; Takahashi, E. The effect of silicic acid on rice in a P-deficient soil. *Plant Soil* **1990**, *126*, 121–125.
18. Ma, J.F.; Takahashi, E. Effect of silicate on phosphate availability of rice in a P-deficient soil. *Plant Soil* **1991**, *133*, 151–155.
19. Ma, J.F.; Takahashi, E. Effect of silicon on the growth and phosphorus uptake of rice. *Plant Soil* **1990**, *126*, 115–119.
20. Matoh, T.; Kairusmee, P.; Takahashi, E. Salt-induced damage to rice plants and alleviation effect of silicate. *Soil Sci. Plant Nutr.* **1986**, *32*, 295–304.
21. Okuda, A.; Takahashi, E. Effect of silicon supply on the injuries due to excessive amounts of Fe, Mn, Cu, As, Al, Co of barley and rice plant. *Jpn. J. Soil Sci. Plant Nutr.* **1962**, *33*, 1–8.
22. Horst, W.J.; Marschner, H. Effect of silicon on manganese tolerance of bean plants (*Phaseolus Vulgaris* L.). *Plant Soil* **1978**, *50*, 287–303.
23. Iwasaki, K.; Matsumura, A. Effect of silicon on alleviation of manganese toxicity in pumpkin (*Cucurbita Moschata* Duch cv. Shintosa). *Soil Sci. Plant Nutr.* **1999**, *45*, 909–920.
24. Cocker, K.M.; Evans, D.E.; Hodson, M.J. The amelioration of aluminium toxicity by silicon in higher plants: Solution chemistry or an in planta mechanism? *Physiol. Plant* **1998**, *104*, 608–614.
25. Ma, J.F.; Sasaki, M.; Matsumoto, H. Al-induced inhibition of root elongation in corn, *Zea Mays* L. is overcome by Si addition. *Plant Soil* **1997**, *188*, 171–176.
26. Brownell, P.E. Sodium in C₄ photosynthesis. In *Plants in Action*; Atwell, B., Kriedemann, P., Turnbull, C., Eds.; Macmillan Education Australia: South Yarra, 1999; 518–520.
27. Ohnishi, J.; Kanai, M. Na⁺-induced uptake of pyruvate into mesophyll chloroplasts of a C₄ plant, *Panicum Miliaceum* L. *FEBS Letter* **1987**, *219*, 347–350.
28. Figdore, S.S.; Gerloff, G.C.; Gabelman, W.H. The effect of increasing NaCl levels on the potassium utilization efficiency of tomatoes grown under low-K stress. *Plant Soil* **1989**, *119*, 295–303.
29. Ohta, D.; Yasuoka, S.; Matoh, T.; Takahashi, E. Sodium stimulates growth of *Amaranthus Tricolor* L. plants through enhanced nitrate assimilation. *Plant Physiol.* **1989**, *89*, 1102–1105.
30. Costigan, P.A.; Mead, G.P. The requirements of cabbage and lettuce seedlings for potassium in the presence and absence of sodium. *J. Plant Nutr.* **1987**, *10*, 385–401.

BIBLIOGRAPHY

1. Ma, J.F.; Takahashi, E. *Soil, Fertilizer, and Plant Silicon Research in Japan*, 1st Ed.; Elsevier Science: Amsterdam, 2000.
2. Ma, J.F. Functions of silicon in higher plants. In *Silicon Biomineralization*; Muller, W.E.G., Ed.; Springer: Berlin, 2003; 127–147.
3. Mitani, M.; Ma, J.F. Uptake system of silicon in different plant species. *J. Exp. Bot.* **2005**, *56*, 1255–1261.

Slaking, Dispersion, and Crust Formation

Hwat Bing So

School of Agriculture and Food Sciences, University of Queensland, St. Lucia, Queensland, Australia

Abstract

The nature of soil structure is that it conveys specific properties to the soil, and any alteration such as the breakdown or structural development will affect the physical properties of the soil. The particles and aggregates of a soil are arranged in a hierarchical order where the lower orders tend to have higher densities and greater internal strength than the higher orders. Slaking (breakdown of the higher order: macroaggregates to microaggregates) and dispersion (breakdown of the lower order: microaggregates to particles) are the fundamental processes that are the cause of most of the soil physical problems encountered in the field that affects plant growth and soil behavior, such as waterlogging, reduced infiltration, crusting, hardsetting, compaction, poor timing of farming operations, and poor germination and establishment. Although generally perceived as distinctly different problems in the field, they can be corrected and managed using the same techniques, i.e., the prevention or suppression of the processes of slaking and dispersion in these soils. These measures include the use of organic ameliorants, the introduction of grasses into the crop rotation, and the use of gypsum and lime.

INTRODUCTION

Soil structure is generally defined as the arrangement, orientation, and organization of the primary particles of sand, silt, and clay into compound aggregates, which exhibit properties that are unequal to the properties of a mass of non-aggregated material with a similar texture. Therefore, the nature of soil structure is that it conveys specific properties to the soil, and any alteration, i.e., breakdown or structural development, to the soil structural units will affect the physical properties of the soil.

ORGANIZATION OF SOIL PARTICLES AND AGGREGATES

The aggregation and organization of the soil particles tend to form a hierarchical order^[1,2] where the lower orders tend to have higher densities and greater internal strength than the higher orders. A schematic diagram of the hierarchical nature of soil structural elements in a clay soil is given in Fig. 1.^[1] Clay particles tend to form domains (packets of parallel clay sheets, generally consisting of 5–7 sheets), and in turn, several domains form clusters, followed by several orders of clusters, i.e., microaggregates and macroaggregates. The hierarchical nature implies that the destruction of a lower order will result in the destruction of all higher hierarchical orders. An example is the dispersion of sodic clay domains, which results in the destruction of all higher orders, resulting in a dense soil with low hydraulic conductivity (K). Hence, the clay domains are the fundamental

building blocks of the soil, and its integrity may determine the soil's physical properties and behavior.

MECHANISMS OF SLAKING, DISPERSION, AND CRUST FORMATION

Soil aggregates break down in response to an external force or energy source such as tillage or raindrop energy or internally generated forces such as when aggregates are subjected to rapid wetting, with the latter being the most common cause occurring in the field. Upon rapid wetting, aggregates break down in two stages. The first is “slaking” where macroaggregates break down into microaggregates (approximately 20–250 μm in diameter). The second stage is “dispersion” where the microaggregates break down into individual particles of sand, silt, and clay.^[3] The result of these processes is the formation of a surface seal at the soil–air interface, associated with an increased packing density of the soil particles and a reduction in pore sizes. These, in turn, give rise to a reduced K or infiltration rates. Upon drying, the surface seal dries out into a surface crust. Surface sealing and crusting is a common phenomenon with cultivated soils (Fig. 2). When slaking and dispersion occur throughout the surface horizon following wetting, the whole surface horizon experienced an increased packing density and a reduction in pore sizes. In cases where the surface horizon hardens rapidly following drying, the soil is referred to as hardsetting.^[5,6] These soils are associated with high contents of silt and fine sand, which do not readily form aggregates.

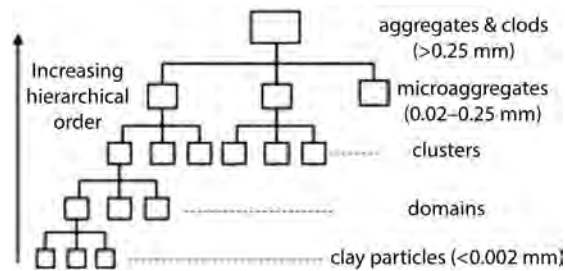


Fig. 1 Hierarchical organization of particles and structural elements.

Source: Adapted from Dexter.^[1] ©1988.

Slaking of the surface soil is not necessarily a detrimental process. In its virgin state, the surface soil of self-mulching Vertisols slakes upon wetting followed by surface sealing, which breaks into small aggregates when dry and forms a mulch layer. These are highly productive soils. However, when slaking is followed by dispersion, the soil's productivity is generally reduced associated with the lowered soil K, which in turn limits the supply of water to the root zone.

Mechanisms of Slaking

Slaking following rapid wetting is the result of the following three distinct processes:

1. Softening of bonds;
2. Differential swelling following wetting of the aggregates from the outside: The stresses formed between the swelling outer layers and the inner parts of the aggregates result in the successive disintegration of the outer layers of the aggregates, which are readily observable when dropped in water; and
3. Increased pressure within entrapped air pockets from the tensions exerted by the water meniscus: Combined with the softening of the bonds, this pressure results in visually observable mini explosions of the soil aggregates when dropped in water.

Mechanisms of Dispersion

Dispersion is the result of swelling pressures derived from the osmotic pressure difference between the ion concentrations within the diffuse double layer between the clay particles and the external solution created by the addition of water. Dispersion occurs when the swelling pressure (the repulsive force) is greater than the attractive force between the clay particles. Dispersion is seen as the cloudy appearance around the aggregates when left in water for some period.^[3] In the field, it can be seen as the cloudy and muddy appearance of water in puddles or in streams and rivers near construction sites or cultivated fields, after a rainstorm. Swelling of the clay domains is enhanced by the

presence of monovalent cations such as sodium (Na) and the presence of low electrolyte (salt) concentrations in the water or soil solution. Swelling is suppressed by the presence of divalent and trivalent cations^[7] and by the presence of high electrolyte (salt) concentrations.

CONSEQUENCES OF SLAKING AND DISPERSION

Slaking and dispersion are the fundamental processes of the degradation of structurally unstable soils, and most cultivated soils exhibit some degree of instability. These degradation processes can be associated with many soil physical problems observed in the field that affect plant growth or soil behavior.^[4,8] Waterlogging, reduced infiltration, crusting, hardsetting, compaction, poor timing of farming operations, and poor germination and establishment are generally perceived as distinctly different problems in the field, but as shown in Fig. 2, they have a common cause in slaking and dispersion and can therefore be corrected using the same treatment. Hence, an ameliorative treatment applied to reduce or prevent dispersion and to correct one particular problem will also correct many of the other physical problems. The most important effect of slaking and dispersion is the reduction of the soil's K associated with the blockage of pores between the larger particles or stable aggregates. A small increase in dispersion may result in large decreases in K.^[9,10]

FACTORS AFFECTING SUSCEPTIBILITY TO SLAKING AND DISPERSION

Since slaking and dispersion are associated with the swelling of clay domains and the inability of the forces of attraction or bonding to hold the clay domains together, they are affected by the nature of the clay minerals, the adsorbed cations, and the bonding mechanism.

Clay Type and Composition of Adsorbed Cations

Swelling clays (montmorillonites, vermiculites, and illites) tend to confer a degree of instability on soil aggregates, whereas non-swelling clays (kaolinites) are generally associated with high structural stability. Swelling is enhanced by the presence of Na and low electrolyte concentrations. Na-dominated clays continue to swell when the water content is increased and float apart in suspension,^[7] which is referred to as spontaneous dispersion. Swelling, slaking, and dispersion will also increase in response to the increasing proportion of Na or exchangeable sodium percentage (ESP). Although there is laboratory evidence for a threshold ESP,^[11] there is no clear evidence that a threshold ESP exists in field soils.^[4] However, for soil classification purposes, a soil is classified as sodic in Australia when the soil's ESP is equal to or greater than 6%^[6] and in the United

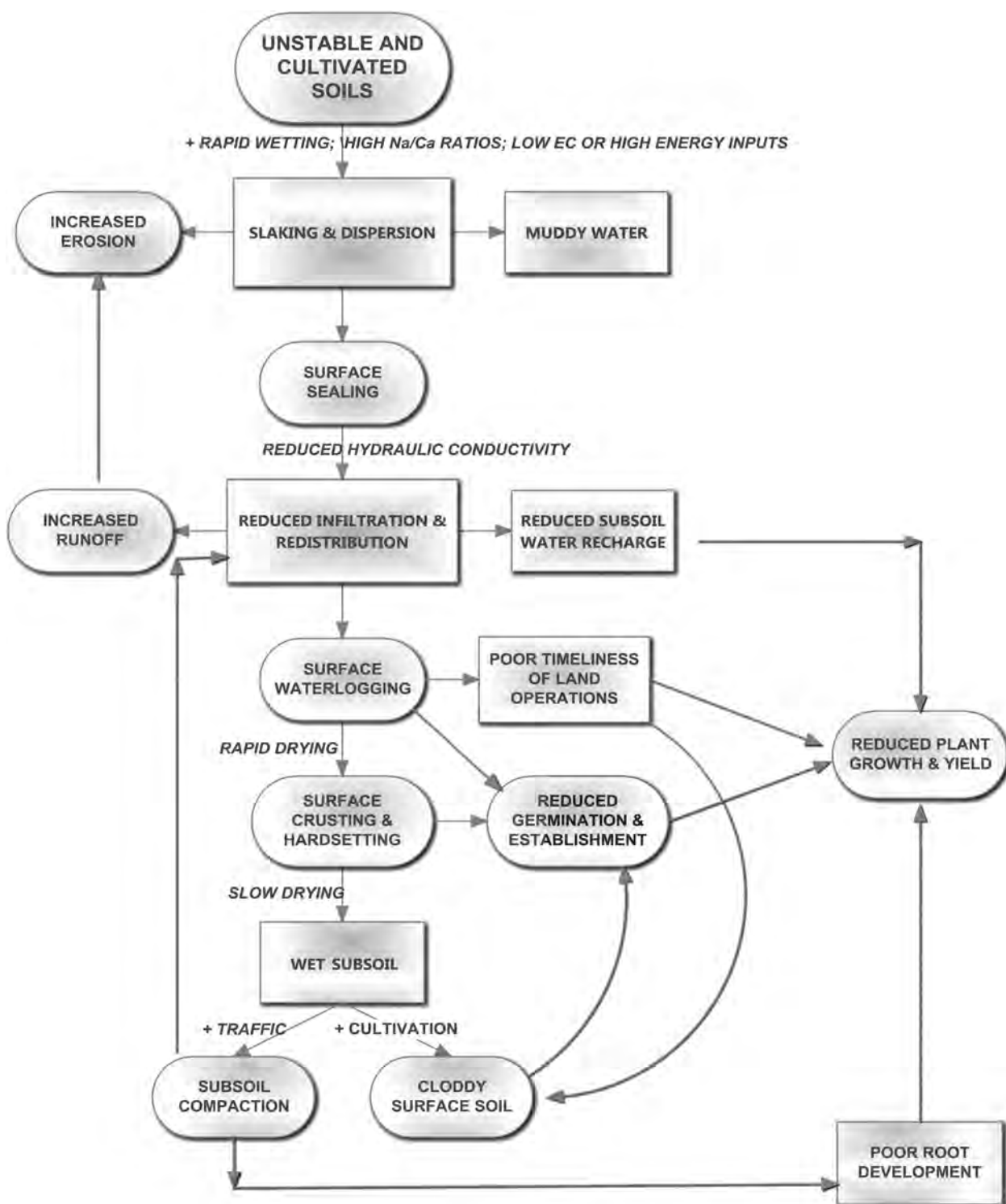


Fig. 2 A schematic diagram showing how slaking and dispersion may result in soil physical problems often observed in the field (shown in the rounded boxes) and plant growth.

Source: Adapted from So & Aylmore.^[4] ©1983.

States when the ESP is equal to or greater than 15%,^[10] indicating the increased probability of physical problems occurring on these soils when used for cropping. On the contrary, dispersion is reduced when the electrical

conductivity (EC) of the bulk solution increases. Therefore, a soil with a high ESP and high salinity may appear as well structured when dry but will immediately disperse following rainfall or when irrigated with low EC water.

Table 1 Strategies for managing slaking and dispersion.

Process	Aggregate hierarchical level affected	Action required	Management options available
Slaking	Macroaggregates	Prevent breakdown of bonds	1. Zero tillage or minimum tillage 2. Slow wetting during irrigation 3. Reduce droplet energy during spray irrigation
		Increase aggregate bonding and bonding strength	1. Incorporate organic matter 2. PVA application 3. PAM application 4. Lime or Ca carbonates application 5. Introduce grasses in crop rotation
Dispersion	Microaggregates and clay domains	Increase flocculation: 1. Replace monovalent cations with divalent or polyvalent cations (permanent exchangeable cation effect) 2. Increase electrolyte concentrations (temporary electrolyte effect)	1. Gypsum application 2. Lime application 3. Gypsum or slaked lime [Ca(OH) ₂] application

Calcium (Ca)-dominated clays exhibit limited swelling. Ca montmorillonite swells to a basal distance between two clays of 18 Å.^[12] At this distance, the electrostatic attractive forces between the clay plates are greater than the repulsive or swelling pressures between the plates. Ca- and magnesium (Mg)-dominated clays tend to flocculate, whereas Na-dominated clays tend to disperse. However, Mg clay does not have the same stability as Ca clay, and its tendency to disperse will depend on its combination with other factors, e.g., Mg clay will disperse following cultivation at much lower water contents than Ca clay.^[13] Soils with similar ESP tend to show greater dispersion when the other cation is Mg compared to Ca.^[14]

Trivalent cations such as Al³⁺ and Fe³⁺ confer the greatest stability onto clay domains. Hence, besides providing bonding materials to the aggregates, the presence of aluminum (Al) and iron (Fe) oxides and hydroxides ensures a large proportion of these cations being present in the diffuse double layer. Highly acidic soils (pH < 5) will have a high concentration of Al in solution, and these will stabilize the clay domains. Hence, Oxisols are generally non-dispersive soils and possess high structural stability.

Nature of Aggregate Bonding

Swelling and dispersion may be prevented if slaking of aggregates is prevented by strong bonds. The most common bonding materials for aggregates are in the order of abundance organic matter, Fe and Al oxides and hydroxides, Ca carbonates, and silica oxides. The strength of bonding increases with the amount present in the soil. The material that is readily affected by human activity is soil organic matter. In the virgin and undisturbed state, soils with high ESP are generally non-dispersive because of the

accumulated soil organic matter in the surface soil. However, cultivation reduces the soil organic matter contents, and when the bond strength from organic matter decreases below the swelling pressures from the sodic clay, dispersion occurs followed by the appearance of soil physical problems. Hence, the addition of adequate organic matter that will readily break down and produce suitable bonding material may prevent or reduce the dispersion of clay domains.

MANAGEMENT OF SLAKING AND DISPERSION

Managing slaking and dispersion of clay soils is the essence of managing soil structure. Table 1 lists a range of management options that have been used. Slaking can be managed at the macroaggregate level by avoiding the breakdown of existing bonds (organic matter) by minimizing soil disturbance (zero tillage) and/or through the addition of bonding material such as organic matter, polymers [such as polyvinyl alcohol (PVA) and polyacrylamide (PAM)], and Ca carbonates (lime). It can also be reduced by incorporating grasses into the existing crop rotation as grass roots, and associated fungal hyphae bind microaggregates to macroaggregates, besides maintaining or increasing soil organic matter contents.

On the other hand, dispersion can be managed at the microaggregate level by using a combination of divalent cations and high EC in the soil solution, e.g., the use of Ca ameliorants such as gypsum (CaSO₄ · 2H₂O) that provides Ca²⁺ and the continuous presence of adequate EC levels associated with its limited solubility (2.2 dS/m at 25°C). Gypsum is most useful on Na-dominated clays and other soils when dispersion occurs, e.g., following excessive cultivation. However, it may not be helpful on clays that are

already strongly dominated by Ca ions. Ca^{2+} replaces Na^+ from the clay exchange sites (exchangeable cation effect) that represent a permanent or residual effect of gypsum. The presence of a high electrolyte where free gypsum is present provides the dominant effect on sodic soils. It represents the temporary electrolyte effect of gypsum.^[4,9] Lime will provide a similar effect, but it is a slow-acting ameliorant.

CONCLUSION

The nature of soil structure is that it conveys specific properties to the soil, and any alteration such as the breakdown or structural development will affect the physical properties of the soil. The particles and aggregates of a soil are arranged in a hierarchical order where the lower orders tend to have higher densities and greater internal strength than the higher orders. Slaking (breakdown of the higher order: macroaggregates to microaggregates) and dispersion (breakdown of the lower order: microaggregates to particles) are the fundamental processes that are the cause of most of the soil physical problems encountered in the field that affects plant growth and soil behavior, such as waterlogging, reduced infiltration, crusting, hardsetting, compaction, poor timing of farming operations, and poor germination and establishment. Although generally perceived as distinctly different problems in the field, they can be corrected and managed using the same techniques, i.e., the prevention or suppression of the processes of slaking and dispersion in these soils.

REFERENCES

1. Dexter, A.R. Advances in characterization of soil structure. *Soil Till. Res.* **1988**, *11*, 199–238.
2. Hadas, A. Long-term tillage practice effects on soil aggregation modes and strength. *Soil Sci. Soc. Am. J.* **1987**, *51*, 191–197.
3. So, H.B. Salt-affected soils: Physical properties and behavior. In *Encyclopedia of Environmental Management*; Jorgensen, S.E., Ed.; Taylor & Francis: New York, 2013; 2334–2344.
4. So, H.B.; Aylmore, L.A.G. How do sodic soils behave? The effects of sodicity on soil physical behavior. *Aust. J. Soil Res.* **1993**, *31*, 761–777.
5. Mullins, C.E.; MacLeod, D.A.; Northcote, K.H.; Tisdall, J.M.; Young, I.M. Hardsetting soils: Behaviour, occurrence and management. *Adv. Soil Sci.* **1990**, *11*, 37–108.
6. Northcote, K.H.; Skene, J.K.M. *Australian Soils with Saline and Sodic Properties*; CSIRO: Melbourne, 1972.
7. Aylmore, L.A.G.; Quirk, J.P. The structural status of clay systems. *Clay. Clay Miner.* **1962**, *9*, 104–130.
8. So, H.B.; Smith, G.D.; Raine, S.R.; Schafer, B.M.; Loch, R.J. *Sealing, Crusting and Hardsetting Soils: Productivity and Conservation*; Australian Society of Soil Science Inc.: Brisbane, 1985; 527 pp.
9. So, H.B.; Tayler, D.W.; Yates, W.J.; McGarity, J.W. Amelioration of structurally unstable grey and brown clays. In *Modification of Soil Structure*; Emmerson, W.W., Bond, R.D., Dexter, A.R., Eds.; John Wiley & Sons: Chichester, 1978; 325–334.
10. So, H.B.; Cook, G.D. The effect of slaking and dispersion on the hydraulic conductivity of clay soils. In *Soil Surface Sealing and Crusting*; Poesen, J.W.A., Nearing, M.A., Eds.; Catena Verlag: Cremlingen, 1993; Vol. 24, 55–64.
11. Shainberg, I.; Letey, J. Response of soils to sodic and saline conditions. *Hilgardia* **1984**, *52*, 1–57.
12. Quirk, J.P. Interparticle forces: A basis for the interpretation of soil physical behavior. *Adv. Agron.* **1994**, *53*, 121–183.
13. Emmerson, W.W. Physical properties and structure. In *Soil Factors in Crop Production in a Semi-Arid Environment*; Russell, J.S., Greacen, E.L., Eds.; University of Queensland Press: St. Lucia, 1977; 78–104.
14. Bakker, A.C.; Emmerson, W.W.; Oades, J.M. The comparative effects of exchangeable calcium, magnesium, and sodium on some soil physical properties of red-brown earth subsoils. I. Exchange reactions and water contents for dispersion of Shepparton soil. *Aust. J. Soil Res.* **1973**, *11*, 143–150.

Social and Behavioral Sciences' Contribution to Soil Science

Sven B. Lundstedt

School of Public Policy and Management, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

The social and behavioral sciences cover most of the ways to study human beings and their cultures and social systems. They are complex ways. Economics is considered by many to be both a social and a behavioral science. Since Keynes' time, economics has been basically divided traditionally into micro and macro levels of analysis. To the extent that soil science sees a need to connect some of its research questions to economic phenomena, it would be useful to study this important distinction.

INTRODUCTION

Inadequate soil management has led to wars of expansion and natural disasters that have, in turn, contributed to the very decline of nations and civilizations. We need but look to our human environment to see this important natural connection. The strong link between soil and human society is discussed further in the entries *Human Society and the Planet: Soil as a Heritage* (p. 1127) and *Human Society and Soil* (p. 1123).

OVERVIEW

Psychology has grown into an overwhelmingly complex, multifaceted discipline with many separate subdisciplines. If one is interested in human behavior and the individual's connection with soil phenomena, research questions might come in the form of new hypotheses about soil and individual people's various psychological and physical needs being met by soil reconstitution.

Sociology is traditionally a discipline that focuses on the social system (society) and all of its manifestations both large and small. Like other branches of science mentioned, it has many subdisciplines as well. Society has had a deep historical influence on the material substance called soil. The question of how social organizations treat the issues of soil management and development is important, especially whether sociological studies help or hinder agricultural problem solving. It is well to point out that sociology and agriculture have traditionally been partners.

Political science is yet another social and behavioral science that has focuses on political systems and governance. To the extent politics, e.g., plays an integral part in soil management, and soil issues have become exceptionally important. From federal to local and state government, agriculture has been working for it, a large and powerful political base in the U.S. society with many

powerful stakeholders and interest groups of varying kinds who participate in policy formation and implementation associated with fundamental soil issues.

Cultural anthropology has a distinguished research history of studying human cultures and subcultures of all kinds worldwide. That would include issues that bear upon soil considered in the light of all relevant facets of a culture. There is surely many human cultural aspects of the uses of soil which have been noted in anthropological studies of primitive and modern agricultural communities.

The concept of "latent policy," therefore, serves as a pivotal social psychological variable useful in explaining, quantitatively and qualitatively, the positive and negative public meaning of secretive policy behavior and its presence under circumstances where it is not customarily expected to exist or, even under law, should not exist.^[1,2]

A final example is one concerning the management of soil and, hence, soil policy as understood in the sense that is very much a part of agricultural policy formation. Policies governing the uses and the forms of management of soil exist in all societies in some form, although some are more advanced than others. Such policies are important because they reflect recognition of not only accumulated scientific knowledge of soil but also the agricultural policies (simply serving as rules) for best practices in its use in growing food, especially while being aware of the social structures and processes that are contexts for soil management practices.

Other societal processes and values play a part as well. Economic values and policies governing soil use practices, e.g., are surely among the most seriously affected by whether they are manifest, latent (transparent), or latent (secret) policies, which then often lead to transactional difficulties in bargaining and conflict management, especially in negotiations where transparency was initially intended but not, in fact, practiced. Lack of forthrightness and transparency suggests dishonesty in negotiation transactions and is a sure formula for failure in an economic transaction if true.

Yet “latent policy transactions” (secret deals) are in constant need of requiring careful application and scrutiny, even though the realists claim they are necessary facts of life and part of the cost of doing business. Those who look upon the inevitability will claim, cynically, for transactions to continue to exist among humans, and they will continue to exist as common attributes of the human uses of soil. Hence, as many insist, there is always some ongoing need for a control point and a certain amount of regulation. The Securities and Exchange Commission, e.g., stands out as an illustration of a public agency that seeks balance in such matters.

An illustration of another interdisciplinary approach is illustrated by Thompson and Mosely's high-Alpine tropical paleoclimatology research which provides examples of increasing global climate change. It suggests that a long-term crisis may arise from water reductions and drought from a decline in melting snow in the high Andes and African mountains such as Kilimanjaro, shrinking of both the Quelccaya Ice Cap, and the shrinking across the world in Africa of the Kilimanjaro ice fields. The social science of cultural anthropology has the methods to be more than quite capable of recording important social changes in communities affected by decreases in usable soil for agriculture.^[3-5] These underlying anthropomorphic changes can be important and are often poorly understood and blindly implemented, leading to serious human policy errors without further collateral validity.

LITERATURE AND SOME OF ITS POTENTIAL LESSONS ABOUT SOIL MANAGEMENT

It is appropriate to study the lessons provided by historic literature. Especially important is how such literature can often enlighten us about many important things, including the power of soil in all societies for both good and evil. John Steinbeck, author of *The Grapes of Wrath*, begins his novel writing about the enormous power of soil management gone wrong, thus triggering the social change and failed technology leading to the enormous human suffering associated with the Dust Bowl migrations of the 1930s in the United States.^[6]

In his first chapter, Steinbeck lays the “groundwork” for the striking human drama of a soil catastrophe with devastating social changes and human suffering to follow.

In the roads where the teams moved, where the wheels milled the ground and the hooves of the horses beat the ground, the dirt crust broke and the dust formed Men and women huddled in their houses, and they tied handkerchiefs over their noses when they went out, and wore goggles to protect their eyes When night came again it was black night, for the stars could not pierce the dust to get down, and the window lights could not even spread beyond

their own yards. Now the dust was evenly mixed with the air, an emulsion of dust and air. Houses were shut tight, and cloth wedged around doors and windows, but the dust came in so thinly that it could not be seen in the air, and settled like pollen on the chairs and tables, on the dishes. The people brushed it from their shoulders. Little lines of dust lay at the door sills.

Once that very dust was a nutrient-laden soil that sustained human agriculture and attacked hunger. With its disappearance came a crashing decline in one form of American agricultural subculture and rural civilization. Steinbeck alerted Americans to the grim reality of poor soil management and reminded many of the second-order economic and societal events such as the “Great Depression” that was coterminous with it.

Keeping this excerpt from Steinbeck's book in mind, recall that his book was published in 1939, a turning point of the “Great Depression.” The following citation, which contains excerpts from real situations reported by the *New York Times* on July 27, 2003, vividly illustrates some of the social conditions in Brazil which reflect soil management policies and promises that failed to be implemented. The setting is Mirante do Paranapanema, approximately 900 km South West of Rio de Janeiro.^[7]

The encampments have sprung up along highways throughout this fertile farming region and are expanding rapidly. Precarious clusters of shacks made of wood, cardboard and plastic, they are inhabited by thousands of peasants who have flocked here with a single unwavering demand: a piece of land.

In some cases, farms, cattle ranches and sugar mills have been occupied, threatening violent clashes. In other areas, squatters have in weeks seized or vandalized government offices, blocked highways, taken hostages and looted trucks to press their demands.

It almost came to this kind of response when the homeless farmers, retreating by necessity from soil erosion destruction, described so vividly and accurately by Steinbeck's novel, were forced west to California facing new difficulties over which they had little control at first. Fortunately, the suffering created by the initial real “dust bowl” experience about which Steinbeck wrote only lasted until World War II, which gradually introduced new resources and agricultural techniques to reverse it. The agricultural problems remained to be worked on long after World War II. However, the slave labor conditions for the “dust bowl” farmers created by a readiness of predatory efforts of some to exploit migrant human labor in California lasted only long enough to find new victims from Mexico and elsewhere south of the border. What were the social, monetary, and human costs of this initial western migration in order to escape from the toxic dust from used-up soils? Almost

incalculable amounts of money and pain, it would seem, when human suffering is considered!

This example of the power of soil for good and evil is the gist of a momentous social problem, essentially formed over soil rights, but it does not stop there; it reaches far beyond it to a human struggle for a new standard of living and quality of life. Perhaps, it is an unfortunate metaphor, but the circumstances can be attributed to the growth of a one-sided form of economic and social pathology in which the main symptoms are social chaos and even the danger of a revolution, rather than peaceful evolution. It is a widening of a dangerous conflict which is the result of not acting much earlier to intervene in peaceful ways, to act competently, and to take appropriate steps to solve this serious societal problem. There have been many warnings throughout the hemisphere in the last few decades, but only few were listened.

Many economists would argue that a robust market-oriented economy, with little governmental interference of course, is the only solution for these kinds of "sustainable development," and the sooner the better it would seem they would say. Yet for this solution to work, it had to happen much earlier in the 20th century, so that it could develop properly to include education. Equally as many other economists, perhaps, would prefer to see a larger government effort in the form of an aid plan using education and new forms of governance to guide the social change and its soil counterpart, it is so necessary for these problems to be solved equitably. According to some who have tired of the constant battles over resources and a workable structure, this intervention is, once again, too little and too late to do much else than to encourage the kind of violence described in this *New York Times* article.

For others, the only solution to these problems is found in using market forces to solve them and the less intervention the better, except, however, to increase trade in services, manufacturing, and agriculture and, of course, education, reflecting the need for trained workers. But if truth be admitted, the prognosis is unknown. In either case, it is an enormously complex situation in which only one or two social and behavioral sciences at work may not be enough, in addition to agricultural innovations and the soil sciences. The problems are not only political and economic. Soil science alone, while an essential science with a distinguished research history, cannot do the whole job.

INTERACTIVE ISSUES

A subject as broad as the social and behavior sciences can enlighten soil scientists by drawing their attention to these suggested intellectual bridge building.

There are several research issues that cross the boundaries of soil and social sciences. Research crossroads illustrate the involving partnerships, the hybrids, such as among these disciplines are growing, not diminishing. Hence, the rise of interdisciplinary "disciplines" involving the social and behavioral sciences and, say, soil science is increasing.^[3,6] One can even go a step further in to point out that many have reached into biology and physics to mention only two possibilities.

CONCLUSION

Social and behavioral sciences and soil science are truly evolutionary sciences. The interconnections formed by them may yet grow to have some additional future subject matter kinship with soil science, a most esteemed intellectual partner for the future. It is a question worth further consideration.

REFERENCES

1. Lundstedt, S. Latent policy. *Working Paper Series: College of Administrative Sciences*; The Ohio State University: Columbus, 1976.
2. Lundstedt, S.B.; Spicer, M.W. Latent policy and the federal communications commission. In *Telecommunications, Values and the Public Interest*; Lundstedt, S.B., Ed.; Ablex Publishing Company: Norwood, 1990; 289–299.
3. Frank, R. Interdisciplinary: The first half-century. *Soc. Sci. Res. Counc.* **1988**, 42 (3), 73–78.
4. Krajick, K. Ice man: Lonnie Thomson scales the peaks for science. *Science* **2002**, 298 (October), 518–593.
5. Thompson, L.; Mosley-Thompson, E.; Davis, M.E.; Henderson, K.A.; Brecher, H.H.; Zagorodnov, V.S.; Mashiotta, T.A.; Lin, P.; Mikhaleuk, V.N.; Hardy, D.R.; Beer, J. Kilimanjaro ice core records: Evidence of Holocene climate change in tropical Africa. *Science* **2002**, 298 (October), 589–593.
6. Steinbeck, J. *The Grapes of Wrath*; Penguin Books: New York, 1939; 3–7.
7. Roter, R. Poor press Brazil's leader on his promise of land. *New York Times Int.* **2003**, (July), 6.

Sodic Soils

Pichu Rengasamy

*Department of Soil and Water Science, University of Adelaide, Waite,
South Australia, Australia*

Abstract

Soils with a high proportion of exchangeable sodium are considered sodic soils. On wetting these soils, clay particles swell and disperse degrading soil structure, and on further drying soils become dense. Poor water storage and restricted movement of air and water in soil profile lead to yield decline of many crops. Sodic subsoils cause accumulation of salts in the root zone. In dryland sodic soils, multiple problems such as salinity and toxic elements in addition to sodicity occur making management decisions difficult

INTRODUCTION

Sodic soils are those with a high proportion of sodium adsorbed to the soil particles compared to other cations such as calcium, magnesium, and potassium. Exchangeable sodium percentage (ESP) is a criterion used to identify sodic soils. When a soil with high ESP is wetted, the bonds between soil particles are weakened. As a result, the clay particles swell and often become detached and disperse at high water content. This leads to poor physical properties of soils. In the classification system developed in United States, a soil with an ESP > 15 is considered as “sodic,” and in Australia, the criterion is >6. These differences are owing to the influence of several soil factors including salt levels, pH, organic matter, and clay mineralogy on the adverse effects of ESP on soil properties. Sodic soils are widespread in arid and semiarid regions of the world extending up to 30% of the total land area (Table 1). Use of saline water, including waste and effluent waters containing sodium salts, for irrigation induces sodicity in soils. Sodicity is a latent problem in many salt-affected soils where deleterious effects on soil properties are evident only when salts are leached below a threshold level.^[1] While soil salinity reduces plant growth, directly affecting physiological functions through osmotic and toxicity effects on plants, sodicity causes deterioration of soil physical properties indirectly affecting plant growth and survival. Sodic soils are subjected to severe structural degradation and exhibit poor soil–water and soil–air relationships; these properties (Table 2) adversely affect root growth, thereby restricting plant production and making it difficult to work in soils when they are wet or dry.

YIELD DECLINE IN SODIC SOILS

Sodic soils make the paddocks prone to waterlogging, poor crop emergence and establishment, gully erosion, and in

some instances tunnel erosion. Because of the heterogeneity in the accumulation of sodium by soil particles, these symptoms may be observed only in certain parts of the paddock. Generally, patchy growth and barren patches are visible in a number of spots in a paddock, while the rest of the field may look normal. However, the effects of sodicity are fully realized in the harvested yield. The actual yield obtained in sodic soils is often less than half of the potential yield expected on the basis of climate, particularly rainfall and evapotranspiration.^[2,3] Relative yield of cereals grown in dryland sodic soils in Australia in relation to average root zone ESP is given in Fig. 1.

Swelling and dispersion of sodic aggregates destroy soil structure, reduce the porosity and permeability of soils, and increase the soil strength even at low suction (i.e., high water content). These adverse conditions restrict water storage and transport. Soils are, therefore, either too wet immediately after rain or too dry within a few days for optimal plant growth. Thus, the range of soil water content that does not limit plant growth and function (“non-limiting water range”) is very small.^[4] Dense, slowly permeable sodic subsoils reduce the supplies of water, oxygen, and nutrients needed for obtaining maximum potential yield. During the rainy season, even with prolonged ponding of water on the surface, only a small increase in water content occurs in subsoil. The low porosity leads to slow internal drainage and water redistribution within the profile.^[5] This reduction in water storage causes crop water stress during prolonged dry periods. Subsoil as a source of water and nutrients becomes more important in dryland cropping regions than in irrigated soils.

SALT ACCUMULATION IN ROOT ZONES OF SODIC SOILS

Soils with sodic subsoils are characterized by moderate to high exchangeable sodium and, in many cases, with high

Table 1 World distribution of sodic soils.

Continent	Country	Area of sodic soils (000 ha)
North America	Canada	6,974
	United States	2,590
South America	Argentina	53,139
	Bolivia	716
	Brazil	362
	Chile	3,642
	Algeria	129
Africa	Angola	86
	Botswana	670
	Cameroon	671
	Chad	5950
	Ethiopia	425
	Ghana	118
	Kenya	448
	Liberia	44
	Madagascar	1,287
	Namibia	1,751
	Niger	1,389
	Nigeria	5,837
	Somalia	4,033
	Sudan	2,736
	Tanzania	583
	Zimbabwe	26
South Asia	Bangladesh	538
	India	574
	Iran	686
North and Central Asia	China	437
	Union of Soviet Socialist Republics	119,628
Australasia	Australia	339,971

Source: From Bui, Krogh, et al.^[6]

pH (>8.5) where carbonate and bicarbonate minerals are present. Subsoil sodicity restricts drainage beyond the root zone and, as a result, salts accumulate in this zone. The concentration of accumulated salts fluctuates with rainfall pattern, input of salt from agronomic practices, and soil weathering, as schematically explained in Fig. 2. Dryland salinity or “seepage salinity” in many countries is associated with rising saline groundwater tables. However, the extent of subsoil salinity, also called “transient salinity,” not associated with saline groundwater is large in many landscapes dominated by subsoil sodicity. A relationship between rainfall, subsoil ESP, and EC_e for northeastern Australian soils has been reported.^[7] In dryland regions with annual rainfall between 250 and 600 mm, sodic subsoils have an EC_e between 2 and 20 that can dramatically

Table 2 Physical and chemical properties of a typical sodic soil profile in South Australia.

Properties	0–20 cm	20–40 cm	40–100 cm
Chemical properties			
pH _{1:5} (water)	7.9	8.9	9.2
EC _e (dS/m)	0.4	3.8	4.9
Organic carbon (%)	1.2	0.6	0.3
Exchangeable sodium (%)	6.2	14.6	24.5
CaCO ₃ (%)	0.1	2.8	4.5
Boron (mg/kg)	1.2	22.0	38.5
Water soluble Al(OH) ₄ [−] (mg/kg)	0.0	1.2	2.6
Physical properties			
Spontaneously dispersed clay (%)	1.2	8.6	9.4
Swelling (mm/mm)	0.04	0.18	0.20
Hydraulic conductivity at saturation (mm/day)	22.8	4.5	2.3
Penetrometer resistance at 100 kPa suction (MPa)	1.8	4.2	4.8
Aeration porosity (%)	9.7	4.8	3.9
Bulk density (Mg/m ³)	2.0	2.2	2.3
Final infiltration rate in the field (mm/hr)	0.2		

affect crop production through osmotic effects during dry periods. Laboratory-measured EC_e will increase several folds under field conditions as the soil layers dry in-between rainy days. The combination of poor water storage and osmotic stress enhances water stress to crops under dryland cropping.

ROOT ZONE CONSTRAINTS IN DRYLAND SODIC SOILS

Multiple problems occur in soils with subsoil sodicity. Soil compaction, crusting, and induration of subsoil layers require “physical” reclamation. Sodicity, salt accumulation, and alkaline pH require “chemical” reclamation.

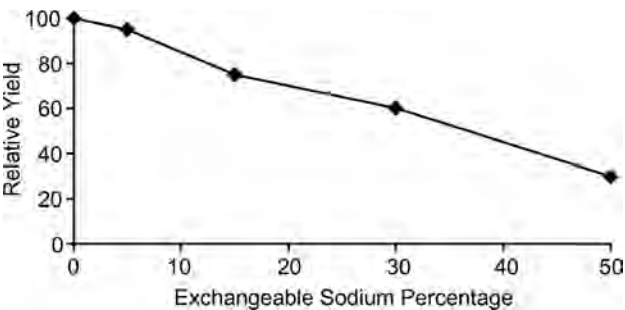


Fig. 1 Relative yield of cereals grown in Australian sodic soils in relation to average root zone ESP.

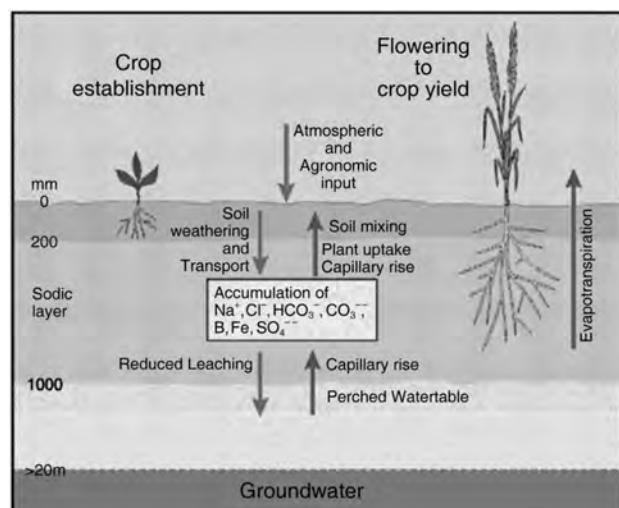


Fig. 2 Salt accumulation in sodic subsoils.

All of these conditions cause, in addition to water stress, macro- and micronutrient deficiency, and toxicity owing to Na^+ , Cl^- , HCO_3^- , CO_3^{2-} , B, $\text{Al}(\text{OH})_4^-$, and others. Low organic matter and biological activity compound these problems encountered in sodic subsoils.

MANAGEMENT OF DRYLAND SODIC SOILS

Major criteria in increasing productivity in dryland sodic soil are improved water storage and transport in the root zone and crop water use efficiency. More information is available on agricultural management in sodic soils that is more relevant to irrigated lands.^[5] Reclamation procedures involving high costs are prohibitive in dryland regions because of low benefit/cost ratio.

Diagnosis of multiple problems with large variations, vertically and horizontally across the paddock, is primarily important. Gypsum is the most commonly used compound to reclaim sodic soils. Subsoil reclamation may involve higher rates of gypsum application or deep placement of gypsum by deep ripping or deep plowing. Salt-tolerant plant species may alleviate subsoil salinity. Plants that can tolerate ion toxicity such as boron, carbonate, sodium, and chloride have also been identified. Strategies to improve subsoil fertility may include: 1) mechanical means of placing nutrients deeper in the profile; 2) using nutrient sources of lower or higher mobility; 3) using deep-rooted legumes to fix nitrogen at depths; and 4) selection of plant species

and genotypes better suited to acquiring nutrients from subsoils. Research is needed on developing plants that modify the rhizosphere and adapt to edaphic conditions. Farming systems should be developed to prevent accumulation of salts and toxic elements in the root zone of sodic soils.

CONCLUSION

Soils with a high proportion of exchangeable sodium are considered sodic soils. On wetting these soils, clay particles swell and disperse degrading soil structure, and on further drying soils become dense. Poor water storage and restricted movement of air and water in soil profile lead to yield decline of many crops. Sodic subsoils cause accumulation of salts in the rootzone. In dryland sodic soils, multiple problems such as salinity and toxic elements in addition to sodicity occur making management decisions difficult. Soil amelioration with gypsum and choice of tolerant crops are useful in farming these soils.

REFERENCES

1. Rengasamy, P.; Olsson, K.A. Sodicity and soil structure. *Aust. J. Soil Res.* **1991**, *31*, 821–837.
2. French, R.J.; Schultz, J.E. Water use efficiency of wheat in a mediterranean-type environment. 1. The relation between yield, water use and climate. *Aust. J. Agric. Res.* **1984**, *35*, 765–775.
3. Rengasamy, P. Sodic soils. In *Methods for Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentine, C., Stewart, B.A., Eds.; CRC Press: New York, 1997; 265–277.
4. Letey, J. The study of soil structure: Science or art. *Aust. J. Soil Res.* **1991**, *29*, 699–707.
5. Oster, J.D.; Jayawardane, N.S. Agricultural management of sodic soils. In *Sodic Soils: Distribution, Properties, Management and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 125–147.
6. Bui, E.N.; Krogh, L.; Lavado, R.S.; Nachtergaele, F.O.; Toth, T.; Fitzpatrick, R.W. Distribution of sodic soils: The world scene. In *Sodic Soils: Distribution, Properties, Management and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 19–33.
7. Shaw, R.J.; Coughlan, K.J.; Bell, L.C. Root zone sodicity. In *Sodic Soils: Distribution, Properties, Management and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 95–106.

Sodic Soils: Formation and Global Distribution

Brian Murphy

New South Wales Department of Land and Water Conservation, Cowra, New South Wales, Australia

Abstract

Sodic soils, which are considered a subset of salt-affected soils, have levels of exchangeable sodium that are sufficiently high enough to cause swelling and clay dispersion in water having low electrolyte concentrations, such as under rainfall and in many irrigation waters. These properties cause serious environmental and productivity problems.

INTRODUCTION

A functional definition of a sodic soil is as follows:

Whenever exchangeable sodium is present in sufficient concentrations to influence soil behaviour, sodicity occurs, a soil is said to be sodic.^[1]

Precise problems depend on where sodicity is in profile. Typical problems associated with sodic soils include:^[2,3]

- dense impermeable subsoils causing waterlogging and restricting root growth;
- surface crusting and sealing restricting emergence and surface infiltration, hardsetting seedbeds;
- development of a toxic chemical environment for root growth;
- high susceptibility to erosion, especially deep gully erosion (Fig. 1);
- contamination of surface waters by sediments and colloid-assisted contaminants; and
- instability in engineering structures; piping or tunneling can occur in structures associated with these soils.^[3]

ASSESSMENT AND RECOGNITION

The occurrence of any of the above problems is often sufficient to indicate at least the possibility that soils are sodic. From a classification viewpoint, sodic soils are traditionally considered to have dense impermeable subsoils with a coarse columnar to prismatic structure. Surface soils are often of a lighter texture than the subsoil and may be bleached in the lower layer (A2 or E horizon) (Fig. 2). These soils are identified in the Food and Agriculture Organization of the United Nations/United Nations Educational, Scientific and Cultural Organization, Russian, Canadian, and old Australian schemes by the terms solonetz or soloth.^[4,5] In *Soil Taxonomy*, this group of soils relates to the natric Alfisols. In a new Australian soil classification,

the order sodosol is used to identify soils with sodic properties.^[6] However, soils with sodic layers are common in many other soil types, especially the Vertisols and Aridisols of *Soil Taxonomy*^[7] and the Vertosols of the Australian Soil Classification System.^[6]

The level of exchangeable sodium percentage (ESP), which is the amount of exchangeable sodium as a percentage of the total cation exchange capacity of the soil, is used to confirm the identification of sodic soils. ESP is used as this gives a strong indication of the actual and potential dispersion behavior of the soil. Two of the most commonly used critical levels of ESP are 15% for U.S. soils^[8,9] and 6% for Australian soils.^[1,6,10] The level of ESP that begins to adversely affect the soil behavior can vary with the level of soluble salts in the soil, the nature of soil clays, soluble minerals present in the soil, and other factors. Sumner^[11] presented a more detailed history of the definition of sodic soils and the development of critical values of ESP.

As dispersion is the most critical property that causes the limitations of these soils, measures of dispersion have been used to identify sodic soils. Dispersion can be measured by aggregate tests or suspension tests.^[1] Spontaneous dispersion of soil aggregates in water is usually a strong indication of the presence of a sodic soil. However, the accuracy of dispersion tests to predict the presence of sodic soils is influenced by the level of electrolyte concentration as described in Fig. 3, with the critical level of ESP that causes dispersion being dependent on the electrical conductivity of the soil solution. As a general rule, classification systems have adopted the convention that the levels of ESP used to identify sodic soils are those that predict the dispersion behavior of the soil in water with a low electrolyte concentration (rainwater or freshwater).^[11]

Sodic and Saline Soils

The close association between saline and sodic soils has led to some confusion when discussing these soils. Sodic



Fig. 1 Severe gully erosion in sodic subsoils formed on granite parent material.

soils are salt-affected soils but are not necessarily saline. Sodicty relates to the level of exchangeable sodium in the soil with the positive charge of the sodium ion being balanced by the negative charge on the clay minerals and organic matter. Soils may be sodic without being saline. Salinity relates to the level of free salt present in the soil solution. In saline soils, the positive change of the soluble cations, such as sodium ions, is balanced by anions such as chloride, sulfate, or bicarbonate present in the soil solution.

In general usage, the following soils have come to be recognized:^[1,11]

1. Sodic and saline soils with a high ESP (>15% in the United States and >6% in Australia) and high soluble salts ($EC_{sat} > 4$ dS/m), where EC_{sat} is the electrical conductivity of the saturation extract.
2. Sodic and non-saline soil with a high ESP (>15% in the United States and >6% in Australia) and low soluble salts ($EC_{sat} < 4$ dS/m).



Fig. 2 Example of a typical profile of a sodic soil profile—solonetzic soil or soloth-sodosol-natric xeralf.

ORIGIN AND FORMATION

Sodium, which on average is 2.64% of the atoms in the earth's crust,^[13] is highly mobile on the landscape and tends to become the dominant cation as solutions move further from their original source. This has a significant effect on the origins and the formation of sodic soils. Ultimately, the only condition for the formation of sodic soils is that sodium becomes accumulated on the clay exchange complex. Therefore, there are numerous mechanisms for the formation of sodic soils.

Traditionally, sodic soils are thought to develop by the following pathways as described by Bui et al.:^[4]

1. Initial salinization of the soil from saline parent material or rising saline groundwater.
2. Leaching of salts and dispersion and downward movement of clays (illuviation) to form an abrupt texture change at the top of the B horizon.

The link to saline and arid areas for sodic soils is evident.

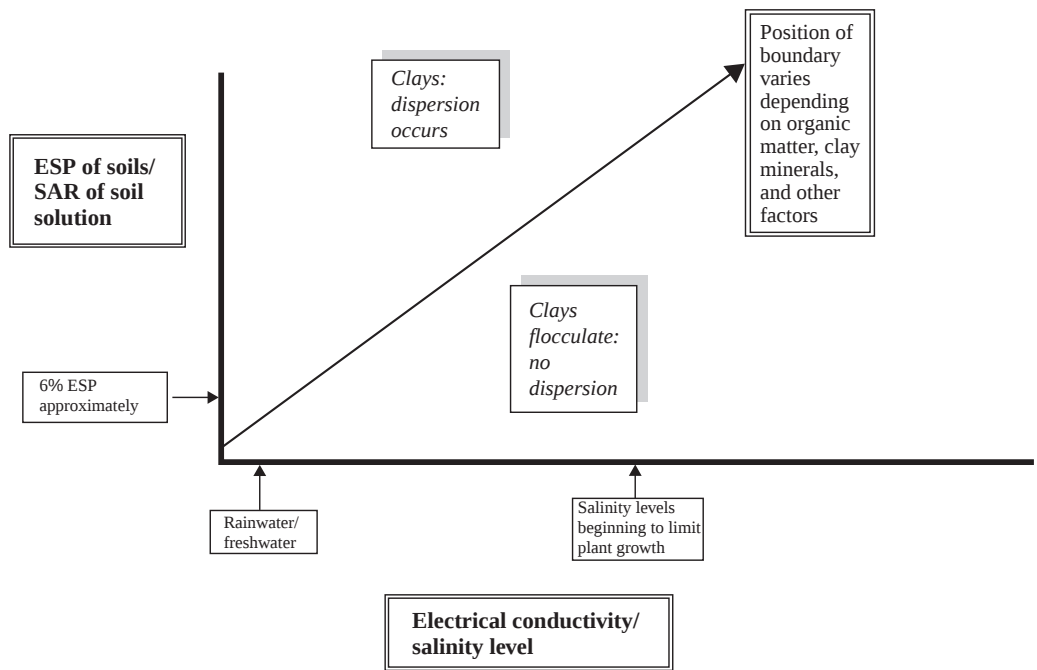


Fig. 3 Schematic diagram of effect of ESP, sodium adsorption ratio, and EC levels on dispersion in sodic soils.
Source: Adapted from Rengasamy, Greene, et al.^[12]

Subsequent work has shown that this apparently simple process can, when combined with weathering and the pedological, geomorphological, and chemical processes occurring on the landscape, lead to a number of pathways for the development of sodic soils.^[4,14]

The Formation of Sodic Soils

A brief summary of the possible pathways for the formation of sodic soils is given below and is summarized in Fig. 4. At any location, more than one pathway for the formation of sodic soils can be operating.

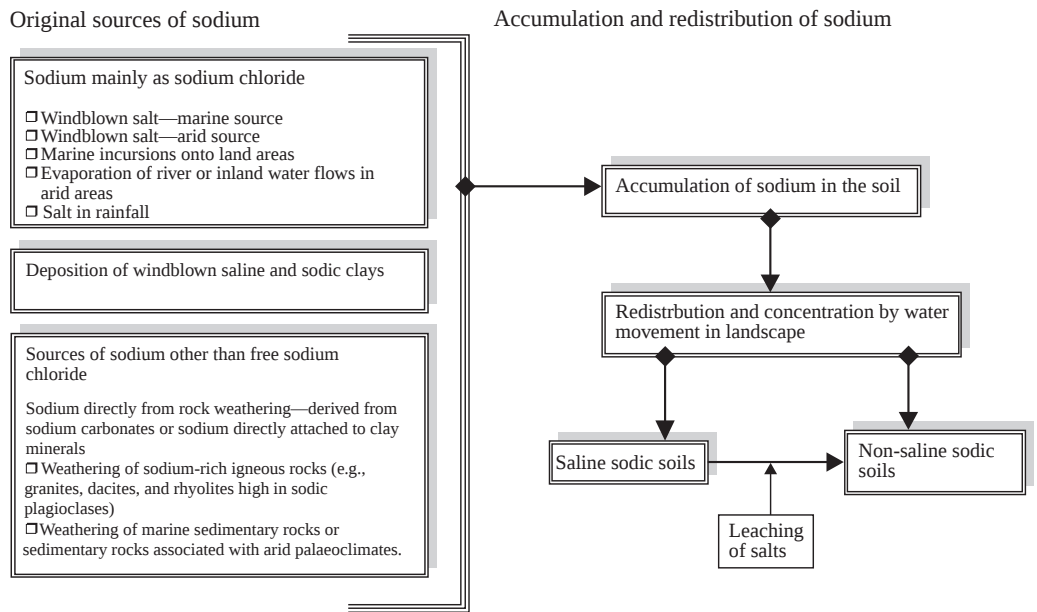


Fig. 4 Schematic diagram summarizing the origins and development of sodic soils.

Geological—long-term development of sodic soils

1. Weathering of rocks with high levels of sodium relative to calcium and magnesium can release sodium, especially from sodic plagioclases and alkali feldspars, and lead to the development of sodic soils associated with these rocks.^[4]
2. Aeolian sources of salt and therefore sodium cations are a major source in many areas.^[14] Ocean is the main source of aeolian salt in some areas, but aeolian salt blown from arid terrestrial sources is highly significant for many areas of sodic soils.
3. The erosion and deposition of sodic clays is another possible pathway for sodic soils to develop on the landscape. Sodic clays are especially susceptible to water erosion, and the erosion and deposition may result in the formation of sodic clays in areas of deposition. Saline sodic clays may be susceptible to wind erosion because of aggregation associated with their salinity.
4. The buildup of sodium and salt in association with marine incursions of the sea in the Quaternary or Tertiary periods.^[5,14]

Short-term development of sodic soils associated with human activities

1. Rising groundwater with high salinity can lead to salinization of the soil and the development of sodic soils.
2. Irrigation with water with high sodium levels relative to calcium and magnesium can lead to the development of sodic soils.

GLOBAL DISTRIBUTION OF SODIC SOILS

Sodic soils are widespread throughout the world, many of them occurring in important agricultural areas. Most of them occur in drier areas but significant amounts also occur in more humid areas. The total area of sodic soils is estimated by Lubbock^[15] to be more than 550 million hectares. Szabolcs^[16] estimated that 932 million hectares was affected by salinity/sodicity worldwide, of which 581 million hectares was sodic soils. Although there are difficulties in estimating such figures because of varying criteria for identifying sodic soils and the lack of specific on-the-ground data, this estimate does give an indication of the potential problem of sodic soils as a world environmental and productivity issue. Sodic soils occur on all the continents, but the largest areas are in the driest continent, Australia.

The broad distribution of sodic soils on a global basis is given in Table 1.^[4]

Table 1 Summary of the global distribution of sodic soils.

Continent	Estimated area (million hectares)
North America	9.6
South America	57.9
Africa	27.1
Europe	22.9
Northern and Central Asia	120.1
Southern Asia	1.8
Australasia	340

Source: From Bui, Krogh, et al.,^[4] Gupta & Abrol,^[5] Northcote & Skene,^[10] Isbell, Reeve, et al.,^[14] Szabolcs.^[16]

CONCLUSION

Sodic soils are a major group of problem soils. They are potentially productive if managed properly, but can be severely degraded or can cause severe environmental problems if managed inappropriately. It is likely that increasing salinization worldwide will increase the areas of these soils. Therefore, knowledge about the origins, formation, distribution, and management of these soils is seen as a valuable input to managing scarce and vulnerable natural resources.

REFERENCES

1. Rengasamy, P.; Churchman, G.J. Cation exchange capacity, exchangeable cations and sodicity. In *Soil Analysis, an Interpretation Manual*; Peverill, K.I., Sparrow, L.A., Reuter, D.J., Eds.; CSIRO Publishing: Melbourne, 1999; 147–158.
2. So, H.B.; Aylmore, L.A.G. How do sodic soils behave? The effects of sodicity on soil behaviour. *Aust. J. Soil Res.* **1993**, *31*, 761–777.
3. Sumner, M.E.; Miller, W.P.; Kookana, R.S.; Hazelton, P. Sodicity, dispersion and environmental quality. In *Sodic Soils, Distribution, Properties, Management and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 149–172.
4. Bui, E.N.; Krogh, L.; Lavado, R.S.; Nachtergaele, F.O.; Toth, T.; Fitzpatrick, R.W. Distribution of sodic soils: The world scene. In *Sodic Soils, Distribution, Properties, Management and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 19–33.
5. Gupta, R.K.; Abrol, I.P. Salt-affected soils: Their reclamation and management for crop production. *Adv. Soil Sci.* **1990**, *11*, 223–288.
6. Isbell, R.F. *The Australian Soil Classification*; CSIRO Publishing: Melbourne, 1996.
7. U.S. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*; USDA, Soil Conservation Service: Washington, 1975.

8. USSSL staff. *Diagnosis and Improvement of Saline and Alkali Soils*; USDA, U.S. Government Printing Office: Washington, 1954.
9. Sposito, G. *The Chemistry of Soils*; Oxford University Press: New York, 1989.
10. Northcote, K.H.; Skene, J.K.M. *Australian Soils with Saline and Sodic Properties*; Soil Publication No. 27; CSIRO Publishing: Melbourne, 1972.
11. Sumner, M.E.; Rengasamy, P.; Naidu, R. Sodic soils: a reappraisal. In *Sodic Soils, Distribution, Properties, Management and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 3–17.
12. Rengasamy, P.; Greene, R.S.B.; Ford, G.W.; Mahanni, A.H. Identification of dispersive behaviour and the management of red-brown earths. *Aust. J. Soil Res.* **1984**, 22, 413–431.
13. Paton, T.R. *The Formation of Soil Material*; George Allen and Unwin: London, 1978.
14. Isbell, R.F.; Reeve, R.; Hutton, J.T. Salt and sodicity. In *Soils—An Australian Viewpoint*; Division of Soils; CSIRO Publishing: Melbourne, 1983; 107–117.
15. Bui, E.N.; Krogh, L.; Lavado, R.S.; Nachtergaele, F.O.; Toth, T.; Fitzpatrick, R.W. Distribution of sodic soils: The world scene. In *Sodic Soils, Distribution, Properties, Management and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 19–33.
16. Szabolcs, I. *Salt-Affected Soils*; CRC Press: Boca Raton, 1989.

Sodic Soils: Humid Regions

Samuel J. Indorante

U.S. Department of Agriculture—Natural Resources Conservation Service
(USDA-NRCS), Du Quoin, Illinois, U.S.A.

Abstract

Sodium-affected soils (SASs) usually occur in arid, semiarid, and subhumid climates. To a much lesser extent, SAS can occur in humid regions with a mean annual precipitation (MAP) > 100 cm. SAS genesis in humid regions is similar to SAS genesis in more arid regions and varies with the climate, parent material, topography, vegetation or land use, and length of the time taken for sodium (Na) soil genesis. The SAS genesis model tends to follow the sequence of salinization, desalinization, solonization, and solodization. Taxonomical and land management classification of SASs is based on soil morphology and chemical properties. Accurately identifying and mapping SAS is the first step in managing these areas to overcome the production problems caused by the high Na concentration and poor physical makeup of SAS.

BACKGROUND INFORMATION

Sodium-affected soils (SASs) or sodium (Na) soils are soils that have been adversely affected by Na salts and/or exchangeable Na. They usually occur in arid, semiarid, and subhumid climates where rainfall is insufficient to leach soluble salts from the soils, or where internal drainage is restricted. To a much lesser extent, SAS can occur in humid regions (Table 1) with a mean annual precipitation (MAP) > 100 cm because of factors that restrict leaching of soluble salts from the soil. The main factors are a source of Na, high water table, clayey dense subsoils, impermeable underlying geologic strata, and seasonal periods of high evapotranspiration.

SASs in humid areas tend to be leached of excess salts, but they contain enough exchangeable Na to disperse the clay particles. The presence of exchangeable Na, as measured by exchangeable sodium percentage (ESP), under low salt conditions, as measured by electrical conductivity of the saturation extract (ECe), causes the breakdown (peptization or dispersion) of soil aggregates (Fig. 1).^[1] These high ESP, low ECe soils tend to be sticky when wet, and highly erodible, and have poor tilth, low porosities, low infiltration rates, and low permeabilities.

Soil moisture recharge in these SAS profiles is slow, and these soils also dry slowly. When these soils dry up, they become hard, cloddy, and crusty (Fig. 2); when wet, they are sticky and difficult to work. The adverse effect of exchangeable Na on soil structure and associated soil–water and soil–plant relationships makes crop production difficult even in humid climates (MAP > 100 cm).

GENESIS OF SAS IN HUMID AREAS

SASs can occur over a wide range of climates, but generally under highly, seasonally contrasted ones of arid to

humid conditions.^[2,3] The rate and degree of SAS genesis vary in each of the climates and also vary with parent materials, topography, vegetation or land use, and the length of time Na soil genesis has taken place. There are several main environmental factors that promote the genesis of SAS in all climates: the presence of shallow saline groundwater or a saline soil solution; textural discontinuities during the deposition of eolian, glacial, alluvial, or colluvial sediments; occurrence of perched water tables within 1 m of the surface; low-slope gradients; and impeded drainage.^[2] These environmental factors in a given climate control water movement over, into, and through the soil profile and the soil landscape. Water movement, in turn, controls the distribution of soluble Na salts and exchangeable Na in Na soil profiles and across the soil landscape.^[4,5]

SAS genesis pathways have been described in arid climates,^[2] semiarid climates,^[3,6] subhumid climates,^[3,5] and humid climates.^[7–10] The sequence of pedogenic processes tends to follow but is not restricted to the classic sodic soil genesis model as summarized by Miller and Pawluk.^[11]

Sodic Soil Genesis Model

Salinization—this first step begins with salt accumulation because of Na-rich parent material or capillary rise of shallow saline groundwater. The periodic alternation of dry and humid conditions is necessary, with high rates of evapotranspiration in the summer.

Desalinization—this step begins with the leaching of salts by water moving through the soil profile.

Solonization—this step begins with the leaching of salt by rainwater, which leads to the dispersion of clay if ESP exceeds 10–15% and total soluble salt content is 0.1–1.5% or less, and the formation of a sodic soil. The translocation

Table 1 Global distribution (km²) of SAS with humid soil moisture regimes.

Country or continent	Area (km ²)
Argentina	143,246
Australia	138,185
China	34,676
Asia, Africa, North America, and South America ^a	85,891
Global total	401,998

^aThe balance of humid region SAS is scattered in these continents.
Source: Data from U.S. Department of Agriculture–Natural Resources Conservation Service, Soil Survey Division, World Soil Resources, 2000.

of dispersed clays will eventually lead to an abrupt textural change between A and B horizons.

Solodization—the final step is Na-induced hydrolysis and eluviation at the top of the slowly permeable B horizon.

The sources of Na in humid climate SAS include Na-bearing minerals in medium-textured Loess,^[9] coarse-textured Loess,^[10] alluvial deposits,^[12] and fluvial terraces.^[7] Humid climate SASs are formed under both forest^[7] and mixed forest prairie and prairie vegetation^[9] and occur on level^[9,10,12] and gently sloping^[7,9] landscapes.

NOMENCLATURE AND CLASSIFICATION OF SAS

SASs are primarily classified in terms of soil morphology, including soil texture, soil structure, and the presence of a

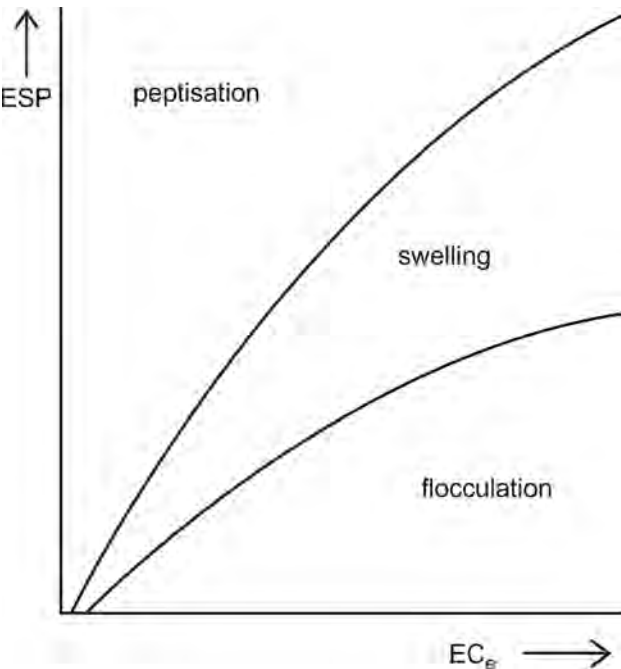


Fig. 1 Schematic diagram illustrating clay behavior as a function of ESP and ECe.
Source: From Kamphorst & Bolt.^[1]



Fig. 2 A SAS spot intermingled with non-SASs in southern Illinois, U.S.A. The climate is humid.

diagnostic horizon. They are also classified in terms of chemical properties including pH, ECe, electrical conductivity of the 1:5 soil to solution ratio (EC_{1:5}), ESP, and sodium adsorption ratio (SAR). The specific criterion used to define and classify sodic soils varies by geographic region and the goal of the classification system. Taxonomical classifications are designed to serve the multiple purposes of soil survey.^[13] Land management classifications group soils for SAS for the purpose of site-specific management of Na soil areas for agronomic and related uses.

Taxonomical Classification

One of the most common taxonomical characteristics for classifying Na soils in all climates is the presence of a natric horizon. This term is defined as a mineral soil horizon that satisfies the requirements of an argillic horizon but also has prismatic, columnar, or blocky structure and a subhorizon having an ESP >15%.^[13,14] The Australian soil classification system^[15] uses the presence of a sodic horizon to define its Sodosols (SAS). The Australian sodic horizon is a clear or abrupt textural B horizon that is sodic (ESP > 6) in all or part of the B horizon, depending on the thickness of the horizon and a pH > 5.5. Some taxonomical classifications of SASs in humid regions are Natraqualfs, Natrudalfs, Natralbolls, Natraquolls, Natraquerts, and natrudolls;^[13] others are hydromorphical solonetztes,^[16] solodic sodosol,^[15] and gleyic and mollic solonetz.^[14]

Land Management Classification of SAS

Common terms used to label, describe, and classify SAS in humid regions for management purposes are as follows:

- Sodic—a non-saline soil containing sufficient exchangeable Na to adversely affect crop production and soil structure under most conditions of plant growth. The SAR of the saturation extract is >13.^[17]

- Sodic non-saline—soils that have an SAR of the 1:5 soil to soil solution ration ($SAR_{1:5}$) between about 3 and 10 but insufficient salt to maintain the clay in a flocculated condition.^[18]
- Very sodic non-saline—Soils having an $SAR_{1:5} > 10$ and very low $EC_{1:5}$ values, which would be highly dispersive and have poor physical properties.^[18]
- Non-saline-alkali—soils for which the ESP is >15 and the ECE is <4 mmhos/cm. The pH usually ranges between 8.5 and 10.^[19]
- Solonetz (in humid regions)—soils that are leached of excess salts and have a well-developed structural B horizon with ESPs from 5% to 20%.^[3]

IDENTIFYING, MAPPING, AND MANAGING HUMID SAS

SASs have primarily been identified and mapped using traditional soil survey methods, which include aerial photo interpretation, the application of local soil-landscape models, soil sampling with the associated field tests, and laboratory analysis. Even when mapping at large scales (1:12,000 to 1:31,680), SASs are often mapped as complexes of SAS–non-SASs because of the irregular shape, relatively small size, and the intricate geographic mix of SAS–non-SASs. Along with the traditional soil survey methods, geophysical techniques, such as electromagnetic induction, have been applied successfully in the identification and detailed mapping of SAS in humid regions^[20,21] and subhumid regions^[22] for both taxonomical classification and land management classification.

Once the SAS have been identified, mapped, and classified, the next step is the management of these areas for specific crops or land uses. An example of the importance of site-specific management and reclamation of SAS for crop production can be seen in a comparison between corn (*Zea mays* L.) and soybean [*Glycine max* (L.) Mer.] yields in SAS and corresponding non-SAS in the humid Midwestern United States.^[23] The two soils are cisne (fine, montmorillonitic, mesic vertic albaqualfs) and huey (fine-silty, mixed, mesic typic natraqualfs)^[13] and are typically mapped as a complex. Under the same climate and management factors, and under non-irrigated conditions, the huey soils (with more than 15% ESP in the subsoil) had an average 18% yield reduction for soybean and an average 38% yield reduction for corn when compared with the cisne soils.

Applications of gypsum ($CaSO_4 \cdot H_2O$), done in conjunction with water table management and tillage, are likely to produce the most success in the reclamation of SAS in humid areas for crop production. These cultural practices should help prevent soil swelling and dispersion; increase porosity, structural stability, hydraulic properties, soil tilth, drainage, and leaching; and reduce dry soil strength.^[24]

CONCLUSION

The adverse effect of exchangeable Na on soil structure and associated soil–water and soil–plant relationships make crop production difficult, even in humid climates. Crop production in these areas provides a unique challenge and opportunity for soil scientists and crop scientists. Through a better understanding of humid region SAS systems, scientists may find better ways to manage these areas to overcome the production problems caused by the high Na concentration and poor physical makeup of SAS.

REFERENCES

1. Kamphorst, A.; Bolt, G.H. Saline and sodic soils. In *Soil Chemistry. A. Basic Elements—Developments in Soil Science 5A*, 2nd Ed.; Bolt, G.H., Bruggenwert, M.G.M., Eds.; Elsevier: New York, 1978; 171–191.
2. Bui, E.N.; Krough, L.; Lavado, R.S.; Nachtergaele, F.O.; Toth, T.; Fitzpatrick, R.W. Distribution of sodic soils: The world scene. In *Sodic Soils—Distribution, Properties Management, and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 19–33.
3. Szabolcs, I. Solonetz soils in Europe. Their formation and properties with particular regard to utilization. In *European Solonetz Soils and Their Reclamation*; Szabolcs, I., Ed.; Akademiai Kiado: Budapest, 1971; 9–34.
4. Richardson, J.L.; Wilding, L.P.; Daniels, R.B. Recharge and discharge of groundwater in aquic conditions illustrated with flownet analysis. *Geoderma* **1992**, *53*, 65–78.
5. Seelig, B.D.; Richardson, J.L. Sodic soil toposquence related to focused water flow. *Soil Sci. Soc. Am. J.* **1994**, *58*, 156–163.
6. Heck, R.J.; Mermut, A.R. Genesis of natriborolls (solonetzic) in a closed basin in Saskatchewan, Canada. *Soil Sci. Soc. Am. J.* **1992**, *56*, 842–848.
7. Pettry, D.E.; Brent, F.V.; Nash, V.E.; Koos, W.M. Properties of natraqualfs in the upper coastal plain of Mississippi. *Soil Sci. Soc. Am. J.* **1981**, *45*, 587–593.
8. Smith, G.D. Intrazonal soils: A study of some solonetz-like soils found under humid conditions. *Soil Sci. Soc. Am. Proc.* **1937**, *2*, 461–469.
9. Wilding, L.P.; Odell, R.T.; Fehrenbacher, J.B.; Beavers, A.H. Source and distribution of sodium solonetzic soils in Illinois. *Soil Sci. Soc. Am. Proc.* **1963**, *27*, 432–438.
10. Lavado, R.S.; Taboada, M.A. Water salt, and sodium dynamics in a Natraquoll in Argentina. *Catena* **1988**, *15*, 577–594.
11. Miller, J.J.; Pawluk, S. Genesis of solonetzic soils as a function of topography and seasonal dynamics. *Can. J. Soil Sci.* **1994**, *74*, 207–217.
12. Horn, M.E.; Rutledge, E.M.; Dean, H.C.; Lawson, M. Classification and genesis of some solonetz (sodic) soils in eastern Arkansas. *Soil Sci. Soc. Proc.* **1964**, *28*, 688–692.
13. United States Soil Survey Staff. *Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting*

- Soil Surveys*; Agriculture Handbook No. 436; U.S. Department of Agriculture-Natural Resources conservation Service, U.S. Government Printing Office: Washington, 1999.
14. FAO-UNESCO. *Soil Map of the World. Revised Legend*; World Soil Resources Report 60, Food and Agricultural Organization: Rome, 1988.
 15. Isbell, R.F. *The Australian Soil Classification*; Australian Soil and Land Survey Handbook; CSIRO: Collingwood, 1996.
 16. VASKhNIL (V.V. Dokuchaev Institute of Soil Science). *Classification and Diagnostics of Soils of the USSR*; Amerind Publishing Co.: New Delhi, 1986. Translated by S. Viswanathan.
 17. SSSA. *Glossary of Soil Science Terms*; Soil Science Society of America: Madison, 1997.
 18. Sumner, M.E.; Rengasamy, P.; Naidu, R. Sodic soils: A reappraisal. In *Sodic Soils—Distribution, Properties, Management, and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 3–17.
 19. United States Salinity Laboratory Staff. *Diagnosis and Improvement of Saline and Alkali Soils*; Agriculture Handbook No. 60; U.S. Department of Agriculture-Agricultural Research Service, U.S. Government Printing Office: Washington, 1954; 160.
 20. Nettleton, W.D.; Bushue, L.J.; Doolittle, J.A.; Endres, T.J.; Indorante, S.J. Sodium-affected soil identification in south-central Illinois by electromagnetic induction. *Soil Sci. Soc. Am. J.* **1994**, *58*, 1190–1193.
 21. Ammons, J.T.; Timpson, M.E.; Newton, D.L. Application of an aboveground electromagnetic conductivity meter to separate natraqualfs and ochraqualfs in Gibson County, Tennessee. *Soil Surv. Horiz.* **1989**, *40*, 66–77.
 22. Bennett, D.L.; George, R.J. Using the EM38 to measure the effect of soil salinity on eucalyptus globules in Southwestern Australia. *Agric. Water Manag.* **1995**, *27*, 69–86.
 23. Olson, K.R.; Endres, T.J. Corn and soybean response to exchangeable sodium in south-central Illinois. *Soil Surv. Horiz.* **1996**, *37*, 49–54.
 24. Shainberg, I.; Sumner, M.E.; Miller, W.P.; Farina, M.P.W.; Pavan, M.A.; Fey, M.V. Use of gypsum on soils: A review. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer-Verlag: New York, 1989; 1–112.

Sodic Soils: Irrigation Farming

David Burrow

Agriculture Victoria, Tatura, Victoria, Australia

Abstract

The area of irrigated soils affected by sodicity is expanding with increasing use of saline–sodic groundwaters and wastewaters. Infiltration and movement of irrigation water through soil are impeded with high exchangeable sodium (Na) percentage and low soil salinity. The accumulation of high Na concentrations in the soil also interferes with calcium and nitrogen uptake in crops. Consequently, management of irrigation water quality with respect to SAR_{iw} and EC_{iw} is critical for soil management. Minimum tillage and the use of cover crops, mulches, gypsum, and dissolved organic polymers in irrigation waters aim to stabilize surface soil aggregates in sodic soils. The modulation of irrigation frequency and volumes can improve crop water availability and leaching of sodic salts from the profile. Slow-wetting irrigation methods reduce the breakdown of surface aggregates. The provision of surface and subsurface drainage is essential for many sodic soils to prevent waterlogging and the accumulation of salts in the root zone. Advances in computer simulation software can improve the management of sodic soils under saline–sodic irrigation.

INTRODUCTION

Sodic soils, which occur naturally in many of the arid and semiarid regions of the world, exhibit poor chemical and physical properties. Soil management and farm operations are often difficult on these soils. Furthermore, farmers are frequently encouraged to use poorer quality saline–sodic wastewaters and groundwaters for irrigation, either alone or blended with surface waters, and this is leading to increased land area affected by sodicity. With increased use of saline–sodic irrigation water and its application to many soil types, farmers and scientists need to better understand the processes underpinning the management of sodic soils for the maintenance or improvement of crop productivity.

PROPERTIES OF SODIC SOILS AFFECTING IRRIGATION FARMING

Hydraulic Properties

The movement of water into the soil following rainfall or surface irrigation, its redistribution within subsurface layers, and its drainage below the root zone are important for productive irrigation farming. Water entry into the soil, measured by the infiltration rate (IR), and through soil layers, measured by the hydraulic conductivity (HC), both decrease with increasing exchangeable sodium (Na) and decreasing soil salinity. For strongly sodic soils, heavy rainfall or low-salinity irrigation may lead to surface waterlogging on flat land or run off on sloping ground.^[1] Similarly, freshwater irrigation of

sodic soils with low HC may lead to shallow depths of wetting, uneven wetting, and low water capacity in plants.^[2] These hydraulic traits, together with lower air-filled porosity and excessive penetration resistance in dry soils, can restrict plant growth.^[3]

The low IR of a strongly sodic soil during irrigation is usually attributed to the formation of a surface seal or crust, which in turn derives from the physiochemical dispersion of clay-sized particles from unstable surface aggregates. These dispersed clay particles migrate with water flow and eventually block conducting pores.

Early studies showed that soil HC varies up to two orders of magnitude, depending on the salt concentration of a percolating solution and soil-exchangeable Na percentage.^[4] Reductions in HC were explained by swelling and dispersion of clay particles, but these factors are modulated by clay mineralogy, sesquioxide and calcium carbonate content, soil texture, organic matter content, pH, soil weathering, and other exchangeable cations.

The effects of soil sodicity on IR and HC, resulting from saline–sodic irrigation, are exacerbated in climatic zones with a marked rainfall season because of the leaching of salts from surface soil horizons. Rainfall with negligible salinity ($EC < 0.1$ dS/m) infiltrates and reduces soil EC below a threshold electrolyte concentration (TEC) in these layers, resulting in clay swelling or dispersion. In the field, these processes are not necessarily detrimental to HC as they may be accommodated by pore geometry and slow wetting rates or circumvented by preferential flow through large cracks. However, the maintenance of soil EC above TEC is an essential first step toward managing sodic soils under irrigation.

Nutrient Status

In sodic soils, high Na to calcium (Ca) and magnesium (Mg) to Ca ratios reduce the amount of Ca available for plant uptake and can induce Ca deficiencies in crops. Data, obtained principally from greenhouse and laboratory studies, indicate that cereal species are sensitive to Ca deficiencies and particularly during seedling growth stages.^[5]

Poor productivity on sodic soils has also been attributed to lowered availability and plant uptake of nitrogen or micronutrients. Nitrogen availability in these soils is restricted under anaerobic conditions,^[6] while micronutrient availability is controlled by adsorption and precipitation reactions.^[7]

The relative solubility of boron (B) and high Na/Ca ratios in some sodic soils also means that B and Na toxicity may contribute to crop production losses. However, the relative importance of nutrient availability and specific ion toxicity on crop production losses for sodic soils is difficult to judge because of the compounding influence of poor soil physical properties on crop growth and yield.

SOIL MANAGEMENT

Specific tillage operations and cropping systems have been designed to overcome some of the poor physical properties of sodic soils.^[1,8] Field grading and raised-bed cropping systems are used in landscapes of low gradient to encourage runoff and limit surface waterlogging. Deep ripping is used to break up massive subsoil structures and for the incorporation of gypsum into subsurface horizons. Shallow tillage, at moisture contents close to the lower plastic limit, is used to generate seedbeds of fine tilth, rather than massive clods. Following crop establishment, zero tillage is typically advocated to minimize mechanical destruction of surface aggregates and subsequent dispersion of clays during irrigation.

Tillage practices, which aim to stabilize surface soil aggregates of an appropriate size distribution, are aided by the use of cover crops, mulch applications, and gypsum applications.^[11] Soil cover crops, such as *Lolium rigidum*,

tend to stabilize topsoil aggregates by means of entwining root systems and the incorporation of root mucilages within the root zone. Mulches are used for the creation of a more favorable soil microclimate that encourages turnover in microbial populations, microbial detritus, and the generation of stabilizing, organic glues. Careful applications of gypsum or other Ca products at the soil surface aim to reduce soil sodicity or maintain soil EC above TEC.

IRRIGATION MANAGEMENT

Water Quality

Water quality guidelines for crop production based on salinity (EC_{iw}) and sodicity (SAR_{iw}) have been developed for infiltration, crop salinity tolerance, and specific ion toxicity.^[8] In contrast, water quality guidelines for soil permeability (HC) and potential dispersion of clay tend to be based on soil chemical properties.

According to infiltration guidelines, calculated on paired EC_{iw} and SAR_{iw} ranges, the quality of irrigation waters used throughout the world (Table 1) should rarely limit their application to land.^[8] However, when the EC of infiltrating water falls below 0.2 dS/m, e.g., as occurs after rainfall, severe infiltration problems are predicted. Crop salinity tolerance guidelines predict severe Na toxicity for sensitive crops, based on $SAR_{iw} > 9$, and severe salinity effects on crop productivity, based on $EC_{iw} > 3$ dS/m. However, these predictions when applied to data in Table 1 only concern irrigation of crops with drainage waters because of their typically high SAR_{iw} and EC_{iw} .

The use of an adjusted SAR_{iw} ($adj\ R_{Na\ iw}$) in Table 1 accounts for the effects of high bicarbonate concentrations in some irrigation waters. Bicarbonate tends to precipitate Ca, and to a lesser extent Mg, as a carbonate solid, with the net result of an increase in SAR_{iw} . The transformations of Ca (and Mg) between soluble, exchangeable, and precipitated forms are modulated by pH and the partial pressure of CO_2 , hence these two factors indirectly affect SAR_{iw} .

Table 1 Mean and range of values for EC_{iw} , SAR_{iw} , and $adj\ R_{Na\ iw}$ of irrigation sources throughout the world.

Measure	Canals	Drains	Groundwaters	Rivers	Wadis	Wells	Wastewaters	All sources
Samples	11	5	4	88	12	115	5	250
Countries	5	3	2	29	3	42	3	54
EC	1.37	4.38	1.45	0.80	2.74	1.78	0.84	1.47
	<i>0.20–4.15</i>	<i>1.06–6.20</i>	<i>0.46–3.80</i>	<i>0.03–7.42</i>	<i>0.30–8.01</i>	<i>0.14–7.93</i>	<i>0.69–1.11</i>	<i>0.03–8.01</i>
SAR	4.75	9.98	1.43	1.87	8.63	5.12	2.46	4.04
	<i>0.60–14.0</i>	<i>3.50–18.0</i>	<i>1.20–1.90</i>	<i>0.10–14.0</i>	<i>0.30–36.0</i>	<i>0.20–49.0</i>	<i>0.40–4.10</i>	<i>0.10–49.0</i>
Adj R_{Na}	3.91	11.74	1.98	1.99	9.3	5.73	2.78	4.42
	<i>0.60–11.2</i>	<i>3.70–22.0</i>	<i>1.30–3.80</i>	<i>0.10–18.0</i>	<i>0.30–38.0</i>	<i>0.20–66.0</i>	<i>0.50–4.70</i>	<i>0.10–66.0</i>

Note: Units: EC_{iw} (dS/m), SAR_{iw} , and $adj\ R_{Na\ iw}$ [(mmol/L)^{0.5}].

^aRange values are shown in italics.

Source: From Ayers & Westcot.^[8]

Irrigation Practices

Specific irrigation practices complement tillage and chemical additions to improve the production potential of sodic soils. These practices include increased frequency of irrigation, preplanting irrigation, and adoption of alternative irrigation systems. Uniform applications of polyacrylamide or finely powdered gypsum to topsoils, for the purpose of stabilizing fragile aggregates, are also best achieved through an initial irrigation.

Increasing the frequency of short-duration irrigations on soils with high initial IR (e.g., cracking clays) and low final IR (due to sodicity) maximizes the periods of high IR to wet the root zone and minimizes the risk of waterlogging. Preplanting irrigation wets the entire rooting depth and if timed correctly allows restoration of soil aeration before crop planting. This technique is especially effective on sodic soils of very low IR. A change from surface irrigation systems (e.g., flood or furrow) to sprinklers or drippers can alleviate waterlogging on low IR soils by matching water delivery rates to the IR. The slow wetting of topsoils also decreases aggregate slaking and clay dispersion under these systems of alternative irrigation.

DRAINAGE OF SODIC SOILS

Drainage is essential for the continued practice of irrigation farming on sodic soils. Surface drainage prevents waterlogging and the development of anaerobic conditions in the topsoil, while adequate subsurface drainage allows leaching of toxic or plant-dehydrating salts from the profile.

For some irrigated regions, the presence of a shallow watertable restricts drainage through a diminished hydraulic gradient. These watertables are frequently saline-sodic and contribute to sodification of soils via capillary transmission of Na salts. Hence, the removal in these regions of subsurface water through drains (open or piped) and pumped bores is essential for Na management. Because the installation of open or piped drains is expensive, the use of cheaper mole drains within subsurface layers has been proposed. However, in unstable sodic soils, mole drains collapse soon and therefore are not generally used.^[9]

Periodic leaching of soil profiles is required for removal of salt or toxic ions. Analytical equations have been developed to predict the leaching requirement for soil sodicity under saline-sodic irrigation.^[10] However, these equations do not fully account for Na-induced changes in IR or HC, which impinge on leaching capability.

Practically, intermittent irrigation is more efficient in salt removal than continuous ponding by promoting diffusion of salts into more favorable leaching pathways. Efficient leaching of sodic soils also requires irrigation with water of low SAR_{iw} and EC_{iw} > TEC in addition to the provision of subsurface drainage for removal of salts.

CONCLUSION

A range of farm management practices on irrigated sodic soils have been briefly described. Adequate control of water quality with respect to SAR_{iw} and EC_{iw} and the provision of drainage to remove excess sodic salts from the root zone are pivotal in controlling soil sodicity. Concurrent management of surface soils with ameliorants (e.g., gypsum, organic polymers, and mulches), cover crops, and appropriate irrigation methods enhances farm operations on these sodic soils at relatively low cost. Adequate modification of drainage and sodicity in subsoils is more costly and therefore less frequently attempted, particularly for cereal and forage crops. Consequently, the control of soil sodicity in irrigated farming remains focused on irrigation water quality, irrigation systems and scheduling, provision of surface runoff, and application of soil ameliorants to the soil surface. The continual emergence of irrigation and soil chemistry simulation software for computers should enhance further scientific research and refine farm management of soil sodicity.

REFERENCES

1. Oster, J.D.; Jayawardane, N.S. Agricultural management of sodic soils. In *Sodic Soils: Distribution, Properties, Management, and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 125–147.
2. Shaw, R.J.; Coughlan, K.J.; Bell, L.C. Root zone sodicity. In *Sodic Soils: Distribution, Properties, Management, and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 95–106.
3. Jayawardane, N.S.; Blackwell, J. The effects of gypsum-enriched slots on moisture movement and aeration in an irrigated swelling clay. *Aust. J. Soil Res.* **1985**, *23*, 481–492.
4. Fireman, M. Permeability measurements on disturbed soil samples. *Soil Sci.* **1944**, *58*, 337–355.
5. Curtin, D.; Naidu, R. Fertility constraints to plant production. In *Sodic Soils: Distribution, Properties, Management, and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 107–123.
6. Cairns, R.R.; Bowser, W.E.; Milne, R.A.; Chang, P.C. Effect of nitrogen fertilization of brome grass on solonetz soil. *Can. J. Soil Sci.* **1967**, *47*, 1–6.
7. Naidu, R.; Rengasamy, P. Ion interactions and constraints to plant nutrition in Australian sodic soils. *Aust. J. Soil Res.* **1993**, *31*, 801–819.
8. Ayers, R.S.; Westcot, D.W. *Water Quality for Agriculture*, FAO Irrigation and Drainage Paper (Rev. 1); Food and Agricultural Organization of the United Nations: Rome, 1985; Vol. 29, 174 pp.
9. Muirhead, W.A.; Christen, E.; Moll, J. *Mole Drains under Slots Reduce Waterlogging and Salinity in an Irrigated Clay Soil of Semi-Arid Australia*, 1994 International Ag. Engineering Conference, Milan, Aug. 29–Sept. 1, 1994. Rep. N94-A-018.
10. Jurinak, J.J.; Suarez, D.L. The chemistry of salt-affected soils and waters. In *Agricultural Salinity Assessment and Management*; Tanji, K.K., Ed.; American Society of Civil Engineers: New York, 1990; 42–63.

Sodic Soils: Physical Properties and Behavior

Hwat Bing So

School of Agriculture and Food Sciences, University of Queensland, St. Lucia, Queensland, Australia

Abstract

The physical properties and behavior of sodic soils are determined not only by the level of exchangeable sodium percentage (ESP) but also by its interaction with the soil solution concentrations generally measured as the electrical conductivity (EC). The principal effect of high ESP is swelling of the clay fraction following wetting, which is exacerbated by low EC and will result in slaking and dispersion and the subsequent blockage of pores. The soils hydraulic and mechanical properties will be adversely affected by these processes, which may be followed by other observable problems in the field such as waterlogging, crusting, and/or hard setting.

INTRODUCTION

When the exchangeable sodium (ES) in the soil is present in sufficient quantities to influence its behavior, the soil is considered to be sodic (see also the entry *Sodic Soils: Formation and Global Distribution*, p. 2032). Exceptions are the sub-plastic soils, which may remain stable and unaffected by high levels of ES.^[1] Although the effect of sodium (Na) on soil behavior increases progressively with the level of ES percentage (ESP) in the laboratory, soils in the field are classified using the concept of a critical level of ESP, wherein soil properties or behavior is likely to be adversely affected by the high ESP level. The most widely used critical levels are an ESP of 15% for U.S. soils^[2] and 6% for Australian soils.^[3] The difference between the two was due to the electrical conductivity (EC) of the water used for testing the soil, during the development of these critical levels.^[4] The original level of ESP >15% selected by workers in the U.S. Salinity Laboratory, California, U.S.A., as the threshold above which the soil structure was adversely affected was based on hydraulic conductivity (HC) measurements using tap water having a threshold electrolyte concentration (TEC) of 3–10 mmol L⁻¹ (EC 0.4–1.3 dSm⁻¹), while that TEC used in Australia was 0.7 mmol L⁻¹ (EC 0.09 dSm⁻¹).^[1] As a result, a much higher ESP was required before degradation was initiated in the California study.

These differences highlight the importance of soil solution concentrations in the manifestation of the effect of ESP on soil behavior. Furthermore, the Australian opinion was based on the analysis of many clay soils, whereas the U.S. opinion was based on the analysis of soils with textures lighter than sandy loam.^[1]

EFFECT OF SODICITY ON SWELLING AND DISPERSION OF CLAYS

In homoionic clay systems, Na-clay swells considerably more than calcium (Ca) clay (Fig. 1), and swelling in

Na-clay increases rapidly as the electrolyte concentration decreases^[5] until clay particles become detached from each other when the electrolyte concentrations become very low and produce spontaneous dispersion of the clay particles.

In normal soils, the clay fraction is associated with a range of cations, which are generally dominated by Ca and magnesium with a smaller complement of Na and potassium (K) ions, and aluminum ions in acid soils with a pH less than 5.5. The principal effect of high sodicity on soils is the swelling of clay fraction following wetting associated with the hydration of Na ions, which leads to slaking of the aggregates and dispersion of the clay size fraction, which consists of clay particles and aggregates <2 microns. The dispersed material is effective in the blockage of pores, leading to a reduction in pore sizes, which in turn affects the hydrologic and mechanical properties and behavior of the soil (see the entry *Slaking, Dispersion, and Crust Formation*, p. 2021). The severity of the effect of dispersion is partly dependent on the clay content of the soil. At lower clay contents (<30%), the soil matrix is sandy with inherently large transmission pores and high HCs. In contrast, soils with higher clay contents (>30%) have a clayey matrix, and the number of transmission pores is determined by the degree of structural development. The same degree of dispersion tends to have a greater effect on soils with a clayey matrix when compared with a soil with a sandy matrix.

The effect of increasing Na on the amount of dispersed clay in Vertisols is shown in Fig. 2.^[6,7] Although it is common to express Na content as ESP and the amount of dispersed clay as dispersion ratio of clay, a better relationship was obtained for dispersible clay (DC) and ES contents, both expressed as a proportion of the bulk soil

$$DC = 0.029 + 0.023(ES); R^2 = 0.77^{***} \quad (1)$$

Furthermore, the inclusion of clay content improves the relationship further, indicating the separate effect of texture on the amount of dispersed material

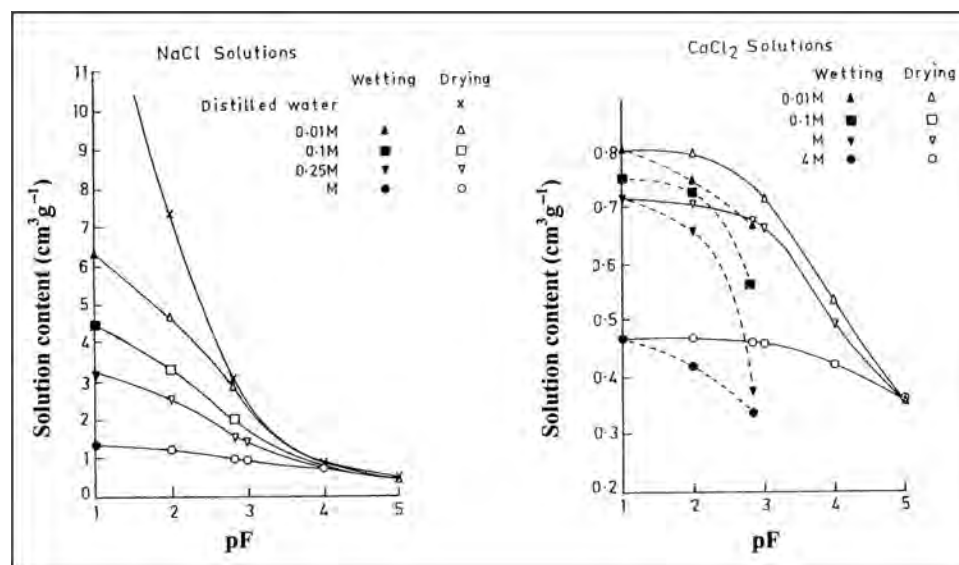


Fig. 1 Swelling of Na and Ca montmorillonite clay, as a function of solution content and water potential expressed as pF.

Source: Adapted from Aylmore & Quirk.^[5]

$$DC = -0.021 + 0.013(ES) + 0.0012(\%Clay); \quad (2)$$

$$R^2 = 0.88^{***}$$

EFFECT OF SODICITY ON THE HYDROLOGIC PROPERTIES AND BEHAVIOR OF SOILS

The soils HC (K) is strongly and negatively correlated to DC content ($R^2 = 0.56^{**}$). Therefore, excessive Na in the soil affects K through its effect on clay dispersion and the

subsequent blockage of pores. K is strongly correlated with ES ($R^2 = 0.44^{**}$) as shown in Fig. 3.^[7] ESP has also been shown to be strongly and negatively correlated with K, independent of clay content and type.^[1]

Critical levels of Na in the soil are rarely shown in the laboratory; however, field assessment of degradation associated with Na is generally based on whether there are detrimental effects on the soil's productivity, which is principally affected by the amount of plant available soil water (ASW) over a crop growth season. ASW is the difference between water input, controlled by the soil infiltration rate, and the water output from the root zone,

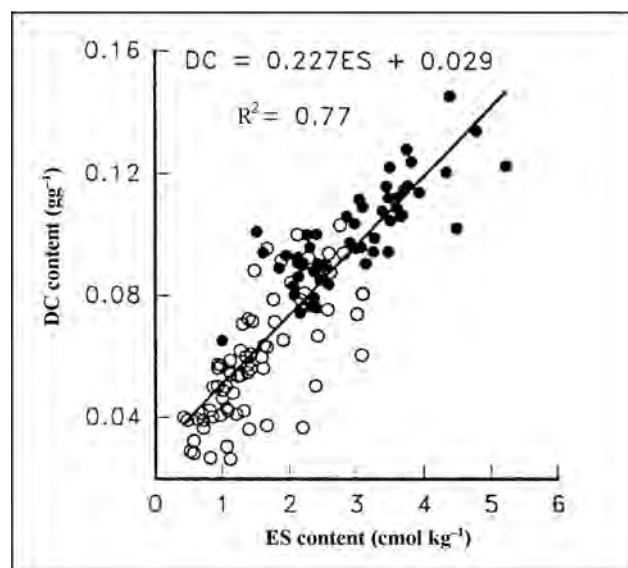


Fig. 2 DC content as a function of ES content on two Vertisols from Queensland, Australia. Solid circles are the Waco soil (clay content 70%) and open circles are the Langlands soil (clay content 54%).

Source: Adapted from So & Aylmore.^[8]

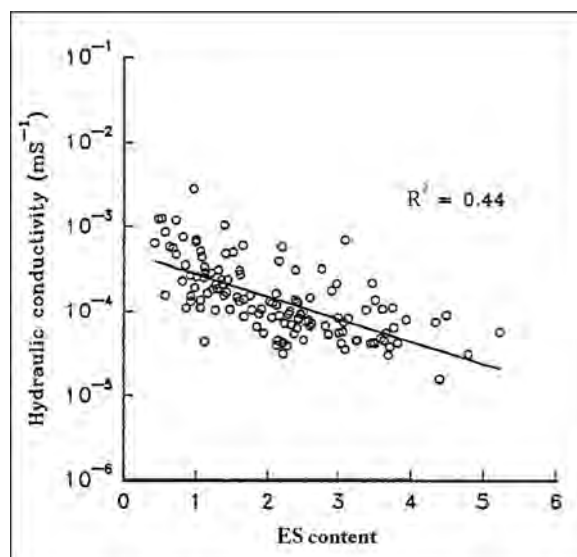


Fig. 3 Soil HC (K) as a function of ES content for two Vertisols from Queensland, Australia.

Source: Adapted from So & Aylmore.^[8]

controlled by the drainage rate, and both rates are affected by the soil HC. As K decreases, the soil infiltration and drainage rates also decrease. Initially, they may slow down the recharge of the soil water storage but remain adequate for the requirement of the plants. However, at the critical ESP, the reduction in K becomes sufficiently large to reduce the amount of water storage such that crop growth becomes affected. On this basis, the critical ESP will vary with the prevailing seasonal rainfall, but this relationship has yet to be quantified. The most common critical level of sodicity used to indicate the probability of such reduction is ESP of 15% in the United States (11) and 6% in Australia.^[3]

A reduction in the rate of drainage in sodic soils will increase the probability of surface waterlogging following rainfall or irrigation, which may significantly reduce germination and establishment of seedlings.

EFFECT OF SODICITY ON THE MECHANICAL PROPERTIES AND BEHAVIOR OF SOILS

Surface waterlogging would reduce soil strength and its ability to support mechanical loads and hence a reduced trafficability. It increases the time when the soil is in a plastic state and not suitable for tillage; therefore, its workability is reduced. On drying, the surface seal forms a crust or a hard setting surface depending on the texture and structural status of the soil. Clayey soils are dominated by highly active clay particles, which interact and form aggregates. The larger the aggregates, the larger are the transmission pores between them and the higher is the HC. However, the presence of excessive Na combined with rapid wetting will result in slaking and dispersion of the immediate surface aggregates, which leads to a

surface seal and a hard surface crust when dry. The higher the ESP, the harder is the crust formed as shown in Fig. 4.^[8] The thickness of the crust depends on the size of the dry aggregates,^[8] while the size of the crust depends on the rate of drying, being larger when the rate of drying is slower. The development of surface crusting leads to a reduction of evaporation and consequently a subsoil below the crust that remains wet for longer periods. When subjected to traffic under such conditions, the load transmitted to this wet subsoil will lead to compaction, while cultivation leads to cloddy surface conditions.

In contrast, fine sandy or silty soils are dominated by particles that are sufficiently large to have very low or no surface activity and, therefore, tend to remain as individual particles. On the other hand, they are small enough to block transmission pores. Therefore, soils with a high content of fine sand and silt in the surface horizon are inherently unstable. Their aggregates readily slake and disperse throughout this horizon when wet. Under wet conditions, these soils are very soft and highly susceptible to deformation and compaction. Because of the low water holding capacity associated with the predominance of coarse particles, these soils rapidly dry into a hard setting soil and are sometimes referred to locally as a “Sunday soil,” being too wet for cultivation on the Saturday and too dry on the Monday. As conditions suitable for cultivation are restricted, sodic soils tend to give rise to cloddy seedbeds. Cultivation when it is too wet will compress the soils into large clods, while cultivation when it is too dry and the tensile strength of soil is high will also result in cloddy seedbeds. Both conditions will require additional cultivation and energy to create a suitable seedbed.

High ESP is associated with a lower liquid and plastic limit and would require more energy to cultivate (see the entry *Slaking, Dispersion, and Crust Formation*, p. 2021) and lead to the perception of a soil with a heavier texture than that indicated by its particle size distribution. The use of gypsum to ameliorate the effect high ESP has on clay soils increases the ease of tillage due to a reduction in soil strength.^[9]

CONCLUSION

The presence of high amounts of Na (high ESP) in the soil gives rise to a variety of physical and mechanical problems in the field through its effect on swelling, slaking, and dispersion.

REFERENCES

1. McIntyre, D.S. Exchangeable sodium, subplasticity and hydraulic conductivity of some Australian soils. *Aust. J. Soil Res.* **1979**, *17*, 115–120.
2. Allison, L.E.; Bernstein, L.; Bower, C.A.; Brown, J.W.; Fireman, M.; Hatcher, J.T.; Hayward, H.E.; Pearson, G.A.;

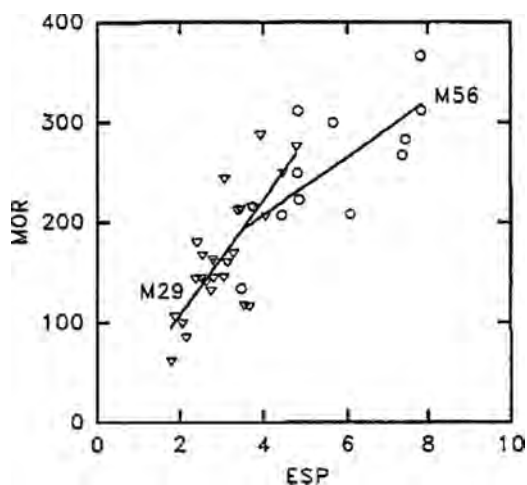


Fig. 4 Modulus of rupture (MOR) as a function of ESP for two clay soils from Western Australia.

Source: Adapted from So & Aylmore.^[8]

- Reeve, R.C.; Richards, L.A.; Wilcox, L.V. Diagnosis and improvement of saline and alkali soils. In *USDA Agricultural Handbook 60*; Richards, L.A., Ed.; Government Printer: Washington, 1954.
3. Northcote, K.H.; Skene, J.K.M. *Australian Soils with Saline and Sodic Properties*; CSIRO Austr. Soils Publication No. 27; CSIRO: Melbourne, 1972.
 4. Sumner, M.E. Sodic soils: New perspectives. *Aust. J. Soil Res.* **1993**, *31*, 683–750.
 5. Aylmore, L.A.G.; Quirk, J.P. The structural status of clay systems. *Clay. Clay Miner.* **1962**, *9*, 104–130.
 6. Cook, G.D.; So, H.B.; Dalal, R.C. Structural degradation of two vertisols under continuous cultivation. *Soil Till. Res.* **1992**, *24*, 47–64.
 7. So, H.B.; Cook, G.D. The effect of slaking and dispersion on the hydraulic conductivity of clay soils. *Catena Supp.* **1993**, *24*, 55–64.
 8. So, H.B.; Aylmore, L.A.G. How do sodic soils behave? The effects of sodicity on soil physical behavior. *Aust. J. Soil Res.* **1993**, *31*, 761–777.
 9. McKenzie, D.C.; So, H.B. Effect of gypsum on vertisols of the Gwydir Valley, New South Wales. 2. Ease of tillage. *Aust. J. Exp. Agr.* **1989**, *29*, 63–67.

BIBLIOGRAPHY

1. Collis-George, N.; Greene, R.S.B. The effect of aggregate size on the infiltration behavior of a slaking soil and its relevance to ponded irrigation. *Aust. J. Soil Res.* **1979**, *17*, 65–73.
2. Guarnieri, A.; Fabbri, A.; Molari, G. Influence of sodicity and salinity on the mechanical properties of two Italian soils. *Biosyst. Eng.* **2005**, *91*, 239–243.

Sodic Soils: Processes, Characteristics, and Classification

Pichu Rengasamy

Department of Soil and Water Science, University of Adelaide, Waite, South Australia, Australia

Abstract

Land managers with paddocks that are prone to waterlogging, poor crop or pasture emergence, gully erosion, or tunnel erosion may be experiencing the effects of sodicity. Sodic soils are formed by the adsorption of Na^+ by the negatively charged sites on soil particles, particularly soil clays from soil solutions containing free sodium (Na) salts such as sodium chloride, sodium carbonate, sodium bicarbonate, and sodium sulfate. The soils are considered to be sodic when the free salts are leached from the soil layers and only exchangeable Na remains adsorbed on soil particles. If free salts are also present, the soils become saline-sodic. Sodic soils are generally found in arid and semiarid regions with high evapotranspiration and low rainfall associated with low leaching that are responsible for salt accumulation.

CHARACTERISTICS OF SODIC SOILS

The negative charge, which depends on the type and amount of clay materials, plays a vital role in the adsorption of Na. Negative charge in soils increases with the content of clay minerals in the order smectites > illites > kaolinites. Soils dominant in positive-charged sites such as Oxisols containing oxides of aluminum and iron, and low pH soils generally have negligible amounts of exchangeable Na. Soils with pH-dependent charge minerals increase their negative charge with increasing pH. The more the negative charge, the more is the adsorption of Na in soils.

Although soil sodicity is a result of chemical reaction (cation exchange) with the salts, it mainly affects soil physical properties and thus plant growth and productivity. It is possible that the release of high amounts of Na^+ in soil solutions can be toxic, which are encountered in saline-sodic soils. Sodic soils have an extremely poor physical condition (Table 1), leading to an inadequate balance between water and air regimes within the soil. The imbalance stems from restricted water acceptance and transmission properties, which result in the soil being too wet or dry for much of the time, and leads to poor root development and crop growth. In addition, sodic soils are difficult to cultivate and have poor load-bearing characteristics. The lack of structural stability in these soils promotes soil hardening throughout soil profile and seal and crust formation at the soil surface, resulting in soil erosion and pollution of water bodies.^[1–3] Poor drainage in sodic soils also causes secondary salinity in subsoil layers.^[4]

SOIL PROCESSES

Swelling and dispersion of clay particles on wetting are the major processes responsible for the deterioration of physical behavior of sodic soils. In saline-sodic soils with high electrolyte concentration, swelling is minimal and clay dispersion is absent. Both swelling and dispersion behavior are governed by the balance between attractive and repulsive forces, arising from intermolecular and electrostatic interactions between solution and the solid phases in the soil. The distinction between saline and sodic soils arises because these forces vary, depending on whether the soil solution is concentrated (saline) or diluted with a high proportion of Na to divalent ions in the solid exchange phase to cause swelling and dispersion (sodic).

Soil scientists have used models based on pure clay minerals involving Lifshitz-van der Waals, ion correlation, hydration, and electrical diffuse double layer forces generated between colloidal particles suspended in water to explain sodic soil behavior.^[5] But soil clay systems that are complex, heterogeneous intergrowths of different clay structures intimately associated with organic and biopolymers do not behave in the same way as their pure clay mineral counterparts. Thus, classical theories of colloidal behavior such as Derjaguin-Landau-Verwey-Overbeek (DLVO) theory^[6] may not satisfactorily explain the behavior of sodic soils.

The processes that occur during the initial wetting of dry aggregates, resulting in swelling to the final stage of aggregate disintegration and leading to dispersion of soil clays when completely wet, are important in understanding

Table 1 Example of properties of sodic soils compared to an ideal soil that is highly productive.

Properties	Sodic soil	Ideal soil
PH _{1:5} (water)	9.2	6.0–8.0
Electrical conductivity (EC) _{1:5} (dS/m)	0.2	<0.4
Organic carbon (%)	0.3	>1.0
Na adsorption ratio (SAR) _{1:5}	9.9	<3.0
Spontaneously dispersed clay (%)	8.7	0
Hydraulic conductivity at saturation (mm/day)	4.0	>80.0
Penetrometer resistance (MPa) at 100 kPa suction	3.8	<2.0
Aeration porosity (%)	5.6	>15
Bulk density (Mg/m ³)	2.2	<1.5
Non-limiting water range (mm ³ /mm ³)	0.38–0.42	0.1–0.5

Source: From Rengasamy.^[7]

sodic soil behavior. The polar nature of water molecules and solvation reactions with the solid phase is primary factors in causing swelling and dispersion. Clay particles in dry soil aggregates are bound together by inorganic and organic compounds involving several types of bonding, which produce strong, attractive pressures of the magnitude of megapascals.^[8] The water stability of an aggregate depends on the strength and persistence of these linkages in the presence of water molecules, which, in turn, are functions of the type of bonding. Bond strength in the presence of water generally decreases in the order: covalent, hydrophobic, Lifshitz-van der Waals, coordination complexing, hydrogen, and finally ionic bonds. In contrast to covalent bonds, ionic bonds are readily solvated by water molecules. Similar to pure Na compounds, Na–clay linkage is easily solvated and ionic bond is broken, whereas calcium and magnesium ions are linked to clay particles by polar covalent (combination of covalent and ionic) bonds, with decreased ionicity compared to Na linkage. The degree of covalency in a bond involving metal cations depends on ionization and ionic potentials.^[3] The tendency to form covalent bonding increases in the order: Na⁺, K⁺, Mg²⁺, Ca²⁺, Fe³⁺, and Al³⁺, for example. Thus, both the type of cation and the nature of clay ligand determine the ionicity of the clay–cation–clay bonds.

Highly ionic bonding in sodic clays leads to extensive hydration and swelling with increasing water content, and finally, with separation of linkages on water saturation, clay particles disperse spontaneously in water, whereas in calcium, or magnesium-saturated clays with polar covalent bonding, limited hydration leads to limited swelling without any separation (or dispersion) of clay particles. Only on mechanical agitation, these clays can disperse. The flocculation of dispersed clay particles is brought about by the addition of electrolyte, which as a result of

its osmotic effect causes dehydration of clay water system, thereby bringing clay particles together. Therefore, saline–sodic soils do not have severe deterioration of physical properties. The order of flocculating power is Ca²⁺ > Mg²⁺ > K⁺ > Na⁺. Thus, gypsum, a calcium compound, is very effective in reclaiming sodic soils. In addition to its flocculating effect, gypsum also promotes removal of Na from clays by way of cation (calcium) exchange.

CLASSIFICATION

There is a need for a uniform system of classification of salt-affected soils that distinguishes between saline and sodic conditions and is useful for soil management. The international nomenclature^[9] includes the terms “sodic,” “alkali,” “solonchak,” “solonetz,” and “solodized solonetz,” and the complex interrelationship between them makes comparison between sodic soils difficult. The classification of saline and sodic soils devised by the U.S. Department of Agriculture widely followed in many countries is as follows:

Saline, non-sodic soils – Exchangeable sodium percentage < 15

EC in saturation paste (ECe) > 4dS/m

Sodic, non-saline – ESP > 15 and ECe > 4dS/m

Non-saline, non-sodic – ESP < 5 and ECe < 4dS/m

The effects of exchangeable Na on soil physical behavior vary from soil to soil and are influenced by several factors such as electrolyte concentration, pH, organic matter, biopolymers, and aggregate stability in water. Therefore, the definitions used on the basis of ESP vary according to practical experience. The following classes have been proposed^[10] on the basis of physical behavior of sodic soils:

Class 1: Dispersive soils. Soils that disperse spontaneously without shaking will have severe problems associated with crusting, reduced porosity, etc. even when subjected to minimum mechanical stress, e.g., under zero tillage.

Class 2: Potentially dispersive soils. Soils that require inputs of mechanical energy (raindrop impact and tillage) to bring about dispersion will experience soil physical problems when mechanically disturbed.

Class 3: Flocculated soils. Soils that contain more than the minimum electrolyte concentration required for flocculation of clays (or prevention of dispersion of clays) will present few physical problems, but salts could be excessive and limit productivity.

Measurements of clay dispersion along with ESP or SAR, EC, and pH will be necessary for managing saline and sodic soils. Based on these principles, a detailed proposal^[11] on

classification of salt-affected soils is available, but this has to be tested for practical applications.

CONCLUSION

Although soil sodicity is a result of chemical exchange reactions of the accumulated Na salts, it mainly affects soil physical properties and thus plant growth and productivity. The physico-chemical processes involved in sodic soils are briefly described. The classification of sodic soils on the basis of soil characteristics and dispersion/flocculation behavior is also discussed.

REFERENCES

1. Quirk, J.P. Some physico-chemical aspects of soil structural stability: A review. In *Modification of Soil Structure*; Emerson, W.W., Bond, R.D., Dexter, A.R., Eds.; John Wiley: New York, 1978; 3–16.
2. Shainberg, I.; Letey, J. Response of soils to sodic and saline conditions. *Hilgardia* **1984**, *52*, 1–57.
3. Rengasamy, P.; Sumner, M.E. Processes involved in sodic behaviour. In *Sodic Soils: Distribution, Properties, Management, and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 35–50.
4. Shaw, R.J.; Coughlan, K.J.; Bell, L.C. Root zone sodicity. In *Sodic Soils: Distribution, Properties, Management, and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 95–106.
5. Quirk, J.P. Interparticle forces: A basis for the interpretation of soil physical behaviour. *Adv. Agron.* **1994**, *53*, 121–183.
6. van Olphen, H. *An Introduction to Clay Colloid Chemistry*; John Wiley: New York, 1977.
7. Rengasamy, P. Sodic soils. In *Methods for Assessment of Soil Degradation*; Lal, R., Blum, W.H., Valentine, C., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1998; 265–277.
8. Rengasamy, P.; Olsson, K.A. Sodicity and soil structure. *Aust. J. Soil Res.* **1991**, *31*, 821–837.
9. Szabolcs, I. *Salt-Affected Soils*; CRC Press: Boca Raton, 1989.
10. Rengasamy, P.; Greene, R.S.B.; Ford, G.W.; Mehanni, A.H. Identification of dispersive behaviour and the management of red-brown earths. *Aust. J. Soil Res.* **1984**, *22*, 413–431.
11. Sumner, M.E.; Rengasamy, P.; Naidu, R. Sodic soils: A reappraisal. In *Sodic Soils: Distribution, Properties, Management, and Environmental Consequences*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 3–17.

Sodic Soils: Reclamation

Jock Churchman

Land and Water, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Adelaide, South Australia, Australia

Abstract

Two main mechanisms are involved in the reclamation of sodic soils: 1) improvement in water flow; and 2) displacement of exchangeable sodium ions.

IMPROVEMENT IN WATER FLOW

Sodic soils show a low permeability to water. Their permeability can be improved quite rapidly by increasing electrolyte concentrations^[1,2] (the “electrolyte effect”). Ions in solution tend to inhibit the swelling and dispersion of fine clay particles by compressing the electric double layers of adjacent particles (Fig. 1). The volume around the particle surfaces within which repulsion takes place is reduced, ultimately to zero.

It has been found^[2] that there is no swelling and dispersion when the total electrolyte concentration exceeds a critical level, i.e., the “threshold electrolyte concentration” that depends on soil properties.^[1,4]

DISPLACEMENT OF EXCHANGEABLE SODIUM (NA) IONS

The deleterious effects of exchangeable Na reflect the more extensive hydration of the Na ion than that of other common exchangeable cations on soil particles (Fig. 2). Exchangeable Na, which forms an ionic association with the clay surface, causes hydration, dissociation of particles, and hence swelling and dispersion.^[5] By contrast, exchangeable calcium (Ca) forms polar covalent bonds with the clay surface.^[5] Ca shows only limited hydration and dissociation of particles and therefore causes only limited dispersion and swelling.^[5] Na⁺ on exchange sites is replaced by Ca²⁺ for the longer-term remediation of sodic soils.

CHEMICAL AMELIORANTS

Gypsum

Soluble sources of Ca ions are the most suitable sodic soil ameliorants. While gypsum is used most often, not all sources of gypsum are similarly effective or similarly suitable for each specific problem from sodicity. It was found^[1,6] that only “by-product” gypsum, as a surface

application, could prevent crusting. Mined gypsum was too slowly soluble for the purpose.^[1] Gypsum is a by-product of many industrial processes.^[6] Furthermore, gypsums from different mines can differ quite markedly in purity.^[7] They can also differ in dissolution rate. A finer particle size generally provides a greater solubility.^[6] As well, Na salts associated with gypsum (Table 1)^[7] decrease its efficiency for amelioration.^[8]

Lime

Lime has also been used as an ameliorant for sodic soils.^[7,9,10] While generally less soluble than gypsum, lime may provide a useful incidental source of Ca ions when used to raise the pH of acid sodic soils. It may provide a cheaper alternative to gypsum or else be used in association with gypsum to extend the lifetime of the Ca reserves.^[7] Because of its increased solubility in the presence of Na salts, it may be more effective than gypsum for the reclamation of saline sodic soils. Lime also contributes less to the salt load than gypsum.^[10]

Reclamation of Calcareous Sodic Soils

Calcareous sodic soils have an abundance of Ca, but it is present as calcium carbonate (CaCO₃) and hence is insoluble at the high pH of these soils. Their sodicity is often overcome through acid additions,^[11,12] either directly or indirectly. Additions of elemental sulfur, pyrite, iron and aluminum sulfates^[11,13] and also organic matter^[14] provide acid through microbial activity for indirect acidification. Plant growth can also help dissolve native CaCO₃.^[13]

Miscellaneous Chemical Ameliorants

Sodic soils may be reclaimed with soluble sources of Ca that include waste materials, e.g., acidic cottage cheese whey,^[1] acid resin by-products of the oil industry, and also CaCl₂ as an industrial by-product.^[4] Addition of polymers can also increase the permeability of soils.^[1]

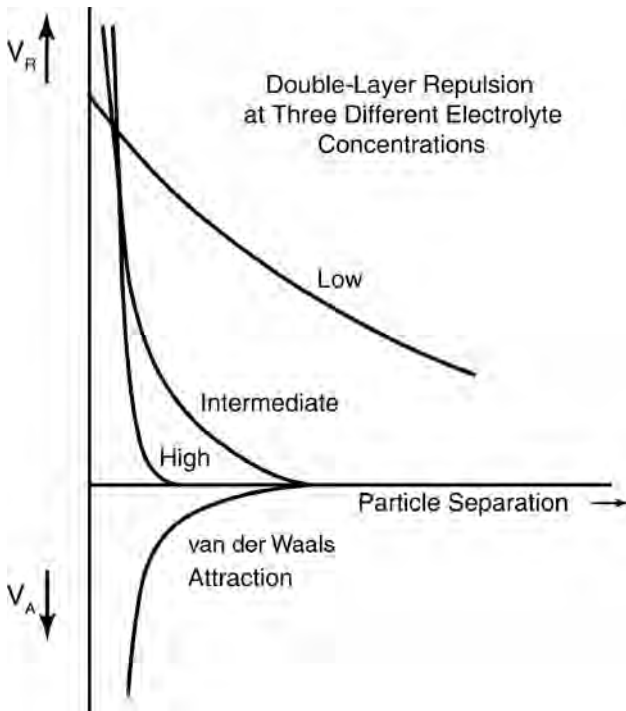


Fig. 1 Repulsive and attractive energy as a function of particle separation at three electrolyte concentrations.

Source: From van Olphen.^[3]

ROLE OF BIOLOGY AND ORGANIC MATTER

Effects of Plant Growth

Rice culture ameliorates sodic soils through the build-up of carbon dioxide and hence dissolution of CaCO_3 , among other effects.^[11] Among crops, rice is particularly tolerant

to sodicity, and some grasses are also highly tolerant^[11,13] (Table 2).

Ameliorants are readily leached out of soils, so losing the benefit of the electrolyte effect. In addition, Na^+ from Na salts often present in soils can replace exchangeable Ca^{2+} . Therefore, long-term reclamation probably requires increased plant growth and hence enhanced biological activity^[14] and stabilization of soil structure. This occurs partly through transient binding agents, e.g., roots, hyphae, polysaccharides, and hydrophobic binding,^[14] and partly through associations between soil organic matter and Ca^{2+} in the soil system. Organic matter has a stronger affinity for Ca than for Na,^[14] so it is most effective when used with gypsum or lime to help restore a stable structure to soils.^[14] There is a resulting increase in stable pores for the transmission of water and nutrients, root growth, and soil stability.

Effects of Additions of Organic Matter

Due to its stabilizing effect on soil structure, organic matter can also help to reclaim sodic soils when added to soils, even though additions of organic matter can sometimes also enhance their dispersion, with resulting physical problems.^[14] The different effects vary with types of soils and also of organic matter.^[14] The addition of readily decomposable organic matter tends to reduce the pH of calcareous sodic soils and hence to aid their reclamation.^[14]

STRATEGIES FOR THE RECLAMATION OF SODIC SOILS

Reclamation without the Addition of Ameliorants

Reclamation can be achieved slowly without ameliorants by many cycles of irrigation and cropping.^[11] As well,

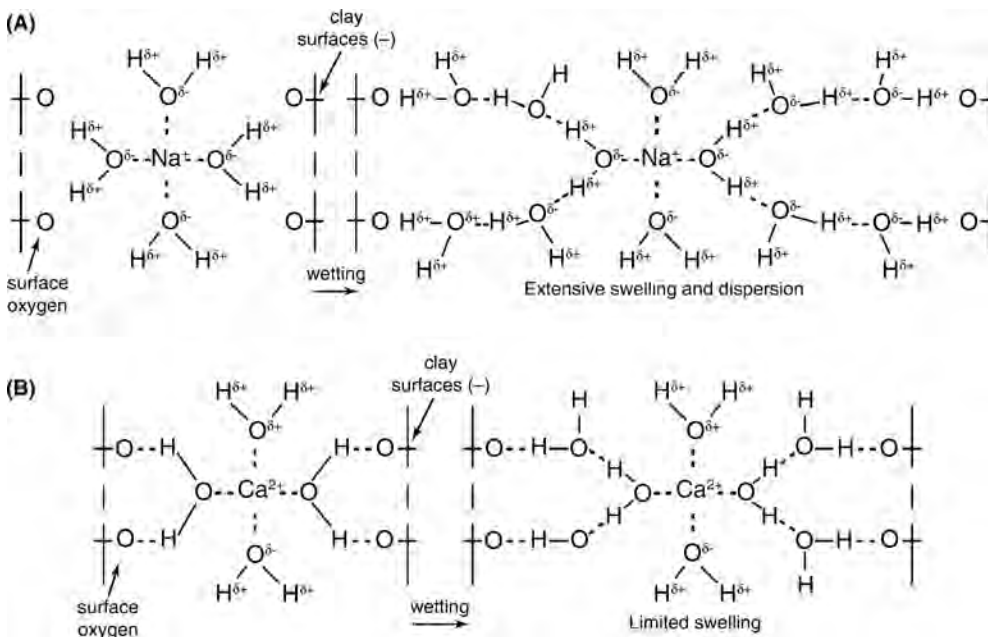


Fig. 2 Schematic representation of the effect of wetting and nature of bonding between cations and clay surfaces on swelling and dispersion: (A) Na^+ aquo ion linking clay particles by ionic bonding, (B) Ca^{2+} aquo ion linking clay particles by polar covalent bonding. Water molecules are linked to cations by hydrogen bonding. **Source:** From Rengasamy & Sumner.^[5]

Table 1 Composition of some mined gypsum.

Sample	Fe ₂ O ₃ (%)	MnO (%)	TiO ₂ (%)	CaO (%)	K ₂ O (%)	SO ₃ (%)	P ₂ O ₅ (%)	SiO ₂ (%)	Al ₂ O ₃ (%)	MgO (%)	Na ₂ O (%)	ZrO ₂ (%)	Sr (%)	SUM (%)	Org C (%)	CaCO ₃ (%)
Plant life	0.48	0.010	0.10	25.7	0.260	30.6	0.057	18.9	1.16	0.47	0.18	<0.001	0.271	78.1		
Neindorf	1.54	0.021	0.19	23.7	0.662	28.9	0.066	17.6	3.32	1.64	0.28	<0.001	0.300	78.3	0.41	1.8
Salt lake	0.03	<0.003	0.02	33.5	0.011	29.7	0.051	0.20	0.07	<0.009	0.05	<0.001	0.241	73.8	0.53	3.4
Austral Pacific	0.26	0.006	0.04	22.6	0.70	28.5	0.047	30.2	0.56	0.04	<0.03	<0.001	0.115	82.4	0.07	0.1
Agra unsieved	0.15	0.004	0.03	31.2	0.093	39.0	0.050	4.50	0.61	0.06	<0.03	<0.001	0.341	76.0		
Agra sieved (A)	0.12	0.010	0.03	31.0	0.067	39.5	0.049	3.90	0.40	0.04	<0.03	<0.01	0.845	75.9	0.02	0.1
Agra sieved (B)	0.11	0.012	0.03	31.0	0.067	39.4	0.053	3.80	0.28	0.06	<0.03	<0.001	0.847	75.6	0.1	
Cresco	0.15	0.004	0.02	31.0	0.089	39.2	0.071	4.20	0.37	0.07	0.04	<0.001	0.357	75.5	0.03	0.1
Top gypsum	0.32	0.006	0.10	23.4	0.340	28.5	0.057	26.3	1.12	0.35	0.19	0.001	0.194	80.9		
Jomoco (A)	0.46	0.010	0.08	28.0	0.237	33.9	0.286	11.8	1.02	0.68	0.17	<0.001	0.403	77.0	n.d.	n.d.
Jomoco (B)	0.44	0.007	0.07	27.9	0.232	33.7	0.277	12.4	1.01	0.67	0.14	<0.001	0.402	77.2	n.d.	n.d.

Source: From Naidu & Merry.^[7]

Table 2 Relative tolerance of crops to sodicity.

ESP range	Crops ^a
10–15	Safflower, mash, pea, lentil, pigeon-pea, and curd bean
16–20	Bengal gram and soybean
20–25	Groundnut, cowpea, onion, and pearl millet
25–30	Linseed, garlic, and guar
30–50	Indian mustard, wheat, sunflower, berseem, hybrid napier, and guinea grass
50–60	Barley, sesbania, saftal, and panicums
60–70	Rice and para grass
70+	Karnal, rhodes, and bermuda grasses

^aYields are about 50% of the potential yields in the respective sodicity ranges.

Source: From Gupta & Abrol^[11] and Oster & Jayawardene.^[13]

underlying calcareous or gypsiferous layers may be incorporated into upper parts of the soil profile through deep plowing.^[4]

Optimal Supply of Ameliorants

The effectiveness of gypsum can be increased by its placement in cracks for their stabilization so that they remain open to conduct water, air, and plant nutrients when soils become wet.^[1] Water flow to subsoil layers, aeration, and hence higher crop yields were achieved by incorporating gypsum in narrow tilled slots.^[13] More generally, gypsum is often incorporated into subsoil layers by tillage.

Ameliorants may also be used in association with other additions. Only a little gypsum applied with acidifying nitrogenous fertilizers led to effective reclamation of a saline-sodic soil.^[1] This combination confers immediate benefits through a higher electrolyte concentration and also medium-term benefits through an increased supply of exchangeable Ca ions. Enhanced plant growth from fertilizer use provides long-term benefits.

Supply of Water for Reclamation

Ameliorants are only effective when dissolved. A supply of water and its flow through the profile are required for effective amelioration. Irrigation and/or rainfall may supply water, but the quality of water available for irrigation can vary. Gypsum can be supplied in electrolyte-free irrigation water, as in parts of California.^[13] Saline-sodic irrigation water, including water commonly derived from groundwater sources, may exacerbate the deleterious effects of sodicity when its application during the dry season is followed by rains, as can occur in Israel.^[13] In this case, (by-product) gypsum is applied to the surface prior to the rainy season.^[13] Adequate drainage to beyond the root zone should accompany irrigation in order to

leach displaced electrolyte, but this should not cause increased salinity in underlying groundwater.^[12]

As long as there is provision for the disposal of the salt water, dilution with salt water^[3,12] is also useful in sodic soil reclamation. Its immediate effect is to increase the hydraulic conductivity of sodic soils by the electrolyte effect, but long-term reclamation will occur provided it also contains sufficient Ca (Ca : total cations ≥ 0.3). This can be added as gypsum.

Rate of Supply of Ameliorants

All additives increase costs, so their optimal use is required. The cost of transporting gypsum often limits application rates. Recommendations from field trial results of the most effective application rates for ameliorants are necessarily specific to soil type and climate, among other factors. Nonetheless, both simulations^[9,15,16] and laboratory studies^[17] have shown that the solubility of gypsum increases as the proportion of exchangeable Na on the soil increases. They have also shown that gypsum is more effective when mixed throughout the soil than when applied to the soil surface alone. Simulations^[10] have shown that reclamation is primarily limited to the depth interval in which gypsum was applied (Fig. 3A). They have also shown (Fig. 3B) that the electrical conductivity (EC) of the solution within and below the gypsum-amended layer decreases as the exchangeable sodium percentage (ESP) within that layer decreases.

Nonetheless, calculations of gypsum requirements have been based on 1) the exchange of Na by Ca, rather than the attainment of a sufficiently high electrolyte concentration EC to maintain a low permeability, and 2) assumptions of chemical equilibrium. The gypsum requirements for the achievement of a suitable permeability by the electrolyte effect also depend on the inherent EC

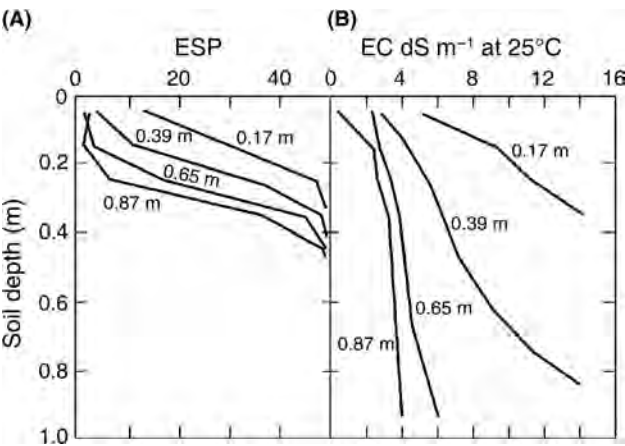


Fig. 3 Computer-model results (ESP and EC) for reclamation of a soil (initial ESP = 50; CEC = 200 mmol_c kg⁻¹) with gypsum and water (EC = 0). Numbers next to each line are depths of applied water.

Source: From Oster & Frenkel.^[10]

of the soil solution.^[12] Because mineral dissolution can maintain a high EC throughout most of the soil profile, gypsum often needs to be added to the soil surface only, mainly to prevent crusting. Assumptions of equilibrium, while strictly invalid, have probably been useful for the calculation of gypsum requirements because of low flow rates of percolating solutions and high surface areas of gypsum particles.^[11] Models of soil and water flow, e.g., UNSATCHEM,^[17] which consider equilibrium and kinetic expressions for ion exchange and the dissolution and precipitation of minerals and also ameliorants, may enable more robust calculations of requirements for ameliorants.

Ultimately, the main purpose of reclaiming sodic soils is to obtain the maximum possible improvement of yield of the particular crop in relation to the cost of ameliorant applied. Work in the Australian sugar industry^[18] has established a quantitative relationship between percentage increase in ESP and consequent loss in sugarcane yield. Sugarcane yield decreased by 1.5–2.1 tonne ha⁻¹ for every 1% increase in ESP.^[18] This relationship forms the basis of a cost–benefit analysis for calculating gypsum application rates from the ESP, the quality of the gypsum used, the quality of irrigation water, and economic factors. These include the cost of gypsum and its application, the price for the product (sugarcane), and the discount rate.^[19] Clearly there are opportunities for more practical calculations of this kind to be carried out for other crops and in other soil and climatic environments.

REFERENCES

- Sumner, M.E. Sodic soils: new perspectives. *Aust. J. Soil Res.* **1993**, *31* (6), 683–750.
- Quirk, J.P.; Schofield, R.K. The effect of electrolyte concentration on soil permeability. *J. Soil Sci.* **1955**, *6* (2), 163–178.
- van Olphen, H. *An Introduction to Clay Colloid Chemistry*; Wiley: New York, 1963.
- Loveday, J. Amendments for reclaiming sodic soils. In *Soil Salinity Under Irrigation: Processes and Management*; Shainberg, I., Shalhevet, J., Eds.; Springer: Berlin, 1984; 353–374.
- Rengasamy, P.; Sumner, M.E. Processes involved in sodic behaviour. In *Sodic Soils*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 35–50.
- Shainberg, I.; Sumner, M.E.; Miller, W.P.; Farina, M.P.W.; Pavan, M.A.; Fey, M.V. Use of gypsum on soils: A review. *Adv. Soil Sci.* **1989**, *9*, 1–112.
- Naidu, R.; Merry, R.H.; Churchman, G.J.; Wright, M.J.; Murray, R.S.; Fitzpatrick, R.W.; Zarcinas, B.A. Sodicity in south Australia. A review. *Aust. J. Soil Res.* **1993**, *31* (6), 911–929.
- Rengasamy, P.; Olsson, K.A. Irrigation and sodicity. *Aust. J. Soil Res.* **1993**, *31* (6), 821–837.
- Levy, G.J.; Shainberg, I.; Miller, W.P. Physical properties of sodic soils. In *Sodic Soils*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 77–94.
- Oster, J.D.; Frenkel, H. The chemistry of the reclamation of sodic soils with gypsum and lime. *Soil Sci. Soc. Am. J.* **1980**, *44* (1), 41–45.
- Gupta, R.K.; Abrol, I.P. Salt-affected soils: their reclamation and management for crop production. *Adv. Soil Sci.* **1990**, *11*, 223–288.
- Oster, J.D.; Shainberg, I.; Abrol, I.P. Reclamation of salt-affected soil. In *Soil Erosion, Conservation and Rehabilitation*; Agassi, M., Ed.; Marcel Dekker: New York, 1996; 315–352.
- Oster, J.D.; Jayawardene, N.S. Agricultural management of sodic soils. In *Sodic Soils*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 125–147.
- Nelson, P.N.; Oades, J.M. Organic matter, sodicity and soil structure. In *Sodic Soils*; Sumner, M.E., Naidu, R., Eds.; Oxford University Press: New York, 1998; 51–75.
- Tanji, K.K.; Deverel, S.J. Simulation modeling for reclamation of sodic soils. In *Soil Salinity Under Irrigation: Processes and Management*; Shainberg, I., Shalhevet, J., Eds.; Springer: Berlin, 1984; 238–251.
- Frenkel, H.; Gerstl, Z.; Alperovitch, N. Exchange induced dissolution of gypsum and the reclamation of sodic soils. *J. Soil Sci.* **1989**, *40* (3), 599–611.
- Suarez, D.L.; Šimunek, J. UNSATCHEM: Unsaturated water and solute transport model with equilibrium and kinetic chemistry. *Soil Sci. Soc. Am. J.* **1997**, *61*, 1633–1646.
- Nelson, P.; Ham, G. Exploring the response of sugar cane to sodic and saline conditions through natural variation in the field. *Field Crop. Res.* **2000**, *66* (3), 245–255.
- Nelson, P.; Fitzgerald, T.; Swan, G.; Brennan, L. *Gypsy, Discounted Cash Flow Analysis for Application of Gypsum to Sodic Soils Under Sugarcane: Manual*; CRC for Sustainable Sugar Production: Townsville, 2000.

Sodicity: Subsoil

Kwong Yin Chan

Organic Wastes Recycling Unit, New South Wales Department of Primary Industries, Richmond, New South Wales, Australia

Abstract

Subsoil sodicity refers to the problems associated with the presence of excessive concentration of exchangeable sodium in the subsoil. Many soils with subsoil sodicity are characterized by low hydraulic properties and plant available water capacity, salinity, alkalinity, and deficiencies of micronutrients and toxic ions.

Large areas in different parts of the world with subsoil sodicity are marginal land with low productivity. However, based on available information, to what extent the low productivity is directly related to subsoil sodicity is not clear. Critical limits and criteria commonly used to assess sodicity of surface soils are not necessarily applicable to the subsoils. The ameliorative techniques are reviewed and further research needs are discussed.

INTRODUCTION

Subsoil sodicity refers to the problems associated with the presence of excessive concentration of exchangeable sodium (Na) in the subsoil. The latter is the layer below the plow layer, i.e., the Ap horizon or its equivalent in the undisturbed soil profile. In the literature, most of the information on sodicity has been derived from surface soils and often the distinction between surface and subsoil sodicity is not made clear. For example, sodic soils are defined as soils with exchangeable sodium percentage (ESP) exceeding 6% in any part of the top meter of the soil profile.^[1] Using the critical ESP value of 6% or 15% as criterion, sodic subsoils are widespread in many parts of Australia, Indo-Gangetic Plains in India, parts of Canada, and the United States.^[2] However, critical limits and criteria commonly used assessing sodicity of surface soils^[1–3] are not necessarily applicable to subsoils.

CHARACTERISTICS AND EFFECTS ON SOIL FUNCTIONS

Many soils with subsoil sodicity are characterized by the following properties, some of which are not directly/indirectly results of high concentration of exchangeable Na:

Low Hydraulic Properties and Plant Available Water Capacity (PAWC)

Many soils with high concentration of exchangeable Na subsoils have impeded infiltration and drainage as well as low plant available water.^[2,3] In surface soils, the poor physical conditions of sodic soils are caused by the processes of swelling and dispersion of clay particles originally bound in

the structural units (aggregates and peds) occurring at the surface.^[4,5] In contrast, low permeability of sodic subsoil is largely attributed to clay swelling rather than dispersion in the subsoil environment. As a result, it is now well recognized that methodology and results of sodicity effects relevant to surface soils are not applicable to subsoils.^[3] A “throttle” with very low hydraulic conductivity was identified in the 25–50 cm of a duplex soil with sodic subsoil using field measurements.^[6] The impeded hydraulic property of the subsoil layer restricts soil water storage from rainfall under dryland conditions. A negative relationship between PWAC and ESP at 0.6 m depth was established for soils in Queensland, Australia^[3]

$$\text{PWAC} = 13.671 e^{-0.0191\text{ESP}}$$
$$r^2 = 0.72 \text{ (for ESP} > 1.5 \text{)}$$

Salinity

In many sodic soils, both ESP and salinity increase with depth. Hence, many sodic subsoils are also saline.^[2,3] A linear relationship was established between saturated electrical conductivity at 0.9 m depth and ESP for soils with annual rainfall between 280 and 2200 mm yr⁻¹ in Australia.^[3] Therefore, soils with high sodicity are likely to be saline in their subsoils, particularly in the lower rainfall areas and for those soils with clay content of about 40–50%, corresponding to the transition from a coarse to a fine matrix.^[3]

Alkalinity

Some sodic subsoil soils have high pH (~10) due to the presence of carbonates and bicarbonates of Na.^[2] The high

pH conditions can result in nutritional troubles in crops^[7] and exacerbate soil structural problems.^[2]

Deficiencies of Micronutrients and Ion Toxicity

Nutritional constraints to plant growth have been identified in some sodic subsoils and include deficiency in micronutrients such as zinc, iron, and manganese, especially in the alkaline sodic soils as well as ion toxicity such as boron and molybdenum.^[7] Availability of micronutrients in alkaline sodic soils is not well understood.

CONSTRAINTS TO CROP PRODUCTION

Large areas in different parts of the world with subsoil sodicity are marginal land with low productivity.^[2,3] However, as many of these soils also have high ESP in the surface layer, it is not clear to what extent the low productivity is directly related to subsoil sodicity. The sodicity vs. yield relationship derived for surface soil^[8] is unlikely to be applicable to subsoils. Furthermore, regression analyses suggest that while large yield responses are obtained by reducing ESP of the surface soil (0–10 cm), they are not significantly affected by variation in the subsoil ESP.^[8] While low productivity of these soils might, to some extent, be related to adverse soil properties such as low PAWC, high bulk density, and low hydraulic conductivity,^[2,3] all of which are caused by the high ESP (sodicity) and the consequent subsoil stability, other properties are often, but not

always, associated with sodic subsoils, namely high salinity, pH, and toxic ions, can also result in poor plant growth and production. For instance, boron toxicity is the major factor-limiting crop yield in some soils with subsoil sodicity.^[7] Subsoil salinity can have opposite effects on crop growth and production. While directly limiting the available water to plant due to osmotic effect of the soil solution on the one hand, it can also help to counteract the adverse effect of sodicity by reducing clay swelling.^[3]

ENVIRONMENTAL IMPACTS

Tunnel erosion leading to gullying in many parts of the world is commonly associated with sodic B horizon.^[2] In Australia, many soils with sodic subsoils are duplex, namely soil profiles with large texture contrast between the A and B horizons. Gullies are formed by dispersion and washing away of the sodic B at points of water entry such as rabbit burrows followed by collapse of the overlying A horizon. The same process can lead to piping and tunneling in earthworks and dam walls.^[9] These gullies, failed earthworks, and dam walls are sources of sediments and water contaminants.

AMELIORATION AND MANAGEMENT

Methods being used to ameliorate subsoil sodicity have been reviewed.^[9,10] They fall into the categories presented in Table 1.

Table 1 The ameliorative methods for overcoming subsoil sodicity.

Methods	Description of technology	Advantages	Limitations
Gypsum application	Surface application of gypsum	Specialized machinery not required, positive responses have been reported for some soil types	Effects on reducing subsoil sodicity occur very slowly
Deep tillage/gypsum	Deep ripping of soil to depth with incorporation of gypsum	Quick responses, using readily available machinery	Uneven mixing of gypsum, large amounts of soil disturbance, variable and even negative responses, improvements often short term due to recompaction, excessive leaching of nutrients
Gypsum slotting	The soil is disturbed to depth only in narrow vertical bands (100–150 mm wide and spaced 1–2 m apart), which are filled with loosened soil mixed with gypsum at rates required to overcome sodicity	Large yield responses and soil improvements have been documented	Costly, special slotting machine needed, significant loss of macroporosity due to lateral swelling has been noted
Cropping strategies	Use of strong-rooted crops in rotation systems such as safflower (<i>Carthamus tinctorius</i>) to loosen subsoil biologically, use of raised beds, more frequency irrigation	Low cost, does not require large disturbance of soil profile, suitable for Vertisols	Subsoil sodicity not removed
Breeding of tolerant species	Breeding of species that are tolerant of sodicity and/or other associated problems, i.e., salinity, boron toxicity	Does not require large disturbance to soil profiles and use of non-renewable resources, large yield responses possible	Long-term research needed, sodicity problems remain

Choice of ameliorative methods is often governed by economics. Expensive methods such as gypsum slotting can only be considered for intensive agricultural enterprises. So far, effects of deep tillage on sodic subsoils have been variable, negative responses have been reported due to rapid recompaction, exposure of subsoil problems such as salt, and toxic minerals. For sodic soils with high ESP throughout the profiles, it might be more cost-effective to first address the problem of surface sodicity. Biological ameliorative methods, e.g., using deep-rooted plants and tolerant species, might be better long-term strategies for improving the productivity and sustainability of these soils.

RESEARCH NEEDS

Critical limits and criteria used for assessing sodicity of surface soils are not applicable to subsoils. More research on quantifying the effect of subsoil sodicity on soil functions and crop productivity is needed. More effective ameliorative and management of these soils require better understanding of the other constraints, e.g., salinity and B toxicity, which are often but not invariably associated with sodicity. These can be achieved by providing more detailed mapping and soil classification with clearer information on the location of subsoil sodicity in the soil profile and better identification of other associated limiting properties.

REFERENCES

1. Northcote, K.H.; Skene, J.K.M. *Australian Soils with Saline and Sodic Properties*; CSIRO: Melbourne, 1972.
2. Szabolcs, I. *Salt-Affected Soils*; CRC: Boca Raton, 1989; 73 pp.
3. Shaw, R.J.; Couglan, K.J.; Bell, L.C. Root zone sodicity. In *Sodic Soils—Distribution, Properties, Management, and Environmental Consequences*; Sumner, M., Naidu, R., Eds.; Oxford University Press: New York, 1998; 95–106.
4. So, H.B.; Aylmore, L.A.G. How do sodic soils behave? The effects of sodicity on soil physical behaviour. *Aust. J. Soil Res.* **1993**, *31*, 761–777.
5. Sumner, M.E. Sodic soils: New perspectives. In *Australian Sodic Soils: Distribution, Properties and Management*; Naidu, R., Sumner, M.E., Rengasamy, P., Eds.; CSIRO: Australia, 1995; 1–33.
6. McIntyre, D.S.; Loveday, J.; Watson, C.L. Field studies of water and salt movement in an irrigated swelling clay soil. II. Profile hydrology during ponding. *Aust. J. Soil Res.* **1982**, *20*, 91–99.
7. Curtin, D.; Naidu, R. Fertility constraints to plant production. In *Sodic Soils—Distribution, Properties, Management, and Environmental Consequences*; Sumner, M., Naidu, R., Eds.; Oxford University Press: New York, 1998; 107–123.
8. McKenzie, D.C.; Bernardi, A.L.; Chan, K.Y.; Nicol, H.I.; Banks, L.W.; Rose, K.L. Sodicity vs yield decline functions for a Vertisol (Grey Vertisol) under border check and raised bed irrigation. *Aust. J. Exp. Agr.* **2002**, *42*, 363–368.
9. Oster, J.D.; Jayawardane, N.S. Agricultural management of sodic soils. In *Sodic Soils—Distribution, Properties, Management, and Environmental Consequences*; Sumner, M., Naidu, R., Eds.; Oxford University Press: New York, 1998; 125–172.
10. Jayawardane, N.S.; Chan, K.Y. The management of soil physical properties limiting crop production in Australian sodic soils—A review. *Aust. J. Soil Res.* **1994**, *32*, 13–44.

Soil Carbon Map: Africa

Arwyn Jones

Roland Hiederer

Luca Montanarella

European Commission, Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy

Abstract

The soils of Africa hold an important soil organic carbon (SOC) pool that is distributed very inhomogeneously over the continent. Values range from very low or negligible levels in the deserts of the Sahara, the Kalahari, and the Horn of Africa to very high in the wetlands of Sudan (Sudd), Democratic Republic of Congo (Ngiri–Tumba–Maingombe), Congo (Sangha–Nouabale–Ndoki), Rwanda, Burundi, Tanzania, and the Barotse Floodplain in Zambia. The soils of the tropical zone contain slightly more carbon (C) than the soils of the savannah regions. From the amended version of the Harmonized World Soil Database, soil C stocks for continental Africa have been calculated as 80.1 Gt C for the layers of 0–30 cm and 74.5 Gt C for the depth of 30–100 cm (this gives a total of 154.6 Gt C for 0–100 cm depth). The soils of the Democratic Republic of Congo and Sudan hold the most SOC with 19.11 and 12.65 Gt, respectively. Various estimates place the total terrestrial C pool of Africa (this includes aboveground vegetation and also all SOC, not just that in the upper 1 m) at between 250 and 280 Gt. In this context, soil holds between 55% and 70% of the total African terrestrial C pool. It should be noted that data for African SOC pools and sinks are quite imprecise and that the uncertainties associated with these values can be very large. Significant additional effort and investment are required to improve these estimates and spatial resolution of the soil biophysical properties in order to better quantify the impact of human activities on the SOC reservoir of Africa.

INTRODUCTION

The Biological Carbon (C) Cycle

One of the most important functions of soil is its role in the terrestrial C cycle by facilitating the conversion of atmospheric carbon dioxide (CO₂) to living matter and then its breakdown and release to the environment. Biology plays an important role in the movement of C between soil and atmosphere. Through photosynthesis, green plants use solar energy and water to turn atmospheric CO₂ into carbohydrates (i.e., sugars, starch, and cellulose), the principle compounds that are necessary to buildup biomass for plants or the bodies of animals. Most C leaves the biosphere through respiration and the decomposition of organic matter (C₆H₁₂O₆, e.g., litter fall and roots). Where oxygen (O₂) is present, aerobic respiration occurs, which frees the energy contained in the sugars and transforms carbohydrate “fuel” back into CO₂, which is released back to the atmosphere as follows:



When O₂ is lacking, e.g., in soil that is saturated with water due to poor drainage, then microbial communities

(e.g., *Methanobacterium*, *Methanosaeta*, and *Methanosarcina*) catalyze the breakdown of organic matter anaerobically. In these conditions, organic matter is broken down to produce methane (CH₄) and CO₂:



The amount of C entering the terrestrial C cycle by photosynthesis and released back into the atmosphere by respiration each year is about 1000 times greater than the amount of C that moves through the geological cycle on an annual basis. CO₂ and CH₄ are important greenhouse gases that affect the earth's climate. This makes soil an important component in the study of climate change.

Soil Organic Matter and C

Organic matter is often described as the “life force” of the soil. It creates pores in the soil that help the roots of plants to develop and store significant amounts of water, thus increasing the soil's capacity to withstand drought conditions. Organic matter can attract the ions of essential nutrients and, as a result, reduce the risk of valuable plant food being leached away. The same process can



Fig. 1 A profile of a Phaeozem from Tanzania—a soil that develops under natural grasslands in subtropical/subtemperate climates. There is enough rainfall to support the vegetation and the organisms that drive the decomposition process. Soils, especially topsoils, that are rich in organic matter appear dark in color and will show a gradual lightening with depth, which is very evident in this figure. Dark-colored soils absorb heat, warm up more rapidly, and cool down less rapidly. Plants such as grasses contribute to soil organic matter through their extensive fibrous root mats and, to a lesser extent, litter fall on the surface. Over time, soil under grasslands, such as Phaeozem or Kastanozems, and Chernozems can accumulate large amounts of soil organic matter. The conversion of natural grasslands to agricultural use causes considerable losses of organic matter.

Source: Photo by Jones, Breuning-Madsen, et al.^[2]

occur with certain pollutants, which limits the potential contamination of soil and groundwater. Finally, the biochemical structure of organic matter can help to buffer excessive acid or alkaline soil conditions (Fig. 1).

The amount of soil organic matter is determined by a balance between biological inputs (e.g., plant residues), the rate of humus formation and its loss, which comprises the decomposition of vegetation, and humus mineralization. Depending on the climate and land use, organic matter can remain stable in soil for long periods. However, the level of organic matter in soil can be manipulated through variations in input or output conditions. Several land management practices, such as the harvesting of crops or drainage, remove C from the soil. Minimal or no-till cultivation, mulching crop residues, crop rotation, and intercropping with perennials all enhance the formation and increased storage of organic matter in the soil. The drainage of peatlands leads to significant losses of organic matter and emissions of CO₂ to the atmosphere and should be avoided.

It is generally accepted that most soil organic matter contains between 40% and 60% C (by dry weight).

Soil Organic Carbon (SOC) and Soil Functions

Low levels of organic C in the soil are generally detrimental to soil fertility and water retention capacity and tend to increase vulnerability to soil compaction, which leads to increases in surface water runoff and erosion. Other effects

of low organic C levels are a reduction in biodiversity and an increased susceptibility to acidification and desertification. Climatic conditions strongly influence both the trends and rates of accumulation and transformation of organic substances in the soil.

Measuring C in Soil

Although there are some promising new technologies, soil C usually cannot be measured directly in the field. Samples must be collected (see below) and sent to a laboratory for analysis. The most accurate method for measuring soil C is through dry combustion in an elemental analyzer. These instruments heat a very small sample of dry pulverized soil to around 1000°C and then measure the CO₂ levels that are released through the combustion process. If the sample contains carbonates or inorganic C, then the soil is pretreated with hydrochloric acid to remove the inorganic compounds. Results are expressed as the percentage of C in the sample.

Traditional procedures for determining SOC are the loss on ignition (LOI) and Walkley–Black tests. However, both are less accurate than the elemental analyzer process described above. LOI measures the loss of weight of organic matter in a dry soil sample after it has been heated in an oven to around 400°C for a few hours. The Walkley–Black method uses potassium dichromate to liberate the C in a sample. As both of these tests only measure organic C, neither can extract all the C that is present in a soil sample.

Problems in Measuring C in Soil

While the C content for a specific location can be calculated accurately using the methodology described above, the result is only applicable to the sampling location. The inherent variability of soil across the landscape means that the soil characteristics may be quite different a few meters away from a sampling site, which means that the C content may also be different. Soil scientists often take several samples within a soil body to determine an average or “representative” C content or density for a particular soil type.

Calculating Soil C Stocks

The amount (or stock—t ha⁻¹) of SOC can be calculated to a given depth using the following equation:^[1]

$$\text{SOC} = C \times \text{BD} \times (1 - \text{VS}/100) \times T \times 10^2 \quad (3)$$

where C is the organic C content for a specific soil type (% weight) for a given depth—determined by laboratory analysis; BD is the soil bulk density (g cm⁻³)—a crucial factor that describes the weight of an undisturbed soil sample. Bulk density can range from 0.1 for light

peats to 1.8 for very dense, compacted mineral soils; VS is the volume of coarse fragments (and/or ice) in the soil (% volume)—a measurement recorded in the field; and T is the thickness of the soil layer (e.g., 0.30 m).

SOC is normally expressed in tonnes of carbon (tC) and gigatonnes (Gt = 1 billion tonnes), but data are also presented in megatonnes [1 million (10^6) tonnes], petagrammes (Pg = 10^{15} g), or teragrammes [Tg = 10^{12} g equivalent to 1 billion (10^9) kilograms].

SOC IN AFRICA: A PERSPECTIVE

One of the most striking features of African soils is the huge range in values and the diversity in the distribution of SOC. The map shown in Fig. 2 was produced for the published Soil Atlas of Africa^[2] and presents the distribution of SOC densities in Africa to a depth of 100 cm from the surface in tonnes C per hectare ($t\ C\ ha^{-1}$) with a resolution of 3 arc seconds. The values shown on the map are based on data from an amended version of the Harmonized World Soil Database (HWSD).^[3]

The colors on the map correspond to the amount of organic C in the soil. Lighter tones indicate the lowest concentrations of organic C, while darker tones indicate higher amounts. SOC varies according to temperature, moisture, local soil conditions, and land use. Most of the C stock of the continent is located in Central Africa. Levels for much of northern and southern Africa are relatively low. Values range from very low or negligible levels in the deserts of the Sahara, the Kalahari, and the Horn of Africa

to very high in the wetlands of Sudan (Sudd), Democratic Republic of Congo (Ngiri–Tumba–Maingombe), Congo (Sangha–Nouabale–Ndoki), Rwanda, Burundi, Tanzania, and the Barotse Floodplain in Zambia.

The following four main zones stand out: the very low arid regions, temperate grasslands, tropical forests and grasslands, and wetlands. The maximum SOC densities were found in Rwanda ($214.6\ t\ C\ ha^{-1}$) and Burundi ($166.7\ t\ C\ ha^{-1}$), while the minimum SOC densities were found in arid Djibouti ($15.1\ t\ C\ ha^{-1}$) and Mauritania ($20.3\ t\ C\ ha^{-1}$). The average SOC content is 35.08 and $31.83\ t\ C\ ha^{-1}$ for the 0–30 cm and 30–100 cm soil layers, respectively.

The soils of the tropical zone contain slightly more C than the soils of the savannah regions. The impact of land use change on C varies depending on the type of change. The conversion of tropical forests into farmland reduces topsoil C content by 20–50% of the original levels through reduced production of detritus, increased erosion rates, and increased decomposition of soil organic matter by oxidation. Deeper layers tend to be less affected. The conversion of forests to pasture appears not to change soil C dramatically and may even increase levels in some cases but may affect other issues (e.g., biodiversity and erosion).^[4–6]

From the amended version of the HWSD, soil C stocks for continental Africa have been calculated as $80.1\ Gt\ C$ for the 0–30 cm layers and $74.5\ Gt\ C$ for the depth 30–100 cm (this gives a total of $154.6\ Gt\ C$ for 0–100 cm depth). The soils of the Democratic Republic of Congo and Sudan hold the most SOC with 19.11 and $12.65\ Gt$, respectively.

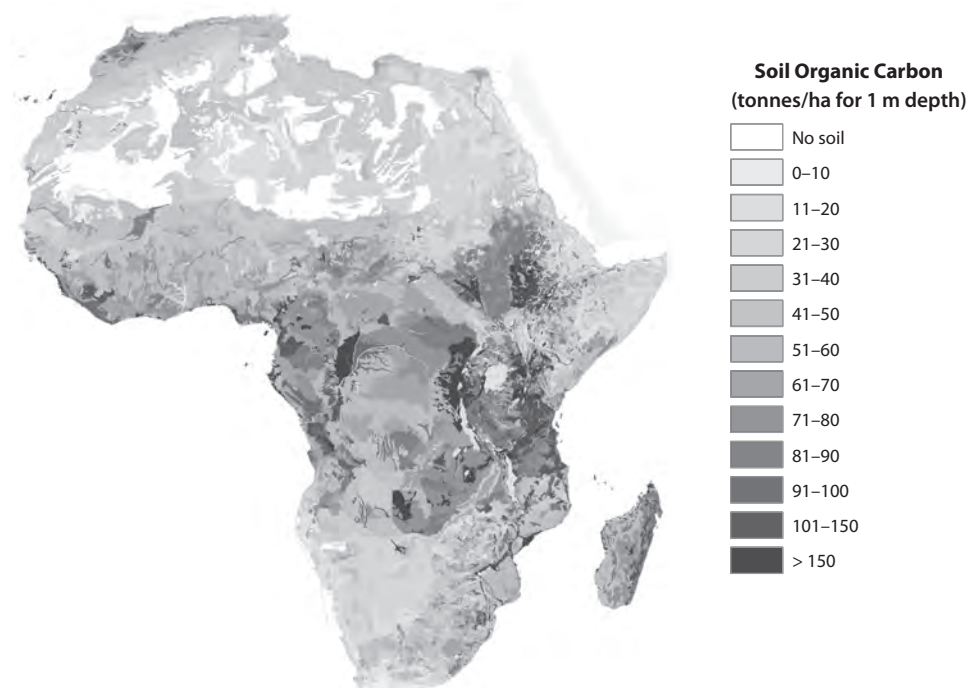


Fig. 2 The map shows organic C in the uppermost 100 cm of soil. Organic C held deeper than 100 cm (e.g., deep peats and estuaries) is not counted in this exercise. Consequently, there is more soil C in reality than shown in this map. Abrupt boundaries in the map may reflect disparate soil data sets.

Source: From Jones, Breuning-Madsen, et al.^[2] and Hiederer & Kochy.^[3]

The reader should note that there are uncertainties in any estimate of soil C stocks at national and continental scales, and the uncertainty of the information presented here is high. More confidence can be attached to information from regions with more detailed soil inventories (the Mediterranean parts of North Africa, South Africa, and parts of Eastern Africa).

In comparison, various estimates^[7] place the total terrestrial organic C pool of Africa (this includes aboveground vegetation and also all SOC, not just that in the upper 1 m) at between 250,000 and 280,000 Gt.^[4] In this context, soil holds between 55% and 70% of the total African terrestrial organic C pool.

SOC and Soil Type

It is clear from the earlier text on soil-forming factors and soil processes that organic C content will vary according to the type of soil. Histosols in Africa (by definition soils rich in organic matter) can contain close to 100 kg C m⁻², while Gypsisols (a desert soil containing high levels of gypsum and found only in very arid climates) typically contain only around 1 kg C m⁻².

In addition, SOC varies with depth and soil type. In Leptosols (shallow soils over hard rock or gravel), virtually all of the SOC are located in the 0–30 cm layer. In contrast, Histosols (organic soils also known as peat) can be several meters thick.

SOC and Ecosystems

The relationship between SOC content and ecosystems shows some interesting features. Using the World Wide Fund for Nature’s Terrestrial Ecoregions of the World database as a base, the soils with the highest mean SOC

content, as expected, are found in the wet mangrove ecosystem with a mean C content of 6.61 kg m⁻². Similarly, there is no surprise that the deserts and dry shrubland ecosystem exhibit the lowest C content with only 1.53 kg m⁻², reflecting the limited biomass production and poorly developed soils of arid regions. Higher net primary production in tropical regions, coupled with deep soils, gives very high C content to the soils of tropical and subtropical moist broadleaf forests (5.7 kg m⁻²). Soils in tropical and subtropical dry broadleaf forests (5.0 kg m⁻²) and flooded grasslands and savannahs (4.17 kg m⁻²) also contain significant levels of SOC, while levels are significantly lower in Mediterranean forests, woodlands and scrub (2.83 kg m⁻²), and montane grasslands and shrublands (3.47 kg m⁻²).^[8]

However, total SOC stocks also depend on the geographical extent of a soil. In this context, Fig. 3 demonstrates that the most important stock of soil C to a depth of 1 m is stored in the tropical and subtropical grasslands, savannahs, and shrubland ecosystems, holding around 80,000 Tg C. Due to their small extent, the temperate coniferous forests and mangrove ecosystems contain relatively small C stocks. Further investigation of the distribution of C within soil shows that in all ecosystems, SOC is concentrated in the topsoil. The slightly higher values in the subsoils of flooded grasslands and tropical moist broadleaved forests may reflect conditions that facilitate the leaching of C deeper into the soil (however, the reader should note that the values for the subsoil relate to more than twice the depth of the topsoil). These figures demonstrate that the topsoil is the most important SOC pool, yet it is also the most susceptible to manipulation by human activities and climatic variations which, together or singularly, can lead to the loss of soil C and degraded soils.

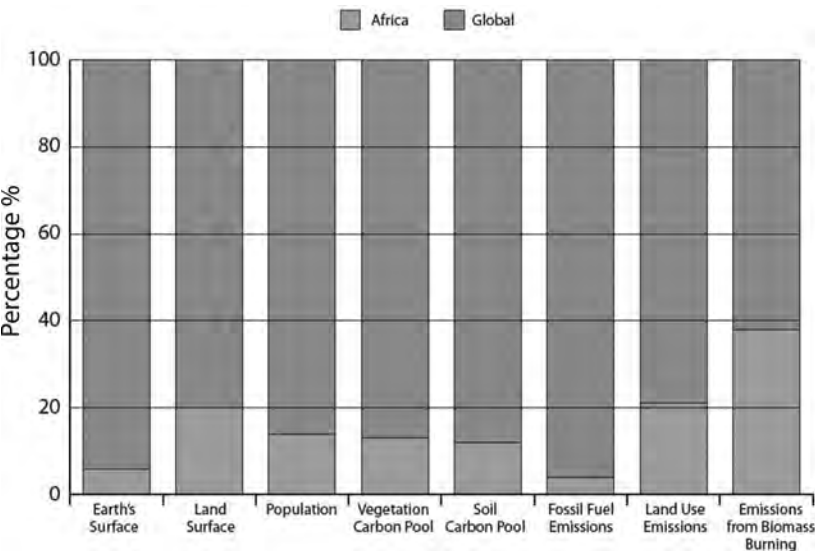


Fig. 3 Topsoil and subsoil organic C stocks in major African ecosystems. Source: From Jones, Breuning-Madsen, et al.^[2]

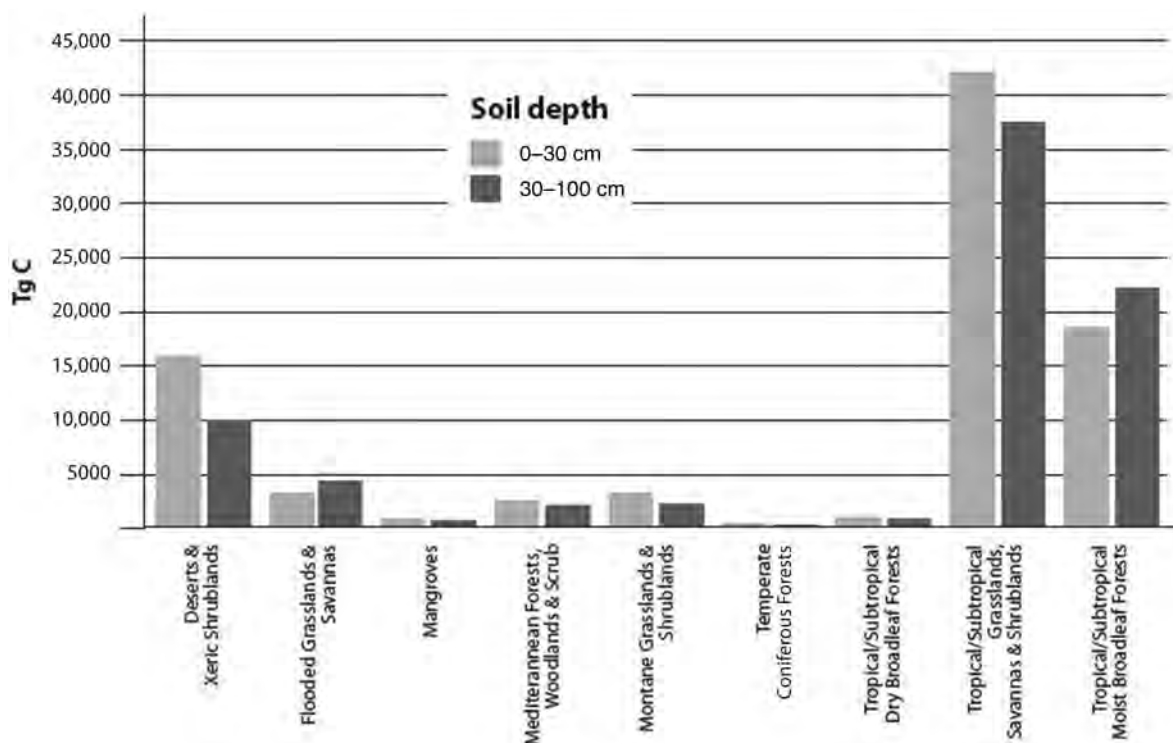


Fig. 4 A global perspective of Africa with regard to C. For comparison, the soils of the northern circumpolar regions (i.e., above a latitude of 50° North) contain about 25% of the global terrestrial vegetation C and approximately 50% of the terrestrial soil C pool. It should be noted that data for African pools and sinks are very imprecise and that the uncertainties associated with these values can be very large.
Source: From Jones, Breuning-Madsen, et al.^[2]

THE SOILS OF AFRICA IN THE GLOBAL C CYCLE

Fig. 4 presents a global view of Africa from a C perspective. While occupying 20% of the earth's land surface and being home to 14% of the world's population, Africa contributes only 4% of the global fossil fuel C emissions that are responsible for rising atmospheric CO₂ concentrations (0.2 Pg C yr⁻¹). This is a disproportionately small fraction but reflects the rural and underdeveloped nature of the African population. However, Africa generates more than one-fifth of the global emissions from land use (i.e., deforestation and land use change) and, staggeringly, nearly 40% of global emissions from burning biomass (i.e., deforestation, shifting cultivation, savannah fires, fuel wood, and agricultural residues). Both the vegetation and the soils of Africa occupy a similar global proportion, about 13% and 10–12%, respectively.^[7,9,10]

In general terms, Africa emits slightly more C on an annual basis than it sequesters. While soil is by far the largest store, holding an estimated 200 Gt of C, soils are also the largest emitter of C in Africa with 11 Gt C primarily due to microbial or heterotrophic respiration and decomposition of organic matter. However, this amount is almost offset by plant productivity (net primary productivity). On the other hand, C emissions due to land clearance and the

burning of biomass are markedly higher than emissions due to the combustion of fossil fuels (e.g., oil and gas). While they affect large areas, emissions from savannah fires are offset by the subsequent regrowth.

Fig. 5 shows the amount of C stored in the soil (to a depth of 1 m) and vegetation of the main ecosystems of the planet.^[11] The figure clearly illustrates the huge capacity of soil to store C compared to vegetation and emphasizes the key role that soil plays in the terrestrial global C cycle. The only ecosystem approaching parity between C stores in vegetation and the soil is the tropical forest. This is due to the lush and continuously growing vegetation, and the rapid recycling of litter and plant remains in the soil. However, the figure also shows clearly that the main reservoirs of global land-based C are the cold permafrost regions and ecosystems such as the boreal forests.

CONCLUSION

Fluxes associated with the SOC reservoir of Africa are significant in the global climate system. In general terms, Africa emits slightly more C on an annual basis than it sequesters. Soil is by far the largest store, holding an estimated 200 Gt of C, with vegetation accounting

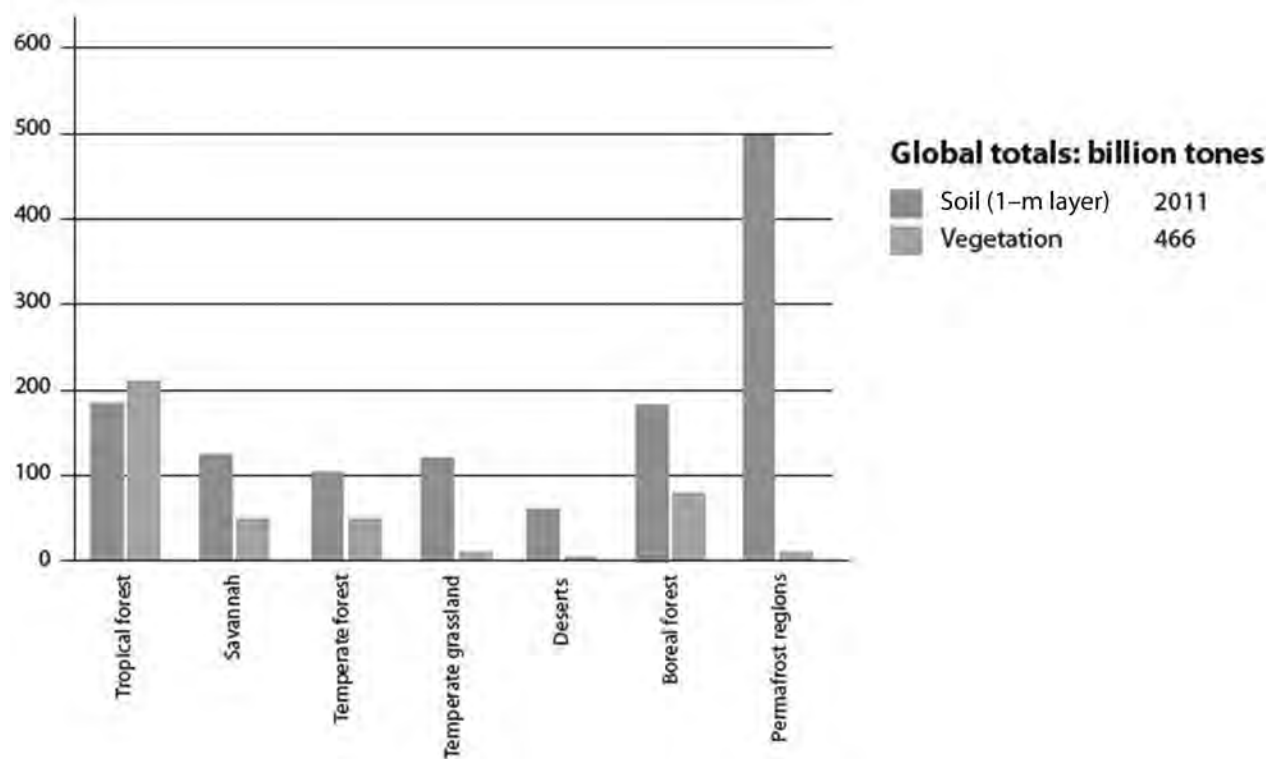


Fig. 5 A comparison of organic C pools in the soil and the vegetation of major ecosystems. Permafrost-affected soils are the principle terrestrial C pool, while in the tropics, there is a balance between the two pools.

Source: EC Joint Research Centre.^[11]

for around 80 Gt C. With 11 Gt C, soils are also the largest emitter of C in Africa primarily due to microbial or heterotrophic respiration and decomposition of organic matter. However, this amount is almost offset by plant productivity (net primary productivity). On the other hand, C emissions due to land clearance and the burning of biomass are markedly higher than emissions due to the combustion of fossil fuels (e.g., oil and gas). While they affect large areas, emissions from savannah fires are offset by the subsequent regrowth.

While these data suggest only a small C imbalance across Africa, climatic fluctuations (especially drought) can lead to sizeable changes in net ecosystem productivity and increase in wildfires. In these situations, the land system of Africa becomes a major source of global C emissions. Considering the existing and increasing pressures on land across Africa (climate variability, land use change, population growth, industrialization, and urbanization), C emissions are likely to undergo substantial changes over the coming years.

The reader should note that there are uncertainties in any estimate of soil C stocks at national and continental scales, and the uncertainty of the information presented here is high. More confidence can be attached to information from regions with more detailed soil inventories (the Mediterranean parts of North Africa, South Africa, and parts of Eastern Africa).

Significant additional effort and investment are required to improve these estimates and spatial resolution of the soil biophysical properties in order to better quantify the impact of human activities on the SOC reservoir of Africa.

ACKNOWLEDGMENTS

This entry builds on the considerable knowledge on soils in Africa that has been amassed through the substantial endeavors of many individuals and organizations from across Africa and Europe. The authors would like to thank everybody who shared their knowledge. In particular, the Joint Research Centre is grateful for the support and commitment offered by the Editorial Board of the Soil Atlas of Africa, the Food and Agriculture Organization of the United Nations, the African Soil Science Society, and the African Union Commission.

REFERENCES

1. Batjes, N.H. The carbon and nitrogen in the soils of the World. *Eur. J. Soil Sci.* **1996**, *47*, 151–163.
2. Jones, A.; Breuning-Madsen, H.; Brossard, M.; Dampha, A.; Deckers, J.; Dewitte, O.; Gallali, T.; Hallett, S.; Jones, R.; Kilasara, M.; Le Roux, P.; Micheli, E.;

- Montanarella, L.; Spaargaren, O.; Thiombiano, L.; Van Ranst, E.; Yemefack, M.; Zougmore, R., Eds. *Soil Atlas of Africa*; European Commission, Publications Office of the European Union: Luxembourg, 2013; 176 pp.
3. Hiederer, R.; Kochy, M. *Global Soil Organic Carbon Estimates and the Harmonized World Soil Database*, EUR 25225 EN; European Union: Luxembourg, 2012.
 4. Sombroek, W.G.; Nachtergaele, F.O.; Hebel, A. Amounts, dynamics and sequestrations of carbon in tropical and sub-tropical soils. *Ambio* **1993**, *22*, 417–426.
 5. Murty, D.; Kirschbaum, M.U.F.; McMurtrie, R.E.; McGilvray, H. Does conversion of forest to agricultural land change soil carbon and nitrogen? A review of the literature. *Glob. Change Biol.* **2002**, *8*, 105–123.
 6. Guo, L.B.; Gifford, R.M. Soil carbon stocks and land use change: A meta-analysis. *Glob. Change Biol.* **2002**, *8*, 345–360.
 7. Williams, C.A.; Hanan, N.P.; Neff, J.C.; Scholes, R.J.; Berry, J.A.; Denning, A.S.; Baker, D.F. Africa and the global carbon cycle. *Carbon Balance Manag.* **2007**, *2*, 3.
 8. Henry, M.; Valentini, R.; Bernoux, M. Soil carbon stocks in ecoregions of Africa. *Biogeosci. Discuss.* **2009**, *6*, 797–823. Available at <http://www.biogeosciences-discuss.net/6/797/2009/> (accessed August 19, 2015).
 9. IPCC. *Fourth Assessment Report (AR4). Working Group II Report "Impacts, Adaptation and Vulnerability" – Chapter 9: Africa IPCC*; Cambridge University Press: Cambridge, 2007; 433–446.
 10. Ciais, P.; Bombelli, A.; Williams, M.; Piao, S.L.; Chave, J.; Ryan, C.M.; Henry, M.; Brender, P.; Valentini, R. The carbon balance of Africa: Synthesis of recent research studies. *Phil. Trans. R. Soc.* **2011**, *369*, 1–20.
 11. EC Joint Research Centre. *Soil Atlas of Northern Circumpolar Region*, EUR 23499 EN; European Union: Luxembourg, 2010.

Soil Carbon Sequestration: Ethiopia

Ambachew Demessie

College of Agriculture, Hawassa University, Awassa, Ethiopia

Bal Ram Singh

Department of Environmental Sciences, Norwegian University of Life Sciences, Aas, Norway

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Sequestration of soil organic carbon (SOC) is vital to increase the SOC stock and quality, decrease soil erosion, and mitigate climate change. In this entry, we reviewed the level of soil degradation and the loss of SOC in Ethiopia. Results of studies on land use types are summarized to sort out those sound types that can potentially reverse the SOC loss in various ecosystems of the country. The SOC loss through soil erosion, deforestation, and land use change is estimated to be from 15 to 1000 kg ha⁻¹ yr⁻¹ in Ethiopia. Much of this loss in SOC can be attributed to reduced inputs of organic matter, increased decomposability of crop residues, and tillage effects that decrease the amount of physical protection to decomposition. The problem of SOC depletion in Ethiopia can be tackled through better management aimed at augmenting the normal function of the soils and SOC sequestration. Land use types such as native forest, grass land, plantation forest, agroforestry, and woodlands better protect the soil from severe erosion and augment the accrual of SOC. Restoration of severely degraded lands through area closure is good option to enhance SOC sequestration if it is practiced in areas where the soil depth class falls below 35 cm. We suggest that appropriate policy should be launched to safeguard very sloppy lands from cultivation. Along with biological and physical conservation measures, cow dung and crop residues should be returned to the cultivated crop fields to enhance the sequestration of SOC and sustain agricultural productivity.

INTRODUCTION

Ethiopia, with an estimated population of 100 million, is located in the Horn of Africa (Fig. 1). It is bordered by Eritrea to the north, Djibouti and Somalia to the east, Sudan and South Sudan to the west, and Kenya to the south. The nation's topographic features are dominated by rugged mountains and lowland plains. Based on major landforms and altitudinal variations, the soils of Ethiopia can be grouped as highland and lowlands soils.

Montane forests are the main constituents of the natural vegetation, of which dry afromontane forms the largest part. The highlands of Ethiopia, in contrast to most mountain systems outside Africa, are very suitable for human inhabitation. About 14% of the population lives in areas above 2400 m (cool climatic zone), and about 75% lives between 1500 and 2400 m a.s.l. (temperate zone). The remaining 11% of the population resides in the lowlands which constitute 60% of the 1,270,000 km² total area of the country. Agriculture is the dominant sector of the nation's economy where the livelihood of the 85% of the population is reliant on. In general, the highland areas support 60% of livestock and 90% of the agriculturally suitable area of the

country. Favorable climate and ecological conditions, sufficient rainfall, moderate temperatures, and well-developed soils were the basis for the early development of agricultural systems in the highland areas. Keeping up with the steady population growth, agriculture generally expanded from gently sloping land onto the steeper slopes of the mountains. The clearing of forests, traditional tillage practices, overgrazing, and the subsequent erosion process over long periods of time gradually destroyed the soil resource.

Soil erosion has reduced the soil organic matter (SOM) layers in the highland areas across the country. Originally, the highland soils of Ethiopia were more than 50 cm deep and probably had rooting depth of more than 100 cm that leads to the conclusion that the range of depth classes shown in Table 1 represent the legacy of soil degradation through erosion process. Deforestation, soil erosion, traditional tillage practices, and overgrazing are ongoing and are reducing the soil quality (health) through the loss of SOM. Soil quality (health) is defined as *the capacity of a soil to function within ecosystem and land use boundaries, to sustain biological productivity and maintain environmental quality*.



Fig. 1 Location of Ethiopia on earth.

SOM influences several critical soil functions and is affected by land use and land management practices. Decreases in SOM contents, through cultivation or tillage intensification, are often related to the deterioration of soil structure. Effects include the loss of aggregate stability, increased crust formation, increased runoff and soil erosion, slower water infiltration, and a slower exchange of water/gasses. Different organic matter pools such as particulate organic carbon (POC; organic fraction >53 μm diameter, recognizable structure) and humic fraction [non-identifiable chemical structure such as humic acids (HAs) and humin (HU)] affect different soil functions.

POC is important in providing energy for biological processes. The humic fraction is an important source of essential soil nutrients and is the principal pool in contributing to the soil's cation exchange capacity.

The loss of SOM and the little or no input to compensate the loss deplete the soil carbon (C) stock [mass of organic C stored in the soil C pool (or reservoir)], which is a phenomenon in the highland and lowland areas of Ethiopia. C exists in the environment in many forms, predominantly as a plant biomass, SOM, geologic deposits, and atmospheric carbon dioxide (CO_2) gas, and dissolved in seawater. On the other hand, sequestration of C (storage of C in a stable solid form) in soil and vegetation is a necessity for improving soil quality and mitigation of climate change. Soil is the largest overall reservoir of C and a major source and sink for C exchanges between the atmosphere, terrestrial vegetation, and aquatic environments.

Depending on land use type and topographic features, soil erosion and cultivation of fragile lands such as sloppy terrain cause decline in soil quality and depletion of soil organic carbon (SOC). Much of this loss in SOC can be attributed to reduced inputs of organic matter, increased decomposability of crop residues, and tillage effects that decrease the amount of physical protection to decomposition. The changing climate on soils also exacerbates the problem of increasing SOC loss through its effect on soil moisture and temperature regimes. In this entry, we discuss the overriding factors that control the SOC loss such as deforestation, soil erosion, and cultivation of fragile lands and summarize some study findings on land use types that can potentially recover the soil C lost as a result of mismatch of management and land use type in Ethiopia.

Table 1 Areas with different soil depths in the highland areas of Ethiopia by administrative region in percentage of the respective highland parts of each region.

Region	Total area in km ² (=100%)	Area >1500 m a.s.l. in percentage of the region	Area of soil depth class in percentage of highland area above 1500 m a.s.l. (high land = 100%)				
			130 cm	75 cm	35 cm	10 cm	Lake area (percentage of highland area)
Tigray	65,900	53.9	11.7	30.1	46.0	12.2	0.0
Welo	80,800	40.4	6.2	4.2	17.6	72.0	0.0
Gonder	73,800	67.8	12.5	38.5	22.1	22.3	4.6
Gojam	61,900	84.5	63.7	20.0	13.5	1.3	1.5
Shewa	80,900	86.6	49.3	18.2	8.7	22.2	1.6
Welega	69,900	58.8	76.7	3.4	17.8	0.7	1.4
Illibabur	43,900	39.3	67.0	14.4	12.2	6.4	0.0
Kafa	52,400	80.8	69.9	22.7	1.0	6.4	0.0
Gomugofa	40,200	76.0	39.5	36.6	9.9	11.1	2.9
Sidamo	1,18,100	38.5	60.5	15.1	17.9	6.2	0.3
Arsi	24,100	70.4	68.3	18.3	1.8	11.6	0.0
Bale	1,23,900	28.5	37.5	28.4	18.9	15.2	0.0
Hararghe	257,400	13.6	20.6	18.4	24.2	36.8	0.0

Source: From Humi.^[1] ©1988 International Mountain Society and the United Nations University.

SOIL DEGRADATION IN ETHIOPIA

The highlands of Ethiopia constitute more than 44% of the total area of the country.^[2] Vegetation and soil degradation are serious problems in both the highlands and lowlands of Ethiopia. Nationally, the conservative deforestation rate of natural forests is estimated to be 0.16–0.20 million ha yr⁻¹, and the natural forest cover is believed to have decreased from 16% in the 1950s to about 2.8% in 1980s.^[3] From the information available, the rate of deforestation in Ethiopia is alarming, and the rate of afforestation is very negligible. A satellite image from 1986 to 2000 shows that Ethiopia's forest cover has been reduced to 2.36% or 27,059 km² from 1990 to 2000.^[4] The growing demand of the ever-increasing population for grazing and arable land, fuel wood, and construction material is the major factor contributing to deforestation in Ethiopia.^[5,6] Fuel wood has become scarce to a large number of the Ethiopian households. Thus, rural people use dung and agricultural residues as alternative fuels instead of putting them back into the soil to increase crop yields. The demand for forest product is growing, and as a result, the remnant forests are under heavy pressure of massive deforestation in the country (Fig. 2).

Deforestation and the subsequent cultivation of fragile lands accelerate the rate of soil degradation through erosion process. Processes of soil degradation include erosion by water and wind, loss of soil physical and chemical properties necessary to maintain soil quality (health), loss of nutrients, and SOC depletion.^[7]

The poor land cover and traditional farming practice have aggravated soil erosion, resulting in the reduction of the soil depth layers across the country. Hurni^[1] has pointed

out that the rooting depth below 50 cm (Table 1) has significant negative effects on soil productivity due to loss of SOM. The loss of SOM reduces the water-holding capacity of soil during periods of intermittent rainfall that makes it difficult to be available at times of draught. Regions that have areas with depths less than 10 cm make them unsuitable even for moderate grassland use or afforestation.^[1,2] Estimates of soil loss from Ethiopian highlands and lowlands vary widely. The rate of soil loss by erosion documented for different land use types in Ethiopia ranges from 16 tons to 300 tons per hectare per year.^[1] Based on this range, the amount of annual soil loss by erosion is estimated to be from 1248 million tons to 23400 million tons per year from the 78 million hectares of cultivated, grazing, and range lands.

The Ethiopian highlands reclamation study^[2] suggested the annual average rate of 100 tons per hectare per year soil loss for different land use types in the country. Based on this rate, the total soil loss is 7800 million tons per year from cultivated, grazing, and range lands of Ethiopia.^[8] The soil degradation through water and wind erosion in lowland soils is caused by the poor farming practices particularly in the areas practicing cereal monoculture farming system, uncontrolled burning of range lands, conversion of woodland and native forests into crop lands, and burning dung and crop residues for household energy needs.^[2] Land degradation is seriously threatening the economic and social development of the country as a whole. Due to degradation, an increasing number of Ethiopians befall to be vulnerable to the effects of draught. The severity of the devastating draught and the resulting famines in the past, present, and beyond can be attributed to an accelerating process of soil degradation.

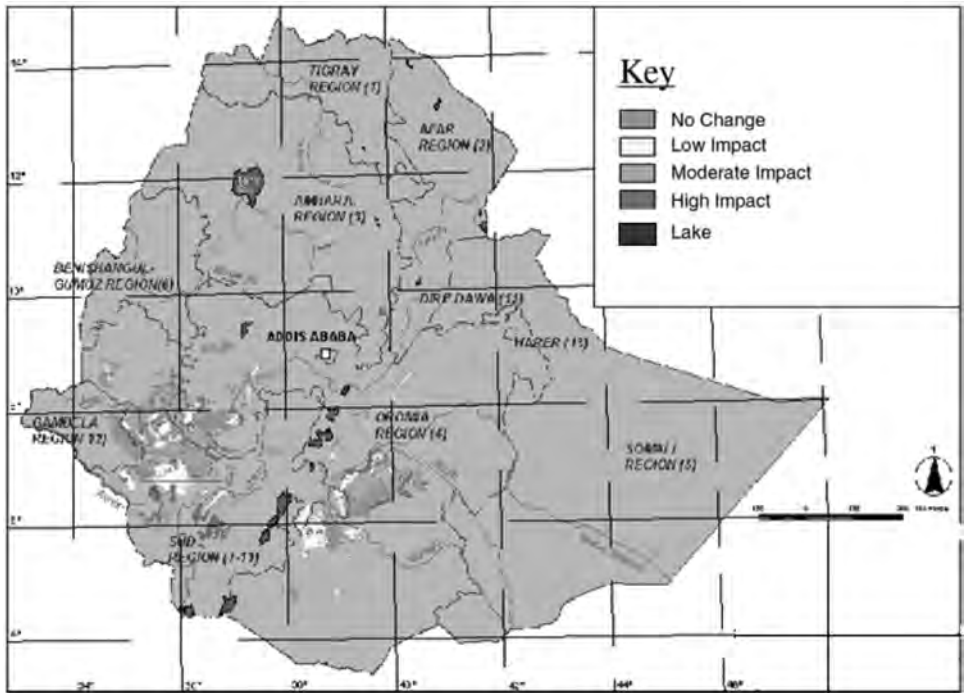


Fig. 2 The relic forests of Ethiopia between 1973 and 1990. Source: From Reusing.^[4] ©2000.

Table 2 Changes of SOC and some properties of soils in different locations of Ethiopia.

Site	Soil order	Vegetation	Texture	pH (H ₂ O)		OC (g kg ⁻¹)	
				Forested	Cultivated	Forested	Cultivated
Awassa	Andisol	High land savanna	LS	5.28	5.45	40.3	16.6
Ghinch	Vertisol	Savanna	C	5.74	5.98	32.8	20.0
Holeta	Vertisol	Savanna	C	5.58	4.88	36.1	16.0
Jimma	Alfisol	Rain forest	C	6.15	4.94	47.3	20.2
Sirinka	Vertisol	Guinea savanna	C	6.02	6.75	27.2	7.7

Note: LS: loam soil; C: clay; OC: Organic Carbon.

Source: Adapted from Spaccini.^[11] ©2005 Elsevier.

LOSS OF SOM IN ETHIOPIAN SOILS

It is well documented that SOM is accumulated more in the upper most soil layer and decreased downward to 1 m depth.^[9,10] The reduction of soil depth from 100 cm to 10 cm (Table 1) implies that the available SOM is washed away downslope. Comparatively within the Ethiopian highlands, the effect of soil erosion is more severe in Welo, Hararghe, and Shewa (Table 1). The legacy of soil erosion is more pronounced in Welo region where 72% of the total highland area has a soil depth class of 10 cm (Table 1). The SOM loss associated with the removal of surface soil ranges from 15 to 1000 kg ha⁻¹ yr⁻¹ which amounts to 1.17 to 78 million tons of organic matter lost per year from 78 million hectares of cultivated, grazing, and range lands. The loss of soil nitrogen (N) ranged from 0.39 to 5.07 million tons per year and phosphorus from 1.17 to 11.7 million tons per year.^[8] The Ethiopian highland reclamation study^[2] put the degraded area on the highlands at 27 million hectares, of which 14 million hectares is very seriously eroded with 2 million hectares of this having reached a point of no return.^[8]

Conversion of native forest and woodlands into cultivated lands increases the loss of SOM (Table 2). Spaccini et al.^[11] assessed the influence of deforestation and the subsequent cultivation on total SOC and humus fraction (the stable fraction of SOM) under different soil and vegetation types from five locations in Ethiopia (Table 2). Spaccini et al. found that organic C concentration was

invariably higher under forest setting compared with that under cultivated fields. The finding also shows that HA, fulvic acid, HU, and humic hydrophobic fraction extracted with an acetone–hydrogen chloride solution were decreased in cultivated lands in all locations, but the magnitude of decrease was site-specific (Table 3). However, data in Tables 2 and 3 shows that there is also variation within the forest and the cultivated fields of the different sites that could be attributed to difference in soil type, rainfall amount, and distribution. The data reported by Spaccini et al.^[11] are in line with the finding reported by Demessie.^[12] Demessie has evaluated the soil C and N stock under natural forest, plantations, agroforestry, and cultivated land use types located under similar site characteristics. The sites were adjacent to each other having similar Andic Paleustalf soil and located within an altitudinal range of 2137 to 2215 m a.s.l. All sites were receiving the same average annual rainfall of 917 mm. Demessie found that the SOC was higher under natural and plantation forests compared with those under traditional agroforestry and monocropped cultivated fields (Fig. 3). The variation seen could be attributed to difference in land use type, implying that conversion of native vegetation to cultivated lands will reduce SOM and other physical and chemical properties of soils regardless of differences in magnitude due to site properties. If properly managed, woodlands in lowland areas sequester more SOC. For example, Lemenih and Itanna^[13] showed that acacia woodland soil sequestered 40.3 Mg C ha⁻¹ and 5.3 Mg N ha⁻¹ under the semiarid

Table 3 Changes in humic fractions of stable SOC in different locations of Ethiopia.

Site	HA	HA	FA	FA	HU	HU	HE	HE
	Forested	Cultivated	Forested	Cultivated	Forested	Cultivated	Forested	Cultivated
Awassa	15.7	11.5	6.0	6.4	2.9	0.7	2.7	1.7
Ghinch	13.3	9.1	2.5	1.1	0.7	0.7	3.1	0.7
Holeta	7.7	3.3	4.0	2.2	1.3	1.3	4.4	2.3
Jimma	12.5	3.8	6.8	0.8	4.2	3.2)	3.6	0.4
Sirinka	10.5	2.3	1.3	1.2	0.3	0.3)	2.4	1.4

Note: HA: humic acid; FA: fulvic acids; HU: humin fraction; HE: humic hydrophobic fraction.

Source: From Spaccini.^[11] ©2005 Elsevier.

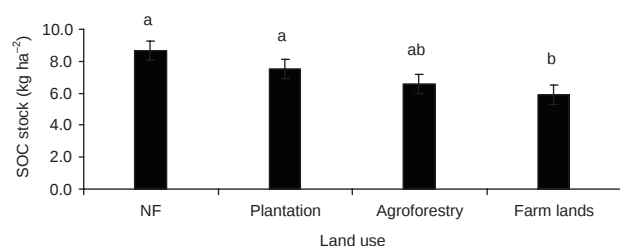


Fig. 3 Total organic carbon stock of 0–20 cm depth under NF (natural forest), plantations agroforestry, and farmland uses ($P < 0.05$).

Source: Adapted from Demessie.^[12] ©2009 Norwegian University of Life Science.

environment in the rift valley of Ethiopia. Similarly, Michelsen et al.^[14] evaluated C stocks, microbial respiration, and biomass in fire-prone tropical grassland, woodland, and forest ecosystems in the lowlands of Western Ethiopia. The forest and woodland ecosystems sequestered more SOC than the other land use types. From the foregoing discussion of research findings, it can be concluded that the storage of SOC in Ethiopia may be enhanced by reducing C losses through restoration of vegetation on degraded lands.

POTENTIAL OF SOIL C SEQUESTRATION THROUGH REHABILITATION OF DEGRADED LANDS IN ETHIOPIAN SOILS

C sequestration refers to long-term storage of CO₂ to mitigate or deter global warming and avoid the harmful effect of climate change. The principal global C pools are in the order of oceanic, geologic, pedologic (soil), biotic, and atmospheric pool. The SOC concentration ranges from a low in soils of the arid regions to high in soils of the mild climatic regions (the highlands for Ethiopia's case) and extremely high in organic or peat soils.^[15] Soil C plays an important role in the maintenance of agricultural productivity under changing climatic conditions and is an element of the mitigation and adaptation response in the land use sector.^[16] In particular, the maintenance of existing C stocks as well as soil C sequestration can contribute to improving and/or sustaining agricultural productivity, reducing greenhouse gas emissions, and increasing the resistance and resilience of agricultural ecosystems against climate change impacts, such as rising temperatures, increased frequency of flooding, and other extreme weather events. All fertile soils have an adequate supply of organic matter that includes plant and animal residues at various stages of decomposition, cells (living and dead) and tissues of microbes, and substances synthesized by soil population.^[17]

Sequestration of SOC is therefore fundamentally important to decrease soil degradation, increase the SOC stock, increase productivity, and mitigate climate change.^[15,18,19] It is well documented that land use types such as native

forest, grass land, plantation forest, agroforestry, and woodlands better augment the accrual of SOC through litter input, lower temperature that reduces high rate of decomposition, and protection of soil from severe erosion.^[9,10,20] Degraded lands can be restored through sound management and judicious land use practices.

Land use, soil type, climate, and vegetation are the drivers of SOM dynamics.^[21] Wherever one of the land use changes decreased soil C, the reverse process increased soil C and vice versa.^[9] For example, a study result reported by Demessie^[12] showed that plantations established on cultivated lands accrued SOC in 25–30 years times the amount very close to the natural forest (Fig. 3). A change in land use from agriculture to forestry is replacing the annual cycle of cultivating and harvesting crops by the much longer forest cycle.^[22] This enables the production of a larger biomass and reduces the degree of soil disturbance. On the other hand, the conversion of natural vegetation to cultivated land results in very rapid declines in SOM, reduced inputs of organic matter, increased decomposability of crop residues, and tillage effects that decrease the amount of physical protection to decomposition.^[22,23]

Consequently, restoration of degraded soils in Ethiopia is a vital strategy to reduce soil erosion and environmental degradation and to improve soil productivity. Such management accentuates the potential of Ethiopian soils to provide terrestrial sinks of C and reduce the rate of enrichment of atmospheric CO₂. SOC contents decrease by 0–63% following deforestation.^[7,10,11] There exists a high potential for increasing SOC through the establishment of natural or improved fallow systems (agroforestry) with attainable rates of C sequestration in the range of 0.1 to 5.3 Mg C ha⁻¹ yr⁻¹.^[7] The accumulation and turnover of SOM are major factors in soil fertility and ecosystem functioning and determine whether soils are sinks or sources of C in the global cycle.^[23]

The turnover of C in soils is controlled mainly by water regimes and temperature but is modified by factors such as size and physicochemical properties of C additions and distribution of litter within the soil matrix and its interaction with clay surfaces.^[24] Substantial fraction (often 30–50% or more) of the energy and C annually fixed in forests is contributed to the forest floor as litter fall (mostly) leaves.^[25]

The sequestration of SOC can be greatly enhanced when degraded soils and ecosystems are restored; marginal agricultural soils are converted to a restorative land use or replanted to perennial vegetation.^[15] When food production is a priority, agroforestry systems can be used as an alternative for economically friendly land use approach in cultivated lands. For example, a study result by Demessie et al.^[10] on parkland type of agroforestry system in Gambo district of Southern Ethiopia suggest that the SOC and N stocks tended to be less in farmlands than in the parkland agroforestry system, although the difference is not significant (Fig. 4). In this particular study site, the soil in

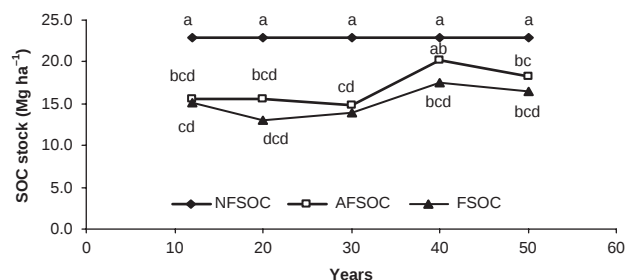


Fig. 4 Total SOC stock of 1 m depth in the chronosequences of agroforestry (AF) and farm (F) land uses. Means followed by the same lower case letter(s) are not significantly different at ($P < 0.05$).

Source: From Demessie, Singh, et al.^[10] 2013 Springer.

parkland agroforestry systems is equally subjected to soil disturbance through tillage practices exposing the SOM for decomposition. However, even the slightest increase of SOM through the addition of litter to the soil will have positive effect on the soil quality compared with the treeless cultivated land where little or no crop residue is returned to the soil. When the soil disturbance through tillage practice is minimum, the C sequestration potential of agroforestry systems is estimated to be between 12 and 228 Mg C ha⁻¹.^[26,27]

Cow dung and crop residues should be returned to the cultivated crop lands along with biological (wind break and shelter belts) and physical conservation measures. Area closure (an area delineated to exclude human interference) should be adapted to areas where the soil depth class falls below 35 cm and appropriate policy should be launched to safeguard very sloppy lands from cultivation. For example, Descheemaeker et al.^[28] have shown that the establishment of area closures has become an important measure to increase SOM and N compared with the degraded grazing land in the Tigray highlands of northern Ethiopia (Fig. 5). This suggests that such sound and environmentally friendly land management practices will possibly enhance the potential sequestration of C in Ethiopian soils. More research has to be done on management practices to explore those suitable to SOC sequestration and maintenance of soil quality for different soils and ecosystems in the country.

REFERENCES

- Hurni, H. Degradation and conservation of the resources in the Ethiopian highlands. *Mt. Res. Dev.* **1988**, *8*, 123–130.
- FAO. *Ethiopian Highlands Reclamation Study (EHRS)*. Final report. Food and Agriculture Organization of the United States (FAO): Rome, 1984; Vol. 1–2.
- EFAP. *Ethiopian Forestry Action Plan: Vol. 1 and 2*; Ministry of National Resources Development and Environmental Protection: Addis Ababa, 1994.
- Reusing, M. Change detection of natural high forests in Ethiopia using remote sensing and GIS techniques

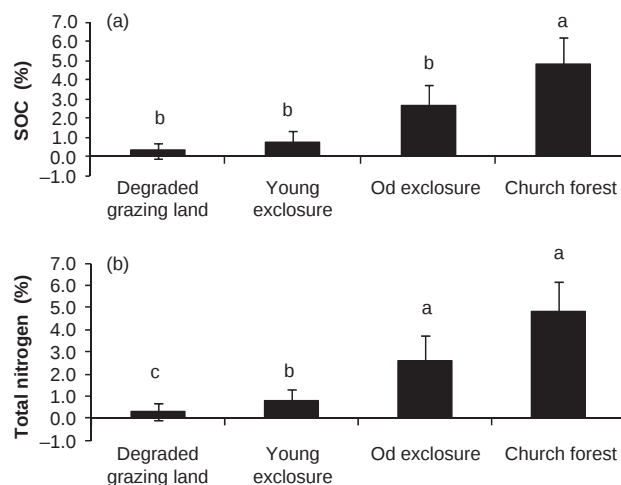


Fig. 5 Soil organic carbon (a) and total nitrogen (b) in different land uses of May Ba'ati in Dogu'a Tembien district of Tigray highlands in northern Ethiopia.

Source: Adapted from Descheemaeker, Nyssen, et al.^[28]

International Archives of Photogrammetry and Remote Sensing. German Technical Cooperation (GTZ): Ethiopia, Vol. XXXIII, Part B7, 2000.

- Mekonnen, A. Rural household biomass fuel production and consumption in Ethiopia: A case study. *J. Forest Econ.* **1999**, *5*, 69–97.
- Bishaw, B. Agroforestry and community forestry for rehabilitation of degraded watersheds on the Ethiopia Highlands. In *International Conference on Development Studies in Ethiopia*, Addis Ababa, Ethiopia, Jul 11–12, 2003.
- Vågen, T.G.; Lal, R.; Singh, B.R. Soil carbon sequestration in sub-Saharan Africa: A review. *Land Degrad. Dev.* **2005**, *16*, 53–71.
- Hawando, T. *Desertification in Ethiopian Highlands*. RALA Report No 200; Norwegian Church AID: Addis Ababa, 1997.
- Guo, L.B.; Gifford, R.M. Soil carbon stocks and land use change: A meta analysis. *Glob. Change Biol.* **2002**, *8*, 345–360.
- Demessie, A.; Singh, B.R.; Lal, R. Soil carbon and nitrogen stocks under chronosequence of farm and traditional agroforestry land uses in Gambo district, Southern Ethiopia. *Nutr. Cycl. Agroecosys.* **2013**, *3* (95), 365–375.
- Spaccini, R.; Mbagwu, C.A.; Conte, P.; Piccolo, A. Changes of humic substances characteristics from forested to cultivated soils in Ethiopia. *Geoderma* **2005**, *132*, 9–19.
- Demessie, A. *Effects of Conservation of Natural Forests to Plantations, Traditional Agroforestry and Cultivated Lands on Carbon Sequestration and Maintenance of Soil Quality in Gambo District, Southern Ethiopia*. PhD thesis, Norwegian University of Life Science: Norway, 2009.
- Lemenih, M.; Itanna, F. Soil carbon stocks and turnovers in various vegetation types and arable lands along an elevation gradient in southern Ethiopia. *Geoderma* **2004**, *123* (1–2), 177–188.
- Michelsen, A.; Andersson, M.; Jensen, M.; Kjoller, A.; Gashew, M. Carbon stocks, soil respiration and microbial

- biomass in fire-prone tropical grassland, woodland and forest ecosystems. *Soil Biol. Biochem.* **2004**, *36* (11), 1707–1717.
15. Lal, R. Soil carbon sequestration to mitigate climate change. *Geoderma* **2004**, *123*, 1–22.
 16. Smith, P. Soils and climate change. *Curr. Opin. Env. Sustain.* **2012**, *4*, 1–6.
 17. Canada Department of Agriculture Glossary of terms in soil science, Canada, Publ. No. 1459, 1972; 66 pp.
 18. Oades, J.M. The retention of organic matter in soils. *Biogeochemistry* **1988**, *5*, 35–70.
 19. Batjes, N.H. *Management Options for Reducing CO₂-concentrations in the Atmosphere by Increasing Carbon Sequestration in the Soil*. Report 410-200-031, Dutch National research programme on global air pollution and climate change and Technical paper 30; International Soil Reference Center: Wageningen, 1999.
 20. Demessie, A.; Singh, B.R.; Lal, R. Soil carbon and nitrogen stocks under plantations in Gambo district, southern Ethiopia. *J. Sustain. Forest.* **2011**, *30* (6), 496–517.
 21. Feller, C.; Beare, M.H. Physical control of soil organic matter dynamics in the tropics. *Geoderma* **1997**, *79*, 69–116.
 22. Vesterdal, L.; Ritter, E.; Gundersen, P. Change in soil organic carbon following afforestation of former arable land. *Forest Ecol. Manag.* **2002**, *169*, 137–147.
 23. Post, W.M.; Kwon, K.C. Soil carbon sequestration and land-use change: Processes and potential. *Glob. Change Biol.* **2000**, *6*, 317–328.
 24. Schjønning, P.; Thomsen, I.K.; Mberg, J.P.; Jonge, H.D.; Kristensen, K.; Christensen, B.T. Turnover of organic matter in differently textured soils I. Physical characteristics of structurally disturbed and intact soils. *Geoderma* **1999**, *89*, 177–198.
 25. Ovington, J.D.; Heitkamp, D. The accumulation of energy in forest plantations in Britain. *J. Ecol.* **1960**, *48*, 639–646.
 26. Dixon, R.K. Agroforestry systems: Sources of sinks of greenhouse gases? *Agroforest. Syst.* **1995**, *31*, 99–116.
 27. Albrecht, A.; Kandji, S.T. Carbon sequestration in tropical agroforestry systems. *Agr. Ecosyst. Environ.* **2003**, *99*, 15–27.
 28. Descheemaeker, K.; Nyssen, J.; Poesen, J.; Haile, M.; Muys, B.; Raes, D.; Moeyersons, J.; Deckers, J. Soil and water conservation through forest restoration in enclosures of the Tigray highlands. *J. Drylands* **2006**, *1* (2), 118–133.

Soil Carbon: Compositional and Isotopic Analysis

James Moran

Pacific Northwest National Laboratory, Richland, Washington, U.S.A.

M. Lizabeth Alexander

Alexander Laskin

Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.

Abstract

Inherent soil heterogeneity makes tracking carbon (C) through these dynamic environments very difficult. Here we discuss three emerging techniques for spatially resolved analysis of organic components in soil. Laser ablation isotope ratio mass spectrometry (LA-IRMS) is a method for performing stable isotope analysis on solid samples. The laser-based sampling associated with LA-IRMS allows targeting of specific regions along a solid surface, with resulting isotope analysis at a spatial resolution of up to 50 μm . Laser ablation aerosol mass spectrometry (LA-AMS) uses a similar, laser ablation-based, sampling approach with up to 2 μm spatial resolution. In contrast to LA-IRMS, which provides isotope measurement but no chemical identification, the high resolution mass spectrometry capabilities of the AMS enable basic identification and classification of organics based on their elemental ratios (C/N, O/C, H/C, etc.), degree of oxidation, or inclusion of functional groups (e.g., amines, hydrated species, etc.). Nanospray desorption electrospray ionization mass spectrometry (nano-DESI) offers additional detail to chemical identification above that provided by AMS. This technique applies a solvent bridge to dissolve organics off of a solid's surface. A high voltage applied between the solvent droplet and detector inlet then aspirates the analytes into the mass spectrometer for chemical identification. Translating the solvent bridge over a sample's surface enables spatial ($\sim 100 \mu\text{m}$) resolution in organic compound analysis. Taken together, the spatial resolution of these emerging approaches, combined with their associated isotopic measurement and chemical identification, provides a suite of tools to enhance the understanding of C accumulation, localization, and transformation in soils.

INTRODUCTION

Soils hold an extremely large reservoir of organic carbon (C), with the overall net balance being both influenced by and potentially directing global-scale climate modification. Despite its central role in C-cycling, a complete set of key principles controlling the fate of soil organic C remains elusive in large part due to the inherent geochemical, organic matter, and microbial heterogeneity of soil systems. Here we present an overview of three spatially resolved techniques that can be used to monitor soil organic C found in such heterogeneous environments. The first technique, laser ablation isotope ratio mass spectrometry (LA-IRMS), can be used to spatially track stable isotope substrates/labels as they migrate into soil systems. Laser ablation aerosol mass spectrometry (LA-AMS) enables the identification of dominant organic compounds at high spatial resolutions. Nanospray desorption electrospray ionization (nano-DESI) permits high mass resolution chemical structure identification of solvent-soluble organics, also at fine-scale spatial resolution. Combined, these emerging tools provide a spatial framework for constraining both the

fundamental organic components of a soil and for testing the reactivity and accumulation of added (and potentially isotopically labeled) C sources into a soil.

LA-IRMS

Stable isotope analysis has a strong history of tracking organic substrate turnover in soil, delineating rhizosphere activity zones, and identifying natural transitions in C inputs to a soil. When coupled with stable isotope probing, isotope analysis can be used to track an added substrate into a soil and help determine its fate—is it simply remineralized, spread throughout the soil system, or focused within specific organisms/spatial locations? Traditional bulk stable isotope methods use elemental analysis (EA) to convert organic C to carbon dioxide (CO_2) coupled to continuous flow isotope analysis by IRMS. While this permits precise isotope measurements, the approach is limited in spatial resolution by a combination of physical separation techniques (manual dissection) and the overall EA-IRMS measurement sensitivity (typically on the order of 10^3 s of $\mu\text{g C}$).

To achieve better spatial resolution, the LA-IRMS uses a laser ablation system coupled with an optical microscope to image the surface of a solid sample, spatially identify a target for analysis, and then selectively sample the region for isotope analysis. Moran et al.^[1] described the basic components of an effective LA-IRMS system. A solid sample is placed in an ablation chamber designed to enable optical imaging (through quartz windows) while immersing the sample in a turbulent helium flow. A neodymium-doped yttrium aluminum garnet (Nd:YAG) laser ($\lambda = 266$ nm) is used to ablate a targeted portion of a sample's surface at a variety of different spot sizes ranging down to 50 μm . Importantly, the ablation process produces almost entirely small particulates (10's nm diameter) with very little production of volatile organic C compounds so that the large majority of the sample is preserved in the particulate phase. The small diameters of these particulates enhance their mobility in the applied helium carrier which entrains and transfers the sample fragments to a micro combustion reactor (940°C) containing nickel and platinum catalysts in the presence of a small oxygen flow. Rapid combustion converts all organic material in the particulates to CO_2 , which is cryogenically trapped in a capillary immersed in liquid nitrogen. Following trapping, the helium flow rate is reduced such that when the sample-derived CO_2 is sublimated, it is entrained in a relatively low amount of carrier gas and transferred to an IRMS for isotopic analysis. This reduction in carrier gas effectively concentrates the sample-derived CO_2 , which enables more of the sample to enter the IRMS, thereby increasing overall sensitivity from 10's μg C (EA-IRMS) to ~ 65 ng C (LA-IRMS). Taken together, the LA-IRMS system offers isotope measurement precision and accuracy comparable to that of EA-IRMS

but with improved measurement sensitivity and, most importantly, the inclusion of spatial resolution.

Despite the spatial and sensitivity advantages, there are some limitations to the LA-IRMS approach. First, samples must be dried prior to analysis which stops microbial processes and may require a set of time series samples to be collected when monitoring the fate of applied substrates. Second, samples need to fit into an ablation chamber for analysis. Moran et al.^[1] described a chamber with a half inch diameter. While chambers could be constructed to enable analysis of larger samples, any such modifications would require attention to maintain a sufficient carrier gas flow to facilitate sample transfer of ablation particulates through the combustion reactor for subsequent analysis. Finally, suitable samples for LA-IRMS must maintain sufficient cohesion such that the violent ablation process does not destroy the sample's structure. Bruneau et al.^[2] applied a polyethylene glycol resin to soil samples to enhance their structural cohesion and reduce damage to the sample's structure during the ablation process. Despite adding exogenous C to the sample, the work demonstrates that the amount of added C did not preclude the tracking of ^{13}C -labeled CO_2 into root biomass.

The spatial resolution of LA-IRMS can be observed in previous ablations of a strand of hair (Fig. 1A) where each of the visible 50 μm ablation pits provided sample material for an individual $\delta^{13}\text{C}$ measurement. The ability to direct the ablation sampling allows any part of the sample's surface to be analyzed while minimizing collateral damage to the adjacent sample. Moran et al.^[3] demonstrated the use of LA-IRMS to track labeled ^{13}C -bicarbonate incorporation into biomass of a hypersaline, photoautotrophic microbial mat. Following the isotope analysis, the group successfully performed DNA extraction for sequencing from around the

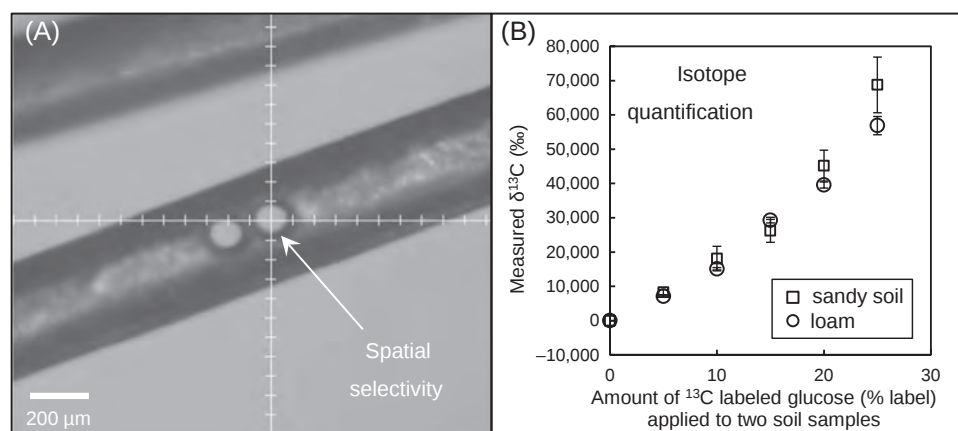


Fig. 1 LA-IRMS image of post-ablated hair samples (A) and data resulting from analysis of ^{13}C -glucose treated soil (B). The position of ablation pits in a strand of horse hair (A) is completely controlled by the user and can be aligned at any position over the sample's surface. Minimal damage was observed around the ablation pit and previous work on microbial samples has demonstrated successful extraction of DNA from sites surrounding the pits. Application of glucose solutions (B) containing different amounts of ^{13}C highlight the ability of LA-IRMS to differentiate between samples of varying ^{13}C content. The accuracy and precision of the approach require a much lower level of labeling for successful tracking (previous accuracy was demonstrated at better than 0.5‰).

ablation pits, demonstrating minimal damage to the surrounding sample by ablation. Thus, while LA-IRMS may not be considered non-destructive, it can be described as minimally destructive outside of the 50 μm ablation target.

One key capability of the LA-IRMS system is the spatial tracking of labeled C as it moves into a soil community. Previous work (Fig. 1B) applied various amounts of ^{13}C -glucose to the surface of two different soil types (a sand and a loam). Once dried, LA-IRMS was directed to the samples to measure the resulting ^{13}C content and, as shown, increasing amounts of label was easily measured in both the types of soil and the matrix effect of the soil did not inhibit isotope quantification. Moran et al.^[3] used a time series approach for tracking the uptake of bicarbonate into a phototrophic microbial mat. They found that integration of the LA-IRMS results across a depth profile of the mat showed isotopic agreement with EA-IRMS analysis of the entire mat segment. The added spatial resolution of LA-IRMS, however, was able to pinpoint the location of bicarbonate accumulation in the mat while spatially linked time series samples provided insight to the migration of the label through different layers of the stratified biomass. By providing spatially resolved isotope quantification, LA-IRMS can be used to track label incorporation into a soil as well as its subsequent migration or localization of the applied C.

LA-AMS

LA-AMS is an outgrowth of previously mentioned work^[1] The Aerodyne AMS was originally developed for characterization of the size and composition of atmospheric aerosols. In the work by Moran et al.,^[1] the AMS was used to demonstrate that ultraviolet laser ablation of organic and biological samples yields particles with a mode aerodynamic diameter of 100–200 nm, similar to the diameters obtained by LA of more refractory samples such as glasses and minerals.^[4] Furthermore, the EA of these particles was

shown by mass spectrometry to reflect that of the bulk sample, leading to the investigation and development (underway) of LA-AMS for the spatial and bulk characterization of biological and other organic samples such as plants, roots, fungi, and organic matter in soil.

LA-AMS uses an Aerodyne high-resolution time-of-flight AMS (HR-ToF-AMS; Fig. 2A) which is capable of providing real-time *in situ* measurements of total mass, size distributions, and chemical composition of aerosol particles.^[5] Even though the AMS undergoes continual development as higher resolution mass spectrometers come to market, the basic particle detection scheme remains consistent. The AMS uses an aerodynamic lens to sample submicron particles into vacuum, where they are aerodynamically sized based on ToF measurements before being thermally vaporized on a heated surface and chemically analyzed via electron ionization (EI) mass spectrometry. Due to the use of thermal vaporization, the HR-ToF-AMS is most effective for detecting non-refractory, more volatile organic species in particulate matter. EI leads to substantial fragmentation of the analyte ions such that the molecular ion appears along with many fragment ions, which is problematic especially for complex organic mixtures. EI has benefits over other techniques, however, by being sensitive to nearly all organic species and having a nearly constant response factor over a wide range of chemical compositions, minimizing the use of calibrations and/or standards. Together, these attributes enable the total mass of complex aerosol mixtures to be measured without calibration of individual species or the use of a large number of standards. Due to sample fragmentation by EI, the mass spectra are composed primarily of fragments of larger compounds. However, by employing high-resolution mass spectrometry, the identities of different ion fragments at the same nominal m/z can be determined, allowing for the determination of classes of compounds including hydrocarbons, nitrates, amines, sulfates, and chlorides as well as the measurement of elemental ratios (e.g., O/C, H/C, and N/C) within organic mixtures.

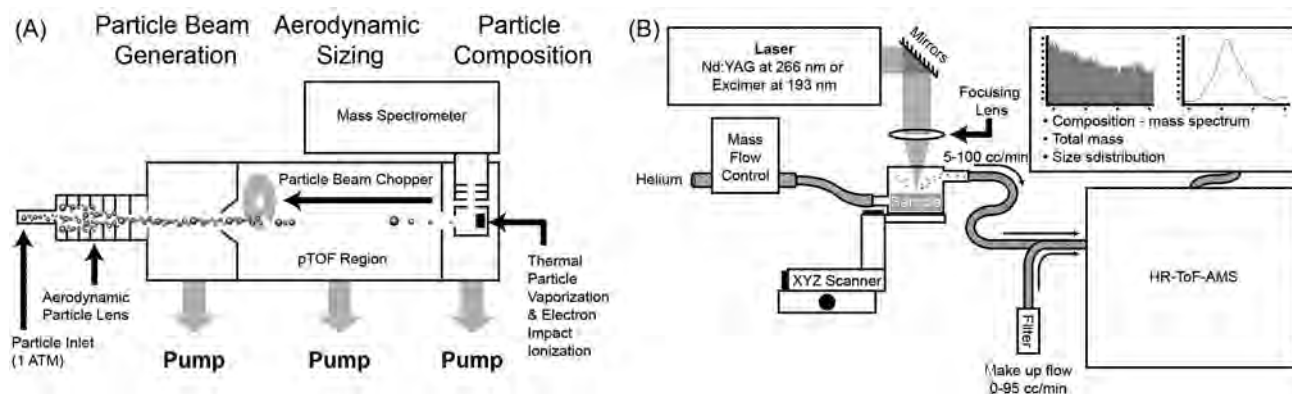


Fig. 2 Schematic of (A) the HR-ToF-AMS showing the sample inlet, aerodynamic lens, particle sizing, vaporization and electron impact ionization, and time-of-flight mass spectrometer and (B) the HR-ToF-AMS interfaced to an LA system. The current LA-AMS uses an excimer-based LA system (Photon Machines G2) with resolution down to 2 μm .

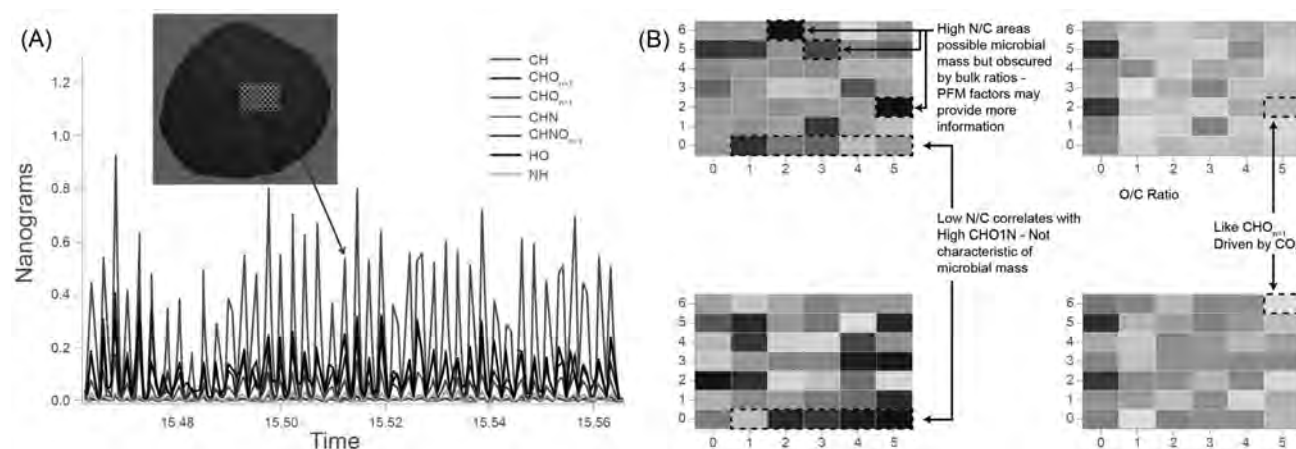


Fig. 3 Spatially linked LA-AMS time series of chemical families identified within a soil aggregate (A) obtained from the ablated sites as pictured in the inset. Chemical families include key ion fragments as determined by the AMS: CH (unoxidized hydrocarbons), $\text{CHO}_{n=1}$ (singly oxidized hydrocarbons), $\text{CHO}_{n>1}$ (multiply oxidized hydrocarbons), CHN (amines), $\text{CHNO}_{n=1}$ (organic nitrates), OH (hydrated species), and NH (ammonia). These can be related to their parent compounds such as alkanes, aldehydes, alcohols, carboxylic acids, and contaminants like polycyclic aromatic hydrocarbons (PAHs). The excimer laser is focused to 10 nm and is operated in a 3-second cycle beginning with a burst of 20 shots (1 second), followed by a 2-second rest to allow the transfer line to clear. The identities and abundances of the ions identified in the spatially linked mass spectra are used to produce chemical maps (B) that are related to spatial properties of the soil aggregate.

In typical applications, the AMS samples directly from the air at atmospheric pressure. The ability to analyze samples under ambient atmospheric conditions permits direct coupling of the AMS to a laser ablation output (Fig. 2B) without requiring any further sample preparation or purification. Furthermore, the rapid real-time response of the HR-ToF, acquiring 10,000 or more mass spectra per second, allows fast scanning of the laser over a sample surface. The known scan rate of the laser can then be used to create a spatial map (Fig. 3A) of elemental and molecular ratios of key species present in the sample with a sensitivity of 0.1 ng or less. Although preliminary, these results demonstrate how LA-AMS can be used to answer crucial questions in soil science such as correlations (Fig. 3B) between species indicative of microbial mass. An additional capability of LA-AMS is the ability to depth profile with a resolution of down to 100 μm , opening up the possibility of 3-D mapping correlating to morphology in combination with a non-destructive method technique such as X-ray tomography.

NANO-DESI

Nano-DESI^[6,7] is a technique for analysis of organic and biological molecules on surfaces. It is complementary to the ability of LA-AMS to measure fragmentation mass spectra at high lateral and depth resolution by having the capability to determine the identity of molecules on a sample surface by soft method electrospray ionization, which causes little or no molecular fragmentation, coupled to an ultrahigh-resolution mass spectrometer. Nano-DESI uses a solvent

bridge (Fig. 4) composed of a small ($\sim 100 \mu$) solvent droplet formed between two capillaries to dissolve molecules off the sample's surface. The analytes are then aspirated into the inlet of a mass spectrometer by a high voltage applied between the mass spectrometer and the primary capillary which also supplies solvent to maintain the droplet on the sample surface. The droplet can be translated across the sample to spatially resolve the sample's molecular composition and different solvents (i.e., polar vs. non-polar) can be applied to selectively dissolve/analyze different chemical classes. Nano-DESI is capable of detecting small ($<10 \text{ ng}$) quantities of material. This method has

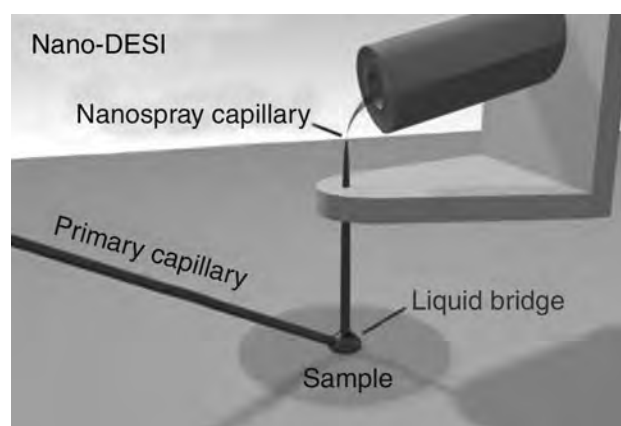


Fig. 4 Schematic of nano-DESI source configuration for analysis of substrate deposited analyte.

Source: From Roach, Laskin, et al.^[7] ©2010 Royal Society of Chemistry. Reproduced with permission.

been applied to a number of sample types including soil, atmospheric aerosols,^[6] and spatially resolved analysis of metabolites in bacteria grown on agar plates.^[8]

In the work by Lanekoff et al.,^[8] nano-DESI was used to spatially profile chemical gradients generated by microbial communities grown on agar plates. Nano-DESI allowed the determination of metabolites from colonies of living *Synechococcus* sp. PCC 7002 and, by not impacting viability of the cultures, enabled time series analysis through the culture's growth. The high sensitivity of nano-DESI allowed the identification of glycolipids not previously reported by using conventional cell extraction methods. Using the spatial profiling capability of nano-DESI, it was found that most of the lipids and metabolites were located on the colony while sucrose and glucosyl were more present in the agar with the gradients of these compounds dependent upon the age of the colony.

CONCLUSIONS

A suite of emerging techniques are poised to provide new data streams for interpreting C migration into and through a soil. By enabling spatially resolved isotope (¹³C) analysis, LA-IRMS offers a method for tracking isotopically labeled substrates into a soil and can identify specific locations where these substrates accumulate or are most rapidly consumed. LA-AMS offers high-throughput, spatially resolved (~2 μm) identification of classes of organic compounds within a soil. This technique can be coupled with LA-IRMS to identify key organics present in sections of a soil showing high ¹³C label accumulation and determine the degree to which the added labeled compound is being chemically modified in this location. Finally, by offering detailed chemical identification of soluble organics, nano-DESI can offer more detailed chemical identification and help identify specific chemical transformations or compound migration in a soil. It is clear from the results of the use of nano-DESI for the determination of chemical gradients produced by microbial communities on agar, as well as the identification of chemical species previously unreported as metabolites, that nano-DESI is a powerful tool with great potential for spatial determination of metabolites of microbes, fungi, roots, and other biological sources in soil.

Taken together, these three methods provide a platform for identifying substrate localization in a soil and, through chemical identification, for tracking the recalcitrance or

reaction products of applied organics within these highly resolved locations.

ACKNOWLEDGMENT

A portion of the research was performed using EMSL, a DOE Office of Science User Facility sponsored by the Office of Biological and Environmental Research and located at Pacific Northwest National Laboratory.

REFERENCES

1. Moran, J.J.; Newburn, M.K.; Alexander, M.L.; Sams, R.L.; Kelly, J.F.; Kreuzer, H.W. Laser ablation isotope ratio mass spectrometry for enhanced sensitivity and spatial resolution in stable isotope analysis. *Rapid Commun. Mass Sp.* **2011**, *25* (9), 1282–1290.
2. Bruneau, P.M.C.; Ostle, N.; Davidson, D.A.; Grieve, I.C.; Fallick, A.E. Determination of rhizosphere C-13 pulse signals in soil thin sections by laser ablation isotope ratio mass spectrometry. *Rapid Commun. Mass Sp.* **2002**, *16* (23), 2190–2194.
3. Moran, J.J.; Doll, C.G.; Bernstein, H.C.; Renslow, R.S.; Cory, A.B.; Hutchison, J.R.; Lindemann, S.R.; Fredrickson, J.K. Spatially tracking (13) C-labelled substrate (bicarbonate) accumulation in microbial communities using laser ablation isotope ratio mass spectrometry. *Environ. Microbiol. Rep.* **2014**, *6* (6), 786–791.
4. Alexander, M.L.; Smith, M.R.; Hartman, J.S.; Mendoza, A.; Koppelaar, D.W. Laser ablation inductively coupled plasma mass spectrometry. *Appl. Surf. Sci.* **1998**, *127*, 255–261.
5. Jayne, J.T.; Leard, D.C.; Zhang, X.F.; Davidovits, P.; Smith, K.A.; Kolb, C.E.; Worsnop, D.R. Development of an aerosol mass spectrometer for size and composition analysis of submicron particles. *Aerosol Sci. Tech.* **2000**, *33* (1–2), 49–70.
6. Roach, P.J.; Laskin, J.; Laskin, A. Molecular characterization of organic aerosols using nanospray-desorption/electrospray ionization-mass spectrometry. *Anal. Chem.* **2010**, *82* (19), 7979–7986.
7. Roach, P.J.; Laskin, J.; Laskin, A. Nanospray desorption electrospray ionization: An ambient method for liquid-extraction surface sampling in mass spectrometry. *Analyst* **2010**, *135*, 2233–2236.
8. Lanekoff, I.; Geydebrekht, O.; Pinchuk, G.E.; Konopka, A.E.; Laskin, J. Spatially resolved analysis of glycolipids and metabolites in living *Synechococcus* sp. PCC 7002 using nanospray desorption electrospray ionization. *Analyst* **2013**, *138*, 1971–1978.

Soil Carbon: Farming and Economic Aspects

Bruce A. McCarl

Agricultural Economics, Texas A&M University, College Station, Texas, U.S.A.

Abstract

A discussion is presented on key issues that will determine whether farming for sale of carbon will ever be an economic proposition. These include the needed market developments that would allow soil carbon to sell, the drawbacks of saturation, and their most likely associated market discounts, market rules, transaction costs, the influence of the biofuel boom, risk, and the potential to bridge to a lower-emission future.

INTRODUCTION

The last years have shown growing interest in soil carbon sequestration. However, soil carbon sequestration is not a significant income source outside of select niche markets [e.g., under the Chicago Climate Exchange (CCX) or the Climate Trust].^[1] This entry reviews the prospects and some obstacles from an economic perspective covering:

- Greenhouse gas (GHG) emission restrictions and the context for making money
- Saturation/new equilibrium and volatility
- Fungibility and market discounts
- Rules and early actor risk
- Soil carbon as a bridge to the future
- Transaction costs, monitoring, and verification
- Signals that are changing farming
- Risk and the no-till panacea

GHG EMISSION RESTRICTIONS AND MONEY

The Intergovernmental Panel on Climate Change^[2] Fourth Assessment Report indicated a high degree of certainty that anthropogenic GHG emissions are contributing to climate change. Furthermore, this report also indicates that climate change is risky and suggests that society reduces GHG emissions. Soil carbon comes into this picture in which carbon dioxide (CO₂), the primary GHG, is released when the soil is disturbed, while carbon content of soil increases (carbon is sequestered) when disturbance is reduced ultimately drawing carbon from the atmosphere. Thus, altering agricultural soil management so as to reduce disturbance generates a net reduction in atmospheric GHGs. This is why there is potential for farming and selling soil carbon.

Market trading is another aspect of soil carbon farming. Economists began suggesting pollution emission rights trading in the late 1960s,^[3,4] allowing firms for which emission reduction is expensive to buy reduction credits from

firms that could reduce more cheaply. This was implemented in the sulfur dioxide trading market where power plants for which sulfur reductions were expensive could buy from other firms in an electronic trading market.^[5] Carbon offers such possibilities. The vast bulk of U.S. emissions come from coal-fired power plants and petroleum use.^[6] Emission reductions in energy use can be very costly raising the issue of whether a market could arise where firms needing offsets could buy from cheaper sources. Agriculture is one potential source. Several estimates indicate that substantial agricultural soil carbon could arise for prices below U.S. \$25 per ton (e.g., McCarl and Schneider^[7] and Antle et al.^[8]). On the other hand, anecdotal estimates of carbon capture and geologic storage of coal power plant emissions are above U.S. \$200 per ton, showing an income earning potential from farming soil carbon.

However, while market trading and GHG regulation are widely discussed, a market will only emerge if we have emission limits. The Kyoto Protocol would have imposed binding emission limits and started a market, but the U.S. Government declined the participation. Instead, in the United States, there is a voluntary “Clear Skies Initiative”^[9] that aims to reduce emission intensity but does not contain features that start a trading program. There are some regional or experimental trading programs. For example, there is voluntarily trading under the CCX and some contemporary or emerging regional binding emissions, which is limited in places like Oregon, California, and the Northeast. However, these are niche markets in which some but not all are covered. Furthermore, U.S. prices have been small being around U.S. \$1–4 per ton CO₂, but European prices have been higher in the vicinity of U.S. \$30. Widespread trading will not occur until some national GHG emission limit is imposed and trading rules developed that permit entry of agricultural soil carbon. (A development that has not really arisen in the Kyoto-based trading in the rest of the world.) Prices may be much higher with projected prices under some atmospheric GHG stabilization schemes being well up into the multiple U.S. \$100s.

SOME ISSUES

Saturation/New Equilibrium and Volatility

Soil carbon is subject to a physical capacity constraint, given a land use as well covered by West and Six^[10] and likely elsewhere in this encyclopedia. When soil disturbance is reduced, then the contribution of carbon to the soil or the destruction of soil carbon is limited through oxidation. Simultaneously, the rate of decomposition starts to increase. When these items equilibrate, a maximum soil capacity is reached and more can be attained only by augmenting the net addition or reducing decomposition by other actions. West and Post^[11] showed that the sequestration from tillage changes may persist for only 10–15 years. Furthermore, reverting to an increased degree of disturbance would rapidly release the carbon, leading soil carbon to be commonly called as volatile or impermanent. The permanence of sequestered carbon has been a major factor in its lack of entry in the international Kyoto trading system.

This approach to equilibrium/saturation has implications for the value of soil carbon and for future farmer options. Farmers must reduce disturbance to get paid but may only be paid while accumulation occurs. In turn, they could not revert to traditional practices without releasing the carbon. This implies a reduced price for carbon if a permanent agreement to maintain reduced disturbance is not negotiated. West and Post^[11] showed that this occurs at a rate of about ¼ ton per acre or a total of around 4 tons over 15 years. Valuing this at the U.S. CO₂ price of U.S. \$4 per ton (about U.S. \$14.70 per ton carbon) implies that the permanent income potential is about U.S. \$60 per acre, while the seven times higher European price yields about U.S. \$420 per acre, either of which may not be large enough for a farmer to permanently commit to changes in disturbance regime. Additionally, Kim, McCarl, and Murray^[12] showed that if farmers are to be paid to maintain the carbon beyond the time it ceases to accumulate or if they lease it for say 20 years, then they are allowed to change practices that the value and returns per acre would be discounted by ½ to 2/3.

Rules, Land Use History, and Early Actor Risk

In reciting a list of possible obstacles and reflecting over conversations with farmers, timing is a practical concern. The Kyoto Protocol has an “additionality” clause saying that the only offsets that should be paid for are those which were stimulated by the market. Numerous farmers have been using reduced disturbance tillage methods for some time, and they often argue that they should be compensated for being good actors. However, strict adherence to additionality would only have tillage changes stimulated by the carbon program being eligible for payment. When a market comes into existence, then it is highly likely that disturbance changes after a certain date are the only ones that

will be paid for; so incentives to adopt are not strong, and in fact, the perverse incentive exists to diminish soil carbon holdings when a market looms on the horizon. Thus, there is a disincentive to farming soil carbon. Furthermore, under the above saturation discussion, the time one is or has been in the regime may be a key factor in payment amount and duration.

Soil Carbon as Part of a Response Agenda

While permanence arguments have been detrimental to opportunities for marketing soil carbon, there are encouraging factors. By no means in a future trading world will soil carbon be dominant. To reduce GHG concentrations, society must reduce emissions from coal-fired power plant and petroleum consumption since they compose roughly 85% of the emissions.^[6] While there are possible ways to do this, they take time to develop and require turnover of capital and the vehicle fleet. Farming soil carbon is something we can do bridging to a lower-emission future.

Transaction Costs, Monitoring, and Verification

Monitoring and verification have been widely discussed as potential impediments to soil carbon sales. Certainly, there is a need to assure that the carbon is accumulating or remaining in place, and thus, there will be a measurement and monitoring cost. Mooney et al.^[13] argued that the cost will be low in the few dollars per ton range. Reportedly, implementation of this is an obstacle in the European model.

In addition, there is a question of contract size. Minimum exchange contracts can be large, i.e., 10,000 tons CO₂, which amounts to at the CCX rating of ½ ton CO₂ per acre per year to the tillage change induced carbon on 20,000 acres or given average farm size of 500 acres, the results from 40 operations, and in a developing country with farm size of a hectare, this would be 8000 operations. This implies the need for a broker as occurring under CCX with Iowa Farm Bureau. Crop insurance agents who pull together groups of farmers typically retain 25% of the premiums they collect, while real estate agents get 6–7%. This portends a substantial price gap between market prices and what farmers receive, further reducing the income opportunities.

Signals That are Changing Farming

While we commonly discuss soil carbon as an option, there are some trends that may be limiting its potential for the future. Fundamentally, this involves the implications of rising energy prices and the biofuel boom. Petroleum prices have risen substantially. This has increased the incentives for farmers to both reduce energy use and produce biofuel feedstocks. Intensive tillage uses substantially more energy than reduced forms, and a substantial increase in reduced

tillage is being observed potentially reducing the likely future tillage change response in a carbon market.

Additionally, the biofuel boom has raised commodity prices and land values and is likely to stay for some time. This provides great incentives to increase land disturbance, e.g., deforesting or converting grasslands into farming, both reducing sequestration. It also limits the potential response to a carbon market.

Risk and the No-Till Panacea

Practical farming considerations also enter into the farming soil carbon picture. Bennett,^[14] in commenting on a paper a number of years ago, alluded to the risk introduced by long-term no-till contracts under the possibility that factors like weed resistance to common herbicides. Furthermore, the work done by Klemme^[15] implies that there are obstacles involved that are limiting reduced tillage adoption.

SUMMARY AND CONCLUSIONS

Whether, and to what extent, carbon will be profitably farmed in agricultural soils will depend on the issues of practical economics and policy design. In some cases, carbon can be enhanced through small alterations in land uses, perhaps to the benefit of all. In other cases, incentives or GHG prices may simply be insufficient to stimulate changes in land use practices. In general, the establishment of a carbon trading market should provide additional economic opportunities for landholders and rural communities. The big questions are as follows:

- Will agricultural soils be approved as a saleable GHG offset with workable rules defined?
- Will the incentives be large enough so that landowners will adopt appropriate practices in their own best interests?
- Can an emissions trading program be designed so the carbon that is paid for remains sequestered for as long as needed?

REFERENCES

1. Chicago Climate Exchange, 2007. Available at <https://www.theice.com/ccx>.
2. Intergovernmental Panel on Climate Change. *Climate Change 2007*, Cambridge University Press: Cambridge, 2007. Available at <http://www.ipcc.ch/>.
3. Dales, J.H. Land, water, and ownership. *Can. J. Economics* **1968**, *1* (4), 791–804.
4. Crocker, T.D. The structuring of atmospheric pollution control systems. In *The Economics of Air Pollution*;

- Wolozin, H., Ed.; W.W. Norton & Co.: New York, 1966; 61–68.
5. Stavins, R.N. Lessons learned from SO₂ allowance trading. *Choices* **2005**, *20* (1), 53–57.
6. US Environmental Protection Agency. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2005*, April 2007, USEPA #430-R-07-002. Available at <http://epa.gov/climatechange/emissions/usinventoryreport.html>.
7. McCarl, B.A.; Schneider, U.A. Greenhouse gas mitigation in U.S. agriculture and forestry. *Science* **2001**, *294*, 2481–2482.
8. Antle, J.M.; Capalbo, S.M.; Paustian, K.H.; Ali, M.K. Estimating the economic potential for agricultural soil carbon sequestration in the central United States using an aggregate econometric-process simulation model. *Climatic Change* **2007**, *80* (1–2), 145–171.
9. Bush, P.G.W. *The Clear Skies Initiative* 2002. Available at <http://www.whitehouse.gov/news/releases/2002/02/clearskies.html>.
10. West, T.O.; Six, J. Considering the influence of sequestration duration and carbon saturation on estimates of soil carbon capacity. *Climatic Change* **2007**, *80*, 25–41.
11. West, T.O.; Post, W.M. Soil organic carbon sequestration rates for crops with reduced tillage and enhanced rotation. *Soil Sci. Soc. Am. J.* **2002**, *66*, 1930–1946.
12. Kim, M.-K.; McCarl, B.A.; Murray, B.C. Permanence discounting for land-based carbon sequestration. *Ecol. Econ.* **2008**, *4*, 763–769.
13. Mooney, S.; Antle, J.M.; Capalbo, S.M.; Paustian, K. Design and costs of a measurement protocol for trades in soil carbon credits. *Can. J. Agri. Econ.* **2004**, *52* (3), 257–287.
14. Bennett, J.F. A commentary on Marland, G., B.A. McCarl, and U.A. Schneider, Soil carbon: Policy and Economics. In *Carbon Sequestration in Soils: Science, Monitoring, and Beyond*, Proceedings of the St. Michaels Workshop; St. Michaels, 1998; Rosenberg, N.J., Izaurrealde, R.C., Malone, E.L., Eds.; Battelle Press: Columbus, 1999; 168–170.
15. Klemme, R.M. A stochastic dominance comparison of reduced tillage systems in corn and soybean production under risk. *Am. J. Agr. Econ.* **1985**, *67*, 550–557.

BIBLIOGRAPHY

1. Iowa Farm Bureau. *Carbon Credit Aggregation Program*. Available at http://soilcarboncenter.k-state.edu/conference_2007/pdf_files_2007/Michael_Walsh-David_Miller.pdf.
2. Marland, G.; McCarl, B.A.; Schneider, U. Soil carbon: Policy and economics. *Climatic Change* **2001**, *51*, 101–117.
3. Oregon Climate Trust. Available at <http://www.climatetrust.org/>.
4. United Nations Framework Convention on Climate Change, *Kyoto Protocol*. Available at <http://unfccc.int/resource/docs/convkp/kpeng.pdf>.

Soil Conservation: Incentives

David Sanders

World Association of Soil and Water Conservation, Bristol, U.K.

Abstract

Soil degradation is one of the most widespread and serious environmental problems faced by humanity, and millions of dollars are being spent annually on soil conservation programs. Some of these programs are succeeding, but generally progress has been far too slow. As a result, a variety of incentives are being used to encourage land users to adopt soil conservation measures.

TYPES OF INCENTIVES

There are many types of incentives, but they can be divided into two broad categories: direct and indirect.^[1] Direct incentives can be provided in the form of wages, grants, subsidies, and loans, or in kind through the provision of food aid, agricultural implements, livestock, trees, seeds, etc., or in a combination of the two. Indirect incentives include fiscal and legislative measures such as tax incentives, guaranteed inputs and input prices, and land tenure arrangements. They include services such as extension services, technical assistance, the use of agricultural equipment, marketing, storage, education, and training. They also include social services, community organization, and the decentralization of decision-making. Of the two, indirect incentives are by far the more important. This particularly applies to land tenure rights, markets and prices, and the decentralization of decision-making.

Equally important are disincentives. Like incentives, disincentives come in many forms, varying from cash or in kind payments to tax disincentives and legal measures.

In the end, the way a land user reacts to soil degradation and soil conservation programs is most likely to be the result of weighing up all the different incentives and disincentives that may be available or in force at the time.

WHEN SHOULD INCENTIVES BE USED?

The effects of land degradation fall within two broad categories: on-site (or on-farm) and off-site (or off-farm). These can be broken down again into the following four categories, depending on whether the preventive measures needed are perceived by those involved to be cost-effective ("economic") or not cost-effective ("uneconomic"):

1. Where the problem is on-site and the treatment is economic, i.e., it is perceived by the land user to more than pay for itself, the task is primarily one of extension and

providing the land user with the correct type of information. No other incentive should be needed. For example, it may be possible to solve the problem by helping farmers change their farming practices from, say, clean cultivation to no-tillage, as has been done very effectively in Brazil through demonstrations and extension.^[2]

2. Where the problem is off-site and the treatment is economic, the task is again one of extension and no other incentive is needed. For example, farmers on lower slopes who are suffering from excessive off-site runoff and erosion can be protected if farmers on the higher slopes are shown through extension how to grow more profitable perennial crops that provide better ground cover than the grown annual crops.
3. Where the problem is on-site and the only effective treatment is uneconomic, i.e., not perceived by the land user to be worth doing (e.g., the construction of an expensive terrace system), three possibilities arise as follows:

The land user treats the problem for the public good but contrary to his/her own economic advantage.

The land user is forced to carry out the required measures (regulation).

The land user is subsidized to do what is required through one or more incentives. This last possibility is the situation where indirect and direct incentives are most commonly used. For example, in the 1960s, a large and effective program was launched in Jordan to change the use of steep, rocky land that was eroding. Food aid was provided to farmers in return for building stonewall terraces and planting their land to olive and fruit trees instead of continuing to cultivate it for annual cereal crops.^[3]

4. The most difficult situation arises where the problem manifests itself off-site and the on-site treatment is uneconomic for the land user to implement, e.g., where the cultivation of steep upper slopes of a catchment

leads to flooding and erosion lower down, in some cases some distance away. Here the use of incentives may prove to be essential, not only in the short term but also on a continuing basis.

CAUSES AND CONSTRAINTS

The importance of fully involving the land user in soil conservation programs is generally recognized, but what are not widely appreciated are the constraints under which land users operate.^[1]

There are extrinsic and intrinsic factors that affect all land users and the decisions that they take. Extrinsic factors include such things as the farmer's resources of land, capital, and equipment. The intrinsic factors include awareness, technical understanding, skills, and attitudes to conservation. Distinguishing between these factors is important if the correct incentives are to be selected. Too often, programs concentrate on the intrinsic factors and therefore use incentives in the wrong way; for example, it may be a waste of time to provide training to farmers in some new conservation technology if the real reason that they are not adopting conservation practices is lack of capital or of long-term access to land. This is a mistake frequently made in developing countries. On the other hand, a wealthy land user may respond better to training in improved farming techniques than to being provided with subsidies to undertake conservation measures. Without this understanding, incentives may easily be misdirected.

GENERAL PRINCIPLES FOR THE USE OF INCENTIVES IN SOIL CONSERVATION PROGRAMS

The adoption of agricultural innovations, in general, and in soil conservation in particular, is a complex process. Few farmers are able to adopt even simple technologies, let alone packages of conservation measures, without adjusting their traditional practices and their livelihood strategies. A study^[4] on the use of incentives in soil conservation therefore concluded, among other things, that a necessary condition for adoption is that changes must be profitable to the farmer. But profit by itself may not be sufficient to stimulate the required change. Profit maximization, although an important driving force, is by no means the only motivational force.

It must also be understood that, to be effective, the objective of incentives should be to alter the long-term behavior of the land user, not simply to boost adoption rates to meet project targets. It is all too easy for incentives to turn into wages without the recipients making the link between the incentives and the desired conservation work. Far too often, incentive schemes, such as food-for-work

programs, turn into work-for-food programs, without altering the land users' behavioral pattern.

Essentially, incentives are needed when the adoption of conservation measures is not profitable to the land users. Broadly, they are justified when the adoption of these measures produces benefits that are external to the farm, and the provider of the incentives receives the benefits. If society benefits, there is a case for society paying.

Incentives are not simply justified by low incomes. If the desired conservation measures are profitable to land users, it is most likely that land users will find the way to finance the necessary changes. If they cannot, then it may be necessary to provide enabling incentives, such as cheap credit or improved land rights. Incentives may be necessary to overcome barriers to the adoption of profitable conservation measures. Experience demonstrates that incentives may be justified for both rich and poor land users where society clearly stands to gain.

Incentives may also be justified on a continuing basis. Numerous examples exist of projects failing after incentives were withdrawn. In these cases, the land users obviously did not consider the conservation practices profitable without the incentives. But if society derives adequate benefits from the conservation activities, then there is a case for the incentives to be continued indefinitely. It may be necessary, and only fair, that those who benefit pay the cost.

THE IMPACT OF INCENTIVES

To evaluate the impact of incentives, it is necessary to carefully monitor their use and effects. Unfortunately, this is difficult to do and, in practice, is seldom done. Changes in behavior patterns are often difficult to assess and, as a result, the benefits attributed to the incentives are hard to measure.

It is also difficult to differentiate between the effects of different stimuli to which farmers respond, including external factors such as weather and markets. A crucial issue is whether land users actually desire a change and are prepared to alter their land use and management systems. Unless the change is genuinely desired, the land user will just "take the money and run."

In a study,^[4] it was significant that contributors kept returning to the issue of profitability. In one way or another, they all tended to conclude that soil conservation must be profitable for the land user if it is to be sustained. To some this meant that soil conservation must be profitable in its own right without outside incentives. To others it meant that outside incentives must be provided as long as the conservation activity was not profitable by itself. But most agreed that, when the measures are profitable to the land user, they are likely to be adopted and maintained. That is, using incentives to make projects profitable contributes to their success. And, importantly, the sustainability of soil

conservation measures depends on their continued profitability, either with or without continued external incentives.

CONCLUSION

It would seem that the most important requirement is the removal of the many disincentives to soil conservation that exist. This is likely to be very difficult to achieve. Many government policies are designed to overcome other problems, such as inadequate food production and low farm incomes, and these unintentionally contribute to soil degradation. The problem is understandable—governments, and society in general, are inclined to view food production and the welfare of farmers as more important than soil conservation. A major problem faced by those working in soil conservation is to be able to reconcile the multiple objectives of society and not simply to argue for soil conservation for its own sake.

Another challenge is to design sustainable incentives. Soil conservation incentives, particularly in the form of subsidies and technical assistance, tend to be temporary measures. Usually, when these measures are withdrawn, conservation activities also cease. There is a need to build the right types of incentives into the social and economic system for the continuance of soil conservation programs.

In addition to this, the effectiveness of using incentives for soil conservation is constrained by the institutions within which they are applied. The challenge is to devise incentives that work within the particular institutional structure in which the problem exists.

Certainly, there is no single incentive appropriate for every problem.

REFERENCES

1. Sanders, D.; Cahill, D. Where incentives fit in soil conservation programs. In *Incentives in Soil Conservation*; Huszar, P.C., Sombatpanit, S., Enters, T., Sanders, D.W., Eds.; Science Publishers, Inc.: Enfield, 1999.
2. FAO. Chapter 3, *Conservation Agriculture—Case Studies in Latin America and Africa*. FAO Soils Bulletin; Food and Agriculture Organization of the United Nations: Rome, 2001; Vol. 78.
3. Sanders, D.W. Food and agriculture organization activities in soil conservation. In *Conservation Farming on Steep Lands*; Moldenhauer, W.C., Hudson, N.W., Eds.; Soil and Water Conservation Society and World Association of Soil and Water Conservation: Ankeny, 1988.
4. Sanders, D.; Huszar, P.; Sombatpanit, S.; Enters, T. Chapter 23, conclusions. In *Incentives in Soil Conservation*; Huszar, P.C., Sombatpanit, S., Enters, T., Sanders, D.W., Eds.; Science Publishers, Inc.: Enfield, 1999.

Soil Conservation: Strategies

Winfried E.H. Blum

Institute of Soil Research, University of Natural Resources and Applied Life Sciences, Vienna, Austria

Abstract

National and international soil conservation strategies are presented, and a new proposal for an Intergovernmental Panel on Land and Soil, in analogy with the Intergovernmental Panel on Climate Change, is presented, aiming at a world soils policy.

INTRODUCTION

Soil conservation means to maintain or to improve the quality of soil for providing goods and services to humankind and the environment. This can best be achieved through participatory approaches involving as many stakeholders as possible. However, successful conservation strategies require a solid scientific and technical background as well as an infrastructure for the transfer of this knowledge to those who need it, e.g., stakeholders and decision makers. Finally, substantial economic resources to sustain soil conservation efforts in the medium and long term are also needed because goodwill alone is not sufficient.

This means that soil conservation is a task for communities at large, on a national and an international basis.

In view of the fact that soils are a limited resource and need to be protected for the benefit of future generations, soil conservation is a task not only for the rural population, particularly farmers and foresters, but also for the whole community. The following deliberations are mainly based on Montanarella.^[1]

NATIONAL SOIL CONSERVATION STRATEGIES

From a historical point of view, several national soil protection strategies and policies have been developed in different countries of the world.

In the United States, the U.S. Soil Conservation Act of 1935 was created, following the catastrophic soil degradation.^[2] The U.S. Soil Conservation and Domestic Allotment Act of 1935 had a long-lasting impact on soil protection strategies and the development of soil conservation actions, both in the United States and at a global scale.^[3] This soil conservation act was later followed by the Food Security Act of 1985 and the Farm Bill, 1996, both indicating that soil conservation in the United States was mainly aiming at combating soil erosion and maintaining soil fertility for agricultural production.

Even with this limited focus, this was a basic step for the development of strategies for soil conservation at national and international levels.

Example of national soil protection strategies is the Federal Soil Protection Act of Germany (1998). This soil protection act is enlarging the focus of soil protection, comprising all the functions of soils, such as natural functions, functions as an archive of natural and cultural history, and functions useful to humans, which means that soil is regarded not only as a basis for the production of biomass but also as a filter, buffer, and transformation substrate, as a habitat for biota and a gene reserve, as well as a physical basis for the development of technical infrastructure and a source of raw materials and geogenic and cultural heritage.

The soil action plan developed by DEFRA^[4] for England goes in the same direction. This first soil action plan for England aims at a comprehensive view on the protection of all soil functions and contains 52 actions on issues ranging from soil management on farms to soils in the planning system, soils and biodiversity, soil contamination, and role of soils in conserving cultural heritage and landscape, and all these actions, taking steps toward more sustainable soil use and protection.

INTERNATIONAL SOIL CONSERVATION STRATEGIES

Increasing concerns about worldwide soil degradation processes, especially in Europe, led to transnational soil conservation strategies, e.g., the Alpine Convention, including eight states with alpine environments in Central Europe, setting legal and administrative measures for the protection of soils, by applying the precautionary principle.

Step forward in international soil protection is the European Union (EU) Thematic Strategy for Soil Protection,^[5] which is based on the recognition of the multifunctionality of soils, in which soils are no longer considered

only as construction surfaces or dumping sites or means for agricultural production, but as a fundamental environmental compartment, similar to air and water, performing vital ecological, social, and economic services for European citizens, including filtering and buffering contaminants, producing clean drinking water, and having a pool of biodiversity, a source of raw materials, a sink of atmospheric carbon dioxide, and an archive of cultural heritage.

Besides these regional policies, global policies aim at strategies for soil conservation, such as the World Soil Charter and the World Soils Policy, and several UN conventions, such as the 1992 Framework Convention on Climate Change, the 1992 Convention on Biological Diversity, and the 1994 Convention to Combat Desertification. However, the strategies behind these global conventions are not strong enough to produce results in achieving global soil protection.

This led to new initiatives in Europe, to create an Intergovernmental Panel on Land and Soil (IPLS), in analogy with the Intergovernmental Panel on Climate Change (IPCC) of the Climate Change Convention. The IPLS should give scientific advice to the public, like the IPCC, and would have the following functions:^[6]

1. To serve as a clearing house for ongoing and periodic assessment of global land and soil degradation
2. To assess and synthesize globally the scientific, technical, and socioeconomic information relevant for the understanding of the risk of human-induced land and soil quality change
3. To address the variety of land use and soil management issues, including desertification as related to environmentally sustainable development
4. To stimulate and involve the scientific community to advance and develop the science of soils in sustainable land use

5. To assist actively national, regional, and global decision makers in developing strategies to assess, monitor, and mitigate negative impacts of land and soil use

CONCLUSION

From these national and international strategies, it becomes clear that a nested system of soil conservation strategies on a worldwide level is needed. The basis of this has to be developed by policy-relevant soil information in order to make knowledge-based decisions. Unfortunately, there is a lack of sufficient data about soils in many parts of the world, including Europe.

The further development of the EU Thematic Strategy for Soil Protection will allow the exploration of new methods for international cooperation in this field.

REFERENCES

1. Montanarella, L. Policies for a sustainable use of soil resources. In *Function of Soils for Human Societies and the Environment*, Special Publication 266; Frossard, E., Blum, W.E.H., Warkentin, P.B., Eds.; Geological Society: London, 2006; 149–158.
2. Hannam, I.; Boer, B. *Legal and International Frameworks for Sustainable Soils: A Preliminary Report*; IUCN: Gland, Switzerland, and Cambridge, 2002.
3. Helms, D. *A Historical Guide to the U.S. Government*; Oxford University Press: New York, 1998; 434–439.
4. DEFRA. *The First Soil Action Plan for England 2004–2006*; Department for Environment, Food and Rural Affairs (DEFRA): London, 2004.
5. EC. *Towards a Thematic Strategy for Soil Protection*; European Commission: Brussels, 2002.
6. W.B.G.U. (German Advisory Council on Global Change). *World in Transition: New Structures for Global Environmental Policy*; Earthscan: London, Berlin, 2001.

Soil Conservation: U.S. Historical Perspective

Dana D. York

Watershed and Landscape Programs Division, Natural Resources Conservation Service, U.S. Department of Agriculture (USDA-NRCS), Washington, District of Columbia, U.S.A.

Abstract

In the United States, Americans generally accept abundant agricultural production and environmental quality as compatible national goals—but it has not always been this way. This attitudinal shift is one facet of the legacy of the Natural Resources Conservation Service (NRCS), a U.S. Department of Agriculture agency established in 1935 as the Soil Conservation Service. This entry explores the agency's creation and the enduring influence of its founding chief, soil scientist Hugh Hammond Bennett. It examines the rise of soil and water conservation districts and their continuing role in helping NRCS to deliver conservation technical and financial assistance to the nation's farmers and ranchers. The achievements of this partnership are reviewed, and the work of Bennett's heirs in addressing modern challenges to conservation agriculture is discussed, underscoring the validity of his advocacy for an ongoing program of soil and water conservation on working lands.

INTRODUCTION

As the U.S. Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) celebrates its 75th anniversary in 2010, it is useful to reflect on the agency's origins, its legacy in ensuring both productive agricultural lands and a healthy environment, and its ongoing role in helping farmers and ranchers to meet their stewardship goals and address modern conservation challenges, including climate change.

HUGH HAMMOND BENNETT AND THE SOIL CONSERVATION SERVICE

The NRCS, a technical agency of the USDA, celebrates its 75th anniversary in 2010. Although the United States began a national program of soil surveying as early as 1899, it did not formalize its Soil Conservation Service (SCS)—the forerunner of today's NRCS—until the mid-1930s. Its establishment was by then long overdue.

Government policies designed to encourage settlement in the near west and poor land management decisions taken by individual farmers had resulted in severe degradation of the American plains, characterized by terrifying dust storms blowing from Colorado's Front Range to the Atlantic Coast. Drought and wind erosion threatened the nation's agriculture-based economy and citizens' quality of life. Faced with these perils, the distinguished soil scientist Hugh Hammond Bennett advocated “retracing our steps

across the land in an effort to correct past mistakes in the interest of the future.”^[1]

A true visionary, Bennett had been laying groundwork for the agency well in advance of its creation. As a federal manager, he lobbied to stand up soil erosion experiment stations and used the results to develop demonstration projects, including comprehensive conservation plans for participating farms. Some of those original plans are in use, in Wisconsin's Coon Creek Watershed and elsewhere (Fig. 1).

But Bennett knew more had to be done, and on a grander scale. He set off on a nationwide speaking tour and went before the U.S. Congress to ask for additional funding and personnel. Initially, his success was limited. Following several dust storms in the spring of 1935 that so blackened the sky it looked like midnight at midday across the plains, Bennett tried again. This time, with detritus from the storms visible from congressmen's windows, his pleas fell on more receptive ears.^[2] Bennett emerged victorious from the hearings. President Franklin D. Roosevelt signed the Soil Conservation Act into law on April 27, 1935, and Bennett became the first chief of the aptly named SCS.

LOCALLY LED CONSERVATION

Two years later, the first local soil conservation district was formed in North Carolina, which included Bennett's family farm. Its founding was a watershed moment, so



Fig. 1 Chief Hugh H. Bennett, Mrs. Bennett, and regional conservator A.E. McClymonds view conservation work on the Frank Milsna farm, Manske Ridge on the Coon Creek Watershed Demonstration Project in Vernon County, Wisconsin, October 25, 1946.

Source: Photo courtesy of USDA NRCS.

to speak, in the history of soil conservation in the United States, as landowners and the government began to share responsibility for conservation on private lands. Government field offices were also set up and staffed with expertise appropriate to the needs of the county being served.

Bennett was convinced of the soundness of this approach: “In this democracy,” he wrote, “national action to conserve soil must be led by these millions of land users. If they are active and willing participants in such a movement, it will endure; otherwise it will fail.”^[1] He also thought participation in agency programs should be entirely voluntary, which it has been to this day.

He was undoubtedly right about the importance of bringing society and soil science together. There are now nearly 3000 conservation districts nationwide that coordinate federal, state, local, and private assistance to enable landowners to put conservation on the ground. As a result of this insightful construct, delivery of conservation programs is connected from policymakers and funding sources in Washington, DC, all the way to individual farmers and ranchers far from the nation’s capital. Moreover, programs are tailored to meet the specific needs of agricultural producers as well as the community at large, ultimately providing benefits to all citizens.

Seventy-one percent of land in the continental United States is in private hands. That equates to nearly 1.4 billion acres (0.57 billion ha).^[3] Thus, the marriage of local leadership and national objectives has been critically important in effecting lasting change for the environment, whether on a single acre or across a landscape. It has also afforded flexibility in program implementation over the years, as science and technology have advanced, funding has ebbed and flowed, and particular regions or resource concerns have become greater or lesser priorities.

FROM SCS TO NRCS

In 1994, the SCS was renamed the NRCS. The work that began in 1935 emphasizing soil health had by that time grown into a broader family of authorities, programs, and strategies, supporting high-quality, productive soils, clean and abundant water, clean air, and healthy plant and animal communities.

The agency, its partners, and cooperating landowners have made the most of NRCS’s expanding role in working lands conservation. Between 1936 and 2009, these entities have acted in concert to install 15 million miles (24 million km) of terraces; put 129 million acres (52.2 million ha) into contour farming and 425 million acres (172.1 million ha) into conservation cropping systems; preserve, restore, or enhance 9 million acres (3.64 million ha) of wetlands; establish or protect 160 million acres (64.8 million ha) of wildlife habitat; and plant 20 million acres (8.1 million ha) of trees (Fig. 2).^[3]

In just a few years, cooperators have installed over 10,000 manure management facilities and preserved 600,000 acres (242,915 ha) through farm and ranch land easement programs.

In the quarter century between 1982 and 2007, conservation programs have also contributed to a 43% decline in soil erosion on cropland nationwide.^[3]

Since the mid-20th century, NRCS and its survey partners have mapped 2.1 billion acres (0.85 billion ha) of soils (Fig. 3). In 2010, the agency conducts soil surveys and other assessments on the World Wide Web, provides online “energy estimator” management tools, and offers both its plants database and field office technical guides electronically, to name a few e-initiatives.

Maps and tables for more than 2300 soil surveys can also be accessed for free on the Internet (<http://www.nrcs.usda.gov/programs/soilsurvey/>), as can the results of



Fig. 2 NRCS helped landowners install terraces, conservation tillage, and conservation buffers to save soil and improve water quality on this farm in northwest Iowa.

Source: Photo courtesy of Lynn Betts, USDA Natural Resources Conservation Service.



Fig. 3 NRCS employees digitizing soils for soil survey information. Digitized layers are used in geographical information systems (GIS).

Source: Photo courtesy of Bob Nichols, USDA Natural Resources Conservation Service.

National Resources Inventories that report on land use and natural resource conditions and trends on U.S. non-federal lands (<http://www.nrcs.usda.gov/technical/NRI/>).

TECHNICAL AND FINANCIAL ASSISTANCE

Thanks to these and other accomplishments, Americans now generally accept abundant agricultural production and environmental quality as compatible national goals—a significant cultural change from Bennett's day.

For the past few years, symbiosis has been expressed through legislation and appropriations informally referred to as “farm bills.” This legislation expires every 5 to 7 years. The latest version, the “Food, Conservation, and Energy Act of 2008,” runs through 2012 and authorizes the largest ever federal investment in natural resources conservation, totaling \$24 billion over 5 years.^[4] For the most part, the funding and authorities resulting from farm bills have supported abundant production while also increasing the capacity of NRCS to get conservation on the ground.

It is worth noting, however, that conservation technical assistance—what the agency views as “the engine of conservation”—is authorized and funded outside farm bill legislation. While there can be no doubt that farm bill financial incentives for farmers and ranchers facilitate adoption of conservation practices, it is technical assistance that makes them feasible and effective.^[5]

SOIL SCIENTISTS: WALKING IN BENNETT'S FOOTSTEPS

Despite changes over the years, the agency's 21st-century soil scientists continue to walk in Bennett's footsteps. They have routinely been leaders in leveraging technology to serve conservation and are front and center helping agriculture confront the challenges of climate change.

The Rapid Assessment of U.S. Soil Carbon for Climate Change and Conservation Planning, now in its initial stages, is a great example. This effort is designed to gather data on soil carbon stocks for the nation, as influenced by soil, agricultural management, land use, and ecosystem characteristics. These data will be available for input into simulation models for erosion prediction. Furthermore, they will be used as a component of conservation planning activities to enhance soil quality, soil carbon sequestration, and ecosystem services.

HELPING PEOPLE HELP THE LAND

Throughout his life, Bennett continued to champion the importance of soil conservation along with the agency he created and shaped. Soil conservation, he wrote in 1943:

is not just an incidental bit of the mechanics of farming; it becomes part and parcel of the whole business of making a living from the land, and is the only way by which we may have permanently productive land for a permanent agriculture to support a permanent nation.^[6]

His legacy is reflected in the vision statement of NRCS, which calls for conservation actions to promote productive lands and a healthy environment, and in its mission of “helping people help the land.”

CONCLUSION

Given the demands of a growing world population with respect to food and other resources, Bennett's message about the importance of conservation is as relevant today as it was in 1935—and just as vital to ensuring sustainable, permanent agriculture for a permanent nation.

ACKNOWLEDGMENTS

The author would like to recognize the contributions of NRCS National Historian Douglas Helms to the agency's—and her own—understanding of, and respect for, its unique place in American agricultural and conservation history. She would also like to acknowledge the assistance of Mary Elder, NRCS Public Affairs.

REFERENCES

1. Bennett, H.H. *Soil Conservation*; McGraw-Hill Book Co., Inc.: New York, 1939; 313–337.
2. Helms, J.D. *Hugh Hammond Bennett and the Creation of the Soil Conservation Service*; U.S. Department of Agriculture Natural Resources Conservation Service: Washington. Available at http://www.nrcs.usda.gov/about/history/articles/hugh_hammond_bennett_and_the_creation_of_the_soil_conservation_service.html (accessed April 2010).
3. U.S. Department of Agriculture Natural Resources Conservation Service. Summary Report: 2007 National Resources Inventory. Available at http://www.nrcs.usda.gov/technical/NRI/2007/2007_NRI_Summary.pdf (accessed April 2010).
4. Congressional Research Service. The 2008 Farm Bill: Major Provisions and Legislative Action, Summary. Available at http://assets.opencrs.com/rpts/RL34696_20081003.pdf (accessed April 2010).
5. Helms, J.D. *Technical Assistance—the Engine of Conservation*; U.S. Department of Agriculture Natural Resources Conservation Service. Available at http://www.nrcs.usda.gov/about/history/articles/CTA_17Mar_Draft3.pdf (accessed April 2010).
6. U.S. Department of Agriculture Natural Resources Conservation Service. NRCS 75th Anniversary Conservation Facts and Figures. Available at <http://www.nrcs.usda.gov/about/NRCS%20Historical%20Facts%20Figures.pdf> (accessed April 2010).

Soil Degradation: Global Assessment

Selim Kapur

Departments of Soil Science, Landscape Architecture, and Agricultural Engineering, University of Cukurova, Adana, Turkey

Erhan Akça

Center of Remote Sensing, Adiyaman University, Adiyaman, Turkey

Abstract

Although soil provides food, fiber, and nutrients that are essential for humans, its value is rarely esteemed. Soils throughout human history were continuously mistreated, which led to the fall of civilizations. Agricultural and industrialization activities trigger erosion, salinity, and contamination, all of which cause a loss of soil quality to irrecoverable levels. Contemporary soil resource exhaustion has resulted in severe land degradation in many parts of the world; more than 25% of usable global land is degraded. Thus, combatting land degradation is humanity's major priority for global security.

INTRODUCTION

One of the major challenges that humanity will face in the forthcoming decades is the need to increase food production to cope with the ever-increasing population, which is indeed the greatest threat for food/soil security.^[1] The worldwide loss of 12 million hectares (ha) of agricultural land per year was reflected in the average annual loss up to 38 billion for EU25^[2] in spite of the strict laws for environmental protection enacted there. This is the unpalatable but important indicator of a steady decline in agricultural production and an increase in soil damage, reflecting the “impaired productivity” in a quarter of the world's agricultural land.^[3–5]

BACKGROUND

Soil can degrade without actually eroding. It can lose its nutrients and soil biota and can become damaged by waterlogging and compaction. Erosion is only the most visible part of degradation, where the forces of gravity, water flow, or wind actively remove soil particles.

Rather than taking the classical view that soil degradation was, is, and will remain an ongoing process, mainly found in countries of the developing world, this phenomenon should be seen as a worldwide process that occurs at different scales and different time frames in different regions. The causes of biophysical and chemical soil degradations are enhanced by socioeconomic interventions, which are the main anthropogenic components of this problem, together with agricultural mismanagement, overgrazing, deforestation, overexploitation, and pollution as reiterated by Lal,^[6] UNEP/ISRIC,^[7] Lal,^[8] Eswaran &

Reich,^[9] Eswaran et al.,^[10] Kapur et al.,^[11] and Cangir et al.^[12] as the main reasons for erosion and chemical soil degradation.

Soil degradation, the threat to “soil security,” is ubiquitous across the globe in its various forms and at varying magnitudes, depending on the specific demands of people and the inexorably increasing pressures on land. Europe provides many telling examples of the fragile nature of soil security and the destructive consequences of a wide range of soil degradation processes. Asia, Africa, South America, and North America are not only partly affected by the non-resilient impacts of soil degradation but also experiencing more subtle destruction of soils through political developments, which seek to provide temporary relief and welfare in response to the demands of local populations.

Europe

The major problems concerning the soils of Europe are the loss of such resources owing to erosion, sealing, flooding, large mass movements as well as local and diffuse soil contamination, especially in industrial and urban areas, and soil acidification.^[13,14] Salinity is a minor problem in some parts of Western and Eastern Europe but with severe effects at the northern and western parts of the Caspian Sea (with low salinity)^[15] mainly because of the shift to irrigated agriculture and destruction of the natural vegetation (Fig. 1).

The urbanization and construction of infrastructure at the expense of fertile land are widespread in Europe, particularly in the Benelux countries, France, Germany, and Switzerland, and such effects are most conspicuously destructive along the misused coasts of Spain, France, Italy,

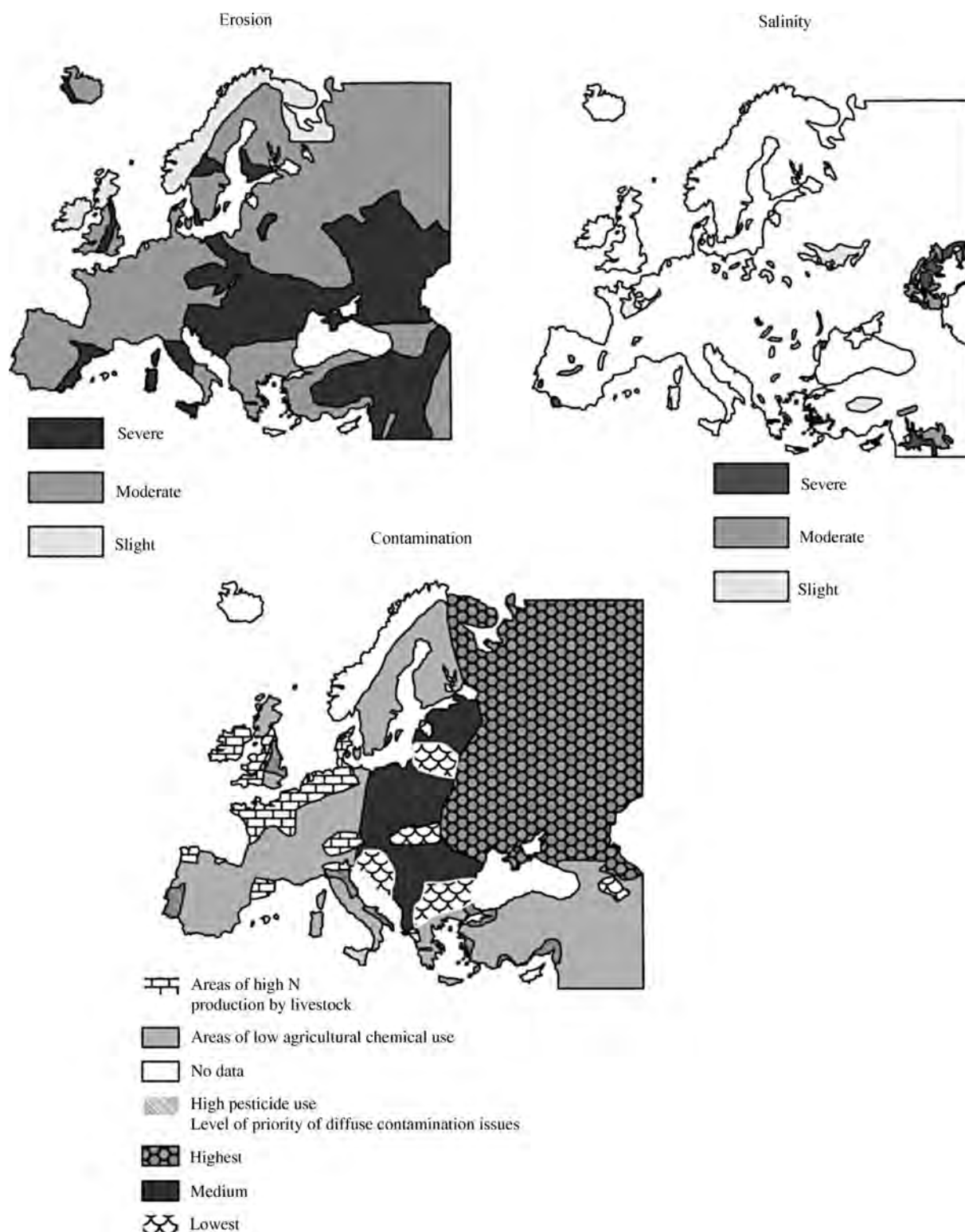


Fig. 1 Soil degradation in Europe.

Source: Adapted from UNEP/ISRIC,^[7] Kobza,^[16] USDA-NRCS,^[17,18] Erol,^[19] Kosmas, Ferrara, et al.,^[20] and Kharin, Tateishi, et al.^[21]

Greece, Turkey, Croatia, and Albania. The drastic increase in the rate of urbanization since the 1980s is expected to follow the Blue Plan, which seeks to create beneficial relationships among populations, natural resources, the main

elements of the environment, and the major sectors of development in the Mediterranean Basin and to work for sustainable development in the Mediterranean region. The very appropriate term “industrial desertification” remains

valid for the once degraded soils of East Europe, under the pressure of mining and heavy industry, as in Ukraine where such lands occupy 3% of the total land area of the country.^[22]

There are three broad zones of “natural” erosion across Europe, including Iceland, as follows: 1) the southern zone (the Mediterranean countries); 2) a northern Loess zone comprising the Baltic States and part of Russia; and 3) the eastern zone of Slovenia, Croatia, Bosnia and Herzegovina, Romania, Bulgaria, Poland, Hungary, Slovakia, the Czech Republic, and Ukraine (Fig. 1). Seasonal rainfalls are responsible for severe erosion owing to overgrazing and the shift from traditional crops. Erosion in Southern Europe is an ancient problem and continues in many places, with marked on-site impacts and with significant decreases in soil productivity as a result of soil thinning. The northern zone of high-quality Loess soils displays moderate effects of erosion with less intense precipitation on saturated soils. Local wind erosion on light textured soils is also responsible for the transportation of agricultural chemicals used in the intensive farming systems of the northern zone to adjacent water bodies, along with eroded sediments. The high erodibility of the soils of the eastern zone is exacerbated by the presence of large state-controlled farms that have introduced intensive agriculture at the expense of a decrease in the natural vegetation. Contaminated sediments are also present in this zone, particularly in the vicinity of former industrial operations/deserts, with high rates of erosion in Ukraine (33% of the total land area) and Russia (57% of the total land area), whose agricultural land has been subjected to strong water and wind erosion ever since the beginning of industrialization.

Localized zones of likely soil contamination through the activities of heavy industry are common in North-western and Central Europe as well as Northern Italy, together with more scattered areas of known and likely soil contamination caused by the intensive use of agricultural chemicals. Sources of contamination are especially abundant in the “hot spots” associated with urban areas and industrial enclaves in the northwestern, southern, and central parts of the continent (Fig. 1). Acidification through the deposition of wind-borne industrial effluents and aerosols has been a long-standing problem for the whole of Europe; however, this is not expected to increase much further, especially in Western Europe, as a result of the successful implementation of emission-control policies.^[3]

The desertification of parts of Europe has been evident for some decades, and the parameters of the problem are becoming clear, with existing emphasis on monitoring of the environmentally sensitive areas^[22,20] on selected sites, seeking quality indicators for 1) soil (particularly organic carbon); 2) vegetation; 3) climate; and 4) human management throughout the Mediterranean basin. Apart from the human factor, these indicators are inherent.

Asia

The most severe aspect of soil degradation on Asian lands has been desertification owing to the historical, climatic, and topographic character of this region as well as the political and population pressures created by the conflicts of the past 500 years or so. Salinization caused by the rapid drop in the level of the Aral Sea and the waterlogging of rangelands in Central Asia owing to the destruction of the vegetation cover by overgrazing and cultivation provide the most striking examples of an extreme version of degradation—desertification caused by misuse of land. Soil salinity, the second colossal threat to the Asian environment, has occurred through the accumulation of soluble salts, mainly deposited from saline irrigation water or through mismanagement of available water resources, as in the drying Aral Sea and the Turan lowlands as well as the deterioration of the oases in Turkmenistan, with excessive abstraction of water in Central Asia (Fig. 2).^[21]

The dry lands of the Middle East have been degrading, since the Sumerian epoch, with excessive irrigation causing severe salinity and erosion-siltation problems,^[21] especially in Iraq, Syria, and Saudi Arabia. Iraq has been unique in the magnitude of the historically recorded build up of salinity levels, with 4.81 million ha saline land, which is 74% of the total arable land surface (i.e., 90% of the land in the southern part of that country). The historical lands of Iran, Pakistan, Afghanistan, India, and China are also subject to ancient and ongoing soil degradation processes, which are subtle in some areas but evident and drastic in others (Fig. 2).

Africa

Africa's primary past and present concern has been the loss of soil security by nutrient depletion, i.e., the decreasing NPK levels (kg ha^{-1}) along with micronutrients in cultivated soils following the exponential growth in population and the resulting starvation and migrations (Fig. 3). The intensification of land use to meet the increased food demands combined with the mismanagement of the land leads to the degradation of the continental soils. This poses the ultimate question of how the appropriate sustainable technologies that will permit increased productivity of soils can be identified. This problem is illustrated by the example of the Sudan, where nutrient depletion has steadily increased through more mechanized land preparation, planting, and threshing without the use of inorganic fertilizers, and legume rotations. Thus, aggregate yields have been falling as it became more difficult to expand the cultivated area without substantial public investments in infrastructure. This decline in yield has occurred at enormous rates in Vertisols of Sudan, with the mean annual sorghum yields decreasing from 1000 to 500 kg ha^{-1} and the wholesale price of sorghum increased 3.7% per month since 2007.^[24] In Burkina Faso, the decreased infiltration and

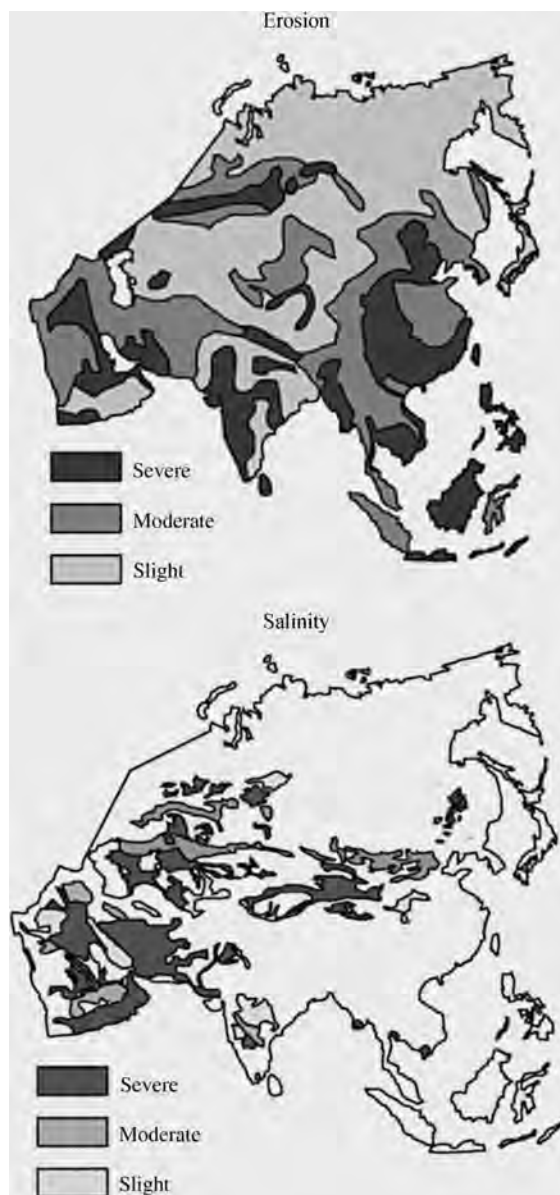


Fig. 2 Soil degradation in Asia.

Source: Adapted from UNEP/ISRIC,^[7] Erol,^[19] Kosmas, Ferrara, et al.,^[20] and Lowdermilk.^[23]

increased runoff causing erosion are further consequences of repeated cultivation. Thus, the technological measures to be identified for these two African examples must include development of water retention technologies in Burkina Faso, while polyculture/rotations with proper manuring and fertilization for cost-efficient provisions of nitrogen and phosphorus and preferably green manuring are all needed in Sudan to permit the balanced management of soil moisture, nutrients, and organic matter [and to enhance carbon (C) sequestration, a main goal for sustainability based on the earth sciences—to ensure the security of both the soil and global climate].^[24,25]

Central and South America

Africa and Latin America have the highest proportion of degraded agricultural land. Water and wind erosion are the dominant soil degradation processes in Central America and South America and have caused the loss of the topsoil at alarming rates because of the prevailing climatic and topographic conditions. Almost as important is the loss of nutrients from the Amazon basin (Fig. 4).^[26] These effects are mainly attributable to deforestation and overgrazing, the former being responsible for the degradation of 576 million ha out of 1000 million ha potential agricultural land. Another important factor has been the ever-increasing introduction of inappropriate agricultural practices derived from the so-called imported technology, which have not been properly adapted to indigenous land use procedures. The traditional methods of permitting the land to recover naturally have been almost totally abandoned and replaced by unsuitable technological measures designed to maintain production levels (temporarily) and to overcome the loss of soil resilience, thus increasing chemical inputs.

The rapid industrialization/urbanization of the limited land resources in the Caribbean region has been expelling agricultural communities to remote and marginal regions that are at present rich in biodiversity and biomass—a major global C sink. Moreover, large-scale livestock herding of Central America and South America is also a major threat to soil security and has been responsible for degrading 1 million km² of Argentinean, Bolivian, and Paraguayan pasturelands.

North America

The most prominent outcome of soil degradation (or more correctly desertification) in the United States is exemplified by the accelerated dust storm episodes of the 1930s—the Dust Bowl years, marked by the “Black Blizzards,” which were caused by persistent strong winds, droughts, and overuse of the soils. These resulted in the destruction of large tracts of farmland in the South and Central United States. Salinization has become an equally severe problem in the western part of the country (Fig. 5) through the artificial elevation of water tables by extensive irrigation, with associated acute drainage problems. An area of about 10 million ha in the west of United States has been suffering from salinity-related reductions in yields, coupled with very high costs in both the Colorado River basin and the San Joaquin Valley.^[27] Unfortunately, new irrigation technologies, such as the center pivot irrigation system (developed as an alternative to the conventional irrigation systems causing the salinity problems), have caused a decline of the water-table levels in areas north of Lubbock, Texas, by around 30–50 m, leading to a dramatic decrease in the thickness of the well-known Ogallala Aquifer by 11% decrease between 2003 and 2011 only.

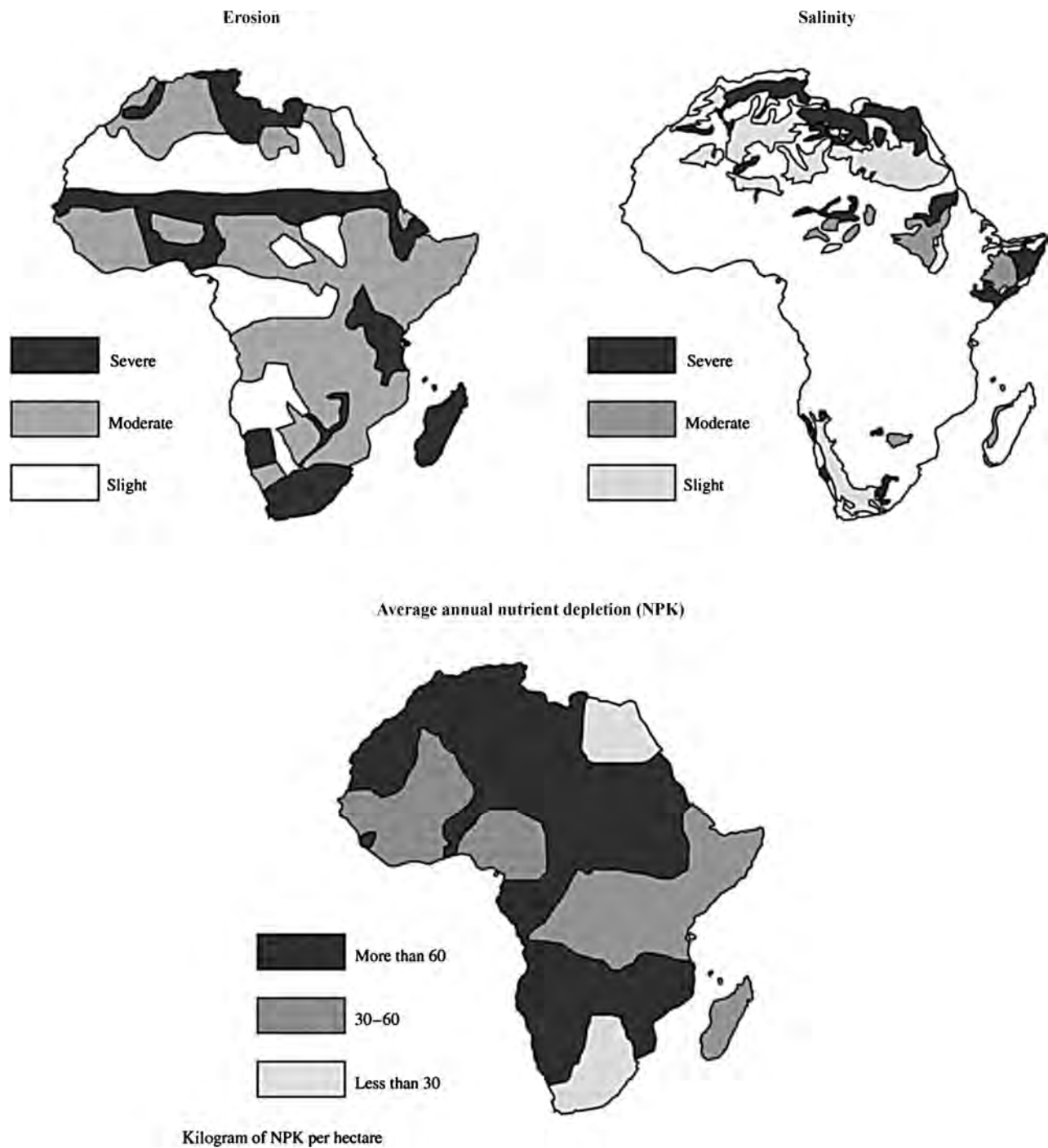


Fig. 3 Soil degradation in Africa.
Source: Adapted from UNEP/ISRIC^[7] and Lowdermilk.^[23]

In some areas, this has been followed by ground subsidence, which is an extreme form of soil structure degradation, i.e., loss of the physical integrity of the soil.

Loss of topsoil, as a result of more than 200 years of intensive farming in the United States, is estimated to vary between 25% and 75% and exceeds the upper limit in some parts of the country.^[18,19] The United States provides good examples of the difficulties involved in erosion control,

with its large-scale intensive agriculture—deteriorating soil structure and increasing erosion of its susceptible soils. This problem could be overcome primarily by the strict introduction of the no-till system. No-till areas have increased from 4 million ha in 1989 to 25.3 million ha in 2004, and they are forecast to follow a linear extrapolation until 48 million ha out of the total 81 million ha of cultivated land will be attained.^[28] Conservation farming is

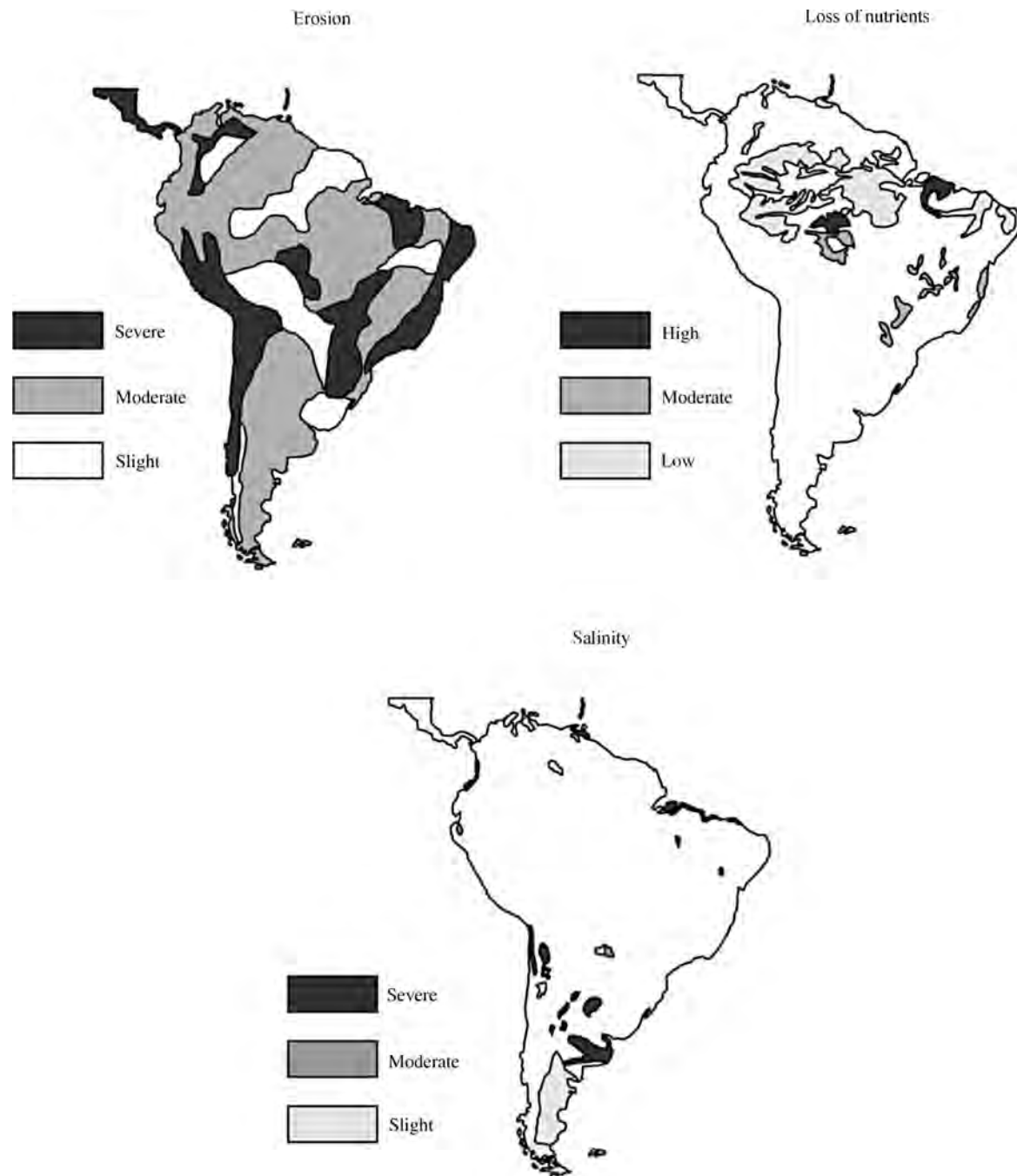


Fig. 4 Soil degradation in Central America and South America.
Source: Adapted from UNEP/ISRIC^[7] and Kharin, Tateishi, et al.^[21]

practiced in only about half of all U.S. agricultural land and on less than half of the country's most erodible cropland. Conservation farmers are encouraged to use only the basic types of organic fertilizers, such as animal and green manure together with compost, mulch farming, improved pasture management, and crop rotation, to conserve soil nutrients.

Canada is a large country where 68 million ha of available land is cultivated, which is only the 7.4% of the

territorial of the country with an average farm size of 450 ha. It is reported that Canada has experienced annual soil losses on the prairies, through wind and water erosions, which are similar to the Asian steppes, amounting, respectively, to 60 and 117 million tons. These annual rates are much higher than the rate of soil formation, resulting in an annual potential grain production loss of 4.6 million tons of wheat. With regard to primary soil salinity, during historic times, the prairies have experienced steady increases

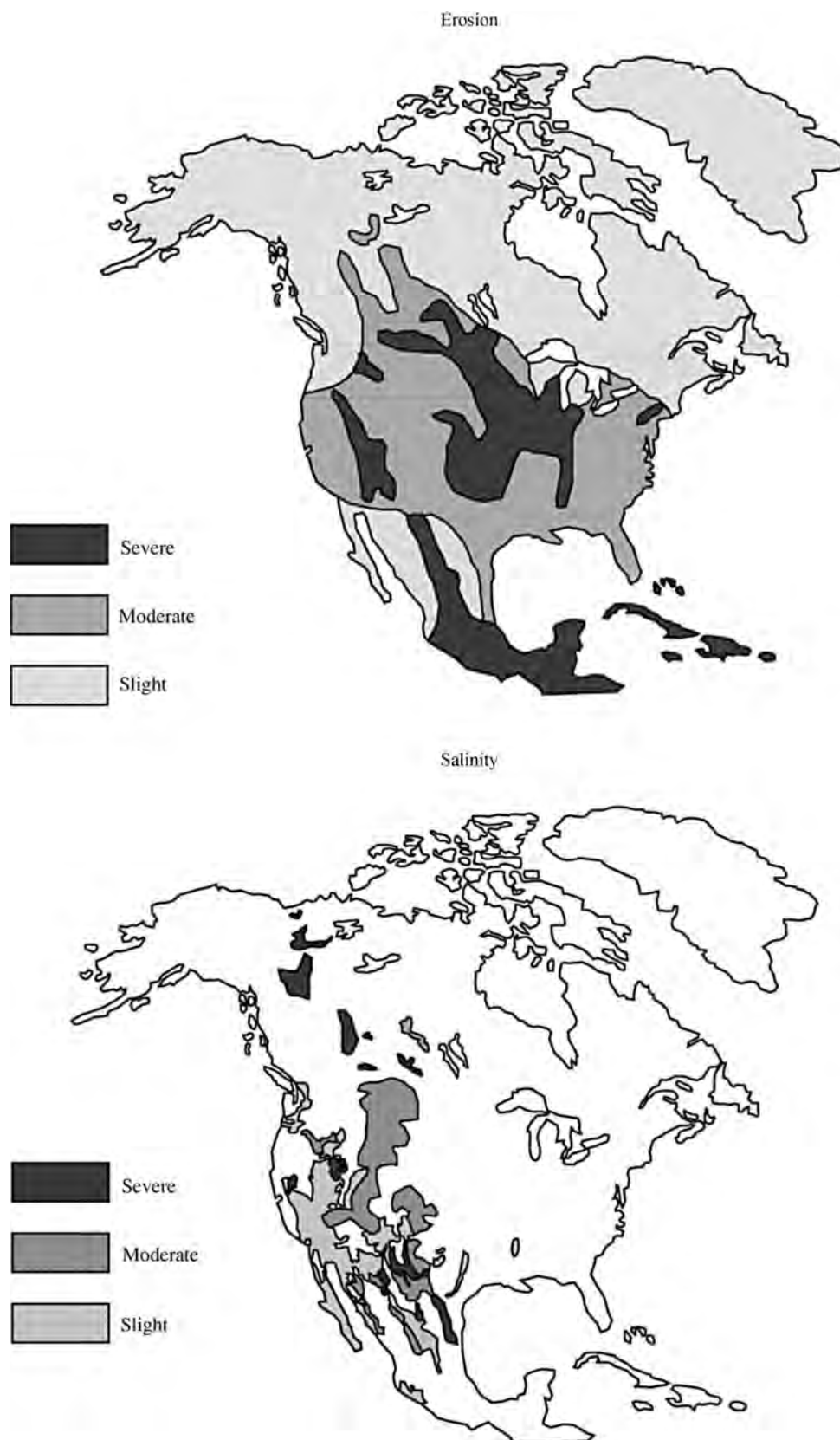


Fig. 5 Soil degradation in North America.

Source: Adapted from UNEP/ISRIC^[7] and Kharin, Tateishi, et al.^[21]

related partially to increasing groundwater levels. Major problems of secondary salinity are estimated to affect 2.2 million ha of land in Alberta, Saskatchewan, and parts of Manitoba, with an immense economic impact each year.

Australia

The Australian agricultural/soil resource base has been endangered, as the “business-as-usual” concept was adopted on the continent to achieve temporary economic betterment. The identification of different types of soil degradation in Australia reveals that erosion has been the main component, primarily via dust storms, which are a serious problem, especially where cropping practices do not include the retention of cover and minimum tillage methods. Water erosion effects are also particularly severe in areas of summer rainfall and topographic extremities (Fig. 6). Although water is a scarce resource in Australia, about 14 million tons of soil is lost in Australia by water erosion. Remedial actions for this include the well-known measures of maintaining adequate cover and changing prevailing attitudes toward stock management, storage feed, redesign of watering sites, and management of riparian areas.

Part of the excess salinity in Australia is of primary origin and was retained in the subsoil by trees, which have been cleared to create soil surfaces for cropping and pastures, allowing the penetration of water to the saline subsoil, then followed by abstraction from the water table, thus leading to the ultimate disaster. About 30% of Australia’s agricultural land is sodic, creating poor physical conditions and impeded productivity. This problem can only be alleviated by massive revegetation programs and by taking extra care of the water table and plant cover. Despite the introduction of costly conventional measures for reclamation, salinity levels continue to increase across Australia in the dry and irrigated soils. The dryland salinity in the continent affects about 5 million ha of farmland and is expanding at a rate of 3–5% per year.^[29]

The retardation of organic matter levels also requires remediation measures, with economically justified fertilizer use strategies to be utilized throughout the continent. Moreover, overgrazing has resulted in the impoverishment of plant communities and loss of habitats as well as the decline in the chemical fertility of the soil by progressive depletion of organic matter in the topsoil, followed by deterioration in the soil structure.

Acidification caused by legume-based mixed farming plus use of ammonia-based fertilizers threatens 55 million ha of Australian land. Liming seems to be the most effective remedy, but is costly, does not lead to rapid recovery, and is impractical for subsoil acidity. Thus, the precise remedies are yet to be developed for the conditions on this continent, utilizing careful, long-term monitoring and the experience of farmers to devise specific treatment and conservation procedures.

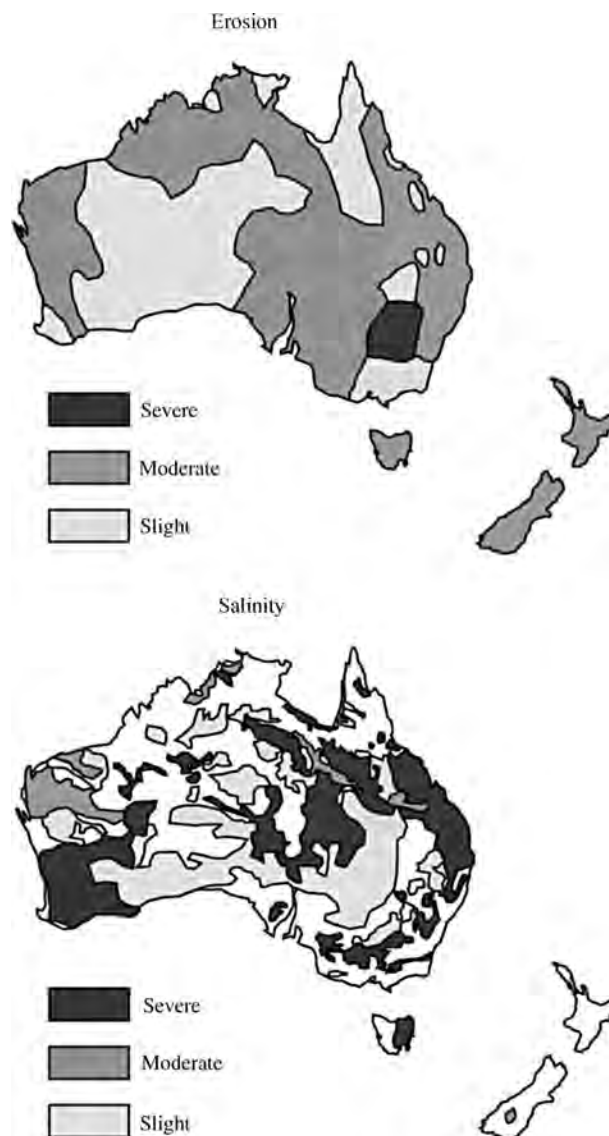


Fig. 6 Soil degradation in Australia.

Source: Adapted from UNEP/ISRIC^[7] and Kharin, Tateishi, et al.^[21]

CONCLUSION

The state of soil degradation and its remedies as a multi-function–multi-impact approach have been identified through a Driving Force–Pressure–State–Impact–Response matrix by the European Environmental Agency^[2] (Fig. 7), leading to sustainable land management (SLM)^[30–32] measures to be taken for the future. SLM is concerned with more soil-friendly farming practices that minimize the erosion potential of soils, together with the adoption by landholders of property management planning procedures that involve community actions undertaken within several concerted action projects of the European Union, Food and Agriculture

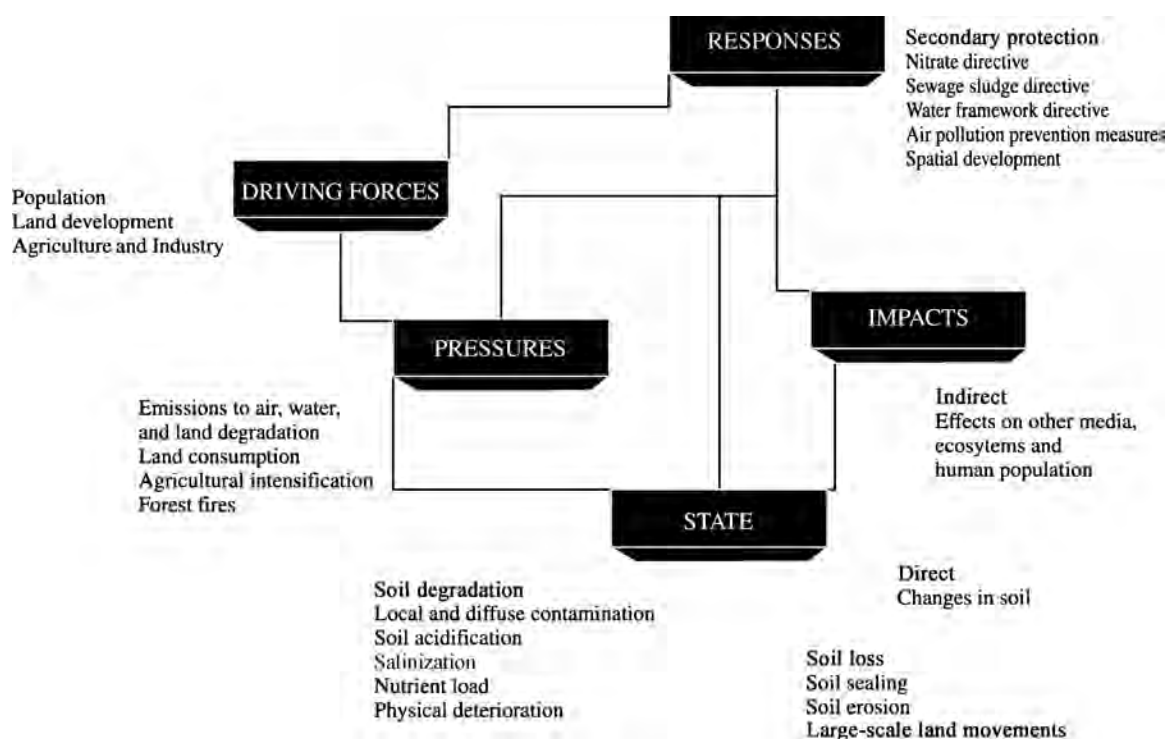


Fig. 7 Driving Force–Pressure–State–Impact–Response framework applied to soil.

Source: Adapted from World Bank.^[30]

Organization, Global Environment Facility, United Nations Environment Programme, International Crops Research Institute for the Semi-Arid Tropics, Canadian International Development Agency, and World Bank.^[33] Moreover, as Smyth and Dumanski^[34] have stated, these combine socioeconomic principles with environmental concerns, so that the production is enhanced, together with the reduction of its level of risk with the protection of natural resources, which would prevent the degradation of soil and water quality to be successfully accepted by the farmer. The methods to be adopted for SLM via community actions include contour farming, terracing, vegetative barriers, and other land use practices amalgamated with indigenous (traditional) technical knowledge (ITK)^[32] as applied to farming and landscape preservation. The impetus to the use of ITK by scientists and local communities in creating new strategies for sustainable resource management was initiated in the United Nations Conference on Environment and Development held in Rio de Janeiro (Brazil) in 1992, and the need for raising awareness and combatting land degradation is man's priority for global social security.

REFERENCES

1. International Food Policy Research Institute. *The Economics of Desertification, Land Degradation, and Drought*. Toward an Integrated Global Assessment, IFPRI Discussion Paper 01086 May, Bonn, 2011; 188 pp.
2. Toth, G.; Montanarella, L.; Rusco, E. *Threats to Soil Quality in Europe*; Institute for Environment and Sustainability, Land Management and Natural Hazards Unit EUR 23438 EN: Milan, 2008; 1–162.
3. Amman, M.; Bertok, I.; Cabala, R.; Cofala, J.; Heyes, C.; Gyarmas, F.; Klimont, Z.; Schöpp, W.; Wagner, F. *A Further Emission Control Scenario for the Clean Air for Europe (CAFE) Programme*; International Institute for Applied Systems Analysis: Laxenburg, Austria, 2005.
4. Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Oxon, 1994; 1–115.
5. Steiner, K.G., Ed. *Causes of Soil Degradation and Development Approaches to Sustainable Soil Management*; GTZ. Weikersheim. Margraf Verlag: Filderstadt, 1996; 1–50.
6. Lal, R. Forest soils and carbon sequestration. *Forest Ecol. Manag.* **2005**, *220*, 242–258.
7. UNEP/ISRIC. *GLASOD World Map of the Status of Human-Induced Soil Degradation, 1:10 M. 3 Sheets*, 2nd Rev. Ed.; ISRIC: Wageningen, 1991.
8. Lal, R. *Methods and Guidelines for Assessing Sustainable Use of Soil and Water Resources in the Tropics*. N.21, SMSS Technical Monograph; U.S. Agency for International Development: Washington, 1994; 1–79.
9. Eswaran, H.; Reich, P. Desertification: A global assessment and risks to sustainability. In *Proceedings of the 16th Int. Cong. Soil Science*, CD-ROM; Montpellier: France, 1998.

10. Eswaran, H.; Beinroth, F.H.; Reich, P. Global land resources and population supporting capacity. *Am. J. Alternat. Agr.* **1999**, *14*, 129–136.
11. Kapur, S.; Atalay, P.İ.; Ernst, F.; Akça, E.; Yetiş, C.; İşler, F.; Öcal, A.D.; Üzel, İ.; Şafak, Ü. A review of the late quaternary history of Anatolia. In *1st International Symposium on Cilician Archaeology*; Durugönül, S., Ed.; Mersin University: Turkey, 1999; 253–272.
12. Cangir, C.; Kapur, S.; Boyraz, D.; Akça, E.; Eswaran, H. Land resource consumption in Turkey. *J. Soil Water Conserv.* **2000**, *3*, 253–259.
13. Millennium Ecosystem Assessment. *Ecosystems and Human Well-Being: Desertification Synthesis*; World Resources Institute: Washington, 2005.
14. Oldeman, L.R.; Hakkeling, R.T.A.; Sombroek, W.G. *World Map of the Status of Human-Induced Soil Degradation: An Explanatory Note*; International Soil Reference Center: Wageningen, 1992; 1–34.
15. Karpinsky, M.G. Aspects of the Caspian Sea benthic ecosystem. *Mar. Pollut. Bull.* **1992**, *24*, 3849–3862.
16. Kobza, J. Monitoring and contamination of soils in Slovakia in relation to their degradation. In *Proceedings of the 1st International Conference on Land Degradation*; Kapur, S., Akça, E., Eswaran, H., Kelling, G., Vita-Finzi, C., Mermut, A.R., Öcal, A.D., Eds.; Adana: Turkey, 1996; 10–14.
17. USDA-NRCS. *Risk of Human Induced Desertification, NRSC*; Soil Survey Division, World Soil Resources, 2003.
18. USDA-NRCS. *Global Desertification Vulnerability, NRSC*; Soil Survey Division, World Soil Resources, 2003.
19. Erol, O. Misuse of the Mediterranean coastal area in Turkey. In *Proceedings of the 1st International Conference on Land Degradation*; Kapur, S., Akça, E., Eswaran, H., Kelling, G., Vita-Finzi, C., Mermut, A.R., Öcal, A.D., Eds.; Adana: Turkey, 1999; 195–203.
20. Kosmas, C.; Ferrara, A.; Briassoulis, H.; Imeson, A. Methodology for mapping environmentally sensitive areas (ESAs) to desertification. In *The MEDALUS Project. Mediterranean Desertification and Land Use Report*; Kosmas, C., Kirkby, M., Geeson, N., Eds.; European Commission: Belgium, 1999; Vol. 199, 31–47.
21. Kharin, N.G.; Tateishi, R.; Harahsheh, H. *Degradation of the Drylands of Asia*; Center for Environmental Remote Sensing, Chiba University: Japan, 1999; 1–81.
22. Mnatsakanian, R. *Environmental Legacy of the Former Soviet Republics*; Center for Human Ecology, University of Edinburgh: Edinburgh, 1992.
23. Lowdermilk, W.C. *Conquest of the Land through Seven Thousand Years. No. 99*; USDA, Soil Conservation Service, U.S. Government Printing Office: Washington, 1997; 1–30.
24. Sanders, J.H.; Southgate, D.D.; Lee, J.G. *The Economics of Soil Degradation: Technological Change and Policy Alternatives, No. 22*; SMSS Technical Monograph; Department of Agricultural Economics, Purdue University: West Lafayette, 1995; 8–11.
25. Croitoru, L.; Sarraf, M. *The Cost of Environmental Degradation; Case Studies from the Middle East and North Africa*; The World Bank Washington: Washington, 2010.
26. Lal, R. Monitoring soil erosion's impact on crop productivity. In *Soil Erosion Research Methods*; Lal, R., Ed.; Soil and Water Conservation Society: Ankeny, 1988; 187–200.
27. Barrow, C.J. *Land Degradation*; Cambridge University Press: Cambridge, 1994.
28. Derpsch, R.; Friedrich, T.; Kassam, A.; Hongwen, L. Current status of adoption of no-till farming in the world and some of its main benefits. *Int. J. Agric. Biol. Eng.* **2010**, *3*, 1–25.
29. State of the Environment 2011 Committee. *Australia State of the Environment 2011*; Independent Report to the Australian Government Minister for Sustainability, Environment, Water, Population and Communities; DSEWPac: Canberra, 2011.
30. World Bank. *Sustainable Land Management. Challenges, Opportunities, and Trade-Offs*; World Bank: Washington, 2006; 112 pp.
31. Eswaran, H.; Beinroth, F.; Reich, P. Biophysical considerations in developing resource management domains. In *Proceedings of the IBSRAM International Workshop on Resource Management Domains, Kuala Lumpur, Aug 26–29, 1996*; Leslie, R.N., Ed.; The International Board for Soil Research and Management (IBSRAM): Thailand, 1996; 61–78.
32. Eswaran, H.; Berberoglu, S.; Cangir, C.; Boyraz, D.; Zucca, C.; Özevren, E.; Yazıcı, E.; Zdruli, P.; Dingil, M.; Dönmez, C.; Akça, E.; Çelik, I.; Watanabe, T.; Koca, Y.K.; Montanarella, L.; Cherlet, M.; Kapur, S. The anthroscape approach in sustainable land use. In *Sustainable Land Management: Learning from the Past for the Future*; Kapur, S., Eswaran, H., Blum, W.E.H., Eds.; Springer: Heidelberg, 2011; 1–50.
33. Zdruli, P.; Steduto, P.; Kapur, S.; Akça, E., Eds. *Ecosystem-based assessment of soil degradation to facilitate land users' and land owners' prompt actions*. In *Medcoastland Project Workshop Proceedings*, Adana, Turkey, Jun 2–7, 2003, 2006.
34. Smyth, A.J.; Dumanski, J. A frame work for evaluating sustainable land management. *Can. J. Soil Sci.* **1995**, *75* (4), 401–406.

BIBLIOGRAPHY

1. Manfred, K.; Christian, P.; Markus, M. Strategic environmental assessment. Environmental report. In *Central Europe Programme 2007*; Austrian Institute of Ecology: Wien, 2013; 44 pp.

Soil Enzymes

Cherukumalli Srinivasa Rao

Minakshi Grover

Sumanta Kundu

Susheelendra Desai

Central Research Institute for Dryland Agriculture, Indian Council of Agricultural Research (ICAR), Hyderabad, India

Abstract

Soil enzymes play a key role in the energy transfer through decomposition of soil organic matter and nutrient cycling, and hence play an important role in agriculture. These enzymes catalyze many vital reactions necessary for the life processes of soil microorganisms and also help in stabilization of soil structure. Although microorganisms are the primary source of soil enzymes, plants and animals also contribute to the soil enzyme pool. Soil enzymes respond rapidly to any changes in soil management practices and environmental conditions. Their activities are closely related to physio-chemical and biological properties of the soil. Hence, soil enzymes are used as sensors for soil microbial status, for soil physio-chemical conditions, and for the influence of soil treatments or climatic factors on soil fertility. Understanding the possible roles of different soil enzymes in maintaining soil health can help in the soil health and fertility management, particularly in agricultural ecosystems. In this entry, we describe the major soil enzymes and their role in nutrient cycling, soil fertility, and yield sustainability. The effects of cropping systems, crop management, and nutrient management practices on soil enzyme activities are discussed. We further discuss the reports on the effect of climatic factors like elevated carbon dioxide, temperature, and drought on soil enzymes activities. Advancements in enzyme activity assay tools are also discussed, followed by prospects in the field of soil enzymes. To conclude, soil enzymatic activity is a useful tool for assessing and managing productivity of an ecosystem.

INTRODUCTION

Soil enzymes are the key players in biochemical process of organic matter recycling in the soil system and their activities are closely related to soil organic matter (SOM), soil physical properties, and microbial activity and/or biomass. Depending on their location, enzymes can be extracellular or intracellular. Intracellular enzymes are found in cell's cytoplasm or bound to the cell walls of living and metabolically active cells, viable but non-proliferating cells (such as spores) and dead cells. Extracellular enzymes released into the soil and are "permanently" immobilized on clay and humic colloids via ionic interactions, covalent bonds, hydrogen bonding, entrapment, and other mechanisms.

Soil enzymes are necessary catalysts for decomposition of SOM and nutrient cycling and, strongly influence energy transformation, environmental quality, and agro-nomic productivity. However, mechanical tillage, monoculture, and residues removal adversely impact enzymatic activity and availability of plant nutrients. In general, enzymatic activity decreases with an increase in soil depth. Further, soil enzymes provide early detection of changes in soil health because they respond to soil management changes and environmental factors much

sooner than other soil quality parameters. Moreover, availability of well-documented assays for a large number of soil enzyme activities makes them the preferred tool for assessing soil health. However, it is necessary to understand the relationship between different enzyme pools and biotic and abiotic factors to predict the potential impact of soil management and environmental changes on ecosystem functions and productivity.

MAJOR SOIL ENZYMES AND THEIR FUNCTIONS

Major soil enzymes used as soil function indicators are presented in [Table 1](#). The activities of these soil enzymes can be used for a meaningful assessment of reaction rates for important soil processes, soil productivity, microbial activity, and inhibiting effects of pollutants, etc.^[1]

EFFECT OF AGRO-TECHNIQUES AND CROPPING SYSTEMS ON SOIL ENZYMES

Soil enzymes, being necessary catalysts for organic matter recycling, strongly influence soil fertility and agronomic

Table 1 Major soil enzymes and their functions.

Enzyme	Source	Reaction catalyzed	End product	Soil function indicated	Factors influencing enzyme activity
α -Amylase	Plants, animals, and microorganisms	Starch hydrolysis	Glucose and/or oligosaccharides	C-cycling	Management practices, type of vegetation, environment, and soil types.
β -Amylase	Mainly plants	Starch hydrolysis	Maltose		
Dehydrogenase	Microorganisms	Oxidation of organic compounds	Transfer of H to NAD or NADP (electron transport system)	C-cycling, microbial oxidative activity	Soil water content, temperature, pesticides, trace elements, management practices, pollution, etc.
Endo-1, 4- β -glucanase	Microorganisms, protozoa, and termites	Cellulose endohydrolysis	Oligosaccharides	C-cycling	Temperature, pH, water, O ₂ contents, quality and location of organic matter, mineral elements, and fungicides.
Exo-1, 4- β -glucanase		Cellulose cleavage at ends	Glucose and cellobiose		
β -glucosidase		Cellobiose hydrolysis	Glucose (sugar)		
Phenol oxidase	Plants and microorganisms	Lignin hydrolysis	C compounds (humic substances)	C-cycling	Soil pH, mean annual precipitation and temperature, SOM content, management practices, N enrichment, etc.
Urease	Microorganisms, plants, and some invertebrates	Urea hydrolysis	Ammonia (NH ₃) and CO ₂	N-cycling	Cropping history, organic matter content, soil depth, management practices, heavy metals, temperature, pH, etc.
Alkaline phosphatase	Mainly bacteria	Hydrolysis of esters and anhydrides of phosphoric acid	Phosphate (PO ₄)	P-cycling	Organic matter content, pH, management practices, pollution, crop species, and varieties.
Acid phosphatase	Plants, fungi, and bacteria				
Arylsulfatase	Microorganisms, plants, and animals	Hydrolysis of sulfate esters	Sulfate (SO ₄ ⁻²)	S-cycling	Heavy metal pollution, pH, organic matter content and composition, and availability of organic sulfate esters
Protease	Microorganisms and plants	N mineralization	Plant available N	N-cycling	Humic acid concentration, availability of C and N, etc.
Chitinase	Plants and microorganisms	Degradation and hydrolysis of chitin	Carbohydrates and inorganic nitrogen	C- and N-cycling	Availability of N, soil depth, atmospheric CO ₂ levels, etc.

NAD: nicotinamide adenine dinucleotide, NADP: nicotinamide adenine dinucleotide phosphate.

productivity. Qualitative and quantitative changes in soil enzymes determine the availability of nutrients and crop productivity. Different agricultural practices like tillage, cropping systems, and nutrient management influence the soil enzyme activities, thereby influencing yield sustainability.^[2]

Adverse impacts of mechanical tillage, cropping systems, and residues removal have been observed in soil enzymatic activities and availability of plant nutrients.^[3] Management-induced changes in soil moisture, temperature, and soil organic carbon (SOC) input influence

microbial biomass carbon (MBC), nutrient availability, and SOC turnover.^[4] Plowing disrupts macroaggregates and accentuates mineralization of the labile SOC pool whereas stable aggregation protects SOC and influences SOM turnover and soil fertility.^[5] There is a close relationship between SOC concentration and soil enzyme activities, which is influenced within particular horizons by other factors (i.e., pH, texture, and gleying). As the enzymatic activity decreases with increase in soil depth, management-induced differences are observed more in the surface than in the subsoil.^[6] Thus enzyme activities can be

used to evaluate the degree of alteration of soils in both natural and agroecosystems.

Aon, Cabello et al.^[7] showed that certain enzymatic activities (acid and alkaline phosphatases, dehydrogenase, fluorescein diacetate (FDA) hydrolysis, β -glucosidase, and urease) and groups of bacteria and fungi exhibit strong links independent of season and crop, despite fluctuations shown by the different microbial and biochemical activities along with oxygen (O₂) and carbon dioxide (CO₂) fluxes as a function of time and space. The spatiotemporal pattern of microbial and biochemical activities exhibited an identifiable path: fungal biodiversity increased, microorganisms stratified as a function of depth unlike all enzymatic activities that showed the maximal stratification.

Fertilization exerts a strong influence on soil quality. Organic amendments such as farmyard manure (FYM), plant residues, and compost are known to improve soil physical and chemical properties, increase SOM, urease activity (UA) and acid phosphatase activity, and enhance soil quality. Amending with organic matter and application of balanced fertilizers improve soil biological properties including microbial biomass and enzymatic activities.^[5,8–10]

Optimum and balanced application of plant nutrients significantly increases the dehydrogenase activity (DHA) that is indicative of the oxidative activity of soil microflora, and hence microbial activity. However, DHA may not be a reliable index of microbial activity for soils receiving high nitrogen (N) inputs, as it is adversely affected by N fertilization. Treatments receiving high rate of N either as chemical fertilizers alone or in combination with organics and the N fertilized wheat (*Triticum aestivum*) cropping system have shown high UA.^[9]

Effects on DHA also differ with source of nutrients applied. For instance, the activity is more with organic N sources as compared to mineral N.^[6] Basu, Mahapatra, and Bhadoria^[11] reported that integrated application of organic or industrial wastes, soil ameliorants, and chemical fertilizer could improve the biological properties of an acid lateritic soil and dry matter production of groundnut, intercropped with sabai grass (*Eulaliopsis binata*). Similarly, application of lime or rice husk ash has been reported to improve DHA and alkaline phosphomonoesterase enzymes and decrease acid phosphomonoesterase activity.^[11]

Cropping systems have a definitive bearing on soil enzymatic activity. In general, soils under legume-based systems contain relatively high MBC than those under other systems. Similarly, integrated use of chemical fertilizers and FYM for 3–7 yr increased MBC and microbial activity under jute (*Corchorus capsularis*)–rice–wheat system in some tropical agricultural soils.^[12] In a study conducted in tropical region of Colombia, Vallejo, Roldan, and Dick^[13] observed that silvi pastoral system stimulated soil MBC and enzymatic (β -glucosidase, UA, and alkaline and acid phosphatase) activities and provided more favorable microbial habitat as compared to monoculture pasture.

Monoculture practices cause decrease in soil enzyme activities when compared with legume-based rotations. Reducing the duration of fallow in a fallow–wheat rotation enhanced enzyme activities and accentuated cycling of C and phosphorus (P).^[14] Soil biological properties and DHA are enhanced by management systems, which add biomass C through legume-based rotations, agroforestry systems, or conservation tillage. It is precisely because of its rapid response to management that the DHA is considered a good biological indicator of soil quality.^[9]

The maintenance of SOC is the main factor that favors the microbial activity and the enzyme activity that correlates with the soil microbial biomass. Bonanomi et al.^[15] reported a positive correlation between enzymatic activities and total SOC content. Forest cleaning, cropping, and management practices affect the soil microbial activity and hence the enzyme activity, a fact attributed to less C inputs. Similarly, the intensively managed soils exhibit decrease in microbial activity in comparison to well-managed pastures.^[16] In high-input management regime soils, a drastic reduction in microbial biomass and fungal mycelium has been observed. The positive correlation between organic C content and both microbial biomass and fungal mycelium suggested that a reduction in the available organic C induced a decrease in the decomposer biomass.^[15] Besides, several other factors having influence on soil enzyme activity are summarized in Table 2.

SOIL ENZYMES AND YIELD SUSTAINABILITY

Many studies have reported significant correlation between soil enzymatic activity and plant yield, whereas some studies show no close relationship. For example, mean grain yields of different crops across rainfed production systems were positively correlated with DHA, arylsulfatase activity (ASA), and UA in soil.^[10] Grain yields of groundnut–finger millet, finger millet mono cropping, winter sorghum, soybean, and upland rice were significantly and positively correlated with DHA, ASA, and UA activities whereas groundnut mono cropping and pearl millet with three enzymes were not.^[18] The probable reason was that the latter two cropping systems were practiced under semiarid conditions with relatively low rainfall (~500 mm annually) and high mean annual temperatures as compared with the other five systems. The sustainable yield index was also positively correlated with all the three enzymes in all production systems except in case of soybean (no correlation with ASA and DHA). However, under irrigated maize–wheat–cowpea system, Manjaiah and Singh^[19] reported significant correlation between yields and all the soil enzymes studied. Similarly, using linear regression models, Lopes, de Sousa et al.^[27] interpreted the enzymatic activities (cellulase, β -glucosidase, acid phosphatase, and ASA) as a function of the relative cumulative yields of corn and soybean. The plant productivity is under the influence of many

Table 2 Response of soil enzyme activity under different ecosystems/practices.

Ecosystem/Practice	Enzymes of C-cycling	Phosphatase	Dehydrogenase	Enzymes N-cycling	Total organic carbon	References
Forest vs. pasture vs. agricultural	Highest (β -glucosidase) in forest, lowest in agricultural soil	Highest (alkaline phosphatase) in forest, lowest in agricultural soil	—	Highest (urease) in pasture lowest in agricultural soil	Highest in forest	Kizilkaya and Dengiz ^[20]
Conservational vs. conventional tillage	High (β -glucosidase)	High	High	High urease and protease	High	Roldan et al. ^[21]
Organic residue with RDF (maize residue in rice and wheat cultivation)	High (invertase)	High alkaline phosphatase	High	High urease and protease	—	Tao et al. ^[17]
Organic vs. unamanded (in bell pepper)	High (β -glucosidase)	High acid phosphatase	High	High urease	High	Gopinath et al. ^[22]
Rehabilitated (stabilized soils (over a 50 yr period) on moving sand dunes	High (α - and β -glucosidase)	High	High	High protease	High	Zhang et al. ^[23]
Degraded vs. native vegetation	Low (cellulose)	—	Low	—	Low	Araújo et al. ^[24]
Mined (coal mine soil vs. forest soil)	—	—	Low	—	Low	Kumar et al. ^[25]
Polluted	—	Low acid and alkaline phosphatase	—	—	—	Ohiri et al. ^[26]

factors like soil quality, plant genotype, biotic and abiotic stresses, etc. Further, in managed ecosystems, many other factors (external inputs) may influence the relationship between enzyme activity and plant productivity.

MEASUREMENT OF SOIL ENZYMES ACTIVITIES

Enzymatic activities have been measured by detecting degradation of the target substrates and the generation of the products. A variety of methods depending on the mode of detection (spectrophotometry, fluorescence, and radiolabeling), the reaction substrates, and conditions (temperature, use of buffers, time of reaction, etc.) have been used for measuring the enzyme activities in soils. The enzymatic assay can be direct where substrate is added to the soil system and the product formed is quantified or indirect where enzyme is extracted from the soil and assayed afterward. Extractability of the soil enzymes is a drawback in an indirect method as a significant portion of the soil enzymes is bound to soil components and is not extracted completely. Assays based on substrate-induced respiration have been developed to measure enzyme activity. In this method, a substrate is added to soil, and production of CO₂ or consumption of O₂ is measured. Simultaneous measurement of substrate-induced respiration of numerous substrates has been made possible by multiwell plate system.

Advances in molecular biology has opened up opportunities to develop genomic, transcriptomic, and proteomic approaches to estimate the potential of a complex microbial community to produce a specific enzyme, expression of enzyme-coding genes, and presence of a specific enzyme in an environment.^[28,29] The abundance of enzyme-encoding genes or transcribed [messenger RNA (mRNA)] sequences has been used for assessing enzyme activity. However, there are many challenges associated with detection and quantification of enzyme-coding genes. For instance, many of the genes are not well conserved and many enzymes have alternate forms with the same function. Use of multiple primers or degenerate primers can help to detect the different forms of a gene. The detection of abundance of an enzyme-coding gene indicates only the potential function and not the expression level of the gene in an environment. The expression of a gene can be indicated by the presence of an mRNA transcript that correlates more closely with enzyme production rates than gene copies. However, the persistence and turnover of the enzyme have to be considered while interpreting the enzyme activity by gene expression.^[29] With the development of GeoChip, thousands of functional genes can be targeted together, thereby covering a significant portion of enzyme-catalyzed processes. Besides, quantitative polymerase chain reaction of gene sequences or mRNA transcripts can be used for quantification of enzyme-encoding genes. However, it is

an indirect method of assessing enzyme activity, and as the genes encoding inactive molecules are also included, accurate assessment of enzyme activity is difficult. *In situ* zymography technique for 2-D distribution of enzyme activities in soil has been applied to map and quantify protease and amylase activity in the rhizosphere of lupine (*Lupinus polyphyllus*).^[30] In this method, thin gels with embedded substrates are incubated with the enzyme and the substrate remaining in the gel after incubation is quantified using calibration curves and digital image analysis. Thus, with some considerations, different assay methods can provide reliable information on the soil enzymes profile.

SOIL ENZYMES AND CLIMATE CHANGE

Increasing atmospheric concentrations of CO₂ and other greenhouse gases is causing global warming and altered precipitation patterns. The activities of soil enzymes are influenced by abiotic (e.g., temperature, water potential, and pH) and biotic (e.g., enzyme synthesis and secretion) factors. Climate change-induced extreme weather events are impacting agriculture and hence can affect quality and quantity of soil enzymatic activity. These changes will have important consequences for ecosystem functions such as decomposition, nutrient cycling, and plant-microbe interactions, which will ultimately affect productivity and net C balance. The impacts of climate change on microbes and their extracellular enzymes, although of profound importance, are not well understood, but may be predicted, assessed, and managed.^[29,31]

Growing concern about the potential consequences of climate change on soil processes, coupled with a desire to develop methods of improving C sequestration, has stimulated experimental research, modeling, and theorizing.^[29] Microbial enzyme activities are responsible for the synthesis as well as decomposition of SOM, though the rate of synthesis and decomposition processes determines net C balance in the soil.

Elevated atmospheric concentrations of CO₂ may affect soil microbial communities directly and indirectly. Increased plant rhizodeposition, water use efficiency, and accelerated nutrient uptake under elevated atmospheric CO₂ strongly affect microbial activities. Increased activities of oxidative enzymes (degrade resistant SOM) and enzymes involved in N and P mineralization (chitinases, peptidases, and phosphatases) have been observed in response to elevated CO₂, whereas no response or decreased activity observed for C-degrading enzymes indicating the influence of elevated CO₂ on microbial responses associated with C-, N-, and P-cycling.^[29,32] Thus, the labile substrate additions through increased rhizodeposition can stimulate the decomposition of more resistant SOM due to extracellular enzymatic activity. However, the effects may vary in different ecosystems. For example, in peatlands where substrate is not a limiting

factor, low O₂ availability and low pH restrict activities of oxidative enzymes (phenol oxidase and peroxidase) lead to accumulation of phenols that inhibit hydrolytic activity and microbial substrate utilization. This, in turn, contributes to organic matter accumulation. On the other hand, in the substrate limited arid systems, photodegradation of surface litter reduces soil input, low redox potentials of phenols due to alkaline pH, and active oxidative enzymes due to arid conditions lead to increased decomposition of recalcitrant C compounds.^[29]

Available scanty literature suggests that warming effects could be either positive or negative depending on the nature and kinetics of the target enzyme being assayed. It has been observed that the temperature sensitivity of extracellular enzymes changes seasonally. A study in Mediterranean indicated that soil UA and β -glucosidase activity were positively correlated with soil temperatures in winter and negatively in summer. Warming increased soil enzyme activities in winter (when soil moisture was highest) and in spring (coinciding with the greatest biological activity).^[33] Steinweg, Dukes et al.^[31] observed difference among seasons and treatments in mass-specific enzyme activity indicating that enzyme production was not directly controlled by size and activity of microbial biomass. Mass-specific enzyme activity increased with increase in temperature from low to medium warming and declined at higher temperature, suggesting that enzyme production increased with temperature or turnover rate decreased. In general, enzyme activity increases with temperature (up to some optimum) and so at least theoretically the rate of enzymatically catalyzed reactions will increase due to warming, assuming that enzyme pool sizes remain constant.^[29] On the other hand, warming may also increase enzyme denaturation rates and the activity of extracellular proteases which would counteract any changes in enzyme pool sizes. In a study by Gong, Zhang et al.,^[34] warming enhanced phosphatase activity (35.8%) but inhibited the cellulase activity (30%) in grassland ecosystem. In addition, warming caused reduction in soil C (7.2%) and available P (20.5%). Further, significant interactive effects of warming and N addition on soil enzyme activity indicated that global change may alter nutrient cycling by influencing soil enzyme activity.

Changing seasonal precipitation patterns may increase drying/wetting events in the soil. The diffusion of enzymes, substrates, and reaction products in the soil depend on soil texture and moisture. Under low moisture conditions, *in situ* enzyme activities are low although in some microsites where solute concentration increases within pore spaces may exhibit high activity. Prolonged droughts are likely to decrease enzyme production resulting in lower measured activities. However, slower enzyme turnover in dry soils, along with continuous production (even at low rates) could lead to increase in pool size during a drought.^[29,31] Rewetting leads to increased availability of organic matter (due to cell lysis and disruption of soil aggregates) which may

result in a pulse of microbial biomass turnover thus causing a temporary increase in extracellular enzyme activities. On the other hand, decreased microbial biomass could lead to a decrease in enzyme production and a decline in the relative abundance of different types of enzymes. Whereas under prolonged precipitation, enhanced plant growth and rhizodeposition result in increased enzymatic activities.

The net effect on enzyme activity depends on how both enzyme production and turnover are affected by changes in climatic conditions. The complexity of interactions between different climatic factors and soil properties makes it difficult to pinpoint the effect of a single abiotic factor on a particular soil enzyme.^[31] As soil enzymes play important role in nutrient cycling, all the climate change studies must consider soil enzymes activities as important parameter while quantifying the outcomes of climate change.

SOIL FAUNA IN RELATION TO SOIL ENZYME ACTIVITY

The role of soil fauna as indicators of soil health is gaining importance. This group comprises the invertebrate community that plays roles in supplying pretransformed organic material to the microorganisms after fragmentation, resulting from their feeding process. Besides increasing the contact surface, the fauna helps in bioturbation of litter and also contributes to soil enzymes. The most representative organisms normally studied as indicators of soil health belong to the mesofauna, which lives in soil macropores and spaces in the soil–litter interface, feeding on fungal hyphae and organic matter, and thus taking part in nutrient cycling and soil aggregation. The macrofauna includes bigger soil organisms which sometimes are active in soil functioning. Increasing number of studies on effects of soil fauna on soil health parameters have been reported with some studies emphasizing on the influence of soil fauna on soil enzymes.^[16]

An increase in invertase, protease (during rice and wheat cultivation), and alkaline phosphatase (during rice cultivation) activity was observed in the presence of earthworms when maize was used as residue whereas no change in DHA observed in the presence of earthworms. Additionally, the five enzyme activities in earthworm casts were significantly higher than those in the surrounding soil, especially DHA and invertase activity. This result suggests that the increased degradation of maize residues by earthworms enhanced substrate concentrations in soil resulting in high microbial activity.^[17] The functional role of soil fauna also varies depending on feeding habit and the physico-chemical properties of plant litter. Mukhopadhyay, Roy, and Joy^[35] compared the feeding impact of *Anoplo-desmus saussurei* (Humbert) feeding on semiliquid portions and *Porcellio laevis* (Latraeille) feeding by scrapping soft tissues, on litter breakdown and soil enzyme activities on decomposing leaf litter of two forest tree species, *Cassia siamea* (litter high in soluble carbohydrates,

cellulose, and hemicelluloses) and *Shorea robusta* (litter high in polyphenols, tannin, and lignin) in microcosms. Feeding by *A. saussurei* and *P. laevis* caused significant weight loss and decline of the main chemical constituents in *C. siamea* litter, whereas the weight loss was much less in *S. robusta*. Soil enzyme activities were influenced by litter quality, with *C. siamea* litter exhibiting higher amylase, cellulase, and invertase activities than *S. robusta*. Soil enzyme activities were also influenced by the presence of detritivores. Amylase activity increased in the presence of both arthropods in *C. siamea* and *S. robusta* litters. Cellulase activity was enhanced only by *A. saussurei* in both litters, whereas both detritivore species contributed to higher invertase activity in *S. robusta* litter. Further the positive impact of *A. saussurei* was greater than that of *P. laevis* for all the enzymes in *S. robusta* litter. The results indicated a direct and species-specific comminuting effect of detritivore arthropods on soil enzyme activities.

Species-specific fungal enzymatic responses have also been recorded in soil.^[36] Lignocellulolytic enzyme production by saprotrophic basidiomycetes colonizing leaf litter increased during macrofauna and collembola activity. This may be attributed to litter comminution and to fungal physiological responses to grazing. *Hypholoma fasciculare* and *Phanerochaete velutina* (exhibiting fast and extensive growth) increased production of cellulolytic and phospholytic enzymes during macroinvertebrate grazing, whereas the slow-growing species, *Rhizoctonia bicolor* reduced enzyme production. Contrasting enzymatic responses of fungal species suggest the impacts of soil fauna on fungal-mediated nutrient mineralization.

However, a meta-analysis of litter box experiments^[37] showed that soil fauna significantly increased litter removal from the litter layer but did not significantly affect overall C mineralization. The rate of leaf litter decomposition is significantly faster than decomposition of macrofauna feces produced from the same litter. This suggests that larger litter mass loss from soil surface caused by soil fauna may be decoupled from overall litter mineralization. Fauna effect of litter decomposition and mineralization is likely to be affected by climatic condition and litter quality but also by diversity and functional complexity of soil food webs as well as the time for which soil gets exposed to soil fauna. Several studies suggested that long-term fauna effect is more likely to promote C sequestration than mineralization. However, more studies are needed on the role of soil fauna in organic matter mineralization.

CONCLUSION AND FUTURE PROSPECTS

Soil enzymes is a vast research area with enormous potential to be exploited and manipulated for agricultural benefits. The importance of soil enzymes in soil biogeochemical processes, soil health, and agricultural productivity is well

known. Soil enzymes are under the influence of physical, chemical, and biological properties of soil which in turn are influenced by soil management practices, environmental factors, etc. However, there is very little information on this subject due to lack of well-equipped laboratories as well as trained manpower to research in this area to understand determinants of soil enzymes activities. There is an urgent need to develop tools and techniques to assay enzymatic activities with precision and research methodology to conduct experiments in this complex area. With the advancement in molecular techniques, measurement of soil enzyme activity has become possible up to functional microbial communities, gene, transcriptome, and protein levels. The limitations associated with *in situ* enzyme activity need to be addressed to make the measurements more precise and practically feasible. Temporal and spacial studies on soil enzymes activities are needed to know the factors controlling enzymes activities so that optimal conditions for the soil enzymes activities may be understood. Soil enzyme profiling under different cropping systems and management practices can help in developing correlations between soil enzymes activities and productivity and soil health of the ecosystem. Climate change-induced abiotic and biotic stresses affect soil enzymes activities. However, basic and strategic research has to be conducted across agroecological regions to understand dynamics of soil enzymatic activities and interplay of cropping system and agro-techniques under changing climatic conditions to generate enough knowledge that can be used to manipulate activities and interventions in order to improve specific ecosystem functions. This information will also help to design coping mechanisms to maintain soil enzyme activities under climate change conditions and thus sustain crop productivity.

REFERENCES

1. Nare, R.W.A.; Savadogo, P.W.; Gnankambary, Z.; Nacro, H.B.; Sedogo, P.M. Effect of three pesticides on soil dehydrogenase and fluorescein diacetate activities in vegetable garden in burkina faso. *Curr. Res. J. Biol. Sci.* **2014**, *6* (2), 102–106.
2. Srinivasarao, C.; Lal, R.; Kundu, S.; Prasad Babu, M.B.B.; Venkateswarlu, B.; Singh, A.K. Soil carbon sequestration in rainfed production systems in the semiarid tropics of India. *Sci. Total Environ.* **2014**, *487*, 587–603. DOI: 10.1016/j.scitotenv.2013.10.006.
3. Celika, I.; Barut, Z.B.; Ortasa, I.; Goka, M.; Demirbasa, A.; Tuluna, Y.; Akpinara, C. Impacts of different tillage practices on some soil microbiological properties and crop yield under semi-arid Mediterranean conditions. *Int. J. Plant Prod.* **2011**, *5* (3), 237–254.
4. Srinivasarao, C.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Vittal, K.P.R.; Kundu, S.; Singh, S.R.; Singh, S.P. Long-term effects of soil fertility management on carbon sequestration in a rice-lentil cropping system of the Indo-Gangetic plains. *Soil Sci. Soc. Am. J.* **2012**, *76* (1), 168–178. DOI: 10.2136/sssaj2011.0184.
5. Srinivasarao, C.; Venkateswarlu, B.; Lal, R.; Singh, A.K.; Kundu, S. Sustainable management of soils of dryland ecosystems of India for enhancing agronomic productivity and sequestering carbon. In *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press: Burlington, 2013; Vol. 121, 253–329.
6. Srinivasarao, C.; Chary, G.R.; Venkateswarlu, B.; Vittal, K.P.R.; Prasad, J.V.N.S.; Kundu, S.; Singh, S.R.; Gajanan, G.N.; Sharma, R.A.; Deshpande, A.N.; Patel, J.J.; Balaguravaiah, G. *Carbon Sequestration Strategies Under Rainfed production Systems of India*; Central Research Institute for Dryland Agriculture (ICAR): Hyderabad, 2009; 102 pp.
7. Aon, M.A.; Cabello, M.N.; Sarena, D.E.; Colaneri, A.C.; Franco, M.G.; Burgos, J.L.; Cortassa, S.I. Spatio-temporal patterns of soil microbial and enzymatic activities in an agricultural soil. *Appl. Soil Ecol.* **2001**, *18* (3), 239–254.
8. Mohammadi, K. Effect of different fertilization methods on soil biological indexes. *World Acad. Sci. Eng. Technol.* **2011**, *78*, 407–410.
9. Pajares, S.; Gallardo, G.F.; Masciandaro, G.; Ceccanti, B.; Etchevers, J.D. Enzyme activity as an indicator of soil quality in degraded cultivated acrisols in the Mexican trans volcanic belt. *Land Degrad. Dev.* **2011**, *22* (3), 373–381.
10. Mandal, A.; Patra, A.K.; Singh, D.; Swarup, A.; Ebhin Masto, R. Effect of long-term application of manure and fertilizer on biological and biochemical activities in soil during crop development stages. *Bioresource Technol.* **2007**, *98*, 3585–3592.
11. Basu, M.; Mahapatra, S.C.; Bhadoria, P.B.S. Influence of soil ameliorants, manures and fertilizers on bacterial populations, enzyme activities, N fixation and P solubilization in peanut rhizosphere under lateritic soil. *Br. Microbiol. Res. J.* **2011**, *1*, 11–25.
12. Chakraborty, A.; Chakrabarti, K.; Chakraborty, A.; Ghosh, S. Effect of long-term fertilizers and manure application on microbial biomass and microbial activity of a tropical agricultural soil. *Biol. Fert. Soils* **2011**, *47*, 227–233.
13. Vallejo, V.E.; Roldan, F.; Dick, R.P. Soil enzymatic activities and microbial biomass in an integrated agroforestry chronosequence compared to monoculture and a native forest of Colombia. *Biol. Fert. Soils* **2010**, *46*, 577–587.
14. Acosta-Martinez, V.; Cruz, L.; Sotomayor-Ramírez, D.; Pérez-Alegria, L. Enzyme activities as affected by soil properties and land use in a tropical watershed. *Appl. Soil Ecol.* **2007**, *35*, 35–45.
15. Bonanomi, G.; D'Ascoli, R.; Antignani, V.; Capodilupo, M.; Cozzolino, L.; Marzaioli, R.; Puopolo, G.; Rutigliano, F.A.; Scelza, R.; Scotti, R.; Rao, M.A.; Zoina, A. Assessing soil quality under intensive cultivation and tree orchards in Southern Italy. *Appl. Soil Ecol.* **2011**, *47*, 184–194.
16. Cardoso, E.J.B.N.; Vasconcellos, R.L.F.; Bini, D.; Miyauchi, M.Y.H.; dos Santos, C.A.; Alves, P.R.L.; de Paula, A.M.; Nakatani, A.S.; de Moraes Pereira, J.; Nogueira, M.A. Soil health: Looking for suitable indicators. What should be considered to assess the effects of use and management on soil health? *Sci. Agric.* **2013**, *70*, 274–289. DOI: 10.1590/S0103-90162013000400009.
17. Tao, J.; Griffiths, B.; Zhang, S.; Chen, X.; Liu, M.; Hu, F.; Li, H. Effects of earthworms on soil enzyme activity in an

- organic residue amended rice–wheat rotation agro-ecosystem. *Appl. Soil Ecol.* **2009**, *42*, 221–226.
18. Saha, S.; Prakash, V.; Kundu, S.; Kumar, N.; Mina, B.L. Soil enzymatic activity as affected by long term application of farm yard manure and mineral fertilizer under a rainfed soybean–wheat system in N-W Himalaya. *Eur. J. Soil Biol.* **2008**, *44*, 309–315.
 19. Manjaiah, K.M.; Singh, D. Soil organic matter and biological properties after 26 years of maize wheat–cowpea cropping as affected by manure and fertilization in a Cambisol in semiarid region of India. *Agr. Ecosyst. Environ.* **2001**, *86*, 155–162.
 20. Kizilkaya, R.; Dengiz, O. Variation of land use and land cover effects on some soil physic-chemical characteristics and soil enzyme activity. *Zemdirbyste.* **2010**, *97*, 15–24.
 21. Roldan, A.; Salinas-García, J.R.; Alguacil, M.M.; Díaz, E.; Caravaca, F. Soil enzyme activities suggest advantages of conservation tillage practices in sorghum cultivation under subtropical conditions. *Geoderma* **2005**, *129*, 178–185.
 22. Gopinath, K.A.; Saha, S.; Mina, B.L.; Pande, H.; Srivastava, A.K.; Gupta, H.S. Bell pepper yield and soil properties during conversion from conventional to organic production in Indian Himalayas. *Sci. Hortic.* **2009**, *122*, 339–345.
 23. Zhang, Y.L.; Chen, L.J.; Chen, X.H.; Tan, M.L.; Duan, Z.H.; Wu, Z.J.; Li, X.L.; Fan, X.H. Response of soil enzyme activity to long-term restoration of desertified land. *Catena* **2015**, *133*, 64–70.
 24. Araujo, A.S.F.; Cezars, S.; Leite, L.F.C.; Borges, C.D.; Tsai, S.M.; Eisenhauer, N. Soil microbial properties and temporal stability in degraded and restored lands of northeast Brazil. *Soil Biol. Biochem.* **2013**, *66*, 175, 181.
 25. Kumar, S.; Chaudhuri, S.; Maiti, S.K. Soil dehydrogenase enzyme activity in natural and mine soil—A review. *Middle-East J. Sci. Res.* **2013**, *13*, 898–906.
 26. Ohiri, R.C.; Agha, N.C.; Nwachukwu, N. Variations in Phosphatase activity of crude oil and used crankase oil polluted agricultural soil. *J. Biol. Agric. Healthcare* **2013**, *3*, 143–149.
 27. Lopes, A.A.C.; de Sousa, D.M.G.; Chaer, G.M.; Junior, F.B.R.; Goedert, W.J.; Mendes, I.C. Interpretation of microbial soil indicators as a function of crop yield and organic carbon. *Soil Sci. Soc. Am. J.* **2013**, *77* (2), 461–472.
 28. Nannipieri, P.; Giagnoni, L.; Renella, G.; Puglisi, E.; Ceccanti, B.; Masciandaro, G.; Fornasier, F.; Moscatelli, M.C.; Marinari, S. Soil enzymology: Classical and molecular approaches. *Biol. Fert. Soils* **2012**, *48*, 743–762.
 29. Burns, R.G.; DeForest, J.L.; Marxsen, J.; Sinsabaugh, R.L.; Stromberger, M.E.; Wallenstein, M.D.; Weintraub, M.N.; Zoppini, A. Soil enzymes in a changing environment: Current knowledge and future directions. *Soil Biol. Biochem.* **2013**, *58*, 216–234.
 30. Spohn, M.; Carminati, A.; Kuzyakov, Y. Soil zymography: A novel *in situ* method for mapping distribution of enzyme activity in soil. *Soil Biol. Biochem.* **2013**, *58*, 275–280.
 31. Steinweg, J.M.; Dukes, J.S.; Paul, E.A.; Wallenstein, M.D. Microbial responses to multifactor climate change: Effects on soil enzymes. *Front. Microbiol.* **2013**, *4*, 146. DOI: 10.3389/fmicb.2013.00146.
 32. Kelley, A.M.; Fay, P.A.; Polley, H.W.; Gill, R.A.; Jackson, R.B. Atmospheric CO₂ and soil extracellular enzyme activity: A meta-analysis and CO₂ gradient experiment. *Ecosphere* **2011**, *2* (8), 1–20. art 96. DOI: 10.1890/ES11-00117.1.
 33. Sardans, J.; Peñuelas, J.; Estiarte, M. Changes in soil enzymes related to C and N cycle and in soil C and N content under prolonged warming and drought in a Mediterranean shrubland. *Appl. Soil Ecol.* **2012**, *39*, 223–235.
 34. Gong, S.; Zhang, T.; Guo, R.; Hongbin, C.; Lianxuan, S.; Guo, J.; Sun, W. Response of soil enzyme activity to warming and nitrogen addition in a meadow steppe. *Soil Res.* **2015**, *53* (3), 242–252.
 35. Mukhopadhyay, S.; Roy, S.N.; Joy, V.C. Enhancement of soil enzyme activities by the feeding impact of detritivore arthropods on tropical forest tree leaf litters. *Trop. Ecol.* **2014**, *55*, 93–108.
 36. Crowther, T.W.; Boddy, L.; Jones, T.H. Functional and ecological consequences of saprotrophic fungus–grazer interactions. *ISME J.* **2012**, *6* (11), 1992–2001.
 37. Frouz, J.; Roubickov, A.; Hedenec, P.; Tajovský, K. Do soil fauna really hasten litter decomposition? A meta-analysis of enclosure studies. *Eur. J. Soil Biol.* **2015**, *68*, 18–24.

Soil Organic Matter (SOM)

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Soil organic matter (SOM) is essential to maintaining soil quality and providing numerous ecosystem services. Its magnitude in soil depends on climate, soil properties, terrain characteristics, vegetation, and landscape position. Its concentration is more in soils of cooler and humid than warmer and dry climates. There are numerous cost-effective and non-destructive methods of determination of SOM. Its magnitude and dynamics can be modeled and related to several index properties. Managing SOM is essential for advancing food security, mitigating climate change, improving water quality, and enhancing biodiversity.

INTRODUCTION

The origin and importance of soil organic matter (SOM) were appropriately summarized in 1973 by Allison:^[1] “Soil organic matter has over the centuries been considered by many as an elixir of life. Ever since the dawn of history, some eight thousand or more years ago, man has appreciated the fact that dark soils, commonly found in the river valleys and broad level plains, are usually productive soils. He also realized at a very early date that color and productivity are commonly associated with organic matter derived chiefly from decaying plant materials.” The SOM comprises the sum of all organic substances in soil and primarily consists of heterogeneous substances of plant, animal, and microbial origin at various stages of decomposition. Its decomposition products include: 1) soluble organic compounds such as sugars, proteins, and other metabolites; 2) amorphous organic compounds such as humic acids, fats, waxes, oils, lignin, and polyuronides; and 3) organomineral complexes, which involve hybrid compounds of organic molecules attached to clay particles through polyvalent cations such as Al^{3+} , Ca^{2+} , and Mg^{2+} .^[2]

Two principal components of SOM are highly active humus and relatively inert charcoal. Humus is a dark brown or black amorphous material. Being a colloidal substance, it is characterized by a large surface area, high charge density, and high affinity for clay. Charcoal is widely present in soils of the fire-dependent ecosystems (i.e., savanna, steppe, and grasslands). The charcoal, also called black carbon (C), is of pyrogenic origin. It comprises lightly charred organic matter, soot particles rich in graphite, and polycondensed aromatic groups. Highly fertile soils of the Amazon, *terra preta do Indios*, are enriched by the addition of charcoal.

SOM DYNAMICS

In natural ecosystems, SOM is in dynamic equilibrium with its environment. Its amount in the soil is stabilized when the

rate of input equals that of the output. The input of SOM is mainly through residues of plants grown *in situ* and from substances brought in by alluvial, aeolian, and colluvial processes. The output of SOM is through decomposition or oxidation, erosion, and leaching. The rate of decomposition depends on temperature and moisture regimes and is more in warmer and wetter than in cooler and drier climates. Soil erosion, by water and wind, preferentially removes SOM because it is lighter (lower density) than mineral fraction and is concentrated in the vicinity of the soil surface where erosional processes are the most effective. The soluble and colloidal components of SOM are leached into the subsoil and eventually into aquatic ecosystems along with the percolating water. Removal of crop residues for alternative uses, such as traditional or modern biofuels (cellulosic ethanol) and livestock feed, reduces input of biomass and decreases the soil organic C pool. Thus, there is no such thing as a free biofuel from crop residues.

Natural factors affecting the SOM concentration include climate, soil type, vegetation, hydrology, landscape position, and soil biodiversity. Principal climatic parameters are precipitation amount and distribution, temperature and its seasonal and diurnal variation, annual and seasonal water budget, and the duration of the growing season as determined by the degree days and hydrologic balance. Predominant soil properties that affect SOM include texture, clay minerals, pH, ionic composition and concentration, and soil depth. Internal drainage of the soil profile, affecting soil moisture regime, is also important. Landscape affects SOM content through the position, drainage density, slope shape, and slope aspect. Slope aspect determines soil temperature, slope position determines moisture regime, and slope shape determines the sedimentation and depositional processes. Analyses of these factors indicate that once SOM has been severely depleted by natural or anthropogenic factors, it is difficult to restore in tropical agroecosystems, coarse-textured soils, drought-prone environments, and

nutrient-deficit [nitrogen (N), phosphorus (P), and sulfur (S)] conditions. Vegetation, and its residues, influences SOM content through the relative concentration of lignin, cellulose, polyphenols, and other recalcitrant compounds. Root system, its depth distribution, and turnover are also important to SOM stock in the profile, especially in the subsoil horizons. Roots contribute more to refractory SOM (relatively stable soil C pool) than the same amount of above-ground residue-derived C.^[3] Presence of stabilizing elements (especially N, P, and S) is important to converting biomass C into humus.^[4] In drier climates, SOM stock is favorably influenced by the hydraulic lift of deep-rooted shrubs.^[5]

ASSESSMENT OF SOM

There is a long history of determination of SOM concentration in the laboratories.^[6] Concentration of SOM in the laboratory is determined by either the wet combustion^[7] or dry combustion method.^[8,9] There are numerous modern techniques of measuring SOM stock *in situ*,^[10] rapidly and

in a cost-effective manner.^[11,12] There are also molecular-level methods for monitoring SOM concentration in relation to climate change,^[13] including nuclear magnetic resonance techniques and SOM biomarker techniques.^[14–19] Several practical indicators of changes in SOM concentration have been developed.^[20,21] While concentration of SOM in the root zone (measured as % or gram per kilogram) has been the unit of assessment for agricultural and forestry land uses, the assessment at regional, watershed, national, and global scales is done in units of megagram per hectare, teragram, or picogram.

MANAGEMENT OF SOM

The quantity and quality of SOM affect numerous soil processes and ecosystem services of significance at local, regional, and global scales (Fig. 1). The SOM concentration of 2–3% dry soil weight is needed to improve the use of efficiency of inputs and enhancing agronomic production.^[22] It is essential to advance global food security^[23] and create a positive C budget by reducing losses and

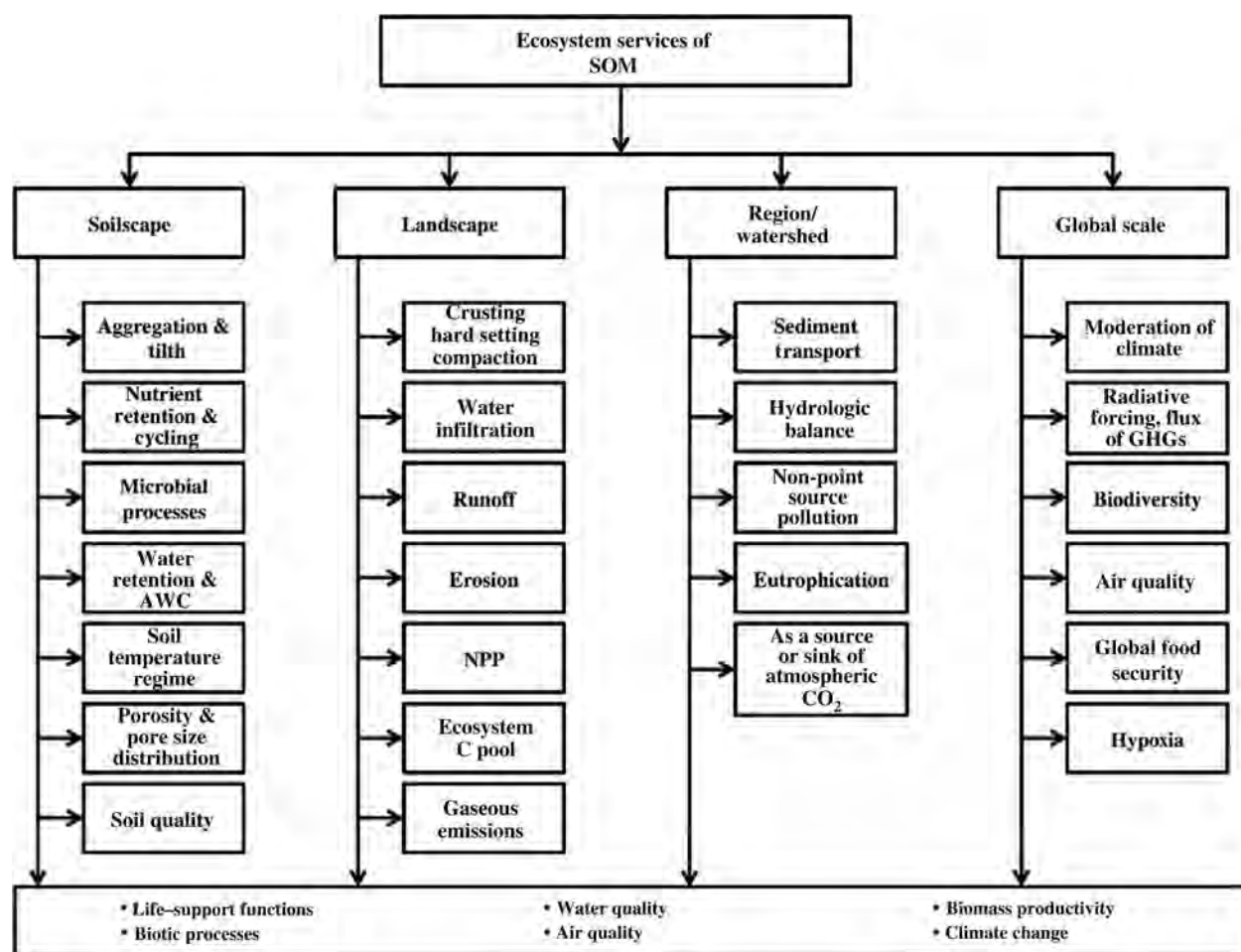


Fig. 1 Ecosystem services provided by SOM. Abbreviations: AWC, available water capacity; GHGs, greenhouse gases (e.g., CO₂, CH₄, and N₂O); NPP, net primary productivity.

Table 1 Soil organic matter as a determinant of specific components of soil quality.

Soil physical quality	Soil chemical quality	Soil biological quality
1. Soil structure and tilth	1. Soil reaction (pH)	1. Food for soil organisms
2. Porosity and pore size distribution; biopores	2. Cation exchange capacity	2. Habitat for biota
3. Plant's available water holding capacity	3. Elemental toxicity (Al^{3+} , Fe^{3+} , and Mn^{3+})	3. Microbial biomass carbon
4. Soil erodibility	4. Electrical conductance and salt concentration	4. Products of decomposition
5. Infiltration, percolation, run off, and leaching	5. Plant available nutrients (i.e., N, P, K, Cu, and Zn)	5. Soil respiration rate
6. Crusting, compaction, and hard setting	6. Nutrient/elemental cycling	6. Methanogenesis
7. Soil temperature regime and thermal conductivity	7. Chemical transformations	7. Nitrification/denitrification
8. Gaseous diffusion (diffusion coefficient)	8. Nutrient diffusion	8. Soil C sink capacity
9. Surface area	9. Organomineral complexes	9. Gaseous composition of soil air
10. Soil strength	10. Redox potential	10. Activity and species diversity of soil flora and fauna

increasing inputs of biomass C. Maintenance of SOM above the critical level enhances physical, chemical, and biological processes, which are strong determinants of the overall soil quality (Table 1 and Fig. 2). The processes are enhanced by the adoption of soil-specific practices. Some recommended management practices in agroecosystems include conservation tillage, mulch farming, cover cropping, integrated nutrient management including use of manures, biofertilizers, and biochar; agroforestry; controlled grazing; and improved pasture. Restoration of wetlands, erosion control, and afforestation of degraded and desertified lands are important options of enhancing the

SOM concentration and pool. In 1938, William Albrecht^[24] emphasized that “soil organic matter is one of our most precious resources; its unwise exploitation has been devastating; and it must be given its proper rank in any conservation policy as one of the major factors affecting the level of crop production in the future.” It was in a similar context, in view of the rising energy costs in late 1970s and the suggestion about the use of crop residues for biofuels, when Hans Jenny^[25] stated in 1980 that “I am arguing against indiscriminate conversion of biomass and organic wastes to fuels. The humus capital, which is substantial, deserves being maintained because good soils are a national asset.”

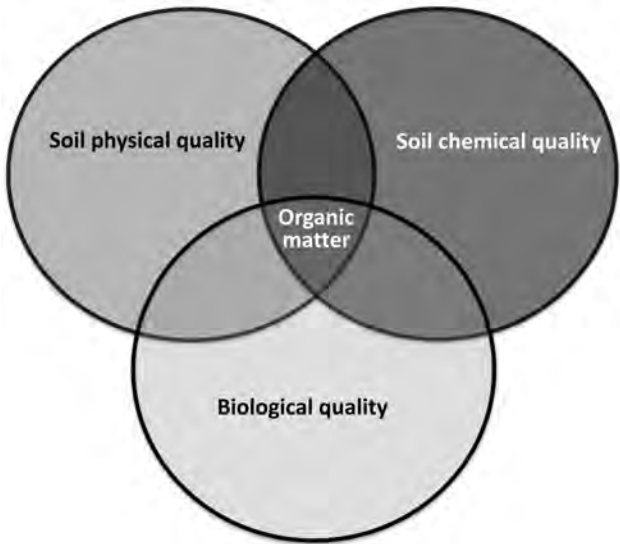
CONCLUSION

Quantity and quality of SOM and its maintenance above the threshold/critical level have been the foundation of past civilizations and are the bases on which the future of mankind depends. Its concentration in the root zone determines numerous ecosystem services including food security, climate change mitigation and adaptation, biodiversity, and water resources. It is the global public good, whose importance to human well-being and ecosystem functions can never be overstated. While strengthening and promoting long-term basic and applied research on processes and practices of its management and assessment at different scales, policy instruments must be in place at national and international levels for its enhancement and sustainable management.

REFERENCES

1. Allison, F.E. *Soil Organic Matter and Its Role in Crop Production*; Elsevier: Amsterdam, 1973.

Fig. 2 Interactive physical, chemical, and biological processes moderated by soil organic matter as a strong determinant of soil quality.



2. Schnitzer, M. Soil organic matter – the next 75 years. *Soil Sci.* **1991**, *151*, 41–58.
3. Kätterer, T.; Bolinder, M.A.; Andrén, O.; Kirchmann, H.; Menichetti, L. Roots contribute more to refractory soil organic matter than above-ground crop residues, as revealed by a long-term field experiment. *Agr. Ecosyst. Environ.* **2011**, *141*, 184–192.
4. Kirkby, C.A.; Kirkegaard, J.A.; Richardson, A.E.; Wade, L.J.; Blanchard, C.; Batten, G. Stable soil organic matter: A comparison of C:N:P:S ratios in Australian and other world soils. *Geoderma* **2011**, *163*, 197–208.
5. Armas, C.; Kim, J.H.; Bleby, T.M.; Jackson, R.B. The effect of hydraulic lift on organic matter decomposition, soil nitrogen cycling, and nitrogen acquisition by a grass species. *Oecologia* **2012**, *168* (1), 11–22.
6. Manlay, R.J.; Feller, C.; Swift, M.J. Historical evolution of soil organic matter concepts and their relationships with the fertility and sustainability of cropping systems. *Agr. Ecosyst. Environ.* **2007**, *119*, 217–233.
7. Allison, F.E. Wet-combustion apparatus and procedure for organic and inorganic carbon in soil. *Soil Sci. Soc. Am. Proc.* **1960**, *24*, 36–40.
8. Nelson, D.W.; Sommers, L.E. Total carbon, organic carbon and organic matter. In *Methods of Soil Analysis, Part 2. Chemical and Microbial Properties*. Amer. Soc. Agron. No. 9 (Part 2). Agronomy Series; Page, R.L., Miller, R.H., Keeney, D.R., Eds.; ASA/SSSA: Madison, 1982; 539–580.
9. Tiessen, H.; Moir, J.O. Total and organic carbon. In *Soil Sampling and Methods of Analysis*; Carter, M.R., Ed.; Lewis Publishers: Boca Raton, 1993; 187–199.
10. Wielopolski, L.; Chatterjee, A.; Mitra, S.; Lal, R. *In situ* determination of soil carbon pool by Inelastic Neutron Scattering: Comparison with dry combustion. *Geoderma* **2011**, *160* (3–4), 394–399.
11. Chatterjee, A.; Lal, R.; Wielopolski, L.; Martin, M.Z.; Ebinger, M.H. Evaluation of different soil carbon determination methods. *Crit. Rev. Plant Sci* **2009**, *28*, 164–178.
12. Lal, R.; Cerri, C.; Bernoux, M.; Etchevers, J.; Cerri, E. *The Potential of Soils of Latin America to Sequester Carbon and Mitigate Climate Change*; The Hawthorn Press: West Hazelton, 2006; 1–554.
13. Feng, X.; Simpson, M.J. Molecular-level methods for monitoring soil organic matter responses to global climate change. *J. Environ. Monit.* **2011**, *13*, 1246–1254.
14. Braun, S.; Ågren, G.I.; Christensen, B.T.; Jensen, L.S. Measuring and modeling continuous quality distributions of soil organic matter. *Biogeosciences* **2010**, *7*, 27–41.
15. Parton, W.J.; Schimel, D.S.; Cole, C.V.; Ojima, D.S. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Sci. Soc. Am. J.* **1987**, *51*, 1173–1179.
16. Nye, P.H.; Greenland, D.J. *The Soil Under Shifting Cultivation*. Technical Communication No. 51; Commonwealth Bureau Soils: Harpenden, 1960; 1–156.
17. Viaud, V.; Angers, D.A.; Walter, C. Toward landscape-scale modeling of soil organic matter dynamics in agroecosystems. *Soil Sci. Soc. Am. J.* **2010**, *74*, 1847–1860.
18. Neill, C. Impacts of crop residue management on soil organic matter stocks: A modeling study. *Ecol. Model* **2011**, *222*, 2751–2760.
19. Kutsch, W.L.; Bahn, M.; Heinemeyer, A. *Soil Carbon Dynamics: An Integrated Methodology*; Cambridge University Press: Cambridge, 2009; 1–286.
20. Sohi, S.P.; Yates, H.C.; Gaunt, J.L. Testing a practical indicator for changing soil organic matter. *Soil Use Manage.* **2010**, *26*, 108–117.
21. Plante, A.F.; Fernández, J.M.; Haddix, M.L.; Steinweg, J.M.; Conant, R.T. Biological, chemical and thermal indices of soil organic matter stability in four grassland soils. *Soil Biol. Biochem.* **2011**, *43*, 1051–1058.
22. Lal, R. Beyond Copenhagen: Mitigating climate change and achieving food security through soil carbon sequestration. *Food Secur.* **2010**, *2*, 169–177.
23. Magdoff, F.; Van Es, H. *Building Soils for Better Crops: Sustainable Soil Management*; SARE/USDA: Beltsville, 2009; 294 pp.
24. Albrecht, W. Loss of soil organic matter and its restoration. In *Soils and Men, USDA Yearbook of Agriculture*; U.S. Government Printing Office: Washington, 1938; 347–360.
25. Jenny, H. Alcohol or humus. *Science* **1980**, *209*, 444.

Soil Organic Matter (SOM): Accumulation

Sylvie A. Quideau

Department of Renewable Resources, University of Alberta, Edmonton,
Alberta, Canada

Abstract

Organic matter plays a central role in soil functioning. It acts as a nutrient storehouse, improves soil physical properties such as water retention and porosity, and provides a habitat for soil microbes. Accordingly, organic matter is often used as a proxy for soil quality. Its accumulation reflects the balance between organic matter additions (e.g., littering) and losses (e.g., decomposition). Its extent, type and rate of accumulation vary widely among soils as a function of the five key forming factors, i.e., climate, organisms, relief, parent material, and time. Long-term rates of carbon accumulation in soils vary from 0.2 to 12 gC m⁻² yr⁻¹. Finally, organic matter is an important criterion for classifying soils, in particular to separate organic from mineral soil orders, and secondly to recognize diagnostic surface and subsurface horizons.

INTRODUCTION

Chemical, biological, and physical processes continually transform organic matter (OM) added to soils in the form of plant or animal detritus. Soil OM includes residual components of original organic tissues, their degradation products, and products synthesized by soil fauna and microbes. Two major types of natural organic compounds are distinguished: 1) non-humic substances, belonging to identifiable chemical classes such as carbohydrates; and 2) humic substances, a series of brown to dark brown, high-molecular weight biopolymers distinctive to soil environments.

OM accumulation in soils results from the combination of several pedogenic processes that can be conceptualized and separated into four broad categories, namely: 1) OM additions; 2) transformations; 3) transfers; and 4) losses. OM accumulation is also directly linked to the genesis of soils. Hence it is dependent on defined environmental factors, also known as soil-forming factors, which include climate, organisms, relief, parent material, and time. Finally, OM is recognized as a key attribute of soils, as it is instrumental in the vast majority of soil physical, chemical, and biological properties. With regard to soil classification, the amount of carbon (C) accumulation is used at the higher level to separate mineral from organic soils. At the lower level, the location and type of OM accumulation within a soil profile help identify a series of diagnostic surface and subsurface horizons.

PEDOGENIC PROCESSES

Additions to Soils

Littering refers to the deposition of dead plant and animal detritus on the soil surface (Table 1). As litter decays,

soluble organic products leach into the mineral soil. Alternatively, particulate litter can be incorporated into the soil by faunal *pedoturbation*, which is soil mixing by animals, including termites, earthworms, ants, and rodents. *Root turnover* also may constitute an important addition of C into the mineral soil. In forests, relatively more C is added as surficial litter; in grassland ecosystems, up to two-third of carbon is added through the decay of roots.

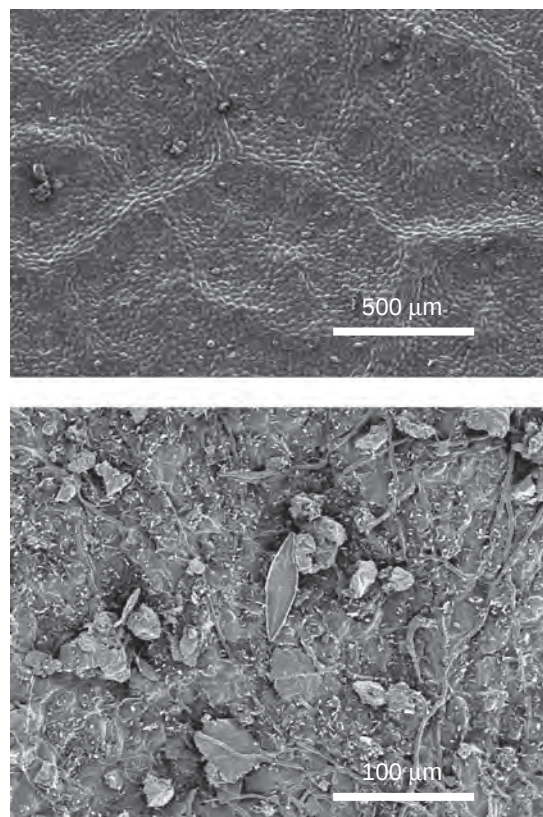
OM Transformations

Decomposition refers to the chemical and biochemical reactions occurring during the decay of plant and animal remains as soil microorganisms colonize them (Fig. 1). Decomposition involves the fragmentation of biopolymers into smaller molecules, with carbon dioxide production as the ultimate step in aerobic soils, and methane production occurring in anaerobic environments. Organic litter is composed of distinct chemical components that decompose at different rates. In addition to focused on litter chemistry, including nutrient concentrations and ratios such as the C-to-nitrogen (N) ratio or lignin-to-N ratio, physical attributes of litters affect their decomposition rates.^[1] *Mineralization* releases soluble or gaseous inorganic constituents during decomposition processes.

Humification is a multistage process. Source materials for humus synthesis include residual components from incomplete decomposition of organic litter and the products of microbial anabolic activities. According to traditional concepts, polyphenols derived from lignin degradation, together with those synthesized by microorganisms, are oxidized to quinones, which undergo self-polymerization or combine with amino compounds to form N-containing polymers.^[2] More concepts recognize the importance of

Table 1 Fundamental pedogenic processes associated with OM accumulation.

Term	Definition	Representative horizon (soil order)
Littering	Accumulation of fresh organic detritus on the mineral soil surface to a depth of <30 cm	Oi ^a or L ^b horizon (all soil orders)
Root turnover	Subsurface C addition through root death	
Decomposition	Breakdown of organic molecules into simpler compounds	Oe and Oa ^a , F and H ^b , and A horizons (all soil orders)
Mineralization	Release of inorganic constituents of OM	
Humification	Formation of humic substances from raw organic materials	
Synthesis	Formation of new organic molecules by combination of elements or constituents	
Paludization	Accumulation of thick (>30 cm deep) organic materials on the mineral soil surface	Folistic, histic horizon (Histosols) ^{a,c} Of, Om, Oh horizons (organic soils) ^b
Ripening	Changes in organic soil promoted by entry of air into previously waterlogged material	
Melanization	Darkening of light-colored initial mineral soil by addition of OM	Mollic horizon (Mollisols ^a , Phaeozems ^c , and Kastanozems ^c) Chernozemic horizon (Chernozems) ^b Chernic horizon (Chernozems) ^c Melanic epipedon (Andisols ^a and Andosols ^c) Umbric epipedon (Umbrisols) ^c
Podzolization	Translocation of OM in the soil profile associated with Al and Fe migration	Spodic horizon (Spodosols ^a and Podzols ^c) Podzolic horizon (Podzols) ^b

^aFrom Soil Survey Staff.^[3]^bFrom Soil Classification Working Group.^[4]^cFrom IUSS Working Group WRB.^[5]**Source:** Modified from Buol, Southard, et al.^[6] ©2003 Iowa State University Press.**Fig. 1** Scanning electron micrographs of a fresh (top) and decomposing (bottom) manzanita leaf left in the field for 1 year.

enzymes of biological origin as important catalysts of the humification pathways.

Accumulation in Organic Soils

Paludization can be considered as a geogenic rather than pedogenic process because it involves deposition of initial parent material. Paludization occurs when conditions impede decomposition, enabling the buildup of a thick mass of organic deposits. Decomposition is hampered by poorly drained conditions, as in Histosols^[3] and organic soils,^[4] and by extreme cold, as in Histels, a suborder of the Gelisols,^[3] and in organic Cryosols.^[4] Under anaerobic conditions, humic substances exhibit a relative accumulation of aromatic C compounds arising from the absence of lignin-degrading fungi. In contrast to paludization, *ripening* refers to the decomposition processes occurring in the organic horizon under oxidizing conditions after exposure to the air.

Surface Accumulation in Mineral Soils

Melanization produces the thick, dark-colored surface horizons characteristic of Mollisols^[3] and Chernozems.^[4,5] The formation of mollic epipedons is promoted by the proliferation of grass roots that constitute a considerable input of plant residues. Another key factor in melanization

is the active faunal community, which contributes to the rapid incorporation of the residues into the mineral soil and favors high initial mineralization rates. In the World Reference Base for Soil Resources,^[5] a chernic horizon is considered a special type of mollic horizon with high biological activity. Subsequent decomposition and humification processes result in the formation of chemically stable, dark-colored humic substances. Multivalent cations, such as calcium, act as bridges between organic colloids and clay particles and stabilize organic substances within the soil matrix. In soils with lower base status, surface accumulation of OM may lead to the formation of an umbric epipedon, as is the case in Umbrepts^[3] and Umbrisols.^[5] In allophanic soils derived from volcanic parent material (Andisols^[3]), OM accumulation is favored by the formation of resistant organic-alumina complexes.

Subsurface Accumulation

Podzolization results in the formation of subsurface horizons of OM accumulation characteristic of Spodosols^[3] and Podzols.^[4,5] Subsurface OM accumulation occurs at the top of the spodic horizon due to the migration of water-soluble organic compounds from the mineral surface.^[7] Soluble organics originate from the decomposition processes of nutrient-poor, acidic organic residues, such as heath and coniferous litter. According to the traditional metal-fulvate theory, organometallic complexes form in the decomposing litter under conditions of low metal saturation, complex more iron (Fe) and aluminum (Al) as they flush down the soil profile with percolating water, and precipitate when the ratio of metal to C reaches a critical level. Other proposed mechanisms for immobilization include flocculation, polymerization, and sorption on mineral surfaces.

ENVIRONMENTAL CONTROLS

The rate of OM accumulation varies greatly among soils, reflecting the influence of environmental factors on pedogenic processes (Table 2):

$$OM = f(\text{climate, organisms, parent material, relief, and time})$$

where *f* stands for “is a function of.”

Climate

Climate influences OM accumulation by controlling the balance between litter production and decomposition rates. On a global basis, soil C content increases with increasing precipitation but decreases with increasing temperature. In situations in which moisture is not a controlling factor, decomposition increases with increasing temperature. Litter production follows the same trend. Therefore, the

Table 2 Long-term accumulation rates of soil C in different ecosystem types.

Ecosystem type	Accumulation interval (yr)	Long-term rate of accumulation (g C m ⁻² yr ⁻¹)
Polar desert ^b	2,600	2.4
Polar desert ^b	9,000	0.2
Xeric scrubland ^a	12,000	0.2
Desert grassland ^b	3,000	0.8
Temperate grassland ^a	3,900	7.4
Temperate grassland ^b	9,000	2.2
Temperate grassland ^a	240,000	0.6
Boreal forest ^b	3,500	11.7
Boreal forest ^b	5,400	0.8
Temperate coniferous forest ^b	1,200	10.0
Temperate deciduous forest ^b	2,000	5.1
Temperate deciduous forest ^a	14,000	0.9–2.5
Tropical forest ^b	3,500	2.5

^aFrom Chadwick, Kelly, et al.^[8]

^bFrom Schlesinger.^[9]

Source: Modified from Chadwick, Kelly, et al.^[8] ©1994 Springer and Schlesinger.^[9] ©1990 Nature Publishing Group (NPG).

worldwide accumulation of soil OM is related more to factors controlling decomposition than to the productivity of ecosystems. On a local and regional scale, OM content decreases exponentially with rising annual temperature at any given level of precipitation:^[10]

$$OM = Ce^{-kT}$$

where *T* is the mean annual air temperature (°C), and *C* and *k* are constants. On the other hand, OM accumulation does not follow a climatic pattern in poorly drained soils where anaerobic conditions impede decomposition processes.

In addition to its influence on total C content, climate may also affect the chemical composition of soil OM. High rainfall favors high leaching regimes and increases alkyl C while reducing the development of aromaticity in soil OM.^[11] Aromaticity also has been reported to be negatively correlated to the precipitation/temperature ratio. For prairie soils of the Great Plains, polysaccharides decreased with increasing temperature but increased with increasing precipitation.^[12] Similarly, OM composition was reported to change in conjunction with total C stocks along a climosequence of boreal forest soils.^[13] Namely, total soil C stocks (0–1 m) increased with decreasing mean annual temperature, while the degree of OM decomposition decreased along the soil sequence.

Organisms

The amount, placement, chemical composition, and physical attributes of the organic residues of vegetation also affect

OM accumulation. Litter production worldwide declines with increasing latitude from tropical to arctic forests, following the same distribution patterns as net primary productivity. The placement of organic residues affects the distribution of OM accumulation within the soil profile. In grassland soils, where belowground production is abundant, OM is more evenly distributed than it is in forest soils, where most accumulation occurs in the uppermost horizon. The chemical composition and physical attributes of litter both exert a significant influence on the accumulation and turnover of soil OM by determining the palatability of the plant material, which in turn can alter the distribution and activity of soil fauna. Soil animals, such as earthworms, may accelerate decomposition rates by contributing to the rapid mixing of fresh plant residues into the mineral soil.

Climatic factors are often considered to have a greater impact on OM composition than does the original composition of the litter inputs, in other words than the type of vegetation growing on site.^[8] While vegetation and climate are considered as two discrete environmental or independent soil-forming factors, they often covary in natural landscapes, and their independent assessment remains challenging. One potential approach to separate the influence of vegetation from that of climate is to compare a strict biosequence (soils developed under a similar climate but different vegetative covers) and a climo/biosequence (where both vegetation and climate vary). Results from work conducted along two Californian soil sequences suggested that vegetation rather than climate was controlling OM composition.^[2]

Parent Material

Parent material may influence OM accumulation through its effect on soil fertility. Soils formed from base cation-rich volcanic rocks (e.g., basalt) are typically more fertile, and thus experience more OM accumulation than soils with lower inherent mineral-derived nutrients, such as those formed from granitic materials. Parent material is also effective through its determination of soil texture. Soil clay content and OM accumulation are positively correlated. Clay content affects soil moisture and water availability, thereby modifying plant productivity and litter production. Additionally, high clay content may induce OM accumulation by stabilizing humic substances formed during decomposition. Clay and OM form organo-mineral complexes that are resistant to further biodegradation. Lastly, the texture of the parent material may indirectly affect OM accumulation through its influence on structure development. For example, fine textured soils tend to contain more aggregates than sandy soils, and OM may preferentially accumulate within these aggregates.

Relief

Topography interacts with microclimate to influence OM distribution in soils. OM accumulation is often favored at

the bottom of hills where conditions are wetter than at mid or upper slope positions, resulting in lower decomposition losses. In a similar fashion, OM accumulation is usually greater on north-facing slopes compared with south-facing slopes (in the northern hemisphere) because temperature is lower; lower temperatures are conducive to both lower primary productivity and decomposition losses.

Time

Long-term rates of OM accumulation in Holocene-aged soils vary from less than 1 to 12 g Cm⁻²yr⁻¹.^[8] OM, however, does not accumulate indefinitely in soils. Depending on other soil-forming factors, an equilibrium level is reached over time. OM encompasses a series of pools with varying turnover rates. Amounts of the young, labile OM may level off in decades because plant biomass stabilizes, while amounts of the more recalcitrant fractions, composed of humic substances often complexed with clay minerals, may continue to increase for tens of thousands of years.

Human Influence

Human influence is considered as part of the “organisms” environmental factor in the original formulation by Hans Jenny,^[10] alongside with microorganisms, vegetation, and animals. Since then, many authors have proposed that humans be recognized as a sixth soil-forming factor. Human influence on OM accumulation in soils ranges from the simple alteration of a single soil horizon, for instance surficial OM enrichment linked to manure addition, to more drastic changes including the complete reconstruction of entire soils, such as is the case during land reclamation following surface mining. Human-affected soils are starting to get included in several soil classification systems, although a consensus is lacking. For instance, the anthropic and plaggen epipedons are two horizons that exhibit OM accumulation as a result of human activities.^[5]

CONCLUSION

Our appreciation of the importance of OM in soil functioning and our understanding of the factors and processes that control OM accumulation have very much developed alongside the discipline of pedology. As early as the 19th century, Vasily Dokuchaev, commonly regarded as the father of pedology, observed that soils change predictably as a result of their environmental settings. Much of his work focused on understanding the conditions favoring OM accumulation in Chernozems of Russia. At the time, interest in OM mostly arose from its pivotal role in the fertility and overall productivity of soils. Discussions of climate change mitigation and C trading have emphasized the role of soils as an important global C pool. Understanding

how OM accumulation and storage respond to changing environmental conditions is becoming increasingly important to accurately model C at the global scale.

OM accumulation in soils varies horizontally and vertically across the landscape and evolves with time through the combined influence of several pedogenic processes. Soil OM encompasses a series of C pools with differing residence times. Labile C may turnover within days or years, while stable C may persist in soils for thousands of years. In between these two extremes, lies C with intermediate (decadal) turnover rates, which may be of most relevance to the overall soil C response to climatic changes over the next century. Hence, much research is targeted at understanding the factors controlling not only the accumulation of OM in soils but maybe more importantly the stability of the accumulated OM. Lastly, C in subsoil horizons has been investigated less than surficial C. Deep soil C (i.e., below 20 cm) represents more than one-half of the total C stored in soils globally. Understanding the mechanisms responsible for deep C accumulation and stability is emerging as one of the key challenges in soil science.

REFERENCES

1. Quideau, S.A.; Chadwick, O.A.; Benesi, A.; Graham, R.C.; Anderson, M.A. A direct link between forest vegetation type and soil organic matter composition. *Geoderma* **2001**, *104*, 41–60.
2. Quideau, S.A.; Graham, R.C.; Oh, S.-W.; Hendrix, P.F.; Wasylishen, R.E. Leaf litter decomposition in a chaparral ecosystem, Southern California. *Soil Biol. Biochem.* **2005**, *37*, 1988–1998.
3. Soil Survey Staff. *Keys to Soil Taxonomy*, 12th Ed.; Natural Resources Conservation Service; U.S. Department of Agriculture: Washington, DC, 2014; 372 pp.
4. Soil Classification Working Group. *The Canadian System of Soil Classification*, 3rd Ed.; Agriculture and Agri-Food Canada Publication 1646: Ottawa, ON, 1998; 187 pp.
5. IUSS Working Group WRB. World reference base for soil resources 2014, international soil classification system for naming soils and creating legends for soil maps. *World Soil Res Rep.* **2014**, *6*, 203.
6. Buol, S.W.; Southard, R.J.; Graham, R.C.; McCracken, R.J., Eds. Biogeochemical processes in soil formation. In *Soil Genesis and Classification*, 5th Ed.; Iowa State University Press: Ames, 2003; 85–118.
7. Browne, B.A. Towards a new theory of podzolization. In *Carbon Forms and Functions in Forest Soils*; McFee, W.W., Kelly, J.M., Eds.; Soil Science Society of America: Madison, 1995; 253–273.
8. Chadwick, O.A.; Kelly, E.F.; Merritts, D.M.; Amundson, R.G. Carbon dioxide consumption during soil development. *Biogeochemistry* **1994**, *24*, 115–127.
9. Schlesinger, W.H. Evidence from chronosequence studies for a low carbon storage potential of soils. *Nature* **1990**, *348*, 232–234.
10. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
11. Preston, C.M. Applications of NMR to soil organic matter analysis: History and prospects. *Soil Sci.* **1996**, *161* (3), 145–166.
12. Amelung, W.; Flach, K.; Zech, W. Climatic effects on soil organic matter composition in the great plains. *Soil Sci. Soc. Am. J.* **1997**, *61*, 115–123.
13. Norris, C.E.; Quideau, S.A.; Bhatti, J.S.; Wasylishen, R.E. Soil carbon stabilization in jack pine stands along the Boreal Forest Transect Case Study. *Glob. Change Biol.* **2011**, *17*, 480–494.

Soil Organic Matter (SOM): Available Water Capacity

Tom G. Huntington

U.S. Geological Survey (USGS), Augusta, Maine, U.S.A.

Abstract

Available water capacity (AWC) is defined as the amount of water (cm^3 water/ 100 cm^3 soil) retained in the soil between field capacity (FC) and permanent wilting point (PWP). FC and PWP are defined as the volumetric fraction of water in the soil at soil water potentials of 10–33 and 1500 kPa, respectively. One of the paradigms of soil science is that AWC is positively related to soil organic matter (SOM) because SOM raises FC more than PWP. SOM enhances soil water retention because of its hydrophilic nature and its positive influence on soil structure. It improves soil structure by enhancing aggregate formation, thereby increasing porosity in the range of pore sizes that retain plant-available water and enhancing infiltration and water retention throughout the rooting zone. Understanding the sensitivity of AWC to SOM is important for assessments of how environmental changes that influence SOM content could affect AWC and sustainability of agricultural productivity.

INTRODUCTION

Available water capacity (AWC) is defined as the amount of water (cm^3 water/ 100 cm^3 soil) retained in the soil between field capacity (FC) and permanent wilting point (PWP).^[1] FC and PWP are defined as the volumetric fraction of water in the soil at soil water potentials of 10–33 and 1500 kPa, respectively. One of the paradigms of soil science is that AWC is positively related to soil organic matter (SOM) because SOM raises FC more than PWP^[2–4] (Fig. 1). SOM enhances soil water retention because of its hydrophilic nature and its positive influence on soil structure.^[5,6] Increasing SOM increases soil aggregate formation and aggregate stability^[7] (Fig. 2), thereby increasing porosity in the range of pore sizes that retain plant-available water and enhancing infiltration and water retention throughout the rooting zone. When SOM decreases, soil aggregation and aggregate stability decrease and bulk density increases.^[8] These changes in physical properties result in lower infiltration rates and higher susceptibility to erosion.^[9,10] This entry reviews the literature on the sensitivity of AWC to soil organic carbon (SOC) and discusses the environmental implications of changes in SOM and AWC.

SENSITIVITY OF AWC TO SOM

Many studies have determined that AWC decreases as SOM decreases, but considerable variation in this relation has also been reported. A majority of studies have used regression analysis to quantify the sensitivity of AWC to SOM. Other estimates of the relation between SOM and AWC have been obtained from changes in water retention

characteristics over time for a specific soil where different management, fertilization, or erosion histories have resulted in changes in SOM. To facilitate comparison among estimates of the relation between AWC and SOM in this review, sensitivity was defined as the average change in AWC (cm^3 water/ 100 cm^3 soil) predicted for a 5% relative decrease in SOM where the initial SOM concentration was in the range of 2–4% by weight. If texture was included in the published regression, it was held constant at 65% silt, 20% clay, and 15% sand. If bulk density was included in the regression, it was assumed that bulk density decreased with increasing SOM following the regression equation:

$$\text{Bulk density} = 1.723 - 0.212(\text{OC})^{0.5} - 0.0006(\text{WC15})^2$$

where OC = percent soil organic carbon and WC15 = water content at 1500 kPa.^[11] WC15 for a soil with the texture of 65% silt, 20% clay, and 15% sand was assumed to be 11.3 cm^3 water/ 100 cm^3 soil. Where data or regression analysis required transformation of soil organic carbon (SOC) to SOM, it was assumed that $\text{SOM} = 1.724 \times \text{SOC}$.

Reported sensitivities fell in the following ranges (0 to +0.27),^[12–14] (0 to –1),^[15–20] (–1 to –2),^[21–24] and (< -2)^[25–27] (cm^3 water/ 100 cm^3 soil) per 5% relative decrease in SOM concentration. The average value for the sensitivity was $-1.1 (\pm 0.34 \text{ SE}) \text{ cm}^3$ water/ 100 cm^3 soil per 5% relative decrease in SOM concentration. Out of the 15 studies, 3 found a negative or no relation between SOM and AWC. Regression models that indicated no relationship between SOM and AWC typically found that SOM resulted in equivalent increases in water retention at both FC and PWP with the result that AWC did not change.

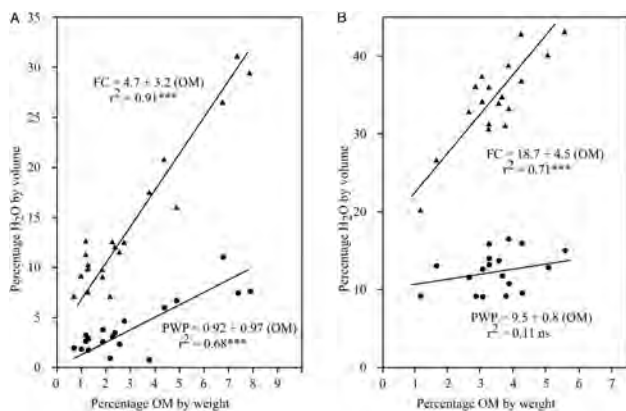


Fig. 1 Water content at FC and PWP for sandy (A) and silt loam (B) textured soils. *** indicates that the relationship is statistically significant with p -value <0.0001 and ns indicates that the relationship was not statistically significant with p -value >0.05 .
Source: Reproduced from Hudson.^[3]

In addition, two large studies have used the U.S. National Soil Inventory database to study the relationship between SOM and AWC where AWC was based on reported values for FC and PWP.^[28,29] One study^[28] showed a positive relationship between SOM and plant AWC that was dependent on soil texture and reported that FC was more sensitive to SOM than PWP. In another study,^[29] a complex relation was reported such that a 1% increase in SOM causes a 2% to >5% increase in soil AWC depending on the soil texture. Both studies

concluded that coarse-textured soils had a higher sensitivity to SOM than fine-textured soils and that sensitivity decreased as SOM content increased. These two studies^[28,29] are consistent with the findings of most of the other cited studies that have found that increases in SOM result in proportionately larger increases in AWC in medium- to coarse-textured soils compared with finer-textured soils. Most studies found that aggregate stability has been shown to increase with increasing SOM at similar rates across a wide range of clay contents^[7] (Fig. 2). However, in that study^[7] it was also shown that with increasing clay content, soils require a higher SOM content to maintain a given aggregate stability value (Fig. 2). As SOM levels decrease, as occurs following chronic high rates of erosion, the effect of incremental decreases in SOM on AWC becomes progressively smaller. In heavily eroded soils, clay contents become increasingly more important than SOM in influencing hydraulic properties.^[10]

The physical and chemical properties of SOM could influence how SOM affects soil water retention. Differences in SOM quality may arise from differences in inputs, such as natural plant residues, manure, or sludge, or they may result from different soil drainage conditions. These and other factors such as mineralogy, exchangeable cations, and surface chemical properties that can influence the formation and stability of aggregates and mesopores complicate the simple relation between AWC and SOM.^[6]

There are several other possible explanations for the high variability in the reported sensitivity of AWC to SOM. Differences in methods used to measure water retention at both FC and PWP could introduce bias. Both *in situ* and laboratory methods have been used for the determination of water retention at FC and PWP. Laboratory methods can involve everything from intact cores to air-dried, sieved, rewetted, and recompact soil samples.^[1] Any pretreatment that changes soil porosity and soil aggregation will affect estimated water retention. There is no agreed-upon standard water potential for the estimation of FC in the laboratory. Different methods have been used to determine SOC and SOM. Differences in texture, mineralogy, bulk density, coarse fragment volume, and quality of the SOM of the soils used in the regression analyses could mask the effects of SOM. Some reports involve controlled regressions where variables such as texture are held constant. It may not be surprising that in uncontrolled regressions, the effects of SOM are obscured.^[3,6]

Soil water retention data from long-term plots, such as classic crop rotations, fertility experiments, and no-till vs. conventional tillage experiments, may be more appropriate for this type of sensitivity analysis. In these types of experiments, SOM changes measurably over time, but other variables are essentially held constant. However, there have been few reports of this kind. At the Sanborn Field in Missouri, continuous cropping for over a century resulted in large decreases in SOM and soil productivity.^[30] Losses

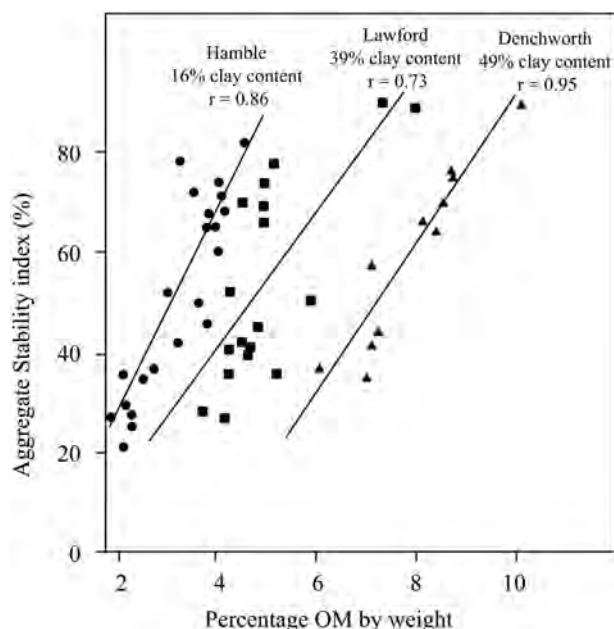


Fig. 2 Relationship between aggregate stability and soil organic matter content for samples from the Hamble, Lawford, and Denchworth soil series from southern England.
Source: Redrawn from Haynes & Beare.^[7]

of SOM at the Sanborn Field could have increased erosion and loss of topsoil that likely decreased AWC.^[30] However, in another study at the Sanborn Field,^[15] no significant differences in water retention among treatments were observed for soil cores collected between 2.5 and 10 cm depth. A study in Denmark reported that after 90 years of manure additions, AWC was substantially higher on the amended plots than on unfertilized reference plots.^[27] In a 32-year continuous corn experiment, cropping treatments that resulted in lower SOM (removal of corn stover and tillage) reduced SOM by 8% and 25% and decreased AWC by 8% and 13%, respectively.^[31] In a shorter-term study, the addition of manures to grasses, cereals, and sugar beets for seven or more years increased AWC at four sites with different soils in the United Kingdom.^[32]

ENVIRONMENTAL IMPLICATIONS

Environmental changes such as land use conversions, changes in agricultural management practices, climate change, and changes in acidic deposition could affect SOM contents and, consequently, AWC. Land use conversions from grasslands or forests to cultivated agriculture^[33] or intensification of agriculture^[34] are likely to result in net losses of SOM, but improvements in agricultural management practices have the potential to substantially enhance SOM contents and increase AWC.^[35] Historical increases in acidic deposition followed by decreases in decades have been associated with concomitant increases and decreases in SOM in forest soils.^[36,37]

It is generally acknowledged that global mean annual temperature will likely increase in the range of 1.4–5.8°C during the 21st century as atmospheric carbon dioxide

(CO₂) concentrations increase.^[38] However, there is substantial uncertainty in how SOM levels will respond to the associated soil warming and elevated CO₂ concentrations. Some modeling^[39–41] and empirical studies where land use and management were held constant^[42–47] are consistent with declines in SOM following increases in temperature, but other environmental factors cannot be ruled out. Other studies suggest only no or small declines in SOM with increasing temperature or increases in SOM if precipitation increases along with increasing temperature.^[48–51] In one study^[52] it was concluded that temperature affects SOM polymerization and adsorption and desorption to mineral surfaces as well as the production and conformation of microbial enzymes. They argued that the net effect of these processes on the availability and degradability of SOM will determine how SOM will be affected by increasing soil temperatures. In another study,^[53] it was reported that warming accelerates decomposition of decades-old carbon in forest soils. In yet another study,^[54] elevated CO₂ was shown to have a priming effect on the decomposition of older SOM and to accelerate the decomposition of new detrital inputs. Complex relations have been reported between SOM and temperature and CO₂ that suggest both increases and decreases in SOM,^[52,55,56] with the result that there is no consensus on the overall effect of these environmental changes on SOM in soils.

In areas where climate change results in a hotter and drier environment with more variability and extreme events of drought and heavy rainfall, as has been predicted for some regions,^[57,58] soils will be subjected to more influence from a number of processes that are conducive to structural degradation.^[59] Soil aggregate stability and soil structural integrity are likely to be adversely affected by losses in SOM that would accompany drying conditions.

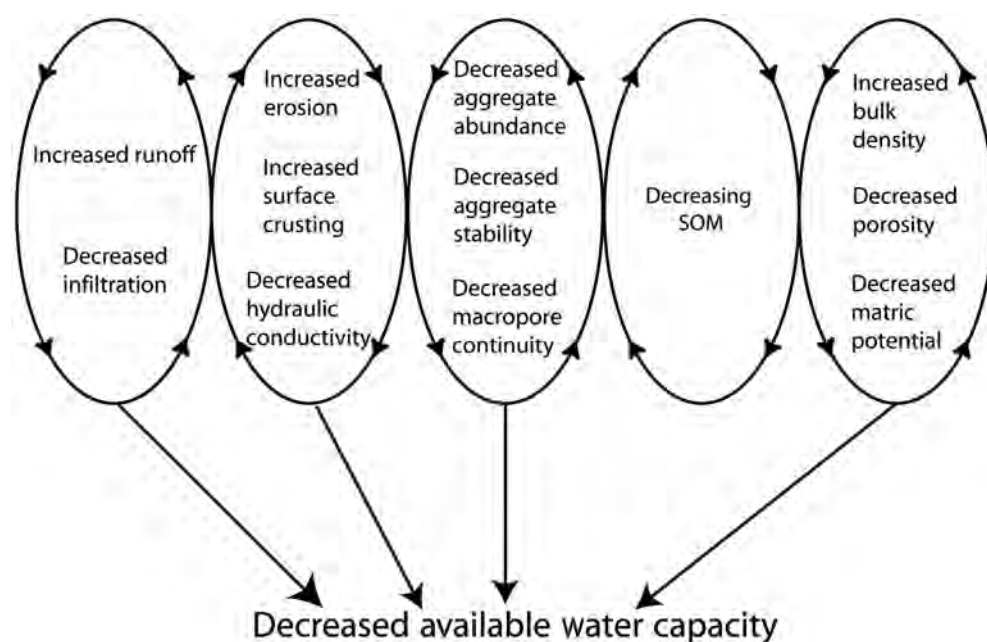


Fig. 3 A conceptual model of the major effects of decreasing SOM on soil physical properties that interact through the loops shown to decrease available water capacity in cultivated, eroding agricultural fields.

Structural degradation is associated with a loss in AWC and crop water use efficiency and increasing susceptibility to erosion. Decreases in SOM content would have adverse effects on soil chemical and physical properties because of the importance of SOM to soil structure and hydrologic properties. SOM is important for the maintenance of good soil structure through its positive influence on soil aggregate formation and stability^[7] (Fig. 2). Aggregation increases soil porosity and provides for improved water retention within the range of soil water potential where water is available to plants. It has been recognized that decreases in SOC could reduce AWC and soil fertility with resulting adverse consequences on soil health and productivity.^[60,61]

Reductions in SOM in the surface soil result in increased bulk density, decreased porosity, and increases in matric potential that, in turn, result in decreased AWC (Fig. 3). Decreases in SOM also result in decreases in soil aggregation, aggregate stability,^[62,7] and macropore continuity, thus causing increases in surface crusting and decreases in hydraulic conductivity (Fig. 3). These changes in physical properties lead to reduced rates of water infiltration and increases in runoff and erosion.^[10] A climate warming-induced intensification of the hydrologic cycle could increase runoff and erosion in some regions.^[63] Decreased infiltration and increased runoff result in less precipitation retained throughout the soil rooting zone so that less precipitation becomes available for plant growth. SOM exerts a strong influence on AWC through these interacting feedback loops that affect many soil physical properties (Fig. 3).

CONCLUSION

Despite considerable uncertainty in the relation between AWC and SOM, most studies are consistent with the classical conceptual understanding that these soil properties are positively related. SOM promotes soil aggregate formation and the development of soil porosity that enhances infiltration and retention of water in a range that is available to plants. Environmental changes that result in increases in SOM will increase AWC. On the other hand, environmental changes that result in decrease in SOM will decrease AWC with resulting adverse consequences for the sustainability of agricultural productivity in some regions. There is a need to improve the quantitative understanding of the sensitivity of AWC to SOM to reduce the level of uncertainty and to provide resource managers with better decision support systems.

ACKNOWLEDGMENTS

Rattan Lal, Harry Vereecken, Alan Olness, and Jeffery Kern provided useful insights and references on the sensitivity of AWC to SOM. Miko Kirschbaum and Gregory Lawrence provided valuable insights that substantially improved this revised entry. Funding for this entry was

provided by U.S. Geological Survey Climate and Land Use Change Research and Development Program.

REFERENCES

1. Romano, N.; Santini, A. Measurement of water retention: Field methods. In *Methods of Soil Analysis, Part 4*; Dane, J.H., Topp, G.C., Eds.; SSSA: Madison, 2002; 721–738.
2. Jenny, H. *The Soil Resource: Origin and Behavior*; Springer-Verlag: New York, 1980.
3. Hudson, B.D. Soil organic matter and available water capacity. *J. Soil Water Conserv.* **1994**, *49*, 189–194.
4. Emerson, W.W. Water retention, organic carbon and soil texture. *Aust. J. Soil Res.* **1995**, *33*, 241–251.
5. Stevenson, F.J. *Humus Chemistry. Genesis, Composition, Reactions*; John Wiley and Sons: New York, 1982.
6. Kay, B.D. Soil structure and organic carbon: A review. In *Soil Processes and the Carbon Cycle*; Lal, R., Ed.; CRC Press: Boca Raton, 1997; 169–197.
7. Haynes, R.J.; Beare, M.H. Aggregation and organic matter storage in meso-thermal, humid soils. In *Structure and Organic Matter in Agricultural Soils*; Carter, M.R., Stewart, B.A., Eds.; CRC Lewis: Boca Raton, 1996; 213–262.
8. Haynes, R.J.; Naidu, R. Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: A review. *Nutr. Cycl. Agroecosys.* **1998**, *51*, 123–137.
9. Ascoug, J.C., II; Baffaut, C.; Nearing, M.A.; Liu, B.Y. The WEPP watershed model: I. Hydrology and erosion. *Trans. ASAE* **1997**, *40*, 921–933.
10. Rhoton, F.E.; Lindbo, D.L. A soil depth approach to soil quality assessment. *J. Soil Water Cons.* **1997**, *52*, 66–72.
11. Manrique, L.A.; Jones, C.A. Bulk density of soils in relation to soil physical and chemical properties. *Soil Sci. Soc. Am. J.* **1991**, *55*, 476–481.
12. Vereecken, H.; Maes, J.; Feyen, J.; Darius, P. Estimating the soil moisture retention characteristics from texture, bulk density and carbon content. *Soil Sci.* **1989**, *148*, 389–403.
13. Lal, R. Physical properties and moisture retention characteristics of some Nigerian soils. *Geoderma* **1979**, *21*, 209–223.
14. Anderson, S.H.; Gantzer, C.J.; Brown, J.R. Soil physical properties after 100 years of continuous cultivation. *J. Soil Water Conserv.* **1990**, *45*, 117–121.
15. Gupta, S.C.; Larson, W.E. Estimating soil water retention characteristics from particle size distribution, organic matter content and bulk density. *Water Resour. Res.* **1979**, *15*, 1633–1635.
16. Thomasson, A.J.; Carter, A.D. Current and future uses of the UK soil water retention dataset. In *Indirect Methods for Estimating the Hydraulic Properties of Unsaturated Soils*, Proceedings of the International Workshop on Indirect Methods for Estimating the Hydraulic Properties of Unsaturated Soil, Riverside, CA, 1992; van Genuchten, T.M., Leij, F.J., Lund, L.J., Eds.; University of California: Riverside, 1992; 355–358.
17. Salter, P.J.; Berry, G.; Williams, J.B. The influence of texture on the moisture characteristics of soils, III: Quantitative relationships between particle size, composition, and available water capacity. *J. Soil Sci.* **1966**, *17*, 93–98.

18. Pidgeon, J.D. The measurement and prediction of available water capacity of ferrallitic soils in Uganda. *J. Soil Sci.* **1972**, *23*, 431–444.
19. Schroo, H. Soil productivity factors of the soils of the Zandry formation in Surinam. *De Surinaamse Landbouw* **1976**, *24*, 68–84.
20. Karlen, D.L.; Wollenhaupt, N.C.; Erbach, D.C.; Berry, N.C.; Swain, J.B.; Eash, N.S.; Jordahl, J.L. Crop residue effects on soil quality following 10 years of no-till corn. *Soil Till. Res.* **1994**, *31*, 149–167.
21. Rawls, W.L.; Brakensiek, D.L.; Saxton, K.E. Estimation of soil water properties. *Trans. ASAE* **1982**, *25*, 1316–1320.
22. Hollis, J.M.; Jones, R.J.A.; Palmer, R.C. The effects of organic matter and particle size on the water retention properties of some soils in the West Midlands of England. *Geoderma* **1977**, *17*, 225–231.
23. Tran-Vinh-An, Nguba, H. Contribution a l'étude de l'eau utile de quelques sols dur Zaire. *Sols Africana* **1971**, *16*, 91–103.
24. Hamblin, A.P.; Davies, D.B. Influence of organic matter on the physical properties of some east Anglian soils of high silt content. *J. Soil Sci.* **1977**, *28*, 11–22.
25. Salter, P.J.; Haworth, F. The available water capacity of a sandy loam soil, II. The effects of farm yard manure and different primary cultivations. *J. Soil Sci.* **1961**, *12*, 335–342.
26. Schertz, D.L.; Moldenhauer, W.C.; Livingston, S.J.; Weesies, G.A.; Hintz, E.A. Effect of past soil erosion on crop productivity in Indiana. *J. Soil Water Conserv.* **1989**, *44*, 604–608.
27. Schjonning, P.; Christensen, B.T.; Carstensen, B. Physical and chemical properties of a sandy loam receiving animal manure, mineral fertilizer or no fertilizer for 90 years. *Eur. J. Soil Sci.* **1994**, *45*, 257–268.
28. Rawls, W.J.; Pachepsky, Y.A.; Ritchie, J.C.; Sobecki, T.M.; Bloodworth, H. Effect of soil organic carbon on soil water retention. *Geoderma* **2003**, *116*, 61–76.
29. Olness, A.; Archer, D. Effect of organic carbon on available water in soil. *Soil Sci.* **2005**, *170*, 90–101.
30. Mitchell, C.C.; Westerman, R.L.; Brown, J.R.; Peck, T.R. Overview of long-term agronomic research. *Agron. J.* **1991**, *83*, 24–29.
31. Moebius-Clune, B.N.; van Es, H.M.; Idowu, O.J.; Schindelbeck, R.R.; Moebius-Clune, D.J.; Wolfe, D.W.; Abawi, G.S.; Thies, J.E.; Gugino, B.K.; Lucey, R. Long-term effects of harvesting maize stover and tillage on soil quality. *Soil Sci. Soc. Am. J.* **2008**, *72*, 960–969.
32. Bhogal, A.; Nicholson, F.A.; Chambers, B.J. Organic carbon additions: Effects on soil bio-physical and physico-chemical properties. *Eur. J. Soil Sci.* **2009**, *60*, 276–286.
33. Paul, E.A.; Paustian, K.; Elliott, E.T.; Cole, C.V. *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*; CRC Press: Boca Raton, 1997.
34. Mattson, K.G.; Swank, W.T.; Waide, J.B. Decomposition of woody debris in a regenerating, clear-cut forest in the southern Appalachian. *Can. J. For. Res.* **1987**, *17*, 712–721.
35. Rosenberg, N.J.; Izaurrealde, R.C.; Malone, E.L. Carbon sequestration in soils: science, monitoring and beyond. *Conference Proceedings: St. Michaels Workshop, Dec.*; Battell Press: Columbus, 1999.
36. Oulehle, F.; Evans, C.D.; Hofmeister, J.; Krejci, R.; Tahovska, K.; Persson, T.; Cudlin, P.; Hruska, J. Major changes in forest carbon and nitrogen cycling caused by declining sulphur deposition. *Glob. Change Biol.* **2011**, *17*, 3115–3129.
37. Lawrence, G.B.; Shortle, W.C.; David, M.B.; Smith, K.T.; Warby, R.A.F.; Lapenis, A.J. Early indications of soil recovery from acidic deposition in U.S. red spruce forests. *Soil Sci. Soc. Am. J.* **2012**, *76*, 1407–1417.
38. Houghton, J.T.; Ding, Y.; Griggs, D.C.; Noguer, M.; van der Linden, P.J.; Dai, X.; Maskell, K.; Johnson, C.A. *Climate Change 2001: The Scientific Basis*; Cambridge University Press: Cambridge, 2001.
39. Rounsevell, M.D.A.; Evans, S.P.; Bullock, P. Climatic change and agricultural soils: Impacts and adaptation. *Clim. Change* **1999**, *43*, 683–709.
40. Jenkinson, D.S.; Adams, D.E.; Wild, A. Model estimates of CO₂ emissions from soil in response to global warming. *Nature* **1991**, *351*, 304–306.
41. Cox, P.M.; Betts, R.A.; Jones, C.D.; Spall, S.A.; Totterdell, I.J. Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. *Nature* **2000**, *408*, 184–187.
42. Carter, J.O.; Howden, S.M. *Global Change Impacts on the Terrestrial Carbon Cycle*; Working Paper Series 99/10, Report to the Australian Greenhouse Office; CSIRO Wildlife and Ecology: Canberra, 1999.
43. Bellamy, P.H.; Loveland, P.J.; Bradley, R.I.; Lark, R.M.; Kirk, G.J.D. Carbon losses from all soils across England and Wales 1978–2003. *Nature* **2005**, *437*, 245–248.
44. Senthilkumar, S.; Basso, B.; Kravchenko, A.N.; Robertson, G.P. Contemporary evidence of soil carbon loss in the U.S. corn belt. *Soil Sci. Soc. Am. J.* **2009**, *73*, 2078–2086.
45. Nafziger, E.D.; Dunker, R.E. Soil organic carbon trends over 100 years in the morrow plots. *Agron. J.* **2011**, *103*, 261–267.
46. Khan, S.A.; Mulvaney, R.L.; Ellsworth, T.R.; Boast, C.W. The myth of nitrogen fertilization for soil carbon sequestration. *J. Environ. Qual.* **2007**, *36*, 1821–1832.
47. Fantappiè, M.; L'Abate, G.; Costantini, E.A.C. The influence of climate change on the soil organic carbon content in Italy from 1961 to 2008. *Geomorphology* **2011**, *135*, 343–352.
48. Smith, T.M.; Weishampel, J.F.; Shugart, H.H.; Bonan, G.B. The response of terrestrial c storage to climate change – Modeling c-dynamics at varying temporal and spatial scales. *Water Air Soil Poll.* **1992**, *64*, 307–326.
49. Schimel, D.S.; Braswell, B.H.; Holland, E.A.; McKeown, R.; Ojima, D.S.; Painter, T.H.; Parton, W.J.; Townsend, A.R. Climatic, edaphic, and biotic controls over the storage and turnover of carbon in soils. *Global Biogeochem. Cy.* **1994**, *8*, 279–293.
50. Torn, M.S.; Lapenis, A.G.; Timofeev, A.; Fischer, M.L.; Babikov, B.V.; Harden, J.W. Organic carbon and carbon isotopes in modern and 100-year-old-soil archives of the Russian steppe. *Glob. Change Biol.* **2002**, *8*, 941–953.
51. Johnston, A.E.; Poulton, P.R.; Coleman, K. Soil organic matter. Its importance in sustainable agriculture and carbon dioxide fluxes. *Adv. Agron.* **2009**, *101*, 1–57.
52. Conant, R.T.; Ryan, M.G.; Ågren, G.I.; Birge, H.E.; Davidson, E.A.; Eliasson, P.E.; Evans, S.E.; Frey, S.D.; Giardina,

- CP.; Hopkins, F.M.; Hyvönen, R.; Kirschbaum, M.U.F.; Lavallee, J.M.; Leifeld, J.; Parton, W.J.; Megan Steinweg, J.; Wallenstein, M.D.; Martin-Wetterstedt, J.Å.; Bradford, M.A. Temperature and soil organic matter decomposition rates – Synthesis of current knowledge and a way forward. *Glob. Change Biol.* **2011**, *17*, 3392–3404.
53. Hopkins, F.M.; Torn, M.S.; Trumbore, S.E. Warming accelerates decomposition of decades-old carbon in forest soils. *Proc. Nat. Acad. Sci.* 2012. Available at <http://www.pnas.org/cgi/doi/10.1073/pnas.1120603109>.
 54. Phillips, R.P.; Meier, I.C.; Bernhardt, E.S.; Grandy, A.S.; Wickings, K.; Finzi, A.C. Roots and fungi accelerate carbon and nitrogen cycling in forests exposed to elevated CO₂. *Ecol. Lett.* **2012**, *15*, 1042–1049.
 55. Smith, P.; Fang, C.; Dawson, J.J.C.; Moncrieff, J.B.; Donald, L.S. *Impact of Global Warming on Soil Organic Carbon, Advances in Agronomy*; Academic Press: San Diego, 2008; 1–43.
 56. Kirschbaum, M.U.F. The temperature dependence of organic matter decomposition: Seasonal temperature variations turn a sharp short-term temperature response into a more moderate annually averaged response. *Glob. Change Biol.* **2010**, *16*, 2117–2129.
 57. Meehl, G.A.; Stocker, T.F.; Collins, W.D.; Friedlingstein, P.; Gaye, A.T.; Gregory, J.M.; Kitoh, A.; Knutti, R.; Murphy, J.M.; Noda, A.; Raper, S.C.B.; Watterson, I.G.; Weaver, A.J.; Zhao, Z.-C. Global climate projections. In *Climate Change: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*; Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L., Eds.; Cambridge University Press: Cambridge, 2007; 747–845.
 58. Tebaldi, C.; Hayhoe, K.; Arblaster, J.M.; Meehl, G.A. Going to the extremes: An intercomparison of model-simulated historical and future changes in extreme events. *Clim. Change* **2006**, *79*, 185–221.
 59. Chan, K.Y. Soil attributes and soil processes in response to the climate change. In *Soil Health and Climate Change*; Singh, B.P., Allen, D.E., Dalal, R.C., Eds.; Springer: Berlin, 2011; Vol. 29, 49–67.
 60. McCarty, J.J.; Canziani, O.F.; Leary, N.A.; Dikken, D.J.; White, K.S. *IPCC Climate Change 2001: Impacts, Adaptation & Vulnerability*; Cambridge University Press: Cambridge, 2001.
 61. Allen, D.E.; Singh, B.P.; Dalal, R.C.; Cowie, A.L.; Chan, K.Y. Soil health indicators under climate change: A review of current knowledge. In *Soil Health and Climate Change*; Singh, B.P., Allen, D.E., Dalal, R.C., Eds.; Springer: Berlin, 2011; Vol. 29, 25–45.
 62. Pimentel, D.; Harvey, C.; Resosudarmo, P.; Sinclair, K.; Kurz, D.; McNair, M.; Crist, S.; Shpritz, I.; Fitton, L.; Saffouri, R.; Blair, R. Environmental and economic costs of soil erosion and conservation benefits. *Science* **1995**, *267*, 1117–1123.
 63. Huntington, T.G. Climate warming-induced intensification of the hydrologic cycle: A review of the published record and assessment of the potential impacts on agriculture. *Adv. Agron.* **2010**, *109*, 1–53.

Soil Organic Matter (SOM): Characterization

Christian Feller

Institute of Research for Development, Montpellier, France

Abstract

Originally, the meaning of “humus” was “soil” in Latin. By the middle of the 18th century, this original sense survived in the European scientific literature along with the concepts of soil layer (the humus-horizon concept) and soil constituent (the humus-constituent concept). The humus-horizon concept was mainly developed by foresters from the mid-19th century, and it contributed to the emergence of pedology. Humus as a constituent was studied by chemical techniques till the end of 19th century. Soon after the emergence of microbiology, the role of microbial metabolites in humic substance formation was questioned, but it was only in the 1960s that the concept of microbial biomass emerged (Jenkinson, 1966) and developed a lot. At the same time, the era of a physical approach to soil organic matter opened.

INTRODUCTION

Widely different meanings are attached to the term humus, which have persisted through the present time.

The entry “Humus” in the “Encyclopaedia Universalis Duchaufour” (1970) reads as follows:

stricto sensu, “humus” refers to the fraction of SOM which has undergone a more or less rapid transformation of biological as well as physicochemical origin ... lato sensu ... (the humus) ... designates the ensemble of organic horizons. (The original French or German quotations are not given here, but only their English translation.)

Thus, the same word is used to describe two very different concepts: humus as a constituent and humus as a soil horizon. This entry deals with the development of these two concepts over the past three centuries.

All references cited in the text for authors and years can be found in the historical review papers.

THE DIFFERENT MEANINGS OF THE WORD HUMUS

For the Roman writers, humus meant soil or earth.^[1,2] Cicero (106–43 B.C.) replaced the word “humus” by “terra,” sourced from French “terre” (earth), “terreau,” or “terre végétale” (mold), which survived until the 18th century.

The word “humus” re-entered the European scientific vocabulary in the 18th century. It was used by Wallerius in his “Mineralogy” (1753). In Diderot and d'Alembert's Encyclopaedia (1765), humus signifies vegetable mold. But in 1781, in the Abbé Rozier's *Cours complet d'agriculture* (Full Courses in Agriculture), a famous French dictionary of agriculture, humus was largely mentioned in numerous

entries. However, its meaning remained still not precise, denoting “vegetable mold,” “mold,” or “constituent.”

In the first edition (1809, in German) of his *Grundsätze der rationellen Landwirtschaft* (Rational Principles of Agriculture, or in French *Principes raisonnés d'agriculture*, 1811), the German agronomist Thaer, the father of the “humus theory,” gives a very precise definition of humus restricted to the notion of constituent, the properties of which (composition, reactivity, and extractability) were greatly detailed by the author through an approach that was very complete for that time and is still acceptable today.

Through the 18th century, investigations were carried out both on the “humus constituent” (in connection with the rise of chemistry and then microbiology) and, later on, on the “humus horizon” (in connection with forestry research and the emergence of pedology).

HISTORICAL ACCOUNT OF THE HUMUS-HORIZON CONCEPT

Hundeshagen (1830) was the first to introduce a morphological classification of forest humus, quoting two types of humus, followed by Emeis in 1875 (three humus types including the present-day mull).^[1,2]

But it was Müller, in his noteworthy works *The Natural Forms of Humus* (1879, 1884, and 1889), who laid the present-day scientific bases of the study on the different forms of humus, and even of a general survey of soil genetic processes in cold and temperate climates. He defined “Mull,” “Mor,” and “Mullartiger Torf” (equivalent to Moder). His book (1889) can be regarded as a treatise, still valid, on the changes of brown soils to Podzols.^[3] Müller has to be considered as one of the main forerunners of pedology. The other names that should be mentioned here include Ramann (1893), who described the “coarse

humus" (Rohhumus), Henry (1908) for biological processes of litter decomposition, Kubiena (1953) for the morphological approach and classifications, and many others such as Wilde (1954 and 1971), Duchaufour (1956), and Delecour (1980).

Delecour's classification inspired the new French system of soil classification [référentiel pédologique Français (RPF, 1990)], and in soil taxonomy (Soil Survey Staff, 1975), the different types of humus are also taken into consideration while characterizing the "diagnostic horizons," the "epipedons," and, especially, the "organic materials."

HISTORICAL ACCOUNT OF THE "HUMUS-CONSTITUENT" CONCEPT

Three main approaches—namely chemical, microbiological, and physical—corresponding to different concepts on soil organic matter (SOM) pool can be distinguished:

The Chemical Approach

Few natural substances attracted chemists' attention as did humus.^[1,2,4–7] Waksman^[5] evokes the "glossary" and the "chaos" of humic substances.

Achard (1786) was probably the first to initiate a long series of studies on the fractionation of SOM by the extraction of peat with alkali and obtention of a dark amorphous precipitate upon acidification. He was followed by Vauquelin (1797) who discovered a blackish substance in the exudates of the wounded bark of elm (Latin "ulm"), named "ulmin" later. In 1822, Döbereiner created the term "humic," and, as early as 1830, the vocabulary became appreciably enriched.

The hymatomelanic acids (alcohol-soluble SOM) were described by Hoppe-Seyler in 1889. Many others can also be cited here. But the qualifier "fulvic," still in continued use along with the terms "humic" and "humin," did not enter the fractionation vocabulary before 1919 with Oden.

Numerous procedures for extracting humic substances were tested in relation to different objectives.^[6–9] It would appear that there was a need at the international level for standardized extraction procedures and agreement about definitions of humic substances (International Humic Substance Society, 1983).

The Microbiological Approach

In 1877, Schloesing and Müntz, using chloroform as an antiseptic agent, discovered the major role of bacteria in the nitrification process.^[2] The origin of humus was thus questioned: plant and/or microbial? Contribution of microorganism metabolites to humic substance formation is ancient: Braconnot (1834) and Lucas (1841) for fungi^[5] and Kostychev (1886) for bacteria.^[6] Maillard (1913) proposed that microorganisms

only played a role during the stage of decomposition of plant residues and the release of monomers but not during the formation of humic substances that are completely chemical. In contrast and at the same time, working independently, both Dumont (1913) and Trusov (1916) distinguished two sources for humus: 1) the "inherited humus" formed by the plant debris or components that resisted humification and 2) the "microbial humus," which they considered as very active especially in nitrogen dynamics (Dumont's nomenclature).^[6]

Jenkinson (1966) was the first to quantify "microbial biomass" by a simple method using soil fumigation with chloroform. This completely new quantitative concept was ultimately based on the forgotten past works such as those of Müntz (1875) and Schloesing and Müntz (1877) and also based on the works of Wollny (1902), Deherain and Demoussy (1896), and Stoklasa and Ernest (1905).

The Physical Approach

Since 1970, the relevance of pure chemical approaches to SOM for the understanding of the role of SOM in soil properties and functioning has been questioned (Anderson et al., 1983; Tiessen and Stewart, 1983); instead, the use of a different SOM characterization based on particle size and/or density fractionations has been developed.^[2,10] But is the latter concept so recent?^[10]

The first elaborate particle size fractionation of SOM was carried out under suitable experimental conditions by Schloesing in 1874, which attracted much criticism; this work could practically be published. Concerning the process of flocculation, Schloesing questioned the distribution of organic matter (OM) between sand and clay in natural samples and thus set out an SOM fractionation in five particle size fractions as follows: "coarse and fine sands," "scales," and two "clay deposits." One of the Schloesing's conclusions was that:

clay contains 69% organic matter; it is enough to be actually modified in the way it acts as a cement.

Henin and Turc in 1950 proposed a density SOM fractionation using benzene–chloroform mixtures of known densities to separate a "light" fraction ($d < 1.75$), named "free OM," essentially constituted of plant debris from a "heavy" fraction ($d > 1.75$), more humified and named "OM bound to inorganic constituents," and "constituted in particular by the clay–humus complex."

This densitometric approach has essentially been developed between 1950 and 1970 by French scientists (Monnier et al., 1962; Duchaufour and Jacquin, 1966; Dabin, 1971) as a preliminary step in humus fractionation. This approach was further advanced by Greenland and Ford (1964) and Ford et al. (1969) who improved the separation of "free OM" by combining sonication (for disrupting aggregates) with densitometry ($d = 2.0$).

Edwards and Bremner, in 1964 and 1967, have shown the effectiveness of sonication in dispersing the soil in water, with no need for a preliminary oxidation of OM or the addition of a dispersing agent. Particle size fractions can thus be separated together with the associated OM. This work is of cardinal importance as it provides a practical method for locating OM among particle size fractions, with no interfering chemical reaction likely to denature the organic or inorganic constituents. This opened the era of particle size fractionation for SOM studies and bio-organomineral interactions.

Rapidly different forms of energy were combined for the process of dispersion, such as chemical dispersants, sonication, shaking with beads (Bruckert et al., 1978; Balesdent et al., 1991), and using Na resins (Feller et al., 1991; Gavi-nelli et al., 1995).

The combinations of both densitometry and granulometry techniques are often used for the characterization of SOM in particle size fraction of aggregate classes.

CONCLUSION

With the two concepts remaining still, the meaning of the word “humus” is unprecise. One can advise the use of it more in the sense of a horizon concept and keep “SOM” for humus as a constituent. Nevertheless, the qualifier “humic” and its derivatives are always used in the sense of a “constituent,” e.g., “humic substances.” But what is important now is to look at how humus was involved in the history of agronomy and ecology (see *Soil Organic Matter (SOM): Historical Concepts*, p. 2139).

REFERENCES

1. Feller, C.; Boulaine, J. La réapparition du mot humus au XVIIe siècle et sa signification agronomique. *Revue Forestière Française* **1987**, 29 (6), 487–495.
2. Feller, C. The concept of soil humus in the past three centuries. In *History of Soil Science*; Yalon, D.H., Berkowicz, S., Eds.; Advances in GeoEcology: Reiskirchen, 1997; Vol. 29, 15–46.
3. Feller, C.; Blanchart, E.; Jabiol, B.; Greve, M.H. Quand l'humus est à l'origine de la pédologie. 1. Les travaux du forestier danois P.E. Müller (1840–1926). *Etude et Gestion des Sols* **2005**, 12, 101–121.
4. Maillard, L.C. *Genèse des matières protéiques et des matières humiques*; Masson: Paris, 1913; 423 pp.
5. Waksman, S.A. *Humus: Origin, Chemical Composition and Importance in Nature*; Baillière, Tindall and Cox: London, 1936; 494 pp.
6. Kononova, M.M. *Soil Organic Matter: Its Nature, Its Role in Soil Formation and in Soil Fertility*, 2nd Ed.; Pergamon: Oxford, 1961; 505 pp.
7. Vaughan, D.; Ord, B.G. Soil organic matter: A perspective on its nature, extraction, turnover and role in soil fertility. In *Soil Organic Matter and Biological Activity*; Vaughan, D., Malcolm, R.E., Eds.; M. Nijhoff and D.W. Junk Publ: Dordrecht, 1985; 1–35.
8. Schnitzer, M. Humic substances: Chemistry and reactions. In *Soil Organic Matter: Developments in Soil Science*; Schnitzer, M., Khan, S.U., Eds.; Elsevier Press: Oxford, 1978; Vol. 1, 1–16.
9. Stevenson, F.J. *Humus Chemistry: Genesis, Composition, Reactions*; J. Wiley and Sons: New York, 1982.
10. Feller, C. Un fractionnement granulométrique de la matière organique des sols en 1874. *Etude et Gestion des Sols* **1998**, 5, 195–200.

Soil Organic Matter (SOM): Composition and Dynamics

Cornelia Rumpel

M.F. Dignac

National Institute of Agronomic Research (INRA), National Center for Scientific Research (CNRS), Thiverval-Grignon, France

Abad Chabbi

National Institute of Agronomic Research (INRA) and Biodiversity (ACBB), Lusignan, France

Abstract

Molecular studies of soil organic matter are extremely useful to analyze its composition, origin, and dynamics. In this entry, we discuss advances in the field of organic matter studies, which have been achieved by modern spectroscopy and spectrometry including the analysis of stable isotopes. Results suggest that chemical recalcitrance may not be of major importance for carbon stabilization in soil.

INTRODUCTION

Maintaining or increasing organic matter levels in soils is a vital task for good soil management. Soil organic matter (SOM) is a continuum of substances in all stages of decay. Some of these substances may be subjected to stabilization and remain in soil for up to several thousand years. To act on SOM levels and on the amount of easily decomposable material, it is important to know which organic constituents are likely to be stabilized in soil and which are not. Out of the wealth of organic materials entering soils, only a small amount of molecules are ultimately subject to stabilization. However, the identification of these molecules by chemical analyses has not been achieved. In 2001, a consortium of experts on the chemistry of organic matter in the environment stated that more than half of the organic matter in soils remains uncharacterized on a molecular basis despite research efforts lasting for several decades.^[1]

Before the introduction of analytical methods that can be used to characterize solid samples, only the soluble part of the organic matter could be subjected to detailed chemical analysis. Solubilization in alkaline media might introduce artifacts and does not give access to all the organic matters present in soil. Therefore, studies based on the chemical solubility of soil organic constituents might give misleading results. Furthermore, such results are not related to the biological functioning of soils or to the dynamics of their organic constituents. With the application of spectroscopic techniques to soil science, complete information on the bulk chemical composition of SOM could be obtained. However, spectroscopic methods are insensitive, while providing limited information on functional groups

present in SOM. More detailed molecular-level information may be obtained by wet chemical analysis and pyrolysis coupled to gas chromatography and mass spectrometry. Substantial progress in the understanding of the relationship between SOM composition and its dynamics was made when these molecular-level techniques were coupled to isotope quantification [of carbon-13 (^{13}C) and carbon-14 (^{14}C)].

In this entry, we present a state-of-the-art discussion on the milestones and the results obtained in SOM research by using molecular studies. Molecular-level information greatly improved our understanding of SOM composition in relation to its origin, function, and dynamics (Fig. 1). We present information obtained from spectroscopic approaches, analytical pyrolysis, wet chemical analysis, and isotopic methods, concerning the bulk chemical composition of SOM and its dynamics in view of SOM function as long-term C stock.

BULK CHEMICAL COMPOSITION OF SOM

According to molecular analysis of humic substances, extracted from soil with sodium hydroxide, SOM was thought to contain a complex mixture of highly aromatic macromolecules, which are resistant to decomposition owing to their chemical nature. With the introduction of solid-state ^{13}C nuclear magnetic resonance spectroscopy, the paradigm of increasing aromaticity during the formation of SOM was challenged. Studies on organic forest floor horizons indicated that the contribution of aromatic C remained constant throughout the soil profile.^[2] The high contribution of aromatic compounds to organic matter of

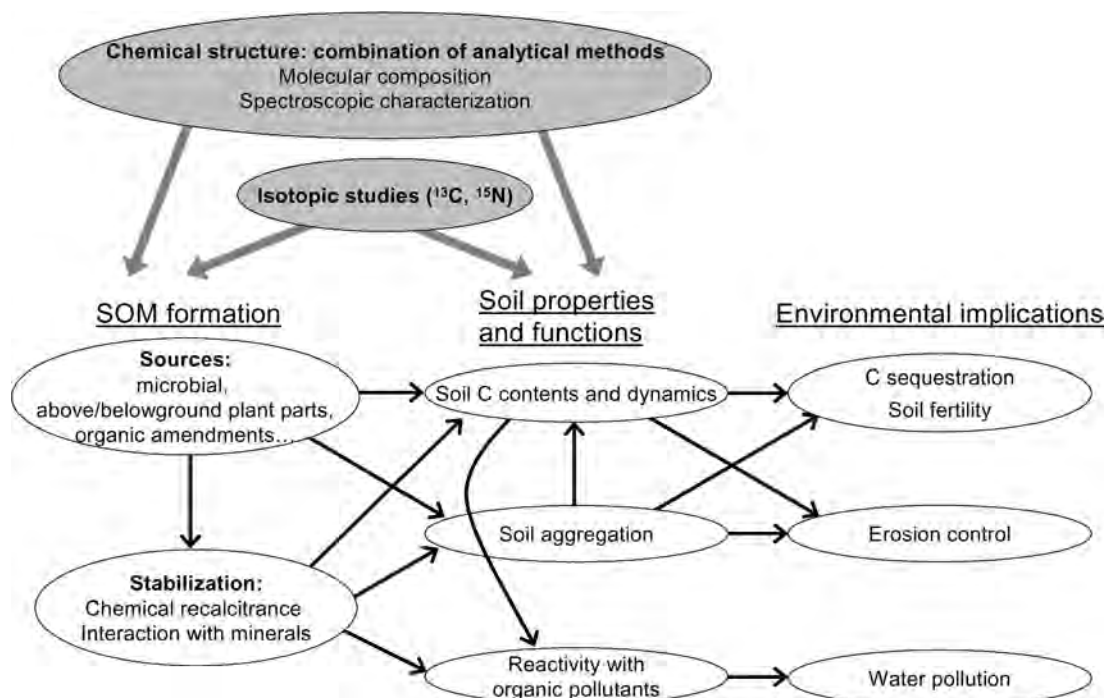


Fig. 1 A combination of methods (chemical characterization and isotopic studies) is necessary to understand the complex properties and functions of SOM, which act at the molecular level and may have major environment implications.

some soils was related to the presence of substantial amounts of charcoal produced by vegetation fires.^[3] Nitrogen (N), which had long been suspected to be of stable heterocyclic nature in soils, was shown to be mainly amide N, derived most probably from proteins.^[4] With these results, the nature of organic matter as a mixture of macromolecular humic substances was questioned. It was shown that instead of macromolecules, substances with lower molecular weight are more likely to survive in soil. For example, studies with analytical pyrolysis showed that polysaccharides and proteins may have a much longer residence time in soils than molecules derived from lignin or other macromolecular substances.^[5] From these findings, a new theory emerged after which the microbial loop could possibly be crucial for organic matter stabilization in soil. Under these conditions, the origin of stabilized C may be seen in a new light. It is probably not the precursor substance as itself, which is preserved in soil, but its constituting C, after integration into new (microbial-derived) molecules.

In soils on which no anthropogenic materials, such as compost or sewage sludge, are added, the ultimate source of SOM is plant material. Plant residues consist of above and belowground material. For the management of plant litter input, the above or belowground origins of soil C are of vital importance. However, this question has rarely been addressed, as the quantification of litter input from roots is extremely difficult. The few studies that exist provide evidence that root-derived C may be present in soil in larger amounts than shoot-derived C.^[3,6–8] Molecular studies taking advantage of cutin and suberin as aliphatic

biomarkers of root- and shoot-derived C are of great use to trace the origin of soil organic C.

DYNAMICS OF SOM

The molecular dynamics of SOM is essentially related to the decomposition and incorporation of its precursors constituting plant material. As such, the dynamics in soil of the main plant litter compounds, lignin, carbohydrates, and lipids has been estimated. For such investigations, coupling of molecular and isotopic analysis is necessary. The most suitable technique is the analysis of the stable C isotope ratio of single molecules after separation by gas chromatography. With this technique, the incorporation of the precursor molecules into SOM may be studied on sites, which were subjected to a C3–C4 vegetation change. To carry out a kinetic study, suitable field trials are necessary. Using this technique, Dignac et al.^[9] were able to show that lignin, which is degraded slowly in the first stages of litter decomposition, turns over in soil at a much higher rate than total C. Along with similar results obtained by Wiesenberger et al.^[8] for the lipid fraction, these findings question the generally admitted idea that aromatic and aliphatic substances are the main precursors of SOM. By contrast, carbohydrates, which were considered among the most labile SOM compounds, were found to turn over at a similar rate as total soil C in some physical fractions of soils.^[10] These results are likely to change our understanding of SOM formation. To this end, it was suggested that microbial

C may be an important factor for the stabilization of SOM. Investigations on the dynamics of microbial C using ^{14}C activity measurements on a molecular level showed that it may be very old.^[7] However, the age of microbial C is related to the age of the substance the microorganisms feed on. The very old age of some microorganisms was related to their capability to degrade ^{14}C -free lignite or coal present in some soils. All these molecular studies of SOM dynamics tend to suggest that inherent chemical recalcitrance of SOM precursors is not responsible for their long-term stabilization in soils.

CONCLUSION

SOM has attracted the attention of soil scientists for its importance for soil quality and C sequestration. Studies of the composition of SOM have evolved during the last decades from solubilization techniques, poorly related to the functions and dynamics of organic constituents, to spectroscopic and molecular-level techniques. Molecular-level studies led to great progress in the understanding of the chemical composition of SOM. Coupling molecular with isotopic techniques induced essential progress in understanding the dynamics of plant litter compounds. Molecular dynamic studies suggested that chemical recalcitrance of plant-derived molecules might not be the determinant factor for their preservation in soils.

REFERENCES

- Hedges, J.I.; Eglinton, G.; Hatcher, P.G.; Kirchman, D.L.; Arnosti, C.; Derenne, S.; Evershed, R.P.; Kögel-Knabner, I.; de Leeuw, J.W.; Littke, R.; Michaelis, W.; Rullkötter, J. The molecularly-uncharacterized component of nonliving organic matter in natural environments. *Org. Geochem.* **2000**, *31*, 945–958.
- Kögel-Knabner, I. Biodegradation and humification processes in forest soils. In *Soil Biochemistry*; Bollag, J.-M., Stotzky, G., Eds.; Marcel Dekker: New York, 1993; Vol. 8, 101–137.
- Skjemstad, J.O.; Clarke, P.; Taylor, J.A.; Oades, J.M.; McClure, S.G. The chemistry and nature of protected carbon in soil. *Aust. J. Soil Res.* **1996**, *34* (25), 251–271.
- Knicker, H. Biogenic nitrogen in soils as revealed by solid-state carbon-13 and nitrogen-15 nuclear magnetic resonance spectroscopy. *J. Environ. Qual.* **2000**, *29*, 715–723.
- Gleixner, G.; Bol, R.; Balesdent, J. Molecular insight into soil carbon turnover. *Rapid Commun. Mass Sp.* **1999**, *13*, 1278–1283.
- Rasse, D.P.; Rumpel, C.; Dignac, M.F. Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. *Plant Soil* **2005**, *269*, 341–356.
- Rethemeyer, J.; Grootes, P.M.; Bruhn, F.; Andersen, N.; Nadeau, M.J.; Kramer, C.; Gleixner, G. Age heterogeneity of soil organic matter. *Nucl. Instrum. Meth. B* **2004**, *223–224*, 521–527.
- Wiesenberg, G.L.B.; Schwarzbauer, J.; Schmidt, M.W.I.; Schwark, L. Sources and turnover of organic matter in agricultural soils derived from n-alkane/n-carboxylic acid compositions and C-isotope signatures. *Org. Geochem.* **2004**, *35*, 1371–1393.
- Dignac, M.-F.; Bahri, H.; Rumpel, C.; Rasse, D.P.; Bardoux, G.; Balesdent, J.; Girardin, C.; Chenu, C.; Mariotti, A. Carbon-13 natural abundance as a tool to study the dynamics of lignin monomers in soil: An appraisal at the Closeaux experimental field (France). *Geoderma* **2005**, *128*, 1–17.
- Derrien, D.; Marol, C.; Balesdent, M.; Balesdent, J. The turnover of carbohydrates in a cultivated soil estimated by ^{13}C natural abundances. *Eur. J. Soil Sci.* **2006**, *57*, 547–557.

Soil Organic Matter (SOM): Global Carbon Cycle

Keith Paustian

Natural Resources Ecology Lab, Colorado State University, Fort Collins, Colorado, U.S.A.

Abstract

Soil organic matter (SOM) comprises an integral component of the global carbon (C) cycle, both as the largest overall repository of C within the terrestrial system and as a major source and sink for C exchanges between the atmosphere, terrestrial vegetation, and aquatic environments. Consequently, SOM plays a significant role in regulating the composition of the atmosphere, particularly with respect to carbon dioxide (CO₂). Thus, there is both concern that the effects of climate and land use change on soils will exacerbate the problem of increasing CO₂ in the atmosphere and hope that through better management, soils can play a part in mitigating increasing CO₂ levels.

CARBON COMPONENTS OF SOIL

Carbon (C) is present in both inorganic and organic forms in soil. With the exception of some soils formed on carbonate-rich parent material or arid soils containing high levels of primary or secondary carbonates, organic forms of C predominate in soil. These organic compounds range from fresh plant residues—which are the primary source of soil organic matter (SOM)—to highly recalcitrant, amorphous humic substances. Plant residues are broken down by the soil biota, chiefly microorganisms, to derive energy (i.e., respiration) and C and nutrient elements needed to grow and reproduce. The soil biota comprises a complex food web in which microbial-, plant-, and animal-derived materials are continually consumed, decomposed, and reformed as soil biomass and other secondary compounds. Plant compounds that are not fully metabolized by the biota, e.g., lignin derivatives, microbial metabolites, and other organic substances, can also recombine through chemical and physical reactions. Thus, the decomposition process can be viewed as a “cascade” of biophysicochemical reactions and products,^[1] resulting in the loss of C via respiration, together with the formation of a wide array of secondary SOM compounds including complex recalcitrant substances that can persist in soil for several hundreds to thousands of years.

C STOCKS IN SOIL

C comprises a relatively minor component of most soils, in terms of mass. Most soils contain from 1% to 10% C by mass in surface horizons, with the majority having C contents in the range of 1–3%. The most notable exceptions are soils formed under waterlogged conditions, which restrict the flow of oxygen to soil organisms, greatly reducing the

rates of SOM decay. Such organic soils or “Histosols” may contain 10–30% or more of their total dry mass as C. Despite this generally low concentration, on an area basis, the C contained in soils usually exceeds that contained in the living and dead vegetation (Table 1). Estimates of the global C stock in soils vary, although estimates are on the order of 1,400–1,600 Pg (petagram = 10¹⁵ g = billion metric tons) organic C^[3,4] and 700–900 Pg inorganic C.^[5] Concentrations of organic C are highest in wetlands and in cold, wet environments (e.g., boreal forest) where decomposition rates are suppressed and are lowest in desert and tundra soils where plant productivity, and hence C additions to soil, is low. Stocks of inorganic C are greatest in desert and semiarid to subhumid grasslands and savannas, where carbonate leaching is restricted and the formation of secondary carbonate minerals is favored.

C FLUXES

The carbon dioxide (CO₂) exchanges between the atmosphere, vegetation, and soils are among the largest annual fluxes in the global C cycle (Fig. 1), which are clearly detectable in the regular, seasonal variations in atmospheric CO₂ concentrations. Global estimates of net primary production expressed as C assimilated by live plants (minus respiration) are on the order of 60 Pg C/yr.^[2] Much of the annually produced biomass is added to the soil and soil surface each year as senesced leaves, roots, and woody debris. Emissions of CO₂ from soils through decomposition processes (via heterotrophic respiration) are thought to be on the order of 50 Pg/yr with additional losses of about 10 Pg/yr from disturbances such as fire.^[2] If terrestrial biome C stocks were at equilibrium, then additions of C through net primary production would

Table 1 Global C stocks in soil and vegetation by major biome types.

Biome	Area (10 ⁶ km ²)	Average C density (Mg ha ⁻¹)	Soil C stock (Pg)	Vegetation C stock (Pg)
Tropical forests	17.6	123	216	212
Temperate forests	10.4	96	100	59
Boreal forests	13.7	344	471	88
Tropical savannas	22.5	117	264	66
Temperate grasslands	12.5	236	295	9
Deserts and semideserts	45.5	42	191	8
Tundra	9.5	13	121	6
Wetlands	3.5	642	225	15
Croplands	16.0	80	128	3
Total	151.2	—	2011	466

Source: Adapted from IPCC.^[2] ©2000 Cambridge University Press.

be fully balanced by losses through decomposition and other emission sources such as fire. However, overall estimates of the global C cycle suggest that a net accumulation of C presently occurs in terrestrial vegetation and soils, at a rate of about 2 Pg C/yr (Table 2). During the 1980s, tropical regions are believed to have been a net source of C emissions, largely through deforestation. Similar global estimates for the 1990s have not been reported, but if the tropics remain a net source of C from

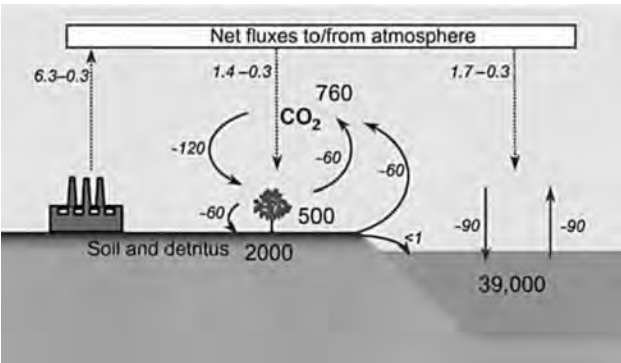


Fig. 1 Simplified depiction of the global C cycle. Total stocks of C in the atmosphere (as CO₂), oceans, vegetation, and soils + detritus are shown in bold and approximate annual gross fluxes between the major biosphere pools are shown in italics. Dashed arrows denote direction and magnitude of the net flux of CO₂ from industrial sources (mainly fossil fuel combustion) and net sinks to terrestrial and marine environments, estimated as averages for 1990–1999. All stock values are in units of Pg (i.e., billion metric tons) and fluxes are Pg/yr.

Source: Adapted from IPCC^[2] ©2000 Cambridge University Press, Schlesinger^[6] ©1995, and IPCC^[7] ©2001 Cambridge University Press.

deforestation, then the size of the terrestrial sink outside the tropics would be correspondingly greater. In the global budget, the terrestrial sink term is calculated as a difference between the estimates of fossil C emissions, ocean uptake, and observed increases in atmospheric CO₂. The existence and relative size of the inferred terrestrial sink are broadly consistent with estimates of a northern hemisphere terrestrial sink, based on atmospheric transport models^[8]—although considerable uncertainty as to the magnitude and geographic distribution of the sink remains.^[9]

SOIL C FLUXES AND GLOBAL CHANGE

While the net terrestrial accumulation of C is small relative to the annual fluxes of C between the atmosphere and land, it is extremely significant in relation to the net change in CO₂ in the atmosphere. In other words, if the terrestrial C sink was to disappear, and everything else remained equal, the rate of growth of CO₂ in the atmosphere would increase by more than 50%, from about 3.2 to over 5 Pg/yr (Table 2).

The origin of and controls on this terrestrial C sink are not well understood. Model estimates^[10] attribute much of the sink to increased plant growth rates due to higher CO₂ concentrations, although other factors such as nitrogen deposition and changing land use contribute as well. The contribution of soils to the overall C sink is uncertain although it is hypothesized that most of the C accumulates as biomass and surface litter pools. In part, this is to be expected due to the lag time between C accumulation in biomass and fresh residue pools, and its subsequent appearance in more stabilized SOM pools, as well as the fact that much of the C added to soils is relatively rapidly decomposed and evolved as CO₂. Global accumulation of refractory humic compounds in mature native ecosystems has been estimated at 0.4 Pg/yr and it has been suggested that these increases roughly balance leaching losses and long-term average organic C transfers to oceans.^[11] Soil C increases attributed to global afforestation and forest

Table 2 Major net sources and sinks of C in the global budget for the 1980s and 1990s.

	1980–1989	1990–1999
Atmospheric increase in CO ₂ –C	3.3 ± 0.1	3.2 ± 0.1
Emission of C from fossil fuel and cement	5.4 ± 0.3	6.3 ± 0.3
Ocean–atmosphere flux	–1.9 ± 0.6	–1.7 ± 0.3
Land–atmosphere flux	–0.2 ± 0.7	–1.4 ± 0.3
Net from land use change	1.7 (0.6–2.5)	NA
Net from other terrestrial sinks	–1.9 (–3.8–0.3)	NA

Note: NA = not available.

Source: Adapted from IPCC.^[7] ©2001 Cambridge University Press.

regrowth, since the 1950s are estimated to be on the order of 0.1 Pg/yr.^[12] Global estimates are lacking for other ecosystems, but inventories for the United States^[13] suggest that soils in cropland and grazing land represent a small sink, on the order of 10–30 Tg/yr (teragram = 10^{12} g = million metric tons), which can be compared to estimates for C increases in U.S. forest biomass of about 210 Tg/yr.^[14]

The potential feedbacks of climate change on soil C stocks are being debated. Since both plant production (hence, C inputs) and decomposition (hence, C losses) are affected by changes in temperature, precipitation, and CO₂ concentrations, the interactions and feedbacks controlling the terrestrial C balance are complex and difficult to predict. Earlier estimates, assuming a stimulation of decomposition due to projected increases in temperature, suggested that soils could become a significant net source of CO₂, on the order of 40–60 Pg over a 50–100-year period, under a climate regime predicted for double CO₂ concentrations.^[6,15] Other studies suggest that including positive effects of CO₂ on plant productivity, management, and together with adaptations in land might largely offset climate-driven increases in decomposition potential.^[16,17] Debates^[18–20] have highlighted the complexity and continuing uncertainty of how temperature increase, and other changes in climate and CO₂, may impact soils and the global C cycle.

The other significant driver of soil C changes is land use and management. It is well established that the conversion of native ecosystems (e.g., forests, grasslands, and wetlands), primarily to agricultural uses, has led to significant losses of C from terrestrial ecosystems, on the order of 120–170 Pg C or more over the past 150–300 years from vegetation and soils combined.^[21] From soils alone, estimates of historical losses over the same period are 50–100 PgC.^[22,23] However, this legacy of land use offers an opportunity to reverse the historical trend and through better management of soils, exploit their potential to become a significant sink for CO₂. A variety of management options are available to increase soil C storage in agricultural^[24] and other managed ecosystems.^[2] Estimates suggest that potential C sequestration through improvements in land use and management, globally, is on the order of 1 Pg/yr.^[2] Thus the role of SOM in the global C cycle, and how it will respond to changes in climate and land use, will be determined by both the natural forces regulating the earth's biosphere as well as the social, economic, and political actions of human kind.

REFERENCES

1. Swift, M.J.; Heal, O.W.; Anderson, J.M. *Decomposition in Terrestrial Ecosystems*; Blackwell Press: Oxford, 1979; 372 pp.
2. IPCC. *Land Use, Land Use Change, and Forestry*. Intergovernmental Panel on Climate Change Special Report; Cambridge University Press: Cambridge, 2000; 377 pp.
3. Post, W.M.; Emanuel, W.R.; Zinke, P.J.; Stangenberger, A.G. Soil carbon pools and world life zones. *Nature* **1982**, *298*, 156–159.
4. Eswaran, H.; Van Den Berg, E.; Reich, P. Organic carbon in soils of the world. *Soil Sci. Soc. Am. J.* **1993**, *57*, 192–194.
5. Eswaran, H.; Reich, P.F.; Kimble, J.M.; Beirnoth, F.H.; Padmanabhan, E.; Moncharoen, P. Global carbon stocks. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2000; 15–25.
6. Schlesinger, W.H. An overview of the carbon cycle. In *Advances in Soil Science: Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1995; 9–25.
7. IPCC. *Climate Change 2001: The Scientific Basis, Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change (IPCC)*; Houghton, J.T., Yihui, D., Eds.; Cambridge University Press: Cambridge, UK.
8. Tans, P.P.; Fung, I.Y.; Takahashi, T. Observational constraints on the global atmospheric carbon dioxide budget. *Science* **1990**, *247*, 1431–1438.
9. Field, C.B.; Fung, I.Y. The not-so-big US carbon sink. *Science* **1999**, *285*, 544–545.
10. McGuire, A.D.; Sitch, S.; Clein, J.S.; Dargaville, R.; Esser, G.; Foley, J.; Heimann, M.; Joos, F.; Kaplan, J.; Kicklighter, D.W.; Meier, R.A.; Melillo, J.M.; Moore, B., III; Prentice, I.C.; Ramankutty, N.; Reichenau, T.; Schloss, A.; Tian, H.; Williams, L.J.; Wittenberg, U. Carbon balance of the terrestrial biosphere in the twentieth century: Analysis of CO₂, climate and land use effects with four process-based ecosystem models. *Glob. Biogeochem. Cycles* **2001**, *15*, 183–206.
11. Schlesinger, W.H. Evidence from chronosequence studies for a low carbon-storage potential of soils. *Nature* **1990**, *348*, 232–234.
12. Post, W.M.; Kwon, K.C. Soil carbon sequestration and land-use change: Processes and potential. *Glob. Change Biol.* **2000**, *6*, 317–327.
13. Eve, M.D.; Paustian, K.; Follett, R.; Elliott, E.T. A national inventory of changes in soil carbon from national resources inventory data. In *Methods of Assessment of Soil Carbon*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2001; 593–610.
14. US Environmental Protection Agency. Inventory of US Greenhouse Gas Emissions and Sinks: 1990–1998. *US EPA 236-R-00-001*; US EPA: Washington, DC, 2000.
15. Jenkinson, D.S.; Adams, D.E.; Wild, A. Model estimates of CO₂ emissions from soil in response to global warming. *Nature* **1991**, *351*, 304–306.
16. Prentice, K.C.; Fung, I.Y. The sensitivity of terrestrial carbon storage to climate change. *Nature* **1990**, *346*, 48–51.
17. Paustian, K.; Elliott, E.T.; Peterson, G.A.; Killian, K. Modelling climate, CO₂ and management impacts on soil carbon in semi-arid agroecosystems. *Plant Soil* **1996**, *187*, 351–365.
18. Giardina, C.P.; Ryan, M.G. Evidence that decomposition rates of organic carbon in forest mineral soil do not vary with temperature. *Nature* **2000**, *404*, 858–861.

19. Davidson, E.A.; Trumbore, S.E.; Amundson, R. Soil warming and organic carbon content. *Nature* **2000**, *408*, 789–790.
20. Thornley, J.H.M.; Cannell, M.G.R. Soil carbon storage response to temperature: A hypothesis. *Ann. Bot.* **2001**, *87*, 591–598.
21. Houghton, R.A. The annual net flux of carbon to the atmosphere from changes in land use 1850–1990. *Tellus* **1999**, *51B*, 298–313.
22. IPCC. Agricultural options for mitigation of greenhouse gas emission. In *Climate Change 1995: Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses; Intergovernmental Panel on Climate Change (IPCC) Working Group II*; Cambridge University Press: Cambridge, UK, 1996; 745–771.
23. Lal, R. Soil Management and restoration for carbon sequestration to mitigate the accelerated greenhouse effect. *Prog. Environ. Sci.* **1999**, *1*, 307–326.
24. Paustian, K.; Collins, H.P.; Paul, E.A. Management controls on soil carbon. In *Soil Organic Matter in Temperate Agroecosystems; Long-Term Experiments of North America*; Paul, E.A., Paustian, K., Elliott, E.T., Cole, C.V., Eds.; CRC/Lewis Publishers: Boca Raton, 1997; 15–49.

Soil Organic Matter (SOM): Global Distribution

Wilfred M. Post

Oak Ridge National Laboratory, Oak Ridge, Tennessee, U.S.A.

Abstract

Over long periods of time, organic matter in soils is the result of climatic, biological, and geological factors. These factors are not independent. Over shorter periods of time, soil carbon varies with vegetation disturbances and changes in land use patterns that affect rates of organic matter input and its decomposition. Evidence from long-term experiments suggests that carbon losses due to oxidation and erosion can be reversed with soil management practices that minimize soil disturbance and optimize plant yield through fertilization.

INTRODUCTION

Globally, the amount of organic matter in soils, commonly represented by the mass of carbon (C), is estimated to be 1200–1500 Pg C (1 Pg C = 10^{15} g C) in the top 1 m of soil.^[1,2] This is 2 to 3 times larger than the amount of organic matter in living organisms in all terrestrial ecosystems.^[1] The exact ratio between living and dead organic matter in terrestrial ecosystems varies, depending on the ecosystem. The amount of C stored in soil is determined by the balance of two biotic processes—the productivity of terrestrial vegetation and the decomposition of organic matter. Each of these processes has strong physical and biological controlling factors. These include climate; soil chemical, physical, and biological properties; and vegetation composition. Interactions among these controlling factors are of particular importance. These biological and physical factors are the same as the ones that influence the aboveground structure and composition of terrestrial ecosystems, so there are strong correspondences between soil organic matter content and ecosystem type.

ORGANIC MATTER INPUTS

Quantity

The amount of C stored in soils is to a great extent determined by the rate of organic matter input through litterfall, root exudates, and root turnover. The main factors that influence vegetation production are suitable temperatures for photosynthesis, available soil moisture for evapotranspiration (ET), and rates of carbon dioxide (CO₂) and water exchange. Dry and/or cold climates support low vegetation production rates and soils under such climates have low organic matter contents.

Where climates are warm and moist, vegetation production is high and soil organic matter contents are correspondingly high. Fig. 1 shows the striking correspondence between soil organic matter content and general climate measurements, which results from the relationship between vegetation production and suitable moisture and temperature conditions.

Vegetation production depends not only on climate but also on nutrient supply from decomposition and geochemical weathering. Walker and Adams^[6] hypothesized that the level of available phosphorus during the course of soil development is the primary determinant of terrestrial net primary production. Numerous workers have examined this hypothesis. Tiessen, Stewart, and Cole^[7] and Roberts, Stewart, and Bettany^[8] found that available phosphorus explained about one-fourth of the variance in soil organic matter in many different soil orders. The relationship between phosphorus and C is strongest during the aggrading stage of vegetation–soil system development.^[9] Initially, the production of acidic products by pioneer vegetation promotes the release of phosphorus by weathering of parent material. Organic matter builds up in the soil, increasing the storage of phosphorus in decomposing organic compounds. Nitrogen (N)-fixing bacteria populations, which depend on a supply of organic C and available phosphorus, can grow to meet ecosystem demands for N. Plant growth is enhanced by this increasing N and phosphorus cycling, resulting in increased rates of weathering. This process continues until the vegetation is constrained by other factors affecting phosphorus availability: Leaching losses become larger than the weathering inputs;^[10] an increasing fraction of the phosphorus becomes unavailable by adsorption or precipitation with secondary minerals;^[11] or N availability (denitrification or leaching is affected) reaching or exceeding N inputs and fixation.^[12] In mature soils, net primary production is more likely to be limited by N. Availability of other

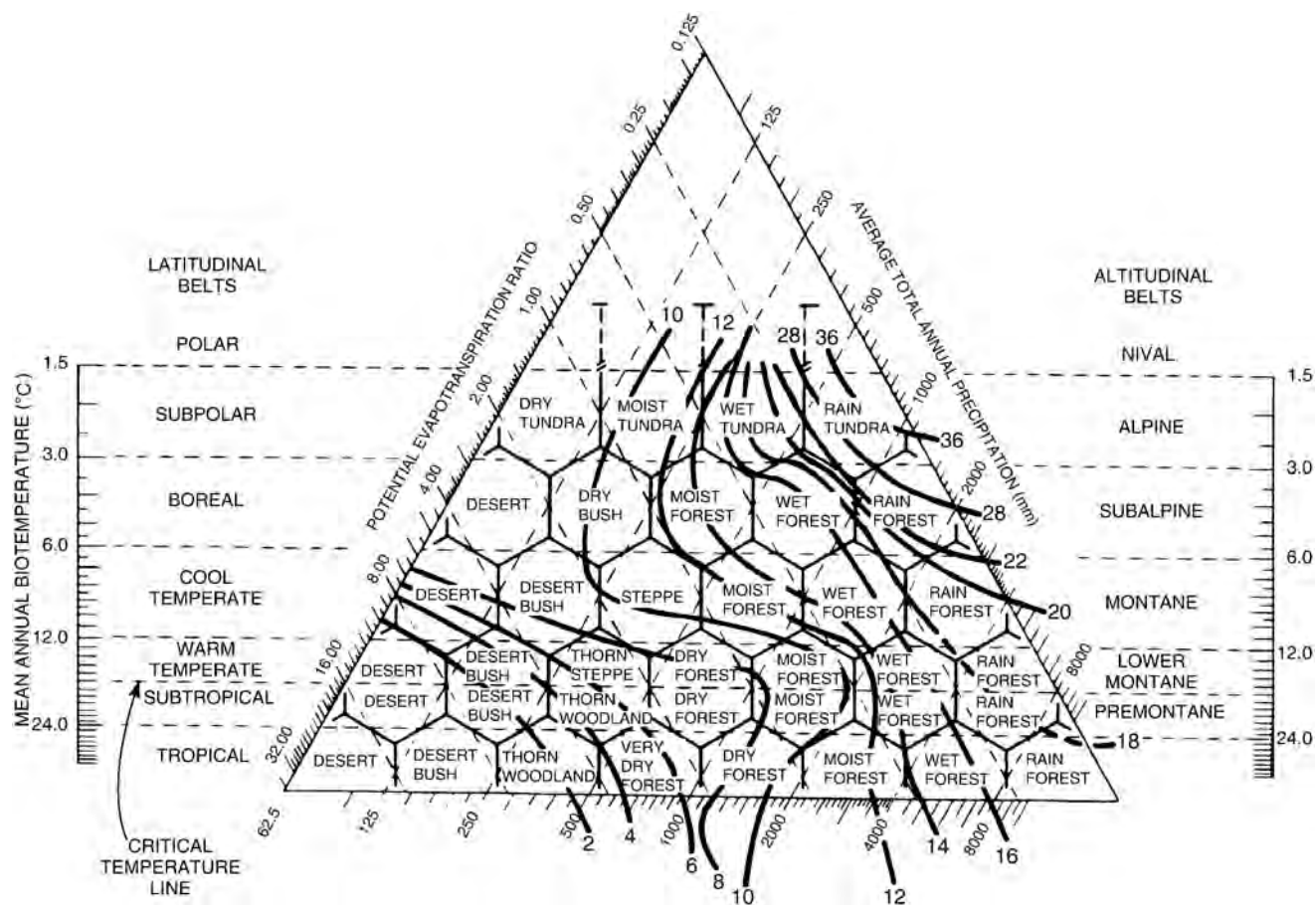


Fig. 1 Contours of soil C density (kg m^{-2}) plotted on Holdridge diagram for world life zone classification. Values of biotemperature and precipitation uniquely determine a life zone and associated vegetation. Contour lines for mean soil C content in the surface meter of soil are determined from data derived from over 3000 soil profiles.

Source: From Holdridge,^[3] Zinke, Stangenberger, et al.,^[4] and Post, Pastor, et al.^[5]

nutrients that are largely derived from parent materials, such as most base cations, may also influence soil organic matter accumulation during early soil development.^[13] Soils derived from base cation-rich volcanic parent materials (Andisols) have much higher C contents on average than soils from other parent materials.^[4]

Species Composition

Biotic factors, in particular plant species composition, also affect soil organic matter dynamics. Production and decomposition rates are to some degree controlled by species composition. Each terrestrial plant species produces different amounts and chemical compositions of leaves, roots, branches, and wood of varying decomposability. This range of decomposability may be summarized by the lignin and N content of the organic material.^[14,15] Litter decay rate is inversely related to C/N and lignin/N ratios and positively related to N content. Species with tissues that have low nutrient or high lignin content produce litter that is slow to decay. N is made available to plants during

the decomposition process. N is a limiting element for productivity in most terrestrial ecosystems so the rate at which it is released during decomposition is an important factor in ecosystem production. Thus, the interactions between processes regulating plant populations and their productivity and microbial processes regulating N availability result in some of the observed variation in soil C and N storage.^[16–19]

Placement

The deeper the fresh detritus is placed in the soil, the slower it decomposes. This is a result of declining decomposer activity and increased protection from oxidation with depth in the soil. Prairies have a somewhat lower productivity than forests and produce no slowly decomposing woody material. Nevertheless, prairies have a very high soil organic matter content because prairie grasses allocate twice as much production to belowground roots and tillers than to aboveground leaves.^[20] The result is high soil organic matter contents with a uniform distribution in the

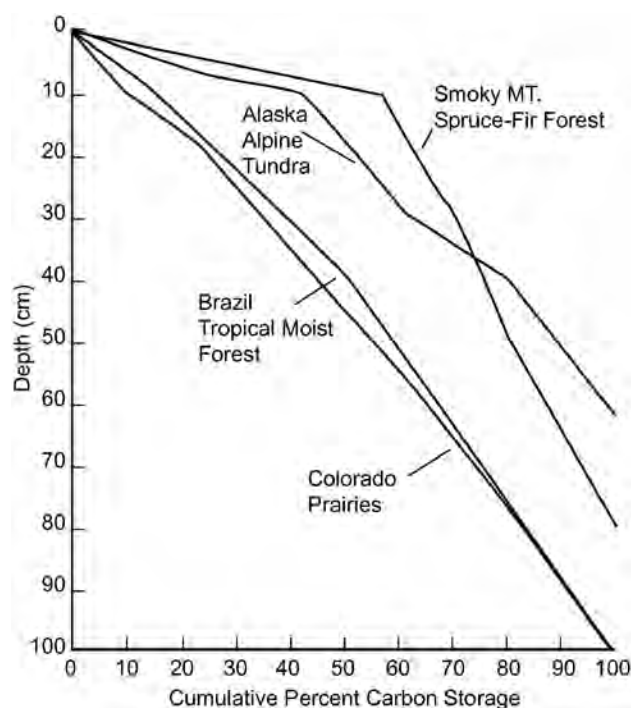


Fig. 2 Cumulative C storage as a function of depth for four ecosystems. Refer text for the explanation of these patterns.

Source: From Zinke, Stangenberger, et al.^[4]

upper 1 m of soil (Fig. 2). In contrast, a spruce–fir forest contains 50 percent of its soil organic matter in the top 10 cm. There are interesting exceptions to the rule that above-/belowground plant allocation determines soil organic matter distribution patterns in soil. Tropical moist forest soils show a uniform depth distribution similar to the depth distribution of temperate grasslands; however, in tropical forests, this is largely due to a long-term accumulation of recalcitrant organic materials at lower depths in the soil rather than increased allocation to roots. Alpine tundra soils support a largely herbaceous flora but show a similar depth distribution as forest soils because of inhibition of surface litter decomposition by low temperatures and high water saturation.

DECOMPOSITION

Climate

Organic matter decay rates can be related to environmental parameters such as temperature and soil moisture. Climatic indices that correlate well with decay rates include plant moisture and temperature indices,^[21,22] linear combinations of temperature and rainfall,^[23] and actual ET.^[15] Warm temperatures and available soil moisture enhance microbial and micro- and macro-invertebrate activity. These environmental

conditions are also correlated with plant production. As a result, the amount of organic matter present in soil is highest in vegetation types with the highest rates of organic matter production. These are ones found in the warm, moist climate regions. The contours of soil C density displayed in Fig. 1 reflect the balance of input by vegetation production and loss from decomposition imposed by climate. Soil C content increases from lower left to the upper right in Fig. 1 as the temperature decreases in the cool temperate, boreal, and subpolar life zones and as precipitation increases in the warm temperate, subtropical, and tropical life zones.

The combined influence of temperature and precipitation is presented by the third axis of the Holdridge diagram (Fig. 1) as the ratio of potential ET (PET) to annual precipitation. When this ratio is less than 1.0, rainfall exceeds PET and vice versa. Life zones bordering the line with the PET is equal to precipitation (PET ratio = 1.0) have soil C contents around 10 kg m^{-2} except in warm temperate and subtropical zones where strong seasonality limits production, but decomposition conditions are favorable for most of the year. Soil C content increases as the PET ratio decreases, indicating that productivity increases faster than the rate of decomposition with increasing moisture availability.

Organic Matter Quality

On global scale, climate may be the most important factor controlling decay rates, but within a given region, substrate chemistry is the more important factor.^[15,24,25] Decay rate is often negatively related to substrate C/N ratio. Litter C/N is initially much greater than microbial C/N but approaches microbial C/N as the microbes release the C as CO_2 while taking up N (N immobilization). The further the initial litter C/N is from microbial C/N, the slower the decay rate. Lignin content or lignin/N ratios may be better predictors of decay rates because lignin itself is difficult to decompose, and it shields N and other more easily degraded chemical fractions from microbes. Concise and simple models of decay rate are based on a combination of chemical and climatic indices.

The effect of litter quality on soil organic matter content is most dramatically expressed in Podzols (Spodosol in the U.S. Department of Agriculture classification). These occur over large areas in boreal zones dominated by evergreen conifers, but often occur in other regions on shallow or sandy soils. Low N content of organic matter inputs and cool temperatures reduce decomposition and soil animal activity. As a result, large surface organic matter accumulations occur over a thin A horizon. Low temperatures combined with leaching of organic acids result in podsolization as the predominant soil-forming process. Leaching of iron, aluminum oxides, and organic matter results in a distinct E horizon near the surface where these materials are removed and deposited in the

Table 1 Mean organic C contents (kg m^{-2}) by Food and Agriculture Organization–the UN Educational, Scientific and Cultural Organization soil units to 1 m depth.

Soil unit	Mean C (kg m^{-2})
Acrisols	9.4
Cambisols	9.6
Chernozems	12.5
Podzoluvisols	7.3
Ferrasols	10.7
Gleysols	13.1
Phaeozems	14.6
Fluvisols	9.3
Kastanozems	9.6
Luvissols	6.5
Greyzems	19.7
Nitisols	8.4
Histosols	77.6
Podzols	24.2
Arenosols	3.1
Regosols	5.0
Solonetz	6.2
Andisols	25.4
Vertisols	11.1
Planosols	7.7
Xerosols	4.8
Yermosols	3.0
Solochaks	4.2

These soil units generally span a wide range of climate conditions and therefore present a different view of soil organic matter content based on additional soil factors. In particular, the high C content of Podzols, Histosols, and Andisols is apparent. Refer text for additional explanation of biological, chemical, and physical factors responsible.

Source: From Batjes.^[2]

B horizon. If the surface organic layers are included, these soils can have substantial organic matter contents, exceeding the expected amount for the climate conditions. Batjes^[2] gives an average value for Podzols of 24.2 kg m^{-2} for the surface meter, which is considerably above the mean for most other soil types (see Table 1).

SIGNIFICANT PHYSICAL AND CHEMICAL INFLUENCES

There are several notable exceptions to the climate-based explanation of variation in soil C content. There are two in particular that have lower rates of decomposition and therefore higher accumulations of organic matter than expected (Table 1). These include Histosols due to hydrological conditions and Andisols due to parent material chemical effects.

Histosols

In landscape positions where water accumulates at or above the surface of the soil for an appreciable part of the growing season, decomposition can be reduced to such an extent that large amounts of undecomposed organic matter can accumulate. This soil type is called a Histosol and can be found in any region in wetlands where decomposition is restricted. The soil surface of mature or old-growth boreal forests over shallow water tables are often covered with *Sphagnum* moss, which may also lead to the development of Histosols. Histosols with the largest areas and thickest accumulations occur in lowland tundra where a mixture of sedges, lichens, and mosses grow at the northern limit of vegetation in the northern hemisphere. Production, decomposition, and evaporation are limited by low temperatures and water-saturated soils. In these cold regions, deeper layers may freeze and not become thawed during the short growing season (permafrost). As a result, Histosols have C contents over 70 kg m^{-2} in the surface meter (Table 1). Some regions have been accumulating organic matter since the last glacial period without any substantial decomposition. Histosols in such regions may be several meters thick and contain over 250 kg C m^{-2} .^[2] Globally, it is estimated that boreal and subarctic Histosols contain 455 PgC that has accumulated during the post-glacial period.^[26]

Andisols

Andisols form on young volcanic stone (basalt lava) rich in nutrients and alkaline. Andisols are weakly weathered soils associated with pyroclastic parent materials that are rich in allophane, ferrihydrite, and other minerals that readily form complexes with humus molecules. These chemical constituents provide conditions promoting high vegetation production and also the retention of organic matter in soil. As a result, Andisols typically have higher soil C contents (25.4 kg m^{-2} , Table 1) than soils with the same environmental conditions, but different parent materials.

CONCLUSION

Over long periods of time, organic matter in soils is the result of climatic, biological, and geological factors. These factors are not independent. In particular, there exists a strong relationship between climate and vegetation type. In Fig. 1, the Holdridge climate-based life zones have names that depict the dominant vegetation of climates. Jobbágy and Jackson^[27] provide a summary of soil data based on biomes that demonstrates similar soil C distribution as that based on climate (Table 2).

Over shorter periods of time, soil C varies with vegetation disturbances and changes in land use patterns that

Table 2 Mean organic C content (kg m^{-2}) by biome to 1 m depth.

Biome	Mean C (kg m^{-2})
Boreal forest	9.3
Crops	11.2
Deserts	6.2
Sclerophyllous shrubs	8.9
Temperate deciduous forest	17.4
Temperate evergreen forest	14.5
Temperate grassland	11.7
Tropical deciduous forest	15.8
Tropical evergreen forest	18.6
Tropical grassland/savanna	13.2
Tundra	14.2

Source: From Jobbágy & Jackson^[27] and Biome classification is based on Whittaker.^[28]

affect rates of organic matter input and its decomposition. Various land uses result in very rapid declines in soil organic matter from the native condition.^[29–32] Losses of 50% in the top 20 cm and 30% for the surface 100 cm are average. Much of this loss in soil organic C can be attributed to erosion, reduced inputs of organic matter, increased decomposability of crop residues, and tillage effects that decrease the amount of physical protection to decomposition. Evidence from long-term experiments suggests that C losses due to oxidation and erosion can be reversed with soil management practices that minimize soil disturbance and optimize plant yield through fertilization. These experimental results are believed to apply to large regions and that organic matter is being restored as a result of establishment of perennial vegetation, increased adoption of conservation tillage methods, efficient use of fertilizers, and increased use of high yielding crop varieties.^[33,34] Additionally, when agricultural land is no longer used for cultivation and allowed to revert to natural vegetation or replanted to perennial vegetation, soil organic C can accumulate by processes that essentially reversing some of the effects responsible for soil organic C losses initially—from when the land was converted from perennial vegetation—and return them to typical amounts for the climate, vegetation, landscape position, and parent material conditions.^[35,36]

ACKNOWLEDGMENTS

This work was sponsored by U.S. Department of Energy, Carbon Dioxide Research Program, Environmental Sciences Division, Office of Biological and Environmental Research, and performed at Oak Ridge National Laboratory (ORNL). ORNL is managed by UT-Battelle, LLC, for the U.S. Department of Energy under contract DE-AC05-00OR22725.

REFERENCES

1. Post, W.M.; Peng, T.-H.; Emanuel, W.R.; King, A.W.; Dale, V.H.; DeAngelis, D.L. The global carbon cycle. *Am. Sci.* **1990**, *78*, 310–326.
2. Batjes, N.H. Total carbon and nitrogen in the soils of the world. *Eur. J. Soil Sci.* **1996**, *47*, 151–163.
3. Holdridge, L.R. Determination of world plant formations from simple climatic data. *Science* **1947**, *105*, 367–368.
4. Zinke, P.J.; Stangenberger, A.G.; Post, W.M.; Emanuel, W.R.; Olson, J.S. *Worldwide Organic Soil Carbon and Nitrogen Data*; ORNL/TM-8857; Oak Ridge National Laboratory: Oak Ridge, 1984.
5. Post, W.M.; Pastor, J.; Zinke, P.J.; Stangenberger, A.G. Global patterns of soil nitrogen storage. *Nature* **1985**, *317*, 613–616.
6. Walker, T.W.; Adams, A.F.R. Studies on soil organic matter: I. Influence of phosphorus content of parent materials on accumulations of carbon, nitrogen, sulfur, and organic phosphorus in grassland soils. *Soil Sci.* **1958**, *85*, 307–318.
7. Tiessen, H.J.; Stewart, W.B.; Cole, C.V. Pathways of phosphorus transformations in soils of differing pedogenesis. *Soil Sci. Soc. Am. J.* **1984**, *48*, 853–858.
8. Roberts, T.L.; Stewart, J.W.B.; Bettany, J.R. The influence of topography on the distribution of organic and inorganic soil phosphorus across a narrow environmental gradient. *Can. J. Soil Sci.* **1985**, *65*, 651–665.
9. Anderson, D.W. The effect of parent material and soil development on nutrient cycling in temperate ecosystems. *Biogeochemistry* **1988**, *5*, 71–97.
10. Jenny, H. *The Soil Resource*; Springer: Berlin, 1980.
11. Walker, T.W.; Syers, J.K. The fate of phosphorus during pedogenesis. *Geoderma* **1976**, *15*, 1–19.
12. Schlesinger, W.H. *Biogeochemistry: An Analysis of Global Change*; Academic Press: New York, 1991.
13. Torn, M.S.; Trumbore, S.E.; Chadwick, O.A.; Vitousek, P.M.; Hendricks, D.M. Mineral control of soil organic carbon storage and turnover. *Nature* **1997**, *389*, 170–173.
14. Aber, J.D.; Melillo, J.M. Nitrogen immobilization in decaying hardwood leaf litter as a function of initial nitrogen and lignin content. *Can. J. Bot.* **1982**, *58*, 416–421.
15. Meentemeyer, V. Macroclimate and lignin control of litter decomposition rates. *Ecology* **1978**, *59*, 465–472.
16. Zinke, P.J. The patterns of influence of individual trees on soil properties. *Ecology* **1962**, *43*, 130–133.
17. Wedin, D.A.; Tilman, D. Species effects on nitrogen cycling: A test with perennial grasses. *Oecologia* **1990**, *84*, 433–441.
18. Hobbie, S.E. Effects of plant species on nutrient cycling. *Trends Ecol. Evol.* **1992**, *7*, 336–339.
19. Hobbie, S.E. Temperature and plant species control over litter decomposition in Alaskan tundra. *Ecol. Monogr.* **1996**, *66*, 503–522.
20. Sims, P.L.; Coupland, R.T. Producers. In *Grassland Ecosystems of the World: Analysis of Grasslands and Their Uses*; Coupland, R.T., Ed.; Cambridge University Press: Cambridge, 1979; 49–72.
21. Olson, J.S. Energy storage and the balance of producers and decomposers in ecological systems. *Ecology* **1963**, *44*, 322–331.

22. Fogel, R.; Cromack, K. Effect of habitat and substrate quality on Douglas fir litter decomposition in western Oregon. *Can. J. Bot.* **1977**, *55*, 1632–1640.
23. Pandey, U.; Singh, J.S. Leaf-litter decomposition in an oak–conifer forest in Himalaya: The effects of climate and chemical composition. *Forestry* **1982**, *55*, 47–59.
24. Flanagan, P.W.; Van Cleve, K. Nutrient cycling in relation to decomposition and organic matter quality in taiga ecosystems. *Can. J. For. Res.* **1983**, *13*, 795–817.
25. McLaugherty, C.A.; Pastor, J.; Aber, J.D.; Melillo, J.M. Forest litter decomposition in relation to soil nitrogen dynamics and litter quality. *Ecology* **1984**, *66*, 266–275.
26. Gorham, E. Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. *Ecol. Appl.* **1991**, *1*, 182–195.
27. Jobbágy, E.G.; Jackson, R.B. The vertical distribution of organic carbon and its relation to climate and vegetation. *Ecol. Appl.* **2000**, *10* (2), 423–436.
28. Whittaker, R.H. *Communities and Ecosystems*; MacMillan: London, 1975.
29. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
30. Davidson, E.A.; Ackerman, I.L. Changes in soil carbon inventories following cultivation of previously untilled soils. *Biogeochemistry* **1993**, *20*, 161–193.
31. Mann, L.K. Changes in soil carbon after cultivation. *Soil Sci.* **1986**, *142*, 279–288.
32. Schlesinger, W.H. Changes in soil carbon storage and associated properties with disturbance and recovery. In *The Changing Carbon Cycle: A Global Analysis*; Trabalka, J.R., Reichle, D.E., Eds.; Springer: New York, 1985; 194–220.
33. Buyanovsky, G.A.; Wagner, G.H. Carbon cycling in cultivated land and its global significance. *Glob. Change Biol.* **1998**, *4*, 131–141.
34. Lal, R.; Kimble, J.M.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Ann Arbor Press: Ann Arbor, 1998.
35. Post, W.M.; Kwon, K.C. Soil carbon sequestration and land-use change: Processes and potential. *Glob. Change Biol.* **2000**, *6*, 317–328.
36. Silver, W.L.; Ostertag, R.; Lugo, A.E. The potential for carbon sequestration through reforestation of abandoned tropical agricultural and pasture lands. *Restor. Ecol.* **2000**, *8* (4), 394–407.

Soil Organic Matter (SOM): Historical Concepts

Christian Feller

Institute of Research for Development, Montpellier, France

Abstract

Soil organic matter (SOM) contributes significantly to the chemical, physical, and biological ecosystem functions of soil. It thus provides agricultural and environmental services, such as soil carbon (C) sequestration, regulation of the water cycle, and detoxification of pollutants. Identification of the functions and services provided by SOM has a long and tumultuous history of scientific discoveries and struggles against false assumptions. This entry reports major steps of this history with emphasis on the role of SOM in: 1) plant productivity, soil fertility, and sustainability of cropping and farming systems and 2) regulating climate change through C sequestration.

INTRODUCTION

The role of soil organic matter (SOM) in controlling the capacity of soil resources to deliver agricultural and environmental services and sustain human societies at both local (e.g., fertility maintenance) and global [e.g., mitigation of atmospheric carbon (C) emissions] scales is scientifically well established. SOM contributes to a range of functions that can be connected to goods and services at the ecosystem level,^[1] but the scientific recognition of this fact and its implications for farming practices and land use management options have fluctuated over time.^[1,2]

This entry presents a brief history of the building of concepts and tools that led to the perception of the interconnection between SOM, soil fertility, and ecosystem sustainability with focus on the following two broad functions of SOM: 1) a productive function, namely, the sustainment of farming and 2) an environmental function, namely, the control of the greenhouse effect, climate change, and soil C sequestration. For more information, see the entry *Soil Organic Matter (SOM): Characterization* (p. 2123).

All references are cited in text for authors and years can be found in the historical review papers.

AGRONOMIC FUNCTION OF SOM

Concepts of plant nutrition varied greatly during antiquity (Browne, 1944).^[1,2] Finally, it was Aristotle's theory that remained influential during the Middle Ages: soil is the main source of nutrients for plants, and soil fertility is dependent on four qualities—warmth, coldness, humidity, and dryness. Some different theories emerged during the 16th and 17th centuries, namely the theory of “salts” from Palissy (1580),^[3] the theory of “water” from Van Helmont

(early 17th century), and the theory of “tillage” from Tull (1733). The idea that the soluble fraction of humus could nourish the plants emerged by the end of the 18th century (Hassenfratz, 1792),^[2] while several authors, founders of the ecophysiology such as Priestley (1777), Ingen-Housz (1780), Senebier (1782), and de Saussure (1804), gave a partial refutation of this theory by experimentally demonstrating a gaseous origin of plant C and the role of light during photosynthesis. Contradictory debates arose on the subject, but many agricultural scientists shared an intermediary point of view and assigned functions in plant nutrition to both SOM and air. This was in particular the view of the famous German agronomist Albrecht Daniel Thaër (1752–1828) known for the “theory of humus” developed in his seminal book *Principles of Rational Agriculture* (1809).^[4]

Thaër's principles were based on the following two unverified theoretical concepts of plant nutrition: 1) the majority of plant dry matter derives from the “soil nutritive juices” contained in the hot water-soluble fraction of soil humus; and 2) plant demand for “juices” is selective and varies with the species cultivated. Therefore, management of soil fertility must be based on the management of the soil-humus balance as well as on that of agricultural techniques and crop succession. The system was completely quantified with an indicator—the “fertility degree”—used for the evaluations of both fertility and economy. Based on a false theory, nevertheless, it was the basis for the first rational and systematic approach to fertilization within the context of sustainable cropping practices (de Wit, 1974).^[2] Thaër's work seriously influenced the thinking of his peers during the first half of the 19th century. Had Thaër focused on mineral rather than organic budgets, he would probably have been regarded as the founder of Western scientific agriculture.

The “mineral theory,” first developed by Sprengel (1838) and then by Liebig (1840) in his authoritative text *Die organische Chemie in ihrer Anwendung auf Agrikultur und Physiologie*, demonstrated the origin of plant dry matter from mineral compounds and not from humus. It was the death of the humus theory and the beginning of mineral fertilization, chemical fertilizer industry, and the “nitrogen–phosphorus–Potassium era.” A practical consequence of this theory is the prevalent idea that organic fertilization is not necessary if mineral fertilizers are available. Grandeau^[5] proposed a theory that bridged the gap between the “humus” and the “mineral” theories by attributing an essential role to humus as a factor of solubilization and thus of assimilation of mineral elements with limited availability for the plant (phosphorus). He created the concept of “bioavailability” that was fully supported by Liebig.^[5] The “mineralist” approach reached its apogee after World War II with a spectacular increase in crop yields and in fertilizer and other input consumptions. However, as early as the 1930s, environmental degradation and loss of ecosystem services appeared: wind and water erosion; air, soil, and water pollution; etc. The notion of a “healthy” function of soil and SOM began to be developed by Balfour and Howard (1940), Balfour (1944), and Rodale (1945). It was the birth of “organic farming” with the partly ideological concept of the biotic pyramid proposed by Albrecht Daniel Thaër (1975): Any soil degradation (at the basement of the pyramid) threatens plant health that threatens animal health and finally human “health.” The philosophy of organic farming is fundamentally holistic. The solutions lie with the abandonment of chemical practices and the predominant use of renewal resources, especially organic matter resources. Organic farming and other agroecological practices are scientifically studied and experimented all around the world in the perspective of a sustainable agriculture: Humus is thus once again in fashion!

AN ENVIRONMENTAL FUNCTION OF SOM: CONTROL OF THE GREENHOUSE EFFECT AND C SEQUESTRATION

Since 1992, increasing attention has been given to the potential of SOM for C storage and to its importance in

the greenhouse gas (GHG) fluxes [mainly carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O)] at the soil–plant–atmosphere interfaces.^[1] The terms “sequestration” and “C sequestration” were proposed to define the aptitude of terrestrial ecosystems to act as sinks for these GHGs. A definition of the “soil C sequestration” including the total balance of the three main GHGs was proposed by Bernoux et al. (2006). The history of “soil C sequestration” (including that of humus or SOM) thus includes those of SOM storage and GHG (especially CO₂) fluxes at different space and timescales. The sole history of CO₂ fluxes is described below.

The first *in situ* measurements of soil CO₂ were made by Boussingault and Lewy (1852, 1853) at depths ranging from 40 to 240 cm. These authors showed that CO₂ concentrations in soils without farmyard manure (FYM) application were 22 to 23 times higher than in the atmosphere and that applying FYM could increase this concentration by 245 times.

The main forerunner to modern CO₂ fluxes measurements at soil–plant–atmosphere level is the Danish ecophysiologist Henrik Lundegårdh (1888–1969; Larkum, 2003). Lundegårdh published considerable data on CO₂ fluxes at the soil–plant interface with two books in 1924 and 1930, respectively, and a vast paper in 1927.^[6]

In these three publications, Lundegårdh reports an impressive amount of quantitative data on *in situ* CO₂ fluxes between atmospheric, plant, and soil components. Data were collected using the instruments of his own conception for the sampling of soil atmosphere (equivalent to our static chamber) or the continuous monitoring of CO₂ fluxes at the plant or atmosphere level. In the 1927 publication, he even described field equipment and experimental designs completely equivalent to “free air CO₂ experiment (FACE),” which are among the most sophisticated experiments for the study of CO₂ fluxes at the field level. Lundegårdh’s findings are remarkably close to the existing data.

CONCLUSION

There is a long history of scientists’ engagement in the study of SOM, as a consequence of their conviction of its

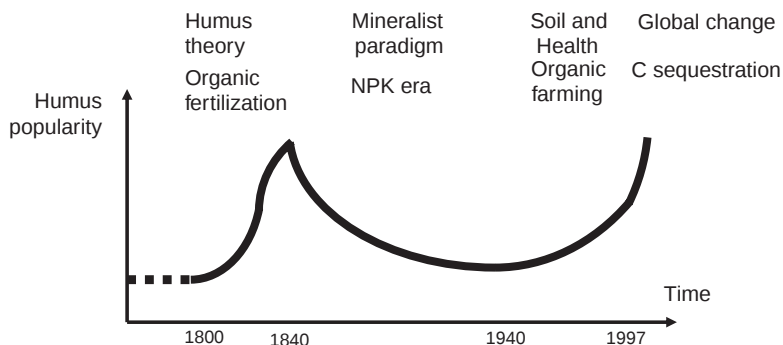


Fig. 1 Ups and downs in the popularity of humus in relation to scientific theories and agricultural practices.

functional value. There are many cases, often forgotten, of perceptions that preview concepts that are accepted as essential for sound management of natural resources, such as that of sustainability (e.g., Thaër) or of developments of new approaches and tools (such as FACE-type experiments) that are readily recognizable by scientists.

The popularity of humus has changed over the past three centuries (Fig. 1), but it is generally accepted by scientists that loss in SOM is one of the major factors, leading to degradation of ecosystem services and loss of ecosystem resilience. In many countries, however, conflicts have arisen between policies for ecosystem protection that embrace sustainable soil management and those targeted at agricultural development. The decision-makers will be convinced by scientists when SOM will be proved to have a quantitative economic value. That is a scientific challenge.

REFERENCES

1. Feller, C.; Manlay, R.J.; Swift, M.J.; Bernoux, M. Functions, services and value of soil organic matter for human societies and the environment: A historical perspective. In *Functions of Soils for Human Societies and the Environment*; Frossard, E., Blum, W.E.H., Warkentin, B.P., Eds.; Geological Society, Special Publication: London, 2006; 9–22.
2. Feller, C.; Thuriès, L.; Manlay, R.; Robin, P.; Frossard, E. The principles of rational agriculture by A.D. Thaër (1752–1828). An approach of the sustainability of cropping systems at the beginning of the 19th century. *J. Plant Nutr. Soil Sci.* **2003**, *166*, 687–698.
3. Feller, C. Une fausse rupture ou l'intérêt du retour aux sources en histoire de l'agronomie. L'exemple de la nutrition minérale des plantes et du «génial» Palissy. In *Histoire et Agronomie. Entre Ruptures et Durée*; Robin, P., Aeschlimann, J.P., Feller, C., Eds.; IRD: Paris, 2007; 181–201.
4. Thaër, A. *Grundsätze der Rationellen Landwirtschaft (1809–1812)*; Realschulbuch: Berlin, 1809.
5. Grandeau, L. Recherches expérimentales sur le rôle des matières organiques du sol dans la nutrition des plantes. In *Annales Station Agronomique de l'Est*; Berger-Levrault et Cie: Paris, 1878; 225–352.
6. Lundegårdh, H. Carbon dioxide evolution of soil and crop growth. *Soil Sci.* **1927**, *23*, 417–453.

Soil Organic Matter (SOM): Landscape

Henry H. Janzen

Lethbridge Research Centre, Agriculture and Agri-Food Canada, Lethbridge, Alberta, Canada

B.H. Ellert

Agriculture and Agri-Food Canada, Lethbridge, Alberta, Canada

D.W. Anderson

Department of Soil Science, University of Saskatchewan, Saskatoon, Saskatchewan, Canada

Abstract

The heterogeneity of organic matter in the landscape and its further rearrangement by human activity determine how ecosystems function. The topsoil, enriched in organic matter, forms a veneer over the landscape, one that varies from place to place and year to year. The performance and persistence of ecosystems depend on this thin layer. And how that layer varies over the landscape, especially in response to management, therefore has long-lasting effects on how productive and resilient the ecosystem will be. We know something about how past management has affected organic matter distribution; we need to learn how restorative practices will shape its patterns across the landscape.

INTRODUCTION

Soil organic matter is composed largely of plant tissues decomposing back to the simple molecules from which they were first formed: carbon dioxide (CO₂), ammonium, water, and various salts. Although often present in large amounts, organic matter is transient—new litter is continually being added, and organic matter already present is always decomposing. So at any time, the amount present reflects the net balance between additions and losses.

Both inputs and losses at a site depend on conditions there. Litter inputs from plant growth and decomposition are both influenced by many factors: temperature, moisture, aeration, nutrients, plant community, and more. And wind and water move organic matter-laden soil about the landscape. So the amount and composition of organic matter differ from place to place in the landscape.

Organic matter varies in several dimensions: It varies vertically within the profile and horizontally across the landscape. It changes across time, often by human influence. How organic matter is distributed over the landscape may be as important as its amount in affecting the way an ecosystem behaves.^[1] In this review, we describe how organic matter is distributed in “natural” landscapes, show how human activities can alter that pattern, and illustrate with a few examples how organic matter distribution (and redistribution) can affect ecosystem function.

ORGANIC MATTER IN LANDSCAPES UNAFFECTED BY HUMAN ACTIVITY

Vertical Distribution

In most soil profiles, organic matter [or organic carbon (C)] concentration is highest near the surface, where most plant litter is added, and then declines with increasing depth.^[2,3] In grasslands, globally, the surface 0.2 m of soil accounts for 42% of the organic C in the first 1 m; in shrublands, the proportion is 33% and in forests, it is 50%.^[4]

The rate of organic matter turnover also changes with depth. Compared to that deeper in the profile, surface soil usually has higher proportions of “young” organic matter from inputs of plant litter. Radiocarbon studies show that the mean turnover time or radiocarbon “age” of organic matter usually increases with depth.^[5–7]

Lateral Distribution

The amount of organic matter in the soil at a given spot is the result of a complex interaction, over time, of parent material, climate, vegetation, and topography.^[8] Among landscapes, at regional scales, organic matter content is controlled largely by precipitation, temperature, soil texture, and vegetation type.^[9,10] But within a landscape, organic matter is also influenced by other factors. Topography, through effects on microclimate and water movement, can produce soils of widely different organic matter

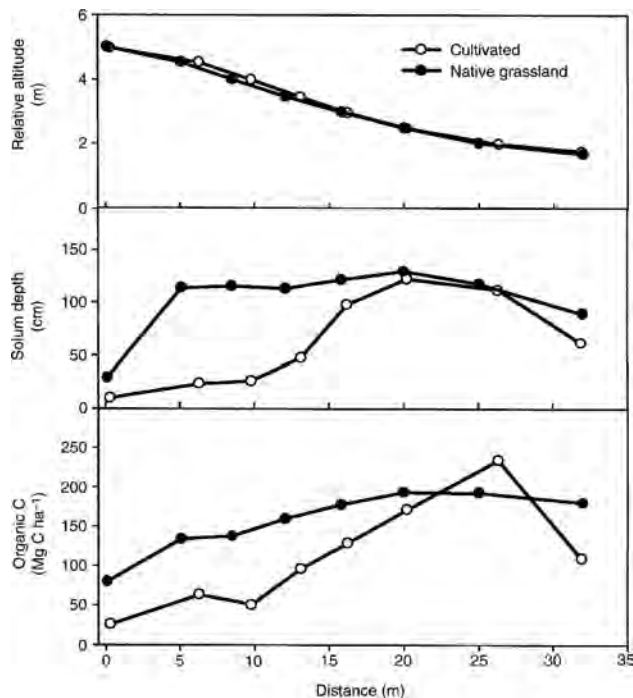


Fig. 1 An illustration of variability in soil organic C in transects across two toposequences, a native grassland and an agricultural field cultivated for about two decades. Organic matter is about 58% C, by weight, so it is often measured as organic C.

Source: Adapted from Gregorich & Anderson.^[11]

within meters (Fig. 1).^[9,12] Usually, the amount of organic matter is lowest near summits and highest in toeslope positions.^[13–15] This pattern occurs for various reasons:^[14,16,17] higher moisture and nutrient status downslope increase litter production; organic matter in lower slope positions may decompose more slowly because of soil conditions (e.g., reduced aeration or accumulated clay); variations in microclimate produce different plant communities across topographical gradients; and erosion may move organic matter downhill.

Apart from topography, localized variations in texture, soil chemistry, vegetation,^[1,18] and other properties may also cause organic matter to vary within landscapes. Even in landscapes that appear uniform, therefore, soil organic matter content varies significantly over scales of several meters. Tiessen and Santos^[19] observed coefficients of variation greater than 50% in organic C and total nitrogen (N) concentrations in the surface soil of a tropical semiarid field ($65 \times 40 \text{ m}^2$) immediately after clearing.

Organic matter varies across landscapes not only in amount but also in form. For example, Paul et al.^[5] observed that radiocarbon “age” of surface soil increased from toeslope to summit positions at two uncultivated grassland sites. And Schimel et al.^[16] found that the relative mineralization rate of organic matter (N mineralized per unit of total N) decreased downslope, even though total mineralization increased, pointing to differences in organic matter quality.

Temporal Distribution

Organic matter in the landscape also changes with time. Soils in early stages of development accumulate organic matter, but the rate of build-up slows as soils approach a steady state, where decomposition roughly balances litter inputs.^[20] Even then, however, organic matter fluctuates during a year because litter additions follow a different pattern from decomposition. It also fluctuates from year to year with changes in weather: in some periods, when litter inputs exceed losses, organic matter accumulates; in others, it is depleted.^[21]

HUMAN INFLUENCE ON DISTRIBUTION

Human activities can profoundly alter soil organic matter content. Historically, disturbance of “natural” ecosystems has almost always resulted in losses.^[22,23] Converting grass- and forestlands to arable agriculture, e.g., typically results in the loss of about 30% of the organic C originally present in the solum.^[24] But with better land management, at least part of the organic matter lost can be restored.^[25]

Human influence on soil organic matter—whether it leads to losses or gains—is rarely uniform over the landscape. Thus, human intervention often alters not only the amount of organic matter, but also its distribution over the landscape. The rearrangement can occur in several ways.

Patchwork Application of Practices

Land practices are often not applied uniformly over the landscape. Forests may be cut in patches; diverse cropping practices may be used in scattered patterns in agricultural landscapes; organic materials such as manures, derived from large, far-flung areas, may be funneled into small areas;^[26] grazing animals may congregate near water or shade, depositing organic matter there.^[27] These, and other practices applied non-uniformly, can redistribute organic matter on the landscape.

Non-Uniform Effects on C Balance

Even where the same management practice or land use change is applied uniformly over a large area, its effect on organic matter may vary from place to place because the landscape is not uniform to begin with. For example, Schimel et al.^[28] observed that proportional losses of organic matter after cultivating grassland were higher in upper- than in lower-slope positions. And the mechanism of organic matter loss, whether by erosion or biological mineralization, may also vary among slope positions.^[11]

Tillage

Farmers cultivate soils to control pests, prepare land for seeding, and bury residues. This tillage dilutes organic

matter-rich soil near the surface by mixing it with soil from deeper layers.^[15] It may also affect organic matter deeper in the profile by altering rooting patterns, leaching, faunal activity, and soil temperature.^[29,30] Consequently, tillage changes organic matter distribution in the profile, even though total amount may not always be affected.^[31]

Tillage also alters lateral distribution of organic matter by physically “dragging” topsoil. Each tillage pass moves soil, but the extent of movement is greater downslope than upslope, so that over the years, tillage moves soil and organic matter to lower-slope positions.^[32]

Erosion

Globally, the predominant human influence on distribution of organic matter in the landscape is via erosion, especially in deforested and agricultural lands. Although erosion is a natural process and occurs also in undisturbed ecosystems, human disturbance may increase erosion by orders of magnitude.^[33,34]

Erosion affects organic matter at a site within the landscape in three ways;^[12] it may remove organic matter; result in mixing of subsoil into the surface layer by stripping surface soil away; or result in deposition of soil eroded from elsewhere in the landscape.

The effects of erosion are not uniform across the landscape. Highest losses usually occur from convex uplands or “shoulder elements.”^[12,14] Much of the eroded soil removed from one location in the landscape may be deposited nearby, especially, in closed watersheds.^[35] But areas of net removal are usually larger than areas of deposition so that erosion often increases the variability of organic matter on the landscape. This is compounded by selective translocation of soil fractions rich in organic matter.^[15,36] The net effect, therefore, is often a removal of organic matter from widespread areas in the landscape and its deposition in low-lying areas.

Erosion not only affects directly the distribution of organic matter on the landscape, but may also have long-lasting secondary consequences through effects on plant growth and litter input.^[15] While erosion can result in extensive translocation of organic matter, its net effect on total amount stored in the landscape is not always clear. If erosion suppresses productivity, thereby limiting replenishment of organic matter, the organic matter may spiral downwards over the long term. But when productivity of eroded areas can be restored, the eroded landscape might eventually contain more organic matter than before, because of the higher storage potential of eroded areas^[37] and the accumulated and buried organic matter in depositional areas.^[15]

Disruption and management of ecosystems by humans often alter irreversibly the amount and distribution of organic matter in landscapes, often increasing variability on the landscape (e.g., Fig. 1).^[38] Our understanding of this redistribution is incomplete.^[39,40]

IMPLICATIONS

The heterogeneity of organic matter in the landscape and its further rearrangement by human activity determine how ecosystems function. To illustrate, we present three examples.

Firstly, the spatial variability of organic matter influences productivity on the landscape. Plant productivity is closely linked to organic matter;^[41] consequently, landscapes with variable organic matter usually show corresponding heterogeneity in productivity (whether high organic matter increases productivity or high productivity increases organic matter is not always clear). In farmlands, therefore, increasing attention has been devoted to “site-specific farming”—adjusting practices spatially to compensate for variability in organic matter and related soil properties.^[42]

Secondly, the way organic matter is distributed across the landscape influences the unintended release of nutrients into the broader environment. For example, organic matter affects transformations of N, both as a source of mineralized N and by effects on microbial activity. Consequently, nitrate leaching or nitrous oxide emissions may be linked to organic matter distribution, especially as sites where organic matter accumulates may also have high moisture.^[43,44]

Thirdly, the heterogeneity of organic matter within the landscape makes it harder to measure changes in C storage. Soil organic matter has been proposed as a potential “sink” for C; widespread adoption of practices that build organic matter could increase C storage in soils, mitigating the increases in atmospheric CO₂ linked to global warming.^[45] But to quantify that “sink” precisely, organic matter distribution across the landscape would have to be taken into account; differences in organic matter among points on the landscape are usually much greater than the expected response to new management.^[46] At one location, Garten and Wullschlegel^[47] estimated that more than 100 samples would need to be taken to detect a change in soil organic C of about 1 Mg C ha⁻¹. The redistribution of organic matter by erosion makes measuring of the C sink even more complicated.^[48] When erosion occurs during the measurement interval, it may be hard to distinguish C exchanged with the atmosphere from that merely redistributed on the landscape.

The topsoil, enriched in organic matter, forms a veneer over the landscape, one that varies from place to place and year to year. The performance and persistence of ecosystems depend on this thin layer. And how that layer varies over the landscape, especially in response to management, therefore has long-lasting effects on how productive and resilient the ecosystem will be. In the past, human activity has often accentuated native variability, removing organic matter from sites where it was already in short supply and depositing it in areas of excess. New approaches may favor the preservation of organic matter both in amount

and distribution across the landscape. We know something about how past management has affected organic matter distribution; we need to learn how restorative practices will shape its patterns across the landscape.

REFERENCES

- Herrick, J.E.; Wander, M.M. Relationships between soil organic carbon and soil quality in cropped and rangeland soils: The importance of distribution, composition, and soil biological activity. In *Soil Processes and the Carbon Cycle*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1997; 405–425.
- Ajwa, H.A.; Rice, C.W.; Sotomayor, D. Carbon and nitrogen mineralization in tallgrass prairie and agricultural soil profiles. *Soil Sci. Soc. Am. J.* **1998**, *62*, 942–951.
- Zhao, Q.; Zhong, L.; Yingfei, X. Organic carbon storage in soils of southeast China. *Nutr. Cycl. Agroecosyst.* **1997**, *49*, 229–234.
- Jobbágy, E.; Jackson, R.B. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecol. Appl.* **2000**, *10* (2), 423–436.
- Paul, E.A.; Follett, R.F.; Leavitt, S.W.; Halvorson, A.; Peterson, G.A.; Lyon, D.J. Radiocarbon dating for determination of soil organic matter pool sizes and dynamics. *Soil Sci. Soc. Am. J.* **1997**, *61*, 1058–1067.
- Trumbore, S. Age of soil organic matter and soil respiration: Radiocarbon constraints on belowground C dynamics. *Ecol. Appl.* **2000**, *10* (2), 399–411.
- Scharpenseel, H.W.; Pfeiffer, E.M.; Becker-Heidmann, P. Ecozone and soil profile screening for C-residence time, rejuvenation, bomb ^{14}C photosynthetic $\delta^{13}\text{C}$ changes. In *Assessment Methods for Soil Carbon*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; Lewis Publishers/CRC Press: Boca Raton, 2001; 207–220.
- Jenny, H. *The Soil Resource: Origin and Behavior*; Springer: New York, 1980; 377 pp.
- Arrouays, D.; Daroussin, J.; Kicin, J.L.; Hassika, P. Improving topsoil carbon storage prediction using a digital elevation model in temperate forest soils of France. *Soil Sci.* **1998**, *163* (2), 103–108.
- Burke, I.C.; Yonker, C.M.; Parton, W.J.; Cole, C.V.; Flach, K.; Schimel, D.S. Texture, climate, and cultivation effects on soil organic matter content in U.S. grassland soils. *Soil Sci. Soc. Am. J.* **1989**, *53*, 800–805.
- Gregorich, E.G.; Anderson, D.W. Effects of cultivation and erosion on soils of four toposequences in the Canadian prairies. *Geoderma* **1985**, *36*, 343–354.
- Pennock, D.J. Effects of soil redistribution on soil quality: Pedon, landscape, and regional scales. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 167–185.
- Burke, I.C.; Elliott, E.T.; Cole, C.V. Influence of macroclimate, landscape position, and management on soil organic matter in agroecosystems. *Ecol. Appl.* **1995**, *5* (1), 124–131.
- Schimel, D.S.; Kelly, E.F.; Yonker, C.; Aguilar, R.; Heil, R.D. Effects of erosional processes on nutrient cycling in semiarid landscapes. In *Planetary Ecology*; Caldwell, D.E., Brierley, J.A., Brierley, C.L., Eds.; Van Nostrand Reinhold: New York, 1985; 571–580.
- Gregorich, E.G.; Greer, K.J.; Anderson, D.W.; Liang, B.C. Carbon distribution and losses: Erosion and deposition effects. *Soil Till. Res.* **1998**, *47*, 291–302.
- Schimel, D.; Stillwell, M.A.; Woodmansee, R.G. Biogeochemistry of C, N, and P in a soil catena of the shortgrass steppe. *Ecology* **1985**, *66* (1), 276–282.
- Cheng, W.; Virginia, R.A.; Oberbauer, S.F.; Gillespie, C.T.; Reynolds, J.F.; Tenhunen, J.D. Soil nitrogen, microbial biomass, and respiration along an arctic toposequence. *Soil Sci. Soc. Am. J.* **1998**, *62*, 654–662.
- Mueller-Harvey, I.; Juo, A.S.R.; Wild, A. Soil organic C, N, S and P after forest clearance in Nigeria: Mineralization rates and spatial variability. *J. Soil Sci.* **1985**, *36*, 585–591.
- Tiessen, H.; Santos, M.C.D. Variability of C, N, and P content of a tropical semiarid soil as affected by soil genesis, erosion and land clearing. *Plant Soil* **1989**, *119*, 337–341.
- Chadwick, O.A.; Kelly, E.F.; Merriitts, D.M.; Amundson, R.G. Carbon dioxide consumption during soil development. *Biogeochemistry* **1994**, *24*, 115–127.
- Campbell, C.A.; Zentner, R.P.; Selles, F.; Biederbeck, V.O.; McConkey, B.G.; Blomert, B.; Jefferson, P.G. Quantifying short-term effects of crop rotations on soil organic carbon in southwestern Saskatchewan. *Can. J. Soil Sci.* **2000**, *80*, 193–202.
- Solomon, D.; Lehmann, J.; Zech, W. Land use effects on soil organic matter properties of chromic luvisols in semiarid Northern Tanzania: Carbon, nitrogen, lignin and carbohydrates. *Agr. Ecosyst. Environ.* **2000**, *78*, 203–213.
- Tiessen, H.; Cuevas, E.; Chacon, P. The role of soil organic matter in sustaining soil fertility. *Nature* **1994**, *371*, 783–785.
- Davidson, E.A.; Ackerman, I.L. Changes in soil carbon inventories following cultivation of previously untilled soil. *Biogeochemistry* **1993**, *20*, 161–193.
- Sampson, R.N.; Scholes, R.J. Additional human-induced activities—Article 3.4. In *Land Use, Land Use Change, and Forestry: A Special Report of the IPCC*; Watson, R.T., Noble, I.R., Bolin, B., Ravindranath, N.H., Verardo, D.J., Dokken, D.J., Eds.; Cambridge University Press: Cambridge, 2000; 180–281.
- Fernandes, E.C.M.; Motavalli, P.P.; Castilla, C.; Mukurumbira, L. Management control of soil organic matter dynamics in tropical land use systems. *Geoderma* **1997**, *79*, 49–67.
- Franzluebbers, A.J.; Stuedemann, J.A.; Schomberg, H.H. Spatial distribution of soil carbon and nitrogen pools under grazed tall fescue. *Soil Sci. Soc. Am. J.* **2000**, *64*, 635–639.
- Schimel, D.S.; Coleman, D.C.; Horton, K.A. Soil organic matter dynamics in paired rangeland and cropland toposequences in North Dakota. *Geoderma* **1985**, *36*, 201–214.
- Cihacek, L.J.; Ulmer, M.G. Effects of tillage on profile soil carbon distribution in the northern great plains of the U.S. In *Management of Carbon Sequestration in Soil*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1997; 83–91.
- Mikhailova, E.A.; Bryant, R.B.; Vassenev, I.I.; Schwager, S.J.; Post, C.J. Cultivation effects on soil carbon and

- nitrogen contents at depth in the Russian chernozem. *Soil Sci. Soc. Am. J.* **2000**, *64*, 738–745.
31. Yang, X.-M.; Wander, M.M. Tillage effects on soil organic carbon distribution and storage in a silt loam soil in Illinois. *Soil Till. Res.* **1999**, *52*, 1–9.
 32. Govers, G.; Lobb, D.A.; Quine, T.A. Tillage erosion and translocation: Emergence of a new paradigm in soil erosion research. *Soil Till. Res.* **1999**, *51*, 167–174.
 33. Meade, R.H.; Yuzyk, T.R.; Day, T.J. Movement and storage of sediment in rivers of the United States and Canada. In *The Geology of North America*; Wolman, M.G., Riggs, H.C., Eds.; Geological Society of America: Boulder, 1990; Vols. 0–1, 255–280.
 34. Oldeman, L.R. The global extent of soil degradation. In *Soil Resilience and Sustainable Land Use*; Greenland, D.J., Szabolcs, I., Eds.; CAB International: Oxfordshire, 1994; 99–118.
 35. Pennock, D.J.; De Jong, E. Rates of soil redistribution associated with soil zones and slope classes in Southern Saskatchewan. *Can. J. Soil Sci.* **1990**, *70*, 325–334.
 36. van Noordwijk, M.; Cerri, C.; Woomer, P.L.; Nugroho, K.; Bernoux, M. Soil carbon dynamics in the humid tropical forest zone. *Geoderma* **1997**, *79*, 187–225.
 37. Izaurralde, R.C.; Nyborg, M.; Solberg, E.D.; Janzen, H.H.; Arshad, M.A.; Malhi, S.S.; Molina-Ayala, M. Carbon storage in eroded soils after five years of reclamation techniques. In *Soil Processes and the Carbon Cycle*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1997; 369–385.
 38. Beckett, P.H.T.; Webster, R. Soil variability: A review. *Soils Fert.* **1971**, *34* (1), 1–15.
 39. Starr, G.C.; Lal, R.; Kimble, J.M.; Owens, L. Assessing the impact of erosion on soil organic carbon pools and fluxes. In *Assessment Methods for Soil Carbon*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; Lewis Publishers, CRC Press: Boca Raton, 2001; 417–426.
 40. Jacinthe, P.A.; Lal, R.; Kimble, J.M. Assessing water erosion impacts on soil carbon pools and fluxes. In *Assessment Methods for Soil Carbon*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; Lewis Publishers, CRC Press: Boca Raton, 2001; 427–449.
 41. Bauer, A.; Black, A.L. Quantification of the effect of soil organic matter content on soil productivity. *Soil Sci. Soc. Am. J.* **1994**, *58*, 185–193.
 42. Beckie, H.J.; Moulin, A.P.; Pennock, D.J. Strategies for variable rate nitrogen fertilization in hummocky terrain. *Can. J. Soil Sci.* **1997**, *77*, 589–595.
 43. van Kessel, C.; Pennock, D.J.; Farrell, R.E. Seasonal variations in denitrification and nitrous oxide evolution at the landscape scale. *Soil Sci. Soc. Am. J.* **1993**, *57*, 988–995.
 44. Pennock, D.J.; Corre, M.D. Development and application of landform segmentation procedures. *Soil Till. Res.* **2001**, *58*, 151–162.
 45. Lal, R.; Bruce, J.P. The potential of world cropland soils to sequester C and mitigate the greenhouse effect. *Environ. Sci. Pol.* **1999**, *2*, 177–185.
 46. Ellert, B.H.; Janzen, H.H.; McConkey, B.G. Measuring and comparing soil carbon storage. In *Assessment Methods for Soil Carbon*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; Lewis Publishers/CRC Press: Boca Raton, 2001; 131–146.
 47. Garten, C.T., Jr.; Wulschleger, S.D., Jr. Soil carbon inventories under a bioenergy crop (Switchgrass): Measurement limitations. *J. Environ. Qual.* **1999**, *2*, 1359–1365.
 48. Pennock, D.J.; Frick, A.H. The role of field studies in landscape-scale applications of process models: An example of soil redistribution and soil organic carbon modeling using century. *Soil Till. Res.* **2001**, *58*, 183–191.

Soil Organic Matter (SOM): Management

R. Cesar Izaurralde

Battelle Pacific Northwest National Laboratory, Washington, District of Columbia, U.S.A.

Carlos C. Cerri

University of Sao Paulo, Sao Paulo, Brazil

Abstract

Soil organic matter (SOM) consists of a complex array of living organisms, such as bacteria and fungi, plant and animal debris in different stages of decomposition, and *humus*—a rather stable brown to black material showing no resemblance to the organisms from which it originates. Because SOM is or has been a part of living tissues, its composition is dominated by carbon (C), hydrogen, and oxygen and—in lesser abundance—by nitrogen, phosphorus, and sulfur among other elements. The levels of SOM are expressed in terms of soil organic carbon (SOC) concentration (g kg^{-1}) or mass per unit area (g m^{-2}) to a given depth. The level of SOC in virgin soils reflects the action and interaction of the major factors of soil formation: climate, vegetation, topography, parent material, and age. These factors control SOC content by regulating the balance between C gains via photosynthesis and losses via autotrophic and heterotrophic respiration, as well as C losses in soluble and solid form. The SOC content usually ranges between 5 and 100 g kg^{-1} in mineral soils. These concentrations appear modest but at 1500 Pg, the amount of organic C stored globally in soils is second only to that contained in oceans and at least twice that found in either terrestrial vegetation or the atmosphere.

INTRODUCTION

Cultivated soils usually contain less soil organic carbon (SOC) than virgin soils^[1] due to the magnification of two biophysical processes: 1) net nutrient mineralization accompanied by the release of carbon dioxide (CO_2) due to microbial respiration and 2) soil erosion. SOC losses of up to 50% have been reported within 30–70 years of land use conversions under temperate conditions.^[2–5] SOC losses reported in subtropical and tropical environments often match or even surpass those observed under temperate conditions.^[6–8] In subtropical and tropical environments, shifting cultivation systems appear to conserve more SOC than forestlands permanently cleared for cultivation.^[9]

MAJOR PROCESSES LEADING TO CARBON (C) LOSSES FROM SOIL

Mineralization Processes

Depending on its frequency and kind, tillage changes the soil biophysical environment in ways that affect the net mineralization of nutrients and the release of C. These changes can be described in terms of increases or decreases in soil porosity, disruption of soil aggregates,

and redistribution in the proportion of soil aggregate size, as well as alteration of energy and water fluxes. All these changes enhance, at least temporarily, the conversion of organic C into CO_2 ^[10] and the net release of nutrients from soil organic matter (SOM). Much of the success of past agricultural practices relied heavily on the control of decomposition processes through tillage operations to satisfy plant nutrient demands. All these came at a price, however, for a heavy reliance on soil nutrients to feed crops without proper replenishment led to the worldwide declines of SOM.^[11]

Soil Erosion Processes

Agricultural ecosystems normally experience soil losses at rates considerably greater than natural ecosystems because of an incomplete plant or residue cover of the soil during rainy or windy conditions. When surface and environmental conditions are right (i.e., bare soil, sloping land, intense rain, and windy weather), the kinetic energy embedded in wind and water is transferred to soil aggregates causing them to be detached and transported away from their original position across fields or downhill. Besides the physical loss of soil particles and on-site impact on soil productivity, the detachment and transport processes also cause aggregate breakdown, thereby exposing labile C to microbial

activity. This aggregate breakdown also facilitates the preferential removal of soil materials comprised mainly of humus and clay or silt fractions. Consequently, water- and wind-borne sediments become enriched in C with respect to the contributing soil. C enrichment ratios ranging from 3 to 360 have been reported.^[12,13] The fate of these C-enriched sediments is not well known, for while transport and burial of C in eroded sediments may lead to “sequestration,”^[14] it may also result in part of it being emitted back to the atmosphere as CO₂.^[15]

RESTORING SOM: THE EMERGING SCIENCE OF SOIL C SEQUESTRATION

The Role of Long-Term Field Experiments

SOM is an essential attribute of soil quality^[16] and has an essential role in soil conservation and sustainable agriculture. Many practices—some involving land use changes—have been shown to increase SOM and thus received considerable attention for their possible role in climate change mitigation.^[17–19] C sequestration in managed soils occurs when there is a net removal of atmospheric CO₂ because C inputs (non-harvestable net primary productivity) are greater than C outputs (soil respiration, C costs related to fossil fuels and fertilizers). Soil C sequestration has the additional appeal that all its practices conform to the principles of sustainable agriculture (e.g., reduced tillage, erosion control, diversified cropping systems, improved soil fertility). Long-term field experiments have been instrumental to increase our understanding of SOM dynamics.^[20,21] The first and longest standing experiment was started at Rothamsted, England, by J.B. Lawes and J.H. Gilbert who in 1843 began documenting the impact of nutrient manipulation on crop yields and soil properties.^[22]

Other experiments were initiated thereafter in the United States, Europe, and Oceania with the goal of discovering interactions among climate, soil, and management practices. The knowledge that emerged from these experiments has been instrumental for the development and testing of agroecosystem and SOM models.^[23]

The Global Importance of Soil C Sequestration

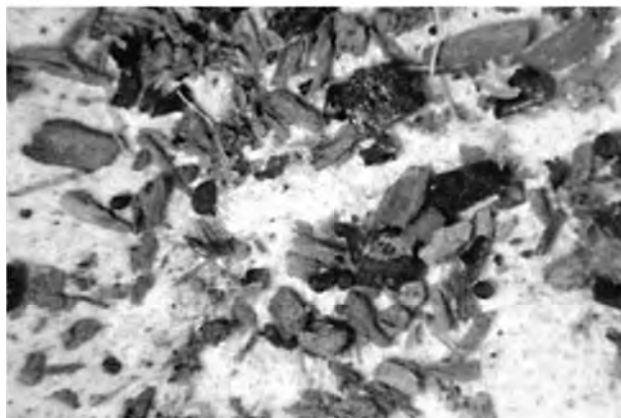
There appears to be a significant opportunity for managed ecosystems to act as C sinks; e.g., results from inverse modeling experiments suggest that during 1988–1992, terrestrial ecosystems may have been sequestering atmospheric C at rates of 1–2.2 Pg y^{−1}.^[24] Some of the likely causes include the growth of new forest in previously cultivated land^[25] and the “CO₂ fertilization effect.”^[26] Globally, agricultural soils have been estimated to have the capacity to sequester C at a rate of 0.6 Pg y^{−1}^[11] during several decades. The realization of this potential C sequestration would not be trivial since it would offset roughly

about one-tenth of the emissions from fossil fuels. In the United States, annual gains in soil C from improved agricultural practices have been estimated at 0.14 Pg yr^{−1}.^[25] Whether or not soil C sequestration practices are widely adopted will depend on their value relative to other C capture and sequestration technologies.

Mechanisms of Soil C Sequestration

Reviews of experimental results have contributed to organize our understanding of the environmental and management controls of soil C sequestration in grassland^[27,28] and agricultural^[29,30] ecosystems. The use of C balance, soil fractionation, and isotope techniques have been instrumental to reveal how new C (from crop residues, roots, and organic amendments) enters soil, resides shortly (for a few years) in labile soil fractions, and finally becomes a long-time constituent (for hundreds of years) of recalcitrant organo-mineral complexes.^[31] Fig. 1 contrasts young

(A)



(B)



Fig. 1 Young organic matters (fine roots and other organic debris) extracted from two Cryoboralfs under cereal cropping for 13 years receiving N at annual rates of 0 kg ha^{−1} (A) and 50 kg ha^{−1} (B). The black material is charcoal.

Source: From Solberg, Nyborg, et al.^[32]

Table 1 Examples of worldwide land use and management impacts on SOC.

Region/country	Climate	Soil	Duration	Crop/land use	Treatment	SOC (kg m ⁻²)		Depth (cm)	References
						Initial	Final		
Argentina	Temperate humid	Argiudoll	17	Corn–wheat–soybean	Moldboard plow	No tillage	4.95 5.46	20	[37]
Chaco, Argentina	Subtropical semiarid	Alfisol	20	Highly restored			7.05	20	[6]
			10	Moderately restored			3.10		
			60	Highly degraded			1.50		
Rondonia, Brazil	Tropical humid			Forest			4.33	50	[38]
			5	Pasture			5.85		
			9	Pasture			5.26		
			20	Pasture			5.28		
			41	Pasture			6.56		
			81	Pasture			6.12		
Georgia, U.S.A.	Temperate humid	Hapludult	5	Bermudagrass	Unharvested	1.39	1.74	6	[39]
					Lightly grazed		2.01		
					Heavily grazed		2.00		
					Hayed		1.59		
Kentucky	Temperate humid	Paleudalf	20	Conventional tillage corn	0 kg N/ha ⁻¹		4.89	30	[40]
					84 kg N/ha ⁻¹		5.63		
					168 kg N/ha ⁻¹		5.64		
					336 kg N/ha ⁻¹		6.14		
				No till corn	0 kg N/ha ⁻¹		5.54		
					84 kg N/ha ⁻¹		5.84		
					168 kg N/ha ⁻¹		5.89		
					336 kg N/ha ⁻¹		6.63		
Kutztown, Pennsylvania	Temperate humid	Fragiudalf	15	Corn	Conventional	4.20	4.30	15	[35]
					Organic	4.40	5.00		
					Manure	4.10	5.30		
Michigan	Cool temperate humid	Hapludalf	7	No crops—natural succession	Tillage		2.06	15	[41]
					Control		2.22		
Swift Current, Canada	Cold semiarid	Haploboroll	10	Cont. wheat	Minimum tillage	3.05	3.52	15	[42]
				Fallow–wheat–wheat		2.99	3.34		

(Continued)

Table 1 Examples of worldwide land use and management impacts on SOC. (Continued)

Region/country	Climate	Soil	Duration	Crop/land use	Treatment	SOC (kg m ⁻²)		Depth (cm)	References
						Initial	Final		
Breton, Canada	Cold subhumid	Cryoboralf	51	Green manure–wheat–wheat		2.89	3.22		
				Wheat–fallow	Nil	2.64	1.81	15	[34]
					Fertilizer		2.13		
					Manure		3.11		
				Wheat–oat–barley–hay–hay	Nil		2.91		
Russia	Cool temperate humid	Mollisol	300		Fertilizer		3.37		
					Manure		4.32		
				Native grassland		2.07		50	[5]
				Hay		2.13			
				Continuous cropping		1.59			
Punjab, India	Subtropical subhumid	Alluvial	50	Continuous fallow		1.51			
			6	Corn–wheat	Minimum tillage, residue retained	0.48		15	[43]
					Minimum tillage, residue removed	0.48			
					Conventional tillage	0.50			
					Conventional tillage	3.20	3.73	20	[44]
Morocco	Warm temperate semiarid	Calcixeroll	11	Continuous wheat and other rotations	No tillage		3.39		
			20	Bush fallow		2.77		15	[45]
			25	Bush fallow		2.96			
			10	Bermudagrass		3.90			
			10	Cultivation		1.29			
Western Nigeria									

(labile) organic matter fractions extracted from two cultivated soils with and without nitrogen (N) fertilizer.^[32] The amounts of labile organic matter—fine roots and other organic debris—present in each soil reflect differences in crop productivity induced by the addition of N at annual rates of 50 kg ha⁻¹ for 13 years. “Terra Preta” soil—in tropical regions of South America and West Africa—represents a prime example of ancient wisdom applied to develop sustainable agriculture through the improvement of soil fertility and SOM.^[33]

The quantity and quality of C entering soil as well as the interaction of this C with the soil biophysical environment are major factors determining the rate and duration of soil C sequestration. The quantity of C added to soil in the form of roots, crop residues, and organic amendments has been shown to play a dominant role in defining the trajectory of SOC over time.^[34] Management practices geared toward optimizing nutrient supply and building nutrient reserves (e.g., fertilization and the use of legumes in crop rotations) are almost guaranteed to increase soil C stocks. The quality of crop residues and the timing of their incorporation to soil also have an influential role on C decomposition and, thus, on soil C storage.^[35] The degree of soil disturbance—through its impact on soil aggregation—constitutes another major factor regulating C decomposition and retention in soil.^[36] In this context, no tillage agriculture has come to represent one of the most significant technological innovations of the last years because it allows farmers the possibility of growing crops economically while reducing erosion and improving both quantity and quality of SOM. A few examples of the management impacts on soil C sequestration from around the world are presented in Table 1.

SOM, Energy, and Full C Accounting

Land is the natural habitat of humans. Humans dwell on it and use it as a resource for the production of food, fiber, and other goods. Simply put, land is managed when there is manipulation of energy and matter flows in order to meet certain economic and social objectives. Farm mechanization and fertilizers are two of the many technical innovations that—though they rely on the utilization of fossil energy—have brought dramatic increases in food production during the 20th century. Changes in management practices that include soil C sequestration as an objective require careful evaluation of their impact not only on soil C gains but also on C costs from the use of fossil energy (e.g., manufacture of fertilizers)^[46,47] and on the net greenhouse gas emissions.^[48]

THE ROLE OF SOM IN THE 21ST CENTURY

SOM has played and will continue to play a central role in sustainable land management. The restoration of SOM at

global scales offers a unique opportunity to mitigate global warming. As population levels and affluence increase, demands on land to produce food, fiber, biomass, and other products will remain high. Because land is finite, important decisions will have to be made in order to balance such demands with functional objectives such as the preservation of natural ecosystems. As part of any climate policy, the impact of land use changes and management on SOM storage should be included as a criterion for making these decisions. Depending on their degree of expansion, several evolving agricultural technologies—such as genetically modified crops, conservation tillage, organic farming, and precision farming—may have important implications for soil C sequestration.^[19] Their ultimate impact on C sequestration will depend not only on the economic benefit realized by individual producers but also on whether society recognizes the value of soil C storage to mitigate global warming.

REFERENCES

- Davidson, E.A.; Ackerman, I.L. Changes in soil carbon following cultivation of previously untilled soils. *Biogeochemistry* **1993**, *20*, 161–193.
- Dalal, R.C.; Mayer, R.J. Long-term trends in fertility of soils under continuous cultivation and cereal cropping in southern Queensland. II. Total organic carbon and its rate of loss from the soil profile. *Aust. J. Soil Res.* **1986**, *24*, 281–292.
- Mann, L.K. Changes in soil carbon after cultivation. *Soil Sci.* **1986**, *142*, 279–288.
- Ellert, B.H.; Gregorich, E.G. Storage of carbon, nitrogen and phosphorus in cultivated and adjacent forested soils of Ontario. *Soil Sci.* **1996**, *161*, 587–603.
- Mikhailova, E.A.; Bryant, R.B.; Vassenev, I.I.; Schwager, S.J.; Post, C.J. Cultivation effects on soil carbon and nitrogen contents at depth in the Russian chernozem. *Soil Sci. Soc. Am. J.* **2000**, *64*, 738–745.
- Abril, A.; Bucher, E.H. Overgrazing and soil carbon dynamics in the western Chaco of Argentina. *Appl. Soil Ecol.* **2001**, *16*, 243–249.
- Lal, R. Deforestation and land use effects on soil degradation and rehabilitation in western Nigeria, II. Soil chemical properties. *Land Degrad. Dev.* **1996**, *7*, 87–98.
- Lobe, I.; Amelung, W.; Du Preez, C.C. Losses of carbon and nitrogen with prolonged arable cropping from sandy soils of the South African Highveld. *Eur. J. Soil Sci.* **2001**, *52*, 93–101.
- Houghton, R.A. Changes in the storage of terrestrial carbon since 1850. In *Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1995; 45–65.
- Reicosky, D.C.; Lindstrom, M.J. Fall tillage method: Effect from short-term carbon dioxide flux from soil. *Agron. J.* **1993**, *85*, 1237–1243.
- Cole, V.; Cerri, C.; Minami, K.; Mosier, A.; Rosenberg, N.J.; Sauerbeck, D. Agricultural options for mitigation of greenhouse gas emissions. In *Climate Change 1995: Impacts*,

- Adaptations and Mitigation of Climate Change*; Watson, R.T., Zinyowera, M.C., Moss, R.H., Eds.; Report of IPCC Working Group II; Cambridge University Press: London, 1996; 745–771.
12. Sterk, G.; Herrmann, L.; Bationo, A. Wind-blown nutrient transport and soil productivity changes in southwest Niger. *Land Degrad. Dev.* **1996**, *7*, 325–336.
 13. Zobeck, T.M.; Fryrear, D.W. Chemical and physical characteristics of wind-blown sediment. *T. Am. Soc. Agr. Eng.* **1986**, *29*, 1037–1041.
 14. Stallard, R.F. Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial. *Global Biogeochem. Cy.* **1998**, *12*, 231–257.
 15. Lal, R. Global soil erosion by water and C dynamics. In *Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1995; 131–141.
 16. Doran, J.W.; Coleman, D.C.; Bezdicsek, D.F.; Stewart, B.A., Eds. *Defining Soil Quality for a Sustainable Environment*; Soil Science Society America Special Publication No. 35; SSSA: Madison, 1994; 244 pp.
 17. Batjes, N.H. Mitigation of atmospheric CO₂ concentrations by increased carbon sequestration in the soil. *Biol. Fert. Soils* **1998**, *27*, 230–235.
 18. Post, W.M.; Kwon, K.C. Soil carbon sequestration and land-use change: Processes and potential. *Glob. Change Biol.* **2000**, *6*, 317–327.
 19. Izaurrealde, R.C.; Rosenberg, N.J.; Lal, R. Mitigation of climatic change by soil carbon sequestration: Issues of science, monitoring and degraded lands. *Adv. Agron.* **2001**, *70*, 1–75.
 20. Powlson, D.S.; Smith, P.; Smith, J.U., Eds. *Evaluation of Soil Organic Matter Models Using Existing Long-Term Datasets*; NATO ASI Series I; Springer: Heidelberg, 1996; Vol. 38, 429 pp.
 21. Paul, E.A.; Paustian, K.; Elliott, E.T.; Cole, C.V., Eds. *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*; NATO ASI Series I; CRC/Lewis Publishers: Boca Raton, 1997; 414 pp.
 22. Jenkinson, D.S. The Rothamsted long-term experiments: Are they still of use? *Agron. J.* **1991**, *83*, 2–10.
 23. Smith, P.; Smith, J.U.; Powlson, D.S.; McGill, W.B.; Arah, J.R.M.; Chertov, O.; Coleman, K.W.; Franko, U.; Frolking, S.; Jenkinson, D.S.; Jensen, L.S.; Kelly, R.H.; Klein-Gunnewiek, H.; Komarov, A.S.; Li, C.; Molina, J.A.E.; Mueller, T.; Parton, W.J.; Thornley, J.H.M.; Whitmore, A.P. A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments. *Geoderma* **1997**, *81*, 153–225.
 24. Fan, S.; Gloor, M.; Mahlman, J.; Pacala, S.; Sarmiento, J.; Takahashi, T.; Tans, P. A large terrestrial carbon sink in North America implied by atmospheric and oceanic carbon dioxide data and models. *Science* **1998**, *282*, 442–446.
 25. Houghton, R.A.; Hackler, J.L.; Lawrence, K.T. The U.S. carbon budget: Contributions from land use change. *Science* **1999**, *285*, 574–578.
 26. Kimball, B.A. Carbon dioxide and agricultural yield: An assemblage and analysis of 430 prior observations. *Agron. J.* **1983**, *75*, 779–782.
 27. Conant, R.T.; Paustian, K.; Elliott, E.T. Grassland management and conversion into grassland: Effects on soil carbon. *Ecol. Appl.* **2001**, *11*, 343–355.
 28. Scott, N.A.; Tate, K.R.; Ford-Robertson, J.; Giltrap, D.J.; Smith, C.T. Soil carbon storage in plantations and pastures: Land-use implications. *Tellus* **1999**, *51B*, 326–335.
 29. Janzen, H.H.; Campbell, C.A.; Izaurrealde, R.C.; Ellert, B.H.; Juma, N.; McGill, W.B.; Zentner, R.P. Management effects on soil C storage on the Canadian prairies. *Soil Till. Res.* **1998**, *47*, 181–195.
 30. Paustian, K.; Collins, H.P.; Paul, E.A. Management controls on soil carbon. In *Soil Organic Matter in Temperate Ecosystems: Long-Term Experiments in North America*; Paul, E.A., Paustian, K., Elliott, E.T., Cole, C.V., Eds.; CRC/Lewis Publishers: Boca Raton, 1997; 15–49.
 31. Jastrow, J.D. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. *Soil Biol. Biochem.* **1996**, *28*, 665–676.
 32. Solberg, E.D.; Nyborg, M.; Izaurrealde, R.C.; Malhi, S.S.; Janzen, H.H.; Molina-Ayala, M. Carbon storage in soils under continuous cereal grain cropping: N fertilizer and straw. In *Management of Carbon Sequestration in Soil*; Lal, R., Kimble, J., Follett, R., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1998; 235–254.
 33. Glaser, B.; Haumaier, L.; Guggenberger, G.; Zech, W. The “Terra Preta” phenomenon: A model for sustainable agriculture in the humid tropics. *Naturwissenschaften* **2001**, *88*, 37–41.
 34. Izaurrealde, R.C.; McGill, W.B.; Robertson, J.A.; Juma, N.G.; Thurston, J.T. Carbon balance of the Breton classical plots after half a century. *Soil Sci. Soc. Am. J.* **2001**, *65*, 431–441.
 35. Drinkwater, L.E.; Wagoner, P.; Sarrantonio, M. Legume-based cropping systems have reduced carbon and nitrogen losses. *Nature* **1998**, *396*, 262–265.
 36. Six, J.; Elliott, E.T.; Paustian, K. Soil macroaggregate turnover and microaggregate formation: A mechanism for C sequestration under no-tillage agriculture. *Soil Biol. Biochem.* **2000**, *32*, 2099–2103.
 37. Alvarez, R.; Russo, M.E.; Prystupa, P.; Scheiner, J.D.; Blotta, L. Soil carbon pools under conventional and no-tillage systems in the Argentine rolling pampa. *Agron. J.* **1998**, *90*, 138–143.
 38. Neill, C.; Cerri, C.C.; Melillo, J.M.; Feigl, B.J.; Steudler, P.A.; Moraes, J.F.L.; Piccolo, M.C. Stocks and dynamics of soil carbon following deforestation for pasture in Rondônia. In *Soil Processes and the Carbon Cycle*; Lal, R., Kimble, J., Follett, R., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1998; 9–28.
 39. Franzluebbers, A.J.; Stuedemann, J.A.; Wilkinson, S.R. Bermudagrass management in the southern piedmont USA: I. Soil and surface residue carbon and sulfur. *Soil Sci. Soc. Am. J.* **2001**, *65*, 834–841.
 40. Ismail, I.; Blevins, R.L.; Frye, W.W. Long-term no-tillage effects on soil properties and continuous corn yields. *Soil Sci. Soc. Am. J.* **1994**, *58*, 193–198.
 41. Richter, D.D.; Babbar, L.I.; Huston, M.A.; Jaeger, M. Effects of annual tillage on organic carbon in a fine-textured Udalf: The importance of root dynamics to soil carbon storage. *Soil Sci.* **1999**, *149*, 78–83.

42. Curtin, D.; Wang, H.; Selles, F.; McConkey, B.G.; Campbell, C.A. Tillage effects on carbon fluxes in continuous wheat and fallow–wheat rotations. *Soil Sci. Soc. Am. J.* **2000**, *64*, 2080–2086.
43. Ghuman, B.S.; Sur, H.S. Tillage and residue management effects on soil properties and yields of rainfed maize and wheat in a subhumid subtropical climate. *Soil Till. Res.* **2001**, *58*, 1–10.
44. Mrabet, R.; Saber, N.; El-Brahli, A.; Lahlou, S.; Bessam, F. Total, particular organic matter and structural stability of a calcixeroll soil under different wheat rotations and tillage systems in a semiarid area of Morocco. *Soil Till. Res.* **2001**, *57*, 225–235.
45. Lal, R. Land use and soil management effects on soil organic matter dynamics on alfisols in western Nigeria. In *Soil Processes and the Carbon Cycle*; Lal, R., Kimble, J., Follett, R., Stewart, B.A., Eds.; CRC/Lewis Publishers: Boca Raton, 1998; 109–126.
46. Schlesinger, W.H. Carbon and agriculture: Carbon sequestration in soils. *Science* **1999**, *284*, 2095.
47. Izaurrealde, R.C.; McGill, W.B.; Rosenberg, N.J. Carbon cost of applying nitrogen fertilizer. *Science* **2000**, *288*, 811–812.
48. Robertson, G.P.; Paul, E.A.; Harwood, R.R. Greenhouse gases in intensive agriculture: Contributions of individual gases to the radiative forcing of the atmosphere. *Science* **2000**, *289*, 1922–1925.

Soil Organic Matter (SOM): Mass Spectrometry Data Analysis

Nikola Tolic

Malak M. Tfaily

Yufeng Shen

Rosalie Chu

Thomas Fillmore

Rui Zhao

Kent Bloodsworth

Errol Robinson

Ljiljana Paša-Tolić

Nancy J. Hess

*Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory,
Richland, Washington, U.S.A.*

Abstract

Considered a general purpose tool for characterization of samples at the molecular level in environmental and life sciences, mass spectrometry (MS) is commonly used for measuring organic matter in air, water, and soil. Broad base of commercially available instrumentation and data analysis tools provide functional analytical platforms for molecular characterization for a variety of applications and samples. MS instrumentation with ultrahigh-resolving power ($m/\Delta m_{\text{FWHM}} > 200,000$) is necessary for broad identification of thousands of molecular components present in organic matter samples. This entry discusses basic principles of data analysis for ultrahigh-resolution MS measurements of small molecules of organic matter.

INTRODUCTION

Elemental composition analysis of natural organic matter (NOM) has a broad set of applications in agriculture, environment, energy, human health, forensics, and warfare, just to name a few. Regardless of whether the objective is broad elemental composition characterization or targeted inquiry for chemical or biomarkers, mass spectrometry (MS) is becoming the dominant analytical platform for rapid survey of NOM at the molecular level. Bibliometric study of MS usage in research articles over the last three decades from 1980 to 2010 of analytical chemistry shows trends of sharp increase across wide range of scientific disciplines including earth and environmental sciences.^[1] Each “omics” conquest by MS strengthens analytical capabilities of MS through development of new methods, instrumentation, and computational tools. Fourier transform ion cyclotron resonance mass spectrometry (FT-ICR MS) is the MS platform offering the highest mass resolving power and accuracy, allowing unique determination of elemental composition from mass measurement alone, for small (<500 Da) organic molecules.^[2]

NOM samples differ in source (atmosphere, hydrosphere, and geosphere), complexity (water, sediments,

aerosol, permafrost, agricultural soil, petroleum, etc.), and applied extraction methods (liquid solvents, ionic liquids, and supercritical fluids) all of which can influence selection for combining chromatography, MS instrumentation, and ionization methods for deeper analyte coverage. Electrospray ionization (ESI) coupled with FT-ICR MS is the optimal and most utilized technique for NOM analysis; a review by Sleighter and Hatcher^[3] offers more detailed information on ionization methods, application of positive and negative ion modes, and interpretation of MS spectra from NOM samples.

DATA PROCESSING OF ULTRAHIGH MASS SPECTRA

Mass spectrometers work by measuring changes in ion current due to fundamental ion properties, i.e., mass, charge, and abundance, resulting in an ion mass-to-charge ratio (“m/z”) and count (intensity) pair. A mass spectrum is a 2-D plot correspondent to measured ions m/z and intensity pairs, commonly referred as peaks. Thousands of peaks representing different molecular species could be measured and identified in a single ESI FT-ICR spectrum from NOM

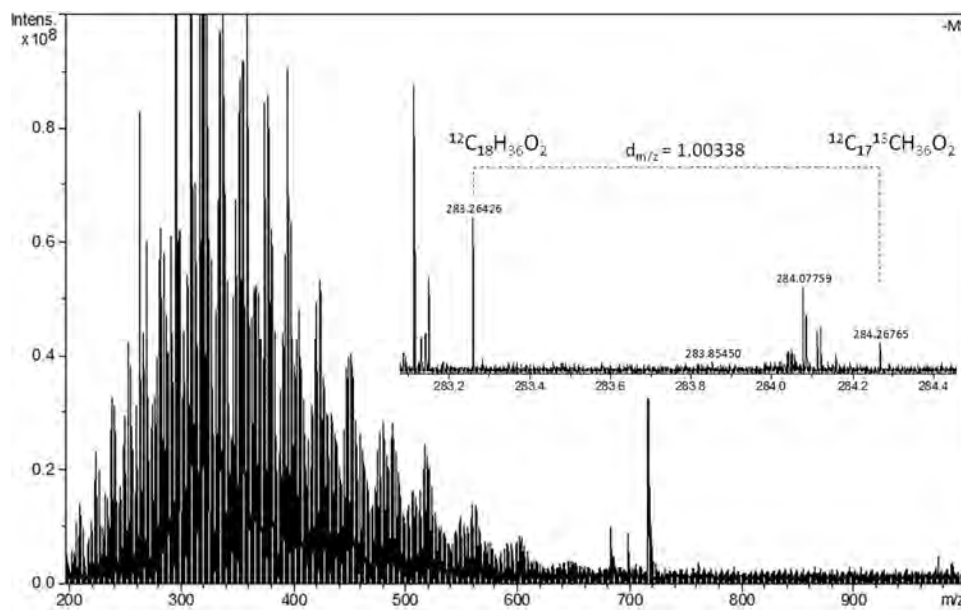


Fig. 1 Negative mode mass spectra from 12T FT-ICR instrument from mud pore water with more than 7,000 resolved peaks in m/z range 200–1,000 with signal-to-noise better than 5 and average peak resolution of 310,000. Inset illustrates distance and precision at which isotopic peaks are located from the mono-isotopic peak.

sample as illustrated in Fig. 1. Measuring samples in both positive and negative ion mode significantly increases coverage for NOM samples that usually contain both basic and acidic molecular species preferentially ionized in the positive and negative modes, respectively. Positive mode spectra are more complex as they contain both hydrogen (H_2) and sodium adducts, making them more challenging for interpretation, i.e., chemical formula assignment. Instrument manufacturers usually provide specialized software functions for low-level signal processing as well as graphical spectrum annotation tools. Preparing the instrumentation for MS experiment according to the “best practice” guide^[4] requires external calibration of the mass spectrometer using standard mixtures as close in time as possible to the actual sample measurements. This is a

necessary step for testing of instrument performance and stability. After low-level peak picking, a list of peaks is internally calibrated to assure necessary mass measurement accuracy (MMA) for molecular formula assignment. Careful and accurate execution of these steps is of paramount importance; missteps at this stage could invalidate or provide misleading results and conclusions. Fortunately, NOM samples often contain known classes of compounds (humics, fatty acids, metabolic products, and contaminants) ideal for targets of internal calibration over wide m/z range. Once m/z values are calibrated to sub-ppm error values, the molecular formula assignment procedure can start. Most data analysis workflows start with low m/z values where unique formula assignments are possible and then extend the formula to higher m/z values based on expected

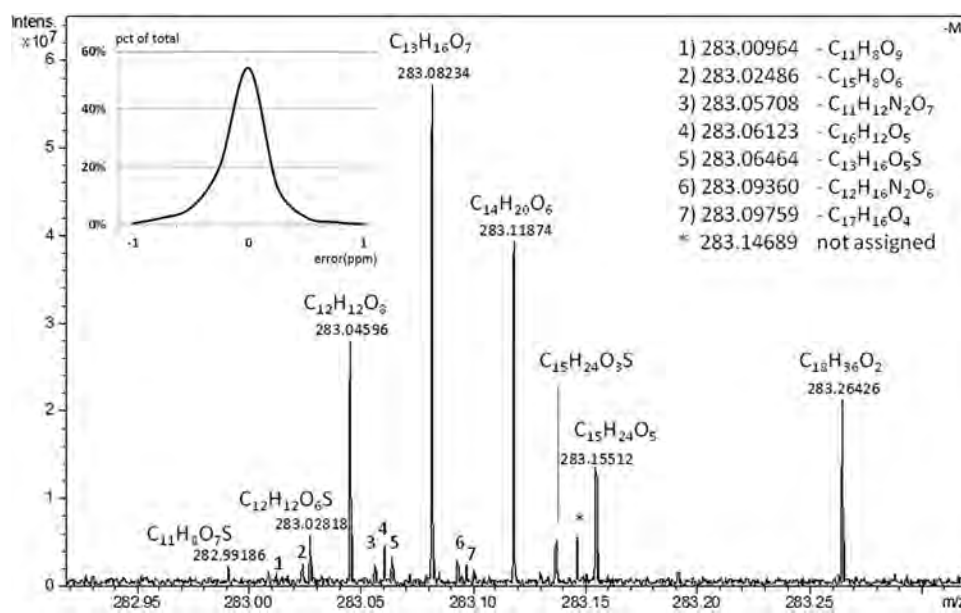


Fig. 2 Assigned molecular formula with mass error within 0.5 ppm at nominal peak 283 using CIA software for spectrum in Fig. 1. In total, more than 5,900 (~84%) molecular formula including isotopic ^{13}C peaks are assigned. Inset shows mass error distribution for assignment across full range.

relationships in the composition of NOM mixture. Adding basic building blocks [methylene (CH_2), H_2 , oxygen (O), etc.] on already assigned formula, amounts to precisely evaluating spacing between peaks and minimizes the necessity for direct formula assignment. This technique of “formula propagation” using basic building blocks is best illustrated using plots exploring the concept of Kendrick mass and defect (KMD) analysis.^[5] Briefly, Kendrick mass is a scaling transformation such that CH_2 assumes integer mass 14 [instead of IUPAC (International Union of Pure and Applied Chemistry) mass of 14.01565] and mass defect 0. Literally, Kendrick mass transformation straightens skewed IUPAC mass defect plots for formulae differing by CH_2 into easily identifiable horizontal lines (Fig. 3A). This simple transformation allows visual analysis of related compound classes based on heteroatom composition and spotting of isolated outlier “identifications,” leading Marshall and coworkers to propose adoption of 2-D KMD plots as standard visualization technique when presenting results from NOM measurements.^[8] Extension of the method using higher order of mass defect analysis was proposed by Roach et al. allowing resolution of compounds with close KMD due to secondary and tertiary compositional differences.^[9]

Measuring organic matter is related to understanding and modeling dynamics of carbon (C) cycling that necessitates increased throughput and automation of data analysis steps. Compound identification algorithm (CIA) developed by Kujawinski and Behn^[10] at Woods Hole Oceanographic Institute is the only general, fully implemented, automated, and freely available formula assignment software for organic matter from high-resolution mass spectra. CIA, implemented in Matlab (MathWorks) programming environment, relies on sub-ppm MMA of FT-ICR MS spectra and precalculated database of molecular formula comprising of

elements C, H, nitrogen (N), O, phosphorus (P), and sulfur (S). The database includes molecular formulae in range of 150–1,500 Da compiled according to “seven golden rules” for filtering chemically plausible molecular formula.^[11] Relationships between peaks are established based on common functional groups and building blocks (CH_2 , H_2 , and O, extendable to others). Formula assignment starts from low m/z range and search for database matches with 1 ppm MMA, if peak assignment is not already established based on relationship with other peaks through specified basic building blocks. In case of ambiguity formula, selection is based on lowest number of non-oxygen heteroatoms N, S, and P. Tuning of algorithm using non-oxygen heteroatom filters and selected formula propagation rules based on prior knowledge about sample or extraction protocols could reduce false positive assignments; for example, limiting numbers of N and S atoms in aerosol samples or requiring certain number of O atoms for each P in soil samples. Algorithm is independent of MS instrumentation, supports both positive and negative ion mode, and includes alignment between different spectra for robust estimation of spectral similarity. CIA is available through source code public repository GitHub.^[10] Replacing universal CIA database with specialized databases tailored for particular NOM samples, e.g., petroleum, lignin, or metabolome, should be in a straightforward manner, offering opportunity for both more confident and faster formula assignments.

RESULTS INTERPRETATION AND VISUALIZATION

Once MS peaks are assigned with molecular formulae, the interpretation of results represents its own challenges. Various indexes based on elemental composition counts

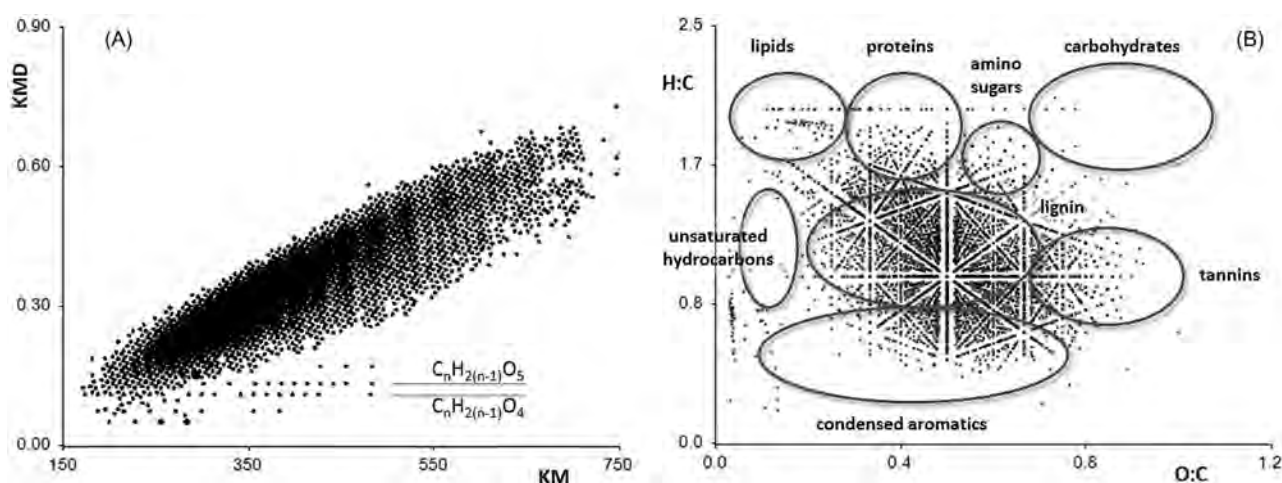


Fig. 3 Kendrick mass defect (A) and van Krevelen (B) plots are most utilized diagrams for interpretation of NOM results from MS measurements. Based on spectrum in Fig. 1, mass defect plot illustrates homologous series represented with straight lines. In van Krevelen plot, expected location of biomolecule classes is indicated with overlaid shapes.

Source: Compiled from Sleighter & Hatcher,^[3] Mann, Chen, et al.,^[6] and Van Krevelen.^[7]

and ratios are used to introduce fuzzy structural information, categorize results, and draw research conclusions. Double-bond equivalent or rings-plus-double-bond equivalent is an established index calculated directly from molecular formula to describe unsaturation and rings in molecule and is often presented in research articles plotted against C-number or H:C ratio.^[11] Similarly, aromaticity index is developed as an identification measure for aromatics and condensed aromatics based on their density of double C–C bonds obtainable from molecular formula.^[12] CHO index is constructed on the concept of oxidation state of organic C offering a better measure of NOM degradation potential.^[6] Some plot variants of these indexes are often found in MS-based NOM research manuscripts, but it would be unusual to find one without a visual representation using van Krevelen diagram. Although initially developed for assessment of structures and processes in various coal samples, van Krevelen diagram^[7] became an unavoidable tool for graphical interpretation of formula assignments from NOM samples. Original van Krevelen diagrams display 2-D plane scatter plots of molecular formulae with X,Y axis measuring O:C and H:C ratios, respectively. Application of van Krevelen diagram on general NOM samples led to mapping of regions of the diagram plane to different classes of organic compounds (Fig. 3B), providing useful insight into NOM structural characteristics.^[13] Three-dimensional variation of the plots was proposed by adding N:C ratio as an additional coordinate axis, effectively allowing visual separation of assigned formula according to the O and N classes.^[14]

CHALLENGES, PROSPECTS, AND FUTURE DIRECTIONS

Quantitative analysis of sample absolute abundance is one of the longest standing challenges with ESI-MS application to complex samples due to the vast differences in ionization efficiencies of analyte components. At the moment, ESI-MS is considered a non-quantitative measure for characterization of complex mixtures, i.e., measured signal intensity does not correlate in a simple or well understood manner with actual analyte concentrations in sample due to the charge competition in electrospray. Most of the attempted solutions propose calibration of ionization sensitivity based on standard, e.g., organic acids, introduced together with analyte; however, since signal suppression occurs in the ionization stage, this approach offers questionable results.

Another serious disadvantage of MS as standalone platform is that MS instrumentation cannot obtain structural information for isomeric ions from a shotgun MS experiment. In tandem mass spectrometry (MS/MS), the parent molecule with a narrow m/z range is fragmented in a secondary MS event using miscellaneous dissociation

methods. The collected fragment ions are then used to infer parent molecule structural information. Application of MS/MS is mostly limited to specialized NOM components like lipids or metabolome products where isolation of ions for fragmentation is not as challenging as for general NOM samples.^[3] Using nuclear magnetic resonance (NMR) in conjunction with MS is suggested as the ultimate technique when full structural information is required; however, at the moment, it lacks throughput for broad NOM characterization, again due to complexity of typical NOM samples. Structural determination by NMR would require deep fractionation of the NOM sample to reduce the number of organic molecules present.

Advancement in FT-ICR instrumentation with higher magnetic fields (two 21T FT-ICR systems are at preproduction in National High Magnetic Field Laboratory in Tallahassee, Florida, U.S.A., and Pacific Northwest National Laboratory, Richland, Washington, U.S.A.) will offer even higher sensitivity and necessary resolving power for going significantly higher than 500 Da boundary for unique elemental composition determination from mass only, set by Kim, Rodgers, and Marshall.^[2] Novel computational tools based on these spectra are expected to make use of fine isotopic structure resolving ambiguities and allowing deeper coverage and direct formula assignments to challenging classes of important environmental components such as organometallics. With growing interest of the research community, one can expect continuous technology advances as well as the construction of MS spectral libraries with searchable metadata information allowing comparison of NOM across geochemical, ecosystem, and temporal gradients. Accumulated knowledge and computational infrastructure will contribute to building better diagnostic and predictive models of the C cycle feedback loops to climate, facilitating resolution of pressing sustainable use of resources and climate change challenges.

CONCLUSION

Ultrahigh resolution MS provides a general purpose analytical platform for wide characterization of organic matter in complex environmental samples. Sub-ppm MMA allows direct molecular formula assignment for small NOM molecules, an important step for better understanding of sample composition and reactivity. Utilization of fragmentation methods and isotopic signatures should further increase reliability of formula assignments and provide insights into NOM structural properties. An ever increasing volume of acquired data presents both challenges and opportunities for development of new data mining algorithms and visualization functions. Continuous advances in FT-ICR instrumentation, extraction protocols, ionization and fragmentation methods, and data processing tools will increase the utilization of ultrahigh-resolution MS in soil and other environmental sciences.

ACKNOWLEDGMENTS

We want to acknowledge Kelly Wrighton from the Ohio State University for providing samples for MS measurements used in figures. MS measurement was performed using EMSL, a DOE Office of Science User Facility sponsored by BER and located at Pacific Northwest National Laboratory (PNNL). PNNL is operated by Battelle for Department of Energy.

REFERENCES

1. Waaijer, C.J.F.; Magnus, P. Bibliometric mapping: Eight decades of analytical chemistry, with special focus on the use of mass spectrometry. *Anal. Chem.* **2015**, *87* (9), 4588–4596.
2. Kim, S.; Rodgers, R.P.; Marshall, A.G. Truly “exact” mass: Elemental composition can be determined uniquely from molecular mass measurement at ~ 0.1 mDa accuracy for molecules up to ~ 500 Da. *Int. J. Mass Spectrom.* **2006**, *251* (2), 260–265.
3. Sleighter, R.L.; Hatcher, P.G. *Fourier Transform Mass Spectrometry for the Molecular Level Characterization of Natural Organic Matter: Instrument Capabilities, Applications, and Limitations*; INTECH Open Access Publisher, 2011.
4. Webb, K.; Bristow, T.; Sargent, M.; Stein, B. *Methodology for Accurate Mass Measurement of Small Molecules. Best Practice Guide*; LGC Limited: Teddington, 2004; 1–8.
5. Kendrick, E. A mass scale based on $\text{CH}_2 = 14.0000$ for high resolution mass spectrometry of organic compounds. *Anal. Chem.* **1963**, *35* (13), 2146–2154.
6. Mann, B.F.; Chen, H.; Herndon, E.M.; Chu, R.K.; Tolic, N.; Portier, E.F.; Chowdhury, T.R.; Robinson, E.W.; Callister, S.J.; Wullschlegel, S.D.; Graham, D.E.; Liang, L.; Gu, B. Indexing permafrost soil organic matter degradation using high-resolution mass spectrometry. *Plos One* **2015**, *10* (6), e0130557.
7. Van Krevelen, D.W. Graphical-statistical method for the study of structure and reaction processes of coal. *Fuel* **1950**, *29* (12), 269–284.
8. Hughey, C.A.; Hendrickson, C.L.; Rodgers, R.P.; Marshall, A.G.; Qian, K. Kendrick mass defect spectrum: A compact visual analysis for ultrahigh-resolution broadband mass spectra. *Anal. Chem.* **2001**, *73* (19), 4676–4681.
9. Roach, P.J.; Laskin, J.; Laskin, A. Higher-order mass defect analysis for mass spectra of complex organic mixtures. *Anal. Chem.* **2011**, *83* (12), 4924–4929.
10. Kujawinski, E.B.; Behn, M.D. Automated analysis of electrospray ionization Fourier transform ion cyclotron resonance mass spectra of natural organic matter. *Anal. Chem.* **2006**, *78* (13), 4363–4373. Available at <http://github.com/KujawinskiLaboratory/findformula>.
11. Kind, T.; Fiehn, O. Seven golden rules for heuristic filtering of molecular formulas obtained by accurate mass spectrometry. *BMC Bioinformatics* **2007**, *8* (1), 105.
12. Koch, B.P.; Dittmar, T. From mass to structure: An aromaticity index for high-resolution mass data of natural organic matter. *Rapid Commun. Mass Sp.* **2006**, *20*, 926–932.
13. Kim, S.; Kramer, R.W.; Hatcher, P.G. Graphical method for analysis of ultrahigh-resolution broadband mass spectra of natural organic matter, the van Krevelen diagram. *Anal. Chem.* **2003**, *75* (20), 5336–5344.
14. Wu, Z.; Rodgers, R.P.; Marshall, A.G. Two-and three-dimensional van Krevelen diagrams: A graphical analysis complementary to the Kendrick mass plot for sorting elemental compositions of complex organic mixtures based on ultrahigh-resolution broadband Fourier transform ion cyclotron resonance mass measurements. *Anal. Chem.* **2004**, *76* (9), 2511–2516.

Soil Organic Matter (SOM): Modeling

Peter Smith

*Institute of Biological and Environmental Sciences, School of Biological Sciences,
University of Aberdeen, Aberdeen, Scotland*

Abstract

There are a number of approaches to modeling soil organic matter turnover including: 1) process-based multicompartment models; 2) models that consider each fresh addition of plant debris as a separate cohort which decays in a continuous way; and 3) models that account for carbon and nitrogen transfers through various trophic levels in a soil food web. These approaches are described in more detail below.

PROCESS-BASED, MULTICOMPARTMENT SOM MODELS

Most models are process based, i.e., they focus on the processes mediating the movement and transformations of matter or energy and usually assume first order rate kinetics.^[1] Early models simulated the SOM as one homogeneous compartment.^[2] Some years later, two-compartment models were proposed^[3,4] and, as computers became more accessible, multicompartment models were developed.^[5,6] Of the 33 SOM models represented within the Global Change and Terrestrial Ecosystems (GCTE) Soil Organic Matter Network (SOMNET) database,^[7–9] 30 are multicompartment, process-based models. Each compartment or SOM pool within a model is characterized by its position in the model's structure and its decay rate. Decay rates are usually expressed by first-order kinetics with respect to the concentration (C) of the pool

$$dC/dt = -kC$$

where *t* is the time. The rate constant *k* of first-order kinetics is related to the time required to reduce by half the concentration of the pool “when there is no input.” The pool's half-life [*h* = (ln 2)/*k*], or its turnover time (*τ* = 1/*k*) are sometimes used instead of *k* to characterize a pool's dynamics: The lower the decay rate constant, the higher the half-life, the turnover time, and the stability of the organic pool.

The flows of C within most models represent a sequence of carbon going from plant and animal debris to the microbial biomass and then to soil organic pools of increasing stability. Some models also use feedback loops to account for catabolic and anabolic processes and microbial successions. The output flow from an organic pool is usually split. It is directed to a microbial biomass pool, another organic pool, and, under aerobic conditions, to CO₂. This split simulates the simultaneous anabolic and catabolic activities and growth of a microbial population feeding on one

substrate. Two parameters are required to quantify the split flow. They are often defined by a microbial (utilization) efficiency and stabilization (humification) factor which control the flow of decayed C to the biomass and humus pools, respectively. The sum of the efficiency and humification factors must be inferior to one to account for the release of CO₂. A thorough review of the structure and underlying assumptions of different process-based SOM models is available.^[6]

COHORT MODELS DESCRIBING DECOMPOSITION AS A CONTINUUM

Another approach in modeling SOM turnover is to treat each fresh addition of plant debris into the soil as a cohort.^[5] Such models consider one SOM pool that decays with a feedback loop into itself. For example, Q-SOIL^[10] is represented by a single rate equation. The SOM pool is divided into an infinite number of components, each characterized by its “quality” with respect to degradability as well as impact on the physiology of the decomposers. The rate equation for the model Q-SOIL represents the dynamics of each SOM component of quality *q* and is quality dependent. Exact solutions to the rate equations are obtained analytically.^[11]

FOOD-WEB MODELS

Another type of model simulates C and N transfers through a food web of soil organisms,^[1,12] such models explicitly account for different trophic levels or functional groups of biota in the soil.^[13–18] Some models that combine an explicit description of the soil biota with a process-based approach have been developed.^[19] Food-web models require a detailed knowledge of the biology of the system to be simulated and are usually parameterized for application at specific sites.

Table 1 Overview of SOM models represented within GCTE-SOMNET in January 2001.

Inputs								
Model	Timestep	Meteorology	Soil and plant	Management	Factors affecting decay rate constants		Soil outputs	References
ANIMO	Day, week, month	P, AT, Ir, EvW	Des, Lay, Imp, Cl, OM, N, pH	Rot, Ti, Fert, Man, Res, Irr, AtN	T, W, pH, N, O		C, N, W, ST, gas	[20]
APSIM	Day	P, AT, Ir	Lay, W, C, N, BD, Wi, PG, PS	Rot, Ti, Fert, Irr	T, W, pH, N		C, N, W, ST, gas	[21]
Candy	Day	P, AT, Ir	D, Imp, W, N, C, Wi, PD, Nup	Rot, Ti, Fert, Man, Res, Irr, AtN	T, W, N, Cl		C, N, W, ST, gas	[22]
CENTURY	Month	P, AT	W, Cl, OM, pH, C, N	Rot, Ti, Fert, Man, Res, Irr, AtN	T, W, N, Cl, pH, Ti		C, BioC, 13C, 14C, N, W, ST, gas	[23]
Chenfang Lin Model	Day	ST	OM, BD, W	Man, Res	T, W, F		C, BioC, gas	[24]
DAISY	Hour, day	P, AT, Ir, EvG	Lay, Cl, C, N, PG, PS	Rot, Ti, Fert, Man, Res, Irr, AtN	T, W, N, Cl		C, BioC, N, W, ST, gas	[25]
DNDC	Hour, day, month	P, AT	Lay, Cl, OM, pH, BD	Rot, Ti, Fert, Man, Res, Irr, AtN	T, W, N, Cl, Ti		C, BioC, N, W, ST, gas	[26]
DSSAT	Hour, day, month, year	P, AT, Ir	Des, Lay, Imp, W, Cl, PS, OM, pH, C, N	Rot, Ti, Fert, Man, Res, Irr	T, W, N, Cl, Ti		C, BioC, N, W, ST	[27]
D3R	Day	P, AT	Y, PS	Rot, Ti, Res	T, W, N, Cv, Ti		Decomposition of surface and buried residue	[28]
Ecosys	Minute, hour	P, AT, Ir, WS, RH	Lay, W, Cl, CEC, PS, OM, pH, N, BD, PG, PS	Rot, Ti, Fert, Man, Res, Irr, AtN	T, W, N, O, Cl, Cv		C, BioC, N, W, ST, pH, Ph, EC, gas, ExCat	[29]
EPIC	Day	P, AT	Lay, Imp, W, Cl, OM, pH, C, BD, Wi	Rot, Ti, Fert, Man, Res, Irr, AtN	T, W, N, pH, Cl, Ce, Cv		C, BioC, N, W, ST	[30]
FERT	Day	P, AT, WS	Des, Lay, W, Cl, OM, pH, C, N, BD, W, Ph, K, Nup, Y, PS	Rot, Ti, Fert, Man, Res, Irr	T, W, N, pH, Cv		C, N, Ph, K	[31]
ForClim-D	Year	P, AT	W, AG	None	T, W		C	[32]
GENDEC	Day, month	ST, W	W, InertC, LQ	Can be used—not essential	T, W, N		C, BioC, N, gas, LQ	[33]
HPM/EFM	Day	P, AT, Ir, WS	W, Cl, PS	Rot, Fert, Irr, AtN	T, W, N		C, BioC, N, W, gas	[34]
ICBM	Day, year	Combination of weather & climate	Many desirable: none essential	C inputs to soil	T, W, Cl		C	[35]
KLIMAT-SOIL-YIELD	Day, year	P, AT, ST, Ir, EvG, EvS, VPD, SH	Des, Lay, Imp, W, Cl, PS, OM, pH, C, N	Fert, Man, Res, Irr	T, W, N, Cl		C, BioC, N, W, ST	[36]
CNSP pasture model	Day	P, AT, Ir	Lay, Imp, W, Cl, CEC, OM, pH, C, N, PS, AS	Fert	T, W, N, pH		C, N, W, ST	[37]

(Continued)

Table 1 Overview of SOM models represented within GCTE-SOMNET in January 2001. (*Continued*)

Model	Timestep	Meteorology	Soil and plant	Inputs			References
				Management	Factors affecting decay rate constants	Soil outputs	
Humus balance	Year	Climate based on P and AT	Des, Lay, PS, OM, pH, C, N	Rot, Fert, Man	N, H, Cl, Cv	C, N	[38]
MOTOR	User specified	P, AT, EvG	Des, OM	Rot, Ti, Fert, Man	T, W, N, Cl, Ti	C, BioC, 13C, 14C, gas	[39]
NAM SOM	Year	P, AT	Des, PS, OM, Ero	Man, Res	T, W, Cl, Cv	C, BioC	[40]
NCSOIL	Day	ST (P, AT)	W, OM, C, N	Fert, Man, Res	T, W, N, pH, Cl, Ti	C, BioC, 14C, N, 15N, gas	[41]
NICCE	Hour, day	P, AT, Ir, WS	Imp, OM, C, N, W, TC, PG	Fert, Man, Res, Irr, AtN	T, W, Cl, N	C, BioC, 13C, 14C, N, 15N, W, ST, gas	[42]
O'Brien model	Year	None	Lay, C, 14C	None	None	C, 14C	[43]
O'Leary model	Day	P, AT	Lay, W, Cl, pH, N	Ti, Fert, Res	T, W, N, Cl, Ti	C, BioC, N, W, ST, gas, ResC, ResN	[44]
Q-soil	Year	Optional	C, N	Rot, Fert, Man, Res, AtN	T, W, N	C, BioC, 13C, N	[10]
RothC	Month	P, AT, EvW	Cl, C, InertC (can be estimated)	Man, Res, Irr	T, W, Cl, Cv	C, BioC, gas, 14C	[45]
SOCRATES	Week	P, AT	CEC, Y	Rot, Fert, Res	T, W, N, Cv, Ce	C, BioC, gas	[46]
SOMM	Day	P, ST	OM, N, AshL, NL	Man	T, W, N	C, N, gas	[47]
Sundial	Week	P, AT, EvG	Imp, Cl, W, Y	Rot, Fert, Man, Res, Irr, AtN	T, W, N, Cl	C, BioC, N, 15N, W, gas	[48]
Verberne	Day	P, AT, Ir, WS, EvS	Des, W, Cl, PS, OM, C, N	Man, AtN	T, W, N, Cl	C, BioC, N, W	[49]
VOYONS	Day, week, month	P, ST	Cl, OM, C, N	Fert, Man, Res, Irr, AtN	T, W, Cl	C, BioC, 13C, 14C, N, gas	[50]
Wave	Day	P, AT, Ir, EvG	Lay, OM, C, N, W, PG	Rot, Ti, Fert, Man, Res, Irr, AtN	T, W, N	C, N, W, ST, gas	[51]

Note: P, precipitation; AT, air temperature; ST, soil temperature; Ir, irradiation; EvW, evaporation over water; EvG, evaporation over grass; EvS, evaporation over bare soil; WS, wind speed; RH, relative humidity; VPD, vapor pressure deficit; SH, sun hours; Des, soil description; Lay, soil layers; Imp, depth of impermeable layer; Cl, clay content; OM, organic matter content; N, soil nitrogen content/dynamics; C, soil carbon content/dynamics; InertC, soil inert carbon content; W, soil water characteristics; Wi, wilting point; PD, soil particle size distribution; CEC, cation exchange capacity; Ero, annual erosion losses; BD, soil bulk density; TC, thermal conductivity; PG, plant growth characteristics; PS, plant species composition; AS, animal species present; AG, animal growth characteristics; Y, yield; Nup, plant nitrogen uptake; LQ, litter quality; AshL, ash content of litter; NL, N content of litter; Rot, rotation; Ti, tillage practice; Fert, inorganic fertilizer applications; Man, organic manure applications; Res, residue management; Irr, irrigation; AtN, atmospheric nitrogen inputs; T, temperature; W, water; N, nitrogen; O, oxygen; Cl, clay; Ce, cation exchange capacity; Cv, cover crop; Ti, tillage; F, fauna; BioC, biomass carbon; 13C, 13C dynamics; 14C, 14C dynamics; 15N, 15N dynamics; gas, gaseous losses (e.g., CO₂, N₂O, and N₂); ResC, surface residue carbon; ResN, surface residue nitrogen; Ph, phosphorus dynamics; K, potassium dynamics; EC, electrical conductivity; ExCat, exchangeable cations; NB, N in the soil inputs and outputs section is used to denote all aspects of the N cycle. Further details regarding optimum decay conditions, SOM components, rate constants, methods of pool fitting, and refractory SOM are given elsewhere.

Source: From Molina & Smith,^[6] Falloon & Smith,^[52] and a metadatabase of all models is available at <http://www.iacr.bbsrc.ac.uk/aen/somnet/index.htm>.^[7]

FACTORS AFFECTING SOM TURNOVER IN MODELS

Rate “constants” (k) are constant for a given set of biotic and abiotic conditions. For non-optimum environmental circumstances, the simplest way to modify the maximum value of k is by multiplication by a reduction factor μ —ranging from 0 to 1. Environmental factors considered by SOM models include temperature, water, pH, nitrogen, oxygen, clay content, cation exchange capacity, type of crop/plant cover, and tillage.

Many studies show the effect of temperature on microbially mediated transformations in soil, either expressed as a reduction factor or the Arrhenius equation; but the assumption that SOM decomposition is temperature dependent has been challenged by a study suggesting that old SOM in forest soils does not decompose more rapidly in soils from warmer climates than in soils from colder regions.^[53] Studies, however, suggest that old SOM is not more resistant than younger pools of SOM.^[54,55] Water and oxygen have a major impact on the microbial physiology. While some models simulate O_2 concentrations in soil explicitly,^[56,57] many define the extent of anaerobiosis based on soil pore space filled with water.^[58,59] Soil clay content and total SOM are correlated. Various schemes simulate the effect of clay on rate equations to obtain SOM accumulation. Nitrogen is an essential element for microbial growth that will be maximal when enough N is assimilated to maintain the microbial C:N ratio.^[60] An overview of the 33 models represented in the GCTE-SOMNET,^[7,8] including the factors affecting SOM turnover that are presented in Table 1.

SOM MODEL EVALUATION

There are many reasons for evaluating the performance of a SOM model. Model evaluation shows how well a model can be expected to perform in a given situation, it can help to improve the understanding of the system (especially where the model fails), it can provide confidence in the model's ability to predict changes in SOM or where there are no data, and it can be used to assess the uncertainties associated with the model's predictions. Models can be evaluated at a number of different levels. They can be evaluated at the individual process level, at the level of a subset of processes (e.g., net mineralization), or the model's overall outputs (e.g., changes in total SOM over time) can be tested against measured laboratory and field data. Models can also be evaluated for their applicability in different situations, e.g., for scaling-up simulated net C storage from a site specific to a regional level.^[61] Many examples of different forms of SOM model evaluation are presented elsewhere.^[6]

In the most comprehensive evaluation of SOM models,^[61] nine models were tested against 12 data sets from

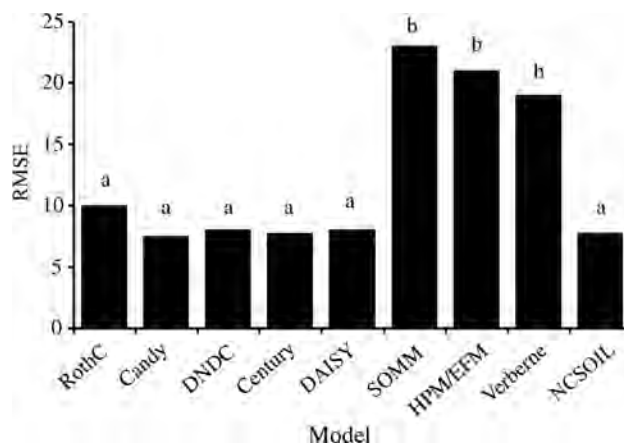


Fig. 1 Overall root mean square error value for nine SOM models when simulating changes in total soil organic carbon in up to 12 data sets from seven long-term experiments. The RMSE values of the models with the same letter (a or b) do not differ significantly (two sample, two-tailed t-test; $p > 0.05$), but the RMSE values of the two groups (a and b) do differ significantly (two sample, two-tailed t-test; $p < 0.05$).

Source: From Smith, Smith, et al.^[62]

seven long-term experiments representing arable rotations, managed and unmanaged grassland, forest plantations, and natural woodland regeneration. The results showed that six models had significantly lower overall errors [root mean square error (RMSE)] than another group of three models (Fig. 1).

The poorer performance of three of the models was related to failures in other parts of the ecosystem models, thus providing erroneous inputs into the SOM module.^[61]

SOM MODEL APPLICATION

SOM models are often used as research tools in that they are hypotheses of the dynamics of C and N in soil and can be used to distinguish between competing hypotheses.^[41] Another increasing application of SOM models is in agronomy; many SOM models are being used to improve agronomic efficiency and environmental quality through incorporation into decision support systems, e.g., SUN-DIAL-FRS,^[48] DSSAT,^[27] and APSIM.^[21]

SOM models are being used, more than ever, to extrapolate our understanding of SOM dynamics both temporally (in to the future) and spatially (to assess C fluxes from whole regions or continents). An early example of a regional scale application was the use of the CENTURY model to predict the effects of alternative management practices and policies in agroecosystems of the Central United States.^[63] Since then, many studies have adopted similar methodologies to assess SOM dynamics at the regional,^[64] national,^[65,66] and global scales.^[67–75] SOM models are increasingly being used by policy makers at the national, regional, or global scales, e.g., in the post-Kyoto

debate on the ability of the terrestrial biosphere to store carbon.^[76] With such an important role in society, it is important that SOM models are transparent, well evaluated, and well documented. There is still a variety of understanding and different hypotheses incorporated in our SOM models. The developments in SOM models will further improve our understanding and allow models to be used truly predictively, without the need for site-specific calibration. These developments will improve estimates of, and reduce, the uncertainty associated with SOM model predictions.

CONCLUSION

SOM models are used for many purposes in soil science, including use as tools to synthesize and explore, explain, and extrapolate experimental data, use for making projections of SOM behavior under environmental conditions, and use for supporting decision making at many levels, by many users. SOM models is one of our most important tools for improving our understanding soil of organic matter dynamics.

REFERENCES

1. Paustian, K. Modelling soil biology and biochemical processes for sustainable agricultural research. In *Soil Biota. Management in Sustainable Farming Systems*; Pankhurst, C.E., Doube, B.M., Gupta, V.V.S.R., Grace, P.R., Eds.; CSIRO Information Services: Melbourne, 1994; 182–193.
2. Jenny, H. *Factors of Soil Formation. A System of Quantitative Pedology*; McGraw-Hill: New York, 1941.
3. Beek, J.; Frissel, M.J. *Simulation of Nitrogen Behaviour in Soils*; Pudoc: Wageningen, 1973.
4. Jenkinson, D.S. Studies on the decomposition of plant material in soil. V. J. Soil Sci. **1977**, *28*, 424–434.
5. McGill, W.B. Review and classification of ten soil organic matter (SOM) models. In *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*; Smith, J.U., Powlson, D.S., Smith, P., Eds.; NATO ASI Vol. I, 38; Springer-Verlag: Berlin, 1996; 111–132.
6. Molina, J.A.E.; Smith, P. Modeling carbon and nitrogen processes in soils. Adv. Agron. **1998**, *62*, 253–298.
7. Global Change and Terrestrial Ecosystems (GCTE). *Soil Organic Matter Network (SOMNET) Database*. Available at <http://www.ufz.de/somnet/>.
8. SOMNET. *Soil Organic Matter Network (SOMNET): 1996 Model and Experimental Metadata*; GCTE Task 3.3.1; Smith, P., Smith, J.U., Powlson, D.S., Eds.; GCTE Report 7; GCTE Focus 3 Office: Wallingford, 1996; 255 pp.
9. Smith, P.; Powlson, D.S.; Smith, J.U.; Glendining, M.J. The GCTE SOMNET: A global network and database of soil organic matter models and long-term datasets. Soil Use Manage. **1996**, *108*, 57.
10. Bosatta, E.; Ågren, G.I. Theoretical analyses of the interactions between inorganic nitrogen and soil organic matter. Eur. J. Soil Sci. **1995**, *46*, 109–114.
11. Bosatta, E.; Ågren, G.I. Theoretical analysis of microbial biomass dynamics in soils. Soil Biol. Biochem. **1994**, *26*, 143–148.
12. Smith, P.; Andrén, O.; Brussaard, L.; Dangerfield, M.; Ekschmitt, K.; Lavelle, P.; Tate, K. Soil biota and global change at the ecosystem level: Describing soil biota in mathematical models. Glob. Change Biol. **1998**, *4*, 773–784.
13. Hunt, H.W.; Coleman, D.C.; Cole, C.V.; Ingham, R.E.; Elliott, E.T.; Woods, L.E. Simulation model of a food web with bacteria, amoebae, and nematodes in soil. In *Current Perspectives in Microbial Ecology*; Klug, M.J., Reddy, C.A., Eds.; American Society for Microbiology: Washington, 1984; 346–352.
14. Hunt, H.W.; Coleman, D.C.; Ingham, E.R.; Ingham, R.E.; Elliot, E.T.; Moore, J.C.; Rose, S.L.; Reid, C.P.; Morley, C.R. The detrital food web in a shortgrass prairie. Biol. Fert. Soils **1987**, *3*, 57–68.
15. Hunt, H.W.; Trlica, M.J.; Redente, E.F.; Moore, J.C.; Detling, J.K.; Kittel, T.G.F.; Walter, D.E.; Fowler, M.C.; Klein, D.A.; Elliott, E.T. Simulation model for the effects of climate change on temperate grassland ecosystems. Ecol. Model. **1991**, *53*, 205–246.
16. de Ruiter, P.C.; Van Faassen, H.G. A comparison between an organic matter dynamics model and a food web model simulating nitrogen mineralization in agro-ecosystems. Eur. J. Agron. **1994**, *3*, 347–354.
17. de Ruiter, P.C.; Van Veen, J.A.; Moore, J.C.; Brussaard, L.; Hunt, H.W. Calculation of nitrogen mineralization in soil food webs. Plant Soil **1993**, *157*, 263–273.
18. de Ruiter, P.C.; Neutel, A.-M.; Moore, J.C. Energetics and stability in belowground food webs. In *Food Webs, Integration of Patterns and Dynamic*; Polis, G.A., Winemiller, K.O., Eds.; Chapman & Hall: New York, 1995; 201–210.
19. McGill, W.B.; Hunt, H.W.; Woodmansee, R.G.; Reuss, J.O. PHOENIX, a model of the dynamics of carbon and nitrogen in grassland soil. In *Terrestrial Nitrogen Cycles. Processes, Ecosystem Strategies and Management Impacts*; Clark, F.E., Roswall, T., Eds.; Ecological Bulletins; Swedish Natural Science Research Council: Stockholm, 1981; Vol. 33, 49–115.
20. Rijtema, P.E.; Kroes, J.G. Some results of nitrogen simulations with the model ANIMO. Fert. Res. **1991**, *27*, 189–198.
21. McCown, R.L.; Hammer, G.L.; Hargreaves, J.N.G.; Holzworth, D.P.; Freebairn, D.M. APSIM: A novel software system for model development, model testing and simulation in agricultural systems research. Agr. Syst. **1996**, *50*, 255–271.
22. Franko, U. Modelling approaches of soil organic matter turnover within the CANDY system. In *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*; Powlson, D.S., Smith, P., Smith, J.U., Eds.; NATO ASI I, Vol. 38; Springer-Verlag: Berlin, 1996; 247–254.
23. Parton, W.J.; Stewart, J.W.B.; Cole, C.V. Dynamics of C, N, P, and S in grassland soils: A model. Biogeochemistry **1987**, *5*, 109–131.
24. Lin, C.; Liu, T.S.; Hu, T.L. Assembling a model for organic residue transformation in soils. Proc. Natl. Council (Taiwan) B **1987**, *11*, 175–186.
25. Mueller, T.; Jensen, L.S.; Hansen, S.; Nielsen, N.E. Simulating soil carbon and nitrogen dynamics with the soil-plant-atmosphere system model DAISY. In *Evaluation of*

- Soil Organic Matter Models Using Existing, Long-Term Datasets*; Powlson, D.S., Smith, P., Smith, J.U., Eds.; NATO ASI I, Vol. 38; Springer-Verlag: Berlin, 1996; 275–281.
26. Li, C.; Frolking, S.; Harriss, R. Modeling carbon biogeochemistry in agricultural soils. *Global Biogeochem. Cy.* **1994**, *8*, 237–254.
 27. Hoogenboom, G.; Jones, J.W.; Hunt, L.A.; Thornton, P.K.; Tsuji, G.Y. *An Integrated Decision Support System for Crop Model Applications*, Paper 94-3025, ASAE Meeting, Missouri, June, 1994; 23 pp.
 28. Douglas, C.L., Jr.; Rickman, R.W. Estimating crop residue decomposition from air temperature, initial nitrogen content, and residue placement. *Soil Sci. Soc. Am. J.* **1992**, *56*, 272–278.
 29. Grant, R.F. Dynamics of energy, water, carbon and nitrogen in agricultural ecosystems: Simulation and experimental validation. *Ecol. Model.* **1995**, *81*, 169–181.
 30. Williams, J.R. The erosion-productivity impact calculator (EPIC) model: A case history. *Phil. Trans. Roy. Soc. Lond. B* **1990**, *329*, 421–428.
 31. Kan, N.A.; Kan, E.E. Simulation model of soil fertility. *Physiol. Biochem. Cultivated Plants* **1991**, *23*, 3–16. (in Russian).
 32. Perruchoud, D.O. Modelling the Dynamic of Non-living Organic Carbon in a Changing Climate: A Case Study for Temperate Forests. Ph.D. Thesis, ETH Diss. No. 11900, 1996; 196 pp.
 33. Moorhead, D.L.; Reynolds, J.F. A general model of litter decomposition in the northern Chihuahuan desert. *Ecol. Model.* **1991**, *56*, 197–219.
 34. Thornley, J.H.M.; Verberne, E.L.J. A model of nitrogen flows in grassland. *Plant Cell Environ.* **1989**, *12*, 863–886.
 35. Andrén, O.; Kätterer, T. ICBM—The introductory carbon balance model for exploration of soil carbon balances. *Ecol. Appl.* **1997**, *7*, 1226–1236.
 36. Sirotenko, O.D. The USSR climate–soil–yield simulation system. *Meteorologia i Gidrologia* **1991**, *4*, 67–73. (in Russian).
 37. McCaskill, M.R.; Blair, G.J. A model of S, P and N uptake by a perennial pasture. I. Model construction. *Fert. Res.* **1990**, *22*, 161–172.
 38. Schevtsova, L.K.; Mikhailov, B.G. *Control of Soil Humus Balance Based on Statistical Analysis of Long-Term Field Experiments Database*; VIUA: Moscow, 1992. (in Russian).
 39. Whitmore, A.P.; Klein-Gunnewiek, H.; Crocker, G.J.; Klir, J.; Körschens, M.; Poulton, P.R. Simulating trends in soil organic carbon in long-term experiments using the Verberne/MOTOR model. *Geoderma* **1997**, *81*, 137–151.
 40. Ryzhova, I.M. Analysis of sensitivity of soil-vegetation systems to variations in carbon turnover parameters based on a mathematical model. *Eurasian Soil Sci.* **1993**, *25*, 43–50.
 41. Molina, J.A.E.; Hadas, A.; Clapp, C.E. Computer simulation of nitrogen turnover in soil and priming effect. *Soil Biol. Biochem.* **1990**, *22*, 349–353.
 42. Van Dam, D.; Van Breemen, N. NICCE—A model for cycling of nitrogen and carbon isotopes in coniferous forest ecosystems. *Ecol. Model.* **1995**, *79*, 255–275.
 43. O'Brien, B.J. Soil organic carbon fluxes and turnover rates estimated from radiocarbon measurements. *Soil Biol. Biochem.* **1984**, *16*, 115–120.
 44. O'Leary, G.J. Soil Water and Nitrogen Dynamics of Dryland Wheat in the Victorian Wimmera and Mallee. Ph.D. Thesis, University of Melbourne, 1994; 332 pp.
 45. Coleman, K.; Jenkinson, D.S.; Crocker, G.J.; Grace, P.R.; Klir, J.; Körschens, M.; Poulton, P.R.; Richter, D.D. Simulating trends in soil organic carbon in long-term experiments using RothC-23.6. *Geoderma* **1997**, *81*, 29–44.
 46. Grace, P.R.; Ladd, J.N. *SOCRATES v2.00 User Manual*; Cooperative Research Centre for Soil and Land Management: South Australia, 1995.
 47. Chertov, O.G.; Komarov, A.S. SOMM—A model of soil organic matter and nitrogen dynamics in terrestrial ecosystems. In *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*; Powlson, D.S., Smith, P., Smith, J.U., Eds.; NATO ASI I38; Springer-Verlag: Berlin, 1996; 231–236.
 48. Smith, J.U.; Bradbury, M.J.; Addiscott, T.M. SUNDIAL: simulation of nitrogen dynamics in arable land. A user-friendly, PC-based version of the Rothamsted nitrogen turnover model. *Agron. J.* **1996**, *88*, 38–43.
 49. Verberne, E.L.J.; Hassink, J.; de Willigen, P.; Groot, J.J.R.; van Veen, J.A. Modelling soil organic matter dynamics in different soils. *Neth. J. Agric. Sci.* **1990**, *38*, 221–238.
 50. André, M.; Thiery, J.M.; Courmac, L. ECOSIMP model: Prediction of CO₂ concentration changes and carbon status in closed ecosystems. *Adv. Space Res.* **1992**, *14*, 323–326.
 51. Vanclooster, M.; Viaene, P.; Diels, J.; Feyen, J. A deterministic evaluation analysis applied to an integrated soil-crop model. *Ecol. Model.* **1995**, *81*, 183–195.
 52. Fallon, P.; Smith, P. Modelling refractory organic matter—A review. *Biol. Fert. Soils* **2000**, *30*, 388–398.
 53. Giardina, C.P.; Ryan, M.G. Evidence that decomposition rates of organic carbon in mineral soil do not vary with temperature. *Nature* **2000**, *404*, 858–861.
 54. Fang, C.; Smith, P.; Moncrieff, J.; Smith, J.U. Similar response of labile and resistant soil organic matter pools to changes in temperature. *Nature* **2005**, *433*, 57–59.
 55. Knorr, W.; Prentice, I.C.; House, J.I.; Holland, E.A. Long-term sensitivity of soil carbon turnover to warming. *Nature* **2005**, *433*, 298–301.
 56. Grant, R.F. A technique for estimating denitrification rates at different soil temperatures, water contents and nitrate concentrations. *Soil Sci.* **1991**, *152*, 41–52.
 57. Sierra, J.; Renault, P. Respiratory activity and oxygen distribution in natural aggregates in relation to anaerobiosis. *Soil Sci. Soc. Am. J.* **1996**, *60*, 1428–1438.
 58. Skopp, J.; Jawson, M.D.; Doran, J.W. Steady-state aerobic microbial activity as a function of soil water content. *Soil Sci. Soc. Am. J.* **1990**, *54*, 1619–1625.
 59. Doran, J.W.; Mielke, L.N.; Stamatiadis, S. Microbial activity and N cycling as regulated by soil water-filled pore space. In *Proceedings of the 11th Conference International Soil Tillage Research Organization*, Edinburgh, Scotland, 1988; Vol. 1, 49–54.
 60. Molina, J.A.E.; Clapp, C.E.; Shaffer, M.J.; Chichester, F.W.; Larson, W.E. NCSOIL, a model of nitrogen and carbon transformations in soil: Description, calibration, and behavior. *Soil Sci. Soc. Am. J.* **1983**, *47*, 85–91.
 61. Izaurralde, R.C.; Haugen-Kozyra, K.H.; Jans, D.C.; McGill, W.B.; Grant, R.F.; Hiley, J.C. Soil organic carbon dynamics:

- Measurement, simulation and site to region scale-up. In *Assessment Methods for Soil Carbon. Advances in Soil Science*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2001; 553–575.
62. Smith, P.; Smith, J.U.; Powlson, D.S.; McGill, W.B.; Arah, J.R.M.; Chertov, O.G.; Coleman, K.; Franko, U.; Frolking, S.; Jenkinson, D.S.; Jensen, L.S.; Kelly, R.H.; Klein-Gunnewiek, H.; Komarov, A.; Li, C.; Molina, J.A. E.; Mueller, T.; Parton, W.J.; Thornley, J.H.M.; Whitmore, A.P. A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments. *Geoderma* **1997**, *81*, 153–225.
 63. Donigian, A.S. Jr.; Barnwell, T.O. Jr.; Jackson, R.B. IV; Patwardhan, A.S.; Weinrich, K.B.; Powell, A.L.; Chinnaswamy, R.V.; Cole, C.V. *Assessment of Alternative Management Practices and Policies Affecting Soil Carbon in Agroecosystems of the Central United States*; US EPA Report EPA/600/R-94/067; Environmental Protection Agency: Athens, 1994; 194 pp.
 64. Falloon, P.; Smith, P.; Smith, J.U.; Szabó, J.; Coleman, K.; Marshall, S. Regional estimates of carbon sequestration potential: Linking the Rothamsted carbon model to GIS databases. *Biol. Fert. Soil* **1998**, *27*, 236–241.
 65. Lee, J.J.; Phillips, D.L.; Liu, R. The effect of trends in tillage practices on erosion and carbon content of soils in the US corn belt. *Water Air Soil Poll.* **1993**, *70*, 389–401.
 66. Parshotam, A.; Tate, K.R.; Giltrap, D.J. Potential effects of climate and land-use change on soil carbon and CO₂ emissions from New Zealand's indigenous forests and unimproved grasslands. *Weather Clim.* **1995**, *15*, 3–12.
 67. Post, W.M.; Emanuel, W.R.; Zinke, P.J.; Stangenberger, A.G. Soil carbon pools and world life zones. *Nature* **1982**, *298*, 156–159.
 68. Post, W.M.; Pastor, J.; Zinke, P.J.; Staggenger, A.G. Global patterns of soil nitrogen storage. *Nature* **1985**, *317*, 613–616.
 69. Post, W.M.; King, A.W.; Wullschleger, S.D. Soil organic matter models and global estimates of soil organic carbon. In *Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets*; Powlson, D.S., Smith, P., Smith, J.U., Eds.; NATO ASI I, Vol. 38; Springer-Verlag: Berlin, 1996; 201–222.
 70. Potter, C.S.; Randerson, J.T.; Field, C.B.; Matson, P.A.; Vitousek, P.M.; Mooney, H.A.; Klooster, S.A. Terrestrial ecosystem production: A process model based on satellite and surface data. *Global Biogeochem. Cy.* **1993**, *7*, 811–841.
 71. Schimel, D.S.; Braswell, B.H. Jr.; Holland, E.A.; McKeown, R.; Ojima, D.S.; Painter, T.H.; Parton, W.J.; Townsend, J.R. Climatic, edaphic, and biotic controls over storage and turnover of carbon in soils. *Global Biogeochem. Cycles* **1994**, *8*, 279–293.
 72. Goto, N.; Sakoda, A.; Suzuki, M. Modelling soil carbon dynamics as a part of the carbon cycle in terrestrial ecosystems. *Ecol. Model.* **1993**, *74*, 183–204.
 73. Esser, G. Modelling global terrestrial sources and sinks of CO₂ with special reference to soil organic matter. In *Soils and the Greenhouse Effect*; Bouwman, A.F., Ed.; John Wiley & Sons: New York, 1990; 247–261.
 74. Goldewijk, K.K.; van Minnen, J.G.; Kreileman, G.J.J.; Vloedveld, M.; Leemans, R. Simulating the carbon flux between the terrestrial environment and the atmosphere. *Water Air Soil Poll.* **1994**, *76*, 199–230.
 75. Melillo, J.M.; Kicklighter, D.W.; McGuire, A.D.; Peterjohn, W.T.; Newkirk, K.M. Global change and its effect on soil organic carbon stocks. In *Role of Nonliving Organic Matter in the Earth's Carbon Cycle*; Zepp, R.G., Sonntag, C.H., Eds.; John Wiley & Sons: New York, 1995; 175–189.
 76. IPCC. Additional Human-Induced Activities-Article 3.4. In *Land Use, Land-Use Change, and Forestry. A Special Report of the IPCC*; Watson, R.T., Noble, I.R., Bolin, B., Ravindranath, N.H., Verardo, D.J., Dokken, D.J., Eds.; Cambridge University Press: Cambridge, 2000; 377 pp.

Soil Organic Matter (SOM): Molecular Simulations

Amity Andersen

Pacific Northwest National Laboratory, Richland, Washington, U.S.A.

Abstract

Molecular simulation is a powerful tool used to gain an atomistic, molecular, and nanoscale level understanding of the structure, dynamics, and interactions from adsorption on minerals and assembly in aggregates of soil organic matter (SOM). Given the importance of SOM fate and persistence in soils and the current knowledge gaps, applications of atomistic scale simulations to study the complex compounds in SOM and their interactions in self-assembled aggregates composed of different organic matter compounds and with mineral surfaces of different types common in soils are few and far between. Here, we describe various molecular simulation methods that are currently in use in various areas and applicable to SOM research, followed by a brief survey of specific applications to SOM research and an illustration of our recent efforts in this area. We conclude with an outlook and the challenges for future research in this area.

INTRODUCTION

A number of molecular simulation methods have been developed to give atomistic and molecular-scale insight in several fields of research including chemistry, biology, biotechnology, solid- and liquid-state physics, and materials science. These cross-cutting techniques can also be applied to modeling efforts in soil science.^[1] The application of molecular simulation methods to soil organic matter (SOM) has been limited, but such methods can provide considerable insight into the molecular structure, dynamics, and processes of SOM including interactions with water, mineral surfaces, ionic species, and other soil organics.

MOLECULAR SIMULATION METHODOLOGIES

These methods can be divided into three broad categories: classical, quantum mechanical, and mixed classical/quantum mechanical methods.

Classical Molecular Mechanics Methods

The term “classical” implies the use of classical Newtonian physics to describe the interactions and motion of the atoms when conducting this class of molecular simulations. In these methods, the atomic interactions, Coulombic (electrostatic), van der Waals, bonding, angles, and dihedral and improper torsion angles are effectively accounted for with empirically or quantum-mechanically derived classical force field parameters. Validated force field parameter sets are widely available to simulate biomolecular/organic systems [e.g., assisted model building with energy refinement (AMBER),^[2] Chemistry at HARvard Macromolecular Mechanics,^[3] optimized potentials for

liquid simulations,^[4] GRONingen MACHine for Chemical Simulations (GROMACS),^[5] generalized AMBER force field,^[6] and condensed-phase optimized molecular potentials for atomistic simulation studies^[7]], inorganic mineral systems (e.g., ClayFF force-field^[8]), and hybrid organic–inorganic systems (e.g., INTERFACE^[9]). Since water is typically present, water force field models are also important (e.g., extended simple point charge model,^[10] 3-site transferable intermolecular potential model,^[11] and 4-site transferable intermolecular potential model^[11]). A representative energy function describing the various atomistic interactions, for instance, in the AMBER force field^[2] is shown in Eq. 1:

$$\begin{aligned} V(r) = & \sum_{\text{bonds}} k_b (d - d_0)^2 + \sum_{\text{angles}} k_a (\theta - \theta_0)^2 \\ & + \sum_{\text{torsions}} \sum_n \frac{1}{2} V_n [1 + \cos(n\omega - \gamma)] \\ & + \sum_{j=1}^{N-1} \sum_{i=j+1}^N f_{ij} \left\{ \epsilon_{ij} \left[\left(\frac{r_{0ij}}{r_{ij}} \right)^{12} - 2 \left(\frac{r_{0ij}}{r_{ij}} \right)^6 \right] \right. \\ & \left. + \frac{q_i q_j}{4\pi\epsilon_0 r_{ij}} \right\} \end{aligned} \quad (1)$$

The first term on the right side describes the harmonic (spring-like) motion of the molecular bonds (two-atom interactions). The second term describes the harmonic motion of the bond angles (three-atom interactions). The third term describes the motion of the torsion angles (four-atom interactions). The fourth term describes the non-bonding Lennard–Jones-type potential for the van der Waals interaction of two atoms and the Coulombic (electrostatic) pair-wise interactions, respectively. Other force fields may have additional interaction terms describing the

coupling of various internal variables such as angles and bond lengths. From a point of view of the computational methods described in this entry, classical molecular dynamics methods are the least demanding computationally since the complex interaction between atoms is described with a simple effective potential as described earlier. Simulations with millions of atoms can be performed on state-of-the-art supercomputers using this approach.^[12] Such simulations also include water-solvated DNA and RNA segments and small proteins interacting with clay mineral surfaces.^[13–19] Most force fields are not able to simulate chemical reactions. The ReaxFF force field of Goddard et al.^[20] is a force field that is designed to model chemical reactions. However, the parameterization of the ReaxFF force field to handle possible SOM reactions is unavailable and may take considerable effort, given the complexity of such systems. Course-graining strategies are also available to group atoms into functional groups with effective potentials to further reduce the number of particles (i.e., atoms), thereby reducing the computational cost of the simulation. Approaches to extend the simulation timescales beyond what can be accomplished with typical molecular dynamics simulations are also available.

Software packages available to perform classical molecular dynamics simulations of SOM and its interactions with minerals include large-scale atomic/molecular massively parallel simulator (LAMMPS),^[21] NANoscale Molecular Dynamics,^[22] GROningen MACHine for Chemical Simulations (GROMACS),^[5] Daresbury Laboratory (DL)_POLY,^[23] and many others (e.g., see http://en.wikipedia.org/wiki/List_of_software_for_molecular_mechanics_modeling).

Quantum Mechanical Methods

Quantum mechanical or quantum chemistry methods are more complex and go far beyond the simple force field description of the atomic interactions and motion found in the classical methods. These methods use quantum mechanical or first-principles techniques to describe the behavior of the electrons and nuclei. Chemical bonding, reactions, and charge transfer processes are naturally described with this approach. These methods involve solving the Schrödinger equation given by,

$$\hat{H}\Psi = E\Psi \quad (2)$$

where E is the total energy of the system, Ψ is the wave function, and $\hat{H}$, the Hamiltonian, is expressed as:

$$\hat{H} = \sum_i -\frac{\hbar^2}{2m_e} \nabla^2 \mathbf{r}_i + V_{\text{ext}}(\{\mathbf{R}_I\}) + V_{e-e}(\{\mathbf{r}_i\}) \quad (3)$$

for the many-body system with the first term describing the electronic kinetic energy, the second term describing the electron–nuclei interactions, and the third term describing

the electron–electron interactions. Quantum chemistry methods are further divided into two categories: *ab initio* (or first principles) and semiempirical. The *ab initio* methodologies include, in increasing degree of complexity and computational expense, Hartree–Fock theory, density functional theory (DFT), Møller–Plesset perturbation theory, coupled-cluster theory, and configuration interaction. Semiempirical methodologies, which are less accurate than the *ab initio* methods, include modified intermediate neglect of differential overlap, Austin Model 1, Parameterized Model 3, and density functional tight-binding (DFTB). The above list of *ab initio* and semiempirical methods is not exhaustive but serves as a starting point for further reading.^[24,25] For the size of systems encountered in SOM, DFT is the most practical and most widely used *ab initio* method. On computing resources, DFT simulations with molecular and mineral surface models of several hundred atoms can be performed routinely, depending on a number of factors including the nature of the electronic system and the size and nature of the basis set (e.g., Gaussian functions, plane waves, and numerical atomic orbitals) employed to represent the electronic wave functions. However, the system sizes typically involved in SOM research are far beyond what any quantum mechanical approach can handle. As a result, suitable model systems have to be designed. These models can include small molecules or fragments of large molecules interacting with small surface areas of minerals. Quantum mechanical approaches also permit one to simulate spectroscopic observables like nuclear magnetic resonance (NMR), ultraviolet/visible, and X-ray absorption spectroscopy (XAS). Semiempirical methods, because of the lower numerical complexity compared with fully *ab initio* methods, are capable of handling much larger system sizes, but this crucially hinges on the availability of suitable and accurate parameter sets. Of the semiempirical Hamiltonian methods, the DFTB method^[26] has emerged as a very promising approach especially for composite organic–inorganic systems such as SOM interacting with mineral surfaces. The DFTB method can simulate systems of thousands of atoms, and there is active research to push these methods to even larger systems comparable to the ones that can be handled by classical force field approaches.

The software packages available to perform *ab initio* quantum chemistry simulations include Northwest Chem (NWChem),^[27] Gaussian,^[28] General Atomic and Molecular Electronic Structure System,^[29] CP2K,^[30] Quantum Espresso,^[31] Vienna Ab initio Simulation Package,^[32] and many others (e.g., see http://en.wikipedia.org/wiki/List_of_quantum_chemistry_and_solid-state_physics_software).

Mixed Quantum Mechanics/Molecular Mechanics (QM/MM) Methods

As described earlier, classical molecular mechanics approaches can model large molecular assemblies such as

biopolymers found in SOM. However, the intermolecular interactions are limited to electrostatic and van der Waals interactions (i.e., no chemical reactions or charge transfer). On the other hand, quantum mechanical methods can address charge transfer and chemical reactions including chemisorption of soil organic molecules to mineral surfaces and other complex interactions. However, the system sizes are limited to small molecules, small representative surfaces, and small fragments of biopolymers. Thus, there have been efforts (in areas other than SOM science) to combine elements of both classical molecular mechanics and quantum chemistry to address large-scale molecular assemblies that require electronic detail. This has resulted in hybrid approaches or QM/MM methods, where one defines a relatively small embedded region of atoms to be performed with more accurate quantum chemistry methods and the large region of atoms encompassing the QM region to be performed with less computationally demanding classical mechanics. These approaches are considered to be first steps in multiscale approaches to bridge spatial and temporal scales from the electronic scale to the macroscopic scale. QM/MM has been extensively utilized in biological problems to study catalytically active sites of protein molecules and in chemistry to study the catalytically active sites of zeolite materials. Such QM/MM methods have rarely been used to study SOM. The challenge for soil science is to have an extensive QM/MM infrastructure that can handle the definition of QM and MM regions, including the possibility of having multiple QM regions to describe more than one potential chemically active interaction for a given SOM system.

Most quantum mechanical software packages mentioned earlier support QM/MM approaches. For a complete review, see, e.g., Lin and Truhlar.^[33]

BRIEF SURVEY OF MOLECULAR MODELING APPLICATIONS TO SOM

Each of the theoretical approaches described above has strengths and limitations, which shape the type of molecular models that can be designed and tackled for a given methodology. Classical molecular dynamics can address models with millions of atoms, the most of the methods described above; however, classical molecular dynamics typically cannot deal with chemical reactions and charge transfer, which involve electronic degrees of freedom that quantum chemistry methods can address. At the other end of the spectrum, quantum mechanical methods can address chemical reactions and charge transfer, but these methods can only handle several hundreds of atoms for the *ab initio* methods and thousands of atoms for the semiempirical Hamiltonian methods. Both methods have been used in the literature to address various aspects of SOM research. The exact structure of SOM is a matter of debate, and the molecular simulation research that has been done and is ongoing

can be divided into two modeling approaches according to the model of SOM adopted. The first modeling approach relies on humic acid atomistic models, notably those of Schulten^[34] and similar models for the molecular model of SOM and dissolved organic matter. The models developed by Schulten are flexible macromolecules of covalently bonded organic groups representing an average structure of SOM based on pyrolysis mass spectrometry data and molecular mechanics sampling of conformations along with quantitative–structure–activity relationship analysis (mass, surface area, volume, partial charges, polarizability, refractivity, hydrophobicity, and hydration energy statistical correlation). These models are able to demonstrate general aspects of SOM such as water and cationic bridging and hydrogen bonding between carboxylic/carboxylate functional groups and other polar and ionic functional groups and organization of complex aggregate structures. The SOM models due to the second modeling approach are derived from biomolecules and/or biopolymers. The premise here is that SOM is not structurally dissimilar from the biomolecular input (i.e., plant, microbial, and animal matter at various stages of decomposition). This model has gained support from NMR^[35] and XAS^[36] experiments.

The computational simulations adopting the Schulten humic acid model or similar models to represent SOM include the work of Sutton and Sposito,^[37] Kirkpatrick et al.,^[38] and Aquino et al.^[39] who used the Schulten humic acid molecular model (with modifications made by Sutton et al.^[40]) to study the interaction of SOM with the Ca²⁺-montmorillonite-layered clay mineral. Kirkpatrick and coworkers^[38,41,42] have used classical molecular dynamics to probe the interaction of a soil organic molecular model smaller than the Schulten humic acid model having similar properties with Ca²⁺ cations^[38,41] and Na⁺-hectorite.^[42] Aquino et al.^[39,43] have used both classical and quantum chemistry methods (DFT and DFTB) to model the interaction of carboxylate-bearing alkyl groups with water^[39] and various cations.^[43] A detailed account of these SOM modeling efforts can be found in the review by Schaumann and Thiele-Bruhn.^[44]

Simulation efforts based on the biomolecule/biopolymer approach are relatively sparse and aimed primarily at the indirect prebiotic topic. However, their insights are applicable in the soil realm as their studies involve interactions of biomolecules/biopolymers with clay mineral surfaces and cationic species. Large-scale classical molecular dynamics studies include the series of DNA–MgAl layered double hydroxide (LDH),^[13] RNA–clay (montmorillonite) surface,^[16] RNA–MgAl LDH,^[15] and ribozyme–clay (montmorillonite) surface^[14] simulations performed by Coveney and coworkers to probe how clay and LDH surfaces and intercalated spaces may be involved in the origin of life. In each of these simulations, Coveney and coworkers monitored the conformational changes of these biopolymers with respect to the clay surface to gauge whether mineral surfaces can enhance folding and mineral

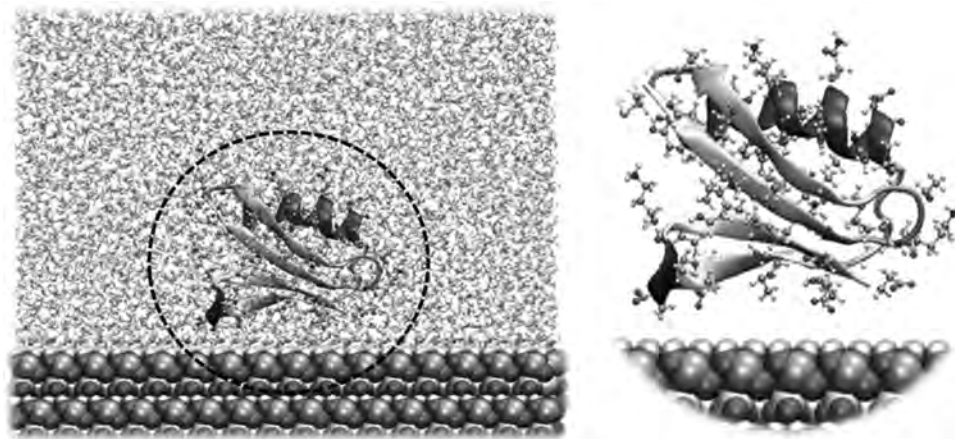


Fig. 1 Rendering of model protein (Gb1 subunit) adsorbed to kaolinite proton-covered (001) basal plane from a classical molecular dynamics simulation.^[17] Left: Simulation snapshot with water molecules shown. Right: Magnified inset showing simulation snapshot with the field of water molecules surrounding the protein and adsorbed to the clay surface removed for clarity. The β -sheet (arrows) and large α -helix (coil) are slightly strained compared to the solution-phase protein with the formation of a mini α -helix at one of the β -sheet turns.

binding outcomes favorable to life. In the intercalated DNA–MgAl LDH case, Coveney and coworkers noted that, at elevated temperatures and pressures (relevant to the origins of life), the intercalation of DNA in the layered mineral enhanced the stability of DNA.^[13]

To address the subject of SOM stability in soil systems, Andersen et al.^[17–19] have embarked on a series of protein–mineral interaction molecular dynamics simulations to explore the potential stabilizing or destabilizing mechanisms of proteinaceous organic matter and the ramifications this may have on soil enzyme activity. A representative picture of one of these large-scale simulations is shown in Fig. 1. Using relatively small biomolecules, hydrophilic glucose and amphiphilic stearic acid, Petridis et al. have demonstrated complex hydrophilic and amphiphilic films on an alumina surface, which suggests the potential for physical protection of certain domains of SOM over mineral surfaces.^[45]

Because of system size limitations mentioned earlier, quantum chemistry application to the study of soil biomolecules and biopolymers has been limited to the interaction of small DNA nucleotides, amino acids, and peptides on small clay cluster or surface models and is used to study chemisorption and chemical reactions of these small molecules on these surfaces.^[46–51]

CHALLENGES AND OUTLOOK

The primary challenge of applying the described methods is the computational limitations on the number of atoms that can be simulated for complex systems such as SOM and the timescales over which the simulations can be performed. As computers become faster and algorithms become more efficient, larger and larger molecular systems will be

feasible for a given method. One burgeoning challenge with the increases in the size of molecular systems that can be simulated is the need for big data solutions to handle massive amounts of simulation data efficiently and for model-building and analysis tools to handle these data. For the simulation methods described here, improvements in performance of algorithms and strategies such as the division of computational work over multiple computer processors, including fast coprocessors such as graphical processing unit and Intel Phi, are in constant research and development.

ACKNOWLEDGMENTS

Dr. Patrick Reardon, Ms. Stephany Chacon, Dr. Nancy Washton, Dr. Niri Govind, Dr. Nikolla Qafoku, Dr. Markus Kleber, and Dr. Nancy Hess are gratefully acknowledged for helpful discussions. This work was funded in part by the “Understanding Molecular-Scale Complexity and Interactions of Soil Organic Matter” Intramural Project at Environmental Molecular Sciences Laboratory (EMSL), a United States Department of Energy (U.S. DOE). Office of Science User Facility, sponsored by the Office of Biological and Environmental Research and located at Pacific Northwest National Laboratory (PNNL). PNNL is operated by Battelle Memorial Institute for the U.S. DOE under DOE contract number DE-AC05-76RL1830. The research also benefited from computational resources provided by EMSL.

REFERENCES

1. Greathouse, J.A.; Johnson, K.L.; Greenwell, H.C. Interaction of natural organic matter with layered minerals:

- Recent developments in computational methods at the nanoscale. *Minerals* **2014**, *4*, 519–540.
2. Case, D.A.; Babin, V.; Berryman, J.T.; Betz, R.M.; Cai, Q.; Cerutti, D.S.; Cheatham, T.E., III; Darden, T.A.; Duke, R.E.; Gohlke, H.; Goetz, A.W.; Gusarov, S.; Homeyer, N.; Janowski, P.; Kaus, J.; Kolossváry, I.; Kovalenko, A.; Lee, T.S.; LeGrand, S.; Luchko, T.; Luo, R.; Madej, B.; Merz, K.M.; Paesani, F.; Roe, D.R.; Roitberg, A.; Sagui, C.; Salomon-Ferrer, R.; Seabra, G.; Simmerling, C.L.; Smith, W.; Swails, J.; Walker, R.C.; Wang, J.; Wolf, R.M.; Wu, X.; Kollman, P.A. *AMBER 14*; University of California: San Francisco, 2014.
 3. Brooks, B.R.; Brooks, C.L.; Mackerell, A.D.; Nilsson, L.; Petrella, R.J.; Roux, B.; Won, Y.; Archontis, G.; Bartels, C.; Boresch, S.; Caflisch, A.; Caves, L.; Cui, Q.; Dinner, A.R.; Feig, M.; Fischer, S.; Gao, J.; Hodoscek, M.; Im, W.; Kuczera, K.; Lazaridis, T.; Ma, J.; Ovchinnikov, V.; Paci, E.; Pastor, R.W.; Post, C.B.; Pu, J.Z.; Schaefer, M.; Tidor, B.; Venable, R.M.; Woodcock, H.L.; Wu, X.; Yang, W.; York, D.M.; Karplus, M. CHARMM: The biomolecular simulation program. *J. Comput. Chem.* **2009**, *30*, 1545–1614.
 4. Jorgensen, W.L.; Tirado-Rives, J. The OPLS [optimized potentials for liquid simulations] potential functions for proteins, energy minimizations for crystals of cyclic peptides and crambin. *J. Am. Chem. Soc.* **1988**, *110*, 1657–1666.
 5. Pronk, S.; Páll, S.; Schulz, R.; Larsson, P.; Bjelkmar, P.; Apostolov, R.; Shirts, M.R.; Smith, J.C.; Kasson, P.M.; van der Spoel, D.; Hess, B.; Lindahl, E. GROMACS 4.5: A high-throughput and highly parallel open source molecular simulation toolkit. *Bioinformatics* **2013**, *29* (7), 845–854.
 6. Wang, J.; Wolf, R.M.; Caldwell, J.W.; Kollman, P.A.; Case, D.A. Development and testing of a general amber force field. *J. Comput. Chem.* **2004**, *25*, 1157–1174.
 7. Sun, H.; Ren, P.; Fried, J.R. The COMPASS force field: Parameterization and validation for phosphazenes. *Comput. Theor. Polym. S.* **1998**, *8*, 229–246.
 8. Cygan, R.T.; Liang, J.-J.; Kalinichev, A.G. Molecular models of hydroxide, oxyhydroxide, and clay phases and the development of a general force field. *J. Phys. Chem. B.* **2004**, *108*, 1255–1266.
 9. Heinz, H.; Lin, T.-J.; Kishore Mishra, R.; Emami, F.S. Thermodynamically consistent force fields for the assembly of inorganic, organic, and biological nanostructures: The INTERFACE force field. *Langmuir* **2013**, *29*, 1754–1765.
 10. Berendsen, H.J.C.; Grigera, J.R.; Straatsma, T.P. The missing term in effective pair potentials. *J. Phys. Chem.* **1987**, *91*, 6269–6271.
 11. Jorgensen, W.L.; Chandrasekhar, J.; Madura, J.D.; Impey, R.W.; Klein, M.L. Comparison of simple potential functions for simulating liquid water. *J. Chem. Phys.* **1983**, *79*, 926–935.
 12. Kunaseth, M.; Kalia, R.K.; Nakano, A.; Nomura, K.-I.; Vashishta, P. A scalable parallel algorithm for dynamic range-limited n-tuple computation in many-body molecular dynamics simulation. In *SC '13 Proceedings of the International Conference on High Performance Computing, Networking, Storage and Analysis, Article No. 71*, Denver, CO, Nov. 17–22, 2013, ACM: New York, 2013; 1–12.
 13. Thyveetil, M.-A.; Coveney, P.V.; Greenwell, H.C.; Suter, J.L. Computer simulation study of the structural stability and materials properties of DNA-intercalated layered double hydroxides. *J. Am. Chem. Soc.* **2008**, *130*, 4742–4756.
 14. Swadling, J.B.; Wright, D.W.; Suter, J.L.; Coveney, P.V. Structure, dynamics, and function of the hammerhead ribozyme in bulk water and at a clay mineral surface from replica exchange molecular dynamics. *Langmuir* **2015**, *31*, 2493–2501.
 15. Swadling, J.B.; Suter, J.L.; Greenwell, H.C.; Coveney, P.V. Influence of surface chemistry and charge on mineral-RNA interactions. *Langmuir* **2013**, *29*, 1573–1583.
 16. Swadling, J.B.; Coveney, P.V.; Greenwell, H.C. Clay minerals mediate folding and regioselective interactions of RNA: A large-scale atomistic simulation study. *J. Am. Chem. Soc.* **2010**, *132*, 13750–13764.
 17. Andersen, A.; Reardon, P.N.; Chacon, S.S.; Washton, N.M.; Qafoku, N.; Kleber, M. Probing the interactions of protein GB1 with mineral surfaces: A theoretical investigation into mechanisms of proteinaceous soil organic matter stabilization/destabilization using molecular dynamics simulations. submitted.
 18. Andersen, A.; Govind, N.; Washton, N.M.; Reardon, P.N.; Chacon, S.S.; Burton, S.D.; Lipton, A.S.; Kleber, M.; Qafoku, N. Elucidating the physical and chemical structural changes of proteins on mineral surfaces using large-scale molecular dynamics simulations in tandem with NMR spectroscopy. In *American Geophysical Union Fall Meeting*, San Francisco, CA, Dec 15–19, 2014.
 19. Andersen, A.; Reardon, P.N.; Chacon, S.S.; Washton, N.M.; Qafoku, N.; Kleber, M. Understanding soil organic matter–mineral interactions with large-scale molecular dynamics simulation of biopolymer–mineral interfaces. In *249th American Chemical Society Spring National Meeting*, Denver, CO, Mar 22–26, 2015.
 20. van Duin, A.C.T.; Dasgupta, S.; Lorant, F.; Goddard, W.A., III. ReaxFF: A reactive force field for hydrocarbons. *J. Phys. Chem. A.* **2001**, *105*, 9396–9409.
 21. Plimpton, S. Fast parallel algorithms for short-range molecular dynamics. *J. Comput. Phys.* **1995**, *117*, 1–19. Available at <http://lammps.sandia.gov>.
 22. Phillips, J.C.; Braun, R.; Wang, W.; Gumbart, J.; Tajkhorshid, E.; Villa, E.; Chipot, C.; Skeel, R.D.; Kale, L.; Schulten, K. Scalable molecular dynamics with NAMD. *J. Comput. Chem.* **2005**, *26*, 1781–1802.
 23. Smith, W.; Todorov, I.; Leslie, M. The DL_POLY molecular dynamics package. *Zeitschrift für Kristallographie* **2009**, *220*, 563–566.
 24. Jensen, F. *Introduction to Computational Chemistry*, 2nd Ed.; John Wiley & Sons Ltd.: Chichester, 2007.
 25. Szabo, A.; Ostlund, N.S. *Modern Quantum Chemistry: Introduction to Advanced Electronic Structure Theory*, 1st Ed., Rev. Ed.; McGraw-Hill: New York, 1989.
 26. Aradi, B.; Hourahine, B.; Frauenheim, T. DFTB+, a sparse matrix-based implementation of the DFTB method. *J. Phys. Chem. A.* **2007**, *111*, 5678–5684.
 27. Valiev, M.; Bylaska, E.J.; Govind, N.; Kowalski, K.; Straatsma, T.P.; Van Dam, H.J.J.; Wang, D.; Nieplocha, J.; Apra, E.; Windus, T.L.; de Jong, W.W. NWChem: A comprehensive and scalable open-source solution for large scale molecular simulations. *Comput. Phys. Commun.* **2010**, *181*, 1477.
 28. Frisch, M.J.; Trucks, G.W.; Schlegel, H.B.; Scuseria, G.E.; Robb, M.A.; Cheeseman, J.R.; Scalmani, G.; Barone, V.;

- Mennucci, B.; Petersson, G.A.; Nakatsuji, H.; Caricato, M.; Li, X.; Hratchian, H.P.; Izmaylov, A.F.; Bloino, J.; Zheng, G.; Sonnenberg, J.L.; Hada, M.; Ehara, M.; Toyota, K.; Fukuda, R.; Hasegawa, J.; Ishida, M.; Nakajima, T.; Honda, Y.; Kitao, O.; Nakai, H.; Vreven, T.; Montgomery, J.A., Jr.; Peralta, J.E.; Ogliaro, F.; Bearpark, M.J.; Heyd, J.; Brothers, E.N.; Kudin, K.N.; Staroverov, V.N.; Kobayashi, R.; Normand, J.; Raghavachari, K.; Rendell, A.P.; Burant, J.C.; Iyengar, S.S.; Tomasi, J.; Cossi, M.; Rega, N.; Millam, N.J.; Klene, M.; Knox, J.E.; Cross, J.B.; Bakken, V.; Adamo, C.; Jaramillo, J.; Gomperts, R.; Stratmann, R.E.; Yazyev, O.; Austin, A.J.; Cammi, R.; Pomelli, C.; Ochterski, J.W.; Martin, R.L.; Morokuma, K.; Zakrzewski, V.G.; Voth, G.A.; Salvador, P.; Dannenberg, J.J.; Dapprich, S.; Daniels, A.D.; Farkas, Ö.; Foresman, J.B.; Ortiz, J.V.; Cioslowski, J.; Fox, D.J. *Gaussian 09*; Gaussian, Inc: Wallingford, 2009.
29. Schmidt, M.W.; Baldrige, K.K.; Boatz, J.A.; Elbert, S.T.; Gordon, M.S.; Jensen, J.H.; Koseki, S.; Matsunaga, N.; Nguyen, K.A.; Su, S.; Windus, T.L.; Dupuis, M.; Montgomery, J.A. General atomic and molecular electronic structure system. *J. Comput. Chem.* **1993**, *14*, 1347–1363.
 30. Hutter, J.; Iannuzzi, M.; Schiffmann, F.; VandeVondele, J. CP2K: Atomistic simulations of condensed matter systems. *Wiley Interd. Rev. Comput. Mol. Sci.* **2014**, *4*, 15–25.
 31. Giannozzi, P.; Baroni, S.; Bonini, N.; Calandra, M.; Car, R.; Cavazzoni, C.; Ceresoli, D.; Chiarotti, G.L.; Cococcioni, M.; Dabo, I.; Dal Corso, A.; de Gironcoli, S.; Fabris, S.; Fratesi, G.; Gebauer, R.; Gerstmann, U.; Gougoussis, C.; Kokalj, A.; Lazzeri, M.; Martin-Samos, L.; Marzari, N.; Mauri, F.; Mazzarello, R.; Paolini, S.; Pasquarello, A.; Paulatto, L.; Sbraccia, C.; Scandolo, S.; Sclauzero, G.; Seitsonen, A.P.; Smogunov, A.; Umari, P.; Wentzcovitch, R.M. Quantum Espresso: A modular and open-source software project for quantum simulations of materials. *J. Phys-Condens. Mat.* **2009**, *21*, 1–19.
 32. Hafner, J. Ab-initio simulations of materials using VASP: Density-functional theory and beyond. *J. Comput. Chem.* **2008**, *29*, 2044–2078.
 33. Lin, H.; Truhlar, D.G. QM/MM: What have we learned, where are we, and where do we go from here? *Theor. Chem. Acc.* **2007**, *117*, 185–199.
 34. Schulten, H.-R. Analytical pyrolysis and computational chemistry of aquatic humic substances and dissolved organic matter. *J. Anal. Appl. Pyrol.* **1999**, *49*, 385–415.
 35. Kelleher, B.P.; Simpson, A.J. Humic substances in soils: Are they really chemically distinct? *Environ. Sci. Technol.* **2006**, *40*, 4605–4611.
 36. Lehmann, J.; Solomon, D.; Kinyangi, J.; Dathe, L.; Wirick, S.; Jacobsen, C. Spatial complexity of soil organic matter forms at nanometer scale. *Nat. Geosci.* **2008**, *1*, 237–242.
 37. Sutton, R.; Sposito, G. Molecular simulation of humic substance–Ca–montmorillonite complexes. *Geochim. Cosmochim. Ac.* **2006**, *70*, 3566–3581.
 38. Kalinichev, A.G.; Iskrenova-Tchoukova, E.; Ahn, W.-Y.; Clark, M.M.; Kirkpatrick, R.J. Effects of Ca²⁺ on supra-molecular aggregation of natural organic matter in aqueous solutions: A comparison of molecular modeling approaches. *Geoderma* **2011**, *169*, 27–32.
 39. Aquino, A.J.A.; Tunega, D.; Pašalić, H.; Schaumann, G.E.; Haberhauer, G.; Gerzabek, M.H.; Lischka, H. Molecular dynamics simulations of water molecule-bridges in polar domains of humic acids. *Environ. Sci. Technol.* **2011**, *45*, 8411–8419.
 40. Sutton, R.; Sposito, G.; Diallo, M.S.; Schulten, H.-R. Molecular simulation of a model of dissolved organic matter. *Environ. Toxicol. Chem.* **2005**, *24*, 1902–1911.
 41. Iskrenova-Tchoukova, E.; Kalinichev, A.G.; Kirkpatrick, R.J. Metal cation complexation with natural organic matter in aqueous solutions: Molecular dynamics simulations and potentials of mean force. *Langmuir* **2010**, *26*, 15909–15919.
 42. Morrow, C.P.; Yazaydin, A.Ö.; Krishnan, M.; Bowers, G.M.; Kalinichev, A.G.; Kirkpatrick, R.J. Structure, energetics, and dynamics of smectite clay interlayer hydration: Molecular dynamics and metadynamics investigation of Na-hectorite. *J. Phys. Chem. C.* **2013**, *117*, 5172–5187.
 43. Aquino, A.J.A.; Tunega, D.; Schaumann, G.E.; Haberhauer, G.; Gerzabek, M.H.; Lischka, H. Proton transfer processes in polar regions of humic substances initiated by aqueous aluminum cation bridges: A computational study. *Geoderma* **2014**, *213*, 115–123.
 44. Schaumann, G.E.; Thiele-Bruhn, S. Molecular modeling of soil organic matter: Squaring the circle? *Geoderma* **2011**, *166*, 1–14.
 45. Petridis, L.; Ambaye, H.; Jagadamma, S.; Kilbey, S.M.; Lokitz, B.S.; Lauter, V.; Mayes, M.A. Spatial arrangement of organic compounds on a model mineral surface: Implications for soil organic matter stabilization. *Environ. Sci. Technol.* **2014**, *48*, 79–84.
 46. Mignon, P.; Sodupe, M. Theoretical study of the adsorption of DNA bases on the acidic external surface of montmorillonite. *Phys. Chem. Chem. Phys.* **2012**, *14*, 945–954.
 47. Mignon, P.; Sodupe, M. Structural behaviors of cytosine into the hydrated interlayer of Na⁺-montmorillonite clay. An ab initio molecular dynamics study. *J. Phys. Chem. C.* **2013**, *117*, 26179–26189.
 48. Mignon, P.; Ugliengo, P.; Sodupe, M. Theoretical study of the adsorption of RNA/DNA bases on the external surfaces of Na⁺-montmorillonite. *J. Phys. Chem. C.* **2009**, *113*, 13741–13749.
 49. Rimola, A.; Sodupe, M.; Ugliengo, P. Aluminosilicate surfaces as promoters for peptide bond formation: An assessment of Bernal's hypothesis by ab initio methods. *J. Am. Chem. Soc.* **2007**, *129*, 8333–8344.
 50. Rimola, A.; Ugliengo, P.; Sodupe, M. Formation versus hydrolysis of the peptide bond from a quantum-mechanical viewpoint: The role of mineral surfaces and implications for the origin of life. *Int. J. Mol. Sci.* **2009**, *10*, 746–760.
 51. Aquino, A.J.A.; Tunega, D.; Gerzabek, M.H.; Lischka, H. Modeling catalytic effects of clay mineral surfaces on peptide bond formation. *J. Phys. Chem. C.* **2004**, *108*, 10120–10130.

Soil Organic Matter (SOM): Nutrient Dynamics

Charles W. Rice

Department of Agronomy, Kansas State University, Manhattan, Kansas, U.S.A.

Abstract

Agriculture in the 1800s and early 1900s relied upon plowing the soil, with low crop yields; crop residues were often removed. This combination of agricultural practices resulted in reducing the replenishment of organic carbon (C) to the soil. Managing agricultural soils for sequestering C will result in additional benefits. Management practices that increase soil C also tend to reduce soil erosion, reduce energy inputs, and improve soil resources.

INTRODUCTION

Soil organic matter (SOM) is a fundamental component of soil and the global carbon (C) cycle. SOM controls many of the chemical, physical, and biological properties of the soil.^[1] The estimated amount of organic C stored in world soils is about 1100–1600 Pg, more than twice the C in living vegetation (560 Pg) or in the atmosphere (750 Pg).^[2] Hence, even relatively small changes in soil C storage per unit area could have a significant impact on the global C balance. SOM is derived mostly from plant residues. Plants convert carbon dioxide (CO₂) into tissue through photosynthesis. Upon their death, plant tissues decompose, primarily by soil microorganisms, and most of the C in the plant material is eventually released back into the atmosphere as CO₂. Between 10% and 20% of the C in plant residue forms SOM, sometimes referred to as “humus.” Some of this C can persist in soils for hundreds and even thousands of years. Associated with the C in SOM are many essential plant nutrients—primarily nitrogen (N), phosphorus (P), and sulfur (S). Concentrations of SOM range from 0.2% to over 80% in peat soils although the typical range for temperate soils is 0.4–10%.^[3] While it is a minor component of most soils, SOM is essential to the living component of soil. SOM provides the energy source for most soil microorganisms and provides the nutrient for plants and the soil biological community. Thus, knowledge of SOM dynamics is crucial for the understanding of global C cycling and plant production.

The accumulation of SOM is dependent on the quantity and quality of organic residue inputs, largely as plant material, the rates of microbial decomposition, and the capacity of the soil to store organic matter. The quality of the plant residue affects both the extent and rate of decomposition. Labile C compounds such as simple sugars degrade relatively rapidly and more completely to CO₂. On the other

end of the spectrum, lignin is more difficult to degrade. Most microorganisms do not have the capacity to completely degrade lignin to CO₂. Thus, many of the partial degradation products form the precursors to SOM. Generally, the C/N ratio is a guide of decomposability. A ratio > 30 slows decomposition and immobilizes N; a ratio < 20 releases N and allows microbial decomposition to proceed. Lignin content or lignin/N ratios are also used to assess organic matter degradability.

The rate of microbial decomposition of plant material and SOM is also a function of climate. As temperature increases, microbial activity increases; generally for every 10°C increase, microbial activity doubles ($Q_{10} = 2$). Soil water content is also important, optimum microbial activity occurs at near “field capacity” which is equivalent to 60% water-filled pore space.^[4] As soil becomes waterlogged, decomposition slows and becomes less complete. Peat soils are often a result of these waterlogged conditions. As soils dry, decomposition is also slowed. The third factor affecting the amount of SOM is soil texture—primarily clay content and mineralogy. Clay content stabilizes SOM by two mechanisms. First, organic molecules are chemically bound to clay surfaces, which retards degradation. Clays with high adsorption capacities, such as montmorillonite clays, are effective in retention of organic molecules. Second, soils with greater clay content increase the potential for aggregate formation. Macroaggregates physically protect organic matter trapped inside the aggregates from microbial degradation. Generally, aggregates > 250 µm provide the greatest protection.^[5,6]

COMPOSITION

SOM is not one definable entity. Since SOM is formed from plant material and microbial decomposition products, it is a myriad of organic compounds. There have been

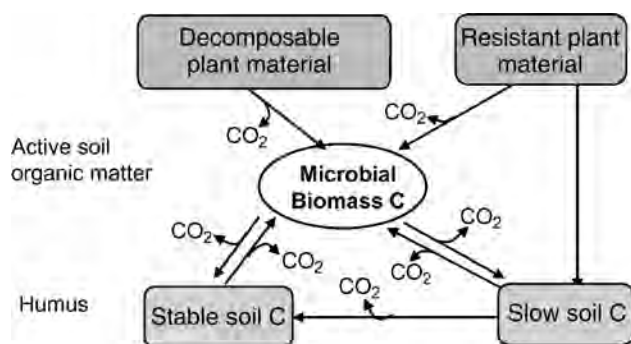


Fig. 1 Schematic of plant decomposition through microbial biomass in the formation of SOM.

Source: Adapted from Paul & Clark.^[7]

several theories on the formation of soil humus, but the most widely accepted theory is that organic residues undergo decomposition by microorganisms.^[7] The altered compounds and new compounds synthesized by soil microbes polymerize through chemical or enzymatic reactions. Thus, SOM is undergoing constant transformation. Typically, most soil organic models define three pools of SOM (Fig. 1). The active pool, which is comprised of microbial biomass and labile organic compounds, makes up less than 5% of the soil organic C. The slow pool usually makes up 20–40% of the total organic C, and the recalcitrant pool makes up 60–70% of the soil C. These fractions are often defined kinetically based on laboratory mineralization.^[5,8] Microbial biomass is the processor, and the slow pool is the one in which much of the plant-associated nutrients reside for mineralization. The recalcitrant pool is material that is difficult to degrade and contains what in the older literature was known as humic and fulvic acids—fractions obtained by chemical fractionation procedures. The active pool has turnover times on the order of months to years, the slow pool takes decades to turn over, while C in the recalcitrant pool takes from hundreds to thousands of years to turn over completely. However, 2–5% of the recalcitrant pool is degraded annually. Since the recalcitrant pool is generally in equilibrium in natural systems, then the rate for formation equals the rate of degradation.

NUTRIENTS

Along with C, SOM contains important plant nutrients. SOM can be a source or sink of plant nutrients. Plant productivity is directly associated with SOM content and turnover. Approximately 90–95% of the soil N, 40% of the soil P, and 90% of the soil S are associated with SOM.^[3] Generally, the C/N/P/S ratio is 100:10:1:1. In agricultural soils, approximately 2–4% of the organic matter is rendered available for plant uptake on an annual basis. As discussed earlier, the more active pools of SOM are likely to be the major source of plant nutrients.

Nitrogen

The plant nutrient that is needed from soil in the greatest quantities is N. Approximately 90–95% of the soil N is in the organic form. The net release of organic N as inorganic N from the soil can be a significant source of N to satisfy plant needs. This process is called net N mineralization, which is the sum of two simultaneous processes, namely N mineralization and N immobilization. N mineralization, or more correctly called “ammonification,” is the conversion of organic N to ammonium (NH_4^+). N immobilization is the conversion of NH_4^+ to organic N. Microorganisms control both these processes, thus, factors that regulate microbial activity, as described earlier, will impact N availability.^[9] In addition to environmental factors, the quality of the organic substrate for microorganisms is important. Low-quality substrate, i.e., high C/N ratio, microorganisms degrading the residue require additional nutrients, primarily N. As a result, soil microorganisms assimilate or immobilize inorganic N. Plants may become deficient in N as the microbes fulfill their N need during decomposition of high C/N residue. Later as the organic material is processed, the N previously assimilated by microorganisms is re-released as inorganic N and can become available to plants. Often the release of organic N to the inorganic forms is in synchrony with plant uptake since favorable temperatures and water availability that promote microbial activity also promote plant growth.^[9,10] In the most native ecosystems and organic agriculture, N mineralization is the major source of plant N needs. In cropland, as much as 11–300 kg N ha⁻¹ can be supplied from organic matter.^[11] Further biological transformations of N occur in the soil, including nitrification (the oxidation of NH_4^+ to NO_3^-) and denitrification (conversion of NO_3^- to N_2); however, these are not directly linked to SOM and will not be discussed in this section. Please refer to Sylvia et al.^[12] for further discussion.

Phosphorus and Sulfur

S follows similar transformations as N. P is more controlled by soil chemistry and chemical transformations. Thus, SOM is not usually a major source of P by plants, except in very high organic soils. However, organic P can represent 80% of the total P in some soils. Significant amounts of P and S are contained within the microbial biomass. As a rule, a C/S or C/P ratio greater than 60 promotes the immobilization of S and P into the microorganisms.

Because of the importance of SOM to the quality of the soil and plant productivity, an understanding of SOM dynamics is critical to preserve natural ecosystems and ensure the long-term productivity of managed ecosystems. Gains and losses of SOM have added significance because it is a reservoir for global C and the associated interaction with climate change.

Table 1 Land use for C sequestration.

Management strategies		
Land use	Soil management	Crop management
Cultivation	Tillage	Varieties
Rangeland	Residue management	Crop rotations
Forestry	Fertility	Cover crops
	Water management	
	Erosion control	

Source: Adapted from Lal, Kimble, et al.^[13]

MANAGEMENT OF SOM TO ENHANCE SOIL QUALITY

Agriculture in the 1800s and early 1900s relied upon the plowing the soil with low crop yields, and crop residues were often removed. This combination of agricultural practices resulted in reducing the replenishment of organic C to the soil. Approximately, 50% of the SOM has been lost from soil over a period of 50–100 years of cultivation. Higher yields, return of crop residues, and development of conservation tillage practices have increased SOM. Table 1 lists several practices affecting the soil’s ability to sequester C.^[13] Examples of rates of soil C increases are summarized in Table 2.^[14] N management that increases crop productivity results in an increase in SOM.^[15] N fertilizer applied at recommended rates for 10 years increase soil C approximately 2 MTC ha⁻¹. Grassland systems can also contribute to C sequestration when properly managed. Under elevated atmospheric CO₂, the soil contained 6% more C to a depth of 15 cm compared with ambient conditions.^[16] The increase in soil C was due to increased plant production followed by incorporation into the soil. The amount of C sequestered over the 8-year experimental period was equivalent to 4 Mg ha⁻¹. Proper fire management may also increase soil C.^[17]

Managing agricultural soils for sequestering C will result in additional benefits. Increasing soil organic C include increased crop productivity and enhanced soil, water, and air quality (Fig. 2). In addition, management

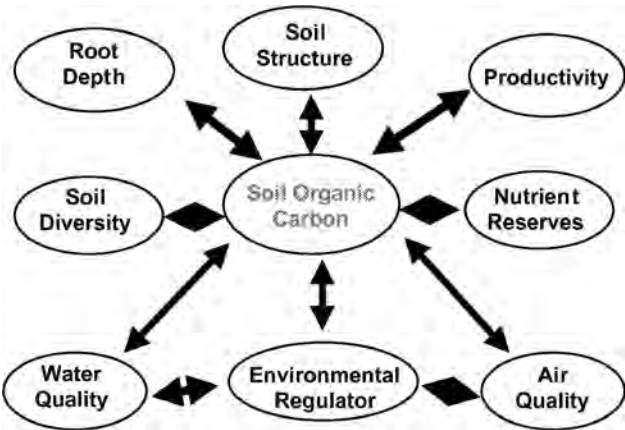


Fig. 2 Relationship between SOM and soil, water, and air quality.

Source: Adapted from Lal, Follett, et al.^[14]

practices that increase soil C also tend to reduce soil erosion, reduce energy inputs, and improve soil resources.

REFERENCES

1. Doran, J.W.; Parkin, T.B. Defining and assessing soil quality. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., Eds.; [Spec. Publ 35]; SSSA: Madison, 1994; 3–21.
2. Sundquist, E.T. The global carbon dioxide budget. *Science* **1993**, *259*, 934–941.
3. Smith, J.L.; Lynch, J.M.; Bezdicek, D.F.; Papendick, R.I. Soil organic matter dynamics and crop residue management. In *Soil Microbial Ecology*; Metting, B., Ed.; Marcel Dekker: New York, 1992; 65–94.
4. Linn, D.M.; Doran, J.W. Effect of water-filled pore space carbon dioxide and nitrous oxide production in tilled and nontilled soils. *Soil Sci. Soc. Am. J.* **1984**, *48*, 1267–1272.
5. van Veen, J.A.; Paul, E.A. Organic C dynamics in grassland soils. 1. Background information and computer simulations. *Can. J. Soil Sci.* **1981**, *61*, 185–201.
6. Jastrow, J.D.; Miller, R.M. Soil aggregate stabilization and carbon sequestration: feed backs through organo-mineral associations. In *Soil Process and the Carbon Cycle*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; CRC Press: New York, 1996; 207–223.
7. Paul, E.A.; Clark, F.E. *Soil Biology and Biochemistry*; Academic Press: San Diego, 1996.
8. Rice, C.W.; Garcia, F.O. Biologically active pools of soil C and N in tallgrass prairie. In *Defining Soil Quality for a Sustainable Environment*; Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., Eds.; [Spec. Publ 35]; SSSA: Madison, 1994; 201–208.
9. Rice, C.W.; Havlin, J.L. Integrating mineralizable N indices into fertilizer N recommendations. In *Soil Testing: Prospects for Improving Nutrient Recommendations*; Havlin, J.L.,

Table 2 Estimates of C sequestration potential of agricultural practices of U.S. cropland.

Agricultural practice	(MTC/ha/yr)
Conservation Reserve Program	0.3–0.7
Conservation tillage	0.24–0.40
Fertilizer management	0.05–0.15
Rotation with winter cover crops	0.1–0.3
Summer fallow elimination	0.1–0.3

Source: Adapted from Lal, Follett, et al.^[14]

- Jacobsen, J.S., Eds.; [Spec. Pub. No. 40]; SSSA: Madison, 1994; 1–13.
10. McGill, W.B.; Meyers, R.J.K. Controls on dynamics of soil and fertilizer nitrogen. In *Soil Fertility and Organic Matter as Critical Components of Production Systems*; Follett, R.F., Stewart, J.W.B., Cole, C.V., Eds.; Spec. Pub. No. 19; SSSA: Madison, 1987; 73–99.
 11. Smith, J.L.; Paul, E.A. The significance of soil microbial biomass estimations. *Soil Biochem.* **1990**, *6*, 357–396.
 12. Sylvia, D.M.; Fuhrman, J.F.; Hartel, P.G.; Zuberer, D.A. *Principles and Application of Soil Microbiology*; Prentice Hall: Upper Saddle River, 1998.
 13. Lal, R.; Kimble, J.R.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Ann Arbor Press: Chelsea, 1998.
 14. Lal, R.; Follett, R.F.; Kimble, J.; Cole, C.V. Managing U.S. cropland to sequester carbon in soil. *J. Soil Water Conserv.* **1999**, *54*, 374–381.
 15. Espinoza, Y. *Dynamics and Mechanisms of Stabilization of C and N in Soil*. Ph.D. dissertation, Kansas State University: Manhattan.
 16. Williams, M.A.; Rice, C.W.; Owensby, C.E. Carbon and Nitrogen dynamics and microbial activity in tallgrass prairie exposed to elevated CO₂ for 8 years. *Plant Soil* **2000**, *227*, 127–137.
 17. Rice, C.W.; Owensby, C.E. Effects of fire and grazing on soil carbon in rangelands. In *The Potential of U.S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect*; Follett, R., Ed.; Lewis Publishers: Boca Raton, 2000; 323–342.

Soil Organic Matter (SOM): Solvent-Based High Resolution Mass Spectrometry

Malak M. Tfaily

Yufeng Shen

Nikola Tolic

Jennifer Kyle

Rosalie Chu

Thomas Fillmore

Rui Zhao

Kent Bloodsworth

Errol Robinson

Ljiljana Paša-Tolić

Nancy J. Hess

Environmental Molecular Sciences Laboratory, Pacific Northwest National Laboratory, Richland, Washington, U.S.A.

Abstract

A limited understanding of the molecular composition of soil organic matter (SOM) inhibits the ability to routinely decipher chemical processes within soil and accurately predict how terrestrial carbon (C) fluxes will respond to changing climatic conditions and land use. To elucidate the molecular-level structure of SOM, organic matter should be extracted from the soil matrix and analyzed using high resolution mass spectrometry (HR MS). Here, we report the use of advanced solvent-based methods and sub-critical fluid extraction protocols for molecular characterization of SOM by HR MS. Combining liquid solvent extraction with super-critical fluid extraction protocols is recommended for extracting a representative fraction of SOM, especially when analysis coverage, recovery, and sensitivity are required. These approaches provide the chemical and structural detail needed to develop mechanistic descriptions of soil C flow processes.

INTRODUCTION

The focus on ecosystem stress and climate change is currently relevant as researchers and policy makers strive to understand the feedbacks between soil carbon (C) dynamics and climate change. Successful development of soil molecular profiles that link microbiology with organic C will have a great impact on assessments of soil ecosystems and vulnerability in response to climate change. Current biogeochemical models predict widely varying C uptake by the terrestrial ecosystem under future climate conditions due to insufficient understanding and representation of soil C processes and dynamics.^[1] Molecular characterization of soil organic matter (SOM) is therefore a necessary first step to develop a more complete understanding of soil C processes and dynamics, their resulting feedbacks to climate change, and a reduction in the uncertainty of predictions of future climate scenarios modeled at the regional and global scales.^[2]

The ability to address fundamental issues associated with biogeochemical cycling of C is limited because the molecular complexity of SOM compounds has prevented routine identification for wide variety of soil types and ecosystems. Early studies of SOM mostly consisted of bulk analysis (i.e., the total weight of soil fractions or elements).^[3] Structural investigations of macromolecular SOM are based on non-destructive spectroscopic techniques, such as nuclear magnetic resonance (NMR) and Fourier transform infrared (FTIR), which provide semi-quantitative analysis of the presence and identity functional groups and types of C bonds (aromatic C, alkyl C, O-alkyl C, etc.). Identification of the molecular composition of SOM typically involves the fragmentation of SOM molecules into smaller units through chemical or thermolytical degradation, which complicates reconstruction of the intact suite of individual molecules present in soils. While gas chromatography mass spectrometry (MS) provides molecular-level information of SOM compounds, only a limited number of compound classes

(i.e., aliphatic and cyclic lipids, low molecular weight phenols, and carbohydrates) are routinely identified using this approach.^[3–5]

Ultra-high resolution mass spectrometry (UHR MS) by Fourier transform ion cyclotron resonance (FTICR) MS coupled with electrospray ionization (ESI) can provide elemental composition of dissolved organic matter (DOM) molecules with ultrahigh-mass resolution and sub-ppm mass accuracy.^[6–11] ESI is based on analyte availability in solution^[12] and uses electrical energy to assist the transfer of ions from solution into the gaseous phase before they are subjected to mass spectrometric analysis, and, therefore, that is why this technique has predominantly been utilized for DOM.^[8,10] For SOM characterization, organic molecules must first be extracted from soil, preferably with little or no chemical alteration. Traditional extraction methods using strong acids and alkaline solutions have been tested on different soils; however, the resulting extracts did not generate well-defined organic matter fractions and potentially distort the SOM through chemical modification.^[13] Moreover, different organic solvents have also been utilized for extracting explosive residues in soil but not for characterization of organic matter composition.^[14] Collectively, none of these studies provided enough information of the different types of compounds extracted by each solvent, as no HR MS instrument was utilized.

Here, we present the developments of novel liquid solvent and sub- and super-critical fluid extraction (SCFE) protocols for a more representative characterization of C compounds in SOM, thereby providing the chemical and structural details needed to develop mechanistic descriptions of soil C flow processes.

Liquid solvent extraction

Liquid solvent extractions (LSEs) with methanol (MeOH), water (H₂O), acetonitrile (ACN), pyridine, chloroform, and hexane were used to extract SOM from different soil types, and then analyzed by 12 Tesla (T) FTICR MS for characterization of SOM molecular composition.^[2] Our experiments have allowed us to identify thousands of individual compounds in complex soil mixtures with a wide range of C content representing diverse ecosystems within the United States. This UHR technique generated large databases of chemical formulas for individual samples, which could be searched for specific compounds and compound classes or integrated across the entire data set to summarize the molecular characteristics of SOM (e.g., aromaticity, elemental ratios, and degree of unsaturation). The yield of the chemical extraction was dependent on: 1) the type of solvent used and its polarity; 2) sample-to-solvent ratios; and 3) the chemical and physical nature of the samples including their origins.

Extraction of specific organic molecular classes can be established by using solvents that differ in polarity. For example, hexane and chloroform are non-polar solvents that extract non-polar aliphatic compounds, such as lipids and

other compounds with high hydrogen (H) to C ratios, while MeOH, a more polar solvent, extracts compounds with intermediate oxygen (O) to C and H to C ratios. ACN, the most polar among the organic solvents tested, extracts compounds with higher O content (e.g., polyphenolic tannin-like compounds). H₂O is selective for compounds with high O to C ratios including carbohydrates and sugars in addition to the hydrophilic aromatic compounds. In testing SOM, our findings suggest that combining the ESI FTICR MS data from the four solvent (hexane, MeOH, H₂O, and ACN) extracts, each conducted on a separate soil sample, resulted in a richer representation of the diversity of organic compounds present in soil than the use of any solvent singly. As a result, we calculated a composite composition by combining the common compounds that were observed in spectra produced by three of the four solvents (MeOH, H₂O, and ACN) and the compounds that were uniquely extracted by each of the other solvents (Fig. 1). In addition, the composite composition was shown to be successful in capturing differences in composition between different soil types.^[2] These observations suggest that SOM from different ecosystems is compositionally different and that a collection of extractions are better able to capturing such differences, in spite of the variability in the classes of compounds each solvent extracts.

To minimize extraction time and sample volume, we built on the results of single solvent extraction techniques to develop sequential extraction protocols and procedures that differed by the solvent type (and hence solvent polarity) and the order of extraction. Sequential extraction methods take

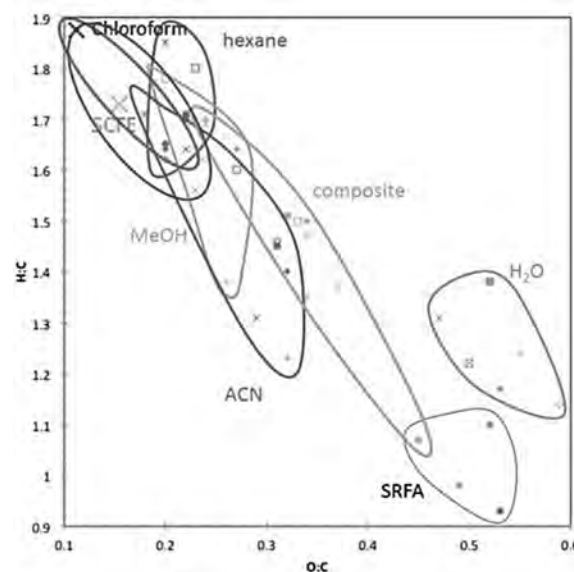


Fig. 1 The CO₂-SCFE and other liquid solvent selectivity for extraction of SOM in specific ranges of O to C and H to C ratios. Composite refers to the sum of the uniquely extracted and common compounds and gives the full representation of compound diversity of organic matter in each soil. SRFA corresponds to Suwannee river fulvic acid standard purchased from International Humic Substances Society.

advantage of the fact that different SOM compounds show dissimilar reactivity toward different solutions. A single soil sample is extracted with a series of extractants, in our case, different polar and non-polar solvents, each chosen to selectively dissolve a single class or group of compounds with similar chemical characteristics. The best coverage, recovery, and selectivity was observed when the sequence was designed so that the most polar compounds are removed first by H₂O, followed by MeOH, and finally, chloroform, a non-polar solvent. The recovery from the single sample sequential extraction protocol utilizing three solvents and 100 mg of soil was comparable to the multiple sample single solvent extraction protocol utilizing six solvents and 600 mg of soil. By utilizing sequential extraction protocol, we decreased the number of solvents used and the time need to analyze the samples by half as well as decreased the amount of starting material required to generate a comparable number of extracted compounds. Both of these extraction techniques expand our depth of knowledge of molecular soil composition leading to a greater understanding to soil ecology and organic C flow.

Sub- and Super-critical Fluid Extraction

Sub- and super-critical fluid (SCFs), such as carbon dioxide (CO₂), is an extraction technique that is complementary to LSE. It has a 10–100-fold faster molecular diffusion rate than liquid solvents without surface tension.^[15] The high diffusion rate of SCFs enables high coverage of SOM by penetrating into and extracting organic molecular species out of porous soil matrices. Additionally, SCF CO₂ is chemically stable and can be removed from the sample simply by relieving the pressure without leaving behind residues in the final extraction products. This extraction technique provides concentrated intact products with high extraction recovery and sensitivity. Altering SCF operation pressure and/or temperature and modifying SCF composition with addition of polar solvents,^[15,16] such as MeOH, can easily modulate the SCFE selectivity, which has a potential to perform sequential SCFE, similar to what was described above for LSE. SCFE was initially widely explored in the 1990s^[15,16] and has recently undergone a revival and is a technique that allows broad characterization of complex organic compounds from solid matrices.

We developed an automated on-line SCFE-micro solid phase extraction (microSPE)—nanoscale liquid chromatography (nanoLC) system coupling to FT ICR MS using a nanoscale ESI interface for SOM analysis. For successful online extraction and characterization of SOM, several challenges must be addressed. The first is the purity of the SCF applied as the extraction solvent. Even high purity SCF, such as SFE-grade CO₂, contains contaminants that can be enriched by online microSPE resulting in high MS background and interfering with accurate MS to assign compounds. A graphitic C LC column can be used to purify SFE-grade CO₂ and the resultant background. Secondly,

microSPE needs to be implemented carefully to provide a sufficient sample capacity without degradation of subsequent LC separation. Implementation of nanoscale LC is favorable for highly sensitive characterization of soils while only consuming of a few milligrams of soil sample.

A tremendous amount of different molecular species (e.g., >24,000) based on their unique masses can be detected by Fourier transform MS even with the use of neat SCF–CO₂ for extraction of SOM from a small size of soil sample (e.g., 8 mg of peat). With use of high mass accuracy (i.e., with ≤1 ppm mass errors) provided by 12T FTICR MS, 7618 molecular formulas can be assigned from a single step, 3 hr, automated experiment. The great power of SCFE for extraction of SOM has been investigated to extract soil residue after been sequentially extracted by H₂O and MeOH. SCFE was able to recover additional compounds not extracted by H₂O and MeOH. The overlap between CO₂–SCFE and H₂O and MeOH extraction is less than 30%, revealing their different selectivity for SOM.

SCFE is a sensitive approach that enables extracting SOM from a small size of a soil sample. We have used a 50 µL sample vessel for SCFE with sample loading capability of 1–35 mg (the soil sample weight loaded depends on the density of soil). For low C content soils, 1 mg sample generates ~800 assigned molecular formulas, whereas 20–35 mg generates ~2600 molecular formulas.

CO₂–SCFE selectivity is similar to hexane and chloroform for extraction of SOM (Fig. 1) and provides rich information for lipids, unsaturated hydrocarbons, and protein compounds. Extraction of SOM cellulose, tannins, and amino sugars by using SCFE requires modifying CO₂ with polar solvents. Modifications that use H₂O and/or MeOH are the optimal, as it extends the SCFE capability to extract more polar molecular species in soil.

CONCLUSION

Coupling the properties of different solvents with FTICR MS enabled a wide range of compound classes to be extracted in complex soil mixtures with a wide range of C content representing diverse ecosystems within the United States. LSE and SCF extraction followed by FTICR MS is a promising tool to understand the dynamics of SOM and provides data for more accurate climate models, and the fate and transportation of C. In-depth characterization of soil biogeochemistry is an essential first step for understanding soil C processes and dynamics, including its possible feedbacks to climate change and impacts on existing biogeochemical models. We described the differences observed in the SOM composition derived from FTICR MS when different organic and aqueous solvents were used for extraction and highlighted a new approach for more representative characterization of the diversity of SOM.

Our exploration for extraction of SOM shows that the SCFE technique is an important member of a family of

extraction methods for characterization of SOM. SCFE can be performed in parallel with conventional extraction methods or used as a complementary tool for recovery of the residues remaining after LSE to enrich content and decrease biases of data sets for SOM study. The results from these experiments show that solvent extraction followed by UHR MS is a promising tool to understand the dynamics of SOM.

ACKNOWLEDGMENTS

The authors acknowledge support for this work by the U.S. Department of Energy (DOE) Office of Science, Office of Biological and Environmental Research (BER). We thank Drs. Jeffrey P. Chanton, James Stegen, and David Kennedy for providing the samples used in this study. We also want to thank Sherry L. Cady for her valuable comments on this entry. This research was performed using EMSL, a DOE Office of Science User Facility sponsored by BER and located at Pacific Northwest National Laboratory (PNNL). PNNL is operated by Battelle for DOE. We thank the TES program for their postdoctoral support for MMT.

REFERENCES

- Lawrence, D.M.; Oleson, K.W.; Flanner, M.G.; Thornton, P.E.; Swenson, S.C.; Lawrence, P.J.; Zeng, X.B.; Yang, Z.L.; Levis, S.; Sakaguchi, K.; Bonan, G.B.; Slater, A.G.J. Parameterization improvements and functional and structural advances in version 4 of the community land model. *Adv. Model. Earth Syst.* **2011**, *3*, M03001.
- Tfaily, M.M.; Chu, R.K.; Tolić, N.; Roscioli, K.M.; Anderton, C.R.; Paša-Tolić, L.; Robinson, E.W.; Hess, N.J. Advanced solvent based methods for molecular characterization of soil organic matter by high-resolution mass spectrometry. *Anal. Chem.* **2015**, *87*, 5206–5215.
- Kogel-Knabner, I. Analytical approaches for characterizing soil organic matter. *Org. Geochem.* **2000**, *31*, 609–625.
- Kögel-Knabner, I. The macromolecular organic composition of plant and microbial residues as inputs to soil organic matter. *Soil Biol. Biochem.* **2002**, *34*, 139–162.
- Derenne, S.; Largeau, C. A review of some important families of refractory macromolecules: Composition, origin and fate in soils and sediments. *Soil Sci.* **2001**, *166*, 833–847.
- Kim, S.; Kramer, R.W.; Hatcher, P.G. Graphical method for analysis of ultrahigh-resolution broadband mass spectra of natural organic matter, the van Krevelen diagram. *Anal. Chem.* **2003**, *75*, 5336–5344.
- Koch, B.P.; Witt, M.R.; Engbrodt, R.; Dittmar, T.; Kattner, G. Molecular formulae of marine and terrigenous dissolved organic matter detected by electrospray ionization Fourier transform ion cyclotron resonance mass spectrometry. *Geochim. Cosmochim. Ac.* **2005**, *69*, 3299–3308.
- Tfaily, M.M.; Hamdan, R.; Corbett, J.E.; Chanton, J.P.; Glaser, P.H.; Cooper, W.T. Investigating dissolved organic matter decomposition in northern peatlands using complementary analytical techniques. *Geochim. Cosmochim. Ac.* **2013**, *112*, 116–129.
- Tfaily, M.M.; Hodgkins, S.; Podgorski, D.C.; Chanton, J.P.; Cooper, W.T. Comparison of dialysis and solid-phase extraction for isolation and concentration of dissolved organic matter prior to Fourier transform ion cyclotron resonance mass spectrometry. *Anal. Bioanal. Chem.* **2012**, *404*, 447–457.
- Tfaily, M.M.; Podgorski, D.C.; Corbett, J.E.; Chanton, J.P.; Cooper, W.T. Influence of acidification on the optical properties and molecular composition of dissolved organic matter. *Anal. Chim. Ac.* **2011**, *706*, 261–267.
- Tremblay, L.B.; Dittmar, T.; Marshall, A.G.; Cooper, W.J.; Cooper, W.T. Molecular characterization of dissolved organic matter in a north Brazilian mangrove porewater and mangrove-fringed estuary by ultrahigh resolution Fourier transform-ion cyclotron resonance mass spectrometry and excitation/emission spectroscopy. *Mar. Chem.* **2007**, *105*, 15–29.
- Fenn, J.B.; Mann, M.; Meng, C.K.; Wong, S.F.; Whitehouse, C.M. Electrospray ionization—principles and practice. *Mass Spectrom. Rev.* **1990**, *9*, 37–70.
- Schmidt, M.W.I.; Torn, M.S.; Abiven, S.; Dittmar, T.; Gugenberger, G.; Janssens, I.A.; Kleber, M.; Kogel-Knabner, I.; Lehmann, J.; Manning, D.A.C.; Nannipieri, P.; Rasse, D.P.; Weiner, S.; Trumbore, S.E. Persistence of soil organic matter as an ecosystem property. *Nature* **2011**, *478*, 49–56.
- Leggett, D.C. Cold Regions Research and Engineering Laboratory (U.S.); U.S. Army Toxic and Hazardous Materials Agency. In *Sorption of Military Explosive Contaminants on Bentonite Drilling Muds*; US Army Corps of Engineers, Cold Regions Research & Engineering Laboratory: Hanover, NH, 1985.
- McHugh, M.A.; Krukonis, V.J. *Supercritical Fluid Extraction*, 2nd Ed.; Butterworth-Heinemann: Stoneham, MA, 1994.
- Fahmy, T.M.; Paulaitis, M.E.; Johnson, D.M.; McNally, M.E.P. Modifier effects in the supercritical fluid extraction of solutes from clay, soil, and plant materials. *Anal. Chem.* **1993**, *65*, 1462–1469.

Soil Organic Matter (SOM): Structure and Characterization

Georg Guggenberger

Institute of Soil Science and Soil Geography, University of Bayreuth, Bayreuth, Germany

Abstract

Soil organic matter (SOM) encompasses all biologically derived organic material found in the soil or on its surface irrespective of: 1) source; 2) whether it is living or dead; or 3) stage of decomposition but excluding the aboveground portion of living plants. This implies large structural heterogeneity and close linkage to SOM functions. Litter composition and its changes during biotic and abiotic transformation are key variables in the processes and the size of carbon (C) sequestration in soil. In the context of the Kyoto Protocol, analysis of SOM structure helps us to understand these processes and to predict changes in the sign and magnitude of terrestrial C fluxes in a changing environment. To reduce SOM heterogeneity, different components of SOM need to be separated into entities that differ in terms of source, composition, and turnover. Evidence suggests that the classical chemical fractionation into fulvic acids, humic acids, and humin, according to solubility characteristics in dilute acid and base, is not useful in this respect. Fractionations that are more promising rely mostly on physical fractionations according to particle size or density and the analysis of dissolved organic matter in the soil solution.

ANALYTICAL METHODS

The analytical approaches can be divided into degradative methods involving chemolysis, thermolysis, or thermochemolysis and non-invasive spectroscopic techniques such as nuclear magnetic resonance (NMR) spectroscopy (Table 1).^[1–6] Degradative methods provide molecular-level information on specific organic compounds accessible to the degradative step, whereas spectroscopic techniques inform on the bulk composition of soil organic matter (SOM). Since all analytical approaches have their drawbacks, a comprehensive picture of the SOM structure can only be obtained using a combination of various methods.^[7]

Degradative Methods

Chemolysis of SOM involves various degradation procedures (hydrolysis, oxidation, and extraction) that are selective in their attack on specific molecular structures (Table 1). The techniques are well suited to follow the often subtle changes in the composition of plant- and microbial-derived biopolymers during microbial transformation processes, in particular when different soil fractions are comparatively investigated.^[9,36] But only a part of SOM present in specific structures—as determined by NMR spectroscopy—can be identified with these techniques, and information on the original macromolecular structure of SOM is limited.^[37]

Analytical pyrolysis uses thermal degradation to cleave bonds in the organic macromolecules and enables a sensitive and rapid characterization of organic constituents.^[23]

Problems may arise due to the complicated pyrolysis behavior of many organic compounds, in particular in the presence of catalytic minerals. Thermal secondary reaction causes considerable modification of the organic compound, which may bias the interpretation of the pyrolysis products with respect to their mother compounds.^[24,25] Some of the problems can be solved by thermochemolysis utilizing tetramethylammonium hydroxide.^[25,38] Hydroxyl and carboxyl groups are converted into their respective methyl ethers and methyl esters to avoid fragmentation of aliphatic and benzenecarboxylic acids.

Spectroscopic Techniques

NMR spectroscopy has a large potential to analyze the different chemical species of ^1H , ^{13}C , ^{15}N , and ^{31}P nuclei in solution and solid (not ^1H) state. With the cross-polarization magic angle spinning pulse sequence, the chemical structure of C and nitrogen (N) can be characterized *in situ* non-destructively, and within a feasible time of analysis.^[28,29] However, even with completion of detailed experiments on rates of signal generation and relaxation for each type of carbon (C),^[30] quantitative analysis of NMR data is not possible in the presence of paramagnetic species. Samples containing charred materials cannot be quantified by the CP technique; instead, the C nuclei must be directly excited using a Bloch decay sequence.^[33] Additional information on the structure of soil organic C can be obtained by specific pulse sequences such as interrupted decoupling (ID), proton spin relaxation editing, and mixing of proton spins (for literature see Baldock and Nelson^[1]). Of the other spectroscopic techniques, infrared spectroscopy provides

Table 1 Important degradative and non-degradative techniques to study SOM structure.

Technique	Components addressed	Applications	Drawbacks	References
Chemolytic techniques	Address defined biomolecules released mostly as their monomer units from the organic matrix by an appropriate chemical treatment	Analysis of defined organic components of plant, microbial and pyrogenic origin often well-suited as biomarkers	Only about 60% (fresh organic residues) to about 20% (transformed, mineral-associated compounds) of SOM is characterized	[7,8]
Acid hydrolysis	Polysaccharides	Distinguishes different types of polysaccharides by sequential extraction Pattern of monosaccharides released gives information on the source (plant vs. microorganisms)	Not all polysaccharides are hydrolyzable Yield differs for different types of with organic matter	[9,10]
	Proteinaceous compounds	Comprises the dominant form of organic nitrogen in soil Assessment of D/L ratios	Only a part of the proteinaceous compounds is hydrolyzable	[11,12]
	Amino sugars	Traces microbial input to SOM Distinguishes between fungal and bacterial residues	Source of some amino sugars is not well defined	[13,14]
Solvent extraction	Extractable lipids	Comprises hydrophobic aliphatic compounds of various origins	Complex and polymerized lipids are only scarcely accessible	[15,16]
Saponification	Cutin and suberin components	Detects aliphatic biopolymers derived from vascular plants	Does not comprise non-ester plant biomacromolecules	[17]
CuO oxidation	Lignin	Traces plant input to SOM Informs about type of plant Informs about lignin decomposition in soil	Only cleaves arylether bondings Yield is not defined and decreases with increasing lignin decomposition	[18,19]
	Cutin and suberin components	Detects aliphatic biopolymers derived from vascular plants	Does not comprise non-ester plant biomacromolecules	[20]
HNO ₃ oxidation	Pyrogenic C	Traces the highly aromatic core of charred organic components	Yield varies with the type of the charred organic components	[21,22]
Analytical pyrolysis	Products of SOM released by thermal degradation	Pattern of pyrolysis products allows detailed information on the chemical composition of the parent molecules Well suited for biomarker analysis	Volatilization of different types of pyrolysis fragments varies Pyrolysis fragments may have different sources Secondary pyrolysis reactions may occur	[23,24]
Py-GC MS	Pyrolysis products that are volatile to be separated by gas chromatography	Separates pyrolysis products that have the same mass/charge (m/z) ratio	Non-volatile pyrolysis products cannot be detected	[25]
Py-FIMS	Pyrolysis products are treated by soft ionization technique	Soft ionization produces predominantly molecular ions of the pyrolysis products	Volatilization decreases with increasing polarity of the fragments	[23]
Thermochemolysis with TMAH	In particular aliphatic and benzenecarboxylic acids as their methyl esters	Pyrolysis/methylation renders polar products volatile for gas chromatographic separation	Reproducibility needs to be improved	[26,27]
NMR spectroscopy	Analysis of all C, H, N, or P species within a solid or soluble sample	Signal intensity relates to the concentration of nuclei creating the signal	Most applications are not quantitative	[28,29,30,31]

(Continued)

Table 1 Important degradative and non-degradative techniques to study SOM structure. (Continued)

Technique	Components addressed	Applications	Drawbacks	References
Solution ^{13}C , ^1H , ^{15}N , and ^{31}P NMR spectroscopy	Analysis of C, H, N, and P species in aqueous samples	Composition of C, H, and N species in the soil solution and in alkaline extracts (humic and fulvic acids)	Insoluble compounds are not detected	[32]
CP MAS ^{13}C , and ^{15}N spectroscopy	Analysis of C and N species in solid samples	Characterization of the chemical structure of organic C and N <i>in situ</i> and non-destructively	Limited quantitative interpretation caused by paramagnetic species, by different C and N relaxation, by spinning side bands, and by the cross-polarization technique	[28,29]
BD ^{13}C NMR spectroscopy	Analysis of C species in solid samples	Direct excitation of ^{13}C spins enables analysis of charred materials	Very time consuming	[33]
IR spectroscopy (FTIR and DRIFT)	Analysis of C species in solid samples	Informs on the type of atoms to which C is bonded and on the nature of the bond	Reveals only a few well-resolved peaks	[34]
			Signals from minerals often dominate	[35,34]

Note: SOM, soil organic matter; Py-GC MS, pyrolysis-gas chromatography mass spectrometry; Py-FIMS, pyrolysis-field ionization mass spectrometry; TMAH, tetramethylammonium hydroxide; NMR, nuclear magnetic resonance; CP MAS NMR, cross-polarization magic angle spinning nuclear magnetic resonance; BD NMR, bloch decay nuclear magnetic resonance; IR, infrared; FTIR, Fourier transform infrared; DRIFT, diffuse reflectance infrared Fourier transform.

information on the type of atoms to which C is bonded and on the nature of the bond,^[35] whereas electron spin resonance spectroscopy gives information about free radicals in SOM.^[39]

CHEMICAL STRUCTURE OF SOM

Individual Plant and Microbial Components

Individual components derived from plants (primary resources) and microorganisms (secondary resources) can be considered as the parent material of SOM. Within plant and microbial tissues, polysaccharides are generally the most important organic components. Analysis of sugar monomers released by hydrolysis or pyrolysis revealed that crystalline and non-crystalline plant polysaccharides are rapidly decomposed and microbial polysaccharides accumulate in soil.^[8,40] Most amino sugars in soil are also of microbial origin, predominantly from fungal chitin and bacterial peptidoglycan.^[13] An important biopolymer in soil is vascular plant-derived lignin with its *p*-hydroxyphenyl, guaiacyl, and syringyl monomeric units. The pattern of lignin monomers can be used to identify the primary resource of SOM; in contrast to angiosperm lignin, gymnosperm lignin does not contain syringyl units. In soil, lignin is decomposed in aerobic environments via side-chain oxidation and ring opening.^[41,42] This results in a shortening of alkyl side chains and an increase in carbonyl and carboxyl groups. Thus, parts of the lignin degradation products are water soluble^[41] and represent, together with

the polysaccharides, the majority of the dissolved organic matter in soil.^[6] The third major component is free and bound lipids of plant and microbial origin, including waxes, organic acids, steroids, glycerides, and phospholipids.^[3,17] Plant cutin and suberin (insoluble polyesters) can be easily decomposed,^[43] while non-saponifiable cutan and suberan (insoluble non-polyesters) appear to be resistant to degradation.^[44] Some aliphatic compounds of microbial origin are also selectively preserved.^[45]

The SOM Continuum

SOM is generally composed of the above-mentioned plant and microbial residues and their transformation products.^[7,46] The latter refers largely to the term “humic substances,” as frequently used in older references.^[47,48] There has been a shift in paradigm away from humic substances. Organic matter degradation is considered to be predominantly a process of attrition, during which relatively resistant biomolecules are selectively concentrated.^[49] In surface soil horizons with large inputs of plant residues, the SOM continuum varying from fresh residues to highly degraded components is dominated by the former material.^[7] In contrast, SOM in subsoil horizons is enriched by highly altered, recalcitrant organic materials that can only be characterized to a minor extent by chemolytic treatments.^[49] Accessibility of these compounds is limited to (chemo)thermolytic and spectroscopic methods.

In aging SOM, Schulten, Leinweber, and Reuter^[50] and Schulten and Leinweber^[23] observed that increasing

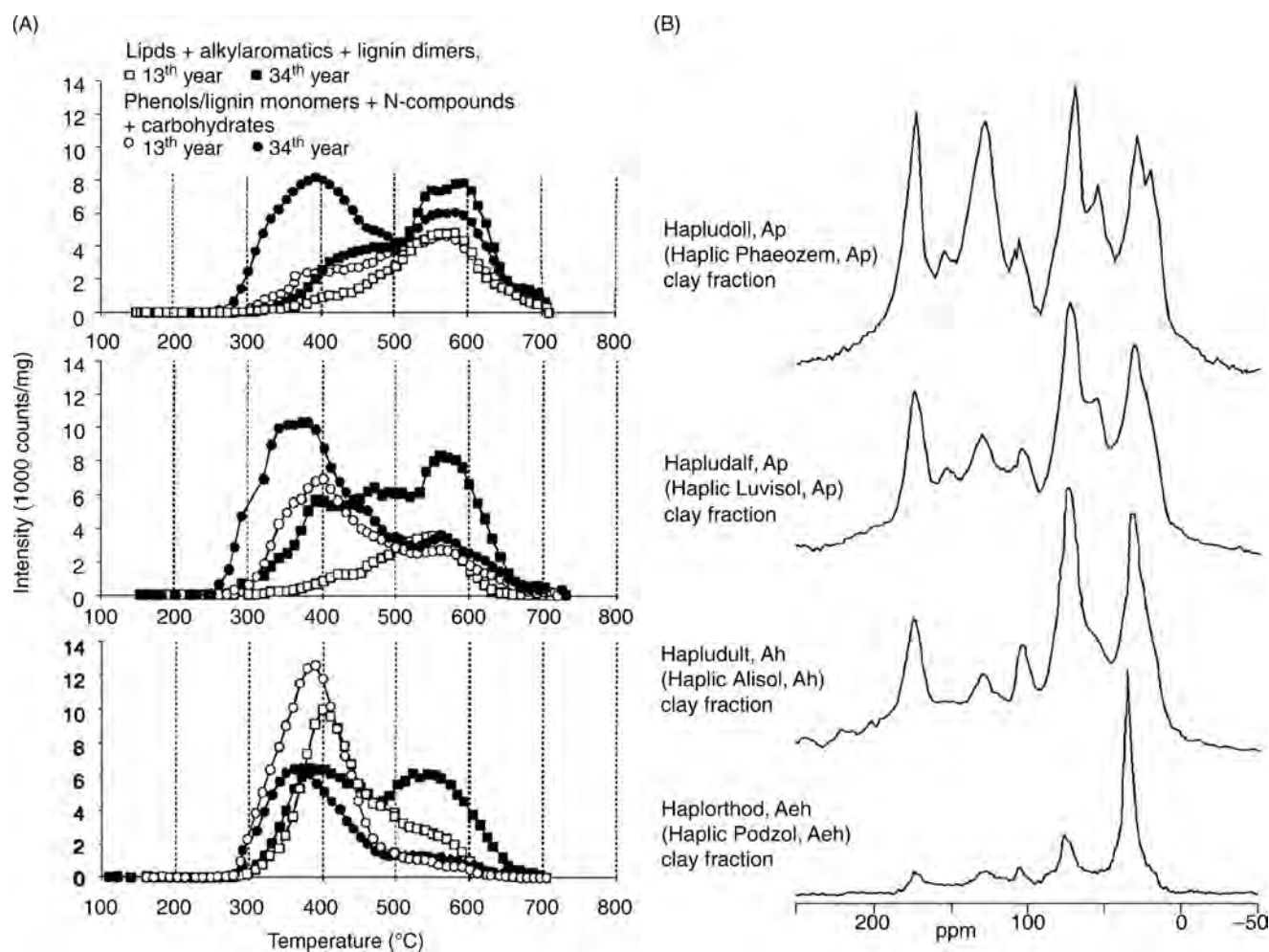


Fig. 1 (A) Thermograms for compound classes evolved by pyrolysis-field ionization from clay (above), fine silt (center), and medium silt (below) from a humus formation experiment after 13 and 34 years of soil development and (B) solid-state ¹³C NMR spectra of the clay fraction isolated from soils of different pedogenesis.

Source: From Kögel-Knabner^[10] and Schulten & Leinweber.^[23]

amounts of thermal energy applied in pyrolysis-field ionization mass spectrometry were required for volatilizing lignin dimers, alkylaromatics, and lipids (Fig. 1A). The authors suggested that the development of stronger chemical bonds was due to either formation of 3-D cross-linking by aryl-alkyl combinations or by formation of strong organic-mineral complexes. Kögel-Knabner^[7] concluded from ID pulse sequences at ¹³C NMR spectroscopy that cross-linked aliphatic compounds accumulate during SOM decomposition.

The pedogenic environment has a large impact on the structure of SOM. This can be particularly confined by solid-state ¹³C NMR spectroscopy of mineral-associated organic matter, while spectra obtained on bulk samples are influenced by plant residues that are rather similar in composition.^[37] The Mollisol in Fig. 1B is characterized by about similar contributions of alkyl (0–50 ppm), O-alkyl (50–110 ppm), aromatic (110–160 ppm), and carboxyl/amide (160–200 ppm) C. In contrast, Alfisols and Ultisols

show a high proportion of alkyl and O-alkyl C, and Spodosols are dominated by alkyl C.

Further work may benefit from the application of microscopic and non-destructive microspectroscopic techniques to investigate SOM within its mineral and microbiological soil environment. For in-depth information on modern methodological approaches and concepts on SOM structure, the excellent reviews of Kögel-Knabner,^[37] Baldock and Nelson,^[1] and Hedges et al.^[49] are recommended.

REFERENCES

1. Baldock, J.A.; Nelson, P.N. Soil organic matter. In *Handbook of Soil Science*; Sumner, M.R., Ed.; CRC Press: Boca Raton, 2000; B-25–B-84.
2. Swift, R.S. Organic matter characterisation. In *Methods of Soil Analysis. Part 3. Chemical Methods*; Sparks, D.L., Ed.; Soil Science Society of America: Madison, 1996; 1011–1069.

3. Stevenson, F.J.; Elliott, E.T. Methodologies for assessing the quantity and quality of soil organic matter. In *Dynamics of Soil Organic Matter in Tropical Ecosystems*; Coleman, D.C., Oades, J.M., Uehara, G., Eds.; University of Hawaii Press: Honolulu, 1989; 173–199.
4. Christensen, B.T. Carbon in primary and secondary organo-mineral complexes. In *Structure and Organic Matter Storage in Agricultural Soils*; Carter, M.R., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1996; 97–165.
5. Golchin, A.; Oades, J.M.; Skjemstad, J.O.; Clarke, P. Soil structure and carbon cycling. *Aust. J. Soil Res.* **1994**, *32*, 1043–1068.
6. Guggenberger, G.; Zech, W.; Schulten, H.-R. Formation and mobilization pathways of dissolved organic carbon: Evidence from chemical structural studies of organic carbon fractions in acid forest floor solutions. *Org. Geochem.* **1994**, *21*, 51–66.
7. Kögel-Knabner, I. Biodegradation and humification processes in forest soils. In *Soil Biochemistry*; Bollag, J.-M., Stotzky, G., Eds.; Marcel Dekker: New York, 1993; Vol. 8, 101–137.
8. Stevenson, F.J. *Humus Chemistry: Genesis, Composition, Reactions*, 2nd Ed.; Wiley-Interscience: New York, 1994.
9. Guggenberger, G.; Christensen, B.T.; Zech, W. Land-use effects on the composition of organic matter in particle-size separates of soils: I. lignin and carbohydrate signature. *Eur. J. Soil Sci.* **1994**, *45*, 449–458.
10. Cheshire, M.V. *Nature and Origin of Carbohydrates*; Academic Press: London, 1979.
11. Chen, C.-N.; Shufeldt, R.C.; Stevenson, F.J. Amino acid analysis of soils and sediments: Extraction and desalting. *Soil Biol. Biochem.* **1975**, *7*, 143.
12. Amelung, W.; Zhang, X. Determination of amino acid enantiomers in soils. *Soil Biol. Biochem.* **2001**, *33*, 553–562.
13. Parsons, J.W. Chemistry and distribution of amino sugars in soils and soil organisms. In *Soil Biochemistry*; Paul, E.A., Ladd, J.N., Eds.; Marcel Dekker: New York, 1981; Vol. 5, 197–227.
14. Amelung, W. Methods using amino sugars as markers for microbial residues in soil. In *Assessment Methods for Soil Carbon*; Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 2001; 233–272.
15. Dinel, H.; Schnitzer, M.; Mehuys, G.R. Soil lipids origin, nature, content, decomposition and effect on soil physical properties. In *Soil Biochemistry*; Bollag, J.-M., Stotzky, G., Eds.; Marcel Dekker: New York, 1990; Vol. 5, 397–429.
16. Capriel, P.; Beck, T.; Borchert, H.; Härter, P. Relationship between soil aliphatic fraction extracted with supercritical hexane, soil microbial biomass, and soil aggregate stability. *Soil Sci. Soc. Am. J.* **1990**, *54*, 415.
17. Kögel-Knabner, I.; Ziegler, F.; Riederer, M.; Zech, W. Distribution and decomposition pattern of cutin and suberin in forest soils. *Z. Pflanzenernähr. Bodenkd.* **1989**, *152*, 409–413.
18. Ertel, J.R.; Hedges, J.I. The lignin component of humic substances: Distribution among soil and sedimentary humic, fulvic, and base-insoluble fractions. *Geochim. Cosmochim. Acta* **1984**, *48*, 2065–2074.
19. Kögel-Knabner, I.; Hatcher, P.G.; Zech, W. Chemical structural studies of forest soil humic acids: Aromatic carbon fraction. *Soil Sci. Soc. Am. J.* **1991**, *55*, 241–247.
20. Goñi, M.A.; Hedges, J.I. Potential applications of cutin-derived CuO reaction products for discriminating vascular plant sources in natural environments. *Geochim. Cosmochim. Acta* **1990**, *54*, 3065–3072.
21. Glaser, B.; Haumaier, L.; Guggenberger, G.; Zech, W. Black carbon in soils: The use of benzenecarboxylic acids as specific markers. *Org. Geochem.* **1998**, *29*, 811–819.
22. Glaser, B.; Haumaier, L.; Guggenberger, G.; Zech, W. The 'Terra Preta' phenomenon: A model for sustainable agriculture in the humid tropics. *Naturwissenschaften* **2001**, *88*, 37–41.
23. Schulten, H.-R.; Leinweber, P. Characterization of humic and soil particles by analytical pyrolysis and computer modeling. *J. Anal. Appl. Pyrol.* **1996**, *38*, 1–53.
24. Saiz-Jimenez, C. Analytical pyrolysis of humic substances: Pitfalls, limitations, and possible solutions. *Environ. Sci. Technol.* **1994**, *28*, 1773–1780.
25. Van Bergen, P.; Flannery, M.B.; Poulton, P.R.; Evershed, R.P. Organic geochemical studies of soils from Rothamsted experimental station: III. Nitrogen-containing organic matter in soil from Geescroft Wilderness. In *Fate of N-Containing Macromolecules in the Biosphere and Geosphere*; Stankiewicz, B.A., van Bergen, P.F., Eds.; Symposium Series 707; American Chemical Society, Oxford University Press: Oxford, 1998; 321–338.
26. del Rio, J.C.; McKinney, D.E.; Knicker, H.; Nanny, M.A.; Minard, R.D.; Hatcher, P.G. Structural characterization of bio- and geo-macromolecules by off-line thermochemolysis with tetramethylammonium hydroxide. *J. Chromatogr.* **1998**, *823*, 433–448.
27. Filley, T.R.; Minard, R.D.; Hatcher, P.G. Tetramethylammonium hydroxide (TMAH) thermochemolysis: Proposed mechanisms based upon the application of ^{13}C -labeled TMAH to a synthetic model lignin dimer. *Org. Geochem.* **1999**, *30*, 607–621.
28. Knicker, H.; Nanny, M.A. Nuclear magnetic resonance spectroscopy. Basic theory and background. In *NMR Spectroscopy in Environmental Science and Technology*; Nanny, M.A., Minear, R.A., Leenheer, J.A., Eds.; Oxford University Press: London, 1997; 3–15.
29. Skjemstad, J.O.; Clarke, P.; Golchin, A.; Oades, J.M. Characterisation of soil organic matter by solid-state ^{13}C nmr spectroscopy. In *Driven by Nature: Plant Litter Quality and Decomposition*; Giller, K.E., Ed.; CAB International: Wallingford, 1997; 253–271.
30. Pfeffer, P.E.; Gerasimowicz, W.V. *Nuclear Magnetic Resonance in Agriculture*; CRC Press: Boca Raton, 1989.
31. Preston, C.M. Applications of NMR to soil organic matter analysis: history and prospects. *Soil Sci.* **1996**, *161*, 144–166.
32. Preston, C.M. Review of solution NMR of humic substances. In *NMR of Humic Substances and Coal*; Wershaw, R.L., Mikita, M.A., Eds.; Lewis Publishers: Chelsea, 1987; 3–32.
33. Skjemstad, J.O.; Clarke, P.; Taylor, J.A.; Oades, J.M.; McClure, S.G. The chemistry and nature of protected carbon in soil. *Aust. J. Soil Res.* **1996**, *34*, 251–271.

34. Parfitt, R.L.; Fraser, A.R.; Farmer, V.C. Adsorption on hydrous oxides III. Fulvic acid and humic acid on goethite, gibbsite and imogolite. *J. Soil Sci.* **1977**, *28*, 289–296.
35. Piccolo, A.; Conte, P. Advances in nuclear magnetic resonance and infrared spectroscopies of soil organic particles. In *Structure and Surface Reactions of Soil Particles*; Huang, P.M., Senesi, N., Buffle, J., Eds.; John Wiley & Sons: Chichester, 1998; 183–250.
36. Hedges, J.I.; Oades, J.M. Comparative organic geochemistry of soils and marine sediments. *Org. Geochem.* **1997**, *27*, 319–361.
37. Kögel-Knabner, I. Analytical approaches for characterizing soil organic matter. *Org. Geochem.* **2000**, *31*, 609–625.
38. Saiz-Jimenez, C. The chemical structure of humic substances: recent advances. In *Humic Substances in Terrestrial Ecosystems*; Piccolo, A., Ed.; Elsevier: Amsterdam, 1996; 1–44.
39. Cheshire, M.V.; Senesi, N. Electron spin resonance spectroscopy of organic and mineral soil particles. In *Structure and Surface Reactions of Soil Particles*; Huang, P.M., Senesi, N., Buffle, J., Eds.; Wiley: Chichester, 1998; 325–376.
40. Huang, Y.; Eglinton, G.; VanderHage, E.R.E.; Boon, J.J.; Bol, R.; Ineson, P. Dissolved organic matter in grass upland soil horizons studied by analytical pyrolysis techniques. *Eur. J. Soil Sci.* **1998**, *49*, 1–15.
41. Haider, K. Problems related to the humification processes in soils of the temperate climates. In *Soil Biochemistry*; Bollag, J.-M., Stotzky, G., Eds.; Marcel Dekker: New York, 1992; Vol. 7, 55–94.
42. Shevchenko, S.M.; Bailey, G.W. Life after death: Lignin–humic relationships reexamined. *Critical Rev. Environ. Sci. Technol.* **1996**, *26*, 95–153.
43. Riederer, M.; Matzke, K.; Ziegler, F.; Kögel-Knabner, I. Inventories and decomposition of the lipid plant biopolymers cutin and suberin in temperate forest soils. *Org. Geochem.* **1993**, *20*, 1063–1076.
44. Tegelaar, E.W.; de Leeuw, J.W.; Saiz-Jimenez, C. Possible origin of aliphatic moieties in humic substances. *Sci. Total Environ.* **1989**, *81/82*, 1–17.
45. Lichtfouse, E.; Chenu, C.; Baudin, F.; Leblond, C.; da Silva, M.; Behar, F.; Derenne, S.; Largeau, C.; Wehrung, P.; Albrecht, P. A novel pathway of soil organic matter formation by selective preservation of resistant straight-chain biopolymers: Chemical and isotopic evidence. *Org. Geochem.* **1998**, *28*, 411–415.
46. Waksman, S.A. *Humus, Origin, Chemical Composition and Importance in Nature*; Ballière, Tindall & Cox: London, 1938.
47. Aiken, G.R.; McKnight, D.M.; Wershaw, R.L.; MacCarthy, P., Eds. *Humic Substances in Soil, Sediment and Water*; Wiley-Interscience: New York, 1985.
48. Hayes, M.H.B.; MacCarthy, P.; Malcolm, R.; Swift, R.S., Eds. *Humic Substances II. In Search of Structure*; Wiley-Interscience: Chichester, 1989.
49. Hedges, J.I.; Eglinton, G.; Hatcher, P.G.; Kirchman, D.L.; Arnosti, C.; Derenne, S.; Evershed, R.P.; Kögel-Knabner, I.; de Leeuw, J.W.; Littke, R.; Michaelis, W.; Rullkötter, J. The molecularly-uncharacterized component of nonliving organic matter in natural environments. *Org. Geochem.* **2000**, *31*, 945–958.
50. Schulten, H.-R.; Leinweber, P.; Reuter, G. Initial formation of soil organic matter from grass residues in a long-term experiment. *Biol. Fert. Soils* **1992**, *14*, 237–245.

Soil Organic Matter (SOM): Turnover

Johan Six

Department of Agronomy and Range Science, University of California—Davis,
Davis, California, U.S.A.

Julie D. Jastrow

Environmental Research Division, Argonne National Laboratory, Argonne, Illinois, U.S.A.

Abstract

Soil organic matter (SOM) is a dynamic entity. The amount (stock) of organic matter in a given soil can increase or decrease depending on numerous factors including climate, vegetation type, nutrient availability, disturbance, land use, and management practices. But even when stocks are at equilibrium, SOM is in a continual state of flux; new inputs cycle—via the process of decomposition—into and through organic matter pools of various qualities and replace materials that are either transferred to other pools or mineralized. An understanding of SOM turnover is crucial for quantifying carbon (C) and nutrient cycles and for determining the quantitative and temporal responses of local, regional, or global C and nutrient budgets to perturbations caused by human activities or climate change.

INTRODUCTION

Soil organic matter (SOM) is a dynamic entity. The amount (stock) of organic matter in a given soil can increase or decrease depending on numerous factors including climate, vegetation type, nutrient availability, disturbance, land use, and management practices. But even when stocks are at equilibrium, SOM is in a continual state of flux; new inputs cycle—via the process of decomposition—into and through organic matter pools of various qualities and replace materials that are either transferred to other pools or mineralized. For the functioning of a soil ecosystem, this “turnover” of SOM is probably more significant than the sizes of SOM stocks.^[1] An understanding of SOM turnover is crucial for quantifying carbon (C) and nutrient cycles and for determining the quantitative and temporal responses of local, regional, or global C and nutrient budgets to perturbations caused by human activities or climate change.^[2]

DEFINITION OF SOM TURNOVER

The turnover of an element [e.g., C, nitrogen, and phosphorus] in a pool is generally determined by the balance between inputs (I) and outputs (O) of the element to and from the pool as shown in Fig. 1. Turnover is most often quantified as the element's mean residence time (MRT) or its half-life ($T_{1/2}$). The MRT of an element in a pool is defined as 1) the average time the element resides in the pool at steady state or 2) the average time required to completely renew the content of the pool at steady state. The term half-life is adopted from radioisotope work, where it is

defined as the time required for half of a population of elements to disintegrate. Thus, the half-life of SOM is the time required for half of the existing stock to decompose.

The most common model used to describe the dynamic behavior or turnover of SOM is the first-order model, which assumes constant zero-order input with constant proportional mass loss per unit time^[3,4]

$$\frac{\partial S}{\partial t} = I - kS \quad (1)$$

where S is the SOM stock, t is the time, k is the decomposition rate, and kS is equivalent to output O. Assuming equilibrium ($I = O$), the MRT can then be calculated as

$$MRT = \frac{1}{k} \quad (2)$$

and MRT and $T_{1/2}$ can be calculated interchangeably with the formula as follows:

$$MRT = T_{1/2} / \ln 2 \quad (3)$$

MEASURING SOM TURNOVER

Most often the turnover of SOM, more specifically the turnover of SOM-C, is estimated by one of four techniques:

1. Simple first-order modeling
2. ^{13}C natural abundance technique
3. ^{14}C dating technique
4. “Bomb” ^{14}C technique

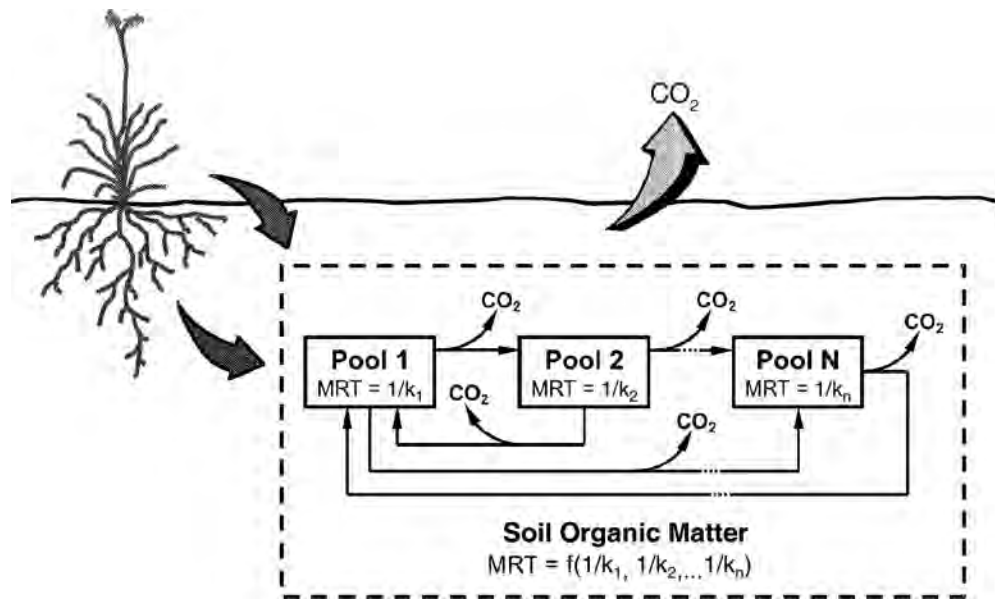


Fig. 1 The turnover of SOM is determined by the balance of inputs and outputs. Total SOM consists of many different pools that are turning over at different rates. The MRT of total SOM is a function of the turnover rates of its constituent pools.

This list does not include tracer studies where a substrate (e.g., plant material) enriched in ^{13}C , ^{14}C , and/or ^{15}N is added to soil, and its fate is followed over time. Most studies of this type (see Schimel,^[5] for a review) use the tracers to quantify the short-term (1–5 years) decomposition rate of freshly added material rather than the long-term turnover of whole-soil C.

Eqs. 1 and 2 form the basis for estimates of SOM turnover derived from first-order modeling; the unknown k is calculated as

$$k = \frac{I}{S}$$

by assuming a steady state

$$\frac{\partial S}{\partial t} = 0.$$

This approach requires estimates of annual C input rates, which can be assumed to be continuous or discrete.^[3] The input can also be written as

$$I = hA$$

where A is the annual addition of C as fresh residue and h (the isohumification coefficient) represents the fraction that, after a rapid initial decomposition of A , remains as the actual annual input to S . An estimate of h is then necessary. A value of 0.3 is commonly used for agricultural crops, but the value can be higher for other materials such as grasses or peat.^[6,7]

Another approach to estimate k by first-order modeling is “chronosequence modeling.”^[8] An increase (or decrease) in C across a chronosequence of change in vegetation, land use, or management practice can be fitted to a first-order model

$$S = S_e \left[1 - \left(\frac{S_e - S_0}{S_e} \right) e^{-kt} \right]$$

which is equivalent to

$$S = S_0 + (S_e - S_0)(1 - e^{-kt}) \quad (4)$$

where t is the time since the change, S_e is the C content at equilibrium, and S_0 is the initial C content before the change ($t=0$). An average value of I can then be calculated

$$I = kS_e$$

but in this case I represents annual inputs of new SOM (hA) rather than inputs of fresh litter or detritus. This approach is also used for chronosequences of primary succession (e.g., on glacial moraines, volcanic deposits, river terraces, and dune systems), in this case $S_0=0$.^[4]

The ^{13}C natural abundance technique relies on 1) the difference in ^{13}C natural abundance between plants with different photosynthetic pathways [Calvin cycle (C_3 plants) vs. Hatch–Slack cycle (C_4 plants)] and 2) the assumption that the ^{13}C natural abundance signature of SOM is identical to the ^{13}C natural abundance signature of the plants from which it is derived.^[9] Thus, where a change in vegetation type has occurred at some known point in time, the rate of loss of the C derived from the original vegetation and the incorporation of C derived from the new vegetation can be inferred from the resulting change in the ^{13}C natural abundance signature of the soil. The turnover of C derived from the original vegetation is then calculated by using the first-order decay model

$$\text{MRT} = \frac{1}{k} = \frac{t}{\ln(S_i/S_0)} \quad (5)$$

where t is the time since conversion, S_t is the C content derived from original vegetation at time t , and S_0 is the C content at $t = 0$. For further details on the technique, see Cerri et al.^[9] and Balesdent and Mariotti.^[10]

The presence of ^{14}C with a half-life of 5570 years in plants and the transformation of this ^{14}C into SOM with little isotopic discrimination allows the SOM to be dated, providing an estimate of the age of the SOM. The ^{14}C dating technique is applicable within a time frame of 200–40,000 years; samples with an age less than 200 years are designated as modern (see Goh,^[11] for further details of the methodology).

Thermonuclear bomb tests in the 1950s and 1960s caused the atmospheric ^{14}C content to increase sharply and then to fall drastically after the tests were halted. This sequence of events created an *in situ* tracer experiment; the incorporation of bomb-produced radiocarbon into SOM after the tests stopped allows estimates of the turnover of SOM. Further details of the technique are described by Trumbore,^[2] Goh,^[12] and Harrison et al.^[13]

RANGE AND VARIATION IN ESTIMATES OF TOTAL SOM TURNOVER

The comparisons of MRT values estimated by the four methods previously described (see Table 1) reveal a wide range of MRTs. Although variations within each method are attributable to differences in vegetation, climate, soil

type, and other factors, the largest variations in observed MRTs are method dependent. For example, MRTs estimated by simple first-order modeling and ^{13}C natural abundance are generally smaller by an order of magnitude than MRTs estimated by radiocarbon dating, because of the different time scales that the two methods measure. The ^{13}C method is generally used in medium-term observations or experiments (5–50 years); hence, this method gives an estimate of turnover dominated by relatively inputs and C pools that cycle within the time frame of the experiment. In contrast, the oldest and most recalcitrant C pools dominate estimates by radiocarbon dating because of the long-term time frame (200–40,000 years) that this method measures.^[11]

FACTORS CONTROLLING SOM TURNOVER

Primary production (specifically, the rate of organic matter transfer belowground) and soil microbial activity (specifically, the rates of SOM transformation and decay) are recognized as the overall biological processes governing inputs and outputs and, hence, SOM turnover. These two processes (and the balance between them) are controlled by complex underlying biotic and abiotic interactions and feedbacks, most of which can be tied in some way to the state factor model of soil formation.^[4] Climate (especially temperature and precipitation) constrains both production and decomposition of SOM. Vegetation type affects

Table 1 Range and average MRTs of total soil organic C in various ecosystem types as estimated by four different methods.

Method and ecosystem	Sites and sources ^a	MRT (years)		
		Low ^b	High ^b	Average ± SE
First-order modeling				
Cultivated systems and recovering grassland or woodland systems	7/7	15 ^[14]	102 ^[15]	67 ± 12
¹³C natural abundance				
Cultivated systems	20/10	18 ^[16]	165 ^[17]	61 ± 9
Pasture systems	12/10	17 ^[18]	102 ^[19]	38 ± 7
Forest systems	2/2	18 ^[20]	25 ^[21]	22 ± 4
Radiocarbon aging^c				
Cultivated systems	21/8 ^d	327 ^[22]	1770 ^[23]	880 ± 105
Grassland systems	4/3 ^e	Modern ^[23]	1040 ^[24]	— ^f
Forest systems	4/3	422 ^[22]	1550 ^[25]	1005 ± 184
“Bomb” ¹⁴C analysis				
Cultivated systems	1/1	1863 ^[13]	1863 ^[13]	1863 ^g
Forest and grassland systems	14/12	36 ^[26]	1542 ^[27]	535 ± 134

Note: SE, standard error.

^aFirst value indicates the number of sites used to calculate average MRT values; second value indicates the number of literature sources surveyed (i.e., some sources provided data for multiple sites).

^bNumber in parentheses indicates reference to literature.

^cValues presented in MRT columns for this technique are radiocarbon ages in years before present.

^dIncludes two sites dating as “modern.”

^eIncludes three sites dating as “modern.”

^fOnly one value available.

production rates and the types and quality of organic inputs (e.g., below- vs. aboveground, amounts of structural tissue, C/N and lignin/N ratios), as well as the rates of water and nutrient uptake—all of which, in turn, influence decomposition rates. The types, populations, and activities of soil biota control decomposition and nutrient cycling/availability and hence influence vegetative productivity. Parent material affects SOM turnover as soil type, mineralogy, texture, and structure influence pH, water and nutrient supply, aeration, and the habitat for soil biota, among other factors. Topography modifies climate, vegetation type, and soil type on the landscape scale and exerts finer-scale effects on temperature, soil moisture, and texture. Lastly, time affects whether inputs and outputs are at equilibrium, and temporal scale influences the relative importance of various state factor effects on production and decomposition.

Disturbance or management practices also exert considerable influence on SOM turnover via direct effects on inputs and outputs and through indirect effects on the factors controlling these fluxes. An example of management effects on MRT is illustrated in Table 2; in most cases, the MRT of whole-soil C is significantly longer under no tillage agriculture than under conventional tillage practices.

TURNOVER OF DIFFERENT SOM POOLS

The previous discussion is focused on the turnover and MRT of whole-soil C; hence, it treats SOM as a single, homogeneous reservoir. But, in fact, SOM is a heterogeneous mixture consisting of plant, animal, and microbial materials in all stages of decay combined with a variety of decomposition products of different ages and levels of complexity. Thus, the turnover of these components varies

continuously, and any estimate of MRT for SOM as a whole merely represents an overall average value (Fig. 1).

Although average MRTs are useful for general comparisons of sites or the effects of different management practices, they can be misleading because soils with similar average MRTs can have very different distributions of organic matter among pools with fast, slow, and intermediate turnover rates.^[2,31] Simulation models that account for variations in turnover rates for different SOM pools are used to generate more realistic descriptions of SOM dynamics. A few models represent decomposition as a continuum, with each input cohort following a pattern of increasing resistance to decay,^[32] but most models are multicompartmental, with several organic matter pools (often 3–5) that are kinetically defined with differing turnover rates. For example, the CENTURY SOM model^[33] divides soil C into active, slow, and passive pools, with MRTs of 1.5, 25, and about 1000 years, respectively, and separates plant inputs into metabolic (readily decomposable; MRT of 0.1–1 years) and structural (difficult to decompose; MRT of 1–5 years) pools as a function of lignin/N ratio. Even though compartmental models are reasonably good at simulating changes in SOM, the compartments are conceptual in nature, and thus it has been difficult to relate them to functionally meaningful pools or experimentally verifiable fractions.^[34,35]

The use of isotopic techniques to analytically determine the MRTs of physically and chemically separated SOM fractions has demonstrated the existence of various turnover rates for different pools. For example, low-density SOM (except for charcoal) invariably turns over faster than high-density, mineral-associated SOM, and hydrolyzable SOM turns over faster than non-hydrolyzable residues.^[36,37] The MRTs of primary organomineral associations generally increase with decreasing particle size, although there are exceptions (particularly among fine

Table 2 Effect of tillage practices on MRT of total soil organic C estimated by the ¹³C natural abundance technique.

Site (References)	Cropping system ^a	Depth (cm)	tb (yr)	MRT (yr)
Sidney, NE ^[28]	Wheat–fallow (NT)	0–20	26	73
	Wheat–fallow (CT)			44
Delhi, Ont. ^[29]	Corn (NT)	0–20	5	26
	Corn (CT)			14
Boigneville, France ^[16]	Corn (NT)	0–30	17	127
	Corn (CT)			55
Rosemount, MN ^[30]	Corn (NT, 200 kg N ha ⁻¹ yr ⁻¹)	0–30	11	118
	Corn (CT; 200 kg N ha ⁻¹ yr ⁻¹)			73
	Corn (NT; 0 kg N ha ⁻¹ yr ⁻¹)			54
	Corn (CT; 0 kg N ha ⁻¹ yr ⁻¹)			72
Average ± SE		NT		80 ± 19
		CT		52 ± 11

Note: SE, standard error.

^aNT, no tillage; CT, conventional (moldboard plow) tillage.

^bTime period of experiment.

Table 3 MRT of macro- and microaggregate-associated C estimated by the ^{13}C natural abundance technique.

Ecosystem (References)	Aggregate size class	μm	MRT (yr)
Tropical pasture ^[44]	M	>200	60
	m	<200	75
Temperate pasture grasses ^[19]	M	212–9500	140
	m	53–212	412
Soybean ^[45]	M	250–2000	1.3
	m	100–250	7
Corn ^[46]	M	>250	14
	m	50–250	61
Corn ^[47]	M	>250	42
	m	50–250	691
Wheat–fallow, no tillage ^[48]	M	250–2000	27
	m	53–250	137
Wheat–fallow, conventional tillage ^[48]	M	250–2000	8
	m	53–250	79
Average $\pm$ SE	M		42 $\pm$ 18
	M		209 $\pm$ 95

Note: M, macroaggregate; m, microaggregate; SE, standard error.

gradations of silt- and clay-sized particles) that have been variously related to climate, clay mineralogy, and fractionation methodology.^[34,38,39]

For a given set of biotic and abiotic conditions, the turnover of different SOM pools depends mechanistically on the quality and biochemical recalcitrance of the organic matter and its accessibility to decomposers. With other factors equal, clay soils retain more SOM with longer MRTs than do sandy soils.^[40] Readily decomposable materials can become chemically protected from decomposition by association with clay minerals and by sorption to humic colloids.^[38,41] Clay mineralogy also plays an important role. For example, montmorillonitic clays and allophanes generally afford more protection than illites and kaolinites.^[42] In addition, the spatial location of SOM within the soil matrix determines its physical accessibility to decomposers. Relatively labile material may become physically protected by incorporation into soil aggregates^[43] or by deposition in micropores inaccessible even to bacteria. Studies of the average MRTs of organic matter in macroaggregates vs. microaggregates show consistently slower turnovers in microaggregates (Table 3). Thus, a much higher proportion of the SOM occluded in microaggregates consists of stabilized materials with relatively long MRTs.

REFERENCES

- Paul, E.A. Dynamics of organic matter in soils. *Plant Soil* **1984**, *76*, 275–285.
- Trumbore, S.E. Comparison of carbon dynamics in tropical and temperate soils using radiocarbon measurements. *Global Biogeochem. Cy.* **1993**, *7*, 275–290.
- Olson, J.S. Energy storage and the balance of producers and decomposers in ecological systems. *Ecology* **1963**, *44*, 322–331.
- Jenny, H. *The Soil Resource—Origin and Behavior*; Springer: New York, 1980; 377 pp.
- Schimmel, D.S. *Theory and Application of Tracers*; Academic Press: San Diego, 1993; 119 pp.
- Buyanovsky, G.A.; Kucera, C.L.; Wagner, G.H. Comparative analyses of carbon dynamics in native and cultivated ecosystems. *Ecology* **1987**, *68*, 2023–2031.
- Jenkinson, D.S. The turnover of organic carbon and nitrogen in soil. *Phil. Trans. R. Soc. Lond. Ser. B* **1990**, *329*, 361–368.
- Jastrow, J.D. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. *Soil Biol. Biochem.* **1996**, *28*, 665–676.
- Cerri, C.; Feller, C.; Balesdent, J.; Victoria, R.; Plenecassagne, A. Application du tracage isotopique naturel en ^{13}C a l'etude de la dynamique de la matiere organique dans les sols. *C.R. Acad. Sci. Paris Ser. II* **1985**, *300*, 423–428.
- Balesdent, J.; Mariotti, A. Measurement of soil organic matter turnover using ^{13}C natural abundance. In *Mass Spectrometry of Soils*; Boutton, T.W., Yamasaki, S., Eds.; Marcel Dekker: New York, 1996; 83–111.
- Goh, K.M. Carbon dating. In *Carbon Isotope Techniques*; Coleman, D.C., Fry, B., Eds.; Academic Press: San Diego, 1991; 125–145.
- Goh, K.M. Bomb carbon. In *Carbon Isotope Techniques*; Coleman, D.C., Fry, B., Eds.; Academic Press: San Diego, 1991; 147–151.
- Harrison, K.G.; Broecker, W.S.; Bonani, G. The effect of changing land use on soil radiocarbon. *Science* **1993**, *262*, 725–726.
- Hendrix, P.F. Long-term patterns of plant production and soil carbon dynamics in a georgia piedmont agroecosystem. In *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*; Paul, E.A., Paustian, K., Elliott, E.T., Cole, C.V., Eds.; CRC Press: Boca Raton, 1997; 235–245.
- Buyanovsky, G.A.; Brown, J.R.; Wagner, G.H. Sanborn field: Effect of 100 years of cropping on soil parameters influencing productivity. In *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*; Paul, E.A., Paustian, K., Elliott, E.T., Cole, C.V., Eds.; CRC Press: Boca Raton, 1997; 205–225.
- Balesdent, J.; Mariotti, A.; Boisgontier, D. Effect of tillage on soil organic carbon mineralization estimated from ^{13}C abundance in maize fields. *J. Soil Sci.* **1990**, *41*, 587–596.
- Vitarello, V.A.; Cerri, C.C.; Andreux, F.; Feller, C.; Victoria, R.L. Organic matter and natural carbon-13 distributions in forested and cultivated oxisols. *Soil Sci. Soc. Am. J.* **1989**, *53*, 773–778.
- Desjardins, T.; Andreux, F.; Volkoff, B.; Cerri, C.C. Organic carbon and ^{13}C contents in soils and soil size-fractions, and their changes due to deforestation and pasture installation in Eastern Amazonia. *Geoderma* **1994**, *61*, 103–118.
- Jastrow, J.D.; Boutton, T.W.; Miller, R.M. Carbon dynamics of aggregate-associated organic matter estimated by carbon-

- 13 natural abundance. *Soil Sci. Soc. Am. J.* **1996**, *60*, 801–807.
20. Martin, A.; Mariotti, A.; Balesdent, J.; Lavelle, P.; Vuattoux, R. Estimate of organic matter turnover rate in a savanna soil by ^{13}C natural abundance measurements. *Soil Biol. Biochem.* **1990**, *22*, 517–523.
21. Trouve, C.; Mariotti, A.; Schwartz, D.; Guillet, B. Soil organic carbon dynamics under *eucalyptus* and *pinus* planted on savannas in the congo. *Soil Biol. Biochem.* **1994**, *26*, 287–295.
22. Paul, E.A.; Collins, H.P.; Leavitt, S.W. Dynamics of resistant soil carbon of midwestern agricultural soils measured by naturally-occurring ^{14}C abundance. *Geoderma* **2001**, *104*, 239–256.
23. Paul, E.A.; Follett, R.F.; Leavitt, S.W.; Halvorson, A.; Peterson, G.A.; Lyon, D.J. Radiocarbon dating for determination of soil organic matter pool sizes and dynamics. *Soil Sci. Soc. Am. J.* **1997**, *61*, 1058–1067.
24. Jenkinson, D.S.; Harkness, D.D.; Vance, E.D.; Adams, D.E.; Harrison, A.F. Calculating net primary production and annual input of organic matter to soil from the amount and radiocarbon content of soil organic matter. *Soil Biol. Biochem.* **1992**, *24*, 295–308.
25. Trumbore, S.E.; Bonani, G.; Wolfli, W. The rates of carbon cycling in several soils from AMS ^{14}C measurement of fractionated soil organic matter. In *Soils and the Greenhouse Effect*; Bouwman, A.F., Ed.; Wiley: London, 1990; 407–414.
26. O'Brien, B.J. Soil organic carbon fluxes and turnover rates estimated from radiocarbon enrichments. *Soil Biol. Biochem.* **1984**, *16*, 115–120.
27. Bol, R.A.; Harkness, D.D.; Huang, Y.; Howard, D.M. The influence of soil processes on carbon isotope distribution and turnover in the british uplands. *Eur. J. Soil Sci.* **1999**, *50*, 41–51.
28. Six, J.; Elliot, E.T.; Paustian, K.; Doran, J.W. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Sci. Soc. Am. J.* **1998**, *62*, 1367–1377.
29. Ryan, M.C.; Aravena, R.; Gillham, R.W. The use of ^{13}C natural abundance to investigate the turnover of the microbial biomass and active fractions of soil organic matter under two tillage treatments. In *Soils and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1995; 351–360.
30. Clapp, C.E.; Allmaras, R.R.; Layese, M.F.; Linden, D.R.; Dowdy, R.H. Soil organic carbon and ^{13}C abundance as related to tillage, crop residue, and nitrogen fertilization under continuous corn management in Minnesota. *Soil Till. Res.* **2000**, *55*, 127–142.
31. Davidson, E.A.; Trumbore, S.E.; Amundson, R. Soil warming and organic carbon content. *Nature* **2000**, *408*, 789–790.
32. Ågren, G.I.; Bosatta, E. Theoretical analysis of the long-term dynamics of carbon and nitrogen in soils. *Ecology* **1987**, *68*, 1181–1189.
33. Parton, W.J.; Schimel, D.S.; Cole, C.V.; Ojima, D.S. Analysis of factors controlling soil organic matter levels in great plains grasslands. *Soil Sci. Soc. Am. J.* **1987**, *51*, 1173–1179.
34. Balesdent, J. The significance of organic separates to carbon dynamics and its modeling in some cultivated soils. *Eur. J. Soil Sci.* **1996**, *47*, 485–493.
35. Christensen, B.T. Matching measurable soil organic matter fractions with conceptual pools in simulation models of carbon turnover: Revision of model structure. In *Evaluation of Soil Organic Matter Models*; Powlson, D.S., Smith, P., Smith, J.U., Eds.; Springer: Berlin, 1996; 143–159.
36. Martel, Y.A.; Paul, E.A. The use of radiocarbon dating of organic matter in the study of soil genesis. *Soil Sci. Soc. Am. Proc.* **1974**, *38*, 501–506.
37. Trumbore, S.E.; Chadwick, O.A.; Amundson, R. Rapid exchange between soil carbon and atmospheric carbon dioxide driven by temperature change. *Science* **1996**, *272*, 393–396.
38. Christensen, B.T. Physical fractionation of soil and organic matter in primary particle size and density separates. *Adv. Soil Sci.* **1992**, *20*, 1–90.
39. Feller, C.; Beare, M.H. Physical control of soil organic matter dynamics in the tropics. *Geoderma* **1997**, *79*, 69–116.
40. Sorensen, L.H. The influence of clay on the rate of decay of amino acid metabolites synthesized in soils during decomposition of cellulose. *Soil. Biol. Biochem.* **1974**, *7*, 171–177.
41. Jenkinson, D.S. Soil organic matter and its dynamics. In *Russell's Soil Conditions and Plant Growth*; Wild, A., Ed.; Wiley: New York, 1988; 564–607.
42. Dalal, R.C.; Bridge, B.J. Aggregation and organic matter storage in sub-humid and semi-arid soils. In *Structure and Organic Matter Storage in Agricultural Soils*; Carter, M.R., Stewart, B.A., Eds.; CRC Press: Boca Raton, 1996; 263–307.
43. Tisdall, J.M.; Oades, J.M. Organic matter and water-stable aggregates in soils. *J. Soil Sci.* **1982**, *33*, 141–163.
44. Skjemstad, J.O.; Le Feuvre, R.P.; Prebble, R.E. Turnover of soil organic matter under pasture as determined by ^{13}C natural abundance. *Aust. J. Soil Res.* **1990**, *28*, 267–276.
45. Buyanovsky, G.A.; Aslam, M.; Wagner, G.H. Carbon turnover in soil physical fractions. *Soil Sci. Soc. Am. J.* **1994**, *58*, 1167–1173.
46. Monreal, C.M.; Schulten, H.R.; Kodama, H. Age, turnover and molecular diversity of soil organic matter in aggregates of a gleysol. *Can. J. Soil Sci.* **1997**, *77*, 379–388.
47. Angers, D.A.; Giroux, M. Recently deposited organic matter in soil water-stable aggregates. *Soil Sci. Soc. Am. J.* **1996**, *60*, 1547–1551.
48. Six, J.; Elliot, E.T.; Paustian, K. Aggregate and soil organic matter dynamics under conventional and no-tillage systems. *Soil Sci. Soc. Am. J.* **1999**, *63*, 1350–1358.

Soluble Salts: Translocation and Accumulation

Jimmie L. Richardson

Department of Soil Science, North Dakota State University, Fargo, North Dakota, U.S.A.

Abstract

Translocation and accumulation of soluble salts in soils encompass both physical transport through a landscape to an accumulation area and soil profile processes of accumulation at that point. Soluble salts accumulate by sequences of mineral precipitation (evaporite sequences) that control the salt types of the resulting evaporite deposits either in or on the soil. These processes are driven by water that is invariably in the soil, not surface water. The water progressively becomes more concentrated in dissolved solids either by dissolving more salts as the water travels in the landscape or by losing water to the atmosphere via evapotranspiration. Either way, the dissolved solids precipitate as salts in zones in discharge areas on the landscape.

TRANSLOCATION IN LANDSCAPES

In a landscape, step one of salt translocation results from removal of labile salts in the recharge and throughflow zones of the landscape and saturated, often-transient flow, to the discharge zones. Dissolution occurs in soils located in the recharge zone of landscapes when rain-water or snowmelt infiltrates and percolates through soil or geologic materials. The landscape sources of water can be divided into local, regional, and irrigation water. Local water moves as transient flow (interflow or throughflow) from nearby areas either in periods of wetness or from local water sources. The regional systems are larger scale created by fracture or aquifer flow to the accumulation areas, often via artesian aquifers that discharge under pressure. Irrigation water is gathered from distant streams, aquifers, or other source areas and transported in engineered channel or pipe systems to the accumulation sites. Landscape translocation removes salts from recharge areas and transfers the dissolved solids to discharge areas where accumulation can occur.

Accumulation, or step two of translocation within the landscape, occurs because inefficient water removal exists during discharge. In semiarid and arid climates, the lack of water and the high potential evaporation rates create conditions with poorly integrated intermittent streams. Thus, when water is present, it cannot be transported externally from the accumulation zones or saline soils. In young or constructional landscapes, such as a Pleistocene or Holocene lake or till plain, an integrated drainage network has usually not developed; without drainage channels water cannot be effectively removed. For example, in Grand Forks County, North Dakota, the subhumid climate with 0.5 m of annual precipitation at about 45 degrees north latitude maintains relatively low evaporation potentials. Over 25% of the land area is variously impacted by saline or sodic soils. Saline soils have a saturated paste electrical

conductivity that exceeds 4 dS/m, and sodic soils have sodium (Na) absorption ratios exceeding 12 dS/m. Some areas that are strong discharge areas or receive abundant water in deserts have very high evaporation potentials and ineffective drainage. These result in hypersalinity or salinity that exceeds an electrical conductivity of 20 dS/m. The Bonneville Salt Flats, which lie around the Great Salt Lake in Utah, are examples.

Accumulation sites often have a high water table either seasonally or periodically during pluvial times of drought-pluvial cycles. At these sites, increases in salinity correlate best to pluvial or wetter periods because this is when the water table is the highest and evapotranspiration losses concentrate salts from the water table. Irrigation water creates a perpetually pluvial condition that often creates a high water table, resulting in persistent salinization. Salinization or sodification potentially occurs rapidly unless adequate drainage has been established before irrigation.

As a short synopsis of influence of the landscape, the translocation of dissolved solids relates closely to periods of saturated flow whereby salts increase as the path of transportation increases and water is lost to evapotranspiration.

TRANSLOCATION AND ACCUMULATION WITHIN SOILS

Translocation of salts within the soil involves a constant flux between the upward water losses by matric and osmotic flow and the downward infiltration and subsequent percolation of rain or irrigation water. The actual accumulation in the root zone of plants, which follows, relates to matric flow and osmotic flow. In matric flow, which is often called capillarity, water moves slowly from wet soil to dry soil. The flow direction is independent of gravity, instead it is dictated by moisture content and the matric potential

possessed by the soil. Soils with abundant surface area, such as those with clay textures, tend to have higher matric potentials at given water contents. In osmotic flow, water moves from low salt content to high salt content. Soil salinity increases hygroscopic water because of the increase in osmotic potential, often to the point where a soil looks moist but plants cannot obtain water.

Salt efflorescences often disappear after a rain event because the water that infiltrates carries the labile salts downward. Often, however, casts of less soluble minerals remain, such as gypsum, are good indicators of saline condition. The salt efflorescence will return after a few rainless days with high evaporation stress. As evaporation stress occurs on the soil, the matric forces move water to the surface where the water vaporizes and the salts precipitate. Few really labile minerals form in the soil. Usually when somebody notes a “salt” in the soil it is gypsum. One exception is the winter precipitation of mirabilite. Mirabilite is a Na sulfate that has 10 waters of hydration. With such a high amount of hydrogen bonding the ice-like crystals form easier at cool temperatures.^[1,2] The usual events that tend to dominate saline soils are evaporative and discharge driven by evaporation with flow reversals during rainfall, irrigation, or snowmelt events.

Saline seeps occur in Western United States and Canada when an aquifer is activated during a pluvial time of the pluvial–drought cycle. The aquifer does not function unless enough water reaches it to create saturated flow. Once saturated flow occurs, the water moves quickly to points of discharge on the landscape and the seepage water dries out in the air or by plant usage, leaving the salts behind. In these cases, the translocation to the soil is lateral from the

landscape. A possible rule of thumb for the recharge area of most small saline seeps is about 100 m or less.

Precipitation of translocated salts occurs in sequence as evaporation or evapotranspiration concentrates water. First to form is calcite that removes 1 mole each of calcium (Ca) and carbonate for every mole of calcite formed. Depending on which of these two ions is dominant, the next to form is either gypsum or magnesium (Mg) calcite or dolomite with a rise in pH. In fact if the pH is over 8.5, one can almost be sure that the dominant carbonate is of Na.^[3]

One mole of gypsum removes 1 mole of Ca and 1 mole of sulfate. If the sulfate is dominant over Ca as gypsum precipitates, the water and the salt efflorescence become high in Na and Mg with sulfate and chloride as the anions.^[3] Although the Eugster and Jones model was for a fully closed basin system, their sequence of evaporites is a good predictive tool in most saline situations because these situations represent groundwater discharge or ineffective drainage that act like a closed basin system or labile ion sink.

REFERENCES

1. Beke, G.J.; Palmer, C.J. Subsurface occurrence of mirabilite in a mollisol of Southern Alberta, Canada: A case study. *Soil Sci. Soc. Am. J.* **1989**, *53*, 1611–1614.
2. Timpson, M.E.; Richardson, J.L.; Keller, L.P.; McCarthy, G.J. Evaporite mineralogy associated with saline seeps in Southwestern North Dakota. *Soil Sci. Soc. Am. J.* **1986**, *50*, 490–493.
3. Eugster, H.P.; Jones, B.F. Behavior of major solutes during closed basin brine evolution. *Am. J. Sci.* **1979**, *279*, 609–631.

Solute Transport

Arne E. Olsen

University of Florida, Gainesville, Florida, U.S.A.

James W. Jawitz

Soil and Water Science Department, University of Florida, Gainesville, Florida, U.S.A.

Abstract

The movement of chemicals through the soil environment has been the primary focus of agricultural and environmental soil physics during the last several decades. Agricultural physicists have focused on the movement of nutrients into and through the rooting zone, with the goal of maximizing the amount of nutrient uptake while limiting the amount of nutrients passing through the root zone. Environmental soil physicists have focused on describing the movement of wastes (industrial, commercial, and residential) through the soil system. The two fields converge with respect to the movement of pesticides and excess nutrients, and these areas have proven to be fertile areas of research. Without regard to the area of study, significant advancements in the field of chemical transport through the soil system have been made in the preceding decades. The intent of this entry is to elucidate the fundamental concepts of chemical transport in the soil environment, including interactions of soil water, soil solids, and the chemicals of interests.

PRINCIPLES OF SOLUTE TRANSPORT

The movement of chemicals through the soil environment is generally described in terms of three fundamental mechanisms: advective transport, diffusive transport, and dispersive transport.^[1] These transport mechanisms are combined with the principle of continuity (or conservation of mass) to arrive at a general transport equation known as the Advection–Dispersion Equation (ADE).

For a 1-D system, the continuity equation has the form:

$$\frac{\partial M_s}{\partial t} = -\frac{\partial J_s}{\partial z} \quad (1)$$

where M_s is the total solute mass per volume of porous media [ML^{-1}], t is the time [T], J_s is the solute flux [$\text{ML}^{-1}\text{i}^{-1}$], and z is the spatial dimension [L]. Eq. 1 states that the change in solute mass storage is equal to the change in solute flux (or flux divergence). For a solute that reacts with the soil solids, the total mass of solute in the system can be expressed as:

$$M_s = \theta C_w + \rho_b S \quad (2)$$

where θ is the water content of the soil on a volume basis [L^3h^{-3}], C_w is the solvent-phase concentration [ML^{-3}], ρ_b is the bulk density of the porous media [ML^{-3}], and S is the mass of solute sorbed per unit mass of porous media [MM^{-1}]. Sorption is the interaction of a solute with a solid surface, generally described as either adsorption (interaction at the surface) or absorption (penetration of the solute into the solid).

The total solute flux is the sum of the advective and dispersive fluxes (J_A and J_D):

$$J_s = J_A + J_D \quad (3)$$

Advection

The advective flux describes the movement of solutes with the bulk solvent flow:

$$J_A = q_w C_w \quad (4)$$

where q_w is the solvent flux (or specific discharge) [LT^{-1}]. The flux of the solvent can be described using either Darcy's equation for saturated conditions or Buckingham's modification to Darcy's equation for unsaturated conditions:

$$q_w = -K(\theta) \frac{\partial h}{\partial z} \quad (5)$$

where $K(\theta)$ is unsaturated hydraulic conductivity [LT^{-1}] and h is the total hydraulic head [L].

Diffusion

Diffusion results from thermally induced agitation of solute molecules (i.e., Brownian motion). In soil solutions where concentration gradients exist, the random movement of molecules will tend to cause molecules in the higher concentration regions to move through the solvent to areas of lower concentration. This process is mathematically

modeled using Fick's First Law, which relates the solute flux density to the concentration gradient and a constant of proportionality that is dependent on the solute of interest. Fick's Law can be written for the 1-D case as:

$$J_{\text{Diff}} = -D_0 \frac{\partial C_w}{\partial z} \quad (6)$$

where D_0 is the diffusion coefficient [$L^2 t^{-1}$].^[2] For diffusion in porous media, the diffusion coefficient must be modified to account for the irregular flow network, thus the diffusion coefficient for flow in soils has been defined as:

$$D_s = D_0 \theta \Gamma \quad (7)$$

where Γ is the tortuosity, an empirical parameter that relates the straight line path to the actual roundabout path through the solvent-filled pores.^[3] In soils, solute movement resulting from molecular diffusion is generally very slow relative to advective and dispersive transport.

Dispersion

Neither pore sizes nor velocities are uniform throughout porous media, and there are a wide range of pore sizes. Advection through pores of differing sizes results in dispersion of the solute. Solute spreading resulting from this differential advection is referred to as mechanical dispersion, D_m [$L^2 T^{-1}$]. The spreading processes of mechanical dispersion and molecular diffusion are linearly combined and referred to as hydrodynamic dispersion, D [$L^2 T^{-1}$]:

$$D = D_s + D_m \quad (8)$$

Conceptually and mathematically, such spreading can be approximated by analogy to molecular diffusion. Thus, the dispersive solute flux can be written as:

$$J_D = -\theta D \frac{\partial C_w}{\partial z} \quad (9)$$

The effects of molecular diffusion are small compared to mechanical dispersion when the solvent velocity is large. As the solvent velocity approaches zero, the diffusion term dominates.

ADVECTION-DISPERSION EQUATION

Substitution of Eqs. 2, 4, and 9 into Eq. 1 yields the ADE for reactive solute transport through porous media:

$$\frac{\partial(\theta C_w + \rho_b S)}{\partial t} = -\frac{\partial q_w}{\partial z} + \frac{\partial}{\partial z} \left(\theta D \frac{\partial C_w}{\partial z} \right) \quad (10)$$

Note that this form of the ADE does not include a source-sink term.

SORPTION

Reactive solutes interact with the solid matrix during flow, as opposed to ideal or nonreactive solutes that are transported passively through the porous medium with no interactions with the solid matrix. Natural soils are composed of charged particles, and these may be clay minerals that have developed surface charge due to isomorphous substitution or organic or clay minerals that have developed charge from pH effects. When the solute and the soil system are oppositely charged, the solute is attracted to the soil surface, resulting in a retarded rate of transport through the soil. Hydrophobic interactions that promote partitioning of non-polar solutes into organic matter in soils may similarly retard solute transport.

For reversible reactions, a solute may be sorbed and desorbed by the porous medium. The solute is therefore distributed between the solvent and solid phases. In order to write the ADE in terms of solvent-phase concentration, it is necessary to develop a functional relationship between C_w and S . Various functional expressions, usually referred to as sorption isotherms, are possible. Reversible, instantaneous sorption isotherms are either linear or non-linear. For the linear case, sorbed-phase and solvent-phase solute concentrations are related by a proportionality constant called the distribution coefficient, K_d [$L^3 M^{-1}$]:

$$S = K_d C \quad (11)$$

Two alternative models are frequently used to describe non-linear sorption isotherms: Freundlich and Langmuir. Freundlich isotherms have the form:

$$S = K_d C_w^N \quad (12)$$

where N is a dimensionless empirical constant. Generally, isotherms with N is a dimensionless empirical constant. Generally, isotherms with orption isotherms: Freundlich and Langmuir. Freundlich isotherms have the form: linear or non. Isotherms with N is a dimensionless empirical constant. Generally, isotherms with orption isotherms: Freundlich and Langmuir. Freundlich isotherms have the form: linear or non. Isotherms with linear case, sorbed-phase and solvent-phase solute S_{max} :

$$S = \frac{S_{\text{max}} k C_w}{1 + k C_w} \quad (13)$$

where k is an empirical constant [$L^3 t^{-1}$].

For steady-state water flow, homogeneous media, and linear sorption, the ADE can be simplified by substitution of Eq. 11 into Eq. 10 resulting in the following equation:

$$R \frac{\partial C}{\partial t} = D \frac{\partial^2 C}{\partial z^2} - v \frac{\partial C}{\partial z} \quad (14)$$

where $v = \text{eq}_w / \theta$ is the solvent pore velocity and R is the retardation factor, given by

$$R = 1 + \frac{\rho_b}{\theta} K_d \quad (15)$$

for non-reactive solutes [K_d or non-R or non-reactive solutes (r , given by $\rho_b K_d$ increases)], R increases and is a measure of the degree of retardation for sorbed solutes.

Eq. 14 has been solved analytically for a variety of boundary conditions (e.g., Wierenga, Parker and van Genuchten, and Huang and van Genuchten^[4-6]). These solutions can be used to describe solute concentration distributions as a function of space or time. Solutions to more complex applications of the ADE such as transient flow conditions, irregular system geometries, and fluid or media heterogeneities require numerical solution techniques. Gray and Pinder^[7] outlined the fundamental principles of numerical modeling of solute transport. Two broad categories of numerical modeling techniques are available, finite difference and finite element.

NONEQUILIBRIUM SORPTION

Observations that the ADE did not replicate a variety of observed concentration distributions led to the development of nonequilibrium transport theories for both chemical and physical phenomenon. Breakthrough curves for nonequilibrium systems are asymmetric because some solute molecules behave ideally, moving rapidly through the system, while others are slowed due to either chemical or physical interactions with the soil.

Physical non-equilibrium results from the partitioning of the flow field into mobile and immobile regions.^[8] Solutes are advected and dispersed in the mobile regions, while the immobile region acts as a source/sink that exchanges solute mass with the mobile region via diffusion. van Genuchten and Wierenga^[9] developed an analytic solution to the physical non-equilibrium steady-state flow problem for reactive solutes.

Chemical non-equilibrium results from the existence of sites within the soil matrix where sorption and desorption are rate-limited. A two-site model has been proposed^[10,11] where some of the sorption sites are in equilibrium with the soil solution while others share a kinetic relation with the soil solution. Lindstrom and Narasimhan^[12] developed analytical solutions to the 1-D non-equilibrium ADE. An alternative to the two-site model approach is to consider the sorption sites as exhibiting a distribution of rate constants. This technique accounts for the heterogeneity of sorption mechanisms and kinetic rates in the soil system without increasing the number of required model parameters.^[13]

TRANSFER FUNCTIONS AND SOLUTE MOVEMENT THROUGH SOIL

Alternatives to the ADE methodology for modeling the movement of solutes through soil have been proposed.

For example, transfer functions have been proposed to describe solute transport without the constraint of describing the causative transport processes, instead shifting the focus to characterizing the output from the system.^[14] The transfer function approach requires knowledge of the input solute distribution and the general nature of the output solute distribution. With this knowledge, the solute concentration exiting the soil system can be determined at any distance from the input location. Innovative approaches to modeling the movement of solutes through the soil system will continue to be the primary focus of soil and environmental physics.

REFERENCES

1. Biggar, J.W.; Nielsen, D.R. Spatial variability and leaching characteristics of a field soil. *Water Resour. Res.* **1976**, *12*, 78–84.
2. Crank, J. *The Mathematics of Diffusion*; Clarendon Press: Oxford, 1975.
3. Hillel, D. *Environmental Soil Physics*; Academic Press: San Diego, 1998.
4. Wierenga, P.J. Solute distribution profiles computed with steady state and transient water movement. *Soil Sci. Soc. Am. J.* **1977**, *41*, 1050–1055.
5. Parker, J.C.; van Genuchten, M.Th. Determining transport parameters from laboratory and field tracer experiments. *VA Agric. Exp. Stat. Bull.* **1984**, 84–3.
6. Huang, K.; van Genuchten, M.Th. An analytical solution for predicting solute transport during ponded infiltration. *Soil Sci.* **1995**, *159*, 217–223.
7. Gray, W.G.; Pinder, G.F. An analysis of numerical solution of the transport equation. *Water Resour. Res.* **1976**, *12*, 547–555.
8. Coats, K.H.; Smith, B.D. Dead end pore volume and dispersion in porous media. *Soc. Petrol. Eng. J.* **1964**, *4* (1), 73–84.
9. van Genuchten, M.Th.; Wierenga, P.J. Mass transfer in sorbing porous media. 1. analytic solutions. *Soil Sci. Soc. Am. Proc.* **1976**, *40*, 473–480.
10. Selim, H.M.; Mansell, R.S.; Elzeftaway, A. Distribution of 2,4-D and water in soil during infiltration and redistribution. *Soil Sci.* **1976**, *121* (3), 176–183.
11. Cameron, D.R.; Klute, A. Convective–dispersive solute transport with a combined equilibrium and kinetic adsorption model. *Water Resour. Res.* **1977**, *13* (1), 183–188.
12. Lindstrom, F.T.; Narasimhan, M.N.L. Mathematical theory of a kinetic model for dispersion of previously distributed chemicals in a sorbing porous medium. *SIAM J. Appl. Math.* **1974**, *24*, 469–510.
13. Chen, W.; Wagenet, R.J. Solute transport in porous media with sorption site heterogeneity. *Environ. Sci. Technol.* **1995**, *29*, 2725–2734.
14. Jury, W.A.; Roth, K. *Transfer Functions and Solute Movement Through Soil: Theory and Applications*; Birkhauser Verlag: Basel, 1990.

Spectral Mixtures: Soil—Plant

Alfredo R. Huete

*Department of Soil, Water and Environmental Sciences, University of Arizona,
Tucson, Arizona, U.S.A.*

Abstract

Approximately 70% of the earth's terrestrial surface consists of open canopies with vegetation overlying various proportions of exposed soil and litter background. The aim of remote sensing is to exploit and model these complex patterns of surface energy interactions for the purpose of mapping and extracting information about the biophysical and biochemical character of soils. Several approaches are used to analyze soil-plant spectral mixtures, including vegetation indices (VIs) and mixture models. VIs are useful in assessing the spatial and temporal dynamics of the vegetation component but yield little information on soil type and soil properties. Mixture models, on the other hand, have been successfully used to discriminate and monitor both soil and vegetation biophysical properties from broadband multispectral sensor systems to hyperspectral sensor imagery. Hyperspectral sensors have greatly improved the identification of numerous soil and plant absorption features.

INTRODUCTION

There exists considerable knowledge about soil optical properties based on extensive laboratory analyses studies.^[1–3] Information about soils are generally obtained by analyses of their spectral signatures, which are a function of the absorption properties of their biogeochemical constituents, moisture condition, and the size, shape, and geometry of the soil particles. At the landscape level, it is much more difficult to monitor and assess soil properties due to their extreme spatial variability and the masking of the soil surface by vegetation and litter. Airborne and satellite remote sensing are used to characterize and map soils in several ways: 1) direct remote sensing measurements of exposed soil surfaces in vegetated (e.g., canopy gaps) and non-vegetated areas (hyperarid zones); 2) employment of spectral mixture models to generate soil component images from remote sensing data over partially vegetated areas; and 3) one can infer soil properties based on remote sensing measurements of the overlying vegetation.

SOIL-VEGETATION CANOPY MIXTURES

In arid, semiarid, and subhumid regions, soil surfaces are rarely spectrally pure and often contain significant quantities of litter and vegetation mixing with the soil signal.^[4] The remotely sensed, spectral responses from these surfaces are a function of the number and type of reflecting components, their optical properties, and their relative proportions. The optical properties and mixture proportions further vary seasonally and with land cover conversions.^[5] Resolving the extent of soil and vegetation cover is important to hydrologic and biogeochemical processes and crucial for functional analyses of ecosystems.^[6–8]

BASICS OF MIXTURE MODELING

Spectral mixture models are used in remote sensing to separate remote sensing measurements into their soil, non-photosynthetically active vegetation (NPV), and vegetation components. Such models have utility in a variety of applications, including biogeochemical studies, leaf water content, land degradation, land cover conversions, fuelwood assessment, and soil and vegetation mapping.^[9–12] Soil-plant spectral mixtures may be modeled as linear or non-linear mixtures.^[13,14] In the linear case, also known as the “checkerboard” model, photons only interact with a single material and the fractions of each material are equivalent to their aerial proportions.^[15] Linear models are easy to use and work quite well over certain types of land cover conditions, such as desert shrub canopies.^[16] Mixture modeling generally involves three steps:

1. assess the dimensionality or number of spectral constituents in an image;
2. identify the physical nature of each of the constituents or “endmembers”;
3. derive the amounts of each component for each pixel in the image.

The first step is generally accomplished with principal components analysis. The second step is achieved with an end-member analysis whereby various reference spectra are used to model the image. Several types of endmember signatures may be utilized in spectral mixture analyses, including image endmembers, bundled spectra endmembers, and laboratory reference endmembers, which are cataloged in “spectral libraries”^[17] (Fig. 1). Typical endmembers used in remote sensing include green vegetation, soil, NPV, and

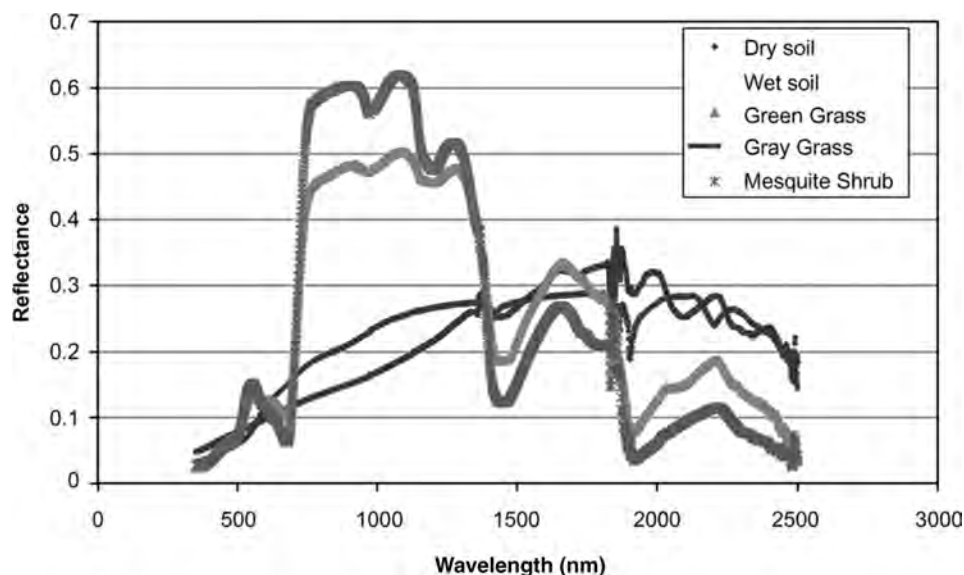


Fig. 1 Examples of reference spectral signatures of soil–plant components commonly found in semiarid canopies of the southwest United States.

“shade.” Shade is used to model pixels not fully illuminated and vary with topography.^[18]

In matrix notation, a linear mixture model is expressed as follows:

$$[D]=[R][C]+[\epsilon], \quad (1)$$

where $[D]$ is the spectral data matrix, $[R]$ is the reflectance matrix of the spectral constituents (endmembers), $[C]$ is the component contributions or “loadings” matrix, and $[\epsilon]$ is the residual errors. One can first model the spectral variability of a scene with several image endmembers followed by the use of reference endmembers to determine the composition of the image endmembers.^[19] Residual images are a quick way to find “outlying” pixels not adequately explained by the mixture model and may indicate the presence of unknown reflecting components. Residuals in the short-wave infrared region, attributable to cellulose and lignin in the vegetation, have been used to resolve and separate the NPV signal from soil.^[20] Residual images also contain the bulk of the noise and error in images. The products of mixture modeling include a set of component fraction images and a set of residual images containing the differences between modeled and measured data.

HYPERSPECTRAL SENSORS

Hyperspectral imaging sensors, with a large number of contiguous bands, carry valuable diagnostic information about soil and vegetation surfaces and have great potential for quantitative retrieval of their biophysical properties. The airborne visible–infrared imaging spectrometer (AVIRIS) sensor is of particular interest in soil–vegetation spectral mixture studies. AVIRIS operates in the 400–2450 nm region collecting 224 spectral bands with a nominal 10 nm spectral response function.^[21] This enables one to observe high-resolution spectra associated with unique absorption features, which are typically lost in

coarser waveband data, such as from Landsat thematic mapper (TM), SPOT, and AVHRR. The Hyperion Hyperspectral Imager onboard the Earth Observing-1 provides 220 contiguous bands covering the spectrum from 400 to 2500 nm.^[22] This sensor is a part of NASA’s New Millennium Program, Earth Observing series designed to test new technologies for Earth System Science studies.

Okin et al.^[23] used AVIRIS data in retrieving soil and vegetation information in arid and semiarid environments. They applied a multiple endmember spectral mixture analysis on AVIRIS imagery collected over the Mohave Desert in California and were able to reliably distinguish and map soil-surface types and vegetation cover. Palacios-Orueta et al.^[24] used hyperspectral AVIRIS imagery to map the spatial variation of soil organic matter and soil iron contents over two watersheds in the Santa Monica Mountains in California. They used a sequential spectral mixture analysis technique known as hierarchical foreground/background analysis to limit the influence of the vegetation signal while extracting the variation in the soil signal with laboratory derived “training” vectors.

INFERRING SOIL PROPERTIES THROUGH VEGETATION

About 25–30% of the land area have significant quantities of forest and dense vegetation with leaf area index values exceeding 1, including tropical and temperate forests and cropland. Where the surface is strongly vegetated, soil properties must be inferred from measurements of the vegetated surface in conjunction with models and field measurements. Spectral vegetation indices (VIs) as well as mixture models have been used to map vegetation characteristics to infer soil properties. Cwick et al.^[25] mapped the forest soils in Manitoba through hyperspectral reflectance measurements of Black Spruce (*Picea mariana*) needles.

They utilized the concentrations of potassium in the needles to infer the potassium distributions in the soil rooting zone with primary variations associated with either poorly or well-drained soils.

CONCLUSION

Approximately 70% of the earth's terrestrial surface consists of open canopies (deserts, grasslands, savannas, and open forests) with vegetation overlying various proportions of exposed soil and litter background.^[4] The aim of remote sensing is to exploit and model these complex patterns of surface energy interactions for the purpose of mapping and extracting information about the biophysical and biochemical character of soils. Several approaches are used to analyze soil-plant spectral mixtures, including VIs and mixture models. VIs are useful in assessing the spatial and temporal dynamics of the vegetation component but yield little information on soil type and soil properties. Mixture models, on the other hand, have been successfully used to discriminate and monitor both soil and vegetation biophysical properties from broadband multispectral sensor systems (Landsat TM, SPOT, and AVHRR) to hyperspectral sensor imagery (AVIRIS, Hyperion). Hyperspectral sensors have greatly improved the identification of numerous soil and plant absorption features, from soil-plant mixtures, related to mineralogy, liquid water, chlorophyll, cellulose, and lignin contents.^[1,7,9]

REFERENCES

- Ben-Dor, E.; Irons, J.R.; Epema, G. Soil reflectance. In *Remote Sensing for the Earth Sciences*; Rencz, A.N., Ed.; Wiley: New York, 1999; 111–188.
- Irons, J.R.; Weismiller, R.A.; Petersen, G.W. Soil reflectance. In *Theory and Application of Optical Remote Sensing*; Asrar, G., Ed.; Wiley: New York, 1989; 66–106.
- Baumgardner, M.F.; Silva, L.F.; Biehl, L.L.; Stoner, E.R. Reflectance properties of soils. *Adv. Agron.* **1985**, *38*, 1–44.
- Graetz, R.D. Remote sensing of terrestrial ecosystem structure: An ecologist's pragmatic view. In *Remote Sensing of Biosphere Functioning*; Hobbs, R.J., Mooney, H.A., Eds.; Springer: New York, 1990; 5–30.
- Adams, J.B.; Sabol, D.E.; Kapos, V.; Almeida Filho, R.; Roberts, D.A.; Smith, M.O.; Gillespie, A.R. Classification of multispectral images based on fractions of endmembers: Applications to land-cover change in the Brazilian Amazon. *Remote Sens. Environ.* **1995**, *52*, 137–154.
- Franklin, J.; Duncan, J.; Turner, D. Reflectance of vegetation and soil in Chihuahuan desert plant communities from ground radiometry using SPOT wavebands. *Remote Sens. Environ.* **1993**, *46*, 291–304.
- Asner, G.P.; Lobell, D.B. A biogeophysical approach for automated SWIR unmixing of soils and vegetation. *Remote Sens. Environ.* **2000**, *74*, 99–112.
- Novo, E.M.; Shimabukuro, Y.E. Identification and mapping of the Amazon habitats using a mixing model. *Int. J. Remote Sens.* **1997**, *18*, 663–670.
- Ustin, S.L.; Smith, M.O.; Adams, J.B. Remote sensing of ecological processes: A strategy for developing and testing ecological models using spectral mixture analysis. In *Scaling Physiological Processes: Leaf to Globe*; Ehleringer, J.R., Field, C.B., Eds.; Academic Press: New York, 1993; 339–357.
- Smith, M.O.; Ustin, S.L.; Adams, J.B.; Gillespie, A.R. Vegetation in deserts. 1. A regional measure of abundance from multispectral images. *Remote Sens. Environ.* **1990**, *31*, 1–26.
- Gillespie, A.G. Spectral mixture analysis of multispectral thermal infrared images. *Remote Sens. Environ.* **1992**, *42*, 137–145.
- Wessman, C.A.; Bateson, C.A.; Benning, T.L. Detecting fire and grazing patterns in tallgrass prairies using spectral mixture analysis. *Ecol. Appl.* **1997**, *7*, 493–512.
- Huete, A.R. Soil-dependent spectral response in a developing plant canopy. *Agron. J.* **1987**, *79*, 61–68.
- Borel, C.C.; Gerstl, S.A.W. Nonlinear spectral mixing models for vegetative and soil surfaces. *Remote Sens. Environ.* **1994**, *47*, 403–416.
- Huete, A.R. Separation of soil-plant spectral mixtures by factor analysis. *Remote Sens. Environ.* **1986**, *19*, 237–251.
- McGwire, K.; Minor, T.; Fenstermaker, L. Hyperspectral mixture modeling for quantifying sparse vegetation cover in arid environments. *Remote Sens. Environ.* **2000**, *72*, 360–374.
- Bateson, C.A.; Asner, G.P.; Wessman, C.A. Endmember bundles: A new approach to incorporating endmember variability in spectral mixture analysis. *Trans. Geosci. Remote Sens.* **2000**, *38*, 1083–1094.
- Adams, J.B.; Smith, M.O.; Johnson, P. Spectral mixture modeling, a new analysis of rock and soil types at the Viking Lander 1 site. *J. Geophys. Res.* **1986**, *91* (B8), B8098–B8112.
- Farrand, W.H.; Singer, R.B.; Merenyi, E. Retrieval of apparent surface reflectance from AVIRIS data: A comparison of empirical line, radiative transfer, and spectral mixture methods. *Remote Sens. Environ.* **1994**, *47*, 311–321.
- Roberts, D.A.; Smith, M.O.; Adams, J.B. Green vegetation, nonphotosynthetic vegetation, and soils in AVIRIS data. *Remote Sens. Environ.* **1993**, *44*, 255–269.
- Vane, G.; Green, R.O.; Vhrien, T.G.; Enmark, H.T.; Hansen, E.G.; Porter, W.M. The airborne visible/infrared imaging spectrometer (AVIRIS). *Remote Sens. Environ.* **1993**, *44*, 127–143.
- Pearlman, J.; Segal, C.; Liao, L.; Carman, S.; Folkman, M.; Browne, B.; Ong, L.; Ungar, S. *Development and Operations of the EO-1 Hyperion Imaging Spectrometer*. Available at http://eo1.gsfc.nasa.gov/new/validationReport/Technology/TRW_EO1%20Papers_Presentations/05.pdf (accessed October 2001).
- Okin, G.S.; Roberts, D.; Murray, B.; Okin, W.J. Practical limits on hyperspectral vegetation discrimination in arid and semiarid environments. *Remote Sens. Environ.* **2001**, *77*, 212–225.
- Palacios-Orueta, A.; Pinzón, J.E.; Ustin, S.L.; Roberts, D. Remote sensing of soils in the Santa Monica mountains: II. Hierarchical foreground and background analysis. *Remote Sens. Environ.* **1999**, *68*, 138–151.
- Cwick, G.J.; Aide, M.T.; Bishop, M.P. Use of hyperspectral and biochemical data from black spruce needles to map soils at a forest site in Manitoba. *Can. J. Remote Sens.* **1998**, *24*, 187–193.

Spodosols

Delbert Mokma

Department of Crop and Soil Sciences, Michigan State University, East Lansing, Michigan, U.S.A.

Abstract

Spodosols have a typical morphology, dark colored O or A horizon, ashy gray colored albic horizon, and dark reddish brown colored B horizon. Organic compounds formed by decomposition of organic matter in the surface form organo-metallic complexes with aluminum and iron that are released by weathering of minerals in the subsurface. These complexes accumulate and coat sand grains in the B horizons and may cement grains together into ortstein. The degree of Spodosol development is determined by parent material, topography, vegetation, climate, and time.

INTRODUCTION

Spodosols are a group of soils that are characterized by the presence of a spodic horizon in which active amorphous organic matter and aluminum (Al), with or without iron (Fe), have accumulated.^[1] The formative element of the word, “Spodosols,” was derived from the Greek word, “spodos,” which means wood ashes. The name is connotative of bleached eluvial (E) horizons that overly distinctive reddish brown spodic (Bhs or Bs) horizons. Most Spodosols are sandy. These materials have many macropores and are rapidly permeable. Sandy properties give Spodosols low available water capacities. Available water capacity increases as the amount of humus and sesquioxides in the spodic horizons increases.^[2] Cation exchange capacity is related to the amount of organic carbon (C) rather than clay. Spodosols are acidic, thus they are naturally infertile.

A typical Spodosol has a distinctive profile with four master horizons (Table 1). O and/or A horizons have black or dark brown colors, which result from accumulation of organic materials. E horizons have ashy gray color albic materials,^[1] as a result of the weathering of the non-resistant minerals and subsequent eluviation of amorphous materials. The Bhs or Bs horizons have dark reddish brown colors resulting from illuvial accumulations of organo-metallic complexes. Reddest hues, darkest values, and lowest chromas usually occur in upper parts of spodic horizons. Dark reddish brown colors of spodic horizons are related to amounts of amorphous materials.^[3] Hues become more yellow, and value and chroma increase with depth. Colors of C horizons tend to have yellower hues and higher values than spodic horizons.

Primarily, E horizons have uncoated sand grains. Immobilized, organo-metallic complexes coat most sand grains in B horizons. These coatings become thicker with time. As vegetation removes water from B horizons, desiccation

causes cracks to form in the coatings. Thin Al and/or Fe oxide coatings occur on sand grains in C horizons.

DISTRIBUTION

Spodosols are most extensive in areas of cool, humid, or perhumid climates.^[1] Some have formed in warm and hot, humid regions especially in areas of quartz-rich sand that have fluctuating water tables. Spodosols are not formed in arid areas.

FORMATION OF SPODOSOLS

Soil-Forming Processes

Translocation of humus and sesquioxides

Formation of Spodosols involves the formation, translocation, and immobilization of organo-metallic complexes. Organic acids form in forest canopies, canopy leachate, and soil organic matter. These acids cause weathering of minerals and release of Al and Fe in upper mineral horizons. Soluble organo-metallic complexes form and move downward through O and A horizons. Al and Fe in several Swedish and Finnish Spodosols are translocated downward as organo-metallic complexes.^[4]

Immobilization of organo-metallic complexes occurs as a result of increases in pH, microbial decomposition of the organic ligand, and when the amount of Al and Fe taken up by organic molecules exceeds a threshold. Vanillic and p-hydroxybenzoic acids dominated surface horizons of Spodosols in Michigan, whereas protocatechuic acid dominated spodic horizons.^[5] Distribution of protocatechuic acid paralleled those of Al and Fe, suggesting that it plays a role in the translocation process.

Table 1 Selected properties of a typical haplorthod.

Horizon	Depth (cm)	Texture	Color	Organic C (%)	ODOE ^a	Al ^b (%)	Fe ^b (%)
A	0–8	S	10YR 3/1	1.8	0.02	0.04	0.05
E	8–23	S	10YR 5/2	0.3	0.01	0.08	0.04
Bhs	23–28	S	5YR 3/2	1.8	0.24	0.40	0.45
Bs	28–41	S	7.5YR 4/4	0.7	0.07	0.43	0.15
BC	41–53	S	10YR 4/6	0.4	0.02	0.19	0.05
C	53–178	S	10YR 6/4	0.1	0.02	0.04	0.01

^aOptical density of oxalate extract.^bExtracted with ammonium oxalate.

Ortstein is a spodic horizon in which sand grains are cemented together by organic matter and Al with minor amounts of Fe and little or no silicon.^[6] Ortstein in somewhat poorly drained Spodosols usually have horizontal orientation, whereas that in well-drained Spodosols usually have vertical orientation.^[7] Ortstein in the former soils is more continuous and more strongly cemented, thereby being more root restrictive than the later soils. Ortstein reduces forest and blueberry production. Crushed ortstein receiving blueberry leaf extract recemented in 10 days in a laboratory study.^[8] Tillage operations to break ortstein are not likely to be permanent solutions. Crushed ortstein receiving blueberry and white pine leaf extracts recemented more completely and more strongly than bracken fern leaf extract.^[9]

Silicon

Silicon may play a role in the translocation of Al through the formation of imogolite.^[10] Imogolite tends to be greater in lower Bs horizons than in upper Bhs or Bs horizons. It may form in spodic horizons after Al had been translocated as organo-metallic complexes,^[11] which may be broken down by soil microbes, thereby releasing ionic Al. The Al could then form imogolite in the B horizon.

Soil-Forming Factors

Parent material

Spodosols developed in sandy to coarse loamy materials. They usually have little silicate clay. Quartz is the primary mineral in sand and silt fractions, with lesser amounts of orthoclase, plagioclase, mica, pyroxene, and amphibole.^[12] Accessory minerals are often highly weathered and are not easily identifiable. Less resistant minerals tend to have weathered from E and B horizons.

As clay content of parent materials increases, the translocation of organic C, Al, and Fe decreases, thickness of E and B horizons decreases, and B horizon color is lighter and less red.^[13] Silicate clays inhibit translocation of organic C, Al, and Fe by adsorbing these materials.

Topography

Spodosols form on surfaces that have shallow to very deep water tables. Spodic horizon development is influenced by depth and duration of water saturation.^[14] Spodic horizons do not readily form in soils that are saturated with water near the soil surface for long periods. Water saturation inhibits translocation processes. Poorly drained soils in Spodosol landscapes usually have less organic C, Al, and Fe in their B horizons than better-drained Spodosols in the hydrosequence. Some poorly and somewhat poorly drained Spodosols, especially those in warmer climates, have little Fe because it is chemically reduced and leached from the soil. Somewhat poorly drained Spodosols usually have more humus and sesquioxides than other members of the hydrosequence. Saturated zones inhibit water movement, thus, organo-metallic complexes are carried from A and E horizons to B horizons, but rarely through B horizons. In well-drained Spodosols, organo-metallic complexes can be carried by percolating waters to and through B horizons to lakes and streams.

Vegetation

Spodosols form under coniferous and deciduous forest but not under grasses. Spodic horizons formed under conifers have more Al and Fe than those under deciduous trees.^[15] Because organic compounds are involved with translocation of Al and Fe, more leaf or needle litter means more organic compounds available to form organo-metallic complexes.

Fire frequency influences spodic horizon development by affecting forest succession.^[16] When intervals between intense fires are 5 and 50 years, jack pine (*Pinus banksiana* L.) survives and regenerates. Much organic matter on the forest floor is destroyed by frequent fires; therefore, less organic compounds are available to form organo-metallic complexes with Al and Fe. Thus, E horizons are thin or non-existent, and B horizons have small amounts of organic C, Al, and Fe. If fire intervals are 50–350 years, white (*Pinus strobus* L.) and/or red (*Pinus resinosa* Ait.) pine succeed jack pine. Less organic matter is destroyed by fewer fires and more organic compounds are available to form complexes with Al and Fe and translocate them from

E horizons to B horizons. If fire intervals exceed 350 years, northern hardwoods (sugar maple, *Acer saccharum* Marsh; beech, *Fagus grandifolia*) succeed the pines. These trees are more shade tolerant and more fire resistant. Translocation of organic C, Al, and Fe continues, but more slowly because less Al and Fe and more Ca are biocycled.

Climate

Spodosols usually form in areas with cool, humid climates. Spodosols of warmer climates have less Fe compared to those of cool climates. Spodosols dominate areas with cryic and frigid soil temperature regimes but are less common in areas with a mesic soil temperature regime in New England.^[17] Strong spodic horizon development is associated with areas of deep snowpack before soils freeze in Michigan. Meltwater from snow can freely percolate through the soil in spring.

Time

Under optimum conditions Bs horizons may form in few 100 years, but usually they form more slowly. Eluvial horizons form in 300 years or less.^[18,19] Weak Bs horizons form in about 2000 years.^[19,20] At least 8000 years are required for most soils on a surface to qualify as Spodosols.

CONCLUSION

Spodosols have a typical morphology, dark colored O and/or A horizon, ashy gray colored albic horizon, and dark reddish brown colored B horizon. Organic compounds formed by decomposition of organic matter in the surface form organo-metallic complexes with Al and Fe that are released by weathering of minerals in the subsurface. These complexes accumulate and coat sand grains in the B horizons and may cement grains together into ortstein. The degree of Spodosol development is determined by parent material, topography, vegetation, climate, and time.

REFERENCES

1. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Survey*, 2nd Ed.; USDA Agriculture Handbook 436; U.S. Government Printing Office: Washington, 1999; Vol. 869.
2. Shetron, S.G. Distribution of free iron and organic carbon as related to available water in some forested sandy soils. *Soil Sci. Soc. Am. Proc.* **1974**, *38*, 359–362.
3. Mokma, D.L. Color and amorphous materials in spodosols from Michigan. *Soil Sci. Soc. Am. J.* **1993**, *57*, 125–128.
4. Lundstrom, U.S.; van Breeman, N.; Bain, D.C.; van Hees, P.A.W.; Giesler, R.; Gustafsson, J.P.; Ilvesniemi, H.; Karlton, E.; Melkerud, P.-A.; Olsson, M.; Riise, G.; Wahlberg, O.; Bergelin, A.; Bishop, K.; Finlay, R.; Jongmans, A.G.; Magnusson, T.; Mannerkoski, H.; Nordgren, A.; Nyberg, L.; Starr, M.; Tau Strand, L. Advances in understanding the podzolization process resulting from a multidisciplinary study of three coniferous forest soils in the Nordic countries. *Geoderma* **2000**, *94*, 335–353.
5. Vance, G.F.; Mokma, D.L.; Boyd, S.A. Phenolic compounds in soils of hydrosequences and developmental sequences of spodosols. *Soil Sci. Soc. Am. J.* **1986**, *50*, 992–996.
6. Lee, F.Y.; Yuan, T.L.; Carlisle, V.W. Nature of cementing materials in ortstein horizons of selected Florida spodosols. I. Constituents of cementing materials. *Soil Sci. Soc. Am. J.* **1988**, *52*, 1411–1418.
7. Mokma, D.L.; Doolittle, J.A.; Ternes, L.A. Continuity of ortstein in sandy spodic horizons of Michigan. *Soil Surv. Horiz.* **1994**, *35*, 6–10.
8. Bronick, C.J.; Mokma, D.L.; Li, H.; Boyd, S.A. Recementation of crushed ortstein by blueberry leaf extract. *Soil Sci. Soc. Am. J.* **2004**, *68*, 558–561.
9. Bronick, C.J.; Mokma, D.L.; Kizilkaya, K.; Li, H.; Boyd, S.A. Recementation of crushed ortstein by leaf extract from podzolizing and depodzolizing species. *Soil Sci.* **2004**, *169*, 306–313.
10. Farmer, V.C.; Russell, J.D.; Berrow, M.L. Imogolite and proto-imogolite allophane in spodic horizons: Evidence for a mobile aluminum silicate complex in podzol formation. *J. Soil Sci.* **1980**, *31*, 673–684.
11. Ugolini, F.C.; Dalgren, R. The mechanism of podzolization as revealed by soil solution studies. In *Podzols et Podzolisation*; Righi, D.; Chauvel, A., Eds.; AFES et INRA: Paris, 1987; 195–203.
12. Haile-Mariam, S.; Mokma, D.L. Mineralogy of two sandy spodosol hydrosequences in Michigan. *Soil Surv. Horiz.* **1995**, *36*, 121–132.
13. Gardner, D.R.; Whiteside, E.P. Zonal soils in the transitional region between the podzol and gray-brown podzolic regions in Michigan. *Soil Sci. Soc. Am. Proc.* **1952**, *16*, 137–141.
14. Mokma, D.L.; Sprecher, S.W. Water table depths and color patterns in spodosols of two hydrosequences in Northern Michigan, USA. *Catena* **1994**, *22*, 275–286.
15. Messenger, A.S.; Whiteside, E.P.; Wolcott, A.R. Climate, time, and organisms in relation to podzol development in Michigan sands. I. Site descriptions and microbiological observations. *Soil Sci. Soc. Am. Proc.* **1972**, *36*, 633–638.
16. Mokma, D.L.; Vance, G.F. Forest vegetation and origin of some spodic horizons, Michigan. *Geoderma* **1989**, *43*, 311–324.
17. Evans, C.V. Preliminary investigations of hydric soil hydrology and morphology in New Hampshire. In *Preliminary Investigations of Hydric Soil Hydrology and Morphology in the United States*; Wakely, J.S., Sprecher, S.W., Lynn, W.C., Eds.; Technical Report WRPDE-13; U.S. Army Engineer Waterways Experiment Station: Vicksburg, 1996; 114–126.
18. Gjems, O. Some notes on clay minerals in podzol profiles in Fennoscandia. *Clay Miner. Bull.* **1960**, *4*, 208–211.
19. Mokma, D.L.; Yli-Halla, M.; Lindqvist, K. Podzol formation in sandy soils of Finland. *Geoderma* **2004**, *120*, 259–272.
20. Franzmeier, D.P.; Whiteside, E.P. A chronosequence of podzols in Northern Michigan. I. Ecology and description of pedons. II. Physical and chemical properties. *Mich. State Univ. Agr. Exp. Sta. Quart. Bull.* **1963**, *46*, 1–36.

Strontium

Silvia H. Haneklaus

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

The contamination of soils, plants, and water bodies with radionuclides is a major threat to human health. Contaminations with radioactive strontium (Sr) are particularly dangerous because of its similarity to calcium (Ca). Experiments on the behavior of radioactive Sr in the soil, its transfer to the food chain, and its interactions with Ca are limited. A solution to this problem is the analysis of stable Sr, which behaves identically to radioactive Sr. This contribution summarizes data on chemical speciation and spatial variability of Sr in soils, its relationship with relevant soil parameters, and factors influencing Sr uptake by plants. Special attention was paid to Ca, as it has to be put on par with Sr.

INTRODUCTION

Strontium (Sr) was discovered by Adair Crawford in 1790 and named after the Western Scottish village of Strontian. There are four stable Sr isotopes, namely, ^{84}Sr , ^{86}Sr , ^{87}Sr , and ^{88}Sr , which occur naturally in a ratio of 0.56:9.87:7.00:82.58, respectively.^[1] Of these four stable Sr isotopes, only ^{87}Sr is radiogenic. Thus, the ^{87}Sr concentration increases over time because of the decay of rubidium (^{87}Rb) to ^{87}Sr (half-life of 4.7×10^{10} yr). However, the radioactive isotope ^{90}Sr from anthropogenic nuclear sources is commonly associated with ^{87}Sr . Plant available concentrations of Sr and calcium (Ca) in soils are important factors that affect the transfer_{soil/plant} of ^{90}Sr .^[2] Consequently, the knowledge of the factors and processes influencing the chemical and spatial speciation of Sr in the soil is of prime interest in risk assessment of undesired ^{90}Sr contaminations.

Sr IN SOILS

Sr occurs abundantly in nature, and its concentration can be as much as 0.034% in most igneous rocks.^[3] Celestite (SrSO_4) and strontianite (SrCO_3) are the predominant Sr minerals. It occurs in large concentrations and in association with potassium in volcanic rocks, alkali rocks, and pegmatites. Sr is removed by weathering from these and other igneous rocks and sediments. Intermediate concentrations of Sr occur in basic magmatic rocks, which contain about two times higher Sr concentrations than granite. Some ultramafic rocks have very low Sr concentrations.^[4] A mean Sr concentration of 375 mg kg^{-1} is observed in igneous rocks, 300 mg kg^{-1} in shales, and 20 mg kg^{-1} in sandstones.^[5]

Sr isotope ratios in parent materials and minerals vary in relation to geological age and geographical location.^[6]

The $^{87}\text{Sr}/^{86}\text{Sr}$ ratio depends on the initial Rb/Sr ratio in the parent material, the age of rock, and the Sr isotope ratio at the onset of rock formation, as ^{87}Sr is radiogenic.^[6] The $^{87}\text{Sr}/^{86}\text{Sr}$ ratio is often used to study biogeochemical processes such as weathering of minerals. However, this ratio must be evaluated critically, as with the age of weathering, surface changes in the $^{87}\text{Sr}/^{86}\text{Sr}$ ratio also occur, presumably because of dynamic variations in weathering rates of different minerals.^[7]

In the soil profile, about 90% of the Sr is generally exchangeable and 10% in non-exchangeable form.^[8] The data in Table 1 summarize total and plant available Sr concentrations in soils of different origins. Reimann et al.^[4] reported distinctly lower total Sr concentrations in soils of northern Europe with a median of 110 mg kg^{-1} Sr in comparison with the world average of 240 mg kg^{-1} Sr. In several countries of northern Europe, median values for total Sr concentrations range from 45 mg kg^{-1} in Poland to 181 mg kg^{-1} in Norway. About 7% of the total Sr concentration in the topsoil is plant available.^[4] The reported plant available Sr concentrations range from 4 mg kg^{-1} in Poland to 15 mg kg^{-1} in Finland.^[4] The corresponding values for Germany are 8 ($\text{NH}_4\text{-Ac}$) and 55 (total) mg kg^{-1} Sr, and somewhat lower than those reported by Haneklaus^[2] in Schleswig-Holstein with 18 ($\text{NH}_4\text{-Ac}$) and 79 (total) mg kg^{-1} Sr (Table 1). The mean rapidly exchangeable Sr concentration in these studies was 4.8 mg kg^{-1} Sr.

Interaction of Sr with Soil Matrix

The advantage of using stable Sr in research is that it provides a direct measure for the spatial variation of pedogenetic soil characteristics and agro-technical measures influencing the uptake of radioactive Sr isotopes by plants. In comparison, studies with radioactive Sr (viz. ^{90}Sr , ^{89}Sr ,

Table 1 Variation in the Sr content in dependence on chemical speciation and geogenic origin.

Fraction	Country	Method	Min. ($\mu\text{g g}^{-1}$)	Max.	Mean	n	References
Sr _{total}	N. Germany ^a	X-RF	15.2	118.4	79.0	155	[2]
	N. Germany ^b	X-RF	57.0	97.0	71.3	154	[9] ^c
	Finland		109	2730	744	221	[10]
	N. Europe ^d	X-RF	10	667	110 ^e	773	[4]
	Scotland ^f	—	60	1000	380	43	[11]
	Russia	—	53	135	92	—	[12]
	India	HF-HNO ₃ -HClO ₄	74	198	127	10	[13]
Sr _{total exchangeable}	United States	—	80	900	400	13	[11]
	N. Germany ^a	1N NH ₄ -Ac, pH 7	4.4	48.9	18.1	155	[2]
	Finland	NH ₄ -Ac, pH 4.65	9.4	16.6	12.9	1944	[14] ^g
	Finland	NH ₄ -Ac, pH 4.65	2.4	63.8	19.2	221	[10]
	N. Europe	NH ₄ -Ac, pH 4.5	0.04	473	8 ^e	773	[4]
Sr _{rapidly exchangeable}	Scotland ^h	1N NH ₄ -Ac, pH 7	3.5	17.5	6.6	7	[11]
	N. Germany ^a	0.025 N CaCl ₂	1.5	10.2	4.8	155	[2]

^aSchleswig-Holstein.
^bIsle of Rügen.
^cSample description (Sr data unpublished).
^dBelarus, Estonia, Finland, Germany, Latvia, Lithuania, Norway, Poland, Russia, and Sweden.
^eMedian.
^fNortheast Scotland.
^gmg L⁻¹.
^hAberdeenshire/Midlothian.

and ⁸⁵Sr) provide *in situ* analysis of radioactive Sr behavior under existing soil conditions. An important point to be noted in Sr experimentation is whether the stable or carrier-free radioactive Sr has been used, which leads to the application of significant amounts of Sr in the former case, compared with only insignificant rates of Sr added to the soil in the latter case.

The correlation matrix between soil pH, clay, organic C, plant available Ca concentration, and Sr species is shown in Fig. 1 for soils from the federal state of Schleswig-Holstein.^[2] Correlation coefficients between total exchangeable Sr and C_{org.} were significant but low. In comparison, distinctly close and significant relationships were observed for soil pH and clay content (Fig. 1). Reimann et al.^[4] and Haneklaus^[2] also reported a similar correlation coefficient of $r = 0.6$ for the relationship between total Sr and NH₄-Ac extractable Sr when total Sr was determined by aqua regia extraction and X-ray fluorescence (X-RF) techniques, respectively. Analogous correlation coefficients were also observed for the relationship between plant available Ca ($r = 0.78$) and plant available Sr ($r = 0.90$) concentrations in the topsoil (Fig. 1).^[2,4]

Rengel estimated a mean Ca loss of 300 kg ha⁻¹ yr⁻¹ by leaching. The Ca/Sr ratio in soils is about 140 for total^[15] and may vary between 250 and 5000 for exchangeable Ca and Sr.^[16] Assuming equivalent leaching rates of Ca and Sr implies Sr losses of <1 kg ha⁻¹ yr⁻¹. Wiklander^[8] observed considerable leaching losses of ⁹⁰Sr in a lysimeter experiment after 42 months.

Sr sorption is dominated by ion exchange, and sorption/desorption is a reversible process; the organic soil fraction preferentially adsorbs Ca, while the inorganic soil fraction (e.g., the clay minerals) preferentially adsorbs Sr.^[17,18]

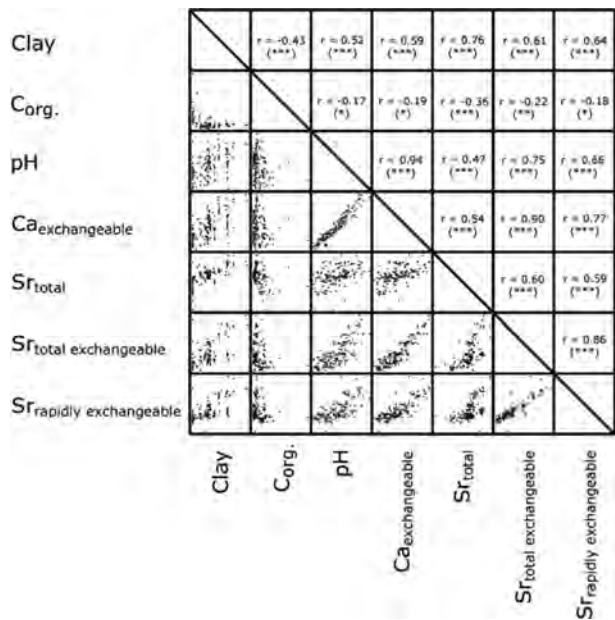


Fig. 1 Correlation between soil parameters and Sr species in topsoil samples of production fields in northern Germany (n=155).
Source: Calculated from data by Haneklaus.^[2]

A small fraction of ^{85}Sr added to the soil is also fixed in non-exchangeable form. Valcke^[17] suggested that this fixation occurs in Sr/organic matter/clay complexes.

The adsorption of Sr on different types of clay minerals has been studied intensively with the objective to decontaminate effluents from contaminated sites. Sr is adsorbed moderately to mineral surfaces and is desorbed when its concentration in the soil solution increases, competing ions such as Ca are present, and soil pH is low.^[19] The application of even small quantities of stable Sr and Ca as amendments may increase the uptake of ^{90}Sr by desorption. However, the application of Ca may positively increase the Ca/Sr ratio.^[20] The Sr sorption decreases in the presence of calcite, and high concentrations of bicarbonate and SrCO_3 may be formed at high concentrations of Sr in the soil.^[19] For a Cambrian blue clay soil, the Langmuir isotherm indicated a constant of 294 and a limiting sorption of $0.034 \text{ mol kg}^{-1}$.^[18] Also Freundlich isotherms reflected an adequate sorption of Sr with a regression coefficient of >0.99 . The sorption ratio is generally higher than 0.95 for all concentrations ranging from 10^{-3} to 10^{-8} M .^[21]

Sr UPTAKE BY PLANTS

Sr uptake by plants is passive and is thus linked to its concentration in the soil solution. Plant available Ca and Sr in the soil are important factors governing the uptake of radioactive Sr. The addition of Ca often results in a negative linear relationship between Sr and Ca in plants.^[2] The Sr to cation exchange capacity (CEC) ratio influences Sr uptake in such a way that any increase in the CEC decreases the Sr uptake.

The Sr concentration in plants varies depending on the plant available concentrations in the soil, root uptake, and the growth or phenological stage. Sr uptake by dicotyledonous crops is, e.g., distinctly higher than that by monocotyledonous crops owing to the higher CEC of the latter.^[20] The Sr concentration in vegetative plant tissue varies between 1 and 150 mg kg^{-1} with a mean value of

25 mg kg^{-1} .^[3] The Sr concentration in seeds is about 5 to 10 times lower than that in vegetative plant material, most likely because of the phloem immobility.^[20] Sr concentrations from 18 to 36 mg kg^{-1} have been reported in oilseed rape seeds, while the average concentration was 7 mg kg^{-1} in wheat.^[2]

Impact of Fertilizer Use on Sr Uptake by Plants

Gabe and Rodella^[22] reported Sr concentrations in different phosphatic fertilizers, limestone, and gypsum. The mean Sr concentration (mg kg^{-1}) occurs in the following order: concentrated apatite (12,121) > rock phosphates (7319) > single super phosphate (5396) > fused phosphates (4965) > gypsum (2984) > limestone (824) > ammonium phosphates (304). The variation in the Sr concentration was the highest in limestone samples ranging from 36 to 4736 mg kg^{-1} Sr. Long-term application of lime significantly increased soil pH along with total Ca and Sr concentrations in an acid Albic Luvisol in Romania (Fig. 2).^[23] Haneklaus^[2] reported that the Sr uptake in plants is influenced more by the Ca concentration in soil rather than by soil pH. Liming of an acid soil can reduce the Sr uptake by 40–45%, as was reported in a lysimeter experiment by Wiklander.^[8] Under field conditions, Haneklaus^[2] estimated a minimum threshold rate of lime application of 1.4 t ha^{-1} CaO for reducing the Sr uptake significantly on an acid, silty loam soil (pH 5.3). Liming combined with an additional Ca supply by gypsum can cause the maximum reduction in Sr uptake by oilseed rape probably because of the precipitation of SrSO_4 .^[2] In contrast, Andersen^[20] reported that a dose of at least 10 t ha^{-1} stable Sr is required to achieve a significant reduction of ^{90}Sr uptake.

CONCLUSION

Sr behavior in soils and uptake by plants are similar to that of Ca. Stable Sr is usually of minor interest, but it may gain

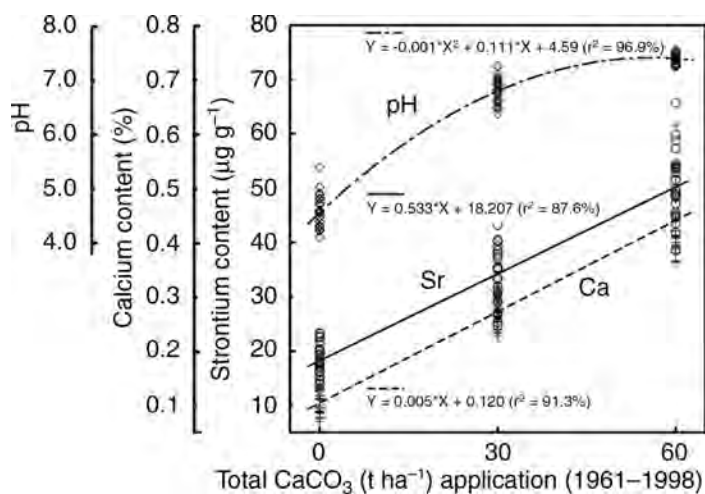


Fig. 2 Long-term influence of regular lime applications of 0, 5, and 10 t ha^{-1} from 1961 to 1998 corresponding to total rates of 0, 30, and 60 t ha^{-1} on soil pH and total Sr and Ca contents in the topsoil of an Albic Luvisol in Romania.

Source: Adapted from Rogasik, Kurtinez, et al.^[23]

relevance with its role in reducing the transfer of radioactive Sr into the food chain. An optimum Ca saturation is a prerequisite to limit the uptake of radioactive Sr by plants. Furthermore, amendments (e.g., lime) add significant amounts of Sr to the soil and thus increase the concentration of competitive stable Sr in soils.

REFERENCES

1. Weast, R.C.; Astle, M.J.; Beyer, W.H. *Handbook of Chemistry and Physics*; CRC Press Inc.: Boca Raton, 1988–1989.
2. Haneklaus, S. Strontiumgehalte in Pflanzen und Böden Schleswig-Holsteins und Bewertung von Düngungsmaßnahmen zur Verminderung der Strontiumaufnahme von Kulturpflanzen, 1989. PhD Thesis, University of Kiel, Kiel; 169 pp.
3. Coughtrey, P.J.; Thorne, M.C. *Radionuclide Distribution and Transport in Terrestrial and Aquatic Ecosystems. A Critical Review of Data*; Balkema: Rotterdam, 1983; Vol. 1, 93–239.
4. Reimann, C.; Siewers, U.; Tarvainen, T.; Bityukova, L.; Eriksson, J.; Gilucis, A.; Gregorauskiene, V.; Lukashev, V.K.; Matinian, N.N.; Pasieczna, A. *Agricultural Soils in Northern Europe: A Geochemical Atlas*. In Bundesanstalt fuer Geowissenschaften und Rohstoffe and Staatliche Geologische Dienste in der Bundesrepublik Deutschland, Eds.; Schweizerbart'sche Verlagsbuchhandlung: Stuttgart, 2003; 279 pp.
5. Lisk, D.J. Trace metals in soils, plants, and animals. *Adv. Agronom.* **1972**, *24*, 267–311.
6. Barbaste, M.; Robinson, K.; Guilfoyle, S.; Medina, B.B.; Lobinski, R. Precise determination of the strontium isotope ratios in wine by inductively coupled plasma sector field multicollector mass spectrometry (ICP-SF-MC-MS). *J. Anal. Atom. Spectrom.* **2002**, *17*, 135–137.
7. Ma, Y.; Liu, C. Sr isotope evolution during chemical weathering of granites. *Sci. China* **2001**, *44*, 726–734.
8. Wiklander, L. Uptake, adsorption and leaching of radiostrontium in a lysimeter experiment. *Soil Sci.* **1964**, *97*, 168–172.
9. Haneklaus, S.; Fleckenstein, J.; Schnug, E. Comparative studies of plant and soil analysis for the evaluation of the sulphur status of oilseed rape and wheat. *J. Plant Nutr. Soil Sci.* **1995**, *158*, 109–112.
10. Lakanen, E.; Silanpää, M. Strontium in Finnish soils. *Ann. Agric. Fenn.* **1967**, *6*, 197–207.
11. Swaine, D.J. The trace element content of soils. *Commonw. Bur. Soil Sci. Tech. Commun.* **1955**, *48*, 157.
12. Protasova, N.A.; Golubev, I.M.; Korobeinikov, N.I. Trace elements in landscapes of the Tambov Oblast and the biogeochemical zoning of its territory. *Eurasian Soil Sci.* **1996**, *29*, 1360–1366.
13. Moslehuddin, A.Z.M.; Laizoo, S.; Egashira, K. Trace elements in Bangladesh paddy soils. *Commun. Soil Sci. Plant Anal.* **1999**, *30*, 1975–1996.
14. Paasikallio, A. Strontium content and strontium-calcium ratio in timothy (*Phleum pratense* L.) and soil in Finland. *Ann. Agric. Fenn.* **1979**, *18*, 174–181.
15. Comar, C.L. Some over-all aspects of strontium-calcium discrimination. In *The Transfer of Calcium and Strontium across Biological Membranes*; Wassermann, R.H., Ed.; Academic Press: New York, 1963; 405–417.
16. Menzel, R.G.; Heald, W.R. Strontium and calcium contents of crop plants in relation to exchangeable strontium and calcium of the soil. *Soil Sci. Soc. Am. Proc.* **1959**, *23*, 110–112.
17. Valcke, E. The behaviour dynamics of radiocesium and radiostrontium in soils rich in organic matter, 1993. PhD Thesis, Katholieke Universiteit Leuven, Belgium; 135 pp.
18. Chirkst, D.E.; Litinova, T.E.; Cheremisina, O.V.; Ivanov, M.V.; Mironenkova, N.A. Isotherm of strontium sorption on clay. *Russ. J. Appl. Chem.* **2003**, *76*, 727–730.
19. Mincher, B.J.; Fox, R.V.; Riddle, C.L.; Cooper, D.C.; Groenewold, G.S. Strontium and cesium sorption to Snake River Plain, Idaho soil. *Radiochim. Acta* **2004**, *92*, 55–61.
20. Andersen, A.J. *Plant Accumulation of Radioactive Strontium with Special Reference to the Strontium-Calcium Relationship as Influenced by Nitrogen*, Riso Report 278; Forskningscenter Risoe: Denmark, 1973; 56.
21. Hakem, N.L.; Al Mahamid, I.; Apps, J.A.; Moridis, G.J. Sorption of cesium and strontium on Hanford soil. *J. Radioanal. Nucl. Chem.* **2000**, *246*, 275–278.
22. Gabe, U.; Rodella, A.A. Trace elements in Brazilian agricultural limestones and mineral fertilizers. *Commun. Soil Sci. Plant Anal.* **1999**, *30*, 605–620.
23. Rogasik, J.; Kurtinez, P.; Panten, K.; Funder, U.; Rogasik, H.; Schroetter, S.; Schnug, E. Kalkung und Bodenfruchtbarkeit. *FAL Agric. Res.* **2005**, *286* (special issue), 71–81.

Structure

Eileen J. Klavivko

Agronomy Department, Purdue University, West Lafayette, Indiana, U.S.A.

Abstract

Soil structure is an important characteristic of the soil. There is no one single definition of soil structure, but most descriptions of structure refer to the “arrangement” (size, shape, and orientation) of particles and the pores between them, or the “stability” of the particle arrangement to some disruptive force (e.g., hand manipulation, water, wind, and wheel traffic). Soil structure affects many properties and processes in the soil.

INTRODUCTION

Soil aggregates are the backbone of soil structure. Aggregates are clusters of organic matter, sand, silt, and clay held together by a variety of forces. A hierarchical order of aggregates is sometimes described with smaller groupings of particles comprising microaggregates and then groupings of microaggregates forming macroaggregates.^[1,2] Pores within aggregates are smaller and are important for water retention, whereas pores between aggregates are larger and are important for water flow and aeration.

DESCRIPTION OF STRUCTURE

The descriptions of soil structure are often approached from the viewpoint of pedology, plant growth, or physics. The pedological view describes the shape, size, and relative strength of soil peds or aggregates. These descriptions reflect long-term soil forming processes as well as shorter-term human-induced changes caused by soil management practices.

From an agronomic point of view, “good” soil structure promotes rapid water infiltration, adequate drainage of excess water, good aeration, water retention, resistance to erosion, and good germination, emergence, and rooting of plants.^[3] “Poor” soil structure may result in surface crusting, decreased infiltration, increased runoff and erosion, poor seedling emergence, poor rooting ability, or inadequate air or water for plants. Much of the soil management research conducted over the decades has been to devise ways to improve soil structure while enhancing cash crop yields.

The soil physics view of soil structure is often to assess the stability of aggregates to disruptive forces that simulate disruptive forces in the field.^[2] Size distribution and stability of aggregates after disruption are used to calculate quantitative indices of stability. Other ways to describe structure include pore size distribution and continuity. The particular

method used to assess or describe soil structure should be matched with the intended use of the information.

AGGREGATE FORMATION

Soil aggregates are formed through biological, chemical, and physical processes. As soil microorganisms decompose plant residues and other organic materials, they exude polysaccharides that tend to glue particles together. Fungal hyphae also enmesh particles, thereby forming aggregates. Earthworms help to aggregate soil through excretion of mucilages, which stick particles together, as well as bringing mineral and organic particles together in their castings. Plant roots also help to aggregate soil through several mechanisms, including root exudation and physical enmeshment by fibrous roots. Chemical conditions that impact aggregation include the concentrations and types of cations present in the soil solution, as they affect flocculation of the clay particles. Physical processes affecting aggregation include wetting/drying cycles, freeze/thaw cycles, and mechanical tillage. These physical processes break down large clods and destroy aggregates, but they also bind particles together to form small aggregates. The net impact of these physical processes on formation and destruction of aggregation is highly dependent on initial soil conditions.

An important point to remember is that soil aggregation is dynamic.^[2,3] Aggregates are not permanent but are continuously being created and destroyed by natural means as well as by human management of the soil. Soil management practices must include regular “maintenance” of soil structure to sustain productivity over the long term. For example, soils maintained as native grasslands generally have higher organic matter contents and greater aggregate stability than similar soils used for row crop production.^[4] When soils are used for crop production, sustainable management practices often include periodic inputs of organic materials (animal manures and crop residues) to the soil to

provide a food source for the soil organisms which in turn form aggregates. The relative stability of different organic binding agents in soil micro- and macroaggregates is discussed in more detail in other sections of this encyclopedia. Growing a fibrous-rooting plant as a cover crop after the main crop is harvested is a way to use the normally fallow time periods of the year to improve near-surface structure. Conservation tillage practices, especially no-tillage, are another way to improve soil aggregation and stability compared with conventional tillage practices.^[5] The increased aggregation results from both greater aggregate formation near the surface, because of the presence of surface crop residues, and less physical disruption of aggregates by the tillage operation itself.

ENVIRONMENTAL SIGNIFICANCE OF SOIL STRUCTURE

Stable soil aggregates contain organic binding agents, as discussed earlier, and therefore are involved in carbon sequestration in soils. Within a given soil, increased soil aggregation is usually associated with increased soil carbon. Conversely, as aggregates are disrupted or destroyed, carbon mineralization and carbon dioxide evolution increase.^[6] Therefore, increasing soil aggregation and aggregate stability can be an important method to increase carbon sequestration in soil. It must be remembered, however, that these higher aggregation levels must be continuously maintained, as previously discussed, or else the sequestered carbon will be released again. Soil organic matter associated with different aggregate size fractions appears to have different turnover times, and the interplay of organic matter chemistry, aggregate size, and physical location within or between aggregates is active areas of research.

Although aggregates are naturally degraded with time as microorganisms feed on the organic binding materials, these are more quickly and noticeably destroyed by tillage operations. Aggregates near the soil surface are also subjected to the disruptive impact of raindrop energy or rapid wetting, which can result in slaking, dispersion, and crust formation. Such surface structural breakdown can have major impacts on plant growth and overall soil functioning. Surface crusts can delay or inhibit seedling emergence, leading to low plant populations and yields. Surface seals can also reduce infiltration and increase runoff and erosion. The partitioning of less water into the soil profile may reduce water availability to the plant at a later date. The structure of soil below the surface may be degraded by tillage or wheel traffic, with potential consequences of poor rooting ability because of compaction, poor drainage and aeration of the rooting zone, and reduced water availability to the plant.

The structure of surface and subsurface horizons affects water flow into and through soil profiles and across landscapes. The partitioning of water between infiltration and surface runoff is controlled largely by near-surface conditions, although subsurface impeding layers may limit infiltration as well. Impeding layers may occur naturally or be formed as a result of management. In fields with a good subsurface drainage system, e.g., water collects temporarily in depressions after a rain. If these depressions are tilled when the soil is too wet, near-surface soil horizons become compacted and impermeable. After the next rain, water stands in these depressions even longer because of the compacted soil. Many soils have sufficient structural development below the surface to exhibit some degree of preferential flow of water, whereby water flows quite rapidly through large, interpedal voids while moving more slowly into and through smaller pores within the aggregates.

CONCLUSION

In summary, it is well known that soil structure has a major influence on the ability of soil to perform different functions. Whether the soil is used to grow plants, accept wastes, or partition rainfall into runoff, or for interflow, groundwater recharge, or build houses and roads, the arrangement of particles and the stability of that arrangement are important factors to consider. Research attempts to more quantitatively describe the *in situ* structure and the response of soil to various disruptive forces, as well as the relationships between soil structure, plant growth, and water flow.

REFERENCES

1. Tisdall, J.M.; Oades, J.M. Organic matter and water-stable aggregates in soils. *J. Soil Sci.* **1982**, *33*, 141–163.
2. Hillel, D. *Fundamentals of Soil Physics*; Academic Press: San Diego, 1980; 413 pp.
3. Karlen, D.L.; Erbach, D.C.; Kaspar, T.C.; Colvin, T.S.; Berry, E.C.; Timmons, D.R. Soil tillage: A review of past perceptions and future needs. *Soil Sci. Soc. Am. J.* **1990**, *54*, 153–161.
4. Six, J.; Elliott, E.T.; Paustian, K.; Doran, J.W. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Sci. Soc. Am. J.* **1998**, *62*, 1367–1377.
5. Klavivko, E.J. Residue effects on soil physical properties. In *Managing Agricultural Residues*; Unger, P.W., Ed.; Lewis Publishers: Boca Raton, 1994; 123–141.
6. Elliott, E.T. Aggregate structure and carbon, nitrogen and phosphorus in native and cultivated soils. *Soil Sci. Soc. Am. J.* **1986**, *50*, 627–633.

Structure: Belowground Management

Ken A. Olsson

Tatura, Victoria, Australia

Bruce Cockroft

Research Consultant, Kialla, Victoria, Australia

Abstract

Researchers have long sought to improve the structure of difficult subsoils by deep tillage, chemical ameliorants, and ameliorative crops, aiming for higher yields through faster infiltration of water, deeper wetting, increased water and nutrient availability to plants, increased drainage, leaching of salts and toxins, lower soil strength, and deeper rooting. Subsoil modification, however, has not consistently increased crop yields, while benefits are often transient and confounded. Many ameliorative measures do not modify the subsoil to a particular specification, while most do not adequately address the stabilization and subsequent management of the modified soil.

INTRODUCTION

Crop yield is strongly correlated with the production and functioning of roots. Soil management aims to produce and maintain conditions in the soil so that roots are provided with low mechanical resistance, non-limiting supplies of water, oxygen, and nutrients, and favorable soil temperature.^[1] Root growth and functioning are promoted by soil structure of low strength and of high porosity that facilitates the necessary transport of water, nutrients, and gases through stable pores.^[2] While specifications for the necessary levels of soil strength and porosity are well documented,^[1,3] they are not generally met in poorly structured subsoils underlying the surface (A) horizon(s). These include hardpans, claypans, and compacted layers of high strength and low porosity. Such layers limit yield by physically restricting root proliferation, drainage, and the storage and transport of water, while making the management of otherwise well-structured surface horizons both difficult and inflexible.^[4,5]

Researchers have long sought to improve the structure of difficult subsoils by deep tillage, chemical ameliorants, and ameliorative crops, aiming for higher yields through faster infiltration of water, deeper wetting, increased water and nutrient availability to plants, increased drainage, leaching of salts and toxins, lower soil strength, and deeper rooting.^[6] Subsoil modification, however, has not consistently increased crop yields, while benefits are often transient and confounded.^[7–9] Many ameliorative measures do not modify the subsoil to a particular specification, while most do not adequately address the stabilization and subsequent management of the modified soil.

IMPROVING AND MANAGING SUBSOIL STRUCTURE

Principles guiding the improvement and management of subsoil structure for the growth and functioning of roots are documented in the literature.^[10] An initial survey of the soil profile provides information on soil physical and chemical properties including soil horizons, textures, and clay mineralogy. Any chemical problems such as salinity, sodicity, and acidity are corrected and nutrients are supplied at non-limiting rates. When soil temperature is not the limitation, the main soil physical factors that directly influence root growth and functioning are soil strength, soil water supply, and aeration. Specifications for each that is non-limiting for roots are given in the literature.^[1] Management to reduce soil strength and to increase porosity is then based on meeting each of these in a defined volume of soil within the root zone.

Soil Strength

For unrestricted elongation, roots require soil strength, measured as penetrometer resistance, of less than 0.5 MPa.^[11] However, subsoils and compacted layers may have penetrometer resistances greater than 2 MPa, even when wet. In dense clay subsoils, strength increases rapidly on drying and roots may grow only during brief periods and over a small range of matric suction. This results in shallow, confined root systems, and low root concentrations in the matrix.^[10] In expansive clays, roots tend to be restricted to cracks.

Soil strength is reduced by loosening the clay and fragmenting hardpans^[4] while producing macropores (>30 μm diameter). The resulting fragments and aggregates assist root proliferation.^[10] Loosening the soil to about 0.5 m depth is considered a practical compromise in reducing the

effects of dense soil,^[1,3] but the depth of modification will depend on the position of restricting layers within the root zone. Ripping is effective in brittle soils, hardpans, and compact sandy layers; it is less effective in loams and clays. Loosening of clay subsoils is best performed using rigid tines working in soil slightly drier than the plastic limit, aiming to achieve brittle failure.^[12] The tines fail the soil upward by means of low angle tips operating at modest increments of depth and at low speed to avoid high confining stresses at the points. Wings behind each point shear and fracture many of the fragments under the confining pressure from the overburden. Deep tillage may be combined with additions of specified quantities of ameliorants, such as gypsum and lime, to help stabilize the loosened soil.^[10]

An alternative technique mixes the soil layers and incorporates ameliorants, crop residues, and fertilizers in parallel slots, 100–150 mm wide at 1–2 m spacing tilled to the required depth.^[13]

The benefits to crops from changes in soil structure brought about by the activity of roots of previous crop and pasture species are limited in the context. Crop responses are often confounded with improved nutrition and drainage of the surface soil and with reduced incidence of disease.^[14]

The loosened subsoil must be protected from compaction by traffic and from rapid wetting. An effective approach to the problem of soil compaction is to separate roots from traffic using beds.^[15] Management for stability in water and mechanical stability of the loosened subsoil also includes slow wetting and rapid drainage to avoid very low matric suctions to ensure an effective stress within aggregates and fragments.

Loosened subsoils under no traffic inevitably harden with time even though much of the improved macroporosity may be retained.^[16] While roots may grow in the macropores formed by tillage, or by the roots of earlier crops, they may find it difficult to penetrate the hard fragments that are >20 mm diameter where these retain the high strength of the original soil.^[17] A consequent slowing of root growth may be expressed physiologically as a slowing of the shoots, even where supplies of water, oxygen, and nutrients are adequate.^[18] Research is needed to find ways to develop soft, porous aggregates that are stable both in water and to applied mechanical stresses. The process involves the penetration of unconfined fragments by roots and fungal hyphae and mellowing by successive cycles of wetting and drying.^[10] Stabilization requires additions of organic matter from roots, added calcium and electrolyte, and a moist, well-drained environment.

Soil Water

In temperate crops, yield shows a close to linear increase with transpiration.^[19,20] Poorly structured subsoils of low macroporosity slow the infiltration and penetration of water. Subsoils may show a predominance of fine pores (<30 μm diameter), which restrict both the storage and flow of water

to crop roots. Storage pores (0.2–30 μm diameter) in subsoils commonly comprise <15% of the soil volume.^[21,22] Water availability from the subsoil is further reduced where roots have difficulty penetrating the strong matrix. Where topsoils are shallow, potential transpiration rates may be sustained for only a few days after irrigation.^[23]

For rapid water entry, free drainage, and gas exchange, the soil must be stable in water and maintain >15% macropores.^[15] Initially, the fragments produced by tillage may be little altered from the original and limit water storage, gas exchange, and water flow to roots, while mechanically restricting root entry. Should roots become largely confined to and clumped in macropores, they may be poorly functional in water uptake as theory predicts a major resistance to water flow in the soil.^[24,25] Management should aim to develop soft, porous aggregates to enable improved hydraulic properties and a more even distribution of roots. Ideally, these aggregates should have >20% storage pores.^[26]

Soil Aeration

The diffusion of oxygen in air-filled pores continuous to the soil surface is necessary for aeration of the subsoil. Dense, slowly permeable subsoils, however, have few interconnected macropores over depth and may restrict oxygen supply to the root, particularly when the clay is swollen.^[27]

The poor aeration of wet clays slows the growth and functioning of roots^[28] and may also increase the incidence of soil-borne disease.^[29]

Management should aim for a stable air-filled porosity of >15% of the soil volume to be reached within 24-hour drainage throughout the modified solum.^[1] Roots require at least 10% oxygen for unrestricted growth.^[17] Surface drainage is essential. Deep tillage then produces the necessary macroporosity to depth. In some cases, the restructured subsoil in the root zone may connect hydraulically with a more permeable layer beneath.^[1] In other situations, artificial drainage using a pipe or mole drain is required.^[30] In irrigated soils, a slow application rate minimizes filling the macropores with water.

Where fragments and aggregates are not larger than about 10 mm or smaller than 1 mm,^[31] and where the oxygen diffusion rate is higher than about 35 $\mu\text{g}/\text{m}^2/\text{s}$,^[32] intraaggregate aeration is unlikely to be a limiting factor for root growth. An increase in root and microbial respiration in fragments undergoing structural change as described earlier should be met by the increased porosity within the fragment.

CONCLUSION

The management of difficult subsoils to achieve and sustain high productivity from crops must provide the necessary low soil strength together with complements of macropores and storage pores, which will not limit the transfer of water and gases, the proliferation of roots, or the supply of water to crops. Mechanical loosening of dense subsoil layers is an

initial step in an ongoing process of development of the prescribed levels of soil strength and porosity. The process is facilitated by chemical ameliorants, the high initial porosity of the loosened soil, high levels of biological activity, and closely controlled wetting under non-trafficked conditions. A quantitative understanding of root and top responses to profile modification in relation to plant production is an important area for research.

REFERENCES

- Cockroft, B.; Olsson, K.A. Case study of soil quality in south-eastern Australia: Management of structure for roots in duplex soils. In *Soil Quality for Crop Production and Ecosystem Health*; Gregorich, E.G., Carter, M.R., Eds.; Elsevier: Amsterdam, 1997; 339–350.
- Letey, J. The study of soil structure: Science or art. *Aust. J. Soil Res.* **1991**, *29*, 699–707.
- Greenland, D.J. Soil management and soil degradation. *J. Soil Sci.* **1981**, *32*, 301–322.
- Wildman, W.E.; Meyer, J.L.; Neja, R.A. *Managing and Modifying Problem Soils, Leaflet 2791*; University of California, Division of Agricultural Sciences: Berkeley, 1979.
- Chartres, C.J.; Cresswell, H.V.; Murphy, B.W.; Greeves, G.W.; Little, I.P.; Gessler, P.E. Distribution of physical and chemical subsoil constraints and their prediction in landscapes. In *National Workshop on Subsoil Constraints to Root Growth and High Soil Water and Nutrient Use by Plants*, Proceedings of National Workshop, Tanunda, South Australia, Aug 30–Sept 2, 1992.
- Unger, P.W. Effects of deep tillage and profile modification on soil properties, root growth and crop yields in the United States and Canada. *Geoderma* **1979**, *22*, 275–295.
- Eck, H.V.; Unger, P.W. Soil profile modification for increasing crop production. *Adv. Soil Sci.* **1985**, *1*, 65–100.
- Soane, G.C.; Godwin, R.J.; Marks, M.J.; Spoor, G. Crop and soil responses to subsoil loosening, deep incorporation of phosphorus and potassium fertilizer and subsequent soil management on a range of soil types. Part 2. Soil structural conditions. *Soil Use Manage.* **1987**, *3*, 123–130.
- Sojka, R.E.; Horne, D.J.; Ross, C.W.; Baker, C.J. Subsoiling and surface tillage effects on soil physical properties and forage oat stand and yield. *Soil Till. Res.* **1997**, *40*, 125–144.
- Olsson, K.A.; Cockroft, B.; Rengasamy, P. Improving and managing subsoil structure for high productivity from temperate crops on beds. In *Subsoil Management Techniques*; Jayawardane, N.S., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 1995; 35–65.
- Cockroft, B.; Barley, K.P.; Greacen, E.L. The penetration of clays by fine probes and root tips. *Aust. J. Soil Res.* **1969**, *7*, 333–348.
- Spoor, G.; Godwin, R.J. An experimental investigation into the deep loosening of soil with rigid tines. *J. Agr. Engng. Res.* **1978**, *23*, 243–258.
- Jayawardane, N.S.; Blackwell, J.; Kirchhof, G.; Muirhead, W.A. Slotting—A deep tillage technique for ameliorating sodic, acid and other degraded subsoils and for land treatment of waste. In *Subsoil Management Techniques*; Jayawardane, N.S., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, 1995; 109–146.
- Cresswell, H.P.; Kirkegaard, J.A. Subsoil amelioration by plant roots—The process and the evidence. *Aust. J. Soil Res.* **1995**, *33*, 221–239.
- Cockroft, B.; Tisdall, J.M. Soil management, structure and root activity. In *Modification of Soil Structure*; Emerson, W.W., Bond, R.D., Dexter, A.R., Eds.; Wiley: Chichester, 1978; 387–391.
- Tisdall, J.M.; Olsson, K.A.; Willoughby, P. Soil structural management and production in a non-cultivated peach orchard. *Soil Till. Res.* **1984**, *4*, 165–174.
- Dexter, A.R. Advances in characterization of soil structure. *Soil Till. Res.* **1988**, *11*, 199–238.
- Masle, J.; Passioura, J.B. The effect of soil strength on the growth of young wheat plants. *Aust. J. Plant Physiol.* **1987**, *14*, 643–656.
- Penman, H.L. Water as a factor in productivity. In *Potential Crop Productivity*; Wareing, P.F., Cooper, J.P., Eds.; Heinemann: London, 1971; 89–99.
- Taylor, H.M.; Jordon, W.R.; Sinclair, T.R. *Limitations to Efficient Water Use in Crop Production*; American Society of Agronomy; CSSA; SSSA: Madison, 1983.
- Olsson, K.A.; Rose, C.W. Hydraulic properties of a red-brown earth determined from *in situ* measurements. *Aust. J. Soil Res.* **1978**, *16*, 169–180.
- Williams, J. Soil hydrology. In *Soils: An Australian Viewpoint*. The Division of Soils CSIRO, Ed.; CSIRO: Melbourne; Academic Press: London, 1983; 507–530.
- Olsson, K.A.; Rose, C.W. Patterns of water withdrawal beneath an irrigated peach orchard on a red-brown earth. *Irrig. Sci.* **1988**, *9*, 89–104.
- Passioura, J.B. Soil structure and plant growth. *Aust. J. Soil Res.* **1991**, *29*, 717–728.
- Tardieu, F.; Bruckler, L.; Lafolie, F. Root clumping may affect the root water potential and the resistance to soil-root water transport. *Plant Soil* **1992**, *140*, 291–301.
- Hall, D.G.M.; Reeve, M.J.; Thomasson, A.J.; Wright, V.F. *Water Retention, Porosity, and Density of Field Soils*. Soil Survey Technical Monograph, No. 9; Rothamsted Experiment Station: Harpenden, 1977.
- Greacen, E.L.; Gardner, E.A. Crop behaviour on clay soils. *Trop. Agric.* **1982**, *59* (2), 123–132.
- Cannell, R.Q.; Jackson, M.B. Alleviating aeration stresses. In *Modifying the Root Environment to Reduce Crop Stress*; Arkin, G.F., Taylor, H.M., Eds.; ASAE: Michigan, 1981; 141–192.
- Bergman, H.F. Oxygen deficiency as a cause of disease in plants. *Bot. Rev.* **1959**, *25*, 418–485.
- MacEwan, R.J.; Gardner, W.K.; Ellington, A.; Hopkins, D.G.; Bakker, A.C. Tile and mole drainage for control of waterlogging in duplex soils of south-eastern Australia. *Aust. J. Exper. Agric.* **1992**, *32*, 865–878.
- Braunack, M.V.; Dexter, A.R. Compaction of aggregate beds. In *Modification of Soil Structure*; Emerson, W.W., Bond, R.D., Dexter, A.R., Eds.; John Wiley & Son: Chichester, 1978; 119–126.
- Stolzy, L.H.; Letey, J.; Szuskiewicz, T.E.; Lunt, O.R. Root growth and diffusion rates as functions of oxygen concentrations. *Soil Sci. Soc. Am. Proc.* **1961**, *25*, 463–467.

Structure: Earthworms

Martin Shipitalo

*U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
Ames, Iowa, U.S.A.*

Tayfun Korucu

*Biosystems Engineering Department, Kahramanmaraş Sutcu Imam University,
Kahramanmaraş, Turkey*

Abstract

Earthworms are an important part of the soil ecosystem and an indicator of soil quality. Sometimes referred to as ecosystem engineers, they play a pivotal role in maintaining soil productivity. Their burrowing, feeding, and casting activities alter the physical, chemical, and biological properties of soil. As a result, earthworms can increase aeration, infiltration, aggregate stability, pH, porosity, and plant-available nutrients. Their abundance and species composition can be affected by soil properties such as pH, texture, water content, and temperature as well as management practices such as tillage, crop rotation and manure, fertilizer, and pesticide applications. The availability of crop residues and other organic materials, which serves as their food source, is usually the most important management factor affecting earthworm abundance. The estimated 3,000–7,000 known species of earthworms can be classified into three ecological groups based on feeding and burrowing habits: litter-dwelling epigeic species, shallow-burrowing endogeic species, and deep-burrowing anecic species. Burrows created by anecic species can play a critical role in facilitating preferential flow of water and agrochemicals through the soil profile. Endogeic and anecic species can accelerate mineralization of plant residues while simultaneously enhancing aggregation and stabilizing organic matter within soil aggregates. Consequently, their net impact on carbon sequestration is uncertain. Earthworm populations and their contributions to soil structure can be maximized by keeping tillage to a minimum, retaining or adding crop residues and manures to soil, and limiting the use of pesticides that are toxic to earthworms.

INTRODUCTION

Earthworms are almost universally regarded by farmers and gardeners as a sign of healthy soil. Entries in farmer-oriented publications frequently include testimonials of how changes in crop and soil management result in the return of earthworms to fields where they were previously absent. The publication of the Soil Biology Primer and the proliferation of websites devoted to earthworms are further recognition of the perceived role of earthworms and other soil fauna in maintaining healthy soils (Table 1). From a scientific viewpoint, however, it is uncertain how much earthworms contribute to soil quality or if they are a consequence or cause of good soil health. Obviously, there are healthy soils that don't have earthworms.

Earthworms affect a number of soil processes including nutrient and carbon cycling, plant growth, and the activity and distribution of microorganisms and are frequently referred to as ecosystem engineers. Perhaps the most noticeable impact of earthworms, however, is their effect on soil structure. Earthworms burrow into and ingest soil and in doing so modify soil porosity, aggregate size, and aggregate stability. The amount of soil ingested is highly dependent on the size, composition, and activity of the earthworm population and is hard to accurately measure because belowground activity is difficult to monitor. Nevertheless, estimated ingestion rates for temperate

region soils are as high as several hundred megagrams per hectare per year. In tropical areas, where climatic conditions are less likely to inhibit activity, ingestion rates as high as 2,600 Mg ha⁻¹ yr⁻¹ have been reported. Similarly, the contributions of earthworms to soil porosity and aggregation, the benefits of earthworm-enhanced soil structure to plant growth, and effects on water quality are difficult to quantify.

TYPES OF EARTHWORMS

Part of the problem in determining the effects of earthworms on soil structure comes from incomplete knowledge of their behavior. Worldwide there are an estimated 3,000 – 7,000 + species of earthworms,^[1] few of which have been investigated in detail. A number of classification schemes have been proposed, which group these species based on various aspects of their behavior. The most widely used system is that of Bouché in which earthworms are divided into three groups.^[1,2] *Epigeic* earthworms are generally found beneath or within accumulations of organic matter on which they feed and rarely burrow into or ingest much soil (Fig. 1). Typical habitats include forest litter or manure piles, thus they have little direct effect on the structure of mineral soils and are rarely found in tilled fields.

Table 1 Web-based resources on earthworms and soil structure.

Source	Contents	Address
U.S. Department of Agriculture-Natural Resources Conservation Service Soil Biology Primer	General information on soil fauna and their effects on soil	http://www.nrcs.usda.gov/wps/portal/nrcs/main/soils/health/biology/
Canada Worm Watch	Information on how to collect and identify earthworms	https://www.naturewatch.ca/wormwatch/
Purdue University	Extension publication on earthworms and crop management	https://www.extension.purdue.edu/extmedia/ay/ay-279.html
University of California	Articles on earthworm biology and sustainable agriculture	http://asi.ucdavis.edu/programs/sarep/research-initiatives/are/ecosystem/earthworm-information
Great Lakes Worm Watch	General earthworm biology plus information on the impact of invasive earthworms on forest ecology	http://www.nrri.umn.edu/worms/
Earthworm Society of Britain	General information on earthworm biology, ecology, and identification	http://www.earthwormsoc.org.uk
Penn State University	Extension publication on earthworms including effects of agricultural practices	http://extension.psu.edu/plants/crops/soil-management/soil-quality/earthworms
University of Florida	General information on earthworms with an emphasis on species found in the southern United States	http://entnemdept.ufl.edu/creatures/MISC/MISC/Earthworm.htm
New South Wales Department of Primary Industries	Information on benefits of earthworms and how to encourage their activity in Australia	http://www.dpi.nsw.gov.au/agriculture/resources/soils/biology/earthworms
Research Institute of Organic Agriculture, FiBL	Information on earthworm effects on soil fertility with an emphasis on organic management practices	https://www.fibl.org/fileadmin/documents/shop/1629-earthworms.pdf

They are usually small, heavily pigmented, reproduce rapidly, and have a short life span. *Endogeic* earthworms burrow extensively belowground and obtain their nutrition by ingesting a mixture of soil and organic matter (Fig. 1). They form extensively branched, subhorizontal networks of burrows in search of food, but most of their activity is in the 10–15 cm upper where organic matter levels are generally highest. Portions of their burrows are often

occluded with their excrement (casts) and they occasionally cast on the soil surface. They are generally medium in size, reproduction rate, and longevity and are lightly pigmented. *Anecic* earthworms are generally larger than the other ecological types, reproduce more slowly, are relatively long-lived, and tend to be pigmented only dorsally. They normally live in permanent or semipermanent burrows that can extend deep into the soil. They feed primarily on decaying surficial organic litter that they frequently pull into their burrows or mix with excrement to form a midden (Fig. 1). The midden blocks the burrow entrance and promotes further decay of the incorporated organic residues. These categories are not absolute, however, as the behavior of many species is intermediate to these groupings and can vary with environmental conditions.^[2]

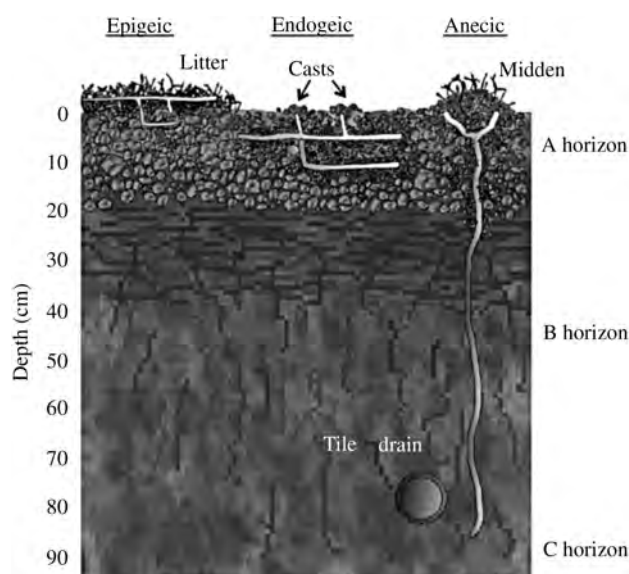


Fig. 1 Diagrammatic representation of the burrows made by the three ecological groups of earthworms as defined by Bouché.

EFFECTS ON SOIL POROSITY

Because they burrow extensively into mineral soil, endogeic and anecic earthworms can substantially alter soil porosity. Although earthworm burrows usually account for a small fraction of the soil volume, due to their continuity, stability, and relatively large size compared to pores formed by most other mechanisms, these macropores can greatly affect the movement of air, water, and solutes. A number of investigators have demonstrated that burrows made by anecic and endogeic earthworms can effectively conduct water.^[3–5] Because most of their activity is confined to surficial soil horizons, however, endogeic earthworms probably do not directly influence movement deep into the profile.^[6] The fact

that portions of their burrows are often occluded with casts, probably further limits their effectiveness.

On the other hand, anecic earthworms have the potential to influence gas, water, and solute movement throughout the profile. For example, burrows created by *Lumbricus terrestris* L. (a widespread anecic species) are normally single, nearly vertical channels up to 12 mm in diameter and 2.4 m deep.^[2] These burrows can have several entrances directly underneath the midden, but these usually coalesce into a single channel within the upper few centimeters of soil (Fig. 1). Although the midden would seem to inhibit entry of water, field studies conducted on burrows with undisturbed entrances indicate that they can transmit substantial amounts of water, up to 10% of rainfall.^[4,7]

Both burrow types can increase infiltration thereby increasing plant-available water and reducing surface runoff. For instance, when earthworms were eliminated from a pasture, a three-fold reduction in infiltration rate and a two-fold increase in runoff were noted.^[8] In cultivated soils, earthworms can also reduce runoff by disrupting surface crusts that impede infiltration.^[9] The contribution of earthworm burrows to infiltration, however, is dependent on a number of factors. High-intensity rainfall and dry soil can increase flow in earthworm burrows.^[4,5,7] Although it seems logical to assume that earthworms might block flow in their burrows, infiltration rates for burrows with the worm removed are similar to those for occupied burrows.^[10] In fact, occupied burrows are probably more effective in transmitting water than abandoned burrows because they are more likely to maintain near-surface continuity. Theoretically, it should be possible to model the contribution of earthworm burrows and other macropores to infiltration based on their distribution and geometrical properties.^[11] This has proved difficult as not all burrows conduct water and their dimensions are not strongly correlated to their infiltration capacity.^[10,12] Nevertheless, investigators have been successful in linking spatial variability in infiltration patterns on a catchment scale to earthworm populations and macropore distribution.^[13]

In rare circumstances, increased infiltration due to earthworm burrows can have negative consequences. Earthworm burrows can contribute to non-uniform distribution of water during furrow irrigation, loss of water through unlined irrigation ditches,^[6] and leakage of manure storage lagoons.^[14] Anecic earthworms can burrow close to tile lines (Figs. 1 and 2) and may increase losses of injected animal wastes in drainage waters.^[12] Earthworm burrows can also increase leaching of surface-applied agrochemicals, particularly when intense storms occur shortly after application on residue-covered no-till soils.^[4] The potential for this to occur is greatly reduced with time and low-intensity, intervening, rainfalls. Ingestion of herbicide-coated residues by earthworms can also reduce leaching losses.^[15] Once the water enters the burrows, the organic matter-rich linings may further reduce herbicide transport by increasing sorption and degradation.^[3]



Fig. 2 This *L. terrestris* burrow, impregnated with plastic and excavated *in situ*, passed within 2 cm of a buried tile, and had an average infiltration rate of 353 ml min⁻¹.

Furthermore, anecic earthworms can alter carbon distribution in soils by physically moving plant residues into the subsoil, where it may contribute to increased organic carbon sequestration.^[16] There is an unsettled debate as to whether earthworms increase greenhouse gas emissions by enhancing organic matter decomposition or whether there is a net increase in carbon sequestration attributable to stabilizing of organic matter in soil aggregates.^[17,18]

EFFECTS ON AGGREGATION

Although earthworms feed on decaying organic matter and the microorganisms that colonize it, the material ingested by endogeic and anecic species during feeding and burrowing is predominately mineral matter.^[3] This mixture is excreted as casts on the soil surface or belowground, depending on the species of earthworm, location of the food source, and soil bulk density.^[19] The casts usually contain more clay and less sand than the surrounding soil due to selective ingestion with the effect more prominent with endogeic species, which tend to be smaller than anecic earthworms.^[3] Moreover, earthworm casts are usually higher in pH, contain more available nutrients, and have higher levels of microbial activity than the uningested soil.^[1,2]

Freshly excreted casts are initially less water stable than uningested soil because digestive secretions and the peristaltic action of the earthworm gut disrupt many of the existing interparticle bonds. During passage through the earthworm, however, the mineral matter is intimately mixed with ingested organic matter. If casts are allowed to age or dry before being subjected to disruption, their stability can

exceed that of the uningested soil, thereby enhancing formation of a desirable, water-stable, granular, soil structure.^[3]

A number of mechanisms can contribute to the increased stability of earthworm casts with aging or drying. These include chemical or mechanical stabilization by: 1) internal secretions of earthworms; 2) plant fibers incorporated into casts; 3) growth of fungal hyphae; 4) bacterially produced gums; 5) bonding by calcium humate or mucilage; 6) wetting and drying cycles; and 7) age-hardening/thixotropic effects combined with organic bonding.^[3] The mechanisms are not mutually exclusive and the relative contribution of a particular process is probably dependent on a number of factors. For the most part, however, incorporation of organic matter into casts is critical either as bonding agent or as promoter of microbial activity that leads to the production of bonding agents. For this reason a positive correlation between organic carbon content and cast stability is frequently noted. Thus, casts of earthworms that have higher organic matter ingestion rates are more stable than those that ingest more mineral-rich mixtures. Additionally, the distribution of the organic bonding agents is probably more important than the total amount of organic matter within the casts.^[3]

Because freshly deposited casts are initially of low stability, they are subject to dispersion and transport if not protected from raindrop impact or the action of flowing water. Thus, earthworm activity can increase infiltration and reduce runoff while increasing losses of soil and sediment associated nutrients from pastures^[8] and cultivated fields.^[19] Furthermore, foraging and midden building by anecic earthworms can reduce surface residue cover, thus exposing more soil and casts to raindrop impact with negative consequences for soil structure.^[20]

CONCLUSIONS

In general, earthworm activity improves soil structure by increasing macroporosity and enhancing aggregation, which in turn can reduce runoff and sediment loss and provide a better environment for plant growth. Under some circumstances, however, increased infiltration, deposition of casts on the soil surface, and excessive residue removal can have undesirable consequences. For the most part, these problems can be minimized by adopting modified management practices.

REFERENCES

1. Lee, K.E. *Earthworms, Their Ecology and Relationships with Soils and Land Use*; Academic Press: New York, 1985; 411 pp.
2. Edwards, C.A.; Bohlen, P.J. *Biology and Ecology of Earthworms*, 3rd Ed.; Chapman & Hall: London, 1996; 426 pp.
3. Shipitalo, M.J.; Le Bayon, R.-C. Quantifying the effects of earthworms on soil aggregation and porosity. In *Earthworm Ecology*, 2nd Ed.; Edwards, C.A., Ed.; CRC Press: Boca Raton, 2004; 183–200.
4. Shipitalo, M.J.; Dick, W.A.; Edwards, W.M. Conservation tillage and macropore factors that affect water movement and the fate of chemicals. *Soil Till. Res.* **2000**, *53* (3–4), 167–183.
5. Trojan, M.D.; Linden, D.R. Microrelief and rainfall effects on water and solute movement in earthworm burrows. *Soil Sci. Soc. Am. J.* **1992**, *56* (3), 727–733.
6. Kemper, W.D.; Trout, T.J.; Segeren, A.; Bullock, M. Worms and water. *J. Soil Water Conserv.* **1987**, *42* (6), 401–404.
7. Edwards, W.M.; Shipitalo, M.J.; Owens, L.B.; Norton, L.D. Water and nitrate movement in earthworm burrows within long-term no-till cornfields. *J. Soil Water Conserv.* **1989**, *44* (3), 240–243.
8. Sharpley, A.N.; Syers, J.K.; Springett, J.A. Effect of surface-casting earthworms on the transport of phosphorus and nitrogen in surface runoff from pasture. *Soil Biol. Biochem.* **1979**, *11* (5), 459–462.
9. Klavivko, E.J.; Mackay, A.D.; Bradford, J.M. Earthworms as a factor in the reduction of soil crusting. *Soil Sci. Soc. Am. J.* **1986**, *50* (1), 191–196.
10. Shipitalo, M.J.; Butt, K.R. Occupancy and geometrical properties of *Lumbricus terrestris* L. burrows affecting infiltration. *Pedobiologia* **1999**, *43* (6), 782–794.
11. Jarvis, N.J. A review of non-equilibrium water flow and solute transport in soil macropores: Principles, controlling factors and consequence for water quality. *Eur. J. Soil Sci.* **2007**, *58* (3), 523–546.
12. Shipitalo, M.J.; Gibbs, F. Potential of earthworm burrows to transmit injected animal wastes to tile drains. *Soil Sci. Soc. Am. J.* **2000**, *64* (6), 2103–2109.
13. van Schaik, L.; Palm, J.; Klaus, J.; Zehe, E.; Schröder, B. Linking spatial earthworm distribution to macropore numbers and hydrological effectiveness. *Ecohydrology* **2014**, *7* (2), 401–408.
14. McCurdy, M.; McSweeney, K. The origin and identification of macropores in an earthen-lined dairy manure storage basin. *J. Environ. Qual.* **1993**, *22* (1), 148–154.
15. Farenhorst, A.; Topp, E.; Bowman, B.T.; Tomlin, A.D. Earthworm burrowing and feeding activity and the potential for atrazine transport by preferential flow. *Soil Biol. Biochem.* **2000**, *32* (4), 479–488.
16. Don, A.; Steinberg, B.; Schöning, I.; Pritsch, K.; Joschko, M.; Gleixner, G.; Schulze, E.-D. Organic carbon sequestration in earthworm burrows. *Soil Biol. Biochem.* **2008**, *40* (7), 1803–1812.
17. Lubbers, I.M.; van Groenigen, K.J.; Fonte, S.J.; Six, J.; Brussaard, L.; van Groenigen, J.W. Greenhouse-gas emissions from soils increased by earthworms. *Nature Clim. Change* **2013**, *3* (3), 187–194.
18. Zhang, W.; Hendrix, P.F.; Dame, L.E.; Burke, R.A.; Wu, J.; Neher, D.A.; Li, J.; Shao, Y.; Fu, S. Earthworms facilitate carbon sequestration through unequal amplification of carbon stabilization compared with mineralization. *Nat. Commun.* **2013**, *4*, 2576. doi: 10.1038/ncomms3576.
19. Binet, F.; Le Bayon, R.C. Space-time dynamics *in situ* of earthworm casts under temperate cultivated soils. *Soil Biol. Biochem.* **1999**, *31* (1), 85–93.
20. Shuster, W.D.; Subler, S.; McCoy, E.L. Foraging by deep-burrowing earthworms degrades surface soil structure of a fluventic hapludoll in Ohio. *Soil Till. Res.* **2000**, *54* (3–4), 179–189.

Structure: Pedological Concepts and Water Flow

Clinton C. Truman

Southeast Watershed Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Tifton, Georgia, U.S.A.

Donald P. Franzmeier

Department of Agronomy, Purdue University, West Lafayette, Indiana, U.S.A.

Abstract

Soil structure is the combination or arrangement of primary soil particles into secondary units or peds. Soil structure involves the geometric arrangement of particles and interparticle forces that act between them. Soil structure influences how a soil behaves, including how water moves into (infiltration), across (runoff), and through (percolation or drainage) a given soil. This entry discusses pedologic classification of soil structure, general mechanisms of ped formation, and the importance of soil structure for water flow and water quality.

CHARACTERIZATION OF STRUCTURE

Soil peds are characterized by their shape, size, and grade.^[1] The shape of peds is mainly spherical or rectilinear (Fig. 1). Spherical peds comprise *granular* structure. Rectilinear peds are *platy* if their vertical dimension is smaller than the horizontal dimension, *prismatic* if the vertical dimension is larger, or *blocky* if peds have about equal vertical and horizontal dimensions. *Columnar* is a special kind of prismatic structure where ped tops are rounded. Blocky structure can be subdivided into *angular blocky* if peds have sharp edges and corners or *subangular blocky* if edges and corners are rounded.

The size and structure of peds are described by the diameter of granules, the thickness of the plates, the width of prisms, or any dimension of the blocks. Size limits vary from <1 to >100 mm (Table 1).

Structure *grade* is described according to how evident peds appear in a soil exposure and how well they hold together when the soil is handled. Four classes of grade are *structureless*, *weak*, *medium*, and *strong* (Table 1). Structureless soils have no visible structure and are subdivided into two contrasting classes. *Single grain* soils are sandy, and individual particles are not associated with other particles. In *massive* structure, particles are strongly bound to each other but in no recognizable pattern. Soil horizons that have undergone little or no soil formation have massive structure. Also, massive structure is often formed when a fine-textured soil is compacted while wet.

Some soils have compound structure in which large peds break up (part) into smaller peds of different shape. For example, a coarse prismatic structure may break into medium blocky structure (Fig. 2).

FORMATION OF SOIL STRUCTURE

Chemical, biological, and physical processes interact to form soil structure. Soil particles, such as clay, act separately in a *dispersed* system, or they group together to form domains in a *flocculated* system. Domains, in turn, group together to form aggregates, and aggregates to create peds. Flocculated systems form stronger structure than dispersed systems because of particle interaction. If the salt content of a soil (Na^+ , Ca^{2+} , Mg^{2+} , Cl^- , SO_4^{2-} , etc. in soil solution) is high, as in the desert, the soil tends to be flocculated. In most soils, however, salt content is not high enough to cause flocculation. In these soils, the *kind* of exchangeable cation affects flocculation. Sodium ions (Na^+) and to a lesser extent, Mg^{2+} , tend to disperse soils, whereas Ca^{2+} and Al^{3+} tend to flocculate soils. Also, iron and aluminum oxides tend to flocculate soils.

Biological processes are mainly responsible for *granular* structure. Ants and termites secrete organic “glues” that hold particles together. Earthworms ingest soil, mix it with organic compounds, and excrete aggregated soil material. Plants also secrete organic materials that bind soil particles together. These organisms mainly affect surface (A) horizons. Thus, these horizons generally have granular structure.

Blocky, platy, and prismatic structures are formed by intersecting horizontal and vertical fissures or planes of weakness in soils, mainly subsoil horizons. Vertical fissures are created when a soil shrinks. For example, a water puddle often has a layer of fine soil particles at the bottom. When the water evaporates and the fine material dries, it breaks up into polygons separated from each other by cracks. The same process takes place in subsoils.^[2] When soil dries and shrinks, smaller sized peds in the east–west

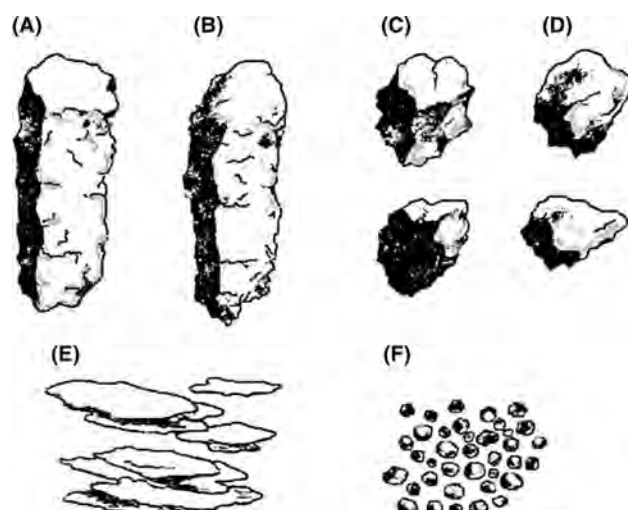


Fig. 1 Sketches of soil structure shapes: (A) prismatic, (B) columnar, (C) angular blocky, (D) subangular blocky, (E) platy, and (F) granular.

Source: From SSSA.^[3]

and north–south directions are balanced by creation of vertical fissures—the field does not get smaller in area. Soil shrinkage creates some, but fewer, horizontal fissures because soil elevation may actually decrease as soil shrinks. Horizontal fissures may also be caused by freezing and thawing. When soil freezes, liquid water moves to the freezing front where a layer of ice forms and displaces soil material. The freezing front tends to be at about the same horizontal level in the soil. When ice melts, a horizontal fissure is left.

Table 1 Classes of grade and size of soil structure.

Class	Definition			
Grade (strength of development)				
Structureless (0)	No observable structure (massive or single grain)			
Weak (1)	Structure barely observable in place; few peds survive digging			
Moderate (2)	Peds are well formed and evident in place; many peds survive digging			
Strong (3)	Structure distinct in place; most peds survive digging			
Class	Platy (mm)	Prismatic and columnar (mm)	Blocky (mm)	Granular (mm)
Size				
Very fine (VF)	<1	<10	<5	<1
Fine (F)	1–2	10–20	5–10	1–2
Medium (M)	2–5	20–50	10–20	2–5
Coarse (C)	5–10	50–100	20–50	5–10
Very coarse (VC)	>10	>100	>50	>10

Source: From Soil Survey Division Staff.^[1]



Fig. 2 Sketch illustrating compound structure, moderate coarse prismatic structure parting to weak medium angular blocky structure.

Platy structure forms where horizontal fissures predominate, such as near-surface E horizons that undergo many freeze–thaw cycles. *Prismatic* structure forms where vertical fissures predominate, such as in clayey subsoil horizons that dry out due to evapotranspiration. *Blocky* structure forms when horizontal and vertical fissures intersect.

Soil structure is defined according to how individual particles (sand, silt, and clay) are joined together to form peds. Soil scientists, however, describe the end product—how the mass of soil in a profile breaks out into peds. Examples of soil structure descriptions from a published soil survey are “moderate medium granular structure” (A horizon) and “strong very coarse prismatic structure parting to moderate coarse angular blocky” (Bx horizon, fragipan).

Often ped surfaces are coated with clay, and the resulting clay films or skins make ped shape more apparent. Clay is mobilized in near-surface horizons, moves down the soil profile as a slurry, and is immobilized on a ped surface as water soaks into the ped and clay is filtered out on its surface. Chemical changes down the profile, such as a change in pH, may also help clay skin formation. Other mobile constituents (iron or manganese oxides) may also coat ped surfaces.

SIGNIFICANCE OF SOIL STRUCTURE

Characterizing soil structure is based on where soil particles are located, but the significance of soil structure relates to the space where there are no soil particles. Spaces between soil particles are called voids or pores. Pores are often divided into two types—macropores and micropores. Voids between peds are macropores and serve as pathways for roots and rapid water movement. Micropores are smaller pores between particles and aggregates. They hold water between rains for later use by plants.

Flat spaces between adjoining platy, blocky, or prismatic peds are planar voids. The ease with which water moves and roots grow through a planar voids depends on ped

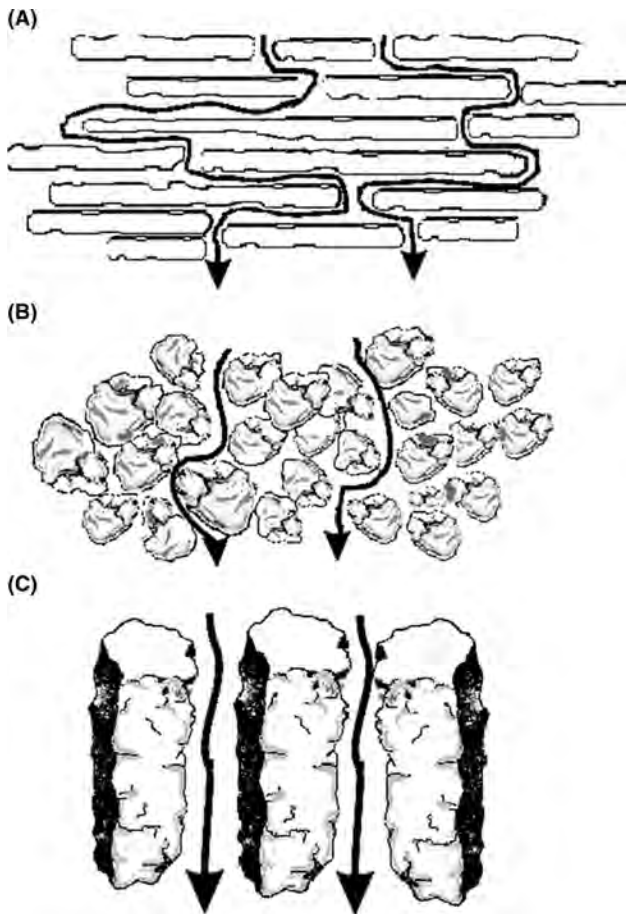


Fig. 3 Sketch illustrating paths of water flow through soils with different soil structure shapes: (A) platy; (B) blocky; and (C) prismatic.

shape (Fig. 3). The route through a platy structure is more tortuous than through a prismatic structure.

Tubular voids, such as root channels and earthworm burrows, are not usually included in the description of structure but are important for water movement and root growth in soils. They act much like planar voids. Often, new roots follow old root channels in which old roots have decayed. Inactive worm channels are especially favorable environments for roots because they are often lined with highly fertile worm castings.

Generally, permeability increases with stronger structural development and smaller peds. Soils with faster permeability are also conducive to better root growth and movement of soil air to roots.

WATER MOVEMENT AND SOIL STRUCTURE

Water moves slowly into and through ped interiors by *matrix flow* and moves rapidly through larger voids as *preferential flow*. Interaction of these two kinds of flow is shown in Fig. 4. In the left part of the diagram, rectangles represent

peds (prisms in this case), and the space between rectangles represents planar voids. The size of the arrow represents the magnitude of rainfall. Thin lines within a prism show matrix flow, and thick lines on the prism surface show preferential flow. Under a gentle rain, the water application, rate 1, all water soaks into (infiltrates) the prism as matrix flow, mainly from the top. With a moderate rain, rate 2, water starts to soak into the prism, but soon the intake rate of the prism is exceeded and preferential or non-matrix flow begins. Then, water also soaks into the side of the prism. Under a hard rain, rate 3, water moves by matrix and preferential flow almost from the beginning of the storm. The same relations are shown in the graph, with the additional information that the infiltration rate into the prism decreases with time. Preferential flow begins when the horizontal line with the arrow crosses the infiltration curve.

SOIL STRUCTURAL STABILITY

Soil structural stability describes the ability of a soil to retain its arrangement of soil and voids when exposed to different stresses and is one of the most important properties that affect a soil's ability to store and transmit water and agrichemicals.^[4] For example, compaction alters soil structure and hydrology by increasing bulk density, breaking down aggregates, decreasing porosity, aeration, and infiltration, and by increasing soil strength and runoff.

SOIL MANAGEMENT FOR WATER FLOW

Runoff, infiltration, and subsequent drainage carry agrichemicals (nutrients and pesticides) to surface and ground waters. Therefore, soil structure affects rates, timing, and amounts of runoff, infiltration, and agrichemical fate and transport. In the subsurface, soil structure influences matrix or preferential flow.^[5]

Agrichemical transport by preferential flow means that water carrying the chemicals travels only through a small portion of the soil volume (not matrix flow).^[6] When infiltration exceeds matrix flow, water and associated agrichemicals overflow into macrochannels causing preferential flow that transmits water and agrichemicals to deeper positions in relatively unsaturated soil materials.^[7]

Although coarse-textured sandy soils generally have little structure, they have good aeration and drainage, promote infiltration, limit runoff, yet lack the capacity to sorb and hold sufficient water and nutrients. They tend to be drought prone and lack fertility. A management practice to improve the structural nature of these soils is to add organic matter, which binds soil particles and promotes water-holding capacity, and can be accomplished by managing residue or addition of amendments such as manures or litter. Structural management of clayey soils is more difficult because potential plasticity and cohesion are high. Increasing aggregation in the surface of fine-textured soils

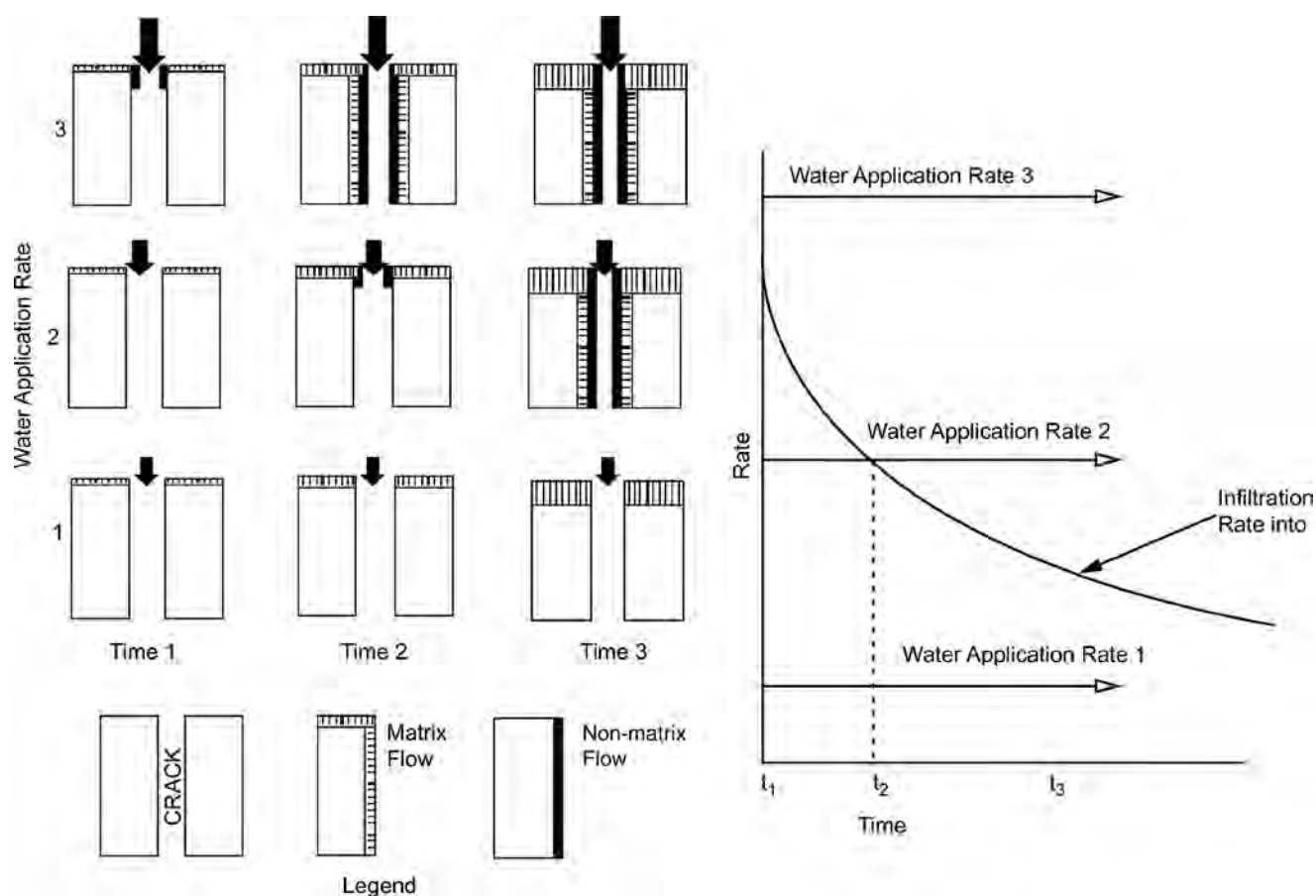


Fig. 4 Diagram illustrating water flow into and through a soil horizon with prismatic structure. Water application (rainfall) rate 1 is small enough that only matrix flow occurs. Application rate 2 initially produces matrix flow. As the soil wets, the infiltration rate into a prism decreases, application rate exceeds the infiltration rate, and non-matrix flow begins. Application rate 3 is fast enough that non-matrix (preferential) flow begins early and continues throughout the rain storm.

Source: From Bouma, Paetzold, et al.^[8]

will have a dramatic effect on rainfall partitioning into infiltration and runoff and sediment delivery, and subsequent agrichemical fate and transport.

Tillage has favorable and unfavorable effects on soil structure. Moldboard plowing destroys soil structure and aggregation in the Ap horizon, disrupts continuous macropores, increases compaction risk, and retards organic matter build up. One management practice that maintains or enhances soil structure and aggregate stability is reduced (or conservation) tillage. This type of tillage allows for residue management and subsequent organic matter build up. Organic matter is most important in modifying soil structural characteristics and can increase the resistance of the pore network to destructive forces. Because of enhanced soil structure, reduced tillage systems tend to decrease runoff and erosion; increase infiltration, soil water holding capacity, and hydraulic conductivity; and reduce agrichemical losses by runoff.^[9–11] Reduced tillage systems also conserve soil structure including preferential flow pathways (earthworm burrows, root channels, fissures, and/or cracks). Therefore, leaching by preferential flow can be

enhanced by reduced tillage. Transport pathways are grouped into runoff and leaching, each a transporting agent for agrichemicals. Runoff and leaching potentials must be considered when evaluating management practices such as

Table 2 Selected topics and corresponding references associated with soil structure.

Topic(s)	References
Methods of quantifying/studying soil structure	[12–14]
Modeling soil structure	[5,10,15,16]
Improving soil structure	[17–19]
Biological effects on soil structure	[20,21]
Tillage and cropping effects on soil structure	[4,22–24]
Soil structure effects on chemical movement	[25–28]
Drainage and soil structure	[26,29]
Soil structure, and Vertisols and cracking	[23,30]
Soil classification, preferential flow, and soil structure	[5,31]

tillage on soil structure, water flow, and water quality. Additional literature on soil structure effects on water flow and water quality is given in [Table 2](#).

REFERENCES

1. Soil Survey Division Staff. *Soil Survey Manual*; USDA–Soil Conservation Service, Agric. Handbook 18; U.S. Government Printing Office: Washington, DC, 1993; 437.
2. Fanning, D.S.; Fanning, M.C.B. *Soil Morphology, Genesis, and Classification*; Wiley: New York, 1989; 395.
3. SSSA. *Glossary of Soil Science Terms*; SSSA: Madison, 1997; 134.
4. Gregorich, E.; Reynolds, W.; Culley, J.; McGovern, M.; Curnoe, W. Changes in soil physical properties with depth in a conventionally tilled soil after no-tillage. *Soil Tillage Res.* **1993**, *26*, 289–299.
5. Vervoort, R.; Radcliffe, D.; West, L. Soil structure development and preferential solute flow. *Water Resour. Res.* **1999**, *35*, 913–928.
6. Flury, M. Experimental evidence of transport of pesticides through field soils—A review. *J. Environ. Qual.* **1996**, *25*, 25–45.
7. McIntosh, J.; McDonnell, J.; Peters, N. Tracer and hydro-metric study of preferential flow in large undisturbed soil cores from the Georgia Piedmont, USA. *Hydrol. Process.* **1999**, *13*, 139–155.
8. Bouma, J.; Paetzold, R.F.; Grossman, R.B. *Measuring Hydraulic Conductivity for Use in Soil Survey*; Soil Survey Investigations Report No. 38; Soil Conservation Service, U.S. Department of Agriculture: Washington, DC, 1982.
9. Leonard, R. Movement of pesticides into surface waters. In *Pesticides in the Soil Environment*; Cheng, H.H., Ed.; Soil Science Society America Book Series 2; SSSA: Madison, 1990; 303–349.
10. Cresswell, H.; Smiles, D.; Williams, J. Soil structure, soil hydraulic-properties and the soil-water balance. *Austr. J. Soil Res.* **1992**, *30*, 265–283.
11. Pagliai, M.; Raglione, M.; Panini, T.; Maletta, M.; La Marca, M. The structure of two alluvial soils in Italy after 10 years of conventional and minimum tillage. *Soil Tillage Res.* **1995**, *34*, 209–223.
12. Bartoli, F.; Bird, N.; Gomendy, V.; Vivier, H.; Niquet, S. The relation between silty soil structures and their mercury porosimetry curve counterparts: Fractals and percolation. *Eur. J. Soil Sci.* **1999**, *50*, 9–22.
13. Gomendy, V.; Bartoli, F.; Burtin, G.; Doirisse, M.; Philippy, R.; Niquet, S.; Vivier, H. Silty topsoil structure and its dynamics: The fractal approach. *Geoderma* **1999**, *88*, 165–189.
14. Hakansson, I.; Lipiec, J. A review of the usefulness of relative bulk density values in studies of soil structure and compaction. *Soil Tillage Res.* **2000**, *53*, 71–85.
15. Gimenez, D.; Perfect, E.; Rawls, W.; Pachepsky, Y. Fractal models for predicting soil hydraulic properties: A review. *Eng. Geol.* **1997**, *48*, 161–183.
16. Connolly, R. Modelling effects of soil structure on the water balance of soil-crop systems: A review. *Soil Tillage Res.* **1998**, *48*, 1–19.
17. Haynes, R.; Naidu, R. Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: A review. *Nutr. Cycling Agroecosys.* **1998**, *51*, 123–137.
18. Kort, J.; Collins, M.; Ditsch, D. A review of soil erosion potential associated with biomass crops. *Biomass Bioener.* **1998**, *14*, 351–359.
19. Nemati, M.; Caron, J.; Gallichand, J. Using paper de-inking sludge to maintain soil structural form: Field measurements. *Soil Sci. Soc. Am. J.* **2000**, *64*, 275–285.
20. Lee, K.; Foster, R. Soil fauna and soil structure. *Austr. J. Soil Res.* **1991**, *29*, 745–775.
21. Shuster, W.; Subler, S.; McCoy, E. Foraging by deep-burrowing earthworms degrades surface soil structure of a fluventic hapludoll in Ohio. *Soil Tillage Res.* **2000**, *54*, 179–189.
22. Kay, B. Rates of change of soil structure under different cropping systems. *Adv. Soil Sci.* **1990**, *12*, 1–51.
23. Oygarden, L.; Kvaerner, J.; Jenssen, P. Soil erosion via preferential flow to drainage systems in clay soils. *Geoderma* **1997**, *76*, 65–86.
24. Angers, D. Water-stable aggregation of Quebec silty clay soils: Some factors controlling its dynamics. *Soil Tillage Res.* **1998**, *47*, 91–96.
25. Beck, A.; Johnston, A.; Jones, K. Movement of nonionic organic-chemicals in agricultural soils. *Critical Rev. Environ. Sci. Tech.* **1993**, *23*, 219–248.
26. Ball, B.; Campbell, D.; Douglas, J.; Henshall, J.; O’Sullivan, M. Soil structural quality, compaction and land management. *Eur. J. Soil Sci.* **1997**, *48*, 501–593.
27. Bergstrom, L.; Stenstrom, J. Environmental fate of chemicals in soil. *Ambio* **1998**, *27*, 16–23.
28. Walton, R.; Volker, R.; Bristow, K.; Smettem, K. Experimental examination of solute transport by surface runoff from low-angle slopes. *J. Hydro.* **2000**, *233*, 19–36.
29. Collis-George, N. Drainage and soil structure—A review. *Austr. J. Soil Res.* **1991**, *29* (6), 923–933.
30. Pillai, U.; McGarry, D. Structure repair of a compacted vertisol with wet–dry cycles and crops. *Soil Sci. Soc. Am. J.* **1999**, *63*, 201–210.
31. Quisenberry, V.; Smith, B.; Phillips, R.; Scott, H.; Northcliff, S. A soil classification system for describing water and chemical transport. *Soil Sci.* **1993**, *156*, 306–315.

BIBLIOGRAPHY

1. Soil Survey Staff. In *Soil Survey Manual*; USDA–Soil Conservation Service, Agric. Handbook 18; U.S. Govt. Print. Office: Washington, DC, 1962.

Structure: Plant Establishment

Douglas L. Karlen

National Laboratory for Agriculture and the Environment, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.

Abstract

Soil structure influences plant establishment and plants influence soil structure. Factors involved in the reciprocal relationship (e.g., plant species, soil mineralogy, and human intervention) are summarized to provide an overview of a very dynamic process that can influence food, feed, fiber, energy production, and water and air quality. Soil structure influences plant germination, emergence, root development, growth, development, and yield. How plants affect soil structure and how soil structure affects plants are discussed in this entry.

PLANT EFFECTS ON SOIL STRUCTURE

As a medium for plant growth and development, soil is a three-dimensional, dynamic, naturally occurring blanket on the surface of the earth.^[1] This medium has an inherent (naturally occurring) soil structure that reflects the forces of climate, and living organisms acting on parent material, over time and within a specific topographic or landscape (slope or relief) position.^[2] The naturally occurring soil structure directly affects, and is affected by, the type of plants (e.g., grass, forest, or agricultural crops) and how they are grown. Soils formed under grass (Fig. 1) generally have higher organic matter content and a structure that is conducive for rapid plant growth.^[3] This type of soil structure develops as perennial grasses replace most of their roots and top growth every year. As the litter layer dies, it falls on the soil surface and may be decomposed or mixed into the upper part of the soil by earthworms and other soil organisms.

Grassland soils are generally less acidic than soils that developed under forest. This occurs as grassland soils were generally formed with less annual precipitation than nearby forest soils and as grasses are more effective than trees in recycling nutrients that can be leached. Forest vegetation is more prevalent under a humid (higher precipitation) climate, which also contributes to greater leaching. Trees return organic matter to the soil surface primarily in the form of fallen leaves, twigs, and eventually the entire trunk and its branches.^[3] As the litter layer decomposes, organic acids are formed, thus increasing the weathering of parent materials and subsequent leaching losses. Most organic matter under forest vegetation is concentrated in the litter layer and remains in the upper few inches of the surface or A horizon. Below that relatively shallow zone in forest-derived soils, organic matter is often more eluviated (leached) than under grassland. The net result of these natural processes is that A horizons in

grassland soils (Mollisols) tend to have stronger, more granular structures, and faster infiltration rates than in forest-derived soils (Alfisols), but forest soils are more likely to have deeper profiles and more strongly developed B horizons than grassland soils.

HUMAN IMPACTS ON SOIL STRUCTURE

Human decisions regarding how to use the land for food, feed, and fiber production (i.e., agriculture) have a major impact on soil structure. The effect of choosing to cut native forests (especially by mechanical methods) or till [e.g., plowing, disking, rotary tillage, or even puddling (compaction) for rice] soils rather than to leave them in their natural condition can affect soil structure in both positive and negative ways depending upon the length of time for which the evaluation is being made. The short-term effects are generally positive with regard to crop production, especially on fine-textured soils. Tillage releases essential plant nutrients from plant residues or the soil organic matter and has traditionally provided for weed control. It can also improve the air and water relationships required for plant growth by breaking surface crusts and/or by disrupting dense compacted layers that can restrict plant emergence and growth.

Negative effects of tillage or other aggressive physical disturbances of the soil (i.e., deforestation or over grazing) become more evident when changes are observed over the long term. These impacts include loss of soil organic matter, reductions in total aggregation, declines in the water stability of the remaining aggregates, and decreases in the volume of macropores and biopores (Fig. 2). The long-term negative impacts actually begin immediately after cultivation as soil aggregates are broken apart and the existing structure is weakened or destroyed as the soil is loosened.



Fig. 1 Well-aggregated soil showing plant root growth throughout the profile.

However, the short-term benefits generally mask the long-term effects when observations are primarily focused on plant performance and above ground responses. The negative long-term effects begin to be

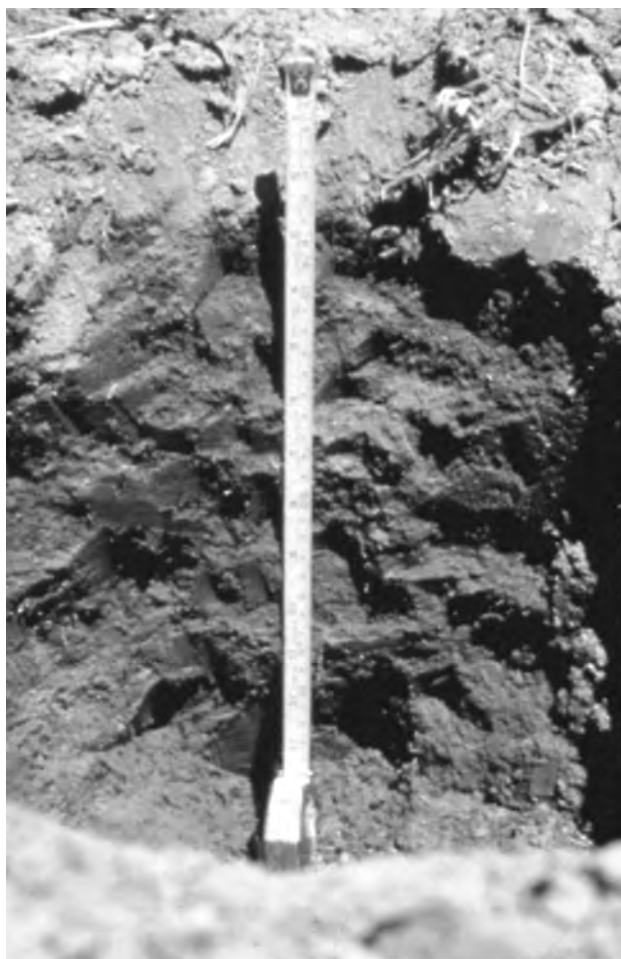


Fig. 2 Compacted soil with very little plant root growth throughout the soil profile.

evident after a short period of time as the soil particles gradually consolidate and create a denser soil than in nearby non-cultivated sites. As soil aggregates are broken or dispersed, the fine particles are gradually filtered into the soil pores where they accumulate and block the narrow passageways, thus decreasing porosity and creating even stronger surface crusts. The soil degradation continues as the crusts restrict water infiltration, thus increasing runoff and subsequently soil erosion.

SOIL STRUCTURE EFFECTS ON PLANT GROWTH

Germination is affected by soil structure, as the two primary factors affecting this plant process are soil temperature and water content. Soil temperature is affected by several factors including the angle of the sun's rays, soil color, surface residue or cover, and the depth and time of measurement. The most favorable temperature for seed germination depends on the plant species, but efforts to optimize those conditions, in addition to weed control prior to the development of herbicides, are among the primary reasons that agricultural soils have traditionally been tilled.^[4] The loose and friable soil created by cultivation at the expense of soil structure loses water through evaporation and warms more quickly than more-dense, non-tilled soil. The effects of increased soil temperature and decreased water content are not always positive, especially in coarse-textured soils.^[5] If the seedbed is too dry, germination can be delayed by the lack of water. Under those conditions, it is possible to actually observe plant growth benefits from a slight amount of soil compaction. Also, if the effect of surface tillage on soil temperature below the seed zone is of primary concern, the increased surface porosity following tillage can actually insulate against temperature change deeper into the soil profile. This occurs as the air space created by the tillage operation conducts heat much less efficiently than by either soil particles or water molecules.

Emergence of plant seedlings is affected by soil structure primarily when the surface soil consolidates and forms a crust. This occurs most frequently when soils have been tilled leaving very little surface residue prior to an intense rainfall or surface irrigation event. The thickness and strength of the crusts that can develop will be determined by several factors including the specific type and mineralogy of the soil, the amount of soil organic matter that is present, the stability of the soil aggregates, the intensity with which water strikes the soil-surface, and the air temperature or rate of drying after the wetting event.^[2]

Root development is another plant process that is strongly influenced by soil structure. Under normal conditions, plant roots during early stages of plant development will elongate and extend into the soil 5 to 10 times

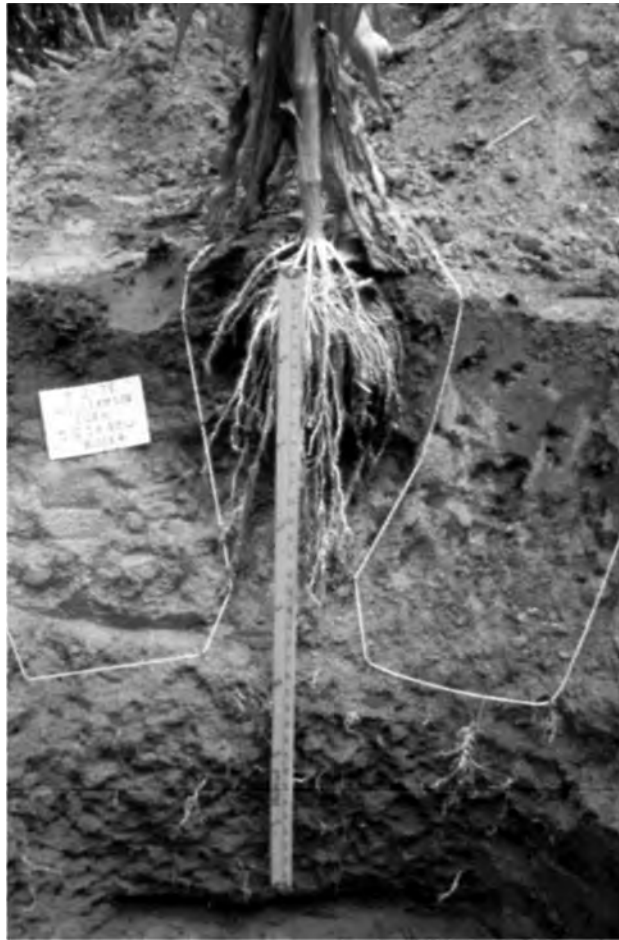


Fig. 3 Corn root growth in the surface horizons of a U.S. south-eastern coastal plain soil confined to a mechanically disturbed area created by in-row subsoiling.

deeper than the above ground height. This occurs because after a seedling emerges, the demand for water increases tremendously and the root system must be sufficiently established to meet those demands. If the soil structure does not allow the roots to expand because of inherent fragipans or eluviated soil horizons (Fig. 3) or because of compaction caused by animal hooves or wheel traffic (Fig. 4), plant growth, development, and ultimately yield or productivity will be reduced. The primary requirements for good root development are simply 1) adequate soil oxygen for physiological functioning of the root; 2) sufficient soil water for above and belowground plant needs; 3) soil temperature within the range suited to the plant species; 4) sufficient nutrients and sunlight for the plant to produce and transport carbohydrates (food and hormones) to the root system; and 5) relatively low or insignificant amounts of toxic chemicals, lethal gases, disease organisms, or insect damage.^[6] Keeping these factors in balance is a primary goal associated with agricultural soil management, when evaluated with regard to soil structure and plant emergence.



Fig. 4 Corn root growth in a U.S. mid-western Mollisol confined to the non-wheel track area (right-hand side).

MANAGING SOIL STRUCTURE FOR PLANT ESTABLISHMENT

Soil management research around the world has shown that the use of heavy machinery and frequent disk tillage can compact subsurface horizons in many irrigated soils. The result of such compaction in Morocco was a 12–23% decrease in wheat grain yield and a 9–20% decrease in straw yield.^[7] The decrease was accompanied by a consistent reduction in the number of shoots per unit area and significant changes in both root growth and distribution. In a U.S. study, improving aeration by subsoiling and by avoiding wheel traffic significantly increased sugar beet yield.^[8] This study showed that regardless of the type of tillage used prior to planting, minimizing wheel traffic was one of the most important management decisions that could be made.

Finally, even with reduced-tillage or no-tillage practices, it is important to understand how subtle changes in soil structure can influence plant establishment. A specific example is known as seed furrow or sidewall smearing.^[9] This situation can occur if mechanical planters equipped with double-disk furrow openers are used when soils are too wet. When sidewall smearing occurs, penetration resistance adjacent to and below the seed is increased, pore size is reduced, and air permeability is decreased. One solution is to use different planter attachments, such as a triple coulter, ahead of the double-disk openers. Another would be to delay planting until the soils were slightly drier, but that may also have negative consequences depending upon the crop, length of growing season, and other management factors.

CONCLUSION

Soil structure develops naturally (over time) in response to mineralogy and method of deposition (parent material),

climate, topography, vegetation, and human intervention. Soil structure affects air and water quality through its effect on water entry (runoff), retention (storage), and availability to plants. Soil structure is dynamic. Therefore, understanding how plants influence soil structure and vice versa is critical for sustainable land management.

REFERENCES

1. Doran, J.W.; Sarrantonio, M.; Liebig, M.A. Soil health and sustainability. In *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press: San Diego, 1996; Vol. 56, 1–54.
2. Henry, D. *Fundamentals of Soil Science*, 8th Ed.; Wiley: New York, 1990; 360 pp.
3. Troeh, F.R.; Thompson, L.M. *Soils and Soil Fertility*, 5th Ed.; Oxford University Press: New York, 1993; 462 pp.
4. Lal, R.; Stewart, B.A. *Soil Management: Experimental Basis for Sustainability and Environmental Quality, Advances in Soil Science*; CRC Press: Boca Raton, 1995; 555 pp.
5. Karlen, D.L. Tillage and planting system effects on corn emergence from Norfolk loamy sand. *Appl. Agr. Res.* **1989**, *4*, 190–195.
6. Trowse, A.C. Jr. Root tolerance to soil impediments. In *Crop Tolerance to Suboptimal Land Conditions*; Jung, G.A., Ed.; ASA Spec. Publ. No. 32; ASA/CSSA/SSSA: Madison, 1978; 193–232.
7. Oussible, M.; Crookston, R.K.; Larson, W.E. Subsurface compaction reduces the root and shoot growth and grain yield of wheat. *Agron. J.* **1992**, *84*, 34–38.
8. Johnson, B.S.; Erickson, A.E. Sugarbeet response to subsoiling and wheel traffic. *Agron. J.* **1991**, *83*, 386–390.
9. Iqbal, M.; Marley, S.J.; Erbach, D.C.; Kaspar, T.C. An evaluation of seed furrow smearing. *Trans. ASAE* **1998**, *41*, 1243–1248.

Structure: Roots

Alvin J.M. Smucker

Department of Crop and Soil Sciences, Michigan State University, East Lansing, Michigan, U.S.A.

Abstract

Soil structure includes a vast array of heterogeneous and separable individual soil aggregates. Soil aggregates are the primary repositories of carbon, water, microbial communities, plant nutrients, and pollutants within the soil profile. Soil aggregates develop and function through complex biological, chemical, and physical interactions with climate, water, and ion activities occurring within and across interfaces of adjacent soil aggregates within the soil profile. Repeated soil wetting and drying cycles develop stronger soil aggregates than the surrounding bulk soil and create smaller internal pores than the surrounding macropores of the bulk soil.

INTRODUCTION

Soil structure includes a vast array of heterogeneous and separable individual soil aggregates. Soil aggregates are the primary repositories of carbon (C), water, microbial communities, plant nutrients, and pollutants within the soil profile. These biophysical polymorph structures, ranging in size from submillimeter to many millimeters across, control the absorption, storage, and losses of most soil constituents. The dynamic properties of soil aggregates are controlled by soil type, anthropogenic inputs, daily changes in the weather, plant roots, and soil animal activities. Separable soil aggregates are formed during the combination of soil minerals, soil organic matter, plant residues, microbes, and other soil biological constituents.^[1] Soil aggregates develop and function through complex biological, chemical, and physical interactions with climate, water, and ion activities occurring within and across interfaces of adjacent soil aggregates within the soil profile. Formation processes of soil aggregates include the cementation of adjacent smaller soil aggregates into larger aggregates^[2] and/or accumulation and cementation of soil minerals, ions, and particulate organic matter onto surfaces of existing aggregates.^[3,4] Repeated soil wetting and drying cycles develop stronger soil aggregates than the surrounding bulk soil and create smaller internal pores than the surrounding macropores of the bulk soil.

PLANT ROOT CONTRIBUTIONS TO SOIL AGGREGATES

Plant species and root architectural contributions to the development of specific soil structural characteristics are well known. Root growth and exudation, fungal associations, and death contribute substantial quantities of C into soil aggregates.^[5] Many of the mechanisms controlling soil aggregation processes and dynamics are unknown.

The research suggests roots opportunistically invade pores between or within soil aggregates depositing soluble C substrates and residues that release C upon decomposition. These root products are energy-rich food sources that stimulate the microbial growth and the production of polysaccharides and other cementing agents that glue soil mineral particles into more stable soil aggregates.^[6–8] In relatively undisturbed soils, root derived C and subsequent decomposition products located within aggregates contribute more to the formation and stabilization of smaller soil aggregates than do plant residues located in the larger pores between aggregates.^[9] Greater numbers of aggregate wetting and drying cycles increase aggregate stability by promoting mineralization of root residues.^[10] Closer proximity of soil minerals, during drying, also increases the bridging strengths of adsorbed organic and inorganic compounds causing greater soil aggregate stabilities.^[11] These and other highly orchestrated plant genetic and environmental interactions govern the majority of these dynamic feed-forward and feed-back root and soil activities.

EVALUATING ROOT–SOIL AGGREGATION INTERACTIONS

Continuous rhizodeposition of C onto soil aggregates is one of the major contributions of crop management to soil structure formation and stability. There seems to be very little information on biogeochemical mechanisms associated with the improvement of C fixed by plants and retained for prolonged periods of time. Sissoko^[8] demonstrated how plant root exudates increase the stability of soil aggregates. Using a 7-day root exudation, incubation, and soil wetting and drying protocol for nine cycles, Sissoko developed soil aggregates containing 70% more microbial biomass and produced 280% more stable aggregates than water-based controls.

Natural Isotopes of C

Evaluating the complexities of C flow from plant roots to their various forms of cementation products within soil aggregates is complicated by the multiple biogeochemical processes of soil C mineralization, the spatial and temporal distributions of various C sources, and the heterogeneity of soil aggregate sizes and stabilities across a vast array of landscapes. Net accumulations of C, originating directly from plant residues or from soil organic matter (SOM), within soil aggregates can be estimated by mass spectrometric evaluations of the natural abundance of ^{13}C in bulk soil. Using soils that produced separate or combined populations of C3 or C4 plants whose metabolisms discriminate in their fixation and accumulation of $^{13}\text{CO}_2$ and $^{12}\text{CO}_2$,^[5] identified SOM and plant root C contributions to the stabilities of multiple soil aggregate sizes. With natural C-13 isotopes, it is possible to identify changes in bulk soil C sources after 6 years of continuous plant growth on the same soil.^[12]

Mechanical Removal of Concentric Layers

Plant-induced changes in soil C accumulations can be identified during shorter time periods by removing and analyzing the C isotope sources in 10–15 mg subsamples from concentric layers of soil aggregates. Santos et al.^[7] identified plant root C accumulations in peeled surface layers of soil aggregates as early as 6 weeks, for corn, using greenhouse containers, or 20 months, for alfalfa, in rotational field studies. Root depositions of ^{13}C and ^{15}N to specific concentric layers within soil aggregates are routinely being reported for aggregates sampled from the rhizosphere soils in surface horizons.^[7,13] Smaller stainless steel soil aggregate erosion (SAE) peeling chambers^[4] that more efficiently produce concentric layers of soil from individual, and smaller aggregates have been developed for identifying C, nitrogen (N), phosphorus (P), pH,^[13–15] and microbial communities^[16] at specific microsites within soil aggregates. These approaches provide numerous opportunities for identifying the rhizodeposition of additional plant compounds from specific C3 and C4 plant species.

Peeled soil layers from aggregates provide information on the deposition rates of both newly deposited labile C and older recalcitrant C within soil aggregates. Kinyangi^[14] demonstrated interdependent and short-term relationships among soil C, N, and P. Using dry combustion analyses of small (10–20 mg) soil samples extracted from concentric layers of soil aggregates, he compared ratios of $^{13}\text{C}/^{12}\text{C}$ soil C, planted to C3 species of *Crotalaria grahamiana* plants whose roots produced a $\delta^{13}\text{C}$ signature of -26.1% , and C4 cultivars of *Zea mays* plants whose roots produced a $\delta^{13}\text{C}$ signature of -10.6% with extractable P from the same peeled layers of aggregates from a Kenyan Oxisol. Using the following equation,

$$\% \text{C from maize} = \frac{[(\delta^{13}\text{C layer}_{\text{final}} - \delta^{13}\text{C layer}_{\text{initial}})]}{(\delta^{13}\text{C crop}_{\text{maize}} - \delta^{13}\text{C layer}_{\text{initial}})}$$

Kinyangi reported that roots of a contemporary maize crop sampled from both field (4 months) and container (5 weeks) studies contributed up to 50% of the measurable soil C in surface layers of Oxisol aggregates, 4–8 mm across. C contents in surface layers of these same soil aggregates declined nearly 23% when fertilizer P was applied to the soil. This study demonstrated a high rate of C deposition by plant roots and transitory C mineralization that controls soil aggregate stability and P availability.

Observations that *Tithonia diversifolia* extracts P from P-deficient Oxisols when soil aggregates are penetrated by very fine roots were confirmed by identifying lower extractable P levels from the centers of peeled soil aggregates that had been penetrated by *Tithonia* roots.^[14] External soil layers of aggregates from P-deficient Oxisols were removed by the SAE method of peeling soil aggregates^[4] and extractable P levels measured. Consequently, greater quantities of soil P can be gleaned from the centers of rigid soil aggregates of P-deficient Oxisols and transferred to food crops via decomposing *Tithonia* plant roots and aboveground residues.^[14]

MICRODENSITOMETRY OF SOIL AGGREGATES

Investigations of interior regions of soil aggregates could lead to a greater understanding of the complex porosities that sequester soil C, microbial communities, and ions. Greater knowledge of water and ion fluxes into and through soil aggregates will lead to knowledge-based opportunities for better managing these complex living centers of the soil. Microdensitometry evaluations of root and soil aggregate interactions are routine analytical options at the GeoSoilEnviro CARS (Consortium for Advanced Radiation Sources) beamline of Sector 13 at the Advanced Photon Source (APS), located within Argonne National Laboratory (ANL) near Chicago, Illinois. Computer algorithms rendered 2-D (Fig. 1) and 3-D images for observations. This image and other synchrotron X-ray images of soil aggregates generate new spatial concepts that expand our knowledge of intra-aggregate and root interfaces. For example, root penetration of these very rigid soil aggregates, 1 mm across, of Kenyan Oxisols permitted root extraction of previously unavailable P from the internal regions of these aggregates.^[14] These types of root-induced macropores (RIMs) also increase macropore connectivities between adjacent soil aggregates, greatly increasing bypass flow through adjacent soil aggregates. Although the APS synchrotron used in these studies is quite large, immobile, and less expensive, new desktop models for routine microtomographic evaluations of porous materials are becoming commercially available. As microdensitometry evaluations of soil aggregates become less expensive, more specific internal porosity comparisons can

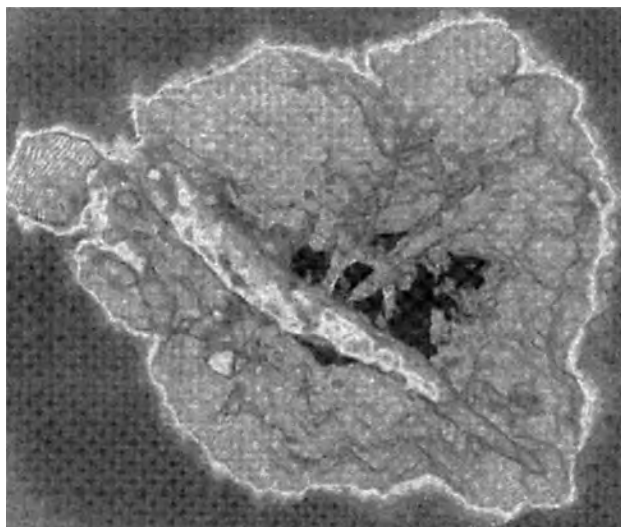


Fig. 1 The 2-D image of several electronic slices through microtomographic reconstructions of an undisturbed prairie grass soil aggregate, 1 mm across. Notice the diagonal RIM from top left to bottom right is the remnant of a dehydrated root that contributed root exudates and other plant C substrates to the soil aggregate. This soil aggregate image was produced by K.M. Kemner, B. Lai, and H.-R. Lee at the X-ray beam in Sector 2 of the APS at ANL. Images were recorded by a CCD camera with a pixel unit cell size 7×6 microns.

be made across different soil types and management systems. These evaluations, although somewhat disruptive, when coupled with X-ray diffraction measurements should lead to greater understanding of the biogeochemical mechanisms controlling root uptake of ions as they are associated with the soil minerals at the root–soil interface.

REFERENCES

- Oades, J.M. Soil organic matter and structural stability: Mechanisms and implications for management. *Plant Soil* **1984**, *76*, 319–337.
- Tisdall, J.M.; Oades, J.M. Organic matter and water-stable aggregates in soils. *J. Soil Sci.* **1982**, *33*, 141–163.
- Dexter, A.R.; Horn, R. Effects of land use and clay content on soil structure as measured by fracture surface analysis. *Z. Pflanzenernahrung Bodenk.* **1998**, *151*, 325–330.
- Smucker, A.J.M.; Santos, D.; Kavdir, Y.; Paul, E.A.; Snider, R. Concentric Gradients within Stable Soil Aggregates, In *Proceedings of the 16th World Congress of Soil Science*, France, August, 1998.
- Jastrow, J.D.; Boutton, W.; Miller, R.M. Carbon dynamics of aggregate-associated organic matter estimated by Carbon-13 natural abundance. *Soil Sci. Soc. Am. J.* **1996**, *60*, 801–807.
- Gale, W.J.; Cambardella, C.A.; Bailey, T.B. Surface residue and root derived carbon in stable and unstable aggregates. *Soil Sci. Soc. Am. J.* **2000**, *64*, 196–201.
- Santos, D.; Murphy, S.L.S.; Taubner, H.; Smucker, A.J.M.; Horn, R. Uniform separation of concentric surface layers from soil aggregates. *Soil Sci. Soc. Am. J.* **1997**, *61*, 720–724.
- Sissoko, F. *Enhancement of Soil Aggregation by the Combined Influences of Soil Wetting and Drying and Root-Microbial Associations*. M.S. Thesis, Michigan State University, East Lansing, 1997; 109 pp.
- Gale, W.J.; Cambardella, C.A.; Bailey, T.B. Root derived carbon and the formation and stabilization of aggregates. *Soil Sci. Soc. Am. J.* **2000**, *64*, 201–207.
- Van Gestle, M.; Ladd, J.N.; Amato, M. Carbon and nitrogen mineralization from two soils of contrasting texture and microaggregate stability: Influence of sequential fumigation, drying and storage. *Soil Biol. Biochem.* **1991**, *23*, 313–322.
- Reid, J.B.; Goss, M.J. Interactions between soil drying due to plant water use and decreases in aggregate stability caused by maize roots. *J. Soil Sci.* **1982**, *33*, 47–53.
- Balesdent, J.; Wagner, G.H.; Mariotti, A. Soil organic matter turnover in long-term field experiments as revealed by carbon-13 natural abundance. *Soil Sci. Soc. Am. J.* **1988**, *52*, 118–124.
- Kavdir, Y. *Distribution of Cover Crop Nitrogen Retained by Soil Aggregates within a Rye-corn Agroecosystem*. Ph.D. Dissertation; Michigan State University, 1999; 157 pp.
- Kinyangi, J. *Carbon, Nitrogen and Phosphorus Sequestration within Soil Aggregates of Maize and Tree-based Agroforestry Systems*. M.S. Thesis; Michigan State University, 2000; 101 pp.
- Santos, D. *Contributions of Roots and Organic Matter to Soil Aggregate Development and Stabilization*. Ph.D. Dissertation; Michigan State University, 1998; 149 pp.
- Blackwood, C.; Smucker, A.J.M.; Paul, E.A. *Microbial Distributions within Concentric Layers of Soil Aggregates*. Abstracts of Papers for the Ecol. Soc. Am., Snowbird, 2000.

Sub-Irrigation

Norman R. Fausey

Soil Drainage Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Columbus, Ohio, U.S.A.

Abstract

Water management has quite a broad scope, including all practices that influence any component of the hydrologic cycle. Within this broad scope are many practices, including things such as cloud seeding to increase precipitation, reservoir management to minimize flood events and to store water for municipal use, the use of plastic mulches to reduce evaporation, and the use of infiltration basins to enhance recharge of groundwater aquifers. Municipal water supply; public safety issues related to flood forecasting, minimization, and urban storm water management; recreational needs; and agricultural production are examples of why water may need to be managed. Agricultural water management practices generally fall into one of following five primary categories: irrigation, drainage, soil erosion control, water supply for animal needs, and waste water disposal. Irrigation involves adding water to assure an adequate supply for crop needs. Irrigation water may be applied on the soil surface or below the soil surface by various methods. One of the methods used to apply water below the soil surface is sub-irrigation.

DEFINITION AND DESCRIPTION OF SUB-IRRIGATION

Sub-irrigation is the practice of adding water to the soil by means of subsurface drains that are also used to drain water from the soil during periods when the soil is too wet. The drains may be open drains (ditches) or closed drains (drainpipes). Water may be supplied from a surface or a subsurface source and is delivered into the subsurface drains and allowed to redistribute within the soil from these subsurface drains. Control structures within the ditches or at the outlet of the closed drains are used to block the water from leaving through the outlet and, thereby, to establish a pressure gradient to cause water to flow from the drains into the soil.

Sub-irrigation is only applicable to areas needing subsurface drainage and having an adequate water supply. These areas typically have high water tables during some times of the year that can be lowered by subsurface drainage. Sub-irrigation depends on being able to reestablish an elevated water table; this requires a substantial amount of water not only to raise the water table, but also to meet the evapotranspiration demand of the crop to maintain the water table at a raised position within the soil.

idea to construct drains to remove excess water from the soil proved successful, the idea of putting water back into the soil through the drains could not have been too far behind. Providing an adequate source of water and a means to move the water against the gravitational gradient would certainly have limited the feasibility prior to the advent of efficient pumping systems. Although not well documented, efforts and progress toward applying sub-irrigation were certainly made, as suggested by an anonymous^[1] quote found in an early publication on drainage, stating: “I want the drains to irrigate with as much as to drain.” An extension bulletin providing guidance for sub-irrigation in Florida was published in 1938,^[2] indicating a recurring early demand for this information in this region. Renfro^[3] summarized and discussed the use of sub-irrigation in the United States up to the mid-1950s. Most of the early applications were on very permeable organic or sandy soils, using open ditches, for high-value crops (vegetables and citrus), and in areas with a readily available water supply, including the Sacramento–San Joaquin Delta in Central California, the Everglades of southern Florida, the San Luis Valley in Colorado, the Flatwoods of the Florida coastal plain, the Cache Valley in northern Utah, the Egin Bench in southern Idaho, and the Great Lakes states.

THE PAST

Agricultural water management using sub-irrigation is not new. It seems reasonable to presume that once the

THE PRESENT

During the past years, agriculture has evolved rapidly. World population pressures, scientific advances, and

changing economic and social values have accelerated a shift in agriculture; diversity has given way to specialization. There are fewer farms and fewer farmers; production per unit of land is greater and is increasing. Risk reduction is a strong driving force in management decisions. Water management has become an important tool for reducing the risk of too much and too little water. This environment has led to a greater awareness of water management options and impacts, and as a result, sub-irrigation has become a topic of increased interest for farmers and researchers.

In 1991, an international conference on sub-irrigation and controlled drainage water management was held in East Lansing, Michigan, U.S.A. The book^[4] that resulted from this conference gives an excellent overview of the status of sub-irrigation around the world. Reports of studies and experience from Canada, China, England, Finland, Italy, the Netherlands, and various locations within the United States are included and illustrate a high level of interest in sub-irrigation.

Sub-irrigation, when properly managed in concert with subsurface drainage, can produce consistently high yields every year regardless of the weather conditions during the growing season. The subsurface drainage function of the system is used to remove excess soil water to assure trafficability for early planting, to lengthen the growing season, and to avoid flooding and lack of oxygen in the root zone, which causes root damage and stunting or death of the plants. The sub-irrigation function of the system is used to avoid deficit water conditions in the root zone, causing stunting and premature senescence of plants. An assured adequate supply of water also allows planning and management for high yields that would not occur when relying on natural rainfall. High yield management involves higher plant populations and more fertilizer application to take full advantage of the available water. With sub-irrigation, yield goals can be raised and still be reached consistently.

Sub-irrigation water management is also beneficial to the environment. An adequate supply of water to meet crop needs encourages maximum growth and, therefore, efficient use of applied nutrients. Nutrients are taken up rather than being left in the soil where they would be subject to being transported to surface water and groundwater as non-point-source pollutants. Sub-irrigation promotes greater production and increases the amount of organic residue returned to the soil. Sustained soil quality and soil health depend on maintaining or increasing the amount of organic matter in the soil. Modern environmental goals also promote the capture and sequestration of carbon in the soil rather than the release of carbon dioxide into the atmosphere. Increasing the soil organic matter content stores more carbon in the soil.

THE FUTURE

Economic and social pressures will likely continue to have a significant impact on agriculture and, consequently, on agricultural water management practices. As society becomes more environmentally aware and sensitive overall, an ethic will emerge for the elimination, or certainly the reduction, of delivery of non-point-source contaminants from agriculture to surface water and groundwater. Some of this has already begun and has resulted in the promotion of uncultivated vegetated corridors along streams designed to slow and filter runoff waters moving to streams. Cost share programs are available in some states to offset the annual loss of production for farmers willing to establish permanent vegetation in these corridors or to assist with the installation of structures to control subsurface drainage discharge to reduce nitrate delivery to streams.

Realistically, it is difficult to control water quality during storm events. Subsurface drainage increases infiltration and decreases surface runoff. The use of sub-irrigation during the summer months would result in less available storage in the soil for rainwater and, therefore, increased runoff, unless the drains are opened quickly and the water table is allowed to fall rapidly ahead of the infiltrating water. Anything that increases runoff encourages the transport of sediments and other pollutants with the runoff water. Water that can be retained in the soil or on the land during storm events will not contribute to runoff and pollutant transport.

Historically, the landscape included more wetland areas where runoff waters were retained or slowed and filtered before reaching streams. Agricultural and cultural development resulted in the loss of many of these wetlands that were also barriers to transportation or breeding grounds for diseases. Removal of the wetlands caused rapid delivery of runoff water to streams resulting in increased flooding and increased transport of sediments and other pollutants to the streams. One way to improve water quality would be to reestablish more wetland areas back into the landscape. Surface runoff and subsurface drainage waters could be directed to these wetlands for treatment and volume reduction before being discharged to streams. Some or all of the water leaving a wetland could be captured/harvested and stored on site in lieu of continuing offsite and downstream. The stored water could meet irrigation or other water supply needs.

Water supply is a critical need for sub-irrigation systems to be feasible and economical. The concept of capture of water during periods when excess water is available and its reuse to meet crop needs during deficit water periods offers an opportunity to make sub-irrigation affordable and practical in many areas. A holistic approach involving the integration of constructed wetlands, water storage facilities, and sub-irrigation of crops offers a unique opportunity to realize

consistent and high crop yields. Such a system will generate more wetland habitat and protect water quality—two of the society's high priority goals. Therefore, the future for sub-irrigation is bright, considering the value to many segments of society to develop and use such integrated systems.

Sub-irrigation as a part of an overall water management plan that protects water quality, increases wetland habitat, and stabilizes crop yields will become the best water management practice.

REFERENCES

1. Anonymous. Tile to irrigate. *Drainage J.* **1890**, 12 (10), 283.
2. Spencer, A.P. Sub-irrigation. *Fla. Agr. Ext. Sta. Bull.* **1938**, 99.
3. Renfro, G. Applying water under the surface of the ground. In *Water: The 1955 Yearbook of Agriculture*; U.S. Government Printing Office: Washington, 1955; 273–278.
4. Belcher, H.W.; D'Itri, F.M. *Sub-irrigation and Controlled Drainage*; Lewis Publishers: Boca Raton, 1995.

Sugarcane Fields: Harvest Systems and Residue Management

Newton La Scala Jr.

Ricardo De Oliveira Bordonal

Eduardo Barretto De Figueiredo

Department of Exact Sciences, College of Agricultural and Veterinarian Sciences,
Sao Paulo State University, Sao Paulo, Brazil

Alan Rodrigo Panosso

Department of Mathematics, College of Engineering of Ilha Solteira,
Sao Paulo State University, Sao Paulo, Brazil

Mara Regina Moitinho

Mariana Marotti Corradi

Department of Exact Sciences, College of Agricultural and Veterinarian Sciences,
Sao Paulo State University, Sao Paulo, Brazil

Abstract

Sugarcane (*Saccharum officinarum*) fields in Southern Brazil have been progressively converted from a burned harvest regime (BH) to a non-burned green mechanized harvest (GH), after which a large amount of crop residue is left on the soil surface. This conversion has resulted in ongoing social, economic, and environmental changes. In this entry, we analyze the aspects of soil physics and chemistry related to this conversion, with special emphasis on carbon dioxide (CO₂) emissions and soil carbon (C) accumulation. We show how small changes in soil C stocks, related to increased soil CO₂ emissions arising from different management options, can impact the C footprint of ethanol. Ethanol is an important option as a replacement for fossil fuels. The results we present and the mechanisms we discuss reveal an optimum land management strategy once the sugarcane areas are converted from a BH to a GH system, and the soil is left covered with sugarcane crop residue. Very different from a bare soil surface scenario, we predict that this is a more appropriate method for sustainable and long-term needs. Here, we maintain that when assessing the long-term effect of sugarcane-based ethanol production, consideration should be given to the long-term effects on the agroecosystem, particularly the soil being the basis for sustainability.

INTRODUCTION

Increased prices and the environmental impact associated with fossil fuel consumption have driven interest in bio-fuels, and concerns regarding food, energy, and the environment have risen considering world production.^[1,2] An estimated 138.8 billion liters of ethanol will be required in 2021 to meet the world's energy demands,^[3] and sugarcane-based ethanol can play an important role in satisfying this need.^[4] Sugarcane is one of the most widely used feedstocks for ethanol production, and Brazil is the largest producer. Production is concentrated in the south-central region and accounts for 90% of national production.^[5] In 2014, sugarcane plantations occupied a total area close to 9.1 million hectares and produced an estimated 28.4 billion liters of ethanol.^[6]

The National Alcohol Program (*Proálcool*) was established in Brazil in 1975 with the purpose of reducing petroleum imports using sugarcane-based ethanol; the environmental benefits were recognized soon after.^[7] This fact, together with the use of hydropower, makes the Brazilian energy matrix far cleaner when compared with most countries where fossil fuels are the main energy source.^[8,9] For example, the widespread introduction of “FlexFuel” vehicles (i.e., designed to run on both gasoline and ethanol) decreased gasoline use in Brazil reducing emissions by 27.5 Tg CO₂eq in 2003.^[10]

Biofuel sustainability has been discussed widely, particularly with regard to preharvest burning during sugarcane agricultural production.^[11,12] There has been much debate on the long-term effects of gaseous emissions from burning on human health.^[13,14] This has resulted in a push from

preharvest burning toward green mechanized harvesting, particularly in São Paulo, even though this management practice has some disadvantages such as using more diesel and fewer workers. Here, we discuss the main aspects associated with the sources and sinks of carbon dioxide (CO₂) emissions under contrasting sugarcane production scenarios, i.e., burned harvest regime (BH) vs. non-burned green mechanized harvest (GH), in Southern Brazil.

Greenhouse Gas (GHG) Balance in Sugarcane Production: Burned vs. Green Harvest

Typically in Brazil, the conversion from a BH to a GH system leaves approximately 10–20 tons of dry matter ha⁻¹ yr⁻¹ on the soil surface after harvest, which represents close to 4.2 and 8.4 tons of carbon (C) ha⁻¹ yr⁻¹, respectively. On the other hand, intensive tillage applied during crop renewal, normally on a five-year planting cycle, can promote high CO₂-C emissions and reduce the potential for long-term soil C sequestration.^[15,16] Reduced tillage and crop rotation during sugarcane renewal are the strategies employed to increase the soil C stock and improve the GHG balance;^[12] a potential GHG mitigation of up to 70.9 Tg CO₂eq has been reported for São Paulo by 2050.^[17]

Change in harvesting methods and leaving crop residues on the soil surface have a huge impact on soil properties: maintaining high soil water content, reducing the soil temperature, reducing erosion, and providing nutrients to the subsequent ratoons.^[18–21] Soil CO₂-C emissions have been studied in sugarcane areas under BH and GH regimes to identify better management practices to reduce emissions

and increase soil C. Some results point to additional emissions of only 250 kg CO₂-C 25 days after tillage as an effect of removing the straw and 1100 kg CO₂-C because of intensive tillage for crop renovation in GH areas.^[16] However, many authors maintain that crop residues should be removed and used as a source of the so-called “second-generation ethanol” or even to generate surplus electricity. There is huge potential for additional power generation in sugarcane straw that can be recovered after harvest. One of the most important Clean Development Mechanism projects applied in Brazil, which uses only sugarcane bagasse after juice extraction, is responsible for 7% of the current electricity generation,^[22] and this is estimated to reach 14% in 2020.^[23]

Approximately, 90% of GHG emissions associated with ethanol production come from the agricultural phase rather than the industrial process.^[5] Thus, when the C footprint of Brazilian ethanol is considered, it is important to take into account the type of management that is applied during sugarcane cultivation (i.e., harvested by burning or non-burning; Fig. 1). De Figueiredo and La Scala Jr^[24] showed that the total CO₂ equivalent emitted in the field during the agricultural production of sugarcane would come mostly from synthetic fertilizers and diesel, and in the case of the BH scenario, the burning of residues accounted for approximately 941 kg CO₂eq ha⁻¹ yr⁻¹. Synthetic nitrogen (N) fertilizers represent approximately 27% of the total GHG balance in GH areas where because of N immobilization by microorganisms, it became necessary to increase their application by 30% during the first years after conversion. Long-term studies reported that small reductions in the amounts of N fertilizer are possible if green mechanized



Fig. 1 Sugarcane production areas, for the last 35 years, indicating the location of GH and BH, located in Guariba city, São Paulo, Brazil. The geographical coordinates are 20° 24' S and 48° 09' W, with mean elevation of 550 m above sea level.

Source: From Alan Rodrigo Panosso; unpublished data.

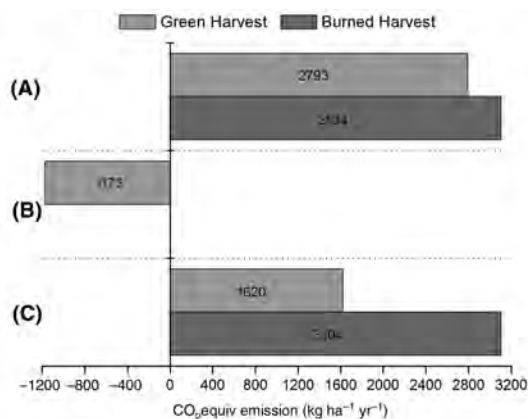


Fig. 2 Estimation of GHG emission (in kg CO₂eq ha⁻¹ yr⁻¹) associated with sugarcane production, considering 1 ha under burned and green harvest systems: A – subtotal: GHG emissions from synthetic N fertilizer, vinasse, filter cake, residues burning, harvest residues, liming, and diesel; B – C sequestration; C – total GHG balance.

Source: Adapted from De Figueiredo and La Scala Jr^[24] ©2011.

harvesting is adopted for a long period, e.g., more than 15 years.^[25] Furthermore, sugarcane yield is strongly dependent on N fertilization, which is itself responsible for most of the GHG emissions. The work done by De Figueiredo and La Scala Jr^[24] showed subtotals of GHG emissions of 2793 and 3104 kg CO₂eq ha⁻¹ yr⁻¹ in GH and BH, respectively. Moreover, a modest soil C accretion of 320 kg C ha⁻¹ yr⁻¹ (1173 kg CO₂eq ha⁻¹ yr⁻¹) in the GH scenario is important and should certainly be accounted for. This then results in a total GHG balance of 1620 and 3104 kg CO₂eq ha⁻¹ yr⁻¹ in the GH and BH scenarios, respectively (Fig. 2).

Recent publications have also shown the high potential of soil C accumulation in sugarcane fields under green cane management.^[17,26] Cerri et al.^[27] reported that the maintenance of crop residues on the soil surface after harvest could accumulate an average of 1.5 Mg C ha⁻¹ yr⁻¹. However, the simple conversion of sugarcane fields from a BH to a GH system does not guarantee significant increases in soil C stocks over time. Studies have shown that tillage events can result in huge soil CO₂-C emissions during sugarcane field renovation, conducted typically every 5 to 6 years after planting.^[21]

The Effect of Sugarcane Crop Residues on Soil CO₂ Emission

Based on the direct and indirect GHG emission inventories from sugarcane production, our estimations show that when determining the budget in agricultural systems, changes in the soil C stock and its relation to soil CO₂ emissions should be a major consideration. Minor changes in soil C stocks in agricultural areas can impact the total C balance

positively or negatively, with soil C accumulation or loss. This could easily be the highest CO₂ source or, on the other hand, may offset other emissions associated with diesel and synthetic N fertilizer use.

Any management change in agricultural areas results in variations in soil CO₂ emissions. A typical emission of 2 μmol CO₂ m⁻² s⁻¹ would result in 27,372 kg CO₂ ha⁻¹ or 7464 kg CO₂-C ha⁻¹ over a year. Soil CO₂ emission is a large component directly related to soil C transfer to the atmosphere by transference of soil organic matter gains due to crop cycling. It is interesting to notice that just a 10% reduction in the mean magnitude of the soil CO₂ emission, for instance, from 2 to 1.8 μmol CO₂ m⁻² s⁻¹, would reduce soil emissions by approximately 2737 kg CO₂ ha⁻¹ yr⁻¹ (= 746 kg CO₂-C ha⁻¹ yr⁻¹). This is a large component compared with other emission sources, including the soil itself as a possible C sink. As presented in our GHG balance (Fig. 2), savings associated with a 10% reduction in soil CO₂ emission would be equivalent to twice that of what we assumed to be the soil C sequestration once sugarcane areas are converted from BH to GH. This emission reduction of 0.2 μmol CO₂ m⁻² s⁻¹ is something that authors do not pay much attention to, as measuring such a difference between treatments in field experiments, even in agricultural areas, is not easy because of the high temporal and particularly spatial variability of soil CO₂ emissions. Reduction in soil CO₂ emissions, sometimes unnoticed, could save soil C within 1 year and is enough to compensate for other emission sources associated with ethanol production, “zeroing” its C footprint at the end of the production chain.

A set of experiments conducted in the sugarcane areas of Southern Brazil indicate contrasting aspects of the soil CO₂ emission in green vs. burned harvested areas. Panosso et al.^[28] showed a mean soil CO₂ emission in an Eutrustox to be 39% higher in a BH compared with a neighboring GH plot 70 days after harvest (e.g., 2.87 and 2.06 μmol CO₂ m⁻² s⁻¹ for BH and GH, respectively; Fig. 1). Spatial variability models of soil CO₂ emission in both sites indicate a more homogeneous emission pattern from the GH compared with the BH, and CO₂ emission being less influenced by rain, for example. De Figueiredo et al.^[16] in an investigation into the influence of soil management (tillage and liming on an Haplustult) on soil CO₂ emission after sugarcane field renovation showed the important effect of additional soil CO₂-C emission of 253 kg C ha⁻¹ (= 927.7 kg CO₂ ha⁻¹) in a no-till plot. This was induced by the simple removal of sugarcane residues from soil surface, over a period of 3 weeks during which time the soil was left bare, after the replanting of the sugarcane field.

The same effect was observed by Corradi et al.,^[29] whereby the removal of the crop residue from the soil surface resulted in an immediate increase in soil CO₂ emissions. During an experiment carried out over 50 days, it was observed that the emissions from residue-covered soils

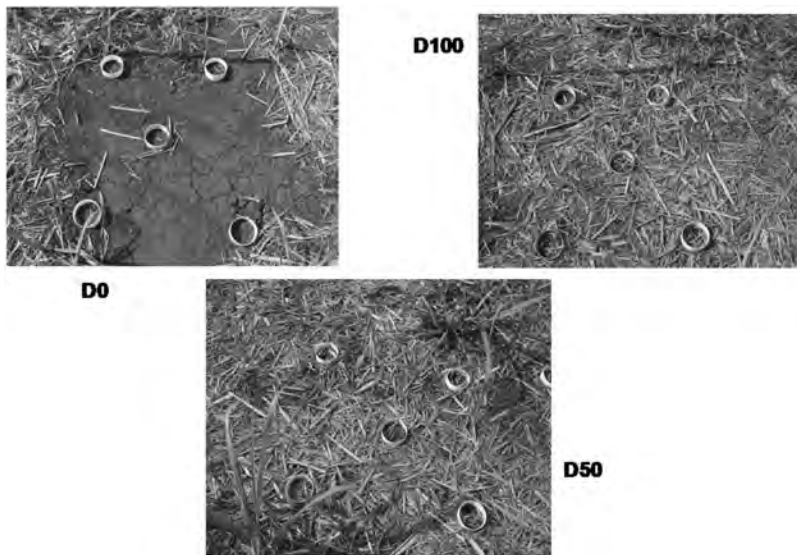


Fig. 3 Experimental plots under different crop residues density on soil surface: 0% (D0), 50% (D50), and 100% (D100) of crop residues.

Source: From Mariana Marotti Corradi.

with densities of 3 (D50) and 6 (D100) t ha⁻¹ were less than those from the soil from which the residues were removed (D0; Fig. 3). During the 50 days after the sugarcane residues were removed, CO₂ emissions were higher in D0 than in D100 and D50 (Fig. 4), despite changes in soil temperature and moisture associated with precipitation events. Total emissions, which are defined by the area below the curves (Fig. 4), were as high as 553.6 ± 47.2 g CO₂ m⁻² (mean and standard error) in D0 and as low as 384.7 ± 31.7 g CO₂ m⁻² in D50, which are comparable to D100 total emission. This difference represents an additional emission close to 460.7 kg CO₂-C ha⁻¹ after the complete removal of the sugarcane residue in just 50 days.

Several studies have evaluated the spatial variability of CO₂ emission in areas of bare soil^[30] and those with vegetation.^[31,32] The spatial variation of soil CO₂ emission is

governed mainly by physical, chemical, and biological soil properties, such as soil organic matter content,^[33] microorganism population density in the rhizosphere,^[34] dry bulk soil density,^[30,31] soil pH,^[35] and the available phosphorus.^[36] Furthermore, on a temporal scale, CO₂ emission is controlled by properties that vary over time such as soil moisture,^[37] air-filled soil porosity,^[38] water-filled soil porosity,^[39] soil oxygenation,^[40] and soil temperature.^[37] Schwendenmann et al.^[41] observed spatial relationships between CO₂ emissions, soil C, and phosphorus concentration; temporal variations in the flow of CO₂ were triggered by the soil water content, and the CO₂ flux decreased during the periods when the soil received high precipitation. Brito et al.^[32] also studied soil respiration in sugarcane areas and reported the influence of different topographic positions in the spatial patterns of soil CO₂ emissions.

Soil moisture demonstrates a relation with soil CO₂ emission; increasing it facilitates microbial activity but decreases the oxygen content inside the soil. Linn and Doran^[39] studied the effect of water-filled porosity on soil CO₂ emissions and found that higher emissions occurred when this was approximately 60%. But in an experiment conducted by Corradi et al.,^[29] the temperature and moisture of soil were also affected by the presence (or removal) of sugarcane crop residues, and in general, covered fields had lower soil temperatures and higher soil moisture levels. These parameters do not entirely explain the higher emissions observed in the D0 plot compared to the D100 over the study period.

According to Razafimbelo et al.,^[42] the green mechanized harvest method contributed to soil C accumulation in upper soil layers. In long-term, an increase in the organic C content of 20% and 15% was reported at the 0–5 cm and 0–10 cm layers, respectively. Furthermore, the incorporation of sugarcane crop residues after soil tillage resulted in higher rates of CO₂ emission, since a greater contact between the soil and crop residues, in combination with

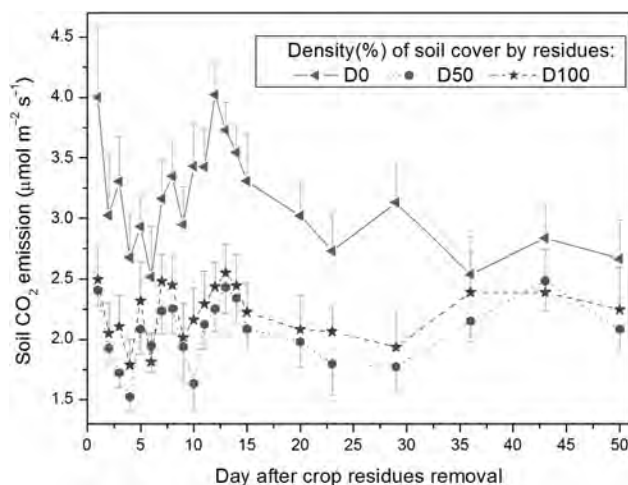


Fig. 4 Mean (± half of standard error) of CO₂ emission in the studied days.

Source: Adapted from Corradi, Panosso, et al.^[29] ©2013.

increased aeration and temperature, enhances soil microbial activity.^[15] Assessing the apparent contributions of resident soil C and aboveground crop residue C input to CO₂-C losses, Pes et al.^[43] observed that the crop residue C input was the primary driver of the short-term soil CO₂-C losses during the crop establishment period, and the mineralized resident soil C released by soil tillage gave a limited contribution to increased soil CO₂ emission.

Some Conceptual Aspects on Soil CO₂ Emission

Soil CO₂ emissions can be modeled as they are directly related to the decay of C in soil organic matter, which follows a first-order decay as presented in Eq. 1:

$$\frac{dC_{\text{soil}}}{dt} = -kC_{\text{soil}} \quad (1)$$

where dC_{soil}/dt is the soil C decay in soil organic matter, k is the so-called “decay factor” (time⁻¹), and C_{soil} is the soil C stock (kg m⁻²). The soil CO₂-C emission ($F_{\text{CO}_2-\text{C}}$) should be proportional to soil C decay in organic matter. This is presented in Eq. 2:

$$F_{\text{CO}_2-\text{C}} \propto -\frac{dC_{\text{soil}}(t)}{dt} \quad (2)$$

By relating Eqs. 1 and 2, we derive Eq. 3, which shows that the soil CO₂-C emission (or CO₂ emission) is proportional to the decay factor vs. the C stock in the soil:

$$F_{\text{CO}_2-\text{C}} \propto kC_{\text{soil}} \quad (3)$$

The decay factor (k) is a key issue that should be noted. This factor is related to soil temperature and soil moisture, oxygen content inside the soil, the resilience of the soil organic matter (related to clay content), and how labile the soil organic matter is, this being mostly related to the C/N ratio. Considering Eq. 3, it is theoretically possible to have soils where the C stocks are high, but emissions are low, because of lower values of k in those soils. On the other hand, even soils with low C stocks could have larger CO₂-C emissions due to high k factors. In Eq. 3, it is possible to see that the ratio between $F_{\text{CO}_2-\text{C}}$ and C_{soil} gives an indication of how stable the soil system is in terms of the stability of its soil C. This is an important aspect, particularly when considering the management options available for increasing agricultural production.

With few works published so far, most of the studies relating soil CO₂ emission to soil properties do not pay attention to the soil oxygen content.^[40] This is one of the most important factors influencing microbial activity and hence CO₂ (or CO₂-C) emission. Furthermore, there is growing evidence of the synergistic effect of applying vinasse in the presence of straw on soil GHG emissions. After increasing the amount of straw on the soil surface, the highest emissions (approximately 3000 kg CO₂eq ha⁻¹ yr⁻¹) were reported for ratoon cane treated with vinasse.^[44]

Leaving or removing crop residues on the soil surface seems to interfere with the soil CO₂ emissions so significantly that this could be a strong component in terms of GHG emission.

Here, we emphasize that keeping crop residues on a soil surface could have an important and immediate effect on the soil oxygen content, reducing soil CO₂ emissions and soil organic matter decay. Our results indicate a significant reduction in CO₂ emissions just by leaving sugarcane residues on the soil surface. This is apparently not directly governed by soil moisture. Empirical equations, based on the impact of crop residue removal on soil C stocks, predict that removing residues to produce additional ethanol would have an impact on soil C stocks and GHG emissions in the United States.^[45] This is another indication that the conservation of soil C in agricultural areas should rely on the presence of sugarcane residues on the soil surface, which is an important aspect to be considered in further studies.

CONCLUSION

When considering the soil C budget associated with sugarcane ethanol production in Brazil, it is important to take into account how the soil is managed, with an emphasis on soil C stocks. Small changes in these stocks could significantly increase the C footprint of sugarcane-based ethanol or decrease it close to zero. The soil C budget is a dynamic system in space and time and depends strongly on soil CO₂ emissions, which are shown to be related to the presence or absence of crop residues on the soil surface. When addressing the sustainability of biofuel production, this factor should be taken into account and kept as an option for further studies, as leaving crop residues on soil surface seems to mitigate further emissions associated with agricultural operations.

ACKNOWLEDGMENT

The authors would like to thank Fundação de Amparo à Pesquisa do Estado de São Paulo – FAPESP (contract grant number 2008/58187-0) and Conselho Nacional de Desenvolvimento Científico e Tecnológico – CNPq (contract grant number 142232/2012-2), Brazil, for financial support.

REFERENCES

1. Tilman, D.; Socolow, R.; Foley, J.A.; Hill, J.; Larson, E.; Lynd, L.; Pacala, S.; Reilly, J.; Searchinger, T.; Somerville, C.; Williams, R. Energy. Beneficial biofuels—the food, energy, and environment trilemma. *Science* **2009**, *325* (5938), 270–271.
2. Popp, J.; Lakner, Z.; Harangi-Rákos, M.; Fári, M. The effect of bioenergy expansion: Food, energy, and environment. *Renew. Sust. Energ. Rev.* **2014**, *32*, 559–578.

3. Goldemberg, J.; Mello, F.F.C.; Cerri, C.E.P.; Davies, C.A.; Cerri, C.C. Meeting the global demand for biofuels in 2021 through sustainable land use change policy. *Energ. Policy* **2014**, *69*, 14–18.
4. Leal, M.R.L.V.; Nogueira, L.A.H.; Cortez, L.A.B. Land demand for ethanol production. *Appl. Energ.* **2013**, *102*, 266–271.
5. Seabra, J.E.A.; Macedo, I.C.; Chum, H.L.; Faroni, C.E.; Sarto, C.A. Life cycle assessment of Brazilian sugarcane products: GHG emissions and energy use. *Biofuel. Bioprod. Bior.* **2011**, *5* (5), 519–532.
6. Conab.. Acompanhamento da safra brasileira de cana-de-açúcar - safra 2014/2015. Companhia Nacional de Abastecimento (CONAB) 2014. Available at <http://www.conab.gov.br> (accessed July 2014).
7. Da Silva, J.G.; Serra, G.E.; Moreira, J.R.; Gonçalves, J.C.; Goldemberg, J. Energy balance for ethyl alcohol production from crops. *Science* **1978**, *201* (4359), 903–906.
8. Cerri, C.E.P.; Sparovek, G.; Bernoux, M.; Easterling, W.E.; Melillo, J.M. Tropical agriculture and global warming: impacts and mitigation options. *Sci. Agr.* **2007**, *64* (1), 83–99.
9. Goldemberg, J.; Coelho, S.T.; Guardabassi, P. The sustainability of ethanol production from sugarcane. *Energ. Policy* **2008**, *36* (6), 2086–2097.
10. Macedo, I.C. *Sugar Cane's Energy: Twelve Studies on Brazilian Sugar Cane Agribusiness and Its Sustainability*, 2nd Ed.; União da Agroindústria Canavieira do Estado de São Paulo (UNICA): São Paulo, 2005; 233 pp.
11. Tsao, C.-C.; Campbell, J.E.; Mena-Carrasco, M.; Spak, S.N.; Carmichael, G.R. Increased estimates of air-pollution emissions from Brazilian sugar-cane ethanol. *Nat. Climate Change* **2011**, *2* (1), 53–57.
12. Bordonal, R.O.; De Figueiredo, E.B.; La Scala, N. Greenhouse gas balance due to the conversion of sugarcane areas from burned to green harvest, considering other conservationist management practices. *Glob. Change Biol. Bioenerg.* **2012**, *4* (6), 846–858.
13. Galdos, M.; Cavalett, O.; Seabra, J.E.A.; Nogueira, L.A.H.; Bonomi, A. Trends in global warming and human health impacts related to Brazilian sugarcane ethanol production considering black carbon emissions. *Appl. Energ.* **2013**, *104*, 576–582.
14. Silveira, H.C.S.; Schmidt-Carrijo, M.; Seidel, E.H.; Scapulatempo-Neto, C.; Longatto-Filho, A.; Carvalho, A.L.; Reis, R.M.V.; Saldiva, P.H.N. Emissions generated by sugarcane burning promote genotoxicity in rural workers: A case study in Barretos, Brazil. *Environ. Health* **2013**, *12* (1), 87.
15. Silva-Olaya, A.M.; Cerri, C.E.P.; La Scala, N., Jr.; Dias, C.T.S.; Cerri, C.C. Carbon dioxide emissions under different soil tillage systems in mechanically harvested sugarcane. *Environ. Res. Lett.* **2013**, *8* (1), 015014.
16. De Figueiredo, E.B.; Panosso, A.R.; Reicosky, D.C.; La Scala, N. Short-term CO₂-C emissions from soil prior to sugarcane (*Saccharum* spp.) replanting in southern Brazil. *Glob. Change Biol. Bioenerg.* **2015**, *7*, 316–327.
17. Bordonal, R.O.; De Figueiredo, E.B.; Aguiar, D.A.; Adami, M.; Rudorff, B.F.T.; La Scala, N. Greenhouse gas mitigation potential from green harvested sugarcane scenarios in São Paulo State, Brazil. *Biomass Bioenerg.* **2013**, *59*, 195–207.
18. Leal, M.R.L.V.; Galdos, M.V.; Scarpere, F.V.; Seabra, J.E.A.; Walter, A.; Oliveira, C.O.F. Sugarcane straw availability, quality, recovery and energy use: A literature review. *Bio-mass Bioenerg.* **2013**, *53*, 11–19.
19. Gava, G.J.C.; Trivelin, P.C.O.; Oliveira, M.W.; Penatti, C.P. Crescimento e acúmulo de nitrogênio em cana-de-açúcar cultivada em solo coberto com palhada. *Pesqui. Agropecu. Bras.* **2001**, *36*, 1347–1354.
20. Fortes, C.; Vitti, A.C.; Otto, R.; Ferreira, D.A.; Franco, H.C.J.; Trivelin, P.C.O. Contribution of nitrogen from sugarcane harvest residues and urea for crop nutrition. *Sci. Agr.* **2013**, *70* (5), 313–320.
21. La Scala, N., Jr.; De Figueiredo, E.B.; Panosso, A.R. A review on soil carbon accumulation due to the management change of major Brazilian agricultural activities. *Braz. J. Biol.* **2012**, *72* (3), 775–785.
22. Agência Nacional de Energia Elétrica (ANEEL). Banco de Informações de Geração: Matriz de Energia Elétrica, 2014. Available at <http://www.aneel.gov.br/aplicacoes/capacidade-debrasil/capacidadebrasil.cfm> (accessed May 2014).
23. União da Indústria de Cana-de-Açúcar (UNICA). Bioeletridade 2011. Available at <http://www.unica.com.br/documentos/publicacoes/bioeletridade> (accessed August 2014).
24. De Figueiredo, E.B.; La Scala, N., Jr. Greenhouse gas balance due to the conversion of sugarcane areas from burned to green harvest in Brazil. *Agr. Ecosyst. Environ.* **2011**, *141*, 77–85.
25. Robertson, F.A.; Thorburn, P.J. Management of sugarcane harvest residues: Consequences for soil carbon and nitrogen. *Aust. J. Soil Res.* **2007**, *45* (1), 13–23.
26. Galdos, M.V.; Cerri, C.C.; Cerri, C.E.P. Soil carbon stocks under burned and unburned sugarcane in Brazil. *Geoderma* **2009**, *153*, 347–352.
27. Cerri, C.C.; Galdos, M.V.; Maia, S.M.F.; Bernoux, M.; Feigl, B.J.; Powlson, D.; Cerri, C.E.P. Effect of sugarcane harvesting systems on soil carbon stocks in Brazil: An examination of existing data. *Eur. J. Soil Sci.* **2011**, *62* (1), 23–28.
28. Panosso, A.R.; Marques, J., Jr.; Pereira, G.T.; La Scala, N., Jr. Spatial and temporal variability of soil CO₂ emission in a sugarcane area under green and slash-and-burn managements. *Soil Till. Res.* **2009**, *105* (2), 275–282.
29. Corradi, M.M.; Panosso, A.R.; Martins Filho, M.V.; La Scala, N., Jr. Crop residues on short-term CO₂ emissions in sugarcane production areas. *Eng. Agric.* **2013**, *33*, 699–708.
30. Teixeira, D.B.; Panosso, A.R.; Cerri, C.E.P.; Pereira, G.T.; La Scala, N., Jr. Soil CO₂ emission estimated by different interpolation techniques. *Plant Soil* **2011**, *345*, 187–194.
31. Xu, M.; Qi, Y. Soil-surface CO₂ efflux and its spatial and temporal variations in a young ponderosa pine plantation in northern California. *Glob. Change Biol.* **2001**, *7* (6), 667–677.
32. Brito, L.F.; Marques, J., Jr.; Pereira, G.T.; Souza, Z.M.; La Scala, N., Jr. Soil CO₂ emission of sugarcane fields as affected by topography. *Sci. Agr.* **2009**, *66*, 77–83.
33. Lal, R. Challenges and opportunities in soil organic matter research. *Eur. J. Soil Sci.* **2009**, *60* (2), 158–169.
34. Duiker, S.W.; Lal, R. Carbon budget study using CO₂ flux measurements from a no-till system in central Ohio. *Soil Till. Res.* **2000**, *54*, 21–30.

35. Reth, S.; Reichstein, M.; Falge, E. The effect of soil water content, soil temperature, soil pH-value and the root mass on soil CO₂ efflux – A modified model. *Plant Soil* **2005**, *268* (1), 21–33.
36. Duah-Yentumi, S.; Rønn, R.; Christensen, S. Nutrients limiting microbial growth in a tropical forest soil of Ghana under different management. *Appl. Soil Ecol.* **1998**, *8*, 19–24.
37. Yuste, J.C.; Baldocchi, D.D.; Gershenson, A.; Goldstein, A.; Misson, L.; Wong, S. Microbial soil respiration and its dependency on carbon inputs, soil temperature and moisture. *Glob. Change Biol.* **2007**, *13* (9), 2018–2035.
38. Panosso, A.R.; Marques, J., Jr.; Milori, D.M.B.P.; Ferraudo, A.S.; Barbieri, D.M.; Pereira, G.T.; La Scala, N., Jr. Soil CO₂ emission and its relation to soil properties in sugarcane areas under Slash-and-burn and Green harvest. *Soil Till. Res.* **2011**, *111* (2), 190–196.
39. Linn, D.M.; Doran, J.W. Effect of water-filled pore-space on carbon-dioxide and nitrous-oxide production in tilled and nontilled soils. *Soil Sci. Soc. Am. J.* **1984**, *48* (6), 1267–1272.
40. Chen, X.; Dhungel, J.; Bhattarai, S.P.; Torabi, M.; Pendergast, L.; Midmore, D.J. Impact of oxygation on soil respiration, yield and water use efficiency of three crop species. *J. Plant Ecol.* **2011**, *4* (4), 236–248.
41. Schwendenmann, L.; Veldkamp, E.; Brenes, T.; O'Brien, J.; Mackensen, J. Spatial and temporal variation in soil CO₂ efflux in an old-growth neotropical rain forest, La Selva, Costa Rica. *Biogeochemistry* **2003**, *64* (1), 111–128.
42. Razafimbelo, T.; Barthès, B.; Larré-Larrouy, M.C.; De Luca, E.F.; Laurent, J.Y.; Cerri, C.C.; Feller, C. Effect of sugarcane residue management (mulching versus burning) on organic matter in a clayey Oxisol from southern Brazil. *Agr. Ecosyst. Environ.* **2006**, *115*, 285–289.
43. Pes, L.Z.; Amado, T.J.C.; La Scala, N., Jr.; Bayer, C.; Fiorin, J.E. The primary sources of carbon loss during the crop-establishment period in a subtropical Oxisol under contrasting tillage systems. *Soil Till. Res.* **2011**, *117*, 163–171.
44. Carmo, J.B.; Filoso, S.; Zotelli, L.C.; De Sousa Neto, E.R.; Pitombo, L.M.; Duarte-Neto, P.J.; Vargas, V.P.; Andrade, C.A.; Gava, G.J.C.; Rossetto, R.; Cantarella, H.; Neto, A.E.; Martinelli, L.A. Infield greenhouse gas emissions from sugarcane soils in Brazil: effects from synthetic and organic fertilizer application and crop trash accumulation. *Glob. Change Biol. Bioenerg.* **2013**, *5* (3), 267–280.
45. Liska, A.J.; Yang, H.; Milner, M.; Goddard, S.; Blanco-Canqui, H.; Pelton, M.P.; Fang, X.X.; Zhu, H.; Suyker, A.E. Biofuels from crop residue can reduce soil carbon and increase CO₂ emissions. *Nat. Climate Change* **2014**, *4* (5), 398–401.

Sulfate and Sulfide Minerals

Delvin S. Fanning

*Department of Environmental Science and Technology, University of Maryland,
College Park, Maryland, U.S.A.*

Steven N. Burch

University of Maryland, College Park, Maryland, U.S.A.

Abstract

Sulfide and sulfate minerals occur only in small quantities if at all in most soils in humid regions, although certain sulfate minerals, especially gypsum occur in some abundance in soils of arid regions. Where they do occur, however, they can have big effects upon the chemistry of the soils. When exposed to oxidizing conditions either by natural or by human actions, the sulfides oxidize to form sulfuric acid. Unless sufficient acid-neutralizing substances are present, active acid sulfate soils form with many detrimental consequences.

INTRODUCTION

Sulfide and sulfate minerals^[1] are rare to non-existent in most soils of humid regions, but sulfate minerals commonly occur in some soils of arid regions. Where present, these minerals have strong effects on soil chemical properties. It is important to know about the conditions under which they form in soils and the effects they may have on soil properties. Sulfide minerals form in certain anaerobic soils, where there is a source of sulfur (S) as dissolved sulfate in the soil water, e.g., from seawater in tidal marsh soils classified as *Sulfaquents* and *Sulfihemists* by *Soil Taxonomy*.^[2]

Sulfate minerals, in contrast to sulfides, occur in soils that are more aerobic in nature. They may form from sulfides as part of the overall process of sulfuricization.^[1] Generally, they are not present in soils of humid regions because most sulfate minerals readily dissolve and leach from such soils. However, they are present even in humid regions in most active acid sulfate soils, classified as “Sulfaquepts” and “Sulfudepts” by *Soil Taxonomy*.^[2,3] and in some postactive acid sulfate soils. They are more common in soils of subhumid and arid regions, from which gypsum and other relatively soluble sulfates are not readily leached because of limited water supply.

Awareness of S-bearing minerals in soils is important because the oxidation of sulfide minerals and the hydrolysis of certain sulfate minerals form sulfuric acid. When this acid is not sufficiently neutralized by carbonate or other minerals, it results in the formation of acid sulfate soils with pH < 3.5, which can be extremely detrimental to plant growth. The water that emanates from such soils has the

properties of acid mine drainage and can cause fish kills and other severe environmental problems. For more information about acid sulfate soils and associated problems, see other sections of this encyclopedia.

SULFIDES

The S in sulfide minerals occurs as the sulfide anion, S^{2-} , as the disulfide anion, S_2^{2-} , or less commonly as a polysulfide anion.^[1] The sulfide anion is large compared to the S^{6+} cation of the sulfate anion, to which sulfide can be converted by oxidation (Fig. 1). Iron (Fe) sulfide minerals, highly insoluble under strong anaerobic conditions, are the only sulfides known to form and occur to any significant extent in soils. The sulfides of other heavy metals are also insoluble under strongly anaerobic conditions and may occur in very small quantities in some soils.

The Fe disulfide pyrite, FeS_2 , which crystallizes in the cubic crystal system, is the main sulfide mineral that forms and occurs in soils.^[1] Pyrite can occur in soils as a result of sulfidization in tidal environments, where it commonly occurs as framboids.^[1] Alternatively, it can occur in macroscopic crystals, in soils, in mine spoils from the mining of coal, lignite, and ore deposits. In such cases, it may also be present as marcasite, the less stable orthorhombic polymorph of pyrite. The Eh(pe)/pH conditions under which pyrite—the main Fe sulfide mineral that occurs in soils—is stable relative to certain Fe sulfates and goethite, the most stable $FeOOH$ mineral in soils, are shown in Fig. 2.

Monosulfide minerals such as mackinawite, FeS , and polysulfide minerals such as griegite and Fe_3S_4 occur in

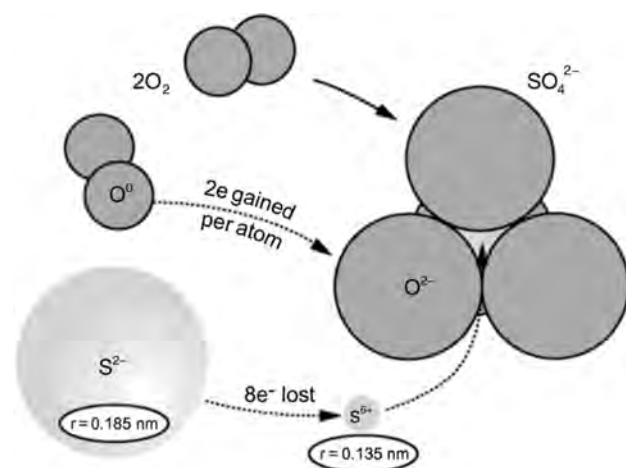


Fig. 1 Idealized oxidation of sulfide anion to produce sulfate anion, where the sizes of the atoms and ions are shown in proportion to their expected sizes in nature.

Source: Adapted from Fanning, Rabenhorst, et al.^[1]

small quantities in some soils undergoing sulfidization,^[1] particularly where the levels of Fe in the soil materials occur in excess relative to the amount of S.^[4] In spite of their presence in low quantities, these minerals have a strong black pigmenting effect on soil materials in which they occur, because of their occurrence in dispersed extremely fine particles. Their presence can be detected by the evolution of foul smelling hydrogen sulfide gas from the materials upon the addition of (dilute) hydrochloric acid, whereas no such evolution occurs with the disulfide minerals.

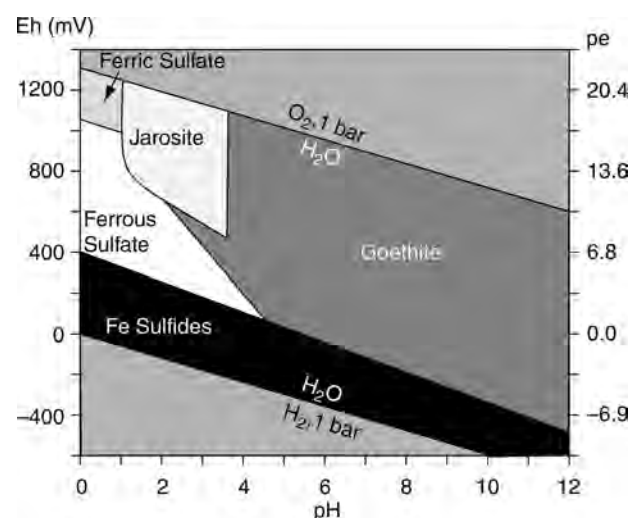


Fig. 2 Idealized Eh(pe)/pH diagram for the Fe-S-O system showing mineral phases that might be expected to be stable under various conditions.

Source: Adapted from Fanning, Rabenhorst, et al.^[1]

SULFATES

Many sulfate minerals are very soluble and behave as salts. However, others such as jarosite and barite are essentially insoluble and can remain in strongly leached soils for many millennia. Fe and aluminum sulfate salts form sulfuric acid upon hydrolysis to form minerals such as goethite, schwertmannite, and jarosite.^[1,5]

Gypsum

Gypsum, $\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$, is essentially the only calcium sulfate mineral that occurs extensively in soils. In arid regions, the quantities in some soil horizons may be sufficient for the recognition of the *gypsic* or *petrogypsic* diagnostic horizons and of the *Gypsis* suborder of Aridisols of *Soil Taxonomy*.^[2,3]

Gypsum forms in acid sulfate and other soils upon the reaction of sulfuric acid with calcium carbonate (CaCO_3) and is also a by-product of industrial neutralization of waste sulfuric acid with CaCO_3 .^[1] It is also readily dissolved and reprecipitated, which enables it to move in response to wetting and drying of soils. Sometimes, gypsum is deliberately added to soils, as a calcium source, e.g., to enhance the growth of certain plants (peanuts) or for the reclamation of soils that contain high levels of exchangeable Na.

Jarosite

Jarosite, $\text{KFe}_3(\text{SO}_4)_2(\text{OH})_6$, and natrojarosite, $\text{NaFe}_3(\text{SO}_4)_2(\text{OH})_6$, form in certain active acid sulfate soils. They are distinctive minerals in soils because of their pale yellow color. Jarosites show the present or past occurrence of acid sulfate soil conditions and, in combination with a soil pH measured in water of 3.5 or less, their presence can enable the identification of a “sulfuric horizon” as defined by *Soil Taxonomy*.^[2,3] Jarosite forms in soils under acid sulfate conditions, when the soil pH is sufficiently low and the Eh/pe is sufficiently high (Fig. 2).

Jarosite in soils is highly insoluble as long as the Eh/pe remains sufficiently high for the ferric, Fe^{3+} , form of Fe to be stable. Because of its durability in soils, it may enable the dating of old postactive acid sulfate soils by potassium-argon or $^{40}\text{Ar}/^{39}\text{Ar}$ (argon/argon) methods,^[1] although soil scientists have to employ this technique. Chemical reducing conditions can convert the ferric Fe of jarosite to the ferrous form and make the mineral soluble, like with Fe oxide and oxyhydroxide minerals.

Relatively Soluble Fe Sulfates

Ferrous sulfate minerals, e.g., rozenite, $\text{FeSO}_4 \cdot 4\text{H}_2\text{O}$ (usually white in color), or melanterite, $\text{FeSO}_4 \cdot 7\text{H}_2\text{O}$ (usually aqua blue in color), or other ferrous sulfate minerals depending on humidity conditions^[1] can form in/on soils when ferrous sulfate-rich solutions (see ferrous sulfate

stability field of Fig. 2) are desiccated. These minerals, or the dissolved ions of them, oxidize and hydrolyze readily to form sulfuric acid and Fe (hydr)oxide minerals and are big contributors to the formation of acid drainage waters in some environments.^[1] Relatively soluble ferric sulfate minerals, such as copiapite, $\text{Fe}_{14}\text{O}_3(\text{SO}_4)_{18} \cdot 63\text{H}_2\text{O}$ and coquimbite, $\text{Fe}_2(\text{SO}_4)_3 \cdot 9\text{H}_2\text{O}$, form and behave in a similar way, although the Fe is already largely oxidized in these minerals.^[1] Halotrichite, a ferrous aluminum sulfate mineral, $\text{FeAl}_2(\text{SO}_4)_4 \cdot 22\text{H}_2\text{O}$, can also behave in a similar manner.

Sodium and Magnesium Sulfates

A number of sodium (Na), e.g., mirabilite, $\text{Na}_2\text{SO}_4 \cdot 10\text{H}_2\text{O}$, and magnesium (Mg), e.g., hexahydrite, $\text{MgSO}_4 \cdot 6\text{H}_2\text{O}$, and mixed Na and Mg, e.g., bloedite, $\text{Na}_2\text{Mg}(\text{SO}_4)_2 \cdot 4\text{H}_2\text{O}$, minerals occur on certain salinized soils such as in North Dakota and Saskatchewan. It is suspected that the S for the formation of these minerals has been released from sulfide minerals deeper in the overall soil-geologic columns upon which these minerals occur.^[1]

Barite

Barite, BaSO_4 , occurs in a few soils.^[1] The origin of this extremely white and very insoluble mineral in soils is poorly understood, although it occurs in some soils that have experienced sulfuricization.^[1] Color pictures and electron micrographs showing the appearance of this mineral in soils are given by Carson et al.^[6]

CONCLUSION

Sulfide and sulfate minerals occur only in small quantities if at all in most soils in humid regions, although certain sulfate minerals, especially gypsum occurs in some abundance in soils of arid regions. Where they do occur, however, they can have big effects upon the chemistry of the soils. Sulfide minerals, especially pyrite, form in tidal marsh soils and in tidal sediments, and they occur in some geologic sediments accumulated under these conditions.

When exposed to oxidizing conditions either by natural or by human actions, e.g., ditching or construction activities, the sulfides oxidize to form sulfuric acid. Unless sufficient acid neutralizing substances are present active acid sulfate soils form with many detrimental consequences as identified elsewhere in this encyclopedia. Sulfate minerals form under oxidizing conditions. Some sulfate minerals are very soluble and are readily leached from soils or are transformed to other minerals. However, certain sulfate minerals, e.g., jarosite and barite, are very insoluble and can remain in soils for long periods of time to attest to acid sulfate soil condition that occurred in the early stages of the formation of some upland soils. The Eh-pe/pH conditions under which certain sulfide and sulfate minerals are stable and are shown in an Eh-pe/pH stability diagram.

REFERENCES

1. Fanning, D.S.; Rabenhorst, M.C.; Burch, S.N.; Islam, K.R.; Tangren, S.A. Sulfides and sulfates. In *Soil Mineralogy with Environmental Implications*; Dixon, J.B., Schulze, D.G., Eds.; SSSA: Madison, 2002; 229–260.
2. Soil Survey Staff. In *Soil Taxonomy*, 2nd Ed.; U.S. Dept. Agric. Handbook 436; U.S. Government Printing Office: Washington, 1999.
3. Soil Survey Staff. In *Keys to Soil Taxonomy*, 9th Ed.; U.S. Dept. Agric., Natural Resources Conservation Service: Washington, 2003. Available at http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_051544.pdf.
4. Rabenhorst, M.C.; James, B.R. Iron sulfidization in tidal wetlands. In *Bio-mineralization Processes of Iron and Manganese*; Skinner, C.W., Fitzpatrick, R.W., Eds.; Catena Verlag: Cremlingen-Destedt, 1992; 203–217.
5. Alpers, C.N.; Jambor, J.L.; Nordstrom, D.K. Sulfate minerals: Crystallography, geochemistry, and environmental significance. *Reviews in Mineralogy and Geochemistry Series*; The Mineralogical Society of America: Washington, 2000; Vol. 40.
6. Carson, C.D.; Fanning, D.S.; Dixon, J.B. Alfisols and ultisols with acid sulfate weathering features in Texas. In *Acid Sulfate Weathering*; Kittrick, J.A., Fanning, D.S., Hossner, L.R., Eds.; Soil Sci. Soc. Am. Spec. Publ. No. 10; SSSA: Madison, 1982; 127–146.

Sulfur

Ewald Schnug

Silvia H. Haneklaus

Elke Bloem

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

Soil fertility is generally defined as the ability of soils to yield crops and is therefore closely intertwined with plant nutrient cycles. The soil sulfur (S) cycle is driven by biological and physico-chemical processes, which affect flora and fauna. The organic matter pool is an important source and sink for S, though its contribution to the mineral nutrition of high yielding crops has been overestimated in the past. Under temperate conditions, it is the spatiotemporal variation of physico-chemical soil properties, which control the plant-available sulfate-S content in the soil via the access of plant roots to S-rich groundwater or capillary ascending porous water. Therefore, soil methods determining plant-available S status will always only deliver information for that instant ignoring relationships with the plant S status or crop yield. A more promising way to evaluate the S supply is a site-specific S budget, which includes information about geomorphology, texture, climatic data, and crop type and characteristics of the local soil water regime.

INTRODUCTION

The major soil properties and external sulfur sources affecting the amount of plant-available sulfate in the soil are shown in Fig. 1.

For a comprehensive understanding of the relationship between sulfur (S) and soil fertility, it is necessary to identify the main factors and processes controlling the plant-available S pool and to appraise the ecotoxicological significance of S within the context of a limiting factor rather than a pollutant. Detailed reviews about the role of S in agroecosystems already exist^[1–9] and will provide further information, not explicitly mentioned here.

S IN SOILS

The concentration of S in parent materials ranges from 0.026% to 1% S with igneous rocks $\leq$ metamorphic rocks $\leq$ magmatic rocks of upper continental crust $\ll$ limestones $<$ sedimentary rocks (sulfides) $\leq$ shales $<$ sedimentary rocks (sulfates) $\ll$ coal.^[10–12] The typical range of S in agricultural soils of humid and semi-humid regions is 100–500 $\mu\text{g g}^{-1}$ or 0.01–0.05% S. This equals 224–1120 kg ha^{-1} S in the A_h horizon.^[8] The total S content of soils may be as low as 20 $\mu\text{g g}^{-1}$ (0.002%) in highly leached and weathered soils of humid regions or as high as 35,000 $\mu\text{g g}^{-1}$ (3.5%) in marine marsh soils and up to 50,000 $\mu\text{g g}^{-1}$ (5%) in calcareous and saline soils of arid and semiarid regions.^[8] Tropical soils generally contain low amounts of S due to their low organic matter content.

BIOLOGICAL ASPECTS OF S IN SOILS

Most of the S in terrestrial soils is bound in the organic fraction, which amounts normally to more than 95% of the total S content.^[2,8] Organic S in soils is a heterogeneous mixture of soil organisms, partly decomposed plant material, animal, and microbial residues. Little is, however, known about the composition of individual chemical compounds. Many different approaches have been developed to separate soil organic S into major fractions. The following approaches to identify distinct forms and properties of soil organic matter were made: chemical extraction followed by physico-chemical separation into humic acids, fulvic acids, and humins reactivity with reducing agents in order to separate carbon-bonded S (C–S) and sulfate esters (C–O–S) and physical separation of organo-mineral size fractions and molecular weight fractionation.^[13]

Sulfate-S is rapidly bound in the form of sulfate ester, which is regarded as a short-term source for S.^[14] The incorporation of S into high molecular weight fractions such as humic acids prevents the rapid mineralization of S. In litter, S is predominately C–S to the humin fraction to a level of about 89%. The incorporation of organic S into complex organic substances followed by association with clay minerals can lead to the relative isolation of S from decayed soil microorganisms. This process is considered a physical protection of organic S against degradation,^[15] and it results in a decreased availability of S to plants.

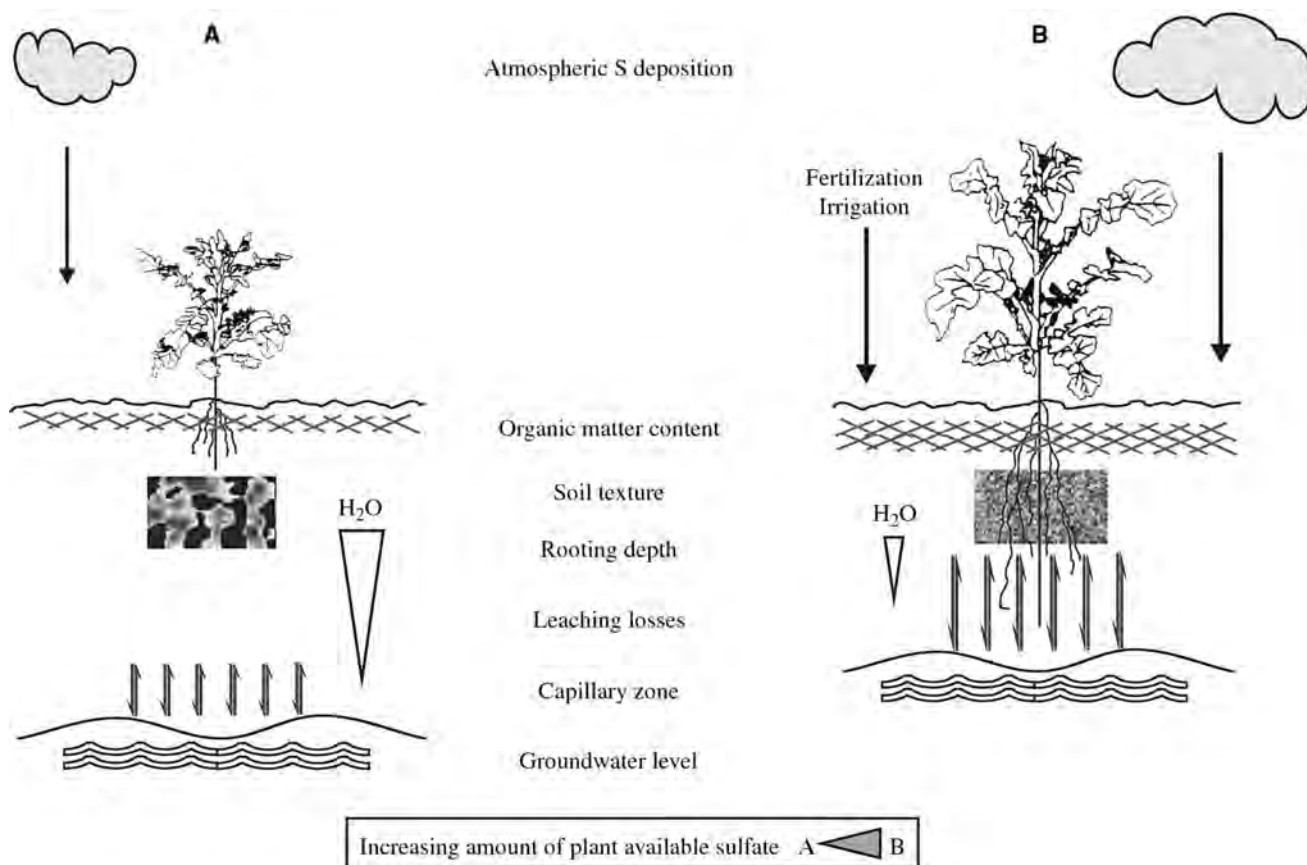


Fig. 1 Overview and efficacy of essential impact factors modifying plant-available sulfate in soils.

S Mineralization

In soils, microbial-mediated processes are mainly responsible for S transformations, so that the factors affecting the microbial activity, such as temperature, moisture, pH, and substrate availability will also influence the processes of mineralization, immobilization, oxidation, and reduction. In aerobic agricultural soils, the main factor in these processes is the release of inorganic, plant-available sulfate from organic matter.

Two types of processes are involved in the mineralization of S: biological and biochemical mineralization.^[16] Biological mineralization is considered to be driven by the microbial need for organic C to provide energy, and S released as sulfate is a by-product of the oxidation of C to carbon dioxide. This process is faster the more recently the organic matter was formed.^[17] Biochemical mineralization relies on the release of sulfate from the sulfate-ester pool through enzymatic hydrolysis.

S in the microbial biomass is actively turned over in soils while comprising only ~2–3% of the total soil S. The turnover of soil microbial biomass is fundamental to the incorporation of sulfate-S into soil organic matter, but quantitative measures to assess this are unavailable.

The contribution of mineralization to the S supply of plants is only small,^[2] because mineralization, immobilization, and possible leaching of S occur concurrently.^[14] The amount of S mineralized within the organic S pool in the soil ranges from 0.5% to 3.1% annually.^[2] The contribution of net mineralization accounts on average for 10–30 kg ha⁻¹ yr⁻¹ S^[18] in soils with carbon contents between 1% and 4%. The studies of Eriksen, Murphy, and Schung^[2] and Bloem^[18] reveal that mineralization is an important, however, not cardinal S pool for plants. High yielding crops cannot satisfy their S demand solely by mineralization and atmospheric S depositions.^[7]

Crops and Crop Rotation

Dead plant material contributes to the organic matter pool but the living plant organic matter that partly remains on the field may be regarded as a storage pool for S, too. S offtake and S demand differ depending on the plant species. Sugarcane has a very low S demand with only 0.3 kg S Mg⁻¹ dry matter yield,^[19] cereals about 1.5–2 kg S Mg⁻¹ of grain and 1–3 kg S Mg⁻¹ of straw yield,^[18] soybean between 4.3 and 8.8 kg S Mg⁻¹,^[20,21] mustard about 16 kg S Mg⁻¹,^[22,23] and oilseed rape up to 20 kg S Mg⁻¹.^[24]

The S content of straw highly influences the mineralization of plant residues. Crops, which leave residues with high S contents, may be considered as catch crops for S and are able to reduce S losses from soil by leaching. Cruciferous crops such as oilseed rape show a high S content of the residual straw,^[24,25] which provide a large amount of rapidly decomposable organic material.^[26] In comparison, plants without a secondary metabolism such as cereals leave smaller amounts of S. Only if the S content of, e.g., barley straw is above 0.13% S, the release of S from the mineralization of straw can be expected.^[27] If the value is below 0.13% S, a net immobilization of S takes place because the microbes need more S for the formation of their microbial biomass. This may exacerbate S deficiency of the following crop, e.g., oilseed rape, particularly on light soils, where macroscopic symptoms may become visible even before winter^[25] because of the net immobilization of S. So far there is no evidence that winter crops benefit during their main growth from the previous crop's organic matter residues in plant with high S.

PHYSICO-CHEMICAL ASPECTS OF S IN SOILS

Sulfate has a high mobility in soils and can be delivered from subsoil or shallow groundwater. The water soluble fraction can be leached, adsorbed, immobilized, or taken up by the plant. Availability of sulfate is therefore more a question of transfer between pools in terms of space and time rather than between biological or chemical systems. The insoluble, organic fraction is only slightly leached and is not directly available for plant uptake.

Soil Water Regime

Under temperate conditions, groundwater is often an important S pool, and there are three ways in which groundwater can contribute to the S nutrition of plants. Firstly, there is a direct S input if the groundwater level is only 1–2 m below the surface. This is sufficient to cover the S requirement of most crops, as plants can utilize the sulfate in the groundwater directly by their root system. An average yielding (3 Mg ha⁻¹ seeds) oilseed rape crop covers more than 50% of its S demand by shallow groundwater or soil water.^[24] Secondly, groundwater contains between 5 and 100 mg L⁻¹ sulfate-S which, if used for irrigation, can supply up to 100 kg ha⁻¹ S to the crop.^[18] Thirdly, the capillary rise of groundwater under conditions of a soil moisture deficit in the upper soil layers leads to a S input. The contribution of this process to the S supply of plants depends on the soil texture within the soil profile and climatic conditions. Generally heavier soils show less frequent S deficiency than lighter ones, as they retain more S-rich pore water. This is proven by a significant relationship between groundwater level and clay content and the sulfate-S content of soils.^[18]

Capillary Rise/Leaching

While rainfall water contains only about 2.5 mg S L⁻¹,^[28] the S content of adsorbed soil water varies between 15 and 100 mg S L⁻¹.^[18] Variation of the sulfate content of water in soil pores is caused by temporal changes of the groundwater level, soil depth, and differences in soil texture.^[18] S-rich water in the soil pore ascends in the soil profile if the evapotranspiration rate is higher than the amount of precipitation.

A higher groundwater level below the surface and a high soil moisture deficit at the lower periphery of the root zone will increase the extent of the capillary rise. Giesel et al.^[29] have calculated an average capillary rise of 0.3 mm d⁻¹ on a sandy soil, 3 mm d⁻¹ on a loamy sand, and 5 mm d⁻¹ on a loamy silt with a groundwater level of 1.5 m below the surface. This means that under extreme conditions on sandy soils with a groundwater level >2 m below the surface virtually, no sulfate will be available to the plant due to capillary rise, but on loamy and silty soils, the S supply of crops may be fully satisfied by capillary water, even if the sulfate concentration of the groundwater is low (10 mg L⁻¹ S).^[18] Spring crops particularly benefit from capillary ascending soil water, as their main growth takes place when capillary rise dominates soil water movements.

The rate of leaching of S under temperate conditions depends on soil type, winter precipitation, and the concentration of sulfate-S in the soil water, during the period of leaching.^[18] S losses through leaching are higher over winter due to higher precipitation rates and the lack of plant uptake. As a result, S stored in soils decreases with increasing precipitation, and on sandy, free-draining soils, all sulfate beneath the capillary zone and rooting depth of the crop may be leached before the beginning of the vegetative period^[18] because of the rapid movement of the leachate through the soil and the low adsorption capacities of such soils. Average sulfate leaching depths are 15 cm with 50 mm of precipitation on a loamy soil and 25 cm on a sandy soil, respectively.^[30] Additionally, leaching is higher on fallow than on cropped soils^[31] and is enhanced by S fertilization. Average leaching losses under temperate conditions are in the range of 30–80 kg ha⁻¹ yr⁻¹ S.^[32,33] In addition to vertical water movements in the soil, lateral fluxes can occur particularly in the landscapes with pronounced differences in geomorphology. These fluxes are the major reason for the high spatiotemporal variability of sulfate in soils.

Soil Compaction

Soil compaction and tillage operations causing soil compaction will also decrease the amount of plant-available S because of the reduced soil volume and consequent impact on soil pore space. Consequently, rooting depth and density are decreased so that S-rich capillary water cannot be used.^[34,35] This effect of soil compaction is supported by

the fact that S deficiency regularly becomes visible first in the headland and along tramlines of fields^[7] because it is sulfate-S from the subsoil that mainly contributes to the S nutrition of crops.^[18]

S EMISSIONS

S-containing atmospheric constituents are principally sulfur dioxide (SO₂), SO₄²⁻, SO₃²⁻, hydrogen sulfide, carbonyl sulfide, dimethyl sulfide, carbon disulfide, and methyl mercaptan. Natural S emissions from biogenic sources and from volcanoes add around 60 Tg S yr⁻¹ to the earth's atmosphere with an upward tendency due to the increasing consumption of fossil fuels in South America, Africa, and Asia.^[9]

Before the industrial revolution, atmospheric S depositions were, on an average, below 10 kg S ha⁻¹ yr⁻¹. As a consequence of increased burning of fossil fuels with industrial development from 1890 to 1980, atmospheric S depositions, mainly as SO₂, increased steadily by 0.47 kg S ha⁻¹ yr⁻¹.^[1] In some rural areas of northern Scotland, however, SO₂ depositions were zero or below 2 kg S ha⁻¹ in the 1970s, while the S input in the midlands of England was about 160 kg S ha⁻¹ at the same time.^[36] At its peak, the negative impacts of SO₂ emissions to humans, plants, soils, and buildings were so serious that S became an unpopular nutrient and was called as the "yellow poison."^[37] The political consequences resulted in Clean Air Act in European countries and North America and the desulfurization of emissions led to a drastic decrease of atmospheric S depositions.

Agriculture adapted and coevolved to increasing S loads. Soil acidification caused by high S deposition required higher amounts of liming materials. Increasing S demands of agricultural crops due to higher yield potential as a result of plant-breeding progress and production technology coincided with increased atmospheric S supply. Additionally, the use of S-containing fertilizers went down nearly to zero.^[25] Consequently, the reduction of atmospheric S deposition had a major effect on the productivity of crops. During the 1990s, macroscopic S deficiency became the most widespread nutrient disorder in northern Europe.^[7]

S FERTILIZERS

The worldwide demand of S fertilizers in 2010 is estimated to be about 11.3 million tons of S.^[38] Numerous S fertilizers and secondary raw material fertilizer products are available for either soil or foliar application in sulfate or elemental form.^[39] Arguments for and against the use of individual products depend on local farming conditions. Organic fertilizers such as manure and slurry contain about 1 and 0.5 kg S Mg⁻¹, respectively.^[40] This equals to 0.07 kg S N⁻¹.

AGROECOLOGICAL ASPECTS OF S

Interaction of Agroecosystems with Other Ecosystems

The S input and off-take in agroecosystems may have a strong influence on neighboring ecosystems. S is often in excess in natural systems, despite the low atmospheric S input because the turnover of S is much lower in these than in agricultural systems. Acid subsoils, e.g., under forest vegetation, and peat soils show pH values <5, which are far below the acidity levels of fertile agricultural soils. Large storage capacities for adsorbed sulfate exist on such sites because the pH value of the soil is lower than the zero point of charge, resulting in the hydration of metal oxides and thus a positive surface.^[41] Although adsorbed sulfate plays only a minor role in the direct S nutrition of agricultural crops, it may contribute positively to the S balance of the whole surrounding landscape. Natural vegetation, land without plant production and forests show a positive S balance even when atmospheric inputs are low.^[2] Agricultural crops may benefit from this S pool, if groundwater reservoirs of both ecosystems are connected. This also means, however, that with an increasing share of agricultural farmland in landscapes, the risk of S deficiency overall also increases.

Sustainability

Sustainable agriculture should use soils in such a way that human needs for food or other agricultural goods are realized, and the quality of the environment and the natural resources remain preserved.^[42] The contamination of groundwater with nitrates is a most serious problem. Nitrogen (N) and S are both involved in protein biosynthesis, and a shortage in the S supply of crops also lowers the utilization of applied fertilizer N and thus deteriorates the crop quality. Non-protein N is accumulated in plant parts and besides poor efficiency for N fertilization, and S deficiencies may increase the loss of N from agricultural soils through volatilization and leaching.^[43] On average, each kilogram of S shortage to satisfy the S demand of the crop causes 15 kg of N to be lost to the environment. Such N inputs endanger strongly the stability of natural communities as, e.g., the growth of algae in water bodies.^[44] Correcting S deficiency by fertilization is environmentally safe as sulfate is, in comparison with N, geogenously abundant.

Global Change

Climate is one of the major factors involved in pedogenesis. Soil formation is directly influenced mainly by temperature and water and indirectly via the climate-depending vegetation.^[45] Changes in climate may change soil types, increase erosion, affect element cycles, and increase the release of greenhouse gases. Increased temperature and humidity

accelerate pedogenesis. Thus, global change would allow soils in the northern hemisphere to proceed faster through the individual stages of development and/or degradation associated with their individual soil series.^[46] Higher temperatures accelerate the decomposition of organic materials and thus decrease the organic matter content of soils. At the same time, the net mobilization of S might increase while the organic S pool decreases. This effect is expected to be more pronounced in cultivated than in range soils.^[47] But at the same time under more humid climate conditions organic matter tends to accumulate in soils.^[47] Therefore, an expected adverse effect of global warming on soil organic matter might be at least partially compensated in those areas where this coincides with increasing humidity.^[48] With increasing carbon accumulation, the other growth-limiting elements, such as N, S, and P, may dilute relatively,^[49] which might result in decreased soil fertility.

Plant Health

Owing to the fact that higher anthropogenic S inputs in the past decades enabled plants to adapt to increasing environmental stress, the decline in the S supply within only one decade might have serious consequences for the stability of the ecosystems.^[49] S metabolism provides several efficient mechanisms by which plants are able to tackle abiotic (e.g., xenobiotics and increasing surface ozone levels) and biotic (e.g., pests and diseases) stress, particularly via the glutathione metabolism that again is closely related to the S supply of the plants.^[3] Other mechanisms involved in response to plant pathogens include the production of S-containing compounds in the secondary metabolism of the agriculturally important *Brassica* species, the release of volatile S compounds, the production of S-rich proteins, the localized deposition of elemental S, and the production of phytochelatin, which detoxify heavy metals by forming complexes.^[50,51]

Certain diseases (e.g., light leaf spot in oilseed rape) occur more frequently particularly in areas with low S input in Europe,^[52] and an improved knowledge of the significance of S metabolites in crop resistance to diseases will be beneficial for the improvement of S fertilizer strategies and could therefore minimize the input of pesticides.

ACKNOWLEDGMENT

The authors cordially thank Dr. Kerr C. Walker (SAC, Aberdeen) for his linguistic efforts on this entry.

REFERENCES

1. Dämmgen, U.; Walker, K.C.; Grünhage, L.; Jäger, H.-J. The atmospheric sulphur cycle. In *Sulphur in Agroecosystems*; Schnug, E., Ed.; Kluwer Academic Publishers: Dordrecht, 1998; 75–114.
2. Eriksen, J.; Murphy, M.D.; Schnug, E. The soil sulphur cycle. In *Sulphur in Agroecosystems*; Schnug, E., Ed.; Kluwer Academic Publishers: Dordrecht, 1998; 39–74.
3. Hell, R.; Rennenberg, H. The plant sulphur cycling. In *Sulphur in Agroecosystems*; Schnug, E., Ed.; Kluwer Academic Publishers: Dordrecht, 1998; 135–174.
4. Howarth, R.W.; Stewart, J.W.B. The interactions of sulphur with other element cycles in ecosystems. In *Sulphur Cycling on the Continents: Wetlands, Terrestrial Ecosystems and Associated Water Bodies, SCOPE 48*; Howarth, R.W., Stewart, J.W.B., Ivanov, M.V., Eds.; John Wiley & Sons: Chichester, 1992; 67–79.
5. Howarth, R.W.; Stewart, J.W.B.; Ivanov, M.V., Eds. *Sulphur Cycling on the Continents: Wetlands, Terrestrial Ecosystems and Associated Water Bodies, SCOPE 48*; John Wiley & Sons: Chichester, 1992; 350.
6. Janzen, H.H.; Ellert, B.H. Sulfur dynamics in cultivated, temperate agroecosystems. In *Sulfur in the Environment*; Maynard, D.G., Ed.; Marcel Dekker Inc.: New York, 1998; 11–94.
7. Schnug, E.; Haneklaus, S. Diagnosis of sulphur nutrition. In *Sulphur in Agroecosystems*; Schnug, E., Ed.; Kluwer Academic Publishers: Dordrecht, 1998; 1–38.
8. Stevenson, F.J. *Cycle of Soil. Carbon, Nitrogen, Phosphorus, Sulfur, Micronutrients*; John Wiley & Sons: New York, 1986; 380.
9. Whelpdale, D.M. An Overview of the atmospheric sulphur cycle. In *Sulphur Cycling on the Continents: Wetlands, Terrestrial Ecosystems and Associated Water Bodies, SCOPE 48*; Howarth, R.W., Stewart, J.W.B., Ivanov, M.V., Eds.; John Wiley & Sons: Chichester, 1992; 5–26.
10. Bowen, H.J.M. *Trace Elements in Biochemistry*; Academic Press: London, 1966.
11. Friend, J.P. The global sulfur cycle. In *Chemistry of the Lower Atmosphere*; Rasool, S.I., Ed.; Plenum Press: New York, 1973; 177–201.
12. Wedepohl, K.H. Chemical fractionation in the sedimentary environment. In *Origin and Distribution of the Elements*; Ahrens, L.H., Ed.; Pergamon Press: London, 1968; Vol. 30, 999–1016.
13. Anisimova, M.; Haneklaus, S.; Schnug, E. Significance of sulfur for soil organic matter. In *Sulfur Nutrition and Sulfur Assimilation in Higher Plants*; Brunold, C., Rennenberg, H., De Kok, L.J., Stulen, I., Davidian, J.-C., Eds.; Paul Haupt Publishers: Berne, 2000; 239–244.
14. Ghani, A.; McLaren, R.G.; Swift, R.S. The incorporation and transformations of ³⁵S in soil: Effect of soil conditioning and glucose or sulfate addition. *Soil Biol. Biochem.* **1993**, *25*, 327–335.
15. Eriksen, J.; Lefroy, R.D.; Blair, G.J. Physical protection of soil organic S studied by extraction and fractionation of soil organic matter. *Soil Biol. Biochem.* **1995**, *27*, 1011–1016.
16. McGill, W.B.; Cole, C.V. Comparative aspects of cycling of organic C, N, S and P through soil organic matter. *Geoderma* **1981**, *26*, 267–286.
17. Ghani, A.; McLaren, R.G.; Swift, R.S. Mobilization of recently-formed soil organic sulphur. *Soil Biol. Biochem.* **1993**, *25*, 1739–1744.
18. Bloem, E. Schwefel- bilanz von agrarökosystemen unter besonderer berücksichtigung hydrologischer und bodenphysikalischer standorteigenschaften. *Landbauforschung Voelkenrode* **1998**, *192*, 1–156.

19. Katyal, J.C.; Sharma, K.L.; Srinivas, K. Sulphur in Indian agriculture. In Proceedings of TSI/FAI/IFA Symposium on sulphur in balanced fertilisation, KS-2/1-KS-2/12, 1997.
20. Aulakh, M.S.; Pasricha, N.S.; Azad, A.S. Phosphorus-sulphur interrelationships for soybeans on phosphorus and sulphur deficient soils. *Soil Sci.* **1990**, *150*, 705–709.
21. Nambiar, K.K.M.; Gosh, A.B. Highlights of research of a long-term fertilizer experiment in India (1971–82). Technical Bulletin No. 1; Longterm Fertilizer Experiment Project, IARI, 1984; 100 pp.
22. Jain, G.L.; Sahu, M.P.; Somani, L.L. Balanced fertilization programme with special reference to secondary and micro-nutrients nutrition of crops under intensive cropping. In Proceeding of FAI/NR Seminar, Jaipur, 1984; 147–174.
23. Aulakh, M.S.; Pasricha, N.S.; Sahota, N.S. Yield, nutrient concentration and quality of mustard crops as influenced by nitrogen and sulphur fertilizers. *J. Agr. Sci.* **1980**, *94*, 545–549.
24. Schnug, E. *Quantitative und Qualitative Aspekte der Diagnose und Therapie der Schwefelversorgung von Raps (Brassica napus L.) Unter Besonderer Berücksichtigung Glucosinolatarter Sorten*; DSC thesis, Christian-Albrechts-University, Kiel, Germany, 1988.
25. Schnug, E.; Haneklaus, S. Sulphur deficiency in brassica napus-biochemistry-symptomatology-morphogenesis. *Landbauforschung Völkenrode* **1994**, *144*, 1–31.
26. Wu, J.A.G.; O'Donnell, Z.L.; Syers, J.K. Microbial growth and sulphur immobilization following the incorporation of plant residues into soil. *Soil Biol. Biochem.* **1993**, *25*, 1567–1573.
27. Chapman, S.J. Barley straw decomposition and S immobilisation. *Soil Biol. Biochem.* **1997**, *29*, 109–114.
28. Dämmgen, U.; Grünhage, L.; Küsters, A.; Jäger, H.J. Konzentrationen von luftinhaltsstoffen. I. Criteria pollutants. *Landbauforschung Voelkenrode* **1996**, *170*, 196–221.
29. Giesel, W.; Renger, M.; Strebel, O. Berechnung des kapillaren aufstiegs aus dem grundwasser in den wurzelraum unter stationären bedingungen. *Z. Pflanzenern.und Bodenkde* **1972**, *132*, 17–30.
30. Kumar, V.; Karwasra, S.P.S.; Singh, M.; Dhankar, J.S. An evaluation of the sulphur status and crop responses in the major soils of Haryana, India. *Sulphur. Agr.* **1994**, *18*, 23–26.
31. Kirchmann, H.; Pichlmayer, F.; Gerzabek, M.H. Sulfur balance and sulfur-34 abundance in a long-term fertilizer experiment. *Soil Sci. Amer. J.* **1996**, *59*, 174–178.
32. Mansfeld, T. *Schwefeldynamik Von Böden Des Dithmarscher Speicherkoogs Und Der Bornhoeveder Seenkette in Schleswig-holstein*. Dissertation, Univ Kiel SchriftenR Inst Pflanzenern Bodenkde, Kiel, 1994.
33. Preuschoff, M. *Untersuchungen Zur Schwefelversorgung Von Weißohl an Zwei Lößstandorten*. PhD thesis, University Hanover, Verlag Ulrich E. Grauer, Stuttgart, 1995.
34. Singh, B.R. *Effect of Soil Compaction on S Availability to Crop Plants*. Abstracts of the COST Action 829; Fundamental, Agronomical and Environmental Aspects of Sulfur Nutrition and Assimilation in Plants: Goslar, 1998.
35. Unger, P.W.; Kaspar, T.C. Soil compaction and root growth: A review. *Agron. J.* **1994**, *86*, 759–766.
36. Semb, A. Sulfur emissions in Europe. *Atmos. Environ.* **1978**, *12*, 455–460.
37. Boelsche, J. *Das gelbe Gift*; Rowohlt Verlag: Reinbek, 1994.
38. Cecotti, S.; Morris, R.J.; Messick, D. A global overview of the sulphr situation: Industry's background, Market trends, and commercial aspects of sulphur fertilisers. In *Sulphur in Agroecosystems*; Schnug, E., Ed.; Kluwer Academic Publishers: Dordrecht, 1998; 175–202.
39. Paulsen, H.M. Produktionstechnische und Ökologische bewertung der landwirtschaftlichen verwertung von schwefel aus industriellen prozessen. *Landbauforschung Voelkenrode* **1999**, *197*, 1–143.
40. Pedersen, C.A.; Knudsen, L.; Schnug, E. Sulphur Fertilisation. In *Sulphur in Agroecosystems*; Schnug, E., Ed.; Kluwer Academic Publishers: Dordrecht, 1998; 115–134.
41. Curtin, D.; Syers, J.K. Extractability and adsorption of sulfate in soils. *J. Soil Sci.* **1990**, *41*, 305–312.
42. Anon. *Nachhaltiges Deutschland: Wege zu Einer Dauerhaft Umweltgerechten Entwicklung*; Umweltbundesamt Berlin, Erich Schmidt Verlag: Berlin, 1997.
43. Schnug, E. Sulphur nutritional status of European crops and consequences for agriculture. *Sulphur Agr.* **1991**, *15*, 7–12.
44. Wild, A. *Umweltorientierte Bodenkunde*; Spektrum Akademischer Verlag: Heidelberg, 1993.
45. Hugget, R.J. *Geoecology: An Evolutionary Approach*; Routledge: London, 1995.
46. Rogasik, J.; Daemmgen, U.; Luettich, M.; Obenauf, S. Wirkungen physikalischer und chemischer klimaparameter auf bodeneigenschaften und bodenprozesse. *Landbauforschung Voelkenrode* **1994**, *148*, 107–139.
47. Burke, I.C.; Yonker, C.M.; Parton, W.J.; Cole, C.V.; Flach, K.; Schimel, D.S. Texture, climate and cultivation effects on soil organic matter content in U.S. Grassland Soils. *Soil Sci. Soc. Am. J.* **1989**, *53*, 800–805.
48. Schnug, E. Response of plant metabolism to air pollution and global change impact on agriculture. In *Responses of Plant Metabolism to Air Pollution and Global Change*; DeKok, L.J., Stulen, I., Eds.; Backhuys Publishers: Leiden, 1998; 15–22.
49. Schnug, E.; Haneklaus, S. Ecological aspects of plant sulphur supply. In Proceedings of 15th International Congress Soil Science, Acapulco/Mexico, 1994; 5a: Comm. IV: Symposia, 1994; 364–371.
50. Resende, M.L.V.; Flood, J.; Ramsden, J.D.; Rowan, M.G.; Beale, M.H.; Cooper, R.M. Novel phytoalexins including elemental sulphur in the resistance of cocoa (*Theobroma cacao* L.) to verticillium wilt (*Verticillium dahliae* Kleb.). *Physiol. Molec. Plant Pathol.* **1996**, *48*, 347–359.
51. Schnug, E. Significance of Sulphur for the nutritional and technological quality of domesticated plants. In *Sulfur Nutrition and Sulfur Assimilation in Higher Plants*; Cram, W.J., DeKok, L.J., Stulen, I., Brunold, C., Rennberg, H., Eds.; Backhuys Publishers: Leiden, 1997; 109–130.
52. Thomas, J. Watch out for light leaf spot. NIAB Fellows Newsletter; 1994.

Sulfur: Chemical Behavior in the Rhizosphere

Zhengyi Hu

Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China

Silvia H. Haneklaus

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

The rhizosphere is a key zone with respect to the mechanisms of soil nutrient dynamics. The effect of plant growth on most soil nutrients in the rhizosphere has been intensively studied. However, only limited data are available for the effect of plant growth on the chemical behavior of sulfur (S) in the rhizosphere. Such information is nevertheless required to assess agronomic and ecological impacts in relation to the local S cycle. In this entry, the state of knowledge about the chemical behavior of S in the rhizosphere is summarized.

INTRODUCTION

Atmospheric sulfur (S) loads are closely linked to soil quality and the S nutrition of plants. An excess of S may have a negative effect on the soil–plant system, e.g., in flooded paddy soils. Here, S will be reduced to hydrogen sulfide, which hinders plant growth. In contrast, atmospheric S deposition cannot satisfy the agronomic demands in the remote areas of China.

CHEMICAL BEHAVIOR OF S IN THE RHIZOSPHERE

Oxidization of S^0 in the Rhizosphere and Non-Rhizosphere

Elemental S (S^0) is often used as a fertilizer to satisfy the S demand of crops. The S^0 needs to be oxidized to SO_4^{2-} before it becomes plant available. The activity of *Thiobacilli* is highly important for the oxidation of S^0 .^[1] Adding *Thiobacilli* together with S^0 to soil yields a significantly faster oxidation rate than the application of S^0 alone.^[2] Heterotrophic microorganisms are also S^0 oxidizers in soils.^[1,3] Grayston and Germida^[4] isolated 273 bacterial phyla and 70 fungal species from the rhizosphere of canola (*Brassica napus*). From these 273 bacterial isolates, 245 (89.7%) oxidized S^0 to thiosulfate or tetrathionate, and 133 (48.7%) oxidized S^0 to SO_4^{2-} . All 70 fungal isolates oxidized S^0 to SO_4^{2-} . Bacterial isolates showed the highest S^0 oxidation rate (Table 1).

A rhizobag culture experiment demonstrated that the oxidation of S^0 in the rhizosphere varied with respect to

soil moisture content and soil type.^[5] The oxidation rate of S^0 was generally lower under waterlogged conditions than under aerobic conditions (Fig. 1). On a paddy soil originating from limestone, the oxidation rate of S^0 was faster in the rhizosphere of rice than in the non-rhizosphere (Fig. 1). However, these differences were not observed on a paddy soil originating from granite. The reason could be related to different contents of plant available S and/or different microbiological species in the two soils.

Soil Microbial Biomass S in the Rhizosphere and Non-Rhizosphere

The microbial biomass S in agricultural soils varies from 5% to 9%.^[6] Despite its small size, the microbial biomass is a highly active fraction that acts as the driving force behind mineralization–immobilization and oxidation–reduction processes. A rhizobag culture experiment demonstrated that the S content of microbial biomass was 6.3 mg S/kg in the non-rhizosphere and 11.8 mg S/kg in the rhizosphere of rice.^[7] The S content of microbial biomass was up to 72% higher in the rhizosphere of rice than in the non-rhizosphere (Fig. 2). In non-vegetated soil samples, the S content of the microbial biomass was usually significantly lower than in vegetated treatments.^[7]

Variations in the Chemical Behavior of S in the Non-Rhizosphere and Rhizosphere

In a rhizobag experiment, it was demonstrated that the distribution of S fractions in a rhizosphere and a non-rhizosphere soil varied with the crop type (Table 2).

Table 1 Number of S⁰-oxidizing bacterial isolates from the rhizosphere and rhizoplane of canola grown in a growth chamber.

Soil	Area of isolation	Total isolates	Number of isolates producing		
			S ₂ O ₃ ²⁻ /S ₄ O ₆ ²⁻	SO ₄ ²⁻	S ₂ O ₃ ²⁻ /S ₄ O ₆ ²⁻ and SO ₄ ²⁻
Haverhill	Rhizosphere	56	49 (87.5%) ^a	25 (44.6%)	25 (44.6%)
	Rhizoplane	43	42 (97.7%)	30 (69.8%)	30 (69.8%)
Carrot River	Rhizosphere	31	26 (83.9%)	15 (48.4%)	14 (45.2%)
	Rhizoplane	40	38 (95.0%)	20 (50.0%)	20 (50.0%)
Asquith	Rhizosphere	19	18 (94.7%)	7 (36.8%)	7 (36.8%)
	Rhizoplane	32	29 (90.6%)	13 (40.6%)	10 (31.2%)
Laird	Rhizosphere	15	11 (73.3%)	6 (40.0%)	5 (33.3%)
	Rhizoplane	37	32 (86.5%)	17 (45.9%)	13 (35.1%)
Total bacteria		273	245 (89.7%)	133 (48.7%)	124 (45.4%)

^aS oxidizers as percentage of total isolates.

Source: From Grayston & Germida.^[4]

More ester-bonded S was found in the non-rhizosphere of oilseed rape and rice (Table 2). Hu et al.^[8] observed similar results for oilseed rape, wheat, and radish. The reason could be a higher arylsulfatase activity in the rhizosphere as it is this enzyme that catalyzes the decomposition of sulfate esters.^[9] Han and Yoshida^[10] found that arylsulfatase activity was higher in the rhizosphere than in the non-rhizosphere of rice.

Amino acids, such as cysteine and methionine, are the major components of carbon-bonded S.^[11] S-containing amino acids do not accumulate in free forms because they are rapidly degraded in aerobic soils.^[8] Paul and

Schmidt^[12] reported that cysteine and methionine were slightly greater in a rhizosphere than in a non-rhizosphere soil.^[12] Other experiments revealed no significant differences between the two compartments.^[7,10] These results indicate that carbon-bonded S is of minor importance for the S nutrition of crops compared to ester sulfate.

The amount of residual S (non-reducible S by either hydrogen iodide or Raney nickel) was significantly greater in the rhizosphere than in the non-rhizosphere of oilseed rape, but the opposite results were found for rice (Table 2). Other crops such as wheat and radish also showed higher levels of residual S in the rhizosphere.^[10] Rice had a higher

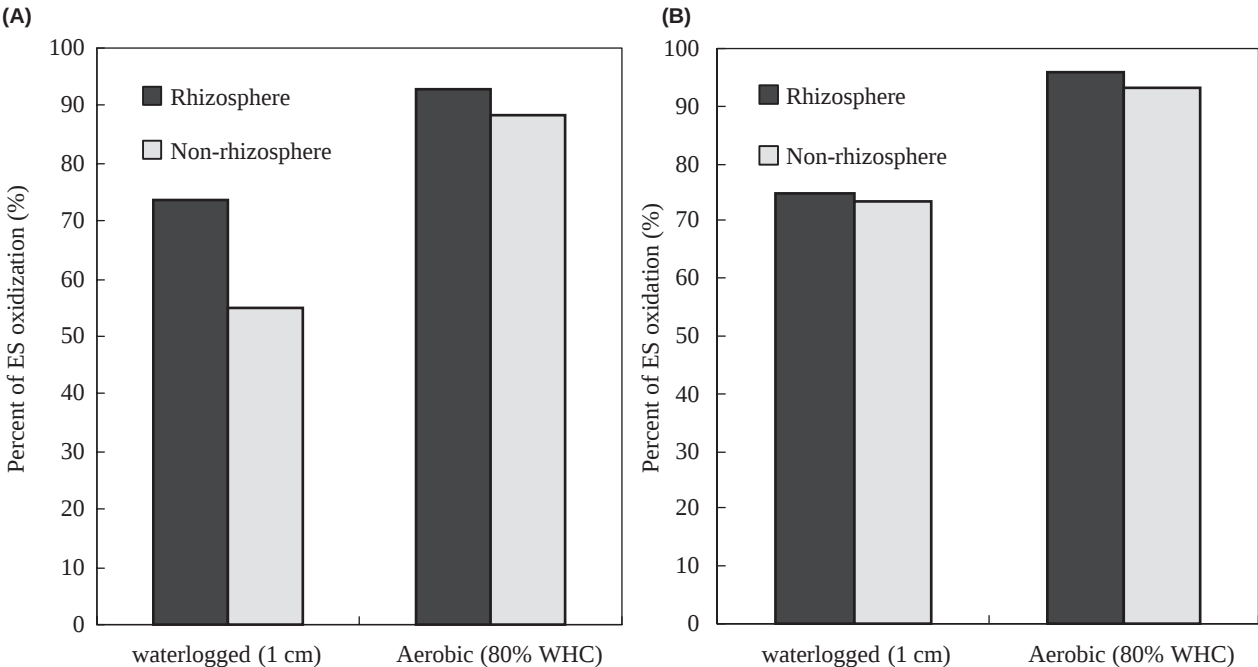


Fig. 1 The oxidation of elemental S (ES) in the rhizosphere of rice with respect to water management and soil type: (A) paddy soil derived from limestone and (B) paddy soil derived from granite.

Source: From Li, Hu, et al.^[5]

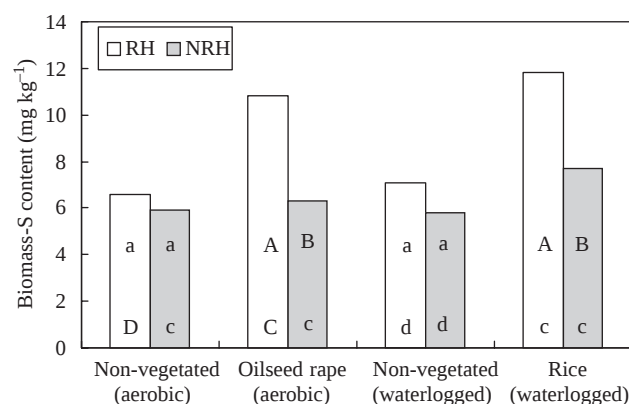


Fig. 2 The concentration of microbial biomass S (MB-S) in the rhizosphere (RH) and non-rhizosphere (NRH). Different letters a and b and A and B indicate significant differences between RH and NRH at $P < 0.05$ and $P < 0.01$ level (Student's *t* test). Different letters c and d and C and D indicate significant differences of MB-S in RH and NRH relative to non-vegetated soils at $P < 0.05$ and $P < 0.01$ level (Student's *t* test), respectively. Source: From Han & Yoshida.^[10]

ability to utilize residual S from the soil, which could be related to its aerenchyma, which transport oxygen from leaves to roots.

The content of soluble SO_4^{2-} -S and adsorbed SO_4^{2-} -S was higher in the rhizosphere than in the non-rhizosphere (Table 2). Enhanced mass flow of SO_4^{2-} -S to the rhizosphere after the mineralization of organic S in the non-rhizosphere is supposedly the reason for this effect. Wind^[13] found that the contents of SO_4^{2-} and $\text{S}_2\text{O}_3^{2-}$ in the rhizosphere of rice were related to rice growth. The same author found more SO_4^{2-} in the rhizosphere (300 $\mu\text{mol/kg}$) of rice than in the non-vegetated (110 $\mu\text{mol/kg}$) treatment (Fig. 3).

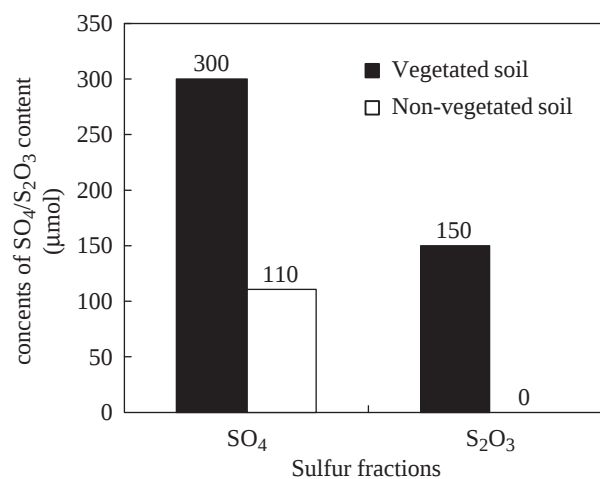


Fig. 3 SO_4^{2-} and $\text{S}_2\text{O}_3^{2-}$ content in the interstitial water from rice vegetated soil and non-vegetated soil. Source: From Wind.^[13]

ECOLOGICAL EFFECTS OF S TRANSFORMATIONS IN THE RHIZOSPHERE

Grayston and Germida^[4] found that canola leaf size and root and pod dry weight at maturity were improved by inoculation with 7 of 14 S-oxidizing bacteria.^[4] The shoot material had higher iron, S, and magnesium contents after inoculation by 2 of 18 bacterial isolates (Table 3). In the case of three isolates, the treatment had a detrimental effect on growth of the fungal pathogens, *Rhizoctonia solani* AG2-1, *R. solani* AG4, and *Leptosphaeria maculans* "Leroy."^[4]

S in nature occurs in valences from -2 to $+6$.^[14] Many types of organic S compounds exist.^[14,15] Internal cycling

Table 2 Contents of different S fractions (mg S/kg) in the rhizosphere (RH), non-rhizosphere (NRH), and the RH to NRH ratios (mean).

Water management	Non-vegetated/cropping	RH/NRH	Total S	S fractions				
				Organic S fractions			Inorganic S fractions	
				Ester-bonded S	Carbon-bonded S	Residual S	Soluble SO_4^{2-} -S	Adsorbed SO_4^{2-} -S
Aerobic condition	Non-vegetated oilseed rape	RH	141.9 a	20.4 a	11.7 a	67.2 a	33.0 a	12.0 a
		NRH	133.9 a	20.1 a	13.8 a	57.2 a	28.8 a	13.7 a
		RH	122.3 a	30.0 b	14.6 a	53.8 A	34.0 a	10.0 a
		NRH	120.4 a	44.5 a	17.3 a	25.8 B	23.0 b	9.8 a
		Ratio	1.02	0.67	0.84	2.08	1.48	1.02
Waterlogged condition	Non-vegetated rice	RH	145.3 a	24.0 a	15.7 a	56.3 a	43.0 a	6.3 a
		NRH	133.0 a	27.0 a	14.4 a	46.4 b	40.4 a	4.8 b
		RH	155.2 a	6.0 B	11.0 a	12.6 B	110.3 A	15.3 A
		NRH	131.2 a	33.6 A	10.7 a	46.9 A	34.7 B	5.3 B
		Ratio	1.18	0.18	1.03	0.27	3.18	2.89

Note: Values followed by different letters a and b and A and B indicate significant differences between RH and NRH at $P < 0.05$ and $P < 0.01$ level (Student's *t* test), respectively.

Source: From Hu, Haneklaus, et al.^[7]

Table 3 S, iron, and magnesium content of canola shoots after seed inoculation with S-oxidizing rhizosphere.

Treatment	Mg (mg/pot)	S (mg/pot)	Fe (μ g/pot)
Control	9.3	21.6	438.5
Isolate no. 13	11.1	23.2	656.6 ^a
Isolate no. 14	11.9 ^a	28.1 ^a	727.1 ^a

^aSignificant increase above control ($P < 0.05$).Source: From Grayston & Germida.^[4]

reactions are responsible for maintaining a biologically available S supply through the mineralization of organic substrates and redox transformation of inorganic species.^[14] The speciation of S in natural organic matter could provide a clear understanding of not only biogeochemical transformations of S but also the role of organic S in the complexation of toxic trace metals.^[16] Here, S-containing functional groups in humic substances may play an important role in complex formation with trace metals.^[16]

CONCLUSION

Basic information about the chemistry of S in the rhizosphere is available, but these data refer to specific soil types and crop species. Besides this, the effects of S on nutrient uptake and heavy metal availability are unclear. Crop response to inoculation with S-oxidizing bacteria depends on various factors, such as the isolates and crop species. Further studies are required to understand the ecological significance of S transformations in the rhizosphere.

REFERENCES

1. Wainwright, M. Sulfur oxidation in soils. *Adv. Agron.* **1984**, 37, 373–378.
2. Fan, X.; Habib, L.; Fleckenstein, J.; Haneklaus, S.; Schnug, E. “*In situ* digestion”: A concept to manage soil phosphate in organic farming. In *Proceedings of the 13th International Fertilizer Symposium, Fertilizers in Context with Resource Management in Agriculture, Tokat, Turkey, 2002*; 219–228.
3. McCaskill, M.R.; Blair, J.G. Particle size and soil texture effects on elemental sulfur oxidation. *Agron. J.* **1987**, 79, 1079–1083.
4. Grayston, S.J.; Germida, J.J. Sulfur-oxidizing bacteria as plant growth promoting rhizobacteria for canola. *Can. J. Microbiol.* **1991**, 37, 521–529.
5. Li, M.; Hu, Z.Y.; Zhang, L.G. Oxidation of elemental sulphur at the rhizosphere of rice and its impacts on nutrients uptake. Unpublished data.
6. Wu, J.; O'Donnell, A.G.; He, Z.L.; Syers, J.K. Fumigation-extraction method for the measurement of soil microbial biomass-S. *Soil Biol. Biochem.* **1994**, 26, 117–125.
7. Hu, Z.Y.; Haneklaus, S.; Wang, S.P.; Xu, C.K.; Cao, Z.H.; Schnug, E. Comparison of mineralization and distribution of soil sulfur fractions in the rhizosphere of oilseed rape and of rice. *Commun. Soil Sci. Plant Anal.* **2003**, 34 (15 and 16), 2243–2257.
8. Hu, Z.Y.; Yang, Z.H.; Xu, C.K.; Haneklaus, S.; Cao, Z.H.; Schnug, E. Effect of crop growth on the distribution of soil sulfur fractions in the rhizosphere. *J. Plant Nutr. Soil Sci.* **2002**, 165, 249–254.
9. Fitzgerald, J.W. Naturally occurring organic sulfur compounds in soils. In *Sulfur in the Environment, Part II, Ecological Impact*; Nriagu, J.O., Ed.; Wiley: New York, 1978; 391–443.
10. Han, K.W.; Yoshida, T. Sulfur mineralization in rhizosphere of lowland rice. *Soil Sci. Plant Nutr.* **1982**, 28, 379–387.
11. Freney, J.R. Forms and reactions of the organic sulfur compounds in soils. In *Sulfur in Agriculture*; Tabatabai, M.A., Ed.; ASA: Madison, 1986; 207–232.
12. Paul, E.A.; Schmidt, E.L. Formation of free amino acids in rhizosphere and nonrhizosphere soil. *Soil Sci. Soc. Am. Proc.* **1961**, 25, 359–362.
13. Wind, T. Sulfur compound, potential turnover of sulfate and thiosulfate, and number of sulfate-reducing bacteria in planted and unplanted paddy soil. *FEMS Microbiol. Ecol.* **1995**, 18 (4), 257–266.
14. Hu, Z.Y.; Xu, C.K. Soil sulfur and environmental quality. In *Behaviors of Chemical Substances in Soils and Environmental Quality*; Chen, H.M., Ed.; Scientific Publishing Company: Beijing, 2002; Chapter 10, 283–307.
15. Morra, M.J.; Fendorf, S.E.; Brown, P.D. Speciation of sulfur in humic and fulvic acids using X-ray absorption near-edge structure (XANES) spectroscopy. *Geochim. Cosmochim. Acta* **1997**, 16 (3), 683–688.
16. Xiao, K.; Weesner, F.; Bleam, W.F.; Bloom, P.R.; Skyllberg, U.L.; Helmke, P.A. XANES studies of oxidation state of sulfur in aquatic and soil humic substances. *Soil Sci. Soc. Am. J.* **1998**, 62, 1240–1246.

Sulfur: Nutrition

Silvia H. Haneklaus

Elke Bloem

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Abstract

Sulfur (S) is a major plant nutrient and its deficiency not only impairs crop productivity and quality but also interferes with environmental demands. Even lesser known is the significance of the S nutritional status for improving the natural resistance of crops against diseases. Under humid conditions, the plant available soil S content is not an appropriate indicator of its availability to crops because of its high spatio-temporal variability even at microscale. In comparison, a comprehensive S balance, which implies meteorological, soil physical, and hydrological parameters, provides a reliable prognosis of the S status. This entry summarizes research information on S nutrition, addressing all agronomic aspects.

INTRODUCTION

Sulfur (S) is normally associated with the acid rain phenomenon and at the peak of industrialization, it became infamous as the yellow poison. Large parts of East Asia suffer in particular from extraordinarily high atmospheric S depositions. In contrast, clean air acts resulted in a drastic decline in S loads and consequently atmospheric depositions declined to zero in rural remote areas of northern Europe.^[1] On soils in these territories, physical and hydrological parameters rather than their soil organic matter (SOM) content influence the size of the plant available sulfate pool in the rhizosphere (see the entry *Sulfur*, p. 2241). Macroscopic symptoms of severe S deficiency are observed routinely on farm lands irrespective of the crop grown (Fig. 1).

Negative impacts on crop productivity and crop quality are severe if not alleviated by timely S fertilization at crop-specific rates. While respective dose-effect relationships are well described in the literature^[2] (Fig. 2), less attention has been paid so far to interactions between S nutritional status and resistance of plants against biotic and abiotic stress. The S-induced resistance (SIR) is one constituent of the complex phenomenon of induced resistance against fungal diseases.^[3] Even more complex seems the cross talk between S nutritional status of crops and their infestation by general and specialist insects.^[4]

An insufficient S supply is directly linked to the nitrogen (N) utilization efficiency, as both nutrients are required in a balanced ratio for amino acid biosynthesis. Under conditions of severe S deficiency, N utilization may be as low as 25%. On an average, each kilogram of S unavailable to

satisfy the demand of the plant leaves 15 kg of N unutilized with the risks to be lost to the environment.

The S supply is of a high ecological importance in the context of green house gases, and it is positively correlated with the release of hydrogen sulfide (H₂S) by plants.^[5] The predicted climate change implies scenarios that might increase the risk of S deficiency by another 11% in the next 70 years. Based on the model calculations of Schnug and Haneklaus^[6], the input of actual climate and crop data show that about 24% of the mean ozone concentration can be degraded by H₂S, provided the S supply is sufficiently high.

Both severe S deficiency and its toxicity may occur in plants, foodstuffs, animal feed, and the human body.^[7] Even more important than detrimental effects of an over-rated S supply on crop parameters is a possibly detrimental effect on animal health. Prominent examples of adverse effects of high S intake on ruminants are polioencephalomalacia, neurological disorder, and hemolytic anaemia.^[8] While deficiency of S in the human diet is rare, Grimble^[9] points out that high intake of L-methionine might increase the homocysteine level in plasma and thus may favor inflammatory centers.

PLANT AVAILABLE S IN SOILS

The variability of sulfate concentrations within one field can be as high as variations between different soil types in diverse climatic zones.^[10] This high spatio-temporal variation of plant available sulfate concentrations under humid conditions is closely related to soil physical and hydrological parameters (see the entry *Sulfur*, p. 2241). The results

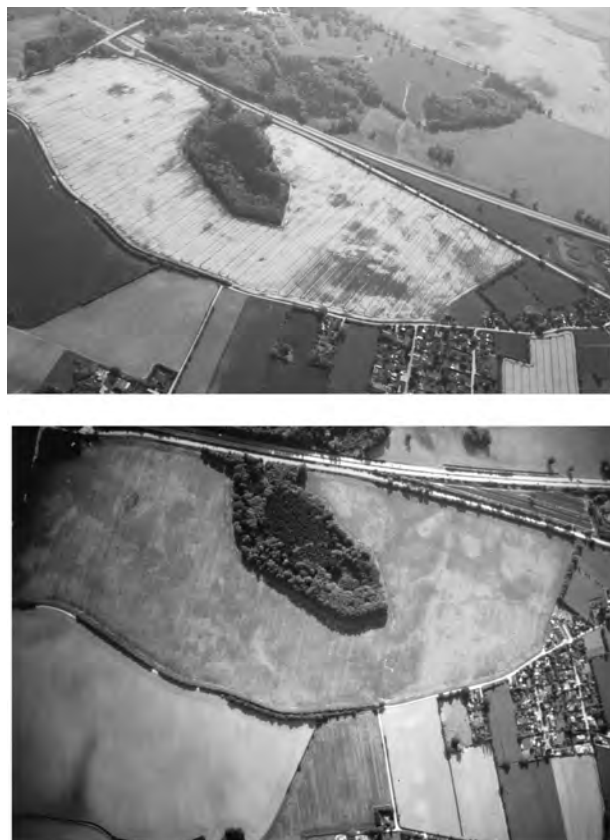


Fig. 1 Repeated patterns of small-scale spatial variability of S deficiency in oilseed rape (top; S-fertilized field in lower right corner) and sugar beet (bottom) on a production field in northern Germany.

presented in Fig. 3 illustrate coherences for different soil types, soil water regime, and crop S status. These studies were conducted at a time when S fertilization was only sporadically used.

Severe S deficiency in crops can occur on all soil types and is generally exacerbated by high agronomic yields, soils with a light soil texture, high permeability and low

SOM content, sites poorly connected to capillary ascending groundwater, leaching, reduced root growth and rooting intensity in acid soils, soil compaction, or low soil temperatures. In addition to the spatial variability, rapid temporal changes in soil sulfate are a causal reason for a lack of relationship between soil analytical data and plant S status or crop yield. For predicting the S supply of crops a site-specific S budget, which includes information about geomorphology, texture, climatic data, crop type, and characteristics of the local soil water regime (see Fig. 1), provides reliable data.

S IN PLANTS

The S deficiency in crops induces characteristic macroscopic symptoms, which are comprehensively described by Haneklaus et al.^[1,10] It is commonly evaluated as being highly biocompliant, but there are indications that overrated S fertilization may reduce crop yield, and the adverse effect is related to crop type.^[7]

S AND CROP PRODUCTIVITY

The determination of the total S content in the vegetative plant tissue is a reliable method to assess the S nutritional status of crops. There is no one exclusive critical nutrient value for each crop, as it depends on the growth conditions, the phenological stage of the plant at sampling, the sampled plant part, the determined S species, the targeted yield, and mathematical approach for calculating it.^[10] Data about symptomatological S values, critical nutrient values, and toxicological values have been compiled and statistically computed for different crops for a better appraisal of the S nutritional status of a crop plant.^[10] The relationship between S concentration in plant tissue and yield reflects the physiological patterns in the internal nutrient utilization, which is specific to each plant species and can be best established by boundary lines.^[10]

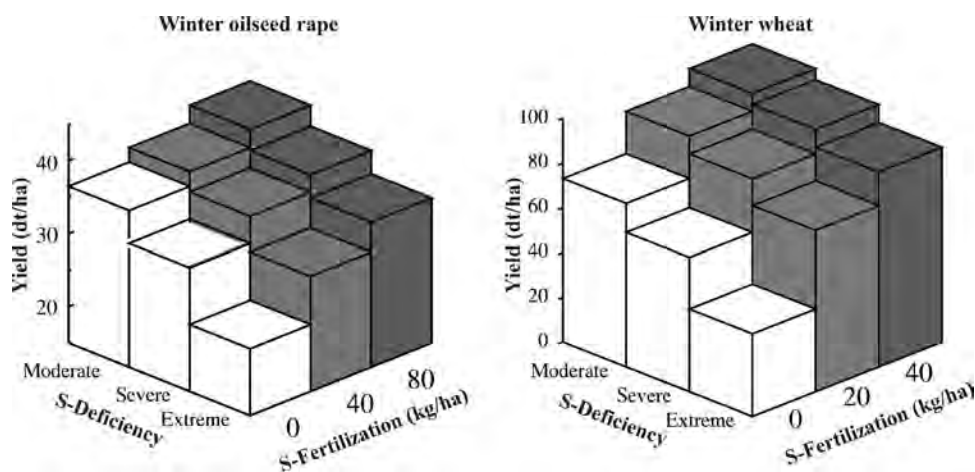


Fig. 2 Influence of S fertilization on seed and grain yield of winter oilseed rape and winter wheat in relation to the S nutritional status.

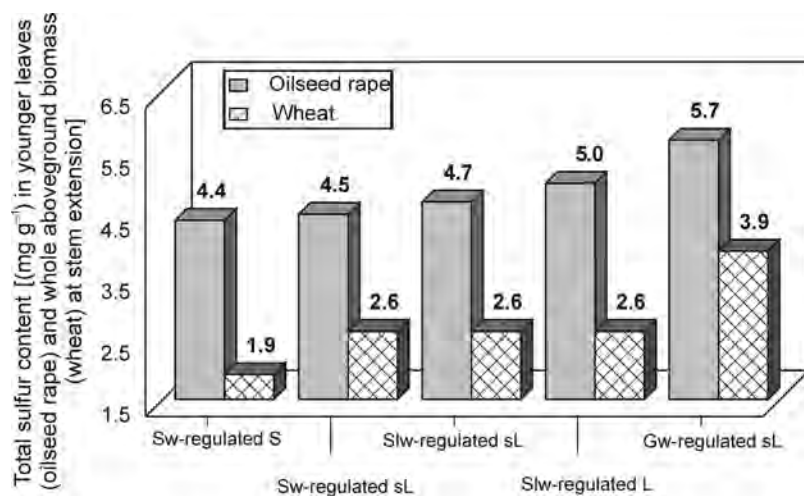


Fig. 3 Influence of soil type and soil water regime on the S nutritional status of oilseed rape ($n = 78$) and winter wheat ($n = 82$) on the Isle of Ruegen, Germany, in 1993 (Gw = ground-water, Slw = slack water, Sw = seepage water, and sL = sandy loam).

The BOUNDARY LINE DEVELOPMENT SYSTEM determines the upper boundary line functions and evaluates optimum nutrient values and ranges.^[10]

S AND CROP QUALITY

A sufficient S supply is required for a high quality of bread-making wheat. Its deficiency yields distinctly firmer and less extensible doughs as the elasticity and resistance against extensibility of the dough are related to the concentration of S-containing amino acids and glutathione.

It is the asparagine content of a crop that determines the formation of potentially carcinogenic acrylamide during processing at high temperatures.^[11] S deficiency causes a substantial accumulation of free asparagine in wheat grains and may constitute up to 50% of the total free amino acid pool.^[11] Consequently, there exists a close correlation between free asparagine in the grain and acrylamide formation in heated wheat flour.

The S supply is a key factor influencing the pharmaceutical quality of medicinal plants where S-containing secondary compounds account for the bioactive principal.^[10] Prominent examples are the significant increase of the glucosinolate content in vegetative and generative tissue of Tropaeolaceae (e.g., *Tropaeolum majus*) and the alliin content in Liliaceae (e.g., *Allium cepa*) by S fertilization.

S AND CROP RESISTANCE AGAINST BIOTIC STRESS

Glucosinolates and/or their degradation products, isothiocyanates, are effective against soil-borne fungal diseases, nematodes, and weeds and their targeted use comes under the term biofumigation.^[7]

The term SIR denotes the reinforcement of the natural resistance of plants against fungal pathogens through

triggering the stimulation of metabolic processes involving S by targeted sulfate-based and soil-applied fertilizer strategies. Molecular, physiological, and agronomic aspects of SIR are comprehensively discussed by Haneklaus et al.^[3] The S nutritional level favored the infestation of oilseed rape by specialist insects, but a practical utilization seems not feasible because of and interferences with crop production and crop quality besides the range of agronomic and environmental factors, which control the abundance of beneficial and pest insects.^[4]

CONCLUSION

S nutrition has many facets: minimum factor that impairs crop productivity, crop quality, and plant health; pollutant or indispensable resource to minimizing nitrogen losses to the environment. Substantial progress in developing fertilizer strategies, which address and transcribe an holistic approach of crop production, can be expected since elaborate analytical facilities, all coming under the term “omics” (e.g., ionomics and metabolomics), are available.

REFERENCES

1. Haneklaus, S.; Bloem, E.; Schnug, E. History of sulfur deficiency in crops. In *Sulfur: A Missing Link Between Soils, Crops, and Nutrition. B. Crops and Sulfur*; Jez, J., Ed.; CSSA-ASA-SSSA: Madison, 2008; 45–58.
2. Schnug, E.; Haneklaus, S. Sulphur deficiency in *Brassica napus*—Biochemistry, symptomatology, morphogenesis. FAL—Agr. Res. (Landbauforschung voelkenrode) **1994**, (special issue 144), 31.
3. Haneklaus, S.; Bloem, E.; Schnug, E. Sulfur and plant disease. In *Mineral Nutrition and Plant Diseases*; Datnoff, L., Elmer, W., Huber, D., Eds.; APS Press: St. Paul, 2007; 101–118.
4. Haneklaus, S.; Bloem, E.; Aljmlil, F.; Buechs, W.; Schnug, E. Influence of sulfur fertilization on the insect inventory in oilseed rape during the vegetation period. In *Advances in*

Plant Sulfur Research Including Links to Agriculture and Environment, Significance of Sulfur in the Food Chain and Regulatory Aspects of Sulfur Metabolism; Sirko, A., Ed.; Margraf Publishers: Weikersheim, 2009; 141–145.

5. Bloem, E.; Haneklaus, S.; Salac, I.; Wickenhäuser, P.; Schnug, E. Facts and Fiction about sulfur metabolism in relation to plant-pathogen interactions. *Plant Biol.* **2007**, *9*, 596–607.
6. Schnug, E.; Haneklaus, S. The role of sulfur in sustainable agriculture. In *Proceedings of the 1st Sino-German Workshop on Aspects of Sulfur Nutrition of Plants*, Shenyang, China, May 23–27, 2004; Schnug, E., De Kok, L.J., Eds.; FAL—Agric. Res. (Landbauforschung Voelkenrode), special issue 283; 131–135.
7. Haneklaus, S.; Bloem, E.; Schnug, E. Sulphur interactions in crop ecosystems. In *Sulfur in Plants—An Ecological Perspective*; Hawkesford, M.J., DeKok, L.J., Eds.; Springer: Dordrecht, 2006; 17–58.
8. Gould, D.H.; Dargatz, D.A.; Garry, F.B.; Hamar, D.H.; Ross, P.F. Potentially hazardous sulfur conditions on beef cattle ranches in the United States. *JAVMA* **2002**, *221*, 673–677.
9. Grimble, R.F. The effects of sulfur amino acid intake on immune functions in humans. *J. Nutr.* **2006**, *136*, 1660–1665.
10. Haneklaus, S.; Bloem, E.; Schnug, E.; De Kok, L.; Stulen, I. Sulphur. In *Handbook of Plant Nutrition*; Barker, A.V., Pilbeam, D.J., Eds.; CRC Press: Boca Raton, 2006; 183–238.
11. Halford, N.G.; Muttucumaru, N.; Curtis, T.Y.; Parry, M.A. Genetic and agronomic approaches to decreasing acrylamide precursors in crop plants. *Food Addit. Contam.* **2007**, *24* (Suppl. 1), 26–36.

Surface Area: Specific

L.A. Graham Aylmore

*Soil Science and Plant Nutrition, University of Western Australia, Nedlands,
Western Australia, Australia*

Abstract

Surface area is an extremely important soil property because a solid can establish contact with another solid, liquid, or a gas, only at its surface. The larger the specific surface area (the surface area per unit mass of material), the larger is the surface energy per unit mass, which is available for both physical and chemical interaction with the surroundings. From the agricultural or environmental point of view, most chemical reactions in soils take place at surfaces within the porous matrix. It is the interaction between the surfaces of the finely divided fraction and the soil water solution, which determines the mechanical or physical properties such as consistency, plasticity, swelling, and shrinking of the soil (i.e., the clay–water interaction). It is on the surfaces of the primary particles that the majority of plant nutrients, heavy metals, and pesticides are held and hence any meaningful study of their retention must, of necessity, be referred back to a unit surface area basis for an understanding of the mechanisms involved.

RELATION WITH PARTICLE SIZE DISTRIBUTION

Surface area depends primarily on the state of subdivision of a particulate material. A system in which the particles are in a fine state of subdivision is termed as a “disperse” system. Large surface per unit mass is a characteristic property of all disperse systems, and the physical and chemical behaviors of such systems depend greatly on the effects, which take place at the interface between two different phases of the system (e.g., between solid and liquid phases). Thus, the clay fraction containing finely divided crystalline and amorphous constituents plays a major role in this regard. For example, 1 g of <2 μm clay possesses approximately 50 times the specific surface area (SSA) of the same quantity of very fine sand and 10 times that of the same quantity of silt; colloidal clay (100 mμ) possesses approximately 20 times the SSA of the same quantity of <2 μm size clay and 1000 times that of the same quantity of very fine sand. It is for this reason that the particles of smallest size in the soil are frequently referred to as the “active fraction.”

DETERMINATION OF SSA

The most universally accepted method for determining the SSA of finely divided porous materials is that of Brunauer, Emmett, and Teller^[1] the BET method. The theoretical description of the process of surface adsorption derived by BET can be written as follows:

$$\frac{p}{v(p_0 - p)} = \frac{1}{v_m c} + \frac{c - 1}{v_m c} \frac{p}{p_0} \quad (1)$$

for adsorption on a free surface, where v is the quantity of gas adsorbed at pressure p , v_m is the amount required to form a monolayer, and c is a constant related to the difference between the heat of adsorption of the first and subsequent layers. The main simplifying assumptions in the derivation of this equation are that the first layer of molecules is adsorbed at a particular energy and that all subsequent layers are adsorbed at an energy equal to that for the condensation of the pure liquid. This is a linear equation and a plot of $p/v(p_0 - p)$ against p/p_0 gives a straight line if the theory is obeyed. The intercept on the vertical axis is $1/v_m c$ and the slope is

$$\frac{c - 1}{v_m c}$$

so that the two constants v_m and c can be obtained from experimental data. In practice, this equation is found to be closely obeyed by most adsorbents over the range as follows:

$$\frac{p}{p_0} = 0.05 - 0.35$$

By measuring v , the volume of gas adsorbed at several pressures within the range over which the equation is applicable, and knowing p_0 , the saturation vapor pressure, we can obtain an estimate of the volume of gas required to form a monolayer over the surface of the soil from the appropriate plot. Knowing the area occupied by one molecule of gas (i.e., molecular area), the surface area of the sorbent material can be accurately estimated.

A number of gases have been utilized in the application of this method to the physical adsorption of gases at low

temperature. When vapor adsorption methods were first developed, Emmett and Brunauer^[2] advocated the use of nitrogen as an adsorbate at 77 K (the boiling point of liquid nitrogen), and this gas has been generally accepted as the most satisfactory. The use of nitrogen as the primary standard has, however, been questioned,^[3,4] there being suggestions that the quadrupole moment of nitrogen can interact with hydroxyl or other polar groups, to varying extents, leading to a change in the cross-sectional area of the adsorbed molecule.

Argon (Ar) adsorption at 77 K or 91 K (the boiling point of liquid oxygen) has been preferred by some workers, as this is a symmetrical non-polar atom, which should be less subject to specific interactions affected by changes in the chemical nature of surfaces. However, argon is less strongly adsorbed than nitrogen, raising some doubts as to whether the BET procedure provides a satisfactory estimate of the monolayer capacity ($c < 50$). Also there are strong suggestions that the effective saturation vapor pressure for argon at 77 K varies between that of solid argon above non-porous materials to that of the supercooled liquid within micropores ($< 20 \text{ \AA}$) and transitional ($20\text{--}200 \text{ \AA}$) pores.^[5,6]

Carbon dioxide adsorption at 196 K has been extensively used in studies on carbon blacks, charcoals, and similar materials.^[7] The surface areas obtained using carbon dioxide on these materials often vastly exceed those obtained by nitrogen adsorption at 77 K, indicating the presence of large volumes of microporous regions inaccessible to the less energetic nitrogen adsorption. Similar effects have been noted with the expanding layer lattice aluminosilicates such as smectite and vermiculite.^[8]

For the measurement of very small surface areas ($< 1 \text{ m}^2/\text{g}$), krypton sorption at 77 K has the advantage of providing a low saturation vapor pressure at the working temperature, and hence a marked reduction in the volume of unadsorbed gas that has to be corrected for in the BET calculation.^[9] Krypton (Kr) adsorption also offers the benefit that its inertness makes it unlikely that it will interact specifically with the solid surface. Because krypton and carbon dioxide sorption at 77 and 196 K, respectively, like that of argon at 77 K, take place below the bulk triple point, the use of these two gases in SSA measurement is likewise subject to some uncertainty in the correct values for saturation vapor pressure to be used in the BET analysis.

The t-Method

Uncertainties in the validity of the BET approach when micro- and transitional pores are present, and in the absolute values of molecular areas and how these will vary with a substrate, which led to the development of alternative methods for comparison of gas adsorption isotherms and surface area determination. Among these, the t-method of Lippens and De Boer^[10] and similar variations^[11,12] have attracted attention as a means of interpreting vapor

adsorption and characterizing the porosity of solid adsorbents. These methods involve the use of “reduced isotherms” in which the volume adsorbed V is plotted against the corresponding statistical thickness (t) of the adsorbed layer on a non-porous reference solid,^[10] or alternatively against some arbitrarily chosen parameter

$$V_s = \frac{V}{V_x}$$

where V_x is the volume adsorbed at^[11]

$$\frac{p}{p_0} = 0.4$$

where adsorption results entirely from monolayer–multilayer formation, straight line plots with slope proportional to the surface area of the sample will be obtained.^[13]

Microporosity and Accessibility of Surfaces

The nitrogen surface areas for the layer lattice aluminosilicate clay materials range from about 10 to 30 m^2/g for the coarser kaolinite minerals through 50–180 m^2/g for the illitic clays. If only the external surface area of the smectites is accessible to the measuring sorbate, as is generally the case for non-polar gases, the surface area involved is frequently of the magnitude of 38 m^2/g (e.g., Wyoming bentonite) to 99 m^2/g (Redhill montmorillonite).^[14] Even these values are, however, influenced by the nature of the dominant exchangeable cation on the clay surface suggesting differences in lamellar packing arrangement.^[8] The use of more strongly sorbed polar molecules such as water, ethylene glycol,^[15] ethylene glycol monoethyl ether (EGME),^[16] and carbon dioxide,^[8,13] and adsorption from solution of cetyl pyridinium bromide (CPB),^[17] methylene blue (MB),^[18] or p-nitrophenol (pNP)^[19,20] provides access to the interlayer surfaces of the smectite clays to varying extents giving substantially higher values (300–800 m^2/g). From chemical analysis and crystallographic parameters, the definitive total SSA can be calculated to be 750 m^2/g .^[21]

The value of the constant c in the BET equation gives a measure of the energy with which the gas molecules are adsorbed by the porous material. For non-polar gas (i.e., electrically neutral) molecules, the presence of the charged cations on the clay surface will have little, if any, effect on the sorption. On the other hand, conducting a similar experiment with polar molecules such as water, the differing effects of different cations upon the adsorption of water are illustrated by the significant differences in sorption energies calculated with different exchangeable cations in the soil. Using this method, Keenan et al.^[22] demonstrated on kaolinitic clay the decreasing effect with increasing size of the monovalent alkali metal ions in the order $\text{Na} > \text{K} > \text{Rb} > \text{Cs}$, as would be expected, the smaller ions having a greater affinity for water of hydration. The obvious

consequence of the presence of cations on the clay surfaces is dramatic variations in the apparent molecular area covered by one molecule as the sorbate clusters around the charged cations.

Similar difficulties are observed with other materials containing micropores, such as the tubular halloysites^[23] and allophanes (SSA ranging from 400 to 900 m²/g),^[24] where the possibility of enhanced sorption of gases through capillary condensation, the physical configuration, and pore shape^[25] make it difficult to interpret sorbed volumes in a meaningful way in terms of surface coverage. Measurement of the SSA of soils is frequently further complicated by the presence of organic matter, which can both significantly reduce the value measured by effectively blocking access of the sorbate to micropores^[20,26,27] or, alternatively, increase it by virtue of its considerable capacity to absorb polar molecules into the structure.^[28,29]

Molecular Areas

Values of molecular area have been determined with fair accuracy for a number of gases, e.g., 0.162 nm² for nitrogen (N₂), 0.138 nm² for Ar, and 0.196 nm² for Kr at 77 K, 0.144 nm² for Ar at 90.1 K, and 0.221 nm² for CO₂ at 195 K. As could be expected, given the tendency of polar molecules to cluster around cation sites, the molecular areas estimated for most such sorbates vary substantially. Derived values include 0.108 nm² for water,^[30] 0.33 nm² for ethylene glycol,^[15] 0.52 nm² for EGME,^[16] 0.54 nm² for CPB (using adsorption from solution),^[17] 1.3 nm² for MB,^[18] and 0.424 nm² for pNP.^[19,20] These derived values are the subject of some controversy.^[21,31]

Negative Adsorption (Co-Ion Exclusion)

In 1947, Schofield^[32] described a method for determining the SSA of clays based on the electrostatic repulsion of anions from the negatively charged clay surface. The volume from which anions (co-ions) are excluded from a negatively charged clay surface in a 1:1 electrolyte solution is given by^[33]

$$V_{\text{ex}} = \frac{2A(1 - e^{\Psi/2})}{K} \quad (2)$$

in which V_{ex} is the volume of exclusion, $2/K$ is the characteristic length in the Debye-Hückel theory of strong electrolytes and is given by $3.04/C^{1/2}$ expressed in Å, C is the concentration, A is the area of the interface, and Ψ is the reduced electrical potential at the Gouy plane associated with the development of the diffuse layer of co-ions. For large electric potentials of the magnitude of -200 mV, the value of $(1 - e^{\Psi/2})$ is 0.98 so the slope of a plot of V_{ex} against $2/K$ would give a good estimate of the surface area. However, potentials of this magnitude are only attained in very dilute solutions. The layer lattice

aluminosilicate clay surfaces are constant charge surfaces for which the Gouy plane potential increases with decrease in concentration. For lithium (Li)-montmorillonite, Quirk and Marcelja^[34] found that the surface potential varied from -36 to -225 mV in lithium chloride solutions ranging from 0.3 to 10^{-4} M. Edwards et al.^[35,36] measured the volume of exclusion for Wyoming bentonite and Fithian illite saturated with the alkali metal cations. The co-ion exclusion plot yielded an SSA of 560 m²/g for Li-montmorillonite and 156 m²/g for Cs-montmorillonite compared with the theoretical hydratable area of some 750 m²/g. For the illite saturated with these cations, the exclusion area varied from 80 to approximately 0 m²/g compared with a nitrogen area of 110 m²/g. The decreasing slope of the co-ion exclusion plots with increasing radius of the ion is associated with the greater accommodation of ions in the Stern layer, thereby reducing the Gouy plane potential and consequently co-ion exclusion. As the basic assumption underlying Schofield's equation that the surface charge is large is not met, co-ion exclusion is clearly not appropriate for measuring the surface area of clays, although it is incorrectly cited in the literature.

CONCLUSION

The surface area available to any given sorbate is determined essentially by surface reactivity and geometrical considerations. Hence their use should be governed by the intent and purpose of the surface area measurement. Despite the reservations with respect to saturation vapor pressures, application of the BET theory to the adsorption of N₂, Ar, and Kr gases at low temperatures provides the standard procedures for the measurement of the external surface areas of finely divided materials using automated systems. More realistic measurement of the surface areas available in hydrated or solvated systems, where interlayer surfaces are accessible, can be obtained by the use of more strongly adsorbed polar molecules such as water, EG, EGME, or CPB. However, the affinity of charged molecules for cations, uncertainty in their surface orientation, and extent of surface coverage severely limit confidence in the accuracy of these methods. For comparative purposes, the use of water vapor sorption at

$$\frac{p}{p_0} = 0.19$$

on homoionic soils probably provides the most useful method of assessing the relative surface areas of soils.

REFERENCES

1. Brunauer, S.; Emmett, P.H.; Teller, E. Adsorption of gases in multimolecular layers. *J. Am. Chem. Soc.* **1938**, *60*, 309–310.

2. Emmett, P.H.; Brunauer, S. The use of low temperature van der waals adsorption isotherms in determining the surface area of iron synthetic ammonia catalysts. *J. Am. Chem. Soc.* **1937**, *59*, 1553–1564.
3. Aristov, B.C.; Kiselev, A.V. Effect of dehydration of silica surface on the adsorption isotherms for gaseous nitrogen and argon. *Russ. J. Phys. Chem.* **1963**, *37*, 1359–1363.
4. Pierce, C.; Ewing, B. Area of uniform graphite surfaces. *J. Phys. Chem.* **1964**, *68*, 2562–2568.
5. Harris, M.R.; Sing, K.S.W. Use of argon adsorption for the determination of specific surface area. *Chem. Ind.* **1967**, 757–758.
6. Carruthers, J.D.; Payne, D.A.; Sing, K.S.W.; Stryker, L.J. Specific and nonspecific interactions in the adsorption of argon, nitrogen and water vapour on oxides. *J. Colloid Interf. Sci.* **1971**, *36*, 205–216.
7. Anderson, R.B.; Bayer, J.; Hofer, L.J.E. Determining surface areas from CO₂ isotherms. *Fuel* **1965**, *44*, 443–452.
8. Aylmore, L.A.G.; Sills, I.D.; Quirk, J.P. Surface area of homoionic illite and montmorillonite clay minerals by the sorption of nitrogen and carbon dioxide. *Clay Clay Miner.* **1970**, *18*, 91–96.
9. Beebe, R.A.; Beckwith, J.B.; Honig, J.M. The determination of small surface areas by krypton adsorption at low temperatures. *J. Am. Chem. Soc.* **1945**, *67*, 1554–1558.
10. Lippens, B.C.; De Boer, J.H. Studies on pore systems in catalysis. V. The t Method. *J. Catal.* **1965**, *4*, 319–323.
11. Sing, K.S.W. Empirical method for analysis of adsorption isotherms. *Chem. Ind.* **1968**, *44*, 1520–1521.
12. Pierce, C. The universal nitrogen isotherm. *J. Phys. Chem.* **1968**, *72*, 3673–3676.
13. Aylmore, L.A.G. Gas sorption in clay mineral systems. *Clay Clay Miner.* **1974**, *22*, 175–183.
14. Aylmore, L.A.G.; Quirk, J.P. The micropore size distributions of clay mineral systems. *J. Soil Sci.* **1967**, *18*, 1–17.
15. Dyal, R.S.; Hendricks, S.B. Total surface of clays in polar liquids as a characteristic index. *Soil Sci.* **1950**, *69*, 421–432.
16. Carter, D.L.; Mortland, M.M.; Kemper, W.D. Specific surface. In *Methods of Soil Analysis. Part 1. Physical and Mineralogical Methods*; Klute, A., Ed.; 2nd Ed.; SSSA: Madison, 1986.
17. Greenland, D.J.; Quirk, J.P. Determination of surface area by adsorption of cetyl pyridinium bromide from aqueous solution. *J. Phys. Chem.* **1963**, *67*, 2886–2887.
18. Pham, T.H.; Brindley, G.W. Methylene blue adsorption by clay minerals. Determination of surface areas and cation exchange capacities. *Clay Clay Miner.* **1970**, *18*, 203–212.
19. Ristori, G.G.; Sparvoli, E.; Quirk, J.P.; Martelloni, C. Measurement of total and external surface area of homoionic smectites by p-nitrophenol adsorption. *Mineral. Petrogr. Acta* **1985**, *29A*, 137–143.
20. Theng, B.K.G.; Ristori, G.G.; Santi, C.A.; Percival, H.J. An improved method for determining the surface areas of topsoils with varied organic matter content, texture and clay mineral contents. *Eur. J. Soil Sci.* **1999**, *50*, 309–316.
21. Quirk, J.P.; Murray, R.S. Appraisal of the ethylene glycol monoethyl ether method for measuring hydratable surface area of clays and soils. *Soil Sci. Soc. Am. J.* **1999**, *63*, 839–849.
22. Keenan, A.G.; Mooney, R.W.; Wood, L.A. The relation between exchangeable ions and water adsorption on kaolinite. *J. Phys. Colloid Chem.* **1951**, *55*, 1462–1474.
23. Churchman, G.J.; Davy, T.J.; Aylmore, L.A.G.; Gilkes, R.G.; Self, P.G. Characteristics of fine pores in some halloysites. *Clay Miner.* **1995**, *30*, 89–98.
24. Hall, P.L.; Churchman, G.J.; Theng, B.K.G. Size distribution of allophane unit particles in aqueous suspensions. *Clay Clay Miner.* **1985**, *33*, 345–349.
25. Aylmore, L.A.G. Hysteresis in gas sorption isotherms. *J. Colloid Interf. Sci.* **1974**, *46*, 410–416.
26. Burford, J.R.; Deshpande, T.L.; Greenland, D.J.; Quirk, J.P. Influence of organic materials on the determination of the specific surface areas of soils. *J. Soil Sci.* **1964**, *15*, 192–201.
27. Sequi, P.; Aringhieri, R. Destruction of organic matter by hydrogen peroxide in the presence of pyrophosphate and its effect on soil specific surface area. *Soil Sci. Soc. Am.* **1977**, *41*, 340–342.
28. Chiou, C.T.; Lee, J.F.; Boyd, S.A. The surface area of soil organic matter. *Environ. Sci. Technol.* **1990**, *24*, 1164–1166.
29. de Jonge, H.; de Jonge, L.W.; Mittelmeijer-Hazeleger, M.C. The microporous structure of organic and mineral soil materials. *Soil Sci.* **2000**, *165*, 99–108.
30. Quirk, J.P. Significance of surface areas calculated from water vapour isotherms by the use of the B.E.T. Equation. *Soil Sci.* **1955**, *161*, 9–21.
31. Tiller, K.G.; Smith, L.H. Limitations of EGME retention to estimate the surface area of soils. *Aust. J. Soil Res.* **1990**, *28*, 1–26.
32. Schofield, R. Kenworthy calculation of surface areas from measurements of negative adsorption. *Nature* **1947**, *160*, 408–410.
33. Chan, D.Y.C.; Pashley, R.M.; Quirk, J.P. Surface potentials derived from co-ion exclusion measurements on homoionic montmorillonite. *Clay Clay Miner.* **1984**, *32*, 131–138.
34. Quirk, J.P.; Marcelja, S. Application of double-layer theories to the extensive crystalline swelling of li-montmorillonite. *Langmuir* **1997**, *13*, 6241–6248.
35. Edwards, D.G.; Posner, A.M.; Quirk, J.P. Repulsion of chloride ions by negatively charged clay surfaces. Part 1. Monovalent cation fithian illites. *Trans. Faraday Soc.* **1965**, *61*, 2808–2815.
36. Edwards, D.G.; Posner, A.M.; Quirk, J.P. Repulsion of chloride ions by negatively charged clay surfaces. Part 2. Monovalent montmorillonites. *Trans. Faraday Soc.* **1965**, *61*, 2816–2819.

Surface Management

Kwong Yin Chan

Organic Wastes Recycling Unit, New South Wales Department of Primary Industries, Richmond, New South Wales, Australia

Abstract

Surface soil controls a number of processes, namely gaseous exchanges, hydrology, carbon, and nutrient cycling, all vital for the functioning of the soil. These are controlled by the structure of the surface soil. In agriculture, management of surface soil structure requires an integrated approach including physical and mechanical, chemical, and biological methods.

INTRODUCTION

Surface soil, the uppermost part of the soil profile ordinarily moved in tillage (i.e., the plow layer, Ap horizon), or its equivalent in uncultivated soils, ranges in depth from 7 to 25 cm.^[1] This is the part of the soil profile: 1) that is in intimate contact with the atmosphere; 2) that regulates and partitions water in the environment; 3) where most of plant roots and soil organic carbon are concentrated; and 4) where most of the biological and nutrient-cycling activities take place. All these processes, which are vital to the functioning of the soil, are directly controlled by the structure of the surface soil. At the same time, surface soil is also most affected by human activities associated with agriculture, mining, and urbanization. Clearing land of native vegetation for agricultural development and inappropriate management practices, both of which affect mostly the surface soil structure, is the causes of worldwide land degradation problems, namely, compaction, erosion, desertification, siltation, dust storms, and eutrophication.^[2,3]

IDEAL SOIL SURFACE STRUCTURAL CONDITIONS

Functionally, the surface soil should have a range of pore sizes suitable for transmission as well as storage, free drainage, and root proliferation.^[4] A system of classifying structural quality of surface soil based on these factors has been defined in [Table 1](#).^[5] Determined mainly by soil bulk density, this system is influenced by texture only for the extreme classes, namely, clays and loamy sands.

Besides having a suitable porosity (structural form), surface soil structure should also be assessed in terms of stability and resilience.^[6] Stability is a measure of the ability to resist structural breakdown when subjected to stresses, while resilience is a measure of the ability of the soil to recover its structural form after disturbance. Both stability

and resilience are, to a varying extent, dependent on soil organic carbon levels.^[6]

DIFFERENT MANAGEMENT METHODS

Many management practices in agriculture and other land use affect soil structure, either directly or indirectly, because of their effects on the factors and processes involved in soil structure formation.^[3,4,6] While soil surface structural degradation problems (such as compaction, crusting, surface sealing, and hardsetting) are often caused by poor management practices such as excessive and inappropriate tillage, bare fallowing, and stubble burning/removal, management practices that are effective in improving/ameliorating soil surface structure also exist. These practices are described under the following categories:

Mechanical and Physical Methods

Tillage

Tillage is traditionally the most common practice used to modify soil surface structure. It can have large impacts, beneficial as well as adverse, on surface soil structural condition. Appropriately used, it is useful in providing suitable seedbeds, alleviating physical impedance (e.g., crusts), and managing crop residue.

On the other hand, inappropriate and excessive tillage has been the cause of widespread land degradation problems in many parts of the world (e.g., the dust bowls in the United States during the 1930s).^[3,7]

The worldwide trend is toward reducing the number of tillage operations (reduced tillage) and even eliminating them altogether in cropping (zero tillage), as advocated by the conservation tillage movement.^[8,9] Significant reduction in soil erosion and improvements in water infiltration and surface soil aggregation have been reported

Table 1 A system for classification of surface soil structural quality.

Class	Air-filled porosity at FC ^a (% v/v)	Water-holding capacity (mm/m)
Very good	>15%	>200
Good	10–15%	150–200
Moderate	5–10%	100–150
Poor	<5%	<100

^aFC = soil water potential (–5 KPa).Source: From Hall & Reeve.^[5]

under reduced/zero tillage systems when compared with conventional tillage systems.^[3,8,9] There is a recognition of the need to rationalize tillage so as to maximize its benefits and minimize its adverse effects. Adopting controlled traffic systems of farming, in which the wheel tracks of all operations are confined to fixed paths, prevents recompaction of soil by traffic outside the selected paths. As the field is divided into cropping and traffic zones, the undesirable effects of tillage on soil structure in the cropping zones can be avoided. Careful consideration of the type of tillage implement, as well as the frequency of tillage, is also important in crop-residue management (amount and orientation), which plays a vital role in wind- and water-erosion control.^[8]

Natural processes

These refer to the natural processes of wetting and drying and freezing and thawing, which can influence soil surface structure.^[10] These processes can be influenced to improve surface soil structure. Many traditional farming practices use these natural processes to manage soil structure. For instance, one aim of autumn cultivation by European farmers is to put the soil into a condition where it will gain maximum benefits from the freezing and thawing of frost action by exposing the maximum clod surface at a suitable moisture content to produce frost tilth.^[11] Combined actions of wetting and drying are responsible for mellowing due to production of incipient cracks.^[10] Both processes of soil restructuring work better for clay soils than for sandy soils.

Chemical Methods

These refer to the application of natural or synthetic chemicals to improve soil structure. Generally, they act either by overcoming the chemical causes of instability (e.g., dispersion) or as stabilizing agents. For example, the application of gypsum to sodic soils can stop dispersion, improve physical condition of the surface soil, and increase crop yield.^[12]

Soil conditioners in forms of synthetic organic polymers [e.g., polyvinyl alcohol (PVA) and polyacrylamide (PAM)]

can stabilize soil structure and improve soil physical conditions.^[4] Hitherto, their uses have been restricted to intensive horticulture because of the high costs involved. However, with the availability of new water-soluble polymers, the application rate, and, therefore, the costs involved have been greatly reduced.^[13]

Biological Methods

These are practices that involve using living organisms or their products for the improvement/maintenance of surface soil structure. The benefits of a pasture phase (ley) as part of the crop production system in maintaining favorable soil physical conditions have long been recognized.^[4,14] The pasture phase improves soil structure because of the activities of the denser root network and higher organic carbon inputs when compared with the crops.^[3,6,15] The pasture roots create new pores as well as enlarge and stabilize existing ones. The increase in structural stability results from a combination of the direct effects of enmeshing and entanglement, as well as the indirect effects of associated microbial activities and the release of binding material by the roots. Improvement in soil structural stability due to increases in soil organic carbon under increasing duration of pasture has been documented (Fig. 1).^[15] Retention of crop residue (straw) provides surface protection against raindrop impact

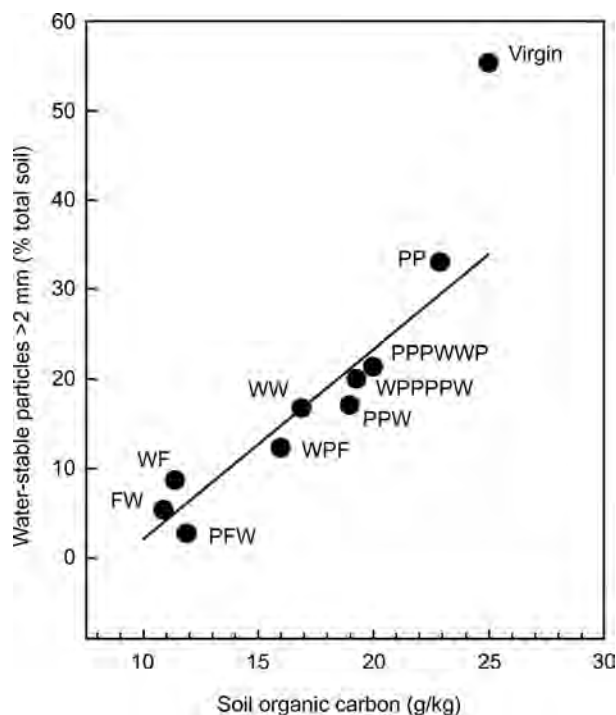


Fig. 1 The relationship between water-stable aggregation and organic carbon levels under different crop/pasture rotation (P = pasture; W = wheat; F = fallow).

Source: From Tisdall & Oades.^[15]

and wind in the short term and assists in maintaining/increasing the soil organic carbon levels, and hence the soil structural stability, in the long term. Application of organic matter as manure is a time-honored practice for maintaining/improving surface soil structure.

Introduction of soil fauna

Activities of many of the soil fauna, such as earthworms, ants, and termites, can modify soil structure. Earthworms can improve soil structure by creating burrows (macropores) and promoting surface aggregation because of their casting activities.^[16,17] Significant improvements in surface soil structure and hence soil physical properties, such as increased water infiltration, porosity, and water-holding capacity, have been demonstrated as a consequence of introducing certain earthworm species to new locations.^[17]

IMPORTANCE OF AN INTEGRATED APPROACH

While many of the aforementioned methods, when implemented separately, have beneficial effects on soil structure and therefore on soil physical properties (Table 2), an integrated approach is required in practice. For effective improvement/maintenance of surface soil structure, these methods complement each other and usually more than one method is needed. For instance, in conservation

tillage both reduction in tillage intensity and retention of crop residue are employed. Chemical/mechanical methods alone are merely short-term solutions that are ineffective and may even result in adverse effects, in the longer term. Excessively tilling a soil that is already low in stability (because of low soil organic matter level) will only make the soil more tillage-dependent, vulnerable to complete structural collapse on wetting, and prone to crop failure.^[19] However, under certain circumstances they can serve as a starting point for overcoming the initial/inherent limiting soil structural conditions (e.g., the use of gypsum to overcome clay dispersion). Macropores, which are important components of the surface soil structure, can be effectively created/maintained only by biological agents (e.g., plant roots, earthworms, and termites) as biopores.^[10] This highlights the importance of biological methods as a part of the long-term soil surface management package.

CONCLUSION

Surface soil controls a number of processes, namely gaseous exchanges, hydrology, carbon, and nutrient cycling, all vital for the functioning of the soil and are controlled by the structure of the surface soil. In agriculture, management of surface soil structure requires an integrated approach including physical and mechanical, chemical, and biological methods.

REFERENCES

1. SSSS. *Glossary of Soil Science Terms*; SSSA: Madison, 1997; 108 pp.
2. Lal, R.; Stewart, B.A. Soil degradation. *Advances in Soil Science Series*; Springer-Verlag: New York, 1990; Vol. 11, 8–12.
3. Chan, K.Y.; Pratley, J. Soil structural declines—How the trend can be reversed? In *Agriculture and the Environmental Imperative*; Pratley, J., Robinson, A., Eds.; CSIRO: Collingwood, 1998; 129–163.
4. Greenland, D.J. Soil management and soil degradation. *J. Soil Sci.* **1981**, 32, 301–322.
5. Hall, D.G.M.; Reeve, M.R.; Thomasson, A.J.; Wright, V.F. *Water Retention Porosity and Density of Field Soils*; Soil Survey Technical Monograph No. 9; Rothamsted: Harpenden, 1977; 55–60.
6. Kay, B.D. Rates of change of soil structure under different cropping systems. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer-Verlag: Heidelberg, 1990; Vol. 12, 1–52.
7. Phillips, S.H.; Young, H.M. *No-Tillage Farming*; Reisman Associates: Milwaukee, 1973; 11–16.
8. Unger, P.W.; McCalla, T.M. Conservation tillage systems. In *Advances in Agronomy*; Brady, N.C., Ed.; Academic Press: New York, 1980; Vol. 33, 2–53.
9. Carter, M.R. *Conservation Tillage in Temperate Agroecosystem Development and Adaption to Soil Climate and Biological Constraints*; CRC: Boca Raton, 1994; 400 pp.

Table 2 Beneficial effects on soil surface structure of different management practices.

Management practices	Beneficial effects on soil structure	References
Mechanical and physical		
No tillage/reduced tillage	Increased water infiltration, soil water storage, aggregation; reduced soil erosion	[3,8,9]
Controlled traffic	Reduced soil compaction; increased biological activity of crop zone	[18]
Natural processes	Increased tilth, friability	[10,11]
Chemical		
Use of ameliorants (e.g., gypsum, organic polymers)	Increased water infiltration, water-stable aggregation; reduced soil erosion, crusting	[12,13]
Biological		
Pasture leys	Increased water-stable aggregation, water infiltration, soil organic matter	[3,4,14,15]
Introduction of soil fauna	Increased macroporosity, water infiltration, water-holding capacity	[17]

10. Dexter, A.R. Amelioration of soil by natural processes. *Soil Till. Res.* **1991**, *20*, 87–100.
11. Payne, D. Soil structure, tilth and mechanical behaviour. In *Russell's Soil Conditions & Plant Growth*; Wild, A., Ed.; Longman: Essex, 1988; 379–411.
12. Shainberg, I.; Sumner, M.E.; Miller, W.P.; Farina, M.P.W.; Pavan, M.A.; Fey, M.V. Use of gypsum on soils: A review. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer-Verlag: New York, 1989; Vol. 9, 2–101.
13. Wallace, A.; Wallace, G.A. Soil and crop improvement with water-soluble polymers. *Soil Technol.* **1990**, *3*, 1–8.
14. Low, A.J. The effect of cultivation on the structure and other physical characteristics of grassland and arable soils (1945–1970). *J. Soil Sci.* **1972**, *23*, 363–380.
15. Tisdall, J.M.; Oades, J.M. The effect of crop rotation on aggregation in a red-brown earth. *Aust. J. Soil Res.* **1980**, *18*, 423–433.
16. Lee, K.E. Physical effects on soils. In *Earthworms: Their Ecology and Relationships with Soils and Land Use*; Academic Press: Sydney, 1985; 33–64.
17. Edwards, C.A.; Bohlen, P.J. Role of earthworms in soil structure, fertility and productivity. In *Biology and Ecology of Earthworms*; Chapman & Hall: London, 1996; 196–212.
18. Chamen, W.C.T.; Longstaff, D.J. Traffic and tillage effects on soil conditions and crop growth on a swelling clay soil. *Soil Use Manage.* **1995**, *11*, 168–176.
19. Greacen, E.L. Tillage research in a semi-arid environment. *Soil Till. Res.* **1983**, *3*, 107–109.

Surface Water: Nitrogen Enrichment

Tim P. Burt

Penny E. Widdison

Department of Geography, University of Durham, Durham, U.K.

Abstract

Nitrogen (N) is essential for plant growth and comprises nearly 79% of the earth's atmosphere in the form of N_2 gas. In order for N to be used for plant growth, it must be “fixed” in the form of ammonium or nitrate. In the terrestrial N cycle, microbes break down organic matter to produce much of the available N in soils. Nitrate is completely soluble in water and it is not adsorbed to clay particles; therefore, it is vulnerable to being leached out of the soil by percolating rainfall or irrigation water.

INTRODUCTION

Generally, the movement of nitrogen (N) can be described in three ways: 1) upward, crop uptake, and gaseous loss; 2) downward, as leaching to groundwater; and 3) lateral, via surface and subsurface flow to surface waters. The N cycle under arable soils is shown in Fig. 1.

NO₃ LEVELS IN SURFACE WATERS

The natural water quality of a river will be determined primarily by the catchment soil type and underlying geology to which water, falling on the catchment as rain, is exposed as it drains to the river. Climate provides an important context for N cycling by controlling the propensity for carbon and N to be stored within the catchment; thus, in the United Kingdom, upland soils tend to conserve organic matter as peat, whereas organic matter tends to decompose much more readily in lowland soils. Deviations from this baseline water quality are generally caused by the influence of human activities through point and diffuse pollution sources. Up to 40% of total N reaches the aquatic system through direct surface runoff or subsurface flow.^[1] N delivery to surface waters is further controlled by: 1) soil structure and type; 2) rainfall; 3) the amount of NO₃ supplied by fertilizers; and 4) plant cover and root activity.^[2]

In pristine river systems, the average level of NO₃ is about 0.1 mg L⁻¹ as N (mg L⁻¹ N). However, in Western Europe, high atmospheric N deposition results in N levels of relatively unpolluted rivers to range from 0.1 to 0.5 mg L⁻¹.^[3] NO₃ concentrations in European rivers have been rising (Fig. 2) and “No progress has been made in reducing the concentration of NO₃ in Europe's rivers.”^[4] High rates of N input to rivers and coastal waters are not

confined to Europe. In an average year, the Mississippi River discharges 1.57 million metric tons of N into the Gulf of Mexico.^[5] About 7 million metric tons of N in commercial fertilizers are applied annually in the basin leading to NO₃ concentrations in agricultural drains of 20–40 mg L⁻¹ or more.^[5] In the United States, in 1998, more than one-third of all river miles, lakes (excluding the Great Lakes), and estuaries did not support the uses for which they were designated under the Clean Water Act.^[6] Table 1 illustrates N inputs to rivers and coasts in areas of America, Africa, and Asia.

It is widely acknowledged that agriculture is the main source of N pollution in surface waters and groundwater in rural areas of Western Europe and the United States.^[2,7,8] The U.K. House of Lords' report *Nitrate in Water*^[9] commented on the conflicts that can arise when the use of land for farming comes into conflict with the use of land for water supply. Concern initially focused on alleged links between high NO₃ concentrations in drinking water and two health problems in humans: the “blue-baby” syndrome methemoglobinemia and gastric cancer. There are also concerns for environmental degradation. Nutrient enrichment in water bodies encourages the growth of aquatic plants (see Fig. 3). Reed beds and other marginal plants may be attractive on a small scale, but when these and, particularly, underwater plant growth are excessive, this can cause a narrowing of waterways and become a nuisance to recreational users of rivers and lakes. Furthermore, eutrophication (a group of effects caused by nutrient enrichment of water bodies) can adversely affect the aquatic ecosystem. An algal bloom may cut out light to the subsurface, and when it dies back, decomposition uses the oxygen supply needed by other species. Some algae are toxic to fish, while others, e.g., cyanobacterial species, are toxic to mammals including domestic pets.^[10]

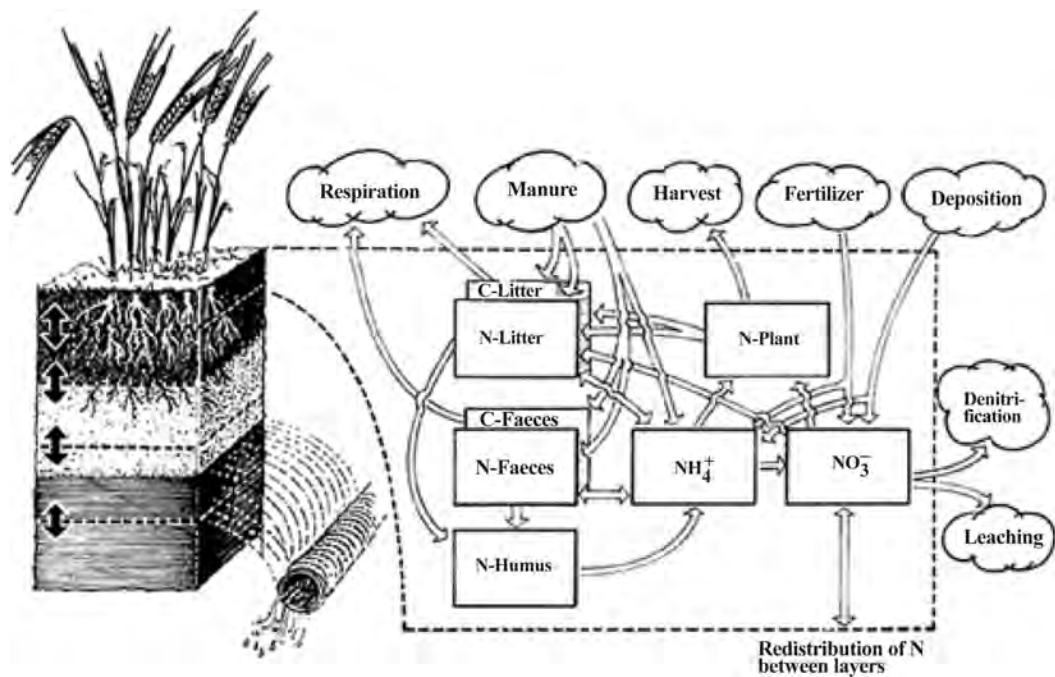


Fig. 1 The N cycle under arable soils.
Source: From Burt, Heathwaite, et al.^[2]

Studies in Asia have demonstrated the link between increasing use of fertilizers and increasing incidence of algal blooms. Table 2 illustrates rates of fertilizer application for selected Asian countries. In some Chinese provinces, fertilizer application is greater than 400 kg ha⁻¹. This is usually applied as a single application, and with crop utilization efficiency as little as 30–40%, a high proportion is lost to rivers, lakes, and coastal waters.^[11] The environmental impact at the

regional level has led to a rise in the incidence of red tides (algal blooms). During the 1960s, less than 10 red tides per year were recorded, but in the late 1990s over 300 per year were recorded.^[11]

The popular misconception that the NO₃ problem is caused by farmers applying too much NO₃ fertilizer is too simplistic. Nevertheless, there is little doubt that the high concentrations of NO₃ in fresh waters noted have mainly resulted from runoff from agricultural land and

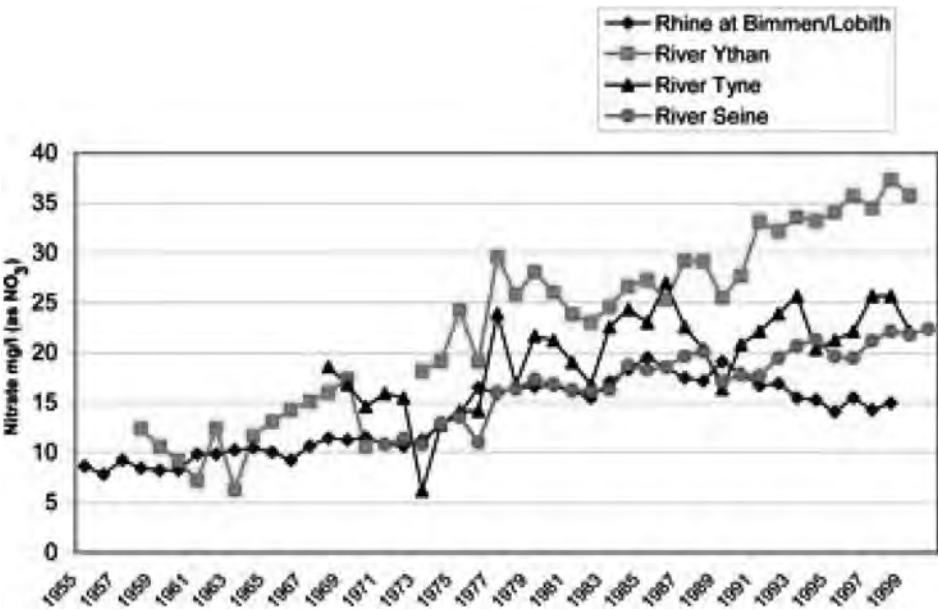


Fig. 2 NO₃ concentration in selected European rivers.
Source: From European Environment Agency.^[4]

Table 1 N inputs to rivers and coastal waters.

River	N inputs to rivers ($\text{kg}^{-2} \text{yr}^{-1}$)	N exports to coastal waters ($\text{kg}^{-2} \text{yr}^{-1}$)
Mississippi	7,489	597
Amazon	3,034	692
Nile	3,601	268
Zaire	3,427	632
Zambezi	3,175	330
Rhine	13,941	2,795
Po	9,060	1,840
Ganges	9,366	1,269
Chang Jiang	11,823	2,237
Juang He	5,159	214

Source: From Norse.^[11]

that the progressive intensification of agricultural practices, with increasing reliance on the use of nitrogenous fertilizer, has contributed significantly to this problem. Since 1945, agriculture in the industrialized world has become much more intensive. Fields are ploughed more frequently; more land is devoted to arable crops, most of which demand large amounts of fertilizer; grassland too receives large applications of fertilizer to ensure a high quality silage for winter feed; stocking densities in general are higher, leading to increased inputs of manure on grassland and problems of disposal of stored slurry; cattle often have direct access to water courses resulting in soil and bank erosion and direct contamination from animal waste; many low-lying fields are underdrained, encouraging more productive use of the land and speeding the transport of leached NO_3 to surface water courses. It is true that lowland rivers close to

Table 2 Average fertilizer use (kg ha^{-1} of cropland 2000).

Country	Average fertilizer use kg ha^{-1} (2000)
China	255.6
Japan	301.0
Korean Rep.	407.3
Vietnam	285.3

Source: From Food and Agriculture Organisation.^[12]

urban areas receive larger quantities of N from sewage effluent, but budgeting studies confirm that agriculture is the main source of NO_3 in river water.^[11,13]

Betton et al.^[14] have mapped NO_3 concentrations for mainland Britain. A marked northwest to southeast gradient is evident, reflecting relief, climatic conditions, and agricultural activity. Upland areas in the north and west are usually characterized by NO_3 concentrations below $1 \text{ mg NO}_3\text{-N L}^{-1}$. This reflects the high rainfall and low temperatures of such areas: upland soils tend to conserve organic matter and mineralization rates are low. In contrast, a decreasing ratio of runoff to rainfall and an increasing intensity of agricultural land use toward the south and east of Britain result in higher mean concentrations of NO_3 in river water. Many of the lowland rivers are characterized by concentrations above $5 \text{ mg NO}_3\text{-N L}^{-1}$; in East Anglia and parts of the Thames basin, mean NO_3 concentrations in rivers are close to the E.C. limit of $11.3 \text{ mg NO}_3\text{-N L}^{-1}$, a level exceeded in some spring waters especially in the Jurassic limestones of the Cotswolds and Lincolnshire Wolds.^[15]

The changing pattern of lowland agriculture since 1945 is reflected in long-term records of NO_3 for surface and ground waters.^[15] For both large and small rivers, there has been a relatively steady upward trend in NO_3 concentrations,

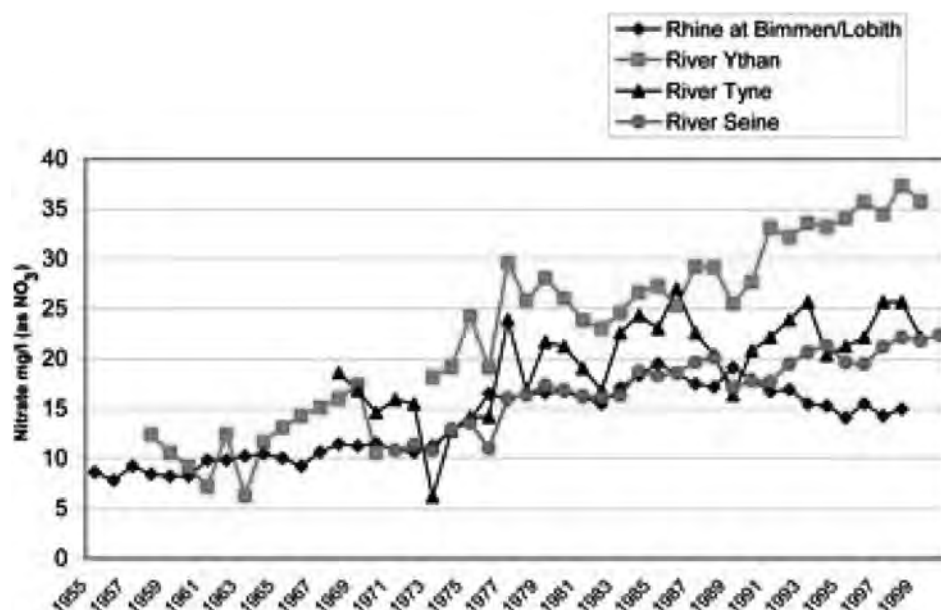


Fig. 3 Choked watercourse, River Skerne, County Durham, August 2002.

Source: From Courtesy of P. Widdison.

often of the order of 0.1–0.2 mg NO₃–N L⁻¹. Analyses for relatively short time series of just a few years (e.g., Betton and Webb^[14] have shown that the upward trend may be interrupted, either because of climatic variability (drier years are associated with lower NO₃ concentrations) or because of land use change. Nevertheless, statistical analysis of long time series shows that the main effect is a steady increase in NO₃ levels over time that is independent of climate.^[15] If trends continue, the mean NO₃ concentration of many rivers in Europe will soon be above the E.C. limit; in many cases, this level is already exceeded during the winter when NO₃ concentrations reach their maximum. In catchments where groundwater is the dominant discharge source, this long-term trend may be prolonged as it may take years for NO₃ to percolate down to the saturated zone. In such basins, NO₃ pollution may remain a problem for decades to come. A number of options have been considered as a means of halting the upward trend.

LAND USE CONTROLS TO REDUCE N ENRICHMENT TO SURFACE WATERS

Trends in water management in Europe include moves toward catchment-level management, improved intersectoral coordination and cooperation, and frameworks facilitating stakeholder participation. This approach is developed by the European Union in its Water Framework Directive, which sets targets for good ecological status for all types of surface water bodies and good quantitative status for groundwater.^[3] More localized schemes, like the U.K. Nitrate Vulnerable Zones, involve greater restrictions on farming practice, such as restricting the amount and timing of organic and inorganic fertilizer application. The EU Common Agricultural Policy is to change the way payments are made to farmers. Single-farm payments will encourage farming in a more environment friendly way. Financial payments may be available to farmers for loss of income or for changing farming practice such as improving slurry storage and fencing off watercourses to restrict livestock access.^[2] Much interest focuses on the use of riparian land as NO₃ buffer zones.^[16]

The terrestrial–aquatic ecotone (boundary zone) occupies the zone between the hillslope and the river channel, usually coinciding with the floodplain. Given their position, nearstream ecotones can potentially function as natural sinks for sediment and nutrients emanating from farmland. Observed denitrification rates in floodplain sediments may be sufficient to remove all NO₃ from groundwater flowing under a riparian woodland, with a floodplain width of 30 m. Saturated, anoxic soils, rich in carbon, are exposed to NO₃-rich groundwater. Rates of denitrification are high within this zone because the nutrients required by denitrifying bacteria are abundant.

Wetlands and wet meadows (defined as areas where the water table is at or above land surface for long enough each year to promote the formation of hydric soils and support the growth of aquatic vegetation) also have potential as N sinks.^[17] High production rates by wetland vegetation result in an abundance of carbon providing an organic substrate for bacterial processes. Wetland plants transport oxygen into anaerobic sediments, which can enhance denitrification leading to losses of N as N₂O or N₂ from wetland sediments.

The type of vegetation found on the floodplain controlling the efficiency of NO₃ absorption is the subject of much debate (see, e.g., Haycock et al.).^[16] Several studies have argued the presence of trees is crucial; yet others state the role of surface vegetation is secondary to the presence of saturated conditions together with carbon-rich sediment. Denitrifying bacteria operate best at the junction of anaerobic/aerobic zones where both carbon and NO₃ are abundant. It is clear that NO₃ losses may be reduced by creating a nutrient-retention zone between the farmland and the river. Given that many floodplains around the world are part of an intensive agricultural system, creating permanently vegetated buffer strips between field and water courses is an idea that should be actively promoted. However, buffer strips will only be successful nutrient sinks if they are managed in an appropriate way. Underlying artificial drainage should be broken or blocked up to prevent a direct route to the watercourse for solutes, and grassland strips need maintenance to prevent them becoming choked with sediment and losing their sediment retention potential.

Solving the problem of nutrient enrichment of surface waters cannot be seen in the short term. Long-term land use change is needed. Taking farm land immediately adjacent to water courses out of production is one option that could go some way to allow modern agriculture and water supply to coexist in the same basin.

REFERENCES

1. Heathwaite, A.L. Nitrogen cycling in surface waters and lakes. In *Nitrate: Processes, Patterns and Management*; Burt, T.P., Heathwaite, A.L., Trudgill, S.T., Eds.; Wiley: Chichester, 1993; 403–416.
2. Burt, T.P.; Heathwaite, A.L.; Trudgill, S.T., Eds. *Nitrate: Processes, Patterns and Management*; Wiley: Chichester, 1993.
3. WHO. *Water and Health in Europe: A Joint Report from the European Environment Agency and the WHO Regional Office for Europe*; World Health Organisation: Copenhagen, 2002.
4. European Environment Agency. *Nitrogen Concentrations in Rivers*; European Environment Agency, 2001.
5. United States Geological Service. *Nitrogen in the Mississippi Basin—Estimating Sources and Predicting Flux to the Gulf of Mexico*; USGS Fact Sheet No. 135-00; USGS, 2000.

6. Ribaud, M. Non-point source pollution control policy in the USA. In *Environmental Policies for Agricultural Pollution Control*; Shortle, J.S., Abler, D.G., Eds.; CAB International: Oxford, 2001; 123–150.
7. Power, J.F.; Wiese, R.; Flowerday, D. Managing farming systems for nitrate control: A research review from management systems evaluation areas. *J. Environ. Qual.* **2001**, *30*, 1886–1880.
8. Royal Society. *The Nitrogen Cycle of the United Kingdom*; Royal Society: London, 1983.
9. House of Lords. *Nitrate in Water: Sixteenth Report from the European Communities*; HMSO: London, 1989.
10. Addiscott, T.M. Fertilizers and nitrate leaching. In *Agricultural Chemicals and the Environment*; Hester, R.E., Harrison, R.M., Eds.; Royal Society of Chemists: Cambridge, 1996; 1–26.
11. Norse, D. Fertilisers and world food demand. Implications for environmental stress. In *IFA-FAO Agriculture Conference*; FAO: Rome, 2003.
12. Food and Agriculture Organisation. *Earth Trends Data Tables: Agricultural Inputs* 2003. Available at <http://earthtrends.wri.org>.
13. Burt, T.P.; Johnes, P.J. Managing water quality in agricultural catchments. *Trans. Inst. Br. Geogr.* **1997**, *22* (1), 61–68.
14. Betton, C.; Webb, B.W.; Walling, D.E. *Recent Trends in NO₃-N Concentration and Load in British Rivers*; IAHS Publication 203; IH Press: Wallingford, 1991; 169–180.
15. Johnes, P.J.; Burt, T.P. Nitrate in surface waters. In *Nitrate: Processes, Patterns and Management*; Burt, T.P., Heathwaite, A.L., Trudgill, S.T., Eds.; Wiley: Chichester, 1993; 269–317.
16. Haycock, N.E., Ed. *Buffer Zones: Their Processes and Potential in Water Protection*; Quest Environmental: Harpenden, 1997.
17. Howard-Williams, C.; Downes, M.T. Nitrogen cycling in wetlands. In *Nitrate: Patterns, Processes and Management*; Burt, T.P., Heathwaite, A.L., Trudgill, S.T., Eds.; Wiley: New York, 1993; 141–167.

Surface Water: Pollution by Nitrogen Fertilizers

Gregory McIsaac

Natural Resources and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, Illinois, U.S.A.

Abstract

The use of industrially manufactured nitrogen (N) fertilizers increased rapidly in developed countries between 1960 and 1980. This facilitated a large increase in the production of feed and food grains (maize, wheat, and rice) per unit of cultivated land, but in some regions it also contributed to enrichment of surface and groundwater with various forms of N. Fertilizer, however, is not the only source of N that can cause contamination of surface waters. Biological N fixation, mineralization of soil organic N, and animal wastes can also contribute to N enrichment of water bodies. Additionally, under some conditions, N applied to the soil may be converted to gaseous or immobile forms of N that do not contribute to surface water contamination. Because of these various sources and transformations of N, the severity of surface water contamination by N fertilizer has been difficult to precisely quantify. Research indicates that the amount of contamination from fertilizer varies depending on the amount of fertilizer applied, and characteristics of the soils, crops, climate, and the receiving water bodies.

PROBLEMS CAUSED BY N POLLUTION OF SURFACE WATERS

There are three water quality concerns associated with different forms of nitrogen (N). Firstly, the combined concentrations of nitrate (NO_3^-) plus nitrite (NO_2^-) in excess of 10 mg N L^{-1} can contribute to methemoglobinemia (“blue baby syndrome”) in infants if ingested.^[1] To guard against this, the U.S. Public Health Service limits NO_3^- plus NO_2^- concentration in public drinking water supplies to 10 mg N L^{-1} . Secondly, unionized ammonia (NH_3) may be toxic to fish at concentrations as low as 0.02 mg N L^{-1} . Finally, elevated total N concentrations (including NO_3^- , NH_3 , and organic forms) in rivers can promote the process of cultural eutrophication in coastal waters, whereby increased production and decomposition of algae, leads to reduced oxygen concentrations. This, in turn, may reduce the abundance and diversity of marine life and may promote the outbreak of nuisance algae.^[2]

SOURCES OF N POLLUTION

N contamination may come from a variety of sources: municipal sewage, animal manure, atmospheric deposition, biological N fixation, soil organic N, and/or N fertilizers. The consequences of contamination in a specific water body will depend upon the amount of contamination from all sources and characteristics of the receiving waters. Shallow rivers, wetlands, lakes, and reservoirs have some capacity to remove N by microbial denitrification. The susceptibility of estuaries and coastal waters to eutrophication depends on temperature, availability of phosphorus and

silica for algae production, and the rate of water exchange with the open ocean.

FERTILIZERS

The contribution of inorganic fertilizer to surface water N contamination increased after 1960 as the widespread and intensive use of inorganic N fertilizers rapidly expanded.^[3] The use of N fertilizer has allowed greater production of feed and food crops per unit area cultivated. In the United States, 75% of N fertilizer is applied to maize, while in other countries, N fertilizer is primarily used on wheat and rice. Prior to 1960, N for crop production was obtained primarily by using crop rotations that included legumes such as clover and alfalfa, which can establish a symbiotic relationship with soil bacteria that can convert atmospheric N_2 gas to biologically available forms of N.

Commercial N fertilizer is primarily manufactured as gaseous NH_3 , using the Haber–Bosch process in which gaseous N is reacted with gaseous hydrogen under pressure. The gaseous NH_3 may be injected into the soil, which is a common fertilizer application practice in the United States. Additionally, a wide variety of granular and aqueous fertilizer products containing N are manufactured from manufactured NH_3 .

WHAT HAPPENS WHEN FERTILIZER IS APPLIED TO SOIL?

Biochemical Processes

In the soil, NH_3 reacts with water and is largely converted to ammonium (NH_4^+), which tends to be strongly adsorbed

on soil particles. This adsorption inhibits the movement of NH_4^+ through the soil. NH_4^+ is an energy rich substance and certain soil bacteria can utilize this energy by decomposing the NH_4^+ to NO_3^- . Unlike NH_4^+ , NO_3^- is not adsorbed to soil particles and, therefore, moves readily with water in the soil. NO_3^- that is not taken up by plant roots or soil microorganisms can be transported to groundwater and surface water by a variety of mechanisms.

Hydrologic Processes

Rainfall, snow melt, or irrigation water input to the soil periodically exceeds the water-holding capacity of the soil in the root zone. Depending on the characteristics of the soil, this may lead to one or more of the following: 1) saturation of the root zone with water; 2) surface runoff; and 3) drainage of water through the soil profile to groundwater and/or surface water bodies. Each of these has different consequences for transport of NO_3^- to surface waters.

If the soil becomes saturated, oxygen may become scarce and in anoxic conditions, denitrifying bacteria may convert the NO_3^- to N gases (NO , N_2O , and N_2). N converted to these gases becomes unavailable for plant uptake or for surface water contamination. Additionally, saturated soil during the growing season is harmful to many crops like maize that cannot tolerate low oxygen concentrations in the root zone for more than a few days.

Surface runoff has the capacity to transport soil, vegetation, and surface applied granular fertilizers from agricultural fields to surface water bodies. If a granular form of N fertilizer had been applied immediately prior to the event that caused the surface runoff to occur, NO_3^- and NH_3 concentrations in runoff can be very high.^[4] However, this does not appear to be a common phenomenon. Small rainfall events are much more common than large events that typically produce surface runoff. After granular fertilizer is applied, it is likely that a series of small rainfall events will dissolve the granules and move the N into the soil profile, where it is less likely to contaminate surface runoff. Surface runoff usually has a low NO_3^- concentration but it can be high in organic and particulate N derived from soil and vegetation.

Drainage of water through the soil profile to groundwater and surface water appears to be the hydrologic pathway that most frequently leads to problematic NO_3^- contamination of surface waters in agricultural watersheds. This can occur in two ways: by natural drainage where ground water contributes to stream flow and river flow and by artificial subsurface drainage, where perforated pipes (sometimes called tile drains) have been buried in the soil for the purpose of removing water to reduce damage caused by saturated conditions and thereby enhance crop production (Fig. 1).

Artificial subsurface drainage improves aeration of the soil root zone and increases the length of time that machinery can be used on the soil.^[5] It is a common practice in the

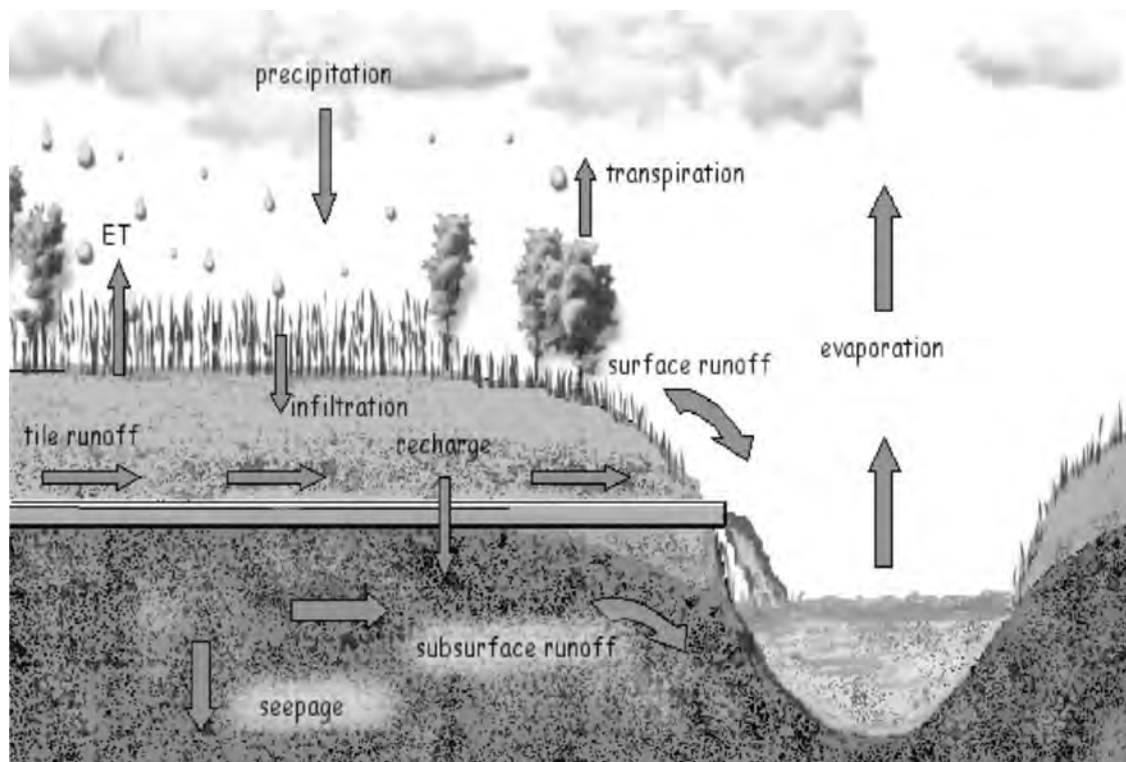


Fig. 1 Illustration of the hydrologic cycle with artificial subsurface drainage (tile runoff) contributing to surface channel flow.

Source: From Steinheimer, Scoggin, et al.^[6]

North Central United States, and in northwestern Europe, where flat and swampy land has been converted to cropland. The water removed from the soil by artificial drainage is usually directed to surface ditches, streams, and rivers. This water can have high concentrations of N, principally in the NO_3^- form, especially where N fertilizers are applied in excess to the amount necessary for crop production.^[7,8] This NO_3^- can also be derived from microbial conversion of soil organic matter to inorganic N in the process of mineralization. Mineralized soil organic N may come from crop residues (unharvested leaves, stalks, and roots). Of course, some of the N in crop residues may have originated from fertilizer applied in previous years, but it can also derive from biological N fixation or animal manures applied on a field.

SPATIAL VARIABILITY

In most agricultural settings, commercial fertilizer provides only one source of N used for crop production. Animal manure, biological N fixation, mineralization from soil organic N, and deposition of N from the atmosphere can also contribute to soil fertility and surface water contamination. Because there are multiple sources and sinks of N in the soil, the relationship between N fertilizer application rate and N loss in drainage water is not always consistent across locations and across studies. If denitrification and plant and microbial uptake of N are large, NO_3^- concentrations in subsurface drainage may be low in spite of high fertilizer N inputs. If mineralization of soil organic matter is large, N in drainage water may be large without N fertilizer input. High rates of mineralization of soil organic N occur

after the initial cultivation of virgin land, and after a leguminous forage crop such as alfalfa or clover is cultivated into the soil. Appropriate use of N fertilizer should take all of these N sources into account, as should studies examining the relationship between N fertilizer use and water quality.

WATERSHED SCALE ANALYSES

Regional N Input–Output Analyses

Howarth et al.^[9] developed an approach for estimating the net N inputs to a region N that is highly correlated with average N transport in the rivers draining temperate regions (Fig. 2). Net N input to a region was defined as sum of N in fertilizer used, biological N fixation of agricultural crops, oxidized N in atmospheric deposition in the region, and the N in food and feed imported to the region minus the N in food and feed exported from the region. This approach assumes that there is no net gain or loss of N from soil organic matter. This assumption appears to be reasonable in regions where most soils have been under continuous cultivation for 60 years or more, at which time, annual mineralization of soil organic N is roughly replaced by organic N returned to the soil in crop residues and microbial biomass.^[10]

In temperate regions, riverine N transport was on average 25% of the net N input to the region. The fate of the other 75% of the net N is unknown, but much of it is probably converted to gaseous forms of N by microbial denitrification. The high net N input in countries draining to the North Sea, most notably the Netherlands, is in part

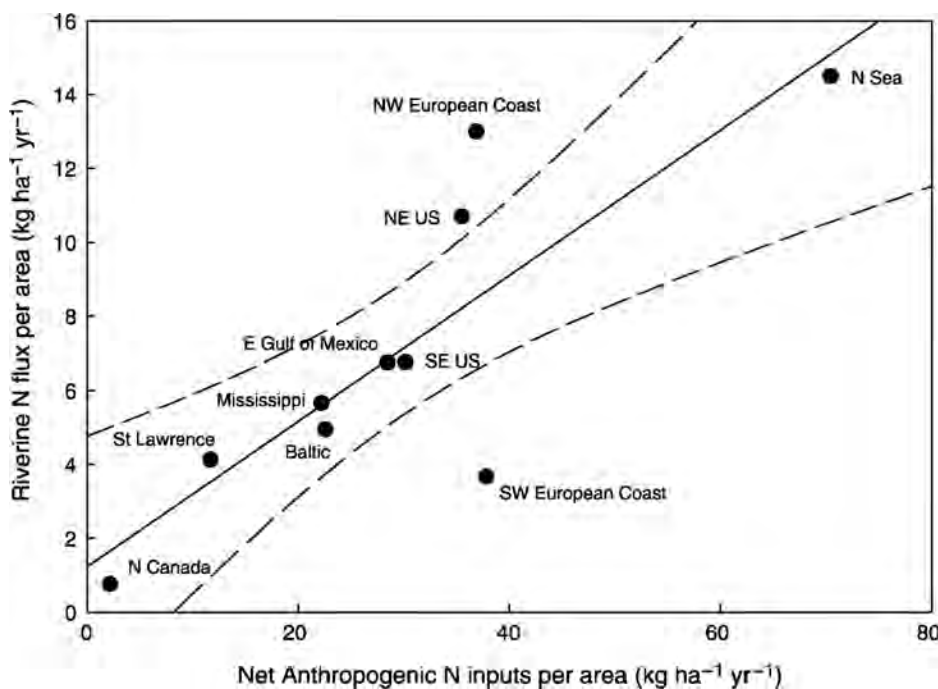


Fig. 2 Average annual riverine total N flux as a function of net anthropogenic N inputs to temperate regions draining to the North Atlantic Ocean.

Source: From Howarth, Billen, et al.^[9]

due to high density of domestic animals as well as use of N fertilizers. In tropical regions, riverine N flux was much greater than 25% of net N inputs, even in regions where little N fertilizer was used. The reasons for this are not precisely known but it is believed to be due, in part, to greater rates of biological N fixation in both cultivated and non-cultivated land in the tropics. This may also be due to the conversion of forest, wetlands, and grasslands to crop production, which leads to high rates of mineralization of soil organic N to NO_3^- that is highly mobile.

Hydrologic Process Models

The quantity of NO_3^- transported in rivers is also related to the quantity of water flowing in the rivers per unit of land area, which is also known as water yield. Caraco and Cole^[11] demonstrated that riverine NO_3^- N transport in major rivers in the world was a function of water yield, fertilizer use, population density, and atmospheric deposition of oxides of N. Building on these results, McIsaac et al.^[12] developed the following model of annual NO_3^- discharge in the Lower Mississippi River 1960–1998:

$$\text{NF}_m = 0.66\text{WY}^{0.93}e^{(0.13\text{NNI}^{2-5} + 0.06\text{NNI}^{6-9})} \quad (1)$$

where NF_m = annual NO_3^- N flux in Lower Mississippi River ($\text{kg N ha}^{-1} \text{yr}^{-1}$), NNI^{2-5} = average annual net N input during the previous 2–5 years ($\text{kg N ha}^{-1} \text{yr}^{-1}$), NNI^{6-9} = average annual net N input during the previous 6–9 years ($\text{kg N ha}^{-1} \text{yr}^{-1}$), and WY = annual water yield (m yr^{-1}).

This equation accounted for 95% of the annual variation in NO_3^- flux in the Mississippi River from 1960 to 1998 and suggested that riverine NO_3^- in a given year was correlated with net N input averaged over the previous 2–9 years. Furthermore, calculations with the model suggest that if the N fertilizer use in the basin had been 12% lower than actual during this period, NO_3^- flux to the Gulf of Mexico would have been 33% less than observed

(Fig. 3), assuming crop yields were not limited by N shortages.

The Role of Fertilizer Use Efficiency

The efficiency of fertilizer used for maize production in the major maize producing states (Illinois, Iowa, Indiana, Minnesota, and Nebraska) in the Mississippi River Basin increased between 1986 and 2000. Maize yields have increased about 20% from 1986 to 2000, while N fertilizer use has remained roughly constant. Between 1976 and 1986, an average of 0.023 kg N of fertilizer was applied for each kilogram of maize harvested. Between 1996 and 2000, an average of 0.018 kg N of fertilizer was applied per kg of maize harvested. If this improvement in fertilizer use efficiency had not occurred, NO_3^- flux in the Mississippi River in 1996–1998 would have been about 50% greater than the measured flux, according to the model of McIsaac et al.^[12] (Fig. 3).

Farmers face two major uncertainties when making fertilizer application decisions: they do not know what their yields will be nor how weather conditions might influence the availability of N fertilizer to the crop. The cost of N fertilizer has been relatively low in relation to the value of the increased yields, and consequently many farmers have believed that applying more N fertilizer than necessary provides “cheap insurance” against the uncertainties. In some instances, farmers did not consider N available from animal manure or from previously harvested legume crops like soybeans.

A number of factors are likely responsible for the increased fertilizer use efficiency. Research and outreach efforts have provided farmers with better information for making N fertilizer decisions. Water quality concerns have focused attention on the need for improved nutrient management. Weather during the 1990s was generally more favorable for corn production than the 1980s, when three major droughts occurred in the corn growing region of the Mississippi River Basin.

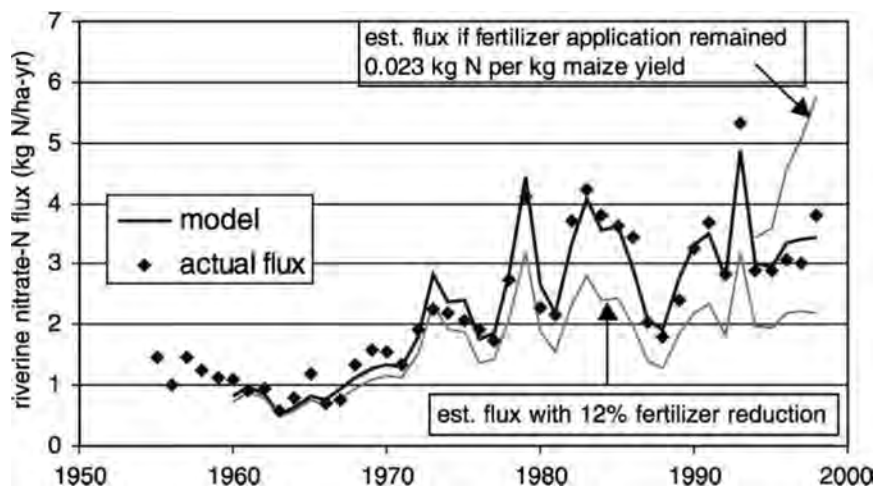


Fig. 3 Annual riverine NO_3^- flux in the Lower Mississippi River at St. Francisville, Louisiana, as determined from measurements (diamonds) as estimated from Eq. 1 (thick line), as estimated from Eq. 1 assuming a 12% reduction in N fertilizer input (thin lower line), and assuming fertilizer applications remained 0.023 kg N per bushel of harvested maize rather than declining to 0.018 kg N per bushel of harvested maize (thin upper line).

Source: From McIsaac, David, et al.^[12]

ADDITIONAL NEEDS AND APPROACHES FOR REDUCING N TRANSPORT

A improvement in the efficiency of N fertilizer use has also been observed in wheat production in the United Kingdom and rice production in Japan.^[3] However, improved fertilizer use efficiency alone may not be sufficient to address water quality problems in some settings. Jaynes et al.^[8] reported that even with N fertilizer rates at recommended levels, NO₃⁻ concentration in tile drainage water sometimes exceeded the drinking water standard of 10 mg N L⁻¹ in Iowa.

Zucker and Brown^[13] recommended several additional practices that can reduce N contributions in tile drainage: water table management, treatment of drainage water in wetlands, and use of crop rotations that reduce N losses. Additional monitoring and documenting the changes in water quality associated with changing fertilizer management practices are needed to improve our understanding of the connections between N fertilizer use and water quality in different geographic settings.

CONCLUSION

In many settings N enrichment of surface water bodies has increased following the increased use of N fertilizers. The precise contribution of N fertilizers to surface water N has been difficult to quantify because there are multiple sources of N contributing to most water bodies, and, depending on environmental conditions, a certain portion of soil N may be converted to gaseous or immobile forms. In general, however, agricultural regions with extensive artificial subsurface drainage systems or with sandy soils tend to have the most N enriched surface waters.

The efficiency of N fertilizer used for crop production increased in many areas in the 1990s and this has very likely limited or reduced the subsequent contamination of surface waters. Continued improvements in fertilizer use efficiency, and the use of wetlands for removing N from surface waters will help alleviate problems caused by N enrichment. Additional monitoring and research are needed to more precisely quantify how N management practices influence surface water N concentrations in different settings. A more precise understanding of the causal relationships between N inputs to the land and the contamination of surface waters could provide more effective guidance for management, policies, and programs intended to protect aquatic resources while maintaining optimal use of land resources.

REFERENCES

1. Skipton, S.; Hay, D. *Drinking Water: Nitrate and Methemoglobinemia ("Blue Baby" Syndrome)*; Publication G98-1369; University of Nebraska Cooperative Extension: Lincoln, 1998. Available at <http://www.ianr.unl.edu/pubs/water/g1369.htm> (accessed February 2002).
2. National Research Council. *Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution*; National Academy Press: Washington, DC, 2000.
3. Smil, V. *Enriching the Earth: Fritz Haber, Carl Bosch and the Transformation of World Food Production*; MIT Press: Cambridge, 2001.
4. Romkens, M.J.M.; Nelson, D.W.; Mannering, J.V. Nitrogen and phosphorus composition of surface runoff as affected by tillage method. *J. Environ. Qual.* **1973**, *2* (2), 292–295.
5. Badiger, S. *Integrated Numerical Modeling of Spatial and Time-variant Hydrologic Response in Subsurface Drained Watersheds*. Ph.D. Thesis; Department of Agricultural Engineering, University of Illinois, Urbana, 2001.
6. Steinheimer, T.R.; Scoggin, D.K.; Kramer, L.A. Agricultural chemical movement through a field-size watershed in Iowa: Subsurface hydrology and distribution of nitrate in groundwater. *Environ. Sci. Technol.* **1998**, *32*, 1039–1047.
7. Stevenson, F.J., Ed. *Agricultural Drainage*; American Society of Agronomy (Monograph # 38): Madison, 1999.
8. Jaynes, D.B.; Colvin, T.S.; Karlen, D.L.; Cambardella, C.A.; Meek, D.W. Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. *J. Environ. Qual.* **2001**, *30*, 1305–1314.
9. Howarth, R.W.; Billen, G.; Swaney, D.; Townsend, A.; Jaworski, N.; Lajtha, K.; Downing, J.A.; Elmgren, R.; Caraco, N.; Jordan, T.; Berendse, F.; Freney, J.; Kudeyarov, V.; Murdoch, P.; Zhao-Liang, Z. Regional nitrogen budgets and riverine N&P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. *Biogeochemistry* **1996**, *35*, 75–139.
10. Stevenson, F.J. Origin and distribution of nitrogen in the soil. In *Nitrogen in Agricultural Soils*; Stevenson, F.J., Ed.; Agronomy Monograph 22; American Society of Agronomy: Madison, 1982; 1–42.
11. Caraco, N.F.; Cole, J.J. Human impact on nitrate export: An analysis using major world rivers. *Ambio* **1999**, *28*, 167–170.
12. McIsaac, G.F.; David, M.B.; Gertner, G.Z.; Goolsby, D.A. Nitrate flux in the Mississippi River. *Nature* **2001**, *414*, 166–167.
13. Zucker, L.A.; Brown, L.C. *Agricultural Drainage: Water Quality Impacts and Subsurface Drainage Studies in the Midwest*; Ohio State University Extension Bulletin 871, 1998. Available at <http://ohioline.osu.edu/b871/> (accessed February 2002).

Sustainable Agriculture: Social Aspects

Frederick H. Buttel

Department of Rural Sociology, University of Wisconsin, Madison, Wisconsin, U.S.A.

Abstract

Agriculture—conventional and sustainable, “traditional” and “modern”—is intrinsically social (or socio-technical) in nature. The social character of agriculture is particularly apparent, particularly widely recognized, and particularly crucial in the case of sustainable agriculture. Any reasonable definition of sustainable agriculture include both social and economic as well as technological and ecological components; the roots of unsustainability and the pathways to sustainability are fundamentally social processes. This entry will provide an overview of the most fundamental social dimensions of sustainable agriculture.

THE DEFINITION OF SUSTAINABLE AGRICULTURE

It is widely agreed that sustainable agriculture is intrinsically a joint social and ecological construct. Thus, e.g., Ikerd^[1] stresses the “anthropocentric” as well as “eco-centric” nature of agricultural sustainability, noting that the essence of sustainable agriculture is that “we [in sustainable agriculture] are concerned about sustaining agriculture for the benefit of humans, both now and into the indefinite future.” Sustainable agriculture is an agriculture that is “ecologically sound, economically viable, and socially responsible.”^[2] Allen similarly emphasizes that a conception of sustainable agriculture that fails to recognize the role of people, social actors, social institutions, and social movements is a limited one.^[3]

Sustainable agriculture and organic farming are very closely related but are not exactly coterminous. Organic farming systems tend to be highly sustainable systems according to the ecological and social components of the definition of sustainable agriculture and are often quite economically viable as well.^[4,5] Many in the sustainable agriculture community, however, feel strongly that sustainability improvements in mainstream or conventional agriculture are as important as the expansion of organic farming and organic agro-food systems.

STRUCTURAL CAUSES OF THE LACK OF SUSTAINABILITY

Agricultural sustainability is not only a vision of the long-term goals for agriculture, which imply measuring sticks for or indicators of its achievement.^[6] Sustainability also implies a critique of or a set of concerns about “conventional” agriculture—that mainstream agriculture has shortcomings in terms of environmental soundness, economic soundness, and social justice—and suggests that there have

been trends over time that jeopardize the achievement of these three ends.

The historical U.S. pattern of abundant land, limited and/or relatively expensive rural labor, and a strong tendency toward overproduction and low commodity prices has encouraged a capital-intensive, chemical-intensive monocultural system with a very high degree of enterprise and spatial homogeneity (or specialization). Commodity, trade, and public research policies have historically reinforced these tendencies to monoculture, specialization, and heavy reliance on chemicals, leading to an agriculture that has significant shortcomings on all of the criteria implied in the definition of sustainable agriculture.^[7]

SUSTAINABILITY AND THE CORPORATE INDUSTRIAL AGRICULTURAL ECONOMY

Agricultural sustainability cannot be considered apart from the globalized and highly concentrated corporate economy within which sustainable agriculture, and agriculture, as a whole, is enmeshed. Corporate concentration on input provision, food processing, and retailing limits the choices available to sustainable and other farmers. Corporate control over agriculture is increasing through processes such as the disappearance of “open markets” for agricultural products; the extension of vertical integration and contractual relationships; and the ever closer alignment of multinational chemical and seed companies on the one hand and food processors, manufacturers, and global trading companies on the other.^[8] There is evidence that the increased corporate domination of agro-food systems has negative implications for achieving agricultural sustainability. Farmer contractees, for example, are typically directed contractually by “integrators” to use particular practices that, more often than not, are unsustainable.

It should be stressed, however, that corporations—often small ones, but also large firms—have been major agents in

extending the scope of organic food production and organic marketing. There are two major nationwide organic chain grocery stores, and a number of corporations have been very successful in producing organic products such as potatoes, grapes, and fresh fruits and vegetables. The corporate organization of the organic food industry represents a major dilemma for proponents of agricultural sustainability. Highly successful sustainable agriculture ideas will be attractive to private corporations, which will often be successful in diffusing products and practices. At the same time, corporate agricultural sustainability practices may undermine certain aspects of the sustainability agenda (e.g., large-scale monocultural production of organic potatoes in the Northern Intermountain states is associated with a considerable threat of soil erosion and loss of biodiversity).

There is considerable debate in the sustainable agriculture community as to whether the best indicator of the growth of sustainable agriculture is the national market share of organic food—much of which is due to the activities of very large corporations such as Whole Foods—or whether the extent of sustainable agriculture is most accurately measured by the prevalence of community-supported agriculture (CSA), farmers markets, local co-ops, community gardens, and direct marketing of food. Those with the strongest commitments to sustainable agriculture are also most likely to see food system localization and the recreation of more local “foodsheds” as the heart and soul of a genuine and enduring sustainable agriculture.^[9]

SUSTAINABILITY AS A SOCIAL MOVEMENT

Agricultural sustainability is as much a social movement as it is a set of technologies and production practices. Social definitions of sustainability—that is, the concerns, views, ideologies, and agendas of movement participants—are as or more important in shaping what sustainability is as ecological or biological indicators such as soil erosion rates, levels of soil organic matter, biodiversity, and levels of runoff. What has put sustainability on the national and global agenda is primarily the organization and activities of the following three major types of groups: farmer-driven sustainable agriculture organizations and movements, consumer-driven or consumer-oriented sustainable agriculture initiatives (such as many CSA farms and most food co-ops and local food councils), and national and global environmental groups. Environmental groups have arguably done the most to increase the visibility and persuasiveness of the overall notion of sustainability, and of the particular concept of agricultural sustainability, though farmer-oriented groups do not always agree with environmentalists’ agendas for agricultural sustainability.

Each of the three main types of sustainable agriculture organizations, however, agrees on the main components of

the movement agenda: the need for more public research on sustainable practices, the need for public policy incentives for sustainability, the desirability of family farming, and the imperative to improve the environmental performance of agriculture. Increasingly, this consensus has extended to issues such as opposition to major trade liberalization agreements (the World Trade Organization and the North American Free Trade Agreement) and opposition to genetic engineering of crops and foods.^[10]

PUBLIC POLICY AND THE FUTURE OF AGRICULTURAL SUSTAINABILITY

Three necessary but insufficient conditions for the continued advance of sustainable agriculture are the development of improved sustainable technology; the presence of a vigorous and dynamic sustainability movement; and the development of a public policy environment that reduces the public policy disincentives to an environmentally sound, socially responsible, and economically viable agriculture. Of these three conditions, the public policy environment of agricultural sustainability is arguably the most important over the long term, even though the macro-public-policy environment is difficult to redirect, and there are areas of public policy disagreement among farmers, consumers, and environmental actors in the sustainable agriculture community; e.g., some sustainability advocates, particularly those in the environmental community (and some farm groups in that community), believe that the federal (and global) levels of action are most important or efficacious, whereas others believe that at this time the local (community or regional) arena is where advocates can most easily make a difference. Nonetheless, it is apparent that over the long term sustainable agriculture cannot advance far in a public policy environment that involves the strong disincentives to sustainability that prevail. There is a need for more active federal, state, and local regulation of both the on-site and the off-site impacts of agriculture; for ending the commodity-driven pattern of federal agricultural policy that shovels the lion’s share of subsidies in the direction of large, monocultural producers of overproduced commodities; and for redirection of the public agricultural research agenda. Some of the most innovative public policy ideas in sustainable agriculture are being developed in Europe. Green payments, taxes on pesticides and fertilizers, and the embracement of a “multifunctionality” approach to government agricultural policy are particularly promising policy instruments.^[11] Each of these policy instruments would support sustainability and yield other benefits (reduced government outlays, rural development, and reduced greenhouse gas emissions). There are no technocratic shortcuts to sustainability. Sustainability is, and must remain, as much a social movement as it is a set of practices and measuring sticks of agro-food system performance.

REFERENCES

1. Ikerd, J.E. Assessing the health of ecosystems: A socioeconomic perspective. In Paper Presented at the First International Ecosystem Health and Medicine Symposium, Ottawa, Canada, 1994, 2 pp. Available at <http://www.ssu.missouri.edu/faculty/jikerd/papers/Otta-ssp.htm> (accessed October 2002).
2. Ikerd, J.E. New farmers for a new century. Paper Presented at the Youth in Agriculture Conference, Ulvik, Norway, February, 2000, 3 pp. Available at <http://www.ssu.missouri.edu/faculty/jikerd/papers/Newfarmer1.htm> (accessed October 2002).
3. Allen, P. Connecting the Social and Ecological in Sustainable Agriculture. In *Food for the Future: Conditions and Contradictions of Sustainability*; Allen, P., Ed.; Wiley: New York, 1993; 1–16.
4. Flora, C.B.; Francis, C.; King, L., Eds. *Sustainable Agriculture in Temperate Zones*; Wiley-Interscience: New York, 1990.
5. Flora, C.B., Ed. *Interactions Between Agroecosystems and Rural Communities*; CRC Press: Boca Raton, 2001.
6. Airstars, G.A., Ed. *A Life Cycle Approach to Sustainable Agriculture Indicators: Proceedings. Center for Sustainable Systems*; University of Michigan: Ann Arbor, 1999.
7. Buttel, F.H. The sociology of agricultural sustainability: Some observations on the future of sustainable agriculture. *Agr. Ecosyst. Environ.* **1993**, *46*, 175–186.
8. Heffernan, W.D. *Consolidation in the Food and Agriculture System*; National Farmers Union: Denver, 1999. Available at http://www.nfu.org/images/heffernan_1999.pdf (accessed October 2002).
9. Kloppenburg, J., Jr.; Hendrickson, J.; Stevenson, G.W. Coming into the foodshed. *Agr. Hum. Values* **1996**, *13*, 33–42.
10. Kirschenmann, F. Questioning biotechnology's claims and imagining alternatives. In *Of Frankenfoods and Golden Rice*; Buttel, F., Goodman, R., Eds.; Wisconsin Academy of Sciences, Arts, and Letters: Madison, 2002; 35–61.
11. Organisation for Economic Cooperation and Development (OECD). *Multidimensionality: A Framework for Analysis*; OECD: Paris, 1998.

Sustainable Agriculture: Soil Quality

John W. Doran

*U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
University of Nebraska, Lincoln, Nebraska, U.S.A.*

Ed G. Gregorich

*Eastern Cereal and Oilseed Research Centre, Agriculture and Agri-Food Canada,
Ottawa, Ontario, Canada*

Abstract

Sustainable agriculture is a way of farming that can be carried out for generations to come. This long-term approach to agriculture combines efficient production with the wise stewardship of the earth's resources, including land management to maintain or enhance the quality of agricultural soils. Over time, sustainable agriculture is expected to meet human needs for food and fiber; protect the natural resource base, preventing the degradation of soil, water, and air quality and conserving biodiversity; use non-renewable resources efficiently; work with natural biological cycles and controls; and assure the economic survival of farming and the well-being of farmers and their families. Thus, the concept of sustainability embraces three distinct but overlapping areas of concern: social, economic, and environmental factors. Taking a holistic ecosystem approach to agriculture is the key to integrating these components of sustainability.

INTRODUCTION

Growing human populations, diminishing non-renewable resources, social and political instability, and environmental degradation threaten the natural processes that sustain the global ecosphere and life on earth.^[1,2] With losses of farmland to degradation and urbanization and little new agricultural land to develop, meeting the needs of future populations for food, fiber, and other agricultural products depends on a significant increase in crop yields. Under food production practices, this yield increase spells greater use of inputs (e.g., energy, fertilizer, and pesticides) and raises the question of the environmental costs of increased production.

AGRICULTURE AND ENVIRONMENTAL SUSTAINABILITY

To a large extent, the economic and social viability of agriculture depends on the sustainability of the resources that agriculture both depends on and affects. Farmers have always intuitively known that good soil quality underpins the success of their operations, and soil conservation programs in many countries throughout the 20th century supported this recognition. Since the late 1980s, interest has grown in a fuller accounting of the environmental costs of agricultural production. These costs include declining water quality and competition for finite water resources, declining air quality, emission of greenhouse gases, loss

of wildlife habitat, and decline in species and genetic diversity.^[3] To a large extent, conserving and enhancing soil quality can mitigate these costs.

Soil

Agricultural land management in the past has taxed the soil system, often withdrawing more than it has returned. In particular, mechanical cultivation and the continuous production of row crops have resulted in the displacement and loss of soil by erosion, large decreases in soil organic matter, and a concomitant release of carbon (C) as carbon dioxide into the atmosphere.^[4] The inventories of soil productive capacity indicate human-induced degradation on nearly 40% of the earth's arable land as a result of land clearing, extensive soil cultivation, soil erosion, atmospheric deposition of pollutants, overgrazing, salinization, and desertification.^[5] The projected doubling of the human population in the 21st century threatens even greater degradation of soils and other natural resources.^[6]

Water

Intensive farming practices have, in many parts of the world, jeopardized the quality of surface water and groundwater through the addition of agriculturally derived nitrate nitrogen (N), phosphorus (P), pesticides, sediment, and pathogens (e.g., bacteria from manure). Agriculture is considered the most widespread contributor to non-point-source water pollution in the United States.^[7] The major

agricultural water contaminant in North America and Europe is nitrate N derived from atmospheric deposition, livestock manures, and commercial fertilizers. Human alterations of the N cycle have almost doubled the rate of N input to terrestrial ecosystems over the past years, resulting in large increases in the transfer of N from land to the atmosphere and to rivers, estuaries, and coastal waters.^[8–10]

Agriculture, competing as it does with many other uses of freshwater, is also at the heart of discussions of water quantity and is a key consideration in the trend toward demand management, rather than supply management, of water resources. Global warming scenarios predict water shortages in some existing agricultural areas in the world, and water surpluses in others. Changing conditions will call for new ways of farming and of resolving conflict among water users.

Controlling soil erosion reduces the amount of sediments entering waterways, along with the chemicals (e.g., pesticides) and pathogens they carry. Nutrient management plans that account for all sources of N and P and match nutrient application more closely to plant needs reduce the risk of N and P entering groundwater and surface water. Maintaining soil quality improves the ability of soil to receive and partition water, making the most efficient use of this resource.

Air and Atmosphere

Atmospheric concentrations of greenhouse gases—particularly nitrous oxide, methane, and carbon dioxide—have been increasing dramatically over the past years, enhancing the greenhouse effect by which the earth's atmosphere is warmed. Agriculture is a significant contributor of these gases to the atmosphere; in Canada, e.g., agriculture accounted for about 10% of total 1996 emissions of these gases, an increase of about 4% since 1981.^[11] Soil management practices such as tillage, cropping patterns, and the use of manure and commercial fertilizers influence atmospheric quality through changes in the soil's capacity to produce or consume greenhouse gases.^[12,13] Under the United Nations Framework Convention on Climate Change and its protocols, signatory countries are considering ways to reduce emissions, including those from agriculture, and also to capitalize on agricultural sinks of greenhouse gases, such as by C sequestration in agricultural soils. The threat of ozone depletion and global climate change necessitates a better understanding of the influence of land management on soil processes.

Agricultural emissions of ammonia (from fertilizer and manure) and particulate matter (dust released during soil tillage and subsequent erosion) have been linked to various environmental effects, such as acidification, eutrophication, and smog. Reducing the emission of these substances involves improving the quality of soils so that they are more resistant to erosion, reducing tillage, increasing the amount

of soil cover, and practicing better N management (both manure and commercial fertilizer).

Agroecosystem Biodiversity

Agriculture benefits from biodiversity in many ways, but it has also reduced biodiversity over the years, mainly through the conversion of natural habitats, but also through effects on soil and water quality. Improving soil quality not only maintains a healthy biological community in the soil itself, along with the functions these organisms carry out, but also supports the other flora and fauna dependent on this community. Soil quality improvements that benefit water quality can also improve the viability of aquatic ecosystems.

ASSESSMENT OF SOIL QUALITY

Assessment of soil quality or health is invaluable in determining the sustainability of land management systems.^[14] Soil quality is the major link between the strategies of conservation management practices and the achievement of the major goals of agriculture.^[15,16]

Assessment of soil quality can be used to identify problem production areas, make realistic estimates of food production, monitor changes in sustainability and environmental quality as related to agricultural management, and assist government agencies in formulating and evaluating sustainable agricultural and land use policies (Fig. 1). Using simple indicators of soil quality and soil health that have meaning to farmers and other land managers is likely the most fruitful means of linking science with practice in assessing the sustainability of land management practices.^[17]

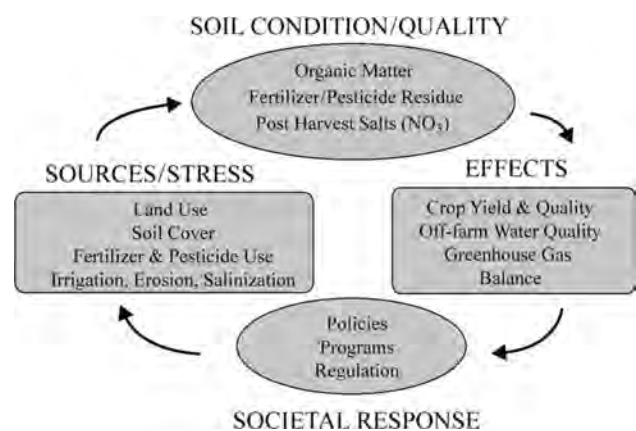


Fig. 1 The politics of soil health. The assessment of soil condition/quality is needed to monitor changes in sustainability and environmental quality as related to agriculture management and to assist governmental agencies in formulating realistic agricultural and land use policies.

Doran et al.^[18] stressed the importance of holistic management approaches that optimize the multiple functions of soil, conserve soil resources, and support strategies for promoting soil quality. They proposed a basic set of indicators to assess soil quality and health in various agricultural management systems. Many of these key indicators are very useful to specialists (e.g., researchers, consultants, extension staff, and conservationists) but are beyond the expertise or time constraints of the producer to measure. Success in developing and implementing standards for assessing soil quality and sustainability hinges on partnership with agricultural producers, who are the primary stewards of the land and the chief decision-makers regarding land use and management. In light of this consideration, Doran et al.^[18] also presented strategies for sustainable management that include a more generic set of indicators to assess soil quality and health that are practical for producers (Table 1). A study conducted in the northern U.S. corn and dairy belt used a similar approach to determine how farmers assess soil quality and health and found that they ranked soil organic matter, crop appearance, and risk to erosion as the three most important properties for describing soil health and sustainable management.^[17] Such strategies maximize the benefits of natural cycles, reduce dependence on non-renewable resources, and help producers identify long-term goals for sustainability that also meet short-term needs for production. Soil quality assessment must be directed at the financial survival of the farm as well as environmental preservation.

Although much remains to be done, useful models exist for translating soil science into practice; e.g., Gomez et al.^[19] provide a practical framework for determining the

sustainability of hill country agriculture in the Philippines. It uses indicators that satisfy both the needs of the farmer (e.g., productivity, profitability, stability, and viability) and the conservation of soil and water resources. Threshold values for sustainability are identified relative to the average local conditions for crop yield, profit, risk of crop failure, soil depth, percent soil cover, and soil organic matter content. This conceptual framework for assessing sustainability could be expanded to include other needs of society and environmental conservation. In particular, adding a category for balancing energy input and output, as well as monetary costs, would better assess the short- and long-term sustainability of management and the value of greater reliance on renewable resources and less dependence on fossil fuels and petrochemicals in enhancing economic, ecological, and environmental resources. Expanding the list of resource conservation variables to include leachable salts (especially nitrate), measured as soil electrical conductivity at the time of fertilization and after harvest, would permit land managers to better quantify the impact of agricultural practices on air and water quality.

Confirmation of the effectiveness of systems for residue management, organic matter formation, N and C cycling, soil structure maintenance, and biological control of pests and diseases will assist in discovering approaches that are both profitable and environmentally sound. The challenge in the future will be to better use the diversity and resiliency of the soil biological community to maintain a quality ecosystem, thus fostering sustainability. Sustainability strategies must be fine-tuned using such practices as crop rotation for greater crop diversity and tighter cycling of nutrients, reduction of soil disturbance to maintain soil

Table 1 Strategies for sustainable agricultural management and proposed indicators of crop performance and soil and environmental health.

Sustainability strategy	Indicators for producers
Conserve soil organic matter through maintaining soil C and N levels by reducing tillage, recycling plant and animal manures, and/or increasing plant diversity where C inputs C outputs	Direction/change in organic matter levels with time (visual or remote sensing by color or chemical analysis) Specific organic matter potential for climate, soil, and vegetation and soil water storage
Minimize soil erosion through conservation tillage and increased protective cover (residue, stable aggregates, cover crops, and green fallow)	Visual (gullies, rills, dust, etc.) Surface soil properties (topsoil depth, organic matter content/texture, water infiltration, runoff, ponding, percent cover)
Balance production and environment through conservation and integrated management systems (optimizing tillage, residue, water, and chemical use) and by synchronizing available N and P levels with crop needs during the year	Crop characteristics (visual or remote sensing of yield, color, nutrient status, plant vigor, and rooting characteristics) Soil physical condition/compaction Soil and water nitrate levels Amount and toxicity of pesticides used
Better use of renewable resources through relying less on fossil fuels and petrochemicals and more on renewable resources and biodiversity (e.g., crop rotations, legumes, manures, and integrated pest management)	Input and output ratios of costs and energy Leaching losses/soil acidification Crop characteristics (as listed above) Soil and water nitrate levels

Source: From Postel.^[2]

organic matter and reduce erosion, and development of systems that make greater use of renewable biological resources. Ultimately, the indicators of soil quality and strategies for sustainable management must be linked to the development of management systems that foster reduction in the inputs of non-renewable resources, maintain acceptable levels of productivity, and minimize impact on the environment.

REFERENCES

1. Bouma, J. Soil environmental quality: A European perspective. *J. Environ. Qual.* **1997**, *26*, 26–31.
2. Postel, S. Carrying capacity: Earth's bottom line. In *State of the World*; Brown, L.R., Ed.; W.W. Norton & Co.: New York, 1994; 3–21.
3. Lal, R. Basic concepts and global issues: Soil quality and agricultural sustainability. In *Soil Quality and Agricultural Sustainability*; Lal, R., Ed.; Ann Arbor Press: Chelsea, 1998; 3–12.
4. Houghton, R.A.; Hobbie, J.E.; Melillo, J.M.; Moore, B.; Peterson, B.J.; Shaver, G.R.; Woodwell, G.M. Changes in the carbon content of terrestrial biota and soils between 1860 and 1980: A net release of CO₂ to the atmosphere. *Ecol. Monogr.* **1983**, *53*, 235–262.
5. Doran, J.W.; Sarrantonio, M.; Liebig, M.A. Soil health and sustainability. In *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press: San Diego, 1996; 1–54.
6. Ruttan, V.W. The transition to agricultural sustainability. *P. Natl. Acad. Sci.* **1999**, *96*, 5960–5967.
7. National research council. *Soil and Water Quality: An Agenda for Agriculture, Committee on Long-Range Soil and Water Conservation, Board on Agriculture, National Research Council*; National Academy Press: Washington, 1993; 516 pp.
8. Matson, P.A.; Parton, W.J.; Power, A.G.; Swift, M.J. Agricultural intensification and ecosystem processes. *Science* **1997**, *277*, 504–509.
9. Socolow, R.H. Nitrogen management and the future of food: lessons from the management of energy and carbon. *P. Natl. Acad. Sci.* **1999**, *96*, 6001–6008.
10. Vitousek, P.M.; Aber, J.D.; Howarth, R.W.; Likens, G.E.; Matson, P.A.; Schindler, D.W.; Schlesinger, W.H.; Tilman, D.G. Human alteration of the global nitrogen cycle: Sources and consequences. *Ecol. Appl.* **1997**, *7*, 737–750.
11. McRae, T.; Smith, C.A.S.; Gregorich, L.J., Eds. *Environmental Sustainability of Canadian Agriculture*; Report of the Agri-Environmental Indicator Project; Agriculture and Agri-Food Canada: Ottawa, 2000; 224 pp.
12. Mosier, A.R. Soil processes and global change. *Biol. Fert. Soils* **1998**, *27*, 221–229.
13. Rolston, D.E.; Harper, L.A.; Mosier, A.R.; Duxbury, J.M. *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*; American Society of Agronomy Spec. Publ. 55; American Society of Agronomy: Madison, 1993.
14. Karlen, D.L.; Mausbach, M.J.; Doran, J.W.; Cline, R.G.; Harris, R.F.; Schuman, G.E. Soil quality: A concept, definition, and framework for evaluation. *Soil Sci. Soc. Am. J.* **1997**, *61*, 4–10.
15. Acton, D.F.; Gregorich, L.J., Eds. *The Health of Our Soils: Toward Sustainable Agriculture in Canada*; Agriculture and Agri-Food Canada: Ottawa, 1995.
16. Parr, J.F.; Papendick, R.I.; Hornick, S.B.; Meyer, R.E. Soil quality: Attributes and relationship to alternative and sustainable agriculture. *Am. J. Alternative Agr.* **1992**, *7*, 5–11.
17. Romig, D.E.; Garlynd, M.J.; Harris, R.F.; McSweeney, K. How farmers assess soil health and quality. *J. Soil Water Conserv.* **1995**, *50*, 229–236.
18. Doran, J.W.; Jones, A.J.; Arshad, M.A.; Gilley, J.E. Determinants of soil quality and health. In *Soil Quality and Soil Erosion*; Lal, R., Ed.; CRC Press: Boca Raton, 1999; 17–36.
19. Gomez, A.A.; Swete Kelly, D.E.; Seyers, J.K.; Coughlan, K.J. Measuring sustainability of agricultural systems at the farm level. In *Methods for Assessing Soil Quality*; Doran, J.W., Jones, A.J., Eds.; Soil Sci. Soc. Am. Spec. Publ. 49; SSSA: Madison, 1996; 401–410.

Sustainable Development and Sustainability

Alan D. Hecht

Office of Research and Development, U.S. Environmental Protection Agency (EPA),
Washington, District of Columbia, U.S.A.

Joseph Fiksel

Center for Resilience, Ohio State University, Columbus, and
Office of Research and Development, U.S. Environmental
Protection Agency (EPA), Cincinnati, Ohio, U.S.A.

Abstract

Sustainability is generally understood to mean a state or condition that allows for the fulfillment of economic and social needs without compromising the natural resources and environmental quality that are the foundation of human health, safety, security, and economic well-being. Sustainable development is both a *goal* of achieving this state for the benefit of existing and future generations and a *process* whereby innovative tools, models, and approaches are adopted to advance economic prosperity while minimizing adverse global impacts. Many case studies of business and government show that adopting sustainable practices can both enhance economic growth and protect the environment and social well-being. The need for sustainable approaches to business and economic development is made more urgent by increased rates of degradation and depletion of natural resources and by growing threats to human health and the environment. These stresses will only increase as the global population grows from 7 billion people to 9 billion by 2050. The practice of sustainable development is essential for reaching the next level of environmental protection, social development, and economic progress. This entry identifies some key origins of sustainability in the United States and around the world and concludes with eight steps toward achieving sustainable development.

INTRODUCTION

Sustainable development was defined in a 1987 UN report as “economic and social development that meets the needs of the present without compromising the ability of future generations to meet their own needs.”^[1] The concept of “sustainability” has been widely characterized as resting on three pillars: social well-being, economic prosperity, and environmental protection (Fig. 1).

As described in a 2011 report from the U.S. National Research Council, sustainability can be viewed as both a *goal* and a *process* that improves the economy, the environment, and society for the benefit of existing and future generations.

As a *goal*, sustainability is aimed at meeting society’s basic economic and social needs without undermining the natural resource base and environmental quality that are necessary for continuing to meet these needs in the future. Sustainability also describes the *process* whereby innovative tools, models, and approaches are applied in order to meet that goal, striving to advance economic prosperity while protecting natural resources. From the perspective of business and finance, sustainability as both a goal and a process aims to create value for shareholders and society

while efficiently using resources and minimizing adverse effects on the environment.

In the United States, the concept of sustainable development has roots that are over a century old. President Theodore Roosevelt relied on Gifford Pinchot, Roosevelt’s chief forester and the “father of American conservation,” for advice to develop public land policies in the late 1890s. Pinchot believed that forests should be made to serve the nation’s future as well as its present. He was an early advocate of sustainable development, writing in his 1910 book, *The Fight for Conservation*, “the central thing for which conservation stands is to make this country the best possible place to live in, both for us and for our descendants.”^[2]

Nearly 50 years later, the National Policy and Environmental Act established in 1969 the national goal of creating and maintaining “conditions under which [humans] and nature can exist in productive harmony, and fulfill the social, economic and other requirements of present and future generations of Americans.”^[3]

Sustainability has been investigated worldwide, as the Club of Rome composed of European economists and scientists published “The Limits to Growth” in 1972.^[4] This controversial report highlighted the negative impact of economic growth on the environment and served as a



Fig. 1 The three pillars of sustainability.

signal warning of the need for strategic planning in economic growth and environmental protection. The concept of sustainability was developed at the 1992 Earth Summit in Rio de Janeiro, where 179 nations endorsed the principles of Agenda 21, which declared, “The right to development must be fulfilled so as to equitably meet developmental and environmental needs of present and future generations.”^[5]

Twenty years after the Earth Summit, corporate, government, and non-governmental leaders converged again in Rio de Janeiro in June 2012 to examine and reaffirm commitments to global sustainability. The meeting provided a timely opportunity to further develop a common understanding of sustainability and how it can improve the lives of people in all countries around the world.

Sustainability is incorporated in many government goals and policy objectives. The European Union has been promoting sustainability since 2000, when it formulated the Lisbon Strategy with the goal of achieving the most competitive and dynamic knowledge-based economy in the world. The European Council has stated, “Clear and stable objectives for sustainable development would present significant economic opportunities and the potential to unleash a new wave of technology innovation, generating growth and employment.”^[6]

Sustainability has been advanced in the United States largely through Presidential Executive Orders such as 13423 under George W. Bush and 13514 under Barack Obama. Both orders set specific targets and goals for how the U.S. government should manage its own facilities. Federal procurement policies also reflect sustainability goals, their requirements for purchasing green and energy-efficient products.^[7]

The United States has sought to promote sustainability internationally through the first presidential “Policy Directive on Global Development,” whose aim is to focus on sustainable development outcomes that “place a premium

on broad-based economic growth, democratic governance, game-changing innovations, and sustainable systems for meeting basic human needs.”^[8]

Around the world, sustainable business practices are advocated by such varied organizations as the Global Reporting Initiative, the CERES coalition, and the United Nations Environmental Programme. At the state and local levels, an association of over 1220 local governments from 70 different countries known as ICLEI—Local Governments for Sustainability provides technical consultation, training, and information services to implement sustainable development.^[9] Many cities in the United States have adopted sustainability goals and a number of states have established governing bodies responsible for advancing sustainable practices.^[10]

A SENSE OF URGENCY

Global Footprint Network’s *Living Planet Report 2010* notes that humanity’s ecological footprint has more than doubled since 1966.^[11] This footprint represents the pressures on land and water area caused by the industrial and lifestyle patterns of human population. A challenge for global society is to “decouple” this global footprint from continued economic growth.

World population is expected to reach 9 billion people by 2050.^[12] Almost all of the increase will take place in the developing nations, where hundreds of millions of people are seeking more secure access to food, clothing, and shelter as well as sanitation, education, health care, energy, communication, and consumer goods. Coupled with the increasingly high levels of resource consumption in developed nations, this will place significant stress on global ecosystems. Ironically, the pressures on natural resources in developing nations could impede industrialization and further aggravate income gaps between rich and poor.

In addition to its impact on global warming, the inexorable growth of the global economy is contributing to a number of disturbing trends: sea level is rising, available fresh water and arable land are growing scarce, forests are disappearing, and biodiversity is threatened by changing natural habitats. The impacts of population growth and economic development are detailed in the 2005 Millennium Ecosystem Assessment, which found that 15 of 24 important global ecosystem services are being degraded or used unsustainably.^[13]

Accomplishing sustainable development in an increasingly interconnected global economy will require a holistic “systems approach” to problem solving, as shown in Fig. 2. The challenge of sustainability is to enable continued economic growth while reducing the rates of both resource use and waste generation. For example, companies that import components and feedstocks from overseas must be mindful of the potential environmental and social impacts in their supply chains. Considering the direct and indirect effects of

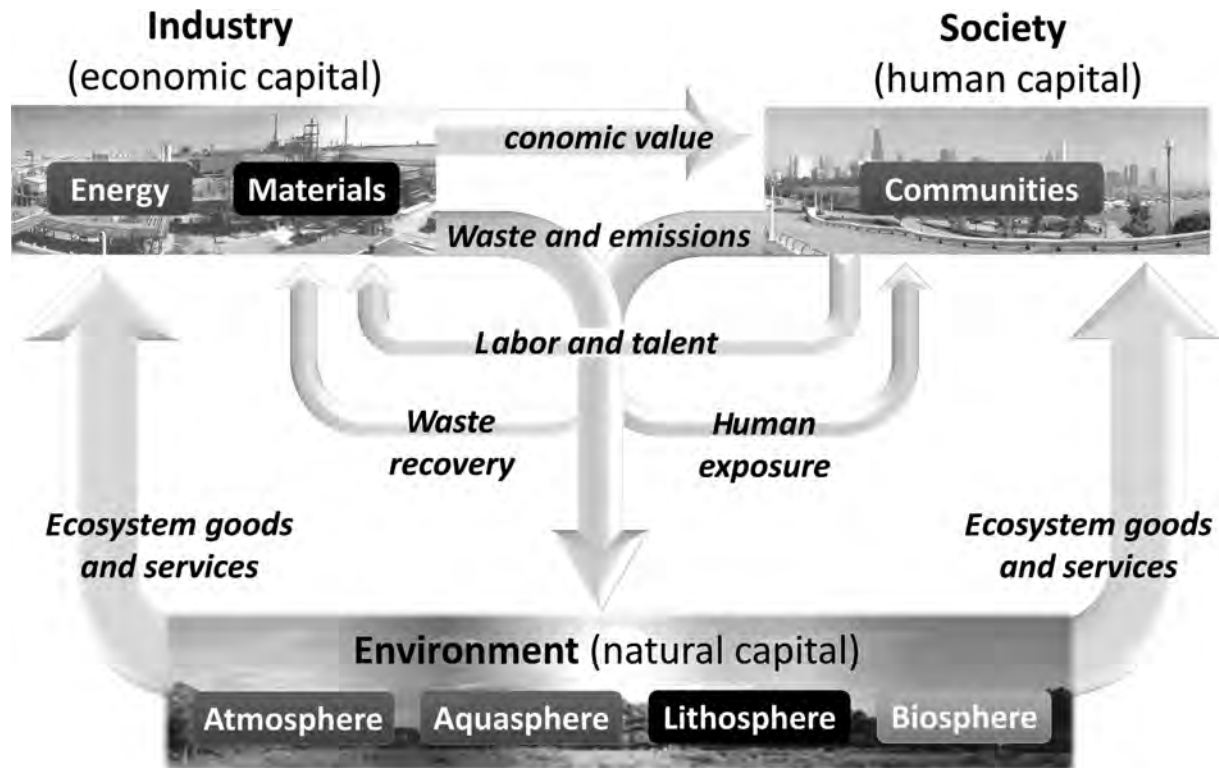


Fig. 2 Systems view of sustainability.

Source: From Center for Resilience, The Ohio State University.

sustainability policies and strategies will help avoid unintended adverse consequences.

OPPORTUNITIES FOR POSITIVE CHANGE

Sustainable practices can enhance and support economic growth. Many businesses have recognized the importance of sustainable practices to support a growing economy. Reducing the use of resources typically leads to reduced purchasing and operating costs, so companies generally view sustainable development as enhancing their bottom line while reducing their environmental footprint. In addition, sustainability enhances shareholder value by improving brand image, reputation, and stakeholder relationships. According to the Dow Jones Sustainability Indexes: "Sustainability is a business approach that creates long-term shareholder value by embracing opportunities and managing risks that derive from economic, environmental and social developments."^[14]

Sustainability strategies have been incorporated into the management objectives of major companies around the world.^[15] A 2009 study published in the *Harvard Business Review* concluded that "sustainability is a mother lode of organizational and technological innovations that yield both bottom-line and top-line returns" and that "there is no alternative to sustainable development."^[16]

The World Business Council for Sustainable Development (WBCSD) has developed an ambitious agenda to assist global industries in moving toward sustainable growth. The WBCSD Vision 2050 report coined the phrase "green race" and outlines a "pathway that will require fundamental changes in governance structures, economic frameworks, business, and human behavior." The report argues that these changes are "necessary, feasible, and offer tremendous business opportunities for companies that turn sustainability into strategy."^[17]

Financial industries including banking, investment, and insurance have also recognized the importance of sustainability and protection of natural resources. For example, the Equator Principles, launched in Washington in June 2003, require that banks assess the social and environmental impacts of projects that they finance. The principles were initially adopted by 10 global financial institutions: ABN AMRO Bank, N.V., Barclays plc, Citi, Crédit Lyonnais, Credit Suisse First Boston, HVB Group, Rabobank Group, The Royal Bank of Scotland, WestLB AG, and Westpac Banking Corporation.

Influenced by advocacy groups such as the Rainforest Action Network, Citigroup has gone beyond the Equator Principles by committing to refuse funding for projects that could result in illegal logging, other environmental damage, or harm to indigenous people.^[18] A report from the non-profit organization CERES lays out a road map for

sustainability based on assessing the company's baseline environmental and social performance, analyzing corporate management and accountability structures and systems, and conducting a materiality analysis of risks and opportunities.^[19]

While these corporate and institutional commitments are valuable, scientific and technological innovation will be critical elements in achieving genuine progress in sustainability. The required scientific knowledge to apply a systems approach will come from many disciplines—including chemistry, biology, and physics as well as economic and social sciences. New approaches such as “transdisciplinary” research and “sustainability science” have been introduced to describe the fusion of multiple disciplines to generate new knowledge, which is important for advancing sustainable innovation.^[20]

There are many excellent examples of innovations that dramatically improve the sustainability of products and processes. For example, traditional wood adhesives often contain formaldehyde, which is toxic to humans. Researchers have been able to develop new adhesives from soy flour, which are environmentally friendly, stronger, and cost-competitive. These new adhesives have replaced more than 50% of formaldehyde-based adhesives and earned a Presidential Green Chemistry Challenge Award.^[21]

Finally, it is important to assure the continued resilience of environmental, economic, and social systems in the face of turbulent change. *Resilience* is defined as the capacity not only to absorb disturbances but also to adapt and reorganize in order to maintain or improve functional capacity.^[22] In order to cope with unpredictable events, whether caused by humans or nature, infrastructure and supply chain systems must be designed for long-term resilience. One example is harnessing ecosystem services to provide “green infrastructure” that helps to protect community-based water resource systems from storms, floods, and other climate disruptions.^[23]

EIGHT STEPS TO SUSTAINABILITY

Sustainability represents the next level of environmental protection.^[24] In previous centuries, the traditional approaches to environmental issues focused on land conservation and risk mitigation. However, the 21st century brings a new set of environmental challenges that are more complex and globally interconnected. These new challenges demand new, more integrated approaches that account for the linkages among environmental resources, national security, human well-being, and economic competitiveness. In addition to regulatory approaches, a variety of strategies to achieve both sustainability and resilience, including public-private collaboration, integrated systems thinking, technological innovation, and adaptive management will be needed.

Effective working toward global sustainability incorporates these eight steps.

Take the Long View

Sustainability requires long-term thinking not only in natural resource management and urban development but also in corporate strategic planning. Taking a long view leads us to consider emerging pressures on ecosystems at different scales of resolution, and how the built environment and ecosystems can be managed in a synergistic way. For industrial innovators, the long view involves thinking about new business models in which material and energy resources can be used more effectively, while wastes and hazardous residuals can be reduced or eliminated.

Understand the System Dynamics

At all levels of society, humans participate in many different complex systems. As a growing population and consumption increase pressures on the ecosystem and the world becomes more tightly connected, small perturbations or changes in any one system can quickly cascade to other systems. Traditional industrial and engineering systems were not created with a systems view and may therefore be vulnerable to unexpected disruptions, such as natural disasters, industrial accidents, sabotage, or terrorism. By understanding system vulnerabilities and leverage points, we can develop cost-effective and resilient management strategies.

Define Sustainability Goals

President Kennedy sets a goal for the United States to land on the moon. This goal was achieved because it was fully supported with the needed resources and policies. Making sustainable development, an explicit policy goal at all levels of government, as well as the private sector, can send a clear message about the need for breakthrough innovation and creative regulatory and compliance approaches that serve the collective interests of the public, business, and government. A commitment to sustainable development can be a driver for a new generation of technology and policy innovation that restores U.S. leadership in the global economy.

Use Effective Tools

In order to practice sustainable development, advanced tools and approaches are necessary. Science and technology are key underlying contributors, along with innovative environmental and economic policies. New instrumentation, data handling, and methodological capabilities have expanded our understanding of the environment and of how complex biological, chemical, and human systems interact. A growing body of economic and environmental

assessment tools is available to support planners and decision makers at all levels of government and industry.

Find the Right Collaborators

Around the world, collaboration and partnerships among stakeholders are crucial to achieving solutions that are less polarized, more economically viable, and focused on balancing short- and long-term goals. Collaboration has begun among government, industry, and non-governmental organizations, and has already yielded many sustainable innovations.

Lead by Example

The profound changes needed to achieve sustainability will require confidence and bold leadership. The most persuasive approach for overcoming uncertainty and hesitation is leading by example—demonstrating that these changes are both realistic and beneficial. Government can help by establishing framework conditions and incentives that encourage innovation and by initiating collaborative pilot projects that bridge knowledge gaps and overcome inertia.

Measure and Track Progress

The use of sustainability indicators is essential for an integrated systems approach to the multifaceted challenges of sustainability. Indicators can help managers and policy makers anticipate and assess key trends, provide early warning of potential disruptions, quantify progress toward sustainability goals, and support decision making about complex trade-offs. A broad spectrum of available indicators is available that captures the economic, environmental, and social attributes of the systems that we strive to manage.

Learn and Adapt

The path to sustainability cannot be planned precisely due to the enormous complexity and uncertainty inherent in global political, economic, social, and natural systems. To navigate this path successfully, we cannot cling to preconceived notions or prescriptive solutions. We must be prepared to learn from experience, rethink our assumptions, and continuously adapt to change. Taking a resilient approach in the short term will enable sustainability in the long term.

CONCLUSION

The approach to energy use and resource management is not sustainable. Moving from the status quo toward sustainability requires new approaches to problem solving that take an entire system into account, rather than isolated

components. The challenge of achieving sustainability is urgent since unsustainable behavior—whether it is economic, social, or environmental—can lead to national and global crises. For example, when withdrawal rates from freshwater aquifers exceed recharge rates, water scarcity is the eventual result. Conversely, sustainable practices can preserve and protect natural capital and enhance economic growth. The challenge of realizing sustainability goals raises a number of practical questions: “What kind of government policies and business strategies can advance us toward sustainability?” “How can business and government work together to advance sustainable practices?” and “What can citizens do to advance the concept of sustainability?” Making sustainability operational is not something a government or business can do alone. A sustainable and thriving economy will require the convergence of four major elements: 1) regulations and policies, 2) advances in science and technology, 3) green business practices, and 4) public support and participation. A global commitment to accelerate movement toward sustainability is essential to safeguard social well-being, shared prosperity, and our natural environment. There is really no alternative.

DISCLAIMER

Views expressed in this entry are those of the authors and do not necessarily reflect the views or policies of the U.S. EPA. Mention of trade names or commercial products does not constitute EPA endorsement or recommendations for use.

ACKNOWLEDGMENT

The authors are grateful to Edward Fallon (Senior Environmental Employment participant at EPA) and Meadow Anderson (American Association for Advancement of Science Fellow at EPA) for their helpful edits and comments.

REFERENCES

1. World Commission on the Environment and Development. *Our Common Future*; Oxford University Press: New York, 1987. Available at <http://www.un.org/documents/ga/res/42/ares42-187.htm> (accessed September 2011).
2. Pinchot, G. *The Fight for Conservation*; Doubleday, Page & Company: New York, 1910.
3. National Environmental Policy Act (NEPA). Available at <http://ceq.hss.doe.gov/nepa/regs/nepa/nepaeqia.htm>.
4. Meadows, D.H.; Meadows, D.K.; Randers, J.; Behrens, W.W. *The Limits to Growth*; Universe Books: New York, 1972.
5. United Nations Division for Sustainable Development. *Agenda 21*, 1992. Available at http://www.un.org/esa/dsd/agenda21/res_agenda21_00.shtml (accessed October 2011).

6. Larsson, A.; Azar, C.; Sterner, T.D.; Strömberg, D.B.; Andersson, B. *Technology and Policy for Sustainable Development*; Centre for Environment and Sustainability at Chalmers University of Technology and the Göteborg University: Göteborg, 2002.
7. Bush, G.W. *Executive Order 13423—Strengthening Federal Environmental, Energy, and Transportation Management 2007*. Available at <http://edocket.access.gpo.gov/2007/pdf/07-374.pdf>; Obama, B.H., Ed.; *Executive Order 13514—Federal Leadership in Environmental, Energy, and Economic Performance, 2009*. Available at <http://edocket.access.gpo.gov/2009/pdf/e9-24518.pdf> (accessed October 2011).
8. ICLEI. *Local Governments for Sustainability*. Available at <http://www.iclei.org/index.php?id=about> (accessed March 2012).
9. Obama, B.H. *Fact Sheet: U.S. Global Development Policy 2010*. Available at <http://www.whitehouse.gov/the-press-office/2010/09/22/fact-sheet-us-global-development-policy> (accessed March and February 2012).
10. Engel, K.H.; Miller, M.L. State Governance: Leadership on Climate Change (Arizona Legal Studies Discussion Paper No. 07-37). In *Agenda for a Sustainable America*; Dernbach, J.C., Ed.; Environmental Law Institute: Washington, 2009; 441 pp. Available at http://papers.ssrn.com/sol3/papers.cfm?abstract_id=1081314 (accessed October 2011).
11. WWF, Global Footprint Network, and ZSL. *Living Planet Report 2010: Biodiversity, Biocapacity, and Development*. Available at <http://www.footprintnetwork.org/press/LPR2010.pdf> (accessed October 2011).
12. UN. *World Population to 2300*. Available at <http://www.un.org/esa/population/publications/longrange2/WorldPop2300final.pdf> (accessed March 2012).
13. Millennium Ecosystem Assessment. *Guide to the Millennium Assessment Reports*; Island Press: Washington, 2005. Available at <http://www.maweb.org/en/index.aspx> (accessed October 2011).
14. Dow Jones Sustainability Indices, in Collaboration with RobecoSAM. *Corporate Sustainability*. Available at http://www.sustainability-index.com/07_html/sustainability/corpsustainability.html (accessed October 2011).
15. Esty, D.; Winston, A.S. *Green to Gold: How Smart Companies Use Environmental Strategy to Innovate, Create Value, and Build Competitive Advantage*; Yale University Press: New Haven, 2006.
16. Nidumolu, R.; Prahalad, C.K.; Rangaswami, M.R. Why sustainability is now the key driver of innovation. *Harvard Bus. Rev.* **2009**, *87*, 57–64.
17. World Business Council for Sustainable Development. *Vision 2020: The New Agenda for Business*. Available at http://www.wbcsd.org/web/projects/BZrole/Vision2050-FullReport_Final.pdf (accessed October 2011).
18. Swiss Re Center for Global Dialogue. Capitalizing on natural resources. In *New Dynamics in Financial Markets*, 9th International Sustainability Leadership Symposium September 10–11, 2008; Swiss Re Centre for Global Dialogue: Rüschlikon, 2008. Available at http://www.sustainability-zurich.org/cm_data/Report_08_V8_final.pdf (accessed October 2011).
19. CERES. *The 21st Century Corporation: The Ceres Roadmap for Sustainability*. Available at <http://www.ceres.org/resources/reports/ceres-roadmap-to-sustainability-2010> (accessed October 2011).
20. Kates, R.W.; T.M. Parris, T.M. Long-term trends and a sustainability transition. *Proc. Nat. Acad. Sci.* **2003**, *100* (14), 8062–8067.
21. U.S. EPA. *The Presidential Green Chemistry Challenge*. Available at <http://www.epa.gov/greenchemistry/pubs/pgcc/presgcc.html> (accessed October 2011).
22. Fiksel, J. Sustainability and resilience: Towards a systems approach. *Sustain. Sci. Pract. Pol.* **2006**, *2* (2), 14–21.
23. U.S. EPA. *Green Infrastructure*. Available at <http://cfpub.epa.gov/npdes/greeninfrastructure/gisupport.cfm> (accessed October 2011).
24. Hecht, A.D. The next level of environmental protection: Business strategies and government policies converging on sustainability. *Sustain. Dev. Law Pol.* **2007**, *7*, 19–25.

Tailings: Minerals Processing Residue Rehabilitation

Lloyd R. Hossner

Soil and Crop Sciences Department, Texas A&M University, College Station, Texas, U.S.A.

Hamid Shahandeh

Texas A&M University, College Station, Texas, U.S.A.

Abstract

There are large areas of abandoned mine tailings from a variety of industrial operations around the world. Tailings contain a large proportion of the original ore that was mined. Tailings are deposited in impoundments with dimensions ranging from several square meters to some that are many square kilometers in area. These tailings have been largely abandoned and allowed to revegetate under natural conditions. There have been both public and scientific awareness of the possible consequences of long-term disposal of tailings. A complete understanding of the physical and chemical properties of tailings is essential for planning successful rehabilitation programs.

INTRODUCTION

Metals have been mined and exploited with the growth of world industry. In particular, iron, lead, zinc, and copper have been mined extensively in regions all over the world. In the process of metal beneficiation, massive heaps of spoil (tailings and waste rock) have been created at mine sites or areas distant from mines (Fig. 1). Tailings and waste rock create esthetic problems in the landscape and affect water, soil, plant, and public health. Tailings is defined as the solid waste product of the milling and mineral concentration process.^[1] Mill tailings are the finely ground host rock materials from which the desired mineral values have been extracted during the concentration process. Generally, tailings are transported from the mill to their place of disposal as a water slurry containing 15–50% solids by weight and discharged by impoundment on land in settling ponds adjacent to the mills, used as backfill in the open pit or underground mine, disposed in deep lakes or offshore, or processed for secondary metal recovery followed by disposal. Tailings impoundments range in size from <10 to >2000 ha and may be stacked as high as 50 m.^[2]

There are a large number of abandoned mine waste and tailings deposits from a wide variety of industries around the world. The total area of land disturbed by mining in China is estimated to be about 2 million hectares.^[3] In the United States between 1930 and 1980 more than 2 million hectares of land were affected by mining operations.^[4] The mine tailings produced in Malaysia, England, Thailand, and Canada are estimated

in billions of tons occupying hundreds of thousands of hectares.^[2]

POTENTIAL ENVIRONMENTAL PROBLEMS

Abandoned spoil and tailings contain the waste products of both mining and ore processing operations. Chemical extractants such as sulfuric acid and sodium bicarbonate are used for uranium ores, cyanide for gold ores, sodium hydroxide for aluminum ores, and sulfuric acid and hydrochloric acids for copper, nickel, and cobalt ores. In addition to these toxic solvents and the dissolved heavy metals, tailings from uranium and phosphate operations can contain radionuclides such as thorium and radium. These materials are often a major source of pollution in the local environment due to dust blow and the potential leaching of the products of mineral weathering into water sources.

Tailings are also subject to the process of weathering and, over time, changes may occur in their properties which could be hazardous to the environment. For example, sulfides are associated minerals that are readily oxidized in the tailings when exposed to air, water, and iron-oxidizing bacteria. One of the oxidation products is sulfuric acid. Tailings containing sulfide minerals may eventually have a pH of 1.5–3.5.^[2] Toxic ions may also contaminate soils and waters adjacent to smelters through seepage, runoff waters, and eroded sediments. Effluents arising from tailings seepage could be toxic in varying degrees to man, animals, and plant life. Also, there is a concern for the amount of heavy metal uptake by plants growing on tailings and its effect on the food chain.^[5]

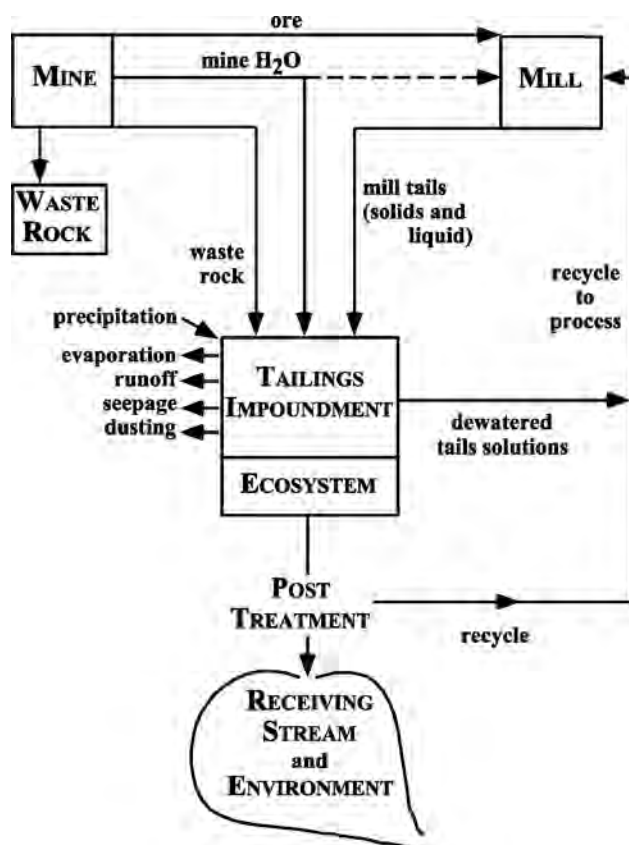


Fig. 1 Mine/mill environment.

Source: From Ritcey.^[2]

MINE WASTE REHABILITATION PRACTICES

Engineering Approach

Mine waste residue and soils contaminated with toxic metals and radionuclides can be remediated and stabilized using engineering approaches including chemical, physical, and thermal techniques (e.g., *in situ* mobilization, immobilization, degradation, and burial; or removal and reburial, vitrification, vacuum extraction, steam flooding, pumping and leaching, electroosmosis, and electroacoustic extraction).^[6] The use of chemical and physical techniques to stabilize mineral wastes against wind and water erosion is limited because of the cost and maintenance.^[7] Physical stabilization of materials such as waste rock from strip mining can be used to reduce wind and water erosion.

Chemical stabilization requires a chemical agent, such as lignin sulfate or resinous adhesive, to react with mine waste to provide a crust resistant to wind and water erosion. Physical stabilization can be performed by protecting or isolating the waste from the environment with physical barriers such as liners and clay caps. Usually, the design and installation of a cover system consists of three zones: an erosion resistant layer, a moisture retention layer, and an

underlying clay barrier to prevent entrance of water into the waste material. The design of the capping sequence and incorporation of a coarse layer to break capillarity are shown to be very important. When waste material poses a threat to groundwater quality or to plant growth, a layer of impermeable plastic or compacted clay may be placed between the waste and the soil cap to prevent movement of water into the toxic wastes. In other materials, it may be necessary to place a diffusion barrier of very coarse material between the toxic waste and the cover soil to break a water column that might permit diffusion of the toxic materials from the waste into the cover soil. Soil caps placed over non-toxic materials may be thinner than soil caps placed over toxic materials.^[8]

Ecological Approach

According to the Center for Minesite Rehabilitation in Australia, the strategies for rehabilitation of mine waste should be based on stabilization and sustainable revegetation of tailings and waste rock in a reconstructed ecosystem.^[9] Tailings must be stabilized before or concurrent with full-scale revegetation. The keys to successful revegetation of metalliferous mill tailings are: 1) obtain a complete understanding of the physical and chemical properties of the tailing material; 2) select an appropriate final land use of the area under consideration; 3) select the most cost effective combinations of amendments, nutrient supplements, cover materials, and plant species to be employed; and 4) monitor the performance of the vegetation and make adjustments as needed. It is widely accepted that the establishment of vegetation is a desirable method for stabilization of mine wastes.^[10,11] Vegetation reduces wind velocity at the surface, captures dust particles, reduces raindrop impact, reduces runoff by increasing infiltration, and reduces the overland flow of water and sediment. Vegetative stabilization also improves the chemical, biological, and physical properties of the mine wastes by increasing the organic matter content, nutrient level, cation exchange capacity, and biological activity.

Vegetation established on mine tailings has several advantages over physical and chemical methods of stabilization. Vegetation is esthetically pleasing and accepted by the public, relatively inexpensive, less disruptive of site remediation, creates a beneficial habitat for wildlife, plant roots and shoots can take up heavy metals, and plants may stimulate microbial immobilization of heavy metals in the rhizosphere. Thus, establishment of vegetation in mining areas has the potential of reducing contamination of adjacent soil, surface water, and ground water.^[8]

The main approaches to revegetation have been summarized in Table 1.^[7] The waste characteristics of a site determine which approach is most suitable. Generally there are two approaches to revegetation which have been used in

Table 1 Approaches to revegetation of minerals processing residue (tailings).

Waste characteristics	Reclamation technique	Problems encountered
Low toxicity: total metal content <0.1%. No major acidity or alkalinity problems	Amelioration and direct seeding with agricultural or amenity grasses and legumes. Using traditional or specialized techniques	Probable commitment to a medium/long-term maintenance program. Grazing management must be monitored
Low toxicity and climatic limitations: toxic metal content <0.1%. No major acidity or alkalinity problems, extreme of temperature, rainfall, etc.	Amelioration and direct seeding with native species. Seed or transplant ecologically adapted native species using amelioration treatments where appropriate	Irrigation often necessary at establishment. Expertise required on the characteristics of native flora
High toxicity: toxic metal content >0.1%. High salinity in some cases	1) Amelioration and direct seeding with tolerant ecotypes. Apply lime, fertilizer and organic matter, as necessary, before seeding 2) Surface treatment and seeding with agricultural or amenity grasses and legumes. Amelioration with 10–50 cm of innocuous mineral waste and/or organic material. Apply lime and fertilizer as necessary	Regular fertilizer application. Few species have evolved tolerance and few are available commercially. Grazing management not possible Regression will occur if depths of amendments are shallow or if upward movement of metals occurs. Availability and transport costs may be limiting
Extreme toxicity: very high toxic metal content. Intense salinity or acidity	Isolation: surface treatment with 30–100 cm of innocuous barrier and surface binding with 10–30 cm of a suitable rooting medium. Apply lime and fertilizer as necessary	Susceptibility to drought according to the nature and depth of amendments. High cost and potential limitations of materials availability

Source: From Tordoff, Baker, et al.^[7]

combination or separately. They are adaptive and ameliorative approaches. The adaptive approach to revegetation is to combat the toxicity of the waste by direct seeding with metal tolerant cultivars. The metal tolerant plants may be of great benefit to developing countries for low-cost revegetation. The other approach is ameliorative and the toxicity is avoided or diluted, rather than tolerated by using some form of covering system. This approach has been widely used and subjected to considerable research. The two types of covering material are ameliorants and inert amendments. Ameliorants are materials such as sewage sludge, compost, domestic refuse, peat, and topsoil. Inert ameliorants include materials such as clay, shale, or gypsum. Inert ameliorants are commonly wastes from other industry or mining activities. This approach has the added benefit of using one category of waste to overcome the problems of other forms of waste.^[12]

Although revegetation is desirable, metal wastes can present a very unfavorable environment for plants because of the presence of many growth limiting factors such as high salinity, metal toxicity, or nutrient deficiency in the mine tailings and soils.^[13]

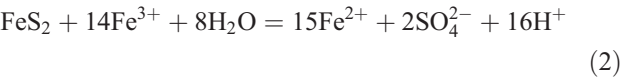
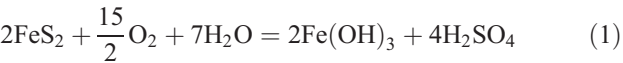
CHARACTERISTICS OF MINE WASTE THAT LIMIT REHABILITATION

Characteristics of tailings from 43 selected mine waste sites are presented in Table 2.^[14] Alleviation or modification of each chemical and physical limitation of the tailing is required prior to the establishment of vegetation.

Chemical Properties

The chemical composition of tailings depends on the original ore mineralogy, extraction techniques, and associated minerals. Among the minerals present in tailings, sulfides are often an important constituent that must be considered in tailings management.

Mining often exposes the sulfide-bearing minerals (pyrite, marcasite, pyrohotite, chalcopyrite, arsenopyrite, cobalite) to the atmosphere. In the presence of water, iron sulfide oxidation by weathering (Eq. 1) and by iron oxidizing bacteria catalysis (Eq. 2) will convert the sulfides to sulfuric acid.^[15]



Large amounts of lime, between 10–150 t ha⁻¹, may be required to neutralize the acidity produced in sulfidic tailings.^[5]

Various salts may appear on the surface of tailings depending on the nature of the original mill process. The factors which are involved in the appearance of salts on the tailings surface include: excess concentrations of soluble salts in tailings materials, availability of shallow subsurface water, salt concentrations within the tailing water, and temperature and chemical potential gradients between the surface and the interior of the tailing. The upward migration of

Table 2 Mean and range of values for selected physical and chemical characteristics of tailings.

Property	Unit	Mean	Range
Particle size distribution			
<2 mm	%	95	20–100
Sand	%	51	1–97
Silt	%	43	0–96
Clay	%	7	0–40
Moisture retention (bar)			
0.1	%	22	0–55
0.3	%	18	0–55
15	%	4	0–20
Available water holding capacity	%	16	0–35
Bulk density	g cm ⁻³	1.5	0.2–3.1
Particle density	g cm ⁻³	2.91	0.01–4.29
pH	6.2	1.8–9.4	
Cation exchange capacity	cmolc/kg	2.63	0.19–46.5
Organic matter	%	2	0.02–25
Electrical conductivity	dS/m	2	0.1–22.4
Available nutrients			
Phosphorus (P)	mg/kg	10	1–400
Potassium (K)	mg/kg	63	1–564
Calcium (Ca)	mg/kg	11,930	40–52,480
Magnesium (Mg)	mg/kg	230	15–1328
Total analysis			
Nitrogen (N)	%	0.013	0.001–0.166
Sulfur (S)	%	4.02	0.001–38.87
Iron (Fe)	%	15.5	0.4–56.81
Aluminum (Al)	%	2.8	0.1–8
Calcium (Ca)	%	1.7	0.01–10.95
Magnesium (Mg)	%	1.2	0.04–5.0
Sodium (Na)	%	0.5	0.01–2.9
Potassium (K)	%	0.7	0.04–3.32
Manganese (Mn)	%	0.2	0.01–4.0
Silicon (Si)	%	22	4–37
Cadmium (Cd)	mg/kg	38	2–280
Chromium (Cr)	mg/kg	1000	70–7,000
Cobalt (Co)	mg/kg	1140	100–9,999
Molybdenum (Mo)	mg/kg	70	10–800
Nickel (Ni)	mg/kg	96	10–546
Lead (Pb)	mg/kg	340	0.3–2,810
Titanium (Ti)	mg/kg	2500	200–10,000
Zinc (Zn)	mg/kg	510	1–5000
Copper (Cu)	mg/kg	130	1–750

salts and their inhibiting effect on root growth and revegetation have been major concerns in the reclamation of many mine tailings. For example, a common problem associated with the establishment of vegetation in asbestos tailings, bentonite tailings, and red mud from bauxite ore processing for alumina is salinity and/or sodicity. These wastes can be highly alkaline (pH > 10), saline (EC > 30 dS m⁻¹), and sodic. Vegetation establishment on these tailings is difficult because of salinity, alkalinity, clay dispersion and low hydraulic conductivity.^[5]

Mine wastes are usually deficient in major plant nutrients and almost universally deficient in nitrogen. Nutrient deficiencies, especially nitrogen, phosphorus and potassium, are often the principal limiting constraints to revegetation of kaolinitic china clay, iron, copper, gold, silver, and other heavy metal mine wastes.^[5] The success of establishing plants on tailings depends on providing an adequate supply of plant nutrients. High levels of the major essential nutrients may reduce the harmful effects of metal ions.

Toxic ions are frequently present in tailings of heavy metals in sufficient concentrations to prevent plant growth unless considerable amelioration is undertaken. Toxic ions decrease root respiration, limit water and nutrient uptake, reduce enzymatic activity and microbial populations, and inhibit cell mitosis in root meristematic regions.^[15] Toxic ions may also contaminate soils and waters adjacent to tailings through seepage, runoff waters, and eroded sediments. Another problem with heavy metal contaminated tailings is the possible uptake of metals by plants in quantities that could be toxic in the general food chain. Radioactivity associated with uranium and phosphate tailings and high concentrations of arsenic, mercury, and cyanide in silver and gold mine tailings are other environmental concerns.

Physical Properties

The physical properties of tailings vary with the mineral being processed, the origin of the ore body, and the process used for mineral concentration. Physical properties are the most important in determining productivity of vegetated mine wastes because of the cost involved in trying to ameliorate particle size distribution and water-holding capacity. Non-uniform texture is the main physical problem of mine wastes which limits the availability of water to plants. Mining generates a wide range of particle size materials. This includes coarse mine wastes, fine clays, flotation tailings, chemical precipitates, and slimes. In the mineral industry the slime size fraction is < 5 µm.

Fine texture is a major problem in gold tailings. Gold tailing is finely crushed (0.01–0.1 µm) because of the large surface area needed for fast reaction with processing chemicals. It is relatively easy to establish vegetation on these tailings but it is difficult to obtain full ground cover because young plants are sand blasted by finely crushed, wind

blown materials. Laying large rock fragments on the slope of sand deposits and installing reed wind breaks on the entire sand deposit has helped the establishment of vegetation.^[2] A critical factor in reclaiming and vegetating a tailing is the availability of moisture for plant growth. Moisture curve characteristics of tailings derived from iron, nickel, copper, lead, zinc, and gold mines showed properties similar to a sandy loam soil, an indication of potential water deficiency for plant growth.^[2]

Vegetation establishment and maintenance on coarse tailings, where there will be little water retention and rapid drainage, are also related to water availability. Establishment of vegetation on very coarse tailings is difficult but it can be achieved by adding a soil cover followed by hydro-mulching and hydroseeding.^[15]

Crusting, cracking, and a general lack of structure are common characteristics of mine tailing brought about by differences in texture, lack of organic matter, and variable mineralogy. Structure determines the bulk density of tailings. Water infiltration is often limited in fine textured tailings due to poor structural characteristics. Root penetration and moisture stress of plants due to limited rooting generally becomes a problem with dry bulk density values above 1.5 Mg m^{-3} in tailings.^[15]

Tailings have a high heat capacity. Tailings exposed to direct solar radiation can have temperatures of 55–65°C at 1–2 cm depth. Internal temperature in a silver mine in Canada reached 45°C during the oxidation process. Hay or straw mulch has been an effective insulator for stabilizing the tailings temperature.^[2]

CONCLUSION

There are large areas of abandoned mine tailings from a variety of industrial operations around the world. Tailings contain a large proportion of the original ore that was mined. Tailings are deposited in impoundments with dimensions ranging from several square meters to some that are many square kilometers in area. These tailings have been largely abandoned and allowed to revegetate under natural conditions. There have been both public and scientific awareness of the possible consequences of long term disposal of tailings. Now many countries have enacted legislation to ensure reclamation of tailings of disposal sites once the mining and milling operations have ceased. A complete understanding of the physical and chemical properties of tailings is essential for planning successful rehabilitation programs.

REFERENCES

1. Richmond, T.C. The revegetation of metalliferous tailings. In *Reclamation of Drastically Disturbed Lands*; Barnhisel, R.I., Darmody, R.G., Daniels, W.L., Eds.; American Society of Agronomy: Madison, 2000; 801–818.
2. Ritcey, G.M. *Tailings Management: Problems and Solutions in the Mining Industry*; Elsevier: New York, 1989.
3. Ye, Z.H.; Wong, J.W.C.; Wong, M.H. Vegetation response to lime and manure compost amendments on acid lead/zinc mine tailings. *Restor. Ecol.* **2000**, *8*, 289–295.
4. Johnson, W.; Paone, J. *Land Utilization and Reclamation in the Mining Industry, 1930–1980*; U.S. Bureau of Mines Information Circular 8862; United States Printing Office: Washington, 1982; 22 pp.
5. Hossner, L.R.; Shahandeh, H. Chemical and physical limitations to mine tailings reclamation. *Trends Soil Sci.* **1991**, *1*, 291–305.
6. Francis, A.J.; Dodge, C.J. Remediation of soils and wastes contaminated with uranium and toxic metals. *Environ. Sci. Technol.* **1998**, *32*, 3993–3997.
7. Tordoff, G.M.; Baker, A.J.M.; Willis, A.J. Current approaches to the revegetation and reclamation of metalliferous mine wastes. *Chemosphere* **2000**, *41*, 219–228.
8. Menzies, N.W.; Mulligan, D.R. Vegetation dieback on clay-capped mine waste. *J. Environ. Qual.* **2000**, *29*, 437–442.
9. Bell, L.C. The Australian centre for minesite rehabilitation research: an initiative to meet the strategic research needs for sustainable mining rehabilitations. *Water Air Soil Pollut* **1996**, *91*, 125–133.
10. Harris, J.A.; Birch, P.; Palmer, J. *Land Restoration and Reclamation: Principles and Practices*; Addison-Wesley/Longman: Harlow/Essex, 1996; 230 pp.
11. Munshower, F.F. *Practical Handbook of Disturbed Land Revegetation*; Lewis/CRC Press: London/Boca Raton, 1994.
12. Bradshaw, A.D.; Johnson, M.S. Revegetation of metalliferous mine waste: The range of practical techniques used in western Europe. In *Minerals, Metals and the Environment*; Anthony, M.T., Ed.; Institute of Mining and Metallurgy: London, 1992; 1–11.
13. Johnson, M.S.; Cooke, J.K.W.; Stevesson, J.K.W. Revegetation of metalliferous wastes and land after metal mining. In *Mining and Environmental Impact; Issues in Environmental Science and Technology*; Hester, R.E., Harrison, R.M., Eds.; Royal Society of Chemistry: London, 1994; 31–48.
14. Murray, D.R. *Pit Slope Manual. Supplement 10–1; Reclamation by Revegetation*; Report No. 77–31, Vol. 1, Mine Waste Description and Case Histories; CANMET (Can. Center Min. Energy Technol.): Toronto, 1977.
15. Hossner, L.R.; Hons, F.M. Reclamation of mine tailings. *Adv. Soil Sci.* **1992**, *17*, 311–349.

Taxonomies: U.S. Soil Taxonomy and Indian Scenario

Tapas Bhattacharyya

Dr Balasaheb Sawant Konkan Krishi Vidyapeeth, Dapoli, India

Dilip Kumar Pal

Division of Soil Resource Studies, National Bureau of Soil Survey and Land Use Planning, Nagpur, India

Abstract

The National Bureau of Soil Survey and Land Use Planning (Indian Council of Agricultural Research) as a premier soil survey institute has been using the U.S. Soil Taxonomy (USST) to group Indian soils for agricultural land use planning. USST is a user-friendly platform for both pedologists and edaphologists because of its an open-ended system, which helped Indian soil scientists over the years to contribute their thoughts to enrich USST by describing Indian tropical soils by improvising its rationale. Major research findings show the Oxisols are more of a concept than reality. Classification of Indian soils has consistently been revised with special reference to low activity clay, saline, sodic, and hydromorphic soils.

INTRODUCTION

The U.S. Soil Taxonomy (USST) of the U.S. Department of Agriculture (USDA) is an elaborate, universally acceptable, hierarchical system of soil classification showing well-defined differentiating criteria based on measurable soil and associated land characteristics. Higher categories in USST depend on various properties that are produced by distinct soil-forming processes which are well defined by the nomenclature of different taxa. Moreover, USST is an open-ended system that accommodates new concepts developed over time. The system, as the soil itself, is dynamic as evidenced by its continuous revision since its inception^[1–3] which was adopted in India in 1969.^[4]

USST SOIL ORDERS IDENTIFIED IN INDIA

The soil resource mapping programme (1986–1996) in India generated database on soils, their area and characteristics following USST.^[5] Seven soil orders are reported from India. The occurrence of soil orders, sub-orders, great groups, subgroups, and families are detailed in Fig. 1. During the last two decades focus of research on Indian soils has changed qualitatively because mineralogical, micromorphological, and age-control tools have been used to measure the subtle pedogenetic processes of the present and past geological periods.^[6] It helped to understand the basic issues in soil genesis through USST for efficient use and agricultural management.

Black Soils (Vertisols) and Their Intergrades

Deep to very deep black soils (Vertisols), popularly known as black cotton soils, dominated by smectitic clays, are

characterized by the presence of either slickensides or wedge-shaped peds with $\geq 30\%$ clay and cracks that open and close periodically. A group of soils belonging to other soil orders possess the characteristics of black soils showing linear extensibility (LE) of 6.0 cm or more. High LE values are caused by smectitic clays that allocate these soils to vertic subgroup. The presence of slickensides is not a must for classifying a soil into vertic intergrades.^[3] Studies show that red Vertisols and their intergrades also exist, suggesting the term black soils for Vertisols and vertic intergrades are no more tenable and shrink–swell (SS) soils are appropriate to address them. The estimation indicates that Indian SS soils occupy nearly 76.4 Mha.^[5] Studies explained that the slow and steady process of haplodization induced by argillipedoturbation inhibits the process of horizonation and favors the development of Vertisols with characteristic cyclic horizons.^[4] Extensive pedological research during the last 15 years on Indian Vertisols shows clay illuviation as a more dominant pedogenic process than the argillipedoturbation thereby suggesting the subsoil horizon designation as B⁴ instead of A¹. Besides, soil taxonomists in India observed characteristics of the “mollic” epipedon in many benchmark Vertisols of central India.^[4,6]

For the manifestation of vertic properties, the minimum threshold value of clay smectite has been reported as 20. By and large, Indian Vertisols are calcareous and these calcareous soils cover 229 Mha area.^[4,6] Keeping in view the ill effects of excess carbonates on crop performance, USST has a provision for carbonatic mineralogy class at the family level for soils containing $>40\%$ carbonates. Interestingly, Indian soils with $>40\%$ carbonates are sometimes endowed with calcium (Ca)-rich zeolites which offset the ill effects of excess carbonates.^[4,6] In view of contemporary natural chemical degradation process in the presence of soil modifiers like Ca-zeolites and gypsum, the mineralogy

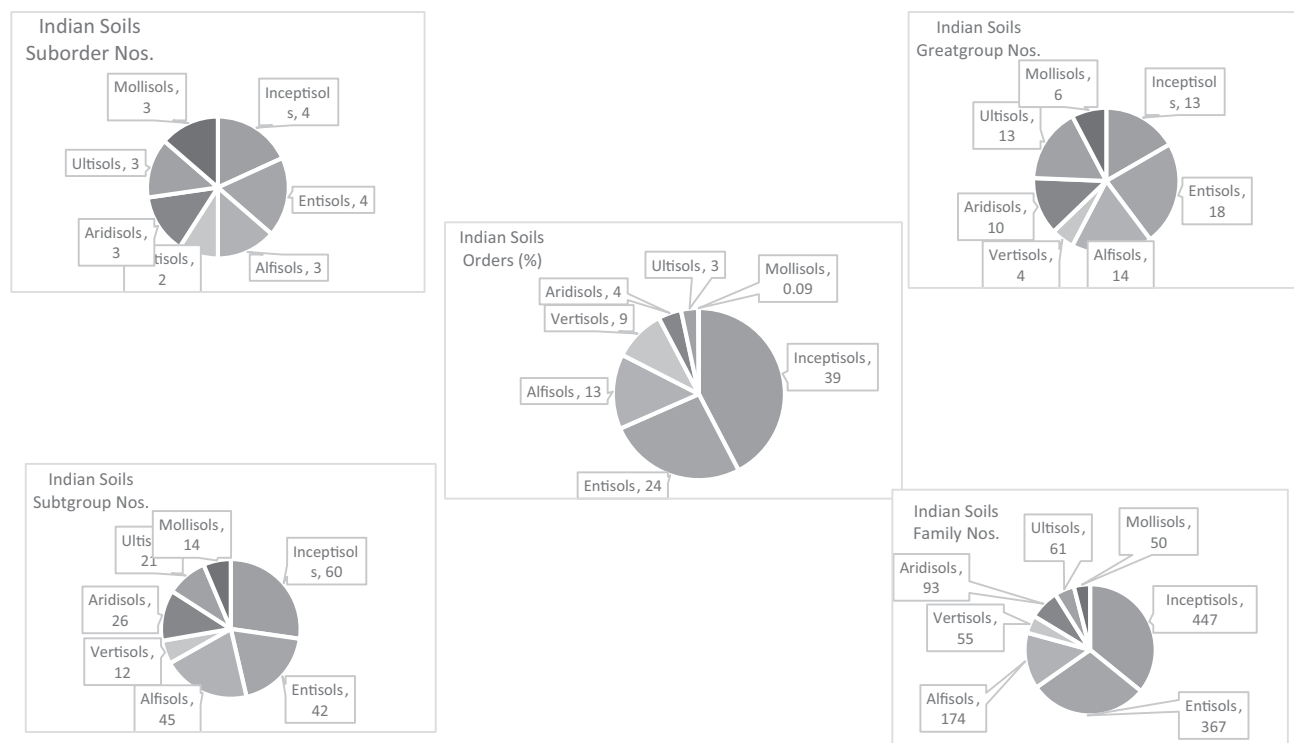


Fig. 1 Total orders (a), sub-orders (b), great groups (c), subgroups (d), and families (e) of Indian soils.

class in USST should, therefore, be based on pedo-edaphic data sets.^[4] The rationale for using exchangeable sodium percent (ESP) as 15 or more in one or more horizons within 40 cm of its upper boundary has been debated for diagnostic subsurface horizons like the natric.^[2] It has been observed that even an ESP of 5, highly smectitic Vertisols develop poor drainage that impairs crop productivity. This is in contrast to non-smectitic soils which do not have the drainage problem even at the ESP of 50 and above,^[4] and Indian Vertisols though that have ESP > 15, indicating their sodic character but do not affect crop growth because of the presence of natural Ca-zeolite.^[8] The lower limit of sodicity at ESP > 40 for soils of the Indo-Gangetic Plains (IGP) at ESP > 5 but < 15 for Indian Vertisols or the limit of ESP 6 for Australian soils or > 15 for all soil types appears to be redundant for the zeolitic sodic Vertisols of India. In view of the pedogenetic processes controlling the hydraulic properties of soils mediated through dispersibility, grouping soils on the basis of saturated hydraulic conductivity (Ks) appears logical using a threshold value of Ks < 1 cm h⁻¹.

Brown Forest Soils (Mollisols) in Tropical Climate

Acidic and fairly weathered Mollisols in the zeolitic Deccan basalt areas were reported in the hills of central and western India under forest in the tropical humid climate.^[9] Formation and persistence of Mollisols in the hills of central and western India expand the basic rationale of the USST for the formation of a group of organic matter-rich, dark-colored, Ca-saturated, soft, clayey, smectitic, but

acidic Mollisols. The formation of Mollisols in the zeolitic basalt landscape over millions of years and their persistence in central and western India demonstrate that the quality of parent materials prevents the transformation of smectite to kaolin, helps in the retention of adequate amount of smectite, and provides continuous supply of bases (Ca²⁺ ions) required for the formation of Mollisols even under tropical humid climate. The foregoing suggests that although the soils in the study area are formed in humid tropical climate for millions of years, they have not reached the advanced stage of weathering represented by Ultisols and Oxisols; instead, they have remained Mollisols.

Alfisols, Ultisols, Oxisols, and Their Classification

Owing to the serious difficulty for field identification of clay skins in highly weathered soils, the kandic horizon introduced in Soil Taxonomy was often used for classification of highly weathered humid subtropical soils in north-eastern India.^[4] The “kandi” group of Alfisols and Ultisols is common in India. The introduction of this subsurface diagnostic horizon and inclusion of “kanhapi” intergrade helped to group many low-activity clay Inceptisols into Alfisols (>35% base saturation) or Ultisols (<35% base saturation). Although the Inceptisols have been grouped properly as Kandi (c) Alfisols and/or Ultisols, the formation and existence of Oxisols have been debated.^[4,9] The kandic horizon provides a basis for differentiation among soils with clay accumulation in the subsoils. The presence of

an argillic horizon could not explain the diagnostic criteria to differentiate all Ultisols and Alfisols from Oxisols and Inceptisols. The proposed kandic horizon is such a diagnostic horizon that it can separate low cation exchange capacity (CEC) Ultisols and Alfisols from the high CEC Ultisols and Alfisols. The important property of Oxisols is that they should be almost devoid of weatherable minerals (<10%), and thus further weathering may not supply nutrients for sustaining plant growth. Moreover, the advanced stage of weathering might obliterate the boundary differentiation between horizons and as such clay films may be absent. The conditions for their formation in the tropical climate with stable landscape and siliceous/acidic parent material are available in the Indian subcontinent; Oxisols are not reported.^[4,9] An in-depth study of international reference on the laterites (Ultisols) in the state of Kerala indicated inconsistency in soil grouping (order) and the assignment of mineralogy class,^[10] and thus it was envisaged that the transformation of Ultisols to Oxisols with time could be difficult to reconcile not only in the tropical part of India but also elsewhere.^[4,9–11] Many of the micaceous soils of the IGP of northwestern India are sodic and have clay enriched textural B horizon without any clay skins and grouped in Inceptisol order. Decrease in clay mica (<2 μm) with depth in the profile can confirm clay illuviation for these soils to group them in Alfisols.^[4,11] A new subgroup (Alumic Hapludults) was proposed for soils of the Shillong Plateau, India, supporting the growth of pine forest with high extractable Al^{3+} content (>50%) in the Bt horizon. The concept of a Modic subgroup in Ultisols, showing an excellent environment for aerobic respiration, has also been proposed.^[4] In the humid tropical weathering environment in India, the presence of vermiculite/high charge smectite is common. Minerals in clay fractions have not weathered to reach the stage of kaolinite suggesting mixed mineralogy class as appropriate.

Inceptisols and their classification

Inceptisols occupy 40% and 15.2% in India and world, respectively.^[10,11] The concept of grouping shallow black soils into a Leptoveritic subgroup^[4] has finally found a place in the Vertisol order as Leptic Haplusterts.^[2] Out of the three types of saturation such as endo, epi, and anthric saturation,^[2] the first two found places in the great group (for instance, in Inceptisols, they are Endoaquepts and Epiaquepts, respectively). Anthric condition has been referred as a variant of episaturation, which is usually associated with controlled flooding (in India, e.g., for crops like wetland paddy). This condition causes reduction process in the saturated and puddled surface soils with alternate oxidation during unsaturated conditions. Although the “Anthraquic” condition has been included in Soil Taxonomy, this special type of saturation does not find any place in the great group/subgroup level. In absence of that, most of the soils in the IGP and other rice growing areas of India qualify for episaturation.^[4]

Salt-Affected Soils and Their Classification

The sodic (alkali) soils of the north-west part of the IGP with high salts, ESP, pH, chromas and yellower hues, key out as Typic or Aquic Calciorthids, Camborthids (Aridisols), and Haplustalfs (Alfisols), and this grouping does not spell out their saline-sodic nature to set these soils apart at some high categoric level in the system required for land use recommendations.^[4] Accordingly structural requirements for the Natric horizon were proposed to be modified to include horizons with high ESP (≥ 40) but having simple blocky structure with or without tongues of eluvial material. New subgroups, viz., Natric, within the orders of Inceptisols, Alfisols, and Aridisols, are suggested for the high sodium-saturated soils lacking Natric horizons. For similar practical considerations, the high concentrations of salts in soils when associated with high ESP pose problems in leaching and consequently new subgroups, viz., Salic and Salic Natric, within the orders of Aridisols and Alfisols, were suggested. Salt-affected soils occupy 6.65 Mha area in India, nearly 36% of which occur in the IGP.^[4] The soils of floodplains irrespective of containing water soluble salts are classified as Aquepts and Ustepts. Since USST^[5] may not serve the purpose of differentiating non-saline/non-sodic soils with saline and/or sodic soils with special reference to the IGP, India,^[4,6] sodic intergrade was proposed for soils with ESP >15. Similarly for the salt-affected soils of active alluvial plains of the IGP, Salic and Sodic intergrades were proposed at the subgroup level for the meaningful interpretation of soil management.^[4]

Paleosols and polygenetic soils and their classification

Paleosols are not covered by the Soil Taxonomy, since it was not endorsed by the USDA.^[2] Paleopedologists made efforts to apply Soil Taxonomy for the study of Paleosols. Paleosols are the soils formed in a geological age and can be either buried or non-buried.^[4] and can be either buried or non-buried. Soil Taxonomy recognizes a soil as buried when it is covered by new soil material of at least 50 cm thickness.^[2] The Paleopedological Sub-Commission recommended that only when the type or direction of soil-forming processes changes, the soils are recognized as Paleosols. Besides, there are polygenetic soils showing different diagnostic features indicating more than one climatic episode. Such soils, for example, ferruginous soils of southern Peninsular India, overlying the saprolites of metamorphic rocks, dominated by kaolinite or dioctahedral smectites have been reported as relict Paleosols^[8] and related color and ratios of various sand sizes with silt to indicate Paleosols from southern Peninsular India.^[4] These relict soils have been influenced by the climatic change from humid to drier conditions during the Plio-Pleistocene transition period.

Alluvial soils which are older than 2,500 years BP of the central IGP are relict Paleosols that experienced three climatic episodes during the Holocene period leading to different types of soils. A series of Vertisols in a climo-sequence (ranging from humid to arid bio climates), from Typic Haplusterts to Udic/Aridic/Sodic Haplusterts and finally to Sodic Calcicusterts, are in place to suggest that USST can be a useful tool to trace the signatures of climate change. Buried soils from Andhra Pradesh (AP), India, formed by allochthonous overburden, represent a double profile.^[4] A Thapto-Tropofluventic subgroup of a Vertisol (Haplustert) and of an Inceptisol (Ustropept/Haplustept) has been proposed. Soil Survey Staff^[5] used thapto (GK Thaptein meaning bury) to represent a buried Histosol or a buried histic epipedon, whereas “thapto” in AP soils was used to indicate any buried Entisol with fluvial characteristics in the tropical climate and also a Thapto-Haplustalfic subgroup for a buried Alfisol.^[8] There are efforts to identify the Paleosols according to the approaches of Soil Taxonomy, first by recognizing the diagnostic horizons and other pedogenic features. However, because many ancient buried Paleosols are often incomplete and difficult to reconstruct, even experienced workers find it difficult to recognize diagnostic features of the Soil Taxonomy or the Food and Agriculture Organization scheme. In spite of this difficulty, the paleopedology community favored developing a separate soil group.^[4]

CONCLUSION

Different kinds of soils in India indicate that the soil diversity is quite large because of variability of several factors of soil formation. USST helped to fine-tune the basic differences among Indian soils, since the concept of each taxonomic class has always remained purpose specific. Since inception, many researchers have found that the USST is useful in terms of grouping a particular type of soil and this has immensely enriched the soil classification literature to understand the basics of soils, in general, and tropical soils, in particular. Amid the renaissance in soil science, a massive demand would be to appropriately manage tropical soils for their restoration and preservation, and the USST has an important role to play in this endeavor.

REFERENCES

1. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, Agriculture Handbook No. 436; Soil Conservation Service, U.S. Dept. of Agriculture: Washington, DC, 1975.
2. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; Agriculture Handbook No. 436; U.S. Govt. Printing Office: Washington, DC, 1999.
3. Soil Survey Staff. *Keys to Soil Taxonomy*, 12th Ed.; U.S. Department of Agriculture, Natural Resource Conservation Service: Washington, DC, 2014.
4. Bhattacharyya, T.; Chandran, P.; Ray, S.K.; Pal, D.K. Soil classification following the US taxonomy: An Indian commentary. *Soil Horizon*. **2015**, 1–16. DOI:10.2136/sh14-08-0011.
5. Bhattacharyya, T.; Sarkar, D.; Ray, S.K.; Chandran, P.; Pal, D.K.; Mandal, D.K.; Prasad, J.; Sidhu, G.S.; Nair, K.M.; Sahoo, A.K.; Das, T.H.; Singh, R.S.; Mandal, C.; Srivastava, R.; Sen, T.K.; Chatterji, S.; Patil, N.G.; Obireddy, G.P.; Mahapatra, S.K.; Anil Kumar, K.S.; Das, K.; Singh, A.K.; Reza, S.K.; Dutta, D.; Srinivas, S.; Tiwary, P.; Karthikeyan, K.; Venugopalan, M.V.; Velmourougane, K.; Srivastava, A.; Raychaudhuri, M.; Kundu, D.K.; Mandal, K.G.; Kar, G.; Durge, S.L.; Kamble, G.K.; Gaikwad, M.S.; Nimkar, A.M.; Bobade, S.V.; Anantwar, S.G.; Patil, S.; Sahu, V.T.; Gaikwad, K.M.; Bhondwe, H.; Dohre, S.S.; Gharami, S.; Khapekar, S.G.; Koyal, A.; Sujatha, P.; Reddy, B.M.N.; Sreekumar, P.; Dutta, D.P.; Gogoi, L.; Parhad, V.N.; Halder, A.S.; Basu, R.; Singh, R.; Jat, B.L.; Oad, D.L.; Ola, N.R.; Wadhwa, K.; Lokhande, M.; Dongare, V.T.; Hukare, A.; Bansod, N.; Kolhe, A.; Khushpure, J.; Kuchankar, H.; Balbuddhe, D.; Sheikh, S.; Sunitha, B.P.; Mohanty, B.; Hazarika, D.; Majumdar, S.; Garhwal, R.S.; Sahu, A.; Mahapatra, S.; Puspamitra, S.; Kumar, A.; Gautam, N.; Telpande, B.A.; Nimje, A.M.; Likhar, C.; Thakre, S. Soil information system: Use and potentials in humid and Semi-Arid Tropics. *Curr. Sci.* **2014**, *107*, 1550–1564.
6. Bhattacharyya, T.; Pal, D.K.; Mandal, C.; Chandran, P.; Ray, S.K.; Sarkar, D.; Velmourougane, K.; Srivastava, A.; Sidhu, G.S.; Singh, R.S.; Sahoo, A.K.; Dutta, D.; Nair, K.M.; Srivastava, R.; Tiwary, P.; Nagar, A.P.; Nimkhedkar, S.S. Soils of India: Historical perspective, classification and recent advances in knowledge; A review. *Curr. Sci.* **2013**, *104*, 1308–1323.
7. Pal, D.K.; Bhattacharyya, T.; Ray, S.K.; Chandran, P.; Srivastava, P.; Durge, S.L.; Bhuse, S.R. Significance of soil modifiers (Ca-zeolites and gypsum) in naturally degraded Vertisols of the Peninsular India in redefining the sodic soils. *Geoderma* **2006**, *136*, 210–228.
8. Bhattacharyya, T.; Chandran, P.; Ray, S.K.; Mandal, C.; Tiwary, P.; Pal, D.K.; Wani, S.P.; Sahrawat, K.L. Processes determining the sequestration and maintenance of carbon in soils: A synthesis of research from tropical India. *Soil Horizons*. **2014**, 1–16. DOI:10.2136/sh14-01-0001.
9. Pal, D.K.; Wani, S.P.; Sahrawat, K.L.; Srivastava, P. Red ferruginous soils of tropical Indian environments: A review of the pedogenic processes and its implications for edaphology. *Catena*. **2014**, *121*, 260–278. DOI:10.1016/j.catena.2014.05.023.
10. Bhattacharyya, T.; Nagar, A.P.; Nimkhedkar, S.S.; Sarkar, D.; Sehgal, J.; Velayutham, M.; Gajbiye, K.S. *Soil Taxonomic Database of India and the States (1:250,000 scale)*. NBSSLUP Publication No. 143; NBSS&LUP: Nagpur, 2009; 266 pp.
11. USDA. *United States Department of Agriculture, Natural Resources Conservation Service; Soil Survey Division-World Soil Resources*: Washington, 2006.

Taxonomy

Mark H. Stolt

Department of Natural Resources Science, University of Rhode Island, Kingston, Rhode Island, U.S.A.

Abstract

Soil, like any natural resource, needs to be classified in order to best utilize and conserve the resource. As long as humans have lived in communities, soils have been classified. In most cases, the classification approach is strongly influenced by the concept that the most important use of soils is for agriculture. The taxonomic classifications of soil resource, however, can be applied to any number of use and management purposes when the classification is based on the physical, chemical, and morphological properties of the soils. As such, use and management assessment of soil resources are made through soil surveys based on taxonomic soil classifications. In this entry, the development and architecture of the soil taxonomic system used in the United States in the past years—Soil Taxonomy, are discussed. The discussion includes references and comparisons to other national classification systems and the international soil classification system known as the World Reference Base. Advances in soil classification and major issues are also discussed.

INTRODUCTION

One could argue that of all the natural resources, soil, water, and air are the only essential resources. Soil, like any natural resource, needs to be classified in order to best utilize and conserve the resource. Fanning and Fanning^[1] suggested fairly simple reasons for classification as follows: to provide a means to organize our knowledge and to provide a system to retrieve the information for any number of purposes. Soils have been classified for almost as long as humans have lived in communities.^[2] One of the primary reasons was to assess the land for agriculture; those with the best soils for agriculture were taxed the greatest amount because they owned the best land. Since such classification approaches were not designed around the natural physical, chemical, and morphological properties of the soils (taxonomic properties), but based on their productivity or other use assessments, these soil classification approaches are termed interpretive classification systems.^[1]

Agriculture is also the root of most soil taxonomic systems. For example, in Soil Taxonomy (the system used in the United States and many other areas of the world), in deciding which properties to be used as criteria for classifying the soil, the property with the most effect on agriculture is the one that is chosen.^[3] Although throughout history agriculture has driven the design of a soil classification system, there are many other uses of Soil Taxonomy. This is clear in the title of the U.S. soil classification system; although typically referred to as just “Soil Taxonomy,” the complete title is *Soil Taxonomy: A Basic System for Making and Interpreting a Soil Survey*. In every U.S. soil survey, there is a list of use and management interpretations for each soil taxa. These include

many non-agronomic interpretations such as use of the soil for construction materials, recreational purposes, supporting roads and buildings, waste disposal treatment, constructing earthen dams, and silviculture. Thus, the ultimate purpose of Soil Taxonomy is to provide a means to assess the potential of the soil resource to serve any number of purposes through soil surveys. In this entry, we will focus on taxonomic soil classification systems.

SOIL TAXONOMY OVER 20TH CENTURY

Soil science and taxonomy, as we know them, are little more than a century old. Prior to the late 1800s, soils were considered a surface mantle of loose and weathered rock.^[4] V.V. Dokuchaev, a Russian geologist working mostly between 1870 and the turn of the 20th century, articulated our existing concept of soil: a natural body that has formed over time through a series of physical, chemical, and biological processes acting upon a given soil parent material. These processes are governed by the landscape setting, climate, and soil biology. Jenny^[5] called these the five state factors of soil formation: climate, organisms, relief, parent materials, and time. Thus, soils, with similar parent materials and biology that have formed under similar climates and landscape settings, will have similar properties and thus taxonomic classification.

Of the taxonomic soil classification systems prior to the turn of the 20th century, Dokuchaev's system was the most influential. His works were translated to German by Glinka^[6] and later translated from German to English by C.F. Marbut.^[1,7–9] Marbut considered many of the ideas and concepts of soils at that time relative to mapping soils and

developed a classification system for the United States that was published in its final form in 1935.^[10] Followers of Dokuchaev, especially Sibertsev,^[11] further developed Dokuchaev's system that later became the foundation of the soil classification system that was used in the United States between 1938 and 1965.^[1] The 1938 Yearbook system^[12] was based primarily on Dokuchaev's concepts of soil orders which were named zonal, intrazonal, and azonal soils in the 1938 Yearbook.^[13] Schaetzl and Anderson^[14] called the 1938 Yearbook the first serious attempt of classifying soils in the United States. Zonal soils essentially matched the concept of "normal soils" described by Dokuchaev and more fully articulated by Marbut.^[15] Soils that have developed in better drained conditions over enough time to be considered to have reached maturity under existing climatic conditions. Azonal soils were the young soils developing in areas such as shallow to bedrock, floodplains, or aridic sands. Intrazonal soils were controlled by wetness or chemical composition such as salts. Examples and summaries of the 1938 Yearbook system can be found in Fanning and Fanning^[1] and Schaetzl and Anderson.^[14]

Although the 1938 Yearbook was used for many years in the United States, serious flaws were widely recognized in the system. There were six categories in the hierarchy, but the suborder (second) and family (fourth) categories went essentially unused.^[16] Likewise, the series-level category failed in the hierarchical scheme since it would not always match with a higher-level taxa. Some soils went unrecognized, such as some of those formed from volcanic ash or shrink-swell clays.^[1] Another issue was that classification was based on unaltered (virgin) soils; thus, a soil that had been cultivated or eroded was classified into a taxa that represented what it would have been like in a virgin form.^[14] Because of these issues, a more quantitative and less subjective system was needed.

In the early 1950s, Guy Smith began to work toward a new classification system.^[17] Over the next decade, six versions (labeled approximations) of the system were circulated among soil scientists mostly in the United States.^[18] The 7th Approximation^[19] was circulated worldwide at the World Soil Congress in 1960. At first, there was little acceptance of the new system, but Simonson^[20] noted regarding his paper published in *Science* "evidence of growing interest was indicated by requests for a few thousand reprints of my 1962 paper." By 1965, the system was adopted for U.S. Department of Agriculture–Natural Resources Conservation Service [USDA-NRCS; formerly known as Soil Conservation Service (SCS)] soil surveys and a decade later published as Soil Taxonomy.^[3]

A list of guidelines for the development of an effective soil classification system was given within the 7th Approximation^[19] and later revised for inclusion in Soil Taxonomy.^[3] In general, the guidelines suggested that the system be flexible enough that soils that were not

identified could easily be added; the system could be used worldwide; the properties used to classify the soil will be measured (not necessarily by instruments); the focal point of the properties measured is soil genesis related; if a choice needs to be made between which of two properties to use in the classification criteria, the one with the most effect on plant growth be used; and properties that are easily changed by anthropogenic activities such as plowing be only used at the lowest of levels when classifying the soil.

THE SOIL THAT WE CLASSIFY

Our concept of soil depends on our background and experiences;^[21] e.g., someone with a potted plant may say the plant is supported by soil, and many would agree. A soil scientist would argue that this concept of soil is analogous to saying a cup of water taken from the lake shore is a lake. No one would agree with that! Knox^[22] reviewed the various bodies of soil materials that could be considered soil. These ranged from individual particles, like a sand grain, to landscape-level bodies that could be mapped out at a 1:24,000 scale. That pot of soil materials described above would be considered a "hand specimen" by Knox.^[22] In this entry, we will use the USDA-NRCS^[23] definition of soil: a collection of natural bodies; composed of mineral and organic materials; and that supports the growth of plants out-of-doors or show evidence of soil-forming (pedogenic) processes. Within this context, our collection of natural bodies is the series of horizons that vary with depth and over an area of the landscape on the order of 1–10 m². Such a 3-D area of soil is called a pedon.^[19,21,24] Most soils are described using a 2-D view of a pedon, called a soil profile (Fig. 1). In most soil taxonomic systems, a pedon is the smallest soil unit that is classified.

SOIL TAXONOMY INTO THE 21st CENTURY

Soil Taxonomy has the following six hierarchical categories: order, suborder, great group, subgroup, family, and series (Table 1). The first five categories are set up as a key, so that every soil can be classified, and the last taxon in the key is the default. In most cases, classification to the order, great group, or subgroup level in mineral soils is predicated on the presence or absence of diagnostic horizons. Those diagnostic horizons forming at the soil surface are called epipedons. Subsurface diagnostic horizons are typically in the position of the B horizons in the profile. The standard for the epipedon is the mollic because soils with mollic epipedons typically are excellent for agriculture. Most of the soils in the Great Plains in the United States and the Steppe regions in Russia have mollic epipedons. These are the soils that Dokuchaev studied in Russia and formulated his early concepts of soils. Mollic epipedons are thick (typically 25 cm or more), dark in color, rich in organic matter, and rich in basic cations (base saturation >50%). Umbric

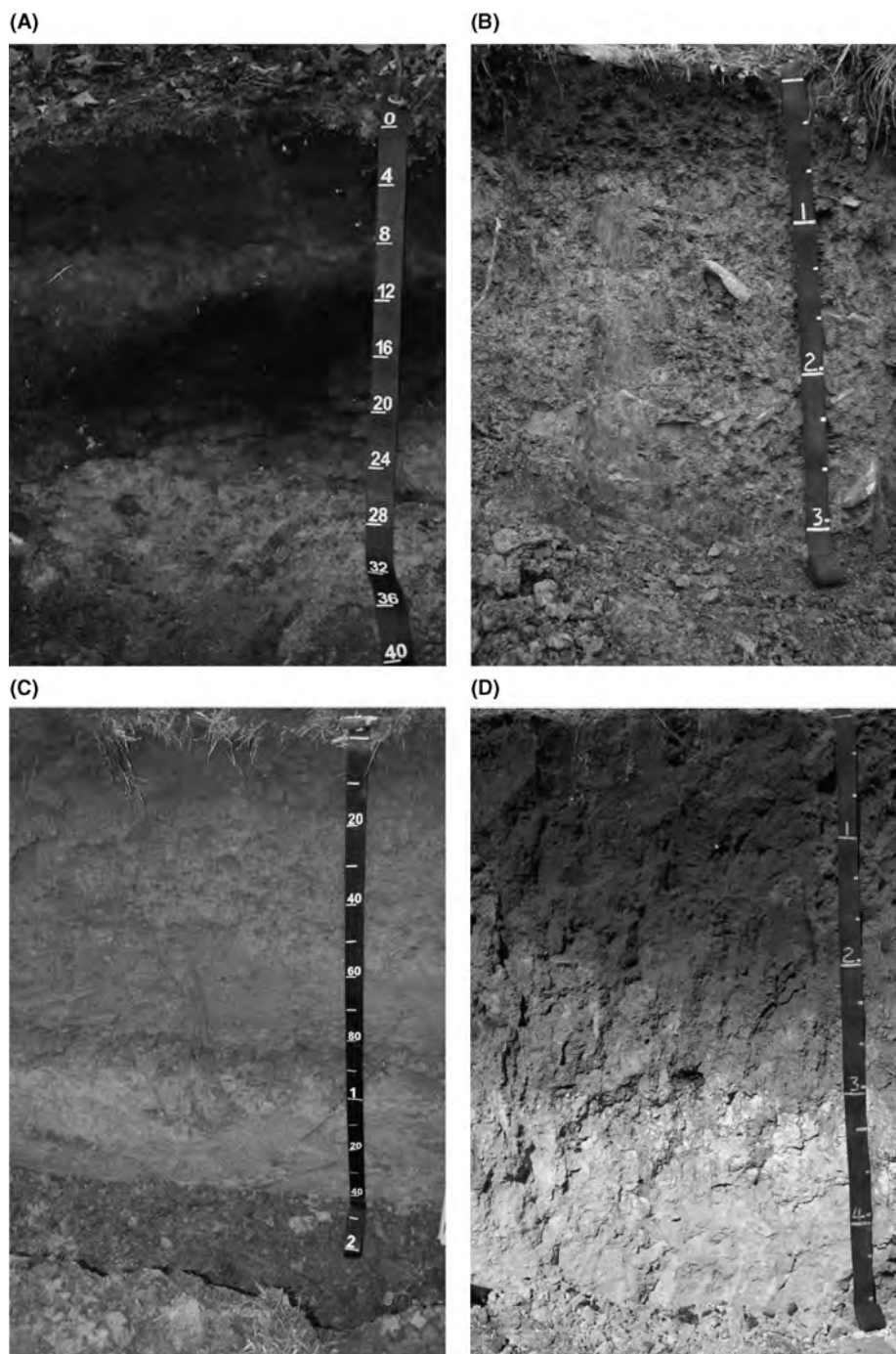


Fig. 1 (A) A Typic Epiaquod in Rhode Island. The tape is in inches. The spodic diagnostic subsurface horizon varies in thickness and depth. Near the tape, the spodic horizon occurs from 10 to 20 in. The soil is saturated to the surface for parts of year and thus the aquic suborder. The water is held, or perched, above a restrictive layer (epi) starting at 36 in. There are no other diagnostic features (typic). (B) An umbric Endoaquult in Pennsylvania. The soil has an umbric epipedon, an argillic diagnostic subsurface horizon, and a base saturation <35%. The soil is saturated throughout the profile and to the soil surface (Endoaqu) for extended periods of the year. The small marks on the tape are 10 cm increments. The large marks are in feet. (C) Fluvaquentic Eutrudept on a Pennsylvania floodplain. The tape is in centimeters. Fluvaquentic indicates that the soil is saturated during the year for a considerable amount of time above 60 cm, and there is buried soil organic carbon (note the buried A horizon between 80 and 90 cm). The soil has a cambic diagnostic subsurface horizon (20–80 cm) with a base saturation >60% (Eutr). The climate is humid and thus an udept. (D) Calcic Haplustalf in Texas. The small increments on the tape are 10 cm apart. The larger marks are one foot apart. The soil has calcic and argillic diagnostic subsurface horizons. The base saturation is greater than 35%, and the moisture regime is semiarid (ustic).

Table 1 The architecture of Soil Taxonomy.

There are six categories from the highest (least detail) to lowest (most detail) level: order, suborder, great group, subgroup, family, and series. The names for classes in the order, suborder, great group, and subgroup categories are all coined terms using Greek, Latin, French, or similar roots. The soil orders are listed in the order they key out; a limited number of examples are provided for the other categories.

Order: Names of the 12 orders consist of three or four syllables and every name ends in the suffix “sol.”

Order name	Formative element	Derivation
Entisol	ent	Nonsense syllable, think of recent
Vertisol	ert	L. <i>Verto</i> , turn, think of invert
Inceptisol	ept	L. <i>inceptum</i> , think of inception
Aridisol	id	L. <i>aridus</i> , dry, think of arid
Mollisol	oll	L. <i>mollis</i> , soft, think of mollify
Spodosol	od	Gk. <i>Spodos</i> , wood ash, think of Podzol*
Alfisol	alf	Nonsense syllable, think of Pedalfer**
Ultisol	ult	L. <i>ultimus</i> , last, think of ultimate
Oxisol	ox	F. <i>oxide</i> , think of oxide
Andisol	and	Gk. <i>Andes</i> , think andesite
Histosol	ist	Gk. <i>histos</i> , tissue, think of Histology
Gelisol	el	L. <i>gelare</i> , to freeze, think of gel

*Podzol—classic term used for soils with spodic horizons; **Pedalfer—one of the two taxa in Marbut's^[10] highest category.

Suborder: The name of every suborder is a two-syllable term consisting of a prefix syllable with some specific connotation plus the formative element from the order to which the suborder belongs. The examples are for Ultisols, Inceptisols, and Vertisols.

Formative element	Connotation	Example
Aqu	Aquic moisture regime	Aquult
Ud	Udic moisture regime	Udept
Ust	Ustic moisture regime	Ustert

Great group: The name of each great group is constructed by adding a second prefix element with a specific connotation to the two-syllable term that is the name of the suborder to which the great group belongs.

Formative element	Connotation	Example
Umbra	An umbric epipedon	Umbraquult
Eutr	High base status	Eutrudept
Calci	A calcic horizon	Calciustert

Subgroup: Names of subgroups are binomial. An adjective (sometimes two) is used to modify the name of the great group to give the name of each subgroup within that great group.

Adjective	Connotation	Example
Plinthic	Presence of plinthite	Plinthic Umbraquult
Arenic	Thick sandy surface	Arenic Eutrudept
Lithic	Shallow to rock	Lithic Calciustert

Family: A sequence of descriptive adjectives is used in family names. The common examples are used to describe particle-size classes, mineralogy classes, activity classes, and soil temperature classes.

An example is coarse-loamy, mixed, active, mesic, typic dystrodept. Coarse-loamy (particle-size class), mixed (mineralogy class), active (activity class), mesic (soil temperature class), aquic dystrodept (subgroup).

Series: It is a family that falls within a given range of characteristics. Series names are typically derived from towns or localities that are near where the soils are found (i.e., Newport). The range in characteristics is related to morphological properties such as thickness of the B horizons, soil colors, rock fragment contents, and soil textures. The range in characteristics is not found in Soil Taxonomy but can be accessed online: <http://soils.usda.gov/technical/classification/osd/index.html>

epipedons are similar to the mollic except more acidic, having a lower base status. Histic epipedons are typically 20–40 cm thick and dominated by organic soil materials. There are also folistic (unsaturated histic epipedons), melanic (associated with volcanic ash), plaggen (heavily manured for centuries), and anthropic (phosphorus rich from long-term human occupation) epipedons. In general, the ochric epipedon is the default, meaning soils that do not meet the criteria for the other six epipedons have an ochric epipedon (very new soils may have no epipedon).

In Soil Taxonomy, there are 18 diagnostic subsurface horizons. Only the more common are briefly discussed in this entry, so that the reader can understand the concept and follow the examples in Table 1. Most diagnostic subsurface horizons are related to the accumulation of soil constituents in the B horizon position of the soil profile. The accumulations may be of silicate clay (argillic), sesquioxides bound to organic matter (spodic), calcium carbonate (calcic), gypsum (gypsic), and salts more soluble than gypsum (salic). Certain diagnostic subsurface horizons are dense or cemented by the constituent that has accumulated such as petrocalcic, petrogypsic, or duripan (cemented by silica that has reprecipitated in the B horizon). Younger soils may not have had enough time to form any of the diagnostic horizons previously mentioned but have B horizons recognized by a simple change in color. These “color” B horizons typically meet the criteria for a cambic diagnostic horizon.

In keying out soils to the order level, Gelisols are the first possibility. The important criterion is the presence of permafrost (generally within a meter of the soil surface). Histosols key out next; these soils are dominated by organic soil materials (at least 40 cm) in the upper 80 cm. Spodosols follow, having spodic diagnostic horizons. Andisols have andic soil properties (high exchange capacity and water-holding capacity). Oxisols are extremely weathered and dominated by iron and aluminum oxides (oxic horizon). Vertisols are dominated by shrink–swell clays that results in a self-mixing soil. Aridisols may have a number of diagnostic horizons such as argillic, calcic, and cambic as long as they are in aridic (dry) moisture regimes. Ultisols have argillic horizons and low base saturation (<35%). Mollisols have mollic epipedons and a base saturation throughout the profile greater than 50%. These soils may have calcic or argillic horizons. Alfisols almost always have argillic horizons, but unlike Ultisols, their base saturation is >35%. Inceptisols are weakly developed soils typically with cambic horizons. Entisols are the least developed, characterized by a lack of diagnostic horizons or epipedons other than ochric, and key out last.

Because soil moisture (or lack of) has a dramatic effect on soil morphology, suborders typically bring the wetness attributes or soil moisture states related to climate (soil moisture regimes) into the classification (see examples in Table 1 and Fig. 1). These moisture regimes include aquatic

(wet), udic (humid), ustic (semiarid), and xeric (dry parts of the year and moist other parts).

There were only a few major changes to Soil Taxonomy over the years. In 1975, there were 10 orders; that was changed to 12 with the addition of Gelisols (soils with permafrost) and Andisols (soils formed in volcanic ash-type materials). The definition of soil was changed in the 2nd edition.^[25] Previously by definition, a soil was required to be able to support rooted plants.^[3] In the new edition, soils, as long as they showed evidence of pedogenesis, were considered soils even if they did not support rooted plants. This departure from the agricultural roots of Soil Taxonomy was designed to accommodate soils in the arctic that were permanently covered with snow or ice^[26,27] or soils that were permanently under water (subaqueous soils).^[28]

In 1983, field versions of Soil Taxonomy were first published. The field versions essentially just included what was needed to key the soil out to the family level, but much of the supplemental materials of the classification system were left out. These versions were called the *Keys to Soil Taxonomy*. As we learned more and more about the soils, changes in the taxonomy were introduced in revised versions of *Keys to Soil Taxonomy*. For example, taxa were included in the 11th edition of *Keys to Soil Taxonomy*^[23] to accommodate subaqueous soils.^[28] Although by design Soil Taxonomy was meant to be flexible enough so that new soils could be added when deemed necessary,^[3] some argue that over time it might be too big and cumbersome to be useful any longer. For example, the first *Keys to Soil Taxonomy*^[29] was a 4 in. × 9 in. paperback that easily fit in your back pocket for use in the field (<250 pages). By 1996, the 7th edition of *Keys to Soil Taxonomy* in the same 4 in. × 9 in. format was well over 600 pages.

One area of soils that continues to be difficult to classify in Soil Taxonomy is the anthropogenic soils (i.e., soils developed in materials moved by man).^[30] In an attempt to resolve this issue, the International Committee on Anthropogenic Soils (ICOMANTH) was formed with the charges defining appropriate classes in Soil Taxonomy for soils that have their major properties derived from human activities. The first activities of this committee were reported in the 1st circular letter dated 1995 (http://clic.cses.vt.edu/icomanth/03-AS_Circulars.pdf). Efforts of the committee resulted in new horizon designations for soils with restrictive properties because of the presence of a continuous man-made material such as cement (master horizon designation M) and horizons containing human artifacts such as glass, brick, or cement (subordinate horizon designation u). These changes appeared in the 10th edition of *Keys to Soil Taxonomy*.^[31] Suggestions for classifying such soils have primarily focused on subgroups such as spolic, urbic, or garbic^[1,32] or a separate order with Soil Taxonomy—Anthrosols.^[33] Efforts toward classifying anthropogenic soils continue through ICOMANTH such that there are seven circular letters describing proposals,

changes, and related activities of the committee (<http://clic.cses.vt.edu/icomanth/circlet.htm>).

OTHER NATIONAL AND INTERNATIONAL CLASSIFICATION SYSTEMS

There are numerous other taxonomic soil classification systems. Buol et al.^[34] reviewed a number of these including the Russian, French, Belgian, British, Canadian, Australian, Brazilian, and Chinese. Many of these systems take elements directly or indirectly from Soil Taxonomy.^[25] For example, at the order level, the Australian system uses “Organosols” instead of Histosols.^[35] Seven of the diagnostic horizons or features in the Chinese system are taken directly from Soil Taxonomy.^[36] The Chinese and Australian systems have soil orders for anthropogenic soils.^[35,36]

The World Reference Base (WRB)^[37] is the only true international soil classification system. This system, known as the WRB, was endorsed and adopted by the International Union of Soil Sciences as the classification system for soil correlation and international communication. The intention of the WRB was not to provide an alternative for any national soil classification system but to serve as a soil classification translator for communication at an international level.^[37] During the development of the WRB, a focused effort was put forth to include as much nomenclature from Soil Taxonomy^[25] and other major national soil classification systems, as possible.

The WRB is a two-tiered system. At the highest category are 32 reference soil groups. A number of these are essentially the same as the soil orders in Soil Taxonomy; examples of these include Vertisols, Histosols, Podzols (Spodosols), Cryosols (Gelisol), and Chernozems (Mollisols). Like Soil Taxonomy,^[25] diagnostic horizons are used as criteria in many of the taxa. Although their definitions are not identical, a number of the WRB diagnostic horizons are conceptually the same as those in Soil Taxonomy. Some examples include mollic, ochric, histic, gypsic, calcic, and cambic.

The second tier of the WRB is a series of qualifiers that are added as a prefix or suffix to the reference soil groups to identify in greater detail properties of the particular soil. Each reference soil group is assigned a list of qualifiers that may be used. In the WRB,^[37] an example is given for a Cryosol soil reference group: “Histic Turbic Cryosol (Reductaquic, Dystric).” Histic and turbic are the prefix qualifiers indicating that the Cryosol has histic epipedon and evidence of cryoturbation (mixing by freeze-thaw), respectively. Reductaquic and dystric are the suffixes indicating that the soil is both saturated and reduced and has a base saturation less than 50%, respectively. Unlike Soil Taxonomy,^[25] climatic parameters are not applied in the classification of soils in the WRB, and the system is not designed for mapping soils at resource-level scales.

SUMMARY AND CONCLUSION

Soil is one of the basic and essential natural resources. Thus, some form of soil classification has been around since humans formed communities. The early systems were primarily based on a use assessment of the soils for a single purpose such as agriculture productivity (i.e., low yields, moderate yields, or high yields). These systems are often referred to as interpretive soil classifications. The taxonomic classification of soils is based on the natural physical, chemical, and morphological properties of the soil and can provide many use and management interpretations. The application of taxonomic classification for soil resources is through the use of soil surveys. The best example of a taxonomic system for mapping and assessment of the soil resource is Soil Taxonomy. This is a hierarchical system that is set up like a key, so that every soil can be classified. There are six categories in the system with order the highest (least amount of detail) and series the lowest (greatest amount of detail). Although Soil Taxonomy is used worldwide, many countries have their own soil classification system. There is also a truly international system called the WRB. The WRB is designed as an international translator for systems such as Soil Taxonomy and was not meant to provide classifications for resource soil mapping.

Issues with Soil Taxonomy appear to be related to the lack of significant taxa for anthropogenic soils, its field applications, and the rapidly expanding number of taxa. Issues with field applications center around the need for considerable laboratory data to correctly classify a soil. A possible resolution is to develop guidelines based on reasonable assumptions that can assist the user in applying Soil Taxonomy in the field without any laboratory data. As we learn more and more about soils, the number of taxa in Soil Taxonomy continues to grow. Proposals are being considered to add taxa to accommodate the classification of some anthropogenic soils. Some users feel that continued additions leave the classification system too cumbersome to key through to find the correct taxa. Whether this is a problem or not is debatable, but correcting such a problem may require reformatting the hierarchical system. An approach to resolve this issue that is being considered is the new classification system using some of the format of the WRB and some of Soil Taxonomy. As the world's population continues to grow, resource managers will turn more and more to soils information to best utilize this essential resource. As such, developing an understanding of Soil Taxonomy will continue to be a need for natural resource users and managers.

REFERENCES

1. Fanning, D.S.; Fanning, M.C.B. *Soil Morphology, Genesis, and Classification*; John Wiley and Sons: New York, 1989.
2. Simonson, R.W. Soil classification in the United States. *Science* **1962**, *137*, 1027–1034.

3. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys, USDA-NRCS Agricultural Handbook*; U.S. Government Printing Office: Washington, 1975; Vol. 436.
4. Simonson, R.W. Historical aspects of soil survey and soil classification. Part 1. 1899–1910. *Soil Surv. Horizons* **1986**, 27 (1), 3–11.
5. Jenny, H. *Factors of Soil Formation*; McGraw Hill Book Co. Inc.: New York, 1941.
6. Glinka, K.D. *Die Typen der Bodenbildung: Ihre Klassifikation und Geographische Verbreitung*; Gebruder Borntraeger: Berlin, 1914.
7. Marbut, C.F. *The Great Soil Groups of the World and Their Development. Translated from Glinka 1914*; Edwards Brothers: Ann Arbor, 1927.
8. Smith, G.D. Historical development of soil taxonomy-background. In *Pedogenesis and Soil Taxonomy*; Wilding, L.P., Smeck, N.E., Hall, G.F., Eds.; Elsevier: Amsterdam, 1983; 23–49.
9. Simonson, R.W. Historical aspects of soil survey and soil classification. Part 2. 1911–1920. *Soil Surv. Horizons* **1986**, 27 (2), 3–9.
10. Marbut, C.F. Soils of the United States. In *Atlas of American Agriculture*; Baker, O.E., Ed.; USDA, U.S. Government Printing Office: Washington, 1935; 12–15.
11. Sibertsev, N.M. *Russian Soil Investigations. Israel Programs for Science Translations Jerusalem, Translated from Russian 1966*; U.S. Department of Commerce: Springfield, 1901.
12. Baldwin, M.; Kellogg, C.E.; Thorp, J. Soil classification. In *Yearbook of Agriculture: Soils and Men*; USDA, U.S. Government Printing Office: Washington, 1938; 979–1001.
13. Simonson, R.W. Historical aspects of soil survey and soil classification. Part 4. 1931–1940. *Soil Surv. Horizons* **1986**, 27 (4), 1–10.
14. Schaezel, R.; Anderson, S. *Soils: Genesis and Geomorphology*; Cambridge University Press: New York, 2005.
15. Marbut, C.F. A scheme of soil classification. In *Proceedings of the 1st International Congress of Soil Science*, Jun 13–22, 1927; American Organizing Committee: Washington, 1928; Vol. 4, 1–31.
16. Simonson, R.W. Historical aspects of soil survey and soil classification. Part 5. 1941–1950. *Soil Surv. Horizons* **1987**, 28 (1), 1–8.
17. Cline, M. *Soil Classification in the United States. Agronomy Memo*, 79–12; Cornell University: Ithaca, 1979.
18. Simonson, R.W. Historical aspects of soil survey and soil classification. Part 6. 1951–1960. *Soil Surv. Horizons* **1987**, 28 (2), 39–46.
19. Soil Survey Staff. *Soil Classification, A Comprehensive System—7th Approximation*; USDA. U.S. Government Printing Office: Washington, 1960.
20. Simonson, R.W. Historical aspects of soil survey and soil classification. Part 7. 1961–1970. *Soil Surv. Horizons* **1987**, 28 (3), 77–84.
21. Simonson, R.W. Concept of soil. *Adv. Agron.* **1968**, 20, 1–47.
22. Knox, E.G. Soil individuals and soil classification. *Soil Sci. Soc. Am. Proc* **1965**, 29, 79–84.
23. Soil Survey Staff. *Keys to Soil Taxonomy*, 11th Ed.; United States Department of Agriculture Natural Resources Conservation Service. Government Printing Office: Washington, 2010.
24. Simonson, R.W.; Gardner, D.R. The concepts and functions of the pedon. T. Int. Congr. Soil Sci. **1960**, 4, 127–131.
25. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; USDA-NRCS Agricultural Handbook 436, U.S. Government Printing Office: Washington, 1999.
26. Bockheim, J.G. Soil development rates in the Transantarctic Mountains. *Geoderma* **1990**, 47, 59–77.
27. Bockheim, J.G. Properties and classification of cold desert soils from Antarctica. *Soil Sci. Soc. Am. J.* **1997**, 64, 224–231.
28. Stolt, M.H.; Rabenhorst, M.C. Subaqueous soils. In *Handbook of Soil Science*, 2nd Ed.; Huang, P.M., Li, Y., Sumner, M.E., Eds.; CRC Press: Boca Raton, 2011; 36.1–36.14.
29. Soil Survey Staff. *Keys to Soil Taxonomy*; United States Department of Agriculture Natural Resources Conservation Service, Government Printing Office: Washington, 1983.
30. Kimble, J.M.; Aherns, R.; Bryant, R. *Classification, Correlation, and Management of Anthropogenic Soils*; USDA-NRCS, National Soil Survey Center: Lincoln, 1999; 227 pp.
31. Soil Survey Staff. *Keys to Soil Taxonomy*, 10th Ed.; United States Department of Agriculture Natural Resources Conservation Service, Government Printing Office: Washington, 2006.
32. Pouyat, R.V.; Ellland, W.R. The investigation and classification of humanly-modified soils in the Baltimore ecosystem study. In *Classification, Correlation, and Management of Anthropogenic Soils*; Kimble, J.M., Aherns, R., Bryant, R., Eds.; USDA-NRCS, National Soil Survey Center: Lincoln, 1999; 141–154.
33. Bryant, R.B.; Galbraith, J.M. Incorporating anthropogenic processes in soil classification. In *Soil Classification: A Global Desk Reference*; Eswaren, H., Rice, T., Aherns, R., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2003; 57–65.
34. Buol, S.W.; Southard, R.J.; Graham, R.C.; McDaniel, P.A. *Soil Genesis and Classification*, 5th Ed.; Iowa State Press: Ames, 2003.
35. Isbell, R.F. *The Australian Soil Classification System. Australian Soil and Land Survey Handbook*; CSIRO Publishing: Collingwood, 1998; 494 pp.
36. Gong, Z.; Zhang, G.; Luo, G. The Anthrosoles in Chinese soil taxonomy. In *Classification, Correlation, and Management of Anthropogenic Soils*; Kimble, J.M., Aherns, R., Bryant, R., Eds.; USDA-NRCS, National Soil Survey Center: Lincoln, 2003; 40–51.
37. Food and Agriculture Organization of the United Nations. *World Reference Base for Soil Resources*; FAO: Rome, 2006.

Temperature Measurement

Kevin McInnes

Department of Soil and Crop Sciences, Texas A&M University, College Station, Texas, U.S.A.

Abstract

There is no set design either for construction or procedure for installation of soil thermometers, other than the requirement that they operate in a continuously moist environment. Soil thermometers may be tailored for measurement of either surface or subsurface temperatures. Surface temperature measurements are desirable because they can be used to estimate subsurface temperatures. Measurement of subsoil temperature is difficult because installation disturbs the soil and alters the thermal regime.

VARIABILITY OF SOIL TEMPERATURE

Soil temperatures generally fall between -20°C and 60°C , except in extreme environments. Mean diel soil surface temperature is usually the same as the mean diel air temperature. Similarly, the mean annual soil temperature is usually equal to the mean annual air temperature.^[1] Differences between extremes of the daily average surface temperatures vary from a few degrees near the equator to upward of 30°C or more in the mid-latitudes. For the diel cycle, differences between the periodic extreme surface temperatures depend on the nature of the soil cover. Bare soil produces the largest differences, with $15\text{--}30^{\circ}\text{C}$ being common. Both diel and annual patterns in temperature at the soil surface travel down through the soil where each magnitude is increasingly attenuated and each phase increasingly retarded with increasing depth.

Water in soil helps buffer against extreme temperature fluctuation. In the case of soil freezing, declining soil temperatures are buffered near 0°C by the latent heat of fusion. Once water is frozen, soil temperatures may drop as low as -40°C in extremely cold environments.^[2] Likewise, as soil temperatures rise under extreme heating, they are buffered by the latent heat of vaporization at temperatures near 95°C .^[3] When all moisture is lost under prolonged heating, as might occur under a slow-moving fire, interfacial soil temperatures may rise hundreds of degrees.^[3]

MEASUREMENT OF SURFACE TEMPERATURE

Surface temperature may be used to calculate subsurface temperatures with a fair degree of accuracy.^[4] Surface temperatures can be measured with or without direct contact. Non-contact thermometers infer temperature from the amount of thermal radiation emitted from the soil. Infrared thermometers (IRTs) measure temperature by capturing and quantifying the amount of thermal radiation emitted in a

narrow band of the infrared spectrum, typically $8\text{--}14\text{ }\mu\text{m}$. Soil temperature is related to the quantity of radiation emitted through the Stefan–Boltzmann equation and knowledge of the soil's thermal emissivity.^[5] As soil temperature is inferred from the amount of infrared radiation captured by the sensor, reflected background radiation introduces a source of uncertainty that must be considered. A soil's thermal emissivity changes with soil water content and surface roughness, whereas background radiation varies with air temperature, humidity, and cloud cover. Monitoring changes in emissivity and background radiation requires additional IRT measurements, adding to the complexity of the soil temperature measurements. A variation of the IRT used extensively in remote sensing is the infrared camera. An infrared camera provides detailed information on spatial variation in surface temperature that otherwise would be difficult to collect.

Contact thermometers used to measure surface temperature must have good thermal contact with the soil surface and minimally affect surface energy exchange. To satisfy both these requirements sensors may be coated with soil.^[6] The added soil provides weight that helps hold the thermometer in contact with the soil and gives the sensor similar emissivity and reflectivity as the soil.

TYPES OF SOIL THERMOMETERS

While non-electric contact sensors are used to measure soil temperature, the advent of accurate and reliable electronics has removed nearly all advantages they held over electric thermometers. There are several types of electric thermometers suited to measuring soil temperatures. Temperature probes made from thermocouples, resistance sensors, and integrated circuits (ICs) are most common. All these sensors easily function over the normal range of soil temperature.

Thermocouples are popular soil temperature sensors because they are robust and easy to construct. Thermocouples consist of two dissimilar metal wires joined to each other at one end, the sensing junction, and to the terminals of a meter at the other end. A difference in temperature between the sensing junction and the meter terminals results in a voltage that can be measured and converted to temperature provided the temperature of the terminals is known. There are eight different combinations of metals and alloys, designated as letter types, for which there are internationally accepted values of the relationship between voltage and temperature difference (referenced to 0°C). Modern thermocouple meters have these relationships built into their software. Industry-accepted errors in wire composition allow a specified degree of uncertainty in voltage output at a given temperature.^[7] Type T wire consists of copper and constantan conductors and is the most popular type for constructing soil thermometers. Manufacture with standard type T wire allows an error $\pm 0.8^\circ\text{C}$ from the calibration equation used in modern thermocouple meter software. This allowable error for type T is better than that for other types of thermocouple wire by a factor of approximately 2. A limited selection of special wires with tolerances half those of the standard wires (e.g., $\pm 0.4^\circ\text{C}$ for type T) is available.

Thermocouples may be connected in a series configuration called a *thermopile* to measure minute temperature differences, such as those associated with gradients inducing heat fluxes in soil or they may be combined in parallel to measure a spatial average temperature.^[8] Caution must be exercised when using a parallel configuration of thermocouples because improper construction may cause serious errors.^[8–10]

IC temperature transducers are a new class of thermometers. IC sensors are available with either a voltage or a current output. Both outputs are linearly related to temperature. Voltage output is typically 10 mV K^{-1} ,^[11] while current is usually regulated at $1\text{ }\mu\text{A K}^{-1}$.^[12] The IC sensors that regulate current proportional to temperature are particularly useful in measuring soil temperatures because they are insensitive to voltage drops over long lead wires. Any shielded and well-insulated twisted pair of conductors is sufficient for operation of the sensor hundreds of meters from the receiving meter. Current is calculated using Ohm's Law and the voltage drop across an inline resistor of known magnitude. Current-regulating sensors may be connected in parallel to measure a spatial average temperature.

Other soil temperature sensors take advantage of the sensitivity of electrical resistance of metals or semiconductors to temperature. Resistance temperature detectors (RTDs) are wire and metal film-type sensors that make accurate soil thermometers. Platinum RTDs are the most common. Resistances of platinum RTDs are usually $100\text{ }\Omega$ at 0°C , although new thin film deposition technology has led to creation of RTDs with resistances $>1\text{ k}\Omega$. The resistance of platinum increases almost

linearly with temperature, approximately 0.4% per $^\circ\text{C}$. Because of this linear relationship, RTDs may be connected in series and are used to measure an average soil temperature. The disadvantage to using RTDs as soil thermometers is that resistances of individual sensors are low enough that three- or four-conductor cables are required to account for the resistances of the lead wires.^[10]

Thermistors are also thermally sensitive resistors, but made from semiconductor materials. They have two distinct advantages over RTDs. First, their nominal resistance is greater so that only two conductors are usually required for lead wires. The second advantage is that their sensitivity to temperature is considerably greater. Meters are not required to measure with the precision necessary for RTDs. A disadvantage limiting the use of thermistors in averaging temperature is that their electrical resistance is usually a highly non-linear function of temperature.

Non-electric thermometers such as liquid-in-glass thermometers, dilatation thermometers, bimetallic thermometers, and manometric thermometers are sluggish and difficult to install in a manner that would minimize errors from heat conduction along the thermometer stem. In addition, thermometers that rely on thermal expansion as a measure of temperature are prone to serious errors in temperature measurement caused by external forces acting on the sensing portion of the device,^[13] such as forces present in soil prone to shrinking and swelling.

SOURCES OF ERROR

Spatial variability leads to uncertainty in measured soil temperature.^[8] The number of temperature sensors required to produce an average within a certain value of the mean must be determined from an *in situ* measure of variability.^[8] Fortunately, once the required number is known, then many thermocouples,^[8] IC temperature sensors,^[12] or RTDs sensors^[8] are easily connected in a configuration that provides an average temperature. By their nature, IRTs measure an average surface temperature integrated over an area that depends on the distance of the thermometer from the surface and its field of view.

Heat conduction down the lead wires or stem of the thermometer to the sensing element may lead to error. Thermal conductivity of copper, e.g., is two orders of magnitude greater than that of soil. To reduce conduction errors, a length of wire or portion of the thermometer stem should be buried along an isotherm of the temperature being measured. With electric thermometers, $0.2\text{--}0.5\text{ m}$ of wire buried along an isotherm is usually sufficient to reduce errors to within acceptable levels. Reducing the diameter of the lead wires near the sensor also will reduce heat conduction errors.

Lack of proper insulation and shielding of electric thermometers may also lead to serious errors. Insulation on sensors and cabling needs to be highly resistant to moisture.

Adhesive-lined heat-shrinkable tubing or resin-filled stainless steel tubing is a commonly used insulator for a sensor. To reduce the potential from electromagnetically induced noise, sensor cabling should be shielded and properly grounded. Except for the self-powered thermocouple, most electric temperature sensors are also susceptible to self-heating errors. To prevent errors caused by Joule heating the current should be kept below the self-heating limit imposed by the heat dissipation coefficient,^[10] but large enough to give adequate resolution on the meter.

INSTALLATION OF SUBSURFACE THERMOMETERS

Soil thermometers may be buried in disturbed soil or they may be installed on supports driven into the soil. Burying is most efficient when the thermometers are to be placed within about 0.15 m of the surface. Small diameter pipe has been used to house soil thermometers installed at greater depth.^[13] Metal pipe may be driven to considerable depth in hard soil, but its considerably different thermal properties affect the local temperature regime. PVC pipe is a better support for soil thermometers because its ability to transfer heat is nearly equal to that of dry soil. Insertion is relatively easy, provided a pilot hole is created. Electric sensors should be embedded and sealed so they are flush with the outside of the pipe. The lead wires should be located inside the tube and, to reduce the potential for errors from heat conduction along the lead wires, a section of the wires should be bunched inside the pipe at the same depth as the sensor. Wooden stakes with embedded thermometers also have been used for support. Like PVC, wood transfers about the same amount of heat in response to a temperature gradient as does soil.

REFERENCES

1. Shul'gin, A.M. *The Temperature Regime of Soils*; Translated by Gourevitch, A. Israel Program for Scientific Translations; Sivan Press: Jerusalem, 1957.
2. Sharratt, B.S. Freeze-thaw and winter temperature of agricultural soils in interior Alaska. *Cold Reg. Sci. Technol.* **1993**, 22 (1), 105–111.
3. Campbell, G.S.; Jungbauer, J.D.; Bristow, K.L.; Hungerford, R.D. Soil temperature and water content beneath a surface fire. *Soil Sci.* **1995**, 159 (6), 363–374.
4. Campbell, G.S. *Soil Physics with BASIC: Transport Model for Soil-Plant Systems*; Elsevier: Amsterdam, 1985.
5. Fuchs, M.; Tanner, C.B. Surface temperature measurements of bare soils. *J. Appl. Meteor.* **1968**, 7 (2), 303–305.
6. Ham, J.M.; Senock, R.S. On the measurement of soil-surface temperature. *Soil Sci. Soc. Am. J.* **1992**, 56 (2), 370–377.
7. American Society for Testing and Materials. *Manual on the Use of Thermocouples in Temperature Measurement*, 4th Ed.; ASTM Manual Series: MNL12 (PCN: 28-012093-40); ASTM: Philadelphia, 1993.
8. Tanner, C.B. *Basic Instrumentation and Measurements for Plant Environment and Micrometeorology*; Department of Soils, Bull. 6; University of Wisconsin: Madison, 1963.
9. Schooley, J.F. *Thermometry*; CRC Press: Boca Raton, 1986.
10. Michalski, L.; Eckersdorf, K.; McGhee, J. *Temperature Measurement*; John Wiley: New York, 1991.
11. National Semiconductor Corporation. *LM35/LM35A/LM35C/LM35CA/LM35D Precision Centigrade Temperature Sensors*; Datasheet 005516; National Semiconductor Corporation: Arlington, 1997.
12. Analog Devices. *Two-Terminal IC Temperature Transducer AD590*; Analog Devices: Norwood, 1997.
13. Garvitch, Z.S.; Probine, M.C. Soil thermometers. *Nature* **1956**, 177 (4522), 1245–1246.

Tepetates: Mexico

Jorge D. Etchevers

Claudia Hidalgo

Postgraduate College, Natural Resources Institute, Montecillo, Mexico

C. Prat

Research Institute for Development (IRD), Montpellier, France

P. Quantin

Research Institute for Development (IRD), Dijon, France

Abstract

Tepetate is a vernacular Mexican term referring to a hardened land with various degrees of infertility. *Tepetates* are derived from volcanic material, mainly tuff. The scientific definition reserves the term *tepetate* for a hardened layer in soils formed from pyroclastic materials. In the central Valley of Mexico, these indurate layers are among the most striking pedological features. Two types have been recognized: the fragipan and the duripan type. The first one can be converted into soil after proper management to alleviate the social pressure for agricultural land.

TEPETATES: HARDENED VOLCANIC SOIL LAYERS

Tepetate is a vernacular Mexican term referring to a soil or hardened material with various degrees of infertility.^[1] Etymologically, *tepetate* derives from the Nahuatl term “tepetlatl” (“tetl” = stone and “petatl” = bed) which means “bed of stone.”^[2] However, in both folk and technical senses, *tepetate* has a wide semantic range of meanings.^[2,3] In the 16th century, the Nahuatl classification of earth materials included the *tepetate* as both a deductive class defined by slightly friable consistence “rock-like materials” and an inductive class of “arable soil.”^[2] It is speculated that the term *tepetate* was adopted to enclose two Nahuatl words: “tepetatl” and “tepetatlali,” used to differentiate between “bed of stone” inapt for agriculture and ameliorated “bed stone,” respectively. This explanation makes it easier to understand the ambiguity of the term. Studies on peasant soil classification make these terms equivalent to non-workable and workable land, respectively.^[4] Although the folk definition is widely used, a more modern scientific definition reserves the term *tepetate* for a hardened layer found in soils formed from pyroclastic materials.^[5,6] The main attribute of *tepetates* is their hardness when dry.

Hardened soil layers are not exclusive from Mexico. Similar formations can be found in Nicaragua, Salvador and Honduras,^[5,7,8] Ecuador,^[9] Chile,^[10] Peru,^[11] and Colombia^[12] under different names “talpetate,” “cangahua,” “tobas and ñadis,” and “sillares” (Table 1).

Pedologically, *tepetates* are surface or subsurface hardened layers derived from tuff,^[17] pyroclastic flows,^[5] or old volcanic ashes.^[18] *Tepetates* have been described as mineral “C” horizons little affected by pedogenetic processes, which correspond to an intermediate state of alteration of a vitric rhyolitic tuff.^[17,19] The main attribute of *tepetates* is its hardness, although there are big differences among different types. They vary in color from gray over yellow to reddish brown, as well as, in thickness and structure.

The best known *tepetates* in Mexico are located in the Valleys of Mexico and Tlaxcala, in the eastern and western slopes of the Sierra Nevada between 19°10' and 19°40' LN and 98°10' and 98°55' LW (Fig. 1).^[13,20,21]

Up to 10 layers of superimposed *tepetates* have been described in Central Mexico.^[21]

The first three series of these volcanic deposits (named T₁, T₂, and T₃) were dated between 10,000 and 40,000 years old.^[21] Peña and Zebrowski^[22] classified the *tepetates* of Central Mexico in accordance with two criteria: the stratigraphic series of origin and their consistency (similar to a fragipan or petrocalcic horizon). The fragipan-type *tepetates*, which are the ones that can be ameliorated for agricultural and forestry uses, were classified as *tepetates* types t₁, t₂, and t₃ with or without calcium carbonate.^[22]

This type of *tepetates* appears in different positions in the landscape, mainly in foothills and glacia, in the eastern and western slopes of the Sierra Nevada. Their characteristics differ in terms of the topographic position. Most of the *tepetates* located in foothills are of the fragipan type but

Table 1 Areas reported to have hardened layers of volcanic origin in different countries of Latin America.

Country	Name of the layer	Area (km ²)
Mexico ^a	Tepetate	30,700
Nicaragua ^b	Talpetate	2,500
El Salvador and Honduras ^c	Talpetate	Not reported
Ecuador ^d	Cangahua	3,000
Chile ^e	Ñadis y tobas	4,750 and not reported
Perú ^f	Sillares	10,000
Colombia ^g	Hardened volcanic formations	12,000–15,000

^aBertaux & Quantin.^[13]

^bPrat & Quantin^[8] and Miehllich.^[14]

^cInformation not available.

^dHidalgo.^[15]

^eZebrowski, Quantin, et al.^[16]

^fNimlos & Zamora.^[11]

^gFaivre & Gaviria.^[12]

do not have all the characteristics required by the Soil Taxonomy to be classified as such, the reason for which they were called fragipan type (Table 2). In the “glacis,” located further down foothills, *tepetates* show calcareous coatings and are more like petrocalcic horizons.^[23]

The most outstanding physical properties of the fragipan-type *tepetates* are an average bulk density of 1.45 Mg m⁻³; total porosity near 40%, with a macroporosity often below 5%; a hydraulic conductivity of 0.3–0.5 mm hr⁻¹; a hardness to dry penetration below 20 kg cm⁻²; and texture from sandy clayey loam to silty clayey loam (with more than 25% clay).^[5,6,21] This kind of *tepetates* expands and gets friable when it is moist and

disintegrates when it is sunk in water. The physical properties, especially hardness and the low values of macroporosity in natural conditions, are serious limitations to farming (root development and water penetration) and favor surface erosion.^[24] Loss due to erosion amounts to 30 Mg ha⁻¹.

The fragipan-type *tepetates* are rich in bases, among which calcium, magnesium, and potassium prevail; they come from a mineral fraction rich in volcanic glasses and plagioclase highly susceptible to pedogenetic alteration. The cation exchange capacity ranges from 20 to 40 cmol kg⁻¹ of fine earth, due to the abundance of 2:1 clays; the percentage of base saturation is high. Reaction is slightly alkaline, pH 7.3–8.2, because of the presence of carbonates. These materials contain only traces of organic carbon (0.2–0.7%), nitrogen (0.04–0.07%), and soluble phosphorus (<3 ppm), which is one of its major chemical problems. These characteristics of this type of *tepetates* result in low fertility, so it is necessary to improve it before their rehabilitation for agricultural or forest use.^[24,25]

The material of origin of the fragipan-type *tepetates* is mainly made up of alkaline rhyolitic glasses, plagioclases, amphiboles, and pyroxenes of different sizes.^[19,21] Secondary minerals are a mixture of 1:1 and 2:1 clays, often interstratified with a 2:1 strong component of the beidellite type. Clays are very well oriented, which favor the cohesion of the fragipan-type *tepetates* and strengthens the compact nature of their matrix. This *tepetate* type presents “free” non-crystalline silica, accumulation of iron and manganese oxides, and occasionally calcite and only traces of allophane. Mineralogical properties explain only partly the behavior and characteristics of these materials.^[5,18,19]

The fragipan-type *tepetates* can be converted into soil after mechanically breaking up the hardened material^[24] to improve their physical conditions (pore space and water

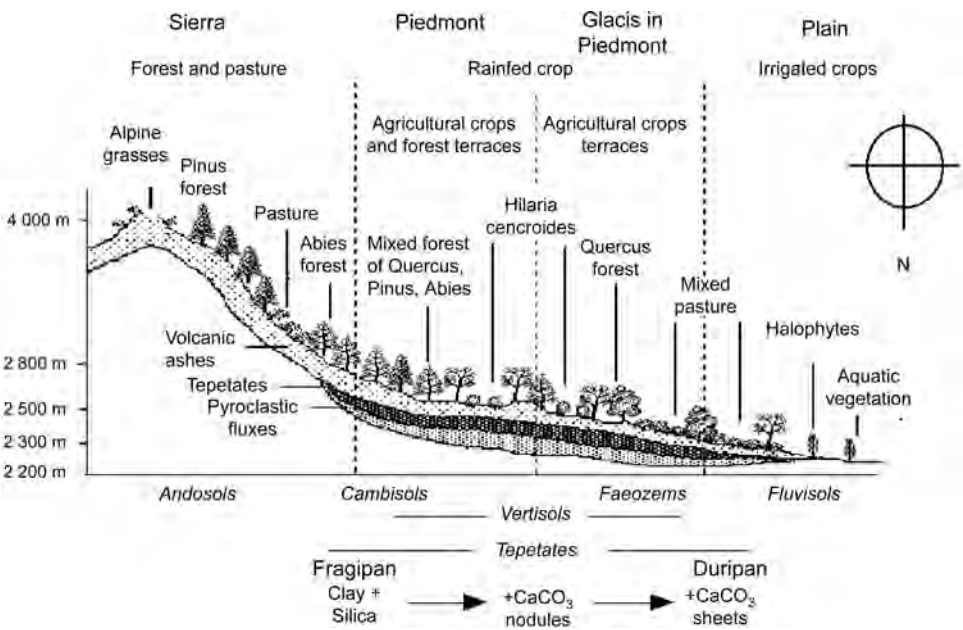


Fig. 1 Localization of the *tepetates* in the landscape in Central Mexico.

Table 2 Similarities and differences between fragipan and *tepetates*.

Characteristic	Fragipan ^a	Tepetate
Layer thick	15 cm or more	6 cm or more
Pedogenesis	Evidence within the horizon or, at minimum, on the faces of structural units	Evidence within the horizon
Layer structure	Very coarse prismatic, columnar, or blocky structure of any grade; weak structure of any size; massive	Massive structure predominant and virtually no structural units, blocky structure of any grade in the t ₃ <i>tepetate</i>
Separations between structural units	Separations allow roots to enter, have an average spacing of 10 cm or more on the horizontal dimensions	Virtually no separations in the massive structure, but the spacing in the t ₃ <i>tepetate</i> can be around 10 cm
Diameter of air-dry fragments of the natural soil fabric	5–10 cm	Virtually no fragments when dry
Water behavior	More than 50% of the horizon slake when they are submerged in water; slowly or very slowly permeable to water	The <i>tepetate</i> slakes when submerged in water, slowly or very slowly permeable to water
Layer resistance	A firm or firmer rupture-resistance class in 60% or more of the volume, a brittle manner of failure at or near field capacity	Hard and very hard when dry and friable to very friable when moist
Roots	Virtually no roots	Virtually no roots
Organic matter content	Relatively low	Very low
Bulk density	High, relative to the horizons above it	Very high, relative to the horizons of the profile
Translocation of clay	Some evidences	Very frequent

^aNatural Resources Conservation Service. *Keys to Soil Taxonomy* (7th edition 1996 and 8th edition 1998).

retention capacity). Chemical and organic fertilizers must also be added to increase the available nitrogen and phosphorus as well as the organic matter content.^[26] *Tepetates* are very poor in the first two; however, their content of the other essential nutrients and clay, as well

as their cation exchange capacity, is similar to that shown by productive soils.^[26]

Manual incorporation of the fragipan-type *tepetates* to agricultural or forest activity began centuries ago. Mechanized methods were introduced by the mid-20th century. The ripped-up layer is 40–50 cm deep. Deep plowing is done when the *tepetate* is dry because fracturing of hard horizons is more complete than under moist conditions.^[27,28]

Ancient experiences gathered by local farmers and modern research experiences allow to propose a series of management practices for the rehabilitation of the *tepetates* of the fragipan type: 1) The *tepetate* layer must be broken at a minimum depth of 40 cm and fragments of a size between 3 and 5 mm must be left after the mechanical or manual preparation (actions leading to a very fine fragmentation, less than 2 mm, must be avoided, as well as a frequent tillage of the *tepetate*); 2) After tillage, organic matter must be added (an application of 40 Mg ha⁻¹ of farmyard manure had a 4-year residual effect); 3) Nitrogen and phosphorus fertilizers must be applied as a complementary practice to the addition of organic matter; rates to be applied must be based on the crop demand and the nutrient supply capacity of the recovered *tepetates*; nitrogen applications are recommended to be made in a fractional manner so as to improve their use and efficiency; 4) The crops showing the best responses in the first years of the *tepetate* incorporation are small-grain cereals such as wheat, barley, and oats and fodder crops such as vicia, trifoliums, and wild medicagos; good responses are obtained only as of the second or third cropping year for maize, beans, and broad beans, mainly planted in association; 5) In the initial phase of establishing a crop, especially if small-grain cereals, it is advisable to increase up to 50% the sowing density in order to have a better plant population; 6) The adoption of a crop rotation pattern containing legumes and Gramineae is recommended.

CONCLUSION

Tepetates are layers of hardened material that develop in profiles of lands formed from volcanic material in Mexico and other countries with volcanic influence and are object of particular scientific interest. Amelioration of the fragipan-type *tepetates* allows small farmers to have access to agricultural land and to obtain a means of support, reducing their migration toward populated areas, and at the same time slowing down erosion and desertification.

REFERENCES

1. Gibson, C.H. *Los Aztecas Bajo el Dominio Español, 1519–1810*, 1st Ed.; Siglo XXI: México, 1967.
2. Williams, B.J. Tepetate in the valley of Mexico. *Ann. Assoc. Am. Geogr.* **1972**, *62*, 618–623.

3. Williams, B.J. "Tepetate" in 16th century and contemporary folk terminology, Valley of Mexico. *Terra* **1992**, *10*, 483–493. (No. especial).
4. Ortiz, C. Los levantamiento etnoedafológicos. Ph.D. Thesis; Colegio de Postgraduados: Mexico, 1999; 212.
5. Quantin, P. L'induration des matériaux volcaniques pyroclastiques en Amérique Latine: Processus géologiques et pédologiques. *Terra* **1992**, *10*, 24–33. (No. especial).
6. Quantin, P. *Étude Des Sols Volcaniques Indurés (Tepetates) Des Bassins de Mexico et Tlaxcala (Mexique)*; Rapport scientifique final. Commission des Communautés Européennes. Contrat CCE/ORSTOM no. TS2-0212; ORSTOM: Paris, 1992; 77 pp.
7. Prat, C. *Etude du "Talpetate", Horizon Volcanique Induré De La Région Centre-Pacifique Du Nicaragua: Genèse, Caractérisation Morphologique, Physico-Chimique Et Hydro-Dynamique, Son Rôle Dans L'érosion Des Sols*; Université de Paris 6: Paris, 1991.
8. Prat, C.; Quantin, P. Origen y génesis del talpetate: Horizonte endurecido de suelos volcánicos de la región Centro-Pacífico de Nicaragua. *Terra* **1992**, *10*, 267–282. (No. especial).
9. Vera, R.; López, R. Tipología de la cangahua. *Terra* **1992**, *10*, 112–119. (No. especial).
10. Luzio, W.; Alcayaga, S.; Barros, C. Origen y propiedades del "Fierrillo" de los suelos volcánicos de drenaje impedido en Chile. *Terra* **1992**, *10*, 67–73. (No. especial).
11. Nimlos, T.J.; Zamora, C.J. Indurated ash soils in Peru. *Terra* **1992**, *10*, 283–289. (No. especial).
12. Faivre, P.; Gaviria, S. Suelos y formaciones piroclásticas endurecidas en los Andes de Colombia. *Terra* **1992**, *10*, 89–99. (No. especial).
13. Bertaux, J.; Quantin, P. Les niveaux indurés des séries pyroclastiques récentes du Mexique Central: Reflets de la géométrie du dépôt et d'une altération pédologique. In *15ème réunion des Sciences de la Terre*; Nancy, 1993; 28 pp.
14. Miehllich, G. Formation and properties of *tepetate* in Central Highlands of Mexico. *Terra* **1992**, *10*, 137–144. (No. especial).
15. Hidalgo, C. Étude d'horizons indurés à comportement de fragipan, appelés tepetates, dans les sols volcaniques de la vallée de Mexico. In *Thèses et documents microfichés 146*; ORSTOM Éditions: Paris, 1996.
16. Zebrowski, C.; Quantin, P.; Arias, H.; Werner, G. Les "tepetates." Récupération et mise en valeur des terres volcaniques indurées au Mexique. *ORSTOM Actual.* **1991**, *33*, 13–18.
17. Werner, G. Suelos volcánicos endurecidos (*tepetates*) en el estado de Tlaxcala: Distribución, rehabilitación, manejo y conservación. *Terra* **1992**, *10*, 318–331. (No. especial).
18. Miehllich, G. *Chronosequences of Volcanic Ash Soils*; Hamburger Bodenkundliche Arbeiten: Hamburg, 1991; Vol. 15.
19. Peña, D.; Zebrowski, C. Los suelos y tepetates de la vertiente occidental de la Sierra Nevada. *Terra* **1992**, *10*, 151–155. (No. especial).
20. Gutiérrez, C.C.; Ortiz, C. Caracterización del tepetate blanco en Texcoco, México. *Terra* **1992**, *10*, 201–209. (No. especial).
21. Zebrowski, C.; Quantin, P.; Trujillo, G. *Suelos Volcánicos Endurecidos*; ORSTOM: Quito, 1997.
22. Etchevers, J. Factores físicos, químicos y biológicos que afectan la productividad de los suelos volcánicos endurecidos. In *Suelos Volcánicos Endurecidos*; Zebrowski, C., Quantin, P., Trujillo, G., Eds.; ORSTOM: Quito, 1997; 155–161.
23. Etchevers, J.; Zebrowski, C.; Hidalgo, C.; Quantin, P. Fertilidad de los tepetates: II. Situación del fósforo y del potasio en tepetates de México y Tlaxcala. *Terra* **1992**, *10*, 386–391. (No. especial).
24. Zebrowski, C.; Sánchez, B. Los costos de rehabilitación de los suelos volcánicos endurecidos. In *Suelos Volcánicos Endurecidos*; Zebrowski, C., Quantin, P., Trujillo, G., Eds.; ORSTOM: Quito, 1997; 462–471.
25. Márquez, A.; Zebrowski, C.; Navarro, H. Alternativas agronómicas para la recuperación de tepetates. *Terra* **1992**, *10*, 465–473. (No. especial).
26. Martín, E. Contribución al conocimiento de la génesis del Talpetate en Nicaragua. Publicaciones Geológicas del ICAITI **1973**, *4*, 123–138.
27. Colmet-Daage, F. *Cartes de Sols de la Sierra Équatorienne à 1/50 000*; MAG-ORSTOM: Quito, 1975–1984.
28. Instituto de Investigaciones Agropecuarias. *Cartas de Suelos*; Ministerio de Agricultura: Santiago, 1985.

Termites

David E. Bignell

*School of Biological Sciences, Queen Mary and Westfield College,
University of London, London, U.K.*

John A. Holt

James Cook University, Townsville, Queensland, Australia

Abstract

Termites are abundant soil invertebrates in much of the tropics and subtropics. The physical effects of termites on soils range from the micromorphological to soil profile evolution and soil structure. There is evidence of enhancement of soil hydraulic conductivity and infiltration rates. Chemical effects include the promotion of litter decomposition, nutrient recycling, and the formation of stable pools of soil organic matter. In some landscapes, termite mounds act as foci for nutrient redistribution and contribute inorganic nitrogen following symbiotic fixation in the insect's gut. Termite mounds are sites of activity for diverse bacteria and fungi.

INTRODUCTION

Termites have a profound effect on soil properties, resulting both from their mound-building and gallery excavating activities and from their modes of feeding. Although other soil animals can affect soil properties, the importance of termites lies in their huge abundance and biomass in the tropics and subtropics and their unique associations with a diversity of microorganisms. Contrary to common assumptions, most termite species are beneficial to plant growth: they promote organic matter decomposition and nutrient cycling and reduce erosion through a strong influence on soil physical, chemical, and biological processes. They also have long-term effects on soil development and the character of landscapes. For a modern synthesis of termite biology refer Abe et al.,^[1] and for a review of economic impact and control methods, see Pearce.^[2]

TYPES

Termites are eusocial, polymorphic insects (order Isoptera) with generally cryptic behavior. They live in large family groups that are composed of reproductive forms (sometimes winged), together with numerous sterile soldiers and workers. Sometimes known inaccurately as “white ants,” termites are derived forms of cockroach which depend on mutualistic intestinal bacteria and, in some higher forms, externally cultivated basidiomycete fungi, for assistance with energy metabolism and the provision of nitrogen (N). Some termites also contain populations of flagellate protozoans in the hindgut, a number of which have the ability to degrade cellulose and other plant polysaccharides. Termites are the only social insects with a true soldier caste having no role other than to

defend the colony. More than 2300 species have been described in 270 genera and 6 families. Most of these are beneficial inhabitants of tropical and subtropical forests, savannas, and rangelands. A small number of relatively primitive species, loosely termed “drywood termites,” “dampwood termites,” and “subterranean termites,” are able to colonize and/or consume timber used for buildings, furniture, transmission poles, and fencing; such pest species are inadvertently distributed by humans and become pantropical, with one or two species extending their range to include cities in temperate zones. Other species are agricultural or silvicultural pests, especially where non-indigenous crops are introduced or where land is cleared of natural plant detritus prior to planting or not fertilized with mulch.^[3] The large majority of termite species are not pests and can be grouped heuristically into four broad functional categories based on their diets: grass-feeders, litter-feeders, wood-feeders, and soil-feeders. The last group dominate lowland tropical forests and the wetter savannas; grass-feeders and wood-feeders become relatively more common as aridity increases; almost all termite species are associated with the soil in some way, using it as either a habitat, a resource, or both. Under most global change scenarios, non-soil-feeding species are likely to become more dominant, but overall termite biomass may decline.^[4,5]

HABITAT

Termite diversity is strongly influenced by latitude (the numbers of naturally occurring taxa declining steadily north and south of the Equator) and by rainfall (species richness and abundance generally increasing with greater rainfall at the same latitude). Constructions made by

termites include: numerous galleries tunnelled through wood and soil; underground chambers containing the queen, king, larval nurseries, and (in the subfamily *Macrotermitinae*) symbiotic fungus gardens; runways attached to the sides of trees, decaying wood, and buildings; soil sheeting covering the surface of the ground; and arboreal or epigeal (i.e., emerging from the surface) mound nests with a complex internal structure. Materials utilized, manufactured, or translocated by termites include: surface soil; subsoil; compacted feces; and carton (a lightweight organic-rich mixture of partially digested cellulose, saliva, and soil). Epigeal mounds can attain a height of more than 7 m but most are much smaller (Fig. 1). Many termites live underground with little indication of their presence at the soil surface. A large number of species are not primary mound builders, either living entirely in diffuse subterranean gallery networks or becoming established in pockets within the mounds of other species as secondary occupants.^[6,7] Arboreal species are often conspicuous, but less important in biomass terms than subterranean forms, except where inundation is frequent. All termites are detritivores, consuming and digesting (in different species) a wide range of freshly dead or decaying plant material including dry grass, leaf litter, sound wood, decaying wood, dung, and humus-rich soil. Lichen and living roots may also be eaten, often without damage to plants. Termite abundance varies from less than 50 individuals m⁻² in arid savannas to more than 7000 m⁻² in some African forests. The corresponding biomass densities range from negligible to more than 100 g live weight m⁻² in exceptional cases. In the tropics as a whole, they are estimated to constitute 10% of all animal biomass (up to 95% of soil insect biomass) and to impact C mineralization (decomposition) to roughly the same extent as all mammalian herbivores and natural fires. Overall, they contribute about 2% of the CO₂ flux to the atmosphere from

all terrestrial sources. Earlier concerns about the possible role of CH₄ production by termites in global warming have now been shown to be unfounded.^[8]

EFFECTS ON SOILS

Through their activities as decomposers and constructors, termites have important roles in the genesis, morphology, and properties of soils throughout the tropics and subtropics.^[5] They also fix N on a significant scale, through mutualistic gut bacteria, and may be the primary providers of this nutrient in many soils and food chains, with termite mounds serving as foci for nutrient redistribution in some landscapes. It is highly probable that termites function as keystone species in ecosystems (creating niche opportunities for a large number of other organisms) and as ecosystem engineers (mediating biological function and maintaining soil structure, fertility and stability). Evidence is accumulating that termites (with earthworms) strongly affect C mineralization, the establishment of stable pools of complex organic material, N fixation, and denitrification. As a consequence, long-term exclusion or poisoning of termites changes the character of plant communities as well as adversely affecting soil porosity, aeration, water-holding capacity, hydraulic conductivity, infiltration rates, and drainage.^[4,7]

The physical effects of termites on soils range from micromorphological to soil profile evolution and structure.^[7] Soil translocation is a major consequence of nest building, and the construction of foraging runways at the soil surface or above the ground. Estimates of the quantities moved vary but may approach 5 ton ha⁻¹ year⁻¹ in some systems; over thousands of years, these activities strongly influence soil profile development, especially lateritic profiles and ferrisols.^[7,9,10] Micromorphological analysis of tropical soils frequently shows the presence of small (<1 mm pellets of biological origin, thought to be produced either by termite fecal deposits or the mixing of soil with termite saliva using the mandibles. The pellets, which may comprise up to 20% of the soil matrix, have a major influence on microporosity, moisture storage capacity, and infiltration rate. Subterranean chambers and galleries are also a conspicuous feature of soil profiles, almost universally detectable in soil cores and accounting for up to 2% of total soil volume, with effects on both infiltration rates and drainage. The behavior of termites in repacking soil particles, augmented with organic material derived from salivary and fecal products, while constructing mounds and galleries, is a key activity.^[7] It can be shown to affect bulk density (this is increased, restricting seed lodgement and growth), structural stability (this is increased when termites construct mounds that are highly organic, e.g., containing hemicellulose or glycoprotein), and hydraulic conductivity (both conductivity and flow rates are reduced if termite galleries are reduced and, correspondingly, increased after mulching that attracts additional termite foraging).



Fig. 1 A termite-dominated landscape: mounds of *Amitermes* *viriosus* in mulga country, West MacDonnell National Park, Northern Territory, Australia.

Source: Photo by D.E. Bignell.

Table 1 Soil properties significantly affected by termites.

Soil property	References
Soil profile development	[7,10–12]
Bulk density	[7]
Hydraulic conductivity/infiltration rates	[7,13]
Soil chemical properties	[5]
Organic matter decomposition	[7]

The chemical properties of soils depend primarily on their parent materials but with additional influence from climate, vegetation cover, and the activities of soil organisms. A notable contribution of termites is the incorporation of cation-rich clay subsoils, together with organic-rich fecal and salivary materials, into constructions.^[5,7,9] In many cases, when these are subsequently eroded, the surface soil becomes enriched in Ca, magnesium, sodium, potassium, together with organic N, phosphorous, and interstratified clay minerals. Several studies have shown that the inorganic nitrogen concentrations of mound soil or of soil reworked by termites are higher than adjacent soils that are unaffected by termites; this nitrogen apparently originates from symbiotic fixation within the termite gut. Leaching of such nitrogen represents a short-term redistribution and can be a significant input to the soil pool that is immediately available to plants.^[7] Non-mutualistic bacteria and fungi may also be present in larger numbers in mound soils, and the assessments of their activities suggest a vigorous metabolism which includes (in separate species) cellulose degradation, nitrification, and denitrification. Many evidences indicate that the presence of termite mounds promotes the accumulation of plant biomass on or around the structure.^[3,7] In semiarid systems, mulching of the soil to promote termite activity can lead to significant improvements in plant cover, plant species number, biomass production, and rainfall use efficiency. A summary of the effects of termites on soils is presented in [Table 1](#).

CONCLUSION

Termites are abundant soil invertebrates in much of the tropics and subtropics. The physical effects of termites on soils range from the micromorphological to soil profile evolution and soil structure. There is evidence of enhancement of soil hydraulic conductivity and infiltration rates. Chemical effects include the promotion of litter decomposition, nutrient recycling, and the formation of stable pools of soil organic matter. In some landscapes, termite mounds

act as foci for nutrient redistribution and contribute inorganic nitrogen following symbiotic fixation in the insect's gut. Termite mounds are sites of activity for diverse bacteria and fungi.

REFERENCES

1. Abe, T.; Bignell, D.E.; Higashi, M. *Termites: Evolution, Sociality, Symbioses, Ecology*; Kluwer Academic Publishers: Dordrecht, 2000; 466 pp.
2. Pearce, M.J. *Termites, Biology and Pest Management*; CAB International: Wallingford, 1997; 172 pp.
3. Wood, T.G. The agricultural importance of termites in the tropics. *Agr. Zool. Rev.* **1996**, *7*, 117–155.
4. Lavelle, P.; Bignell, D.E.; Lepage, M. Soil function in a changing world; the role of invertebrate ecosystem engineers. *Eur. J. Soil Biol.* **1997**, *33*, 159–193.
5. Black, H.I.J.; Okwakol, M.J.N. Agricultural intensification, soil biodiversity and agroecosystem function in the tropics: The Role of Termites. *Appl. Soil Ecol.* **1997**, *6*, 37–53.
6. Noirot, C.; Darlington, J.P.E.C. Termite nests. Architecture, regulation and defence. In *Termites: Evolution, Sociality, Symbioses, Ecology*; Abe, T., Bignell, D.E., Higashi, M., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 121–139.
7. Holt, J.A.; Lepage, M. Termites and soil properties. In *Termites: Evolution, Sociality, Symbioses, Ecology*; Abe, T., Bignell, D.E., Higashi, M., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 389–407.
8. Sugimoto, A.; Bignell, D.E.; MacDonald, J.A. Global impact of termites on the carbon cycle and atmospheric trace Gases. In *Termites: Evolution, Sociality, Symbioses, Ecology*; Abe, T., Bignell, D.E., Higashi, M., Eds.; Kluwer Academic Publishers: Dordrecht, 2000; 409–435.
9. Lobry de Bruyn, L.; Conacher, A.J. The role of termites and ants in soil modification: A Review. *Aust. J. Soil Res.* **1990**, *28*, 55–93.
10. Lal, R. *Tropical Ecology and Physical Edaphology*; John Wiley and Sons: Chichester, 1987; 732 pp.
11. Stoops, G. Relict properties of soils in humid tropical regions with special reference to central Africa. *Catena Suppl.* **1989**, *16*, 95–106.
12. Tardy, Y.; Roquin, C. Geochemistry and Evolution of Lateritic Landscapes. In *Weathering, Soils & Paleosols*; Martini, I.P., Chesworth, W., Eds.; Elsevier: London, 1992; 407–443.
13. Ouedraogo, P.; Lepage, M. Rôle des Termitières de *Macrotermes subhyalinus* dans une Brousse Tigrée (Yatenga, Burkina Faso). In *Fonctionnement et Gestion des Écosystèmes Forestiers Contractés Sahéliens*; Herbès, M., Ambouta, J.M.K., Peltier, R., Eds.; John Libbey Eurotext: Paris, 1997; 91–94.

Terra Preta (Black Earths)

Kurt Spokas

U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
St. Paul, Minnesota, U.S.A.

Jose Marques, Jr.

Department of Soil and Fertilizers, Sao Paulo State University, Sao Paulo, Brazil

Newton La Scala, Jr.

Department of Exact Sciences, College of Agricultural and Veterinarian Sciences,
Sao Paulo State University, Sao Paulo, Brazil

Ed Nater

Department of Soil, Water and Climate, University of Minnesota, St. Paul,
Minnesota, U.S.A.

Diego S. Siqueira

Department of Soil and Fertilizers, Sao Paulo State University, Sao Paulo, Brazil

Abstract

Fertile dark-colored soils in the Amazon (anthropogenic dark earths or *terra preta de Indio*) have been attached to historic anthropogenic soil modification through black carbon and other organic amendments from both agricultural and waste management activities of ancient Amazonian natives. This hypothesis was supported by the presence of artifacts (e.g., ceramic pieces) located within these soils and by the assumption that these deposits were unique to the Amazon region. This model of *terra preta* development has fueled the renaissance into recreating bolstered soil fertility through intentional black carbon additions.

Fertile dark-colored soils in the Amazon [anthropogenic dark earths or *terra preta de Indio* (TPI)] have been attached to historic anthropogenic soil modification through black carbon and other organic amendments from both agricultural and waste management activities of ancient Amazonian natives.^[1,2] This hypothesis was supported by the presence of artifacts (e.g., ceramic pieces) located within these soils^[3] and by the assumption that these deposits were unique to the Amazon region.^[4] This model of *terra preta* development has fueled the renaissance into recreating bolstered soil fertility through intentional black carbon additions.^[5] However, improving soil is an intricate process involving not solely the addition of an individual chemical component but also the interactions between all of the following soil-forming factors: climate, topography, parent material, organisms (vegetation), and time.^[6] This entry expands on the observations of a new possible interaction between fire, time, and iron mineralogy that might play a significant role in the development of these TPI soils.

Terra preta soils are chemically distinguished by the presence of Ca^{+2} , phosphorus (P), Mg^{+2} , high base

saturation, and carbon contents and by the absence of extractable Al^{+3} compared to surrounding soils and geologic materials.^[7,8] Due to the contrasting soil chemistries of surrounding and deeper layered soils, the presence of calcium, P, and the pottery fragments has typically provided an evidence for anthropogenic influences in the development.^[9] However, the exact soil formation history is not fully elucidated. The most prominent feature of the *terra preta* is the dark brown to black coloration (e.g., Munsell® colors 10YR3/1; 5YR2/2), which is in stark contrast to the yellow to orange coloration of the surrounding geologically derived soil (e.g., 5YR5/8).^[3,8,9] Despite the fact that there is evidence of anthropogenic activity in some of the *terra preta* soils,^[4] not all of the *terra preta* soils have this characteristic. Based on the observations of other soils in Brazil^[11] and other continents,^[10] our suggestion here is that the *terra preta* (and other black carbon rich) soils are derived from fire residuals as a soil parent material and not solely linked to anthropogenic alteration.

In Southern Brazil (Monte Alto, São Paulo state; 21.3°S 48.5°W, elevation 700 m), distant from the Amazonian region, there are other examples of dark fertile soils in

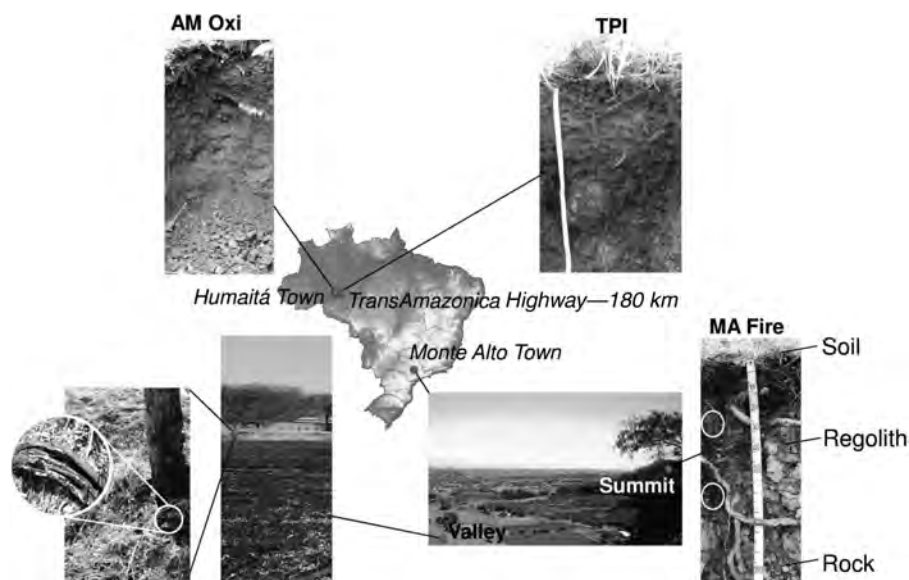


Fig. 1 Illustrations of location of the soil profile in Humaitá Town (AM Oxi),^[6,11] TransAmazonica Highway—180 km (TPI),^[6,11] and Monte Alto Town (MA Fire) in Brazil, the carbon-rich soil material comprising the first 15–30 cm of soil at the Monte Alto site (white circle in the soil profile MA Fire) at a topographic summit, in particular the movement of the black-colored soil by erosion down the vertical face of the exposed outcrop, the thick (>50 cm) fertile soil located in the depositional valley, and the charred tree stumps and remaining forest debris that clearly show the evidence of fire activity at the site.

tropical climates. We investigated one of these sites that possess a history of natural fires which occurs during drought periods (Fig. 1). This site is on a topographic high, void of any potential agricultural amendment practices. The residuals from this reoccurring natural fire activity have produced a veneer surface of carbon-rich soil with contrasting soil chemistries compared to the surrounding and deeper layer geological derived soil. This new carbon-rich soil, derived from parent material with very low or zero magnetic expression, ranges in depth from 10–35 cm in the topographic high areas to >50 cm in the base of the valley, in Southern Brazil. This soil possesses elevated concentrations of P, Ca^{+2} , and Mg^{+2} , high base saturation levels, and elevated soil carbon, when compared to surrounding soils, the typical Southern Brazilian or Amazonian Oxisols (AM Oxi; Fig. 1). Hence, these soil properties of Monte Alto (MA) fire site shared with the *terra preta* soils in the Amazon (TPI compared to MA Fire) are in stark contrast to native soil properties (AM Oxi). In addition, the iron oxides of maghemite and magnetite are also present confirming the occurrence of fire,^[12] which is corroborated by the visible charring of the trunks and forest debris at the Monte Alto site. The valleys in Monte Alto area contain fertile black earth soils (Mollisols; Fig. 1). Most striking is the similarity in soil chemistries of these newly formed soils from fire residues and the *terra preta* soils (Fig. 2), which are also shared with soils in the Monte Alto valley. Total organic carbon and P contents in shallow soil layers in TPI and MA Fire sites are with much higher contents than the typical Oxisols. The same applies for bases, aluminum (Al) + hydrogen (H), Ca^{+2} , and other elements, which have higher contents at shallower layers (lighter colors) compared to the deeper soil layers (darker colors) in MA Fire and TPI sites.

The magnetic susceptibility (MS) values (also presented in Fig. 2 and Table 1) suggest a higher presence of

ferromagnetic crystals (like maghemite) in MA Fire and TPI soils. The geologic parental material at the MA Fire is a carbonated rock, with low iron content and low MS value of $22 \times 10^{-8} \text{ m}^3 \text{ kg}^{-1}$. In this environment, carbon in soil organic matter is hypothesized to be preserved due to high pH values. But with fire occurring, soil carbon could additionally form complexes with ferromagnetic iron minerals. MS values are higher in MA Fire ($97 \times 10^{-8} \text{ m}^3 \text{ kg}^{-1}$), as ferromagnetic iron minerals formed by fire are leached, accumulating between the fire residues on surface and the rock. However, the overall presence of MS in the soil profiles also suggests substantial mixing in these soils, which could result as a consequence of fire-impacted areas possessing a high erosion potential and a propensity for landslides.^[13,14] This infrequent but massive movement of soil as being the major contribution of dissolved sediment in tributary rivers in the Amazon has been confirmed.^[14] The higher MS value of AM Oxi ($70 \times 10^{-8} \text{ m}^3 \text{ kg}^{-1}$) is a result of the higher iron content of parent material, different from the MS value of MA Fire, but both have their MS increased due to this interaction between soil carbon, iron, fire, and climate.

These data support a conclusion that both the Amazonian *terra preta* and Monte Alto soils have resulted from similar parent materials, in other words, fire residuals from natural vegetation fires. Natural wildfires do occur in tropical rain forests, with typical concentration around the transitional edges of the forest (cerrado or savanna), as well as topographic highs. These fires are triggered by lightning during seasonal water stress, which can be assumed to have occurred in the geologic past, in both the mainland of Brazil and the Amazon.^[13–15]

The fundamental assertion here is that these *terra preta* soils originated directly from this fire residual parent material. The use of these naturally fire-cleared areas by the past Amazonian natives would also make logistical

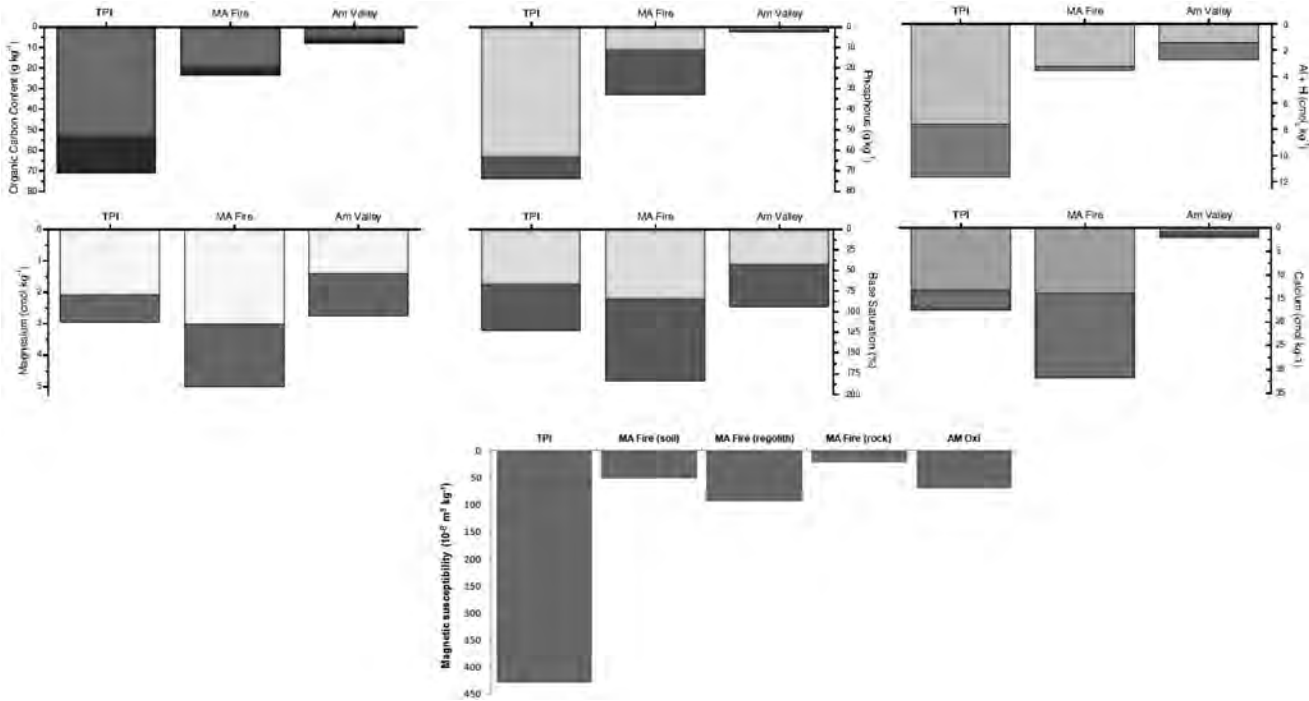


Fig. 2 Chemical data from the average of *terra preta* soils in the Amazon (TPI),^[16] MA Fire,^[17] and native Amazonian Valley (AM Oxi) illustrating the distinct chemical differences of both the MA Fire and TPI soils compared to the typical oxisol soil for organic carbon content, P, Al + H, base saturation, magnesium, calcium, and MS. Surface soil (0–20 cm) is shown in lighter shade, and deeper intervals are shown in darker shading.

sense, because nature strives for the path of least resistance. Following these fire events, the soils are also highly susceptible to erosion by massive debris flows (landslides)

during storm events. This overall mechanism agrees with the data collected from dissolved sediments in the upper Amazonian headwaters,^[14] confirming that the majority

Table 1 Surface (0–20 cm) and deeper soil data (as in Fig. 2) from the average of *terra preta* soils in the Amazon (TPI),^[16] MA Fire,^[17] and native Amazonian Valley (AM Oxi) illustrating the distinct chemical differences of both the MA Fire and TPI soils compared to the typical oxisol soil for organic carbon content, P, Al + H, base saturation, magnesium, calcium, and MS.

Organic carbon (g kg ⁻¹)			P (g kg ⁻¹)		Al + H (cmol kg ⁻¹)	
Site	0–20 cm	Deeper	0–20 cm	Deeper	0–20 cm	Deeper
TPI	53.18	17.78	63.25	10.44	7.6	4.05
MA Fire	19.8	3.8	11	22	3.2	0.3
AM Oxi	5.15	2.8	1.4	1.1	1.4	1.35
Magnesium (cmol kg ⁻¹)			Base saturation (%)		Calcium (cmol kg ⁻¹)	
Site	0–20 cm	Deeper	0–20 cm	Deeper	0–20 cm	Deeper
TPI	2.08	0.87	66.23	56.5	13.13	4.38
MA Fire	3	2	84	99	13.7	18.1
AM Oxi	1.4	1.35	42.5	51.5	0.8	1.15
MS (10 ⁻⁸ m ³ kg ⁻¹)						
TPI	428.8					
MA Fire (soil)	52					
MA Fire (regolith)	94					
MA Fire (rock)	22					
AM Oxi	70.3					

of sediment enters in very brief but substantial loading events that are infrequent and not solely linked to seasonal patterns.

In the classic pedogenic model, Jenny^[18] equated climate, topography, parent material, organisms (vegetation), and time as the factors determining soil formation, with parent material being thought of as physical and chemical transformation of geologic formations. However, our new observations in Southern Brazil coupled with other reports of other *terra preta*-like soils in other continents^[10] support the conclusion that these black earth soils should be considered as derived from fire residual parent materials only and not from the modification of existing geologic soils, and ferromagnetic minerals could be used as a pedogenic indicator of soil evolution in the genesis of TPI soils or any other soils formed by fire residues. This new parent material classification would provide harmony and clearer linkages between other localized occurrences of increased black carbon soils present across all of Brazil as well as on other continents.

CONCLUSION

Our results suggest a new vision on the evolution and formation of soils derived from fire, which have broader evidences from other sites but have similar aspects such as iron content in soil plus fires. The MS has the potential to be used as an indicator of the quality and evolution of soils formed in the presence of fire. The understanding of the formation of iron oxides and their changes induced by fire in those natural environments could assist in understanding the potential soil carbon accumulation of those sites.

REFERENCES

1. Glaser, B.; Haumaier, L.; Guggenberger, G.; Zech, W. The 'Terra Preta' phenomenon: A model for sustainable agriculture in the humid tropics. *Naturwissenschaften* **2001**, *88*, 37–41.
2. Mann, C. The real dirt on rainforest fertility. *Science* **2002**, *297*, 920–923.
3. Lima, H.N.; Schaefer, C.E.R.; Mello, J.W.V.; Gilkes, R.J.; Ker, J.C. Pedogenesis and pre-Colombian land use of "Terra Preta Anthrosols" ("Indian black earth") of Western Amazonia. *Geoderma* **2002**, *110* (1–2), 1–17.
4. Heckenberger, M.J.; Petersen, J.B.; Neves, E.G. Village size and permanence in Amazonia: Two archaeological examples from Brazil. *Lat. Am. Antiq.* **1999**, *10* (4), 353–376.
5. Marris, E. Black is the new green. *Nature* **2006**, *442*, 624–626.
6. Santos, L.A.C.; Campos, M.C.C.; Aquino, R.E.; Bergamin, A.C.; Silva, D.M.P.; Marques, J., Jr.; França, A.B.C. Caracterização de terras pretas arqueológicas no sul do estado do Amazonas. *Rev. Bras. Ciênc. Solo* **2013**, *37* (4), 825–836.
7. Artaxo, P.; Oyola, P.; Martinez, R. Aerosol composition and apportionment in Santiago de Chile. *Nucl. Instrum. Methods* **1999**, *150*, 409–416.
8. Glaser, B.; Balashov, E.; Haumaier, L.; Guggenberger, G.; Zech, W. Black carbon in density fractions of anthropogenic soils of the Brazilian Amazon region. *Org. Geochem.* **2000**, *31* (7–8), 669–678.
9. Eden, M.J.; Bray, W.; Herrera, L.; McEwan, C. Terra Preta soils and their archaeological context in the Caquetá Basin of southeast Colombia. *Am. Antiquity* **1984**, *49*, 125–140.
10. Downie, A.E.; Van Zwieten, L.; Smernik, R.J.; Morris, S.; Munroe, P.R. Terra Preta Australis: Reassessing the carbon storage capacity of temperate soils. *Agr. Ecosyst. Environ.* **2011**, *140* (1–2), 137–147.
11. Campos, M.C.C. *Pedogeomorfologia aplicada à ambientes amazônicos do Médio Rio Madeira*. Available at http://ufrpe.br/pgs/portal/files/teses/2009/Milton_Costa_Campos.pdf (accessed September 2014).
12. Schwertmann, U.; Fechter, H. The influence of aluminum on iron oxides: XI. Aluminum-substituted maghemite in soils and its formation. *Soil Sci. Soc. Am. J.* **1984**, *48*, 1462–1463.
13. Sanford, R.L. Jr.; Saldarriaga, J.; Clark, K.E.; Uhl, C.; Herrera, R. Amazon rain-forest fires. *Science* **1985**, *227* (4682), 53–55.
14. Cochrane, M.A. Fire science for rainforests. *Nature* **2003**, *421*, 913–919.
15. Cochrane, M.A.; Alencar, A.; Schulze, M.D.; Souza, C.M. Jr.; Nepstad, D.C.; Lefebvre, P.; Davidson, E.A. Positive feedbacks in the fire dynamic of closed canopy tropical forests. *Science* **1999**, *284* (5421), 1832–1835.
16. Campos, M.C.C.; Ribeiro, M.R.; Souza Júnior, V.S.; Ribeiro Filho, M.R.; Souza, R.V.C.C.; Almeida, M.C. Characterization and classification of archaeological dark earths from the Middle Madeira River Region. *Bragantia* **2011**, *70*, 598–609.
17. Marques, J. Jr.; Lepsch, I.F. Depósitos superficiais neoceno-zóicos, superfícies geomórficas e solos Monte Alto, SP. *Geociências* **2001**, *19*, 90–106.
18. Jenny, H. Arrangements of soil series and types according to functions of soil forming factors. *Soil Sci.* **1946**, *61*, 375–391.

Terra Preta de Indio

Johannes Lehmann

Department of Crop and Soil Science, Cornell University, Ithaca, New York, U.S.A.

Abstract

Terra preta soils are fertile and carbon (C)-rich soils found throughout Amazonia. These soils have been created by Amerindian populations before the arrival of Europeans. The key to their unique properties is the high charcoal (also called black C or biochar) contents that have great stability and ability to improve both chemical and biological properties of soils. Important lessons can be learned from terra preta about the development of civilization in the Amazon, basic functioning of soils, C sequestration, and sustainable land use.

INTRODUCTION

Terra preta de indio soils, also called Amazonian dark earths,^[1] are found throughout most of the Amazon Basin. Terra preta means “black earth” in Portuguese, and their dark color makes them distinctly different from the surrounding soils. The interest in terra preta arises from their unique properties, namely their high soil fertility and production potential as well as high carbon (C) contents, which have implications both for the study of the development of civilization in the Amazon and for modern sustainable land use.^[1] Terra preta soils were described in the scientific literature for the first time by Charles F. Hartt,^[2] although in-depth scientific research documenting its unique properties started only in the middle of the 20th century. This entry gives a brief overview of the properties, origin, and distribution of terra preta soils.

PROPERTIES OF TERRA PRETA DE INDIO

The dark color of terra preta results from large amounts of both organic matter and charcoal (also called “black C” in biogeochemical literature and “biochar” when linked to purposeful soil management).^[3] The source of the charcoal can be linked to occupation by Amerindian populations, and the charcoal is therefore several hundred to a few thousand years old as seen from the ¹⁴C dates obtained from charcoal fragments (Fig. 1). The high concentrations of charcoal in terra preta are also a testament to its high stability against mineralization. Indeed, the organic matter in terra preta is highly resistant against microbial decay and that recalcitrance barely changes over millennial timescales.^[4]

Charcoal is the reason for the high cation exchange capacity of terra preta soils^[5] with high concentrations of plant-available nutrients and often higher pH than that

of the surrounding soils^[1] (Fig. 1). The elevated nutrient concentrations are also a result of burial activities, food preparation and waste (especially phosphorus (P)- and calcium (Ca)-rich fish waste), animal and possible human excrements, debris of housing materials such as straw or palm leaves, and various other activities (dyes, oils, fiber from palms and bark, etc.).^[6] The characteristics are the often very high P and Ca contents in terra preta, but also magnesium, manganese, zinc, and copper and to a lesser extent nitrogen and potassium contents are typically elevated.^[1]

The chemical properties and chiefly the high charcoal contents also change the environment for biological processes in terra preta. For example, the diversity of microbial populations commonly shows a greater species richness.^[7] The mechanisms by which abundance and ecology of soil biota are affected are not sufficiently clear. It is plausible that better nutrient availability or higher pH values enhance the activity and reproduction of microorganisms and alter species composition. Observations about the interactions between charcoal and microorganisms also suggest specific effects of charcoal on soil ecology.^[8] In many instances, terra preta soils contain large amounts of pottery shards, indicating past occupation that may also modify soil properties.

THE ORIGIN OF TERRA PRETA DE INDIO

The origin of terra preta has been the subject of much debate.^[9] While early reports speculate about volcanic ash deposits or sedimentation processes, it is by now accepted that terra preta soils were created by human activity^[10] and are therefore aptly called “terra preta de indio”—black earth made by Amerindians. The main arguments for the human origin are the abundance of artifacts in combination with high charcoal and P contents in terra preta soils (Fig. 1) and similarity in mineralogy of clays between terra preta

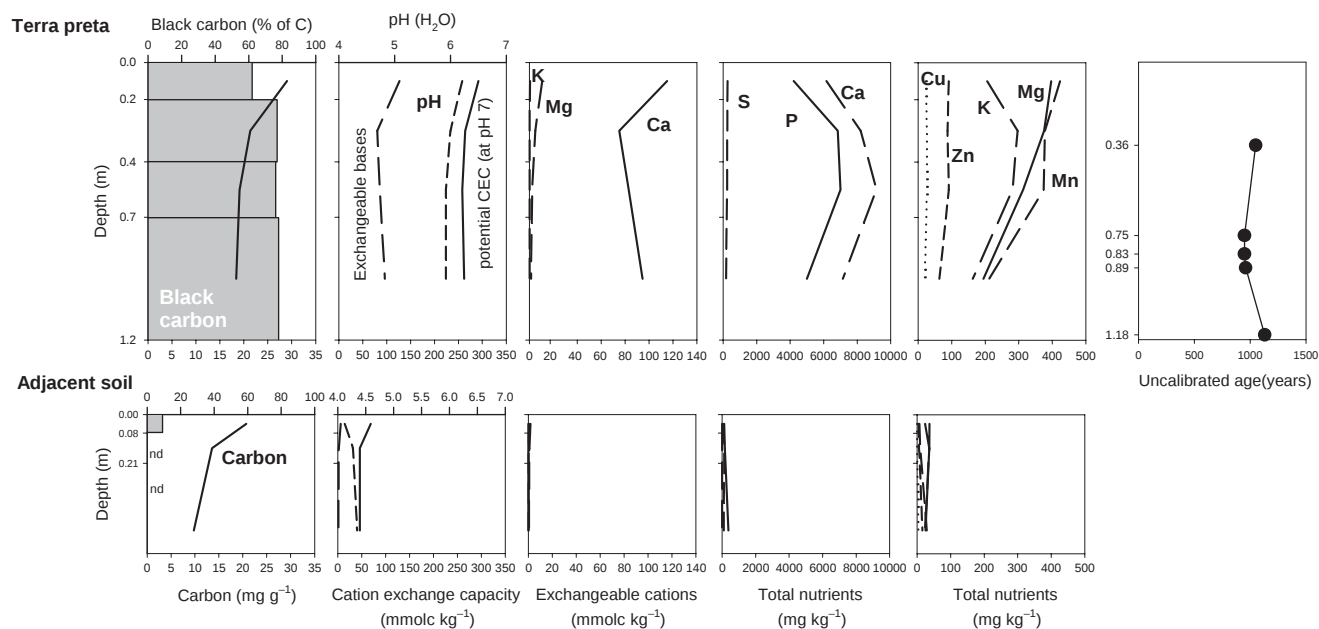


Fig. 1 Physical and chemical properties of a terra preta profile and the adjacent soil near Lago Grande, Manaus, Brazil (location and chemical analyses and uncalibrated ^{14}C dates of charcoal; nd: not determined).

Source: From Liang, Lehmann, et al.,^[4] Liang, Lehmann, et al.,^[5] and Neves, Petersen, et al.^[6]

and the surrounding soils.^[4] While some uncertainty will likely remain as to whether terra preta soils were created intentionally as part of soil management for agriculture or they are a by-product of habitation, it is certain that their occurrence is inextricably linked to Amazonian civilizations before the arrival of the Europeans. Amazonia was densely populated around the year 1492, being home to sophisticated populations who lived in urban clusters.^[11] Given that shifting cultivation is an unlikely land use system before steel axes made frequent slashing of trees possible,^[12] a purposeful soil management is a more likely explanation for the high and sustained crop productivity needed to feed larger urban centers. Terra preta soils were created between 500 and 8000 years B.P.,^[6] but not to any significant extent after the arrival of the first European explorers in 1541. They are therefore remnants of ancient civilizations in the Amazon Basin.

DISTRIBUTION OF TERRA PRETA DE INDIO

Terra preta soils are widely distributed in the Amazon Basin, but their occurrence is poorly documented. It appears that terra preta soils are most widespread in the Central Amazon between Manaus and Santarém (Fig. 2). Most terra preta sites have been mapped close to major streams that provide transportation by boat and food through fishing activities. Sites are located preferably on bluffs, which improve control over the surrounding area in case of warfare.^[9] Some studies, however, find more than half the sites to be far from any major river.^[10] As surveys

extend further into remote areas and away from waterways, the available information may be refined.

Terra preta occurs on all major soil types,^[13] indicating that physical and chemical attributes of the soil prior to occupation did not play a major role in their distribution. The enrichment with organic C and nutrients typically reaches much deeper in terra preta than in the surrounding soils (Fig. 1). In an assessment of about 190 soils, 57% of the profiles had an A horizon that reached to a depth of between 0.2 and 0.6 m, which is significantly deeper than those of adjacent forest soils with about 0.1–0.15 m.^[13] In some cases, the dark coloration reaches to depths of more than 2 m.

Some sites show vast expanse of several hundred hectares, such as the terra preta underlying the city of Santarém. But most documented sites are small with sizes of less than 2 ha.^[13] The challenge in these assessments lies in the identification of what constitutes a terra preta, as the boundaries are ill-defined, and soil changes are often difficult to detect in the field without chemical analyses.

Another factor is that soil changes may occur gradually from terra preta to uninfluenced soils and that such transitional soils may deserve their own category. These so-called “terra mulatta” (Portuguese for “brown earth”) have been described as a distinct soil type^[14] and often surround terra preta. The area of terra mulatta can be expansive and is estimated to exceed the area occupied by terra preta (Fig. 3).

The chemical properties of terra preta and terra mulatta are different. Typically, terra preta soils are darker than terra

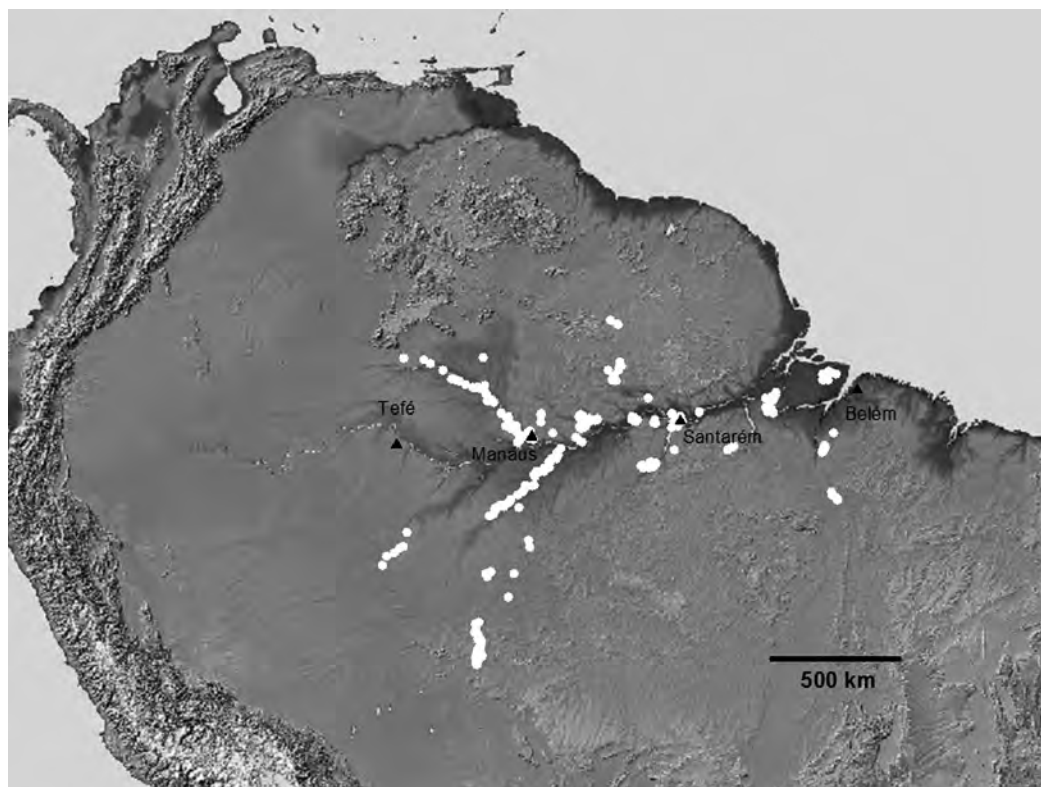


Fig. 2 Distribution of terra preta de índio (white dots) and major cities (black triangles) in the Central and Eastern Amazon region.
Source: From Lehmann, Kern, et al.,^[1] Kern, D'Aquino, et al.,^[13] and background map from NASA Images at www.nasaimages.org.

mulatta, even though organic C contents are often similar.^[15] Terra mulatta have lower nutrient availability, principally of P and Ca, than terra preta,^[15] and do not contain artifacts such as pottery shards. Therefore, such terra mulatta are not middens but may actually constitute agricultural areas^[15] that were generated through charring both during field establishment and on an annual basis using crop residues.

Terra preta and terra mulatta are archaeologically important testaments to the culture of ancient civilizations in the Amazon Basin. They provide information about the way Amerindian populations lived and utilized their environment, sustaining seemingly large and complex societies. Also the mere existence of C-rich and fertile Terra preta de índio several millennia after their creation demonstrates how soils function and suggests the ways for improving soil

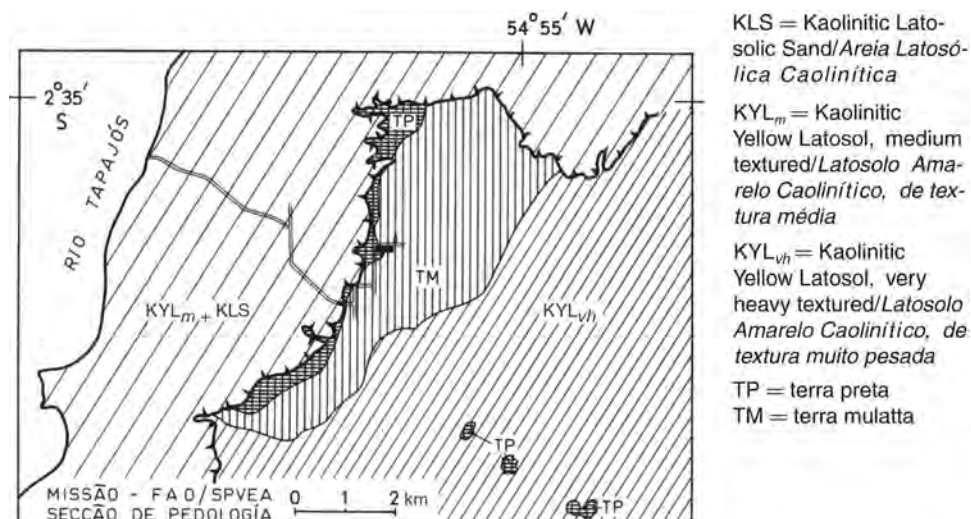


Fig. 3 Distribution of terra preta de índio and terra mulatta at the Belterra Estate on the lower Tapajós River, near Santarém, Brazil.

Source: From Sombroek.^[14]

sustainability. Principally, the discovery of the role of charcoal as a key ingredient in terra preta has led to the development of biochar soil management to improve soil fertility and mitigate global warming through long-term C sequestration.^[16] Such modern biochar management affords the possibility of withdrawing carbon dioxide from the atmosphere while achieving sustainable soil improvement.

REFERENCES

1. Lehmann, J.; Kern, D.C.; Glaser, B.; Woods, W.I. *Amazonian Dark Earths: Origin, Properties, Management*; Kluwer Academic Publishers: Dordrecht, 2003.
2. Hartt, C.F. Preliminary report of the Morgan exhibition, 1870–1871—Report of a reconnaissance of the Lower Tapajos. *Bull. Cornell Univ. (Science)* **1874**, *1*, 1–37.
3. Glaser, B.; Haumaier, L.; Guggenberger, G.; Zech, W. The Terra Preta phenomenon—A model for sustainable agriculture in the humid tropics. *Naturwissenschaften* **2001**, *88*, 37–41.
4. Liang, B.; Lehmann, J.; Solomon, D.; Sohi, S.; Thies, J.E.; Skjemstad, J.O.; Luizão, F.J.; Engelhard, M.H.; Neves, E.G.; Wirick, S. Stability of biomass-derived black carbon in soils. *Geochim. Cosmochim. Ac.* **2008**, *72*, 6069–6078.
5. Liang, B.; Lehmann, J.; Solomon, D.; Kinyangi, J.; Grossman, J.; O'Neill, B.; Skjemstad, J.O.; Thies, J.; Luizão, F.J.; Petersen, J.B.; Neves, E.G. Black carbon increases cation exchange capacity in soils. *Soil Sci. Soc. Am. J.* **2006**, *70*, 1719–1730.
6. Neves, E.G.; Petersen, J.B.; Bartone, R.N.; da Silva, C.A. Historical and socio-cultural origins of Amazonian dark earths. In *Amazonian Dark Earths: Origin, Properties, Management*; Lehmann, J., Kern, D.C., Glaser, B., Woods, W.I., Eds.; Kluwer Academic Publishers: Dordrecht, 2003; 29–50.
7. Kim, J.-S.; Sparovek, G.; Longoc, R.M.; De Melo, W.J.; Crowley, D. Bacterial diversity of terra preta and pristine forest soil from the Western Amazon. *Soil Biol. Biochem.* **2007**, *39*, 684–690.
8. Pietikäinen, J.; Kiikkilä, O.; Fritze, H. Charcoal as a habitat for microbes and its effect on the microbial community of the underlying humus. *Oikos* **2000**, *89*, 231–242.
9. Petersen, J.B.; Neves, E.G.; Heckenberger, M.J. Gift from the past: Terra preta and prehistoric Amerindian occupation in Amazonia. In *Unknown Amazonia*; McEwan, C., Barreto, C., Neves, E.G., Eds.; The British Museum Press: London, 2001; 86–105.
10. Smith, N.J.H. Anthrosols and human carrying capacity in Amazonia. *Ann. Assoc. Am. Geogr.* **1980**, *70*, 553–566.
11. Heckenberger, M.J.; Russell, J.C.; Fausto, C.; Toney, J.R.; Schmidt, M.J.; Pereira, E.; Franchetto, B.; Kuikuro, A. Pre-columbian urbanism, anthropogenic landscapes, and the future of the Amazon. *Science* **2008**, *321*, 1214–1217.
12. Denevan, W.M. Stone vs metal axes: The ambiguity of shifting cultivation in prehistoric Amazonia. *J. Steward Anthropol. Soc.* **1992**, *20*, 153–165.
13. Kern, D.C.; D'Aquino, G.; Rodrigues, T.E.; Frazão, F.J.L.; Sombroek, W.; Myers, T.P.; Neves, E.G. Distribution of Amazonian Dark Earths in the Brazilian Amazon. In *Amazonian Dark Earths: Origin, Properties, Management*; Lehmann, J., Kern, D.C., Glaser, B., Woods, W.I., Eds.; Kluwer Academic Publishers: Dordrecht, 2003; 51–75.
14. Sombroek, W. *Amazon Soils—A Reconnaissance of Soils of the Brazilian Amazon Region*; Agricultural Publications and Documentation: Wageningen, 1966.
15. Woods, W.I.; McCann, J.M. The anthropogenic origin and persistence of Amazonian Dark Earths. *Yearbook Conf. Lat. Am. Geogr.* **1999**, *25*, 7–14.
16. Lehmann, J.; Joseph, S. *Biochar for Environmental Management: Science and Technology*; Earthscan: London, 2009.

Testing

Yash P. Kalra

Northern Forestry Centre, Canadian Forest Service, Edmonton, Alberta, Canada

Jagtar S. Bhatti

Canadian Forest Service, Northern Forestry Centre, Edmonton, Alberta, Canada

Abstract

As an agronomic tool, soil testing represents an index of measurement of nutrient availability. Soil testing, combined with other agronomic information, helps determine the best way to manage nutrients. There is an interest to use soil testing to determine the pollution of surface water and groundwater from fertilizer/waste material application. However, to identify the potential impact of pollutants, a comprehensive approach should be taken on the environment and soil testing. This entry describes the steps in soil testing.

INTRODUCTION

Soil testing is generally defined as any chemical and physical measurement that is made on a soil. The term soil testing may be a rapid chemical analysis to assess the plant-available nutrient status, salinity, and elemental toxicity of a soil or may represent a program that includes interpretations, evaluations, and fertilizer and amendment recommendations based on the results of chemical analysis.^[1,2] As an agronomic tool, soil testing represents an index of measurement of nutrient availability. The objectives of soil testing are as follows: 1) to provide an index of nutrient supply in a soil; 2) to predict the probability of obtaining a profitable response to fertilizers and/or lime; 3) to provide a basis for recommendations on the amount of fertilizer and/or lime to apply; and 4) to evaluate the fertility status of soils by the use of soil test summaries.^[3] Soil testing, combined with other agronomic information, helps determine the best way to manage nutrients. There is an interest to use soil testing to determine the pollution of surface water and groundwater from fertilizer/waste material application. However, to identify the potential impact of pollutants, a comprehensive approach should be taken on the environment and soil testing. Different steps in soil testing follow.

SOIL SAMPLING

Soils are heterogeneous (both horizontally and vertically), and it is, therefore, important that the sample collected for analysis be representative of the field.^[1] Errors in soil sampling fall into the following three general categories: sampling error, selection error, and measurement error. Sampling error is caused by the inherent variation and can be avoided by including the total population in the sample. Selection error arises from any tendency to select some units of the population with greater or lesser probability than was

intended. Measurement error is caused by the failure of the observed measurement to be true to value for the unit. It includes both the random errors of measurement and biases.

The most critical aspect of soil testing is obtaining a soil sample that is representative of the site.^[3] Statistical concepts determine which of the following three commonly used sampling strategies are used: 1) simple random sampling; 2) stratified random sampling; and 3) systematic or grid sampling. Other important factors to consider are the depth of sampling, degree of field variability, number of cores needed to constitute a single composite sample, tools used to collect soil sample cores, type of containers, frequency of sampling, and the effect of season on sample collection.^[4]

PREPARATION OF SOIL SAMPLES FOR LABORATORY ANALYSIS

Sample preparation is critical in obtaining accurate results.^[4] The field-moist soil samples must be dried promptly to minimize microbial activity. The drying temperature can have a significant effect on the physiochemical properties of soils. Generally, the soils are air-dried. If an air-circulating oven is used to speed up the process, temperature should not exceed 38°C. The soils are ground to 2 mm size for routine analysis. For certain analyses, the samples need to be ground further to 1 mm size or finer.

APPARATUS FOR SOIL ANALYSIS

A soil laboratory would contain the following apparatus: soil grinder, sieves, auto-dispensers, funnel racks, titrator, time-controlled stirrer, time-controlled shaker, and balances. It would also have ovens, muffle furnace, pH meter, conductivity meter, sulfur analyzer, nitrogen analyzer, continuous flow analyzer (air-segmented and non-segmented

stream instruments), ultraviolet-visible spectrophotometer, ion chromatograph, atomic absorption spectrophotometer, flame emission spectrophotometer, and X-ray fluorescence spectrophotometer. Finally, the laboratory would have a potentiometric instrument; inductively coupled plasma atomic emission spectrophotometer; and glassware, plasticware, and other apparatus generally considered common to soil analysis laboratories. Computers, integral parts of modern laboratory instruments, are used for setting instrument parameters, controlling the operation of the instruments, and data capture and storage.

METHODS OF SOIL TESTING

Chemical soil tests are designed to estimate the amounts of plant nutrients available for plant growth. Different reagents are being used to extract different nutrients as listed in Table 1 and analyzed using the instruments given in the previous section. Apart from nutrient contents, loss on ignition or a potassium dichromate wet oxidation procedure routinely determines organic matter. Soil texture is determined by sedimentation procedure in the laboratory and hand texturing method in the field.^[4] Soil pH is determined in a soil–water slurry of 1:1 or 1:2 or in a soil slurry

of 0.01 M CaCl₂ or 1 M KCl. Exchangeable acidity is determined by using the barium chloride–triethanolamine (TEA) extractant buffered to pH 8.2. The Adams–Evans buffer^[5] and the Shoemaker, Mclean, and Pratt buffer^[6] methods are used to determine the lime requirements of coarse-textured and fine-textured soils, respectively.

Fixed extraction ratios (e.g., 1:2 and 1:5) and saturation extract methods are used for the measurement of electrical conductivity. However, an estimation of the soluble salt content of soils is carried out using a 1:2 soil/water ratio. The sodium adsorption ratio is an useful index of soil sodicity based on the ratio of basic cations, which can be calculated from chemical analysis of the soil extract.^[4]

Soil structure, air porosity, water desorption characteristics, bulk density, saturated hydraulic conductivity, and infiltration rate are measured *in situ* and in laboratory.^[4]

CORRELATION AND CALIBRATION

Correlation is the process of determining whether there is a relationship between the soil test results and the plant uptake of a nutrient or yield. A soil test for a given nutrient has historically involved three steps: 1) selecting an extractant; 2) correlating the amount of nutrient extracted with the amount taken up by the plant; and 3) calibrating the test values with crop yields. Nutrients extracted in soil testing procedures are frequently referred to as “available nutrients.” The Glossary of Soil Science Terms^[7] defines available nutrients as the amount of soil nutrient in chemical forms accessible to plant roots or compounds likely to be convertible to such forms during the growing season. For the purpose of soil testing correlation, the rate at which a nutrient is taken up by the plant becomes more important than the amount of the available nutrient. Thus, the correlation of estimated nutrient uptake based on soil testing and other pertinent factors with total uptake by plant seems to be a logical approach.^[8] This can be accomplished by greenhouse experiments and field trials, along with the use of empirical and mechanistic modeling approaches.

Calibration is the process of determining the probability of getting a growth response to applied nutrients. Calibration of a soil test is basic to a good soil testing program. Soil test extractants were designed to rapidly and accurately assess the available nutrient status and/or elemental toxicity in soils. They were also designed to provide a quantitative basis for recommendations of the rates of plant nutrients that should be added as fertilizers, manures, or other materials to economically achieve optimum yield. A properly calibrated soil test should provide information on the degree of deficiency or sufficiency of an element and identify how much of the element should be applied if it is deficient. There are additional criteria of a successful soil test. It should be easy to perform to ensure rapid turnaround time, cost-effective to promote wide usage, and reproducible.

Table 1 Commonly used extractants for plant nutrients.

Soil nutrient	Extractant
NO ₃ -N	2.0 M KCl
NH ₄ -N	2.0 M KCl
P	0.03 M NH ₄ F + 0.025 M HCl (Bray-P1) 0.05 M HCl + 0.0125 M H ₂ SO ₄ (Mehlich-1) 0.5 M NaHCO ₃ , pH 8.5 (Olsen-P) 0.2 M Acetic acid (CH ₃ COOH) + 0.25 M NH ₄ NO ₃ + 0.015 M NH ₄ F + 0.013 M HNO ₃ + 0.001 M EDTA (ethylene diamine tetraacetic acid; Mehlich 3) 0.25 M CH ₃ COOH + 0.15 M NH ₄ F (Kelowna)
Cation exchange capacity (Ca, Mg, K, and Na)	1.0 M ammonium acetate (NH ₄ C ₂ H ₃ O ₂), pH 7.0
S	Ca (H ₂ PO ₄) ₂ , containing 500 mg/L P 0.01 M CaCl ₂ 0.25 M CH ₃ COOH + 0.15 M NH ₄ F (Kelowna)
Fe, Mn, Cu, and Zn	0.005 M DTPA (diethylene triamine pentaacetic acid) + 0.01 M CaCl ₂ , and 0.1 M TEA (HOCH ₂ CH ₂) ₃ N adjusted to pH 7.30 ± 0.05 with 1:1 HCl
B	Hot water
Cl	Water

Source: From Carter.^[4]

INTERPRETATION AND RECOMMENDATIONS

Soil testing interpretation is the process by which one assesses the fertility status of a field based upon a set of known factors. It involves an evaluation of chemical test results in terms of basic soil–plant relationship phenomena. More complex interpretation may include consideration of factors such as the crop to be grown, yield goal, individual soil characteristics, climate, tillage system, and environmental quality.^[9] Many fertilizer recommendations are made using interpretation of both the concepts and designs to fertilize the soil rather than the plants. Recommendations based on basic cation saturation ratio are designed to alter the existing ratio to achieve the ideal K/Ca/Mg ratio, regardless of the existing and potential yield levels. The recommendations are based on soil test results and other agronomic conditions.^[9]

QUALITY CONTROL PROCEDURES FOR SOIL ANALYSIS

The quality of soil test results is evaluated on the basis of the following two components: bias and precision. Bias refers to the deviation of analytical result from the true value and measures accuracy of the result. Precision refers to the reproducibility of a given test value. A primary means to evaluating soil test quality procedures, as well as performance, is through a comparison of laboratories performing the same test procedure on the same soil. The use of appropriate reference soils to assess, monitor, maintain, and ensure the accuracy of analytical results should be part of a soil testing procedure. Incorporation of reference materials is one cost-effective facet of a data quality program to ensure the accuracy of analytical results.^[4]

It is vital that laboratories monitor and document the quality of their analytical results on a regular basis. For this purpose, many laboratories participate in proficiency testing programs. The Soil and Plant Analysis Council, Inc. (formerly the Council on Soil Testing and Plant Analysis) was founded in 1969 to address the issues of uniformity in the analytical procedures and fertilizer recommendations.^[10] The AOAC International is an association of scientists and organizations devoted to promoting methods validation and quality measurements in the analytical sciences. In 1990, the Soil Science Society of America and the AOAC International established a joint committee to conduct validation studies for methods of soil analysis. The first validation was for soil pH methods.^[11]

FUTURE OF SOIL TESTING

The future of soil testing is bright. There will be greater automation in the laboratory and wider use of multielement extractants. The search for better methods to evaluate bio-availability of elements in soils will continue to be an important concern and will affect both the agronomic and analytical aspects of soil testing. No unusually drastic change is anticipated concerning laboratory procedures, but there is an expected change in the future use of soil tests, as less of a diagnostic procedure and more of a monitoring one.

ACKNOWLEDGMENT

The authors are grateful to Drs. J.D. Beaton, J.B. Jones, Jr., T.D. Peck, and B. van Raij for their help in the preparation of the entry.

REFERENCES

1. Westerman, R.L. *Soil Testing and Plant Analysis*, 3rd Ed.; SSSA Book Series Number 3; Soil Science Society of America: Madison, 1990; 1–784.
2. Walsh, L.M.; Beaton, J.D. *Soil Testing and Plant Analysis*; Soil Science Society of America: Madison, 1973; 1–491.
3. Havlin, J.L.; Beaton, J.D.; Tisdale, S.L.; Nelson, W.L. *Soil Fertility and Fertilizers: An Introduction to Nutrient Management*; Prentice Hall: Upper Saddle River, 1999; 1–499.
4. Carter, M.R. *Soil Sampling and Methods of Analysis*; CRC Press: Boca Raton, 1993; 1–823.
5. Adams, F.; Evans, C.E. A rapid method for measuring the lime requirement of red–yellow podzolic soils. *Soil Sci. Soc. Am. Proc.* **1962**, 26, 355–357.
6. Shoemaker, H.E.; Mclean, E.O.; Pratt, P.F. Buffer methods for determining the lime requirement of soils with appreciable amounts of extractable aluminum. *Soil Sci. Soc. Am. Proc.* **1961**, 25, 274–277.
7. SSSA. *Glossary of Soil Science Society of America Terms*; Soil Science Society of America: Madison, 1996; 1–134.
8. Jones, J.B., Jr. Soil test methods: Past, present, and future. *Commun. Soil Sci. Plant Anal.* **1998**, 29, 1543–1552.
9. Brown, J.R. *Soil Testing: Sampling, Correlation, Calibration, and Interpretation*; SSSA Special Publication Number 21; Soil Science Society of America: Madison, 1987; 1–244.
10. Jones, J.B., Jr.; Kalra, Y.P. Soil testing and plant analysis activities—The United States and Canada. *Commun. Soil Sci. Plant Anal.* **1992**, 23, 2015–2027.
11. Kalra, Y.P. Determination of pH of soils by different methods: Collaborative study. *JAOAC Int.* **1995**, 78, 310–324.

Texture

Harold R. Geering

*Department of Agricultural Chemistry, University of Sydney, Sydney,
New South Wales, Australia*

Hwat Bing So

*School of Agriculture and Food Sciences, University of Queensland, St. Lucia,
Queensland, Australia*

Abstract

Soil texture is the consistency of a moist soil felt between the fingers and is associated with its particle size distribution. Texture is a fundamental property that affects many of the physical and chemical properties and behavior of the soil, such as the soil water and nutrient holding capacities, hydraulic conductivity, and resistance to cultivation. Soil texture is affected by clay content and type, silt and organic matter contents of the soil as well as the ion composition on the surfaces of the clay fraction.

INTRODUCTION

Besides color, one of the principal descriptors of a mineral soil (i.e., one with less than 10% organic matter content) is its texture, which is well correlated with the particle size distribution of the fine earth fraction (<2 mm fraction). The distribution of particle size is conveniently partitioned by particle size analysis (PSA) into subfractions of sand, silt, and clay, the proportions of which will have a dominant influence on many of the practical soil properties important to agriculture, the environment, and engineering purposes, such as ease of cultivation, nutrient and water holding capacities, transmission characteristics, earth dam construction, and susceptibility to erosion. However, with respect to particle size ranges, the subdivision into sand, silt, and clay fractions varies with country and professional institution.^[1] Examples of four of these particle size classifications are shown in Fig. 1.^[2] In defining soil texture, the first three systems shown in Fig. 1 are most widely used in textural class designation.

For describing a soil and/or soil horizon portion of a soil profile, the soil is allocated to a textural class. The textural class descriptors provide an indication of which particle size fraction is dominant, with the exception of a loam where no single size range exerts a dominant influence. In the field, the textural class is estimated by the hand feel of a moist sample of fine earth fraction, when molded between the thumb and forefingers at just below the “sticky point.” In this moisture condition, the soil just begins to stick to the fingers and its moisture content is approximately equivalent to field capacity or the amount of water retained at a water

potential of -33 kPa.^[3] Because the process of hand molding destroys the structure of the soil, the estimate of the textural class will also be related to the particle size distribution. Initially, the soil is remolded several times to form a uniform moist bolus of about the size of a golf ball. Then, the bolus is deformed into threads (see Fig. 2) or ribbons of approximately 3 mm thick (see Fig. 3). The force necessary to deform and shear the bolus and the length of these threads or ribbons increase with clay content. At the same time, if the sample contains a particular size fraction that is readily identifiable by feel to the fingers, this is taken into account in deciding on the soil texture class, e.g., the grittiness of coarse sand or the slippery silky feel of silt. Alternatively, medium and coarse sand can also be seen with a hand lens, and fine sand can be heard by its crunchy grinding sound when remolding and shearing are done close to the ears. Finally, organic matter when present as humus at >10% usually confers cohesion to sandy textures and greasiness to clayey textures^[6] or tends to make sandy and clayey soils feel more loamy in texture as described by Clarke.^[7]

Taking all of the above-mentioned description and criteria into account, soil texturing by hand is essentially a consistency test (see the entry *Consistence*, p. 471) at the sticky point.^[8] It then follows that since the amount of work and cohesion increase in the order of the three principal texture class of sand, loam, and clay, they are often well correlated with the texture classes derived from PSA.^[1,9] Because additional criteria are used in describing the texture of a soil by hand, other than just particle size distribution per se, there are usually more

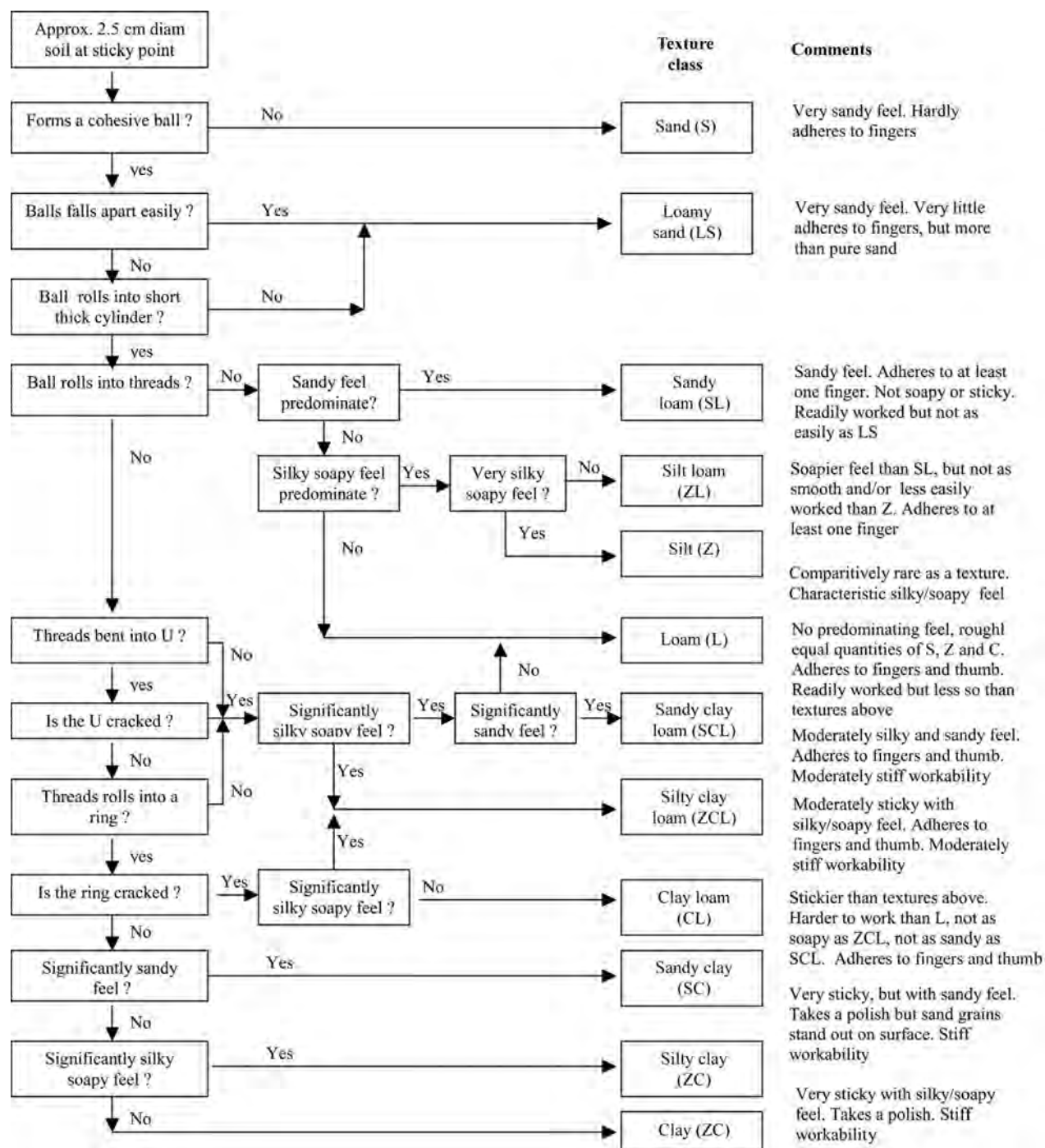


Fig. 2 A guide to field assessment of texture for mineral soils in the United Kingdom by S. Northcliff, Reading University, and J. R. Landon, Booker Agricultural International.

Source: From Rowell.^[4]

(see the entry *Surface Area: Specific*, p. 2255). Smaller particles possess larger SSA, e.g., SSA for sand is approximately $0.01\text{--}0.1\text{ m}^2\text{ g}^{-1}$, silt approximately $1\text{ m}^2\text{ g}^{-1}$, and clay may range from 10 to $800\text{ m}^2\text{ g}^{-1}$. As most soils contain a considerable amount of clay, it is characterized by large SSA. All physical and chemical processes in the

soil occur at the interface of the soil solids; hence, the larger the SSA, the greater will be the cumulative effect of these processes, and hence the clay fraction plays the major role in this regard. Clays are referred to as the active fraction of the soil, whereas sand and silt are referred to as the skeleton of the soil.

Field texture groups	Ribbon length (mm)	Coherence of the bolus at sticky point	Feel	Other features	Texture grade (synonymous with texture class)	Approx clay %
1 The Sands	Nil	Nil	Sandy	Single sand grains adhere to fingers	1. Sand (S)	Commonly <5
	5	Slight	Sandy	Discolors fingers with an organic stain	2. Loamy Sand (LS)	5–10
	5–15	Slight	Sticky	Sand grains sticks to fingers and discolors with a clay stain	3. Clayey Sand (CS)	5–10
2 The Sandy Loams	15–25	Just coherent	Very sandy	Medium Sand readily visible	4. Sandy Loam (SL)	10–20
	15–25	Just coherent	Very sandy	Fine sand may be heard	5. Fine Sandy Loam (FSL)	10–20
	20–25	Strong	Sandy	Medium Sand easily visible	6. Light Sandy Clay Loam (SCL)	15–20
3 The Loams	About 25	Coherent	Spongy and greasy	No obvious sandiness	7. Loam (L)	25
	About 25	Coherent	Slightly Spongy	Fine sand	8. Loam Fine Sandy (Lfsy)	25
	About 25	Coherent	Smooth	Silky; very smooth when manipulated	9. Silt Loam (SiL)	25 (>25% silt)
	25–40	Strong	sandy	Medium Sand in fine matrix	10. Sandy Clay Loam (SCL)	20–30
4 The Clay Loams	40–50	Strong	Smooth	No obvious sand grains	11. Clay Loam (CL)	30–35
	40–50	Coherent	Smooth	Silky feeling	12. Silty Clay Loam (SiCL)	30–35 (>25% silt)
	40–50	Coherent	Smooth and sandy	Fine sand can be felt and heard	13. Fine Sandy Clay Loam (FSCL)	30–35
5 The Light Clays	50–75	Coherent	Plastic	Fine to medium sand	14. Sandy Clay (SC)	35–40
	50–75	Coherent	Plastic	Smooth and silky	15. Silty Clay (SiC)	35–40 (>25% silt)
	50–75	Coherent	Plastic	Smooth with slight resistance to shearing	16. Light Clay (LC)	35–40
	> 75	Coherent	Plastic	Smooth with a little resistance to shearing	17. Light Medium Clay (LMC)	40–45
6 The Medium & Heavy Clays	> 75	Coherent	Plastic	Fair resistance to shearing	18. Medium Clay (MC)	45–55
	> 75	Coherent	Plastic	Firm resistance to shearing	19. Heavy Clay (HC)	>50

Fig. 3 A summary guide for assessment of soil texture for mineral soils in Australia, silt fraction is 0.02–0.002 mm.

Source: From McDonald, Isbell, et al.^[5]

Ease of Cultivation

The ease of cultivation for soils of different textures gives rise to the expressions of “light-textured soils” and “heavy-textured soils” being used for sandy and clayey soils, respectively. Coarse-grained, sandy soils tend to be loose, well aerated, and easy or light to cultivate. Fine-textured soils tend to absorb much water and may become plastic and sticky when wet. Hence, they require much more energy and are heavy to cultivate. Upon drying, they tend to become dense and cohesive.^[14]

Nutrient and Water Holding Capacity and Transmission Properties

The SSA and the magnitude of the electrical charges per unit area of clay and silt determine the cation exchange capacity (CEC) of the soil, which is the number of cations that a unit mass of soil can hold (see the entry *Ion Exchange*, p. 1241). The larger the SSA, the larger is the soil’s nutrient holding capacity, e.g., CEC for kaolinite, illite, and montmorillonite clays range from 3 to 15 mmol kg⁻¹, 10 to 40 mmol kg⁻¹, and 60 to 120 mmol kg⁻¹, respectively. However, fine particle size distribution is generally

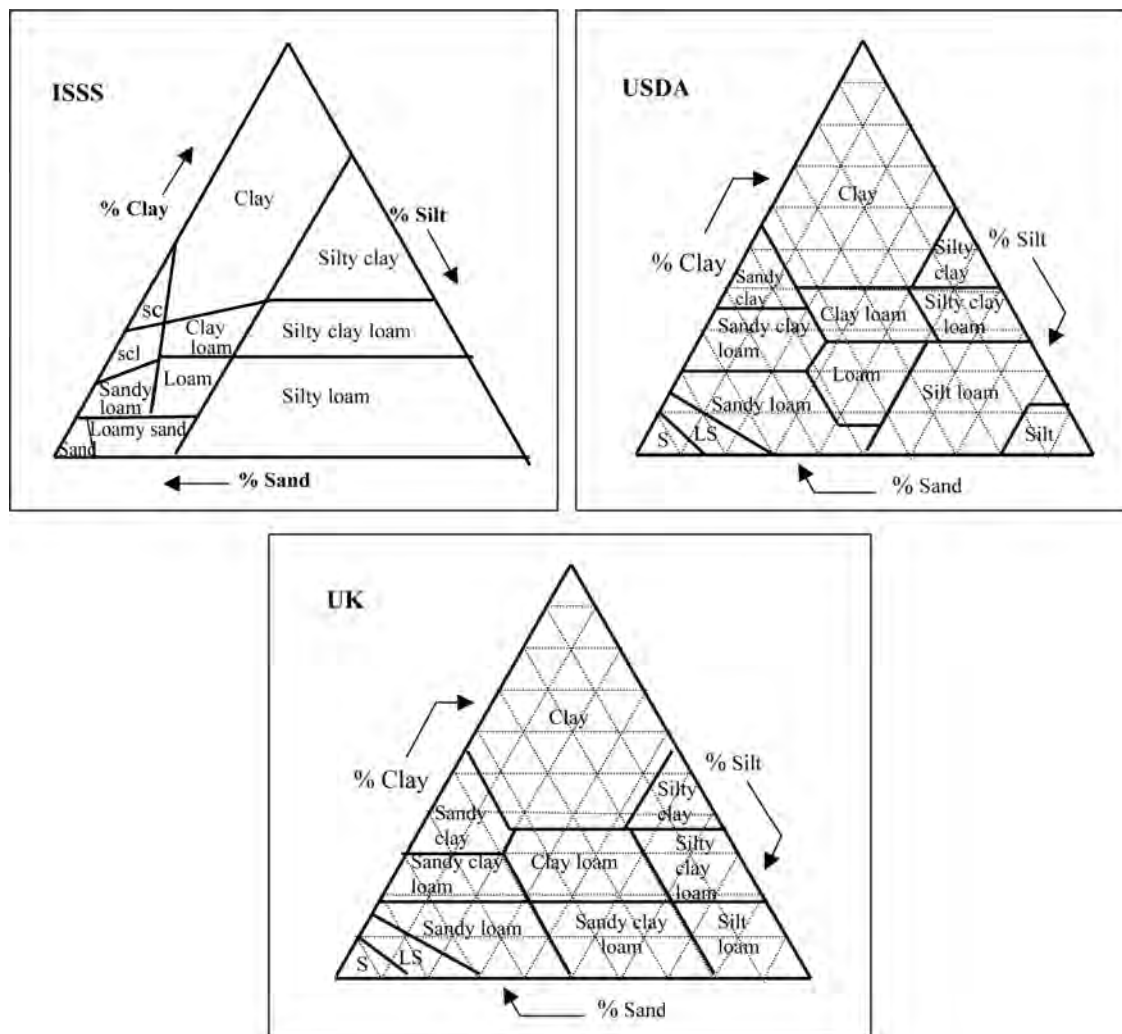


Fig. 4 Three of the most frequently used texture triangles. The ISSS is based on particle size of sand (2–0.02 mm), silt (0.02–0.002 mm), and clay (<0.002 mm). The USDA triangle is based on silt fractions of 0.05–0.002 mm and the U.K. triangle on silt fractions of 0.06–0.002 mm.

Source: From Rowell,^[4] McDonald, Isbell, et al.,^[5] Gee & Bauder,^[12] and Soil Survey Staff.^[13]

associated with lower infiltration and drainage rates. Therefore, sandy soils tend to have higher infiltration rates and generally experienced greater leaching, resulting in acid pH and low base saturation. On the other hand, clayey soils tend to have lower infiltration rates and less leaching, resulting in neutral to alkaline pH and high base saturation.

Particle size distribution and SSA are important determinants of the soil's water holding capacity. Water is adsorbed on soil surfaces, and water will accumulate in the pore space between the particles as capillary water, which is held in the soil by the force exerted by the water menisci across the pores between particles. This force is inversely proportional to the size of the pore; hence, water is readily removed or drained from the larger pores. At field capacity, which is the water content of an undisturbed soil at a water potential of –10 kPa, sandy soils tend to have lower water

contents than clayey soils. Similarly, at the wilting point, which is the water content where water is adsorbed onto soil surfaces by a force of –1.5 MPa, sandy soils have lower water contents than clayey soils. However, the plant available water capacity (PAWC), or the amount of water held between field capacity and wilting point, is also lower in sandy soils. The approximate relationship between PAWC and texture is shown in Fig. 5.^[15] Since water at wilting point is adsorbed on the surface of soil particles, it is predominantly affected by the particle size distribution or texture class. However, water at field capacity is adsorbed and is capillary water. The latter is strongly affected by soil structure; this effect increases with surface area and clay content. Where the soil is poorly structured or finely structured, field capacity tends to be highly associated with the predominance of fine pores such as the black earths (curve A in Fig. 5), whereas strongly structured soils have a

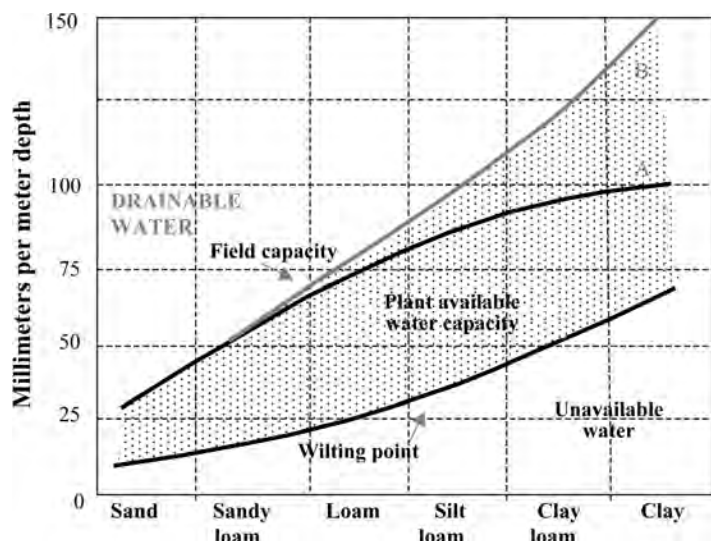


Fig. 5 Approximate water holding capacities of different textured soils.

Source: From Foth.^[15]

predominance of large pores and field capacity tends to be lower for the same clay content such as the red Latosols (curve B in Fig. 5). Particle size distribution will affect the pore size distribution of a soil, and consequently, it will influence the soil's transmission characteristic, which is termed as the soil's hydraulic conductivity (see the entry *Hydraulic Conductivity*, p. 1133).

Packing of Particles and Hard Setting Characteristic of Soils

Particle size distribution also affects the potential packing density of the soil or soil aggregates.^[16–18] The presence of sufficient quantities of small particles that will fill the space between the larger particles will result in a massive structureless soil. This generally occurs on soils with 30–35% sand content. One specific soil characteristic associated with the presence of silt and fine sand particles (silty or fine sandy-textured soils) is hard setting, which is a soil characteristic where the aggregates readily break down upon wetting. The structure will not be reformed, and the soil dries out rapidly into a hard consistency. It softens considerably upon rewetting.^[19] Hard setting soils are characterized by low water holding capacities and they alternate rapidly between soft and hard consistencies, often resulting in poor soil physical conditions for most plants.

CONCLUSION

Soil texture is the consistency of a moist soil felt between the fingers and is associated with its particle size distribution. It is affected by clay content and type, silt and organic matter contents of the soil as well as the ion composition on the clay fraction. Texture is a fundamental property that determines the SSA of the soil, which in turn affects many of the physical and chemical properties and behavior of the

soil, such as the soil water and nutrient holding capacities, hydraulic conductivity, and resistance to cultivation.

REFERENCES

1. Hodgson, J.M. *Soil Sampling and Soil Description*; Oxford University Press: Oxford, 1978.
2. Brady, N.C. *The Nature and Properties of Soils*, 8th Ed.; Macmillan Pub. Co. Inc.: New York, 1974.
3. Cassel, D.K.; Nielson, D.R. Field capacity and available water capacity. In *Methods of Soil Analysis. Part 1. Physical and Mineralogical Methods*; Klute, A., Ed.; American Society of Agronomy Inc. and Soil Science Society of America Inc.: Madison, 1986; Chapter 36, 901–926.
4. Rowell, D.L. *Soil Science, Methods and Applications*; Longman Scientific & Technical, UK and John Wiley & Sons Inc.: New York, 1994; Chapters 1 and 2.
5. McDonald, R.C.; Isbell, R.F.; Speight, J.G.; Walker, P.J.; Hopkins, M.S. *Australian Soil and Field Handbook*, 2nd Ed.; Inkata Press: Melbourne, 1990; 115–123.
6. Northcote, K.H. *A Factual Key for the Recognition of Australian Soils*; CSIRO, Rellim Technical Publications: Adelaide, 1979.
7. Clark, G.R. *The Study of Soil in the Field*; Clarendon Press: Oxford, 1971; 46 pp.
8. Collis-George, N. The mechanical and structural properties of soil. In *The Fundamentals of Modern Agriculture*; Blake, C.D., Ed.; The University of Sydney Press: Sydney, 1967; Chapter 3, 61–88.
9. Hodgson, J.M. *Soil Survey Field Handbook*, Technical Monograph No. 5; Soil Survey of England and Wales: Harpenden, 1974.
10. Klute, A., Ed. *Methods of Soil Analysis. Part 1. Physical and Mineralogical Methods*; American Society of Agronomy Inc. and Soil Science Society of America Inc.: Madison, 1986.
11. Loveland, P.J.; Whalley, R.W. Particle size analysis. In *Soil Analysis, Physical Methods*; Smith, K.A., Mullins, C.E., Eds.; Marcel Dekker Inc.: New York, Basel, Hong Kong, 1991; 271–328.

12. Gee, G.W.; Bauder, J.W. Particle-size analysis. In *Methods of Soil Analysis. Part 1. Physical and Mineralogical Methods*; Klute, A., Ed.; American Society of Agronomy Inc. and Soil Science Society of America Inc.: Madison, 1986; 383–412.
13. Soil Survey Staff. *Soil Taxonomy, a Basic System of Soil Classification for Making and Interpreting Soil Surveys*; USDA Soil Conservation Service, R.E. Krieger Pub. Co.: Malabar, 1988.
14. Hillel, D. *Introduction to Soil Physics*; Academic Press, Inc.: New York, 1982.
15. Foth, H.D. *Fundamentals of Soil Science*, 6th Ed.; John Wiley & Sons: New York, 1978.
16. Bodman, G.B.; Constantin, G.K. Influence of particle size distribution in soil compaction. *Hilgardia* **1965**, *36*, 567–591.
17. Gupta, S.C.; Larson, W.E. A model for predicting packing density of soils using particle size distribution. *Soil Sci. Soc. Am. Proc.* **1979**, *19*, 758–764.
18. Smith, G.D.; Coughlan, K.J.; Fox, W.E. The role of texture in soil structure. In *Modifications of Soil Structure*; Emerson, W.W., Bond, R.D., Dexter, A.R., Eds.; John Wiley & Sons: New York, 1978; 79–86.
19. So, H.B.; Smith, G.D.; Raine, S.R.; Schafer, B.M.; Loch, R.J. *Sealing, Crusting and Hardsetting Soils: Productivity and Conservation*; Queensland Branch of the Australian Society of Soil Science Inc.: Brisbane, 1995; 527 pp.

Tillage Erosion: Description and Process

Michael J. Lindstrom

USDA-ARS-MWA, North Central Soil Conservation Research Laboratory, Morris, Minnesota, U.S.A.

Abstract

Tillage erosion is directly proportional to the degree and scale of topographic complexity. Soil conservation measures that do not include a reduction in tillage erosion will not be effective in controlling soil loss on upper slope landscape positions of cultivated agricultural lands. To reduce soil loss caused by tillage erosion, frequency, tillage intensity (speed and depth), and the size of tillage implements must be reduced.

INTRODUCTION

Tillage erosion is a problem that has been present since the dawn of cultivation. The problem has intensified with increased tillage speed, depth, and size of tillage tools and with the tillage of steeper and more undulating lands. Evidence of tillage erosion is commonly observed as a difference in soil color between hilltops and adjacent lower slope positions. Tillage erosion is defined by the Soil Science Society of America as the downslope displacement of soil through the action of tillage. It is easy to visualize that forward soil movement will be less when tillage operations are conducted in the upslope direction than when conducted in the downslope direction (Fig. 1). This difference in soil translocation distance is a function of gravity. Assuming that the tillage direction often occurs equally in the upslope and downslope directions, then a net downslope displacement of soil will take place. However, it is not an easy matter to move from this simple concept to one that suggests that the soil loss from hilltops in undulating landscapes because of soil translocation by tillage can exceed levels that would be considered sustainable for crop production.

Tillage erosion has often been described in qualitative rather than quantitative terms. Evidence of the mass downslope movement of soil by tillage has been present for years. One example frequently cited comes from the Palouse region of the Pacific Northwest of the United States^[1] where soil banks, 3–4 m high, have developed at fenceline locations on steep side slope. These soil banks are the result of moldboard plowing, where the tillage above the fenceline turned the furrow slice toward the fenceline, and tillage below the fenceline turned the furrow slice away from the fenceline. In studies designed to measure the effect of soil variability across landscapes on crop production potentials, tillage has been implicated as the cause for observed downslope soil movement and an increased variability of soil properties.^[2,3]

Examination of stereoscopic aerial photographs taken in 1947 and 1991 in the Loam Belt of Belgium showed a severe surface lowering on the top of the hillslopes and on hillslope convexities. Deposition occurred on the lowermost parts of the hillslope in hillslope concavities and in topographic-defined convergence lines. The observed pattern differed markedly from that expected from water erosion processes, indicating that soil redistribution was dominated by tillage operations.^[4]

DETERMINATION OF TILLAGE EROSION

A simple linear regression of the form $Y = a + b(S)$ has been developed,^[5,6] which describes the relationship between slope gradient (S) and mean soil translocation distance (Y) in the direction of tillage. Slope gradients were considered positive when tilling upslope and negative when tilling downslope. Expanding on this relationship, it has been proposed^[6] that tillage translocation could be considered a diffusion-type geomorphological process, similar to rain splash and soil creep, and characterized by a single constant, the tillage transport coefficient (k).

$$k = -D\rho_b B \quad (1)$$

where D is the depth of tillage (m), ρ_b is the soil bulk density (kg/m^3), and B is the slope of the linear regression equation of the relationship between soil displacement (m) and slope gradient (m/m). Using this relationship, the unit soil transport rate in the direction of tillage (Q_s) at any specific point in a field can be calculated as:

$$Q_s = kS \quad (2)$$

where S is the slope gradient (m/m). Representative tillage transport coefficients (k values) for moldboard plow tillage have ranged between 230 kg/m and 330 kg/m .^[6] Agricultural fields commonly undergo a series of tillage operation resulting in k values of 400–600 kg/m .

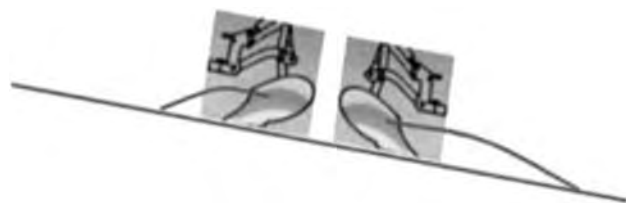


Fig. 1 Relative soil displacement distances when the thrust is upslope vs. downslope.

It is not possible to directly calculate soil erosion using Q_s , because this value essentially represents a soil flux rate at a cross section for a specific tillage operation or series of operations. However, soil loss or gain will result when, for an elementary slope segment of unit width, the incoming flux is different from the outgoing flux:

$$E = (Q_{s, \text{in}} - Q_{s, \text{out}})/X \quad (3)$$

where E is the tillage erosion rate (kg/m^2) and X is the length (m) of the elementary slope segment under consideration. Because Q_s is directly proportional to the slope gradient, soil loss or gain will be proportional to the change in slope gradient. Soil translocation by tillage will result in soil loss in convex slope positions such as crests and shoulder slopes because there is an increase in slope gradient and thus an increase in soil transport rate. Conversely, soil deposition will take place in concave slope positions in the foot- and toe-slope positions. When slope gradients between adjacent elemental slope segments are equal, irrespective of their magnitude, no net soil loss or gain takes place because the $Q_{s, \text{in}}$ equals $Q_{s, \text{out}}$. Thus, in backslope positions where slope gradients are commonly the greatest, exhibiting the greatest soil transport rate, net soil loss or gain will be minimal provided slope gradients remain constant. Therefore, the rate of soil gain or loss will depend on the unit transport rate and the degree of change in slope gradients:

$$E = \Delta Q_s / X \quad (4)$$

The magnitude of soil erosion rates by tillage vs. water is affected by many variables, i.e., topography, rainfall intensity, tillage intensity (depth and frequency), and land use. Examination of the relationship between a range of topographic parameters and ^{137}Cs -derived erosion rates, from fields in the United Kingdom,^[7] showed that the highest correlation was between erosion rate and landscape curvature at four of the five sites investigated. These results were not consistent with the dominance of water erosion, where slope angle and upslope lengths, or areas, are the primary influences. In a study comparing the roles of tillage and water erosion in landform development on agricultural land in Belgium,^[8] it was suggested that if water erosion were the dominant process, the landscape would be characterized by increased incision of the

concavities and convergent waterways. A gradual increase in slope angles on upland convex slopes was also noted. In contrast, tillage produces maximum erosion on convex slopes, leading to reduced slope angles and infilling of concavities and hollows. The pattern of landform development observed was an infilling of the slope concavities and convergent waterways by sediment displaced through tillage that then more compensated for the lower frequency, but more visible, rill and gully incision. The pattern indicates that, despite high susceptibility of the area to water erosion, landform development in this agricultural landscape is dominated by tillage erosion processes. These processes result in a reversal of the expected landscape evolution, with a gradual obliteration of topographic features. Other studies have indicated that tillage erosion rates are of the same order of magnitude as water erosion.^[9,10]

EFFECTS OF TILLAGE EROSION

The tillage transport coefficient (k value) is a measure of the mean distance a mass of soil per unit width is moved by tillage, in a specified direction relative to the direction of tillage. The soil mass is translocated in the forward direction (parallel to the direction of tillage) but is also translocated in the lateral direction (perpendicular to the direction of tillage). Determination of k values has mostly been in the forward direction. However, using the mean displacement distances does not fully describe soil translocation. To illustrate, a single pass with a chisel plow may move 70 kg of soil forward per meter width of tillage. The mean forward displacement of this 70 kg of soil may be 40 cm, but significant quantities of soil may be moved as little as 5 cm or as much as 300 cm. Soil displacement will vary across the width of a tillage implement because of the spacing and arrangement of the individual tillage tools. This variation in distance over which soil is translocated is important because it affects the distance that soil constituents (amendments and contaminants) are dispersed or mixed by tillage.

The rate of soil loss by tillage erosion within topographically complex landscapes is several times more than is considered sustainable for crop production. Soil loss in a convex slope position in the Ontario Province of Canada was estimated to be 54 t/ha/yr.^[11] Estimates made using resident ^{137}Cs indicate that between 70% and 100% of soil lost in convex slope positions is the direct result of tillage erosion.^[12] Crop yield reductions of 40–50% have been associated with these eroded landscape positions throughout southwestern Ontario.^[13]

Although tillage erosion can result in considerable soil loss and accumulation within fields, soil is not directly lost from fields by tillage erosion. However, tillage erosion exposes subsoil material on upper slope positions, which may become more susceptible to wind and water erosion. Furthermore, the redistribution of soil by tillage erosion

delivers topsoil to areas of concentrated overland water flow on both the microtopographic scale (i.e., rills) and the macrotopographic scale (i.e., convergent landforms).^[11,14] As such, tillage erosion acts as a delivery mechanism of soil, which is then subject to water erosion.

Soil translocation by tillage produces maximum erosion at abrupt convex slope positions, causing a reduction in slope angles, and an infilling of hollows, resulting, over time, in a gradual obliteration of topographic features. As tillage erosion proceeds, the erosion process occurs over an increasingly larger area. In contrast, when water erosion is the dominant process, the landscape is characterized by increased incision of concavities and ephemeral gullies. A gradual increase in slope angle on convex slope positions also occurs.

CONCLUSION

Tillage erosion is directly proportional to the degree and scale of topographic complexity. Soil conservation measures that do not include a reduction in tillage erosion will not be effective in controlling soil loss on upper slope landscape positions of cultivated agricultural lands. To reduce soil loss caused by tillage erosion, frequency, tillage intensity (speed and depth), and the size of tillage implements must be reduced.

REFERENCES

- Papandick, R.I.; Miller, D.E. Conservation tillage in the Pacific Northwest. *J. Soil Water Conserv.* **1977**, *32*, 40–56.
- Kachanoski, R.G.; Rolston, D.E.; deJong, E. Spatial variability of a cultivated soil as affected by past and present microtopography. *Soil Sci. Soc. Am. J.* **1985**, *49*, 1082–1087.
- Cao, Y.Z.; Coote, D.R.; Rees, H.W.; Wang, C.; Chow, T.L. Effects of potato production on soil quality and yield at a benchmark site in New Brunswick. *Soil Till. Res.* **1994**, *29*, 23–34.
- Vandaele, K.; Vanommeslaeghe, J.; Muylaert, R.; Govers, G. Monitoring soil redistribution patterns using sequential aerial photographs. *Earth Surf. Proc. Land.* **1995**, *21*, 353–364.
- Lindstrom, M.J.; Nelson, W.W.; Schumacher, T.E. Quantifying tillage erosion rates due to moldboard plowing. *Soil Till. Res.* **1992**, *24*, 243–255.
- Govers, G.; Vandaele, K.; Desmet, P.J.J.; Poesen, J.; Bunte, K. The role of tillage in soil redistribution on hillslopes. *Eur. J. Soil Sci.* **1994**, *45*, 469–478.
- Quine, T.A.; Walling, D.E. Use of caesium-137 measurements to investigate relationships between erosion rates and topography. In *Landscape Sensitivity*; Thomas, D.S.G., Allison, R.J., Eds.; John Wiley: New York, 1993; 31–48.
- Quine, T.A.; Desmet, P.J.J.; Govers, G.; Vandaele, K.; Walling, D.E. A comparison of the role of tillage and water erosion in landform development and sediment export on agricultural land near Leuven, Belgium. In *Variability in Stream and Sediment Transport*, Proceedings of the Canberra Symposium, Canberra, Australia, Dec 1994; 77–86.
- Quine, T.A.; Walling, D.E.; Chakela, G.K.; Mandiringana, O.T.; Zhang, X. Rates and patterns of tillage and water erosion on terraces and contour strips: Evidence from caesium-137. *Catena* **1999**, *36*, 115–142.
- Govers, G.; Quine, T.A.; Walling, D.E. The effect of water erosion and tillage movement on hillslope profile development: A comparison of field observations and model results. In *Farm Land Erosion: In Temperate Plains Environment and Hills*; Wicherek, S., Ed.; Elsevier: Amsterdam, 1993; 285–300.
- Lobb, D.A.; Kachanoski, R.G.; Miller, M.H. Tillage translocation and tillage erosion on shoulder slope landscape positions measured using ¹³⁷Cs as a tracer. *Can. J. Soil Sci.* **1995**, *75*, 211–218.
- Lobb, D.A.; Kachanoski, R.G. Modeling tillage translocation using step, linear-plateau and exponential functions. *Soil Till. Res.* **1999**, *51*, 317–330.
- Battison, L.A.; Miller, M.H.; Shelton, I.J. Soil erosion and corn yield. I. Field evaluation. *Can. J. Soil Sci.* **1987**, *67*, 731–745.
- Govers, G.; Quine, T.A.; Desmet, P.J.J.; Walling, D.E. The relative contribution of soil tillage and overland flow erosion to soil redistribution on agricultural land. *Earth Surf. Proc. Land.* **1996**, *21*, 929–946.

Tillage Erosion: Measurement Techniques

David A. Lobb

Department of Soil Science, University of Manitoba, Winnipeg, Manitoba, Canada

Abstract

Tillage erosion is the redistribution of soil that occurs within a landscape as a direct result of tillage. Variations in the amount of soil moved by tillage cause loss and accumulation. Tillage translocation is the movement of soil by tillage; variability in translocation is affected by the design and operation of tillage implements and by the topographic and soil properties of landscapes. A wide variety of materials and methods have been used to measure tillage erosion. There are two general methods of measuring tillage erosion: measurement of soil loss and accumulation at points over the surface of a landscape and measurement of tillage translocation at points over a landscape and the difference between those points.

INTRODUCTION

Tillage erosion is the redistribution of soil that occurs within a landscape as a direct result of tillage. Variations in the amount of soil moved by tillage cause loss and accumulation. Tillage translocation is the movement of soil by tillage; variability in translocation is affected by the design and operation of tillage implements (implement width and length, tool shapes and arrangement, tillage depth and speed, available power to maintain operating speed and depth under changing conditions) and by the topographic and soil properties of landscapes (slope gradient and curvature and soil texture, bulk density, and moisture content). Typically, tillage results in the progressive downslope movement of soil, causing severe soil loss on upper slope positions and accumulation in lower slope positions (Fig. 1). A wide variety of materials and methods have been used to measure tillage erosion.

The earliest scientific study of tillage erosion took place in the United States in the 1930s.^[1] Since then, 40–50 studies have been conducted around the world, most undertaken between 1995 and 2005.^[2] Very few of these studies used identical materials and methods, reflecting the evolution of experimental techniques and the diversity of experimental conditions. There are two general methods of measuring tillage erosion: measurement of soil loss and accumulation at points over the surface of a landscape and measurement of tillage translocation at points over a landscape and the difference between those points.

TRANSLOCATION MEASUREMENTS

Tillage translocation is normally measured with a tracer that is incorporated into the soil in plots. The distributions of the tracer before and after tillage are used to calculate translocation—forward translocation from the distribution

parallel to the direction of tillage and lateral translocation from the distribution perpendicular to the direction of tillage (Fig. 2).

There are two methods of calculating translocation from tracer distributions. For the more common method, the quantity of soil translocated per unit width of tillage is calculated directly from the tracer distributions.^[3] For the less common method, a summation curve is generated from a tracer distribution by employing convolution, and translocation per unit width of tillage is calculated from this curve.^[4,5] Both methods provide accurate measures of translocation but the latter provides additional information regarding the distance over which translocated soil is dispersed.^[6] Translocation can be expressed as a length (averaged over the depth of tillage), a volume, or a mass.

Tillage erosion is expressed similarly to wind and water erosion, i.e., a change in soil mass per unit area or a change in elevation. The calculation of tillage erosion from translocation measurements is simply the difference in translocation between two points, divided by the separating distance—An increase or decrease of kilograms of soil translocated per meter width of tillage divided by the meters of distance between measurements.

Within a complex soil landscape, numerous measurements of tillage translocation may be required. Plots should be placed so that the variability in translocation is characterized in detail. Where slope gradient has been assumed to be the dominant factor affecting soil movement, plots should be located over the full range of gradients. Greater attention should be paid to those areas where slope gradient is changing (i.e., convex and concave areas), because changes in slope gradient result in soil loss and accumulation. A pair of plots positioned at the steepest point along a slope profile—one tilled upslope and the other tilled downslope—provides measures of the minimum and maximum translocation, net downslope translocation, and the total soil loss from the upper slope and the total soil

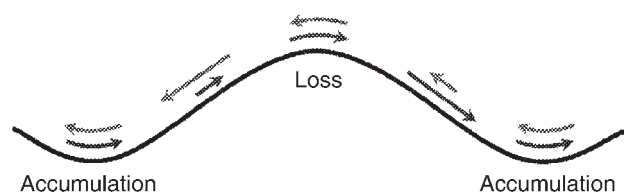


Fig. 1 Variability of translocation in hilly landscape (upslope–downslope tillage): Soil loss occurs on convex areas and accumulation occurs on concave areas. (Translocation is indicated by arrows, its magnitude by their lengths.)

accumulation on the lower slope. Although the minimum and maximum values of translocation occur at this point on the hillslope, if the gradient in this region of the slope is uniform, no soil loss or accumulation will occur at this point.

Tracers consist of plot tracers and point tracers. Point tracers are individually labeled tracers of various shapes and materials, such as steel nuts^[7] or plastic spheres.^[8] Plot tracers are used to label a volume of soil; they can be physical (e.g., gravel^[9]) or chemical (e.g., ^{134}Cs ^[3] and chlorine^[4]). Point tracers also represent a volume of soil but they have the distinct advantage that they can be used to characterize the complexity of soil movement in 3-D. However, plot tracers provide a more accurate measure of bulk soil movement in 2-D.

After tillage, soil is sampled and analyzed to determine the distribution of the tracer. Sampling usually involves destruction of the site, eliminating the possibility of successive translocation measurements from recurrent tillage operations. Nondestructive sampling methods do exist.^[4] Sampling must be carried out as soon as possible after tillage to isolate the movement of soil by tillage from that by other processes. Other measures can also be taken to isolate soil movement by tillage.^[5]

The precise relocation of plots and the recovery of the tracer are important considerations when sampling. Precise plot relocation is best achieved by plot making materials where the base of the plot is left in place throughout the measurement process.^[10] This plot base is used as a reference for tracer placement and movement. An acceptable level of tracer recovery is greater than 95%. Tracer that is not recovered is usually the result of not sampling deep enough or to the full extent of forward and lateral movement.

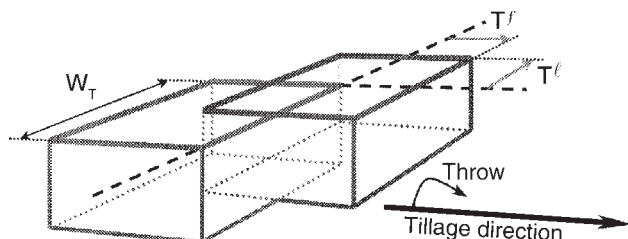


Fig. 2 Simplified illustration of soil translocation by tillage: T^f = forward translocation; T^l = lateral translocation; and W_T = unit width of tillage.

Every effort should be made to sample the full extent of tracer movement because small quantities of unrecovered tracer at the furthest extent can greatly impact the accuracy of erosion measurements.

The design of plots is possibly the most important consideration when using tracers. Even under controlled operating conditions, the movement of soil during tillage is inherently variable because of the nature of soil–tillage interactions. Soil strains and fails during tillage, and tools and implements flex and shift—The “flow” of soil during tillage is not steady. For accurate measurement of soil movement by tillage, plots must be sufficiently long and wide to filter out this “noise.” The length of plots used in tillage erosion studies has ranged from a few centimeters to a few meters. As stated previously, a longer plot provides a better measure of the average movement of soil by tillage. The movement of soil varies across the width of tillage as a function of the tool type, arrangement, and spacing. For accurate measurement of soil movement by tillage, the width of the plot must be a multiple of the unit width of tillage. For a single pass of a simple tillage implement, the unit width is the distance between two adjacent tools. However, for an implement with multiple tool types or for a sequence of different implements, establishing the unit width of tillage may not be possible. To reduce measurement errors associated with plot width, the plot must be as wide as possible. The paths of a tillage implement overlap in the field (as much as 10%), with the exception of implements such as the moldboard plow, causing additional soil movement at the edges of each path. To account for this additional movement, plot widths must be equal to the spacing of implement paths; however, this is not practical in most cases. The depth of plots must exceed the maximum depth of the tillage operation(s) under examination. Tillage depth can vary greatly within a landscape, so caution must be used in estimating the maximum tillage depth—it is better to overestimate tillage depth than to underestimate.

Tillage translocation can also be measured using detailed elevation surveys of the soil surface. The redistribution of soil volume, along with soil bulk density, can be used to measure soil translocation.^[1,9] There are two good examples of this technique. In the study of tillage erosion by contour moldboard plowing, the profiles of furrow slices turned upslope are compared to those of furrow slices turned downslope—The difference is a measure of net downslope translocation. In the study of tillage erosion by hoeing on steep slopes (soil is only hoed downslope), the profiles of the furrow created at the top of the slope and the ridge created at the bottom are measures of downslope translocation.

MEASUREMENTS OF LOSS AND ACCUMULATION

Soil loss and accumulation by tillage erosion can be estimated from changes in soil properties or surface elevation. In most

cases, the loss or accumulation from a single tillage event or an annual sequence of events would be too small to measure accurately; consequently, measurements are usually made over a period of many years. The distribution of these changes over the surface of the landscape allows the assessment of soil redistribution.

Soil surface elevation is commonly used as an indicator of recurrent tillage erosion. The elevation within a tilled portion of the landscape is compared to an adjacent non-eroding landscape feature, such as a fenceline or hedgerow. Decreases in surface elevation on upper slope landscape positions and increases in elevation on lower slope positions are classic evidences of tillage erosion. Using tillage history and elevation changes, it is possible to estimate the average annual rate of tillage erosion.

Soil properties, such as the amount of organic matter and carbonates, are commonly used as indicators of recurrent tillage erosion. Organic-poor, carbonate-rich subsoil exposed on the surface of upper slope landscape positions is a classic evidence of soil loss by tillage erosion. Accumulation of organic-rich soil in convergent slope positions is also a classic evidence of tillage erosion. However, such familiar soil properties are not considered reliable estimators of soil erosion. Radioisotopes found in soil are used to accurately estimate soil erosion; the most frequently used are ^{137}Cs and ^{210}Pb .^[5,11] ^{210}Pb naturally occurs and ^{137}Cs originates from human use of nuclear radiation. Both isotopes contact the soil through atmospheric deposition and remain concentrated in the surface soil. The spatial distribution of radioactivity is compared to a baseline distribution or to a single value based on a non-eroded reference site. This technique is suited to timescales of 5–40 yr for ^{137}Cs and 50–150 yr for ^{210}Pb . Soil magnetic susceptibility^[12] and fly ash concentrations have been used to estimate soil erosion.

The use of long-term soil loss and accumulation to estimate rates of tillage erosion is problematic in that it is not possible to isolate individual erosion processes. Furthermore, erosion processes interact. The contribution of tillage erosion to the total soil redistribution within a landscape must be determined based on the signature patterns of redistribution by individual erosion processes.^[13]

CONCLUSION

The assessment of soil erosion in cultivated landscapes requires the measurement of tillage erosion. Tillage erosion is measured directly by measuring the movement of soil by tillage – tillage translocation. Translocation is measured using plot tracers or point tracers. Careful consideration

must be given to several aspects of the techniques used to measure tillage translocation and tillage erosion to ensure accurate and meaningful information. Techniques for measuring tillage translocation and tillage erosion are well developed but are undergoing enhancements.

REFERENCES

1. Mech, S.J.; Free, G.R. Movement of soil during tillage operations. *Agric. Eng.* **1942**, *23*, 379–382.
2. Govers, G.; Lobb, D.A.; Quine, T.A. Tillage erosion and translocation: Emergence of a new paradigm in soil erosion research. *Soil Till. Res.* **1999**, *51*, 167–174.
3. Quine, T.A.; Govers, G.; Poesen, J.; Walling, D.E.; van Wesemael, B.; Martinez-Fernandez, J. Fine earth translocation by tillage in stony soils in the Guadalentin, southeast Spain: An investigation using caesium-134. *Soil Till. Res.* **1999**, *51*, 279–301.
4. Lobb, D.A.; Kachanoski, R.G.; Miller, M.H. Tillage translocation and tillage erosion on shoulder slope landscape positions measured using ^{137}Cs as a tracer. *Can. J. Soil Sci.* **1995**, *75*, 211–218.
5. Lobb, D.A.; Kachanoski, R.G.; Miller, M.H. Tillage translocation and tillage erosion in the complex upland landscapes of southwestern Ontario, Canada. *Soil Till. Res.* **1999**, *51*, 189–209.
6. Lobb, D.A.; Quine, T.A.; Govers, G.; Heckrath, G. Comparison of methods used to calculate tillage translocation using plot-tracers. *J. Soil Water Conserv.* **2001**, *56*, 321–328.
7. Lindstrom, M.J.; Nelson, W.W.; Schumacher, T.E. Quantifying tillage erosion rates due to moldboard plowing. *Soil Till. Res.* **1992**, *24*, 243–255.
8. Govers, G.; Vandaele, K.; Desmet, P.J.J.; Poesen, J.; Bunte, K. The role of tillage in soil redistribution on hillslopes. *Eur. J. Soil Sci.* **1994**, *45*, 469–478.
9. Turkelboom, F.; Poesen, J.; Ohler, I.; Ongprasert, S. Reassessment of tillage erosion rates by manual tillage on steep slopes in northern Thailand. *Soil Till. Res.* **1999**, *51*, 245–259.
10. Zhang, J.H.; Li, Y.; Lobb, D.A.; Liu, G.C. Assessment for tillage translocation and tillage erosion by hoeing tillage in hilly subtropical regions, southwestern China. *Soil Till. Res.* **2003**, *75*, 99–108.
11. Walling, D.E.; He, Q.; Quine, T.A. Use of fallout radionuclide measurements in sediment budget investigations. *Géomorphologie* **1996**, *3*, 17–28.
12. de Jong, E.; Nestor, P.A.; Pennock, D.J. The use of magnetic susceptibility to measure long-term soil redistribution. *Catena* **1998**, *32*, 23–35.
13. Govers, G.; Quine, T.A.; Desmet, P.J.J.; Walling, D.E. The relative contribution of soil tillage and overland flow erosion to soil redistribution on agricultural land. *Earth Surf. Processes.* **1996**, *21*, 929–946.

Tillage Erosion: Properties and Productivity Effects

Tom E. Schumacher

*Department of Plant Science, South Dakota State University, Brookings,
South Dakota, U.S.A.*

Abstract

The relation between soil productivity and soil erosion is complex. Soil productivity is dependent on the ability of the soil to provide water, nutrients, and oxygen to the plant root system as well as to limit the exposure of the root system to toxic gases or solutions. Tillage erosion, the downslope displacement of soil through the action of tillage operations, is a specific form of soil erosion. The impact of tillage erosion on soil productivity is primarily related to soil removal from a specific landscape position and deposition in another part of the landscape. As a result, many of the causes of change in soil productivity attributed to other forms of erosion also apply to tillage erosion. Tillage erosion acts on soil productivity through the loss of effective rooting depth, loss of plant nutrients, and loss of plant-available water.

INTRODUCTION

The relation between soil productivity and soil erosion is complex. Soil erosion is defined as the wearing away of the land surface by natural or anthropogenic agents that abrade, detach, and remove soil from one point on the earth's surface and deposit it elsewhere.^[1] The capacity of a soil to produce a certain yield of crops or other plants within a specified system of management is defined as soil productivity.^[1] Soil productivity is dependent on the ability of the soil to provide water, nutrients, and oxygen to the plant root system as well as to limit the exposure of the root system to toxic gases or solutions. The processes involved in the supply of beneficial and toxic compounds are dependent on storage capacities and transport rates that are determined by an array of soil properties. Soil and topographic properties that control the inherent productivity of a specific field location include slope, effective root zone depth, bulk density, available water-holding capacity, plant-available nutrient content, soil aeration, water infiltration and percolation, soil pH, aluminum and manganese concentration, root zone cation exchange capacity, aggregate stability, salinity, and nutrient cycling by soil biological organisms.^[2–4] As soil erodes, soil properties of the root zone often are altered. Consequently, storage capacity and transport rates of compounds critical for plant growth also change. Several reviews have been written on various aspects of the subject of soil erosion, soil properties, and productivity.^[5–7]

Tillage erosion, the downslope displacement of soil through the action of tillage operations,^[1] is a specific form of soil erosion. The impact of tillage erosion on soil productivity is primarily related to soil removal from a specific landscape position and deposition in another part of the landscape. As a result, many of the causes of change in soil

productivity attributed to other forms of erosion also apply to tillage erosion. Lal^[8] lists several direct effects of soil erosion on crop yield. These include a reduction in effective rooting depth, loss of plant nutrients, loss of plant-available water, loss of land area (due to gully formation and unfavorable conditions in depositional areas), and damage to seedlings from the action of the erosion agent. Of these, tillage erosion acts on soil productivity through the first three, namely: loss of effective rooting depth, loss of plant nutrients, and loss of plant-available water.

TILLAGE EROSION AND SOIL PROPERTIES

Soil productivity changes due to tillage erosion at eroding landscape positions are dependent on the properties of the underlying soil horizons and the severity of soil loss. Lal^[8] describes three scenarios of erosion-induced change in soil productivity that depend on subsoil properties. Some soils may have no change in soil properties affecting crop growth. These soils usually are deep soils with relatively uniform soil properties in and below the root zone. A second scenario involves rare cases where soil properties in subsurface soil horizons are more favorable for plant growth than in the A horizon. In these soils, soil productivity actually increases with erosion. A third scenario, which is by far the most common in agricultural fields, involves soils that drop in productivity as soil is removed from the surface. These soils have subsurface horizons with soil properties that are less beneficial to plant growth than soil in the A horizon.

Tillage erosion changes soil properties and thus soil productivity by at least two mechanisms. One mechanism involves the net loss of topsoil from field locations where the slope forms a convex shape such as the shoulder

landscape position. A second mechanism involves the mixing of topsoil with subsoil exposed by tillage erosion as soil is translocated by the tillage operation beyond the eroding landscape position.

Over a period of time, tillage erosion can result in the exposure of subsoil. Tillage operations spread and mix subsoil with topsoil as soil is translocated downslope even though there may be no net loss of soil in the translocation zone.^[9,10]

This mixing of topsoil with subsoil spread by tillage equipment can result in a change in soil properties that reduces soil productivity in adjacent non-eroding landscape positions. The subsoil materials over time will become mixed from the zone of soil loss (convex slopes) to the zone of deposition (concave slopes).^[11]

LANDSCAPE POSITION AND SOIL PRODUCTIVITY

The effects of different erosion agents on soil productivity can often be better differentiated by their location of

action within the landscape than by unique changes in soil properties. Tillage erosion occurs in topographic positions that are convex such as shoulder positions.^[12] Water erosion primarily occurs in mid-to-lower portions of a backslope.^[13] Wind erosion occurs primarily on coarse-textured soils and is most severe on shoulder positions facing into the prevailing wind.^[14] Tillage erosion appears to be responsible for most of the soil erosion occurring in shoulder positions where wind erosion is not a predominate agent of erosion.^[15] Therefore, soil productivity losses observed in shoulder positions and on knolls in humid and semihumid regions are likely the result of tillage erosion.

The shoulder position prior to agriculture often has lower solum thickness.^[16] As a result, changes in soil productivity from tillage erosion may occur more rapidly and with greater intensity per unit of soil erosion because the soil tends to be naturally shallower in the shoulder position. Fig. 1 illustrates a 32-ha field from Central South Dakota showing variation in winter wheat (*Triticum aestivum* L.) yield, predicted soil gain/loss from moldboard plow tillage, and available phosphorous based on the Olsen P test.^[17]

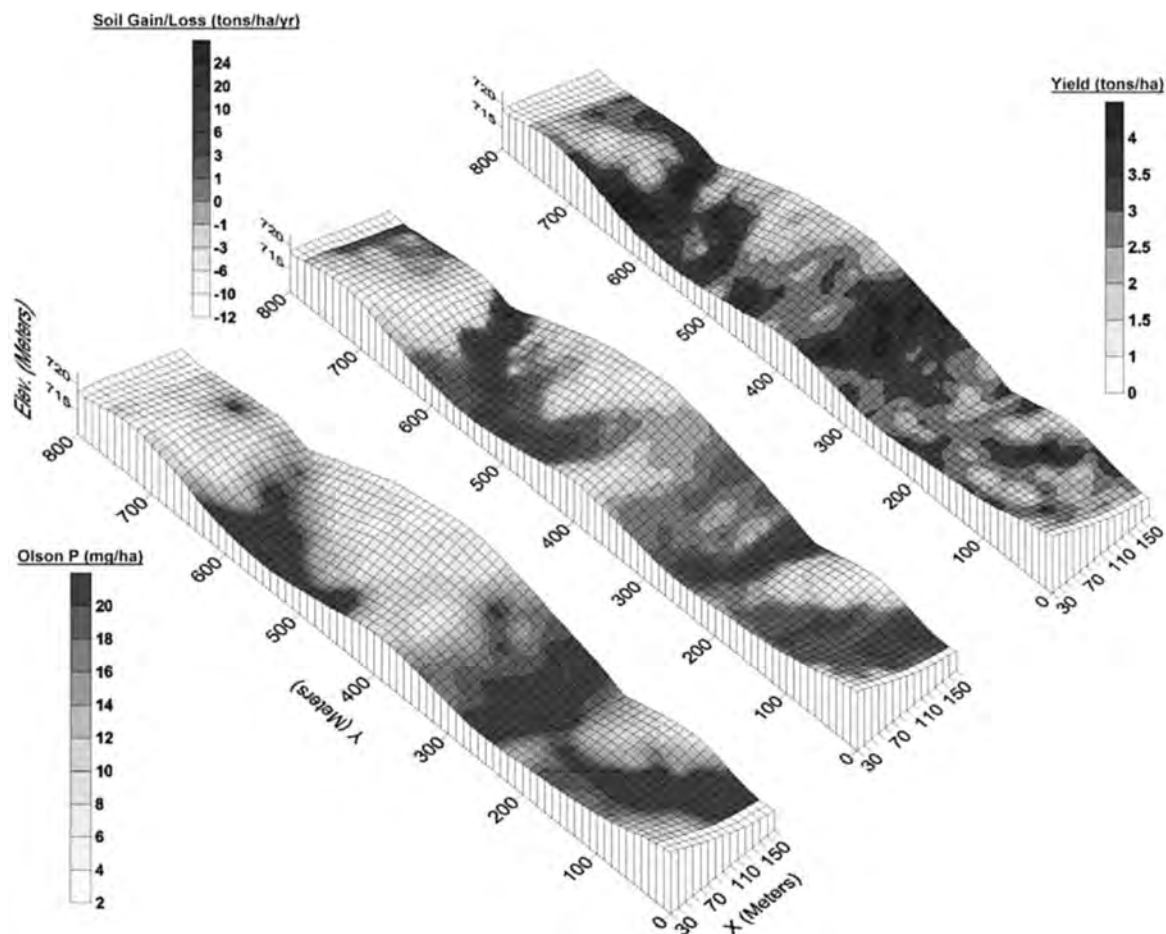


Fig. 1 Predicted soil loss from tillage erosion, winter wheat yield, and Olsen P test from a 32-ha field in Central South Dakota. Soil gain (deposition) is represented by positive values in the middle graph, while soil loss (erosion) is represented by negative values.

Lower yields in this field tended to be associated with field areas that had high rates of tillage erosion, lower plant-available phosphorous, and exposed subsoil material.

MODELING TILLAGE EROSION AND SOIL PRODUCTIVITY

The complexity of the soil erosion–productivity relationship is such that few generalizations can be made. The mechanisms by which tillage erosion impacts soil productivity will change depending on soil and climate. Site-specific soil property measurements can be very expensive and time-consuming if applied to large-scale fields. Modeling erosion events, erosion-induced changes in soil properties within the root zone, and soil property–productivity relationships is one approach that may be used to evaluate fields. In one study, a tillage erosion model^[18] and water erosion model^[19] were used to evaluate the effects of erosion on soil property distribution in a closed depressional landscape.^[20] The erosion-induced change in soil properties was then coupled with a productivity index,^[21] a model that relates soil properties to soil productivity. Model results showed that considerable variability in soil productivity was introduced into the landscape by the combined action of tillage and water erosion.

In most cases, soil erosion is because of an interaction of various erosive agents such as tillage and water, tillage and wind, or a combination of all the three agents. An evaluation of soil productivity within eroded/eroding landscapes must take into account the interaction of all erosive agents on soil properties and the specific effect of soil properties as related to the supply of water, oxygen, and nutrients to the root systems of growing plants. Using less intensive tillage operations has the beneficial effect of reducing soil loss from all the three erosion agents, thus reducing the rate of change in soil productivity. However, a reduction in tillage operations by itself is less likely to reverse changes in soil properties and productivity caused by erosion. The restoration of productivity to eroded soils is often difficult, time-consuming, and likely to require the integrated application of older and yet-to-be developed technology.

REFERENCES

1. Soil Science Society of America. *Glossary of Soil Science Terms*; SSSA: Madison, 1997; 134 pp.
2. Letey, J. Relationship between soil physical properties and crop production. *Adv. Soil Sci.* **1985**, *1*, 277–294.
3. Pierce, F.J.; Lal, R. Monitoring soil erosion's impact on crop productivity. In *Soil Erosion Research Methods*; Lal, R., Ed.; Soil Water Conservation Society: Ankeny, 1994; 235–264.
4. Shaffer, M.J.; Schumacher, T.E.; Ego, C.L. Simulating the effects of erosion on corn productivity. *Soil Sci. Soc. Am. J.* **1995**, *59*, 672–676.
5. Follet, R.F.; Stewart, B.A., Eds. *Soil Erosion and Crop Productivity*; American Society of Agronomy: Madison, 1985; 533 pp.
6. Lal, R. Soil erosion impact on agronomic productivity and environment quality. *Crit. Rev. Plant Sci.* **1998**, *17*, 319–464.
7. Lal, R. Effects of soil erosion on crop productivity. *CRC Crit. Rev. Plant Sci.* **1987**, *5*, 303–367.
8. Lal, R. Monitoring soil erosion's impact on crop productivity. In *Soil Erosion Research Methods*; Lal, R., Ed.; Soil Conservation Society: Ankeny, 1988; 187–200.
9. Lobb, D.A.; Kachanoski, R.G. Modeling tillage translocation using step, linear-plateau, and exponential functions. *Soil Till. Res.* **1999**, *51*, 317–330.
10. Sibbesen, E.; Skjoth, F.; Rubaek, G.H. Tillage caused dispersion of phosphorous and soil in four 16-year old field experiments. *Soil Till. Res.* **2000**, *54*, 91–100.
11. Alba-De, S.; Lindstrom, M.; Schumacher, T.E.; Malo, D.D. Soil landscape evolution due to soil redistribution by tillage: A new conceptual model of soil catena evolution in agricultural landscapes. *Catena* **2004**, *58*, 77–100.
12. Govers, G.; Vandaele, K.; Desmet, P.; Poesen, J.; Bunte, K. The role of tillage in soil redistribution on hillslopes. *Eur. J. Soil Sci.* **1994**, *45*, 469–478.
13. Young, R.A.; Mutchler, C.K. Effect of slope shape on erosion and runoff. *T. ASABE*. **1969**, *12*, 231–233.
14. Chepil, W.S.; Siddoway, F.H.; Armbrust, D.V. Wind erodibility of knolly terrain. *J. Soil Water Conserv.* **1964**, *19*, 179–181.
15. Govers, G.; Lobb, D.A.; Quine, T.A. Tillage erosion and translocation emergence of a new paradigm in soil erosion research. *Soil Till. Res.* **1999**, *51*, 167–174.
16. Hall, G.F. Pedology and Geomorphology. Pedogenesis and soil taxonomy: Concepts and interactions. In *Developments in Soil Science*; Wilding, L.P., Smeck, N.E., Hall, G.F., Eds.; 11A Ed.; Elsevier: Amsterdam, 1983; 117–138.
17. Bly, A.; Gerwing, J.; Gelderman, R.; Mitchel, V. Influence of starter P and Zn fertilizer on yield of no-till corn across strips with residual P rates. In *Soil Progress Report 97–16, Soil Water Science Research 1997 Annual Report*; South Dakota State University: Brookings, 1997; 1–3.
18. Lindstrom, M.J.; Schumacher, J.A.; Schumacher, T.E. TEP: A tillage erosion prediction model to calculate soil translocation rates from tillage. *J. Soil Water Conserv.* **2000**, *55*, 105–108.
19. Flanagan, D.C.; Nearing, M.A. *USDA—Water Erosion Prediction Project Hillslope Profile and Watershed Model Documentation*, NSERL Report No. 10; USDA-ARS National Soil Erosion Research Laboratory: West Lafayette, 1995.
20. Schumacher, T.E.; Lindstrom, M.J.; Schumacher, J.A.; Lemme, G.D. Modeling spatial variation in productivity due to tillage and water erosion. *Soil Till. Res.* **1999**, *51*, 331–339.
21. Pierce, F.J.; Larson, W.E.; Dowdy, R.H.; Graham, W.A.P. Productivity of soils: Assessing long-term changes due to erosion. *J. Soil Water Conserv.* **1983**, *38*, 39–44.

Tillage Erosion: Relationship to Water Erosion

Gerard Govers

Laboratory for Experimental Geomorphology, Catholic University of Leuven, Leuven, Belgium

Abstract

Tillage and water erosion interact in various ways. It has long been known that tillage affects the soil's resistance to detachment and transport by water. Soil tillage alters the soil structure, reduces soil's bulk density, breaks up the soil crust, and incorporates superficial residue into the soil. However, these phenomena are not directly related to the fact that during tillage operations, soil displacement and tillage erosion occur. Tillage translocation and erosion have specific effects on the water erosion processes. Conversely, water erosion processes may affect tillage translocation and erosion. We discuss the interactions between tillage erosion and water erosion that occur at various timescales.

TILLAGE EROSION AND WATER EROSION

On a short timescale, tillage erosion acts as a delivery mechanism for water erosion. Water erosion is often the strongest along the central axis of hill slope concavities (hollows); here, a large amount of surface runoff is concentrated, whereas slope gradients are often relatively high.^[1] This often leads to ephemeral gully erosion.^[2] At the same time, important amounts of soil are deposited here as a result of tillage. The delivery of sediment by tillage may exceed the average water erosion rate, in which case net deposition will occur.

Conversely, if soil delivery by tillage is lower than the average water erosion rate, the result will be net incision. The balance depends upon the relative intensity of both processes and the landscape morphology. Deposition by tillage will increase with increasing slope concavity. On the other hand, water erosion rates will increase with increasing slope gradient and upslope water contributing area. The interaction between tillage and water erosion is not one way; when ephemeral gullies form, farmers will often undertake special tillage operations to fill them in, which then leads to an increased soil displacement by tillage.

On a longer timescale, tillage translocation and erosion affect topography. Tillage translocation leads to the deposition of soil in concavities (hollows) as well as on the upslope side of field borders. It causes soil erosion on convexities (shoulders) as well as the downslope side of field borders. Therefore, the process will lead to a diminution of the overall relief energy within a field and the water erosion risk, as the latter strongly depends on the slope gradient.

The accumulation and erosion of soil at field borders can be fairly rapid, leading to the development of soil

banks of several meters high over a time period of a few decades if the soil is consistently thrown downslope during tillage. In intensive tillage systems in Western Europe, concavities are filling at a rate of 5–20 mm yr⁻¹ because of tillage deposition, while convexities are typically eroding at rates of 0.5–2 mmyr⁻¹.^[3] Considering a typical slope of 100 m having an average gradient of 10%, it will take 44–180 years to reduce average slope gradient by 1% point, neglecting the effects of water erosion on topography. Changes in relief energy will be more rapid when convexities and concavities are more pronounced, as is the case on many moraine landscapes in Northern Europe and America.^[4]

In many agricultural systems, slopes are interrupted by vegetative borders and/or water diversion structures to reduce the water erosion risk. In such systems, soil translocation by tillage will lead to much more rapid and important changes in topography. For example, tillage translocation will gradually decrease the slope of the field strips because of the deposition of soil at the downslope end of the field and erosion at the upslope end (Fig. 1).^[5] Again, the speed at which these changes occur depends on the initial topography and the intensity of the tillage erosion process. Changes will be more rapid when initial slopes are high, tillage is intense, and field borders are closely spaced. Calculations for 6-m wide, slow-forming terraces in Ecuador show that tillage erosion will reduce the initial slope gradient of 30% to approximately 6% in 25 years using traditional tillage techniques. In some cases, this effect may be deliberately speeded up by consistently throwing the soil downslope when tilling the field strips.

Some positive effects of the reduction of within-field slope gradient may be counteracted by erosion processes specifically related to the soil banks. In some cases,

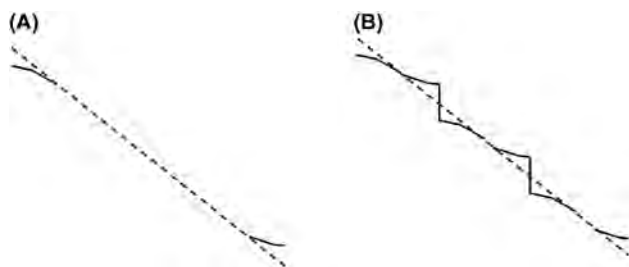


Fig. 1 Schematic representation of the evolution of the slope profile by tillage erosion with (B) and without (A) the presence of diversion structures or terraces.

gully erosion may form on the terrace banks. Such bank gullies can be initiated by piping caused by animal activity, cracking, and/or mass transport processes occurring on the banks. The risk of bank failure and the speed at which bank gullies will develop depend, to a large extent, on the properties of the soil profile.

Evidently, water erosion may also lead to topographical changes affecting tillage erosion rates and patterns. Generally, water erosion will tend to incise the landscape, leading to steeper slope gradients and more pronounced convexities and concavities. This can be seen in its most extreme form in badlands. As a consequence, the potential for tillage erosion is increased. However, the reverse may also occur when the within-field deposition of sediment by water is important. This deposition will lead to a reduction in relief energy and, eventually, a reduction in tillage erosion.

Tillage erosion also affects water erosion by changing soil properties. The extent and nature of such changes depend on the initial properties of the soil. Generally, such changes will be more important when the soil's A horizon is shallow and when there is a large difference in properties between the A horizon and the lower soil horizons.

In the first phase, tillage erosion leads to the exposure of subsoil on convexities and at upslope field borders. If tillage erosion continues for a sufficiently long period, this subsoil material will be transported further downslope. During downslope transport, the subsoil material is mixed with the original topsoil. After a long time period, the properties of the plough layer will be predominantly determined by the properties of the original subsoil at convexities and upslope field borders. They will gradually change to those of the original topsoil in the downslope direction. The area where subsoil properties dominate will increase with time.

In the final stage, a soil profile inversion may occur over large parts of the landscape, whereby the original topsoil material is buried by a layer of subsoil material eroded further upslope. The effect of these changes on water erosion rates and patterns may vary. On stony soils, tillage may increase stone cover on convexities, thereby

leading to a considerable decrease in water erosion risk from the protective effect of the stones.^[6] However, in most cases, the properties of the subsoil are such that tillage erosion will lead to an increase in the water erosion risk. Very often, the subsoil material has a lower structural stability than the original topsoil, which causes a higher sensitivity for crusting and more runoff generation in the areas of exposure.

Because the subsoil material is exposed in upslope landscape positions, this will increase the erosion risk over the whole landscape. The runoff generated in the upslope landscape positions increases soil erosion further downslope. This process may be further enhanced by reduced vegetative growth on the less-productive tillage eroded areas, and poorer soil cover and increased erosion rates. In concave landscape positions, the potential negative effect of water erosion on soil quality is to some extent compensated for by tillage deposition. Tillage deposition causes the depth of the A horizon to either decrease more slowly or increase in these landscape positions, thereby potentially promoting carbon sequestration.^[7] However, in the long term, the original A horizon may become buried by subsoil material.

Changes in soil properties like soil texture and organic matter content caused by water erosion are unlikely to affect tillage erosion to a great extent. Results of tillage erosion experiments suggest that tillage speed and depth, tillage tool geometry, and dynamic soil properties, like bulk density and moisture content, have a far greater influence on tillage displacement and erosion.^[8]

Finally, different spatial signatures of tillage and water erosion have implications for the calculation of average soil erosion rates at the field scale and for the assessment of effects on soil quality. Tillage erosion causes only soil redistribution within a field, so that the average soil erosion rate over the field is zero. The intensity of tillage erosion may be better assessed by calculating an average rate over the eroding part of the landscape, as is often done for water erosion. To calculate an overall soil erosion rate, erosion and deposition rates for tillage and water erosion must be calculated using a spatially distributed model, so that soil losses and gains can be assessed at each landscape position. These may be combined to produce a map of the total soil erosion or deposition rate at each point.^[9] This rate can then be calculated by taking the average of the values for the eroding landscape positions.

Because of the compensating nature of water and tillage erosion, the total average soil erosion rate that is calculated using this procedure will generally be much lower, but it will also be more realistic than the sum of the separately calculated average water erosion rate and the average tillage erosion rates. Similarly, the assessment of the soil quality effects of erosion and deposition by tillage and water is only possible using a spatially distributed model that accounts for both processes. Both water erosion and tillage erosion contribute to the redistribution of soil

carbon, albeit in a significantly different way: The effect of erosion on the overall carbon budget of arable land can only be understood if both processes are correctly accounted for.^[7]

CONCLUSION

Water and tillage erosion interact in various ways. On the long term, the processes mainly interact through their differential effect on topography. While tillage erosion essentially tends to reduce the water erosion risk by smoothing the topography, water erosion acts inversely. On a shorter timescale, the effect of tillage erosion on the spatial variation of soil properties at the field scale has a significant effect on the intensity and spatial variability of water erosion. The interaction between water and tillage erosion not only needs to be considered when calculating the overall erosion risk but is also important for a correct understanding of the effect of water and tillage erosion on the carbon budget of arable land.

REFERENCES

1. Govers, G.; Quine, T.A.; Desmet, P.J.J.; Walling, D.E. The relative contribution of soil tillage and overland flow erosion to soil redistribution on agricultural land. *Earth Surf. Proc. Land.* **1996**, *2*, 929–946.
2. Vandaele, K.; Poesen, J.; Govers, G.; Van Wesemael, B. Geomorphic threshold conditions for ephemeral gully erosion. *Geomorphology* **1996**, *16*, 161–173.
3. Govers, G.; Van Daele, K.; Desmet, P.J.J.; Poesen, J.; Bunte, K. The role of soil tillage in soil redistribution on hillslopes. *Eur. J. Soil Sci.* **1994**, *45*, 469–478.
4. Lobb, D.A.; Kachanoski, R.G.; Miller, M.H. Tillage translocation and tillage erosion on shoulder slope landscape positions measured using ¹³⁷Cs as a tracer. *Can. J. Soil Sci.* **1995**, *75*, 211–218.
5. Dabney, S.M.; Liu, Z.; Lane, M.; Douglas, J.; Zhu, J.; Flanagan, D.C. Landscape benching from tillage erosion between grass hedges. *Soil Till. Res.* **1999**, *51*, 219–232.
6. Poesen, J.; Van Wesemael, B.; Govers, G.; Martinez-Fernandez, J.; Desmet, P.J.J.; Vandaele, K.; Quine, T.A.; Degraer, G. Patterns of rock fragment cover generated by tillage erosion. *Geomorphology* **1997**, *18*, 183–197.
7. Van Oost, K.; Govers, G.; Quine, T.; Heckrath, G. Comment on “managing soil carbon.” *Science* **2004**, *305*, 1567.
8. Van Muysen, W.; Govers, G.; Bergkamp, G.; Roxo, M.; Poesen, J. Measurement and modelling of the effects of initial soil conditions and slope gradient on soil translocation by tillage. *Soil Till. Res.* **1999**, *51*, 303–316.
9. Van Oost, K.; Govers, G.; Desmet, P.J.J. Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. *Landscape Ecol.* **2000**, *15*, 577–589.

Tillage Erosion: Terrace Formation

Seth M. Dabney

National Sedimentation Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Oxford, Minnesota, U.S.A.

Dalmo A.N. Vieira

National Sedimentation Laboratory, Arkansas State University, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Jonesboro, Arkansas, U.S.A.

Abstract

All tillage moves soil laterally as well as vertically. With each tillage operation, soil preferentially moves downslope, and some soil usually leaves the tilled zone in the form of clods deposited along field borders. Over time, these clods, along with stones removed from the tilled area and sediment deposited by runoff, coalesce to form terraces or lynchets. The formation of such terraces may or may not be desirable but can hardly be avoided if field boundaries are fixed for long periods of time. This entry reviews research into terrace formation by tillage, identifies ways in which these terraces can be advantageous to soil and water conservation, and discusses problems that have been recognized with tillage terraces.

INTRODUCTION

All tillage moves soil laterally as well as vertically. While a few implements such as a moldboard plow or a one-way disk plow throw soil to only one side, most tillage implements—including tandem disks, chisel plows, harrows, and cultivators—throw soil to both sides. With such implements, a tillage operation along the contour moves some soil uphill, more soil downhill, and still more soil along the contour in the direction of travel.^[1–3] Within an undulating field, each tillage operation flattens the topography, as soil is progressively removed from convexities and is deposited in depressions and other concavities.

Field boundaries interrupt soil fluxes caused by tillage and can result in the formation of two distinct classes of terraces. On sloping land with untilled strips, tillage moves soil toward a down-gradient untilled strip and away from an up-gradient untilled strip. This leads to aggradation of soil upslope of untilled strips and degradation of soil downslope of them and causes the gradual formation of bench terraces. The speed of bench terrace formation is greater where the spacing between the untilled strips is narrower, where the initial slope is steeper, and where tillage is more intense. The other class of terraces formed at field boundaries is due to local ridges and furrows. Depending on tillage tool design and operation, some soil often leaves the tilled zone in the form of clods deposited along field borders. The combination of clods thrown from the tilled area and dead furrows left behind creates small terrace channels at the margins of tilled areas that can intercept and alter runoff flow patterns. The impact of these small gradient terraces on hydrology is greatest on gently sloping lands. This entry

reviews research into terrace formation by tillage, identifies ways in which these terraces can be advantageous to soil and water conservation, and discusses problems that have been recognized with tillage terraces.

ANCIENT LYNCHETS

Archeologists use the term “lynchets” to refer to soil banks that are believed to be remnants of ancient agricultural activities. These ancient soil banks have been studied intensively for over 100 years. Lynchets dating from the bronze age, iron age, and medieval periods are all generally recognized to be the morphological response on a hillslope to the presence of field boundaries in cultivated landscapes.^[4] While medieval lynchets tend to be oriented along the slope, older “Celtic fields” were often square and bounded by lynchets on all sides,^[5] and some lynchets are oriented up and down the slope.^[6] Lynchets form at all field boundaries either bounded by untilled grass strips (balks), hedge rows, paths, fences, or ditches.^[6,7] Frequently, lynchets contain piles of stones at their core. These stones may have marked property boundaries or may simply represent convenient disposal locations as stones were removed from fields.

CONTEMPORARY LYNCHETS

Contemporary lynchet formation has been observed and described around the globe.^[7–23] The presence of a physical boundary such as field borders, fences, hedge rows, or vegetative barriers creates a local interruption of the soil

movement created by tillage, resulting in morphological changes in the vicinity of the boundary. Erosion or deposition occurs depending on the orientation of terrain slopes with respect to the boundary. When the tilled terrain slopes away from the boundary, the net soil translocation in the downhill direction creates erosion near the boundary, while soil deposition occurs where the terrain slopes toward the boundary. Localized erosion also occurs wherever tillage implements first engage the soil, and deposition occurs at locations where tillage tools are removed from the ground and turned around. The aggrading side of a field boundary is often termed a “positive lynchets,” while the degrading side (the upslope edge of a tilled area) is termed a “negative lynchets” (Fig. 1). Positive lynchets have deeper, more fertile, and more productive soils than those on negative lynchets, where subsoil may be exposed.^[12,19,21,22,24]

A modern example of lynchets formation is provided by a field subdivided by grass hedges planted on the contour in North Mississippi, U.S.A., which was surveyed periodically over 16 years (Fig. 1). Tillage between the hedges led to erosion in the downslope side of each hedge, as soil was displaced in the downhill direction by tillage, and the hedges caused deposition of sediment transported by water and tillage on their upslope side. Lynchets developed rapidly once contour tillage with disks and chisel plows began between the grass hedges.^[25,26] Changes in elevation reflected the combined influences of tillage and water erosion. At each grass hedge, lynchets of up to 0.8 m formed during a period of 16 years, and the average slope of field was reduced from 7.2%, when the hedges were first established, to 3.7%.^[27]

Hand tillage with a hoe differs from that associated with draft animals or tractors because soil usually moves

in a direction opposite of the workers’ travel. Because hoeing is hard work, it is common for soil to move mainly downhill, with work starting at the bottom of a field and proceeding upslope.^[19] Again, the net result is gradual terrace formation.

Soil erosion by water generally increases with increasing slope length because of an accumulation of runoff. This should lead to the development of concave hillslope profiles with little change in elevation near hilltops.^[28] In contrast, the quantity of soil moved by tillage is independent of field size and leads to rapid soil loss at upslope field boundaries. Since erosion rates are generally considered in terms of mass per unit area, the amount of soil “eroded” by tillage is much greater on narrow fields than on long slopes.^[19,23,29] Thus, the more strips into which a field is divided, the greater the contribution of tillage translocation to terrace formation, and the more rapid terrace formation will be.

ENGINEERED TERRACES

Engineered terraces are designed to manage runoff from a certain contributing area. Therefore, when a field is to receive more than one terrace, it is common practice to start at the top of the field and work down so that the bottom terrace does not get overloaded by a storm that might occur before upslope terraces are completed. While specialized equipment such as bulldozers and scrapers are generally used for terrace construction, ordinary farm machinery can also be used to form broad-base terraces,^[30] and proper tillage operations are needed as part of routine maintenance to preserve functionality of all terraces.^[8]

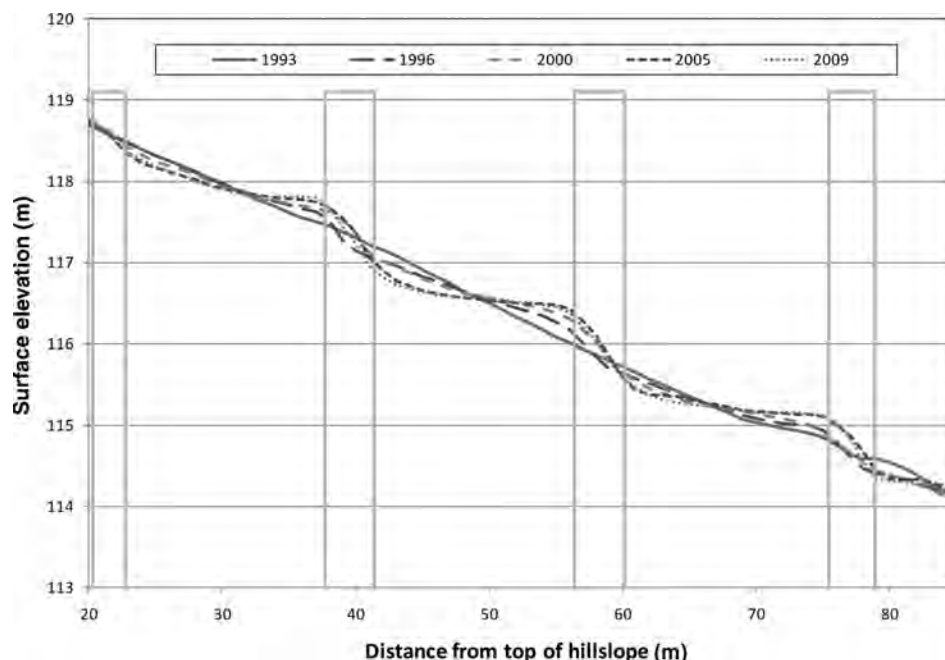


Fig. 1 Patterns of erosion and deposition created by tillage and water erosion and deposition in a field near Coffeeville, Mississippi, U.S.A. Tillage translocation leads to erosion downslope of each grass hedge, creating the negative lynchets, while deposition due to tillage and water transport reinforces the positive lynchets.

Bench Terraces

For deep soils, tillage translocation can be used to reduce the cost and increase the farmability of bench terraces. On irregular fields, starting at the bottom of the slope may facilitate efficient development of straight parallel terrace systems. First, a bulldozer pushes up soil from below into a terrace with extra height where it crosses existing gullies.^[31,32] A tile outlet near the thalweg drains impounded runoff, leaving sediment trapped in the low area. When the thalweg area is sufficiently filled (up to 5 m of fill in 3 years), further sediment is cut off by construction of the next upslope terrace, parallel to the first. In one case, downhill moldboard plowing between terraces accounted for 50% of the soil moved, reducing a slope from 14% to 4% over a 15-year period. Similar to the natural development of lynchets that occurs when tillage is performed between strips of permanent vegetation, bench terraces can be developed by soil moved by tillage operations. Tillage smooths the benches, and the vegetation conserves soil and water while benches gradually develop.

A similar process occurs when fields are tilled between parallel vegetated strips, as described with reference to lynchets. The vegetation conserves soil and water while benches gradually develop. This process has been formalized by the U.S. Department of Agriculture–National Resources Conservation Service (USDA-NRCS) in the vegetative barrier national conservation practice standard.^[33] The vertical interval between vegetated strips should not exceed 2 m, and the planned vegetated back slope should not be steeper than 1 on 2, horizontal to vertical, which is consistent with historical practice.^[8]

Gradient Terraces

Gradient terraces are ridge-and-channel systems that reduce erosion by intercepting surface runoff and redirecting it to a stable outlet at reduced velocities. Tillage adjacent to vegetated strips may create small berms that form when soil is thrown beyond the implement width into and against the vegetation and deposited next to it and within it, later coalescing into a stable ridge. If the tillage boundary remains unchanged, subsequent operations increase the berm height until soil cannot be thrown atop the berm. When the vegetated strip deviates from the contour, these tillage-induced channels can act as gradient terraces and divert an appreciable fraction of runoff.

A microtopography survey was conducted at 28 locations along the upslope edge of several grass strips in a field with 6–9% slope steepness,^[25] farmed with tandem disks and chisel plows. At every location, asymmetrical triangular channels (Fig. 2) formed during seven years of commercial farming. The hydraulic radius of the smallest channel was 0.04 m. At 0.2% grade and Manning's $n = 0.025$, this smallest channel could carry 25 mm/hr of runoff from a 15-m wide cropped strip more than 100 m long. Since the

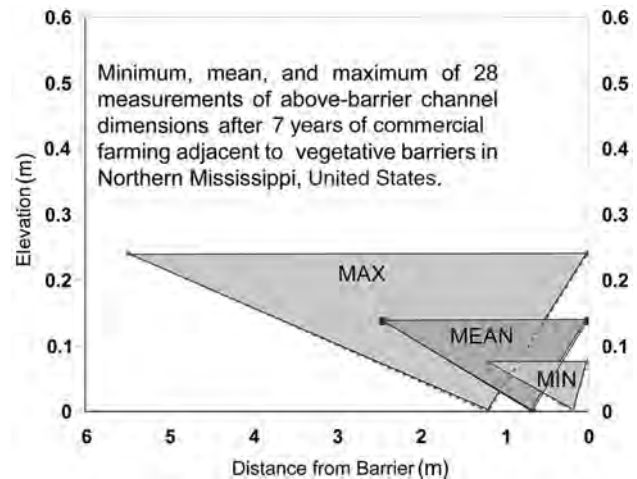


Fig. 2 Typical channel cross sections observed after several years of tillage with disks and chisel plows. When vegetated strips deviate slightly from the contour, small channels can act as small gradient terraces and divert an appreciable fraction of the runoff.

distance between local depressions in the field is usually less than 200 m, these tillage-induced channels could divert most of the fields' runoff to these depressions, where a stable outlet could be provided. Reducing downslope runoff accumulation interrupts slope length and reduces soil erosion by water.

A more detailed study showed that when tillage was conducted with tandem disks and chisel plows immediately adjacent to the vegetation, berms quickly formed along grass hedges planted close to the contour at the bottom of experimental plots with 22.1 m long, 5% slopes.^[27] Four years after tillage adjacent to hedges had begun, berms of triangular shape attained an average height of 0.13 m (Fig. 3), forming a shallow channel in the upslope side of the vegetation that diverted runoff. Observations indicated when no soil berm was present, most (>95%) runoff passed through grass hedges planted on 0.3% gradient from a true contour. Once tillage berms were formed, however, runoff from small storms no longer crossed the vegetation but was diverted to flow along the upslope side of the berm and grass hedge. The resulting berms diverted more than 70% of runoff from runoff events less than 10 mm/day and more than 55% for events less than 80 mm/day.^[34]

Tillage berms may or may not form, depending on the design and adjustment of tillage tools, which affect how soil is displaced to the sides. When soil is thrown beyond the implement's width, a line of deposition is created; if soil is moved away from the boundary, a dead furrow is formed. The presence of a grass hedge or other fixed boundary ensures that the position of the first pass near the boundary is maintained, facilitating the formation of berms or furrows. Tillage implement type and speed, slope steepness, and the superposition of sequential tillage operations determine the formation of berms and their dimensions. On flat lands, the creation of dead furrows

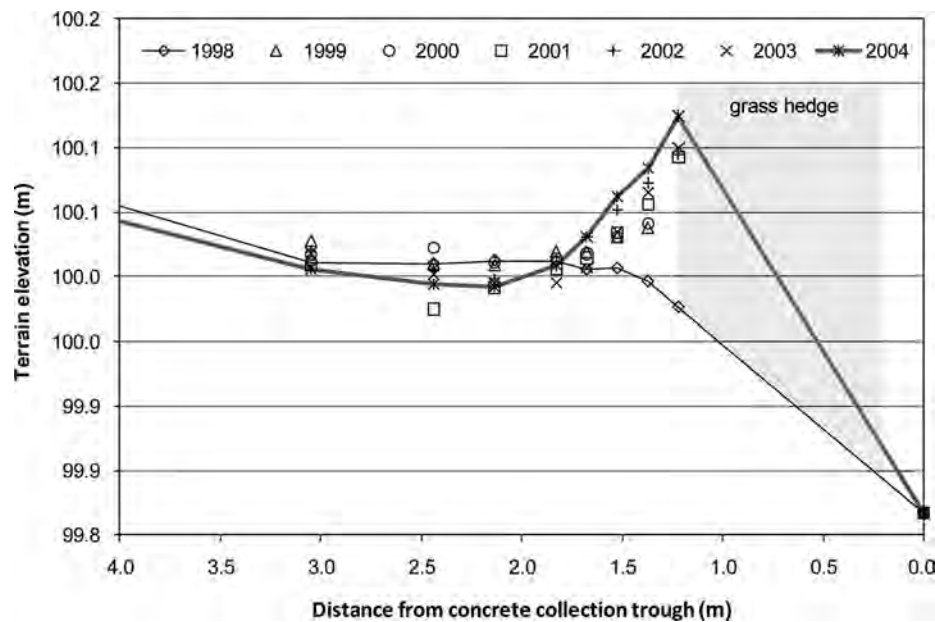


Fig. 3 Evolution of topography in the lowest 4 m of research plots with grass hedges and an average contributing area steepness of 5%. Chisel and disk tillage was kept a distance away from hedges from 1996 to 2000 and was moved adjacent to the hedges from 2001 to 2004.

Source: From Vieira & Dabney.^[27]

and ridges associated with the layout of lands used to till the fields makes the tillage patterns clearly visible in light detection and ranging topographic data^[35] and can greatly alter surface hydrology.

TERRACE BENEFITS AND PROBLEMS

Bench terraces facilitate contour farming operations. On very steep slopes, mechanical cultivating is difficult, and contour plowing can be dangerous due to the risk of overturning. The formation of bench terraces reduces the slope steepness and therefore makes contour tillage easier and safer. When tractor or animal power is limited, it may be easier to plow or hoe downhill.^[11,29] However, when sufficient power is available, it is most efficient to plow on the

contour because a uniform load can be matched to a tractor or team of animals.

Contour tillage reduces runoff and soil erosion, thereby conserving plant nutrients and making more water available to grow crops.^[36] On the other hand, large fertility differences can develop between the negative and positive lynchets.^[12,19,24] This reduces uniformity of crop growth where fertilizer and liming amendments are not available. In these situations, farmers often cut soil from the terrace back slope and distribute it on the degraded negative lynchet. If the upper field is owned by a different landowner, an entire terrace may be undermined in this fashion.^[12] Manuring, fertilization with seaweed, and liming with marl were ancient responses to similar problems.^[37]

While tillage berms acting as gradient terraces can improve soil conservation where runoff is diverted to a

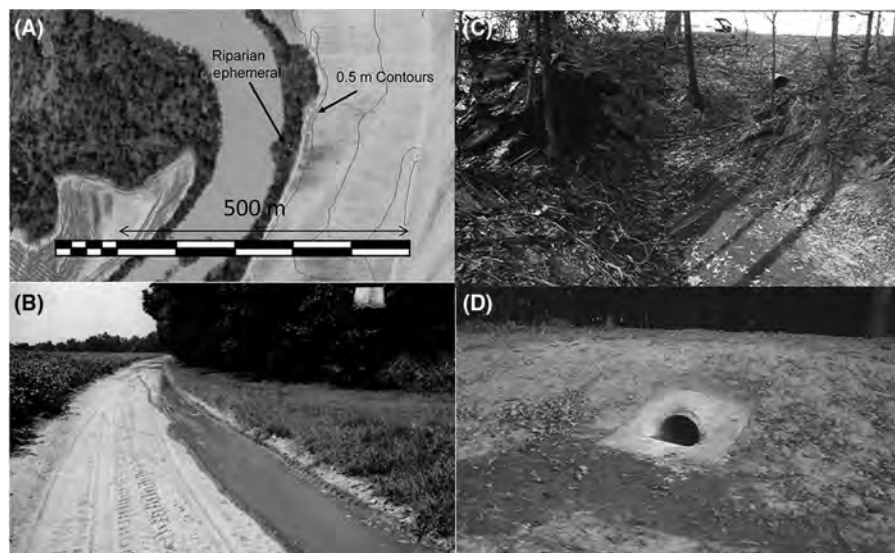


Fig. 4 Runoff reaching (A) a filter strip/riparian forest buffer surrounding an oxbow lake in the Mississippi Delta was (B) redirected by a small tillage berm to (A and C) a riparian ephemeral flow channel. Storm flow through this channel was controlled by (C and D) a slotted-inlet pipe and associated earthen pad, which created backwaters that allowed the riparian buffer to function.

Source: From Dabney, Moore, et al.^[38]

stable outlet, they can have negative consequences on the effectiveness of some conservation practices. For example, if berms keep a significant fraction of runoff water from entering a grassed waterway, erosion on the margin of the waterway may be increased. Tillage berms may keep water from entering a vegetated filter strip on very flat fields (Fig. 4), which may preclude the filter from improving water quality from small runoff flows. This problem is the main reason why the USDA-NRCS restricts the establishment of filter strips to fields with an upslope contributing area steepness greater than 1%.^[39] Tillage berms are also implicitly recognized in the USDA-NRCS contour buffer strip standard^[40] that states, "If sediment accumulates just below the upslope edge of the buffer strip to a depth of 6 inches or more ... relocate the buffer/cropped strip interface location."

CONCLUSION

Tillage erosion has come to be recognized as an important contributor to the evolution of agricultural landscapes and, in some cases, the degradation of soil resources. With each tillage operation, a tilled area becomes flatter, and some soil leaves the tilled zone in the form of clods deposited along field borders. Over time, these clods, along with stones removed from the tilled area and sediment deposited by runoff, coalesce to form terraces or lynchets. If a field is divided into several tilled strips, then the consequence will be the gradual development of bench terraces, unless the untilled areas are periodically relocated. Also, tillage-induced berms located at the upslope edge of the untilled areas may act as small gradient terraces that alter runoff flow patterns. On flat lands, tillage berms may dominate surface drainage patterns. The formation of such bench and gradient terraces due to agricultural tillage operations may or may not be desirable but can hardly be avoided, if field boundaries on sloping lands are fixed for long periods of time.

REFERENCES

1. Lindstrom, M.J.; Nelson, W.W.; Schumacher, T.E.; Lemme, G.D. Soil movement by tillage as affected by slope. *Soil Till. Res.* **1990**, *17* (3–4), 255–264.
2. Lobb, D.A.; Kachanoski, R.G.; Miller, M.H. Tillage translocation and tillage erosion in the complex upland landscapes of southwestern Ontario, Canada. *Soil Till. Res.* **1999**, *51*, 189–209.
3. Van Oost, K.; Govers, G.; de Alba, S.; Quine, T.A. Tillage erosion: A review of controlling factors and implications for soil quality. *Prog. Phys. Geogr.* **2006**, *30* (4), 443–466.
4. Fisher, P.F. Pedogenesis within the archaeological landscape at South Lodge Camp, Wiltshire, England. *Geoderma* **1983**, *29*, 93–105.
5. Fowler, P.J.; Evans, J.G. Plough-marks, lynchets and early fields. *Antiquity* **1967**, *41*, 289–301.
6. Smith, R.T. Early agriculture and soil degradation. In *The Effect of Man on the Landscape: The Highland Zone*; Evans, J.G., Limey, S., Cleere, H., Eds.; CBA Res. Rep. No. 11; Derry and Sons, Ltd.: Nottingham, 1975; 27–37.
7. MacNab, J.W. British strip lynchets. *Antiquity* **1965**, *39*, 279–290.
8. USDA–Soil Conservation Service. *A Manual of Conservation of Soil and Water*; Agriculture Handbook No. 61; U.S. Department of Agriculture: Washington, 1954; 79–107.
9. Papendick, R.I.; Miller, D.E. Conservation tillage in the Pacific Northwest. *J. Soil Water Conserv.* **1977**, *32*, 49–56.
10. Abujamin, S.; Abdurachman, A.S. *Contour Grass Strips as a Low Cost Conservation Practice*. Food and Fertilizer Technology Center Extension Bulletin No. 221; ASPAC: Taipei City, 1985; 1–7.
11. Quine, T.A.; Walling, D.E.; Zhang, X.; Wang, Y. Investigation of soil erosion on terraced fields near Yanting, Sichuan Province, China, using cesium-137. In *Erosion, Debris Flows and Environment in Mountain Regions*, Proceedings of the Chengdu Symposium, Jul 1992; IAHS Publ. no. 209; 155–168.
12. Lewis, L.A. Terracing and accelerated soil loss on Rwandian steeplands: A preliminary investigation of the implications of human activities affecting soil movement. *Land Degrad. Rehabil.* **1992**, *3*, 241–246.
13. Govers, G.; Vandaele, K.; Desmet, P.J.; Poessen, J.; Bunte, K. The role of tillage in soil redistribution on hillslopes. *Eur. J. Soil Sci.* **1994**, *45*, 469–478.
14. Revel, J.C.; Guirresse, M. Erosion due to cultivation of calcareous clay soils on the hillsides of south west France. I. Effect of former farming practices. *Soil Till. Res.* **1995**, *35*, 147–155.
15. Morgan, R.P.C.; Rickson, R.J. Water erosion control. In *Slope Stabilization and Erosion Control: A Bioengineering Approach*; Morgan, R.P.C., Rickson, R.J., Eds.; E & FN Spon: London, 1995; 133–190.
16. Kiepe, P. Cover and barrier effect of *Cassia siamea* hedge-rows on soil conservation in semi-arid Kenya. *Soil Tech.* **1996**, *9*, 161–171.
17. McEntee, M.A. Colluvial processes and soil variation at field boundaries in County Down. *Ir. Geogr.* **1998**, *31* (1), 55–69.
18. Thapa, B.B.; Cassel, D.K.; Garrity, D.P. Ridge tillage and contour natural grass barrier strips reduce tillage erosion. *Soil Till. Res.* **1999**, *51*, 341–356.
19. Turkelboom, F.; Poesen, J.; Ohler, I.; Ongprasert, S. Reassessment of tillage erosion rates by manual tillage on steep slopes in northern Thailand. *Soil Till. Res.* **1999**, *245*–259.
20. Van Oost, K.; Govers, G.; Desmet, P. Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. *Landscape Ecol.* **2000**, *15*, 577–589.
21. Nyssen, J.; Haile, M.; Moeyersons, J.; Poesen, J.; Deckers, J. Soil and water conservation in Tigray (northern Ethiopia): The traditional daget technique and its integration with introduced techniques. *Land Degrad. Dev.* **2000**, *11*, 199–208.
22. McBratney, A.; Minasny, B.; Follain, S.; Walter, C. Simulation of soil thickness evolution in a complex agricultural

- landscape at fine spatial and temporal scales. *Geoderma* **2006**, *133*, 71–86.
23. Dercon, G.; Govers, G.; Poesen, J.; Sánchez, H.; Rombaut, K.; Vandenbroeck, E.; Loaiza, G.; Deckers, J. Animal-powered tillage erosion assessment in the southern Andes region of Ecuador. *Geomorphology* **2007**, *87*, 4–15.
 24. Dercon, G.; Deckers, J.; Govers, G.; Poesen, J.; Sánchez, H.; Vanegas, R.; Ramírez, M.; Tacuri, E.; Loaiza, G. Spatial variability in crop response under contour hedgerow systems in the Andes region of Ecuador. *Soil Till. Res.* **2006**, *86*, 15–26.
 25. Dabney, S.M.; McGregor, K.C.; Meyer, L.D.; Grissinger, E.H.; Foster, G.R. Vegetative barriers for runoff and sediment control. In *Integrated Resources Management and Landscape Modification for Environmental Protection*; Mitchell, J.K., Ed.; ASAE: St. Joseph, 1993; 60–70.
 26. Dabney, S.M.; Liu, Z.; Lane, M.; Douglas, J.; Zhu, J.; Flanagan, D.C. Landscape benching from tillage erosion between grass hedges. *Soil Till. Res.* **1999**, *51*, 219–231.
 27. Vieira, D.A.N.; Dabney, S.M. Modeling edge effects of tillage erosion. *Soil Till. Res.* **2011**, *111*, 197–207.
 28. Meyer, L.D.; Kramer, L.A. Erosion equations predict land slope development. *Agric. Eng.* **1969**, *50*, 552–523.
 29. Poesen, J.; van Wesemael, B.; Govers, G.; Martinez-Fernandez, J.; Desmet, P.; Vandaele, K.; Quine, T.; Degraer, G. Patterns of rock fragment cover generated by tillage erosion. *Geomorphology* **1997**, *18*, 183–197.
 30. Samsel, L.G. Building and maintaining terraces with ordinary farm machinery. *Agric. Eng.* **1943**, *24*, 337–338, 342.
 31. Jacobson, P. New methods of bench terracing steep slopes. *T. ASAE* **1963**, *6*, 257–258, 261.
 32. Jacobson, P. Improvements in bench terracing. In *the Proceedings of the 60th Annual Meeting of the ASAE, Saskatoon*, Saskatchewan, Canada, Jun 27–30, American Society of Agricultural Engineering: St. Joseph, 1967; 1–11.
 33. USDA-NRCS. Vegetative Barrier, Code 601, 2010. Available at http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs143_026353.pdf (accessed August 25, 2015).
 34. Dabney, S.M.; Wilson, G.V.; McGregor, K.C.; Vieira, D.A. N. Runoff through and upslope of contour switchgrass hedges. *Soil Sci. Soc. Am. J.* **2012**, *76*, 210–219.
 35. Li, S.; Lobb, D.A.; Tiessen, K.H.D. Modeling tillage-induced morphological features in cultivated landscapes. *Soil Till. Res.* **2009**, *103*, 33–45.
 36. Wadsworth, R.; Swetnam, R. Modeling the impact of climate warming on the landscape scale: Will bench terraces become economically and ecologically viable structures under changed climate? *Agric. Ecosyst. Environ.* **1998**, *68*, 27–39.
 37. Bell, M. The prehistory of soil erosion. In *Past and Present Soil Erosion: Archaeological and Geographical Perspectives*; Bell, M., Boardman, J., Eds.; Oxbow Monograph 22; Oxbow Books: Oxford, 1992; 21–35.
 38. Dabney, S.M.; Moore, M.T.; Locke, M.A. Integrated management of in-field, edge-of-field, and after-field buffers. *J. Am. Water Resour. Assoc.* **2006**, *42* (1), 15–24.
 39. USDA-NRCS. Filter Strip, Code 393, 2010. Available at http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs144p2_024456.pdf (accessed August 25, 2015).
 40. USDA-NRCS. Contour Buffer Strips, Code 332, 2010. Available at http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_006620.pdf (accessed August 25, 2015).

Tillage: Gas Exchange

Donald C. Reicosky

North Central Soil Conservation Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Morris, Minnesota, U.S.A.

Abstract

Concern for soil productivity and greenhouse gas emissions requires new knowledge to minimize agriculture's impact on the environment. Soil carbon is the foundation of a healthy environment and sustainable agriculture. This is highly dependent on management decisions that influence the intensity of tillage and the amount and placement of residues. Conservation tillage or no-till systems have shown increases in soil organic matter within 10 to 12 years of consistent use. The increase in soil organic matter depends on a delicate balance between the residue inputs of the previous crops and the tillage intensity associated with establishing the next crop. Farmers are faced with serious decisions with respect to environmental consequences of maintaining sustainable production and managing this delicate balance.

INTRODUCTION

Agriculture is the economic foundation of rural America and has a major influence on components of industry, world trade, and global ecology. In traditional agricultural production, tillage of the soil has been an integral part of the production process. Tillage is the mechanical manipulation of soil and crop residue to prepare a seedbed where crop seeds are planted, sprout, take root, and grow into plants to produce grain. Intensive tillage loosens soil, buries crop residue, enables the soil to warm and dry, enhances the release of soil nutrients for crop growth, kills the weeds that compete with crop plants for water and nutrients, facilitates root growth in compacted soil, and improves the flow of water and air within the soil. The enhanced gas exchange affects the processes that impact the accumulation and loss of soil carbon (C) in agricultural systems. Tillage increases water infiltration and increases the soil porosity, especially large pores, which allows greater movement of soil gases through the soil. Diffusion allows movement of gases into or out of the soil from higher to lower concentrations.

Concern for environmental quality and tillage-induced greenhouse gas emissions [carbon dioxide (CO₂), methane, nitrous oxide] requires new knowledge to minimize agriculture's impact on the environment. In the United States, the moldboard plow has been a significant symbol of agriculture over the past years and is being reevaluated in many parts of the world as new conservation tillage techniques are developed and researched.

CROP PRODUCTION AND SOIL C LOSS

The link between global warming and atmospheric CO₂, a greenhouse gas, has heightened interest in soil C storage, as

soil organic matter, in agricultural production systems. Agricultural soils and agricultural production play an important role in C sequestration or storage and thus can help mitigate global warming.^[1] Intensive tillage has mineralized or oxidized 30% and 50% of the native soil C or soil organic matter since the pioneers brought the soils into cultivation. Tillage processes and mechanisms leading to C loss are directly linked to soil productivity, soil properties, and environmental issues.^[2] Soil C dynamics can have an indirect effect on climate change through net absorption or release of CO₂ from soil to the atmosphere in the natural C cycle. In agriculture, C comes into the system through photosynthesis and is returned to the atmosphere as CO₂ through human and microbial respiration. Good soil C management is vital because of its role in maintaining soil fertility, physical properties, and biological activity required for food production. Good soil C management is also needed to partially offset greenhouse gas emissions from manufacture and use of acid fertilizers, liming, fossil fuels, and the release of more potent nitrous oxide and methane from agricultural systems. Minimizing agriculture's impact on the global increase of CO₂ requires that we sequester and maintain high C levels in soil.

TILLAGE-INDUCED CO₂ LOSS

Tillage affects soil microbial activity, organic matter decomposition, and soil C loss in agricultural systems. Much of the C is lost as CO₂, which is the end product of microbial feeding on soil organic matter. Reicosky and Lindstrom^[3] showed major short-term gaseous loss of C, immediately after tillage, which partially explains long-term C loss from tilled soils. Gas exchange was measured using a large, portable chamber to determine CO₂ loss from

various types of tillage. Moldboard plow was the most intensive tillage and caused more CO₂ loss than less intensive tillage methods. No till or no soil disturbance lost the least amount of CO₂, suggesting minimal environmental impact. Moldboard plowing loosens and inverts soil and allows rapid CO₂ loss and oxygen entry. It also incorporates and mixes residues to enhance microbial decomposition and respiration (oxidation).^[4] Stirring the soil in tillage is analogous to stirring the coals in a fire. Plowing accelerates microbial decomposition and soil aggregate breakdown to cause decreased soil C content in the surface layer. Ellert and Janzen^[5] and Rochette and Angers^[6] found similar results for different soils and less intensive tillage methods.

Reicosky^[7] reported that average short-term CO₂ loss from four conservation tillage tools was 31% of the CO₂ from the moldboard plow. The moldboard plow lost 13.8 times more CO₂ as the soil not tilled, while conservation tillage tools averaged about 4.3 times more CO₂ loss. The smaller CO₂ loss from conservation tillage tools was significant and suggests progress in equipment development for enhanced soil C management. Conservation tillage reduces the extent, frequency, and magnitude of mechanical disturbance caused by the moldboard plow and reduces the large air-filled pores or holes in the soil to slow the rate of gas exchange and C oxidation.

Strip tillage tools are designed to minimize soil disturbance. Different strip tillage tools and moldboard plow were compared to quantify short-term tillage-induced CO₂ loss relative to tillage intensity.^[8] Less intensive strip tillage reduced soil CO₂ losses. No till had the lowest CO₂ loss, and moldboard plow had the highest immediately after tillage. Forms of strip tillage had an initial soil CO₂ loss related to tillage intensity intermediate between the extremes of plowing and no till. The cumulative CO₂ losses for 24 hours were directly related to the soil volume disturbed by the tillage tool. Reducing the volume of soil disturbed by tillage should enhance soil and air quality by increasing the soil C content and suggests that soil and environmental benefits of strip tillage be considered in soil management decisions. The CO₂ released immediately after moldboard plowing suggests little C sequestration. Conservation tillage methods that leave most of the crop residue on the surface with limited soil contact yield better C sequestration to enhance environmental quality.

MECHANICS OF GAS EXCHANGE

Tillage affects all physical soil conditions, especially aeration.^[9] Tillage can drastically change the configuration, continuity, and size of soil pores. The moldboard plow is probably the most efficient implement to loosen a large volume of soil and to break up dense, massive soil clods into smaller units. After plowing, all plants and plant residues are buried and partially mixed into the soil, simplifying subsequent tillage and planting operations.

Several tillage-related factors affect soil gas exchange, especially the soil porosity and air permeability. The exchange of air between the soil and the atmosphere is bidirectional and can occur by two different mechanisms called diffusion and convection. In diffusion, the moving force is the gradient of partial pressure or concentration of the specific gas that causes the unevenly distributed molecules to randomly migrate from a zone of high concentration to low concentration. In convection, also called mass flow, the moving force consists of a gradient of total gas pressure and results in the entire mass of air streaming from a zone of high pressure to a zone of low pressure. Barometric pressure changes, soil temperature gradients, and wind gusts over the loosened soil surface can create pressure differences between soil air and the external atmosphere, thereby inducing convective flow into or out of the soil. Whether diffusion or mass flow is the dominant gas flow mechanism from soil depends on the total pressure gradient and pore size and pore continuity. When the soil is consolidated with only small pores, gas exchange can occur primarily by diffusion. When the soil has large pores, gas exchange can also occur still by mass flow. Out in the field, both processes are occurring simultaneously.

The degree to which air pressure fluctuations and convective flow can exchange gas between soil and the atmosphere has long been debated among soil physicists. Most believe that diffusion, rather than convection, is the more important gas exchange mechanism. Available evidence suggests that convection can, in certain circumstances following intensive tillage, contribute significantly to gas exchange and soil aeration, particularly at shallow depths and in soils with large pores.^[10] The tillage-induced CO₂ loss demonstrates the role of mass flow as a cause of C loss from tilled soils. The magnitudes of CO₂ fluxes were too large to be accounted for by simple diffusion from the soil.^[3] The tillage-induced change in soil air permeability showed that convection contributes significantly to gas exchange.^[11]

While the effects of mass flow are intermittent and variable, they tend to be particularly significant immediately after a tillage event up to the time that the soil reconsolidates. External factors that cause soil reconsolidation may include secondary tillage, raindrop impact, or wheel track compaction. The real concern follows an intensive tillage operation, where the change in the soil physical properties increases soil air permeability and changes the gaseous loss from a diffusion-controlled process to a convectively controlled process. Methods for measuring change in soil air permeability on large scales have not been developed.

SOIL PRODUCTIVITY AND ENVIRONMENTAL BENEFITS

While moldboard plowing and other forms of intensive tillage have done much to increase U.S. crop production

over the past years, the increase in production has been accompanied by unseen costs of decreased soil quality from erosion and increased greenhouse gas emissions.^[1,12] The organic matter of many of the prairie's soils has declined from that present under virgin conditions. The unseen, unmeasured costs that result from intensive tillage include loss in soil C due to enhanced oxidation and depletion of soil fertility reserves. The magnitude of these unseen costs depends primarily on the intensity of tillage, the quantity and quality of crop residue returned to the soil, and the crop rotation. Intensive tillage, primarily moldboard plowing, decreases soil C in virtually all crop production systems.

CONCLUSION

Concern for soil productivity and greenhouse gas emissions requires new knowledge to minimize agriculture's impact on the environment. Soil C is the foundation of a healthy environment and sustainable agriculture. This is highly dependent on management decisions that influence intensity of tillage and the amount and placement of residues. Conservation tillage or no-till systems have shown increases in soil organic matter within 10 to 12 years of consistent use. The increase in soil organic matter depends on a delicate balance between the residue inputs of the previous crops and the tillage intensity associated with establishing the next crop. Farmers are faced with serious decisions with respect to environmental consequences of maintaining sustainable production and managing this delicate balance.

REFERENCES

1. Lal, R.; Kimble, J.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Sleeping Bear Press: Ann Arbor, 1998; 128 pp.
2. Paustian, K.; Collins, H.P.; Paul, E.A. Management controls on soil carbon. In *Soil Organic Matter and Temperate Ecosystems: Long-Term Experiments in North America*; Paul, E.A., Paustian, K., Elliot, E.T., Cole, C.V., Eds.; CRC Press: Boca Raton, 1997; 15–49.
3. Reicosky, D.C.; Lindstrom, M.J. Impact of fall tillage and short-term carbon dioxide flux. In *Soil and Global Change*; Lal, R., Kimble, J., Levine, E., Stewart, B.A., Eds.; Lewis Publishers: Chelsea, 1995; 177–187.
4. Reicosky, D.C.; Kemper, W.D.; Langdale, G.W.; Douglas, C.L., Jr.; Rasmussen, P.E. Soil organic matter changes resulting from tillage and biomass production. *J. Soil Water Conserv.* **1995**, *50* (3), 253–261.
5. Ellert, B.H.; Janzen, H.H. Short-term influence of tillage on CO₂ fluxes from a semi-arid soil on the Canadian prairies. *Soil Till. Res.* **1999**, *50*, 21–32.
6. Rochette, P.; Angers, D.A. Soil surface carbon dioxide fluxes induced by spring, summer, and fall moldboard plowing in a sandy loam. *Soil Sci. Soc. Am. J.* **1999**, *63*, 621–628.
7. Reicosky, D.C. Tillage-induced CO₂ emissions from soil. *Nutr. Cycl. Agroecosys.* **1997**, *49*, 273–285.
8. Reicosky, D.C. Strip tillage methods: Impact on soil and air quality. In *Proceedings of the Australian Society of Soil Science Incorporated National Soils Conference*, Brisbane, Apr 27–30, 1998; Australian Society of Soil Science Inc.: Brisbane, 1998; 56–60.
9. Erickson, A.E. Tillage effects on soil aeration. In *Predicting Tillage Affects on Soil Physical Properties and Processes*; Unger, P.W., Van Doren, D.M., Eds.; ASA Special Publication No. 44; ASA: Madison, 1982; 91–104.
10. Renault, P.; Mohrath, D.; Gaudu, J.C.; Fumanal, J.C. Air pressure fluctuations in a prairie soil. *Soil Sci. Soc. Am. J.* **1998**, *62*, 553–563.
11. Reicosky, D.C.; Lindstrom, M.J. Fall tillage methods: effect on short-term carbon dioxide flux from soil. *Agron. J.* **1993**, *85* (6), 1237–1243.
12. Schlesinger, W.H. Changes in soil carbon storage and associated properties with disturbance and recovery. In *The Changing Carbon Cycle: A Global Analysis*; Trabalha, J.R., Reichle, D.E., Eds.; Springer-Verlag: New York, 1985; 194–220.

Tilth

Jerry L. Hatfield

National Laboratory for Agriculture and the Environment, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.

Abstract

Soil tilth has and continues to be an interesting term. The term intrigues people because of its connection with the soil and confuses people because of their inability to provide an exact definition or measurement of the concept. As a term that describes a soil property, it is better visualized than quantified; however, the biological, chemical, and physical principles that underlie tilth are fundamental to a quality soil resource. Soil tilth has potential agronomic, biological, and engineering aspects that further increase the richness of being able to understand the dynamics of the soil.

Tilth embodies the fundamental understanding of soil structure changes in response to tillage, organic matter additions, and crop rotations. Soils will not respond the same to different inputs and management practices because of fundamental differences among soils in their basic composition of sand, silt, and clay. Improving tilth will benefit the water exchange, gas exchange, nutrient cycling, and ability of a plant to explore the profile, support the biodiversity of biological system in the soil, and provide the structural support for land management operations. The dependence of humankind on the soil as a source of food requires that we begin to understand how to manage our soils for their long-term improvement and enhancement.

INTRODUCTION

Tilth describes the physical attributes of the soil but is more than a description of the physical properties. Tilth is more of a description of the properties that describe its ability to support plant growth, biological processes, and exchange processes and provides a stable base for engineering applications. This entry explores the history of the concept of tilth and its measurement, the role of tilth in the functions of the soil, and the potential for the improvement of tilth. Understanding these processes will help us understand the value of soil as a critical resource for the future of humankind.

HISTORY OF TILTH

One of the first references to tilth was provided by Keen.^[1] He provided a summary of previous observations regarding the importance of tilth for the planting of crops into the proper seedbed, the impact of frost on the soil, how hard rains after plowing would create a crust, and how grazing animals could compact the soil. It is interesting that through his observations, he gleaned information that described the biological, chemical, and physical responses of soil to various forces. Keen's treatise on the physical conditions of the soil was followed by a paper by Yoder^[2] who described soil tilth as the soil conditions that determined the "fitness" of the soil for the growth and development of crop plants. The cornerstone of his definition of tilth was based on soil structure. Yoder^[2] defined the ideal tilth as offering minimum resistance to root penetration; allowing for good infiltration

and storage of soil water; having a good exchange of gases between the soil and the atmosphere with adequate air supply in the soil; having the proper balance of water and air in the pore space of the soil; resisting erosion; promoting microbiological activity; facilitating the placement of green manure crops and organic residues; and providing a stable foundation for farm implements. This is a very inclusive list that covers all aspects of the functions of soil but more importantly demonstrates the potential interaction among the physical, biological, and chemical processes that are occurring within the soil. Yoder recognized and emphasized the value of green manure crops and organic residue addition to the soil, as one of the key components in the development of tilth. There were other very early observations that soil organic matter (carbon) in the upper soil surface declined as a result of tillage operations. Whiteside and Smith^[3] reviewed the literature and found tilled soils had less organic matter than untilled grasslands or forest areas. The change in organic matter in soils is shown in Fig. 1 and shows that the state of organic matter content is near steady state. There are efforts underway to potentially increase organic matter in soils through the use of conservation tillage systems coupled with manure. These practices are based on the observations that were detailed in how to increase the tilth of the soil in the 1960s.

Yoder^[2] also made another interesting observation in which he found that poor soil tilth was associated with soil structure because factors that cause problems with stable soil structure were the same as those that caused problems with tilth. Soil structure is a critical component of soil tilth and cannot be overlooked in the process of how to

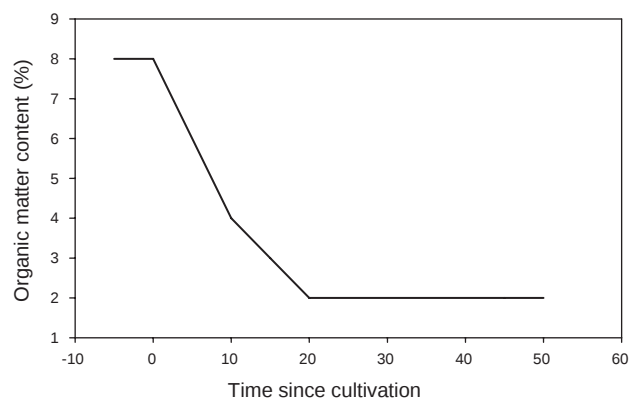


Fig. 1 Schematic representation of changes in soil organic matter as native soil is subjected to cultivation.

improve soil tilth. Changes in soil structure and the impact on soil tilth are probably most quickly recognized by gardeners because they work their soil either by hand or with small tools where the effect of improving soil tilth is readily apparent, and hard soil requires more energy to prepare and produces less vigorous plants.

IMPROVEMENT OF SOIL TILTH

Karlen et al.^[4] summarized the literature on soil tilth and proposed several areas for potential research on soil tilth. They suggested that an examination be made of the tilth-forming processes as a combination of physical, chemical, and biological processes that interact within the soil. They proposed a definition of tilth as “The physical condition of a soil described by its bulk density, porosity, structure, roughness, and aggregate characteristics as related to water, nutrient, heat, and air transport; stimulation of microbial and microfauna populations and processes; and impedance to seedling emergence and root penetration.” This is an inclusive definition that illustrates the major factors affecting soil tilth; however, the impact of these different factors can only be realized as the result of complex interactions between the basic foundation of soil, i.e., sand, silt, and clay particles, and the biological, chemical, and physical forces that create a stable soil aggregate. Creating soil tilth requires time because these processes carry the additional element of time for them to exert their impact on the soil. There is a large hysteresis effect in that it takes little time to degrade soil but a much longer time to enhance soil. Another dimension to this problem is that not all soils will respond the same to the addition of organic material through either plant roots or manures and there needs to be an increased understanding of how cropping systems interact with tillage across soils and climates to quantify the impacts on tilth. Manure and organic amendments remain the most effective additions to soil to begin the pathway toward enhancing the soil.

ROLE OF SOIL TILTH

Good soil tilth provides a stable base for agricultural production. The ability of a plant to explore the soil profile to extract water, nutrients, and oxygen reduces a potential limitation to plant growth. A limited soil profile equates to a limited amount of plant growth and less vigorous plant. The vast majority of the world’s agriculture depends upon rainfall to produce a reliable supply of food, feed, or fiber. When the soil has an unstable structure that easily crusts, infiltration of rainwater into the soil is limited. This is critical when rainfall occurs in intense storms. The other limitation comes when soil water-holding capacity is diminished by reductions in organic matter and the rainfall leaches through the profile and is not available to the plant. The simple change in the soil creates a limitation to plant growth and yield because of reduced water availability. Fields with poor soils often show poor plant growth because of water limitations even in years that have normal rainfall for that location. Efficient use of stored soil water is critical to providing food, feed, and fiber for the world’s population, and the process begins with a soil that has good tilth. Soil with good tilth has improved nutrient cycling because of the role of soil organic matter nurturing microbiological activity within the soil and supplying nutrients to plants is as critical as water for sustaining plant productivity.

There are also indirect effects of soil tilth on plant production systems that are often ignored. Healthy soils with good tilth have fewer pressures on plants from insects and diseases. This aspect has not been fully studied but offers an additional piece of evidence for the proper management of the soil resource.

It is critical to consider soil tilth as part of the ecosystem that contributes to improved ecosystem health and ecosystem services, e.g., water quality, air quality, biodiversity, and efficient use of resources. Soil is a foundation for the existence of humankind; however, good soil tilth increases the value of that foundation to provide a reliable source of food, feed, fiber, and fuel from the soil. Soil is precious, and good tilth increases the value of the soil for all future generations.

REFERENCES

1. Keen, B.A. *The Physical Properties of the Soil*. Rothamsted Monographs on Agricultural Science; Longmans, Green and Co: London, 1931.
2. Yoder, R.E. The significance of soil structure in relation to the tilth problem. *Soil Sci. Soc. Am. Proc.* **1937**, 2, 21–33.
3. Whiteside, E.P.; Smith, R.S. Soil changes associated with tillage and cropping in humid areas of the United States. *Agron. J.* **1941**, 33, 765–777.
4. Karlen, D.L.; Erbach, D.C.; Kaspar, T.C.; Colvin, T.S.; Berry, E.C.; Timmons, D.R. Soil tilth: A review of past perceptions and future needs. *Soil Sci. Soc. Am. J.* **1990**, 54, 153–161.

Time and Space: Soils in

Neil E. Smeck

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

C. Lee Burras

Department of Agronomy, Iowa State University, Ames, Iowa, U.S.A.

Abstract

Soils vary greatly in their characteristics and potential to meet human needs. The most essential function of soil from a human perspective is that soil serves as the reservoir of water and nutrients essential for sustained plant growth. Plants serve as our food, either via direct production of fruit, vegetables, and grain or indirectly as feed for livestock. Soils also serve as the foundation for societal infrastructure and as a medium for the disposal and amelioration of wastes. The regional and global differences in soil characteristics and potentials for use are due to differences in energy fluxes in the various ecosystems of the world. The objective of this entry is to discuss the evolution of soils as a function of time and factors that result in the spatial distribution of the multitude of soils that mantle the earth.

INTRODUCTION

Soil, the relatively thin organic-mineral entity at the earth's surface, forms at the interface between the atmosphere and lithosphere by the assimilation of energy. The sun not only warms the earth's surface which increases the rate of chemical reactions but also drives the photosynthetic production of biomass. Death and decay of biomass result in the accumulation of humus in the upper horizon(s) of soil. Soil formation is also driven by the weathering of minerals in the lithosphere which releases energy utilized in soil development and nutrients that are essential for the growth of plants. Gravitational energy provides the driving force for the infiltration and percolation of water into and through the earth's surface. The magnitude of humus accumulation, mineral weathering, and translocation or leaching of solutes and colloids by water moving through the lithosphere plays a major role in differentiating the great diversity of soils that cover the face of the earth.

SOIL EVOLUTION AND DEGRADATION AS A FUNCTION OF TIME

Soil formation is initiated the moment a portion of the lithosphere is exposed to the atmosphere following major depositional or erosional geologic events. The geologic materials deposited by water, wind, ice, or gravity or exposed by erosion are referred to as "parent material" because they are the precursor of soil. Common parent materials are alluvium, glacial till, glacial

outwash, lacustrine and marine sediments, Loess (wind-blown silt), volcanic ash, pedisegment, colluvium, and residuum (weathered bedrock). Very youthful soils differ little from their parent material but as time progresses, soils with characteristics distinctive of the ecosystems in which they form will evolve (such soils are considered mature soils). The evolution of a youthful to a mature soil proceeds due to additions and losses of materials and translocation and transformation of components within the soil.^[1] Specific processes include the weathering of primary minerals to secondary minerals (clays, salts, and oxides) and soluble ions, leaching of soluble components from the soil, accumulation of organic matter in the soil, and the translocation and accumulation of secondary minerals within the soil. All of these processes contribute to the formation of unique sequences of soil horizons characteristic of the various mature soils.

Young soils exhibit weak expression of horizons, little accumulation of organic matter above that in the geologic parent material, and minimal mineral weathering. Such soils generally exhibit only A and C horizons (lack B horizons) and classify as Entisols in *Soil Taxonomy*.^[2] With additional time, such soils will develop weak B horizons, become more intensely weathered, and accumulate organic matter. These developments will qualify the soils as Inceptisols in *Soil Taxonomy*.^[2] Eventually as weathering and developmental processes continue, a mature soil will form. A mature soil can be defined as a soil that has attained a steady state with the energy fluxes in the ecosystem in which it occurs. This implies that there are sufficient energy influxes from the

ecosystem to maintain the soil in its state for an indefinite time but insufficient energy for additional soil formation. Most of the soil orders in *Soil Taxonomy* represent such steady states. The distribution of these mature orders will be discussed in the second section of this entry.

Rates of soil formation are a function of the ecosystem in which soil formation occurs and the existing stage of soil development. Soil formation occurs more rapidly in high energy ecosystems (tropical) and in youthful, relatively shallow soils. As soil formation progresses toward maturity, increases in soil depth gradually slow and eventually cease because weathering, leaching, and energy inputs are reduced at depth. Buol et al.^[3] list soil formation rates, obtained from the literature, ranging from 0.1 to 750 yr/cm; the former at the surface of a recent mudflow and the latter at a depth of 1 m in an Oxisol. Although rudimentary soils can form in <100 years, thousands of years are necessary for mature soils to form. Because more time is required to acquire the energy necessary for intensely weathered and/or highly developed soils to attain maturity, soil orders can be arrayed by the amount of time required to attain a steady state (Fig. 1). The steady-state condition attained by mature soils necessitates that such soils are being regenerated at the same rate as they are being degraded by weathering or truncated by geologic erosion. Troeh et al.^[4] suggest “a rate of 1 mt/ac yr may be considered typical for geologic erosion from gently sloping soils.” Using this geologic erosion rate, a steady-state condition could be maintained with a rate of soil formation of 100 yr/cm of soil.

Accelerated erosion caused by human activities on the landscape, however, can remove soil at rates that are many times that of the capacity for soil regeneration. Whereas maximum “tolerable” soil loss rates have been established at 11 mt/ac yr by the U.S. Department of Agriculture—Natural Resources Conservation Service (10 times the rate of geologic erosion), accelerated erosion rates exceeding 450 mt/ac yr have been documented.^[4]

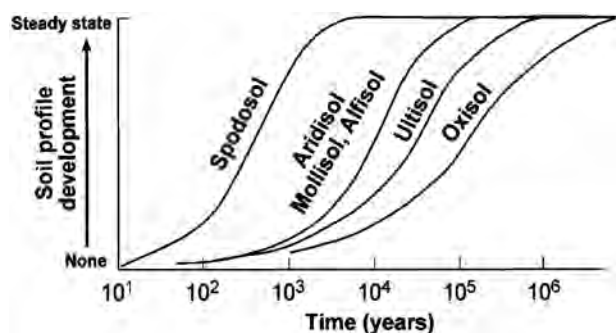


Fig. 1 Schematic diagram showing the variations in time to attain the steady state for various soil orders.

Source: From Birkeland.^[5]

Wischmeier and Smith^[6] defined tolerable soil loss as “the maximum level of soil erosion that will permit a high level of crop productivity to be sustained economically and indefinitely.” It is obvious that mismanagement of soils can result in the loss of soil at a rate greater than that of soil formation.

Reduction of the world’s soil resources poses a serious threat for the longevity of humankind. Accelerated erosion is not the only process leading to soil degradation; compaction, oxidation of organic matter due to tillage and drainage, and salinity due to poorly managed irrigation systems all contribute to soil degradation. On the basis of the rate of soil formation relative to the length of a human lifetime, soil must be considered a non-renewable resource. Soil loss due to accelerated erosion will not be regenerated in our lifetime. Degradation of the quality of soil due to inappropriate management practices, however, can be reversed by the use of best management practices; thus, soil can be considered a renewable resource with respect to soil quality. The most unique attribute of soil is that soil can be used indefinitely without deterioration if managed for sustainability.

SPATIAL DISTRIBUTION OF MATURE SOILS

Soil results from the interactions among climate, vegetation, geological parent material, topography, and time. These five “state factors,” as formulated by Jenny,^[7] determine the nature and distribution of soils over the face of the earth. The relative importance of a given state factor’s influence on soil properties is a function of scale. Climate is generally the most important state factor at a global scale, whereas vegetation is the most important factor at a regional scale. At a local scale, topography and parent material separately or in combination often control the distribution of soils.

Precipitation and temperature are the most important climatic factors controlling the weathering and, ultimately, global distribution of mature soils (Fig. 2). Warm, wet climates generally result in highly weathered soils, whereas dry, cold climates result in minimally weathered soils. Climates between these extremes result in the formation of soils with intermediate degrees of weathering. This relationship can be illustrated by considering a few major climatic areas of the world. Wet tropical regions such as the rainforests of the Amazon Basin of South America and the Congo Basin of Africa are noted for deep, highly weathered, nutrient-poor soils classified as Oxisols in the U.S. soil classification system.^[2] Many Oxisols, however, occur in dry climates as relicts of paleotropical climates. Oxisols are often red in color due to the concentration of iron and aluminum oxides, as other chemical components are released by weathering and leached from the soil. On the opposite end of this spectrum, the polar regions of North America, Europe, and Asia have

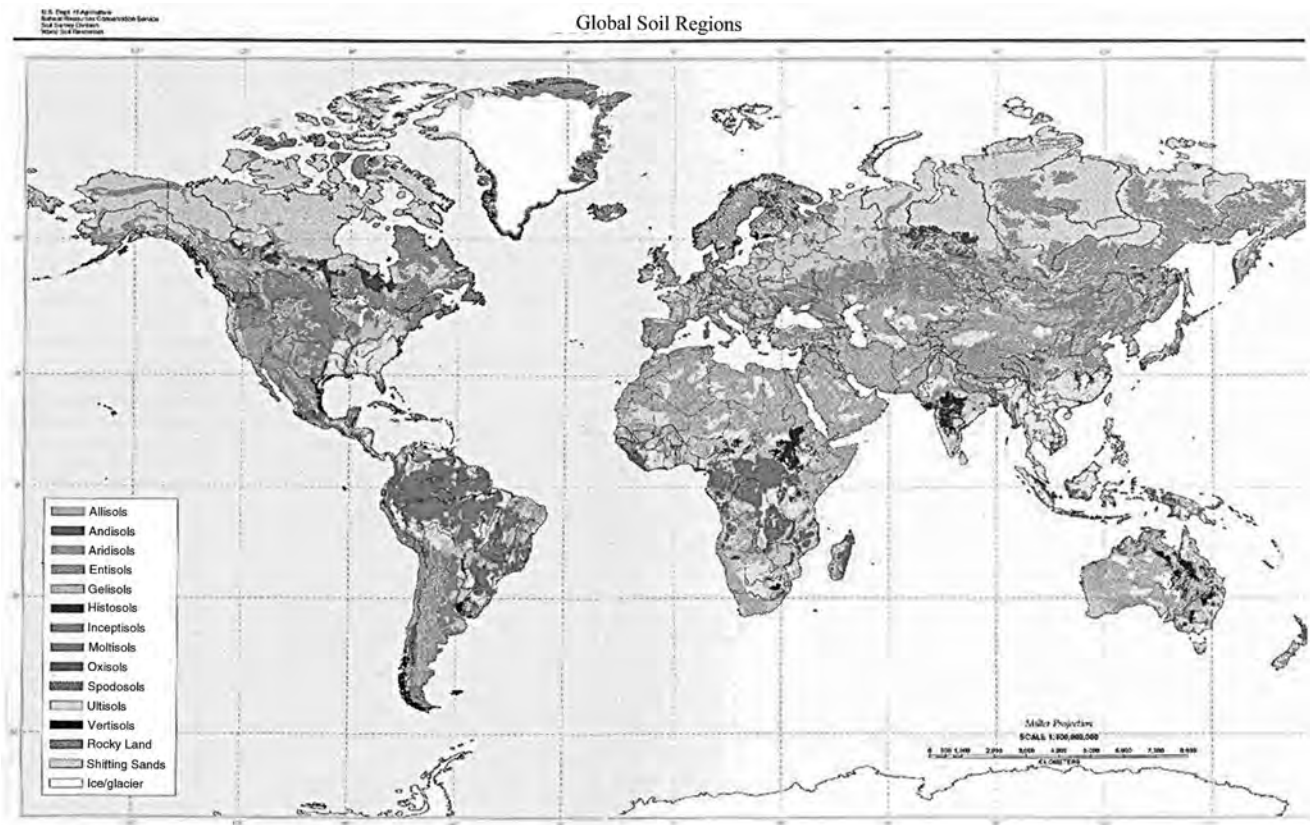


Fig. 2 World soil map.

Source: From USDA–NRCS.^[8]

large areas of Gelisols,^[2] which are permanently frozen soils. Because little weathering occurs when a soil is frozen, Gelisols are generally minimally weathered soils. Both Gelisols and Oxisols very readily degrade due to mismanagement but for contrasting reasons. Gelisols are fragile because they develop and regenerate at such an exceedingly slow rate. Oxisols are so highly weathered that they are depleted of primary minerals and nutrients; thus, there is little internal energy remaining for regeneration.

On progressing from humid-tropical to humid-temperate climates, soils generally become less weathered. The following soil orders can be considered a weathering continuum: Oxisols (tropical)–Ultisols–Alfisols (temperate). Ultisols generally occur in humid, temperate to semitropical climates, but may also occur in wet tropical locations where there has been insufficient time for Oxisols to form. Ultisols are concentrated in Central America, the southeastern quadrant of the United States, Asia, and Africa. The key properties of Ultisols are low-nutrient and nutrient-bearing mineral contents, an A horizon with low organic matter content, and a thick, prominent B horizon. The B horizon is a zone of clay concentration due to both clay illuviation and formation. Alfisols are found in humid-temperate climates (with deciduous forests) that are too cool for

Ultisols to form. These regions include much of eastern North America, Central Europe and Great Britain, and China. Alfisols are more fertile and contain more nutrient-bearing minerals than Ultisols due to less intense weathering. Alfisols are also characterized by clay accumulation in B horizons, but the accumulation is less pronounced than in Ultisols. In addition, the clay accumulation in Alfisols is more dependent on clay illuviation and less dependent on clay formation than that of the B horizons in Ultisols.

Wet and/or cool climates such as in Scotland and Ireland result in the formation of Histosols.^[2] Histosols are organic soils developed in wet, anaerobic locations where plant production of organic matter far exceeds decomposition of the litter produced. Histosols can be tens of meters deep, although they are generally 1–3 m deep. They are commonly referred to as peats and mucks. Historically, these soils have been mined for use as fuel. The arid and semiarid climates of the world result in the presence of either Aridisols or Entisols.^[2] Aridisols and Entisols in arid regions cover nearly 28% of the globe due to extensive dry areas in Africa, North America, South America, Asia, and Australia. Aridisols range from moderately developed soils that are relicts of past more humid climates to weakly developed soils

commonly containing soluble salts due to minimal leaching. Such diverse soils are grouped together as Aridisols primarily due to sufficient moisture for only very limited vegetative growth without irrigation. Entisols, which commonly consist of shifting sands and consequently show very little soil development, dominate the driest portions of the earth's surface. Entisols also occur in other climatic regions on very young or weathering resistant parent materials.

Vertisols^[2] are very clayey soils associated with seasonally wet and dry climates. They are widespread in western India, eastern Australia, and along the northwestern coast of the Gulf of Mexico. During the dry season, Vertisols develop wide, deep cracks that swell shut during the wet season due to the expansive nature of the clays. The swelling pressures in the subsoil cause it to be thrust toward the surface resulting in heaving of the soil surface. Repetition of this cyclic process between seasons results in churning or inversion of the soil.

The distribution of vegetative communities is a function of climate, comparable to the impact of climate on the distribution of soils; however, vegetative communities superimpose unique soil forming processes and characteristics on those imposed by climate. This can be illustrated by considering the distribution of Spodosols, Alfisols, and Mollisols,^[2] which are strikingly different soils. Spodosols occur in ecosystems dominated by conifers (e.g., Maine and other northern New England states, Michigan, Wisconsin, Minnesota, and upper New York state), Alfisols occur in ecosystems dominated by deciduous trees (e.g., New York, Pennsylvania, and Ohio) and Mollisols in prairie ecosystems common throughout the Central United States (e.g., North Dakota to Texas and Illinois to Colorado). Spodosols are generally shallow, sandy, low fertility, and acidic soils characterized by a prominent bleached E horizon over B horizons that have accumulated iron and aluminum oxides and/or humic materials due to the chelation and translocation of these components by compounds released by the coniferous vegetation or decomposition products of the litter. Alfisols are fertile soils characterized by an A horizon with low organic matter content and clayey B horizons. The translocation of clay from near surface horizons to the B horizon is attributed to the high degree of acidity produced by the decomposition of the leaves of deciduous trees. Mollisols are fertile soils characterized by thick, dark, humus-rich A horizons that form in prairie ecosystems. The annual production of a deep, fibrous root system by prairie grasses in conjunction with the calcium-rich chemistry of the soils results in the accumulation of stable humic compounds in the soil to considerable depth.

The distribution of soils on a landscape scale is generally related to the distribution of vegetation, topography, and parent materials or a combination thereof. An example of vegetation determining soil distribution occurs in some Iowa landscapes where soils formed in deciduous forest,

savannah, and prairie ecosystems exist contiguously. Alfisols formed in the forested ecosystems, Mollisols in the prairie ecosystems, and soils with properties intermediate between Alfisols and Mollisols in the savannah ecosystem. Because of the causative relationship between ecosystem biology and soil pattern, such soil distributions are referred to as "biosequences."

The impact of parent materials on the distribution of soils occurs at any scale but is most evident on large scale maps. Sharpsburg, Lindley, Gosport, and Nodaway soil series form an association in the various parent materials that occur in the hilly landscapes of southern Iowa and northern Missouri (Fig. 3). The soils comprising this association are quite different as indicated by their classification into different soil orders: Sharpsburg derived from Loess is a Mollisol, Lindley derived from glacial till is an Alfisol, Gosport derived from shale is an Inceptisol, and Nodaway formed in alluvium is an Entisol. Although differences in age of the four parent materials influence the classification of soils at the order level, the soils derived from the different parent materials would classify as different soil series based solely on properties inherited from the parent materials.

Another parameter influencing soil distributions, especially at the field scale, is the impact of water flow, both surface and subsurface, in the landscape. Much of the precipitation that falls on sloping landscape components is removed by surface runoff, whereas depressional soils accumulate surface run-on and retain all precipitation. Potential water infiltration and percolation through the soils are, therefore, least on sloping landscape components, intermediate on level areas, and greatest in the lowest portions of the landscape. The depth to a water table (zone of saturation in the lithosphere) also varies across landscapes. The shallowest depth to the water table generally occurs in the lowest landscape positions which serve as the primary point for aquifer recharge, whereas, the greatest depths to the water table occurs in the landscape components with the highest elevations (Fig. 4). Due to differential water distribution and depths to water tables within landscapes, soil wetness and drainage classes vary within landscapes.

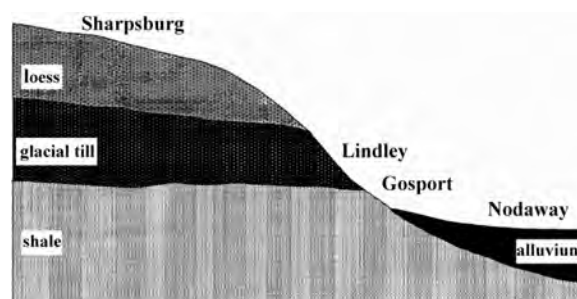


Fig. 3 Cross section of a landscape showing the relationship between soils and geological parent materials from southern Iowa and northern Missouri.

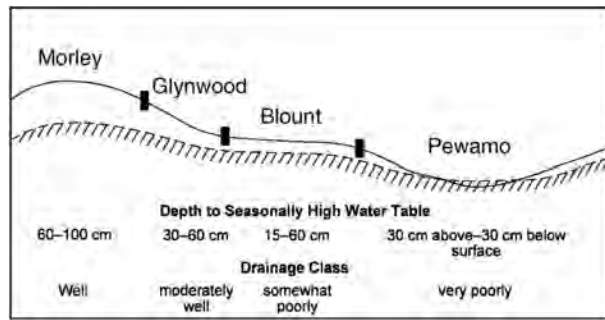


Fig. 4 Cross section of landscape showing relationship between soil distribution and topography, drainage class, and depth to seasonally high water table (undrained condition) in Morley–Glynwood–Blount–Pewamo association of western Ohio.

Wetness and drainage characteristics have a major impact on the suitability of soils for various uses, both agricultural and nonagricultural (construction, sewage disposal, home sites, recreational activities, etc.).

CONCLUSION

In summary, the world's soils are distributed in a highly systematic manner. Soil properties and distribution are predictable with knowledge of the five state factors and their impact on soil formation. Time is required for mature soils, those steady-state entities comprising the uppermost portion of the lithosphere, to evolve from geological parent material. The relative importance of the other four factors

is a function of scale. Climate and vegetation are most important in explaining global soil patterns, whereas parent material, topography, and time are more important at landscape scales.

REFERENCES

1. Simonson, R.W. Outline of a generalized theory of soil genesis. *Soil Sci. Soc. Am. Proc.* **1959**, 23, 152–156.
2. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; Agriculture Handbook No. 436; USDA/Natural Resources Conservation Service: Washington, 1999; 869 pp.
3. Buol, S.W.; Hole, F.D.; McCracken, R.J.; Southard, R.J. Time as a factor of soil formation. In *Soil Genesis and Classification*, 4th Ed.; Iowa State University Press: Ames, 1997; 179–194.
4. Troeh, F.R.; Hobbs, J.A.; Danahue, R.L. *Soil and Water Conservation Productivity and Environmental Protection*, 3rd Ed.; Prentice Hall: Upper Saddle River, 1999; 610 pp.
5. Birkeland, P.W. *Soils and Geomorphology*; Oxford University Press: Oxford, 1984; 372 pp.
6. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses: A Guide to Conservation Planning*; Agriculture Handbook 537; USDA/Agricultural Research Service: Washington, 1978.
7. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941; 281 pp.
8. USDA–NRCS. *World Soil Resources*. Available at <http://www.nhq.nrcs.usda.gov/WSR/mabindex/metadata/Maps/ORDERS.JPG> (Reviewed August 2001).

Torrential Erosion: Himalayas

Ram Prashad Yadav

*Regional Center, National Bureau of Soil Survey and Land Use Planning,
New Delhi, India*

Abstract

Torrential erosion is an extreme case of accelerated erosion wherein a large mass of soil is detached from catchment/hill and is transported rapidly with large volume runoff through rainy seasons streams called torrents and deposited in torrent beds and low lying fields. Though of very short duration during monsoon season, it causes immense damage to life, property and the ecosystem in the Himalayan region of India, Nepal and Bhutan. Intense flash floods in Lower Himalayas (Ropar, Punjab state) in 1988 and Middle Himalayas (Kedarnath, Uttarakhand state) in 2013 are the most glaring examples of natural devastation that caused heavy losses of human lives, livestock and infrastructures beside land and vegetation. Both natural factors (geology, physiography, seismic activities, and torrential rains) and anthropogenic activities (population, deforestation, infrastructure construction, and unscientific cultivation on hill slopes) make the region susceptible to mass erosion. It is possible to delineate mass erosion prone areas based on geo-environmental factors in the Himalayan region. While land hazard zonation is an effective technique in Middle and Upper Himalayas, vulnerable Shivalik/Lower Himalayan regions can demarcated based on a single value stability indicator. Amelioration techniques for treatment of mass eroded areas are very costly and require large investments for vast areas. A three-pronged integrated participatory approach comprising of catchment area treatment, torrent control measures, and rehabilitation of torrent beds and adjoining fields has proved effective for sustainable management of the fragile ecosystem. Central and state governments have implemented many projects with internal resources and external funding to check the menace of torrential erosion.

INTRODUCTION

Torrents are mountain streams that flow with high velocity and are flashy, usually carrying large size bed load particles. They transport heavy loads of sediments derived from landslides, landslips, or other unstable areas. Typically, torrents occur in the valleys of steep, small catchments or watersheds that experience periods of torrential rainfall and are susceptible to extensive and rapid soil erosion from the landscape. In Nepal, these are described as debris torrents as they involve rapid movement of water charged soils, rocks, and organic materials down steep slopes. In India, torrents have various local names and are known as “*choes*” in Punjab and Haryana, “*kholas*” in Himachal Pradesh and Nepal, “*Jhora*” in Sikkim and West Bengal, and “*Dong/Jhora*” in Assam and Bhutan.

In India, torrents have been taken as small mountainous rainy season streams rushing down the slope with flashy floods, often loaded with sediments. Typically, perennial streams fed by the melting of snow are not considered torrents. Flash floods in the torrents during peak storms in the rainy season (four to five times in a year) cause severe problems of land erosion and floods in the command area. The floodwater, about a meter or so in depth, comes

all of a sudden, and though it lasts for 2–3 hours only, causes immense damage to land, vegetation, and infrastructures.

TORRENT VS. TORRENTIAL EROSION

Though torrents and torrential erosion are two different terms, they are often but erroneously used synonymously. Torrential erosion describes the complete process of erosion (detachment, transportation, and deposition) and includes all the areas affected by erosion in catchments, torrents, and land inundated by flash floods. Torrential erosion is an extreme case of accelerated erosion wherein a large mass of soil is detached from hills and a large volume of runoff laden with debris rushes down the hills and spreads in the plains leading to the formation of torrents. Thus, torrents are the valley streams that transport eroded materials from a fragile watershed that experience mass erosion through landslides, landslips, and land sledging during torrential rains. During transportation, these cause problems of channel flow, deposition, meandering/shifting of course, inundation of bank, etc., in their course. The genesis of torrents, damage caused by them, and their control measures have been documented by various

organizations like Food and Agriculture Organization^[1,2] and researchers.^[2,3] Aggradation and degradation in torrents is therefore just a part of torrential erosion. Thus, the taming of torrents in isolation without treating catchments is a futile exercise.

EXTENT OF EROSION IN HIMALAYAN REGION

The Himalayas extend from the Indus in the west to the Brahmaputra in the east. They form an arc between two extremes, covering a distance of 2500 km. From east to west, the width of the Himalayas varies from 150 to 400 km. The northern most ranges of the Himalayas have the highest peaks and are known as Greater Himalayas or Upper Himalayas. To their south lies the Middle Himalayas of intermediate heights. The southern most ranges are known as the Outer or Lower Himalayas and commonly referred to as the Shivalik ranges/hills. These occupy significant portion of Nepal, Bhutan, and the Indian states of Assam and West Bengal in the east and Jammu and Kashmir, Himachal Pradesh, Punjab, Haryana, and Uttarakhand in the west.

A major part of Greater and Middle Himalayas is either covered with snow or devoid of any soil (rock outcrops). The Shivalik Hills (Outer Himalayas) are composed of various strata of loose conglomerates beds, friable sandstone, siltstone, and claystone. In these regions, few showers of an intensity higher than 12.5 mm hr^{-1} account for 85% of the rainfall received during the four monsoon months (June to September), the main rainy season. The steep unstable slopes, torrential rains, fragile geology, and rapid deforestation make the Shivalik ecosystem most susceptible to torrential erosion.

Precise information about the extent of torrential erosion for the entire Himalayan region is not available. The National Commission of Agriculture compiled available information and obtained an estimate of 2.73 Mha in India.

GENESIS OF TORRENTIAL EROSION

Removal of protective vegetative cover favors gully formation, which expand toward the head horizontally and get deeper because of rapid movement of runoff water in high slopes. The deep gullies with steep side slopes promote mass erosion through landslides, landslips, deepening of channel bed, and bank caving in the fragile hills. Faulty construction of roads, railway tracks, building, and canals or uncontrolled mining also causes unstable hill slopes, which result in landslides. Thus, excessive debris loads are carried to hill torrents down below. Torrent development is closely related with landslides in the catchments. In Nepal, the watersheds producing torrents experience landslides produce an average soil loss of $38.3 \text{ Mg ha}^{-1} \text{ yr}^{-1}$. On the contrary, watersheds covered

with dense *Shorea robusta* forests, which do not experience landslides produced an average soil loss of $<1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, had no torrents downside.^[4]

Flashy floods laden with debris come rushing down the hill slope and develop into channels of large dimensions and with lower flow intensities. Because of the greatly reduced carrying capacity of the discharge in the channels located in the plains, coarse material gets deposited in gravel bars and isolated islands. As a result, the discharge tends to migrate laterally cutting the ill-defined banks of the channels, and creating meander patterns that are typical of streams, not totally confined by the valley walls. During extreme storms, the discharge is so large that it inundates low lying agricultural fields and deposits large amount of coarse sand and gravels. The torrent beds widen many times that needed to convey effective discharge or even large floods. Often a large number of torrents discharge into the main rivers causing flash floods. The Soan river, during its run of 55 km in Himachal Pradesh, is fed by about 73 torrents with combined catchment area of $1,204 \text{ km}^2$.

The magnitude of damage caused by flash floods depends on the intensity of rainstorm. In 1988, the Shivalik region of Punjab received 700 mm rainfall in 3 days in the last week of September and experienced severest form of flood. Beside extensive damage to standing crops, livestock, and infrastructures, large amounts of sand were deposited in the agricultural fields along 200-km long stretch from Chandigarh to Pathankot. The State government incurred an expenditure of 2.64 billion rupees to clear sand from worst affected area of 2,988 ha. Rainfall events on June 16 and 17, 2013 caused flooding of Saraswati and Mandakini rivers in the Rudraprayag district of Uttarakhand (Fig. 1). Prolonged heavy down pour on these dates resembled a “cloud burst” (except for amount of precipitation of 100 mm hr^{-1}) type event in the Kedarnath valley and surrounding areas



Fig. 1 Vast devastation caused by torrential erosion in Chamoli district, Uttarakhand state of India in 2013.

Source: From Anonymous.^[5] ©2013 Deccan Chronicle Holding Limited, Hyderabad.

that damaged the banks of river Mandakini for 18 km between Kedarnath and Sonprayag and completely washed away the towns of Gaurikund, Rambara and Kedarnath. The roads and footpath between Gaurikund and Kedarnath were also damaged.^[5] There were reports of loss of a large number of human lives and damage to the property and livestock.^[6]

CAUSES OF TORRENTIAL EROSION

Landslides in the Upper and Middle Himalayas and landslips/vertical cliffs in Lower Himalayas are mainly responsible for torrential erosion. A landslide is a downward movement of a mass of rock or soil as a result of slope failure. The main reason for landslide occurrence is gravity force involving failure of earth materials under shear stress, i.e., when the shear stress exceeds the shear resistance of soil mass. Natural factors like steep slopes, a fragile geology, high seismic activity, and good rainfall make the region highly susceptible to landsliding. The problem has been further aggravated by anthropogenic activities like deforestation, unscientific cultivation on steep slopes, and developmental activities like road construction, mining, and rampant building constructions on hill slopes.

Water seeping through disturbed slopes, toe cutting by rivers and torrents, and pore water pressure in the soil zone are major agents responsible for landslides. Torrents, especially in their early stages, because of excessive velocities of flow and turbulence carry out quick removal of debris and cutting of sides. The landslides mostly take place after heavy rainfall. Construction of roads with slope deformities also accelerate mass erosion in the hills. These roads have been observed to suffer from about 4 to 10 landslides and 8 to 20 slumps per km, engulfing an area of about 15–20 ha.^[7] Major landslides in the Himalayas are estimated to cause an annual loss of more than 50,000 man hours per km and 5,000 vehicle hours per km on hill roads due to disruption of communication alone. The country suffers a loss of Rs. 1.50 billion annually toward the cost of removal of debris from the roads and loss of vehicle hours.

Majority of the Himalayan region falls under the highly earthquake prone “Zone V,” where earthquakes of magnitude 8 or more on the Richter scale could occur. Cracks developed in the underground rock layers, following an earthquake, may result in landslides during a monsoon due to reduced shear strength of underground strata. The two fault lines passing across the Himalayan region, the main central thrust (MCT) and main boundary fault (MBF), have been active for the past 400,000 to 500,000 years and are responsible for the formation of new landslides near the region of these fault lines. Chamoli and Uttarkashi fall on the MCT while MBF passes through Shimla–Dehradun–Almora.

Landslides are often caused by the cumulative effects of both natural and man-made factors. Landslides and

floods occurring in higher Central Himalayas of Chamoli district (Uttarakhand), which serve as the source for the Alaknanda river, have been attributed to inadequate vegetation cover coupled with the sheared carbonate rocks.^[8] Using remotely sensed data, the authors established a linear correlation between the magnitude of landslides and the forest cover. Many a time, man-made disturbances in a natural ecosystem may be the overriding cause of landslide hazards. The devastating landslides and associated floods of Alaknanda in 1970 have been attributed to the clearing of 6477 ha of forest land between 1959 and 1969 in the watersheds of the Patalganga, Garudganga, and Birahiganga of Chamoli districts.

DELINEATION OF SUSCEPTIBLE AREA

The landslides may cause immense damage to human life and property and untold misery to people as was seen during landslides occurring in Uttarakhand state. Much of the damages to human life, property, and developmental works can be avoided, if the likely landslide prone areas are identified and restored by the appropriate bioengineering technology on a watershed basis.

Landslide susceptibility mapping is the identification of areas within the landscape that have characteristics that could make them susceptible to landsliding.^[9] Landslide susceptibility mapping methods are diverse and numerous and often vary with the region, landscape, purpose, and finances available for the project. In the Middle and Upper Himalayan region, landslide hazard zonation (LHZ) is one technique that involves evaluation of hazard or risk caused due to slope failure processes. It considers various geo-environmental factors affecting the stability of a slope like considered land use, soil thickness, drainage, and slope condition. The LHZ mapping based on the landslide hazard elevation factor rating scheme, accepted by the Bureau of Indian Standard as an IS code, can also be used. It is a macro approach based on the basic causative factors, such as lithology, structure, slope morphology, relative relief, land use, and hydrogeological conditions. The Geological Survey of India has prepared a large number of LHZ maps of different basins in the Himalayan ecosystem.

Hardly any studies have been done for characterization and delineation of areas susceptible to mass or torrential erosion in the Lower Himalayas/Shivaliks. A study in the vertical cliff zone of the Shivalik region has been taken to develop a single-value stability indicator based on geology, physiography, and land management practices.^[10] It involved the development of a single-value stability indicator based factors that are assigned weights and normal values. Indicators have been identified at two levels, i.e., at the level of vertical cliff and at the level of strata. The sites of mass erosion-prone areas were assessed to arrive at a single value of stability based on their characteristics and strata composing them. The single value derived from this

Table 1 Giving weights to various indicators of mass erosion prone sites.

Indicator level I (properties of cliff)			Indicator level II (properties of strata)		
Parameter	Weight	Score	Parameter	Weight	Score
Number of strata	0.20	0.20			
Angle of strata	0.20	0.20			
Stability of strata	0.60		Cementation	0.60	0.36
			Clay content	0.20	0.12
			Aggregate size (MWD)	0.20	0.12
Total	1.00				

Source: From ICAR News.^[10] ©2011 Indian Council of Agricultural Research.

assessment ranged from 0.242 to 0.934 (Table 1). On the basis of these values, sites were classified as fragile, unstable, stable, and highly stable for values of <0.40, 0.40–0.60, 0.60–0.80, and >0.80, respectively.

PREVENTIVE AND CONTROL MEASURES

State Interventions

The menace of torrents was first noticed in the middle of the 19th century, and various types of administrative, legislative, and technological control measures have been implemented. A special “Choe” (Torrent) Control Act was passed during 1902 in Punjab. Under the act, villages situated at the Himalayan foothills were evicted and goat and sheep were banned in 126 villages of the Ropar district. More than 110 Cooperative Choe Reclamation Societies were constituted to ensure people’s participation in Choe reclamation measures.

The Government of India laid major emphasis on the control of torrential erosion after independence and, with the funding by the World Bank, implemented several projects in the western part of Himalayas (Shivalik foothills) comprising the states of Punjab, Himachal Pradesh, Haryana, Jammu and Kashmir, and Uttarakhand states. The projects were christened as Kandi Watershed Area Development Project during 1980–1988, National Watershed Development Project for Rainfed Agriculture during 8th (1992–1997) and 9th (1998–2003) Five-Year Plans, and Integrated Watershed Development Project from 1990 to 2005 and 1999 to 2005.

Technological Package

The technological package includes integrated watershed management to reduce runoff and soil loss based on

the following three-pronged treatment approach: 1) catchment’s landscape measures; 2) torrent control measures; and 3) regeneration of reclaimed beds and adjoining fields. The first approach addresses the cause of the problem and is key to long-term success. The second approach addresses problems associated with the torrents and is likely to have mixed success, as it replaces natural healing processes with the need to stabilize, harden, and fix the location of channel banks to protect anthropogenic activities and infrastructures. The third approach improves the ecology and productivity of these systems and is another key to developing self-sustaining systems. Often the success of this measure will depend on the success of other approaches, the ability to manage all activities within a watershed, and whether these ecosystems have sufficient time to recover before being impacted by extreme rainfall events. A broad outline of each measure is presented in the next section together with information on the benefits of the measures.

Catchment area treatments

The measures aim to reduce runoff and soil loss and, thus, reduce the peak discharge going to the torrents. The treatments may include staggered contour trenching, terracing in the catchments, and loose stone check dams, crate wire structures, drop structures, silt retention and water harvesting structures, grade stabilizers, etc., in gullies and streams coupled with vegetative measures. Staggered contour trenching followed with vegetative measures and protection from animal grazing by local community rehabilitated the degraded catchments of Sukhna lake in Haryana. Sediment-free runoff from rehabilitated catchments can be harvested by the construction of earthen embankments. The harvested runoff not only controlled flow to the torrent but it was also recycled for sustainable agricultural production in the command area (Fig. 2).^[11]



Fig. 2 Construction of an earthen embankment at Bunga in Haryana Shivaliks stopped flash flood downstream and provided supplemental irrigation for sustainable agricultural production.

Torrent control treatments

Torrent control measures are required to control bank erosion and reclaim the area along torrent bed in excess of that needed for safe disposal of water. Spurs, gabion structures constructed transverse to river flow extending from one of the bank at an angle to the flow, are generally used to train the flow along a desired course, retain sediments around them near bank, and prevent damage to buildings, roads, etc. Based on the function served and angle to the flow direction from downstream, the spurs may be of repelling ($>90^\circ$), attracting ($<90^\circ$), or deflecting (90°) type. Repelling and deflecting spurs promote quick sedimentation on both upstream and downstream sides, but sediment is deposited only at the downstream side of attracting spurs. Construction of retaining walls to stabilize steep slopes and protect buildings on torrent banks and cross barriers to break the flow velocity are the other measures required for complete taming of torrents.

In a few studies, attracting, deflecting, and repelling type of spurs have proved effective in guiding flow and protecting banks in different locations.^[12–14] Deflecting spurs are prone to damage by scour action at their nose during heavy flow. These types of spurs require construction of aprons of larger dimension, which further enhances the cost. Vegetative reinforcement of structures/spurs proved effective in improving their efficiency and reducing the cost of treatment. Average length saved by spurs (attracting and deflecting) increased from 1.92 to 3.13 times of their length 1 year after the establishment of vegetative measures and it may improve further. Rehabilitation of the Darer Rao torrent, with 1750 ha catchment area, gave a benefit-cost ratio of 2.7:1.^[15]

Regeneration of reclaimed beds and adjoining fields

The mechanical measures in themselves are not sufficient to provide permanent solution. It is the vegetation, which offers regeneration, permanent solution, and sustainability of conservation measures for treatment of torrents. Organic matter and fertility buildup are very low in torrent beds and soil structure is almost single grained, resulting in very poor retention of water and nutrients. Among the plant species tested, shrubs of medium height proved very effective in stabilizing banks, reducing flow velocity, and promoting sedimentation.^[13] These species rehabilitated degraded lands in the torrent beds through addition of organic residues and nutrients. Fast growing multipurpose tree-based silvi-pastoral systems provide ample returns to the farmers besides rejuvenating the torrent beds.

CONCLUSION

The post-independence era (1947 onwards) of India has witnessed the rapid expansion of areas prone to torrential

erosion because of deforestation, increased biotic pressure, and construction activities in the Himalayan hills; 2.73 Mha area is affected by torrential erosion in the Indian Himalayas alone. These areas experience soil loss up to $550 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ in mine spoiled hills and runoff up to 65% of monsoon rains causing flash floods and sand deposition in the fields. The areas vulnerable to torrential erosion can be delineated using geo-environmental factors in Upper and Middle Himalayas and single-value stability indicator in Lower Himalayas/Shivaliks. Technology package for control of torrential erosion comprising suitable soil conservation measures for denuded catchments and shallow streams has proved effective in checking the torrential erosion and regeneration of degraded lands. These measures have shown a marked reduction in runoff and soil loss.

REFERENCES

1. F.A.O. *Torrent Control Terminology*, Conservation Guide No. 6; F.A.O.: Rome, 1981; 165 pp.
2. Hattinger, H. *Torrent Control in the Mountains with Respect to the Tropics*. Conservation Guide No. 2; F.A.O.: Rome, 1976; 119–134.
3. Galay, V. *Erosion and Sedimentation in the Nepal Himalayas—An assessment of River processes*. WECS Report; His Majesty's Government of Nepal: Kathmandu, 1987.
4. Dhital, M.; Kahnal, N.; Thapa, K.B. *The Role of Extreme Weather Events, Mass Movements and Land Use changes in Increasing Natural Hazards*; ICIMOD: Kathmandu, 1993.
5. Anonymous. Uttarakhand disaster: Armed forces step up rescue operations, Kedarnath evacuated. Deccan Chronicle. April 10, 2015.
6. Dobhal, D.P.; Gupta, A.K.; Mehta, M.; Khandelwal, D.D. Kedarnath disaster: Facts and plausible causes. *Current Sci.* **2013**, *105* (2), 171–174.
7. Saxena, P.B.; Jauhari, R.; Sharma, P. An eco-system approach for appraising the environment degradation in the Himalayan water divides and their conservation by eco-top therapy. In *Torrent Menace—Challenges and Opportunities*; Sastry, G., Sharda, V.N., Juyal, G.P., Samra, J.S., Eds.; Central Soil and Water Conservation Research and Training Institute: Dehradun, 1995; 75–81.
8. Kimothi, M.M.; Juyal, N. Environmental impact assessment of a few selected watersheds of the Chamoli district (Central Himalayas) using remotely sensed data. *Int. J. Remote Sens.* **1996**, *17* (7), 1391–1405.
9. Morrocco, S.; Bones, R. *Landslide Susceptibility Mapping: Literature Review and Findings*; Geological Survey of Ireland: Dublin, 2011; 26 pp.
10. ICAR News. *Stability Indicators at Chandigarh for Mass Erosion Prone Sites*; Indian Council of Agricultural Research: Delhi, 2011.
11. Arya, S.L.; Samra, J.S. *Socio-economic Implications and Participatory Appraisal of Integrated Watershed*

- Management Project at Bunga, Technical Bull. No. T-27/C-6; Central Soil and Water Conservation Research and Training Institute: Research Centre: Chandigarh, 1995; 112 pp.*
12. Juyal, G.P.; Ghosh, B.N.; Bihari, B.; Rathore, A.C. Development of cost effective technology for treatments of torrents in Shiwaliks of Uttaranchal, extended abstracts. In *National Conference on Resource Conserving Technologies for Social Upliftment*, New Delhi, December 7-9, 2004, Association of Soil and Water Conservationist: Dehradun, 2004; 162–165.
 13. Tiwari, A.K.; Aggarwal, R.K.; Arya, S.L.; Sharma, P.; Prasad, R. Bio-engineering measures for control of torrents in Shiwaliks, Extended Abstracts. In *National conference on Resource Conserving Technologies for Social Upliftment*, New Delhi, December 7-9, 2004, Association of Soil and Water Conservationist: Dehradun, 2004; 103–104.
 14. Patnaik, U.S.; Singh, R.K.; Sharda, V.N.; Dhyani, S.K. Reclamation of torrents in Doon Valley. In *Torrent Menace: Challenges and Opportunities*; Shastri, G., Sharda, V.N., Juyal, G.P., Samra, J.S., Eds.; Central Soil and Water Conservation Research and Training Institute: Dehradun, 1995; 260–272.
 15. Bisht, M.S. Torrent control measures in Darer Rao (Dehradun)—A Case study. In *Torrent Menace: Challenges and Opportunities*; Shastri, G., Sharda, V.N., Juyal, G.P., Samra, J.S., Eds.; Central Soil and Water Conservation Research and Training Institute: Dehradun, 1995; 186–194.

Trace Metal Contamination

Åsgeir Rossebø Almås

Bal Ram Singh

*Department of Environmental Sciences, Norwegian University of Life Sciences,
Aas, Norway*

Abstract

Soil contamination caused by the addition of compounds that result in adverse effect on soil functioning is a matter of great concern. Trace metals include a great number of elements in the periodic table of elements, and their biogeochemistry and impact on living organisms can be very different. It is well known that some are essential up to a critical limit where they become toxic, and others are directly toxic even in small concentrations. Over the years, the research has focused on unraveling their sources, reactions in soils, and impact on the ecosystem. Several assessment techniques have been applied to estimate the biological risks, and some of those are discussed here. Contamination is caused by both geogenic and anthropogenic sources, which are described. The source of contamination is important to understand the biological response to environmental concentrations determined. The determination of metal concentrations includes total digestions of soils with concentrated acids under high temperature and pressure, extractions using complexing agents and weak extraction solutions, evacuation of soil solutions, and installation of devices accumulating metal ions directly from moist soils. The latter have been made possible by applying the diffusive gradients in thin films and the technique is discussed in relation to its capacity in estimating bioavailability. We present studies where the speciation of metals in soil solution has been carried out using the speciation models Windermere Humic Aqueous Model/Model VI, in combination with biological impact studies (pollution-induced community tolerance). Such combined studies provide good evidence on the fate of metal contamination in the ecosystem.

INTRODUCTION

Soil contamination is caused by the addition of compounds that result in adverse effect on soil functioning, while soil pollution is reserved to the cases where contamination is severe and leads to malfunctioning of soil function resulting in soil degradation. Soil degradation could be physical, chemical, or biological but the focus in this entry will be on chemical degradation caused by trace metals and metalloids. Chemical degradation, defined as combined negative effects of chemicals and chemical processes on those properties that regulate the life process in the soil, can be caused by natural processes or by anthropogenic activities. Trace metals with specific densities greater than 5 g/cm³, such as lead (Pb), zinc (Zn), cadmium (Cd), mercury (Hg), and chromium (Cr), are often referred to as heavy metals. Metalloids such as arsenic (As), antimony (Sb), and selenium (Se) form oxyanions and their physicochemical behavior in the environment is very different from the trace metals. Trace metals and metalloids are persistent in the environment, non-biodegradable, non-thermo-degradable and thus readily accumulate to toxic levels.

Entry of soil-borne metals and metalloids into the food chain depends on the amount and source of input, properties rate and magnitude of uptake by plants, and the extent of absorption by animals. Soil metal pollution has become a

severe problem in many parts of the world. Historically, trace metal toxicity to human health received attention because of a series of widespread poisoning. For example, the hundreds of tragic cases of human poisoning of Minamata Bay in Japan (Minamata disease) in the late 1950s were believed to have occurred from the ingestion of fish containing methyl mercuric compounds probably derived through biomethylation of mercuric salts by aquatic organisms. High concentrations of trace metals and metalloids, such as Cd, Cu, Pb, Zn, and As, in soils and crops have often been reported in a high number of countries. For example, significant adverse impacts of As-rich ground water on human health have been recorded in Bangladesh, India, and China. It is claimed that millions of people are potentially at risk from As poisoning also through food crops. Similarly, Cd accumulation in the grazing animals in New Zealand and Australia made it unsuitable for human consumption and affected access of meat products to overseas markets. There have also been concerns about urban development of horticultural sites, which contained toxic concentrations of metals and metalloids such as Cu, Pb, and As in soils resulting from excessive use of fungicides and herbicides that are rich in these elements. Following rapid social and economic development over the past several decades, soil pollution by trace metals has been both serious and widespread in China. This entry presents a brief overview of trace metal contamination, sources, their

reaction in soils, and assessment techniques to find the risk of biological impact.

Sources of Contamination

Soil can be contaminated by trace metals released from smelters, mines, waste incinerators, industrial wastewater, and from the application of sludge or municipal compost, pesticides, and fertilizers. Irrespective of their sources in the soil, accumulation of trace metals can degrade soil quality, reduce crop yield and the quality of agricultural products, and thus, negatively affect the health of human, animals, and the ecosystem.^[1]

Trace metals reach the soil environment through both pedogenic and anthropogenic processes. Most trace metals occur naturally in soil parent materials, chiefly in forms that are not readily bioavailable for plant uptake. Unlike pedogenic inputs, trace metals added through anthropogenic activities typically have a higher bioavailability.^[2] Anthropogenic activities primarily associated with industrial processes, manufacturing, the disposal of domestic and industrial waste materials, and the application of phosphorus (P) fertilizers are the major source of metal enrichment in soils.^[3] In China, sources of trace metals in urban soils and urban road dusts are mainly derived from traffic sources and industrial sources. However, the sources of trace metals in agricultural soils are mainly influenced by parent materials, mining, fertilization, pesticide application, etc.^[4]

Geogenic

Most of the trace metals occur in nature, the major source of which is weathering of soil parent materials including igneous and sedimentary rocks and coal. For example, coal is estimated to release 45,000 tons of As annually, whereas human activities release approximately 50,000 tons.^[5] Although the anthropogenic As source is increasingly becoming important, the episode of extensive As contamination of ground waters in many countries including Bangladesh, India, China, and Mexico is of geological origin. Originally transported by rivers from sedimentary rocks in the Himalayas over tens of thousands of years.^[6]

Volcanic and geological activities mobilize natural Hg from deep reservoirs in the earth to the atmosphere. Annual emission of Hg from global mercuriferous belts, the zone along plate tectonic including western North America, Central Europe, and southern China was estimated up to 500 Mg/year.^[7] In Norway, large areas of soils developed on alum shales contain substantially higher concentration especially of Cd, Cu, and nickel.^[8]

Anthropogenic

Anthropogenic activities primarily associated with industrial processes, manufacturing, and the disposal of domestic

and industrial waste materials are the major source of metal enrichment in soils. Atmospheric pollution from Pb-based petrol was a major issue in many countries where there was no constraint in the usage of leaded gasoline. While biosolids are the major source of metal inputs in Europe and North America, P fertilizers are considered the major source of trace metal input, especially Cd, in Australia and New Zealand.^[3]

Phosphate compounds contain a range of metals, which originate primarily from the phosphate rocks used for fertilizer production. Although many countries have formulated threshold levels for Cd and other trace metal accumulation in soils due to the use of sewage sludge effluents and municipal biosolids, such limits have not been established for fertilizer use. Copper (Cu) may accumulate in agricultural soils from continuous use of Cu fungicides and biosolids.

METAL BEHAVIOUR IN SOIL

Trace metals in soils are mostly partitioned in organic and inorganic fractions and not readily available to soil living organisms. However, the fraction of metals bound to solid soil fractions quickly reacting to soil physico-chemical changes is termed geochemically active. Such changes occur typically under shifts in pH and redox conditions, changes in the composition of pore water solutes like dissolved organic matter (DOM), or even changes in the quality and quantity of solid material such as iron (Fe) oxides or solid soil organic matter (SOM). The latter can take place through soil amendments of sewage sludge or different sources of Fe-bearing minerals.

The sum of geochemically active metals and the chemically labile metal fractions in pore water are often termed bioavailable (Fig. 1). The bioavailable metals can potentially be taken up by plants or other soil living organisms. The controlling mechanisms are complex as the uptake is controlled by the organism it selves, its requirements for mineral nutrients, its physiological status, uptake strategies, and internalization rates. Whether the metal uptake is controlled by diffusion or mass flow in the rhizosphere,^[9] the metal speciation in pore water and the geochemistry

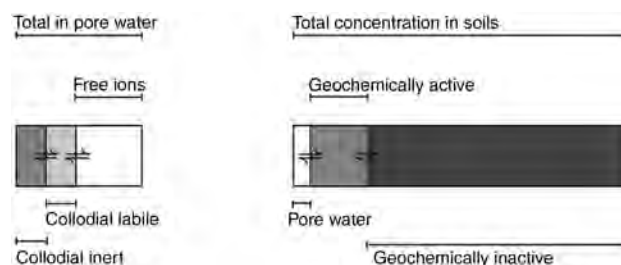


Fig. 1 Schematic presentation of trace metal partitioning in soil (right side) and speciation in pore water of soils (left side).

of the different metals in the bulk soil^[10] are also the contributory factors.

The metal geochemistry depends on factors as described in a number of studies and modeling approaches.^[11–15] The most important soil factors controlling the concentrations of metals in pore water are the total concentration in soil, the concentration of solid SOM, the soil pH, and varying concentration of metal-binding solutes (ligands) in the pore water.^[15,16] The relative importance of the anionic pore water components varies, but the concentration of DOM is most important due to its high metal-binding capacity.

METHODS OF ASSESSMENT

Chemical Assessments

The total digestion of soil and sediment samples can be carried out by digesting ground material with ultrapure concentrated acid under high pressure and temperature. For direct assessment of bioavailability, the results from total digestions of soils in acid are of little value. Hence, a number of extraction protocols have been developed and implemented. Generally, these methods make use of weak acids [1.0 M ammonium (NH₄) acetate, 0.1 M hydrogen chloride, and 0.1 M NH₄ lactate], complexing agents (e.g., ethylenediaminetetraacetic acid, diethylene triamine pentaacetic acid, and trichloroacetic acid), or dilute salts (0.1 M potassium nitrate, 0.01 M calcium chloride, etc.) for the estimation of bioavailable or geochemically active metal fractions. Other more sophisticated are of methods using isotopic dilution or even sequential extraction (SE) regimes. The use of SE protocols, mostly based on the Tessier^[17] protocol, aims at describing the metal binding to different soil fractions, from water soluble through exchangeable to inert organic and inorganic soil components. The SE precision of metal partitioning between the suggested metal associations to different solid soil fractions determined with this approach has, however, been questioned.^[18,19] Despite some limitations, SE technique is widely used in pollution research and in studying the partitioning of metals in solid soil fractions.

Most functional fractionation of metals in soils seems to aim at identifying metal concentration and speciation in pore water and the concentrations being geochemically active. The remaining metal fractions are identified as geochemically inactive (Fig. 1). For the assessment of bioavailability, the use of diffusive gradients in thin film (DGT) technology has gained importance as the DGT units can be placed directly in moist soils,^[20] free water, and sediments. This provides several advantage, and the two most obvious ones are the time-integrated flux and accumulation of labile trace elements through a size discriminating gel into a resin (Chelex or Fe oxide). Second, the solid solution ratio during exposure is realistic with respect to field situations. Like uptake through plant roots, the

accumulation of trace elements in the DGT lowers the concentrations in soil solution at the surface of the device. This induces the resupply of trace elements from the geochemically active fractions in solid soil. Moreover, the DGT design restricts trace element complexes from uptake. Several studies have provided sound evidence showing unique functional applicability of the DGT technique for estimating metal bioavailability.^[21,22]

Modeling

Models capable of handling the geochemistry of soil solution require simulation of processes of solubility equilibria, including precipitation and dissolution reactions, specific adsorption on “oxide-like” surfaces, and cation exchange. The mathematical approaches to predict the free metal activity in soil solution are often done using adsorption and ion-exchange models for the various soil constituents and compare these predictions with measurements. Thus, the free metal concentration and the geochemical active metal fractions in soil can be estimated. Humic substances are recognized to play an important role in the chemical speciation of waters, sediments, and soils. Their interactions with protons and trace metals are possible to assess with an appropriate speciation code. Incorporation of humic substances into trace metal speciation models presents a major challenge due to the polydisperse nature of these materials. Humic substances are traditionally divided into three operational fractions: the soluble fulvic and humic acids and the insoluble humin. From the standpoint of speciation modeling, the two major types of metal-binding organic sites are strongly acidic carboxylic groups (ionized in the acidic pH region) and the weaker acidic phenolic groups (ionized in the higher pH region). The non-ideal competitive adsorption (NICA)-Donnan model^[13] is a combination of the Donnan-type non-specific binding of electrolyte ions in combination of the NICA model for specific binding to produce a model for metal ion binding to humic substances. The Windermere Humic Aqueous Model (WHAM) uses the site heterogeneity of DOM together with electrostatic effects and competition among protons and metal ions to describe metal binding by humic matter.^[14,23]

As the controlling variables are well known, simpler approaches are provided in the assessments of metal solubility in soil systems. These variables are, besides the total concentration of metals (M_{tot}) in soils, the soil pH, concentration of SOM, and also other metal-binding soil surfaces express as either metal oxides or clays.^[15,24,25] Sometimes these binding properties are simply expressed through the cation exchange capacity. These variables are optimized using known data on metal concentrations in pore water and hence have the tendency in being conditional to the soils used.^[12] These variables are combined into multiple regression models. Well-parameterized regression models can be applied more easily to provide at least good estimates of metal solubility in soils with

shifting concentrations of SOM or pH^[26] as long as the parameter values are within the range used for optimization.

Biological Assessments

Uptake in plants

The bioavailability of trace metals in soils has for long been connected to the uptake in agricultural crops for human and animal consumption. Cd is de facto the most studied non-essential trace metal in soil–plant ecosystem. Other metals of concern have been Hg and Pb and the metalloids As and Se. Due to the impact from trace industry on productive land and use of pesticides, the biogeochemistry of Zn and Cu has also followed closely and a large number of studies have been published. From one of our own pot studies conducted using smelter-contaminated soils, close correlation between DGT available Cd and Zn and its uptake in rye grass and spinach was reported.^[10] The speciation of Cd and Zn in soil pore water was estimated using the WHAM/Model VI.^[14] The free ion activity in pore water correlated well with the concentrations in plants, but the correlation between C_E (effective concentration) for Cd and Zn was even better. A strong linear relationship was found between C_E -Zn and the Zn concentration in ryegrass ($r^2 = 0.96$, $p < 0.001$). Since a DGT unit is unaffected by toxicity, the DGT uptake was linear with C_E only in non-toxic range of the concentration scale. The uptake stopped in spinach when the bioavailable concentrations in soils became toxic, whereas for rye grass the C_E was below toxic critical concentration. This indicates another matter of importance that the biological impact of trace metal soil contamination is species dependent.

Pollution-induced community tolerance (PICT)

Additionally, to investigating the uptake in plants and toxicity measured as reduction in growth, the development of metal tolerance in soil microbial communities has been reported.^[27,28] Microbial functions in soils are quite resilient toward metal contamination as they often recover after initial inhibition.^[29] This probably is connected to the fact that tolerant organisms replace the sensitive once, thus changing the community composition and flowingly increase its conditional metal tolerance. Almås et al.^[27] presented evidences that such metal tolerance had been developed in soil microbial communities withdrawn from the 10 different soils sampled from the same sites as investigated in the pot experiment cited above. The soils varied in physical and chemical characteristics, but the C_E and metal concentrations in pore water were determined and the correlations with acquired metal tolerance expressed as LC₅₀ [(LC50 = 50% of Lethal Concentration) 50% inhibition using the ³H thymidine incorporation method] were positive and strong. The Mtot in soils as a single parameter had no impact on LC₅₀. This showed that not only the

metal contamination but also more important the biogeochemistry of metals in soils controlled the development of PICT for Zn and Cd. Later, soil microbial communities were sequentially extracted from four soils to investigate if the strength of attachment to soil had any impact on PICT development. This was carried out by sequential dispersion and density gradient centrifugation to separate free and loosely attached cells (FLA) from strongly attached cells (SA).^[30] The results showed two remarkable findings. First, the FLA communities had developed a high metal tolerance compared to that of SA, and it was positively correlated with Cd²⁺ and Zn²⁺ in pore water ($r^2 = 0.98$ and 0.99 for FLA-IC₅₀ values against the Cd²⁺ and Zn²⁺, respectively). In contrast, SA shows no such sign of PICT. Second, the *in situ* concentration of free Cd²⁺ calculated from input parameters of pore water from field soils showed that the PICT for Cd in the test was order of magnitudes higher than Cd²⁺ in field condition. Nor the concentrations of Cd²⁺ or the total Cd in pore water in field could have affected even the most sensitive communities. The observed increase in PICT for Cd was thus not due to direct exposure of Cd. The plausible explanation for this PICT was that Zn had selected for organisms that are also Cd tolerant. These findings illustrate the importance in understanding the total biogeochemistry of metals in soils. For instance, the apparent lack in PICT development for bacterial cells identified as SA may be important when assessing the impact of metal contamination on microbial functions. Tolerant organisms may very well maintain the microbial functions, but these results indicate that investigating the biogeochemistry as a whole is of utmost importance for understanding the apparent robustness often seen in studies of microbial functions in metal contaminated soils.

REFERENCES

1. Nagajyoti, P.C.; Lee, K.D.; Sreekanth, T.V.M. Heavy metals, occurrence and toxicity for plants: A review. *Environ. Chem. Lett.* **2010**, *8* (3), 199–216.
2. Lamba, D.T.; Ming, H.; Megharaj, M.; Naidu, R. Heavy metal (Cu, Zn, Cd and Pb) partitioning and bioaccessibility in uncontaminated and long-term contaminated soils. *J. Hazard. Mater.* **2009**, *171* (1–3), 1150–1158.
3. Bolan, N.S.; Adriano, D.C.; Naidu, R. Role of phosphorus in (im)mobilization and bioavailability of heavy metals in the soil–plant system. *Rev. Environ. Contam. T.* **2003**, *177*, 1–44.
4. Wei, B.; Yang, L. A review of heavy metal contaminations in urban soils, urban road dusts and agricultural soils from China. *Microchem. J.* **2010**, *94* (2), 99–107.
5. Mahimairaja, S.; Bolan, N.S.; Adriano, D.C.; Robinson, B. Arsenic contamination and its risk management in complex environmental settings. *Adv. Agron.* **2005**, *86*, 1–82.
6. Smedley, P.L.; Kinniburgh, D.G. A review of the source, behaviour and distribution of arsenic in natural waters. *Appl. Geochem.* **2002**, *17* (5), 517–568.
7. Lindqvist, O.; Johansson, K.; Aastrup, M.; Andersson, A.; Bringmark, L.; Hovsenius, G. Mercury in the Swedish

- environment—recent research on causes, consequences and corrective methods. *Water Air Soil Poll* **1991**, *55* (1–2), 1–126.
8. Singh, B.R. Trace element availability to plants in agricultural soils, with special emphasis on fertilizer inputs. *Environ. Rev.* **1994**, *2* (2), 133–146.
 9. Degryse, F.; Smolders, E.; Zhang, H.; Davison, W. Predicting availability of mineral elements to plants with the DGT technique: A review of experimental data and interpretation by modelling. *Environ. Chem.* **2009**, *6* (3), 198–218.
 10. Almås, Å.R.; Lombnæs, P.; Sogn, T.A.; Mulder, J. Speciation of Cd and Zn in contaminated soils determined by DGT-DIFS, and WHAM Model/ VI in relation to uptake by spinach and ryegrass. *Chemosphere* **2005**, *62* (10), 1647–1655.
 11. Almås, Å.R.; Loftis, S.; Tipping, E.; Mulder, J. Solubility of major cations and Cu, Zn and Cd in soil extracts of some contaminated agricultural soils near a zinc smelter in Norway: Modelling with a multisurface extension of WHAM. *Eur. J. Soil Sci.* **2007**, *58* (5), 1074–1086.
 12. Ivezic, V.; Almås, Å.R.; Singh, B.R. Predicting the solubility of Cd, Cu, Pb and Zn in uncontaminated Croatian soils under different land uses by applying established regression models. *Geoderma* **2012**, *170*, 89–95.
 13. Kinniburgh, D.G.; Milne, C.J.; Benedetti, M.F.; Pinheiro, J.P.; Filius, J.; Koopal, L.K. Metal ion binding by humic acid: Application of the NICA-Donnan Model. *Environ. Sci. Tech.* **1996**, *30* (5), 1687–1698.
 14. Tipping, E. Humic ion-binding Model VI: An improved description of the interactions of protons and metal ions with humic substances. *Aquat. Geochem.* **1998**, *4*, 3–48.
 15. Tipping, E.; Rieuwerts, J.; Pan, G.; Ashmore, M.R.; Loftis, S.; Fargo, M. The solid solution partitioning of heavy metals (Cu, Zn, Cd, Pb) in upland soils of England and Wales. *Environ. Pollut.* **2003**, *125* (2), 213–225.
 16. Smolders, E.; Oorts, K.; van Sprang, P.; Schoeters, I.; Janssen, C.R.; McGrath, S.P. Toxicity of trace metals in soil as affected by soil type and aging after contamination: Using calibrated bioavailability models to set ecological soil standards. *Environ. Toxicol. Chem.* **2009**, *28* (8), 1633–1642.
 17. Tessier, A.; Campbell, P.G.C.; Bisson, M. Sequential extraction procedure for the speciation of particulate trace metals. *Anal. Chem.* **1979**, *51* (7), 844–851.
 18. D'Amore, J.J.; Al-Abed, S.R.; Scheckel, K.G.; Ryan, J.A. Methods for speciation of metals in soils: A review. *J. Environ. Qual.* **2005**, *34* (5), 1707–1745.
 19. Young, S.D.; Zhang, H.; Tye, A.M.; Maxted, A.; Thums, C.; Thornton, I. Characterizing the availability of metals in contaminated soils. I. The solid phase: Sequential extraction and isotopic dilution. *Soil Use Manage* **2005**, *21* (Suppl 2), 450–458.
 20. Zhang, H.; Zhao, F.J.; Sun, B.; Davison, W.; McGrath, S.P. A new method to measure effective soil solution concentration predicts copper availability to plants. *Environ. Sci. Technol.* **2001**, *35* (12), 2602–2607.
 21. Nolan, A.L.; Zhang, H.; McLaughlin, M.J. Prediction of zinc, cadmium, lead, and copper availability to wheat in contaminated soils using chemical speciation, diffusive gradients in thin films, extraction, and isotopic dilution techniques. *J. Environ. Qual.* **2005**, *34* (2), 496–507.
 22. Nowack, B.; Koehler, S.; Schulen, R. Use of diffusive gradients in thin films (DGT) in undisturbed field soils. *Environ. Sci. Technol.* **2004**, *38* (4), 1133–1138.
 23. Loftis, S.; Tipping, E. Assessing WHAM/Model VII against field measurements of free metal ion concentrations: Model performance and the role of uncertainty in parameters and inputs. *Environ. Chem.* **2011**, *8* (5), 501–516.
 24. Guggenberger, G.; Christensen, B.T.; Zech, W. Land-use effects on the composition of organic matter in particle-size separates of soil. 1. Lignin and carbohydrate signature. *Eur. J. Soil Sci.* **1994**, *45* (4), 449–458.
 25. Sauvé, S.; Hendershot, W.; Allen, H.E. Solid-solution partitioning of metals in contaminated soils: Dependence on pH, total metal burden, and organic matter. *Environ. Sci. Technol.* **2000**, *34* (7), 1125–1131.
 26. Ivezic, V.; Singh, B.R.; Almås, Å.R.; Loncaric, Z. Water extractable concentrations of Fe, Mn, Ni, Co, Mo, Pb and Cd under different land uses of Danube basin in Croatia. *Acta Agr Scand B-S P.* **2011**, *61* (8), 747–759.
 27. Almås, Å.R.; Bakken, L.R.; Mulder, J. Changes in tolerance of soil microbial communities in Zn and Cd contaminated soils. *Soil Biol. Biochem.* **2004**, *36* (5), 805–813.
 28. Bååth, E. Measurements of heavy metal tolerance of soil bacteria using thymidine incorporation into bacteria extracted after homogenization-centrifugation. *Soil Biol. Biochem.* **1992**, *24*, 1167–1172.
 29. Holtan-Hartwig, L.; Bechman, H.; Høyås, T.R.; Linjordet, R.; Bakken, L.R. Heavy metals tolerance of soil denitrifying communities: N₂O dynamics. *Soil Biol. Biochem.* **2002**, *34* (8), 1181–1190.
 30. Almås, Å.R.; Mulder, J.; Bakken, L.R. The trace metal exposure of soil bacteria depends on their position in the soil matrix. *Environ. Sci. Technol.* **2005**, *39* (16), 5927–5932.

Tropics and Subtropics

Hari Eswaran

*U.S. Department of Agriculture—Natural Resources Conservation Service
(USDA-NRCS) (Retired), Washington, District of Columbia, U.S.A.*

Eswaran Padmanabhan

Geoscience, University Teknologi Petronas, Bandar Seri Iskandar, Malaysia

Abstract

In this entry, the tropics are defined as the area with a mean summer to winter soil temperature difference of less than 6°C. With a relatively uniform temperature, subdivisions of the biome may be made on the availability of moisture. Though climate is a major determinant of the biological character of the tropics, significant differences are also introduced by the soils that are present.

INTRODUCTION

The tropics straddle the equator and by definition do not have large differences in temperature in the months corresponding to “winter or summer” of the higher latitudes. In this section, the tropics are defined as the area with a mean summer to winter soil temperature difference of less than 6°C as per the definition in Soil Taxonomy.^[1] The wide diversity of soils of the tropics is elaborated by Eswaran et al.^[2] With increases in elevation, the mean annual temperature decreases, resulting in temperature zones with corresponding differences in floristic composition. With a relatively uniform temperature, subdivisions of the biome may be made on the availability of moisture. The semiarid tropics border the deserts and occupy about 15.5% of the area, while the humid tropics include the intertropical convergence zone and occupy 11.1% of the tropical biome zone.

Within the humid area, there is a subzone where the evapotranspiration does not exceed the precipitation in any month of the year. This is, the extremely wet part of the tropics and typifies the “tropical rain forest.” In these perhumid areas, there are no dry months and annual precipitation frequently exceeds 4 m and may be as high as 12 m. Tree growth is luxuriant with several layers of vegetation. The canopy is closed, and the under-storey layers are usually very high in humidity. High humidity promotes the growth of epiphytes. Tropical rainforests provide a large amount of leaf litter, but these are rapidly humified with insignificant accumulation of organic matter in the soil. Diversity of plants is large and most of the families occur on all the continents. The humid and perhumid areas support a wide variety of animal life that clusters around specific layers of the forest. As a rule, grass-eating animals are scarce but increase where the forest grades into the savanna.

A tropical dry forest occurs in the semiarid areas. Although this is the climax vegetation, it is quickly destroyed and does not have enough resilience to regenerate. Degeneration is evident in the semiarid parts of Africa where previous forest lands lie barren after deforestation. Species diversity is also lower in the dry forests, which are generally more fragile. The dry forests give way to grasslands or savanna. In these landscapes, gallery forests where the forests follow riparian courses are frequent. In the semiarid areas, there is also considerable plant adaptation to drought conditions. Long tap roots and thick barks characterize the shrubs. The branches of the shrubs are also generally twisted and gnawed because of frequent fires. Herbivores dominate, but carnivores are also present to maintain a balance in the animal population.

Though climate is a major determinant of the biological character of the tropics, significant differences are also introduced by the soils that are present. A classical example is illustrated by areas with highly weathered Oxisols belonging to great groups such as the Acrudox and Acrustox. These soils have a very low nutrient and water-holding capacity, and consequently, even in an area with more than 5 m of rain and no dry season, the vegetation may comprise stunted trees or grass. In soils with excess water, as in some wetland areas, there is a completely different association of vegetation and animal life.

The dominant soils and the role they play in providing the character of the biomes are illustrated with a few examples. [Table 1](#) provides the area occupied by the suborders in tropical biomes.

OXISOLS

Oxisols^[3] are present in both the wet and dry tropics. Large contiguous areas of Oxisols are on the Amazon

Table 1 Area ($\times 1,000 \text{ km}^2$) occupied by soil suborders in tropical biomes.

Soil order	Suborder	Tropical			
		Semiarid		Humid	
		Area	%	Area	%
Histosols	Sapristis	37	0.11	281	0.81
		37	0.11	281	0.81
	Aquods			13	0.04
	Cryods				
Spodosols	Humods			29	0.08
	Orthods			18	0.05
				61	0.17
	Vitrands	139	0.40	48	0.14
Andisols	Ustands	59	0.17		
	Udands			187	0.54
		198	0.57	235	0.68
	Aquox	300	0.86	20	0.06
Oxisols	Ustox	3,087	8.87		
	Perox			1,010	2.90
	Udox			5,169	14.85
		3,387	9.73	6,199	17.81
Vertisols	Aquerts			1	0.00
	Usterts	1,170	3.36		
	Uderts			86	0.25
		1,170	3.36	87	0.25
Aridisols	Argids	44	0.13		
	Calcids			1	0.00
	Cambids				
		44	0.13	1	0.00
Ultisols	Aquults	714	2.05	331	0.95
	Humults	24	0.07	254	0.73
	Udults			2,656	7.63
	Ustults	3,633	10.44		
Mollisols	Xerults				
		4,371	12.56	3,240	9.31
	Aquolls			1	0.00
	Rendolls			121	0.35
Alfisols	Ustolls	136	0.39		
	Udolls			54	0.16
		136	0.39	176	0.51
	Aqualfs	387	1.11	20	0.06
	Ustalfs	3,777	10.85		
	Udalfs			618	1.77
		4,165	11.96	638	1.83
	Aquepts	758	2.18	744	2.14
	Cryepts				

(Continued)

Table 1 Area ($\times 1,000 \text{ km}^2$) occupied by soil suborders in tropical biomes. (Continued)

Soil order	Suborder	Tropical			
		Semiarid		Humid	
		Area	%	Area	%
Inceptisols	Ustepts	2,807	8.06		
	Udepts			1,757	5.05
		3,566	10.24	2,502	7.19
	Aquents	74	0.21	29	0.08
Entisols	Psamments	1,947	5.59	558	1.60
	Fluvents	521	1.50	396	1.14
	Orthents	683	1.96	111	0.32
		3,225	9.26	1,093	3.14
Total		20,299	58.31	14,514	41.69

Shield in South America and in the Congo Basin in Central Africa. Small areas occur on the volcanic islands of the Pacific and in Southeast Asia, where they form on weathering products of basic and ultrabasic rocks. On the older surfaces of Africa and South America, peneplanation has flattened the land surface. The Oxisols are generally formed on preweathered and transported sediments. One feature they have is a stone line, marking differential deposition.

The soils have low nutrient retention capacity and low water-holding capacity, and in many countries, these features have earned them the name “droughty soils.” The ability to retain anions such as phosphates, nitrates, and sulfates is high. Extremes in cation and anion retention are seen in Oxisols with a net positive charge. These properties are detrimental to plant growth, and therefore, vegetation production is very low and sometimes only adapted plants grow on these soils. The Oxisols of the high plateaus of Central Africa have a subsurface dark horizon called a sombric horizon. They are distinctive in these non-descript soils and have caught the attention of local farmers who prefer them because of their better fertility.

ULTISOLS

Occupying 7.6 million square meter (21.87% of the tropics), the Ultisols are the second most extensive soils. Although there are a few members on recent erosion surfaces, most of the Ultisols are highly weathered with few weatherable minerals and occur on stable geomorphic surfaces.^[4] The landscape is generally undulating. Ultisols occur in association with Oxisols and also share many of the physicochemical properties. Water and nutrient retention is also low, and some, as the Oxisols on old landscapes, have stone lines.

The Ultisols have an argillic or a kandic horizon, and the latter soils are generally associated with older landscapes. On the old surfaces of Central Africa and the Amazon

Shield, lithological discontinuities, sometimes with stone lines, are common in the Kandiusults and Kandiidults. Clay content differentiation in the upper part of the soil, a defining characteristic for Ultisols, is generally due to geomorphic processes: clay translocation and accumulation to form an argillic horizon, and it is a relict feature in many Ultisols or confined to the soils on younger surfaces.

Organic matter content is related to moisture saturation. Some Ultisols (humults), which occur at high elevations in cooler environments, also have high organic matter. For low-input farming, these organic matter-rich soils are frequently the first choice for food production. Ultisols are preferred to Oxisols for grain production, but the latter soils are generally highly rated for perennial crops such as rubber (*Hevea brasiliensis*).

Many Ultisols (and some Oxisols and Alfisols) have plinthite.^[5] Plinthite is a sesquioxide-rich, organic matter poor material, which has a reticulate pattern with reddish and white colors. It forms under aquic conditions, and when the water table is lowered and the surface exposed, it can harden to ironstone. Soils with plinthite are considered as degraded because their productivity is low, and they are also difficult to manage.

ALFISOLS

The Alfisols are the pedological equivalent of the Ultisols with a high pH or base saturation. They are preferred to Ultisols for their better nutrient status and easy workability. The Alfisols are more frequent in the semiarid tropics. In the humid tropics, they occur on specific rock types, such as limestone and/or basic rocks, or at locations where there is an external source of bases. The dust from the Sahara (Harmattan winds) and Kalahari deserts add large quantities of carbonates to the soils of Central Africa, and so Alfisols are extensive in Africa.

Alfisols, particularly the Ustalfs, which have a longer period of moisture deficit, are widely used for food crop production. In the wet season, they grow maize, sorghum, and soybeans. As moisture stress increases, the variety of crops grown in the absence of irrigation includes chickpeas and millet. In the Ustalfs and in some of the Udalfs, sugarcane is an important crop. In the drier tropics, the Alfisols and the other soils are used for grazing. Managed pastures are modern, and in many countries, the carrying capacity for animals is frequently exceeded. With an adequate supply of water and nutrients, particularly nitrogen (N) and phosphorous (P), these are one of the most productive soils of the tropics. The other very productive soils are the Molisols, but these are not very extensive.

INCEPTISOLS AND ENTISOLS

The alluvial soils of valleys and coastal plains have been the sites for agricultural development in Asia. These soils

are naturally wet or can be puddled for rice cultivation. The Entisols occur on the flood plains, while the better developed Inceptisols are on older surfaces. In the semiarid tropics, rice is grown during the wet season while an upland crop is attempted, if adequate moisture is available, for much of the growing period. In Africa, wet Inceptisols and Entisols were traditionally avoided because of heavy infestation with pests and diseases. The African farmer prefers to cultivate the better drained upland soils.

On the coastal plains of the tropics, acid sulfate soils occur, and they are more extensive in the tropics than in temperate areas. Being formed under brackish water conditions, they support unique vegetation and are also a unique ecosystem. These soils occur in association with the peats or Histosols. The characteristic feature of acid sulfate soils is the presence of a mineral called pyrite (FeS_2). In the Entisols (sulfaquents), the process of FeS_2 formation is active. The Entisols commonly occur close to tidal flats and are generally continuously reduced. The bearing capacity of such soils is low, and due to the presence of FeS_2 , the soils are not cultivated. Drainage of such soils, or where the soils have been naturally drained by downcutting of rivers, results in oxidation of the upper part of the soil with a concomitant ripening process. The FeS_2 oxidizes and alters to a yellow mineral called jarosite and in the process liberates sulfuric acid that lowers soil pH. This chemical transformation is considered to be the greatest acidity change in soils. The resultant soil is an Inceptisol (sulfaquept) and has the dubious status of being the most acid soil. Many sulfaquepts are used for rice production with low production and even crop failures.

Large areas of the tropics (Table 1) are covered with sandy soils (Psamments). The largest extent is in the desert margins of the Sahara and the Kalahari, in Africa. The Kalahari sands have been blown eastward into parts of Botswana, Zimbabwe, and Zambia, where they bury former Ultisols and Oxisols. Toward the north, the Kalahari sands occur in Congo as large, broad tongues. Most of the soils on these broad deposits are Psamments, but in the depressions, they are associated with Spodosols.

The upland Inceptisols and Entisols occur on steep sloping land. The soils are shallow because of continuous downwearing by water erosion. The tropical udepts and ustepts and the orthents are commonly found under shrub or forests. Land conditions prevented their cultivation, although these soils are also being used for agriculture because of scarcity of land in many of the poorer countries.

VERTISOLS

Only the Vertisols^[6,7] of the tropics have been used for agriculture in low-input systems. These soils are sticky and plastic when wet and extremely hard when dry and so cannot be managed with the traditional bullock-drawn plows. In India, e.g., when mechanical power became

available, the soils were only planted at the end of the rainy season.

Vertisols are very productive soils for agriculture.^[8] The pH is between 6 and 7.5, and the soil is well supplied with nutrients. Vertisols do not have adverse quantities of toxic elements and are not deficient in any nutrients, but they must be supplied with N and P for sustained productivity. They are derived from basic parent materials, and the largest extent is in India (on the Deccan Plateau) and bordering the Nile in Sudan.

OTHER SOILS

Mollisols, Aridisols, Histosols, Spodosols, and Andisols occur in small areas in the tropics. Mollisols are present in the cooler areas, with an isothermic soil temperature regime. Tropical Aridisols form a small band in Africa and in India. Their properties are similar to Aridisols of other biomes.

The Histosols^[9] are very extensive in Southeast Asia and mainly in Sumatra and Kalimantan of Indonesia. They are found on the coastal plains of these islands. They also occur at similar physiographic settings in the other countries. Unlike the Histosols of northern latitudes, the tropical Histosols are woody and have buried timber. Some also have acid sulfate features. These soils are extremely difficult to manage because of their very acid nature and the many nutrient (specifically, trace element) deficiencies. The beach ridges on the coast create a basin inland where the Histosols form under aerobic conditions. Associated with the Histosols are the Spodosols of the coastal platforms in the tropics. Spodosols are formed on sands and have a bleached surface E horizon with an organic matter rich subsurface horizon. A lateral seepage of organic colloid-rich water from the peat swamps into the sand result in the formation of a “spodic-like” B horizon.

The Andisols occur on the circum-Pacific belt of volcanoes. They are present in all countries of this orogenic zone. Generally, they share the properties of the temperate Andisols but have a lower content of organic matter. Some of the Andisols with udic soil moisture regimes, particularly those occurring on the older surfaces, have a net positive charge, similar to the Acrudox. The upper part of the soil may have a net negative charge as long as there is organic matter, and

so organic matter management is crucial to the productivity of these soils.

REFERENCES

1. Soil Survey Staff. *Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; U.S. Dept. Agric. Handbook 436; Government Printing Office: Washington, 1999.
2. Eswaran, H.; Beinroth, F.H.; Kimble, J.; Cook, T. Soil diversity in the tropics: implications for agricultural development. In *Myths and Science of Soils of the Tropics*; Lal, R., Sanchez, P.A., Eds.; Special Publ. No. 29; Publ. Soil Sci. Soc. Amer.: Madison, 1992; 1–16.
3. Buol, S.W.; Eswaran, H. Oxisols. *Adv. Agron.* **2000**, *68*, 152–195.
4. Lim, J.S.; Khalsi, M.S.; Eswaran, H. Alfisols and Ultisols with low activity clays in Malaysia. In *Proceedings of Second International Soil Classification Workshop. Part II*, Beinroth, F.H., Panichapong, S., Eds.; Publ. Dept. of Land Development: Bangkok, 1979; 107–118.
5. Eswaran, H.; De Coninck, F.; Varghese, T. Role of plinthite and related forms in soil degradation. *Adv. Soil Sci.* **1990**, *11*, 109–128.
6. Dudal, R.; Eswaran, H. Distribution, properties and classification of Vertisols. In *Vertisols: Their Distribution, Properties, Classification and Management*; Wilding, L.P., Puentes, R., Eds.; Publ. Soil Management Support Services, US Department of Agriculture, Natural Resources Conservation Service: Washington, 1988; 1–22.
7. Mermut, A.R.; Padmanabhan, E.; Eswaran, H.; Dasog, G.S. Pedogenesis. In *Vertisols and Technologies for Their Management. Developments in Soil Science Series*; Ahmad, N., Mermut, M., Eds.; Elsevier Science: Burlington, MA, 1997; Vol. 24, 43–61.
8. Eswaran, H.; Virmani, S.M.; Abrol, I.P. Issues and challenges of dryland agriculture in southern Asia. In *Agriculture and the Environment: Bridging Food Production and Environmental Protection in Developing Countries*; Juo, A.S.R., Freed, R.D., Eds.; ASA Special Publication No. 60; American Society of Agronomy: Madison, 1995; 161–180.
9. Eswaran, H. Classification and management of Histosols. In *Proceedings of the 2nd International Symposium on Peat Land Soils, Thailand/Malaysia*, Eswaran, H., Panichapong, S.M., Eds.; Publ. Dept. of Land Development: Thailand, 1986; 15–26.

Turbations

Brad D. Lee

Department of Plant and Soil Sciences, University of Kentucky, Lexington, Kentucky, U.S.A.

Robert C. Graham

Soil and Water Sciences Program, University of California, Riverside, California, U.S.A.

Abstract

Mixing of soil material through natural mechanical action is collectively referred to as turbation. The turbations of some kind operate in most soils. Often effects are minimal but in some soils, turbations are a dominant process that strongly impacts the soil morphology and may even overwhelm the effects of other processes. Turbations can be categorized as bioturbations, cryoturbations, and pedoturbations. Bioturbation is the mixing of soil material by organisms. It can occur slowly through root growth or quickly through tree throw, burrowing invertebrates and mammals. Bioturbation by mammals and invertebrates promotes the formation of thick, organic-rich A horizons and retards the differentiation of illuvial horizons. Cryoturbation, the dominant soil process in many Gelisols, is the mixing of soil material through freeze–thaw processes. This process orients rock fragments within the soil, transports organic-rich A horizon material into the subsoil, and disrupts the lateral continuity of soil horizons. Pedoturbation is the disruption of soil horizonation by the self-mixing of soil material. This process is common in smectite-rich soils with a high shrink–swell capacity where surface material falls into large, deep cracks during dry climatic periods. The most extreme examples of this process in clay-rich soils are Vertisols. Pedoturbation also occurs in petrocalcic horizons. The precipitation of calcium carbonate within voids in these cemented horizons causes expansion and internal shattering, resulting in cracks that are sites for more calcite precipitation, and the process is continued.

INTRODUCTION

Various processes act within soils to physically disturb and mix the soil material. These natural processes are in general referred to as *turbations* and prefixes are added to more specifically categorize them as *bioturbations*, *cryoturbations*, and *pedoturbations*. The turbations of some kind operate in most soils. Often their effects are relatively minimal, and the development of soil horizons is primarily under the influence of such processes as leaching and illuviation. In some soils, turbations are a dominant process that strongly impacts the soil morphology and may even overwhelm the effects of other processes. The mechanisms responsible for the different kinds of turbations, and the resulting effects on the soils in which they operate, are explored in the following sections.

BIOTURBATION

Mixing of soil caused by organisms, either flora or fauna, is known as bioturbation. Soil is the basic rooting medium for

plants and, while root growth itself disturbs soil fabric, the process is generally so slow and localized that a mixing effect is not obvious. A much greater impact results from tree throw, i.e., the uprooting of trees, usually by wind. When a tree falls, a root wad, consisting of soil and rock enmeshed by coarse roots, is pulled out of the ground (see Fig. 1). The soil material held in the root wad is eventually eroded and released in a mixed heap on the soil surface. Forest fires speed the release process by burning the fallen tree and its roots. Where tree throw is common, relatively weakly developed soils result in a hummocky landscape.^[1] In some cases, tree throw lifts soil horizons intact and lays them back as an inverted profile rather than fully mixing them.^[2]

Animals digging in or burrowing through soil can be very effective mixing agents. The mixing power of earthworms has long been recognized.^[3] Earthworms bring organic material from the surface down into the mineral soil. As they travel through the soil, they ingest, and intimately mix, both organic and inorganic materials throughout the zone in which they are active. This generally produces a thick, dark, organic-rich A horizon (mollic or umbric epipedon) and can counteract the



Fig. 1 An example of bioturbation: Tree throw has lifted root wads of soil and rock that will eventually be deposited in a mixed mound on the soil surface.

development of illuvial horizons.^[4] Bioturbation has also been attributed to other invertebrates, including ants, termites, and crayfish.^[5]

Burrowing mammals are powerful bioturbation agents. Pocket gophers are among the most active burrowers, as they continually explore the upper several decimeters of soil for food and habitat. Other burrowers, such as prairie dogs and ground squirrels, have more fixed tunnel systems but still effectively mix the soil, often digging into underlying soft bedrock, thereby deepening the soil–bedrock boundary.^[6] Some predators, such as badgers, coyotes, and bears, pursue these small burrowing animals by attempting to excavate them, and in the process, they disrupt even larger volumes of soil. Bioturbation by mammals, as with earthworms, tends to promote the formation of thick, organic-rich A horizons and to retard the strong differentiation of illuvial horizons. Bare soil thrown out of burrows is subject to raindrop impact and erosion, but thorough mixing of the surface soil, as accomplished by gophers, may improve aeration and water infiltration.

Bioturbation by mammals can have direct impacts on environmental quality and cultural resources. Burrowing can move environmental contaminants from the surface to deeper within the soil^[7] and can break through subsurface barriers constructed to retain contaminants in landfills.^[8] Pocket gopher and other macrofauna activity in archaeological sites can destroy the stratification of artifacts, confounding interpretations.^[9]

CRYOTURBATION

The churning and mixing of soils caused by freezing and thawing is known as cryoturbation. Freezing of water in the soil causes expansion, subsequent thawing causes collapse, and together the resulting forces mix the soil in complex patterns (Fig. 2). Cryoturbation

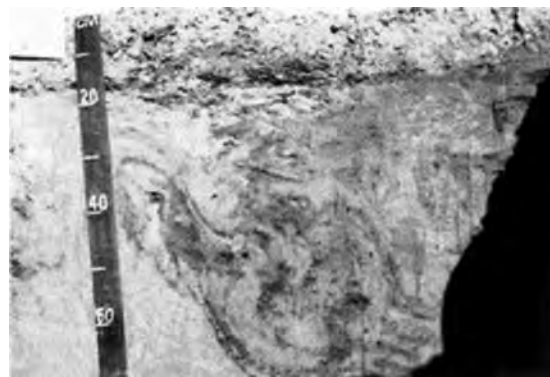


Fig. 2 An example of cryoturbation: Freeze–thaw processes in a Gelisol have mixed organic matter into the subsoil in convoluted patterns. Scale is in centimeters.

Source: Photo provided by J.G. Bockheim.

dominates in many Gelisols, where an “active layer” overlies the permafrost. The active layer, usually <2 m thick, is frozen during the cold season but thaws in the summer and is a zone of intense cryoturbation.^[10] During freezing, liquid water migrates to freezing fronts, both near the surface and at the underlying permafrost, and the soil mass expands at those locations when the water freezes. Cryoturbation orients rock fragments within the soil, transports organic-rich A horizon material into the subsoil and upper permafrost, and disrupts the lateral continuity of soil horizons.^[11] Ice wedges in the permafrost also promote cryoturbation.^[12] When soil temperatures drop below -30°C in the winter, the already solid ice in the permafrost recrystallizes and shrinks by 3%, forming vertically oriented cracks. In the spring, water from the active layer migrates into these cracks and freezes, with a 9% increase in volume (at temperatures between 0°C and -30°C). The resulting ice wedges slowly expand over the years and generate subsoil pressures that must be resolved in an upward direction, thereby contributing to cryoturbation. The effects of cryoturbation in permafrost areas can also be seen at the soil surface in the form of patterned ground, with polygonal nets of sorted stones, frost-heave ridges, and frost boils.

Although cryoturbation is most strongly expressed in permafrost regions, it also occurs in temperate zone soils where freezing is limited to the upper part of the soil. Frost heave churns the upper several centimeters of these soils, lifting rock fragments and uprooting seedlings. The effect is pronounced where the soil is bare, not insulated by snow cover or leaf litter.

PEDOTURBATION

Some soils undergo substantial swelling when they are wetted and shrink upon drying. The resulting disruption

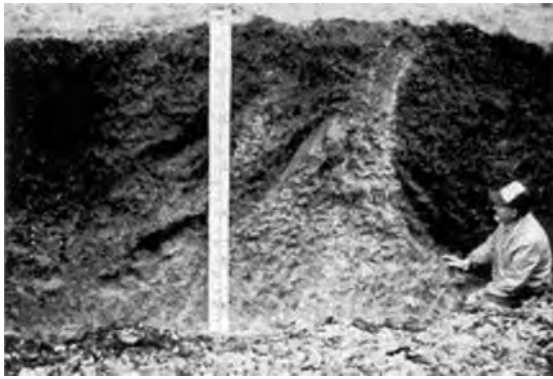


Fig. 3 An example of pedoturbation: Shrink–swell processes in a Vertisol have forced lighter-colored subsoil material toward the surface.

Source: Photo provided by T.D. Cook.

of the soil is called pedoturbation and is generally associated with clayey soils that are rich in smectite, the most extreme expressions of which are Vertisols. When the soil swells, often with differential wetting patterns,^[13] its fabric is sheared upward along slickensides, bringing subsoil material toward the surface (Fig. 3).^[14] When the soil dries and shrinks, cracks are formed that can be in the order of 10 cm wide at the soil surface and extend to over a meter in depth. These cracks promote mixing because surficial materials fall into them and are thus added directly to the subsoil.^[15] The morphology of Vertisols is dominated by the effects of pedoturbation, but the mixing is not necessarily so rapid that it obliterates horizons formed by leaching and illuviation, such as calcic horizons.^[16] As with bioturbation, pedoturbation in Vertisols can rapidly mix anthropogenic contaminants to some depth.^[17]

Pedoturbation is not restricted to Vertisols and other clayey soils. Minor shrink–swell behavior is present in most soils, and to some degree, it can alter the soil fabric. Narrow shrinkage cracks (<5 mm) under desert pavements allow eolian dust to infiltrate into shallow B horizons where it is incorporated into the soil fabric, slowly building the soil upward.^[18] In another example, the rock fabric of granodiorite was disrupted and converted to soil fabric by minor shrink–swell activity in the uppermost part of the weathered bedrock profile.^[19] Another type of pedoturbation happens in strongly developed petrocalcic horizons. The precipitation of calcium carbonate within voids in these cemented horizons causes expansion and internal shattering, whereupon the resulting cracks are sites for more calcite precipitation, and the process is continued. The result is a petrocalcic horizon composed of turbated and re cemented angular fragments of the original petrocalcic horizon.^[20]

CONCLUSION

Many of the mechanisms and effects of turbation on soil morphology have been explored, but the broad-scale distribution of soils impacted by tree throw and the extent of turbation effects within a soil profile have not been well documented. The mechanisms of turbation are climate dependent (macroflora and burrowing macrofauna distribution, ice wedges and freezing depth, and wetting–drying cycles). Climatic changes will impact the quantity of soils impacted by turbation, the prevalent type of turbation, and the extent to which turbation effects soil development. Pedologic investigations into the extent of turbation in varying climatic conditions are needed to predict how turbation will impact soil development in the future.

REFERENCES

1. Meyers, N.L.; McSweeney, K. Influence of treethrow on soil properties in northern Wisconsin. *Soil Sci. Soc. Am. J.* **1995**, *59*, 871–876.
2. Schaetzl, R.J. Complete soil profile inversion by tree uprooting. *Phys. Geogr.* **1986**, *7*, 181–189.
3. Darwin, C. *The Formation of Vegetable Mould through the Action of Worms, with Observations on Their Habits*; John Murray: London, 1881.
4. Graham, R.C.; Wood, H.B. Morphologic development and clay redistribution in lysimeter soils under chaparral and pine. *Soil Sci. Soc. Am. J.* **1991**, *55*, 1638–1646.
5. Hole, F.D. Effects of animals on soil. *Geoderma* **1981**, *25*, 75–112.
6. Munn, L.C. Effects of prairie dogs on physical and chemical properties of soils. In *Proceedings of the Symposium on the Management of Prairie Dog Complexes for the Reintroduction of the Black-Footed Ferret*; Oldemeyer, J.L., Biggins, D.E., Miller, B., Crete, R., Eds.; Biological Report 13; USDI–Fish and Wildlife Service: Washington, 1993; 11–17.
7. Mace, J.E.; Graham, R.C.; Amrhein, C. Anthropogenic lead distribution in rodent-affected and undisturbed soils in southern California. *Soil Sci.* **1997**, *162* (1), 46–50.
8. Suter, G.W.; Luxmoore, R.J.; Smith, E.D. Compacted soil barriers at abandoned landfill sites are likely to fail in the long term. *J. Environ. Qual.* **1993**, *22*, 217–226.
9. Balek, C.L. Buried artifacts in stable upland sites and the role of bioturbation: A review. *Geoarchaeology* **2002**, *17* (1), 41–51.
10. Bockheim, J.G.; Tarnocai, C. Gelisols. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; E-256–E-269.
11. Ping, C.L.; Michaelson, G.J.; Kimble, J.M.; Romanovsky, V.E.; Shur, Y.L.; Swanson, D.K.; Walker, D.A. Cryogenesis and soil formation along a bioclimate gradient in Arctic North America. *J. Geophys. Res.* **2008**, *113*, G03S12.
12. Rieger, S. *The Classification and Genesis of Cold Soils*; Academic Press: New York, 1983.
13. Weitkamp, W.A.; Graham, R.C.; Anderson, M.A.; Amrhein, C. Pedogenesis of a vernal pool Entisol–Alfisol–Vertisol

- catena in southern California. *Soil Sci. Soc. Am.* **1996**, *60*, 316–323.
14. Wilding, L.P.; Tessier, D. Genesis of Vertisols: Shrink–swell phenomena. In *Vertisols: Their Distribution, Properties, Classification and Management*; Wilding, L.P., Puentes, R., Eds.; Texas A & M University Printing Center: College Station, 1988; 55–81.
 15. Buol, S.W.; Hole, F.D.; McCracken, R.J.; Southard, R.J. *Soil Genesis and Classification*, 4th Ed.; Iowa State University Press: Ames, 1997.
 16. Southard, R.J.; Graham, R.C. Cesium-137 distribution in a California Pelloxerert: Evidence of pedoturbation. *Soil Sci. Soc. Am. J.* **1992**, *56*, 202–207.
 17. Graham, R.C.; Ulery, A.L.; Neal, R.H.; Teso, R.R. Herbicide residue distributions in relation to soil morphology in two California Vertisols. *Soil Sci.* **1992**, *153* (2), 115–121.
 18. Anderson, K.C.; Wells, S.G.; Graham, R.C. Pedogenesis of vesicular horizons, Cima Volcanic Field, Mojave Desert, California. *Soil Sci. Soc. Am. J.* **2002**, *66*, 878–887.
 19. Frazier, C.S.; Graham, R.C. Pedogenic transformation of fractured granitic bedrock, southern California. *Soil Sci. Soc. Am. J.* **2000**, *64*, 2057–2069.
 20. Birkeland, P.W. *Soils and Geomorphology*, 3rd Ed.; Oxford University Press: Oxford, 1999.

Turfgrass: Protection of Soil and Water

James A. Murphy

Department of Plant Science, Rutgers University, New Brunswick, New Jersey, U.S.A.

Stephanie L. Murphy

Soil Testing Laboratory, Rutgers University, New Brunswick, New Jersey, U.S.A.

Abstract

When man alters a landscape, changes in the soil and vegetation take place. Perennial grasses often play an important role in the structural redevelopment of the altered ecosystems. Turf is established and maintained in landscapes to enhance the environment and quality of life through stabilization and accelerated restoration of soil. Turfgrasses are frequently a part of the secondary succession on abandoned farmland and other disturbed landscapes that may take hundreds or thousands of years to restore the soil, depending on the loss or degradation of soil.

INTRODUCTION

Turfgrasses, turfs, and lawns often bring to mind connotations of a society's affluence or playgrounds of the idle rich; however, the turfed areas of many landscapes have more humble beginnings and play an important role in the protection of the earth's soil mantle. The origin of turfed areas in managed landscapes probably started along with man's effort to domesticate animals. Many centuries ago, human dwellings were close to the wilderness where predators roamed, and domesticated animals were herded and tethered close to dwellings to prevent escape and predation. Clearings eliminated cover for stalking predators, and grass provided incentive for the domesticated animals to stay put. Thus, a grazing system was developed that was necessary for survival of both man and his animals.^[1] Sheep and goats grazed the village green of ancient times, and the well-knit turf was esthetically inviting, encouraging residents to gather and socialize. These village greens became the squares or parks of many older towns and cities. Thus, this closely cropped system has become the basis for many landscapes around modern-day homes, businesses, roadways, in parks, and in other places of beauty. The total area of turfgrass in the United States has been estimated at 31 million acres, with more than 80% of the total in lawns.^[2]

FUNCTION OF TURF IN LANDSCAPES

The usefulness of closely cropped perennial grasses was probably recognized early in developing civilizations because of the important benefits in the stabilization and enhanced durability of soil under traffic. The ability to perform daily chores and recreational activities without interference from mud and dust would have been clearly recognized

by early humans. The fibrous roots of perennial grasses are recognized as the most effective means of stabilizing soil.^[3] This outstanding capacity of grasses to stabilize the soil mantle, particularly under repeated treading, enables humans to utilize the land in many ways, including parks, sports, playgrounds, and multiple other uses.^[4] The first airport runways were covered with turfgrasses until the aircraft became too large and heavy.^[5,6] Turfed runways are used for small aircraft and gliders in rural areas. The prehistoric Americans and homesteaders of the North American plains used sod to construct dwellings.^[7] Turf areas serve as recreational space for young and old. The grazed grounds around early human communities and homes probably provided for the first areas of play and sports. The game of lacrosse was modified from a sport played by Native Americans on grassed plains.^[7] Many sporting games including bowls, golf, and cricket have been played for centuries on turfgrass.^[8] Football, soccer, and baseball are the dominant sports that rely on the durability and safety of turfgrass fields. Many burial grounds are covered with turf, and many modern memorial gardens are designed into beautiful park-like settings with turf serving as the backdrop that ties together the landscape. Furthermore, a large proportion of nearly 4 million miles of public roads in the United States^[9] are bordered by low maintenance turf for reasons of safety as well as soil stabilization.

SOIL CONSERVATION

As the human population increases worldwide, the utilization of our land and soil resources continues to expand. Population growth tends to be greater in the developing countries and puts intense pressure on these societies to intensively utilize much of the available soil resources in

agricultural operations, housing, or infrastructure. Continued prosperity in developed countries fuels the expansion of suburban land development around redeveloping urban centers. These agricultural and construction activities of man result in extensive land disruption, which exposes soil to the erosive forces of wind and water. Soil erosion is a direct or contributing cause of declining soil productivity; destruction of land; sedimentation of streams, reservoirs, and storm drains; damage to roadways; mounting floods; and degradation of air and water quality.^[10] Although many individuals were conscious of the problems of soil erosion earlier, it was not until the early 1900s that the agricultural community in the United States began to regard soil conservation as fundamental to sound farming practice.^[11] Up to that time, vegetative methods of erosion control were given little attention and limited incidental use. Land development and use in urban centers have received greater attention for its impact on soil losses and the subsequent degradation of water quality, sedimentation of waterways, and severity of flooding.^[12,13]

Turfgrasses are playing an increasingly important role in enhancing our physical environment, conserving our soil and water resources, and providing recreation.^[14,15] There is considerable effort being expended to repair the landscape in China that was intentionally destroyed during the Cultural Revolution of the 1960s. Turf and trees are being replanted in China to halt and eventually reverse the severe environmental degradation of smog and dust pollution, soil erosion, water pollution, and heat buildup in cities.^[16] Turf provides a relatively inexpensive durable ground cover that protects the valuable non-renewable soil resource from water and wind erosion. Protection of soil by thick-growing vegetation takes place through several processes.^[17] One important way in which close-growing vegetation protects the land is through the interception of rainfall by the plant surfaces. This prevents the soil from the beating action of raindrops that dislodges soil particles and allows soil to move in water runoff. This positive effect can be frequently observed in runoff that is clear from turfed areas, whereas water runoff from land under cultivation or development is laden with silt, clay, organic matter, nutrients, and other materials potentially present in soil.

The maintenance of high humus or organic matter content in the soil is another benefit of keeping land in grass cover. Disruption of land is known to rapidly destroy soil organic matter. Organic matter is critical to create or maintain soil structure, the system of aggregates and pores in soil. Soils high in organic matter content typically have a rate of water infiltration that is considerably greater than that in soil low in organic matter.^[18] Greater infiltration of water into soil subsequently reduces the amount of runoff, further reducing the erosion hazard.

Another aspect of grass cover in erosion control is that it slows the movement of water over the land surface, thus reducing the erosive power of the water. Slower movement of water also provides greater time for water to infiltrate the

soil, thus reducing runoff. The dense cover of vegetation provided by turfgrass also reduces the wind velocity at the soil surface compared to bare soil and hence is effective in reducing erosion by wind; taller vegetation can reduce wind velocity to a greater degree; erosion control may depend on the extent of bare areas between plants. Grass roots are better than the roots of annual weeds and other plant species in binding soil particles together.^[3,19] These principles of reducing soil erosion—including raindrop interception, improved infiltration, slowing runoff, and binding of soil particles—in grassland systems have proven to be just as effective in turfgrass settings.^[20–23]

WATER QUALITY

Non-point source pollution is considered an important cause of decreased water quality in the United States.^[13] Soil erosion losses are a significant component of non-point source pollution. Erosion depletes land of important nutrients and organic matter content and removes the most physically suitable soil for biologic sustenance. The eroded sediment that makes its way into streams, reservoirs, and other water bodies carries nutrients such as nitrogen and phosphorus that can significantly reduce the water quality. Close-growing vegetation, including turfgrasses, plays an essential role in capturing water and filtering the sediment and nutrients in runoff.^[20,24] Turf is recommended as the vegetation to enhance environmental quality in urbanized landscapes through the stabilization of exposed soil; slowing of runoff water; promotion of infiltration; and removal of pollutants such as sediment, organic matter, nutrients, and heavy metals from runoff water.^[13,25]

SOIL ORGANIC CARBON

The loss of organic matter (carbon) from cultivated soils and other man-induced activities contributes to increased atmospheric carbon dioxide (CO₂), one of the so-called greenhouse gases.^[26] Perennial grasses are important components of ecosystems that conserve or enhance soil organic matter,^[27,28] and it is thought that a grassland system can be managed to lessen the increases in atmospheric CO₂ by a process called carbon sequestration.^[29] Perennial grass cover on soil is more effective than cover by weedy species in maintaining high soil organic matter levels.^[27]

CONCLUSION

When man alters a landscape, changes in the soil and vegetation take place. Perennial grasses often play an important role in the structural redevelopment of the altered ecosystems. Turf is established and maintained in landscapes to enhance the environment and quality of life through stabilization and accelerated restoration of soil. Turfgrasses are

frequently a part of the secondary succession on abandoned farmland and other disturbed landscapes that may take hundreds or thousands of years to restore the soil, depending on the loss or degradation of soil. Other benefits of turfgrasses to humans and the environment include temperature moderation, reduction of noise and glare, reduction of pest populations, safety via firebreaks and increased visibility zones on roadsides, airfields and security-sensitive locations, and creation of surfaces for sport and leisure activities, as well as esthetic benefits.^[15]

REFERENCES

- Hillel, D. *Out of the Earth. Civilization and the Life of the Soil*; University of California Press: Berkeley, 1992.
- The Lawn Institute. *How the Environment Benefits from a Well-Maintained Lawn*. Available at http://www.turfgrasssod.org/lawninstitute/environmental_benefits.htm.
- Carter, M.R.; Angers, D.A.; Kunelius, H.T. Soil structural form and stability and organic matter under cool-season perennial grasses. *Soil Sci. Soc. Am. J.* **1994**, *58*, 1194–1199.
- Asay, K.H.; Sleper, D.A. Contributions from breeding forage and turf grasses—An overview. In *Contributions from Breeding Forage and Turf Grasses*; Sleper, D.A., Asay, K.H., Eds.; CSSA: Madison, 1989; Vol. 15, 1–20.
- Morrish, R.H.; Alton, A.E.; Cale, E.B. Airfields and flight strips. In *Grass: The Yearbook of Agriculture 1948*; Stefferud, A., Ed.; USDA, U.S. Government Printing Office: Washington, 1948; 319–323.
- Morrish, R.H.; Harrison, C.M. The establishment and comparative wear resistance of various grasses and grass-legumes mixtures to vehicular traffic. *Agron. J.* **1948**, *40*, 168–179.
- Roberts, E.C.; Huffine, W.W.; Grau, F.V.; Murray, J.J. Turfgrass science—Historical overview. In *Turfgrass*; Waddington, D.V., Carrow, R.N., Shearman, R.C., Eds.; Agron. Monogr. ASA, CSSA, and SSSA: Madison, 1992; Vol. 32, 1–27.
- Beard, J.B. *Turfgrass: Science and Culture*; Regents/Prentice Hall: Englewood Cliffs, 1973.
- U.S. Department of Transportation/Federal Highway Administration. *Highway Statistics 2003: Public Road Length—2003*. Available at <http://www.fhwa.dot.gov/policy/ohim/hs03/pdf/hm20.pdf>.
- National Resource Conservation Service. *National Resource Inventory: 2001 Annual NRI*; 1997. Available at <http://www.nrcs.usda.gov/technical/land/nri01/nri01eros.html>.
- Bennett, H.H.; Lowdermilk, W.C. USDA, General aspects of the soil-erosion problem. In *Soils and Men: Yearbook of Agriculture 1938*; USDA, U.S. Government Printing Office: Washington, 1938.
- U.S. Environmental Protection Agency. *Nonpoint Source Pointers (Factsheets)*; 1996. Available at <http://www.epa.gov/OWOW/NPS/facts> (Revised August 18, 2003).
- U.S. Environmental Protection Agency. *National Water Quality Inventory: 1996 Report to Congress*; 1996. Available at <http://www.epa.gov/305b/2000report.html> (accessed March 2001).
- Meyer, W.A.; Funk, C.R. Progress and benefits to humanity from breeding cool-season grasses for turf. In *Contributions from Breeding Forage and Turf Grasses*; Sleper, D.A., Asay, K.H., Eds.; Spec. Publ. 15; CSSA: Madison, 1989; Vol. 15, 31–48.
- Beard, J.B.; Green, R.L. The role of turfgrasses in environmental protection and their benefits to humans. *J. Environ. Qual.* **1994**, *23*, 452–460.
- International Turf Producers Foundation. In *Water Right—Conserving Our Water Preserving Our Environment*; Rolling Meadows, IL 2001. Available at <http://www.turfgrasssod.org/Water.pdf>.
- Enlow, C.R.; Musgrave, G.W. USDA, Grass and other thick-growing vegetation in erosion control. In *Soils and Men: Yearbook of Agriculture 1938*; USDA, U.S. Government Printing Office: Washington, 1938; 615–633.
- Smith, F.B.; Brown, P.E.; Russell, J.A. The effect of organic matter on the infiltration capacity of Clarion loam. *J. Am. Soc. Agron.* **1937**, *29*, 521–525.
- Weaver, J.E.; Harmon, G.W. *Quantity of Living Plant Materials in Prairie Soils in Relation to Runoff and Soil Erosion*; Bull. 8; Nebr. Univ. Conserv. Dept.: Nebraska, 1935.
- Gross, C.M.; Angle, J.S.; Welterlen, M.S. Runoff and sediment losses from turfgrass. *J. Environ. Qual.* **1990**, *19*, 663–668.
- Gross, C.M.; Angle, J.S.; Hill, R.L.; Welterlen, M.S. Runoff and sediment losses from tall fescue under simulated rainfall. *J. Environ. Qual.* **1991**, *20*, 604–607.
- Morton, T.G.; Gold, A.J.; Sullivan, W.M. Influence of overwatering and fertilization on nitrogen losses from home lawns. *J. Environ. Qual.* **1988**, *17*, 124–130.
- Watschke, T.L.; Mumma, R.O. *The Effect of Nutrients and Pesticides Applied to Turf on the Quality of Runoff and Percolating Water*; ER 8904; Pennsylvania State University, Environmental Resources Research Institute: University Park, 1989.
- Mendez, A. Sediment and nitrogen transport in grass filter strips. *J. Am. Water Resour. As.* **1999**, *4*, 867–875.
- U.S. Environmental Protection Agency. In *Erosion, Sediment, and Runoff Control for Roads and Highways*; USEPA 841-F-95-008d; U.S. Government Printing Office: Washington, 1995. Available at <http://www.epa.gov/OWOW/NPS/education/runoff.html>.
- Lal, R.; Kimble, J.M. Conservation tillage for carbon sequestration. *Nutr. Cycl. Agro-ecosys.* **1997**, *49*, 243–253.
- McHenry, J.R. Influence of some perennial grasses on the organic matter content and structure of an eastern Nebraska fine-textured soil. *J. Am. Soc. Agron.* **1947**, *39*, 981–994.
- Paustian, K. Agricultural soils as a sink to mitigate CO₂ emissions. *Soil Use Manage.* **1997**, *13*, 230–244.
- Malhi, S.S.; Nyborg, M.; Harapiak, J.T.; Heier, K.; Flore, N.A. Increasing organic C and N in soil under bromegrass with long-term N-fertilization. *Nutr. Cycl. Agro-Ecosys.* **1997**, *49*, 255–260.

Ultisols

Joey N. Shaw

Agronomy and Soils, Auburn University, Auburn, Alabama, U.S.A.

Abstract

The presence of Ultisols suggests appreciable pedogenic development, although preweathered parent materials often confound this relationship. Although not extensive, these soils can be found in many parts of the world. Unfortunately, some of these soils have been abused since the 19th century and have limited genetic ability to rebound. In some parts of the world, shifting agriculture is practiced on Ultisols because amendment application is not possible. This shifting cultivation is mostly caused by limitations in soil chemical properties and not by poor physical properties or climate. If managed properly, Ultisols are capable of providing and sustaining substantial food and other resources.

INTRODUCTION

Soils classified in the Ultisol order possess a subsurface horizon (argillic or kandic horizon) with more clay than overlying horizons and low base saturation [the percentage of exchangeable calcium (Ca), magnesium (Mg), potassium (K), and sodium] in the lower portion of the soil. In some regions, Ultisols commonly possess dense, root, and water-restrictive subsurface horizons (fragipans), and hardened accumulations of iron (Fe) oxides mixed with other minerals (plinthite and ironstone). Ultisols are found in all climates except deserts. Ultisols occupy approximately 9.2% of the earth's land surface, are concentrated in tropical regions (80%), and are found in South America, Africa, China, Australia, the Pacific Islands, coastal India, and the southeastern United States^[1,2] (Fig. 1). Conceptually, the presence of Ultisols suggests: 1) in many cases, an advanced degree of soil weathering; 2) the land surface has been stable for a significant time; 3) long-term periodic wetting and drying of the soil; and 4) the soils are relatively infertile and have limited ability to replenish nutrients through inherent mechanisms. Although extensive cultivation of these soils occurs, Ultisols require that soil fertilizers and lime be added for sustainable crop production. Readers are encouraged to refer the works by West^[1], Miller^[3], Buol et al.^[4] and West et al.^[5] for detailed information.

MORPHOLOGY OF ULTISOLS

An idealized soil horizon sequence for a well-drained Ultisol consists of a darkened surface horizon (A) where soil organic matter (SOM) has accumulated; a lightened or bleached horizon (E) where materials have been removed;

a subsurface horizon (B) where clay, Fe, and aluminum (Al) oxides have concentrated; and an underlying horizon (C) that is relatively unweathered and comprised mostly of soil parent material. Transitional horizons exhibiting characteristics of both over- and underlying horizons are common.

SOM in Ultisols

Most Ultisols occur in temperate or tropical regions that possess a finite capacity to accumulate SOM due to aerobic microbial activity. Ultisols typically possess less SOM than Mollisols, Oxisols, or Vertisols, but often possess quantities comparable to Alfisols.^[1] Surface horizons (A) of poorly drained (wet) members (Aquults) as well as some Ultisols found either at high elevations or in portions of the tropics (humults) contain relatively high SOM quantities.

Horizons of Ultisols

The A and E horizons comprise the eluvial portion of the soil where leaching and erosion have removed both soluble and colloidal soil constituents. The B and corresponding transitional horizons are considered the zone of illuviation where materials have accumulated. Besides possessing higher amounts of clay than overlying horizons, most B horizons of Ultisols possess translocated clay as evidenced by clay films, coatings on ped surfaces, or clay bridges between sand grains. These subsurface horizons qualify as argillic horizons, which in combination with a low base saturation, are diagnostic of Ultisols. Kandic horizons, the other subsurface diagnostic horizon commonly found in the southeastern U.S. Ultisols, may not possess clay films.

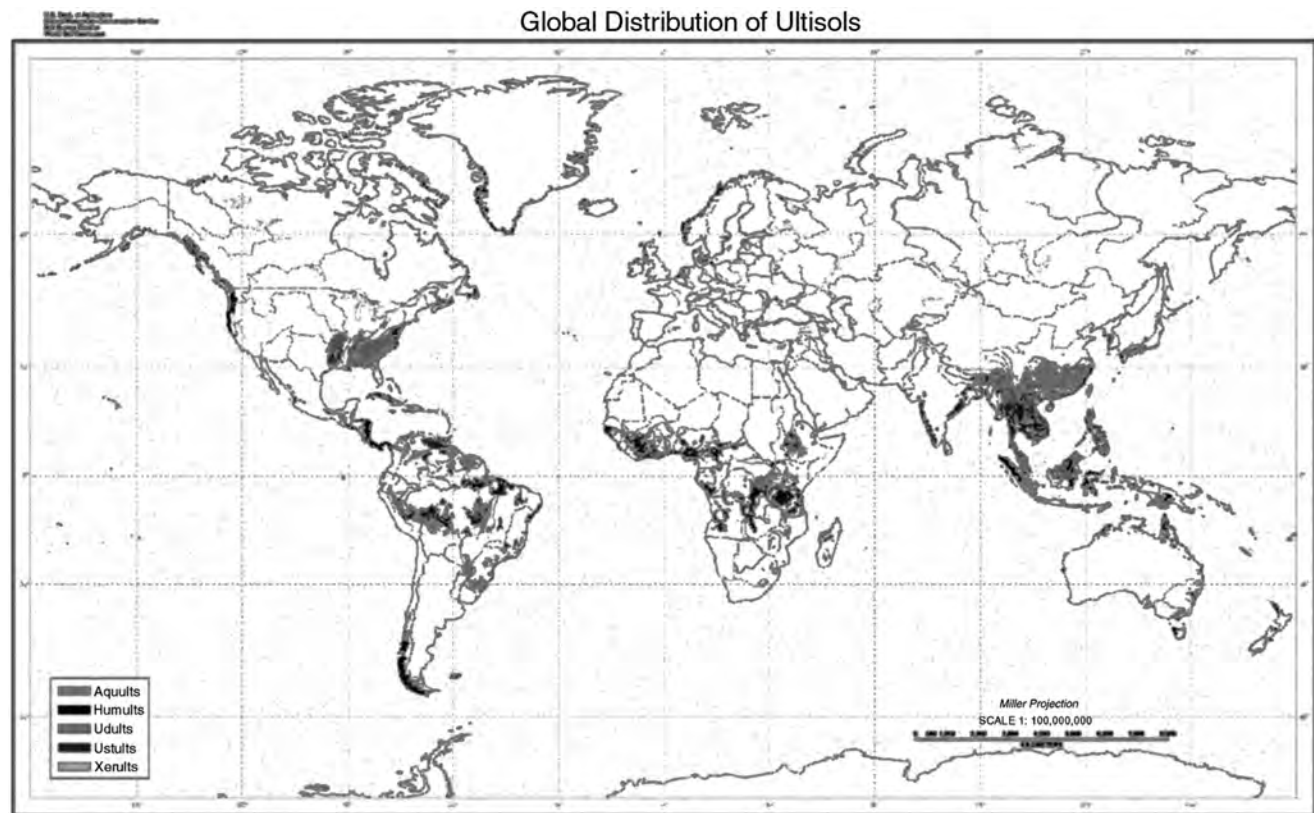


Fig. 1 Ultisol suborder distribution.

Source: From Soil Survey Staff.^[2]

Argillic and Kandic Horizons

Argillic and kandic horizons possess more clay than the overlying eluvial horizons. Argillic horizons are thought to form through soil wetting, colloid dispersion, downward movement of clays from eluvial horizons, and truncation of movement in illuvial horizons induced by soil drying or other mechanism. Clay films are often better expressed in the lower portions of the argillic horizon because of either 1) destruction of shallow films over long periods due to eluviation or 2) mixing and destruction (bioturbation) of clay films by organisms.^[4] Because the destruction of clay films occurs in older soils, kandic horizons sometimes do not exhibit evidence of translocated clay. Although clay translocation is a formative process in Ultisol development, most soil clays likely move only short distances (few centimeters), and the majority of argillic horizon clay is derived *in situ* through mineral weathering.^[3] In addition, parent materials possessing high clay content (e.g., fine-textured fluvio-marine sediments, shales, and argillaceous limestones) contribute significant subsurface clay. Argillic horizons of well-drained Ultisols also typically possess Fe and Al oxide minerals. Argillic horizons range in color from deep red (10R hues) to yellow (5Y hues) to gray depending mostly on the quantity, type, and degree of hydration of the Fe oxides present.

C horizons of Ultisols consist mostly of parent materials that include, but are not limited to, saprolite formed from igneous and metamorphic rocks, fluvio-marine sediments, wind-blown Loess materials, local alluvium or colluvium, and thin, slightly weathered layers of sedimentary rocks (e.g., limestone, shale, or sandstone).

SOIL-FORMING FACTORS

State Factor Approach

The state factor approach proposed by Jenny^[6] surmises that soil formation is a function of climate, organisms, relief, and parent materials, acting over time. Parent materials on a local scale and climate on a regional scale, coupled with stable landscapes (function of relief), are considered the critical factors for Ultisol formation.^[3]

Parent material

Because Ultisols possess low amounts of exchangeable bases (primarily Ca, Mg, and K) by definition, parent materials low in both bases and weatherable minerals can induce their development;^[1] e.g., Gamble and Daniels^[7] suggested the weathered status of fluvio-marine parent materials of

the North Carolina Coastal Plain mandates that Ultisols be the dominant order of that region. However, Ultisols also commonly develop on parent materials high in bases and weatherable minerals.^[4] In this situation, humid environments with seasonal surpluses of moisture enhance mineral weathering and base leaching. Therefore, seasonal drying and wetting of soils facilitate argillic horizon formation, base leaching, and subsequently Ultisol development.^[3,5]

SOIL CHEMICAL PROPERTIES

Base Saturation in Ultisols

The low base criterion required for Ultisol placement (< 35% base saturation in the lower portion of the solum by the pH 8.2 sum of cations method^[2]) was originally designed to separate soils with argillic horizons that could sustain prolonged cultivation without amendment addition (Alfisols) and soils that require nutrient additions (Ultisols).^[8] In reality, the base saturation criteria are largely for classification purposes, and the genetic and fertility significance of this value is unknown.^[4] Typically, base saturation decreases with depth in Ultisols, whereas it increases with depth in Alfisols (Fig. 2). Conceptually, base recycling by vegetation is thought to control base distribution within Ultisols, whereas base recycling coupled with mineral weathering is thought to sustain bases in Alfisols. The base saturation criteria have consistently separated soils formed in acid, felsic terrains (Ultisols), from soils derived from ultramafic parent materials (primarily Alfisols) in the southeastern U.S. Piedmont.^[4]

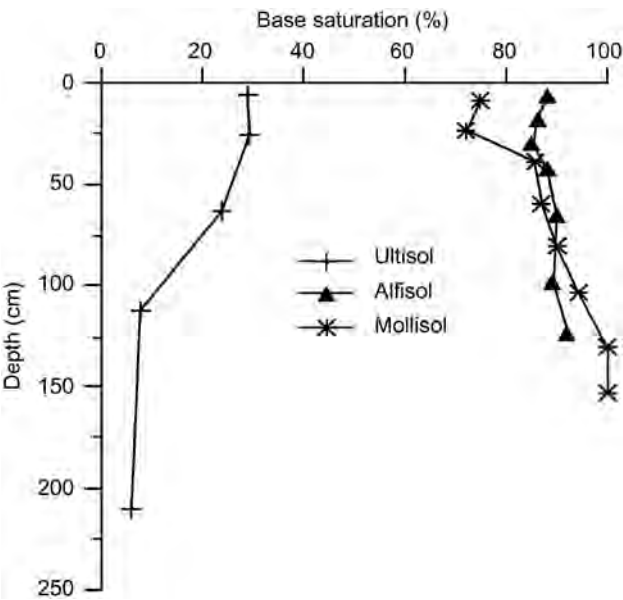


Fig. 2 Base saturation (%) for soils with argillic horizons representing the Ultisol (Typic Kandiodult), Alfisol (Typic Haplustalf), and Mollisol (Pachic Argiustoll) orders.
Source: From Soil Survey Staff.^[2]

Table 1 Response of crop yields to lime application for selecting Ultisols.

Soil	Crop	Yield (kg/ha)	
		Unlimed	Limed
Typic Paleudults	Seed-cotton ^[10]	990	1920
Typic Hapludults	Seed-cotton ^[10]	2430	2800
Typic Paleudults	Corn-grain ^[11]	2970	3920
Typic Hapludults	Corn-grain ^[11]	4700	5100
Typic Paleudults	Soybean-grain ^[12]	750	2030
Typic Hapludults	Soybean-grain ^[12]	1210	2550
Typic Paleudults	Wheat-grain ^[9]	1140	1610
Typic Hapludults	Wheat-grain ^[9]	1710	3020

Typic Paleudult is from the Coastal Plain Province of Alabama; Typic Hapludult is from the Appalachian Plateau Province of Alabama.
Source: Adapted from Adams.^[9]

Soil Acidity

Ultisols are typically acidic (pH between 4.5 and 5.5) and possess relatively high exchangeable Al. In acid soils, the synergistic effects of Al, hydrogen, and manganese toxicity, coupled with Ca, Mg, and molybdenum deficiency, result in reduced root growth and a concomitant yield decrease.^[9] Therefore, increased yields upon lime application are common for Ultisols (Table 1). This is in part caused by the amelioration of Al toxicity because of the precipitation of Al forms at pH > 5.5.

COMMON MINERALS

Low-Activity Minerals

Kaolinite, hydroxy-interlayered minerals, and Fe and Al oxyhydroxides are commonly found in the clay fraction of Ultisols. Sand and silt fractions are often dominated by quartz, with lesser quantities of mica, feldspars, ferromagnesian, and other weatherable minerals. This results in soils with a relatively limited capacity (compared to most other orders) to 1) sorb positively (+) charged nutrients [termed cation exchange capacity (CEC)]; and 2) supply bases through mineral weathering. Similarly, kandic horizons are defined by having higher clay than eluvial horizons and an extremely low CEC. Conversely, Ultisols with high Fe and Al oxide contents are capable of retaining negatively (–) charged species such as phosphate. The adsorption of phosphate on Fe oxides occurs both as a specific attraction and as a ligand exchange governed mainly by pH.

Smectites in Ultisols

Although Ultisols with low-activity clays are the most typical, areas of Ultisols containing appreciable smectite

with relatively high CEC have been found in the southeastern U.S. Coastal Plain.^[13] In this situation, the smectite is inherited from the parent material.

SOIL MANAGEMENT AND USE

Cultivation of Ultisols

The southeastern U.S. Piedmont is an example of an anthropogenically degraded soil resource. The majority of this region was in intensive cotton cultivation subsequent to European settlement. Sandy loam-textured surface horizons with appreciable SOM blanketed the region. However, monoculture cultivation and poor erosion control resulted in the destruction of these surface horizons and a degradation in this region's ability to produce row crops viably. Subsequently, this region is predominantly in pulpwood production. Forest harvesting of Ultisols can also result in nutrient losses. Buol et al.^[4] speculate that deforestation has accelerated the conversion of forest lands to savannas on Ultisols in the tropics because of the excessive removal of nutrients during timber harvesting.

Extensive acreage of cotton, corn, soybean, and peanuts in the Coastal Plain region of the United States illustrates the productivity potential of Ultisols. Cultivation of Ultisols is facilitated in this region by relatively long growing seasons, sufficient available moisture, good access to soil amendments, and favorable soil physical characteristics. Agronomic management systems that utilize reduced tillage, cover crops, and residue management that ultimately increase SOM can help sustain the productivity of cultivated Ultisols.^[5,14]

Use of Ultisols

Well-drained Ultisols that are not on steep slopes or shallow to bedrock provide few limitations for many urban applications. However, Ultisols can range from excessively drained to very poorly drained, depending on landscape position, climate, and soil and parent material permeability. Excessively drained sandy Ultisols that allow rapid leaching of land-applied chemicals, soils on steep slopes, poorly drained soils that possess seasonal high water tables for significant portions of the year, and soils with high quantities of shrink-swell clay provide severe limitations for many urban and agronomic applications. Soil Survey reports produced by the Natural Resource Conservation Service should be consulted for evaluating soil for a particular use.

SUBORDERS

At the suborder level (the next level of soil classification below order), Ultisols are separated by soil climate. Aquults occupy 0.8% of the land surface area, can be

found in most climates (except deserts), and possess seasonal high water tables for extended periods of the year that render suitability for most severe applications.^[2] These soils typically reside in depressions or other low lying positions. Their identification requires the interpretation of shallow-occurring soil wetness features (termed redoximorphic features). Humults possess high quantities of SOM in the upper portion of the soil, occupy 0.2% of the land surface area, and are mostly restricted to cooler, wetter environments.^[2] Humults occur either in mountainous terrains where aerobic decomposition of SOM is thought to be inhibited by temperature and moisture or in the tropics, where extensive root growth may lead to excessive SOM accumulations.^[3] Ustults are found in semiarid regions that typically possess sufficient precipitation during the growing season for dryland cultivation. Xerults are found in Mediterranean climates characterized by cool, moist winters, and dry, warm summers. It is thought that the excessive winter-time precipitation during lower evapotranspiration periods promotes leaching and subsequent Ultisol development in these climates. Udults are the most extensive Ultisol suborder (8.1%). These soils are found in humid environments, are fairly low in SOM, but possess sufficient subsoil moisture for dryland crop cultivation during most years. However, certain subsoil properties within udults (e.g., plinthite, traffic or tillage pans, and fragipans) may restrict root development as well as limit suitability for some urban uses.

CONCLUSION

The presence of Ultisols suggests appreciable pedogenic development, although preweathered parent materials often confound this relationship. Although not extensive, these soils can be found in many parts of the world. Unfortunately, some of these soils have been abused since the 19th century and have limited genetic ability to rebound. In some parts of the world, shifting agriculture is practiced on Ultisols because amendment application is not possible.^[2] This shifting cultivation is mostly caused by limitations in soil chemical properties and not by poor physical properties or climate. If managed properly, Ultisols are capable of providing and sustaining substantial food and other resources.

REFERENCES

1. West, L.T.; Beinroth, F.H. Ultisols. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 1999; E358–E372.
2. Soil Survey Staff. *Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys*; Handbook No. 436; USDA–NRCS: Washington, 1999.

3. Miller, B.J. Ultisols. In *Pedogenesis and Soil Taxonomy II. The Soil Orders*; Wilding, L.P., Smeck, N.E., Hall, G.F., Eds.; Development in Soil Science 11B; Elsevier: Amsterdam, 1983; 283–323.
4. Buol, S.W.; Hole, F.D.; McCracken, R.J.; Southard, R.J. Ultisols: Low base status soils. In *Soil Genesis and Classification*, 4th Ed.; Iowa State University Press: Iowa, 1997; 352–362.
5. West, L.T.; Beinroth, F.H.; Sumner, M.E.; Kang, B.T. Ultisols: Characteristics and impacts on society. *Adv. Agron.* **1997**, *63*, 179–236.
6. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941.
7. Gamble, E.E.; Daniels, R.B. Parent material of upper and middle coastal plain soils in North Carolina. *Soil Sci. Soc. Am. Proc.* **1974**, *38*, 633–637.
8. Smith, G.D. Ultisols. In *The Guy Smith Interviews: Rationale for Concepts in Soil Taxonomy*; Forbes, T.R., Ed.; Tech. Monogr. No. 11; Soil Management Support Services, USDA: Washington, 1986; 260 pp.
9. Adams, F. Crop response to lime in the southern United States. In *Soil Acidity and Liming*, 2nd Ed.; Adams, F., Ed.; Agronomy Number 12; American Society of Agronomy: Madison, 1984; 211–266.
10. Adams, F. *Response of Cotton to Lime in Field Experiments*; Bulletin #376; Alabama Agriculture Experiment Station: Auburn, 1968.
11. Adams, F. *Response of Corn to Lime in Field Experiments*; Bulletin #39; Alabama Agriculture Experiment Station: Auburn, 1969.
12. Rogers, H.T.; Adams, F.; Thurlow, D.L. *Lime Needs of Soybeans on Alabama Soils*; Bulletin #452; Alabama Agriculture Experiment Station: Auburn, 1973.
13. Karathanasis, A.D.; Hurt, G.W.; Hajek, B.F. Properties and classification of montmorillonite-rich Hapludults in the Alabama Coastal Plains. *Soil Sci.* **1986**, *142*, 76–82.
14. Reeves, D.W. The role of organic matter in maintaining soil quality in continuous cropping systems. *Soil Till. Res.* **1997**, *43*, 131–167.

Uranium Contamination

Ewald Schnug

Institute for Crop and Soil Science, Julius Kuhn Institute (JKI), Braunschweig, Germany

Roberta Bottino Montolar Sparovek

Luiz de Queiroz College of Agriculture (ESALQ) Piracicaba, University of Sao Paulo, Piracicaba, Brazil

Maria del Carmen Lamas

Institute of Soil Science, University of Buenos Aires, Buenos Aires, Argentina

Sylvia L. Kratz

Jurgen Fleckenstein

Susanne Schroetter

Institute of Plant Nutrients and Soil Science, Federal Agricultural Research Centre (FAL), Braunschweig, Germany

Abstract

Uranium (U) is a natural constituent of the rocks of the earth's crust. During weathering, it is released to soils and waters, where it is normally found in lower concentrations than in the basic rock materials. The oceans are the largest environmental sinks for weathered U. U is a toxic radionuclide and the heaviest of all naturally occurring elements. Its concentration is often increased by anthropogenic activities.

GENERAL ISSUES OF URANIUM (U)

Radionuclides are part of our natural environment. Their biological impact is mostly considered as negative, but some authors also claim a positive effect of low radiation for life processes, called "hormesis,"^[1] which may be the active principle of a couple of spas and mineral waters.^[2] Natural U is a mixture of the three isotopes ^{238}U , ^{235}U , and ^{234}U in the weight proportions 99.27%, 0.72%, and 0.006% and half lifetimes of 4.468×10^9 , 703.8×10^6 , and 244,500 years, respectively. An amount of 1 mg of natural U has an activity of 25.28 Bq [1 Becquerel (Bq) = 1 radioactive decay per second]. U species occur in the following four valences: U^{3+} (III), U^{4+} (IV), UO_2^+ (V), and UO_2^{2+} (VI). The predominant states in the environment are U(IV) and U(VI).^[3] This number of different valences is also one explanation for the potential toxicity of U in comparison to other heavy metals.^[4] With a specific mass of 18.9 g/cm^3 , U is the heaviest natural element. It occurs generally in low concentrations which may be elevated as a result of leaching from natural deposits or because of anthropogenic activities, such as mining, military operations employing ammunition with depleted uranium (DU; the fissionable ^{235}U has been extracted for nuclear fuel processing) penetrators, manufacturing and use of nuclear fuels, the

deposition of industrial and medical wastes, and the use of phosphate (P) fertilizers in agriculture.^[5–10]

HAZARDS OF U

Humans and animals can be contaminated by U through inhalation, ingestion, and skin contact. Inhalation is not as important as ingestion. Skin contact is particularly a threat for those working in the U industry. The ingestion of U occurs by drinking water and through the food chain via crop plants, animal feed, and animal products.

Mammals have a particularly high sensibility to U.^[3] Once the U is in the organism, it is transferred to the extra cellular fluids and transported with the blood to other organs. Uranyl (UO_2^{2+}) is the soluble form transported, and it forms complexes with protein and anions. U tends to accumulate in the body and has two modes of damaging the organism: either by radioactivity or by its biochemical toxicity as a heavy metal. Liver and kidneys are the predominant places of U accumulation. Radiation is of particular risk for lungs and bones. The most remarkable effect of U toxicity going along with low and medium contaminations is cancer. The dangers arising from the biochemical toxicity of U are generally considered to predominate the risks from its

radioactivity.^[11] The uptake with the natural element U cannot be avoided. The daily average U intake of humans with solid food is 2–4 µg. The total intake, however, is very much dependent on the U concentrations in the additional drinking water. Even at low concentration levels, drinking water accounts to more than 80% of the total U ingestion.^[7] This underlines the need to limit the uptake of soluble U compounds with waters for personal risk reduction.^[12] With the increased consumption of mineral waters, being advertised as the “better drinking water,” the relative importance of U as a threat to human health is increasing.^[13]

U IN SOILS AND PLANTS

In soils, the average U concentration ranges from 0.1 mg/kg to 11 mg/kg,^[14] with an average of 1.8 mg/kg. Mineral soils with a high content of clay can have U concentrations much above this average because clay minerals are prominent adsorbers of U.^[15] In terms of masses, water is the most significant vector for the transport of U (in solutions or by erosion) in the environment. The transport of U to natural waters occurs by diffusion or mass flow either dissolved or suspended.^[16] The solubility of U in soils depends on many factors such as, for instance, pH, redox potential, temperature, texture, organic and inorganic compounds, moisture, and microbial activity.^[15] U introduced into soils by anthropogenic activities is easily mobilized and transferred into the food chain.^[17] Soils with low fertility show higher transfer rates than do fertile soils (Fig. 1).

U IN FERTILIZERS

P fertilization is the main source for U contamination of agricultural soils. The Federal Agricultural Research Centre database for heavy metals in fertilizers records means for U concentrations (mg/kg U): 125 for single superphosphates, 219 for triple superphosphates, 43 for soft rock P, and 75 for compound fertilizers with P but less than 3 for manures. Thus, an average P dressing of 22 kg/ha P charges the soil with >20 g/ha U when triple superphosphate is used and <3 g/ha U when manures are used. In comparison, the maximum U uptake expected by crops is <1 g/ha U.

U IN WATERS

In the aquatic environment, U occurs in concentrations from 0.1 µg/L to 10 µg/L,^[18] mainly as uranyl carbonate complexes. In confined aquifers, U is found in the tetravalent stage [U^{4+} (IV)], but in surface waters it is found in the hexavalent state [UO_2^{2+} (VI)].^[4] Almost all tetravalent (uranous) compounds have a very low solubility and are less mobile than the hexavalent (uranyl) ones. The formation of hydrolysis products increases the solubility of U oxides by

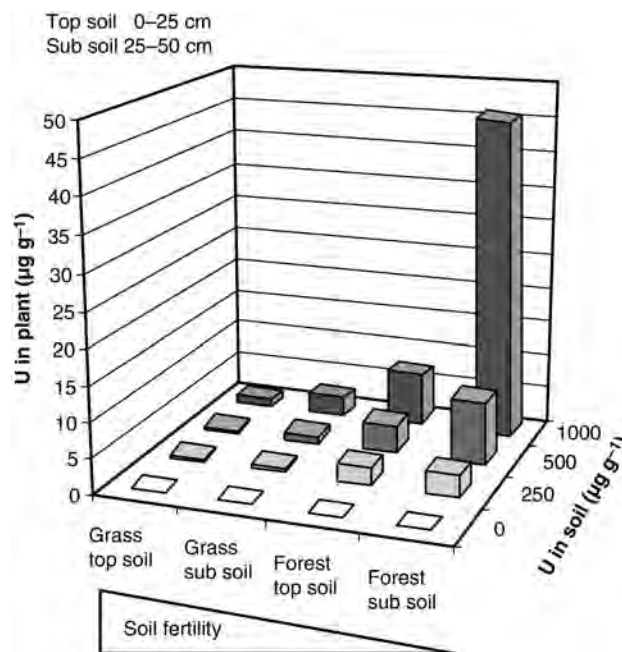


Fig. 1 U concentrations in *Lolium perenne* (4th cut, 40 weeks after contamination) grown on top and subsoil material of a podzolic brown soil (sandy loam, Alfisol) under grass and forest cultivation with different fertility and U contamination. [1.5 kg soil per pot, contamination with finely pulverized green modification of U_3O_8 . Soil substrates: grassland topsoil, grassland subsoil, forest topsoil, forest subsoil; pH ($CaCl_2$): 5.9, 4.8, 3.5, and 3.8; total carbon (%): 1.2, 0.5, 2.0, and 0.6; total nitrogen (mg/g N): 1.0, 0.4, 1.1, and 0.4; plant available U (AAAC according to the study by Sillanpää,^[19] mg/kg U): 0.02, 0.03, 0.04, and 0.03; plant available P (calcium ammonium lactate, mg/kg P): 108, 20, 48, and 20; plant available potassium (K; calcium ammonium lactate, mg/kg K): 261, 246, and 25, 5.] U determination by ICP-QMS following microwave digestion with nitric acid. U concentrations are means of four replicates.

Source: From Lamas, Fleckenstein, et al.^[20]

a complex between a metal and one or more ligands. The amount of hydrolysis products in solutions is directly related to the activity of UO_2^{2+} and the pH.^[15] Hydrolysis also competes with the formation of organic and inorganic complexes.^[7] At pH values below 4, the uranyl ion is complexed by fluorite; between pH values 4 and 7.5, it is complexed by P; and at higher pH values, complexation with carbonate to a carbonate–uranyl complex predominates.^[21] In the natural environment with typical pH values around 6 or below and in the presence of oxygen, humic acid, and fulvic acid, very stable complexes are formed.

U at higher concentrations in groundwater comes along with low pH, low adsorption capacity, lack of anions, and the presence of U ores. In comparison to this, surface waters usually show low concentrations of U because of precipitation and adsorption to bottom sediments especially at the presence of organic matter and under reducing conditions.^[15,22,23]

PROTECTION OF SOILS FROM U CONTAMINATION

Soils can be charged with surplus U by mining activities, by fertilizer use, and by the use of DU ammunition in military operations. In the first two cases, U is dispersed in the environment, whereas DU penetrators represent a typical hot spot situation; however, the extreme long half lifetime of the main isotope ^{238}U makes any contamination a permanent threat. Because of the only small offtake of U by crop plants ($<2\text{ g/ha}$), phyto-remediation (removal of U with harvests) is only a hypothetical option. Reasonable effects are only achieved by measures to further reduce the U transfer into the food chain. Most promising here is increasing the P concentrations in the soil because U has a strong affinity to form low soluble complexes with P in soils. However, the amount of P needed for significant effects is far higher than that usually applied for satisfying the P demand of crops.^[24] Together with the fact that U transfer generally decreases with soil fertility (Fig. 1), U contamination through military activities seems especially tragic for populations in crisis areas where soil fertility is usually poor. Taking all these facts into account, a clear favor should be given to strategies avoiding or reducing U contaminations of soils, such as the shutdown of U effluxes from mining operations, the removal of U from mineral fertilizers or an extended and more efficient use of manures, and finally the international ban of DU military ammunition.

REFERENCES

1. Luckey, T.D. *Radiation Hormesis*; CRC Press Inc.: Boca Raton, 1991.
2. Hevesy, G.; Paneth, F.A. *A Manual of Radioactivity*; 2nd Ed.; Oxford University Press: Oxford, 1938.
3. Fellows, R.J.; Ainsworth, C.C.; Driver, C.J.; Cataldo, D.A. Dynamics and transformations of radionuclides in soils and ecosystem health. In *Soil Chemistry and Ecosystem Health*; Spec. Pub; SSSA: Madison, 1998; Vol. 52, 85–112.
4. Ribeira, D.; Labrot, F.; Tisnerat, G.; Narbonne, J.F. Uranium in the environment: Occurrence, transfer, and biological effects. *Rev. Environ. Contaminat. Toxicol.* 1996; 146, 53–89.
5. Azouazi, M.; Ouahidi, Y.; Fakhi, S.; Andres, Y.; Abbe, J.C.; Benmansour, M. Natural radioactivity in phosphates, phosphogypsum and natural waters in Morocco. *J. Environ. Radioactiv.* 2001, 54, 231–242.
6. Barisic, D.; Lulic, S.; Miletic, P. Radium and uranium in phosphate fertilizers and their impact on the radioactivity of waters. *Water Res.* 1992, 26, 607–611.
7. Cothorn, C.R.; Lappenbusch, W.L. Occurrence of uranium in drinking water in the U.S. *Health Phys.* 1983, 45, 89–99.
8. Schnug, E.; Haneklaus, S.; Schnier, C.; Scholten, L.C. Issues of natural radioactivity in phosphates. *Commun. Soil Sci. Plant Anal.* 1996, 27 (3 and 4), 829–841.
9. Spalding, R.F.; Sackett, W.M. Uranium runoff from the Gulf of Mexico. Distribution province anomalous concentration. *Science* 1972, 175, 629–631.
10. UNEP. *Depleted Uranium in Kosovo. Post-Conflict Environmental Assessment*; United Nations Environmental Programme: Nairobi, 2001.
11. Milvy, P.; Cothorn, C.R. Scientific background for the development of regulations for radionuclides in drinking water. In *Radon, Radium and Uranium in Drinking Water*; Cothorn, C.R., Rebers, P., Eds.; Lewis Publishers: Chelsea, 1990; 1–16.
12. Harduin, J.C.; Royer, P.; Piechowski, J. Uptake and urinary-excretion of uranium after oral-administration in man. *Radiat. Prot. Dosim.* 1994, 53, 245–248.
13. Sparovek, R.B.M.; Fleckenstein, J.; Schnug, E. Issues of uranium and radioactivity in natural mineral waters. *Landbauforschung Völkenrode* 2001, 51 (4), 149–157.
14. Kabata-Pendias, A. *Trace Elements in Soils and Plants*, 3rd Ed.; CRC Press: New York, 2000.
15. Harmsen, K.; de Haan, F.A.M. Occurrence and behaviour of uranium and thorium in soil and water. *Neth. J. Agr. Sci.* 1980, 28, 40–62.
16. Drever, J.I. *The Geochemistry of Natural Waters*, 2nd Ed.; Prentice-Hall: Englewood Cliffs, 1988.
17. Meyer, M.C.; Fleckenstein, J.; McLendon, T.; Price, D.; Schnug, E. Uptake of munitions-derived depleted uranium by three grass species. *J. Plant Nutr.* 2004, 27, 1415–1429.
18. Brits, R.J.N.; Smit, M.C.B. Determination of uranium in natural waters by preconcentration on anion-exchange resin and delayed-neutron counting. *Anal. Chem.* 1977, 49, 67–69.
19. Sillanpää, M. Micronutrients and the Nutrient Status of Soils: A Global Study. *FAO: Rome*, 1982, 48 pp.
20. Lamas, M.; Fleckenstein, J.; Schroetter, S.; Sparovek, B.; Schnug, E. Determination of uranium by means of ICP-QMS. *Commun. Soil Sci. Plant Anal.* 2002, 33 (15–18), 3469–3479.
21. Erikson, R.L.; Hostetler, C.J.; Divine, J.R.; Price, K.R. *A Review of the Environmental Behavior of Uranium Derived from Depleted Uranium Alloy Penetrators*; Pacific Northwest Laboratory: Richland, 1990.
22. Casas, L.; de Pablo, J.; Gimenez, J.; Torrero, M.E.; Bruno, J.; Cera, E.; Finch, R.J.; Ewing, R.C. The role of pH and carbonate on the solubility of UO_2 and uranite under normally reducing conditions. *Geochim. Cosmochim. Acta* 1998, 62, 2223–2231.
23. Higgs, J.J.W.; Falck, W.E.; Eobkerj, P.J. Sediment-groundwater systems froze the natural analogue sites of Needle's Eye and Broubster. *Fluid Processes Series*, Technical WE/89/40; British Geological Survey, 1989.
24. Lamas, M. Factors affecting the availability of uranium in soils. *Landbauforschung Voelkenrode (FAL—Agric. Res.)* 2005, Special Issue Nr. 280.

Urban Lands: Abandoned

Joshua W. Beniston

Jornada Experimental Range, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Las Cruces, New Mexico, U.S.A.

William D. Shuster

National Risk Management Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency (EPA), Cincinnati, Ohio, U.S.A.

Abstract

The loss of industrial manufacturing, migration to suburban areas, and economic and natural disasters have all led to large population reductions in many cities in the United States and Europe. After losing large proportions of their peak populations, a consequent land use change is often large-scale demolition of degraded residential and commercial infrastructure. The resulting large areas of vacant land represent significant challenges and opportunities for cities to reutilize this unique type of landscape. The soils that comprise vacant urban land can present a mixture of native soils affected by some early disturbance history (due to initial development) and imported soil fill material (as backfill for demolition excavation). Therefore, the specific condition of urban soils in vacant land, their quality, and associated suitability for redevelopment as green spaces can vary widely. The assessment of vacant lot soils is key to determining how these sites might be used to provide ecosystem services, including food production, stormwater runoff and flood control, biodiversity, recreation, and education. Such assessments also point to the specific management gaps that can help to guide productive utilization of these landscapes in cities where vacant land is extensive.

INTRODUCTION—THE CONTEXT OF VACANT URBAN LAND

The soils found in vacant urban lots are the unique pedogenic products of larger patterns in socioeconomic change. During the final quarter of the 20th century, economic globalization made it advantageous for industrial manufacturing operations to relocate to areas outside of traditional manufacturing centers in North America and Europe. The direct loss of employment opportunities in these cities led to massive population declines. Within these postindustrial or legacy cities, population loss was further facilitated by migration from urban core areas to suburban developments. These legacy cities are most dense in the northeastern and Central United States, where a number of cities have lost half or more of their peak populations (Table 1). Following a decline in population, the residential housing stock that comprised neighborhoods becomes vacant and falls into disrepair (Fig. 1). Demolition of these structures is touted as one way to discourage the spread of blight, leaving behind large areas of vacant land. Data on the area of vacant urban land in the United States are sparse, but surveys indicate that the total area of vacant land in some U.S. cities is significant (often >1000 ha per city; Table 1). Vacant urban land is not

unique to these legacy cities or demographic trends. The real estate foreclosure crisis and natural disasters, such as Hurricane Katrina, have also driven large increases in vacant infrastructure and land in other regions of the United States, namely California's central valley and New Orleans, Louisiana. Cities in a number of European regions (Germany, Northern England, and Eastern Europe) have also lost large proportions of their population due to similar trends of deindustrialization and migration to suburban areas.^[1] Globally, declining urban populations have been documented due to epidemics (sub-Saharan Africa), conflict (Middle East), and environmental disasters (Soviet Union).^[1] Our review of the literature, however, suggests that very little data have been published on the amount and extent of vacant urban land outside of the United States.

Vacant urban lands are dynamic socioecological systems. Vacant land is spatially distributed in a mixture of hot spots in blighted neighborhoods. Blighted neighborhoods are then dispersed across a cityscape according to more specific socioeconomic histories and trajectories. Urban areas with high vacancy rates are commonly associated with increases in crime and decreased the value of adjacent properties. On the other hand, vacant land is a natural resource that exists in some abundance and is often

Table 1 Change in peak population and vacant land in cities with the highest population losses in the United States.

City, State	% Change from peak to population in 2010	Number of vacant lots	Vacant land area (ha)
St. Louis, Missouri	-62.7	20,000	N/A
Detroit, Michigan	-61.4	90,000	6,500
Youngstown, Ohio	-60.6	23,000	2,800
Cleveland, Ohio	-56.6	18,000	1,500
Buffalo, New York	-55.0	16,000	1,300
Pittsburgh, Pennsylvania	-54.8	14,000	N/A
Flint, Michigan	-48.0	12,000	N/A

Source: From Dewar & Thomas^[2] ©2013 University of Pennsylvania Press; Youngstown Neighborhood Development Corporation; Cuyahoga County, Ohio Land Bank; Groundwork Buffalo; Genesee County, Michigan Land Bank.

present in contiguous landscapes that are well suited to the classical definition of green infrastructure. Nassauer and Raskin^[3] provided a synthesis of conditions common to urban areas with high levels of vacancy: 1) occupied structures, abandoned structures, and vacant land occur together in irregular spatial patterns; 2) ecological legacies, or the lasting effects on a landscape of past events, are widespread, including soil contamination and degradation; and 3) due to risk adversity and lack of an economic underpinning, financial investment for redevelopment is unlikely.

There is widespread interest in transforming vacant land into functional green spaces capable of providing

an enhanced suite of ecosystem services (ES). ES are the recognized or measurable, beneficial outcomes that result from the multitude of abiotic and biotic interactions that constitute ecosystems. The concept of ES lies at the nexus of *provisioning* services, e.g., water volume inputs to the urban hydrologic cycle; *regulating* services, largely provided by soils, which can regulate the cycling of water as per the balance between infiltration and runoff; *supporting* services that include efficiency of water use by plant communities and nutrient cycling; and *cultural* services, where soils and their interaction with water resources can provide a multitude of recreational and aesthetic opportunities. Urban agriculture, stormwater infiltration landscapes, and neighborhood parks have all been identified as potential end points for vacant land conversion to green space, with each capable of enhancing ES in the core areas of legacy cities. However, the development of vacant land as green spaces must address the assessment and management of soil to realize the ecological potential of these sites.^[4] Land use history results in both high spatial variability and lasting impacts (ecological legacies) on soil conditions in urban areas. Thus, understanding the soil conditions at the site level can accelerate the success of site selection and management for green space projects in vacant urban land. This entry focuses on vacant lot soils as the product of the circumstances leading up to landscapes shifting to vacant urban land; their possible limitations and opportunities for management; and ultimately, their potential to yield ES. We will concentrate on insights provided by the assessment work on vacant urban land from the north-central and eastern United States.

DIFFERENTIATION OF URBAN AND RURAL SOILS

The properties of urban soils are affected by the type and extent of development. Although urban soils can include soils constituted for parks and sports fields (e.g., golf courses), we focus on the disturbed soils that are affected by the product of development (and demolition) in cities. These urban soils are generally considered anthropogenic soils or soils modified by human activity. The anthropogenic influence often results in the incorporation of artifact materials (building materials, wastes, etc.) into the soil profile.^[5] Historical land use and the associated disturbances are an overarching factor that influences the morphology and distribution of anthropogenic urban soil types. Long-term effects of anthropogenic disturbance commonly observed in urban soils include removal of topsoil layers, surface compaction, discontinuous soil profiles, buried soils, and physical and chemical properties altered by the presence of large quantities of debris materials. Land cover and parent material have also been found



Fig. 1 Vacant houses (A) and a vacant house being demolished (B) in Youngstown, Ohio.

Source: Photos (A) and (B) from Youngstown Neighborhood Development Corporation.

to explain variation in soil properties in urban areas.^[6] Overall, urban soils are patterned by development and demolition, whereas their rural soil counterparts are affected by agroecosystem management or degree of cover by native vegetation. Rural or predevelopment soil conditions are often used as a qualitative reference condition relating the degree to which urbanization has affected soil taxonomy, physical and chemical characteristics, and hydrology.

The construction and demolition of buildings and infrastructure are key historical disturbances that influence the properties and management of soils in vacant urban land. Residential demolitions impact soil conditions across entire residential lots, and due to the reach of heavy equipment employed in demolitions, beyond the area formerly covered by the building.^[7] The negative effects of demolition on soils may persist for many years in unmanaged vacant lots. A survey of vacant land in Cleveland, Ohio, suggested that the management potential of some sites was limited by the presence of large amounts of debris and high levels of soil compaction that were often the result of demolitions that occurred >20 years prior to the study.^[8] Alternately, vacant lot soils that have been under long-term perennial vegetation have demonstrated favorable chemical and biological properties.^[9]

MANAGING VACANT URBAN LAND FOR IMPROVED ECOSYSTEM SERVICES

The selection and addition of soil or fill materials and the removal of debris following construction and demolition, including building foundations and basements, are key management activities that affect the potential for ES on vacant land. Covering vacant land with asphalt or other impervious surfaces following demolition is among the worst outcomes for ES, as it prevents water infiltration and plant growth in the covered area. The persistence of unremoved foundations and basements in vacant lots also results in impervious areas in both soil surface and subsurface layers in the former housing area. In Cleveland, Ohio, many sites have been filled with material that is either coarse sands or fine, silty clay loam lacustrine material, neither of which retains water or nutrients well, so any attempt at intentional plantings suffers. Using a structured, sandy loam soil as postdemolition fill material in vacant urban lots, however, provides the potential for the site to support water infiltration, water retention, and plant growth. The grading of the fill materials is another key stage in the restoration process. Beniston et al.^[10] observed extremely high compaction and low microbial activity in the soil surface (0–10 cm) layer immediately after vacant houses were demolished. Heavy compaction, as well as material that is not well packed, is a somewhat common condition following demolition in vacant land. Ideally, the site should be compacted to achieve a bulk

density in the range suitable for plant growth (approximately 0.9–1.4 Mg m⁻³, depending on soil texture).

Vacant lots offer pervious surface area for infiltration and redistribution of soil moisture. The processes that mediate between infiltration and runoff regulate the potential for a vacant lot to be used as a passive, infiltrative sink for excess urban runoff volume. The role of vacant lots in the urban hydrologic cycle may offer value to local governance and utilities that can ill-afford the additional runoff volume generated by vacant areas and underutilized streets as inputs to overburdened centralized wastewater infrastructure. The potential for reuse of vacant lots as stormwater volume management is tied to the imprinting of anthropogenic disturbance as the primary soil-forming factor of vacant lot soils. When these areas were developed, the regular grid of housing or other development utilized gutters and downspouts to move runoff from impervious area (roofs) to the street and the centralized sewer collection system. The small amount of pervious area (yards) were usually sloped toward the street to encourage drainage to the street. These landforms often persist past the demolition, so that even if the soils were permeable, any runoff that did form would be routed to the street and to the centralized collection system. Shuster et al.^[7] found that soil physical and hydrologic characteristics were influenced by different levels of disturbance, namely lower (parcel area with remnant, undisturbed soils), higher (fill areas on vacant lots within the footprint of the structure), and demolition technique (burying debris on-site vs. removing debris); and in some cases, both factors influenced vacant lot hydrology. Factors that can affect the formation of runoff from a vacant lot include the proportion of vegetative cover, the absence of which can allow raindrop impact and splash erosion (sealing of the soil surface) and lower the resistance to overland flow. The sealing of the soil surface can decrease or prevent infiltration, increasing the proportion of rainfall converted to runoff volume. Apart from establishing vegetative cover on infiltrative soils, the grade of a vacant lot should be level or in the shape of shallow bowl to encourage infiltration.

Vacant land also supports urban biodiversity and directly influences the biodiversity-driven ES of pollination and pest control by providing habitat for both pollinator species and species that act as predators of herbivorous pest insects.^[11,12] A key component to realizing these ES is vegetation management, as pollinator and beneficial predator insect species benefit from increased plant diversity, increased structural diversity of vegetation, and diverse phenology of flowering species that spans the growing season.^[12] Studies of the public perception of vacant land, however, consistently report that neighboring residents prefer green spaces where vegetation is regularly maintained, as opposed to unmanaged ecological succession, on vacant urban lots.^[11] Research has also demonstrated that biodiversity

of insect and arthropod species can be maintained in highly managed urban green spaces. In a study comparing predatory insect diversity between unmanaged vacant urban lots and community gardens, Gardiner et al. reported that conversion to gardens reduced the abundance of only a few predator groups, while unmanaged lots and gardens contained similar levels of flowering plant diversity and insect diversity.^[13] Similar patterns in belowground biodiversity have been reported. A study of nematode food webs and soil properties found that the initial conversion of unmanaged vacant lots to urban gardens impacted the structure of nematode food webs, while older gardens contained food webs similar to those in unmanaged lots.^[9] That study also reported that overall soil quality was higher in gardens than in the unmanaged lots.^[9] In general, soil management practices that increase organic matter and improve soil structure ultimately support increased biodiversity by improving habitat for both aboveground plant species and belowground microbes and invertebrates.

Contamination is a key issue of concern in soils of vacant urban land. Lead (Pb) contamination is commonly observed in urban soils, with urban areas accumulating and concentrating Pb and other heavy metals (e.g., zinc and copper) during the 20th century. Sources included exhaust particulates from automobiles burning leaded gasoline, brake linings, air deposition from co-located industry, and leaded paints commonly used prior to 1978 for all manner of construction and maintenance. The U.S. Environmental Protection Agency has set a screening level of 400 mg kg⁻¹ Pb for soils in residential areas, and soils exceeding that level require further testing and additional management considerations. A survey of more than 60 vacant lots in Cleveland, Ohio, reported a wide range in the concentration of soil Pb, and many sites exceeded 400 mg kg⁻¹,^[14] though the bioavailability of Pb compounds remains an open question in need of a comprehensive assessment.

Urban soils may also be contaminated by persistent organic compounds that pose significant human health risks, including by incomplete combustion processes and are associated with traffic, any type of burning or polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs).^[15] PAHs are produced by incineration, domestic heating, and heavy industry. PCBs are industrial compounds that are no longer produced commercially, but have been deposited in the environment through widespread historical use of a variety of industrial and automotive products. Soil is a major reservoir of persistent organic compounds, and concentrations of both of these compound groups tend to be the highest in soils near sources, including highway corridors and industrial manufacturing areas.

Direct ingestion of soil and inhalation of soil born dust particles are key exposure pathways for both heavy metals and persistent organic compounds in urban soils, so establishing vegetative or mulch cover on vacant land is a key

management practice for limiting exposure. Due to the large range of variability in site history, and therefore soil physical and chemical characteristics, soil testing for fertility and selected contaminants (depending on the intended use of soils) is recommended prior to planning green space projects on vacant urban land. Soil tests for Pb are increasingly available in more locations at a reasonable cost, while testing for persistent organic compounds remains highly specialized and expensive.

URBAN AGRICULTURE ON VACANT LAND

Urban agriculture has emerged as a popular method of utilizing vacant urban land.^[16] Creating and maintaining gardens and urban farms on vacant land have the potential to provide many benefits to the surrounding communities, including fresh fruits and vegetables, improved soil quality and ES, unique educational opportunities, and the potential for supplemental income. Urban gardens can produce vegetable crop yields similar to rural sites,^[10,16] and high levels of soil quality have been observed at intensively managed urban farms. Participation in urban agriculture has been linked directly with improved diets. In cities with large areas of vacant land, it may be possible to implement urban agriculture at a scale that is capable of impacting the city's food system and food security (Fig. 2). The studies in Cleveland, Ohio, and Detroit, Michigan, estimated that a significant percentage of the fruits and vegetables consumed by the populations within these cities could be produced by using a sizable proportion of their existing vacant land for urban agriculture (Table 2).^[17,18]

Site assessment and selection are key steps in establishing urban gardens. Sites that have undergone heavy disturbances, such as demolitions, are likely to have degraded soils and require more intensive management to be productive. Soil contamination, as described earlier, is a concern, and all sites should be tested for soil heavy metal concentrations. Soil nutrient content and soil organic carbon (SOC) content are strong predictors of crop growth in urban soils, and soil testing can inform management to improve agronomic conditions.^[10,19] The selection of crops, application of amendments, adjustment of soil pH, and other aspects of site management should be based on heavy metal and nutrient test results.

In situations where vacant lot soils have been physically degraded by demolition or by other disturbances, intensive management is required to improve soil quality for crop production and improved ES.^[16] A unique soil management opportunity exists in cities, where the materials necessary to create amendments occur in great abundance and the land areas requiring intervention are relatively small. Thus, large applications of amendments such as compost and mulch can be made to soils in vacant land during the installation phase of urban agriculture or green spaces



Fig. 2 A 0.80-ha urban farm established on formerly vacant land in Cleveland, Ohio.

(Fig. 3). These applications are both capable of producing immediate impacts on soil properties and unfeasible for the larger land areas of rural farms where available quantities of materials and their cost are prohibitive. Beniston et al.^[10] reported that significant improvements to soil quality and

crop growth resulted immediately from a large (equivalent to 150 Mg ha⁻¹) application of compost produced from yard wastes to a heavily compacted urban soil. Compost application to urban soils also results in long-term increases in SOC at amended sites.^[20] SOC is a key indicator of soil quality, and increases in SOC can result in measurable enhancement of many ES. The production of compost from urban yard wastes and its use as a soil amendment for green space projects in vacant land provide multiple benefits by removing material from the waste stream and improving soils.

Cover cropping is another management strategy capable of improving the ecological function of soils in vacant land. In Youngstown, Ohio, intensive cover cropping that utilized both warm season and cool season grasses produced large amounts of biomass and resulted in measurable improvements to soil structure in a compacted urban soil.^[10] Cover crops, particularly those grown during the cooler months in northern climates, are also very effective at retaining and cycling nutrients. One caveat regarding the large compost applications described earlier is that repeated, large applications of

Table 2 Potential for crop production from urban agriculture to meet consumption demands through cultivation of vacant land in two cities in the United States.

City	Land area required	Food Production Potential	References
Cleveland, Ohio	1240 ha	22–48% fresh produce 25% Poultry 100% Honey	[18]
Detroit, Michigan	406 ha (current consumption) 1364 ha (recommended consumption)	31% Vegetables 17% Fruit	[17]

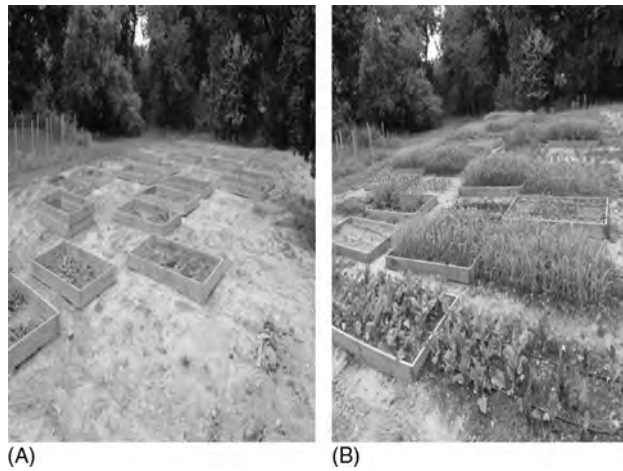


Fig. 3 An urban agriculture experiment on contiguous vacant lots in Youngstown, Ohio. The garden was established immediately following the demolition of houses in spring 2011, and the soil was heavily compacted following the demolition (A). A large quantity of compost produced from yard waste was applied to the site, and crops are growing well 6 weeks later (B).

Source: From Beniston, Lal, et al.^[10] ©2014.

compost can lead to soil nutrient levels that are in excess of crop demand. Excessive soil phosphorus (P) levels from compost application can increase the risk of water pollution from urban agriculture. This is a legitimate concern in vacant land of the United States, as urban stormwater drainage systems provide rapid transfer of runoff to surface waters. Further, many cities with large vacant land areas are in close proximity to the Great Lakes and other freshwater resources.^[3] Again, soil testing can be valuable to urban site managers to prevent excessive nutrient application via compost, fertilizer, or other amendments. Sites with excessive P levels can utilize cover cropping to retain nutrients on-site and improve soil quality, while no longer importing nutrients.

CONCLUSION

Through assessments of the soils in vacant land in postindustrial cities, we report that these landforms can provide a base of natural resources in urban areas. The soils that underlay vacant lots can present as a broad range of mixtures among soil materials and demolition debris. The variation in soil properties that results from these materials and their land use histories regulates their suitability for green infrastructure. These same assessments also illuminate the management gap and suggest ways in which demolitions might be better executed to yield vacant lots that could be used for agriculture, stormwater management, or other green spaces (Table 3). Soil testing (for chemical attributes) can be used in conjunction with soil physical and

Table 3 Best management practices for soils of vacant urban land.

During the demolition process

- Removal of building materials/debris, including basement structures
- Use of debris free, structured soil as fill material
- Grading of site to level or gently concave slope
- Minimize compaction during grading
- Apply organic matter, such as compost, to the soil surface following grading
- Establish vegetation

Site assessment for greenspaces and urban agriculture

- Conduct initial soil testing for nutrients, heavy metals, and organic matter
- Develop and implement management plan to improve soil constraints
- Continue regular soil testing

Source: Adapted from Shuster, Dadio, et al.^[7] ©2014 and Beniston & Lal^[16] ©2012 Springer.

hydrologic data to guide specific management strategies. If soils are properly evaluated and managed, then vacant land may be effectively utilized as landscapes that provide valuable ES in urban areas.

REFERENCES

1. Reckien, D.; Martinez-Fernandez, C. Why do cities shrink? *Eur. Plan. Stud.* **2011**, *19* (8), 1375–1397.
2. Dewar, M.; Thomas, J.M. Introduction: The city after abandonment. In *The City After Abandonment*; Dewar, M., Thomas, J.M., Eds.; University of Pennsylvania Press: Philadelphia, 2013; 1–17.
3. Nassauer, J.I.; Raskin, J. Urban vacancy and land use legacies: A frontier for urban ecological research, design and planning. *Landscape Urban Plan.* **2014**, *125*, 245–253.
4. Dominati, E.; Patterson, M.; Mackay, A. A framework for classifying and quantifying the natural capital and ecosystem services of soils. *Ecol. Econ.* **2010**, *69* (9), 1858–1868.
5. Lehmann, A.; Stahr, K. Nature and significance of anthropogenic urban soils. *J. Soil. Sediment.* **2007**, *7*, 247–260.
6. Pouyat, R.V.; Yesilonis, I.D.; Russell-Anelli, J.; Neerchal, N.K. Soil chemical and physical properties that differentiate urban land-use and cover types. *Soil Sci. Soc. Am. J.* **2007**, *71*, 1010–1019.
7. Shuster, W.D.; Dadio, S.; Drohan, P.; Losco, R.; Shaffer, J. Residential demolition and its impact on vacant lot hydrology: Implications for the management of stormwater and sewer system overflows. *Landscape Urban Plan.* **2014**, *125*, 48–56.
8. Shuster, W.D.; Barkasi, A.; Clark, P.; Dadio, S.; Drohan, P.; Furio, B.; Gerber, T.; Houser, T.; Keltz, A.; Losco, R.; Reinbold, K.; Shaffer, J.; Wander, J. Moving beyond the Udothent: A proposed protocol for assessing urban soils to service data needs for contemporary urban ecosystem management. *Soil Surv. Horizons* **2011**, *52*, 1–30.

9. Grewal, S.S.; Cheng, Z.; Masih, S.; Wolboldt, M.; Huda, N.; Knight, A.; Grewal, P.S. An assessment of soil nematode food webs and nutrient pools in community gardens and vacant lots in two post-industrial American cities. *Urban Ecosyst* **2011**, *14* (2), 181–194.
10. Beniston, J.W.; Lal, R.; Mercer, K.L. Assessing and managing soil quality for urban agriculture in a degraded vacant lot soil. *Land Degrad. Dev.* **2015**. DOI:10.1002/ldr.2342.
11. Gardiner, M.M.; Burkman, C.E.; Prajzner, S.P. The value of urban vacant land to support arthropod biodiversity and ecosystem services. *Environ. Entomol.* **2013**, *42* (6), 1123–1136.
12. Lin, B.B.; Philpott, S.M.; Jha, S. The future of urban agriculture and biodiversity-ecosystem services: Challenges and next steps. *Basic Appl. Ecol.* **2015**, *16* (3), 189–201.
13. Gardiner, M.M.; Prajzner, S.P.; Burkman, C.E.; Albro, S.; Grewal, P.S. Vacant land conversion to community gardens: Influences on generalist arthropod predators and biocontrol services in urban greenspaces. *Urban Ecosyst.* **2014**, *17*, 101–122.
14. Minca, K.K.; Basta, N.T.; Scheckel, K.G. Using the Mehlich-3 soil test as an inexpensive screening tool to estimate total and bioaccessible lead in urban soils. *J. Environ. Qual.* **2013**, *42* (5), 1518–1526.
15. Cachada, A.; Pato, P.; Rocha-Santos, T.; da Silva, E.F.; Duarte, A.C. Levels, sources, and potential human health risks of organic pollutants in urban soils. *Sci. Total Environ.* **2012**, *430*, 184–192.
16. Beniston, J.; Lal, R. Improving soil quality for urban agriculture in the North Central U.S. In *Carbon Sequestration in Urban Ecosystems*; Lal, R., Augustin, B., Eds.; Springer: Dordrecht, 2012; 279–314.
17. Colasanti, K.J.A.; Hamm, M.W. Assessing the local food supply capacity of Detroit, Michigan. *J. Agr. Food Syst. Commun. Dev.* **2010**, *1*, 41–58.
18. Grewal, S.S.; Grewal, P.S. Can cities become self-reliant in food? *Cities* **2012**, *29*, 1–11.
19. Knight, A.; Cheng, Z.; Grewal, S.S.; Islam, K.R.; Kleinhenz, M.D.; Grewal, P.S. Soil health as a predictor of lettuce productivity and quality: A case study of urban vacant lots. *Urban Ecosyst.* **2013**, *16*, 637–656.
20. Brown, S.; Miltner, E.; Cogger, C. Carbon sequestration potential in urban soils. In *Carbon Sequestration in Urban Ecosystems*; Lal, R., Augustin, B., Eds.; Springer: Dordrecht, 2012; 173–196.

Urban Lands: and Construction Sites

Wolfgang Burghardt

Soil Technology, University of Essen, Essen, Germany

Abstract

Land use will inevitably change soil properties. Disturbance of soils and the new substrates added to soils change the morphology of the soil profile and the processes in soil. Urban soils have, therefore, different quality characteristics than natural soils. In the urban ecosystem, a new soil landscape develops, but still, this was not the subject of serious study.

INTRODUCTION

Cities in North and Latin America, Europe, and Asia are the home of more than 80% of the world's population. The demands placed on soils and therefore the targets for soil rehabilitation in urban areas must recognize the high degree of intervention by humans and the benefits of soils to the urban population and its economy.

To varying degrees (20–100%), the urban soils are covered by buildings, pavements and road surfaces, and communications and utility lines. Periodically, new free spaces will be created and rehabilitated, while others will disappear by conversion to another use. Thus, urban soils can change radically over time and acquire new properties.

URBAN SOILS

Types of Substrates

In the city, few soils retain their natural state. Other soils are young and not yet differentiated into horizons by the processes of soil development. Their properties are dominated by the characteristics of substrate of the soil. The majority of urban soils can be grouped into four ways^[1] according to the anthropogenesis of the substrates involved:

Soils modified by treatment or environmental exposure and use

The results will be of compaction; stratification; mixing; reshaping of structure; enrichment by organic substances, such as sand, gasoline, oil, and tar; contamination by toxic amounts of hazardous compounds, such as pathogenic microorganisms; acidification or alkalization; breaking of material; and sorting and separating of substrate components.

Mode of deposition of soil substrates

The methods of deposition include the following: pouring of material (e.g., rail berth); methods of tipping materials (e.g., from shovel, wheel barrows, conveyor belt, or trucks); leveling by heavy machinery; lagoon deposits; spills (e.g., harbor sludge and fly ash); blast furnace sludge; casting waste (e.g., slag), dilapidation, and demolition (e.g., of buildings); the deposition of rubble from war raids; sealing (underground and surface cover); and thorough mixing following repeated excavation and tipping.

Technogenic substrates

The technogenic substrates are from man-made material such as rubble, ashes and cinder, slag, foundry waste, industrial residues and waste, town refuse, sewage sludge, industrial sludge and dust, road paving materials, mine waste, and material from soil remediation (e.g., biological, thermal, and extractive treatment).^[2,3]

Reduction of fine earth material by increase in skeletal soil fractions

Fine earth is often mixed with building material and other technogenic substrates that are coarse-sized. Hence, the mass of fine earth in the soil volume is reduced, as is its influence on the soil characteristics.

Degradation

Most of the changes in urban soils involve a decrease in the capability of soils to fulfill the functions for the welfare of humans. The quality of urban soil can be defined by the degree to which it offers: protection against the occurrence and transfer to humans of hazardous compounds; improvement of the urban green space, climate, groundwater renewal, and concentration of chemicals; decay and safe storage of waste; and satisfactory features

for economic and social purposes. The soil characteristics required to achieve these functions are very diverse and in part contradictory.

In the urban environment, soil degradation arises from a diverse array of processes. Soil degradation can occur by enrichment with compounds from emissions of households, industry, waste tips, traffic, and maintenance work. Amendment of vegetable gardens with ash, sludge, and slag contributes to this too. Small service enterprises such as petrol stations, dry cleaners, car washes, garages, and diverse workshops are also sources of pollutants. Leakage and accidental spills from tanks, pipelines, gas pipes, sewage canals, and road tankers pollute the soils too. Fires can release harmful compounds to soils. Latrines not connected to the urban sewage canal system and the manure of dogs and other animals cause chemical soil degradation by eutrophication.

Soil degradation can be from the lack or imbalance of nutrients in plants of many urban substrates, which often results in the deficits of phosphorus, nitrogen, and micronutrients. The atmospheric deposition of protons results in acidification. Similar effects arise from tipping of acid or acid producing substrates, e.g., of high sulfide content. Alkaline dust and substrates will raise the pH. Salinity can occur from uncontrolled irrigation in dry regions or use of de-icing salts in cold parts of the world.

Physical impacts are from an increase in coarse particles (stone and gravel), which will reduce fine earth volume. Most of the man-made substrates are wastes or construction materials, which contain high amounts of coarse materials. Large areas of the city are degraded by sealing. In addition, unsealed ground is often strongly compacted. The compaction is not only confined to the topsoil as in rural sites. It also penetrates considerable depth. Soil structure can be destroyed by dispersing agents and may inhibit wetting. Decline of soil structure can result in water and wind erosion and landsliding.

The loss of humus-containing topsoil horizons and of subsoil by excavation in cities is one of the largest forms of soil loss. Excavation is the anthropogenic form of erosion.

Degradation can also mean a strong spatial heterogeneity of soil features, or the destruction and splitting of water catchment areas.

REHABILITATION OF URBAN SOILS

Hazardous Compounds

The rehabilitation of polluted urban soils has, besides others, the goal to avoid the transfer of hazardous compounds from soils to humans. The uptake of harmful compounds can result from the direct consumption of polluted soil by dermal uptake through contact of the skin with soil; oral uptake by the ingestion of dirty vegetables and fruits, drinking muddy water or direct

ingestion of soil from dirty hands; pulmonary uptake by the inhalation of fine dust from the fraction finer than 0.010 mm in diameter (PM10); and uptake through open wounds to the skin.

The primary vehicle for the uptake of hazardous compounds will be food and drinking water and the release of volatile gases and vapor from polluted soils.

The most sensitive segment of the population is children. Direct soil uptake by children occurs in playgrounds, school yards, and sports grounds. All bare soil and sand pits should contain less than the limits for compounds dangerous to children. Therefore, polluted soil and soil infected by pathogenic microorganisms^[4] must be replaced. The depth of the new soil layer should be at least 35 cm, greater than the depth to which children will dig.^[5-7] Between the topsoil and the subsoil, a geotextile could be placed to avoid deeper digging.

Earthen road materials are common in many suburban areas. They are sources of fine dust (PM10), which can be inhaled. This will be a large problem in arid and semiarid areas. Rehabilitation requires the stabilization of the road surface by binding agents or covering it by a fine gravel layer.

Polluted vegetable gardens are a source of contaminated food. There are several rehabilitation strategies practiced. The polluted soil can be replaced by clean soil. In case of heavy metal pollution, the mobility of metals can be reduced by liming. The pH values above 5.0 for most metals and 6 for cadmium and nickel will minimize the amount of plant uptake. For vegetable gardens, the uptake of pollutants can also be diminished by reducing the cultivation area of vegetables in a garden or by the growth of vegetables, which accumulate low amounts of the hazardous compounds in the plant parts consumed by humans. A possible, but rarely practiced, strategy will be the addition of compounds to soils apart from lime, which decrease the plant availability of heavy metals. Iron minerals can have such an effect.^[8] Another novel, but inadequately tested, strategy would be phytoremediation by the growth of crops that accumulate hazardous compounds and decrease the mobile fraction of the pollutant in the soil. Pollutant uptake by subsequent crops will be lowered.^[9]

Urban sites, which are degraded by the release of gases from aromatic polyhydrocarbons and mercury vapor, are generally rehabilitated by the removal of the polluted soil and its replacement by an unpolluted substrate.

From atmospheric deposition, soils of most areas of cities will be slightly polluted. Acute toxicity may not be the problem for the soils per se, but it might be of importance for long-term residents who gradually accumulate pollutants to harmful levels. Possibly, the most effective rehabilitation measures for the whole city will be a dense vegetation cover that stabilizes the soil, prevents wind and water erosion of the soil, and minimizes direct contact by humans with the pollutants.

Greenery and Street Trees

The quality of life in the cities will be enhanced by the greenery, which consists mainly of city forest, parks, lawns, and street trees. The rehabilitation of green areas can be achieved by optimizing the pH either by liming or by adding sulfur. The nutrient content can be improved by fertilizing and manuring with compost. Restrictions to the root soil volume or to aeration and available water content can be corrected by soil amelioration before planting.

Street trees live under extreme conditions, and the soil in which they grow will often be degraded.^[10] Street tree patches are often too small for aeration and water uptake. Sizes of at least $2 \times 2 \text{ m}^2$ are recommended for each street tree. When soils of street tree patches are stony or of man-made substrates of low soil quality, they can be replaced by suitable materials. For reducing the compaction and improvement of aeration, the soil around the tree can be loosened by devices that release compressed air from lances pushed into the soil. Aeration will be also improved by inserting perforated drainage pipes near the tree into the rooting zone.

Salinity

Salinity problems occur in two major forms in cities. In cold climates, streets are kept free from ice and snow by spreading salts. De-icing salts often contaminate street tree patches. The increases in sodicity may damage the soil structure of loam, clay, and organic soils. Rehabilitation measures are liming and gypsum addition, loosening or replacing loam, clay, and organic soil by sandy soil.

Soils of cities located in arid areas will often be irrigated. From groundwater and perched water near the soil surface, evapotranspiration will be increased. In arid areas, an accumulation of salts dissolved in the irrigation water will result. The secondary salinity by irrigation has to be reclaimed by loosening dense soil layers, destroying water percolation barriers caused by stone layers under fine textured horizons, and controlling groundwater depth by drainage.

In arid and semiarid areas, gypsum- and carbonate-containing soils are common. Gypsum and carbonates can form compacted layers. Excess irrigation will dissolve gypsum and also carbonates creating voids. Voids are prone to collapse and may damage the stability of buildings and streets. When irrigation water is given in small doses, and only the topsoil is wetted, the consequences can be minimized or avoided.

Sealing

A serious problem of urbanization is the sealing of soils, which limits water infiltration and green area. To a degree,

rehabilitation can be effected by changing the paved surface from impermeable to permeable materials, e.g., from tarmac to paving stone or lawn stone.

Much of the city landscape is sealed by buildings. Soil rehabilitation by unsealing is not possible. But above the sealed area, a soil layer can be established. Examples are soil covers on roofs for rooftop plantings^[11] or dense arrangements of potted plants on sealed surfaces.

Compaction

Compaction reduces water infiltration and water storage capacity of soils that are important land characteristics in cities for determining the volumes of drainage water as well as the groundwater levels and the prevalence of floods in rivers flowing through cities. It determines also the storage capacity for storm water and influences the occurrence of surface water ponding. Topsoil compaction can be ameliorated by loosening the soil before establishing the vegetation cover. On vegetated sites, deep rooting plants and liming to improve the soil structure and to favor the growth of earthworms are effective measures for improving soil porosity. However, deep compaction can only be rehabilitated by deep excavation and refilling the substrate again without compacting it.

Aeration

A significant function of soils in cities is as burial grounds. For the decay of the corpse, the soil must be well aerated. Therefore, soils from silt and clay, of very high organic matter content (>8%) and compacted soils, have generally low suitability as burial grounds.

To achieve good aeration, the soil must also be free of groundwater and stagnant water approximately to a depth of 2.5 m; alternatively, burial grounds may need drainage or a raised surface by infilling.

REHABILITATION OF CONSTRUCTION SITES

The rehabilitation of construction sites begins with demolishing buildings and ends with cleaning and improving the ground for a new use. The rehabilitation of these sites will reverse the sealing of soils.

One principle of rehabilitation of construction sites is to leave as much of the material as possible at the site. This will minimize transport cost and will not damage other sites by transferring the problems of the rehabilitation site.

Construction materials will, however, need to be inspected to separate clean materials from those that are contaminated. The different kinds of the material will also be separated for the purpose of recycling. What is not sold can be crushed to sizes that can be reused at the site either for construction, filling material, or as a substrate for

revegetation. The soil substrate will therefore often be stony, dry, calcareous, and poor in nutrients.

The management of contamination at construction sites requires documentation on individual layers of the profile and mapping of their spatial distribution. Contamination will occur in patches of variable size depending on the characteristics of the layers. The layers can be of materials that already have a certain content of pollutants (e.g., ashes, cinder, rubble, dust, and slag from nonferrous metals), or they may have a high capacity for reducing solubility or for sorption (e.g., well-buffered near-neutral pH, high carbonate, organic matter, and iron content^[12]) or high pore volume^[13] for the storage of compounds. Another significant property is the gravel content that reduces the amount of fine earth. Compounds moving through the reduced amount of fine earth will be concentrated and will increase their content in the fine earth mass.^[12]

The amount of contaminated material, which must be either cleaned or treated for immobilization of the pollutants^[14] or removed to a waste tip, should be minimized. This will be achieved by a substrate survey^[2] and excavating the individual substrate layers very carefully. For example, at a textile plant at Nordhorn, Germany, 69% of the excavated soil was separated as unpolluted or only slightly polluted.^[15] For a 65-ha area of a marshalling yard near Ruhr, Germany, the amount of aromatic polyhydrocarbon contaminated substrate was determined to be less than 10% of the total amount of the excavated substrate of the site.^[16]

Mixing of the material of different layers raises the risk of uncontrolled distribution of hazardous compounds and the creation of future unforeseen hazards. Therefore, mixing contaminated materials must be strictly avoided.

Excess material, especially if treated for immobilization of pollutants, may be heaped to create a green area at the rehabilitated site. Necessary security constructions and control measures to prevent any release of residual pollutants can be established at the heap.

For recreation areas on only slightly polluted soil substrate or of substrates with low potential for a vigorous vegetation cover, a 0.5–1 m covering of soil is usually sufficient. The soil cover should be applied without soil compaction. Therefore, the use of heavy machinery should be avoided. Alternatively, green area planning can use the different kinds of substrates for the establishment of varied vegetation types on rocky regions, raw soils (soils at an early stage of pedogenesis), acid- or carbonate-containing soils, and soils of low and high nutrient content, dry, wet, and peaty patches.

For substrates that are highly polluted, to minimize the diffusion of hazardous compounds surface or subsoil sealing is required. The soils under the impermeable sealed layer will not be exposed to leaching by precipitation. This will only work when the groundwater is well below the sealed layer.

The rehabilitation of soils polluted by organic compounds may use the potential of soils for natural attenuation due to the *in situ* degradation by microorganisms. This will probably only work for soils freshly contaminated and with low to medium contamination levels. For soils contaminated many years earlier, it can be assumed that the easily degraded compounds are already decomposed. An open question is whether the access of microorganisms to the residual polluted soil particles can be improved by *in situ* treatment. This would be an asset for the remediation of waste tips, slightly polluted soils under a soil cover and the more intensively polluted soil under sealed layers.

REFERENCES

1. Burghardt, W. The German double track concept of classifying soils by their substrate and their anthropo-natural genesis: The adoption to urban areas. In *Proceedings of the First International Conference on Soils of Urban, Industrial, Traffic and Mining Areas*, University Essen, Germany, Jul 12–18, 2000; Burghardt, W., Dornauf, C., Eds.; Working Group SUITMA/SU of IUSS: co. Fb9, Angewandte Bodenkunde/Soil Technology; University of Essen: Essen, 2000; Vol. 1, 217–222.
2. Burghardt, W. Substrate der bodenbildung urban, gewerblich und industriell überformten flächen [Substrates of the soil formation of the urban, commercial and industrial transformed sites]. In *Urbaner Bodenschutz [Urban Soil Protection]*; Arbeitskreis Stadtböden der Deutschen Bodenkundlichen Gesellschaft, Ed.; Springer: Berlin, 1996; 25–44.
3. Hiller, D.A.; Meuser, H. *Urbane Böden [Urban Soils]*; Springer: Berlin, 1998; 161 pp.
4. Giammanco, G.; Marranzano, M.; Giannino, L.R. The soil of public gardens as salmonella reservoir. *Igiene Moderna* **1984**, 82, 762–765.
5. MAGS NRW. *Metalle auf Kinderspielplätzen [Metals on Playgrounds of Children]*. Erlaß VB4-0292.5.3; Ministerium für Arbeit; Gesundheit und Soziales Nordrhein Westfalen: Düsseldorf, 1990.
6. Bachmann, G. *Fachliche Eckpunkte zur Ableitung von Bodenwerten im Rahmen des Bundesbodenschutzgesetzes [Professional Principles for the Deduction of Soil Characteristic Values in the Frame of Federal Soil Protection Law]*; Erich Schmidt: Berlin, 1988; 121 pp.
7. Federal Republic of Germany (FRG). Bundesbodenschutz- und altlastenverordnung (BBodSchV) [Soil protection and contaminated site decree of the federal republic of Germany]. *Bundesgesetzblatt Jahrgang 1999*, I (36).
8. Alloway, B.; Warren, G.; Lepp, N.; Singh, B.; Penny, C.; Bocheran, F.; Gregory, L.; Dourado, A. Minimising the risk of heavy metal exposure from contaminated soil by reducing bioavailability using adsorbitive minerals and selected garden vegetables. In *Proceedings of the First International Conference on Soils of Urban, Industrial, Traffic and Mining Areas*, University Essen, Germany, Jul 12–18, 2000; Working Group SUITMA/SU of IUSS: co.

- Fb9, Angewandte Bodenkunde/Soil Technology; Burghardt, W., Dornauf, C., Eds.; University of Essen: Essen, 2000; Vol. 3, 765–769.
9. Schwartz, C.; Perronnet, K.; Morel, J.L. Phytoremediation of urban and industrial soils contaminated by heavy metals. In *Proceedings of the First International Conference on Soils of Urban, Industrial, Traffic and Mining Areas, University Essen*, Germany, Jul 12–18, 2000; Working Group SUITMA/SU of IUSS: co. Fb9, Angewandte Bodenkunde/Soil Technology; Burghardt, W., Dornauf, C., Eds.; University of Essen: Essen, 2000; Vol. 3, 771–776.
 10. Freie und Hansestadt Hamburg. Untersuchung im öffentlichen grün [Investigation in the public green—Rehabilitation of environmental damaged street trees and park trees]. *Naturschutz und Landschaftspflege in Hamburg* **1988**, 22, 320.
 11. Liesecke, H.-J., Ed. *Dachbegrünung—Beiträge zur Extensivbegrünung [Rooftop Planting—Contributions to Extensive Greening]*; Patzer: Berlin, 1985; 130 pp.
 12. Burghardt, W.; Böhm, B.; Dornauf, C.; Rabearisoa, R. Verteilung von Stoffen aus Einträgen in Stadtböden [Distribution of compounds from deposits in urban soils]. *Bodenschutz* **1998**, 3, 92–97.
 13. Baedjer, N.; Burghardt, W. The influence of man-made materials on the solute concentration of percolating water. In *Proceedings of the First International Conference on Soils of Urban, Industrial, Traffic and Mining Areas, University Essen*, Germany, Jul 12–18, 2000; Working Group SUITMA/SU of IUSS: co. Fb9, Angewandte Bodenkunde/Soil Technology; Burghardt, W., Dornauf, C., Eds.; University of Essen: Essen, 2000; Vol. 2, 471–476.
 14. Testa, S.M. *The Reuse and Recycling of Contaminated Soil*; Lewis Publishers: Boca Raton, 1997; 243 pp.
 15. Strasser, H.; Holland, K.; Schuller, D.; Rongen, P. *Bewertungskriterien für die Folgenutzung eines Altstandortes am Beispiel des Sanierungsfalles—Nordhorn Povel [Valuation Criteria for the Future Use of a Soil Contaminated Former Industrial Site Exemplified by a Case of Restoration—(Textile Factory) Nordhorn Povel]*; UBA Texte, 89-078; Umweltbundesamt: Berlin, 1989; 131 pp.
 16. Hiller, D.; Jessen, V.; Koeppner, T.; Stenpass, R.; Burghardt, W. Characteristics of soils of a railway marshalling yard in the heavy industry area of Duisburg (Ruhr area, Germany). In *Proceedings of the First International Conference on Soils of Urban, Industrial, Traffic and Mining Areas, University Essen*, Germany, Jul 12–18, 2000; Working Group SUITMA/SU of IUSS: co. Fb9, Angewandte Bodenkunde/Soil Technology; Burghardt, W., Dornauf, C., Eds.; University of Essen: Essen, 2000; Vol. 3, 943–946.

Urban Lands: Management

Klaus Lorenz

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

The importance of urban soils and their ecosystem services for the well-being of mankind are increasing as the global urban population is growing. Specifically, urban soils support nutrient cycling, primary and especially food production, carbon storage and regulation, water storage, pollution attenuation, and regulation of biodiversity. However, awareness for those ecosystem services must be raised among stakeholders to support urban land use planning and management for enhancing the resilience of urban ecosystems.

INTRODUCTION

Earth's urban population is growing, and thus, more people depend on the management of ecosystem services arising from urban soil functions. Among the supporting services of urban soils is primary production, i.e., the support of plant growth, in particular, in urban forests, gardens, and parks. Urban green spaces contribute to climate and air quality regulation by moderating the urban heat island and trapping airborne particulate matter. Vegetation growing on pervious urban soils promotes infiltration of water and, thus, reduces the risk of surface runoff and flooding. Vegetated urban soil supports also biodiversity maintenance. Further, many opportunities for recreation in urban areas depend on green spaces. Urban soils fulfill also many specific functions such as carrying gray infrastructure (e.g., buildings and roads) and supporting waste management. How the functions and ecosystems services of urban soils can be maintained and enhanced for the well-being of the urban population to meet their increasing demands is a major challenge for urban ecosystem management. Urban soil properties are strongly affected by demolition and construction activities, and this provides an opportunity for urban soil management toward enhancing urban ecosystem services. In this entry, soil functions and ecosystem services are presented, and it will be discussed how urban soils can be managed to enhance 1) soil carbon (C) storage; and 2) food production by urban agriculture.

URBAN SOIL FUNCTIONS AND ECOSYSTEM SERVICES

Soil is among the living foundations for the human civilization as it serves essential human needs such as those for healthy and nutritious food, freshwater, and waste absorption.^[1] Urban soils, in particular, perform critical soil functions that must be considered to support urban development.^[2] Soil functions can be distinguished with

regard to their significance in urban environments.^[3] Soil functions occur in spite of the land use as, e.g., rainwater must be dispersed or regulated, and landscaping plant roots must have air available for growth. The ecosystem services provided by urban soils are the result of inherent and manageable soil properties.^[4] Urban soil ecosystem services can be distinguished based on the Millennium Ecosystem Assessment typology. Supporting ecosystem services of urban soils (e.g., nutrient cycling and primary production) are regarded less important compared to those of non-urban soils.^[5] However, increased consumption of food produced by urban agriculture may substantially enhance primary production and associated nutrient cycling in urban soils. The importance of urban soils for the provisioning ecosystem services of food production and C storage/regulation are increasing, though, to a lower extent than those of non-urban soils. Also, flood regulation by urban soils is regarded as being of low importance on a regional scale but with a large potential for enhancement for local water storage, reduction in surface runoff, and flood control.^[5] Other regulating ecosystem services of urban soils are pollution attenuation and regulation of biodiversity.^[4] Important cultural services of vegetated urban green space which rely on soils for their biogeochemical cycling include recreation/tourism and cultural heritage. Urban soils fulfill also a range of carrying functions not addressed by the Millennium Ecosystem Assessment such as carrying structures and piped utilities.^[5]

MANAGING URBAN SOIL ECOSYSTEM SERVICES

Urban soils are used by humans for specific purposes such as green spaces, housing, and other infrastructure.^[6] Within short distances of one another, urban soils deliver very different functions and provide very different ecosystem services compared to the soil natural capital. By best planning and management practices, ecosystem service

provision by urban soils can be optimized.^[7] Soil C plays a key role for supporting, regulating, provisioning, and cultural services of urban soils. Thus, how C storage of urban soils can be managed for optimizing ecosystem services and how food production on urban soils can be enhanced will be discussed in the following sections.

Managing Urban Soil C Stock for Enhancing the Provision of Ecosystem Services

Urban land use alters soil C comprised of soil inorganic carbon (SIC) and soil organic carbon (SOC). The urban soil C stock contributes to the provisioning ecosystem service C store/regulation and the regulating service climate/temperature.^[5] Climate change can be mitigated, in particular, by storing some C derived from atmospheric carbon dioxide (CO₂) in urban soils for millennia. This long-term storage occurs as bicarbonate and carbonate ionic forms as well as carbonated salts in the solid phase. Those are among the predominant stable forms of SIC.^[8] C dating of pedogenic carbonates indicates long C residence times of >30,000 years.^[9] Similarly, radiocarbon ages of 1000–>10,000 years have been reported for SOC, especially in subsoil layers, and SOC turnover times increase with an increase in soil depth.^[10] The long-lasting legacy of urban investment decisions, growth of urban activities that produce CO₂ emissions, and increasing concentration of urban population drives urgency for urban climate change policy^[11] and for soil C-friendly urban soil and land use management.^[12] However, the recommendations listed below for enhancing urban soil C stocks should be regarded as preliminary as technologies for increasing urban soil C stocks are less well known compared to those for non-urban soils.^[13]

SIC

The urban SIC stock contributes to the provision of C storage/regulation and climate/temperature regulation. The SIC comprises both primary carbonates inherited from the parent material or deposited as dusts and secondary carbonates that form by precipitation of carbonate ions derived from root, microbial and faunal respiration, and calcium (Ca) and magnesium (Mg) ions from weathering or brought from outside the urban land unit.^[14] Carbonate-bearing soil parent materials commonly occur in urban soils.^[15] Further, demolition waste (particularly cement and concrete) may also contribute to urban SIC storage.^[16] Especially, the coarse fraction (>2 mm) of urban soils may substantially contain SIC in the form of demolition waste, limestone, and chalk fragments.^[17]

The urban SIC stock can be maintained by protecting existing stocks from losses and can be enhanced by increasing stocks (Table 1). Protection includes practices to 1) reduce soil erosion losses, in particular, during construction activities when urban soil is left uncovered. Further, SIC losses from urban areas can be reduced when 2) both excavated carbonate-containing soil and coarse fraction fragments are used *in situ* for landfilling and landscaping activities within the same urban area. Special attention should be given to urban soils heavily affected by enhanced demolition activities following military conflicts (e.g., World War II) and economic decline (e.g., Rust Belt cities in the United States) as those soils may contain high amounts of SIC as a product of carbonation of deposited concrete-/cement-derived material. Also, urban soils of arid and semiarid regions should be carefully managed for SIC maintenance as they may naturally contain high SIC stocks.

Table 1 Management practices for enhancing urban soil carbon, soil inorganic carbon, and soil organic carbon stocks.

	Protecting stocks	Enhancing stocks
Soil carbon stocks	Minimizing losses by decomposition, mineralization, and erosion during construction activities Using excavated urban soil <i>in situ</i> for landfilling and soil construction	Addition of organic material
Soil inorganic carbon stocks		Cultivating vegetation species producing Ca ²⁺ - and Mg ²⁺ -rich litter Addition of Ca ²⁺ - and/or Mg ²⁺ -containing soil amendments and waste materials Irrigation with alkaline water
Soil organic carbon stocks		Establishing vegetation on bare soil Optimizing net primary production and especially root production Urban forestry Burial of organic matter and black carbon in soil under impervious surfaces and at deeper soil depths under pervious surfaces Cultivating plant species that divert root carbon into stable mineral-associated soil organic carbon fractions

The CO₂-enriched water in the pore space may precipitate Ca²⁺ and Mg²⁺ released during decomposition of organic matter (OM) aside reacting with soil-derived Ca²⁺ and Mg²⁺. Increasing SOC stock may also amplify carbonate weathering and precipitation by promoting microbial activity which in turn causes an increase in the release of CO₂, Ca²⁺, and Mg²⁺.^[18] Thus, urban SIC stocks may also be enhanced by 3) increasing the SOC stock by adding OM or 4) by increasing urban vegetation cover producing large amounts of Ca²⁺- and Mg²⁺-rich litter. The formation of new soil carbonates requires outside Ca²⁺ and/or Mg²⁺ sources such as fertilizers, liming material (i.e., containing calcite or dolomite), or other soil amendments.^[19] Thus, 5) amending urban soils with those materials may cause an increase in SIC stocks. Other common sources and processes for carbonate occurrence in urban soils are precipitation from CO₂-enriched natural surface or irrigation water containing Ca²⁺ and/or Mg²⁺ and precipitation in soils from groundwater that has moved through carbonate-containing soils or sediments.^[20] Thus, 6) irrigating urban soils with alkaline water may result in an increase in SIC stocks as was observed, e.g., in urban soils of China.^[21,22] Waste materials in urban soils derived from construction materials may contain Ca silicate and Mg silicate minerals. Naturally, those minerals occur in basic igneous rocks including those quarried for aggregate.^[23] Those minerals are also produced in cement and iron-making and steel-making slag. Weathering of such materials (i.e., Ca-rich minerals such as silicates, hydroxides, and sulfates) in urban soils may be subsequently followed by pedogenic carbonate precipitation. Materials include Ca-rich components such as artificial mortars, plaster, concrete, natural basic rock aggregates, and slag.^[9] Thus, another recommendation to enhance SIC stocks in urban soils is 7) to use Ca- and Mg-rich materials as soil amendments.

SOC

Aside contributing to climate change mitigation, protection and increasing urban SOC stocks may also support critically important soil-derived ecosystem services, including the supporting service nutrient cycling; the provision of food, freshwater, wood, and fiber; and the regulating ecosystem services of flood control and attenuation of water quantity.^[5] The benefits of SOC are well known and related to its fundamental role for the function and fertility of terrestrial ecosystems.^[24] Specifically, there are strong interactions between SOC and soil, water, and air quality.^[25]

Urban soil management may be directed toward maintaining or enhancing SOC stocks (Table 1). The persistence of SOC is largely due to complex interactions between SOC and its environment, such as the interdependence of compound chemistry, reactive mineral surfaces, climate, water availability, soil acidity, soil redox state, and the presence of potential degraders in the immediate micro-environment.^[26] By altering those factors, urban soil

management may be optimized for SOC persistence. In urban ecosystems, SOC-enhancing management practices may be used to restore SOC in bare soils by 1) revegetation and in vegetated green space soils by 2) fertilization, irrigation, reduced soil disturbance, and residue management (i.e., returning grass clippings). Further, urban soils can potentially be optimized for SOC accumulation by 3) adding organic amendments such as biosolids, yard, and food waste during soil construction activities.^[27] In particular, lawn SOC stocks may benefit from practices targeted specifically at SOC as urban lawns are often intensively managed. Also, garden management practices, such as the addition of peat, composts, and mulches, and cultivation of trees and shrubs contribute to greater SOC stocks in urban gardens.^[28] Specifically, 4) tree planting and management of existing tree cover by urban forestry practices likely provide the greatest scope for enhancing urban SOC stocks.^[23] Urban sites covered by trees are often relatively undisturbed compared to other vegetated urban sites allowing long time for SOC accumulation. However, how to increase C restitution from tree biomass to SOC storage in urban soils is less well known.^[29]

Urban soils have often an impervious cover under which soil microbial activities are reduced, and thus, buried OM and SOC may decompose slower than those of pervious urban soils. Thus, 5) burial of large amounts of OM during the construction of imperviously covered urban soil may be a practice to enhance the provisioning ecosystem service C store/regulation. The SOC stock can be further strengthened by 6) adding “recalcitrant” compounds such as combustion derived black carbon (BC) as those may dominate SOC of imperviously covered soils.^[30] Adding BC compounds such as biochar or char to urban soils has the potential to enhance SOC stocks and improve urban soil quality.^[31–33] Further, asphalt, coal, coal ash, and tar can contribute to urban soil BC.^[34–36] However, research into the use of BC and, especially, biochar to enhance SOC and improve soil quality is in its early stages.^[37] Little thought has been given, in particular, to biochar application to urban soils and gardens.^[23]

Open urban soils should be vegetated and the vegetation maintained to optimize net primary production as this process is the primary source of soil C inputs such as plant litter and exudates and dissolved organic carbon.^[10] Best management practices (e.g., reduced physical disturbance, fertilization, and irrigation) should, in particular, 7) optimize root growth and root turnover as SOC may be primarily originating from root-derived C inputs.^[38] Further, 8) burial of soil and/or carbonaceous materials at deeper soil depths can contribute to the persistence of the provisioning ecosystem service C store/regulation.^[39] For example, enrichment of urban soil profiles mainly with anthropogenic OM up to 1.9 m depth was reported for some soils in Stuttgart, Germany.^[35] Additional storage of SOC at deeper soil depths in urban areas may also be possible by 9) cultivating plants with deep and prolific root systems.^[40] The strategy

for enhancing SOC is to select plant species that better divert root-derived C into stable mineral-associated SOC fractions. Thus, selection of appropriate vegetation in planning and design of new urban developments may be important for enhancing SOC stocks. Further, 10) adding clay-sized soil fractions to urban soils may provide preferential spots for SOC accumulation.^[41] In summary, vegetation species, soil compaction, and sward/litter management must be controlled for optimum SOC accumulation in open urban soils.^[23]

Enhancing the Provision of Food by Management of Urban Soils for Agricultural Production

Urban soils support only the provision of a small proportion of global food production as a result of urban agricultural practices. This proportion may increase as urban agriculture has been proposed as a solution for producing food in places where population density is highest, reducing transportation costs, connecting people directly to food systems, and using urban areas and their soils more efficiently. However, this expansion would require one-third of the total global urban area alone to meet the global vegetable consumption of urban dwellers, and those estimates didn't even consider how much urban area may actually be suitable and available for urban agriculture.^[42] Further, there is only a limited potential contribution of urban agriculture to global cereal production as the global annual harvested area for cereals is about 10 times greater than the global urban area. Nevertheless, urban agriculture is an important reality for many households, especially in the developing countries.^[43]

Agricultural production in urban ecosystems is challenging as urban soils are often less fertile than

non-urban soils and their biological, chemical, and physical properties are degraded. Specifically, urban soils are often compacted, low in soil organic matter (SOM), high in inorganic and organic contaminants, and characterized by altered soil moisture characteristics. As discussed previously, SOM in urban soils can be enriched for enhancing agricultural production by applying composts made from biosolids, yard, and food wastes and amending soils with BC compounds such as biochar (Table 2). Further, cover cropping and reduced tillage are intensive management strategies to increase SOM contents for urban agricultural production.^[44] Adding OM and intensive cover cropping with deep-rooted cover crops or crops having an extensive root profile are also appropriate management strategies to reduce soil compaction and improve urban soil quality. Similar to the alleviation of compaction in agricultural soils, subsoiling or deep tillage practices may be used to reduce compaction in soils for urban agricultural. In cases where it is not economically feasible to reduce soil compaction (e.g., by removing impervious surfaces such as asphalt), soils may be created on top of the compacted layers. For example, large quantities of compost, wood chips, and other forms of OM are used to create raised beds over compacted urban soil layers. However, knowledge about urban agricultural practices using raised beds is scanty.^[44]

The moisture characteristics of urban soils are altered by physical disturbance; removal or desurfacing; burial or coverage of soil by fill material or impervious surfaces; and soil, water, and vegetation management practices (e.g., fertilization, irrigation, mowing, and drainage).^[45] For poorly drained urban soils, both surface and subsurface drainage are management options to improve the soil moisture characteristics for urban agriculture.^[46] On the

Table 2 Enhancing agricultural production by managing urban soils.

Strategy	Expected outcome
Applying composts, biosolids, mulches, yard, and food wastes; adding black carbon compounds (including biochar); cover cropping; reduced tillage	Enrichment of soil organic matter; improvement infiltration and water-holding capacity
Applying organic matter, planting deep-rooted cover crops or crops with extensive root profiles, subsoiling, deep tillage, and raised beds	Reduction in soil compaction, improvement in soil quality
Surface and subsurface drainage	Improvements in soil moisture characteristics
Excavation and soil washing, bioremediation, or thermal treatment	Ex situ decontamination
Applying liming compounds or organic matter	Soil stabilization of cadmium, nickel, zinc, copper, lead, chromium, and organic pollutants
Applying fertilizers and improving soil conditions for rapid growth of microorganisms	Bioremediation of organic pollutants by microorganisms
Growing and harvesting of hyperaccumulator plant species	Phytoextraction of cadmium, nickel, and zinc
Planting trees with extensive and deep root systems	Phytodegradation of organic pollutants by microorganisms
Importing clean soil and establishing raised beds	Safe production of vegetables
Natural attenuation	Reduced concentrations of organic pollutants

other hand, urban soils may also exhibit elevated water deficit for plant growth. The previously discussed soil addition of OM in the form of compost, mulch, and other amendments can both improve infiltration and increase the water-holding capacity of soils for urban agricultural use.

Urban soils for agricultural use may be contaminated with heavy metals and organic pollutants, e.g., petroleum hydrocarbons, polycyclic aromatic hydrocarbons and polychlorinated biphenyls.^[47] Possible contamination sources are urban dust deposition and anthropogenic deposits, e.g., in derelict land previously used for industrial purposes (brownfields). Further, anthropogenic activities such as fertilization and application of sewage sludge, wastewater, and pesticides may also contribute to contamination of urban soils and restrict their use for agriculture. For decontamination, urban soils may be excavated and treated *ex situ* by using technical devices for soil washing, bioremediation, and thermal treatment (Table 2).^[48] However, more cost-effective are *in situ* methods involving stabilizing approaches for contaminants. For example, applying liming compounds or OM is recommended remediation techniques for stabilizing cadmium (Cd), nickel (Ni), zinc (Zn), copper, lead, chromium, and some organic pollutants in soils. Organic pollutants soluble to some degree in water can be treated *in situ* by bioremediation which depends on the activity of soil microorganisms. Phytoextraction that describes the uptake of contaminants by plant roots and subsequent plant harvest may also be used to reduce the concentration of Cd, Ni, and Zn in soils for subsequent urban agricultural use.^[49] Phytodegradation leading to microbial metabolism in the plant rhizosphere and subsequent degradation of organic pollutants may also be used for decontamination of urban soils. For example, poplar trees (*Populus* spp.) are hardy against many soil contaminants and growing conditions and enhance the microbial degradation of hydrocarbons.^[50] Further, organic pollutants may be removed by phytovolatilization leading to gaseous losses of organic contaminants after metabolism in plant tissues.^[49] If budgets are limited, but sufficient time is available to clean up a soil before it is used for urban agriculture, natural attenuation can be used to reduce the concentration of organic pollutants *in situ*.^[48] However, if urban soils cannot be decontaminated to levels for safe production of vegetables, establishing urban agriculture on imported clean soil or raised beds on top of the contaminated soil may be an option. As deposition of contaminants with urban dust can occur, monitoring soil contaminant concentrations in imported soil and raised beds is needed and, if necessary, soil decontamination for growing vegetables in urban areas.^[51]

CONCLUSION

More and more people depend on the ecosystem services arising from urban soil functions as earth's urban

population is growing. This increasing importance of urban soils for human well-being must be considered to support urban development. Supporting ecosystem services of urban soils include nutrient cycling and primary production. Further, urban soils are increasingly managed for food production and can contribute to C storage/regulation. Urban soils have a large potential for enhancement of local water storage, reduction in surface runoff, and flood control. Other regulating services of urban soils also increasing in significance are pollution attenuation and regulation of biodiversity. By best soil management practices, the inorganic and organic C stocks of urban soil can be enhanced, and urban agricultural production can be optimized. However, it is critically important to raise awareness among stakeholders for urban soils, their functions, and ecosystem services. The enhanced recognition of urban soils will support urban land use planning and management for enhancing the resilience of urban ecosystems.

REFERENCES

1. Montgomery, D.R. *Dirt: The Erosion of Civilizations*; University of California Press: Berkeley, 2007.
2. James, P.; Holden, M.; Lewin, M.; Neilson, L.; Oakley, C.; Truter, A.; Wilmoth, D. Managing metropolises by negotiating urban growth. In *Institutional and Social Innovation for Sustainable Urban Development – Routledge Studies in Sustainable Development*; Mieg, H.A., Töpfer, K., Eds.; Routledge: Abingdon, 2013; 217–232.
3. Lehmann, A.; Stahr, K. The potential of soil functions and planner-oriented soil evaluation to achieve sustainable land use. *J. Soil. Sediment.* **2010**, *10*, 1092–1102.
4. Morel, J.L.; Chenu, C.; Lorenz, K. Ecosystem services provided by soils of urban, industrial, traffic, mining, and military areas (SUITMAs). *J. Soil. Sediment.* **2015**, *15*, 1659–1666.
5. Rawlins, B.G.; Harris, J.; Price, S.; Bartlett, M. A review of climate change impacts on urban soil functions with examples and policy insights from England, UK. *Soil Use Manage.* **2015**, *31*, 46–61.
6. Setälä, H.; Bardgett, R.D.; Birkhofer, K.; Brady, M.; Byrne, L.; de Ruiter, P.C.; de Vries, F.T.; Gardi, C.; Hedlund, K.; Hemerik, L.; Hotes, S.; Liiri, M.; Mortimer, S.R.; Pavao-Zuckerman, M.; Pouyat, R.; Tsiafouli, M.; van der Putten, W.H. Urban and agricultural soils: Conflicts and trade-offs in the optimization of ecosystem services. *Urban Ecosyst.* **2014**, *17*, 239–253.
7. Chen, Y.; Day, S.D.; Wick, A.F.; McGuire, K.J. Influence of urban land development and subsequent soil rehabilitation on soil aggregates, carbon, and hydraulic conductivity. *Sci. Total Environ.* **2014**, *494–495*, 329–336.
8. Macías, F.; Arbestain, M.C. Soil carbon sequestration in a changing global environment. *Mitig. Adapt. Strateg. Glob. Change* **2010**, *15*, 511–529.
9. Renforth, P.; Manning, D.A.C.; Lopez-Capel, E. Carbonate precipitation in artificial soils as a sink for atmospheric carbon dioxide. *Appl. Geochem.* **2009**, *24*, 1757–1764.

10. Rumpel, C.; Kögel-Knabner, I. Deep soil organic matter—A key but poorly understood component of terrestrial C cycle. *Plant Soil* **2011**, *338*, 143–158.
11. Marcotullio, P.J.; Sarzynski, A.; Albrecht, J.; Schulz, N.; Garcia, J. The geography of global urban greenhouse gas emissions: An exploratory analysis. *Climatic Change* **2013**, *121*, 621–634.
12. Seto, K.C.; Güneralp, B.; Hutyra, L.R. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *P. Natl. Acad. Sci. U.S.A.* **2012**, *109*, 16083–16088.
13. Whitmore, A.P.; Kirk, G.J.D.; Rawlins, B.G. Technologies for increasing carbon storage in soil to mitigate climate change. *Soil Use Manage.* **2015**, *31*, 62–71.
14. Lal, R.; Kimble, J.M. Pedogenic carbonates and the global carbon cycle. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2010; 291–302.
15. Lehmann, A.; Stahr, K. Nature and significance of anthropogenic urban soils. *J. Soil. Sediment.* **2007**, *7*, 247–296.
16. Washbourne, C.L.; Renforth, P.; Manning, D.A.C. Investigating carbonate formation in urban soils as a method for capture and storage of atmospheric carbon. *Sci. Total Environ.* **2012**, *431*, 166–175.
17. Rawlins, B.G.; Henrys, P.; Breward, N.; Robinson, D.A.; Keith, A.M.; Garcia-Bajo, M. The importance of inorganic carbon in soil carbon databases and stock estimates: A case study from England. *Soil Use Manage.* **2011**, *27*, 312–320.
18. Bronick, C.J.; Lal, R. Soil structure and management: A review. *Geoderma* **2005**, *124*, 3–22.
19. Nordt, L.C.; Wilding, L.P.; Drees, L.R. Pedogenic carbonate transformations in leaching soil systems: Implications for the global C cycle. In *Global Climate Change and Pedogenic Carbonates*; Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A., Eds.; CRC Press: Boca Raton, 2010; 43–64.
20. Denef, K.; Stewart, C.E.; Brenner, J.; Paustian, K. Does long-term center-pivot irrigation increase soil carbon stocks in semi-arid agro-ecosystems? *Geoderma* **2008**, *145*, 121–129.
21. Xu, N.; Liu, H. Spatial distribution of soil inorganic carbon in urbanized territories. *Adv. Materials Res.* **2013**, *726–731*, 188–193.
22. Xu, N.; Liu, H.; Wei, F.; Zhu, Y. Urban expanding pattern and soil organic, inorganic distribution in Shanghai, China. *Environ. Earth Sci.* **2012**, *66*, 1233–1238.
23. Renforth, P.; Edmondson, J.; Leake, J.R.; Gaston, K.J.; Manning, D.A.C. Designing a carbon capture function into urban soils. *P. I. Civil Eng. Urban Des. Plann.* **2011**, *164*, 121–128.
24. Janzen, H.H. The soil carbon dilemma: Shall we hoard it or use it? *Soil Biol. Biochem.* **2006**, *38*, 419–424.
25. Smith, P.; Haberl, H.; Popp, A.; Erb, K.H.; Lauk, C.; Harper, R.; Tubiello, F.N.; Pinto, A.D.; Jafari, M.; Sohi, S. How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals? *Glob. Change Biol.* **2013**, *19*, 2285–2302.
26. Schmidt, M.W.I.; Torn, M.S.; Abiven, S.; Dittmar, T.; Guggenberger, G.; Janssens, I.A.; Kleber, M.; Kögel-Knabner, I.; Lehmann, J.; Manning, D.A.C.; Nannipieri, P.; Rasse, D.P.; Weiner, S.; Trumbore, S.E. Persistence of soil organic matter as an ecosystem property. *Nature* **2011**, *478*, 49–56.
27. Brown, S.; Miltner, E.; Cogger, C. Carbon sequestration potential in urban soils. In *Carbon Sequestration in Urban Ecosystems*; Lal, R., Augustin, B., Eds.; Springer: Dordrecht, 2012; 173–196.
28. Edmondson, J.L.; Davies, Z.G.; McCormack, S.A.; Gaston, K.J.; Leake, J.R. Land-cover effects on soil organic carbon stocks in a European city. *Sci. Total Environ.* **2014**, *472*, 444–453.
29. Scharenbroch, B.C. Urban trees for carbon sequestration. In *Carbon Sequestration in Urban Ecosystems*; Lal, R., Augustin, B., Eds.; Springer: Dordrecht, 2012; 121–138.
30. Raciti, S.M.; Hutyra, L.R.; Finzi, A.C. Depleted soil carbon and nitrogen pools beneath impervious surfaces. *Environ. Pollut.* **2012**, *164*, 258–261.
31. Beesley, L.; Dickinson, N. Carbon and trace element fluxes in the pore water of an urban soil following greenwaste compost, woody and biochar amendments, inoculated with the earthworm *Lumbricus terrestris*. *Soil Biol. Biochem.* **2011**, *43*, 188–196.
32. Ghosh, S.; Yeo, D.; Wilson, B.; Ow, L.F. Application of char products improves urban soil quality. *Soil Use Manage.* **2012**, *28*, 329–336.
33. Scharenbroch, B.C.; Meza, E.N.; Catania, M.; Fite, K. Biochar and biosolids increase tree growth and improve soil quality for urban landscapes. *J. Environ. Qual.* **2013**, *42*, 1372–1385.
34. Rawlins, B.G.; Vane, C.H.; Kim, A.W.; Tye, A.M.; Kemp, S.J.; Bellamy, P.H. Methods for estimating types of soil organic carbon and their application to surveys of UK urban areas. *Soil Use Manage.* **2008**, *4*, 47–59.
35. Stahr, K.; Stasch, D.; Beck, O. *Entwicklung von Bewertungssystemen für Bodenressourcen in Ballungsräumen*. BWPLUS-Projekt BWC 99001. Available at <http://www.fachdokumente.lubw.baden-wuerttemberg.de/servlet/is/40148/?COMMAND=DisplayBericht&FIS=203&OBJECT=40148&MODE=METADATA> (accessed September 2014).
36. Yang, Y.; Mahler, B.J.; Van Metre, P.C.; Ligouis, B.; Werth, C.J. Potential contributions of asphalt and coal tar to black carbon quantification in urban dust, soils, and sediments. *Geochim. Cosmochim. Acta* **2010**, *74*, 6830–6840.
37. Lorenz, K.; Lal, R. Biochar application to soil for climate change mitigation by soil organic carbon sequestration. *J. Plant Nutr. Soil Sci.* **2014**, *177*, 651–670.
38. Rasse, D.P.; Rumpel, C.; Dignac, M.-F. Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. *Plant Soil* **2005**, *269*, 341–356.
39. Chaopricha, N.T.; Marín-Spiotta, E. Soil burial contributes to deep soil organic carbon storage. *Soil Biol. Biochem.* **2014**, *69*, 251–264.
40. Kell, D.B. Large-scale sequestration of atmospheric carbon via plant roots in natural and agricultural ecosystems: Why and how. *Philos. T. R. Soc. Lon. B.* **2012**, *367*, 1589–1597.
41. Churchman, G.J.; Noble, A.; Bailey, G.; Chittleborough, D.; Harper, R. Clay addition and redistribution to enhance carbon sequestration in soils. In *Soil Carbon*, Progress in Soil Science; Hartemink, A.E., McSweeney, K., Eds.; Springer International Publishing Switzerland: Bern, 2014; 327–335.

42. Martellozzo, F.; Landry, J.-S.; Plouffe, D.; Seufert, V.; Rowhani, P.; Ramankutty, N. Urban agriculture: A global analysis of the space constraint to meet urban vegetable demand. *Environ. Res. Lett.* **2014**, *9* (6), 064025.
43. De Bon, H.; Parrot, L.; Moustier, P. Sustainable urban agriculture in developing countries. A review. *Agron. Sustain. Dev.* **2010**, *30*, 21–32.
44. Beniston, J.; Lal, R. Improving soil quality for urban agriculture in the North Central U.S. In *Carbon Sequestration in Urban Ecosystems*; Lal, R., Augustin, B., Eds.; Springer: Dordrecht, 2012; 279–314.
45. Lorenz, K.; Lal, R. Terrestrial carbon management in urban ecosystems and water quality. In *Carbon Sequestration in Urban Ecosystems*; Lal, R., Augustin, B., Eds.; Springer: Dordrecht, 2012; 73–102.
46. Craul, P.J. *Urban Soil in Landscape Design*; John Wiley: New York, 1999.
47. Meuser, H. *Contaminated Urban Soils, Environmental Pollution*, Vol. 18; Springer: Dordrecht, 2010.
48. Meuser, H. *Soil Remediation and Rehabilitation – Treatment of Contaminated and Disturbed Land, Environmental Pollution*, Vol. 23; Springer: Dordrecht, 2013.
49. Koptsik, G.N. Problems and prospects concerning the phytoremediation of heavy metal polluted soils: A review. *Eurasian Soil Sci.* **2014**, *47*, 923–939.
50. Fisk, S. Soil microbes and plants work together to clean up contaminated soils. *Soil Horizons* **2014**, *55*, doi:10.2136/sh2014-55-5-f.
51. Säumel, I.; Kotsyuk, I.; Hölscher, M.; Lenkerei, C.; Weber, F.; Kowarik, I. How healthy is urban horticulture in high traffic areas? Trace metal concentrations in vegetable crops from plantings within inner city neighbourhoods in Berlin, Germany. *Environ. Pollut.* **2012**, *165*, 124–132.

Urban Wastewater for Irrigation: Heavy Metals

P.S. Minhas

Punjab Agricultural University, Ludhiana, India

K. Lal

Central Soil Salinity Research Institute, Karnal, India

Abstract

Wastewater primarily produced from municipalities and industries contains a variety of pollutants, and their usage has certain negative environmental impacts in the form of pathogens causing diseases, heavy metal accumulation, and soil salinization. In the developing countries, where wastewater is generally used for irrigating high-value crops like vegetables and fodders, the bioavailability and mobility of heavy metals is a matter of great concern as these parameters govern their uptake by plants and, in turn, food chain contamination. To limit the adverse effects, the guidelines for the treatment and usage of wastewater must be strictly observed. Due to resource constraints and prohibitive costs of conventional treatment in the developing countries, larger portions of wastewater are disposed either raw or after partial treatment in streams and land, or they are used for irrigation in peri-urban agriculture. Heavy metals, unlike organic pollutants, cannot be destroyed or converted into other harmless forms. Both phyto- and georemediation measures have been proposed for minimizing the concentration of heavy metals in the food chain. The afforestation, comprising trees of high economic value, efficient microbial strains, and wet lands, cultivation of remunerative but inedible crops, and application of organic and inorganic amendments, biofilters, conjunctive and regulated water use, are some of the useful strategies to mitigate or reduce the adverse impacts of wastewater irrigation.

INTRODUCTION

Sustainable management of soil and water resources in agriculture is the key and the most essential strategy for providing food and livelihood security. However, these natural resources are being contaminated with heavy metals because of the usage of large quantities of wastewater produced from continually expanding populations along with rapid industrial growth and urbanization. Wastewater is the water that has been used in domestic or industrial processes, and during the utilization process, its quality deteriorates because of excessive soluble salts, metals, metalloids, pathogens, and other organic and inorganic pollutants. Its use calls for sound management strategies to protect these resources from further degradation. Because of scarcity of freshwater especially in the arid and semiarid regions, even the highly polluted (e.g., selenium (Se), arsenic (As), cadmium (Cd), and chromium (Cr)) waters, including urban wastewaters, are being utilized for irrigation. The practice of wastewater irrigation is widely adopted by resource-poor farmers of the developing countries ignoring the potential health risks. Wastewater will also be increasingly used in the future because of higher freshwater allocation to industry and municipalities. This entry discusses some issues related to toxic constituents of waters, contamination of soils by heavy

metals, biomagnification in food chain by heavy metals, and some remedial measures.

URBAN WASTEWATER A SOURCE FOR LIVELIHOOD

With rapid urbanization and industrialization, huge volumes of the wastewater are generated, which need safe disposal. Only 15–25% of the water supplies to urban and industrial sectors are consumed and the remainder returned as wastewater. Peri-urban agriculture uses raw sewage, sludge, and industrial effluents in cultivated fields. These sources of water are used for irrigating high-value vegetables, fruits, food grains, fodder, and industrial crops. In poor economies, dependence on wastewater is a fact of life not a matter of choice. About two million hectares of agricultural land is irrigated with raw or partially treated, diluted, or treated wastewater.^[1] In closed basins, such as in some Middle Eastern countries, up to 70% of irrigation requirement is met using wastewater.^[2] Furthermore, the use of wastewater is increasing because of the combination of many interacting factors: water scarcity, assured irrigation, lack of alternate sources, proximity to market for perishable items, and affordable source of nutrient. Considerable increases in crop yields (up to 37%) in

different cropping systems have been obtained with wastewater use compared to those with freshwater and chemical fertilizer application. The increase in crop yields with the use of wastewater may be ascribed to added nutrients, e.g., the 10-cm wastewater irrigation per hectare supplies variable amount of nutrients: 16–62 kg nitrogen, 4–24 kg phosphorus, 2–69 kg potassium, 18–208 kg calcium, 9–100 kg magnesium, and 27–182 kg sodium, depending on the nutrient concentration and quantities of wastewater applied. However, it is seldom known which nutrients are the most beneficial and what their optimum quantities are, suggesting the strong need for periodic monitoring to avoid imbalanced nutrient supply.

SOURCES OF HEAVY METALS

Heavy metals [lead (Pb), Cd, Cr, mercury (Hg), nickel (Ni), Se, zinc (Zn), copper (Cu), and As] are the elements having a specific gravity of about 6.0 g cm^{-3} or greater and atomic weight more than that of iron (Fe). Not all the heavy metals are toxic, and in small quantities, many are essential to plant growth (Fe, manganese (Mn), molybdenum (Mo), Zn, and Cu). Although metals and metalloids are usually present in soil parent materials, there are several other anthropogenic sources (Table 1) such as mining, sludge production from industrial effluents, fossil fuel combustion, metallurgical industries, paints and pigments, and batteries. Among industries, Ni or chrome plating, engineering, tannery, textile, paper mill, power generating, and distillery are major contributors. Disposal of domestic, municipal, and industrial wastes is one of the major anthropogenic sources, which produces significant quantities of substances that harm environment. In addition to the abovementioned sources, agricultural activities, such as impurities in fertilizers, application of sewage sludge as a source of plant nutrients, pesticides, and irrigation with metal-rich waters, also contribute metal ions significantly to their total concentrations in soils especially in regions of intensive farming.

Treating the wastewaters before disposal as per guidelines is rather expensive. Thus, wastewater treatment comprises only 35% in Asia, 14% in Latin America, and negligible amount in Africa.^[3] The rest is disposed without treatment, thereby exacerbating the problem of soil and water contamination. It is always easier and less expensive to treat the wastewater at point source, i.e., industry level, than they are mixed resulting in large volumes of wastewater with complex composition and soil contamination. The stringent effluent emission standards formulated are rarely adhered by the industries causing contamination of soil–plant–animal continuum. In the developing countries, 70% of industrial wastes are dumped without treatment. Even after treatment, 40–60% of the heavy metals remain in wastewater. The magnitude of contamination depends on the amount and composition of wastewater disposed from

Table 1 List of anthropogenic sources of some selected metals and metalloids.

Chemical	Symbol	Major uses and sources ^a
Arsenic	As	Pesticides, animal feed additives, and manures from intensive animal production, desiccants, fossil fuel combustion, electronics, pigments and paints, medical uses, and groundwater
Cadmium	Cd	Electroplating, pigments for plastics and paints, batteries, sewage sludge, fossil fuel combustion, polymer stabilizers, waste disposal, and impurities in fertilizers
Chromium	Cr	Chrome-plated metals, pigments, electronics, fossil fuel combustion, impurities in fertilizers, wood preservatives, stainless steel
Copper	Cu	Fly ash, wind-blown copper containing dust, impurities in fertilizers, warfare and military training, waste disposal, medical uses, electronics, sewage sludge, wood preservatives, pesticides, and fungicides
Lead	Pb	Fossil fuel combustion, batteries, pigments and paints, polymer stabilizers, printing, sports shooting, warfare, and military training, waste disposal, impurities in fertilizers, and pesticides
Mercury	Hg	Pesticides, chlorine manufacture, medical uses, batteries, and electronics
Nickel	Ni	Alloy manufacture, electroplating, sewage sludge, batteries, and catalyst
Selenium	Se	Pigments, paints, printing, medical uses, and groundwater
Zinc	Zn	Metallurgical industries, electronics, pesticides, impurities in fertilizers, sewage sludge, pigments and paints, polymer stabilizers, printing, and medical uses

^aIn addition to mining of ore bodies in the earth's crust, smelting of scrap material, and weathering of soil minerals.

domestic and industrial sources, their mixing ratio, kind of raw material used by the industries, and degree of treatment used prior to disposal.

HEAVY METAL BIOAVAILABILITY AND CONTAMINATION OF FOOD CHAIN

Bioavailability and mobility of these metals rather than their absolute concentration in soils is a matter of great concern as these parameters govern the heavy metal uptake by plant and the attendant food chain contamination. The use of sewage and industrial effluents enhances the available metal status of agricultural soils by 2 to 100 times.^[4] Heavy metals are readily fixed in soils, and their contents

Table 2 Health problems associated with heavy metal accumulations in humans.

Heavy metal	Health problem
Cr, Cd, Hg, Pb, As, and Zn	Carcinogenic
As	Jaundice, reduced blood cells, and facial edema
Pb	Kidney damage, reduced reproductive capacity, disturbance in blood chemistry, and neurological disorders
Cr	Affects sugar metabolism
Ni	Skin cancer, asthma, and bronchial cancer
Zn	Affects elementary canal, metal fume fever or Monday morning fever, protein metabolism, and bronchial cancer
Hg	Affects kidney, heart, blood, and muscles
Fe	Blood chemistry, hemorrhagic gastritis, bronze diabetes, heart disease, and damage to kidney

are negatively correlated with soil pH and positively with organic carbon and clay content of soils. Although most of the added heavy metals accumulate in few centimeters of surface soil, leaching of contaminants below the plant root zone may, in the long run, affect the quality of groundwater resources. The actual threat depends on a host of factors including depth of water table, soil characteristics, and scale of heavy metals being added.

The indiscriminate and excessive use of wastewater causes a gradual heavy metal contamination in soil to the levels that might become toxic to crops. The heavy metal bioavailability depends on their potentially available amount, the activity in the soil solution, and the rate of

transfer from soil to plant roots. There is an unambiguous relationship between heavy metal accumulation in food crops and its source in wastewater from industries, municipalities, and domestic sources. Dietary intake of contaminated food and consumption of meat and milk obtained from the animal fed on fodder produced with wastewater are the major sources of food chain contamination, which have serious implications for the health and livelihood of the consumers and for the poor, in particular. Some of the important health problems associated with heavy metals are given in Table 2.

Crops also show preferential absorption and specificity toward heavy metals. Leafy vegetables (lettuce and spinach) and root crops are generally hyperaccumulator of heavy metals. The maximum accumulation seems to occur in roots followed by leaves, and this trend is very conspicuous in case of Ni. Thus, plants themselves act as filters to check the translocation of heavy metals from soils to consumable plant parts, i.e., seeds/fruits.^[5] This is especially true for Ni, Cu, and Pb. Further, Ni and Cu have added protective mechanism in terms of preemergence of phytotoxicity, where plants die or fail to grow much before they accumulate the metal ions to levels which are toxic to human beings (Table 3). However, Cd, which is carcinogenic and causes *itai-itai* disease (severely painful joints), escapes the “soil–plant barrier” and tends to accumulate at higher levels with the increasing Cd concentration in soils. The filtering at the root level may be less effective, but Cd tends to be partially excluded from fruits, as its contents are generally higher in foliar tissues than in fruits, seeds, and roots.

To contain heavy metal buildup, wastewaters should be evaluated for possible hazards. For safe crop production

Table 3 Recommended maximum concentration of heavy metals in irrigation water, maximum allowable limits, annual loading rates, recommended safe intake, and toxic limit.

Element	Recommended maximum concentration (mg l ⁻¹) in irrigation water ^a	Maximum allowable limit in soil (mg kg ⁻¹) ^b	Cumulative loading rate (kg ha ⁻¹) ^c	Recommended safe intake ^d	Toxic limit ^d
As	0.10	–	41	15–25 µg day ⁻¹	3 mg day ⁻¹ for 2–3 weeks
Cd	0.01	3–5	39	<70 µg day ⁻¹	200 µg kg ⁻¹ of fresh weight
Cr	0.10	50–100	3,000	50–200 µg day ⁻¹	–
Cu	0.20	100	1,500	2 mg day ⁻¹	12 mg day ⁻¹
Fe	5.0	–	–	14 mg day ⁻¹	>45 mg day ⁻¹
Ni	0.20	50–100	420	7.3 µg kg ⁻¹ body weight	–
Pb	5.0	100	300	20–282 µg day ⁻¹	>500 µg day ⁻¹
Se	0.02	–	100	100–200 µg day ⁻¹	9 mg day ⁻¹
Zn	2.0	300	2,800	15 µg day ⁻¹	150 µg day ⁻¹

^aMaximum permissible limits in irrigation water (quoted by Pescod^[6]).

^bThe maximum allowable limits for European countries (Austria, Poland, and Great Britain; quoted by Kabata-Pendias^[7]).

^cMaximum lifetime cumulative loading rates (U.S. Environmental Protection Agency^[8]).

^dRecommended safe and toxic intake limits for an adult human (Oliver^[9]).

and soil resource protection, the maximum permissible limits of heavy metals in irrigation waters^[6] and the maximum allowable limits and lifetime cumulative loading rates in soil have been proposed.^[7,8] These are presented in Table 3 including the recommended safe intake and toxic levels of heavy metal for an adult human being.^[9]

MANAGEMENT OPTIONS

Dealing with pathogens is much easier than with the heavy metals in wastewater. The industrial effluents must be collected separately and should not be mixed with the domestic wastewater. Polluter must pay, and therefore, industries must be encouraged to treat effluents at site and minimize its disposal.

Inorganic amendments (e.g., lime, phosphate, and alum), which form insoluble complexes or raise the pH, reduce heavy metal toxicity in soil or groundwater. Similarly, enhancing aeration of sewage-irrigated soils is important. These soils usually remain wet and anaerobic due to high contents of organics. Thus, drainage, regular tillage, and deficit irrigation should help in the formation of inactive oxidized forms of several toxic elements. Oxidized forms are generally less soluble and thus less available for plant uptake. Biosorbent organic materials with high specific area (e.g., saw dust, lignite, and peat) also reduce the heavy metal load of wastewater. Hyperaccumulating plant species, such as willow, mustard, tobacco, *Thlaspi*, and *Alyssum*, are the low-cost and plant-based phytoextraction measures. Even after the treatment, it must be ensured that wastewater is not applied to crops intended for human consumption. To avoid the contamination of the food chain, it is appropriate to use wastewater for cut flowers, aromatic grasses, cotton, and high-value forest trees like mahogany, mulberry, and poplar, which could be remunerative alternatives for wastewater use.

Wetland plants, such as *Eichhornia*, *Typha*, and *Phragmites*, have the capacity to remove heavy metals from wastewater. Under laboratory conditions, efficient fungi

and bacteria have been identified for absorption of heavy metals. Some of the georemediation technologies utilize chemical treatment for converting heavy metals into their insoluble minerals, bounding into the alumino-silicate portion of the soil or sediment, and immobilizing in the clay and smectitic lattice. A typical example is binding by specific nanoparticles, e.g., hematite, that have future potential to control the heavy metal pollution from diverse sources.

REFERENCES

1. Jimenez, B.; Asano, T. Acknowledge all approaches: The global outlook on reuse. *Water* **2004**, *21*, 32–37.
2. Raschid-Sally, L.; Carr, R.; Buechler, S. Managing wastewater agriculture to improve livelihoods and environmental quality in poor countries. *Irrig. Drain.* **2005**, *54*, S11–S22.
3. Scott, C.A.; Faruqi, N.I.; Raschid-Sally, L., Eds. Wastewater use in irrigated agriculture: Management challenges in developing countries. In *Wastewater Use in Irrigated Agriculture: Confronting Livelihood and Environmental Realities*; CAB International in Association with IWMI: Colombo, IDRC: Ottawa, 2004.
4. Minhas, P.S.; Samra, J.S. *Waste Water Use in Peri-Urban Agriculture: Impacts and Opportunities*, Bulletin No. 2; CSSRI: Karnal, 2004.
5. McBride, M.B. Toxic metals in sewage sludge-amended soils: Has promotion of beneficial use discounted the risks. *Adv. Environ. Res.* **2003**, *8*, 5–19.
6. Pescod, M.B. *Wastewater Treatment and Use in Agriculture*, FAO Irrigation and Drainage Paper 47; FAO: Rome, 1992.
7. Kabata-Pendias, A. Agricultural problems related to excessive trace metal contents of soil. In *Heavy Metals (Problems and Solution)*; Salomons, W., Forstner, U., Madar, P., Eds.; Springer Verlag: Berlin, 1995; 3–18.
8. United States Environmental Protection Agency. 40 CFR, Part 503B. Fed. Regist. **1993**, *58* (32), 9248–9415.
9. Oliver, M.A. Soil and human health: A review. *Eur. J. Soil Sci.* **1997**, *48*, 573–592.

Urea: Subsoiling and Controlled Release

Tangyuan Ning

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A. and Chinese National Engineering Research Center for Slow/Controlled Release Fertilizers, Shandong Agricultural University, Taian, China

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Zengjia Li

Min Zhang

Chinese National Engineering Research Center for Slow/Controlled Release Fertilizers, Shandong Agricultural University, Taian, China

Abstract

Soil tillage and nitrogen (N) management are important to crop production. No-till (NT) can increase the soil carbon pools but reduce yield in some intensive agricultural regions by subsoil compaction and shallow root development. Subsoiling can reduce these problems caused by NT. Compared with conventional urea, use of controlled release urea (CRU) can significantly improve crop yields and also N use efficiency (NUE) while reducing environmental pollution. But the release of CRU is difficult to control because of numerous factors (e.g., the nature of the coating material, the type of CRU, and the agronomic conditions). The coating materials include sulfur, polymer, superabsorbent/water retention, and biocomposite, among which polymer has large potential. Subsoiling is performed at a 35–45 cm depth keeping a suitable interval between rows before or at the same time of sowing. CRU can be side-dressed or placed at 10 or 15 cm depth. Crops such as maize (*Zea mays*), wheat (*Triticum aestivum*), rice (*Oryza sativa*), barley (*Hordeum vulgare*), and canola (*Brassica napus*) under different conditions should use different controlled release period of CRU. Subsoiling can be used once after 2 to 3 years of NT to reduce the cost. Coupled use of subsoiling and CRU can improve yields by –4.1% to 63.7%. Additional research should focus on the optimal controlled release period of CRU for different crops in the field situation, low-cost and environmentally friendly coating material, and mechanization for using the subsoiling and CRU at the same time with sowing.

INTRODUCTION

Soil tillage and nitrogen (N) management are important to crop production.^[1] Tillage, which can modify soil's physical, mechanical, and biological properties by cutting, mixing, overturning, and loosening, has been an important technological development in the evolution of agriculture.^[2] Even well-graded soils are prone to close packing of soil particles and reduction of total porosity.^[3] Soil tillage was first used and continues to be the most common method to alleviate the soil compaction. However, any benefits of mechanical tillage are reduced by wheel traffic compaction by farm operations in many soil types and climatic regions.^[4] The wheel-induced soil compaction is widely recognized as a constraint to high crop yields when large agricultural vehicles are used for agronomic management.^[4] Thus, no-till (NT), also known as direct drilling and zero tillage, is increasingly being used in the United States, South America, China, and elsewhere,^[2] in which crops are sown

without any prior loosening of the soil by cultivation other than the very shallow disturbance (<5 cm) by the passage of the drill coulters after which usually >30% of the surface remains covered with residues.^[5] NT systems have a positive effect on soil structure, soil organic carbon (SOC) pool, and crop yield for a time. However, some negative effects on soil condition and crop growth have been observed after the long-term use of NT, which exacerbates the risk of subsoil compaction and shallow rooting.^[5] Soils compacted by vehicular traffic or natural processes often benefit from subsoiling, defined as tillage below a depth of 35 cm, by creating macropores that increase rooting and infiltration^[3,6] and enhance cereal production.^[7,8]

The increasing use of N fertilizer can significantly increase crop production. However, using too much N fertilizer could also decrease the N use efficiency (NUE) with a waste of precious resources, economic losses, and adverse impacts on the environment.^[9] Compared with conventional urea, use of controlled release urea (CRU) can

significantly improve crop yields, increase NUE, and reduce environmental pollution.^[10,11]

In general, coupled use of subsoiling and CRU increases the NUE and maize (*Zea mays*) grain yield.^[11,12] Yet, research data on the coupled use of subsoiling and CRU on diverse crops and regions are scanty. Thus, the objective of this entry is to collate, review, and synthesize the available research information on the coupled use of subsoiling and CRU for maize, wheat (*Triticum aestivum*), rice (*Oryza sativa*), barley (*Hordeum vulgare*), and canola (*Brassica napus*) in different countries.

SUBSOILING

Application of Subsoiling

Subsoiling is performed at a 35–45 cm depth keeping a suitable interval between rows before or at the time of

sowing (Fig. 1). To increase the SOC content, crop residues should be returned to the field prior to subsoiling. For some crops, such as maize, subsoiling can also be performed during early growth stages. In this scenario, the interval between rows should be the same as row interval of crops.

Reduce Cost of Subsoiling

In conservation systems, subsoiling is often conducted only in the row and referred to as in-row subsoiling or strip tillage. The strategy is to minimize surface disturbance and reduce the subsoiling cost.^[3] Despite the high cost of in-row subsoiling, significant gains in crop yield may be adequate to offset additional costs. There exists a range of options to reduce the cost of subsoiling, primarily through reductions in energy costs through fuel consumption, including performance of farm operations at field soil moisture, using subsoiler shanks for the designed depth, reducing frequency of subsoiling, and controlling vehicular traffic.^[3] In general, subsoiling is required once after 2 to 3 years of NT.^[13,14]

CONTROLLED RELEASE UREA

Choose Suitable CRU

An ideal CRU is coated with a natural or seminatural, environmentally friendly macromolecule material that retards urea release to such a slow rate that a single application to the soil can meet nutrient requirements for the crop growth. Thus, CRU enhances NUE and increases crop yields.^[15] The multistage diffusion model indicates that

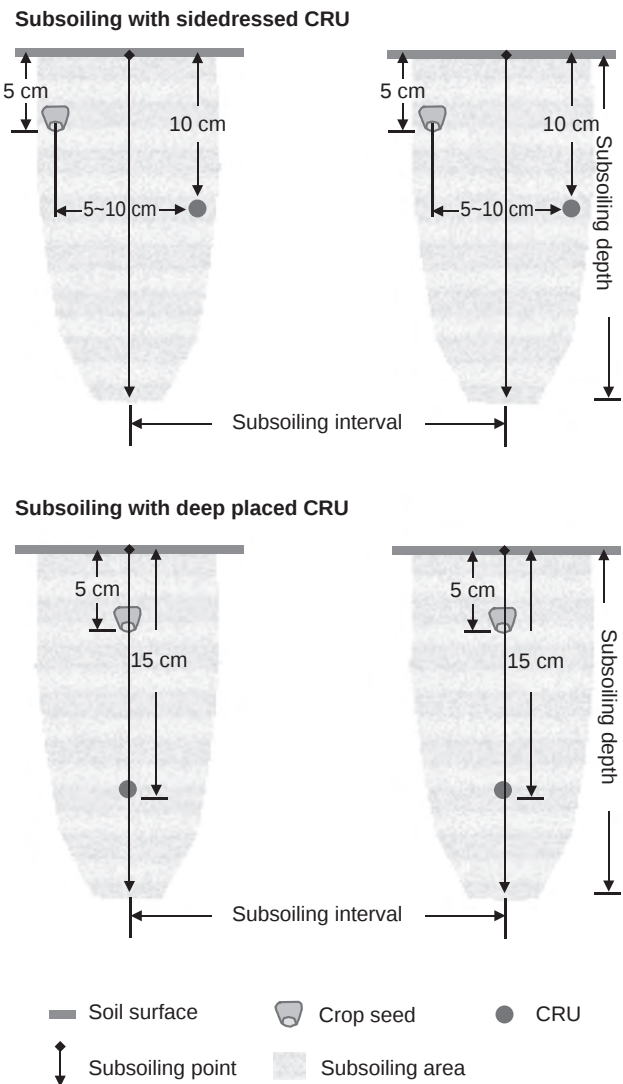


Fig. 1 Coupled use of subsoiling and CRU.

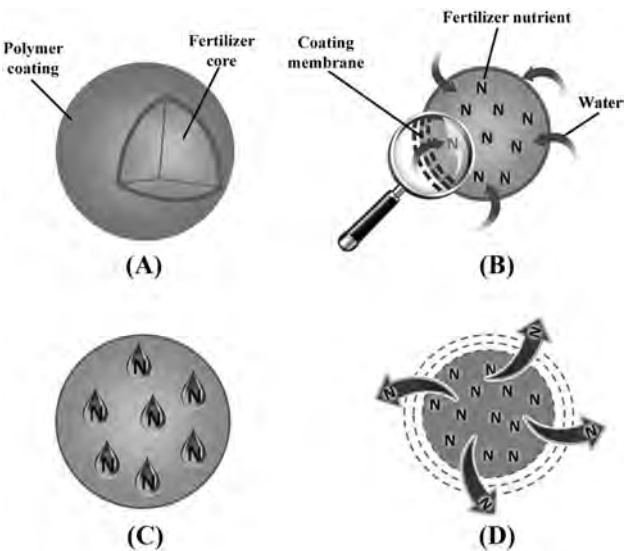


Fig. 2 Diffusion mechanism of controlled release: (A) Fertilizer core with polymer coating, (B) water penetrates into the coating and core granule, (C) fertilizer dissolution and osmotic pressure development, (D) controlled release of nutrient through swollen coating membrane.

Source: Adapted from Azeem, KuShaari, et al.^[15] ©2014 Elsevier.

Table 1 Use quantities and methods of CRU for crops.

Crop type	Region	Longitude and latitude	Soil type	Controlled release period	Used quantity	Used method	Yield increase	N use efficiency	References
Maize	China	36°10'N, 117°9'E	Cambisol	60–90 days	0.5–0.75 × commonly used	Banded at 10–15 cm soil depth	8~10%	Increase	[9,10,12]
Rice	India	23°59'N, 82°25'E	Sandyloam	100 days	105 kg N ha ⁻¹	Band application at about 2–5 cm	7.1~63.7%	Increase apparent N recovery fraction	[16]
Rice	China	36°09'N, 117°09'E	–	120–150 days	200 kg N ha ⁻¹	Placing with seeds	3~5.9%	Use less N fertilizers for greater rice grain yield	[17]
Winter wheat	China	34°55'N, 118°20'E	Calci-aquic Vertisols	90 days with 180 days	150 kg N ha ⁻¹	A mixture of 180 days and 120 release period of CRUs at the ratio of 3:7	6.5%	Reduced N fertilizer by one-third	[18]
Winter wheat	Canada	–	Clay loam	40 days	120 kg N ha ⁻¹	Seed-placed	Similar	Spring broadcast application of CRU was less effective than conventional urea	[19]
Barley, wheat, canola	Canada	–	The locations covered a wide range of soil types and panned a distance of 1600 km from east to west	60–90 days	Commonly used	Banded beneath the soil surface/broadcast	–4.1% to 22%	–	[20,21]

Note: –, not mentioned.

soil/irrigation water penetrates the coating and condenses on the solid fertilizer core followed by a partial nutrient dissolution (Fig. 2). The effects of CRU depend on numerous factors such as the nature of the coating materials, CRU type, and agronomic and edaphic conditions.^[15] Commonly used coating materials include sulfur, polymer, superabsorbent/water retention, and biocomposite.^[15] Among the newly developed commercial fertilizers, polymer-coated urea (PCU) is promising. In comparison with the large amount of conventional N fertilizers used in agriculture worldwide, total use of PCU is rather insignificant. The main obstacle to the wider use of PCU is the high cost associated with the complex manufacturing process and the expensive coating materials.^[22] Using the recyclable styrofoam waste as the coating material is economic and environmentally benign. Thus, it is important to choose a suitable CRU or PCU with consideration of the cost and release stage under the specific condition.

There are also differences between CRU and slow release urea (SRU), because the pattern of nutrient release of SRU is difficult to predict and remains subject to changes in soil type and climatic conditions. In contrast, the pattern, quantity, and time of release of CRU can be predicted, within some limits. Thus, CRU has some advantages compared with SRU.

Application of CRU

CRU can be side-dressed or applied at 10 or 15 cm depth after or at the time of subsoiling (Fig. 1). Further, CRU is most effective when placed at 15 cm depth. However, the growth rate of crop is slower during the seeding stage of crops when used with CRU because of its lower release compared with that of the conventional urea. Nonetheless, a higher growth rate is observed at a later stage.^[10] Split application of short controlled release period CRU, half at the sowing time and half side-dressed at tillering, can increase NUE,^[23] but also increases the energy input associated with split application. Thus, a blended application of 30–50% conventional urea and 70–50% CRU, and also a blended application of CRU with short and long controlled release periods, can be applied to improve crop growth for the entire growth period.^[20] By so doing, different crops under diverse conditions use different controlled release periods of CRU to enhance NUE (Table 1). Most importantly, the release in the field is >80% before the peak crop N uptake. For maize in North China, the favorable controlled release period of CRU is 60–90 days after seeding.^[9,10,12] For rice in India, the favorable controlled release period of CRU is 100 days but better for 120–150 days in China.^[16,17] In China, a mixture of 180- and 120-day release period of CRU at the ratio of 3:7 is benefit for winter wheat.^[18] In Canada, a controlled release period of 40–60 days is suitable for winter wheat, and that of 60–90 days is adequate for barley and canola.^[19–21] In general, using CRU can increase the yield and NUE for equivalent or lower rate of N application.

RESEARCH NEEDS AND FUTURE DIRECTIONS

Additional research is needed to identify an optimal controlled release period of CRU in the field situation for different crops under diverse soil and weather conditions. Mechanization of subsoiling and applying CRU at the time of sowing is very important for farmers' acceptance of the coupled use of subsoiling and CRU. Identifying economic and environmentally friendly coating materials for CRU is a high researchable priority.

CONCLUSIONS

There are positive coupled effects of subsoiling and CRU on crop growth, NUE, and environment. Such effects can increase the water and NUE and improve agronomic yields by –4% to 64% (Table 1). Thus, the coupled use of subsoiling and CRU is an important strategy in regions with limited resources of water and fertilizer, or in regions with the need to improve the water and N use efficiencies, which improves agronomic production.

ACKNOWLEDGMENT

Financial supports are gratefully acknowledged for the Special Research Funding for Public Benefit Industries (Agriculture) of China (201103001) and the Nature Science Fund of China (30900876).

REFERENCES

1. Lal, R. Agricultural activities and the global carbon cycle. *Nutr. Cycl. Agroecosys.* **2004**, *70* (2), 103–116.
2. Zhang, H.L.; Lal, R.; Zhao, X.; Xue, J.; Chen, F. Opportunities and challenges of soil carbon sequestration by conservation agriculture in China. *Adv. Agron.* **2014**, *124*, 1–36.
3. Raper, R.L.; Bergtold, J.S. In-row subsoiling: A review and suggestions for reducing cost of this conservation tillage operation. *Appl. Eng. Agric.* **2007**, *23* (4), 463–471.
4. Nevens, F.; Reheul, D. The consequences of wheel-induced soil compaction and subsoiling for silage maize on a sandy loam soil in Belgium. *Soil Till. Res.* **2003**, *70* (2), 175–184.
5. Soane, B.D.; Ball, B.C.; Arvidsson, J.; Basch, G.; Moreno, F.; Roger-Estrade, J. No-till in northern, western and south-western Europe: A review of problems and opportunities for crop production and the environment. *Soil Till. Res.* **2012**, *118*, 66–87.
6. Pikul, J.L.; Aase, J.K. Water infiltration and storage affected by subsoiling and subsequent tillage. *Soil Sci. Soc. Am. J.* **2003**, *67* (3), 859–866.
7. Martinez, I.G.; Prat, C.; Ovalle, C.; del Pozo, A.; Stolpe, N.; Zagal, E. Subsoiling improves conservation tillage in cereal production of severely degraded Alfisols under Mediterranean climate. *Geoderma* **2012**, *189–190*, 10–17.
8. Izumi, Y.; Yoshida, T.; Iijima, M. Effects of subsoiling to the non-tilled field of wheat-soybean rotation on the root system

- development, water uptake, and yield. *Plant Prod. Sci.* **2009**, *12* (3), 327–335.
9. Ning, T.Y.; Shao, G.Q.; Li, Z.J.; Han, H.F.; Hu, H.G.; Wang, Y.; Tian, S.Z.; Chi, S.Y. Effects of urea types and irrigation on crop uptake, soil residual, and loss of nitrogen in maize field on the North China Plain. *Plant Soil Environ.* **2012**, *58* (1), 1–8.
 10. Shao, G.; Li, Z.; Ning, T.; Zheng, Y. Responses of photosynthesis, chlorophyll fluorescence, and grain yield of maize to controlled-release urea and irrigation after anthesis. *J. Plant Nutr. Soil Sc.* **2013**, *176* (4), 595–602.
 11. Qin, S.; Zhang, Z.; Ning, T.; Ren, S.; Su, L.; Li, Z. Absciscic acid and aldehyde oxidase activity in maize ear leaf and grain relative to post-flowering photosynthetic capacity and grain-filling rate under different water/nitrogen treatments. *Plant Physiol. Bioch.* **2013**, *70*, 69–80.
 12. Hu, H.; Ning, T.; Li, Z.; Han, H.; Zhang, Z.; Qin, S.; Zheng, Y. Coupling effects of urea types and subsoiling on nitrogen-water use and yield of different varieties of maize in northern China. *Field Crop Res.* **2013**, *142*, 85–94.
 13. Raper, R.L.; Schwab, E.B.; Balkcom, K.S.; Burmester, C.H.; Reeves, D.W. Effect of annual, biennial, and triennial in-row subsoiling on soil compaction and cotton yield in Southeastern US silt loam soils. *Appl. Eng. Agric.* **2005**, *21* (3), 337–343.
 14. Tian, S.; Ning, T.; Zhao, H.; Wang, B.; Li, N.; Han, H.; Li, Z.; Chi, S. Response of CH₄ and N₂O emissions and wheat yields to tillage method changes in the North China plain. *Plos One* **2012**, *7* (12), e51206.
 15. Azeem, B.; KuShaari, K.; Man, Z.B.; Basit, A.; Thanh, T.H. Review on materials & methods to produce controlled release coated urea fertilizer. *J. Control. Release* **2014**, *181*, 11–21.
 16. Kondo, M.; Singh, C.V.; Agbisit, R.; Murty, M.V.R. Yield response to urea and controlled-release urea as affected by water supply in tropical upland rice. *J. Plant Nutr.* **2005**, *28* (2), 201–219.
 17. Yang, Y.; Zhang, M.; Li, Y.C.; Fan, X.; Geng, Y. Controlled release urea improved nitrogen use efficiency, activities of leaf enzymes, and rice yield. *Soil Sci. Soc. Am. J.* **2012**, *76* (6), 2307–2310.
 18. Yang, Y.-C.; Zhang, M.; Zheng, L.; Cheng, D.-D.; Liu, M.; Geng, Y.-Q. Controlled release urea improved nitrogen use efficiency, yield, and quality of wheat. *Agron. J.* **2011**, *103* (2), 479–485.
 19. McKenzie, R.H.; Bremer, E.; Middleton, A.B.; Pfiffner, P.G.; Dowbenko, R.E. Controlled-release urea for winter wheat in southern Alberta. *Can. J. Soil Sci.* **2007**, *87* (1), 85–91.
 20. Grant, C.A.; Wu, R.; Selles, F.; Harker, K.N.; Clayton, G.W.; Bittman, S.; Zebbarth, J.; Lupwayi, Z. Crop yield and nitrogen concentration with controlled release urea and split applications of nitrogen as compared to non-coated urea applied at seeding. *Field Crop Res.* **2012**, *127*, 170–180.
 21. Khakbazan, M.; Grant, C.A.; Finlay, G.; Wu, R.; Malhi, S.S.; Selles, F.; Clayton, G.W.; Lupwayi, N.Z.; Soon, Y.K.; Harker, K.N. An economic study of controlled release urea and split applications of nitrogen as compared with non-coated urea under conventional and reduced tillage management. *Can. J. Plant Sci.* **2013**, *93* (3), 523–534.
 22. Yang, Y.C.; Zhang, M.; Li, Y.; Fan, X.H.; Geng, Y.Q. Improving the quality of polymer-coated urea with recycled plastic, proper additives, and large tablets. *J. Agr. Food Chem.* **2012**, *60* (45), 11229–11237.
 23. Malhi, S.S.; Soon, Y.K.; Grant, C.A.; Lemke, R.; Lupwayi, N. Influence of controlled-release urea on seed yield and N concentration, and N use efficiency of small grain crops grown on Dark Gray Luvisols. *Can. J. Soil Sci.* **2010**, *90* (2), 363–372.

Value of Soil to Humans

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

The soil plays a major role in nearly all human activities. Plants growing in the soil provide most of our food, material for clothing and building, and medicinal and industrial chemicals. The soil is a storehouse for many chemicals. Carbon is stored in the soil from decaying plant residues; both cations and anions (organic and inorganic) are held on exchange sites within the soil and held from leaching or releasing to the atmosphere. The soil is also the home for millions of microorganisms, which degrade organic wastes, and chemicals such as pesticides.

INTRODUCTION

The soil is the foundation upon which buildings, roads, and other structures are formed. The soil provides recreational sites for playing fields, golf courses, outdoor camping and picnics, wildlife habitat, and biological diversity.

The soil is inextricably linked to water and air. Water running off the soil or percolating into aquifers is used by animals and is cleansed and made fit for direct human consumption. It is also used for navigation and recreation. Water remaining in the soil can later provide water for plant use. The soil interacts with water to influence the atmosphere. Carbon from decaying plant residues is stored in the soil and thus does not contribute to atmospheric buildup of carbon dioxide (CO₂). The soil (and the water therein) interacts with atmospheric radiation and mitigates surface temperatures.

THE SOIL AS A PRODUCER OF FOOD AND FIBER

Approximately 11% of the land area, or 1461 million hectares of the world, are cultivated. In the United States, nearly 186 million hectares are cultivated.^[1] Crop and rangeland produce most of the food and fiber for the nation's human needs and for export. The United States' agricultural exports have exceeded \$50 billion.

The quality of the soil and landscape varies greatly. Nearly 19.8 million hectares of cropland are irrigated, mostly in the western states. Soil and landscape characteristics interact with weather to create a number of hazards for continued use of the land without serious degradation. Of all nature's hazards, erosion from wind and water is probably the most serious, although salinity, compaction,

acidity, and nutrient and organic matter depletion are also important.

Of the crops produced in the United States, those with the largest area include corn (29.6 million hectares), wheat (25.4 million hectares), soybeans (25.7 million hectares), hay (24.7 million hectares), sorghum (4.8 million hectares), cotton (5.2 million hectares), barley (2.7 million hectares), and oats (1.1 million hectares).^[1] Grain is used for direct human consumption, as feed for animals, and for the production of alcohol, sugar, and industrial chemicals. Cotton and other fiber crops are used for the making of cloth, and some seed crops are used for oil extraction and as feed for livestock. Many other crops are grown, including sugar beets, sugarcane, potatoes, canola, vegetables, fruits, nuts, and other products, for direct human consumption, animal feed, and specialty uses.

In the Western United States, about 239 million hectares of land is used for rangeland and pastures,^[1] producing much of the nation's beef, mutton, and dairy products. As with cropland, rangeland can be damaged from overgrazing, erosion, and other causes. Some of the rangeland are interspersed with forestland and cropland.

THE SOIL AS A PRODUCER OF FORESTS

The United States has almost 262 million hectares of forests,^[1] which are owned by the federal, state, and local governments and by private individuals and institutions. These forests are a source of wood and chemicals and are used for recreation and wildlife habitat.

Worldwide forests cover much of the landscape from the tropical rainforests to the boreal forests of the north and south hemispheres. Harvest and clearing procedures have been decreasing forestland in the tropics as well as in the United States.

Forests contribute a number of important functions for humans in addition to the production of wood and industrial products. Forests are a large sink for carbon, which is tied up in the vegetation and in the forest litter. Forests mitigate water balance, slowing runoff and releasing water slowly to surface and groundwaters. In mountainous areas, the forests store snow in the winter, and the melt water is slowly released to streams and groundwater in the summer.

THE SOIL AND ITS INTERACTION WITH WATER

Nearly, all atmospheric precipitation falls on the soil. As it strikes the surface, it either runs off the land and enters surface water bodies (lakes, streams, and wetlands) or enters the soil. If it enters the soil, it may be stored for later use by plants or may continue downward to enter groundwater. Thus, to a great extent, the condition of the soil (particularly at the surface) determines the ultimate disposition of water. Water moving across soil often carries with it eroded soil particles that may damage the soil for plant growth and may cause off-site damage leading to undesirable consequences. Sediment may be deposited in water supplies used for human consumption, or in road ditches, reservoirs, lakes, and other locations.

In 1992, the 48 contiguous states of the United States had approximately 50 million hectares of wetlands, down from 89 million hectares in 1780.^[1]

The soil is also an excellent sorbent for chemicals. Exchange sites for both cations and anions can remove chemicals from water as it percolates downward. After moving through the mantle of soil and underlying earth, water is usually purified, making it useable for human consumption. However, the capacity of soils to sorb chemicals varies greatly. Soils high in clay and/or organic matter usually have a greater sorbing capacity than sandy soils. Likewise, chemicals differ in their ability to attach to soil particles.

In the United States, as well as many other parts of the world, water supplies have sometimes been contaminated with excessive amounts of chemicals. Nitrates, phosphorus, and pesticides have been of most concern although chemicals from industry and human wastes are also of concern.

THE SOIL AND ITS INTERACTION WITH THE ATMOSPHERE

The soil has an important bearing on atmospheric conditions, particularly the atmosphere close to the ground. The color of the surface soil affects the balance between radiation sorbed by the soil or reflected back into the atmosphere. The water content of the soil determines how much of the radiant energy is used for heating the soil and air, or evaporating water from the soil.

In addition, the soil is a sink for carbon, nitrogen, and other gaseous elements that may be converted to volatile forms and released into the atmosphere. Carbon in the form of plant residues is continually added to the soil. Whether it remains in the soil as organic matter or is decomposed to CO₂ and released depends on the characteristics of the soil and its management. When the native soil was first brought into cultivation, the organic matter within the soil was usually drastically reduced. With better management, the organic matter is in many cases being increased. This has important global implications on the reduction of atmospheric CO₂ and lessening possible climate change.^[2]

THE SOIL AS A BASE FOR BUILDINGS, LAWNS, AND GARDENS

Many different types of structures are built on soils and the underlying materials. The different types of structures included are as follows: 1) shallow excavations consisting of trenches and holes usually dug to a maximum of 1–2 m; 2) dwellings and small commercial buildings; 3) roads and streets; and 4) lawns and landscaping. In addition, several types of sanitary facilities are built in the soil including: 1) septic tank absorption fields; 2) sewage lagoons; 3) trench sanitary landfills; 4) area sanitary landfills; and 5) daily cover for landfills. In 1992, the United States had nearly 21 million hectares of urban and built-up land.

Soil properties and site features determine, to a large extent, the performance of these structures. Type and content of clay, mineralogy of the silt and sand fraction, the kinds of adsorbed cations, erodibility, permeability, corrosivity, shrink–swell potential, available water capacity, and other behavioral characteristics affect engineering uses. Important landscape characteristics include depth to bedrock, soil wetness, depth to a seasonal high water table, slope, likelihood of flooding, natural soil structure, and soil density.

Of the 21 billion hectares of urban and built-up land, gardens and lawns comprise a considerable share. This land is usually highly managed for the production of fruits, vegetables, lawns, and ornamentals.

THE SOIL FOR USE IN RECREATION AND SPORTS

The soil is the material upon which most outdoor sports and recreational facilities are built. The facilities are as follows: 1) fields upon which a wide variety of sports are played (e.g., baseball, football, and soccer); 2) picnic grounds and hiking trails; 3) camping sites; 4) golf courses; and 5) parks and open spaces. The United States has 16,743 golf courses occupying almost 1 million

hectares of land. Several million hectares are probably devoted to picnic and local park areas.

Each of the recreation and sport uses requires either natural or engineered soil and/or landscape conditions. Sport fields usually require good drainage and the ability of the soil to support a grass cover. The soil is often graded to achieve the desired landscape features.

THE SOIL AS A MEDIUM FOR DIVERSITY

The soil is a medium in which a multitude of flora and fauna grow.^[3] Flora range from simple microorganisms to higher herbaceous plants, such as agricultural plants and trees. There are millions of microorganisms harbored in each gram of soil. Bacteria, actinomycetes, fungi, and algae are included. Many fauna (protozoa, worms, and arthropods) live in the soil, including higher animals such as mice, gophers, ground squirrels, and prairie dogs. Birds, such as the burrowing owl, access the soil through holes made by animals. Soil insects are an important source of food for many birds (e.g., worms for robins).

Because the soil is home to so many species, it is a repository for a great deal of genetic material. This material is a valuable resource that needs to be preserved. Under agricultural and highly managed forests, the genetic diversity of species is often limited and changed.

THE SOIL AS AN ARCHIVE OF NATURAL AND HUMAN HISTORY

Soil science can contribute to the understanding of natural and human history and thus provide valuable lessons for

management of natural resources. The detection of buried organic-rich horizons (Paleosols) and carbon dating of these horizons in relation to other horizons, as well as other chemical and physical properties, geomorphic, and landscape studies, are methods soil scientists use to reveal the past. Lowdermilk^[4] documents the deterioration of land through erosion, siltation, salinity, and nutrient depletion in the Middle East region. Lowdermilk estimated that at least 11 empires have risen and fallen in the 7000-year history of the Tigris and Euphrates Valleys. While the area supported large populations in the past, it depends on oil exports to sustain its people. The decline of the Mayan Empire in Central America more than 1000 years ago has been related to soil deterioration from poor management.^[5] Other examples of the decline of civilization as a result of soil deterioration are in Turkey and the Negev Desert in Israel.

REFERENCES

1. USDA-ERS. *Agricultural Resources and Environmental Indicators, 1996–1997*; . Agricultural Handbook Number 712; U.S. Department of Agriculture, Economic Research Service: Washington, 1997.
2. Lal, R.; Kimble, J.M.; Follett, R.F.; Cole, C.V. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*; Sleeping Bear Press: Chelsea, 1998.
3. Russell, E.W. *Soil Conditions and Plant Growth*, 10th Ed.; Longman Group Limited: London, 1973.
4. Lowdermilk, W.C. *Conquest of the Land Through Seven Thousand Years*; Agriculture Information Bulletin 99; U.S. Department of Agriculture—SCS: Washington, 1953.
5. Olson, G.W. Archaeology: Lessons on future soil use. *J. Soil Water Conserv.* **1981**, *36*, 261–264.

Variability

Josef H. Görres

Department of Plant and Soil Science, University of Vermont, Burlington, Vermont, U.S.A.

Abstract

Spatial variability occurs at all scales—from micrometer scale (1–10 μm) relevant to microbial processes to continental scale (1000 km) relevant to the distribution of soil orders and climate change. The order of magnitude of the coefficient of variation of soil properties ranges from a few percent to greater than 100%. Jenny's soil-forming factors, such as climate, topography, organisms, parent material, and time, represent the fabric of the natural variability of soil that is modified by anthropogenic properties resulting in a complex distribution of soil characteristics. Very small-scale soil property variations may affect biological and chemical processes and may even explain soil ecological puzzles such as the high level of functional redundancy and diversity of soil biota. Spatial and temporal variations in soil properties may affect the spatial distribution of important soil processes. The spatial analysis of these variations is often done using semivariance analysis and mapping with kriging, an unbiased, optimal interpolation technique developed for estimating ore deposits from limited field data. Variable rate agriculture, in which Global Positioning System technology is used to map crop yield, uses spatial analysis to predict spatially explicit fertilizer applications to improve yield and lower environmental impacts. Spatial analysis is also used in environmental management, environmental modeling, and a multitude of research investigations. All applications need to consider the patterns and scale of variations in order to obtain accurate estimates of the variability of soil properties and processes.

INTRODUCTION

All soil properties, whether they be physical, biological, or chemical in nature, show considerable multiscale variability, ranging from microbial scales (1–1000 μm) to regional (10–100 km) and even continental scales (thousands of kilometers). Variability at all scales is high. The coefficients of variation (CVs—ratio of the standard deviation and mean) for individual properties may range from several percent to several hundred percent. Warrick^[1] classified variations of physical soil properties with CVs less than 15%, between 15% and 50%, and greater than 50% as low, medium, and high, respectively. Bulk density and porosity were classed as low, whereas electrical conductivity, hydraulic conductivity, and solute concentrations were classified as high variability. For biological properties, similar ranges are reported in the literature; e.g., CVs for phosphatase activity were reported as 39%.^[2] For forest soils, the CV for nematode abundance was of the order of 100% as compared to moisture and organic matter with CVs of 10–40%.^[3] Where earthworms are present, the CV of water percolation may vary from 10% to 100%.^[4,5] To make matters more complex, some soil properties and their variability change seasonally^[3] or indeed on a daily basis.

Soil scientists engaged in soil mapping, classification, and morphology may have to integrate over several scales to identify soil taxa. Soil taxonomy recognizes that translocations and translocations of soil materials connect soil

horizons in depth. Evaluating the 3-D variation of soil properties in combination at the pedon scale (meter scale) may thus produce thousands of unique profiles, each representing a soil series, the finest taxonomic resolution in soil classification systems and the pedology analog of species in Linnaean taxonomy.

Both natural processes and human activity contribute to the spatial variability encountered in soils. Jenny's equation of soil formation is the most commonly used model of pedogenesis that attempts to understand differences among soils in terms of five spatially variable factors. These five factors are responsible for landscape to continental-scale variations in soil properties.^[6] The mosaic of different soil series created by the superposition of Jenny's factors may be regarded as the fabric of soil variability which is further modified by land management.

Soil biological processes create very much smaller, microscale variations in properties other than those considered in soil classification. These variations can be tracked to particular origins;^[7] e.g., it has long been known that soil biota are more numerous in soil surrounding roots^[8] and burrows of soil animals^[9] than in the bulk soil. Microscale heterogeneity may cause the spatial separation of key nutrient cycling transformations that may drive many below-ground ecological processes.^[10,11]

Any record of the spatial distribution of properties is a snapshot in time. Soil processes are strongly dependent on meteorological and climatic forces and thus respond to temporal variations in temperature, precipitation, wind

speed, insolation, and relative humidity. Denitrification, the conversion of nitrate into a suite of nitrogen (N) gases, e.g., depends on the distribution of carbon at the microscale. However, denitrification is mainly an anaerobic process that is strongest under saturated conditions. Large rainfall events thus cause an increase in denitrification activity. Nitrification, on the other hand, is an aerobic process and thus occurs when soils are drying. As biochemical processes often occur at microsites, the diffusion of biochemical reaction products further links space and time.^[11]

Soil variability has practical consequences for experimental designs, farming, and the implementation and design of best management practices; e.g., precision farming employs variable rate technologies that strive to apply agrochemicals to soils based on georeferenced spatial variations of field fertility parameters. This may save farm resources and improve the environmental quality by reducing inputs of fertilizers and pesticides. Yet, environmental and economic impacts of variable rate agriculture may depend on many factors and it is not clear whether this technology always leads to a reduction in application rate on the field scale.^[12] In environmental application, land managers and soil evaluators often face questions associated with landscape-scale variations, such as how site-specific soil evaluations need to be when determining soil suitability for land use projects.

What all applications that account for the spatial variability of soils have in common is that the scale of variation ought to be a focus. The scale germane to an application will have to be isolated to design an appropriate sampling strategy and determine the volume (sample support) of an individual soil sample.^[13] Soil variability represents interesting challenges to soil scientists, making knowledge of spatial distribution of soil properties and processes paramount to the practice in both fundamental and applied pedology.

FACETS OF SPATIAL VARIABILITY IN SOILS

Soils, Soil Properties, Soil Processes, and Time

When discussing the spatial variability of soils, a distinction has to be made between the variability of soils, i.e., among taxonomic units, and soil properties, a measurable quantity. Pedologists classify terrestrial and subaqueous soils based on the vertical distribution of matter and its properties within a representative soil volume called a pedon.^[14] Each soil series thus has a unique sequence of horizons with specific diagnostic properties that are intertwined with spatially and temporally variable processes within and outside of the soil. Soil survey maps are quintessentially 2-D as the nature of a soil series is given by the relationship and interpretation of properties in the vertical as well as the horizontal dimensions of its profile. Soil classification relies on the spatial, 3-D arrangement of measurable and relatively stable properties within a soil profile.^[14]

However, stability and equilibrium are not one of the hallmarks of many soil properties. Spatial distributions change over time. Daily variations in soil moisture and temperature may engender rapid changes in food web dynamics and biochemical transformation rates responsible for much of the belowground decomposition and nutrient cycling. Variations also occur at seasonal and interannual scales. Even longer-term environmental variations relevant to climate and vegetation affect soil development in a way that may have tangible effects on soil loss.^[15] Among these decade-long cycles, space weather, in particular the 22-year cycles of the solar magnetosphere and the concomitant variation in cosmic ray fluxes, may be responsible for droughts such as the Dustbowl episode in the Southern United States in 1930.^[16] While devastating for farmers, these drought episodes are responsible for a spatial redistribution of top soil materials. However, even at a single point in time, snapshots of soils present observers with sufficiently puzzling spatial variability.

The importance of the dependence of variability on time is best illustrated by how changes in soil properties may affect the variability of soil processes. It is well known that small differences in soil moisture content may translate into large differences in soil processes. The affected processes are physicochemical and biological in nature. Moisture saturation at the center of aggregates in an otherwise dry soil may result in considerable denitrification rates.^[17] Also, only small differences in moisture or the presence and location of macropores in soils may result in large differences in water fluxes through the soil.

Factors of Soil Formation: Intrinsic Sources of Spatial Variability

Early soil scientists were intrigued by the differences in soils that they encountered in their travels. Dukochaev (1846–1903), Hilgard (1833–1916), and Jenny (1899–1992) formulated theories in which independent factors could explain geographic variations in soils.^[18] These factors are intrinsic or natural sources of variability. In an attempt to create a more quantitative foundation for soil science, Jenny's equation^[6] expressed the magnitude of soil-forming processes in terms of five variables or factors, namely: climate, parent material, topography, organisms, and time.

$$\text{Process} = F(\text{climate, topography, parent material, organisms, time}) \quad (1)$$

The five factors act on all soil-forming processes with each factor imbuing different qualities to the soil. Overlaying these qualities across the landscape differentiates among soil series, the spatial variability considered by soil taxonomy. At the continental scale, large-scale variations in climate, vegetation, and parent material give rise to characteristic patterns of soil orders, the coarsest soil

Global Soil Regions

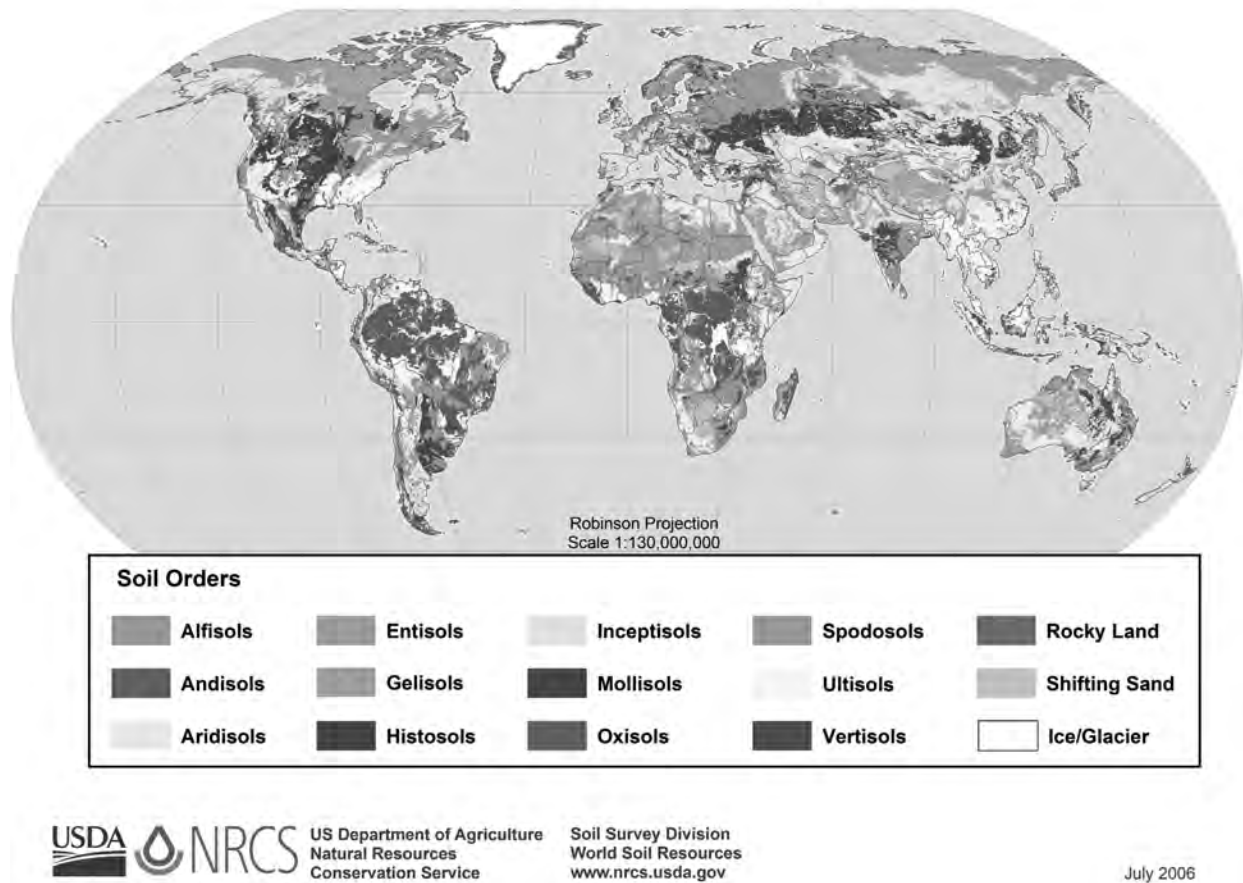


Fig. 1 The planetary distribution of soil orders across the land area. Distinct geographic patterns of soil orders emerge based on climate, topography, organisms, parent material, and time.

Source: Reproduced with permission from Dr. Paul F. Reich, Geographer–World Soil Resources, USDA Natural Resources Conservation Service.

taxonomic scale (Fig. 1); e.g., most Spodosols are found in northern, temperate climates with boreal, coniferous forests, where precipitation is high and glaciation has left deposits of sands, leading to distinct patterns of reduced and oxidized horizon environments (Fig. 2A). In contrast, Inceptisols have not developed very distinctive horizons due to a lack of time that the other factors have acted upon them. In Fig. 2B, an agricultural Inceptisol is shown with a horizon altered by plough action. This young soil (only thousands of years old) developed after the Laurentide ice sheet of the last glaciation melted and receded. In this soil, the combination of two parent materials, a fine silt mantle overlying glacial sand and gravel deposits, results in characteristic reduction oxidation horizons that dominate the spatial variability in the depth of the soil. The fine silt material may retain water long enough above the gravel deposit to reduce iron oxides, resulting in a horizon where the distinctive orange color of iron oxides is absent.

The five factors do not act independently, and Jenny's equation can thus not be understood as a linear algebraic

equation. Toposequences are a good example of the interrelationship between soil-forming factors at the landscape scale. A special toposequence called a hydrosequence is shown in Fig. 3. The landscape position along this hydrosequence not only determines elevation and, thus, the distance to ground water and the drainage class of the soil, but also determines vegetation, decomposition, and production rates of the ecosystems as the land surface slopes from an upland area into a wetland. Together, these factors are responsible for the variation of soils along the toposequence.

Action of *Homo sapiens*: Extrinsic Factors of Variability

Superimposed on the patchwork of intrinsic variation are extrinsic spatial patterns introduced by land management. At the landscape scale, the mosaic of agricultural fields (Fig. 4) may impose an additional variability of soils. In addition, within each field, management such as tillage and irrigation may introduce further variability at the field scale.

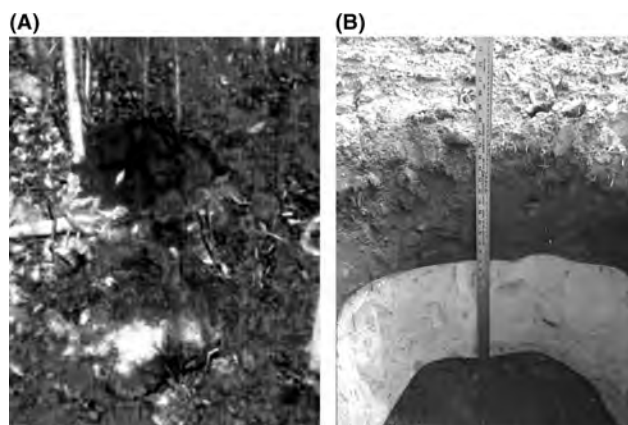


Fig. 2 (A) Spodosol at Hubbard Brook, New Hampshire. Distinct horization with a “bleached” E horizon above a characteristic dark area Bhs horizon, indicating the translocation of iron and humus from the E and O horizons. Provided by Dr. Joel Tilley of the Agricultural and Environmental Testing Lab at the University of Vermont. (B) Inceptisol at the Peckham Farm, Kingston, Rhode Island. A silt cap deposited over coarse sand gravel deposits created conditions that favor reduction of iron oxides within the silt horizon resulting in a white, colorless layer and a streak of white iron oxides at the contact between the silt and the sand and gravel.

Notably, extrinsic patterns may persist for many years even when the practice causing the initial variation is abandoned; e.g., ridge and furrow tillage in the 19th century may still have affected the crop yield pattern in 1911 at Rothamsted.^[19]

Apart from the desired patterns of fertilizer application and planting, the use of machinery causes compaction that

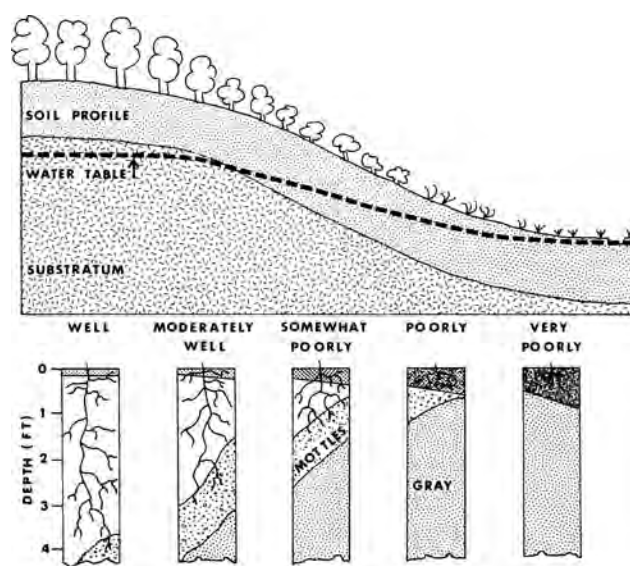


Fig. 3 Soil profiles along a toposequence vary by elevation or more importantly with the distance to the water table.

Source: From Rhode Island Agricultural Research Station 1988, Wright and Sautter, *Soils of the RI Landscape*, reproduced with permission of the Director, Rhode Island Agricultural Research Station.



Fig. 4 Landscape-scale field planting pattern, an extrinsic variation contributing to the spatial variability of soils.

can also persist for many years.^[20] Likewise, the hoof fall of large grazing animals may affect soil compaction in pastures with characteristic spatial variations that depend on grazing pattern, paddock geography, and animal stocking rate. Dung deposition patterns in the pasture add to the variability of pasture soils^[21] and render the distribution of nutrient cycling processes uncertain.

These examples suggest that land management creates additional variability at the field scale. However, it can also be argued that land conversions may make soils more homogeneous; e.g., the conversion from a heterogeneous land use mosaic to large fields that support monocultures such as corn or sunflowers would smooth variations at the landscape scale while increasing heterogeneity at the field scale.

Variation at the Microscale

The patterns of soil heterogeneity observed by soil surveys, i.e., those that are caused by soil-forming factors, essentially integrate soil properties over the pedon that has a volume of several meters cubed. Yet, the diversity of processes at much smaller scale is apparent even when observing differences at the soil surface. The pattern of vegetation may reveal small-scale variations in soil properties, a fact that is used in plant indicator systems.^[22] Equally obvious at the soil surface are the patterns of excavation mounds of burrowing animals like ants and earthworms, which may coexist within centimeters of each other but have quite divergent microbial communities.^[23]

Very small-scale spatial variability is also regarded as the key to better understanding why soil faunal communities maintain high levels of diversity with much apparent functional redundancy.^[24] Pore networks comprise a diversity of pore neck and chamber diameters that allow access

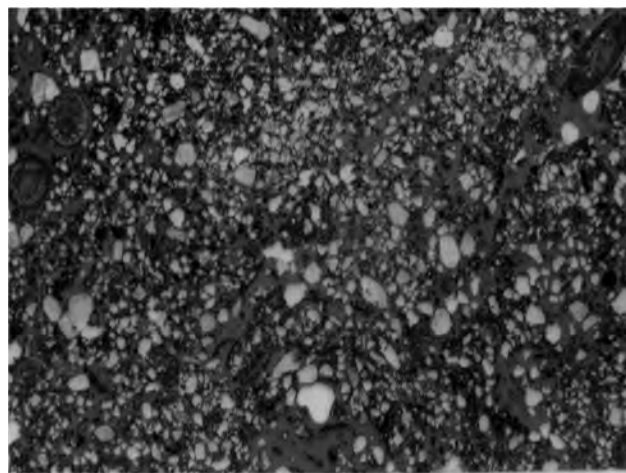


Fig. 5 Micrograph of a thin section of sandy loam core showing the small-scale spatial variation in pore sizes. Pores are segmented in a light color. The network shows how the physical pore habitat varies even at the millimeter scale. The vertical extent of the image is 1 cm.

only to those organisms that fit into the pores. The variability of pore diameter at the micrometer scale is depicted in Fig. 5, where the light-colored segmented areas represent the pore network of a sandy loam collected in Rhode Island. The physical pore structure interacts with biologically active “spheres,” where soil organisms exert their influence on the soil and its microbial and faunal communities.^[7] Drilosphere and rhizosphere are the soils surrounding earthworm burrows and plant roots, respectively. The enrichment of the drilosphere with detrital resources welded to the burrow walls by earthworms and modification of the soil pore structure^[25] causes a large increase in biological activity.^[9,26] Similarly, resource enrichment can cause increases in microbial populations in the rhizosphere.^[8] Here, local photosynthate exudations prime soil biological and chemical activity.

Hattori and Hattori^[27] introduced the concept of microhabitats of protozoa and microorganisms that are located within soil aggregates, which Beare et al.^[7] called the “aggregatusphere.” With molecular techniques, it is possible to establish the spatial variability of microorganisms within single aggregates revealing great diversity at the millimeter scale.^[28]

The spatial distribution of pore spaces (also called the “porosphere”) and its partitioning into air and water-filled pores may select for specific soil organisms. Moisture controls the level of competition and predator–prey relationships that occur among soil organisms. Naturally, the relative fraction of air-filled and water-filled pores changes as soils dry and wet up, and thus, the mosaic of habitat of water and air-based soil organisms also changes. The concept of habitable pore space^[29,30] encapsulates this variability at scales of pore diameters ranging from tens to hundreds of micrometers. At these scales, soil pores are

microhabitats for protozoa^[27] and nematodes.^[25,31] However, there are different interpretations as to what happens to these habitable spaces when drying and wetting cycles rapidly alter their habitability for the aerial and the aquatic communities that inhabit soil pores.^[32]

The idea that microsites can significantly affect bulk soil processes suggests that measurement may be scale dependent. Parkin^[10] found that a soil core (100 cm³ scale) had very high denitrification activity. A single, small woodchip was isolated as the source of the high denitrification rate measured for the core. Parkin’s experiment demands that sampling support be considered in designs that are to estimate a soil property or process. Similarly, the moisture distribution within aggregates should be considered when examining denitrification in an otherwise dry soil.^[17]

Concepts of Variability

Variability is often conceptualized as the superposition of intrinsic and extrinsic factors, some of which were discussed earlier. However, there are other ways to look at soil variability. One such approach distinguishes between stochastic variations and deterministic variations that when added together are responsible for the variation in soil properties.

Deterministic variations are those that can be predicted from soil processes, and they may be spatially dependent at various scales. Spatial dependence is introduced by the translocation of matter and energy from one location to another or by the extent of large parent material deposits or even by vegetation patterns. Stochastic variations are those that cannot be predicted and are much like white noise and thus not spatially dependent. For spatial interpretations, the deterministic variables may have to be removed so that random variations can be properly characterized.

Webster^[33] argued that with sufficient knowledge about all soil processes, all variability is deterministic and that only incomplete knowledge and the sheer complexity of interacting soil processes may make variations in soil appear random. Notwithstanding philosophical argumentation about random and deterministic variations, apparent spatial dependence or independence may require different methods of estimating a central tendency and the variation about it.

One measure that is often used to describe variability is the CV, which measures the variation as standard deviation, s , standardized by the mean, μ :

$$CV = \frac{\sigma}{\mu} \quad (2)$$

The CV, however, requires that sample points are independent and may be falsely applied when there is any degree of spatial dependence. This casts some doubt on CVs estimated at different scales as soil properties may have different levels of spatial dependence at different scales.

Sampling and Quantifying Variability

Quantification of the variability of soil properties is more complex than it may appear. Even when designing the sampling method, important decisions need to be made. Apart from the classical equation for the number of samples that need to be taken to determine the mean of a sample distribution at a given confidence level and accuracy, the physical size of the sample support (the physical volume or mass of a sample specimen) becomes important as it will determine the smallest variations that can be resolved.

Sometimes, the nature of the assay will determine the sample support. For enzyme activity assays, only a couple of grams of soil is necessary. To get reliable enzyme activity information, several soil samples need to be homogenized and several replicates may need to be analyzed. The purpose of homogenization and analytical replication is to smooth small-scale variations and increase the certainty of estimation at a given scale. For measurements of bulk density, cores are taken that are typically 5–10 cm in diameter and 10 cm long. Variations that are contained within that volume are averaged out, or smoothed. Large-scale examples of spatial averaging come from remote sensing where luminosity is averaged over the area of pixels that are tens to thousands of meters in length. Selection of the satellite and sensor allows matching the scale of resolution with the research or management need. Spatial averaging is an essential tool not only for improving the reliability of data, but also focusing on the scale of the relevant variation.

Given a sufficient number of samples that were randomly collected, first-order statistics can describe soil variability in terms of statistical moments of the sample distribution: mean, variance, skewness, and kurtosis. This assumes that measured soil properties are independent of each other at the sample locations and for the scale represented by the sample support.

Many soil properties are spatially dependent. First-order, or classical, estimates of mean and variance can become biased when spatial relationships, the so-called second-order properties, are not taken into account. A second-order measure commonly used in soil science is the semivariogram, although other methods such as power spectra derived from Fourier harmonic decomposition have also been used.^[19] A semivariogram measures one half of the covariance between soil property values of sample pair classes that are classed by pair separation distances called lags. The further samples are apart, the greater their semivariance and the less related they are in space unless there are repeating patterns. Theoretically, the semivariance is limited to a maximum value that is reached when the lag exceeds the range of spatial dependence. To parameterize the semivariogram, an analytical semivariance function is fitted that best describes the variation. For spatially dependent samples, the semivariance functions increase as lag increases until the variance reaches a constant value called the “sill” (Fig. 5). The lag value at which the sill is reached

is called “range” and gives the maximum distance up to which the measured soil property is spatially correlated. Thus, for truly random variations, the sill variance is reached at lag 0. The semivariance parameter at lag 0 is also called “nugget variance.” It serves as a good aggregate estimator of error due to sampling uncertainty, unresolved variations, and analytical errors.

However, the estimates of second-order properties should be carried out with some care because just as there are rules for sampling spatially independent sample sets, there are rules for measuring the variability of spatially dependent samples. The Nyquist sampling theorem states that any variation potentially resolved by the sampling support needs to be sampled with a frequency of greater than twice the wavelength or characteristic length of the variation. If the rule is not obeyed, the signal cannot be adequately reconstructed or worse; the signal is misrepresented by an *alias* signal. For soil science, this means that under sampling a deterministic variation can result in a biased estimate of the mean and variance and potentially a misrepresentation of a pattern. An example for this is the soil moisture distribution across a plowed field (Fig. 6). The solid line in Fig. 7 shows the original, exhaustive 230-point sample set for soil moisture. The sample interval was 10 cm. The stippled line is based on 32 data points randomly selected from the original. This represents an average sampling interval of just under the wavelength, associated with a ridge and furrow pattern. The range of the original is 50 cm, but 90 cm for the undersampled variation. In addition to the overestimation of the range distance, the sill variance is overestimated by 30%. This error is not captured by the nugget variance, which usually

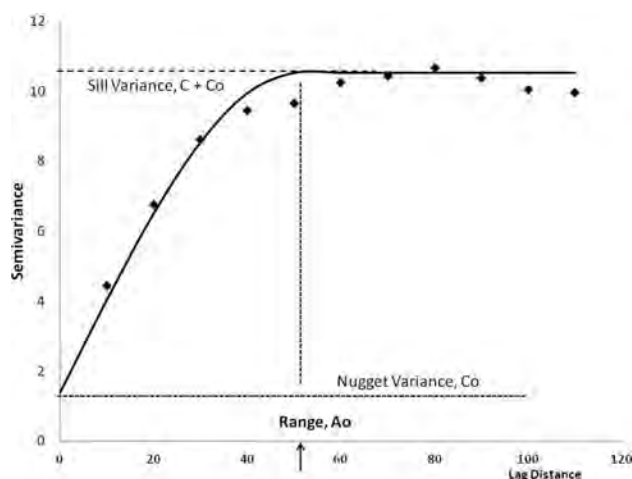


Fig. 6 Diagram of a semivariance function. There are three spatial parameters most often estimated with semivariance analyses. The distance, A_0 , where the data are no longer spatially dependent is called range. The maximum semivariance, $C + C_0$, is called sill and comprises the variance due to spatial variations, C , plus the nugget variance, C_0 , which is an aggregate measure of uncertainty due to analytical and sampling errors.

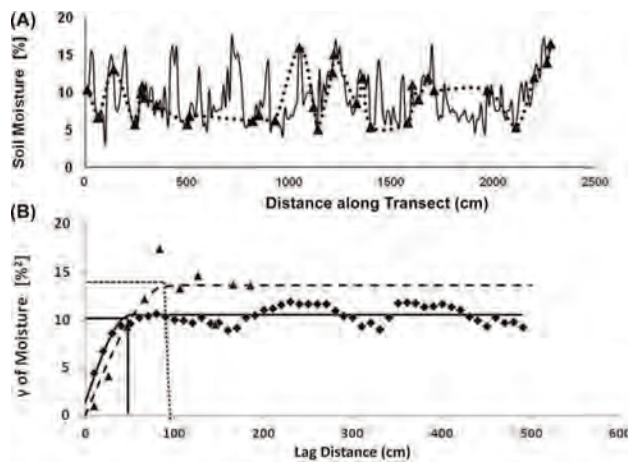


Fig. 7 (A) Exhaustive trace of soil moisture variation across a plowed field (solid line) with a randomly selected subset of data points (stippled line and triangular markers) at an average spacing just less of the wavelength of the ridge-furrow pattern. This is a violation of the Nyquist sampling theorem. The sampled variation mimics or aliases a variation with much longer spatial characteristics. (B) The semivariance function of the more exhaustively sampled moisture profile (solid line and diamond markers) has a spatially correlated range of 50 cm, while the semivariance of the aliased variation (triangular markers and broken line) is 90 cm long overestimating both the correlated range and the sill variance. Note that the nugget variance of the aliased signal is zero, showing that when signals are undersampled, the nugget variance is not a good indicator of the estimation error made.

Source: (A) Gorres, unpublished data.

is a good estimator of spatial estimation errors, but is zero for the undersampled moisture variation in this case.

For complete sample sets where sample points abut each other, the Nyquist theorem does not need to be invoked as the sample support would smooth out or integrate over any small unresolved variations. This kind of smoothing can be accomplished by taking several samples from each location, followed by combining and homogenizing the samples. Prior to analysis, satellite technology may also provide unabridged sample sets, where small-scale variations in luminosity are integrated over the pixel support. Apart from several specialized measurements (e.g., light detection and ranging), environmental satellite imagery is of low resolution and suitable only for large-scale investigations. Much progress needs to be made before remotely sensed data can replace *in situ* sampling for the analysis of some properties.

While it is clearly important to obtain as many samples as possible to satisfy needs for accuracy and improved inferential power, the cost of sampling and analysis often competes with the requirements of the central limit theorem and the Nyquist sampling theorem. Cost may prohibit high-frequency sampling across an area of interest. As a result, a bias toward long wavelength, variations may develop, exposing the estimates to aliasing of patterns. To include

and improve the estimates at shorter scales, random sampling schemes have been devised that include nests of closely spaced variations.^[26] In their study of physical and biological properties, 25×25 m plots were subdivided into twenty-five 5×5 m plots. A random core sample was taken from each plot. However, at five locations, nine samples were collected in 3×3 sample nests. The nested samples were 10 cm apart. In this way, variations at the meter and centimeter scale could be resolved.^[34]

Kriging

To estimate regional means of spatially correlated soil properties, the semivariance values are used in a spatial interpolation method called kriging. Kriging gives unbiased means and minimizes the error of estimation of regional parameters. The method was developed as a way to estimate the yield of ore deposits. In geological investigations, depositional patterns are large scale and aliasing may not be as much of a problem as in soils. Thus, the true error of the estimation of regional means with kriging depends strongly on the quality of the data collected and the care given to considering the spatial scale of patterns, even though kriging will mathematically minimize estimation error.

Some Applications of Spatial Estimation Methods

Spatial variations in soil properties are particularly important when the agricultural or environmental management requires targeted measures; e.g., nutrients need to be managed differently on a sandy soil than on a clay soil. Sometimes environmental risks can be better assessed when a spatially explicit distribution of soil properties is available. There is a wide range of applications that use the spatial variability of soils in order to optimize the estimates of properties or assess environmental risk.

One of the foremost commercial applications of the analysis of spatial patterns of soil properties is in variable rate agriculture, also known as precision agriculture. Improved productivity has been attributed to applying fertilizers at rates tailored to the previous year's yields or soil fertility status measured and spatially referenced using Global Positioning System. More fertilizer is applied in field areas where fertility is low than that in areas where it is high. Not only does this approach improve productivity and economic return, but it can also reduce nutrient loading to groundwater and surface water.

Losses of nutrients from fields are not only a risk to water quality but they also represent valuable farm resources. Spatial variations of soil properties are thus considered in farm nutrient management plans. Soil management tools such as the N leaching index and P index are used to estimate the risk of nutrient losses. In the United States, many states assign the nitrate (NO_3) leaching index based on the hydrologic group of the soil, a landscape-scale

measure of infiltration potential, as well as the regional rainfall patterns.

At much larger scales, the spatial variability of soil alkalinity has been incorporated into approaches to regulating acid air pollution. The critical load concept is central to this. The critical load of acid deposition is reached when observable changes in the ecosystem occur, i.e., when the acid challenge can no longer be buffered by the ecosystem. The critical load concept relates the atmospheric deposition of SO_x and NO_x to the rate of base cation release from soil minerals during weathering as well as other factors. Weathering is a soil-forming process, and thus, climate, parent material, organisms, and topography impose a spatial pattern, which has to be accounted for when mapping critical loads of acid deposition for a region. The typical resolution of critical load maps is one to several kilometers. Once critical loads have been determined, exceedance of the critical load for a map unit can be calculated and emissions can be regulated.

Increasingly, variations in soil properties are being used in combination with soil process models such as the Leaching Estimation and Chemistry Model (LEACHM) or Nitrogen Loss and Environmental Assessment Package in pollution risk assessment. For some applications, it is sufficient to treat the distribution of soil properties as independent. In this case, the means and measures of uncertainty of a particular soil process can be simulated by randomly selecting soil properties from their respective cumulative distribution functions as inputs to a soil process model. The repetition of these simulations gives a distribution of values for the soil process from which statistical moments can be derived; e.g., LEACHN, a submodel of LEACHM that estimates N and carbon dynamics, was parameterized using soil properties randomly selected from cumulative distribution functions of measured soil organic matter, bulk density, texture, and field capacity to simulate the mean of $\text{NO}_3\text{-N}$ in water extracted at a hypothetical production well.^[35] This approach also allowed an estimate of the probability that the water pumped at the well exceeded the Environmental Protection Agency drinking water standard of 10 mg $\text{NO}_3\text{-N/L}$ under different land use scenarios.

Watershed-scale model estimates of runoff and erosion have been improved using spatially explicit representations of the landscape. These models use remotely sensed data or spatially referenced soil survey data to establish a spatial context rather than assuming random distributions of soil property values across the landscape; e.g., the incorporation of soil information into a watershed model improved the estimate of storm hydrographs by accounting for delays in runoff contributions to stream flow.^[36]

CONCLUSION

Soils exhibit high levels of variability. CVs vary from less than 10% to greater than 100%. The magnitude of the CV

varies with the scale of measurement. Classical statistics can parameterize spatial distributions of spatial variations in terms of first-order statistical moments when the variations are spatially independent. However, more often than not, variations are spatially dependent, making it necessary to evaluate a spatial correlation between points. Any analysis of spatial heterogeneity that does not consider the properties of spatial patterns in the design of measurement and monitoring schemes will likely produce erroneous information.

REFERENCES

1. Warrick, A.W. Spatial variability. In *Environmental Soil Physics*; Hillel, D., Ed.; Academic Press: New York, 1998; 655–676.
2. Amador, J.A.; Glucksman, A.; Lyons, J.; Görres, J.H. Spatial distribution of phosphatase activity within a riparian forest. *Soil Sci.* **1997**, *162* (11), 808–825.
3. Görres, J.H.; DiChiaro, M.; Lyons, J.; Amador, J.A. Spatial and temporal patterns of soil biological activity in a forest and an old field. *Soil Biol. Biochem.* **1998**, *30* (2), 219–230.
4. Subler, S.; Baranski, C.A.; Edwards, C.M. Earthworm additions increased short term nitrogen availability and leaching in two grain-crop agro ecosystems. *Soil Biol. Biochem.* **1997**, *29* (3), 413–421.
5. Dominguez, J.; Bohlen, P.J.; Parmelee, R.W. Earthworms increase nitrogen leaching to greater soil depths in row crop agroecosystems. *Ecosystems* **2004**, *7* (6), 672–685.
6. Jenny, H. *Factors of Soil Formation: A System of Quantitative Pedology*; McGraw Hill Book Company: New York, 1941; 1–281.
7. Beare, M.H.; Coleman, D.C.; Crossley, D.A.; Hendrix, P.F.; Odum, E.P. A hierarchical approach to evaluating the significance of soil biodiversity to biochemical cycling. *Plant Soil* **1995**, *170* (1), 5–22.
8. Rouatt, J.W.; Katznelson, H.; Payne, T.M.B. Statistical evaluation of the rhizosphere effect. *Soil Sci. Soc. Am. J.* **1960**, *24* (4), 271–273.
9. Savin, M.C.; Görres, J.H.; Amador, J.A. Dynamics of soil microbial and micro faunal communities as influenced by macropores, litter incorporation, and the anecic earthworms *Lumbricus terrestris* (L.). *Soil Sci. Soc. Am. J.* **2004**, *68* (1), 116–124.
10. Parkin, T.B. Soil microsites as a source of denitrification variability. *Soil Sci. Soc. Am. J.* **1987**, *51* (5), 1194–1199.
11. Drury, C.F.; Voroney, R.P.; Beauchamp, E.G. Availability of $\text{NH}_4^+\text{-N}$ to microorganisms and the soil internal N-cycle. *Soil Biol. Biochem.* **1991**, *23* (2), 165–169.
12. Batte, M.T. Factors influencing the profitability of precision farming systems. *Soil Water Conserv.* **2000**, *55* (1), 12–18.
13. Parkin, T.B.; Starr, J.L.; Meisinger, J.J. Influence of sample size on measurement of soil denitrification. *Soil Sci. Soc. Am. J.* **1987**, *51* (6), 1492–1501.
14. Soil Survey Staff. *Soil Taxonomy: A Basic Guide for Soil Classification for Making and Interpreting Soil Surveys*, 2nd Ed.; USDA-NRCS: Washington, 1999.

15. Nearing, M.A.; Pruski, F.F.; O'Neal, M.R. Expected impacts of climate change on soil erosion rates. *Soil Water Conserv.* **2005**, *59* (1), 43–50.
16. Kallenrode, M.-B. Droughts in the Western U.S. In *Space Physics: An Introduction to Plasmas and Particles in the Heliosphere Magnetospheres*; Springer: Berlin, 2004; 397–398.
17. Sexstone, A.J.; Revsbech, N.P.; Parkin, T.B.; Tiedje, J.M. Direct measurement of oxygen profiles and denitrification rates in soil aggregates. *Soil Sci. Soc. Am. J.* **1985**, *49* (3), 645–651.
18. Fanning, D.S.; Fanning, M.C.B. The factors of soil formation—Overview. In *Soil: Morphology, Genesis and Classification*; John Wiley and Sons: New York, 1989; 287–290.
19. McBratney, A.B.; Webster, R. Detection of a ridge and furrow pattern by spectral analysis of crop yield. *Int. Stat. Rev.* **1981**, *49* (1), 45–52.
20. Raghavan, G.S.V.; McKeyes, E.; Chassé, M.; Merineau, F. Development of compaction patterns due to machinery operation in an orchard soil. *Canad. J. Plant Sci.* **1976**, *56* (3), 505–509.
21. Afzal, M.; Adams, W.A. Heterogeneity of soil mineral nitrogen in pasture grazed by cattle. *Soil Sci. Soc. Am. J.* **1991**, *56* (4), 1160–1166.
22. Ellenberg, H.; Weber, H.E.; Duell, R.; Wirth, V.; Werner, W.; Paulissen, D. Indicator values of plants in Central Europe. *Scripta Geobot.* **1992**, *18*, 160–166.
23. Amador, J.A.; Görres, J.H. Microbiological characterization of the structures built by earthworms and ants in an agricultural field. *Soil Biol. Biochem.* **2007**, *39* (8), 2070–2077.
24. Ettema, C.H.; Wardle, D.A. Spatial soil ecology. *Trends Ecol. Evol.* **2002**, *17* (4), 177–183.
25. Görres, J.H.; Amador, J.A. Partitioning of habitable pore space in earthworm burrows. *J. Nematol.* **2010**, *42* (1), 68–72.
26. Görres, J.H.; Savin, M.; Amador, J.A. Dynamics of carbon and nitrogen mineralization, microbial biomass, and nematode abundance within and outside the burrow walls of anecic earthworms (*Lumbricus terrestris*). *Soil Sci.* **1997**, *162* (9), 666–671.
27. Hattori, T.; Hattori, R. The physical environment in soil microbiology: An attempt to extend principles of microbiology to soil microorganisms. *CRC Crit. Rev. Microbiol.* **1976**, *4* (4), 423–461.
28. Mummey, D.; Holben, W.; Six, J.; Stahl, P. Spatial stratification of soil bacterial populations in aggregates of diverse soils. *Microb. Ecol.* **2006**, *51* (3), 404–411.
29. Elliott, E.T.; Anderson, R.V.; Coleman, D.C.; Cole, C.V. Habitable pore space and microbial trophic interactions. *Oikos* **1980**, *35* (3), 327–335.
30. Hassink, J.; Bowmann, L.A.; Zwart, K.B. Relationship between habitable pore space, soil biota and mineralization rates in grassland soils. *Soil Biol. Biochem.* **1993**, *25* (1), 47–55.
31. Neher, D.A.; Weicht, T.R.; Savin, M.C.; Görres, J.H.; Amador, J.A. Grazing in a porous environment. 2. Nematode community structure. *Plant Soil* **1999**, *212* (1), 85–99.
32. Görres, J.H.; Savin, M.C.; Neher, D.A.; Weicht, T.R.; Görres, J.H. Grazing in a porous environment: 1. The effect of soil pore structure on C and N mineralization. *Plant Soil* **1999**, *212* (1), 75–83.
33. Webster, R. Is soil variation random? *Geoderma* **2000**, *97* (3–4), 149–163.
34. Amador, J.A.; Wang, Y.; Savin, M.C.; Görres, J.H. Small-scale variability of physical and biological soil properties in Kingston, Rhode Island. *Geoderma* **2000**, *98* (1), 83–94.
35. Görres, J.H.; Gold, A. Incorporating spatial variability into GIS to estimate nitrate leaching at the aquifer scale. *J. Environ. Qual.* **1996**, *25* (3), 491–498.
36. Zhu, A.X.; Mackay, D.S. Effects of spatial detail of soil information on watershed modeling. *J. Hydrol.* **2001**, *248* (1), 54–77.

Variability: Spatial

Brian Slater

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Soils are highly variable in time and space. Scientists and environmental managers need to understand soil variability to design appropriate strategies to maintain the capacity of soils to provide essential environmental services. Variability also implies diversity, which can be viewed as a positive attribute of the soil cover. Pedometric techniques for quantifying variability and depicting its spatial characteristics are becoming essential tools for more precise and effective management of soils and productive enterprises that depend on them.

INTRODUCTION

Soil is a product of natural processes, which vary greatly in space and time, and is highly variable. It is this soil heterogeneity that makes it worthy of scientific attention. “Were the soil uniform, we should simply acknowledge the fact and shift our attention to something more interesting.”^[1] The major pedological endeavors—to understand the history of soil development, to classify the soil, and to depict soil geography—are all fundamentally directed toward knowledge of soil variability. Likewise, understanding and managing services provided by the soil resource for human benefit require knowledge of the nature of soil variability and the development of methods to measure, predict, and manage it.

Spatial variability has mostly been treated as a challenge to be controlled. Efficient farmers may wish to be able to apply consistent management across significant areas from year to year. However, soil variability may demand more adaptable management based on understanding how some soil properties (e.g., moisture-holding capacity) vary spatially within and between fields and how other properties (e.g., the actual amount of stored water) vary within and between growing seasons. Variability can be both a challenge and an asset.^[2]

VARIABILITY, DIVERSITY, AND HETEROGENEITY

It is well understood in ecology that diversity (the corollary of variability) creates the potential for stability, resilience, and a hedge against risk. In the longer term, a dryland farmer may benefit from the presence of contrasting soils (and soil properties such as soil moisture behavior) within a farm or field because rainfall may be highly variable and management-inconsistent. Having areas of light-textured soils may provide rapid crop response to lighter falls of rain; heavier soils may store more water during prolonged drought, and medium-texture soils may provide maximum

moisture availability and high yields in the best years. This diversity may result in efficient environmental function; e.g., areas of poorer infiltration may promote runoff, which is accepted in adjacent soils with higher permeability. Soil variability also factors into the diversity of natural and managed environments, of which soil is an integral part, diverse soils support diverse landscapes and a variety of habitats for living organisms and people.

Spatial variability implies that there are places where soil properties are best suited for specific purposes. An understanding of spatial variability leads to better matching soil management and soil characteristics to maintain the capacity of the soil to provide environmental services. Technology transfer and the scaling of known responses from sites, research plots, and fields to broader areas depend on knowledge of spatial variation. Agronomic research has evolved from attempts to control variability in field experiments to its embrace in precision agriculture.

SCALES OF VARIATION

Variability exists across the entire range of spatial scales. At the submicroscopic and microscopic scales, contrasts exist between various soil minerals, organic and living components, gases in the soil atmosphere, and the soil solution, creating soil fabric. Soil structure provides the architecture for soil water movement through pores and solid surfaces for adsorption and exchange of nutrients. At a broader scale, similar properties may cluster as a result of similar genetic materials and processes, and adjacent soils with contrasting properties are frequently arrayed together in landscapes.

ORIGINS OF SOIL VARIABILITY

Many early observations of soil variability are related to differences in plant yield and vigor and contrasts in mechanical and physical properties of soil as a material for

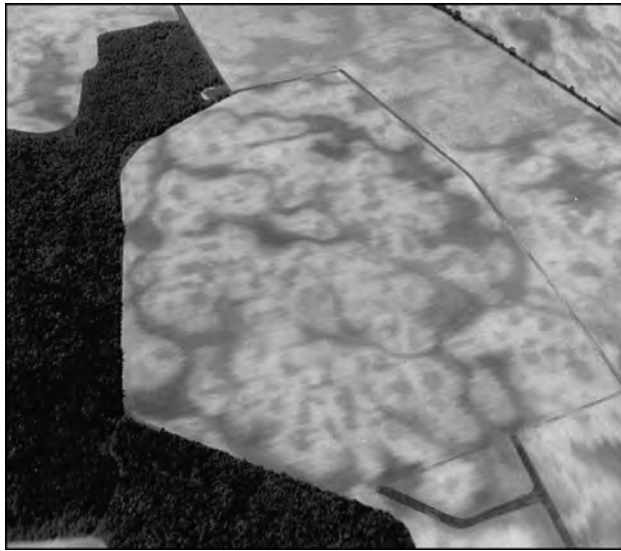


Fig. 1 Aerial image of farm fields in Central Ohio, showing soil variability associated with contrasts in drainage conditions.

construction (Fig. 1). People needed to understand and predict these contrasts to improve crop production and safely fabricate buildings and roads. With the development of pedology in 19th-century Russia,^[3] the idea that soil developed according to the interaction of factors of soil formation provided a basis for understanding differences in the morphology and behavior of soils observed in the field. Observations that similar attribute values and soil qualities clustered spatially and were often related more or less predictably to other environmental features such as types of geological materials, landform and landscape positions, patterns of surface and subsurface water movement, and vegetation provided impetus for soil survey efforts.^[4]

Patterns include chronosequences related to surfaces of contrasting geomorphic age and stability, lithosequences related to varying types of parent material and bedrock, toposequences resulting from contrasting relief on similar parent materials, and biosequences responding to changes in flora and fauna. These differences provided the basis for early forms of soil classification and the development of geographic units delineated on soil maps. As well as these more serial (anisotropic) patterns, repetitive patterns of varying regularity in some soils result from directional forces. Examples include gilgai microrelief in Vertisols, tree throw, and cryoturbation features.

Geologic materials and transported sediments exhibit major differences in composition and structure and associated physical and chemical properties related to their lithologic origin and history of development and transport. The weathering and reworking of materials and products further add to the diversity of soil-forming precursors. Geomorphic differentiation results from the effects of contrasting hydrologic conditions, microclimate effects

including solar radiation differences, and variable erosion and accretion on surfaces of variable stability. Finally, biological activity within and on the soil contributes to soil evolution and variation.

Many of these spatial contrasts are visible or measurable directly, although patterns present at short or long ranges have become clear with the advent of remote sensing techniques, instruments, and imagery, from microscopes to aerial photography and the variety of satellite-based imaging platforms. Much soil variability exists in the third dimension and hence can be best understood only through excavation of the soil in the field. Over time, soils develop horizons, approximately horizontal layers that reflect specific patterns and processes of soil genesis. Even at a single site, soil properties and functions vary spatially in the vertical direction. Variability can be greatly different between different horizons and for different attributes within an individual horizon.^[5]

Human influences have modified natural patterns and established new forms of heterogeneity. Land clearing and forestry operations, tillage, fertilizer and amendment applications, accelerated erosion and compaction, earth shaping, drainage, building of roads, structures, and other impervious surfaces have altered natural patterns.

SYSTEMATIC AND RANDOM VARIATION

While highly variable, considerable orderliness exists in the pattern of soil cover. This orderliness contributes to the utility of soil survey and classification.^[6] To be captured on soil maps, there must be contiguity of entities with similar properties and some external expression of boundaries (e.g., locations of rapid, coincident variation of multiple attributes). Conventional soil survey relies on the presence of relatively homogeneous patches of similar soil with a limited range of attribute variation, with more variation at and across boundaries.^[7] However, in reality, abrupt changes may be rare (dependent on scale); soil attributes often have a more continuous, gradual, or noisy spatial distribution.^[7,8] Conventional soil survey depictions of discrete polygons (*choropleth* maps) most consistently capture abrupt patterns of soil variation, where there is a strong correlation between soil properties and sensible external expression of boundaries such as changes in vegetation, landform, and remotely sensed image patterns. Even where a particular attribute changes abruptly in geographic space, other attributes may vary independently or at different rates or different scales. Soil surveys based on external classification systems rather than “taken from the landscape” locally^[6] delineate impure map units. Individual bodies of soil (pedons and polypedons) are not expected to exhibit the same range of variation in all attributes defined by taxonomic classes, and the geographic projection of these classes will frequently not yield large contiguous polygons, but rather smaller patches interspersed with

either contrasting bodies or collections of sites where one or more attributes lie outside the range defined by the class limits.^[9] Many studies of soil variability by pedologists have concentrated on defining the variation within mapping units.^[10] Mapping systematic variation may be useful for spatial generalizations and interpretations.

MEASURING SOIL VARIABILITY

While pedologists have concentrated on understanding systematic variability, the aim of many agronomic researchers has been to control or eliminate it. Where the soil population can be assumed to meet traditional requirements of normality, randomness, and independence,^[11] variability can be measured by parametric statistics such as the coefficient of variation (CV):

$$CV = \frac{s}{\bar{x}} \times 100 \quad (1)$$

where s is the standard deviation and $\bar{x}$ is the mean.

The CV has frequently been used to compare variability of collections of samples such as those taken from within different mapping units for the purpose of comparison. Samples for comparing variability of map units have often been taken on linear transects or according to various “design-based” sampling schemes aimed at gaining sufficient information to estimate mean attribute values with known levels of confidence.^[12]

Systematic soil variability may often be difficult to categorize deterministically with sufficient confidence, and variation may appear chaotic. Patterns may be obscured or impossible to measure and predict because of random or unidentified sources of variation or unavoidable measurement error. The need to quantify and propagate uncertainty in models of soil spatial variability has provided impetus to the growing field of soil science known as pedometrics.

Spatial clustering of similar attribute values is evidence of spatial dependence. Geostatistical methods have been widely used to model spatial dependence stochastically. Models that treat the soil as a product of random spatial processes have proved successful in providing a basis for measuring, mapping, and managing soil spatial variability.^[13]

This geostatistical model is often stated:^[14]

$$Z(x) = \mu + \varepsilon(x) \quad (2)$$

where $Z(x)$ is a *regionalized variable*, μ is the mean, and $\varepsilon(x)$ is a random component or *residual*.

The residuals are considered to be spatially (auto) correlated; sample locations of greater proximity are more likely to have similar attribute values. The *semivariance* is a quantity that represents the degree of spatial autocorrelation:

$$\gamma(h) = \frac{1}{2} E\{[Z(x) - Z(x+h)]^2\} \quad (3)$$

where $\gamma(h)$ is semivariance, E is the expectation, and h is a vector representing the distance between observations (the *lag*).

In practice, the semivariance is estimated from samples; e.g., for a linear transect of n sites, the semivariance at lag h is:

$$\hat{\gamma}(h) = \frac{1}{2(n-h)} \sum_{i=1}^{n-h} \{Z(x_i) - Z(x_i + h)\}^2 \quad (4)$$

where h is the distance separating pairs of sample sites.

Spatial dependence can be expressed by a function that relates the correlation between pairs of observations to the distance between observations. The function that relates the semivariance to the lag is the *semivariogram* (or just *variogram*; Fig. 2). Graphical realizations of the variogram are often used to gauge the structure of spatial variation. Typically, the semivariance increases with increasing lag, frequently reaching a maximum (the *sill*) at a finite lag distance (the *range*) beyond which spatial dependence cannot be measured. The variogram typically does not pass through the origin, as it would be for a purely continuous process,^[1] but has an ordinal intercept known as the *nugget variance* that is most often interpreted as a random uncorrelated short-range error component, perhaps a result of measurement inaccuracy. Models can be fit to the function produced from a series of observations (e.g., by an automated procedure such as minimizing the sum of squares). The fitted relationship can then be used to interpolate attribute values for locations between sample stations, a technique known as *kriging*. Kriging uses a linear weighting based on the lag distances from measured points and weights determined to minimize the variance of the prediction error.

The value of an attribute Z at a location x_0 is:

$$\hat{Z}(x_0) = \sum_{i=1}^n \lambda_i Z(x_i) \quad (5)$$

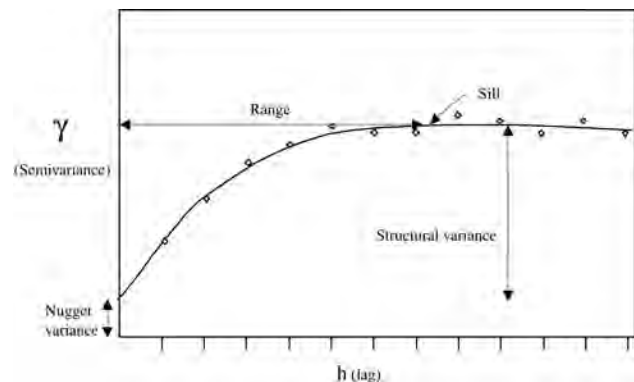


Fig. 2 Schematic semivariogram for soil properties.

where $Z(x_i)$ are a series of point observations and λ_i are weights.

Kriging is considered an optimal interpolator in that it minimizes prediction error (for well-behaved models) and is unbiased. Kriging has been used to estimate attribute values at unvisited sites, to make continuous maps, or to estimate values over defined blocks (block kriging). Methods have been developed to incorporate prior data and covarying attributes (cokriging), to map occurrences of attribute values meeting defined criteria (indicator kriging), and to simulate multiple realizations of stochastic outcomes.^[15]

MANAGING VARIABILITY

Soil diversity means that we cannot know everything everywhere about soils. As a result, there is uncertainty and hence a need to sample, interpolate, and map variation. Soil variability demands differential, site-specific management.

Soil management needs to be responsive to the variation in the resource to minimize economic and environmental costs. Soil managers and conservationists have long understood that contrasts in soil characteristics result in differences in the capacity of soils to produce and provide services and also demand differences in treatment. Soil and land capability maps and farm conservation plans are examples of simple spatial depictions of soil variability and consequences for management.

Increasing recognition of short-range variability has fostered the development of precision farming techniques, which aim to direct management responses to specifically matched soil and environmental conditions at within-field scales. Precision management involves collection of information on soil spatial and temporal variability and the tailoring of inputs and practices to the capacities and demands of the resource, with the aim of reducing costs, increasing production, and minimizing leakage of inputs (such as fertilizers) from the system.

CONCLUSION

Soils are highly variable in time and space. Scientists and environmental managers need to understand soil variability to design appropriate strategies to maintain the capacity of soils to provide essential environmental services. Variability also implies diversity, which can be viewed as a positive attribute of the soil cover. Pedometric techniques for

quantifying variability and depicting its spatial characteristics are becoming essential tools for more precise and effective management of soils and productive enterprises that depend on them.

REFERENCES

1. Heuvelink, G.B.M.; Webster, R. Modeling soil variation: Past, present, and future. *Geoderma* **2001**, *100*, 269–301.
2. Heuvelink, G.B.M.; Webster, R. Reply to comment on “Modeling soil variation: Past, present, and future,” by Philippe Baveye. *Geoderma* **2002**, *109*, 295–297.
3. Simonson, R.W. Early teaching in USA of Dokuchaiev factors of soil formation. *Soil Sci. Soc. Am. J.* **1997**, *61*, 11–16.
4. Hudson, B.D. The soil survey as a paradigm-based science. *Soil Sci. Soc. Am. J.* **1972**, *56*, 836–841.
5. Kachanoski, R.W. Processes in soils—From pedon to landscape. In *Scales and Global Change*; Rosswall, T., Woodmansee, R.G., Risser, P.G., Eds.; John Wiley and Sons Ltd.: Chichester, 1988; 153–177.
6. Butler, B.E. *Soil Classification for Soil Survey*; Clarendon Press: Oxford, 1980.
7. Burrough, P.A. Soil variability: A late 20th century view. *Soils Fert.* **1993**, *56*, 529–562.
8. Burrough, P.A.; Bouma, J.; Yates, S.R. The state of the art in pedometrics. *Geoderma* **1994**, *62*, 311–326.
9. McBratney, A.B.; de Gruijter, J.J.; Brus, D.J. Spatial prediction and mapping of continuous soil classes. *Geoderma* **1992**, *54*, 39–64.
10. Wilding, L.P.; Drees, L.R. Spatial variability and pedology. In *Pedogenesis and Soil Taxonomy. 1. Concepts and Interactions*; Wilding, L.P., Smeck, N.E., Hall, G.F., Eds.; Elsevier: Amsterdam, 1983; 3–116.
11. Upchurch, D.R.; Edmonds, W.J. Statistical procedures for specific objectives. In *Spatial Variabilities of Soils and Landforms*; Mausebach, M.J., Wilding, L.P., Eds.; SSSA Special Publication, Soil Science Society of America: Madison, 1991; Vol. 28, 49–72.
12. Burrough, P. Sampling designs for quantifying map unit composition. In *Spatial Variabilities of Soils and Landforms*; Mausebach, M.J., Wilding, L.P., Eds.; SSSA Special Publication, Soil Science Society of America: Madison, 1991; Vol. 28, 89–126.
13. Webster, R. Is soil variation random? *Geoderma* **2000**, *97*, 149–163.
14. Webster, R.; Oliver, M.A. *Geostatistics for Environmental Scientists*; Wiley: Chichester, 2001.
15. Goovaerts, P. *Geostatistics for Natural Resources Evaluation*; Oxford University Press: New York, 1997.

Variable Charge Soils: Mineralogy and Chemistry

Eric Van Ranst

Geology and Soil Science, Ghent University, Ghent, Belgium

Nikolla P. Qafoku

Pacific Northwest National Laboratory, Richland, Washington, U.S.A.

Andrew Noble

*Integrated Water and Land Management and Ecosystems Program,
International Center for Agricultural Research in the Dry Areas (ICARDA), Amman, Jordan*

Ren-kou Xu

*State Key Laboratory of Soil and Sustainable Agriculture, Institute of Soil Science,
Chinese Academy of Sciences, Nanjing, China*

Abstract

Soils rich in particles with amphoteric surface properties in the Oxisols, Ultisols, Alfisols, Spodosols, and Andisols orders are considered to be variable charge soils. They have developed under intensive weathering in subtropical and tropical regions or from volcanic ash parent material. The term “variable charge” is used to describe organic and inorganic soil constituents with reactive surface groups whose charge varies with pH, and ionic concentration and composition of the soil solution. The variable charge is developed on the surface groups as a result of adsorption or desorption of solid-like ions that are constituents of the solid phase, i.e., H^+ , and the adsorption or desorption of solid-unlike ions that are not constituents of the solid phase. Variable charge soils are heterogeneous charge systems. Variations in relative proportions and chemical composition of the major mineral soil constituents of highly weathered soils and volcanic ash soils influence the surface reactivity, which is the key factor controlling nutrient availability. The coexistence and interactions of soil particles and colloids with net opposite surface charges confer interesting properties with complex attributes with respect to physical and chemical behavior, compared to homogeneously charged soil systems of temperate regions. In this entry, the mineralogy, chemistry and management of variable charge soils are reviewed and discussed, focusing on the chemistry of these fascinating soil systems. Because surface charge properties are of central importance in the management of variable charge soils, different management practices that may manipulate charge characteristics and retention capacities are discussed.

INTRODUCTION

Soils rich in particles with amphoteric surface properties in the Oxisols, Ultisols, Alfisols, Spodosols, and Andisols orders^[1] are considered to be variable charge soils (Table 1).^[2] The term “variable charge” is used to describe organic and inorganic soil constituents with reactive surface groups whose charge varies with pH and ionic concentration and composition of the soil solution. Such groups are the surface carboxyl, phenolic, and amino functional groups of organic materials and surface hydroxyl groups of iron (Fe) and aluminum (Al) oxides, allophane, and imogolite. The hydroxyl surface groups are also present on edges of some phyllosilicate minerals such as kaolinite, mica, and hydroxyl-interlayered vermiculite. The variable charge is developed on the surface groups as a result of adsorption or desorption of ions that are constituents of the

solid phase, i.e., H^+ , and the adsorption or desorption of solid-unlike ions that are not constituents of the solid phase.

Highly weathered soils and subsoils (e.g., Oxisols and some Ultisols, Alfisols, and Andisols) may undergo iso-electric weathering and reach a “zero net charge” stage during their development. They usually have a slightly acidic to acidic soil solution pH, which is close to either the point of zero net charge (PZNC)^[4] or the point of zero salt effect.^[4] They are characterized by high abundances of minerals with a point of zero net proton charge (PZNPC)^[4] at neutral and slightly basic pHs; the most important being Fe and Al oxides and allophane. Under acidic conditions, the surfaces of these minerals are net positively charged. In contrast, the surfaces of permanent charge phyllosilicates are negatively charged regardless of ambient conditions. Variable charge soils, therefore, are heterogeneous charge systems.

Table 1 Global areas and percentages of suborders of the soil orders with variable charge soils.

Orders	Suborders	Area [km ² (×10 ³)]	Subtotal [km ² (×10 ³)]	Proportion (%)	Subtotal (%)
Alfisols	Aqualfs	836		0.7	
	Cryalfs	2,518		1.9	
	Ustalfs	5,664		4.3	
	Xeralfs	897		0.7	
	Udalfs	2,706	12,621	0.2	9.6
Andisols	Cryands	255		0.2	
	Torrands	2		0.01	
	Xerands	32		0.01	
	Vitrands	281		0.2	
	Ustands	63		0.1	
	Udands	279	912	0.2	0.7
Oxisols	Aquox	320		0.2	
	Torrox	31		0.01	
	Ustox	3,096		2.4	
	Perox	1,162		0.9	
	Udox	5,201	9,810	4	7.5
Spodosols	Aquods	169		0.1	
	Cryods	2,460		1.9	
	Humods	58		0.01	
	Orthods	667	3,354	0.5	2.5
Ultisols	Aquults	1,281		1.0	
	Humults	344		0.3	
	Udults	5,540		4.2	
	Ustults	3,870		3.0	
	Xerults	19	11,054	0.01	8.5

Source: Data from USDA-NRCS, Soil Survey Division, World Soil Resources, 1998 (modified after Wilding^[3]).

The coexistence and interactions of oppositely charged surfaces or particles confer a different pattern of physical and chemical behavior on the soil, relative to the homogeneously charged soil system characteristic of temperate regions. In some variable charge soils (Oxisols and some Ultisols developed on ferromagnesian-rich parent materials), the surfaces of phyllosilicates are coated to a lesser or greater extent by amorphous or crystalline, oppositely charged nanoparticles of Fe and Al oxides. These coatings exhibit a high reactive surface area and help cementing larger particles with one another. Because of these electrostatic interactions, stable microaggregates that are difficult to disperse are formed in the variable charge soils.

Most highly weathered soils have reached the “advanced stage” of the Jackson–Sherman weathering sequence that is characterized by the removal of sodium, potassium, calcium (Ca), magnesium (Mg), and Fe(II), the presence of Fe and Al polymers, and very dilute soil solutions with an ionic strength (IS) of less than 1 mmol L⁻¹. The interpenetration or overlapping of the diffuse double layers on oppositely charged surfaces may occur in these dilute systems. These diffuse layer interactions may

decrease the magnitude of the effective charge, i.e., the counterion charge,^[5] and may also inhibit the natural tendency for acidification of the soils.^[6] In addition, salt adsorption, which is defined as the simultaneous adsorption in equivalent amounts of the cation and anion of an electrolyte with no net release of other ions into the soil solution, appears to be a common phenomenon in these soils. They act as cation and anion exchangers and as salt-sorbers. The magnitude of salt adsorption depends strongly on initial IS in the soil solution and the presence of appreciable amounts of oppositely charged surfaces. The overlapping of the diffuse layers on oppositely charged surfaces may be responsible for the salt adsorption in variable charge soils.^[7,8]

Among the authors who have made illustrious contributions toward a better understanding of these fascinating soil systems are S. Matson,^[9] R.K. Schofield,^[10] M.E. Sumner,^[5,11–15] van Olphen,^[16] G.W. Thomas,^[17] G.P. Gillman,^[7,18–22] B.K.G. Theng,^[2] J.W. Bowden,^[23] G. Uehara,^[7] K. Wada,^[24] G. Sposito,^[4,25] N.J. Barrow,^[26] R.J. Hunter,^[27] J. Chorover,^[25] L.W. Zelazny,^[28] T.R. Yu,^[29] P. Bertsch,^[30] E. Van Ranst,^[31–33] N. Qafoku,^[5,13–15,33,34]

A.D. Noble,^[22,33] and R.K. Xu.^[6,8,35–38] This entry is mainly based on publications by these authors.

MINERALOGY OF VARIABLE CHARGE SOILS

The most important mineralogical components in variable-charge soils are kaolinite, Fe oxides, and gibbsite in Oxisols; kaolinite, hydroxyl-interlayered vermiculite, muscovite, Fe and Al oxides, and minor amount of smectite in Ultisols (quartz in the sand and silt fractions); kaolinite and smectite in the highly weathered Alfisols (mica and Fe oxides in smaller abundances and quartz and feldspars in the coarse fractions); minor amounts of mica, chlorite, kaolinite, and vermiculite in the clay fraction of Spodosols (orthoclase, plagioclase, mica, pyroxene, and amphibole in the sand fraction; and quartz and traces of plagioclase in the silt and sand fraction); and allophane, imogolite, poorly crystalline Fe oxides (e.g., ferrihydrite), volcanic glass (which is a mixture of aluminosilicates and traces of ferromagnesian minerals), and secondary silicon (Si) minerals (cristobalite, opaline silica) in Andisols.

The clay fraction mineralogy of variable charge subsoils is mostly dominated by the quintet: kaolinite, gibbsite, goethite, hematite, and short-range-order minerals (e.g., allophane, imogolite, and ferrihydrite). However, the contents and proportions of mineralogical constituents, particle size distribution, and specific surface areas are different in unlike soils. As a result, they manifest a significant diversity in their properties. Variable charge subsoils are likely to have one of the following five mineralogical characteristics when they reach the apparent steady state during their development.^[13,29] 1) large amounts of kaolinite and crystalline and less reactive Fe and Al oxides, mainly hematite and gibbsite (typical examples are some Oxisols from Africa, South America, Australia, and Southern China); 2) large amounts of kaolinite and very reactive Fe and Al oxides, mainly Al-substituted goethite and gibbsite (typical examples are Ultisols from the Southeastern United States, South America, Africa, and Southern China); 3) large amounts of extremely reactive short-range-order minerals such as allophane, imogolite, and ferrihydrite (typical examples are Andisols from Indonesia, Pacific Rim countries, and New Zealand); 4) almost monophase mineralogy dominated by kaolinite and small amounts of Fe oxides (typical examples are some Ultisols from Southeastern United States, Southern Africa, and Brazil and some Oxisols from Hawaii); and 5) almost monophase mineralogy dominated by not very reactive Fe and Al oxides and, in some cases, Ni oxides (typical examples are some Oxisols from New Caledonia and Jamaica). Because kaolinite, Fe and Al oxides, and allophane are the most typical mineralogical components of these soils, they are treated in more detail below.

Kaolinite is a 1:1 phyllosilicate that has an Al octahedral sheet and a Si tetrahedral sheet. Each layer of kaolinite has

two surfaces: the surface of oxygens of the Si tetrahedral sheet and the surface of hydroxyls of the Al octahedral sheet. Both these surface groups are fully charge satisfied and have low reactivity. However, the edges of kaolinitic particles have unsatisfied and quite reactive, singly coordinated aluminol and silanol variable charge groups. Kaolinite has the lowest surface charge (about 1–5 cmol_(c) kg⁻¹) among common clay minerals because of the relatively small number of isomorphic substitutions in its structure. It has a PZNC at pH about 3.5 and a PZNPC at pH about 5; it also has a low specific surface area (between 5 and 39 m² g⁻¹) and, commonly, a pseudohexagonal plate shape.

Fe and Al oxides are the most important sources of variable charge in soils of the humid tropics. Their amphoteric surfaces are capable of sorbing and desorbing protons depending on pH and IS of the soil solution. Their PZNPC is at pHs over 7, so they have a net surface positive charge under acid conditions typical for variable charge soils. Goethite and hematite are the most important Fe oxides in highly weathered soils. The substitution of Al for Fe is more frequent in these soils. This affects surface charge as it generally results in a lower degree of crystallinity, a smaller particle size, and hence a significant increase in surface area and sorption capacities. They occur as nanoparticles with different shapes and high specific surface area.

Under the name of allophane are included a group of aluminosilicate minerals with primarily short-range structural order that occur as spherical nanoparticles 3–6 nm in diameter, with a chemical composition of silica and alumina. The specific surface area varies between 700 m² g⁻¹ and 900 m² g⁻¹. Allophane may have both permanent and variable surface charge. The variable charge results from protonation and dissociation of surface aluminol and silanol functional groups, with aluminol groups having negative, neutral, or positive charge and the more acidic silanol groups having either neutral or negative charge. Allophane interacts effectively with soil organic matter (OM).

CHEMISTRY OF VARIABLE CHARGE SOILS

With respect to the origin of the surface charge, the soil minerals common to variable charge soils can be grouped in^[1] constant surface charge and^[2] variable charge soil minerals.

Constant Charge Minerals

In constant surface charge minerals, the permanent charge is due to imperfections in the lattice structure resulting from isomorphous substitution. The substitution is generally restricted to a lower valence element for those with a higher valence, e.g., a divalent Mg for a trivalent Al, resulting in a negative charge at the particle surface. This negative charge is then compensated by the accumulation at the crystal surface of the counterions that are oppositely charged,

hydrated or non-hydrated ions. As those defects occur in the interior of the crystal lattice, the resulting surface charge is permanent and it is not influenced by pH, electrolyte concentration, dielectric constant and ionic composition of the aqueous phase, and temperature. In this case, the fundamental equation of the Gouy–Chapman theory of the double layer may be written as follows:^[28]

$$(8.6 \times 10^{-3} \text{ RT } \epsilon \epsilon_0 \text{C})^{1/2} \sinh \frac{ze\Psi_0}{2\text{RT}} = \sigma_{\text{permanent}} = \text{const.}$$

where

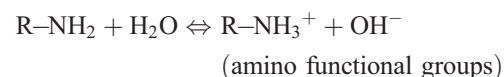
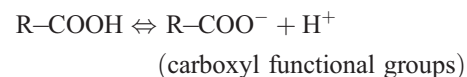
$\sigma_{\text{permanent}}$ = permanent surface charge (cmol kg^{-1});
 Ψ_0 = surface potential (volts);
 e = electronic charge ($1.6021 \times 10^{-19} \text{ C}$);
 z = valence of the adsorbed ions (dimensionless but carries charge);
 C = electrolyte concentration (M);
 ϵ = dielectric constant of aqueous phase (78.5 at 298 K);
 ϵ_0 = permittivity of free space ($8.854 \times 10^{-12} \text{ C V}^{-1} \text{ mol}^{-1}$);
 R = molar gas constant ($8.314 \text{ J mol}^{-1} \text{ K}^{-1}$); and
 T = absolute temperature (K).

This equation implies that surface potential, however, is affected by all the factors mentioned above, i.e., electrolyte concentration, temperature, and dielectric constant and ionic composition of the aqueous phase. The most important constant surface charge minerals are the 2:1 phyllosilicates with variable basal spacing, such as vermiculites and smectites. Smectites, if present, only occur in small abundances in highly weathered tropical soils. In volcanic ash soils, they are usually present in minor amounts but occasionally in substantial or dominant amounts.

Variable Charge Minerals

In variable charge soil minerals, the surface charge is created by the adsorption of the potential-determining ions (pdi) or other specifically adsorbed ions onto the surface. The charging process requires the presence of these ions in the soil solution in sufficient quantities for adsorption to occur. Variable charge or amphoteric constituents in soils may have organic and/or inorganic origin.

The origin of charge on the organic fraction is due to ionization of active carboxyl, amino, and phenolic functional groups as a result of their protonation or deprotonation as it is shown in the following reactions:



Acidic functional groups are the main source of the exchange capacity in the organic fraction. Carboxyl groups

probably account for the highest proportion of the cation exchange capacity (CEC) of soil OM at pH 7.^[32]

In the highly weathered tropical soils, inorganic constituents with amphoteric surfaces are Fe and Al oxides, clay minerals with edges with broken bonds, and complexes of Fe and Al with OM. Again these functional groups acquire their charge by either protonation or deprotonation. The surface charge may also originate from surface complex formation of specifically adsorbed ions. The Gouy–Chapman equation of the double layer theory for symmetrical electrolyte may be written in this case as follows:

$$(8.6 \times 10^{-3} \text{ RT } \epsilon \epsilon_0 \text{C})^{1/2} \sinh \frac{ze}{2\text{RT}} (\text{const.}\Psi_0) = \sigma_{\text{variable}}$$

The surface potential does not vary with electrolyte concentration, dielectric constant of the aqueous phase, and temperature. The surface potential is also independent of ionic composition of the aqueous phase, if the ions are not pdi or have not the ability to form inner sphere complexes with the surface reactive groups. Surface potential changes with pH or when activity of ions that may potentially be specifically adsorbed changes in the aqueous phase. We should emphasize here that most soil variable charge surfaces such as Fe and Al oxides, hydroxides, and oxy-hydroxides do not obey the Nernst equation, which means that their surface potential changes by less than 59 mV for each unit change in pH.^[27] The fundamental assumption involved when the Nernst equation is derived is that the activity of the pdi on the surface remains constant as the surface acquires charge. This is not true in the case of Fe and Al oxides, which have few charged surface groups at their PZNPC. The interface between the solid Fe or Al oxide surfaces and the aqueous phase in contact with them in variable charge soils is depicted in Fig. 1.

MANAGEMENT OF VARIABLE CHARGE SOILS

Variable charge soils are generally acidic and have low CECs and the propensity to fix phosphorus (P) and boron.^[40] They support tropical rain forests that have an intrinsically tight nutrient cycling capacity. Soil OM that effectively acts as a slow-release nutrient source mediates this cycling. When these ecosystems are disturbed and placed under agronomic production, OM is rapidly lost due to continuous mixing of surface soils, and consequently, there is a rapid decline in fertility. This decline is associated with an accelerated acidification and a reduction in the CEC, thereby limiting the ability of the soil to hold cationic macronutrients and micronutrients, which are rapidly lost through leaching. Insufficient negative charge in the subsoil will also lead to poor root development, making plants incapable of using subsoil moisture during droughty periods. On the other hand, these soils develop appreciable anion exchange capacity (AEC) under acidic conditions, which retain otherwise leachable anions such as nitrate and

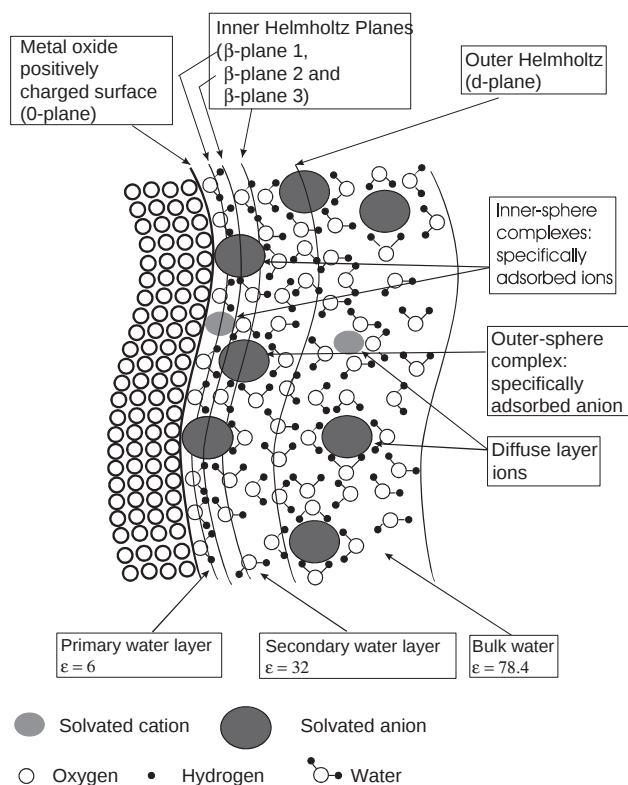


Fig. 1 Schematic presentation (not in proportional dimensions) of the electrical double layer at the Fe oxide: aqueous solution interface. The 0-plane is defined by the location of protonated or deprotonated surface sites. The β -planes extend from surface to the centers of specifically adsorbed anions or cations. The d-plane corresponds to the beginning of the diffuse layer of counterions and coions.

Source: Modified after Brown and Parks.^[39]

sulfate particularly in subsoils. In addition, some experiments seem to show that adsorption of low-affinity cations, such as Ca, reduced surface charge and even resulted in charge reversal. Due to these intrinsic characteristics, surface charge properties are of central importance in the management of variable charge soils (Table 2).

The surface charge characteristics and retention capacities in variable charge soils may be manipulated via:

1. Increasing OM content: The CEC and amounts of nutrients stored in highly weathered and volcanic ash topsoils are mainly related to the OM content. OM content decreases progressively with time under cultivation in these soils, and the loss of OM through oxidation or erosion occurs especially during prolonged cropping; this reduces the ability to supply nitrogen, P, and sulfur and to retain cations in exchangeable form.^[32] Agricultural practices aiming at increasing OM content and decreasing OM removal through erosion are therefore important. However, under humid tropical conditions, increasing soil OM and maintaining it are problematic as it is rapidly mineralized.

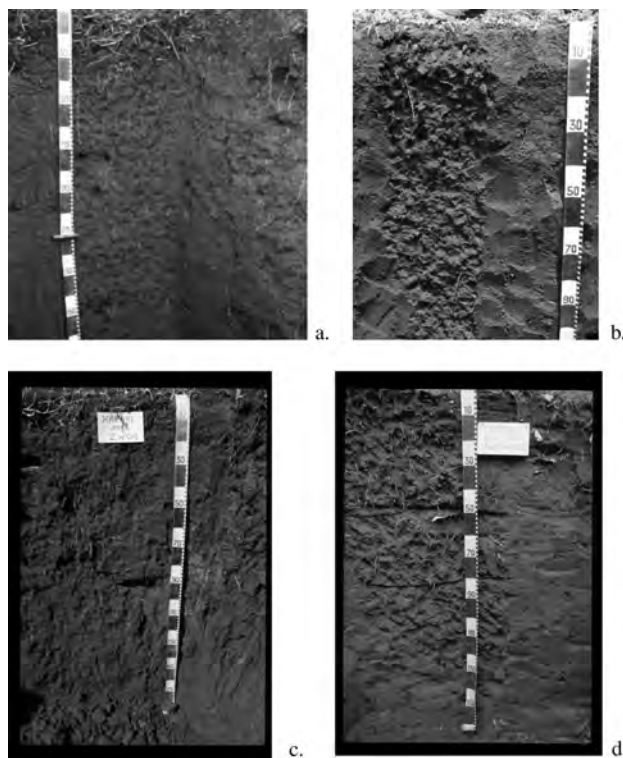


Fig. 2 Some typical variable charge soils: a Typical Haplustox from West Cameroon, an Oxyaquic Paleudalf from Southwest Ethiopia, a Rhodic Paleustalf from the Harare region in Zimbabwe, and an Acrudoxic Hapludand from Central Java in Indonesia.

Source: Photos courtesy of E. Van Ranst, Ghent University, Belgium.

2. Decreasing PZNPC in soils: One way of lowering PZNPC in soils would be to increase the OM content in the soil. OM has a low PZNPC ($\text{pH} \sim 4$), thus contributing to decreasing the soil PZNPC. The large organic anions may also be adsorbed on positively charged surfaces, reducing the amount of positive charge in subsoil. The adsorption of organic anions of low molecular weight organic acids secreted from plant roots and produced through the decomposition of plant residues in variable charge soils can also decrease PZNPC of the soils through increasing negative surface charge and decreasing positive surface charge.^[35] Phosphate (PO_4) anions may react with the surface of soil colloids by adsorption reactions and reverse charge. Consequently, in situations where it is not possible to build up OM levels, inorganic soil amendments such as PO_4 could be used to lower the PZNPC. However, this is a rather expensive intervention in tropical countries.
3. Changing soil pH: Increasing the soil pH through the addition of liming materials will result in the precipitation of phytotoxic Al and an increase in CEC. The increase in soil pH enhances the soil capacity to hold and supply essential plant nutrients. However, in order to increase the soil pH to 6 or above, as is often advocated in permanently charged temperate soils,

Table 2 Constraints and management options for variable charge soils.

Constraint	Management options
I. Soil acidity—Al toxicity	Routine prophylactic applications of calcitic or dolomitic limestone. Soil pH should be maintained at pH 5.5 to reduce the risk of Al^{3+} toxicity. If manganese (Mn) toxicity is suspected as pH should be raised to 6.0. Overliming decreases the availability of P, boron, zinc, and Mn and results in yield declines. In addition, there is a decline in soil structure. Judicious use of acidifying nitrogenous fertilizers will minimize acid generation through the conversion of NH_4^+ to NO_3^-. Alternatively, the use of NO_3^--based sources of nitrogen would eliminate any acid generation associated with mineralization. Retention of crop residues and organic matter to increase the pH buffering capacity. Application of organic materials with high ash alkalinity is an alternative source of alkalinity where conventional liming materials are not available. Application of biochar to increase soil pH and exchangeable base cations and decrease soil exchangeable Al^{3+}. Application of gypsum increases levels of soluble Ca^{2+} and reduces levels of toxic Al^{3+} in subsoils.^[41] Subsoil acidity can be corrected through a combination of growing deep-rooted acid-tolerant grass species (<i>Andropogon spp.</i>) and in combination with leaching, and nitrate fertilizations will result in the remediation of subsoil acidity.
II. Low cationic retention from fertilizers	Regular small amounts of inorganic fertilizer should be applied to reduce the risk of leaching loss and increase efficiency. In addition, fertilizer additions will increase the ionic strength, thereby increasing the surface charge. Increasing the negativity of the exchange complex can be achieved through increasing the soil pH or decreasing PZNPC. Lowering of PZNPC can be achieved by sorption of large anions onto particle surfaces, thereby masking a portion of the positive charge. Continued regular applications of organic matter and biochar will increase the CEC of soils as well as effectively lower the PZNPC. In situations where organic matter levels are not possible to increase, inorganic soil amendments such as phosphate and silicates could be used to lower PZNPC. When increasing the negative charge of these soils, the anion exchange capacity will decline, thereby reducing the ability of these soils to retain anionic species such as nitrate.
III. Strong fixation of anionic fertilizers such as P	Try to reduce the reactivity of surfaces for PO_4. This could be achieved by increasing the amounts of colloids with low affinity for PO_4 fixation, the addition of other anions that compete with PO_4, and the application of organic matter. In addition, sources of silica, ground basalt, volcanic ash, and Ca silicate can be used. Banded applications of inorganic P fertilizers. Use of rock phosphate under acid conditions
IV. Low nutrient-supplying capacity	The application of superphosphate may promote the release of nutrients through the breakdown of silicate minerals through the formation of H_3PO_4. Si deficiency on highly weathered soils can be rectified through the addition of silicate materials.

enormous inputs of lime are required, which is inefficient because of the high buffering capacity of these soils; it may also result in micronutrient deficiencies. By maintaining these soils at a pH of 5.5, phytotoxic Al is eliminated from the exchange complex, and a significant amount of negative charge is generated. Frequent small applications of lime are advocated to achieve this objective. This may also be done through the application of basic rock dusts that act as a slow-release source of Ca and Mg as well as being a liming material.^[19] In addition, laboratory and field experiments with ground, calc-alkaline pyroclastic materials from the Nyiragongo volcano (Democratic Republic of Congo) added to highly weathered soils have shown that the negative surface charge may increase in these soils.^[31] Nevertheless, it is of practical importance to maintain the pH of variable charge soils at levels

sufficiently high to precipitate Al and to increase CEC especially in Ultisols of the humid tropics that contain 2:1 clay minerals with variable basal spacing. Andisols are generally characterized by low amounts of exchangeable Al, and overliming them commonly results in manganese and zinc deficiencies. One should be careful when applying management techniques that aim at changing the pH in soils because an increase in soil solution pH would increase the CEC but decrease the AEC. An increase in CEC would help agricultural production, but the corresponding decrease in AEC would decrease the capacity of soils, especially subsoil, to retain contaminant anions such as nitrate. Management practices that contribute to increasing the negative permanent charge in soils are, therefore, more desirable in some cases. The application of permanently charged clay minerals (i.e., bentonite) has been

shown to permanently increase the negative charge component and productivity of degraded Oxisols.^[22] Gypsum can also increase negative charge.^[42]

4. Increasing both soil pH and CEC through incorporation of biochar: Biochar is a carbon (C)-rich, trendy yet ancient soil amendment, produced by slow thermochemical pyrolysis of biomass materials. It is formed when organic materials, such as livestock manures, sewage sludge, crop residues, and compost, are heated at temperatures between 300°C and 1000°C under relatively low oxygen conditions. It is possible that the application of biochar to soils would help in the long-term maintenance of the soil's organic C content and soil fertility. In addition, biochars can have an unusually high CEC, which increases the adsorption capacity of soils for selected nutrients and toxic heavy metals decreasing their mobility. Biochars normally have alkaline pH and can be used to ameliorate soil acidity and to increase the pH buffering capacity of acid soils.^[37] Biochar is stable and difficult to decompose in soils compared with its feedstock and thus a better amendment for acidic variable charge soils than plant residues.

CONCLUSION

Variable charge soils have unique surface charge characteristics and mineralogy. Kaolinite and sesquioxides are the most common minerals in the clay fraction, and little or no weatherable minerals are present in the silt and sand fractions. Variations in relative proportions and chemical composition of the major mineral soil constituents of highly weathered soils and volcanic ash soils influence the surface reactivity, which is the key factor controlling nutrient availability. Nutrient management of these soils effectively dictates small and frequent applications of inorganic fertilizers. The most typical properties of variable charge soils are low effective CEC, high effective AEC, low OM, low available water-holding capacity, high hydraulic conductivity, very stable microaggregates, high P appreciable AEC that significantly retards the adsorption, and relatively high point of zero charge. A rational management can be obtained by trying to change the nature of the colloidal surfaces, leading to an increase in CEC. This might be achieved by blocking positively charged sites, e.g., application of P fertilizers and silicates and incorporation of young pyroclastic materials and biochars, or by a better management of the OM and a more efficient erosion control.

REFERENCES

1. Soil Survey Staff. *Keys to Soil Taxonomy*, 12th Ed.; Natural Resources Conservation Service, USDA: Washington, DC, 2014; 359 pp.
2. Theng, B.K.G. *Soils with Variable Charge*; New Zealand Society of Soil Science: Lower Hutt, 1980.
3. Wilding, L. Classification of soils. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, FL, 2000; E-175–E-182.
4. Sposito, G. *The Surface Chemistry of Soils*; Oxford University Press: New York, 1984.
5. Qafoku, N.P.; Sumner, M.E. Adsorption and desorption of indifferent ions in variable charge subsoils: The possible effect of particle interactions on the counter-ion charge density. *Soil Sci. Soc. Am. J.* **2002**, *66* (4), 1231–1239.
6. Li, J.Y.; Xu, R.K.; Zhang, H. Iron oxides serve as natural anti-acidification agents in highly weathered soils. *J. Soil. Sediment.* **2012**, *12* (6), 876–887.
7. Uehara, G.; Gillman, G.P. *The Mineralogy, Chemistry and Physics of Tropical Soils with Variable Charge Clays*; Westview Press: Boulder, 1981.
8. Li, S.Z.; Xu, R.K.; Li, J.Y. Electrical-double layer interaction between oppositely charged particles in variable charge soils as related to salt adsorption. *Soil Sci.* **2009**, *174*, 27–34.
9. Mattson, S. The laws of colloidal behavior: VI. Amphoteric behavior. *Soil Sci.* **1931**, *32*, 343–365.
10. Schofield, R.K. Effect of pH on electrical charges carried by clay particles. *J. Soil Sci.* **1949**, *1* (1), 1–8.
11. Sumner, M.E. Effect of alcohol washings and pH value of leaching solution on positive and negative charges in ferruginous soils. *Nature* **1963**, *198*, 1018–1019.
12. Alva, A.K.; Sumner, M.E.; Miller, W.P. Salt absorption on gypsum amended acid soils. In *Plant-Soil Interactions at Low pH*; Wright, R.J., Ed.; Kluwer Academic Publications: Dordrecht, 1991; 93–97.
13. Qafoku, N.P.; Sumner, M.E.; West, L.T. Mineralogy and chemistry of some variable charge subsoils. *Commun. Soil Sci. Plan* **2000**, *31*, 1051–1070.
14. Qafoku, N.P.; Sumner, M.E.; Radcliffe, D.E. Anion transport in column of variable charge subsoils: Nitrate and chloride. *J. Environ. Qual.* **2000**, *29* (2), 484–493.
15. Qafoku, N.P.; Sumner, M.E. Retention and transport of calcium nitrate in variable charge subsoils. *Soil Sci.* **2001**, *166* (5), 297–307.
16. van Olphen, H. *An Introduction to Clay Colloid Chemistry*, 2nd Ed.; John Wiley & Sons: New York, 1977.
17. Thomas, G.W. Historical developments in soil chemistry: Ion exchange. *Soil Sci. Soc. Am. J.* **1977**, *41* (2), 230–238.
18. Gillman, G.P. A proposed method for the measurement of exchange properties of highly weathered soils. *Aust. J. Soil Res.* **1979**, *17*, 129–139.
19. Gillman, G.P. The effect of crushed basalt scoria on the cation exchange properties of a highly weathered soil. *Soil Sci. Soc. Am. J.* **1980**, *44*, 465–468.
20. Gillman, G.P. Effects of pH and ionic strength on the cation exchange capacity of soils with variable charge. *Aust. J. Soil Res.* **1981**, *19*, 93–96.
21. Gillman, G.P. Using variable charge characteristics to understand the exchangeable cation status of oxic soils. *Aust. J. Soil Res.* **1984**, *22*, 71–80.
22. Noble, A.D.; Gillman, G.P.; Nath, S.; Srivastava, R.J. Changes in the surface charge characteristics of degraded soils in the tropics through the addition of beneficiated bentonite. *Aust. J. Soil Res.* **2001**, *39*, 991–1001.

23. Bowden, J.W.; Nagarajah, S.; Barow, N.J.; Posner, A.M.; Quirk, J.P. Describing the adsorption of phosphate, citrate, and selenite on a variable charge mineral surface. *Aust. J. Soil Res.* **1980**, *18*, 49–60.
24. Wada, K.; Okamura, Y. Net charge characteristics of Dystrandept B and theoretical prediction. *Soil Sci. Soc. Am. J.* **1983**, *47*, 902–905.
25. Chorover, J.; Sposito, G. Surface charge characteristics of kaolinitic tropical soils. *Geochim. Cosmochim. Ac.* **1995**, *5*, 875–884.
26. Barrow, N.J. *Reactions with Variable Charge Soils*; Martinus Nijhoff Publishers: Dordrecht, 1987.
27. Hunter, R.J. *Introduction to Modern Colloid Science*; Oxford University Press: New York, 1993.
28. Zelazny, L.W.; Liming, H.; Vanwormhoudt, A. Charge analysis in soils and anion exchange. In *Methods of Soil Analysis, Part 3-Chemical Methods*; Sparks, D.L., Ed.; Soil Science Society of America, Inc.: Madison, 1996; 1231–1253.
29. Yu, T.R. *Chemistry of Variable Charge Soils*; Oxford University Press: New York, 1997.
30. Bertsch, P.M.; Seaman, J.C. Characterization of complex mineral assemblages: Implications for contaminant transport and environmental remediation. *P. Natl. Acad. Sci. USA.* **1999**, *96*, 3350–3357.
31. Van Ranst, E. Rational soil management in the humid tropics. *Meded. Zitt. K. Acad. Overzeese Wet.* **1995**, *40* (1994–2), 209–233.
32. Van Ranst, E.; Shamshuddin, E.J.; Baert, G.; Dzwowa, P.K. Charge characteristics in relation to free iron content and organic matter of soils from Bambouto Mountains, Western Cameroon. *Eur. J. Soil Sci.* **1998**, *49*, 243–252.
33. Qafoku, N.P.; Van Ranst, E.; Noble, A.; Baert, G. Variable charge soils: Their mineralogy, chemistry and management. *Adv. Agron.* **2004**, *84*, 159–215.
34. Qafoku, N.P. Terrestrial nanoparticles and their controls on soil/geo processes and reactions. *Adv. Agron.* **2010**, *107* (2), 33–91.
35. Xu, R.K.; Zhao, A.Z.; Ji, G.L. Effect of low molecular weight organic anions on surface charge of variable charge soils. *J. Colloid Interf. Sci.* **2003**, *264* (2), 322–326.
36. Wang, Y.; Jiang, J.; Xu, R.; K.; Tiwari, D. Phosphate adsorption at variable charge soils/water interfaces as influenced by ionic strength. *Aust. J. Soil Res.* **2009**, *47*, 529–536.
37. Yuan, J.H.; Xu, R.K. The amelioration effects of low temperature biochar generated from nine crop residues on an acidic Ultisol. *Soil Use Manage.* **2011**, *27* (1), 110–115.
38. Li, J.Y.; Xu, R.K. Inhibition of acidification of kaolinite and an alfisol by aluminum oxides through electrical double-layer interaction and coating. *Eur. J. Soil Sci.* **2013**, *64* (1), 37–45.
39. Brown, G.E.; Parks, G.A. Sorption of trace elements on mineral surfaces: Modern perspectives from spectroscopic studies, and comments on sorption in the marine environments. *Int. Geol. Rev.* **2001**, *43*, 963–1073.
40. Terraza Pira, M.F.; Sumner, M.E.; Cabrera, M.L.; Thompson, A. Boron adsorption on Guatemalan south coast soils with andic properties. *Soil Sci. Soc. Am. J.* (submitted).
41. Sumner, M.E. Gypsum and acid soils: The world scene. *Adv. Agron.* **1993**, *51*, 1–32.
42. Sumner, M.E. Amelioration of subsoil acidity with minimum disturbance. In *Subsoil Management Techniques*; Jayawardane, N.S., Stewart, B.A., Eds.; Lewis Publishers: Boca Raton, FL, 1995; 147–186.

Vertisols

Wouter A. Blokhuis

*International Soil Reference and Information Centre (ISRIC), Wageningen,
the Netherlands*

Abstract

Vertisols are clay soils in which the clay fraction is dominated by swelling clay minerals, mainly smectites. Vertisol formation is favored by a warm climate with alternating wet and dry seasons. Vertisols have a characteristic morphology resulting from the swelling and shrinking of the clays upon changes in moisture content. Soil properties have a distinct impact on agricultural practice. The soils are used for rain-grown and irrigated cropping. They require specific agronomic techniques and land use systems.

INTRODUCTION

Vertisols are dark-colored clay soils with a specific structure and with surface characteristics and deep cracks when dry. The soil swells upon wetting and shrinks when drying. Clay percentages, by definition 30 or higher,^[1] are often over 60. Smectite minerals (montmorillonite and others) dominate the clay fraction. Soil parent materials are either silty/clayey deposits, rich in smectite minerals, or weathering materials from mainly basic rock types (marl, limestone, basalt, and basic igneous rock) that produce swelling clay minerals upon further weathering. Both the formation of the parent materials and the formation of the typical Vertisol profile are favored by a warm climate with alternating wet and dry seasons.

The preservation of basic ions during weathering and soil formation is essential for the formation of smectites, and this explains the occurrence of most Vertisols in poorly drained, often level terrain. Vertisols developed from basic rocks generally occupy sloping positions; external drainage is normal, but internal drainage is sufficiently slow and leaching restricted, so that the smectite clays are formed. Most Vertisols are found between 45° north and south latitudes in a semiarid to subhumid environment. The largest areas are in India (79 mha), Australia (70.5 mha), and Sudan (50 mha). Vertisols cover extensive areas in several other African countries, the Americas, and China and smaller areas in Southern Asia and Southern Europe (Table 1). They are also described in arctic semiarid regions, e.g., in Canada and the United States.^[4] Vertisols and other soils with vertic properties may cover up to 320 mha worldwide.^[3] Because of their physical properties and the resulting problems with soil and water management, Vertisols are correctly labeled “problem soils.” This should not, however, distract from the fact that with proper management Vertisols are among the most productive soils in semiarid agriculture.

Problems are also met in the transfer of research results and technologies from other soils to Vertisols. The

applications of techniques that are effective in non-swelling soils have often produced misleading results. Concepts such as field capacity, permanent wilting point, available water capacity (AWC), and bulk density have a different or more restricted meaning, and Vertisols sometimes require specific laboratory or field investigations.

SOIL PHYSICAL AND CHEMICAL PROPERTIES RELEVANT TO AGRICULTURAL USE

Soil Structure

Most structural features of Vertisols have seasonal aspects. This implies non-permanency of their porous system, a striking difference contrasting with rigid soils.^[5] When dry, most Vertisols have subangular and wedge-shaped angular peds, increasing in size with depth. Cracks, some centimeters wide at the soil surface, extend to depths between 60 and 100 cm. Cracks are commonly obscured at the surface by fine crumb to granular soil aggregates (Fig. 1). This surface mulch is developed because of repeated wetting and drying.^[6] Other Vertisols develop a hard surface crust, i.e., 2–3 cm thick.

Factors That Influence Soil Structure

Clay percentage and amount of swelling
clay minerals

Generally, there is a positive correlation between clay percentage and percentage of smectite clay minerals in the clay fraction and degree of structure development.^[7] Vertisols with over 60% of mainly smectite clay are well structured: they have a surface mulch, a finely aggregated surface soil, a blocky structure with wedge-shaped peds, and many slickensides in the subsoil. Vertisols with relatively low-clay and high-sand contents are poorly structured.

Table 1 Distribution of dark clay soils.

Countries and areas	Estimated extent (mha)
<i>Angola</i> : valleys of the Cunene and Cunbango, region of Catete, and southwestern part of the country	0.5
<i>Argentina</i> : mainly in the northeastern part of the country in the province of Entre Rios, in the northeastern department of Buenos Aires, in south Corrientes, Santa Fe, and the Eastern Chaco	6.0
<i>Australia</i> : mainly in Queensland (Darling Downs), northern plains, east-central part and coastal areas of the Northern Territories, patches in south Australia (near Adelaide), northwestern Australia, and in Tasmania	70.5
<i>Benin</i> : northern part of the country and in the region of Divo	0.1
<i>Bolivia</i> : mainly in the eastern part of the country, Livingstone, and Tuli; probably large areas around the Okovambo and Makarikari swamps	2.0
<i>Botswana</i> : west of Livingstone and Tuli; probably large areas around Makarikari swamps	0.5
<i>Brazil</i> : southwestern and western regions of Rio Grande do Sul (Bage, Uruguaiana, Alegrese, and d. Pedrito districts); in the western parts of the country bordering Uruguay, Paraguay, and Bolivia; in north eastern Brazil	4.5
<i>Burkina Faso</i> : Souron Valley; poorly drained basin deposits spread over the country	0.4
<i>Cameroon</i> : Logone Chari basin and part of the Chad Basin; penepplain of Kaele and Marona region	1.2
<i>Chile</i> : depressional areas associated with non-calcic brown soils in Santiago and O'Higgins provinces and also associated with Prairie and Chestnut soils in Magallanes Province	0.5
<i>China</i> : Central China	12.0
<i>Ecuador</i> : hilly lowland and valley bottoms in western part of the country (provinces of Guayas, Manabi, Esmeraldas)	1.0
<i>Egypt</i> : Nile delta	1.0
<i>Ethiopia</i> : rift valley and Ethiopia plateau	13.0
<i>Ghana</i> : mainly Accra, Ho-Keta, and Winneba Plains; scattered patches near Kpandu, Kwamen, and Kwesi	0.2
<i>India</i> : Central and South Central Deccan plateau (parts of Bombay, Hyderabad, and Madhya Pradesh state)	79.0
<i>Indonesia</i> : mainly in Central Java (Semarang, Demak, Bojonegoro, Surabaya area), East Java and Lesser Sunda Islands (Lombok, Timor, Sumbawa, and Flores)	1.8
<i>Ivory Coast</i> : in northern part of the country and in the region of Divo	2.8
<i>Kenya</i> : Athi Plains near Nairobi and other areas	Unavailable
<i>Lesotho</i> : Drakensberg	2.3

(Continued)

Table 1 Distribution of dark clay soils. (Continued)

Countries and areas	Estimated extent (mha)
<i>Madagascar</i> : some valley depressions in the western part of the country and on the uplands in the western and northwestern part	0.8
<i>Malawi</i> : Chyre valley and areas around Nyasa and Chilwa lakes	1.6
<i>Mali</i> : Niger valley, and a large area along the borders with Mauritania	0.7
<i>Marocco</i> : mainly in the northwest (Gharb) and in the doukkalas (south of Casablanca); scattered spots	0.2
<i>Mozambique</i> : alluvial plains of the Limpopo and Inkomati rivers and surrounding uplands; Zambezi valley upstream of Tete	1.1
<i>Namibia</i> : probable occurrence in the Caprivi Strip along the Cubango river and around the Etosha Pan	0.7
<i>Niger</i> : central Niger valley and several scattered patches	0.1
<i>Nigeria</i> : northeast Bornu and Benoue river basin	4.0
<i>Paraguay</i> : depressional areas in basaltic and limestone plateaus of the eastern region; in the Paraguay river basin and large areas in the Chaco region	1.5
<i>Portugal</i> : depressional area on diorite and limestone in Alentejo	0.1
<i>Senegal</i> : Senegal valley, lowland of the northwest	0.2
<i>Somalia</i> : in the plains extending between the Juba and the Shebeli rivers	0.8
<i>South Africa</i> : Bushveld and Springbok Flats (Transvaal)	2.1
<i>Sudan</i> : region between the White and the Blue Nile extending east of the Blue Nile into Ethiopia and covering an area west of the White Nile: widespread in South Sudan, Bahr el Ghazal, Upper Nile, and Equatoria Province	50.0
<i>Swaziland</i> : mainly in the Middelveld, eastern Lowveld and Lebombo areas	0.2
<i>Syria</i> : Jezireh and basaltic plateaus, south of Damascus	0.6
<i>Tanzania</i> : valley of the Mayowsi and Malagarasi, Great Ruaha valley areas near Tendigo swamps and Rukwa lake; scattered upland areas	7.0
<i>Togo</i> : in the northern part of the country and in the Mono valley	0.1
<i>Uganda</i> : valley of the Semliki, areas near George, Albert, and Edward lakes, areas in the eastland northeast of the country	1.7
<i>Union of Soviet Socialist Republics</i> : compact Chernozems in the Caucasus between Krasnodar and Grozny	0.8
<i>United States of America</i> : "Blackland" in Central Texas from the Red River bottomland on the north	18.0

(Continued)

Table 1 Distribution of dark clay soils. (Continued)

Countries and areas	Estimated extent (mha)
and northeast to the Rio Grande plain in the San Antonio area on the southwest; belt extending from Lowndes County in Missouri to Perry County in Alabama; basaltic plateaus in Arizona; scattered spots in California, N. Dakota, and Montana; islands of Oahu, Kauai Molokai in Hawaii	
Uruguay: in the southwest, northwest, and south-central part of the country and along the border with Brazil (Department of Cerro Largo)	1.0
Venezuela	1.5
Zaire: Ruzizi Plain, valley of the Semliki and Lufira	0.3
Zambia: valley of the Luangwa, Lukushi, and Zambezi; Kafue flats	5.0
Zimbabwe: part of the valley of the Zambezi; Livingstone area, the southern part of the country	1.8

Source: From Ahmad^[2] and Dudal & Eswaran.^[3]

They have a thin surface crust and a coarse blocky structure with few large slickensides in the subsoil.

Organic matter

Organic matter in Vertisols ranges from 5 to 100 g/kg and is commonly between 5 and 50 g/kg.^[8] Organic matter functions, such as binding nutrients and water, are, in Vertisols, provided by active clay surfaces. Organic matter in the soil (crop residues, weeds, decaying roots, and waste compost) stimulates biological activity and supplies organic colloids to form organo-mineral composite aggregates. Organic matter has an ameliorative effect on notably the physical properties of the soil such as structure development and aggregate stability,^[7,9,10] and it reduces dry bulk density, penetration resistance, and shear strength and increases saturated hydraulic conductivity, porosity, and soil water and nutrient retention.^[11–14] Blanchart et al.^[15] mention the important role of

roots and earthworms in restoring the physical properties of degraded Vertisols.

Coulombe, Wilding, and Dixon^[8] consider clay-organic complexes to be fundamental to the formation and stability of soil aggregates. Research in Australia, summarized by McGarry,^[7] has shown that an improved structural stability was linked with the larger amounts of the “active fraction,” i.e., the more labile types of organic matter rather than with the total amount.

Cation exchange capacity, exchangeable cations, and electrolyte concentration of the soil solution

Vertisols generally have high cation exchange capacities and high base saturations. Commonly, calcium or magnesium dominates, depending on the parent material.^[16] In Vertisols with a high proportion of divalent cations, the clays are flocculated and the soils well structured. Vertisols with high exchangeable sodium (sodic soils) have surface crusts and widely spaced cracks enclosing compact large aggregates. They tend to disperse upon wetting.^[8]

Coupling electrical conductivity (EC) with the exchangeable sodium percentage (ESP) or sodium absorption ratio is necessary to predict clay dispersion.^[17] Crescimanno et al.^[18] found that at low cationic concentration, physical soil degradation occurs in a 2–5 ESP range, and that at increasing ESP, soil behavior appears as a continuum. Heber, Shedaksharappa, and Patil^[19] observed that hydraulic conductivity decreased and dispersion increased with increasing ESP and decreasing electrolyte concentration. The presence of some minerals such as gypsum can overcome the dispersing action of sodium and flocculate soil particles.^[20]

Climate and landform

The length of the rainy season, the amount and intensity of annual rainfall, and the frequency of wet–dry cycles impact the soil structure of Vertisols. In Sudan, the strongly self-mulching soils with fine aggregation are in the northern arid part of the Vertisol belt.^[21–24] Generally, these soils have free carbonates and gypsum; calcium is the dominant cation. In higher-rainfall regions, increased soil leaching results in lower pHs, whereas there are no free carbonates and gypsum; mulch development is reduced and soil structural aggregates are coarser. Crusty Vertisols are restricted to sodic spots in irrigated fields and topographic low situations with standing water after heavy rains.^[25]

SOIL CONSISTENCE

Soil consistence directly impacts tillage operations. Generally, Vertisols are sticky and very plastic when wet, hard to very hard when dry, and friable when moist. Moist consistence is the optimum for tillage operations, but unfortunately the time span that the soil is moist is usually short and its



Fig. 1 Cracks and surface mulch in a Vertisol.

occurrence is uncertain because it depends on often erratic rainfall.

Sodic Vertisols are smearier when wet, more easily dispersed, and extremely hard when dry. Sandy Vertisols have lower plasticity and stickiness as compared to modal Vertisols, whereas the large aggregates—or large clods resulting from tillage—are extremely hard and do not crumble into a mulch upon wetting and drying.

SOIL–WATER RELATIONS

Infiltration and Hydraulic Conductivity

Infiltration into a dry surface mulch or prepared seedbed is rapid. Subsurface moistening is rapid as long as water can enter open cracks. When cracks close after wetting, infiltration is greatly reduced. Water in cracks passes through fissures between ped surfaces (including slickensides) and through biopores into the soil matrix.^[26] Infiltration is promoted by the retention of crop residue or fallow weeds, either as a cover or plowed into the soil.

It was observed in furrow irrigation^[27] that water in cracks was 10 m ahead of water on the surface. Shrinkage fissures connect cracks, so that irrigation water passes through quickly. If a rainy season starts with heavy storms, wetting through cracks and connecting fissures is rapid. Lighter rains give a more gradual surface wetting whereby crack entrances are closed, and cracks fail to function as preferential pathways.^[5]

The hydraulic conductivity in a moist Vertisol is slow, whereas in a wet Vertisol, it is very slow. Measuring hydraulic conductivity in Vertisols is problematic, as there is no horizontal wetting front and pore-size changes with the moisture content.^[28]

Moisture Retention

Vertisols have a relatively high moisture retention at all water potentials. The AWC defined as the difference between the water contents at 0.033 MPa (0.33 bar; field capacity) and 1.5 MPa (15 bar; permanent wilting point) has been reported as 110–250 mm for the surface 1 m depth.^[29] At field capacity and wilting point, a positive correlation was found between water content and both clay percentage and amount of smectite minerals in the clay fraction.^[30,31] Table 2 gives chemical and physical characteristics, including water contents at field capacity and permanent wilting point and the AWC of a typical Vertisol in Sudan.

Kutilek^[5] found that Vertisols with ESP between 10 and 20 showed a drastic increase of the lower boundary of available water and suggested to abandon the concept of the 1.5 MPa wilting point in sodic Vertisols and use the direct experiment on plant wilting instead. He concluded that AWC in Vertisols cannot be measured by procedures empirically derived in non-vertic soils. Farbrother^[32] proposed to consider the saturated water content as the upper boundary of AWC in Vertisols of the semiarid zone.

The concept of plant available water capacity (PAWC) was introduced by Gardner et al.^[33] PAWC is defined as the amount of water that can be extracted from an initially fully wet soil by a crop before a chosen stress symptom is observed.^[34] PAWC varies with crop, root development, depth of rooting, and other soil properties and gives a more realistic figure for the storage of soil water that is available to the plant roots in a given situation than does AWC (Fig. 2). Kirchhof, Smith, and Hyson^[36] mention the effect of soil textural class, soil moisture, and soil depth (an overburden effect) on readily plant available water.

Table 2 Chemical and physical characteristics of a Typic Pellustert (S81 FN 835–9) from near Damazin, Sudan (11°45'N and 32°47'E).

Water content									
Depth (cm)	pH ^a	EC ^c (mmhos cm)	Color moist	Clay (%)	Structure ^b	FC ^c (%)	PWP ^d (%)	Bulk ^e density (g/cm ³)	Available ^f water (cm/cm)
0–5	7.6	0.96	10YR 3/1	64.2	3cpl/3msbk	46.7	33.3	1.06	0.14
5–25	8.2	0.59	10YR 3/1	57.6	3cpl/2msbk	45.0	32.0	1.07	0.14
25–55	8.6	0.69	10YR 3/1	68.7	3cpl/2mabk	32.2			
55–90	8.8	0.94	10YR 3/1	71.8	3cpl/2mabk	47.6	33.3	1.09	0.16
90–125	8.5	1.45	10YR 3/1	74.7	Massive	34.3			
125–150	8.3	1.52	10YR 3/1	77.4	Massive	52.8	35.1	1.08	0.19

^aMeasured in a 1:1 soil to water ratio.

^bStructure codes are as follows: 3: strong; 2: moderate; c: coarse; m: medium; pl: platy; /: breaking to; sbk: subangular blocky; and abk: angular blocky.

^cFC = field capacity; water content on a weight basis at 0.3 bar.

^dPWP = permanent wilting point; water content at 15 bar.

^eBulk density at 0.3 bar tension.

^fAvailable water on a volume basis measured between FC and PWP.

Source: From Soil Management Support Services.^[36]

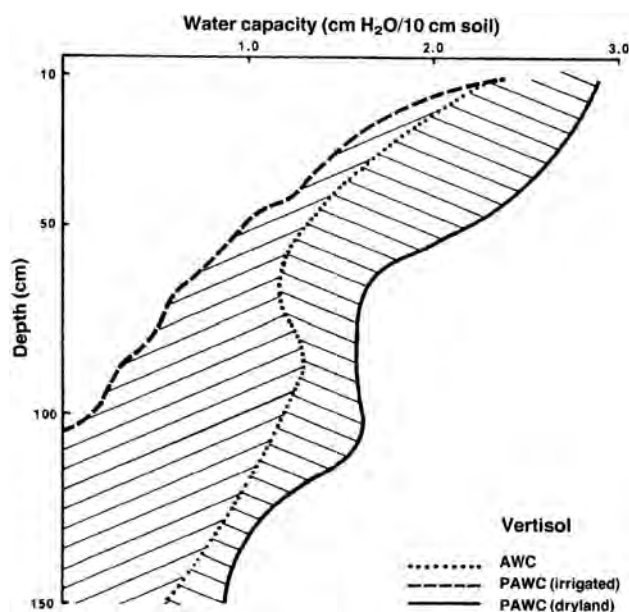


Fig. 2 Distribution with soil depths of the AWC and the PWAC at two levels of plant stress for a Typical Pellustert. **Source:** From Gardner, Coughlan, et al.^[35]

Groundwater in Vertisols?

At the bottom of cracks, infiltration may stagnate and form transient perched water tables. Groundwater tables in Vertisols cannot be defined as lateral movement of water is negligible.^[37] However, a groundwater level could develop in a more permeable substratum underlying a Vertisol. Dowling et al.^[38] report on groundwater tables at the interface of Vertisols and the underlying basalt. Groundwater tables changing with the seasons are observed in Vertisols in Israel (see Ben Hur et al.).^[39]

SOIL PROPERTIES AND THE AGRICULTURAL USE OF VERTISOLS

Soil Management and Tillage Operations

The management of Vertisols under dryland conditions is discussed by Kalane,^[40] the sustainable management of Vertisols in African countries by Syers, Penning de Vries, and Nyamudeza,^[41] and soil management of irrigated cotton by McKenzie et al.^[42]

Deep tillage of dry Vertisols requires high draft. Plowing or harrowing wet Vertisols is almost impossible. A moist soil can be tilled by hand or by animal-drawn and machine-drawn implements.

Plowing, especially deep plowing, has several disadvantages for soil quality: soil structure is destroyed,^[43] soil faunal activity (earthworms and termites) is disrupted, and the continuity of pores are reduced. This has a negative effect on water and air permeability. The use of heavy

machinery compacts wet soils and plowing them forms shear planes or plow pans.^[6,27,44,45]

The surface mulch that develops in many Vertisols after repeated wetting and drying is a great asset because it forms a natural seedbed and promotes the infiltration of rain or irrigation water. Structure regenerates by the process of self mulching. Pillai and McGarry^[46] studied clod bulk density, images of soil structure, and evaporation in samples from compacted wheel furrows. A distinct repair of soil structure was observed. Rapidity of restoration differed between crops and increased with the number of wet-dry cycles.

Soil tillage is technically difficult and costly in Vertisols, and several studies have been done of reduced tillage. Minimum, or zero, tillage and retention of stubble or fallow weeds show promise to improve soil structure, increase organic matter,^[47] and soil water storage,^[48] improve chemical fertility,^[49] optimize operation timeliness, and conserve water and soil in irrigated cotton.^[50]

The subsurface drainage systems in Vertisols have received much attention, often in connection with control of salinity and sodicity in irrigated agriculture.^[51–54]

Turpin et al.^[55] found that the combination of zero tillage and stubble retention increased the stability of soil aggregates and so increased the storage of rainfall during the fallow period. Minimum tillage in irrigated cotton produced a less dispersive soil with lower ESP and greater organic matter content in comparison with traditional plowing.^[56] McGarry, Bridge, and Radford^[57] found that no-tillage changes pore-size distribution and specifically increased the amount of continuous soil pores and counts of earthworms and termites in the upper 24 cm of soil. They related the higher hydraulic conductivity and infiltration in the no-tillage trials to this higher faunal activity.

Water Management

In the arid to semiarid areas, rain-fed cropping is strongly dependent on an efficient use of rain water. Water management in these regions must aim at maximizing transpiration and minimizing evaporation, runoff, and drainage.^[58]

In the subhumid and semiarid regions after heavy rainfall, excess water on flat terrain may cause flooding and prevent timeliness in seedbed preparations. On sloping terrain, runoff (often accompanied by soil erosion) is a danger.^[59]

Artificial internal drainage is generally not an option because of the low hydraulic activity and the internal soil movements (churning or pedoturbation) that will disrupt mole drains and break tile drains. However, successful mole and tile draining has been reported.^[45,60,61] Subsurface trenches were effective in yield increase.^[62]

Efficient use of irrigation waters requires data on plant available and readily available water.^[36] Water use regulation, or budgeting, in irrigation schemes can only be approximate: the very low hydraulic conductivity and the absence of a horizontal wetting front make the calculation

of the necessary volumes of water and timing of irrigations difficult.^[27] The final moisture addition at any specific depth approximates the original cross section of the cracks at that depth.^[32] This controls the amount of water taken by the soil at each watering and has been referred to as the “self-regulating” characteristic of the clay.

Rice cultivation under flood irrigation requires specific soil and nutrient management.^[63] Puddled fields give a higher yield than unpuddled fields, but puddling has a negative effect on the soil physical condition.^[64]

Soil Erosion

Vertisol surface soils are usually well aggregated, or there is a surface crust. Wind erosion of the dry surface soil is, therefore, unlikely. However, under extreme conditions, wind erosion does occur. Harris^[45] mentions wind erosion in southeastern Texas in Vertisols with calcareous surface soils tending to aggregate into sand-sized particles. Mermut, Patterson, and McDaniel^[4] report wind erosion from cold Vertisols in the United States and Canada. The surface granulation of soil is probably encouraged by freeze/thaw cycles in addition to drying and wetting.

Yule and Willcocks^[27] considered water erosion as the main limitation to sustain the production from Vertisols worldwide. Water erosion occurs on slopes as small as 1%. Some of the factors that make Vertisols highly susceptible to water erosion are as follows:

Much of the sediment erodes as aggregated material that has a lower bulk density than particles of a dispersed soil.

Wet Vertisols have low infiltration rates causing high energy runoff and soil particle removal and transport.^[65]

Crop or fallow weed cover, loose stubble, and roughing by harrowing help control soil erosion by water.^[66]

Slopes may also be reduced in length and percentage by the layout of graded, free-draining furrows, and thus disposing of runoff at non-eroding velocities.

Gilgai

Gilgai is a repetitive pattern of small mounds and depressions found in many Vertisols. In India and Sudan, it is restricted to the higher-rainfall regions.^[67,21] There are various forms of gilgai microrelief.^[68,69] Marked differences exist in morphological, chemical, and physical properties of soils associated with mounds and depressions and with intervening soils between them.^[70] With repeated leveling, the microrelief will eventually disappear. However, the soil profile differences in a gilgai complex (the area with a specific type of gilgai) are permanent features that continue to affect the growth of plants.^[68] Uneven growth is likely to be a consequence.

LAND USE SYSTEMS

Rain-Grown Annual Crops

The specific properties of Vertisols have prompted cultivators to devise appropriate practices. Harris^[45] considers timing of operations and water management the most important keys to successful production systems. In Australia, erosion control was the major innovation in the last several decades.^[27] Much of the cultivated land is contoured. Residue retention and reduced tillage have become standard practices.

A commonly used land layout is a system of alternating small ridges and furrows, or cambered beds separated by larger furrows (Fig. 3). The furrows not only collect rainwater but also serve as small irrigation ditches and surface drains. Examples are given by Ahmad.^[6]

In traditional cultivation, the deep Vertisols in India are fallowed during the rainy season and cropped on stored soil moisture in the post-rainy season. Weeds are controlled by tillage during the fallow period. Runoff and erosion may cause high water and soil losses. Toward the end of the rainy season, the land is prepared for planting.^[5]

A cultivation system with broad bed and furrows has the advantage over flat cultivation that it has a higher percentage of transmittent pores, as was found by Jayasree and Rao^[71] in Vertisols of India.

In the broadbed-and-furrow (BBF) system, developed at International Crops Research Institute for the Semi-Arid Tropics, Hyderabad, India, there is a rainy season crop and a post-rainy season crop.^[72,73] A cropping calendar is developed from a detailed study of the rainfall data over the years. Commonly, an early start of the rainy season crop allows a second crop, often by means of intercropping, using residual moisture. The land is laid out in semipermanent-graded BBFs on gradual slopes (usually



Fig. 3 Typical land layout for the cultivation of vegetable and small root crops. The beds are 3–4 m wide, and the drains serve for drainage or irrigation as needed.

Source: From Ahmad.^[6]

0.4–0.8%).^[74] Beds are slightly raised, acting as “mini-bunds” for moisture conservation and erosion control. The furrow is shallow but provides good surface drainage to prevent water logging of the crops growing on the bed. Excess water is drained through a system of field drains and grassed waterways. The excess water is collected in ponds and is used for complimentary irrigation of lower-lying fields.

Opportunity cropping refers to planting a crop whenever surface moisture conditions are satisfactory for seedling emergence, provided the soil moisture store equals or exceeds some specified fraction of PAWC. Carroll et al.^[50] showed that opportunity cropping, in combination with zero or reduced tillage to control episodic soil erosion, was the most appropriate system to manage the variable rainfall in Central Queensland, Australia.

Irrigated Crops

In Australia, Sudan, and other countries, cotton and several other annual crops are grown under irrigation. Fields are often laid out as cambered beds for crops alternating with furrows for flood irrigation. Irrigation gives full control of the cropping calendar and the water management in the dry season.

The quality of irrigation water is of particular importance in Vertisols. Soluble salts are not easily leached to deeper layers, and thus sodicity and salinity can develop. The Sudan Gezira irrigation scheme owes its success to high-quality irrigation water from the Blue Nile.

The watercourses and embankments of irrigation schemes are vulnerable to slumping and cracking. However, unlined irrigation canals and channels function well in the impermeable Gezira clays, provided the embankments are kept moist most of the time.

Pasture

Worldwide, most Vertisols are used for extensive rearing of livestock. The full productive potential of the soils is not realized in this land use system.^[6] The unavailability of forage, the water shortage in the dry season, and the inaccessibility during the rainy season are serious constraints for the exploitation of land, even in this non-intensive way. Overgrazing, sometimes triggering soil erosion, can be a problem near settlements. Livestock grazing caused the deterioration of soil physical properties in Ethiopian highland Vertisols.^[75]

Forestry and Agroforestry

The shortage of timber and fuelwood in many traditional farming areas on Vertisols has received increasing attention in the last decades. Shrubs and trees, which provide a crop or forage as well as wood, have been introduced in agroforestry systems. Larger woodlands have been established

with *Acacia*, *Eucalyptus*, *Casuarina*, and other species in Africa and Australia. Teak forests (*Tectona grandis*) are successful in higher-rainfall areas in Trinidad^[6] and Indonesia. An *Eucalyptus globulus*-based agroforestry system appeared to be the best option to cultivate seasonally waterlogged highland Vertisols in Ethiopia.^[76] Young^[77] gives an outline of agroforestry practices that can be effective in the control of erosion and maintenance of soil fertility.

GEOTECHNICAL PROBLEMS

Buildings and roads built on Vertisols need special precautions against the effects of swell–shrink cycles and the associated movements of the soil. Walls of houses crack and roads of any kind become uneven and intractable. Fredlund^[78] discusses the geotechnical problems in urban land use on Vertisols.

CONCLUSION

Vertisols are clay soils in which the clay fraction is dominated by swelling clay minerals, mainly smectites. Vertisol formation is favored by a warm climate with alternating wet and dry seasons. Vertisols have a characteristic morphology resulting from the swelling and shrinking of the clays upon changes in moisture content. Soil properties have a distinct impact on agricultural practice. The soils are used for rain-grown and irrigated cropping. They require specific agronomic techniques and land use systems. Minimum or zero tillage has a positive effect on soil structure and soil/water relations.

REFERENCES

1. Soil Survey Staff. *Keys to Soil Taxonomy*, 8th Ed.; U.S. Department Agriculture, Natural Resources Conservation Service: Washington, 1998; 325 pp.
2. Ahmad, N. Occurrence and distribution of Vertisols. In *Vertisols and Technologies for Their Management, Developments in Soil Science 24*; Ahmed, N., Mermut, A., Eds.; Elsevier: Amsterdam, 1996; 1–42.
3. Dudal, R.; Eswaran, H. Distribution, properties and classification of Vertisols. In *Vertisols: Their Distribution, Properties, Classification and Management*; Wilding, L.P., Puentes, R., Eds.; Texas A&M University Printing Center, College Station: Texas, 1988; 1–22.
4. Mermut, A.R.; Patterson, D.D.; McDaniel, P.A. Cold Vertisols and their management. In *Vertisols and Technologies for Their Management, Developments in Soil Science 24*; Ahmad, N., Mermut, A., Eds.; Elsevier: Amsterdam, 1996; 479–497.
5. Kutilek, M. Water relations and water management of Vertisols. In *Vertisols and Technologies for Their Management, Developments in Soil Science 24*; Ahmad, N., Mermut, A., Eds.; Elsevier: Amsterdam, 1996; 201–230.

6. Ahmad, N. Management of Vertisols in rainfed conditions. In *Vertisols and Technologies for Their Management, Developments in Soil Science 24*; Ahmad, N., Mermut, A., Eds.; Elsevier: Amsterdam, 1996; 363–428.
7. McGarry, D. The structure and grain size distribution of Vertisols. In *Vertisols and Technologies for Their Management, Developments in Soil Science 24*; Ahmad, N., Mermut, A., Eds.; Elsevier: Amsterdam, 1996; 231–259.
8. Coulombe, C.E.; Wilding, L.P.; Dixon, J.B. Overview of Vertisols: Characteristics and impacts on society. In *Advances in Agronomy*; Sparks, D.L., Ed.; Academic Press: San Diego, 1996; Vol. 751, 289–375.
9. Ohu, J.O.; Ekwue, E.J.; Folorunsa, O.A. The effect of addition of organic matter on the compaction of a Vertisol from Northern Nigeria. *Soil Technol.* **1994**, 7 (2), 155–162.
10. Bellakki, M.A.; Badanus, V.P.; Setty, R.A. Effect of long-term integrated nutrient management on some important properties of a Vertisol. *J. Ind. Soc. Soil Sci.* **1998**, 46 (2), 176–180.
11. Yuksel, O.; Kavdr, Y.; Bahtiyar, M. The effect of municipal waste compost on physical characteristics of clay soils. *Fresenius Environ. Bull.* **2004**, 13 (11a), 1094–1098.
12. Ohu, J.O.; Arku, A.Y.; Nankham, M.D.; Pishiria, L.W. Improving north-eastern Nigeria Vertisol for small-scale irrigation schemes. In *18th International Congress on irrigation and Drainage*, Montreal, Canada, 2002; 1–12.
13. Zhan, Q.-H.; Yuan, C.L.; Zhang, X.-P. Ameliorative effect and mechanism of organic materials on Vertisol. *Acta Pedologica Sinica.* **2003**, 40 (3), 420–425.
14. Venkatesh, M.S.; Hebsur, N.S.; Satyanarayana, T. Distribution of available micronutrient cations in some oilseed growing Vertisols of North Karnataka. *Karnataka J. Agr. Sci.* **2001**, 14 (3), 615–619.
15. Blanchart, E.; Albrecht, A.; Chevallier, T. The respective roles of roots and earthworms in restoring physical properties of Vertisols under a *Digitaria decumbens* pasture (Martinique, W.I.). *Agr. Ecosyst. Environ.* **2004**, 103 (2), 343–355.
16. Ahmad, N. Vertisols. In *Pedogenesis and Soil Taxonomy II. The Soil Orders, Developments in Soil Science 11 B*; Wilding, L.P., Smeck, N.E., Hall, G.F., Eds.; Elsevier: Amsterdam, 1983; 91–123.
17. Wilding, L.P.; Tessier, D. Genesis of Vertisols: Shrink–Swell phenomena. In *Vertisols: Their Distribution, Properties, Classification and Management*; Wilding, L.P., Puentes, R., Eds.; Texas A & M University Printing Center, College Station: Texas, 1988; 55–81.
18. Crescimanno, G.; Iovino, M.; Provenzano, G. Influence of salinity and sodicity on soil structural and hydraulic characteristics. *Soil Sci. Soc. Am. J.* **1995**, 59 (6), 1701–1708.
19. Hebsur, N.S.; Shedaksharappa, G.S.; Patil, C.V. Interacting influence of SAR and electrolyte concentration on the Hydraulic conductivity and dispersion of soils. *Madras Agr. J.* **1993**, 80 (4), 202–206.
20. Coulombe, C.E.; Wilding, L.E.; Dixon, J.B. Vertisols. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 2000; E 269–E 286.
21. De Vos t.N.C., J.H.; Virgo, K.J. Soil structure in Vertisols of the blue Nile clay plains, Sudan. *J. Soil Sci.* **1969**, 20, 189–206.
22. Abdulla, H.H. Vertisols in the Semiarid Region of Sudan. In *Vertisol Management in Africa*, Proceedings Workshop in Harare, Zambia, January, 16–21, IBSRAM Proceedings no. 9, 1989; 283–296.
23. Jewitt, T.N.; Law, R.D.; Virgo, K.J. Vertisols of the tropics and subtopics: their management and use. *Outlook Agr.* **1979**, 10, 33–40.
24. Blokhuis, W.A. Morphology and genesis of Vertisols. In *Vertisols and Rice Soils of the Tropics; Symposium Papers II*, Trans. 12th Intern. Congr. Soil Sci., New Delhi, 1982; Vol. 3, 23–45.
25. Blokhuis, W.A. *Vertisols in the Central Clay Plain of the Sudan*; Doctoral thesis, Agricultural University, Wageningen, 1993; XVI + 418 pp.
26. Lin, H.S.; McInnes, K.J. Water flow in clay soil beneath a tension infiltrometer. *Soil Sci.* **1995**, 159 (6), 375–382.
27. Yule, D.F.; Willcocks, T.J. Tillage and cultural practices. In *Vertisols and Technologies for Their Management, Developments in Soil Science 24*; Ahmad, A., Mermut, A., Eds.; Elsevier: Amsterdam, 1996; 261–302.
28. Bouma, J.; Loveday, J. Characterizing soil water regimes in swelling clay soils. In *Vertisols: Their Distribution, Properties, Classification and Management*; Wilding, L.P., Puentes, R., Eds.; Texas A & M University Printing Center, College Station: Texas, 1988; 83–96.
29. Virmani, S.M.; Sahrawat, K.L.; Burford, J.R. Physical and chemical properties of Vertisols and their management. In *Vertisols and Rice Soils of the Tropics; Symposium Papers II*. Trans. 12th Intern. Congr. Soil Sci., New Delhi, 1982; Vol. 3, 80–93.
30. Haider, G.; Hordofa, T.; Bekele, E. Irrigation water management for cotton on Vertisols in the middle awash region of Ethiopia. In *Management of Vertisols in Sub-Saharan Africa, Proceedings of a Conference Held at ILCA, Addis Ababa, Ethiopia, Aug 31–Sept 4, 1987*; Jutzi, S.C., Haque, I., McIntire, J., Stares, J.E.S., Eds.; ILCA: Addis Ababa, 1988; 166–182.
31. Padole, V.R.; Kalane, R.L.; Agarkar, S.C. Studies on relationships of moisture retention with soil texture and clay mineralogy of black soils. *Ann. Plant Physiol.* **1996**, 10 (2), 212–214.
32. Farbrother, H.G. Supplementary irrigation. In *Management of Vertisols Under Semiarid Conditions*, Proceedings of the First Regional Seminar, Kenya, December, 1–6, 1986, IBSRAM, 1987; 267–282.
33. Gardner, E.A.; Shaw, R.J.; Smith, G.D.; Coughlan, K.J. Plant available water capacity: concept, measurement and prediction. In *The Properties and Utilization of Cracking Clay Soils*, Proceedings of a Symposium held at the University of New England, Armidale, New South Wales, Australia, Aug 24–28, 1981; McGarity, J.W., Hoults, E.H., So, H.B., Eds.; Reviews in Rural Science 5, University of New England: Armidale, 1984; 164–175.
34. Coughlan, K.J.; Smith, G.D.; Yule, D.F. Soil physical research for improved dryland crop production on Vertisols in Queensland, Australia. In *Management of Vertisols for Improved Agricultural Production; Proceedings of an IBSRAM Inaugural Work Shop*, February 18–22, 1985; ICRI-SAT Center, Patancheru, A.P. 502324, India; ICRISAT: Patancheru, India, 1989; 87–99.

35. Gardner, E.A.; Coughlan, K.J.; Silburn, D.M. Soil water measurement and management on Vertisols in Queensland, Australia. In *Management of Vertisols in Sub-Saharan Africa, Proceedings of a Conference Held at ILCA, Addis Ababa, Ethiopia, Aug 31–Sept 4, 1987*; Jutzi, S.C., Haque, I., McIntire, J., Stares, J.E.S., Eds.; ILCA: Addis Ababa, 1988; 131–165.
36. Kirchhof, G.; Smith, P.; Hyson, L. Variation in plant-available water and hydraulic conductivity along transects of different textural soils. In *Proceedings of the 5th International Conference on Precision Agriculture*, Bloomington, MN, Jul 16–19, 2000, USDA, 2001; 1–16.
37. Soil Management Support Services, USDA-SCS, Soil Survey Administration, Democratic Republic of the Sudan. *Tour Guide*. In Fifth International Soil Classification Workshop. Soil Survey Administration: Khartoum, Sudan, 1982.
38. Blokhuis, W.A. Relationships between morphological properties and drainage in Vertisols. In *Taxonomy and Management of Vertisols and Aridisols*, Proceedings of 5th International Soil Classification, Workshop, Sudan, Nov 2–11, 1982; Part I: Papers; Soil Survey Administration: Khartoum, Sudan, 1982; 231–242.
39. Dowling, A.J.; Yule, D.F.; Lisle, A.T. The salinization of Vertisols with shallow water tables in the emerald irrigation area. In *The Properties and Utilization of Cracking Clay Soils*, Proceedings of a Symposium held at the University of New England, Armidale, New South Wales, Aug 24–28, 1981; McGarity, J.W., Hault, E.H., So, H.B., Eds.; Reviews in Rural Science 5, University of New England: Armidale, 1984; 285–291.
40. Ben Hur, M.; Li, F.H.; Keren, R.; Ravina, I.; Shalit, G. Water and salt distribution in a field irrigated with marginal water under high water table conditions. *Soil Sci. Soc. Am. J.* **2001**, *65* (1), 191–198.
41. Kalane, R.G. Management of slowly permeable soils under dryland conditions. In *Agrotechnology for Dryland Farming*; Dhopte, A.M., Ed.; Scientific Publishers: Jodhpur, 2002; 101–115.
42. Syers, J.K.; Penning de Vries, F.W.T.; Nyamudeza, P. *The Sustainable Management of Vertisols*; CABI Publishing: Wallingford, 2001; xiv + 304 pp.
43. McKenzie, D.C.; Shaw, A.J.; Rochester, I.J.; Hulugalle, N.R.; Wright, P.R. *Soil and Nutrient Management for Irrigated Cotton*; AGFACTS, NSW Agriculture: Orange, Australia, 2002. P 5.2.6., 40 pp.
44. Daniells, I.; Larsen, D.L.; McKenzie, D.C.; Anthony, D.T. W. SOILpak: A successful decision support system for managing the structure of Vertisols under irrigated cotton. *Aust. J. Soil Res.* **1996**, *34* (6), 879–889.
45. Mahmoud, M.A. Rainfed agriculture and cropping systems on Vertisols in Sudan. In *Management of Vertisols in Sub-Saharan Africa, Proceedings of a Conference Held at ILCA, Addis Ababa, Ethiopia, Aug 31–Sept 4, 1987*; Jutzi, S.C., Haque, I., McIntire, J., Stares, J.E.S., Eds.; ILCA: Addis Ababa, 1988; 335–336.
46. Harris, B.L. Management of Cropland Vertisols in Texas. In *Vertisol Management in Africa*, Proceedings of a Workshop in Harare, Zambia, Jan, 16–21, 1989, IBSRAM Proceedings, no. 9, 1989; 253–267.
47. Pillai, U.P.; McGarry, D. Structure repair of a compacted Vertisol with wet-dry cycles and crops. *Soil Soc. Am. Proc.* **1999**, *63* (1), 201–210.
48. Blair, N.; Crocker, G.J. Crop rotation effects on soil carbon and physical fertility of two Australian soils. *Aust. J. Soil Res.* **2000**, *38* (1), 71–84.
49. O'Leary, G.J.; Connor, D.J. Stubble retention and tillage in a semiarid environment: 1. Soil water accumulation during fallow. *Field Crops Res.* **1997**, *52* (3), 209–219.
50. Chan, K.Y.; Hulugalle, N.R.; Arshad, M.A. Changes in some soil properties due to tillage practices in rainfed hard-setting alfisols and irrigated Vertisols of eastern Australia. *Soil Till. Res.* **1999**, *53* (1), 49–57.
51. Carroll, C.; Halpin, M.; Burger, P.; Bell, K.; Sallaway, M.M.; Yule, D.F. The effect of crop type, crop rotation and tillage practice on runoff and soil loss on a vertisol in central queensland. *Aust. J. Soil Res.* **1997**, *35* (4), 925–939.
52. Bharambe, P.R.; Shelke, D.K.; Jadhav, G.S.; Vaishnav, V.G.; Oza, S.R. Management of salt-affected Vertisols with sub-surface drainage and crop residue incorporation under soybeans-wheat cropping system. *J. Ind. Soc. Soil Sci.* **2001**, *49* (1), 24–29.
53. Jayawardana, N.S.; Biswas, T.K.; Blackwell, J.; Cook, F.J. Management of salinity and sodicity in a land FILTER system for treating saline waste water on a saline-sodic soil. *Aust. J. Soil Res.* **2001**, *39* (6), 1247–1258.
54. Rana, J.S.; Sewa Ram; Rao, K.V.G.K. Effect of subsurface drainage on soil salinity and crop yields in Vertisols of Chambal Command. In *Role of drainage and challenges in 21st century, Proceedings of the 8th ICID, International Drainage Workshop*; New Delhi, India, 31/1-4/2/2000; 2000, Vol. I, 389–401.
55. Kumar, H.R.; Manjunatha, M.V.; Hebbaraa, M.; Patil, S.G. Studies on the performance of sub-surface drainage system in saline Vertisols. *Karnataka J. Agr. Sci.* **2001**, *14* (4), 1010–1014.
56. Turpin, J.E.; Bridge, B.J.; Orange, D.; Thompson, D.J. Water and bromide movement in a Vertisol under four fallow management systems. *Aust. J. Soil Res.* **1999**, *37* (1), 75–89.
57. Hulugalle, N.R.; Entwistle, P. Soil properties, nutrient uptake and crop growth in an irrigated Vertisol after nine years of minimum tillage. *Soil Till. Res.* **1997**, *42* (1–2), 15–32.
58. McGarry, D.; Bridge, B.J.; Radford, B.J. Contrasting soil physical properties after zero and traditional tillage of an alluvial soil in the semiarid subtropics. *Soil Till. Res.* **1999**, *53* (2), 105–115.
59. Yule, D.F. Water Management of Vertisols in the Semiarid Tropics. In *Management of Vertisols Under Semi-Arid Conditions*, Proceedings of the First Regional Seminar, Kenya, December, 1–6, 1986, IBSRAM, 1987; 107–123.
60. Silburn, D.M.; Glanville, S.F. Management practices for control of runoff losses for cotton farmers under storm rainfall. I. runoff and sediment on a black Vertisol. *Aust. J. Soil Res.* **2002**, *40* (1), 1–20.
61. Muirhead, W.A.; Humphreys, E.; Jaywardane, N.S.; Moll, J.L. Shallow subsurface drainage in an irrigated Vertisol with a perched water table. *Agr. Water Manag.* **1996**, *30* (3), 261–282.

62. Holsambre, D.G.; Varade, S.B.; Acharya, H.S.; Rapte, S.L. Drainage characteristics of Vertisols. *J. Ind. Soc. Soil Sci.* **1982**, *40*, 116–121.
63. Beyene, D.; Regassa, H. Research work on the management of Vertisols in Ethiopia: experience of the institute of agricultural research. In *Vertisol Management in Africa*, Proceedings of a Workshop in Harare, Zambia, January, 16–21, 1989, IBSRAM Proceedings, No. 9, 1989; 165–171.
64. Kondo, M.; Ota, T.; Wanjogu, R. Physical and chemical properties of Vertisols and soil nutrient management for intensive rice cultivation in the Mwea area in Kenya. *Jpn J. Trop. Agr.* **2001**, *45* (2), 126–132.
65. Mohanty, M.; Painuli, D.K. Land preparatory tillage effect on soil physical environment and growth of yield of rice in a Vertisol. *J. Ind. Soc. Soil Sci.* **2003**, *51* (3), 217–222.
66. Freebairn, D.M.; Loch, R.J.; Silburn, D.M. Soil erosion and soil conservation for Vertisols. In *Vertisols and Technologies for Their Management, Developments in Soil Science 24*; Ahmad, N., Mermut, A., Eds.; Elsevier: Amsterdam, 1996; 303–362.
67. Sharratt, B.S.; Lindstrom, B.J.; Benoit, G.R.; Young, R.A.; Wilts, A. Runoff and soil erosion during spring thaw in the northern U.S. Corn belt. *J. Soil Water Conserv.* **2000**, *55* (4), 487–494.
68. Sehgal, J.L.; Bhattacharjee, J.C.J. Typic Vertisols of India and Iraq, their characterization and classification. *Pédologie* **1988**, *38*, 67–95.
69. Probert, M.F.; Fergus, I.F.; Bridge, B.J.; McGarry, D.; Thompson, C.H.; Russell, J.S. *The Properties and Management of Vertisols*; CAB International (IBSRAM): Oron, 2001; 36 pp.
70. Hubble, G.D.; Isbell, R.F.; Northcote, K.H. Features of Australian soils. In *Soils: An Australian Viewpoint*; The Division of Soils CSIRO, Ed.; CSIRO, Melbourne; Academic Press: London, 2001; 17–47.
71. Thompson, C.H.; Beckmann, G.G. Gilgai in Australian black earth and some of its effects on plants. *Trop. Agri. (Trinidad)* **1982**, *59*, 149–156.
72. Jayasree, G.; Rao, Y.N. Effect of different long-term management practices on micro-structure of Vertisols and Alfisols. *J. Ind. Soc. Soil Sci.* **2002**, *50* (4), 452–456.
73. Virmani, S.M.; Rao, M.R.; Srivastava, K.L. Approaches to the management of Vertisols in the semiarid tropics: The ICRISAT experience. In *Management of Vertisols for Improved Agricultural Production, Proceedings of an IBSRAM Inaugural work shop*, Feb 18–22, 1985; ICRISAT Center, Patancheru, A.P. 502 324, India; ICRISAT: Patancheru, 1989; 17–33.
74. Kampen, J. An approach to improved productivity on deep Vertisols. In ICRISAT Information Bulletin No. 11, India ICRISAT: Patancheru, July 1982; 14 pp.
75. Swindale, L.D. Developing, testing and transferring improved Vertisol technology: The Indian Experience. In *Management of Vertisols in Sub-Saharan Africa*, Proceedings of a Conference Held at ILCA, Addis Ababa, Ethiopia, Aug 31–Sept 4, 1987; Jutzi, S.C., Haque, I., McIntire, J., Stares, J.E.S., Eds.; ILCA: Addis Ababa, 1988; 13–43.
76. Taddese, G.; Mohamed Saleem, M.A.; Ayalneh, W. Effect of livestock grazing on physical properties of a cracking and self-mulching Vertisol. *Aust. J. Exp. Agr.* **2002**, *42* (2), 129–133.
77. Kidanu, S. *Using Eucalyptus for Soil and Water Conservation on the Highland Vertisols of Ethiopia*; Doctoral Thesis Wageningen University: Wageningen, The Netherlands, 2004.
78. Young, A. The potential of agroforestry for soil conservation—with special reference to Vertisols. In *Management of Vertisols Under Semiarid Conditions*, Proceedings of the First Regional Seminar, Dec 1–6, 1986, IBSRAM: Oron, 1987; 187–199.
79. Fredlund, D.G. Geotechnical problems associated with swelling clays. In *Vertisols and Technologies for Their Management; Development in Soil Science 24*; Ahmad, N., Mermut, A., Eds.; Elsevier: Amsterdam, 1996; 499–524.

Vertisols: Surface Crack Management

J. Somasundaram

Nishant K. Sinha

Ashok K. Patra

Indian Council of Agricultural Research, Indian Institute of Soil Science, Bhopal, India

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

High smectite clay causes the formation of cracks that are a striking feature of black Vertisols, which are also distinguished by the presence of potholes. Soil cracks improve water recharge through preferential flow during the rainy season. A network of surface and subsurface cracks is also a prerequisite to the formation of a pothole. Cracks also extend the soil–air interface in the soil profile. Thus, the evaporation loss is high from surface and subsurface cracks, which also increase the number of wetting/drying cycles. Loss of soil moisture in rainfed regions can severely jeopardize crop productivity. Thus, there exists a strong need to understand the cracking patterns in Vertisols for the management of cracks and to minimize loss of profile soil moisture. Cracking patterns can be altered by management practices such as tillage, addition of soil amendments, and different crop covers/cropping systems. Despite their importance, scientific knowledge on managing surface cracks in Vertisols is very scanty. Different management practices, namely conservation tillage, residue retention, and addition of soil amendments, were effective in reducing crack formation in Vertisols. This entry lucidly revolves around the surface cracks and its management in Vertisols.

INTRODUCTION

The world's soil resource has been broadly classified into 12 distinct soil orders. Out of these, Vertisols (Latin *verto*, “turn”), which cover ~2.4% of global ice-free land areas, are uniquely characterized with a high content of shrinking–swelling clay, dark colored, and prominent surface cracking varying with moisture contents. Cracks are the unique feature of Vertisols and are used as one of the criteria in defining Vertisols in Soil Taxonomy.^[1] When Vertisols dry, mainly the following four shrinkage phases are possible: structural shrinkage, basic or normal shrinkage, residual shrinkage, and zero shrinkage leading to the formation of cracks. Basic shrinkage, traditionally referred to as normal shrinkage, accounts for most of the shrinkage in agricultural soils, generally between soil water potentials of -33 to < -1500 kPa.^[2] In this shrinkage phase, water is lost from the interclay domains causing shrinkage to be equidimensional resulting in crack formation. The soil blocks between the cracks contain all the structural and residual porosities, whereas the cracks (including the cracks above the soil surface) contain only the basic or normal shrinkage porosity.^[3]

DIFFERENT TYPES OF CRACKS IN VERTISOLS

Grossman et al.^[4] have grouped cracks in Vertisols into the following three sets:

1. Vertically oriented cracks that outline large blocks or prisms at the upper part of the soil. The cracks are wide, about 5–10 mm, and become progressively deeper as the soil dries out.
2. Cracks that form angular or blocky elements at the soil surface. These form at high water tensions, perhaps close to the wilting point.
3. Cracks that form deeper in the soil and are related to the internal pedoturbation associated with the slickensides.

The first two sets of cracks exhibit properties that are important in land use and management. Vertisols with a granular surface soil mulch (the first set) tend to have lower bulk densities, perhaps due to a slightly higher organic matter content and to the space between the granules. Soils with angular surface structure (the second set) are easier to till, and roots can permeate the spaces and move deeper. In addition, the filled crack spaces are probably the most

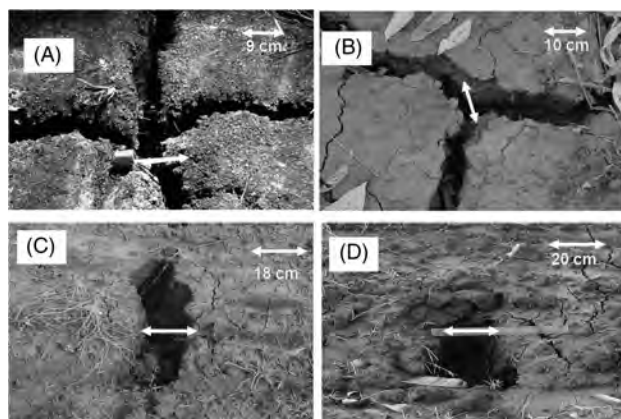


Fig. 1 Surface black soil showing (A and B) intersecting cracks and (C and D) potholes.

likely areas for roots to establish during the next season because water flows easily through these areas.

Cracks have several direct as well as indirect effects on crop performance. Because the rhizosphere is dehydrated last, the cracks normally form away from the stubble of the previous crop that locates at the center of the polygon. In this case, dislodging of the plant is not a problem, but when the rhizosphere also dries out, soil shrinkage could strangle or shred crop roots. Cracks also retard surface wetting from any off-season rains. At the beginning of the rainy season, much of the water is not available to the plants because the water is rapidly evacuated by the void system. During the initial rain showers, the subsoil below the zone of the cracks is moistened. Successive rains moisten the top few centimeters of the soil, causing it to swell and seal the surface. Subsequent rains cause ponding, making tillage difficult, and facilitate faster soil erosion.^[5] Soil cracks may improve water recharge through preferential flow during the rainy season. A network of surface and subsurface cracks (Fig. 1A and B) also leads to pothole formation (Fig. 1C and D). The size of the cracks gets larger when two or more such cracks meet. Consequently, the runoff drains into these cracks, where a deep and wide hole is formed during the rainy season. Such holes are locally known as potholes or pipe holes. In this process, the top soil at the edge of the

cracks gets slumped down into the crack, resulting in the formation of potholes. Through subsurface cracks/channels, the eroding soil may join somewhere with the adjoining ravine systems posing a serious problem of gully head extension that results in the loss of arable land and crop productivity.^[6] Cracks also extend the soil–air interface in the soil profile. Consequently, the evaporation loss is high from surface and subsurface cracks, which also increases the number of wetting/drying cycles. Loss of soil water in rainfed regions can severely jeopardize crop productivity. Thus, the management of cracks in Vertisols is inevitable for sustainable productivity in these areas.

MANAGEMENT OF SURFACE CRACKS

Different agronomic practices such as tillage systems and nutrient management indirectly affect the cracks formation, by directly affecting soil water content and soil organic matter. In India, a systematic survey was conducted on cracks/pothole dimensions (length, width, and depth) under various land conservation treatments, namely contour furrows (CFs), conservation bench terraces (CBTs), residue (wheat straw) incorporation (RI), control (without conservation measures), during dry spells after the onset of the monsoon under various field treatments.^[7] The greater volume of potholes was observed in uncultivated fields than in cultivated ones (Table 1). It was due to the unbroken, gentle slopes that allowed silt-laden runoff to travel longer distances before entering cracks to form potholes. The data further reveal that among the cultivated fields, the control field had the greatest volume of potholes followed by CBT, RI, and CF. The CBT treatment had a large volume of potholes that is attributed to gentle terrace slopes (1%), which results in greater runoff into the cracks and leads to more pothole formation. In the RI treatment, the volume of potholes was less than that of the CBT, and this is as a result of better soil aggregation and porosity, which offer better protection against sudden slumping or erosion into potholes. The lower volume of potholes in the CF treatment is explained by the higher water content in the profile because of greater absorption of rainfall across the field. This shows that conservation measures are very effective in

Table 1 Pothole dimensions under different treatments in Vertisols (Typic Chromustert).

S. No.	Treatments	Length (m)	Width (m)	Depth (m)	Volume (m ³ ha ⁻¹)
I	Cultivated field				
	CF	0.24 ± 0.04 (34.22)	0.20 ± 0.02 (22.36)	0.17 ± 0.02 (25.14)	119.7
	CBTs	0.18 ± 0.02 (22.70)	0.19 ± 0.02 (24.97)	0.46 ± 0.04 (20.91)	231.8
	Residue incorporated	0.29 ± 0.03 (21.03)	0.17 ± 0.02 (17.32)	0.22 ± 0.03 (22.77)	142.8
	Control (cultivated)	0.32 ± 0.03 (19.60)	0.27 ± 0.05 (20.10)	0.39 ± 0.06 (28.61)	450.1
II	Uncultivated field	0.58 ± 0.05 (18.89)	0.41 ± 0.06 (32.72)	0.31 ± 0.09 (62.59)	573.3

Note: ±, standard error; values in parentheses are coefficient of variation.

Source: Adapted from Somasundaram, Singh, et al.^[7] ©2011 Wiley & Sons.

reducing crack/pothole formation in the black soils of southeastern Rajasthan, India. It is also possible that cultivated fields had a smaller volume of potholes than uncultivated fields because of regular ploughing that mixed accumulated clay in the lower subsoil. In uncultivated fields, a higher amount of clay accumulates that could have resulted in a greater occurrence of potholes.

Nevertheless, potholes and cracks are omnipresent in these black Vertisols. The application of manures and amendments such as crop residues helps to minimize cracks by promoting soil aggregation. The creation of surface dust mulch also reduces cracks, which helps successful crop growth. Tillage and crop rotation directly affect soil structure. Thus, the selection of appropriate tillage practices is imperative for these black soils to minimize the occurrence of soil cracks. Clay accumulates where cracks intersect (slickensides) and also accumulates in the subsurface soil at 60–70 cm. Enormous pressure is developed as a result of swelling (when wet) and shrinking with wide cracks formed during the dry season. Ploughing immediately after the harvest during the postrainy season helps to prevent weeds from producing seeds, exposes hibernating insects and disease-causing organisms, improves soil fertility, fills cracks, and pulverizes the soil, thereby improving moisture retention. Intercropping of cereals and pulses such as in the sorghum/pigeon pea system is effective in controlling crack

formation because of the contrasting root patterns, which aids aggregate formation and the breakdown of the subsurface hardpan. Selection of short-duration crops that complete their dough stage (grain filling stage) before crack development is another option to minimize the problem. *In situ* moisture conservation practices, such as construction of contoured or graded bunds, also help reduce crack formation due to changes in the moisture regime.

Worldwide, several studies came into conclusion that conservation agricultural practices such as no-tillage, reduced tillage, and residue retention on surface, helps in increasing profile moisture and soil organic carbon. High soil carbon facilitates soil aggregation and improves micro-environment for soil microorganism and macroporosity, which resulted in high soil profile moisture and reduces possibilities of crack formation under conservation agriculture practices compared to traditional practices. In contrast, higher crack volume is found in no-tillage system compared to conventional tillage system (Fig. 2A).^[8] Observations were recorded under soybean–wheat cropping system in month of November after 2 years of imposition of tillage treatments. Possibilities of higher crack volume in no-tillage system were due to no residue generation in soybean crop, which left soil bare to evaporation. However, in another study, lower volume of cracks was observed in no-tillage compared to conventional tillage system.^[9]

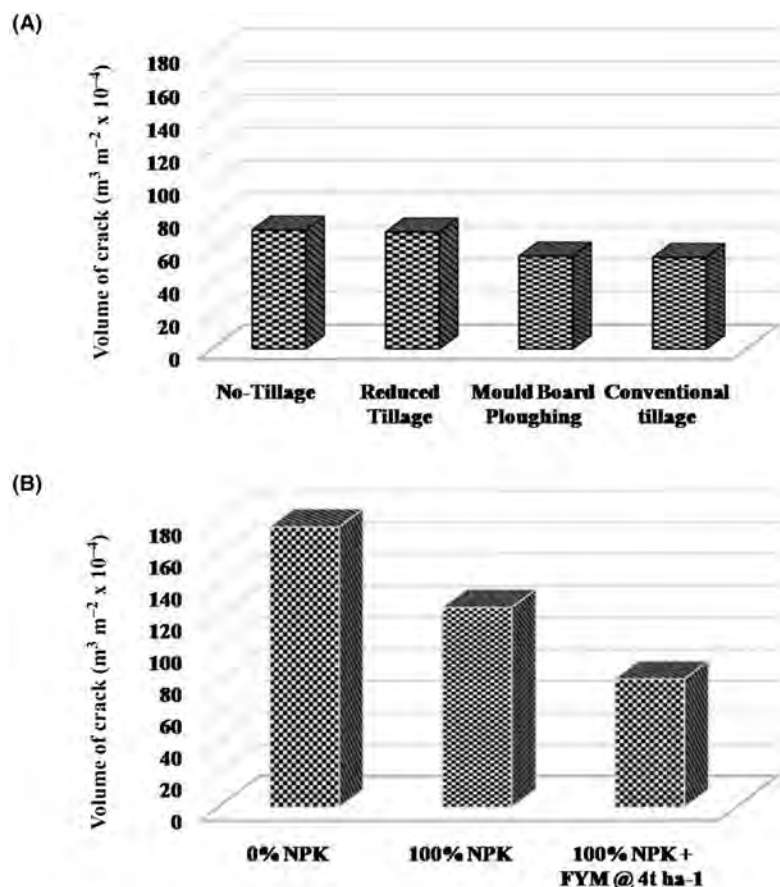


Fig. 2 Crack volume under different (A) tillage systems in soybean–wheat rotation and (B) nutrient management in soybean–linseed rotation.

Source: Redrawn from Bandyopadhyay, Mohanty, et al.^[8] ©2003 Elsevier B.V.

Table 2 Impact of crop covers on crack volume in Vertisols (Typic Haplustert).

Treatment	Crack volume ($\text{m}^3 \text{ ha}^{-1}$)
Soybean	22.5
Maize	25.0
Pigeon pea	27.4
Soybean + maize (1:1)	27.5
Soybean + pigeon pea (2:1)	28.4
Maize + pigeon pea (1:1)	27.4
Cultivated fallow	33.3
CD ($P = 0.005$)	NS

Note: NS, Non-Significant; CD, Critical Difference.

Source: Adapted from Singh, Chaudhary, et al.^[11] ©2014 IASWC, Dehradun.

Among different nutrient management practices, results indicated that the least cracks in 100% nitrogen, phosphorus, and potassium (NPK) + farm yard manure (FYM) at 4 t ha^{-1} . The effects of nutrients on crack volume may be attributed to the fact that NPK + FYM registered significantly greater root development than other nutrients treatments.^[8] The root produced might have reduced the shrinkage process by anchoring the soil mass.^[10] Crack volume was also recorded under different crop covers after two crop cycles (Table 2), which revealed that higher crack volume was registered under cultivated fallow.^[11]

In soil science, amendments are frequently used to reclaim soil; to improve soil physical, chemical, and biological health; and to promote plant growth and vigor. A soil amendment can be any material added to a soil with an aim to provide a better environment for plant roots. Commonly used amendments in area of soil science are the residual materials or by-products from other processes and include municipal biosolids, manures, litters, wood ash or saw dust, coal combustion products, neutralizing lime products, compost, and traditional agricultural fertilizers. Of these, five soil amendments, namely wheat straw (S1), FYM (S2), sawdust (S3), gypsum (S4), and fly ash (S5), with a control (S0) were selected to conduct a preliminary study at Indian Council of Agricultural Research-Indian

Institute of Soil and Water Conservation (formerly ICAR-CSWCRTI), India, on crack occurrences.^[12] Amendments were mixed into the soils at the rate of 0, 2.5, 5.0, and 10 g kg^{-1} and kept in plastic trays (Fig. 3). These trays were kept in shade without any disturbances for monitoring the crack development. It was evident from the study that the addition of soil amendments was very effective in reducing the crack formation/volume. Irrespective of the soil amendments, increased rate of application of soil amendments had significant effect on reduction of crack volume (Fig. 4). Among various amendments studied, wheat straw recorded the lowest crack volume followed by FYM, sawdust, gypsum, and fly ash.^[12]

Similarly, soil moisture content also recorded higher value under wheat residue amended soil at 10 g kg^{-1} of soil compared to other treatments and control. Data on percent decrease of crack volume in amendments added at 2.5 g kg^{-1} of soil over control have revealed that the highest reduction (-74%) was observed under wheat straw, and the lowest reduction (-23%) was registered under fly ash. Beneficial effect of residue addition on the reduction of crack parameters was also reported.^[13,14] As the dose of amendments increased (i.e., from 2.5 to 10 g kg^{-1} of soil), the percent reduction was also greatly increased. Results indicated that the addition of soil amendments results in reduction of crack volume. Based on the results obtained, a follow-up research was carried out on experimental farm, and the result indicated that all amendments reduced crack volume in cropped and fallow plots during winter (Table 3). Wheat straw being most effective during winter months, however, no effect of amendments was observed during summer. Use of amendments improved crop yields. Among different soil amendments studied, wheat straw was most effective in improving grain (40%) and biomass yields (33%). Cost-benefit analysis showed that FYM and fly ash were the only remunerative options, which increased returns by 123% and 43%, respectively. Use of amendments was not economical for grass production; however, fly ash may be recommended, if cracking behavior of soils is required to be managed.

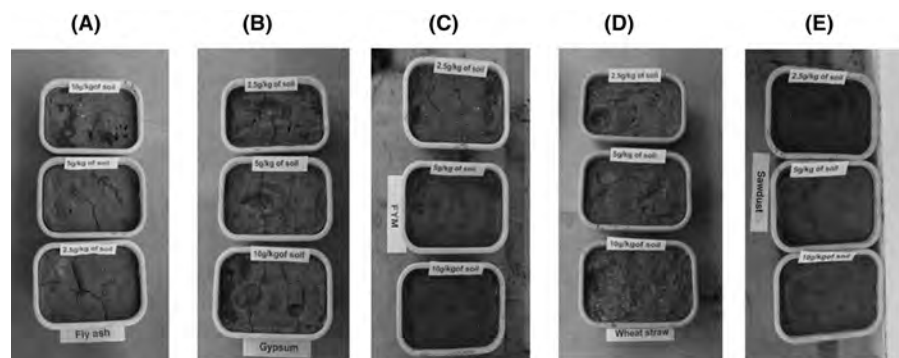


Fig. 3 Cracking as influenced by different soil amendments: (A) fly ash, (B) gypsum, (C) FYM, (D) sawdust, and (E) wheat straw in medium-deep black Vertisols.

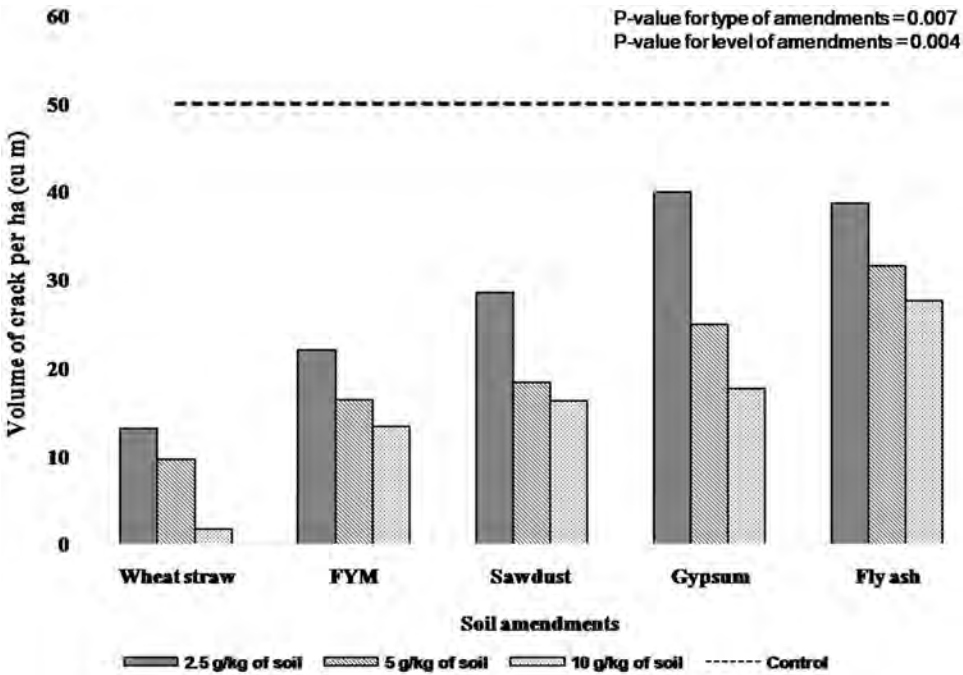


Fig. 4 Volume of crack as influenced by different soil amendments.

CONCLUSIONS

Cracking is a striking phenomenon of Vertisol and has several positive and negative effects on agricultural soils. The positive effects of cracks include improved drainage, improved infiltration, and improved soil structure, whereas the negative effect of cracking is the rapid transport of water with solutes through soil cracks, which often leads

to crop water and nutrient stress and sometimes leads to groundwater contamination. Therefore, suitable land use with good soil management practices such as adopting conservation tillage, leaving crop residue on soil surface, using different soil amendments, and selecting appropriate crops in the crop cycle would be the best option to reduce the formation of wide cracks. Moreover, it also helps in preventing loss of fertile topsoil through potholes during the

Table 3 Effect of soil amendments application on crack volume under different land uses.

		Crack volume (m ³ ha ⁻¹)			
Land uses	Soil amendments	October 23, 2008	January 2, 2009	March 3, 2009	May 11, 2009
Agriculture	Control	28.55	137.53	161.94	245.29
	Fly ash	25.44	82.74	141.42	208.01
	Wheat straw	19.02	45.68	108.44	201.60
	Gypsum	20.43	47.94	123.95	215.22
	FYM	24.61	91.80	163.97	227.42
Grassland	Control	31.33	192.02	232.62	250.62
	Fly ash	27.67	71.21	155.89	140.05
	Wheat straw	20.77	73.02	123.78	140.52
	Gypsum	32.70	104.15	98.29	233.24
	FYM	27.63	87.32	130.40	257.49
Fallow land	Control	28.02	149.65	171.40	246.69
	Fly ash	19.19	117.84	141.26	190.60
	Wheat straw	14.18	82.70	119.91	211.79
	Gypsum	19.75	80.24	130.34	242.06
	FYM	21.10	94.92	136.68	220.11

Source: Adapted from Annual Report, ICAR-Indian Institute of Soil and Water Conservation (formerly ICAR-CSWCRTI) © Director, ICAR-IISWC.^[15]

rainy season and also in preventing water loss through evaporation in the post-rainy season, thus increasing moisture availability to the next crop.

ACKNOWLEDGMENT

We express our gratitude and sincere thanks to Dr. V.N. Sharda, Member (ASRB) & Former Director (ICAR-IISWC), Dehradun; Dr. A. Subba Rao, Former Director (ICAR-IISS), Bhopal, India, for providing scientific facilities and guidance.

REFERENCES

1. Soil Survey Staff. *Keys to Soil Taxonomy*, 11th Ed.; USDA-Natural Resources Conservation Service: Washington, 2010.
2. Yule, D.F.; Ritchie, J.T. Soil shrinkage relationships of Texas Vertisols. I. Small cores. *Soil Sci. Soc. Am. J.* **1980**, *44*, 1285–1291.
3. McGarry, D.; Yule, D.F. Soil shrinkage. In *Encyclopedia of Soil Science*; Lal, R., Ed.; Marcel Dekker: New York, 2002; 1197–1200.
4. Grossman, R.B.; Nettleton, W.D.; Brasher, B.R. Application of pedology to plan response prediction for tropical vertisols. Proceedings of the Fifth International Soil Classification Workshop; Nov 2–11, 1982, Soil Survey Administration: Sudan, 1982; 97–116.
5. Jutzi, S.C.; Haque, I.; McIntire, J.; Stares, J.E.S., Eds. Management of Vertisols in Sub-Saharan Africa. In *Proceedings of a Conference held at ILCA, Addis Ababa, Ethiopia*, Aug 31–Sept 4, 1987; ILCA: Addis Ababa, 1988.
6. Somasundaram, J.; Chaudhary, R.S.; Lakaria, B.L.; Saha, R.; Sinha, N.K.; Singh, R.K.; Jha, P.; Singh, R.K.; Subba Rao, A. Pothole formation and occurrence in black vertisols of central and Western India. *Agr. Res.* **2014**, *3* (1), 87–91.
7. Somasundaram, J.; Singh, R.K.; Prasad, S.N.; Sethy, B.K.; Kumar, A.; Ramesh, K.; Lakaria, B.L. Management of black vertisols characterized by pot-holes in the Chambal region, India. *Soil Use Manage* **2011**, *27*, 124–127.
8. Bandyopadhyay, K.K.; Mohanty, M.; Painuli, D.K.; Misra, A.K.; Hati, K.M.; Mandal, K.G.; Ghosh, P.K.; Chaudhary, R.S.; Acharya, C.L. Influence of tillage practices and nutrient management on crack parameters in a vertisol of central India. *Soil Till. Res.* **2003**, *71*, 133–142.
9. Munoz-Romero, V.; López-Bellido, L.; López-Bellido, R.J. The effects of the tillage system on chickpea root growth. *Field Crop. Res.* **2012**, *128*, 76–81.
10. Mitchell, A.R.; Van Genuchten, M.T. Shrinkage of bare and cultivated soil. *Soil Sci. Soc. Am. J.* **1992**, *56*, 1036–1042.
11. Singh, R.K.; Chaudhary, R.S.; Somasundaram, J.; Rashmi, I.; Hati, K.M.; Sinha, N.K.; Subba, R.A. Impact of crop covers on soil properties, runoff, soil-nutrients losses and crop productivity in vertisols of Central India. *Indian J. Soil Cons.* **2014**, *42* (2), 268–275.
12. Somasundaram, J.; Singh, R.K.; Prasad, S.N.; Kumar, A.; Sinha, N.K.; Mohanty, M.; Saha, R.; Mishra, D.M.; Mandal, D.; Sharda, V.N. Management of surface cracks in black vertisols (Typic chromusterts) by soil amendments and land uses in Chambal region, India. *Indian J. Soil Cons.* **2015** (Under Consideration).
13. Bhusan, L.; Sharma, P.K. Long-term effects of lantana (*Lantana* sp. L) residue additions on soil physical properties under rice-wheat cropping. I. Soil consistency, surface cracking and clod formation. *Soil Till. Res.* **2002**, *65*, 157–167.
14. Mohanty, M.; Painuli, D.K. Cracking of a vertisol as influenced by puddling and residue management under rice-wheat cropping system. *J. Indian Soc. Soil Sci.* **2006**, *54*, 452–460.
15. Annual Report (2009–2010). *ICAR-Indian Institute of Soil and Water Conservation (formerly ICAR-CSWCRTI)*; Annual Report, Dehradun, 2009–2010.

Vesicular Crusts

Patrick Drohan

Department of Crop and Soil Sciences, Pennsylvania State University, University Park, Pennsylvania, U.S.A.

Abstract

The vesicular horizon is a surface, or near surface, soil horizon composed of tiny non-connected circular, ovoid, or prolate pores commonly called vesicles. The horizon has been found to form in soils of a variety of parent materials deposited via eolian processes as dust and is often composed of fine textures. Vesicle pore formation is hypothesized to occur by several different processes with the most widely accepted being entrapped air and soil particle movement following soil wetting (air entrapment due to a wetting front from rain or snowmelt). Field vesicular porosity has been noted to occur within platy and columnar structure. While fragile and easily disturbed by most fauna and humans, vesicular horizons have been noted to persist in environments where surface stability is greatest; increasing surface stability can be due to crusts, pavements, or landscape age. The vesicular horizon has been hypothesized to restrict plant growth with one possible mechanism being that non-interconnected pores found in the horizon may restrict water movement through the horizon, especially in thicker horizons. Field research on hydraulic properties of vesicular horizons has noted significantly lower hydraulic conductivities in vesicular horizons on surfaces of increasing geomorphic age because of greater accumulations of clay. Owing to its effect on water movement into arid soils, the vesicular horizon may be the most influential soil horizon in arid landscape geomorphology and pedogenesis.

INTRODUCTION

The vesicular horizon is a surface, or near surface, soil horizon composed of tiny non-connected circular, ovoid, or prolate pores commonly called vesicles.^[1–4] In some cases, a thin (1 mm to 5 cm; Fig. 1A) horizon or silt cap may over the horizon.^[5] The vesicular horizon is found in arid environments^[1,3,6,7] and can form in association with mineral crusts, biological crusts, and stone pavements. As dating of the vesicular horizon has proved to be difficult, soil ages are believed to range from the Holocene to the Pleistocene.^[1,5,8] Vesicles in the horizon are typically less than 1 mm in diameter,^[3,7] however, in the Canadian High Arctic, vesicles up to 30 mm in diameter have been observed.^[7] The shape of vesicles can be quite variable and may be influenced by soil drainage and surface stability.^[7] Bunting^[7] hypothesizes that irregular drying, or the diffusion of air or water under pressure, may contribute to irregularly shaped vesicles.

FORMATION

Vesicle pore formation is hypothesized to occur by several different processes: 1) entrapped air and soil particle movement following soil wetting (entrapment due to a wetting front from rain or snowmelt),^[1,7] 2) heated air,^[3] 3) release of carbon dioxide via drying of carbonate soils and solution

(Paletskaya et al., 1958, as cited in Evenari et al.^[3]); and 4) ice crystal formation and melting. Several researchers have demonstrated vesicle formation in the laboratory using sieved soil from the field that originally had vesicles.^[1,3,7] Soil particle movement not only plays an important role in vesicle formation but also in the movement of rock fragments to the soil surface, which is believed to be one mechanism explaining arid lands pavement formation.^[1] Field research has confirmed this laboratory result.^[9]

Vesicular porosity has been noted to occur under irrigation furrows,^[2] in cracks and fissures of Lithosols in the southern Sinai,^[3] in tropical environments as a fossil feature where laterite soils are present,^[10] in termite mounds (Lee and Wood, 1977, as cited in Bunting^[7]), and in humid soils where earthworms are present.^[7]

STRUCTURE

Field vesicular porosity has been noted to occur within platy^[1,11] and columnar structure.^[5,8,9] Vesicular porosity has been found to quickly reform in the laboratory following disturbance^[1–3] and to more closely resemble *in situ* vesicular porosity with numerous laboratory wetting and drying cycles; repeated wetting and drying decreases the number of pores but increases the total porosity.^[12] While field vesicle shape is often ovoid, less rounded shapes have been observed with laboratory rewetting experiments.^[12]

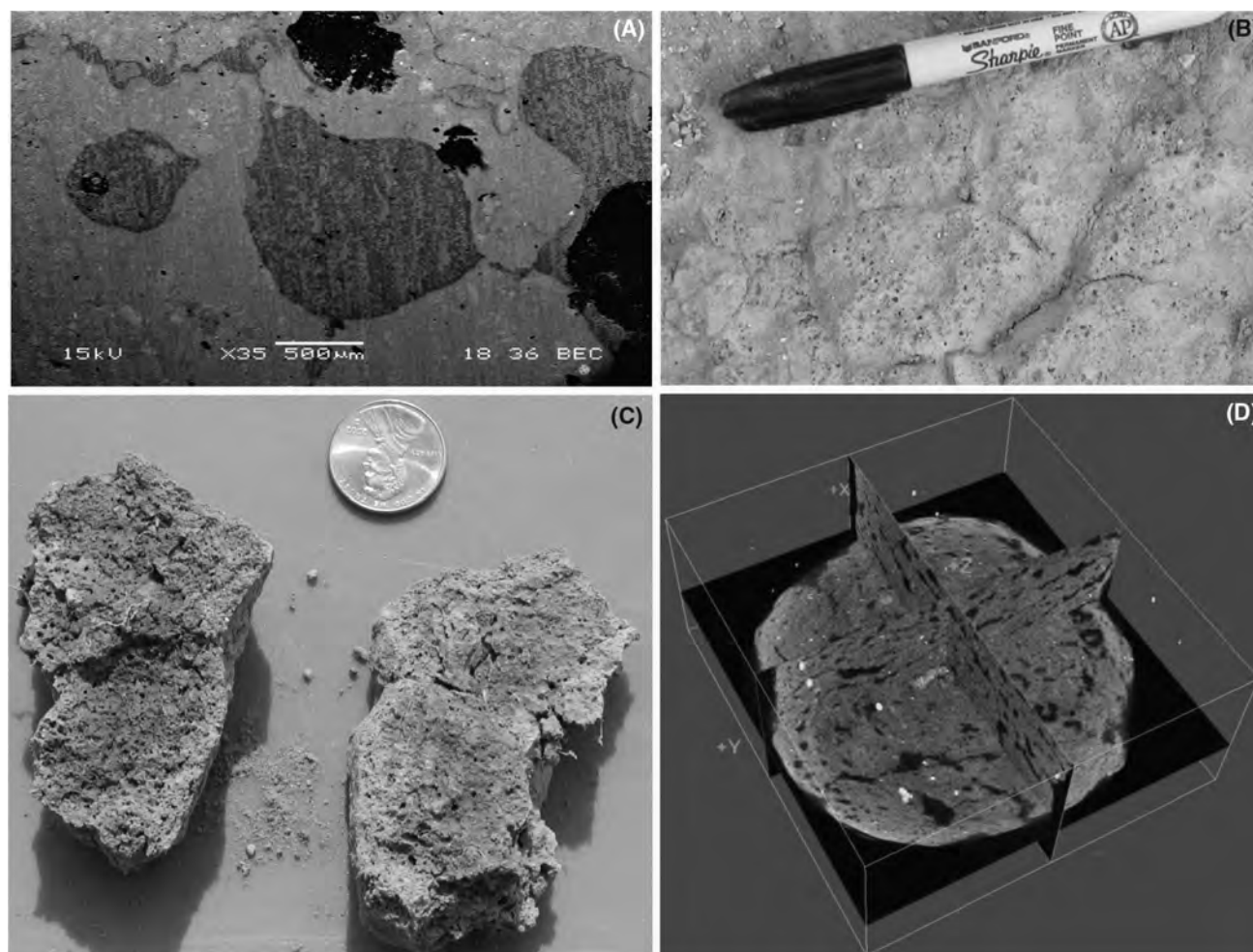


Fig. 1 (A) Scanning electron microscope image of vesicular horizon ped from Callville Wash, Lake Mead NRA, Nevada, U.S.A. (B) Vesicular horizon surface from ~75 ka soil at the Mojave National Preserve, California, U.S.A. (pavement removed). (C) Same soil, vesicular horizon ped cross section. (D) Same soil, computed tomography scan of vesicular horizon ped interior.

Source: Photo (A) courtesy of Maureen Yonovitz, University of Nevada, Las Vegas and photos B, C, and D courtesy of Dr. Michael Young, Desert Research Institute, Las Vegas, Nevada, U.S.A.

Laboratory experiments have shown that pores near the soil surface undergo microerosion and microdeposition owing to percolation.^[12] Coatings of non-laminated and fine-textured particles,^[12] clays,^[5] and calcite^[3] have been observed on vesicle surfaces. Plate-like silt and clay particle vesicles with a pavement-like appearance^[5,13] and layers of differing thicknesses have also been observed within vesicles.^[13]

STABILITY

While fragile and easily disturbed by most fauna and humans, vesicular horizons have been noted to persist in environments where surface stability is greatest; increasing surface stability can be due to crusts, pavements,^[3] or landscape age.^[14] Vesicle stability over time has also been attributed to cementation by calcium carbonate^[3] and

possibly to rapid dewatering of the horizon. Evenari et al.^[3] and McFadden et al.^[5] suggest that the formation of silt and clay coatings interior to vesicles can increase vesicle stability. However, if unprotected by a crust or pavement, the vesicular porosity is prone to destruction. On the basis of laboratory experiments, Springer^[1] hypothesized that vesicular porosity was unstable itself in which soil with vesicles was repeatedly wetted, resulting in vesicle destruction and soil rearrangement.

PHYSICAL, CHEMICAL, AND MINERALOGICAL PROPERTIES

The physical, chemical, and mineralogical properties of the vesicular horizon distinguish it from other soil horizons and help explain its genesis and ecosystem role. The thickness of the horizon may range from a few millimeters to several

centimeters,^[3] and soil bulk densities of the horizon range from 0.9 to 1.52 g cm⁻³.^[1,2] The horizon has been found to be formed in soils of a variety of parent materials deposited via eolian processes^[1,5] as dust and is often composed of a wide range of soil textures (fine sand, silt, silt loam, and clayey silt),^[3,7] very fine sandy loam,^[1] and sandy loam.^[5] In the laboratory, coarse sand has been observed to produce vesicular porosity.^[3] The vesicular horizon has been observed to contain few, if any, stones.^[7-9]

From a literature survey, McFadden et al.^[5] report vesicular horizon soil pH to be between 7.0 and 8.9 and conductivity to be 0.1–0.5 mS cm⁻¹. Mineralogy consists of quartz dominantly in the coarse fraction; with finer fractions having a very fine silt, micritic calcite, and unoriented silicate clay.^[5] Clay-size mineralogy consists of a mixed assortment of mixed-layer clay minerals, smectite, kaolinite, illite, quartz, and rare occurrences of palygorskite (McFadden, 1982; McFadden et al., 1986; Reheis et al., 1995, as cited in McFadden et al.^[5]). In soils from Patagonia, an exchangeable sodium percentage >15 was important in vesicle formation.^[15] Soil color varies depending on geographic location where formed and whether one is interior or exterior of a ped.^[9] In arid landscapes, vesicular horizon colors are typically lighter due to less organic matter.^[5]

HYDRAULIC PROPERTIES

The vesicular horizon has been hypothesized to restrict plant growth^[8] with one possible mechanism being the non-interconnected pores found in the horizon,^[12] which may restrict water movement especially in thicker horizons. Field research on the hydraulic properties of vesicular horizons has noted significantly lower hydraulic conductivities in vesicular horizons on surfaces of increasing geomorphic age due to greater accumulations of clay. Surface age has been shown in the Mohave Desert, U.S.A., to be a good predictor of vesicular horizon-saturated hydraulic conductivity.^[8]

CONCLUSION

Owing to the nature of the vesicular horizon's non-interconnected pores restricting water movement through the horizon, many researchers have speculated that the vesicular horizon could be the most influential soil horizon in arid landscape geomorphology and pedogenesis. From an ecological standpoint, the horizon likely plays a significant role in arid ecosystems, and further research should address the spatial extent of such horizons in various arid landscapes and its role in controlling water and nutrient movement into the soil. From a human health perspective, vesicular horizons once disturbed may pose a human health

risk because of atmospheric dispersion of fine particles that can contribute to respiratory distress. Further research should explore the horizon's disturbance in proximity to urban development and human health consequences and its resilience to disturbance.

REFERENCES

1. Springer, M.E. Desert pavement and vesicular layer of some soils of the desert of the Lahontan basin, Nevada. *Soil Sci. Soc. Am. Proc.* **1958**, 22, 63–66.
2. Miller, D.E. Formation of vesicular structure in soil. *Soil Sci. Soc. Am. Proc.* **1971**, 35, 635–637.
3. Evenari, M.; Yaalon, D.M.; Gutterman, Y. Note on soils with vesicular structure in deserts. *Z. Geomorph. N.F.* **1974**, 18 (2), 162–172.
4. FitzPatrick, E.A. *Micromorphology of Soils*; Chapman & Hall: New York, 1984; 433 pp.
5. McFadden, L.D.; McDonald, E.V.; Wells, S.G.; Anderson, K.; Quade, J.; Forman, S.L. The vesicular layer and carbonate collars of desert soils and pavements: formation, age, and relation to climate change. *Geomorphology* **1998**, 24 (2–3), 101–145.
6. Marbut, C.F. Soils of the United States. In *Atlas of American Agriculture, Part 3*; USDA Bureau of Soils: Washington, 1935; 98 pp.
7. Bunting, B.T. The occurrence of vesicular structures in arctic and subarctic soil. *Z. Geomorph. N.F.* **1977**, 21 (1), 87–95.
8. Young, M.H.; McDonald, E.V.; Caldwell, T.G.; Benner, S.G.; Meadows, D.G. Hydraulic properties of a desert soil chronosequence in the Mojave Desert, USA. *Vadose Zone J.* **2004**, 3, 956–963.
9. Anderson, K.; Wells, S.; Graham, R. Pedogenesis of vesicular horizons, Cima Volcanic Field, Mojave Desert, California. *Soil Sci. Soc. Am. J.* **2002**, 66, 878–887.
10. Pullan, R.A. A morphological classification of lateritic ironstones and ferruginised rocks in northern Nigeria. *Niger. J. Sci.* **1967**, 1 (2), 161–174.
11. Allen, B.L. Micromorphology of aridisols. In *In Soil Micromorphology and Soil Classification*; Douglas, L.A., Thompson, M.L., Eds.; Soil Science Society of America: Madison, 1985; 197–216.
12. Figueira, H.; Stoops, G. Application of micromorphometric techniques to the experimental study of vesicular layer formation. *Pedologie* **1983**, 33 (1), 77–89.
13. Sullivan, L.A.; Koppi, A.J. Morphology and genesis of silt and clay coatings in the vesicular layer of a desert loam soil. *Aust. J. Soil Sci.* **1991**, 29, 579–586.
14. Haff, P.K.; Werner, B.T. Dynamical processes on desert pavements and the healing of surficial disturbances. *Quart. Res.* **1996**, 45, 38–46.
15. Bouza, P.; Del Valle, H.F.; Imbellone, P.A. Micromorphological, physical, and chemical characteristics of soil crust types of the central Patagonia region, Argentina. *Arid Soil Res. Rehabil.* **1993**, 7, 355–368.

Village Bamboos

Arun Jyoti Nath

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A. and Ecology and Environmental Science, Assam University, Silchar, India

Ashesh Kumar Das

Department of Ecology and Environmental Science, Assam University, Silchar, India

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

It has been recognized that bamboos growing in the wild play an important role in soil nutrient buildup and maintenance of soil fertility. However, the cultivated one that has been managed by farmers in Asia over millennia for their livelihood was little studied with respect to its role in soil nutrient status. Therefore, the specific aim of this entry is to describe the usefulness of bamboo plantations to improve soil quality and sustain agronomic productivity. Field measurement of soil properties indicates soil bulk density decreases with an increase in age of bamboo plantation from 1.42 Mg m⁻³ under 2-year-old plantations to 1.27 Mg m⁻³ under 20-yr-old plantations with an overall decline of 11% in soil compactness. On the contrary, there is an increase in water holding capacity (30.3–41.5%), and of the concentration of soil organic carbon (SOC; 0.68–1.17%), total nitrogen (0.11–0.28%), available phosphorus (1.73–4.53 mg 100 g⁻¹), and exchangeable potassium (0.20–0.67 cmol (+) kg⁻¹) with an increase in plantation age by 37, 72, 155, 162, and 235%, respectively, from soil under 2–20-year-old plantation. The SOC stock and sequestration rate of village bamboo plantation to 10 cm depth range from 9.66 to 14.86 Mg ha⁻¹ and 0.14 to 0.39 Mg ha⁻¹ yr⁻¹, respectively. The increasing trend of nutrient and SOC stock suggests the potential of village bamboo management in restoring soil quality through onsite enrichment, conservation, and cycling of carbon and nutrients.

INTRODUCTION

Bamboo, the giant grass, is a vernacular term for members of the subfamily Bambusoideae of the family Poaceae. In Asia, the history of bamboo is inextricably interwoven with human history so much that parts of Asia could be described as a “bamboo civilization.” Bamboo has unique rhizomal growth feature by which culms (individual bamboo) in the clump (cluster of culms) are interconnected and reproduce asexually to produce new culms every year. This characteristic distinguishes bamboo from most other woody plants. Based on the bamboo production area, for example, rural landscape or forest areas can be termed as “village bamboos” or “forest bamboos.”^[1] In contrast with forest bamboos, village bamboos are cultivated and managed in a traditional homegardening system (an age old tropical agroforestry system) to fulfill diverse livelihood requirements and provide numerous environmental services to rural communities. Furthermore, village bamboos protect traditional homesteads from winds, provide construction materials and fuel wood.

Bamboo plays an important role in maintaining and improving the nutrient status of the soil.^[2] From a comparative study, it was reported that the presence of bamboo in

the forest significantly affected the physical and chemical properties of soil.^[3] Nutrient content in soil was positively related to yield and explained much of the variation in yield across bamboo sites and regions in China.^[4–6] Hence, bamboo growth and biomass are positively related to soil organic matter, which is the primary source of nutrients in bamboo cultivation sites in Korea.^[7,8] Bamboo can grow in relatively poor soil and efficiently make use of the available nutrients and build up relatively fertile soil around the clumps.^[9] While studying the relationship between soil conditions and fountain bamboo (*Sinarundinaria fangiana*), it was observed that bamboo grew well on acid soil with low base saturation, deep, and low gravel content of soils but died in the alkaline, shallow, calcareous soil with high gravel content.^[10] A canonical correlation analysis for bamboo growth showed that surface soil depth, total nitrogen (N), and soil organic matter content had high positive correlation, and clay content and cation exchange capacity were negatively correlated with the bamboo growth.^[8] Studies related to soil and bamboo revealed emphasis has been given to forest bamboos, whereas village bamboos remain unexplored although it forms an important component of the agroforestry systems of south Asia, especially in India and Bangladesh. In traditional

agroforestry systems, bamboos are grown on soils of poor quality or degraded site of the holdings. Therefore, bamboo has been traditionally used to reclaim degraded lands. Despite the significant impact of village bamboos to the livelihood and wellbeing of farmers, there exists the need of scientific understanding of the role of traditional village bamboo management on sustaining soil nutrient status. Therefore, this entry is to understand how village bamboo plantations impact soil quality.

Traditional Village Bamboo Management System

In village landscapes, bamboo plantation (Fig. 1) is the inherent social and ecological attribute of local farming practices. For optimum utilization of traditional homegardens, farmers have divided it into different landforms depending on the suitability of the species for different soil quality. Each land form varies by certain environmental factors and specific vegetation types. Traditional management system for plantation development is outlined in Fig. 2. Under the traditional management system, propagation is practiced through offset (belowground vegetative

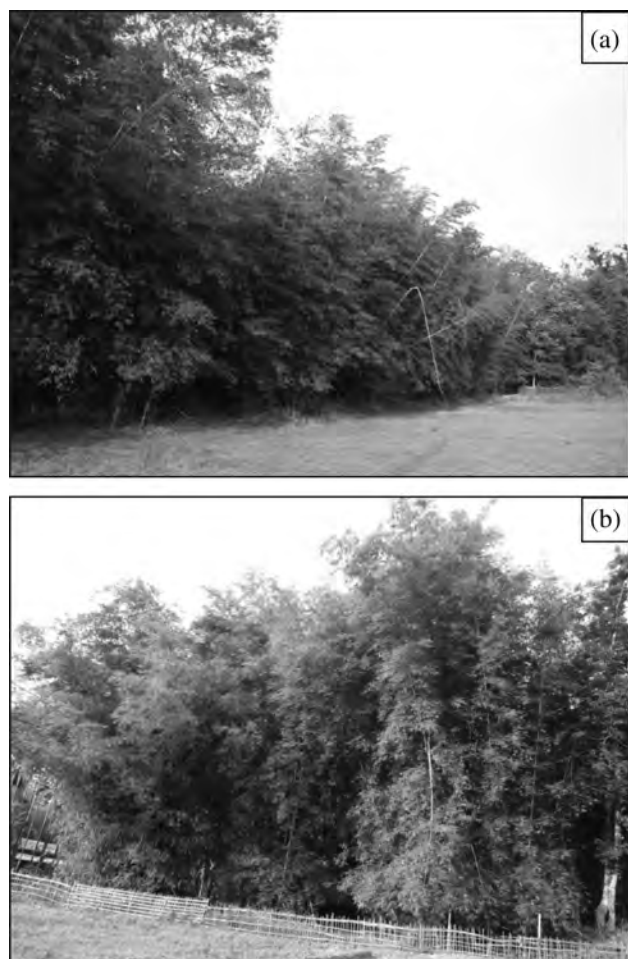


Fig. 1 Village bamboo plantations in NEI.

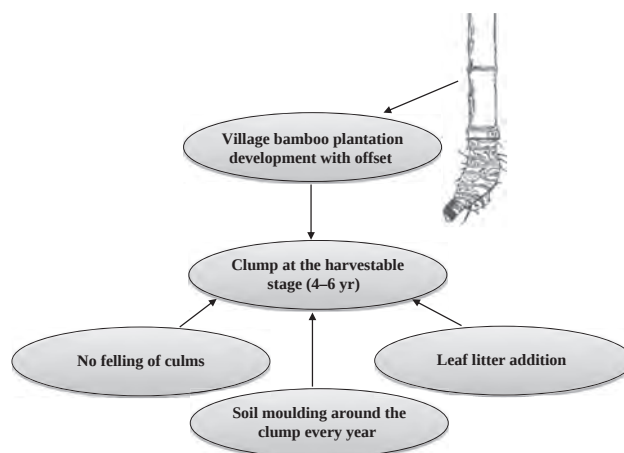


Fig. 2 Traditional management system for raising bamboo plantation.

part). Offset from 1- to 2-year-old culms is cut at about 1.5–2.0 m height, excavated along with a portion of rhizome, and planted during the rainy season. Bamboo plantation management fulfills both subsistence and commercial needs of the bamboo growers and generates employment at the community level (Fig. 3). Due to its site specificity in the homegarden, bamboo builds up a complex interplay between soil and vegetation implying the prevalence of a different microclimate under bamboo canopy than that under other parts of the homegarden.

SOIL PROPERTIES AND BAMBOOS

Data from North East India (NEI) on soil physical and chemical properties with respect to age chronosequences (2-, 5-, 10-, 15-, and 20-year-old clumps) of bamboo plantation (*Bambusa cacharensis*) are presented in Fig. 4. Soil bulk density (BD) decreases with increase in clump age, and all parameters exhibit a strong negative correlation with the age (Fig. 4). Such a decreasing pattern is attributed to the increasing amount of soil organic carbon (SOC) from input of litter fall and root biomass with increase in clump age. A significant effect of bamboo species (*Guadua angustifolia*) on BD and SOC concentration has also been reported in soils of montane region of Ecuador.^[11] A similar trend is also reported for *Dendrocalamus strictus* from some coal mined areas in Central India^[9] and Moso bamboo (*Phyllostachys pubescens*) of Japan.^[12] The number of culms per clump increases with the increase in clump ages^[3] which increases the litter fall under aged clumps.^[13]

Increase in clump age accumulates more SOC from litter mass that subsequently influences soil water holding capacity ($R^2 = 0.99$, $P < 0.001$). There exists a statistically significant correlation of SOC content with clump age ($R^2 = 0.95$, $P < 0.001$; Fig. 4). The SOC is an important determinant of soil quality, especially of nutrient reserves and soil fertility.^[14] Bamboo growth and biomass are

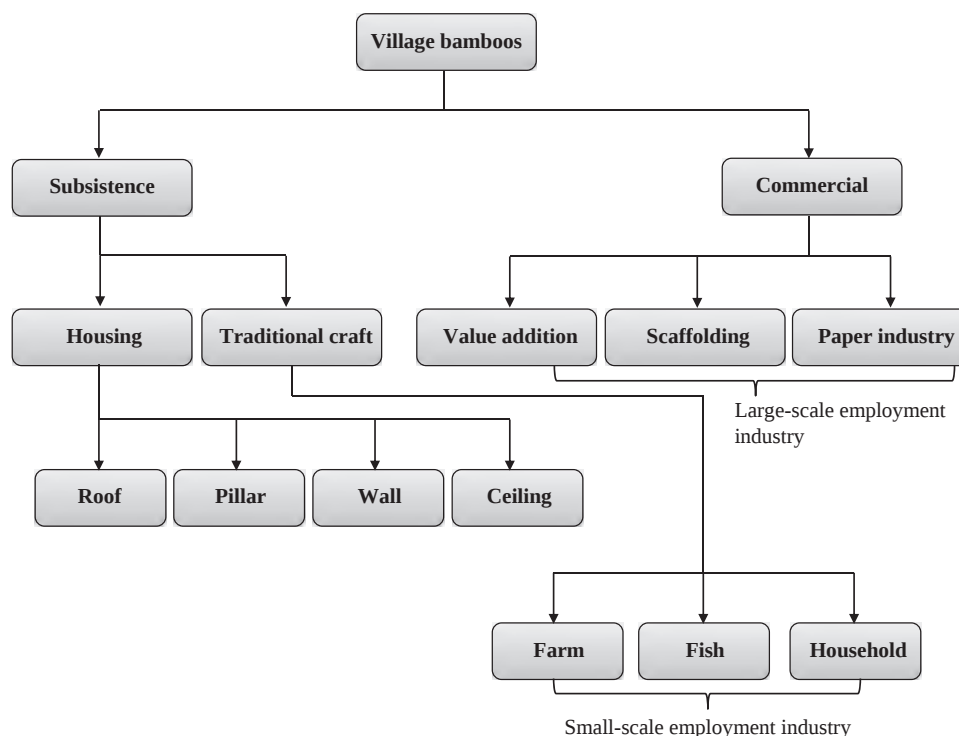


Fig. 3 Diversity in utilization of village bamboos.

positively correlated with SOC content because it is a primary source of nutrients in bamboo cultivation.^[9,15] Total N content also increases with increase in clump age. A similar trend is observed for *D. strictus* plantations in Central India.^[9] Increase in N content is strongly correlated with the SOC content ($R^2 = 0.98$, $P < 0.001$) of the bamboo plantation. Slow rate of bamboo litter decomposition and subsequent nutrient release replenish soil carbon (C) and N stocks and positively influence the soil microenvironment^[16,17] and, therefore, signify the role of bamboo litter in enhancing C and N contents under its canopy. Concentration of available phosphorus (P) and exchangeable potassium (K^+) increases with increase in clump age, and these variables are strongly correlated (Fig. 4). The bamboo stand (*P. pubescens*) mixed with other trees has high productivity and strong nutrient recycling ability,^[18] indicating the importance of bamboo agroforestry in enhancing P and K^+ contents in soil. Increase in the concentration of P and K^+ under managed bamboo plantation (*Phyllostachys praecox*) has also been reported in China.^[19] Further, SOC concentration is positively correlated with that of P ($R^2 = 0.95$, $P < 0.001$) and K^+ ($R^2 = 0.92$, $P < 0.001$) under bamboo canopy. For a bamboo forest in the NEI, over harvesting of bamboo culms for commercial purposes impoverishes the soil nutrient status^[20] and thus reduces soil fertility. Bamboo culms are harvested selectively under bamboo agroforestry system.^[13,21] This strategy develops a balance between harvest and production for maintaining a steady stock, which subsequently increases litter deposition over time, enhancing SOC and, therefore, P and K^+ contents in soil.

SOC Stock and Sequestration Rate

In bamboo forests, the SOC stock is about twice as much as in the aboveground vegetation.^[22] The SOC stock (up to 10 cm soil depth) under bamboo in NEI ranges from 9.7 Mg ha⁻¹ under 2-year-old plantations to 14.9 Mg ha⁻¹ under 20-yr-old plantations and increases linearly with increase in age ($R^2 = 0.94$, $P < 0.05$). In Moso bamboo plantation, the average SOC stock of 94 Mg ha⁻¹ up to 1 m depth has been reported from China.^[23] In montane region of Ecuador, SOC stock of *Guadua* bamboo forest is as much as 52–96 Mg ha⁻¹ up to 30 cm soil depth.^[11] The SOC stock ranges from 25 to 35 Mg ha⁻¹ up to 30 cm soil depth in a managed bamboo (*P. pubescens*) in Zhejiang Province, China.^[23] Soil C stock ranges from 23.2 to 29.1 Mg C ha⁻¹ in a poplar (*Populus deltoides*)-based agroforestry systems in India.^[24] The SOC stock of 21.6–25.6 Mg C ha⁻¹ has been reported in soil under acacia (*Acacia nilotica*), white catechu (*Acacia polyacantha*), and mangium (*Acacia mangium*) agroforestry systems after 5 years of plantation establishment.^[25] The estimated SOC stock of bamboo plantation in NEI (10–15 Mg ha⁻¹, 10 cm depth) is comparable to those for other species in 0–10 cm depth, signifying the role of village bamboos in SOC stock management. The SOC sequestration rate (0–10 cm depth) in NEI varies with plantation age. The highest SOC sequestration rate is 0.39 Mg ha⁻¹ yr⁻¹ for 5–10-year-old plantation that gradually decreases to 0.14 Mg ha⁻¹ yr⁻¹ in 15–20-year-old plantation. This signifies saturation of SOC sink capacity with increase in the plantation age. However, this trend is based on a few sampling points from a few

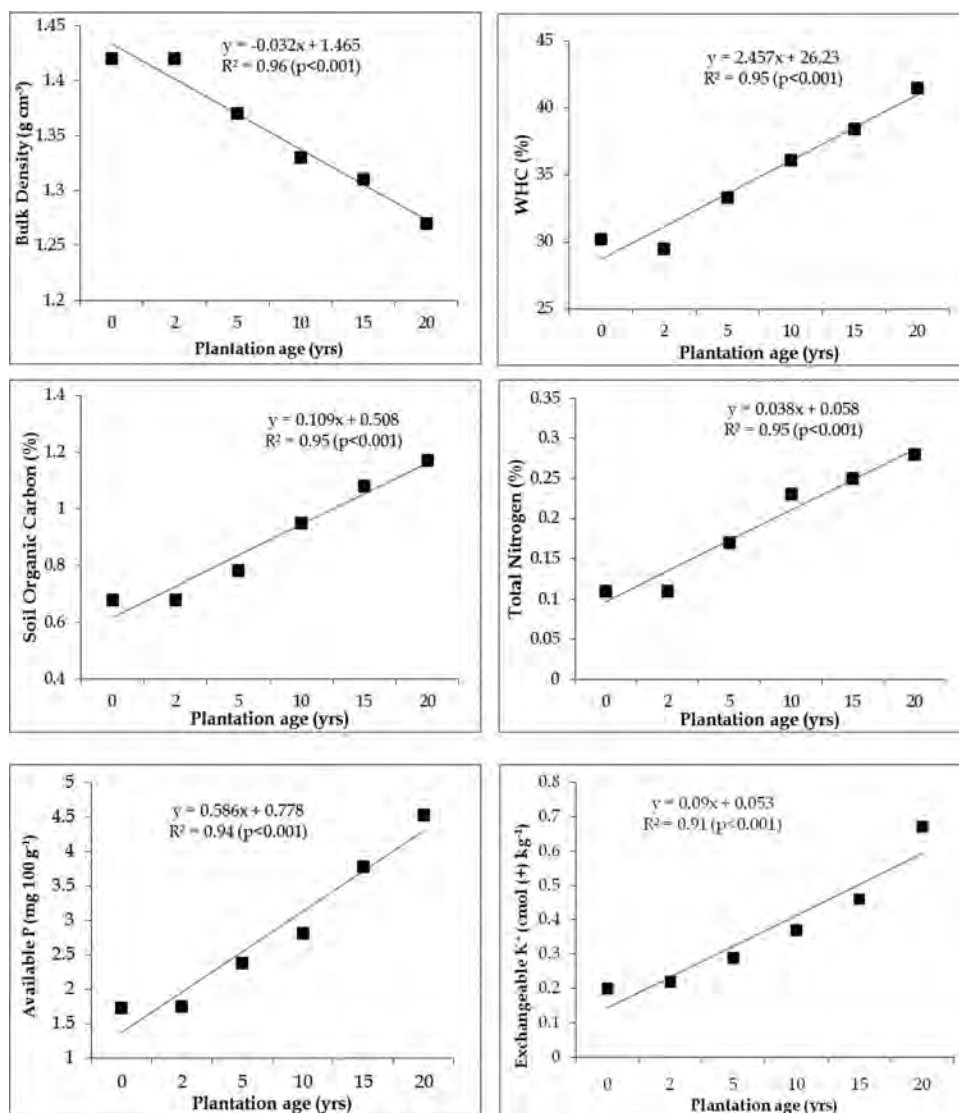


Fig. 4 Correlation of bamboo plantation age with different soil parameters in bamboo agroforestry system in NEI.

bamboo clumps. Therefore, a well-designed study is needed to strengthen the data bank. A third-order polynomial equation ($Y = 0.0815X^2 + 0.5134X - 0.4081$, $R^2 = 0.95$, $P < 0.05$) best explains the SOC sequestration rate with respect to age chronosequences of a bamboo plantation. The SOC sequestration rate of $0.44 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ is reported from bamboo-based agroecosystems in upland of Vietnam.^[15] A study from China shows that SOC sequestration rate can increase up to $6.3 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ (up to 40 cm soil depth) in an intensively managed bamboo forest (*P. praecox*).^[26] Therefore, opportunities exist to enhance SOC sequestration rate in village bamboo stand by adopting scientific management systems.^[27]

CONCLUSION

An increasing trend in nutrient and SOC stock with increase in clump age under the bamboo canopy suggests

the potential of bamboo in soil restoration, onsite nutrient conservation, nutrient cycling, and C sequestration. Measured SOC stock and sequestration rate are comparable to those of other agroforestry and bamboo-based agroecosystems elsewhere for similar soil depths, suggesting the high potential of traditionally managed village bamboos in sustaining productivity and restoring soil quality. However, more research is needed to evaluate the management systems that can further promote SOC sequestration rate and improve soil quality under village bamboo plantations.

ACKNOWLEDGMENT

Senior author greatly acknowledges the research fellowship granted by the Department of Biotechnology, Government of India, in the form of Overseas Associateship.

REFERENCES

- Holttum, R.E. *The Bamboos of the Malay Peninsula*; The Gardens' Bulletin: Singapore, 1958; Vol. 16.
- Kleinhenz, V.; Midmore, D.J. Aspects of bamboo agronomy. *Adv. Agron.* **2001**, *74*, 99–153.
- Christanty, L.; Mailly, D.; Kimmins, J.P. Without bamboo, the land dies: Biomass, litter fall, and soil organic matter dynamics of a Javanese bamboo talon-kebun system. *Forest Ecol. Manag.* **1996**, *87* (1–3), 75–88.
- He, L.M.; Ye, Z.J. Application of multiple statistical analysis on the study of bamboo (*P. pubescens*) soils. *J. Bamboo Res.* **1987**, *6* (4), 28–40. (in Chinese).
- Hong, S.S. Multiple-year response of bamboo forest to fertilization. *Interciencia* **1994**, *19*, 394–398.
- Shanmughavel, P.; Peddappaiah, R.S.; Muthukumar, T. Biomass production in an age series of *Bambusa bambos* plantations. *Biomass Bioenerg* **2001**, *20* (2), 113–117.
- Jin, H.S.; Chong, H.P. Studies on fertilizer-managements and growth analysis in the rejuvenating bamboo grove. *J. Korean Forest Soc.* **1982**, *56*, 51–65. (in Korean).
- Chung, Y.G.; Ramm, C.W. Relationships between soil-site properties and bamboo (*Phyllostachys bambusoides*) growth. *J. Korean Forest Soc.* **1990**, *79*, 16–20.
- Singh, A.N.; Singh, J.S. Biomass, net primary production and impact of bamboo plantation on soil redevelopment in a dry tropical region. *Forest Ecol. Manag.* **1999**, *119*, 195–207.
- Zhang, X.; Zhang, X.M.; Wang, J.P. Mass distribution and nutrient content circulation of short rotation *Phyllostachys nidularia* forest. *J. Bamboo Res.* **1996**, *15* (3), 67–81. (in Chinese).
- Tian, G.; Justicia, R.; Coleman, D.C.; Carroll, C.R. Assessment of soil and plant carbon levels in two ecosystems (woody bamboo and pasture) in montane Ecuador. *Soil Sci.* **2007**, *172*, 459–468.
- Hiraoka, M.; Onda, Y. Factors affecting the infiltration capacity in bamboo groves. *J. Forest Res-Jpn.* **2012**, *17*, 403–412.
- Nath, A.J.; Das, A.K. Carbon storage and sequestration in bamboo-based smallholder homegardens of Barak Valley, Assam. *Curr. Sci. India* **2011**, *100* (2), 229–233.
- Lal, R. Soil carbon sequestration impacts on global climate change and food security. *Science* **2004**, *304* (5677), 1623–1627.
- Ly, P.; Pillot, D.; Lamballe, P.; Neergaard, A.D. Evaluation of bamboo as an alternative cropping strategy in the northern central upland of Vietnam: Above-ground carbon fixing capacity, accumulation of soil organic carbon, and socio-economic aspects. *Agr. Ecosyst. Environ.* **2012**, *149*, 80–90.
- Nath, A.J.; Das, A.K. Decomposition dynamics of three priority bamboo species of homegardens in Barak Valley, Northeast India. *Trop. Ecol.* **2011**, *52* (3), 325–330.
- Tu, L.; Hu, H.; Hu, T.; Zhang, J.; Li, X.; Liu, L.; Xiao, Y.; Chen, G.; Li, R. Litterfall, litter decomposition, and nutrient dynamics in two subtropical bamboo plantations of China. *Pedosphere* **2014**, *24* (1), 84–97.
- Liu, G.; Fan, S.; Qi, L.; Xiao, F.; Huang, Y. Nutrient distribution and biological cycle characteristics in different types of *Phyllostachys pubescens* forest in northwest Fujian. *Chinese J. Ecol.* **2010**, *29* (1), 2155–2161. (in Chinese).
- Zhang, T.; Li, Y.; Chang, X.; Jiang, P.; Zhou, G.; Liu, J.; Lin, L. Converting paddy fields to Lei bamboo (*Phyllostachys praecox*) stands affected soil nutrient concentrations, labile organic carbon pools, and organic carbon chemical compositions. *Plant Soil* **2013**, *367* (1), 249–261.
- Nath, A.J.; Das, A.K. Population status and regeneration of a tropical clumping bamboo *Schizostachyum dullooa* under two management regimes. *J. Forest. Res.* **2011**, *22* (1), 43–46.
- Nath, A.J.; Lal, R.; Das, A.K. Managing woody bamboos for carbon farming and carbon trading. *Global Ecol. Conserv.* **2015**, *3*, 654–663.
- Song, X.; Zhou, G.; Jiang, H.; Yu, S.; Fu, J.; Li, W.; Wang, W.; Ma, Z.; Peng, C. Carbon sequestration by Chinese bamboo forests and their ecological benefits: Assessment of potential, problems, and future challenges. *Environ. Rev.* **2011**, *19*, 418–428.
- Zhou, G.M.; Xu, J.M.; Jiang, P.K. Effect of management practices on seasonal dynamics of organic carbon in soils under bamboo plantations. *Pedosphere* **2006**, *16*, 525–531.
- Gupta, N.; Kukal, S.S.; Bawa, S.S.; Dhaliwal, G.S. Soil organic carbon and aggregation under poplar based agroforestry system in relation to tree age and soil type. *Agroforest. Syst.* **2009**, *76* (1), 27–35.
- Kimaro, A.A.; Isaac, M.E.; Chamshama, S.O.A. Carbon pools in tree biomass and soils under rotational woodlot systems in eastern Tanzania. In *Carbon Sequestration Potential of Agroforestry Systems: Opportunities and Challenges*. Advances in Agroforestry; Kumar, B.M., Nair, P.K. R., Eds.; Springer: Dordrecht, 2011; Vol. 8, 129–143.
- Zhou, G.; Zhuang, S.; Jiang, P.; Xu, Q.; Qin, H.; Wong, M.; Cao, Z. Soil organic carbon accumulation in intensively managed *Phyllostachys praecox* stands. *Bot. Rev.* **2011**, *77* (3), 296–303.
- Nath, A.J.; Lal, R.; Das, A.K. Ethnopedology and soil quality of bamboo (*Bambusa* sp.) based agroforestry system. *Sci. Total Environ.* **2015**, *521–522*, 372–379.

Waste Rock and Overburden Dumps: Rehabilitation

Martin V. Fey

R.D. O'Brien

A.J. Mills

Department of Soil Science, University of Stellenbosch, Matieland, South Africa

Abstract

Improved mining technology over the last few decades has resulted in an increasing trend toward open-cast methods, thereby leading to the disturbance of large areas of land to access the natural resources beneath the surface. This entry focuses on principles that would apply to the whole range of spoil materials—from those produced by conventional quarrying to those from strip mining of vast ore deposits over hundreds of square kilometers. The nature of the spoil may vary from relatively hard, coarse, unweathered rock fragments to fine, soft, easily erodible earthy materials. In all cases, the common thread is the need to first remove whatever soil cover exists before mining can commence. This soil cover may either have been lost or, at least if stockpiled for later use in rehabilitation work, severely disturbed through mixing of topsoil with deeper horizons and sterilized to some degree as a result of stockpiling. A wide variety of practices have been applied to restore landscapes that have been affected by mining. The focus of this entry will be on modern methods and recognized principles of rehabilitation.

DETERMINATION OF REHABILITATION GOALS BEFORE MINING COMMENCES

A prerequisite to mining should be the execution of a baseline survey that identifies and maps the existing soils and establishes the potential land use as a means of guiding the formulation of rehabilitation goals. While the planned rehabilitation program is unlikely to completely dictate mining methods, the greater the cooperation between the mine and rehabilitation planners, the better the chances of successful rehabilitation. Physical and chemical characterization of each overburden layer needs to be done prior to mine development. This will reveal the potential for compaction, erosion, and satisfaction of plant growth requirements in terms of water availability, nutrition, and toxicity problems. Soil scientists and mining engineers often disagree over what constitutes “topsoil;” the onus should be on the soil scientist to demonstrate the special character of true topsoil in terms of the contribution made by humus to soil structure, aeration, water holding capacity, plant nutrient supply, and toxicity buffering, and also in terms of microbiological activity. Topsoil may also constitute a seed bank that may be of enormous value during revegetation.

The final land use must be decided upon before the development of the mine and should consider factors such as stability and degree of permanence and not just the level of productivity to be achieved. The final land use will also influence the landscape design, fertilizer, liming, and maintenance requirements, and the cost of reclaiming the area. However, the final land use should be seen as the

minimum requirement for rehabilitation and not inhibit research to develop improved methods that could lead to a more stable end land use. The aim of landscaping of waste dumps is to produce slopes that are not prone to erosion and are compatible with the proposed land use and the surrounding landscape. Erosion of waste dumps can have severe effects on the success of the reclamation, leading to exposure of encapsulated waste rock, high sediment loads, and lower water quality in receiving streams. Various computer models have been developed to aid the design of waste dumps^[1] by predicting erosion and soil loss from waste rock dumps. These models range in complexity from short-term stability during the construction stage to topographic evolution models simulating erosion over centuries.

There is no typical design or shape of waste rock and overburden dumps. Economic circumstances often dictate, however, that the amount of landscaping be kept to a minimum, in which case there is still a considerable slope angle on the sides of the dump. This may have important implications for revegetation because of differential temperature and moisture regimes on north and south aspects and the need for special measures such as terracing to trap water and reduce erosion. The latter can in some cases be counterproductive because of the tendency of contour terraces to concentrate the flow of water and to become breached if rainfall is too intense. The consequences can then be quite devastating in the form of deep gully erosion. [Fig. 1](#) gives an example of the landscaping challenges presented by waste rock dumps.



Fig. 1 Waste rock overburden alongside open-cast coal mining on the South African highveld.

RESTORING A GROWING MEDIUM FOR PLANTS

There are physical, chemical, and biological factors that need to be considered in restoring the capacity of overburden materials to sustain plant growth.

One of the biggest physical problems is compaction arising from the use of machinery during overburden placement after mining. Some degree of soil compaction is inevitable as soils are stripped and stored for later use. It is generally agreed that soil handling has a marked effect on the success of any reclamation scheme. In an attempt to remove the effects of compaction on the replaced soil material, various forms of subsoiling or ripping using tines drawn by crawler tractors are used. These methods are not always successful, and alternative methods of soil placement may need to be developed. One such method, called loose tipping,^[2] involves the tipping of soil material from dump trucks, with the heaps being spread by an excavator operating from the overburden surface. With this method, there is no need for vehicles to travel over newly restored soil. In some cases, the heaps are left in place to serve as windbreaks, assist in recruitment of windblown seeds, and reduce both erosion and sandblasting effects on new vegetation in areas exposed to strong winds. Eventually, natural forces reduce the relief of the mounds.

Another serious physical problem is the tendency of many spoils to disperse readily under raindrop impact, developing a surface crust that inhibits seed germination and seedling emergence and reducing the infiltration of water and diffusion of oxygen into underlying layers. Severe erosion can result from crust formation. Organic or gravel mulches, or aggregate-promoting chemicals applied on the surface, such as gypsum or polyacrylamide, are variously employed in such circumstances. Some spoils, even though finely divided and showing dispersive tendencies, may contain a certain proportion of coarse gravel and the value of this should be recognized: Initially severe erosion may rapidly abate as the lag concentrate of surface gravel develops into stable mulch.

The placement of topsoil, although valuable, is not a requirement for reclamation success. The quality of topsoil

that has been stockpiled may deteriorate rapidly^[3] and the material often acidifies, requiring liming and fertilizer application to ensure optimal plant growth. The viability of seeds of many plants may decrease sharply after a few months of burial in a stockpile, and microbiological activity may also subside considerably. The importance of using freshly removed topsoil, therefore, needs to be emphasized by the rehabilitation specialist when dealing with mine planners.

In areas with insufficient topsoil, suitable waste rock or overburden is identified as a topsoil substitute. Such material will often be greatly improved by addition of organic matter and soil amendments such as lime (to neutralize acidity if the spoils are sulfidic) or gypsum (to improve permeability to water). Acidic rock drainage (ARD) can be particularly problematic in some mining wastes in view of the gradual nature of pyrite oxidation, leading to the slow release of acid, and the difficulty of accurately estimating the amount of lime needed for neutralization. Burial beneath 0.5 m or more of soil can be quite effective in minimizing ARD by cutting off a ready supply of oxygen from the pyritic soil.

The addition of fertilizers to topsoil substitutes may be needed until the nutrient cycle between soil and plants has been established. Slow-release formulations of nitrogen fertilizers may be especially valuable, although the cost may prohibit their use in some cases. A leguminous component in the vegetation cover can make a big difference to longer term nitrogen supply by harnessing the nitrogen-fixing capacity of symbiotic bacteria. Stimulating the population of soil fauna (e.g., bacteria, fungi, earthworms, termites, and rodents) is seldom pursued through direct inoculation or introduction of the desired species but can usefully be encouraged by understanding their special requirements (e.g., a suitable soil pH in the case of earthworms) and ensuring that these are met to some degree. The loosening and aerating effects of burrowing animal activity may well have been underestimated as prerequisites of successful rehabilitation. Even natural soils in their undisturbed state would tend, after long periods, to become increasingly compacted and structurally inferior were it not for the continual reworking of the soil by resident fauna.

REVEGETATION

The choice of vegetation cover depends on the end use, climatic conditions, and the potential productivity of the growing medium. Where cultivated crops, pastures, or forestry existed prior to mining, the goal is usually to restore productivity by re-establishing the same pattern of land use as far as possible. Often, the goal is partly aesthetic in achieving harmony with the surrounding landscape.

In many rehabilitation projects, it is desirable to reinstate the natural plant and animal communities of the area. This may be because the land is required for conservation or wildlife habitat purposes. Alternatively, it may be decided that natural plant communities will constitute the most

effective rehabilitation because of low maintenance requirements as well as resilience to drought and other extreme climatic events. Plant species in undisturbed ecosystems are an expression of the prevailing soil and climate. Consequently, if premining soil conditions can be approximated, plant species from surrounding natural areas are likely to be suitable for achieving revegetation of the disturbed area. Seeds should ideally be collected from areas that are as near to the rehabilitation site as possible to maintain the genetic integrity of surrounding ecosystems and to ensure that the reintroduced plants are adapted to the local conditions. The number of plant species and functional types on the rehabilitation site should preferably be maximized, given the ecological principle that ecosystem stability tends to increase with increasing biodiversity.^[4]

If topsoil is limited, it may not be possible to reinstate the local flora in its entirety. The chemical status of rock wastes is often not conducive to plant growth and if roots escape the topsoil layer and come into contact with high metal concentrations, acidity, alkalinity, or salinity of the waste materials, revegetation efforts may be thwarted. In such a situation, plant species should be carefully selected for their tolerance to the particular chemical conditions of the dump. Plant species that have colonized old dumps in the area are often suitably adapted to the chemistry of the waste material and constitute a valuable seed source for rehabilitation.

The rehabilitation of a waste dump and the maintenance of a recreated ecosystem may not depend solely on the reintroduction of local plant species. Effective management of the vegetation may include prescribing suitable fire regimes or herbivore stocking densities. Besides their bioturbation role mentioned above, invertebrates may also play a role in key ecosystem processes such as pollination, seed dispersal, and nutrient cycling. Provision of organic mulch on the soil surface can assist colonization of the soil by these invertebrates. Monitoring populations of invertebrates such as ants, nematodes, and spiders is useful for comparing rehabilitated land with adjacent ecosystems. Such studies can be used for assessing the extent to which the rehabilitation process is moving toward a premining ecosystem state.

ENSURING SUSTAINABILITY THROUGH AN UNDERSTANDING OF SOIL-FORMING PROCESSES

A key factor, of which reclamation ecologists are often unaware, is the value of studying natural soil profiles in the undisturbed landscape for clues as to the intensity of soil-forming processes that have given rise to the development of soil horizons. Features such as cemented calcic or duripan horizons in arid landscapes, plinthite in the subhumid tropics, and gleying in hydromorphic soils often reveal subtleties of climatic and topographic effects that are key ingredients in the functioning of the ecosystem. Inability to recognize their significance may doom the rehabilitation

effort to failure in the long run. In arid and semiarid ecosystems, a large proportion of nutrients supplied to roots from the mineralization of soil organic matter is derived from the top few centimeters of topsoil.^[5] This soil skin or pedoderm^[6] has various properties showing greatly enhanced expression relative to the bulk of the topsoil, such as content of organic matter and plant nutrients, activity of microorganisms, and stability of soil aggregates. It may also have features that are not evident in the remainder of the topsoil horizon, such as physical crusting, biological crusts, vesicular layers, bleaching, or salt crystals. The pedoderm is the interface between the atmosphere and the soil body and affects ecosystem function in a disproportionate manner because it controls water and air entry into the soil. Rehabilitation plans should take cognizance of its potentially critical role in sustaining ecosystem function and should aim to restore its unique physical and chemical characteristics.

CONCLUSION

Land disturbed severely through open-caste mining and other activities can be restored successfully by adhering to appropriate techniques and observing certain well-established principles of reclamation, which include the restoration of a growing medium for plants, the selection of vegetation species that is compatible with the local environment, and creating opportunities for a functioning, resilient ecosystem to develop as quickly as possible. Failure to either restore or simulate certain special features of the local soil, which existed prior to disturbance, such as less permeable or wetter layers within the profile as well as soil surface (pedoderm) characteristics, may make it impossible to mimic the original ecosystem.

REFERENCES

1. Evans, K.G. Methods for assessing mine site rehabilitation design for erosion impact. *Aust. J. Soil Res.* **2000**, *38*, 231–247.
2. Moffat, A.J.; Bending, N.A.D. Replacement of soil and soil-forming materials by loose tipping in reclamation to woodland. *Soil Use Manage* **2000**, *16*, 75–81.
3. Davies, R.; Hodgkinson, A.; Younger, A.; Chapman, R. Nitrogen loss from a soil restored after surface mining. *J. Environ. Qual.* **1995**, *24*, 1215–1222.
4. Tilman, D.; Wedin, D.; Knops, J. Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* **1996**, *379*, 718–720.
5. Woods, L.E. Active organic matter distribution in the surface 15 cm of undisturbed and cultivated soil. *Biol. Fert. Soils* **1989**, *8*, 271–278.
6. Mills, A.J.; Fey, M.V. Frequent fires intensify soil crusting: physico-chemical feedback in the pedoderm of long-term burn experiments in South Africa. *Geoderma* **2004**, *121*, 45–64.

Wastewater: Treatment and Utilization in Irrigation

Yona Chen

Jorge Tarchitzky

Department of Soil and Water Sciences, Faculty of Agricultural, Food and Environmental Quality Sciences, Hebrew University of Jerusalem, Rehovot, Israel

Abstract

Wastewater is predominantly utilized water discharged from municipal sewage water systems. Wastewater treatment is essential for the protection of public health and environmental quality. The effluent obtained after wastewater treatment is often utilized for irrigation. Attention has to be paid to various organic, inorganic, and microbial components in these effluents, since some are beneficial while others may adversely affect soil properties, crop quality and yield, or public health. These components in treated wastewater, regulations related to its utilization, and its potential desirable and adverse effects are discussed in detail.

INTRODUCTION

Wastewater or sewage water is utilized water discharged from homes, office buildings, cities, industry, and agriculture. Wastewater treatment is essential for the protection of public health and environmental quality. Treated wastewater (TWW; effluent) can be discharged in either rivers or the open sea, or can be utilized to irrigate agricultural crops. All these outcomes require a certain degree of treatment that will meet specific health and environmental standards. Large regions around the world, such as North Africa, the Middle East, and major parts of Asia, face severe water scarcity combined with the contamination of water resources. In a world where about 65% of total water withdrawal for human use is for irrigation, TWW reuse presents a highly desirable alternative with great potential for solving water shortages.

WASTEWATER CONSTITUENTS

The physical, chemical, and biological properties of wastewater are important parameters in the design of treatment facilities and are crucial in the prerequisite decision-making processes involved in building these plants. The major components of concern in wastewater are volatile and fixed suspended solids, biodegradable organics, pathogens (bacteria, viruses, and parasites), nutrients [nitrogen (N), phosphorus (P), and potassium (K)], heavy metals (e.g., cadmium, zinc, nickel, and mercury), refractory organics, and dissolved inorganics [total dissolved salts, sodium (Na), calcium, magnesium, chlorine (Cl), and boron (B)].^[1] The composition of untreated wastewater depends on the quality of the freshwater (FW) from which it comes and on the ratio between domestic, commercial, and

industrial wastewater in the treatment plant influent. The addition of chemical or biological components to FW during its domestic use is generally called “pick-up.” For some of the constituents, mainly organic materials, pathogens, nutrients, and heavy metals, their concentrations in raw wastewater are a direct result of the pick-up resulting from the specific use of the FW, because in FW they are usually negligible. In contrast, the concentration of dissolved inorganic solids (e.g., salts) is determined by their pick-up during FW use and by the composition of the incoming FW supply. The mineral pick-up in domestic water use in the United States has been reported to include 20–50 mg/L Cl, 40–70 mg/L Na, 150–400 mg/L total dissolved solids, and 0.1–0.4 mg/L B.^[1] These concentrations do not take into account the salt contribution of home water softeners. In Israel, the domestic salt increment is in the range of 80–100 mg/L Cl, 50–80 mg/L Na, and 0.1–0.3 mg/L B. The composition of domestic wastewater from a particular community is roughly constant over time. A typical composition of raw wastewater is presented in [Table 1](#).

WASTEWATER TREATMENT

The early objectives of wastewater treatment were 1) removal of suspended and floatable material; 2) treatment of biodegradable organics; and 3) elimination of pathogenic organisms.^[2] Later, the removal of nutrients such as N and P, as well as heavy metals, was also initiated, pursuant to environmental concerns. More efforts are focused on removing dissolved inorganic salts and organic micropollutants.

The treatments used to achieve the described objectives can be grouped into the following categories: 1) Physical: based on the application of physical forces such as

Table 1 Typical composition of untreated wastewater, TWW at different treatment levels and the overall removal.

Parameter	Raw wastewater	Primary effluent	Secondary effluent	Tertiary effluent	Overall removal
	mg/L				%
BOD (biological oxygen demand)	185	149	13	4.3	98
TSS (total suspended solids)	219	131	9.8	1.3	99+
Ammonia-N	22	21	9.5	9.3	58
Nitrate-N	0.1	0	0	0	0
Total Kjeldahl-N	31.5	30.6	13.9	14.2	56
Phosphate-P	6.1	5.1	3.4	0.1	98
B	0.35	0.38	0.42	0.31	13
Chloride	240	232	238	284	0
Na	198	192	198	211	0

Source: Modified from Metcalf & Eddy, Inc.^[2]

screening, mixing, flocculation, sedimentation, flotation, filtration, and gas transfer; 2) chemical: based on the addition of chemicals or on chemical reactions, such as precipitation, adsorption, and disinfection; and 3) biological: based on biological activity, used to remove biodegradable organic substances as well as nutrients (N and P).

Wastewater treatment plants most commonly employ a combination of different treatment units based on these principles, according to the desired quality of the TWW expected from the plant. The composition of raw wastewater, the resulting TWW quality after different levels of treatment, and the percentage removal of given components are presented in Table 1.

The conventional treatments are very efficient at removing organic components (chemical oxygen demand; biochemical oxygen demand—BOD), nutrients (N and P), and some of the heavy metals. In contrast, ions originating from mineral salts, such as chloride, Na, and B, are not removed during conventional treatments. Only advanced wastewater treatment plants can efficiently remove them.

TWW UTILIZATION

Following treatment, the effluent is transported to reservoirs and from there, when needed, to irrigation plots. Transport and distribution pipeline systems have to meet the following conditions: 1) clear and consistent identification of TWW pipelines and accessories via defined color coding (usually purple) and 2) a minimum horizontal separation between the TWW and FW pipeline systems to prevent cross-contamination, with separation distance varying according to local regulations (usually 3 m; 10 ft).

The TWW reclamation has to contend, via reservoirs, with the gap between TWW flow rate from the treatment plant and TWW demand by consumers. Wastewater flow has daily, weekly, and seasonal fluctuations. These fluctuations can be influenced by storm water inflow, seasonal changes in population due to tourism, or seasonal water

usage in industries. In addition, TWW demand fluctuates due to rainfall and crop water consumption.

REGULATIONS

TWW reuse regulations are based on several considerations and aspects, such as public health, defined utilization areas, environmental aspects, aesthetics, economics, public policy, and public acceptance.^[3] When crop irrigation is considered, the main concern is public health. In California, e.g., regulations are based on the concentration of total coliforms, water turbidity, and required treatment. For food crops, the requirements are 2.2 total coliform/100 ml, two nephelometric turbidity units, oxidation, coagulation, filtration, and disinfection. The World Health Organization guidelines are significantly less restrictive, with a requirement of 6–7 log pathogen reduction and ≤ 1 helminth egg per liter.

IRRIGATION WITH TWW

Nutrients of Importance to Plant Growth in TWW

TWW contains nutrients that are not present in FW, such as organic and ammonium-N, P, K, and micro-elements. N and P concentrations depend on the treatment level since their main source is organic, while K concentration does not change during treatment. It is, however, somewhat higher than in the FW source.

The N in sewage water is mainly organic (protein, amide, amino acid, and urea). Therefore, the main form of N in TWW is organic and ammonium-N. Different treatment plants employ different processes that may change the N forms in the effluent and in case of intensive aeration, nitrate-N appears as well.

P in sewage water and TWW also originates from organic compounds, and its concentration decreases with increasing treatment level. Fifty to ninety percent of the

total P in TWW is in soluble form. In plots irrigated with TWW, P is often applied in higher quantities than in plots irrigated with FW. Consequently, the P level in the soil in these plots may be higher^[4] than recommended for many crops. In plots irrigated with TWW, P may accumulate deeper in the profile. The vertical movement becomes more significant as the clay content in the soil decreases. Excessive P levels in soils may damage crops by inducing micronutrient deficiencies. It may also cause environmental damage by contaminating surface water and inducing eutrophication in water bodies.

K in sewage and TWW is a cation, and its source is pick-up of 5–15 mg/L.

Salt Content in TWW—Effects on Plant Growth

The quality of the TWW is an important factor in adapting its use for the farming community. Potential damage can be incurred by irrigation with low-quality effluents. The main effects to be taken into account are 1) total salt concentration—osmotic effect; 2) concentrations of chloride and B; 3) sodicity; and 4) hydrophobicity.

Total salt concentration—osmotic effect

As a general approximation, the pick-up value of the total salt concentration due to domestic use, measured by electrical conductivity (EC) of the water, is about 0.5 dS/m. This reflects all salts added to the FW supply during domestic and industrial activity. Excess salinity in the root zone adversely affects the growth of established plants as reflected by a general reduction in growth rate. Depression of the external osmotic potential by high salt concentrations tends to narrow the gap between the external (in the soil) and the internal (in the plant) water potentials. Salt stress essentially increases the energy that must be expended by the plant to extract water from the soil and to make the biochemical adjustments necessary for growth relative to non-saline conditions.^[5,6] Salinity effects are closely analogous to those of drought as both result in water stress and reduced growth. Maas and Hoffman^[7] proposed that the response curve be represented by two segments: one, a tolerance plateau with slope zero, and the other, a concentration-dependent line whose slope indicates the yield reduction per unit increase in salinity. The point at which the two lines intersect determines the threshold value. This two-section model provides a reasonably good fit for relative yields when plotted against the EC of the saturated soil extract (EC_s) in units of decisiemens per meter (dS/m). Thus, two values are needed to define a crop's sensitivity to salts: 1) the EC_s tolerance threshold and 2) the percent yield decrease per additional EC_s unit. Relevant tables are presented by Maas;^[8] e.g., avocado, representing a salt-sensitive crop, and cotton, a salt-tolerant crop, have threshold values of 1.0–1.3 and 7.7 dS/m,

respectively, and percent yield decreases of 15–20 and 5.2% per unit dS/m, respectively.

Specific toxicity

Specific toxicity differs from salinity (osmotic effect) in that it occurs within the plant itself and is not caused by a shortage of water. Toxicity normally results when certain ions are taken up with the soil water and accumulate in the plant (usually the leaves) during water transpiration, to an extent that results in damage to the plant. The degree of damage depends upon irrigation period, toxic agent concentration, crop sensitivity, and crop water use. Crop failure due to specific toxicity can be severe and at times lead to total loss. The most common toxic ions are chloride, B, and Na.^[9]

Chloride. Most herbaceous crops are not particularly sensitive to chloride. Woody plant species, on the other hand, are generally susceptible to chloride toxicity. Tolerance varies among species and even among varieties or rootstocks within species.^[10] The data presented in Table 2 indicate the maximum chloride tolerance concentration limits for some crops.^[11] These concentrations refer to saturated soil extracts and to soil concentrations. The concentration of soluble salts in the soil is a result not only of the chloride concentration in the irrigation water, but also of other factors such as soil texture, evapotranspiration, and irrigation management. For some crops, these concentrations may exceed the osmotic threshold.

Boron. Low concentrations of 0.5–1.5 mg/L B are essential for plant growth. B is needed for cell division and root tissue elongation and plays an important role in sugar

Table 2 Crop tolerance to chloride concentration in the saturated soil extract and in the soil.

Crop	Toxicity threshold		Crop	Toxicity threshold	
	mmol Cl/L ^a	mg Cl/kg soil ^b		mmol Cl/L	mg Cl/kg soil
Strawberry	10	250	Potato	15	500
Bean	10		Cucumber	25	600
Carrot	10		Tomato	25	600
Avocado/ West Indian	15		Wheat	60	600
Cabbage	15	500	Sorghum	70	700
Citrus/ Troyer	20		Sugar beet	70	1600
			Cotton	75	1600

^aMaximum Cl concentration in saturated soil extracts without loss in yield.

^bMaximum soil Cl concentration above which a yield decline to 95% of the maximum yield is observed.

Source: From Xu, Magen, et al.^[11]

Table 3 Crop tolerance to B concentration in the irrigation water and in the soil solution.

Crop sensitivity	Irrigation water B concentration (mg/L)	Crop sensitivity	Soil solution B concentration (mg/L)
Sensitive	0.3–1.0	Sensitive	0.5–1.0
Moderately tolerant	1.0–2.0	Moderately sensitive	1.0–2.0
Tolerant	2.0–4.0	Moderately tolerant	2.0–4.0
		Tolerant	4.0–6.0

Source: From Ayers & Westcot.^[9]

transport. B deficiency causes anatomical, physiological, and biochemical changes in plants.^[12] Toxic symptoms of B are similar in most plant species, with the older leaves exhibiting them first. Initial visible symptoms include chlorosis, accompanied by necrosis of the edges and tips of the plant leaves. B concentration in leaves has been found to be in direct proportion to the B concentration in the soil solution.

Crops respond differently to B concentration in both the irrigation water and the soil solution. Accordingly, tables defining plant sensitivity to B in the irrigation water and soil solution have been published.^[5,13,14] Maas^[5] classified crops into groups according to their tolerance to B concentration in the irrigation water and in the soil solution (Table 3). The group of sensitive crops includes lemon, avocado, orange, grapefruit, apricot, peach, persimmon, grape, and onion, among others.

Salt Content in TWW—The Influence on Soil Physical Properties

Sodicity

An important concern in TWW irrigation is excessive Na concentration, which can result in structural deterioration of the soil. Excessive exchangeable Na allows swelling processes in clay minerals: aggregate stability is reduced and water infiltration rates decrease. This becomes a problem when the infiltration rate drops to levels at which the crop is no longer adequately supplied with water or when the hydraulic conductivity of the soil profile becomes too low to provide adequate drainage. A lack of oxygen to plant roots can also result. The sodium adsorption ratio (SAR) of irrigation water is generally a good indicator of the exchangeable Na status of the soil. Together with the adverse effects of sodic conditions, the presence of suspended solids and dissolved organic matter (DOM) in the effluents can affect the physical and hydraulic properties of the soil. In general, the SAR of TWW is higher by 1–3 units due to Na pick-up during domestic use.

A number of researchers have found reduced hydraulic conductivity in soils to which TWW was applied.^[15] Such changes may have physical, biological, or chemical causes.

Clogging of the soil surface layer has also been reported to result from the accumulation of suspended solids. Processes by which living organisms, bacteria in particular, influence soil hydraulic conductivity include slime production and accumulation under anoxic conditions, gas production, in particular by denitrifiers and possibly by methanogens, and plugging of the soil pores by bacterial cells.

Hydrophobicity

The coating of soil particles with DOM can change their surface area. Dissolved organic carbon adsorbs to the soil particles, and consequently, particle behavior may change. Farmers utilizing TWW in northern Israel reported a unique type of water distribution regime in drip-irrigated soils, consisting of 1) a limited wetted area on the soil surface and 2) small saturated areas around and below the dripper in the TWW-irrigated soil, in contrast to the even, onion-shaped wetting profile formed under FW irrigation. Following these observations in the field, it was hypothesized, then proven, that TWW irrigation introduces water-repellent organic constituents into the soil.^[16,17]

Tests performed to characterize the water distribution showed that the diameter of the saturated area on the soil surface and its water content (at a depth of 0–10 cm) were smaller with TWW than with FW irrigation. TWW accumulated on the soil surface in small lenses and then flowed rapidly into the ground. The repellency of soils irrigated with FW and TWW was measured using waterdrop penetration time and additional tests. Soils irrigated with FW were hydrophilic, whereas those irrigated with TWW exhibited hydrophobicity.^[17]

CONCLUSION

TWW is a valuable resource that has the potential to substitute much of the irrigation water used in agriculture. This substitution is essential for appropriate management of precious water resources, in particular in arid and semiarid countries. TWW is rich in nutrients, mostly N, P, and K, and therefore, its application requires adjustments on the farmers' side to avoid overfertilization and damage to the environment. Irrigation with TWW may also result in adverse effects on soil and groundwater quality. The most important factors that need to be addressed are mineral salts additions, chloride- and B-specific toxicity, enhanced SAR values, dispersive effects of DOM, DOM-induced hydrophobicity, and pathogens. Appropriate wastewater treatment practices, disinfection regulation, and caution can ensure that adverse effects are avoided.

REFERENCES

1. Pettygrove, G.S.; Asano, T. *Irrigation with Reclaimed Municipal Wastewater—A Guidance Manual*; Prepared by Dept. of Land, Air, and Water Resources, University of Calif., Davis; Lewis Publishers, Inc.: Chelsea, 1985.
2. Metcalf & Eddy, Inc. *Wastewater Engineering: Treatment, Disposal and Reuse*, 3rd Ed.; Revised by Tchobanoglous, G., Burton, F.; McGraw-Hill, Inc.: New York, 1991.
3. Asano, T.; Burton, F.L.; Leverenz, H.L.; Tsuchihashi, R. *Water Reuse: Issues, Technologies, and Applications*, 1st Ed.; Metcalf & Eddy, Inc., Copyright, Ed.; McGraw-Hill: New York, 2007.
4. Tarchitzky, J.; Loewengart, A.; Bar-Hai, M.; Zilberman, A. *National Wastewater Effluent Survey—2003–2005*; Ministry of Agriculture and Rural Development (in Hebrew): Bet Dagan, 2006.
5. Maas, E.V. Salt tolerance of plants. In *Handbook of Plant Science in Agriculture*; Christie, B.R., Ed.; CRC Press: Boca Raton, 1984; Vol. II, 57–75.
6. Maas, E.V. Crop tolerance to saline sprinkling waters. *Plant Soil* **1985**, *89*, 273–284.
7. Maas, E.V.; Hoffman, G.J. Crop salt tolerance—Current assessment. *J. Irrig. Drain. Div.-ASCE* **1977**, *103*, 115–134.
8. Maas, E.V. Crop salt tolerance. In *Agricultural Salinity Assessment and Management*; Tanji, K.K., Ed.; American Society of Civil Engineers: New York, 1990; 262–304.
9. Ayers, R.S.; Westcot, D.W. *Water Quality for Agriculture*; FAO Irrigation and Drainage Paper 29, Rev. 1; FAO United Nations: Rome, 1994.
10. Hoffman, G.J.; Rhoades, J.D.; Letey, J.; Sheng, F. Salinity management. In *Management of Farm Irrigation Systems*; Hoffman, G.J., Howell, T.A., Solomon, K.H., Eds.; The American Society of Agricultural Engineers: St. Joseph, 1990; 667–715.
11. Xu, G.; Magen, H.; Tarchitzky, J.; Kafkafi, U. Advances in chloride nutrition of plants. *Adv. Agron.* **2000**, *68*, 97–150.
12. Keren, R.; Bingham, F.T. Boron in water, soils and plants. *Adv. Soil Sci.* **1985**, *1*, 229–276.
13. US Salinity Laboratory Staff. *Diagnosis and Improvement of Saline and Alkali Soils*, Handbook No. 60; USDA, US Government Printing Office: Washington, 1954.
14. Feigin, A.; Ravina, I.; Shalhevet, J. *Irrigation with Sewage Effluent*; Advanced Series in Agricultural Sciences; Springer-Verlag: Berlin, 1991; Vol. 17.
15. Tarchitzky, J.; Golobati, Y.; Keren, R.; Chen, Y. Wastewater effects on montmorillonite suspensions and hydraulic properties of sandy soils. *Soil Sci. Soc. Am. J.* **1999**, *63*, 554–560.
16. Chen, Y.; Lerner, O.; Tarchitzky, J. Hydraulic conductivity and soil hydrophobicity: Effect of irrigation with reclaimed wastewater. In *9th Nordic IHSS Symposium on Abundance and Functions of Natural Organic Matter Species in Soil and Water*; Lundstrom, U., Ed.; Nordic Chapter of the International Humic Substances Society (IHSS), Mid Sweden University: Sundsvall Sweden, 2003; Vol. 19, 19 pp.
17. Tarchitzky, J.; Lerner, O.; Shani, U.; Guilboa, A.; Loewengart-Acyicii, A.; Brener, A.; Chen, Y. Water distribution pattern in treated-wastewater-irrigated soils: Hydrophobicity effect. *Eur. J. Soil Sci.* **2007**, *58*, 573–588.

Water Conservation

Paul W. Unger

Conservation and Production Research Laboratory, U.S. Department of Agriculture (USDA), Bushland, Texas, U.S.A.

Abstract

Soil–water conservation is important for crop production in all climatic regions but is critical in subhumid, semiarid, and arid regions. Increasing demands for water for agricultural and non-agricultural purposes make water conservation increasingly important in some regions. Improved soil–water conservation depends on increasing infiltration and reducing evaporation, transpiration (by weeds), and deep drainage losses.

INTRODUCTION

Although this entry emphasizes water conservation for agriculture, it is also important for residential, industrial, and recreational uses. Hence, all users should conserve water, especially where supplies are limited.

Adequate water is essential for optimum growth and development of all crops. For terrestrial crops, water is stored in soil between successive precipitation or irrigation events. Precipitation is usually adequate for crops in humid regions but becomes increasingly limited and erratic when going through subhumid and semiarid regions into arid regions. Periods without effective precipitation (precipitation that is beneficial to crops) often occur during the growing season of most crops in the drier regions. Ratios of precipitation to potential evapotranspiration are >0.75 , $0.50\text{--}<0.75$, $0.20\text{--}<0.50$, and <0.20 in humid, subhumid, semiarid, and arid regions, respectively.^[1]

In humid regions, plants seldom experience water deficiency because precipitation is usually frequent enough. However, even 7–10 days without rain may cause severe plant water deficiencies on soils having a low water-holding capacity. Major droughts cause severe damage regardless of a soil's water-holding capacity. Therefore, soil–water conservation is important in humid regions, especially where irrigation is not practiced.

Precipitation provides most water for crops in subhumid regions, with some water being available for irrigation. In semiarid regions where precipitation is more limited, crops often are irrigated where water is available. Precipitation is usually ineffective (of little or no benefit to crops) in arid regions, and irrigation is the prime water source for crops. Soil–water storage is essential for successful crop production under all climatic conditions, whether the water is derived from precipitation or irrigation, except with drip or daily sprinkler irrigation for which small amounts of water are applied frequently. Water conservation is important under irrigated conditions because of increasing

competition for water between agricultural, urban, industrial, and recreational users,^[2] especially where water supplies are limited or being exhausted.

SOIL–WATER CONSERVATION PRINCIPLES

Soil–water conservation is influenced by water infiltration into soil, evaporation from soil, use by non-crop plants (weeds), and deep drainage. To improve soil–water conservation, water infiltration must be increased, and losses because of evaporation, use by weeds, and deep drainage must be reduced.

Increasing Infiltration

Effective infiltration depends on soil conditions being favorable for adequate water flow into soil and on sufficiently slow runoff to provide adequate time for infiltration. Runoff water does not benefit crops unless captured downstream and subsequently used for irrigation. Runoff is eliminated when the infiltration rate equals or exceeds the precipitation or irrigation rate. Conditions favoring infiltration include the presence of a surface cover to intercept and dissipate the energy of raindrops, thus reducing soil aggregate dispersion and surface seal development that promote runoff; a rough soil surface that temporarily detains water and retards runoff; a soil profile free of horizons that impede water flow; and a low-antecedent soil–water content. Once a soil is filled to its storage capacity, no additional storage can occur. A soil's water storage capacity depends on its texture (sand, silt, and clay content), organic matter content, profile depth, and horizon characteristics. Water infiltrating a soil in excess of its storage capacity is lost by deep percolation unless an impermeable layer is present. When a soil becomes saturated, runoff will increase and some crops may be damaged.

Growing plants or crop residues (straw, stover, etc.) retained on soil through use of conservation tillage (reduced- or no-tillage) can provide a surface cover. Besides reducing the adverse effects of raindrops striking the surface, such cover also retards the rate of runoff across the surface, thus providing more time for infiltration. Growing plants use soil water and, therefore, the soil is usually drier, which increases the potential for infiltration. Other means to reduce runoff and increase infiltration include tillage to disrupt surface seals caused by previous precipitation, create depressions for temporary water storage on the surface (surface roughness, contour tillage, and furrow diking), reduce the rate of runoff from the surface (graded furrow), and disrupt restrictive layers in the soil profile (deep plowing or chiseling). Infiltration may also be increased by applying phosphogypsum (PG) to the soil surface^[3] and injecting an anionic polymer into irrigation water,^[4] which stabilizes the soil surface.

Reducing Evaporation

Soil–water losses by evaporation are the greatest during first-stage evaporation, which depends on the net effect of water flow to the surface and environmental conditions (wind speed, temperature, relative humidity, and radiant energy). Losses decrease rapidly during second-stage evaporation when the soil–water supply decreases, and the rate depends mainly on soil conditions that control water flow to the surface. Third-stage evaporation is low and controlled mainly by adsorptive forces at the solid–liquid interface. Water moves to the surface as vapor in the third stage. The potential for decreasing evaporation is the greatest during the first stage.^[5] Practices that decrease evaporation include decreasing turbulent water vapor transfer into the atmosphere, decreasing capillary continuity or water flow to the surface, and decreasing the water-holding capacity of the surface soil layer. A reduction of turbulent transfer of soil water to the atmosphere is most easily achieved by applying a surface mulch. Many materials can serve as a mulch,^[6] but crop residues and plastic films are most commonly used. Shallow tillage can reduce capillary water flow to the surface and, therefore, reduces the evaporation of soil water under some conditions.

Reducing Use by Non-Crop Plants (Weeds)

Soil water transpired by weeds (including volunteer crop plants) hinders crop production under most conditions. Weeds present before planting decrease the soil–water supply for later crop use, while those present during the growing season compete directly with crops for soil water, space, light, and nutrients. Effective weed control is especially important for dryland (non-irrigated) crops^[7] and efficient use of irrigation water.^[2] Weeds can be controlled with tillage, with the applications of appropriate herbicides, or by hoeing.

Reducing Deep Drainage

Deep drainage occurs when infiltrated precipitation or irrigation water exceeds a soil's water storage capacity. Such drainage reduces the amount that crops can use because it penetrates to depths beyond crop rooting depths. The water may also move some nutrients beyond the rooting depth, thus resulting in inefficient use of applied nutrients and possibly polluting underground water supplies.

To reduce drainage losses, crop-growing seasons should closely match the season when the potential for excess water is the greatest. Encouraging deeper plant rooting, as with deep tillage, and planting deep-rooting crop species also reduces deep drainage. Precipitation cannot be controlled, but irrigations at timely intervals using appropriate amounts of water can minimize the losses. Other practices include deep plowing to bring materials to the surface that retain more water,^[8] installing subsurface barriers,^[9] and adding organic matter to increase the water-holding capacity.^[10,11]

SOIL–WATER CONSERVATION ACHIEVED UNDER DIFFERENT CONDITIONS

Extensive soil–water conservation research has been conducted, but only a few examples will be given because of space limitations. Under semiarid conditions at Bushland, Texas, U.S.A., moldboard-, rotary-, disk-, sweep-, and no-tillage treatments were imposed after harvesting irrigated winter wheat (*Triticum aestivum* L.) to manage the residues during the fallow period until planting dryland grain sorghum [*Sorghum bicolor* L. (Moench)] 10–11 months later.^[12] Plant-available soil–water contents averaged 149, 143, 158, 179, and 207 mm at sorghum planting, and sorghum grain yields averaged 2.56, 2.19, 2.37, 2.77, and 3.34 Mg/ha with the respective treatments. Greater water contents and yields with sweep-tillage and especially no-tillage resulted from more residues retained on the surface than with other treatments during fallow and the sorghum-growing season. The residues resulted in greater infiltration and lower evaporation, but the individual contribution of these processes could not be determined. Weed control was similar with all treatments, and deep drainage was too small to affect the results.

Soil–water contents were similar to the 15 cm depth one day after a 13.5 mm rain at Akron, Colorado, U.S.A., where conventional-, minimum-, and no-tillage treatments were imposed after harvesting winter wheat. Residue amounts were 1.2, 2.2, and 2.7 Mg/ha with the respective treatments. After 34 rainless days, water contents were $<0.1 \text{ m}^3/\text{m}^3$ to 12, 9, and 5 cm depths with conventional-, minimum-, and no-tillage, respectively.^[13] This field study clearly showed the value of surface residues for reducing evaporation, which was also shown under laboratory conditions.^[14,15] Under no-tillage conditions in Colorado,

increasing the cropping intensity from a wheat–fallow to a wheat–corn (*Zea mays* L.) or sorghum–fallow system resulted in a drier soil root zone. This, in turn, resulted in storing more water from precipitation in the soil because of greater infiltration and lower soil–water evaporation and deep drainage.^[16]

When PG was applied at 10 Mg/ha to a ridged sandy soil in Israel, runoff was reduced sixfold as compared with that from untreated soil.^[3] This showed the value of a soil-stabilizing material for reducing runoff and increasing infiltration. When PG was applied at 3 Mg/ha to a clay loam under laboratory conditions, runoff was less than that from bare soil but greater than where wheat straw was applied at 2.2 Mg/ha.^[17] Injecting anionic polymers into water used for furrow irrigation on a silt loam in Idaho reduced soil loss of 70% when applied at 0.7 kg/ha per irrigation and 97% when applied at 10 g/m³ of water. Infiltration was increased probably because surface sealing and sediment transport were reduced.^[4,18]

CONCLUSION

Soil–water conservation is important for crop production in all climatic regions but is critical in subhumid, semiarid, and arid regions. Increasing demands for water for agricultural and non-agricultural purposes make water conservation increasingly important in some regions. Improved soil–water conservation depends on increasing infiltration and reducing evaporation, transpiration (by weeds), and deep drainage losses. Practices for increasing infiltration include keeping the soil covered with plants or a mulch to reduce surface seal development and to impede runoff; applying materials to stabilize the soil surface; and using tillage to disrupt sealed soil surfaces, provide for temporary depressional water storage on the surface, and disrupt subsurface layers that impede water movement. Evaporation can be reduced by a surface mulch (crop residues or plastic films) or by a shallow tillage. Weeds can be controlled by tillage, herbicides, or hoeing. Deep drainage can be minimized by matching crop-growing seasons to seasons when water excesses are most likely increasing soil–water storage capacity, loosening the soil to allow deeper plant rooting, planting deep-rooting crops, and installing subsurface barriers that impede deep drainage.

REFERENCES

- United Nations Educational, Scientific and Cultural Organization (UNESCO). *World Map of Desertification*. United Nations Conference on Desertification; FAO: Rome, 1977.
- Unger, P.W.; Howell, T.A. Agricultural water conservation—A global perspective. *J. Crop Prod.* **1999**, 2 (2), 1–36.
- Agassi, M.; Shainberg, I.; Warrington, D.; Ben-Hur, M. Runoff and erosion control in potato fields. *Soil Sci.* **1989**, 148, 149–154.
- Lentz, R.D.; Shainberg, I.; Sojka, R.E.; Carter, D.L. Preventing irrigation furrow erosion with small applications of polymers. *Soil Sci. Soc. Am. J.* **1992**, 56 (6), 1926–1932.
- Lemon, E.R. The potentialities for decreasing soil moisture evaporation loss. *Soil Sci. Soc. Am. Proc.* **1956**, 20 (1), 120–125.
- Unger, P.W. Role of mulches in dryland agriculture. In *Production and Improvement of Crops for Drylands*; Gupta, U.S., Ed.; Oxford & IBH Publishing Co. Pvt. Ltd.: New Delhi, 1995; 241–270.
- Lavake, D.E.; Wiese, A.F. Influence of weed growth and tillage interval during fallow on water storage, soil nitrates, and yield. *Soil Sci. Soc. Am. J.* **1979**, 43 (3), 565–569.
- Miller, D.E.; Aarstad, J.S. *Effect of Deep Plowing on the Physical Characteristics of Hezel Soil*. Circ. 550; Washington Agriculture Experiment Station: Pullman, 1972.
- Saxena, G.K.; Hammond, L.C.; Robertson, W.K. Effects of subsurface asphalt layers on corn and tomato root systems. *Agron. J.* **1973**, 65 (2), 191–194.
- Jamison, V.C. Changes in air–water relationships due to structural improvement of soils. *Soil Sci.* **1953**, 76 (2), 143–151.
- Unger, P.W. *Relationships between Water Retention, Texture, Density and Organic Matter Content of West and South Central Texas Soils*; Misc. Publ. MP-1192C; Texas Agriculture Experiment Station: College Station, 1975.
- Unger, P.W. Tillage and residue effects on wheat, sorghum, and sunflower grown in rotation. *Soil Sci. Soc. Am. J.* **1984**, 48 (4), 885–891.
- Smika, D.E. *Seed Zone Soil Water Conditions with Reduced Tillage in the Semi-Arid Central Great Plains*, Proceedings of the 7th Conference of the International Soil Tillage Research Organization, Sweden, 1976.
- Unger, P.W.; Parker, J.J. Evaporation reduction from soil with wheat, sorghum, and cotton residues. *Soil Sci. Soc. Am. J.* **1976**, 40 (6), 938–942.
- Ji, S.; Unger, P.W. Soil water accumulation under different precipitation, potential evaporation, and straw mulch conditions. *Soil Sci. Soc. Am. J.* **2001**, 40 (2), 442–448.
- Farahani, H.J.; Peterson, G.A.; Westfall, D.G.; Sherrod, L.A.; Ahuja, L.R. Soil water storage in dryland cropping systems: The significance of cropping intensification. *Soil Sci. Soc. Am. J.* **1998**, 62 (4), 984–991.
- Benyamini, Y.; Unger, P.W. Crust development under simulated rainfall on four soils. *Agronomy Abstracts*; ASA: Madison, 1984; 243–244.
- Trout, T.J.; Sojka, R.E.; Lentz, R.D. Polyacrylamide effect on furrow erosion and infiltration. *Trans. ASAE* **1995**, 38 (3), 761–765.

Water Content

Tom J. Jackson

Hydrology Lab, U.S. Department of Agriculture—Agricultural Research Service
(USDA-ARS), Beltsville, Maryland, U.S.A.

Abstract

Soil water content is a basic hydrologic state variable. It is used in applications that range in scale from irrigation management to flood forecasting. Soil water content has also increased in importance in weather forecasting and climate modeling, as these disciplines have begun to recognize its importance in large-scale water cycles.

INTRODUCTION

As important as soil water content is in human activities and food supply, it has not been as widely observed and monitored as other variables such as precipitation and temperature. This is caused by inadequate instrumentation as well as the numerous scales of variability that affect this state variable.

Soil water content generally refers to the volumetric soil water content. This is the percentage or fraction of a known volume occupied by water (θ_v). This is related to the gravimetric soil water (θ_g) and the bulk density of the soil (ρ_{soil}).

$$\theta_v = \frac{\theta_g \rho_{\text{soil}}}{\rho_{\text{water}}} \quad (1)$$

Soil bulk density is the mass of dry soil within a known volume. Typical values will range from 1 to 1.5 g/cm³. The measurement and estimation of ρ_{soil} are described elsewhere in this encyclopedia as well as in the study by Grossman and Reinsch,^[1] ρ_{water} is the density of water and is close to a value of 1 g/cm³ and is usually dropped from the equation.

VARIABILITY AND SAMPLING

A problem with both characterizing and sampling soil water content is that it exhibits variability that is related to physical characteristics and forcing variables that vary on scales of soil pores to continents. These sources and scales are discussed in further detail in the work done by Hawley et al.,^[2] Kutilek and Nielsen,^[3] Warrick and Van Es,^[4] and Vinnikov et al.^[5] At the field scale, important factors are the soil texture, vegetation, and topography. The next level is related primarily to precipitation and to a lesser degree to evapotranspiration. Finally, larger scale climatology will influence soil water content over regions and continents. From these works, it is evident that variations within a field can be as large as within a 100 km region.

METHODS FOR MEASURING SOIL WATER CONTENT

There is a wide range of techniques available for measuring soil water content. These are described in depth in the work done by Kutilek and Nielsen,^[3] Campbell and Mulla,^[6] Topp and Ferre,^[7] Hillel,^[8] and Or and Wraith.^[9] Some of these are reviewed in the following section. It is important to note that there is no universal standard method (gravimetric with oven drying is the closest). This is because of the difficulties in capturing the variability, instrument design, and logistics. The advantages and disadvantages discussed in the following sections are summarized in [Table 1](#).

Gravimetric Sampling and Oven Drying

Gravimetric sampling combined with oven drying is generally accepted as the standard technique for characterizing soil water content. In this approach, a sample from a specific depth interval is removed from the field and placed in a container of known tare weight. The sealed sample is weighed, opened, and placed in an oven at 105°C for 24 hr. It is then weighed again. The gravimetric soil water content is computed as follows

$$\theta_g = \frac{\text{Wet weight} - \text{dry weight}}{\text{Dry weight} - \text{tare}} \quad (2)$$

If the sample is collected using a tool of known volume, the volumetric soil moisture can be determined directly.

$$\theta_v = \frac{(\text{Wet weight} - \text{dry weight})/\rho_{\text{water}}}{\text{volume}} \quad (3)$$

Otherwise, an estimate of the bulk density must be used to convert θ_g to θ_v .

Coring tools can be used to extract a sample of known volume. This method works well for small diameter samples from moderate depth intervals on compacted soils.

Table 1 Comparison of some methods of measuring soil water content.

Technique	Advantages	Disadvantages
Gravimetric	Only method providing a direct measurement	Destructive
	Inexpensive equipment	Labor required
	Accurate	Time delay for drying
TDR		Small sample size
	Most accurate indirect method	Expensive
	Rapid	Salinity
Capacitance	Some spatial averaging	Depth resolution for surface layer
	Low cost	Sensitive to installation
	Rapid	
Heat dissipation	Probe design very flexible	
	Low cost	Limited range of moisture
	Easy to install	Calibration hysteresis
Radioactive source	Provides water potential	Small sampling area
		Slow response
	Some spatial averaging	Radiation hazard
Remote sensing		Expensive
		Calibration
		Can't measure surface layer
	Local, regional, and global mapping	Poor spatial resolution
	Spatial averaging	Only surface information

For instance, it is difficult to sample shallow surface layers on tilled soils with this method. Scoops and augers can be used to collect gravimetric water content samples. Many of these devices can be seen at the following web site: <http://www.soilsample.com/>.

As noted in methods manuals, care must be exercised when using this method. Samples should be over 100 g, and consideration should be given to the general soil conditions and oven conditions when choosing a drying period. The major drawback to gravimetric sampling is that it is time consuming. In addition, it is destructive. Repetitive sampling at a location could modify conditions.

In Situ and Portable Measurement Techniques

A brief overview of these methods is presented here. An excellent source of additional information is the web site (<http://www.sowacs.com/>), which includes links to manufacturers as well as comparisons.

Time domain reflectometry

Time domain reflectometry instruments represent a fairly advanced technology at this point in time. These devices are capable of measuring two properties of the soil, namely, apparent dielectric constant and conductivity. The discussion here only considers the dielectric constant.

The soil dielectric constant is a function of the volume fractions and dielectric constants of its constituents: water, soil, and air. There is a very large difference between the dielectric constants of water and the other components, which makes these devices highly sensitive to soil water content.

A time domain reflectometry (TDR), as typically used in soil water devices, measures the time it takes for an electrical wave to travel down parallel rods and reflect back. The rods are embedded in the soil and can be used *in situ* or as a portable sensor. The travel time will vary with the sensor configuration and the dielectric properties of the media. There are limitations on how short and how long the rods can be.

Ferre and Topp^[10] and numerous other investigators have found that, for most field soils, a single calibration function can be used. When the soils have high clay contents (or organic matter), a separate calibration is required.^[11]

Therefore, TDR is a relatively robust method for measuring soil water content and can be incorporated as an *in situ* or as a portable sensor. Earlier system designs used low-cost sensors with a relatively expensive cable tester unit. Newer designs include all components in each individual sensor at a modest cost.

Capacitance devices

The dielectric constant of the soil media can also be measured using capacitance sensors. As described above, this dielectric constant is a function of the volumetric soil water content. A capacitance sensor consists of two or more electrodes with the soil media between them. An electronic oscillator is used to determine the oscillator frequency, which is a function of the dielectric constant (as well as the probe configuration).

These devices have become quite popular, and there are several commercially available systems. One difference between these systems is the capacitor design, which affects the way the device can be used. Some devices utilize horizontal plates separated by a low dielectric material. This probe unit is placed in a low dielectric tube (polyvinyl chloride pipe) installed in the soil. Probes can be left in place for *in situ* measurement^[12] or used in a fashion similar to neutron probes, carrying from site to site and logging the profile as the probe is lowered in the tube.

Another capacitance sensor design utilizes parallel rods.^[12] These can be used as *in situ* or portable sampling



Fig. 1 A handheld capacitance type soil water content probe.

tools. One advantage of this device as opposed to the TDR is that shorter rods can be used (5–6 cm are typical). This means measurement closer to the soil surface is possible and that there are likely to be fewer problems when the device is used as a portable surface sampler. Fig. 1 shows a handheld portable version of the capacitance probes. Capacitance probes will be subject to the same calibration problems as TDR devices as related to soil texture and possible mineralogy. Most instruments are supplied with a general calibration.

Heat dissipation

The thermal conductivity of a soil is a function of its water content. Heat dissipation sensors are based on this relationship.^[13] A heater and a thermocouple are embedded in a ceramic probe and placed in the soil. The initial temperature is recorded, and then the heater is turned on for a short period of time and the final temperature is recorded. The change in temperature is related to the soil water content. The ceramic probe is assumed to be in matric potential equilibrium with the surrounding soil. This device is generally used to measure the matric potential and not the soil water content. Once calibrated for matric potential, the probe can be installed in any soil. However, to obtain soil water content, the probe must be either calibrated for the particular site or the matric potential must be related to soil water content using additional calibration or pedotransfer functions.

Radioactive source devices

These methods have been in use for a number of years.^[14] One of the most common is the neutron probe. Here a radioactive source and a detector are lowered down a tube in the soil. Water within the sphere of influence affects the response (neutron scattering). Another radioactive device is the gamma attenuation probe. A source is lowered down a tube while a detector is lowered down a parallel tube. The attenuation is related to the water content. Considering the range of low cost, reliable alternatives available, and the risks associated with radiation, radioactive source methods are not recommended.

Remote sensing

Remote sensing refers to non-contact methods that generally utilize responses within portions of the electromagnetic spectrum. For soil water content measurements, the soil can be viewed from the surface, using these devices. Sensors can be installed on towers, aircraft, and satellites. A key advantage of remote sensing is that systems can be designed to map large regions quickly on a frequent basis.

Although all regions of the electromagnetic spectrum have been explored for soil water measurement,^[15] only microwave techniques can provide a robust measurement. Microwave sensors appropriate for soil water content measurement can see through clouds and most vegetation. Observations can be made at any time of day as solar illumination isn't required. The microwave signal originates within a depth of the surface soil. The lower the frequency used, the deeper is the depth. However, there are restrictions on the frequencies that can be used, which limit this depth to approximately 5 cm.

Two methods can be used, namely, active and passive. Active methods or radars send out a microwave signal, which interacts with the earth's surface (soil and vegetation). A portion of this signal is reflected back to the sensor. The response depends upon the dielectric properties of the soil, among other factors. As described previously, the dielectric properties are functionally related to the soil water content.

Passive methods measure the earth's natural emission at these frequencies. This emission is called the brightness temperature, and it depends upon the physical temperature and the emissivity. Emissivity is a function of the dielectric constant and thus soil water content.

The examples of soil moisture maps derived over a portion of Oklahoma are shown in Fig. 2.^[16] These cover a watershed (600 km²) at a spatial resolution of 200 m. Here the patterns are related to soil type differences that affect the soil water content. The other image is of the same region (10,000 km²) at a spatial resolution of 800 m. Here the spatial patterns are dominated by rainfall features.

DATA SOURCES

For the reasons discussed in the previous sections, there have been few attempts to measure soil water content on a routine basis such as is done with precipitation. Therefore, there are very few data resources and networks. Much of the historic records have been compiled in the study by Robock et al.,^[17] as part of the Global Soil Moisture Data Bank. This information can be obtained through the following website: http://climate.envsci.rutgers.edu/soil_moisture/.

The entry done by Robock et al.^[17] also describes most of the existing networks that collect soil moisture data. Of these, the soil climate analysis network (SCAN) operated

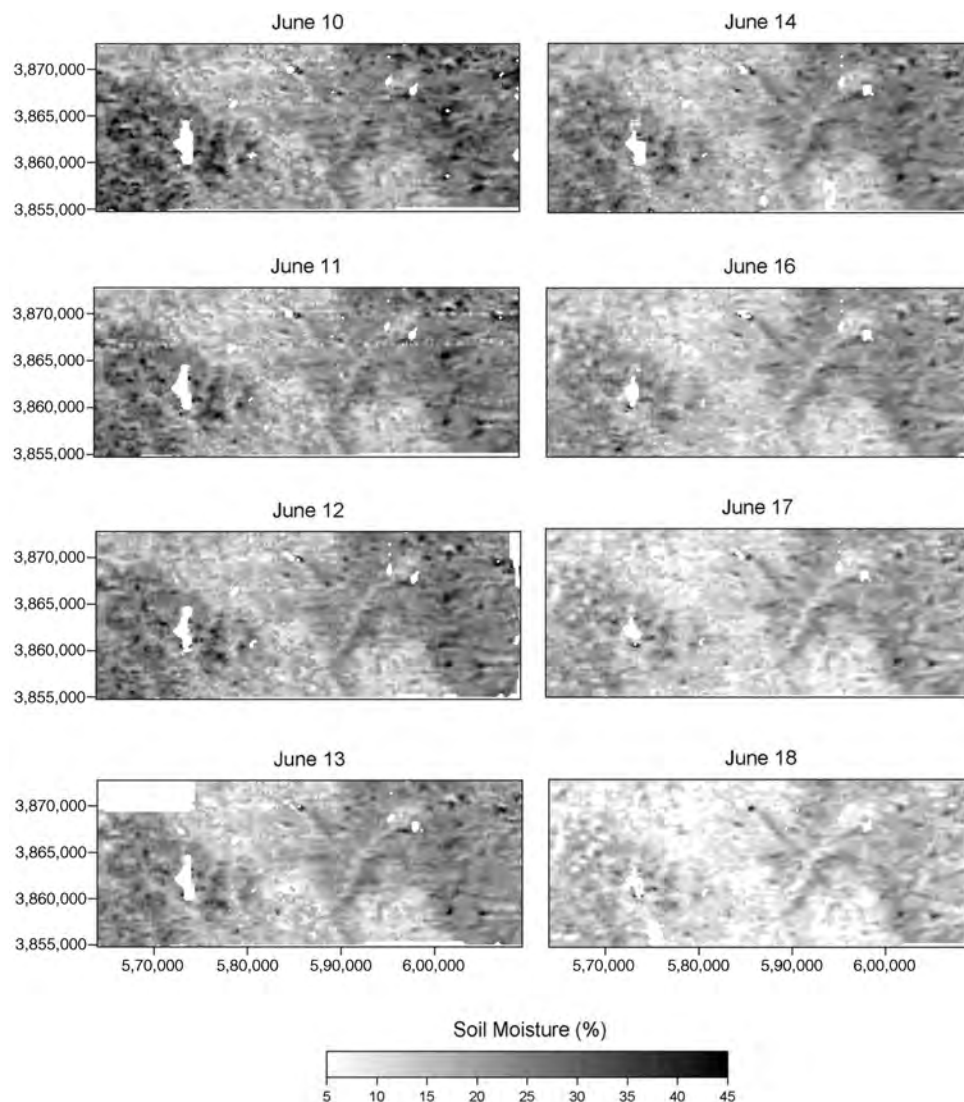


Fig. 2 Soil moisture maps for the Little Washita Watershed, Oklahoma, 1992.

by the U.S. Department of Agriculture is the most readily available. SCAN provides real time and historic (a few years at this point) hourly soil moisture and meteorological data at over 30 stations in the United States. These data can be obtained through the following web site: <http://www.wcc.nrcs.usda.gov/scan/>.

Remotely sensed soil moisture data sets from several previous experiments are available from the following web site: <http://hydrolab.arsusda.gov/>. In addition, since 2002 the Aqua satellite has been producing a global soil moisture product (<http://nsidc.org/daac/amsre/>).

CONCLUSION

In situ soil moisture technology has advanced in years. However, the proliferation of techniques and fundamental differences in the physical characteristic measured has contributed to the lack of standards and transferability of results. There is a great need for improved best practices.

At the present time, the capacitance type probes appear to be a robust approach.

A major change that will occur over the next years is the implementation of satellite remote sensing systems dedicated to the measurement of soil moisture. These include the National Aeronautics and Space Administration Hydros mission^[18] and the European Space Agency Soil Moisture Ocean Salinity Mission.^[19] Routine, daily global mapping of soil moisture, especially if supported by *in situ* soil moisture networks, will benefit many applications.

REFERENCES

1. Grossman, R.B.; Reinsch, T.G. Bulk density. In *Methods of Soil Analysis. Part 4*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America: Madison, 2002; 201–228.
2. Hawley, M.E.; McCuen, R.H.; Jackson, T.J. Soil moisture sampling volume vs. accuracy. *J. Irrig. Drain. Div. ASCE* **1982**, *108*, 1–11.

3. Kutilek, M.; Nielsen, D.R. *Soil Hydrology*; Catena Verlag: Cremlingen-Destedt, 1994.
4. Warrick, A.W.; Van Es, H.M. Soil sampling and statistical procedures. In *Methods of Soil Analysis. Part 4*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America: Madison, 2002; 1–14.
5. Vinnikov, K.Y.; Robock, A.; Qiu, S.; Entin, J.K. Optimal design of surface networks for observation of soil moisture. *J. Geophys. Res.* **1999**, *104*, 19743–19749.
6. Campbell, G.S.; Mulla, D.J. Measurement of soil water content and potential. In *Irrigation of Agricultural Crops*; Agronomy Monograph 30; American Society of Agronomy: Madison, 1990; 127–142.
7. Topp, G.C.; Ferre, P.A. Water content. In *Methods of Soil Analysis. Part 4*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America: Madison, 2002; 417–421.
8. Hillel, D. *Environmental Soil Physics*; Academic Press: San-Diego, 1998.
9. Or, D.; Wraith, J.M. Soil water content and water potential relationships. In *Handbook of Soil Science*; Sumner, M.E., Ed.; CRC Press: Boca Raton, 1999; A-53–A-85.
10. Ferre, P.A.; Topp, G.C. Time domain reflectometry. In *Methods of Soil Analysis. Part 4*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America: Madison, 2002; 434–445.
11. Topp, G.C.; Reynolds, W.D. Time domain reflectometry: A seminal technique for measuring mass and energy in soil. *Soil Till. Res.* **1998**, *47*, 125–131.
12. Starr, J.L.; Paltineanu, I.C. Capacitance devices. In *Methods of Soil Analysis. Part 4*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America: Madison, 2002; 463–474.
13. Scanlon, B.R.; Andraski, B.J.; Bilskie, J. Miscellaneous methods for measuring matric or water potential. In *Methods of Soil Analysis. Part 4*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America: Madison, 2002; 643–670.
14. Hignett, C.H.; Evett, S.R. Neutron thermalization. In *Methods of Soil Analysis. Part 4*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America: Madison, 2002; 501–520.
15. Schmugge, T.J.; Jackson, T.J.; McKim, H.L. Survey of methods for soil moisture determination. *Water Resour. Res.* **1980**, *16*, 961–979.
16. Jackson, T.J.; Le Vine, D.M.; Swift, C.T.; Schmugge, T.J. Large scale mapping of soil moisture using the ESTAR passive microwave radiometer. *Remote Sens. Environ.* **1995**, *53*, 27–37.
17. Robock, A.; Vinnikov, K.Y.; Srinivasan, G.; Entin, J.K.; Hollinger, S.E.; Speranskaya, N.A.; Liu, S.; Namkhai, A. The global soil moisture data bank. *Bull. Am. Meteorol. Soc.* **2000**, *81*, 1281–1299.
18. Entekhabi, D.; Njoku, E.; Houser, P.; Spencer, M.; Doiron, T.; Belair, S.; Crow, W.; Jackson, T.; Kerr, Y.; Kimball, J.; Koster, R.; McDonald, K.; O'Neill, P.; Puttz, T.; Running, S.; Shi, J.C.; Wood, E.; Zyl, J. An earth system pathfinder for global mapping of soil moisture and land freeze/thaw: The hydrosphere state (Hydros) mission concept. *IEEE Trans. Geosci. Remote Sens.* **2004**, *42*, 2184–2195.
19. Wigneron, J.P.; Waldteufel, P.; Chanzy, A.; Calvet, J.C.; Kerr, Y. Two-dimensional microwave interferometer retrieval capabilities over land surfaces (SMOS mission). *Remote Sens. Environ.* **2000**, *73*, 270–282.

Water Content: Passive and Active Microwave Remote Sensing

Eric D. Warner

*Environmental Resources Research Institute, Pennsylvania State University,
University Park, Pennsylvania, U.S.A.*

Gary W. Petersen

*Department of Agronomy, College of Agricultural Sciences, Pennsylvania State
University, University Park, Pennsylvania, U.S.A.*

Abstract

Soil moisture measurement with *in situ* methods over large areas and at frequent time intervals is not practical for most purposes. Remote sensing from air- and space-borne platforms permits data collection from large areas and at frequent time intervals, matching the dynamic nature of soil moisture. Remote sensing with the microwave spectrum, wavelengths ranging from 1 mm to 1 m, is attractive as data can be collected day or night and through clouds. Terrestrial applications of microwave sensing typically use wavelengths from 3 to 75 cm, although the convention of microwave remote sensing is to refer to frequency rather than wavelength. The topic of microwave sensing is a broad one as it not only includes two general types of instruments, but also a variety of applications spanning fields such as agronomy, soil science, hydrology, meteorology, and astronomy.

INTRODUCTION

This discussion will only include passive microwave sensing, synthetic aperture radar (SAR), and their use for detecting soil moisture. There are other types of radar instruments; however, only SAR systems have come to be widely accepted for terrestrial applications. Active and passive microwave systems are briefly described, excluding many critical details about instruments, data collection and processing, and applications. The reader should consult Ulaby et al.^[1–3] for a comprehensive examination of many of these issues. More brief, but very informative, descriptions are available from the work of Elachi,^[4] Rees,^[5] and Lillesand and Kiefer.^[6]

THE PHYSICAL BASIS FOR MICROWAVE REMOTE SENSING OF SOIL MOISTURE

Microwave remote sensing is similar to sensing in other parts of the electromagnetic spectrum in that a sensor is used to detect the results of the interaction of energy from a part(s) of the electromagnetic spectrum with some type of media. Soil is a composite media of solid, water, and air components. As the amount of water in the soil changes, the emission and reflection of energy from the soil also change. The influence of soil moisture on emission and reflection is described numerically by the

dielectric constant. Although the dielectric constant is referred to as a constant, it actually changes with the amount of water in the soil and is indicative of the differences in dielectric properties of dry soil and water. For example, the dielectric constant of water is around 80, while those of soils can vary from 3 (very dry soil) to around 30 (very wet soil).

Fig. 1 shows the change in dielectric constant with volumetric soil moisture at different frequencies for one texture. Most applications of microwave sensing address soil moisture on a volumetric basis to minimize differences due to texture. As can be seen, the relationship between dielectric constant and volumetric moisture content is nonlinear. The rapid increase in dielectric constant with volumetric moisture content can be attributed to the amount of water in direct contact with the soil. At drier moisture contents the water is bound more tightly to the surface of the soil particles, which limits their ability to propagate energy. As the moisture content increases, the water is less bound and permits energy to propagate more freely resulting in an increase in dielectric constant. As can be seen, the dielectric constant is described by a real (ϵ') and an imaginary (ϵ'') component. The real dielectric constant is that which describes the propagation of the energy at a given wavelength or a range of wavelengths. The imaginary part of the dielectric constant describes losses, like friction in which energy is converted to heat, that occur during the interaction of a wave with a material.

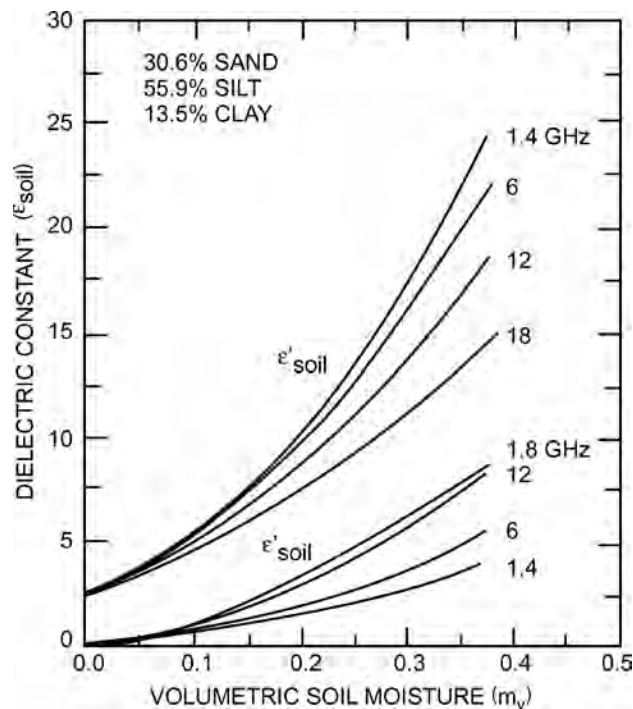


Fig. 1 Variation of dielectric constant with volumetric soil moisture.

Source: Adapted from Lillesand & Kiefer.^[3]

While both passive and active microwave remote sensing of soil moisture are based on the dielectric properties of soil, the physical mechanisms that are the basis for detection by these instruments are very different. A brief background to passive and active microwave sensing is provided as preparation for a description of the application of the instruments for soil moisture detection.

SAR SYSTEMS AND DATA

Active microwave systems employ RADAR, an acronym for radio detection and ranging, technology for generating radio waves for illuminating areas or objects. The radio waves are directed away from a source antenna, which is also used to detect waves reflected from incident surfaces. Fig. 2 shows the general operation of a RADAR system, in this case a side looking airborne radar (SLAR). The detection of reflected waves does not have to be done by the same antenna that generated them, but nearly all commercial and government systems use this approach.

Pulses of known electromagnetic properties are generated by the RADAR hardware and propagated by the antenna. End-to-end knowledge of pulse electromagnetic properties is the foundation of SAR data processing, which can produce the fine azimuth resolution that differentiates SAR from real aperture RADAR systems. The pulse properties on reception provide information about an area from which they were reflected. An area is illuminated by many

pulses as the antenna is moved along a trajectory. SAR processing takes the intensity of each received pulse and sums them. Pulses can contribute positively or negatively depending on the wave properties at detection. Ultimately, an image of values is assembled, indicative of the intensity of the energy received at the antenna and the output of the summation operation executed during processing. Graphically, an image is composed of pixels, giving SAR data an appearance similar to that acquired by the SPOT and Thematic Mapper optical and near-infrared instruments.

The magnitude of the intensity values, also called the backscatter cross-section, in an image is the result of the interaction of the transmitted pulses with the scattering and dielectric properties of the object(s) encompassed in the area defined by a pixel. Fig. 2^[4] shows the difference between the intensity of pulses received from different features present in one image line. For example, the hill has the largest cross-section of all features imaged. The cross-section does not necessarily have to be related to a feature's size. There are conditions in which relatively few small scattering surfaces can generate a large cross-section. Factors influencing backscatter cross-section include surface conditions (soil roughness, soil moisture, and vegetation density and canopy properties) and illumination characteristics (incidence angle, azimuth angle, wavelength, and polarization).

Some additional processing is done to yield an image of values known as the normalized backscatter cross-section. Backscatter cross-section is a ratio of power received over that incident on a surface and also includes antenna and illumination characteristics. For terrestrial applications, backscatter cross-section is usually normalized by the area spanned by the pulses of power summed for each pixel in an image. Targets with area are referred to as distributed, due to the spread of transmitted power across the earth's surface. The spread of power over an area matches the extensive arrangement of most terrestrial targets, such as crop and tree stands.

The normalized backscatter cross-section is a dimensionless quantity equal to the average cross-section per unit area. Normalized backscatter cross-section can vary over orders of magnitude, hence is usually calculated in decibels (10 times the log backscatter cross-section). Note that the normalized backscatter cross-section is the ratio of average scattered power density over the average incident power density over the scattering surface. It is not the total scattered power divided by the total incident power, which means that the normalized backscatter will vary with different incidence and azimuth angles and can be greater than 1 for certain orientations.

PASSIVE MICROWAVE SYSTEMS AND DATA

Passive microwave sensing is based on the detection of radiation by an antenna that is constructed only to

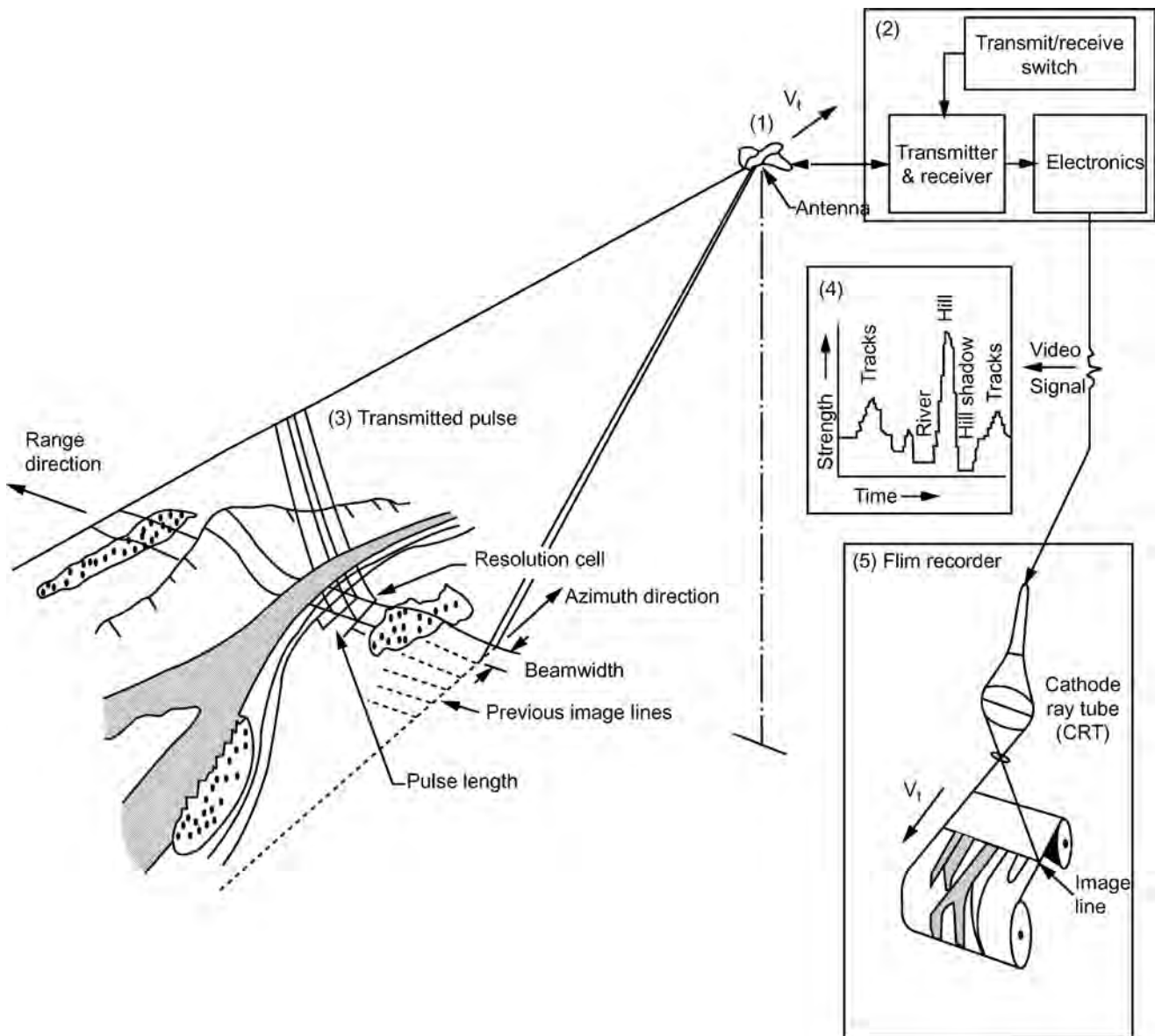


Fig. 2 SLAR instrument operation and data processing.

Source: Adapted from Ulaby, Moore, et al.^[6]

receive energy. The passive instrument then detects the intensity of microwave energy that originates from a source that is not the antenna. Passive instruments are referred to as radiometers, the same terminology used for thermal instruments sensitive to shorter wavelengths. Passive microwave and shorter wave thermal radiometry share many of the same physical and instrument concepts.

The basic structure of a microwave radiometer is shown in Fig. 3. The passive microwave radiometer has three major components. The first component is an antenna that receives the incoming energy for a given instrument orientation. The second component is a receiver that detects and amplifies the signal from the antenna. The receiver is designed to be sensitive to energy of a selected frequency range and polarization.

The third component of a microwave radiometer is a data handling system that has two major parts, a switch and a data recorder. The switch alternately passes data to the data recorder from the receiver and a thermal reference. Data from the receiver are in units of V. The thermal reference attains the temperature that occurs at the surface in order to achieve the detected voltage at the antenna. The nearly simultaneously acquired signals allow for the determination of the apparent antenna temperature, which is a conversion of the antenna acquired voltage to temperature, usually expressed in K. Other recorded data assist post-processing, including antenna pattern and atmospheric corrections. After post-processing, the apparent temperature of the antenna is converted to a quantity referred to as brightness temperature.

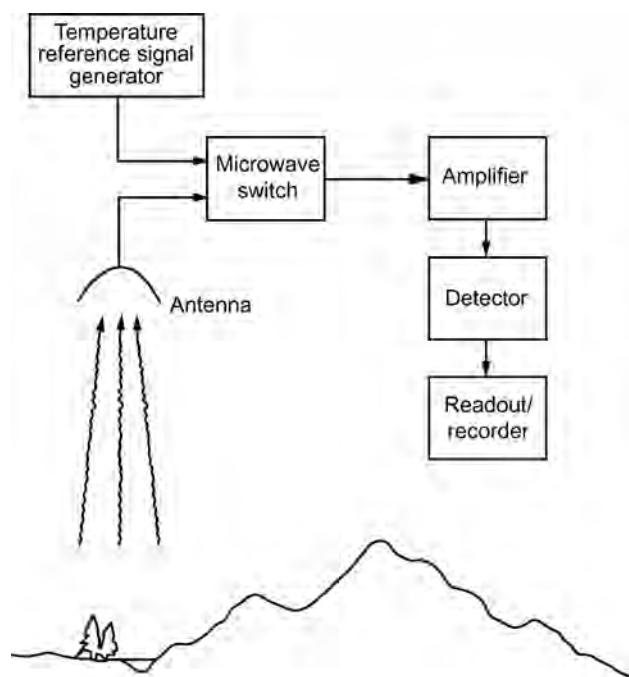


Fig. 3 Passive microwave radiometer sensing system.

Source: Adapted from Lillesand & Kiefer.^[3]

If a scanning radiometer is used to acquire data, then an image can be constructed from the brightness temperature values. The passive microwave image is composed of pixels, the same structure as described for SAR images. Energy detected with passive microwave instruments is low in power and, hence, the spatial resolution of the acquired images is very coarse when compared to SAR systems, usually in the range of kilometers for orbiting instruments.

In the absence of instrument effects, brightness temperature is the product of radiation from sunlight reflected by the earth and atmosphere, and energy emitted from the earth's surface (Fig. 4). For frequencies used for terrestrial

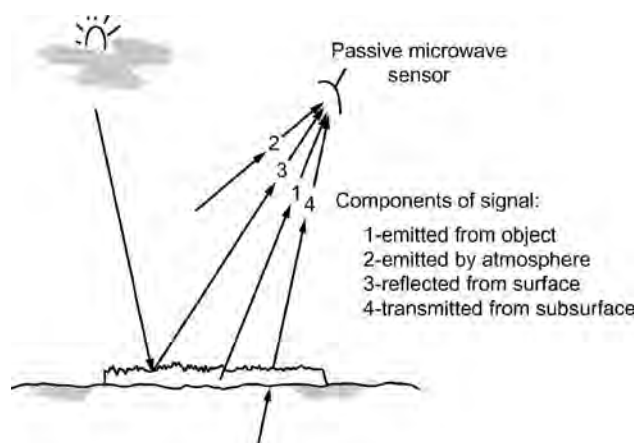


Fig. 4 Signal components for passive microwave sensing.

Source: Adapted from Ulaby, Moore, et al.^[6]

remote sensing, the contribution from the atmosphere is negligible. Reflection and emission from the earth's surface are then the dominant sources of radiation detected by the antenna. Given the variety of sources of energy detected at the antenna, the application of brightness temperature for soil moisture sensing can be complicated. One similarity between brightness temperature and backscatter cross-section is that they are influenced by the same scene and illumination factors.

SOIL MOISTURE SENSING WITH SAR AND PASSIVE MICROWAVE INSTRUMENTS

Many investigations have established the sensitivity of normalized backscatter cross-section and brightness temperature to soil moisture. However, passive microwave brightness temperature and SAR normalized backscatter cross-section are also influenced by soil roughness and vegetation, which complicates the soil moisture detection. It is not possible to describe passive and active sensing of soil and vegetation thoroughly here, but more complete reviews of these sensitivities can be found in the work of Engman,^[7] Engman and Chauhan,^[8] Schmugge et al.,^[9] and Schmugge.^[10] A quick summary of these would imply that backscatter and brightness temperature increase with soil moisture and roughness, and vegetation canopy density.

However, the exact nature of the relationship between normalized backscatter cross-section and brightness temperature, and soil moisture and roughness, and vegetation canopy is also highly dependent on illumination factors (incidence angle, instrument frequency, and polarization, etc.). Also note that soil and vegetation vary greatly over short distances and are independent of each other, further complicating the interpretation of active and passive microwave data. The emphasis of the research is to quantitatively estimate soil moisture from normalized backscatter cross-section and brightness temperature even given the difficulties presented by the soil moisture sensing problem. It is hoped that remotely sensed estimates of soil moisture can be used for applications such as crop management, flood prediction, and regional and global hydrologic research.

Quantitative Soil Moisture Estimation

A variety of modeling approaches have been used to quantitatively estimate soil moisture from SAR normalized backscatter and passive microwave brightness temperature. No approach has been found to be universally applicable. The development and evaluation of quantitative soil moisture algorithms are hampered by the need for extensive supporting ground and sensor data.

The most rigorous investigations have been undertaken through the coordinated effort of a few organizations using airborne multiinstrument sensing of an area, which is

simultaneously sampled for a number of soil and plant properties. The Wahita '92 and '94 programs, using a study area in south-central Oklahoma, are excellent examples of such efforts. Summary of the programs, references to published articles and access to generated data are available through the World Wide Web at <http://hydrolab.arsusda.gov>. Jackson et al.^[11] realized promising results from an airborne passive microwave investigation to estimate soil moisture over a large area of the same section of Oklahoma. See van Oevelen and Hoekman,^[12] for a description of the use of airborne and orbiting SAR instruments to predict soil moisture of sites in Spain and the Sahel.

Progress in the quantitative estimation of soil moisture has also been hindered by the instruments used to collect data. For example, the only orbiting SAR instruments are single frequency/polarization, which restricts the information that can be extracted from the collected data. The lack of information can be compared to attempting to conduct land cover investigations from a single band instrument that only collects data in the green portion of the spectrum. Orbiting passive microwave instruments acquire multifrequency data; however, the spatial resolution of the data is coarse and the frequencies are not ideal for soil moisture detection. The estimation of soil moisture from spaceborne instruments will then rely on having the right instruments as much as having an appropriate method.

REFERENCES

1. Ulaby, F.T.; Moore, R.K.; Fung, A.K. *Microwave Remote Sensing: Active and Passive, Vol. I: Microwave Remote Sensing Fundamentals and Radiometry*; Artech House: Dedham, 1981; 456 pp.
2. Ulaby, F.T.; Moore, R.K.; Fung, A.K. *Microwave Remote Sensing: Active and Passive, Vol. II: Radar Remote Sensing and Surface Scattering and Emission Theory*; Artech House: Dedham, 1984.
3. Ulaby, F.T.; Moore, R.K.; Fung, A.K. *Microwave Remote Sensing: Active and Passive, Vol. III: From Theory to Application*; Artech House: Dedham, 1986.
4. Elachi, C. *Introduction to the Physics and Techniques of Remote Sensing*; Wiley: New York, 1987.
5. Rees, W.G. *Physical Principles of Remote Sensing*; Cambridge University Press: Cambridge, 1990.
6. Lillesand, T.M.; Kiefer, R.W. *Remote Sensing and Image Interpretation*; Wiley: New York, 1979.
7. Engman, E.T. Applications of microwave remote sensing of soil moisture for water resources and agriculture. *Remote Sens. Environ.* **1991**, *35*, 213–226.
8. Engman, E.T.; Chauhan, N. Status of microwave soil moisture measurements with remote sensing. *Remote Sens. Environ.* **1995**, *51*, 189–198.
9. Schmugge, T.J.; Jackson, T.J.; McKim, H.L. Survey of methods for soil moisture determination. *Water Resour. Res.* **1980**, *16* (6), 961–979.
10. Schmugge, T.J. Remote sensing of soil moisture: Recent advances. *IEEE Trans. Geosci. Remote Sens.* **1983**, *GE-21* (3), 336–344.
11. Jackson, T.J.; Le Vine, D.M.; Hsu, A.Y.; Oldak, A.; Starks, P.J.; Swift, C.T.; Isham, J.D.; Haken, M. Soil moisture mapping at regional scales using microwave radiometry: The southern great plains hydrology experiment. *IEEE Trans. Geosci. Remote Sens.* **1999**, *37* (5), 2136–2150.
12. van Oevelen, P.J.; Hoekman, D.H. Radar backscatter inversion techniques for estimation of surface soil moisture: EFEDA-Spain and HAPEX-Sahel case studies. *IEEE Trans. Geosci. Remote Sens.* **1999**, *37* (1), 113–123.

Water Erosion

David Favis-Mortlock

Environmental Change Institute, University of Oxford, Oxford, U.K.

Abstract

Erosion by water is the redistribution and removal of the upper layers of the soil, both by the action of falling rain and by water flowing over the soil during and after rain or following snow melt. It occurs on both agricultural and undeveloped landscapes. While erosion by water is a natural phenomenon, removal of the soil's protective cover of vegetation at times of heavy rains greatly increases erosion rates, so that they exceed rates of soil formation. This is known as accelerated erosion and is always the result of human actions such as unwise agricultural practices, overgrazing, or construction activity. Accelerated water erosion is a problem over much of the earth's surface and has both on-site and off-site impacts. The main on-site impact is the reduction in soil quality which results from the loss of the nutrient-rich upper layers of the soil and the reduced water-holding capacity of many eroded soils. In affluent areas of the world, accelerated water erosion's on-site effects upon agricultural soils can be mitigated by increased use of artificial fertilizers; however, this is not an option for much of the earth's population. Water erosion's main off-site effect is the movement of sediment and agricultural pollutants into watercourses, leading to the silting-up of dams, disruption of the ecosystems of lakes, and contamination of drinking water. In some cases, increased downstream flooding may also occur due to the reduced capacity of eroded soil to absorb water.

PROCESSES OF EROSION BY WATER

Water erosion is driven by the gravitational energy of rain-fall and flowing water. When rain falls upon unprotected soil, its kinetic energy may detach soil particles, and this is one of the subprocesses^[1] of water erosion (RD-ST in Fig. 1) and is often referred to as “rainsplash erosion” or “splash erosion.” A more accurate term is “rainsplash redistribution,” since although considerable quantities of soil can be moved by splash, almost all of it is merely redistributed on the soil's surface, i.e., the net downslope movement of splashed soil is generally small. Rainsplash redistribution is most effective where rainfall intensities are high,^[2] e.g., as a result of convective rainstorms in the world's equatorial regions. Low-intensity rainfall is often of frontal origin. Where such rainfall is common (e.g., northwest Europe), rainsplash is ineffective.

Some of the rain may infiltrate into (i.e., be absorbed by) the body of the soil. In dry conditions, all rain will infiltrate: the result is then rain with no runoff (Fig. 1). Runoff, that part of the rainwater which has not infiltrated, tends to flow downhill under the action of gravity. This thin diffuse film of water has lost virtually all the kinetic energy which it possessed as falling rain, thus it moves only slowly, has low flow power, and is generally incapable of detaching or transporting soil particles (“no erosion” in Fig. 1).

The microtopography (i.e., small-scale pattern of irregularities) of the soil's surface tends to cause this overland flow to concentrate in closed depressions which slowly fill, and this is known as “detention storage” or “ponding.”

Both the flowing water and the water in detention storage protect the soil from raindrop impact, so that rainsplash redistribution usually decreases over time within a storm as the depth of surface water increases. There are, however, complex interactions between rainsplash and overland flow (Fig. 1).

If the rain continues, the increasing depth of water will eventually overtop the depressions. Overland flow that is released in this way is likely to flow downhill more quickly and in greater quantities (i.e., possess more flow power as a result of its kinetic energy) and so may be able to begin transporting (RD-RIFT, RD-FT and FD-FT in Fig. 1) and even detaching (FD-FT in Fig. 1) soil particles. Where it does so, the soil's surface will be lowered slightly. Lowered areas form preferential flow paths for subsequent flow, and these flow paths are in turn eroded further. Eventually, this positive feedback^[3] results in small, well-defined linear concentrations of overland flow (“microrills” or “traces”).

In many cases, individual microrills become ineffective over time due to sedimentation. A subset, however, grow further to become rills^[4] and a smaller subset may go on to develop into gullies. This process of “competition” between microrills and rills leads to the self-organized formation^[3] of networks of erosional channels (dendritic on natural soil surfaces;^[5] constrained by the direction of tillage on agricultural soils^[6]), which form efficient pathways for the removal of water from hillslopes. It is in such erosional channels that water erosion also operates most effectively to detach and remove soil by its kinetic energy.^[7] In most situations, erosion by concentrated flow is the main

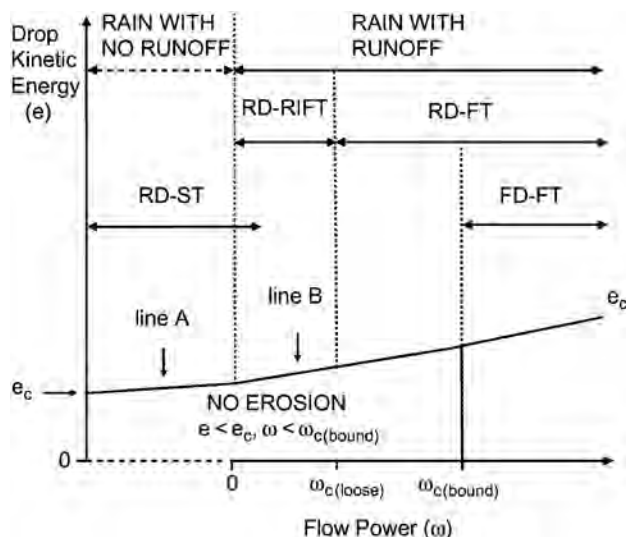


Fig. 1 Detachment and transport processes associated with variations in raindrop and flow energies. Line e_c : critical raindrop energy to cause erosion. Line A: raindrop energy prior to flow (this often increases slightly with time, as a soil crust develops). Line B: raindrop energy when flow occurs (increasing with time, as more drop energy is used to penetrate the deepening flow). $\omega_{c(\text{loose})}$: critical stream power for transporting loose material. $\omega_{c(\text{bound})}$: critical stream power for detaching soil from surface of soil matrix. RD-ST: raindrop detachment, splash transport. RD-RIFT: raindrop detachment, raindrop induced flow transport. RD-FT: raindrop detachment, flow transport. FD-FT: flow detachment, flow transport.

Source: From Kinnell.^[11]

agent of erosion by water. The flow-dominated erosional channels are separated by interrill areas,^[8] where the dominant processes are rainsplash and diffuse overland flow; however, boundaries between rill and interrill areas are both ill-defined and constantly shifting.

In some circumstances, subsurface flow may be important in determining where channel erosion will begin and develop (e.g., at the base of slopes^[9] and in areas of very deep soils such as tropical saprolites^[10]). Meltwater from thawing snow operates in a broadly similar way to rain-derived overland flow,^[11] detaching and transporting unfrozen soil in areas of concentrated flow; however, snowmelt erosion is less well studied and less well understood.

As erosional channels increase in size (i.e., become large rills and gullies),^[12] processes such as the gravitational collapse of channel walls and heads increase in importance.^[13] Runoff and sediment from rills and gullies may be moved into ditches, stream, and rivers and so transported well away from the point of origin. However, sediment may also be deposited within the rill or gully or beyond the rill or gully's confines in a depositional fan, at locations where the gradient slackens. Here it may be stored for a variable period of time,^[14] possibly being reworked by tillage activity until a subsequent erosion event is of sufficient size to



Fig. 2 Impacts of erosion by water at the microscale: overland flow between millimeter-scale soil aggregates. The finger is pointing at an area where concentrated flow is just beginning to incise a microrill.

Source: Photo from a rainfall simulation experiment by A.J.T. Guerra and D.T. Favis-Mortlock, 1997.

re-erode the stored sediment. It may then be redeposited further downstream or make its way into a permanent watercourse and thence to lake or ocean.

EROSION BY WATER: SPATIAL AND TEMPORAL SCALE

Water erosion's complex hierarchy of subprocesses (Fig. 1) means that erosion by water operates (and is studied) over a wide range of spatial scales.^[15] Rainsplash redistribution and the initiation of microrills and rills occur at a scale of millimeters (Fig. 2). Rill erosion on agricultural hillslopes operates at a scale of meters to tens of meters (Fig. 3), while gully erosion can occur on a scale of hundreds of meters or even kilometers. The off-site impacts of erosion can affect very large areas, sometimes hundreds or even thousands of square kilometers (Fig. 4)

At every spatial scale, however, erosion is highly patchy. Even in areas of severe erosion, rates of soil loss can vary greatly from point to point on the landscape as the vagaries of topography and land use concentrate erosive flows on a wide range of spatial scales.^[16] Obvious erosion in one field can be found side by side with virtually untouched areas, and within an eroded field, the severity of erosion can vary markedly.

Erosion by water is often also variable across a range of temporal scales. Soil loss from water erosion occurs both incrementally, as a result of many small rainfall events, and more dramatically, as a result of large but relatively rare storms. Dramatic erosion events can produce large gullies and create flooding and property damage, which hits the news headlines. However, a significant proportion of total soil loss may be due to small but frequent erosion events, which nonetheless have a notable cumulative impact.^[17]



Fig. 3 Impacts of erosion by water at the field scale: a large rill on agricultural land in Germany.
Source: Photo from Katharina Helming, 2001.

THE GLOBAL PROBLEM OF EROSION BY WATER

While soil erosion by water has been occurring naturally for some 450 million years (since the first land plants formed the first soil), accelerated erosion is of much more existing origin. Yet on a human timescale, accelerated erosion is old. There is considerable archaeological evidence from many parts of the world that accelerated erosion by water is often associated with early agriculture.^[18,19] Water erosion's association with unwise agricultural practices was first noted within a scientific context during the second and third decades of the 20th century by pioneers of soil conservation such as Hugh Hammond Bennett in the United States, and subsequently by workers in other parts of the globe. During the period of colonialism, the imposed adoption of European agricultural methods frequently led to accelerated erosion in developing countries, a problem which continues.^[20,21] During the last few decades of the 20th century, a move toward intensive agricultural technologies which leave the soil bare during times of heavy rainfall meant that previously problem-free areas of the world,

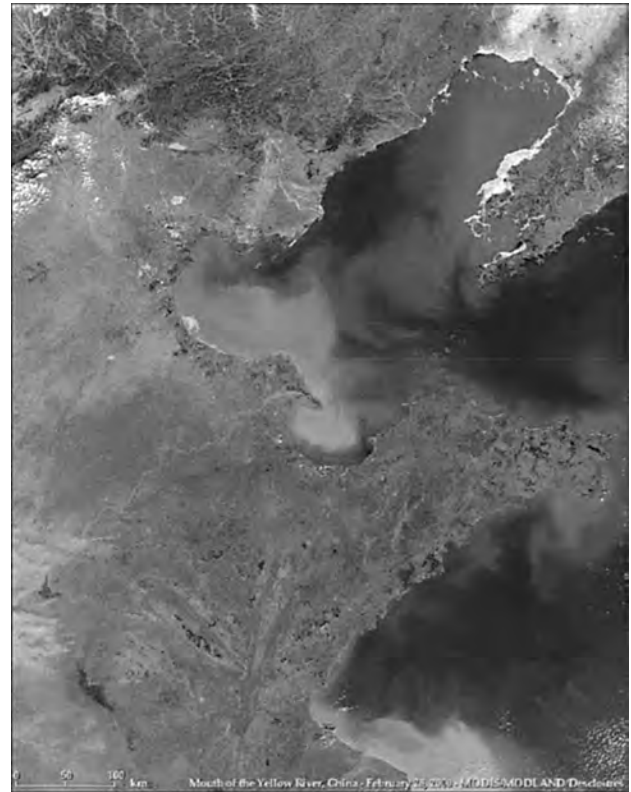


Fig. 4 Impacts of erosion by water at the global scale: a sediment plume covering tens of thousands of square kilometers at the mouth of the Yellow River, China. This results from erosion by water on the Chinese Loess Plateau.

Source: Photo from NASA, 2000, http://modis.gsfc.nasa.gov/MODIS/IMAGE_GALLERY/MODIS1000152_md.html.

such as northwest Europe, began to experience notable increases in water erosion.^[22]

Despite the global nature of the problem of erosion by water, yet we do not have good information regarding the global extent of erosion by water.^[23,24] Data on the severity of erosion are also often limited.^[25]

IMPACTS OF EROSION BY WATER

Impacts of erosion by water can be categorized into on-site (i.e., affecting only the place where the erosion occurs) and off-site (i.e., affecting locations other than that at which the erosion occurs) problems.

Erosion's removal of the upper horizons of the soil results in a reduction in soil quality,^[26] i.e., a diminution of the soil's suitability for agriculture or other vegetation. This is because the eroded upper horizons tend to be the most nutrient rich. Also, because the finest constituents of eroded soil tends to be transported furthest, eroded soils become preferentially depleted of their finer fraction over time; this often reduces their water-holding capacity.^[26] In other words, "erosion removes the cream of the soil."

Increased use of artificial fertilizers may to an extent, and for a time, compensate for erosion-induced loss of soil quality where economic circumstances are favorable. However, this is not usually feasible in developing countries.^[20,21] Loss of soil quality is a long-term problem; globally, it is water erosion's most serious impact.^[27,28]

Movement of sediment and associated agricultural pollutants into watercourses is the major off-site impact resulting from erosion. This leads to silting-up of dams, disruption of the ecosystems of lakes, and contamination of drinking water.^[22] Rates of erosion do not have to be high for significant quantities of agricultural pollutants to be transported off-site.^[29] This is a shorter term impact than loss of soil quality; in the more affluent areas of the world, it can be the main driver for soil conservation policy initiatives.^[24] A more minor off-site effect can occur in situations where eroded soil has a decreased capacity to absorb water; increased runoff may lead to downstream flooding and local damage to property.^[30]

REFERENCES

- Kinnell, P.I.A. *A Discourse on Rainfall Erosion Processes and Modelling on Hillslopes*; Centre for Australian Regolith Studies: Canberra, 2000.
- Nearing, M.A.; Bradford, J.M.; Holtz, R.D. Measurement of waterdrop impact pressures on soil surfaces. *Soil Sci. Soc. Am. J.* **1987**, *51* (5), 1302–1306.
- Favis-Mortlock, D.T.; Boardman, J.; Parsons, A.J.; Lascelles, B. Emergence and erosion: A model for rill initiation and development. *Hydrol. Process.* **2000**, *14* (11–12), 2173–2205.
- Slattery, M.C.; Bryan, R.B. Hydraulic conditions for rill incision under simulated rainfall: A laboratory experiment. *Earth Surf. Proc. Land.* **1992**, *17*, 127–146.
- Helming, K.; Roth, C.H.; Wolf, R.; Diestel, H. Characterization of rainfall–microrelief interactions with runoff using parameters derived from digital elevation models (DEMs). *Soil Technol.* **1993**, *6*, 273–286.
- Planchon, O.; Esteves, M.; Silvera, N.; Lapetite, J.-M. Raindrop erosion of tillage-induced microrelief: Possible use of the diffusion equation. *Soil Till. Res.* **2000**, *15–16*, 1–14.
- Nearing, M.A.; Norton, L.D.; Bulgakov, D.A.; Larionov, G.A.; West, L.T.; Dontsova, K. Hydraulics and erosion in eroding rills. *Water Resour. Res.* **1997**, *33* (4), 865–876.
- Govers, G.; Poesen, J. Assessment of the interrill and rill contributions to total soil loss from an upland field plot. *Geomorphology* **1988**, *1*, 343–354.
- Huang, C.; Laflen, J.M. Seepage and soil erosion for a clay loam soil. *Soil Sci. Soc. Am. J.* **1996**, *60* (2), 408–416.
- Fernandes, N.F.; Coelho Netto, A.L. Subsurface hydrology of layered colluvium mantles in unchannelled valleys—south-eastern Brazil. *Earth Surf. Proc. Land.* **1994**, *19*, 609–626.
- Sharratt, B.S.; Lindstrom, M.J.; Benoit, G.R.; Young, R.A.; Wilts, A. Runoff and soil erosion during spring thaw in the northern U.S. corn belt. *J. Soil Water Conserv.* **2000**, *55* (4), 487–494.
- Poesen, J.; Vandaele, K.; van Wesemael, B. Gully erosion: Importance and model implications. In *Modelling Soil Erosion by Water*; Boardman, J., Favis-Mortlock, D.T., Eds.; Springer: Berlin, 1998; 285–312.
- De Ploey, J. A model for headcut retreat in rills and gullies. *Catena Suppl.* **1989**, *14*, 81–86.
- Trimble, S.W. A sediment budget for coon creek basin in the driftless area wisconsin 1853–1977. *Am. J. Sci.* **1983**, *283*, 454–474.
- Kirkby, M.J.; Imeson, A.C.; Bergkamp, G.; Cammerat, L.H. Scaling up processes and models from the field plot to the watershed and regional areas. *J. Soil Water Conserv.* **1996**, *51* (5), 391–396.
- Favis-Mortlock, D.T.; Boardman, J.; MacMillan, V.J. The limits of erosion modeling: Why we should proceed with care. In *Landscape Erosion and Evolution Modeling*; Harmon, R., Doe, W.W. III, Eds.; Kluwer Academic/Plenum Publishing: New York, 2001; 477–516.
- Boardman, J.; Favis-Mortlock, D.T. Frequency-magnitude distributions for soil erosion, runoff and rainfall—A comparative analysis. *Zeitschrift Für Geomorphologie N.F. Supplementband* **1999**, *115*, 51–70.
- Bell, M.G.; Boardman, J., Eds. *Past and Present Soil Erosion*; Oxbow Books: Oxford, 1992.
- O's Hara, S.L.; Street-Perrott, F.A.; Burt, T.P. Accelerated soil erosion around a mexican highland lake caused by pre-hispanic agriculture. *Nature* **1993**, *362*, 48–51.
- Hallsworth, E.G. *Anatomy, Physiology and Psychology of Soil Erosion*; Wiley-Interscience: New York, 1987.
- Stocking, M. Soil erosion in developing countries: Where geomorphology fears to tread. *Catena* **1995**, *25* (1–4), 253–267.
- Evans, R. *Soil Erosion and Its Impacts in England and Wales*; Friends of the Earth: London, 1996.
- United Nations Environment Program. *Global Land Degradation Map*. Available at <http://www.unep.no/db/maps/prod/level3/id-1238.htm> (accessed July 2001).
- European Environment Agency. *Down to Earth: Soil Degradation and Sustainable Development in Europe*; Office for Official Publications of the European Communities: Luxembourg, 2000. Available at http://reports.eea.eu.int/Environmental_issue_issue_series_16/en (accessed July 2001).
- Boardman, J. An average soil erosion rate for Europe: Myth or reality? *J. Soil Water Conserv.* **1998**, *53* (1), 46–50.
- Lal, R. Monitoring soil erosion's impact on crop productivity. In *Soil Erosion Research Methods*; Lal, R., Ed.; Soil and Water Conservation Society: Ankeny, 1989; 126–137.
- Larson, W.E.; Pierce, F.J.; Dowdy, R.H. The threat of soil erosion to long-term crop production. *Science* **1983**, *219*, 458–465.
- Pimentel, D.; Allen, J.; Beers, A.; Guinand, L.; Hawkins, A.; Linder, R.; McLaughlin, P.; Meer, B.; Musonda, D.; Perdue, D.; Poisson, S.; Salazar, R.; Siebert, S.; Stoner, K. Soil erosion and agricultural productivity. In *World Soil Erosion and Conservation*; Pimental, D., Ed.; Cambridge University Press: Cambridge, 1993; 277–292.
- Harrod, T.R. Runoff, soil erosion and pesticide pollution in cornwall. In *Conserving Soil Resources: European Perspectives*; Rickson, R.J., Ed.; CAB International: Wallingford, 1994; 105–115.
- Boardman, J.; Ligneau, L.; de Roo, A.; Vandaele, K. Flooding of property by runoff from agricultural land in north western Europe. *Geomorphology* **1994**, *10*, 183–196.

Water Erosion: Empirical Models

John M. Laflen

(Retired), U.S. Department of Agriculture—Agricultural Research Service
(USDA-ARS), Buffalo Center, Iowa, U.S.A.

Abstract

Empirical models of soil erosion by water are used to determine the extent of soil erosion, to identify areas needing soil erosion control measures, to select appropriate practices to minimize damages resulting from soil erosion, and to estimate downstream delivery of sediment and other materials. Examples of damages might include reduced productivity, exposure of landfill materials, obstacles to farming, living plant damages, increased maintenance of locks and dams, reduced reservoir capacity, reduced aquatic habitat, deposited nuclear materials, and increased eutrophication of surface waters.

INTRODUCTION

An empirical model is a representation of data.^[1] Although the models described here were developed based on relationships from data, and while these relationships are explainable to a certain extent based on our understanding of the erosion process, they came from data. The models described do not include simulation models, even those that might include empirical components, nor the application of empirical models in Geographical Information Systems.

EMPIRICAL EROSION MODEL DEVELOPMENT

Early erosion research literature describes the effect of various factors—climatic, topographic, soil, management and conservation practices, on soil erosion, and empirical models include most of these factors. Early models were relatively simple relationships with factor values generally determined based on a limited set of experimental observations and involving only a few factors. As the science progressed, the complexity increased, with relationships of factor values to factor characteristics being developed based on experimental data.

One of the early developments was establishment of erosion plots at numerous locations in the United States. These plots provided an immense data set that is used for evaluating soil erosion and hydrologic models. Over 10,000 plot years of data were collected on these plots.^[2] Additionally, many plots were located on experimental farms that contained small watersheds on which conservation practices were installed, monitored, and evaluated. These experimental areas provided the necessary data for developing the relationships in water erosion empirical models.

Erosion plots have also been established in other regions of the world. Hudson^[3] described field experiments involving erosion plots in Rhodesia. Erosion plots can be found on every continent (except Antarctica) and are frequently used to evaluate the applicability of empirical and non-empirical erosion models to specific regions and to develop appropriate factor values for practices common to those regions.

The development of empirical models has been a classic example of the operation of science and its application to solving real world problems. The steady progression of science is illustrated in [Table 1](#) where the steps in development of the U.S. empirical erosion modeling are given chronologically. The work initiated with Zingg continues, each step built on top of earlier work. Meyer^[12] gives an excellent review of the development of the Universal Soil Loss Equation (USLE) in the United States.

EMPIRICAL MODELS

Universal Soil Loss Equation

The USLE is written as

$$A = RKLSCP \quad (1)$$

where, A is soil loss per unit area (units depend on units used for R and K); R is the rainfall and runoff factor, the number of rainfall erosion index units, plus factors for runoff from snowmelt or applied water. K is the soil erodibility factor, the soil loss rate per erosion index unit on a plot 72.6 ft long of uniform 9% slope maintained in a clean tilled fallow. L is the slope length factor, the ratio of soil loss for a given slope length to a 72.6 ft long slope. S is the slope steepness factor, the ratio of soil loss for

Table 1 Evolution of U.S. empirical water erosion models.

Reference	Soil loss	Coefficient or rainfall factor	Soil erodibility factor	Length—slope factor	Cropping management factor	Conservation practice factor
Zingg ^[4]	A	C'	$L^{0.6}S^{1.4}$			
Smith ^[5]	A	C'	$L^{0.6}S^{1.4}$	P		
Browning ^[6]	A	C'''	K'	$L^{0.6}S^{1.4}$	P	
Musgrave ^[7]	A'	$P_{30}^{1.75}$	K'	$L^{0.35}S^{1.35}$	C*	P
USLE ^[8]	A	EI_{30}	K	$(L/72.6)^{0.5} (0.065 + 0.045 S + 0.0065 S^2)$	C	P
USLE ^[2]	A	EI_{30}	K	$(L/72.6)^{0.5} (65.4 \sin^2 \Theta + 4.56 \sin \Theta + 0.065)$	C	P
MUSLE ^[9]	A*	$95 (Qq_p)^{0.56}$	K	$(L/72.6)^{0.5} (65.4 \sin^2 \Theta + 4.56 \sin \Theta + 0.065)$	C	P
OF ^[10]	A	$0.5 EI_{30} + 15 Q_0 q_0^{0.33}$	K	$(L/72.6)^{0.5} (65.4 \sin^2 \Theta + 4.56 \sin \Theta + 0.065)$	C	P
RUSLE ^[11]	A	EI_{30}	K	$(L/72.6)^m (a \sin \Theta + b)$	C	P

A, soil loss (ton/acre); A', soil loss (in./yr); A*, sediment delivery (ton/yr); P_{30} , maximum precipitation amount (in.) falling in 30 minutes in a storm; Q, storm runoff volume (acre-ft); q_p , peak runoff rate (cubic ft per sec); Q_0 , storm runoff volume (in.); q_0 , peak runoff rate (in. per hr); E, storm rainfall energy (hundreds of ft-ton per acre); I_{30} , maximum rainfall intensity in a 30-minute period within a storm (in. per hr); K', K', K, soil erodibility factors; L, slope length (ft); S, slope (%); Θ , slope angle (°); m, exponent on length term; values depend on slope or slope and rill/interrill ratio; a, b, coefficients in function making up slope term; values depend on slope; C', C', C''', coefficients; C*, egetal cover factor; C, cropping and management factor (before 1978) and cover and management factor (after 1978); P, conservation practice factor.

a given slope to soil loss from a 9% slope. C is the cover and management factor, the ratio of soil loss for a given land use and management to soil loss from a continuous clean tilled fallow. P is the support practice factor, the ratio of soil loss with a support practice, such as contouring, terracing, or stripcropping, to soil loss for up and down the slope.

Much more detail on factor values, and assistance in computing these values can be found in Wischmeier and Smith.^[2]

The USLE was the culmination of over two decades of work in developing a national model of soil erosion by water. As shown in Table 1, the form and factors were long established before the USLE was developed. The defining scientific finding in the development of the USLE was that of Wischmeier and Smith^[13] when they isolated a single rainfall variable that could be used over the United States to model soil erosion from rainstorms. This allowed the application of factor values in one region to be used in another region. Climatic databases were developed for the entire United States, and tables and charts were published to allow use of the USLE for most U.S. conditions.^[2,8,14] The USLE technology was the water erosion model of choice over much of the world through the 20th century. It is being replaced by Revised Universal Soil Loss Equation (RUSLE) in the United States.^[11,15]

While the USLE is a good predictor of average annual erosion, it is a poor predictor of individual storm soil erosion because it does not contain variables associated with the hydrologic condition of the soil at the time the storm occurs. Additionally, the USLE does not consider deposition on a field, and does not predict sediment yield, a needed estimate for many applications.

The USLE's area of application is from the top of a hillslope (or divide) to where deposition begins, or runoff enters a channel.

Modified Universal Soil Loss Equation (MUSLE)

MUSLE is written as

$$A^* = 95(Qq_p)^{0.56} KLSCP \quad (2)$$

where A* is sediment delivery, Q is total volume of runoff, and q_p is peak rate of surface runoff, and K, L, S, C, and P are the same as for the USLE.

MUSLE was developed to predict individual storm sediment yield. A major shortcoming of the USLE for many applications was that while it estimated soil erosion, it did not predict sediment yield, and it was a poor predictor of individual storm soil erosion. Williams et al.^[16] determined that for individual storms, sediment concentration for five small watersheds near Riesel, Texas, was highly correlated with the ratio of peak runoff rate per square mile to total volume of runoff. Williams and Berndt^[17] applied the USLE to average annual sediment yield prediction for the same five small watersheds using a delivery ratio based on watershed characteristics. They weighted the various factors in the USLE (except for the rainfall factor) based on drainage area of the various factor values, channel lengths, channel slopes, and cultivated area.

Williams^[9] examined three forms of a runoff factor and determined that the runoff factor $95(Qq_p)^{0.56}$ (with Q, the volume of runoff given in acre-feet, and q_p , the peak flow rate given in cubic feet per seconds) was the best form for estimating individual storm sediment yield from small

watersheds. In the analysis, he used the procedures of weighting the USLE factors pioneered in 1972. The 18 watersheds ranged in size from 2.7 to 4380 acres. Two watersheds were located at Hastings, Nebraska, and the remainder were located at Riesel, Texas, and included the five small watersheds used in earlier analyses.

MUSLE requires inputs—storm volume of runoff and peak rate of runoff, not used in the USLE. This information can be generated by simulation models or it can be estimated for design storms.

MUSLE continues to be heavily used in various modeling activities. It is an option in the EPIC model^[18] for computing sediment yield. While MUSLE is widely used, there are few publications that detail the accuracy of MUSLE, outside of the original work performed by Williams in its development.

ONSTAD-FOSTER (OF) MODIFICATION OF THE USLE

The OF modification of the USLE is written as

$$A = (0.5R + 15Q_o q_o^{0.33}) KLSCP \quad (3)$$

where A, R, K, L, S, C, and P are the same as for the USLE, and Q_o is runoff volume in inches and q_o is peak flow rate in inches per hour.

Foster et al.,^[19] using basic erosion principles, derived a soil loss equation that included a runoff parameter and a rainfall parameter to replace the rainfall factor in the USLE. This factor was $0.5R + 15Q_o q_o^{0.33}$. Onstad and Foster^[10] found good results when the factor was tested using watershed data from Ohio and Iowa.

Again, as for MUSLE, hydrologic information must be available to estimate peak rate and total volume of storm runoff. A major difference between MUSLE and OF are the approaches used in developing the equations—MUSLE variables were selected and tested, while OF variables were determined based on analysis of fundamental erosion processes. Hence, the difference in forms. It has never been established that one is a superior predictor to the other.

OF has received limited use, although it is also an option for use in some modeling activities, including the EPIC model.

Soil Loss Estimation Model for Southern Africa (SLEMSA)

Elwell^[20] introduced SLEMSA as a reliable management tool developed from limited resources. The development team evaluated the USLE for adoption in southern Africa but were concerned about adopting factor values developed in America.^[21] Additionally, there was considerable concern about differences in farming practices. They felt that

they did not have the resources to implement the USLE, and a simpler approach was needed.

SLEMSA is written as

$$Z = KCX \quad (4)$$

where Z is mean annual soil loss, K is mean annual soil loss from a standard field plot 30 m × 10 m at 4.5% slope for a soil of known erodibility under a weed-free bare fallow, C is the ratio of soil lost from a cropped plot to that lost from bare fallow land, and X is the ratio of soil lost from a plot length L and slope percent S to that lost from the standard plot.

K values are computed as

$$\ln K = \ln E + a \quad (5)$$

where E is seasonal rainfall energy, and a and b are given by

$$a = 2.884 - 8.1209 F \quad (6)$$

$$b = 0.4681 + 0.7663 F \quad (7)$$

where F is soil erodibility. Based on soil texture, F is assigned a value between 4 (light textured, or sandy soil) and 6 (heavy textured, or clay soil). Additional adjustments are made based on hydrologic characteristics of the soil, tillage, soil structure, and crusting, giving a range of F values from 1 to 9.

C is computed as $C = e^{(-0.06i)}$ when $i < 50\%$, and as $C = (2.3 - 0.01i)/30$ when $i \geq 50\%$, where i is the percentage rainfall energy intercepted by the crop (used crop cover %).

X is computed as

$$X = (l)^{1/2} (0.76 + 0.53s + 0.076s^2) / 25.65 \quad (8)$$

This is derived from the USLE LS factor in use in the 1970's. The equation above converted the USLE LS factor to the metric system, and adjusted it to a 4.5%, 30m long slope. The variables l and s are length (m) and slope (%).

The USLE and SLEMSA are very similar in many respects. The major differences are in the climate variables and the cropping management approaches. The USLE climate factor contains both rainfall energy and rainfall intensity, whereas SLEMSA contains only rainfall energy. The USLE C value incorporates both tillage and cropping, while the C value for SLEMSA is a function of the interception of rainfall energy by crops. However, SLEMSA does capture some of the tillage component in the F value.

ABAG (German USLE)

Auerswald^[22] described the application of the USLE in Germany. Experimentally, most of the USLE factors were validated, and few if any modifications were needed to apply it under German conditions. One modification was that soils highly enriched in potassium were found to be

more erodible than expected, based on the soil erodibility nomogram of Wischmeier et al.^[23]

While no modification of the USLE was found necessary, ABAG factors were necessary for German conditions, specifically the R and C factors. Schwertmann et al.^[24] published a book for extension workers in Germany detailing the use of the USLE in Germany.

SOIL LOSS

Rosewell and Edwards^[25] developed a computer program that used the USLE approach to estimate average annual soil losses and to give recommendations on ways to reduce soil loss through changes in land and crop management practices. SOILOSS was developed for use in New South Wales, Australia. It used a slightly different equation for computing rainfall energy than that used in the USLE. It also used both the USLE nomogram for soil erodibility, plus experimentally derived values for NSW. In a later revision,^[26] Rosewell adopted LS values from RUSLE. From the earliest version of SOILOSS, the subfactor approach of Laflen et al.^[27] that was implemented in RUSLE was used to estimate the C value. It also contained P value estimates similar to those in the USLE, but some were from experimental work in NSW. It also included a benefit from sediment deposited in bank (terrace) channels.

Keats and Rosewell developed a SOILOSS user guide^[28] for high school students. It included the program, an introduction to the USLE, and complete parameterization for most Australian conditions. It included worksheets and examples.

In a number of ways, SOILOSS could be considered an early RUSLE, using much of the science and approaches that were eventually used in RUSLE. It was also developed in the time frame where RUSLE was developed.

Revised Universal Soil Loss Equation

RUSLE^[11] maintains the form of the USLE in terms of the factors. But, since it is operated on a computer, many improvements in computations, particularly those including interactions, have been incorporated. Major improvements in the revised USLE include improved R value map for the United States, time variation in soil erodibility, improved LS factor that incorporates modern science, a subfactor approach for computing the cover and management factor, and P values based on modeling analyses that include storm severity, ridge height, drainage, and off-grade contouring.

RUSLE has been implemented for use in erosion prediction by the Natural Resource Conservation Service in the United States. It is also used in several simulation models. RUSLE development continues, with

improvements in science, ease of use, supporting databases, and applications related to sediment deposition and delivery.

CONCLUSION

Empirical erosion modeling has been applied on much of the earth's land surface. The approaches have almost exclusively been to quantify factors that influence soil erosion, and then to combine these factors, usually as products of the factors. The development of empirical models has clearly been an evolutionary process, and almost all models owe all or much of their technology to the USLE.

REFERENCES

1. Haan, C.T. *Statistical Methods in Hydrology*; Iowa State Press: Ames, 1977; 394 pp.
2. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall-Erosion Losses—A Guide to Conservation Farming*; Agricultural Handbook No. 537; U.S. Department of Agriculture: Washington, 1978; 58 pp.
3. Hudson, N.W. Erosion control research. *Rhod. Agr. J.* **1957**, *54* (4), 297–323.
4. Zingg, A.W. Degree and length of land slope as it affects soil loss in runoff. *Agr. Eng.* **1940**, *21* (2), 59–64.
5. Smith, D.D. Interpretation of soil conservation data for field use. *Agr. Eng.* **1941**, *22* (5), 173–175.
6. Browning, G.M.; Parish, C.L.; Glass, J. A method for determining the use and limitations of rotation and conservation practices in the control of soil erosion in Iowa. *J. Am. Soc. Agron.* **1947**, *39* (1), 65–73.
7. Musgrave, G.W. The quantitative evaluation of factors in water erosion—A first approximation. *J. Soil Water Conserv.* **1947**, *2* (3), 133–138.
8. Wischmeier, W.H.; Smith, D.D. *A Universal Equation for Predicting Rainfall-Erosion Losses—An Aid to Conservation Farming in Humid Regions*; ARS Special Report 22–66; Agricultural Research Service, U.S. Department of Agriculture: Washington, 1961; 11 pp.
9. Williams, J.R. Sediment yield prediction with universal equation using runoff energy factor. In *Present and Prospective Technology for Predicting Sediment Yields and Sources*; Sediment-Yield Workshop, USDA Sedimentation Laboratory, Oxford, Mississippi, Nov 28–30, 1975; ARS-S-40; Agricultural Research Service, U.S. Department of Agriculture: Washington, 1975; 244–252.
10. Onstad, C.A.; Foster, G.R. Erosion modeling on a watershed. *Trans. Am. Soc. Agric. Eng.* **1975**, *18* (2), 288–292.
11. Renard, K.G.; Foster, G.R.; Weesies, G.A.; McCool, D.K.; Yoder, D.C. *Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE)*; Agricultural Handbook No. 703; U.S. Department of Agriculture: Washington, 1997; 384 pp.
12. Meyer, L.D. Evolution of the universal soil loss equation. *J. Soil Water Conserv.* **1984**, *39* (2), 99–104.

13. Wischmeier, W.H.; Smith, D.D. Predicting rainfall erosion losses—A guide to conservation planning. *Trans. Am. Geophys. Un.* **1958**, *39* (2), 285–291.
14. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall-Erosion Losses from Cropland East of the Rocky Mountains—Guide for Selection of Practices for Soil and Water Conservation*; Agricultural Handbook No. 282; U.S. Department of Agriculture: Washington, 1965; 47 pp.
15. Renard, K.G.; Foster, G.R.; Weesies, G.A.; Porter, J.P. RUSLE: Revised Universal Soil Loss Equation. *J. Soil Water Conserv.* **1991**, *46* (1), 30–33.
16. Williams, J.R.; Hiler, E.A.; Baird, R.W. Prediction of sediment yields from small watersheds. *Trans. Am. Soc. Agric. Eng.* **1971**, *14* (6), 1157–1162.
17. Williams, J.R.; Berndt, H.D. Sediment yield computed with universal equation. *J. Hydr. Div. Proc. Am. Soc. Civil Eng.* **1972**, *98* (HY12), 2087–2098.
18. Sharpley, A.N.; Williams, J.R. *EPIC-Erosion/Productivity Impact Calculator: 1. Model Documentation*; Technical Bulletin No. 1768; U.S. Department of Agriculture: Washington, 1990; 235 pp.
19. Foster, G.R.; Meyer, L.D.; Onstad, C.A. An erosion equation derived from basic erosion principles. *Trans. Am. Soc. Agric. Eng.* **1977**, *20* (4), 678–682.
20. Elwell, H.A. *Modeling Soil Losses in Zimbabwe Rhodesia*. Paper Number 79–2051; Institute of Agricultural Engineering: Harare, 1979.
21. Elwell, H.A. Modeling soil losses in southern Africa. *J. Agr. Eng. Res.* **1978**, *23*, 117–127.
22. Auerswald, K. Empirical soil erosion models. *Personal Letter*, 4-12-2000, 2000.
23. Wischmeier, W.H.; Johnson, C.B.; Cross, B.V. A soil erodibility nomograph for farmland and construction sites. *J. Soil Water Conserv.* **1971**, *26* (5), 189–193.
24. Schwertmann, U.; Vogl, W.; Kainz, M. *Bodenerosion durch Wasser-Vorhersage des Abtrags und Bewertung von Gegenmaßnahmen*; Ulmer Verlag: Stuttgart, 1987; 64 pp.
25. Rosewell, C.J.; Edwards, K. *SOILOSS—A Program to Assist in the Selection of Management Practices to Reduce Erosion*; Technical Handbook No. 11; Soil Conservation Service of New South Wales: Sydney, 1988; 71 pp.
26. Rosewell, C.J. *SOILOSS-A Program to Assist in the Selection of Management Practices to Reduce Erosion*, 2nd Ed.; A Technical Handbook No. 11; Soil Conservation Service of New South Wales: Sydney, 1993; 108 pp.
27. Laflen, J.M.; Foster, G.R.; Onstad, C.A. Simulation of individual storm soil loss for modeling impact of soil erosion on crop productivity. In *Soil Erosion and Conservation*; El-Swaify, S.A., Moldenhauer, W.C., Lo, A., Eds.; Soil Conservation Society of America: Ankeny, 1985; 285–295.
28. Keats, J.; Rosewell, C. *SOILOSS-High School User Guide*; Department of Conservation and Land Management: Sydney, 1993; 87 pp.

Water Erosion: Hybrid Models

Jeffrey G. Arnold

Grassland Soil/Water Research Laboratory, U.S. Department of Agriculture (USDA), Temple, Texas, U.S.A.

Kevin W. King

U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Temple, Texas, U.S.A.

Jimmy R. Williams

Texas A&M University, Temple, Texas, U.S.A.

Abstract

Simulation models are among the best tools available for analyzing soil erosion and conservation management. Models can project the consequences of alternative management, conservation planning, or policy-level activities and substantially reduce the cost of managing soil and water resources. Numerous models have been developed to estimate soil erosion and the impact of land management on erosion. Equations have been integrated into comprehensive hybrid models that directly simulate hydrology, plant growth, nutrient cycling, and land management impacts on soil erosion.

INTRODUCTION

Each year, soil erosion causes billions of dollars of damage in the world.^[1] Simulation models are among the best tools available for analyzing soil erosion and conservation management. Models can project the consequences of alternative management, conservation planning, or policy-level activities and substantially reduce the cost of managing soil and water resources. Numerous models have been developed to estimate soil erosion and the impact of land management on erosion. Early models such as the universal soil loss equation (USLE)^[2] were empirical and predicted average annual erosion rates using simple nomograph procedures. The USLE and its subsequent improvements have been modified and enhanced.^[3,4] These equations have been integrated into comprehensive hybrid models that directly simulate hydrology, plant growth, nutrient cycling, and land management impacts on soil erosion. Then, the models use predicted soil erosion to estimate nutrient, pesticide, and bacteria transport; change in soil productivity; and carbon dynamics.

EROSION EQUATIONS

The USLE has been used throughout the world to estimate average annual sheet and rill erosion. Several improved forms of the USLE have been developed that include the Onstad–Foster equation^[4] and modified universal soil loss equation (MUSLE).^[3] The equations are identical except for their energy components. The USLE strictly depends on rainfall as an indicator of erosive energy. The MUSLE and

its variations use only runoff variables to simulate erosion and sediment yield. Runoff variables increase the prediction accuracy, eliminate the need for a delivery ratio, and enable the equation to give single-storm estimates of sediment yields. The USLE gives only average annual estimates. The Onstad–Foster equation contains a combination of the USLE and MUSLE energy factors. The empirical erosion model equations are of the following form:

$$Y = \chi(K)(C)(P)(LS) \quad (1)$$

$$\chi = EI \quad \text{for USLE} \quad (2)$$

$$\chi = 0.646EI + 0.45(Qq_p)^{0.33} \quad \text{for Onstad–Foster} \quad (3)$$

$$\chi = 1.586(Qq_p)^{0.56}A^{0.12} \quad \text{for MUSLE} \quad (4)$$

where Y is the sediment yield (t/ha), K is the soil erodibility factor, C is the crop management factor, P is the erosion control practice fact, LS is the slope length and steepness factor, Q is the runoff volume (mm), q_p is the peak runoff rate (mm/h), and A is the watershed area (ha).

COMPREHENSIVE HYBRID MODELS

Numerous hybrid models have been developed including Revised USLE (RUSLE),^[5] groundwater loading effects of agricultural management systems (GLEAMS),^[6] and agricultural nonpoint source pollution model (AGNPS).^[7] Three comprehensive hybrid models with varying spatial

scales will be discussed in more detail: field scale—environmental policy impact calculator (EPIC), small watershed scale—agricultural policy/environmental extender (APEX), and river basin scale—soil and water assessment tool (SWAT). All three models use modified versions of USLE for erosion and link with components for plant growth, hydrology, and management. APEX and SWAT also include components for channel sediment routing and pond and reservoir deposition.

The EPIC model^[8] was developed in the early 1980s to assess the effect of erosion on productivity. The model has since been expanded and refined to allow the simulation of many processes important in agricultural management.^[9] EPIC is a continuous simulation model (daily time step) that can be used to determine the effect of management strategies on water quality. The drainage area considered by EPIC is generally a field-sized area, up to 100 ha, where weather, soils, and management systems are assumed to be homogeneous. The major components in EPIC are weather simulation, hydrology, erosion, nutrient cycling, pesticide fate, plant growth, soil temperature, tillage, economics, and plant environment control. EPIC can be used to compare management systems and their effects on sediment nitrogen, phosphorus, and pesticides. The management operations that can be simulated are crop rotations, tillage operations, irrigation scheduling, drainage, furrow diking, liming, grazing, manure handling, and nutrient and pesticide application rates and timing.

The APEX model^[10] was developed to extend the EPIC model capabilities to whole farms and small watersheds. In addition to the EPIC functions, APEX has components for routing water, sediment, nutrients, and pesticides across complex landscapes and channel systems to the watershed outlet. A watershed can be subdivided to assure that each subarea is relatively homogeneous in terms of soil, land use, management, etc. The routing mechanisms provide for the evaluation of interactions between subareas involving surface run-on, return flow, sediment deposition and degradation, and nutrient transport. Water quality in terms of nitrogen (ammonium, nitrate, and organic), phosphorus (soluble and adsorbed/mineral and organic), and pesticide concentrations may be estimated for each subarea and at the watershed outlet. APEX is limited in watershed size because of the detailed crop management system and because daily rainfall is uniformly distributed over the entire watershed.

SWAT^[11] is a complex, conceptual model with spatially explicit parameterization. It is a continuous time model that operates on a daily time step. The objective in model development was to predict the impact of management on water, sediment, and agricultural chemical yields in ungauged basins. To meet this objective, the model is physically based (as calibration is not possible on ungauged basins) on readily available inputs, is computationally efficient to operate on large watersheds and in a reasonable time, and is capable of continuous simulation over long time periods necessary for computing the effects of management changes. Major model

components include weather, hydrology, erosion/sedimentation, soil temperature, plant growth, nutrients, pesticides, groundwater flow, land management, stream routing, and pond/reservoir routing. SWAT allows a large basin to be subdivided into grid cells or subwatersheds and allows point source inputs and reservoirs to be included in the drainage network. Basins with size ranging from 100 to 100,000 km² have been simulated.^[12] Geographic information system (GIS) interfaces have been developed using GRASS and ArcView to automate model input development and spatially display model outputs.^[13]

MODEL COMPONENTS THAT INTERACT WITH EROSION EQUATIONS

Fig. 1 shows major model components and their interaction with other components and management. The major process components that influence erosion are hydrology, plant growth, residue decay, and management (rotations, tillage, irrigation, fertilizer, and manure).

Hydrology

The primary driver of soil erosion by water is surface water runoff that is caused by rainfall, snowmelt, and irrigation. To simulate surface runoff accurately, an estimate of soil moisture at the start of the runoff event is critical. For continuous time models, the components of the soil water balance must be simulated between events. The main components of the soil water balance are canopy storage, surface runoff, infiltration, snow fall and melt, evapotranspiration (ET), percolation, and lateral soil flow (Fig. 2). The hybrid models discussed here use a daily time step for most water balance processes with the option of subdaily time step for the rainfall/runoff process. Hybrid models using empirical erosion models such as MUSLE and Onstad–Foster require peak rate and total runoff volume from the hydrology routines.

Plant Growth

Simulation of plant growth is crucial for accurate prediction of soil erosion. Plant growth impacts water extraction from the soil (ET), canopy and residue cover, and nutrient uptake. In most climates, the largest component of the water balance is ET. ET indirectly affects erosion by modifying soil moisture between rainfall events and thus impacting soil erosion. Canopy and residue cover have direct impact on the erosion equations. For the hybrid models, a relationship between cover (canopy and residue) and the USLE C-factor was developed.^[8]

Residue Decay

Decay of plant residue (including standing, flat, buried, and roots) reduces the amount of residue protecting the soil surface from erosion. Decay is normally simulated as a function of temperature; soil moisture; and soil carbon,

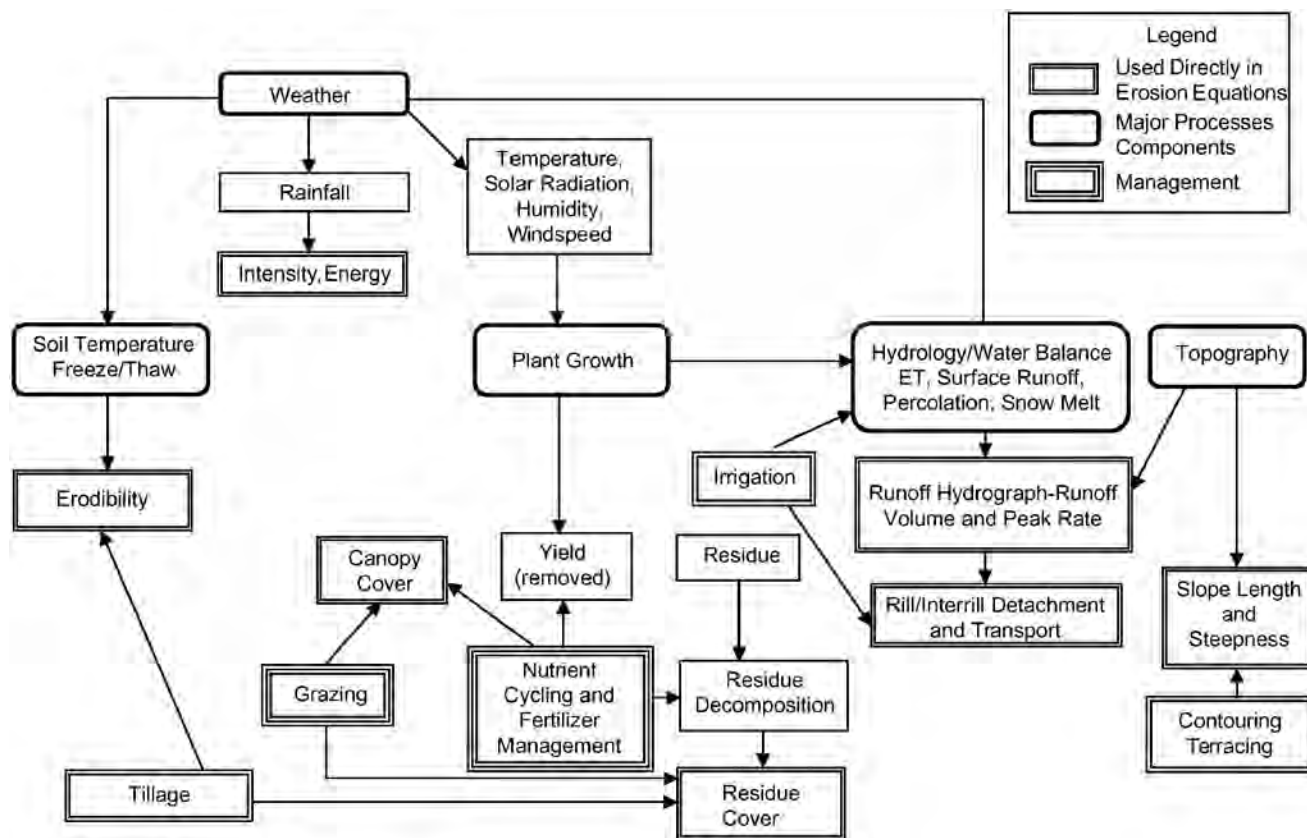


Fig. 1 Major hybrid model components and their interactions with management practices.

nitrogen and phosphorus contents. The hybrid models simulate nutrient cycling and track nutrient pools in the soil and nutrient concentrations in the plants and residue. The decay rates can then be calculated as a function of the soil C/N and C/P ratios and the composition of plant residue.

Management

Plant growth management is simulated by most comprehensive models. Crop rotations, planting and harvest dates, the length of growing season, cuttings, and grazing all impact the water balance and the amount of canopy and ground cover.

Tillage

Tillage operations remove plant residue from the soil surface (standing and flat) and incorporate residue into the soil, reducing cover. Tillage also mixes nutrients and changes soil properties such as erodibility, bulk density, and hydraulic conductivity.

Irrigation

Surface irrigation often produces surface runoff and erosion. Irrigation also affects the soil water balance by increasing soil moisture and thus increasing surface runoff and erosion during rainfall events.

Fertilizer and manure management

Application of manure and commercial fertilizers influences soil carbon, nitrogen, and phosphorus and thus influences residue decomposition. Increased fertilizer application can increase plant growth and potentially increase canopy and residue cover. Soil properties such as erodibility and bulk density are altered by fertilizer and manure applications.

MODEL APPLICATIONS

Applications of hybrid soil erosion models include farm planning, erosion productivity,^[14] sustainable farming systems, carbon sequestrations,^[15] climate change,^[16] and total maximum daily loads (TMDLs).

EPIC Applications

1985 RCA appraisal

The USDA used EPIC to simulate the soil-climate-plant management processes in agricultural production and to estimate the impact of soil erosion on resource productivity and fertilizer requirements for the 1985 Resources Conservation Act appraisal.^[14] EPIC results showed that

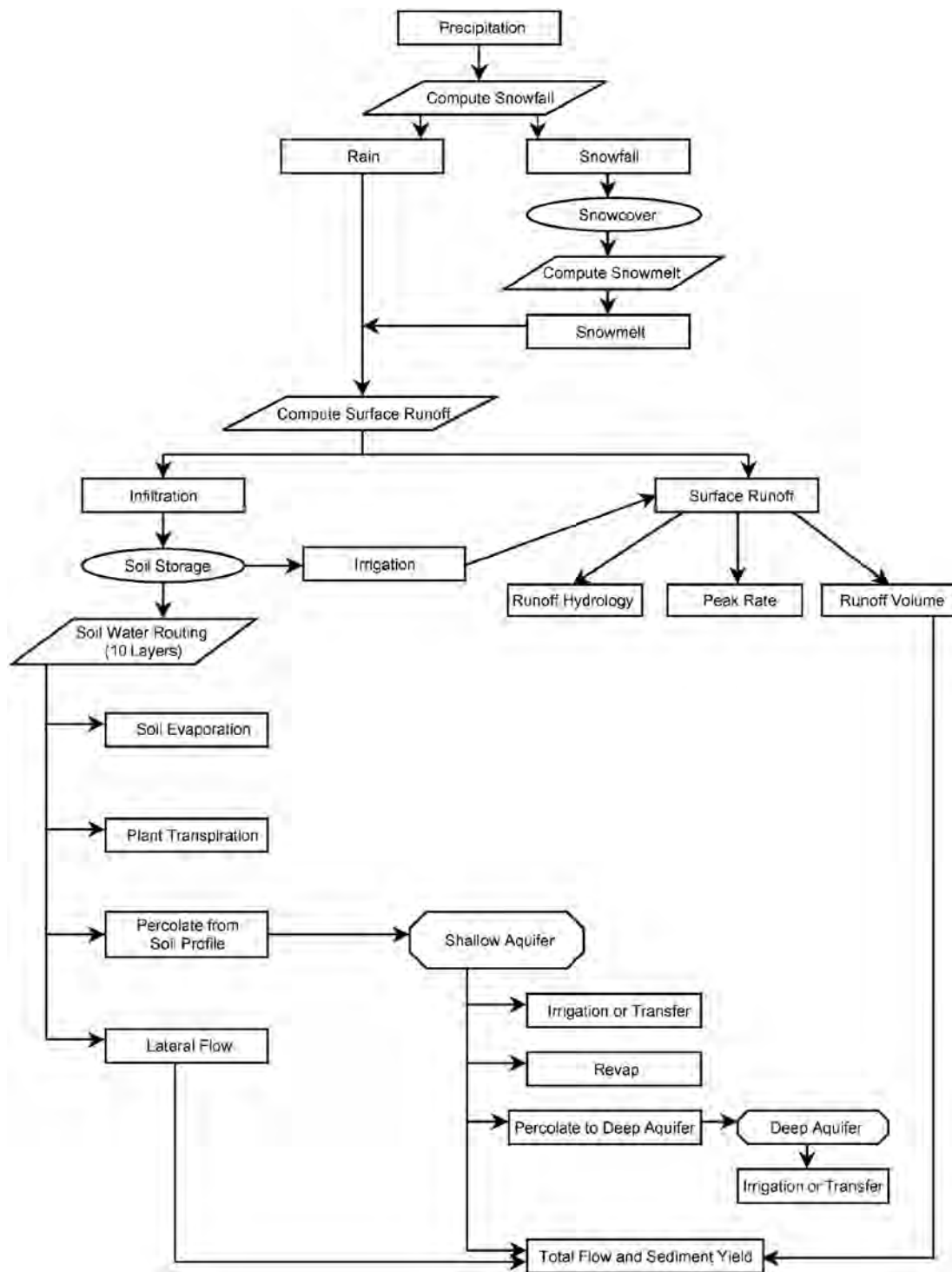


Fig. 2 Major components of the hydrologic balance in hybrid models.

if cropping patterns and the mix of management, tillage, and conservation practices continued for 100 years, sheet and rill erosion will exceed soil loss tolerance on 127 million acres. Annual fertility requirements are estimated to increase by 798 and 672 million pounds of nitrogen and phosphate, respectively.

Environmental protection agency (EPA) nutrient movement study

EPA used EPIC to examine the impacts of tillage on nitrogen and phosphorus movement. Four tillage strategies were examined using 100-year EPIC simulations at 100 sites in

Illinois. Simulated crop yields closely corresponded to expected yields. No till corn/soybean rotations were found to reduce sediment-attached nitrogen and phosphorus in runoff relative to conventional till continuous corn.^[17]

APEX Applications

Animal waste recycling

The APEX model has been used to study animal water disposal systems in the Upper North Bosque watershed in Texas. Various forage production systems and soils were simulated to illustrate the interactions among crops, manure recycling, and nutrient losses.

Buffer analysis

APEX was used to evaluate buffer grass strip efficiency under a variety of soils, climate, slope, and management across the United States. Relationships of buffer trapping efficiency of sediment and nutrients were developed for use in regional buffer analysis studies and for use in watershed models.^[18]

SWAT Applications

Hydrologic unit model of the United States

The Natural Resources Conservation Service (NRCS) used the SWAT model for the 1997 Resource Conservation Assessment. The model was linked to national economic models and used for national policy planning of water supply and quantity of the 18 major river basins of the United States. GISs were utilized to integrate SWAT with national soils, land use, and elevation databases.^[19]

TMDL analysis

There are approximately 15,000 water bodies in the United States identified by EPA as impaired for various uses. For each of these, states must estimate the severity of the problem (develop a TMDL) and determine potential solutions. SWAT has been selected by EPA for use in TMDL analysis and is being used to determine the impact of proposed management practices on attaining water quality standards.^[20]

REFERENCES

- Committee on Conservation Needs and Opportunities. *Soil Conservation: Assessing the National Resource Inventory*; National Academy Press: Washington, 1986; Vol. 1, 114 pp.
- Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses—A Guide to Conservation Planning*; Agric. Handbook No. 537; U.S. Department of Agriculture: Washington, 1978.
- Williams, J.R.; Berndt, H.D. Sediment yield prediction based on watershed hydrology. *T. ASAE* **1977**, *20* (6), 1100–1104.
- Onstad, C.A.; Foster, G.R. Erosion modeling on a watershed. *T. ASAE* **1975**, *18* (2), 288–292.
- Renard, K.G.; Ferreira, V.A. RUSLE model description and data base sensitivity. *J. Environ. Qual.* **1993**, *22* (3), 458–466.
- Leonard, R.A.; Knisel, W.G.; Still, D.A. GLEAMS: Groundwater loading effects of agricultural management systems. *T. ASAE* **1987**, *30* (5), 1403–1418.
- Young, R.A.; Onstad, C.A.; Bosch, D.D.; Anderson, W.P. *AGNPS, Agricultural Nonpoint Source Pollution Model—A Watershed Analysis Tool*; Conservation Research Report 35; U.S. Department of Agriculture: Washington, 1987.
- Williams, J.R.; Jones, C.A.; Dyke, P.T. A Modeling approach to determining the relationship between erosion and soil productivity. *T. ASAE* **1984**, *27*, 129–144.
- Sharpley, A.N.; Williams, J.R. *EPIC—Erosion/Productivity Impact Calculator: Model Documentation*; U.S. Department of Agriculture Technical Bulletin No. 1768, USDA: Washington, 1990; 235 pp.
- Williams, J.R.; Arnold, J.G.; Srinivasan, R.; Ramanarayanan, T.S. APEX: A new tool for predicting the effects of climate change and CO₂ changes on erosion and water quality; NATO ASI Ser I. **1998**, *55*, 441–449.
- Arnold, J.G.; Srinivasan, R.; Muttiah, R.S.; Williams, J.R. Large area hydrologic modeling and assessment. Part I Model development. *J. Am. Water Resour. As.* **1998**, *34* (1), 73–89.
- Arnold, J.G.; Muttiah, R.S.; Srinivasan, R.; Allen, P.M. Regional estimation of base flow and groundwater recharge in the upper Mississippi basin. *J. Hydrol.* **2000**, *227* (2000), 21–40.
- DiLuzio, M.; Srinivasan, R.; Arnold, J.G. Watershed Oriented Non-Point Pollution Assessment Tool. In *7th Annual Conference on Computers in Agriculture*; ASAE: Orlando, 1998; 7 pp.
- Putnam, J.; Williams, J.R.; Sawyer, D. Using the erosion productivity impact calculator (EPIC) model to estimate the impact of soil erosion for the 1985 RCA appraisal. *J. Soil Water Conserv.* **1985**, *43* (4), 321–326.
- Izaurrealde, R.C.; Haugen-Kozyra, K.H.; Jans, D.C.; McGill, W.B.; Grant, R.F.; Hiley, J.C. *Soil Dynamics: Measurement, Simulation and Site-to-region Scale-up*; Lewis: Boca Raton, 2001.
- Rosenberg, N.J.; Epstein, D.L.; Wang, D.; Vail, L.; Srinivasan, R.; Arnold, J.G. Possible impacts of global warming on the hydrology of the Ogallala aquifer region. *J. Climate* **1999**, *42*, 677–692.
- Phillips, D.L.; Hardin, P.D.; Benson, V.W.; Baglio, J.V. Using the national resources inventory and the EPIC model to evaluate the impact of alternative agricultural management practices in Illinois. *J. Soil Water Conserv.* **1993**, *48* (5), 449–457.
- Rodriguez, O.; Williams, J.R.; Arnold, J.G. Buffer grass strip efficiency evaluated with apex as input for regional applications (Draft). 2001.
- Srinivasan, R.S.; Arnold, J.G.; Jones, C.A. Hydrologic modeling of the United States with the soil and water assessment tool. *Water Resour. Devel.* **1998**, *14* (3), 315–325.
- Saleh, A.; Arnold, J.G.; Gassman, P.W.; Hauck, L.W.; Rosenthal, W.D.; Williams, J.R.; McFarland, A.M.S. Application of SWAT for the upper north Bosque watershed. *T. ASAE* **2000**, *43* (5), 1077–1087.

Water Erosion: Modeling

Calvin W. Rose

Faculty of Environmental Sciences, Griffith University, Nathan Campus, Nathan, Queensland, Australia

Abstract

An outline is given to the development from earlier statistically based models of water-driven soil erosion to models designed to represent the physical processes involved. Examples of dynamic and steady-rate process models are also given.

INTRODUCTION

Modeling soil erosion by water was initially developed in the context of agricultural productivity and land degradation.^[1–4] Modeling was subsequently stimulated by off-site concerns for the water quality of streams, lakes, and oceans.^[5,6] Soil erosion by water is also recognized as one process in the (most) descriptive long-term, land-forming models of geomorphologists and geographers,^[7,8] which include mass movement and other forms of erosion. The concern that soil erosion might ultimately expose buried contaminants has encouraged modeling as a possible landscape change over thousands of years.^[9,10]

Early erosion models used statistical techniques to summarize large bodies of data, giving the effect of environmental and agricultural management factors on net soil loss from agricultural plots.^[1,2] The Revised Universal Soil Loss Equation (RUSLE),^[11] a familiar example, developed from the USLE summarizes erosion experiments in U.S. field plots.^[1] RUSLE allows prediction of annual expected net soil loss from a plot of uniform slope throughout an entire crop rotation, using databases for climate, vegetation, and field-management operations that are available for the United States. The annual soil loss per unit area, A , is given by as follows:

$$A = RKLSCP$$

where R is a rainfall-runoff factor, K is soil erodibility factor, L is slope length factor, S is slope steepness factor, C is cover management factor, and P is support practices factor.

Later, soil erosion modeling developments primarily use deterministic models, which seek to represent the physical, hydrological, and sediment transport–deposition processes involved.

SOIL EROSION PROCESSES

Water-induced soil erosion involves downslope flow of water with a sediment concentration that depends on the net

balance of two rapid but opposing types of processes.^[12–14] Sediment removed from the soil surface adds to the sediment concentration, a process countered by deposition, which continuously returns sediment to the soil surface at a rate depending predominantly on the distribution of size and density of the sediment.^[13] The rate of soil removal depends on an interaction of rainfall and overland-flow characteristics with soil strength-related properties, which can be affected by subsurface hydrology.^[15–21]

The impact of raindrop on unprotected soil is commonly the initial soil-removal mechanism.^[13,22] However, when an excess of rainfall over infiltration rate occurs, the resulting overland flow both transports sediment and exerts a shear stress on the soil, τ . This shear stress arises chiefly from turbulence,^[15,17,18] and its rate of action is called the stream power, Ω

$$\Omega = \tau V$$

where V is the velocity of flow.

For cultivated soil, overland flow-driven erosion processes usually become dominant over rainfall-driven processes when Ω exceeds approximately 0.2 W m^{-2} .^[23]

DYNAMIC STOCHASTIC AND DETERMINISTIC MODELS OF SOIL EROSION

The rates of sediment removal from, and return to, the soil surface in deposition can be described stochastically.^[24] The probability density of resting on the soil surface is a function of time alone, but the probability to describe motion involves both space and time. This leads to two simultaneous partial differential equations, one related to the deposited or stationary mass and the second to sediment in motion.^[24] These two equations are formally identical with the two partial differential equations derived using deterministic mass-conservation considerations.^[16,24]

Fig. 1 gives a deterministic representation of rainfall-driven erosion, recognizing that soil consists of a range of

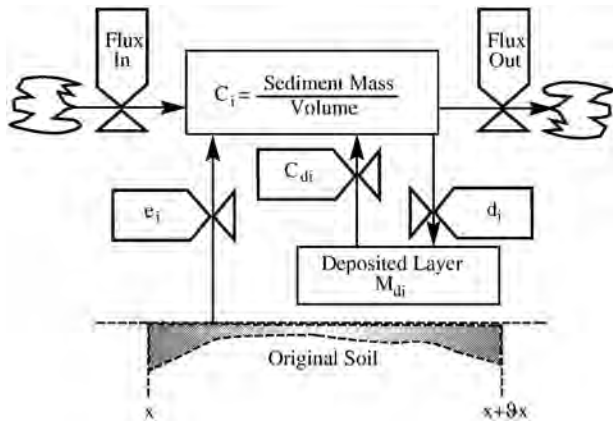


Fig. 1 Forrester-style flow diagram describing the interaction of rainfall detachment and redetachment processes among the original soil, the layer formed by deposition, and the sediment flux. The water layer with sediment concentration c_i (for size class i) and the deposited layer of mass per unit area m_{di} are shown artificially elevated for clarity. Rates of processes exchanging sediment are shown by valve symbols.

Source: From Hairsine & Rose.^[16]

particle sizes. For a typical size class (i), e_i is the rate of detachment from the original soil matrix. Deposition at rate (d_i) from sediment in the overland flow forms a deposited layer of mass per unit area (m_{di}), from which sediment is redetached at rate e_{di} . Let c_i denotes the concentration of sediment in i at any time (t) in depth of water (D), with volumetric flow per unit width (q) in the x direction. Then mass conservation of sediment of i flowing over the soil surface requires that^[16,25]

$$\frac{\partial}{\partial t}(Dc_i) + \frac{\partial}{\partial x}(qc_i) = e_i + e_{di} - d_i \quad (1)$$

Mass conservation of sediment in the deposited layer (Fig. 1) requires that

$$\frac{\partial}{\partial t}(m_{di}) = d_i - e_{di} \quad (2)$$

Solution of Eqs. 1 and 2 (or corresponding equations when flow-driven processes are also important) involves numerical or approximate analytical methods.^[25–28] A major purpose of such models is to test the physical understanding of basic dynamic soil erosion and deposition processes. Dynamic models also help in choosing approximations for simpler steady-rate models, which are more suitable for application in field-predictive applications. Use of steady-rate models avoids the need to solve partial differential equations in space and time.

Especially at long-time scales and catchment space scales, there is a change in land surface elevation that results from net erosion or net deposition, an issue ignored

in equations such as Eqs. 1 and 2, but crucial in seeking to describe landscape evolution.^[7,8] A quantitative general model of channel network growth and hillslope evolution has been developed whose output resembles many observed landscape features.^[29–32] In this model, surface denudation is represented as the sum of fluvial transport processes, dependent on discharge and slope, and a slower analog of diffusive processes dependent on slope alone. Digital terrain modeling is used to determine drainage areas, and the landform is adjusted through time in response to erosion processes.

Three physically based steady-rate models that have received significant application in land use contexts are outlined below.

STEADY-RATE MODELS OF WATER EROSION

Water Erosion Prediction Project

Developed by the U.S. Department of Agriculture, the hillslope version of Water Erosion Prediction Project (WEPP) is a comprehensive, continuous simulation model that draws on U.S. databases describing climate, soils, tillage, and crop parameters.^[33,34] Runoff is generated from rainfall input using an infiltration equation.^[34] The peak predicted runoff rate is the steady-rate value assumed for the erosion event.

Sediment sourced from the inter-rill area is delivered to rills, and net erosion within rills can add to transported sediment. If predicted total sediment transport exceeds the maximum transport limit, a separate equation acknowledges that net deposition will occur, representing a mass conservation of transported sediment. A flow chart of the WEPP profile model is shown in Fig. 2.

U.S. databases provide the 64 parameters needed to run the model. However, experience outside the United States indicates that infiltration and erodibility parameters, in particular, must be obtained from in-country experiments if predictions of soil and water loss are to be of useful accuracy.^[35] More versions of the WEPP model allow for simulation of small watersheds.^[36]

Griffith University Erosion System Template

Program Griffith University Erosion System Template (GUEST)^[37] is based on physical theory.^[13,16–18,38] The simpler of two options in GUEST defines an empirical erodibility parameter β and uses the mean velocity of sediment settling as a soil characteristic.^[13] GUEST also addresses rilling in a physical manner.^[13,38] If total runoff amount is alone measured, an effective runoff rate for an erosion event can be calculated from rainfall rate.^[39,40] If β has been previously determined or can be estimated, then the program can be used to predict soil loss.

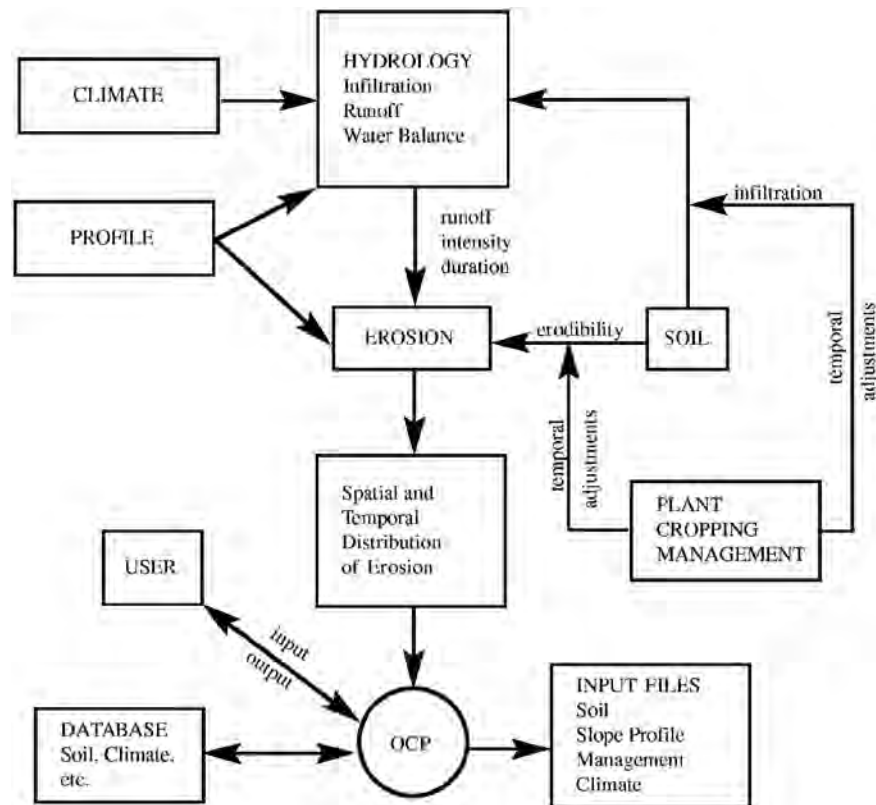


Fig. 2 Flow chart illustrating the components of the hillslope profile version of WEPP.

Source: From Lane & Nearing.^[34]

GUEST assumes that a constant fraction of the excess stream power $\Omega - \Omega_0$, where Ω_0 is a threshold value, is used in flow-driven erosion.^[13,18] The maximum sediment concentration or transport limit (c_t) is derived theoretically.^[13,18] The erodibility parameter β is given by

$$\beta = (\ln \bar{c}) / (\ln \bar{c}_t) \quad (3)$$

where $\bar{c}$ is the average sediment concentration and $\bar{c}_t$ is the average value of c_t during an erosion event. How the relation between $\bar{c}$ and Ω is controlled by the value of β is shown in Fig. 3.

European Soil Erosion Model

European soil erosion model (EUROSEM)^[41,42] describes erosion for single events on single-slope segments (as does GUEST), uses an experimentally based relationship for the transport limit,^[43] and is linked to the KINEROS hydrologic model.^[44] The numerical-solution procedure in KINEROS provides the basis for sediment discharge calculation. Net deposition occurs if the calculated sediment concentration exceeds the transport limit. Sediment removed by rainfall impact is delivered to rills, which transport the sediment (as in WEPP).

EUROSEM represents canopy protection against rainfall impact, adding drop impact due to throughfall. The effect of different cultivation practices on surface water storage is also represented.

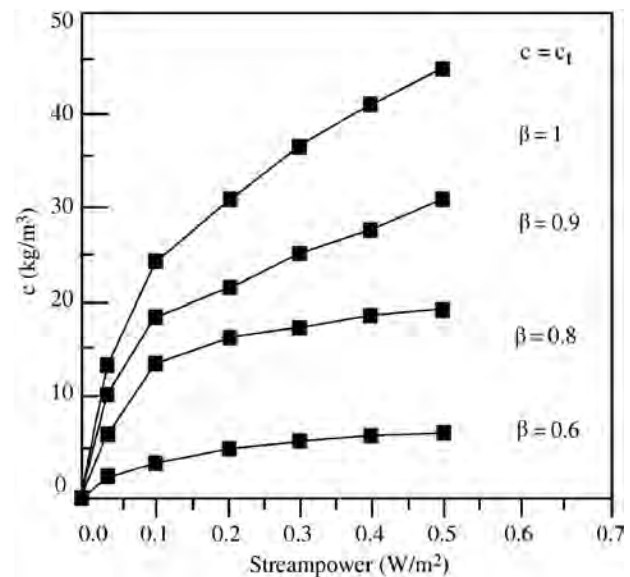


Fig. 3 The relationship between sediment concentration, c , and streampower for various values of the erodibility parameter β given in Eq. 3. For $\beta = 1$ (the upper curve), $c = c_t$, the sediment concentration at the transport limit. (After Rose, where assumed values of other parameters are given.)

Source: From Rose.^[14]

CONCLUSION

Theoretically driven process-based models are available that have demonstrated their ability to interpret and guide field experimentation on soil erosion driven by rainfall and runoff. Examples of such models are outlined, and similarities and differences between such models are noted.

REFERENCES

1. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses—A Guide to Conservation Planning*; Agriculture Handbook No. 537; U.S. Department of Agriculture: Washington, 1978.
2. Romkens, M.J.M. The soil erodibility factor: A perspective. In *Soil Erosion and Conservation*; El-Swaify, S., Moldenhauer, W., Lo, A., Eds.; Soil Conservation Society of America: Ankeny, 1985; 237–347.
3. El-Swaify, S.; Moldenhauer, W.; Lo, A. *Soil Erosion and Conservation*; Soil Conservation Society of America: Ankeny, 1985.
4. Hurni, H.; Tato, K., Eds. *Erosion, Conservation, and Small-Scale Farming*; Geographica Bernesia Wadsworth Publishing Company: Missouri, 1992.
5. Knisel, W. Jr., Ed. *CREAMS: A Field-Scale Model for Chemicals, Runoff and Erosion from Agricultural Management Systems*; Conser. Res. Rep. No. 26; U.S. Department of Agriculture: Washington, 1980.
6. Ghadiri, H.; Rose, C.W. *Modeling Chemical Transport in Soils: Natural and Applied Contaminants*; Lewis Publishers: London, 1992.
7. Kirkby, M.J. Hillslope process-response models based on the continuity equation. In *Slopes: Form and Process*; Brunsdon, D., Ed.; Spec. Publ. 3; Institute of British Geographers: London, 1971; 15–30.
8. Huggett, R.J. Dissipative systems: Implications for geomorphology. *Earth Surf. Process.* **1998**, *13*, 45–49.
9. Evans, K.G. Methods for assessing mine site rehabilitation design for erosion impact. *Aust. J. Soil Res.* **2000**, *38* (2), 231–248.
10. Hancock, G.R.; Evans, K.G.; Willgoose, G.R.; Moliere, D.R.; Saynor, M.J.; Loch, R.J. Medium-term erosion simulation of an abandoned mine site using the SIBERIA landscape evolution model. *Aust. J. Soil Res.* **2000**, *38* (2), 249–263.
11. Renard, K.G.; Foster, G.R.; Weesies, G.A.; Porter, J.P. RUSLE: Revised universal soil loss equation. *J. Soil Water Conserv.* **1991**, *46*, 30–33.
12. Rose, C.W. Developments in soil erosion and deposition models. In *Advances in Soil Science*; Stewart, B.A., Ed.; Springer-Verlag: New York, 1985; Vol. 2, 1–63.
13. Rose, C.W. Erosion and sedimentation. In *Hydrology and Water Management in the Humid Tropics—Hydrological Research Issues and Strategies for Water Management*; Bonell, M., Hufschmidt, M.M., Gladwell, J.S., Eds.; Cambridge University Press: Cambridge, 1993; 301–343.
14. Rose, C.W. Research progress on soil erosion processes and a basis for soil conservation practices. In *Soil Erosion Research Methods*; Lal, R., Ed.; 2nd Ed.; Soil and Water Conservation Society: Ankeny, 1994; 159–178.
15. Foster, G.R. Modeling the erosion process. In *Hydrologic Modeling of Small Watersheds*; Hann, C.T., Ed.; Monograph No. 5; Am. Soc. Agr. Eng.: St. Joseph, 1982; 297–379.
16. Hairsine, P.B.; Rose, C.W. Rainfall detachment and deposition: Sediment transport in the absence of flow-driven processes. *Soil Sci. Soc. Am. J.* **1991**, *55*, 320–324.
17. Hairsine, P.B.; Rose, C.W. Modeling water erosion due to overland flow using physical principles: I. Uniform flow. *Water Resour. Res.* **1992**, *28*, 237–243.
18. Hairsine, P.B.; Rose, C.W. Modeling water erosion due to overland flow using physical principles: II. Rill flow. *Water Resour. Res.* **1992**, *28*, 245–250.
19. Proffitt, A.P.B.; Hairsine, P.B.; Rose, C.W. Modeling soil erosion by overland flow: Application over a range of hydraulic conditions. *Trans. ASAE* **1993**, *36*, 1743–1753.
20. Rose, C.W.; Hairsine, P.B.; Proffitt, A.P.B.; Misra, R.K. Interpreting the role of soil strength in erosion processes. *Catena Supp.* **1990**, *17*, 153–165.
21. Huang, C.; Gabbard, D.S.; Norton, L.D.; Laflen, J.M. Effects of hillslope hydrology and surface condition on soil erosion. In *Advances in Geology*; Rohdenburg, M., Ed.; 31st Ed.; Catena Verlag GMBH: Reiskirchen, 1998; 257–262.
22. Proffitt, A.P.B.; Rose, C.W.; Hairsine, P.B. Rainfall detachment and deposition: Experiments with low slopes and significant water depths. *Soil Sci. Soc. Am. J.* **1991**, *55*, 325–332.
23. Proffitt, A.P.B.; Rose, C.W. Soil erosion processes I. The relative importance of rainfall detachment and runoff entrainment. *Aust. J. Soil Res.* **1991**, *29*, 671–683.
24. Lisle, I.G.; Rose, C.W.; Hogarth, W.L.; Hairsine, P.B.; Sander, G.C.; Parlange, J.-Y. Stochastic sediment transport in soil erosion. *J. Hydrol.* **1997**, *204*, 217–230.
25. Sander, G.C.; Hairsine, P.B.; Rose, C.W.; Cassidy, D.; Parlange, J.-Y.; Hogarth, W.L.; Lisle, I.G. Unsteady soil erosion model, analytical solutions and comparison with experimental results. *J. Hydrol.* **1996**, *178*, 351–367.
26. Rose, C.W.; Hogarth, W.L.; Sander, G.; Lisle, I.; Hairsine, P.B.; Parlange, J.-Y. Modeling processes of soil erosion by water. *Trends Hydrol.* **1994**, *1*, 443–452.
27. Rose, C.W.; Hogarth, W.L. Process-based approaches to modeling soil erosion. In *Modeling Soil Erosion by Water*; Boardman, J., Favis-Mortloch, D., Eds.; Springer: Berlin, 1998; 259–270.
28. Parlange, J.-Y.; Hogarth, W.L.; Rose, C.W.; Sander, G.C.; Hairsine, P.; Lisle, I. Addendum to unsteady soil erosion model. *J. Hydrol.* **1999**, *217*, 149–156.
29. Willgoose, G.R.; Bras, R.L.; Rodriguez-Iturbe, I. A physically based coupled network growth and hillslope evolution model. 1. Theory. *Water Resour. Res.* **1991**, *27*, 1671–1684.
30. Willgoose, G.R.; Bras, R.L.; Rodriguez-Iturbe, I. A physically based coupled network growth and hillslope evolution model. 2. Applications. *Water Resour. Res.* **1991**, *27*, 1685–1696.
31. Willgoose, G.R.; Bras, R.L.; Rodriguez-Iturbe, I. A physical explanation of an observed link area-slope relationship. *Water Resour. Res.* **1991**, *27*, 1697–1702.

32. Willgoose, G.R.; Bras, R.L.; Rodriguez-Iturbe, I. Results from a new model of river basin evolution. *Earth Surf. Processes* **1991**, *16*, 237–254.
33. Nearing, M.A.; Foster, G.R.; Lane, L.J.; Finkner, S.C. A process-based erosion model for USDA water erosion prediction project technology. *Trans. ASAE* **1989**, *32*, 1587–1593.
34. Lane, L.J.; Nearing, M.A., Eds. *USDA Water Erosion Prediction Project: Hillslope Profile Model Documentation*; NSERL Report No. 2; National Soil Erosion Laboratory, USDA-ARS: West Lafayette, 1989.
35. Yu, B.; Ciesiolka, C.A.A.; Rose, C.W.; Coughlan, K.J. A validation test of WEPP to predict runoff and soil loss from a pineapple farm on a sandy soil in subtropical Queensland, Australia. *Aust. J. Soil Res.* **2000**, *38*, 537–554.
36. Flanagan, D.C.; Nearing, M.A., Eds. *USDA Water Erosion Prediction Project: Hillslope Profile and Watershed Model Documentation*; NSERL Report No. 10; National Soil Erosion Research Laboratory, USDA-ARS: West Lafayette, 1995.
37. Misra, R.K.; Rose, C.W. Application and sensitivity analysis of process-based erosion model GUEST. *Eur. J. Soil Sci.* **1996**, *47*, 593–604.
38. Ciesiolka, C.A.A.; Coughlan, K.J.; Rose, C.W.; Escalante, M.C.; Hashim, G.M.; Paningbatan, E.P.; Sombatpanit, S. Methodology for a multi-country study of soil erosion management. *Soil Technol.* **1995**, *8*, 179–192.
39. Yu, B.; Rose, C.W.; Ciesiolka, C.A.A.; Coughlan, K.J.; Fentie, B. Toward a framework for runoff and soil loss prediction using GUEST technology. *Aust. J. Soil Res.* **1997**, *35*, 1191–1212.
40. Yu, B.; Rose, C.W. Application of a physically based soil erosion model, GUEST, in the absence of data on runoff rate. I. Theory and methodology. *Aust. J. Soil Res.* **1999**, *37*, 1–12.
41. Morgan, R.P.C.; Quinton, J.N.; Rickson, R.J. *EUROSEM: Documentation Manual*; Silsoe College: Silsoe, 1992.
42. Morgan, R.P.C. The European soil erosion model: An update on its structure and research base. In *Conserving Soil Resources, European Perspectives*; Rickson, R.J., Ed.; CAB International: Wallingford, 1994; 428 pp.
43. Govers, G. Empirical relationships on the transporting capacity of overland flow. *Int. Assoc. Hydrol. Sci. Publ.* **1990**, *189*, 45–63.
44. Woolhiser, D.A.; Smith, R.E.; Goodrich, D.C. *KINEROS: A Kinematic Runoff and Erosion Model: Documentation and User Manual*; USDA Agricultural Research Service ARS-677, 1990.

Water Erosion: Process-Based Modeling

Mark A. Nearing

Watershed Research Center, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Tucson, Arizona, U.S.A.

Abstract

Soil erosion models can play a critical role in addressing problems associated with soil protection and conservation. They are used for assessment and inventory purposes when financial and time costs of obtaining useful soil erosion measurements are prohibitive. They provide assessments of both on-site loss of topsoil and off-site delivery of sediments from fields and catchments. Thus, they have value for addressing both declining soil productivity on agricultural lands and problems associated with non-point source pollution. Models are used for purposes of conservation planning, primarily for selecting the most appropriate conservation measures from a range of options for a particular field or geographical region. Models are also used with increasing frequency by governmental agencies at all levels to set regulations for erosion control practices for agriculture, construction, and forestry operations. Finally, models are used to increase and synthesize our knowledge of soil erosion and conservation science.

INTRODUCTION

Historical models of soil erosion by water have been primarily either empirically based or process based (sometimes referred to as physically based). The first models of soil erosion were empirically based. The prime example of the empirically based model is the Universal Soil Loss Equation (USLE).^[1,2] More models have been based on equations which describe the physical, biological, and/or chemical processes which cause or affect soil erosion. It is important to understand that both the process-based and the empirically based models possess a major empirical component, in the sense that the constitutive equations use parameters based on experimental data.

PROCESS-BASED MODELS

Process-based erosion models address soil erosion on a relatively fundamental level using mass balance differential equations for describing sediment continuity on a land surface. The fundamental equation for mass balance of sediment in a single direction on a hillslope profile is given as follows:

$$\partial(cq)/\partial x + \partial(ch)/\partial t + S = 0 \quad (1)$$

where c (kg/m^3) is sediment concentration, q (m^2/s) is unit discharge of runoff, h (m) is depth of flow, x (m) is distance in the direction of flow, t (s) is time, and S [$\text{kg}/(\text{m}^2 \text{s})$] is the source/sink term for sediment generation. Eq. 1 is an exact 1-D equation. It is the starting point for development of all process-based models. The differences in various erosion models are primarily: 1) whether the partial differential

with respect to time is included and 2) differing representations of the source/sink term, S . If the partial differential term with respect to time is dropped, then the equation is solved for the steady state, whereas the representation of the full partial equation represents a fully dynamic model. The source/sink term for sediment, S , is generally the greatest source of differences in soil erosion models. It is this term that may contain elements for soil detachment, transport capacity terms, and sediment deposition functions. It is through the source/sink term of the equation that empirical relationships and parameters are introduced.

One of the earliest process-based erosion models was presented by Meyer and Wischmeier.^[3] This model considered soil detachment by rain and by runoff and then compared the sediment load generated from this detachment to the sediment transport capacity of the flow. If sediment load exceeded sediment transport capacity, then deposition was calculated. The ideas from this early work were expanded and revised by Foster and Meyer^[4] and ultimately were worked into the Chemicals, Runoff, and Erosion from Agricultural Systems (CREAMS) model,^[5] which is a hybrid model in the sense that it uses process-based equations with parameters from the USLE. Experience from developing CREAMS then led to the development of the fully process-based Water Erosion Prediction Project (WEPP) model.^[6]

CONTINUOUS SIMULATION MODELS

Process-based models for soil erosion by water may be either continuous simulation or event models. Continuous simulation models predict soil erosion for series of

individual storms and have auxiliary components for updating system parameters between storms. Event models simply predict erosion for individual storm events but require the user to provide system information for each event. As such, the continuous simulation model is useful for analyzing the effects of crop management systems on erosion, whereas the event model is relatively limited in that regard.

The WEPP model is an example of a continuous simulation model. WEPP includes the following nine major components: climate, infiltration, water balance, winter process and snowmelt erosion, plant growth and residue decomposition, irrigation, surface runoff, rainfall erosion, and channel routing for watersheds.^[6] System parameters which are updated on a daily basis within the continuous simulation include soil moisture, soil density, soil resistance to erosion, plant canopy, surface residue, soil surface roughness, soil surface sealing and crusting, buried residue amounts, frozen soil layers, snow cover, and roots. All of these system parameters define the antecedent conditions for each erosion event.

The WEPP climate model has been tested for erosion and well parameterized for the United States.^[7] The infiltration component of the hillslope model is based on the Green and Ampt equation as modified by Mein and Larson,^[8] with the ponding time calculation for an unsteady rainfall.^[9] The water balance and percolation component of the profile model is based on the water balance component of Simulator for Water Resources in Rural Basins,^[10,11] with some modifications for improving the estimation of percolation and soil evaporation parameters. The plant growth component of the model simulates plant growth and residue decomposition for cropland and rangeland conditions. The residue and root decomposition model simulates the decomposition of surface residue (both standing and flat), buried residue, and roots for the annual crops specified in the WEPP User Requirements^[12] plus perennial crops of alfalfa and grasses. Surface runoff is calculated using a kinematic wave equation. Flow is partitioned into broad sheet flow for inter-rill erosion calculations and concentrated flow for rill erosion calculations. The erosion component of the model uses a steady-state sediment continuity equation that calculates the net values of detachment or deposition rates along the hillslope profile.^[13] The erosion process is divided into rill and inter-rill components where the inter-rill areas act as sediment feeds to the rills, or small channel flows. The model is applicable to hillslopes and small watersheds.

Because the model is based on all of the above-mentioned processes, and more, it is possible with a continuous simulation, process-based model to have an enormous array of possible system interactions represented in the simulations. Just to name a very few examples, slope length and steepness effects are functions of soil consolidation, surface sealing, ground residue cover, canopy cover, soil water content, crop type, and many other factors. Ground residue cover is a function of biomass production

rates, tillage implement types, residue type, soil moisture, temperature and solar radiation, previous rainfall, and many other factors. Rill erosion rates are a function of soil surface roughness, ground cover, the consolidation of soil, soil physical and chemical properties, organic matter, roots, inter-rill erosion rates, slope, and runoff rates, among other factors. The lists continue ad infinitum. These are interactions which are simply not possible to represent with an empirical model. The continuous simulation, process-based model is quite complex in this sense.

The disadvantage of the process-based model is also the complexity of the model. Data requirements are huge, and with every new datum element comes the opportunity to introduce uncertainty, as a first-order error analysis would clearly indicate. Model structure interactions are also enormous in number, and with every structural interaction comes the opportunity for error, as well. In a sense, the goal in using the process-based model is to capture the advantages of the complexity of model interactions, while gaining the accuracy and dependability associated with the simpler empirically based model. This can be done and was done with the WEPP model, using a combination of detailed sensitivity analyses and calibration of the model to the large database of natural runoff plot information used to develop the USLE. Without the tie between model and database, and without knowledge of the sensitive input variables so as to know where to focus efforts, turning a complex model such as WEPP into a useful conservation tool would not be possible. Thus, in a sense, even though WEPP routines are process-based descriptors of various components of the erosional system, ultimately the model must be empirically based on the same type of data as was used to develop the USLE, along with additional experimental data collected specifically for WEPP.

Examples of process-based event models of soil erosion by water include the European Soil Erosion Model (EUROSEM),^[14] the Hairsine and Rose model,^[15,16] and Griffith University Erosion System Template (GUEST).^[17] EUROSEM is a fully dynamic model which simulates the erosion, transport, and deposition of sediment over the land surface by inter-rill and rill processes. The model requires break-point rainfall data, ideally of one-minute resolution, as input along with a detailed description of the soils, slopes, and land cover of the watershed. The model computes the interception of rainfall by the plant cover, the generation of runoff as infiltration excess, soil detachment by raindrop impact and runoff, the transport capacity of runoff, and the deposition of sediment. EUROSEM uses the runoff generator and the water and sediment routing routines of the Kinematic Runoff and Erosion Model.^[18] The model of Hairsine and Rose^[15,16] develops the ideas presented earlier by Rose and colleagues,^[19,20] where erosion is calculated as a net balance of instantaneous processes of

sediment entrainment, deposition, and re-entrainment. The GUEST model^[17] is a practical conservation tool developed from these same concepts.

REFERENCES

1. Wischmeier, W.H.; Smith, D.D. A Universal soil-loss equation to guide conservation farm planning. *Trans. Int. Congr. Soil Sci.* **1960**, *1*, 418–425.
2. Wischmeier, W.H.; Smith, D.D. *Predicting Rainfall Erosion Losses. A Guide to Conservation Planning*; Agriculture Handbook No. 537; USDA-SEA, U.S. Govt. Printing Office: Washington, 1978.
3. Meyer, L.D.; Wischmeier, W.H. Mathematical simulation of the process of soil erosion by water. *Trans. Am. Soc. Agric. Eng.* **1969**, *12* (6), 754–758.
4. Foster, G.R.; Meyer, L.D. A closed-form soil erosion equation for upland Areas. In *Sedimentation (Einstein)*; Shen, H.W., Ed.; Colorado State University: Ft. Collins, 1972; 12:1–12:19.
5. Knisel, W.G., Ed. *CREAMS: A Field Scale Model for Chemicals, Runoff, and Erosion from Agricultural Management Systems*; USDA-ARS: Washington, 1980; 643 pp.
6. Flanagan, D.C.; Nearing, M.A. *USDA-Water Erosion Prediction Project: Hillslope Profile and Watershed Model Documentation*; NSERL Report No. 10; USDA-ARS National Soil Erosion Research Laboratory: West Lafayette, 1995.
7. Baffaut, C.; Nearing, M.A.; Nicks, A.D. Impact of climate parameters on soil erosion using CLIGEN and WEPP. *Trans. ASAE* **1996**, *39*, 447–457.
8. Mein, R.G.; Larson, C.L. Modeling infiltration during a steady rain. *Water Resour. Res.* **1973**, *9* (2), 384–394.
9. Chu, S.T. Infiltration during an unsteady rain. *Water Resour. Res.* **1978**, *14* (3), 461–466.
10. Williams, J.R.; Nicks, A.D. SWRRB, a simulator for water resources in rural basins: An overview. In *Proceedings of the Natural Resources Modeling Symposium*, Pingree Park, Oct 16–21, 1983; DeCoursey, D.G., Ed.; USDA-ARS: Washington, 1985; 17–22.
11. Arnold, J.G.; Williams, J.R.; Nicks, A.D.; Sammons, N.B. *SWRRB: A Basin Scale Simulation Model for Soil and Water Resource Management*; Texas A&M University Press: College Station, 1990; 142 pp.
12. Flanagan, D.C.; Livingston, S.J. *USDA-Water Erosion Prediction Project: WEPP User Summary*; NSERL Report No. 11; USDA-ARS National Soil Erosion Research Laboratory: West Lafayette, 1995; 47097–1196.
13. Nearing, M.A.; Foster, G.R.; Lane, L.J.; Finkner, S.C. A process-based soil erosion model for USDA-water erosion prediction technology. *Trans. ASAE* **1989**, *32* (5), 1587–1593.
14. Morgan, R.P.C.; Quinton, J.N.; Smith, R.E.; Govers, G.; Poesen, J.W.A.; Auerswald, K.; Chisci, G.; Torri, D.; Styczen, M.E. The European soil erosion model (EUROSEM): A dynamic approach for predicting sediment transport from fields and small catchments. *Earth Surf. Processes.* **1998**, *23*, 527–544.
15. Hairsine, P.B.; Rose, C.W. Modeling water erosion due to overland flow using physical principles: 1. Sheet flow. *Water Resour. Res.* **1992**, *28* (1), 237–243.
16. Hairsine, P.B.; Rose, C.W. Modeling water erosion due to overland flow using physical principles: 2. Rill flow. *Water Resour. Res.* **1992**, *28* (1), 245–250.
17. Misra, R.K.; Rose, C.W. Application and sensitivity analysis of process-based erosion model GUEST. *Eur. J. Soil Sci.* **1996**, *47*, 593–604.
18. Woolhiser, D.A.; Smith, R.E.; Goodrich, D.C. *KINEROS: A Kinematic Runoff and Erosion Model: Documentation and User Manual*; USDA Agricultural Research Service ARS-77; 1990.
19. Rose, C.W.; Williams, J.R.; Sander, G.C.; Barry, D.A. A mathematical model of soil erosion and deposition processes: I. Theory for a plane land element. *Soil Sci. Soc. Am. J.* **1983**, *47*, 991–995.
20. Rose, C.W. Developments in soil erosion and deposition models. In *Advances in Soil Science*, Stewart, B.A., Ed.; Springer-Verlag: New York, 1985; Vol. 2, 1–63.

Water Infiltration

Manoj K. Shukla

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Infiltration is the vertical downward entry of water into the soil. It is an important component of the hydrological cycle, which recharges the soil and the groundwater resources underneath. The rate of infiltration of water into soil controls the amount of water storage, runoff, and erosion of the soil. Knowledge of water infiltration into soil is essential for determining strategies of soil and water conservation and minimizing the risk of nonpoint source pollution.

INTRODUCTION

Infiltration—the entry of water into the soil—is one of the important components of the hydrological cycle (Fig. 1). It recharges the soil and the groundwater resources underneath. The rate of infiltration of water into soil matrix governs the amount of water storage in soil. It also controls the amount of runoff and erosion. Therefore, knowledge of water infiltration into soil is essential for determining strategies of soil and water conservation and minimizing the risk of nonpoint source pollution.

INFILTRATION

The rate of water infiltration into soil determines the following: 1) the time at which superficial water appears on the soil surface and 2) the amount of runoff that will form over the soil surface during rainfall or irrigation. The infiltration rate of a soil depends on soil texture, structure, moisture status prior to infiltration, continuity and stability of pores, and soil suction. Soil management including tillage influences these factors. Initially, water infiltrates into the soil at a rapid rate, but as time elapses, the infiltration rate attains a steady or asymptotic state, which approximates the saturated hydraulic conductivity of soil, K_s (Fig. 1). The high initial infiltration rate is observed when soil is dry. Under this situation, large suction-gradient exists (suction at the soil surface is zero or atmospheric pressure, and inside soil, it can be 2000–15,000 cm of H_2O depending on dryness of soil), which forces the water rather quickly into the soil.

INFILTRATION PROCESS

When rain falls or water is applied through sprinklers, the water supply rate may be either less than or greater than K_s of the surface soil. If the rate is less than K_s , all the water

falling on soil surface enters the soil. In this case, the infiltration rate is equal to the water supply rate; the rate of supply of water determines the infiltration rate, and the process of infiltration is known as “flux controlled.” On the other hand, if water is supplied at a rate higher than the maximum infiltration rate of soil, the soil–water transmission properties determine the amount and rate of actual infiltration. Infiltration rate in this situation is called “profile controlled.” When water infiltrates into a dry soil, the progress of water movement is observed by the darkened color of soil, as it gets wetter. There exists a sharp downward moving boundary between the wet region and the underlying dry region, which is known as the “wetting front.”

If the water supply rate is greater than K_s and the soil is dry, then for a while all the water enters the soil. In this case, the rate at which water enters the soil is greater than K_s . This occurs because water flows in response not only to gravity but also to soil suction. Sooner or later, the supply rate begins to exceed the capability of the soil to absorb the water. At this point, water begins to build up on the soil surface and runoff begins. Runoff can also occur if the soil becomes saturated above an impermeable layer, or if the soil has a layer in which K_s is less than that of the layers above it.

The time between the start of the rainfall and the initiation of runoff is known as the “time to ponding.” The infiltration rate continues to decrease asymptotically and approaches K_s . The steady-state infiltration rate (i_c) is also termed the “steady-state infiltrability.” It is approximately the same as the field saturated K_s of the surface soil. Basic infiltration rates for some soils are given in Table 1.

FACTORS AFFECTING THE INFILTRATION PROCESS

1. Soil properties: The i_c is approximately equal to K_s . Therefore, soils with higher K_s tend to have more

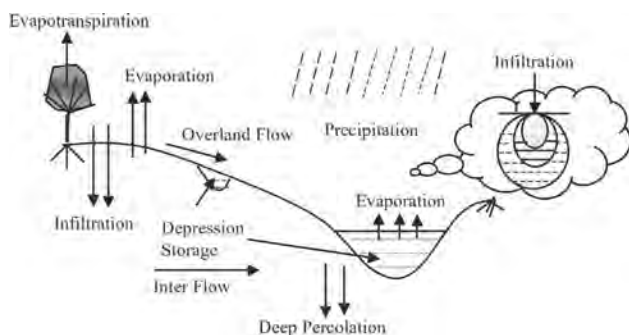


Fig. 1 A schematic representation of processes during a rainfall or irrigation event along with the components of hydrological cycle.

infiltration and less runoff. In addition, the pore-size distribution influences the rate of change of infiltrability. Generally speaking, the wider the range of pore sizes, the more gradual the change in the infiltration rate. The pore-size distribution is a mirror image of the particle-size distribution.

2. Initial moisture content: If the initial moisture content of soil is high, the initial infiltration rate of soil is low. For saturated soils, the infiltration rate approaches K_s almost instantaneously.
3. Rainfall: For rainfall or water supply rates less than K_s in a deep homogeneous soil, infiltration may continue indefinitely. For a given rainfall rate, the longer is the time to ponding, more gradual is the change in infiltration rate. Extremely high rainfall rates may cause slaking of aggregates at the soil surface leading to surface sealing or the formation of soil crusts.
4. Surface sealing and crusting: Change in K_s of the surface soil by slaking has a strong influence on the infiltration rate. The formation of a 5-mm-thick seal can lead up to a 75% decrease in infiltration rate of soil.
5. Layered soils: When the wetting front in the soil reaches a layer with either a coarser or a finer texture, there is a decrease in the infiltration rate for some time. If the layer has a coarser texture, the infiltration rate will recover when the large pores in that layer become saturated. For soil in which the layer is fine textured, the infiltration rate will stay low.

Table 1 Basic infiltration rates for various soil types.

Soil type	Steady-state infiltration rate (mm h ⁻¹)
Sand	<30
Sandy loam	20–30
Loam	10–20
Clay loam	5–10
Clay	1–5

Source: Adapted from Hillel.^[1]

6. Entrapped air: If air is trapped in the soil, K_s is reduced. Water infiltration into the soil is also restricted.

INFILTRATION MODELS

A range of conceptual and empirical models has been developed to predict water infiltration rate of soil as a function of time or the total quantity of water infiltrated into the soil. Common symbols used in these models are as follows: i for the infiltration rate, I for cumulative infiltration, t for time, and i_c for equilibrium rate of infiltration. When t is large, i_c approaches K_s .

Green–Ampt Infiltration Model

Green and Ampt^[2] made several simplifying physical assumptions to develop a mathematical relationship for predicting infiltration rate. The soil–water profile of infiltration was considered to be a step-like profile, and infiltration into the soil was assumed to be piston flow going progressively deeper with time. The soil profile was considered homogeneous and isotropic; therefore, K_s is unique:

$$i = K_s + bI^{-1} \Leftrightarrow i = i_c + bI^{-1} \quad (1)$$

where b is a constant. The Green and Ampt^[2] model uses an approximate description of actual flow regimes.

Kostiakov Equation

The Kostiakov^[3] equation for cumulative infiltration (I) and infiltration rate (i) can be expressed as follows:

$$I = Bt^{-n} \quad (2)$$

$$i = Bt^{-n-1} \quad (3)$$

where B and n are constants. These parameters do not have a physical meaning and can be obtained by fitting the equation to the experimental data. It can be inferred from Eq. 3 that at $t=0$, i approaches ∞ . However, as t increases, i tends to become zero. Therefore, the equation explains the horizontal infiltration, and the Kostiakov^[3] equation for vertical infiltration is inadequate.

Horton Equation

Horton^[4] equations for cumulative infiltration (I) and infiltration rate (i) are given as follows:

$$I = i_c t + \frac{i_0 - i_c}{\alpha} [1 - e^{-\alpha t}] \quad (4)$$

$$i = i_c + (i_0 - i_c)e^{-\alpha t} \quad (5)$$

where i_0 is the initial infiltration rate at $t=0$. The constant α determines the rate at which i_c approaches i_0 . Unlike other

Table 2 Comparison of various infiltration equations.

Parameter	Green–Ampt ^[2]	Kostiakov ^[3]	Horton ^[4]	Philip ^[5]	Holtan ^[6]
Theory	Physically based	Empirical	Empirical	Physically based	Empirical
Surface ponding	Required	Not required	Not required	Required	Not required
Initial infiltration	Infinite	Infinite	Finite	Infinite	Finite
At large time	Steady infiltration = K_s	Zero	Steady infiltration = K_s	Steady infiltration = K_s	Steady infiltration = K_s

Note: K_s , saturated hydraulic conductivity of soil.

equations, the Horton equation has a finite infiltration rate, i_0 at $t=0$. The equation is somewhat cumbersome as it has three constants that need to be evaluated experimentally. The decrease in infiltration rate with time is attributed to a number of factors, such as the closure of soil pores by a swelling soil, erosional deposit, and compaction due to raindrop impact.^[4]

Philip Equation

Philip^[5] provided a rigorous mathematical solution of flow equation applied to vertical infiltration into an infinite deep uniform soil in terms of a power series. In practice, first two terms of the power series are enough to describe steady infiltration through a soil profile as follows:

$$I(t) = St^{0.5} + At \quad (6)$$

$$i(t) = \frac{1}{2}St^{-0.5} + A \quad (7)$$

where S is the sorptivity. According to Philip,^[5] as t approaches infinity, the infiltration rate decreases to a finally asymptotic value, and the coefficient A can be replaced by K_s of upper layer.

Holtan Equation

The Holtan equation^[6] for infiltration rate in soil is again a two-form mathematical equation as follows:

$$i = i_c + af^n \quad (8)$$

where i_c is the equivalent infiltration rate and f is the available porosity as depleted by infiltration volumes, which can be expressed as follows:

$$f = M - I \quad (9)$$

where M is the water storage capacity of the soil above the first impeding stratum. It can also be expressed as total porosity—antecedent water content of soil, in depth units. I is the cumulative infiltration at that time.

$$i = i_c \quad \text{for } I > M \quad (10)$$

As long as $0 \leq I \leq M$, Eq. 9 is consistent; however, for $I > M$, the $(M - I)^n$ becomes positive or negative depending on the value of exponent n . Also, in the absence of an impeding layer, Holtan^[6] did not discuss the meaning of M . A comparison of all the infiltration equations discussed above is given in Table 2.

CONCLUSION

Infiltration is the vertical downward entry of water into the soil. It is an important component of the hydrological cycle, which recharges the soil and the groundwater resources underneath. The rate of infiltration of water into soil controls the amount of water storage, runoff, and erosion of the soil. Knowledge of water infiltration into soil is essential for determining strategies of soil and water conservation and minimizing the risk of nonpoint source pollution. Several mathematical relationships are available in literature, which can predict the rate and accumulative amount of water infiltration into soil.

REFERENCES

1. Hillel, D. *Applications of Soil Physics*; Academic Press: New York, 1980.
2. Green, W.H.; Ampt, G.A. Studies on soil physics: I. Flow of air and water through soils. *J. Agr. Sci.* **1911**, *4*, 1–24.
3. Kostiakov, A.N. On the dynamics of the coefficient of water percolation in soils and on the necessity of studying it from a dynamic point of view for purposes of amelioration. *T. Commun. Int. Soc. Soil Sci.* **1932**, *17*–21.
4. Horton, R.E. An approach towards a physical interpretation of infiltration capacity. *Soil Sci. Soc. Am. Proc.* **1940**, *5*, 399–417.
5. Philip, J.R. Theory of infiltration: 4. Sorptivity and algebraic infiltration equations. *Soil Sci.* **1957**, *84*, 257–264.
6. Holtan, H.N. A concept for infiltration estimates in watershed engineering. *U.S. Dept. Agr. Agr. Res. Serv.* **1961**, 41–51.

Water Management: Computer Modeling

Laj R. Ahuja

U.S. Department of Agriculture (USDA), Fort Collins, Colorado, U.S.A.

Liwang Ma

Great Plains Systems Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.

Abstract

Water is a vital resource for food and fiber production. Irrigated farmlands in the semiarid to arid regions of the world contribute about 36% of the total production. As the human population increases, more irrigation water will be needed to meet the greater food demand. However, the supplies of usable and renewable freshwater are scarce. Agriculture is already facing a stiff competition for freshwater supplies from drinking, industrial, and recreational uses, even in the United States that has vast resources. The only answer is to conserve water through judicious management for all uses. In agriculture, the management of soil water based on scientific principles can help to increase water use efficiency, reduce losses, and prevent secondary salinization of groundwater resources, a common problem in irrigated areas of the world.

WATER FLOW

The theories of water flow in the integrated soil–plant–atmosphere system or continuum have vital field applications for soil water management under both rainfed and irrigated agriculture. During the last years, the increase in our basic understanding of soil water entry (infiltration), redistribution, retention, and uptake by plants has changed the old concepts of field capacity (water-holding capacity) of soils and the water available for plant growth ingrained in the traditional soil water management for plant production. The field capacity is no longer considered a constant intrinsic property of soil, but rather a continually decreasing soil water content, albeit at a slow rate of internal drainage.^[1,2] Factors such as the impeding or coarse-textured layers in the soil profile, pre-infiltration wetness, depth of the groundwater table, and surface evaporation rate influence the field capacity value.^[3–5] Similarly, the concept that soil water is equally available to plants in the range of soil water contents from field capacity to permanent wilting^[6] has been shown to be invalid^[7] and replaced by a dynamic concept of water availability to plants. It is postulated that the rate of water uptake that would sustain normal plant growth at any time depends not only on soil water status but also on the atmospheric conditions and properties of the plants.^[8–10] Atmospheric conditions dictate the evapotranspiration (ET) demand, or the rate at which the plant is required to transpire and absorb water from the soil to maintain its turgidity. Rooting depth, density, proliferation, extension, physiological adjustment of plant to water stress, and soil's unsaturated conductivities determine the soil water content at which the actual ET falls below the demand ET. These conditions also determine the

water content or soil water suction at which the plants will wilt. This new dynamic concept of soil water availability has tremendous implications for soil water management. This concept requires that irrigation applications be more flexible, based on meteorological conditions, plant growth stage, root system, and soil properties.

The knowledge of water flow in the soil–plant–atmospheric continuum needs to be combined with the knowledge of heat flow, nutrient processes, plant growth, and management effects in order to attain the best scientific management of soil water. However, the interactions among these components of the agricultural system are highly complex, dynamic, and difficult to comprehend. Fortunately, the modern technology has allowed us to develop computer models of the entire agricultural systems that incorporate these interactions.^[11–14] Therefore, computer models of this nature are useful as research and management guides for efficient use of water and nutrients under varying weather and climatic conditions. Modeling may be used for all areas of soil water management in a variety of ways—from designing an irrigation and drainage system for uniformity and effectiveness, to comparison of soil and water conservation practices, to simulation of the entire soil–plant–atmosphere continuum on a day-to-day basis for making decisions on water, fertilizer, or pesticide applications. Although there are many gaps in our knowledge at present, the models are developed well enough to use them as guides for optimum use of resources. This entry illustrates the applications of models in soil water management for plant production through examples. (Readers interested in the theories and assumptions underlying these models and the extent of their validation should see the cited references.)

Table 1 Management practices, timing, and application options in RZWQM.

Management practice	Timing options	Application options
Tillage operations	Plant growth cycle Specified date	29 Different implements User-specified intensity
Manure applications	Plant growth cycle Specified date	Surface broadcast Broadcast incorporated Injected fumigation Chemigation 15 Different types of manure available
Fertilizer applications	Plant growth cycle Specified date Preplant with topdress Preplant and topdress with plant demand trigger	Surface broadcast Broadcast incorporated Injected $\text{NH}_3\text{-N}$ Fertigation Nitrification inhibitor additives
Pesticide applications	Plant growth cycle Specified date	Surface broadcast Broadcast incorporated Injected fumigation Chemigation Slow release formulations Foliar or soil surface only
Crop planting and harvesting	Multiple year same crop Multispecies rotations	Five Harvesting operations External crop models can be used
Irrigation applications	Fixed interval Specified dates Root zone depletion	Fixed amount Varying amounts Refill root zone

EXAMPLES OF APPLICATION

The U.S. Department of Agriculture-Agricultural Research Service Root Zone Water Quality Model (RZWQM) is an

agricultural system model with a Windows-based graphical user interface. It has been developed and used for a wide range of agricultural management problems, including both water quantity and quality.^[14,15] Table 1 shows the various

Table 2 Average yearly summary for manure and water management based on RZWQM.

Management option	Manure	Water applied (cm/event)	Silage yield ($\text{Mg ha}^{-1} \text{ yr}^{-1}$)	N uptake ($\text{kg N ha}^{-1} \text{ yr}^{-1}$)	N seepage ($\text{kg N ha}^{-1} \text{ yr}^{-1}$)	Denitrification ($\text{kg N ha}^{-1} \text{ yr}^{-1}$)	Volatilization ($\text{kg N ha}^{-1} \text{ yr}^{-1}$)
1	Full rate every year	20	25.2	254	196	130	11.1
2	Half rate every year	20	21.9	182	106	48	3.5
3	Full rate every 2 years	20	22.8	192	91	57	4.7
4	Full rate every 3 years	20	19.0	153	83	34	3.2
5	Full rate, split application	20	25.1	250	188	140	7.1
6	Full rate every year	10	25.8	307	116	141	11.1
7	Half rate every year	10	22.7	228	49	50	3.5
8	Full rate every 2 years	10	22.9	232	39	59	4.7
9	Full rate every 3 years	10	20.1	186	40	35	3.2
10	Full rate, split application	10	25.6	302	109	150	7.1

Note: Full manure application rate was 44.8 Mg ha^{-1} , and half rate was 22.4 Mg ha^{-1} . Manure was applied on October 15 for one application and October 15/April 1 for split application (22.4 Mg ha^{-1} each).

Source: From Ma, Shaffer, et al.^[16]

management practices simulated in RZWQM that affect soil water management for plants. This model can be used to address questions such as the amounts and timings of irrigation water, fertilizer, and pesticides applications under various tillage and cropping schemes in different soil types (Table 1). Table 2 shows the results of an example application of RZWQM for water and manure management in eastern Colorado from Ma et al.^[16] The field has been used for corn silage production for more than a decade with manure application every fall after harvesting. Irrigation water was applied 4 to 6 times during the months of July and August at a rate of 20 cm per event. Management Option 1 was the simulation result under management practices. As shown in Table 2, reducing water irrigation to 10 cm per event not only increased yield and nitrogen (N) uptake, but also decreased N losses.

Another example application of RZWQM is to answer questions on when and how much to irrigate. A common practice is to apply irrigation at a fixed time interval and a fixed amount of water for each irrigation. This practice disregards the water actually used by the crop in response to the atmospheric ET demand. A more scientific way is to link irrigation timing and amount to soil water contents in the root zone; e.g., the irrigation is triggered when the soil water storage falls below a certain lower limit of the so-called available water, arbitrarily defined as soil water content in equilibrium with 33 kPa (1/3 bar) water suction minus the soil water content at 1500 kPa suction. The irrigation is stopped when the soil water storage is filled to a certain upper limit of available water. What the lower and upper limits of available water for triggering and stopping irrigation should be depends on the soil type, crop, and climatic conditions in the area. A system model will be a good guide to determine optimal limits for an area. For the eastern Colorado site noted previously, on a Vona sandy loam soil, RZWQM-simulated average corn silage yields, N uptake, and N leaching at different irrigation amounts are shown in Fig. 1. The lower limit of available water to trigger irrigation was varied from 10% to 90%, and the upper limit to stop irrigation was varied from 50% to 90%. The irrigation period lasted up to 100 days after planting. These results indicate that the irrigations triggered when the soil water content falls to about 20% of the available water and stopped when the soil water storage is filled to about 50% of available water gave the best results—maximum N uptake and silage yield and minimum nitrate-N leached.

A third example is the use of the GOSSYM model for cotton crop management from Lemmon.^[12] GOSSYM was developed by a team of scientists in Mississippi and South Carolina, in the United States.^[17] It simulates the growth and development of cotton on an organ-by-organ basis (roots, stems, leaves, blooms, squares, and bolls), as well as the transport and uptake of water and nutrients through the soil profile. The model is driven by daily weather variables, such as solar radiation, rainfall, and

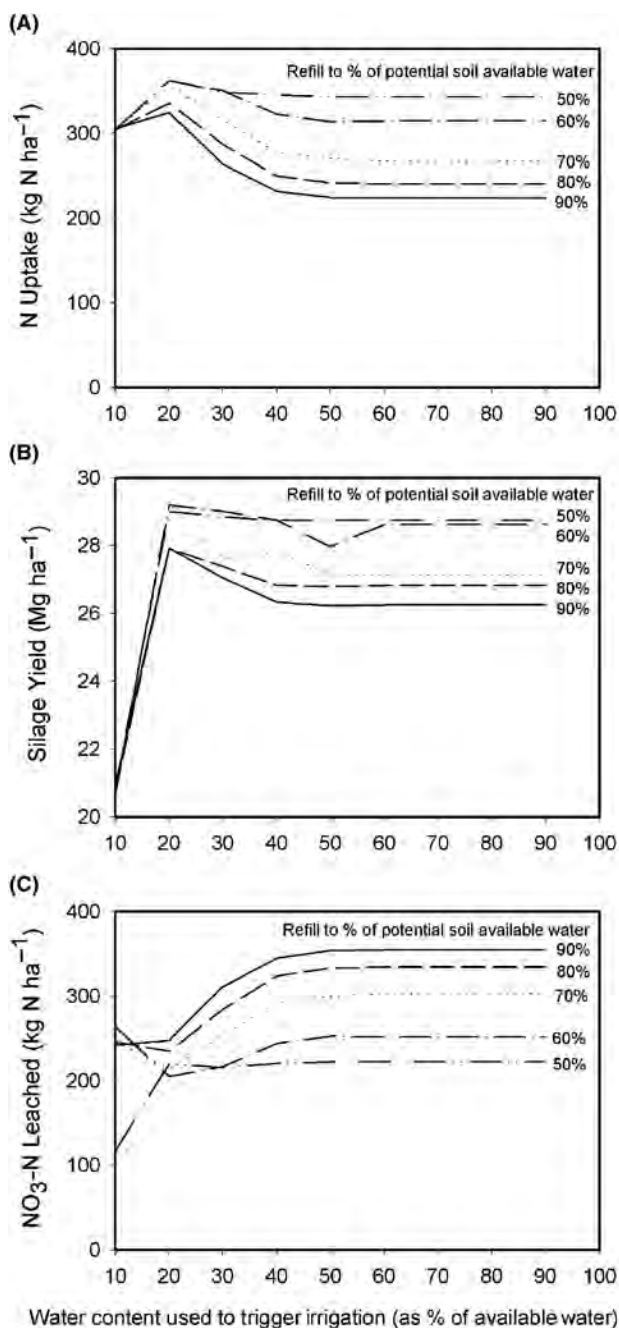


Fig. 1 The responses of yearly average N uptake (A), silage yield (B), and nitrate-N leached (C) to irrigation according to degree of root zone water depletion, as simulated by RZWQM. **Source:** From Ma, Shaffer, et al.^[16]

maximum–minimum air temperatures, and requires input data on soil properties and soil fertility. GOSSYM is used to determine irrigation schedules, N requirements, and crop maturity dates. An example of how GOSSYM simulation results are used is shown in Fig. 2.^[12] The N and water stress values are ratios of the amount of N or water used to the amount needed for full growth at different times. The first row of graphs was produced after the third N application. These results show N and water stresses

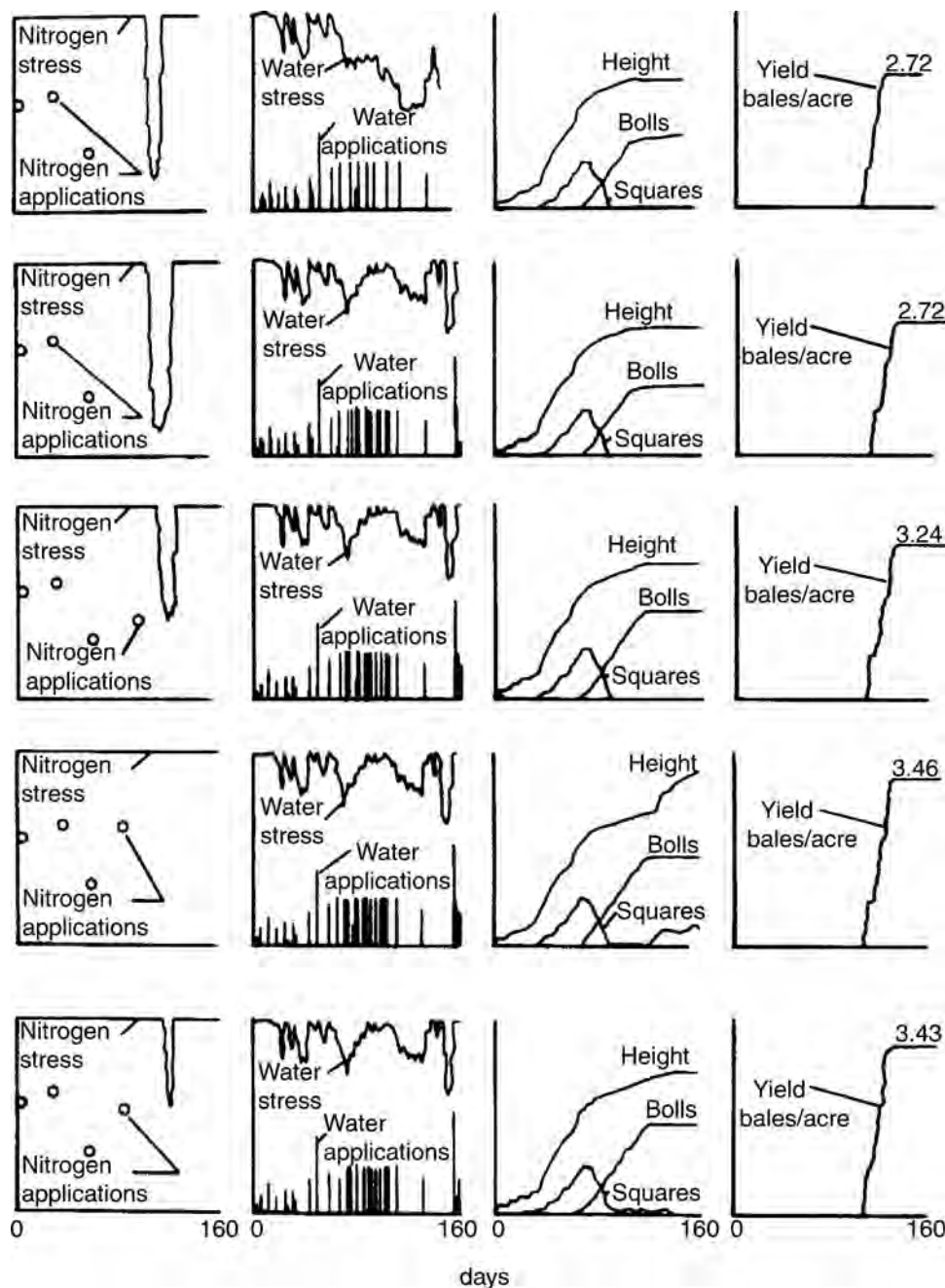


Fig. 2 Graphs produced by Commax from the results of GOSYM simulations, showing the process whereby Commax reduces the water stress and then the N stress.

Source: From Lemmon.^[12]

and their projected effects on yield organs and cotton. The second row of graphs is the simulation results of increasing irrigation schedule to reduce water stress. However, increased irrigation results in increased N stress. The third row shows the simulation result of adding a small N application (fourth N application), which reduces N stress and increases yield. In the fourth row, N application was increased for the fourth N application, which eliminated N stress. However, this increased application also resulted in some new vegetative growth and squares near the harvest time. Thus, the fifth row presents optimal results with a moderate N application rate. These simulation results have convinced Mississippi cotton growers to apply

additional N that was not planned originally and resulted in increased yield.

THE FUTURE

The results presented show the value of the models to guide soil water management for plant production in complex agricultural systems. At the dawn of the 21st century, computers are playing a central role in managing industries, businesses, and many other aspects of life. Their use in managing water and other natural resources and environmental quality is increasing. It is time that computer models

of agricultural systems be integrated with all field research, so that the models continually capture new knowledge as it becomes available. The models should then be used to guide agricultural management under varying climatic and weather conditions.

REFERENCES

1. Richards, L.A.; Gardner, W.R.; Ogata, G. Physical processes determining water loss from soils. *Soil Sci. Soc. Am. Proc.* **1956**, *20*, 310–314.
2. Ogata, G.; Richards, L.A. Water content change following irrigation of bare field soil that is protected from evaporation. *Soil Sci. Soc. Am. Proc.* **1957**, *21*, 355–356.
3. Miller, D.E. *Flow and Retention of Water in Layered Soils*; Conserv. Res. Rep. 13; USDA-ARS: Washington, 1969.
4. Biswas, T.D.; Nielsen, D.R.; Biggar, J.W. Redistribution of soil water after infiltration. *Water Resour. Res.* **1966**, *2*, 513–514.
5. Ratliff, L.F.; Ritchie, J.T.; Cassel, D.K. Field-measured limits of soil water availability as related to laboratory measured properties. *Soil Sci. Soc. Am. J.* **1983**, *47*, 770–775.
6. Veihmeyer, F.J.; Hendrickson, A.H. Soil-moisture conditions in relation to plant growth. *Plant Physiol.* **1927**, *2*, 71–78.
7. Richards, L.A.; Wadleigh, C.H. Soil water in plant growth. In *Soil Physical Conditions and Plant Growth*; Shaw, B.T., Ed.; Agronomy Monograph, no. 2; Academic Press, Inc.: New York, 1952; Vol. 2, 13 pp.
8. Penman, H.L. The dependence of transpiration on weather and soil conditions. *J. Soil Sci.* **1949**, *1*, 74–89.
9. Philip, J.R. The physical principles of soil water movement during the irrigation cycle. 3rd Congr. Int. Commun. Irrig. Drain. Quest. **1957**, *8*, 125–154.
10. Gardner, W.R. Dynamic aspects of water availability to plants. *Soil Sci.* **1960**, *89*, 63–73.
11. Watts, D.G.; Hanks, R.J. A Soil-water-nitrogen model for irrigated corn on sandy soils. *Soil Sci. Soc. Am. J.* **1978**, *42*, 492–499.
12. Lemmon, H. Commax: An expert system for cotton crop management. *Science* **1986**, *233*, 29–33.
13. Shaffer, M.J.; Larson, W.E. *NTRM: A Soil-Crop Simulation Model for Nitrogen, Tillage, and Crop Residue Management*, Conserv. Res. Rep. 34–1; USDA-ARS: Washington, 1987.
14. Ahuja, L.; Rojas, K.W.; Hanson, J.D. *Root Zone Water Quality Model: Modeling Management Effects on Water Quality and Crop Production*; Water Resources Publications, LLC: Littleton, 2000; 372 pp.
15. Ma, L.; Ahuja, L.R.; Ascough, J.C., II; Shaffer, M.J.; Rojas, K.W.; Malone, R.W.; Cameira, M.R. Integrating system modeling with field research in agriculture: Applications of the Root Zone Water Quality Model (RZWQM). *Adv. Agron.* **2000**, *71*, 233–292.
16. Ma, L.; Shaffer, M.J.; Boyd, J.K.; Waskom, R.; Ahuja, L.R.; Rojas, K.W.; Xu, C. Manure management in an irrigated silage corn field: Experiment and modeling. *Soil Sci. Soc. Am. J.* **1998**, *62*, 1006–1017.
17. Baker, D.N.; Lambert, J.R.; McKinion, J.M. GOSSYM: A simulator of cotton crop growth and yield. *Tech. Bull.* 1089; S. C. Agr. Exp. Stn. **1983**.

Water Management: Harvesting

Gary W. Frasier

U.S. Department of Agriculture (USDA), Fort Collins, Colorado, U.S.A.

Abstract

Collecting precipitation for reuse is commonly called water harvesting. Water harvesting is an ancient practice. This entry presents the uses for harvested water, types of systems, and cropping considerations for water harvesting.

INTRODUCTION

Without water, land has no value and man cannot live. Yet with few exceptions, in most places on earth, there is sufficient water available on an annual basis for man not only to survive but also to live. This water comes in the form of precipitation, and all that is required is a means of collecting and storing this precipitation from the sky. Accessing this type of water supply—collecting precipitation for reuse—is commonly called *water harvesting*. In more technical terms, water harvesting can be defined as “the process of collecting and storing precipitation (rain or snow) for beneficial use from an area that has been treated or modified to increase precipitation runoff.”^[1]

Water harvesting is an ancient practice dating back at least 9000 years when it was used in the Edom Mountains of Southern Jordan.^[2] It is probable that the first water harvesting system was nothing more than a simple depression that filled with water runoff from an uphill area. Evidence of extensive water harvesting systems used 3000–4000 years ago have been found in various areas of the Middle East. Many of these ancient systems were located in areas where the annual precipitation was less than 200 mm per year.^[3] Often in arid climates, small rain-filled depressions can be found at the bottom of rock outcroppings which provide drinking water for wildlife.^[4] Rain-filled depressions are being used for domestic drinking water supplies (Fig. 1). Water harvesting techniques are used in areas such as Israel, Egypt, Jordan, Mexico, Australia, and throughout the parts of the United States.

GENERAL DESCRIPTION

Water harvesting systems are frequently grouped into two categories, depending on the intended use of the collected water. Collected water can be used for domestic purposes (e.g., drinking by humans and animals) or crop growing. All water harvesting systems, whether used for drinking water supplies or crop growing, have two major components as

follows: a precipitation collection area (i.e., catchment or runoff area) and a water storage facility. Surfaces of the catchment areas range from rock outcroppings or smoothed–compacted soil to chemically treated or membrane-covered soil, and building rooftops. The storage facility can be a pond, cistern, tank or, in the instance of crop growing, the soil profile.^[5]

RUNOFF FARMING

Runoff farming is a form of water harvesting used specifically for growing crops. Essentially, runoff farming maximizes the effect of limited precipitation over a large area by collecting the surface runoff and applying it to a smaller area. In its simplest form, a portion of land is dedicated to precipitation collection, and the collected water is diverted onto a crop-growing area where it is allowed to infiltrate into the soil for use by plants. The crop area is usually smaller than the water collection area. The collected water can be applied directly to the crop area from the catchment area during the precipitation event or stored for later application using an irrigation system. Runoff farming is not necessarily a technique for increasing crop production but rather a method for sustaining crop production or reducing the risk of crop failure caused by drought.

Types of Systems

Runoff farming systems can be grouped into three general types, depending on the physical arrangement of the catchment area and the crop-growing area (Table 1). The first method, called *floodwater farming*, diverts runoff flowing down a channel during a storm event directly onto a crop area (Fig. 2).

The second method is called *micro-catchment farming*. With this method, each plant or group of plants is assigned a small runoff area situated directly upslope of the growing area. The runoff water is held in the growing area by small earthen berms until it has infiltrated into the soil. This



Fig. 1 A water harvesting system used for drinking water supplies on First Mesa of the Hopi Indian Reservation in Northern Arizona. The catchment surface is a natural rock outcropping, and the collected water is stored in an excavated cistern at the lower end.

technique has been used extensively for tree crops such as pistachio, olive, and almond.

The third method of runoff farming involves a combination of both on-site runoff water and irrigation with excess runoff water from a stored source. The land is formed into a series of large ridges and furrows, and crops, such as fruit trees or grapes, are planted in the furrow bottom. Water runs from the side slopes of the ridges into the crop area. Excess runoff water not directly infiltrated into the planted area continues down the center of the furrow into a storage facility. At a later date, the water is pumped back onto the crop area as needed, using an irrigation system.^[3]

Site Characteristics

The soil in the area of a runoff farming system must serve two diametrically opposed purposes. On the runoff collecting area, the soil must have a low rate of water infiltration and be resistant to soil erosion by water. The soil in the cropping area should have a high infiltration rate with a high water-holding capacity. Many Loess soils are ideally suited for runoff farming applications because they have a high water-holding capacity yet form surface crusts that promote precipitation runoff.^[6]

The relative sizes of the catchment area to the cropping (run-on) area depend on the size and intensities of the



Fig. 2 Floodwater farming near Matrouh, Egypt.

precipitation events, the infiltration rate on the catchment area, and the quantity of water required. Large storms (>25 mm) of moderate to high intensity can provide significant quantities of water from relatively small areas. Conversely, low-intensity storms on soils with high infiltration rates will require larger runoff areas to provide the same quantity of runoff. Typically, the runoff area is 5 to 20 times larger than the cropping area.

Catchment area slopes should be steep enough to maximize runoff and minimize surface storage in depressions, yet flat enough to prevent soil erosion.^[7] To minimize potential soil erosion on the runoff collecting area, overland flow distances should be minimized and the flow maintained as shallow sheet flow. Concentrated water flows should only be used in areas where channels are protected from erosion and designed to carry the flow.^[8] Landform areas with undulating, gently sloping land are best suited for runoff farming. The steeper portions of the land provide the water collection function, while the flatter areas are used for growing crops. In some instances, the crop area is shaped into a series of small terraces with essentially no slope in the planted area. This allows for a better distribution of the collected water in the soil profile.

Successful runoff farming systems include both adequate water management and cropping practices suited to the local area. Good water management includes reducing seepage and evaporative losses of the collected water. For maximum long-term effectiveness, water harvesting systems must have scheduled timely maintenance and repair. Many systems have been adequately designed and constructed, yet have failed to supply the anticipated quantities of water within a relatively short time interval because of little or no maintenance. Usually, maintenance or repair can be accomplished in a short period of time without great expense.

CROPPING CONSIDERATIONS

Cropping practices must include plants that are capable of efficiently using available water yet withstanding

Table 1 Types of runoff farming systems.

Type	Technology level	Crops	Organization
Floodwater farming	Moderate to high	Annual grains and trees	Community
Micro-catchments	Low to moderate	Trees and shrubs	Family
Combination	High	Trees and vines	Community

prolonged time intervals of limited or non-existent water. Good cropping practices also must recognize that water requirements for plant establishment frequently differ from water requirements for mature plants. During the plant establishment phase, rooting systems are usually shallow, which necessitates that water be available in the upper layers of the soil profile. Under these conditions, there is also the potential for significant loss of soil water by evaporation from the unprotected (non-shaded) soil surface. Meeting these water requirements usually means applying water at a frequent interval during the plant establishment phase. Once the plant has achieved some aboveground biomass which shades the soil surface, the direct loss of water from the soil surface is lessened. Also, the deeper rooting system of mature plants allows a larger volume of soil to be used to retain the collected water until it is needed.

A feature that is not controlled but is important to the success of runoff farming systems is the precipitation characteristics of the area. The maximum net effectiveness of the water harvesting system is achieved if the precipitation occurs during cooler weather when evapotranspiration rates are the lowest. Precipitation intensities must be greater than the infiltration rate of the catchment area. The timing of the precipitation compared with the water needs of the crops must be considered. Less extensive systems—especially water storage—are required if precipitation occurs during the cropping season. This reduces the length of time necessary to store the collected water. Selecting crops that are suited for or immediately following the normal rainy periods will improve the chances for a successful runoff farming system.

REFERENCES

1. Frasier, G.W. Water harvesting for rangeland water supplies: A historic perspective. In *Proceedings of a Symposium on Environmental, Economic, and Legal Issues Related to Rangeland Water Development*; Tempe, Arizona, Nov 13–15, 1997; The Center for the Study of Law, Science, and Technology, Arizona State University: Tempe, AZ, 1998; 17–24.
2. Bruins, H.M.; Evenari, M.; Nessler, U. Rainwater-harvesting agriculture for food production in arid zones. The challenges of the African Famine. *Appl. Geogr.* **1986**, *6*, 13–33.
3. Frasier, G.W. Water harvesting. In *The Encyclopedia of Water Science*, 2nd Ed.; Trimble, S.W., Ed.; Taylor & Francis: New York, 2008; 1320–1323.
4. Frasier, G.W. Water supply for arid and semiarid regions. *J. Arid Land Stud.* **2000**, *10* (S), 17–20.
5. Frasier, G.W. Water for animals, man and agriculture by water harvesting. In *Rainfall Collection for Agriculture in Arid and Semiarid Regions*; Dutt, G.R., Hutchinson, C.F., Anaya Garduno, M., Eds.; Commonwealth Agricultural Bureaux, Farnham House: Farnham Royal, 1981; 83–86.
6. Fink, D.H.; Ehrler, W.L. The runoff farming agronomic system: Applications and design concepts. In *Proceedings of Arizona Section, American Water Resources Association and the Hydrology Section, Arizona-Nevada Academy of Science* Tucson, AZ, Apr 7, 1984; Vol. 14, 33–40.
7. Flug, M. Production of annual crops on microcatchments. In *Rainfall Collection for Agriculture in Arid and Semiarid Regions*; Dutt, G.R., Hutchinson, C.F., Anaya Garduno, M., Eds.; Commonwealth Agricultural Bureaux, Farnham House: Farnham Royal, 1981; 39–42.
8. Reij, C.; Mulder, P.; Begemann, L. *Water Harvesting for Plant Production*; World Bank Technical Paper No. 91; The World Bank: Washington, 1948; 1–120.

Water Management: Use Efficiency

David C. Nielsen

U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS),
Akron, Colorado, U.S.A.

Abstract

In semiarid regions where lack of water is limiting crop yield, water use efficiency needs to be maximized to get the most from this scarce resource. This can be accomplished through several management techniques to improve the cropping environment that includes reducing tillage to maintain surface crop residues, minimize evaporation, and trap snow; applying irrigations only during critical growth stages; matching fertilizer application rates to the expected available water condition; and employing crop rotations that reduce or eliminate fallow periods.

INTRODUCTION

In semiarid regions of the world, water is generally the most limiting factor to crop production. Therefore, a primary objective of cropping systems in these regions is to improve water use efficiency, i.e., get the most production of marketable yield for each unit of water used by the cropping system. There are many definitions of water use efficiency,^[1,2] but this discussion uses the following: water use efficiency is the ratio of crop grain yield to the evapotranspiration used to produce that yield. The following discussion will deal mostly with enhancements to water use efficiency that are a result of altering and improving the cropping environment in semiarid regions.

An extremely important method for enhancing water use efficiency is to get more water to be used for plant transpiration and less for evaporation. This can take two forms in a dryland cropping system: increasing precipitation storage efficiency and growing a crop in place of a fallow period.

REDUCING TILLAGE/MAINTAINING CROP RESIDUES

Increasing precipitation storage efficiency through soil-surface water evaporation reduction during the non-crop period of a crop rotation can be accomplished by 1) reducing tillage to minimize the number of times moist soil is brought to the surface and 2) maintaining residues to shade the soil surface, reduce convective exchange of water vapor at the soil-atmosphere interface, reduce runoff by preventing surface crusting, and catch snow.

Precipitation storage efficiency increases as tillage is reduced during the summer fallow period (Fig. 1A).^[3,4] The increased soil water storage is a result of both maintaining crop residues on the soil surface and reducing the number of times that moist soil is brought to the surface.

Data from western Kansas showed water use efficiency increasing by 28% for corn, 17% for sunflower, 10% for soybean, and 7% for grain sorghum. This efficiency was found when a no-till production system was used in place of a conventional tillage system,^[5] using three- or four-sweep plow tillage operations to control weeds in the fallow period prior to crop establishment.

The effect of residue amount on precipitation storage efficiency using data from the three Great Plains sites is shown in Fig. 1B.^[7] Increased precipitation storage efficiency during the 14-month fallow period of a winter wheat-fallow system resulted from increasing amount of surface wheat residue. Crop residues reduce soil water evaporation by shading the soil surface and reducing convective exchange of water vapor at the soil-atmosphere interface.^[8,9] Additionally, reducing tillage and maintaining surface residues reduce precipitation runoff and increase infiltration, thereby increasing precipitation storage efficiency.^[10]

The reduction in convective exchange of water vapor is more effective in standing crop residues (compared to flat residues) because wind speed is reduced near the soil surface by the standing residue.^[11–14] Reductions of up to 50% in potential surface soil water evaporation have been demonstrated when wheat stubble height after harvest was increased from 0.1 to 0.5 m, or when stem populations of short (0.1 m) standing wheat stems were increased from 100 to 600 stems m⁻².^[15]

Standing crop residues also increase the soil water available for crop production in the central and northern Great Plains by increasing snow deposition during the overwinter period.^[6,16–19] Reduction in wind speed within the standing crop residue allows snow to drop out of the moving air stream. The greater the silhouette area index (SAI = height × diameter × number of stalks per unit ground area) through which the wind must pass, the greater the snow deposition. Typical populations and heights of standing sunflower stalks after harvest (SAI = 0.03–0.05 m² m⁻²)

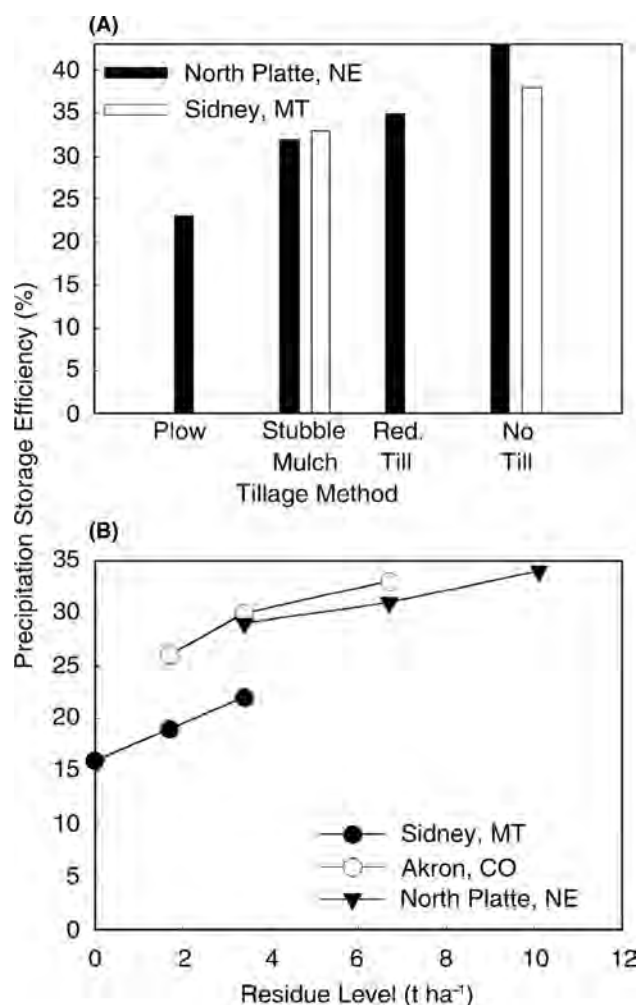


Fig. 1 Precipitation storage efficiency as influenced by (A) tillage method during the 14-month fallow period in a winter wheat–fallow production system and (B) crop residue level on the soil surface.

Source: Data from Smika & Wicks,^[3] Tanaka & Aase,^[4] and Caprio.^[6]

can result in an overwinter soil water increase of about 10–12 cm when winter precipitation is near normal in amount and number of blizzards in northeast Colorado^[20] (Fig. 2). Overwinter soil water storage in North Dakota increased 0.24 cm for each centimeter increase in wheat stubble height because of greater snow trapping.^[21] Because of the drifting and packing of snow that occurs in standing crop residues, precipitation storage efficiencies can be very high. Overwinter precipitation storage efficiencies in excess of 100% in standing sunflower stalks have been reported in some years^[20] as stalks collect snow blown in from adjacent areas without standing residues. Average overwinter precipitation storage efficiency of 55% has been reported for wheat stubble in northeast Colorado.^[22]

The increased precipitation storage efficiency from maintaining residues on the soil surface will result in greater amounts of stored soil water. These higher soil

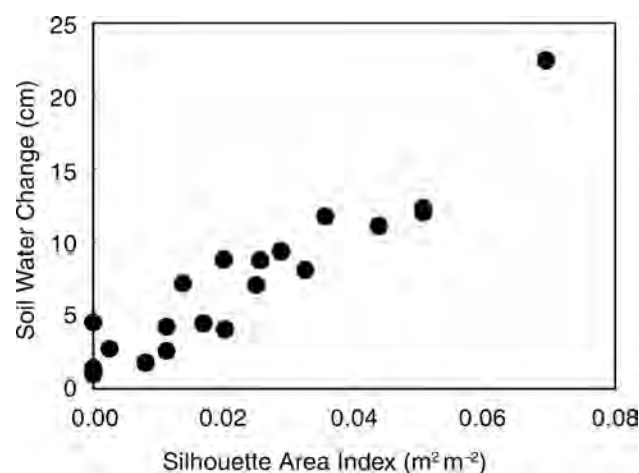


Fig. 2 Influence of sunflower SAI on overwinter soil water change at Akron, Colorado.

Source: From Nielsen.^[20]

water contents at planting have been shown to lead to higher crop yields for winter wheat and proso millet.^[23] The water use efficiency also increases with increased available soil water at planting, but only in years with normal or below normal growing season precipitation. In above-average precipitation years, when growing season precipitation makes up a higher percentage of total water use, there is no increase in water use efficiency with increasing water content at planting.

SHIFTING WATER USE TO CRITICAL GROWTH STAGES

Another method for enhancing water use efficiency is to shift growing season water use so that a greater proportion of the total is occurring during the more critical growth stages of flowering and grain filling and less in the not-so-sensitive vegetative growth stage. When a growth retardant was applied to restrict vegetative development and leaf area in corn, water use during the vegetative stage was reduced, increasing the amount of water available for use during reproductive stages, resulting in a 16% increase in water use efficiency.^[24] Dryland winter wheat water use efficiency in the southern High Plains ranged widely from 0 to 36 kg ha⁻¹ cm⁻¹ due to the wide range in amount and seasonal distribution of precipitation.^[25] Winter wheat water use efficiency at Akron, Colorado, increased linearly as the amount of precipitation falling between 15 May and 25 June (jointing through grain filling) became a higher percentage of the total growing season precipitation.

Similar results are reported for experiments with limited irrigation in which water is withheld during vegetative growth and applied during flowering and grain filling. When a set amount of limited irrigation was applied to

corn, sorghum, and wheat during flowering and grain filling, as opposed to applications throughout the entire growing season, water use efficiency increased by 19%, 42%, and 29%, respectively.^[26] Similarly, water applied to winter wheat during the boot to anthesis stages averaged twice the yield response of water applied during tillering through jointing, without substantially increasing seasonal evapotranspiration.^[25]

APPROPRIATE FERTILIZER APPLICATION

In most cases when water supply is fixed, any management factor that increases yield will increase water use efficiency because evapotranspiration is little affected by the management.^[2,27] For example, dryland winter wheat in northeast Colorado had increased water use efficiency with increased level of applied nitrogen (N) fertilizer (Fig. 3).^[28] However, water use efficiency can decline when high levels of N fertilization produce excessive vegetative growth that uses up the limited soil water supply early in the growing season.^[29] When water availability is low, plants grown under extremely high N fertility conditions may undergo more water stress during the more critical flowering and grain-filling stages than plants grown under lower fertility. Consequently, grain yield was reduced at the two highest fertilizer application rates while water use remained nearly the same for the highest N application levels.

REDUCING OR ELIMINATING FALLOW

One of the most productive ways of improving cropping system water use efficiency is to eliminate fallow periods. Cropping systems in the Great Plains have traditionally relied on the practice of summer fallowing in which one

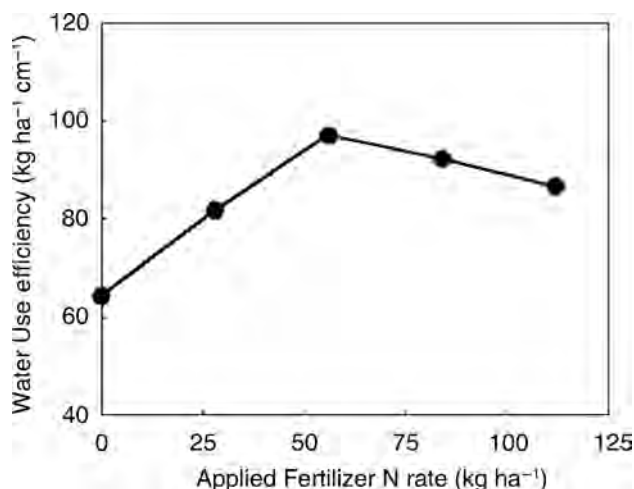


Fig. 3 Influence of N fertilizer application rate on winter wheat water use efficiency at Akron, Colorado.

Source: From Nielsen & Halvorson.^[28]

crop of winter wheat is grown every other year. The purpose of this was primarily to reduce the erratic yields (and sometimes total crop failure) associated with annual cropping. However, this practice is extremely inefficient in its storage and use of precipitation.^[30] Most of the precipitation received during the summer of fallow just preceding wheat planting in the wheat–fallow system is lost to evaporation. Intensifying cropping systems by reducing or eliminating fallow periods increases water use efficiency of the system.

Data averaged over 6 years at Akron, Colorado, show an 18% increase in water use efficiency through reduction of tillage in the wheat–fallow system, going from conventional tillage (35.5 kg ha⁻¹ cm⁻¹) to no tillage (41.9 kg ha⁻¹ cm⁻¹). When corn is added to the system such that two crops are produced in 3 years (wheat–corn–fallow), the water use efficiency increased by 55% (to 54.9 kg ha⁻¹ cm⁻¹). With the continuous wheat–corn–millet system, the water use efficiency increased by 82% (to 64.4 kg ha⁻¹ cm⁻¹). An even higher system water use efficiency (87.3 kg ha⁻¹ cm⁻¹) was reported in a continuous dryland cropping experiment (8 years) with barley, corn, and winter wheat, also at Akron, Colorado.^[31]

CONCLUSION

In semiarid regions where lack of water is limiting to crop yield, water use efficiency needs to be maximized to get the most from this scarce resource. This can be accomplished through several management techniques to improve the cropping environment that includes reducing tillage to maintain surface crop residues, minimize evaporation, and trap snow; applying irrigations only during critical growth stages; matching fertilizer application rates to the expected available water condition; and employing crop rotations that reduce or eliminate fallow periods.

REFERENCES

1. Tanner, C.B.; Sinclair, T.R. Efficient water use in crop production: Research or re-search? In *Limitations to Efficient Water Use in Crop Production*; Taylor, H.M., Jordan, W.R., Sinclair, T.R., Eds.; ASA–CSSA–SSSA: Madison, 1983; 27.
2. Stewart, B.A.; Steiner, J.L. Water use efficiency. In *Dryland Agriculture: Strategies for Sustainability*; Singh, R.P., Parr, J.F., Stewart, B.A., Eds.; Springer: New York, 1990; Vol. 13, 151–173.
3. Smika, D.E.; Wicks, G.A. Soil water storage during fallow in the central great plains as influenced by tillage and herbicide treatments. *Soil Sci. Soc. Am. Proc.* **1968**, *32*, 591–595.
4. Tanaka, D.L.; Aase, J.K. Fallow method influences on soil water and precipitation storage efficiency. *Soil Till. Res.* **1987**, *9*, 307–316.

5. Norwood, C.A. Water use and yield of dryland row crops as affected by tillage. *Agron. J.* **1999**, *91*, 108–115.
6. Caprio, J.M. Potentials for harvesting water from snow. In *Great Plains Agricultural Council Publication No. 120*, Proceedings of the Symposium on Snow Management for Agriculture, Swift Current, Saskatchewan, July 1985; Steppuhn, H., Nicholaichuk, W., Eds.; University of Nebraska: Lincoln, 1986; 109–131.
7. Greb, B.W.; Smika, D.E.; Black, A.L. Effect of straw mulch rates on soil water storage during summer fallow in the great plains. *Soil Sci. Soc. Am. Proc.* **1967**, *31*, 556–559.
8. Aiken, R.M.; Flerchinger, G.N.; Farahani, H.J.; Johnsen, K.E. Energy balance simulation for surface soil and residue temperatures with incomplete cover. *Agron. J.* **1997**, *89*, 405–416.
9. Van Doren, D.M., Jr.; Allmaras, R.R. Effect of residue management practices on the soil physical environment, microclimate, and plant growth. In *Crop Residue Management Systems*; Oschwald, W.R., Ed.; ASA Spec. Publ. 31; ASA-CSSA-SSSA: Madison, 1978; 49–83.
10. Unger, P.W.; Stewart, B.A. Soil management for efficient water use: An overview. In *Limitations to Efficient Water Use in Crop Production*; Taylor, H.M., Jordan, W.R., Sinclair, T.R., Eds.; ASA-CSSA-SSSA: Madison, 1983; 419–460.
11. Bilbro, J.D.; Fryrear, D.W. Wind erosion losses as related to plant silhouette and soil cover. *Agron. J.* **1994**, *86*, 550–553.
12. Siddoway, F.H.; Chepil, W.S.; Armbrust, D.V. Effect of kind, amount, and placement of residue on wind erosion control. *Trans. ASAE* **1965**, *8*, 327–331.
13. Nielsen, D.C.; Aiken, R.M. Wind speed above and within sunflower stalks varying in height and population. *J. Soil Water Conserv.* **1998**, *53*, 347–352.
14. Smika, D.E. Soil water changes as related to position of wheat straw mulch on the soil surface. *Soil Sci. Soc. Am. J.* **1983**, *47*, 988–991.
15. McMaster, G.S.; Aiken, R.M.; Nielsen, D.C. Optimizing wheat cutting height for harvest efficiency and soil and water conservation. *Agron. J.* **2000**, *92*, 1104–1108.
16. Steppuhn, H.; Erickson, D.; Zentner, R.P.; Nicholaichuk, W. Benefit/cost ratios for snow management techniques in semiarid climates. In *Great Plains Agricultural Council Publication No. 120*, Proceedings of the Symposium on Snow Management for Agriculture, Swift Current, Saskatchewan, July 1985; Steppuhn, H., Nicholaichuk, W., Eds.; University of Nebraska: Lincoln, 1986; 613–656.
17. Bauer, A.; Tanaka, D. Stubble height effects on non-growing season water conservation. In *Great Plains Agricultural Council Publication No. 120*, Proceedings of the Symposium on Snow Management for Agriculture, Swift Current, Saskatchewan, July 1985; Steppuhn, H., Nicholaichuk, W., Eds.; University of Nebraska: Lincoln, 1986; 255–272.
18. Smika, D.E.; Page, A.B.; Mickelson, R.H. Snow water management for crop production in the central great plains. In *Great Plains Agricultural Council Publication No. 120*, Proceedings of the Symposium on Snow Management for Agriculture, Swift Current, Saskatchewan, July 1985; Steppuhn, H., Nicholaichuk, W., Eds.; University of Nebraska: Lincoln, 1986; 335–344.
19. Campbell, C.A.; McConkey, B.G.; Zentner, R.P.; Selles, F.; Dyck, F.B. Benefits of wheat stubble strips for conserving snow in southwestern Saskatchewan. *J. Soil Water Conserv.* **1992**, *47*, 112–115.
20. Nielsen, D.C. Snow catch and soil water recharge in standing sunflower residue. *J. Prod. Agric.* **1998**, *11*, 476–480.
21. Reis, R.F.; Power, J.F. Increased soil water storage and herbage production from snow catch in North Dakota. *J. Range Manag.* **1981**, *34*, 359–362.
22. Greb, B.W. *Snowfall and Its Potential Management in the Semiarid Central Great Plains*, USDA-SEA Agric. Rev. Manage ARM-W-18; U.S. Government Print Office: Washington, 1980; 16–17.
23. Nielsen, D.C.; Anderson, R.L.; Bowman, R.A.; Aiken, R.M.; Vigil, M.F.; Benjamin, J.G. Winter wheat and proso millet yield reduction due to sunflower in rotation. *J. Prod. Agric.* **1999**, *12*, 193–197.
24. Shanahan, J.F.; Nielsen, D.C. Influence of growth retardants (anti-gibberellins) on corn vegetative growth, water use, and grain yield under different levels of water stress. *Agron. J.* **1987**, *79*, 103–109.
25. Musick, J.T.; Jones, O.R.; Stewart, B.A.; Dusek, D.A. Water-yield relationships for irrigated and dryland wheat in the U.S. southern plains. *Agron. J.* **1994**, *86*, 980–986.
26. Shawcroft, R.W. Limited irrigation may drop yield, up profit. *Colorado Rancher Farmer* **1983**, *37*, 35–38.
27. Viets, F.G., Jr. Fertilizers and efficient use of water. *Adv. Agron.* **1962**, *14*, 223–264.
28. Nielsen, D.C.; Halvorson, A.D. Nitrogen fertility influence on water stress and yield of winter wheat. *Agron. J.* **1991**, *83*, 1065–1070.
29. Power, J.F. Soil management for efficient water use: Soil fertility. In *Limitations to Efficient Water Use in Crop Production*; Taylor, H.M., Jordan, W.R., Sinclair, T.R., Eds.; ASA-CSSA-SSSA: Madison, 1983; 461–470.
30. Farahani, H.J.; Peterson, G.A.; Westfall, D.G. Dryland cropping intensification: A fundamental solution to efficient use of precipitation. *Adv. Agron.* **1998**, *64*, 197–223.
31. Halvorson, A.D.; Reule, C.A. Nitrogen fertilizer requirements in an annual dryland cropping system. *Agron. J.* **1994**, *86*, 315–318.

Water Measurement: Methods

Des McGarry

Natural Resource Sciences, Queensland Department of Natural Resources and Mines, Indooroopilly, Queensland, Australia

Abstract

This entry compares four water measurement methods—thermogravimetric, neutron thermalization, frequency domain reflectometry, and time domain reflectometry.

INTRODUCTION

The measurement of soil water is fundamental to many agricultural, forestry, engineering, landscape dynamics, meteorological, and hydrological investigations,^[1] aimed at understanding soils' chemical, physical, mechanical, and biological functions and relationships.^[2] Measurements may be direct or indirect, field or laboratory based, occasional or continuous and include the size and time course of the soil moisture content and water movement into, through, retained by and passing out of a soil profile or specific soil layer. Several measurement techniques have been devised in response to these needs, and this entry will review aspects of four—thermogravimetric, neutron thermalization, capacitance [frequency domain reflectometry (FDR)], and time domain reflectometry (TDR). Less common techniques of measuring soil water, including ground penetrating radar, active and passive microwave, electromagnetic induction, and heat dissipation, whether *in situ* or by remote sensing, are reviewed elsewhere.^[3,4]

Here, emphasis will be on a comparison of the strengths and weaknesses of the four measurement techniques (Table 1). This draws on several, detailed reviews of soil water content and its measurement.^[1–4,9–15] Comparisons of the three indirect methods have been conducted.^[16–18] A detailed review of neutron, capacitance, and TDR probes and their use in measuring and monitoring soil water status are available,^[19] revised with a new set of case studies.^[7]

To provide a basis for comparison of the measurement techniques reviewed here, only a specific form of each apparatus will be discussed, namely apparatus that is used to measure soil water content profiles, i.e., water content at discrete, user-chosen depth intervals, commonly to 1 or 2 m depth. This is particularly important for the capacitance and TDR devices where there are many different designs. For the thermogravimetric technique, the determination of soil water profiles is conducted using either one continuous or many samples of individual, selected depths, obtained using thin-walled metal cylinders. Continuous cores are precisely cut into designated lengths to facilitate bulk

density calculation, and hence volumetric water content. For the three indirect measures, to obtain a similar profile of soil water contents as the thermogravimetric technique, the form of each apparatus discussed will be: for the neutron moisture meter (NMM), a “profiling meter”;^[6] for the capacitance probe, one or more pairs of cylindrical rings separated by a non-conducting plastic ring;^[20] and for the TDR, a “segmented probe” consisting of two parallel metal bars with resin between them, which utilizes diode shorting to achieve horizontal probe segmentation.^[21]

SOIL WATER CONTENT

Soil consists of a three-phase system composed of solid (soil particles), liquid (water and solutes), and gas (air), where the solid is not a continuous mass but is broken into individual grains or aggregates with spaces (soil pores) between them that can be filled with the other two phases (water and air). The per mass or per volume fraction of water in the soil is expressed as “water content” or alternatively “soil wetness.”^[2]

DIRECT AND INDIRECT MEASUREMENT OF SOIL WATER CONTENT

Direct techniques of determining soil water content are those where the water in a sample of soil is removed or separated from the soil matrix by evaporation, leaching, or chemical reaction^[15] with subsequent direct measurement of the amount of water removed. The first will only be considered here, where the water in a sample is removed by oven drying and the amount removed is determined by measurement of the change in mass of the sample.

Indirect techniques involve measurement of some physical or chemical property of the soil that is affected by or is a function of soil water content.^[1,4] The most common use of indirect techniques is to measure either the distribution of soil water content at one time or the

Table 1 Strengths and weaknesses of four methods of soil water measurement.

Thermogravimetric	Neutron	Capacitance	TDR
Strengths			
The standard, for characterizing soil water content An absolute, directly measured quantity Calibration standard for indirect methods Simple field and laboratory methods	Not destructive (after initial installation of access tube or probe) No need for laboratory drying, weighing, etc., after initial calibration		
	Can be calibrated to great precision	Robust and stable instrumentation	
	Operate to far greater depths than other methods, e.g., 96 m ^[5]	Fast response times—rapid data gathering (Fig. 1)	
	Zone of influence is largest of all indirect methods; radius range: 15–50 cm ub wet and dry soils, respectively ^[6]	Accuracy with site-specific calibration and good probe-soil contact No health and safety problems	
	Least affected by soil disturbance, smearing, and cracks (air gaps) near access tube	Automatic and continuous logging of water data over long periods (Fig. 1) Remote (telemetry) collection of data possible	
Weakness			
Destructive sampling required Large labor requirements Time consuming: transport, weighing, and drying of samples Provides neither immediate nor continuous water content Accurate bulk density measurement needed to convert to θ_v	Accuracy and precision of indirect methods depend on the uniquenesses of the relationship between the measured property and θ_v , and the minimal influence of other soil properties that are not soil water ^[4] Cost ^[7] Perceived health issues Statutory safety requirements: For all users To acquire, possess, use, relocate, travel with, and supply devices Licensing requirements	Small zone of influence of the electrodes Air gaps beside access tube give dielectric confusion; poor data Care required in access tube installation to achieve a tight fit with no air gaps Temperature sensitivity of some capacitance devices	Single calibration equation suitable for a wide range of soils Less sensitive to the several interferences commonly present in soils that effect capacitance

(Continued)

Table 1 Strengths and weaknesses of four methods of soil water measurement. (Continued)

Thermogravimetric	Neutron	Capacitance	TDR
Soil shrink—swell effects: Bulk density changes so θ_v requires determination over wide water content range Cracks cause problems with representative sample locations (Fig. 2) Necessary to present water data against a mass (not depth) coordinate ^[8]	Probes cannot be left unattended, so: Remote data collection impossible Inter-site travel laborious Repeated lifting of heavy equipment Lengthy stays at sites to manually work probe Wet weather access problems—leads to data gaps (Fig. 3) Repeated visits to probe location gives site tramping that may effect results Calibration required, though usually linear ^[6] Data not continuous (Fig. 3) and data collection cannot be automated Shrinkage cracks can effect values ^[9] Cannot perform measurements at several depths concurrently	Shrinkage crack influences may preclude use in moist to dry Vertisols Calibration required for accurate water content measurement, though factor calibration useable to assess relative differences in water content ^[7] Measure sensitive to <i>in situ</i> soil electrical conductivity—as salinity increases soil appears wetter that it is	Care in probe installation to achieve a tight fit Shrinkage crack influences may preclude use in moist to dry Vertisols Calibration necessary in “extreme” soil conditions, e.g., large clay content, ferritic soils (Oxusols), large organic matter levels

Source: From Gardner, Bell, et al.,^[1] Hillel,^[2] White & Zegelin,^[3] Topp & Ferre,^[4] Pikul,^[10] Evett,^[11] Starr & Paltineanu,^[12] Evett,^[13] Gardner,^[14] Gardner,^[15] Evett, Ruthardt, et al.,^[16] and Evett, Laurent, et al.^[17]

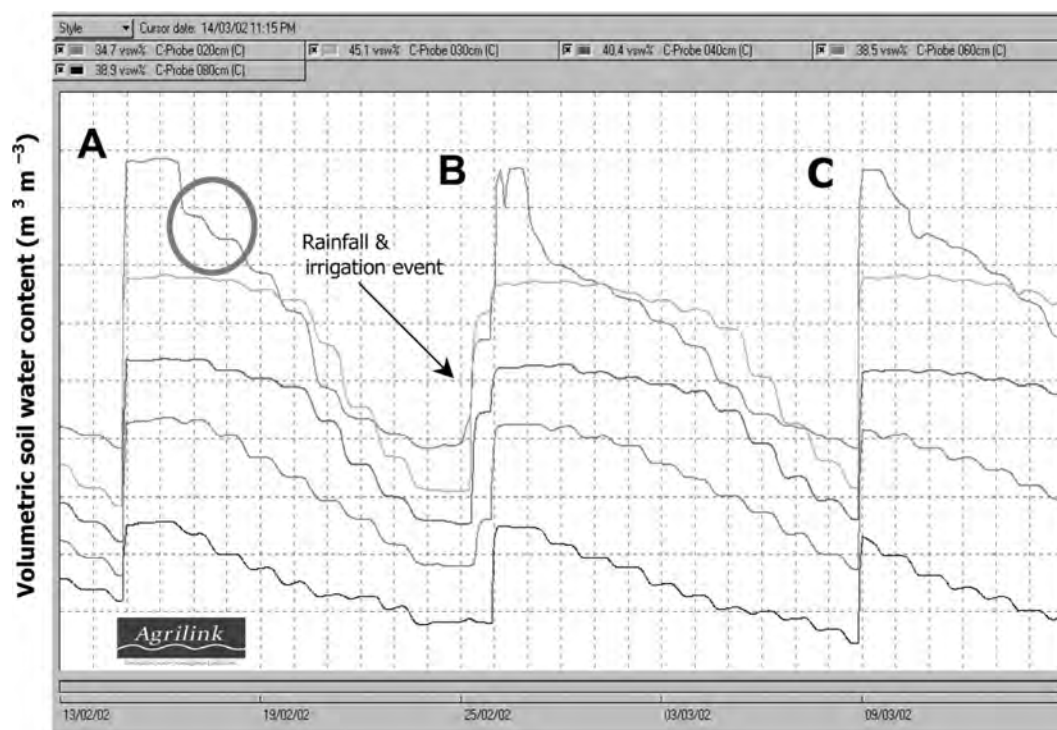


Fig. 1 Capacitance probe data collected at five depths (20, 30, 40, 60, and 80 cm) over a one-month period in an actively growing, irrigated cotton crop on a Vertisol, near St. George, Australia. The data for each depth are an arithmetically smoothed average of readings every 15 minutes (96 readings a day), collected remotely using telemetry to a telephone landline and then computer downloaded. The irrigation events are given (A, B, and C). Evident is 1) water infiltration to each depth at each irrigation, 2) even drying of soil water at all depths—suggesting actively extracting roots at all depths, and 3) diurnal fluctuations (a “night–day–night” example circled) in water extraction showing active and less-active plant growth (hence water extraction) during the day and at night.

Note: The lack of a site-specific calibration causes inability to present actual values of volumetric water contents for the y-axis. The y-axis is presented as a volumetric soil moisture percentage and is a repeatable number. The data are sufficient to plan the time and quantity of irrigation at the farm level.

Source: Courtesy of Agrilink Company.

change in soil profile water content over time at one or several places in a field or plot.^[1]

Thermogravimetric Technique

Gravimetric sampling combined with oven drying is generally accepted as both the standard, direct technique for characterizing soil water content as well as the calibration standard for many indirect soil water measurement techniques. Sampling, laboratory procedures, and calculations are described elsewhere in this encyclopedia and related volumes.^[10] In simple terms, field (or laboratory) wet soil samples are oven dried (commonly 105°C) to a constant mass and the gravimetric water content (θ_g) of the initial sample is expressed as the mass of water (g) per unit mass (g) of oven-dried soil.

Volumetric water content (θ_v) expresses soil water content in terms of the volume of water per unit volume of soil. The only direct method of determining volumetric water content is to obtain undisturbed soil samples, the volume of which is known. This is done most acceptably using

thin-walled metal cylinders of known volume.^[22] θ_v is calculated by multiplying θ_g with the dry soil bulk density^[10] and is the preferred expression of soil water content when considering water contents of different soils, water use under crops, and application rates in irrigation situations,^[19] as θ_v may be expressed as the equivalent depth of water because θ_v is numerically equal to the ratio of the depth of stored water per unit depth of soil. This greatly facilitates computations of soil water budgets where inputs of water (commonly in millimeters of water) from rain or irrigation are compared with outputs due to evapotranspiration and drainage.

The thermogravimetric technique has several important strengths as well as weaknesses (Table 1). Positives include the universally accepted methodology, utilizing commonly available laboratory equipment (balances and ovens), as well as the ability to measure unique, small depth (or spatial) increments (down to millimeter scale) if required. Negatives include the need for destructive soil sampling (particularly if replication done), transportation of samples (often of

considerable weight and bulk) to the laboratory, weighing, oven drying (to constant mass), and subsequent data calculations. The outcome tends to be small, time-specific (non-continuous) data sets that require significant physical effort and time to obtain (Fig. 4). Similar to all methods of determining soil water content, Vertisols (cracking clays) pose specific sampling challenges, particularly obtaining representative bulk volumes in dry soils (Fig. 2).

Neutron Thermalization

This indirect technique of determining soil water employs the NMM. The NMM was first used in the 1950s and

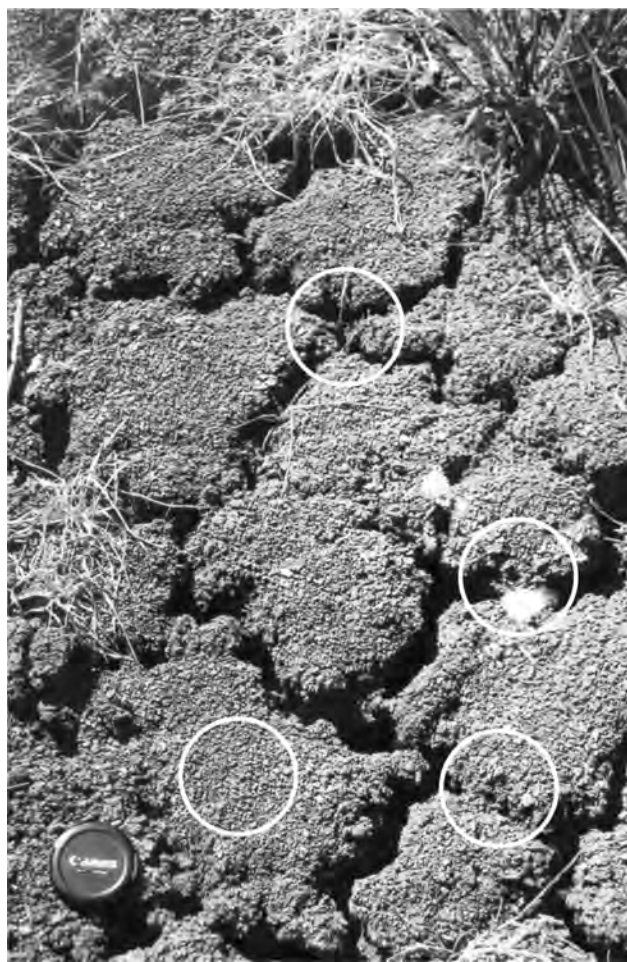


Fig. 2 An example of the challenges of obtaining a representative sample of soil water content in swell/shrink soils, exemplified on the soil surface of a Vertisol in an irrigated (but dry) wheat field, near Narrabri, New South Wales, Australia. The circles (each 100-mm diameter) represent four possible locations of a 100-mm diameter sampling ring. The “correct” soil water content would require sampling of, at best, a composite of all four. In addition, the selection of sampling location based on visible surface cracks may inaccurately sample lower depths as surface cracks do not necessarily continue vertically to depth.

became popular in the 1980s, when it was used extensively by consultants for irrigation scheduling.^[19] Principles, field procedures, calibration, and the associated derivation of data are described in detail elsewhere in this encyclopedia, in related volumes,^[11] and in several reviews.^[6,19,23–25] In simple terms, the neutron moderation technique is based on the response of fast moving neutrons that are slowed (thermalized) by an elastic collision with existing hydrogen particles in the soil. Hydrogen is present in the soil as a constituent of soil organic matter, clay minerals, and water. However, water is the form of hydrogen that is most likely to change from measurement to measurement.^[7] The volume fraction of water in the soil is obtained from a calibration equation that relates volumetric water content (measured by destructive soil core sampling) with the count ratio where the count ratio is the ratio of slow neutrons detected to the number of fast neutrons emitted when the probe is placed in a reproducible standard medium, commonly pure water.^[6] As with all the indirect methods presented here, consideration needs to be given to the “zone of influence” of the device, defined as that area in which a change in the moisture content of the soil will result in a detectable change in the detection systems of the different devices. Common to all devices, the zone of influence both surrounds and extends along some length of the probe and varies inversely with water content. Changes in the water content of the soil immediately adjacent to the probe have a far greater effect on the instrument readings than equivalent changes in water content occurring further away. The zone of influence for a NMM is approximately a sphere with radii of 15 and 50 cm in wet and dry soil, respectively.^[6]

In terms of calibration, and common to all of the indirect measurements, if absolute volumetric water content data are required (compared with changes in soil water content), then specific site calibration is required (rather than the “universal” or soil type specific calibrations supplied by some manufacturers). Site-customized calibration removes the potential effects that several sources of interference may have on the concentration of thermalized neutrons, including bulk density, alternate sources of hydrogen ions in the soil (e.g., organic matter and certain clay minerals), soils high in iron, magnetite, and chloride salts.^[6] Strengths and weaknesses of the technique are detailed in Table 1. Positives include linear calibration of count ratio and volumetric water contents over a wide range of water contents, the device is considered as the most accurate (indirect) method of determining soil water content (with site-specific calibration),^[6] and the large zone of influence (considered by some disadvantage as it averages spatial variation in water content) causes less effect of air gaps around access tubes. Negatives include cost (relative to capacitance probes), licensing and transport restrictions, inability to automate what can be lengthy data collection times, and the need for repeated site access, which can alter the hydrological representativeness of the monitored location.

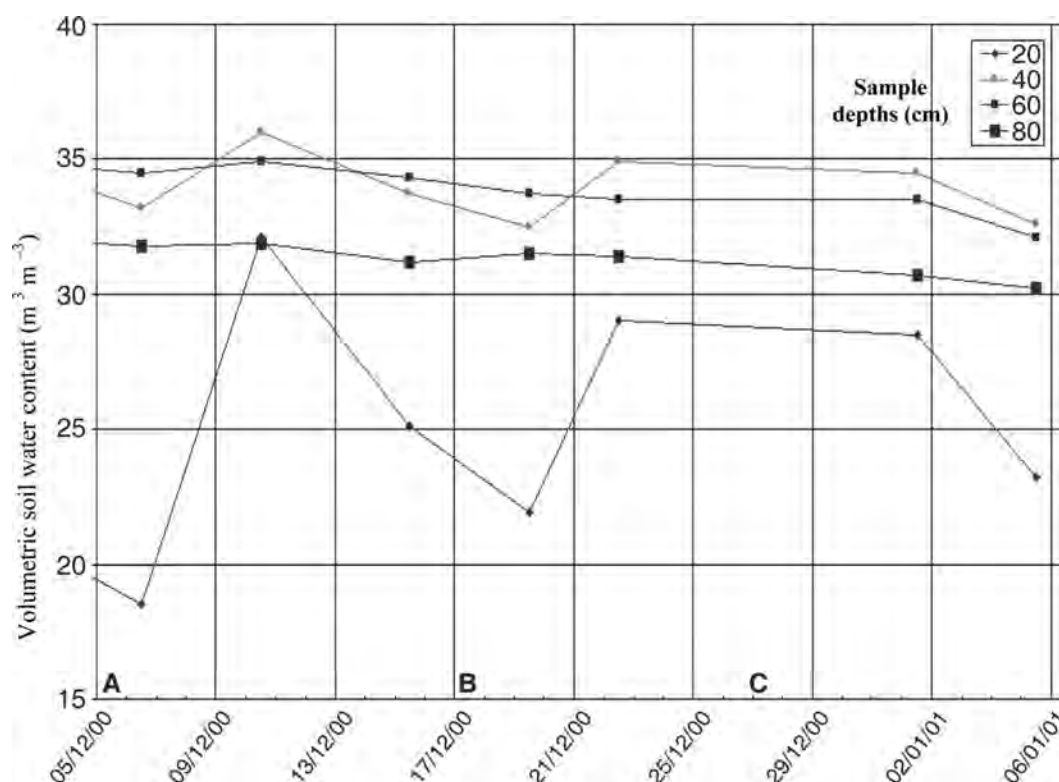


Fig. 3 Time series data collected at four depths over a one-month period by a commercial cotton crop consultant using a NMM in an irrigated cotton field, St George, Queensland, Australia. The three irrigation events are given as A, B, and C. Evident is the inability of the consultant to sample frequently enough to monitor the soil response(s) to the irrigations, despite an attempt to sample on a five-day interval. Excess workload caused a missed reading event on January 28, 2000.

Source: Courtesy of Agrilink Company.

FDR (Capacitance) and TDR

The second and third indirect techniques of determining soil water—FDR and TDR—are both based on measuring the dielectric constant of the soil water media.^[4] The dielectric constant is a measure of the capacity of a non-conducting material to transmit electromagnetic waves or pulses.^[7,19] The dielectric constant of water is relatively large, whereas the values for both soil solids and air are far smaller. The value of the dielectric constant is largely determined by the fraction of water present—as the water content of the soil increases, the dielectric constant increases.^[2] Studies of capacitance and soil dielectric properties to measure soil water content date from the 1930s, but advances in reliability and affordability of electronics and microcomputers have greatly increased this usage.^[3]

Frequency Domain Reflectometry

FDR measures the soil dielectric by placing the soil (in effect) between two electrical plates to form a capacitor.^[7]

When an alternating voltage is applied to the electrodes, a frequency can be measured. This frequency varies with the soil dielectric constant. Using calibration protocols, the

functional relationship between oscillation frequency and water content can be determined.^[7,12] Capacitance probes are the subject of several reviews of their theory, field procedures, and calibration elsewhere in this encyclopedia, in related volumes,^[12] and in several reviews.^[1,3,4,16–20] There are several probe designs^[7] although only one will be considered here, where one or more pairs of cylindrical metal rings separated by a non-conductive plastic ring are mounted on a plastic rod.^[20] This probe is designed for semipermanent (commonly season-long) insertion into a previously installed plastic access tube^[12] with the multiple sensors located at set depths to monitor soil water up to 2 m depth. The zone of influence for this configuration is primarily within 10 cm radius of the access tube and about 10 cm along the rod, centered at the insulator between the two metal rings.^[20] However, 90% of the influence is within a zone that stretches from about 3 cm above and below the center of the plastic ring and 3 cm in the radial direction.^[26] Strengths and weaknesses of the technique are detailed in Table 1. Positives include low-cost, highly sensitive, robust, and stable instrumentation, fast response time, ability to measure time steps seconds apart, ease of use, loggable and amenability to automated and telemetry operated real-time monitoring.^[4] Negatives include smaller sphere of influence (than NMM),

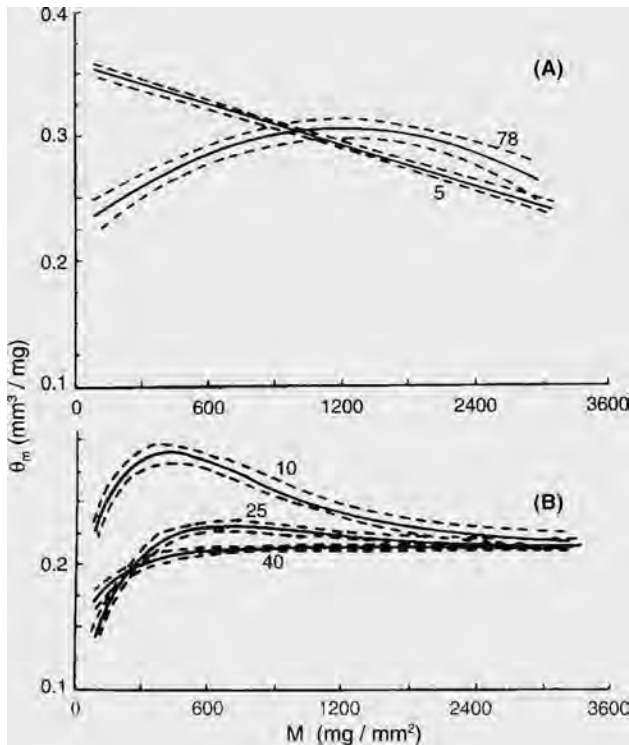


Fig. 4 An example of soil water data from the thermogravimetric method. In this example, curved lines (with standard error as dashed lines) have been fitted to the original data. The data are from a field trial on a Vertisol, so are presented on a mass coordinate basis. Soil cores were collected in triplicate in 0.1 m increments to 1.5 m after each of two crop irrigations: on each of (A) 10, 25, and 40 days after one and on (B) 5 and 78 days after another. The total number of soil samples was 245; individually weighed, before and after oven drying, and the data manually entered for water content calculation; a very large workload for minimal output.

Source: From McGarry & Malafant.^[8]

hence measurement reliability influenced by air gaps around access tubes, and difficulties with soil sampling (for volumetric water content) within the device-measured volume. Temperature effects are under discussion.^[4]

Time Domain Reflectometry

There have been many reviews of TDR theory, the underlying physics, the various apparatus available, their field and laboratory usage, and the advantages and disadvantages of the technique.^[13,7,16–19,21,27] TDR measures the time taken for an electromagnetic pulse (a fast rise voltage step) to travel along metal rods placed in the soil, with the travel time being a function of the apparent dielectric constant of the soil.^[28] Because the travel time of the signal depends on the dielectric constant of the soil, it is possible to obtain a linear calibrated relationship between the travel time and the soil water content.

A variety of forms of TDR soil probes with different application capabilities are available.^[7,21] Parallel rods

were the most common form and facilitated water content measurement at selected depths. The rods were either pushed into the soil surface or inserted at selected depths in the soil profile, commonly into the sides of an access trench. Further, segmented probes have been devised to facilitate concurrent measurement of water content at up to five soil depths, so providing the water content profile and the total stored water profile from a single installation.^[21] This type of soil moisture probe is designed to emulate a two-wire, parallel conductor transmission line. It has a rectangular cross section ($16 \times 12.5 \text{ mm}^2$) and is constructed of two stainless steel conductors laminated to an epoxy core. It is available in various configurations with different lengths and numbers of segments. Segmented probes provide the opportunity for real-time, remote soil water monitoring, and graphical data presentation similar to capacitance probes.

Strengths and weaknesses of the technique are detailed in Table 1. An advantage of the TDR devices is that the determination of the travel time excludes the influence of low-frequency components of the electric pulse; hence systems employing TDR are relatively less sensitive to interferences of soil temperature, salinity, and clay content and type than are capacitance systems. A universal calibration for TDR probes for a wide range of soils has been developed, based on a laboratory-developed empirical relationship between the soil dielectric constant and the volumetric water content.^[21] However, similar to the other indirect devices, improved data accuracy will be achieved with site-specific calibration. Other positives are similar to those of the capacitance devices, given above, including the potential for automated water content profile determinations. Negatives are cost and as with capacitance, the smaller sphere of influence of TDR (than NMM) and associated problems given above.

CONCLUSION

The more modern techniques of capacitance and TDR, if used within a reasonable range of restrictions, provide attractive benefits over the other two techniques reviewed. With both capacitance and TDR, an attraction is the ability to automate the collection of remote (datalogger and telemetry), essentially continuous soil water data to 2 m depth. However, their small zone of influence necessitates very careful installation, and data collection should be restricted to non-swelling soil, or wetter, non- or minimally cracked soil to avoid problems with air gaps around capacitance access tubes and TDR probes, with resultant poor data. Neutron probes provide the best available method for repeated measurement of soil profile volumetric water content,^[6] have the largest zone of influence, and can access depths far greater than the other two indirect techniques. However, operational restrictions, perceived health fears from radioactive materials, and their inability to provide *in situ* automatic measurements (making them labor intensive) make them unattractive to some users.

REFERENCES

- Gardner, C.M.K.; Bell, J.P.; Cooper, J.D.; Dean, T.J.; Hodnett, M.G.; Gardner, N. Soil water content. In *Soil Analysis—Physical Methods*; Smith, K.A., Mullins, C.E., Eds.; Marcel Dekker, Inc.: New York, 1991; 1–73.
- Hillel, D. Content and potential of soil water. In *Environmental Soil Physics*; Academic Press: San Diego, 1998; 129–142.
- White, I.; Zegelin, S.J. Electric and dielectric methods for monitoring soil-water content. In *Handbook of Vadose Zone Characterisation & Monitoring*; Wilson, L.G., Lorne, G.E., Cullen, S.J., Eds.; Lewis Publishers: Boca Raton, 1995; 343–385.
- Topp, G.C.; Ferre, P.A. The soil solution phase. In *Methods of Soil Analysis, Part 4: Physical Methods*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America, Inc.: Madison, 2002; 417–534.
- Kramer, J.H.; Cullen, S.J.; Everett, L.G. Vadose zone monitoring with the neutron moisture probe. In *Handbook of Vadose Zone Characterisation and Monitoring*; Wilson, L.G., Lorne, G.E., Cullen, S.J., Eds.; Lewis Publishers: Boca Raton, 1995; 291–309.
- Hignett, C.; Evett, S.R. Neutron thermalisation. In *Methods of Soil Analysis, Part 4: Physical Methods*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America, Inc.: Madison, 2002; 501–521.
- Charlesworth, P. Soil water monitoring. In *Irrigation Insights*, No. 1, 2nd Ed.; CSIRO Land and Water: Canberra, 2004.
- McGarry, D.; Malafant, K.W.J. A cumulative mass coordinate to determine water profile changes in variable volume soil. *Soil Sci. Soc. Am. J.* **1987**, *51*, 850–854.
- Li, J.; Smith, D.W.; Fityus, S.G. The effect of a gap between the access tube and the soil during neutron probe measurements. *Aust. J. Soil Res.* **2003**, *41*, 151–164.
- Pikul, J.L. Soil water, gravimetric measurement. In *Encyclopedia of Water Science*; Stewart, B.A., Howell, T.A., Eds.; Marcel Dekker, Inc.: New York, 2003; 879–881.
- Evett, S.R. Soil water measurement by neutron thermalization. In *Encyclopedia of Water Science*; Stewart, B.A., Howell, T.A., Eds.; Marcel Dekker, Inc.: New York, 2003; 889–893.
- Starr, T.J.; Paltineanu, I.C. Soil water measurement by capacitance. In *Encyclopedia of Water Science*; Stewart, B.A., Howell, T.A., Eds.; Marcel Dekker, Inc.: New York, 2003; 885–888.
- Evett, S.R. Soil water measurement by time domain reflectometry. In *Encyclopedia of Water Science*; Stewart, B.A., Howell, T.A., Eds.; Marcel Dekker, Inc.: New York, 2003; 894–898.
- Gardner, W.H. Water content. In *Methods of Soil Analysis, Part 1*; Black, C.A., Ed.; American Society of Agronomy: Madison, 1965; 82–187.
- Gardner, W.H. Water content. In *Methods of Soil Analysis, Part 1*; Klute, C.A., Ed.; 2nd Ed.; American Society of Agronomy: Madison, 1986; 493–544.
- Evett, S.R.; Ruthardt, B.B.; Kottkamp, S.T.; Howell, T.A.; Schneider, A.D.; Tolk, J.A. Accuracy and precision of soil water measurements by neutron, capacitance and TDR methods. In *Proceedings of the 17th World Congress of Soil Science*, Bangkok, Thailand, Aug 14–21, 2002; American Chemical Society: Washington, 1996; 318 pp.
- Evett, S.R.; Laurent, J.P.; Cepuder, S.; Howell, P.; Hignett, C. Neutron scattering, capacitance and TDR soil water measurements compared on four continents. In *Proceedings of the 17th World Congress of Soil Science*, Bangkok, Thailand, Aug 14–21, 2001; American Chemical Society: Washington, DC, 1996; 1021.
- Evett, S.R. Some aspects of time domain reflectometry (TDR), neutron scattering, and capacitance methods of soil water content measurement. In *Comparison of Soil Water Measurement Using the Neutron Scattering, Time Domain Reflectometry and Capacitance Methods*; IAEA-TECDOC-1137, 2000, International Atomic Energy Agency: Vienna, 2003; 5–49.
- Charlesworth, P. Soil water monitoring. *Irrigation Insights*, No. 1; CSIRO Land and Water: Canberra, 2000.
- Starr, J.L.; Paltineanu, I.C. Capacitance devices. In *Methods of Soil Analysis, Part 4: Physical Methods*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America, Inc.: Madison, 2002; 463–474.
- Ferre, P.A.; Topp, G.C. Time domain reflectometry. In *Methods of Soil Analysis, Part 4: Physical Methods*; Dane, J.H., Topp, G.C., Eds.; Soil Science Society of America, Inc.: Madison, 2002; 434–446.
- Lowery, B.; Hickey, W.J.; Arsha, M.A.; Lal, R. Soil water parameters and soil quality. In *Methods for Assessing Soil Quality*; Doran, J.W., Jones, A.J., Eds.; Soil Science Society of America Special Publication Number 49; Madison, 1996; 143–155.
- Greacen, E.L. *Soil Water Assessment by the Neutron Method*; CSIRO: Australia, 1981.
- IAEA. *Nuclear Techniques to Assess Irrigation Schedules for Field Crops*; International Atomic Energy Agency: Vienna, 1996.
- IAEA. *Neutron and Gamma Probes: Their Use in Agronomy*; International Atomic Energy Agency: Vienna, 2002.
- Kelleners, T.J.; Soppe, R.W.O.; Robinson, D.A.; Schaap, M.G.; Ayars, J.E.; Skaggs, T.H. Calibration of capacitance probe sensors using electric circuit theory. *Soil Sci. Soc. Am. J.* **2004**, *68*, 430–439.
- Noborio, K. Measurement of soil water content and electrical conductivity by time domain reflectometry: A review. *Comput. Electron. Agric.* **2001**, *31*, 213–237.
- Heathman, G.C.; Starks, P.J.; Brown, M.A. Time domain reflectometry field calibration in the Little Washita River Experimental Watershed. *Soil Sci. Soc. Am. J.* **2003**, *67*, 52–61.

Water Range: Least-Limiting

Alvaro Pires da Silva

Departamento de Solos e Nutrição de Plantas, University of Sao Paulo, Piracicaba, Brazil

Beverly D. Kay

Department of Land Resource Science, University of Guelph, Guelph, Ontario, Canada

Cássio A. Tormena

Departamento de Agronomia, State University of Maringa, Maringa, Brazil

Silvia Imhoff

Facultad de Ciencias Agrarias, National University of the Litoral, Esperanza, Argentina

Douglas L. Karlen

National Laboratory for Agriculture and the Environment, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Ames, Iowa, U.S.A.

Joseph G. Benjamin

Central Great Plains Research Station, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Akron, Colorado, U.S.A.

Abstract

The least limiting water range (LLWR) has been developed as an index of the soil structural quality. The LLWR was defined as the region bounded by the upper and lower soil water content over which water, oxygen, and mechanical resistance become major limitations for root growth. Thus, it combines the effects of critical soil physical properties on biological processes into a single variable. Studies showed LLWR was related to shoot and roots growth and crop production. It also was effective to identify loss of soil quality caused by compaction due to animal trampling, tillage practices, and wheel traffic. The LLWR associated with measurements of microbiological processes rates may help predict the efficiency of different management systems to recover the quality of degraded soils.

INTRODUCTION

Soil quality evaluation should underline the importance of both intrinsic and dynamic soil properties and processes that have the potential to limit plant growth and development. The rates of many biological processes in plants vary with soil water content in a curvilinear manner, with slower rates at both low and high water contents. At low water content, the rates of these processes may be limited by the availability of water, soil strength, and accessibility of substrates or nutrients. At high water contents, the rates are generally limited by inadequate aeration. Thus, two important indicators of soil quality are the soil's ability to store water between rainfall and/or irrigation events and soil strength. The latter is essential for anchoring plants but can limit root growth if soil

strength becomes too high due to compaction caused by wheel traffic or natural reconsolidation.

To help quantify improvement or degradation of the soil physical properties, an index of the soil structural quality known as the least limiting water range (LLWR) has been developed. This index is soil specific and varies with texture or organic matter. It also varies with soil horizons as well as profile depth but collectively help describe effects of soil water content on many biological processes in soils. In addition, it has been recognized that LLWR combines the effects of critical soil physical properties on biological processes into a single variable.

One way to characterize the shape of a LLWR curve is by the horizontal spread between the upper and lower limits, i.e., the range in water contents at which the rate of a specific process is proportional to the maximum rate.

Under similar environmental conditions (e.g., temperature or rainfall/irrigation regimes), LLWR curve shape reflects the impact of soil structure on different biological properties and processes. Initially, the focus was on a non-limiting water range for plant growth. The LLWR concept was extended to evaluate this water content range as an index of structural soil quality. The concept of limiting water ranges for microbiological processes was also explored. Although the criteria for predicting upper and lower limits of LLWR for processes such as net nitrogen (N) mineralization or carbon dioxide (CO₂) emissions are not as well defined as those for plant growth, experimental measurement of the limits is more easily accomplished. The LLWR was combined with the SoilFlex model. This new model allows for predictions of changes in LLWR due to compaction caused by agricultural machines, and thus in soil conditions for plants growth.

The LLWR was defined as the region bounded by the upper and lower soil water content over which water, oxygen, and mechanical resistance become major limitations for root growth. LLWR is delimited by the supply of oxygen to the roots at the wet end, and the combined effects of water supply to the roots and mechanical resistance to root growth at the dry end. As soils become more dense, the LLWR becomes narrower, with mechanical resistance more commonly becoming the limitation at the dry end and poor aeration at the wet end.

METHODOLOGY AND APPLICATIONS

To construct the LLWR index, soil water contents (θ) at the critical limits of matric potential, soil strength, and air-filled porosity must be known for the range of soil bulk densities (ρ_b) likely to occur under the field conditions being evaluated.^[1]

Critical limits for LLWR have been defined as: 1) matric potential at field capacity (at either -10 kPa or -33 kPa) and permanent wilting point (-1500 kPa water potential); 2) air-filled porosity of less than 10%; and 3) soil strength >2.0 MPa.^[1–3] Water contents are then determined at the critical limits. For each soil sample, values of water contents at field capacity (θ_{FC}) and permanent wilting point (θ_{WP}) are measured directly or obtained from the water release curve (WRC, θ vs. ψ). Water content corresponding to 10% air-filled porosity (θ_{AP}) is calculated from the total porosity values. Water content corresponding to soil strength >2.0 MPa (θ_{SS}) is calculated from the relationship between soil strength, as measured by cone index, ρ_b and θ , assuming root penetration becomes negligible after cone penetrometer resistance exceeds 2 MPa. This relationship is known as soil resistance curve (SRC). Values of θ_{FC} , θ_{WP} , and θ_{SS} may also be estimated from pedotransfer functions.^[2]

Undisturbed soil samples are used to determine the functional relationships. Therefore, one of the most crucial

steps is sample collection, because to be meaningful, the variation of soil structural conditions, expressed by the soil bulk density, must be quantified to accurately model WRC and SRC. More details about models and statistical procedures used to obtain the WRC and SRC have been published.^[1,2] Once the functional relationships have been defined, the LLWR computation is carried out for each undisturbed soil sample as illustrated in Fig. 1.

The range defined by water content at field capacity (θ_{FC}) in the upper limit (Fig. 1A) and water content at permanent wilting point (θ_{WP}) in the lower limit (Fig. 1C) characterizes the traditional concept of available water capacity (AWC), which is widely used in soil science, ecophysiology, and irrigation. The AWC concept focuses on the functional relationship water content–water potential (θ – ψ) as a limiting physical factor for plant growth but as the oxygen diffusion rate in soil decreases, root growth can decrease even with elevated values of θ in the soil. In that situation, further soil drying is necessary to reach an appropriate aeration level. Similarly, in dry soils, soil strength may reach a limiting value before θ reaches the permanent wilting point.

Estimates of soil water content at upper critical aeration levels (θ_{AP}) are obtained from the relationship air-filled porosity–water content (AP– θ ; Fig. 1B). A simple approach involves the subtraction of a critical value of air-filled porosity [usually $0.10 \text{ cm}^3/\text{cm}^3$ (i.e., 10%)] from total soil porosity.

In soils with degraded structure, the wet end of the LLWR is commonly determined by θ_{AP} instead of θ_{FC} . In these cases, at θ_{FC} , plant growth is theoretically conditioned by restricted O₂ diffusion in the soil. The air-filled porosity is strongly dependent on soil structural, being linear and negatively related with ρ_b . At the dry end, depending on soil structural conditions, θ_{WP} may be substituted by θ_{SS} , which is calculated from the SRC (Fig. 1D). Water contents at the critical limits for a soil sample with $\rho_b = 1.27 \text{ g cm}^{-3}$ are shown in Fig. 1E.

Fig. 2, which illustrates the LLWR delimitation for a set of soil samples, shows that θ_{AP} substitutes θ_{FC} at $\rho_b = 1.35 \text{ g cm}^{-3}$, while θ_{SS} substitutes θ_{WP} starting from values $\rho_b = 1.37 \text{ g cm}^{-3}$. For this condition, LLWR is strongly reduced until reaching a value of 0. The maximum LLWR value coincides with the AWC range in this case.

The LLWR is zero when water content values at both the wet and dry end are numerically the same. The value of ρ_b at which LLWR = 0 is defined as the critical bulk density for that specific soil (ρ_{bc}).^[1] For the soil illustrated in Fig. 2, LLWR is zero when $\rho_{bc} = 1.55 \text{ g cm}^{-3}$. At this ρ_b value, plant growth is severely restricted by inadequate aeration and excessive soil strength. It has been demonstrated that corn shoot growth rate was correlated with the LLWR^[4] and that LLWR is effective for quantifying effects of soil property changes on plant root growth.^[3,5] This finding was supported by other studies that showed LLWR was related to crop production.^[6–8] A decrease in LLWR means that

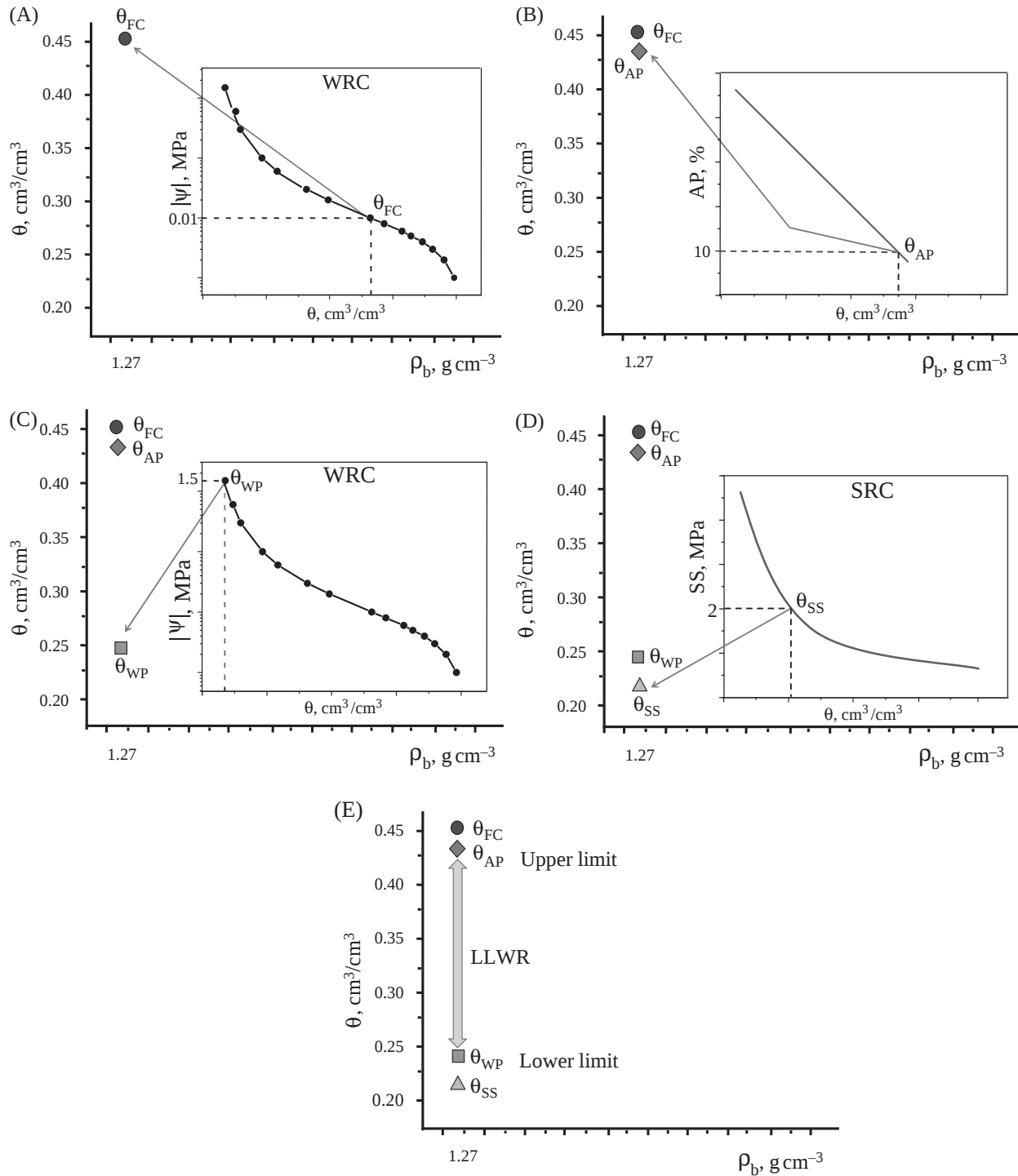


Fig. 1 Illustration of the critical values determination of water content at field capacity (θ_{FC} ; A), air-filled porosity (θ_{AP} ; B), wilting point (θ_{WP} ; C), and soil strength (θ_{SS} ; D), which define the upper and lower limits of the LLWR (E) for one soil sample (e.g., with $\rho_b = 1.27$ g cm⁻³).

plants are increasingly susceptible to water-related environmental stresses and that productivity might be reduced. As LLWR becomes narrower, the frequency that soil structure may contribute to limit moisture conditions becomes wider (i.e., the soil water content will more likely fall outside the LLWR affecting crop production).^[9]

Fig. 3 shows the variation in the LLWR with ρ_b (LLWR curve) for the same set of soil samples used to illustrate the LLWR delimitation in Fig. 2. The LLWR at each ρ_b is the difference between the upper and lower water content. The former is the drier θ of either θ_{FC} or θ_{AP} , and the latter is the wetter θ of either θ_{WP} or θ_{SS} .

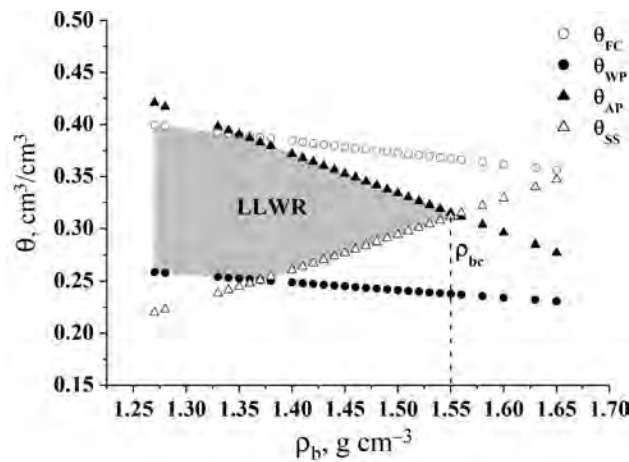


Fig. 2 Water content (θ) variation with soil bulk density (ρ_b) at critical levels of field capacity (θ_{FC}), wilting point (θ_{WP}), air-filled porosity (θ_{AP}), and soil strength (θ_{SS}). Gray area represents the LLWR. ρ_{bc} is the critical soil bulk density.

Factors Affecting the LLWR and Management Strategies to Enhance the LLWR

The LLWR is mainly affected by texture, organic carbon content, and bulk density (probably the most important for a given soil). This would include increases in ρ_b caused by wheel traffic or animal trampling compaction as well as that arising from natural consolidation during drying of soils that are structurally unstable because of low organic matter and/or the type of clay minerals.^[10–12] Fig. 4 illustrates changes in the LLWR curves due to soil degradation caused by combination of loss of soil organic matter and increased soil bulk density due to tillage compared with native soil.^[13]

Management practices to enhance the LLWR would include: 1) tillage designed to reintroduce macro- and mesopores in compacted soil; 2) controlling wheel traffic (traffic lanes) thereby restricting subsequent compaction to

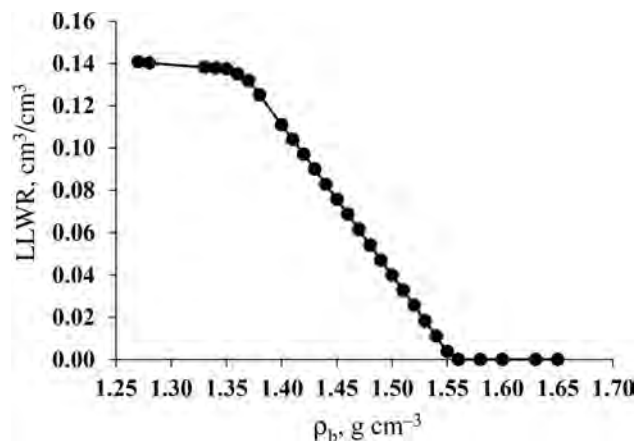


Fig. 3 Variation in the LLWR with soil bulk density (ρ_b).

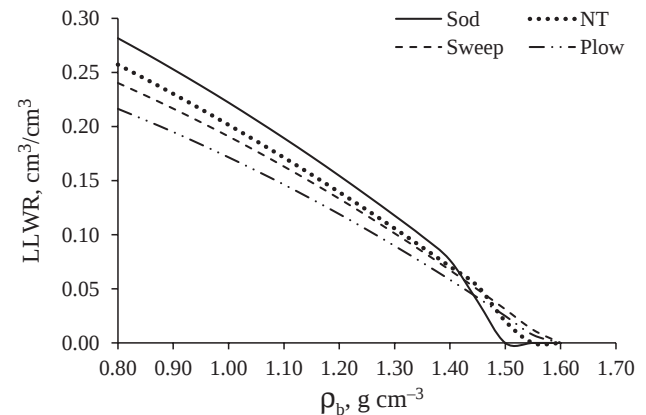


Fig. 4 Variation of the LLWR with soil bulk density (ρ_b) for different soil management systems.

Source: Adapted from Benjamin & Karlen.^[13]

localized zones in field; 3) reducing axle load and using larger tires for decreasing compaction; 4) avoiding animal trampling on wet soils; and 5) making soil less susceptible to compaction and natural consolidation by increasing organic matter content. One strategy to accomplish the latter would be to increase the amount of transient or temporary organic cementing materials that are effective in increasing wet aggregate stability and resistance to compaction. This could be done for some soils by incorporating crops with fine roots and fungal hyphae or cover crops into the rotation, animal or green manure, or applying urban/industrial wastes with high carbon but low metal or other phytotoxic content.

Opportunities and Limitations of the LLWR

Some merits of using LLWR to quantify soil physical quality are:

- It integrates four static soil characteristics (aeration, field capacity, soil mechanical resistance, and permanent wilting point) into a single variable.
- It describes the integrated effects of those characteristics in a way that is readily understood by farmers and advisory personnel and provides a semiquantitative measure of soil and crop management effects on soil structure.
- The LLWR can be combined with temporal variation in soil water content to indicate the frequency that a structure-induced stress may be encountered by a growing crop in soil with a given LLWR under the prevailing water budget conditions (this is crucial because the impact of management-induced structural change on crops varies with climate). This information can also be used to help those using irrigation determine better target water content levels to reduce total water use and optimize irrigation scheduling.

- The LLWR may be used to differentiate soils on the basis of the sensitivity of N mineralization to variation in water content.^[14]
- The LLWR associated with measurements of C–CO₂ emissions from the soils may help predict the efficiency of management systems to sequester carbon into the soils.^[15]
- The LLWR combined with compaction models may estimate the impact of soil loading and soil physical conditions for root growth, such a case of the SoilFlex-LLWR model.^[16]

Research needed to address limitations and unexplored aspects of the LLWR approach should include:

- Dynamic soil characteristics, such as parameters describing transport of water or oxygen.
- The role of limiting soil resistance levels within the soil matrix and their impact on calculated LLWR values.
- Assessments of the relevance of LLWR in compacted soils with abundance of macropores. This may be important because root growth occurs mainly in those pores rather than in the soil matrix.
- Determining more practical methods for quantifying limiting values since this is very time consuming.
- Improving our understanding of how soil characteristics such as soil resistance and aeration affect LLWR.
- Defining how rates of physiological processes that have the greatest influence on yield are related to the extent of the LLWR values.
- Quantifying susceptibility of different crops to changes in the soil environment. Finally, although interactions between limiting characteristics (e.g., effect of reduced aeration on limiting values of soil resistance) could be accounted for within the LLWR concept, these interactions are not generally considered.

CONCLUSION

The LLWR is effective in identifying soil quality constraints in plant growth because of different tillage, traffic, animal trampling, and cropping systems. The LLWR is also effective in identifying similar constraints on microbial processes such as the net mineralization of N or the CO₂ emissions. Knowledge of the LLWR for a soil can help the farmers improve the growing conditions by scheduling irrigation as well as implementing remedial management practices.

ACKNOWLEDGMENT

In Memoriam of Dr. B.D. Kay.

REFERENCES

1. da Silva, A.P.; Kay, B.D.; Perfect, E. Characterization of the least limiting water range of soils. *Soil Sci. Soc. Am. J.* **1994**, *58*, 1775–1781.
2. da Silva, A.P.; Kay, B.D. Estimating the least limiting water range of soil from properties and management. *Soil Sci. Soc. Am. J.* **1997**, *61*, 877–883.
3. Benjamin, J.G.; Nielsen, D.C.; Vigil, M.F.; Mikha, M.M.; Calderon, F.J. A comparison of two models to evaluate soil physical property effects on corn root growth. *Agron. J.* **2013**, *105*, 713–720.
4. da Silva, A.P.; Kay, B.D. The sensitivity of shoot growth of corn to the least limiting water range of soils. *Plant Soil* **1996**, *184*, 323–329.
5. Siegel-Issem, C.M.; Burger, J.A.; Powers, R.F.; Ponder, F.; Patterson, S.C. Seedling root growth as a function of soil density and water content. *Soil Sci. Soc. Am. J.* **2005**, *69*, 215–226.
6. Sharma, P.K.; Bhushan, L. Physical characterization of a soil amended with organic residues in a rice-wheat cropping system using a single value soil physical index. *Soil Till. Res.* **2001**, *60*, 143–152.
7. Benjamin, J.G.; Nielsen, D.C.; Vigil, M.F. Quantifying effects of soil conditions on plant growth and crop production. *Geoderma* **2003**, *116*, 137–148.
8. Lapen, D.R.; Topp, G.C.; Gregorich, E.G.; Curnoe, W.E. Least limiting water range indicators of soil quality and corn production, eastern Ontario, Canada. *Soil Till. Res.* **2004**, *78*, 151–170.
9. da Silva, A.P.; Kay, B.D. Effect of soil water content variation on the least limiting water range. *Soil Sci. Soc. Am. J.* **1997**, *61*, 884–888.
10. Betz, C.L.; Allmaras, R.R.; Copeland, S.M.; Randall, G.W. Least limiting water range: Traffic and long-term tillage influences in a Webster soil. *Soil Sci. Soc. Am. J.* **1998**, *62*, 1384–1393.
11. Tormena, C.A.; da Silva, A.P.; Libardi, P.L. Soil physical quality of a Brazilian Oxisol under two tillage systems using the least limiting water range approach. *Soil Till. Res.* **1999**, *52*, 223–232.
12. Chen, G.; Weil, R.R.; Hill, R.L. Effects of compaction and cover crops on soil least limiting water range and air permeability. *Soil Till. Res.* **2014**, *136*, 61–69.
13. Benjamin, J.G.; Karlen, D.L. LLWR techniques for quantifying potential soil compaction consequences of crop residue removal. *Bioenerg. Res.* **2014**, *7*, 468–480.
14. Drury, C.F.; Zhang, T.Q.; Kay, B.D. The non-limiting and least limiting water ranges for soil nitrogen mineralization. *Soil Sci. Soc. Am. J.* **2003**, *67*, 1388–1404.
15. Medeiros, J.C.; da Silva, A.P.; Cerri, C.E.P.; Giarola, N.F.G.; Figueiredo, G.C.; Fracetto, F.J.C. Linking physical quality and CO₂ emissions under long-term no-till and conventional-till in a subtropical soil in Brazil. *Plant Soil* **2011**, *338*, 5–15.
16. Keller, T.; da Silva, A.P.; Tormena, C.A.; Giarola, N.F.B.; Cavalieri, K.M.V.; Stettler, M.; Arvidsson, J. SoilFlex-LLWR: Linking a soil compaction model with the least limiting water range concept. *Soil Use Manage.* **2015**, *31*, 321–329.

Water Retention

Satish C. Gupta

Dong Wang

Department of Soil, Water, and Climate, University of Minnesota, St. Paul, Minnesota, U.S.A.

Abstract

Soil water retention refers to the mechanism and processes related to changes in soil water content and its energy status. The relationship between these two quantities is called the soil water retention characteristics curve. This entry outlines the methods of measuring soil water content and its energy levels, causes of potential energy in soil water and how it varies with soil texture, mathematical models for describing soil water retention characteristic curves, prediction of soil water retention from easily measurable soil properties, hysteresis in soil water retention, and the definition of plant-available water in soils.

INTRODUCTION

Soil water retention refers to the mechanisms and processes related to changes in soil water content vs. its energy status. There are two components in the soil water retention as follows: 1) the amount of water held in soil; and 2) the potential energy with which the water is held. The relationship between the amount of water and its potential energy is called the soil water retention characteristic curve (Fig. 1). The water retention characteristic curves are different for different soil types (Fig. 2). These differences are caused by the variation in soil particle-size distribution and their packing arrangement. Both these factors influence the soil water retention by affecting the pore-size distribution and the number of a given pore size in each size class.

BACKGROUND

The maximum amount of water held in a soil depends on the surface area of soil particles and the fraction of pores in a given volume of soil (porosity). The potential energy of the soil water, on the other hand, depends on the size distribution of the junction between pores. The junction between pores can be viewed as a small capillary tube, the radius of which determines the force needed to drain the pore (Fig. 3).

The energy with which the water is held in a capillary tube can be described by the following capillary rise equation (Fig. 4):

$$h = \frac{2\sigma}{\rho g r} \cos \theta \quad (1)$$

where h is the height of rise of water in a capillary tube, σ is the surface tension force of water, ρ is the density of

water, g is the acceleration due to gravity, r is the radius of the capillary tube, and θ is the contact angle of water with the capillary tube. The smaller the radius of a capillary tube, the greater is the rise in water height. Applying this principle to soil water retention, the smaller the radius of the junction between two pore cavities, the greater is the energy needed to drain the pore restricted by the pore junction.

The energy with which water is held in soil is also called the soil matrix potential energy or simply the soil matrix potential. Because of the analogy between the capillary tube and the junction between soil pores, this energy is also called the capillary potential energy or simply the capillary potential. There are several ways to measure the potential energy of the soil. These include the tensiometer and the thermocouple psychrometer. The tensiometer consists of a small ceramic cup that is connected to a PVC tube on which a measuring device (vacuum gauge or mercury manometer) is mounted for recording the potential energy (Fig. 5).

In field applications of tensiometers, the ceramic cup and the PVC tube are first filled with de-aired water and sealed with a cap. The tensiometer is then installed in the soil and allowed to reach equilibrium with soil water. The negative pressure at equilibrium corresponds to the energy with which water is held in a soil.

The thermocouple psychrometer (Fig. 6) measures the relative humidity (RH) of the soil air that is in equilibrium with liquid soil water and its solutes. In the thermocouple psychrometer, water is first condensed at the thermocouple junction and then allowed to evaporate. The rate of evaporation is controlled by the RH of the chamber that is in equilibrium with the potential energy of the soil water.^[1]

The water potential (ψ_w) is related to RH by the following relationship:

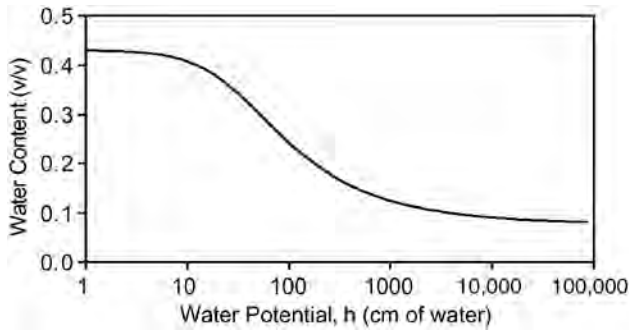


Fig. 1 The schematic representation of soil water retention characteristic curve.

$$\psi_w = \frac{RT}{M} \ln(RH) \quad (2)$$

where R is the ideal gas constant, T is the absolute temperature, and M is the molecular weight of soil solution (water and solutes). Tensiometers are useful for measuring soil matric potential from 0 to -80 kPa, whereas thermocouple psychrometers are effective for measuring soil water potential from -1500 kPa to air-dry soil conditions.

The quantity of water held in the soil is expressed as the mass of water per unit mass of soil or the volume of water per unit volume of soil. The standard method of measuring soil water is by the gravimetric approach. In this case, the wet soil sample is weighed both before and after oven drying at 105°C for 24 hours or to a constant weight. The loss of water per unit weight of soil or per unit volume of soil thus quantifies the amount of water in the soil. The other methods include the neutron probe, the gamma ray attenuation, the time domain reflectometry (TDR), and the heat dissipation unit. The neutron probe is based on the principle that a fast neutron, when released in soil, will collide with hydrogen atoms, thus resulting in slowed neutrons. Because most of the hydrogen atoms in soil are from water, the number of slowed neutron thus determines the amount of water in soil.^[2] Gamma ray attenuation is based on the principle that an increase in soil water will decrease the number of gamma particles passing through the soil and

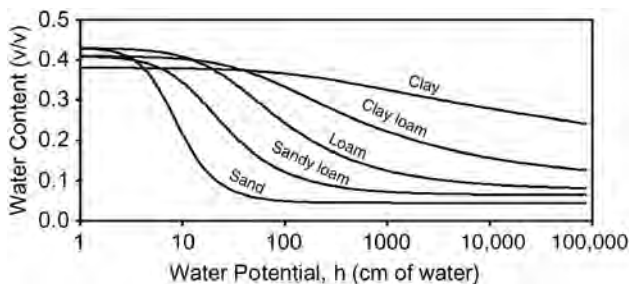


Fig. 2 The water retention of five different textured soils.
Source: From Carsel & Parrish.^[7]

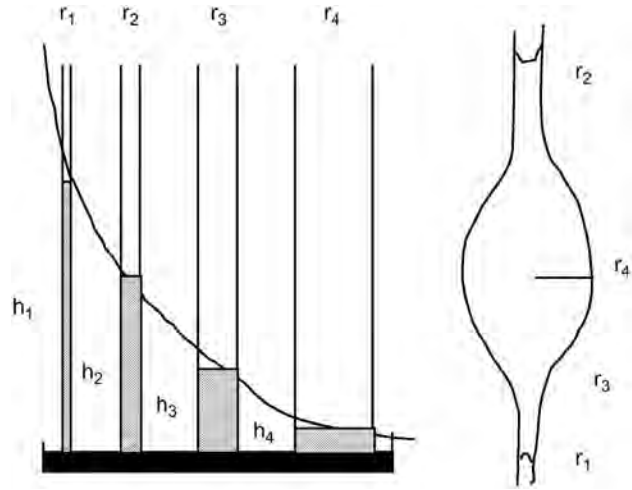


Fig. 3 Height in rise of water in capillary tubes corresponding to various neck radii in a pore.

vice versa.^[3] TDR measures the dielectric constant of the moist soil. Because the dielectric constant of liquid water is 80 and that of the soil matrix is 4, changes in the soil dielectric constant reflect the changes in soil water content.^[4] The heat dissipation method works on the principle that at higher soil water content there is faster dissipation of heat and thus lower rise in soil temperature.^[5]

MODELS OF SOIL WATER RETENTION

In this entry, there are several methods available for estimating the water retention characteristics of soils. These methods are generally based on the premise that there exists some relationship between the soil water retention characteristics and the soil particle-size distribution, soil packing density, and soil organic matter content. These are either empirical or semiphenomenological. The empirical relationships are also called the “pedotransfer” functions. Among the semiphenomenological methods, the most notable is the method of Arya and Paris.^[6] The basis of this method is that the shape of the pore-size distribution curve is similar to the particle-size distribution curve. Then, it is a matter of translating the

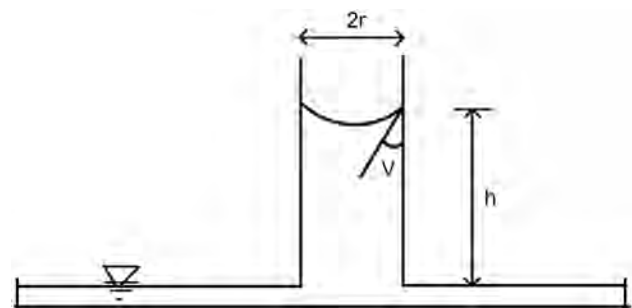


Fig. 4 Relationship between water rise, surface tension, and capillary radius.

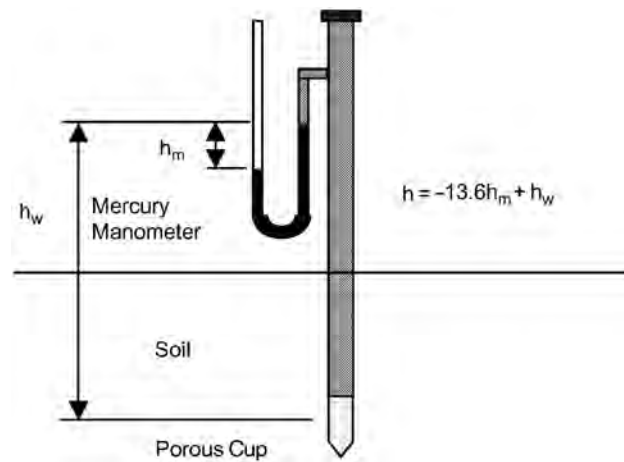


Fig. 5 The schematic representation of tensiometry for soil water potential measurement.

particles of a given size to the amount of water held by those particles and the particle size to a corresponding pore size:

$$V_i = \left(\frac{w_i}{\rho_s} \right) e \quad (3)$$

$$r_i = - \frac{\sigma \cos \theta \sqrt{6}}{\rho_g R_i \sqrt{e n_i^{(1-\alpha)}}} \quad (4)$$

where V_i and w_i are the pore volume and weight of i^{th} fraction, respectively; ρ_s is the particle density; e is the void ratio; r_i and R_i are the mean pore and particle radii of the i^{th} fraction, respectively; n_i is the number of particles in i^{th} fraction, and α is an empirical constant ranging in value from 1.35 to 1.40.

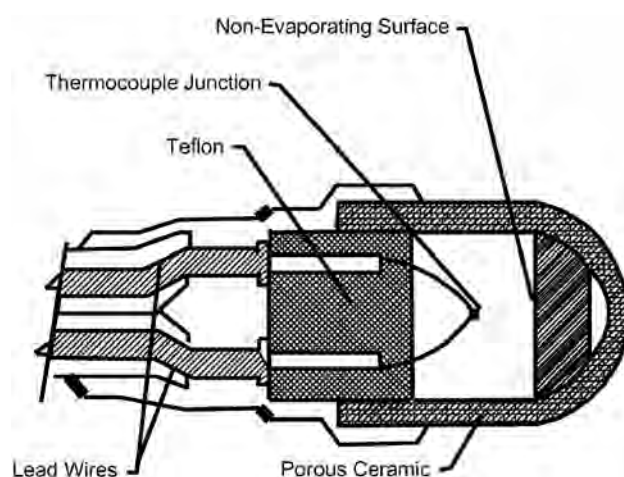


Fig. 6 The schematic representation of thermocouple psychrometer.

Source: From Hanks & Ashcroft.^[8]

Pedotransfer Functions

The pedotransfer functions, or the empirical methods of predicting soil water retention characteristics, use the easily measurable soil properties such as particle-size distribution (sand, silt, and clay contents), bulk density, organic matter content, etc. as surrogates of water retention in soil.^[9] Using the measured database, an empirical equation that predicts the water retention at each soil matric potential is developed. Among the most widely used empirical methods of predicting water retention characteristic is the method proposed by Rawls and Brakensiek.^[10] This empirical method is based on water retention measurements of over 2000 soil horizons. In this method, the empirical relationships predict soil water content at 12 matric potentials (Table 1). The relationships are provided for three levels of input parameters. The first level assumes that the only inputs available are the percent sand, percent silt, percent clay, bulk density, and the organic matter content. The second level of relationships assumes that in addition to the input parameters for the first level of regression equation, the water content of the soil at -1500 kPa matric potential is also known. The third level of regression further assumes that the soil water content at -33 kPa soil matrix potential is also known. Over the years, the method has been modified and improved by the inclusions of additional data sets and/or other input variables.^[11–13]

Mathematical Description

Various mathematical functions have been used to describe the complete soil water retention characteristic curve. Notable among those are the functions suggested by Brooks and Corey^[14] and van Genuchten.^[15] The Brooks and Corey function is:

$$\Theta = \frac{\theta - \theta_r}{\theta_s - \theta_r} = \left(\frac{h_e}{h} \right)^{-b} \quad (5)$$

whereas the van Genuchten function is:

$$\Theta = [1 + \alpha(-h)^n]^{-m} \quad (6)$$

$$m = (1 - n^{-1}) \quad (7)$$

where θ is the normalized water content; θ_r is the residual water content; θ_s is the saturated water content; h is the apparent soil matric potential; h_e is the air-entry matric potential; and α , b , n , and m are constants for a given soil. Because of their mathematical superiority, these two functions have been widely used in many soils and hydrologic modeling studies. Pedotransfer functions have also been developed for predicting parameters of the Brooks and Corey and van Genuchten functions based on easily measurable soil properties such as percent sand, silt, clay, organic matter, and bulk density.^[11–13]

Table 1 Coefficient of linear regressions for predicting soil water contents at 12 matric potentials.

Matric potential (kPa)	Regression coefficients								Correlation coefficient (R)
	A	b	c	d	e	f	g	h	
-4	0.7899	-0.0037			0.0100	-0.1315			0.58
	0.6275	-0.0041			0.0239			-0.08	0.57
	0.1829				-0.0246	-0.0376	1.89	-1.38	0.77
-7	0.7135	-0.0030		0.0017		-0.1693			0.74
	0.4829	-0.0035			0.0263			0.25	0.74
	0.0888	-0.0003			-0.0107	1.53	-0.81		0.91
-10	0.4118	-0.0030		0.0023	0.0317				0.81
	0.4103	-0.0031			0.0260			0.41	0.81
	0.0619	-0.0002			-0.0067		1.34	-0.51	0.95
-20	0.3121	-0.0024		0.0032	0.0314				0.86
	0.3000	0.0024			0.0235			0.61	0.89
	0.0319	-0.0002					1.01	-0.06	0.99
-33	0.2576	-0.0020		0.0036	0.0299				0.87
	0.2391	-0.0019			0.0210			0.72	0.92
	0.2065	-0.0016		0.0040	0.0275				0.87
-60	0.1814	-0.0015			0.0178			0.80	0.94
	0.0136					-0.0091	0.66	0.39	0.99
	0.0349		0.0014	0.0055	0.0251				0.87
-100	0.1417	-0.0012			0.0151			0.85	0.96
	-0.0034				0.0022		0.52	0.54	0.99
	0.0281		0.0011	0.0054	0.0200				0.86
-200	0.0986	-0.0009			0.0116			0.9	0.97
	-0.0043				0.0026		0.36	0.69	0.99
	0.0238		0.0008	0.0052	0.0190				0.84
-400	0.0649	-0.0006			0.0085			0.93	0.98
	-0.0038				0.0026		0.24	0.79	0.99
	0.0216		0.0006	0.0050	0.0167				0.81
-700	0.0429	-0.0004			0.0062			0.94	0.98
	-0.0027				0.0024		0.16	0.86	0.99
	0.0205		0.0005	0.0049	0.0154				0.81
-1000	0.0309	-0.0003			0.0049			0.95	0.99
	-0.0019				0.0022		0.11	0.89	0.99
-1500	0.0260			0.0050	0.0158				0.80

Note: $\theta_x = a + b [\text{sand (\%)}] + c [\text{silt (\%)}] + d [\text{clay (\%)}] + e [\text{organic matter (\%)}] + f [\text{bulk density (Mg/m}^3\text{)}] + g [(\text{water content at } -33 \text{ kPa})] + h [(\text{water content at } -1500 \text{ kPa})]$ where θ_x is predicted water content (m^3/m^3) at a given potential, sand=0.5–2 mm, silt=0.02–0.05, clay ≤ 0.002 mm. Sand + silt + clay = 100%.

Source: From Rawls & Brakensiek.^[10]

METHODS OF CHARACTERIZING SOIL WATER RETENTION

The measurement of soil water retention requires simultaneous determination of soil water content and potential. In the laboratory, the traditional methods of measuring soil water retention involve establishing a series of equilibrium between water in the soil sample and known potentials.^[16] Two of the most common methods that use

this principle are the hanging water column and the pressure plate apparatus. In case of hanging water column, an initially saturated soil sample placed on a ceramic plate is allowed to drain by gravity until there is no more drainage. The soil water content at equilibrium thus corresponds to the gravitational forces equivalent to the distance between the soil sample and the water reservoir. The pressure plate apparatus consists of a ceramic plate and a pressure chamber. The soil sample is placed on a

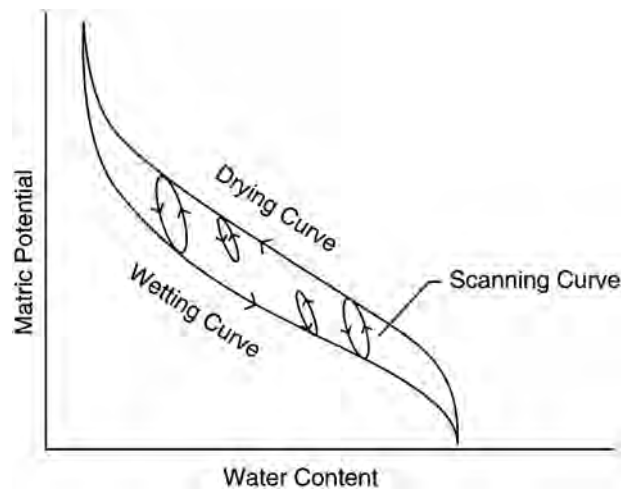


Fig. 7 Hysteresis in soil water retention.
Source: From Hillel.^[19]

ceramic plate; the plate and the soil sample are allowed to saturate overnight and then are desorbed by applying a known air pressure to the chamber. When no more water comes out of the soil sample, the sample is in equilibrium with the applied air pressure, and thus, the energy of the soil water is equivalent to the applied air pressure. The soil sample is taken out and its water content measured gravimetrically.

Soil water retention characteristics can also be characterized *in situ* using tensiometers and various water content measuring devices. A widely used approach is the instantaneous profile method where a flooded area is allowed to drain, and then the soil water content and matric potential are measured simultaneously using a neutron probe and tensiometers.^[17] Wang, Yates, and Ernst^[18] used a combination of small tensiometers and TDR probes to measure *in situ* water retention. In this setup, the authors used a tension infiltrometer as a source of water supply.

HYSTERESIS

For a given soil, the soil water retention characteristic curve varies depending on whether the curve was obtained by desorbing a saturated soil or by wetting a dry soil (Fig. 7).

This behavior in water retention characteristics is called the “hysteresis.” The main reason for the hysteresis is the so-called “ink-bottle” effect, which refers to narrow points of connection between large cavities of the adjoining pores (Fig. 8).

During drying, the water content in the soil is controlled by the narrow points between pores (Fig. 8A), whereas during wetting water retention is controlled by the widest point in the pore cavity (Fig. 8B). As a result of this, the soil at any given potential retains more water

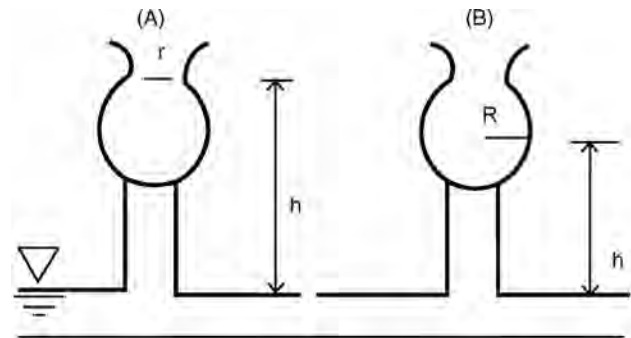


Fig. 8 “Ink-bottle” effect on water imbibition (A) and drainage (B) in large soil pores.
Source: From Hillel.^[19]

while drying than wetting. The wetting and drying curves in Fig. 7 are called the primary curves, whereas the curves connecting the primary curves are called the scanning curves. Scanning curves define the path that a soil takes when switching from wetting to drying and vice versa.

PLANT-AVAILABLE WATER

From the plant growth perspective, two points on the soil water retention curve are significantly important. These are the water retention at soil matric potential of -33 kPa (field capacity water content) and the water retention at soil matric potential of -1500 kPa (permanent wilting point or PWP). The field capacity water content refers to soil water content after a saturated soil has been allowed to drain for 24–48 hours without any significant evaporation. Field measurements have shown that for most soils, the matric potential at field capacity water content corresponds to -33 kPa. If the water retention was measured on a disturbed soil, then the field capacity water content corresponds to a soil matric potential of -10 kPa. PWP refers to soil water content when plants fail to recover its turgidity on watering. Field measurements have shown that this water content corresponds to about -1500 kPa for most plants. Water retention between soil matric potentials of -33 and -1500 kPa is referred to as the available water for plant growth.

As expected, available water capacity of soil varies with soil texture (Table 2). Generally, medium texture soils have higher available water capacity than either the fine-textured or the coarse-textured soils. This is due to both relatively higher field capacity and lower PWP water content of medium texture soils than the coarse-textured or the fine-textured soils. In coarse-textured soil, water content at PWP is low but so is the water content at field capacity. In fine-textured soils, water content at field capacity is higher but so is the water content at PWP.

Table 2 Available water capacity of soils of various textural groups.

Soil texture	Available water (m ³ /m ³)
Clay	0.076
Clay loam	0.121
Loam	0.077
Loamy sand	0.003
Silt	0.168
Sandy loam	0.136
Silty clay	0.071
Silty clay loam	0.142
Sand	0.001
Sandy clay	0.097
Sandy clay loam	0.058
Sandy loam	0.019

Source: From Carsel & Parrish.^[7]

CONCLUSION

Soil water retention refers to the mechanism and processes related to changes in soil water content and its energy status. The relationship between these two quantities is called the soil water retention characteristics curve. This entry outlines the methods of measuring soil water content and its energy levels, causes of potential energy in soil water and how it varies with soil texture, mathematical models for describing soil water retention characteristic curves, the prediction of soil water retention from easily measurable soil properties, hysteresis in soil water retention, and the definition of plant-available water in soils.

REFERENCES

- Rawlins, S.L.; Campbell, G.S. Water potential: Thermocouple psychrometry. In *Methods of Soil Analysis, Part 1, Physical and Mineralogical Methods*; Klute, A., Ed.; 2nd Ed.; ASA-SSSA: Madison, 1986; 597–618.
- Gardner, W.R.; Kirkham, D. Determination of soil moisture by neutron scattering. *Soil Sci.* **1952**, *73*, 392–401.
- Gurr, C.G. Use of gamma rays in measuring water content and permeability in unsaturated columns of soil. *Soil Sci.* **1962**, *94*, 224–229.
- Topp, G.C.; Davis, J.L.; Annan, A.P. Electromagnetic determination of soil water content: Measurement in coaxial transmission lines. *Water Resour. Res.* **1980**, *16*, 574–582.
- Phene, C.J.; Hoffman, G.J.; Rawlins, S.L. Measuring soil matric potential *in situ* by sensing heat dissipation within a porous body. I. Theory and sensor construction. *Soil Sci. Soc. Am. Proc.* **1971**, *35*, 27–32.
- Arya, L.R.; Paris, J.F. A physicoempirical model to predict the soil moisture characteristics from particle size distribution and bulk density data. *Soil Sci. Soc. Am. J.* **1986**, *45*, 1023–1030.
- Carsel, R.F.; Parrish, R.S. Developing joint probability distributions of soil water retention characteristics. *Water Resour. Res.* **1988**, *24*, 755–769.
- Hanks, R.J.; Ashcroft, G.L. *Applied Soil Physics*; Springer: Berlin, 1980.
- Gupta, S.C.; Larson, W.E. Estimating soil water retention characteristics from particle size distribution, organic matter, and bulk density. *Water Resour. Res.* **1979**, *15*, 1633–1635.
- Rawls, W.J.; Brakensiek, D.L. Estimating soil water retention from soil properties. *J. Irrig. Drain. Div. Proc. ASCE* **1982**, *108* (IR2), 167–171.
- Rawls, W.J.; Brakensiek, D.L. Prediction of soil water properties for hydrologic modeling. In *Proceedings of the Symposium on Watershed Management in the Eighties*, Denver, CO, Apr 30–May 1, 1985; Jones, E.B., Ward, T.J., Eds.; Denver, CO.; Am. Soc. Civil Eng.: New York, 1985; 293–299.
- Rawls, W.J.; Ahuja, L.R.; Brakensiek, D.L. Estimating soil hydraulic properties from soil data. In *Indirect Methods for Estimating the Hydraulic Properties of Unsaturated Soils*, Proceedings of the International Workshop, Oct 11–13, 1989; van Genuchten, M.Th., Leij, F.J., Lun, S., Eds.; University of California: Riverside, 1992.
- Rawls, W.J.; Gish, T.J.; Brakensiek, D.L. Estimating soil water retention from soil physical properties and characteristics. *Adv. Soil Sci.* **1991**, *16*, 213–234.
- Brooks, R.H.; Corey, A.T. Properties of porous media affecting fluid flow. *J. Irrig. Drain. ASCE* **1966**, *92*, 61–88.
- van Genuchten, M.Th. A closed form equation for predicting the hydraulic conductivity of unsaturated soils. *Soil Sci. Soc. Am. J.* **1980**, *44*, 892–898.
- Klute, A. Water retention: Laboratory methods. In *Methods of Soil Analysis, Part 1, Physical and Mineralogical Methods*; Klute, A., Ed.; 2nd Ed.; ASA-SSSA: Madison, 1986; 635–662.
- Watson, K.K. An instantaneous profile method for determining the hydraulic conductivity of unsaturated porous materials. *Water Resour. Res.* **1966**, *2*, 709–715.
- Wang, D.; Yates, S.R.; Ernst, F.F. Determining soil hydraulic properties using tension infiltrometers, TDR, and tensiometers. *Soil Sci. Soc. Am. J.* **1998**, *62*, 318–325.
- Hillel, D. *Fundamental of Soil Physics*; Academic Press: New York, 1980.

Water: Plant-Available

Argyrios Gerakis

Laboratory of Agronomy, Faculty of Agriculture, Aristotle University of Thessaloniki, Thessaloniki, Greece

Joe T. Ritchie

Department of Crop and Soil Sciences, Michigan State University, East Lansing, Michigan, U.S.A.

Abstract

The definitions of the upper and lower limits of plant available water are difficult to establish because of water flow into and out of the root zone and because of incomplete extraction by sparse roots at the lower boundaries of the root zone. If the limits of plant available water cannot be measured, they can be estimated based on routinely measured soil properties. This entry presents a simple model that uses few parameters. The model can make it easier to approximate soil water limits in water balance simulation, irrigation scheduling, and watershed management.

INTRODUCTION

Accurately evaluating the available soil water reservoir is vital to developing optimum management for rain-fed crop production in marginally dry regions. Often there is a linear relationship between plant available water and yield^[1] and between plant available water and leaf growth, within limits.^[2] Phenology can be delayed due to preanthesis drought stress.^[2]

Two forces retain water in the soil: adhesion and cohesion. Adhesion is the attraction of water molecules to soil particles. Cohesion is the attraction of water molecules to each other. These two forces oppose the effect of gravity that removes soil water by drainage. The maximum storage of useful water to the plants depends on the equilibrium between soil water retention forces and gravity. Mostly, the minimum storage depends on the retention forces. There is some water that is held so tightly in the soil that the plants cannot extract (Fig. 1).

DEFINING THE LIMITS OF PLANT AVAILABLE WATER

We define potentially plant available water or potentially plant extractable water (θ_p) as the difference between the drained upper limit (θ_d) and the lower limit (θ_l). The drained upper limit is the highest, field-measured water content of a soil after thorough wetting and draining until drainage becomes practically negligible. The θ_d corresponds closely to water content at “field capacity” and to matric suction in the range of 10–33 kPa. The lower limit, θ_l , is the lowest, field-measured, volumetric water content

of a soil after plants stop extracting water due to premature death or dormancy as a result of water deficit. The lower limit corresponds closely to the water content at the “permanent wilting point” and to matric suction of 1.5 MPa. Potentially plant available water is similar across a wide range of soil textures (Fig. 2). The mean and standard deviation of field measured θ_p for a wide range of soils are shown in Table 1.

MEASURING THE LIMITS OF PLANT AVAILABLE WATER

The traditional approach to evaluating the limits of available water has been to measure the “permanent wilting point” and “field capacity” of soil samples removed from the field through use of pressure chambers. Usually, soil water contents are measured at suctions of 1.5 MPa for the wilting point and 10–33 kPa for field capacity. Samples are taken from the soil at various depths to the measured or estimated rooting depth. The difference between the upper and lower limits of availability is summed over the rooting depth to determine the total potentially plant available water.

Several criticisms have been made of the above rather static definitions of available soil water limits. Water above the upper limit can be taken up while drainage is occurring. Plant growth can be retarded before the lower limit is reached. Water extraction by roots may continue beyond the 1.5 MPa range in some instances. Where root density is low, especially in deep soils, it is difficult to separate the effects of root distribution from the effect of water flow through the soil on available water.

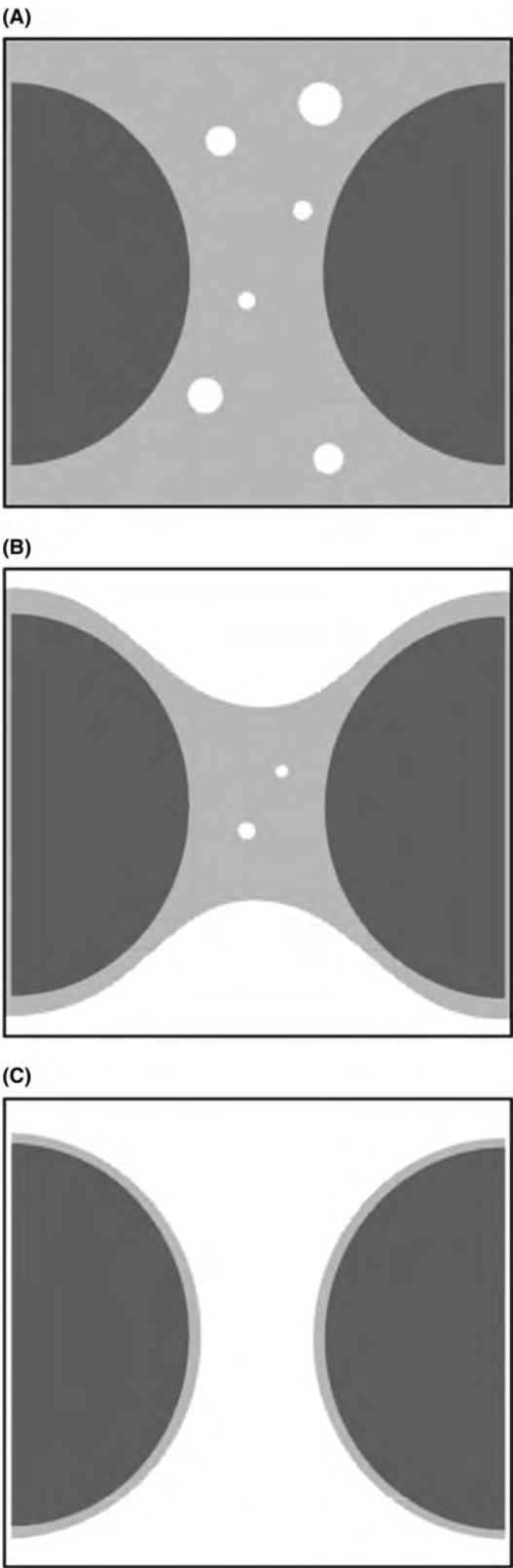


Fig. 1 Schematic of water in soil pores at three water contents: (A) saturation (entrapped air bubbles persist), (B) drained upper limit (larger pores empty, menisci form between soil particles), and (C) lower limit (thin films around soil particles).

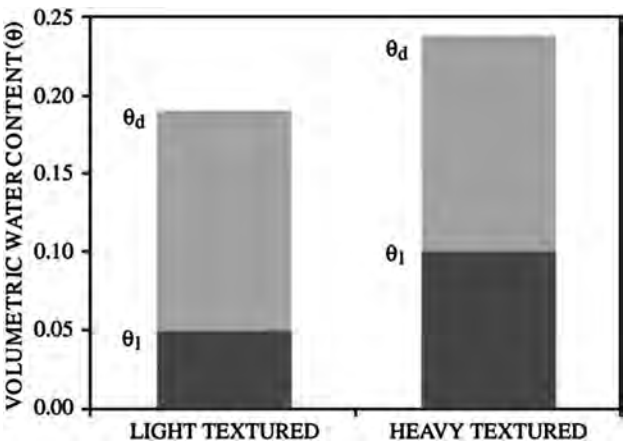


Fig. 2 Potentially extractable soil water $\theta_d - \theta_l$ is similar among soils with a wide variety in texture. Heavy textured soils have higher proportion of smaller pores, therefore both the upper (θ_d) and the lower limit (θ_l) are higher than in light textured soils.

FIELD MEASUREMENT OF THE UPPER LIMIT

The drained upper limit is a necessary input in water balance models. The “tipping bucket” approach in water balance models such as in DSSAT^[4] can be sensitive to the value chosen for θ_d .^[5] Because water can be taken up by plants while drainage is occurring, the drained upper limit is not always the appropriate upper limit of soil water availability. Therefore, it is necessary to measure or reasonably estimate the drainage rates while they are significant.

Table 1 Potentially plant available water for soils of different texture as measured in the field. The values range from a minimum of $8.0 \pm 3.1\%$ for sands to 14.8% for just one observation for the silt. The mean plant available water values for the remaining textural classes are within a narrow range of only $11.0\text{--}14.3\%$.

Texture	No. of samples	Mean and standard deviation for θ_p
		(volume percent)
Sand	76	8.0 ± 3.1
Loamy sand	7	12.9 ± 3.6
Sandy loam	31	13.2 ± 2.2
Loam	51	13.6 ± 3.0
Silt loam	83	14.3 ± 3.3
Silt	1	14.8
Silty clay loam	53	13.0 ± 2.1
Clay loam	41	12.5 ± 3.2
Sandy clay loam	24	11.0 ± 3.5
Sandy clay	0	
Silty clay	31	13.4 ± 3.0
Clay	3	12.9 ± 3.6

Source: From Ratliff, Ritchie, et al.^[3] ©1983.

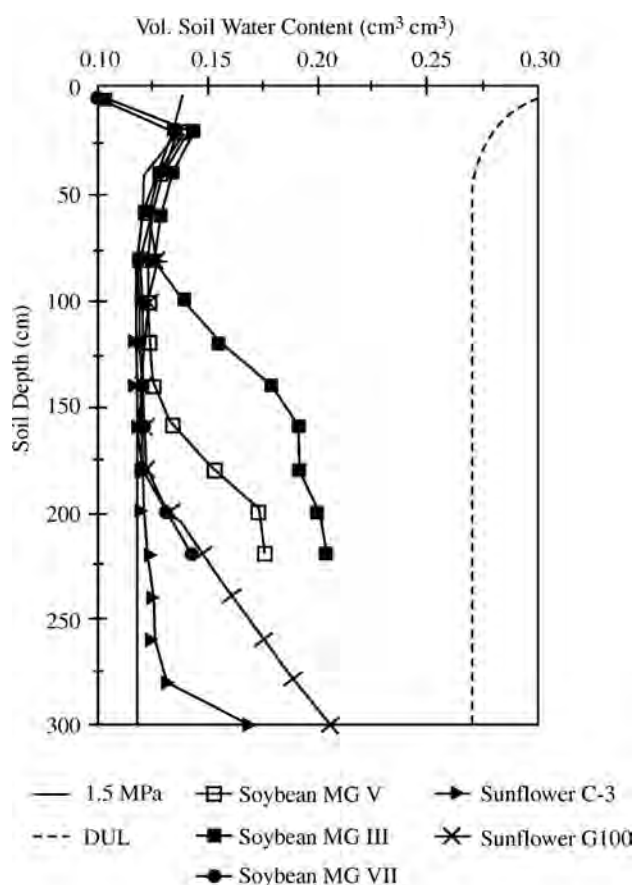


Fig. 3 Minimum soil water content reached by sunflower cultivar G 100, sunflower cultivar Contiflor 3, soybean cultivar Asgrow 3127, soybean cultivar Asgrow 5308, and soybean cultivar R.A. Seven hundred and two planted on a silt loam soil (Entic Haplustoll) in Cordoba, Argentina, during the 1992–1993 growing season. The dashed line indicates the field measured drained upper limit, and the solid heavy line indicates water content measured in the laboratory at 1.5 MPa suction.

Source: From Dardanelli, Bachmeier, et al.^[7]

There are functions to estimate drainage flux that are relatively simple to evaluate from field data.^[6]

FIELD MEASUREMENT OF THE LOWER LIMIT

Field measurements of the lower limit of water availability are more difficult than measuring the upper limit because root growth dynamics are different between and within crops and also because conditions for the lower limit may not often naturally occur.

The best value for the lower limit is obtained when crops reach their maximum vegetative size without stress and then grow on stored soil water until plants are visibly severely distressed. Because leaf senescence and natural death of many crops occur at maturity, it is important that plants have undergone premature severe stress. On the other hand, any stress that causes a reduction in root growth

before establishment of a “normal” rooting profile can cause incomplete root water extraction. Nutrient, drought, and aeration stresses, as well as chemical boundaries in soil, may restrict root growth and rooting depth.

Crop species and cultivars within a species may exhibit differing final water content values at crop maturity. Information on the lowest measured water content for several crops experiencing terminal drought is in Fig. 3. Results indicate that this soil is uniform with depth and the 1.5 MPa suction was quite close to the lowest measured water content when the crop had sufficient time and rooting density to remove the water down to the θ_l . The true θ_l was reached in the 20–80 cm depth for all crop situations but was not obtained below that depth for a short maturity soybean. In contrast, the long season sunflower Contiflor 3 extracted water to the θ_l to a depth of about 260 cm.

ESTIMATING THE LIMITS OF PLANT AVAILABLE WATER

When the cost and time of primary data collection is prohibitive, it is necessary to estimate reasonable values for the limits of the soil water reservoir. Many studies have found statistical relationships between plant available water and one or more soil properties such as texture, organic matter content, coarse fragment content, and bulk density.^[8–11] To achieve a good fit, statistical models can be very complex and may behave erratically. There are many soil properties that are related to the water limits. The more predictors there are in a model, the more specific the model becomes to a particular data set and its generality is being lost. Also, the additional data required to use a complex model are not always available.

To derive a simple model, we used a database of field measured soil water limits.^[3] The database contained 401 soil samples from 15 states of the United States. Seven soil orders were represented, but over 60% of the soils were Mollisols or Alfisols. Histosols, Oxisols, and Spodosols were not represented. There were few data from the midwest and southeast United States. The fact that these regions experience frequent precipitation during the crop growing season precludes the casual collection of field-measured lower limit.

We selected the gravimetric drained upper limit (w_d) as the first dependent variable to estimate, to make the model sensitive to bulk density, and to avoid the error associated with the measurement of bulk density. The model of w_d was

$$w_d = 0.186(\text{sand/clay})^{-0.141} \quad (1)$$

where w_d is the gravimetric water content and “sand” and “clay” are percent sand and clay. Then θ_d can be calculated from

$$\theta_d = w_d \rho_b / \rho_w \quad (2)$$

where θ_d is the volumetric water content, ρ_b is the soil bulk density, and ρ_w is the density of water.

The second dependent variable that we estimated was θ_p , the volumetric, potentially plant extractable water content. We picked sand content as the independent variable once we realized that θ_p is approximately the same across soils, except for soils with high sand content.^[3] The model was

$$\theta_p = 0.132 - 2.5 \times 10^{-6} e^{0.105 \text{ sand}} \quad (3)$$

where 0.132 is the mean θ_p for soils with less than 65% sand.

Then, we modified the estimates of θ_d and θ_l for coarse fragment content (particles larger than 2 mm in diameter) and for unusually high organic C content. The equations were tested with a limited independent data set with good results.^[12]

The θ_l can be calculated as the difference between θ_d and θ_p . The plant extractable water for the soil profile can be calculated as the product of θ_p times soil depth, in units of soil depth. If a profile consists of layers of different composition, then the plant extractable water of the profile is calculated as the sum of the extractable water of each layer. It is important to know the rooting depth and the root density of the crop. The concept of plant extractable water is meaningful only in that depth where roots are dense enough to effectively extract water.

PLANT AVAILABLE WATER AND SOIL-WATER MANAGEMENT

Irrigation scheduling is an obvious application of plant available water information. Through proper irrigation scheduling, it should be possible to apply only the water that matches the evapotranspiration or to meet some other desired criteria.^[13,14] A rule of thumb is that irrigation is needed when the soil water content drops to 50% of potentially extractable soil water for crops such as alfalfa, corn, and spring grains.^[13] However, reduction in growth has been observed at a water content as low as 85% of potentially extractable soil water, in sandy soils.^[2]

Precision agriculture is the management of spatial and temporal variability in all aspects of agricultural production to improve crop performance and environmental quality.^[15] Ideal management variables are those with high spatial dependence and low temporal variance. Plant available water probably has high spatial dependence and high temporal variance, so it is not an ideal candidate for precision management, yet there are two possibilities for precision water management that are related to plant available water: variable rate irrigation and matching agronomic inputs to plant available water defined by soil and/or landscape properties.

Variable sprinkler irrigation systems exist, usually coupled with chemigation. Limited experience so far indicates that the benefits of such systems have not

been fully realized. The potential usefulness of matching agronomic inputs to plant available water increases with spatial variability in plant available water. Mapping small-scale variability is expensive, so perhaps a more viable approach is to map relief or some landscape attribute that is indirectly related to plant available water.^[15]

Tillage seems an important factor in the conservation of plant available water in dryland production. In one case, no-tillage resulted in improved soil water conservation during fallow compared to stubble mulch tillage.^[16] No-tillage afforded substantially higher soil water contents during planting of the crop after fallow. This could be because no-tillage results in less evaporation.

CONCLUSION

The definitions of the upper and lower limits of plant available water are difficult to establish because of water flow into and out of the root zone and because of incomplete extraction by sparse roots at the lower boundaries of the root zone. Pressure extraction equipment used on soil samples removed from a field often does not provide reliable estimates of the limits of water availability when comparing it to observations in the field. Therefore, for accurate soil water balance, it is important to measure the upper and lower limits in the field.

If the limits of plant available water cannot be measured, they can be estimated based on routinely measured soil properties. We present a simple model that uses few parameters. The model can make it easier to approximate soil water limits in water balance simulation, irrigation scheduling, and watershed management. A challenge will be to manage the spatial and temporal variability in plant available water in precision agriculture systems to improve crop performance and environmental quality.

REFERENCES

1. Jones, O.R.; Johnson, W.C. Cropping practices: Southern great plains. In *Dryland Agriculture*; Dregne, H.E., Willis, W.O., Eds.; Agron. Monograph 23; ASA and SSSA: Madison, 1983; 365–385.
2. NeSmith, D.S.; Ritchie, J.T. Short- and long-term responses of corn to a pre-anthesis soil water deficit. *Agron. J.* **1992**, *84*, 107–113.
3. Ratliff, L.F.; Ritchie, J.T.; Cassel, D.K. Field-measured limits of soil water availability as related to laboratory-measured properties. *Soil Sci. Soc. Am. J.* **1983**, *47*, 770–775.
4. Jones, C.A.; Ritchie, J.T.; Kiniry, J.R.; Godwin, D.C. Subroutine structure. In *CERES-Maize: A Simulation Model of Maize Growth and Development*; Jones, C.A., Kiniry, J.R., Eds.; Texas A&M Press: College Station, 1986; 49–105.

5. Buttlar, I.W.; Riha, S.J. Water fluxes in oxisols: A comparison of approaches. *Water Resour. Res.* **1992**, *28* (1), 221–229.
6. Black, T.A.; Gardner, W.R.; Thurtell, G.W. The prediction of evaporation, drainage, and soil water storage for a bare soil. *Soil Sci. Soc. Am. Proc.* **1969**, *33*, 655–660.
7. Dardanelli, J.L.; Bachmeier, O.A.; Sereno, R.; Gil, R. Rooting depth and soil water extraction patterns of different crops in a silty loam haplustoll. *Field Crops Res.* **1997**, *54*, 29–38.
8. Cassel, D.K.; Ratliff, L.F.; Ritchie, J.T. Models for estimating *in situ* potential extractable water using soil physical and chemical properties. *Soil Sci. Soc. Am. J.* **1983**, *47*, 764–769.
9. Cosby, B.J.; Hornberger, G.M.; Clapp, R.B.; Ginn, T.R. A statistical exploration of the relationships of soil moisture characteristics to the physical properties of soils. *Water Resour. Res.* **1984**, *20*, 682–690.
10. Ghosh, R.K. Estimation of soil–moisture characteristics from mechanical properties of soils. *Soil Sci.* **1980**, *130*, 60–63.
11. Gupta, S.C.; Larson, W.E. Estimating soil water retention characteristics from particle size distribution, organic matter percent, and bulk density. *Water Resour. Res.* **1979**, *15*, 1633–1635.
12. Ritchie, J.T.; Gerakis, A.; Suleiman, A. Simple model to estimate field-measured soil water limits. *Trans. ASAE* **1999**, *42* (6), 1609–1614.
13. Hill, R.W. Irrigation scheduling. In *Modeling Plant and Soil Systems*; Hanks, J., Ritchie, J.T., Eds.; Agron. Monograph 31; ASA and SSSA: Madison, 1991; 491–509.
14. Ritchie, J.T.; Amato, M. Field evaluation of plant extractable soil water for irrigation scheduling. *Acta Hort.* **1990**, *278*, 595–615.
15. Pierce, F.J.; Nowak, P. Aspects of precision agriculture. *Adv. Agron.* **1999**, *67*, 1–85.
16. Jones, O.R.; Hauser, V.L.; Popham, T.W. No-tillage effects on infiltration, runoff, and water conservation on dryland. *Trans. ASAE* **1994**, *37* (2), 473–479.

Water-Repellent Soils

Anna Eynard

Ohio State University, Columbus, Ohio, U.S.A.

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Keith D. Wiebe

Resource Economics Division, U.S. Department of Agriculture (USDA), Washington, District of Columbia, U.S.A.

Abstract

Relative amounts of hydrophilic and hydrophobic constituents in the structure of the soil determine soil water repellency or wettability. Changes in the spatial distribution of hydrophilic and hydrophobic soil constituents, in their mutual interactions, and in their interactions with water change soil water repellency. Some degree of water repellency is favorable to the conservation and to the productivity of the soil. However, remarkable production and soil losses occur in severely water repellent soils. In addition, water repellency can be a major concern for the contamination of groundwater with pollutants not retained on hydrophobic soil surfaces. Therefore, preventive and remedial management practices are needed for alleviating problems of water repellent soils.

INTRODUCTION

Soil water repellency is the inability of the soil to become wet and is the opposite of wettability. Soil and water quality and crop yields can be strongly affected by soil water repellency through decreased moisture availability to plants, accelerated soil erosion, and nutrient and pesticide transport. Soil water repellency is determined to a large extent by the hydrophobicity (repulsion for water) and the hydrophilicity (affinity for water) of soil surfaces (Fig. 1). The liquid–solid contact angle and the wetting time tend to zero on hydrophilic surfaces, so that water molecules lay parallel to hydrophilic surfaces. The liquid–solid contact angle increases toward 180° with an increase in hydrophobicity. Soil water repellency can be quantified by measuring the contact angle between soil surfaces and water. Otherwise, water repellency can be expressed by the rate of water entry in unsaturated soil. Amount, sizes, geometry, orientation, connectivity of soil pores, and hydrophobicity of pore walls affect soil water repellency. Water repellency is a dynamic soil property that varies with time following the soil structural and water potential changes during wetting and drying processes. “Actual” and “potential” water repellency characterize the wetting behavior of the soil because soils may be wettable when moist and be highly water repellent when dry.^[1] Water repellency shows a wide spatial variability changing with landscape position at a topographic scale, with soil depth

and area at a field scale, and along pore surfaces at a microscale. In general, soils are not completely water repellent, but their wetting rate can be relatively slow. Subcritical water repellency is the term used for soils with limited wettability.^[2]

EXTENT AND ORIGIN

Soil water repellency may affect soils of any texture in any climatic conditions around the world. From Australia to Canada, Egypt, India, the Netherlands, Japan, Spain, Colombia, New Zealand, South Africa, Turkey, Taiwan, Great Britain, Chile, and Mali among many other countries, problems of soil water repellency have concerned agricultural, range, and forest lands.^[3,4] Wide spread occurrence of water repellency has been recorded also in the United States from Oregon to Florida, Wisconsin, Arizona, California, New Mexico, Colorado, Nevada, and Michigan.^[3,4] Major sources of water repellency are organic compounds bound to soil minerals, or organic coatings as well as organic debris including hydrophobic particulate organic matter in the sand fraction and hydrophobic fine-structured organic particles in the clay fraction.^[5] Soil water repellency is not related to a specific size fraction. However, sandy soils are more susceptible to hydrophobic coating by organic matter than soils containing ≥15–20% clay minerals that have highly hydrophilic

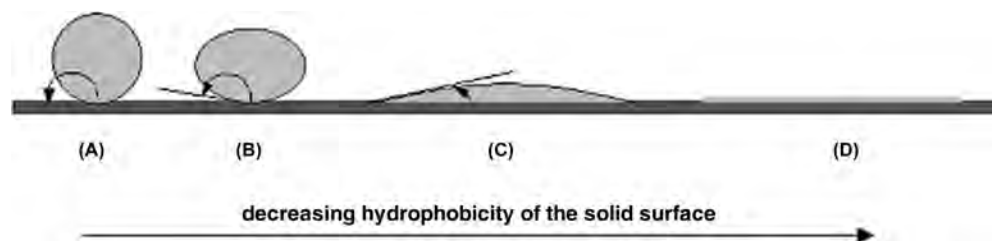


Fig. 1 Water drops on a solid surface. The hydrophobicity of the surface and the liquid–solid contact angle decrease from (A) to (D).

specific surface area.^[6] Organic matter may affect wettability by: 1) increasing the hydrophobic surface and/or 2) stabilizing the soil pores.^[7] The stabilization of soil pores is likely to prevail in clay soils. The origin and the degree of decomposition of the organic matter affect soil water repellency.^[8] Fungal growth and partial alteration of plant organic matter are favorable to the production of hydrophobic compounds. Fire-induced water repellency results from burning of organic matter. During fires, soil particles and aggregates can be coated through the deposition of volatilized compounds. Fires with temperatures from 170°C to 280°C at the mineral soil surface are optimal to produce hydrophobic depositions in subsurface cooler layers.^[9] Water repellency changes with the type and configuration of organic matter. Amphipathic molecules are likely to be involved in temporal changes of wettability. Organic polar groups exposed in wet soils may mutually interact in dry water repellent soils. Hydrophobic organic compounds, such as lipids, can cause severe water repellency even in small amounts in combination with hydrophilic organic compounds. The extraction of both water repellent and water-soluble organic fractions can effectively increase the wettability of the soil.^[10] No specific hydrophobic substances have been detected in fire-induced organic coatings.^[9] In most soils, hydrophilic organic compounds dominate, and they can decrease or increase soil water repellency. Highly hydrophilic molecules such as polysaccharides can decrease soil rehydration rate and wettability.^[11,12] Organic compounds with polar groups interacting with clay minerals can limit clay swelling and increase pore stability. Strengthening of soil pores can increase soil water repellency because aggregate slaking is prevented and water cannot enter quickly into the soil as through planes of failure created during aggregate collapse in slaked soil.^[13]

PRACTICAL IMPLICATIONS

Increased overland flow results from decreased infiltration in water repellent soils. Splash and rill erosions may be severe in water repellent soils especially in the case of fire-induced water repellency where runoff is favored by the lack of vegetation and pore plugging by ash. A water repellent fire-induced subsurface layer can slow down water percolation. When the surface layer becomes saturated, failure tends to occur at the boundary between water

repellent and wettable layers, and the soil may slide down the slope.^[9] Similar erosion losses have been recorded on unburnt water repellent sloping soils.^[14] Water repellent soils tend to be drier than wettable soils with the reductions of crop growth. Delayed germination in drier areas of pasture and crops favors wind erosion and increases weed competition. Crop and pasture production is reduced by water repellency on 5 million ha in Australia.^[15] Preferential flow may cause nutrient losses and contamination of groundwater with fertilizers and pesticides percolating rapidly through wettable fingers in water repellent soils (Table 1).

On the other hand, water repellency can reduce wetting rates and consequent stresses on soil aggregates increasing wet aggregate stability. Subcritical water repellency creates the most favorable conditions for structural stability and plant growth.^[16]

The spatial variability of water repellency needs to be taken into account in evaluating soil wettability of the soil in the field. Presence of thin water repellent layers in the profile may little contribute to the entire water repellency when they underlie hydrophilic layers, and the depth of the water above them creates a pressure head exceeding the breakthrough pressure head. Animal burrows, root channels, cracks, and hydrophilic soil pockets can reduce the effects of water repellency on runoff because they allow fingered water infiltration.^[14]

High water repellency has practical applications in water harvesting. Techniques for coating soil particles with water repellent films have been developed.^[3]

MANAGEMENT OF WATER REPELLENT SOILS

Forage species adapted to water repellent soils have been selected for increasing the productivity of water repellent pastures.^[15] Shrubs and trees can be productive in water repellent soils because, once established, they can uptake nutrients and water from the subsoil. Furrow sowing in moist soil below the dry surface or trench planting of

Table 1 Possible effects of severe soil water repellency.

Increased runoff, splash, and rill erosions
Reduced infiltration and soil moisture
Increased preferential flow, nutrient, and pesticide leaching

Table 2 Management practices for water repellent soils.

Use of tolerant plants and perennial, deep-rooted plants

Furrow sowing and trench planting

Mulching

Liming

Application of surfactants

Application of clay minerals (kaolinite)

Prevention by

1. Regular use of prescribed fires in forests
2. Frequent irrigation of crops and turfs

seedlings in wettable subsurface layers may help crop establishment. Furrow sowing and trench planting take advantage of the remaining strips of water repellent soil to harvest water, which can flow to the seeds. Narrow furrows with mulching, possibly refilled after plant establishment, are convenient in climates with wet periods to prevent excess of water in the furrows, excess of preferential flow, and consequent nutrient and pesticide leaching (Table 2).

Fire-induced water repellency can be prevented by regular use of prescribed fires that minimizes soil heating. Soil chemical stabilizers and wetting agents, although expensive, have been locally applied to reduce post-fire erosion, whereas mechanical techniques such as disking are generally impractical in sloping and stony lands.^[9] The prevention of water repellency in turfgrass systems can be achieved by maintaining high soil moisture and fertility with frequent irrigations and fertilizer applications. The alleviation of water repellency in intensively managed turfs can be obtained by the applications of surfactants. In water repellent cropped fields or pastures, surfactants may be applied in bands in the furrows where seeds are planted to improve wetting. Especially non-ionic formulations of surfactants can be effective against soil water repellency without causing phytotoxic secondary effects.^[17]

Clay minerals have been successfully added to amend water repellent sandy soils. Kaolinite may be preferable because kaolin crystals spread out covering the hydrophobic surfaces of soil particles and aggregates, whereas smectitic clays tend to aggregate upon drying.^[18] Added clays reduce soil water repellency, increase soil moisture retention, and may also increase soil fertility by improving the cation exchange capacity. Clays need to be evenly incorporated in the water repellent soil for best results. Claying has improved the productivity of 37,000 ha in South Australia, in general doubling crop yields, increasing pasture quality and forage yields from 400 to >3000 kg ha⁻¹.^[19] Enhanced microbial degradation of hydrophobic materials may contribute to the improvement of wettability by clay additions. Liming may alleviate water repellency especially when fungal hydrophobic

compounds are prevalent by shifting the composition of the microbial population of the soil.

No-tillage and mulching can reduce soil drying, while old roots and more abundant faunal macropores can provide channels for water infiltration. Moreover, no-till systems can reduce risks of nutrient leaching and groundwater contamination through organic matter retention and microbial degradation of pesticides.^[15]

CONCLUSION

Relative amounts of hydrophilic and hydrophobic constituents in the structure of the soil determine soil water repellency or wettability. Changes in the spatial distribution of hydrophilic and hydrophobic soil constituents, in their mutual interactions, and in their interactions with water change soil water repellency. Some degree of water repellency is favorable to the conservation and to the productivity of the soil. However, remarkable production and soil losses occur in severely water repellent soils. In addition, water repellency can be a major concern for the contamination of groundwater with pollutants not retained on hydrophobic soil surfaces. Therefore, preventive and remedial management practices are needed for alleviating problems of water repellent soils.

REFERENCES

1. Dekker, L.W.; Ritsema, C.J. Wetting patterns and moisture variability in water repellent dutch soils. *J. Hydrol.* **2000**, *231–232*, 148–162.
2. Tillman, R.W.; Scotter, D.R.; Wallis, M.G.; Clothier, B.E. Water-repellency and its measurement by using intrinsic sorptivity. *Aust. J. Soil Res.* **1989**, *27*, 637–644.
3. DeBano, L.F. Water repellency in soils: A historical overview. *J. Hydrol.* **2000**, *231–232*, 4–32.
4. Jaramillo, D.F.; Dekker, L.W.; Ritsema, C.J.; Hendrickx, J.M.H. Occurrence of soil water repellency in arid and humid climates. *J. Hydrol.* **2000**, *231–232*, 105–111.
5. Chenu, C.; Le Bissonnais, Y.; Arrouays, D. Organic matter influence on clay wettability and soil aggregate stability. *Soil Sci. Soc. Am. J.* **2000**, *64*, 1479–1486.
6. Farmer, V.C. Water on particle surfaces. In *The Chemistry of Soil Constituents*; Greenland, D.J., Hayes, M.H.B., Eds.; John Wiley & Sons: New York, 1978; 405–448.
7. Quirk, J.P. The nature of aggregate stability and implications for management. In *Soil Physical Properties and Crop Production in the Tropics*; Lal, R., Greenland, D.J., Eds.; John Wiley & Sons: New York, 1979; 57–74.
8. Doerr, S.H.; Shakesby, R.A.; Walsh, R.P.D. Soil hydrophobicity variations with depth and particle size fraction in burned and unburned *Eucalyptus Globulus* and *Pinus Pinaster* forest terrain in the agueda basin, Portugal. *Catena* **1996**, *27*, 25–47.
9. DeBano, L.F. The role of fire and soil heating on water repellency in wildland environments: A review. *J. Hydrol.* **2000**, *231–232*, 195–206.

10. Horne, D.J.; McIntosh, J.C. Hydrophobic compounds in sands in New Zealand—extraction, characterization and proposed mechanisms for repellency expression. *J. Hydrol.* **2000**, *231–232*, 35–46.
11. Czarnes, S.; Hallett, P.D.; Bengough, A.G.; Young, I.M. Root and microbial-derived mucilages affect soil structure and water transport. *Eur. J. Soil Sci.* **2000**, *51*, 435–443.
12. Chenu, C. Clay- or sand-polysaccharide associations as models for the interface between micro-organisms and soil: Water related properties and microstructure. *Geoderma* **1993**, *56*, 143–156.
13. Panabokke, C.R.; Quirk, J.P. Effect of initial water content on stability of soil aggregates in water. *Soil Sci.* **1957**, *83*, 185–195.
14. Shakesby, R.A.; Doerr, S.H.; Walsh, R.P.D. The erosional impact of soil hydrophobicity: Current problems and future research directions. *J. Hydrol.* **2000**, *231–232*, 178–191.
15. Blackwell, P.S. Management of water repellency in Australia, and risks associated with preferential flow, pesticide concentration and leaching. *J. Hydrol.* **2000**, *231–232*, 384–395.
16. Ellies, A.; Grez, R.; Ramirez, C. Wettening potential and stability of aggregates in soils with different management. *Agr. Tech.* **1995**, *55*, 220–225.
17. Kotska, S.J. Amelioration of water repellency in highly managed soils and the enhancement of turfgrass performance through the systematic application of surfactants. *J. Hydrol.* **2000**, *231–232*, 359–368.
18. McKissock, I.; Walker, E.L.; Gilkes, R.J.; Carter, D.J. The influence of clay type on the reduction of water repellency by applied clays a review of some west Australian work. *J. Hydrol.* **2000**, *231–232*, 323–332.
19. Cann, M.A. Clay spreading on water repellent sands in the south east of South Australia—Promoting sustainable agriculture. *J. Hydrol.* **2000**, *231–232*, 333–341.

Watershed Approach

Timothy R. Green

Great Plains Systems Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Fort Collins, Colorado, U.S.A.

Jan van Schilfgaarde

U.S. Department of Agriculture (USDA), Fort Collins, Colorado, U.S.A.

Abstract

Soils are open systems with natural and human-induced complexities. Water is an important change agent and state variable in soil systems that interact over a broad range of space-time scales. Thus, soil and water management must be understood and practiced in the appropriate context of the landscape and hydrologic system.

THE HYDROLOGIC CYCLE

The hydrologic cycle is defined as the movement of water from the ocean and other sources into the atmosphere, then to the land surface, yielding flow and storage over and through soils and other geologic media to groundwater aquifers, streams, and rivers, and finally returning to the ocean. The water's physical state (liquid, solid, or vapor), energy state, and chemical composition (dissolved constituents) may be altered at various points along its journey through the cycle. Its particular state as it enters, transits, and exits, the soil system plays an important role in the physical, chemical, and biological processes that occur in the soil, including their rates. The hydrologic processes of a particular system may vary significantly in space and time, and much of the local variability is linked to the global climate system. Various aspects of the global climate are not well understood (and poorly predicted), but multidisciplinary and international efforts are aimed at improving this aspect of the science and policy.^[1]

The hydrologic cycle may be considered a closed system (in terms of conservation of mass) that encompasses open systems of soils (Fig. 1). Water moves into and out of a soil system in multiple directions, even though one may tend to think of (vertical) surface infiltration and evapotranspiration dominating the water balance. Water is also stored in the soil–plant system for various periods of time. The residence time of this water and its constituents is important for biochemical transformations, and the rate of water flow affects the transport of key chemicals within and through soils.

A watershed, in turn, is defined as a region of land that contributes to a river system or reservoir. It is a compartment of the full hydrologic cycle that may cover an area of a few hectares to many thousands of square kilometers.

Water flow and storage within a watershed can be related to its geology, landscape topography, and soil hydraulic properties. The surface topography generally determines the energy gradients for movement of incipient precipitation over and through the ground. The flow paths may be confounded by complex geology and development of the parent material, along with the partitioning of water between infiltration and runoff based on the soil hydraulic properties. The location of a soil system within a watershed will determine the soil physical properties and the potential for inflow and outflow of water (e.g., inflow of water from upland areas, exchange of soil water with shallow groundwater, and the degree of natural and human-induced drainage). The landscape position also affects the chemical status of the soil that controls processes such as plant growth and biological toxicity. Thus, the location of an open soil system within a watershed is paramount to how the soil system and processes of interest should be managed.

NATURAL VARIABILITY

Soil and water management on a watershed scale must consider the high degree of variability in soils and related factors within a watershed. Soil formation and development have been studied extensively, and the foundational work of Jenny^[2] qualitatively identified five factors of soil formation: parent material, topography, climate, biota, and time. Additional factors include human activities (see below). While some soil attributes vary slowly compared with the time period of interest, the dynamics of other attributes cannot be ignored. From a practical point of view, some fraction of the natural variability cannot be explained; thus predictions must be assigned some level

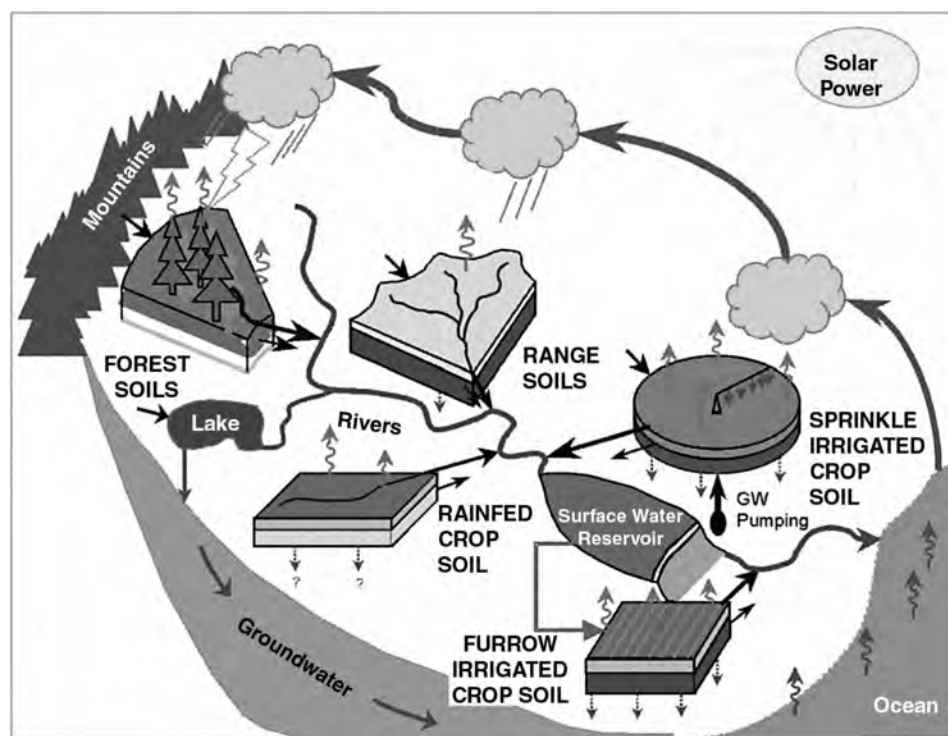


Fig. 1 A schematic view of open soil systems in the context of a watershed and hydrological cycle: upland forests, rangelands, dry-land cropping systems, and pivot- and furrow-irrigated cropping systems.

of uncertainty that the producers, land managers, or other stakeholders must accept.

SCALE: MOVING UP FROM POINT TO PLOT TO FIELD TO WATERSHED

Various properties of a soil may be measured or characterized on a small scale that is essentially a point in comparison with the variables or processes of interest. Many research studies in agronomy have been performed on experimental plots, typically on the order of a few tens of square meters. One of the most important challenges in the earth, soil, and agronomic sciences is transferring information from one spatial scale to another (i.e., scaling) for management purposes. There are two problems with scaling within a watershed: aggregation of land units (upscaling) and disaggregation to individual land unit responses (downscaling). Upscaling provides important information about cumulative (off-site) effects on water quantity and quality over a watershed, while downscaling provides the necessary information to prescribe spatial management and predict on-site effects. The theoretical challenge involves both statistical scaling—based on measurements over a broad range of scales—and process-based, spatially distributed prediction. By addressing these issues and developing new computational methods to combine with measurements on real landscapes, practical prediction tools (with estimated uncertainties) for land management at appropriate space and time scales can be provided.

MANAGEMENT EFFECTS

The effects of historical land management must be borne in mind when devising management plans on a large scale. Most of the land masses of the earth have been impacted to some degree by human land management practices. These include burning and clearing for agriculture, forestry, water diversion and storage, urbanization, transportation infrastructure, mining, and various agricultural and industrial land uses. In agriculture, widespread irrigation and drainage have converted deserts and wetlands, respectively, into croplands. Overirrigation and lack of adequate drainage have resulted in salinity problems and waterlogging. Less dramatic, yet significant, impacts on rainfed lands can be related to tillage practices, crop rotation, and grazing rates.

Advances in satellite-based remote sensing and geographical positioning have led to high-resolution mapping of landscape properties and state variables. This can be applied to spatial soil water management practices in the hope that average agricultural production will increase while adverse environmental effects decrease. For example, the practice of varying agronomic input rates within a farm field, known as “precision agriculture,” touts such technology-dependent promises (see van Schilfgaarde^[3]; www.precision.agri.umn.edu). Although the technology has improved dramatically, there is a need for improved scientific prediction to aid management decisions. Furthermore, the long-term feedback between landscape variability and spatial management will need to be addressed.

STRATEGY

A strategy for soil and water management should be based on a watershed approach to account for the various inputs that control the soil system as well as to determine off-site impacts of the management practices used. The watershed approach recognizes multiple land uses within a given watershed such that management practices on one land area affect another. These interactions in space and time have been explored in some situations but generally require further consideration.

Potential strategies are discussed below for croplands (irrigated and rainfed), rangelands, and forests. The soil scientist must recognize all important (or potentially important) uses and devise a means for accommodating all existing and planned uses. Diverse uses are likely to interact and often cause conflict. This requires developing technically sound alternative proposals, but even more importantly, engaging in conflict resolution by involving all interested parties in meaningful negotiation. Thus, community acceptance is added to the physical problem.

Irrigated Croplands

Some 37% of global crop production comes from the 12% of arable land that is irrigated, much of it in arid climates. Irrigation uses a large portion of available water resources and consequently may deprive cities or industries of water resources. Reduced water in streams may affect fish and wildlife adversely, and return flow from irrigation always degrades the quality of the remaining water resource. Thus, the strategy in managing irrigated agriculture must be to find a balance between crop production and other uses. One component of such a strategy may be to increase irrigation efficiency so as to make more water available for other purposes; another is to concentrate polluting salts in a smaller volume of drainage water or return flow to simplify separate disposal. In a report on agricultural salinity, van Schilfgaarde^[4] addressed the question looming over irrigated agriculture: "Is it sustainable?" Due to the inevitable degradation of water quality in streams and terminal basins and the associated costs, the question of sustainability was restated as, "Are we prepared to maintain irrigated agriculture indefinitely?"

Rainfed Croplands

Many rainfed agricultural areas have been (or need to be) drained to enhance production. Reducing wetlands, however, deprives migrating birds of nesting and resting places, disturbing the natural order. Fish and wildlife specialists need to work with farm interests to find a balance. This may include the preservation of some wetlands, restoring fish habitat in streams and drainage channels, and using crop rotations that encourage diverse fauna. Income from hunting may offset possible losses from reduced crop

production, and ecological criteria may supersede economic benefits.

Rangelands

Rangelands serve many functions: They store snow and modulate runoff to enhance stream flow, they support wildlife and diverse flora, and of course they provide grazing lands for cattle, among others. Overgrazing can affect the vegetation, hydrology, and soil erosion. Cattle on stream banks may also cause erosion and pollute the water. A strategy for sustainable management goes beyond providing the maximum number of grazing days for a herd of cattle.

Forests

Issues of forest management are not to be overshadowed by the above issues for agricultural lands. A disadvantage in terms of soil water management is that forest soils tend to be overshadowed by the trees—literally and figuratively. (See www.soilslab.cfr.washington.edu/S-7/HistForSoilNA.html for a historical perspective.) From the existing perspective, however, forests have the advantage of being readily viewed in a watershed context. This is due in large part to the scale of operations and management of a forest system. The primary environmental indicators of success or failure are stream water quality and associated aquatic habitat. This has led to an emphasis on riparian zones and buffer strips to retain sediment and nutrients. Such an approach automatically integrates land area responses, but it is complicated by multiple users and stakeholders.

Multiple Use Conflicts

When water resources are managed to achieve a specific purpose, such management is likely to preclude, or at least interfere with, alternative uses. Irrigation, drainage, and erosion control are examples of water management practices that support sustainable agriculture. However, irrigation may exhaust water resources desired for urban or industrial uses or reduce water quality and thus adversely affect fish and wildlife. Similarly, drainage may remove wetlands and eliminate nesting areas for migrating birds. Thus, major conflicts can arise. Whereas soil and hydrologic sciences can often explain or even predict the consequences of specific actions, resolving the conflicts generally requires input from the social sciences.

Many countries, states, and regions are adjusting their soil water practices to account for diversity in the opinions of stakeholders. Some of the policy direction is implicitly or explicitly directed by a watershed approach, simply because one cannot neglect interactions between land areas related to geomorphic and hydrologic factors. Development of appropriate strategies and specific practices is a site-specific endeavor that will lead to balanced resource use,

not only to protect the environment but also to satisfy the people depending on the resource. Policy decisions will vary by region according to political circumstances, social desires, and economics as well as the physical and biological situation. To be effective, nature demands that an integrated watershed approach be used.

EXAMPLES OF HISTORICAL PRACTICES

There are many examples of soil and water management practices from around the world that could illustrate the importance of a watershed approach. If space permitted, one could explore case studies on drainage of the Nile River Delta in Egypt, soil erosion of the Loess Plateau in China, historical land clearing and intensive agriculture throughout Europe, salinity and phosphorus issues in landscapes across Australia—including the Murray–Darling river basin and the wheat belt of Western Australia—and a variety of problems requiring (in hindsight, at least) an integrated watershed management approach. This entry limited to three important concepts briefly illustrated by the following case studies.

Colorado River, United States

The Colorado River Basin includes parts of seven U.S. states as well as a part of Mexico. The river rises as a pristine series of streams in the high Rocky Mountains and eventually drains into the Gulf of California. The average annual flow of the river is about $1.7 \times 10^{10} \text{ m}^3$. Based on a relatively wet period of record, however, the courts have decreed that the seven U.S. states are entitled to $1.8 \times 10^{10} \text{ m}^3$, and by treaty it has been agreed that Mexico is entitled to $1.8 \times 10^9 \text{ m}^3$. Furthermore, water quality standards have been imposed in terms of salinity (total dissolved solids). The origin of the salts is dissolution of minerals from the geologic formations and soils through which the water flows. The concentration of salts is increased when irrigated crops transpire pure water and thus pass on all the salts in a smaller volume of return flow.

Colorado River water is used for irrigation, as a municipal supply in cities like Los Angeles, and for many other uses. When the river finally reaches Mexico, the flow is minimal and the salt concentration tends to be high. Clearly, the legal entitlements cannot be met with the water that nature provides. This water shortage and its distribution have been litigated for three-quarters of a century.

An example of an effort to reduce the problem somewhat is pertinent here. In the Grand Valley of Colorado, some 28 million hectares were irrigated with Colorado River water when the Colorado River Salinity Control Program was started in the late 1970s. It was considered the largest single source of salt to the Colorado River. To maintain the agriculture and at the same time protect the river, a large project was implemented to line delivery canals,

replace some of the ditches with pipe, and improve on-farm irrigation efficiency by the introduction of a number of modern practices. These local practices integrated over the Grand Valley had a significant impact on the quantity and quality of water contributing to the Colorado River. The U.S. Bureau of Reclamation provides additional information at dataweb.usbr.gov/html/grandvalley2.html.

San Joaquin Valley, California, United States

In the 1920s, plans were developed to transport substantial amounts of water from northern California to the San Joaquin Valley, primarily for irrigation. Although engineers warned that drainage needed to be provided to make the irrigation enterprise viable, political and financial obstacles prevented the building of a drainage facility for decades. When parts of the nearly 20 million hectares Westlands Irrigation District started to suffer substantially from waterlogging and salinity, one-half of a large drain (the upper half only!) was installed, with its water discharged “temporarily” into a low area euphemistically named the Kesterson Wildlife Refuge. Whereas the drain appeared to be successful as a salt-removal system, even if not completed, it soon became evident that selenium originating in the soils was transported into the Kesterson Reservoir at concentrations sufficient to greatly damage, or kill, aquatic life and birds landing on the water. Finding a solution acceptable to all parties has been difficult. The interests involved include naturalists, sports fishermen and hunters, and farmers. Space does not permit further exploration other than to note that similar adverse effects from trace elements leached from open soil systems have been discovered in other watersheds.

Kissimee River, Florida, United States

Humid regions also present challenges for soil and water management. One such environment in south Florida is a system of streams and wetlands flowing into Lake Okeechobee, just to the north of the well-known Everglades National Park. Florida is very flat, and in its natural state, the shallow Kissimee River meandered from the south end of Lake Kissimee to Lake Okeechobee through wetlands that absorbed and cycled nutrients from agricultural runoff. Occasional heavy rains caused the river to overflow its natural banks and flood adjacent lands.

In the 1960s, the U.S. Army Corps of Engineers was commissioned for half a million 1962 U.S. dollars to dredge a straight canal (called C-38) with water-control gates and boat locks. The dimensions of the river were changed from approximately 166 to 90 km long, from 30 to 91 m wide, and from 3 to 10 m deep. This provided for navigation and flood control but dramatically changed the water quality and ecology. As a result, large-mouth bass

and water fowl diminished in the Kissimee River ecosystem, and nutrients transported to Lake Okeechobee increased eutrophication problems.

In 1990, the State of Florida adopted the South Florida Water Management District's restoration plan (see www.sfwmd.gov for more details), and the U.S. Congress funded 50% of the estimated \$372 million cost through the 1992 Water Resources Development Act. The plan includes acquisition of approximately 360 km² of land, backfilling large portions of the canal, and attempting to return the Kissimee River to its natural state. Further details of this project can be found at fcn.state.fl.us/eog/govdocs/opbenv/saveglades/everglades/html/kissimee.htm.

What Lessons Have We Learned?

The discussion here gives the reader an overview of soil and water management as an integrated process using the watershed approach. While it is true that the watershed approach integrates several processes and landscape factors, it also involves disaggregation of the watershed into its various geographical parts. Both issues are challenging, primarily due to natural and human-induced variability and uncertainty in space and time. However, the advent of readily available spatial data is making research on these topics feasible, and in time sound management practices should follow.

The watershed approach to soil and water management also addresses political and economic issues of the

stakeholders involved in regional, national, and international policy and decision making. Downstream water users have long recognized the importance of upstream land use practices, and it seems policy makers are increasingly facing the challenges of integrated watershed management for water quantity and quality. As history shows, practices of soil and water management are based primarily on social concerns and politics with varying degrees of import being placed on scientific understanding and engineering considerations. Regardless of the technical and political reasoning, some compromises or trade-offs will be required.

REFERENCES

1. IPCC. *Land Use, Land Use Change and Forestry. Special Report of the Intergovernmental Panel on Climate Change*; Watson, R.T., Noble, I.R., Bolin, B., Ravindranath, N.H., Verardo, D.J., Dokken, D.J., Eds.; Cambridge University Press: Cambridge, 2000; 377 pp.
2. Jenny, H. *Factors of Soil Formation*; McGraw-Hill Book Company Inc.: New York, 1941.
3. van Schilfgaarde, J. Is precision agriculture sustainable? *Am. J. Alternative Agr.* **1999**, *14* (1), 43–46.
4. van Schilfgaarde, J. Irrigated agriculture: Is it sustainable? In *Agricultural Salinity Assessment and Management*; Tanji, K.K., Ed.; ASCE Manual & Report on Engineering Practice, No. 71; American Society of Civil Engineers: New York, 1990; 584–595.

Watershed Management: Remote Sensing and GIS Applications

A.V. Shanwal

Department of Soil Science, Chaudhary Charan Singh Haryana Agricultural University, Hisar, India

S.P. Singh

National Bureau of Soil Survey and Land Use Planning, Indian Council of Agricultural Research (ICAR), New Delhi, India

Abstract

Remote sensing (RS) and geographic information system (GIS) are important technologies for addressing challenges in sustainable management of natural resources. Remotely sensed data (e.g., aerial photographs and satellite imagery) can be used to obtain information on soils, land use, vegetation, slope gradient, runoff, erosion, etc. However, developments in a multispectral scanner, radar system, and a multitude of quantitative techniques for analyzing and processing such data provide opportunities for data acquisition through RS and an array of techniques for data analyses.

INTRODUCTION

The need for natural resources conservation must be considered in any agricultural development plans involving conversion of new land use to increase production. An action plan to minimize natural resources degradation must be based on the principles of sustainability so that soil can be handed over to the next generation under better conditions than received from the previous generation. Thus, management practices must be ecologically sound, economically feasible, and socially and politically acceptable.

Natural resources degradation can be contained by adopting the watershed as a hydrological unit for development and management. This approach is multidisciplinary, broad based, and intensive vis-à-vis the simple “Seed and Fertilizer Approach.” Despite the investment, it is an economic approach in the long run.^[1]

The degradation of improperly used watersheds happens because of an increase in soil erosion, decline in soil productivity, reduction in livestock-carrying capacity, decline in forest cover and perturbation in ecological equilibrium, and reduction in biodiversity. The problem requires a high-tech solution with back-up policy decisions.

Geographic information system (GIS) is an important tool for tracking spatial data. GIS draws the composite map by superimposing the data and image files obtained from traditional methods and satellite imageries; allows us to develop, analyze, and display spatially explicit information; and gives us the ability to deal with the larger spatial scales in process such as soil erosion and drainage (≤ 500 acres) and regional landscape (several million acres).

INVENTORYING AND MONITORING

Despite efforts at inventorying and monitoring, conventional techniques provide only sketchy information on resources in a watershed, their location, and spatial and dynamic distribution. Of inventorying and monitoring, the latter is probably the major new thrust for resource managers and scientists.

Improved management practices are not thought of spontaneously prior to testing and implementation. Such practices have evolved over time through experimentation, experience, and trial and error. A case in point is the Bunga watershed in Amabala, Haryana, India.^[2] As was expected, numerous components did not work properly. Neither the rate of runoff nor the outflow from the reservoir was in harmony in the initial stage. High rate of siltation of Sukhana Lake near Chandigarh, Haryana, India, due to severe soil erosion in the catchment area is another example of a dire need for inventorying of data prior to starting any project.^[3] Because of high siltation, the ponded area of Sukhana Lake decreased by about 30 ha between 1980 and 2000. During this period, the courses of Sukhana and Kansal streams feeding the lake also changed considerably. The siltation of Sukhana Lake is the result of poor vegetation cover and heavy runoff in the catchment within the Shiwalik Hills.^[3]

It is thus important that remote sensing (RS) and GIS specialist and the resource managers must work together to determine the specific information needed (e.g., the nature of earth surface cover and their characteristics; the location, size, terrain, and other characteristics of the watershed area involved). Relevant information needed includes the format

(e.g., maps, tables, and scales); time frame for both collecting and processing the data; level of accuracy and reliability; and costs of obtaining and interpreting/processing the data.

RESEARCH AND DEVELOPMENT

Using watershed, landscape or ecosystem approaches have broad support as a means in achieving sustainable use of natural resources and integrating objectives on a practical scale. Addressing watershed management issues on large scales requires enhanced technical capabilities and modern set of tools and facilities.

Landsat-1 was launched in 1972 and the Landsat thematic mapper (TM) type of data has been in existence only since 1984. Thus, a tremendous amount of progress has been made over a short time in developing effective methods of processing and analyzing such data. The last decade of the 20th century witnessed rapid advances and significant increase in the operational use of remotely sensed data.^[4]

Landsat-7 launched in 1998 carries enhanced thematic mappers and pointable sensor with improved spatial

resolution and signal-to-noise ratio (Table 1). The Earth Observing System consists of a morning (AM) and an afternoon (PM) components and carries five separate sensors including MODIS, a 36-narrow band imager with 1-km spatial resolution.^[6] These sensors are significantly improved in their capability to map watershed information classes. These advanced sensors and computational capability have new requirements for satellite data processing algorithms. Added to the image analysis system, there is also a need for RS algorithms for estimating biophysical parameters necessary to drive and validate the watershed process models.^[5]

During the 1980s, advances in computer hardware, particularly speed and data storage, catalyzed the development of software for handling spatial data. One of the most significant products of this period of rapid technological change was the development of GIS (Table 2). It has made a tremendous impact in identifying strategies of watershed development by manipulating and analyzing individual “layers” of spatial data and providing tools for analyzing and modeling the interrelationship among layers. The GIS also provides a means of displaying complex watershed information in a comprehensible manner.

Table 1 RS satellites with advanced sensors.

Satellite	Country	Sensor		
		Name	Spectrum	IFOV
Landsat-7	U.S.A.	ETM ⁺	0.45 ~ 12.50 μm (8 bands)	30 m
SPOT-4	France	HRVIR	0.50 ~ 1.75 μm (5 bands)	10 m
IRS-1D	India	PAN	0.50 ~ 0.75	5.8 m
		LISS-III	0.45 ~ 0.86 1.55 ~ 1.70 (5 bands)	24.0 m
RADARSAT	Canada	WIFS	0.67 ~ 0.86	188 m
		SAR	5.3 GHz	25 × 28 m
TRMN	Japan	VIRS	0.63 ~ 10.7 (5 bands)	2 km
		TMI	10.65 ~ 85.5 GHz	5.6 km × 3.8 km
		CERES	0.3 ~ 50 μm (3 bands)	25 km
		LIS	0.7774 μm	5 km
NOAA-M	U.S.A.	AVHRR/3	0.58 ~ 12.40 μm (6 bands)	0.5 km
CRSS	U.S.A.	PAN	0.45 ~ 0.90 μm	0.82 m
		MSS	0.45 ~ 0.90 μm (4 bands)	3.20 m
ADEOS	Japan	OCTS	0.402 ~ 12.5 μm (12 bands)	700 m
		AVNIR	0.40 ~ 0.92 μm (5 bands)	8 m
EOS-AM	U.S.A.	ASTER	0.52 ~ 11.3 μm (3 bands)	15 m
EOS-PM	U.S.A.	MODIS-N	0.659 ~ 14.24 μm (5 bands)	250 m
EOSAT	U.S.A.	PAN	0.45 ~ 0.90 μm	1 m
		MS	0.45 ~ 0.90 μm (4 bands)	4 m

Note: ETM, enhanced thematic mapper; HRVIR, high resolution visible and middle infrared; LISS, linear imaging self-scanner system; SAR, synthetic aperture radar; VIRS, visible infrared scanner; TMI, TRMM microwave imager; CERES, clouds earth's radiation energy system; AVHRR, advanced very high resolution radiometer; OCTS, ocean color and temperature scanner; AVNIR, advanced visible and near infrared radiometer; ASTER, advanced space borne thermal emission and reflectance radiometer; MODIS, moderate resolution imaging spectrometer-nadir; PAN, panchromatic; MS, multispectral; IFOV, instantaneous field of view.

Source: JSRS Remote Sensing Note.^[5]

Table 2 Different types of GIS.

GIS	Origin	CPU		Data model		Applications		
		PC	WS	Vector	Raster	Analysis	DTM	Network
ARC/INFO	ESRI	✓	✓	✓	✓	✓	✓	✓
MGE	Intergraph		✓	✓	✓	✓	✓	✓
Geo/SQL	Generation 5 Technology	✓	✓	✓		✓	✓	✓
GFIS	IBM	✓	✓	✓	✓	✓		✓
IDRISI	Clark University, U.S.A.	✓		✓	✓			
GRASS	GRASS Information Center	✓	✓		✓	✓		
ERDAS	ISRI	✓	✓		✓	✓	✓	✓
GRAMM ⁺⁺	IIT Bombay, India	✓	✓	✓	✓	✓		✓

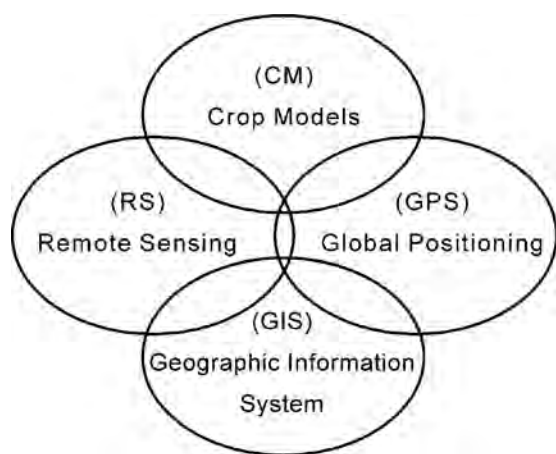
Source: From Maguire^[7] and Morehouse.^[8]

The GIS also provides a means of predicting the outcomes of alternative courses of action, from both spatial and temporal perspective and in a timely and cost-effective manner. However, it does not preclude the need for monitoring the ground truth as a guide to development of management practices.

The last decade of the 20th century witnessed rapid advances and a significant increase in the operational use of RS and GIS in watershed management.^[9] Much of this increase in operational use is due to the continued integration of RS, GIS, global positioning system (GPS), and crop model (CM) techniques.^[10] These four technologies form a powerful, interrelated combination (Fig. 1). In watershed management, there has been a continuous increase in the use of RS data to provide input to new GIS database, upgrade existing database, and monitor land use changes.

In addition to providing input to GIS database, the RS data enable significant improvements in classification accuracies. Further, GPS capabilities provide effective cartographic control to the GIS database and will enable field plots to be located efficiently and accurately in the data set.

The integration of GIS and CMs has proven very successful for alternative land use planning in the watershed.

**Fig. 1** Relationship between RS, GIS, GPS, and CM.

The important ones are Decision Support System for Agro-technology Transfer, Agricultural and Environmental Information System for Windows, Intelligent Decision Support System, and Decision Support System Engine. These simulation models are capable of predicting the potential yields of crops (rice, cassava, potato, sugarcane, sunflower, maize, wheat, barley, millet, sorghum, soybean, peanut, drybeans, tomato, and chickpea) in different physiographic units of a particular watershed. In addition, several GIS-based simulation models have been developed for natural resources management and are used in watershed management.^[11] The important ones are Scheduling and Network Analysis Program, Land Use Analysis System, and LANDSIM and Land Information System. All these models involve database (non-spatial), GIS, model base (simulation analysis), and a graphic user interface.

CHALLENGES AND OPPORTUNITIES AHEAD

Rapid advances in RS and GIS technologies can bring about quantum leap in watershed management. The challenge lies in assuring that RS and GIS technologies continue to serve the practitioners and users, but not vice versa.

However, GIS and RS are both a panacea and a Pandora's box. These are panacea because of the promise to meet the challenges of resource inventory and monitoring, and planning and policy analysis. These are Pandora's box because of the numerous pitfalls of using the tools wrongly, capturing the data poorly, miscommunicating information, conveying incorrect results, and overselling the capabilities.

The dilemma can be addressed by a careful integration of RS, GIS, and other simulation models in watershed management and with due consideration of the following:

1. Designing the classification system as totally exhaustive, mutually exclusive, and hierarchical.
2. Determining the temporal and spatial scale of the watershed by accommodating GIS and RS at multiple

scales without compromising flexibility and quality of a project.

3. Identifying the appropriate data sources (such as video, aerial photography, satellite imagery, and airborne scanner) for different land cover.
4. Assessing and reporting the accuracy of the data needs.
5. Limiting the scope of the project according to the budget and schedule.
6. Standardizing the formats needed for the exchange of information across projects and eliminating duplication.

Further changes may include the following:

1. Developing effective techniques of integration and analysis of data from various sources such as advanced very high resolution radiometer and Landsat TM or SPOT data or Landsat TM plus satellite radar data.
2. Converting research into operational applications in watershed management.
3. Developing effective expert system to assist the analyst.
4. Educating and training the user community in principles and theory of these technologies so that they can use these powerful tools wisely, appropriately, and effectively.

Anticipated developments and opportunities in these technologies comprise the following:

1. Economic availability of satellite optical sensor data with improved spatial, spectral, radiometric, and temporal resolutions.
2. Availability of operational multifrequency, multipolarization synthetic aperture radar data from satellite altitudes.
3. Improvement in computer storage and processing capabilities.
4. Better understanding and use of combined data from optical, microwave, and other remote sensors.
5. Integration of RS, GIS, GPS, and CM technologies.
6. Increased use of expert systems for data analysis.

These developments will improve the quality and characteristics of the data and analytical capabilities in the area of watershed management. A combination of knowledgeable resource managers and practical farmers with the technological tools and data available to them holds great promise for identifying effective watershed management technologies.

CONCLUSION

A variety of information about the characteristics and condition of the area is needed for judicious management of watershed. Aerial photography has been used since the 1950s for obtaining information on soils, land use, vegetation, slope, runoff, erosion, etc. The advent of space

research through satellite, multispectral scanner, and radar data, and quantitative analytical techniques of processing such data has increased the array of data, analysis procedure, and results that can be obtained using RS capabilities.

During the 1980s, advances in computer hardware for processing and data storage catalyzed the development of software for handling spatial and image digital data. These technologies played an important role in the development of GIS for natural resource management, especially in preparing composite map superimposing the data and image files.

During the 1990s, significant development in GIS technology and integration of RS, GIS, GPS, and crop simulation model techniques have created additional complexity and opportunities of using various data sources and analysis techniques for obtaining the information needed by the resource managers. There are some critical issues that need to be addressed for integration of RS, GIS, and other technologies to obtain information needed for sustainable natural resource management through watershed development.

REFERENCES

1. Dhruva Narayana, V.V.; Sastry, G.; Patnaik, U.S. *Watershed Management*; Indian Council of Agricultural Research: New Delhi, 1990.
2. Singh, G.; Mittal, S.P.; Dhruva Narayana, V.V.; Agnihotri, R.C. *Watershed Management Plan (Vil. Bunga)*; Central Soil and Water Conservation Research and Training Institute: Dehra Dun, 1983.
3. Sohal, H.S. Soil and water conservation measures for sediment control in Sukhana Lake, Chandigarh. In *Fifty Years of Research and Sustainable Resource Management in Shiwalik*; Mittal, S.P., Aggarwal, R.K., Samra, J.S., Eds.; Central Soil and Water Conservation Research and Training Institute Research Center: Chandigarh, 2000; 259–266.
4. Hoffer, M.R. Challenges in development and applying remote sensing to ecosystem management. In *Remote Sensing and GIS in Ecosystem Management*; Sample, V.A., Ed.; Island Press: Washington, D.C., 1994; 25–40.
5. JSRS Remote Sensing Note. *Japan Association on Remote Sensing*; University of Tokyo: Japan, 1993.
6. Hall, G.F. Adaptation of NASA remote sensing technology for regional-level analysis of forested ecosystem. In *Remote Sensing and GIS in Ecosystem and Management*; Sample, V.A., Ed.; Island Press: Washington, 1994; 287–314.
7. Maguire, D.J. The Raster GIS design model—A profile of ERDAS. *Comput. Geosci.* **1992**, *18* (4), 463–470.
8. Morehouse, S. The ARC/INFO geographic information system. *Comput. Geosci.* **1992**, *18* (4), 435–441.
9. Murai, S. *GIS Work Book*; Japan Association of Surveyors: Tokyo, 1996; 169 pp.
10. Hoogenboom, G.; Wilkins, P.W.; Tsuji, G.Y. *Discussion Support System for Agrotechnology Transfer*; University of Hawaii: Honolulu, 1999; Vol. 4.
11. Sample, V.A. *Remote Sensing and GIS in Ecosystem and Management*; Island Press: Washington, 1994.

Weathering

Dominique Righi

Hydrology, Clay, Soils and Alterations, National Center for Scientific Research (CNRS), Poitiers, France

Abstract

Clay minerals are the principal products of weathering and soil formation, which are essentially water–rock interaction processes. Clay formation occurring at the earth's surface is a slow reaction process, meaning that some initial materials are only partially reacted and are not in equilibrium with the medium in which they are located. This creates the potential for further transformation of altered products. If the chemistry of the ambient soil or surface materials is changed, e.g., by pollution containing chemicals, there will be a high potential for change in the clay mineralogy of the surface material. Thus, the type of clay minerals produced by weathering reactions and their rate of formation are of the greatest importance in environmental problems.

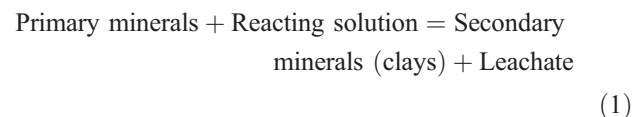
INTRODUCTION

Weathering is the alteration of materials at, or near, the earth's surface in response to the conditions that prevail there.^[1] It results from a complex set of interactions among the lithosphere, atmosphere, hydrosphere, and biosphere. The minerals that constitute the lithosphere are conventionally divided into primary and secondary phases (Fig. 1). *Primary* phase minerals, produced at depth by high-temperature and/or high-pressure processes, are found in igneous and metamorphic rocks. *Secondary* phase minerals are transformations (secondary minerals that result from partial breakdown and reorganization of the primary phase minerals) or neoformations (secondary minerals that precipitate directly from solutions in contact with the rock) produced at the expense of primary phase minerals as a result of weathering. In most earth-surface systems, secondary phase minerals include various clay minerals (e.g., aluminosilicates), oxides, and hydroxides. The aqueous phase (hydrosphere) enters the weathering system as atmospheric precipitations, whose chemistry and abundance are determined by climatic and other atmospheric factors. Within the weathering system, the aqueous phase is rapidly modified by the interaction with solids, including biomass. The dissolution of the most soluble minerals in the system (weathering phases) and precipitation of the least soluble minerals (new phases) tend to control the chemical composition of the aqueous phase. Residence time of the water in the weathering system is another important factor, so that a complex set of interactions connects the aqueous phase with other weathering factors.

Biomass is a major influence on the chemistry and behavior of water in weathering systems.^[2] The major effect of organic compounds is concentrated in the upper meter of the soil. Biomass yields important organic products containing

chemical functional groups, such as carboxyl and phenol, that provide protons to the weathering system. Many of these products can attack a mineral surface directly to release ions into solution. Moreover, complex organic acids, such as oxalic and citric acids, are efficient in mineral breakdown because they hold cations (e.g., Al^{3+} and Fe^{3+}) in solution that would otherwise precipitate.

Weathering is a form of reaction that may be written schematically^[3] as follows:



This process is strongly influenced by the chemical composition of the rock (i.e., the nature of its primary minerals) and environmental factors (e.g., climate, living organisms, and hydrodynamics). The environmental factors act through the aqueous phase, which is the first requirement of weathering.

MAIN WEATHERING MECHANISMS

Weathering mechanisms are determined by the highly variable physicochemical characteristics of the reacting solutions. Therefore, it is helpful to classify them as a function of the ionic concentration and the pH of these solutions.^[2,3]

Acidolysis and Acidocomplexolysis

The low pH of <5 of this weathering alteration is obtained through the dissociation of organic acids. However, the effect on the mineral fraction is determined by the type of

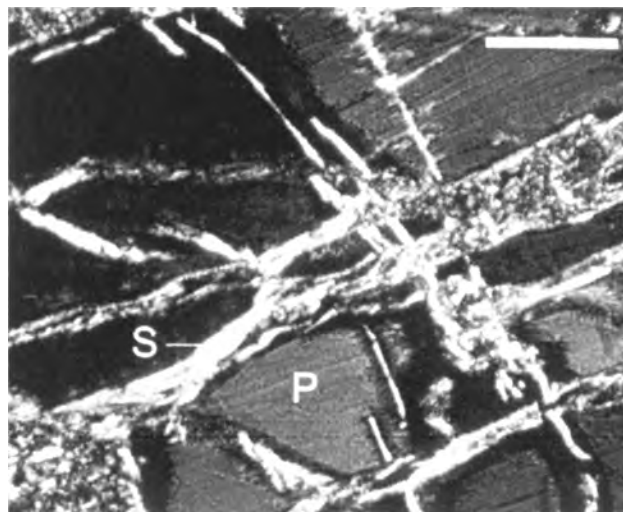


Fig. 1 Photograph from a thin section of a weathered rock (amphibolite) showing remnants of primary mineral (P) which is amphibole. Indicated by (S) are the secondary minerals (clay minerals) produced by reaction between the primary mineral and fluid percolating through the network of pores and fissures. Bar is 50 μm .

Source: Photo courtesy of Dr. Dominique Proust, Faculté des Sciences, Poitiers Cedex, France.

acid involved. Acidolysis, caused by non-complexing organic acids, acts similarly to hydrolysis (see below), whereas acidocomplexolysis, which involves strongly complexing acids (e.g., oxalic and citric), leads to the destruction of all minerals except quartz and others that are highly resistant. Aluminum (Al) and iron (Fe) are solubilized as metalorganic complexes. Acidocomplexolysis is typical for the pedogenetic process called podzolization, in which organic acidity and complexing ability are the important factors.

Hydrolysis

In this alteration, weathering is by pure water or water with dissolved carbon dioxide, which induces a pH of 4.5–6. Hydrolysis may be complete or only partial. Complete hydrolysis leads to the formation of Al hydroxide (gibbsite) and Fe oxides and oxyhydroxides (hematite and goethite), whereas partial hydrolysis leads to the formation of kaolinite or smectite clays as secondary minerals.

Salinolysis

Solutions in this weathering alteration are slightly alkaline and have a high ionic strength. This type of alteration is common in semiarid climatic zones, where there is a strong tendency for the aqueous phase to evaporate. The result is the precipitation of evaporite minerals (sulfates and chlorides) and the formation of various smectite clays.

Alkalinolysis

The aqueous solution in this alteration is salty and strongly alkaline, with a pH of 9 or more. As with salinolysis, evaporitic minerals form, but Al and silicon (Si) may reach relatively high concentrations in solution. This allows the formation of sodium smectites and alkaline zeolites.

Secondary minerals produced through salinolysis and alkalinolysis mechanisms are essentially by precipitation from solution processes (neoformation). However, the required high Si and magnesium (Mg) concentrations in the soil solution are most often reached by the alteration of pre-existing silicate minerals. For example, the dissolution of quartz at a high pH ensures that substantial soluble Si is available for the neoformation of secondary phyllosilicate phases.

BASIC STRUCTURE OF A WEATHERING PROFILE

Over time, weathering processes lead to the development of a weathering profile.^[4] The basic structure of the weathering profile of a basement rock shows successive zones or horizons (Fig. 2). In the crystalline rock, which is considered the basic starting unit of this rock, all the minerals become unstable within the earth's surface conditions and are replaced by clays (secondary minerals), and ions are lost to the percolating aqueous solution.

Rock

The zone in which water–rock interactions begin is the initial unaltered rock. All rocks contain many cracks and fissures caused by the release of thermal and pressure constraints produced at depth. These cracks and fissures are the initial paths of penetration of the altering fluid. In the first stages of weathering, zones near cracks show intense alteration, whereas the zones furthest from the cracks remain unaltered. In the upper horizon of the bedrock, initial formation of clay minerals is localized around the initial cracks in the rock. Thus, in the initial stages of weathering, a rock sample is heterogeneously altered.

Saprock

The first zone of generalized alteration is called saprock. In this zone, the minerals are partially or totally transformed into clays (secondary minerals), but the petrographic structure of the rock (i.e., the location and size of the initial mineral grains) is largely maintained. The overall quantity of clays is greatly increased over that of the basement rock through the alteration and transformation of the original minerals. The most important feature of this zone is the

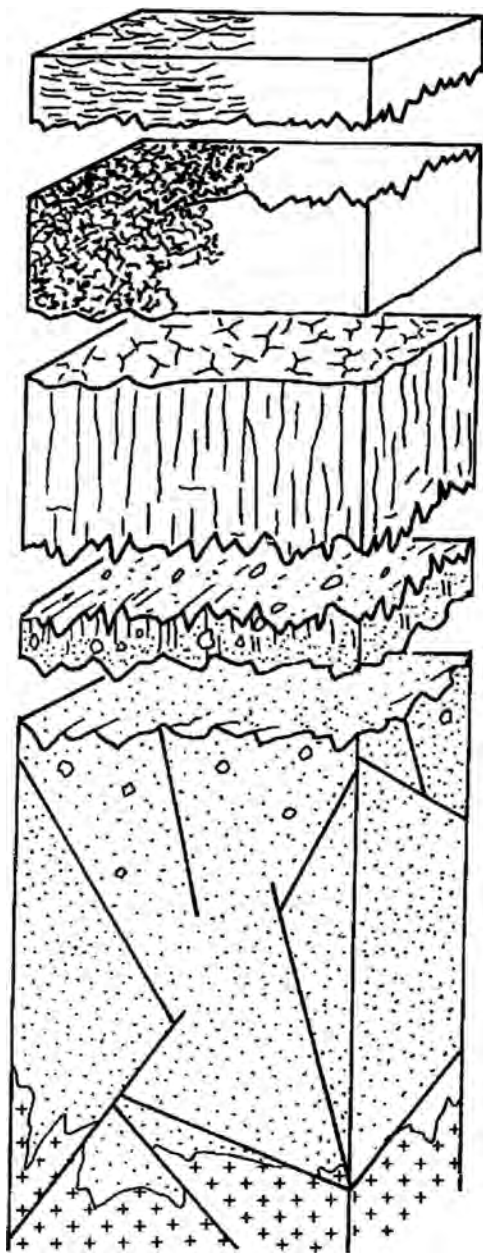


Fig. 2 Schematic representation of a weathering profile showing the successive zones or horizons.

Source: From Righi & Meunier.^[4]

extreme heterogeneity of the mineralogy, with many types of clays and large portions of unaltered or little-altered rock minerals.

Saprolite

The next zone is that of saprolite, where there is little or no rock structure left and clays predominate. The effect of chemical transformation is the greatest here; most of the original minerals are greatly altered. Due to a high degree of mineral transformation in the upper part of the saprolite

zone, the rock structure often collapses, and the initial fractures and fissures are lost.

Organo-Mineral Soil

The uppermost weathering zone commonly contains the soil. This is where, in a natural and undisturbed state, the biogeochemical processes become predominant. The chemical and physical properties of the system are dominated by pore structure. This is a secondary structure due to the weathering transformations which have totally effaced any rock structure. When porosity is sufficiently advanced to allow the establishment of organisms, their secretions and breakdown products induce a more rigorous weathering and the formation of soil type characteristics of the prevailing environmental conditions and vegetation.

BASIC FACTORS IN WEATHERING

Weathering is the result of a complex set of processes, which is controlled by a set of interacting factors. Soil-forming factors, including rock material, climate, vegetation, topography, and time, are relevant for weathering.^[5] In some cases, human activity disturbs natural soils so much that it could be considered an additional factor of soil formation.

Four variables govern weathering processes.^[6] The first is *rock type*, which is a chemical factor. The second, *climate*, is composed of rainfall (a chemical factor) and temperature (a physical factor). The third, *flow rate*, is a chemical factor because it determines, through drainage, the ratio of aqueous phase to rock and its contact time with rock. The fourth variable is the *age* of a weathering sequence, which is a physical parameter (time). The chemical factors of rock composition and flow rate are probably the most important in determining the type of secondary minerals (clay minerals) that will be produced.

On the *continental scale*, flow rate is determined by climate. In free-flowing water systems, drainage can be considered the driving force of weathering. Removing the leachate creates a mass action effect favoring the breakdown of the primary phase minerals. In humid zones, where drainage is free, such subtractive weathering processes lead to the accumulation of Si, Al, and Fe, and the elimination of all other major cations. Kaolinite (an Al-rich clay mineral), gibbsite (Al hydroxide), and goethite and hematite (Fe oxyhydroxide and oxide) are the secondary minerals that form. Thick weathering profiles (20 m or more) characterized by this simple clay mineral assemblage, often called laterite,^[7] cover much of the intertropical zone. Under drier conditions, the aqueous phase may develop high ionic concentrations, leading to the precipitation of salts and/or formation of smectite or fibrous clays.

On the *watershed scale*, flow rate is determined by slope gradient. Similar to the effect of climate, the slope controls

the water residence time and, hence, the chemistry of the reactions. The greater the slope, the more rapid the displacement of the aqueous solutions over the weathering zone. Moreover, material leached out of the upper part of the slope may deposit in the lower part. Therefore, the general type of secondary minerals changes with the slope. In the upper areas where water residence time is short, oxides of Fe and Al tend to form. As one proceeds down-slope, kaolinite is more abundant and, in the lower regions, minerals such as smectites form.

On the *rock fragment* or *mineral scale*, porosity induces similar effects. In the larger pores of the system (1 mm or more in diameter), where time of contact between minerals and aqueous phase is short, kaolinite or Al and Fe oxides may form. In smaller pores (<1 μm diameter), longer residence time for the aqueous phase leads to higher concentrations of cations, which explains why smectites form at grain boundaries in solid rocks. In these systems of low water–rock ratio and long contact time, the chemical composition of the rock or mineral is very important in determining the secondary minerals (type of clay minerals) that will form. Acid rocks having a large proportion of potassium feldspar and quartz will produce kaolinite, aluminous smectite, or illite (mica-like) clays. On the other hand, basic and ultrabasic rocks, which contain a large proportion of ferromagnesian minerals, will produce saponite, vermiculite, and chlorites (Mg- and/or Fe-rich clays) as weathering products. The parent rock effect is greater during the early stages of weathering or pedogenetic evolution. As weathering progresses (time increases), chemical leaching removes all but the most refractory components.

CONCLUSION

Clay minerals are the principal products of weathering and soil formation, which are essentially water–rock interaction

processes. Clay formation occurring at the earth's surface is a *slow reaction process*, meaning that some initial materials are only partially reacted and are not in equilibrium with the medium in which they are located. This creates the potential for further transformation of altered products. If the chemistry of the ambient soil or surface materials is changed, e.g., by pollution containing chemicals, there will be a high potential for change in the clay mineralogy of the surface material. Thus, the type of clay minerals produced by weathering reactions and their rate of formation are of the greatest importance in environmental problems.

REFERENCES

1. Chesworth, W. Weathering systems. In *Weathering, Soils & Paleosols*; Martini, I.P., Chesworth, W., Eds.; Developments in Earth Surface Processes II; Elsevier: Amsterdam, 1992; 19–40.
2. Macias, F.; Chesworth, W. Weathering in humid regions, with emphasis on igneous rocks and their metamorphic equivalent. In *Weathering, Soils & Paleosols*; Chesworth, W., Martini, I.P., Eds.; Developments in Earth Surface Processes II; Elsevier: Amsterdam, 1992; 283–306.
3. Pédro, G. Caractérisation générale des processus de l'altération hydrolytique. Base des méthodes géochimique et thermodynamique. *Sci. Du Sol.* **1979**, 2 (3), 93–105.
4. Righi, D.; Meunier, A. Origin of clays by rock weathering and soil formation. In *Origin and Mineralogy of Clays, Clays and the Environment*; Velde, B., Ed.; Springer-Verlag: Berlin, 1995; 43–161.
5. Jenny, H. *Factors of Soil Formation*; McGraw-Hill: New York, 1941; 281 pp.
6. Velde, B. Origin of clays. In *Introduction to Clay Minerals*; Chapman & Hall: London, 1992; 101–154.
7. Tardy, Y. Diversity and terminology of lateritic profiles. In *Weathering, Soils & Paleosols*; Martini, I.P., Chesworth, W., Eds.; Developments in Earth Surface Processes II; Elsevier: Amsterdam, 1992; 379–405.

Weathering: Carbon Sequestration

William Berry Lyons

Ohio State University, Columbus, Ohio, U.S.A.

Jerry M. Bigham

School of Environment and Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Anne E. Carey

Ohio State University, Columbus, Ohio, U.S.A.

Rattan Lal

Carbon Management and Sequestration Center, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

The chemical weathering of geologic materials by carbonic acid, including both aluminosilicate and carbonate minerals, converts carbon dioxide (CO₂) to bicarbonate/carbonate ions that are later precipitated as carbonate minerals in soils and sediments, thus removing and sequestering CO₂ from the atmosphere. Over periods of geological time, this process is a major control on the partial pressure of CO₂ in the atmosphere. Chemical weathering rates, and hence CO₂ consumption, are influenced by a variety of geological, climatological, and biological factors, such as temperature, precipitation, vegetation cover, bedrock type, depth of the regolith, and tectonic activity. Further work has demonstrated a significant relationship between chemical and physical weathering and a tight coupling between tectonic processes and chemical weathering. In addition to the direct consumption of CO₂ by the weathering reaction, the chemical weathering is the major process releasing soluble nutrient elements such as potassium, calcium, and phosphorus (P) to ecosystems. The production of these nutrients helps promote biomass growth and development, which in turn sequesters CO₂ by photosynthesis. Because rates of weathering and therefore rates of base cations and P production in soils are influenced by both climatic and tectonic regimes, an important linkage between geological processes and ecosystem development exists.

INTRODUCTION

Chemical weathering is the chemical alteration of minerals into new minerals and/or dissolved ions. The reaction involves water and/or an acid. Although various anthropogenic activities, including fossil-fuel burning, have introduced compounds such as NO_x and sulfur dioxide into the atmosphere and these components can react with water to form strong acids, namely nitric acid and sulfuric acid (i.e., “acid rain”), the primary acid consumed in the chemical weathering of both carbonate and aluminosilicate minerals is carbonic acid (H₂CO₃). Upon the dissociation of H₂CO₃, the hydrogen ions produced to attack the minerals present in soil, and chemical weathering occurs. If carbonate minerals such as calcium carbonate (CaCO₃) are present, congruent weathering takes place and the entire mineral is dissolved (CaCO₃ + H₂CO₃ → Ca²⁺ + 2HCO₃⁻). Most aluminosilicate minerals weather incongruently, such that both soluble ions and an insoluble weathering product, a clay mineral, are produced

(2KAlSi₃O₈ + 9H₂O + 2H₂CO₃ → 2K⁺ + 2HCO₃⁻ + 4H₄SiO₄ + Al₂Si₂O₅(OH)₄). In this reaction, 2 moles of potassium ion (K⁺) and bicarbonate are produced along with 4 moles of silicic acid (dissolved Si) for every 2 moles of K feldspar consumed. In addition, 1 mole of insoluble kaolinite clay is left behind in the soil.

Because the carbon dioxide (CO₂) in H₂CO₃ originates from the atmosphere (either directly from the atmosphere or indirectly from the root respiration or decomposition of plant materials), the consumption of CO₂ via chemical weathering is recognized to be a major control or “sink” for CO₂ over time and hence an important driver of climate change. In terrestrial systems, two major processes sequester atmospheric CO₂ as follows: the photosynthetic production of organic carbon (C) and its subsequent burial and the chemical weathering.^[1] Over long periods of geological time, the chemical weathering of aluminosilicate minerals is a primary control of CO₂, while on shorter timescales (few thousand years), the weathering of carbonate minerals also

serves as a “sink” for CO_2 .^[2] The CO_2 consumption rate via weathering is $\sim 2.4 \times 10^{13} \text{ mol C yr}^{-1}$.^[3] Of this, 51% is from carbonate mineral weathering. Empirical data from both available studies and the geological past indicate that chemical weathering rates are indeed directly proportional to the amount of CO_2 present in the system.^[4,5] See also the entry *Clay Minerals: Weathering and Alteration of*, p. 430.

WEATHERING RATES AND CO_2 CONSUMPTION

All the minerals do not weather at the same rate. Minerals formed under conditions furthest removed from earth surface conditions [i.e., high tritium and phosphorus (P)] weather the fastest. Therefore, the processes that formed these minerals and their chemical composition have important influences in the rate of chemical weathering at any given location. Olivine group minerals $[(\text{Mg,Fe})\text{SiO}_4]$, which are found in many volcanic rocks, are among the least stable at surface conditions, while quartz found, e.g., in sandstones is one of the most stable and most resistant to weathering.^[4] In general, continental igneous rocks with more felsic minerals weather at about half the rate of volcanic rocks with mafic minerals.^[6] In the earth's history, the chemical weathering of volcanic rocks could have accounted for more than 80% of the total chemical weathering of the continents.^[1] Because of this, volcanic rocks are an important component to overall global aluminosilicate weathering and the consumption of CO_2 .^[6]

Lasaga^[7] compared the dissolution rates of several silicate minerals at 25°C and pH 5 and was able to calculate the time required for complete dissolution of crystals 10^{-3} m in size. The result for forsterite (Mg_2SiO_4), an olivine group mineral, was 600,000 years compared with 2.7 million years of muscovite and 34 million years of quartz. Appreciable differences often exist when rates of mineral weathering obtained from laboratory experiments are compared with rates estimated from field data. Discrepancies are to be expected given that many factors help control the rate of weathering, and hence CO_2 consumption. These include climate and hydrology, depth of the weathering profile, abundance of vascular plants, and whether the landscape is geologically active.

The last of those processes is referred to in geological nomenclature as tectonics. Tectonics implies, in part, the uplift of geological material and occurs primarily at convergent crustal plate boundaries. The uplift of crust to form mountains leads to increased erosion rates as relief is increased (and in many cases, increased orogenic precipitation occurs), so that a dynamic balance is reached between these two processes when a stable topography is achieved.^[8] Thus, chemical and physical weathering rates are closely coupled.^[9] Modeling indicates that the chemical weathering in continental crustal lithologies such as granite can range over more than an order of magnitude from tectonically

active regions to those that are geologically stable.^[8] These regions of high uplift rates include high-standing islands in the Western Pacific Ocean such as New Zealand and Taiwan that have very high CO_2 consumption rates per unit area via chemical weathering.^[9] This linkage between physical erosion and chemical weathering suggests that regions where rapid erosion occurs can be important regions of CO_2 consumption through the weathering process. Numerous authors have demonstrated the importance of vascular plants in enhancing chemical weathering as well as their overall role in C dynamics.^[10]

In addition to CO_2 sequestration through direct consumption by the weathering process, the chemical weathering provides soluble nutrients such as K, calcium (Ca), and P in soils that promote and enhance plant growth. This solubilization of nutrients by chemical weathering indirectly aids in C sequestration by aiding in the increase of ecosystem biomass through natural fertilization.^[8] In this regard, the production of nutrient elements is especially rapid in the regions of active tectonic uplift or new crustal production such as volcanism.^[11,12] In these tectonically active areas, fresh minerals are continuously supplied at a rapid rate, thereby exerting important influence in ecosystem development.^[12] In forested ecosystems in Taiwan, only ~9% of the P solubilized by chemical weathering is lost via stream discharge, as the rest is retained within the ecosystem.^[12] On the other hand, the atmosphere is the major source of fixed nitrogen (N) to the landscape. The N/P ratio of the inputs to the Taiwanese forests is 6:1, thus maintaining the ecosystem in a probable state of N deficiency, as predicted by the model of Vitousek.^[11] In areas where there is little tectonic activity, P and other essential nutrients like base cations can be rapidly (a few thousand years) weathered from rocks and soils, leaving them deficient in these important components.

WEATHERING OF CARBONATE MINERALS

CaCO_3 dissolves congruently upon reaction with H_2CO_3 . Although this reaction initially consumes CO_2 , after transfer to the oceans, CaCO_3 is eventually formed and CO_2 is produced via the back reaction. Thus, on geological timescales, carbonate mineral weathering has not been considered an ultimate sink for CO_2 .^[13] The widespread addition of lime to agricultural soils has led to significant carbonate mineral dissolution on agricultural lands, and hence enhanced CO_2 sequestration and HCO_3^- input to the oceans.^[14]

CHEMICAL WEATHERING AND CO_2 OVER GEOLOGICAL TIME

Modeling calculations, based in part on chemical weathering constraints, indicate that the CO_2 concentration of the earth's atmosphere has varied greatly over the past 540 million years. Values as high as 15 to 20 times

compared with pre-anthropogenic times occurred ~520 million years ago, and existing values compared with pre-anthropogenic times occurred between ~350 million years and 260 million years ago.^[6] These modeling exercises suggest that a CO₂ threshold below ~500 ppm is needed for the initiation of widespread glacial conditions and that a tight relationship between global temperature and atmospheric CO₂ concentration existed throughout the last 540 million years of earth's history.^[1]

SOIL CARBONATES

Meteoric water falling on exposed land is the primary agent of weathering reactions. Baumgartner and Reichel^[15] reported that the average annual rainfall on exposed lands is 74.6 cm yr⁻¹ but noted that only 36% (26.6 cm yr⁻¹) actually percolates through the soil to the groundwater. Thus, huge variations in "effective" percolation occur as a function of climate and relief. In dry regions, atmospheric CO₂ may be sequestered through the formation of pedogenic carbonates, especially when Ca is derived by the weathering of aluminosilicate minerals. Pedogenic carbonates occur in different morphological forms as needles, laminated layers, or precipitates in interstitial spaces and as coatings on the underside of stones. They also occur as threads, concretions (glebules), and pendants below peds and around roots.^[16] There are four principal mechanisms for the formation of pedogenic carbonates:^[17]

1. The per descendum model: carbonates are precipitated in the subsoil following dissolution of preexisting carbonates or aluminosilicates in the upper soil layers followed by the vertical translocation of soluble elements to the subsoil with the local wetting front.
2. The per ascendum model: the capillary rise of Ca-carrying water from a shallow water table leads to the deposition of pedogenic carbonates in the upper soil layers.
3. The *in situ* model: there are *in situ* dissolution and reprecipitation of carbonates in proximity to the parent material or bedrock.
4. The biogenic model: secondary carbonates are formed aided by biological factors (e.g., termites, microbes, and plant roots).

The rate of formation of carbonates in some soils of the United States and Canada ranges from 1.5 to 21.5 kg C ha⁻¹ yr⁻¹ (Table 1). Ryskov et al.^[18] estimated that Chernozems and chestnut soils of Transbaikalia, Russia, bound 40–70 and 32 kg C ha⁻¹ yr⁻¹, respectively. Chen^[19] reported that in Australia, 0.5–1.0 m deep gypcrete horizons may form in 35,000–100,000 years. While the rates of formation of pedogenic carbonates are low at a decadal scale in comparison with those of sequestration by organic C as humus (100–1500 kg C ha⁻¹ yr⁻¹), the formation of

Table 1 Rate of C sequestration as pedogenic carbonates in North America.

Location	C sequestration rate (kg C ha ⁻¹ yr ⁻¹)
Mormon Mesa, Nevada	2.6–16.3
Whipple, California	1.5
Avra Valley, Arizona	5.8–8.5
Rio Grande Valley, New Mexico	1.5–18.0
Utah, New Mexico	1.7–6.1
Mojave Desert	1.2–4.2
Saskatchewan	12.5–21.5

Source: Recalculated from *Carbonates: Secondary and Pedogenic* (p. 335), Landi, Mermut, et al.,^[20] Machette,^[21] and Schlesinger.^[22]

pedogenic carbonates is an important mechanism of binding atmospheric CO₂ at centennial and millennial scales in dry regions.

REFERENCES

1. Berner, R.A. GEOCARBSULF: A combined model for Phanerozoic atmospheric O₂ and CO₂. *Geochim. Cosmochim. Ac.* **2006**, *70*, 5653–5664.
2. Berner, R.A.; Kothavala, Z. GEOCARB III: A revised model of atmospheric CO₂ over Phanerozoic time. *Am. J. Sci.* **2001**, *301*, 182–204.
3. Gaillardet, J.; Dupré, B.; Lourat, P.; Allègre, C.J. Global silicate weathering and CO₂ consumption rates reduced from the chemistry of large rivers. *Chem. Geol.* **1999**, *159*, 3–30.
4. Navarre-Sitcher, A.; Thyne, G. Effects of carbon dioxide on mineral weathering rates at earth surface conditions. *Chem. Geol.* **2007**, *243*, 53–63.
5. Smith, M.E.; Carroll, A.R.; Mueller, E.R. Elevated weathering rates in the Rocky Mountains during the early Eocene climate optimum. *Nat. Geosci.* **2008**, *1*, 370–374.
6. Berner, R.A. Inclusion of the weathering of volcanic rocks in the GEOCARBSULF model. *Am. J. Sci.* **2006**, *306*, 295–302.
7. Lasaga, A.C. Chemical kinetics of water-rock interactions. *J. Geophys. Res.* **1984**, *89* (B6), 4009–4025.
8. Waldbauer, J.R.; Chamberlain, C.P. Influence of uplift, weathering and base cation supply on past and future CO₂ levels. In *A History of Atmospheric CO₂ and Its Effects in Plants, Animals and Ecosystems*; Ehleringer, J.R., Cerling, T.E., Dearing, M.D., Eds.; Springer: New York, 2005; 166–184.
9. Lyons, W.B.; Carey, A.E.; Hicks, D.M.; Nezat, C.A. Chemical weathering in high-sediment-yielding watersheds, New Zealand. *J. Geophys. Res.* **2005**, *110*, F01008.
10. Moulton, K.L.; West, J.; Berner, R.A. Solute flux and mineral mass balance approaches to the quantification of plant effects in silicate weathering. *Am. J. Sci.* **2000**, *300*, 539–570.

11. Vitousek, P. *Nutrient Cycling and Limitation: Hawaii as a Model System*; Princeton University Press: Princeton, 2004; 223 pp.
12. Carey, A.E.; Lyons, W.B.; Owen, J.S. Significance of landscape age, uplift and weathering rates to ecosystem development. *Aquat. Geochem.* **2005**, *11*, 215–239.
13. Raymond, P.A.; Oh, N.-H.; Turner, R.E.; Broussard, W. Anthropogenically enhanced fluxes of water and carbon from the Mississippi River. *Nature* **2008**, *451*, 449–452.
14. West, T.O.; McBride, A.C. The contribution of agricultural lime to carbon dioxide emissions in the United States: Dissolution, transport and net emissions. *Agric. Ecosyst. Environ.* **2005**, *108*, 145–154.
15. Baumgartner, A.; Reichel, E. *The World Water Balance. Mean Annual Global, Continental, and Maritime Precipitation, Evaporation and Run-Off*; Elsevier: Amsterdam, 1975; 179 pp.
16. Schaetzl, R.J.; Frederick, W.E.; Tornes, L. Secondary carbonates in three fine and fine-loamy alfisols in Michigan. *Soil Sci. Soc. Am. J.* **1996**, *60*, 1862–1870.
17. Monger, H.C. Pedogenic carbonates: Links between biotic and abiotic CaCO_3 . In 17th World Congress of Soil Science Symposium #20, Oral Paper #891, Bangkok, Thailand, Aug 14–21, 2002.
18. Ryskov, Y.G.; Tsybzhitov, T.K.; Tsybikodorzhev, T.T.; Oleinik, S.A.; Ryskova, E.A. Russia's soil: Is it a CO_2 source or sink? *Geochem. Int.* **2001**, *39*, 577–584.
19. Chen, X.-Y. Pedogenic gypcrete formation in arid central Australia. *Geoderma* **1997**, *77*, 39–61.
20. Landi, A.; Mermut, A.R.; Anderson, D.W. Origin and rate of pedogenic carbonate accumulation in Saskatchewan soils, Canada. *Geoderma* **2003**, *117*, 143–156.
21. Machette, M.N. Calcic soils of the southwestern U.S. In *Soils of Quaternary Geomorphology of Southwestern United States*; Special Paper, Vol. 203; Weidl, D.L., Ed.; Geology Society of America: Boulder, 1985; 1–21.
22. Schlesinger, W.H. The formation of caliche in soils of the Mohave desert, California. *Geochim. Cosmochim. Ac.* **1985**, *49*, 57–66.

Wetlands

Ralph W. Tiner

U.S. Fish and Wildlife Service (FWS), Hadley, Massachusetts, U.S.A.

Abstract

Wetland is a universal term used to describe the collection of flooded or saturated environments that have been referred to as marshes, swamps, bogs, fens, salinas, pocosins, mangroves, wet meadows, sumplands, salt flats, varzea forests, igapo forests, bottomlands, sedgeland, moors, mires, potholes, sloughs, mangals, palm oases, playas, muskegs, and other regional and local names. It has been defined as a basis for inventorying these natural resources, for conducting scientific studies, and, in some countries, for regulating uses of these areas. Given that wetlands include a diverse assemblage of ecosystems, classification schemes have been developed to separate and describe these different systems and to group similar habitats. Wetlands provide a number of functions that are considered valuable to society (e.g., surface water storage to minimize flood damages, sediment retention and nutrient transformation to improve water quality, shoreline stabilization, streamflow maintenance, and the provision of vital habitat for fish, shellfish, wildlife, and plants that yield food and fiber for people). Because of these values and the widespread recognition of wetlands as important natural resources, numerous wetland definitions and classification systems have been developed to inventory these resources around the globe. The purpose of this entry is to provide readers with an understanding of what wetlands are (wetland definition), how they vary globally (wetland types), and their extent as determined by various inventories. This entry should serve as a starting point for learning about wetlands, with the listed references being sources of more detailed information.

WETLAND DEFINITIONS

Wetlands are aquatic to semiaquatic ecosystems where permanent or periodic inundation or prolonged waterlogging creates conditions favoring the establishment of aquatic life. Wetlands are often located between land and water and have, therefore, been referred to as ecotones (i.e., transitional communities). However, many wetlands are not ecotones between land and water since they are not associated with a river, lake, estuary, or stream.^[1] Wetlands may derive water from many sources, including groundwater, river overflow, surface water runoff, precipitation, snowmelt, tides, melting permafrost, and seepage from impoundments or irrigation projects.

While the term “wetland” has many definitions, all definitions have common elements (see [Table 1](#); some definitions even include deepwater habitats). The presence of water in wetlands may be permanent or temporary. Their water may be salty or fresh. Wetlands may be natural habitats or artificially created. They range from shallow water environments to temporarily wet (i.e., flooded or saturated) areas. All are wet long enough and often enough to, at least, periodically support hydrophytic vegetation and other aquatic life (including anaerobic microbes), to create hydric soils or substrates, and to activate biogeochemical processes associated with wet environments.

WETLAND TYPES

Differences in climate, soils, vegetation, hydrology, water chemistry, nutrient availability, and other factors have led to the formation of a multitude of wetland types around the globe. In general, wetlands are characterized by their hydrology (e.g., tidal vs. non-tidal, inundation vs. soil saturation, and frequency and duration of wetness), the presence or absence of vegetation (vegetated vs. non-vegetated), the type of vegetation (forested or treed, shrub, emergent, or aquatic bed), and soil type (e.g., organic vs. inorganic and peatland vs. non-peatland). [Table 2](#) presents brief descriptions of some North American types, and [Fig. 1](#) shows examples of vegetated wetlands.

Various countries have devised classification systems for describing differences among their wetlands and for categorizing wetlands for natural resources inventories. Scientists have created systems to organize certain wetlands into meaningful groups for analysis and management (e.g., peatland classifications; see the study by Tiner^[1] for details). The Ramsar Convention Bureau^[2] published a multinational classification system to provide consistency for inventorying wetlands and designating wetlands of international importance. This system includes 11 types of marine or coastal wetlands (i.e., shallow water and intertidal habitats: permanent shallow marine waters; marine subtidal aquatic beds; coral reefs;

Table 1 Examples of wetland definitions used for inventories.

Country/organization	Wetland definition (source)
International/Ramsar	“areas of marsh, fen, peatland, or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish, or salt, including areas of marine water the depth of which at low tide does not exceed 6 m ... may incorporate riparian and coastal zone adjacent to wetlands, and islands or bodies of marine water deeper than 6 m at low tide lying within the wetlands.” ^[2]
Australia	“areas of seasonally, intermittently, or permanently waterlogged soils or inundated land, whether natural or artificial, fresh or saline, e.g., waterlogged soils, ponds, billabongs, lakes, swamps, tidal flats, estuaries, rivers, and their tributaries.” ^[3]
Canada	“land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation, and various kinds of biological activity which are adapted to a wet environment.” ^[4]
The United States	“lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water.” Wetland attributes include hydrophytic vegetation, undrained hydric soil, or saturated or flooded substrates. ^[5]

rocky marine shores; sand; shingle or pebble shores; estuarine waters; intertidal mud, sand, or salt flats; intertidal marshes; intertidal forests; coastal brackish/saline lagoons; and coastal freshwater lagoons). This system also includes 19 inland wetland types (i.e., permanently flooded aquatic habitats to intermittently flooded sites are represented: permanent inland deltas; permanent rivers/streams/creeks; seasonal, intermittent, or irregular rivers/streams/creeks; permanent freshwater lakes; seasonal or intermittent freshwater lakes; seasonal or intermittent saline/brackish/alkaline lakes and flats; permanent saline/brackish/alkaline marshes and pools; seasonal or intermittent saline/brackish/alkaline marshes and pools; permanent freshwater marshes and pools; seasonal or intermittent freshwater marshes and pools; non-forested

peatlands; alpine wetlands including meadows and temporary snowmelt waters; tundra wetlands; shrubby-dominated wetlands; freshwater tree-dominated wetlands on inorganic soils; forested peatlands; freshwater springs and oases; geothermal wetlands; and subterranean karst and cave hydrological systems). Finally, nine man-made wetland types (aquaculture ponds; ponds; irrigated land including rice paddies; seasonally flooded agricultural land; salt exploitation sites; water storage areas including impoundments generally more than 8 ha; excavations; wastewater treatment areas; and canals and drainage channels) are also included in the system.

In North America, the Canadian and U.S. wetland classification systems were developed by government agencies interested in wetland conservation and management. The

Table 2 Brief non-technical descriptions of some wetland types in North America.

Wetland type	General description
Marsh	Herb-dominated wetland with standing water through all or most of the year, often with organic (muck) soils
Tidal marsh	Herb-dominated wetland subject to periodic tidal flooding
Salt marsh	Herb-dominated wetland occurring on saline soils, typically in estuaries and interior arid regions
Swamp	Wetland dominated by woody vegetation and usually wet for extended periods during the growing season
Mangrove swamp (Mangal)	Tidal swamp dominated by mangrove species
Peatland, mire, moor, or muskeg	Peat-dominated wetland
Bog	Nutrient-poor peatland, typically characterized by ericaceous shrubs, other woody species, and peat mosses
Fen	More or less nutrient-rich peatland, often represented by sedges and/or calciphilous herbs and woody species
Wet meadow	Herb-dominated wetland that may be seasonally flooded or saturated for extended periods, often with mineral hydric soils
Bottomland	Riverside or streamside wetland, usually on floodplain
Flatwood	Forested wetland with poorly drained mineral hydric soils located on broad flat terrain of interstream divides, common on coastal plains and glaciolacustrine plains
Farmed wetland	Wetland cultivated for rice, cranberries, sugarcane, mints, or other crops

Note: These types may be defined differently in other regions.

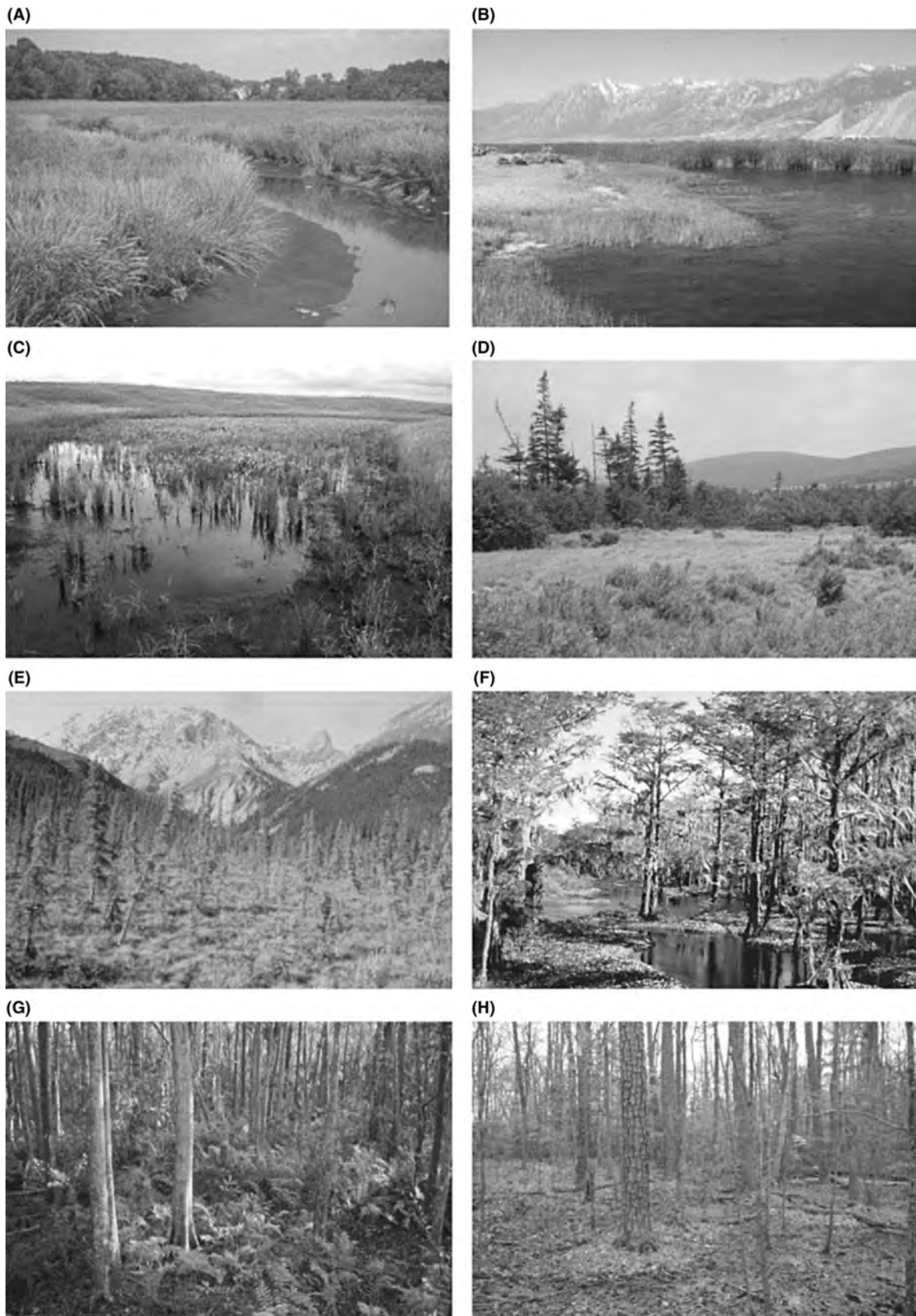


Fig. 1 Some examples of North American wetlands: (A) tidal salt marsh, (B) inland marsh, (C) pothole marsh, (D) wet meadow/shrub swamp, (E) northern peatland, (F) bottomland swamp, (G) hardwood swamp, and (H) flatwood wetland.

Source: Photographs courtesy of U.S. Fish and Wildlife Service.

Canadian system emphasizes wetland origin (class), form, and vegetation in describing the wetland types.^[4] Five wetland classes are recognized as follows: bog, fen, marsh, swamp, and shallow water. Within each class, different forms and types are characterized. Eight general vegetation types are defined by the presence or absence of vegetation (treed, shrub, forb, graminoid, moss, lichen, aquatic bed, and non-vegetated). These types may be subdivided into other types (e.g., treed into coniferous or deciduous; shrub into tall, low, and mixed; graminoids into grass, reed, tall rush, low rush, and sedge; and aquatic bed into floating and submerged). The U.S. Fish and Wildlife Service's wetland and deepwater habitat classification^[5] is the official federal system used for mapping wetlands and for reporting the status and trends of wetlands in the United States. The features separating wetlands include general ecological and physical factors and specific features such as vegetation, soil/substrate composition, hydrology, water chemistry, and human alterations. Classification follows a hierarchical approach with five main levels designated as follows: ecological system (marine, estuarine, lacustrine, riverine, and palustrine), subsystem, class (vegetated: forested, scrub-shrub, emergent, and aquatic bed; non-vegetated: unconsolidated shore, rocky shore, streambed, and reef), subclass, and modifiers. The modifiers are used to describe a wetland's hydrology (water regime), pH and salinity (water chemistry), soils, and the influence of humans and beaver (special modifiers). Common types include estuarine intertidal emergent wetlands (e.g., salt and brackish marshes), estuarine intertidal unconsolidated shore (e.g., tidal flats and beaches), palustrine emergent wetlands (e.g., marshes, fens, and wet meadows), palustrine forested

wetlands, and palustrine scrub-shrub wetlands (e.g., shrub bogs and shrub swamps).

A hydrogeomorphic approach (HGM) to wetland classification has also been developing in the United States.^[16] The HGM system emphasizes the abiotic features important for assessing wetland functions. Seven hydromorphic classes are identified as follows: riverine, depressional, slope, mineral soil flats, organic soil flats, lacustrine fringe, and estuarine fringe. The U.S. Fish and Wildlife Service has adapted the HGM approach to provide additional modifiers to its classification system on a pilot basis. These HGM-type descriptors include landscape position (i.e., lotic, lentic, terrene, estuarine, and marine), landform (i.e., slope, basin, interfluvium, floodplain, flat, island, and fringe), and water flow path (i.e., inflow, outflow, throughflow, bidirectional flow, isolated, and paludified).^[17] These descriptors provide the required information to aid the evaluation of functions of wetlands across watersheds and large geographic areas.

EXTENT OF WETLANDS

Comprehensive wetland inventories do not exist in most countries. There are many inconsistencies among the inventories (e.g., different levels of effort, focus on particular types, and artificial wetlands such as rice paddies are often not included in wetland inventories).^[18] Consequently, comparative analysis is fraught with problems. Nonetheless, Table 3 provides some perspective on the extent of wetlands in many regions. Most of the data came from a series of reports produced for the Bureau of the Ramsar Wetlands Convention.^[6] Globally, estimates for wetlands range from about 750 million ha^[18] to about 1.5 billion ha. Ten countries have over 2 million ha of peatlands alone, with Canada leading at nearly 130 million ha (represents about 18% of the country) followed by the former U.S.S.R. at 83 million ha.^[19,20] About a third of Finland is covered by peatlands (10 million ha). The Pantanal of South America, perhaps the largest wetland in the world, reportedly covers about 200,000 km² (or 2 million ha) during the wet season.^[21]

Table 3 Estimates of the extent of wetlands in different regions of the world.

Region/country	Wetland extent (ha)	References
Africa	121,321,683–124,686,189	[6]
Asia	211,501,790–224,117,790	[7]
Central America		
Mexico	3,318,500 (very incomplete)	[8]
Europe		
Eastern	225,849,930	[9]
Western	28,821,979	[10]
Middle East	7,434,790	[11]
Neotropics	414,996,613	[12]
North America		
Canada	127,199,000–150,000,000	[13]
The United States	114,544,800	[1]
Oceania	35,748,853	[14]
South America		
Tropical region	200,000,000	[15]

REFERENCES

1. Tiner, R.W. *Wetland Indicators: A Guide to Wetland Identification, Delineation, Classification, and Mapping*; Lewis Publishers, CRC Press: Boca Raton, 1999; 392 pp.
2. Ramsar Convention Bureau. *Information Sheet on Ramsar Wetlands*; Gland: Switzerland, 1997; 170.
3. Wetland Advisory Committee. *The Status of Wetlands Reserves in System Six*; Report of the Wetland Advisory Committee to the Environmental Protection Authority: Australia, 1977.
4. National Wetlands Working Group. *Wetlands of Canada*; Ecological Land Classification Series No. 24; Land Conservation Branch, Canadian Wildlife Service; Environment Canada: Ottawa, 1988; 452 pp.

5. Cowardin, L.M.; Carter, V.; Golet, F.C.; LaRoe, E.T. *Classification of Wetlands and Deepwater Habitats of the United States*; FWS/OBS-79/31; U.S. Department of the Interior, Fish and Wildlife Service: Washington, 1979; 131 pp.
6. Stevenson, N.; Frazier, S. Review of wetland inventory information in Africa. In *Global Review of Wetland Resources and Priorities for Wetland Inventory*; Finlayson, C.M., Spiers, A.G., Eds.; Supervising Scientist Report 144: Canberra, 1999; 94 pp.
7. Watkins, D.; Parish, F. Review of wetland inventory information in Asia. In *Global Review of Wetland Resources and Priorities for Wetland Inventory*; Finlayson, C.M., Spiers, A.G., Eds.; Supervising Scientist Report 144: Canberra, 1999; 26 pp.
8. Olmstead, I. Wetlands of Mexico. In *Wetlands of the World: Inventory, Ecology, and Management Volume I*; Whigham, D.F., Dykyjova, D., Heiny, S., Eds.; Kluwer Academic Publishers: Dordrecht, 1993; 637–677.
9. Stevenson, N.; Frazier, S. Review of wetland inventory information in Eastern Europe. In *Global Review of Wetland Resources and Priorities for Wetland Inventory*; Finlayson, C.M., Spiers, A.G., Eds.; Supervising Scientist Report 144: Canberra, 1999; 53 pp.
10. Stevenson, N.; Frazier, S. Review of wetland inventory information in Western Europe. In *Global Review of Wetland Resources and Priorities for Wetland Inventory*; Finlayson, C.M., Spiers, A.G., Eds.; Supervising Scientist Report 144: Canberra, 1999; 57 pp.
11. Frazier, S.; Stevenson, N. Review of wetland inventory information in the Middle East. In *Global Review of Wetland Resources and Priorities for Wetland Inventory*; Finlayson, C.M., Spiers, A.G., Eds.; Supervising Scientist Report 144: Canberra, 1999; 19 pp.
12. Davidson, I.; Vanderkam, R.; Padilla, M. Review of wetland inventory information in the neotropics. In *Global Review of Wetland Resources and Priorities for Wetland Inventory*; Finlayson, C.M., Spiers, A.G., Eds.; Supervising Scientist Report 144: Canberra, 1999; 35 pp.
13. Davidson, I.; Vanderkam, R.; Padilla, M. Review of wetland inventory information in the North America. In *Global Review of Wetland Resources and Priorities for Wetland Inventory*; Finlayson, C.M., Spiers, A.G., Eds.; Supervising Scientist Report 144: Canberra, 1999; 35 pp.
14. Watkins, D. Review of wetland inventory information in Oceania. In *Global Review of Wetland Resources and Priorities for Wetland Inventory*; Finlayson, C.M., Spiers, A.G., Eds.; Supervising Scientist Report 144: Canberra, 1999; 26 pp.
15. Junk, W.J. Wetlands of tropical South America. In *Wetlands of the World: Inventory, Ecology, and Management Volume I*; Whigham, D.F., Dykyjova, D., Heiny, S., Eds.; Kluwer Academic Publishers: Dordrecht, 1993; 679–739.
16. Brinson, M.M. *A Hydrogeomorphic Classification for Wetlands*. Wetlands Research Program Tech. Rep. WRP-DE-4; U.S. Army Waterways Expt. Station: Vicksburg, 1993; 103 pp.
17. Tiner, R.W. *Keys to Waterbody Type and Hydrogeomorphic-Type Wetland Descriptors for U.S. Waters and Wetlands, Operational Draft*; U.S. Department of the Interior, Fish and Wildlife Service; Northeast Region: Hadley, 2000; 20 pp.
18. Finlayson, C.M.; Davidson, N.C. Global review of wetland resources and priorities for wetland inventory: Summary report. In *Global Review of Wetland Resources and Priorities for Wetland Inventory*; Finlayson, C.M., Spiers, A.G., Eds.; Supervising Scientist Report 144: Canberra, 1999; 9 pp.
19. Taylor, J.A. Peatlands of the British Isles. In *Mires: Swamp, Bog, Fen, and Moor; Regional Studies*; Gore, A.J.P., Ed.; Elsevier: Amsterdam, 1983; 1–46.
20. Botch, M.S.; Massing, V.V. Mire ecosystems in the U.S.S.R. In *Mires: Swamp, Bog, Fen, and Moor; Regional Studies*; Gore, A.J.P., Ed.; Elsevier: Amsterdam, 1983; 95–152.
21. Swarts, F.A., Ed. *The Pantanal of Brazil, Bolivia, and Paraguay*; Waterland Research Institute; Hudson MacArthur Publishers: Gouldsboro, 2000; 287 pp.

Wetlands: Biodiversity

Jean-Claude Lefeuvre

Laboratory of the Evolution of Natural and Modified Systems, University of Rennes, Rennes, France

Virginie Bouchard

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Wetlands are highly complex ecosystems subjected to a variety of hydrologic regimes, climatic conditions, soil formation processes, and geomorphologic settings. They include swamps, bogs, marshes, mire, fens, salt marshes, mangroves, and other types of ecosystems saturated by water during all or part of the growing season. They are found on every continent, except Antarctica, and in various climes, from the tropics to the tundra. The extent of the world's wetland is generally estimated to be from 7 to 9 million km² or about 4–6% of the earth. The variety of seasonal and perennial wetlands provides environmental conditions for highly distinctive fauna and flora.

ROLE OF HYDROLOGY

Wetlands differ significantly in their water source and seasonal hydrologic regime. Hydrological patterns (i.e., flooding frequency, duration, and hydroperiod) influence physical and chemical characteristics (e.g., salinity, oxygen and other gas diffusion rates, reduction–oxidation potential, and nutrient solubility) of a wetland. In return, these internal parameters and processes control flora and fauna distribution as well as ecosystem functions. Plants, animals, and microbes are often oriented in predictable ways along the hydrological gradient (Fig. 1). Conversely, the biotic component affects the hydrology by eventually modifying flow or water level in a wetland.^[1,2] Species also influence nutrient cycles and other ecosystem functions.^[3]

A LANDSCAPE PERSPECTIVE

Although the hydrology is part of the ecological signature of an individual wetland, wetlands are considered as neither aquatic nor terrestrial systems. They have characteristics from both systems and are defined as ecotones placed under this dual influence.^[4] Because wetlands are located at the interface of multiple systems, they assure vital functions (e.g., wildlife habitats) beneficial at the landscape level. Reduction of wetland area often reduces biodiversity in the landscape.^[2,5] Increases in biodiversity occur when wetlands are created or restored in a disturbed landscape.^[6]

WETLAND FLORA

Wetland plants are adapted to a variety of stressful abiotic conditions (e.g., immersion, wave abrasion, water level

fluctuation, and low oxygen conditions). Identical adaptations to common environmental features have led taxonomically distinct species to sometimes look similar in terms of morphology, life cycle, and life forms.^[7] Traditionally, wetland plants have been classified into groups of different life forms, primarily in relation with hydrological conditions. Helophytes are defined as plant species with overwintering buds in water or in the submerged bottom.^[8] They are differentiated from hydrophytes in that their vegetative organs are partially raised above water level.^[8] Hejny's classification is based on relatively stable vegetative features that determine the ability of wetland plants to survive two unfavorable conditions, namely cold and drought.^[7] This classification uses the types of photosynthetic organs present in both the growth and flowering phases. Other classifications include both life form and growth form. When a species has a range of growth form, it is classified under the form showing the greatest achievement of its potential.^[7]

Plant communities in wetlands can be more or less homogeneous, mosaic-like, or distributed along a gradient resulting in a clear zonation of species. A gradient exists if one or several habitat parameters change gradually in space. This phenomenon is common in freshwater marshes that present a gradient of water depth and water-saturated soils (Fig. 1). Such a gradient is often accompanied by differences in peat accumulation that is influenced by waves or currents. General principles of the zonation of aquatic plants have been largely described.^[7,9,10] Littoral vegetation can belong to several types of communities, which derived from the general principle that, from deeper water to the shore, we may expect successively submergent, floating, and emergent macrophytes. The most important habitat factor is water depth, depending on slope and

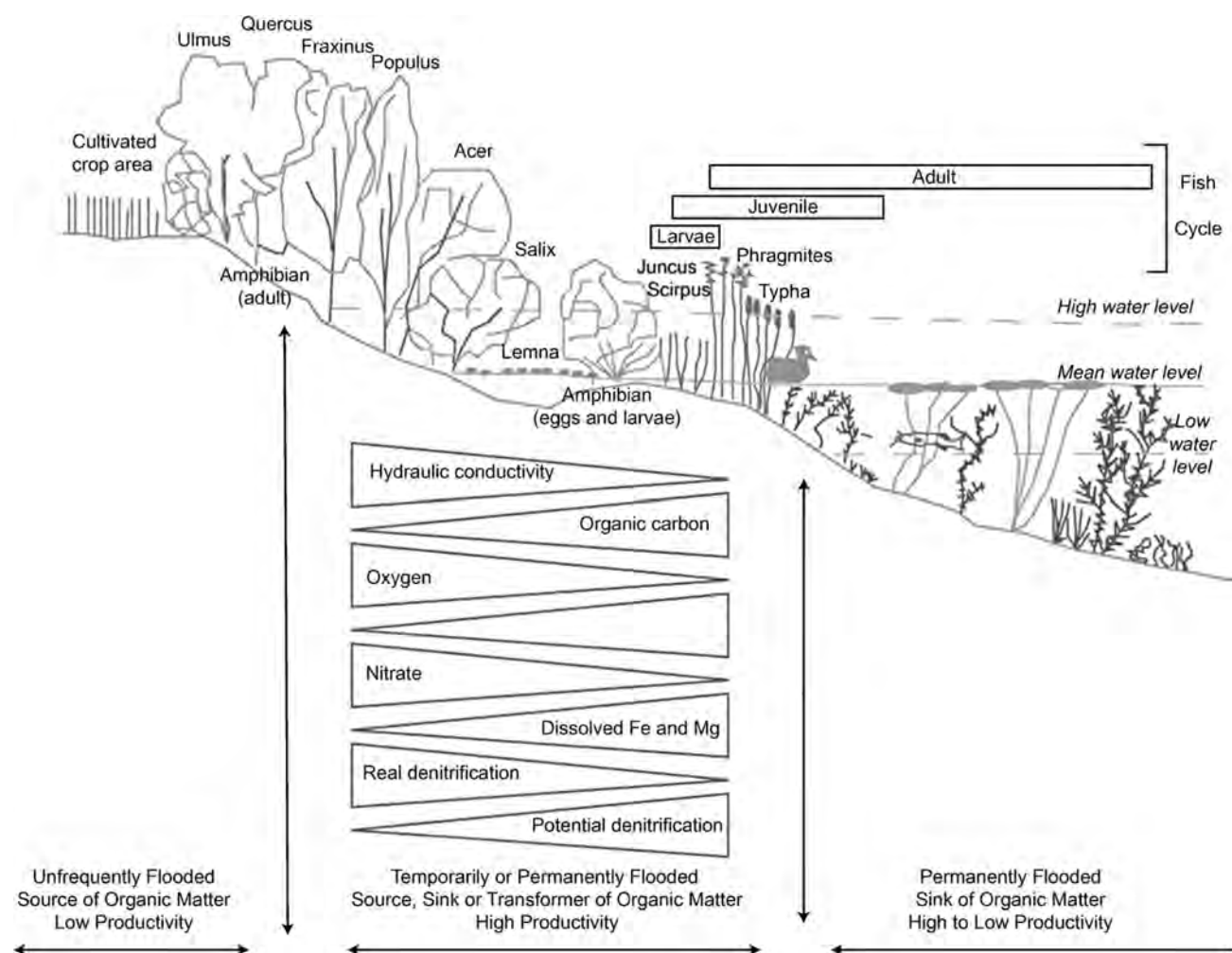


Fig. 1 Species distribution along the hydrological gradient in a freshwater marsh.

peat accumulation.^[10] Other factors may be poor irradiance caused by high turbidity or exposure to waves or flow.^[7,10]

Riparian ecosystems are found along streams and rivers that occasionally flood beyond confined channels or where riparian sites are created by channel meandering in the stream network. Riparian or bottomland hardwood forests contain unique tree species that are flood tolerant. Species distribution is associated with floodplain topography, flooding frequency, and flooding duration.^[11,12] In Southeastern U.S. bottomland, seasonally flooded forests are colonized by *Platanus occidentalis* (sycamore), *Ulmus americana* (American elm), and *Populus deltoides* (cottonwood) and are flooded between 2% and 25% of the growing season (Fig. 1). Other species such as *Fraxinus pennsylvanica* (green ash), *Celtis laevigata* (sugarberry), and *Carya aquatica* (water hickory) are colonized bottomlands that are flooded by less than 2% of the growing season.^[11] Freshwater marshes are dominated with emergent macrophytes rooted in the bottom with aerial leaves (i.e., helophytes). Species such as *Typha* (cattail), *Phragmites* (reed grass), and *Scirpus* (bulrush) are often clonal.

A plant community is usually organized in sequence of patches that are dominated by one species. The second plant groups are the rooted plants with floating leaves (*Nymphaeid*). Lotus (*Nelumbo*) and water lilies (*Nymphaea*) have very identical morphology (i.e., similar leaves and flowers), but a genetic analysis showed that lotus is more closely related with plane tree than with water lilies.^[10] Submerged plants include elodeids (i.e., cauline species whose whole life cycle can be completed below the water surface or where only the flowers are emergent) and isoetids (i.e., species growing on the bottom whose whole life cycle can be completed without contact with the surface). Submerged species include species such as coontail (*Ceratophyllum demersum*) and water milfoil (*Myriophyllum* spp.). Plant species found in salt marshes are called halophytes (i.e., plants that complete their cycle in saline environments). A saltmarsh can be divided into low, middle, and upper marsh, according to flooding frequency and duration. Each zone is dominated by different plant species according to their tolerance to saline immersion.^[10]

A dominant competitive species—often a clonal species—can modify the theoretical zonation. Change in water chemistry (i.e., eutrophication) or hydrology may favor a particular species over the natural plant community. Highly competitive species are often invasive and aggressive in displacing native species. The expansion of *Phragmites australis* into tidal wetlands of North America causes a reduction in biodiversity as many native species of plants are replaced by a more cosmopolitan species.^[13] In riparian ecosystems, biodiversity is usually higher in the intermediate zones, whereas it is lower upstream and downstream (Fig. 2). The percentage of exotic species is low in upstream areas, but can represent up to 40 in downstream zones (Fig. 1).^[14]

WETLAND FAUNA

Diversity of vertebrate and invertebrate species is the result of a diverse community composed of resident and transient species, which use the space differently and at various times of day and year. The density and variety of animal populations at a particular wetland site are also explained by climatic events that affect geographic areas on a large scale; e.g., the population of waterfowl during winter is largely dependent on climate variations in the northern part of the continents.^[15]

Resident species are often dependent on the type of vegetation. Animal communities are generally distributed along a zonation pattern, parallel to the plant communities,

which is driven by the hydrological gradient. A few species depend entirely on a single plant species for their survival. One beetle (*Donacia* spp.) may depend on reeds, at least during its larval stage where another beetle (*Galerucella* spp.) uses only water lilies as their habitat and diet. For many other residents, their habitats extend to several plant communities during their life span. It is generally the case for many vertebrates such as amphibians, rodents, passerines, and waterfowl.

Many amphibian species depend on wetland or riparian zones for reproduction and larval stage. Likewise, wet meadows are necessary as a reproduction zone and nursery for a number of freshwater fishes. About 220 animals and 600 plant species are threatened by a serious reduction of wetlands in California, and the state's high rate of wetland loss (91% since the 1780s) is partly responsible.^[15] Many waterfowl species are sensitive to areas of reduction, patch size and distribution, wetland density, and proximity to other wetlands.^[16] When the Marais Poitevin (France), one of the principal wintering and passages sites for waterfowl in the Western Europe, underwent agricultural intensification in the 1980s, the population of ducks and waders declined tremendously. This decline was partly due to a 50% reduction of wet meadows between 1970 and 1995, primarily caused by the conversion to arable farmlands.^[17] Thus, maintenance of biodiversity depends on the existence of inter connections between wetlands and between aquatic and terrestrial ecosystems. In fact, some authors have pointed out that increase in biodiversity occurs when wetlands are created or restored in a disturbed landscape.^[6,18]

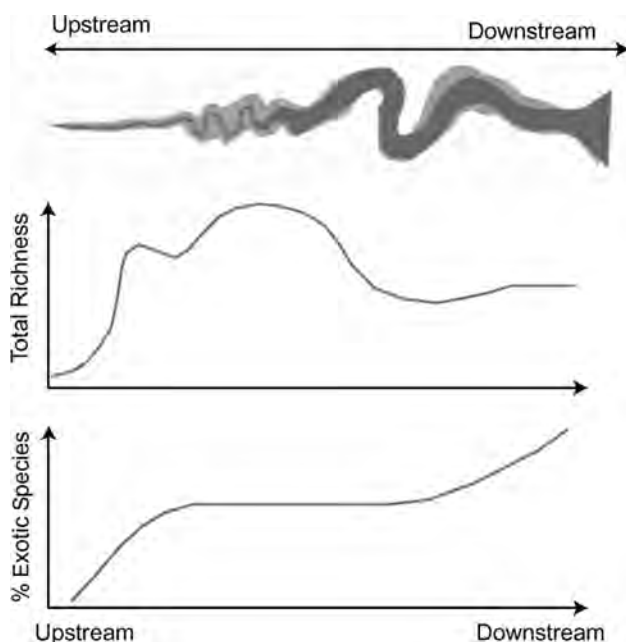


Fig. 2 Total richness and invasive species distribution along a stream longitudinal gradient.

Source: From Planty-Tabacchi.^[14]

CONCLUSION

Despite the importance of wetlands, conservation efforts have ignored them for a long time. It is urgent to conserve the existing wetlands and also to restore and create wetland ecosystems. A wide range of local, state, federal, and private programs are available to support the national policies of wetland “No Net Loss” in the United States and around the world. From a biodiversity perspective, wetland protection policies may not be working because restored or created wetlands are often very different from natural wetlands.^[19] Created wetlands often result in the exchange of one type of wetland for another and result in the loss of biodiversity and functions at the landscape level.^[19] We know that it takes more water to restore a wetland,^[20] even if an important place should be given to the ability of self-design of wetland ecosystems.^[21]

REFERENCES

1. Naiman, R.J.; Johnston, C.A.; Kelley, J.C. Alteration of north American streams by beaver. *BioScience* **1988**, *38*, 753–762.

2. National Research Council. *Wetlands: Characteristics and Boundaries*; National Academy Press: Washington, 1995; 306 pp.
3. Naiman, R.J.; Pinay, G.; Johnston, C.A.; Pastor, J. Beaver influences on the long-term biogeochemical characteristics of boreal forest drainage networks. *Ecology* **1994**, *75*, 905–921.
4. Hansen, A.J.; Di Castri, F., Eds. *Landscape boundaries. In Consequences for Biotic Diversity and Ecological Flows*; Ecological Studies; Springer: Berlin, 1983; 452 pp.
5. Gibbs, J.P. Wetland loss and biodiversity conservation. *Conserv. Biol.* **2000**, *14*, 314–317.
6. Weller, M.W. *Freshwater Marshes*, 3rd Ed.; University of Minnesota Press: Minneapolis, 1994; 192 pp.
7. Westlake, D.F.; Kvet, J.; Szczepanski, A., Eds. *The Primary Ecology of Wetlands*; Cambridge University Press, 1998; 568 pp.
8. Raunkiaer, C. *The Five Forms of Plants and Statistical Plant Geography*; Oxford University Press: Oxford, 1934; 632 pp.
9. Grevilliot, F.; Krebs, L.; Muller, S. Comparative importance and interference of hydrological conditions and soil nutrient gradients in floristic biodiversity in flood meadows. *Biodivers. Conserv.* **1998**, *7*, 1495–1520.
10. Mitsch, W.J.; Gosselink, J.G. *Wetlands*, 3rd Ed.; Wiley: New York, 2000; 920 pp.
11. Clark, J.R.; Benforado, J., Eds. *Wetlands of Bottomland Hardwood Forest*; Elsevier: Amsterdam, 1981; 401 pp.
12. Keogh, T.M.; Keddy, P.A.; Fraser, L.H. Patterns of tree species richness in forested wetlands. *Wetlands* **1999**, *19*, 639–647.
13. Chambers, R.M.; Meyerson, L.A.; Saltonstall, K. Expansion of *Phragmites Australis* into Tidal wetlands of North America. *Aquat. Bot.* **1999**, *64*, 261–273.
14. Planty-Tabacchi, A.M. *Invasions Des Corridors Riverains Fluviaux Par Des Espèces Végétales D'origine Étrangère*; These University Paul Sabatier Toulouse III, 1993; France, 177 pp.
15. Hudson, W.E., Ed. *Landscape Linkages and Biodiversity*; Island Press: Washington, 1991.
16. Leibowitz, S.C.; Abbruzzese, B.; Adams, P.R.; Hughes, L.E.; Frish, J. *A Synoptic Approach to Cumulative Impact Assessment: A Proposed Methodology*; Environmental Research Laboratory, U.S. Environmental Protection Agency: Corvallis, OR, 1992, EPA/600/R-92/167.
17. Duncan, P.; Hewison, A.J.M.; Houte, S.; Rosoux, R.; Tournebize, T.; Dubs, F.; Burel, F.; Bretagnolle, V. Long-term changes in agricultural practices and wildfowling in an internationally important wetland, and their effects on the guild of wintering ducks. *J. Appl. Ecol.* **1999**, *36*, 11–23.
18. Hickman, S. Improvement of habitat quality for resting and migrating birds at the Des Plaines River Wetlands Demonstration Project. *Ecol. Eng.* **1994**, *3*, 485–494.
19. Whigham, D.F. Ecological issues related to wetland preservation, restoration, creation and assessment. *Sci. Total Environ.* **1999**, *240*, 31–40.
20. Zedler, J.B. Progress in wetland restoration ecology. *Trends Ecol. Evol.* **2000**, *15* (10), 402–407.
21. Mitsch, W.J.; Wu, X.; Naim, R.W.; Weihe, P.E.; Wang, N.; Deal, R.; Boucher, C.E. Creating and restoring wetlands: A whole ecosystem experiment in self-design. *BioScience* **1998**, *48*, 1019–1030.

Wetlands: Carbon Sequestration

Virginie Bouchard

Matthew Cochran

School of Natural Resources, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

The primary production in wetland ecosystems is greatly enhanced by hydrological fluxes, which leads to high biomass, primary production, and accretion rates. The carbon (C) is either fixed stored in standing biomass, released by soil decomposing microorganisms, or stored in soil sediments. With the continual destruction of wetlands worldwide, more of this sequestered C is released to the atmosphere. The protection of wetlands will preserve the amount of C already stored in these ecosystems.

INTRODUCTION

The increase in the concentration of “greenhouse gases” in the troposphere and its relation to human activities is well documented.^[1] Carbon (C) fixation via photosynthesis and C release during decomposition have always been two important processes regulating the concentrations of carbon dioxide (CO₂) and methane (CH₄) in the atmosphere, even in prehistoric and preindustrial times. In addition to these internal processes, wetland ecosystems receive and release organic C through hydrologically driven mass fluxes. Thus, the C pool within any wetland ecosystem is in balance between primary production, microbial decomposition, and C fluxes within interconnected ecosystems (Fig. 1). Wetlands play a particularly complex role in controlling greenhouse gases, as these ecosystems are intimately associated with all aspects of the production and consumption of both CO₂ and CH₄ (Fig 2).

C SEQUESTRATION WITH PRIMARY PRODUCTION

The concept of primary productivity is directly related to the ideas of energy flow in ecosystems. A portion of the photosynthetically active radiation, which is radiation in the 400–700 nm wave band, received by an ecosystem is absorbed by autotrophic organisms (photosynthetic plants and microorganisms). The absorbed energy is reradiated, lost as latent heat, or stored by the activity of photosynthesis in organic substances. This last flow of energy corresponds to net primary production (NPP). Fundamental ecological questions relating to the global C budget, the location of the missing C sink, and predictions of global climate change rely on obtaining good estimates of NPP.

Wetland ecosystems can have very high standing biomass values and correspondingly high NPP.^[2,3] The annual aboveground NPP of macrophytes is reported to be up to 5 kg dry matter in the most productive sites.^[2] This production varies according to species, wetland type, and latitude^[2,3] and is often well correlated with the maximum aboveground standing crop. Belowground production is much more difficult to estimate because roots and rhizomes grow and die at different rates and times and because materials are translocated to and from shoots. The ratio of belowground to aboveground production can vary from 0.2 to 2.5 according to various studies.^[2] The connectivity of a wetland to hydrological fluxes (i.e., saltmarshes flooded by tides, coastal freshwater marshes flooded by seiche, and riparian bottomland forests flooded by floods) is one of the most important factors enhancing primary productivity.^[4]

Net C capture from herbaceous vegetation is minimal compared to the long-term accumulation of C in bottomland hardwood forests. At the end of the growing season, most of the C trapped in the biomass of herbaceous macrophytes is found in dead plant litter and is released as CO₂ or CH₄ during decomposition or exported as dissolved or particulate organic matter to adjacent systems. Bottomland hardwood forests store C in tree biomass for a much longer period of time. The productivity of bottomland forest ranges between 200 and 2000 g dry organic matter per year.^[5]

C SEQUESTRATION IN SOILS

C fixed in photosynthesis either remains in the sediment as C accretion or is decomposed to CO₂ and CH₄ by a suite of fermentative microbes involved in soil organic matter decomposition. A diversity of biological, chemical, and physical mechanisms is also known to selectively

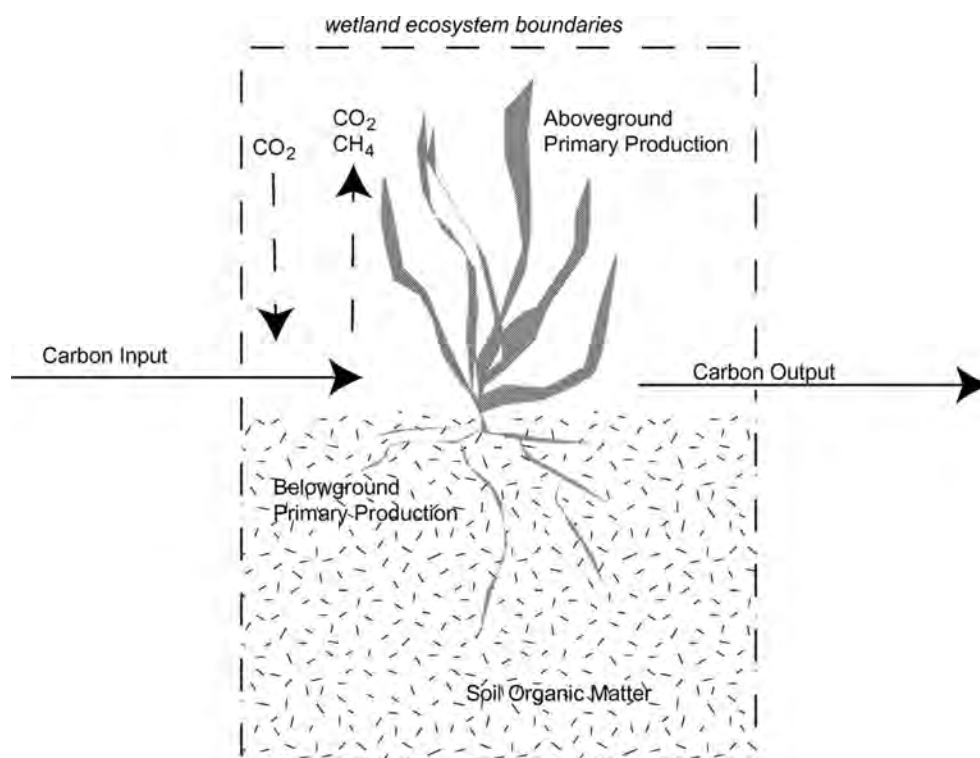


Fig. 1 Conceptual model showing the fluxes of organic C in wetlands.

“protect” different pools of soil organic matter from decomposition by soil microorganism. Most of the recalcitrant C destined for sediment accretion is derived from heavily lignified biomass. Organic soil is a result of the anaerobic conditions created by standing water or poorly drained conditions.

C is even better protected in acidic environments, in marine sediments, and under low temperatures. Peat accumulation is a result of reduced oxidation of the biomass produced in wetlands. The northern peatlands have accumulated 25–38 Pmol C since the last glaciation, equivalent to 50–70% of the total amount of C

present in the atmosphere.^[6] Peat accumulation is greater in bogs than in fens due to differences in nutrient availability. Decomposition rates are enhanced in minerotrophic fens receiving nutrients from adjacent mineral soils relative to ombrotrophic bogs, which are fed only by rainwater.

Canadian wetland soils store about 154 GtC, representing an average of 60% more C than what is stored in Canadian forests (95 Gt C in biomass and soils), which in turn is two orders of magnitude larger than the agricultural soil C pool^[7] (Table 1). Peatlands are a sink for between 20 and 30 g C m⁻² yr⁻¹, which means that Canadian peatlands sequester about 0.03 Gt C yr⁻¹ (Table 1). The sequestration rate of peatlands is smaller than the rate computed for boreal forests. However, the major difference between forests and peatlands is that a forest sink is transient and cannot be sustained continuously. If peatlands also have a theoretical limit to growth, it is reached only after many thousands of years.^[8] However, the fluxes of CO₂ and CH₄ from peatlands vary highly according to seasons, years, species, and sites. Interannual measurement schemes—such as those that have been developed for forest ecosystems with the Ameriflux and Euroflux programs—are needed to establish more precise fluxes of CO₂.

IMPACT OF HUMAN ACTIVITIES ON C STORAGE

Rates of plant production are limited by phosphorus supply in freshwater systems and by nitrogen (N) supply in



Fig. 2 View of a freshwater wetland.

Table 1 Summary of C stocks and fluxes in Canadian wetlands/peatlands, forests, and agricultural fields.

Land use	Area (km ²)	C stock (Gt C)	Fluxes (Gt C yr ⁻¹)	Potential duration (years)
Peatlands/wetlands soils	1.24 × 10 ⁶	154	-0.03	1000 (?)
Forests	5.15 × 10 ⁶	95	-0.2	50
Agriculture fields	0.45 × 10 ⁶	6	0.0034	50

Source: Adapted from Roulet.^[7]

saltwater and terrestrial systems.^[9,10] An excess of nutrients leads to an increase of both primary production and decomposition in terrestrial systems.^[11] Decomposition rates increase, however, at a slower rate than primary production, leading to a potential increase of C storage in soil.^[11] In wetlands, little is known about the effect of excess nutrients on the denitrification process, which is directly linked to the amount of soil organic matter.^[12]

Wetlands have been drained on all continents due to human development. Rapid changes occur in organic soils after drainage as aerobic microbial decomposition is enhanced, releasing large amounts of CO₂ into the atmosphere. C losses can lead to subsidence of the soil profile, changes in bulk density, and decreases in C content of the remaining soil.

CREATION OF WETLANDS TO SEQUESTER C

Wetlands—particularly bottomland hardwood forests—are considered as potentially excellent “C sinks” because they take CO₂ out of the atmosphere and store it in living plant tissues and soil organic matter. Wetland creation might provide an opportunity to positively address wetland habitat losses while also addressing global warming.^[8,13] However, a question, remains: are we capable of constructing wetlands that function in similar ways as natural wetlands? How well these created and constructed wetlands mimic natural wetlands is being debated.^[14,15]

Various studies have indicated that created wetlands might not function in the same capacity as adjacent reference wetlands.^[16] In created *Spartina alterniflora* salt marshes, vegetation rapidly achieves 100% cover although soil nitrogen and organic matter are slow to accumulate. Salt marshes constructed in North Carolina have lower soil organic carbon (SOC) and lower total N reservoirs than a 2000-year-old natural marsh.^[16]

Their C accumulation rates are similar to those of reference sites, but N accumulation rates are higher; thus, C/N ratios have declined over time. In Oregon, 95 restored freshwater marshes had lower soil organic matter than natural marshes and no evidence of accumulation.^[17]

Because bottomland hardwood forests combine both a long-term C storage in tree biomass and a slow release of SOC under flooded conditions, they are considered as

excellent ecosystems that could potentially be used to sequester C. The restoration of bottomland hardwood forests on marginal agricultural lands in the Mississippi River Valley offers significant net C sequestration in the south-central region of the United States.^[13] However, hardwood plantings have problems with slow initial growth and excessive early mortality rates. The use of herbaceous weed control and bedding may offer the potential to overcome these difficulties.^[13] Under the initiative of the U.S. Department of Energy, an 80-acre pilot study is underway in Louisiana. Abandoned marginally productive agricultural fields were planted in January 1997 with seven bottomland hardwood species. Following planting, the site will be monitored for planting success and survival. When the trees attain heights of >4.5 ft, growth data and C sequestration per acre will be calculated.^[13]

CONCLUSION

The primary production in wetland ecosystems is greatly enhanced by hydrological fluxes, which leads to high biomass, primary production, and accretion rates. The C is either fixed stored in standing biomass, released by soil decomposing microorganisms, or stored in soil sediments. With the continual destruction of wetlands worldwide, more of this sequestered C is released to the atmosphere. The protection of wetlands will preserve the amount of C already stored in these ecosystems.

The creation and restoration of new wetlands will certainly contribute to the sequestration of C and should be considered as one way to mitigate greenhouse emissions.

REFERENCES

1. Houghton, J.T.; Meira Filho, L.G.; Bruce, J.; Hoesung, L.; Callander, B.A.; Haites, E.; Harris, N.; Maskell, K., Eds. IPCC (Intergovernmental Panel on Climate Change), climate change 1994. In *Radiative Forcing of Climate Change and an Evaluation of the IPCC IS92 Emission Scenarios*; Cambridge University Press: Cambridge, 1995.

2. Westlake, D.F.; Kvet, J.; Szczepanski, A. *The Primary Ecology of Wetlands*; Cambridge University Press: Cambridge, 1998; 568 pp.

3. Mitsch, W.J.; Gosselink, J.G. *Wetlands*, 3rd Ed.; Wiley: New York, 2000; 920 pp.
4. Odum, W.E.; Odum, E.P.; Odum, H.T. Natures pulsing paradigm. *Estuaries* **1995**, *18* (4), 547–555.
5. Conner, J. Effects of forest management practices on southern forested wetland productivity. *Wetland* **1994**, *14*, 27–40.
6. Gorham, E. Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. *Ecol. Appl.* **1991**, *1*, 182–195.
7. Roulet, N.T. Peatlands, carbon storage, greenhouse gases, and the kyoto protocol: Prospects and significance for Canada. *Wetlands* **2000**, *20* (4), 605–615.
8. Clymo, R.S. Models of peat growth. *SUO* **1993**, *43*, 127–136.
9. Tamm, C.O. *Nitrogen in Terrestrial Ecosystems*; Springer: Berlin, 1991.
10. Vitousek, P.M.; Howarth, R.W. Nitrogen limitation on land and in the sea: How can it occur? *Biogeochemistry* **1991**, *75*, 87–115.
11. Vitousek, P.M.; Aber, J.D.; Howarth, R.W.; Likens, G.E.; Matson, P.A.; Schindler, D.W.; Schlesinger, W.H.; Tilman, D.G. Human alterations of the global nitrogen cycle: Sources and consequences. *Ecol. Appl.* **1997**, *7* (3), 737–750.
12. Davidsson, T.E.; Stahl, M. The influence of organic carbon on nitrogen transformation in five wetland soils. *Soil Sci. Soc. Am. J.* **2000**, *64*, 1129–1136.
13. Williams, J.R. Addressing global warming and biodiversity through forest restoration and coastal wetlands creation. *Sci. Total Environ.* **1999**, *240*, 1–9.
14. Mitsch, W.J.; Wu, X.; Nairn, R.W.; Weihe, P.E.; Wang, N.; Deal, R.; Boucher, C.E. Creating and restoring wetlands: A whole ecosystem experiment in self-design. *BioScience* **1998**, *48*, 1019–1030.
15. Zedler, J.B. Progress in wetland restoration ecology. *Trends Ecol. Evol.* **2000**, *15* (10), 402–407.
16. Craft, C.; Reader, J.; Sacco, J.N.; Broome, S.W. Twenty-five years of ecosystem development of constructed *Spartina Alterniflora* (Loisel) marshes. *Ecol. Appl.* **1999**, *9* (4), 1405–1419.
17. Shaffer, P.; Ernst, T. Distribution of soil organic matter in freshwater emergent/open water wetlands in the Portland, Oregon metropolitan area. *Wetlands* **1999**, *19*, 505–516.

Wetlands: Conservation Policy

Clayton Rubec

Center for Environmental Stewardship and Conservation, Ottawa, Ontario, Canada

Abstract

Meeting the challenge of conserving wetlands requires comprehensive national policies to provide a foundation for domestic action and a framework for international and national cooperation. Such policy is valuable as countries seek to address the management and habitat requirements for wildlife and other natural resources, such as soil and water, as well as human needs. The implementation of a national wetland policy is a key feature of the wise use principles of the Convention on Wetlands of International Importance (the Ramsar Convention). However, such policy seems an elusive goal for many of the 145 nations that (May 2005) are Contracting Parties to this global environmental treaty.

INTRODUCTION

Responding to recommendations by the Convention, in 1998–1999, the author led a team of writers in preparing guidelines for developing and implementing national wetland policies.^[1] These policy guidelines complement the Convention's guidance on wetland legislation.^[2] This entry provides highlights of the guidelines and reports on the status of wetland policy development around the world, effective mid-1999.

RELATIONSHIP BETWEEN POLICY AND WISE USE

The wise use principles are a hallmark of the Ramsar Convention. "Wise use" applies not only to sites listed as wetlands of international importance (as of May 2005 covering over 125 million hectares at 1429 sites) but also to all wetlands in the territory of Contracting Parties. These *principles* help Contracting Parties improve institutional and organizational arrangements, address legislative and policy needs, increase knowledge and awareness of wetland values, inventory and monitor the status of wetlands, identify program priorities, and develop action plans for specific sites as components of a national wetland policy.

The formulation of national wetland policies sometimes involves a lengthy and complex process. Political, jurisdictional, institutional legal, and financial constraints affect policy formulation in addition to social and economic factors that continue to contribute to wetland loss, while the policy process is under way.

WHAT ARE WETLAND POLICIES AND ARE THEY NEEDED?

A policy can simply be a document. However, making is also a process involving consensus building, the encapsulation of

ideas and commitments, implementation, accountability, and review complemented by legislation, strategies, and operational programs. It is a mechanism for an administration to capture the public will or mandate on an issue and refine it with its own vision.

A National Wetland Policy is nationwide in scope, but it may be developed at several levels of government. In Australia and Canada, e.g., both the federal government and state/provincial governments have developed wetland conservation policies. This reflects the federal nature of these two nations, wherein constitutional authority for natural resources management (including wetlands) is divided between the levels of government.

Wetlands are seldom explicitly covered in other natural resource management policies such as for water, soil, forest, land, or agriculture at a national level. Development of a "stand-alone" wetland policy and/or strategy can be an important step in recognizing and solving wetland problems. A wetland policy recognizes wetlands as ecosystems requiring different approaches not masked under other sectoral management objectives. The articulation of goals and objectives for these ecosystems identifies clear responsibilities of the government and a public expectation that the government will deliver these commitments.

Wetland policy objectives need to focus on a variety of themes, as they become the image of the policy. However, the practical implementation of the policy may result in only one or two of these objectives receiving the greatest public attention. For example, Canada's announcement of its federal wetland policy in 1992 contained seven objectives, but "no net loss of wetland functions" has proven to be its catch phrase.

THE GUIDELINES

The guidelines review the key steps and issues that may arise in both developing and implementing a National

Wetland Policy. These include over 20 detailed sections defining the purpose of such an initiative, organizing a suitable process, deciding how to present the content of the policy document, and developing strategies for implementation and monitoring. The text is complemented by the following seven wetland policy essays: 1) *defining stakeholders*; 2) *consultations*; 3) *wetland policies within a federal state*; 4) *sectoral policies and legislation*; 5) *compliance strategies*; 6) *role of nongovernment organizations*; and 7) *the development and coordination process*. The guidelines provide a reference against which all nations can review their wetland action plans and strategies at the national level.

A team of contributors with governmental or non-governmental work experience and expertise in wetland policy development prepared the guidelines. The team included writers from Ramsar national authorities in Australia, Canada, Trinidad and Tobago, Uganda, and the United States. Contributors from BirdLife International, University of Massachusetts, IUCN Environmental Law Centre, and Wetlands International were also involved.

Implementation Strategies

A National Wetland Policy includes specific implementation strategies that demonstrate the priorities of the

government and also fosters the cooperation and involvement of other interests. Linkages between these strategies and national water, soil, biodiversity, and sustainable development policy initiatives are explored in the guidelines.

An analysis of the strategies used in selected National Wetland Policies is summarized in Table 1. These include the policies/action plans of Australia, Cambodia, Canada, Columbia, Costa Rica, Finland, France, Jamaica, Malaysia, Peru, Trinidad and Tobago, and Uganda. These initiatives have many common strategic approaches as follows: 1) ensuring public awareness and education; 2) developing cooperation and partnerships between levels of government from national to local; 3) developing and supporting legislation and interrelated land and water use policies and programs; 4) implementing wetland site management responsibilities; 5) developing a sound basis for the policy through scientific research and expertise; 6) developing institutional and financial capacity for policy implementation; and 7) meeting international commitments. These strategies have been drafted to evoke a clear vision and acceptance across the nation.

Global Review of the Status of Wetland Policies and Strategies

Significant progress is evident globally in the development of National Wetland Policies since the Ramsar Convention focused attention on this issue in 1987.^[1,3,4] Meetings of the Contracting Parties every 3 years allow regular review of the status of wetland policies. The Contracting Parties last met in November 2002 when there were 123 Contracting Parties, while (May 2005) there are 145 Contracting Parties.

Policies

As of November 2002, 60 (56%) of the 107 Ramsar Contracting Parties that submitted national reports to the Eighth Meeting of the Contracting Parties to the Convention on wetlands indicated that they were engaged in development or implementation of a National Wetland Policy. Between 1987 and 2002, the number of nations with a National Wetland Policy officially adopted grew from 0 to 41. An additional 19 nations indicated that a National Wetland Policy was in draft or under consideration. However, 47 (44%) of the Ramsar Contracting Parties did not yet report any actions being taken in support of National Wetland Policy development. A number of nations, particularly those with a commonwealth or federal makeup, reported wetland policies and strategies at the subnational level also. National Wetland Policies have also been called a “national wetland strategy” or an “action plan.”

Table 1 Implementation strategies in selected national wetland policies.

Country	1	2	3	4	5	6	7	8	9	10	11	12
Australia	X	X	X	X	X	X						
Cambodia	X	X	X	X	X	X	X					
Canada	X	X	X	X	X	X						
Colombia	X	X	X	X	X	X	X	X				
Costa Rica	X	X	X	X	X	X	X					
Finland	X	X	X	X	X	X	X					
France	X	X	X	X	X							
Jamaica	X	X	X	X								
Malaysia	X	X	X	X	X	X	X	X	X			
Peru	X	X	X	X	X	X						
Trinidad and Tobago	X	X	X	X	X	X						
Uganda	X	X	X	X	X	X	X	X				

Note: Policy strategies: 1) management of national wetland networks; 2) integration with other policies such as water, soil, and forests; 3) public awareness and education; 4) partnerships; 5) science, monitoring, assessment, and research; 6) international commitments; 7) managing special sites; 8) administration and institutions, capacity building; 9) enforcement, regulation, and legislation; 10) financial mechanisms; 11) restoration of degraded sites; and 12) sustainable use and conservation.

Table 2 Evolution of Ramsar Convention on national wetland policies/strategies.^a

Status of national wetland policies or strategies	1987 Regina COP3	1990 Montreux COP4	1993 Kushiro COP5	1996 Brisbane COP6	1999 San José COP7	2002 Valencia COP8
National wetland policies						
Policy/strategy adopted	0	0	3	6	12	41
Policy/strategy in draft or under consideration	2	5	12	21	30	19
No policy/strategy activity reported	15	40	36	65	72	47
Number of national reports tabled at this COP	17	45	51	92	98	107

^aAs of May 2005, the Ramsar Convention had 145 Contracting Parties.

Table 2 summarizes the status of the development and adoption of National Wetland Policies from 1987 to November 2002. This table was developed by reviewing the reports and conference papers that summarize the activities of the Convention every 3 years by each country. In 1987, only two nations indicated that they were involved in developing any sort of National Wetland Policy/Strategy. By 2002, this number grew to at least 60 nations. Table 3 summarizes the countries by Ramsar's regions (Africa, Asia, Europe, Neotropics, North America, and Oceania) that have either a *national wetland policy* or *national wetland strategy* adopted, being drafted or considered up to May 2005.

CONCLUSION

The development and implementation of National Wetland Policy in the 145 countries that have acceded to the Ramsar Convention is proceeding throughout the world. The Convention's Wise Use Principles and Guidelines on National Wetland Policy,^[1] complemented by guidelines for wetland legislation,^[2] are effective tools in fostering the completion and use of National Wetland Policies and Strategies as important cornerstones of this Convention. At least 56% of the Ramsar Convention's Contracting Parties are continuously implementing or developing National Wetland Policies/Strategies.

One of the most interesting aspects of the Ramsar Convention is its capacity to foster sharing of experience. Interchanges by wetland policy experts are occurring internationally that involve short-term invited visits or sabbaticals, informal exchange of documents, confidential advice, and review of draft policy. Consultation workshops, working with non-government groups, meeting with senior government officials, exploring funding mechanisms, and assisting in drafting the text, have also been involved. Experience gained in one nation's development of these policies

Table 3 Summary of national wetland policies and strategies by Ramsar regions (2002).

Ramsar region	National Wetland Policy or Strategy Adopted by Government	National Wetland Policy or Strategy in Preparation or Under Consideration
Africa	Congo Republic, Côte d'Ivoire, Senegal, South Africa, and Uganda	Algeria, Botswana, Burkina Faso, Chad, Comoros, Democratic Republic of the Congo, Egypt, the Gambia, Ghana, Guinea, Kenya, Malawi, Mali, Morocco, Namibia, Niger, Togo, and Zambia
Asia	Indonesia, Japan, Thailand, and Vietnam	Bangladesh, Cambodia, China, Georgia, India, Republic of Korea, Malaysia, Mongolia, the Philippines, Russia, ^a and Turkey
Europe	Belgium, Bulgaria, Denmark, Estonia, Finland, France, Greece, Iceland, Malta, Monaco, the Netherlands, Norway, Romania, Sweden, Switzerland, and the United Kingdom	Austria, Belarus, Croatia, Czech Republic, Germany, Hungary, Ireland, Italy, Latvia, Lithuania, Poland, Portugal, Russia, ^a Slovak Republic, Slovenia, Ukraine, and Federal Republic of Yugoslavia
Neotropics (Central and South America and Caribbean)	Colombia, Costa Rica, Jamaica, Peru, and Trinidad and Tobago	Argentina, Bahamas, Chile, Ecuador, Guatemala, Honduras, Nicaragua, Panama, Paraguay, and Venezuela
North America	Canada and the United States	Mexico
Oceania (Australia and Pacific)	Australia and New Zealand	—

^aRussia straddles Asia and Europe.

can be shared, and so local expertise can be enhanced, filling a need among the Ramsar family.

REFERENCES

1. Rubec, C.D.A.; Pritchard, D.; Mafabi, P.; Nathai-Gyan, N.; Phillips, B.; Mahy, M.; Lynch-Stewart, P.; Chew, R.; CINTRON, G.; Larson, J.; Ramakrishna, S. *Guidelines for Developing and Implementing National Wetland Policies*; Report to COP7 Ramsar Convention. Handbook No. 2; Ramsar Bureau: Gland, 2000.
2. Shine, C.; Glowka, L. *Reviewing Laws and Institutions to Promote the Conservation and Wise Use of Wetlands*; IUCN Environmental Law Centre Report to COP7 Ramsar Convention. Handbook No. 3; Ramsar Bureau: Gland, 2000.
3. Rubec, C.D.A. Status of national wetland policy development in Ramsar nations. In *Proceedings, Sixth Meeting of the Conference of the Contracting Parties. Convention on Wetlands*, Brisbane, 1996, Vol. 10/12A, 22–29.
4. Ramsar Convention Secretariat. *National Wetland Policies in Ramsar Contracting Parties*; Information Report to Eighth Meeting of the Contracting Parties to the Convention on Wetlands; Ramsar Convention Secretariat: Gland, 2002.

Wetlands: Economic Value

Gayatri Acharya

World Bank, Washington, District of Columbia, U.S.A.

Abstract

The valuation of wetland goods and services has become an important tool of land use analysis and water resource management. With improved understanding of how benefits may be derived directly or indirectly from wetlands, economic tools have been applied to elicit values associated with these benefits. This information is increasingly being used to inform public policy and investment decisions.

INTRODUCTION

Wetlands have historically been and continue to remain centers of human civilization and economic activity, supporting vast populations both directly and indirectly. As such, wetlands have been used, transformed, or converted over time, and their capacity in some cases has been diminished by the impacts of land use changes.

The conversion of multiple use ecosystems to single land uses may signify a welfare loss through reduced access to, or availability of, valuable goods and services. Whether or not such conversions are merited can be assessed only with sufficient information on the relative efficiency of converting or conserving the wetland ecosystem vis-à-vis alternative land uses. Economic values are a measure of how much people are willing to pay for and how much better or worse off they would consider themselves to be as a result of changes in the supply of different goods and services. Valuation provides a means of quantifying the benefits that people receive from wetlands. The net benefits of investing in land uses that are compatible with wetlands conservation, relative to those economic activities, which contribute to wetlands degradation, can then be assessed. Values generated through economic analysis are limited by their inability to capture intrinsic values associated with wetlands and the level of understanding, both economists and natural scientists have, with regard to the level of use of goods and services, and variability in the natural system. Furthermore, economic values measure preferences and welfare impacts associated with changes in the natural system or the availability of a service or product, but they do not, in themselves, suggest anything about inter- or intra-generational distributional effects of conservation or development investments.

Wetland goods and services are also particularly difficult to value. This is because: 1) many goods are not marketed but traded or consumed directly; 2) wetland services such as water quality or groundwater recharge often occur in areas away from the physical location of the wetland and

may not be easily attributable to the wetland; 3) many wetlands are transboundary resources and data on the use and consumption of goods and services are difficult to obtain; and 4) many wetlands are public property. For these and other related reasons, the economic benefits generated by wetlands and the economic costs associated with wetland degradation or loss are frequently unknown and omitted in project or policy analysis. As a result, the potential of wetlands to be acknowledged as contributors to economic growth, income generating activities, and as sources of goods and services, has been underestimated in many parts of the world, resulting in the loss of valuable species, services, and livelihoods. Further, with growing recognition of the links between ecological and economic functions of these ecosystems, there are increasing efforts being made by researchers and planners to factor in the value of the goods and services of wetlands in economic decision making.

ECONOMIC VALUES OF WETLANDS

Wetlands cover an estimated 6% of the world's land surface and may be of various types such as marshes, floodplains, freshwater ponds, lakes, swamps, etc. While these ecosystems are associated with a diverse and complex array of direct and indirect uses, the precise functions and benefits of a specific wetland can be determined only after careful study of the hydrological, physical, and biological characteristics of the site, over a period of time. Mitsch and Gosselink^[1] have long highlighted the importance and complexity of these ecosystems, noting that when trying to understand their value "one must consider whether the values of wetlands are based on biological populations living next to and within the wetlands, the wetland ecosystem itself, or the entire biosphere of which wetlands are a small part."

Increased knowledge of the direct or indirect benefits attributed to these ecosystems has resulted in mounting

concern about the state of the world's wetlands and an increased focus on how to value these ecosystems. As noted by Barbier,^[2] valuing ecosystem services requires two specific sources of knowledge, namely: 1) the ecological processes, components, and functions that generate useful services and 2) the way in which these services translate into specific benefits.^[1] Since the economic value of an ecosystem service (or good) is determined by the contribution it makes to maintain the present level of human well-being, it is calculated as a measure of a change in well-being, or *social welfare*. Therefore, valuing a good or service requires us to study the change in a person's welfare due to a change in the availability of the resource. Economic value expressed in currency units is the monetary expression of the tradeoff between identified alternative land uses (see the work of Freeman^[3] among others for more on valuation theory).

To fully assess these tradeoffs, the ultimate goal of valuation should be to assess the total economic value (TEV) of a system. TEV is defined as the sum of use value plus existence value where use value is comprised of: 1) direct use value (consumptive or non-consumptive); 2) indirect use value; and 3) option value. Box 1 defines each of these values while Fig. 1 shows how direct and indirect benefits can be identified in a wetland system. While valuation attempts to take account of all the components of the total economic benefit of wetlands, it is generally constrained by data availability, and therefore, partial valuation studies are more common (see the work of Barbier^[4] for more on the differences in total and partial valuation).

Direct uses include the use of the wetland for water supply and harvesting of wetland products such as fish and plant resources. Commonly recognized values of wetlands include those captured from the use of raw materials and physical products that are generally used in economic

BOX 1: TYPES OF BENEFITS VALUED BY ECONOMIC VALUATION TECHNIQUES.

Direct benefits: These include the raw materials and physical products that are used directly for production, consumption, and sale including those providing energy, shelter, foods, water supply, transport, and recreation.

Indirect benefits: These include ecological functions which maintain, protect, and support natural and human systems through services such as maintenance of water quality, flow and storage, flood control and storm protection, nutrient retention, and micro-climate stabilization, and other productive and consumptive activities.

Option benefits: These refer to the premium placed on maintaining a pool of wetlands species and genetic resources for future uses such as for leisure or in commercial, industrial, agricultural and pharmaceutical applications and water-based developments, some of which may not be used or known in the present but may be used in the future.

Existence benefits: The intrinsic value of wetlands species and areas regardless of their current or future use possibilities, such as cultural, aesthetic, heritage and bequest significance.

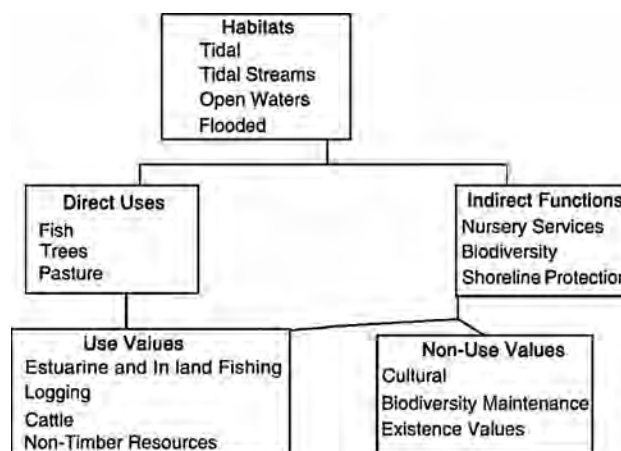


Fig. 1 Identifying direct and indirect benefits in a wetland system.

activities such as fisheries, water supply, and agriculture. Where wild foods, grasses, bird life, and other naturally occurring species are exploited for significant economic returns, such uses are also easily identifiable and may be valued in existing markets.

Direct uses, however, represent only a small proportion of the total value of wetlands, which generate economic benefits in excess of just physical products. *Indirect benefits* are derived from environmental functions that wetlands may provide, depending on the type of wetland, soil, and water characteristics, and associated biotic influences. Wetland functions are defined as a process or series of processes that take place within a wetland, including habitat and hydrological and water quality functions. Such functions have an economic value if they contribute to economic activity and enhance human welfare.

VALUING WETLANDS GOODS—TECHNIQUES AND EXAMPLES

A combination of the available techniques used to value environmental goods and services can be used to approximate a partial or total value of any one ecosystem, taking care that we can identify the various components of the system and have adequate economic and ecological data to do so. The following methods have been used in valuing wetland resources:

Market prices: to estimate economic values for ecosystem products or services that are bought and sold in markets.

Change in productivity: to estimate economic values for ecosystem products or services that contribute to the production of marketed goods or services.

Travel cost: to estimate economic values associated with ecosystems/sites used for recreation based on the assumption that the value of a site is reflected in how much people are willing to pay to travel to visit the site.

Hedonic pricing: to estimate economic values for ecosystem or environmental services that directly affect market prices of some other good, e.g., housing prices may reflect the value of local environmental attributes or amenities.

Contingent valuation method: to estimate economic values for products and ecosystem services. This can be used to estimate non-use values by asking people to directly state their willingness to pay (WTP) for specific environmental services, based on hypothetical scenarios or contingent markets.

Contingent choice, contingent ranking, contingent behavior: to estimate economic values for products and for ecosystem services by asking people to make tradeoffs between sets of ecosystem or environmental services or characteristics.

Benefit transfer: to infer economic values by transferring benefit estimates from localized studies to similar areas under similar market conditions.

Damage cost avoided, replacement cost, and substitute costs: to estimate economic values based on costs of avoided damages resulting from lost or diminished ecosystem services, replacing ecosystem services, or providing substitute services.

Examples of how these techniques have been applied to elicit values of various wetland benefits are given in Boxes 2 and 3 and in Table 1. These studies illustrate the range of values associated with different types of goods and services, wetlands provide.

In some parts of the world, the benefits of protecting natural watersheds and river systems are being recognized and acted upon through government decisions. For

BOX 2: USING CONTINGENT VALUATION AND TRAVEL COST METHODS TO ASSESS THE RECREATIONAL VALUE OF LAKE NAKURU, KENYA.

Lake Nakuru National Park is an important international tourist destination in Kenya. Fees currently charged to enter the park underestimate the total value that tourists place on the wetland and its component species, especially flamingos. A travel cost survey of visitors elicited information about length of stay, travel costs, place of origin, and visitation rates, distinguishing between resident and nonresident tourists. The contingent valuation survey used in this study asked visitors what their personal total costs of travel were; how much they would be willing to increase their expenditures to visit the park; how much they would contribute to a fund to clean-up and control urban pollution affecting the park; and how much they would contribute to a project to conserve flamingos. The survey also asked respondents for the minimum reduction in trip costs that they would be willing to accept should there be no flamingos. Results suggests that the annual recreational value of wildlife viewing in Lake Nakuru National Park was between U.S.\$7.5 and \$15 million and over a third of this value was accounted for by the Willingness to Pay for Viewing flamingos.

Source: Adapted from Navrud & Mungatana.^[5]

BOX 3: VALUING THE HADEIJA NGURU WETLANDS, NIGERIA.

In the Hadeija-Jama' are floodplain region in northern Nigeria, more than one half of the wetlands have already been lost to drought and upstream dams. Ecosystem valuation has been used in this area to weigh the costs and benefits of development projects that would divert some more water away from the floodplain for irrigated agriculture in upstream areas. The net benefits of such a diversion are estimated at U.S.\$29 per hectare. In comparison, the floodplain, under the present flooding regime, provided U.S.\$167 per hectare in benefit to a wider range of local people engaged in farming, fishing, grazing livestock, or gathering fuelwood and other wild products—benefits that would be greatly diminished by the project.

A study of the groundwater recharge function of the wetlands confirms furthermore that the wetlands play an important role by maintaining groundwater recharge in the floodplain. Groundwater recharge supports irrigated agricultural production in the floodplain. Irrigated agriculture using water from the shallow groundwater aquifer has a value of 36,308 Naira (U.S.\$413) per hectare for the study area. A value of at least 2863 Naira or U.S.\$32.5 per farmer per dry season or U.S.\$62 per hectare is attributable to the present rate of groundwater recharge. In terms of maintaining water supply resources, the value of the recharge function is 1,146,588 Naira or U.S.\$13,029 per day for the wetlands.

Source: Adapted from Barbier & Adams,^[4] Acharya,^[6] and National Research Council.^[7]

example, the city of New York found that it could avoid spending U.S.\$6–8 billion in constructing new water treatment plants by protecting the upstate watershed that has traditionally provided these purification services. Based on this assessment, the city invested U.S.\$1.5 billion in protecting its watershed by buying land around reservoirs and providing other incentives for land use.^[8] This decision ensures that in addition to getting a relatively cheap treatment system for its water supply, the city also maintains the watershed for recreation, wildlife habitat, and other ecological benefits derived from a healthy ecosystem. Similarly, to restore balance within the Murray–Darling river basin, the Australian government plans to purchase U.S.\$3.2 billion's worth of water rights over a period of 10 years. This will restore approximately 2750 billion liters of water each year to the Murray–Darling basin river system and its wetlands. The ambitious Comprehensive Everglades Restoration Plan (CERP) aims to achieve ecological restoration by reestablishing hydrologic characteristics as close as possible to their predrainage conditions in what remains of the Florida Everglades ecosystem. The CERP, to be implemented over three decades, includes more than 40 major projects and 68 project components to be constructed at an estimated cost of U.S.\$10.9 billion in 2004 dollars (<http://www.evergladesrestoration.gov>).

A growing literature on wetland valuation has contributed to our understanding of how valuation could influence land use policy and investment decisions. WTP for the services offered by wetlands in Taiwan^[9] and Korea,^[10] mangroves in Micronesia,^[11] marine parks in the Seychelles.^[12] Studies also illustrate the regional economic impacts of continuing wetland loss. With increased

Table 1 Economic values of wetlands: Some examples.

Valuation approach	Good or service valued	References	Value	Location
Contingent valuation, replacement cost	Nitrogen abatement	[13]	US\$59/kgN reduction capacity	Gotland, Sweden
Choice Modelling WTP	improvement in waterbird habitat	[6]	\$126.63–\$198.15 per household/yr	Coorong/Murray River, Australia
Production function	Production function method to value coastal wetlands	[2]	NPV of welfare loss from reduced mangrove support for fisheries US\$1.5–2.0 million	
Market Prices	Artisanal fisheries, aquarium fish, sea cucumbers, shells, tourism	[14]	US\$2.14 million/yr	Kisite Marine National Park and Mpunguti Marine National Reserve, Kenya
Preventive expenditure	Coastal zone/shoreline protection	[15]	US\$603,248/yr	Mahe, Seychelles
Choice modeling WTP	Improvement in waterbird habitat	[16]	\$126.63–\$198.15 per household/yr	Coorong/Murray River, Australia

understanding of the benefits associated with wetland ecosystems, the rehabilitation and restoration of wetlands is also increasing across the globe. The Malian government is working to restore the inner Niger River Delta which supports the livelihoods of millions of people in Mali, and has been subjected to severe degradation. Similarly, Tanzania is looking to reverse the degradation of Lake Victoria which supports 46 million inhabitants and the world's largest freshwater fishery.

Some caveats to valuation also apply. A number of papers have attempted to value the world's ecosystems, some by extrapolating localized studies of specific resource losses to capture the impact of a global loss of such resources. Even if local values are calculated using the best possible data, extrapolating these values to all wetland resources on the planet creates problem not least because there is wide variation in the types of wetlands and their component products. Heimlich et al.^[17] presented the results of over 30 studies carried out on wetlands. After accounting for geographical variation and other differences across the studies by calculating the value over a 50-year period discounted at 6% for all the studies and then converting these values to an annual basis, they find significant variation among the different studies. For example, general recreational values ranged from U.S.\$105 to U.S.\$9859 per acre; waterfowl hunting from U.S.\$108 to U.S.\$3101 per acre; and recreational fishing from U.S.\$95 to U.S.\$28,845 per acre.

Table 1 gives a few examples of various studies and types of goods or services valued. As the figures show, there is a wide variation among the types of wetlands valued and the values attributable to them. Great caution must, therefore, be exercised when extrapolating localized studies to other geographical areas. Resource values are influenced by local consumptive and productive use patterns, market conditions, and the institutional arrangements affecting the

use of the resources. The good news is that there are hundreds of studies of wetland values (see the work by Brander et al.^[18] and Marsden Jacob Associates^[19] for an extensive review) that we can refer to for some guidance on the range of values associated with wetlands.

CONCLUSION

Benefits of some wetlands will always be difficult to quantify and measure primarily because the required scientific, technical, or economic data are difficult to obtain and also that certain intrinsic values are not measurable by existing economic valuation methods. However, as the studies reported in this entry suggest, various wetland goods and services are extremely valuable, and in many cases, a measurable economic value can be obtained. The economic value of ecological functions in sustaining the livelihood and cultures of human societies clearly cannot be disregarded in development and conservation policy because these values allow us to make informed decisions about tradeoffs and help in the conservation of natural resources to enhance human welfare. Hence, valuation studies need to be carried out wherever possible and with collaboration between economists and natural scientists to fully capture the economic and ecological linkages that make wetlands valuable.

REFERENCES

1. Mitsch, W.J.; Gosselink, J.G. The value of wetlands: Importance of scale and landscape Setting. *Ecol. Econ.* **2000**, *35*, 25–33.
2. Barbier, E.B. Valuing ecosystem services as productive inputs. *Econ. Policy* **2007**, *22*, 177–229.

3. Freeman, A.M. *The Measurement of Environmental and Resource Values: Theory and Methods*; Resources for the Future: Washington, DC, 1993.
4. Barbier, E.B.; Adams, W.M.; Kimmage, K. An economic valuation of wetland benefits. In *The Hadejia–Nguru Wetlands: Environment, Economy and Sustainable Development of a Sahelian Floodplain Wetland*; Hollis, G.E., Adams, W.M., Aminu-Kano, M., Eds.; IUCN: Gland, 1993; 191–209.
5. Navrud, S.; Mungatana, E. Environmental valuation in developing countries: The recreation value of wildlife viewing. *Ecol. Econ.* **1994**, *11*, 135–151.
6. Acharya, G.; Barbier, E.B. Valuing groundwater recharge through agricultural production in the Hadejia–Nguru wetlands in Northern Nigeria. *Agr. Econ.* **2000**, *22*, 247–259.
7. Acharya, G. Valuing the hidden hydrological services of wetland ecosystems. *Ecol. Econ.* **2000**, *35*, 63–74.
8. National Research Council (NRC). *Watershed Management for Potable Water Supply: Assessing the New York City Strategy*. Committee to review the New York city watershed management strategy; National Academy Press: Washington, DC, 2000.
9. Hammitt, J.K.; Liu, J.-T.; Liu, J.-L. Contingent valuation of a Taiwanese wetland. *Environ. Develop. Econ.* **2001**, *6* (2), 259–268.
10. Kwak, S.-J.; Yoo, S.-H.; Lee, C.-K. Valuation of the Woopo wetland in Korea: A contingent study. *Environ. Develop. Econ.* **2007**, *12* (2), 323–328.
11. Naylor, R.; Drew, M. Valuing mangrove resources in Kosrae, Micronesia. *Environ. Develop. Econ.* **1998**, *3*, 471–490.
12. Mathieu, L.F.; Langford, I.H.; Kenyon, W. Valuing marine parks in a developing country: A case study of the Seychelles. *Environ. Develop. Econ.* **2003**, *8* (2), 373–390.
13. Gren, I.M.; Folke, C.; Turner, K.; Bateman, I. Primary and secondary values of wetland ecosystems. *Environ. Resour. Econ.* **1994**, *4* (1), 55–74.
14. Emerton, L.; Tessema, Y. *Economic Constraints to the Management of Marine Protected Areas: The Case of Kisite Marine National Park and Mpunguti Marine National Reserve*; IUCN—The World Conservation Union: Kenya, 2001.
15. Emerton, L. *Economic Tools for Valuing Wetlands in Eastern Africa*; IUCN—The World Conservation Union: Kenya, 1998.
16. MacDonald, D.H.; Morrison, M.D.; Rose, J.M.; Boyle, K.J. Valuing a multistate river: The case of the River Murray. *Aust. J. Agri. Res. Econ.* **2011**, *55* (3), 374–392.
17. Heimlich, R.E.; Wiebe, K.D.; Claassen, R.; Gadsby, D.; House, R.M. *Wetlands and Agriculture: Private Interests and Public Benefits*. Agriculture Economic Report No. 765; Resource Economics Division, Economic Research Service, USDA: Washington, DC, 1998.
18. Brander, L.M.; Florax, R.J.G.M.; Vermaat, J.E. The empirics of wetland valuation: A comprehensive summary and a meta-analysis of the literature. *Environ. Res. Econ.* **2006**, *33*, 223–250.
19. Marsden Jacob Associates. *Literature Review of the Economic Value of Ecosystem Services that Wetlands Provide Final Report prepared for the Department of Sustainability, Environment, Water, Population and Communities*; 2012. Available at <https://www.environment.gov.au/system/files/resources/fb918be6-fd56-43a5-9e61-f4e63d455e0c/files/review-ecosystem-services-report.pdf> (accessed October 20, 2015).

Wetlands: Gaseous Emissions

S.K. Nag

Fishery Resource and Environmental Management Division, Central Inland Fisheries Research Institute, Barrackpore, Indian Council of Agricultural Research (ICAR), Kolkata, India

Abstract

Wetlands are one of the very important ecosystems known for their ability to act as a carbon (C) sink. Nearly one-third of the total storage of C in earth's soil is stored in wetlands and they play a vital role in global C-cycle. However, wetlands are also a potential source of greenhouse gas (GHG) emission, particularly methane (CH₄). About 20–25% of global CH₄ emission is contributed by wetlands. CH₄ emission from wetlands is dependent on several abiotic and biotic factors like temperature, moisture, water level, redox potential, type of wetland, C-content, presence or absence of vegetation, type of vegetation, comparative ratios of methanogenic and methanotrophic bacteria, etc. Emission of carbon dioxide (CO₂) from wetlands depends on the nature of wetlands and their management practices. There are more production and emission of CO₂ due to aerobic decomposition of organic matter favored by seasonal drying of wetlands as a result of less rainfall, prolonged draught, and harvesting of peat. Wetlands are, however, not a significant source of nitrous oxide (N₂O), the most potent GHG having 310 times more global warming potential than CO₂. But, if moisture level in the wetlands drops significantly when they resemble upland soil and also if water dries up in seasonal wetlands or is drained off in managed systems, there may be more emission of N₂O than usual.

INTRODUCTION

Wetlands are areas of marsh, fen, peatland, or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish, or salty, including areas of marine waters, the depth of which at low tide does not exceed 6 m.^[1] They are actually the interface of aquatic and terrestrial environments and most important natural resource and ecosystems on earth. The extent of wetlands ranges from 7 to 10 million km² or about 5–8% of the land surface of the earth, distributed in all climatic zones ranging from the tropics to the tundra. There are different types of wetlands like peatlands (bog and fen), marsh, swamp, etc.

Although wetlands cover 5–8% of earth's land area, they account for almost one-third of total terrestrial carbon (C) stock^[2,3] and thus play an important role in global C-cycle. They have large potential for sequestering C in their soils because of a positive balance due to a high primary productivity and a less rate of decomposition. But at the same time wetlands can also act as source of greenhouse gas (GHG). Thus, the balance of C uptake and emissions determines whether a wetland is a source or sink of C.

GLOBAL WARMING AND GHGS

Global warming, as is evident from increase in mean annual temperature over the last few decades at the rate of 0.15°C per decade since 1975,^[4,5] is the result of

increased atmospheric concentration of GHGs—three of the most important being carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O). Atmospheric concentration of CO₂ has increased by around 43% from 270–280 ppm in late 1700s to around 400 ppm in 2014.^[6] CH₄ concentration increased 2.5 times from 722 ppb during preindustrial times to 1,800 ppb in 2011.^[4] Since the late 1990s, its concentration in the troposphere is increasing at the rate of 1.1% per year, which is a serious concern.^[7] The concentration of N₂O increased by 16% from 270 ppb at pre-1750 to the level of 325 ppb.^[8] These gases absorb infrared radiation that may lead to increase in mean surface temperature and potential change in climate. About 70% of the world GHG emission comprises of CO₂, followed by CH₄ (20%), N₂O (8%), and other gases (3%). Although CH₄ and N₂O are present at lower concentration in the atmosphere but their global warming potential (GWP) is 21 times higher for CH₄ and 310 for N₂O than that of CO₂.

WETLANDS AND GHG EMISSION

Wetlands are the world's largest natural source of CH₄^[9] and also a source of N₂O but are a potentially important sink of atmospheric CO₂.^[10] Wetlands contribute about 20–25% of the global CH₄ emissions. When submerged rice fields are included in the wetlands definition, then together they are estimated to emit 33–37% of global CH₄.^[11,12] Wetlands are the largest natural CH₄ source,

whereas ruminants, paddy rice cultivation, and fossil fuel use are the largest anthropogenic sources.

Wetlands are also a source of CO₂ when decomposition of organic matter (OM) exceeds production. In addition, seasonal wetlands and those which are sometimes drained for agricultural and other purposes in many parts of the world become a significant source of atmospheric CO₂.^[13]

N₂O, the most potent GHG, results from denitrification that is generated in waterlogged soils by the availability of C and nitrogen (N). Wetlands receiving excessive amounts of nitrates as a result of anthropogenic activities have increased the rate of denitrification and, therefore, may be significant source of N₂O to the atmosphere.

The gaseous C budget of a wetland depends on the CO₂ uptake during photosynthesis and the amount of C lost as either CO₂ from plant and soil respiration or CH₄ emitted by methanogenesis.^[14–16]

Gas fluxes to and from the atmosphere in wetlands can occur via three pathways: the gas-filled tissue or aerenchyma of emergent vegetation, bubbles rising from the substrate to the water surface (ebullition), and hydrodynamic transport of dissolved gas through the surface water.

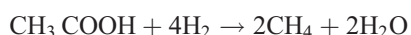
PRODUCTION OF CH₄ IN WETLANDS

Wetlands contribute 109–145 Tg CH₄ yr⁻¹, i.e., 20–26% of global CH₄ emission (550 Tg CH₄ yr⁻¹) with boreal and tropical wetlands contributing the most.^[2,17] However, the estimates suggest that wetlands may contribute about 180 Tg CH₄ yr⁻¹ 76% of which comes from tropical wetlands.^[18,19] Submerged rice fields, which are also sometimes included in the wetlands, emit about 60–80 Tg CH₄ yr⁻¹ (1 Tg = 10¹² g).

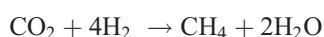
The most comprehensive assessment of global sources of atmospheric CH₄ and their dynamics conducted by Intergovernmental Panel on Climate Change suggests that natural wetlands are responsible for approximately 21% of global CH₄ emissions.

Because of its high GWP, estimation of CH₄ from different sources is one of the urgent tasks in addressing the problem of climate change. Being a major emission source of CH₄, precise estimation of its emission requires credible assessment of wetlands in global or continental scale.

CH₄ is produced in wetlands when labile fraction of OM undergoes mineralization under anaerobic condition by microorganisms called Archaea. CH₄-producing bacteria or methanogens use low-molecular-weight compounds like acetate coming from anaerobic fermentation of OM and ferment them into CH₄ which is the end product of anaerobic decomposition of soil OM.



Alternatively CH₄ is produced from reduction of CO₂.



Methanogenesis requires strictly anaerobic and extremely reduced condition, i.e., redox potential below -200 mV. The presence of other electron (e)-acceptors like NO₃⁻, Fe³⁺, and SO₄²⁻ can inhibit CH₄ production.^[20]

Emission of CH₄ from wetlands to the atmosphere does not depend only on CH₄ production but also on its oxidation which takes place in the upper oxidized soil layer. During convective flow or diffusive transport of CH₄, produced in the lower anaerobic and highly reduced horizon, to the upper layer, the gas comes across with obligate aerobic methanotrophic bacteria present in aerobic soil layer which use molecular oxygen to oxidize CH₄ to CO₂. Thus, emission of CH₄ to the atmosphere is limited.^[12,20] Methanotrophs are estimated to consume about 30 Tg CH₄ yr⁻¹,^[12] an amount close to net annual increase in atmospheric CH₄.^[21]

DRIVERS OF CH₄ PRODUCTION FROM WETLANDS

A large number of physical and biological controlling factors have been identified that are responsible for the huge spatial and temporal variation in CH₄ fluxes from the soil into the atmosphere. CH₄ production, its oxidation, and ultimate emission from wetlands are controlled by several biological, chemical, and physical factors like temperature, moisture, water level, redox potential, time of the day, wetland types, C-content, vegetation, etc.

Temperature

Soil temperature is an important factor, either alone or in combination with other environmental factors, and affects CH₄ flux.^[22] A positive correlation exists between soil temperature and CH₄ emission in temperate freshwater wetlands.^[23–26] In their study on seasonal dynamics of CH₄ flux from natural and constructed wetlands, Singh et al.^[7] observed maximum CH₄ flux during summer season followed by that in rainy and winter seasons which is attributable to a positive correlation with temperature as higher temperature exists in the summer followed by that in rainy and winter seasons. Based on the difference in summer and winter temperatures, Singh et al. reported a Q10 value of 3 which is close to Q10 value of 2.5–3.5 for CH₄ production obtained by others.

Moisture and Water Level

Both the water table height and the soil temperature that influence production and oxidation rates have routinely been identified as main drivers of the CH₄ flux in field studies for more than 20 years. Soil moisture and water levels influence CH₄ flux. Many studies have shown that soil water content is one of the most important factors

influencing GHG fluxes because of its direct effect on microbial activity.^[27,28]

Redox Potential

The oxidation–reduction potential of wetlands is an important factor in determining the GHG emission. Submerged soils are characterized as lacking sufficient oxygen to act as e-acceptor for microbial respirations. A redox potential of –150 mv is essential to methanogenesis by archaeal bacteria.

Diurnal Variation

CH₄ emissions from wetlands vary significantly not only on a seasonal basis but also diurnally.^[29] Ding and Cai^[30] observed that CH₄ emission from freshwater marshes and swamps varied greatly at different times of the day showing increasing rate during the morning reaching a peak in the afternoon and then decreasing in the evening. This diurnal variation becomes more important when plants are growing in the wetland^[31] and the ability of the plant species to transport gas increases.^[32]

The diurnal variation in CH₄ flux in *Carex*-dominated wetland is related to soil temperature variation. Hirota et al.^[31] did not observe any clear relationship between daily CH₄ flux and soil temperature in three emergent plant-dominated zones (*Hippuris* dominated, *Scirpus* dominated, and *Carex* dominated) and one submerged plant

(*Potamogeton*) dominated. In many studies, the relationship between temperature and CH₄ flux from wetlands is reported to be complex ranging from linear, log-linear, or exponential to no correlation.^[33]

Type of Wetland

Most of the CH₄ emission studies have been conducted in peatlands (i.e., bogs, fens, and freshwater marshes). Deep ponds have higher CH₄ flux rates than other types of boreal wetlands. In case of bogs and fens, neutral fens emit more CH₄ than acid fens and bogs. Mangrove wetlands and coastal salt marshes are less emitter than freshwater swamps^[19] which might be due to the fact that salinity inhibits methanogenesis to some extent. Variation in CH₄ emission from diverse types of tropical and subtropical wetlands is depicted in Table 1.

Availability of Organic C

Since the substrate for methanogenesis is organic C compounds in the soil that come from photosynthesis via root exudation, root turnover, and litter production, net ecosystem production is the master variable for the control of CH₄ emissions from wetlands. CH₄ production rates are linearly correlated with water soluble C in the soil or with readily mineralizable C in both field and laboratory conditions. Higher CH₄ emission rates are

Table 1 Examples of rates of GHG flux from different wetlands.

Location	Wetland type	CH ₄ (g m ⁻² yr ⁻¹)	CO ₂ (g m ⁻² yr ⁻¹)	N ₂ O (g m ⁻² yr ⁻¹)	References
Lucknow, India	Subtropical		—	—	Singh, Kulshreshtha, and Agnihotri ^[7]
	Flooded forest	42			
	Macrophyte mat	9			
Venezuela	Tropical floodplain open	9	—	—	Smith et al. ^[34]
Louisiana, U.S.A.	Subtropical bottomland hardwood forest	249	—	—	Yu, Faulkner, and Baldwin ^[35]
South China	Tropical mangrove	1.67–38.5	2015–4155	0.05–9.2	Chen, Tam, and Ye ^[36]
Australia	Tropical mangrove	52–330	—	0.25–1.77	Allen et al. ^[37]
Earth, Costa Rica	Tropical	—	—	—	Nahlik and Mitsch ^[38]
Le Selva, Costa Rica	Forested marsh	44	—	—	
Palo Verde, Costa Rica	Rainforest swamp	293	—	—	
	Alluvial marsh	350	—	—	
		—	—	—	
Canada	Coastal salt marsh	0.17–0.31	3486–4249	0.01–0.11	Chmura, Kellman, and Guntenspergen ^[39]
Wales, U.K.	Coastal salt marsh	78.8	2917–3679	—	Ford et al. ^[40]
NE China	Freshwater inland marsh	65	3155	0.11	Yang et al. ^[41]
Ohio, U.S.A.	Temperate riverine open	102	508.3	1.93	Nag et al. (to be published)

observed from wetlands having higher organic C in the sediment.^[7]

Effect of Vegetation and Plant Species on CH₄ Dynamics

Vegetation is one of the most important drivers of CH₄ dynamics in wetland systems. The mere presence or absence of vegetation as a gas-channel heavily impacts the magnitude of CH₄ flux from sediments to the atmosphere. CH₄ emission from the vegetated surface is many times higher than that from the unvegetated surface of the same water body.^[7] Removal of sedges from arctic wetland can reduce CH₄ emissions and increase accumulation of CH₄ in flooded soils. Sedges provided the main transport pathway for CH₄ release into the atmosphere. Over 85% of net CH₄ emissions from the *Scirpus* and *Phragmites* wetland can be a result of CH₄ releasing to the atmosphere through the root-shoot system.^[42]

CH₄ dynamics varies across plant species growing in the wetlands and their growth characteristics. Presence of some species causes more emission of CH₄ than others even within the same plant families, and for some plants within that family, there may be no detectable emission.^[43] In general, CH₄ emission decreases with increase in plant biomass.

Microbial community analysis of rice rhizospheres indicates high enrichment of a diverse pool of methanogenic archaeal populations.^[44] Rhizosphere acetate levels can differ greatly among plant species and is considered an important precursor to CH₄ production.^[45] Plants in anoxic soil environments significantly alter net CH₄ emissions by increasing its diffusion or flow from flooded sediments via physical traits.^[42]

Effect of Elevated Level of CO₂

CH₄ emissions from wetlands increase in response to higher concentration of CO₂ in the atmosphere.^[46,47] The possible reason for this relation between the two GHGs may be due to increased exudation of plant-derived C in the presence of increased CO₂ or it may be a result of other plant traits that influence net CH₄ production. Plants stimulated by CO₂ fertilization can increase primary production, resulting in higher exudation of organic C.^[46,48] The increase in root-derived C, in turn, directly stimulates the activity and growth of acetoclastic methanogens which produce CH₄.^[49]

Most of the plant physical traits which are affected by elevated CO₂ also impact CH₄ flux. Therefore, changes in the parameters that allow for greater diffusion and flow of CH₄ from soil, through the plant, and into the atmosphere, can increase net CH₄ emissions. Increase in biomass under elevated CO₂ may raise emissions of CH₄ by increasing the rate or volume of CH₄ ventilated through the plant, while greater root biomass could allow more diffusion of

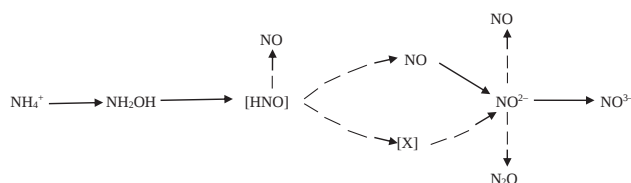
dissolved CH₄ into the roots and aerenchyma where CH₄ ventilation proceeds.^[46]

The development of aerenchyma tissues of aquatic plants provides conduits for CH₄ and supplies their roots with oxygen for CH₄ consumption. Methanogens belong to the domain Archaea, which requires strict anaerobiosis and low redox potentials. However, soil-plant systems also can oxidize a large fraction of the CH₄ produced by methanotrophs inhabiting the root zone.

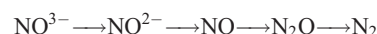
N₂O Emission

N₂O is an important GHG with an irradiative forcing effect of 310 times that of CO₂ and a life time in the troposphere of approximately 120 yr. Part of N₂O is converted to NO in the stratosphere and so contributes to depletion of ozone. Emission of NO and N₂O is the result of microbial nitrification and denitrification in soils. Nitrification is the main source of N₂O emission under aerobic condition and denitrification under anaerobic condition. Both nitrification and denitrification processes in soil can occur simultaneously and the production of N₂O depends on the balance between these two microbial processes.^[50]

Nitrification



Denitrification



About 55% of global N₂O is emitted from natural ecosystem.^[51] Ocean, estuaries, and rivers account for 5.4 Tg N yr⁻¹ out of estimated emission of around 12.1 Tg N yr⁻¹ from all natural sources. Wetlands are not an important source of N₂O emission unless water levels in these systems drop significantly in which case they behave as upland soil and may emit more N₂O.^[52]

In continuously submerged wetlands under sufficiently reducing condition and available organic substrates, rates of nitrification and subsequent denitrification can be substantial but the denitrification proceeds toward N₂ as the end product and little N₂O is produced en route. But if the wetland is seasonal (i.e., water dries up or drained off in managed systems), there may be more N₂O emission due to increase in the availability of oxygen.

Major environmental factors controlling the N₂O emission are soil organic carbon (SOC) content, vegetation type, soil pH, soil temperature, water content, and soil N particularly NO₃⁻, nitrite, and ammonia. Emission of N₂O

increases with increase in SOC. As soil pH increases and bulk density decreases, N₂O emission decreases. The presence of vegetation also affects the emission of N₂O from wetlands just like CH₄ emission. Without inundation, however, N₂O flux is similar in both the wetlands, vegetated and without vegetation.^[53]

CONCLUSION

Wetlands are among the most unique natural resources on the earth providing a wide array of ecosystem services and are a C sink over long term and thus partly offsetting anthropogenic emissions. But there are uncertainties in the global wetland area, the amount of C stored in their soils as well as the quantity of GHG flux from the wetlands. Few data on rates of GHG flux from different wetlands across the world are given in Table 1.

Most of the studies on GHG emission from wetlands have been conducted in temperate regions which are different in characteristics, climatic, and environmental conditions compared to wetlands in the tropics. Also, there is a lack of estimates of N₂O and CH₄ fluxes from subtropical and tropical climates. Although models developed for prediction of emission of GHGs (e.g., CH₄ and N₂O) work with reasonable accuracy at the site or system for which they were developed, their potential for application in other systems with different management practices has not been widely validated.

Climate change impacts the GHG emission. Rising sea level and ingress of sea water increase the salinity of water in the wetlands of coastal and estuarine belts and reduce CH₄ emission. Similarly changes in land use and the area under wetland or changes in the management practices (like intermittent drying or wetting) affect the volume of GHG emission and the relative proportion of CO₂ and CH₄ produced and emitted from wetlands.

ACKNOWLEDGMENT

The author is highly thankful to Dr. Rattan Lal, Distinguished Professor, The Carbon Management and Sequestration Center, School of Environment and Natural Resources, The Ohio State University, Columbus, Ohio, U.S.A. for providing valuable guidance in the development of manuscript.

REFERENCES

1. Ramsar Convention. *The Ramsar Convention on Wetlands*; Ramsar Convention: Iran, 1971.
2. Mitsch, W.J.; Gooselink, J.G. *Wetlands of the World*. In *Wetlands*, 4th Ed.; John Wiley Sons, Inc.: New York, 2007; 43–104.
3. Lal, R. Carbon sequestration. *Philos. T. R. Soc. B.* **2008**, *363* (1492), 815–830. DOI:10.1098/rstb.2007.2185.
4. IPCC. *Climate Change 2013. Climate Change Impacts, Adaptation and Vulnerability* Working Group IIIPCC: Geneva, 2013.
5. WMO. *World Meteorological Organization Provisional Statement on Status of Climate in 2013*. Available at www.wmo.int (accessed July 2014).
6. NOAA/ESRL. *Trends in Carbondioxide. National Oceanic and Atmospheric Administration. Earth System Research Laboratory Global Monitoring Division*. Available at www.esrl.noaa.gov (accessed July 2014).
7. Singh, S.N.; Kulshreshtha, K.; Agnihotri, S. Seasonal dynamics of methane emission from wetlands. *Chemosphere Glob. Change Sci.* **2000**, *2* (1), 39–46. DOI: 10.1016/S1465-9972(99)00046-X.
8. Blasing, T.J. Recent green house gas concentrations. *Carbondioxide Information Analysis Center* **2014**. DOI: 10.3334/CDIAC/atg.032. Available at www.cdiac.ornl.gov/pns/current_ghg.html.
9. Li, C.; Cui, J.; Sun, G.; Trettin, C. Modeling impacts of management on carbon sequestration and trace gas emissions in forested wetland ecosystems. *Environ. Manage.* **2004**, *33* (Suppl 1), S176–S186. DOI:10.1007/s00267-003-9128-z.
10. Denman, K.L.; Brasseur, G.; Chidthaisong, A.; Ciais, P.; Cox, P.M.; Dickinson, R.E.; Hauglustaine, D.; Heinze, C.; Holland, E.; Jacob, D.; Lohmann, U.; Ramachandran, S.; da Silva Dias, P.L.; Wofsy, S.C.; Zhang, X. Couplings between changes in the climate system and biogeochemistry. In *Climate Change 2007—The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*; Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L., Eds.; Cambridge University Press: New York, 2007; 499–587.
11. Megonigal, J.P.; Hines, M.E.; Visscher, P.T. Anaerobic metabolism linkages to trace gases and aerobic processes. In *Biogeochemistry*; Schlesinger, W.H., Ed.; Elsevier-Pergamon: Oxford, 2004; 317–424.
12. Whalen, S.C. Biogeochemistry of methane exchange between natural wetlands and the atmosphere. *Environ. Eng. Sci.* **2005**, *22* (1), 73–94. DOI:10.1089/ees.2005.22.73.
13. Beringer, J.; Livesley, S.J.; Randle, J.; Hutley, L.B. Carbon dioxide fluxes dominate the greenhouse gas exchanges of a seasonal wetland in the wet-dry tropics of northern Australia. *Agr. For. Meteorol.* **2013**, *182–183*, 239–247. DOI:10.1016/j.agrformet.2013.06.008.
14. Whiting, G.J.; Chanton, J.P. Greenhouse carbon balance of wetlands methane emission versus carbon sequestration. *Tellus B.* **2001**, *53* (5), 521–528.
15. Cornell, J.A.; Craft, C.B.; Megonigal, J.P. Ecosystem gas exchange across a created salt marsh chronosequence. *Wetlands* **2007**, *27* (2), 240–250. DOI:10.1672/0277-5212(2007)27[240:EGEAAAC]2.0.CO;2.
16. Yvon-Durocher, G.; Montoya, J.M.; Woodward, G.; Jones, J.I.; Trimmer, M. Warming increases the proportion of primary production emitted as methane from freshwater mesocosms. *Glob. Change Biol* **2011**, *17* (2), 1225–1234. DOI: 10.1111/j.1365-2486.2010.02289.x.
17. IPCC. *Climate Change 2007. In Climate Change 2007 The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*; Solomon, S., Qin, D., Manning,

- M, Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L., Eds.; Cambridge University Press: Cambridge, 2007; 19–92.
18. Bergamaschi, P.; Frankenberg, C.; Meirink, J.F.; Krol, M.; Dentener, F.; Wagner, T.; Platt, U.; Kaplan, J.O.; Körner, S.; Heimann, M. Satellite cartography of atmospheric methane from SCIAMACHY on board ENVISAT: 2. Evaluation based on inverse model simulations. *J. Geophys. Res.* **2007**, *112* (D2), D02304. DOI:10.1029/2006JD007268.
 19. Mitsch, W.J.; Nahlik, A.M.; Wolski, P.; Bernal, B.; Zhang, L.; Ramberg, L. Tropical wetlands: Seasonal hydrologic pulsing, carbon sequestration, and methane emissions. *Wetl. Ecol. Manag.* **2010**, *18*, 573–586. DOI:10.1007/s11273-009-9164-4.
 20. Le Mer, J.; Roger, P. Production, oxidation, emission and consumption of methane by soils: A review. *Eur J Soil Biol.* **2001**, *37* (1), 25–50. DOI:10.1016/S1164-5563(01)01067-6.
 21. IPCC. *Climate Change 2001. The Scientific Basis. Published for the Intergovernmental Panel on Climate Change*; Cambridge University Press: Cambridge, 2001.
 22. Gedney, N.; Cox, P.M.; Huntingford, C. Climate feedback from wetland methane emissions. *Geophys Res Lett.* **2004**, *31* (20), 1–4. DOI:10.1029/2004GL020919.
 23. Altor, A.E.; Mitsch, W.J. Pulsing hydrology, methane emissions, and carbondioxide fluxes in created marshes: A 2-year ecosystem study. *Wetlands* **2008**, *28* (2), 423–438. DOI:10.1672/07-98.1.
 24. Nahlik, A.M.; Mitsch, W.J. Methane emissions from created riverine wetlands. *Wetlands* **2010**, *30* (4), 783–793. DOI: 10.1007/s13157-010-0038-6.
 25. Nahlik, A.M.; Mitsch, W.J. Erratum to: Methane emissions from created riverine wetlands. *Wetlands* **2011**, *31* (2), 449–450. DOI:10.1007/s13157-011-0164-9.
 26. Sha, C.; Mitsch, W.J.; Mander, Ü.; Lu, J.; Batson, J.; Zhang, L.; He, W. Methane emissions from freshwater riverine wetlands. *Ecol. Eng.* **2011**, *37* (1), 16–24. DOI: 10.1016/j.ecoleng.2010.07.022.
 27. Ambus, P.; Zechmeister-Boltenstern, S.; Butterbach-Bahl, K. Sources of nitrous oxide emitted from European forest soils. *Biogeosciences* **2006**, *3* (2), 135–145. DOI: 10.5194/bg-3-135-2006.
 28. Tang, X.; Liu, S.; Zhou, G.; Zhang, D.; Zhou, C. Soil-atmospheric exchange of CO₂, CH₄, and N₂O in three subtropical forest ecosystems in southern China. *Glob. Change. Biol.* **2006**, *12* (3), 546–560. DOI:10.1111/j.1365-2486.2006.01109.x.
 29. Brix, H.; Sorrel, B.K.; Lorenzen, B. Are phragmites-dominated wetlands a net source or net sink of greenhouse gases? *Aquat Bot.* **2001**, *69* (2–4), 313–324. DOI:10.1016/S0304-3770(01)00145-0.
 30. Ding, W.X.; Cai, Z.C. Methane emission from natural wetlands in China: Summary of years 1995–2004 studies. *Pedosphere* **2007**, *17* (4), 475–486. DOI:10.1016/S1002-0160(07)60057-5.
 31. Hirota, M.; Tang, Y.; Hu, Q.; Hirata, S.; Kato, T.; Mo, W.; Cao, G.; Mariko, S. Methane emissions from different vegetation zones in a Qinghai-Tibetan plateau wetland. *Soil Biol. Biochem.* **2004**, *36* (5), 737–748. DOI:10.1016/j.soilbio.2003.12.009.
 32. Ding, W.; Cai, Z.; Tsuruta, H.; Li, X. Key factors affecting spatial variation of methane emissions from freshwater marshes. *Chemosphere* **2003**, *51* (3), 167–173. DOI: 10.1016/S0045-6535(02)00804-4.
 33. Rask, H.; Schoenau, J.; Anderson, D. Factors influencing methane flux from a boreal forest wetland in Saskatchewan, Can. *Soil Biol. Biochem.* **2002**, *34* (4), 435–443. DOI: 10.1016/S0038-0717(01)00197-3.
 34. Smith, L.K.; Lewis, W.M., Jr.; Chanton, J.P.; Cronin, G.; Hamilton, S.K. Methane emissions from the Orinoco River floodplain, Venezuela. *Biogeochemistry* **2000**, *51* (2), 113–140. DOI:10.1023/A:1006443429909.
 35. Yu, K.; Faulkner, S.P.; Baldwin, M.J. Effect of hydrological conditions on nitrous oxide, methane, and carbon dioxide dynamics in a bottomland hardwood forest and its implication for soil carbon sequestration. *Glob. Change Biol.* **2008**, *14* (4), 798–812. DOI:10.1111/j.1365-2486.2008.01545.x.
 36. Chen, G.C.; Tam, N.F.Y.; Ye, Y. Summer fluxes of atmospheric greenhouse gases N₂O, CH₄ and CO₂ from mangrove soil in South China. *Sci. Total Environ.* **2010**, *408* (13), 2761–2767. DOI:10.1016/j.scitotenv.2010.03.007.
 37. Allen, D.; Dalal, R.C.; Rennenberg, H.; Schmidt, S. Seasonal variation in nitrous oxide and methane emissions from subtropical estuary and coastal mangrove sediments, Australia. *Plant Biol.* **2011**, *13* (1), 126–133. DOI:10.1111/j.1438-8677.2010.00331.x.
 38. Nahlik, A.M.; Mitsch, W.J. Methane emissions from tropical freshwater wetlands located in different climatic zones of Costa Rica. *Glob. Change Biol.* **2011**, *17* (3), 1321–1334. DOI:10.1111/j.1365-2486.2010.02190.x.
 39. Chmura, G.L.; Kellman, L.; Guntenspergen, G.R. The greenhouse gas flux and potential global warming feedbacks of a northern macrotidal and microtidal salt marsh. *Environ. Res. Lett.* **2011**, *6* (4), 04416 DOI:10.1088/1748-9326/6/4/044016.
 40. Ford, H.; Garbutt, A.; Jones, L.; Jones, D.L. Methane, carbon dioxide and nitrous oxide fluxes from a temperate salt marsh: Grazing management does not alter global warming potential. *Estuar. Coast. Shelf. S.* **2012**, *113*, 182–191. DOI:10.1016/j.ecss.2012.08.002.
 41. Yang, J.; Liu, J.; Hu, X.; Li, X.; Wang, Y.; Li, H. Effect of water table level on CO₂, CH₄ and N₂O emissions in a freshwater marsh of northeast china. *Soil Biol. Biochem.* **2013**, *61*, 52–60. DOI:10.1016/j.soilbio.2013.02.009.
 42. Van der Nat, F.J.; Middelburg, J.J. Methane emissions from tidal freshwater marshes. *Biogeochemistry* **2000**, *49* (2), 103–121. DOI:10.1023/A:1006333225100.
 43. Kao-Kniffin, J.; Freyre, D.S.; Balser, T.C. Methane dynamics across wetland plant species. *Aqua. Bot.* **2010**, *93* (2), 107–113. DOI:10.1016/j.aquabot.2010.03.009.
 44. Wantanabe, T.; Kimuram, M.; Asakawa, S. Diversity of methanogenic archaeal communities in Japanese paddy field ecosystem, estimated by denaturing gradient gel electrophoresis. *Biol. Fertil. Soils* **2010**, *46*, 343–353.
 45. Strom, L.; Ekberg, A.; Mastepanov, M.; Rojle Christensen, T. The effect of vascular plants on carbon turnover and methane emissions from a tundra wetland. *Glob. Change Biol.* **2003**, *9* (8), 1185–1192. DOI:10.1046/j.1365-2486.2003.00655.x.

46. Inubushi, K.; Cheng, W.G.; Aonuma, S.; Hoque, M.M.; Kobayashi, K.; Miura, S.; Yong Kim, H.; Okada, M. Effect of free-air CO₂ enrichment (FACE) on CH₄ emission from a rice paddy field. *Glob. Change Biol.* **2003**, *9* (10), 1458–1464. DOI:10.1046/j.1365-2486.2003.00665.x.
47. Kao-Kniffin, J.; Freyre, D.S.; Balser, T.C. Increased methane emissions from an invasive wetland plant under elevated carbon dioxide levels. *Appl. Soil. Ecol.* **2011**, *48* (3), 309–312. DOI:10.1016/j.apsoil.2011.04.008.
48. Freeman, C.; Fenner, N.; Ostle, N.J.; Kang, H.; Dowrick, D.J.; Reynolds, B.; Lock, M.A.; Sleep, D.; Hughes, S.; Hudson, J. Export of dissolved organic carbon from peatlands under elevated carbon dioxide levels. *Nature* **2004**, *430* (6996), 195–198. DOI:10.1038/nature02707.
49. Lu, Y.; Wassmann, R.; Neue, H.U.; Huang, C. Dynamics of dissolved organic carbon and methane emissions in a flooded rice soil. *Soil Sci. Soc. Am. J.* **2000**, *64* (6), 2011–2017. DOI:10.2136/sssaj2000.6462011x.
50. Butterbach-Bahl, K.; Kesik, M.; Miehe, P.; Papen, H.; Li, C. Quantifying the regional source strength of N-trace gases across agricultural and forest ecosystems with process based models. *Plant Soil.* **2004**, *260* (1/2), 311–329. DOI:10.1023/B:PLSO.0000030186.81212.fb.
51. Forster, P.; Ramaswamy, V.; Artaxo, P.; Bernsten, T.; Betts, R.; Fahey, D.W.; Haywood, J.; Lean, J.; Lowe, D.C.; Myhre, G.; Nganga, J.; Prinn, R.; Raga, G.; Schulz, M.; Van Dorland, R. Changes in atmospheric constituents and in radiative forcing. In *Climate Change 2007 The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*; Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L., Eds.; Cambridge University Press: Cambridge, 2007; 129–234.
52. USEPA. *Methane and Nitrous Oxide Emissions from Natural Sources*; United States Environment Protection Agency Office of Atmospheric Programs: Washington, DC, 2010.
53. Hernandez, M.E.; Mitsch, W.J. Influence of hydrologic pulses, flooding frequency, and vegetation on nitrous oxide emissions from created riparian marshes. *Wetlands* **2006**, *26* (3), 862–877. DOI:10.1672/0277-5212(2006)26[862:IOHPFF]2.0.CO;2.

Wetlands: Methane Emission

Anna Ekberg

Department of Ecology, Plant Ecology, Climate Impacts Group, Lund University, Lund, Sweden

Torben Røjle Christensen

Climate Impacts Group, Department of Ecology, Lund University, Lund, Sweden

Abstract

Methane (CH₄) is a trace gas that plays an important role in atmospheric chemistry and the energy balance of the earth. Because of the substantial contribution from natural wetlands, knowledge about the dynamics leading to CH₄ formation and emission is of great importance to understand how these ecosystems interact with the climate and how they would respond to climate change.

INTRODUCTION

Methane (CH₄) is a radiatively active trace gas that plays an important role in atmospheric chemistry and the energy balance of the earth. Its contribution to the greenhouse effect is about 22%, which can be compared to carbon dioxide (CO₂) which contributes approximately 65% to the climate forcing of all long-lived greenhouse gases (excluding water vapor).^[1] In addition to the direct warming effects, where infrared radiation is absorbed and returned to the earth's surface, chemical and photochemical reactions with CH₄ in the troposphere and stratosphere indirectly cause greenhouse warming. The preindustrial atmospheric concentration of CH₄ was about 0.75 ppmv (parts per million by volume), and since then it has more than doubled to approximately 1.73 ppmv.^[1] Anthropogenic sources of CH₄ include cattle (about 15% of the annual CH₄ release), rice paddies (20%), coal mining and oil production (14%), biomass burning (10%), natural gas leaks, and landfills and sewage disposal, while the major natural sources include wetlands (20–25%) and termites (5%).^[2] Because of the substantial contribution from natural wetlands, knowledge about the dynamics leading to CH₄ formation and emission is of great importance to understand how these ecosystems interact with the climate and how they would respond to climate change.

CH₄ PRODUCTION AND CONSUMPTION

CH₄ is produced by strictly anaerobic archaeobacteria that are limited to the use of only a few simple substrates for biosynthesis and energy production.^[3] If available carbon compounds are not directly supplied, they depend on other groups of microorganisms for the initial breakdown of more complex organic structures into simpler molecules.

The reduction of CO₂ with hydrogen is a common pathway to CH₄ formation, but acetate and formate are also important precursors of CH₄ in natural environments. Growth of methanogenic bacteria further requires a very low redox potential (E_h below -400 mV), and the ideal environmental conditions are often met in permanently waterlogged wetlands. However, CH₄ efflux from wetlands to the atmosphere depends not only on the rate of production (methanogenesis) but also on the extent of CH₄ consumption (methanotroph) that may occur in oxic surface layers and in the close vicinity of plant roots.^[4–6] In this process, CH₄ is oxidized to CO₂, and the rate of CH₄ emission that can be measured at the surface is hence the net result of two counteracting processes—methanogenesis and methanotroph.

ENVIRONMENTAL CONTROLS ON CH₄ EMISSIONS

Water Table Position

Water table position in relation to the surface is considered to be the most important factor controlling CH₄ flux because it indicates the boundary between anaerobic CH₄ production and aerobic CH₄ consumption (Fig. 1). Negative correlations between water table depth and rates of CH₄ emission have frequently been reported.^[7–9] It has also been found that the lowering of the water table may result in increased CH₄ emissions when episodic releases of CH₄ trapped in pore water occur and the diffusivity of gases in air-filled pore spaces increases.^[10,11] Field experiments in peat-forming wetlands reveal a high spatial variability of CH₄ emissions caused by distinct micro-topographical features on the wetland surface.^[12] Interactions between vascular plants and mosses create differences in elevation and therefore distance

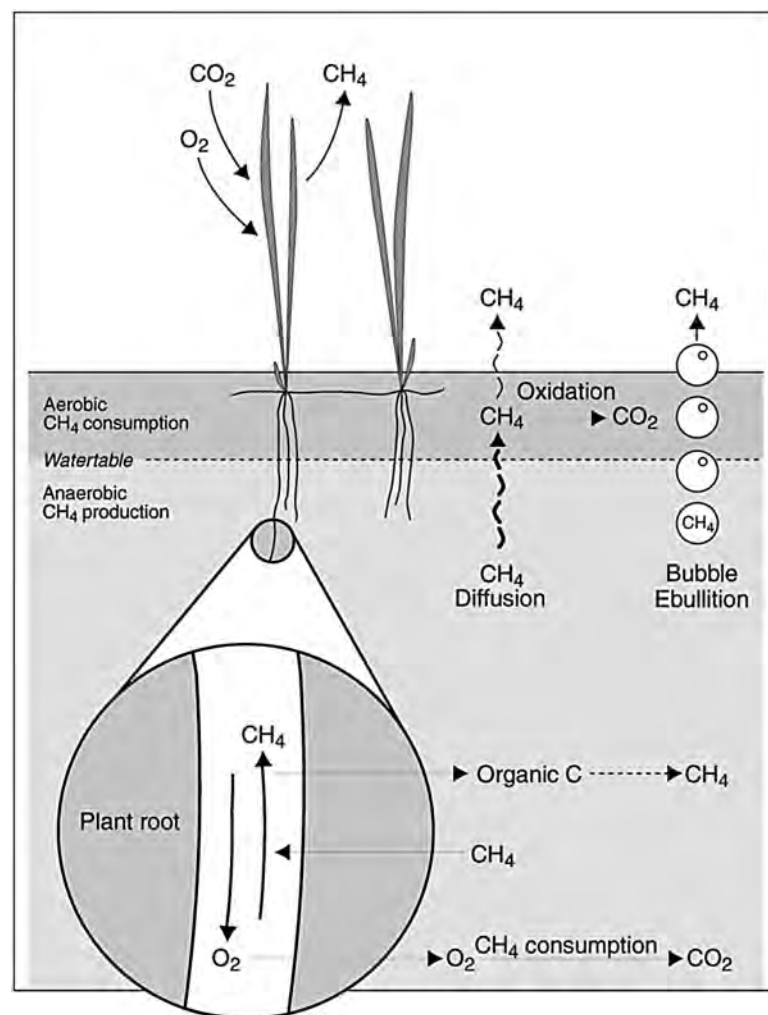


Fig. 1 The depth of the water table is important in determining net CH_4 emissions from natural wetlands because it indicates the boundary between anaerobic CH_4 production and aerobic CH_4 consumption. CH_4 may be transported from the soil to the atmosphere via diffusion through the soil profile, by bubble ebullition, or by plant-mediated transport. In the first case, CH_4 is subjected to methanotrophic oxidation as it passes through the oxic surface layer, and this acts to decrease net CH_4 emissions. Bubbles are quite stable, meaning that little oxidation takes place, and the same is also true when CH_4 is transported through vascular plants. Vascular plants are sources of methanogenic substrate because they release labile carbon compounds into the soil via root exudation and root turnover. Leakage of O_2 from roots may lead to the inhibition of methanogenesis and methanotrophic CH_4 consumption in the otherwise anoxic soil.

to the water table within a radius of less than a meter.^[13] Scaling up of spot CH_4 emission measurements to larger areas without careful consideration of experimental plot location may therefore be misleading. Laboratory incubations have shown that there is a potential for CH_4 production both above and below the water table,^[14] but the actual production in the field peaks at approximately 5–15 cm below the water table.^[8] For methanotrophic CH_4 consumption to occur, a simultaneous supply of both CH_4 and oxygen is necessary. The abundance and activity of methanotrophic bacteria are therefore highest at the interface between anoxic and oxic conditions, i.e., at the depth of the water table.^[5,14] Watson et al.^[5] used a model to show a high capacity for CH_4 oxidation in peat, where methanotroph accounted for 85% of the oxygen uptake potential when both oxygen and CH_4 were present in excess. In the absence of CH_4 , maximum oxygen uptake potential was near the surface.

Soil Temperature

Correlations between soil temperature and CH_4 production and emission have been found at scales ranging from

laboratory incubations and monolith experiments^[8,11,15] to field investigations at the plot^[16] and landscape scale.^[9]

There is a direct effect of temperature on the metabolic rate of CH_4 -producing microorganisms, but the sensitivity to temperature change is different for methanogens and methanotrophs.^[11,17] In a study carried out with peat slurries under laboratory conditions, Dunfield et al.^[17] found that CH_4 production in peat samples from temperate and subarctic areas was more sensitive to changes in temperature as compared to CH_4 consumption in the same samples. Increasing or decreasing temperatures also interact with other parameters with potential to control CH_4 production and emission to the atmosphere. One such interactive effect would be the relationship between temperature and water table position, where increasing temperatures lead to enhanced evaporation and plant transpiration, with lowering of the water table as a consequence.^[18] It has also been suggested that the temperature dependence of CH_4 production is constrained by substrate limitation and that temperature has an effect on methanogenesis mainly via its influence on substrate availability.^[19] In peat samples from a Swedish acid mire, CH_4 production was stimulated by

increased temperatures only when substrate (glucose) concentrations were simultaneously increased.^[15] It has further been found that the decomposition of labile material is more strongly controlled by temperature than the decomposition of more recalcitrant components.^[18] From the same study, it was also concluded that substrate availability lagged behind rapid temperature changes and that the thermal/hydrological history of the soil was important to determine the rate of CH₄ production.

CH₄ Transport: Diffusion, Bubble Ebullition, and Transport through Plants

The solubility of CH₄ in water is low, and it escapes through waterlogged soil to the atmosphere by diffusion, by bubble ebullition, or by transport through vascular plants (Fig. 1). The rate of CH₄ emission is largely controlled by the mode of transport because the different transportation pathways are associated with more or less extensive CH₄ consumption in the soil profile. CH₄ diffusing through oxic environments is subjected to methanotrophic oxidation to CO₂, and a water table depth of only a few centimeters may cause considerable reductions of CH₄ emissions to the atmosphere. Bubbles form when the rate of CH₄ production causes the sum of the partial pressures of dissolved gases to exceed the value of the hydrostatic pressure in the soil. However, the concentration of CH₄ in bubbles will be at equilibrium with dissolved pore water CH₄. At low rates of methanogenesis and with no vascular plants present, the main release of CH₄ to the atmosphere is likely to be by diffusion, but bubbles begin to form when the production rate exceeds the capacity for diffusive loss.^[20] Once formed, bubbles are quite stable, meaning that little methanotrophic CH₄ oxidation occurs even as they pass through oxic environments. Studies conducted in rice paddies have shown that bubble formation tends to decrease as the plants mature because the progressively better developed rooting system acts as gas conduits and efficiently transports gases out of the soil.^[21] However, a close relationship between vascular plant production and the availability of methanogenic substrate (discussed in the next section) may also speed up the rate of methanogenesis to such an extent that bubble formation is promoted. In order to supply oxygen for respiration to submerged structures, certain vascular plant species develop lacunae in stems, roots, and rhizomes.^[22]

Depending on the species, the ventilation of growing roots is carried out either by pressurized bulk flow or by simple diffusion that follows concentration gradients between the atmosphere and the soil. Leakage of oxygen may give rise to CH₄ oxidation and inhibition of methanogenesis in the close vicinity of the roots.^[5,6] However, net CH₄ emission is generally enhanced by the presence of vascular plants with a deep rooting system because of a “chimney effect,” where CH₄ transported from the soil to the atmosphere within plant lacunae is withdrawn from consumption in oxic conditions.^[12,23,24]

Vascular Plants and Substrate for CH₄ Production

Vascular plants are sources of substrates for CH₄ production because they easily release degradable carbon compounds into the soil via root exudation and root turnover (Fig. 1).^[20,24,25] In peat-forming wetlands, the organic matter becomes increasingly recalcitrant with depth,^[26] and several studies have pointed out that plant-derived inputs of labile carbon compounds could significantly contribute to increased CH₄ production.^[20,23,27] This “loading” of substrates into the soil may further be correlated to the photosynthetic rate of the vascular plants in the ecosystem^[24,28] because the amount of carbon allocated to belowground plant structures and eventually released to the soil is likely proportional to the CO₂ fixation rate.^[24] The overall primary productivity in the ecosystem may therefore be of great importance for the CH₄ emission rates from a wetland area.

REFERENCES

1. Lelieveld, J.; Crutzen, P.J.; Dentener, F.J. Changing concentration, lifetime and climate forcing of atmospheric methane. *Tellus* **1998**, *50B*, 128–150.
2. Chappelaz, J.A.; Fung, I.Y.; Thompson, A.M. The atmospheric CH₄ increase since the last glacial maximum. 1. Source estimates. *Tellus* **1993**, *45B*, 228–241.
3. Oremland, R.S. Biogeochemistry of methanogenic bacteria. In *Biology of Anaerobic Micro-organisms*; Zehnder, A.J.B., Ed.; John Wiley: New York, 1988; 641–703.
4. King, G.M. Ecological aspects of methane oxidation, a key determinant of global methane dynamics. *Adv. Microb. Ecol.* **1998**, *12*, 431–468.
5. Watson, A.; Stephen, K.D.; Nedwell, D.B.; Arah, J.R.M. Oxidation of methane in peat: Kinetics of CH₄ And O₂ removal and the role of plant roots. *Soil. Biol. Biochem.* **1997**, *29* (8), 1257–1267.
6. Popp, T.J.; Chanton, J.P.; Whiting, G.J.; Grant, N. Evaluation of methane oxidation in the rhizosphere of a *Carex* dominated fen in north central Alberta, Canada. *Biogeochemistry* **2000**, *51*, 259–281.
7. Christensen, T.R.; Friborg, T.; Sommerkorn, M.; Kaplan, J.; Illeris, L.; Soegaard, H.; Nordstroem, C.; Jonasson, S. Trace gas exchange in a high-arctic valley. 1. Variations in CO₂ and CH₄ flux between tundra vegetation. *Global Biogeochem. Cy.* **2000**, *14* (3), 701–713.
8. Daulat, W.E.; Clymo, R.S. Effects of temperature and water table on the efflux of methane from peatland surface cores. *Atmos. Environ.* **1998**, *32* (19), 3207–3218.
9. Hargreaves, K.J.; Fowler, D. Quantifying the effects of water table and soil temperature on the emission of methane from peat wetland at the field scale. *Atmos. Environ.* **1998**, *32* (19), 3275–3282.
10. Kettunen, A.; Kaitala, V.; Alm, J.; Silvola, J.; Nykänen, H.; Martikainen, P.J. Cross-correlation analysis of the dynamics of methane emissions from a boreal peatland. *Global. Biogeochem. Cy.* **1996**, *10* (3), 457–471.

11. Moore, T.R.; Dalva, M. The influence of temperature and water table position on carbon dioxide and methane emissions from laboratory columns of peatland soils. *J. Soil Sci.* **1993**, *44*, 651–664.
12. Frenzel, P.; Karofeld, E. CH₄ emission from a hollow-ridge complex in a raised bog: The role of CH₄ production and oxidation. *Biogeochemistry* **2000**, *51*, 91–112.
13. Waddington, J.M.; Roulet, N.T. Carbon balance of a boreal patterned peatland. *Glob. Change Biol.* **2000**, *6*, 87–97.
14. Moore, T.R.; Dalva, M. Methane and carbon dioxide exchange potentials of peat soils in aerobic and anaerobic laboratory incubations. *Soil. Biol. Biochem.* **1997**, *29* (8), 1157–1164.
15. Bergman, I.; Svensson, B.H.; Nilsson, M. Regulation of methane production in a Swedish acid mire by pH, temperature and substrate. *Soil Biol. Biochem.* **1998**, *30* (6), 729–741.
16. Verville, J.H.; Hobbie, S.E.; Chapin, F.S.; Hooper, D.U. Response of tundra CH₄ and CO₂ flux to manipulation of temperature and vegetation. *Biogeochemistry* **1998**, *41*, 215–235.
17. Dunfield, P.; Knowles, R.; Dumont, R.; Moore, T.R. Methane production and consumption in temperate and subarctic peat soils: Response to temperature and pH. *Soil Biol. Biochem.* **1993**, *25* (3), 321–326.
18. Updegraff, K.; Bridgham, S.D.; Pastor, J.; Weishampel, P. Hysteresis in the temperature response of carbon dioxide and methane production in peat soils. *Biogeochemistry* **1998**, *43*, 253–272.
19. Valentine, D.W.; Holland, E.A.; Schimel, D.S. Ecosystem and physiological controls over methane production in northern wetlands. *J. Geophys. Res.* **1994**, *99* (D1), 1563–1571.
20. Chanton, J.P.; Whiting, G.J. Trace gas exchange in freshwater and coastal marine environments: Ebullition and transport by plants. In *Biogenic Trace Gases: Measuring Emissions from Soil and Water*; Matson, P.A., Harriss, R.C., Eds.; Blackwell Science: Oxford, 1995; 98–125.
21. Holzapfel-Pschorn, A.; Conrad, R.; Seiler, W. Effects of vegetation on the emission of methane from submerged paddy soils. *Plant Soil* **1986**, *92*, 223–233.
22. Armstrong, W.; Justin, S.H.F.W.; Beckett, P.M.; Lythe, S. Root adaptation to soil waterlogging. *Aquat. Bot.* **1991**, *39*, 57–73.
23. Greenup, A.L.; Bradford, M.A.; McNamara, N.P.; Ineson, P.; Lee, J.A. The role of *Eriophorum Vaginatum* in CH₄ flux from an ombrotrophic peatland. *Plant Soil* **2000**, *227*, 265–272.
24. Joabsson, A.; Christensen, T.R.; Wallén, B. Vascular plant controls on methane emissions from northern peat-forming wetlands. *Trends Ecol. Evol.* **1999**, *14*, 385–388.
25. Jones, D.L. Organic acids in the rhizosphere—A critical review. *Plant Soil* **1998**, *205*, 25–44.
26. Christensen, T.R.; Jonasson, S.; Callaghan, T.V.; Havström, M. On the potential CO₂ releases from tundra soils in a changing climate. *Appl. Soil Ecol.* **1999**, *11*, 127–134.
27. Joabsson, A.; Christensen, T.R. Methane emissions from wetlands and their relationship with vascular plants: An arctic example. *Glob. Change Biol.* **2001**, *7* (8), 919–932.
28. Whiting, G.J.; Chanton, J.P. Plant-dependent CH₄ emission in a subarctic Canadian fen. *Global Biogeochem. Cy.* **1992**, *6* (3), 225–231.

Wetlands: Sedimentation and Ecological Engineering

Timothy C. Granata

Department of Civil and Environmental Engineering and Geodetic Science, Ohio State University, Columbus, Ohio, U.S.A.

Jay F. Martin

Department of Food, Agricultural, and Biological Engineering, Ohio State University, Columbus, Ohio, U.S.A.

Abstract

Wetlands are intimately tied to soils through sedimentation processes. The accretion of sediments accelerates the aging of wetlands, reduces infiltration through bottom substrates increasing water heights, and removes phosphorous, a limiting nutrient in many freshwater ecosystems. In this entry, the mechanics of sedimentation are discussed in relation to natural and constructed wetlands, and ecological engineering principles are suggested as a way to mitigate the effects of excessive sedimentation.

INTRODUCTION

Wetlands are highly efficient at storing and transforming chemicals. The primary input of these chemicals is from overland flow. The characteristics of these chemicals are dependent on the sources in the watershed. The rate of input to riparian zones, however, is dependent on landscape features, such as slope and surface resistance and soil properties, including grain size and cohesive strength. For large flows and loose soils, a significant portion of these chemical inputs can be a result of sediment transport.

PROBLEMS OF SEDIMENTATION IN WETLANDS

Sediment Accretion

The accretion of particulate material in wetlands limits the lifetime of these systems, advancing their succession to a terrestrial ecosystem as a function of sedimentation rates. Three sources of particulate material can be identified as follows: suspended particles derived from the inflow, the production of microbial particles in the wetland, and the production of detritus from macrovegetation. The latter two increase with increased primary production in wetlands. The former is a function of hydraulic loading, defined as the flow, Q , per unit area of wetland, A , i.e., QA^{-1} . Natural wetlands are prominent components of riparian ecosystems that exist either as floodplains, receiving periodic submer-sion, or as part of the riparian corridor, receiving continuous, though varying, inflow. In both cases, flow transports suspended sediments to wetlands. Suspended material is often deposited as current velocities decrease in broad, shallow expanses of wetlands. Sediment transport is a

function of total flow (and thus rainfall intensity) in a watershed and also depends on the slope of the landscape and the susceptibility of soils to erosion. Steep slopes with highly unconsolidated soils will contribute to large suspended sediment loads during high runoff events. For low-flow events, suspended sediment concentrations do not exceed 20 mg/L while for high flows they can exceed 80 mg/L.^[1] In extreme cases of accretion, sedimentation rates of 26.67 cm/yr can bury emergent grasses because sediment accumulation outpaces plant growth.^[2] In contrast, constructed wetlands are designed to treat wastewater having steady and high suspended solid concentrations, ranging from 20 to 75 mg/L, and volumetric ratios of settleable solids from 5 to 20 ml/L.^[3]

Reduced Conductivity of the Bed

Wetlands can experience reduced infiltration rates as particulate matter clogs bottom substrates. Generally, a reduction of hydraulic conductivity occurs with increasing hydraulic loading to the wetland. As a consequence of lower current velocities and higher sedimentation rates with increasing distance from the inlet, the hydraulic conductivity is lowest near the inlet and highest near the outlet. A general design criterion for wetlands is a hydraulic conductivity 1500 m/day based on the bed of 1.25–2.5 cm diameter gravel.^[4]

Phosphorus (P) Accumulation

The accretion of organic material on the bed results in wetlands with a high ion exchange capacity. The beds are partially responsible for the removal of phosphates by

sorption until they become saturated. The movement of P through the bed is slower than the hydraulic conductivity as a result of the storage (exchange) capacity of the bed.^[5] Plant uptake of phosphates is limited because a reduced hydraulic conductivity decreases infiltration and thus transport of phosphates to plant roots.^[6] A more consistent process for phosphate removal is by adsorption onto suspended sediments and their subsequent deposition.^[7] P loading is directly proportional to suspended sediment concentrations and hydraulic loading. It is estimated that 40 mg P accompanies each gram of suspended solids entering wetlands.^[1] Higher loading rates occur for flows over landscapes rich in P, such as agricultural lands. During periods of inundation, wetlands are characterized by aerobic and anaerobic soil zones. In anaerobic zones, the decreased redox potential leads to an increase in the solubility of particulate P, which may be discharged from the wetlands. This dynamic feedback between removal of P by sedimentation and release of P after particulate accretion often leads to the variable retention of P by wetlands. Despite the mineralization of particulate P, wetlands are generally sinks for P, with removal as high as 97%.^[8] The retention of P, however, decreases as P loading increases^[9] and often decreases through the life of the wetland.^[10]

PARTICLE TYPES AND SEDIMENTATION PROCESSES IN WETLANDS

The size distribution of particles in a wetland varies from nanometer-sized colloidal material through micrometer phytoplankton and heterotrophic organisms to millimeter- and centimeter-sized leaf detritus and sediments. For discrete particles, settling results from the force balance between particle weight (F_w) and drag on the particle such that:

$$F_w = F_D = \rho C_D A_{pr} \frac{w^2}{2} = (\rho_p - \rho) V g \quad (1)$$

where C_D is the drag coefficient on a 2-D surface, A_{pr} is the projected area of the particle, w is the settling speed, ρ_p and ρ are the particle and water densities, respectively, V is the volume of the particle, and g is the gravitational acceleration (9.8 m/sec²). Solving for settling rate, w , gives:

$$w = \sqrt{2 \frac{(\rho_p - \rho) V g}{\rho A_{pr} C_D}} \quad (2)$$

Thus, the settling rate of a particle is proportional to the excess density of the particles over that of water and the length scale of the particle defined by VA_{pr}^{-1} . Providing flow is not turbulent, specified as a Reynolds number $Re < 0.1$, and particle shape is unimportant to the drag; C_D can be approximated as follows:

$$C_D = \frac{24}{Re} = \frac{24v}{2Rw} \quad (3)$$

where R is the radius of the particle. Assuming a spherical particle, the above equation reduces to:

$$w = \frac{2\rho'gR^2}{9v} \quad (4)$$

the celebrated Stokes' Law for sinking particles. Here the excess density is written as follows:

$$\rho' = \frac{\rho_p - \rho}{\rho} \quad (5)$$

For biological material, excess density is less than 0.1%, and size is the determining factor. For this reason, small microbial organisms will sink slower than larger detrital material. The excess density of suspended sediments can reach 260%, indicating that the density is responsible for the high settling rates of suspended sediments.

Sedimentation in a wetland will occur if the settling speed is greater than the surface loading rate per area of wetland, or $w > QA^{-1}$. Given the large spectrum of particle types in natural flows, w is usually calculated based on the size and density of the particle type to be removed. Dividing the depth by the sinking rate gives the retention time in the wetland. Particles that are vertically well mixed as they enter a wetland will have a uniform concentration distribution comprising the spectrum of particle sizes. If the settling rate of the particle, w_p , is greater than w , the particle will be deposited in the wetland cell. For particles with $w_p < w$, the fraction of particles that will be removed is X_p . Thus, the total particle removal by a wetland can be calculated as follows:

$$FR = (1 - X_c) + QA^{-1} \sum X_r \Delta x \quad (6)$$

where $1 - X_c$ is the fraction of particles removed with sinking speeds $w_p > w$ and $QA^{-1} \sum X_r \Delta x$ is the fraction with a settling speed $w_p < w$ that are removed over a distance Δx , along the path of the flow.^[3] The crucial step in determining the sedimentation rate in wetlands is first determining the settling rate of each concentration fraction, either by settling columns or by size-concentration measurements.

Often the aggregations of particles form in wetlands,^[11] a process called flocculation. Sedimentation rates of aggregates are dominated by organic particles of fractal dimension.^[12] Aggregates tend to settle faster than their discrete component particles, and the only way to determine removal rate is by carefully transferring flocs to a settling column and determining the percent removal as a function of the height of the column and time.^[3]

In wetlands with a high concentration of suspended solids, settling is affected by the contact between particles. This would occur in the bottom sediments of most wetlands where two settling processes can be identified: hindered and compression settling.^[3] The hindered zone is marked

by a large gradation in particle concentrations, which is less than the total concentrated in the compression zone. By plotting the concentration of particles over the height of a settling column as a function of time, a break point can be found in the time from hindered settling to compression settling.^[3]

Because large, heavy particles settle out first, the highest sedimentation rates in wetlands are near the inlets. As mentioned above, the hydraulic conductivity through the bottom sediment decreases as clogging occurs. Because clogging is also a function of distance from the inlet, the most accurate estimate of the hydraulic conductivity is to measure the change in water surface elevation with increasing distance from the inlet. However, as Kadlec and Watson^[11] show, this is not a simple function of Darcy's Law but must include evapotranspiration in the wetland. For newly constructed wetlands, the hydraulic conductivity of the bottom substrate (gravel, sand, and mud) cannot be used because clogging will result from sediment accretion.^[4]

ECOLOGICAL ENGINEERING SOLUTIONS TO SEDIMENTATION

To keep suspended particles entrained by flows from accruing in wetlands and clogging bottom sediments, one or more settling basins can be included between the inlet channel and the wetland cell. This would have the following two effects: first, to collect all but the finest and least dense suspended particles and, second, to remove P from the inflow.

To overcome the problem of detrital accumulation in wetlands, woody plants could be substituted for grasses and periodically harvested. This would not only reduce the amount of biomass accrued in the wetland but also increase the efficiency of nutrient uptake in the unclogged root zone and provide a potentially marketable resource. Because phytoplankton are the most abundant nutrient filters in a wetland and have intrinsically low sinking rates, a wetland could be designed with the low retention of suspended plankton to further reduce sedimentation while improving nutrient removal. A settling basin could then be sized to accumulate these nutrient-rich particles for harvesting before nutrients are remineralized. To keep flocs of particles from forming in wetlands, cells could be mixed with aerators or inlet pumps. This would break up flocs, which would be exported as discrete particles.

CONCLUSION

Because wetlands are always shallow to promote macrophyte growth, they essentially act as flat plate collectors of

sinking particles. The dominant particles in wetlands are suspended particles in the inflow and biomass of vegetation resulting from growth in the wetlands. The processes of sedimentation can reduce a wetland's storage capacity, its efficiency to retain nutrients, and its lifetime. Future efforts should be aimed at enhancing nutrient removal while reducing sedimentation. This can be done with engineered structures, such as settling basins and grit chambers, or by less costly technologies such as stilling wells. More innovative green solutions are on the horizon.

REFERENCES

1. Porter, K.S. Nitrogen and phosphorous. In *Food Production, Waste, and the Environment*; Ann Arbor Science Publishers Inc.: Ann Arbor, 1975; 372 pp.
2. Chung, C.-H. Ecological engineering of coastlines with salt-marsh plantations. In *Ecological Engineering: An Introduction to Ecotechnology*; Mitsch, W.J., Jørgenson, S.E., Eds.; John Wiley & Sons: New York, 1989; 472 pp.
3. Metcalf and Eddy, Inc. *Wastewater Engineering: Treatment, Disposal, Reuse*; McGraw Hill: New York, 1991; 1334 pp.
4. Watson, T.J.; Choate, K.D. Hydraulic conductivity of onsite constructed wetlands. In 9th National Symposium of Individual and Small Community Sewage Systems, Fort Worth, Texas, Mar 11–14, 2001.
5. Kadlec, R.H.; Knight, R.L. *Treatment Wetlands*; CRC Press/Lewis Publishers: Boca Raton, 1996; 839 pp.
6. Davies, T.H.; Cottingham, P.D. Phosphorus removal from wastewater in a constructed wetland. In *Constructed Wetlands for Water Quality Improvement*; Moshiri, G.A., Ed.; Lewis Publishers: Boca Raton, 1993; 632 pp.
7. Richardson, C.J.; Craft, C.B. Efficient phosphorus retention in wetlands: Fact or fiction? In *Constructed Wetlands for Water Quality Improvement*; Moshiri, G.A., Ed.; Lewis Publishers: Boca Raton, 1993; 632 pp.
8. Mitsch, W.J.; Gosselink, J.G. *Wetlands*; Van Nostrand Reinhold Co.: New York, 2000; 920 pp.
9. Mitsch, W.J.; Reeder, B.C.; Klarer, D.M. Wetlands for nutrient control: Western Lake Erie. In *Ecological Engineering: An Introduction to Ecotechnology*; Mitsch, W.J., Jørgenson, S.E., Eds.; John & Wiley Sons: New York, 1989; 472 pp.
10. White, J.S.; Bayley, S.E.; Curtis, P.J. Sediment storage of phosphorus in a northern prairie wetland receiving municipal and agro-industrial wastewater. *Ecol. Eng.* **2000**, *14*, 127–138.
11. Kadlec, R.H.; Watson, J.T. Hydraulics and solids accumulation in a gravel bed treatment wetland. In *Constructed Wetlands for Water Quality Improvement*; Moshiri, G.A., Ed.; Lewis Publishers: Boca Raton, 1993; 632 pp.
12. Kilps, J.R.; Logan, B.E.; Alldredge, A.L. Fractal dimensions of marine snow determined from image analysis of *in situ* photographs. *Deep-Sea Res.* **1994**, *41* (8), 1159–1169.

Wind Erosion: Climate Change

Alan Busacca

Department of Crop and Soil Sciences, Washington State University, Pullman, Washington, U.S.A.

David Chandler

Plant, Soils and Biometeorology Department, Utah State University, Logan, Utah, U.S.A.

Abstract

Wind erosion is a common phenomenon in areas of arid and semiarid climate and sparse vegetation. As much as 30% of the earth's land area is susceptible to wind erosion. Human impacts, such as farming, grazing, deforestation, urbanization, and withdrawal of groundwater, may contribute as much as 20–30% of the total. As human land use encompasses ever greater areas of earth's arid and semiarid lands, the control of wind erosion will increasingly require decision making that accounts fully for climate change and climate variability.

INTRODUCTION

Wind erosion has been driven by cycles of climate change over geologic time, resulting in the transport and accumulation of eolian sediments. As an important process in the Pleistocene Epoch, or Ice Ages, it left conspicuous eolian deposits throughout the world. Wind erosion, driven by shorter timescale changes in climate, has occurred during the interglacial period of the last 12,000 years. This time period has also seen the rise of human societies, the development of agriculture, and, in the last 200 years, the technological age in which human population increased from less than one billion to more than six billion. The methods we use to study climate change and wind erosion shift at the major glacial–interglacial climatic transition that occurred 12,000 years ago. That is, to reconstruct wind erosion events during the Pleistocene Epoch and their causes, such as climate change, we have relied on geologic studies of the deposits resulting from wind erosion. To reconstruct events that occurred 12,000 years ago, however, we have variably detailed archeological and written records of wind erosion and climate change, in addition. In the last several decades, we have added increasingly detailed observations of meteorological and wind-erosion events from global networks of satellites and remote sensors. Since the 1980s, we have added regional- to global-scale computer models to investigate both ancient and modern wind erosion events and their causes.^[1]

Wind erosion is a common phenomenon in areas of arid and semiarid climate and sparse vegetation. As much as 30% of the earth's land area is susceptible to wind erosion. The total annual production of mineral aerosols from wind erosion is estimated as one to two billion metric tons.^[2]

Human impacts, such as farming, grazing, deforestation, urbanization, and withdrawal of groundwater, may contribute as much as 20–30% of the total. Strikingly, during the cold phases or glacials of the Pleistocene Epoch, wind erosion, as judged by reconstructions from dust in polar ice caps and in deep-sea sediments, may have been one order of magnitude or more greater than that of today.^[3]

WIND EROSION AND CLIMATE CHANGE IN THE GEOLOGIC RECORD

Eolian deposits associated with ancient wind erosion events have been identified back to the Proterozoic (600–2500 Ma).^[3] It is especially difficult to reconstruct ancient environments and the paleoclimatic significance of eolian deposits before the Devonian Period (360–410 Ma), when land surfaces were first colonized by plants.^[3]

About 10% of the land area of the earth is covered by Loess (i.e., deposits of windblown silt; [Fig. 1](#)) in several different climatic settings,^[3–5] and perhaps a similar area is covered by sand dunes and sand seas in semiarid and arid climate areas. To a great extent, these are the tangible products of very active wind-erosion processes triggered by global climatic changes beginning in the Pleistocene Epoch and continuing through the interglacial period. Eolian dust, an important component of soils downwind of areas of wind erosion, may play a role in renewing plant nutrients in areas of rapid leaching in the tropics (e.g., the input of Saharan dust to soils of the Amazon basin).^[1] Thus, global pedological and biogeochemical cycles may be affected by climatically induced dust aerosols. Deep-ocean sediments

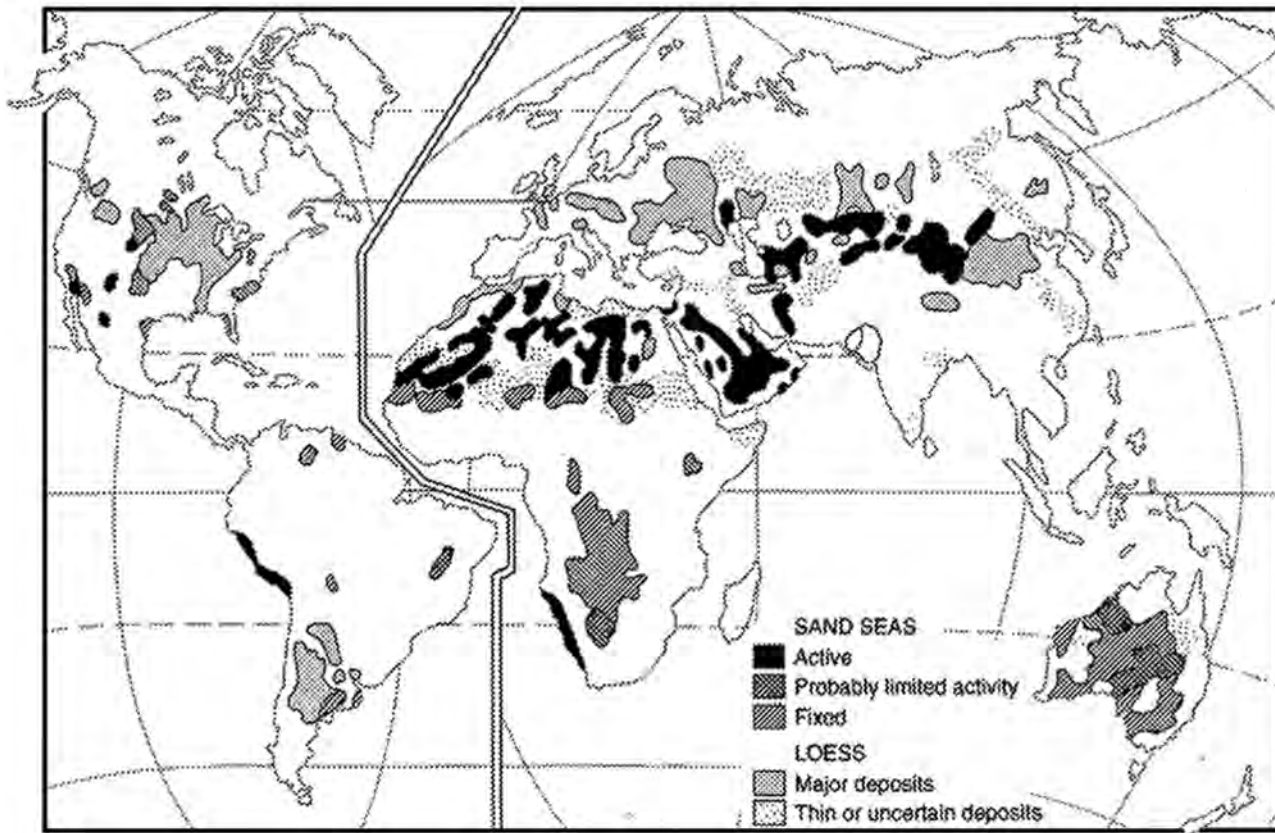


Fig. 1 The global distribution of major eolian deposits.

Source: From Thomas.^[3]

dating from the last glacial maximum (LGM) in many places contain up to 20% quartz from eolian dust (Fig. 2).^[3,4]

Wind erosion, driven by climate change, has played an important role in human history and development over the last 12,000 years. By some measures of dust aerosols, however, eolian sediment fluxes during the LGM about 18,000 years ago were “10–40 times” higher than during the present day.^[5,6] The principal means by which geologists and paleoclimatologists have reconstructed past episodes of wind erosion and linked them to changing climates has been the stratigraphy, geomorphology, and chronology of eolian deposits associated with arid, semiarid, and periglacial areas. These studies began in the mid-19th century and accelerated with new age-dating techniques after the World War II. Starting in the 1960s, more refined reconstructions of past wind erosion and links to climate change were made possible by examination of eolian materials, mainly dust, in cores from Arctic and Antarctic glacial ice and in deep-sea sediments.^[1,6]

Stratigraphic studies of ice cores, deep-sea sediments, and Loess demonstrate that atmospheric dust deposition changed markedly over glacial to interglacial as well as over shorter timescales. Dust deposition shows millennial-scale variation that tracks the patterns of other

paleoenvironmental indicators, such as atmospheric CO₂ concentrations,^[1] thereby suggesting a direct, but not necessarily simple, link to climate change over several time-scales. Stratigraphic studies confirm that the LGM had much greater wind erosion, with periglacial and peridesert zones of the temperate latitudes and peridesert zones of Africa showing dramatic increases in dust fluxes,^[1,5] while other regions showed decreases in dust flux.

Three causes may explain why wind erosion and dust loading into the atmosphere were higher during glacials than interglacials: increased wind speeds, reduced intensity of the hydrologic cycle, and expansion of eroding areas.^[1] Increased wind speeds would increase the probability of exceeding the threshold velocity for sediment entrainment. Reduced intensity of the hydrologic cycle (less precipitation) could allow dust to remain longer in the atmosphere and be carried farther from the source before rainout would occur. Source areas probably expanded by: 1) exposure of sediments on continental shelves with lowered sea level; 2) deposition of large volumes of outwash from continental and alpine glaciers, especially in the northern hemisphere; and 3) reduction in soil moisture and/or reduction in plant cover. Simulations of the atmosphere during glacial climates show an increase in wind intensities, but direct evidence of higher wind speeds is equivocal.^[1] Some models

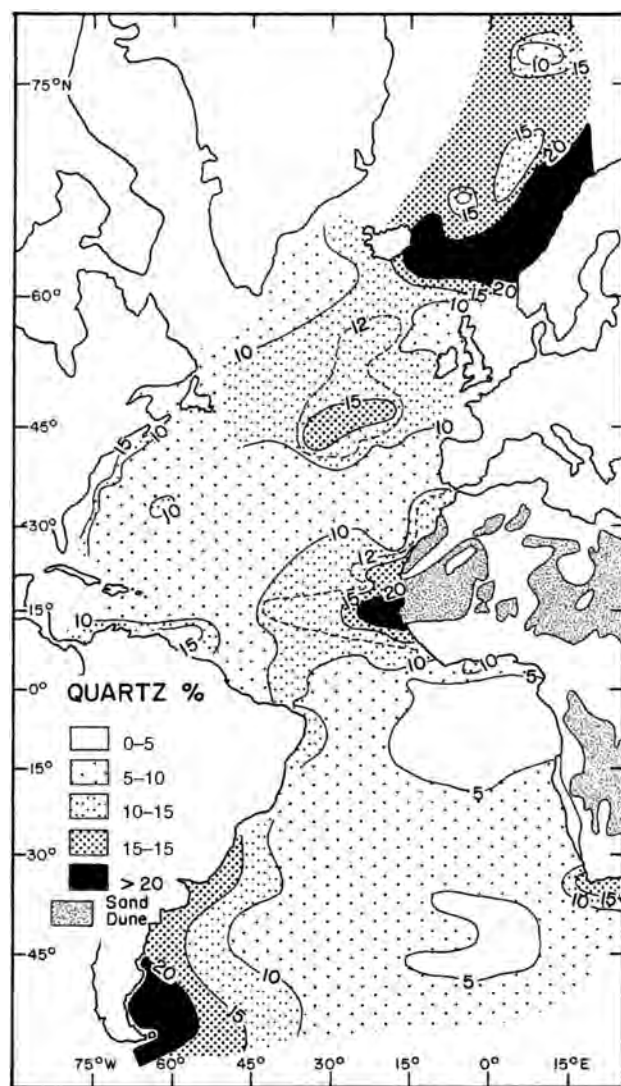


Fig. 2 Distribution of weight percent quartz (carbonate free) in Atlantic sediments of the LGM (18,000 years B.P.).

Source: From Pye.^[4]

predict that there were strong cyclonic winds in areas of periglacial outwash in front of major northern hemisphere ice sheets.^[1] Temperature depression at the LGM is estimated to have 10°C or more in temperate latitudes. These estimates are based on pollen evidence for changes in vegetation. Such decreases in temperature would have resulted in decreased intensity of the hydrologic cycle. Evidence regarding lake levels suggests that runoff was less than it is over much of the globe, including the Sahel in northern Africa.^[5] Drier soils and colder temperatures also drove the expansion of desert, grassland, and shrubland at the expense of forests, resulting in a worldwide increase in the area susceptible to wind erosion. Widespread Loess deposits in North America, Europe, Asia, and South America^[4,5] are of glacial age, which also suggests that the LGM was more arid or that sediment production was more robust (e.g., glacial erosion).

Detailed paleoenvironmental studies of Saharan sand seas have produced process-response models of environmental change driven by arid-humid cycles that result from orbital forcing of climate (Fig. 3).^[7] For example, in Fig. 3, the model predicts that during humid phases, sediment production by chemical weathering and alluviation dominates, and eolian systems are stabilized by soil moisture and vegetation. In contrast, during arid phases, transport capacity and sediment availability increase, resulting in time-lagged deflation of sediments accumulated during the humid phase with construction of eolian deposits.^[7]

The deserts and the Loess Plateau in northern China (Fig. 1) form perhaps the most famous example of an eolian system coupled to glacial-interglacial changes in the global climate system. As many as 40 Loess units are interstratified with Paleosols, thereby representing 2.5 million years of record. The cyclical alternation of Loess and Paleosols is thought to represent fluctuations in the dominance of the Asian winter (dust transporting) and summer (precipitation bringing) monsoons associated with variations in earth's orbital parameters that drove the Ice Ages. At the desert margins, sand beds within the Loess suggest that the desert margin advanced many kilometers to the south by active dune migration during cold glacial periods of intensified winter monsoon. Conversely, Loess or Paleosols in desert sites suggest northern movement of the boundary.^[8]

The results of simulation modeling of radiative forcing today and during the LGM are intriguing. They suggest that dust aerosols in the atmosphere from wind erosion, anthropogenic sources, and volcanic eruptions can "effect" climate change as well as "result from" climate change (e.g., the cooling of global temperatures after the eruption of Mount Pinatubo in 1991). The dustiness of the LGM atmosphere could have produced an additional cooling of 3°C beyond predictions, based on orbital forcing, ice-sheet albedo, and CO₂ forcing alone,^[11] although different models produce different latitudinal distributions of predicted temperature change.

WIND EROSION AND CLIMATE CHANGE IN HUMAN HISTORY AND TODAY

"Climate change" refers to statistical changes in climatic variables such as temperature, evaporation, or precipitation over a long period, relative to human records, such as decades to centuries. "Climate variability" refers to statistical changes in climate variables over shorter timescales, such as months to years. In recorded human history, wind erosion has related both to dry periods within the natural cycles of climate variability and to progressive, long-term climate change. Throughout history and around the world, humankind has responded to the agricultural challenge posed by climate variability and climate change by developing hydraulic infrastructure, especially in areas of arid and semiarid climates. The fall of ancient hydraulic systems,

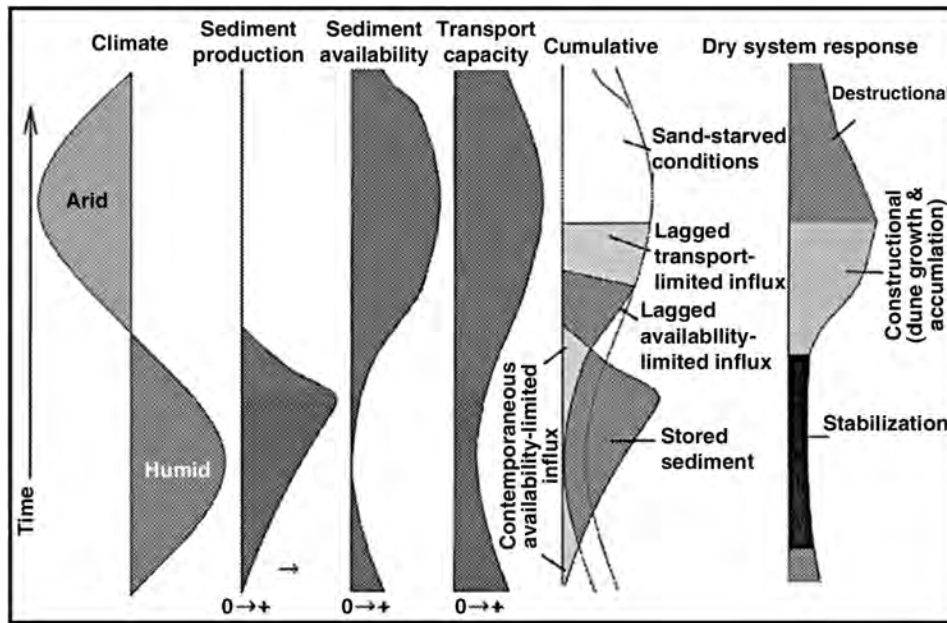


Fig. 3 A climate change driven process-response model based on sediment production, sediment availability (supply), and transport capacity of wind for Saharan sand seas.

Source: From Kocurek.^[7]

such as those in Mesopotamia and along the Indus River, and the subsequent wind erosion, soil degradation, and societal collapse have often been attributed to climate change.^[9] Chinese records document quasi-periodic variations in rainfall, temperature, and dustfall for the past 2000 years, with periods ranging from 2 to 100 years.^[10] Periods of high dustfall were strongly correlated with periods of drought.

When human impacts to the natural and agricultural environment are added to natural climate variability and climate change, greater wind erosion is likely to occur than would be the case under climate change alone. The world's arid and semiarid areas have the greatest interannual variability of rainfall (Fig. 4).^[9] Although ancient sand seas

continue to shift by dune migration, generation of dust aerosols is generally low because deflation has already occurred.^[5] Areas at greatest risk of wind erosion with soil degradation are those at desert margins. The fine- to medium-textured soils deposited previously by deflation at desert margins are often ideal for agriculture but particularly susceptible to entrainment by wind. Human disturbance of eolian soils, such as fallow rotations in dryland agriculture, can lead to accelerated wind erosion, as in the U.S. Great Plains during the Dust Bowl years of the 1930s.

Drought can persist for decades in some arid and semi-arid areas, with dramatic effects on plant cover, soil moisture, and consequent wind erosion. In the Sahel of West Africa, where decadal-scale fluctuations in climate are

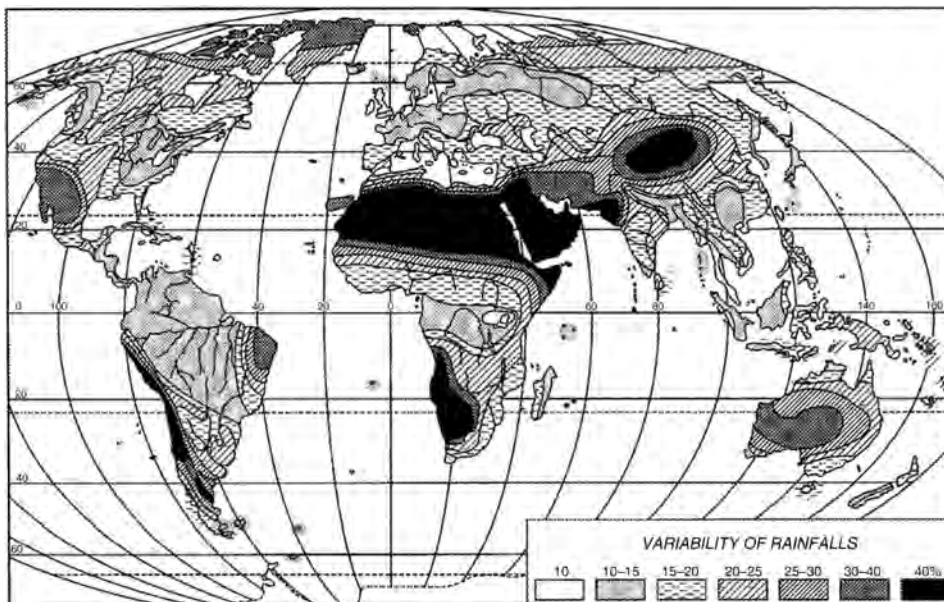


Fig. 4 Interannual variability of rainfall on the surface of the earth. Source: From Mainguet^[9] and Nicholson.^[11]

particularly strong,^[12] mean annual rainfall decreased by 25–40% from 1968 to 1997. West Africa is the source of nearly half of the mineral dust in the atmosphere today. Analysis of dust storms over West Africa shows a steady increase in the frequency of dust occurrence since the early 1970s, which parallels negative rainfall anomalies during this period. The source is principally the semiarid Sahel, not the Sahara desert as once was assumed.^[12] Paradoxically, some measurements and models suggest that atmospheric dust over the Sahel has an important climatic feedback effect that may prolong drought cycles by scattering or absorbing solar radiation or even by modifying the African Easterly Jet.^[12] Dust from the Sahel is carried into the upper atmosphere and transported across the Atlantic Ocean to Bermuda, the Caribbean islands, and even the Amazon Basin.^[11]

Redistribution of surface water also can drive local and regional changes in surface hydrology that result in wind erosion. The diversion of the Owens River to Los Angeles, California, in the early 1900s, turned Owens Lake into a 300-km² dry playa, known for wind erosion and dust generation. A agreement to flood parts of the lakebed avoids decades of litigation and sets a goal to meet federal air quality standards. In Central Asia, large-scale river diversions for irrigation led to the desiccation of Lake Aral. The dramatic reduction in the lake's surface area diminished its effect as a regulator of regional climate and exposed over two million hectares of salinized lake bottom sediments. The combined effect of a shorter growing season, degraded soils, higher winds, and exposed sediments results in erosion of millions of tons of dust and salts annually.^[9] For the lands above the rapidly depleting Ogallala Aquifer in southwestern Kansas and the Oklahoma–Texas Panhandle, deep well irrigation has been key to limiting the recurrence of the Dust Bowl.^[13,14]

Individual severe wind erosion events can also be triggered by extreme synoptic weather patterns, with or without antecedent drought. For example, in 1998, on April 15 and 19, two intense dust storms were generated over the Gobi Desert by springtime cold weather systems. The April 15 dust cloud was removed by precipitation over East Asia. The April 19 dust cloud was transported across the Pacific in five days. Part of the dust subsided to the surface between British Columbia and California, producing average Asian dust concentrations over the West Coast of about 20–50 µg/m³. The 1998 Asian dust event was larger than any other similar event in western North America in the 20th century.

WIND EROSION AND CLIMATE CHANGE IN THE FUTURE

There is considerable interest in the prediction of future climate change, especially in light of the increasing rate of global warming over the past several decades. Several

general circulation models (GCMs) have been applied to estimate climate change driven by rising atmospheric CO₂ levels. These models have more been coupled with the effects of atmospheric aerosols such as sulfate and anthropogenic dust.^[11] The best estimate for the year 2100, under a doubled concentration of CO₂ and incorporating aerosols, is a temperature increase of 1.0–3.5°C,^[14,15] with an increase in global average precipitation ranging from 5% to 15%. Paradoxically, the GCMs also predict that mid-continental land areas in the middle latitudes of the northern hemisphere may undergo a drying trend in summer.^[14] This could have the effect of increasing the potential for wind erosion in critical areas, such as the Great Plains of North America and the desert basins of Asia, especially if human land use results in lesser plant or crop cover. Different predictions come from models that take into account the physiological effects of increased CO₂ on plant growth and climatic warming. These models predict substantial reductions in the area of arid and semiarid vegetation and thus reductions of about 20% in source areas for potential wind erosion.^[11]

Under greenhouse warming, changes will occur not only in the mean values of climate variables but also in the frequency and severity of extreme weather events,^[14] owing to the increased energy in the atmosphere. The predicted increase in variability of temperature, rainfall, and wind speed, already high in arid and semiarid areas (Fig. 4), may exacerbate wind erosion of mid-continental agricultural, range, and desert lands. Although smaller changes in mean values of climatic variables are predicted for tropical latitudes, wind erosion could increase in these areas as well.

Under greenhouse warming scenarios, the severity of drought in the Great Plains of North America may well exceed that experienced during the Dust Bowl.^[14] In the mid-latitudes and in other arid and semiarid areas at low latitudes, increased incidence of high-wind events could begin to “pirate” or erode the large deposits of eolian sands and Loess that were emplaced during the LGM, if plant cover is reduced. According to estimates, present wind speeds exceed the threshold wind velocity 30–60% of the time,^[5] thus the eolian soils of the Great Plains are poised for renewed wind erosion if existing plant cover or land use patterns are significantly disturbed. Dropping water tables in the Ogallala aquifer may eventually force the abandonment of up to six million hectares of irrigated cropland on soils sensitive to wind erosion.^[13] A similar situation may occur in the irrigated lands around Lake Aral. The legacies of ancient hydraulic societies, which rose to high levels of achievement but then collapsed upon climate change, raise questions about the future of hydraulic societies.^[14] If the competition for freshwater resources continues to increase and the availability of groundwater resources declines, especially under climate change scenarios, a reduction or relocation of irrigated croplands is possible.

Improved stochastic climate modeling, coupled with proper land use management of the agricultural lands at the desert margins, may provide the means to limit wind erosion of those lands under future climates. As human land use encompasses ever greater areas of earth's arid and semiarid lands in the future, the control of wind erosion will increasingly require decision making that accounts fully for climate change and climate variability.

REFERENCES

1. Harrison, S.P.; Kohfeld, K.E.; Roelandt, C.; Claquin, T. The role of dust in climate changes today, at the last glacial maximum and in the future. *Earth Sci. Rev.* **2001**, *54* (1–3), 43–80.
2. D'Almeida, G.A. Desert aerosol: Characteristics and effects on climate. In *Palaeoclimatology and Palaeometeorology: Modern and Past Patterns of Global Atmospheric Transport*; Leinen, M., Sarnthein, M., Eds.; Kluwer Academic Press: Dordrecht, 1989; 311–338.
3. Thomas, D.J.T. *Arid Zone Geomorphology*; 2nd Ed.; John Wiley and Sons: New York, 1997.
4. Pye, K. *Aeolian Dust and Dust Deposits*; Academic Press: London, 1987; 334 pp.
5. Goudie, A.S.; Livingstone, I.; Stokes, S. *Aeolian Environments, Sediments, and Landforms*; John Wiley and Sons, Ltd.: West Sussex, 1999.
6. Maggi, V. Mineralogy of atmospheric microparticles deposited along the Greenland Ice Core Project core. *J. Geophys. Res.* **1997**, *102* (C12), 26725–26734.
7. Kocurek, G. Aeolian system response to external forcing factors—a sequence stratigraphic view of the Saharan region. In *Quaternary Deserts and Climatic Change*; Alsharhan, A.S., Glennie, K.W., Whittle, G.L., Kendall, C.G., St. C., Eds.; Balkema: Rotterdam, 1998; 125–153.
8. Ding, Z.; Sun, J.; Rutter, N.; Rokosh, D. Changes in sand content of loess deposits along a north–south transect of the Chinese loess plateau and the implications for desert variations. *Quat. Res.* **1999**, *52*, 56–62.
9. Mainguet, M. *Aridity, Droughts, and Human Development*; Springer-Verlag: Berlin, 1999; 302 pp.
10. De'er, Z. Historical records of climate change in China. *Quat. Sci. Rev.* **1991**, *10*, 551–554.
11. Nicholson, S. Land surface processes and Sahel climate. *Rev. Geophys.* **2000**, *38* (1), 117–139.
12. Opie, J. The drought of 1988, the global warming experiment, and its challenge to irrigation in the old Dust Bowl region. *Agr. Hist.* **1992**, *66* (2), 279–336.
13. Houghton, J.T.; Meira Filho, L.G.; Callander, B.A.; Harris, N.; Kattenberg, A.; Maskell, K., Eds. Intergovernmental panel on climate change. In *Climate Change 1995: The Science of Climate Change*; Cambridge University Press: Cambridge, 1996; 572 pp.
14. Rozensweig, C.; Hillel, D. *Climate Change and the Global Harvest*; Oxford University Press: New York, 1998.
15. Mainguet, M. *L'homme et la Sécheresse*; Masson: Paris, 1995.

Wind Erosion: Control Measures

Gary L. Tibke

*U.S. Department of Agriculture—Wind Erosion Research Unit (USDA-WERU),
Kansas State University, Manhattan, Kansas, U.S.A.*

Abstract

Wind erosion can be controlled by maintaining vegetative residues on the soil surface, reducing field width, utilizing stable soil aggregates, roughing of the land surface, and reshaping the land (i.e., the five basic principles of wind erosion control). While individual conservation practices can be successful in controlling erosion, a combination of practices should generally be considered when a wind erosion control system is being planned.

INTRODUCTION

Wind erosion can be controlled by reducing the wind forces at the soil surface or by creating surface conditions more resistant to wind forces.^[1] This can be accomplished by maintaining vegetative residues on the soil surface, reducing field width, utilizing stable soil aggregates, roughing of the land surface, and reshaping the land (i.e., the five basic principles of wind erosion control). While individual conservation practices can be successful in controlling erosion, a combination of practices should generally be considered when a wind erosion control system is being planned. Droughts will cause a shortage of residue in the stubble mulch program, erosive winds will not always blow in the prevailing direction when barriers are used, and soil clods and ridging for erosion control are temporary control measures and best serve as supplemental practices. Conditions conducive to wind erosion exist when the soil surface is loose, dry, and finely granulated, when the soil surface is smooth and vegetative cover is sparse or absent, and when the susceptible area is sufficiently large. Wind erosion control is usually needed in areas with low and variable precipitation and frequent droughts and in areas where high winds, high temperatures, and consequent high evaporation prevail.^[2] Control is also needed in areas where crops that are easily damaged by moving soil particles are grown on highly erodible soils. Research has shown that it is possible to alter some of the components of the erosion process and that this can be accomplished with one or more of the five basic principles of wind erosion control.^[3]

ESTABLISHING AND MAINTAINING VEGETATION OR VEGETATIVE RESIDUES TO PROTECT THE SOIL

On unprotected fallow fields, applying the principle of vegetative cover can best be done by practicing no-till or

stubble mulch farming.^[4] The importance of vegetative protection on the land cannot be overstressed.

Stubble Mulching

Stubble mulching is a form of subsurface conservation tillage, where the soil is tilled without inversion. Tillage tools, such as chisels, field cultivators, sweeps, or blades, are used (Fig. 1).

Weed control is accomplished with cultivation and/or herbicides. The level of protection afforded by stubble mulching is proportional to the kind, quantity, and orientation of crop residue maintained on the soil surface. Excessive use of tillage implements that bury crop residues is the major cause of inadequate vegetative cover for erosion control on cropland. Also, tillage operations that flatten crop residues must be minimized. Standing residues are 5 to 10 times more effective in controlling wind erosion than flat residue. Conservation tillage systems must be designed to avoid excessive loss of surface residues, if effective wind erosion control is to be accomplished.

No-Tillage

Since 1990, no-till farming has grown from 6% to more than 16% of the planted acres in the United States (Fig. 2). This is an increase of over 32 million acres.^[5] No-till is a procedure whereby a crop is planted directly into a seedbed that has not been tilled since harvest of the previous crop. Only the immediate seed zone is disturbed, and no additional tillage occurs. Many studies have shown that a no-till program is a way to virtually eliminate erosion problems (Table 1).^[6]

The bodies of plants and animals that have finished their life cycle are involved in the processes of soil formation and help to generate fertile and productive soils. No-till largely approximates this ingenious work of nature since it in fact makes use of crop residue for this purpose.



Fig. 1 Stubble mulch tillage.
Source: Courtesy of U.S. Department of Agriculture (USDA).

In many farming systems, the plow can be replaced efficiently by biological activities. Plants, roots, worms, arthropods, and other living beings are natural cultivators of the soil. In one study, after 7 years of no-till, the earth worm population was 36 times higher than the population in a conventionally tilled field.^[7] The switch to no-till causes many soil changes. The type and extent of changes depend on soil type, climate, and farming history. Farmers who have no-tilled for some years usually notice more soil moisture, better seedbed tilth, more organic matter and earthworms, less soil erosion, and improved trafficability.^[8]

Cover Crops

The purpose of a cover crop is to produce vegetative protection for the land against wind and water erosion. Cover crops are usually planted when protective residues are inadequate, and winter and spring winds are high. Cover crops



Fig. 2 No-till corn.
Source: Courtesy of USDA.

Table 1 Estimated soil losses with four tillage systems in Nebraska.

Tillage system	Estimated soil loss (t ha ⁻¹ yr ⁻¹) ^a
Conventional	20.2
Disk and plant	10.5
Till plant	4.5
No-tillage	1.8

^at ha⁻¹ × 0.446 = t acre⁻¹.
Source: From Rice.^[6]

are also planted between rows to provide protection for vegetable or other crops that are highly susceptible to abrasive injury in the seedling stage. Cover crops are best suited to humid and higher rainfall areas or irrigated land. In many of the semiarid and arid areas of the Great Plains and intermountain valleys, cover crops are not practical because they compete for nutrients and limited soil moisture. Generally, cover crops are used in these dry non-irrigated areas only to control erosion on erosion-susceptible knolls or where low residue crops have been grown.

REDUCING FIELD WIDTHS BY ESTABLISHING BARRIERS OR STRIP CROPPING

Windbreaks and wind barriers contribute to wind erosion control by reducing wind speed, on their leeward side, below the threshold required for initiation of soil movement and by decreasing the field length along the erosive wind direction (Fig. 3).

The main effect of reducing downwind field length is a reduction in breakdown of clods and crust by abrasion from saltating (hopping) particles. Abrasion of soil surface components and breakage of saltating aggregates increase the



Fig. 3 Windbreaks and strip cropping.
Source: Courtesy of USDA.

soil flow with distance downwind. This in turn reduces surface roughness and permits even higher rates of soil transport for a given wind speed. The downwind rate of soil movement varies directly with erodibility of the surface and the width of the eroding field. The effect of any barrier in reducing the rate of soil movement depends on the wind velocity and direction, the threshold velocities needed to initiate soil movement and the barrier shape, width, height and porosity. For planning purposes, the distance of 10 barrier heights (h) has been widely adopted as the minimum protected area on the leeward side of barriers.

The rate of soil movement by wind varies directly as the cube of the wind speed (wind erosion force), above the threshold, and inversely with surface soil water. Considering this, it is apparent that a small decrease in wind speed, some distance from the barrier, could still result in a significant decrease in wind erosion forces.^[9] In one test, the wind speed at 12-h leeward of a barrier was 62% of open-field velocity, whereas wind speed cubed (wind erosion force) was only 25% of that in the open field.^[10]

Tree Windbreaks

One advantage of windbreaks over most other types of wind erosion control is their relative permanency. Siddoway and Fenster^[11] made the point that during a series of drought years, windbreaks may be the only effective and persistent control measure on cropland. Single-row plantings are most common in field windbreaks, and 2- to 5-row plantings are used for farmstead and wildlife protection (Fig. 4). Chepil^[12] found single-row field windbreaks more suitable than double or multiple rows because they occupied the least amount of land area for the amount of protection derived from them.

It usually takes many years for tree windbreaks to reach sufficient height and density for good protection, especially in semiarid areas. Therefore, if used, windbreaks require a



Fig. 4 Single-row field windbreak.
Source: Photo courtesy of USDA.

combination of other conservation practices if they are to be successful.

Herbaceous Wind Barriers

Herbaceous barriers (annual and perennial) have worked well for controlling wind erosion, trapping snow, and reducing evaporation in dryland cropping areas (Fig. 5).

Additional advantages of perennial grass barriers, as Black and Bauer^[13] explained, are ease of establishment, low cost, and lack of potential injury from herbicides used for broadleaf weed control. Research in Montana by Aase, Siddoway, and Black^[14] on tall wheatgrass (*Thinopyrum ponticum*) barriers showed that the barrier-protected land would only experience 6.6% of open-field erosion for the entire year.

Artificial Barriers

Artificial barriers such as snow fencing, board walls, bamboo (*Bambusa bambos*) and willow (*Salix* spp.) fences, earthen banks, hand-inserted straw rows, and rock walls have been used for wind erosion control but only on a very limited basis. There is usually a very high cost in material and labor to construct these barriers, and their use generally restricted to high-value crops.

Strip Cropping

Strip cropping is dividing a large field into strips that are narrow enough to help control wind erosion. Usually strips of erosion-resistant crops or standing stubble are alternated with other crops or unprotected fallow fields. Strip cropping reduces the ever increasing downwind abrasion caused by wind eroding fields and is a very effective erosion control method used extensively in the northern Great



Fig. 5 Herbaceous wind barriers.
Source: Photo courtesy of USDA.

Plains of the United States, Canada, and the spring wheat (*Triticum aestivum*) region of Russia.

MAINTAINING STABLE AGGREGATES OR CLOUDS ON THE SOIL SURFACE

Texture is the dominant soil characteristic relative to inherent erodibility. In general, susceptibility to wind erosion increases as soil texture becomes coarser. Sandy or coarse-textured soil classes lack sufficient amounts of silt and clay to bind sand particles together and form soil aggregates or clods resistant to wind erosion. Consequently, such soils form a “single grain” structure or weakly cemented clods, a condition that is quite susceptible to erosion by wind. A soil surface of stable aggregates of about 70% clods >0.84 mm in diameter resists erosion from all but the highest winds.^[9] Clods of this size are not easily moved by wind, and they protect smaller clods and particles in their lee. Loams, silt loams, and clay loams tend to form the most stable aggregates and are, therefore, the least affected by erosive winds. The type of tillage equipment used has a definite influence on soil cloddiness and surface roughness. Smika and Greb^[15] found that tillage by machines other than the chisel tend to reduce the non-erodible soil aggregation. One-way, offset, or tandem disks leave a smooth surface. Subsurface sweeps, because they do not disturb the soil surface, do not create a rough, ridged soil surface, but they do maintain a greater vegetative roughness by allowing some of the vegetation to remain erect.^[16]

ROUGHENING THE LAND SURFACE

Soil surface roughness is composed of anchored vegetative material, soil ridges, soil clods, or the combinations of all three. All help to control wind erosion by lowering the wind velocity near the soil surface and by sheltering erodible soil fractions.^[17] Tillage implements form ridges and depressions that alter wind velocity. The depressions behind the ridges trap saltating soil particles and stop the normal buildup of eroding material downwind.

Emergency Tillage

Emergency tillage is a last-resort wind erosion control practice that can provide a rough, cloddy surface. It is usually carried out when vegetative cover is depleted by excessive grazing, drought, and improper or excessive tillage, or by growing crops that produce little or no residue (Fig. 6).

Emergency tillage is an inadequate wind erosion control measure, and its only purpose is to create a temporary erosion-resistant soil surface. Implements such as listers, chisels, shovels, and “sand fighters” should traverse fields at right angles to erosive winds to roughen the soil surface and bring clods to the surface.^[9] Listers and narrow chisels were found by Chepil and Woodruff^[16] to have the most



Fig. 6 Field following emergency tillage.

Source: Photo courtesy of USDA.

effective tillage points for emergency tillage. Listers provide a high degree of roughness, and in extremely sandy soils, where clods can be produced only by deep tillage, which are the most effective tools available. Chisel cultivators are more widely used because they require less power and destroy less growing crop than listers.

RESHAPING THE LAND TO REDUCE EROSION ON KNOLLS

Reshaping the land by leveling knolls and benching slopes to shorten the unsheltered distance is an option in wind erosion control but is usually not economical or practical. Because land reshaping is very costly, other effective control measures, such as no-till and seeding to permanent grass, are usually options that are more viable. Hills and knolls affect tillage system requirements indirectly by influencing wind shear stress. When the wind blows over a hill, streamlines of airflow are squeezed together, which increases the wind velocity and shear stress, thereby increasing the erosion potential on the windward slope and hilltop. Consequently, this increases the amount of residue, cloddiness, or roughness needed to control wind erosion on the knoll.

OTHER WIND EROSION SITUATIONS

Wind Erosion on Irrigated Land

Wind erosion on irrigated land can be a serious problem in areas characterized by variable high-wind velocities, where the soils are organic or quite sandy and low in organic matter or where crop residues are inadequate. Under certain conditions, it is impractical and wasteful of water to irrigate frequently enough to prevent a finely pulverized surface

soil from blowing. The depth of drying may only be a fraction of an inch and the soil below this may be wet, but if the immediate surface is dry and the wind is strong enough, the top layer can erode unless the soil particles are consolidated into clods or protected by vegetation. The basic element in erosion control by tillage on irrigated land, as on dryland, is the creation of a rough, cloddy surface which will resist the force of the wind, decrease its velocity at the ground level, and trap moving soil. Sandy soils, usually found in irrigated areas, are far more difficult to protect by emergency measures than fine-textured soils.

Wind Erosion Control on Sand Dunes and Other Problem Areas

Wind erosion control on sand dunes and other problem areas require measures that are more intensive to get sand dunes in check. Dunes lack a soil profile because they are unstable and underdeveloped. The sand is fine, loose, and easily moved by wind. It has no organic matter, consequently retains little moisture for plants, and has inherently low fertility. Sand dunes and drift areas often require artificial barriers or cover for stabilization before vegetation can be established. These include oil, clay gravel, picket fence, brush, straw, and hay. Clay is effective but is expensive. Hay or straw can be used as temporary mulches on blowout or small areas of dune sand at road cuts, around dwellings, and other disturbed areas. They provide some organic matter, which is critical for successful dune plantings. The establishment of permanent vegetation is the final objective in the stabilization of dunes.

Other Non-Vegetative Erosion Protection

Some of the non-vegetative and processed vegetative materials used are gravel and crushed rock, various surface films such as resin-in-water emulsion (petroleum origin), rapid-curing cutback asphalt, asphalt-in-water emulsion, starch compounds, latex-in-water emulsion (elastomeric polymer emulsion), by-products of the paper pulp industry, and wood cellulose fiber.^[2] Several of these spray-on adhesives are available for temporary wind erosion control of vegetable seedlings on mineral soils. Some of the adhesives are relatively expensive, but a few are economically feasible on high-value crops threatened by serious blowing that cannot be controlled by other methods.

REFERENCES

1. Lyles, L. Basic wind erosion processes. *Agr. Ecosyst. Environ.* **1988**, 22/23, 91–101.
2. Woodruff, N.P.; Lyles, L.; Siddoway, F.H.; Fryrear, D.W. *How to Control Wind Erosion*, U.S.D.A., SEA Agric. Inf. Bull. No. 354; U.S.D.A., A.R.S.: Washington, 1972; 22 pp.
3. Greb, B.W. *Reducing Drought Effects on Cropland in the Western-Central Great Plains*, U.S.D.A., SEA Agric. Inf. Bull. No. 420; U.S.D.A., A.R.S.: Washington, 1979; 31 pp.
4. Haas, H.J.; Willis, W.O.; Bond, J.J. *Summer Fallow in the Western United States*. Agriculture Research Service Conservation Research Report No. 17. U.S.D.A., 149–160. Washington, D.C., 1974.
5. Watson, S., Ed. *Kansas No-Till Handbook*; Kansas State University: Manhattan, 1999; 72 pp.
6. Rice, R.W. *Fundamentals of No-Till Farming*; American Association for Vocational Instructional Materials: Winterville, 1983; 148 pp.
7. Crovetto, C. *Stubble over the Soil: The Vital Role of Plant Residue in Soil Management to Improve Soil Quality*; American Society of Agronomy: Madison, 1996; 245 pp.
8. Domitruk, D.; Crabtree, B., Eds. *Zero Tillage. Advancing the Art*; Manitoba-North Dakota Zero Tillage Farmers Association: Brandon, 1997; 40 pp.
9. Tibke, G. Basic principles of wind erosion control. *Agr. Ecosyst. Environ.* **1988**, 22/33, 103–122.
10. Skidmore, E.L.; Hagen, J.L. Reducing wind erosion with barriers. *T. ASAE* **1977**, 20, 911–915.
11. Siddoway, F.H.; Fenster, C.R. Soil conservation: Western great plains. In *Dryland Agriculture*, Monograph No. 23; Dregne, H.E., Willis, W.O., Eds.; American Society of Agronomy: Madison, 1983; 231–246.
12. Chepil, W.S. Wind erosion control with shelterbelts in North China. *Agron. J.* **1949**, 41, 127–129.
13. Black, A.L.; Bauer, A. Soil conservation: northern great plains. In *Dryland Agriculture*, Monograph No. 23; Dregne, H.E., Willis, W.O., Eds.; American Society of Agronomy: Madison, 1983; 247–257.
14. Aase, J.K.; Siddoway, F.H.; Black, A.L. Effectiveness of grass barriers for reducing wind erosiveness. *J. Soil Water Conserv.* **1985**, 40, 354–357.
15. Smika, D.E.; Greb, B.W. Nonrodible aggregates and concentration of fats, waxes and oils in soils as related to wheat straw mulch. *Soil Sci. Soc. Am. Proc.* **1975**, 39, 104–107.
16. Chepil, W.S.; Woodruff, N.P. The physics of wind erosion and its control. *Adv. Agron.* **1963**, 15, 211–302.
17. Armbrust, D.V.; Chepil, W.S.; Siddoway, F.H. Effects of ridges on erosion of soil by wind. *Soil Sci. Soc. Am. Proc.* **1964**, 28, 557–560.

Wind Erosion: Environmental Effects

R. Scott Van Pelt

Wind Erosion and Water Conservation Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Big Spring, Texas, U.S.A.

Abstract

Wind erosion is the movement and loss of soil, resulting from the interaction of a bare, loose, and dry soil surface with wind. Although wind erosion occurs in all continents and in all climates, it is most prevalent and its effects are most profound in arid and semiarid areas of the northern hemisphere. Wind erosion is a soil-degrading process that reduces soil productivity, damages crops, and may, in extreme cases, result in the burial of fertile soil horizons and structures. The most visible evidence of wind erosion is fugitive dust (soil-derived dust emanating from and transported beyond the boundaries of a source field) that may be entrained in global circulation patterns and transported hundreds or thousands of kilometers from the source. Global estimates of soil dust flux to the atmosphere range from 250 to 5000 Tg yr⁻¹ on an annual basis. Estimates of return flux to the surface of land areas range from <10 to >200 t km⁻² yr⁻¹, and yearly estimates of total flux to the world's oceans range from 532 to 851 Tg yr⁻¹. Some of the dust returns to the earth by the settling of larger particles from the atmosphere, but much more is scavenged by cloud formation and rainfall. Fugitive dust contains higher concentrations of organic carbon; basic cations such as calcium ion, magnesium ion, and sodium ion; plant nutrients such as nitrogen, phosphorus, potassium, and iron; and numerous trace metals than the soils from which it was derived. These chemical species contained on dust, especially when hydrated, have been documented to react with and catalyze reactions with other atmospheric particulates, colloids, and gases and therefore have an influence on rainfall chemistry, the chemistry of surface waters, and the fertility of midlatitude oceans. Soil dust has been documented to transport living organisms including bacteria, fungi, and algae. Atmospheric dust may also have an effect on climate forcing by absorbing solar radiation in the atmosphere and reducing the radiation flux on the earth's surface. This entry explores many ways that wind erosion and the resulting fugitive dust impact various environments on earth.

INTRODUCTION

Wind erosion is the result of wind impacting a dry, bare, and loose soil surface. Approximately 500 million hectares worldwide are susceptible to wind erosion, primarily in the arid and semiarid regions of the northern hemisphere. Wind erosion is a natural process that has resulted in the sculpting of landscapes and the deposition of fine soil particles in regions downwind of the actual erosion. Locally, wind erosion is a soil-degrading process resulting in the deflation of fertile topsoil, decrease in soil productivity and agricultural sustainability, disruption of commerce, and burial of fertile topsoil and man-made structures.

The fine particles that become entrained in the wind and transported great distances contain higher concentrations of organic carbon; basic cations such as sodium ion (Na⁺), calcium ion (Ca²⁺), and magnesium ion (Mg²⁺); plant nutrients such as nitrogen (N), phosphorus (P), potassium (K), and iron (Fe); trace metals; and soil contaminants than the soils from which they originated. Fugitive dust (soil-derived dust emanating from and transported beyond the boundaries of a source field) is an effective transporter of

living organisms such as bacteria, fungi, and algae and has been linked with human and environmental health problems. Soil dust reacts with and catalyzes reactions with atmospheric gases and pollutants, thus reducing the deleterious effects of acid rain and cleaning the atmosphere of other anthropogenic contaminants.

WIND EROSION AND DEPOSITION PROCESSES

Wind erosion and eolian (wind interacting with the earth's surface) processes are natural phenomena that have shaped the earth since time began. The eolian re-sorting of sand and smaller mineral particles from braided streams and beach environments has resulted in deep deposits of Loess in many regions of the earth. It has only been during the 20th century that wind erosion has been studied in depth and that an understanding of the controlling processes has been emerged. Furthermore, the specific study of the direct and indirect environmental effects is only a few decades old.

As wind blows over a surface, ephemeral gusts move particles in the sand to fine sand size range, which either

creep along the soil surface or become entrained in the wind. Most sand-sized particles do not rise more than 30 cm before returning to the surface, accelerated by the wind, with a significant horizontal component of motion. These saltating grains strike the surface with considerable force, releasing more saltating particles and abrading finer soil particles from soil aggregates and crusts resulting in plumes of dust.^[1] These finer particles are easily entrained into the turbulent eddies to heights often exceeding 2–3 km. The median diameter of entrained particles decreases as altitude increases. As the particles are transported downwind, the higher terminal velocity of the larger particles and the shallower transport depth cause them to settle from the atmosphere first and thus closer to the source. Particles <20 μm in diameter have much lower terminal velocities and may remain entrained and transported in the atmosphere for days or weeks. Particles return to the earth's surface primarily by gravity settling (dry deposition) and by inclusion in rain drops (wet deposition). A classic and thorough treatise on wind erosion processes is presented in *The Physics of Blown Sand and Desert Dunes*.^[2]

SOURCES AND MAGNITUDE OF SOIL DUST

The principal source regions of soil dust are arid and semi-arid regions of the northern hemisphere^[3] (Fig. 1). Large expanses of bare, dry, and often loose soil susceptible to wind erosion exist in these regions. It is estimated that more than half a billion hectare worldwide contributes to the atmospheric dust load. The exact amount of dust that is entrained from these regions is uncertain due to the variability among source regions, within individual source regions, among years, and the variability of estimation methods. Estimates of annual dust entrainment vary from 250 to >5000 Tg yr^{-1} . Deposition rate estimates vary from 10 Tg yr^{-1} to 200 $\text{kg km}^{-2} \text{ yr}^{-1}$ for land areas, and total deposition flux to the oceans has been estimated at 532–851 Tg yr^{-1} . Dust transported from North Africa is deposited in Europe, the Atlantic Ocean, North America, South America, Asia, and other locations in Africa. Dust transported from the great source regions of Asia is deposited in the North and Tropical Pacific basins, North America, Greenland, and Southeast Asia. Evidence from eolian Loess deposits worldwide indicates that the magnitude of dust transport was much greater in the geological past.

ON-SITE EFFECTS OF WIND EROSION

Areas subject to wind erosion may be impacted directly by the blowing sand and dust. Among the short-term direct effects near source areas are reduced visibilities, which impair commerce, endanger travelers, and temporarily destroy scenic enjoyment and recreation opportunities. On cropland, plants may be damaged or killed by sandblast injury. Although these effects impact human activities and

economic success, it is the long-term effects that are of the greatest importance. Wind erosion is a soil-degrading process that selectively removes the fine, chemically active portion of the soil. Fine particles have greater surface-area-to-volume ratios than the bulk soil and thus are more important to nutrient cycling and water retention. The loss of fine mineral particles and soil organic carbon results in a less productive soil and less sustainable agroecosystem.^[4]

Coarser particles are often deposited near the field margins and may bury fences or clog drainage features. In extreme cases of widespread soil degradation, wind erosion can contribute to desertification and desert encroachment.^[5] The destabilization of fragile sandy soils may result in the burial of more fertile soils (Fig. 2) and man-made structures, including buildings, irrigation canals, highways, and rail routes. Migrating dune fields may travel several kilometers per year. Desert encroachment has displaced human populations in several regions around the earth and continues to threaten human population centers and economic enterprise.

OFF-SITE EFFECTS OF WIND EROSION

Soil dust composed of very fine soil particles <20 μm in diameter may be transported hundreds or thousands of kilometers by global atmospheric circulation patterns before being deposited on the earth's surface.^[3] The characteristics of soil dust are determined by the surface from which it was entrained.^[6] As the primary source regions for soil dust are arid and semiarid areas, it is not surprising that the dust from these regions contains an abundance of basic cations such as Ca^{2+} , Mg^{2+} , and Na^+ , carbonate and bicarbonate minerals, and soluble salts. Soil dust also contains enriched concentrations of plant nutrients such as N, P, K, Fe, and trace metals important to biogeochemical cycles of ecosystems.^[7] Carbon in many forms, including soluble organic compounds, and pesticides and daughter products, is also transported on soil dust.^[8] Viable biological organisms such as bacteria and fungi^[9] have been documented on soil dust. These chemical and biological “hitchhikers” contribute nutrients to agroecosystems, open ocean areas, and watersheds, which may be deficient in one or more of these nutrients, and to the cycling of these nutrients. The fertilization of nutrient-deficient waters of midlatitude oceans may result in the proliferation of algal populations and thus may have a minor impact on global carbon cycles. It has been suggested that dust emanating from the Sahara Desert may be a cause of coral diseases in the Caribbean Sea.^[10]

Fine particles tend to be hygroscopic in nature, absorbing available water vapor from the surrounding air mass, and are very efficient condensation nuclei. As water condenses around the dust particle, soluble salts and organic compounds dissolve. The film of water around the particles in clouds also facilitates chemical reactions between the minerals present and the atmospheric gases and non-soil

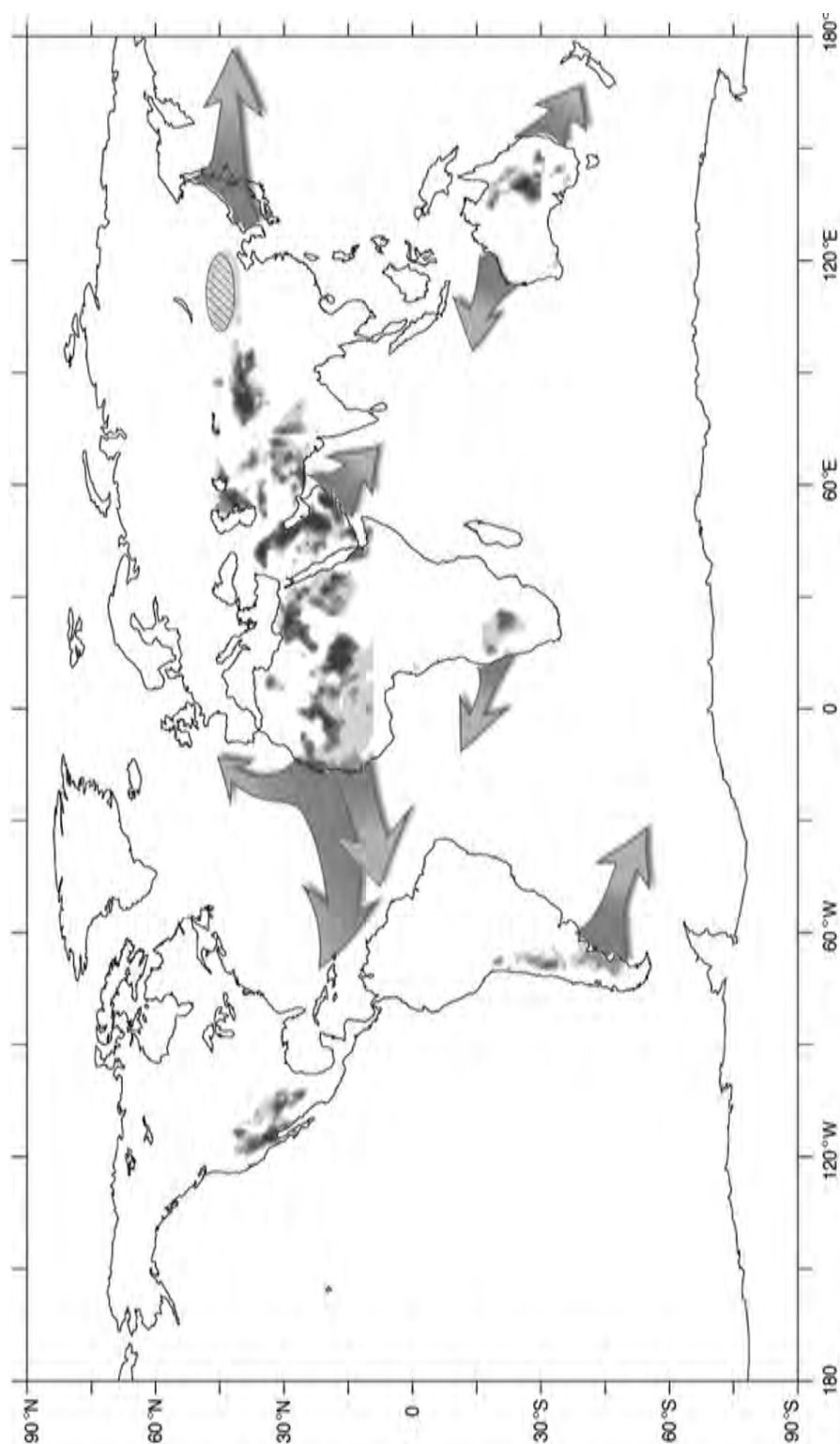


Fig. 1 The global distribution of major dust sources and dust transport paths based on the Total Ozone Mapping Spectrometer aerosol product. The darker gray tones show those regions where substantial concentrations of dust are lifted into the air more than 50% of the time during the dusty seasons of the year. The arrows show the main transport paths over the oceans. The arrow size is not indicative of the magnitude of the transport or the distance that dust is carried.

Source: From Prospero, Ginoux, et al.^[3]



Fig. 2 Encroachment of migrating dunes from the Tengger Desert in Minqin County, Gansu Province, China.

particulates. In industrial areas with high fossil-fuel use, oxides of S and N are present in larger concentrations and are also dissolved in the rain, resulting in rainfall with much higher acidity. This acid rain has been credited with damage to forest and aquatic ecosystems. In areas frequented by high loads of soil dust, soluble basic minerals, primarily carbonates, partially or totally neutralize the acidity.^[11] Finally, the role of soil dust in radiative forcing of climate and the extent to which present and projected future dust levels may influence the earth's energy balance and temperature are in debate.^[12]

CONCLUSIONS

Wind erosion affects a large portion of the earth's surface, with direct effects in the source regions. The fine soil dust results from wind erosion may be transported to great distances and impacts the regions through which it passes and on which it is deposited. The input of basic cations, plant nutrients, soluble salts, soluble organics, and trace elements may be locally important sources for deficient ecosystems but may also have a deleterious effect on the quality of surface waters. However, dust may positively impact water quality in areas prone to acid precipitation. More research is

needed to quantify the effects of dust on environmental quality and climate.

REFERENCES

1. Zobeck, T.M.; Van Pelt, R.S. Wind-induced dust generation and transport mechanics on a bare agricultural field. *J. Hazard. Mater.* **2006**, *132*, 26–38.
2. Bagnold, R.A. *The Physics of Blown Sand and Desert Dunes*; Methuen: London, 1941; 265 pp.
3. Prospero, J.M.; Ginoux, P.; Torres, O.; Nicholson, S.E.; Gill, T.E. Environmental characterization of global sources of atmospheric dust identified with the Nimbus 7 Total Ozone Mapping Spectrometer (TOMS) absorbing aerosol product. *Rev. Geophys.* **2002**, *40* (1), 2–1–2–37.
4. Nordstrom, K.F.; Hotta, S. Wind erosion from cropland in the USA: A review of problems, solutions, and prospects. *Geoderma* **2004**, *121*, 157–167.
5. Zhang, K.; Qu, J.; Zu, R.; Fang, H. Temporal variations of sandstorms in Minqin oasis during 1954–2000. *Environ. Geol.* **2005**, *49*, 332–338. DOI:10.1007/s00254-005-0096-x.
6. Pye, K.E. *Aeolian Dust and Dust Deposits*; Academic Press: London, 1987; 334 pp.
7. Van Pelt, R.S.; Zobeck, T.M. Chemical constituents of fugitive dust. *Environ. Monit. Assess.* **2007**, *130*, 3–16. DOI:10.1007/s10661-006-9446-8.
8. Van Pelt, R.S.; Zobeck, T.M. Carbonaceous materials in soil-derived dusts. In *Soil Carbon Sequestration and the Greenhouse Effect*, 2nd Ed.; Lal, R., Ed.; Soil Science Society of America: Madison, 2009; 365–392.
9. Gorbushina, A.A.; Kort, R.; Schulte, A.; Lazarus, D.; Schnetger, B.; Brumsack, H.-J.; Broughton, W.J.; Favet, J. Life in Darwin's dust: Intercontinental transport and survival of microbes in the nineteenth century. *Environ. Microbiol.* **2007**, *9* (12), 2911–2922.
10. Weir-Brush, J.R.; Garrison, V.H.; Smith, G.W.; Shinn, E.A. The relationship between gorgonian coral (Cnidaria: Gorgonacea) diseases and African dust storms. *Aerobiologia* **2004**, *20*, 119–126.
11. Loye-Pilot, M.D.; Martin, J.M.; Morelli, J. Influence of Saharan dust on the rain acidity and atmospheric input to the Mediterranean. *Nature* **1986**, *321*, 427–428.
12. Tegen, I.; Werner, M.; Harrison, S.P.; Kohfeld, K.E. Relative importance of climate and land use in determining present and future global soil dust emissions. *Geophys. Res. Lett.* **2004**, *31*, L05105. DOI:10.1029/2003GL019216.

Wind Erosion: Field Measurement

R. Scott Van Pelt

Wind Erosion and Water Conservation Research Unit, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), Big Spring, Texas, U.S.A.

Ted M. Zobeck

U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS) (Retired), Lubbock, Texas, U.S.A.

Abstract

Wind erosion and deposition result when wind moves soil from a bare susceptible surface to another location downwind. Although placement of permanent vertical references such as pins or rods has been used to measure soil redistribution, it is more commonly measured by capturing sediment moving during an event. Through the years, several sampling devices have been developed to trap sediment in transport or to measure fine dust that may linger after a wind erosion event. Some of the devices are passive, meaning that they use the force of the wind and the momentum of the sediment to fill the sampler, and some are active, meaning that a stream of air is drawn through the sampler by a vacuum source. In addition to sampling sediment in transport and suspended in the atmosphere, methods have emerged that allow estimation of redistribution rates by comparing soilborne environmental tracer content on a landscape with the tracer content of a location in which no erosion or deposition has occurred.

INTRODUCTION

Soil is redistributed when the erosive force of the wind impacts a bare susceptible surface, entraining particles that travel a distance and then return to the surface. Accurate and reliable measurement of this redistribution is necessary in order to confirm and validate erosion models, to assess the magnitude of soil redistribution,^[1] and for other applications. Soil erosion is a degradative process that impacts soil productivity at the source, and the resultant fugitive dust impacts visibility, commerce, and environmental and human health. Of particular interest is the role that microorganisms transported on dust play in ecosystem processes and methods to sample for living microorganisms from dust suspended in the atmosphere.^[2] Wind-entrained particles are transported in three modes—Heavier, larger particles tend to roll along the surface in creep mode, slightly smaller particles tend to bounce along the surface in saltation mode, and the finest particles are levitated on wind eddies and may travel from a few meters to millions of meters in suspension mode. In general, there is no single sampling method that works equally well for all modes of transport.

SAMPLING CONSIDERATIONS

Measuring wind erosion may be accomplished by repeated measurement of the soil surface with respect to a reference such as a rod or pin driven into the soil^[3] or, more

commonly, by sampling and extrapolating from the amount of wind-entrained sediment moving during a wind erosion event. The exact type of sampling equipment used to collect wind-entrained sediment during and after an erosion event will vary depending on the mode and direction of particle transport. Sampling particles moving horizontally by creep and saltation may require sampling equipment that will quickly align with the direction of the wind, whereas sampling fine dust rising on atmospheric eddies may be accomplished with samplers that are not aligned with the wind field. Fine fugitive dust resulting from an erosion event may remain suspended after the end of the event and be deposited in a vertical direction during calm wind conditions or be washed from the atmosphere by rain.

Changes in the direction or velocity of the wind field may affect sampler efficiency. An ideal sampler for sediment moved by the wind should not only quickly orient the sampler opening into the wind but should also have openings that are isokinetic and relatively insensitive to rapid fluctuations of wind direction.^[4] Isokinetic means that wind enters the sampler at the same velocity encountered adjacent to the sampler. Due to the limitations of some sampler designs, samplers vary in terms of their sampling efficiency. If the sampler collects all the sediment that is in the wind field, then it is 100% efficient, and if it collects only half of the sediment in the wind field, then the efficiency is reduced to 50%. Samplers may be nearly 100% efficient for one range of particle diameters and much less efficient for other ranges of particle diameters, leading to sampling bias.

Many different types of sediment samplers and sensors have been used to measure wind-induced soil erosion and deposition.^[5,6] Samplers are either passive, relying on ambient wind fields to collect sediment samples, or active, relying on an external source of vacuum to pull sediment-laden air into the sampler. In general, passive samplers are primarily used to sample sediment in the creep and saltation mode, whereas active samplers are commonly used to measure fine suspended dust.

Passive Samplers

Passive samplers have the advantage for field measurement of not requiring external power sources and are thus relatively inexpensive to purchase and maintain. Passive samplers for measuring creep have either vertically oriented openings at ground surface or horizontally oriented openings very close to the surface. Perhaps the simplest creep sampler is a depression or series of depressions in the surface in which the creeping material is deposited. Inaccuracies may occur in this simple approach, as original surface material may be confused with the material transported by wind. A simple creep sampler may be made by burying a jar with a small hole in the lid so that the lid is even with the soil surface. Unfortunately, scour of surrounding material may result in the lid becoming superior to the soil surface and losing its supply of sediment transported in creep mode.

Combinations of creep and saltation samplers with horizontal openings have been designed by many researchers including Bagnold^[7] to measure creep and the vertical profile of saltating sediment particles. The Bagnold sampler (Fig. 1) is simply a vertical series of inclined pockets oriented into the expected direction of the erosive wind. The lowest opening had its opening at or slightly below the ground level and was connected to a subterranean sediment reservoir. Pockets above the lowest opening have their own reservoir at the bottom of the incline, but the sampler had a solid back and was not isokinetic. The Bagnold sampler has been found to be only about 20% efficient.^[5] However, by loosening the connection between the lowest slot and the reservoir, the efficiency may increase to 50% as less back pressure will be created at the sampler opening.^[5] Chepil improved on the Bagnold design by mounting the sampler on a rotating plate and adding a wind vane to keep the openings oriented into the wind.^[8] Other improvements through the years have included changing the shape to a wedge with the small end oriented into the wind and adding fine screen to improve the air flow into and through the sampler. A wedge-shaped slot sampler with an opening 50 cm high and 2 cm wide with a fine screen on the back has been found to be over 90% efficient for creep and saltation-sized particles.^[3]

U.S. Department of Agriculture has developed a creep and near-surface saltation sampler (Fig. 2) that is installed flush with the surface and has openings at 0–3, 3–9, and 9–20 mm above the surface.^[9] The sampler is built so that

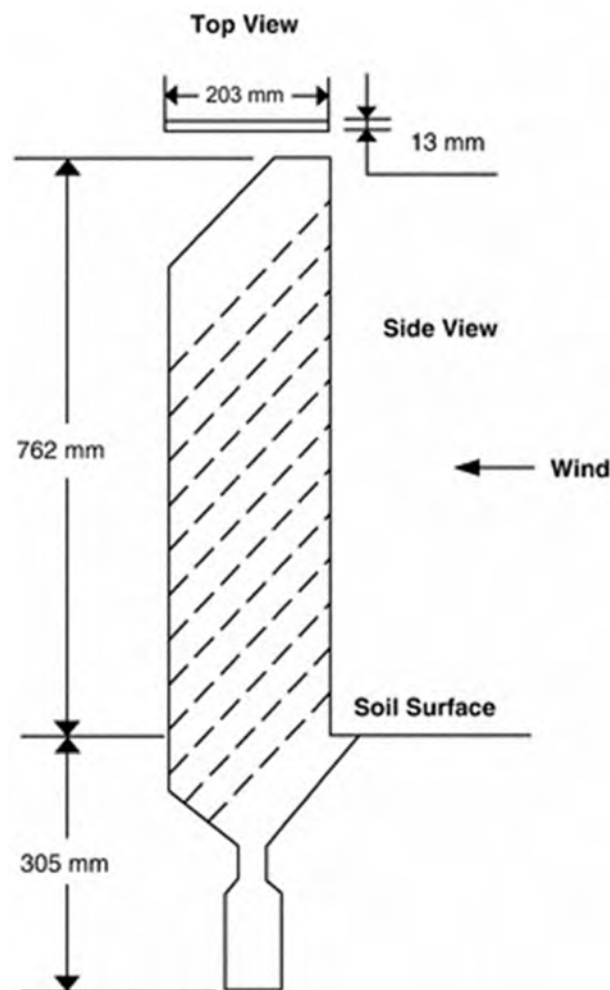


Fig. 1 Schematic diagram of a Bagnold sampler.

Source: From Horikawa & Shen.^[5] ©1960 U.S. Army Corps of Engineers, Beach Erosion Board; Bagnold.^[7] ©1941 William Morrow & Co.

the openings and subtending sediment reservoirs can rotate and stay oriented into the direction of the wind. Companion saltation and suspension samplers for this near-surface sampler are wedge-shaped with screened air outlets and may be mounted on a vertical shaft at multiple arbitrary heights above the surface.^[10] These samplers have become very popular for wind erosion studies globally and are known as Big Spring Number Eight (BSNE) samplers. They may be purchased or fabricated with single large openings or with three BSNE samplers linked together with smaller openings for collecting samples near the ground (Fig. 3). The efficiency of the BSNE has been found to exceed 90%.^[10]

A very inexpensive and popular sampler for measuring saltation and suspension is the Modified Wilson and Cooke (MWAC) samplers.^[11] These are easily fabricated using bendable tubing and bottles (Fig. 4). Since the early 1990s, MWACs and BSNEs have been the saltation and suspension samplers of choice by the international Aeolian

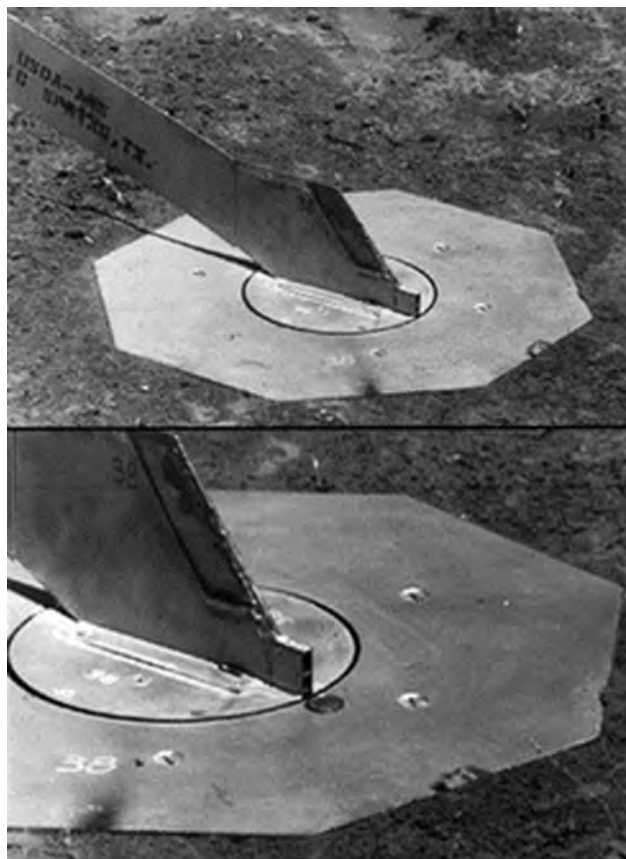


Fig. 2 Creep-saltation sampler (top) and close-up of the sampler openings (bottom); a dime is placed in front of the sampler openings for scale.

community. Both samplers have high efficiencies and are easy to use in field studies but collect the samples over a time period defined by the frequency of sample collection. Passive samplers have been developed that measure the high-frequency temporal variations in sediment transport by placing the saltation and creep sampler over a buried box containing either a microbalance^[12] or a tipping bucket.^[13] Other passive samplers are available with load cells integrated into their design so that sediment is automatically weighed as it is collected (Fig. 5).

Temporal variations of saltation may also be measured with electronic sensors of various types. Among the first electronic sensors, the Saltiphone (Fig. 6) utilized a microphone capsule behind a protective cover that was mounted on a finned tube to pivot into the wind. An impact by a saltating sand grain or aggregate created a sound resulting in an electrical impulse that could be recorded by a data logger. Other sensors such as the Sensit (Fig. 7) and Safire utilize piezoelectric crystal sensors to create the electrical impulse. They have an advantage over the Saltiphone by being cylindrical and not needing to pivot into a rapidly changing wind field. These saltation impact sensors have been tested, and their performance compared in a laboratory wind tunnel utilizing different particle diameters and



Fig. 3 BSNE sampler mast. Note that the lowest section is composed of three samplers mounted together, and the lowest two samplers have smaller openings.

wind speeds.^[14] Electronic sensors utilizing optical technology (Fig. 8) have been developed and used in Aeolian studies.^[15] The user should clean the optics often to obtain reliable saltation data with these sensors. The rapid response of these electronic sensors has permitted the development of a localized definition of threshold wind velocity based on the percentage of time that sediment is being transported by an observed wind field.^[16]

Suspension Samplers

Particles finer than 100 μm in diameter may become entrained in turbulent eddies, transported horizontally and vertically, and lifted to great heights during wind erosion events. Once lifted to high altitude, the particles, especially the finer ones, may travel hundreds to thousands of kilometers in a dust haze before returning to the earth by gravity. Passive samplers such as moss and pans or buckets containing marbles (Fig. 9) have been used to measure dust fall. Pan-type samplers containing marbles to prevent resuspension of the captured dust are easily and inexpensively



Fig. 4 MWAC sampler mast. Individual samplers are easily constructed using tools and items locally available.

built using an aluminum angel food cake pan with 250 mm diameter and 100 mm depth and thus have been used extensively in regional dust deposition studies.^[17] These are typically mounted atop a pole at a height of 2 m above the ground in a non-eroding, vegetated area. Some method to discourage bird perching on the sampler should be employed to prevent fecal contamination.



Fig. 5 BSNE with hinged support and load cell to facilitate the continuous weighing of collected sediment.

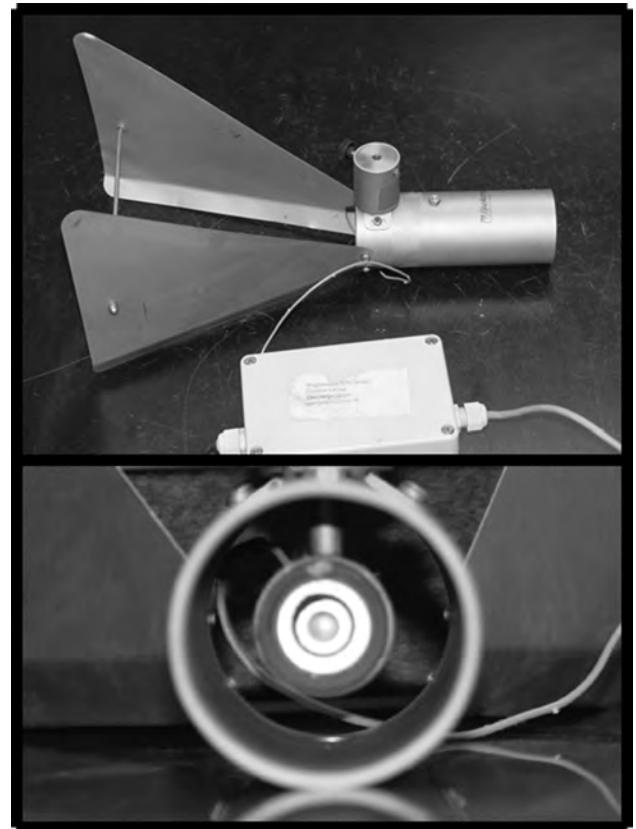


Fig. 6 Saltiphone acoustical impact sensor. The top picture shows the overall assembly including the pivot point and wind fins, and the bottom picture details the covered microphone capsule.

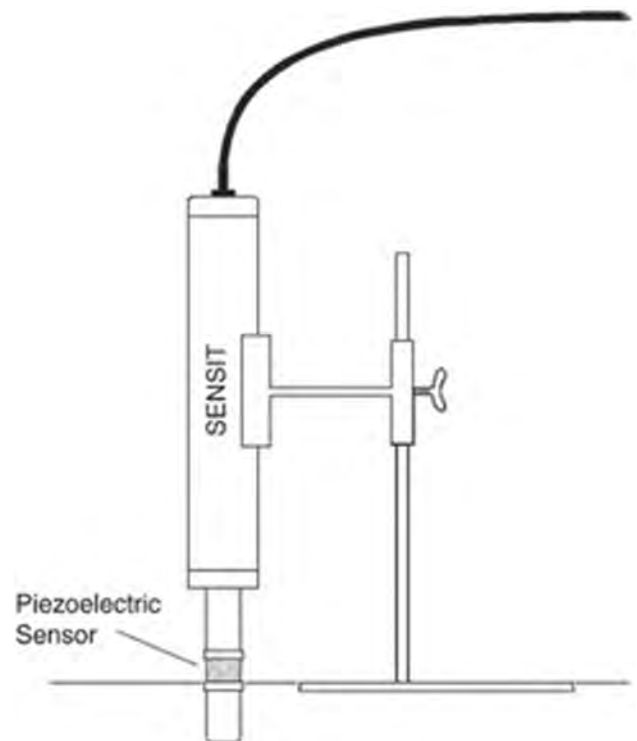


Fig. 7 Sensit particle impact sensor.



Fig. 8 Wenglor optical particle counter mounted for field studies.

Saltation samplers such as the BSNE and MWAC also collect some horizontally moving suspension-sized sediment. The MWAC has been found to be nearly twice as efficient at trapping fine suspension-sized sediment compared to the BSNE.^[1] This study also indicated that



Fig. 9 Pan-type dust collector with marbles. The pan diameter is 250 mm, and the metal bands crossing over the top discourage bird perching.

the efficiency of the BSNE was constant, whereas the efficiency of the MWAC varied with wind speed.

Active sampling devices using mechanically generated suction are often used to sample suspension-sized sediment from moving and stationary air bodies. Many of the devices draw air through a filter that is periodically removed and weighed. Among the first to use this method of sampling, Chepil and Woodruff used aspirated 48-mm filters mounted at 4 heights to measure dust in Kansas and Colorado.^[18] Today researchers can buy large or small self-aspirated filtering devices that many municipalities use to monitor air quality. A relatively new development is the Tapered Element Oscillating Microbalance (TEOM) that draws air through a small filter and weighs that filter at regular intervals to provide a history of dust concentration. For more frequent measurement of dust concentration in the air, optical sensors designed to monitor air quality in industrial

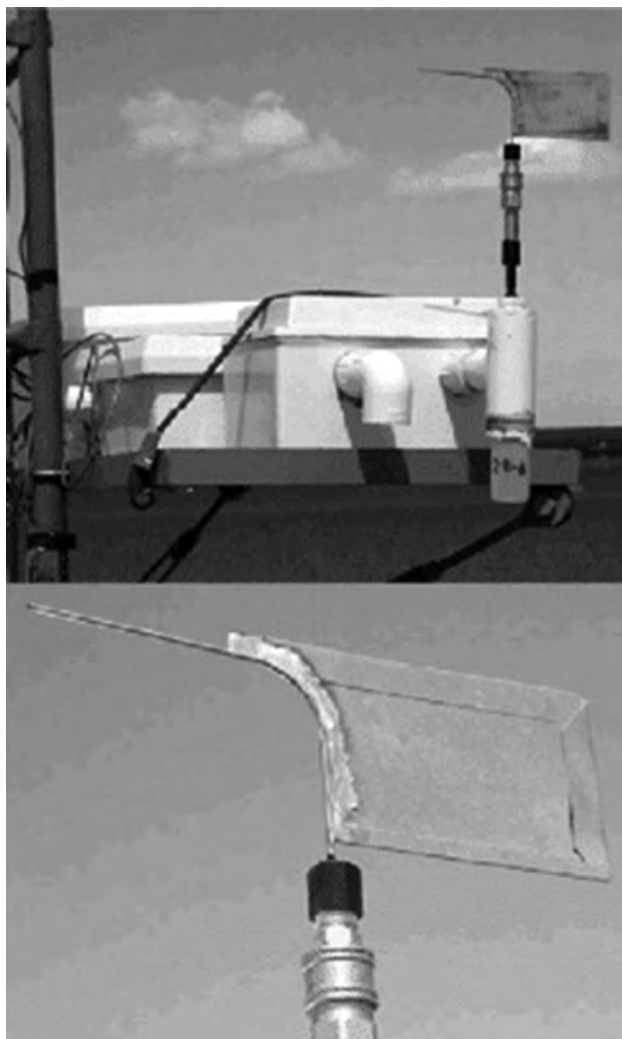


Fig. 10 USDA-ARS fine particle dust sampling head attached to a dust monitoring system (top) and close-up of sampling head (bottom).

settings may be used. Such devices with horizontal tubes that pivot into the wind (Fig. 10) have been used to frequently and quasi-isokinetically sample dust at multiple heights above the ground at several downwind locations over an eroding field.^[19] The entry successfully showed a link between temporal variability of saltation and vertical dust flux.

TRACER-BASED METHODS

Soilborne tracers that have appeared at a unique time in history may be used to estimate soil redistribution rates on the landscape at a decadal timescale. An example of this would be the use of coal fly ash in the soils adjacent to rail lines in which coal was burned in locomotives from the middle 19th century to the middle 20th century. By comparing the amount of time-stable tracer material on a landscape with suspected soil redistribution with the content in the soil of a known stable landscape such as a well-grassed hilltop or courthouse lawn, the difference in tracer content can be linked to soil redistribution. The advantage of this approach includes the necessity for only one trip to sample the soil profile at several locations in the landscape of interest. The most commonly used tracers for this type of soil redistribution measurement are anthropogenic radioisotopes such as ¹³⁷Cs, an isotope of cesium that did not exist in more than trace amounts prior to the atmospheric testing of nuclear weapons in the late 1950s and early 1960s. Anthropogenic radioisotopes have been used to estimate average rates of soil redistribution on nearly every continent of the world.^[20] In addition to anthropogenic radioisotopes, naturally occurring isotopes with relatively short half-lives such as ²¹⁰Pb and ⁷Be have also been successfully used to estimate soil redistribution rates at less than decadal timescales. Finally, chemical tracers such as rare earth elements may be applied to the surface soil layer and tracked on the landscape.

CONCLUSION

Many sediment sampling devices, both passive and active, as well as other methods exist to measure various spatial and temporal aspects of wind erosion. Many of the passive devices are easily and inexpensively constructed and have been extensively used by researchers globally. Other technologies are somewhat more expensive and used for specialized studies of Aeolian processes. Robotics (Fig. 11) offer a unique opportunity to deploy sampling equipment^[21] to investigate Aeolian processes at multiple spatial scales. With advances in technology, routine rapid measurement of wind fields, saltation, and dust concentration may become more affordable and subsequently routine.



Fig. 11 Hexapod robot with two-axis sonic anemometer sampling the wind fields in a cluster of coppice dunes.

REFERENCES

1. Goossens, D.; Offer, Z.Y. Wind tunnel and field calibration of six aeolian dust samplers. *Atmos. Environ.* **2000**, *34*, 1043–1057.
2. Acosta-Martinez, V.; Van Pelt, R.S.; Moore-Kucera, J.; Bad-dock, M.C.; Zobeck, T.M. Microbiology of wind-eroded sediments: Current knowledge and future research directions. *Aeolian Res.* **2015**, *18*, 99–113. Available at <http://dx.doi.org/10.1016/j.aeolia.2015.06.001>.
3. Gibbens, R.P.; Tromble, J.M.; Hennessey, J.T.; Cardenas, M. Soil movement in mesquite dunelands and former grasslands of southern New Mexico from 1933 to 1980. *J. Range Manage.* **1983**, *36* (2), 145–148.
4. Nickling, W.G.; McKenna Neumann, C. Wind tunnel evaluation of a wedge-shaped aeolian sediment trap. *Geomorphology* **1997**, *18*, 333–345.
5. Horikawa, K.; Shen, H.W. *Sand Movement by Wind Action: On the Characteristics of Sand Traps*, Tech Memorandum No. 119; U.S. Army Corps of Engineers, Beach Erosion Board: Vicksburg, 1960; 51 pp.
6. Mark, D.; Hall, D.J. Recent developments in airborne dust monitoring. *Clean Air* **1994**, *23*, 193–217.
7. Bagnold, R.A. *The Physics of Blown Sand and Desert Dunes*; William Morrow & Co.: New York, 1941; 265 pp.
8. Chepil, W.S. *Width of Field Strips to Control Wind Erosion*, Tech Bull. 92; Kansas State College Agricultural Experiment Station: Manhattan, 1947; 16 pp.
9. Zobeck, T.M.; Sterk, G.; Funk, R.; Rajot, J.-L.; Stout, J.E.; Van Pelt, R.S. Measurement and data analysis methods for field-scale wind erosion studies and model validation. *Earth Surf. Proc. Land.* **2003**, *28* (11), 1163–1188.

10. Fryrear, D.W. A field dust sampler. *J. Soil Water Conserv.* **1986**, *41* (2), 117–120.
11. Wilson, S.J.; Cooke, R.U. Wind erosion. In *Soil Erosion*; Kirky, M.J., Morgan, R.P.C., Eds.; John Wiley and Sons: New York, 1980; 217–251.
12. Jackson, D.W.T. A new, instantaneous aeolian sand trap design for field use. *Sedimentology* **1991**, *43* (5), 791–796.
13. Bauer, B.O.; Namikas, S.L. Design and field test of a continuously weighing tipping-bucket assembly for aeolian sand traps. *Earth Surf. Proc. Land.* **1998**, *23* (13), 1171–1183.
14. Van Pelt, R.S.; Peters, P.; Visser, S. Laboratory wind tunnel testing of three commonly used saltation impact sensors. *Aeolian Res.* **2009**, *1* (1), 55–62.
15. Hugenholtz, C.H.; Barchyn, T.E. Laboratory and field performance of a laser particle counter for measuring aeolian sand transport. *J. Geophys. Res.* **2011**, *116* (F1), F01010. doi:10.1029/2010JF001822.
16. Stout, J.E. A method for establishing the critical threshold for aeolian transport in the field. *Earth Surf. Proc. Land.* **2004**, *29* (10), 1195–1207.
17. Reheies, M.C.; Kihl, R. Dust deposition in southern Nevada and California, 194–1989: Relations to climate, source area, and source lithology. *J. Geophys. Res.* **1995**, *100* (D5), 8893–8918.
18. Chepil, W.S.; Woodruff, N.P. Sedimentary characteristics of dust storms: II Visibility and dust concentration. *Am. J. Sci.* **1957**, *255*, 104–114.
19. Zobeck, T.M.; Van Pelt, R.S. Wind-induced generation and transport mechanics on a bare agricultural field. *J. Hazard. Mater.* **2006**, *132* (1), 26–38.
20. Van Pelt, R.S. Use of anthropogenic radioisotopes to estimate rates of soil redistribution by wind I: ^{137}Cs . *Aeolian Res.* **2013**, *9*, 89–102.
21. Saranlı, U.; Buehler, M.; Koditscheck, D. RHex: A simple and highly mobile robot. *Int. J. Robot Res.* **2009**, *20* (7), 616–631.

Wind Erosion: Global Hot Spots

Andrew Warren

Department of Geography, University College London, London, U.K.

Abstract

“Natural” can be differentiated from “induced” erosion; this entry concentrates on the latter. Hot spots vary both in time and space; they come and go and their intensity varies from place to place. A hot spot is assumed to be where the rate of wind erosion is well above the mean, even though it may not have serious economic impact. Hence, the Dust Bowl of the Great Plains in the 1930s was probably a hot spot, yet, agriculture is prosperous there. There and elsewhere, as in the West African Sahel, the Mallee of south-eastern Australia, a great deal of soil may be lost without much impact on yield, until the last few centimeters of soil are left.

NATURAL EROSION

The Sahara, the Central Asian Deserts, and the Chinese deserts are the sites of very high rates of “natural” erosion. Much more sediment leaves the Sahara in wind than in water. Contemporary outputs from the Sahara are of the order of 10^9 tons per year.^[1] Dry Australia, the southwestern United States, and southern South America are less important but still significant natural hot spots. The dust reaches far into the oceans and beyond, as from northern African to the United States and from Australia to New Zealand (Fig. 1). Marine and land sediments (Loess and stabilized sand dunes) show that the hot spots were hotter in the Pleistocene, and that they got hotter as it progressed. In the Central Sahara, Egypt, western Argentina, eastern Iran, and parts of western China, great grooves were cut at those times by wind erosion, some in hard rock. There were also more hot spots, as in parts of Northern Europe and Central North America. Ice core data from Greenland, the Tibetan Plateau, and the Antarctic corroborate the picture. The contemporary global source areas are all very dry. Explanation of the temporal change is more disputed.^[2]

Seen at a finer scale, most “natural” erosion in these deserts occurs in soft sediments, particularly those in ancient, dry lake basins. The northern Chad Depression in the central Sahara and the Lake Eyre basin in Central Australia are major natural hot spots.^[3] In the late Pleistocene, the bare outwash plains around glaciers were the sources of Loess and sand that formed dunes (mostly stabilized) in northern Europe, Asia, and northern Canada. People have little influence on rates of wind erosion in deserts. There are exceptions. Off-road vehicles create a minor hot spot in the Mojave of southern California; and wars and training for wars have created other hot spots in North Africa during the 1940s and during the Gulf War.^[2]

INDUCED EROSION

Identifying global hot spots of induced wind erosion is more problematic. Measurements of emissions are very few indeed, measurements of rates of erosion are even fewer, and measurements of economic impact are the rarest of all. This leaves some results from modeling, historical accounts built on disparate evidence, and informed guesses. Modeling and measurement suggest short-term rates of the order of $30\text{--}60 \text{ t ha}^{-1} \text{ yr}^{-1}$ on dry High Plains sites, depending on winds, rainfall, and land use. Caesium-based measurements in the Sahel of West Africa, on sandy soils under fallow cultivation cycling, and with mean annual rainfalls of about 550 mm give rates of up to $40 \text{ t ha}^{-1} \text{ yr}^{-1}$ over a 30-year period.^[4]

Some historical accounts are well documented and compelling. The Dust Bowl of the American Great Plains in the 1930s was an undisputed hot spot.^[5] Dust storms were more frequent and violent and many farmers went out of business, though more from drought than erosion. There were renewed hot spots here in the 1950s, 1960s, and 1970s, on the evidence of agricultural surveys and dust emissions. Probably more serious, though less well documented, at least in English, were the hot spots in the Soviet Union, specially northern Kazakhstan and neighboring parts of Russia, at about the same time^[6] (although the earliest recorded paper in Russian on wind erosion control was written in 1768). More surprising, even to some experts, are histories of wind erosion in Western Europe. An even earlier paper on wind erosion, titled “A curious and exact relation of a sand cloud which hath lately overwhelmed a great tract of land in the county of Suffolk” appeared in the *Philosophical Transactions of the Royal Society of London* in 1669. In Sweden, no less a person than the great naturalist Linnaeus recorded destitution because of wind erosion in Skania in the 18th century. There is similar evidence in northern Germany, The Netherlands,

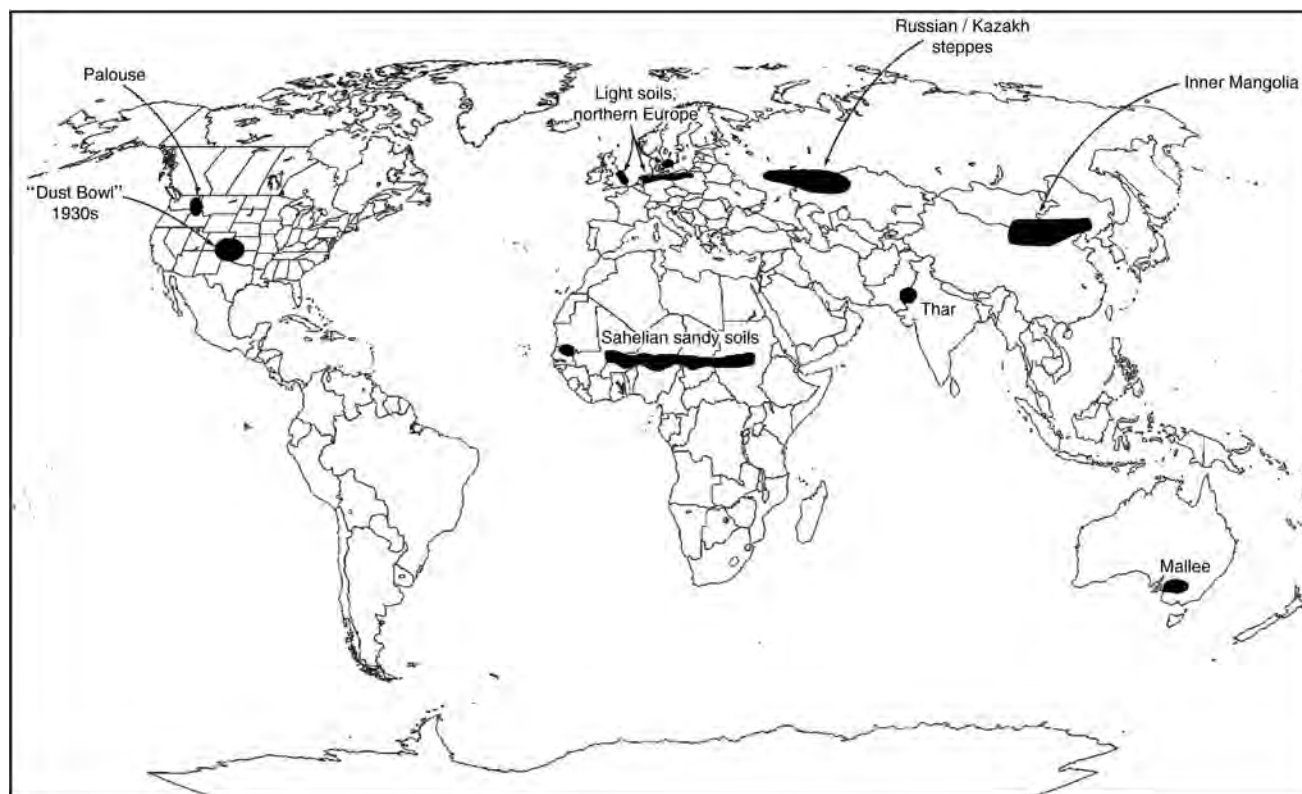


Fig. 1 Global hot spots for wind erosion.

Denmark, Poland, and Hungary. Archaeology shows that there were hot spots as far back as the Neolithic, when ancient farms were buried in sand in southeastern England and The Netherlands. In all these areas, it was and is sandy soils, inherited from glacial outwash, that are the most susceptible. The period of desiccation in the Sahel of West Africa is said to have been a hot spot for wind erosion. Some evidence lies in dust-storm data^[1] and, as in the Great Plains of the 1930s, there was undoubtedly great hardship. However, here too, the exact role of wind erosion is debatable. Much of the dust may have come from quite small perimeters round dust-measuring stations, and the hardship was to do more with drought than erosion.

Informed guesses are a more dubious source. The literature is no great help, even with the wind erosion bibliography (listing works till 1995) and its 2700 odd entries.^[7] It is undoubtedly biased toward work in English and toward areas where funds have been more available and to places where there has been a strong scientific tradition. For example, research in the United States, and the Great Plains in particular, where massive funds for research were released and maintained after the 1930s, overwhelms the bibliography, yet the aforementioned historical evidence suggests that the problem may have been more serious in the Union of Soviet Socialist Republics. Indeed, Petrov's bibliography of relevant work in Russian (1768–1950) has 852 entries. Germany, with its long scientific tradition, apparently dominates research in Europe, yet it is known that the

problem is as serious in neighboring countries. Southeastern Australia is also, by this evidence, a hot spot,^[8] yet in terms of damage to cropland, the Sahel of West Africa is probably more of one (the Sahel is only belatedly receiving scientific attention). Using the literature, Crosson^[9] estimated that the Sahel was a major wind erosion hot spot, yet there are few data on which to base this conclusion.^[4]

Only one systematic attempt at informed guessing on a worldwide scale has been made. This is GLASOD (The *GL*lobal *AS*essment of *SO*il *D*egradation). It is most accessible in the *World Atlas of Desertification*, which is in its second edition.^[10] The assessment was for “polygons” of the order of size of southeast England, the province of Skania in Sweden, or a quarter of an average U.S. State. Experts who knew each polygon were asked a set of carefully constructed questions about the extent and severity of soil degradation, including wind erosion. The result is the best guess we have of the severity of wind erosion globally. But it has serious flaws: in Africa and many other parts of the world, there were no measurements at the time of the questionnaire against which to compare the guesses; the distinction between induced and natural erosion is unclear; some experts have interpreted the questions differently from others; the identifications are generally of potential rather than actual erosion; and the polygons are huge.

The expert opinion used in GLASOD is based on our scientific knowledge about wind erosion. If, as a simple framework, we take the factors in the famous Wind Erosion

Equation of Woodruff and Siddoway, we can follow it in predicting the hot spots.^[11] Three factors are almost equally important. First is the climatic (C) factor. Most induced wind erosion occurs in areas with semiarid climates, as corroborated by maps of the C factor in the Sahel of West Africa and in the United States. This is also true in the Union of Soviet Socialist Republics, Australia, and north-western Europe. However, wind erosion can and does take place in much wetter climates. In Wales and in the Pennine Hills of England, peat is blown away after fires and in long dry spells. Erosion occurs in parts of England, where the mean annual rainfall is of the order of 1500 mm. At roughly the same mean annual rainfall, there is also significant wind erosion on the coastal plain of South Carolina. Wind erosion is reported in the Austrian Tyrol, where rainfalls approach 2000 mm. Even in Iceland, where evapotranspiration is very low, there is some serious wind erosion. The C factor is also important in the temporal dimension: when there are dry years, wind erosion hot spots appear, as in the Dust Bowl and the Sahel.

The K, or soil, factor in the Wind Erosion Equation is the second important control, for wind erosion rarely affects fine-texture soils, although a few may be "pelletized" into aggregates of sand size, which may then be moved by the wind. Areas with agriculture on sandy soils are, worldwide, the common locations of hot spots, because they lack binding agents for aggregation. Peats which also may be loose are also eroded in Western Europe and Florida, when they are drained, cleared, and cultivated. Light Loess soils, as in the Palouse of Washington State, parts of the Great Plains and the Russian steppes, are other hot spots.

The last crucial factor is vegetation cover, and this is the most susceptible to management. The plowing or harrowing of a field can raise considerable quantities of dust,^[12] but its clearance of vegetation in dry and windy seasons is the more generally damaging. The reasons why farmers clear susceptible fields are exceedingly complex. Ignorance may sometimes produce a minute hot spot, but it is an inexperienced farmer indeed who does not realize that erosion follows clearance. In the Sahelian hot spot, for example, the laying of millet (*Pennisetum glaucum*) stalks to counter erosion is a standard practice. Economic forces produce the main hot spots for induced erosion. In Europe, at least, the short-term economic impact of wind erosion does not provide a strong enough incentive to persuade farmers to manage it. Worster believed that the huge hot spot in the U.S. Dust Bowl was mainly due to farmers being driven to cultivate by other economic forces, such as the

need to pay off loans from banks and from machinery salesmen. The massive Soviet hot spots are said to have been the result of yet another kind of economic force: centralized and ill-informed attempts to reach U.S. standards of economic output. In our study of a village in Niger, economics takes yet another guise: wind erosion is the outcome of a gamble by those farmers who can afford to lose a crop, because they have other economic options.^[13]

REFERENCES

1. Middleton, N.J. Climatic controls on the frequency, magnitude and distribution of dust storms: Examples from India/Pakistan, Mauritania and Mongolia. In *Palaeoclimatology and Palaeometeorology: Modern and Past Processes of Global Atmospheric Transport*; Leinen, M., Sarnthein, M., Eds.; Kluwer: Dordrecht, 1989; 97–132.
2. Livingstone, I.; Warren, A. *Aeolian Geomorphology: An Introduction*; Addison-Wesley Longman: Harlow, 1996.
3. McTainsh, G.H. Dust processes in Australia and West Africa: A comparison. *Search* **1985**, *16* (3–4), 104–106.
4. Chappell, A.; Warren, A.; Oliver, M.A.; Charlton, M. The utility of ¹³⁷Cs for measuring soil redistribution rates in South-West Niger. *Geoderma* **1998**, *81* (3–4), 313–338.
5. Worster, D. *Dust Bowl: The Southern High Plains in the 1930s*; Oxford University Press: Oxford, 1979.
6. Alayev, E.B.; Badenkov, Y.P.; Karavaeva, N.A. The Russian plain. In *The Earth as Transformed by Human Action*; Turner, B.L. II, Clark, B.K., Bob, W., Richards, J.F., Matthews, J.T., Meyer, W.B., Eds.; Cambridge University Press: Cambridge, 1990; 453–560.
7. Wind Erosion Bibliography, <http://www.csrl.ars.usda.gov/wewc/apm.htm>.
8. McTainsh, G.H.; Boughton, W.C., Eds. *Land Degradation Processes in Australia*; Longman Cheshire: Melbourne, 1993.
9. Crosson, P.R. Will erosion threaten agricultural productivity? *Environment* **1997**, *39* (8), 4–31.
10. United Nations Environment Programme (UNEP). *World Atlas of Desertification*, 2nd Ed.; Arnold: London, 1997.
11. Woodruff, N.P.; Siddoway, F.H. A wind erosion equation. *P. Soil Sci. Soc. Am.* **1965**, *29* (5), 602–608.
12. Lee, J.A.; Tchakerian, V.T. Magnitude and frequency of blowing dust on the southern high plains of the United States, 1947–1989. *Ann. Assoc. Am. Geogr.* **1995**, *85* (4), 684–693.
13. Warren, A.B.; Simpon, P.J.; Osbahr, H. Soil erosion in the sahel of West Africa: A review of research issues and an assessment of new approaches. *Global Environ. Chang.* **2000**, *11* (1), 79–95.

Wind Erosion: Micrometeorology

Giles F.S. Wiggs

Department of Geography, University of Sheffield, Sheffield, U.K.

Abstract

Wind erosion of soils and sediments can be considered at a number of temporal scales ranging from sub-second to annual or greater. Agricultural scientists often refer to amounts of erosion in terms of tonnes per hectare per year or equivalent, but, while such terms are useful for comparison between fields, they mask many of the dynamic processes operating on the soil at much smaller spatial and temporal scales. In order to fully appreciate wind erosion, it is important to have an understanding of the smaller-scale processes that are operating at timescales ranging from seconds to minutes or hours. It is processes operating at this microscale that control the local erosivity (erosive power) of the atmosphere and can have an impact on the erodibility (susceptibility to erosion) of soils. Such meteorological factors include wind shear stress, wind turbulence, thermal stability, and humidity.

VELOCITY PROFILE, WIND SHEAR STRESS, AND SURFACE ROUGHNESS

As wind moves over a soil surface, it is retarded at its base by friction from the roughness of that surface. This creates a velocity profile with a region very close to the surface of zero velocity and wind high in the profile (or boundary layer) unaffected by the surface friction (called free stream velocity). The depth of this zero-velocity layer (termed z_0) is called the aerodynamic roughness length. It is an important parameter because it partly controls the gradient of the velocity profile, which is proportional to the shear stress (or force) of the wind on the surface. When wind shear stress at the surface reaches a critical level, wind erosion will occur.^[1]

Under normal atmospheric conditions, a wind velocity profile plots as a straight line on a logarithmic chart, the gradient of which is proportional to the surface shear stress. It is simpler to measure velocity profiles than it is to measure surface shear stresses, and so the gradient of a velocity profile (equivalent to shear velocity, u_*) is often used as a surrogate for surface shear stress.

The relationship between surface shear stress (τ_0) and shear velocity (u_*) is given by the following formula:

$$u_* = \sqrt{\tau_0 / \rho} \quad (1)$$

where ρ is air density.

From measurements of a simple mean velocity profile, an assessment can therefore be made of the shear stress imparted by the wind to the surface. High shear stresses are associated with high mean wind speeds, but many other factors also have an influence.

CHANGES IN SURFACE ROUGHNESS

One of the most important controls on shear stress is the roughness of the surface and its effect on the value of aerodynamic roughness, z_0 . Any change in the value of z_0 has a subsequent effect on the velocity profile gradient because of the alteration in the depth of the zero-velocity region. With all other factors remaining the same, an increase in z_0 will lead to an increase in u_* (and hence surface shear stress) and vice versa (Fig. 1).

The wind will respond to surface roughness at a variety of spatial scales. At a small scale, the individual soil clods create a drag on the airflow, as do ploughed furrows or crops. At a larger scale, shelterbelts, fences, and hummocky terrain also affect the velocity profile (Table 1). As the wind blows across a single field, it will respond to these changing surface roughnesses and so too will the shear velocity, surface shear stress, and the ability of the wind to entrain and transport soil particles. Rough surfaces should therefore be more susceptible to wind erosion than smooth surfaces. However, the relationship is complicated by the fact that increases in surface shear stress caused by a larger value of z_0 may not necessarily lead to increases in sediment entrainment and transport. This is because any increase in surface roughness may result in more grains lying within the depth of z_0 and hence protected from erosion in a layer of zero-velocity flow. This is the reason why fields ploughed perpendicular to the mean wind direction can often be stable and resistant to wind erosion, as the aerodynamic roughness height caused by the furrows allows the protection of fine surface soil. Typical values of z_0 for different surfaces are shown in Table 1.

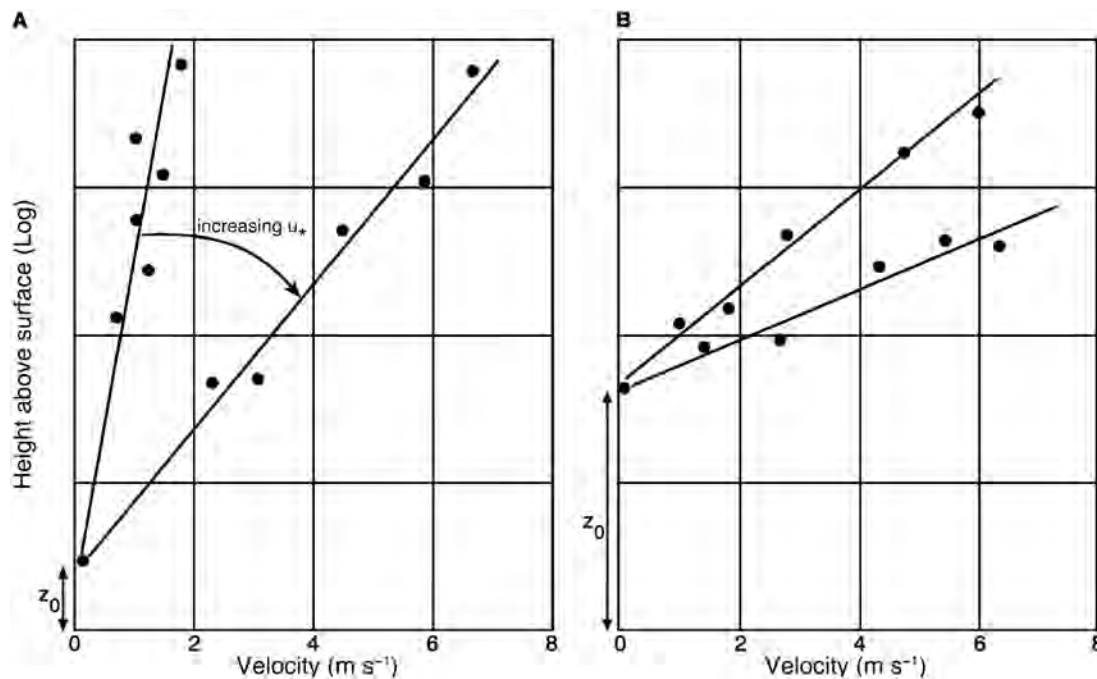


Fig. 1 Velocity profiles showing a focus at height z_0 , the aerodynamic roughness length. The aerodynamic roughness in (A) is less than in (B); hence, shear velocities (u_*) in (B) are greater.

Source: From Wiggs.^[1]

ROLE OF TURBULENCE

The semilogarithmic wind velocity profile (Fig. 1) only occurs under mean wind conditions, averaged over time periods ranging from perhaps 30 seconds or greater. However, under most natural conditions, wind in the environment is turbulent and consists of high-velocity gusts and low-velocity troughs. For this reason, velocity profiles measured over timescales of the order of 1 second do not display a semilogarithmic relationship with height.

In a turbulent velocity profile, momentum transfer through the profile is achieved by turbulent eddies of varying sizes. These operate in a cascade from eddies as large as the boundary layer to viscous energy dissipation at the

surface. These eddies move between different layers of fluid and impart or absorb energy depending on whether the eddy is faster or slower moving than the surrounding air.^[3] This type of momentum transfer is very efficient and results in high mean shear stresses being imparted to a soil surface.

Research^[3,4] suggests that sediment entrainment and transport can be greatly influenced by peak values in instantaneous shear stress provided by these turbulent flow structures (Fig. 2).

Three types of important turbulent structure have been identified as follows: 1) eddies of low momentum fluid at the surface elevated into faster air above (“burst” events); 2) high momentum fluid from high in the velocity profile impacting the surface (“sweeps”); and 3) flow-parallel longitudinal vortices (“streaks”).

Each of these flow structures may have periodicities of around 20 Hz and is considered to be critical in wind erosion because they may provide instantaneous peaks in shear stress which might exceed the threshold of grain entrainment, even where the mean wind velocity is just below that threshold. In these circumstances, turbulent flow structures may provide high instantaneous shear stresses which could entrain a few grains into saltation. These grains may then be kept in transport by the mean wind velocity (as the energy required for sediment transport is less than that required for entrainment) and, on impacting the surface, splash other grains into transport. In this way, a few saltating grains entrained by turbulent

Table 1 Typical aerodynamic roughness (z_0) values for different surfaces.

Surface	z_0 (m)
Sand	0.0003
Rippled sand	0.003
Soil	0.001–0.01
Grass	0.003–0.1
Agricultural crops	0.04–0.2
Forests	1.0–6.0

Source: Adapted from Oke.^[2]

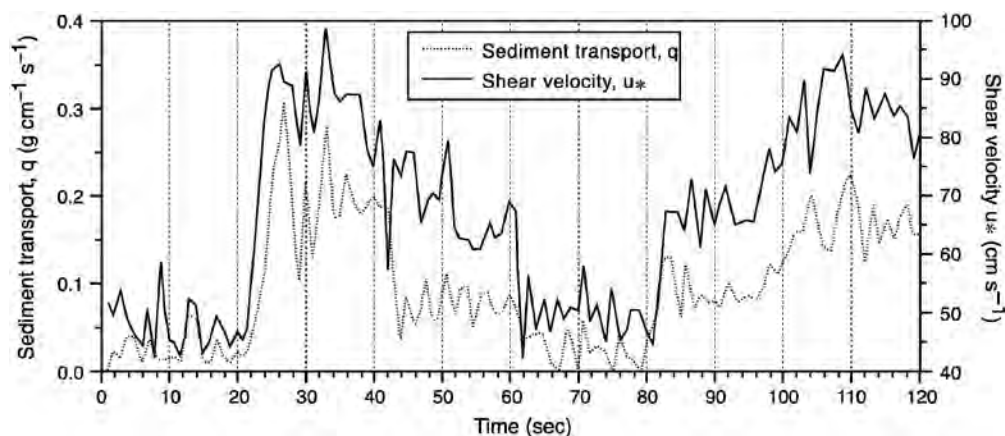


Fig. 2 The response of sediment transport (q) to fluctuations in instantaneous shear velocity (u_*).

Source: From Butterfield.^[4]

motion may provide a downwind cascade of entrainment and result in significant wind erosion.^[1]

THERMAL STABILITY

The logarithmic relationships shown in Fig. 1 only arise in atmospheric conditions with neutral buoyancy. Such conditions are found with cloudy skies (which reduce radiative heating) and strong winds (which promote atmospheric mixing and prevent temperature stratification).^[2] On clear and sunny days (especially in arid or semiarid areas), strong radiative heating may result in thermal instability (with a steep temperature gradient), which increases buoyancy effects and vertically stretches turbulent eddies. This results in a reduction of the wind gradient (becoming more vertical with height, in Fig. 1) with atmospheric turbulence driven by buoyancy effects rather than mechanical forces. Conversely, atmospheric stability (often occurring at night with radiative cooling of the surface) tends to squeeze turbulent eddies vertically resulting in a strong wind gradient (becoming more horizontal with height in Fig. 1) with little vertical mixing.

Both atmospheric stability and instability therefore have an impact on the structure of the velocity profile and hence on the shear stresses evident at the surface. The impact of thermal stability or instability on wind erosion is a function of the role of turbulent eddies. Atmospheric instability is associated with greater vertical mixing of air, and so fast-moving upper air is commonly brought closer to the surface resulting in an increase in shear stress. In contrast, a stable atmosphere is associated with relatively low surface shear stresses because turbulent mixing is damped down.^[5]

Sediment transport and wind erosion are therefore more likely to be evident where the atmosphere displays thermal instability (rather than stability) because buoyancy forces result in strong vertical mixing and increased turbulence. However, atmospheric instability is also associated with low mean wind speeds, and such conditions are unlikely to result in significant erosion. Wind erosion is more closely associated with neutral atmospheric buoyancy where a reasonable amount of turbulent mixing is combined with high mean wind

speeds. As noted, such conditions are generally associated with overcast skies and high winds during the daytime.

CHANGES IN HUMIDITY

Humidity refers to the water vapor content of the atmosphere, and it is an important variable in wind erosion because of the two-way transfer of water vapor between the soil surface and the near-surface atmosphere. This transfer has large implications for the erodibility of a soil.

Grain entrainment is a function of the ratio of resisting forces to dynamic forces.^[1] While grain size is the most important of the resisting forces, considerable resistance to movement can be provided by moisture in the upper layers of the soil. The exact physical nature of the relationship between grain entrainment threshold and moisture content has yet to be quantitatively defined, but there is a theoretical basis for suggesting that the threshold rises as a function of surface tension associated with pore moisture.^[6] Humidity has an important role to play here because an unsaturated air mass (low humidity) will have a significant drying effect on a soil and thus increase its erodibility (lowering the threshold of grain entrainment), while a saturated air mass will have the opposing effect. Research has suggested that the threshold of grain entrainment might be very sensitive to changes in moisture content below about 4–8%.^[6] Fairly small changes in atmospheric humidity may therefore have a critical impact on grain entrainment and wind erosion.

FUTURE DIRECTIONS FOR MICROMETEOROLOGICAL RESEARCH

A major challenge in wind erosion research is to successfully link the differing scales of investigation and applied activity. It is clear that an understanding of erosion processes at the microscale is necessary to successfully comprehend the wind erosion system, and many advances have been made toward this end. However, the application of this understanding is often at the field scale or larger, and it is in the transference of

scales where failure has occurred in the past. Process measurements operating over timescales of seconds or hours cannot easily be linked to changes in field erosion which may occur over timescales of months to years. As scales of interest increase, the reliability and appropriateness of small-scale process work on wind erosion decrease. It has been argued that, where appropriate, small-scale process measurements should become checkpoints for model calibration.^[7] In this respect, micrometeorology is likely to become increasingly important in soil erosion models for the prediction and assessment of wind erosion.

REFERENCES

1. Wiggs, G. Sediment mobilisation by the wind. In *Arid Zone Geomorphology*; Thomas, D.S.G., Ed.; Wiley: London, 1997; 351–372.
2. Oke, T.R. *Boundary Layer Climates*, 2nd Ed.; Routledge: London, 1987.
3. Clifford, N.J.; French, J.R.; Hardisty, J. *Turbulence: Perspectives on Sediment Transport*; Wiley: Chichester, 1993.
4. Butterfield, G.R. Sand transport response to fluctuating wind velocity. In *Turbulence: Perspectives on Sediment Transport*; Clifford, N.J., French, J.R., Hardisty, J., Eds.; Wiley: Chichester, 1993; 305–335.
5. Frank, A.; Kocurek, G. Effects of atmospheric conditions on wind profiles and aeolian sand transport with an example from white sands national monument. *Earth Surf. Processes*. **1994**, *19*, 735–745.
6. Sherman, D.J. Discussion: Evaluation of aeolian sand transport equations using inter-tidal zone measurements. *Sedimentology* **1990**, *37*, 385–392.
7. Sherman, D.J. Problems of scale in the modelling and interpretation of coastal dunes. *Mar. Geol.* **1995**, *124* (1–4), 339–349.

Wind Erosion: Modeling

Lawrence J. Hagen

Wind Erosion Research Unit, U.S. Department of Agriculture—Center for Grain and Animal Health Research (USDA-ARS-CGAHR), Manhattan, Kansas, U.S.A.

Abstract

Models of wind erosion are used to investigate fundamental processes and guide resource management. Many models are similar in that temporal variables control soil wind erodibility, erosion begins when friction velocity exceeds a threshold, and transport capacity for saltation/creep is proportional to the cube of friction velocity. The conservation of mass equations that incorporate erosion processes has been developed to calculate soil loss, transport, and deposition. Components of the Wind Erosion Prediction System model are illustrated as an example.

INTRODUCTION

Numerous models of wind erosion exist. In regional- and global-scale models, dust generation is often coupled to diffusion, advection, and deposition models.^[1] Their applications include the prediction of dust impacts on past climates,^[2] military operations,^[3] health problems,^[4] and existing weather, particularly in East Asia.^[5] Model results also demonstrate that dust is an important contributor to climate forcing.^[6] There are numerous challenges in modifying field-scale models for use on large areas.^[1,7–9]

Field-scale erosion models are used to predict soil loss, plan conservation systems, and assess off-site impacts of wind erosion.^[10,11] Many models exist to simulate specific effects such as sand ripples.^[12] Models also serve to increase and synthesize our knowledge about wind erosion of soil.

WIND EROSION PROCESSES

The major factors that control soil wind erodibility are mainly temporal (Table 1), and their range of variation depends on weather, soil management, and soil intrinsic properties such as texture.

To begin erosion, friction velocity must overcome the forces holding particles on the surface. Threshold friction velocities for dry, monodisperse particles have been measured^[13] and theoretically analyzed (Fig. 1).^[14] For mixtures of sizes, thresholds may be slightly lower due to particles perched on a surface or higher when sheltered by immobile aggregates.^[15] Both surface wetness and high humidity also tend to increase threshold velocities.^[16,17] Saltation is not needed to entrain 0.07–0.10-mm-diameter particles. Abrasion creates mobile aggregates, and abrasion susceptibility of clods/crust can be characterized by an abrasion coefficient. It can be directly measured or estimated from the crushing energy.^[18,19] Intermittent field

erosion in the short term is often caused by wind gustiness,^[20] but longer term variations may be caused by surface armoring during erosion events.^[21]

Modes of soil transport include creep (rolling, 0.8–2.0-mm diameter), saltation (hopping, 0.1–0.8 mm), and suspension (<0.1 mm). The creep/saltation transport capacity q ($\text{kg m}^{-1} \text{s}^{-1}$) is proportional to the cube of friction velocity u_* (m s^{-1}). Many formulas for q have been proposed,^[15] and one formula is as follows

$$q = \frac{C\rho_a}{g} u_*^2 (u_* - u_{*t}) \quad (1)$$

where C is a coefficient, ρ_a (kg m^{-3}) is air density, g (m s^{-2}) is acceleration of gravity, and u_{*t} (m s^{-1}) is threshold friction velocity. The transport capacity for suspended particles is large, and their downwind discharge can be

Table 1 Factors that control wind erodibility of a dry, bare soil.

Soil state	Factors
Aggregated (tilled) surface	Aggregate size distribution by layer
	Dry stability of immobile aggregates by layer
	Breakage coefficient of mobile aggregates
	Rock volume fraction (>2 mm diameter) by layer
Crusted and/or consolidated surface	Random and oriented surface roughness
	Crust cover fraction
	Dry stability of crust/consolidated zone
	Thickness of consolidated zone
	Mass, cover, and breakage coefficient of loose soil on crust
	Other parameters same as aggregated surface

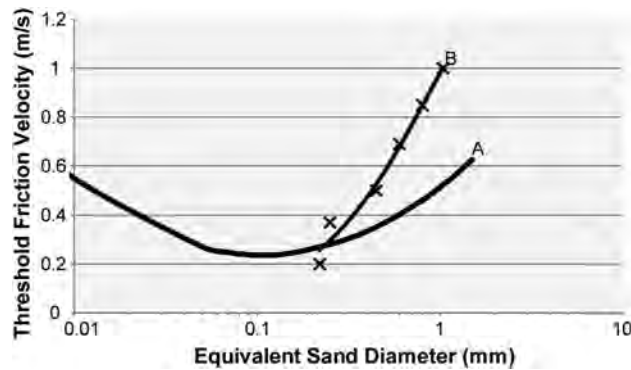


Fig. 1 Threshold friction velocities for monodisperse particles (A) and a particle mixture containing 15% immobile aggregates with maximum diameter of transported particles shown (B). **Source:** From Greeley & Iversen^[13] and Chepil.^[15]

several times that of saltation/creep as illustrated by measurements at dry Lake Owen.^[22]

Wind erosion on a field scale can be modeled using the conservation of mass equations in successive control volumes each a few meters in length (Fig. 2).^[23] Friction velocity above standing biomass is depleted to simulate wind friction velocity at the soil surface. Sources of saltation/creep discharge include entrainment of loose aggregates by wind drag and splash impacts, and abrasion from immobile clods and crust. Sinks for saltation/creep discharge include trapping by surface roughness, interception by standing biomass, and breakage to suspension-sized particles.

Similarly, sources for suspension discharge include entrainment of loose aggregates, abrasion from clods and crust, and breakage of saltation/creep aggregates. Sinks

for suspension include interception by standing biomass and deposition on downwind immobile surfaces. Emission, abrasion, and breakage are modeled separately because the interparticle-binding energy increases by roughly an order of magnitude between each process. Moreover, not all processes may be present in a given surface condition, but each can be measured separately using a wind tunnel.^[24,25] Analytic solutions for quasi-steady-state conservation of mass equations have been developed for saltation/creep and suspension discharge^[23] and the predictions validated using measured data from small fields.^[26–28]

Field-Scale Wind Erosion Models

The most widely used model has been the empirical Wind Erosion Equation (WEQ).^[29] The long-term annual soil loss per unit area (E) is given by the following formula

$$E = ICKLV \quad (2)$$

where the factors are soil wind erodibility (I), climate (C), surface roughness (K), field length (L), and vegetation (V).

A continuous, process-based model, the Wind Erosion Prediction System (WEPS), has been developed (Fig. 3) to replace WEQ.^[30] In WEPS, weather simulators drive five submodels that simulate surface conditions on a daily basis and erosion on a subhourly basis.

Additional models used to predict erosion include the Revised Wind Erosion Equation, Texas Tech Erosion Analysis Model, Wind Erosion on European Light Soils, Australian Land Erodibility Model, and Integrated Wind Erosion Modeling System.^[31–35]

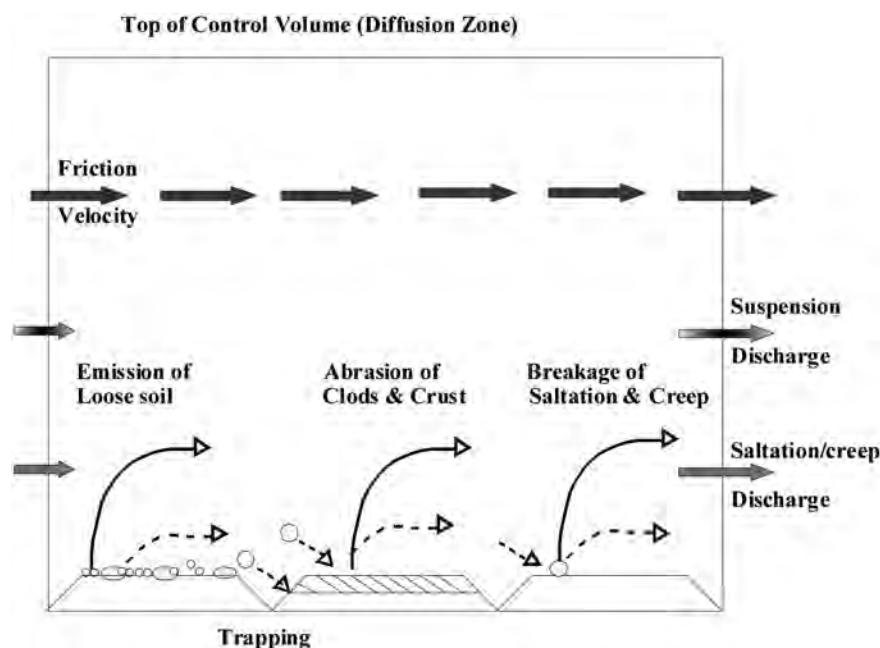


Fig. 2 Schematic representation of control volume illustrating major wind erosion processes on bare soil.

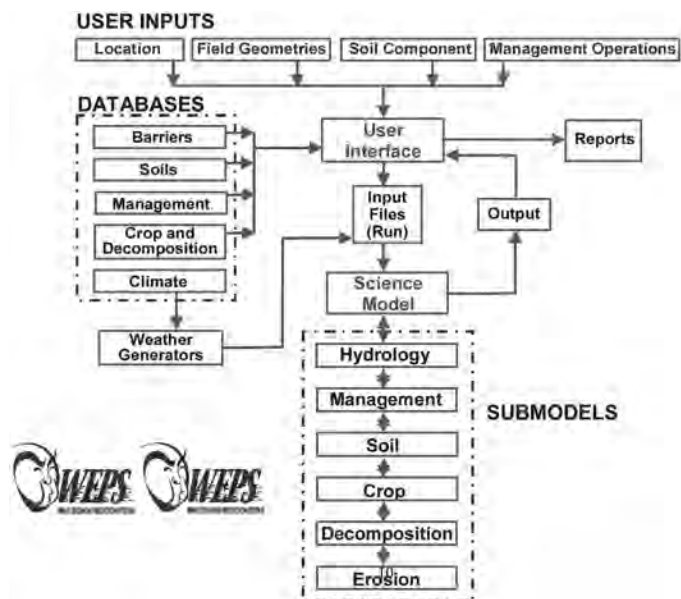


Fig. 3 Flow chart illustrating the components of WEPS field-scale wind erosion model.

ACKNOWLEDGMENT

This entry received contribution from U.S. Department of Agriculture, Agricultural Research Service in cooperation with Kansas Agricultural Experiment Station, and contribution (08-106J) from the Kansas Agricultural Experiment Station, Manhattan, Kansas.

REFERENCES

- Shao, Y. *Physics and Modeling of Wind Erosion*; Kluwer Academic Publishers: Dordrecht, 2000.
- Werner, M.; Tegen, I.; Harrison, S.P.; Kohfeld, K.E.; Prentice, I.C.; Balkanski, Y.; Rodhe, H.; Roelandt, C. Seasonal and interannual variability of the mineral dust cycle under present and glacial climate conditions. *J. Geophys. Res.* **2002**, *107* (D24), 4744–4763.
- U.S. Navy. NRL/Monterey Aerosol Page. Available at <http://www.nrlmry.navy.mil/aerosol> (accessed 2007).
- Draxler, R.R.; Gillette, D.A.; Kirkpatrick, J.S.; Heller, J. Estimating PM10 concentrations from dust storms in Iraq, Kuwait, and Saudi Arabia. *Atmos. Environ.* **2001**, *35*, 4315–4330.
- Zhou, C.H.; Gong, S.L.; Zhang, X.Y.; Wang, Y.Q.; Niu, T.; Liu, H.L.; Zhao, T.L.; Yang, Y.Q.; Hou, Q. Development and evaluation of an operation SDS forecasting system for East Asia: CAUACE/DUST. *Atmos. Chem. Phys. Discuss.* **2007**, *7*, 7987–8015.
- Tegen, I.; Lacis, A.A.; Fung, I. The influence on climate forcing of mineral aerosols from disturbed soils. *Nature* **1996**, *380*, 419–422.
- Raupach, M.R.; Lu, H. Representation of land-surface processes in Aeolian transport models. *Environ. Model. Softw.* **2004**, *19*, 93–112.
- Zobeck, T.M.; Parker, N.C.; Haskell, S.; Guoding, K. Scaling up from field to region for wind erosion prediction using a field-scale wind erosion model and GIS. *Agric. Ecosyst. Environ.* **2000**, *82* (1), 247–259.
- Feng, G.; Sharratt, B. Scaling from field to region for wind erosion prediction using WEPS and GIS. *J. Soil Water Conserv.* **2007**, *62* (5), 321–328.
- Hagen, L.J. A wind erosion prediction system to meet user needs. *J. Soil Water Conserv.* **1991**, *46* (2), 106–111.
- Wagner, L.E.; Hagen, L.J. Application of WEPS generated soil loss components to assess off-site impacts. In *Sustaining the Global Farm*, Proceedings of 10th International Soil Conservation Organization Conference, West Lafayette, May 24–29, 1999; Stott, D.E., Mohtar, R.H., Steinhardt, G.C., Eds.; Purdue University: West Lafayette, 2001; 935–939.
- Gill, T.E.; Warren, A.; Stout, J.E. *Bibliography of Aeolian Research*. Available at <http://www.csrl.ars.usda.gov/wewc/biblio/bar.htm>. 2007.
- Greeley, R.; Iversen, J.D. *Wind as a Geological Process on Earth, Mars, Venus, and Titan*; Cambridge University Press: New York, 1985.
- Shao, Y.; Lu, H. A simple expression for wind erosion threshold friction velocity. *J. Geophys. Res.* **2000**, *105* (D17), 22437–22443.
- Chepil, W.S. *Soil Conditions That Influence Wind Erosion*, U.S. Dept. Agric., ARS, Tech. Bull. No. 1185: Washington, 1985.
- Saleh, A.; Fryrear, D.W. Threshold wind velocities of wet soils as affected by wind blown sand. *Soil Sci.* **1995**, *160*, 304–309.
- Ravi, S.; Zobeck, T.M.; Over, T.M.; Okin, G.S.; D'Odorico, P. On the effect of moisture bonding forces in air-dry soils on threshold friction velocity of wind erosion. *Sedimentology* **2006**, *53*, 597–609.
- Hagen, L.J. Wind erosion mechanics: Abrasion of aggregated soil. *Trans. ASAE* **1991**, *34* (4), 831–837.
- Hagen, L.J.; Skidmore, E.L.; Saleh, A. Wind erosion: Prediction of aggregate abrasion coefficients. *Trans. ASAE* **1992**, *35* (6), 1847–1850.

20. Stout, J.E.; Zobeck, T.M. Intermittent saltation. *Sedimentology* **1997**, *44*, 959–970.
21. Hagen, L.J. Updating soil surface conditions during wind erosion events using the wind erosion prediction system (WEPS). *Trans. ASAE* **2008**, *51* (1), 129–137.
22. Gillette, D.A.; Fryrear, D.W.; Xiao, J.B.; Stockton, P.D.O.; Helm, P.J.; Gill, T.E.; Ley, T. Large-scale variability of wind erosion mass flux rates at Owens Lake. 1. Vertical profiles of horizontal mass fluxes of wind-eroded particles with diameter greater than 50 μm . *J. Geophys. Res.* **1997**, *102* (D22), 25977–25987.
23. Hagen, L.J.; Wagner, L.E.; Skidmore, E.L. Analytical solutions and sensitivity analyses for sediment transport in WEPS. *Trans. ASAE* **1999**, *42* (6), 1715–1721.
24. Hagen, L.J. Assessment of wind erosion parameters using wind tunnels. In *Sustaining the Global Farm*, Proceedings of 10th International Soil Conservation Organization Conference, West Lafayette, May 24–29, 1999; Stott, D.E., Mohtar, R.H., Steinhardt, G.C., Eds.; Purdue University, 2001; 742–746.
25. Mirzamostafa, N.; Hagen, L.J.; Stone, L.R.; Skidmore, E.L. Soil and aggregate texture effects on suspension components from wind erosion. *Soil Sci. Soc. Am. J.* **1998**, *62*, 1351–1361.
26. Hagen, L.J. Evaluation of the wind erosion prediction system (WEPS) erosion submodel on cropland fields. *Environ. Modell. Softw.* **2004**, *2*, 171–176.
27. Funk, R.; Skidmore, E.L.; Hagen, L.J. Comparison of wind erosion measurements in Germany with simulated losses by WEPS. *Environ. Modell. Softw.* **2004**, *2*, 177–183.
28. Feng, G.; Sharratt, B. Validation of WEPS for soil and PM10 loss from agricultural fields within the Columbia Plateau of the United States. *Earth Surf. Processes* **2007**, *32*, 743–753.
29. Woodruff, N.P.; Siddoway, F.H. A wind erosion equation. *Soil Sci. Soc. Am. J.* **1965**, *29* (5), 602–608.
30. Tatarko, J.T.; Wagner, L.E.; et al. *WEPS Users Manual (Draft)*; 2007.
31. Fryrear, D.W.; Bilbro, J.D.; Saleh, A. RWEQ: Improved wind erosion technology. *J. Soil Water Conserv.* **2000**, *55* (2), 183–189.
32. Gregory, J.M.; Wilson, G.R.; Singh, U.B.; Darwish, M.M. TEAM: Integrated, process-based wind-erosion model. *Environ. Modell. Softw.* **1994**, *19* (2), 205–211.
33. Warren, A. *Wind Erosion on European Light Soils (WEELS)*. Final Report to the European Union Commission; 2000.
34. Webb, N.P.; McGowan, H.A.; Phinn, S.R.; McTainsh, G.H. AUSLEM (Australian Land Erodibility Model): A tool for identifying wind erosion hazard in Australia. *Geomorphology* **2006**, *78* (3–4), 179–200.
35. Lu, H.; Shao, Y. Toward quantitative prediction of dust storms: An integrated wind erosion modeling system and its applications. *Environ. Modell. Softw.* **2001**, *16*, 233–249.

Wind Erosion: Principles

Larry D. Stetler

Department of Geology and Geological Engineering, South Dakota School of Mines and Technology, Rapid City, South Dakota, U.S.A.

Abstract

Erosion of sediment and soil by wind arises when the kinetic energy in the passing airstream exceeds the energy employed in adhering the grain to the surface. Initial grain motion is achieved at this instant, and the grain will be transported in saltation as a series of short hops along the surface. Each time the grain impacts the surface, additional grains are released into the flow until at some distance the carrying capacity of the wind is reached. From this point on, the mass of sediment in transport remains relatively constant, i.e., it is self-balanced. However, mass is lost from the process primarily through the deposition of heavy particles behind surface roughness elements. In addition, suspendable particles of fine dust are released vertically through the saltation layer. Many additional factors, such as climate and surface conditions, complicate the simple mechanics involved.

EROSION PROCESSES

Erosion of fine-grained sediment and soil particles by wind is a natural geologic process occurring worldwide on a nearly continuous basis. Throughout the geological record, evidence of wind processes eroding, transporting, and depositing sediment exists primarily as vast accumulations of eolian (wind-blown) sand—both modern sand dune fields and ancient eolian sandstone formations—and Loess (wind-blown silt). About 30% of the total land surface of the planet consists of semiarid and arid lands which give rise to most of the transported sediment and dust. Modern agriculture and industrial activities are also a significant contributor to transported sediment and dust due to induced or mechanical destabilization of the land surface.

Conceptually, the mobilization of loose sediment and soil particles occurs during wind storms or wind “events,” where the mean wind speed exceeds a critical threshold value based on some erodible fraction of the sediment. Individual wind events may be of short (less than one to a few hours) or long (multiple hours to days) duration. In terms of distance and total mass moved, a typical erosion event will transport the greatest mass a short distance and a significantly lesser mass a greater distance. The perception of moving sediment, however, may make this statement appear to be contradictory since there is, generally, an abundance of the smaller and lighter particles which are visually acute when airborne. Exceptional, or non-typical, wind events are able to transport fine particles across entire continents and oceans to distant lands where visual impacts to air quality may be readily observed.

Knowledge of particle entrainment and transport studies were greatly advanced through the work of Bagnold who used field observations in the deserts of Egypt and later wind tunnel studies to detail the mechanics of the erosional and transportation process. This work resulted in his well-known and widely used treatise *The Physics of Blown Sand and Desert Dunes*.^[1] He observed that once a particle begins to move it will be transported by one of three modes: creep, saltation, or suspension. The primary difference in these three transport modes is a relatively simple relation between the particle size and the wind force which, unfortunately, becomes a complex mechanical problem that is not fully understood.

WIND DYNAMICS

Wind is by nature a random process and as such, the internal structure of a natural wind contains random variations known as turbulence (except in possibly very stable conditions). Turbulence develops in natural flows as inertial forces, which provide forward momentum (motion), react to viscous forces, which impede motion due to fluid properties. This results in a flow that is strongly sheared.

The unit of measure for wind speed is the mean wind velocity, $\bar{U}$, defined as an average value for a continuous time series of instantaneous velocities (Fig. 1)

$$\bar{U} = \frac{\sum u_1 + u_2 + \dots + u_i}{T} \quad (1)$$

where $u_1 + u_2 + \dots + u_i$ are components of instantaneous velocity and T is the time interval over which velocity values are summed.

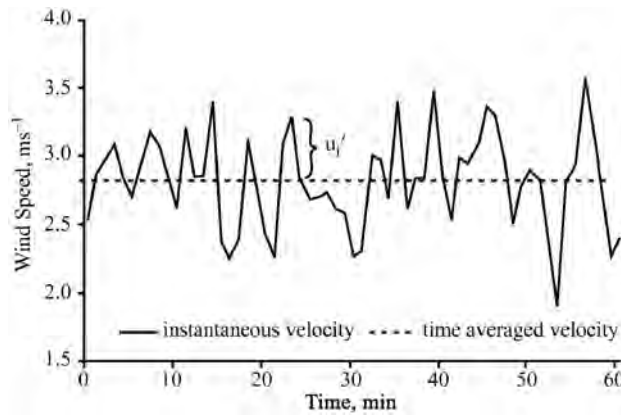


Fig. 1 Time series of 1-minute wind speeds illustrating the principles of determining mean wind speed, instantaneous, and fluctuating values.

Source: Unpublished data from author.

As shown in Fig. 1, at every instant a fluctuating component of velocity, u_i' exists defined as the difference between the instantaneous and the mean velocity values

$$u_i' = u_i - \bar{U} \quad (2)$$

It is these velocity fluctuations, whether positive or negative in regard to the mean velocity, that give rise to flow instability and thus turbulence. As such, turbulence is not a property of the fluid but of the flow of the fluid. Turbulent flows are, therefore, unique in their ability to mix momentum and kinetic energy and to erode and transport particles.

PARTICLE ENTRAINMENT

Bagnold described the instant of initial particle motion by a threshold fluid velocity which represents the wind speed at which the most susceptible particle becomes entrained. As a flow passes over a stationary particle, turbulent kinetic energy is imparted to the grain which builds a reservoir of stored potential energy.^[2] Further increases in this energy reservoir cause the grain to oscillate and at the critical point where potential energy exceeds the energy of adherence, the grain lifts, or more accurately is ejected in a near vertical trajectory away from the surface^[2,3] initiating saltation. For a single grain of diameter d , Bagnold showed that the threshold fluid velocity, v_t , required to initiate saltation motion is related to particle diameter by

$$v_t = 5.75A \sqrt{\frac{\rho_p - \rho_f}{\rho_f}} - gd \log \frac{z}{k} \quad (3)$$

where A is isrequired to initi with the value with the valuenitiate saltati, ρ_p is particle density, ρ_f is fluid

density, g is gravity force, z is the height for which threshold velocity is being calculated, and k is the surface roughness height equal to about 1/30 the surface grain diameter.

For the case of a naturally irregular surface, such as a field or a sand dune, fluid threshold is not defined by a single wind speed or particle size but by a continuum of wind speeds over a length of time where grains of various diameter become entrained almost simultaneously.^[4]

Once these grains are ejected into the overriding flow, they attain a high-energy ballistic trajectory that depends on particle size and wind velocity^[1,5,6] (Fig. 2). Heavier and larger particles will travel relatively close to the surface (A) contrasted to lighter and smaller particles that initially are ejected higher into the wind profile (B). At some distant point downstream, these grains (impactors) will impact on the sediment bed and either rebound upward again (C), eject new impactors (D), or lodge in the loose sediment. In either case, it is probable that upon impact sufficient kinetic energy will be transferred to one or several additional grains that are “splashed,” or dislodged a short distance (E) known as reptation,^[5,6] a low-energy process that includes a small number of grains traveling upstream. Reptation includes what Bagnold referred to as creep—grains that are too large to be entrained aerodynamically and move by rolling or sliding along the surface. Impact force is also a primary method leading to the disaggregation of larger particle agglomerates by breaking particle bonds or the particles themselves into smaller pieces, enabling aerodynamic transport.

Saltation, which is defined as movement in a series of short hops, includes particles in the high-energy population (A, B, C, and D). These particles are initially moved by wind speeds exceeding the fluid threshold and subsequently through surface impacts that eject new particles at a point known as the impact threshold. Bagnold suggested that the impact threshold involves the same mechanics included in Eq. 3 except $A=0.08$. In other words, after saltation is initiated, additional grains are placed in motion at the threshold impact velocity that is about 80% of the threshold fluid velocity. This infers that once motion has begun, further motion can be sustained at wind speeds less than those required for initial grain motion.

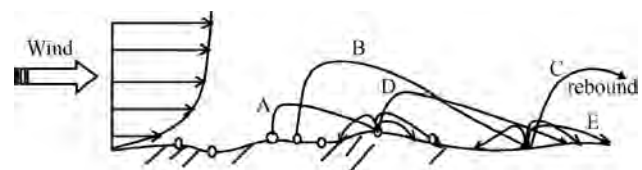


Fig. 2 Particle transport paths in response to the wind profile (at left). A, B, C, and D are high-energy impacting grains moving in saltation. E are low-energy reptating particles including those moving by creep.

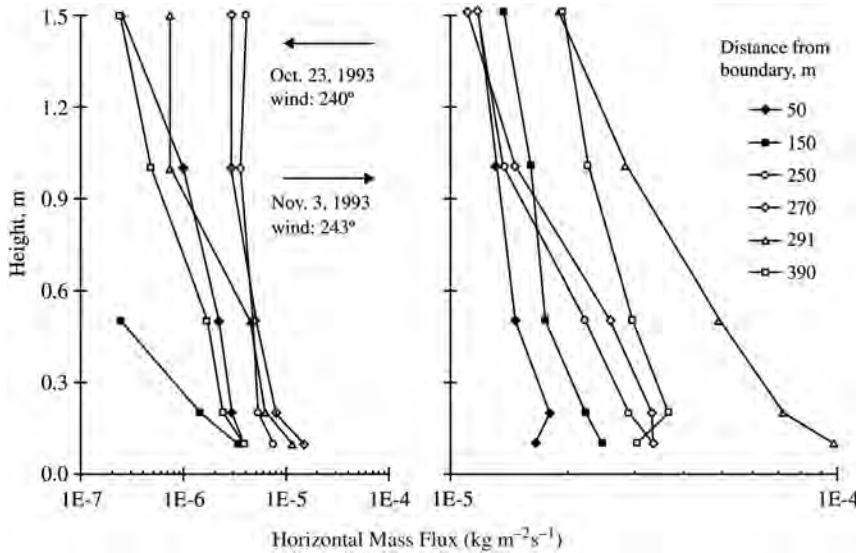


Fig. 3 Vertical and horizontal mass flux profiles for two wind events illustrating the decrease and increase of mass with height and downstream distance, respectively.
Source: Data from Stetler & Saxton.^[10]

The third transport mode, suspension, occurs in grains that are small enough to respond directly to turbulent fluctuations and thus are lofted above the surface saltation layer.

SELF-BALANCING CONCEPT

Energy required to transport particles comes from the extraction of turbulent momentum contained in the flow and a reduction in shearing forces near the surface.^[7,8] Thus, as the flow adjusts to the presence of particles, a self-balancing condition in mass flux develops downstream of the point of initial grain motion.^[9] In other words, the carrying capacity of the wind has been realized. This does not imply additional particles are not eroded, but that as larger grains become deposited behind surface roughness elements, the loss of mass is balanced by erosion of new grains from the surface, primarily by impact and abrasion forces.

The self-balancing concept for maximum horizontal mass flux, $f_{\max}$, has been analyzed by Stout,^[8] reducing to the following formula

$$f_x = f_{\max}(1 - e^{-\frac{x}{b}}) \quad (4)$$

where f_x is horizontal mass flux at downstream distance, x , from a non-eroding boundary and b is the length scale for stability in horizontal mass flux and is applied to height-specific mass flux measurements. In utility, Eq. 4 states that beginning from a non-eroding boundary at $x = 0$, f_x increases with x until $f_{\max}$ is attained. Downstream of this point, horizontal mass flux is relatively unchanged. Typical vertical and horizontal mass flux profiles from two wind events are shown in Fig. 3.^[10] Two components require discussion. First, vertical mass flux profiles, from 0.1 m to 1.5 m height, show mass decreases with height. This is

consistent with the discussion above, i.e., saltating particles having less mass loft higher into the flow. For any single profile, mass is the greatest near the surface and decreases vertically. Second, horizontal mass flux increases downstream until the maximum transport capacity is reached. In the figure, mass continues to increase downstream not yet having attained $f_{\max}$.

CONCLUSION

Complications to the abovementioned processes arrive due to numerous natural and human-induced conditions including soil type, particle size distribution and fraction of erodible particles, climate, and land use. In cultivated fields, surface roughness is a primary consideration complicating the entrainment process as well as the establishment of the saltation layer. Rough surface elements have the effect of reducing wind shear forces, thereby limiting erosion contrasted to a smoother surface that is favorable for a higher erosion rate.^[11] Surface roughness can be either from soil clods, vegetation, or a combination of both.^[12] Additionally, given all the factors discussed above, increases in soil moisture will cause an increase in the wind speed required to initiate grain motion.^[13]

The prediction of wind erosion is, therefore, an inexact science, even with the advent of sophisticated computer models. The numerous and complex relations between all the critical parameters imply a degree of uncertainty, even in the best of calculations.

REFERENCES

1. Bagnold, R.A. *The Physics of Blown Sand and Desert Dunes*; Methuen and Co., Ltd.: London, 1941; 265 pp.

2. Reeks, M.W.; Hall, D. Deposition and resuspension of gas-borne particles in recirculating turbulent flows. *J. Fluid. Eng.* **1988**, *110*, 165–171.
3. Greely, R.; Iverson, J.D. *Wind as a Geological Process on Earth, Mars, Venus and Titan*; Cambridge University Press: New York, 1985; 333 pp.
4. Nickling, W.G. The initiation of particle movement by wind. *Sedimentology* **1988**, *35*, 499–510.
5. Anderson, R.A.; Sorenson, M.; Willets, B.B. A review of recent progress in our understanding of Aeolian sediment transport. *Acta Mech. Suppl.* **1991**, *1*, 1–19.
6. Unger, J.E.; Haff, P.K. Steady state saltation in air. *Sedimentology* **1987**, *34*, 289–299.
7. Anderson, R.S.; Haff, P.K. Simulation of Eolian saltation. *Science* **1988**, *241*, 820–823.
8. Stout, J.E. Wind erosion within a simple field. *Trans ASAE* **1990**, *33* (5), 1597–1600.
9. Owen, P.R. Saltation of uniform grains in air. *J. Fluid. Mech.* **1964**, *20*, 225–242.
10. Stetler, L.D.; Saxton, K.E. Wind erosion, PM₁₀ emissions, and dry-land farming on the Columbia Plateau. *Earth Surf. Processes.* **1996**, *21*, 673–685.
11. Stout, J.E.; Zobeck, T.M. The Wolfforth field experiment: A wind erosion study. *Soil Sci.* **1996**, *161* (9), 616–632.
12. Horning, L.B.; Stetler, L.D.; Saxton, K.E. Surface residue and soil roughness for wind erosion protection. *Trans. ASAE* **1998**, *41* (4), 1061–1065.
13. Pye, K. *Aeolian Dust and Dust Deposits*; Academic Press: London, 1987; 334 pp.

Wind Erosion: Soil Quality and Productivity

John Leys

*Department of Land and Water Conservation, Center for Natural Resources,
Gunnedah, New South Wales, Australia*

Abstract

Wind erosion of agricultural land has negative effects on soil quality and soil productivity. The winnowing of the clay, silt, and organic fractions from the surface soil and its transport off-site during wind erosion result in nutrient decline and a reduction in water-holding capacity of the soil. Sandy soils are more prone to wind erosion and suffer the greatest impact of soil and nutrient loss. The loss of soil productivity through wind erosion impacts the subsequent crop production. The level of decline in crop production depends on the crop, soil, and climatic factors. The use of inorganic fertilizers to replace lost nutrients assists in maintaining crop production but does not fully compensate for eroded particles and nutrients. Minimizing erosion is the best way to maintain soil quality and productivity.

INTRODUCTION

Accelerated erosion on agricultural lands has adverse effects on soil quality and productivity through the removal of soil particles and nutrients. The majority of reviews on this topic have concentrated on the impact of water erosion on soil and crop productivity.^[1,2] However, there are a growing number of researches on the impact of wind erosion on soil quality and productivity, which is the focus of this entry.

Wind erosion reduces soil quality and production using a mechanism different from that of water erosion. Water erosion removes the soil en masse, while wind erosion winnows the finer/lighter particles from the surface, leaving the larger (generally inert) particles behind. As a result, wind erosion removes topsoil^[3] and reduces the soil clay and silt content^[4] and the organic matter.^[5] Wind erosion also has an impact on crop productivity. It sandblasts emerging crops,^[6] reshapes the land surface, thereby making it difficult to traverse with wide agricultural implements, buries or undermines infrastructure such as fences and roads, and buries adjacent land with sand drift. This results in limiting the drifted land's production in the short term.^[7] Off-site impacts will not be discussed here, although wind erosion also has considerable off-site impact—e.g., reduces visibility,^[8] deposits unwanted dust and off-farm associated contaminants, and raises airborne particulate levels,^[9] with particle sizes less than 10 μm (PM_{10}), which can have adverse health effects.^[10]

SOIL QUALITY

Soil quality is generally adversely affected by wind erosion via the removal of soil fines (clay, silt, and organic fractions). Soil texture changes are largely irreversible, unless

topsoil is imported to the site. The quantification of the changes in soil texture, as well as identifying the eroded fractions in the eroded sediments, indicates the magnitude of the decline in soil quality brought about by wind erosion. The descriptions of soil texture changes and the eroded sediments will be presented in the next two subsections.

Soil Texture Changes

The impact of erosion on soils has been measured over many years by using many different methods. Long-term analysis of soils exposed to wind erosion^[11] showed a decline in the fertility and particle size distribution (PSD) of the surface soil. Over a 36-year period, a 6.5% increase in the sand fraction of the top 0–10 cm has been reported for Midwestern United States.^[12] The comparison of the PSD of a soil that had been farmed for 30 years and had suffered periodic erosion with that of an adjacent soil under native vegetation^[13] revealed a loss of fines in the 10–100 μm fraction, as well as an increase in the coarse 350–1000 μm fraction (Fig. 1). Increases in the saltation fraction (>250 μm), and reduction in the 75–210 μm fraction, have been reported for the top 1-cm soil layer over a 15-week period for southeastern Australia.^[3] Further research at the same site using a portable field wind tunnel indicated an increase in the dominant sediment population of PSD of the surface soil layer (approximately 0–500 μm depth) after a 30-minute simulated erosion event. The analysis indicates that for the soil cultivation ridges, the proportion of the 300 μm sediment population increased from 60% in the parent soil to 86% after the simulated erosion. In the cultivation furrows, the 420 μm population increased from 0% to 85%. The analysis of the eroded

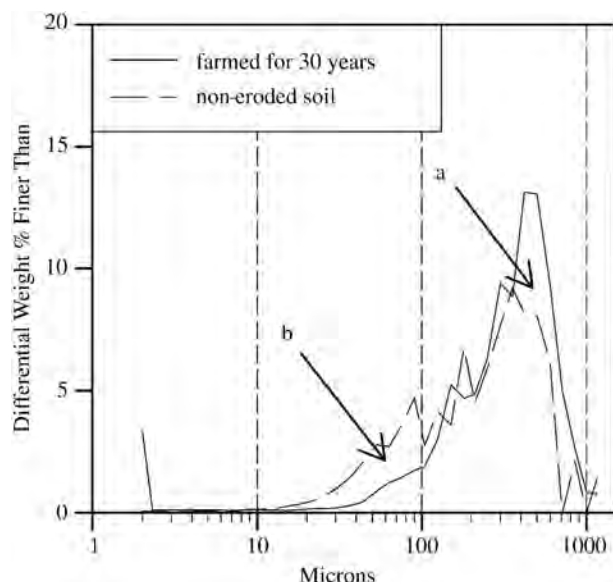


Fig. 1 PSD of two adjacent soils. One has been farmed for 30 years and has undergone repeated erosion; the other is under native vegetation. The PSDs show an increase in the coarse fraction (arrow a) and a decrease in the finer fraction (arrow b) in the eroded soil.

sediments indicated that >78% of material being eroded fell within the 180 μm population, compared with 29% in the parent soil. The increase in the coarser fractions of the surface soil and the high proportion of finer fractions in the eroded sediment imply that wind erosion is winnowing the fines.

Eroded Sediments

A large number of studies in North America,^[14] Belgium,^[15] Nigeria,^[16] Australia,^[3] and China^[17] show that the particle size decreases as the height increases, indicating a selective sorting of the eroded particles. When the vertical component of the wind exceeds the fall velocity of the eroded particles, they are removed from the site along with their associated nutrients. Australian results indicate that for a 1-week period of monitoring, 27% of the total eroded sediment is in the suspension fraction and thus is removed from the eroded field.^[3] These studies further highlight the winnowing action of the wind erosion processes and the potential loss of soil and nutrients.

SOIL PRODUCTIVITY

When soil is eroded from a site, there is often a subsequent decrease in rooting depth and available water-holding capacity.^[13] There is also a loss of nutrients and a subsequent decline in soil productivity. So if there is wind erosion, where does the nutrient go and how much impact does it have on crop production?

Soil Nutrient Loss

There is considerable evidence that wind-eroded sediments are enriched with nutrients. The nutrient loss has been measured at the site of dust emission, at various heights above the eroded surface, and at downwind sites of the eroded area, immediately adjacent to drift banks of soil and from deposited dust. It is important to examine the magnitude of this nutrient loss.

The nutrient content and particle size of eroded sediments collected at 0–0.5 m height during wind tunnel tests were similar to source sediments.^[18] However, if the sediments are sieved and the <90 μm fraction of the sediments are analyzed, then there are significant enrichment ratios, where the enrichment ratio is the ratio of the nutrient concentration in the eroded sediment to the nutrient concentration of the source soil. The sieve size of 90 μm was chosen because the modal particle size of sediments collected by using wind vane samplers^[19] from field trials in Australia, where the wind tunnel studies were undertaken, was all less than 90 μm for sediments collected at or greater than 0.5 m.^[3]

The mean enrichment ratio (standard deviation in parentheses) of eroded sediments sampled in a wind tunnel with a 4-m working section, for 25 sites, each replicated 10 times, for a range of farming systems and soil types in the drier agricultural areas of South Australia, was 3.43 (1.89) for total nitrogen (N), 2.53 (1.09) for total phosphorus (P), and 5.00 (2.46) for organic carbon after 1-minute duration of 75 km/hr wind.^[20] Therefore, there is significant winnowing of nutrients and enrichment of eroded sediments, even over short distances, i.e., 4 m.

The mean (standard deviation in parentheses) emission rates of nutrients from the above mentioned 25 sites (in $\text{g}/\text{m}^2/\text{sec}$) were 0.0007 (0.0008) for total N, 0.0002 (0.0002) for total P, and 0.0086 (0.0108) for organic carbon after a 1-minute duration of the 75 km/hr wind.

Analysis of dust collected in passive dust samplers at the source area shows that with increasing height above the ground, nutrient concentration increases, along with a decrease in particle size (Fig. 2).^[13] This indicates that the greater the sorting of the sediments, the greater the nutrient concentration in the eroded sediments.

Enrichment ratios of deposited sediment increase with distance from the erosion source because the sediments become more sorted. Measurement of wind-eroded material deposited near eroding fields has enrichment ratios in the order 1.21 for total N and 1.25 for organic carbon;^[21] therefore, some nutrient is redistributed locally. For one study in southeastern Australia, the enrichment ratios of 7.84 for total N and 2.96 for total P have been reported with the corresponding deposition rates of 0.0034 $\text{g}/\text{m}^2/\text{day}$ for total N and 0.0008 $\text{g}/\text{m}^2/\text{day}$ for total P.^[22]

The previous discussion reveals that a significant enrichment of eroded sediments occurs and that this enrichment

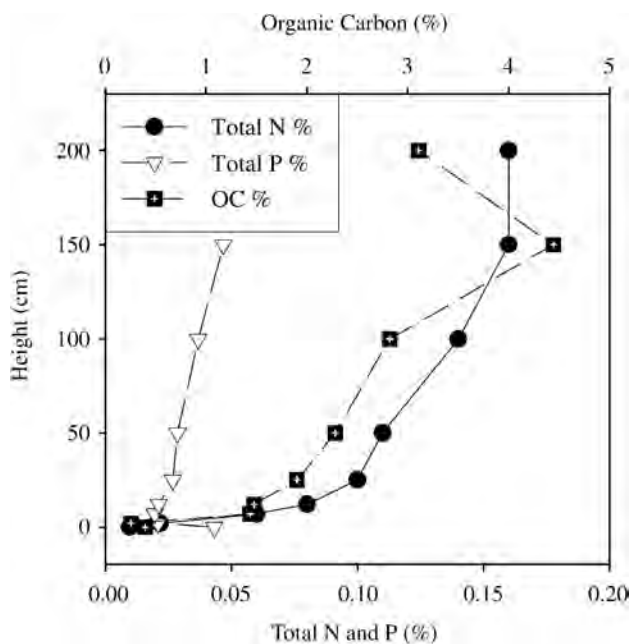


Fig. 2 Relationship between nutrient concentration and height for eroded sediments.

Source: Adapted from Leys & McTainsh.^[13]

increases with distance from source, both vertically and horizontally. The magnitude of this loss is large enough to influence subsequent crop production.

Crop Production Decline

There have been many experiments to test the assumption that soil loss causes crop production decline. The basic method has been to scalp off the surface soil and measure the subsequent changes in crop production. However, this approach is not entirely applicable to wind erosion because of the selective nature of the wind erosion processes, as previously described. The problem is compounded because plant response to erosion is complex.

Wind erosion winnows the finer fractions from the soil surface. When these sediments are analyzed, they do show considerable enrichment in nutrients compared to the soil from which they were derived. But is this nutrient loss significant?

Erosion can impact crop production because there is a reduction in the soil depth,^[23] a direct reduction in the amount of nutrients and water availability,^[13] or a combination of both.

Trial results are inconclusive, with some crops in some years showing no yield effects.^[24] Canadian research shows a linear decrease with increasing wind erosion severity.^[25] For spring wheat, the results demonstrate that for every 1 m across a 200-m field, there was a decline in grain yield of 3.6 kg/ha. When the experiment was repeated for canola the following year, no yield depression was

observed. Similarly, results in the United States revealed yield depression for two consecutive years for grain sorghum and kenaf, but this was not seen in each year for cotton and forage sorghum, owing to the climatic variations between years.^[24] It has also been suggested that above-average rainfall may also compensate for nutrient loss.^[26] This would further mask the effects of nutrient loss on soil production.

Nutrient Restoration

The obvious solution to nutrient loss is to replace it with the next crop. Previous studies demonstrate that additional fertilizers only partially compensate for nutrient loss from erosion.^[27] The effects of fertilizer are highly dependent on soil type, soil depth, the level of erosion, the initial fertilizer level, and the chemistry of the soil (i.e., P can be bound up in soils with high calcium carbonate).

CONCLUSION

Wind erosion of agricultural land has negative erosion effects on soil quality and soil productivity. The winnowing of the clay, silt, and organic fractions from the surface soil, and its transport off-site during wind erosion result in nutrient decline and a reduction in water-holding capacity of the soil. Sandy soils are more prone to wind erosion and suffer the greatest impact of soil and nutrient loss. The loss of soil productivity through wind erosion impacts the subsequent crop production. The level of decline in crop production depends on the crop, soil, and climatic factors. The use of inorganic fertilizers to replace lost nutrients assists in maintaining crop production but does not fully compensate for eroded particles and nutrients. Minimizing erosion is the best way to maintain soil quality and productivity.

REFERENCES

1. Lal, R. *Soil Quality and Agricultural Sustainability*; Ann Arbor Press: Chelsea, 1998; 1–378.
2. Williams, J.R.; Allmaras, R.R.; Renard, K.G.; Lyles, L.; Moldenhauer, W.D.; Langdale, G.W.; Meyer, L.D.; Rawls, W.J. Soil erosion effects on soil productivity: A research perspective. *J. Soil Water Conserv.* **1981**, *36*, 82–90.
3. Leys, J.F.; McTainsh, G.H. Sediment fluxes and particle grain-size characteristics of wind-eroded sediments in south-eastern Australia. *Earth Surf. Processes* **1996**, *21*, 661–671.
4. Zobeck, T.M.; Fryrear, D.W. Chemical and physical characteristics of windblown sediment quantities and physical characteristics. *T. ASAE* **1986**, *29* (4), 1032–1036.
5. Daniel, H.A.; Langham, W.H. The effect of wind erosion and cultivation on the total nitrogen and organic matter content of soil in the southern high plains. *J. Am. Soc. Agron.* **1936**, *28* (8), 587–596.
6. Armbrust, D.V. Wind and sandblast injury to field crops: Effect of plant age. *Agron. J.* **1984**, *76*, 991–993.

7. Gaynor, J.D.; MacTavish, D.C. Movement of granular simazine by wind erosion. *Hortic. Sci.* **1981**, *16*, 756–757.
8. Hagen, L.J.; Skidmore, E.L. *Wind Erosion and Visibility Problems*; No. 76-2019; American Society of Agricultural Engineers: St. Joseph, 1976; 1–15.
9. Saxton, K.E. Wind erosion and its impact on off-site air quality in the Columbia plateau—an integrated research plan. *T. ASAE* **1996**, *39*, 1031–1038.
10. Choudhury, A.H.; Gordian, M.E.; Morris, S.S. Associations between respiratory illness and PM10 air pollution. *Arch. Environ. Health* **1997**, *52* (2), 113–117.
11. Lyles, L. Wind erosion: Processes and effect on soil productivity. In *1976 Annual Meeting American Society of Agricultural Engineers*; No. 76–2016; American Society of Agricultural Engineers: Michigan, 1976; 23 pp.
12. Lyles, L.; Tatarko, J. Wind erosion effects on soil texture and organic matter. *J. Soil Water Conserv.* **1986**, *41*, 191–193.
13. Leys, J.F.; McTainsh, G.H. Soil loss and nutrient decline by wind erosion—cause for concern. *Aust. J. Soil Water Conserv.* **1994**, *7* (3), 30–35.
14. Vories, E.D.; Fryrear, D.W. Vertical distribution of wind-eroded soil over a smooth, bare field. *T. ASAE* **1991**, *34* (4), 1763–1768.
15. Goossens, D. The granulometrical characteristics of a slowly moving dust cloud. *Earth Surf. Processes.* **1985**, *10*, 353–362.
16. Sterk, G.; Raats, P.A.C. Comparison of models describing the vertical distribution of wind-eroded sediments. *Soil Sci. Soc. Am. J.* **1996**, *60*, 1914–1919.
17. Chen, W.; Fryrear, D.W. Sedimentary characteristics of drifting sediments above an eroding loessal sandy loam soil as affected by mechanical disturbance. *J. Arid Environ.* **1998**, *39*, 421–440.
18. Leys, J.F.; Heinjus, D.R. *Simulated Wind Erosion in the South Australian Murray Mallee*; Research Report; Soil Conservation Service of NSW: Sydney, 1991.
19. Shao, Y.; McTainsh, G.H.; Leys, J.F.; Raupach, M.R. Efficiencies of sediment samplers for wind erosion measurement. *Aust. J. Soil Res.* **1993**, *31*, 519–532.
20. Leys, J.; Packer, S.; Butler, P. *Wind Erosion Research at Keith in the Upper South East of South Australia*; Department of Land and Water Conservation: Sydney, 1997; Vol. 28.
21. Cihacek, L.J.; Sweeny, M.D.; Deibert, E.J. Characterization of wind erosion sediments in the red river valley of North Dakota. *J. Environ. Qual.* **1993**, *22* (2), 305–310.
22. Leys, J.F.; McTainsh, G.H. Dust and nutrient deposition to riverine environments of southeastern Australia. *Zeitschrift für Geomorphologie Supplementband* **1999**, *116*, 59–76.
23. Larney, F.J.; Izaurralde, R.C.; Janzen, H.H.; Olson, B.M.; Solberg, E.D.; Lindwall, C.W.; Nyborg, M. Soil erosion—crop productivity relationships for six Alberta soils. *J. Soil Water Conserv.* **1995**, *50* (1), 87–91.
24. Zobeck, T.M.; Bilbro, J.D. Crop productivity and surface soil properties of a severely wind-eroded soil. In *Sustaining the Global Farm*; Nearing, M., Ed.; Purdue University, West Lafayette, Indiana Soil Conservation Organization: Ankeny, 2000.
25. Larney, F.J.; Bullock, M.S.; Janzen, H.H.; Ellert, B.H.; Olson, E.C.S. Wind erosion effects on nutrient redistribution and soil productivity. *J. Soil Water Conserv.* **1998**, *53* (2), 133–140.
26. Arce-Diaz, E.; Featherstone, A.M.; Williams, J.R.; Tanaka, D.L. Substitutability of fertilizer and rain fall for erosion in spring wheat production. *J. Prod. Agric.* **1993**, *6*, 72–76.
27. Larney, F.J.; Janzen, H.H.; Olson, B.M. Efficiency of inorganic fertilizers in restoring wheat yields on artificially eroded soils. *Can. J. Soil Sci.* **1995**, *75*, 369–377.

Windblown Dust

Dale Gillette

U.S. Environmental Protection Agency (EPA), Research Triangle Park, North Carolina, U.S.A.

Abstract

Windblown dust may be understood in terms of generation, transport, and deposition. By mass, windblown dust is probably the single largest contributor to particles in the earth's atmosphere. The dominant mechanism for dust emission is "sandblasting," wherein sand-sized grains are initially made airborne; these sand grains either blast dust particles into the air from coatings of aggregated fine particles on larger particles, or splash fine particles from a reservoir of loose and fine particles. Size distributions of dust particles are broadly similar, despite differences in parent soil material, wind power, and other environmental factors.

INTRODUCTION

The composition of dust is similar to the composition of its parent material. However, enrichments of certain elements or minerals are explained by the size distribution of parent soil minerals and the activity of soluble material. Dust residence time in the atmosphere is approximately less than 30 days, but windblown dust can be transported over phenomenal distances (e.g., across the Atlantic from the Sahara Desert to the Caribbean and North America).

Windblown dust rarely rises above the troposphere, and wet and dry deposition act to remove it from the atmosphere. Long-term mean deposition rates of windblown dust vary widely approximately $10 \text{ g m}^{-2} \text{ yr}^{-1}$ for undisturbed desert areas in the United States; $10\text{--}100 \text{ g m}^{-2} \text{ yr}^{-1}$ for disturbed land areas and urban locations; and several hundred $\text{g m}^{-2} \text{ yr}^{-1}$ for locations within a few hundred kilometers of strong dust areas such as the Sahara.

EMISSION OF DUST

Although sufficiently strong winds can entrain dust directly, the predominant mechanism for emission of dust particles smaller than $10 \mu\text{m}$ (PM_{10}) is abrasion of larger particles that previously became airborne. There is a minimum threshold friction velocity (u_{*t}) for individual particles to be entrained into the airstream from a smooth surface.^[1] This minimum u_{*t} occurs for a particle diameter of approximately $100 \mu\text{m}$. Dust particles require much larger wind forces to remove them from the surface compared with sand-sized particles with diameters of approximately $100 \mu\text{m}$. Because loose particles of $100 \mu\text{m}$ are often found on soil surfaces, a "sandblasting" mechanism is almost always available for unprotected soils exposed to high winds. Physically, sand

grains localize kinetic energy onto small target areas compared with that of the fluid energy transfer with the surface. Experiments have confirmed that particle-to-particle interaction (sandblasting) on an erodible surface is an important and probably dominating mechanism that produces suspension particle flux.

A model for the particle flux of dust produced by the impacts of saltating particles showed that the vertical mass flux of dust particles (F_a) is proportional to the horizontal sand flux (q_{tot}) integrated over the total depth of the sand layer, multiplied by the mass per particle and divided by a binding energy Ω . The quantity Ω is characteristic of the strengths that hold dust particles in the soil. In other words, the model says that the ratio F_a/q_{tot} is related to the particle size of the saltating grains along with the binding energy of the dust particles.

Simultaneous data^[4] on horizontal fluxes of sand (q_{tot}) and vertical fluxes of PM_{10} (F_a) on agricultural soils in Texas formed a body of experimental work with which to evaluate the above model for F_a/q_{tot} . These results (Fig. 1) show that the ratio F_a/q_{tot} for "sand" textures is highly variable but has a consistency regardless of friction velocity, which is proportional to wind speed. That is, for sandy-soil textures, the sandblasting model for dust emission seems consistent with observations.

Sources of windblown dust are natural and anthropogenic. Natural sources include deserts, such as the Sahara, where there is lack of vegetation and (in some areas) soil crusting, coarse gravel and boulders, and presence of fine and sandy material in the soil. In the United States and other locations, deserts have more protection by vegetation, soil crusting, and coarse elements in the soil so that total dust emissions are smaller per unit area of desert. Highly active wind elements such as dust devils produce intense but highly local emissions when they encounter bare and loose soil.

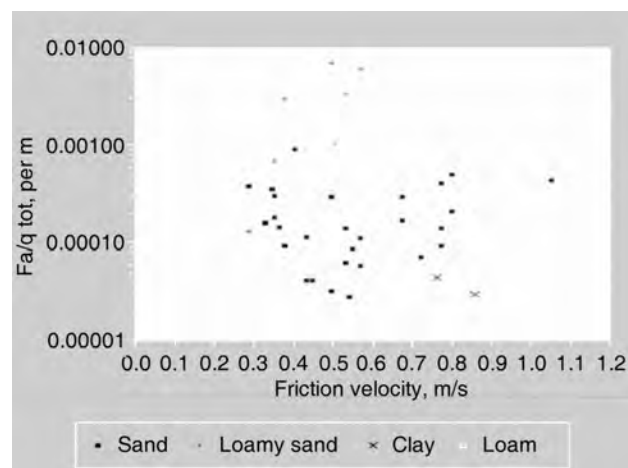


Fig. 1 Ratio of vertical flux of dust to horizontal flux of sand (per meter) vs. friction velocity (meters per second).

Source: From Gillette, Fryrear, et al.^[4]

Disturbances by man that leave large areas of loose soil exposed to high winds are some of the largest causes of wind-blown dust emissions. Examples of these are agriculture, diversion of rivers, and vehicular travel in arid and semiarid lands.

SIZE DISTRIBUTIONS AND OPTICAL PROPERTIES OF DUST

The distributions of dust mass concentration have been observed to be normal in logarithm of particle size (i.e., lognormal) with two modes. For many different wind speeds above the threshold (lower limit), researchers^[5] have found that there is a constant shape of dust mass size distribution for particles smaller than 10 μm . Scanning electron microscope analysis of typical dust samples shows that many particles are aggregated and that individual particles have differing mineralogies.

An explanation^[6] of the constant shape for dust size distributions is that the kinetic energy of individual saltating grains, along with the kinetic energy required to release suspendable particles, determines the quantities of different sizes of emitted dust. The huge variation in particle bond strengths and particle sizes explains the variability of F_a/q_{tot} as shown in Fig. 1, despite the overall similarity of dust size distributions produced over widely different soil locations and wind strengths.

Optical properties of the dust sampled in the United States may be summarized as a typical complex index of refraction for U.S. soil dust for visible light (488 nm); the real part of the index of refraction = 1.525 and the imaginary part of the index of refraction = 0.005i. For Saharan dust, the index of refraction was $1.55 + 0.008i$. Visibility studies using the formula $M = C/V$, where M is the mass concentration of soil aerosols (g m^{-3}) and V is

horizontal visibility (km), measured the constant C as $0.011 \text{ g m}^{-3} \text{ km}$.

COMPOSITION OF DUST

Mineralogically (chemically), a general model^[7] of dust for particles 1–10 μm consists of (in order of the most abundant): quartz, mica, kaolinite, mixed-layer phyllosilicates, and feldspars. For particles smaller than 1 μm , the order is: mica, kaolinite, quartz, and mixed-layer phyllosilicates. Particles from 10 to 100 μm are usually almost exclusively quartz grains characteristically coated with clay. The finer mode particles (1–10 μm) are similar to clay platelets that stick to the surface of larger quartz particles. Collisions of sand-sized particles with other particles (sandblasting) act to release portions of aggregated particles and break up the crystalline structure of mineral particles. One soil containing more clay than another “sandier” soil had the proportions of the 1–10 μm mode increase with wind speed relative to the 10–100 μm mode.^[8] But the sandier soil did not.

Usually, desert soils contain higher percentages of calcium carbonates than soils receiving greater rainfall. Because surface calcium carbonates are largely contained as fine particles, coatings, and interstitial material, the mechanisms of dust emission increase (enrich) the amount of calcium present in dust compared with that of the soil from which it was derived. Similar enrichments are found for other trace elements, such as the aluminum associated with clay composition. Soluble material, such as sodium chloride, is enriched in some barren desert soils. This concentration takes place through transport by water to low places, where the water is then lost largely through evaporation. Enrichment of salts in dust is also observed, especially at playas (drainage basins). Because windblown dust is rich in calcium carbonates, the percentage of carbonate may range from 5% to 25% by mass in deposited dust. Also, because soluble materials are concentrated at the surface in arid regions, salt content ranges from 0% to 20% by mass in deposited windblown dust.

TRANSPORT OF WINDBLOWN DUST

After being emitted from the surface to the air, windblown dust has the potential for transport. An approximate size limit for dust having the potential for long-range transport was defined as the size at which the fall (sedimentation) velocity is less than or equal to 0.1 times the friction velocity of the wind.^[9] As the smallest friction velocity for the threshold of particle movement is approximately 20 cm s^{-1} , the size limit corresponds to a particle having a sedimentation velocity of 2 cm s^{-1} . For soil-derived particles with densities of approximately 2500 kg m^{-3} , this corresponds roughly to particle diameters between 10 and 20 μm . Particles of this size with potential for long-range transport

could be depleted by deposition within a few minutes following their emission.^[10]

Although most dust is deposited near its point of origin, a small fraction of the dust emitted can be transported great distances. For example, dust has been shown to have transported from the Sahara Desert to Miami, Florida.^[11]

DEPOSITION OF WINDBLOWN DUST

Studies in desert regions that are not considered major global contributors of dust have sampled dust deposition, taking great care to exclude bird contributions and sand particles that are probably of local origin. The results showed that typical values of deposition for southern Nevada and southeastern California are 4.3–15.7 g m⁻² yr⁻¹.^[12] Typical values for south-central New Mexico (northern Chihuahuan desert) are 7.6–28.1 g m⁻² yr⁻¹.^[13] Depositions in urban desert areas of the United States are typically higher (e.g., 48 g m⁻² yr⁻¹);^[14] these values reflect dust contributions from anthropogenic disturbances of the soil followed by wind erosion on the disturbed soil.^[15]

Deposition rates downwind of agricultural areas, such as in western Texas, range from 10.8 to 13.7 g m⁻² yr⁻¹,^[16] whereas in the Great Plains of the United States rates range from 20.2 to 90.8 g m⁻² yr⁻¹.^[17] These higher deposition rates reflect sporadic large depositions from dust storms eroding bare farm fields. Deposition rates near major global atmospheric dust contributors, like the Sahara, are higher than other locations. Deposition in the Negev (Israel) ranges from 120 to 300 g m⁻² yr⁻¹.^[18] Rates at Niger, West Africa range from 115 to 138 g m⁻² yr⁻¹.^[19] However, far downwind from major dust sources, deposition rates of 1.3 g m⁻² yr⁻¹ were reported for dust from Africa reaching Miami, Florida.^[11]

REFERENCES

1. Bagnold, R. *The Physics of Blown Sand and Desert Dunes*; Methuen and Co. Ltd: London, 1941; 265 pp.
2. Shao, Y.; Raupach, M.R.; Findlater, P.A. The effect of saltation bombardment on the entrainment of dust by wind. *J. Geophys. Res.* **1993**, *98* (D), 12719–12726.
3. Houser, C.A.; Nickling, W.G. The emission and vertical flux of PM₁₀ from a disturbed clay-crustured surface. *Sedimentology* **2001**, *48*, 255–268.
4. Gillette, D.A.; Fryrear, D.W.; Gill, T.E.; Ley, T.; Cahill, T.A.; Gearheart, E.A. Relation of vertical flux of particles smaller than 10 µm to total Aeolian horizontal mass flux at Owens lake. *J. Geophys. Res.* **1997**, *102* (D), 26009–26015.
5. Gillette, D.A. Fine particulate emissions due to wind erosion. *Trans. Am. Soc. Agr. Eng.* **1977**, *29*, 890–897.
6. Alfaro, S.C.; Gaudichet, A.; Gomes, L.; Maille, M. Modeling the size distribution of a soil aerosol produced by sandblasting. *J. Geophys. Res.* **1997**, *102* (D10), 11239–11249.
7. Gillette, D.A.; Clayton, R.N.; Mayeda, T.K.; Jackson, M.L.; Sridhar, K. Tropospheric aerosols from some major dust storms of the southwestern United States. *J. Appl. Meteor.* **1978**, *17*, 832–845.
8. Gillette, D.A.; Walker, T.R. Characteristics of airborne particles produced by wind erosion of sandy soil, high plains of west Texas. *Soil Sci.* **1977**, *123*, 97–110.
9. Gillette, D.A.; Blifford, I.H., Jr.; Fryrear, D.W. The influence of wind velocity on size distributions of soil wind erosion aerosols. *J. Geophys. Res.* **1974**, *79*, 4068–4075.
10. Johnson, T.; Gillette, D.; Schwiesow, R. Fate of dust particles from unpaved roads under various atmospheric conditions. In *Precipitation Scavenging and Atmosphere-Surface Exchange*; Schwartz, S., Slinn, W.G.N., Eds.; Hemisphere Publishing Co.: Washington, 1992; 933–948.
11. Prospero, J.; Nees, R.; Uematsu, M. Deposition rate of particulate and dissolved aluminum derived from Saharan dust in precipitation at Miami, Florida. *J. Geophys. Res.* **1987**, *92* (14), 723–14731.
12. Reheis, M.; Kihl, R. Dust deposition in southern Nevada and California, 1984–1989: Relation to climate, source area, and source lithology. *J. Geophys. Res.* **1995**, *100*, 8893–8918.
13. Gile, L.; Hawley, J.; Grossman, R. Soils and geomorphology on the basin and range area of southern New Mexico—guidebook to the desert project, New Mexico bureau of mines mineralogical. Res. Memorandum **1981**, *39*, 222.
14. Pe'we', T.; Pe'we', E.; Pe'we', R.; Journaux, A.; Slatt, R. Desert dust: Characteristics and rates of deposition in Central Arizona, U.S.A. In *Desert Dust: Origin, Characteristics, and Effect on Man*; Pe'we', T., Ed.; Geological Society of America: Boulder, 1981; 169–190.
15. Holcombe, T.L.; Ley, T.; Gillette, D. Effects of prior precipitation and source area characteristics on threshold wind velocities for blowing dust episodes, Sonoran desert, 1948–1978. *J. Appl. Meteor.* **1997**, *36*, 1160–1175.
16. Rabenhorst, M.; Wilding, L.; Girdner, C. Airborne dusts in the Edwards plateau region of Texas. *Soil Sci. Soc. Am.* **1984**, *48*, 621–627.
17. Smith, R.; Twiss, P.; Krauss, R.; Brown, M. Dust deposition in relation to site, season, and climatic variables. *Soil Sci. Soc. Am. Proc.* **1970**, *34*, 112–117.
18. Offer, Z.; Goosens, D. Ten years of Aeolian dust dynamics in a desert region (Negev desert, Israel): Analysis of airborne dust concentration, dust accumulation and the high magnitude dust events. *J. Arid Environ.* **2001**, *47*, 211–249.
19. Drees, L.; Manu, A.; Wilding, L. Characteristics of Aeolian dusts in Niger, West Africa. *Geoderma* **1993**, *59*, 213–233.

Wind-Driven Rain: Erosion

Gunay Erpul

Department of Soil Science, Ankara University, Ankara, Turkey

L. Darrell Norton

National Soil Erosion Research Laboratory, U.S. Department of Agriculture—Agricultural Research Service (USDA-ARS), West Lafayette, Indiana, U.S.A.

Donald Gabriels

Department of Soil Management and Soil Care, Ghent University, Ghent, Belgium

Abstract

Wind can make significant changes in the physical characteristics of rains and hence in soil erosion processes. The rainsplash detachment and transport under wind-driven rainfall would differ from that under windless rain. The rainsplash transport may be a significant process to the extent that it may not be negligible in accurately predicting water erosion, and this process may result in a net transportation in the prevailing wind direction.

INTRODUCTION

Wind-driven rain can be described as raindrops falling through a wind field at an angle different from the vertical under the effects of both gravitational and drag forces. A schematic representation of wind-driven rain incidental to a sloping soil surface is given in [Fig. 1](#).

A study of wind-driven rain erosion is an attempt to investigate the combined effect of wind and rain on soil erosion processes in situations where wind and rain occur at the same time.

EFFECTS OF WIND ON PHYSICAL CHARACTERISTICS OF RAINS

Raindrop Size Distribution

In the assessment of the distribution of small simulated raindrops in a wind tunnel, Erpul, Gabriels, and Janssens^[1] obtained a narrower raindrop distribution and observed a distinct increase in mean drop diameter under wind-driven rain compared with windless rain. Collisions between small drops occurred more frequently as a result of their greater number per unit volume in air, leading to an increase in mean drop size. For large drops, however, this would not occur, as large drops are less stable and the wind caused some of them to break up into smaller drops.^[2,3] Basically, the effect of wind on raindrop size distributions is a potentially important effect that needs to be considered when estimating the rainfall erosivity.

RAINDROP IMPACT ENERGY

Wind-driven raindrops gain some degree of horizontal velocity, which increases their resultant impact velocity. The kinetic energy load of the rainfall may be expected to change as a result of increased velocity and the altered size of the raindrops. The exponential relationship between the horizontal wind velocity and the kinetic energy of raindrops was found in natural rains.^[4] The effect of wind on the horizontal component of small raindrops would be greater, so a greater percentage increase in kinetic energy would be expected for small raindrops than for large raindrops.^[5]

Raindrop Impact Angle

Wind-driven raindrops strike the soil surface with an angle deviated from the vertical because of their horizontal and vertical velocities. This inclination depends on the magnitude of wind velocity. In mid-latitudes, rain mostly falls at considerable inclination from the vertical, and the resultant angles of 40–70° have been found in rains driven by wind velocity of 10 m/s.^[6] Gabriels et al.^[7] reported that the mean angles of rain inclination were 52°, 66°, and 67° for the simulated rains in a wind tunnel driven by 6, 10, and 12 m/s of wind velocities, respectively. For these rains, a median drop size was approximately 1.50 mm. Little is known about the physical impact of raindrops on a soil in situations where this impact is not vertical. It is also not known whether inclined raindrops have stronger erosive effects than vertical ones. The extent and magnitude of the

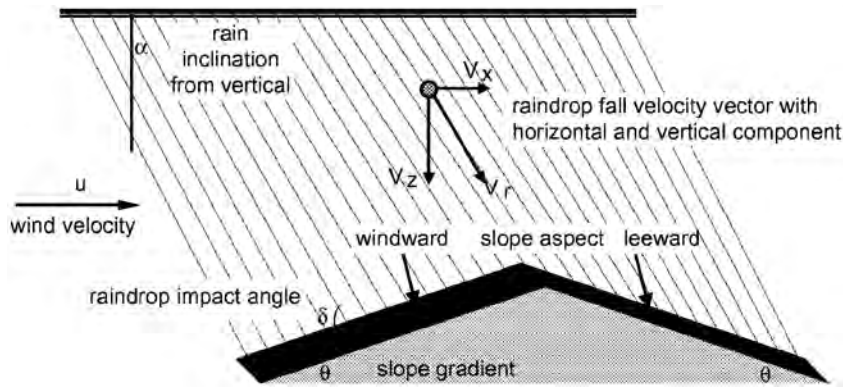


Fig. 1 Schematic representation of wind-driven rain with an angle from the vertical and incident on sloping soil surface.

rainsplash detachment increased as the angle of deviation increased within the range of 5–30°^[8] whereas, with greater deviations, the impact angle could be so small that raindrops would hardly hit the soil surface. Especially when impact angles were less than 30°, rainsplash detachment rate highly decreased (Table 1).

Raindrop Impact Frequency

The distribution and the intensity of rain on sloping surfaces differ depending on wind direction and velocity. In fact, the angle of rain incidence (π), which is a function of rain inclination (α), slope gradient (θ), and slope aspect, determines the rain intensity in wind-driven rains. As an example, a windward-facing slope can receive two times more rain intensity than a leeward-facing slope, or even exceed it in extreme cases for rain inclinations of

40–70°.^[6] When the rain inclination and the slope gradient increase, the discrepancies in the rain intensity between wind slope and leeward slope become greater (Table 1).

EFFECTS OF WIND ON RAINSPLASH DETACHMENT AND TRANSPORT

Rainsplash Detachment

Similar to the effects on rain characteristics, wind movement and velocities can have a profound effect on some aspects of the soil erosion process. When wind accompanies rain, rainsplash detachment tends to increase owing to the increased energy of raindrops. Pedersen and Hasholt^[4] obtained a better correlation between the rainsplash detachment and erosivity indexes, when wind velocity was taken

Table 1 The effect of wind on rain energy, rain intensity, and rainsplash detachment and transport of a silt loam soil.

u (m/sec)	KE (J/m ² /mm)	Windward							Leeward				
		α (°)	θ (°)	π (°)	I (mm/hr)	D (g/m ² /sec)	X _c (m)	q _s (g/m/ec)	π (°)	I (mm/hr)	D (g/m ² /sec)	X _c (m)	q _s (gm/sec)
0	2.23	0	4	4	143	0.23	0.35	0.08	4	161	0.47	0.34	0.16
			9	9	141	0.32	0.38	0.12	9	172	0.75	0.35	0.26
			11	11	132	0.41	0.45	0.18	11	180	0.98	0.36	0.35
6	4.37	52	4	48	92	0.41	0.61	0.25	56	126	0.82	0.73	0.60
			9	44	106	0.51	0.61	0.31	61	107	0.28	0.74	0.21
			11	41	112	0.77	0.62	0.48	63	97	0.30	0.70	0.21
10	13.20	66	4	62	125	2.01	1.26	2.53	70	90	0.78	1.12	0.87
			9	53	137	2.01	1.37	2.75	75	61	0.27	1.05	0.28
			11	55	141	2.91	1.40	4.07	78	50	0.29	0.95	0.28
12	17.94	67	4	63	93	2.26	1.92	4.34	71	66	0.94	1.66	1.56
			9	59	114	4.39	1.86	8.17	76	42	0.28	1.50	0.42
			11	56	120	5.47	1.88	10.28	78	32	0.24	1.17	0.28

Note: u, horizontal wind velocity; KE, kinetic energy [the exponential relationship found between KE and u is $E(u)=1.9723e^{0.1796u}$ (the relationship between columns 1 and 2)]; α , rain inclination from vertical; θ , slope gradient; π , angle of rain incidence calculated using rain inclination, slope gradient, and slope aspect by cosine law: $\cos(\pi)=\cos(\alpha-\theta)=\cos\alpha\cos\theta+\sin\alpha\sin\theta$ for windward slopes and $\cos(\pi)=\cos(\alpha+\theta)=\cos\alpha\cos\theta-\sin\alpha\sin\theta$ for leeward slopes; I, rain intensity; D, rainsplash detachment rate; X_c , mean rainsplash distance calculated by the center of gravity of mass distribution curves; and q_s , rainsplash transport estimated by $q_s=DX_c$.

into account in the kinetic energy calculation. However, the wind not only increased the raindrop impact energy, but also altered the angle of rain incidence, resulting in the variations in the raindrop impact frequency and the raindrop impact angle. In other words, raindrop impact energy, impact frequency, and impact angle determine the magnitude of rainsplash detachment under wind-driven rainfall. The influence of each factor on the process is not exclusive and not clearly distinguished, because all are closely related, and each factor is a function of the wind speed. The highest rainsplash rate occurred with the highest impact frequency and the highest impact angle at a given raindrop impact energy under wind-driven rains (Table 1).

Rainsplash Transport

Wind, as well as slope and overland flow, is another possible factor capable of transporting detached particles by raindrop impact. It is possible to find similarities between the movement and trajectory of sand saltation by wind and the movement and trajectory of soil particles by rainsplash under wind-driven rainfall. In saltation, the hitting sand grains, once ejected, initiate the motion of uplift,^[9] whereas hitting raindrops on the soil surface with an angle initiate a jumping movement of soil particles in wind-driven rainfall.^[10] A threshold is given by impacting raindrops, and wind does not account for the upward movement except that it changes the energy, frequency, and angle of raindrop impact. Once soil particles are entrained in the splash droplets that have risen into the air by raindrop impact, wind velocity gradient will transport these particles. It is found that, together with wind velocity gradient, the impact angle is also playing a significant role in determining the extent of the process. At the same wind speed, longer mean rainsplash distances (X_c) were observed when the impact angle became greater (Table 1). There was a very discernible decrease in X_c in leeward-facing slopes, which was mostly associated with the impact angles of less than 30° . These

results showed that the raindrop impact angle might determine the soil particle ejection angle.

As mentioned above, the amount of soil particles to be transported by the wind will depend on the raindrop impact energy, impact frequency, and impact angle. Subsequently, the soil particle ejection angle and the wind velocity gradient will determine the extent of the process (Fig. 2).

CONCLUSION

Wind could make significant changes in the physical characteristics of rains and hence in soil erosion processes. The rainsplash detachment and transport under wind-driven rainfall would differ from that under windless rain. The rainsplash transport could be a significant process to the extent that it may not be negligible in accurately predicting water erosion, and this process could result in a net transportation in the prevailing wind direction.

REFERENCES

- Erpul, G.; Gabriels, D.; Janssens, D. Assessing the drop size distribution of simulated rainfall in a wind tunnel. *Soil Till. Res.* **1998**, *45*, 455–463.
- Lyles, L.; Disrud, L.A.; Woodruff, N.P. Effects of soil physical properties, rainfall characteristics, and wind velocity on clod disintegration by simulated rainfall. *Soil Sci. Soc. Am. Pro.* **1969**, *33* (2), 302–306.
- Disrud, L.A.; Lyles, L.; Skidmore, E.L. How wind effects the size and shape of raindrops. *Agr. Eng.* **1969**, *50* (10), 617.
- Pedersen, H.S.; Hasholt, B. Influence of wind speed on rainsplash erosion. *Catena* **1995**, *24*, 39–54.
- Erpul, G.; Gabriels, D.; Janssens, D. The effect of wind on size and energy of small simulated raindrops: A wind tunnel study. *Int. Agrophys.* **2000**, *14*, 1–7.
- Sharon, D. The distribution of hydrologically effective rainfall incident on sloping ground. *J. Hydrol.* **1980**, *46*, 165–188.
- Gabriels, D.; Tack, K.; Erpul, G.; Cornelis, W.M.; Norton, L.D.; Biesemans, J. Effect of wind-driven rain on splash detachment and transport of a silt loam soil: A short slope wind tunnel experiment. In *Proceedings of the International Workshop on Technical Aspects and Use of Wind Tunnels for Wind-Erosion Control, and Combined Effect of Wind and Water on Erosion Processes*, Ghent, Belgium, Nov 17–18, 1997; Gabriels, D., Cornelis, W.M., Eds.; I.C.E. Special Report No. 1998/1; International Center for Eromology, University of Ghent: Belgium, 1997; 87–93.
- Van Heerden, W.M. Splash Erosion as Affected by the Angle of Incidence of Raindrop Impact. Unpublished Ph. D. Thesis, Purdue University, Lafayette, 1964.
- Bagnold, R.A. *The Physics of Blown Sand and Desert Dunes*; 1973 Reprint; Chapman & Hall: London, 1941; 265 pp.
- De Lima, J.L.M.P.; Van Dick, P.M.; Spaan, W.P. Splash-saltation transport under wind-driven rain. *Soil Technol.* **1992**, *5*, 151–166.

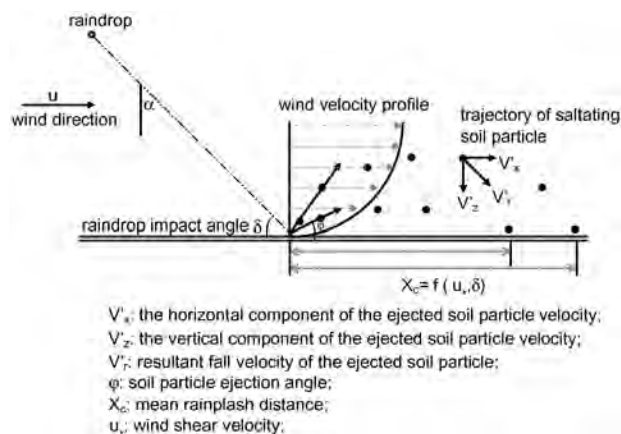


Fig. 2 Rainsplash transport: raindrop-induced and wind-driven splash trajectories of soil particles falling through a wind profile.

World Reference Base for Soil Resources

Peter Schad

Soil Science, Technical University of Munich, Freising-Weihenstephan, Germany

Stefaan Dondeyne

Earth and Environmental Science, University of Leuven, Leuven, Belgium

Abstract

The international soil classification system, World Reference Base (WRB) for Soil Resources, is edited by a Working Group of the International Union of Soil Sciences (IUSS) and published by the Food and Agriculture Organization of the United Nations (FAO). The third edition was released in 2014. The WRB has two hierarchical levels. The first level comprises 32 Reference Soil Groups (RSGs), which are identified using a key. Many RSGs represent specific soil-forming processes or major soil regions. At the second level, the soil names are constructed by adding qualifiers to the name of the RSG. In total, 184 qualifiers are defined. Some can be combined with many RSGs, others with only a few. For every RSG, a list of the available qualifiers is provided. They are subdivided into principal (ranked and given in an order of importance) and supplementary qualifiers (not ranked, but in alphabetical order). In the WRB, diagnostic horizons, properties, and materials are defined. Diagnostic horizons and properties reflect common results of soil-forming processes or indicate specific conditions of soil formation. In addition, diagnostic horizons require a certain thickness. Diagnostic materials significantly influence pedogenetic processes or are indicative of them. The definitions of many RSGs and qualifiers refer to the presence or absence of diagnostics. In addition, many definitions refer to individual characteristics such as clay content. To name a soil, the RSG has to be provided with all applying qualifiers. For map legends, the number of qualifiers depends on scale and purpose of the map.

INTRODUCTION

The third edition of the international soil classification system, World Reference Base (WRB) for Soil Resources, was released in June 2014. In 2015, an update of the third edition of the WRB was published with some minor corrections and amendments.^[1] The third edition is based on the second (2006) and the first (1998) editions. The WRB builds on the Legends (1974, 1988) of the Soil Map of the World (FAO–UNESCO 1971–1981). The body responsible for revising and editing the WRB is a Working Group of the International Union of Soil Sciences (IUSS). In 1998, the IUSS Council endorsed the WRB as its officially recommended terminology to name and classify soils.

In WRB, soils are classified at two categorical levels. The first level has 32 Reference Soil Groups (RSGs), which are identified using a key. In the second level, the soil names are constructed by adding a set of qualifiers to the name of the RSG.

THE FIRST LEVEL

RSGs are groups of soils sharing morphologic, physical, and chemical characteristics, as they have undergone similar formation processes or have been formed from particular parent material or represent major soil regions. [Table 1](#)

presents the 32 RSGs in the sequence of the key, together with a brief and simplified description.

THE SECOND LEVEL

For the lower level, 184 qualifiers have been defined. Some can be combined with many RSGs, others with only a few or even with just one. The qualifiers available for use with a particular RSG are listed in the key, along with the RSG. They are divided into principal and supplementary qualifiers. Principal qualifiers are regarded as being the most significant for a further characterization of soils of the particular RSG. They are ranked and given in an order of importance. Supplementary qualifiers give some further details about the soil. They are not ranked and must be used in alphabetical order. The Cambisols have the longest list with 67 possible qualifiers, and the Nitisols have the shortest list with 34 possible qualifiers. Many qualifiers are mutually exclusive, and, in practice, most classified soils have fewer than 10 qualifiers.

THE DIAGNOSTICS

In the WRB, diagnostic horizons, properties, and materials are defined. Diagnostic horizons and properties are

Table 1 Simplified guide to the RSGs (IUSS Working Group WRB 2014, Update 2015).

Brief description/principal characteristics	RSG
1. Soils with thick organic layers	Histosols
2. Soils with strong human influence	
With long and intensive agricultural use	Anthrosols
Containing significant amounts of artefacts	Technosols
3. Soils with limitations to root growth	
Permafrost affected	Cryosols
Thin or with many coarse fragments	Leptosols
With a high content of exchangeable sodium	Solonetz
Alternating wet–dry conditions, shrink–swell clays	Vertisols
High concentration of soluble salts	Solonchaks
4. Soils distinguished by Fe/Al chemistry	
Groundwater affected, underwater and in tidal areas	Gleysols
Allophanes or Al–humus complexes	Andosols
Subsoil accumulation of humus and/or oxides	Podzols
Accumulation and redistribution of Fe	Plinthosols
Low-activity clay, phosphorus fixation, many Fe oxides, strongly structured	Nitisols
Dominance of kaolinite and oxides	Ferralsols
Stagnating water, abrupt textural difference	Planosols
Stagnating water, structural difference, and/or moderate textural difference	Stagnosols
5. Pronounced accumulation of organic matter in the mineral topsoil	
Very dark topsoil, secondary carbonates	Chernozems
Dark topsoil, secondary carbonates	Kastanozems
Dark topsoil, no secondary carbonates (unless very deep), high base status	Phaeozems
Dark topsoil, low base status	Umbrisols
6. Accumulation of moderately soluble salts or non-saline substances	
Accumulation of, and cementation by, secondary silica	Durisols
Accumulation of secondary gypsum	Gypsisols
Accumulation of secondary carbonates	Calcisols
7. Soils with a clay-enriched subsoil	
Interfingering of coarser-textured, lighter coloured material into a finer-textured, stronger coloured layer	Retisols
Low-activity clays, low base status	Acrisols
Low-activity clays, high base status	Lixisols
High-activity clays, low base status	Alisols
High-activity clays, high base status	Luvissols

(Continued)

Table 1 Simplified guide to the RSGs (IUSS Working Group WRB 2014, Update 2015). (Continued)

Brief description/principal characteristics	RSG
8. Soils with little or no profile differentiation	
Moderately developed	Cambisols
Sandy	Arenosols
Stratified fluvial, marine, and lacustrine sediments	Fluvisols
No significant profile development	Regosols

Note: Al = aluminum; Fe = iron.

characterized by a combination of attributes that reflect widespread common results of the processes of soil formation or indicate specific conditions of soil formation. Their features can be observed or measured, either in the field or in the laboratory, and require a minimum or maximum expression to qualify as diagnostic. In addition, diagnostic horizons require a certain thickness, thus forming a recognizable layer in the soil. Diagnostic materials are materials that significantly influence pedogenetic processes or are indicative of them. Their features may stem from the parent material or may be the result of pedogenetic processes. The definitions of many RSGs and qualifiers refer to the presence or absence of diagnostics at a certain depth. In addition, many definitions refer to individual features such as the base saturation or clay content.

NAMING A SOIL (A PROFILE)

The first step is detecting diagnostic horizons, properties, and materials. In the second step, the RSG is identified using the key. In the third step, all applying principal and supplementary qualifiers are allocated. The principal qualifiers are added before the name of the RSG without brackets and commas. The sequence is from right to left, i.e., the uppermost qualifier in the list is placed closest to the name of the RSG. The supplementary qualifiers are added in brackets after the name of the RSG and are separated from each other by commas. The sequence is from left to right, i.e., the first qualifier according to the alphabet is placed closest to the name of the RSG.

Constructing the second level by adding qualifiers to the RSG rather than by using a dichotomic identification key has several advantages:

- For every soil, the RSG has the appropriate number of associated qualifiers.
- Most of the soil's properties are incorporated into an informative soil name.

- The system is robust. When some data are not available, the classification does not lead to a fundamental error in the soil name. If a qualifier is erroneously added or omitted due to incomplete data, the rest of the soil name remains correct.

CREATING MAP LEGENDS

Whereas for naming a soil, all applying qualifiers must form part of the soil name, for elaborating map legends, the number of qualifiers will depend on the scale and purpose of the map. The WRB recognizes four scale levels:

- for very small map scales, only the RSG is used.
- for next larger map scales, the RSG plus the first applicable principal qualifier is used.
- for next larger map scales, the RSG plus the first two applicable principal qualifiers are used.
- for next larger map scales, the RSG plus the first three applicable principal qualifiers are used.

If fewer qualifiers are applicable than described above, the lesser number is used. The principal qualifiers are placed before the name of the RSG according to the rules for naming a soil. Other qualifiers may be added optionally depending on the purpose of the map or according to national traditions, and irrespectively of the map scale. These additional qualifiers may be principal qualifiers from further down the list, or supplementary qualifiers, and must be used according to the above-mentioned rules for supplementary qualifiers.

For most maps, it is desirable that a map unit does not show a single soil but an association of soils. The WRB distinguishes:

- dominant soils represent $\geq 50\%$ of the soil cover.
- codominant soils represent $\geq 25\%$ and $< 50\%$ of the soil cover.
- associated soils represent $\geq 5\%$ and $< 25\%$ of the soil cover.

For codominant or associated soils, fewer numbers of qualifiers than indicated above (or even no qualifier) may be appropriate.

SUBQUALIFIERS

Qualifiers may be combined with specifiers (e.g., Epi-, Proto-) to form subqualifiers (e.g., Epiarenic, Protocalcic). Depending on the specifier, the subqualifier fulfills all the criteria of the respective qualifier, or it deviates in a defined way from its set of criteria. The use of specifiers does not change the position of the qualifier in the soil name. Only if principal qualifiers are combined with the specifiers, Bathy-(greater depth), Thapto-(buried), or Proto-(weaker

expressed), the created subqualifier shifts to the supplementary qualifiers. The use of subqualifiers (instead of qualifiers) that deviate from the set of criteria of the respective qualifier is compulsory. The use of subqualifiers (instead of qualifiers) that fulfill all the criteria of the respective qualifier is optional. They are recommended especially for soil-naming soils. Their use is not recommended for principal qualifiers in map units or wherever generalization is important.

Qualifiers that have depth requirements can be combined with the specifiers, Epi-, Endo-, Amphi-, Ano-, Kato-, Panto-, and Bathy- to create subqualifiers (e.g., Epicalcic, Endocalcic) further expressing the depth of occurrence. If a diagnostic horizon or a layer with a diagnostic property belongs to a buried soil that does not meet the requirements of the related RSG (see “Buried Soils”), the Thapto- specifier can be used. For soils having a consolidated layer at shallow depths, subqualifiers with the specifier Supra- can be used to describe the soil material above, if the thickness or depth requirements of a qualifier or of its respective diagnostics are not fulfilled, but all other criteria are fulfilled throughout in the soil material above. Besides these subqualifiers constructed individually, the WRB also provides subqualifiers with a given definition (e.g., Protosalic, Neocambic).

BURIED SOILS

A buried soil is a soil covered by younger deposits. Where a soil is buried, the following rules apply:

1. The overlying material and the buried soil are classified as one soil if both together qualify as a Histosol, Anthrosol, Technosol, Cryosol, Leptosol, Vertisol, Gleysol, Andosol, Planosol, Stagnosol, Arenosol, Fluvisol, or Regosol.
2. Otherwise, the overlying material is classified with preference if it is ≥ 50 cm thick or if the overlying material, if it stood alone, satisfies the requirements of a Folic Regosol or of a RSG other than a Regosol. For depth requirements in the overlying material, the lower limit of the overlying material is regarded as if it were the upper limit of continuous rock.
3. In all other cases, the buried soil is classified with preference. For depth requirements in the buried soil, the upper limit of the buried soil is regarded as its soil surface.
4. If the overlying soil is classified with preference, the name of the buried soil is placed after the name of the overlying soil adding the word “over” in between, e.g., Skeletic Umbrisol (Siltic) over Albic Podzol (Arenic).
5. If the buried soil is classified with preference, the overlying material is indicated with the Novic qualifier.

EXAMPLES

A soil named *Hypereutric Albic Endogleyic Mollic Stagnosol* (*Hyperhumic*, *Inclitic*, *Pantoloamic*) belongs to the RSG of the Stagnosols, which implies that it has a topsoil with perched water, reducing conditions and a typical distribution pattern of iron oxides. Following their ranking order, the principal qualifiers provide more information on the nature of the soil: *Mollic* indicates the presence of a thick, dark, structured mineral topsoil horizon with a high base saturation; *Endogleyic* indicates that the subsoil is influenced by groundwater and shows the related distribution pattern of iron oxides; *Albic* indicates the occurrence of whitish soil material relatively depleted of iron oxides; *Hypereutric* reflects a very high base saturation throughout a depth from 20 to 100 cm. The supplementary qualifiers give additional information—a high average content of organic matter up to a depth of 50 cm (*Hyperhumic*), the occurrences of subsurface water flow (*Inclitic*), and a soil texture that is loamy throughout (*Pantoloamic*).

If we use the same example for creating map units, depending on the scale we have:

- first map scale level: Stagnosol
- second map scale level: Mollic Stagnosol
- third map scale level: Gleyic Mollic Stagnosol
- fourth map scale level: Albic Gleyic Mollic Stagnosol.

(The specifiers have not been used to help for a better aggregation of the soils of a map unit.)

If we want to add additional qualifiers, we may have, for example, the following map units:

- Stagnosol (Gleyic)
- Mollic Stagnosol (Gleyic)
- Gleyic Mollic Stagnosol (Hypereutric, Pantoloamic)
- Albic Gleyic Mollic Stagnosol (Hypereutric, Pantoloamic).

THE MAJOR CHANGES IN THE WRB 2014

Having qualifier sequences and rules for qualifier usage that are suitable for both naming soils and creating map legends is one of the advantages of the third edition compared to the second. Furthermore, the new edition integrates better the needs for classifying soils in landscapes that had been neglected before, e.g., very continental permafrost regions. The WRB as international soil classification system should be able to express characteristics regarded as important in national systems. Some amendments have been made to allow for the better representation of soil units in the WRB, for example, from the Australian and the Brazilian systems. Great efforts were undertaken to write the definitions in a more logical and a more didactical way. Precise rules have been introduced for the use of specifiers to define subqualifiers.

ACKNOWLEDGMENTS

The editors of the third edition of the WRB are: Peter Schad (Germany), Cornie van Huyssteen (South Africa), and Erika Micheli (Hungary). Members of the WRB Board are: Lúcia Anjos (Brazil), Carlos Cruz Gaistardo (Mexico), Seppe Deckers (Belgium), Stefaan Dondeyne (Belgium), Einar Eberhardt (Germany), Maria Gerasimova (Russia), Ben Harms (Australia), Arwyn Jones (European Commission), Pavel Krasilnikov (Russia), Thomas Reinsch (the United States), Ronald Vargas (FAO), and Ganlin Zhang (China). Numerous other soil scientists contributed to the third edition of the WRB.

REFERENCE

1. IUSS Working Group WRB. *World Reference Base for Soil Resources 2014, Update 2015*. World Soil Resources Reports No. 106; Prepared by Schad, P.; van Huyssteen, C.; Micheli, E.; FAO: Rome, 2014–2015; 192 pp.



Taylor & Francis

Taylor & Francis Group

<http://taylorandfrancis.com>

Index

A

- Aackia karakoramensis*, 449
- Ab initio* methodologies, quantum mechanical methods, 2167
- Above mean sea level (AMSL), 1383–1384
- Abscissic acid (ABA), 694
- Absorption
- with thermal and fast neutrons, 320
 - and translocation, phosphorus, 1703
- Acacia auriculiformis*, 1324
- Acacia* spp., 827
- Acariformes, 1488
- Acarina, 206
- Accelerated biodegradation, 1773
- Accelerated erosion, 2354
- Accelerator mass spectroscopy (AMS), 1451
- Accounting system. *See* **Emergy, analysis**
- Accumulation, 2192
- precipitation, 93
 - SOM
 - environmental controls, 2114–2115
 - littering, 2112
 - in mineral soils, 2113–2114
 - OM transformations, 2112–2113
 - in organic soils, 2113
 - pedoturbation, 2112
 - root turnover, 2112
 - subsurface, 2114
- Acetobacter diazotrophicus*, 222
- Acetylene reduction, 221
- Achaeta eiseni*, 724
- Achlorophyllous plants, 1503
- Acid(s)
- ammonium oxalate, 1471
 - deposition
 - anthropogenic disturbances, 1821
 - control program, 11
 - drainage, 37, 38
 - gas removal processes, 1801
 - mines oil remediation, *in situ*, 1463–1464
 - phosphatase in soil, 494–495
 - soils, 1396
 - sulfate, 8, 17
- Acid–base titration, 1430
- Acidic complexolitic andosolization, 1650
- Acidic deposition, 11–15
- precursor pollutants, 13
- Acid rock drainage (ARD), 2465
- Acidic soils, 14
- Acidification, 13, 559, 880, 1393
- of aquatic ecosystems, 15
- Acidithiobacillus*, 9
- Acidithiobacillus ferrooxidans*, 9, 1833
- Acidithiobacillus thiooxidans*, 9
- Acidity, 1362
- complex, 233
 - containment and neutralization, 34
 - limits, 558
 - pH, 563–564
- Acid mine drainage (AMD), 6, 1464
- chemistry, 6–8
 - environmental impacts, 9
 - microbiology, 9
 - mineralogy, 8–9
 - prevention, 9
 - treatment, 9–10
- Acidobacteria*, 179
- Acidolysis and acidocomplexolysis, 2559–2560
- Acidophilic bacteria, 8
- Acid rain, 11
- ecosystem impacts, 13
 - agricultural ecosystems, 14
 - aquatic ecosystems, 15
 - forest ecosystems, 14–15
 - soils, 13–14
 - human health effects, 13
 - reducing acidic deposition effects, 15
 - sources and distribution, 11–13
 - structural impacts, 13
- Acid rock drainage (ARD), 31
- Acid sulfate soils (ASS)
- biogeochemistry of sulfide mineral formation, 26
 - oxidizable organic carbon, 26
 - reactive Fe, 27
 - reducing/saturated conditions, 26–27
 - sulfate, 26
 - sulfate-reducing bacteria, 27
- classification in other systems, 18
- defined, 17, 36
- distribution and extent, 20–21
- regional distribution, 21–22
- environments of sulfide formation and accumulation, 28
- management, 31
- acidity containment and neutralization, 34
 - approaches, 33
 - avoidance, 34
 - education and assessment, 33–34
 - oxidation prevention, 34
 - problems for, 31–33
- problems, 36
- acid drainage, 37, 38
 - agriculture, 37
 - dredging, 37
 - heavy liming, 38
 - land disturbance, 36
 - mining, 36–37
 - neutralization, 38
 - non-disturbance, 37–38
 - plant growth, effects on, 37
 - remediation strategies, 37–38
 - ripening or flooding, 38
 - urban development, 37
 - wetlands, 38
- soil taxonomy, 17–18
- sulfidic materials, 18
 - sulfuric horizon, 18
 - sulfurization, sulfide oxidation, 28–29
- Acquired immunodeficiency syndrome (AIDS), 1115
- Acrisols, 392, 409
- Acromyrmex lobicornis*, 151
- Actinobacteria*, 40, 179
- dispersal of, 40
 - factors influencing distribution, 40
 - gene pool, 40–41
 - seasonality of distribution, 40
 - secondary metabolite production, 41–42
 - in soil environment, 41
 - versatility of activities, 41
- Actinobacterial diversity, 41
- Actinomadura*, 41
- Actinomycetes, 138, 1435
- Active migration, nematodes, 1520
- Active plinthite formation, 1744
- Activity-ratio diagram, 1482
- Adenosine triphosphate (ATP), 220, 221, 363
- extraction method, 1431
- Adhesive-lined heat-shrinkable tubing, 2304
- Adsorption
- cation exchange, 1427
 - chemisorption, 1427–1428
 - concentration and affinity, 1426
 - diffuse layer ions, 1427–1428
 - Freundlich isotherm model, 1427
 - H-curve, 1426
 - inner-sphere surface complexes, 1427–1428
 - isotherms for metal ions, 1426–1427
 - L-curve, 1427
 - metal-ion adsorption, 1427
 - metal-ion bridging, 1428
 - molecular-scale mechanisms, 1427–1428
 - outer-sphere surface complexes, 1427–1428
 - oxyanion, 1427–1428
 - pH dependence, 1427
 - phyllosilicate minerals, 1427
 - plant nutrients and contaminants, 427–428
- Adsorption–desorption hysteresis, 428
- Advanced ceramics, 1418–1419
- Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER), 630
- Advanced very high resolution radiometer (AVHRR), 582
- Advection, 2194
- coefficient, 1675
- Advection–Dispersion Equation (ADE), 2194–2195
- AEC. *See* **Anion exchange capacity (AEC)**
- Aeolian dust deposition, 103
- soil organic matter and biological activity, 103
 - weathering and soil formation rate, 103
- Aeration, 44, 85, 562, 681
- aggregate strength
 - on crop and environment, 51–52
 - measurement of, 52–53
 - to soil properties, 52
 - and tillage, 51
 - capacity measurement, 44
 - intensity measurement, 44–45
 - subsurface tillage effects, 48–49
 - tensile strength of aggregates, 51

- Aeration—
tillage
 effects, 47–48
 and greenhouse gases, 49
 transport rate measurement, 45–46
- Aerobic biodegradation, 202
- Aerobic microbial activity, 596
- Aerobic respiration, 177
- Aerosols, 119
- Afforestation, 286
- African drought-prone areas, 402
- Aggregate erosion, soil (SAE), 2226
- Aggregate formation, soil, 1506
- Aggregates, 1629
 stability and carbon sequestration, 1435
- Aggregation, 53
 direct characterization, 61
 formation, 55
 aluminosilicate clays, 56
 earthworms, 55
 fungi and bacteria, 55
 inorganic cations, 56
 iron and aluminum oxides, 56
 roots and vesicular–arbuscular mycorrhizal (VAM), 55
 termites, 55–56
 transient and persistent organic matter, 55
 wetting and drying and freezing and thawing, 56
 fragmentation procedures, 62
 dry sieving, 62
 energy input, 62
 wet sieving, 62
 hierarchy, 55
 macropores, 1388
 and organization, 2021
 sampling and sample preparation, 61–62
 size distribution indices, 62
 SOC sequestration, 1467
 soil quality, 1505–1506
 soil structural stability, 62–63
 structural stability measurement
 fragments, and particles, 61
- Agricultural chemicals, 721
- Agricultural ecosystems, 14, 205
- Agricultural geology, 1104
- Agricultural geophysics, 1004
- Agricultural lands, 65
 crater lakes, adjacent gullies, and thick sand deposits, 68
 earthen levees and floodwalls, 66
 farm building and homes, damage to, 70
 growing crops and residue on erosion and deposition, 69
 land scouring and gully fields, 68–69
 levee repair, sand delta removal, and crater lake filling, 70
 levee saturation and topping, 66–67
 protected and unprotected agricultural lands, 70
 river bottomlands, agriculture, and levees, 65–66
 sand boils, 67–68
 sediment deposition in road and drainage ditches, 69–70
- Agricultural landscapes, 1345
- Agricultural Loess soil, 1373
- Agricultural management
 root functions, 72
 root growth, 72
 factors influencing, 72–74
- Agricultural production
 energy analysis, 2–3
- Agricultural revolution, 1128
- Agricultural soil management, 1914–1915
- Agricultural sustainability, 1
 mulch farming, 1500
- Agricultural technology, 1492
- Agricultural transformation, 1112–1113
- Agriculture
 and environmental sustainability
 agroecosystem biodiversity, 2277
 air and atmosphere, 2277
 soil, 2276
 water, 2276–2277
 expansion, 1311
 origins of, 1308
 sustainability, 1309–1311
- Agriculture–Natural Resources Conservation Service, 1410
- Agriculture–Soil Conservation Service, 471
- Agrobacterium tumefaciens*, 181
- Agro-ecological niche, 530
- Agroecosystems, 442
 distress syndrome, 1077, 1078
 sustainability, 1077
- Agro-ecozones, 1383
- Agroforestry, 291–292, 1223, 1224, 1383
 and intercropping, 896
 management, 662
- Agroforestry systems (AFSs), 1385–1386
 agrosilvopastoral system, 1379
 arboreal species, 1379
 carbon management index (CMI), 1380
 forage, 1379
 Gliricidia sepium, 1380
 leguminous species, 1379–1380
 objective, 1379
 planting in alleys, 1379
 production stability and diversity, 1379
- Agronomic biofortification of cereal grains, 934
 chickpea (*Cicer arietinum*), 936–937
 maize, 936
 oats (*Avena sativa*), 936
 rice, 934–935
 wheat, 935–936
- Agronomic management for better nutrition, 1447
- Agronomic yields, 1421
- Agrosilvopastoral system, 1379
- Air, soil, 85
- Air biofiltration, 203
- Airborne radiometric survey, 1874
- Airborne survey, 1873
- Airborne surveys, 1874
- Air-drying, 62
- Air-dry soil aggregates, 114
- Air-filled porosity, 1782
- Air-filled porosity (ϵ_a), 87, 258, 562
- Air permeability, 85–86
 apparatus, 87
 coefficient, 86
 measurement, 86
 acoustic methods, 87
 steady-state methods, 86
 transient methods, 86–87
 measurement methods, 87
 principle, 86
- Air pollution control law, 1825
- Alaska, boreal forest soils, 248
- Albedo, 89, 217
 factors affecting, 89–90
 measurement, 90
 of natural surfaces, 90
- Albeluviols, 409
- Albeluvisols, 392, 410
- AlcoaWorld Alumina, 1937
- Alfalfa (*Medicago sativa*), 225, 227, 359, 492
- Alfisols, 92, 105, 410, 507, 1161–1162, 2292–2293, 2371
 argillic horizon, formation of, 94–95
 clay illuviation process, 92–93
 pedological translocations/transformations of clay coatings, 93–94
 quantification, 93
 recognition, 93
 distribution of, 94
 geographical distribution and areal extent, 95
 Podzólicos Vermelho-Amarelo Eutrófico, 111
- Al–humus complexes, 99
- AL³⁺ ions, soil pH, 1695
- Alisols, 392, 409
- Alissolos, 381, 382
- Alkaline dissolution of silicates, 318
- Alkaline phosphatases
 (phosphomonoesterases), 739
- Alkalinity, subsoil sodicity, 2056
- Alkalinization, 560
- Alkalinolysis, 2560
- Allergens, transfer of, 245
- Alley cropping (AC), 1384
- Allobophora parva*, 236
- Allocthonous particles, 103
- Allophanes, 97, 136, 1469, 1793
 occurrence, 98
 properties, 99
 structure, 97
 allophanes, 97–98
 imogolite, 97
- Allophanic soils, 99
- Allorhizous root system, 72
- Alluvial plain, IGP, 1170–1172
- Alluvial soils, 65
- Alluvium, 1365, 1368
- Alocrisols, 393
- Al oxides
 gibbsite formation, 2005–2006
 minerals, 2005
 properties, 2006
- Alpine
 belt environment, 101
 soils in, 103
 grassland, 101
 meadow, 101
 soils, 103
 tundra, 101
- Alpine soils, 101
 aeolian dust deposition, 103
 soil organic matter and biological activity, 103
 weathering and soil formation rate, 103

- erosion and rejuvenation, 101
- genesis, 101
- and global climate change, 104
- land use and management, 104
- periglacial phenomena, 101–103
- Al toxicity, 2437
- Aluminol, 426
- Aluminosilicate, 97
 - clays, 56
- Aluminum (Al), 105
 - chemistry of, 105–106
 - occurrence and uses, 105
 - polymerization, 106
 - resistance and tolerance mechanisms in plants, 107
 - solution extracted from soils, 106
 - toxicity, 106, 1362
 - amelioration of, 107
 - in plants, 106–107
- Aluminum hydroxide [Al(OH)₃], 131
- Aluminum (hydr)oxides, 56
- Amaranth (*Amaranthus hypochondriacus*), 495
- Amazon basin soils
 - flood plains, 112–113
 - soil temperature and moisture regimes, 111
 - upland soils (terra firme), 111–112
- Amazonian forest, 208
- AMD. *See* Acid mine drainage (AMD)
- Ameliorants, 114, 2052
 - rate of supply of, 2054–2055
 - reclamation, without the addition of, 2052–2054
 - supply of, 2054
- Ameliorants for improving soil structure
 - gypsum, 115
 - lime, 115
 - organic matter, 115
 - and plant growth, 115
- Amelioration, 833, 1763
 - of compacted soils, 518–520
 - and management, 2057–2058
- Amendments, 824, 1766–1767
- American agricultural subculture, 2027
- American elm (*Ulmus americana*), 2573
- American Society for Testing and Materials (ASTM), 457
- American Society of Agronomy, 680
- AMF. *See* Arbuscular mycorrhizal fungi (AMF)
- Amide fertilizers, 912
- Ammonia (NH₃), 117, 1403, 1773
 - losses, 119
 - quality, 1850
- Ammonia-oxidizing bacteria (AOB), 1970
- Ammonia volatilization, 117–118, 886
 - emissions
 - animals and their wastes, 118
 - biomass burning, 119
 - cropping systems, 118–119
 - global significance, 119
 - measurement, 118
- Ammonification, 1850, 2173
- Ammonium, 175
- Ammonium ions (NH₄), 839
- Ammonium (NH₄⁺) plant release, 1422
- Ammonium nitrate, 921, 1600
 - fertilizers, 912
- Ammonium–nitrogen, leachate, 1323
- Ammonium phosphate, 1773
- Ammonium polyphosphates, 920
- Ammonium sulfate (sulfate of ammonia), 921
- Amoozemeter, 120, 121
 - comparison of methods for measuring K_{SAT}, 124–125
 - constant-head well permeameter method, 120
 - description of, 120–122
 - field data collection, 122
 - models for calculating K_{SAT}, 122–124
- Amorphous minerals
 - crystalline minerals, 1469
 - identification and characterization, 1471
 - physical and chemical properties, 1470
 - properties of soils containing, 1469–1471
 - varieties of, 1469
- Amphiboles, 1473–1474
- Amplitude, 1918
- Anaerobic degradation
 - municipal solid waste, 1322–1323
- Anaerobic processes, 126
 - anaerobic reactions in soils, 127–128
 - biological activity, 126–127
 - development of, 126
 - O₂ availability, 126
 - soil morphology, 128
- Ancient lynchets, 2342
- Ancient societies, 1123–1124
 - soil occupation or militaries, 1124
 - soil parasitism, 1124
 - soil pathogens, 1124
- Andic soils and cryoturbation, 136
- Andisols, 99, 159, 410, 1469–1470, 2136
 - classification, 130
 - Iceland, 134
 - andic soils and cryoturbation, 136
 - classification and soil characteristics, 134–135
 - freely drained soils, 136
 - physiography, 134
 - soil erosion, 136–137
 - Vitrisol classification perspectives, 136
 - wetland soils, 135–136
 - land use and management, 132
 - pedogenic factors, 130
 - physical–chemical properties, 130
 - chemistry, 131–132
 - mineralogy, 130–131
 - physical properties, 132
 - soil order, 97
- And land degradation, drylands and non-drylands, 608–609
- Andosols, 136, 137, 392, 409
- Anecic earthworms, 2214
- Anemia, 1446
- Angkor monuments, 1417
- Anguina tritici*, 1518, 1523
- Anhydrous ammonia, 919
- Animal productivity, 1862
- Animals, soil, 138
 - decomposition, 139
 - degree of presence, 139
 - impact of key
 - earthworms, 140
 - protozoa, 141
 - termites, 140–141
 - influence on soil functions
 - decomposition and nutrient cycling, 139
 - pest control, 140
 - water cycling, 139–140
 - soil biodiversity, 141
 - spatial distribution, 138–139
- Animal wastes, 184
- Anion and cation exchange capacities (AEC and CEC), 1471
- Anion exchange, 1243–1244
- Anion exchange capacity (AEC), 1578, 2005, 2435
- ANNs. *See* Artificial neural networks (ANNs)
- Annual cropping, 1421
- Annual precipitation, 1375, 1753
- Annual vegetation, 356
- Anoxia, 526
- Anoxic limestone drains, 38
- Anoxic or anaerobic soils, 126
- ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation), 814
- Antagonism, 41
- Antagonistic interactions, 1436
- Antecedent soil conditions, 517
- Anthropogenic carbon dioxide (CO₂) emissions, 315
- Anthropogenic disturbances
 - acid deposition, 1821
 - climate change, 1821
 - harvesting, 1820–1821
 - mining, 1821
 - site preparation, 1821
- Anthropogenic processes, 1113
- Anthropogenic soil disturbances, 790–792
- Anthropogenic soils (*Terra Preta do Indio*), 113
- Anthroscapes, 1416
- Anthrosols, 389, 392, 409
- Antibiotic-resistant genes, 245
- Antierosion, 549
- Antigorite, 1999
- Antimony (Sb), 146
 - abundance and sources, 146
 - chemistry, 146
 - mobility of, 146–148
 - physical and chemical properties, 147
 - toxicity and biological effect, 148
 - whether bioaccumulation, 148
- Antimony trioxide (Sb₂O₃), 146
- Antinutrients management, 1447
- Antioxidants, 1446
- Ants, 150, 205, 237, 878
 - microtopography, 150–151
 - nests, 150
 - in soil
 - biota, effects on, 152
 - heterogeneity of physical and chemical properties, 151
 - turnover, 151–152
 - water relations, 152
- AOB. *See* Ammonia-oxidizing bacteria (AOB)
- Aphaenogaster barbigula*, 151
- Apomixis, 245
- Aquents, 728
- Aqueous NH₃, 1801–1802
- Aqueous secondary waste, 1323
- Aquepts, 1152–1153
- Aquifers, 120, 900, 1990, 1992

- Arable cropping system, 289–290
 Arable crops, 72
 Arab soil classification, 370
 Aragonite, 1362
 Arboreal species, agroforestry system, 1379
 Arbuscular mycorrhizal fungi (AMF), 1504, 1506–1507
 Arbuscular mycorrhizas, 1502
Archaea, 175
 Archaeology, 154
 field studies, 154–155
 laboratory soil characterization, 156
 soil morphology, 155–156
 soil science in, 154
 Archeopedology, 154
 Areias Quartzosas, 380
Arena nobilis, 396
 Arenosol, 392
 Arenosols, 409
Argilla communis, 396
 Argillan, 1360
Argilla vitriolacea, 17
 Argillic horizon, formation of, 94–95
 Argissolos, 381, 382
 Arid aeolian plain, IGP, 1173
 Aridic (torric) moisture regime, 158
 taxonomy suborders, 159
 Aridisols, 158, 159, 410, 1162
 Aridosols, 389
 Arid region soils, 1211–1212
 Arid soils, 157
 ecological significance, 159
 river floodplains, 157
 types, 158–159
 on uplands, 157–158
 Armoring, 38
 Aromaticity, 2114
 Aromaticity index, 2157
 Arsenic, 1759
 Arsenic (As), 161, 1898
 bioavailability and toxicity, 161
 contamination in groundwater
 China (including Taiwan), 162–163
 India, 162
 Indonesia, 163
 Lao PDR, 163
 Myanmar, 163
 Nepal, 163
 Pakistan, 163
 Thailand, 163
 Vietnam and Cambodia, 163–164
 extent and severity
 Bangladesh, 161–162
 West Bengal, 162
 in soils, 164–165
 Art and soil, 168
 Hieronymus Bosch and his disciples, 169
 land art movement, 169
 as profile, 168–169
 as surface, 168
 Artesian, 1992–1993
Arthrobotrys nematode-trapping fungus, 1520
Arthropleona, 448
 Arthropods, 140
 Artificial barriers, 2611
 Artificial neural networks (ANNs), 1684
 Ascochyta blight (*Ascochyta rabei*), 229
 Ascomycotina, 979
 Ashbed effect, 927
 Asian cropping systems, 940
 Asparagus (*Asparagus officinalis* L.), 241
 ASS. *See* Acid sulfate soils (ASS)
 Assam (Brahmaputra) valley, IGP, 1174–1175
 Assessment, quality
 by interpolation techniques, 1844–1847
 biological properties, 1844
 compartmentalized framework, 1845
 computer-assisted production, 1844
 definition, 1844
 geographical information system (GIS), 1845
 indicators, 1844–1845
 mapping accuracy, 1845
 methods, 1845
 pedotransfer functions (PTFs), 1844
 science-based techniques, 1845
 wet combustion method, 1844
 by spatial interpolation
 decision trees, 1846
 geostatistical analyses, 1846
 interpolation efficacy, 1847
 remote sensing, 1846–1847
 sampling, 1846
 Associative N₂ fixation, 222
 Assyrian civilization, 168
 Astigmata, 1489
 Astigmatid mites, in soil food webs, 1627
 ASW. *See* Available soil water (ASW)
 Atmosphere–soil interface, 823
 Atmospheric C, 1610
 Atmospheric carbon (C), 307
 Atmospheric carbon dioxide (CO₂), 11, 1212
 Atmospheric derivation, 1204
 Atmospheric N deposition, 1542
 Atmospheric pressure, 672
 Atmospheric stability, 2630
 ATP extraction, 1431
 Atrazine, 1717, 1774
Atriplex vesicaria shrubland, 151
Atta cephalotes, 151
 Atterberg. *See* Consistence
 Atterberg limits, 1740–1741
 application, 173–174
 classification of soils, 173
 factors influencing, 172–173
 index of consistency (I_c), 172
 index of plasticity (I_p), 171–172
 index of shrinkage (I_{sh}), 172
 limitations, 174
 liquid limit, 171
 mechanical properties of soils, 173
 plastic limit, 171
 shrinkage limit, 171
 Attribute data, 1314
 Attributes and constraints, soil, 985
 Augite, 1473–1474
 Australian, 376
 Australian soil classification, 376
 Autocorrelation, 1812
 Autoregulation, 233
 Auto-steering systems, 477, 478
 Autotrophic fixation, 711
 Autotrophic nitrification, 1849
 Autotrophic respiration, 1928
 Auxiliary processes, 1803
 Available soil water (ASW), 2045
 Available water capacity (AWC), 1920, 2531
 SOM
 effects, conceptual model, 2119
 environmental implications, 2119–2120
 sensitivity, 2117–2119
 Available water-holding capacity (AWHC), 339
 AWC. *See* Available water capacity (AWC)
 Aylmore, 1073
Azadirchta indica, 827
Azolla, 233
 Azolla and anabaena, 895–896
Azotobacter, 232
Azotobacter chroococcum, 220
Azotobacter vinelandii, 221
B
 Bacillariophyta, 238
Bacillus subtilis, 1970
 Backett's quantity and intensity
 relationship, 1794
 Backslope, 1327
 Bacteria, 138, 175
 bacterial diversity, 178–180
 bacterial life in soils
 bacterial functions in soil–plant systems, 181–182
 soil microenvironment and ecological interactions, 180–181
 culture-based microbiology, 176
 DNA-based characterization of communities
 without culturing, 178
 DNA-based classification of cultured isolates, 177–178
 microbiology, 176
 process-based microbiology, 176–177
 respiration, 177
 in soil food webs, 1624–1625
 Bacterial biomass, 1429
 Bacterial cells, 9
 Bacterial diversity, 178–180
 Bacterial isolates, 176
 Bacterial life in soils
 bacterial functions in soil–plant systems, 181–182
 soil microenvironment and ecological interactions, 180–181
 Bacterial phylum, 178
 Bacterivorous nematodes, 1525
Bacteroidetes, 181
 Bait minnow, 275
 Balanced fertilization, 73, 257
 Balanced nutrition, 1424
 Bamboo plantations, 1384
 Bananas, 118
 Barite, 2240
 Bark from conifers (soft wood), 1056
 Barley (*Hordeum vulgare* L.), 362, 530
 Basalt weathers, 130
 Baseline steady-state assumption, 1611–1612
 Baseline value, 1853
 Base saturation (BS), 1184, 1578
 Basidiomycetes, 1502
 Basidiomycotina, 979
 Basin flood irrigation, 783
 Bassanite, 1068

- Bauxite, 1900
 Bayer process, 105
 Bayesian maximum entropy approach, 1678–1679
 Bean (*Phaseolus vulgaris*), 1283
 Bedrock (Precambrian shield), 963
 Bedulo (*Ficus clavata*), 734
 Beech (*Fagus* spp.), 185
Belemnitell americana, 336
 Belowground management, structure
 aeration, 2210
 crop yield, 2209
 improvement and management, 2209–2210
 strength, 2209–2210
 water, 2210
 Bench terraces, 748–750, 1988, 2344
 Bentonite backfill, 1993
 Benzene, toluene, ethylbenzene, and xylenes (BTEX), 1719
 Benzenepolycarboxylic acids (BPCA), 1455
Betaproteobacteria, 181
 Bicarbonates, 10
 Bidirectional reflectance distribution function, 90
 Bioaugmentation, 202
 Bioavailability, 1392
 persistent organic pollution
 enhancing, 1773
 sorption and sequestration, 1772–1773
 Bioavailable micronutrients, 1447
 Bioavailable nutrient, 1588
 Bioavailable water, 192
 Bio-banking, 1356
 Biochar, 184–185, 256, 296, 1040
 effect on reducing non-CO₂ soil greenhouse gas emissions, 195
 meta-analysis of, 195
 plant yield and nutrition, 195
 soil biota, 196
 soil carbon and fertility, 193
 biochar C stability, 193–195
 C cycling effects, 193–195
 soil C sequestration, 193–195
 on soil chemical properties, 185
 soil fertility in agricultural systems, 195
 soil physical effects, 196
 on soil physical properties, 185–187
 on soil physics, 195
 soil quality, 186, 189–192
 See also Pyrogenic carbon (PyC)
 Biochemical oxygen (O₂) demand, 1323
 Biocide Directive, 1825
 Biodegradation, 128, 1323
 processes, 198
 Biodiversity, 103, 141, 159, 205–207, 2572
 hydrology, 2572
 landscape perspective, 2572–2574
 markets, 1355
 mulch farming, 1499
 peatlands, 1672
 wetland fauna, 2574
 wetland flora, 2572
 Biodynamic agriculture, 707
 Bioenergy, 1458, 1460, 1461
 crops, 210
 benefits of, 210–211
 C sequestration, 211
 influence at temporal scale, 210
 Biofertilizers, 257, 1223
 Biofortification, 1447
 Biofortified seeds, 1443
 Biofuels
 versus agricultural soils, 213
 biofuel production, 214–215
 conserving soil, 213–214
 production, 214–215
 sustainability, 2231
 Biogenic opaline phytoliths, 1469
 Biogeochemical cycling
 C-cycling (glycosidases), 738–739
 N-cycling (amidohydrolases), 739
 P cycling (phosphatases), 739
 S-cycling (arylsulfatase), 739
 Biogeochemistry of sulfide mineral
 formation, 26
 oxidizable organic carbon, 26
 reactive Fe, 27
 reducing/saturated conditions, 26–27
 sulfate, 26
 sulfate-reducing bacteria, 27
 Biointensive agriculture, 707
 Biological activity, pore system, 1785
 Biological carrying capacity (BCC), 207
 Biological degradation, 553–554
 crop management
 crop rotation, 555
 depletion of soil organic matter, 555–556
 fallowing, 555
 stubble burning and grazing, 555
 factors affecting
 chemicals on soil, 554–555
 soil management, 554
 soil tillage, 554
 soil resilience, 556
 See also Degradation
 Biological nitrogen fixation (BNF), 220, 1552–1553
 agricultural systems, 223
 contributions by different N₂-fixing organisms, 223
 in farmers' fields, 224–225
 inputs of fixed N by legumes, 223–224
 management, impact of, 225
 associative N₂ fixation, 222
 ATP, requirement for, 221
 biochemistry of N₂ fixation, 221
 cell-free N₂ fixation, 221
 environmental factors, 232
 nutrient limitations, 233
 regulation at ecosystem scale, 234
 regulation at physiological scale, 233
 toxicities, 233
 factors affecting, 227
 legume yield, 227–228
 legume agronomy, 228
 pest control, 229
 plant density and row spacing, 229
 planting time, 228–229
 rhizobial inoculation, 228
 soil constraints and plant nutrients, 229–230
 soil water, 228
 species/cultivar choice, 228
 microbial communities, 1435
 %Ndfa and, 230
 land use and crop sequence, 230
 tillage, 230
 N₂-fixing organisms, 232
 N input processes, 1542–1543
 nitrogenase
 control of, 222
 genetics of, 222
 purification of, 221
 tertiary structure, 222
 substrates other than N₂, 221
 symbiotic and free-living N₂ fixers, 220–221
 types, 232
 Biological oxygen demand (BOD), 923
 Biological soil crusts (BSCs), 216
 albedo, 217
 resistance and resilience, 218–219
 soil fertility and vascular plants, 217–218
 soil physical structure and stabilization, 216–217
 soil–water relations, 217
 weathering, 216
 Biological wastewater treatment processes, 199
 Biomass, 2517
 burning, 1501, 1556–1557
 fuels, 210
 off-farm, 194
 overexploitation, 1530–1532
 production cycle, 1532
 productivity, 1123
 removal, 1820
 Biomass-derived black C, 296
 Biomes, 150
 boreal forest biome, 1023
 desert biome, 1023
 polar biome, 1021–1023
 soils, 1020–1021
 temperate grassland and forest biome, 1023
 tropics biome, 1023
 Bio-oil fraction, 190
 Biopores, source, 1388
 Bioreactors, soil, 203
 Bioremediation
 processes, 198–199
 technologies, 202
 deep-sea bioremediation, 203
 engineered biopiles, 202
 in situ subsurface bioremediation, 202
 intrinsic remediation, 202
 landfarming, 202
 monitored natural attenuation, 202
 of shorelines, 203
 soil bioreactors, 203
 soil composting, 202
 Biosequences, 2356
 Biosolids, 299, 1407–1408
 manures, composts, and by-products, 1224
 Biostimulation, 201–202
 Biota, 235
 animals, role of, 236–238
 disease-suppressive soils, 239–240
 Gliocladium spp., 240
 organic matter amendments and disease suppression, 240–241
 Pseudomonas spp., 240
 Trichoderma spp., 240
 plants, role of, 235–236
 role in soil food webs

- Biota—
 bacteria, 1624–1625
 collembola, 1626–1627
 fungi, 1625
 macrofauna, 1627–1628
 mesofauna, 1626
 microfauna, 1625–1626
- Biotechnology, 243
 applications and prospects, 243–245
 definition, 243
 dilemmas and concerns, 245
 antibiotic-resistant genes, 245
 gene escape and genetic pollution, 245
 herbicide and insecticide resistance, 245
 social, moral, and ethical issues, 245
 transfer of allergens, 245
- Biotransforming enzymes, 199
- Bioturbation, 2373–2374
- Birch (*Betula pubescens*), 623
- Birch seedlings (*Betula pendula*), 143
- Birnessite, 2006
- Black box approach to soil microbiology, 177
- Black carbon (BC), 189
 See also Pyrogenic carbon (PyC)
- Black Chernozem, 386
- Black matrix, 454
- Black soils, 452
 and their intergrades, 2291–2292
- Blanket peats, 1107
- Blatong pantad, 403
- Blended fertilizers, 921–922
- Blocky structure, 2217
- Blowout holes, 70
- BNF, leguminous crop rotations, agroforestry, 1223–1224
- Bog, 2568
- Boggy-meadow soils, 415
- Bonding energy, 61
- Boreal forest, 247
 biome, 1023
 and coniferous forest, 963
 environmental conditions, 247
 global ecological significance, 250
 land use, 249
 soil classification, 247
 soil formation, 247–248
 soils
 Alaska, 248
 Canada, 248
 Central and East Siberia, 248
 European Russia, 248
 Scandinavia, 248
 West Siberia, 248
- Boreal podzol, 1748
- Boric acid, 355
- Boron, 2469
- Boron
 correcting deficiency, 253–254
 macronutrients, 252
 physiological role, deficiency, and toxicity symptoms, 253
 soil properties, 252
 soil reaction and pH, 252
- Bosque Chaquenho, vegetation, 1646
- Bottomland, 2568
- Boundary conditions and nutrient entry, 1591–1592
- Bowen ratio, 858–860
- Bowen ratio energy balance (BREB), 859–860
- Box-model. *See* Compartment-model approach
- Bozkır, 403
- BPCA. *See* Benzenepolycarboxylic acids (BPCA)
- Brachiaria pasture, 208
- Brassica* crops, 1039
- Brassica rapa*, 1324
- Brazilian classification systems, 379–382
- Brazilian semiarid
 land use in, 1377–1378
- Brazilian soil classification system, 111
- Brazilian Soil Science Society (SBSCS), 379
- Brazil nut (*Bertholletia excelsa*), 1283
- Breaches, 66
- Brick making, 255
 economic impacts, 256
 effects on soil properties and nutrients, 255–256
 global scenario, 256
 management strategies, 256–257
- Brick soils, 413
- Broad-base terraces, 2343
- Broadcasting, 904
- Brooks–Corey bubbling pressure, 1189
- Brooks–Corey equation, 1978
- Brown Andosols (BA), 134
- Brown forest soils in tropical climate, 2292
- Brunisolic soil, 385
- BSC. *See* Biological soil crusts (BSCs)
- Bubble tube, 121
- Buckwheat (*Fagopyrum esculentum*), 495
- Buckwheat (*Triticum aestivum* L.), 362
- Buffalo grass (*Buchloe dactyloides*), 828
- Buffers
 capacity, 1580, 1897
 nutrients in soil solution, 1577
 power, 1732–1733
 and protected environment, 721
 strips
 on flow hydraulics, 1024–1025
 on sediment transport, 1025–1027
 strips and riparian strips, 829–830
- Bulk chemical composition, SOM, 2126–2127
- Bulk density, 258, 516, 561, 597, 680, 1414, 2475
 determination
 direct methods, 258–259
 radiation methods, 259
 measurement
 aeration status, 260
 gravimetric data to volumetric data, 260
 mass of large volume of soil, 260
 soil structure and soil quality, 259
 and porosity, 259
- Bulking agents, 1407
- Bulk soil resistance, 861
- Bun*, 1383
- Bunch planting, 261–263
- Bundle of rights, law, 1355
- Buoyancy, 296, 2630
- Buoyancy-driven migration, 281
- Burkholderia* spp., 181
- Burning
 vs. mulching, 508–509
 of savannas, 1039
- Burrowing mammals, 2374
- Burrow types earthworms, 2214
- By-products, industrial, 1178
 coal combustion by-products, 1180
 construction wastes, 1180
 land application, 1180–1181
 metal refining and processing, 1178
 mineral processing, 1178–1179
 pulp and paper manufacture, 1180
 regulatory agencies, 1181–1182
- C
- Caatinga, 1377
- Cabbage (*Brassica oleracea* L.), 365
- Caenorhabditis elegans*, 1518
- Calcareous crusts, 1756
- Calcareous sodic soils, 2051
- Calcareous soils, 1579
- Calcic horizons
 petrocalcic horizons, 1691
- Calcification, 264
 components, 264
 manifestations of, 265–266
 mechanisms, 264–265
- Calcined dolomite, 1393
- Calcined magnesite, 1393
- Calcisols, 392, 409
- Calcite (CaCO_3), 335, 711, 1450
 soil pH, 1696
- Calcitic limestone, 1363
- Calcium ammonium nitrate (CAN), 921
- Calcium (Ca), 255, 267
 Ca/Mg ratio, 268
 deficiency, 1362
 depletion, 959–961
 forest ecosystems, 961
 exchangeable, 267
 soil ameliorant
 amelioration of acidic soils, 268
 amelioration of sodic soils, 268
 soil solution, 267–268
 uptake by plants, 268–269
- Calcium (Ca)-dominated clays, 2024
- Calcium carbonate (CaCO_3), 190, 1213
 filling, 156
 liming materials, 1362–1363
- Calcium (Ca)-rich water, 905
- Calcium hydroxide, 115
- Calcium oxide (CaO), 115, 1363
- Calcretes, 1691
- CALDEP, 336
- Calibration, 2321
- Calliandra calothyrsus*, 827
- Cambisols, 392, 409
- Cambissolos, 381, 382
- Camponotus punctulatus*, 150
 mounds, 150
- Canada, boreal forest soils, 248
- Canadian classification systems, 383
 correlation with other soil, 387–388
 outline, 385–387
 pedon, 383
 soil horizons, 383–384
 taxa, defining, 384–385
- Canadian Land Suitability Rating System, 1863
- Candidate phyla, 179
- Canola (*Brassica* sp.), 243, 530

- Capacitance devices, 2476–2477
- Capacity factor, 1714
- Capacity of soil legislation, 1353
- Capillary rise/leaching, sulfur, 2243
- Capping, 34
- Carbonates, 331, 1213
- and clay accumulations, 157
 - coarse fragments, 339
 - formation, 333
 - occurrence in soil, 332–333
 - PCs, 335–336
 - formation, 336–337
 - morphology of, 336
 - stable isotopes, calculation using, 336
 - properties, 333–334
 - redistribution, 1421
 - structure and composition, 331–332
 - and sulfate minerals, 1478–1479
 - water and, 339
 - bedrock and petrocalcic horizons, 340–341
 - dispersed carbonates, 339–340
 - formation, 339
 - indurated carbonate nodules and clasts, 340
- Carbonation, 317
- Carbon (C)
- assessment, diffuse reflectance, 270
 - chemometrics, 270–271
 - potential for soil analysis, 272
 - simultaneous determination of multiple analytes, 271–272
 - soil calibration example, 272
 - auditing, mapping soil carbon and, 1409
 - compositional and isotopic analysis
 - laser ablation aerosol mass spectrometry (LA-AMS), 2075–2076
 - laser ablation isotope ratio mass spectrometry (LA-IRMS), 2073–2075
 - nanospray desorption electrospray ionization (nano-DESI), 2076–2077
 - credit suppliers, 80
 - cycle, 175
 - carbon components, 2129
 - C fluxes, 2129–2130
 - soil C fluxes and global change, 2130–2131
 - farming and economic aspects, 78
 - early actor risk, 2079
 - energy prices, 2079
 - GHG emission restrictions and money, 2078
 - Kyoto Protocol, 2079
 - land use history, 2079
 - risk and the no-till panacea, 2080
 - saturation/new equilibrium and volatility, 2079
 - soil carbon, 2079
 - tillage, 2079–2080
 - transaction costs, monitoring, and verification, 2079
 - financing, 194
 - gases quality
 - carbon dioxide, 1848–1849
 - methane, 1849
 - maps, 1410
 - mineralization, 103, 152
 - natural isotopes, 2226
 - pools, 1756
 - saturation index, 1410
 - total mass, 1675
- Carbon (C) and nitrogen (N) cycling (C/N cycling), 538
- anthropogenic factors disrupting, 541–542
 - bacteria and fungi control, 538–539
 - diversity in organisms and substrates, 539–540
 - natural *versus* anthropogenic controls, 540–541
- Carbon (C) losses from soil, SOM
- energy and full C accounting, 2151
 - erosion processes, 2147–2148
 - long-term field experiments, role, 2148
 - mineralization processes, 2147
 - soil C sequestration
 - global importance, 2148
 - mechanisms, 2148–2151
 - in 21ST century, 2151
 - worldwide land use and management impacts, 2149–2150
- Carbon dioxide (CO₂), 49, 80, 85, 126, 138, 191, 194, 210, 294, 620, 778, 1036, 1048
- capture, 1798–1800
 - concentration, 279
 - quality, 1848–1849
 - soil pH, 1698
- Carbonic acid (H₂CO₃), 12, 2563
- Carbon management index (CMI), 1380
- Carbon map, Africa
- biological carbon (C) cycle, 2059
 - ecosystems, 2062–2063
 - global C cycle, 2063
 - measurement, 2060
 - problems in measuring, 2060
 - and soil functions, 2060
 - soil organic carbon (SOC), 2060–2062
 - soil organic matter and C, 2059–2060
 - soil type, 2062
 - stocks calculation, 2060–2061
- Carbon monoxide (CO), 191
- Carbon nanotubes (CNT), 1514
- Carbon sequestration, 76, 194, 211, 274, 279, 294, 1409, 2576
- aquaculture activities, 274–275
 - creation of wetlands to, 2578
 - defined, 279–281
 - in Ethiopia
 - climate and ecological conditions, 2066
 - deforestation, 2066, 2068
 - degraded lands, rehabilitation, 2070–2071
 - erosion, 2066
 - location, 2067
 - loss of SOM in, 2069–2070
 - Montane forests, 2066
 - organic matter pools, 2067
 - soil degradation in, 2068–2069
 - soil organic matter (SOM), 2066
 - soil quality (health), 2066
 - geologic, 281–282
 - human activities, impact of, 2577–2578
- Iceland, 284–285
- afforestation, 286
 - geological processes, 286
 - land use, cooperation with farmers, and funding, 285–286
 - potential, 285
 - restoration of wetlands, 286
- India, semiarid regions of, 299–300
- carbon management strategies, 303–304
 - critical carbon input, 303
 - nutrient management, 300
 - productivity enhancement, 300–303
 - rainfed cropping systems, 300
 - soil carbon stocks, 300
- nutrient management, 288–289
- in agriculture, 289
 - agroforestry, 291–292
 - arable cropping system, 289–290
 - global C potential, 288–289
 - in grassland and fodder crops, 290–291
 - horticultural crops, 291
- OC concentration in sediment, 276
- in pedosphere, 279–281
- primary production, 2576
- rate of, 276–277
- and respiration, 1929–1930
- sediments, 275–276, 294
- core analysis, 275
 - environmental impacts, 296–297
 - management strategies, 297
 - marine sediments, 296
 - terrestrial, 294–296
- in soils, 2576–2577
- urban ecosystems, 307–308
- climatic effects on, 308–310
 - hidden costs, 311–312
 - influence of soil properties, 310–311
- Carbon sequestration-enhancing practices, 81
- Carbon sink capacity, 315
- SIC sink capacity, 317
 - base cations availability, 317–318
 - neutral and alkaline pH, 317
 - SOC sink capacity, 315
 - clay mineralogy, 316–317
 - soil depth and bulk density, 317
 - soil particle size distribution, 316
 - soil structure, 315–316
- Carbon trading, 1355
- Carcass composting, 1407
- Care of soil, 342
- multistakeholder engagement, 342–343
 - building land ethic, 343–344
 - landcare approach, 343
- Carl Woese's pivotal discovery, 175
- Carnivores, 138
- Carps, 275
- Carya aquatica* (water hickory), 2573
- Casagrande Plasticity Chart, 1740
- Cashew (*Anacardium occidentale*), 1283
- Cassia siamea*, 827
- Castanopsis fissa*, 1323, 1325
- Casuarina* spp., 827
- Catalyst-filled membrane tubes, 1803
- Catastrophic acidification, 20
- Catchment area
- slopes, 2516
 - treatments, 2361
- Cat clay soils, 17
- Catena, 415, 1328, 1339
- Catfish, 276
- Cation exchange capacity (CEC), 92, 184, 190, 347, 410, 426, 880, 1054, 1109, 1393, 1476–1477, 1646, 1665, 2435, 2442

- Cation(s)
 and anion exchanges, 1578–1579
 exchange, 1241, 1427
 capacity, and base cation saturation, 1241–1242
 equilibria and cation selectivity, 1242–1243
 nature and origin of, 1241
 plant nutrition, 1243
 saline and alkali soils, amelioration of, 1243
 soil development and acidification, 1243
 plasticity, 1741
 valence, 824
- Cattle feeders, 81
- Caustic soda (NaOH), 9
- C balance, 104
 non-uniform effects, 2143
- CCHP. *See* Amoozemeter
- C costs, mycorrhiza of forest ecosystems, 1503
- CCs. *See* Cover crops (CCs)
- Cd contamination, 1760
- CEC. *See* Cation exchange capacity (CEC)
- Cell-free N₂ fixation, 221
- Celtis laevigata* (sugarberry), 2573
- Cementation, 1417, 1744
 materials, 1659
- Cement–soil mortars, 1417
- Central and East Siberia, boreal forest soils, 248
- Central uplands, 1157
- CENTURY SOM model, 2189
- Ceramics
 advanced ceramics, 1418–1419
 pottery, 1418
- Cereal crops, 934
- Cereal–legume rotation, 1424
- Cernovitoviella atrata*, 724
- Cerrado, 345
 biomass production, constraints to, 346
 climate, vegetation, and geology, 345–346
 principal soils, 346
 sustainable land use, 346–347
- Certified organic crops, 1600
- Cesium-137, 771, 801–802
- C footprint, no-till, 1379
- Chaetomium globosum*, 980
- Chambers, 859
- Channel catfish, 275
- Chapadas, 346
- Charcoal, 2316
See also Biochar; Pyrogenic carbon (PyC)
- CH₄ conversion factor (MCF), 1401–1402
- Checkerboard model, 2197
- Chelating agents, phytoremediation, 1722
- Chelatobacter heintzii*, 1879
- Chelicerata, 1488
- Chemical ameliorants
 calcareous sodic soils, 2051
 gypsum, 2051
 lime, 2051
- Chemical bonding, 353
- Chemical composition, 348
 soil-forming factors, 349
 biota, 351
 climate, 350
 parent rock, 349–350
 time, 350
 topography, 350–351
- soil-forming processes, 348
 additions, 348–349
 losses, 349
 transformations, 348
 translocations, 348
- Chemical degradation, 558
 soil pH and development of soil acidity, 558–559
 alkalinity, 559–560
 depletion of soil organic matter, 560
 salinity, 559
See also Degradation
- Chemical extractants, 2286
- Chemical fertility, 558
- Chemical fertilizers. *See* Mineral fertilizers
- Chemical flows, absorb, buffer, and transform, 1868
- Chemical indicators, 1862
- Chemical looping, 1802–1803
- Chemical oxygen (O₂) demand, 1323
- Chemical protection, 1632
- Chemical stabilization, 2287
- Chemical weathering, 1480
 processes, 997–998
- Chemisorption, 353, 1427–1428
 chemical bonding, 353
 concentration, 354
 desorption, 355
 ions, behavior of, 355
 pH effects, 354–355
 reacting ions, 353–354
 soil components, 353
 specificity and sorption, 353
 time, effect of, 355
- Chemistry–plant nutrition, soil, 903, 1700
- Chemodenitrification, 1563, 1849
- Chemometrics, DRS, 1887
- Chernossolos, 381, 382
- Chernozems, 356–357, 392, 409
 composition and properties, 357–359
 dehumidification, 359–361
 mineralogy, 357
 soil, 385
- Chicago Climate Exchange (CCX), 77, 2078
- Chickpea (*Cicer arietinum*), 224, 228, 936–937
- China Organic Food Certification Center (COFCC), 1619
- Chinese cabbage (*Brassica rapa* L.), 363
- Chinese classification systems, 389
 alpine region soils, 390
 Ancient China, 389
 Anthrosols, 389
 Aridosols, 389
 diagnostic characteristics, 389
 diagnostic horizons, 389
 Ferrosols, 389–390
 modern China, 389
 nomenclature and mapping, 390
- Chloride, 2469
 concentrations in plants, 364
- Chlorine, 362
 biochemical and physiological functions in plants, 363–365
 chloride in soil, 362
 interaction with other nutrients, 365
 plant yield and quality responses, 362–363
- Chlorites, 424, 1478
- Chloroaromatics, 199
- Chlorobi, 179
- Chloroflexi, 179
- Chloroform fumigation
 incubation and extraction methods, 1429–1430
 microbial communities, 1434
- Chlorophyll, Mg in plants, 1392–1393
- Chloroplast transformation, 245
- Christiansen's uniformity coefficient, 1259
- Chromium, 1898
- Chronic acidification, 14
- Chronosequence modeling, 2187
- Chronosequential sampling, 881
- Chute spillways, 751
- CIA. *See* Compound identification algorithm (CIA)
- Cinnamomum camphora*, 1325
- Cinzeno, 380
- Clarias catfish, 275
- Classical gully
 in China, 1060
 erosion, 1061
- Classical molecular mechanics methods, 2166–2167
- Classification systems, 367, 375–376
 Arab, 370
 Australian, 376
 Brazilian, 379–382
 Canadian, 383
 correlation with other soil, 387–388
 outline, 385–387
 pedon, 383
 soil horizons, 383–384
 taxa, defining, 384–385
- Chinese, 389
 alpine region soils, 390
 Ancient China, 389
 Anthrosols, 389
 Aridosols, 389
 diagnostic characteristics, 389
 diagnostic horizons, 389
 Ferrosols, 389–390
 modern China, 389
 nomenclature and mapping, 390
- communication and, 368–369
- French
 history, 391
 RP and WRB, 391–393
- historical developments, 395–397
- Indian, 398
 climate-based soil classification, 399
 fertility-based classification, 398
 revenue system, 399–400
 suitability of crops, 398–399
- indigenous, 402
 perceptual approach, 403–404
 physical dimension, 402–403
- international and national, 368–369
- major, 406
 WRB for soil resources, 408–410
- Moroccan vernacular soil names, 370–374
- national soil classification systems, 376
- Netherlands, 412
 Dutch system, 413–414
 history, 412–413
 nomenclature, 414

- New Zealand, 376–377
 properties, 367–368
 Russian, 415–417
 single or general-purpose, 368
 South Africa, 418
 binomial system, 418–419
 taxonomic system, 419–420
 Tugela basin classification, 418
- Clay, 157
 coatings, 93, 94, 1417
 content, 1662, 1741
 content and clay type, 1741
 floculation, 824
 fraction, 1469
 illuviation process, 92–93, 94, 1030, 1359–1360, 1421
 pedological translocations/transformations of clay coatings, 93–94
 quantification, 93
 recognition, 93
 microstructures, 504
 mineralogy determines, 56
 particle deposition, 1360
 percentage, 2440–2442
 swelling, 428
 tactoids, 56
- Clayey soils, 89
- Clay minerals, 421, 1629, 1662
 charge properties
 adsorption of plant nutrients and contaminants, 427–428
 clay swelling, 428
 floculation processes, 428
 high-activity and low-activity clays, 427
 origins of charge, 426–427
 surface charge density, 427
 classification, 422
 hydroxy-Al interlayered vermiculites and smectites, 424
 1:1 layer minerals, 422–423
 2:1 layer minerals, 423
 2:1:1 layer minerals, 424
 mixed minerals, 424
 composition, 1954
 contents, 1417–1418
 formation, 1755
 identification and quantification, 425
 isomorphous substitution, 421–422
 and organic molecules, 1631
 structure, 421
 weathering and alteration
 mechanisms and rates, 433–434
 mineral resistance to, 430–431
 mineral weathering pathways, 431–433
 practical implications, 433–434
 thermodynamic formulations, 433
- Clay–organic matter complexes, 504
- Claypans, 140
- Clay-rich Paleosols, 1373
- Cleistogamy, 245
- Climate, 157, 435–437
 condition, Loess Plateau, 1373
 and hydrology, 790
 and landform, 2442
 stabilization, 280
- Climate change, 620
 agroforestry system, 1379–1380
- analytical aspects
 gas chromatography, 440
 micrometeorological methods, 440–441
 nonisotopic tracer methods, 441
 photo-acoustic-infrared detector (PAID), 440
 ultra-large chambers with long-path infrared spectrometers, 441
- anthropogenic disturbances, 1821
- no-till system and green manure, 1378–1379
- nutrient management, 1570–1571
- sampling, 119, 438
 chamber techniques, 438
 closed and open chambers, 438–439
 manual or automated sampling and analysis, 438–440
- soil carbon sequestration, 442
 keeping carbon in the soil, 443–444
 role of man, 442–443
 role of nature, 443
- Climate-resilient soils, 356
- Climatical effects, 620
- Climatic deserts, 657
- Cloning, 243
 microbial communities, 1434
- Clorophyta, 237
- Clostridium pasteurianum*, 220
- Coal bed methane (CBM), 995
- Coal-burning electric utilities, 12
- Coal combustion by-products, 1180
- Coal mines in Pennsylvania, 8
- Coal reactivity analysis, 1454
- Coal seams, 995
- Coarse-grained soil, 456, 1417
- Coarse-textured sandy soils, 2218–2219
- Coastal ecosystems, 295
- Coastal wetlands, reclamation, 37
- Cobalt (Co), 445, 1729
 deficiency, 446
 forms of, 445
 plant availability of, 445–446
- Cocomposting of biosolids, 923
- Coefficient of consolidation (c_v), 474
- Coefficient of linear extensibility (COLE), 2010
- Coefficient of permeability, 474
- CO₂ emission, conceptual aspects on, 2235
- CO₂-enhanced root growth, 443
- COFCC. *See* China Organic Food Certification Center (COFCC)
- Coffee colored waters, 112
- Cognettia sphagnetorum*, 724
- Cohesion, 62
 soil plasticity, 1740–1741
- Cohesive stress, 171
- Coir, 1056
- Cökek, 403
- Cold-hardy insects, 1217
- COLE. *See* Coefficient of linear extensibility (COLE)
- Collembola, 206, 448
 abundance, 450
 distribution, 449–450
 diversity, 450
 economic importance, 448–449
 effects on soil structure, 450
 longevity, 449
 morphology, 448
- role in decomposition, 450
 in soil food webs, 1626–1627
 species of, 449
 swarming, 450
- Colluviosols, 393
- Colonialism, legacy of, 1309
- Colony-forming units (CFU), 40
- Color, soil, 90, 452
 causes for, 453–454
 characteristics, 454
 conditions indicated, 454–455
 determinations, DRS, 1886
 measuring, 452–453
 soil environments associated with Fe oxides, 453
- Colorado Department of Health (CDH)
 regulations, 1408
- Colorado River, United States, 2553
- Columnar, structure, 2216
- Commercial forests, 443
- Comminution, 1489
- Commission of Pedology and Cartography of Soil of France, 370
- Commodity price support, 594
- Common vetch (*Vicia faba*), 220
- Community legislation, EU protection
 policies, 1824
- Community-supported agriculture (CSA), 2274
- Compactability, 517
- Compaction, soil, 52, 456, 791, 921, 1821
 effect, 1460
 management
 amelioration of compacted soils, 518–520
 minimizing hazards, 518
 principles, 456–457
 SOC sequestration, 1467
 sulfur, 2243–2244
 testing, 457
 field tests, 457
 laboratory tests, 457
 nuclear gauge, 458
 sand cone, 458
 yield, 516–517
 consequences, 517
 crop response, 517–519
- Compartment-model approach, 1213
- Competition interactions, 1436
- Complex crop rotations, 1183
- Compost, 1407
 wetlands, 10
- Composting, soil, 199, 202
- Compound fertilizer, 921
- Compound identification algorithm (CIA), 2156
- Comprehensive Everglades Restoration Plan (CERP), 2586
- Concavity, 1328
- Concentric layers, mechanical removal, 2226
- Conditioning index, 459
 soil conditioning index (SCI), 459–460
 development, 460
 testing, 460–461
 use of, 460
 soil organic matter (SOM), 459
- Coniferous forests, 963
- Conjugated xenobiotics, 1720
- Conservation, soil, 895
 agriculture, 1383, 1386

- Conservation—
 global assessment
 Africa, 2092–2093
 Asia, 2092
 Australia, 2097
 Central and South America, 2093
 Driving Force–Pressure–State–Impact–
 Response framework, 2097–2098
 Europe, 2090–2092
 North America, 2093–2097
 incentives
 causes and constraints, 2082
 conservation programs principles, 2082
 impact of, 2082–2083
 types of, 2081
 use of, 2081–2082
 management, 1493
 practices, 1040
 practitioners, 485
 strategies, 2084–2085
 tillage, 1422–1424, 1914–1915
 U.S. historical perspective, 2086–2089
 Conservation agriculture (CA), 304
 Conservation Reserve Program (CRP), 522
 Conservation service, soil (SCS), 753,
 1492, 2296
 demonstration project, 1311
 Conservation tillage, 497, 717, 762–763
 concept, 462
 economics, 463
 effects on soil quality, 463
 grasslands, 463
 historic benefits, 463
 utilization, 463
 history, 462
 management intensive grazing (MIG), 464
 conservation and economics, 464–465
 overgrazing, 464
 research, 464
 soil erosion, 462–463
 Conservative mineralization, 194
 Conserved lands, 466–467
 land protection and stewardship, 467
 management goals and plan, 468–469
 monitoring, 468–469
 site assessment and baseline inventory,
 467–468
 synthesis, reflection, and adaptive
 stewardship, 469
 Conserving soil, 213–214
 Consistence, soil, 471–473, 1664–1665,
 2442–2443
 limits of, 172
 Consolidation, 474–476
 Constant flux method, 86
 Constant-head tube, 122
 Construction wastes, 1180
 Consultative Group for International
 Agricultural Research (CGIAR), 933
 Contact thermometers, 2302
 Containment landfill, 1322
 Contaminants, 1721–1722, 1779, 1993
 bioavailability, 201
 biodegradability, 199–200
 toxicity, 200–201
 Contaminated sites, 1681
 Contamination
 of interfacing ecosystems, 1077
 with organic pollutants, 704
 Contamination in groundwater, Arsenic (As)
 China (including Taiwan), 162–163
 India, 162
 Indonesia, 163
 Lao PDR, 163
 Myanmar, 163
 Nepal, 163
 Pakistan, 163
 Thailand, 163
 Vietnam and Cambodia, 163–164
 Contemporary lynchet, 2342–2343
 Continental monsoon climate, 1373
 Contour(ing), 1986
 buffer strips, 763
 bunds, 748
 farming, 763
 lining, 661
 ridges and stone terraces, 1986–1987
 stone barriers, 1988
 strip cropping, 763
 tillage, 2345
 Controlled grazing, 1040
 Controlled-release fertilizer, 1564
 Controlled traffic, 477
 benefits, 478
 implementing, 478–479
 corn and other row crops, 479
 cotton, 479
 examples, 479–480
 much till, 478
 no-till, 477
 planning ahead, 480
 ridge till, 477–478
 Controlled traffic system, 513
 Conventional tillage (CT), 1384
 Convention on Biological Diversity
 (CBD), 1825
 Convergence of evidence, 1313
 Convexity, 1327
 Cool-season pulses, 228
 Copper (Cu), 481
 containing proteins, 482
 deficiency, 482–483
 as essential element, 481–482
 ions (Cu^{2+}), 481
 plant growth
 on Cu-deficient soils, 482
 on high-Cu soils, 482–483
 and plants, 481
 Coprecipitation, 1428
 Corak, 403
 Coring tools, 2475
 Corn (*Zea mays* L.), 118, 214, 493, 1283
 and other row crops, 479
 production, 215
 solar transformities, 3
 Correlation
 and calibration, 2321
 Inceptisols, 1152
 Cosmogenic nuclide approach, 1337
 Cotton (*Gossypium hirsutum* L.), 479, 501, 529
 Coupling hydrogeology, 1142
 Cover crops (CCs), 484–485, 746–747, 763, 2610
 conservation
 air quality, 486
 soil quality, 486
 water quality, 485
 and green manures, 895
 Cover/management and support practices
 factors, 817
 Coversoil
 acid spoils, 1463
 resources, 1462–1463
 suitability criteria, 1462, 1463
 thickness requirements, 1463
 Cow manure/litter, 184
 Cowpea (*Vigna unguiculata*), 300
 C quantification
 chemical methods, 1455
 ^{13}C stable isotope approach, 1456
 inorganic C, 1450
 molecular biomarkers, 1455
 optical and microscopic technique, 1451–1454
 radiocarbon analysis technique, 1450–1451
 SOC separation and quantification, 1450
 spectroscopic techniques, 1455–1456
 thermal oxidation techniques, 1454–1455
 Cracking nature, 513
 Cratering, 1884
 Crater lakes, 68
 adjacent gullies, and thick sand deposits, 68
 Cr contaminations, 1760–1761
 CREAMS (Chemicals, Runoff and Erosion
 from Agricultural Management
 Systems), 814
 C-rich amendments, 1612–1613
 Cristobalite, 1469
 Criteria for selecting indicators, quality
 combining, 1861
 interpretation, 1859–1860
 minimum data set, 1859
 time and spatial aspects, 1859
 trends, 1861
 Critical carbon input, 303
 Critical limits
 classes of, 1854
 concept, 1853
 properties and parameters selection, 1853
 setting, 1853–1854
 site specific, 1854–1855
 Critical nutrient ranges, 1577
 Critical religion, 1903
 Critical toxicity concentration of Cl^-
 (CTCC), 363
 Crop(s)
 and crop rotation, sulfur, 2242–2243
 damages, 69
 evapotranspiration, 488–491
 and irrigation water circulation, 490–491
 monitoring, 491
 grown, 256
 management, 492
 crop residue, 492–493
 soil management, 493
 management
 crop rotation, 555
 depletion of soil organic matter, 555–556
 fallowing, 555
 stubble burning and grazing, 555
 plants, phosphorus availability, 494
 improvement of P efficiency, 494–495
 production decline, 2642

- productivity and sulfur, 2252–2253
- quality and sulfur, 2253
- resistance against biotic stress and sulfur, 2253
- response to P fertilizer, 1701
- tolerance to chloride, 2469
- yield decline and soil compaction, 598
- yields, 1620
- Crop-growing requirements, 717
- Croplands, 522
 - cropping intensity, 524
 - expansion, 525
 - irrigation and fertilizers, 524
 - planning, 900–901
 - soil degradation, 524
 - in United States, 522–523
 - in World, 523–524
- Crop-livestock fodder, 1384
- Cropping
 - intensity, 524, 1383
 - systems, 746
 - systems and enzyme activity, 2102
 - Vertisols, 512
- Crop residue, 256, 1495–1496
 - management, 1915
 - management systems. *See* Conservation tillage
- Crop residue management, 497, 764
 - environmental benefits, 497–498
 - soil, water, and air quality, 498–499
 - soil erosion reduction, 498
 - soil, climate, and crop relationships, 497
 - survey data, 497
- Crop root
 - length, 501
 - response to temperature, 501
 - development, 502–503
 - growth, 501
 - orientation, 503
- Crop rotation, 505, 763, 1424, 1601
 - temperate agriculture, 504
 - organic additions, fertilization, and Ca, 505–506
 - perennial forages and pastures, 505
 - short-term rotations and cover crops, 505
 - soil cultivation and conservation tillage, 506
 - soil structural quality in temperate farming systems, 505
 - soil structure-forming agents, 504–505
 - soil structure in temperate zones, 504
- Crosscorrelation, 1812–1813
- Crotalaria grahamiana* plants, 2226
- Crusts, 599, 1439
- Cry9C protein, 245
- Cryepts, 1153
- Cryogenic churning (cryoturbation), 436
- Cryoprotectants, 1217
- Cryosolic soils, 385
 - Canada, 388
- Cryosols, 102, 392, 409
- Cryoturbated permafrost-affected soil, 1754
- Cryoturbation, 136, 990, 1755, 2374
- Crystalline aluminosilicates, 132
- Crystallized halloysite, 1469
- CSA. *See* Community-supported agriculture (CSA)
- C-sequestration, 1413, 1423, 1435
- Cu, mineral nutrition disorders, 1446
- Cubical, 1655
- Cubical packing, 1654–1655
- Cu depletion in man, 1446
- Cultivated pasture, 1424
- Cultivation of soils, 294
- Cultural anthropology, 2026, 2027
- Cultural eutrophication, 855–856
- Culture-based microbiology, bacteria, 176
- Cumulative water deficit (CWD), 698
- Cuneiform, 1113
- Current-regulating sensors, 2303
- Currie method, 672, 673
- Cyanobacteria, 179, 180, 218, 232, 233, 840, 1435
- Cycling of elements, 997–998
- Cyclodextrins amendments, 1721
- Cytochrome *c* oxidase, 482
- Cytosolic Al effects, 107
- Cytosolic (intracellular) Ca concentration, 107
- D**
- Dambasi (fertile soils), 403
- Dam bursting, 66
- Dampwood termites, 2309
- DAP, 921
- Darcy's law, 120
 - for laminar flow, 86
- Darcy velocity, 1991
- Daresbury Laboratory (DL) POLY, 2167
- Databases land cover, 1316
- Data collection system principle, 489–490
- Dazkir, 403
- 1-D consolidation settlement, 474
- Death of tree, 236
- Debris, 205
 - flow, 544, 1350–1351
 - classification, 544–545
 - hazards mitigation, 546
 - initiation, 545–546
 - movement and deposition, 546
- Decay factor, 2235
- Deccan plateau, 1159
- Dechloromonas hortensis*, 236
- Decision trees
 - assessment, quality, 1846
- Decomposition, 2112
 - and nutrient cycling, 139
 - rates, 1525
 - SOM
 - climate, 2135
 - organic matter quality, 2135–2136
- Deep ditch-high bed systems, 529
- Deep drainage, 2473
- Deep ripping, 2042
- Deep-sea bioremediation, 203
- Deep-sea sediments, 294
- Deficiency symptoms, phosphorus, 1703
- Deforestation, 659, 661, 1116, 1309, 1491–1492, 1517, 1821
- Degradation, 255, 524, 548, 1352, 1492
 - acidity (pH), 563–564
 - cause–effect relations, 550
 - causes of, 551
 - concepts and evolution, 548
 - historical perspective, 548–549
- data accuracy and reliability, 605–606
- desertification, 550
- and desertification, 1353
- economic incentives and investments, 565
 - conceptual framework, 565–566
 - empirical examples, 566
 - intensive agriculture in Asia, 566
- effects of, 604–605
 - by enzymes, 106
 - and erosion processes, 1925–1926
- factors and causes, 604
- farm-level implications, 568–570
- farm-level optimization model, 568
- food aid needs in low-income countries, 571
 - improving agricultural productivity, 571–573
- global extent, 605
- greenhouse effect, 578
 - and carbon, 578–580
- hydraulic properties, 561
 - infiltration and hydraulic conductivity, 562
 - water retention, 561–562
- intrinsic processes, 550
- measurements, 575–576
- natural vs. anthropogenic processes, 602–604
- NDVI, 551–582
 - and applications, 582
 - global assessment of land degradation, 582–588
 - global greenness, 583
 - global negative trend in RUE-adjusted, 583
- off-farm economic implications, 593
 - benefits and damages, 594
 - policy implications, 594–595
 - social optimization model, 593–594
- physical, 596
 - compaction, 596–597
 - loss of soil structure, 596–597
- poverty and food security, 574–575
- processes and causes, 550–551
- quality and, 602
 - biological processes, 602
 - chemical processes, 602
 - physical processes, 602
- reversible or irreversible, 564
- salinity and sodicity, 563
- soil aeration
 - air-filled porosity, 562
 - electrochemical measurements, 562
- soil conservation, 549–550
- soil strength
 - friability, 563
 - penetrometer resistance, 563
 - precompaction stress, 563
- soil structure, 561
 - soil bulk density, 561
- See also* Brick making
- Degradative and aggradative trajectories, 643
- Degradative chain reaction, 659
- Degraded drylands, 648
- Degraded land, 1458
 - extent, 1458
- Degree of aggregation, 315
- Degree of correlation, DRS, 1888
- Degree of desertification, 654–655
- Degree of enzyme activity inhibition, 740

- Degree of erosion, 795
Degree of fitness of soil, 1076
Degree of pyritization, 27
Degree of soil compaction, 2465
Degrees of hydrophobicity, 1148
Degrees of limitation, 1853
Dehumidification, 359–361
Dehydration
 postponement, 694
 tolerance, 694
Dehydrogenase activity (DHA), 2102
Deinococcus radiodurans, 200
Deinococcus-Thermi, 179
Delineations, 1017
Deltaic sand deposits, 70
Del Villar soil classification system, 371–372
Democracy, non-violence, and peace, 849
Demolition waste, 1324
Denaturing gradient gel electrophoresis (DGGE), 1774
Dendrobaena attemsi, 236
Dendrodrius rubidus, 236
Denitrification, 47, 886, 1036, 1402, 1422, 1543–1544, 1555–1556, 1563, 2133–2134
Dense non-aqueous phase liquids, 1993
Densification, 456
Densimetric techniques, organo-mineral associations, 1629–1630
Deoxyribonucleic acid (DNA), 175
Department of Public Health Engineering (DPHE), 161–162
Depletion of soil organic matter, 555–556
Depositional (moraines) glacial landform, 101
Depression storage, 513
Depth, soil
 restoration, resilience, 1926
Depth distribution, 1610
Derjaguin-Landau-Verwey-Overbeek (DLVO) theory, 2048
Desalinization, 2037
Descendum model, 2565
Desertification, 550, 608, 614, 1113
 assessing the degree of desertification, 654–655
 cause of LDD, 609–614
 constraints and challenges, 637–639
 defined, 633–635
 extent, 616–618
 areal, 618–619
 greenhouse effect, 620
 biological changes, 620
 climatic changes, 620
 greening of earth Hypothesis, 621
 humid environment in Iceland, 623
 causes and consequences, 624
 climate and soils, 623–624
 green island, 624
 nature and extent of soil erosion, 626–627
 stages of desertification, 626
 vicious cycle, 624–626
 impact, 628
 indicators, 629–630
 manifestations, 628–629
 San Pedro River basin, 630–631
 technology, measurement, and analysis, 630
 initial mapping efforts, 635
 irreversible extension of desert land, 657
 agroforestry, 662
 air pollution, 658
 contour lining, 661
 definition, 657
 degradation processes, 658–661
 desertization, 658
 deserts, origin, 657–658
 facts and fallacies, 657
 fluctuations and trends of climatic variables, 658
 land surface albedo, 658
 managed restoration, 661
 natural amelioration, 661
 range reseeding, 662
 ripping and subsoiling, 661–662
 surface roughness, 661
 wildlife management, 662
 and land degradation, drylands and non-drylands, 608–609
 mapping constraints and challenges, 633
 mitigating desertification and its consequences, 639–640
 monitoring rehabilitation, 655
 prevention and restoration, 642–643
 implementation of mitigation, 643–644
 integrated evaluation, 643
 primary productivity, 645–646
 redefined, 654
 remote sensing and GIS, 635–636
 restoration, 648
 animals to desertified ecosystems, 651–652
 and drylands, 648–649
 reinstating drainage and flooding patterns, 652
 removal of disturbances, 649
 restoring vegetation, 650–651
 soil microbes, 651
 reversal, 654
 reverse desertification, 655
 secondary productivity, 646
 woody plant range expansions
 modern studies, 621–622
 prehistory to industrial revolution, 621
 world atlas of desertification (WAD), 636–637
Desertization, 660
Deserts, 608, 1365, 1366, 1368
 biome, 1023
Desorption, 355
 phosphorus, 1702
Desorptivity D_E , 862
Destructuration (degradation), 359
Desulfobacter, 27
Desulfotomaculum, 27
Desulfovibrio, 27
Desurfacing, 256
Detection
 array, 320
Detention
 storage, 2485
Detoxification of pollutants, 1123
Developing countries, 664–665
 See also Economics of soil management
Dewatering, 1993
DGGE. *See* Denaturing gradient gel electrophoresis (DGGE)
Diabrotica beetle species, 1217
Diagnosis and recommendation integrated system (DRIS), 891
Diagnostic epipedons, 407
Diagnostic horizons, 668
 FAO–UNESCO soil map, 669–670
 of FitzPatrick, 669
 in the United States, 668–669
 WRB of International Union of Soil Sciences, 670–671
Diagnostic subsurface horizons, 407
Diatom alga, 238
Diatoms, 1469
Dichlorodiphenyltrichloroethane, 1779
Dicotyledonous plants, 72
Dictyostelium mucoroides, 1828
Dicyrtomina ornata, 448
Dietary deficiency, 933
Dietary modification, 1446–1447
Diethylenetriaminepentaacetic acid, 1879
Differential pressure sensor, 46
Diffuse layer ions, 1427–1428
Diffuse reflectance infrared Fourier transform spectroscopy (DRIFTS), 270
Diffuse reflectance spectroscopy (DRS)
 application, 1889
 chemometrics, 1887
 color determinations, 1886
 degree of correlation, 1888
 merits and limitations, 1888–1889
 micronutrient concentrations, 1888
 multicollinearity, 1887
 multivariate calibration methods, 1887
 non-linearity, 1887
 physics, 1886–1887
 soil organic carbon, 1888
 soil reference methods, 1887–1888
 visible–near-infrared spectra, 1887
Diffusion, 1581, 2194–2195
 nutrients, 1588–1589
Diffusion of gases, 672
 measurement of soil gas diffusion coefficient, 672–673
 predictive equations, 672–673
 principles and equations, 672
Digital maps, 1314
 precision agriculture, 1805
Digital orthophoto quadrangles (DOQ), 1018
Digital soil mapping, 675, 1409–1410, 1680
 example, 677–678
 input data, 675
 model, 676–677
 rationale, 676
 soil inference systems, 675
Dilution, soil pH, 1698
Dimethylarsinous acid, 161
Dimethylselenide (DMSe), 1718
Dioctahedral iron illites, 423
Dioctahedral micas, 1791
Dips, 1342
Direct counting, microbial communities, 1434
Direct enumeration, protozoa, 1829
Direct hauled soil, 1462
Direct hydrogen ion (H^+) toxicity, 1362
Direct seeding, 1306

- Disease infestations, 1436
- Dispersed carbonates, 339–340
- Dispersed system, 2216
- Dispersion, 1073, 1359, 1993, 2021–2024, 2032, 2195
- soil, measurement, 1074
 - See also* Slaking and dispersion
- Dispersion-flocculation processes, 61
- Dispersive soils, 2049
- Disposal methods, radionuclides, 1876
- Disruption, 53
- Dissolution, 348
- reactions, 6
 - of minerals, 1480–1482
- Dissolved organic carbon (DOC), 726, 1614, 1670, 1749
- Dissolved organic matter (DOM), 2177
- Distribution coefficient (K_d), 147
- Disturbances, 790
- anthropogenic soil disturbances, 790–792
 - fire and water repellency, 790
- Diurnal freezing and thawing, 973
- field studies, 973–974
 - simulation studies, 974
 - soil freezing and water movement, 973
- Diversion drains, 750
- Diversion or graded terraces, 1988
- Diversity of vertebrate, 2574
- DNA analysis, microbial communities, 1434
- DOC. *See* Dissolved organic carbon (DOC)
- Documented on-farm and offsite damages, 1493
- Dölek, 403
- Dolomite ($\text{CaMg}(\text{CO}_3)_2$), 711, 1362–1363, 1393
- Dolomitic limestone, 1362
- Domestication and fencing, 464
- Dominant chemical weathering processes, 1755
- Dominant rainfall pattern, 1373
- Double-bond equivalent, 2157
- Double-cropping of rice, 1383
- Downslope displacement, 2336
- Drainable pores, 1783
- Drainage, 1038
- coefficient, 689
 - pattern, 1313
 - of peatlands, 1672
 - sodic soils, 2043
 - soil, 1481
- Drainage, water quality
- environmental conditions
 - climate/precipitation, 689
 - soils/hydrology, 689
 - management practices
 - controlled drainage, 691
 - cropping, 691
 - tillage systems, 691
 - pollutant properties
 - adsorption/filtration, 690–691
 - persistence, 690
 - transport processes, 690
- Drainage (accelerated drainage), 680
- anaerobiosis, 683
 - drainage to reduce, 685–686
 - plant responses to poor aeration, 685
 - soil aeration, 683–685
 - factors that affect soil–air–water relationships
 - aeration, 681
 - bulk density, 680
 - pore size distribution, 681
 - soil structure, 681
 - soil texture, 680–681
 - trafficability, 681–682
- Drainage-ditch water control measures, 1539
- Dredged materials (DM), 37
- Dredging, 37
- Drier soils, 173
- Drinking As-contaminated water, 161
- Drop inlet spillways, 751
- Drop spillway, 751
- Drought, 693
- precipitation, evapotranspiration, and soil moisture, 698–699
 - resistance traits, 693
 - dehydration postponement, 694
 - dehydration tolerance, 694
 - escape, 693–694
 - level of organization, 694
 - Passioura water model, 694–695
 - time scale, 694
 - tolerance, 1324
 - tolerant varieties, 1813
- Drylands, 630
- forest, 208
 - grain sorghum, 263
 - tropics, 299
- Dry matter and nutrient accumulation
- production, 1378
- Dryness (arid soil), 158
- Dry sieving, 62
- Dry structural stability, 63
- Drywood termites, 2309
- Dual porosity, 1782
- Dunes–eolian deposition, 1339–1340
- Duripans, 2013
- Durisol, 392, 409
- Dwelling nematode orders, 1519
- Dyes, 1388
- Dynamic soil
- landscape system, 1001
 - quality, 1863–1864
- Dynamic viscosity, 86
- E**
- Earth
- and belief systems, 1904
 - charter and proposed soil charter, 848–849
 - soils, 413
 - spirituality, 852–853
- Earthen levees, 65–66
- and floodwalls, 66
- Earthfill dams
- materials and soils in, 1417–1418
- Earthmoving activity, 36
- Earthwork for terracing, 749
- Earthworms (*Oligochaeta*), 55, 140, 196, 236, 701, 877
- Aporrectodea caliginosa*, 143
 - burrows, 877
 - casts, 877
 - classification, 701
 - dwelling, 236
 - ecology
 - communities, 702
 - ecological groups, 701–702
- effects on soil properties, 702
- organic matter breakdown and nutrient cycling, 702–703
 - physical properties of soil, 703
 - plant growth, 703
 - undesirable effects, 703
- in soil food webs, 1627–1628
- and soil management
- effects of pesticides and soil pollutants, 704
 - influence of agricultural practices, 703–704
 - soil reclamation, 704
 - waste management, 704
- soil organic matter, 1628
- structure
- aggregation, effects on, 2214–2215
 - anecic, 2214
 - burrow types, 2214
 - contribution, 2212
 - endogeic, 2213
 - freshly deposited casts, 2214–2215
 - infiltration, 2214
 - porosity, effects on, 2213–2214
 - stability, 2215
 - types of, 2212–2213
 - web-based resources on, 2213
- Ease of cultivation, 1665
- Eastern plateau, 1159
- Eco-agriculture, 706, 707
- agroecosystems/agroecology, 707
 - Brazil, 709
 - organic agriculture, 706–707
 - Pakistan, 709
 - South Africa, 709
 - specific systems, 707
 - sustainable land use, 708
 - tillage-focused eco-agriculture, 708
- Eco-justice, 849
- Ecology, 711
- C and N cycles, 711
 - environmental influences on soil microbial activity, 711–712
 - soil temperature, 712
 - soil texture, 712
 - soil water content, 712
 - spatial distribution of organic substrates, 712–713
 - fixation of C and N, 714
 - losses of N from soil, 714
 - organic substrate quality, 713–714
 - soil organic matter, 714
- Ecological complexes, 205
- Ecological damage, 15
- Ecological indicators, 104
- Ecological integrity, 849–850
- Ecological restoration, 648, 649
- abiotic limitations, 1934
 - assessment, 1934
 - biotic manipulations, 1934
 - degrading processes, 1933
 - dynamic systems and restoration goals, 1932–1933
 - Eucalyptus marginata* forest, 1933
 - food-web complexity, 1934
 - measures of success, 1934
 - processes involved in, 1933
 - stressors, 1933

- Economic incentives, 594
- Economics of soil management, 664, 716
- adoption in the United States, 717
 - conservation tillage, 717
 - crop rotations, 717–718
 - implementation, 718
 - monitoring soil properties, 717
- assessing the problem, 664–665
- off-site impacts of soil management, 716–717
- soil degradation management, 716
- soil fertility management, 716
- understanding farmer behavior, 665
- Ecosystem(s), 1758
- agricultural chemicals, 721
 - buffered and protected environment, 721
 - composition and nature of soil, 719
 - distress syndrome, 1077
 - dynamics of water and air, 719–720
 - impacts, acid rain
 - agricultural ecosystems, 14
 - aquatic ecosystems, 15
 - forest ecosystems, 14–15
 - soils, 13–14
 - influence on, 1503
 - life attributes and soil environment, 719
 - mites, 1489
 - of reference, 1935
 - services, 612
 - soil ecology, 721–722
 - soil temperature, 721
 - stability, 625
 - tillage, 721
- Ecosystem function analysis (EFA), 1936
- Ectomycorrhiza and ericoid mycorrhiza, 1503
- Ectomycorrhizal forests, 1502
- Ectomycorrhizal plant, 1502
- Ectoparasitic nematodes, 1520
- Ecto-plant-parasite, 1525
- Eddy covariance, 488
- Edelman approach, 412
- Effect of buffer strips on chemical transport, 1027
- Eisenia submontana*, 236
- Elasticity, 1918
- Electric sensors, 2304
- Electric thermometers, 2303
- Electrochemical measurements, 562
- Electrode methods, 85
- Electrokinetics, 1770
- Electrolyte concentration (EC), 757, 824
- Electromagnetic induction (EMI), 1005–1006
- Electromagnetic spectrum, 1006
- Electromagnetic velocity, 1009
- Electromigration, 1770
- Electron activity, 1896–1897
- Electron capture detector (ECD), 440
- Electronic sensors, 2620
- Electronic waste, 1763
- Electrophoretic techniques, 870
- Electrospray ionization (ESI), 2177
- Electro ultra filtration (EUF), 1794
- Elemental C. *See* Pyrogenic carbon (PyC)
- Eluviation, 1517
- Embodied solar energy, 1
- Embryonic nodes, 72
- Emerging capture technologies
- aqueous NH₃, 1801–1802
 - chemical looping, 1802–1803
 - improved auxiliary processes, 1803
 - membranes, 1802
 - novel liquid sorbents, 1802
 - solid sorbents, 1802
- Emergy, 2
- analysis
- agricultural production, 2–3
 - defined, 1–2
 - environmental decision making, 2
 - forest production, 3–4
 - timber production, 4
- evaluations of forests managed for timber production, 4
- indices, Italian agriculture, 2, 3
- investment ratio, 2
- soil, 1, 2, 3
- sustainability index, 2
- yield ratio, 2
- Emissions
- sulfur, 2244
 - uniformity, 1259
- Empirical erosion model, 2489
- ABAG (German USLE), 2491–2492
 - MULSE, 2490
 - SLEMSA, 2491
 - universal soil loss equation, 2489–2490
- Empirical scaling
- inverse modeling, 1979
 - pedotransfer functions and soil–landscape relationships, 1979
 - transition from laboratory to plot scale, 1979
- Empower, 2
- Empresa Brasileira de Pesquisa Agropecuária (EMBRAPA), 380
- Enchytraeidae (potworms), 723
- ecological importance, 725–726
 - feeding behavior, 725
 - identification, 723
 - sampling, 723–724
 - spatial distribution and population dynamics, 724–725
- Enchytraeids, 206
- Endemic goiter, 1446
- Endeostigmata, 1488, 1489
- End land use, 1304–1305
- ecological restoration and, 1325
- Endogeic earthworms, 2213
- Endoparasitic nematodes, 1520
- Energy balance, soil, 90
- Energy dispersive X ray (EDAX), 156
- Energy flux, 858
- Engineered biopiles, 202
- Engineered terraces, 2343
- Enhanced coal bed methane (ECBM), 995
- Enhanced oil recovery (EOR), 995
- Enrichment index (EI), 1527
- Entisols, 159, 410, 728, 2371
- genesis, 730
 - biota, 730
 - climate, 730
 - parent material, 730
 - relief, 730
 - time, 730
 - suborders, 728–729
- Entomopathogenic nematodes (EPNs), 1521, 1523–1524
- Environmental accounting, 1
- Environmental aspects, Loess, 1368
- Environmental decision making, emergy analysis, 2
- Environmental Defense Fund, 77
- Environmental Impact Assessment Directive and the Strategic Environmental Assessment Directive, 1825
- Environmental indicators, 1935–1936
- Environmental Liability Directive, 1825
- Environmental loading ratio, 2
- Environmental Protection Agency (EPA), 146
- Environmental quality, 1598–1599
- Environmental services, 205
- Enzymes, soil, 198, 737, 738
- agro-techniques and cropping systems, 2100–2102
 - in biogeochemical cycling
 - C-cycling (glycosidases), 738–739
 - N-cycling (amidohydrolases), 739
 - P cycling (phosphatases), 739
 - S-cycling (arylsulfatase), 739
 - characteristics
 - enzyme protein concentrations in soil, 738
 - grouping, 737
 - location, 737–738
 - and climate change, 2104–2105
 - functions and, 2100
 - measurement of, 2103–2104
 - soil fauna in relation to, 2105
 - soil metabolic functioning
 - enzyme activities and soil management relationship, 739–740
 - enzyme activities and soil pollutants relationship, 740
 - enzyme activities and soil properties relationship, 739
 - and yield sustainability, 2102–2103
- Enzyme–substrate complex, 737
- Eolian, 134
- Ephemeral gully erosion, 1061
- Ephemeral gully in United States, 1060
- Epoxy, pores cast, 1388
- Epsom salt (MgSO₄), 1393
- Ericoid mycorrhizas, 1502
- Eroded sediments, 2641
- Erodibility, 51, 63, 815, 845–847
- Erosion, 136–137, 560, 742, 767, 1330, 1517, 1926
- aggregate breakdown, 757
 - crust formation, 759
 - differential swelling, 758
 - and erodibility, 759
 - physico-chemical dispersion, 757
 - slaking, 757–758
 - soil fragments release, 759–760
 - soil properties influencing, 758
- agronomic practices, 762
- conservation tillage, 762–763
 - contour buffer strips, 763
 - contour farming, 763
 - contour strip cropping, 763
 - cover crops, 763
 - crop rotation, 763
 - erosion control with surface cover, 763–764
 - filter strips, 763
 - grassed waterways, 763–764
 - residue management, 762

- on agronomic productivity, 360
- assessment, 744, 769–773
- assessment of erosion-induced CO₂ fluxes
 - mass balance approach, 778–779
- biological control measures
 - contour cultivation, 746
 - cover crops, 746–747
 - cropping systems, 746
 - hedge row barriers, 747
 - vegetative filter strips, 747
 - vegetative grass barriers, 747
- carbon (C) losses from soil, SOM, 2147–2148
- carbon dioxide (CO₂), 777–780
- climate and hydrology, 790
- under coffee and cocoa, 819–820
- control, 762, 821
 - on Loess plateau, 1375–1376
- control with surface cover, 763–764
- crop productivity
 - and human life, 794–796
 - relationship, 796
 - techniques for evaluating, 796
- crop residue management with conservation
 - tillage, 764–766
- crop yield, 786–788
- disturbances, 790
 - anthropogenic soil disturbances, 790–792
 - fire and water repellency, 790
- episodes, 1372
- eroded SOC, 777–778
- erosional effects, 794
- erosion–productivity relationships, 786–788
- estimates of erosion-caused CO₂ emissions, 779–780
- fly-ash spheres as time marker, 798–801
 - analytical methods, 798
 - Cesium-137, 801–802
 - in depositional studies, 802
 - on SOC, 801
 - time marker in erosion studies, 800–801
- global change, 804
 - accelerate soil erosion, 807
 - in China and in the World, 804
 - degrading soil quality, 805–806
 - ecological civilization, 806–807
 - soil and water conservation in china under global climate change, 807–808
- gravity-induced, 744
- hilly ecosystems, 746
- hot spots, 786
- in Iceland, 627
- impacts, 744–745, 768–769
- irrigation-induced, controlling, 782
 - conversion to sprinklers, 782–783
 - site modification, 783
 - soil protection and tillage, 783
 - water properties, 783–784
- landscape, SOM, 2144
- Loess, 1368
- Loess Plateau, 1370, 1374–1375
- lower Himalayas, 1383
- mass planting of vegetative cover, 827
- measurement techniques, 814
 - cover/management and support practices factors, 817
 - interrill erosion, 816
 - plot design and layout, 816
- rainfall erosivity, 814–815
- rainfall simulators, 815–816
- rill erosion, 816–817
- small plot, 815
- soil erodibility, 815
- USLE standard runoff plot, 815
- water erosion assessment, 817–818
- mechanical measures, 747–748
 - bench terraces, 748–750
 - contour bunds, 748
 - diversion drains, 750
 - graded bunds, 748
 - grassed waterways, 750
 - gully control drainage line treatment structures, 750–751
- natural disturbances, 1820
- and nitrate runoff, 211
- no-till, 1566
- off-site impacts, 773–774, 812
- under oil palm, 819
- on-site impacts, 811–812
- patterns, 1336
- prediction, 792
- quality
 - biological effects, 1857–1858
 - chemical effects, 1857
 - physical effects, 1856–1857
- rainfall analysis, 751–752
- rate variability, 1337
- reduction, 498, 782
- risks, 1330
- scale and extent, 768
- sedimentation control vegetative techniques, 825–827
- snowmelt, 831
 - amelioration, 833
 - modeling, 833
 - precipitation, 831
 - runoff events, 832–833
 - snow and freezing conditions, 832
 - soil, 831–832
- SOC, impact on, 801
- SOC sequestration, 1467
- soil amendments, 823
 - gypsum, 824
 - lime, 823–824
 - manure, compost, and organic sludge, 825
 - mulches, 826
 - synthetic polymer, 824–825
- on soil chemical properties, 821
- soil conservation, 834–836
- surface cover, 792
- under tea, 820–821
- timescales, 1336–1337
- tolerance (T), 753
 - basis for T values, 753–754
 - existing concerns, 754–755
- and truncation, 1743
- in the United States, 214
- variability, 792
- vegetative materials, 827
- vegetative strips, 828
 - buffer strips and riparian strips, 829–830
 - filter strip terraces, 829
 - grass strips, 828
 - settling basins, 830
 - vegetated waterways, 828–829
- by water, 742–744
- water quality, 838
 - nitrogen (N), 839
 - nutrients, effects of, 838–839
 - pesticides, effects of, 841–842
 - phosphorus (P), 839–841
 - sediments, effects of, 838
- by wind, 742
- Erosion–productivity relationships, 788
- Erosivity and erodibility
 - process-based approaches, 845–847
 - universal soil loss equation (USLE), 845
- Erwinia carotovora*, 181
- Escherichia coli*, 42, 177
- ESP. See [Exchangeable sodium percentage \(ESP\)](#)
- Espodossolo, 382
- Espodossolos Gleissolos, 381
- Essential mineral elements, 1728
- Ethanol, 214–215
- Ethanol production, 214
- Ethics, global covenant, 848
 - differentiated responsibilities, 852
 - earth charter and proposed soil charter, 848–849
 - earth spirituality, 852–853
 - eco-justice, 849
 - ecological integrity, 849–850
 - precautionary principle, 851–852
 - stewardship, 850–851
- Ethiopia, soil carbon sequestration
 - climate and ecological conditions, 2066
 - deforestation, 2066, 2068
 - degraded lands, rehabilitation, 2070–2071
 - erosion, 2066
 - location, 2067
 - loss of SOM in, 2069–2070
 - Montane forests, 2066
 - organic matter pools, 2067
 - soil degradation in, 2068–2069
 - soil organic matter (SOM), 2066
 - soil quality (health), 2066
- Ethylenediaminetetraacetic acid (EDTA), 1879
- Ettringite, 1417
- Eucalypts and legumes, 1938
- Eucalyptus citriodora*, 1324
- Eucalyptus* spp., 827
- Eucalyptus torelliana*, 1324
- EUF. See [Electro ultra filtration \(EUF\)](#)
- Eukarya*, 175
- European Centre for Ecotoxicology and Toxicology of Chemicals, 1543
- European emission trading system (ETS), 77
- European Russia, boreal forest soils, 248
- European Soil Erosion Model (EUROSEM), 773, 814, 1881, 2501
- European Union (EU) protection policies, 1826
 - air pollution control law, 1825
 - Biocide Directive, 1825
 - community legislation, 1824
 - distribution of, 1824
 - EC chemicals law and water protection law, 1825
 - flood risk management policy, 1825
 - international context, 1825
 - Nitrate Directive, 1825
 - Pesticide Directive, 1825

- European Union (EU) protection policies—
 pollutant-referred supplements, 1825
 Sewage Sludge Directive, 1824
 waste management, 1824–1825
 EUROSEM. *See* European Soil Erosion Model (EUROSEM)
 Eutric Brunisol, 386
 Eutrophication, 855, 917, 1406
 cultural, 855–856
 effects of, 856
 rate of, 856
 reduction and management, 856–857
 Evaporation, 202, 858, 1079, 2473
 analytical models, 861–865
 measurement methods
 chambers, 859
 heat pulse probes, 860
 micro-Bowen ratio systems, 859–860
 microlysimeters, 859
 soil water balance, 859
 numerical models, 860–861
 Evapotranspiration, 1669
 Evapotranspiration (ET), 261, 559, 686, 689, 698, 1538, 1812, 2510
 Excavators, 69
 Exchangeable sodium (ES), 2044
 Exchangeable sodium percentage (ESP), 115, 757, 1965, 2022, 2029, 2032–2034, 2056
 sodium-affected soils (SASs), 2037
 Exergy, 2
 Expert system interface in IDL, 1315
 Exploiting soil cracking, 513
 Explosives, 1717
 Extent and severity, Arsenic (As)
 Bangladesh, 161–162
 West Bengal, 162
 External nutrient requirement, 1736
 Extractants for plant nutrients, 2321
 Extracted C, 1430
 Extraction
 adenosine triphosphate extraction
 method, 1431
 chloroform fumigation incubation and
 extraction methods, 1429–1430
 freeze-dried soil extraction method, 1430
 microbial communities, 1434
 phospholipids fatty acids (PLFA) extraction
 method, 1431
 Exxon Valdez oil spill, 201, 203
- F**
 Faba beans (*Vicia faba*, L.), 224, 530
 Facultative microbes, microbial
 communities, 1434
Fall of Icarus, The, 169
 Fallow, 508
 FAO–UNESCO Soil Map, 20
 Farm building and homes, damage to, 70
 Farmed wetland, 2568
 Farming/cropping systems, 1050
 Farming profitability, 1868
 Farm-level decisions, 593
 Farm management practices, 1038
 Farmyard manure (FYM), 1385, 1401, 1970, 2102
 Fault scarps–tectonic uplift, 1340
 Fauna, 871, 1439–1440
 diversity and biomass, 872–873
 organisms, 871–872
 roles in ecosystem functions, stability, and
 sustainability, 873–874
 FDR (Capacitance) and TDR, 2527
 Fe, mineral nutrition disorders, 1446
 Fe-deficiency-created anemia, 1443
 Federal Soil Protection Act of Germany, 2084
 Feeding, 1443
 Feldspars, 1474, 1791–1792
 Fen, 2568
 Fe-oxidizing bacteria, 9
 Fermentation technology, 244–245
 Ferralsols, 392, 409
 Ferrihydrite ($\text{Fe}_3\text{HO}_8 \cdot 4\text{H}_2\text{O}$), 8, 9, 136, 1469, 2003
 Ferrolisis, 128
 Ferromagnesian minerals, 1473
 Ferrosols, 389–390
 Fertilization, 905
 Fertile Oxisols, 1636
 Fertility, 1471
 agricultural sustainability, 884–885
 biological assessment, 890
 contribution, 190
 decline, 880
 chemical properties, 881–882
 expert knowledge, 880–881
 nutrient budgets and balances, 882
 definitions, 880
 degradation/improvement, 1225
 environmental impact, 892
 environmentally compatible
 management, 884
 essential nutrients, 885
 of forest soils, 957
 heavy metals and food concerns
 contamination sources, 886–887
 management practices for minimizing
 metal transfer to food crops, 888
 phytouptake and phytotoxicity, 887
 soil contamination with metals, 887
 low-input soil fertility management, 894
 arable soil conditions for plant growth, 894
 nutrient availability and cycling, 894–895
 management, 716
 management of environment, 885
 air quality, 886
 soil quality, 885–886
 water quality, 885–886
 North China plains, 899
 and nutrient management
 applications timing, 1584
 base saturation (BS), 1578
 buffer capacity, 1580
 buffering nutrients in soil solution, 1577
 calcareous soils, 1579
 cation and anion exchanges, 1578–1579
 essential plant nutrients, 1576–1577
 lyotropic series, 1578
 mineral solubility in soils, 1579–1580
 nutrient mobility in soil, 1581–1582
 nutrient placement methods, 1583–1584
 nutrient source identification, 1583
 organic materials, 1579–1580
 plant nutrient requirement assessment, 1582
 quantify optimum nutrient rate, 1582
 salts in alkaline soils, 1579
 site-specific nutrient management, 1584
 soil–plant–atmosphere system, 1577
 supplying capacity, 1582
 transport from soil to roots, 1580–1581
 perspectives, 892
 plant analysis, 890–891
 restoration, resilience, 1926
 and SOC status
 lower Himalayas, 1383–1384
 soil, 185, 342
 soil quality management in HHH plain, 899–900
 soil quality of cropland in HHH plain
 ecological practices, 901
 reclamation, 901
 SOC stock, 901–902
 soil test–crop response correlation, 891–892
 soil testing, 891
 spectral sensing and site-specific soil fertility
 evaluation, 892
 and stability, 219
 sustainable management of soil quality in
 HHH plain, 900–901
 techniques for sustainable soil fertility
 management, 895
 agroforestry and intercropping, 896
 azolla and anabaena, 895–896
 cover crops and green manures, 895
 integration of livestock, 896
 mulches, 896
 nutrient recycling, 895
 reduced tillage, 896
 soil conservation, 895
 and vascular plants, 217–218
 Fertilization, 311, 1038, 1210–1211
 in crop rotation, 360
 and P level, 1701
 precision agriculture
 decision-making strategies, 1809
 georeferenced soil and crop information, 1808–1809
 technical equipment and data, accuracy
 of, 1809
 Fertilizers, 40, 919
 consumption and production, 1557
 and irrigation, 1492
 leaching losses, 907
 magnitude of, 908–909
 measurement and modeling, 907–908
 mechanisms and pattern, 907
 mitigation and management of fertilizer
 nutrient losses, 909
 magnesium, 1393–1394
 nutrients and products, 910
 N fertilizers, 911
 phosphate fertilizers, 911–912
 potash fertilizers, 912
 secondary and micronutrients, 912–913
 simple vs. complex, 910–911
 organic, 916–917
 animal residues, 917
 biosolids, 918
 by-products and wastes, 918
 effluents, 917–918
 manures, 917

- municipal wastes, 917
 - plant residues, 917
- rates, 1809
- satellite-aided navigation, 1809
- sulfur, 2244
- types of products, 919
 - blended fertilizers, 921–922
 - compound fertilizer, 921
 - fluid fertilizers, 919–920
 - gaseous fertilizers, 919
 - slow and controlled release fertilizers, 921
 - solid fertilizers, 920–921
- urban waste and sludges, 923
 - agricultural management, 924
 - biosolids production and quality, 923
 - composting of biosolids and MSW, 923–924
 - land application of biosolids, 925
 - production, 923–924
 - regulations covering land application, 924
 - types, 907, 923
- use, 913
 - efficiency, 913–915
 - by nutrients and products, 913–915
 - trends, 913
- Fertilizer applications, 903
 - methods, 903–904
 - broadcasting, 904
 - fertigation, 905
 - foliar feeding, 905
 - placement/banding, 904–905
 - plant nutrients, 1733
- Fertilizer management, 1227–1228
 - nitrogen, 1228
 - other nutrients, 1229–1230
 - phosphorus, 1228–1229
 - potassium, 1229
- Fertilizer phosphorus requirement (FPR), 1738
- Fertilizer P reactions
 - absorption and translocation, 1703
 - deficiency symptoms, 1703
 - desorption, 1702
 - initial rapid reactions, 1702
 - plants nutrition, 1703
 - plants requirement, 1703
 - slow reaction phase, 1702
 - soil P availability, 1702–1703
- Fibrobacteres, 179
- Fibrous high-temperature minerals, 1419
- Fick's first law, 672
- Fick's law, 986
- Fick's second law, 1214
- Field capacity, 561
- Field grading and raised-bed cropping, 2042
- Field layout, 1309
- Field monitoring, 771
- Field pea (*Pisum sativum*), 224
- Field-scale wind erosion models, 2633–2634
- Filter strips, 763, 1024
- Filter strip terraces, 829
- Fine-grained soil, 457, 1417
- Fingermillet (*Eleusine coracana*), 300
- Fire
 - natural disturbances, 1819–1820
 - and water repellency, 790
- Fire and soils, 926
 - ash additions, 927
 - atmospheric losses of elements, 926–927
 - biological properties, changes in, 928
 - effects of soil heating, 927
 - following fire, management of, 928
 - nutrient cycling, changes in, 928
 - physical processes, changes in, 927
 - regimes and impacts, 926
 - soil chemical properties and processes, 927–928
- Fire ants (*Solenopsis invicta*), 151
- Fire-induced water repellency, 2548
- Firmicutes, 179
- Fishery ponds, 257
- Fish farming, 1383
- Fixed extraction ratios, 2321
- Flagellates, 237
- Flashy floods, 2359
- Flat spaces, 2217–2218
- Flatwood, 2568
- Flies, 205
- Flocculated soils, 2049
- Flocculated system, 2216
- Flocculation, 1360
 - processes, 428
 - test, 824
- Flood
 - channel boundaries, 1973
 - irrigation of gypsic soils, 1066
 - risk management policy, 1825
- Flooded soils, 1556
- Flooding tolerance, crops, 526
 - genetic transformation, 527–528
 - genetic variability, 527
 - molecular plant breeding, 527
 - morphological and anatomical adaptation, 527
 - physiological, biochemical, and molecular adaptation, 527
 - plant breeding, 527
- Flood-irrigated pastures, 1038
- Floodplains, 65, 112–113
 - soils, 157
- Floodwall, 66
- Floodwaters, 66
 - farming, 2515
 - pressure, 65
- Flow, slope failure, 1349
- Flow detachment–flow transport (FD–FT) system, 847
- Flow net, 1992
- Flue gas desulfurization waste, 1996
- Fluid fertilizers, 919–920
- Fluid flow modeling, 930
 - application, 930
 - challenges and achievements, 930–931
 - governing equations, 931
 - process scale, 931–932
- Flush of mineralization, 1632
- Fluvents, 729
- Fluvisols, 392, 409
- Flux, 1991
 - chambers, 438
 - discontinuity, 1344–1345
- Fly-ash, 1996
 - spheres as time marker, 798–801
- Foliar application
 - selenium, 1996
- Foliar feeding, 905
- Foliar lesions, 14
- Folsomia candida*, 142, 448
- Food and Agriculture Organization (FAO), 18, 419, 548, 788, 942, 1444
 - data reported as arable land (FAOSTAT), 522
 - World Reference Base (WRB), 134
- Food crops, 933–934
 - agronomic biofortification of cereal grains, 934
 - chickpea (*Cicer arietinum*), 936–937
 - maize, 936
 - oats (*Avena sativa*), 936
 - rice, 934–935
 - wheat, 935–936
- Food packaging, 105
- Food production, 20
- Food security, 939, 942
 - agricultural supply, 951
 - Africa, 951–952
 - Asia, 951
 - Central America, 952
 - economic growth
 - Africa, 952
 - Asia, 952
 - Central America, 952
 - incentives for soil use and conservation, 943–944
 - land degradation, 939
 - low-input and intensive agriculture, 939–940
 - per-capita cropland requirement, 946–947
 - soil degradation, 951
 - soil degradation and food security, 943
 - soil degradation on globalscale
 - decline in productivity, 948–949
 - limitations of methodology, 947–948
 - soil quality and agricultural productivity, 943
 - solutions, 940
 - soil quality, fertilizer use, and nutrient management, 940
 - sustainable and efficient water use, 941
 - trends, 942
 - underlying causes, 946
 - losses and gains of agricultural land, 946
 - misuse of land, 946
 - pressure on land, 946
 - undernourishment, 945
 - vulnerable areas, 945–946
- Food web, 138, 139
- Food webs, 1624
 - biota role in
 - bacteria, 1624–1625
 - collembola, 1626–1627
 - fungi, 1625
 - macrofauna, 1627–1628
 - mesofauna, 1626
 - microfauna, 1625–1626
 - zones of influence (ZOI), 1624
- Food webs, nematodes in, 1626
- Footslope, 1328
- Forage, agroforestry system, 1379
- Forests, 2552
 - ecosystems, 928
 - nutrient cycling, 957
 - fires, 1819
 - production
 - energy analysis, 3–4

- Forestry practices, soil physical properties, 1821
- Forest savannah (*cerradão*), 345
- Forest soils, 955–957, 966, 969
- aeration, 969–971
 - boreal and coniferous forests, 963
 - CA, MG, K, P, and trace elements, 971–972
 - calcium (Ca) depletion, 959–961
 - forest ecosystems, 961
 - fertility of, 957
 - forest ecosystem nutrient cycling, 957
 - influence of abiotic factors, 967–968
 - influence of biotic factors, 966–967
 - influence of climate, 966
 - moisture, 969
 - N availability, 972
 - nutrient supplies and availabilities, 971
 - off-site effects of forest management, 957–958
 - physical properties, 969
 - role of disturbance, 968
 - shrub and woodland, 965
 - soil development and soil-site relationships, 957
 - in soil science, 957
 - temperate deciduous forests, 964
 - temperate rain forests, 964–965
 - temperature, 971
 - tropical monsoon/deciduous forests, 965
 - tropical rain forests, 965
- Forest-type boundaries, 1315
- Formation process
- Loess plateau, 1372–1373
- Formica aquilonia*, 152
- Formica podzolica* mounds, 150
- Formicidae, 206
- Forsterite (Mg_2SiO_4), 2564
- Fortification, 1446
- Fossil fuels, 244
- based emissions, 79
 - burning, 307
 - combustion, 1048
- Fourier transform infrared (FTIR)
- spectroscopy, 1145
- Fourier transform ion cyclotron resonance (FTICR), 2177
- FPR. *See* Fertilizer phosphorus requirement (FPR)
- Fractional density, 1654
- Fragipan-type *tepetates*, 2306
- Fragmentation procedures, 61
- dry sieving, 62
 - energy input, 62
 - wet sieving, 62
- Framework for evaluating sustainable land management (FESLM), 835
- Fraxinus pennsylvanica* (green ash), 2573
- Free acids, soil pH, 1695
- Free-living bacterial-feeding, 1523
- Free-living nematodes, 1525–1526
- Freely drained soils, 136
- Freeze and thaw, diurnal, 973
- field studies, 973–974
 - simulation studies, 974
 - soil freezing and water movement, 973
- Freeze-dried soil extraction method, 1430
- Freeze susceptibility (supercooling), 1217
- Freeze/thaw cycles, 102, 974, 1081
- Freezing, soil, 1079, 1080–1081
- Freezing-susceptible overwintering stages, 1217
- Freezing–thawing, 504
- French classification
- RP and WRB, 391–393
- French soil classification
- history, 391
- Frequency domain reflectometry, 2527–2528
- Freshwater prawn, 275
- Freundlich isotherm model, 1427
- Friability, 563
- Frozen soils, 492, 976, 1753
- applications, 976–977
 - relationships, 976
- FTICR. *See* Fourier transform ion cyclotron resonance (FTICR)
- Fuel additives, 1717
- Fuel wood collection, 662
- Fulvate theory, podzol, 1749
- Fulvic acids, 1359
- Fungal-feeding Collembola, 143
- Fungi, 138, 979
- abundance and diversity, 980
 - and bacteria, 55
 - and ecosystem function, 980–981
 - fungal classification, 979
 - growth and reproduction, 979–980
 - (Mycorrhizae), 206
 - in soil food webs, 1625
- Fungus root, 1436
- Funnel ants, 151
- Furrow diking, 1566
- Furrow irrigation, 1262
- runoff, 1263
- Fusarium lateritium*, 143
- Future of soil science, 982
- altered ecosystems, 983
 - changing demographics, 984
 - continuing needs, 984
 - functions of soil, 983–984
 - influencing, 984
 - pedosphere, 982–983
- Fuzzy sets, pedometrics, 1678
- FYM. *See* Farmyard manure (FYM)
- G**
- Gabion check dams, 751
- Gadgil effect, 1503
- Gamma*proteobacteria, 181
- Gamma-ray spectrometry, 1873
- Garbage compost, 1759
- Gas and vapor phase transport, 986
- soil-gas diffusion coefficient, 986–987
 - predictive equations, 987
 - sieved and repacked soil, 987–988
 - undisturbed soil, 987
 - transport physics and equations, 986
- Gas chromatography (GC), 45, 440
- Gas diffusion, 672
- measurement of soil gas diffusion coefficient, 672–673
 - predictive equations, 672–673
 - principles and equations, 672
- Gas diffusion coefficient, 987
- Gaseous fertilizers, 919
- Gas exchange, 1389, 1390
- Gas exchange, tillage
- crop production and soil C loss, 2348
 - greenhouse gas emissions, 2348
 - mechanics of, 2349
 - soil productivity and environmental benefits, 2349–2350
 - tillage-induced CO_2 loss, 2348–2349
- Gas flux, 438
- density, 672
- Gas transport, 987
- Gaudin–Schuhman distribution, 62–63
- Gelepts, 1153
- Gelisols, 102, 410, 989
- classification of, 989–991
 - and land use, 991
 - pedon as basic soil unit, 989
- Gemmatimonadetes, 179
- Gene escape and genetic pollution, 245
- General Directorate (GD) of Agricultural Reform, 1413
- General spatial soil prediction model, 677
- Genesis, soil, 1154–1155
- Genetically modified (GM) crops, 243, 244
- Genetically modified organisms (GMOs), 1600–1601
- Genetic engineering, 243
- Genetic engineering for phytoremediation, 1722–1723
- Genetic transformation, 527–528
- Genetic variability, 527
- Genomic (chromosomal) DNA, 177
- Geobiont, 236
- GeoChip, 2103
- Geogenic organic carbon (GOC), 1449–1450
- quantification techniques, 1450
 - radiocarbon analysis technique, 1450
- Geognosie, 1104
- Geographical information system (GIS), 1845
- Geographic coordinate system, 1318
- Geographic information system (GIS), 1314, 1806
- integration, 1319
 - in land use planning activities, 1318–1320
 - and models, linking, 1319–1320
 - reliability of, based result, 1320
- Geographic information systems (GIS), 630, 676, 881, 1017–1019, 2555–2558
- Geological sequestration, 279
- Geological storage, 279
- Geologic carbon capture and storage, 992–993
- GS targets, 993
 - coal seams, 995
 - oil and gas reservoirs, 995
 - saline formations, 993–994
 - monitoring, verification, and accounting (MVA), 995–996
- Geology and soil, 997
- atmosphere, hydrosphere, and lithosphere, 997
 - chemical weathering processes, 997–998
 - cycling of elements, 997
 - parent material, 998–999
- Geometric mean water retention, 1683
- Geomorphic description systems
- attributes, 1331
 - environments and landscapes groupings, 1333
 - landscape classifications, 1333, 1334
 - mountains vs. plains, 1331

- soil, 1332
- soils and soil geomorphology, 1334
- in soil science, 1333
- Geomorphic descriptors, 1340–1342
- Geomorphology, 1000, 1333–1334
 - concepts, 1000–1001
 - models, 1001–1002
- Geophiles, 236
- Geophysical methods, 1004–1005
 - applications to soil science
 - precision farming, 1008–1009
 - soil salinity monitoring, 1010–1011
 - soil water content measurement, 1009–1010
 - USDA/NRCS soil surveys, 1007–1008
 - electromagnetic induction, 1005–1006
 - ground-penetrating radar, 1006
 - optical reflectance (UV/VIS/NIR/MIR), 1006–1007
 - resistivity, 1005
 - γ -ray spectroscopy, 1007
- Geo-referenced soil observations, 675
- Geospatial mapping of harvested yields, 1009
- Geostatistics, 1012
 - application in soil science
 - optimal interpolation and isarithmic mapping, 1014–1015
 - optimizing sampling schemes, 1015
 - uncertainty analysis, 1015
 - modeling of the random function, 1012–1014
 - theory of the regionalized variable, 1012
- Geostatistics, pedometrics, 1678–1679
- Geotextiles, 1989
- Geoxenes, 236
- GGP. *See* Grain for Green Project (GGP)
- GHG. *See* Greenhouse gases (GHGs)
- GHG emission restrictions and money, 2078
- GHG emissions and grazing lands
 - abiotic factors, 1036
 - biotic factors, 1038–1039
 - burning of savannas, 1039
 - farm management practices, 1038
 - grazing type and intensity, 1039
 - leguminous pastures, 1039
 - manure, 1039
 - rangelands, 1039
 - soil moisture, 1037
 - soil pH, 1037
 - soil temperature, 1036
 - soil texture, 1037
 - species, 1038
 - weather patterns, 1037
- Gibbsite ($\text{Al}(\text{OH})_3$), 105, 425, 1476
 - formation, 2005–2006
- Gibbsitic Oxisol, 316
- Gilgai, 348, 2445
- Ginderi (*Premna integrifolia*), 734
- Giotto (Florentine painter), 168
- GIS. *See* Geographic information system (GIS); Geographic information systems (GIS)
- GIS-based LESA systems, 1320
- GIS integration and remote sensing, 1017–1019
- GIS/model graphical user interface (GUI), 1319
- Glacial meltwater and Loess, 1365
- Glaciers, 1366, 1368, 1753
- GLASOD, 772
- Gleisols, 1646–1647
- Gleissolo, 382
- Gleyed soils, 1955
- Gleyic Andosols (GA), 134
- Gley soils, 392
- Gleysolic soil, 385
- Gleysols, 409
- Gliocladium* spp., 240
- Gliricidia sepium*, 304, 747, 827, 1380
- Global Assessment of Human-Induced Soil Degradation (GLASOD), 947
- GLobal Assessment of Soil Degradation (GLASOD), 2626
- Global carbon cycle
 - carbon components, 2129
 - C fluxes, 2129–2130
 - soil C fluxes and global change, 2130–2131
- Global C cycle, 1203
 - atmospheric derivation, 1204
 - human activities, 1204–1205
 - mean residence time (MRT), 1203
 - weathering, 1203–1204
- Global Change and Terrestrial Ecosystems (GCTE), 2159
- Global climatic changes, 1745
- Global distribution, SOM
 - Andisols, 2136
 - decomposition
 - climate, 2135
 - organic matter quality, 2135–2136
 - Histosols, 2136
 - mean organic C content, 2137
 - organic matter inputs
 - placement, 2134–2135
 - quantity, 2133–2134
 - species composition, 2134
 - physical and chemical influences, 2136
- Global diversity of species, 206
- Global food security challenge, 1545–1546
- Global Inventory Modeling and Mapping Studies (GIMMS), 582
- Global land use patterns, 1309–1311
- Global latitude, 1329
- Global positioning system (GPS), 1005, 1018, 1804
- Global resources, 1020
 - biomes, 1020
 - boreal forest biome, 1023
 - desert biome, 1023
 - polar biome, 1021–1023
 - soils, 1020–1021
 - temperate grassland and forest biome, 1023
 - tropics biome, 1023
- GlobalSoilMap, 1410
- Global warming, 442
 - agriculture and environmental sustainability, 2277
- Glomales (Zygomycetes), 980
- Glomalin, 1507–1508
- The Glossary of Soil Science Terms, 1388
- Glover equation, 122
- Glover model, 120
- Glutathione peroxidase, 1446, 1729
- GMDH. *See* Group method of data handling (GMDH)
- GMOs. *See* Genetically modified organisms (GMOs)
- GOC. *See* Geogenic organic carbon (GOC)
- Goethite, 8–9, 316, 317, 2003
- Goethite (FeOOH), 127, 425
- Goethite minerals, 1476
- Goethite (α - FeOOH), 8
- Goiter, 1446
- Gold tailings, 2289–2290
- Good soil structure, 114
- GOSSYM model, 2512
- Grade, structure, 2216
- Graded bunds, 748
- Graded-furrow tillage, 1566
- Gradient terraces, 2344–2345
- Grain for Green Project (GGP), 1375–1376
- Grain production, 1313
- Grain sorghum (*Sorghum bicolor* L. Moench), 261
- Grain yield, 1737
- Grain yields, 261
- Grama or mesquite grasses (*genus Bouteloua*), 828
- Gramineouswoody savannah (*campo sujo*), 345
- Grands Groupes de Références (GER), 391
- Granular soils, 1417
- Granulometric clay, 93
- Grassed waterways, 750, 763–764
- Grasses and forbs, 1020
- Grasslands (*campo limpo*), 345, 463
 - and fodder crops, 290–291
 - historic benefits, 463
 - utilization, 463
- Grasslands soils, 1029
 - classification, 1032
 - genesis of, 1030–1032
 - importance, location, and extent, 1029
 - properties, 1029–1030
- Grass strip hydrology, 1024
 - buffer strips on flow hydraulics, 1024–1025
 - buffer strips on sediment transport, 1025–1027
 - effect of buffer strips on chemical transport, 1027
 - theoretical interpretation of sediment transport, 1027
- Grass strips, 828
- Grass tetany, 1393
- Gravimetric sampling and oven drying, 2475–2476
- Gravimetric soil water, 2475
- Gray Luvisol, 386
- γ -ray spectroscopy, 1007
- Grazing, 1313
 - intensity, 1040
 - management practices, 1040
 - type and intensity, 1039
- Grazing lands, 1034–1035
 - GHG emissions, factors affecting
 - abiotic factors, 1036
 - biotic factors, 1038–1039
 - burning of savannas, 1039
 - farm management practices, 1038
 - grazing type and intensity, 1039
 - leguminous pastures, 1039
 - manure, 1039
 - rangelands, 1039
 - soil moisture, 1037
 - soil pH, 1037
 - soil temperature, 1036
 - soil texture, 1037

- Grazing lands—
 mitigation strategies, 1039–1040
 biochar/agrichar, 1040
 controlled grazing, 1040
 grazing intensity, 1040
 grazing management practices, 1040
 nitrification inhibitors, 1040
 rotational grazing, 1040
 soil conservation practices, 1040
 nitrogenous emissions, 1036
 carbon dioxide, 1036
 methane, 1036
 nitrous oxide, 1035
 mechanism of N₂O production, 1035–1036
- Great Green Wall, 650
- Green alga, 237
- Green-Ampt infiltration model, 1978, 2508
- Green-Ampt model, 1187, 1188, 1189
- Green ash (*Fraxinus pennsylvanica*), 2573
- Green fertilizer, 1313
- Green Food program, 1618
- Greenhouse effect, 620, 1048
 agricultural activities and GHG emissions, 1048–1049
 biological changes, 620
 climatical changes, 620
 greening of earth Hypothesis, 621
 managing world soils for mitigation, 1050–1051
 mulch farming, 1499–1500
 potential of world soils as C sink, 1049–1050
- Greenhouse gases (GHGs), 76, 77, 184, 195, 441, 886, 899, 926, 1034
 balance, 2232–2233
 emissions, 1377–1378
 peatlands, 1672
- Green leaf manure (GLM), 304
- Green malt, 1671
- Green manuring, 256
 no-till, 1379
- Green revolution, 555
- Green revolution: China North Eastern Plains, 1042–1043
 advancing maximum attainable yield in different regions by improving arable land quality, 1046
 close the arable land potential productivity gap, 1045
 decrease in ecosystems multifunction, 1043
 mapping city growth boundary, 1044
 mapping permanent prime cultivated land zone, 1044
 maximum attainable crop yields by providing subsidy incentive and prompting rural development, 1046
 raise yield to crop biophysical limits in different climate–soil–groundwater zones, 1045–1046
 restoration of land ecosystems multifunction services, 1043
 restore excessive marginal arable land expansion, 1043–1044
 restore local LCCMU, 1044–1045
 sustainable intensification, 1046–1047
 yield to biophysical limits by introduction of improved crop–soil management, 1046
- Green Revolution technologies, 1601, 1618
- Green technology, 244
- Green wood rights, 1356
- Grevillea robusta*, 827
- Greyzems, 392
- Griffith University Erosion System Template (GUEST), 2500–2501
- GRONingen MACHine for Chemical Simulations (GROMACS), 2166, 2167
- Gross climatic differences, 139
- Gross domestic product (GDP), 664
- Ground conductivity meter (GCM), 1005
- Ground limestone, 115
- Groundnut (*Arachis hypogea*), 224, 300
- Groundnut–finger millet rotation, 300
- Ground-penetrating radar (GPR), 1005, 1006
- Ground-truth data, 1316
- Groundwater, 2444
 depletion, 900
 remediation, 203
- Group method of data handling (GMDH), 1684
- Growing crops and residue on erosion and deposition, 69
- Growing media, 1053
 chemical and biological properties, 1054–1055
 inorganic substrates, 1055–1056
 materials used and growing media choice, 1055
 new developments, 1056–1057
 organic substrates, 1056
 physical properties, 1053–1054
 testing of, 1057
- Growing point, roots, 72
- Growth-promoting bacteria, 198
- GUEST. *See* Griffith University Erosion System Template (GUEST)
- Guidance religion, 1903
- Gujarat plain, 1159–1160
- Gully erosion, 1059
 classical, 1060
 control of, 1062–1063
 ephemeral, 1059–1060
 prediction of, 1062
 techniques of measurement, 1060–1062
- Gully control drainage line treatment structures, 750–751
- Gully fields on O'Bryan Ridge, 68
- Gully stabilization structures, 1988–1989
- Gypsic soils, 1064
 constraints, 1066
 fertility, 1066
 physical and engineering, 1066
 hypergypsic soils, 1064–1066
 occurrences and properties, 1064
 in world, 1064
- Gypsiferous soils, 1069–1070
- Gypsisols, 392, 409
- Gypsum (CaSO₄ · 2H₂O), 115, 824, 1068, 1178, 1205, 1362, 1478–1479, 2031, 2039, 2042, 2051, 2193, 2239
 coprecipitation, 1069
 crust, 1068
 formation, 1068
 global distribution, 1069
 properties, 1069–1070
 sulfate, genesis and origin of, 1068–1069
- H**
- Haber–Bosch process, 524, 1309
- Hairy vetch (*Vicia villosa*), 220
- Halloysite, 1476–1477
- Hand tillage, 2343
- Haploturbel, 990
- Hard-setting characteristic, 1666
- Hard-setting processes, 599
- Hard-setting soils, 1666
 assessment, 1073
 formation, occurrence and processes of, 1072–1073
 dispersion, 1073
 slaking under rapid wetting, 1073
 interparticle bonding, 1073
 MOR, 1073
 soil dispersion, measurement, 1074
 tensile strength, 1074
 wet sieving measurements, 1074
- Harmonized World Soil Database (HWSD), 2061
- Harvesting
 anthropogenic disturbances, 1820–1821
- H-curve, 1426
- Head loss, 1990
- Health
 and quality, 1076
 context of ecosystem health, 1076–1077
 recognition, healthy soil, 1076–1077
- Health management, soil
 agroforestry systems (AFSs), 1385–1386
 conservation agriculture, 1386
 integrated farming system, 1385
 integrated nutrient management, 1385
 liming, 1386
 micronutrient management, 1386
 organic farming, 1386
- Heat and water movement
 coupled, 1080
 soil freezing, 1080–1081
 thermal advection, 1081–1082
 water vapor movement, 1081
 theory, 1079
 soil freezing, 1079
 thermal advection, 1080
 thermally induced liquid flow, 1080
 water vapor movement, 1079–1080
- Heat capacity, 1083–1085
 factors affecting, 1085
- Heat dissipation, 2477
- Heat flux, 1086
 factors affecting, 1086–1087
 measurement techniques, 1087–1088
 surface energy balance, 1086
- Heat pulse probes, 860
- Heavy fraction, 1953
- Heavy liming, 38
- Heavy metal-contaminated arable land, 1763
- Heavy metals, 1089, 1426
 abundance in rocks and soils, 1089–1091
 biological effects, 1091–1092
 general chemistry, 1089
 in organic fertilizers, 1622
 physical and chemical properties, 1090
 plant uptake of, 1622
- Heavy-textured soils, 1665
- Hedge row barriers, 747

- Hematite, 2003
- Heme- and non-heme-Fe diet, 1447
- Hemihydrate ($\text{CaSO}_4 \cdot 0.5\text{H}_2\text{O}$), 1068
- Henin index, 63
- Herbaceous wind barriers, 2611
- Herbicides, 499, 1567, 1717
costs, 245
and insecticide resistance, 245
spraying, 229
- Herbivores as plant-feeding nematodes, 1526
- Heterogeneity
landform creation, 1344–1346
landform modification, 1346–1347
- Heterostigmata, 1489
- Heterotrophic oxidation, 127
- Hg contaminations, 1760–1761
- Hidden carbon costs (HCCs), 307
- Hidromórfico, 380
- Hieronymus Bosch and his disciples, 169
- High-activity and low-activity clays, 427
- High-altitude belt, 101
- High-energy stress, 61, 62
- High rainfall, 1383, 1386
- High resilience, 1919
- Highspeed computer processing, precision
agriculture, 1804
- High-speed computer processing systems,
precision agriculture, 1805
- High-tech monoculture farming, 1121
- Hills, 1339–1340
ecosystem, 1383
ranges (ghats), India, 1157
- Hillslope, 1340
landscapes, 1372
model
elements, 1328
water movement in, 1329
- Hilly gully landscapes, 1372
- Himalayan mountains and Siwaliks, 1158
- Hindered gas exchange, 1954
- Hisingerite, 1469
- Histels, 990
- Histic Andosols, 134, 136
- History, 1093
1927–2000, 1098–1101
9000 B.C.E. – 1500 C.E., 1093–1094
1500 C.E. – 1880 C.E., 1094
1880 C.E. – 2000 C.E., 1094–1096
early to mid-20th century, 1103
agricultural chemistry, 1103
agricultural geology, 1103–1104
emergence of soil science, 1104–1105
- Histosols, 105, 128, 130, 134, 136, 159, 392,
409, 410, 2136
definition, 1107
description, 1107
distribution, 1107
formation, 1108
properties, 1108–1109
uses and problems, 1109–1110
- Holistic management, 707
- Hollandite, 2006
- Holtan equation, 2509
- Homogeneous manure solids, 1407
- Hordelymus europaeus*, 143
- Horizontal–vertical sieving, 62
- Hornblende, 1473–1474
- Horticultural crops, 291
- Horton equation, 2508–2509
- Hot peroxide acidity, 6
- Hotspots of global erosion, 768
- Huang–Huai–Hai (HHH) plain, 899
- Human-assisted rehabilitation, 661
- Human-caused disturbances. *See*
Anthropogenic soil disturbances
- Human culture, 1112
agricultural transformation, 1112–1113
early development, 1112
imperative to improve soil management, 1113
land degradation, 1113
- Human health, 1118–1119
agronomic management for better
nutrition, 1447
antitnutrients management, 1447
biofortification, 1447
cultural practices, 1120–1121
dietary modification, 1446–1447
effects, acid rain, 13
fortification, 1446
healthy soil, healthy food, 1119–1120
microenriched fertilization/ferti-fortification
of crops, 1447
micronutrients content in seeds and
bioavailability, 1447
soil contact, 1119
supplementation, 1446
- Human immunodeficiency virus (HIV), 1115
- Human-induced erosion periods, 794
- Human-inhabited regions, 1113
- Human interventions, spiritual aspect of,
1904–1905
- Humanity, 1113
- Human-operated land management
systems, 1119
- Human society and planet, 1127
contemporary perspective, 1129–1130
historical perspective, 1127–1129
- Human society and soil, 1123
ancient societies, 1123–1124
soil occupation or militaries, 1124
soil parasitism, 1124
soil pathogens, 1124
modern society, 1124–1125
soil–human symbiosis, 1125
- Humid climate SAS, 2038
- Humid environments, 1210
- Humid region irrigation, 1211
- Humid tropics, 1745
- Humification, 1050, 1359, 2112–2113
- Hummock retrogression, 151
- Humo-Ferric Podzol, 387
- Humus, 558, 1747
components, 1359
humus-constituent concept, 2124–2125
humus-horizon concept, historical account,
2123–2124
meanings of, 2123
and sesquioxides, translocation, 2200–2201
- Humus ruralis*, 396
- HWSD. *See* **Harmonized World Soil Database (HWSD)**
- Hybrid poplar (*Populus* spp.), 210
- Hydrated lime [$\text{Ca}(\text{OH})_2$], 9–10
- Hydrate formation zone (HFZ), 296
- Hydration, 348
- Hydraulic adjustment, 1024
- Hydraulic conductivity, 120, 132, 173, 510,
562, 1133–1135, 1683, 1991–1992
infiltration, 1189
- Hydraulic gradient, 66, 1342, 1990
- Hydraulic properties, 561
infiltration and hydraulic conductivity, 562
sodic soils, 2041
water retention, 561–562
- Hydrobiotite, 1477
- Hydrocarbon degradation, 1773
- Hydrogen bonding, 423, 426
- Hydrolases, 737
- Hydrologic cycle, 1137, 2550
local perspective at smaller space and time
scales, 1137
dynamics of runoff source areas, 1137
partial wetting, 1138
predicting infiltration excess runoff,
1137–1138
soils and runoff generation processes, 1137
time of runoff generation, 1138
regional scale perspective for large space and
time scales, 1137
- Hydrologic fluxes of nutrients, 703
- Hydrology, 2572
- Hydrolysis, 6, 348, 2560
- Hydrolytic reactions, 1772
- Hydromorphic podzol, 1749
- Hydropedology, 1140
applications and links to other disciplines,
1141–1143
characteristics, 1140–1141
fundamental issues, 1141
- Hydrophilic compounds, 1144–1145
- Hydrophobic compounds, 1144–1145
- Hydrophobic contaminants, 202
- Hydrophobicity, 194, 1147, 2470, 2546
assessment of, 1148
degrees of, 1148
effects of, 1148–1149
natural soils, 1149
wettability, and capillarity, 1147–1148
- Hydrophobic membrane probe (HMP), 45
- Hydrothermal carbonization (HTC), 1056
- Hydrous micas, 1477
- Hydrous oxides, 1471
for Al, 558
- Hydroxide ions, 354
- Hydroxy-Al, soil pH, 1695–1696
- Hydroxy-Al interlayered vermiculites and
smectites, 424
- Hydroxy-Al polymeric components, 424
- Hydroxy-Al polymers, 1478
- Hydroxyapatite, 1512
- Hydroxy-Fe polymers, 1478
- Hydroxy-interlayered vermiculite (HIV) and
smectite, 1478
- Hydroxylamine (NH_2OH)
chemical decomposition of, 1564
- Hydroxyls, 353
- Hyperaccumulator, 1719–1720, 1721
- Hypergene silicates
structure, 1954
transformation, 1954
- Hyphal fragments, 504

- Hypogastrura*, 449
 Hypogastruridae, 50
 Hypomagnesemia, 1393
 Hypoxia, 526
 Hysteresis, 2539
- I**
- I, mineral nutrition disorders, 1446
- Iceland, 134
 afforestation, 286
 andic soils and cryoturbation, 136
 classification and soil characteristics, 134–135
 C sequestration potential, 285
 freely drained soils, 136
 geological processes, 286
 land use, cooperation with farmers, and funding, 285–286
 physiography, 134
 restoration of wetlands, 286
 soil erosion, 136–137
 Vitrisol classification perspectives, 136
 wetland soils, 135–136
- Icelandic deserts, 136
- Icelandic soils, 134, 624
- IGCC. *See* Integrated gasification combined cycle (IGCC)
- IGP. *See* Indo-Gangetic plain (IGP)
- Illites, 1477
- Illuviation
 of clays, 1359–1360
- Ilmenite, 1475
- ImagePro image analysis software, 800
- Immobile nutrient, 1713
- Immobilization, 192
 N, 711
- Imogolite, 97
- Inactivated enzymes, 1758
- Incentives, soil conservation
 causes and constraints, 2082
 conservation programs principles, 2082
 impact of, 2082–2083
 types of, 2081
 use of, 2081–2082
- Inceptisols, 105, 410, 1160–1161, 2293, 2371
 Aquepts, 1152–1153
 correlation, 1152
 Cryepts, 1153
 Gelepts, 1153
 occurrence, 1151–1152
 soil genesis, 1154–1155
 Udepts, 1153–1154
 use and management, 1155
 Ustepts, 1153
 Xerepts, 1153
- Incident radiation, 90
- Incubation
 chloroform fumigation, 1429–1430
 microwave irradiation incubation and extraction methods, 1430
- Index of consistency (I_c), 172
- Index of plasticity (I_p), 171–172
- Index of shrinkage (I_{Sch}), 172
- India, 1157
 Alfisols, 1161–1162
 Aridisols, 1162
 central uplands, 1157
 Deccan plateau, 1159
 Eastern plateau, 1159
 Gujarat plain, 1159–1160
 hill ranges (ghats), 1157
 Himalayan mountains and Siwaliks, 1158
 Inceptisols, 1160–1161
 indo-gangetic plains (IGP), 1157–1158
 islands, 1158
 Mollisols, 1162
 northeastern hill ranges and valleys, 1158
 soil–physiography relationship, 1157
 Ultisols, 1162
 Vertisols, 1162–1163
 western plains, 1159
- Indian classification systems, 398
 climate-based soil classification, 399
 fertility-based classification, 398
 revenue system, 399–400
 suitability of crops, 398–399
- Indian Council of Agricultural Research (ICAR), 1157
- Indian mustard and canola (*Brassica napus*), 1718
- Indicators, quality
 criteria for selecting
 combining, 1861
 interpretation, 1859–1860
 minimum data set, 1859
 time and spatial aspects, 1859
 trends, 1861
 productivity, 1862
- Indicators, soil quality related to water quality
 influence of management, 1868
 selection, 1868
- Indigenous soil management, 1164
 classifications and fertility indicators, 1165
 examples, 1165
 integrated soil management, 1164
 flexibility and dynamism, 1164–1165
 multifunctionality, 1164
 practices, 1165–1166
 managing limited labor, 1166
 managing soil fertility and soil water, 1166
- Indigenous systems of soil structural management, 510
- Indigenous technical knowledge (ITK), 732
 applications of ITK TO SLM, 734–735
 mediterranean, 732–734
 plow implement of millennia, 732–734
 sustainable land management (SLM), 732
- Indigofera spicata*, 820
- Indigofera teysmannii*, 747
- Indirect quantification of macropores, 1389
- Indirect tension, 52–53
- Indo-Gangetic plain (IGP), 1168–1169
 alluvial plain, 1170–1172
 potentials and constraints, 1172–1173
 arid aeolian plain, 1173
 potentials and constraints, 1173
 Assam (Brahmaputra) valley, 1174–1175
 geographic settings, 1169
 geological age of soils, 1169–1170
 physiography and agroecological subregions (AESRs), 1170
 piedmont plain and tarai region, 1170
 potentials and constraints, 1170
 potentials and constraints, 1175
 semiarid transitional aeolian plain, 1173–1174
 potentials and constraints, 1174
 soils of, 1170
- Indo-gangetic plains (IGP), 1157–1158
- Indole-3-acetic acid (IAA), 363
- Induced erosion, 2625–2627
- Induced systematic resistance, 1436
- Indurated carbonate nodules and clasts, 340
- Industrial waste, 1178
 industrial by-products, 1178
 coal combustion by-products, 1180
 construction wastes, 1180
 land application, 1180–1181
 metal refining and processing, 1178
 mineral processing, 1178–1179
 pulp and paper manufacture, 1180
 regulatory agencies, 1181–1182
 industrial products, 1178
 minimizing production, 1182
- Industrial waste materials, liming, 1363
- Inelastic neutron scattering (INS), 319
 basics, 319–320
 results, 320–321
- Inelastic scatterings, 320
- Inference systems, 675, 1679–1680
- Inferential method, 881
- Infertile Quartzipsammments (*Solo Arenosquartzos Profundos*), 112
- Infertile sediment, 111
- Infiltration, 1183, 1187, 2472–2473
 model parameters, 1187–1188
 effective hydraulic conductivity, 1189
 initial water content, 1189
 porosity, 1188–1189
 wetting front suction, 1189
 models, 1187
 soil surface degradation, 1183
 base saturation, 1184
 land use and fertilization, 1183–1184
 salinization, degradation by, 1184–1185
 soil tillage systems, 1185
 subsoil compaction control, 1185
 trampling, 1185
- Infiltration and hydraulic conductivity, 562, 2443
- Infiltration and runoff, 2550
- Infrared gas analyzers (IRGA), 859
- Infrared (IR) spectroscopy, 97
- Infrared thermometers (IRTs), 2302
- Infusoria, 237
- Inhabiting mites, 1488
- Inherent soil properties, 1862–1863
- Initial rapid reactions, phosphorus, 1702
- Initial water content, infiltration, 1189
- Inner-sphere surface complexes, 1427–1428
- Inorganic C, 1450
- Inorganic carbon, 1199, 1210
 arid region soils, 1211–1212
 atmospheric carbon dioxide (CO_2), 1212
 composition, 1199–1200
 fertilization, 1210–1211
 formation, 1200
 biogenic model, 1201–1202
 carbonate formation, models, 1201
 pedogenic vs. geogenic carbonate, 1200–1201

- per ascensum model, 1201
 - per descensum model, 1201
 - in situ* model, 1201
- humid region irrigation, 1211
- isotopes in calcic soils, 1214–1215
- land clearing and cropping, 1210
- pedogenic carbonate accumulation, 1213–1214
- sodic soil reclamation, 1212
- Inorganic cations, 56
- Inorganic chemicals, 1776–1779
- Inorganic mulches, 1495
- Inorganic nutrient transformations, 1435
- Inorganic pollutants
 - classes, 1769–1770
 - concentration ranges, 1769–1770
 - pathways of contamination, 1770
- In-season N management, 1540
- Insects, 138
 - survival, 1217
 - behavioral adaptation, 1217
 - corn rootworms, 1218–1219
 - freeze tolerance vs. freeze susceptibility, 1217–1218
 - physiological adaptation, 1217
- Insects and disease
 - natural disturbances, 1820
- In situ* residue retention (IRR), 1384
- In situ* soil reclamation
 - in situ* acid mines oil remediation, 1463–1464
 - in situ* sodic mines oil remediation, 1463
- In situ* subsurface bioremediation, 202
- Insoluble P-containing minerals, 1700
- Institute for Land Reclamation and Improvement (ILRI), 20
- Institutions, 1355, 1356
- Integrated farming system, 1385
- Integrated gasification combined cycle (IGCC), 1800–1801
- Integrated management systems, 1952
- Integrated nutrient management (INM), 940, 1221–1222, 1383, 1385
 - components, 1222
 - biofertilizers, 1223
 - biosolids, manures, composts, and by-products, 1224
 - BNF, leguminous crop rotations, agroforestry, 1223–1224
 - mineral and synthetic fertilizers, 1222
 - nitrogen, 1222–1223
 - phosphorus, 1223
 - potassium, 1223
 - environmental concerns
 - nutrient losses, 1224–1225
 - soil fertility degradation/improvement, 1225
 - toxic accumulation, 1225
 - fertilizer management, 1227–1228
 - nitrogen, 1228
 - other nutrients, 1229–1230
 - phosphorus, 1228–1229
 - potassium, 1229
 - sustainable agriculture, 1227
- Integrated soil management, 1164
 - flexibility and dynamism, 1164–1165
 - multifunctionality, 1164
- Integrated Survey in Australia, 1000
- Integration of livestock, 896
- Intensity factor, plant nutrients, 1731–1732
- Interaggregate porosity, 1782
- The Intergovernmental Panel on Climate Change, 1546
- Intergovernmental Panel on Climate Change (IPCC), 76, 2085
- Intergovernmental Panel on Land and Soil (IPLS), 2085
- Intermediate-level microterrace, 296
- Intermittent irrigation, 2043
- Internal P requirement, 1737
- International Centre for Research in Agroforestry (ICRAF), 343
- International Committee on Anthropogenic Soils (ICOMANTH), 2299
- International Congress of Soil Science, 1105
- International Council of Control of Iodine Deficiency Disorders (ICCIDD), 1120
- International Desertification Conference, 630
- International Federation of Organic Agriculture Movement (IFOAM), 1619
- International Geographical Union (IGU), 1240
- International Land Reclamation Institute (ILRI), 17
- International law on soil, 1352–1353
- International Organization for Standardization, 1855
- International Society of Soil Science (ISSS). *See* International Union of Soil Sciences (IUSS)
- International soil conservation strategies, 2084–2085
- International Soil Reference and Information Centre (ISRIC), 1232
 - applied research, 1234–1235
 - history, 1232
 - world data centre for soils, 1232–1234
 - world soil museum, 1232
- International Soil Science Society, 1665
- International Union for Conservation of Nature (IUCN), 651
- International Union of Biochemistry and Molecular Biology (IUBNB), 1240
- International Union of Biological Sciences (IUBS), 1240
- International Union of Geodesy and Geophysics (IUGG), 1240
- International Union of Geological Sciences (IUGS), 1240
- International Union of Soil Sciences (IUSS), 596, 1104, 1237
 - establishment of, 1236–1239
 - historical context, 1237–1238
 - new scientific structure, 1239–1240
- Interparticle bonding, 1073
- Interpolation efficacy
 - assessment, quality, 1847
- Interpretation, 1313
- Interrill erosion, 816
- Inter-rill erosion, 846
- Intervinal chlorosis, 1393
- Intra-aggregate porosity, 51, 1782
- Intrinsic permeability, 1991
- Intrinsic remediation, 202
- Inverse modeling, empirical scaling, 1979
- Invertebrates and fungi, 238
- Iodine (I), 446
 - deficiency, 446–447
 - forms of, 446
 - plant availability, 446
- Ion exchange, 1241
 - anion exchange, 1243–1244
 - cation exchange, 1241
 - amelioration of saline and alkali soils, 1243
 - nature and origin of, 1241
 - plant nutrition, 1243
 - soil development and acidification, 1243
 - cation exchange equilibria and cation selectivity, 1242–1243
 - exchangeable cations, cation exchange capacity, and base cation saturation, 1241–1242
- Ionic bonding, 2049
- IPCC. *See* Intergovernmental Panel on Climate Change (IPCC)
- IPLS. *See* Intergovernmental Panel on Land and Soil (IPLS)
- Irrigation water use efficiency, 1260
- Iron, 56
 - and aluminum oxides, 56, 1476
- Iron disulfide (FeS₂)
 - formation, 1832
 - framboidal, scanning electron microscopic image, 1832
 - implications, 1833
 - mineralogy and crystal chemistry, 1831
 - occurrence, 1831–1832
 - reactivity, 1832–1833
 - schematic image, 1832
 - semiconductor, 1831
 - sulfuric acid, 1831
- Iron (Fe) oxides
 - adsorption of oxyanions, 2005
 - chemical and physical properties, 2005
 - minerals, 2003
 - pH dependent charge, 2005
 - properties of, 2003–2005
 - wet soil features, 2005
- Iron oxides, 1245
 - classification and occurrence, 1245
 - isomorphous substitution, 1245
 - formation and chemistry, 1247–1248
 - methods of identification, 1245
 - bulk analyses, 1245–1246
 - selective extractions, 1246–1247
 - nano Fe minerals, 1248
- Ironstone, 1743–1744
- Irreversible extension of desert land, 657
 - adaptive arid land strategy, 662
 - agroforestry, 662
 - air pollution, 658
 - contour lining, 661
 - definition, 657
 - degradation processes, 658–661
 - desertization, 658
 - deserts, origin, 657–658
 - facts and fallacies, 657
 - fluctuations and trends of climatic variables, 658
 - land surface albedo, 658
 - managed restoration, 661
 - natural amelioration, 661
 - range reseeding, 662

- Irreversible extension of desert land—
ripping and subsoiling, 661–662
surface roughness, 661
wildlife management, 662
- Irrigated agriculture, 159
nutrient management, 1571
- Irrigated croplands, 2552
- Irrigation, 1038, 1128, 1250, 1264
ancient origins and importance, 1264–1265
consumptive use and design, 1250–1251
existing status, 1265–1266
and fertilizers, 524
issues affecting the future, 1266–1267
modernization, 1265
soil moisture management, 1251–1252
irrigation depth, 1252
irrigation interval, 1252
system design, management, and
scheduling, 1252
application depth and frequency,
1254–1255
design concepts, 1252–1253
drainage, 1254
salinity control, 1253–1254
water, 119, 561
water storage capacity of soils, 1250
root depth, 1250
soil water-holding capacity, 1250
- Irrigation, rice paddies, 1951
- Irrigation, urban wastewater for, 2407
heavy metals, 2408
bioavailability and contamination of food
chain, 2408–2410
for livelihood, 2407–2408
management options, 2410
- Irrigation and cropping system techniques, 1539
- Irrigation efficiency, 1256
application efficiency, 1257
irrigation uniformity, 1258–1259
Christiansen's uniformity coefficient, 1259
emission uniformity, 1259
low-quarter distribution uniformity, 1259
seasonal irrigation efficiency, 1257–1258
storage efficiency, 1257
water conveyance efficiency, 1256–1257
water use efficiency, 1260
irrigation water use efficiency, 1260
- Irrigation farming, sodic soils
drainage, 2043
hydraulic properties, 2041
irrigation practices, 2043
nutrient status, 2042
soil management, 2042
water quality, 2042
- Irrigation-induced erosion, 782–785
- Irrigation-induced soil erosion, 1261
sprinkler irrigation, 1262
surface irrigation, 1261–1262
surface irrigation and rainfall erosion
differences, 1262–1263
- Irrigation water quality and fertilizer
application, 1598
- IRTs. *See* Infrared thermometers (IRTs)
- Isomorphous substitution, 422
- Isotherms
for metal ions, 1426–1427
- Isotopes emitting gamma radiation, 1874
- IUCN commission on environmental law, 1354
- Iznik Ceramic, 1419
- J**
- Jarosite [(H₃O, K, Na) Fe₃(OH)₆(SO₄)₂],
8, 2239
- Jarrah (*Eucalyptus marginata*) woods, 185
- Jean Dubuffet, 169
- Jhum, 1273–1274, 1383
fertility and nutrient budget under shifting
agriculture, 1281–1282
knowledge systems for land use management,
1282–1283
in North East India, 1274–1276
proposed alternative program, 1278
soil erosion and nutrient loss, 1276–1278
shifting agriculture and sustainability, 1283
contour pathway, 1283
incremental pathway, 1283
- Jungle vegetation, 113
- K**
- Kabro (*Ficus lacor*), 734
- Kalibugan, 403
- Kames–glacial deposition, 1340
- Kaolinite, 105, 316, 422, 426, 1743
clay, 114
and halloysite, 1476–1477
- Kastanozem Phaeozem, 392
- Kastanozems, 409
- Kattekleigronde, 17
- Keli (fertile soils), 403
- Kendrick mass and defect (KMD)
analysis, 2156
- Kentucky N and phosphorous (P) index,
1547–1548
- Kepir, 403
- Keshan disease, 1446
- K-feldspars, 1791
- K-fertilizer-consuming countries, 1229
- K fertilizers, 912
- K fixation, 1794
- Khanayo (*Ficus semicordata*), 734
- Kissimee River, Florida, United States,
2553–2554
- Kiwifruit (*Actinidia deliciosa* L.), 362
- Kizkalesi (Korykos), 732
- Koch, Robert, 176
- Kocuria rhizophila*, 42
- Köppen–Geiger–Pohl and Meigs systems, 157
- Kostiakov equation, 2508
- Kozeny–Carman law, 1978
- Kozeny–Carman theory, 1135
- K (potassium), 255
- K-rich farm effluents, 1393
- Kriging, 1015, 2431
- Kyoto Protocol, 76, 77, 79, 81, 2078–2079
- L**
- LA-AMS. *See* Laser ablation aerosol mass
spectrometry (LA-AMS)
- Labels, food producers types, 1619
- Labile P, 1701
- Lactobacillus*, 244
- Lag sediments, 1643
- LA-IRMS. *See* Laser ablation isotope ratio mass
spectrometry (LA-IRMS)
- Land capability, 1285, 1286
classification, 1288
classes, 1289–1290
historical perspective, 1288–1289
subclasses, 1290
units, 1290
kinds of systems, 1285
development potential, 1285–1286
land capability, 1286
soil potential, 1287
- Land clearing and cropping, 1210
- Land cover
databases, 1316
mapping, 1313–1314
maps, accuracy of, 1316
monitoring, 1314–1316
- Land degradation, 640, 648, 1113, 1383
in arid, 628
drylands and non-drylands, 608–609
- Land degradation and desertification (LDD), 608
cause of, 609–611
driven poverty, 612
- Land desertion, 658
- Land disturbance, 36
- Land ethics, 342–344
- Land evaluation, 1291
development, 1292
historical overview, 1291
soil surveys, 1292–1293
- Land evaluation, pedometrics, 1680–1681
- Land Evaluation and Site Assessment
(LESA), 1320
- Landfarming, 202
- Landfill
cap, 1322
cells, 1322
design, 1322
design and operation, 1322
factors affecting plant growth, 1323–1324
gas, 1323
leachate, 1323, 1324
- Landforms, 1305–1306, 1333, 1339, 1372, 1373
classification, 1295
description and morphometry, 1294–1295
development, 1295
geomorphic process environments,
1331–1332
and mapping, 1295–1296
modification, 1346–1347
- Land husbandry, 1297
beneficial effects, 1300
defining, 1297–1298
agro-ecologic components, 1298–1300
socioeconomic components, 1300
implications, 1300–1302
- Land management, 15, 1327–1330
- Land manager, 1936
- Land Resource Information Systems
(LRISs), 1320
- Land restoration
biophysical factors limiting, 1305–1306
components, 1304
end land use, 1304–1305
mine rehabilitation, 1306
plant establishment, 1306
substrate properties, 1306
sustainability, 1306

- Landsat multispectral scanner (MSS) data, 1973
- Landscape, 1332
- backslope, 1327
 - classification, 1333–1334
 - evolution and climatic change, 1745
 - explicit geomorphic context, 1331
 - footslope, 1328
 - geomorphic descriptors, 1340–1342
 - heterogeneity, 1344–1347
 - hills, 1339–1340
 - hillslope, 1340
 - informal context, 1331
 - landform, 1339
 - and land use, 1329–1330
 - morphology, 1328–1329
 - on-farm and *in situ* water management, 1330
 - physiographic context, 1331
 - position and soil productivity, 2337–2338
 - regrading, 1462
 - relationships, 1979
 - sensitivity, 1344
 - shoulder, 1327
 - slope orientation/aspect, 1329
 - soil, 1339
 - soil erosion, risks, 1330
 - in soil inventory, 1332
 - SOM
 - C balance, non-uniform effects, 2143
 - erosion, 2144
 - implications, 2144–2145
 - lateral distribution, 2142–2143
 - practices, patchwork application of, 2143
 - temporal distribution, 2143
 - tillage, 2143–2144
 - vertical distribution, 2142
 - spatial scales, 1337–1338
 - summit, 1327
 - terminology, 1327
 - timescales, 1336–1337
 - toeslope, 1328
 - unit boundaries, 1339
 - and vegetation associations, 1409
 - water, 1140
- Landscape function analysis (LFA), 1901
- Land-scoured ridges, 69
- Land scouring, 69
- and gully fields, 68–69
- Landslide
- causes, 1350
 - characteristics, 1349–1350
 - classifications, 1350
 - disaster on Highway 3 in Taiwan, 1350
 - distribution, 1350
 - hazard zonation (LHZ), 2360
 - investigation, 1350
 - mitigation measures, 1350
 - slope failure and, 1349
 - susceptibility mapping, 2360
- Land susceptible to desertification, 618
- Land use
- agriculture, origins of, 1308
 - and agroclimatic conditions, 1383
 - in Brazilian semiarid, 1377–1378
 - and climate change impacts, 1378–1380
 - colonialism, legacy of, 1309
 - databases, 1316
 - distributions, 1017
 - and fertilization, 1183–1184
 - fluctuations in agricultural uses, 1308–1309
 - global land use patterns, 1309–1311
 - indexing/ranking schemes, 1320
 - and landscape, 1329–1330
 - lumped parameter models, 1320
 - making land more productive, 1309
 - mapping, 1313–1314
 - maps, accuracy of, 1316
 - monitoring, 1314–1316
 - systems, 1316
 - forestry and agroforestry, 2446
 - irrigated crops, 2446
 - pasture, 2446
 - rain-grown annual crops, 2445–2446
- Large-scale atomic/molecular massively parallel simulator (LAMMPS), 2167
- La Rochelle surroundings area (France), 677, 678
- Larvae of beetles, 205
- Laser ablation aerosol mass spectrometry (LA-AMS), 2075–2076
- Laser ablation isotope ratio mass spectrometry (LA-IRMS), 2073–2075
- Laser-induced breakdown spectroscopy (LIBS), 319, 323
- high-resolution LIBS methodology, 323–324
- Lasius flavus*, 152
- Lasius flavus* mounds, 150
- Lasius neoniger*, 150
- Lasius niger*, 152
- Latent heat, 1079
- Latent policy, 2026, 2027
- Lateral flow of drop water, 1884
- Laterite, 1416–1417
- Laterites, 1743
- Lateritic gravel, 1743–1744
- Latitudinal distribution of the areas of alpine life zone, 102
- Latosols, 1397
- Latossolo Amarelo, 380
- Latossolo Bruno, 380
- Latossolo Ferrífero, 380
- Latossolos, 111, 381, 382
- Latossolo Variação Una, 380
- Law
- markets, 1356
 - myths, 1356
 - property, 1355–1356
- Layered (platy) petrocalcic horizons, 1691
- 1:1 Layer minerals, 422–423
- 2:1 Layer minerals, 423
- 2:1:1 Layer minerals, 424
- Layer silicate clays, 128
- LCM. *See* Leaf chlorophyll meter (LCM)
- L-curve, 1427
- LDD. *See* Land degradation and desertification (LDD)
- Leachate, 190, 1323, 1324
- Leaching, 560, 1253, 1358, 1544, 1821, 2133–2134
- Leaching Estimation and Chemistry Model (LEACHM), 1319, 2624
- LEACHN, 2426
- Lead arsenate, 1759
- Leaf chlorophyll meter (LCM), 1540
- Leaf-cutting ants, 151
- Leaf tip feeding, 1553
- Least limiting water range (LLWR), 2530–2531
- methodology and applications, 2531–2533
 - opportunities and limitations, 2533–2534
 - strategies to enhance, 2533
- Leaving plant residues, 917
- Legal and institutional conservation
- mechanisms for soil, 1353–1354
- Legislation, soil, 1352
- Legume agronomy, 228
- Legume/rhizobium symbioses, 233
- Legumes, 223–224, 1703
- Legume yield, BNF, 227–228
- legume agronomy, 228
 - pest control, 229
 - plant density and row spacing, 229
 - planting time, 228–229
 - rhizobial inoculation, 228
 - soil constraints and plant nutrients, 229–230
 - soil water, 228
 - species/cultivar choice, 228
- Leguminous CCs, 485
- Leguminous crops, 256
- Leguminous pastures, 1039
- Leguminous plants, 47
- Leguminous species, agroforestry system, 1379–1380
- Lemna minor*, 1324
- Lentil (*Lens culinaris*), 220, 224
- Leopold, Aldo, 849
- Lepidium sativum*, 1324
- Lepidocrocite, 2003
- Leptosols, 134, 136, 392, 409
- Leptospirillum*, 9
- Leptospirosis, 238
- LESA. *See* Land Evaluation and Site Assessment (LESA)
- Lettuce, 1703
- Leucaena leucocephala*, 827
- Leuconostoc*, 244
- Levantine forests, 732
- Levee-protected bottomlands, 68, 70
- Levee repair, sand delta removal, and crater lake filling, 70
- Levee saturation and topping, 66–67
- LFA. *See* Landscape function analysis (LFA)
- LIDAR. *See* Light detection and ranging (LIDAR)
- Ligands, 1481
- Light detection and ranging (LIDAR), 1062, 1806
- Light non-aqueous phase liquids, 1993
- Light-textured soils, 1665
- Lime, 115, 823–824, 2051
- Lime crusts, 1691
- Lime-induced reductions in tissue, 1394
- Limestone
- bedrock, 340
 - efficiency, 1363
 - purity and fineness, 1363–1364
- Liming, 1386
- materials, 1362–1364
- Linear regression analysis, 1813
- Linking hydrogeology, 1142
- Lipid peroxidation, 1729
- Liquidambar formosana*, 1323
- Liquid limit (atterberg limits), 171
- Liquid–solid contact angle, 2546

- Liquid solvent extractions (LSEs), 2177–2178
 Lithiophorite, 2006
 Litter, 2112, 2115
 decomposition, 1435
 inputs, 1675
 nutrient content, 1938
 Litter p, roduction, 1938
 Littleton/Englewood biosolids, 1408
 Livestocks
 grazing, 159, 624
 land use in, 1378
 manure for global warming
 animal wastes, 1401
 biomass carbon transformation, 1401
 GHG emission, 1400
 manure produced annually, 1401
 production, 1034
 systems, 118
 Lixisols, 392, 409
 Lizardite, 1999
 LLWR. *See* Least limiting water range (LLWR)
 Loess
 characteristics, 1367–1368
 cultural and environmental aspects, 1368
 deposition, 1372–1373
 distribution, 1366–1367
 origin of, 1365–1366
 vertical slope in, 1367
 wind-blown silt, 1368
 Loess–Paleosol profile, 1373
 Loess plateau
 area location, 1371
 climate condition and water resource, 1373
 erosion control on, 1375–1376
 formation, 1372–1373
 geology and landform, 1373
 high plain with deep-cutting gullies, 1371
 hilly gully landscapes, 1372
 Loess–Palesol sequence, 1370
 sediment delivery, 1370
 soil erosion, 1370, 1374–1375
 vegetation and soil, 1373
 Loess Plateau of China, 744
 Loop-within-a-loop, N cycle, 1849
 Loosening, 2209–2210
 Loss, soil, 1330
 Loss on ignition (LOI), 2060
 Low-Altitude Stationary Surveillance
 Instrumental Equipment, 1808
 Low-C agriculture technologies, 1377
 Low-cation exchange capacity, 1821
 Low cationic retention from fertilizers, 2437
 Low-energy fragmentation, 61
 Low energy precision application (LEPA)
 irrigation system, 1812
 Lower Himalayas, soil management
 annual soil erosion, 1383
 land use and agroclimatic conditions, 1383
 North Eastern Region (NER), 1382
 rainwater management, 1386
 shifting cultivation, 1383
 slash and burn agriculture, 1383
 SOC under various land use systems,
 1384–1385
 soil fertility and SOC status, 1383–1384
 soil health management, 1385–1386
 soil organic matter (SOM) content, 1382–1383
 soil quality, maintaining and enhancing, 1383
 temporal and spatial variations, 1382
 Lowland ice-wedge polygon landscape, 1755
 Low nutrient-supplying capacity, 2437
 Low-quarter distribution uniformity, 1259
 LRISs. *See* Land Resource Information Systems (LRISs)
 LSEs. *See* Liquid solvent extractions (LSEs)
 Lucerne flea, 449
Lumbricus terrestris, 237
 Lupine (*Lupinus albus*), 224, 227, 494
 rhizosphere of, 2104
 Lupins (*Lupinus* sp.), 530
 Luvisolic soil, 385
 Luvisols, 392, 409
 Luvisols, 381, 382
 Luxury consumption, 1577
 Lyases, 737
 Lyotropic series, 1578
 Lysimeter measurement principle, 489
 Lysimeter method, 488–489
M
Machilus thunbergii, 1325
 Mackey, Brendan, 849
 Macroaggregate formation, 180
 Macrofauna, 138, 871, 877, 1031
 ants, 878
 earthworms, 877
 burrows, 877
 casts, 877
 in soil food webs, 1627–1628
 termites, 877–878
 Macroinvertebrates, 207
 Macronutrient nanofertilizers, 1511–1512
 Macropores, 681
 density and continuity, 1388
 indirect quantification of, 1389
 mapping, 1389, 1390
 origin of, 1388
 quantification of, 1388–1389
 water movement and chemical and gas
 transport, 1389
 Macroporosity, 505, 1388, 1783
 See also Macropores
 Maghemite, 2003
 Magnesian calcites, 335
 Magnesium
 in animals, 1393
 availability to plants, factors affecting, 1394
 fertilizers, 1393–1394
 in plants, 1392–1393
 in soils, 1392
 Magnesium in plant nutrients, 1392–1393
 Magnetite, 2003
 Maibolt, 17
 Main boundary fault (MBF), 2360
 Main central thrust (MCT), 2360
 Maintenance of soil cover, 513
 Maize, 936
 Maize (*Zea mays* L.), 299, 501, 523
 Malaz, 403
 Male sterility, 245
 Malnutrition
 agronomic management for better
 nutrition, 1447
 antitnutrients management, 1447
 biofortification, 1447
 dietary modification, 1446–1447
 fortification, 1446
 microenriched fertilization/ferti-fortification
 of crops, 1447
 micronutrients content in seeds and
 bioavailability, 1447
 supplementation, 1446
 Management, rice paddies
 management approaches, 1951–1952
 Management intensive grazing (MIG), 464
 conservation and economics, 464–465
 Managing limited labor, 1166
 Managing soil fertility and soil water, 1166
 Mandragora, 169
 Manganese (Mn)
 in Brazilian soils, 1397–1398
 minerals, 1743
 in soils, 1396
 toxicity, 1362
 toxicity for sensitive plants, 1396–1397
 values, 1396–1398
 Mangrove swamp, 2568
 Mantle deposits, rocks
 chemical composition and particle size
 distribution, 1954–1955
 mineral components, 1953–1954
 transformation, 1955
 Manual or automated sampling and analysis,
 438–440
 Manure, 1406–1407
 gases produced from livestock manure
 ammonia, 1403
 methane, 1401–1402
 nitric oxide, 1403
 nitrous oxide, 1402
 non-CH₄ volatile organic compounds, 1403
 gas production, factors influencing, 1401
 GHG mitigation strategies, 1404
 livestock manure for global warming,
 1400–1401
 management strategies, 1403–1404
 mitigation strategies, evaluation of, 1404
 Manure, 1039
 Manure, compost, and organic sludge, 825
 Manure and diet modification
 nutrient management, 1705–1706, 1706–1707
 Manure application rates, 1706
 Manure management systems, 195
 Manure phosphorus concentration, 1705–1706
 Manuring, compost, and fertilizer
 application, 509
 MAP, 921
 Map class boundaries, 1316
 Map depicted Zn-deficiency, 1444
 Mapping constraints and challenges, 633
 Mapping soil carbon
 carbon auditing and, 1409
 peat soils, 1410
 pools of carbon, 1410
 purpose, 1409
 scorpan, 1409–1410
 soil carbon maps, 1410
 soil carbon saturation index, 1410
 soil–landscape and vegetation
 associations, 1409
 soil variables, 1410

- Marasmius androsaceus*, 142
- Marginal land, 1458, 1460
- Marine sediments, 296
- Mariotte bottle system, 120
- Mariotte principle, 121
- Markets, 1356
- Market trading, soil carbon farming, 2078
- Marsh, 2568
- Mass error, assigned molecular formula, 2155
- Mass flow, 1581
- and diffusion from soil to plant roots, 1589–1591
- Mass flux balance approach, 1910
- Mass measurement accuracy (MMA), 2155
- Mass movement, soil, 1348–1349
- Mass movement processes, 1341
- Mass planting, 827
- Mass spectrometry data analysis, SOM
- aromaticity index, 2157
 - challenges, prospects and future directions, 2157
 - compound identification algorithm (CIA), 2156
 - double-bond equivalent, 2157
 - Kendrick mass and defect (KMD) analysis, 2156
 - mass error, assigned molecular formula, 2155
 - mass measurement accuracy (MMA), 2155
 - negative mode mass spectra, 2155
 - results interpretation and visualization, 2156–2157
 - ultrahigh mass spectra, data processing, 2154–2156
- Mass wasting, 1820
- Mastigomycotina, 979
- Materials and soils
- ceramics, 1418–1419
 - in earthfill dams, 1417–1418
 - laterite, 1416–1417
 - making cement, 1417
 - soil/land, 1416
- Matrix, soil, 1388
- Matrix flow, 2218
- Maturity index (MI), free-living nematodes, 1527
- Maximum CWD (MCWD), 698
- Maximum density, 1656
- Maximum grain yield (MGY), 1422
- Maximum initial respiratory response (MIRR), 1430
- Maximum permitted concentration (MPC), 887
- Mayan city of Tikal, Guatemala, 155
- MDS. *See* Minimum data set (MDS)
- Mean annual precipitation (MAP), 300, 309
- Mean annual rainfall (MAR) isohyets, 660
- Mean annual temperature (MAT), 308
- Mean organic C content, 2137
- Mean residence time (MRT), 299, 1203, 2186
- Mean residence times (MRTs) of organic matter, 1749
- Mean weight diameter (MWD), 510, 760, 1508
- Measurements of loss and accumulation, 2334–2335
- Mechanical properties, pore system, 1785
- Mechanization, 762
- Mechanized systems, 599
- Medicago*, self-regenerating legumes, 1422
- Mediterranean ITK, 732–734
- Medium fine topsoil, 1683
- Megadroughts, 698
- Megafauna, 871
- Melanization, 2113–2114
- Melilite-mepheline basalt, 349
- Mesisol, 387
- Mesofauna, 138, 723, 871
- in soil food webs, 1626
- Mesopotamia, 1129
- Mesostigmata, 1488
- Metabolically inactive bacteria, 178
- Metabolites, 200
- Metal-depleted humus, 1749
- Metal–humus complexes, 136
- Metal-ion adsorption, 1427
- Metal-ion bridging, 1428
- Metalloids, inorganic pollutants, 1769–1770
- Metal organic frameworks (MOFs), 1802
- Metal phytoavailability, 1722
- Metal-reducing bacteria, 1878
- Metal refining and processing, 1178
- Metal-rich sulfuric acid (H₂SO₄) solutions, 6
- Metal toxicity, 1460, 1758
- Meteoric water falling, 2565
- Methane
- CH₄ conversion factor (MCF), 1401–1402
 - emissions, rice paddies cultivars, 1941
 - diel and seasonal patterns, 1940
 - inorganic fertilization, 1941
 - organic amendments, 1940–1941
 - photosynthetic activity, 1940
 - soils, 1941
 - water management, 1941
- enteric fermentation, 1401
- landfill gas, 1323, 1325
- manure storage, 1401
- methanogenesis, 1401
- methanogens and CH₄-oxidizing bacteria, 1402
- production from manure management, 1402
- quality, 1849
- Methane (CH₄), 45, 49, 85, 195, 658, 1036, 1040, 1048
- Methanogenesis, 128, 1401
- distribution from rice agriculture, 1947–1948
 - globally observed seasonal emissions, 1948–1949
 - mitigation, 1949
 - production and emission, 1947
- Methylcarbamates, 1758
- Micaceous minerals, 1792
- Micas, 1474
- Microaggregates of bacteria, 1630
- Microbes, 200
- Microbial activity, microbial communities, 1434
- Microbial approaches to remediation of contaminants, 201
- bioaugmentation, 202
 - biostimulation, 201–202
- Microbial biomass, 1306
- direct methods to measure, 1429
 - indirect methods to measure
 - adenosine triphosphate extraction method, 1431
 - chloroform fumigation incubation and extraction methods, 1429–1430
 - freeze-dried soil extraction method, 1430
 - microwave irradiation incubation and extraction methods, 1430
 - phospholipid fatty acids extraction method, 1431
 - rehydration method, 1430
 - substrate-induced respiration (SIR) Method, 1431
- S, rhizosphere and non-rhizosphere, 2247
 - UV spectroscopic method, 1431
- Microbial biomass and microbial quotient, 1938
- Microbial biomass carbon, 256
- Microbial biomass carbon (MBC), 2101
- Microbial communities
- actinomycetes, 1433
 - algae, 1433–1434
 - determination, 1434
 - factors affecting
 - management practices, 1434–1435
 - organic residues, amount, quality, and placement, 1434
 - soil moisture, aeration and temperature, 1434
 - soil reaction, 1434 - interactions
 - antagonistic interactions, 1436
 - competition interactions, 1436
 - mutualistic interactions, 1436 - role
 - biological N fixation, 1435
 - inorganic nutrient transformations, 1435
 - litter decomposition, 1435
 - nutrient availability, 1435
 - plant protection and diseases, 1436
 - soil aggregate stability and carbon sequestration, 1435
 - toxic and xenophobic compounds, 1435
 - soil composition, 1433–1434
 - terrestrial ecosystem, 1433
- Microbial ecology and biodegradation, 1774–1775
- Microbial-feeding nematodes, 1520
- Microbial interactions and nutrient cycling, 142
- dispersal of materials, 144
 - nutrient cycling and plant growth, 143–144
 - selective feeding by soil fauna, 142–143
- Microbial oxidation, 1435
- Microbial P and organic matter management, 1708–1709
- Microbial phospholipids, 177
- Microbiotic crusts, 1436
- Micro-Bowen ratio systems, 859–860
- Micro-catchment farming, 2515
- Microchemistry, 1440–1441
- Microcoleus vaginatus*, 218
- Microenriched fertilization/ferti-fortification of crops, 1447
- Microfauna, 138, 871
- in soil food webs, 1625–1626
- Micro-invertebrates, 652
- Microlysimeters, 859
- Micrometeorological methods, 440–441
- Micrometeorology, 2628
- changes in humidity, 2630
 - surface roughness, 2628
 - thermal stability, 2630

- Micrometeorology—
 turbulence, 2629
 velocity profile, 2628
 wind shear stress, 2628
- Micromorphology, 156
- Micromorphology and soil quality
 aggregates, 1439
 crusting, 1439
 fauna, 1439–1440
 microchemistry and submicroscopic studies, 1440–1441
 organic matter, 1440
 pores, 1439
 tillage, 1439
- Micronutrient deficiency, 1386, 1444–1445
- Micronutrient management, 1386
- Micronutrient nanofertilizers, 1512
- Micronutrients content in seeds and bioavailability, 1447
- Micronutrients-deficiency, 1444
- Micronutrients malnutrition in human, correction strategies
 agronomic management for better nutrition, 1447
 antinutrients management, 1447
 biofortification, 1447
 dietary modification, 1446–1447
 fortification, 1446
 microenriched fertilization/ferti-fortification of crops, 1447
 micronutrients content in seeds and bioavailability, 1447
 supplementation, 1446
- Microorganisms, environmental pollution detectors, 1780
- Micropedology, 1439
- Microphytic feeders, 138
- Micropores, filtering function, 1360
- Microporosity and accessibility, SSA, 2256–2257
- Microwatershed-based integrated farming system (IFS), 1383
- Microwave incubation, microbial communities, 1434
- Microwave (MW) irradiation, 1430
- Microwave remote sensing, 2480
- Middle Mississippian Indians, 1491
- Mid-infrared (mid-IR) spectroscopy, 270
- Millet (*Pennisetum glaucum*), 2627
- Millington–Quirk equation, 987
- Mined gypsum, 114
- Mined land reclamation, 1458–1459
- Mine drainage chemistry, 8
- Mineral and organic N fertilization, 1543
- Mineral and synthetic fertilizers, 1222
- Mineral fertilizer
 mycorrhizae and compost application, 1508
- Mineral fertilizers, 1222
 nutrients and products, 910
 N fertilizers, 911
 phosphate fertilizers, 911–912
 potash fertilizers, 912
 secondary and micronutrients, 912–913
 simple vs. complex, 910–911
- Mineralization, 560, 777, 2112
 to CO₂, 315
 of nutrients, 142
 sulfur, 2242
- Mineral nutrients, 1728
 for humans, 1730
- Mineral nutrition disorders
 Cu, 1446
 Fe, 1446
 I, 1446
 Se, 1446
 vitamin A deficiency (VAD), 1446
 Zn, 1445
- Mineralogy, 111
- Mineral P, active mobilization, 1709
- Mineral processing, 1178–1179
- Minerals
 amorphous. *See* Amorphous minerals
 dissolution reactions of, 1480–1482
 primary. *See* Primary minerals
 secondary. *See* Secondary minerals
- Mineral stability series, 1359
- Mineral substratum, 238
- Mineral weatherability, 1358
- Mineral weathering, 560
 mechanisms and rates, 433–434
 pathways, 431–433
 practical implications, 434
 resistance, 430–431
 thermodynamic formulations, 433
- Mine reclamation
 mines oil aggregation and C sequestration, 1467
 mines oil development and SOC sequestration, 1466
- Mine rehabilitation, 1306
- Minesoil
 chemical soil properties, 1460–1461
 coal mining spoils, 1449
 C quantification
 chemical methods, 1455
¹³C stable isotope approach, 1456
 inorganic C, 1450
 molecular biomarkers, 1455
 optical and microscopic technique, 1451–1454
 radiocarbon analysis technique, 1450–1451
 SOC separation and quantification, 1450
 spectroscopic techniques, 1455–1456
 thermal oxidation techniques, 1454–1455
 degraded land extent, 1458
 development, 1449
 general properties, 1459
 grading of, 1459
 mined land reclamation, 1458–1459
 miscanthus production in, 1460
 physical characteristics, 1459
 physical soil properties, 1460
 soil organic carbon (SOC), 1449
 surface mining operations, 1449
- Mine wastes, 9
- Minimum data set (MDS), 1853
- Mining
 anthropogenic disturbances, 1821
- Minirhizotron
 definition, 1484
 image acquisition device, 1485
 image analysis, 1486–1487
 root, continuous non-destructive assessment, 1484
 tube characteristics and installation, 1484–1485
- Miocene-age soil, 1755
- Miocene-age soil weathering, 1755
- Mires, peatlands, 1668
- MIRR. *See* Maximum initial respiratory response (MIRR)
- Miscanthus, 1460
- Miscanthus giganteus*, 1458
- Miscellaneous feeders, 138
- Mississippi and Ohio River bottomlands, 69
- Mississippi River, 65
- Misunderstandings, property rights, 1356
- Mites
 distribution and abundance, 1488–1489
 function, 1489
 in soil food webs, 1626–1627
- Mitigating desertification and its consequences, 639–640
- Mitigating desertization, 662
- Mitigation measures, 1350–1351
- Mixed forests, 964
- Mixed quantum mechanics/molecular mechanics (QM/MM) methods, 2167–2168
- Mixture modeling, 2197–2198
- MMA. *See* Mass measurement accuracy (MMA)
- Mn oxides
 minerals, 2006
 properties, 2006
- Modeling, SOM
 application, 2162–2163
 decomposition as continuum, Cohort model, 2159
 evaluation, 2162
 factors affecting turnover in, 2162
 food-web models, 2159
 process-based, multicompartment models, 2159
 representation in GCTE-SOMNET, 2160–2161
- Modeling soil erosion, 2499
 dynamic stochastic and deterministic models, 2499–2500
 processes, 2499
 steady-rate models of water erosion, 2500
 EUROSEM, 2501–2502
 GUEST, 2500–2501
 WEPP, 2500
- Modeling soil formation, 1680
 pedometrics, 1680
- Moderate resilience, 1919
- Moderate Resolution Imaging Spectroradiometer (MODIS), 582
- Modern civilization and soils, 1491–1492
- Modified universal soil loss equation (MUSLE), 2490–2491
- Modified Wilson and Cooke (MWAC), 2619
- Modular peat biofilter system, 1110
- Modulus of rupture (MOR), 1073, 2046
- MOFs. *See* Metal organic frameworks (MOFs)
- Moisture, 317
 content, 513
 content on aggregate strength, 52
 remote sensing, precision agriculture, 1817
 retention, 2443–2444
 soil moisture characteristic (SMC), 1783–1784
 soil moisture content (SMC), 698

- soil moisture regimes (SMRs), 111
- soil moisture sensing, 2483
 - quantitative soil moisture estimation, 2483–2484
- soil respiration, 1929
- Mojave Desert of California, 1204
- Moldenhauer, 815
- Molecular areas, SSA, 2257
- Molecular plant breeding, 527
- Molecular recalcitrance, persistent organic pollution
 - hydrolytic reactions, 1772
 - oxygenase enzymes, electrophilic attack by, 1771
 - steric hindrance, 1772
- Molecular simulations, SOM
 - challenges and outlook, 2169
 - classical molecular mechanics methods, 2166–2167
 - mixed quantum mechanics/molecular mechanics (QM/MM) methods, 2167–2168
 - quantum mechanical methods, 2167
 - survey on applications of, 2168–2169
- Mollisols, 111, 159, 410, 1162, 2299
- Mollusca, 206
- Molten metal, pores cast, 1388
- Molybdenum–iron (MoFe) protein, 221
- Molybdenum (Mo)
 - correcting deficiency, 254
 - deficiency, 233
 - parent material and soil pH, 252–253
 - physiological role, deficiency, and toxicity symptoms, 253
 - soil properties, 253
- Monin–Obukhov similarity theory, 864
- Monitored natural attenuation, 202
- Monitoring, verification, and accounting (MVA), 995–996
- Monks Mound, 1491
- Monoclonal antibodies, 243
- Monocotyledonous plants, 72
- Monoculture practices and enzyme activity, 2102
- Monomethylarsonous acid, 161
- Monomethylstibine (CH₃SbH₂), 148
- Monotropoid mycorrhizas, 1502
- Montevideo Programme, 1353
- Montgomery, 768
- Montmorillonite, well-crystallized, 1638
- MOR. *See* Modulus of rupture (MOR)
- Moroccan vernacular soil names, 370–374
- Morphological and anatomical adaptation, 527
- Morphology
 - landscape, 1328–1329
- Mortality composting, 1407
- Mortierella isabellina*, 142, 726
- Mortmass, 236
- Mossi of Burkina Faso, 402
- Most probable number (MPN), 1429
- Mounding, 1038
- Mountainous regions, 101
- Mountainous terrain, 1342
- MPN. *See* Most probable number (MPN)
- Mudflow, 1350–1351
- Mulches, 826, 896, 2042
- Mulch farming
 - arable land, mulch procurement for
 - crop residues, 1495–1496
 - planted fallows and cover crops, 1496
 - benefits of, 1496–1497
 - biomass burning, 1501
 - greenhouse effect, 1499–1500
 - limitation of, 1500–1501
 - mulching and agricultural sustainability, 1500
 - mulch types, 1495
 - nutrient cycling and soil fertility enhancement, 1497–1498
 - soil biodiversity, 1499
 - soil-surface management and erosion control, 1497
 - soil temperature management, 1498–1499
 - water management, 1497–1498
- Mulching, 1085, 1602
- Mulch-till, 764
- MULSE. *See* Modified universal soil loss equation (MUSLE)
- Multicollinearity, DRS, 1887
- Multi-GHG, 79
- Multinutrient fertilizers, 912
- Multiple erosion–deposition cycles, 1636
- Multiple use conflicts, 2552–2553
- Multistripe laser triangulation (MLT) scanning, 1388–1389
- Multivariate calibration methods, DRS, 1887
- Municipal solid waste (MSW), 923
 - anaerobic degradation, 1322–1323
- Municipal wastewater, 1762
- Munsell's Color Charts, 1954
- Munsell Soil Color Charts, 452, 453
- Murray–Darling Basin (MDB), 32, 1113, 2586
- Murray River, 32
- Muscovite, 1474
 - and chlorites, 1747
- Mutualistic interactions, 1436
- Mutualists, 979
- MWD. *See* Mean weight diameter (MWD)
- ΔMWD, 63
- Mycelium of fungus, 1507
- Mycena galopus*, 142
- Mycobacterium*, 41
- Mycobacterium tuberculosis*, 41, 176
- Mycorrhizae, 1436, 1703
 - hyphae, glomalin, and soil quality, 1507–1508
 - inoculations on soil quality under long-term field conditions, 1508–1509
 - PyC influence, 1835–1836
 - role in soil, 1506
 - and roots effect on soil quality, 1506–1507
 - symbiosis, 554
- Mycorrhiza inoculations effect
 - on soil quality, 1508–1509
- Mycorrhizal fungi, 979, 1709–1710
- Mycorrhizal fungi, 979, 1709–1710
 - rehabilitation, indicators and monitoring, 1900
- Mycorrhizal fungi, 553
- Mycorrhiza–microbial interactions, 1503
- Mycorrhiza of forest ecosystems
 - C costs, 1503
 - nutrient uptake, 1502–1503
 - role, 1502
 - seedling infection, 1503–1504
 - soil ecosystems, influence on, 1503
 - types, 1502
- Mycorrhizas, 1625
- Mycorrhization, 1503
 - helper bacteria, 1503
- Myrmica scabrinodis*, 152
- Myths, 1356
- N**
- Na fertilizers, 2019
- NaI(Tl) detectors, 1873
- Nano-ball allophane, 98
- Nano-DESI. *See* Nanospray desorption electrospray ionization (nano-DESI)
- Nanofertilizers
 - macronutrient, 1511–1512
 - m micronutrient, 1512
 - nanomaterial-enhanced fertilizers, 1512–1513
 - nanomaterials and, 1511
 - nanoparticulate plant growth enhancers with unclear mechanisms, 1513–1514
- Nanomaterial-enhanced fertilizers, 1512–1513
- Nanomaterials and nanofertilizers, 1511
- Nanoparticulate fertilizer carriers, 1512–1513
- Nanoparticulate plant growth enhancers with unclear mechanisms, 1513–1514
- Nanoscale Molecular Dynamics, 2167
- Nanospray desorption electrospray ionization (nano-DESI), 2076–2077
- Napier grass (*Pennisetum purpureum*), 828
- Nardus stricta*, 143
- National Acid Precipitation Assessment Program (NAPAP), 11
- National Alcohol Program, 2231
- National Bureau of Environmental Protection (NBEP), 1619
- National Centre for Soil Research (CNPS), 380
- National Commission of Soils (CNS), 379
- National Cooperative Soil Survey (NCSS), 1007
- National Oceanic and Atmospheric Administration (NOAA), 582
- National soil classification systems, 376
- National soil conservation strategies, 2084
- National soil law, 1353
- National Soil Survey and Conservation Service (SNLCS), 379
- Native vegetation, 443
- NaTPB. *See* Sodium tetraphenyl boron (NaTPB)
- Natric horizon, 92
- Natural acid rock drainage, 6
- Natural allophanes, 98
- Natural amelioration, 661
- Natural-contained alkalinity, 34
- Natural disasters, 1492
- Natural disturbances
 - erosion, 1820
 - fire, 1819–1820
 - insects and disease, 1820
 - windthrow, 1820
- Natural ecosystems, 234
- Natural farming, 703
- Natural religions, 1903
- Natural resource management (NRM), 342
- Natural resources
 - soils
 - and civilization, 1517

- Natural resources—
 formation and degradation of, 1517
 properties of, 1516–1517
- Natural Resources Conservation Service (NRCS), 69, 459, 753, 1547–1549, 2086–2089
- Natural resources of peatlands
 biodiversity, 1672
 biotic resources, 1671–1672
 carbon resources, 1672
 soil resources, 1670–1671
 water resources, 1670
- Natural vegetative strips (NVSs), 343
- N-based fertilizers, 311
- N-containing herbicide, 1773–1774
- N-deficient soil, 1324
- ¹⁵N dilution method, 1552
- NDVI. *See* Normalized difference vegetation index (NDVI)
- Necrosis, 1393
- Neelipleona*, 448
- Neem (*Azadirachta indica*), 734
- Negative adsorption (co-ion exclusion), SSA, 2257
- Negative mode mass spectra, 2155
- Nematoda, 206
- Nematodes
 beneficial
 entomopathogenic nematodes, 1521
 microbial-feeding nematodes, 1520
 predacious and omnivorous nematodes, 1520–1521
 biogeography
 distribution and abundance, 1520
 species diversity, 1520
 collection and extraction methods, 1521
 harmful, 1520
 history, 1518
 life history classification, 1518
 morphology, 1519
 and soil
 nematode assemblages and response to disturbance, 1526–1527
 nutrient cycling, 1525–1526
 pH, 1525
 soil biota, impact of, 1526
 temperature, 1525
 texture, structure and moisture regime, 1524–1525
 soil-dwelling nematode orders, 1519
 in soil food webs, 1626
 taxonomic classification, 1518
 trophic classification, 1518
- Nematodes (*phylum Nematoda*), 81
- Neolithic Revolution, 1112
- Neossolos, 381, 382
- Nepalese alder (*Alnus nepalensis*), 1283
- Netherlands classification systems, 412
 Dutch system, 413–414
 history, 412–413
 nomenclature, 414
- Net primary productivity (NPP), 582, 1929
- Net radiation, 1086
- Neutral buoyancy, 2630
- Neutralization, 38
- Neutralizing value (NV), 1363
- Neutral soil, 1696
- Neutral to alkaline mine drainage (NAMD), 6
- Neutron activation analysis (NAA), 319
- Neutron thermalization, 2526–2527
- New Zealand soil classification, 376–377
- Nexus approach
 biomass overexploitation, 1530–1532
 biomass production cycle, 1532
 interlinkages management, 1532–1533
 organic wastes integration, 1532
 sub-Saharan Africa (SSA), 1530
- N fertilization, 1423
- N fertilizers, 911, 920
- N-fixing communities, 1436
- N-fixing green manures, 1620
- N-heterocyclic derivatives, 1774
- NH₃-oxidizing bacteria (AOB), 1402
- Nickel contaminations, 1760
- Nile Valley, 1129
- Nitisols, 409
- Nitosols, 392
- Nitossolos, 380, 381, 382
- Nitratea, 1406
- Nitrate Directive, 1825
- Nitrate esters, 199
- Nitrate leaching, 1422
- Nitrate leaching index (NLI)
 framework, 1536
 hydrologic group, 1536
 nutrient management planning, 1536
 percolation index (PI), 1536
 percolation potential, 1535
 processes, 1535
 seasonal index (SI), 1536
 technologically based index, 1536
- Nitrate leaching management
 contribution, 1538
 drainage-ditch water control measures, 1539
 irrigation and cropping system techniques, 1539
 nitrification inhibitors, 1539
 primary techniques for, 1538–1539
 tillage practices, 1539
 within-season monitoring techniques, 1540
- Nitrate (NO₃), 484
- Nitric oxide (NO), 1403
 and NO_x
 quality, 1849–1850
- Nitrification, 714, 1036, 1543–1544, 1555, 1556, 1563
 and denitrification, 1969–1970
 inhibitors, 1040, 1423, 1539, 1564
 rates, 928
- Nitritotriacetic acid (NTA), 1879
- Nitriteoxidizing bacteria (NOB), 1402
- Nitrogen (N), 1222–1223, 1228, 1516
 agronomic efficiency, 1422
 availability, 132
 concentrations, 311
 cycling, 175
 deposition. *See* Acid rain
 fertilization, 211
 fertilizer, 118, 1492–1493
 fixation, 41
 gases quality
 ammonia, 1850
 nitric oxide (NO) and NO_x, 1849–1850
 heterocycles, 199
 and its transformations
 microorganisms, 1541
 N cycle, 1541–1542
 N input processes, 1542–1543
 nitrification, 1541–1542
Nitrobacter, 1541–1542
Nitrosomonas, 1541
 N loss processes, 1543–1544
 N uptake by plants, 1543
 organic matter, 1541
 soil water and aeration, 1541–1542
 temperature effect, 1542
 stimulating effect of, 73
- Nitrogen-15
 biological N₂ fixation, 1552–1553
 fertilizer rate, 1551–1552
 isotope, 1551
 rhizodeposition estimation, 1553
- Nitrogen, soil, 1938
- Nitrogenase, 221, 233
 control of, 222
 genetics of, 222
 purification of, 221
 tertiary structure, 222
- Nitrogen cycle for grazing land, 1035
- Nitrogen derived from the fertilizer (Ndff), 1552
- Nitrogen index
 environmental challenges and N management, 1546
 for fertilizer recommendations, 1549
 global food security challenge, 1545–1546
 Kentucky N and phosphorous (P) index, 1547–1548
- Natural Resources Conservation Service (NRCS), 1547–1549
 590 nutrient management plans, 1549
 previous work, 1546–1547
 recommendation screen of, 1548
 for risk assessment, 1547–1549
 version 4.5, 1547
- Nitrogen (N)-fixing
 bacteria, 206
 bacteria populations, 2133
 organisms, 223, 224, 232, 233
 in soil, 237
- Nitrogen (N) immobilization, 1055
- Nitrogen oxides (NO_x), 11
- Nitrogen physiological efficiency (NPE), 1422
- Nitrogen-rich feedstocks, 195
- Nitrogen use efficiency (NUE), 1422
 crop management, 1423
 mediterranean regions, 1421–1422
 N fertilization, 1423
 N monitoring in soils and crops, 1423
 rainfed mediterranean agriculture, 1421
 rotations, 1423–1424
- Nitrosomonas europaea*, 1970
- Nitrosomonas* spp. bacteria., 1040
- Nitrospirae, 179
- Nitrous oxide (N₂O), 45, 49, 85, 195, 1048, 1402
 emissions, 78
- Nitrous oxide emissions
 agricultural fields, 1563
 agricultural soils, 1969
 biomass burning, 1556–1557
 from denitrification, 1555–1556

- fertilizer consumption and production, 1557
- flooded soils, 1556
- management practices to decrease, 1557
- from nitrification, 1556
- ammonia-oxidizing bacteria (AOB), 1970
- atmospheric distribution, 1562
- nitrification and denitrification, 1969–1970
- paddy soils, 1559–1561
- production mechanism
 - chemical decomposition of hydroxylamine (NH_2OH), 1564
 - chemodenitrification, 1563
 - controlled-release fertilizer, 1564
 - denitrification, 1563
 - measurement of, 1564
 - mitigation strategy, 1564
 - nitrification, 1563
 - nitrification inhibitors, 1564
- salinity, 1970
- salt-affected soils, 1969
- sinks for, 1562
- sources of, 1562–1563
- NLI. *See* Nitrate leaching index (NLI)
- N-limited soil, 143
- N mineralization, 143
- ^{15}N natural abundance method, 1552–1553
- Nocardia*, 41
- Nocturnal soil freezing, 973
- Non-allophanic Andisols, 1650
- Non-allophanic andosolization process, 1650
- Non-aqueous phase liquid (NAPL), 1772–1773
- Non-bonding Lennard–Jones-type potential, 2166
- Non-boreal podzol, 1748
- Noncarbonate soil matrix, 340
- Non- CH_4 volatile organic compounds (NMVOCs), 1403
- Non-conserving cropping, 794
- Non-crop plants (weeds), 2473
- Non-crystalline aluminosilicates, 1471
- Non-electric contact sensors, 2302
- Non-electric thermometers, 2303
- Nonequilibrium sorption, 2196
- Non-erodible layer, 743
- Non-fixing communities, 1436
- Non-imaging systems, 1017
- Non-invasive geophysical methods, subsurface macropores, 1389
- Nonirrigated agriculture, nutrient management, 1571
- Nonisotopic tracer methods, 441
- Non-labile P, 1701
- Non-linearity, DRS, 1887
- Non-photosynthetically active vegetation (NPV), 2197
- Non-plinthic Oxisols, 346
- Non-point pollutant, 78
- Nonpoint-source pollution, 210
- Non-point source pollution (NPSP)
 - contaminant interactions, 1765
 - global challenges and responsibility, 1767
 - management and or remediation, 1766–1767
 - preventing water pollution, 1767
 - sampling requirements, 1766
 - soil and environmental quality, implications, 1765–1766
- Non-polar organic pollutants, 1632
- Non-renewable energy, 2
- Non-renewable fuels, 3
- Non-resilient, 1919
- Non-symbiotic N fixers, 1223
- Non-tropical peatlands, 1669
- Non-vegetative erosion protection, 2613
- Normalized backscatter cross-section, 2481
- Normalized difference vegetation index (NDVI), 581, 582, 1316, 1901
- Northeastern hill ranges and valleys, 1158
- North Eastern Region (NER), 1382–1383
- Northern Circumpolar Soils (NCPS), 1753, 1756
- Northern Circumpolar Soil Carbon Database, 1753
- No-till
 - advantages and disadvantages, 1566–1567
 - in arboreal stratum, 1378
 - biomass production, 1378
 - C footprint, 1379
 - dry matter and nutrient accumulation production, 1378
 - green manure, 1379
 - plant cocktail, 1378–1379
 - results achieved by using, 1567–1568
 - root cycling, 1378
 - salinization process, 1378
 - spontaneous vegetation, 1378
- No-tillage, 2609–2610
- No-till (NT), 1384
- No-till planting, 462
- No-till/strip-till, 764
- Novel liquid sorbents, 1802
- Nozzle-type rainfall simulator, 815
- NPE. *See* Nitrogen physiological efficiency (NPE)
- N pollution of surface waters
 - biochemical processes, 2268–2269
 - fertilizers, 2268
 - hydrologic processes, 2269–2270
 - needs and approaches, 2272
 - problems caused by, 2268
 - sources, 2268
 - spatial variability, 2270
 - watershed scale analyses
 - efficiency of fertilizer, 2271
 - hydrologic process models, 2271
 - regional input–output analyses, 2270–2271
- NPP. *See* Net primary productivity (NPP)
- NPV. *See* Non-photosynthetically active vegetation (NPV)
- N rhizodeposition, 1553
- Nuclear electronics, 320
- Nuclear magnetic resonance (NMR) spectra, 105
- Nuclear magnetic resonance (NMR) spectroscopy, 97
- Nugget variance C_0 , 1013
- Nullah* plugs in India, 788
- Nursery-raised seedlings, 1306
- Nutrients, 2468–2469
 - boundary conditions and nutrient entry, 1591–1592
 - deficiency and toxicity symptoms
 - diagnostic symptoms, 1586–1587
 - functions of elements, 1585
 - mobility of the elements, 1585–1586
- diffusion, 1588–1589
- interactions in soil–plant system, 1594
 - chloride or sulfate, 1594
 - citrate, 1595
 - oxidation–reduction status, 1595
 - phosphorus deficiency, 1595
 - in plants, 1595–1596
 - precipitation effects, 1594
 - in rhizosphere, 1594–1595
 - silicon and Al, 1595–1596
- mass flow and diffusion from soil to plant roots, 1589–1591
- root hairs, 1591
- soil plant system, 1588
- water interactions, 1597–1599
- water use efficiency (WUE) and, 1597–1598
- Nutrient- and water-holding capacities, 1665–1666
- Nutrient availability, 1435, 1471
- Nutrient cycling, 966, 1460
 - influence of abiotic factors, 967–968
 - influence of biotic factors, 966–967
 - influence of climate, 966
 - precision agriculture
 - geostatistical tools, 1812
 - landscape crop water and nitrogen use, 1812–1814
 - limiting factor, 1811
 - root restricting layer, 1811
 - site-specific management, 1811
 - soil textural differences, 1811
 - role of disturbance, 968
 - and soil fertility enhancement, 1497–1498
- Nutrient deficiency, 890
- Nutrient-depleted soil, 114, 213
- Nutrient depletion, 880
- Nutrient dynamics, SOM
 - composition, 2172–2173
 - C sequestration, land use for, 2174
 - microbial biomass, 2173
 - nitrogen, 2173
 - phosphorus and sulfur, 2173
 - plant decomposition, microbial biomass, 2173
 - soil quality enhancement, 2174
- Nutrient elements, soil, 1306
- Nutrient fertilizers, 1773
- Nutrient fixation capacity, 903
- Nutrient-loaded zeolite nanofertilizers, 1512
- Nutrient losses, 1224–1225, 2641–2642
- Nutrient management, 288–289
 - advances in, 1571–1573
 - in agriculture, 289
 - agroforestry, 291–292
 - arable cropping system, 289–290
 - climatic change, 1570–1571
 - conservation approach and, 1573–1574
 - definition, 1574
 - and environmental challenges, 1571
 - fertility and
 - applications timing, 1584
 - base saturation (BS), 1578
 - buffer capacity, 1580
 - buffering nutrients in soil solution, 1577
 - calcareous soils, 1579

- Nutrient management—
 cation and anion exchanges, 1578–1579
 essential plant nutrients, 1576–1577
 lyotropic series, 1578
 mineral solubility in soil, 1579–1580
 nutrient mobility in soil, 1581–1582
 nutrient placement methods, 1583–1584
 nutrient source identification, 1583
 organic materials, 1579–1580
 plant nutrient requirement assessment, 1582
 quantify optimum nutrient rate, 1582
 salts in alkaline soils, 1579
 site-specific nutrient management, 1584
 soil–plant–atmosphere system, 1577
 supplying capacity, 1582
 transport from soil to roots, 1580–1581
 global challenges, 1570–1571
 global C potential, 288–289
 in grassland and fodder crops, 290–291
 horticultural crops, 291
 irrigated agriculture, 1571
 manure and diet modification, 1705–1706, 1706–1707
 nonirrigated agriculture, 1571
 in water resources, 1571
 590 nutrient management plans, 1549
 Nutrient mining, 880
 restoration, resilience, 1926
 Nutrient mobility in soil, 1581–1582
 Nutrient muscular dystrophy, 1446
 Nutrient placement methods, 1583–1584
 Nutrient-poor soils, 346
 Nutrient recycling, 895
 Nutrient requirement, 1735–1736, 1738
 Nutrient restoration, 2642
 Nutrient source identification, 1583
 Nutrient sufficiency, 1735–1738
 Nutrient toxicity, 891
 Nutrient transformation, 1435
 Nutrient uptake, 1502–1503
- O**
 O–Ah–E–Bh–Bs horizons, 1747
 Oak (*Quercus* spp.), 185
 Oats (*Avena sativa* L.), 501, 530, 936
 Odai (*Acacia planifrons*), 734
 Odometer, 474
 ODR measurement, 45
 OFDC. *See* Organic Food Development Center (OFDC)
 Off-site effects of forest management, 957–958
 Oil and gas reservoirs, 995
 Oil palm (*Elaeis guineensis* Jacq.), 224, 362
 Oil radish (*Raphanus sativus* L.), 73
 Oilseed rape (*Brassica napus*), 494
 Oligochaeta, 206, 701
 Oligotrophic state, 855
 Olivines, 1473
 OM. *See* Organic matter
 OM mineralization, 1036
 On-farm and *in situ* water management, 1330
 On-farm production costs, 939
 Onion (*Allium cepa* L.), 364
Onychiurus armatus, 142
 OP11, TM7, 179
 Open cut mines
 acid mine drainage, 1464
 coversoil resources, 1462–1463
 coversoil thickness requirements, 1463
 landscape regrading, 1462
in situ soil reclamation, 1463–1464
 steep slope reclamation, 1464
 Open-type carbon dioxide (CO₂), 491
 Operational taxonomic units (OTUs), 178
 Optical reflectance (UV/VIS/NIR/MIR), 1006–1007
 Ordination, pedometrics, 1678
 Organic agriculture
 crop and pest performance, 1601–1602
 history, 1600–1601
 worldwide statistics, 1601
 Organic Agriculture Movements, 1600
 Organic amendment and mycorrhizae, 1507
 Organic carbon (OC), 1107, 1362
 compounds, microbial communities, 1434
 in grassland soils, 1031
 pools, 1756
 Organic certification agencies, 1600
 Organic chemicals, 1779
 Organic contaminants, 200
 Organic degradation, 175
 Organic farmers, 1600–1601
 Organic farming (OF), 1384, 1386
 challenges and future prospects, 1620
 definition, 1618
 history, 1618–1619
 status, 1619–1620
 Organic fertilization, 1621–1623
 Organic fertilizers, 916–917, 1423
 animal residues, 917
 biosolids, 918
 by-products and wastes, 918
 effluents, 917–918
 heavy metal concentrations in, 1622
 municipal wastes, 917
 negative anthropogenic activities, 1622
 sources of, 1621–1622
 Organic Food Development Center (OFDC), 1619
 Organic interlayer complexes, 1630
 Organic manure, 1543
 Organic materials, 1579–1580
 Organic matter (OM), 115, 1107, 1440
 associations, 1629
 decomposition, 51
 decomposition rates, 1632
 inputs
 placement, 2134–2135
 quantity, 2133–2134
 species composition, 2134
 and moisture levels, 1806
 protection, 1632
 soil pH, 1697–1698
 Organic mulches, 1495
 Organic pesticides, 1768
 Organic pollutants, 1720
 bioremediation, 1768–1769
 chemicals, 1768
 potential impacts of, 1768
 Organic residuals, 1408
 Organic soil, 385
 Organic wastes integration, 1532
 Organisms, 51
 activity, 140
 Organochlorine, 704
 Organometallic complexes, precipitation, 1359
 Organo-mineral complexes, 1050
 Organo-mineral relationships
 characterization, 1629–1631
 microaggregates of bacteria, 1629–1630
 organic matter, biodegradation rate of, 1632
 organo-mineral associations
 binding forces within, 1630–1631
 microstructure, 1629–1630
 nature and scale, 1630
 soil minerals properties, 1631–1632
 Organo-mineral soil, 2561
 Organophosphates, 199
 Organophosphoric insecticides, 704, 1758
 Organossolos, 381, 382
 Oribatid mites, 1489, 1627
 Oribatid soil mites, 1489
 Original oil in place (OOIP), 995
 Orthels, 1753
 Orthents, 729
Orthogonius latus, 142
 Orthoxic Tropudult, 820
 Ortstein, 1359
 Osmotic adjustment (OA), 694
 Other nutrients, 1229–1230
 Outer-sphere complexes, 427–428
 Outer-sphere surface complexes, 1427–1428
 Oven-dry bulk densities, 1656
 Owendry soil, particle density, 1652
 Overcultivation, 661, 662
 Overfertilization, 886
 Overgrazing, 464, 661
 Overirrigation, 2551
 Overstocking, 662
 Ovoviviparity, 1523
 Owners, property right, 1355
 Ownership, 1355–1356
 Oxbow, 68
 Oxidation prevention, 34
 Oxidation–reduction, 6, 348
 Oxidation–reduction reactions, 1729
 Oxides, primary minerals
 Fe, 1475
 Ti, 1475
 Oxidic ratio, 1476
 Oxidizable organic carbon, 26
 Oxidoreductases, 737
 Oxisol-dominated landscapes, 1637
 Oxisols (*Latossolos*), 111, 159, 410, 1362, 1397, 2292–2293, 2299, 2369–2370
 formation processes, 1636
 geologic settings, 1635
 human use, 1636–1637
 hydraulic conductivity, 1635
 location, 1635, 1636
 soil moisture regime (SMR), 1635
 suborders, 1635
 Oxyanions, 1427–1428, 1766–1767
 Oxy-combustion, 1801
 Oxygenase enzymes, electrophillic attack
 by, 1771
 Oxygen diffusion rate (ODR), 45, 47–48, 562, 684
 concept and principle, 1725
 definition, 1725
 and plant response

- root response, 1726
 - seedling emergence, 1725–1726
 - shoot response, 1726–1727
 - soil parameters and, 1725
- Oxygen (O₂), 44
- Oxygen paradox, 232
- Ozone, 13
- P**
- Packing, types, 1656
- Packing density, 1655
- Paddy soils, nitrous oxide emissions
 - fertilizer application, 1560
 - mitigation and challenge, 1560–1561
 - tillage and rice residue retention, 1560
 - water regime, 1559–1560
- Paintings of the Renaissance, 168
- Pakhuri (*Ficus glaberrima*), 734
- Paleohydropedology, 1143
- Paleolithic period, 1112
- Paleo-pollen analysis, 1373
- Paleosols, 155, 1328, 2293
- Palmer Drought Severity Index, 698
- Palmyra (*Borassus flabellifer*), 734
- Palouse wheat growing area, 774
- Paltothyreus tarsatus*, 152
- Paludization, 2113
- The Pampas
 - climate, 1638
 - flooded, 1638
 - management options for sustainable land use, 1641
 - rainfall amounts, 1638
 - research and development priorities, 1641–1642
 - soil associations within subregions, 1640
 - soil related constraints to crop production, 1639–1641
 - soils of, 1638–1639
- Panicum miliaceum*, 1729
- The Pantanal
 - area occupied and percentages by main classes, 1645
 - geological and climatic aspects, 1643
 - soils of, 1644–1647
 - subdivision and hydrological aspects, 1643–1644
 - subregions, 1645
- Paramos, 101
 - climate, 1648
 - in different countries, 1649
 - ecological reserve, 1650
 - ecosystem, world distribution, 1649
 - geology–geomorphology, 1649–1650
 - location, 1648
 - soils
 - and C content, 1650
 - degradation, 1651
 - organic matter production, 1650
 - and water cycle regulation, 1650–1651
 - vegetation, 1648
- Parasites, 138, 206
- Parasiticide formulations, 1602
- Parasitism, 1124
- Parent material, 998–999
- Parent materials, 1396–1398
- Park savannah (*campo cerrado*), 345
- Partial wetting, 1138
- Particle density, 1783
 - active soil measured in water, 1653
 - non-polar inorganic liquid, 1653
 - in non-polar liquid, 1653
 - ovendry soil, 1652
 - particle settling velocity, 1652
 - pycnometer, 1652
 - sedimentation parameter calculation, 1652
 - submersion method, 1652–1653
 - suspension density, 1653
- Particle packing
 - indices of, 1654
 - models of, 1655–1656
 - packing density, 1655
 - particle shape effects, 1656–1657
 - types of, 1654–1655
- Particle settling velocity, 1652
- Particle shape
 - measurement methods, 1659–1660
 - pedogenesis, 1660
- Particle shape effects, 1656–1657
- Particle size
 - designation, 1662
 - distribution, 1661–1662
 - distribution, calcitic liming materials, 1364
 - particle size analysis (PSA), 1662
 - soil properties affecting field texture, 1662–1663
 - soil texture relevance
 - ease of cultivation, 1665
 - hardsetting characteristic, 1666
 - nutrient- and water-holding capacities, 1665–1666
 - soil consistency, 1664–1665
 - specific surface area (SSA), 1663–1664
 - transmission properties, 1665–1666
- Particle size distribution, 1756
- Particulate organic matter (POM), 315
- Partition and regulate water flow, 1866–1868
- Passioura water model, 694–695
- Passive microwave systems and data, 2481–2483
- Pasteur, Louis, 176
- Pasture phases to rebuild organic carbon, 514
- Patchiness of vegetation in drylands, 651
- Pathogens, 206, 1124
- Pathogens, 979
- PAWC. *See* Plant available water capacity (PAWC)
- Pb contaminations, 1760–1761
- PCBs. *See* Polychlorinated biphenyls (PCBs)
- PCs. *See* Pedogenic carbonates (PCs)
- Pea plants (*Pisum sativum*), 220, 227
- Pearlmillet (*Pennisetum glaucum*), 300
- Pearl millet (*Pennisetum typhoides* S. & H.), 502
- Peat, 1056
- Peatlands, 295, 2568
 - evapotranspiration, 1669
 - mires, 1668
 - natural resources of
 - biodiversity, 1672
 - biotic resources, 1671–1672
 - carbon resources, 1672
 - soil resources, 1670–1671
 - water resources, 1670
 - non-tropical, 1669
- restoration, 1672–1673
- sensitivity to disturbance, 1669
- Sphagnum lenense*, 1669
- swamps and marshes, 1668
- wetlands, 1668
- Peat soils, 413
 - mapping soil carbon, 1410
- Pedobioclimatic belts, 437
- Pedogenesis, 729
- Pedogenetic processes, 408, 436
- Pedogenic carbonates (PCs), 335–336, 714, 1204
 - formation, 336–337
 - morphology of, 336
 - stable isotopes, calculation using, 336
- Pedogenic processes, 561
- Pedogenic silica
 - dissolution, 2012
 - morphology, 2013
 - movement and precipitation, 2012
 - sources, 2012
- Pedo-geomorphic process, 1344
- Pedological concepts and water flow, structure
 - blocky structure, 2217
 - characterization, 2216
 - compound structure, 2217
 - flat spaces, 2217–2218
 - formation, 2216–2217
 - grade and size, classes, 2217
 - moderate coarse prismatic structure, 2217
 - permeability, 2218
 - platy structure, 2217
 - prismatic structure, 2217
 - shapes, sketches, 2217
 - shrinkage, 2217
 - significance, 2217–2218
 - soil management for water flow, 2218–2220
 - stability, 2218
 - tubular voids, 2218
 - water movement and, 2218
- Pedological modeling, 1674–1676
- Pedological translocations/transformations of
 - clay coatings, 93–94
- Pedologists, 154
- Pedology, 391, 1104
- Pedometrics
 - contaminated sites, 1681
 - digital soil mapping, 1680
 - fuzzy sets, 1678
 - geostatistics, 1678–1679
 - history, 1677–1678
 - land evaluation, 1680–1681
 - modeling soil formation, 1680
 - numerical and continuous classification, 1678
 - ordination, 1678
 - precision agriculture, 1680
 - soil inference systems, 1679–1680
 - soil sampling, 1679
 - in soil science subdisciplines, 1680–1681
- Pedon, 383
- Pedosphere, 982–983
- Pedotransfer functions (PTF), 1684–1685, 2537
 - accuracy of, 1684–1685
 - definition, 1682
 - hydraulic characteristics, 1682
 - landscape position, 1684
 - mechanical properties and shrink–swell parameters, 1684

- Pedotransfer functions (PTF)—
 methods to develop
 artificial neural networks (ANNs), 1684
 group method of data handling (GMDH), 1684
 regression analysis, 1684
 regression tree algorithm, 1684
 organic matter content, 1684
 particle size distribution, 1684
 porosity or bulk density, 1684
 predictor variables, selection, 1683–1684
 quality, 1840
 reliability of, 1684–1685
 soil hydraulic characteristics, 1682
 soil properties, 1683
 soil structure and morphology
 descriptors, 1684
 water retention and hydraulic conductivity
 characteristics, 1682–1683
- Pedotransfer functions (PTFs)
 assessment, quality, 1844
- Pedoturbation, 2112, 2374–2375
- P-efficient rice cultivars, 1737
- Pelletize manures, 917
- Penetration resistance (PR), 185
- Penetrometers, 563
 resistance, 563
- Perennial grasses, 2378
- Perennial vegetation, 181
- Periglacial phenomena, 101–103
- Periodic leaching, 2043
- Permaculture, 707
- Permafrost, 1753
 active layer, 1688
 characteristics, 1687–1689
 impermeable to water movement, 1689
 living with, 1689
 schematic representation, 1688
 and seasonally frozen ground, 1689–1690
 surface energy balance, 1689
- Permafrost-affected soils, 1754–1755
- Permanent wilting point, 561
- Permanganate oxidation of soil, 1607
- Permeability, 2218
- Persistence, 1918
- Persistent organic pollutants (POPs), 1717
- Persistent organic pollution
 bioavailability, 1772–1773
 microbial ecology and biodegradation, 1774–1775
 molecular recalcitrance, 1771–1772
 nutrient availability, 1773–1774
- Pest control, 140, 229
- Pest-free seeds and transplants, 1602
- Pesticide
 and runoff, 841
 in stream and water bodies, 842
- Pesticide Directive, 1825
- Pesticides, 499, 690, 740
 and salt, 593
- Pesticides, behavior of, 1759
- Pest management in organic farming systems, 1601–1602
- Petrocalcic horizons, 341
 calcic horizons, 1691
 calcium carbonate, accumulation, 1692
 lateral distribution of, 1692–1694
 soil use, 1694
 types of, 1691–1692
 vertical succession of, 1692
- Petroleum hydrocarbons (PHCs), 1721
- Petroplinthite. *See* Plinthite
- Peyrosols, 393
- P fixation capacity, 918
- PH
 AL³⁺ ions, 1695
 CaCO₃, 1696
 factors that affect soil
 carbon dioxide, 1698
 dilution, 1698
 salt content, 1698
 suspension effect, 1698
 free acids, 1695
 hydroxy-Al, 1695–1696
 neutral soil, 1696
 organic matter, 1697–1698
 sodium carbonate, 1696
 in soil, 1695
 soil acidity, strength, 1697
- Phacelia, 73–74
- Phacelia (*Phacelia tanacetifolia*), 495
- Phaeozems, 409
- Phalaris aquatica*, 828
- pH dependence, adsorption, 1427
- Philip equation, 2509
- Phosphatase enzymes, 1508
- Phosphatases role, 1709
- Phosphate, 175
- Phosphate fertilizers, 911–912
- Phosphates, primary minerals, 1475
- Phosphogypsum, 1179
- Phospholipid fatty acid marker, 1829
- Phospholipids fatty acids (PLFA) extraction
 method, 1431
- Phosphoric acid, 355
- Phosphorus, 1228–1229
 equilibria, 1700–1701
 manure and diet modification
 impact on nutrient management, 1706–1707
 in nutrient management practices, 1705–1706
 plant nutrition and microorganism
 contribution
 future prospects, 1710
 microbial P and organic matter management, 1708–1709
 mineral P, active mobilization, 1709
 mycorrhizal fungi, 1709–1710
 phosphatases role, 1709
 soil-fertilizer P reactions
 absorption and translocation, 1703
 deficiency symptoms, 1703
 desorption, 1702
 initial rapid reactions, 1702
 plants nutrition, 1703
 plants requirement, 1703
 slow reaction phase, 1702
 soil P availability, 1702–1703
 spatial speciation in agricultural soils
 autocorrelation parameters, 1713–1714
 capacity factor, 1714
 chemical speciation, 1712
 heterogeneous soil, 1714
 homogeneous soil, 1714
 mobility in soils, 1713
 P fractions, 1714
 plant available P, 1714
 site-specific adsorption capacity, 1714
 site-specific anthropogenic and
 environmental factors, 1714
 variogram maps, 1715
- Phosphorus (P), 184, 1223
- Phosphorus pentoxide (P₂O₅), 910
- Photo-acoustic-infrared detector (PAID), 440
- Photogrammetric process, 1313
- Photo-oxidation processes, 202
- Photosynthesis, 210, 436
- Photosynthetically active radiation (PAR), 582
- Phreatic surface, 1990
- Phyllosilicates, 424, 426, 1360, 1427
- Physical degradation, 597
- Physico-chemical separation, sulfur, 2241
- Physics-based scaling
 fractal scaling, 1978–1979
 similitude-based scaling, 1976–1978
 water transport equations, scale invariance
 of, 1978
- Phytase, 1706
- Phytodegradation, 1720–1721
- Phytoextraction, 203, 1719–1720
- Phytomass decomposition, 235
- Phytophages, 138
- Phytoplankton, 15
- Phytoplankton cells, 276
- Phytoremediation, 203
 contaminants in soil and water, 1718
 enhancement of, 1722
 factors affecting, 1721–1722
 genetic engineering for, 1722–1723
 mechanisms of
 phytodegradation, 1720–1721
 phytoextraction, 1719–1720
 phytostabilization, 1719
 phytovolatilization, 1718–1719
 rhizofiltration, 1721
- Phytostabilization, 1719
- Phytotoxicity, 14, 1324
- Phyto-toxicity of aluminum (Al), 823
- Phytouptake and phytotoxicity, 887
- Phytovolatilization, 1718–1719
- Picard's methods, 930
- Pineapple (*Ananas comosus*), 1283
- Pinnacled crusts, 216
- Piping, 66
- Piping, slope failure, 1348–1349
- Pit checking method, 488
- Pit mine, 1462–1463
- Plaggen epipedon, 92
- Plagioclase, 1474
- Plains
 geomorphic components, 1342
- Planassolos, 381
- Planctomycetes, 179
- Planosols, 392, 409, 1646
- Planossolo, 382
- Plant–animal–soil system, 1035
- Plant available water, 2541
 limits, 2541
 defined, 2541
 estimation, 2543–2544
 field measurement of lower limit, 2543

- field measurement of upper limit, 2542–2543
- measuring, 2541–2542
- water management, 2544
- Plant-available water capacity (PAWC), 1665–1666
- Plant available water capacity (PAWC), 2056
- Plant breeding, 527
- Plant breeding for tolerance, 1362
- Plant cocktail, 1378–1379
- Plant community, 2573
- Plant debris, 560
- Plant density and row spacing, 229
- Plant establishment, 1306
- Plant establishment, structure
 - human impacts on, 2221–2222
 - plant effects on, 2221
 - plant growth and, 2222–2223
 - soil structure management, 2223–2224
- Plant–fungal symbiosis, 494
- Plant growth
 - on Cu-deficient soils, 482
 - on high-Cu soils, 482–483
- Plant growth (biomass yield), 228
- Planting time, 228–229
- Plant micronutrients, 1512
- Plant nutrient, 560
- Plant nutrient requirement assessment, 1582
- Plant nutrients, 190
 - buffer power, 1732–1733
 - classification and general function, 1729
 - definition, 1728–1729
 - fertilizer applications, 1733
 - historical retrospect, 1728
 - intensity factor, 1731–1732
 - magnesium in, 1392–1393
 - mineral nutrients for humans, 1730
 - nutrient requirement, 1735–1736, 1738
 - nutrient sufficiency, 1735–1738
 - quantity factor, 1732
- Plant nutrients, soil constraints and, 229–230
- Plant-parasitic nematodes (PPNs), 1520, 1523
- Plant pathogens, 138, 140
- Plant protection and diseases, 1436
- Plant residue, 189
- Plants absorb contaminants, 1720
- Plants nutrition, phosphorus, 1703
- Plant–soil system, 181
- Plants requirement, phosphorus, 1703
- Plant susceptibility, 14
- Plant system, nutrients, 1588
- Plasma membrane, 106
- Plaster, pores cast, 1388
- Plasticity
 - cations, 1741
 - clay content and clay type, 1741
 - organic carbon, 1741
 - pH, 1741
- Plastic limit (atterberg limits), 171
- Plastic properties
 - definition, 1740
 - factors affecting plasticity
 - cations, 1741
 - clay content and clay type, 1741
 - organic carbon, 1741
 - pH, 1741
 - plasticity, types of, 1741–1742
 - soil plasticity, theory of
 - cohesion, 1740–1741
 - water films, 1740
- Platanus occidentalis* (sycamore), 2573
- Plateau, 1363
- Platinum RTDs, 2303
- Platy structure, 2217
- Pleistocene soils, 157
- Plinthite, 346
 - definitions, 1743–1744
 - formation and morphology, 1744
 - management implications, 1744–1745
 - recementation, 1744
- Plinthosols, 392, 409, 1646
- Plintissolos, 381
- Plintossolo, 380, 382
- Plot design and layout, 816
- Ploughing of compacted soils, 74
- Plow implement of millennia, 732–734
- Plow-pan formation, 52
- Pocosins, 1107
- Podzols, 392
- Podura aquatica*, 449
- Podzólico Acinzentado, 380
- Podzólico Amarelo, 380
- Podzólicos, 112
- Podzólico Vermelho-escuro, 380
- Podzolic soil, 385
- Podzolization, 963, 1358–1359, 2114
- Podzols, 409, 1358–1359
 - agricultural use, 1752
 - formation of, 1748–1750
 - hydrosequence, 1750
 - macromorphology, 1747
 - micromorphology, 1747
 - mineralogy, 1747
 - occurrence of, 1751–1752
 - organic coatings in, 1751
 - organic matter chemistry, 1750–1751
 - pellet-like organic matter, 1750
 - pyrolysis-gas chromatography–mass spectrometry, 1750
- Podzol soils, 413
- Podzolvisols, 410
- Pogonomyrmex occidentalis*, 152
- Pogonomyrmex rugosus* nest disks, 151
- Point of zero charge (PZC), 2005
- Point of zero net charge (PZNC), 131
- Point of zero net proton charge (PZNPC), 2432
- Point source pollution
 - assessment, 1780
 - contaminants, nature and sources of, 1776
 - global challenges and responsibility, 1780
 - management and/or remediation, 1780
 - sampling for, 1779–1780
 - soil and environmental quality, implications to, 1779
 - soil and water, contaminant interactions in, 1776–1779
- Poiseuille's law, 759
- Polar biome, 1021–1023
- Polarographic soil O₂ sensors, 45
- Polar regions
 - environmental setting, 1753
 - soil characteristics
 - NCPS, 1756
 - southern circumpolar soils, 1756
 - soil-forming processes, 1753–1755
- Political science, 2026
- Pollen feeders, 1488
- Pollutant-referred supplements, 1825
- Pollution, 9
 - Cd, Pb and Cr contamination, 1760
 - Hg contaminations, 1760–1761
 - human impact, 1758–1760
 - industrial waste, 1762–1764
 - nickel contaminations, 1760
 - non-point source, 1765–1767
 - organic and inorganic, 1768–1770
 - persistent organic, 1771–1775
 - pesticides, behavior of, 1759
 - point source, 1776–1780
 - source materials causing, 1759
- Pollution-free and green food production systems, 1620
- Pollution-induced community tolerance (PICT), 2367
- Pollution offsets, 1356
- Polyacrylamide applications, 2043
- Polyacrylamide (PAM), 782, 825
- Polycarbonate tube, 121
- Polychlorinated biphenyls (PCBs), 199, 1779
- Polyculture ponds, 276
- Polycyclic aromatic hydrocarbon (PAH), 198
- Polygenetic soils and their classification, 2293–2294
- Polygon-based soil maps, 677
- Polymerase chain reaction amplification, microbial communities, 1434
- Polymeric (polycationic) Al, 106
- Polysaccharides, 55, 504
- Polyvalent cations, 757
- Poly (vinyl chloride) (PVC), 859
- Polyvinyl chloride (PVC) pipe, 121
- Ponding, 2485
- POPs. *See* Persistent organic pollutants (POPs)
- Pores, 1439
- Pore size, 1782
- Pore size distribution, 681
 - measurement methods, 1783–1784
- Pore system
 - definitions, 1782–1783
 - functions of, 1784–1785
 - measurement methods, 1783
- Pore walls, 1360
- Porosities, 1655, 1991
 - infiltration, 1188–1189
 - pore size distribution, 1782–1785
 - of soil, 139, 140
- Porosity (ϵ), 191, 258, 295
- Porous media, 87
- Portable measurement techniques, 2476
- Portlandite, 1417
- Positive-charge deficiency, 1241
- Positive lynchets, 2343
- Positive soil fertility management, 905
- Postactive acid sulfate soils, 18
- Postclassification comparison, 1314–1315
- Postcombustion capture, 1798–1800
- Postfire residues, 927
- Potash fertilizers, 912
- Potash fertilizer trials, 1796
- Potassium, 1223, 1229
 - dynamics and mineralogy
 - feldspars, 1791–1792

- Potassium—
 micaceous minerals, 1792
 transitional clay minerals and allophanes, 1792–1793
 fertility, 1789
 K dynamics, 1793–1794
 nutrition in semi-arid tropics, 1796–1797
 in plants, 1786
 in plant–soil interactions, 1788–1789
 in soils, 1786–1788
 soil test crop response, 1794
 Potassium chloride (KCl), 362, 912
 Potassium chloride (muriate of potash), 921
 Potassium magnesium sulfate, 921
 Potassium nitrate (KNO₃), 365, 921
 Potassium oxide (K₂O), 910
 Potassium sulfate (sulfate of potash), 921
 Potato cyst nematode, 1523
 Potato (*Solanum tuberosum* L.), 362
 Potential acidity, 33
 Potential ET (PET), 2135
 Potential evapotranspiration (PET), 216
 Potential evapotranspiration ratio (PET), 660
 Potentially dispersive soils, 2049
 Pottery, 1418
 Potworms. *See* *Enchytraeidae* (potworms)
 Poultry manure biochar, 185
 Power plants
 carbon dioxide capture, 1798–1800
 emerging capture technologies
 aqueous NH₃, 1801–1802
 chemical looping, 1802–1803
 improved auxiliary processes, 1803
 membranes, 1802
 novel liquid sorbents, 1802
 solid sorbents, 1802
 oxy-combustion, 1801
 postcombustion capture, 1798–1800
 precombustion capture, 1800–1801
 P pools, 1701
 Pra ambao, 403
 Prawn (polyculture), 275
 Precambrian rock, 111
 Precipitates, 8
 Precipitation, 6
 Precipitation (PPT), 698, 1359
 Precipitation storage efficiency, 2519
 Precipitator dusts, 1363
 Precision agriculture
 digital maps, 1805
 fertilization
 decision-making strategies, 1809
 georeferenced soil and crop information, 1808–1809
 technical equipment and data, accuracy of, 1809
 future, 1806–1807
 geographic information systems (GISs), 1806
 global positioning system (GPS), 1804
 highspeed computer processing, 1804
 high-speed computer processing systems, 1805
 nutrient cycling, 1811–1814
 paradigm for, 1804–1805
 remote sensing, 1815–1817
 sensing for, 1806
 Precision agriculture, pedometrics, 1680
 Precision farming, 1008–1009
 Precombustion capture, 1800–1801
 Precompaction stress, 563
 Predacious and omnivorous nematodes, 1520–1521
 Predatory nematodes, 140
 Predictive equations, 987
 Predictive equations for the soil gas diffusion coefficient, 673
 Predictive soil mapping, 676
 Predrainage, 1993
 Preferential flow, 1388–1390, 2218
 Pregastric digesters, 463
 Pregnant women, 1443
 Presidedress soil nitrate test (PSNT), 1540
 Pressure head, 1990
 Pressurized groundwater, 1349
 Primal religion, 1903
 Primary minerals
 oxides
 Fe, 1475
 Ti, 1475
 phosphates, 1475
 silicates
 feldspars, 1474
 micas, 1474
 olivines, 1473
 pyroxenes and amphiboles, 1473–1474
 quartz, 1474–1475
 zircon, 1473
 solubility, 1480
 Prismatic structure, 2217
 Process–response connotation, 1328
 Production–utilization cycle, 211
 Productivity
 anthropogenic disturbances, 1820–1821
 natural disturbances, 1819–1820
 quality
 dynamic soil quality, 1863–1864
 indicators, 1862
 inherent soil quality, 1862–1863
 soil quality and sustainable production, 1864–1865
 soil, 356
 and tillage erosion, 2338
 Productivity changes approach (PCA), 256
 Propane flame burning, 1602
 Properties, soil, 155–156
 Property, 1355–1356
 rights, 1355, 1356
 Prostigmata, 1488
 Proteaceae, 494
 Protected and unprotected agricultural lands, 70
 Protection of SOC, 327
 physical protection, 327–328
 constraints, 329–330
 spatial isolation, 328–329
 Protein manipulation, 243
 Proteobacteria, 179
 Proteolysis, 1791–1792
 Proto-imogolite allophane, 98
 Proto-imogolite complexes, 1359
 Proton activity, 1896–1897
 Protozoa, 141, 206
 activity and composition, factors affecting, 1828–1829
 bacterivory, 1829
 culturing techniques, 1829
 density-gradient centrifugation, 1829
 direct enumeration, 1829
 groups, 1829
 importance, 1828
 in laboratory bioassays, 1829
 phospholipid fatty acid marker, 1829
 Protozoans, 236, 237, 1625–1626
 Provitamin A (beta-carotene), 244
 Proximal gamma-ray spectrometers, 1873
 Psammets, 729
 Pseudoboehmite, 1469
Pseudomonas species, 200
Pseudomonas spp., 181, 240
 Pseudorutile, 1469
 Pseudosand, 56
 Pseudosilt, 56
Pseudosinella decipiens, 449
 Pseudotuberculosis, 238
 PSNT. *See* Presidedress soil nitrate test (PSNT)
 P-solubilizing compounds, 494
 Psychology, 2026
 Public Health Engineering Department (PHED), 162
Pueraria phaseoloides, 510
 Pulp and paper manufacture, 1180
 Pulsed neutron generator (PNG), 319, 320
 Pulverization, 597
 Punas, 101
 Purification of water, 1123
 Puro Pantad, 403
 Pycnometer, 1652
 Pyrite, 7
 formation, 1832
 framboidal, scanning electron microscopic image, 1832
 implications, 1833
 mineralogy and crystal chemistry, 1831
 occurrence, 1831–1832
 reactivity, 1832–1833
 schematic image, 1832
 semiconductor, 1831
 sulfuric acid, 1831
 Pyrite decomposition, 8
 Pyrite oxidation, 8
 Pyrogenic carbon (PyC), 1836
 impacts on climate change, 1836
 influence on soil, 1834–1836
 measurement and characterization, 1834
 occurrence in soil, 1836
 Pyrolysis, 184, 194
 Pyroxenes, 1473–1474
- Q**
 Quality
 assessment by interpolation techniques, 1844–1847
 biological properties, 1844
 compartmentalized framework, 1845
 computer-assisted production, 1844
 definition, 1844
 geographical information system (GIS), 1845
 indicators, 1844–1845
 mapping accuracy, 1845
 methods, 1845
 pedotransfer functions (PTFs), 1844

- science-based techniques, 1845
- wet combustion method, 1844
- assessment by spatial interpolation
 - decision trees, 1846
 - geostatistical analyses, 1846
 - interpolation efficacy, 1847
 - remote sensing, 1846–1847
 - sampling, 1846
- basic soil functions, 1837
- carbon gases
 - carbon dioxide, 1848–1849
 - methane, 1849
- critical limits and standardization, 1853–1855
- erosion, 1856–1858
- evaluation, 1838
- framework for assessing, 1837–1838
- functions, crop production, 1838
- indicators, 1859–1861
- indicators of, 1839
- inherent and dynamic soil, 1837
- input variable, 1839
- land management practices, 1838
- monitoring, 1838–1840
- nitrogen gases
 - ammonia, 1850
 - nitric oxide (NO) and NO_x, 1849–1850
- pedotransfer functions, 1840
- productivity, 1862–1865
- soil and water
 - decision making and policy, 1869–1870
 - functions related to water, 1866–1868
 - indicators of, 1868
 - scales of, 1868–1869
- soil quality index (SQI) assessment, 1837–1840
- sustainable development, 1837
- Quality, soil
 - aggregation, 1505–1506
 - definition, 1505
 - mycorrhizae and roots effect on, 1506–1507
 - mycorrhizae role in, 1506
 - mycorrhiza inoculations effect on, 1508–1509
 - mycorrhizal hyphae, glomalin, and, 1507–1508
 - soil quality index (SQI) assessment concept, 1841
 - conceptual framework for, 1842
 - method, 1841–1842
 - soil physical, chemical and biological index, 1842
- Quantify optimum nutrient rate, 1582
- Quantitative Evaluation of Fertility of Tropical Soils (QUEFTS) model, 1738
- Quantity factor, 1712
 - plant nutrients, 1732
- Quantum mechanical methods, 2167
- Quartz, 1469, 1474–1475, 1743
- Quartzpsamments, 346
- Quicklime (CaO), 823
- R**
- Radiation sensor (pyranometer), 90
- Radioactive elements and inorganic pollutants, 1769
- Radioactive source devices, 2477
- Radioactive waste, 1766
- Radiometric survey
 - airborne survey, 1873
 - applications, 1874–1875
 - background radiation and interactions, 1872–1873
 - proximal survey, 1873–1874
 - theory, 1872
- Radionuclides, 1337
 - fate and transport processes
 - geochemical processes, 1877
 - hydrologic processes, 1876–1877
 - microbial processes, 1877–1879
- Raindrop detachment–flow transport (RD–FT) system, 847
- Raindrop impact
 - energy, 2647
 - pressure, 1881–1882
 - stress associated with lateral flow, 1882
- Raindrop size distribution, 2647
- Rainfall, 832
- Rainfall erosivity, 814–815
- Rainfall-runoff (erosivity), 845
- Rainfall simulators, 815–816
- Rainfed areas, 299
- Rainfed croplands, 2552
- Rainsplash, 2485
 - detachment and transport, 2648–2649
- Rain use efficiency (RUE), 582
- Rainwater management
 - lower Himalayas, soil management, 1386
- Raised bed planting systems, 531
- Raised-bed system of cultivation, 529
 - modern applications, 530–531
 - need and ecological niche, 529–530
 - research and development challenges, 531–532
- Ralstonia eutropha*, 200
- Rangelands, 1039, 2552
- Rape (*Brassica napus* L.), 501
- Rapid soil analysis. *See* Diffuse reflectance spectroscopy (DRS)
- Rare earth elements (REEs)
 - adsorption, 1892
 - application, yield increase, 1894
 - Ce content in grains and seeds, 1893
 - chemical speciation
 - species of, 1891
 - total content, 1891
 - concentrations and coefficient of variation, 1892
 - crop response, 1894
 - mean content of, 1892, 1893
 - origins, 1891
 - plant-available, critical values of, 1894
 - symbol and atomic numbers, 1892
 - total concentration, 1893
 - translocation, 1892–1893
 - uptake by plants, 1894
- Raster-based GIS systems, 1320
- RCC. *See* Roller-compacted concrete (RCC)
- Reaction, soil, 1434
- Reactive Fe, 27
- Reactor-based processes, 201
- Reaggregation policies, 650
- Real-time kinematic (RTK-GPS) systems, 1805
- Real-time sensors, 1540
- ReaxFF force field, 2167
- Recalcitrant contaminants, 202
- Recent organic carbon (ROC), 1449–1450
- Reclaimed minesoils (RMS), 1450
- Reclamation, 792, 1821
- Reclamation, sodic soils
 - biology and organic matter role, 2052
 - chemical ameliorants
 - calcareous sodic soils, 2051
 - gypsum, 2051
 - lime, 2051
 - crops, relative tolerance of, 2054
 - displacement of exchangeable sodium ions, 2051
 - rate of supply of ameliorants, 2054–2055
 - supply of ameliorants, 2054
 - supply of water for, 2054
 - water flow, 2051
 - without the addition of ameliorants, 2052–2054
- Recommended dose of fertilizer (RFD), 1384
- Recommended management practices (RMPs), 299
- Recycling of elements, 1123
- Red clover plants (*Trifolium pratense*), 221
- Red edge effect, 1334, 1341
- Redox concentrations, 128
- Redox phenomena
 - applications of, 1897–1898
 - electron and proton, nature and role, 1896–1897
- Redox potentials, 27, 126, 562
- Redox processes (pseudogley), 93
- Redox sensitive radionuclides, 1878–1879
- Red soils, 452
- Reduced tillage, 896
- Reduced till/zero till/no-till agriculture, 707
- Reductants and oxidants, 1481
- Reduction–oxidation cycles, 730
- Reduction–oxidation potential, 2572
- REEs. *See* Rare earth elements (REEs)
- Reference methods, soil, 1887–1888
- Reference value, 1853
- Référentiel Pédologique* (RP), 391, 392
- Reflectance measurements, 1816
- Reflection hyperbola analysis, 1010
- Reforestation, 1745
- Regions of interest (ROIs), 1873–1874
- Regosolic soil, 385
- Regosols, 392, 409
- Regression coefficient, 845
- Regulation, 1356
- Regulatory agencies, industrial by-products, 1181–1182
- Rehabilitation, 655, 2464
 - development, 1936
 - restoring growing medium for plants, 2465
- Rehabilitation, indicators and monitoring
 - bauxite, 1900, 1901
 - biological components and processes, 1900
 - characteristics, 1899
 - functions, 1899
 - invertebrates, 1900
 - landscape function analysis (LFA), 1901
 - microbial biomass, 1899
 - mycorrhizal fungi, 1900
 - nutrient supply and uptake, 1900
 - plant productivity, 1901
 - respiration, 1899

- Rehabilitation—
 soil microbial biomass, 1899
 soil properties, 1899–1900
 total C content, 1900
 vesicular–arbuscular mycorrhizal fungi, 1901
- Rehydration method, 1430
- Relative resistivity/conductivity boundary shifts, 1009
- Relevant soil depth (RSD) characteristics, 1910
- Religious aspects, soil–human relationships
 critical religion, 1903
 definition, 1903
 earth and belief systems, 1904
 guidance religion, 1903
 natural religions, 1903
 primal religion, 1903
 salvation religion, 1903
 spiritual aspect of human interventions, 1904–1905
 translations, 1903–1904
- Remediation, soil, 1763–1764
- Remediation strategies, 37–38
- Remediation technologies, 1780
- Remolded soils, 171
- Remote sensing, 771, 1313, 1806, 2555–2558
 assessment, quality, 1846–1847
 precision agriculture
 data sources, 1816
 soil moisture, 1817
 soil organic matter, 1816
 soil texture, 1816–1817
- Remote-sensor system, 1316
- Rendzinas, 415, 1907–1908
- Renormalization methods, 931
- Replacement, soil, 1459
- Replacement cost approach (RCM), 256
- Residual shrinkage, 171, 2009
- Residue management, 492, 762
 practices, 497
 rice paddies, 1951–1952
- Resilience
 definition, 1910
 factors affecting, 1910–1911
 land use and soil management
 agricultural soil management, 1914–1915
 concept, 1913–1914
 principles and practices of, 1915–1917
 mass flux balance approach, 1910
 nature, 1909–1910
 as process, 1909
 quality and
 assessment of, 1920–1922
 concepts and definitions, 1918–1920
 dynamic equilibrium, 1919
 exogenous and endogenous factors, 1921
 factors affecting, 1920
 response to input of, 1920
 and soil restoration, 1922
 sustainability, 1922
 temporal changes in, 1920
 types of, 1919
 relevant soil depth (RSD)
 characteristics, 1910
 restoration, 1924–1927
 state, 1910
- Resilience and restoration of ecosystems, 1123
- Resin-impregnated soil, 1389
- Resistance and resilience, 218–219
- Resistance temperature detectors (RTDs), 2303
- Resistivity, 1005
- Resource islands, 650
- Resources Conservation and Recovery Act (RCRA), 1181
- Respiration
 autotrophic, 1928
 and carbon sequestration, 1929–1930
 climate warming, 1928
 components, 1928
 controlling factors of, 1929
 humus, 1928
 importance, 1928
 management and season effect on, 1930
 measurements, 1928–1929
 organic carbon mineralization, 1930
- Restoration
 peatlands, 1672–1673
 resilience
 compaction, 1926
 definition, 1924
 degradation and erosion processes, 1925–1926
 erosion, 1926
 fertility, 1926
 and lack of degradation, 1925
 nutrient mining, 1926
 salinization, 1926
 sealing of surface, 1926–1927
 services, 1924
 soil depth, 1926
 sustainable soil and land management, 1925
 success and completion criteria
 AlcoaWorld Alumina, 1937
 characteristics, 1935
 Ecosystem Function Analysis (EFA), 1936
 ecosystem of reference, 1935
 environmental indicators, 1935–1936
 eucalypts and legumes, 1938
 human-induced degradation, 1936
 jarrah forest, 1938
 land manager, 1936
 litter nutrient content, 1938
 litter production, 1938
 microbial biomass and microbial quotient, 1938
 problems identification, 1938
 rehabilitation development, 1936
 soil nitrogen, 1938
 total ecosystem nitrogen, 1938
 vital ecosystem attributes, 1936
- Restoration ecology, 874
 abiotic limitations, 1934
 assessment, 1934
 biotic manipulations, 1934
 degrading processes, 1933
 dynamic systems and restoration goals, 1932–1933
Eucalyptus marginata forest, 1933
 food-web complexity, 1934
 measures of success, 1934
 processes involved in, 1933
 stressors, 1933
- Restoration of wetlands, 286
- Retaining soil cover, 513
- Revegetation, 2465–2466
 erosion, 1820
 of minerals processing residue, 2288
See also Landfill
- Revised universal soil loss equation (RUSLE), 460, 461, 833
- Revised USLE (RUSLE), 845, 1881
See also Universal soil loss equation (USLE)
- Rhizobacteria, 1436
- Rhizobia, 228, 232
- Rhizobial inoculation, 228
- Rhizobium*, 714, 1436
- Rhizobium bacteria, 105, 1039
- Rhizobium bacteria, 553
- Rhizobium japonicum*, 220
- Rhizobium leguminosarum*, 220
- Rhizodegradation, 1721
- Rhizodeposition
 estimation, 1553
- Rhizofiltration, 1721
- Rhizopoda, 237
- Rhizosphere, 207, 722
 respiration, 1928
 sulfur
 chemical behavior, 2247–2249
 ecological effects, transformations in, 2249–2250
- Rhodococcus*, 41
- Rhodospirillum rubrum*, 222
- Ribosomal ribonucleic acid (rRNA), 175, 177
- Rice (*Oryza sativa*), 118, 299, 934–935
 crop, 1228
 cultivars, 527
- Rice paddies
 methane emissions
 controlling, 1943
 cultivars, 1941
 diel and seasonal patterns, 1940
 flooded rice cultivation, 1946
 inorganic fertilization, 1941
 mitigating, 1943–1945
 organic amendments, 1940–1941
 organic matter management, 1944
 photosynthetic activity, 1940
 problems and feasibility, 1945–1946
 production, oxidation and emission, 1944
 soil amendments and mineral fertilizers, 1943–1944
 soils, 1941
 water management, 1941, 1943, 1945
- methanogenesis
 distribution from rice agriculture, 1947–1948
 globally observed seasonal emissions, 1948–1949
 mitigation, 1949
 production and emission, 1947
- soil management
 characteristics, 1950–1951
 management approaches, 1951–1952
- Rice–wheat–jute system, 303
- Richards' equation, 931
- Richness factor, 1712
- Ridge-till, 764
- Rigidity, 1355
- Rill and interrill erosion, 1987
- Rill erosion, 816–817, 1061
 rates, 2505

- Riparian forest, 208
 Ripening/aerobic breakdown, 1108
 Ripening or flooding, 38, 2113
 Ripping, 2210
 River bottomlands, 70
 agriculture, and levees, 65–66
 River floodplains, 157
 River microterraces, 295
 RMS. *See* Reclaimed mine soils (RMS)
 Road maintenance, 113
 Robert Koch's techniques for isolating pathogens, 176
 ROC. *See* Recent organic carbon (ROC)
 Rocks, 2560
 mantle deposits
 chemical composition and particle size distribution, 1954–1955
 mineral components, 1953–1954
 transformation, 1955
 phosphate, 1492
 weathering, 1953
 Rocky Mountains, 101
 Rofabard, 627
 ROIs. *See* Regions of interest (ROIs)
 Roller-compacted concrete (RCC), 1417
 Rolling topography, 1345
 Roman–Byzantine periods, 733
 Root biomass, 211
 Root–cell membrane, 481
 Root development, 502–503
 Root-feeding nematodes, 138
 Root growth, 51, 501
 Root-induced macropores (RIMs), 2226
 Root initiation, 502
 Root interception, 1580–1581
 Root-knot nematodes, 1523–1524
 Root nodule bacteria (*Rhizobium* species), 220
 Root orientation, 503
 Roots
 cycling, no-till, 1378
 feeding, 1553
 growth, soil–root interface pressure
 axial root growth force, 1958–1959
 axial root growth pressure, measurements of, 1958–1959
 in compacted soils, 1957–1958
 developmental zones in, 1958
 estimation, 1957
 maximum axial growth pressure, 1958
 penetrometer resistance, 1957
 root diameter, 1958
 soil deformation, 1957
 soil penetrometer resistance, 1958
 hairs, nutrients, 1591
 monitoring system
 excavation method, 1960–1961
 minirhizotron method, 1960, 1961
 nuclear magnetic resonance method, 1962
 observation of *in situ* root system, 1963
 observing system, 1962–1963
 plexiglas soil columns, 1962
 profiling methods, 1960
 root-coring method, 1961
 soil-coring, 1960
 proliferation and P level, 1703
 response, ODR, 1726
 structure
 contributions to soil aggregates, 2225
 root–soil aggregation interactions, 2225–2226
 soil aggregates, microdensitometry, 2226–2227
 turnover, 2112
 Roots and vesicular–arbuscular mycorrhizal (VAM), 55
 Root Zone Water Quality Model (RZWQM), 2511–2512
Rosa rugosa, 829
 Rosin–Rammeler distribution models, 62
Rossius kessleri, 237
 Rotational grazing, 1040
 Rotations to manage soil moisture deficit, 513
 Roundness assessments, 1659
 Roundness grade scales, 1660
 Roundup-ready soybean, 243
 Row cultivation, 1493
 RTDs. *See* Resistance temperature detectors (RTDs)
 Rubber (*Hevea brasiliensis*), 224
 plantations, 819
 Rubrožem, 380
 Rugose crusts, 216
 Ruilverkaveling, 1309
 Runoff, 782, 1262
 events, 832–833
 farming, 2515
 generation processes, 1137
 source areas, 1137
 Rural civilization, 2027
 Rural–urban fringes, 255
 Russian Chernozem, 357
 Russian classification systems, 415–417
 Rye (*Secale cereale* L.), 482, 501
 Ryegrass (*Lolium westerwoldicum*), 495
 S
 Sage and mesquite plant communities, 1333
 Salic horizon, 559
 Saline
 fluvo-aquic Inceptisols, 900
 formations, 993–994
 seeps, 2193
 Salinity, 559, 1421
 control, 1253
 monitoring, 1010–1011
 and sodicity, 1766
 soil, 1269–1270
 irrigation and drainage, 1270–1271
 irrigation and drainage management, 1271
 subsoil sodicity, 2056
 Salinization, 659, 662, 2037
 degradation by, 1184–1185
 process, no-till, 1378
 restoration, resilience, 1926
 Salinolysis, 2560
 Salt
 in alkaline soils, 1579
 concentration, 354
 content, 2469–2470
 soil pH, 1698
 efflorescences, 2193
 marsh, 2568
 Salt-affected soils, 1969
 crop tolerance, 1966–1967
 extent and distribution, 1965–1966
 nitrous oxide emissions, 1969–1970
 Salt-affected soils and their classification, 2293
 Saltating soil, 742
 Saltation mode, 2618
 Salvador Dali, 169
 Salvation religion, 1903
 Sample locations and interpolated, 1314
 Samples preparation for laboratory analysis, 2320
 Sampling, 2320
 depths and pollution, 1779–1780
 from paired sites, 881
 soil, 1679
 Sand
 boils, 67–68
 delta, 68
 drown, 1393
 Sandblasting, 2645
 Sandy podzols, 1747
 Sandy soils, 681
 San Joaquin Valley, California, United States, 2553
 San Pedro Riparian National Conservation Area, 630
 Saprock, 2560–2561
 Sapolite, 419, 2561
 Saprophages, 138
 Saprotophs, 979
 SAR. *See* Sodium adsorption ratio (SAR)
 SAR systems and data, 2481
 SASSs. *See* Sodium-affected soils (SASSs)
 Satellite mapping
 direct evidence, 1973
 indirect evidence, 1973–1974
 sensors, 1972–1973
 Saturated hydraulic conductivity, 85
 Saturation
 extract methods, 2321
 soil, 526
 Sawdust, 1056
 Scale
 landscape qualifier, 1334
 spatial, 1337–1338
 timescales, 1336–1337
 Scaling effects
 empirical scaling
 inverse modeling, 1979
 pedotransfer functions and soil–landscape relationships, 1979
 transition from laboratory to plot scale, 1979
 physics-based scaling
 fractal scaling, 1978–1979
 similitude-based scaling, 1976–1978
 water transport equations, scale invariance of, 1978
 Scandinavia, boreal forest soils, 248
 Scanning electron microscopy (SEM), 156
 Scarcity of rain, 157
 Scavenge acidic, 15
 SCFs. *See* Sub- and super-critical fluid (SCFs)
 Schwertmannite [Fe₈O₈(OH)₆SO₄], 8
Scorpan, 1409–1410
 Scorpan factors, 677
 SDSS. *See* Spatial decision support system (SDSS)
 Se, mineral nutrition disorders, 1446

- Sealing mine shafts, 9
- Sealing of surface
restoration, resilience, 1926–1927
- Seasonal irrigation efficiency, 1257–1258
- Seasonal maxima, protozoa, 1828
- Secondary carbonate accumulations, 335
- Secondary clay minerals, 1659
- Secondary minerals
carbonate and sulfate minerals, 1478–1479
chlorites, 1478
hydroxy-interlayered vermiculite (HIV) and smectite, 1478
illites, 1477
iron and aluminum oxides, 1476
kaolinite and halloysite, 1476–1477
smectites, 1478
solubility, 1480
vermiculite, 1477–1478
- Second-generation biofuel crop, 1458
- Sediment, 69, 294, 297, 771
aquatic life influence
composition and particle size, 1982
graphical representation of, 1982
inputs, causes and consequences, 1981–1982
sedimentation, 1984
suspended, 1982–1984
basins, 791
bodies, 1339
deposition in road and drainage ditches, 69–70
transport and supporting hydrologic data, 464
- Sedimentation, 464, 772, 1517, 1984
control and erosion
conservation structures, 1988
contouring, 1986
contour ridges and stone terraces, 1986–1987
geotextiles, 1989
gully stabilization structures, 1988–1989
terraces, 1986
waterways, 1987–1988
parameter calculation, 1652
- Sedum alfredii*, 1720
- Seed bank, 868
habitats and regions, 868
- Seed bank, soil
genetics, 869–870
seed longevity in soil, 868–869
variability, 869
in space, 869
in time, 869
- Seedbed preparation, 51
- Seed germination, 51
- Seed-harvesting ants, 151
- Seedling emergence, 51, 132
ODR, 1725–1726
- Seedling infection, 1503–1504
- Seed treatment
selenium, 1996
- Seepage
applications
dams and levees, 1992
dewatering, 1993
water supply and groundwater contamination, 1992–1993
hydraulic conductivity, 1991–1992
principles, 1990–1991
zones, 1328
- Seepage salinity, 2030
- Seepage velocity, 1991
- Selenious acid, 355
- Selenium (Se)
bioavailability, 1997
cycles of, 1996
field treatment
foliar application, 1996
guidelines, 1997
in large scale, 1996–1997
risk assessments, 1997
seed treatment, 1996
soil application, 1995–1996
geographical distribution, 1995
sources, 1996
toxicity, 1997
uptake by plants, 1995
- Selenomethionine (SeMet), 1718
- Self-balancing, 2638
- Self-regenerating legumes, 1422
- Semiarid bioclimate, 300
- Semiarid or drier climatic regimes, 317
- Semiarid transitional aeolian plain, IGP, 1173–1174
- Semiarid tropics, 299
- Semiempirical methodologies, quantum mechanical methods, 2167
- Seminal roots, 72
- Semivariogram, 1812
- Sensible heat flux, 1086
- Sensing for precision agriculture, 1806
- Sensitivity
landscape, 1344
- Sensors, 45
- Sequestration, 79, 294, 1466–1468
- Serpentine barrens, 1999
- Serpentinic soils
antigorite, 1999
lizardite, 1999
management strategies, 2000
minerals identification, 1999
properties, 1999–2000
serpentine barrens, 1999
thermogravimetric analysis, 1999
vegetation, 1999
- Serradella (*Ornithopus sativus*), 495
- Sesbania sesban*, 827
- Sesquioxides, 1476
Al oxides
gibbsite formation, 2005–2006
minerals, 2005
properties, 2006
iron (Fe) oxides
adsorption of oxyanions, 2005
chemical and physical properties, 2005
minerals, 2003
pH dependent charge, 2005
properties of, 2003–2005
wet soil features, 2005
Mn oxides
minerals, 2006
properties, 2006
properties of, 2004
relationships between oxide minerals, 2006–2007
- Settling basins, 830
- Sewage
sludge, 1996
treatment plants, 917
water. *See* Wastewater
- Sewage Sludge Directive, 1824
- Shadow shielding, 320
- Shallower soils, 206
- Shear stress, 1882
- Shifting agriculture, 1281
- Shifting cultivation
lower Himalayas, soil management, 1383
- Shoot response, ODR, 1726–1727
- Shorelines, 203
- Shortrange order (SRO) solids, 131
- Short-term water deficiency, 73
- Shoulder, 1327
- Shrimps, 275
- Shrinkage, 2217
antecedent water content, 2008
factors affecting, 2008
measurement
natural soil aggregates, 2010
remolded samples, 2009–2010
vertical volume change, 2010
mechanisms, 2008–2009
representations, 2008–2009
residual, 2009
soil classes, cut-off values, 2010
whole soil shrinkage, 2009
zero, 2009
- Shrinkage intensity, 172
- Shrinkage limit (Atterberg limits), 171
- Shrinking–swelling silicate clay minerals, 1032
- Shrub and woodland, 965
- Side looking airborne radar (SLAR), 2481
- Sieved and repacked soil, 987–988
- Silage-making processes, 199
- Silhouette area index (SAI), 2518
- Silica, pedogenic accumulation
dissolution, 2012
morphology, 2013
movement and precipitation, 2012
sources, 2012
- Silica cementation, 2013
- Silica polymerization, 97
- Silicates
primary minerals
feldspars, 1474
micas, 1474
olivine, 1473
pyroxenes and amphiboles, 1473–1474
quartz, 1474–1475
zircon, 1473
weatherability, 1358
- Siliceous crystalline rocks, 999
- Silicic acid, 98
- Silicon, 2201
active uptake, 2015–2016
application on rice growth, 2017
beneficial effects, 2016–2017
mineral stress alleviation, 2018–2019
passive uptake, 2015
photosynthesis, Si-stimulated, 2017
rejective uptake, 2015–2016
resistance to climatic stress, 2018
silica cell and silica body, 2016

- silicate fertilizers, 2019
- susceptibility to disease and insect damage, 2017–2018
- transpiration of panicle, 2018
- water stress alleviation, 2018
- Sills, 1073
- Silt content, 1662–1663
- Silt-sized particles, Loess, 1365–1366
- Silty soils, 681
- Single-celled animals, 141
- Single grain soils, 2216
- Single-period crop soil management, 716
- Single superphosphate, 921
- Site preparation
 - anthropogenic disturbances, 1821
- Site-specific farming (SSF), 1018
- Site-specific nutrient management, 1584
- Site suitability analyses, 1319
- Skogar (woods), 624
- Slaking, 114, 758, 1359
- Slaking and dispersion
 - aggregate bonding nature, 2024
 - consequences of, 2022
 - factors affecting susceptibility to, 2022–2024
 - mechanisms of, 2021–2022
 - particles and structural elements, hierarchical organization of, 2022
 - schematic diagram, 2023
- Slaking under rapid wetting, 1073
- Slash and burn agriculture, 1383
- SLEMSA. *See* Soil loss estimation model for southern Africa (SLEMSA)
- Slight resilience, 1919
- Slope
 - components and hillslope profile, 1328
 - gradient, 1330
 - orientation/aspect, 1329
 - shapes with flow lines, 1329
- Slope complexity, 1340
- Slope failure
 - flow, 1349
 - landslide and, 1349
 - piping, 1348–1349
 - pressurized groundwater, 1349
 - surface erosion, 1349
 - surface saturation, 1348
- Slope gradient, 1341
- Sloping Agricultural Land Technology (SALT), 1283
- Slough hay, 1671
- Slow and controlled release fertilizers, 921
- Slow reaction phase, phosphorus, 1702
- Slurry bioreactors, 203
- Small grains, 479
- Small plot, 815
- Small-scale processes, 931
- SMB. *See* Soil microbial biomass (SMB)
- Smectites, 423, 1478
 - clay, 114
- Sminthurus viridis*, 449
- Snow and freezing conditions, 832
- Snowmelt
 - amelioration, 833
 - modeling, 833
 - precipitation, 831
 - runoff events, 832–833
- snow and freezing conditions, 832
- soil, 831–832
- SOC. *See* Soil organic carbon (SOC)
- Social and behavioral sciences
 - American agricultural subculture, 2027
 - cultural anthropology, 2026, 2027
 - economic values and policies, 2026
 - interactive issues, 2028
 - latent policy, 2026
 - latent policy transactions, 2027
 - market oriented economy, 2028
 - political science, 2026
 - psychology, 2026
 - rural civilization, 2027
 - sociology, 2026
- Social and economic justice, 849
- Social–ecological interactions, 643
- Social forestry, 257
- Sociology, 2026
- SOC mapping
 - land management, 1414
 - materials and methods, 1413–1414
 - soil map of Turkey, 1413
 - stocks according to regions, 1414
 - stocks of Turkey, 1413
- SOC under various land use systems
 - Lembucherra, Tripura (low altitude), 1384
 - Tadong, Sikkim (high altitude), 1384–1385
 - Umiam, Meghalaya (mid-altitude), 1384
- Sodic and saline soils, 2032–2033
- Sodication, 560
- Sodicity, 560, 2470
- Sodicity effect
 - hydrologic properties and behavior, 2045–2046
 - mechanical properties and behavior, 2046
 - swelling and dispersion, 2044–2045
- Sodic mines oil remediation, *in situ*, 1463
- Sodic soil
 - characteristics, 2048
 - classification, 2049–2050
 - exchangeable sodium percentage (ESP), 2029
 - formation and global distribution
 - geological—long-term development, 2035
 - global distribution of, 2035
 - origin and formation, 2033–2034
 - short-term development, 2035
 - sodic and saline soils, 2032–2033
 - humid regions, 2037–2039
 - irrigation farming, 2041–2043
 - management of dryland, 2031
 - physical and chemical properties of, 2030
 - physical properties and behavior, 2044–2046
 - processes, 2048–2049
 - reclamation, 1212, 2051–2055
 - root zone constraints in dryland, 2030–2031
 - salt accumulation in root zones, 2029–2031
 - world distribution of, 2030
 - yield decline in, 2029
- Sodium
 - beneficial effects, 2019
 - essentiality, 2019
 - fertilizers, 2019
- Sodium adsorption ratio (SAR), 784, 1463, 1965–1967, 2038, 2321
- Sodium-affected soils (SASs)
 - factors, 2037
 - genesis of, 2037
 - identification, 2039
 - land management classification, 2038–2039
 - managing, 2039
 - mapping, 2039
 - nomenclature and classification, 2038
 - sodic soil genesis model, 2037–2038
 - soil moisture recharge in, 2037
 - taxonomical classification, 2038
- Sodium and magnesium sulfates, 2240
- Sodium carbonate, soil pH, 1696
- Sodium dithionite-citrate-bicarbonate, 1397
- Sodium iodide scintillation crystals, 1873
- Sodium tetraphenyl boron (NaTPB), 1793
- Soh-phlong, 1283
- Soils
 - accumulation rate, 1336
 - acidity, strength, 1697
 - agriculture effect, 1493
 - boundaries, 1339
 - chemical condition, 1358
 - and civilization, 1517
 - definition, 1675
 - formation and degradation of, 1517
 - geomorphology, 1334
 - landscape, 1339
 - modern urbanization and industrialization effect, 1493
 - pore space, 1359–1360
 - properties of, 1516–1517
- Soil-air-water relationships, drainage
 - aeration, 681
 - bulk density, 680
 - pore size distribution, 681
 - soil structure, 681
 - soil texture, 680–681
 - trafficability, 681–682
- Soilborne pathogens, 1525
- Soil Fe, 2003
- Soil-forming, 92, 155, 348, 349
 - additions, 348–349
 - biota, 351
 - climate, 350
 - losses, 349
 - parent rock, 349–350
- Spodosols
 - climate, 2202
 - humus and sesquioxides, translocation, 2200–2201
 - parent material, 2201
 - silicon, 2201
 - time, 2202
 - topography, 2201
 - vegetation, 2201–2202
 - time, 350
 - topography, 350–351
 - transformations, 348
 - translocations, 348
- Soil-gas diffusion coefficient, 986–987
 - predictive equations, 987
 - sieved and repacked soil, 987–988
 - undisturbed soil, 987
- Soil inorganic carbon (SIC), 315, 335, 1048, 1191, 1206, 1449
 - chemical reactions, 1191, 1195
 - gravimetric techniques, 1191–1192

- Soil inorganic carbon (SIC)—
 - titrimetric techniques, 1192
 - volume/pressure techniques, 1192
- and climate, 1195–1196
- ecological regions, 1208–1209
- flux of SIC stocks, 1209
- integration of climate and time, 1197
- magnitude of SIC stocks, 1206–1207
- morphology of SIC stocks, 1206
- sink capacity, 317
 - base cations availability, 317–318
 - neutral and alkaline pH, 317
- soil moisture conditions, 1208
- soil taxonomy orders and suborders, 1207–1208
- thin sections, 1193
- and time, 1196–1197
- X-ray diffraction, 1192–1193
- X-ray fluorescence and microprobe, 1193
- Soil law
 - effectiveness of, 1353–1354
 - national, 1353
- Soil-less culture, 1053
- Soil-less systems, 533
 - economic considerations, 536
 - opportunity and challenges, 533–534
 - and product quality, 534–536
 - systems without a solid medium, 534
 - and yield, 534
- Soil loss estimation model for southern Africa (SLEMSA), 2491
- Soil microbial biomass (SMB)
 - direct methods to measure, 1429
 - indirect methods to measure
 - adenosine triphosphate extraction method, 1431
 - chloroform fumigation incubation and extraction methods, 1429–1430
 - freeze-dried soil extraction method, 1430
 - microwave irradiation incubation and extraction methods, 1430
 - phospholipid fatty acids extraction method, 1431
 - rehydration method, 1430
 - substrate-induced respiration (SIR) Method, 1431
 - UV spectroscopic method, 1431
- Soil-nesting ants, 150, 152
- Soil organic carbon (SOC), 307, 315, 777, 801
 - carbonates, 1604–1606
 - colorimetric determination, 1607
 - combustion methods for, 1606
 - DRS, 1888
 - enhancement in mine lands, 1466
 - mapping
 - land management, 1414
 - materials and methods, 1413–1414
 - soil map of Turkey, 1413
 - stocks according to regions, 1414
 - stocks of Turkey, 1413
 - near-infrared reflectance spectroscopic (NIRS) analysis, 1606–1607
- PyC, 1836
- rapid dichromate oxidation methods, 1606
- sequestration, 1466–1467
 - agricultural lands, 1610
 - application, 1609–1610
 - baseline steady-state assumption, 1611–1612
 - C-rich amendments, 1612–1613
 - depth distribution, 1610
 - experimental designs, 1610–1611
 - levels determination, 1610
 - microbial role, 1612
 - pretreatment measurements, 1611
 - stock change rate, 1611
- sequestration rates with different integrated
 - nutrient management practices, 52, 210, 299, 301–302
- sink capacity, 315
 - clay mineralogy, 316–317
 - depth and bulk density, 317
 - particle size distribution, 316
 - structure, 315–316
- soil quality, 1505–1506
- total C methods, 1604
- Soil Organic Matter Network (SOMNET)
 - database, 2159
- Soil organic matter (SOM), 40, 151, 177, 210, 356, 885, 971, 1035, 1144, 1440, 1467, 1610, 1639, 1868
 - accumulation
 - environmental controls, 2114–2115
 - littering, 2112
 - in mineral soils, 2113–2114
 - organic matter (OM) transformations, 2112–2113
 - in organic soils, 2113
 - pedoturbation, 2112
 - root turnover, 2112
 - subsurface, 2114
- assessment of, 2109
- available water capacity, 2117–2120
- biomarker techniques, 2109
- bulk chemical composition, 2126–2127
- characterization, 2123–2125
- chemical characterization and isotopic studies, 2127
- content, 1505
 - lower Himalayas, 1382–1383
- dynamics, 2108–2109, 2127–2128
- ecosystem services provided, 2109
- global carbon cycle
 - carbon components, 2129
- C fluxes, 2129–2130
- soil C fluxes and global change, 2130–2131
- global distribution, 2133–2137
- historical concepts
 - agronomic function, 2139–2140
 - environmental function, 2140
- hydrophobicity of, 1144
- landscape, 2142–2145
- management of, 2109–2110, 2147–2151
- mass spectrometry data analysis, 2154–2157
- modeling, 2159–2163
- molecular simulations, 2166–2169
- nutrient dynamics, 2172–2174
- remote sensing, precision agriculture, 1816
- solvent-based high resolution mass spectrometry, 2176–2179
- specific components of soil quality, 2110
- structure and characterization, 2180–2184
 - degradative methods, 2180–2182
 - individual plant and microbial components, 2182
 - SOM continuum, 2182–2183
 - spectroscopic techniques, 2180–2182
 - turnover, 2186–2190
- Soil P availability, 1702–1703
- Soil-P cycle, 1701
- Soil–physiography relationship, 1157
- Soil–plant–atmosphere system, 1577
- Soil resistance curve (SRC), 2531
- Soil Resource Mapping (SRM) project, 1157
- Soilscapes, 1339
- Soil science subdisciplines
 - pedometrics in, 1681
- Soil's nutrient-holding capacity, 1665
- Soil-surface management and erosion control, 1497
 - mulch farming, 1497
- Soil temperature regimes (STRs), 111
- Soil test P (STP) concentration, 1705
- Soil-to-seed contact, 462
- Soil–water–chemical interactions, 689
- Soil water content (SWC), 683
- Soil–water relations, 217
- Soil–water release curves (SWRCs), 339
- Soil–weather–plant interactions, 514
- Sokolov, I.A., 437
- Solenopsis richteri*, 150
- Solid fertilizers, 920–921
- Solid mineral, 1481–1482
- Solid sorbents, 1802
- Solid waste disposals, 1319
- Solodization, 2038
- Solodized solonetz, 1646
- Solonchaks, 392, 409
- Solonetz, 392, 409
- Solonetztes, 415
- Solonetzic soil, 385, 387
- Solonization, 2037–2038
- Solos Litólicos, 380
- Solubility
 - minerals, 1358
- Soluble Fe sulfates, 2239–2240
- Soluble salts
 - stemming, 559
 - translocation and accumulation in landscapes, 2192
 - within soils, 2192–2193
- Solum, 392
- Solute transport
 - nonequilibrium sorption, 2196
 - principles of
 - advection, 2194
 - Advection–Dispersion Equation (ADE), 2194–2195
 - diffusion, 2194–2195
 - dispersion, 2195
 - sorption, 2195–2196
 - transfer functions and solute movement, 2196
- Solution–precipitation mechanism, 1478
- Solvent-based high resolution mass spectrometry, SOM
 - dissolved organic matter (DOM), 2177
 - electrospray ionization (ESI), 2177
 - Fourier transform ion cyclotron resonance (FTICR), 2177
 - liquid solvent extractions (LSEs), 2177–2178

- molecular characterization, 2176
- structural investigations, 2176
- sub- and super-critical fluid (SCFs), 2178
- super-critical fluid extraction (SCFE)
 - protocols, 2177
- ultra-high resolution mass spectrometry (UHR MS), 2177
- Sorghum (*Sorghum bicolor*), 230, 299, 300
 - hybrids, 1796–1797
- Sorption, 202, 2195–2196
 - of anions, 355
 - and sequestration, bioavailability, 1772–1773
- Sorption-based textural control, 316
- Sorption quantity–intensity (Q/I)
 - relationships, 1702
- Sound transmission measurements, 87
- Sources of error, temperature measurement, 2303–2304
- South African classification systems, 418
 - binomial system, 418–419
 - taxonomic system, 419–420
 - Tugela basin classification, 418
- South American tropical savanna ecosystem, 345
- Southern circumpolar soils, 1756
- Southern Great Plains, 1113
- S-oxidizing bacteria, 7
- Soybean, 214
- Soybeans (*Glycine max* L.), 220, 223, 227, 300, 493
- Sparse shrubs, 1066
- Spatial databases, 1318–1319
 - integration, 1319
- Spatial decision support system (SDSS), 1319
- Spatial distribution, 2354–2357
- Spatial information system, 675–676
- Spatial recognition, 1314
- Spatial variability, 2303
- Species/cultivar choice, 228
- Species-specific fungal enzymatic responses, 2105
- Specific cell component analysis, microbial communities, 1434
- Specific discharge, 1991
- Specific emergy, 2
- Specific heat, 1083
- Specific surface area (SSA), 1663–1664, 2324–2325
 - determination
 - microporosity and accessibility, 2256–2257
 - molecular areas, 2257
 - negative adsorption (co-ion exclusion), 2257
 - t-method, 2256
 - particle size distribution, relation with, 2255
- Spectral libraries, 2197–2198
- Spectral mixtures, soil–plant
 - hyperspectral sensors, 2198
 - inferring soil properties through vegetation, 2198–2199
 - mixture modeling, 2197–2198
 - soil–vegetation canopy mixtures, 2197
- Spectroscopic techniques, C quantification, 1455–1456
- Sphagnum*-dominated plant, 1673
- Sphagnum lenense*, 1669
- Spherical peds, 2216
- Sphericity, 1656–1657, 1659
 - and roundness scales, 1660
- Sphericity index, 1657
- Spinach (*Spinacia oleracea* L.), 365
- Splash
 - corona, formation and characteristics of, 1882
 - droplets, 1882–1883
 - loss of soil material by, 1883–1884
- Spodosols, 105, 159, 410, 1470
 - distribution, 2200
 - soil-forming factors
 - climate, 2202
 - parent material, 2201
 - time, 2202
 - topography, 2201
 - vegetation, 2201–2202
 - soil-forming processes
 - humus and sesquioxides, translocation, 2200–2201
 - silicon, 2201
- Spontaneous dispersion, 2032
- Spontaneous vegetation, no-till, 1378
- Spring elasticity model, 1909
- Spring soil temperatures, 493
- Springtails. *See* Collembola
- Spring wheat (*T. aestivum* L.), 362
- Sprinkler irrigation, 1262
- SQI assessment. *See* Quality, soil, soil quality index (SQI) assessment
- 16S rRNA genes, 179
- Stability
 - constant, 63
 - index, 63
- Staggers, 1393
- Standardization, 1855
- Standard Proctor Test (ASTM D-698), 457
- Starch-degrading enzymes, 142
- Staring, W.C.H., 412
- Steady-state flow rate, 122
- Steady-state infiltration rate, 122
- Steady-state methods, 86
- Steady-state rate, 120
- Steam/water granulation, 921
- Steep slope reclamation, 1464
- Stefan–Maxwell equations, 986
- Stem cells, 243
- Stem infiltration, 1553
- Steric hindrance, 1772
- Stewardship, 850–851
- Stibnite (Sb₂S₃), 146
- Stiff grass (*Miscanthus sinensis*), 828
- Stock change rate, 1611
- Stomatal processes, 858
- Storage efficiency, 1257
- Storm profile, 562
- Stratification, 93
- Stratification, landfill gas, 1323
- Stream terraces, 1341
- Streptococcus*, 244
- Streptomyces* sp., 41
- Streptomyces thermoautotrophicus*, 232
- Stress soil productivity, 884
- Strip cropping, 2610–2611, 2611–2612
- Strip tillage, 493
- Strong fixation of anionic fertilizers, 2437
- Strontium
 - cation exchange capacity (CEC) ratio, 2205
 - fertilizer use, impact of, 2205
 - interaction with soil matrix, 2203–2205
 - regular lime applications, long-term influence, 2205
 - in soil, 2203
 - uptake by plants, 2205
- Structural/aggregate stability, 62
- Structural impacts, acid rain, 13
- Structural resiliency, 504
- Structure
 - aggregate formation, 2207–2208
 - belowground management, 2209–2211
 - description, 2207
 - earthworms, 2212–2215
 - environmental significance, 2208
 - pedological concepts and water flow, 2216–2220
 - plant establishment, 2221–2224
 - roots, 2225–2227
 - soil, 51, 1388–1390
 - surface and subsurface horizons, 2208
- Structure-forming agent, 505
- Structure index (SI), 1527
- Structure limitations, 114
 - management options, 114–115
 - processes, 114
- Stubble burning and grazing, 555
- Stubble mulching, 2609
- Sub- and super-critical fluid (SCFs), 2178
- Sub-irrigation, 2228–2230
- Submersion method, particle density, 1652–1653
- Suboxic soils, 126
- Subplastic soils, 1741–1742
- Sub-Saharan Africa, 943
- Sub-Saharan Africa (SSA), 1530
- Subsoiling
 - application, 2412
 - reduce cost of, 2412
- Subsoil organic carbon (OC)
 - composition of, 1615
 - global rooting depths and vertical distribution, 1615
 - quantity of, 1615
 - sources of, 1614–1615
 - turnover times, 1616
 - vertical distribution, global estimates, 1615
- Subsoils, 181
 - compaction, 513, 516, 598
 - compaction control, 1185
 - reclamation, 2031
 - sodicity, 2030
 - alkalinity, 2056
 - amelioration and management, 2057–2058
 - constraints to crop production, 2057
 - environmental impacts, 2057
 - low hydraulic properties, 2056
 - micronutrients and ion toxicity, deficiencies, 2057
 - plant available water capacity (PAWC), 2056
 - research needs, 2058
 - salinity, 2056
 - swelling, 114
- Substrate-induced respiration, microbial communities, 1434

- Substrate-induced respiration (SIR)
method, 1431
- Substrate properties, 1306
- Subsurface
thermometers, installation, 2304
tillage, 48–49
water movement, 1327
- Subterranean clover (*Trifolium subterraneum*), 229
- Subterranean termites, 2309
- Sugar beet (*Beta vulgaris* L.), 363
- Sugarberry (*Celtis laevigata*), 2573
- Sugarcane, 118, 214
- Sugarcane fields, harvest systems and residue management
biofuel sustainability, 2231
CO₂ emission, conceptual aspects on, 2235
crop residue, 2233–2234
decay factor, 2235
greenhouse gas (GHG) balance, 2232–2233
National Alcohol Program, 2231
synthetic nitrogen (N) fertilizers, 2232
- Sugarcane (*Saccharum officinarum*), 223, 493, 1283
- Sulfate, 26
- Sulfate and sulfide minerals
barite, 2240
gypsum, 2239
jarosite, 2239
sodium and magnesium sulfates, 2240
soluble Fe sulfates, 2239–2240
sulfides, 2238–2239
- Sulfate-reducing bacteria, 26, 27
- Sulfate reduction, 28
- Sulfate-S, 2241
- Sulfide-bearing soil, 38
- Sulfide mineral oxidation, 8
- Sulfide minerals (Sulfaquents), 728
- Sulfide oxidation, 6, 28–29
- Sulfides, 36, 2238–2239
- Sulfidic materials, 18
- Sulfidization, 28
- Sulfur
agroecological aspects
global change, 2244–2245
interaction, 2244
plant health, 2245
sustainability, 2244
biological aspects of
crops and crop rotation, 2242–2243
mineralization, 2242
overview and efficacy of, 2242
physico-chemical separation, 2241
sulfate-S, 2241
chemical behavior in rhizosphere, 2247–2250
fertilizers, 2244
nutrition, 2251–2254
physico-chemical aspects
capillary rise/leaching, 2243
emissions, 2244
soil compaction, 2243–2244
soil water regime, 2243
in soil, 2241
- Sulfur dioxide (SO₂), 11
- Sulfuric acid, 6, 11, 19, 36
digestion, 1397
- Sulfuric horizon, 17, 18
- Sulfuricization, 29
sulfide oxidation, 28–29
- Summer legumes, 224
- Summit, 1327
- Sunday soil, 1666, 2046
- Sunflower (*Helianthus annuus* L.), 494, 502
field, 208
- Super-critical fluid extraction (SCFE)
protocols, 2177
- Superoxid dismutase, 1729
- Superoxide, 232
- Superplastic soils, 1741–1742
- Supervised classification, 1314
- Supplementation, 1446
- Suppressive soils, 1526
- Surface abrasion, 1553
- Surface charge density, 427
- Surface clusters and surface precipitates, 1428
- Surface degradation, 1183
base saturation, 1184
land use and fertilization, 1183–1184
salinization, degradation by, 1184–1185
soil tillage systems, 1185
subsoil compaction control, 1185
trampling, 1185
- Surface-derived drinking water, 13
- Surface drainage, 2210
- Surface elevation, 2335
- Surface erosion
slope failure, 1349
- Surface hydration, 173
- Surface-irrigated fields, 1261
- Surface irrigation, 1261–1262
and rainfall erosion differences, 1262–1263
- Surface layer compaction, 598
- Surface microtopography, 1138
- Surface mining, 1458
- Surface Mining and Control Act of 1977 (SMCRA), 37
- Surface Mining Control and Reclamation Act (SMCRA), 1458
- Surface relative humidity, 864
- Surface-retained crop residues, 530
- Surface roughness, 89
- Surface saturation
slope failure, 1348
- Surface temperature measurement, 2302
- Surface tillage on aeration, 47–49
- Surface waterlogging, 2042
- Surface waters, N pollution of
biochemical processes, 2268–2269
fertilizers, 2268
hydrologic processes, 2269–2270
needs and approaches, 2272
problems caused by, 2268
sources, 2268
spatial variability, 2270
watershed scale analyses
efficiency of fertilizer, 2271
hydrologic process models, 2271
regional input–output analyses, 2270–2271
- Susceptibility to degradation, 1919–1920
- Susceptible (erodible) soils, 786
- Suspended sediment
biodiversity, 1983
hypoxic zones, 1983
oxygen availability, 1983
reported in species-specific responses, 1983–1984
respiration in aquatic organisms, 1983
scattering of light, 1982
turbidity, 1983
wavelength-dependent scattering, 1983
- Suspension density, particle density, 1653
- Suspension effect, soil pH, 1698
- Suspensions, 920
- Sustainability, 207–208, 884–885, 1115, 1306, 1377, 1380
management of soil quality and variability, 1116
socioeconomic determinants, 1115–1116
sulfur, 2244
- Sustainable agriculture, 1227
politics of soil health, 2277
social aspects
and corporate industrial agricultural economy, 2273–2274
definition, 2273
lack of, structural causes, 2273
public policy and future, 2274
as social movement, 2274
soil quality
agriculture and environmental sustainability, 2276–2277
assessment of, 2277–2279
strategies and indicators for producers, 2278
- Sustainable development and sustainability
goal and process, 2280
government goals and policy objectives, 2281
opportunities for positive change, 2282–2283
pillars of, 2281
sense of urgency, 2281–2282
steps to
collaboration and partnerships, 2284
effective tools use, 2283–2284
goals, 2283
lead by example, 2284
learn and adapt, 2284
measure and track progress, 2284
system dynamics, 2283
taking long view, 2283
- Sustainable land management (SLM), 732, 1413
- Sustainable soil and land management restoration, resilience, 1925
- Sustainable soil fertility management agroforestry and intercropping, 896
azolla and anabaena, 895–896
cover crops and green manures, 895
integration of livestock, 896
mulches, 896
nutrient recycling, 895
reduced tillage, 896
soil conservation, 895
- Sustainable use of soil, 1352
- Swamps, 2568
and marshes, peatlands, 1668
- Sweet peas (*Lathyrus odoratus*), 220
- Swelling, 172, 757
clays, 2022
- Swell–shrink activity, 93
- Switchgrass (*Panicum virgatum*), 828

- Switchgrass (*Panicum virgatum* L.), 210, 211, 214
- Sycamore (*Platanus occidentalis*), 2573
- Symbiotic and free-living N₂ fixers, 220–221
- Symphyleona*, 448
- Synthetic aperture radar (SAR), 2480
- Synthetic nitrogen (N) fertilizers, 2232
- Synthetic organic chemicals, 1898
- Synthetic polymer, 824–825
- Syntrichium endobioticum, 143
- Systematic packing, types, 1656
- Systematic spatial variation, 1346
- System of rice intensification (SRI), 710
- T**
- Taiga. *See* Boreal forest
- Tailings, minerals processing residue rehabilitation
- definition, 2286
 - mine waste rehabilitation practices
 - characteristics, 2288–2290
 - ecological approach, 2287–2288
 - engineering approach, 2287
 - potential environmental problems, 2286
- Tamarind (*Tamarindus indica*), 734
- Taproot systems, 1388
- Taxonomies
- architecture, 2298
 - azonal soils, 2296
 - hand specimen, 2296
 - national and international classification systems, 2300
 - normal soils concept, 2296
 - over 20th century, 2295–2296
 - over 21th century, 2296–2300
 - U.S. soil taxonomy and Indian scenario. *See* U.S. Soil Taxonomy (USST)
- Taxonomy, 17–18
- sulfidic materials, 18
 - sulfuric horizon, 18
- Tea (*Camellia sinensis*), 107
- Tea husbandry, 1383
- TEC. *See* Threshold electrolyte concentration (TEC)
- Tecto aluminosilicates, 1791
- TEM. *See* Transmission electron microscope (TEM)
- Temperate agriculture, 504
- organic additions, fertilization, and Ca, 505–506
 - perennial forages and pastures, 505
 - short-term rotations and cover crops, 505
 - soil cultivation and conservation tillage, 506
 - soil structural quality in temperate farming systems, 505
 - soil structure-forming agents, 504–505
 - soil structure in temperate zones, 504
- Temperate deciduous forests, 964
- Temperate grassland and forest biome, 1023
- Temperate rain forests, 964–965
- Temperature, 126
- and moisture regimes, 111
- Temperature, soil respiration, 1929
- Temperature management
- mulch farming, 1498–1499
- Temperature measurement
- sources of error, 2303–2304
 - subsurface thermometers, installation, 2304
 - surface temperature, 2302
 - thermometers types, 2302–2303
 - variability, 2302
- Tempest, 169
- Tensile strength, 53, 219, 1074
- Tepetates*, Mexico, 2305–2307
- Tephra sediments, 134
- Terminator technology, 245
- Termite-assisted hand-pitting across a surficial pan, 662
- Termites (isoptera), 55–56, 140–141, 205, 877–878
- effects on soils, 2310–2311
 - habitat, 2309–2310
 - in soil food webs, 1627
 - types, 2309
- Terra Bruna Estruturada, 380
- Terraces, 1341, 1986
- formation, tillage erosion
 - ancient lynchets, 2342
 - bench terraces, 2344
 - benefits and problems, 2345–2346
 - contemporary lynchets, 2342–2343
 - engineered terraces, 2343
 - gradient terraces, 2344–2345
- Terra preta* (black earths)
- chemical data, 2314
 - classic pedogenic model, 2315
 - features, 2312
 - fire residual parent material, 2313–2314
 - illustrations, 2313
 - magnetic susceptibility (MS) values, 2313
 - natural wildfires, 2313
 - surface and deeper soil data, 2314
- Terra Preta de Indio*
- distribution, 2317–2319
 - origin, 2316–2317
 - physical and chemical properties, 2317
 - properties of, 2316
 - terra mulatta, 2317
- Terra preta* soils, 193
- Terra Preta* soils, 189
- Terrestrial biosphere, 443
- Terzaghi's theory, 474
- Testing
- apparatus for soil analysis, 2320–2321
 - correlation and calibration, 2321
 - extractants for plant nutrients, 2321
 - fixed extraction ratios, 2321
 - future of, 2322
 - interpretation and recommendations, 2322
 - methods, 2321
 - quality control procedures, 2322
 - samples preparation for laboratory analysis, 2320
 - sampling, 2320
 - saturation extract methods, 2321
 - sodium adsorption ratio, 2321
- Tetradontophora bielensis*, 448
- Texture, 680–681
- gold tailings, 2289–2290
 - particle size analysis, 2324
 - relevance of
 - clays, 2325
 - ease of cultivation, 2326
 - field assessment, 2325
 - guide for assessment, 2326
 - hard setting characteristic, 2328
 - nutrient and water holding capacity and transmission properties, 2326–2328
 - particles packing, 2328
 - specific surface area (SSA), 2324–2325
 - remote sensing, precision agriculture, 1816–1817
 - soil properties affecting, 2324
- Texture relevance
- ease of cultivation, 1665
 - hardsetting characteristic, 1666
 - nutrient- and water-holding capacities, 1665–1666
 - soil consistency, 1664–1665
 - specific surface area (SSA), 1663–1664
 - transmission properties, 1665–1666
- Thawing, diurnal freezing and, 973
- field studies, 973–974
 - simulation studies, 974
 - soil freezing and water movement, 973
- Thematic mapper (TM) sensor, 1973
- Theophrastus, 884
- Theoretical interpretation of sediment transport, 1027
- Thermal advection, 1080, 1081–1082
- Thermal capacity. *See* Heat capacity
- Thermal conductivity, 1087
- Thermal conductivity detector (TCD), 440
- Thermal gradient, 1081
- Thermally induced liquid flow, 1080
- Thermal oxidation techniques, 1454–1455
- Thermal stability, 2630
- Thermistors, 2303
- Thermocouples, 2303
- Thermogravimetric technique, 2525–2526
- Thermometers types, 2302–2303
- Thermonuclear bomb tests, 2188
- Thermopile, 2303
- Thiobacillus ferrooxidans*, 28, 200
- Threshers and self-propelled tractors, 799
- Threshold electrolyte concentration (TEC), 2041, 2044
- Threshold value, 1853
- Thufur (hummocks), 136
- Tidal marsh, 2568
- Tilapia, 275
- Tillage, 74, 493, 1439, 1492, 2219
- erosion
 - definition, 2330
 - determination, 2330–2331
 - effects of, 2331–2332
 - evidence of, 1346
 - measurement techniques, 2333–2335
 - monuments of, 1345
 - properties and productivity effects, 2336–2338
 - relationship to water erosion, 2339–2341
 - relative soil displacement distances, 2331
 - slope gradients, 2330
 - terrace formation, 2342–2346
 - transport coefficients, 2330
 - gas exchange, 2348–2350
 - intensity, 1423
 - landscape, SOM, 2143–2144
 - vs. no tillage, 509
 - operations, 1388, 2444

- Tillage—
 practices, nitrate leaching management, 1539
 practices, rice paddies, 1951
 and roughness, 513
 systems, 1185, 1602
 translocation, 2339
- Tilth, soil
 history of, 2351–2352
 improvement of, 2352
 role of, 2352
- Timber production
 emergy analysis, 4
- Time and space, soils in
 accelerated erosion, 2354
 biosequences, 2356
 evolution and degradation, 2353–2354
 landscape, cross section, 2356–2357
 parent material, 2353
 soil formation rate, 2354
 spatial distribution, 2354–2357
 tolerable soil loss, 2354
 vegetative communities, 2356
 Vertisols, 2356
 wet and/or cool climates, 2355–2356
 wetness and drainage characteristics, 2357
 world soil map, 2355
- Time-domain reflectometers (TDR), 858, 2476, 2528
- Titanium dioxide NPs, 1513–1514
- Tithonia diversifolia*, 2226
- t-method, SSA, 2256
- Tobacco (*Nicotiana tabacum* L.), 363
- Toeslope, 1328
- Tolerable soil loss, 2354
- Tomocerus minor*, 143
- Topdressing, 904
- Topography, 1337
- Topsoil compaction, 1639–1640
- Torrent
 beds, 2362
 control treatments, 2362
- Torrential erosion
 causes of, 2360
 genesis of, 2359–2360
 in Himalayan region, 2359
 preventive and control measures
 state interventions, 2361
 technological package, 2361–2362
 susceptible area, delineation of, 2360–2361
 torrent vs., 2358–2359
- Total carbon (TC), 1449
- Total economic value (TEV), 2585
- Total ecosystem nitrogen, 1938
- Total porosity, 1783
- Total soil respiration, 1928
- Total suspended solid (TSS), 1109
- Toxic accumulation, 1225
- Toxic and xenophobic compounds, 1435
- Toxic ions, 2286, 2289
- Toxicity concentrations, 1736
- Toxicity-testing methods, 200–201
- Toxic metals, 13
- Trace metal contamination, 2364–2365
 anthropogenic, 2365
 geogenic, 2365
 metal behavior in soil, 2365–2366
- methods of assessment
 biological assessments, 2367
 chemical assessments, 2366
 modeling, 2366–2367
 pollution-induced community tolerance (PICT), 2367
 sources of contamination, 2365
- Tracers, 2334
- Trading soil carbon, 76–77
 actions, 81–82
 agriculture, 78
 challenges, 78
 attitudes, 80–81
 market demand factors, 79–80
 physical factors, 78–79
 trading volumes and prices, low, 77
- Traditional ceramics, 1418–1419
- Traditional ecological knowledge (TEK), 1282
- Trafficability, 681–682
- Tragedy of the Commons, 1356
- Trampling, 1185
- Transantarctic Mountains, 1753
- Transferases, 737
- Transfer functions and solute movement, 2196
- Transformity, 2, 4
- Transgenic *Arabidopsis*, 528
- Transgenic mitigation, 245
- Transient and persistent organic matter, 55
- Transient salinity, 2030
- Transitional clay minerals and allophanes, 1792–1793
- Translations, religious aspects, 1903–1904
- Translocation, 2310
 and accumulation, soluble salts
 in landscapes, 2192
 within soils, 2192–2193
 measurements, 2333–2334
- Transmission characteristics, pore system, 1785
- Transmission electron microscope (TEM), 1454
- Transmission properties, soil texture relevance, 1665–1666
- Transport from soil to roots, nutrient, 1580–1581
- Trapezoidal embankments, 748
- Treated wastewater (TWW), 2467
- Tree windbreaks, 2611
- Trichloroethylene (TCE), 1717
- Trichoderma* spp., 240, 241
- Trifolium*, self-regenerating legumes, 1422
- Trigger value, 1853
- Trihalomethane, 840
- Trimming of the soil core flush, 258
- Trinitrotoluene, 1717
- 2,4,6-Trinitrotoluene (TNT), 199
- Trioctahedral micas, 1791, 1792
- Triple superphosphate, 921
- Tristania conferta*, 1324
- Triticum monococcum*, 734
- Tropeptic Hapludox, 820
- Tropheryma* sp., 41
- Tropical monsoon/deciduous forests, 965
- Tropical rain forests, 965
- Tropical smallholder farm systems, 507
 burning vs. mulching, 508–509
 clearing method, 508
 climatic zone, 508
 fallow, 508
- fallow length, 508
- improved planted fallows, 509–510
- indigenous systems of soil structural management, 510
- manuring, compost, and fertilizer application, 509
- site selection, 508
- and soil structure, 508
- tillage vs. no tillage, 509
- Tropics and subtropics, 2369
 Alfisols, 2371
 Entisols, 2371
 Inceptisols, 2371
 Oxisols, 2369–2370
 Ultisols, 2370–2371
 Vertisols, 2371–2372
- Tropics and subtropics and swelling soils, 512–513
 managing soil structure, 513
 exploiting soil cracking, 513
 pasture phases to rebuild organic carbon, 514
 retaining soil cover, 513
 rotations to manage soil moisture deficit, 513
 tillage and roughness, 513
 zonal wheel traffic to reduce compaction, 513
- Tropics biome, 1023
- Tubular voids, 2218
- Tugela basin classification, 418
- Tugela (Thukela) river, 418
- Turbations, 2373
 bioturbation, 2373–2374
 cryoturbation, 2374
 pedoturbation, 2374–2375
- Turbels, 990, 1753
- Turbid Cryosol, 386
- Turbulence, 2629
- Turbulent flow, 93
- Turfgrass, 2377
 landscapes, functions in, 2377
 SOC, 2378
 soil conservation, 2377–2378
 water quality, 2378
- Turnover, SOM
 biotic and abiotic conditions, 2190
 CENTURY SOM model, 2189
 chronosequence modeling, 2187
 definition, 2186
 disturbance or management practices, 2189
 factors controlling, 2188–2189
 inputs and outputs balance, 2187
 macro- and microaggregate-associated C estimation, 2190
 mean residence time (MRT), 2186
 measuring, 2186–2188
 pools, 2189–2190
 range and variation in estimates of total, 2188
 thermonuclear bomb tests, 2188
- TWW. *See* Treated wastewater (TWW)
- Tying the ridges, 529
- Type T wire, 2303
- Typic Hapludox, 820
- Typic Paleudult, 820
- Tyre-inflation pressure, 517

U

- UAV thermal imaging, 1009
- Udepts, 1153–1154
- Ulmus americana* (American elm), 2573
- ULSE. *See* Universal soil loss equation (USLE)
- Ultisols (*Podzólicos Vermelho-Amarelo Distrófico*), 112, 410, 1162, 2292–2293, 2299, 2370–2371, 2380
 - common minerals
 - low-activity minerals, 2382
 - smectites, 2382–2383
 - morphology of, 2380
 - argillic and kandic horizons, 2381
 - horizons, 2380
 - SOM in, 2380
 - soil chemical properties, 2382
 - base saturation, 2382
 - soil acidity, 2382
 - soil-forming factors
 - state factor approach, 2381–2382
 - soil management and use
 - cultivation, 2383
 - use, 2383
 - suborders, 2383
- Ultrahigh mass spectra, data processing, 2154–2156
- Ultra-large chambers with long-path infrared spectrometers, 441
- Umbric Leptosols, 103
- Umbrisols, 392, 409
- Unburnt eucalypt fuels, 927
- UNCCD. *See* UN Convention to Combat Desertification (UNCCD)
- Unconfined aquifer, 1993
- Uncontrolled seepage, 66
- UN Convention to Combat Desertification (UNCCD), 1825
- Underfertilization, 886
- Undisturbed soil, 987
- UNFCCC. *See* UN Framework Convention on Climate Change (UNFCCC)
- UN Framework Convention on Climate Change (UNFCCC), 1825
- United Nations Childrens Emergency Fund (UNICEF), 162
- United Nations Conference on Desertification (UNCOD), 657
- United Nations Convention to Combat Desertification (UNCCD), 609
- United Nations Environment Programme (UNEP), 657
- Universal Soil Loss Equation (USLE), 772, 792, 833, 845, 1868–1869, 1881
- Unlayered (non-platy) petrocalcic horizons, 1691
- Unmanned aerial vehicles (UAVs), 1009
- Unsupervised processing, 1314
- Unsustainable grazing, 625
- Untreated wastes, 1763
 - See also* Pollution
- Unweathered soil, 1324
- Uplands, 157–158
- Upland soils (*terra firme*), 111–112
- Uppsala (*humus rubra*), 396
- Uranium contamination
 - in fertilizers, 2386
 - general issues, 2385
- hazards, 2385–2386
 - protection of soils, 2387
 - in soils and plants, 2386
 - in waters, 2386
- Urban development, 37, 159
- Urbanization, 307, 662
- Urban lands, 2388–2389
 - agriculture on vacant land, 2391–2393
 - and construction sites, 2395
 - degradation, 2395–2396
 - substrates, 2395
 - ecosystem services, 2390–2391
 - management, 2400
 - ecosystem services, 2400–2401
 - food by management of urban soils for agricultural production, 2403–2404
 - functions and ecosystem services, 2400
 - SIC, 2401–2402
 - SOC, 2402–2403
 - rehabilitation of construction sites, 2397–2398
 - rehabilitation of urban soils
 - aeration, 2397
 - compaction, 2397
 - greenery and street trees, 2397
 - hazardous compounds, 2396
 - salinity, 2397
 - sealing, 2397
 - and rural soils, 2389–2390
- Urban sewage, 1765–1766
- Urban soils, 308
- Urban wastewater for irrigation, 2407
 - heavy metals, 2408
 - bioavailability and contamination of food chain, 2408–2410
 - for livelihood, 2407–2408
- Urea, 920–921, 2411–2412
 - controlled release, 2412–2414
 - application, 2414
 - subsoiling
 - application, 2412
 - reduce cost of, 2412
- Urea ammonium nitrate (UAN), 919
- Urea hydrolysis, 1036
- Uropodina, 1488, 1489
- U.S. Army Corps of Engineers (USACE), 65
- U.S. Department of Agriculture–National Resources Conservation Service (USDA–NRCS), 2344–2346
- U.S. Department of Agriculture (USDA), 214, 459, 460, 1007, 2086–2089
- U.S. Environmental Protection Agency, 1546
- U.S. Environmental Protection Agency (USEPA), 200, 1406–1408
- U.S. Geological Survey (USGS), 1018
- U.S. National Guard, 70
- U.S. Soil Taxonomy (USST)
 - soil orders identified in India
 - Alfisols, 2292–2293
 - black soils and their intergrades, 2291–2292
 - brown forest soils in tropical climate, 2292
 - Inceptisols, 2293
 - Oxisols, 2292–2293
 - Paleosols, 2293
 - polygenetic soils and their classification, 2293–2294

- salt-affected soils and their
 - classification, 2293
- Ultisols, 2292–2293

USDA/NRCS soil surveys, 1007–1008

U-shaped valley, 101

USLE. *See* Universal Soil Loss Equation (USLE)

USLE standard runoff plot, 815

USST. *See* U.S. Soil Taxonomy (USST)

Ustepts, 1153

UV spectroscopic method, soil microbial biomass (SMB), 1431

V

- Vacuum infiltration, 1553
- VAD. *See* Vitamin A deficiency (VAD)
- Vadose zone, 930
- Vague soils, 413
- Valley erosion, 1372
- Value of soil to humans, 2416
 - archive of natural and human history, 2418
 - base for buildings, lawns, and gardens, 2417
 - interaction with atmosphere, 2417
 - interaction with water, 2417
 - medium for diversity, 2418
 - producer of food and fiber, 2416
 - producer of forests, 2416–2417
 - in recreation and sports, 2417–2418
- Van Genuchten–Durner approach, 1054
- Van Leeuwenhoek, Antonie, 176
- Vapor diffusivity, 1079
- Vapotranspiration, 1107
- Variability, 2419–2420
 - concepts of, 2423
 - kriging, 2425
 - sampling and quantifying, 2424–2425
 - spatial estimation methods, 2425–2426
 - diversity and heterogeneity, 2428
 - facets of spatial, 2420
 - extrinsic factors of variability, 2421–2422
 - soil formation, 2420–2421
 - soils, soil properties, soil processes, and time, 2420
 - variation at microscale, 2422–2423
 - managing, 2431
 - measuring soil variability, 2430–2431
 - origins of, 2428–2429
 - scales of variation, 2428
 - systematic and random variation, 2429–2430
- Variability, temperature measurement, 2302
- Variable charge soils, 2432–2434
 - chemistry of, 2434
 - constant charge minerals, 2434–2435
 - variable charge minerals, 2435
 - management of, 2435–2438
 - mineralogy, 2434
- Variable-charge surfaces, 353
- Variables, mapping soil carbon, 1410
- Variogram surface, 1013
- Vegetated waterways, 828–829
- Vegetation, 90
 - communities, 629
 - desert shrublands, 159
 - Loess plateau, 1373
 - and reforestation, 650
 - regeneration, 790, 791
 - and soils in Iceland, 623

- Vegetation indices (VIs), 2198–2199
- Vegetative barriers, 827
- Vegetative barriers/strips, 828
- Vegetative filter strips, 594, 747
- Vegetative grass barriers, 747
- Vegetative stabilization, 2287
- Vegetative strips
- buffer strips and riparian strips, 829–830
 - filter strip terraces, 829
 - grass strips, 828
 - settling basins, 830
 - vegetated waterways, 828–829
- Vegetative structure, 979
- Vehicle-mounted proximal sensor system, 1874
- Velocity profile, 2628
- Veracrisols, 393
- Vermiculite, 1477–1478
- Vernacular soil names, 372–374
- Vernadite, 2006
- Verrucomicrobia, 179
- Verticillium chlamydosporium*, 1520
- Vertisolic soil, 385
- Vertisols, 159, 300, 392, 409, 410, 512, 1162–1163, 1647, 2299, 2371–2372, 2440
- cracking nature, 513
 - geotechnical problems, 2446
 - land use systems
 - forestry and agroforestry, 2446
 - irrigated crops, 2446
 - pasture, 2446
 - rain-grown annual crops, 2445–2446
 - soil consistence, 2442–2443
 - soil physical and chemical properties relevant to agricultural use, 2440
 - cation exchange capacity, 2442
 - clay percentage, 2440–2442
 - climate and landform, 2442
 - organic matter, 2442
 - soil properties and the agricultural use
 - gilgai, 2445
 - soil erosion, 2445
 - soil management and tillage operations, 2444
 - water management, 2444–2445
 - soil–water relations, 2443
 - groundwater, 2444
 - infiltration and hydraulic conductivity, 2443
 - moisture retention, 2443–2444
 - surface crack management, 2450–2454
- Vertissolos, 381, 382
- Vesicular–arbuscular mycorrhizal fungi, 1901
- Vesicular–arbuscular mycorrhizal (VAM), 55
- Vesicular crusts, 2456
- formation, 2456
 - hydraulic properties, 2458
 - physical, chemical, and mineralogical properties, 2457–2458
 - stability, 2457
 - structure, 2456–2457
- Vetiver grass (*Vetiveria zizanioides*), 828
- Vetiver (*Vetiver zizanioides*) grass, 747
- Village bamboos, 2458–2459
- management, 2460
 - SOC stock and sequestration rate, 2461–2462
 - soil properties, 2460–2461
- Vinyl chloride (VC), 199
- Virgin soils, 561
- Visible–near-infrared spectra, DRS, 1887
- Vital ecosystem attributes, 1936
- Vitamin A deficiency (VAD)
- mineral nutrition disorders, 1446
- Vitrisols, 134, 136, 137
- classification perspectives, 136
- Volatile organic carbon (VOC), 202
- Volatilization, 117, 1422
- Volatilization, Se, 1718
- Volcanic eruptions, 134, 137
- Volcanic island, 134
- Volcanic tephra, 97
- Volumetric heat capacity, 1083–1084
- Volumetric water content (VWC), 1006
- W**
- Wadell's procedure, 1657
- Walkley–Black tests, 2060
- Wassents, 729
- Waste generation, pollution, 1763
- Waste management, 1824–1825
- Waste rock and overburden dumps, 2464
- rehabilitation, 2464
 - restoring growing medium for plants, 2465
 - revegetation, 2465–2466
- Wastewater, 2467
- constituents, 2467
 - irrigation with TWW, 2468
 - hydrophobicity, 2470
 - nutrients, 2468–2469
 - salt content, 2469–2470
 - sodicity, 2470
 - regulations, 2468
 - treatment, 2467–2468
 - treatment processes, 198, 924
 - TWW utilization, 2468
- Water and nutrient availability, 1597
- Water conservation, 2472
- soil–water conservation principles, 2472, 2473–2474
 - increasing infiltration, 2472–2473
 - reducing deep drainage, 2473
 - reducing evaporation, 2473
 - reducing use by non-crop plants (weeds), 2473
- Water content, 2480
- measurement, 1009–1010
 - methods for measuring, 2475
 - capacitance devices, 2476–2477
 - data sources, 2477–2478
 - gravimetric sampling and oven drying, 2475–2476
 - heat dissipation, 2477
 - portable measurement techniques, 2476
 - radioactive source devices, 2477
 - remote sensing, 2477
 - time domain reflectometry, 2476
 - microwave remote sensing, 2480
 - passive microwave systems and data, 2481–2483
 - SAR systems and data, 2481
 - soil moisture sensing, 2483
 - variability and sampling, 2475
- Water conveyance efficiency, 1256–1257
- Water cycling, 139–140
- Water erosion, 777, 778
- assessment, 817–818
 - empirical erosion model, 2489
 - ABAG (German USLE), 2491–2492
 - MULSE, 2490
 - SLEMSA, 2491
 - universal soil loss equation, 2489–2490
 - global problem, 2487
 - hybrid models, 2491
 - impacts of, 2487–2488
 - Onstad–Foster (OF) modification of the USLE, 2491
 - patterns, 1374
 - process-based modeling, 2504
 - continuous simulation models, 2504–2506
 - processes, 2485–2486
 - soil loss, 2492
 - RUSLE, 2492
 - spatial and temporal scale, 2486–2487
 - tillage erosion and, 2339–2341
- Water erosion, hybrid models, 2494
- comprehensive hybrid models, 2494–2495
 - erosion equation, 2494
 - model applications, 2496
 - APEX applications, 2498
 - EPA nutrient movement study, 2497–2498
 - 1985 RCA appraisal, 2496–2497
 - SWAT application, 2498
 - model components, 2495
 - hydrology, 2495
 - management, 2496
 - plant growth, 2495
 - residue decay, 2495–2496
- Water Erosion Prediction Project (WEPP), 845
- Water extraction, 652
- Water-filled pore space (WFPS), 1035
- Water films, soil plasticity, 1740
- Water flux, 1140
- Water–gas shift (WGS) reaction, 1800
- Water hammer pressure, 1881
- Water harvesting, 2515
- cropping considerations, 2516–2517
 - defined, 2515
 - runoff farming, 2515
 - site characteristics, 2516
 - system, types of, 2515–2516
- Water hickory (*Carya aquatica*), 2573
- Water-holding capacity (WHC), 66, 185, 206, 463, 1214, 1421
- soil, 504
- Water hyacinth, rhizofiltration, 1721
- Water infiltration, 2507
- factors affecting, 2507–2508
 - models, 2508
 - Green–Ampt infiltration model, 2508
 - Holtan equation, 2509
 - Horton equation, 2508–2509
 - Kostiakov equation, 2508
 - Philip equation, 2509
 - process, 2507
- Water infiltration, tilling, 2290
- Waterlogging, 526, 659, 683, 1324
- rice paddies, 1951
- Water management, 2444–2445, 2510
- application, 2511–2513
 - mulch farming, 1497–1498
 - water flow, 2510

- Water measurement methods, 2522
 - direct and indirect, 2522–2525
 - FDR (Capacitance) and TDR, 2527
 - frequency domain reflectometry, 2527–2528
 - neutron thermalization, 2526–2527
 - thermogravimetric technique, 2525–2526
 - time domain reflectometry, 2528
 - soil water content, 2522
- Water movement and soil structure, 2218
- Water permeability, 823
- Water regime, sulfur, 2243
- Water release curve (WRC), 2531
- Water-repellent soils, 1651, 2546
 - extent and origin, 2546–2547
 - management, 2547–2548
 - practical implications, 2547
- Water resource
 - Loess plateau, 1373
- Water retention, 561–562, 2535–2536
 - hysteresis, 2539
 - of Icelandic Andisols, 135
 - methods of characterizing, 2538–2539
 - models of, 2536–2537
 - mathematical description, 2537–2538
 - pedotransfer functions, 2537
 - plant-available water, 2539–2540
- Watershed approach, 2550
 - historical practices, 2553
 - Colorado River, United States, 2553
 - Kissimee River, Florida, United States, 2553–2554
 - San Joaquin Valley, California, United States, 2553
 - hydrologic cycle, 2550
 - management effects, 2551
 - natural variability, 2550
 - scale, 2551
 - strategy, 2552
 - forests, 2552
 - irrigated croplands, 2552
 - multiple use conflicts, 2552–2553
 - rainfed croplands, 2552
 - rangelands, 2552
- Watershed management
 - remote sensing and GIS applications, 2555
 - challenges and opportunities, 2557–2558
 - inventorying and monitoring, 2555–2556
 - research and development, 2556–2557
- Water-soluble pesticides, 1758
- Water-soluble P (WSP), 1229
- Water-solvated DNA and RNA segments, 2167
- Water-stable aggregation (WSA), 1508
- Water-stable macroaggregation, 505
- Water storage in soil, 1784–1785
- Water suction, 85
- Water supply and groundwater contamination, 1992–1993
- Water table, 1990
- Water tension, 1883
- Water transport equations, scale invariance of, 1978
- Water use efficiency (WUE), 1260, 1422, 1597–1598, 2518
 - eliminating fallow, 2520
 - enhancing effect, 443
 - fertilizer application, 2520
 - irrigation water use efficiency, 1260
 - reducing tillage/maintaining crop residues, 2518–2519
 - shifting water use to critical growth stages, 2519–2520
- Water vapor, convective exchange of, 2518
- Water vapor movement, 1079–1080, 1081
- Waterways, 1987–1988
- Wax, pores cast, 1388
- Wax-impregnated graphite electrodes (WIGEs), 45
- Weathering, 202, 216, 997, 1203–1204, 2559, 2564
 - of carbonate minerals, 2564
 - CO₂ over geological time, 2564–2565
 - factors, 2561–2562
 - mechanisms, 2559
 - acidolysis and acidocomplexolysis, 2559–2560
 - alkalinolysis, 2560
 - hydrolysis, 2560
 - salinolysis, 2560
 - of mica, 423
 - rates and CO₂ consumption, 2564
 - SOC sequestration, 1467
 - soil carbonates, 2565
 - structure, 2560
 - organo-mineral soil, 2561
 - rock, 2560
 - saprock, 2560–2561
 - saprolite, 2561
- Weathering of mica, 1792
- Weathering processes, 1755
- Weed management, 1602
- Weeds, 205
- Weighing system principle, 489
- Well-aerated soil, 85
- WEPP (Water Erosion Prediction Project), 814
- Western plains, India, 1159
- West Siberia, boreal forest soils, 248
- Wet combustion method
 - assessment, quality, 1844
- Wet combustion of soil, 1604
- Wetlands, 38, 295, 2567, 2589
 - conservation policy, 2580
 - global review, 2581–2582
 - guidelines, 2580–2581
 - implementation strategies, 2581
 - and wise use, 2580
 - definitions, 2567
 - ecological engineering, 2602
 - economic valuation techniques, 2585
 - economic values, 2584–2585
 - extent of, 2570
 - fauna, 2574
 - flora, 2572
 - GHG emission, 2589–2590
 - goods, 2584
 - methane, 2590
 - CO₂, elevated level of, 2592
 - diurnal variation, 2591
 - moisture and water level, 2590–2591
 - N₂O, 2592–2593
 - organic C, 2591–2592
 - redox potential, 2591
 - temperature, 2590
 - vegetation and plant species, 2592
 - wetland, types, 2591
- methane emission, 2596
 - diffusion, bubble ebullition, and transport through plants, 2598
 - production and consumption, 2596
 - soil temperature, 2597–2598
 - vascular plants and substrate, 2598
 - water table position, 2596–2597
- particle types and sedimentation processes, 2601–2602
- peatlands, 1668
- sedimentation, 2600
 - accretion, 2600
 - phosphorus (P) accumulation, 2600–2601
 - reduced conductivity, 2600
- soils, 135–136
- types, 2567–2570
 - valuing wetlands goods, 2585–2587
- Wet meadow, 2568
- Wetness, soil, 44
- Wet sieving, 62
- Wet sieving measurements, 1074
- Wet soil features, 2005
- Wetting, 1073
- Wetting and drying and freezing and thawing, 56
- Wetting–drying, 504
- Wetting front, 2507
- Wetting front suction, infiltration, 1189
- Wheat (*Triticum aestivum*, L.), 118, 299, 523, 530, 935–936
- White clover (*Trifolium repens*), 225
- White muscle disease, 1446
- Whole soil shrinkage, 2009
- Wildlife colonization, landfill sites, 1325
- Willow (*Salix* spp.), 210
- Wilting, 482
- Windblown dust, 2644
 - composition, 2645
 - deposition, 2646
 - dust emission, 2644–2645
 - size distributions and optical properties, 2645
 - transport, 2645–2646
- Wind-driven rain, 2647
 - raindrop impact energy, 2647
 - impact angle, 2647–2648
 - impact frequency, 2648
 - raindrop size distribution, 2647
 - rainsplash detachment and transport, 2648–2649
- Wind erosion, 1365, 1374–1375, 1639, 2603 and climate change, 2603–2608
 - control measures, 2609
 - artificial barriers, 2611
 - cover crops, 2610
 - herbaceous wind barriers, 2611
 - no-tillage, 2609–2610
 - sand dunes, 2613
 - strip cropping, 2610–2611, 2611–2612
 - stubble mulching, 2609
 - tree windbreaks, 2611
 - control on sand dunes, 2613
 - and deposition processes, 2614–2615
 - dynamics, 2636–2637
 - environmental effects, 2614
 - field measurement, 2618
 - passive samplers, 2619–2620
 - sampling considerations, 2618–2619

- Wind erosion—
 - suspension samplers, 2620–2623
 - tracer-based methods, 2623
 - global hot spots, 2625–2627
 - induced erosion, 2625–2627
 - on irrigated land, 2612–2613
 - on knolls, 2612
 - micrometeorology, 2628
 - changes in humidity, 2630
 - surface roughness, 2628
 - thermal stability, 2630
 - turbulence, 2629
 - velocity profile, 2628
 - wind shear stress, 2628
 - modeling, 2632
 - field-scale wind erosion models, 2633–2634
 - processes, 2632–2633
 - natural erosion, 2625
 - non-vegetative erosion protection, 2613
 - off-site effects, 2615–2617
 - on-site effects, 2615
 - particle entrainment, 2637–2638
 - processes, 2636
 - roughening the land surface, 2612
 - emergency tillage, 2612
 - self-balancing, 2638
 - soil dust, 2615
 - soil productivity, 2641
 - crop production decline, 2642
 - nutrient loss, 2641–2642
 - nutrient restoration, 2642
 - soil quality, 2640
 - eroded sediments, 2641
 - texture changes, 2640–2641
 - stable aggregates or clods on the soil surface, 2612
 - Wind erosion equation (WEQ), 460, 2633
 - Wind erosion prediction system (WEPS), 2633
 - Wind shear stress, 2628
 - Windthrow
 - natural disturbances, 1820
 - Wind-transported dust, 1359
 - Wind–water erosion, 1375
 - Winter pulses, 224
 - Winter rainfall, 1421
 - Winter wheat (*Triticum aestivum* L.), 73, 492, 900
 - Wisconsin Soil Health Scorecard, 1078
 - Within-season monitoring techniques, 1540
 - Wood, 189
 - Wood by-products, 189
 - Wood chips, 1056
 - Wooded savannah (*cerrado strictu sensu*), 345
 - Wood fiber, 1056
 - Woody plant range expansions
 - modern studies, 621–622
 - prehistory to industrial revolution, 621
 - Woody yucca (*Manihot esculenta*), 1283
 - World Agroforestry Centre, 343
 - World Bank, 1443
 - World Health Organization (WHO), 1443
 - World reference base (WRB)
 - buried soils, 2652
 - changes in 2014, 2653
 - diagnostics, 2650–2651
 - examples, 2653
 - first level, 2650
 - map legends, 2652
 - naming a soil (a profile), 2651–2652
 - second level, 2650
 - subqualifiers, 2652
 - World Reference Base (WRB), 391, 406, 640, 1414, 2300
 - Worms mix organic matter, 238
 - WRB. *See* World reference base (WRB); World Reference Base (WRB)
 - WSA. *See* Water-stable aggregation (WSA)
 - WUE. *See* Water use efficiency (WUE)
- X**
- Xenobiotica, 1758
 - Xerepts, 1153
 - Xiphinema bergeri*, 1525
 - Xiphinema diversicaudatum*, 1525
 - X-ray computed tomography, 1389
 - X-ray diffraction (XRD), 425, 1192–1193
 - patterns, 1396, 1398
 - X-ray fluorescence and microprobe, 1193
- Y**
- Yellow perch, 275
 - Yellow River
 - annual average runoff volume, 1373
 - Loess Plateau area, 1370
 - sediment source, 1374
 - soil erosion control, 1370–1371
 - Yield sensors, 1806
 - Yield sustainability
 - and enzymes, soil, 2102–2103
 - Yves Tanguy, 169
- Z**
- Zai controls, 510
 - Zea mays* plants, 2226
 - Zero net land degradation (ZNLDD), 613
 - Zero shrinkage, 2009
 - Zero-tillage
 - advantages and disadvantages, 1566–1567
 - results achieved by using, 1567–1568
 - Zero-tillage systems, 1301
 - Zero transpiration, 561
 - Zinc (Zn)
 - deficiency, 1444–1445
 - mineral nutrition disorders, 1445
 - Zipellé*, 510
 - Zircon, 1473
 - Zn-fortified food grains, 1443
 - ZOI. *See* Zones of influence (ZOI)
 - Zonal wheel traffic to reduce compaction, 513
 - Zones of influence (ZOI), 1624
 - Zone V, 2360
 - Zoomass, 236
 - Zooplankton, 15
 - Zygomycotina, 979